Chapter 7

Discussion

7.1   <u>Towards maximal objectivity and optimized (geo)statistical approaches in Habitat Mapping</u>

In all of the chapters, methodologies were put forward that strive towards a **maximal objectivity in habitat mapping**. To that end, **(geo)statistical methods** were tested and applied to the marine datasets, presented in this thesis.

In **Theme 1**, the coverage data for habitat mapping were optimized with the use of geostatistical interpolation techniques (or Kriging). Kriging techniques are objective in the sense that they make use of the spatial correlation between neighbouring observations, to predict values at unsampled places (Goovaerts 1999). These techniques give an indication of the errors and uncertainties associated with the interpolated values, based on a variance surface of the estimated values (Burrough and McDonnell 1998).

If there is a linear relation between the sedimentological variable (e.g. grain-size) and a secondary variable (e.g. bathymetry), **multivariate geostatistical techniques,** such as Kriging with an external drift (KED), can be applied.

In **Chapter 2**, KED was tested for the first time on sedimentological data of the entire Belgian part of the North Sea (BPNS). The multivariate geostatistics were limited to a **single secondary dataset**, being the **bathymetry**. Because of the linear relation between the bathymetry and the median grain-size, the interpolation could be improved significantly.

In **Chapter 3**, the same technique (KED) was applied to a set of sedimentological variables (d_s10, d_s50, d_s90 and silt-clay %) for a small study area on the BPNS. Instead of one single secondary dataset, a **whole set of secondary data** was used; **multi-scale topographical datasets** such as slope, aspect and curvature, on different spatial scales, were derived from the bathymetry using spatial analyses. Those datasets were used as secondary datasets for KED, after being reduced by Principal Components Analysis (PCA); this avoided multicollinearity between the data. Compared to other Kriging techniques (e.g. the most commonly used Ordinary Kriging or OK), the KED models performed better; this was demonstrated in Chapters 2 and 3. Due to the KED technique, the model output took into account the topographical variability of the seafloor, most often a steering parameter in the prediction of soft substrata habitats. This is not possible with OK.

Examples of geostatistics in the context of marine habitat mapping are rare in literature. Jerosch et al. (2006) used indicator kriging for the mapping of the spatial distribution of benthic communities at the deep-sea submarine Håkon Mosby Mud Volcano, located on the Norwegian–Barents–Svalbard continental margin. Pesch et al. (2008) applied Ordinary Kriging to interpolate abiotic variables, as input for decision trees for the prediction of benthic communities in Germany. Pesch et al. (2008) recommended the use of multivariate geostatistical methods for the prediction of sedimentological maps, as applied in Chapter 3.

Previously, the sedimentological map of the BPNS was limited showing median grain-size with only 3 classes: very fine sand (63-125 µm), fine sand (125-250 µm) and medium to coarse sand (250-2000 µm) (Lanckneus et al. 2001). This map was made using a simple inverse distance interpolation. Other sedimentological maps are often expressed in classes as well. The most common **sediment classification** is that

of **Folk** (Folk 1974), which is based on the relative proportions of gravel, sand and mud. The advantage of this classification system is its broad applicability and use, making it easy to compare sedimentological maps of different areas. This classification was used for the marine landscape mapping of the UK (Golding et al. 2004; and Connor et al. 2006). However, median grain-size is also a commonly available sedimentological parameter and is readily available for wider areas. Ideally, the analysis would be based on raw data acquired using similar methodologies; however, this is usually not possible and various datasets have to be merged to create a sufficiently dense grid of sample data. For new initiatives, the **MESH Standards and Protocols** (Coggan et al. 2007) and the **MESH Recommended Operational Guidelines** (White and Fitzpatrick 2007) for Seabed Habitat Mapping are recommended. A major **disadvantage of a classification system** (such as the Folk classification), is that the resulting output is **not well suited for modelling purposes**. Modelling techniques require, in general, **continuous datasets** (e.g. 0-100 %), with more variation than only a few classes. The sedimentological maps of Chapters 2 and 3 do not have classes, showing a separate value for each pixel. The resulting data grids can be used easily for other modelling initiatives (e.g. the use of the sediment data grid in sediment transport modelling or as input to the prediction of sandwave dimensions).

In **Theme 2**, both **abiotic and biotic datasets** were **integrated** in view of habitat mapping. Various methodologies exist for this integration, ranging from simple cross tabulation to complex statistical habitat modelling methods.

In **Chapter 4**, emphasis was put on the integration of abiotic datasets, to produce an ecologically relevant map of **marine landscapes** on the BPNS. The methodology was based on a combination of **PCA** and **cluster analysis**. It offered a more objective strategy to cluster a diverse range of information. The methodology, worked out in Chapter 4, allowed working with **continuous datasets**, instead of with classified layers (e.g. weak, moderate or strong maximum tidal stress for the UK seas in Connor et al. (2006); photic and non-photic zone for the Baltic Sea in Al-Hamdani and Reker (2007)) and did not oblige the user to make choices about the number and the breaks of the classes or about the maximum **number of input layers**. In Al-Hamdani and Reker (2007), only three abiotic datasets (sedimentology, light zone and salinity) were included to keep the landscape classification ecologically relevant, but also to limit the number of potential combinations to a manageable number. The PCA in Chapter 4 created 6 Principal Components, being linear combinations of the original 16 abiotic datasets. Still, the most difficult issue in marine landscape mapping, is choosing the appropriate number of landscapes. The maximum number of landscapes can be very high (e.g. 5 sediment classes, 2 light zones and 6 salinity classes resulted into 60 marine landscapes in Al-Hamdani and Reker (2007)). However, it is doubtful that all landscapes have an ecological relevance. Chapter 4 proposed a **statistical criterion (Calinski-Harabasz index)** to obtain the **optimal number of landscapes**. Also here included was a test whether all of the landscapes were relevant ecologically. This validity was tested, using an **indicator species analysis**, with a biological dataset containing macrobenthic species data (Marine Biology Section, Ugent – Belgium, 2008). A correlation between the defined clusters and the occurrence of the macrobenthic species was shown. Still, it must be clear that it is never the absolute aim of the marine landscape mapping to predict the biology as such; for that purpose other and better predictive modelling techniques exist (Guisan and Zimmermann 2000, for an overview). Marine landscapes give an **indication about the biology**,

based on abiotic datasets only and are most interesting in areas where biological data are scarce or absent.

An interesting marine landscape in Chapter 4 is 'cluster 8' (Figure 4.2), characterized by its **gravel occurrence**. Only 6 biological samples were available in this landscape to validate the ecological relevance. Moreover, the sampling technique (Van Veen grab sample) is not suitable to sample gravel, resulting in a biased ecological validation of this landscape. However, the marine landscape map does provide insight in **possible ecological valuable areas**; areas that may be unknown (or forgotten) and biologically under-sampled today. Van Beneden (1883) and unpublished data from Gilson (Royal Belgian Institute of Natural Sciences, 1899-1910) examined the former co-existence of several important ecosystem functions, associated to forgotten offshore "boulder fields" in the area of the Westhinder sandbank (Houziaux et al. 2007a). For this area, Houziaux et al. (2007a) inferred that high levels of epibenthic species richness and taxonomic breadth, wild beds of the European flat oyster (*Ostrea edulis*), and a spawning ground for the North Sea herring (*Clupea harengus*) seemed to occur in the 19th and at the beginning of the 20th century. It is as well in this area (at the SE side of the Westhinderbank) that the marine landscapes map indicates the presence of gravel fields. Houziaux et al. (2007a) showed that even today a high biodiversity of epibenthos exists in this area (e.g. 70 species of bryozoans at one single sampling station). Although not ground-truthed by the available biological samples or the macrobenthic communities on the BPNS, the marine landscapes map of Chapter 4 shows the presence of a gravel landscape in this area, with a possibly high biodiversity. This is not only the case for the Westhinder area, where Houziaux et al. (2007a) already demonstrated the high biodiversity. Other offshore areas might be valuable as well, as are the swales of some parts of the Hinderbanken, the swale at the SE side of the Oosthinderbank, and the swale between the Goote- and Thorntonbank. In the northernmost part of the BPNS, the patch of 'gravel' consists of shell-rich sand.

The strength of two **statistically-based habitat suitability modelling** (HSM) **techniques** was demonstrated in Chapters 5 and 6.
In **Chapter 5**, HSMs were produced for the 4 macrobenthic communities on the BPNS. The modelling technique used **discriminant function analysis** (DFA), to determine which variables discriminate between two or more naturally occurring groups. A limited number of abiotic datasets was used as input for the model (median grain-size, silt-clay %, depth, slope and distance to coast). The model selected median grain-size and silt-clay % as significant explanatory variables for the modelling of the macrobenthic communities. A three-fold **cross-validation** showed that the agreement between the model predictions and observations was very good and consistent. Moreover, the percentage correctly classified instances (CCI) and the Cohen's kappa index confirmed this agreement.
Important for the resulting models is that a suitable habitat, for a species or community, means that its composing species have the possibility of colonizing the habitat, but may as well be absent because of anthropogenic impacts, such as fisheries, or natural temporal variability. Habitat suitability thus predicts the specific ecological potentials of a habitat rather than the realized ecological structure (Degraer et al. 1999b). The value of DFA for HSMs of marine benthic communities has been demonstrated as well in Shin (1982); Vanaverbeke et al. (2002); and Caeiro et al. (2005).

In **Chapter 6**, another method for habitat suitability modelling was demonstrated. Using **ecological niche factor analysis** or ENFA, the occurrence of the macrobenthic species *Owenia fusiformis* was predicted for the BPNS. This modelling technique had the advantage of its capability to handle an unlimited number of abiotic datasets or ecogeographical variables (EGVs). This number was again reduced by **factor analysis** (ENFA), selecting the ecologically most relevant factors for *O. fusiformis*. Different algorithms and subsets of EGVs were applied, resulting in different HSMs. The best model was retained by cross-validation. Earlier, ENFA had been applied successfully for marine HSMs in the following examples: for deep-water gorgonian corals on the Atlantic and Pacific Continental Margins of North America (Bryan and Metaxas 2007); for squat lobsters in the Macnas Mounds in Porcupine Seabight, SW Ireland (Wilson et al. 2007) and for the northern gannet in Bass Rock, western North Sea in Scotland, UK (Skov et al. 2008). In addition, to showing the value of ENFA, Chapter 6 demonstrated the value of **multi-scale EGVs** as input for the HSMs. The importance of considering variations in spatial scale to predict the occurrence of marine benthic species was demonstrated also in Murray et al. (2002); Ysebaert and Herman (2002); Baptist et al. (2006); and Wilson et al. (2007).

The map of the median grain-size of the BPNS, constructed using KED (Chapter 2), was extended to the **Southern North Sea** (Figure 7.1a), including the southern part of the Dutch continental shelf and the southeastern part of the English continental shelf. This was justified on the basis of a linear relation between the median grain-size and the bathymetry for the whole study area. A map of the silt-clay% using OK, was completed for this area as well (Figure 7.1b). From this, the HSMs of the 4 macrobenthic communities (Chapter 5) could be applied to a wider North Sea region (Figure 7.1c-d-e-f). Although these maps are only preliminary versions, some first remarks can be formulated: 1) the Southern North Sea belongs to the same biogeographical area as the BPNS with comparable sediment type and communities; 2) the shading areas on the 4 HSMs are zones outside the limits of the model; this can be due to the presence of gravel (and as an abiotic variable representing gravel is no input for the model, typically associated fauna can not be predicted); 3) the high suitabilities of the *A. alba* community around the shading areas are modelling artefacts; 4) transborder modelling is usually not straightforward because of different resolutions of input data and different sampling and processing methods; and 5) the absence of the *M. balthica* community around Western Scheldt estuary can be due to possible lower estimations of Dutch silt-clay percentages.

The proposed methodology of Chapter 4 was not tested yet outside of the study area. This is due mainly to the unavailability of standardized abiotic datasets over larger areas. For initiatives in this direction, reference is made to Connor et al. (2006) and Al-Hamdani et al. (2007) who compiled marine landscape maps for the UK seas and the entire Baltic Sea (comprising 7 countries), respectively.
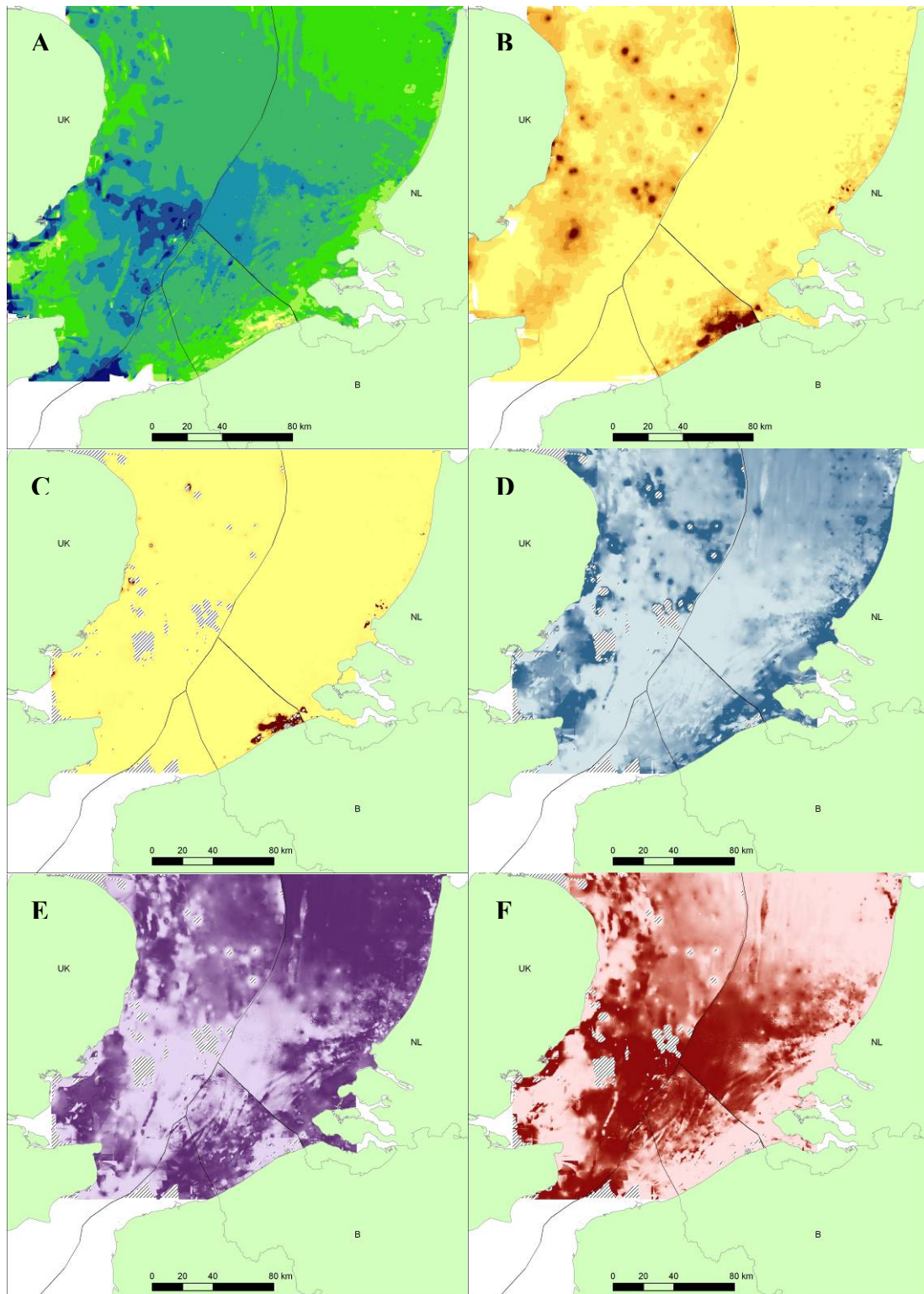
Figure 7.1: Methodologies of Chapters 2-5 applied to the Southern North Sea. A) Median grain-size of the sand fraction; B) Silt-clay %; C) HSM of *Macoma balthica* community; D) HSM of *Abra alba* community; E) HSM of *Nephtys cirrosa* community; and F) HSM of *Ophelia limacina* community. The maps are orientated to the North. Colour scales range from pale to dark and represent gradients of low to high values (63-2000 μm for A and 0-100% for B, C, D, E and F).

7.2 Reliability of abiotic data coverages and habitat maps

In the context of marine habitat mapping, the **importance of high quality abiotic data** is often overlooked. A high quality abiotic dataset or coverage implies that e.g. the spatial and temporal scale, ground-truth data, interpolation and model techniques are appropriate for the dataset and that they have an ecological relevance for the species under consideration. Habitat modellers tend to have a high confidence in the abiotic data they use, and one step further, managers rely particularly on the habitat maps to make decisions on the allocation of marine protected areas or for spatial planning purposes. However, if the topography, the substrate of the seafloor or the hydrodynamic regime is reconstructed from limited coverage, or if inappropriate scales are being used, the HSM application will likely poorly represent the species' or the community's spatial distribution.

Coverages are usually based on **models** (e.g. bottom current velocities or maximum bottom shear stress), or on **interpolations** of ground-truth samples (e.g. median grain-size, Chapters 2 and 3). Even if complex techniques are used to create high quality EGVs (e.g. multivariate geostatistics, Chapter 2), one has to be aware of the **limitations of the datasets**. In the case of models or interpolations, the reliability of the data is decreasing rapidly, at an increasing distance from the location of the ground-truth samples. This is demonstrated in Chapter 2 (Figure 2.12), where a map of the estimation variance of the kriging analysis gives an indication of the overall reliability of the interpolation. However, it gives more an indication of the sampling density than an absolute measure of reliability of the kriging estimate (Journel 1993; Armstrong 1994; and Goovaerts 1997). Still, this map shows where the data density is sufficient or not to obtain a reliable map of the grain-size. Furthermore, **error propagation**, resulting from the combination of erratic abiotic datasets, can further deteriorate the model. Datasets of **full coverage imagery** (e.g. multibeam, satellite imagery, lidar altimetry), are the **most reliable**, as they contain mostly directly measured data for each pixel of the dataset. When the data are available directly from the imagery (e.g. depth data or mathematically derived topographical coverages from multibeam), the reliability can be considered as high. Still, interpreted data from full coverage imagery (e.g. acoustic seabed classification, derived from multibeam backscatter in Van Lancker et al. 2007), will always include a measure of uncertainty. Finally, if relationships exist between coverages and species/communities, but if the coverages are unavailable (e.g. because of high costs) or the relationships are absolutely unknown, any habitat maps will be a flawed reflection of the actual situation.

Foster-Smith et al. (2007b) describe how the accuracy and confidence of marine habitat maps can be assessed. The **accuracy** of a habitat map is a mathematical measure of the **predictive power of a map** to predict correctly the habitat for a particular point (or pixel). By overlaying ground-truth data on the predicted habitat map, an error matrix can be made, by counting 'hits' and 'misses' and calculating accuracy indices (Foster-Smith et al. 2007b, for an overview). **Confidence** is more subjective than accuracy; it is an assessment of the **reliability of a map given its purpose**. Foster-Smith et al. (2007b) proposed a **multi-criteria approach** to determine the confidence one can have in habitat maps. This approach was implemented in the **MESH Confidence Tool**. Three main questions are posed:

- How good is the **ground-truthing**?
- How good is the **remote sensing**?
- How good is the **interpretation**?

The 3 groups correspond to the 3 first steps of the habitat mapping scheme (Figure 1.2), where remote sensing data correspond to coverage datasets. For each group, a number of questions are posed, resulting in a group score, expressed as a percentage. Questions are related to the techniques (appropriate or not for the habitat), coverage (e.g. tracklines spread widely or full-coverage), positioning (e.g. differential GPS or chart based navigation), standards applied (internal or internationally agreed), age of datasets, interpretation (e.g. is the interpretation method documented or not), level of detail of the resulting map and the map accuracy.

The tool is intended to evaluate habitat maps, and preferably for habitat classifications based on acoustical data and validated with biological samples. Still, for the marine landscape map and the HSMs of Chapters 4, 5 and 6, respectively, the tool can be applied as well (Table 7.1).

Table 7.1: Confidence assessment of the habitat maps of Chapters 4, 5 and 6 using the MESH Confidence Tool (Foster-Smith et al. 2007b). Scores are expressed as percentages (GT = ground-truthing; RS = remote sensing; IN = interpretation; BPNS = Belgian part of the North Sea; HSM = habitat suitability model; *O. fusiformis = Owenia fusiformis*).

Chapter	Description	GT score	RS score	IN score	Overall score
4	Marine landscapes (BPNS)	65	73	58	**66**
5	HSMs macrobenthic communities (BPNS)	65	73	92	**77**
6	HSM *O. fusiformis* (BPNS)	65	73	92	**77**

The marine landscape map has, relatively to the other habitat maps, the worst score of confidence (66 %). This is due mainly to the limited availability of samples to validate the marine landscapes (called 'interpretation' by the Confidence Tool). The HSMs of both the macrobenthic communities and *O. fusiformis* on the BPNS have the same score (77 %), because a comparable methodology on the same spatial scale was applied, integrating biological and abiotic data from the beginning. Still, the HSM of *O. fusiformis* is based on more abiotic variables than the HSM of the *A. alba* community (respectively 9 abiotic variables, representing the topographical nature; and 2 abiotic variables, representing the sedimentology only). As such, the first model can be considered as more reliable, but this is not accounted for in the Confidence Tool.

As demonstrated for the HSMs of *O. fusiformis* and the *A. alba* community, these scores have to be interpreted carefully, because of possible subjective assignments of the scores or of criteria that were not considered.

7.3 Comparison of habitat maps on the BPNS

7.3.1 Comparison of marine landscapes with habitat suitability models on the BPNS

The sedimentological maps of Chapters 2 and 3 and the habitat maps of Chapters 4, 5 and 6 are not fully comparable in the sense that the former were used as input for the latter. Still, the marine landscape map and the HSMs of both the macrobenthic communities and of *O. fusiformis* on the BPNS, can be overlain and this result can be evaluated.

Boxplots comparing the marine landscapes (Chapter 4, Figure 4.2) and the HSM of *O. fusiformis* (Chapter 6, Figure 6.3), show a high suitability for this species of landscapes 1 and 2, whereas landscapes 3, 4 and 8 are less suitable (Figure 7.2). The remaining landscapes are unsuitable.
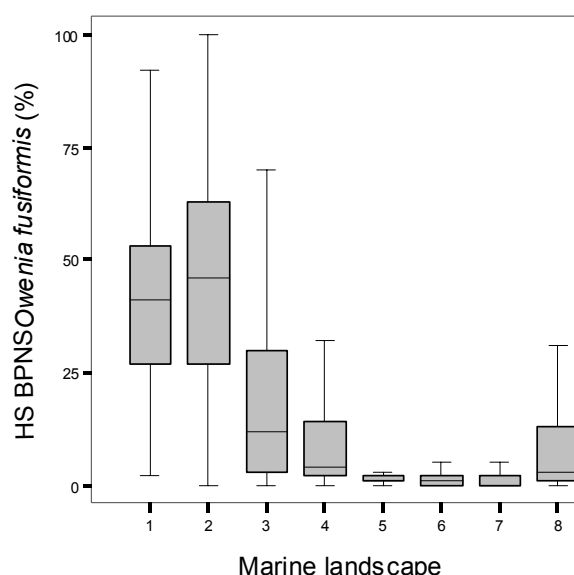

Figure 7.2: Boxplots of HSM of *Owenia fusiformis* on the BPNS, overlain on the marine landscape map, showing that landscapes 1 and 2 have the highest suitability for *O. fusiformis*. The middle line in the box is the median, the lower and upper box boundaries mark the first and third quantile. The whiskers are the vertical lines ending in horizontal lines at the largest and smallest observed values that are not statistical outliers (values more than 1.5 interquantile range).

Landscapes 1 and 2 correspond to "shallow, high silt-clay percentage, high current velocity, high bottom shear stress, turbid, high Chl a concentration" and "shallow NW orientated flats and depressions, fine sand, slightly turbid, high Chl a concentration", respectively. This corroborates the habitat preferences of *O. fusiformis* as presented in Chapter 6, although a moderately high silt-clay %, a moderate maximum bottom shear stress and moderate maximum current velocities were now added as well. Landscapes 3, 4 and 8, respectively, correspond with "shallow SE orientated sandbanks, fine to medium sand, slightly turbid, high Chl a concentration", "deep NW orientated flats and depressions, medium sand" and "high percentage of gravel – shell fragments".

When the same exercise is done for the habitat suitabilities of the macrobenthic communities (Chapter 5, Figure 5.3), overlain on the marine landscapes (Figure 7.3), the following observations can be made:

(1) The *Macoma balthica* community has a pronounced preference for landscape 1; this is logical, as this species was known already to occur mainly in fine sediments with a high silt-clay %; all other communities show much more variation;

(2) The *A. alba* community shows highest suitabilities in landscapes 2 and 3, corresponding more or less with the preference of *O. fusiformis* (Figure 7.2); still, the *A. alba* community shows more variation in the other landscapes;

(3) The *Nephtys cirrosa* community has similar preferences for all of the landscapes, except for landscape 1, meaning that this community has a very broad niche and that its habitat is not at all well defined; and

(4) The *Ophelia limacina* community has a preference for the combination of landscape 4 ("Deep NW orientated flats and depressions, medium sand"), 5 ("Deep SE orientated flats and depressions, medium sand"), 6 ("Crests of sandbanks, medium sand"), 7 ("Slopes of sandbanks, medium sand") and 8 ("High percentage of gravel – shell fragments"). Still, the preferences are very well pronounced.
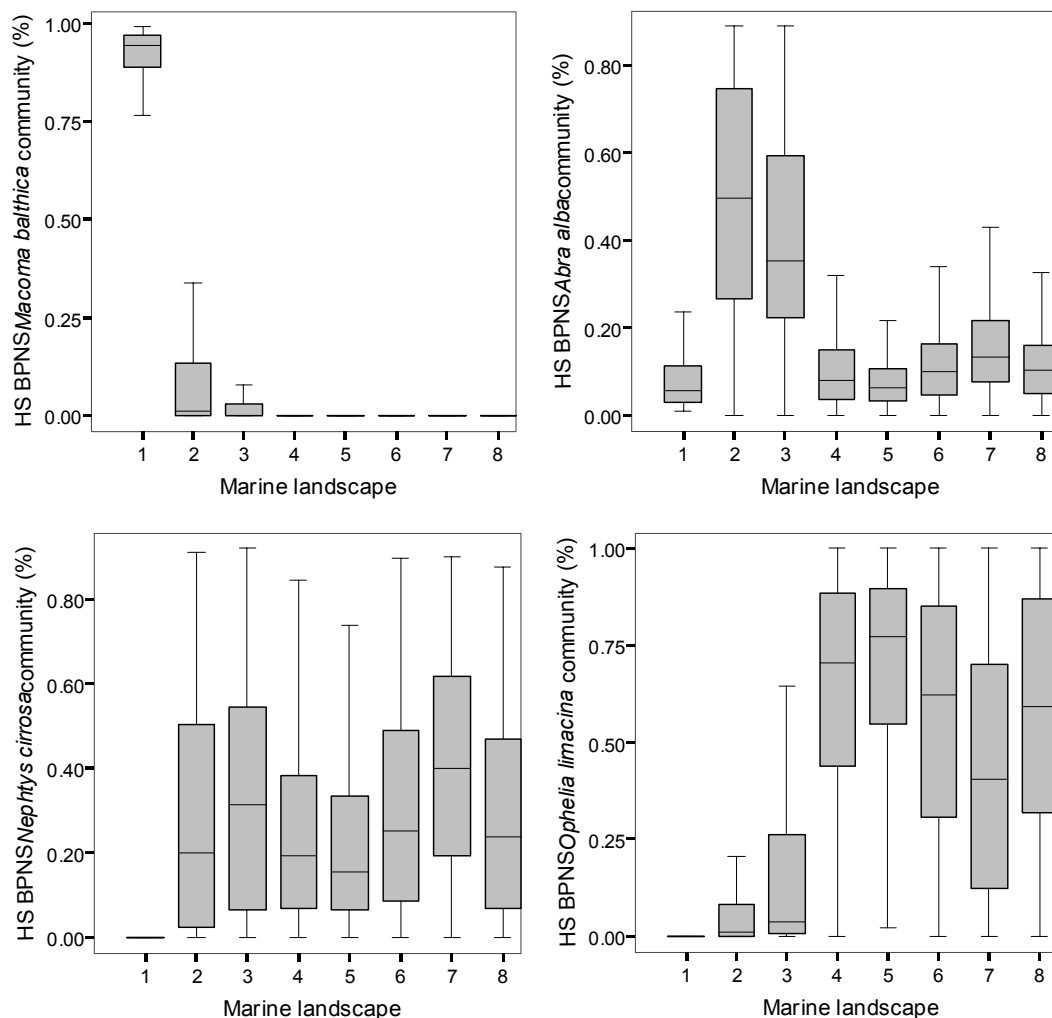

Figure 7.3: Boxplots of HSM of the 4 macrobenthic communities on the BPNS, overlain on the marine landscape map.

As such, there is a discrepancy between the number of macrobenthic communities (4) and the number of marine landscapes (8). However, as only 4 communities have been defined on the BPNS, it is possible that more communities or gradations between communities are present. As discussed in Section 7.1, marine landscape mapping can reveal other ecologically interesting areas that are still unknown today or that are difficult to sample (e.g. landscape 8, representing a 'High percentage of gravel – shell fragments').

7.4 <u>Habitat maps to support a sustainable management of the marine environment</u>

Generally, habitat maps are produced by scientists. Their translation towards simple maps, of use for all stakeholders, is often a difficult task. Still, habitat maps are a crucial tool in the context of a **sustainable management of the marine environment**. In particular, they are useful for:
- the designation of **Marine Protected Areas** (MPAs) (e.g. improved knowledge on valuable habitats with a high biodiversity);
- setting-up **baselines** for future impacts of new anthropogenic activities (e.g. windmill parks);
- **marine spatial planning** (e.g. where to avoid certain activities);
- setting-up new **monitoring studies** (e.g. indication of possible interesting habitats);
- **informing the stakeholders** (e.g. fishermen, aggregate extraction industry, coastal managers, inhabitants of coastal villages…);
- the implementation of **European Directives** (e.g. new proposals for Habitats Directive areas).

For stakeholders, it can be difficult to interpret habitat maps correctly. Generally, they have a preference for classified habitat maps with clear boundaries between valuable and less valuable areas, although this does not correspond with the real situation. It should be the task of scientists to educate stakeholders how to translate or interpret the results, whether or not classified habitat maps or maps with gradual scales are available.

A first attempt was made to translate the 4 HSMs of the macrobenthic communities (Chapter 5) to a EUNIS classification (Schelfaut et al. 2007), being a pan-European classification system of habitats (EUNIS 2002). However, only 1 of the classes corresponded with an existing EUNIS class, while 3 classes could not be correlated. This is the case for many countries (e.g. France) and pleads for an overall updating of the EUNIS classification system (Foster-Smith et al. 2007a).

As described in the legal characterization of the BPNS (Section 1.2.6), there are **5** legally designated Belgian **MPAs**: 3 Special Protection Areas (SPAs), in the framework of the Birds Directive; 1 Special Area of Conservation (SAC), in the framework of the Habitats Directive; and 1 specific marine nature reserve (Figure 1.12). Initially a second SAC was designated on the Vlakte van de Raan (SAC2), but this has been cancelled in February 2008, because of a complaint of the energy company Electrabel.

The designation of the SPAs on the BPNS, is based on selected important bird areas (Haelters et al. 2004). The designation of SAC1 and the former SAC2 is based on the

consultation of experts on different ecosystem components (macro- and epibenthos, fish, seabirds and sea mammals) and the investigation of habitat data (Derous et al. subm. b). The main issue remains the estimation of the biological value of certain areas which is based ideally on all marine ecosystem components and taking into account a number of criteria for their valuation (Derous et al. 2007). For the BPNS, Derous et al. (subm. a) proposed a **biological valuation map** (BVM), based on datasets of macro-, hyper- and epibenthos, demersal fish and avifauna. The HSMs of the 4 macrobenthic communities (Chapter 5; Figure 5.3), served as a major input for the macrobenthos of the BVM, next to point data on densities and species richness. Derous et al. (subm. b) compared the high valuable zones of the Belgian BVM (Derous et al. subm. a) with the Belgian SACs and SPAs of the European Directives and found a good overlap between both, although some valuable areas of the BVM, were not covered by the SACs and SPAs. Derous et al. (subm. b) recommends proposing other priority areas under the Habitats Directive: e.g. the area at the northern side of the former SAC2 (Vlakte van de Raan) (Introduction; Figure 1.12) or the sandbank complex of the Thorntonbank (Introduction; Figure 1.3).

The HSM of *O. fusiformis* on the BPNS (Chapter 6; Figure 6.3), being an important indicator species of the ecologically important *A. alba* community (Van Hoey 2004), confirms the importance of the area at the **northern side of the Vlakte van de Raan** as a **possible priority area**.

The marine landscapes map (Chapter 4; Figure 4.2) shows a clear distinction between the coastal and the more offshore area (with landscape 2 and 3 as most interesting for the *A. alba* community). Moreover, sandbanks, swales, slopes and gravel areas are distinguished, indicating possible interesting habitats that should be examined to discover their ecological relevance. Especially the potential gravel habitat should be explored further (see Section 7.1). **Possible priority areas** could include the area in the swale at the **SE side of the Westhinder sandbank** or the coarser sediment patch in the swale at the **SE side of the Noordhinder sandbank** (Chapter 4; Figure 4.2; cluster 8). Another potential area is located in the swale between the Goote- and Thorntonbank. In Van Lancker et al. (2007), the distribution of possible gravel areas (Introduction; Figure 1.8), was even more extensive than on the marine landscapes map. This is due to the fact that the gravel input layer of the marine landscapes map was based purely on quantitative sample data, whilst the potential gravel area of Van Lancker et al. (2007) is based additionally on geological, multibeam backscatter and diving information. Based on Van Lancker et al. (2007), gravel is expected in the **entire swale system of the southern part of the Hinder Banks**. However, it must be stressed that the gravel on the BPNS is not a continuous layer; it occurs in patches mainly and in many cases the gravel is covered with a sand veneer (Van Lancker et al. 2007). Especially in areas where sand dynamics are high, species richness and diversity decreases and mostly only the *O. limacina* community might be present (Van Lancker et al. 2007). This community is indeed typical for medium to coarse sediments and is often associated with gravel and shell fragments. In Figure 7.4, the **HSM of the *A. alba* community** is combined with the potential gravel areas of the marine landscapes map and of Van Lancker et al. (2007). As the HSM of the *A. alba* community is similar to that of *O. fusiformis*, only the former is presented on the map to avoid confusion. Moreover, *O. fusiformis* is an important indicator species of the *A. alba* community (Van Hoey et al. 2004). The whole zone of high suitability of the *A. alba* community (being the dark orange zone), combined with the potential gravel areas can be considered as potentially valuable seabed habitats.
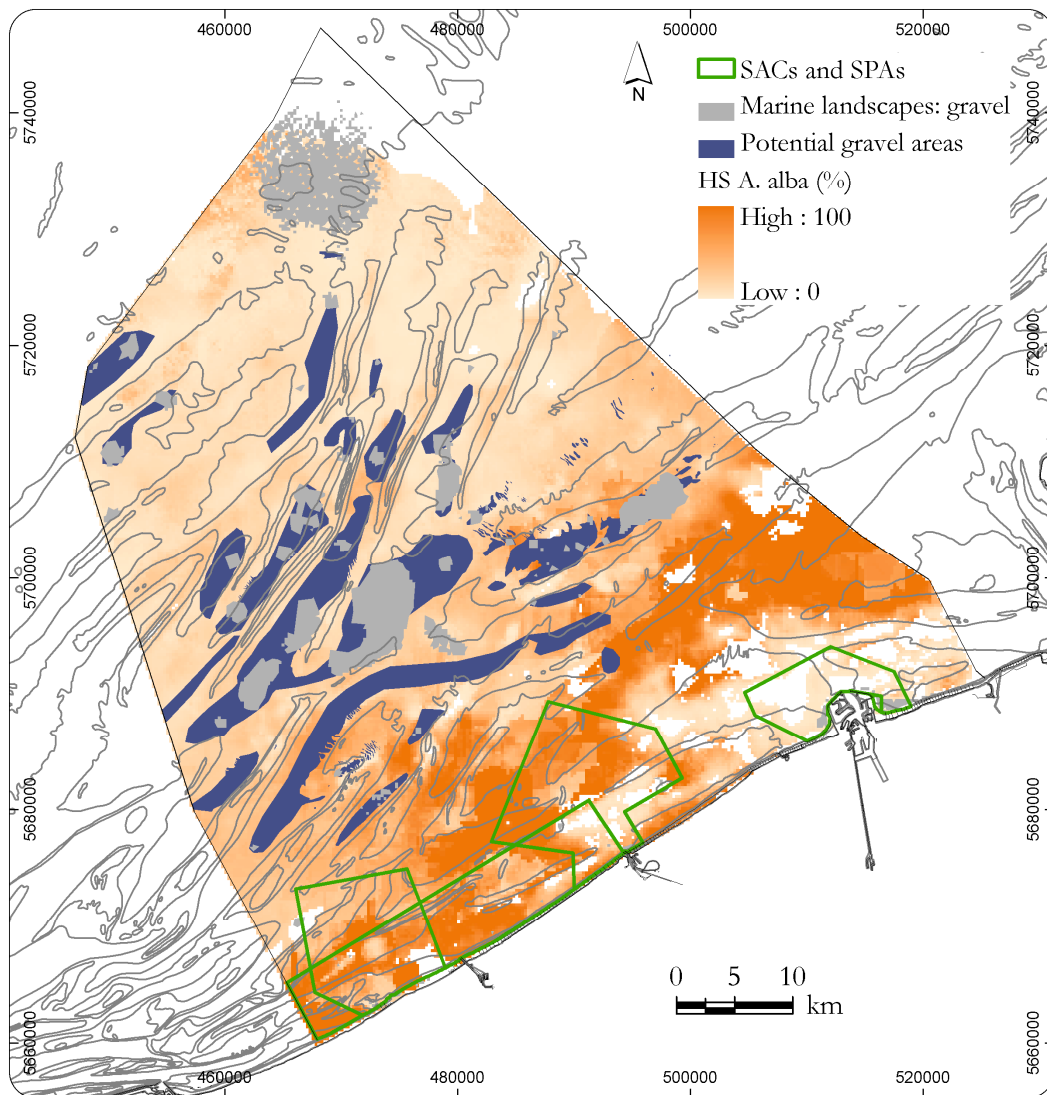
Figure 7.4: Combination of the habitat suitability model of the *A. alba* community, gravel patches from the marine landscapes map and potential gravel areas from Van Lancker et al. (2007).
Based on this map, the whole dark orange area, with the possible gravel areas are potentially valuable seabed habitats. SACs = Special Areas of Conservation; SPAs = Special Protection Areas; HS = habitat suitability; *A. alba = Abra alba*.

However, nature conservation is not the only 'user' of the BPNS; as such the **anthropogenic activities** on the BPNS are overlain on the same datasets (Figure 7.5). If all anthropogenic activities are left unchanged, **little space** is left **for nature conservation**. Still, it must be clear, that this space is not constantly occupied (e.g. the use intensity of military exercises is for most of the designated zones less than 11 exercise days/year/km² (Maes et al. 2005)) and multiple occupation of already designated zones could be considered.
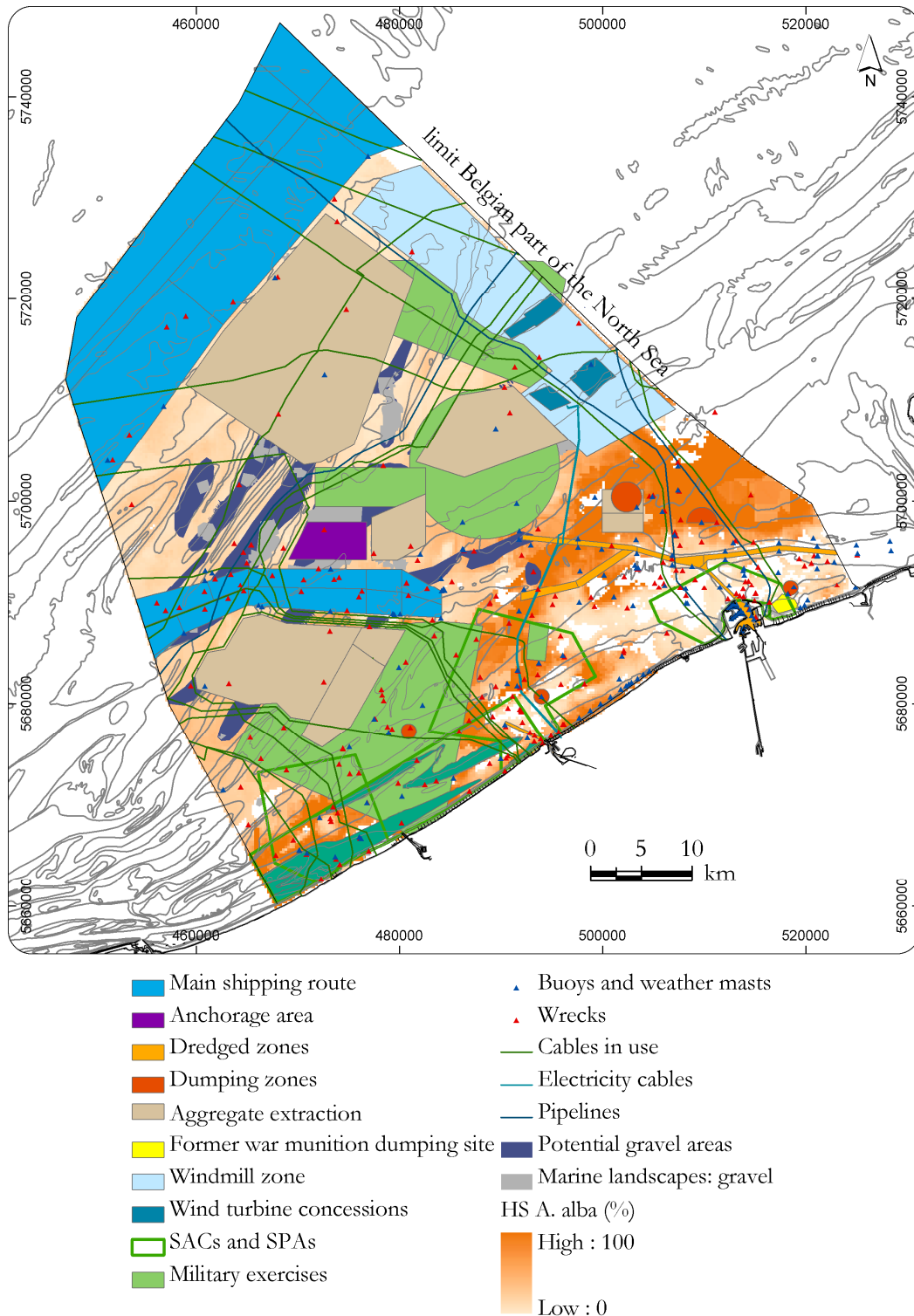
Figure 7.5: Overlay of anthropogenic activities on the habitat suitability model of the *A. alba* community, gravel patches from the marine landscapes map and potential gravel areas from Van Lancker et al. (2007). SACs = Special Areas of Conservation; SPAs = Special Protection Areas; HS = habitat suitability; *A. alba* = *Abra alba*.

In Figure 7.6, all areas of highest suitabilities of the *A. alba* community and possible gravel areas, that are not overlapping with anthropogenic activities (except for cables, pipelines, wrecks, buoys and weather masts), are marked as potentially valuable

seabed habitats. For zones 1 until 3, the ecological relevance has been shown in Chapter 5 for the *A. alba* community and in Chapter 6 for *O. fusiformis* (although the highest suitability zones are somewhat less extensive for *O. fusiformis* than for the *A. alba* community). The areas 4 until 11 are based purely on abiotic knowledge from the marine landscapes (Chapter 4) and the potential gravel patches from Van Lancker et al. (2007). Still, comparing these areas with the species richness data from Houziaux et al. (2007b), it is observed that areas 5, 9, 10 and 11 coincide well with their zones of largest taxonomic breadth (Figure 7.7).
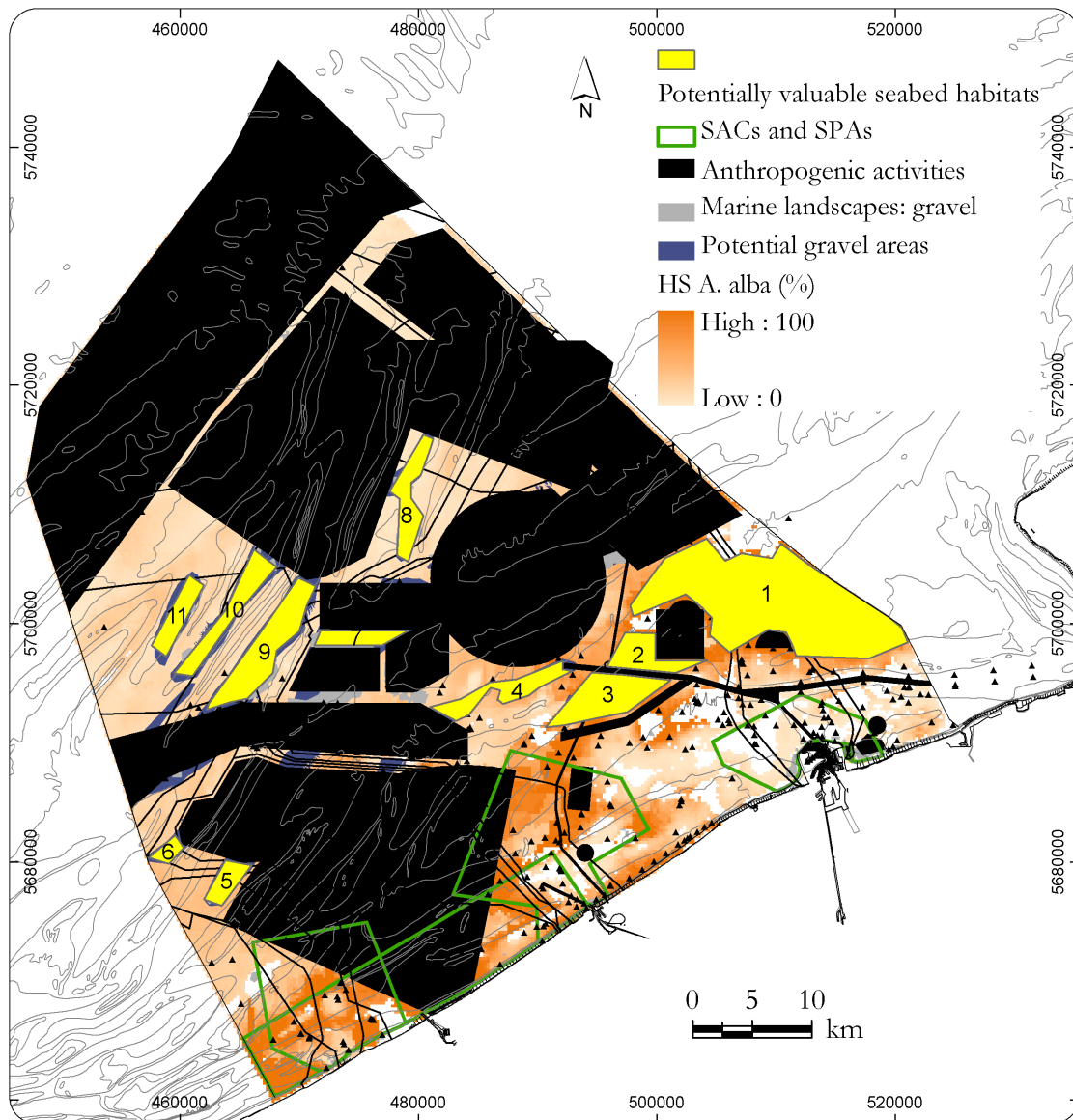

Figure 7.6: Potentially valuable seabed habitats,
not overlapping with anthropogenic activities and in addition to the already designated SACs and SPAs. The zones correspond to the highest suitabilities of the *A. alba* community (zone 1, 2 and 3) and the gravel patches from the marine landscapes map and from Van Lancker et al. (2007) (zone 4, 5, 6, 7, 8, 9, 10 and 11). SACs = Special Areas of Conservation; SPAs = Special Protection Areas; HS = habitat suitability; *A. alba* = *Abra alba*.
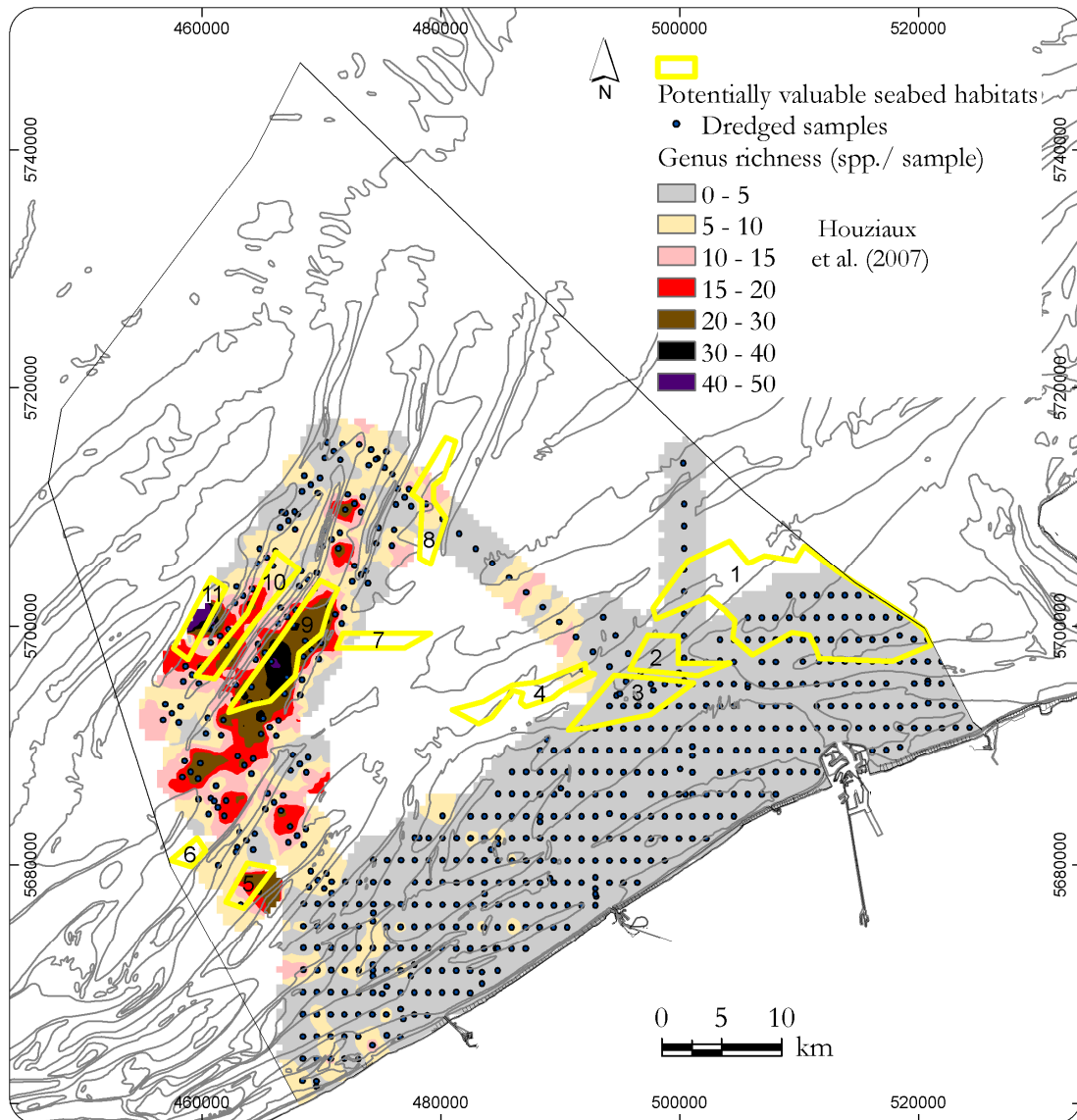
Figure 7.7: Potentially valuable seabed habitats overlain on the species richness map on genus level of epibenthos (excluding infauna), based on historical datasets of Gilson from the beginning of the 20th century (Houziaux et al. 2007b).

Table 7.2 is a summary of the potentially valuable seabed habitats, with their location on the BPNS, their ecological relevance (based on macrobenthos HSMs from Chapter 5 and 6; and epibenthos data excluding infauna, from Houziaux et al. (2007b)) together with possible conflicts with anthropogenic activities. The ecological relevance of avifauna or sea mammals is not taken into account.

Haelters et al. (2007) proposed an OSPAR[19] MPA, called the 'Westhinder MPA'. For this delimitation, historical information about the benthic biodiversity and the extent of former oyster beds in the 19th century was used (Lanszweert 1868; and Houziaux et al. 2007b), together with recent information on biodiversity (Houziaux et al. 2007b) and the presence of gravel (Van Lancker et al. 2007). The area follows the southern slope of the Westhinder sandbank and the northern slope of the Oostdijck sandbank. For the northeastern and western delimitation, the position of the Westhinder anchorage area and the position of some buoys has been used (Haelters et al. 2007).

Figure 7.8 represents the valuable seabed habitats as a result from this research together with their ecological relevance. From this, there are 5 zones with a high ecological relevance: zone 1, 2, 3, 9 and 11.

As both zones 2 and 3 are situated in a highly important area for navigation, they are less suitable. For zone 1, the biodiversity of the **macrobenthos** is high to very high, as also for birds (Derous et al. subm. a). The delineation of zone 2 and 3 is based on the results of a **predictive model** of habitat suitability of the *A. alba* community. With any model (also demonstrated in 7.2), it is clear that the predictions need sound validation in terms of ground-truthing.

Regarding the **species richness of epibenthos**, **zones 9 and 11** seem most valuable. Zone 9 corresponds mostly with the proposed 'Westhinder MPA' of Haelters et al. (2007).

Table 7.2: Potentially valuable seabed habitats, with their location; ecological relevance (limited to macrobenthos), with a scale ranging from low to high (+ to +++) with reference; and possible conflicts with anthropogenic activities (anthropogenic activities based on Maes et al. 2005).

Zone	Location	Ecological relevance		Possible conflicts
1	N & central part VVR, E end AB	+++	Chapter 5, 6	Fisheries
2	NW part VVR, SE part AB	+++	Chapter 5, 6	Fisheries
3	S of central AB	+++	Chapter 5, 6	Shipping, fisheries
4	W & central part AB	?	/	Shipping, fisheries
5	Swale OD - BR	++	Houziaux et al. 2007b	/
6	Swale OD - BeB	?	/	/
7	Swale OH - GB	?	/	Fisheries (minor part)
8	Swale BB - OH	+	Houziaux et al. 2007b	Fisheries (partly)
9	Swale OH - WH	+++	Houziaux et al. 2007b	Fisheries (partly)
10	Swale WH - NH	++	Houziaux et al. 2007b	Fisheries (partly)
11	Swale NH - FB	+++	Houziaux et al. 2007b	Fisheries (partly)

VVR = Vlakte van de Raan; AB = Akkaertbank; OD = Oostdijck; BR = Buiten Ratel; BeB = Berguesbank; OH = Oosthinder; GB = Gootebank; BB = Bligh Bank; WH = Westhinder; NH = Noordhinder; FB = Fairybank; ? = unknown ecological relevance.
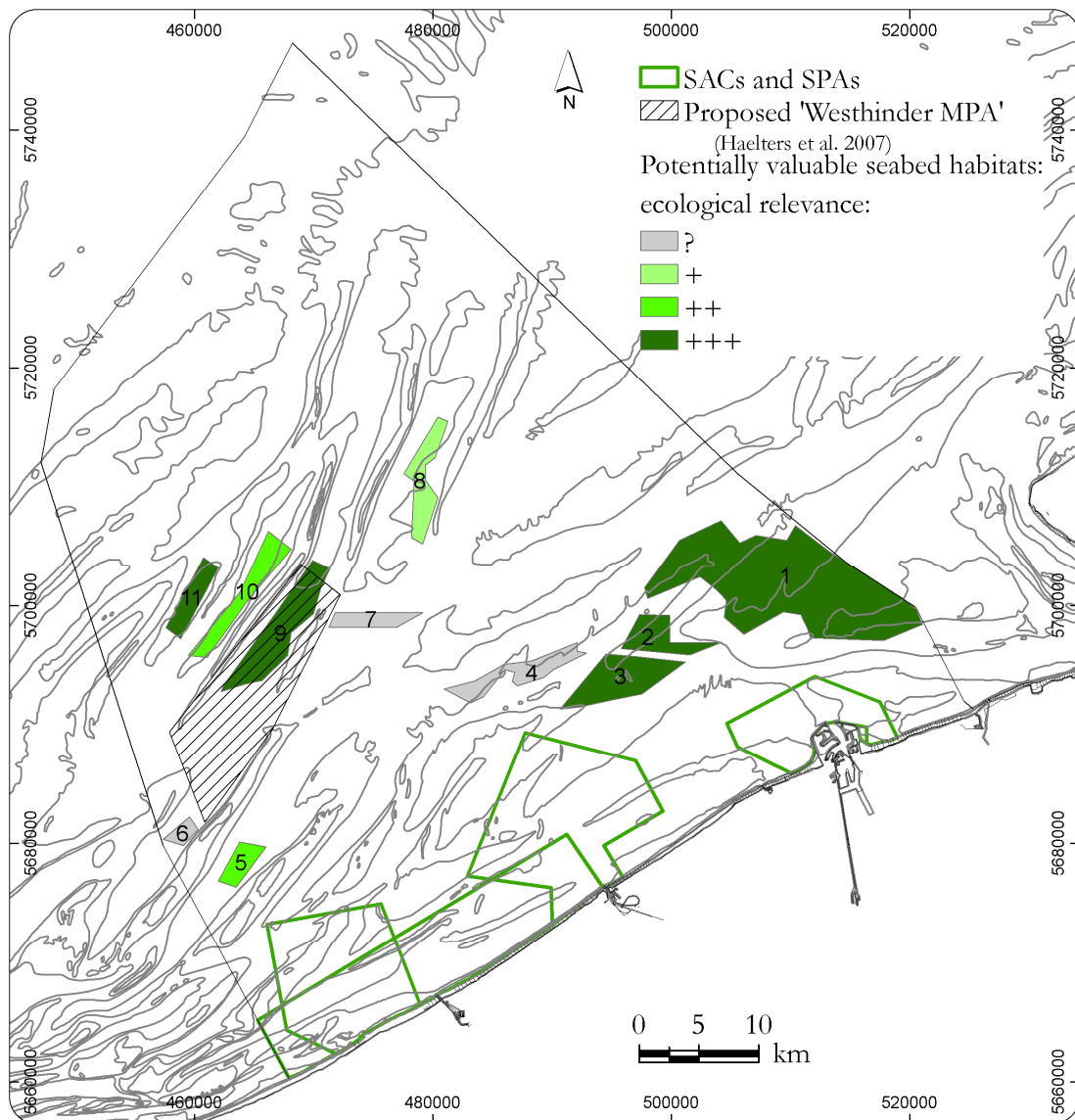
Figure 7.8: Potentially valuable seabed habitats, with their ecological relevance (scale ranges from unknown (?) to high (+++)) (Table 7.2); existing Special Areas of Conservation (SACs) and Special Protection Areas (SPAs); and proposed 'Westhinder MPA' of Haelters et al. (2007).

Finally, it is hoped that this research will contribute to the future implementation of the **EU Marine Strategy Directive**. This Directive aims at both the implementation of an ecosystem-based approach in marine waters and a sustainable use of marine goods and services (CEC 2005). According to Annex II of this Directive, as many biological, physical and chemical characteristics should be included in the assessment of the environmental status of the sea. Habitat maps, at various spatial and temporal scales, and the integration of various datasets will likely be needed to improve management practices. In addition, the techniques and approaches, presented in this thesis, can assist in the prediction of ecologically valuable areas in the marine environment.

Chapter 8

Conclusion

8 Conclusion

As formulated in the research objectives, the general objective of this thesis was to develop **spatial distribution models** as **input for habitat mapping**. Results focussed on the creation of: **abiotic coverages** and **habitat maps**, the latter being the result of the **integration of ground-truth data and the abiotic coverages**. The methodologies were set-up, following an **objective** and **(geo)statistically sound** approach, with emphasis on **intercomparison and validation.**

More specific research objectives were the following:
    ii. Proposing a **new approach for marine landscape mapping**, being simple, statistically sound and easy applicable to other regions. The proposed methodology should be a step forward in standardising marine landscape mapping throughout Europe.
    iii. Developing methodologies that are straightforward, **objective** and **statistically sound**.
    iv. Using a **maximum input of ecogeographical variables** (EGVs), for the modelling of the marine landscapes and habitat suitability modelling (HSM) and as secondary variables for multivariate geostatistics.
    v. **Optimising the reliability of EGVs** by applying multivariate geostatistics for the sedimentological maps and **optimising the reliability of habitat maps** by applying statistically sound methodologies; **validating** the results obtained by the (geo)statistical methodologies.
    vi. Deriving **multi-scale topographical EGVs** from the bathymetry, for both the modelling of sedimentological maps and for the modelling of the HSMs of the species *Owenia fusiformis*.

As explained in the Introduction, the habitat mapping process can be subdivided into 4 steps:
    1) Getting the best out of ground-truth data;
    2) Selecting and getting the best out of coverage data;
    3) Integration of ground-truth and coverage data;
    4) Habitat map design and layout.

Main emphasis of the research was on Step 2 and Step 3 of the habitat mapping process; as such the thesis was subdivided into 2 themes, corresponding with:
    1) Best coverages for habitat mapping;
    2) Integration of datasets in the view of habitat mapping.

Theme 1: Best coverage data for habitat mapping

In the first theme, emphasis was put upon the **application of multivariate geostatistical techniques**; these are used increasingly in certain research domains (e.g. soil science and climatology), but are still unexplored in marine science. To obtain reliable sedimentological maps of scattered ground-truth data, it is crucial to apply the most suitable interpolation techniques. If there is a **linear relation** between the **sedimentological variable** and one or more **secondary variables**, multivariate

geostatistics can be applied. **Kriging with an external drift** or KED (Goovaerts 1997) is an example of such a technique.

KED was applied successfully on sedimentological data of 2 study areas: 1) the Belgian part of the North Sea (BPNS) in Chapter 2; and 2) a small study area at the southeastern part of the BPNS in Chapter 3. For the first study area, the median grain-size of the sand fraction (d_s50) was interpolated with only one secondary variable, the bathymetry. For the second study area, a whole set of secondary variables was used for the interpolation of the d_s10 (10th percentile of the sand fraction), d_s50, d_s90 (90th percentile of the sand fraction) and silt-clay% (fraction below 63 μm). These secondary variables comprised the bathymetry and its multi-scale derivatives (e.g. slope, curvature, fractal dimension on different spatial scales).

Both results were **compared** and **validated** against results obtained by linear regression (Chapter 2) and Ordinary Kriging (OK) (Chapter 2 and 3). Validation indices revealed that all results, based on KED, gave significantly better results (except for the map of the silt-clay%, which correlated very weakly with the secondary variables).

The result for both study areas were **highly reliable sedimentological maps**, reflecting well the natural morphology of the seabed. Moreover, the followed methodology was **objective**, **straightforward** and **scientifically sound**.

Theme 2: Integration of datasets in the view of habitat mapping

The second part of the thesis comprised the **integration of abiotic datasets with biological data.** In this theme, the main aim was to apply **statistical methods** to obtain **highly reliable habitat maps**.

In Chapter 4, marine landscapes were modelled, by integrating a whole set of abiotic variables. Therefore, **principal components analysis** (PCA) was used to reduce the number of abiotic variables to a set of non-correlating components. Moreover, a **cluster analysis** was used to cluster these components to meaningful marine landscapes. To determine the number of landscapes, the Calinski and Harabasz (1974) index was used, designated by Milligan and Cooper (1985) as giving the best results out of a number of indices. Only at the end of the process, biological data were used to test the ecological meaning of the marine landscapes. This was confirmed for the marine landscapes of the BPNS.

HSMs were produced in Chapter 5 and 6. Therefore, two different statistical modelling techniques were applied: **Discriminant Function Analysis** (DFA) and **Ecological Niche Factor Analysis** (ENFA).

DFA was applied in Chapter 5 for the modelling of the 4 macrobenthic communities on the BPNS (Van Hoey et al. 2004). Moreover, a three-fold **cross-validation** showed that the agreement between the model predictions and observations was very good and consistent.

In Chapter 6, ENFA was applied for the modelling of the species *Owenia fusiformis*, on the BPNS. As this species is an important indicator species of the ecologically important *A. alba* community (Van Hoey et al. 2004), the HSM of the entire BPNS corresponded, in essence, with the HSM of the *A. alba* community, although subtle differences were observed. A cross-validation was used to select the best model out of a set of models, based on different modelling algorithms and different subsets of EGVs.

In the Discussion (Chapter 7), results of this research were compared, overlain and integrated.

The **reliability of the habitat maps** was calculated following a multi-criteria approach (Foster-Smith et al. 2007b). The reliability of all of the habitat maps (Chapter 4, 5 and 6) ranged between 66 and 77 %, with the lowest score for the marine landscapes map and the higher scores for the HSMs on the scale of the BPNS.

The methodologies of Chapter 2 (multivariate geostatistics) and Chapter 5 (Discriminant Function Analysis) were tested successfully on the **Southern North Sea** to obtain maps of the median grain-size, silt-clay% and HSMs of the 4 macrobenthic communities.

Based on the HSM of the ecologically important *A. alba* community (Chapter 5), combined with the potential gravel areas from the marine landscapes (Chapter 4) and from Van Lancker et al. (2007), propositions were made for the **designation of potentially valuable seabed habitats** on the BPNS.