Robustness parameters habitat assessment tools

Study as a part of LTV Research & Monitoring 'Natural development' for the Schelde estuary

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Front: The 'Hooge Springer' tidal flat (Westerschelde) during low tide (obtained from archive Monitor Taskforce).

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Summary

The ‘long-term vision (LTV)’ on the Schelde aims at maintaining and/or improving ‘safety’, ‘accessibility’ and ‘natural development’ in the estuary by 2030. An essential part of the program is the development of an adequate and robust habitat assessment tool. Presently, the Saltwater Ecotope System (ZES) is the basis for prediction and monitoring of changes in ecological functioning of the Schelde estuary. ZES combines a set of abiotic parameters (bathymetry, exposure time, dynamics, sediment composition and salinity) to describe ecotopes that in turn link to the presence of specific communities. The current study investigates the robustness of the abiotic descriptive parameters; how determining are the physical-chemical parameters for biological communities, and how accurate should and can they be measured or modeled.

Our study makes use of 3112 macrobenthos samples taken between 1978 and 1997 in the polyhaline and mesohaline part of the Schelde estuary. Species response curves, based on presence – absence data, to each of the above mentioned abiotic parameters have been calculated for the 20 most dominant species within the research area (Ysebaert et al., 2002). As we are interested in the probability of change in communities with changing environmental conditions, we calculated the derivatives of the responses with respect to the environmental variables (these express how much the probability of occurrence of a species changes with one unit change of the environmental variable) and combined those for all species. From these we calculated the change in environmental variable that would be sufficient to provoke an average deviation of 5 % in occurrence probability of all species in the community, which represents the ‘Essential Accuracy (EA)’ of measurement of the environmental variable at a given value.

With respect to **depth**, it is found that the current models of interpolation result in an accuracy of 50 cm area covering for bathymetry. In the litoral zone an accuracy of at least 40 cm should be achieved, whereas in the sublitoral zone the accuracy of depth measurements becomes quickly irrelevant making the interpolated measurements sufficient. Improvements in bathymetric accuracy can be reached by using Laser-altimetry for the litoral zone (accuracy around 20 cm), and use the interpolated bathymetries directly instead of going through the gridding used in models. The same accuracy as for bathymetry in the intertidal zone accounts for exposure time which is derived from height and depth.

Concerning **dynamics**, it is concluded that especially the expanding of the number of available series of field measurements can lead to substantial improvement of the modeled current velocities, over tidal flats in particular. Locally installed EMF and ADCP equipment, or ADCP attached to boats can deliver sufficient information to close the gap of the current velocity uncertainties which are now unclear (probably several 10s of cm/s) to the essential 5 to 10 cm/s at velocities below 80 cm/s. Especially the use of proxies based on morphological characteristics of the bed gives uncertainties and should be validated and affirmed with field measurements. Collecting a large and consistent field data set on current velocities in shallow areas and on tidal flats should receive preference over the refining of current modeling grids.

For **sediment composition**, communities are most sensitive around a median grainsize of 190 µm, corresponding to the transition between pure sands with very low mud content and muddy sands. Here the EA is smallest, in the order of 20 µm. It seems to matter less for communities, how coarse the sand is at coarser sand types. Communities appear to be very sensitive to mud percentage in the range of 0 to 20 % mud, where the EA should be around 4 %. Around 50 % mud the EA is much larger, whereas the EA decreases again with further increasing mud contents which indicates very inhospitable habitats for most species. The EA for median grain sizes is probably nowadays well reached, but where analytical precision is in the range of a few percents mud content measuring sample sites, the small-scale natural variability (cm to m) is definitely higher than the accuracy required. Also, interpolation of sediment composition from the sparse measurements to area-covering maps increases the uncertainty considerably. We see potential in the application of remote sensing methods to derive area-covering maps of sediment composition. However, these methods are currently unable to provide the necessary accuracy of mud percentage and should be improved if possible.
Especially in the brackish zone, a fairly accurate modeling or measurement of **salinity** is called for. Analytical precision in salinity measurements is far below 1 PSU unit, and measurements are based on very frequent time series, leading to far more precision than the EA of 1 to 2.
We discuss **other environmental variables** with a high potential to improve the ecotope classification system. Especially chlorophyll-a content of the sediment, an indicator for microphytobenthos, is a promising variable that is, moreover, easy to assess from remote sensing. We give an example showing that dominant benthic species are well predictable from this variable. A second potential variable is steepness of slopes, especially subtidally. As this is easy to calculate from bathymetric data, it should be easy to further test its usefulness.

Most environmental variables are strongly correlated among them, and moreover show spatial autocorrelation. We demonstrate that the results derived from the univariate logistic models on single variables, are generally also valid for the multivariate case. We further illustrate spatial autocorrelation in a few examples, and show that interpolated values of mud percentage rapidly lose precision away from the sampling points. Further study is needed to see how much of this interpolation uncertainty can be removed by combining field sampling with remote sensing.

Finally, we summarize the main conclusions of the study and suggest a priority listing of future research in order to improve the ecotope classification system and its field validation.

1. Introduction

As part of the long-term vision on the Schelde estuary, scoping developments until 2030, the research program LTV O&M (Long-term visions Research and Monitoring) was started in 2003. LTV O&M consists of three themes: ‘safety’, ‘accessibility’, en ‘natural development’. The current study is part of LTV O&M ‘natural development’. The long-term vision aims at defining the conditions necessary to maintain, at the same time, the physical characteristics of the estuary, maximum protection against flooding, optimal accessibility of the Schelde harbors, a healthy and dynamic ecosystem, and a governmental, political and operational cooperation between Flanders and the Netherlands. LTV O&M ‘natural development’ specifically focuses on the joint physical and biological monitoring of the Schelde estuary. It will define how the morphological evolution should be monitored and how the physical characteristics can be conserved. It will develop a scientific framework for plans, programs and projects in the estuary. It will also define science-based methods for monitoring of effects of ongoing and executed projects.

The following management questions related to natural development of the Schelde estuary are of interest to the ‘Directorate-General for Public Works and Water Management (RWS) Zeeland’ and the Flemish Government, Department of Mobility and Public Works (MOW), Section Maintenance Maritime Waterways; the two managing and executing institutions:

- What are the effects of (anthropogenic) physical alterations in the Schelde estuary on ecology?
- Is it possible to combine the morphologic management of the waterways with the ecosystem-related targets for the Schelde estuary?

To answer these questions it is essential to understand the relation between habitat parameters and the various organisms living in or dependent of the area. It is, therefore, necessary to enlarge the existing knowledge on relations between physical, biological and chemical processes and habitats, and the relations between habitats and organisms. With these insights, ecological conservation targets can be translated into a management plan for the Schelde estuary. To summarize, the following questions should be answered:

- What habitat parameters are of real importance, where and how much of each type of habitat is wanted, and what will be the response of species and communities and their distributions to certain measures?
- What is limiting the ecological carrying capacity of the Schelde estuary system for species?

As a result, the Technical Schelde Commission (TSC) asked for the execution of the following projects within the scope of LTV O&M ‘natural development’:

- ‘Comparative estuary study’.
- The development of habitat assessment tools.

The development of habitat assessment tools can be subdivided in research on the ‘Robustness of the parameters’ which define the current habitat assessment tool (the Saltwater Ecotope System (ZES); Bouma et al., 2005), and research on the ‘Techniques of validation’ of ZES. The current study focuses on the robustness of parameters in ZES, extended with an evaluation of other potentially useful parameters and new techniques to measure or assess relevant parameters for the habitat assessment tool.

Currently, the use of ‘ecotopes’, physically defined habitat types with a specific community, is central in the ecological management of the estuary. The use of ecotopes facilitates mapping and monitoring of changes in ecosystems. It is, however, essential that ecotopes are suitably defined. They should be (1) distinguishable based on field measurements, and (2) specific enough to harbor a unique community that is, moreover, (3) different from the communities in other ecotopes. In other words, there is a need for clear or at least recognizable borders between species assemblages which separate those in a logical way.

The study of Wijnhoven et al. (2006) shows significant differences in macrobenthos parameters between some of the ecotopes, but differences between other ecotopes are not found. This can be the result of poorly measured or irrelevant separations in physical-chemical parameters to define different communities. However, the accuracy with which these parameters can be measured or assessed might also be insufficient to describe a certain area as a mosaic of ecotopes, leading to transitions on the map which might be located differently in the field. There are, thus, two different aspects to the choice of physical-chemical parameters for their use in ecotope definitions: their ability to distinguish between biological communities, and the accuracy and precision by which they can be measured or modeled for each point in the estuary. These criteria are also important for the choice of alternative innovative techniques.

This study will predominantly focus on the five abiotic subdivisive parameters which are in use in ZES: bathymetry, exposure time, hydrodynamics, sediment composition and salinity. The study will focus on the question: How sensitive (or how robust) is the ecotope system towards deviations and variations in assessments of abiotic parameters, or within which error of deviations should the parameters being assessed to come to an unambiguous qualification of ecotopes?

The following questions will be answered in this study:

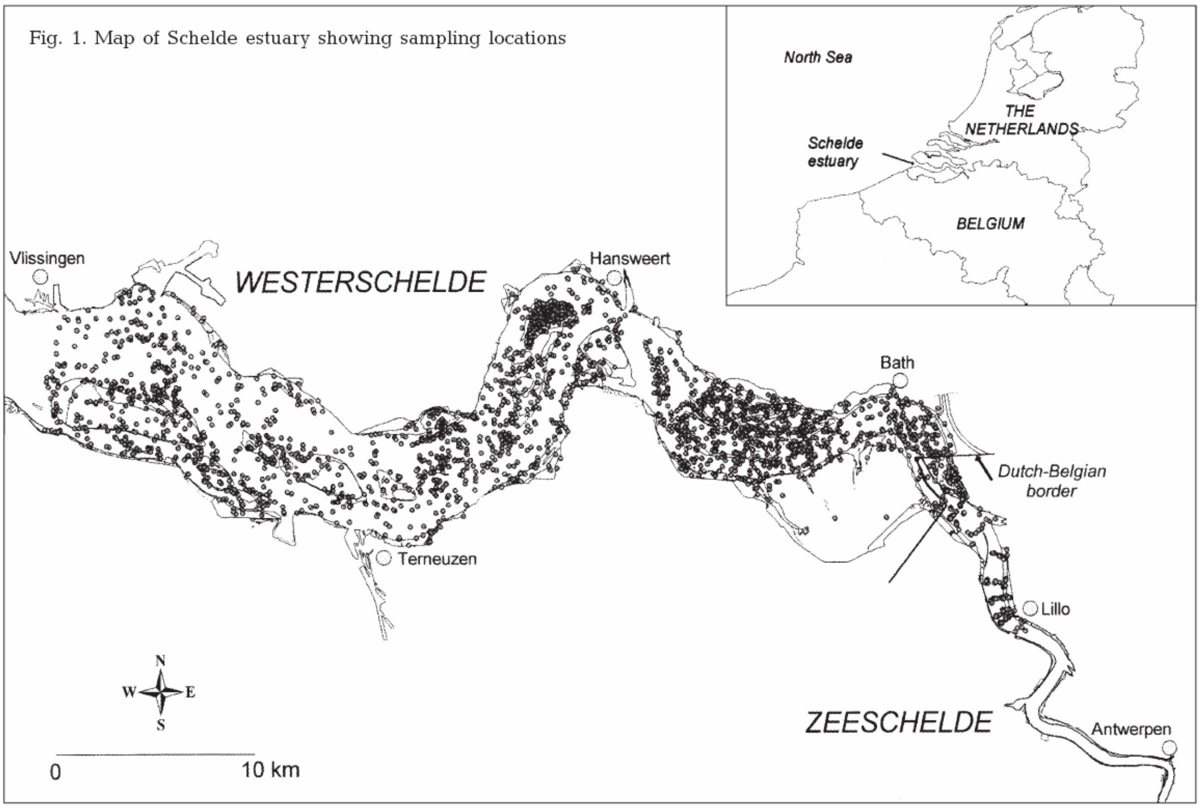
- What is the minimum essential accuracy with which each of the five ecotope parameters should be qualified and mapped?
- What is the current status of each of the parameters in relation to measurability and/or model ability?
- For which parameters do we need more accurate observations to make accurate maps?
- Are there techniques which have the potential to achieve an improvement in the short term?

2. Material and methods

2.1. Study area

As the current study is part of the LTV O&M, it specifically focuses on the Schelde estuary. The robustness of parameters can be investigated in all kind of salt and brackish waters, which might have different outcomes as total ranges and gradients of parameters differ, the relative importance of environmental parameters differ, and other species/communities might react differently to changes in environmental conditions. The Schelde estuary is a turbid, nutrient rich, heterotrophic system which has a complete salinity gradient over its total 160 km length from the mouth near Vlissingen (the Netherlands) to Ghent (Belgium) (Ysebaert et al., 2002). The current study focuses on the complete polyhaline and mesohaline part, existing of the complete 'Westerschelde' (the Netherlands) and a small part of the 'Zeeschelde' (Belgium) (Fig. 1). The average maximum difference between spring and neap tide increases from 3.8 m at Vlissingen to 5.0 m near the border. The lower and middle estuary (the Westerschelde) is a well mixed region characterized by a complex morphology with flood and ebb channels surrounding several large intertidal flats and salt marshes. Approximately 35 % of the total Westerschelde area is intertidal. Upstream of the Dutch/Belgian border the estuary is characterized by a single channel (Ysebaert et al., 2002).

Figure 1:
Map of the Schelde estuary research area showing the sampling locations (extracted from Ysebaert et al., 2002)



2.2. Data set

2.2.1. Macrobenthos samples

The data set we used for our calculations is the same as described by Ysebaert & Meire (1999) and Ysebaert et al. (2002). In total 3112 macrobenthos samples were available for calculations. Although all data are collected between 1978 and 1997, with 90% of the samples collected after 1989, the data set keeps its value as it is a well described extensive long-term set of coupled biotic and abiotic data. Most of the macrobenthos samples are collected by the Centre for Estuarine and Marine Ecology (NIOO-CEME) and the Institute of Nature Conservation, mainly in cooperation with the Institute for Marine and Coastal Management (RWS-RIKZ). Sampling occurred throughout the year, but especially in autumn and spring, and some stations were sampled several times or repeatedly in long-term monitoring programs. Subtidal samples are generally taken with either a 'van Veen' grab or a 'Reineck' box corer, and intertidal samples are generally taken as multiple sediment cores; all sieved over a 1 mm mesh. Although sampled surfaces are generally within the same range, between 0.01 and 0.023 m^2, only the presence or absence of species is taken into account, to compensate for the minor effect of sample size. This also diminishes the seasonal effect, which is much larger on density and biomass. Further only densities of >50 ind. m^{-2} are treated as present (with an exception for *Nephtys cirrosa* and *N. hombergii*, species typically observed in low densities; and *Arenicola marina*, a very large species) to compensate for accidental observations. Species with difficult determination keys were not always determined to species level. Therefore for the genera *Bathyporeia* (*B. elegans*, *B. pilosa*, *B. sarsi*), *Polydora* (*P. ciliate*, *P. ligni*) and *Spio* (*S. filicornis*, *S. goniocephala*, *S. martinensis*), all individuals were lumped and put under the one genus name. This will have an effect on the 'species' (which are probably multiple species) to abiotic parameters relations, but will be of minor importance on the community level in which we are interested. Finally 20 macrobenthic 'species' were selected on basis of their role as indicator species for macrobenthic assemblages, with different types of distributions, and contributing substantially to the total density and biomass observed in the Schelde estuary. Therefore the 'presence – absence data set' exists of the earlier mentioned 'species' and *Capitella capitata*, *Cerastoderma edule*, *Corophium arenarium*, *Corophium volutator*, *Eteone longa*, *Heteromastus filiformis*, *Hydrobia ulvae*, *Macoma balthica*, *Mya arenaria*, *Nereis diversicolor*, *Nereis succinea*, *Pygospio elegans*, *Scrobicularia plana*, *Tharyx marioni*. Further details on species and species selections can be found in Ysebaert & Meire (1999).

2.2.2. Abiotic variables

For each of the macrobenthos samples, the following abiotic environmental variables were added when available: depth (or height), salinity (model salinity and temporal salinity), current velocities (maximum ebb and maximum flood current velocities) and sediment characteristics (median grain size and mud content (fraction <63 µm)).

At the subtidal stations, depth was recorded at the time of sampling. The height of the intertidal stations was sometimes measured directly in the field, but was otherwise obtained from the RIKZ

Geographical Information System (GIS), in which all bathymetric data of the area are stored. Depth and height were recorded relative to the Dutch Ordnance Level (NAP).
Salinity was estimated for each sampling location with the 2D-hydrodynamical model SCALDIS400 (Lievense, 1994; Van der Meulen and Sileon, 1997) with a spatial resolution of 400 meters. Model calculations are based on long-term values for an average tide, under average, minimum and maximum river discharge conditions, giving the related salinity values. Salinity measurements with high spatial resolution are further called the 'model salinities'. To indicate temporal variation in salinity, also monthly to fortnightly measurements at nine stations along the longitudinal gradient are taken. These 'temporal salinities' are calculated as the average salinities of the three months prior to the sampling, which are therefore measurements at a much broader spatial scale.
Current velocities, both maximum ebb and flood current velocities were estimated with the SCALDIS100 model (Dekker et al., 1994) at a spatial resolution of 100 meters, performed by RWS-RIKZ.
In almost half of the sample occasions, besides macrofauna, sediment was also sampled, upon which grain size analysis was performed by laser diffraction. From these the median grain size and the mud content values could be calculated (Ysebaert & Meire, 1999).

2.3. Calculations

As responses of species to environmental variables are generally non-linear, logistic regression was used to model the response of species occurrence (presence = 1; absence = 0) to the abiotic parameters. In a logistic regression model, the binary response variable is related to a predictor variable through a logistic function. Using the maximum likelihood estimates of the regression parameters, the probabilities of a given state of the response variable can be calculated for different levels of the predictor variable. The probability p(x) that a species occurs as a function of an environmental variable x is then described by:

$$p(x) = \left\{\exp(b_0 + b_1 x + b_2 x^2\right\} / \left\{1 + \exp(b_0 + b_1 x + b_2 x^2\right\} \qquad \text{eq. (1)}$$

in a second-order polynomial (with b0, b1 and b2 the regression parameters), which will approximately result in a bell-shaped function (Gaussian logit curve), which is an ecologically realistic response. We have to keep in mind that theoretically a skewed or more complex response curve might also be possible for certain species; in that case we are dealing with a simplification approaching reality. The calculations have yielded response curves of each of the 20 incorporated species to each of the 7 descriptors for the abiotic environment within the ranges for the research area. The concordance of the models with the field observations was tested using the Fisher Exact test. Further details on calculations of the response curves are given in Ysebaert & Meire (1999) and Ysebaert et al. (2002).
In this study we are not interested in the response curves itself but especially in the accuracy with which environmental parameters should be measured. We approached this by investigating the derivatives of the response curves with respect to the environmental variables. These derivatives

express how the probability of occurrence (p(x) in eq. (1)) changes per unit of change in the environmental variable. The derivative of the logistic function in eq. (1) is given by:

$$p'(x) = p(x)\,[1 - p(x)]\,[b_1 + 2b_2 x] \qquad \text{eq. (2)}$$

For every environmental variable, the derivative (eq. 2) was calculated for every species over the entire range of the environmental variable. In order to reduce the output of these calculations, and to gain insight into the change of communities with changes of the environmental variable, we calculated the root mean square deviation of p(x) for all species in the community. This was complicated, however, by the fact that not all species occur over the full range of all variables, and that it does not make sense to incorporate the (zero) expected change in occurrence of species that never occur in the studied range of the environment. We therefore weighted the mean squared response of all species by their probability of occurrence, to obtain the following multivariate response indicator of the community to changes in the environmental variable:

$$PM(x) = \sqrt{\frac{\sum_{i=1}^{n} p_i(x) * p'^2_i(x)}{\sum_{i=1}^{n} p_i(x)}} \qquad \text{eq. (3)}$$

Note that the derivatives p'(x) can be positive or negative. They are squared to make them all positive. Subsequently a weighted mean over all species is calculated, and the square root of this mean gives the weighted deviation of the community to changes in the environmental variable when the latter is at value x.

In order to present these results in a useful way, especially taking into account that all environmental variables are given in different units, we finally calculated the change in environmental variable that would be sufficient to provoke a (weighted) average deviation of 5 % in the occurrence probability p of all species in the community. As PM gives the weighted average change in probability of occurrence of the species in the community per unit change of the environmental variable, the required ‘Essential Accuracy’ of the environmental variable is given by:

$$EA = 0.05 / PM \qquad \text{eq. (4)}$$

It will be clear that the boundary of 5 % average change in probability of occurrence is an arbitrary measure. However, we preferred this presentation of the results since the value EA is given in the same units as the environmental variable, and immediately provides an order of magnitude of the required accuracy in the measurements or calculations. Taking the example of depth, when EA would be in the order of several meters, it is clear that no additional effort to enhance precision of depth measurements would be needed. If it would be in the range of millimeters, it is equally obvious that no method exists that could ever yield this precision for the entire estuary. Thus, with EA, we obtain an intuitive measure of required accuracy that can be compared to known accuracy of different methods.

2.4. Possible shortcomings of the approach

Calculations in this report are based on response curves, which were themselves using the best available, but definitely imperfect, data sets on environmental variables. We use these calculations to derive desired accuracy of the measurements of environmental variables, and therefore there is a danger of circularity in the reasoning. We distinguish between three different possible flaws in the measurements to discuss their potential impact on our calculations:

1. Systematic bias in the measurements.
   If some environmental measurements would be fundamentally flawed (e.g. yielding systematic overestimations of mud (%), this would affect our analysis only very little, since they are based on derivatives. The curves of essential accuracy as a function of the value of the variable would simply be shifted horizontally along the x-axis, but the y-values would not be affected
2. Imprecise measurements – large random errors in the values of the environmental variables.
   The effect of imprecise measurements on the response curves would be to flatten these curves with respect to their ‘real’ shape. This would, in general, lower the derivatives of these curves. Thus, it is possible that we underestimate systematically the essential accuracy of the measurements because of this effect. This could give rise to the paradoxical situation that, after improvement of the acquisition of environmental data, re-fitting of response curves and repetition of the calculations on essential accuracy, an even higher accuracy would be required. However, in the absence of better measurements, it is impossible to evaluate the extent of this problem. The current analysis should be considered as a step in an iterative approach leading to a better modeling capacity of macrobenthic occurrence as a function of the environment
3. Non-incorporation of direct causative variables
   It is possible that the currently used set of environmental variables is incomplete, and that some of these variables influence the macrobenthos only indirectly. For instance, sediment composition may exert its influence through microphytobenthos growth and food availability, but the latter was not included in the analysis. Inclusion of microphytobenthos (which is one of our suggestions for improvement, see section 3.7) could then decrease the intensity of response to sediment composition. Again, the extent of this problem is difficult to evaluate at present. It should be looked at in a following iterative cycle.

Overall, we think that the estimates of essential accuracy as produced in this paper are minimum estimates. However, since the conclusions are based on a robust data set (large sample size, presence-absence data only) we do not expect that they will be fundamentally affected by these considerations. Moreover, improvement of the estimates will anyway only be possible after improvement of the environmental database.

3. Results & discussion

3.1. Bathymetry

3.1.1. Measurability and/or model ability

Depth (or height) in estuaries will clearly influence the occurrence of macrobenthic species, as this factor is related to the physical environmental stress in estuaries. Of course, there is a pronounced difference between the subtidal and the intertidal zone, because in the intertidal depth is closely related to exposure time (see details in 3.2). In both zones, however, depth relates to several physiologically important parameters of the environment, e.g. light, oxygen availability, pressure, food availability and predator presence.

At present, the input for depth-elevation maps for the Westerschelde is gathered from single-beam measurements taken in lines with a mutual distance of 200 meters, completed with RTK-GPS measurements on the tidal flats. These data are further interpolated in DIGIPOL to a 20 by 20 meter grid map (Wiegmann et al., 2005). The accuracy for individual single-beam measurements is 15 cm (2 σ) at a depth of 15 meters (which is the average depth for the Westerschelde), but shallower parts can be measured with an accuracy of up to 10 cm (2 σ). Ten measures per second are taken with a boat speed of 4 m/s. Therefore the average depth of a grid cell can be assessed based on approximately 50 measurements, which gives an accuracy of 2 cm (2 σ) per measured grid cell. However, 90 % of the grid cells are interpolated from the measured ones. It is calculated that the interpolated average depths have a precision of 50 cm (2 σ), but this can be more (even up to 1.5 meter) at steep parts or in pits or on bumps. The precision of the RTK-GPS for the measurements on tidal flats depends on the distance between the mobile receptor and the reference station. The GPS is operated with a station that can locally be placed in the field. The precision of the measurements is than 4.0 cm + 2 mm/km distance (2 σ). When executed on relatively smooth flats an interpolation accuracy of 50 cm (2 σ) is reached, (interpolation similar to single-beam measurements). Restrictions are not to the type of sediment, but more whether spots can be reached. The method is especially useful for small areas (regional use). It has to be noticed that the same report by Wiegmann et al. (2005) mentions in a summary scheme that the accuracy for the interpolation of single-beam and RTK-GPS in DIGIPOL is 40 cm; we will maintain 50 cm accuracy in our report.

Local measurements at sample sites in intertidal areas can easily be done with DGPS using fixed land-based reference stations, which give an accuracy of 10 cm in height and 5 cm in the horizontal positioning (Van der Wal et al., 2007).

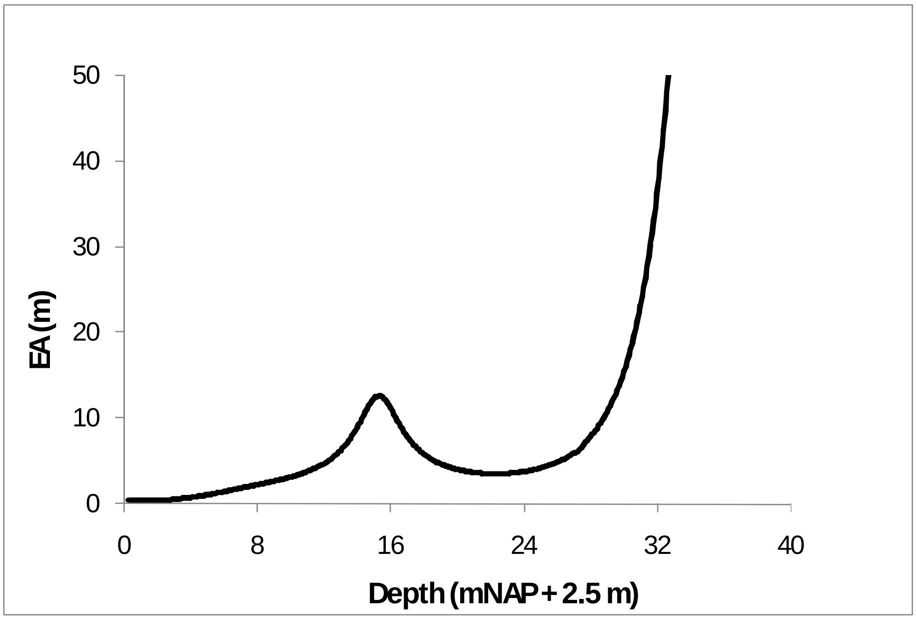
3.1.2. Essential accuracy

To make all values of measurements positive, which is necessary as we work with a quadratic term in our calculations, in an area with an approximate tidal range of at maximum 5 m in the eastern part of our research area, we measured depth as m NAP + 2.5. This resulted in a depth range of 0.3 to 58.9 m for the research area. Figure 2 shows clear differences in community responses to depth changes.

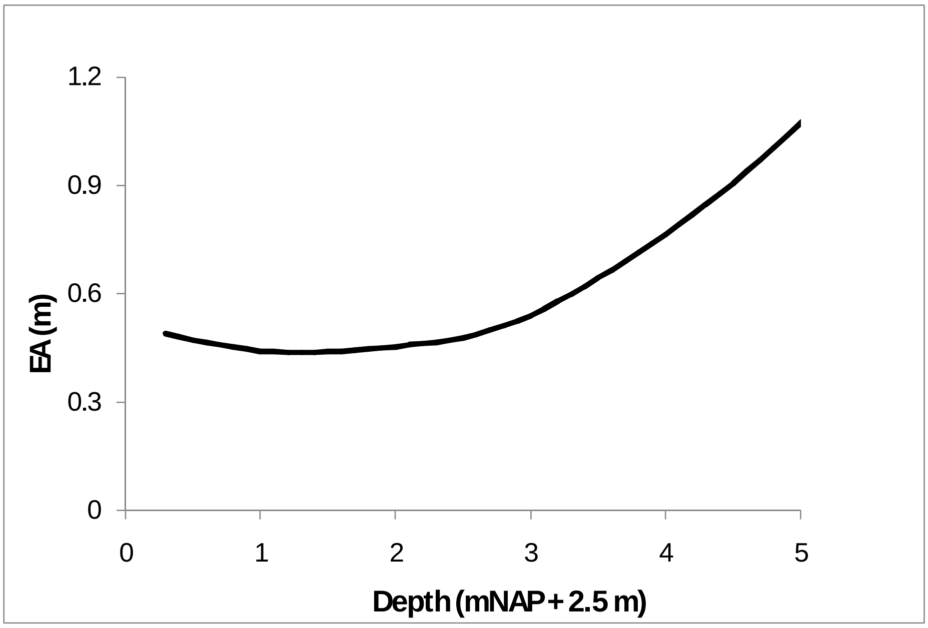
In the first 4.6 m (from +2.2 m NAP to -2.4 m NAP), this is roughly in the intertidal zone, communities change 5 %) for a depth change of 40 to 50 cm. This means that in this depth range an accuracy of

Figure 2:
Essential accuracy for depth measurements (m) as a function of depth (m) in the estuary. Note that depth is expressed as NAP + 2.5 m to make all depth measurements positive.(a) total depth range (b) focus on shallow depths

a)



b)



at least 40 cm should be achieved to keep the probability of changes in communities smaller than 5 %. Taking the current borders between ecotopes into account; NAP level, as well as the border between the litoral and supralitoral zone and the border between the litoral and sublitoral zone (which is in our graph between 4.4 and 4.7 m depth) should be determined with this 40-50 cm accuracy. However, in the sublitoral zone, accuracy of depth measurements very quickly becomes irrelevant. We find very

little changes in community with depth in this entire zone, and it can be questioned whether further subdivision of the sublitoral is required or even justified when classifying ecotopes.
Taking the current methods of single-beam and RTK-GPS measurements interpolated in DIGIPOL into account (Wiegmann et al., 2005), this means that the accuracy of interpolated measurements is generally sufficient for the whole range of depths and elevations in the Westerschelde, but is at the border of essential accuracy. Especially the interpolated elevation in the litoral zone for steep or difficult areas seems to be insufficient.

3.1.3. Potentials for improvements

As described above, the current methods of single-beam and RTK-GPS measurements and interpolation in DIGIPOL are approximately accurate enough for the whole depth range in the Westerschelde. Moreover, the probability of supply, uniformity and continuity is scored 'very high' by Wiegmann et al. (2005); see Annex 2.
An alternative method available is the interpolation in BAS (Bathymetry Assessment System), which makes use of radar (ERS-2 satellite or taken from planes) and, where possible, optical images, and a restricted number of echo loadings (single-beam). Also some current velocity data are necessary. The advantage is that single-beam lines can be taken at a distance of 600 meters which will reduce the costs. Radar images are especially useful under limited wind and strong tidal current conditions, as the mechanism of imaging is based on variations in current velocities and current directions of the tide, waves initiated by current velocities determining the roughness of the sea surface and radar scattering due to the waves (Wiegmann et al., 2005). However, the accuracy is slightly lower than for the DIGIPOL integration methodology currently in use. As BAS is especially useful for the shallow parts, the accuracy is 60 cm (2 σ) for depths till -2.5 m NAP, and 78 cm including also deeper parts. A variation on BAS standard with parallel single-beam lines, is the use of contour and a few crossing lines. The accuracy however decreases to 86 cm (2 σ) for depths till -2.5 m NAP, and 158 cm including also deeper parts, in favor of a reduction of the costs. Radar images which are currently available are those of the C-band of satellites, which are actually only useful for the tidal flats. Under ideal conditions the whole Westerschelde can be mapped, but these images are rare. Other options for the future might be the use of radar images of higher resolution and/or long wavelengths, like for instance the L-band of E-SAR images (Vogelzang, 2005), which show less noise, and therefore improve the BAS interpolation slightly. Vogelzang (2005) calculates that where interpolation with DIGIPOL over 200 m distance parallel single-beam line measurements shows an accuracy of 44 cm (2 σ), and the interpolation over 600 m; 120 cm (2 σ), the accuracy can be reduced to 66 to 74 cm (2 σ), using the C-band and 60 to 70 cm (2 σ), using the L-band combined with 600 m distance parallel single-beam lines, dependent of the enhancement factor used to create the maps.
For ecological purposes, these alternative methods are accurate enough for the sublitoral zone, but are rather too rough for the intertidal. Note, however, that in the sublitoral higher accuracy may be required for other purposes, e.g. morphological developments!

As it is in the name, multi-beam gives much more information, almost covering whole surfaces, than single-beam. Multi-beam echo-sounders scan the surface of water bottoms fan-shaped (Stelwagen et al., 1997). The precision is the same as for the single-beam; 15 cm on average for the Westerschelde for the individual measurements (Wiegmann et al., 2005). After determining the average value from the measurements within grid cells the accuracy will be about 3 to 4 cm. Due to uncertainties in transmission speed, the systematic error can be 5 cm. Multi-beam is less efficient in shallower parts, and at very flat bottoms like mud bottoms, as less signal is reflected in that case. The use of either single- or multi-beam is a trade-off between a larger efficiency and covering whole surfaces, respectively. A variation on the single-beam technique which became less popular with the arising of the multi-beam, but which is especially useful in small areas which should be mapped in detail, is the multi-channel echo-sounder, which are actually rows of single-beams measuring vertically on both sides of a ship (Stelwagen et al., 1997).

A last method is Laser-altimetry, which is roughly comparable to multi-beam in dry areas. The accuracy of measurements, which are 1 measurement per 9 m^2, is about 20-30 cm in sparsely vegetated areas with not to much differences in steepness (Stelwagen et al., 1997; Wiegmann et al., 2005). The systematic error of the whole dataset is always smaller than 5 cm, which is a big plus compared to the wet measurements. The interpolation accuracy is less than the measure accuracy, 20 to 60 cm, but seems to be about 20 cm for the tidal flats in the Westerschelde. An average value for 20 by 20 meter grids is calculated from about 40 measurements, which makes the accuracy 3-4 cm (Wiegmann et al., 2005). This is largely sufficient for ecological purposes. Due to a lack of suitable flight days; the system is vulnerable for clouds and misty circumstances (Stelwagen et al., 1997), or the delivering by the companies who fly (collect the data) of data with not enough quality; the probability of supply is scored ‘intermediate’ (Annex 2). Comparable to Laser-altimetry, there is a technique called Laser-bathymetry, using ‘green’ (530 nm) instead of ‘red’ laser which penetrates sufficiently through the water. However this technique does not behave well in turbid waters and is therefore generally not suitable for use in the Netherlands (Stelwagen et al., 1997).

Comparing the potential techniques to the current practice of the single beam and RTK-GPS combination interpolated in DIGIPOL, only Laser-altimetry will lead to a significant improvement of the accuracy. It is also applicable to the areas where current interpolation accuracy is only marginally sufficient for ecological purposes. None of the methods can be used for the whole depth range, therefore techniques should be combined. Wiegmann et al. (2005) suggests combinations of multi-beam, single-beam (to reduce the costs) and either laser-altimetry or RTK-GPS, which will slightly improve the accuracy, but will cost more.

Bathymetries used in physical models (e.g. SOBEK, ESTMORF, SCALWEST2000, KUSTZUID3 and KUSTZUID 4) are generally too rough (smallest grid cells of size 50 x 75 m^2) for use as a basis for ecological/macrobenthic interpretation (Graveland, 2005). It seems preferable to use the interpolated bathymetries that serve as input to these models, directly instead of going first through the gridding used for the models.

3.2. Exposure time

3.2.1. Measurability and/or model ability

Exposure time or the opposite (tidal inundation time) varies in the first few meters of the intertidal zone. Here a difference in effects of tidal and wave currents is related to depth by exposure time and current strength. Further, exposure time is linked to food availability, predation pressure, risk of drying, temperature variability and exposure to rainwater. In the ecotope system there are 2 sharp borders, dividing the supralitoral, litoral and sublitoral zone, respectively characterized by 'not every tide flooded', 'every tide flooded', 'permanently under water'. The sharp borders are therefore 'average high water of neap tide' and 'average low water of spring tide' (Bouma et al., 2005). The borders between the 3 zones are relatively stable as they are long-term averages of water levels. They are subject to long-term periodic changes (the longest significant period is 18.2 years), and to long-term rise in relative sea level due to land subsidence and (climate-induced) sea level rise. Shorter-term variations influencing water levels (e.g. currents, wave actions, wind conditions, precipitation and river runoff) are on average stable, although they can fluctuate from year to year. However, anthropogenic influences like waterworks in rivers and inlets, embankment constructions, and land use in inland areas can have effects on water levels. This has resulted in a mean high water level rise of approximately 0.4 cm/y (Oenema et al., 1988), or for instance a rise in maximum difference in tide between spring and neap tide near Bath of 20 to 30 cm between 1970 and 1980 (De Kramer, 2002). In general, changes will not be abrupt, and long-term monitoring at a few stations in this case along the Westerschelde will be sufficient to define the 'average high water of neap tide' and 'average low water of spring tide' levels. Of course, changes in depths will have effects on where (spatial) borders between ecotopes are situated. How to measure/monitor these is already described under 3.1.

3.2.2. Essential accuracy

Although the observed tide is built up from approximately 100 tide components, the average maximum difference in tide between spring and neap tide in the Westerschelde mouth is quite regular. The average difference between average low water at neap tide (-1.81 m NAP) and average high water at spring tide (+2.05 m NAP) is 3.86 meter in the mouth of the Westerschelde near Vlissingen (De Kramer, 2002). The average maximum difference in tide between spring and neap tide, increases land inwards to approximately 5.3 m near Rupelmonde, after which it decreases again. Near the border, the eastern border of our research area, this is approximately 5.0 m. As calculated in 3.1, the accuracy of the determination of the border should be around 40 cm to keep the changes within the communities below 5 %.

3.2.3. Potentials for improvements

Water levels can quite easily be monitored at some stations along the Westerschelde to show the 'average high water of neap tide' and 'average low water of spring tide' there, from which a long-term trend can be calculated, which can be extrapolated to the whole Westerschelde. The precision of measurements is more than sufficient for ecological purposes. However, as some systematic basal knowledge of the Schelde tide is lacking, the predictive models to calculate effects of anthropogenic influences, like waterworks and dredge and pours activities are not very accurate (De Kramer, 2002).

3.3. Hydrodynamics

3.3.1. Measurability and/or model ability

The hydrodynamic conditions include current velocities, bed shear stress and wave action. These factors are important for transport of sediment, food and juvenile macrofauna (larval settlement and post-settlement transport), but also influence sediment stability affecting macrofauna. In ZES, three parameters are distinguished, determining whether a location has high dynamics, low dynamics, or no dynamics (stagnant waters). In the case of the Westerschelde, we only have the two dynamic classes. Deviations are made based on fetch length for waves ('strijklengte'), the maximum linear current velocity, and the maximum orbital velocity. The first descriptor is actually a descriptor of the space of open water surrounding a location at a coast or embankment, which determines the uninterrupted distance over which the wind can freely interact with the water surface, resulting in wind waves. According to Nienhuis (1976), locations at which fetch length in a corner of at least 20 degrees reaches 80 to 240 kilometers, are considered semi-exposed and are classified to have high dynamics. As there are no such locations east of the line Vlissingen – Breskens, this descriptor is unimportant for the Westerschelde, and only plays in the ebb-tidal delta. The maximum linear current velocity, the second descriptor discriminates high from low dynamic locations at a theoretical-physical value of 0.8 m/s, assessed during an average spring tide, independent of whether it is high or low tide.
The linear current velocity is the only descriptor of hydrodynamics in the sublitoral zone. In the litoral zone also the third descriptor, maximum orbital velocity determines the classification. The discriminating value of this descriptor is 0.2 m/s. It is (should be) measured in the field. The velocity is four times lower than for the linear current velocity as wave action is much more irregular causing turbulence (Bouma et al., 2005). An area is classified as high-dynamic when at least one of the three descriptors (fetch length, maximum linear current velocity or maximum orbital velocity) is above the border value. When this is not the case, it is classified as low-dynamic.
It appears that in practice the border of 0.2 m/s orbital velocity is seldom reached in the Westerschelde, except for the mouth which was already classified as highly dynamic by fetch length. For a small area with a relatively uniform wind field and no significant wave propagation from elsewhere, fetch is the dominant factor determining wave climate, and a high degree of correlation between the two measures can be expected.
With respect to linear current velocity, there are two different problems in the Westerschelde. First, there is some doubt about the best critical values. De Jong (1999) suggested that the border between high and low dynamics should be at a lower value for linear current velocity, at least for the Westerschelde, where it was found that 0.5 m/s might be more suitable. Second, there is the problem of how to measure or model the appropriate field values, especially in shallow areas where they are the most relevant.
Current velocities are now calculated with hydrodynamics models like DETWES, E-WESTII and SCALDIS100 and SCALDIS400 which are extensive versions of the first mentioned. The models are

integrated 2D hydrodynamics models with which water levels, current velocities and water- and sediment-transport can be modeled. In the Westerschelde, the SCALDIS models, model to a grid of 100 x 100 meters. Calculations are calibrated on water level time series measured at fixed stations along the Westerschelde. Freshwater input is included as decade averages for four larger entries (Schelde Gent + Rupeltak, harbor Antwerpen, lake Zoommeer, and Channel Gent – Terneuzen), and the model is based on a bathymetric map constructed from loadings. An initial salinity distribution has to be provided as input, otherwise it takes months before a salinity distribution related to the hydrodynamics has been installed (Dekker et al., 1994).

Calibration of hydrodynamic models on observed water levels implicitly overemphasizes the calibration in the deep channels compared to the shallow areas. Deep channels transport almost all the water. Models are much more sensitive to calibration parameters (mainly roughness of the bed) in these channels than to parameters characterizing the shallow areas and intertidal flats. Therefore, discrepancies between observed and modeled current velocity are mainly noticeable over the tidal flat. The problem is difficult to tackle in general, and in the Westerschelde in particular it is aggravated by the lack of observed data of current velocity over tidal flats. There is obviously a need for such series. Current velocities at sample sites, especially in the intertidal zone where equipment can easily be installed during low tide, can be measured accurately using EMF (electromagnetic current velocity meter) (e.g. van der Wal et al., 2007; Bouma et al., 2005) or ADCP (Acoustic Doppler Current Profilers) equipment. In order to resolve the vertical gradients in current velocity, several EMF lined vertically are needed per measurement station. ADCP measures current velocity over the vertical, but has the disadvantage of a 'dead zone' close to the instrument. The extent of the dead zone is a function of the total depth resolved. Alternatives are meters (e.g. ADCP) attached to boats, with which transects can be measured (Bell et al., 1997). Drogues, which are tracked at certain intervals, can also be used (e.g. De Jonge & van Beusekom, 1995), but the locations measured are dependent of the currents guiding the drogues through the waters and the probability of hitting shallow areas is not very large.

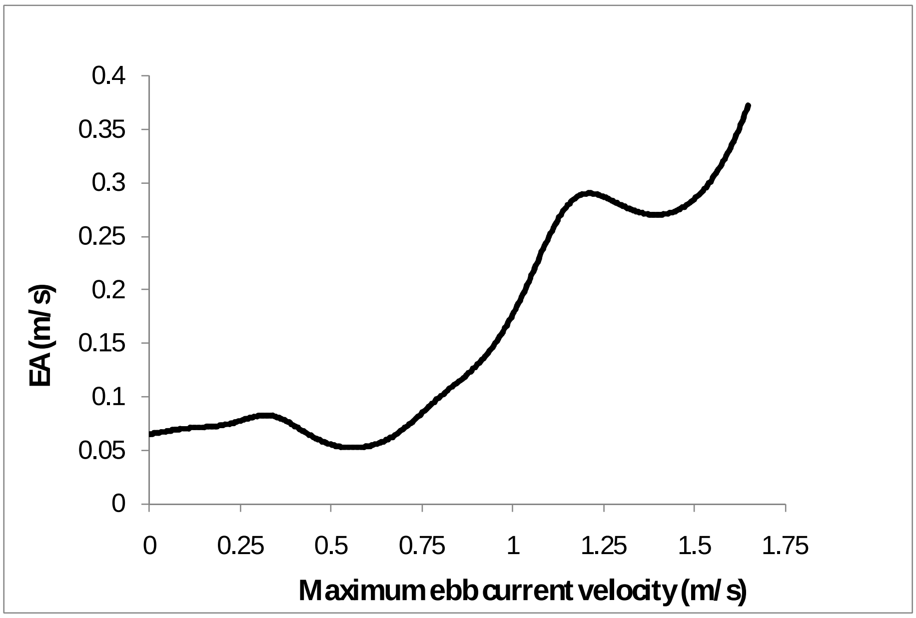
For all field measurements, it should be noted that they can only provide information on a limited number of sample points, and that major efforts would be required to measure at many points simultaneously. Given the state of development of hydrodynamic models, it seems more reasonable to use field measurements as calibration data for the hydrodynamic models. Serious improvements of the behavior of these models over shallow and intertidal areas should be possible, given a sufficiently large and diverse set of measurement points, preferably over at least one spring-neap cycle per point.

3.3.2. Proxies for current velocity

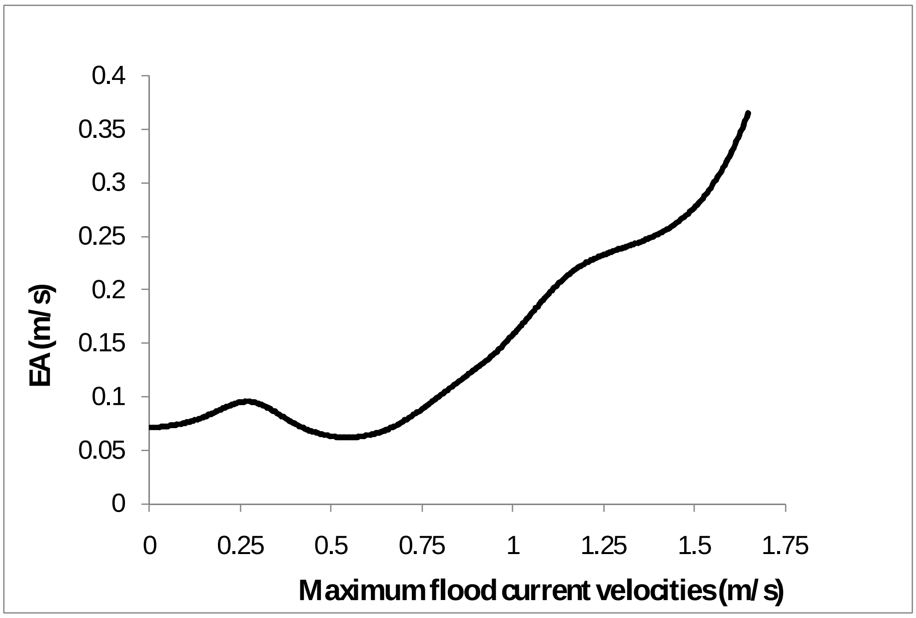
Given the perceived inadequacy of hydrodynamic models, and the lack of calibration and validation data on direct current velocity measurements, proxies have been used in the past to estimate how dynamic areas are. These proxies are based on morphological characteristics of the bed. In particular, the presence of sand mega-ripples has been used. Optical inspection, via aerial pictures taken at water levels as low as possible, has been the preferred method to obtain area-covering indices of dynamics. At low-dynamic circumstances the surface will be almost flat, whereas under high dynamic

Figure 3:
Essential accuracy for current velocity estimations (m/s) as a function of current velocity (m/s) in the estuary. (a) maximum ebb current velocity (b) maximum flood current velocity

a)



b)



conditions (mega-)ripple patterns become visible. From these also shallow neighboring parts which are not visible in pictures are interpreted as low or high dynamic, which involves of course an uncertainty (Bouma et al., 2005). It is not known how well these optical interpretations correspond to real current velocity in the field, and there seems to be little point in correlating them to (unsure) model output, since this will not reveal what causes possible mismatches. Also, it remains uncertain at what current velocity sand mega-ripples start to occur, nor whether this is a similar current velocity at all sites. The use of optical inspection is a rapid and reliable method in itself, but more study is warranted to properly calibrate it. We cannot analyze the required accuracy of the method at present.

3.3.3. Essential accuracy

As current velocities at locations will be dependent of the tide, which means that at a certain location the ebb tide can be stronger than the flood tide, or the other way around; both current velocities are related to shifts in communities in Figure 3. The response of communities to maximum ebb and flood current velocities appears to be highly correlated as is shown by the similar trends for the two. The essential accuracy for current velocity is in the order 5-10 cm/s in the range of current velocity below 80 cm/s. For higher current velocity, less accuracy is required.

At the transition between high and low dynamic ecotopes which is in current use (0.8 m/s), the essential accuracy is 0.1 m/s. Accuracy should be a bit better when the transition point would be changed to 0.5 m/s, but the difference is not large.

It has to be kept in mind, that the effect of current velocity on macrobenthic communities is related to the sediment type. Not only will sediment often reflect the current velocity, but in one type of sediment, a certain velocity can result is sediment turbulence, while in the other sediment, the bottom will remain stable (Warwick & Uncles, 1980).

3.3.4. Potentials for improvements

Dekker et al. (1994) mentions in the evaluation of the SCALDIS100 model that current velocity measurements can deviate about 5 to 10 % from reality. After modeling, the deviation will be 10 to 20 % and can be up to a 100 % on flats and near gullies. Our calculations, which are actually also based on the same model, suggest that deviations in the order of not more than 5 to 10 % (depending on the value of the current velocity – the percentage is higher for lower predicted current velocity) would be required for consistent prediction of the community composition. This is at, or below, the accuracy which present models can provide.

Other models which can calculate current velocities and/or turbulence are SCALWEST2000 and DELFT3D. Both models can get to grids of 50 x 75 meters, however the models do not behave very well in shallow waters, and the scale does not seem to be sufficient for ecological interpretation (especially for turbulence) (Graveland, 2005). In Graveland (2005) it is also suggested to improve the outcome of SCALWEST2000 by fixing the current velocity for the mud flats and the sand flats with (mega-)ripples; i.e. to include visual observations. When ripples are indeed a good representation of the current velocities, then SAR (Synthetic Aperture Radar) imagery is potentially a good technique to scan larger (intertidal) areas for current velocities as there is a nice correlation with SAR backscattering (Van der Wal et al., 2005).

We suggest that the most promising improvement is to collect a relatively large and consistent field data set on current velocity on tidal flats and in shallow areas, so that a qualitative improvement of the calibration of the models becomes possible. We think that this should have preference over the refining of the modeling grids. It may involve incorporation of sediment composition data into the model calibration, as a means to better spatially resolve differences in bottom roughness. Improving (e.g. by nesting) the bathymetric gridding for shallow areas only, may also be a way to improve the model predictions. Such an effort would especially be worthwhile when combined with efforts to improve the modeling of mud transport in the estuary.

As mentioned above, it is difficult to independently validate the usefulness of optical detection of sand mega-ripples as a proxy for 'dynamics'. However, the presence of these mega-ripples clearly influences bottom roughness and therefore hydrodynamics. As the extent of mega-ripple fields is relatively easy to determine from aerial photographs, we suggest that this information could be incorporated in efforts to improve model accuracy by locally adapting bottom roughness.

3.4. Sediment composition

3.4.1. Measurability and/or model ability

Sediment composition is often described as a measure of sediment grain size and a measure of mud content. Both are the result of sedimentation, erosion and transport processes and are therefore linked to prevailing currents in combination with (river) supply. Mud content might be related to the organic fraction of the sediment, and therefore to food availability for several species (e.g. deposit feeders). Further sediment type will influence the efficiency of food uptake and burrowing and tube building possibilities. The impact of currents on macrobenthic communities is also related to the sediment type, not only because currents influence sediment distributions, but certain bottoms are also more stable than others (Warwick & Uncles, 1980), which leads to differences in the impact of the same hydrodynamics. Also macrobenthos itself has an effect on sedimentation and erosion, leading to differences in these processes dependent of the prevailing communities.
In the current ecotope system, environments are first split in hard and soft-bottom substrates. Here we focus only on soft-sediment conditions. Second, sediments are classified in muddy and sandy substrates with different median grain sizes. Borders of different classes in median grain sizes are 250 and 2000 μm and the border for mud content (< 63 μm) lies at 25 % for the Westerschelde (for the North Sea the border lies at 10 %). The resulting ecotopes are: muddy (mud content ≥ 25 %); fine sand (median grain size ≤ 250 μm, mud content < 25 %); coarse sand (median grain size 250 - 2000 μm, mud content < 25 %); and gravel (median grain size > 2000 μm, mud content < 25 %) (Bouma et al., 2005). In principle it is quite easy to determine to what type a sediment sample of a sample site belongs, however, the methodology appears to be crucial. The sediment of the top 10 cm should be analyzed. Then, there are various ways to treat the sediment before analysis, especially those using acids or peroxide to distract the particles from each other and to dissolve chalk and salts. In Bouma et al. (2005) it is stated that none of these pre-treatments should be used. Nothing is said about ultra-sonic baths used in several laboratories to standardize the treatment of the samples, as samples can be delivered to labs in various stages from wet to dry, which influences the aggregation status.
Traditionally, samples were measured using sieve and pipette methodology. Nowadays, several types of instruments are in use, which can deliver a whole range of distributions of the grain sizes available in the sample. A problem, however, is that there are large deviations between the various instruments, especially concerning small grain sizes. Bouma et al. (2005) state that samples should be analyzed on the Malvern 2600L Laser Particle Sizer, or at least that when other instruments are used, a series of samples should be measured on both instruments, which makes it possible to recalculate the results to Malvern 2600L LPS standards. There will however still be deviations due to differences in adjustments and sensitivity of different instruments, even of the same type. When available, mud

contents of summer and autumn should be used, as there is some seasonality in prevailing mud levels, which are generally highest in autumn.

Several hydrodynamic models like DELFT3D-MOR (ZEEKENNIS), SOBEK- MOR, ESTMORF do also include a sedimentation or sediment distribution module, however scales seem to be to large, and sediment types are not always specified in detail to use them for ecological estimations. Further there are several numeric mud content models (1D, 2D and 3D) for the Westerschelde, which have been used within LTV O&M. However, their spatial scales deviate with an accuracy of several tens of kilometers (1D) to one kilometer (3D), which does not reach the accuracy necessary to assess present communities. The relative accuracy of these models is estimated on 50 % in mud content (Graveland, 2005).

3.4.2. Essential accuracy

Although both sediment parameters, median grain size and mud content, are usually well correlated, the ranges of both are analyzed for changes in communities. Median grain sizes in the research vary between 16 and 664 µm (Ysebaert et al., 2002). Around a median grain size of 190 µm, the EA is smallest, in the order of 20 µm. This point more or less corresponds to the transition between pure sands with very low mud content, and muddy sands, a real transition point in the communities. For coarser sands, the EA increases. Communities in coarse sands are relatively species-poor, and it seems to matter less how coarse the sand is.

A different pattern is shown for mud content. Apparently, communities are very sensitive to the difference between no mud and a little mud in the sediment (range 0-20 % mud). Small changes in mud content in this range are reflected in considerable changes in communities. Around 50 % mud the EA is much larger, indicating that a difference between e.g. 40 and 60 % mud is much less important than the difference between 0 and 20 %. This increase of the EA is intuitively very understandable. The decrease again of the EA with further increasing mud content is caused by the fact that many species have a steeply decreasing probability of occurrence at the very high mud contents. In the Westerschelde these are usually compact clays without a sand component, and a very inhospitable habitat for most species.

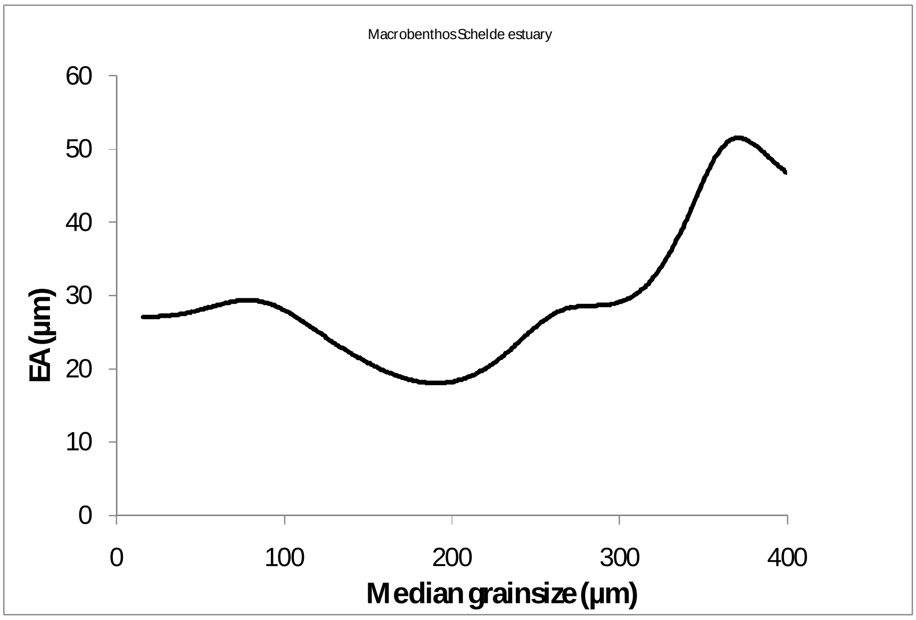
The current border in use between ecotopes lies at 25 % mud content. This means that an accuracy of around 4 % in estimating the mud content should be reached. Analytical precision (replicate measurements on the same well-mixed sample with the same instrument) is probably in the range of a few percent mud content, but small-scale natural variability (scales of cm to m) is definitely higher than the accuracy apparently required.

3.4.3. Potentials for improvements

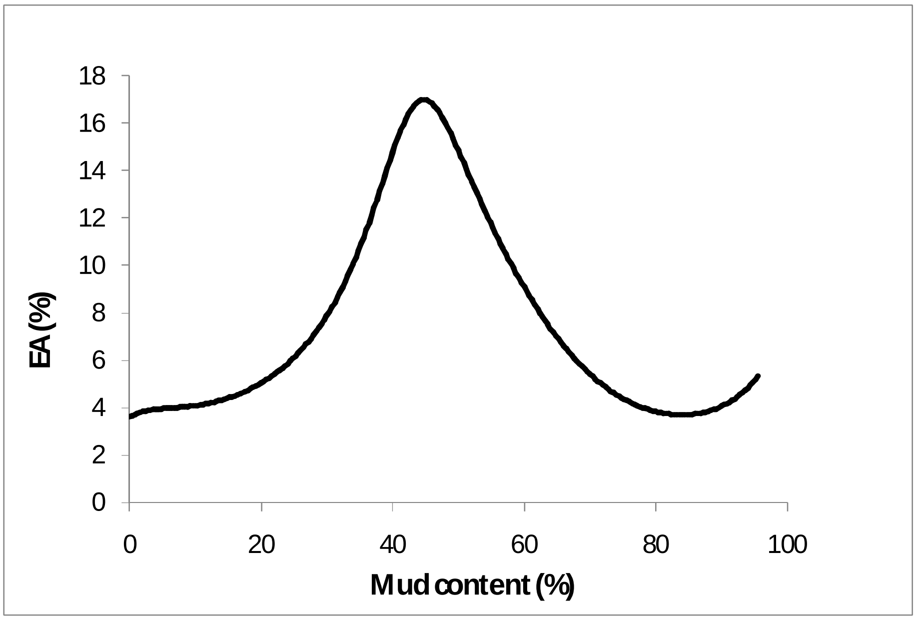
There are potentials to monitor larger areas on sediment composition, but those will never be more accurate than measurements (with laser-diffraction techniques) in soil samples, as those function as the basis. Different methodologies can result in differences in measurements of several percentages. When reference measurements reach certain accuracy, the use of SAR (Synthetic Aperture Radar) imagery in intertidal areas might be useful for interpolation; e.g. ERS SAR (satellites of the European

Figure 4:
Essential accuracy for sediment composition measurements as a function of sediment composition in the estuary. (a) Median grain size (µm) (b) mud content (%)

a)



b)



Space Agency with radar onboard). Van der Wal et al. (2005) showed significant negative correlation between backscattering of the radar (C-band) and mud contents of the sediment, and significant positive correlation with median grain size of the sediment. However, correlations are mainly the result of correlations with surface roughness; which is the ripple structure of the bed. Also surface water, interstitial water, salinity and temperature have to be taken into account, as those also have an effect on the backscatter of the radar, however via either the vertical roughness, correlation length or the dielectric constant. Improvements of the predictive power can probably be reached by combining images with different polarization and incident angles. In the study for instance 69 % of the variation in mud content was explained by the model. This means that the accuracy might not always be sufficient to assess the community, but certain borders, for instance the 25 % mud content border might be

detected quite accurately. As shown in Van der Wal & Herman (2007), even better results might be obtained from the combination of SAR and remote-sensing, either optical (near-infrared; VNIR) or shortwave infrared (SWIR) reflectance. The results were very useful within the same area for time series, but should be validated and calibrated for other areas. Best results were obtained with the combination ERS SAR and CASI (carried by plane), but also the combination ERS SAR and Landsat TM (satellite) appeared to be useful. Best results are obtained when ground-truth data and image data are acquired at the same time, or within a limited time-span. Also the combination ERS SAR and field spectro-radiometer (e.g. TRIOS RAMSES) will be accurate, but only limited areas or local spots will be covered with the field spectro-radiometer, making interpolation necessary, giving additional uncertainties. Using multiple sensor observations, 43 to 74 % of the variation in median sediment grain size can be explained.
The absolute error of mud (%) estimations by remote-sensing is difficult to establish, as Van der Wal and Herman (2007) used log-transformed mud (%) whereas the logistic models use untransformed percentages. However, some order-of-magnitude estimation is possible. The standard error of single sample estimates from remote sensing is in the order of 0.6 for ln(mud). Comparing this to the values of ln(x+EA)-ln(x) for a range of mud percentages x, learns that the accuracy of the remote sensing estimation is only sufficient for small mud percentages (< 10 %), but approximately a factor 2 higher than essential accuracy for the remainder of the range. However, for area-covering estimates, the accuracy decreases rapidly with distance from sample points (see below, section 3.6.3) and thus the remote-sensing accuracy will quickly be higher than interpolated accuracy from scarce sample points. The best strategy seems to combine remote-sensing with field sampling, and use the field samples to constrain the remote-sensing estimations.

3.5. Salinity

3.5.1. Measurability and/or model ability

In estuaries salinity is one of the most deterministic factors for the occurrence of a certain macrobenthic species. Species are due to their physiology restricted to a certain salinity range, and are therefore typical freshwater, brackish water or salt water species. There is a strong relation between salinity and species richness, which is first described by Remane (1934), as relative to fresh- and salt water conditions, only a few species occur in brackish waters, typically around a salinity of 5 to 7. Not only the (average) salinity level, but also the salinity variation or minimum salinity determines which species can be found. In the current ecotope system, salinity is included as year average salinity at high tide with average freshwater discharge (especially river discharges). Salinity variation is included as a calculated value of salinity variation over the same year at high tide by; ((4 x standard deviation salinity) / average salinity) x 100 %. Water with an average salinity above 18 is classified as salt water, while salinity between 5.4 and 18 is classified as brackish water. In practice, there are often just a limited number of salinity measurements available for certain locations. In that case the salinity variation can be calculated by ((maximum salinity - minimum salinity) / average salinity) x 100 %, in which average salinity = (maximum salinity + minimum salinity) / 2. When the difference between maximum salinity and minimum salinity is larger than the average salinity, the salinity variation is classified as large; otherwise the salinity is classified as small (Bouma et al., 2005). As salinity variation is expected to be more important than average salinity when there is a large variation (e.g. Attrill, 2002), three classes are present in the current ecotope system; brackish water with low salinity variation; salt water with low salinity variation; and brackish or salt water with high salinity variation. Salinity can be measured relatively easy using a salinity meter, which should be done at the bottom during high water. However, several measurements, ideally a time series, are necessary to estimate the average salinity during high tide at average freshwater discharge, and this also accounts for the estimation of maximum and minimum salinity, to determine the variation. Salinity is measured periodically (monthly) at several stations and continuously at a few stations, after which measurements can be interpolated in hydrodynamical models. The 2D model SCALDIS400, on which the data used in this study (model salinity data) are also based, estimates to a grid of 400 x 400 meters which might be sufficient for ecological interpretation, as average salinity is not very variable on a spatial scale (Ysebaert & Meire, 1999). Also other hydrodynamic models, like SOBEK can be used. However, such models do not include temporal (seasonal) variation in salinity. Further, they neglect 3D effects, and might therefore underestimate the effects of for instance dredging activities on salinity distributions. 3D Modeling is therefore recommended by Graveland (2005), but we doubt whether 3D-effects, with vertical variations in the Westerschelde of only a few PSU units, are really relevant for the purpose of ecotope modeling. Variation in salinity only partially takes minimum salinity into account, although especially minimum salinity (for instance at extreme discharges) might be crucial for several species, and can be missed by the current way of modeling.

3.5.2. Essential accuracy

Especially in the brackish zone (salinity below 10) a fairly accurate modeling or measurement of salinity is called for. Essential accuracy is in the range 1-2. The occurrence models are more sensitive to the modeled salinity than to the ‘temporal’ or measured salinity. The effect of salinity on the distribution of species is mainly important at a large scale: species have distinct preferences for low, intermediate or high salinity and zonation in the estuary is accordingly.

Both models and observations seem to be able to produce salinity estimates well within the range of required accuracy. Analytical precision in salinity measurements is far below 1 PSU unit. Measurements are moreover based on very frequent (every ten minutes) time series, so that averages have a far higher precision even than individual observations. Also for models, predicting average salinity within a range of 1 should be feasible, since salinity distribution is an important calibration parameter for the models.

3.5.3. Potentials for improvements

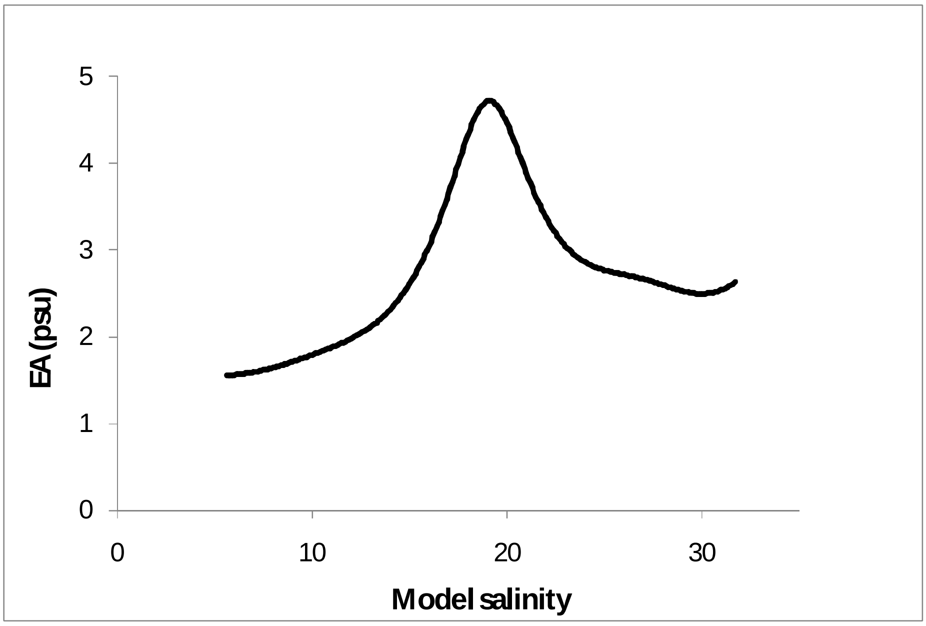
For the two variables investigated here (model salinity and observed salinity prior to the sampling), little improvement in the estimates seems to be possible or needed.

Improvement however seems possible in the use of salinity data for modeling ecotopes. This applies in particular to measures of salinity variability. Existing data also should permit to better quantify temporal variability of salinity, based either on direct observations at the fixed points in the

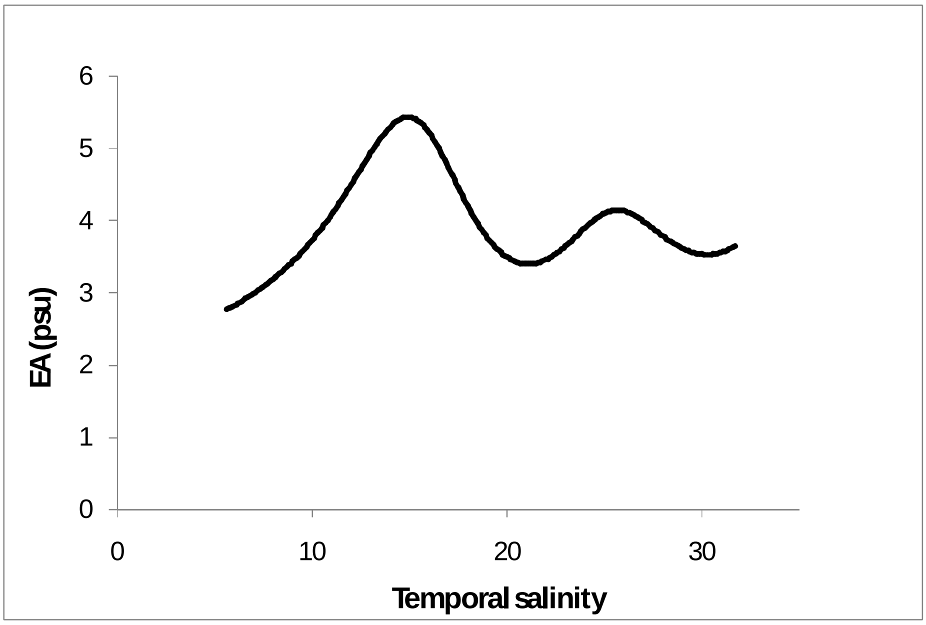
Westerschelde, or on model runs under different forcing scenarios. This analysis has not been performed. It would, be worthwhile to extract variability of salinity at different time scales from the observed series, and use these as new independent variables to explain occurrences in the same data set.

Figure 5:
Essential accuracy of salinity measurements or estimations (-) as a function of salinity (-) in the estuary. (a) Model-derived (time-averaged) salinity (b) measurement-derived actual salinity

a)



b)



3.6. Correlations and their effects

Correlations may affect the results presented here in several different ways, which we will discuss shortly.

3.6.1 Are the univariate-based measures of essential accuracy valid in the multivariate case?

The first potential problem is that measurements of environmental variables are not independent of one another. In our data set, significant correlations were found between many of the variables. An overview of significant correlations in several studies in the Westerschelde can be found in Annex 3 to this report.

These correlations may influence the range given above for the essential accuracy of the environmental variables. One can imagine that, if most environmental variables have been measured accurately, but one variable has a value deviating from the real value, then the effect on the overall prediction might be limited if one uses a multiple logistic regression curve. We have tested this hypothesis, using the multivariate regression curves established for the same data set that was also used previously.

The procedure was as follows. For a sample of species (we used *Arenicola marina, Bathyporeia spp., Cerastoderma edule and Corophium volutator* – the general picture did not differ much between the species so we stopped adding more species) we established the partial derivatives of the multiple logistic regressions with respect to all variables included in these regressions (note that these regressions were established using a stepwise procedure, so not all variables are always included in the analysis). These partial derivatives are given as:

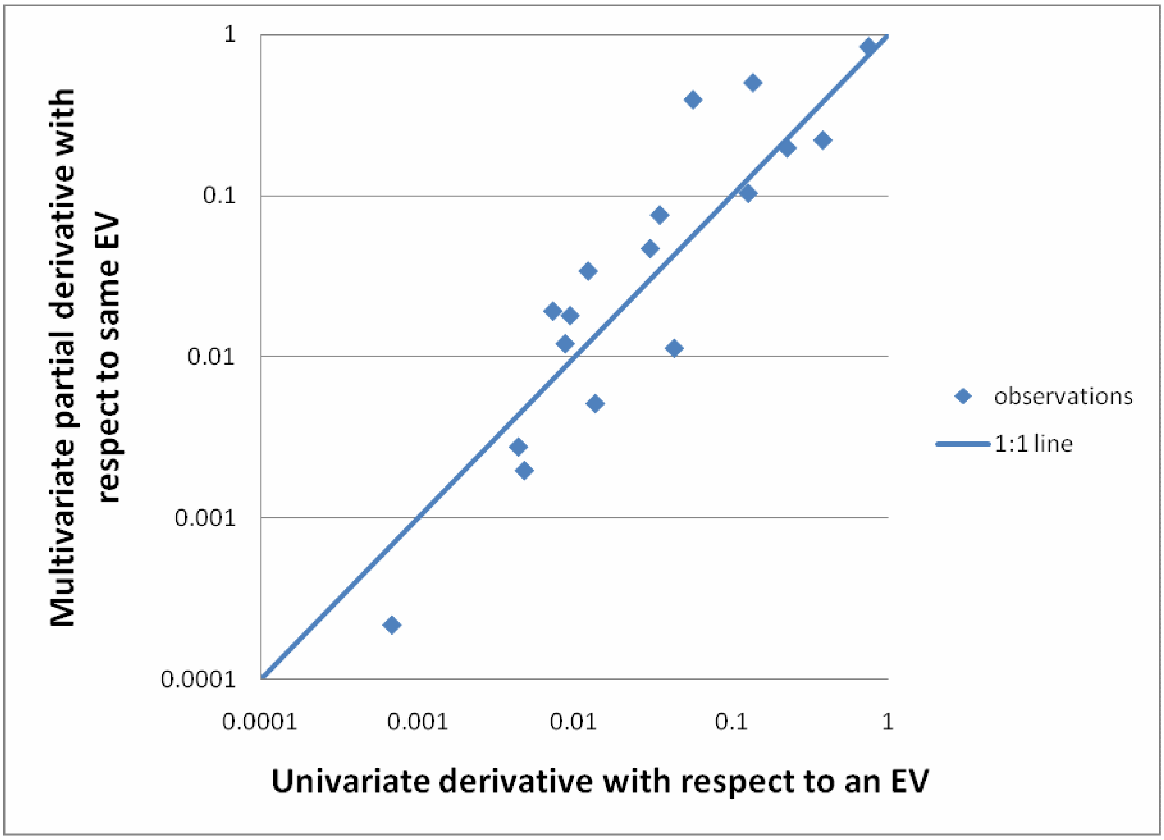
$$\frac{\partial\ p(X)}{\partial\ x_i} = p(X)\,[1 - p(X)]\,[b_{i1} + 2b_{i2}x_i]$$

Where X denotes the vector of environmental variables included in the analysis ($X = [x_1,..x_p]$), x_i is the particular environmental variable under study, and it is assumed that deviation in the measurement of x_i is independent from deviations in the other parameters, i.e. $\frac{\partial\ x_j}{\partial\ x_i} = 0$ for i different from j. This is the situation of a measurement error in one variable, while the others are not affected by this error.

For each of the species and each of the relevant variables, we compared the partial derivatives with the derivatives of the univariate regression for that variable, as used above in this report. As the (partial) derivatives depend on the value of all environmental variables, we performed the analysis for a typical intertidal situation, characterized as: temporal salinity = model salinity = 15; depth = 2 (0.5 m above NAP); maximum ebb current velocity = maximum flood current velocity = 0.4 m/s; mud = 20 %; median = 125 μm.

The resulting graph of partial derivatives versus derivatives (Fig. 6) shows that most values are close to the 1:1 line. Thus the accuracies determined on the basis of the univariate regressions also seem to be valid for the multivariate case.

Figure 6:
Comparison of partial derivatives of multiple logistic regressions with the corresponding derivative of the univariate regression model. All relevant environmental parameters for four selected species are used (see text for details). Line is the 1:1 line



3.6.2. Are all environmental variables equally needed?

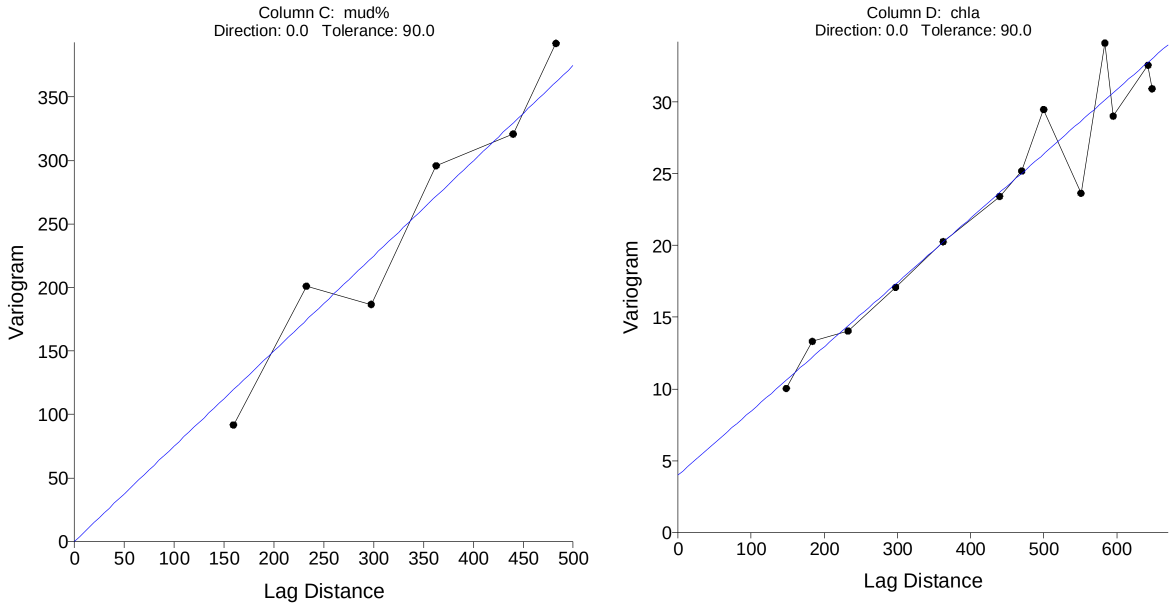
A second aspect of correlation is that environmental variables contain information about each other. When a sample point is characterised by sediment type, height and salinity, it is possible that maximum current velocity does not add much new information and therefore does not improve the predictions made substantially. In this way, it could be possible to eliminate expensive environmental variables without impeding the quality of community predictions very much.

An overview of the multivariate logistic regressions, obtained by stepwise procedures, learns that all environmental variables have been used for at least five or more species. No obvious 'superfluous' variable could be found. In individual models, it is however found that mud (%) and median grain size occur seldomly together. The same is true for maximum ebb and maximum flood current velocity. However, since these variables come as pairs from one analysis, no reduction of costs is obtained by choosing one and dropping the other. For reasons of parsimony and consistency of the prediction models, this could however be a simplifying option in the modeling process. We conclude that the presently used set of environmental variables is a minimum requirement for ecotope classification.

3.6.3. The importance of spatial autocorrelation in environmental variables for ecotope mapping

A final aspect of correlation between environmental variables is the spatial correlation structure. Environmental variables are correlated spatially, implying that a measurement at one particular point also contains information on surrounding points. The typical pattern of autocorrelation is shown by the variograms of mud content and chlorophyll-a concentration for samples in a dense grid on the Molenplaat (Westerschelde) in Figure 7.

Figure 7:
Variograms of mud (%) (left) and chlorophyll-a (right) based on a sampling grid over the Molenplaat in September 1995. 78 samples were used, the basic grid spacing was approximately 150 m

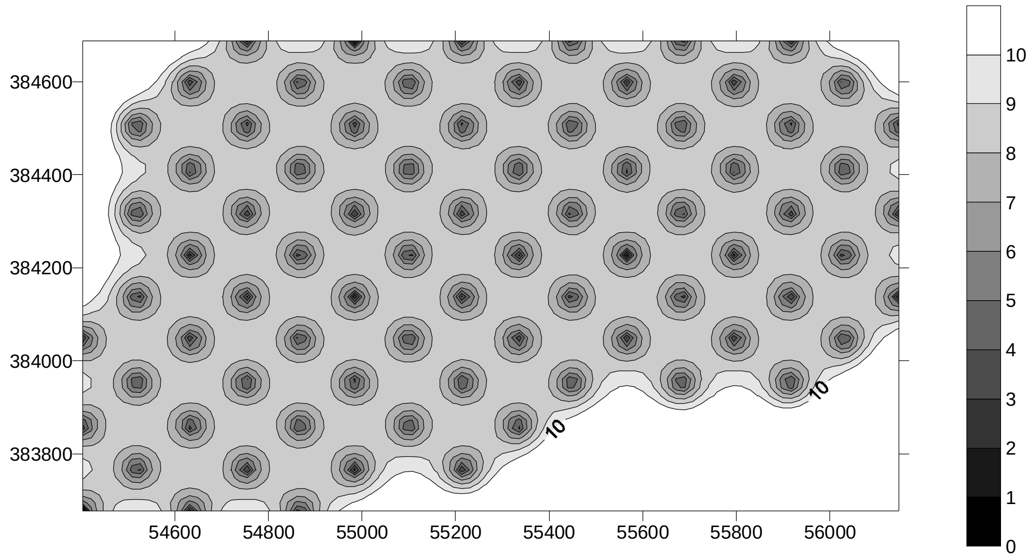


The variogram represents, roughly speaking, how the average difference between sample pairs increases as a function of the physical distance between these pairs. It can be seen that for mud (Fig. 7a), this difference tends to zero as the distance goes to zero: based on this sampling grid at a 100 m scale, we expect replicate samples very near to one another to be very similar. This is usually (but not always) the case. The variogram for chlorophyll-a (Fig. 7b) does not tend to the (0,0) point. It shows a so-called ‘nugget effect’, that is, small-scale variance between samples taken near to one another. Both variograms show that samples become more dissimilar as the distance between them increases. Eventually, this must stabilize at the large-scale variance, but the sampling grid in this instance was not sufficiently large to reach that point.

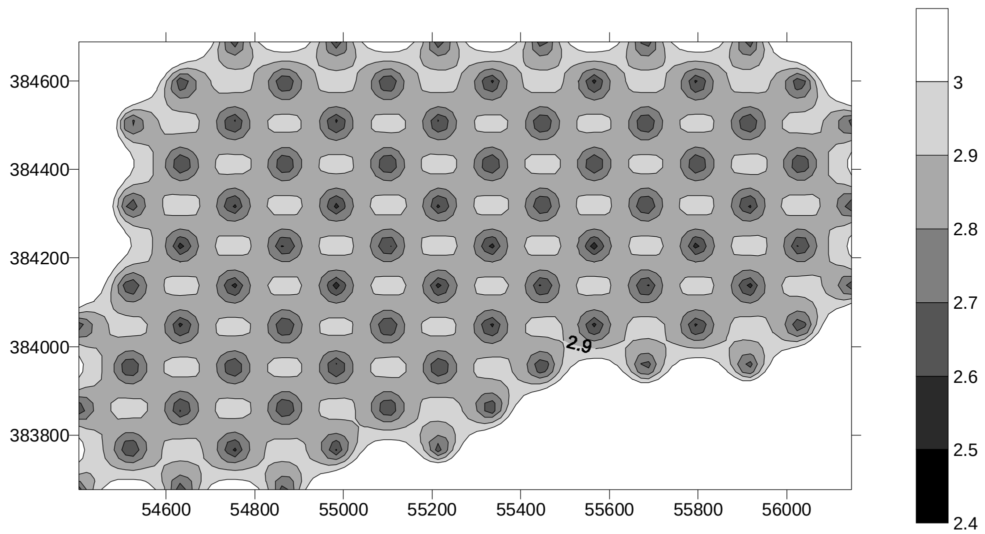
This type of analysis has implications for sampling programs aiming at producing consistent maps of environmental variables, as is required for an ecotope classification. Two requirements are to be met for field programs: samples should be optimally located, and a minimum number of samples should be taken to ensure a certain quality of interpolation, i.e. a certain accuracy of the interpolated points. Figure 8b shows the standard deviation of the interpolated values of chlorophyll-a of the Molenplaat, based on kriging using the variogram in Figure 7b. In this case a regular sampling grid was used. It can be seen that the standard deviation of the interpolation estimate is minimal around the sampling points (it attains a value there corresponding to the square root of the nugget variance), and maximal at the points in between the samples. How fast this standard deviation increases with distance from the samples, is determined by the degree of autocorrelation in the data set, as expressed by the variogram. For comparison, Figure 8a shows the standard deviation of the kriging estimates of mud percentage for the Molenplaat. Standard deviations are very low at the sampling points, but rapidly rise to about 10 % (mud content) in between the points. Compared to the essential accuracy of mud percentages, even this dense grid of observations does not meet requirements in between samples.

Figure 8:
Standard errors of the kriging interpolation estimates for (a) mud% and (b) chlorophyll a on the Molenplaat, based on the same data set as Figure 7. The variograms in Fig. 7 were used as the basis for the kriging interpolation

a)



b)



At present we do not have sufficient detailed data sets to investigate these relations for all areas in the Westerschelde and for all environmental variables. We have not attempted in this project to make an extensive analysis. Sufficient data are available at NIOO for two tidal flats: Molenplaat and Plaat van Walsoorden. Furthermore, an upcoming project in the framework of LTV O&M may yield sufficient data on subtidal areas, at least in the eastern part of the Westerschelde (based on multi-beam images).

An interesting option is to apply remote-sensing results, e.g. from satellite radar images or from CASI flights, in order to reduce the uncertainty of interpolation. This would clearly constitute an improvement over current methods based on (sparse) field samples.

3.7. Other promising parameters

Important factors missing in the ecotope system, which are indeed partially correlated to one or several of the descriptors, but which are probably not completely covered are: food availability (e.g. phytoplankton, particulate organic matter) and the slope of the bottom.

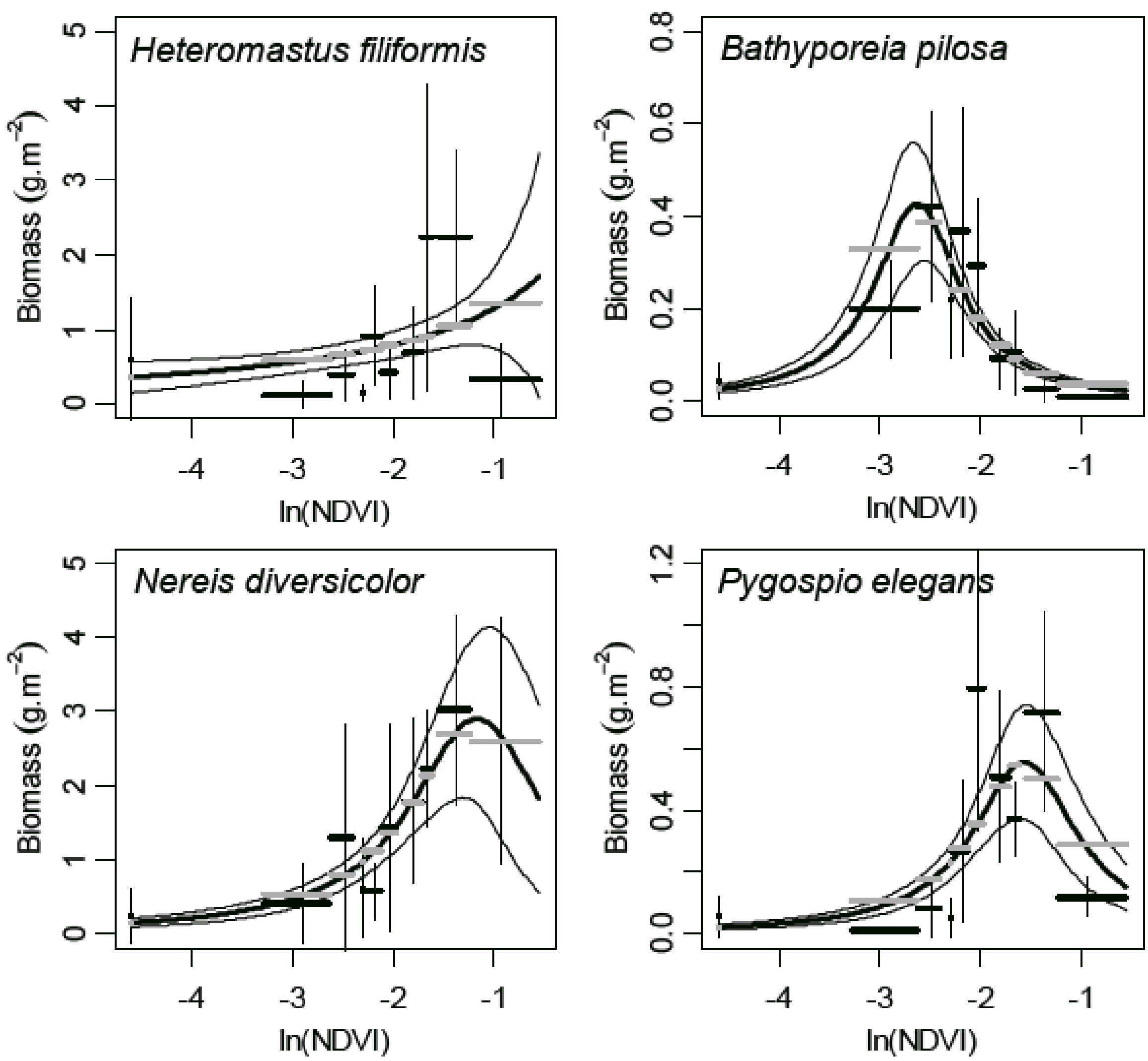
3.7.1. Food availability

An important descriptor for macrobenthic communities might be food availability, as this will highly influence species composition by affecting the number of species and the evenness. The most important food sources for macrobenthos are phytoplankton (for suspension feeders) and microphytobenthos (for deposit feeders). Other macrobenthic species will indirectly be dependent on algal biomass, as they are attracted by the benthic species responding directly (e.g. predators). Herman et al. (2000) showed that microphytobenthos was a very important food source for deposit feeders, however suspension feeders especially fed on pelagic algae and possibly detrital carbon. It might therefore be useful to measure directly food availability in the field. During ebb tide, the microphytobenthos remains at the sediment surface of the intertidal flats where it can be observed. As described in Van der Wal et al. (2007), vegetation, including and on intertidal flats only, microphytobenthos, contains a specific reflectance – absorbance spectrum as it contains chlorophyll-a. Combining reflectance measurements calibrated with concentration measurements of chlorophyll-a in sediment samples makes it possible to estimate chlorophyll-a concentrations for large areas. Chlorophyll-a of the top 1 cm of sediment, can be measured using reverse-phase High Performance Liquid Chromatography (HPLC) methods, measuring the peak at 664 nm. Reflectance in the field can be measured using a hyper-spectral field radiometer (e.g. TRIOS Ramses) for sample locations, or a CASI sensor from a plane for large areas. Chlorophyll-a measurements are based on the absorbance of energy in the red part, and reflection of energy in the near-infrared part. A vegetation index (NDVI) which has a higher value when biomass, coverage or health of the vegetation (microphytobenthos) increases is calculated by $(R_{866nm} – R_{683nm}) / (R_{866nm} + R_{683nm})$, at which; R_{683nm} is the average of the wavelengths 678 nm to 689 nm, which equals CASI band 11; and R_{866nm} is the average of the wavelengths 857 nm to 874 nm, which equals CASI band 17. Van der Wal et al. (2007) recognizes 3 microphytobenthos classes, determined as ‘very high’ (NDVI > 30), ‘high’ (25 < NDVI < 30) and ‘moderate’ (20 < NDVI < 25) concentrations. Four other classes can also be determined, which are ‘moist areas’, ‘salt marshes’, ‘fine sand’ and ‘sand’ surfaces, indicating that the data are also of use for determination of the transition between intertidal and supratidal areas.

On the Plaat van Walsoorden, Van der Wal et al. (2007) found a significant correlation between NDVI and the biomass of dominant macrobenthic species. For *Heteromastus filiformis*, a deep-feeding species, the relation was not significant. For the other species, however, it explained a significant part of the variation in biomass. Figure 9 shows the response curves of the species with respect to NDVI.

For groups of samples representing 1/10th of the total sample, it compares average biomass (dark horizontal lines +/- standard error) with predictions (light grey horizontal line).

Figure 9:
Response curves of biomass of dominant species on remote-sensing derived NDVI on the Plaat van Walsoorden. See text for details



3.7.2. Slope of the bottom

Related to the ecotope system for coastal waters, there has been some discussion about a differentiation according to the slope of the sea bottom. It is suggested to distinguish two classes in the subtidal zone; those with slopes larger than 1:100, and slopes less steep than those (Dankers et al., 2001). Kornman et al. (2001) indicates that for instance for the settlement of Bivalves, the slope of the bottom is important. For bivalves the slope should be 1:(several hundreds to thousands). The slope will probably not only influence settlement, but also food availability and the dynamics and the sediment. In the intertidal zone, the slope can influence predation risk, as several predators forage especially in a small layer of water, which will only be present for a short time at steep slopes. There is not much known about the effect of slopes on communities. Although slopes are relatively easy to measure, as they result from the bathymetric measurement, more specific research to differences in communities related to the steepness of slopes should be executed.

4. Conclusions & recommendations

Based on an examination of the essential accuracy needed to faithfully predict occurrences of macrobenthic species, we can formulate the following conclusions:

1. For most environmental variables (e.g. depth, exposure time, sediment composition, salinity) the analytical precision and accuracy of determinations at *single sample points* is sufficient for the delineation of meaningful ecotopes. This does, however, not seem to be the case for ‘hydrodynamics’ and in particular for current velocity as calculated by models. Models seem to be inaccurate in shallow parts, but there is such a scarcity of measurement data that even that cannot be proven with certainty.
2. The environmental variables depth, exposure time and salinity are measured and/or modeled with sufficient spatial coverage as to produce *reliable maps* directly convertible in ecotope maps. This is not the case for sediment composition, where interpolation leads to rapid increase of standard errors away from data points, to a level that is insufficient for ecological purposes. In view also of the scarcity of sediment composition data, we propose that the use of remote-sensing methods can improve the maps of sediment composition. This topic would require more research as it has, until now, been restricted to a few tidal flats only in the Westerschelde.
3. For the quantification of the variable ‘hydrodynamics’, current practice either relies on the use of (uncertain) model output, or on the use of proxies (e.g. visual detection of sand mega-ripples). Neither of these methods has been properly validated against reliable measurements of current velocity, sediment stability or other variables. We propose that the collection of current velocity data and a better calibration of hydrodynamic models is the best way to solve this problem. Validating the proxies seems more difficult, but the proxies could be useful for spatially differentiating bottom roughness parameters in the models.
4. Our analysis shows that the sensitivity of macrobenthic communities to changes in environmental variables can differ dramatically over the range of values found for the variable. An extreme example is depth. Communities are very sensitive to depth changes in the intertidal, but not at all in the subtidal. We propose that this information on sensitivity should be used when defining and delineating ecotopes. We see no point in distinguishing several subtidal ecotopes according to depth, since depth is not a discriminating variable at all in the subtidal. The information in this report can be used for a revision of the ecotope system and the ecotope boundary definitions.
5. We discuss several potential extensions of the environmental variable system as a basis for ecotope discrimination. In particular, chlorophyll-a (which is observable by remote sensing), slope of the bed (to be derived from bathymetry) and bed form (observable from multi-beam) are candidate variables that are not very difficult to obtain and that can potentially help to discriminate ecotopes. Especially chlorophyll-a has already been demonstrated to correlate well with macrobenthos occurrence, density and biomass.
6. With respect to future further studies, we suggest the following priority list:

- Collection of sufficient data series on current velocity in shallow areas, and re-calibration of models to these data sets.
- Improvement of the mapping of sediment composition, especially in intertidal areas, by combining remote-sensing and field data.
- Reconsideration of the ecotope system based on known species responses. Simplification of the system where possible or needed.
- Improvement of characterization of the environment by inclusion of chlorophyll-a concentration in intertidal areas, to be detected by field sampling and remote-sensing.
- Better use of existing bathymetric data by also including slope as a (derived) variable.

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Annexes

Annex 1

Regression parameters per species and abiotic parameter extracted from Ysebaert & Meire (1999).

	Model salinity			Temporal salinity		
	b0	b1	b2	b0	b1	b2
A. marina	-4.4876	0.2359	-0.00529	-16.0576	1.2208	-0.0254
Bathyporeia spp.	-6.9094	0.6907	-0.0186	-2.6619	0.262	-0.00905
C. capitata	-7.0033	0.4205	-0.00811	-4.5302	0.2017	-0.00334
C. edule	-11.7787	0.845	-0.0169	-6.4215	0.4145	-0.00848
C. arenarium	-6.335	0.32	-0.00654	-3.3022	0.0296	0
C. volutator	1.3865	-0.1766	0	-0.2788	0	-0.00505
E. longa	-7.5659	0.4934	-0.0102	-4.4046	0.2217	-0.00458
H. filiformis	-2.4277	0.2869	-0.00799	-0.8323	0.131	-0.00476
H. ulvae	-7.6278	0.6499	-0.0155	-3.5035	0.2279	-0.00785
M. balthica	-0.4889	0	-0.00064	-0.4171	0	-0.00096
M. arenaria	-2.4598	0	0	-4.6947	0.3424	-0.0107
N. cirrosa	-13.6466	0.8705	-0.0153	-14.9834	1.0643	-0.0204
N. hombergii	-13.8417	0.7561	-0.011	-11.1041	0.65	-0.0106
N. diversicolor	1.1206	-0.1758	0.00299	0.7385	-0.161	0.00291
N. succinea	-7.6936	0.6526	-0.0172	-3.6232	0.2523	-0.00835
Polydora spp.	-3.6535	0.1891	-0.00539	-2.2791	0	0
P. elegans	-4.5216	0.3878	-0.00923	-0.859	0	0
S. plana	-7.5441	0.4355	-0.00865	-3.1038	0	-0.00143
Spio spp.	-8.8219	0.5793	-0.0113	-11.6328	0.8162	-0.0158
T. marioni	-20.4502	1.4531	-0.0273	-5.0642	0.2131	-0.00278

	Depth			Maximum ebb current velocities		
	b0	b1	b2	b0	b1	b2
A. marina	-0.9723	-0.3062	0	-3.3084	11.841	-16.3873
Bathyporeia spp.	-0.5482	-0.1748	0.00139	-0.8305	-0.7429	0
C. capitata	-2.0053	0	0	-1.6555	0	-0.7265
C. edule	-0.8972		-0.0494	-3.7158	17.764	-25.4578
C. arenarium	-1.1454	-0.5638	0	-1.3746	0	-5.1007
C. volutator	0.3489	-0.697	0.0118	0.4807	-4.4698	0
E. longa	-0.6993	-0.3582	0	-4.4046	0.2217	-0.00458
H. filiformis	1.1226	-0.3303	0.00811	0.9316	0	-2.7518
H. ulvae	0.4673	-0.5655	0.00868	-1.4178	6.6523	-10.9256
M. balthica	1.6521	-0.6899	0.0115	-0.1159	5.7402	-11.3351
M. arenaria	-1.0204	-0.3831	0	-4.2368	14.3481	-18.5355
N. cirrosa	-3.6365	0.3439	-0.0112	-4.912	6.7694	-3.3002
N. hombergii	-2.184	-0.0217	0	-2.5242	3.5969	-4.5532
N. diversicolor	1.704	-0.9131	0.0115	2.1693	-6.2043	0
N. succinea	-1.2077	-0.2525	0.00629	-2.0212	2.5264	-4.0152
Polydora spp.	-1.3967	-0.2641	0.00753	-0.8481	-2.5872	0
P. elegans	1.0818	-0.5437	0.0115	0.0158	3.8796	-8.7569
S. plana	-3.3876	2.3558	-0.6535	-2.4124	10.3366	-20.5947

Spio spp.	-2.8752	0.2436	-0.00872	-3.524	5.0769	-3.2829
T. marioni	-1.1171	-0.1985	0	-2.7354	9.7184	-13.942

	Maximum flood current velocities			Median grainsize		
	b0	b1	b2	b0	b1	b2
A. marina	-1.9378	5.3515	-9.5841	-3.5036	0.025	-0.00008
Bathyporeia spp.	-0.7845	-0.8207	0	-5.8042	0.0484	-0.00011
C. capitata	-2.2591	1.9609	-1.8766	-1.8791	0	0
C. edule	-2.0575	9.7087	-16.7715	-2.3111	0.0246	-0.00011
C. arenarium	-2.232	3.4138	-7.623	-8.9815	0.0892	-0.00029
C. volutator	0.5384	-6.0559	1.8435	0.5151	0	-0.00008
E. longa	-2.1425	6.7511	-10.6249	-3.0924	0.0319	-0.00013
H. filiformis	0.8926	0	-2.4952	1.6892	-0.00984	0
H. ulvae	-0.4466	3.0057	-8.4305	-1.1361	0.00952	-0.00005
M. balthica	0.4058	2.7547	-7.7741	0.2037	0.0151	-0.0001
M. arenaria	-1.688	3.7942	-9.9149	-2.7921	0.0278	-0.00013
N. cirrosa	-4.8131	6.6613	-3.2807	-7.1578	0.0383	-0.00006
N. hombergii	-3.1516	5.634	-5.5198	-3.4341	0.0164	-0.00006
N. diversicolor	1.5673	-5.2126	0	0.8026	0	-0.00005
N. succinea	-0.7451	-2.705	0	-1.676	0.00926	-0.00006
Polydora spp.	-0.9719	-2.4457	0	-0.9367	0	-0.00005
P. elegans	0.3586	1.6555	-5.9637	-1.4216	0.0267	-0.00011
S. plana	-2.7641	11.3836	-21.0065	-2.3574	0.0235	-0.00015
Spio spp.	-4.2677	7.185	-4.4092	-6.7564	0.0523	-0.00013
T. marioni	-1.841	5.1387	-8.6742	-2.4274	0.0225	-0.00011

	Mud content		
	b0	b1	b2
A. marina	-2.0977	0.06	-0.00117
Bathyporeia spp.	-0.2651	-0.0716	0
C. capitata	-1.3901	-0.0455	0.000398
C. edule	-2.0283	0.0744	-0.00106
C. arenarium	-2.5604	-0.021	0
C. volutator	-2.6872	0.1171	-0.00109
E. longa	-1.706	0	0
H. filiformis	-0.6586	0.0919	-0.00104
H. ulvae	-0.9163	0	0
M. balthica	-1.207	0.104	-0.00117
M. arenaria	-1.8458	0	0
N. cirrosa	-1.34	-0.1701	0
N. hombergii	-2.5649	0	0
N. diversicolor	-1.662	0.1023	-0.00103
N. succinea	-2.5173	0.1076	-0.00159
Polydora spp.	-3.1689	0.1036	-0.00114
P. elegans	-0.4733	0	0
S. plana	-3.5404	0.0967	-0.00102
Spio spp.	-1.1985	-0.0925	0.00074
T. marioni	-2.5818	0.0961	-0.00133

Annex 2

Comparison of depth/elevation assessment techniques potentially useful for the Westerschelde, extracted from Wiegmann et al. (2005).

Methodology	Depth range	Accuracy after assessing point precision (2 σ)	Accuracy after assessing per grid cell (2 σ)	Probability of supply	Uniformity	Continuity	Costs[1]	Regional usefulness
- Single-beam + RTK-GPS interpolation of lines on 200 m distance in DIGIPOL	Whole range	15 cm	50 cm	Very high	Very high	Very high	k€ 57.8	Not useful
- Single-beam	subtidal	15 cm	50 cm	Very high	Very high	High	k€ 39.2	Very limited
- RTK-GPS	intertidal	6 cm	50 cm	Very high	Very high	Very high	k€ 18.6	Useful
- BAS standard	Above -2.5 m NAP		60 cm	Under investigation	Very high	Under investigation	k€ 32.0	Limited
- BAS contour & crossing	Above -2.5 m NAP		86 cm	Under investigation	Very high	Under investigation	k€ 27.0	Limited
- Multi-beam	< -5 m NAP	4 cm	50 cm	Very high	Very high	High	k€ 82.8	Very useful
- Laser-altimetry	intertidal	20 cm	30 cm	Intermediate	Very high	Intermediate	k€ 24.0	Very useful

[1]Costs calculated for 'vak 3'; a representative part of the Westerschelde; details in Wiegmann et al. (2005).

Annex 3a

Significant correlations between environmental parameters as observed in the Schelde estuary (after Ysebaert et al., 2003)

	Model salinity	Temporal salinity	Depth (m)	Median grain size (µm)	Mud content (%)	Max ebb current velocity (m/s)	Flood current velocity (m/s)
Model salinity							
Temporal salinity	r=0.86, p<0.01 (n=3112)						
Depth (m)							
Median grain size (µm)			r=0.46, p<0.01 (n=1436)				
Mud content (%)			*r=-0.39, p<0.01 (n=1326)*	*r=-0.84, p<0.01 (n=1386)*			
Max ebb current velocity (m/s)			r=0.76, p<0.01 (n=2827)	r=0.45, p<0.01 (n=1455)	*r=-0.37, p<0.01 (n=1340)*		
Flood current velocity (m/s)			r=0.75, p<0.01 (n=2827)	r=0.45, p<0.01 (n=1455)	*r=-0.37, p<0.01 (n=1340)*	r=0.83, p<0.01 (n=3037)	

Negative correlations indicated in italics

Annex 3b

Significant correlations between environmental parameters as observed specifically on intertidal flats of the Schelde estuary between 1994 and 2000 (after Ysebaert & Herman, 2002)

	Median grain size (0-2 cm) (µm)	Median grain size (0-10 cm) (µm)	Mud content (0-2 cm) (%)	Mud content (0-10 cm) (%)	Bed level height (m)	Chlorophyll -a (0-2 cm) (µg/g)	Max ebb current velocity (m/s)	Flood current velocity (m/s)
Median grain size (0-2 cm) (µm)								
Median grain size (0-10 cm) (µm)	r=0.80, p<0.01 (n=209)							
Mud content (0-2 cm) (%)	*r=-0.86, p<0.01 (n=209)*							
Mud content (0-10 cm) (%)		*r=-0.78, p<0.01 (n=209)*	r=0.64, p<0.01 (n=209)					
Bed level height (m)								
Chlorophyll-a (0-2 cm) (µg/g)	*r=-0.4, p<0.01 (n=209)*		r=0.52, p<0.01 (n=209)					
Max ebb current velocity (m/s)					*r=-0.45, p<0.05 (n=30)*	*r=-0.76, p<0.01 (n=30)*		
Flood current velocity (m/s)					*r=-0.45, p<0.05 (n=30)*	*r=-0.76, p<0.01 (n=30)*	r=0.88, p<0.01 (n=30)	

Negative correlations indicated in italics

Annex 3c

Significant correlations between environmental parameters as observed for the mesohaline part of the Schelde estuary (after Ysebaert et al., 2005)

	Organic content (%)	Mud content (%)	Bulk density sediment (kg/m^3)	Elevation (m)
Organic content (%)				
Mud content (%)	r=0.68, p<0.001 (n=20)			
Bulk density sediment (kg/m^3)	*r=-0.71, p<0.0004 (n=20)*	*r=-0.64, p<0.002 (n=20)*		
Elevation (m)				

Negative correlations indicated in italics