Chapter 7

Using carrying capacity compensation scenarios to quantify ecological conservation objectives for mudflats: a general approach applied on the Zeeschelde (Scheldt estuary, Belgium)

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Abstract

Designating restoration goals for dynamic systems like estuaries requires intrinsic flexibility in the restoration concept as well as an expression of the restoration goals in manageable units. An approach is presented to obtain quantified estuarine conservation objectives, using carrying capacity as a central concept. Different scenarios were constructed based on trophic relations, area availability and waste loads. The distinction between 'good' reference and 'bad' compensative scenarios was determined through criteria concerning species diversity. The approach was applied on the Belgian part of the Scheldt estuary, the Zeeschelde. In the Zeeschelde, dissolved oxygen was the first limiting factor on diversity. Using a catchment model in combination with the diversity restrictions, reference scenarios revealed themselves as a pristine scenario and a scenario representing the waste load situation of the year 1950. It was calculated that the Zeeschelde needs about 500 ha extra mudflat area to compensate for lost macrobenthic production. Waste load reductions were also proposed, taking into account catchment derived nutrient ratios.

7.1 Introduction

In and around the estuaries of the developed world and especially of NW-Europe, space is highly demanded for various societal needs. Most estuaries harbour harbours as gates of commerce and trade that feed industry and densily populated areas around them. Agriculture is intense and land prices are in general relatively high. How much area of a habitat is needed? This question is often uttered by policy makers and ecosystem managers who need to budget spatial resources.

Amidst a whole range of such society relevant functions, estuaries support many functions that are more closely related to the system itself: biogeochemical cycling and movement of nutrients, purification of water, mitigation of floods, maintenance of biodiversity, biological production, etc. (Meire *et al.*, 2005). Many functional needs can be translated into physical entities. Water storage capacity volumes can be calculated as a function of flood risk, harbour space as a function of traffic needs, etc. It is of crucial importance that ecological needs can likewise be translated to manageable units such as space.

In Europe the recognised ecological value of estuaries is crystallised in protective legislation. The European Bird Directive (79/409/EEG) and Habitat Directive (92/43/EEG) are important juridical imperatives providing protected areas in estuaries. For areas under the Habitat Directive a good state of conservation is required. Therefore every member of the European community is bound to construct Conservation Objectives that guarantee the presence of the protected habitats and viable populations on the long term. The Water Framework Directive (2000/60/EG) requires that a good ecological status for transitional and coastal waters must be reached in 2015. The ecological status must be formulated based on phytoplankton, macro-algae, angiosperms, benthic invertebrates and fish. This status must be evaluated against a (theoretical) undisturbed reference condition (*e.g.* Borja *et al.*, 2000).

Conservation objectives can be very strong instruments, linking the present and potential ecological health with clear management objectives, provided that they are well constructed. However, construction of conservation objectives or reference conditions for estuarine habitats faces complications, as estuarine habitats are far from static. These transitional water systems are geomorphologically very dynamic and ephemeral, influenced both by sea and land changes, forming a complex and ever evolving mixture of many different habitat types, exposed to human induced changes in water quality and various other kinds of disturbances (Meire *et al.*, 2005) According to the dominating flow pattern, mudflats can either erode to subtidal areas or change into pioneering marshes, young marshes can grow old and old marshes can drown by erosion (Van de Koppel *et al.*, 2005). As the restless nature of estuaries also persists in the long run, both natural evolution and human impacts are intertwined as causes of morphological transition. As such it becomes impossible to refer to a temporal reference state '*sensu stricto*' in order to assess the ecological condition of an estuary. Within this fluid framework, the quantification of habitat needs requires an approach transcending ambiguities resulting from static '*hic et nunc*' protective recommendations.

Up till now, conservation objectives for estuarine systems have only been expressed in general terms, *e.g.* stating that parameters should not deviate significantly from an established base line, subject to natural change (Elliott, University of Hull, written communication, 2008). Such an approach is depending on the definition of 'baseline' and the interpretation of 'significantly', complicating in this way the objectivity of the approach. Up till now conservation objectives of estuarine habitats, expressed in manageable units, have never been reported. It is the double challenge of this article to 1) overcome the issue of system dynamism in expressing conservation objectives and 2) to express conservation objectives in quantified terms of space so that they are easily feasible for management. It is the aim of the present paper to present a coherent method or approach to derive such conservation objectives. Although elaborated for the Zeeschelde, the Belgian part of the Scheldt estuary, we believe that the approach is applicable not only on this selected case but on many estuaries.

First, the outline of the conceptual approach is explained. Then the approach is applied on the well documented Zeeschelde.

7.2 Conceptual approach

7.2.1 Conservation objectives, carrying capacity, ecosystem functioning and space

How much area of a habitat is needed? The question contains the presumption that space is the main determinant of a good state of the habitat, *i.e.* its production, quality and the diversity of the life it carries. This is true for certain environments (Paine, 1966), but certainly not all. The definition of conservation objectives requires that the system can sustain itself and that the populations that live in it are viable on the long run. The concept of carrying capacity is closely related to this formulation of objectives. Many definitions of carrying capacity have been elaborated (overviews *e.g.* in del Monte-Luna *et al.*, 2004 and Elliott *et al.*, 2007). Carrying capacity was formerly and more usually used as an ecological concept but it is extendable in terms of both environmental and societal demands *i.e.* what the natural system wants and can accommodate and what are society's aspirations (Cohen, 1997; Elliott & Cutts, 2004; MacLeod & Cooper, 2005; Yozzo *et al.*,, 2000; Van Cleve *et al.*, 2006), or to system function, ecosystem goods and services, as listed by De Groot *et al.* (2002). The choice of definition depends on what we want to consider for restoration and conservation. Keeping in mind that the eventual results must be expressed in function of area, then the classic definitions are suitable like *e.g.* from Baretta-Bekker *et al.* (1998): 'the maximum population size possible in an ecosystem, beyond which the density cannot increase because of environmental resistance'. It is synonymous with the general productivity of an ecosystem. However, when linking the concepts of carrying capacity and conservation objectives, discordance emerges. Conservation objectives require the determination of minimal conditions for a system to be sustainable, while carrying capacity is about determining maximal possible entities that can be sustained. But the fact that carrying capacity is not a constant on the long term (Seidl and Tisdell, 1999) can be used to develop conservation scenarios. It is in our approach assumed that carrying capacity is a constant during a five year period, and that five year period scenarios can be compared as different states of equilibrium. The strength of linking the carrying capacity and conservation objective concept is that they both share the same duality: not only the production, population, standing stock, crop or other entities that are scoped, need to be considered, but also the factors that control them and that affect quality. The challenge is to assemble all quality needs in the population or production size calculations, and to quantify these relations from the viewpoint of spatial aspects.

7.2.2 Mudflats and benthos

Mudflats are very illustrative for the dynamic and ephemeral character of the estuarine ecosystem. Their outline is set vaguely by the level of high and low water, which probably contributed to the fact that mudflat area evolution is less documented than that of tidal marshes (Meire *et al.*, 2005). Nevertheless, according to De Groot *et al.* (2002) mudflats have important functions. They reduce dike abrasion by wave action, dissipate tidal energy and are potential hot spots for denitrification (Middelburg *et al.* 1996). They host a major part of the estuarine benthic invertebrates (Ysebaert *et al.*, 2005), supporting numerous overwintering wading-birds and different guilds of adult and juvenile fish. The production of benthos is crucial for these higher trophic levels, and this function is the epicentre of our approach. Carrying capacity of wading birds is not scoped as its determination is more complicated than for benthos. The development of competitive interference between wading birds has indicated that food resource competition alone underestimates the demands for space (Stillman *et al.*, 2005). Furthermore it is assumed that the ecotrophic efficiency, *i.e.* the fraction of the benthos production that is utilized within the system for predation or export (Christensen & Pauli, 1998), is a fix percentage of production for all scenarios.

The carrying capacity (CC) for higher trophic levels (waders, fish) in an estuarine ecosystem, depends on the biomass of benthic invertebrates as the maximal system averaged standing stock, is expressed as:

$$CC = B*A \qquad (1)$$

with A the total system habitat area, and B the system averaged benthic biomass per area unit, resulting from all factors of which it is influenced. We assume a linear relation between carrying capacity and habitat area, restricting our approach to the many cases where mudflats are fringing habitats. Both mudflat area and benthic biomass are not constant in time. Mudflat area is prone to morphologic evolution, land reclamation etc., benthic biomass can alter under different water and sediment quality factors. In this approach we assume that all changes in carrying capacity result from human interference. By putting the natural carrying capacity as a constant 'pristine state', it becomes possible to budget changes. The carrying capacity between two scenarios can be thus be compared as:

$$B_i * A_i = B_j * \left(A_j + A_c\right) \qquad (2)$$

with i a reference scenario and j a scenario to be assessed (Fig. 7.1). One scenario covers a five year period in which a carrying capacity equilibrium is assumed. A scenario can be situated in the present, the past or the future. The equation is matched by the area compensation term A_c. A_c represents the area that is needed to compensate scenario j for scenario i. The following scenarios were selected:

- The 'pristine' scenario represents a hypothetical state of the Scheldt basin before any significant human disturbance. It corresponds to a watershed entirely covered by forest.

Low soil leaching and erosion as well as direct litter fall in the tributaries are the only external inputs of nutrient considered. This scenario stands for the 'very good' state of the estuary. The mudflat area is for this scenario unknown.

- The scenarios '1950' to '2000' consist of a reconstruction of the evolution of agriculture, industrial and urban wastewater management policies over the last 50 years, as explained in detail by Rousseau *et al.* (2005). The time range covers the evolution to the worst water quality ever recorded for the Scheldt and its subsequent recovery (Soetaert *et al.*, 2006).
- The '2015' scenario is a prospective scenario assuming that the requirements of all European directives on wastewater treatment and water management are met everywhere in the basin. In particular, this scenario takes into account a 90% abatement of the organic load of urban wastewater by secondary treatment, and an abatement of 90% of the phosphorus load and 70% of the nitrogen load by tertiary treatment. This scenario represents, admittedly, a quite optimistic view of the future situation of the Scheldt hydrographic district.

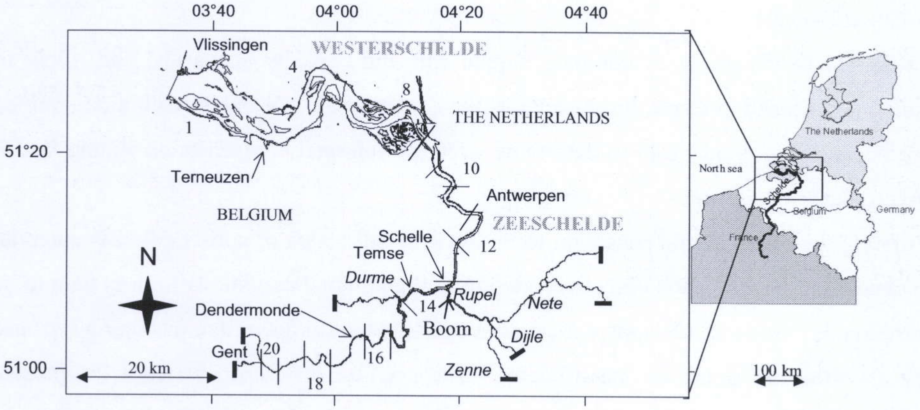

Fig. 7.1: Map of the Zeeschelde; compartments according to Soetaert & Herman (1995)

These scenarios represent average hydrological conditions, characterising a certain 'historical' state of land use and human activity. The light climate in the water column was considered equal for all scenarios, as it could not be reconstructed quantitatively.

7.2.3 Coupling with other factors

Macrofaunal biomass is related with many factors (Ysebaert *et al.*, 2005). On ecosystem scale, however, when plotted for several estuaries, a relation with primary production was found (Herman *et al.*, 1999), namely:

$$B = -1.5 + 0.105 * P \qquad (3)$$

with B in g AFDW m^{-2} and P the system averaged net primary production density (in g C m^{-2} y^{-1}).

Primary production is in turn linked with the nutrient load, as far as no other factors, such as light, are limiting. Primary production has been incorporated in many ecological models (*e.g.* Soetaert *et al.*, 1995; Hofmann *et al.*, 2008). These ecological models have the benefit that they can be used to reconstruct historic primary production scenarios of which no monitoring data are available. Equation (3) thus allows the extrapolation of historic scenario model results for water quality and primary production towards the higher trophic level of benthic macrofaunal biomass.

Equations (1) to (3) allow to compare simple carrying capacity scenarios, and allow to determine compensation terms, but as long as the scenarios are not linked with a determined quality status, there is no mean to determine what the minimal compensation should be for any given scenario to assess.

Determining the minimal compensation to obtain a 'good' status of conservation is essential in the concept of conservation objectives, requiring the minimal conditions for a system to be self sustainable. As such, the approach needs a decision tool to determine whether a scenario can be classified as 'good' or 'insufficient'. The smallest difference between the present situation scenario and any of the 'good' scenarios represents the least compensation need. As the production compensation is covered by carrying capacity equations, the decision tool is based on the qualitative requirements for conservation, *i.e.* habitat quality or species diversity. It is known that eutrophication can cause a collapse of benthic production; in anoxic systems the macrozoobenthic community will be reduced to zero. The reference for a 'good' diversity is assumed to stand also as a reference of production, as the conditions for a 'good' macrozoobenthic production are less well documented for our example, the Zeeschelde.

The present situation can so be evaluated amongst different scenarios, allowing, in case of a bad condition, a quantification of the effort that is necessary for recovery, expressed either in terms of surface, or quality, expressed as a required biomass density increase. It is perfectly possible that an assessment of the present situation scenario could turn out to have more benthic standing stock than *e.g.* the pristine scenario, for instance if the present biomass production would be relatively higher than any corresponding loss of habitat.

The elements required for applying our presented approach are habitat area evolution, benthos biomass production evolution through modelling of primary production, the determination of the limiting factor of habitat quality or species diversity and the effect of this factor on biomass production.

7.3 Application of the conceptual approach

7.3.1 Study site: the Zeeschelde

The watershed of the river Schelde is approximately 21.863 km^2. With about ten million people or 477 inhabitants km^{-2} it is one of the most densely populated watersheds in the world. The Scheldt is a typical rain fed lowland-river, stretching over 355 km from source (St. Quentin in the north of France) to the mouth (Vlissingen). The estuary of the river Scheldt (Fig. 7.1) extends from the mouth in the North Sea at Vlissingen (km 0) till Gent (km 160), where sluices stop the tidal wave in the Upper Scheldt. The tidal wave also enters the major tributaries Rupel and Durme, providing the estuary with approximately 235 kilometres of tidal river. The Zeeschelde, the Belgian part of the Scheldt estuary (105 km long), is characterized by a single ebb/flood channel, bordered by relatively small mudflats and marshes (28% of total surface). The surface of the Zeeschelde amounts to 44 km^2. A freshwater zone (limnetic plus oligohaline) zone stretches from Gent down to about Antwerp (82 km from Gent). Between Antwerp and the Dutch Belgian border the water is mesohaline with considerable salinity changes (Van Damme *et al.*, 2005). A more detailed description of the Scheldt estuary is given in Meire *et al.* (2005). This study is restricted to the Zeeschelde. By cutting away the Dutch part of the estuary, the error of not knowing the scenario values of the sea boundary is minimized.

7.3.2 Mudflat area evolution

Although the overall loss of intertidal habitat of the Scheldt estuary is fairly well known (Meire *et al.*, 2005), no detailed information on the loss of mudflat area was available. This was reconstructed by careful analysis of several maps.

The loss of tidal flats in the Zeeschelde was reconstructed by digitising with ArcGIS 8 (ESRI) old maps and areal photographs. The oldest material is the so called Van der Malen map of 1850. The mudflat area of 1950 was determined in the same way using the map of 'Depot de la guerre' (1950). In 1990 and 2004, orthophoto's were taken from the whole Zeeschelde at low tide in order to construct vegetation maps. A digital elevation model (DEM, a combination of high tide bathymetric sonar and low tide altimetric laser data) that was made in 2001 was used to interpret and refine the results from the orthophoto's. Changes in the intertidal mudflat area through embankment, river straightening, dike construction, industrial infrastructure works, bank fortification, erosion and de-embankment were calculated for 1850, 1950, 1990 and 2004. There was no way to determine the mudflat area before 1850. For lack of better estimates, it was assumed that the mudflat area remained constant from pristine times till 1850. Between 1950 and present, missing values for time intervals were interpolated linearly. It was assumed that for the '2015' scenario, the mudflat area remained the same as the '2000' scenario, so that the compensation area of the future is set relative to the present situation.

The evolution of the total mudflat area of the Zeeschelde, including all tidal parts of the tributaries, is biased by missing information for some compartments, especially in 1990 (Table 7.1). In spite of these gaps, the trend is clear: since 1850 more than 900 ha of intertidal mudflats were lost in the Zeeschelde, corresponding with approximately only one third of the habitat available in 1850.

Between 1850 and 1950, the main loss could be attributed to land winning, from 1950 to 1990 infrastructure works and dike construction showed to be the main factors of mudflat area reduction. Since 1990, intertidal mudflats were probably mostly lost by erosion.

Table 7.1: Area evolution (ha) in the Zeeschelde and tidal tributaries; compartments according to Soetaert & Herman (1995)

Section	1850	1950	1990	2004
compartment 9	757	257	241	197
compartment 10	169	169	116	96.3
compartment 11	183	183	126	81.8
compartment 12	103	91.0	57.8	35.7
compartment 13	56.7	51.0	41.7	29.1
compartment 14	83.5	82.2	71.0	40.5
compartment 15	24.1	31.6	21.4	19.2
compartment 16	17.7	17.7		8.23
compartment 17	17.6	17.6		7.49
compartment 18	10.8	10.8		0.63
compartment 19	10,2	10.2		1.15
Dead end Melle-Gentbrugge		25.0		23.9
Durme			34.6	24.7
Rupel	38.7	38.7		26.1
Dijle-Zenne-Nete				0.31
Total	**1472**	**985**	**709**	**592**

7.3.3 Limitation of estuarine diversity

Indications that the oxygen concentration of the Zeeschelde is the prime factor that has affected its species diversity are amply available, *e.g.* the species composition of the benthic macrofauna (Seys *et al.*, 1999), the fish fauna (Maes *et al.* 2004), or the distribution pattern of the copepod *Eurythemora affinis* (Appeltans *et al.*, 2004). However, quantified oxygen demands of species or communities that belong to the Zeeschelde are scarcely documented, and estuarine oxygen standards . Although water quality standards are amply available, no standard method exists to derive oxygen concentration standards with respect to estuarine whole system diversity. Relations between oxygen are restricted to. Therefore the only way to derive such standard is by combining all individual studies that link species sensitivity or community composition with oxygen, including physiological and ecotoxicological single species studies (*e.g.* Ross *et al.*, 2001) and correlations between dissolved oxygen concentration and community composition. Arguments are listed for fish, benthos and zooplankton.

The response of fish species on increasing dissolved oxygen concentrations, expressed as the probability that a fish is caught in a fike over a 24 hour period, has been modelled by Maes *et al.* (2004). The response results can be divided into two groups: migrating species (execpt

Anguilla anguilla and *Gasturosteus aculeatus*) showed a significant response while most freshwater species or estuarine resident species showed no response (Maes *et al.*, op.cit.). Maes *et al.* (2008) modelled that a minimum oxygen concentration of 5 mg.L^{-1} can restore the migration route of *Alosa fallax*. Initially an oxygen concentration, corresponding with 50% probability to catch the fish of a certain species, is proposed as criterion for a good oxygen condition for the species. But the fike catch results were obtained independently of the seasonal migration pattern. This criterium corresponds with the estuarine criteria propsed by the US Environmental Protection Agency. A correction for temperature and seasonal occurrence was therefore presented, showing a considerable differentiation for *e.g. Allosa fallax* and *Liza ramada* (Maes *et al.* in prep.). The resulting criteria for migrating species range between 1.5 mg L^{-1} for *Anguilla anguilla* to 7.5 mg L^{-1} for *Liza ramada*. As *L. ramada* is considered to be typical for coastal zones and estuaries, migration of this species to the fresh water zone would be rather elective, so that the high criterion for this species can be questioned. The other criteria correspond well with criteria for comparable American species, *e.g.* 5 mg L^{-1} for Alosa sapidissima (Stier & Crance 1985) and 6 mg L^{-1} for several Osmeridae (Dean & Richardson, 1999). Although juvenile fish is generally more sensitive to oxygen than adults, they are capable of avoiding oxygen poor conditions (Wannamaker & Rice, 2000; Richardson *et al.*, 2001), or show physiologic adaptations to withstand hypoxia during short periods (Ross *et al.*, 2001).

In 1964, before a long period of severe deterioration of the water quality, a very diverse macrozoobenthos community was found in the freshwater tidal area of the Biesbosch (the Netherlands), with Shannon Wiener indices between 1 and 2, at oxygen summer concentrations between 50 and 70 % saturation (Wolff, 1973). In the impacted fresh water zone of the Scheldt estuary such diversity was never recorded while recorded oxygen summer conditions only very recently amounted up to such levels of saturation.

In the oligohaline intertidal zone of the Elbe estuary, the presence of 68 macrozoobenthic species could be linked with dissolved oxygen concentrations between 5 and 6 mg.L^{-1} (Krieg, 2005).

The zooplankton species *Eurythemora affinis* shifted from the brackish to the freshwater zone of the Scheldt when dissolved oxygen concentrations increased from around 1 to around 3 mg.L^{-1} (Appeltans *et al.*, 2003.

Based on all previous arguments, taking into account species specific tolerance levels, migration periods and swimming capacities, and community diversity related to oxygen concentrations, it is proposed to put forward an oxygen concentration of 6 mg L^{-1} (= 187 μmol L^{-1}) between November 1st and April 30th and a weekly average of 5 mg L^{-1} (= 156 μmol L^{-1}) for the corresponding summer half year.

In case oxygen concentrations would increase in the Scheldt, other diversity limiting factors should be assessed. For instance, physical disturbance has explained diversity reduction of benthos in soft bottom sediments of the brackish part of the estuary (Ysebaert *et al.* 2000).

7.3.4 Reconstruction of historic estuarine primary production and benthic macrofauna biomass

The **RIVERSTRAHLER model** has been used to model immissions from the watershed into the Zeeschelde. It is a simplified model of the biogeochemical functioning of river systems at the basin scale allowing to relate water quality and nutrient fluxes to anthropogenic activity in the watershed (*e.g.* Billen *et al.*, 1994). The model has recently been applied to the Scheldt river system (Billen *et al.*, 2005), thus reconstructing the respective role of hydrology and human activity in the watershed during the last 50 years.

The RIVERSTRAHLER model has been applied to the upper Scheldt watershed (including the Dender river) on the one hand, to the Rupel watershed on the other hand (Fig. 7.2). The flux results represent integrated values of the fluxes discharged at Temse and Boom respectively. Because hydraulic regulation of the Leie river in the region of Ghent entirely discarded its flow from the lower Scheldt course over different canals, the Leie basin is not included in the analysis. Based on the analysis of the long term rainfall data for the Scheldt watershed over the last 50 years, the following conditions were chosen as representative of 3 classes of hydraulicity:

1995 (804 mm y^{-1}) for the 'mean' conditions, *i.e.* a mean discharge of 185 $m^3\ s^{-1}$ at Schelle

1984 (1275 mm y^{-1}) for the 'wet' conditions, *i.e.* a mean discharge of 250 $m^3\ s^{-1}$ at Schelle

1976 (541 mm y^{-1}) for the 'dry' conditions, *i.e.* a mean discharge of 65 $m^3\ s^{-1}$ at Schelle

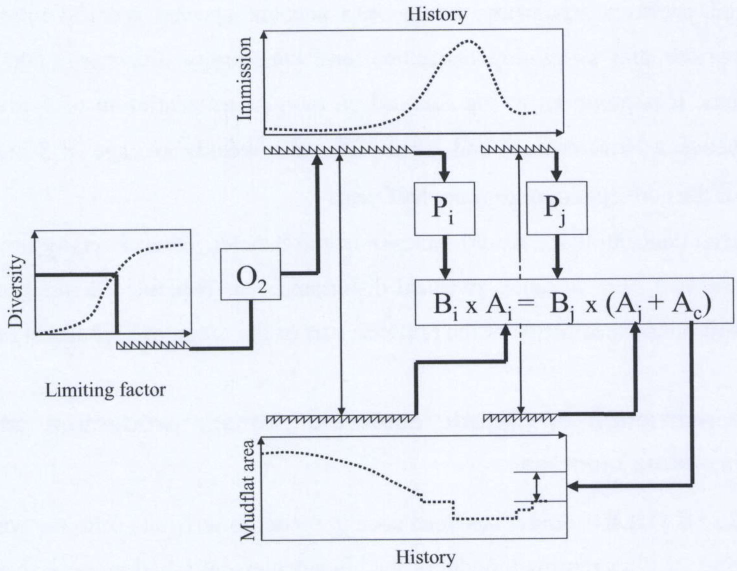

Fig. 7.2: Scheme of the scenario comparison approach. B = Macrofaunal benthic biomass production, A = area, P = primary production, i = reference scenario, j = scenario to be assessed, c = compensation

The estuarine **MOSES model** is a simplified simulation box compartment model using fixed dispersion coefficients, allowing to predict chemical and biological alterations that can take place in dissolved substances that reside in the estuary. The model is described in Soeatert & Herman (1995 a, 1995 b & 1995 c), and has since then been improved by recalibrating on data of 1980-2002, implementing the lateral input of tributaries in a better way, and reformulating the transport in the upper compartments (Cox *et al.*, in prep.).

The Riverstrahler results for the different scenarios have been used as input for the improved MOSES model. In that way the effect of specific estuarine processes could be reconstructed for the present and historical immission scenarios. The MOSES model was run on scenarios with average hydraulicity. The model results of the present '2000' scenario were, according to Hofmann *et al.* (2008), calibrated on data of 2001, a year with a mean discharge of 191 $m^3 s^{-1}$ at Schelle.

The RIVERSTRAHLER and improved MOSES models have been coupled for 4 scenarios: 'pristine', '1950', '2000' and '2015'. System averaged scenario values were averaged over the model compartments (Soetaert *et al.*, 1995), weighed for compartment volume or surface as required per parameter.

The results from the RIVERSTRAHLER model showed that the immission from the watershed has known an increase of human impact from pristine times up till the eighties

(Fig. 7.3). Detritic carbon and phosphorous show the same pattern, reflecting maximal immissions in the period 1970-1980, when the estuarine water quality was indeed bad (Van Damme *et al.*, 2005). In the nineties a period of recovery started. The limits of the progress, as set by policy makers, are marked by the future '2015' scenario. This scenario shows drastic improvement: both carbon and phosphorous immissions become smaller than the '1950' scenario and are nearing the pristine scenario. For the immission of total nitrogen, dominated by nitrate, this is, however, not the case; its immission in the estuary steadily increases, even when carbon and phosphorous loads are decreasing. The future '2015' scenario showed maximal nitrate immissions of history. A more detailed presentation and analysis of the results is given in Billen *et al.* (2005).

Using the RIVERSTRAHLER results as input in the MOSES model gave satisfactory results: The '2000' scenario of the MOSES model results show that the modelled oxygen concentrations fitted well the observed data (Fig. 7.4). The reconstructed oxygen concentrations reached our diversity standard of 156 μmol L^{-1} in the 'pristine' and the '1950' scenario (Fig. 7.4). In the '1950' scenario the summer standard was not met only in July in the most upstream compartment of the Zeeschelde. In the '2000' and '2015' scenarios, summer concentrations dropped below 100 μmol L^{-1}, and winter concentrations in these scenarios dropped below 150 μmol L^{-1} during five consecutive months. According to the European Water Framework Directive (2000/60/EG), the 'pristine' scenario corresponds to the 'very good' status for water quality. The '1950' scenario meets the requirements for good diversity, and can thus be put forward as a 'good' status scenario. Both scenarios can thus be used as a reference to assess the present situation. The '2000' and '2015' scenarios are scenarios to be assessed relative to the reference scenarios.

The primary production results showed another ranking of scenarios than the oxygen results. The production in the '2015' scenario dropped below the production in the '1950' scenario, indicating that implementation of the European Directives will resort a strong effect (Fig. 7.5). In the upstream part of the Zeeschelde , the 'pristine' and '2015' scenarios have production rates about half as big as in the '1950' and '2000' scenarios. In all scenarios primary production dropped to almost zero in the most downstream brackish compartments, which are characterised by strongly varying salinity values (Van Damme *et al.*, 2005).

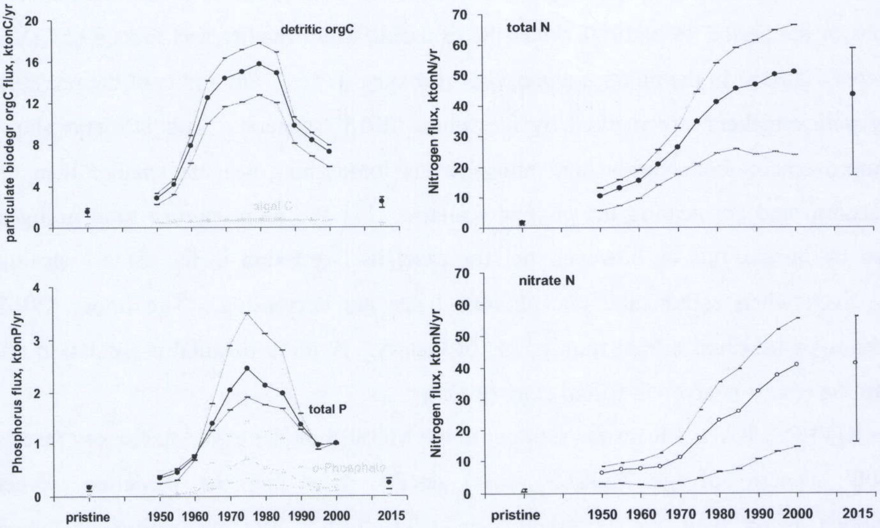

Fig. 7.3: Immissions in the Zeeschelde, as calculated with the Riverstrahler model for different scenarios

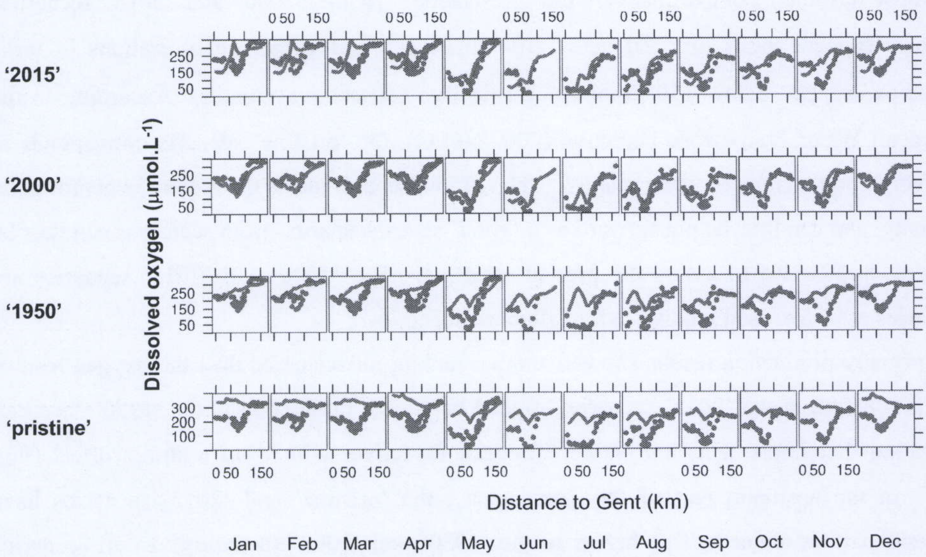

Fig. 7.4: Dissolved oxygen concentrations in the Zeeschelde, as calculated with the MOSES model for every month of 4 scenarios. Solid lines are modeled results, dots are measured data of 2001.

System averaged scenario values were highest for the '2000' scenario, 55.4 g C m^{-2} y^{-1} and lowest for the 'pristine' scenario, 19.1 g C m^{-2} y^{-1} (Table 7.2). The corresponding chlorophyll a concentrations showed a similar pattern as primary production (Fig. 7.5). Concurrent with the immission values of total nitrogen and nitrate (Fig. 7.3) the concentrations of total dissolved inorganic nitrogen (TDIN) and nitrate were almost equal for the '2000' and '2015' scenarios (Fig. 7.5). High rates of nitrification in the Zeeschelde (Vanderborght *et al.*, 2002 ; Hofmann *et al.*, 2008) are explaining the low oxygen concentrations for the '2015' scenario. The system averaged macrozoobenthic production, as calculated from the system averaged primary production (Table 7.2) with equation (3) amounts from 2.6 % ('pristine' scenario) to 7.7 % ('2000') of the system averaged primary production.

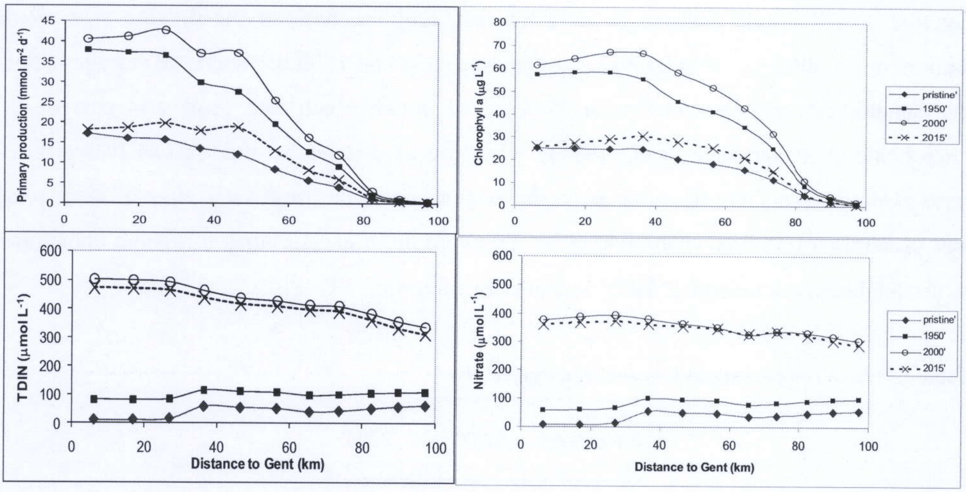

Fig. 7.5: Primary production, chlorophyll a, total dissolved inorganic nitrogen (TDIN) and nitrate in the Zeeschelde, as calculated with the MOSES model for 4 scenarios

Table 7.2: Values used for scenario comparisons

Parameter		'pristine'	'1950'	'2000'	'2015'
System averaged primary production	($g\ C\ m^{-2}\ y^{-1}$)	19,1	43,6	55,4	27,3
System averaged benthos production	($g\ AFDW\ m^{-2}\ y^{-1}$)	0,5	3,1	4,3	1,4
Total system mudflat area	(ha)	1472	985	592	?

7.3.5 Mudflat area assessment

Using the reconstructed system averaged macrozoobenthic production and area evolution (Table 7.2), area needs were calculated with equation (2) for the '2000' and '2015' scenarios, with the 'pristine' and the '1950' scenario as reference (Table 7.3). If the '1950' scenario is used as reference, then the compensation demands amount to 111 hectares of mudflats for the present situation, or to 1635 ha when the improved water quality allows a primary production of only half that of the '2000' scenario (Table 7.3). When the 'pristine' scenario is used as reference, the '2000' and '2015' scenarios offer enough carrying capacity for benthos, as the area needs are negative. The habitat area of the pristine scenario was set equal to the oldest documented habitat area, which is of 1850. If the pristine habitat area would turn out to be at least 125 ha larger than in 1850, then the area demand for the '2015' scenario would become positive, and it would become positive for the '2000' scenario if the pristine area would amount up to 5000 ha. It is known that between 1650 and 1800 the intertidal storage area of the Scheldt estuary decreased with 99 km^2, of which about one third was part of the Zeeschelde (Van der Spek *et al.*, 1997). The ratio of marshes vs mudflats of these former areas is not known. On the other hand, in pristine times, the freshwater zone of the estuary was probably a non tidal riverine stretch. To avoid these uncertainties it is more convenient to use the better documented '1950' scenario as reference.

Table 7.3: Area compensations between scenarios (in ha)

Reference	'2000'	'2015'
'1950'	111	1635
'pristine'	-420	-47

7.3.6 Validation

This paper presents a method to quantify the estuarine mudflat area demand corresponding with a good state of conservation combined with conditions for a good ecological status. The method relies on a comparison of carrying capacity scenarios, and on a trophic relation between primary production and macrofaunal biomass production. Such a direct relationship between primary production and carrying capacity for higher trophic levels has been identified for several ecosytems. In pelagic systems such relations even matched on species level: herring populations were related with primary production although they feed on zooplankton as intermediate trophic level (Perry & Schweigert, 2008). For benthic communities the relation might be more biased. A validation of the results is therefore essential.

The presented approach to determine mudflat area budgets relies mainly on modeled results, on the relation between primary production and macrozoobenthos and on the soundness of the conceptual approach in general.

Model validation of both Riverstrahler (Billen *et al.*, 2005) and MOSES models (Soetaert *et al.*, 1996) has proven satisfactory, but this validation refers to the '2000' scenario only, with data of 2001 (*e.g.* Fig. 7.4). As illustrated before (Fig. 7.5) the water quality of the Scheldt estuary is actually changing. A regime shift has taken place in 2004, leading to higher oxygen concentrations, even supersaturation levels in the freshwater zone (Cox *et al.*, 2009). At decreasing nutrient concentrations an increase of chlorophyll a concentrations was noted. A possible factor could be the end of ammonia intoxication with decreasing ammonium levels. The regime shift has two implications. First, it is expected that oxygen will cease to be the limiting factor for diversity. Van Eck *et al.* (1993) predicted that a recovered oxygen status might trigger a massive dissolution of heavy metals. Preliminary calculations indicate that it is not certain if heavy metals will take over the role as diversity limiting element (Teuchies *et al.*, in prep). It is not even known if the next limitation on diversity is chemical or physical or anything else. Our approach remains applicable at changing diversity limitations as far as they are quantified. Second, regime shift behaviour of primary production is, admitted, not incorporated in the used models. Many factors were considered constant between scenarios, such as estuarine and riverine morphology and light climate. In the Scheldt estuary light is more limiting for primary production than nutrients (Soetaert *et al.*, 1995b), but reconstructing changes among scenarios lead no further than a semi-quantitative estimate that suspended matter concentrations were about a quarter to one third less before the embankments in the Scheldt estuary between 1650 and present (Van Damme *et al.*, 2009).

The interestuarine trophic relation between primary production and benthic macrofaunal production that was used in this study (equation 3), was based on data of the Scheldt estuary, but only of the polyhaline and mesohaline zone (Herman *et al.*, 1999). The Zeeschelde was considered as being one ecosystem, despite the different characteristics of both primary production and benthic macrofauna along the salinity gradient. In order to assess the validity of the trophic relation and to validate the modelled results, data of primary production and macrobenthic fauna need to be compared taking into account the salinity gradient. In the mesohaline part of the Zeeschelde primary production decreases with increasing salinity from the values in the freshwater zone to zero or negative values in the mesohaline zone (Kromkamp & Peene, 1995; Kromkamp *et al.*, 2005), as the phytoplankton communities showed strongest changes in species composition and decrease in biomass of both marine and freshwater populations between 0.5 and 5 psu salinity (Muylaert *et al.*, 2005). An overview of primary production data in the freshwater part of the Zeeschelde and other estuaries is given by Van Damme *et al.* (2009). The primary production in the freshwater zone of the '2000' scenario ranged between 162 to 193 g C m^{-2} y^{-1}. This is in the range of primary production rates of 108 (Muylaert *et al.*, 2005), over 388 (Kromkamp & Peene, 1995) to 500 g C m^{-2} y^{-1} (Kromkamp *et al.*, 2005). The modeled primary production values of the '2000' scenario were thus in agreement with other data.

The changes of diversity and species composition of the soft-sediment benthic macrofauna along the salinity gradient of the Scheldt estuary have been described by Ysebaert *et al.* (1993; 2003). The average biomass in the mesohaline part of the estuary was in the subtidal zone 0.94 g AFDW m^{-2} (Ysebaert *et al.*, 2000) and in the intertidal zone 6.7 g AFDW m^{-2} (Ysebaert *et al.*, 2005). In the oligohaline part of the estuary a sharp decrease in species richness and biomass was observed. In the freshwater tidal part of the Zeeschelde the benthic macrofauna of soft sediments is mainly constituted of Oligochaeta (Ysebaert *et al.*, 1993; Seys *et al.*, 1997). The presence of this impoverished benthic fauna was mainly caused by the bad water and sediment quality of this part of the estuary. The average biomass in the intertidal zone varied between 3.8 g AFDW m^{-2} in 1996 and 1.7 g AFDW m^{-2} in 2002. For the subtidal zone only sufficient data are available for 1996, showing an average biomass of 1.6 g AFDW m^{-2}. For the whole freshwater tidal zone of the Zeeschelde we estimated the average biomass at 2.5 g AFDW m^{-2}. For the Westerschelde, the system averaged biomass was estimated at 10 g AFDW.m^{-2} (Herman *et al.*, 1999).

Summarizing for the whole Zeeschelde we estimated the average yearly total benthic biomass at 3.1 g AFDW m^{-2}. Based on a weighted average, considering the relative surface of the intertidal and subtidal zone, a rough estimate of 1.7 g AFDW m^{-2} is obtained for the whole Zeeschelde. It can be concluded that equation (3) can be called representative for the Zeeschelde as a system. This is not evident, as the relation might be biased by some factors, as the evolution of light limitation as metioned earlier. In the freshwater part the anthropogenic fraction of the particulate organic carbon has been estimated at around 45% during summer and 80% during winter (Hellings *et al.*, 1999; Van Damme *et al.*, 2005). The dominant benthic group in the freshwater and oligohaline part of the Zeeschelde consists of Oligochaetes, which are detrivores. Apparently, the biomass increase due to consumption of antropogenic detritus was not larger than the error margins of the approach. The reduced

residence time of the water in discharge dominated sections, and hence a restriction of plankton availability for benthos, could also alter the relation for the fresh water zone.
Despite the inevitable knowledge gaps, the approach is sound enough as a best possible estimate to derive conservation objectives for estuarine management.

7.3.7 Management

At first sight the results are ambiguous for estuarine managers. Depending on the choice of reference scenario, compensations can either be positive or negative. Even when the option for the better documented '1950' scenario as reference is made, compensation areas differ an order of magnitude if the '2000' or '2015' scenarios are assessed (Table 7.3). The immissions of the '2015' scenario are, however, close to the '1950' scenario, as only the nitrogen immissions need to be reduced (Table 7.4). In contrast with the modelled results, however, the trend of TDIN in the Zeeschelde has been decreasing since the eighties (Soetaert *et al.*, 2006) and this trend persisted also during the last decade (Fig. 7.6). At the downstream boundary of the Zeeschelde (93 km from Gent) the decrease was 16% during the last decade, while in the freshwater and oligohaline zone (km 0 - 52 from Gent) the decrease was about 33%.

Table 7.4: Reduction of immission (in %) required to meet the immissions of the '1950' scenario, for a wet, dry and mean year (BPOC = particulate biodegradable organic carbon, DOC = dissolved organic carbon)

	'2000' scenario			'2015' scenario		
	dry year	mean year	wet year	dry year	mean year	wet year
BPOC	61	60	58	0	0	0
DOC	38	34	34	0	0	0
NH_4^+-N	71	60	58	0	0	0
NO_3^--N	81	84	85	83	84	85
N tot	76	79	80	67	76	78
PO_4^{3+}-P	89	85	81	0	0	0
P tot	77	65	60	0	0	0

Adding mudflat area to the system has a positive feedback on reducing the nitrogen load through benthic denitrification. Taking into account Middelburg *et al.* (1996) and Van Damme *et al.* (this work, chapter 5), a system averaged denitrification rate of 10 mol m^{-2} y^{-1} is acceptable, leading to the result that adding 1635 ha in the '2015' scenario would reduce the immited nitrogen load with 5.6 %. This is not enough to reach the required 84 % reduction (Table 7.4), but together with the observed TDIN reduction over the last decade, the goal for '2015' is about half met. Recalculating the '2015' scenario taking into account the observed reduction yields an area compensation result of 785 ha. If this decrease would persist up till 2015, then an area claim of around 500 ha would be sufficient to compensate for loss of macrozoobenthic stock since '1950'. This claim of 500 ha is presented as a management target, on the condition that this area must be suitable for macrofaunal benthos, but also for the wader birds that feed on them. Additional mitigation measures and initiatives are needed, such as avoidance of disturbance, spatial area distribution, deriving an optimal slope for mudflat habitat, taking into account foraging density by birds, etc.

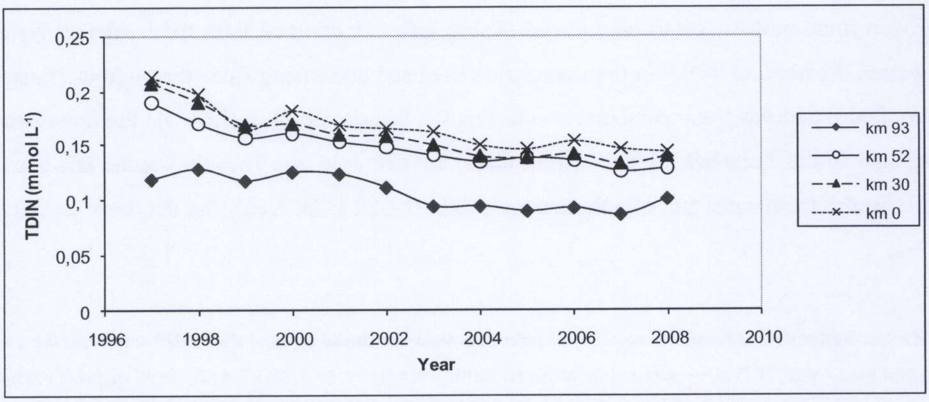

Fig. 7.6: Concentrations of total dissolved inorganic nitrogen (TDIN) over the past decade at 4 locations the Zeeschelde

7.4 Conclusions

A method to assess the need of mudflat habitat area has been presented, and was applied on the Zeeschelde, leading to a quantified area claim for benthic macrofauna as food for birds and fish. The Zeeschelde proved to be a complex case, due to the presence of both a brackish and a fresh water tidal zone of which the hydrology is partly dominated by discharge events. The concept will be easier applicable on more saline transitional waters. Nevertheless, the approach could be validated. The role of water quality, primary production and species diversity has been quantified, but other factors linked with system dynamics, morphology and specific habitat also need to be taken into account. If all these required data are covered, then the integration of datasets through this method could offer a strong universal tool to assess habitat need in a wide array of estuarine systems, as the method is based on fundamental trophic relationships. The benefit of this approach is situated in the combined implementation of both the European Habitat - and Water Framework Directives, as reference conditions and quantifications of good ecological state conditions were incorporated in the quantification of a good state of conservation of priority habitat. The approach might in other perspectives also be applied on the more open coastal systems, *e.g.* to check if benthic biomasses have been suppressed by human impacts.

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