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Enabling Delta Life

Using habitat classification systems to assess impacts on ecosystems

Validation of the ZES.1 for the Westerschelde

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Summary
Voor het onderzoeksprogramma LTV O&M (Lange Termijn Visie Onderzoek en Monitoring) in de Westerschelde, is een validatie van het Zoutewateren Ecotopen Stelsel (ZES.1), uitgevoerd. Deze validatie heeft op twee verschillende niveaus plaats gevonden. Als eerste is de hiërarchische volgorde van de abiotische parameters, zoals gebruikt in het ZES.1, onderzocht met behulp van een CCA. Hiervoor is gebruik gemaakt van benthos data en abiotische data. De laatste is in sommige gevallen gemeten in het veld en in andere gevallen voorspeld met behulp van modellen. Vervolgens zijn mogelijke splitsingswaarden tussen klassen verkend door voor elke abiotische variabele de meest optimale splitsingwaarde te bepalen.

Het belang van de verschillende abiotische variabelen hangt samen met de schaal waarop wordt gekeken. In deze studie wordt op estuarium schaal gekeken en dan blijkt zoutgehalte de meest bepalende factor te zijn bij het verklaren van de soortensamenstelling. Zout wordt op de voet gevolgd door stroomsnelheid. Slibgehalte en diepte blijken minder gewicht in de schaal te leggen. Dit komt wat betreft het zoutgehalte goed overeen met eerdere studies. Stroomsnelheid wordt in andere onderzoeken niet altijd meegenomen als potentieel structurerende factor. Dit heeft tot gevolg dat diepte een belangrijke factor wordt, omdat diepte en stroomsnelheid nauw met elkaar verbonden zijn. Zo zijn de stroomsnelheden vaak het hoogst in de diepe geulen. De waarden van de abiotische parameters die nu worden gebruikt voor het onderscheiden van verschillende ectopen zijn in dit rapport voor het eerst kwantitatief getest. Hieruit komen waarden die meestal redelijk in de buurt liggen bij de waarden zoals die momenteel in het ZES.1 staan. Echter, omdat de waarden wel afwijken wordt aanbevolen om met de nieuwere abiotische en biotische data (in 2008 zijn ecotopenkaarten gemaakt) nogmaals dezelfde analyse uit te voeren. Pas daarna zou besloten kunnen worden om voor de Westerschelde deze waarden aan te passen.

References

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1 Introduction

1.1 Using habitat classification systems to assess impact on ecosystems

Human pressures on coastal, estuarine and marine environments are innumerable. However, it is of vital importance to maintain healthy and productive ecosystems in these areas, to preserve ecosystem services. For example, estuaries play an important role in water management and safety of the population in delta areas largely depends on the proper functioning of an estuary for water storage. Further, water quality and the linked quality of consumable resources, such as fish and shellfish, can only be maintained within healthy, functioning estuaries and seas. To manage and protect these systems during times of continuous economic pressure asks for clear goals and measurable targets. Today, this constitutes one of the hardest exercises for ecologists, as it has proven extremely difficult to quantify impacts of measures on ecosystems.

A way to deal with increasing demand for measurable units to asses impacts on ecosystems is by making use of habitat classification systems. Using habitat classification is becoming increasingly popular across the world now that data is available on larger spatial scales due to more advanced mapping techniques. For example, satellite imaging allows for extensive mapping of terrestrial habitats, whereas various sonar techniques result in extensive submarine habitat information. Several countries make use of their own classification schemes, such as Britain/Ireland (Connor et al. 2004) and France (Dauvin et al. 1994). Also, in the US several distinct habitat classification schemes for marine and coastal habitats are applied (NOAA 2000). Within Europe there is a single habitat classification scheme, developed by the European Environment Agency (EEA) as part of its EUNIS system (**EU**ropean **N**ature **I**nformation **S**ystem). This system encompasses terrestrial, fresh water and marine habitats, is hierarchical and contains six levels. The marine system differentiates between zones first (littoral, infralittoral, circalittoral etc.). Secondly, it divides between substrate types, and then, hydrodynamic energy, environmental variables (salinity) and characterising species. The Netherlands uses its own classification system for salt water habitats (Bouma et al. 2005). This system resembles the EUNIS systems to a certain extent as it used the same discriminatory criteria, However, the system is used specifically for more shallow water systems, which are prevalent in the Netherlands and for estuaries. Therefore, criteria such as salinity and depth are emphasized more.

1.2 Habitat classification with the Dutch classification system in the Westerschelde

The Westerschelde (the Netherlands) is a dynamic area with many different ecosystem functions, such as protection against flooding, accessibility of the present harbors, and maintenance of a healthy and dynamic ecosystem. To maintain quality and function of the Schelde estuary the research program LTV O&M (Long-Term Visions research and Monitoring) was started in 2003. The program consists of the themes “safety”, “accessibility” and “natural development” and tries to incorporate all these different interests into a long-term vision.

Integration of these conflicting functions asks for adequate tools to evaluate effects of management. Currently, habitat maps are often used to give insight into effects of human pressures on distribution of natural habitat. These maps are useful tools to answer policy related questions, such as evaluation of changes in management. The habitat maps are produced by making use of Dutch system for classification of ecotopes in salt water systems (ZES.1). The ZES.1 uses the term ‘ecotope’, but here we will use the term ‘habitat’, which is .

more often used in a international context. The ZES.1 is a hierarchical classification of abiotic parameters, which classify areas as belonging to distinct habitats (Bouma et al. 2005). A habitat (or ecotope) is defined as a spatially limited ecological unit. Generally, characteristics of habitat are determined by both abiotic, biotic and anthropogenic factors. In the ZES.1 the main determinant in appointing habitats is abiotic conditions (Figure 1).

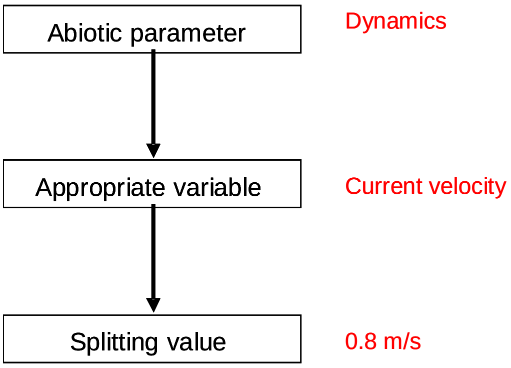

Figure 1.1 Schematic representation of different levels of accuracy of abiotic parameters in ZES.

The abiotic parameters that constitute the ZES.1 are: salinity, substrate, depth, hydrodynamics, and then again depth and substrate, but into more detail. The first substrate parameter divides habitats into hard and soft substrate, the second encompasses sediment grain size dividing between sand and silt. Concerning depth, first habitats are roughly divided by intertidal, supratidal and subtidal and second divisions between high, mid and low intertidal are suggested. For each abiotic parameter, certain values were determined that distinguish between distinct habitats. These splitting values could potentially differ between different water bodies.

The hierarchy of parameters and their splitting values that constitute the Dutch habitat system for salt waters, are merely based on expert judgment, meaning that the habitat system itself was never validated based on field data. Maps that are produced making use of the ZES.1 are frequently validated using macrobenthic monitoring data (Bagelaar et al. 2006), gathered during yearly monitoring programs. This data can be utilized for validation of the habitat system as well. Previous studies that were part of the LTV O&M Natural Development program, investigated ways to validate this system (Van Wesenbeeck 2007), the availability of data for validation (Troost 2007) and robustness of measured abiotic parameters (Wijnhoven, Herman et al. 2007). In the present study we make a first attempt at validation of the habitat system, using macrobenthos monitoring data.

Although the ZES.1 was used in the Wadden Sea (Wijsman & Verhage 2004), most applications of the ZES.1 are in the Westerschelde (Bagelaar et al. 2006, Wijnhoven et al. 2006).The regional character of the present study also limits validation of the ZES.1 to the Westerschelde.

1.3 Approach

In this study it is examined whether the habitat classification system (ZES.1) provide a useful tool for assessing effects of human pressures versus natural processes. To answer this question a validation based on field data, of the ZES.1 needs to be executed. During this validation several questions are leading:

1. Which environmental variables, and in what order, influence benthic species composition?

2. Is this in agreement with the variables and their order used in the ZES.1?
3. Can optimal values for abiotic conditions to divide data into separate groups (or habitats) be quantitatively determined?
4. Are these splitting values in agreement with values used in the ZES.1?

To answer these questions the ZES.1 is validated on two different levels. First, the hierarchical order of abiotic parameters is investigated. Second, to obtain more information on possible splitting values between different habitats, the distribution of species is examined as a function of each separate abiotic parameter. It is important to realize that analyses in this study are done independently of the existing classification system and that it is not the aim to produce habitat maps. Only macrobenthic data of soft bottom communities linked with data of abiotic conditions are used to derive an independent evaluation of structuring effects of abiotics on species composition in the Westerschelde.

2 Methods

2.1 Biotic and abiotic data

Benthic data is collected by the Centre for Estuarine and Marine Ecology (NIOO-CEME), in permanent monitoring programs, which are instructed by the Directorate for Public Works and Water Management (RWS). The monitoring database contains samples from several waters in the Netherlands from 1978 onwards, but for this study only data from the Westerschelde in the years 2003, 2004 and 2005 is used.

Sampling in the field is random within different depth strata and is performed using a box-corer. From each box-core three cores of 8 cm are sieved on a 1 mm sieve. Samples are further selected and determined onto a species level in the lab. For more details on the origin of biotic data see Appendix A.

Analyses in this project require coupled biotic and abiotic data. So, benthos data are coupled with values for environmental conditions. Monitored environmental conditions are sediment characteristics and depth. With each benthos sample a sediment sample is analyzed using a Malvern Mastersizer. From this analysis D50 and the percentage silt smaller than 63 mu is used in further analyses. Depth is measured while sampling from the boat. However, depth can also be derived from bathymetry maps that are used for modeling other environmental variables, such as current velocity.

Current velocity, emergence time and salinity are derived from models, which are acquired from the Directorate for Public Works and Water Management (RWS). Current velocity is calculated using Scalwest model with a spatial resolution of 100m. To calculate current velocities in 2004 a bathymetry map of 2004 and a standard tidal cycle (from 5-5-1996) are used. Salinity data is obtained with the Waqua model Scaldis400. Salinity is modeled for the discharge situation in the year 1992, which is considered an average year considering annual discharge in the period 1970-2000 (A. van Snik, 2006). For more details on the origin of all abiotic data see Appendix B.

2.2 Data exploration

Abiotic conditions from 2004 are coupled to species samples of the years 2003, 2004 and 2005. In this subset a selection of species is made following Ysebaert and Herman (2002). To remove effects of outliers and rare species in the ordination analysis, species occurring at less than six stations are deleted. Similarly as in Ysebaert and Herman (2002) only taxa that are determined until the genus level are used (except for *Nemertea* and Oligochaeta), and some taxa are also grouped at the genus level (*Polydora*, *Eteone*, *Malacoceros* and *Anaitides*). Next to this, samples of large mobile epifauna, such as crabs, are omitted and samples within mussel or oyster beds are deleted as well. The latter two are considered to have strong influences on sediment composition, which interferes with the aim of this study to determine whether we can use abiotic conditions to predict benthic species composition and biomass. Finally, samples without any species are omitted as well as multivariate cannot be performed with data that contains only zeros. Linking these samples with abiotic parameters already shows that most of these samples occur in deep gullies with high current velocities. The remaining dataset contains 861 unique data records and 59 species. Biomass data is log(x+1) transformed and density data is log transformed to meet assumptions for normality.

A correlogram is constructed to examine relationships between abiotic variables. Highly correlated environmental variables are not incorporated both in ordination analyses, as they will explain similar patterns in the species data.

Effects of season, region and monitoring program are examined visually. These effects are not incorporated into ordination analyses, as regions and monitoring programs show spatial variability that is also reflected in salinity and depth (or emergence time). Further, differences in biomass, density or species composition between different seasons are not of interest to our present study. All data analyses were executed in R 2.9.1, which is freely available software.

2.3 Multivariate exploration of hierarchy in abiotic conditions

A multivariate approach is chosen to determine hierarchy of environmental factors influencing species distribution. For community analysis multivariate techniques are often used in ecology (ter Braak 1986, Jongman et al. 1995). Multivariate ordination techniques can give insight into the relationship between species distribution and environmental variables. First, a DCA is performed to examine gradient length (in standard deviation units). Gradient length is over 3 SD, which implies that data is unimodally distributed and in further analyses specific techniques fit for this distribution can be used, such as canonical correspondence analyses (CCA) (ter Braak 1986, Jongman et al. 1995). Besides a CCA containing four environmental variables (salinity, depth, silt and current velocity), a series of partial CCA's are performed to examine relative strength of each environmental variable. This is done by running a partial CCA for a single environmental variable, while including the other environmental variables as covariables, and thus, filtering for their effects. For more background on these analyses we refer to van Wesenbeeck (2007) and to Jongman and ter Braak (1995).

2.4 Validation of splitting values of abiotic parameters

The aim of this analysis is to examine whether "natural" thresholds for environmental variables can be detected that predict distinct species communities. In other words, which value of each environmental variable divides species data (represented by a single value, such as biomass or density) in the most optimal groups, so that variation within these groups is minimal and variation between these groups is maximal. A simple one-way ANOVA tests whether differnces between two separate groups are significant. Plus, an ANOVA outputs the Total Sum of Squares, which is a proxy for the total amount of variance, and the Residual Sum of Squares, which represents the variance remaining unexplained by dividing data into two separate groups. If the residual sum of squares is close to the total sum of squares, their ratio is close to 1, implying that variation is badly explained by both groups. If variation between groups is high, the residual sum of squares is low, implying that, once divided by the total sum of squares, the value is close to zero. This ratio is calculated for all possible values of each environmental variable, This way the most optimal value to divide between groups can be obtained. This procedure was automatized in R 2.9.1 and was executed before by Ysebaert et al. (2009).

The ANOVA procedure to obtain optimal 'splitting values', as described above, is executed for five different response parameters; species density, number, biomass, an 'ecological richness' parameter and cca site scores. Biomass and density are log transformed to meet standards for normality. The ecological richness parameter is constituted from biomass, density and species number (Ysebaert et al 2009). These three parameters are standardized by substracting the mean of all obervations from every single observation and dividing this value by the standard deviation. The average of these three standardized variables forms the 'ecological richness' variable. CCA site scores are used as a response parameter to obtain a measure for species composition. This is done by executing a CCA for a single variable and then using the site scores as a univariate measure for species composition. Examined environmental variables are similar to those in the CCA: Salinity, current velocity, depth and silt content. For each value of these variables the data set is split in two groups, below the specific value and above. The minimum number of samples in each group should be 2.

3 Results

3.1 Exploratory data analyses

To explore basal relationships in abiotic data, correlations between different abiotic parameters are examined in a correlogram (Figure 3.1).

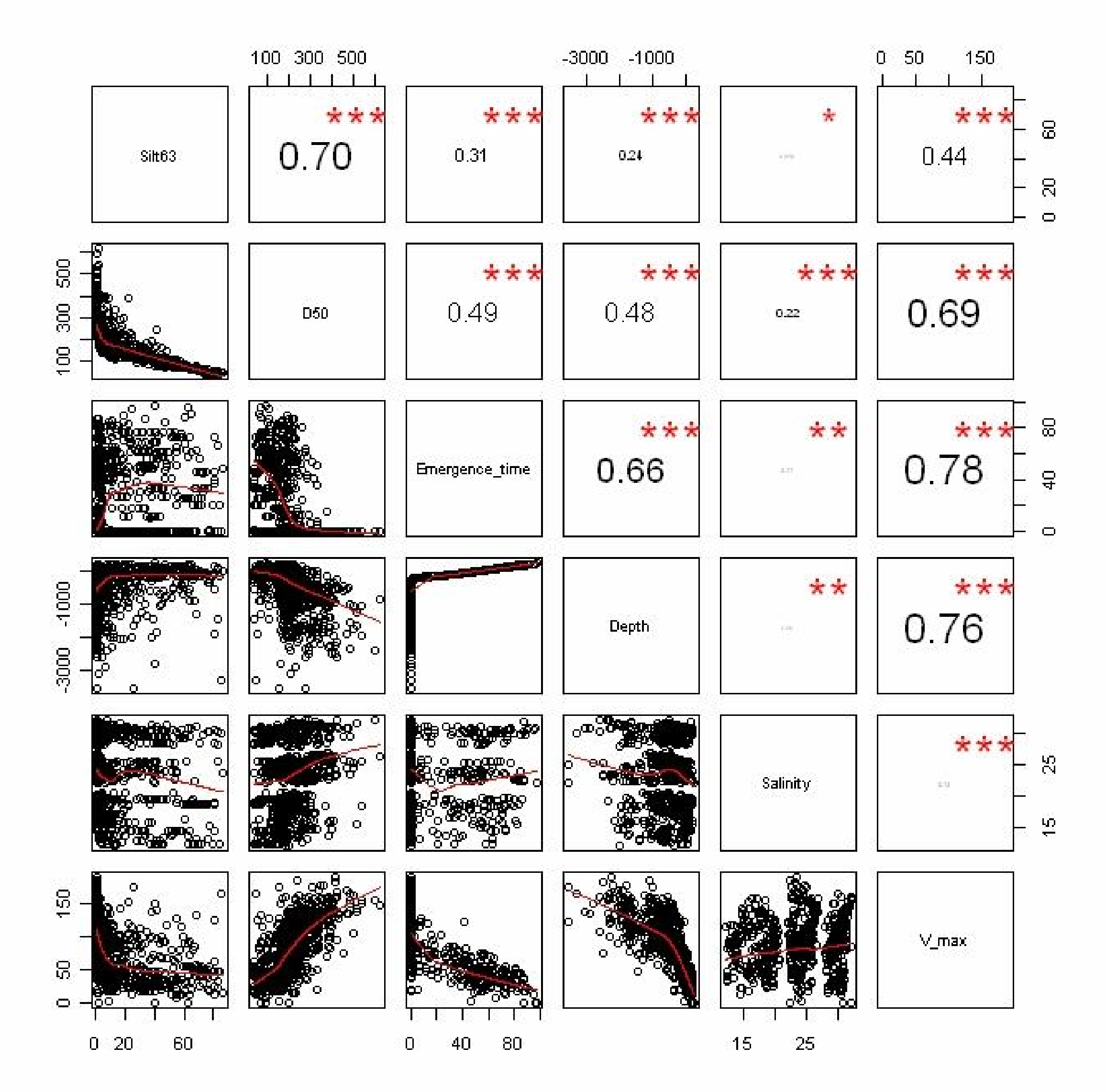

Figure 3.1 Correlations for different environmental variables with R^2 values and stars indicating significance (p<0.05, **p<0.01, ***p<0.001).*

All examined correlations are significant, due to the high number of observations. However, not all are relevant. For example, both emergence time and current velocity are directly derived from depth. Thus, it is no surprise that both variables correlate with depth (Figure 3.1). Compared to depth, emergence time contains less information as it is zero in all points that are below minimum low tide. Using this factor would cause a loss of information in subtidal areas and introduce a large number of zeros in our data, which complicates statistical analyses. Therefore, emergence time is omitted from further analyses. Another straightforward correlation is found between silt content and median grain size. There is some debate which of these parameters performs better in explaining benthic species composition and abundance. Here median grain size is excluded as it seems to correlate slightly better with other environmental variables. Another expected correlation is between current velocity and silt content (Figure 3.1), as high currents prevent precipitation of small particles. There are a few known locations in the westerschelde, where silt precipitates even in high flow conditions due to local eddies. However, this is not expected to affect the relation between flow velocity and silt content on the scale of the whole system. Finally, salinity does not show correlations with other factors that are meaningful and have an R-value larger than 0.22. Salinity data seems clustered in three different groups, which can be explained by the data reflecting spatial distribution of monitoring points along the estuary. For more exploratory data analyses see Appendix C.

3.2 Hierarchy of abiotic variables

To detect relationships between environmental parameters and species composition a CCA is performed.

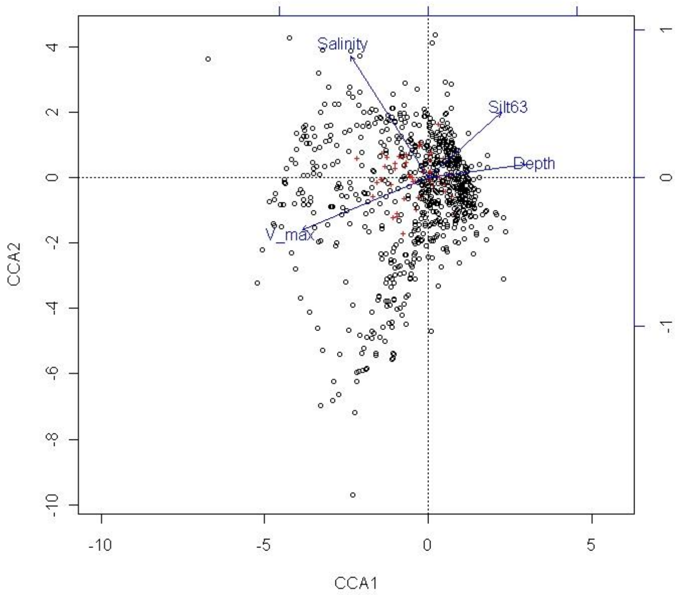

Figure 3.2 Canonical correspondence analysis (CCA) ordination diagram for macrobenthos species biomass data

Figure 3.2. shows that the first CCA axis is mainly determined by current velocity (according to the length of the V_max arrow), which is negatively correlated with depth and silt content (arrows for these variables are pointing in opposite directions). Salinity exerts the largest influence on the second axis (see upward pointing arrow for salinity). In total the CCA explains 5.86 % of the variation in the species data (see Appendix D for calculation), which is a reasonable amount for this type of analyses, due to the many factors that influence distribution of several species.

Partial CCA analysis is executed to examine effects of abiotic variables independent of the other three variables. Therefore, the other three variables were included as covariables. Visual inspection of the CCA (figure 3.2) leads to believe that current velocity is the most important parameter in explaining data patterns as it is the largest variable that constitutes the first axis. However, revealing the contribution of each independent variable after filtering the effect of other variables, shows that salinity explains most variation and current velocity is of second importance (Table 3.1).

Table 3.1 Values for all environmental variables resulting from partial CCA analyses. Variables are ordered in descending order of importance in determining species composition. The constrained column gives the inertia for the single variable. The conditional column gives the inertia for the other three variables.

Variable	Total	Conditional	Unconstrained	Constrained
Salinity	10.2866	0.2658	9.8241	0.1967
V_max	10.2866	0.1894	9.9541	0.1431
Silt63	10.2866	0.26582	9.93381	0.08701
Depth	10.2866	0.26582	9.95407	0.06676

3.3 Optimal ANOVA model

Figure 3.3 represents the ANOVA tests for the four abiotic variables with different response parameters (density, biomass, species number, ecological richness and species site scores). As can be seen in the figure, results do not differ much for density, biomass, species number and ecological richness. For depth, current velocity and silt content, clear results for optimal splitting values are obtained. Only for salinity, optimal models are not very optimal as the ration between the sum of squares and the total sum of squares is around 0.9 implying that most variation is found within groups. Therefore, we decided to introduce a measure for species composition as we found in the CCA that salinity has a large influence on species composition. The CCA site scores give similar results for the other environmental variables, but for salinity the optimal model improves considerably. In table 3.2 the exact splitting values are given for all response variables and for all abiotic conditions. Combining this table with figure 3.3 allows us to determine which splitting value is the best (the one with the lowest ratio value on the y-axis). The most optimal values are printed bold in table 3.2. In general, comparing y-axes of different graphs shows that density and species number yield more significant results compared to biomass. For silt content and salinity best values are obtained by using species site scores, the proxy for species composition. Current velocity links good with three different response parameters: Density, species number and ecological richness. Surprisingly, biomass is not the best response parameter to assess threshold values for environmental parameters. From these results, it seems that construction of an ecological richness parameter is not extremely useful, as it does not lead to significant better explanation of variation in our dataset compared to the other response parameters.

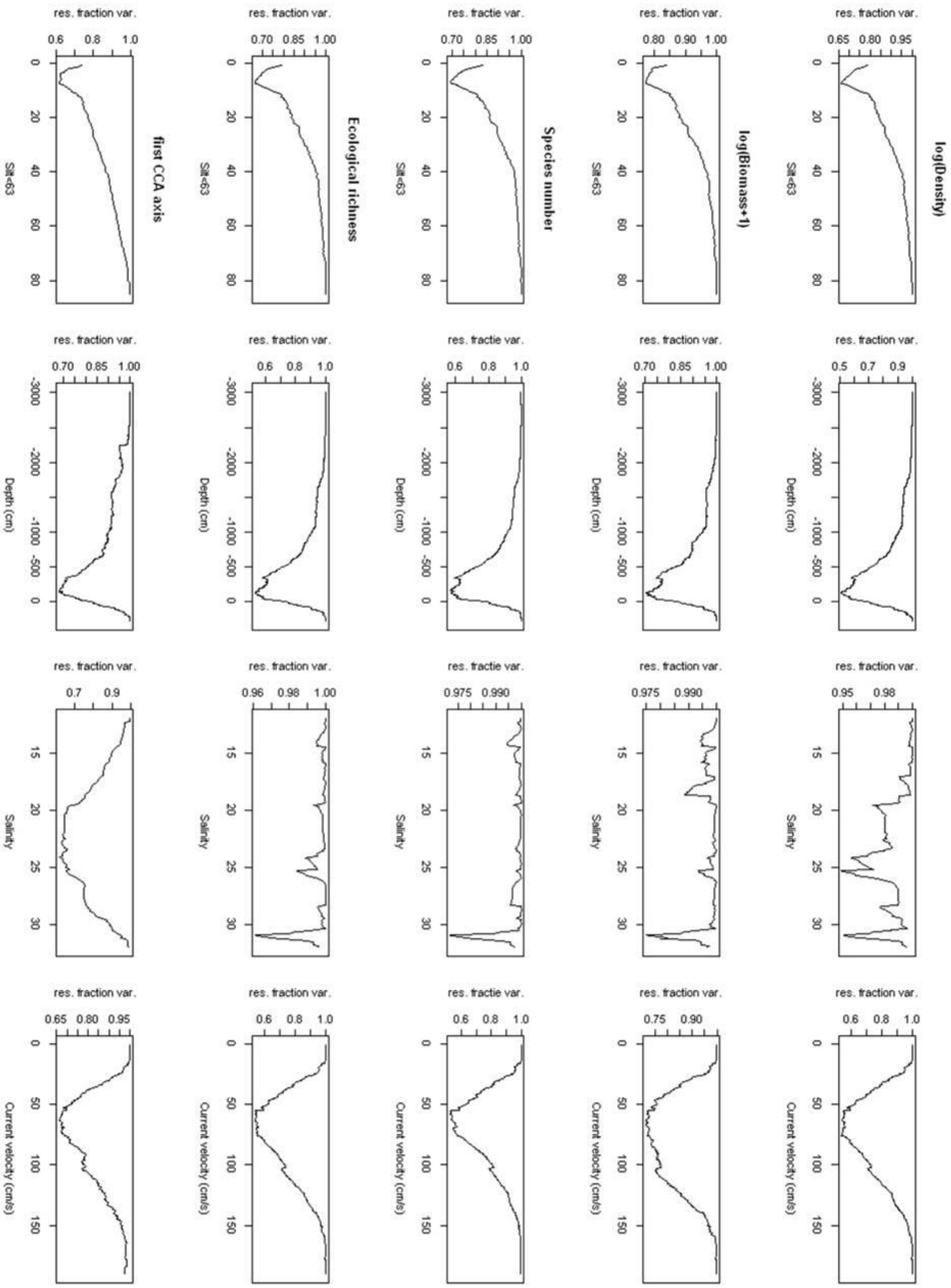

Figure 3.3 *Critical values determined by a one-way ANOVA for four abiotic variables with density, biomass, species number, richness and sites scores from a CCA analysis.*

Table 3.2 Critical values for abiotic conditions where variation between groups is largest and within groups is minimal according to a one-way ANOVA. Bold values yielded best results according to Figure 3.3.

	Silt	Depth	Salinity	Current velocity
Density	7.5	**-135**	25.3	**75.5**
Biomass	6.5	-120	30.9	62.5
Number	7	-175	30.9	**55.5**
Richness	7.5	-135	30.9	**62.5**
CCA site scores	**7**	-175	**24.1**	62

3.4 Previous analyses on hierarchy of abiotic variables

Several studies investigate macrobenthos species response on environmental conditions in the Westerschelde. (Table 3.3). Scale of these studies vary considerably and, thus, results vary as well. On small scales, such as a single intertidal flat, most important factors are: sediment composition, bed level height and presence of microphytobenthos (chlorophyll a). On larger scales salinity and depth are more decisive in determining species composition and abundance. These scales should be reflected by the hierarchy in a habitat classification system. Table 3.4 shows classes and their divisions in the ZES.1 (Bouma et al 2005).

Table 3.3 Previous papers and conclusions on hierarchy of factors in the Westerschelde.

Paper	Spatial scale	Time range	Response variable	Analysis	Factors
Ysebaert & Herman 2002	1-10000 m	1994-2000	Biomass, abundance	Univariate, Multiple regression	Mud content, chlorophyll a, bed height, salinity
			Abundance	Multivariate CCA	1. mud content 2. salinity 3. chlorophyll a
Ysebaert et al. 2003	estuary	1978-1997	Diversity, biomass, abundance	Univariate Two-way ANOVA	Salinity and depth
			Biomass or abundance	Multivariate CCA	1. depth (which reflected hydrodynamic conditions) 2. salinity 3. mud content
Van der Wal et al. 2008	intertidal flat ($12km^2$)	2004-2006	Biomass and species richness	Univariate GLM	Microphytobenthos in interaction with sediment composition

Table 3.4 Current splitting values in ZES.1

Variabelen	Klassen	Grenzen	
Zoutgehalte	brak	5.4-18	
	zout	>18	
	variabel	>5.4 en variabel	
Substraat 1	hard	Steen, hout,veen	
	zacht	sediment	
Diepte 1	sublitoraal	< GLWS	
	litoraal	GLWS-GHWD	
	supralitoraal	>GHWD	
Hydrodynamiek	hoogdynamisch	> 0.8 m/s	
	laagdynamisch	< 0.8 m	
	stagnant	0 m/s	
Diepte 2	sublitoraal: zeer diep	>15 m (Noordzee>30 m)	
	sublitoraal: diep	5-15 m (Noordzee 20-30 m)	
	sublitoraal: ondiep	5m-GLWS (Noordzee 20 m- GLWS)	
	litoraal: laaglitoraal	GLWS-75%	
	litoraal: middenlitoraal	75-25%	
	litoraal: hooglitoraal	25%-GHWD	
	supralitoraal: pionierzone	GHWD tot > 300 keer per jaar	
	supralitoraal: lage kwelder	300-150 keer per jaar	
	supralitoraal: midden kwelder	150-50 keer per jaar	
	supralitoraal: hoge kwelder	50-5 keer per jaar	
Substraat 2	slibrijk	Slib63>25%	
	fijn zand	Slib63<25%	D50<250 µm
	grof zand	Slib63<25%	D50 250-2000 µm
	grind	Slib63<25%	D50>2000 µm

3.5 Combining splitting values results and ZES.1

Table 3.5 is constructed to aid in comparing between splitting values derived in this study and the values that are used in the ZES.1. The ZES.1 divides on more levels than included here. However, in the current study data restricts analyses between certain borders, as data of soft sediment communities is used and data did not cover complete environmental gradients, which is for example the case for salinity, as no data for low salinities is analyzed.

Table 3.5 Splitting values for environmental conditions as suggested by this study compared to splitting values in the ZES.1

	This study	**ZES.1**
Salinity	24.1	18
Current velocity	62.5-75.5 cm/s	80 cm/s
Silt	7 %	25 %
Depth	-135 cm NAP	GLWS

4 Discussion

Several studies look at benthic species distribution in relation to abiotic parameters in the Westerschelde. Obviously, from a policy point of view, it is desirable to use benthic species composition as a tool to predict occurrence of habitats, and, more importantly, effects of management measures on distribution of habitats. However, although significant progress is made in understanding distribution of soft-bottom benthic fauna, it is debatable whether benthic macrofauna is a good and reliable indicator for changes within an estuary.

First, benthic fauna in an estuary is known to be distributed along a continuum, instead of in discrete units. Thus, definition of distinct communities is complex. This is visualized by the performed CCA analysis, where species are distributed continuously along the second axis. Second, previous studies on benthic species composition found that variability was mostly explained by variation between stations and the interaction between stations and years (Ysebaert & Herman 2002). This implies that local changes account for most of the variation in the data, which is reflected by the low percentage of variation that is explained by the CCA analysis. However, although benthic species composition is still difficult to predict by abiotic conditions, some general conclusions can be drawn from this study and previous studies.

4.1 Hierarchy

First of all, partial CCA shows that salinity is the major component explaining species distribution on the scale of an estuary. This is in line with the hierarchy of the ZES.1, where salinity is suggested as the first factor that should be taken into account (Bouma et al 2005). However, whether salinity is relevant also depends on the spatial scale of the study. On the scale of a single intertidal flat, salinity is not expected to show significant variations and, thus, does not have a major influence on species composition and abundance (van der Wal et al 2008).

Our present study points at the importance of current velocity in explaining species composition. In previous studies this factor was regularly overlooked, resulting in high importance of depth (Ysebaert et al 2003). It seems advisable for future work to take into account a qualitative parameter representing hydrodynamics. This is underlined by recent work of Ysebaert et al. (2009) and Plancke et al. (2009), where current velocity is shown to be a structuring parameter for subtidal communities. These recent results plead for including current velocity in analysis of benthic communities. Although current velocity models are heavily criticized, using these models might be preferable to using subjective qualitative measures, such as the presence or absence of bed forms as a proxy for hydrodynamics. In other projects (Deltakennis) an effort has been made to analyze why the current hydrodynamic models perform poorly in intertidal areas and which steps should be taken to improve these models (Dekker, 2009). Considering that flow velocity might be a major independent determining factor in benthos distribution, it would at least be worth implementing this in other studies to look further into it.

On smaller scales, such as a single intertidal flat, factors such as sediment composition and bed height become more important (van der Wal 2008, Ysebaert et al 2009). In the partial CCA sediment composition and depth explain little variation. However, both factors are strongly correlated with current velocity. This explains why these factors seem more important in other studies, where current velocity was not quantified. Comparing our results with the current hierarchy in the ZES.1 some discrepancies are clear. The ZES.1 goes into much detail defining different depth classes. It seems legitimate to ask whether this much detail will ever be reflected by available data. With this respect some simplifications of the ZES.1 might improve its applicability. Similar for other parameters such as submergence time, silt content and median grain size. Silt content and median grain size are strongly correlated, and, thus, it is not desirable to include them both in the same analysis. Similar arguments are valid for submergence time and depth. They are strongly correlated and therefore, cannot be used both.

4.2 Subdivision of classes

Splitting values as calculated in this study deviate slightly from splitting values as proposed in the ZES.1. Using density and CCA scores as response parameters yielded most optimal results. In literature there is an ongoing debate whether density or biomass might be a better predictor for benthic species occurrence. However, based on our results we can not make general assumptions on this. For silt the most optimal splitting value was 7 %, using the CCA site scores. This value is rather distinct from values used in the ZES.1. The form of the graph points at the importance of silt for species composition with very low concentrations. The 25% silt content which is used in the ZES.1 to split between different groups might be a rather high estimate. Our analysis for current velocity showed that splitting values could vary in a range between 55.5 to 75.5 cm/s. Again the 80 cm/s that is mentioned as a standard in the ZES.1 might be a fairly high estimate to distinguish between high and low hydrodynamic conditions. However, the values of 55-75 cm/s predicted by Scalwest may in reality be higher, as Scalwest tends to under-predict velocities in intertidal areas (Dekker et al, 2009).
Considering depth our analysis defines 135 centimeter below NAP as the most appropriate value to separate between different groups. This value divides between subtidal and intertidal samples in particular. This corresponds fairly good with the mean low tide during springtide level (GLWS) in the ZES.1. Finally, for salinity no significant splitting value was found using univariate response parameters, such as density, biomass and richness. Therefore, the CCA site scores were included. For these scores (representing community composition in a univariate measure) salinity can most optimally be divided into two groups at 24.1. This value is higher than the suggested value of 18, which might be caused by the fact that salt variation was not included in our analyses. This factor is included in the ZES and is known to have a large influence on species composition as salinity fluctuations immediately result in mortality of certain species.

4.3 Recommendations for future research

In the present study, only a small percentage of the variation I the benthic data can be explained using the available abiotic information. This may partly be a consequence of inaccuracies in the data. However, it is also likely that abiotic factors alone are not sufficient to provide accurate predictions on benthic species composition. The predictability of benthos composition can possibly be enhanced if certain biotic parameters are taken into account, such as food availability (Chlorophyll-a concentrations, or primary production data). Although this is a deviation from the original habitat approach, this may deserve some attention in the future. A disadvantage of the habitat approach (based on abiotic factors alone) is that the system is seen as relatively static. Including some biotic parameters (preferably in terms of productivity or if that is unachievable in terms of biomass) may partially counteract this limitation. Prerequisite for such an extension of this study is that reliable, spatially explicit data are available productivity or algal biomass. At present the availability of such data is limited, but in the course of other, currently running LTV-projects we expect such data to become available in the near future. It is therefore advisable not to initiate such an analysis immediately, but postpone this until these projects have delivered the required results.

As the current study is one of the first studies that validates the salt water habitat system by using quantitative data it is strongly recommended to extent this analysis using other datasets. New abiotic and biotic data is already available as in 2008 new habitat maps were produced. New data on productivity will be available by the end of this year. All this data can constitute the basis for a similar analysis on validation of the habitat system in 2011. Concluding, from the current study it seems valuable to determine and test hierarchy and splitting values of abiotic conditions that form the base of an habitat classification with real-time data. Even if there are inconsistencies and uncertainties in this data, very general conclusions can be obtained that aid in improving classification systems and will make them more suitable for application.

4.4 Conclusions

Looking back at questions asked in the introduction of this study, compact answers are formulated:

1. Which environmental variables and in what order, influence benthic species composition?

a. Salinity
b. Current velocity
c. Silt content
d. Depth

2. Is this in agreement with the variables and their order used in the ZES.1?

More or less, but the ZES.1 defines different depth classes at two separate levels. The first level where depth is introduced follows directly after salinity. However, we find a larger importance of current velocity, which is confirmed by other studies. Therefore, it is advisable to improve data on current velocities, by improving models and by executing field measurements. To a large extent this is already happening. However, the next step should be to evaluate the effect of new data on species compoNext, this new data should be evaluated in relation to benthic species composition.

3. Can we determine quantitatively the optimal values for abiotic conditions to divide data into separate groups (or habitats)?

Yes. We used a method similar to Ysebaert et al. (2009) to quantify the best value for splitting between distinct classes of environmental conditions.

4. Are these splitting values in agreement with values used in the ZES.1?

Not entirely. However, it seems straightforward that splitting values should not be adapted based solely on this analysis. Moreover, splitting variables are supposed to be variable, especially among different systems.

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A Biotic data

Benthic data of the Wester Scheldt was collected at Rijkswaterstaat (RWS) and at the Netherlands Institute for Ecology, Centre for Marine and Estuarine Ecology (NIOO-CEME). The dataset consists of data that were obtained in three different monitoring programs. These programs are known as BIOMON, MOVE and MOVE-Ecotopes. The first program sampled at different depth strata. Within these strata sample locations were randomly located. The MOVE project is a project to monitor the effects of the deepening of the navigation channel of the Western Scheldt. In this program sampling was also executed on random locations within certain depth strata, but in the intertidal also some fixed points along transect were monitored. Finally, the MOVE-Ecotope program, was started in 2007 and is a test to adjust sampling based on previously studied maps of distribution of ecotopes.

In the BIOMON and MOVE programs sampling in the field was performed using a box-corer. From each box-core three cores of 8 cm were sieved on a 1 mm sieve. Samples were further selected and determined onto a species level in the lab.

For the years used in this study (2003, 2004 and 2005) samples were only gathered in the BIOMON and the MOVE project. The sample data from both these projects and sediment data were put together in one database (BIS-database) by NIOO. To obtain benthosdata that was already coupled with sediment samples, this BIS-database of the NIOO was acquired. Unfortunately this database contained several inconsistencies. For the sedimentdata the main problems were that the obtained samples were analysed with three different Malvern Mastersizers. Data obtained with these different Malverns were not directly comparable.

Although these problems were partly addressed still some recalculating and homogenizing of the data was required. For now, data measured with the new NIOO Malvern was transformed to the old NIOO Malvern, using the same method as van der Wal et al. (2005). The data from the BIS database contained several inconsistencies. These resulted in creation of extra classes, making the database unsuitable for analysis. Therefore, considerable cleaning up of the data was executed. The following small inconsistencies were observed and removed:

1. inconsistent spelling (percentage and precentage)
2. distinct names for identical analysis
3. classes containing question marks (<?? mu)
4. in the years 2006 and 2007 many samples were present twice but with tiny differences (44445 and 44445.3). The most accurate sample was used in this study.
5. the dates are sometimes noted as day/month/year, but sometime as day/month/year/hour/minute/second. This causes problems because databases are linked with dates. The format day/month/year was applied for the complete database.

In the species database there were several records with high species numbers but without measured dry weight. Those records were deleted. Records with low species numbers and a dry weight of zero were changed in a dry weight of 0.001. Records with no benthic animals present were given the value of 0 for dry weight.

B Abiotic data

The last available ecotope map originates from 2004. Therefore, abiotic data of this year were used. These abiotic parameters were combined with benthic data from three years (2003, 2004 and 2005). Three years were used as it can be assumed that in general abiotics of one year can be linked with benthic data of surrounding years. This way a larger amount of benthic data was obtained making the used data less susceptible to outliers and abnormalities. Figure 2.1 shows the sample locations used in the research.

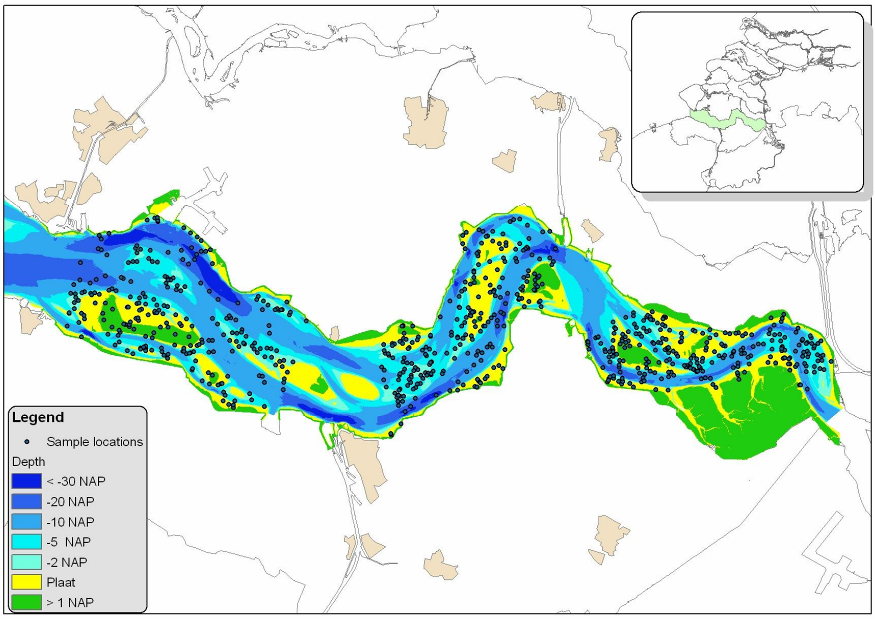

Figure 5.1 Sample locations

5.2 Abiotic data

Abiotic data such as bathymetry, current velocity, emergence time and salinity were available in raster format. First, all raster maps were converted to a 20 m grid. Then abiotic information was derived from maps for each sample location, using ArcGis9.2. Abiotic, benthic and sediment information was collected in an excel file for further analysis.

Bathymetry
Bathymetry in cm relative to NAP was given in a grid map with a grid size of 20x20m. Input for the bathymetry map is collected from single-beam measurements taken along transects with a mutual distance of 100 meters. These data were further interpolated in DIGIPOL to a 20 by 20 meter grid map. On shallow tidal flats, single-beam measurements cannot be taken. Therefore, laser altimetry measurements, with a grid size of 2x2m, were used to fill in missing information (A. van Snik, 2006).

Maximum current velocity
Current velocity was calculated using the ScalWest model. This model was calibrated for the situation in 1996 using the bathymetry from 1996, and a single tidal curve, measured at 5-5-

1996 at fixed stations along the Western Scheldt. For the current velocity in the year 2004, bathymetry of 2004 was used in the model. The same tidal cycle of 5-5-1996 was used in all models. Modeling results were corrected for an average spring tide, as maximum current velocity occurs during spring tide. A factor was used which describes the proportion of the tide difference of an average spring tide versus the tide difference of the tide which was used in calculation. The values for an average spring tide were calculated with the measured tide data of the year 1991 (A. van Snik, 2006).

Emergence time

The total time the intertidal flats are flooded with water is called the emergence time. In this study the opposite was used; the time that the intertidal flats are not submerged. This time was given as a percentage. The emergence time, and the opposite dry time, was derived from the next parameters:

Measured tidal cycle: For calculation of the emergence time in 2004, tidal data of 5 years before 2004 at measuring stations Vlissingen, Terneuzen, Hansweert and Bath was used. In these stations water level relative to NAP was registered every ten minutes.

Elevation: Elevation relative to NAP was used.

Duration of the tidal wave: the time difference in the high water event between the different measuring stations.

Salinity

Salinity was obtained with the use of the Waqua model Scaldis400, resulting in a map with a grid size of 100 x 100 m. The model calculates the salinity simultaneously with the water level movement. The model was calibrated with salinity measurements from the year 1990. The salinity in the Westerschelde is, amongst other things, dependent on the discharge of the Sea Scheldt.

Salinity was modeled for the discharge situation in the year 1992. This year was chosen because the discharge in this year is representative for the average annual discharge in the period 1970-2000. The discharges were imposed as a decade averages. As tidal boundary conditions the available data set of the whole year 1990 was used (A. van Snik, 2006).

C Exploratory data analyses

Benthos data from 2003, 2004 and 2005 was explored for dominant patterns. Figure C1 shows that biomass differs per year and per season. Benthic biomass is always higher in autumn compared to spring, and yearly fluctuations in benthic biomass occur regularly. Besides natural variations effects of monitoring can be visible in data as well. Enormous differences in *Laniche* biomass in 2004 compared to 2003 and 2005 seem to be caused by a single sample.

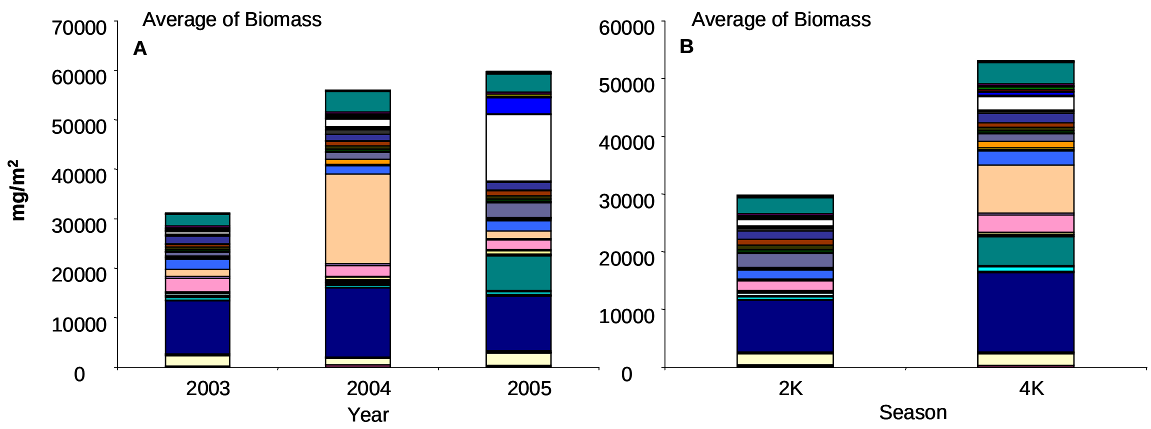

Figure C.1 *Average biomass for different years and for different seasons.A. Third column from top till bottom: White represents Nereis Virens, pink represents Laniche congilega (second column in graph A), turquoise/greenish represents Ensis Americanus and dark blue is Cerastoderma edule.*

From Figure C2 it becomes clear that invasion of *Ensis americanus* contributed to overall rises in biomass from 2003 until 2005. Other shellfish species fluctuate slightly, but remain more stable over these three years.

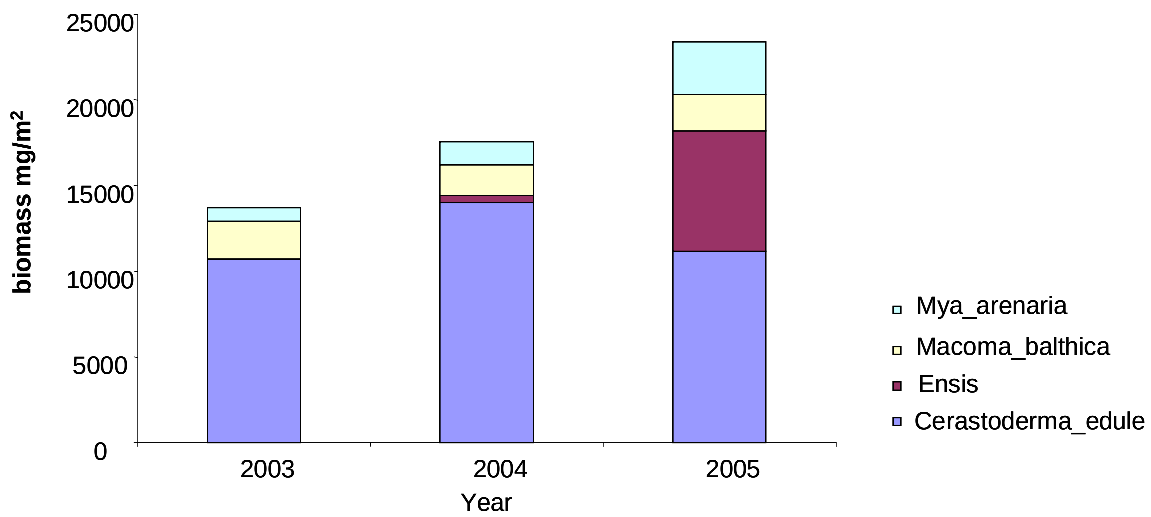

Figure C.2 *Average biomass of several shellfish species in the years 2003, 2004 and 2005.*

Further, differences in biomass and density can be found between areas and between monitoring programs (Figure C3). In Figure C3 and C4 it shows that shellfish communities are dominating in the western part of the Westerschelde, explaining higher biomass and lower densities. *Corophium volutator* is most abundant in the central part and *Pygospio elegans* is most abundant in the Eastern part, although this is probably not significant (Figure C4).

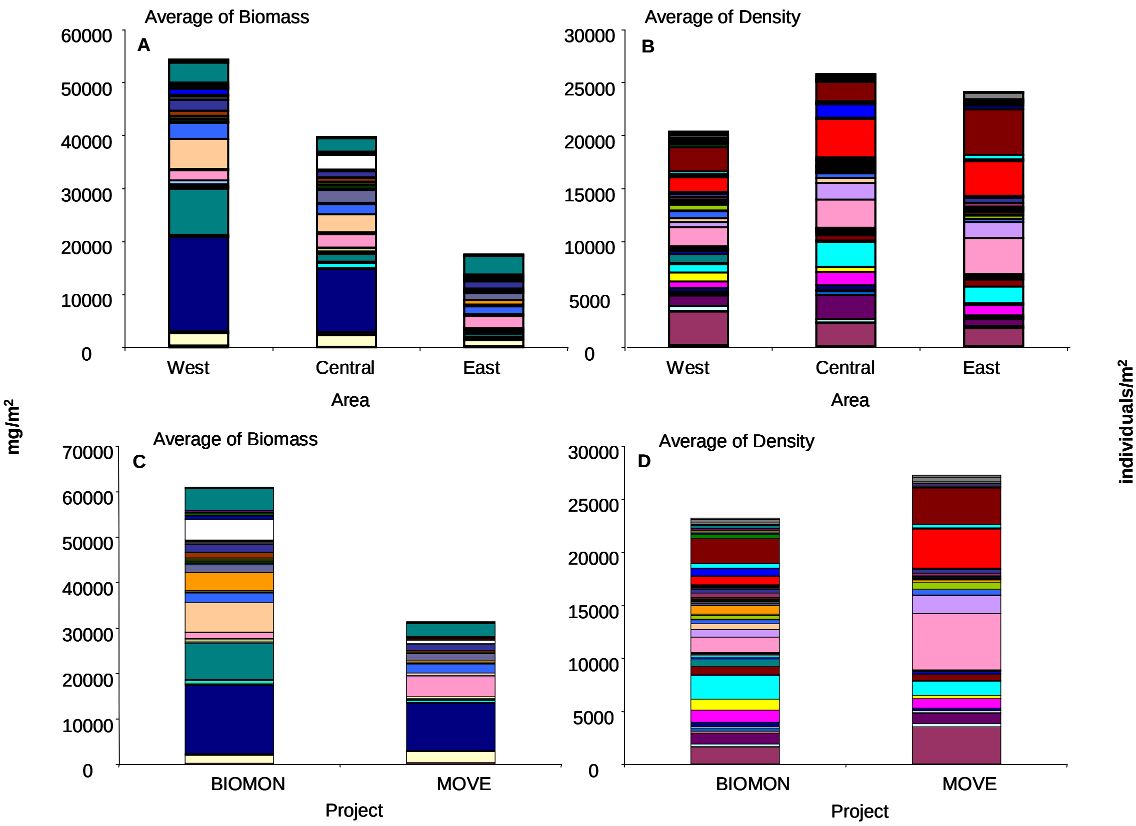

Figure C.3 *Average density and biomass for all species (different colours are separate species) in different areas and in different monitoring programs. First column in graph C from top till bottom: White represents Nereis Virens, pink represents Laniche congilega, turquoise/greenish represents Ensis Americanus and dark blue is Cerastoderma edule.*

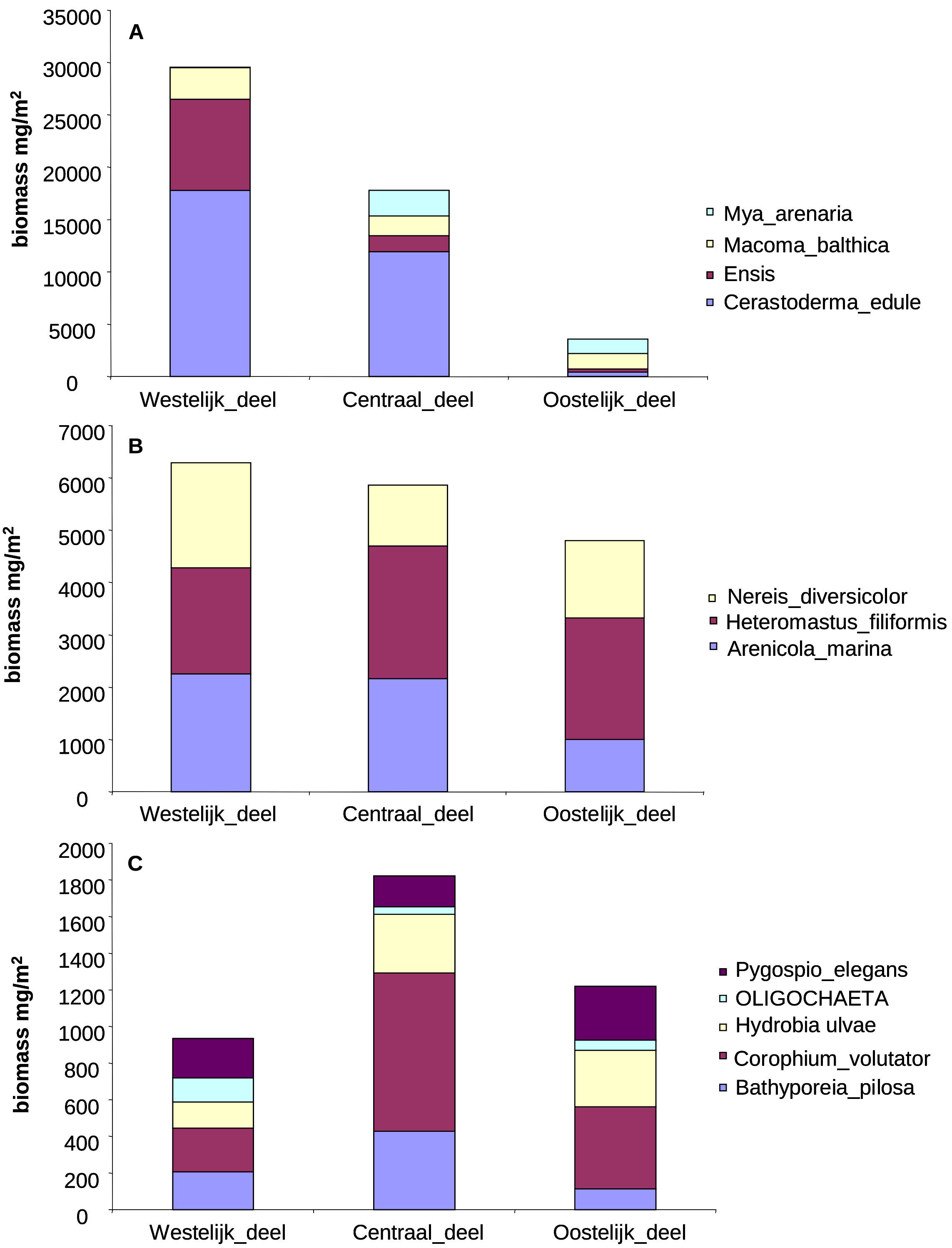

Figure C.4 *Average biomass of several species for the West, central and Eastern part of the Westerschelde. Figure A shows average biomass for shellfish species. Figure B shows biomass for three common species of Polychaeta.*

D Multivariate ordination results

Table 3.1 presents Eigen values of the CCA. For the CCA the percent of variation that is explained by the constrained ordination can be calculated. This is done by dividing the total constrained intertia by the total intertia (0.6033/10.2866= 0.0586). This implies that our four constrained axis together explain 5.86 % of the variation in the dataset. To check whether there are strong interdependencies amongst parameters the variance inflation vectors are checked (Table 3.2). A VIF > 10 indicates that a variable is strongly dependent on others and does not have independent information. VIF's in the current analysis are well below 10. Finally, an ANOVA indicates that all abiotic variables are significant (Table 3.3). As can be seen from figure 3.3 species are well distributed along both CCA axes.

Table 5.1 Results of CCA

***CCA**

	Inertia	Rank
Total	10.3677	
Constrained	0.6074	4
Unconstrained	9.7603	57

Inertia is mean squared contingency coefficient

Eigen values for constrained axes:

CCA1	CCA2	CCA3	CCA4
0.31809	0.17654	0.09025	0.02256

Eigen values for unconstrained axes:

CA1	CA2	CA3	CA4	CA5	CA6	CA7	CA8
0.5758	0.5442	0.4774	0.4336	0.4078	0.3926	0.3868	0.3722

(Showed only 8 of all 57 unconstrained Eigen values)

Table 5.2 Results of VIF analysis

*** VIF**

Silt63	Depth	Salinity	V_max
1.161747	2.647605	1.002063	2.906240

Table 5.3 ANOVA for all abiotic variables

Model: cca(formula = species ~ Silt63 + Depth + Salinity + V_max, data = abiotics)

	Df	Chisq	F	N.Perm	Pr(>F)
Silt63	1	0.1477	13.0604	499	0.002 **
Depth	1	0.1686	14.9030	499	0.002 **
Salinity	1	0.1964	17.3648	499	0.002 **
V_max	1	0.0905	8.0029	499	0.002 **
Residual	856	9.6833			

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

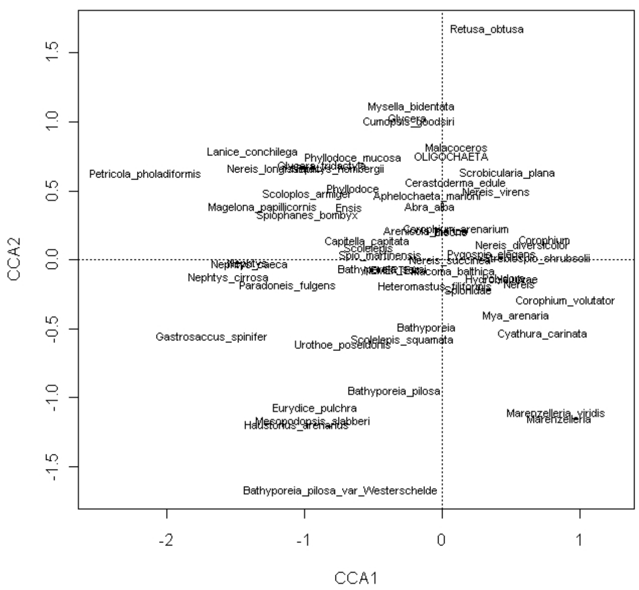

Figure 5.2 Species distribution derived from the CCA