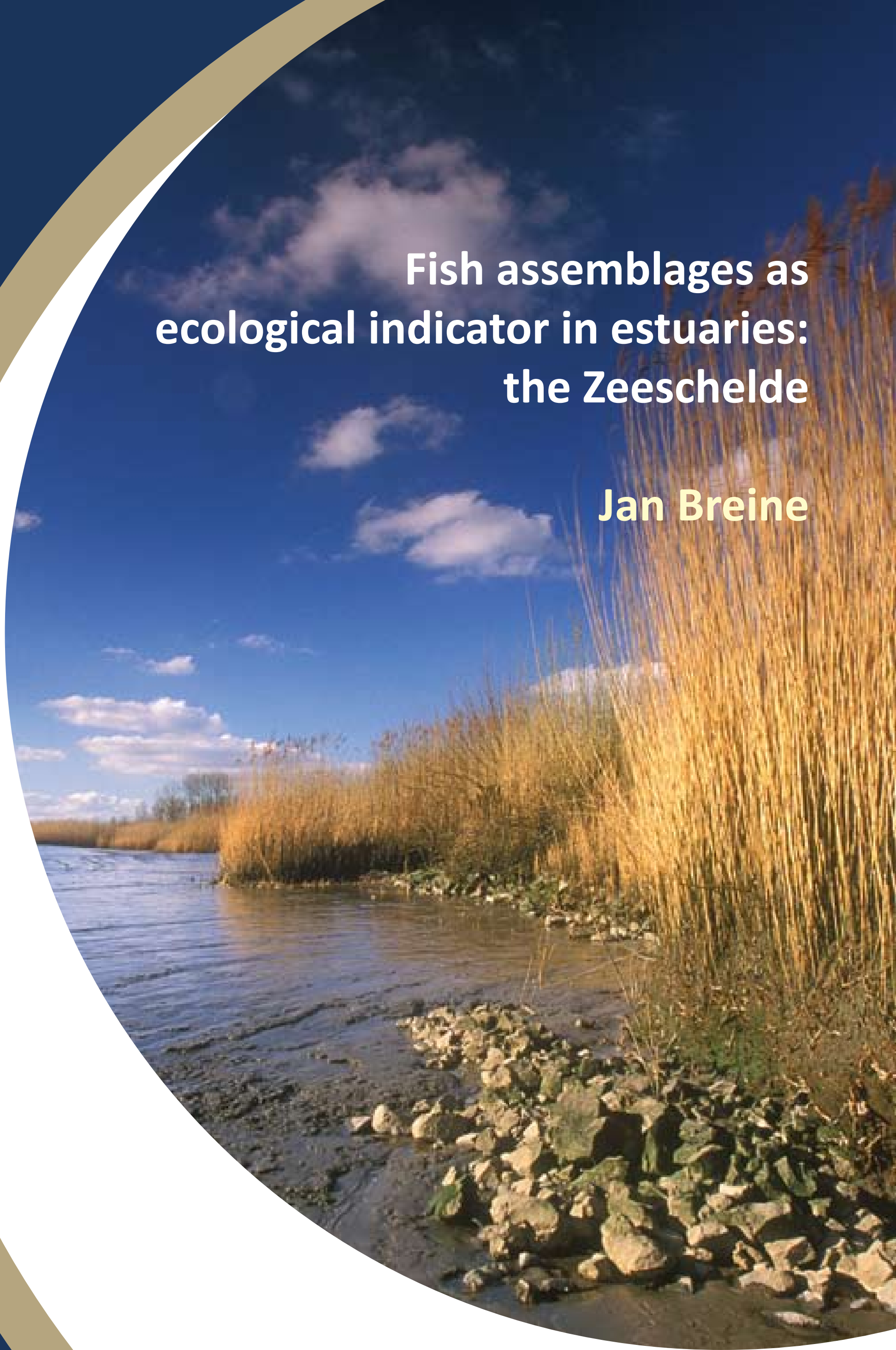

Fish assemblages as ecological indicator in estuaries: the Zeeschelde

Jan Breine

inbo

FISH ASSEMBLAGES AS ECOLOGICAL INDICATOR IN ESTUARIES:
THE ZEESCHELDE (BELGIUM)

VISGEMEENSCHAPPEN ALS ECOLOGISCHE INDICATOR VOOR ESTUARIA:
DE ZEESCHELDE (BELGIË)

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Proefschrift voorgedragen tot het behalen van de graad van Doctor in de wetenschappen

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Estuaries cannot continue to meet society's needs, or the needs of living organisms, if humans continue to regard estuarine management as a purely political or engineering challenge.

(Karr & Chu, 2000)

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Content

List of abbreviations

AIC	Akaike's Information Criterion
AMF	Average Misclassification Fraction
AMIS	Algemene Milieu Impact Studie Sigmaplan
ANB	Agentschap voor Natuur en Bos
AUC	Area Under the Curve
Be	Benthic species (stratum)
BHD	Birds and Habitats Directives
CBD	Convention on Biological Diversity
CIS	Common Implementation Strategy
CF	Catch Frequency
CRT	Controlled Reduced Tide
CPUE	Catch Per Unit Effort
Di	Diadromous species
De	Demersal species (stratum)
Dev	Deviance
df	degrees of freedom
DO	Dissolved Oxygen
DPSIRR	Drivers, Pressure, State, Impact, Response & Recovery
E	Eurytopic species
Es	Estuarine species
EBI	Estuarine index of Biotic Integrity
EC	European Community
EG	Ecological Goals
EI	Ecological Integrity
EIA	Environmental Impact Assessment
EII	Environmental Integrative Indicator
EQR	Ecological Quality Ratio
EU	European Union
EUFG	Estuarine Use Functional Group
FCA	Flood Control Areas
FMFG	Feeding Mode Functional Group
FP	Flow Preference
FS	Fragmentation Sensitive
Fw	Freshwater species
GEP	Good Ecological Potential
GES	Good Ecological Status
GIS	Geographic Information System
GREM	Groot schoor Grembergen
HAM	Groot schor Hamme
HD	Habitats Directive
HMWB	Heavily Modified Water Bodies
HN	Habitat Needs
HS	Habitat structure Sensitive
Hscore	Habitat score (pre-classification)
IBI	Index of Biotic Integrity
ICZM	Integrated Coastal Zone Management
INBO	Research Institute for Nature and Forest
INSV	Insectivorous
INVV	Invertivorous
ISC	International Schelde Commission
ISEH	International Society for Ecosystem Health
LTVS	Long Term Vision for the Schelde estuary
MEP	Maximal Ecological Potential
MDGs	Millennium Development Goals
MM	Marine Migrants
MONEOS	integrated monitoring plan for the Schelde estuary
MOW	Mobility and Public Works
MS	Marine Stragglers
MSS	Marine Seasonal Species
MSRL	Maximum Species Richness Line
MHW	Mean High Water
NOT	Tidal marsh the Notelaar
OMES	Onderzoek Milieu-Effecten Sigmaplan
OMN	Omnivorous
OC-GIS	Ondersteunend Centrum – Geografisch Informatie Centrum
PARA	PARAsitic species
PCA	Principal Component Analysis
Pe	Pelagic (stratum)
PISC	PISCivorous
PLAV	PLAnktiVorous
PROSES	Schelde Estuary Development Project
PS	Pressure Status
PSite	Pre-classification class on a site level
RIZA	Dutch Ministry of Transport, Public Works and Water Management
Rha	Rheophilic A species
Rhb	Rheophilic B species
RSD	Reproduction Special Demands
RTK-GPS	Real Time Kinematic-Global Positioning System
S	Stratum
SEA	Strategic Environmental Assessments
SOD	Schor Ouden Doel
Spa	Specialised spawners
SPPA	Strategic Planning for the Port of Antwerpen
TFW	Tidal Freshwater Zone
UK	United Kingdom
US	United States of America
VIBNA	Association of Industrial Companies of North Antwerpen
VLAREM	Flemish Regulation concerning Environmental Licences
VMM	Flemish Environment Agency
WFD	Water Framework Directive
Z-EBI	Zone-specific Estuarine index of Biotic Integrity

List of fish names

A.alb.	*Alburnus alburnus*	L.lip.	*Liparis liparis*
A.ang.	*Anguilla anguilla*	L.ram.	*Liza ramado*
A.bae.	*Acipenser baeri*	M.mer.	*Merlangius merlangus*
A.bra.	*Abramis brama*	M.sco.	*Myoxocephalus scorpius*
A.cat.	*Agonus cataphractus*	M.sur.	*Mullus surmuletus*
A.fal.	*Alosa fallax*	O.epe.	*Osmerus eperlanus*
A.min.	*Aphia minuta*	O.myk.	*Oncorhynchus mykiss*
A.pre.	*Atherina presbyter*	P.fle.	*Platichthys flesus*
A.tob.	*Ammodytes tobianus*	P.flu.	*Perca fluviatilis*
B.bjo.	*Blicca bjoerkna*	P.loz.	*Pomatoschistus lozanoi*
C.car.	*Cyprinus carpio*	P.max.	*Psetta maxima*
C.carr.	*Carassius carassius*	P.mic.	*Pomatoschistus microps*
C.gib.	*Carrasius gibelio*	P.min.	*Pomatoschistus minutus*
C.gob.	*Cottus gobio*	P.par.	*Pseudorasbora parva*
C.har.	*Clupea harengus*	P.pla.	*Pleuronectes platessa*
C.lab.	*Chelon labrosus*	P.pun.	*Pungitius pungitus*
C.luc.	*Chelidonichthys lucernus*	P.spe.	*Pomatoschistus sp.*
C.lum.	*Cyclopterus lumpus*	R.rut.	*Rutilus rutilus*
C.mus.	*Ciliata mustela*	R.ser.	*Rhodeus sericeus*
D.lab.	*Dicentrarchus labrax*	S.acu.	*Syngnathus acus*
E.enc.	*Engraulis encrasicolus*	S.ery.	*Scardinius erythrophthalmus*
E.luc.	*Esox lucius*	S.gla.	*Silurus glanis*
E.vip.	*Echiichthys vipera*	S.luc.	*Sander lucioperca*
G.acu.	*Gasterosteus aculeatus*	S.rho.	*Scophthalmus rhombus*
G.cer.	*Gymnocephalus cernuus*	S.ros.	*Syngnathus rostellatus*
G.gob.	*Gobio gobio*	S.sal.	*Salmo salar*
G.mor.	*Gadus morhua*	S.sol.	*Solea solea*
L.cep.	*Leuciscus cephalus*	S.spr.	*Sprattus sprattus*
L.del.	*Leucaspius delineatus*	S.tru.	*Salmo trutta*
L.flu.	*Lampetra fluviatilis*	T.lus.	*Trisopterus luscus*
L.gib.	*Lepomis gibbosus*	T.tin.	*Tinca tinca*
L.ide.	*Leuciscus idus*	T.tra.	*Trachurus trachurus*
L.lim.	*Limanda limanda*	Z.viv.	*Zoarces viviparus*

List of metric abbreviations

ManSha	Shannon-Weiner	MpiErs	Percentage of estuarine individuals
MnsBen	Total number of benthic species	MpiEur	Percentage of eurytopic individuals
MnsBra	Total number of species without freshwater species	MpiExo	percentage of invasive fish
MnsDia	Total number of diadromous species	MpiFlo	percentage of flounder
MnsErs	Total number of estuarine species	MpiFra	Percentage of fragmentation sensitive individuals
MnsEur	Total number of eurytopic species	MpiFws	Percentage of freshwater individuals
MnsFra	Total number of fragmentation sensitive species	MpiHab	Percentage of habitat sensitive individuals
MnsFws	Total number of freshwater species	MpiInt	Percentage of pollution intolerant individuals
MnsHab	Total number of habitat sensitive species	MpiInv	Percentage of invertivorous individuals
MnsInd	Total number of individuals	MpiMjm	percentage of marine juvenile migrating fish
MnsInt	Total number of pollution intolerant species	MpiMms	Percentage of marine migrating individuals
MnsInv	Total number of invertivorous species	MpiOmn	Percentage of omnivorous individuals
MnsMms	Total number of marine migrating species	MpiPis	Percentage of piscivorous individuals
MnsOmn	Total number of omnivorous species	MpiRha	Percentage of rheophilic (A) individuals
MnsPis	Total number of piscivorous species	MpiRhb	Percentage of rheophilic (B) individuals
MnsRha	Total number of rheophilic (A) species	MpiSme	percentage of smelt
MnsRhb	Total number of rheophilic (B) species	MpiSpa	Percentage of specialised spawner individuals
MnsSpa	Total number of specialised spawners	MpiTol	Percentage of pollution tolerant individuals
MnsTol	Total number of pollution tolerant species	MvaTol	Total tolerance value
MnsTot	Total number of species	MvdDiv	Simpson unbiased diversity index D
MpiBen	Percentage of benthic individuals	MvdSha	Shannon diversity H’
MpiDia	Percentage diadromous individuals	MvdSim	Simpson dominance index

If it looks like an estuary, smells like an estuary and behaves like an estuary, then there is a good chance that it is an estuary.

Elliott & McLusky, 2002

Chapter 1

General introduction

Our research on fish assemblages in the Zeeschelde is first situated in the frame of international agreements which are intended to halt environmental deterioration and loss of biodiversity. A brief description is given of the Zeeschelde, including some of its management issues. This is followed by an introduction of the concept of a fish-based multimetric index and a presentation of the objectives and outline of this thesis.

1 Conventions, directives and decrees to prevent biodiversity loss

The deterioration of ecosystems and their associated biodiversity is of worldwide concern. Not only biodiversity as such is at stake, but also the ecosystem goods and services on which we rely for survival are endangered. Sustainability has become an explicitly stated and legislatively mandated goal of natural resource management. The ecosystem approach has been adopted as a philosophy for managing the human uses and effects on systems (Christensen *et al.,* 1996; Secretariat of the Convention on Biological Diversity, 2004). Worldwide, international and national initiatives have led to the ratification and implementation of conventions, directives and laws in an attempt to stop the negative evolutions and to protect what is left. On a global scale the *Ramsar Convention* (1971) *on the protection of wetlands as waterfowl habitat,* was the first landmark. It provides a framework for national action and international cooperation for the conservation and wise use of wetlands and their resources with special emphasis on migrating water birds along their flyways. Likewise, the *Bonn Convention on the conservation of migratory species* (CMS, UN, 1979) enhances international cooperation to take action for migratory wild animals with unfavourable conservation status. The next major step was the adoption in Rio de Janeiro by all member states of the United Nations of the *Convention on Biological Diversity* (CBD, UN, 1992). This convention is designed to reconcile economic development with the need to preserve all aspects of biological diversity. It establishes three main goals: the conservation of biological diversity, the sustainable use of its components and the fair and equitable sharing of the benefits arising from the use of genetic resources (European Commission, 2008). Environmental sustainability is also one of the eight *United Nations Millennium Development*

Goals (MDGs, UN, 2000), drawn to end poverty and to be achieved in 2015. Two of the targets intend to reverse the loss of environmental resources and to reduce the rate of loss of biodiversity significantly by 2010. One of the outcomes during the Johannesburg Summit (2002) was the commitment to implement sustainable development and to expedite the achievement of the time-bound, socio-economic and environmental targets contained therein.

In Europe the *Convention on the Conservation of European Wildlife and Natural Habitats*, known as the *Bern Convention*, was adopted in 1979 and came into force in 1982. The EC adopted the *European Directives on the Conservation of Wild Birds* (BD, the EC birds directive 79/409/EEC) in 1979 and on the *conservation of Natural habitats and of Wild Fauna and Flora* (HD, the EC habitats directive 92/43/EEC) in 1992. These directives provide for the establishment of a European ecological network of protected areas, known as Natura 2000, to ensure the conservation of natural habitats and wild fauna and flora on land, at the coast and in the sea. Species and habitats of special interest are listed in the annexes of these directives.

To protect the European aquatic environment the *European Water Framework Directive* (WFD, EC directive 2000/60/EEC) established a framework to prevent and reduce pollution, promote sustainable water use, improve the status of aquatic ecosystems and mitigate the effects of floods and droughts. The aim of the WFD is to ensure that all European surface waters (coastal, transitional, rivers and lakes) and groundwater bodies will be in good ecological status by 2015. It constitutes a new view on water resources management in Europe: with ecosystems at the centre of the management decisions, assessment of the ecological status is based on biological quality elements with hydro-morphology and physico-chemistry as supportive elements. Likewise, the *Marine Strategy Framework Directive* (MSFD, EC directive 2008/56/EEC) establishes a framework to protect and conserve Europe's marine ecosystems and to ensure the ecological sustainability of economic activities linked to the marine environment. It aims to achieve good environmental status of the EU's marine waters by 2021, when the first evaluation of the WFD River Basin Management plans should take place. Member states must determine the "good environmental status" at the level of the marine region or subregion, on the basis of criteria such as biodiversity, the presence of non-indigenous species, stock health, the food chain, eutrophication, changes in hydrographic conditions and concentrations of contaminants, the amount of waste and noise pollution.

In the North-East Atlantic region the *Convention for the Protection of the Marine Environment* (OSPAR Convention) was opened for signature at the Ministerial Meeting of the former Oslo and Paris Commissions in Paris on 22 September 1992 and entered into force on 25 March 1998. It was established to prevent and eliminate pollution and to protect the marine environment against the adverse effects of human activities.

The BeNeLux implemented the Bonn convention specifically for fish with the decree M(96)5 of 26 April 1996. This decree aimed at free migration of fish species in the BeNeLux hydrographical network by 2010 and was recently amended by decree M(2009)1. Elimination of migration barriers are prioritised and given a timing for remediation with deadlines for the years 2015, 2021 and 2027 in concordance with the WFD deadlines.

The above mentioned conventions, directives and decrees call for specific monitoring, assessment and evaluation, measures and reporting to the competent authorities. Our study focuses on fish assemblages as an ecological indicator for estuaries with special emphasis on the WFD and HD. The WFD requires ecological quality goals to be met whereas the HD requires conservation goals to be met. The Zeeschelde, the Belgian part of the Schelde estuary, is presented as a case study.

For the WFD estuaries are categorised as transitional waters and for the HD they are considered as a separate habitat type (1130). For both typologies the upstream delimitation of this habitat is not well defined. An estuary is either defined as ”a semi-enclosed coastal body of water which has a free connection with the open sea, and within which seawater is measurably diluted with freshwater derived from land drainage” (the ”saline” definition of Pritchard 1967) or as ”an inlet of the sea reaching into a river valley as far as the upper limit of tidal rise” (the “tidal” definition of Fairbridge 1980). For Pritchard the tidal freshwater zone (TFW) is a tidal river and excluded from the estuary; under the Fairbridge definition it is categorized as the freshwater zone of an estuary. Processes within the TFW may have a strong influence on fluvial inputs to estuaries. They therefore justify specific attention in estuarine ecosystem management (Van den Bergh *et al*., 2009). From a scientific point of view, considering the importance of dynamic, complex ecological and geomorphological estuarine processes, considering also the need for common definitions to allow cross matches across EU directives for management purposes, we delimit habitat type 1130 under the HD and estuarine transitional water bodies under the WFD by the limit of the tidal influence. This includes tidal marshes, mudflats and plates as well as the channel in the salt, brackish and fresh tidal area. Guidances to both directives however, leave several options for member states to decide differently (European Commission 2003, 2007).

2 The Schelde river and its estuary

The Schelde river is a rain fed lowland river with a length of 355 km from source to mouth. It originates in the north of France (St-Quentin) at 110 m above sea level and flows into the North Sea near Vlissingen (The Netherlands) (Fig. 1.1). Its catchment area, approximately 21,800 km^2, has about ten million inhabitants (477 inhabitants km^{-2}). The river is divided in three zones: The Westerschelde in The Netherlands (58 km) between the mouth at Vlissingen and the Dutch-Belgian border, followed by the Zeeschelde (105 km) to Gent and the Upper Schelde upstream Gent. The tidal Schelde extends to Gent, where sluices interrupt the tidal wave. The tributaries of the Durme and Rupel, with the Nete, Dijle and Zenne, are also tidally influenced and are considered part of the estuary. The estuary of the Schelde has tidal mudflats and marshes along a complete and uninterrupted salinity gradient. The Westerschelde is characterized by flood and ebb channels, separated by intertidal sand and mudflats. Where the Zeeschelde starts, the river changes quite rapidly into a single ebb/flood channel, bordered by relatively narrow tidal mudflats and marshes. Due to the funnel shape of the estuary the maximum vertical tidal range is about 100 km upstream, near the Durme confluence in the freshwater zone. The mean tidal amplitude varies from 3.8 m near the mouth to a maximum of 5.43 m and back to 2 m near Gent. The estuary is well mixed with a smooth transition between salt and fresh water. The polyhaline zone (salinity (g l^{-1}) 18-30) is 40 km long, the mesohaline (5-18) and oligohaline (0.5-5) sections are 32 and 27 km respectively. The freshwater tidal part, including tributaries, has a total length of 135 km. As the longitudinal salinity profile is also determined by the river discharge, the salinity gradient can shift seasonally over a distance of 20 km. Turbidity in the Schelde estuary is high and two maximum turbidity zones might be observed: a first one at the freshwater/seawater interface (mesohaline/oligohaline zone) and a second one originating from tidal asymmetry in the fresh tidal zone (Baeyens *et al.*, 1998; Herman & Heip, 1999; Meire *et al*., 2005; Van Damme *et al*., 2005; Van den Bergh *et al*., 2005).

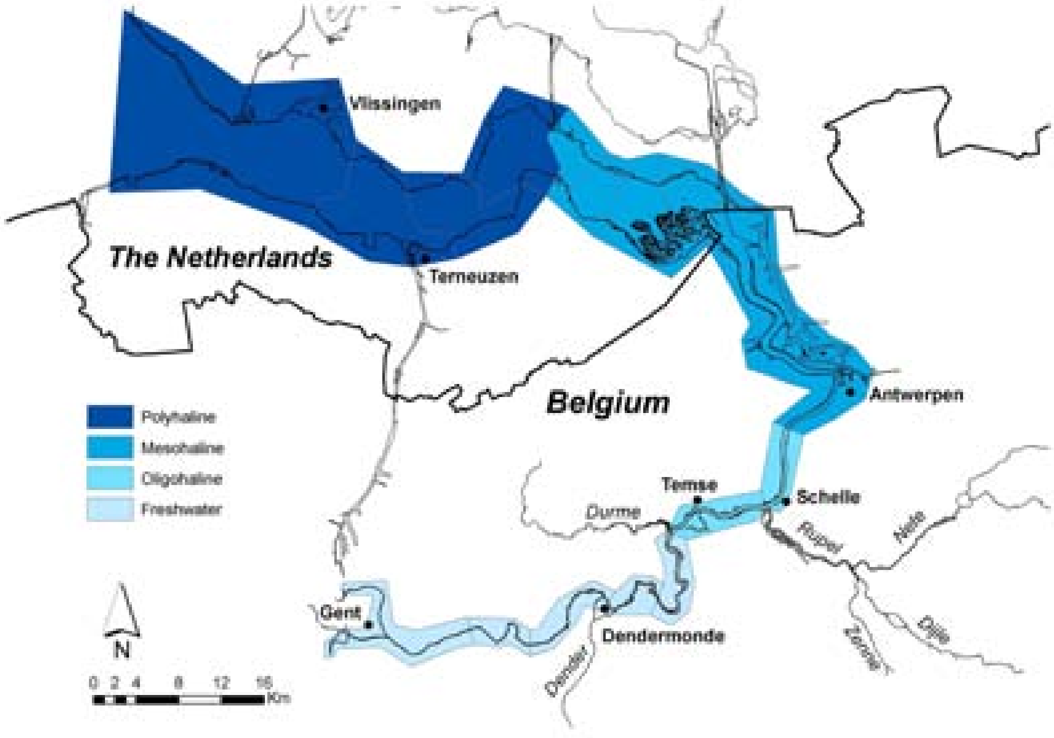

Figure 1.1: Geographical and salinity map of the Schelde estuary.

3 The Zeeschelde

The Zeeschelde, together with its tributaries under tidal influence has a total surface area of almost 53 km² with 890 ha of tidal marshes and 950 ha of tidal mudflats (Fig. 1.2). Although its ecological values are potentially very high, the Zeeschelde is heavily impacted anthropogenically. Historically tidal marshes were reclaimed for agriculture and agglomerations and industries developed close to the riverbank. Tidal wetlands also disappeared due to the large dikes of the SIGMA flood control plan. The navigation requisites for the Zeeschelde, and especially for the port of Antwerpen, call for extended maintenance dredging. Estuarine dynamics increased under the combination of reduced space for the estuary, channel deepening and widening, and the general sea level rise. As a result also indirect habitat loss through erosion was observed in the past decade (Van den Bergh *et al*., 2005a). The importance of this habitat loss for fish is further elaborated in chapters 4 and 9.

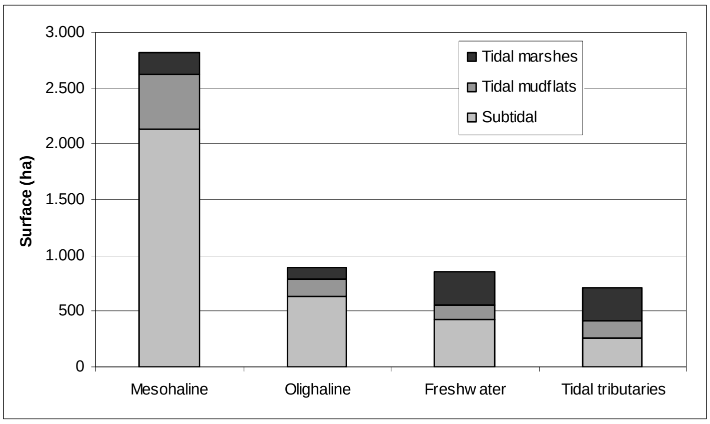

Figure 1.2: Absolute surface (ha) of habitats in the Zeeschelde and of its tidal tributaries.

The Zeeschelde is subject to severe eutrophication as it receives high inputs from domestic, industrial and agricultural activities. Until 2007 the untreated sewage from Brussel reached the Zeeschelde through the Zenne and the Rupel. Since a new sewage treatment plant in Brussel is functioning spectacular changes are being observed in the estuary (Chapter 2). Fishery in the Zeeschelde was extensive until the 1925 (Vrielynck *et al.*, 2003) causing the disappearance of various species such as the sturgeon (*Acipenser sturio*) and allis shad (*Alosa alosa*). At present occasionally one trawler is active.

3.1 Management issues

The ecological values and nature conservation interests of the Zeeschelde are subject of a series of (inter)national agreements and legal commitments. The Schelde River Basin District in terms of the WFD extends to The Netherlands, Flanders, Wallonia, Brussels and France and is shared by six different governments (the adjoining coastal water is partly under the jurisdiction of the Belgian Federal and partly under the jurisdiction of the Dutch Government) (Fig. 1.3).

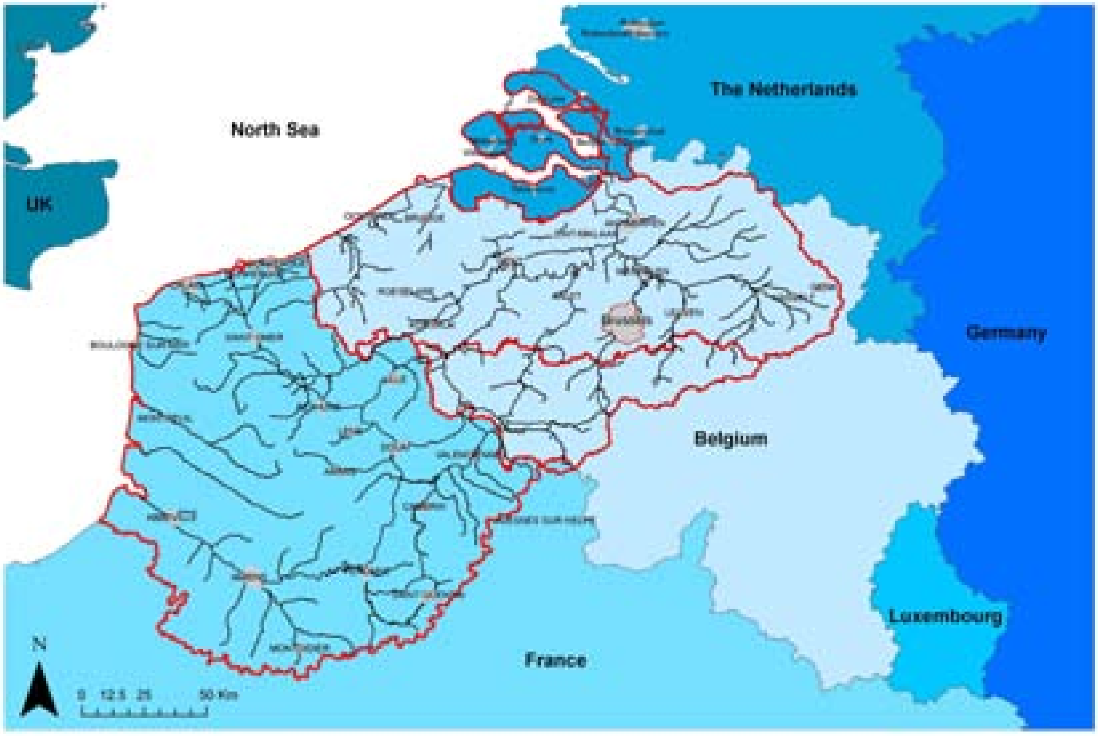

Figure 1.3: Schelde River Basin District and its surface water bodies (source, ISC).

The estuary is shared by The Netherlands and Flanders; its political management is formalised in the *Long Term Vision for the Schelde estuary* (LTVS), a Dutch-Flemish management plan that was established when Flanders started to negotiate about the third deepening of the Westerschelde. The LTVS sets quality targets for the estuary by the year 2030 from three central perspectives (accessibility, flood control and ecosystem health) and the management measures to achieve them. On a regional level the *Updated Sigmaplan* provides the framework for integrated water management and combines flood control and ecological rehabilitation in the Zeeschelde. On a local level the *Strategic Planning for the Port of Antwerpen* (SPPA) assures the optimal multifunctional spatial planning in the harbour area, including the conservation goals for Natura 2000 within the port area. These different management plans apply partly to the estuary or to parts of the estuary (Fig. 1.4). They overlap in space, time and subject and the initiatives originate from different authorities. As a result responsibility for the management i.e. regulation, protection and development of the estuary is embedded in a patchwork of local, regional, and international agencies.

Figure 1.4: Management plans in the Zeeschelde. (LTVS: Long Term Vision for the Schelde Estuary; SPPA: Strategic Planning for the Port of Antwerpen; US: Updated Sigmaplan).

International and national legislations differ in scale and focus on different levels of ecosystem functioning. The WFD aims at good environmental and ecological quality on a river basin district level, while LTVS aims at estuarine processes. The Natura 2000 network has a quantity and connectivity issue for specific habitats and species. In an ideal situation harmonised management would be nested in space, time and issues, different sectors would share delimitations of management units and congruent typologies would be used for different initiatives. The International Schelde Commission (ISC) provided a transnational management approach for the Schelde river basin, bringing together six countries and regions, with the application of the ISC to include surface water, ground water and coastal water throughout the river basin as well as joint consultation on the mitigation of calamities and drought and flooding effects. However, this transnational ecosystem approach was not continued with the administrative implementation of the WFD and HD. For the former the Schelde estuary was divided into 8 distinct water bodies. Moreover, the fresh tidal water bodies were categorised as rivers. For the latter, tidal mudflats and marshes of Zeeschelde and Durme only were designated as habitat type 1130. Moreover, the channel in the fresh tidal area of the

Zeeschelde was not designated. According to the annex in the HD twaite shad (*Alosa fallax*) is extirpated and no habitat was designated in Flanders.

3.2 Prospects for the Zeeschelde

LTVS ecological goals for the Zeeschelde were specified and quantified as an integrated part of the *Updated Sigmaplan* for flood control. Restoration measures to achieve the conservation objectives were combined into an ecological rehabilitation plan based on ecosystem functioning, resilience, goods and services. Simultaneously an optimal flood control plan 'the *Updated Sigmaplan*' was designed. After integration with the conservation goals and measures for *Natura 2000* in the port of Antwerpen and consultation with a wide range of stakeholders both plans were combined in what was called 'The preferred alternative to the *Updated Sigmaplan*'. Its major issue was the coupling of ecological rehabilitation and sustainable nature with flood control measures and navigation requisites (Van den Bergh *et al.*, 2009). This plan is now being implemented and should be completed by the year 2030. It includes the creation of 1400 ha tidal wetland through managed realignment (i.e. tidal wetland restoration by dike relocation), 1100 ha tidal wetland under reduced controlled tide in flood control areas (FCA-CRT), 1500 ha of 'winter bed' for the upper reaches and 2000 ha of non-tidal wetlands, 1000 ha of which are located in flood control areas (FCA-Wetland) (Fig. 1.5). Meire *et al.* (2005) evaluated that with this plan the conservation goals for a healthy and resilient estuary will be realised, on condition that all measures are realised according to the plan. In the general discussion I demonstrate the importance of wetlands for fish.

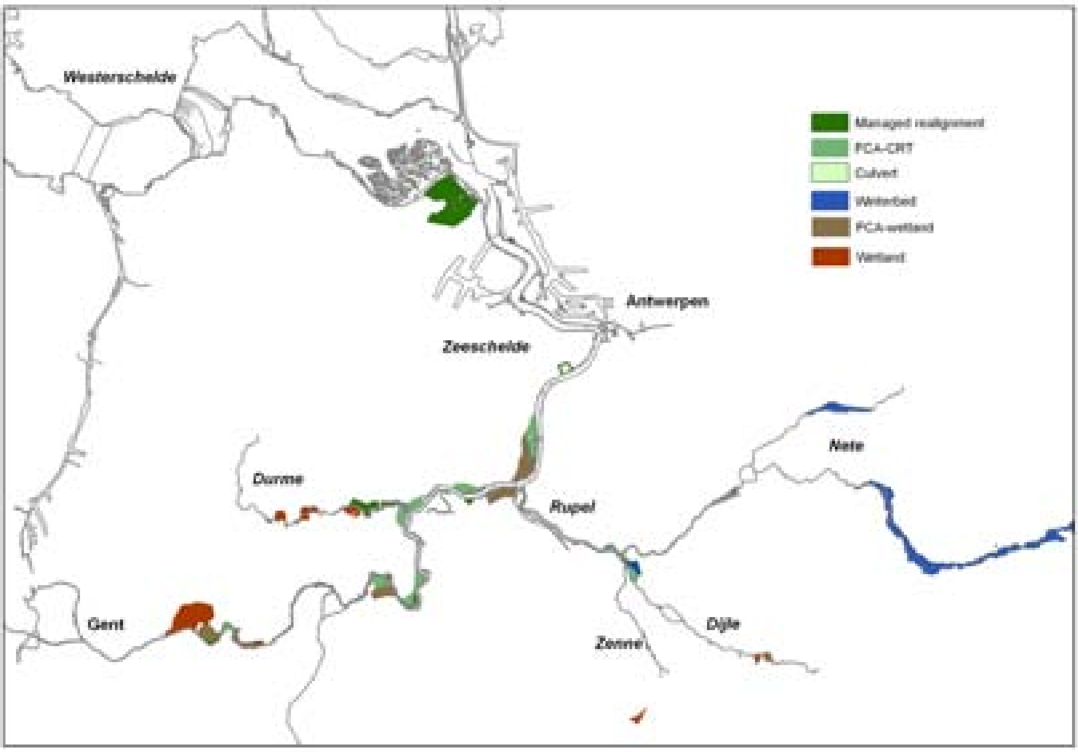

Figure 1.5: Ecological restoration measures for the updated Sigmaplan.

4 Fish as a management assessment tool

The evaluation of the implementation requires monitoring of biota for most of the above mentioned agreements. For all management plans with ecological goals or obligations the derivation of applicable indicators of ecosystem health is important for their assessment. Whilst measurements of physical, chemical and biological components of the estuarine ecosystem have been routinely carried out for decades, the identification of determinants which are sufficiently robust at describing functional health is less easy. Such indices of quality and function need to be able to inform the managers and address whether specific ecological goals and associated habitat needs are being met. In this thesis fish assemblages are used as a tool to measure the ecological status of the estuary. The main advantage is that fish assemblages integrate the direct and indirect effects of stress on the entire aquatic ecosystem and manifest the ecological significance of the perturbation (Fausch *et al*., 1990).

In the WFD fish is a biotic quality element for the assessment of the ecological status in transitional waters taking into account species composition, abundance and the proportion of disturbance-sensitive species. Any distortion in these must be attributable to anthropogenic

impacts and is calculated by means of the Ecological Quality Ratio (EQR) representing the difference between monitored data and reference conditions. This EQR contains five quality classes ranging from bad (close to 0) to high status (close to 1). River lamprey (*Lampetra fluviatilis*), sea lamprey (*Petromyzon marinus*), bitterling (*Rhodeus sericeus amarus*), striped mudminnow (*Cobitis taenia*), weatherfish *(Misgurnus fossilis*), twaite shad (*Alosa fallax*), bullhead (*Cottus gobio*), European sturgeon (*Acipenser sturio*), allis shad (*Alosa alosa*), houting (*Coregonus oxyrhynchus*) and Atlantic salmon (*Salmo salar*) are the fish species on the Annex II of the HD with relevance for the Zeeschelde.
The Natura 2000 site BE 2300006 (Schelde form the Dutch border till Gent including the Durme) was designated for striped mudminnow, bitterling and river lamprey only.

4.1 Introduction to the fish-based index of biotic integrity: a useful tool for biological evaluation

In the US there was a need to address the American watershed legislation outlined in the Clean Water Act (1972). Its goal was to restore and maintain the chemical, physical and biological integrity of the Nation's waters. Apart from chemical and physical quality scientists needed to understand quantity and quality of faunistic elements. Cairns *et al*. (1973) stated that the assessment of pollution should include an integration of chemical, physical and biological monitoring. The most important reason for biological monitoring was the consideration that aquatic organisms function as natural monitors. Organisms intolerant to stress will be destroyed when the water quality declines, causing a change in the community structure (Patrick, 1949).

Verneaux (1981) pioneered the first multimetric index using fish communities and benthic macroinvertebrates to assess the water quality in French rivers. It was based on similar principles to those already published in 1976 (Verneaux *et al*., 1976; Verneaux & Tuffery, 1976). This fish model considered only qualitative aspects of the fish assemblages. Based on the assumption that the relative health of a fish community is a sensitive indicator of the relative health of its aquatic ecosystem Karr proposed to use fish assemblages to assess the ecological quality of running waters (Karr, 1981). He developed a multimetric index including two categories: 1) species composition and richness and 2) ecological assets including trophic composition and abundance. He selected 12 metrics integrating information on different levels from individual specimen to ecosystem level. These metrics are sensitive to a range of biological stresses and allow discrimination between anthropogenic changes and the natural background "noise" of healthy system variations. A fish-based index of biotic integrity (IBI) is a quantitative expression of a number of known relationships between anthropogenic disturbance and the characteristics of the resident fish community. It can be used to evaluate current conditions at a site, determine trends, compare sites and to some extent to identify the cause of local degradation (Karr *et al*., 1986). According to Karr one could, by carefully monitoring fish, rapidly assess the health ("biotic integrity") of a local water resource. Biotic integrity was defined as the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organisation comparable to that of a natural habitat of the same region (Frey, 1975; Karr & Dudley, 1981). System integrity is reflected in the biotic elements and the processes that generate and maintain these biota, whereas diversity describes only the elements (Karr & Kerans, 1992; Angermeier & Karr, 1994). When impacts are recorded Karr suggested implementing a more complete monitoring programme in search of causative agents. He summed seven advantages to use fish as indicator organisms for biological monitoring. However, he did not hide the disadvantages but concluded that regular use of fishes would improve the resolutions of monitoring and assessment programmes previously encountered in the US. His system consisted of six discrete classes to evaluate fish communities: excellent, good, fair, poor, very poor and no fish.

His first assumption was that species composition and richness metrics will react in a predictable way in the presence of human impacts.

A second assumption was that human impacts affect the production and consumption dynamics and these changes can be assessed by examining the trophic structure of the community. Changes in water quality or other habitat conditions result in shifting availabilities of many food resources. Fausch *et al.* (1990) elaborated further on these assumptions (Table 1.1).

At this stage Karr did not incorporate metrics from reproductive guilds, age structure and growth rate, because the necessary information was not easily obtained. He suggested nevertheless that these metrics might be used to assess rivers. Barbour *et al.* (1995) observed a reduction in mean size of organisms caused by environmental stressors. Karr *et al.* (1986) described five major classes of environmental factors that affect aquatic biota in rivers: 1) energy source; 2) water quality; 3) habitat quality; 4) flow regime and 5) biotic interactions. Angermeier and Karr (1994) adjusted these five major sources of variation in aquatic environments to: 1) physicochemical conditions; 2) trophic base; 3) habitat structure; 4) temporal variation and 5) biotic interactions.

Table 1.1: Assumptions about biological patterns associated with increasing human effects on stream biota (modified from Fausch *et al*., 1990).

Stream biota	**reaction**
Number of native species	declines
Number of specific taxa on habitat guilds	declines
Number of intolerant species	declines
Proportion of tolerant individuals	increases
Proportion of trophic specialists	declines
Proportion of trophic generalists	increases
Fish abundance	declines
Proportion of individuals in reproductive guilds requiring silt-free coarse spawning	declines
Incidence in hybrids	increases
Incidence of external evident disease, parasites and morphological anomalies	increases
Proportion of introduced individuals	increases

According to Karr the accumulated information across a range of metrics provides a greater resolving power for the overall index than each metric separately. One metric can not assess all forms of degradation and be sensitive across the full range of degradation. The level of accuracy will improve by combining metrics. One should be aware that the sensitivity of metrics is different e.g. total number of species declines monotonically across the range of degradation while intolerant species disappear before degradation has proceeded very far (Karr, 2006). Such degradation patterns can be used to identify responsible degradation factors. Sites with similar IBI scores can indeed have a different breakdown of metric scoring.

Thompson and Fitzhugh (1986) were possibly the first to develop an IBI for estuarine fish communities in coastal Louisiana. Using the acquainted knowledge about fish reactions in rivers, Deegan *et al.* (1997) assessed how estuarine fish assemblages respond to habitat degradation. The authors hypothesised 15 responses of top trophic levels to disturbance in estuaries of which 8 were confirmed. These responses were integrated in their estuarine index. This estuarine index kept the main IBI categories: species composition and richness, trophic composition and abundance, but modified the metrics to reflect estuarine habitats and fish assemblages. They pre-classified different sites in the estuary and assigned fish species to life-history, trophic and location categories corresponding with the later work of Franco *et al*.

(2008) (life-history: freshwater species, diadromous species, resident species spawning in the estuary, nursery species, marine species, adventitious species; trophic category: filter feeders, zooplankton pickers, invertivores, piscivores, omnivores; location: benthic or pelagic). I explain in chapter 7 how the first estuarine index based on fish communities was developed in Flanders.

4.2 Sampling

The use of resident biota of streams provides an integrative view of human effects and a rich variety of signals that can be used to diagnose the causes of degradation (Karr, 2006). When using such an assessment system one must assume that the fish sample is a balanced representation of the fish community at the sample site. In addition the sample site must be representative of the larger area of interest. There has been a lot of research to define how many samples and how many sites must be surveyed in order to fulfil these assumptions (Breine *et al.*, 2001; Hughes & Herlihy, 2007; Simoens *et al.* (in prep); Van Liefferinge *et al.* (in prep)). Deegan *et al.* (1997) stated that complete seasonal sampling in the estuaries (Waquoit and Buttermilk bay, Massachusetts, US) was not necessary to characterise the major trends and suggested to sample in summer when the cumulative impacts on the fish is highest. Karr (1991) stated that natural streams tend to have small seasonal variations while disturbed areas tend to be unstable (proven by Paller, 2002). Therefore he suggested early summer for a primary sample time as it is the least variable period from year to year. He also stated that an assessment is likely to detect degradation, regardless of the factor responsible for it. This is because ecosystems with a high biotic integrity have the resilience to recover from most natural perturbations (Cairns, 1975). A fish survey methodology for estuaries is described in chapters 7, 8 and 9.

4.3 Classification process

The key problem is to define a type specific baseline for classification. Indeed expectations vary among systems e.g. upstream waters support fewer species than downstream areas. Initially Karr (1981) assigned to each fish sample a grade (minus, zero or plus) for each metric and assigned values to each grade (1, 3 or 5). By summing up these values an index score was obtained. He emphasized that this method was preliminary and that an evaluation must be made with respect to the expectations for a relatively undisturbed natural habitat for that region. Ideally a regional reference site is used as a benchmark. Deegan *et al.* (1979) had no reference sites and could not set the metric threshold values representing a high quality score. For metrics that were different between low- and medium-quality habitats critical values were defined using the 10^{th} or 25^{th} percentile of metric values in these habitats. For the Zeeschelde a reference is absent and a similar approach was adopted to develop the first estuarine index (Chapter 7). However, a reference was defined (Chapter 3) and used to set threshold values for the selected metrics (Chapter 8).

5 Scope and outline

The Zeeschelde and its basin contain important Ramsar and Natura 2000 sites and the *European Water Framework Directive* (WFD, 2000) needs to be implemented. However, the system is neither in a favourable conservation status (HD) nor in a good ecological status (WFD) (Paelinckx *et al.*, 2008; Speybroeck *et al.*, 2008). Surface as well as quality of tidal mudflats and marshes are decreasing, first due to embankment and infrastructure works, more recently also because of destructive erosion (Van den Bergh *et al*., 2005a). It became apparent that restoration measures are needed and a more integrated management approach is required. The LTVS, SPPA and the *Updated Sigmaplan* are three initiatives in the Schelde estuary in the pursuit of this goal. They set conservation goals and the measures to achieve them, each from a different perspective and scale, yet taking each other into account. These management efforts must be assessed. A first step is the assessment of the ecological status of the estuarine ecosystem.

In this thesis I focus on fish as a bio-indicator for such assessment. Four major goals were defined:

1) to describe the current fish assemblages in the Zeeschelde
2) to define a fish reference list for the Zeeschelde based on historical and contemporary data for the different salinity zones
3) to define ecological goals and associated habitat needs for fish assemblages in the Zeeschelde
4) to develop a fish-based tool to assess the ecological status of the Zeeschelde

This thesis is composed of three parts.

The first part (chapters 2 and 3) deals with two questions:

1 What fish assemblages occur in the Zeeschelde? (current situation)

2 What fish species should be present in the Zeeschelde? (ideal situation)

In **chapter 2** the current fish assemblages in the Zeeschelde are described based on fyke net catches. An ecological approach is adopted rather than restrict the analysis to an annotated checklist. Fish were grouped in functional guilds. The term guild is used as synonym for a functional group, which represents ecosystem processes the species perform through resource exploitation (Blondel, 2003). These functional guilds cover estuarine use, mode of feeding and reproductive strategy, thus describing the utilisation of transitional waters by fish (Franco *et al.*, 2008). This information is needed to define whether the fish assemblages show a significant spatial and temporal pattern, which could have an impact on the development of a fish-based assessment tool for the Zeeschelde (Chapters 7 & 8). Secondly, this information is also combined with historical data to define fish reference lists (Chapter 3). Thirdly, these fish data are used to calculate metric values a necessary step in the process of the development of a fish-based index (Chapters 7 & 8).

In **chapter 3** fish species lists are produced, indicating expected species when the Zeeschelde estuary would be at its good or maximal ecological potential. The maximal ecological potential (MEP) is considered as the reference condition for a heavily modified surface water (Borja & Elliott, 2007). The good ecological potential (GEP) has a slightly lower quality than the MEP. An empirical approach is proposed in which actual data (Chapter 2) are combined with historical data. The MEP lists include recent and historical recorded species including occasionally recorded ones (<5% catch frequency in recent data). The GEP lists do not include these rarely recorded species. Only species belonging to the GEP reference lists are considered for the development of ecological goals and associated habitat needs (Chapter 4). These lists also contribute to the development of the fish index. Only reference species from the GEP lists are considered to be assigned to a metric. In addition, threshold values for metrics are based on these GEP lists (Chapter 8).

The **second part** answers the question: “What does an improved fish assemblage look like in an estuary?”

This part assists managers in understanding the estuarine ecosystem functioning from a fish perspective. It contains three chapters.

In **chapter 4** ecological goals and associated habitat needs for estuarine fishes are defined using the Zeeschelde as an example for other North-East Atlantic estuaries. The term ecological emphasizes that these are functional targets within the ecosystem, including interactions among the different fish species and between fish and their environment. These ecological goals and associated habitat needs are defined for specific fish functional guilds on four spatial scales of interest: the regional scale, the river basin scale, the estuary scale and the habitat scale. Ecological goals for estuarine fishes in the GEP lists (Chapter 3) are defined as targets that should be reached in order to ensure a healthy and dynamic fish community in that ecosystem in casu the GEP status. The associated habitat needs ensure spawning, breeding, feeding and growth to maturity. Even though the habitat needs are qualitative or semi-quantitative, suggestions are given for their achievement. The importance of two specific habitat needs is described in chapters 5 and 6.

In **chapter 5** a model for the oxygen requirements of diadromous fish species is presented as a case study to quantify associated habitat needs. For remaining populations of diadromous species, a data driven logistic model is parameterized. The presence or absence of fish species is modelled as a function of temperature, dissolved oxygen concentration, river flow and season. Necessary measures with respect to watershed management are proposed.

In **chapter 6** the life history function of tidal marshes in the Zeeschelde is assessed with special emphasis on the fresh tidal ones. Different fishing protocols are applied in order to assess: 1) spatial and temporal effects and 2) the influence of creek characteristics on the fish assemblages. The results are relevant for tidal wetland development schemes. In addition the results contribute to the challenging task of defining the carrying capacity (i.e. fish biomass) of the estuary.

In the **third part** I answer the question: “How can we assess the ecological status of the Zeeschelde?”

To assess the ecological status a classification tool especially developed for that purpose is needed. This third part contains two chapters describing the development of a fish-based index. Chapter 7 should be regarded as a methodological paper. It describes an approach to select metrics such that the quality assessment errors (Type I and II) are small and balanced.

However, while the approach is universal, it needs calibration data from sites of different habitat status (from bad to high). As some of these data are unavailable the developed index is limited (from bad to moderate) and another strategy has to be developed in order to obtain a full scale index (Chapter 8). The main differences are that here a “good” reference (GEP) is used and that the index assesses the estuary on a larger scale, including the mesohaline, oligohaline and freshwater zones.

In **Chapter 7** a new approach is presented to define an optimal combination of metrics for creating a fish-based estuarine biotic index (EBI) for defining the quality status of a brackish estuarine area. Metric values are calculated using data described in chapter 2. A pre-classification to score and select metrics for further statistical analysis is applied. By balancing misclassification errors a set of metrics that constitute the EBI is selected. The developed index assesses the mesohaline zone in the Zeeschelde on a site level using monthly data.

In **Chapter 8** a zone-specific fish-based multimetric estuarine index of biotic integrity (Z-EBI) is developed. First a pre-classification exercise is performed and data from surveys along the complete salinity gradient in the Zeeschelde (Chapter 2) are screened to calculate the metric values. Then metric values are calculated using only fish species occurring in the reference lists (Chapter 3). Initial metrics are selected using the pre-classification, also taking into consideration spatial and temporal patterns as observed in chapter 2. Final metrics are selected based on statistical analysis and ecological knowledge and the reference list is used to calculate the metric threshold values. The index assesses the three salinity zones in the Zeeschelde.

Finally, **Chapter 9** contains the general discussion where I further incorporate the results obtained in previous chapters. I highlight particular issues indicating gaps in ecological knowledge and how the results can be translated into management measures and incorporate recommendations for further research.

Chapter 2

Fish assemblages across a salinity gradient in the Zeeschelde estuary (Belgium)

Jan Breine, Joachim Maes, Frans Ollevier & Maarten Stevens

Abstract

Between 1991 and 2008 a total of 71 fish species was recorded within the Zeeschelde estuary. The results were obtained from fish surveys from the cooling water filter screens of the power plant at Doel (between 1991 and 2008) and fish surveys along the length of the estuary (collected with 'paired-fyke' nets between 1995 and 2008). Species abundance in the salinity zones was analysed using the fyke net data only. The ten most abundant species represent 90.8% of the total number of individuals caught. In decreasing order of abundance: flounder *(Platichthys flesus),* roach (*Rutilus rutilus*), herring (*Clupea harengus*), eel (*Anguilla anguilla*), pike-perch (*Sander lucioperca*), sole (*Solea solea*), common goby (*Pomatoschistus microps*), seabass (*Dicentrarchus labrax*), three-spined stickleback (*Gasterosteus aculeatus*) and white bream (*Blicca bjoerkna*). Species richness ranges from 33 species in the tidal freshwater zone, 43 species in the oligohaline zone to 59 species in the mesohaline zone. Each salinity zone is characterised by a typical fish assemblage, although some species are shared between all three salinity zones: e.g. three-spined stickleback (*Gasterosteus aculeatus*), Prussian carp (*Carrasius gibelio*), roach (*Rutilus rutilus*) and eel (*Anguilla anguilla*). Freshwater species comprise about 70% of the species in the freshwater zone. In the oligohaline zone the contribution of the freshwater species to the species richness is less while marine migrants become more abundant. As expected, the contribution of marine migrants and estuarine species is higher in the mesohaline zone. The recorded differences in fish assemblages between the different zones could be attributed to habitat differences and hence the definition of different estuarine fish guilds. The recent increase in species richness in the freshwater and oligohaline zone coincides with a remarkable increase in dissolved oxygen since 2007. It is argued that the fish community in the Zeeschelde would benefit from the creation of a diversified habitat with connected floodplains, mudflats and tidal marshes.

Keywords: fish assemblages, salinity gradient, spatial variation, Zeeschelde

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1 Introduction

Estuaries play a key role in nutrient cycling and transformation, and are an essential habitat in the life cycle of many organisms, in particular fish and waterfowl (Colclough *et al*., 2005; McLusky & Elliott, 2004). An estuary is that part of a river which is under tidal influence and is characterised by a continuous salinity gradient (Fairbridge, 1980). Hence fish assemblages in estuaries are very diverse and composed of marine, estuarine, freshwater and migrating species (Henderson, 1988; Lobry *et al*., 2003). Elliott and Dewailly (1995) assessed the fish assemblage structure in 17 European estuaries. They identified functional guilds according to the habitat use of each fish species encountered. This guild approach facilitates the comparison of fish assemblages across different estuaries (e.g. Lobry *et al*., 2003). Recently Franco *et al*. (2008) validated the functional guild approach. Estuaries in Northwest Europe have been the subject of considerable research focussing on the functioning of the different habitats (e.g. Dolbeth *et al.*, 2007; Elliott *et al.*, 2007b). Their role as nursery and feeding areas, refuges and migration routes have been described for specific estuaries such as the Zeeschelde (Maes *et al.*, 2007, 2008), the mudflats in the Westerschelde (Cattrysse *et al.*, 1994), the Forth estuary (Elliott *et al.*, 1990) and in general by Elliott & Hemingway (2002) and McLusky & Elliott (2004). Other research focused on spatiotemporal patterns in fish composition and assemblage structure indicating that fish communities differ in space and time (Potter *et al.*, 1997; Marshall & Elliott, 1998; Araújo *et al*., 1999; Thiel & Potter, 2001; Jovanovick *et al.,* 2007; Selleslagh & Amara, 2008; Selleslagh *et al*., 2009). Spatial patterns in estuarine species assemblages are mainly correlated with salinity (Henderson, 1989), while temporal variations are mostly the result of migration of young fish (Maes *et al*., 1998a; Thiel & Potter, 2001).

The fish community in the Zeeschelde, the Belgian part of the Schelde estuary, has been studied since the 1990s. However, studies in the earlier years are mostly limited to the mesohaline and oligohaline zone, only occasionally including one site in the freshwater zone (Van Damme *et al.*, 1994; Maes *et al.*, 1997; Maes *et al.*, 1998a,b; Peeters *et al.*, 1998; Maes *et al*., 1999; Peeters *et al.*, 1999; Ercken *et al.,* 2002; Maes *et al.*, 2003, 2004a,b, 2005; Stevens, 2006; Stevens *et al.*, 2006; Cuveliers *et al.*, 2007; Buysse *et al.*, 2008 and Guelinckx *et al.*, 2008). Vrielynck *et al.* (2003) give a historical overview of fish species present in the salt and brackish parts of the Zeeschelde and its tributaries. The Rupel (oligohaline tributary) and Durme (freshwater tributary) have been surveyed annually since 2004 (Breine & Van

Thuyne, 2004, 2005; Breine *et al.*, 2006, 2007; Van Thuyne & Breine, 2008). Since 2007 volunteers monitor fish all year round at different sites along the salinity gradient of the Zeeschelde, including the tidal freshwater zone.

The main aim of this study is to describe the fish assemblage along the salinity gradient in the Zeeschelde estuary based on sampling results in the mesohaline, oligohaline and freshwater zone and, to provide an overview of its temporal and spatial variation (measured as species richness). We assessed seasonal patterns and tested the oxygen concentration impact on these fish assemblages.

Estuaries are among the most impacted aquatic environments. Many fish species are vulnerable to the effects of human impacts and can therefore be employed as indicators of the ecological quality of the estuarine ecosystem (Breine *et al*., 2007). One essential step in the assessment of ecological quality is the development of a reference list which can be used as a benchmark. In chapter 3 we develop a reference species list for the Zeeschelde combining historical data with data on the current situation here presented.

2 Material and methods

2.1 Study area

The river Schelde is a tidal lowland river with its origin in the northern part of France (St. Quentin), and its mouth in the North Sea near Vlissingen, The Netherlands. With a total length of 355 km, the fall is approximately 100 m and the mean depth about 10 m (Baeyens *et al.*, 1998). The main river and tributaries are rain-fed, with a minimal discharge in summer and autumn, causing the salt water to penetrate further upstream in these seasons. Tides with an average amplitude of 4.5 m penetrate 160 km upstream. The salinity profile is mainly determined by the river discharge and to a far lesser extend by the tidal action (Baeyens *et al.*, 1998).

In the Zeeschelde (the Belgian part of the estuary, Fig. 2.1) three zones are distinguished: a mesohaline zone between Zandvliet and Antwerpen, an oligohaline zone between Antwerpen and Temse, including the Rupel tributary, and a tidal freshwater zone till Gent including the Durme tributary. In Gent the effect of the tide is abated by a complex of sluices. The Rupel is an oligohaline tributary; the tidal part of the Durme is interrupted downstream Lokeren in the 1960’s and functions now as a large freshwater tidal creek of the main river. The tidal amplitude in the Durme is quite important (average 5.40 m at Tielrode), therefore habitat conditions change drastically between incoming and outgoing tides. Both Rupel and Durme have important mudflats (26 and 24 ha) and marshes (43 and 100 ha).

The oligohaline zone has been impacted for decades by untreated sewage water from metropolitan Brussels. From 1925 onwards fish was absent in the Rupel river (Vrielynck *et al.*, 2003). The industrial areas of Lille (France), Gent and Antwerpen (Belgium) and Vlissingen (The Netherlands) have a major negative impact on the water quality (Van Eck *et al.*, 1991). For years the Zeeschelde off Antwerpen remained anoxic, creating an effective barrier for diadromous fish (Maes *et al.*, 2007, 2008). As water treatment efforts increased and diffuse pollution along the river reduced, the water quality ameliorated and some shifts in oxygen regime and nutrient cycling were observed (Maris *et al.*, 2008, Van Damme *et al.*, 2005; Van den Bergh *et al.*, 2005). Since March 2007 most sewage water from Brussels is treated and since then some improvement in water quality of the Rupel river has been observed (Van Thuyne & Breine, 2008; Stevens *et al.*, 2009). However, the Zeeschelde still receives significant discharges of raw industrial and domestic waste water, as well as diffuse pollution from agricultural runoff, resulting in a poor water quality within a large part of the estuary.

2.2 Data collection

Data were collected at 32 different sites in the Zeeschelde and its tributaries (Fig. 2.1). Surveys occurred with fyke nets (detailed description see p. 22) between 1995-1999 and 2001-2008 within the mesohaline zone and between 1997-1999 and 2001-2008 within the other salinity zones. Collections at the cooling water intakes of the power station at Doel (N° 23, Fig. 2.1) were made between 1991-2001 and 2003-2008. All field work was performed by trained fish biologists and trained volunteers using a standardised protocol (see Breine *et al*., 2007). All fish were identified to species level, counted and returned into the estuary. Occasional cross examination in the laboratory assured the quality of the fish identification.

Data were collected by assignments from Flemish Environmental Agency (VMM), Association of Industrial Companies of North Antwerpen (VIBNA, Vereniging van de Industriële Bedrijven van Noord-Antwerpen), MOW – Department of Mobility and Public Affairs, division Maritime Access and the Research Institute for Nature and Forest (INBO, Instituut voor Natuur- en Bosonderzoek). For the period 1997-1999 data from the mesohaline reach nearby Sieperdaschor were obtained from H. De Wilde. Table 2.1 gives an overview of the survey campaigns at the sites, illustrating differences of sampling effort between the zones.

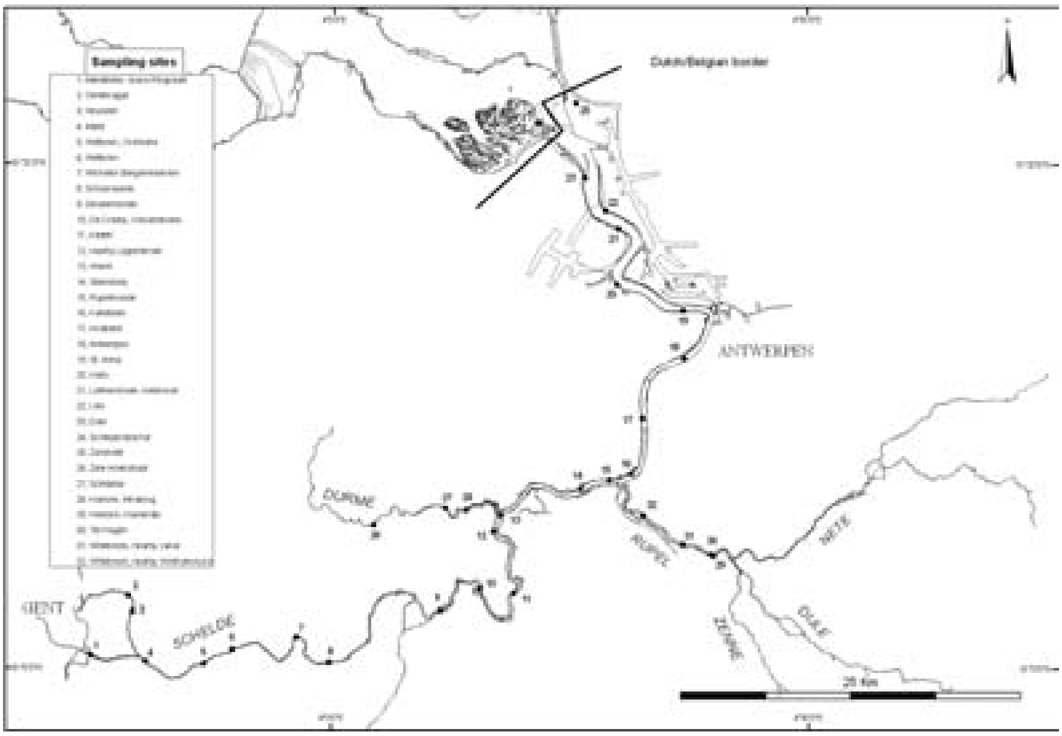

Figure 2.1: Map of the Zeeschelde Estuary with indication of the sampling sites.

Monthly oxygen concentrations in the different salinity zones were obtained from the VMM at Zandvliet (mesohaline), Steendorp (Oligohaline) and Kastel (Freshwater). For each year the annual average values were calculated using these data.

Table 2.1: An overview of the sites surveyed between 1991 (including Doel) and 2008. All sites were surveyed with fyke nets except the cooling circuit at Doel

River	Site (number in Fig. 2.1)	Period	Frequency	Number of samples
Schelde	Merelbeke (1)	2003	campaign	2
Schelde	Merelbeke – sluice Ringvaart (1)	2002	monthly	12
Schelde	Gentbrugge (2)	1997	campaign	1
Schelde	Heusden (3)	2002	monthly	11
Schelde	Melle (4)	1997	each season	4
Schelde	Melle (4)	2002	monthly	12
Schelde	Overbeke, Wetteren (5)	2007-2008	twice a year	4
Schelde	Wetteren (6)	2007	each season	4
Schelde	Uitbergen, Wichelen (7)	2008	twice a year	4
Schelde	Schoonaarde (8)	1997	campaign	2
Schelde	Dendermonde (9)	1997	each season	4

Table 2.1: Continued.

River	Site (number in Fig. 2.1)	Period	Frequency	Number of samples
Schelde	De Cramp, Vlassenbroek (10)	2007-2008	permanent	44
Schelde	Kastel (11)	2002-2007	twice a year	17
Schelde	Lippenbroek (12)	2006-2008	permanent	158
Schelde	Weert (13)	2007-2008	permanent	43
Schelde	Steendorp (Notelaar) (14)	2008	campaign	6
Schelde	Steendorp (14)	2002-2007	twice a year	14
Schelde	Steendorp (14)	1997	each season	4
Schelde	Steendorp (14)	1998	monthly	8
Schelde	Steendorp (14)	2001	monthly	5
Schelde	Rupelmonde (15)	2007-2008	permanent	62
Schelde	Kallebeek (16)	1997	once only	1
Schelde	Kruibeke (17)	1997	each season	4
Schelde	Antwerpen (18)	1997	each season	4
Schelde	Antwerpen (18)	1998	monthly	8
Schelde	Antwerpen (18)	2001	monthly	6
Schelde	Antwerpen (18)	2002-2007	twice a year	12
Schelde	Antwerpen (18)	2007-2008	permanent	398
Schelde	St. Anna (19)	2004-2005	permanent	304
Schelde	Kallo (20)	1995-1998	monthly	11
Schelde	Kallo (20)	2008	permanent	25
Schelde	Liefkenshoek, Ketenisse (21)	2007-2008	permanent	185
Schelde	Lillo (22)	1995	four times	4
Schelde	Doel (23)	1991-2008	monthly	170
Schelde	Sieperdaschor (24)	1997-1999	seasonally	9
Schelde	Zandvliet (25)	1995-2004	permanent	197
Schelde	Zandvliet (25)	2005-2007	twice a year	6
Durme	Zele (26)	2004-2008	yearly	5
Durme	Sombeke (27)	2004-2007	yearly	4
Durme	Hamme, Mirabridge (28)	2004-2008	yearly	5
Rupel	Heidonk, Hamerdijk (29)	2004-2008	yearly	5
Rupel	Heidonk, Hamerdijk (29)	2007-2008	weekly	56
Rupel	Ter Hagen (30)	2007-2008	weekly	29
Rupel	Willebroek, near canal (31)	2004-2008	yearly	5
Rupel	Willebroek, Wintham sluice (32)	2004-2008	yearly	5

2.3 Sampling gear

2.3.1 Fyke nets

At each location one or two ‘paired-fyke’ nets (type 120/80) were deployed near the low-tide mark for at least two tidal cycles (24 h) and emptied the next day (Fig. 2.2). Each paired fyke consists of two fyke ends of 2.2 m long, linked by an 11 m leader net. The largest hoop measures 0.8 m and has an oblate basis of 1.2 m to make sure that the net stays upright. Fishes are directed by the leader into the fyke and collected in the last chamber with a mesh size of 8 mm. Some sites were surveyed during two successive days.

Figure 2.2: Example of a 'paired fyke' net. The net is deployed at the low-tide mark and emptied 24 hours later.

2.3.2 Intake screens at the power station Doel

The Doel data set (1991-2008) was used to complete the species list of the mesohaline zone (presence/absence). The cooling water is drawn through a multiple intake system (25 m^3s^{-1}) at 2 m above the bottom of the estuary and filtered by two vertical travelling screens with a mesh size of 4 mm. The screens prevent larger organisms and debris to obstruct the condensers (Maes *et al.*, 2004b). We took three successive hours to complete one fish survey.

2.4 Data analysis

The numbers of individuals caught with fyke nets were transformed to catch per unit effort (CPUE); i.e. the total number of individuals is divided by the number of fykes used and the number of days. CPUE data were pooled per month, season and year, log (x+1) transformed and analysed with a non-metric multidimensional scaling (NMDS) ordination to examine the spatial organization of the fish assemblage. Only data from the common month and year in all three zones were used for the analysis. For the period 1997-2008 this corresponds per zone with 28 pooled CPUE data. Dissimilarity matrices were calculated from log (x+1) transformed fish abundance data, using Bray-Curtis distances. The NMDS ordination was created using random starting configurations and iterated until solutions converged. The vegan package in R 2.6.2 was used for the analysis (Oksanen *et al*., 2006, R Development Core Team). To reduce the effect of rare species only the 15 species with the highest abundance in one of the salinity zones were included for analysis (i.e. 22 species). To test the spatial differences in fish assemblages a Discriminant Analysis (DA) was applied to the same data. The estimated distinctiveness of fish assemblages was calculated using Wilk's Lambda criterion (λ) (Castillo-Rivera *et al*., 2002). This value ranges from 1 (similar groups) to 0 (different groups). A PCA with spring and autumn catches assessed the species contribution within each salinity zone. Spring and autumn were chosen as yearly data for these seasons were available for the periods considered.

3 Results

3.1 Dissolved oxygen

Minimum, maximum and average annual values, for dissolved oxygen in the different salinity zones for the years 1997-2008 are presented in table 2.2. They indicate in general and for all zones an increase in dissolved oxygen over this period. The increase in average annual dissolved oxygen concentration during the observation period is highest for the freshwater zone. The lowest minimum and average values are recorded in the oligohaline zone.

Table 2.2: Annual minimum (min), maximum (max) and average dissolved oxygen values (mg l^{-1}) for the different zones in the tidal Zeeschelde between 1997 and 2008 (VMM data).

	Freshwater			**Oligohaline**			**Mesohaline**		
year	**min**	**average**	**max**	**min**	**average**	**max**	**min**	**average**	**max**
1997	1.4	3.3	7.4	1.0	2.2	6.0	5.6	5.6	7.1
1998	1.7	4.6	9.2	1.7	3.4	8.1	4.8	6.8	9.1
1999	0.9	2.8	11.7	0.8	2.1	4.6	3.1	6.0	6.7
2000	1.6	2.3	3.3	2.6	4.3	8.2	3.1	5.6	10.0
2001	1.2	3.6	6.6	1.9	4.2	5.1	3.7	5.3	6.0
2002	4.0	6.1	8.5	1.7	4.0	7.2	4.8	6.4	10.1
2003	4.1	5.1	8.0	3.2	4.4	5.5	5.3	7.7	11.8
2004	6.2	5.5	13.4	2.5	4.0	6.3	2.2	6.5	10.2
2005	5.3	5.8	9.4	1.2	2.5	9.4	1.9	3.9	9.8
2006	1.7	5.7	10.2	0.9	3.0	6.7	2.6	7.9	10.8
2007	4.5	7.0	9.0	1.2	5.2	8.5	1.9	7.0	8.6
2008	4.3	7.8	10.2	2.5	6.1	9.2	5.1	7.1	8.9

As an illustration the average seasonal DO values in the oligohaline zone for the period 1997-2008 are given in figure 2.3. Seasons were defined using the meteorological approach. An increase in DO values is clear. The lowest DO values are recorded in summer and autumn, the highest in winter.

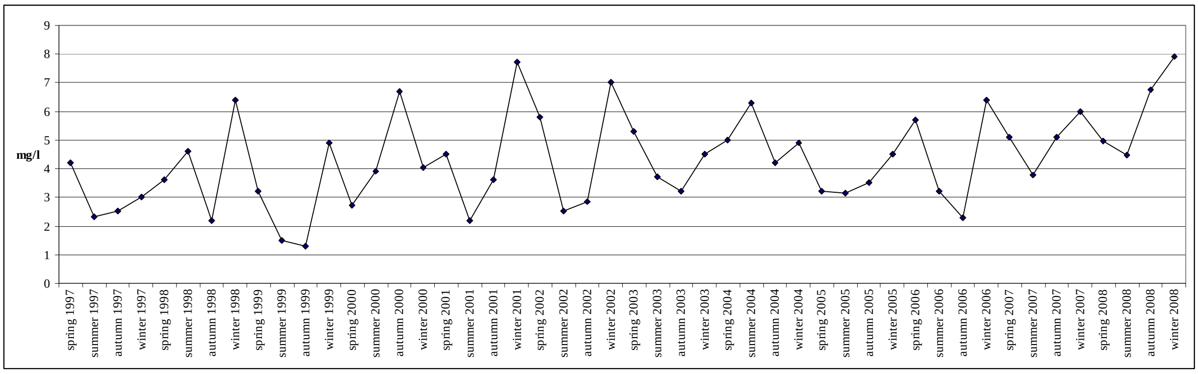

Figure 2.3: Average seasonal DO values (mg l^{-1}) in the oligohaline zone between 1997 and 2008.

3.2 Fish inventory

3.2.1 Fyke net catches

In total 66 species were caught between 1995 and 2008 (Table A, annex). Within the mesohaline zone 59 species were caught on 741 fishing occasions (day catches) between 1995 and 2008. In the oligohaline 43 species were collected on 632 fishing occasions between 1997 and 2008. In the freshwater zone 33 species were caught on 336 fishing occasions between 1997 and 2008.

3.2.1.1 Zone differences

Between 1997 and 2008, 28 fishing occasions took place in the same month in all the zones (Table 2.3). During these surveys, in total 59 species were caught. Forty species were selected for a PCA analysis with log (x+1) transformed number of individuals caught.

Table 2.3: Common fishing occasion in the three salinity zones.

Year	1997			2002		2003	2004			2005		2006	2007									2008						
Month	3	9	12	3	9	9	3	4	9	3	4	9	3	4	5	6	7	8	10	11	12	1	4	5	6	7	8	9

The NMDS ordination shows a clear distinction between the different zones (Fig. 2.4). The catches in the different salinity zones form three distinct groups. For the freshwater zone (dots) summer and autumn catches form two separate groups. The spring catches are scattered alongside these two groups. In the oligohaline zone (triangles) summer and spring catches form two separate groups. The winter and autumn catches are situated along these groups. In the mesohaline zone (squares) seasonal catches are spread over the plot with spring and summer catches forming separate groups. Combinations with other PCA factors give similar patterns.

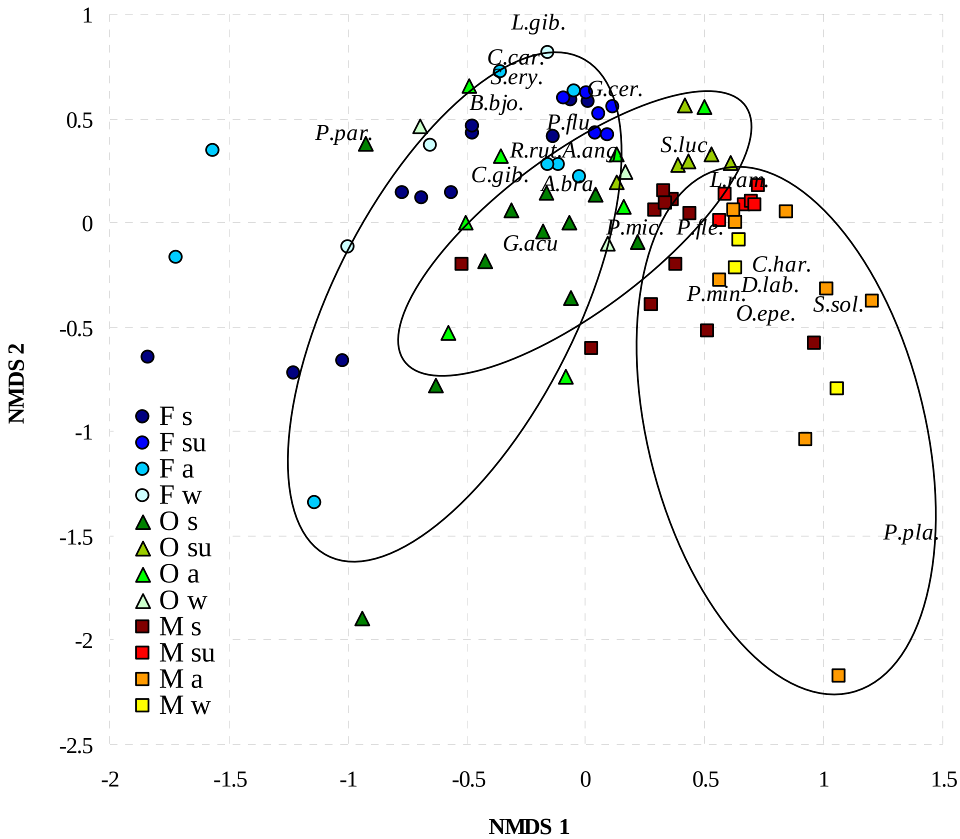

Figure 2.4: Non-metric multidimensial scaling (NMS) ordination of fish abundance data for the different salinity zones of the Zeeschelde estuary between 1997 and 2008 (n=84) (F: freshwater ●; O: oligohaline ▲; M: mesohaline ■; s: spring; su: summer; a: autumn; w: winter).

The NMDS differentiates clearly the three different zones. Species as plaice (*Pleuronectes platessa*), herring (*Clupea harengus*), seabass (*Dicentrarchus labrax*) and smelt (*Osmerus eperlanus*) are typical for the mesohaline zone. In the oligohaline zone the presence of common goby (*Pomatoschistus microps*) and herring is responsible for the differentiation from the freshwater zone, while the presence of freshwater species is responsible for the separation from the mesohaline zone. Some catches in the different zones are closely located due to species with a comparable abundance in all zones e.g. three-spined stickleback (*Gasterosteus aculeatus*), white bream (*Blicca bjoerkna*), Prussian carp (*Carassius gibelio*), roach (*Rutilus rutilus*) and eel (*Anguilla anguilla*).

The DA on the log transformed abundance of species with zone as grouping variable revealed a significant difference between the zones: $\lambda = 0.029$, $p<0.0001$, with more than 95% correctly classified cases.

3.2.1.2 Freshwater zone

In the freshwater zone 33 species were collected between 1997 and 2008 (Table A, annex). We grouped fish into guilds or functional groups (Table A, annex) according to Franco *et al.* (2008) to facilitate comparison between the salinity zones. Freshwater species comprised 69.7% of the total species richness and contributed 78.9% to the total number of individuals recorded (Fig. 2.5). The marine migrants contributed only 0.04% to the total number caught and were only recorded during 2008. Diadromous species make up 18.2% of the species richness and 19.3% of the individuals recorded. In 1997 only a few diadromous specimens were caught but this guild was well represented from 2005 onwards. Two estuarine species (common goby *Pomatoschistus microps* and sand goby *P. minutus*) have been encountered yearly in the freshwater zone since 2006. They were already occasionally recorded in 1997 and 2004. Estuarine species contributed 1.7% to the total number of individuals caught. The annual guild contribution (relative percentage) is given in figure 2.5.

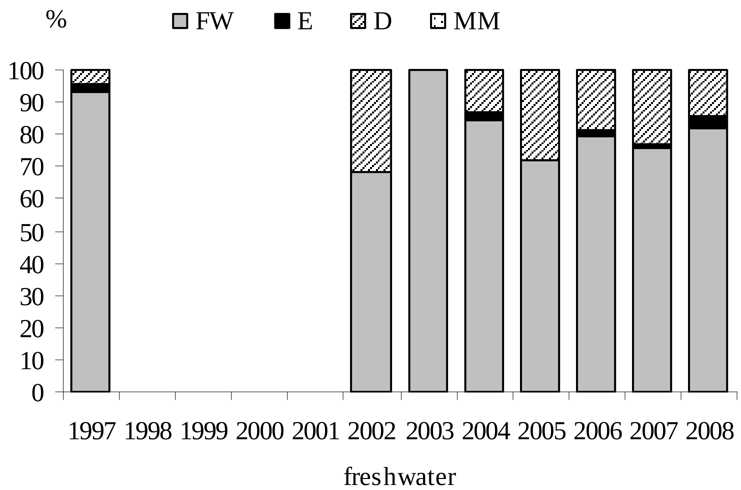

Figure 2.5: The annual contribution of the estuarine use functional guilds in the freshwater zone of the Zeeschelde between 1997 and 2008. (F: freshwater species; E: estuarine residents; D: diadromous species; MM: marine migrants).

A PCA with annual spring and autumn catch data (CPUE, log (x+1) transformed) groups all the results (except 2002) obtained before 2007 (Fig. 2.6). Factor 1 explains 39.8% and the second factor 17.9% of the variance.

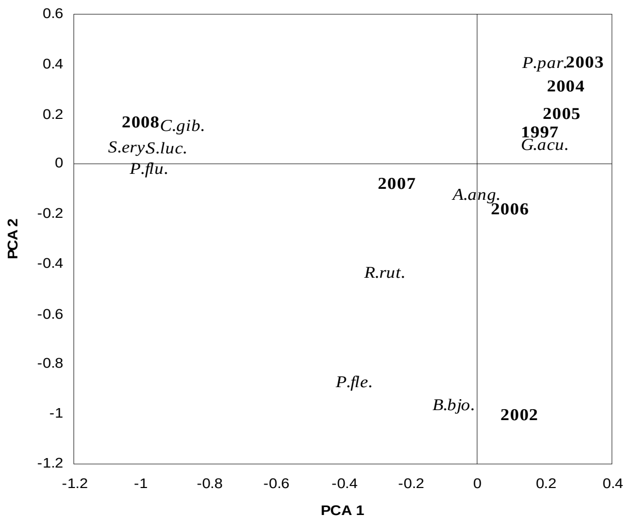

Figure 2.6: Scatterplot of factor loadings categorized by year obtained by PCA with log (x+1) transformed number of individuals caught (CPUE) and factor loadings of the ten most abundant species in the freshwater zone of the Zeeschelde estuary between 1997 and 2008 (spring and autumn catches, n=26). Abbreviations see Table A in annex.

The gradual increase in number of individuals caught separates the 2007 and 2008 catches from the previous years. The 2002 catches are separated because of the presence of white bream (*Blicca bjoerkna*, factor loadings -0.07;-0.95) and flounder (-0.36;-0.86). The year 2002 was a very wet one (Maris *et al*., 2008). The presence of chub separates 2007 (*Leuciscus cephalus*, -0.22;-0.06). The catch results in 2008 are similar to those in 2007 but are separated in the scatterplot mainly because of pike-perch (-0.94;0.07), rudd (*Scardinius erythrophthalmus*, -0.98;0.04), Prussian carp (*Carrasius gibelio*, -0.95;0.14) and perch (*Perca fluviatilis*, -0.95;-0.01).

The CPUE log (x+1) transformed data in the freshwater zone of the Zeeschelde are represented in figure 2.7. All survey data are here included as it is the purpose to show the catch results over the years. These data were not used for any further analysis. The figure shows that since 2004 an increasing number of individuals and species was caught and that roach is the most abundantly recorded species in the freshwater zone. An increasing number of white bream, rudd, pike-perch and flounder were caught since 2005.

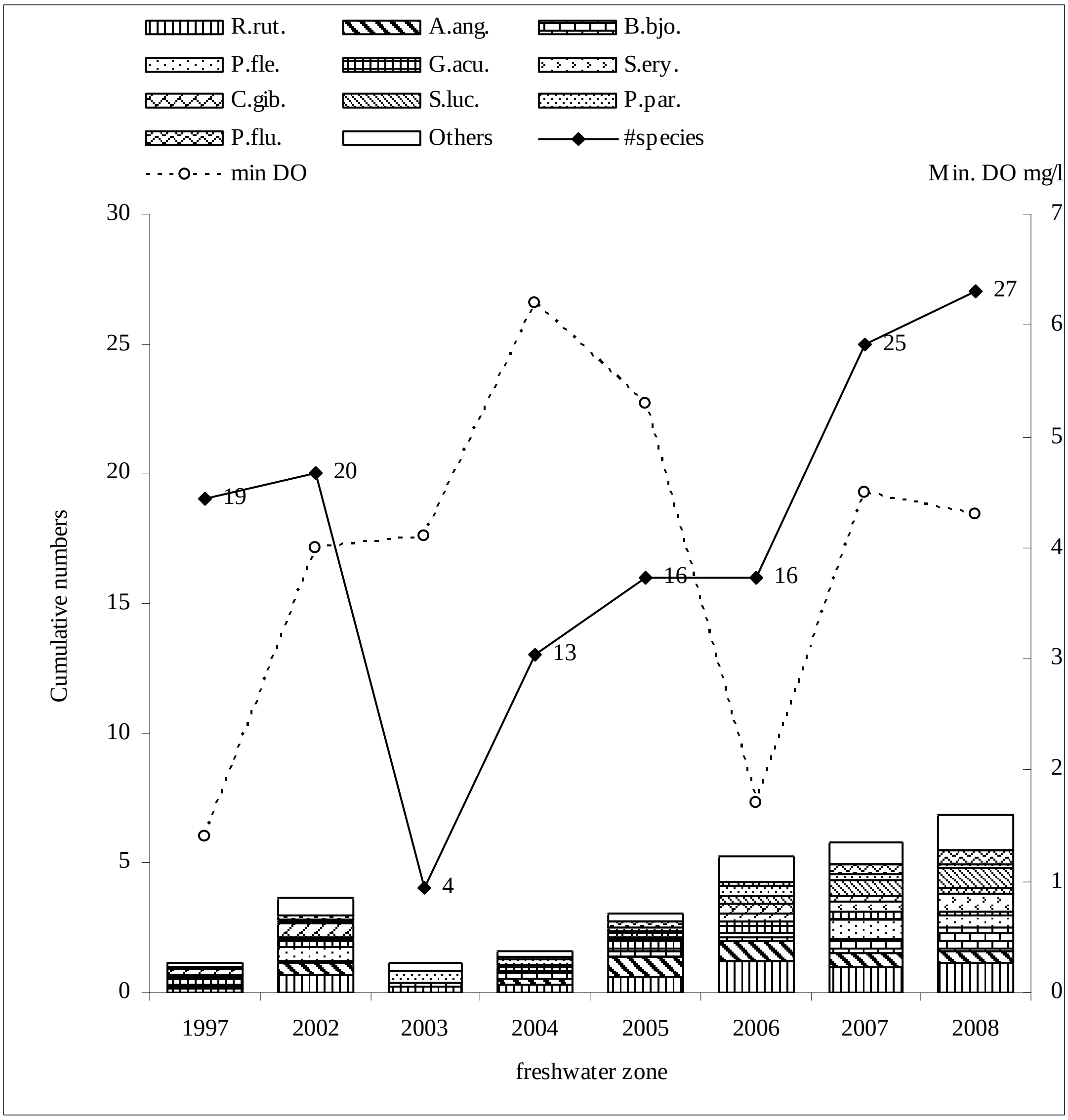

Figure 2.7: Species richness (figures on full line) and the catch per unit effort (cumulative numbers, log (x+1) transformed) for fish species caught in the freshwater zone of the Zeeschelde between 1997 and 2008 (abbreviations see Table A). Only the on average 10 most abundant species are indicated with a specific pattern. Dotted lines connect the minimum recorded DO for a particular year.

3.2.1.3 Oligohaline zone

In the oligohaline zone 43 species were caught between 1997 and 2008 (Table A). Some 53.5% are freshwater species, contributing 62.9% to the total abundance. Nine marine migrants contribute 5.3% to the total abundance, while the contribution to the species richness is 20.9%. Some of the marine migrants, e.g. herring (*Clupea harengus*), were collected yearly but the highest numbers of marine migrants were caught in 2007 and 2008. Diadromous species make up 19.9% of the species and 14% of the individuals caught. Of this guild only eel and flounder were caught in all years, the other diadromous species were caught regularly since 2007. The two estuarine species, common and sand goby, were recorded in the oligohaline zone since 2003 and 1997 respectively. Since 2007 the greater pipefish (*Syngnathus acus*) was also caught and contributes together with the two gobies, 11.9% to the total abundance. Occasionally marine stragglers venture in the oligohaline zone, e.g. Lozano's goby (*Pomatoschistus lozanoi*) and the lesser weever (*Echiichthys vipera*). The annual guild presence (relative percentage) is given in figure 2.8.

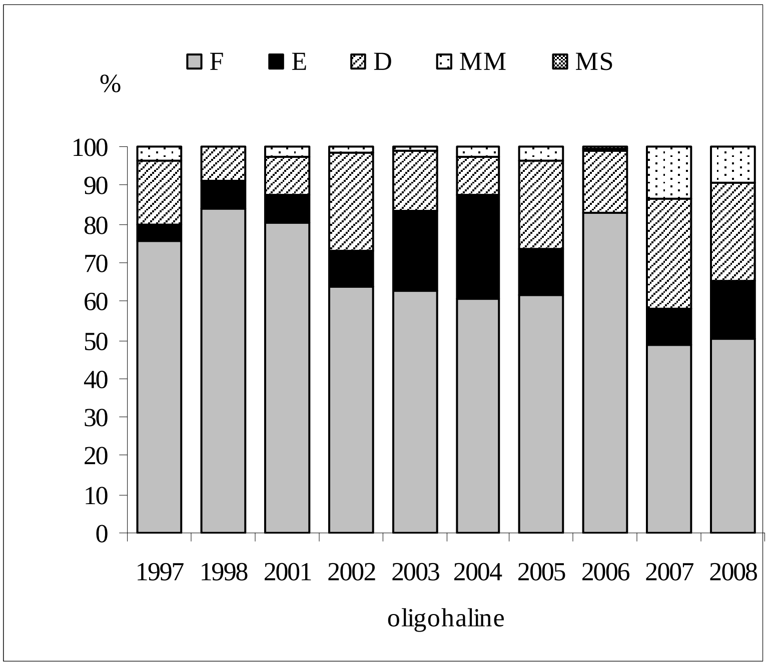

Figure 2.8: The annual contribution of the estuarine use functional guilds in the oligohaline zone of the Zeeschelde between 1997 and 2008. (F: freshwater species; E: estuarine residents; D: diadromous species; MM: marine migrants; MS: marine stragglers).

Figure 2.9 shows the scatterplot of a PCA with annual catch data (spring and autumn CPUE, log (x+1) transformed). The figure shows one group of most annual catches, but the catch results of 2007 and 2008 are separated from them and from each other. Factor 1 explains 41.2% and the second factor 17.0% of the variance.

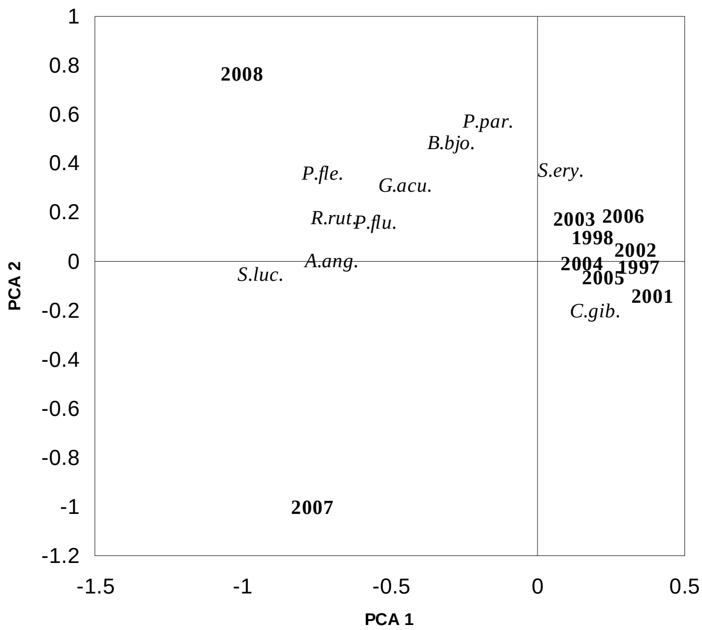

Figure 2.9: Scatterplot of factor loadings categorized by year obtained by PCA with log (x+1) transformed number of individuals caught (CPUE) and factor loadings of the ten most abundant species in the oligohaline zone of the Zeeschelde estuary between 1997 and 2008 (spring and autumn catches, n=26). Abbreviations see Table A in annex.

In 2007 and 2008 more species and individuals were caught. They are separated from the other catches mainly by the presence of smelt (*Osmerus eperlanus*, factor loading=-0.87;0.34), seabass (*Dicentrarchus labrax*, -0.82;-0.55), pike-perch (*Sander lucioperca*, -0.93;-0.04) and herring (*Clupea harengus*, -0.95;0.12). The difference between these two years is the result of differences in numbers caught and because whiting (*Merlangius merlangus*), sole (*Solea solea*), lesser weever (*Echiichthys vipera*) and pouting (*Trisopterus luscus*) were caught in 2007 and not in 2008. In 2008 cod (*Gadus morhua*) was caught, but absent in the 2007 catches.

The CPUE log (x+1) transformed data in the oligohaline zone of the Zeeschelde shows a remarkable increase in number of individuals in 2007 and 2008 (Fig. 2.10). Roach is again the most frequently caught species. The pike-perch and rudd catches increase since 2006. Again the purpose is to illustrate species present and therefore all survey data were used. Species richness increased in 2007 but decreased again in 2008. Still compared to previous years species richness is higher in 2007 and 2008.

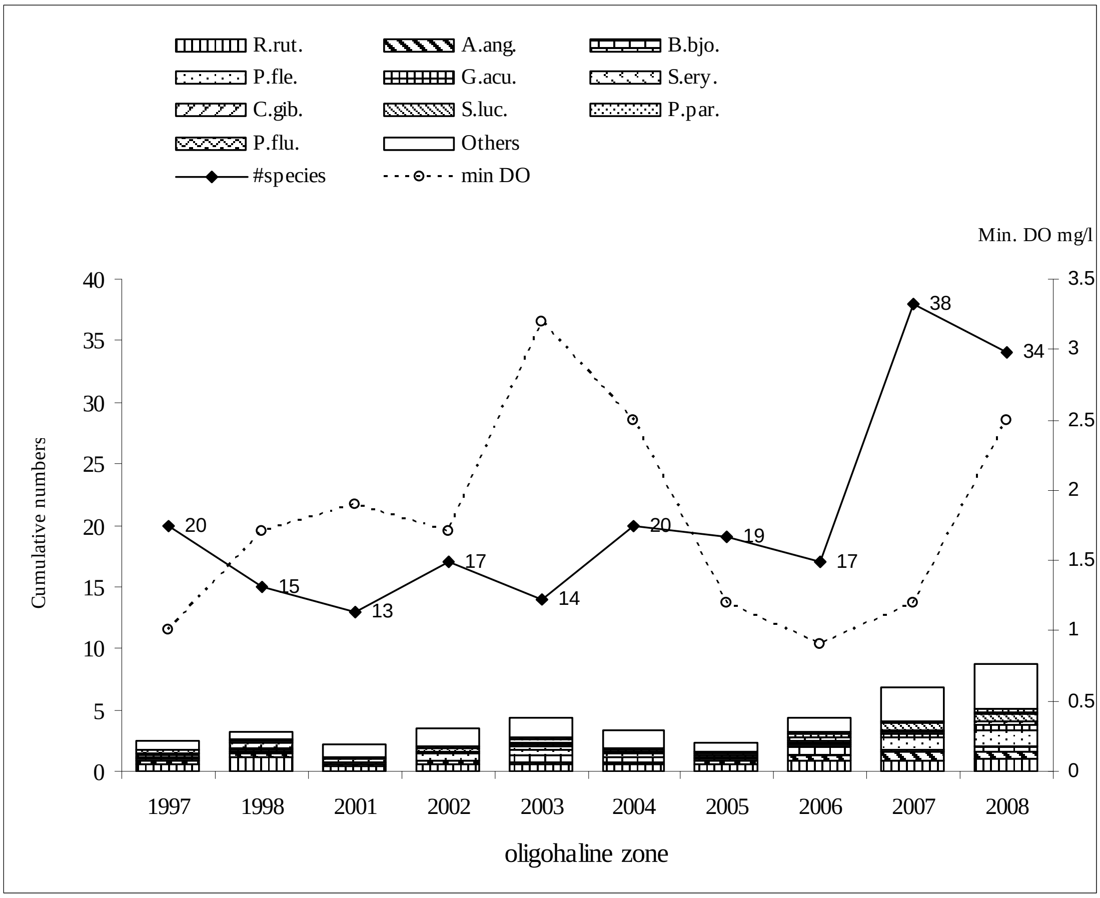

Figure 2.10: Species richness (figures on full line) and the catch per unit effort (cumulative numbers, log (x+1) transformed) for fish species caught in the oligohaline zone of the Zeeschelde between 1997 and 2008 (abbreviations see Table A). Only the on average 10 most abundant species are indicated with a specific pattern. Dotted lines connect the minimum recorded DO for a particular year.

3.2.1.4 Mesohaline zone

In the mesohaline zone 59 species were collected between 1995 and 2008 (Table A). Of these 33.3% were freshwater species, contributing 19.3% to the total abundance. Marine migrants were well represented, comprising 26.6% of the species and contributing 44.5% to the total number of individuals caught. The marine migrants occurred in all annual catches. Some 15% of the species were diadromous species, contributing 27% of the total number of individuals caught. Diadromous species were always present in the annual catches. Ten estuarine species (16.6% of total species number) were sampled, contributing 8.8% to the total individuals caught. The marine stragglers contributed 8.3% to the species and 0.2% to the total number of individuals caught. The annual guild distribution (relative percentage) is shown in figure 2.11.

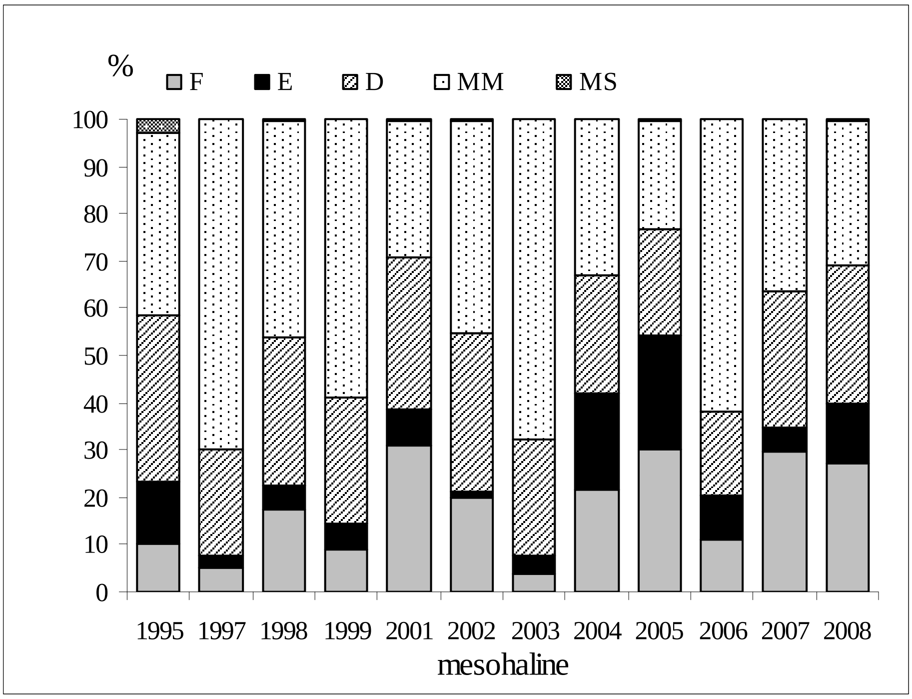

Figure 2.11: The annual contribution of the estuarine use functional guilds in the mesohaline zone of the Zeeschelde between 1995 and 2008. (F: freshwater species; E: estuarine residents; D: diadromous species; MM: marine migrants; MS: marine stragglers).

The scatterplot from a PCA with annual data (spring and autumn CPUE log (x+1) transformed) shows a more dispersed pattern than the ones observed in the freshwater and oligohaline zones (Fig. 2.12). Factor 1 explains 18.8% and the second factor 16.0% of the variance. The scatterplot indicates that the catch results obtained in 2001 and 2007 are separated from the other results. Catches in 2008 are less distinct than in the other salinity zones due to a decrease in numbers of individuals caught (Fig. 2.13) and in species richness. In 2001 thinlip mullet (*Liza ramado*, factor loading= 0.65;0.48), tub gurnard (*Chelidonichthys lucernus*, 0.35;0.72) and brill (*Scophthalmus rhombus*, 0.53;0.76) are responsible for the separation. The catches of lumpsucker (*Cyclopterus lumpus*, 0.47;-0.60) and turbot (*Psetta maxima*, 0.47;-0.60) contributed to the separation of the 2007 catches. These species were not or less caught in the other years. The more dispersed general pattern reflects the higher annual catch variations in the mesohaline zone compared with the other zones.

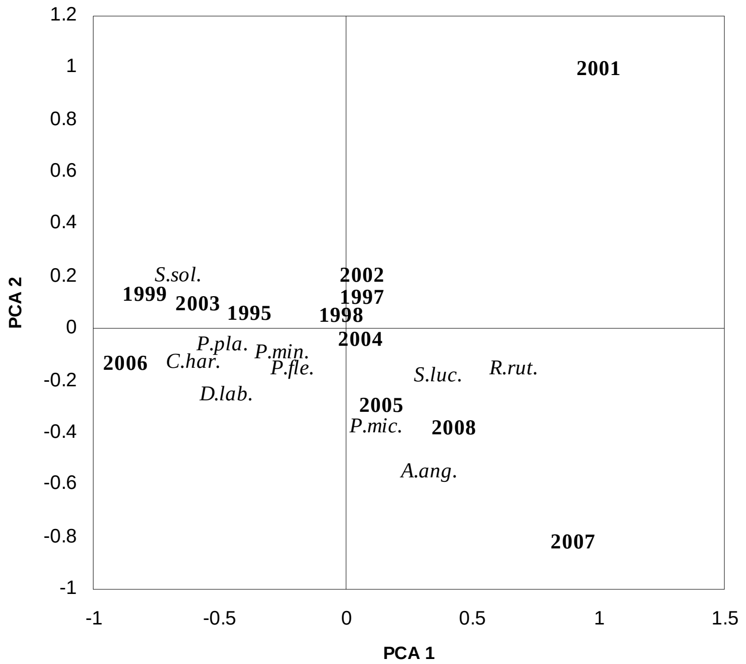

Figure 2.12: Scatterplot of factor loadings categorized by zone obtained by PCA with log (x+1) transformed number of individuals caught (CPUE) and factor loadings of the ten most abundant species in the mesohaline zones of the Zeeschelde estuary between 1995 and 2008 (spring and autumn catches n=48). Abbreviations see Table A in annex.

The catch per unit effort (log (x+1) transformed CPUE data) in the mesohaline zone of the Zeeschelde is given in figure 2.13. The figure shows an increase in CPUE till 2001 followed by lower catches until 2005. In 2006 and 2007 the annual CPUE was high, but in 2008 again a decrease was observed. All survey data were used to illustrate species presence.

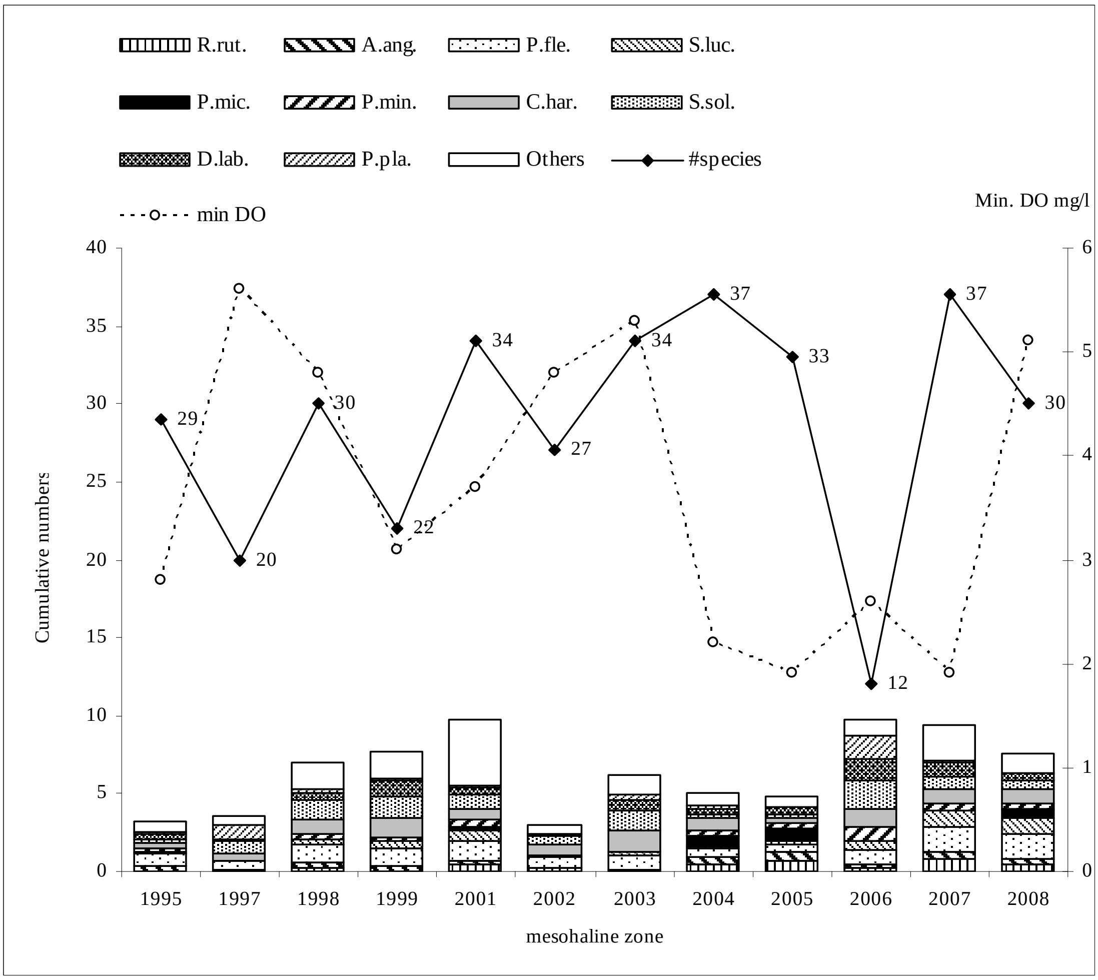

Figure 2.13: Species richness (figures on full line) and the catch per unit effort (cumulative numbers, log (x+1) transformed) for fish species caught in the mesohaline zone of the Zeeschelde between 1995 and 2008 (abbreviations see Table A). Only the on average 10 most abundant species are indicated with a specific pattern. Dotted lines connect the minimum recorded DO for a particular year.

3.2.2 Doel

At the intake screens of the power station at Doel 66 species were collected between 1991 and 2008 of which snake pipefish (*Entelurus aequoreus* (Linnaeus, 1758)), solenette (*Buglossidium luteum* (Risso, 1810)), painted goby (*Pomatoschistus pictus* (Malm, 1845)), dragonet (*Callionymus lyra* (Linnaeus, 1758)) and great sandeel (*Hyperoplus lanceolatus* (Le Sauvage, 1824)) were not caught with fykes. This brings the total of fish species caught in the Zeeschelde estuary to 71.

4 Discussion

4.1 Dissolved oxygen

The main abiotic factor influencing the presence of fish within an estuary is the dissolved oxygen concentration (DO) (Maes *et al*., 1998a; Araújo *et al*., 2000; Turnpenny *et al*., 2006; Maes *et al*., 2007, 2008). Not only does oxygen have an impact on the presence of fish, it is also the driver for many water quality improvements. DO within the Zeeschelde has increased continuously since 1996 (Table 2.2, Maris *et al*., 2008). The changes observed in the fish assemblages in the freshwater and oligohaline zones become evident from 2007 onwards (Figs. 2.7 and 2.10). The activation of the water purification plant (Brussels North, March 2007) increased abruptly the oxygen concentration of the Zenne river, a tributary of the Rupel, and fish started to inhabit this river (Van Thuyne & Breine, 2008). A similar improvement was observed in the Thames estuary where the return of fish species was a striking feature linked with the recovery from pollution (Wheeler, 1969, 1979; Andrews & Rickard, 1980; Attrill, 1998). The average oxygen concentration in the oligohaline zone remained below 5 mg l^{-1} between 1997 and 2006. Although in 2007 an improvement was recorded, still 54.6% of the OMES records were below this norm (Maris *et al.*, 2008). The mesohaline zone has a higher oxygen concentration due to oxygen rich water coming in from the Westerschelde. This could explain why, compared to the other zones, in this zone no significant increase in fish catches was observed between 1995 and 2008. In the late 1970s temporal anoxia was common in the upstream part of the Zeeschelde (Soetaert *et al.*, 2006). In the freshwater part of the Zeeschelde an improvement of oxygen concentration is noted between 1998 and 2002 and is due to a higher discharge (wet years) and a higher primary production during the summer months (Maris *et al*., 2008). Between 2002 and 2007 a decrease in biological oxygen demand is noted in the freshwater part (Maris *et al*., 2008). Although the freshwater zone has for the period 2007-2008 high DO concentrations, even in summer, the norm of 5 mg l^{-1} (Vlarem II, 1995) is not always reached (Maris *et al.*, 2008).

4.2 Zone differences in fish assemblages

The difference in species richness and composition between the different zones is illustrated by the NMDS and the DA. A gradual change in guild distribution is observed with the salinity gradient (Figs. 2.5, 2.8 and 2.11). Though this shift is gradual, our results show the relevance to distinguish three salinity zones for fish assemblages within the Zeeschelde: mesohaline, oligohaline and freshwater. Van Damme *et al.* (1999) made also the distinction between a fresh water zone with long and short retention time but this was not reflected in the fish assemblages. The observed shift in distribution is consistent with other estuaries in the North Sea area. Thiel & Potter (2001) recorded a sequential change in the species composition from the most downstream site (high salinity) to the most upstream one (oligohaline) in the Elbe estuary. Selleslagh *et al*. (2009) reported a variable catch composition between intermediate and upper stations in the Somme estuary. For each salinity zone species with a clear seasonal pattern are further highlighted.

4.2.1 Freshwater zone

As expected the fish assemblages in the freshwater zone of the Zeeschelde are dominated by freshwater species (68-100%), corresponding with observations in tidal freshwater along the Atlantic coast of North America (Odum *et al.*, 1988) and in a freshwater estuary in Estonia (Vetemaa *et al.*, 2006). Freshwater individuals contributed 82.7% to the total catch between 1997 and 2008. The tidal freshwater zone is essentially a habitat for freshwater and diadromous species. An essential fish habitat consists of both the water column and underlying surface of a particular area. It contains all habitat characteristics essential to the long-term survival and health of particular fishes. Although this zone is characterised by the presence of freshwater species, its fish community is different from non tidal freshwater rivers due to morphological characteristics, dynamics and its connection with the oligohaline zone. However, between 1997 and 2008 the number of estuarine species and marine migrants is limited to 3.5 and 0.2% respectively. In a highly polluted river like the Zeeschelde oxygen deficiency strongly affects the fish community structure. However, over the years a gradual improvement in species richness is observed. A significant and steady increase in species richness and number of individuals is noted since 2004, the worst year observed being 2003 (Fig. 2.7). In addition concordant with the water quality (DO) improvements, a shift in fish assemblage structure occurred. In 1997 resistant freshwater species such as three-spined stickleback, Prussian carp and roach were dominant in numbers. At present the most abundantly caught species are flounder, common goby, pike-perch, roach and white bream. Another indication of the water quality improvement is the presence of twaite shad, recorded in spring 2007. Other diadromous species observed since 2007 are smelt and thinlip mullet. These results were predicted by Maes *et al*. (2007, 2008). Since summer 2007 herring and seabass (marine migrants) frequent this zone with abundance peaks in summer. For some species a seasonal pattern in frequency of occurrence and abundance can be distinguished. Ide (*Leuciscus idus*), Wells catfish (*Siluris glanis*), smelt, thinlip mullet, rudd, eel and pike-perch show a peak in summer. Some species such as lampreys are underestimated because of the low catch efficiency of fykes for this particular group. Concerning trophic level, omnivorous species such as roach, rudd, Prussian carp and eel are dominant in numbers. This discrepancy is an indication that although the water quality improved the habitat quality is still not optimal (Manolakos *et al*., 2007).

4.2.2 Oligohaline zone

This zone is characterised by a return of fish due to a continuous amelioration of the water quality (DO; Maris *et al.*, 2008). Since 2007 species richness is higher than in the freshwater zone and estuarine species and marine migrants have increased their importance which corresponds with previous research in oligohaline waters (e.g. Rozas & Hackney, 1983). Between 1997 and 2008 the number of estuarine individuals contributed 11.8%, diadromous 19.8% and the marine migrants 5.3% to the total catch. During this period freshwater individuals contributed 62.9% to the total catch. We therefore consider the oligohaline zone essentially a habitat for freshwater, estuarine, diadromous and marine migrant species. As already discussed, since the treatment of Brussels' waste water a higher oxygen concentration is observed in the oligohaline zone, enhancing the presence of fish (Fig. 2.10). During 1994-1995, dissolved oxygen was almost completely absent during the major part of the year and only during winter 12 fish species were caught at the cooling water inlets in Schelle, all of which were freshwater species except smelt and eel as diadromous species (Maes *et al.*, 1998a). Between 1995 and 2007 a gradual increase in species was recorded (Maes *et al.*, 1997; Peeters *et al.*, 1999; Ercken *et al.*, 2002; Maes *et al.*, 2004a; Maes *et al.*, 2005; Stevens *et al.*, 2006; Cuveliers *et al.*, 2007; Guelinckx *et al.*, 2008). Although in general no significant seasonal effects were observed some species in the oligohaline zone show nevertheless such effects. Sole is never recorded during winter, which corresponds with observations in other estuaries (Martinho *et al.*, 2007). Since 2008 twaite shad is occasionally recorded (Stevens *et al*., 2009). The anadrome lampreys are easily missed with fyke nets, but they are caught in summer at the lock-weir complex in Gent (Stevens *et al*., 2009). Upstream spawning grounds for twaite shad and lampreys are absent or inaccessible due to barriers (Stevens *et al*., 2009).

4.2.3 Mesohaline zone

This is the area where estuarine fish complete their life cycle and where fish from upstream, from the Westerschelde and from the North Sea seek refuge and food. Especially mudflats provide food for juveniles (Hiddink & Jager, 2002; Stevens, 2006). As such we find representatives from all estuarine use functional groups. Van Damme *et al.* (1994) presented a checklist of 23 fish species for the mesohaline zone of the Zeeschelde belonging to five ecological guilds: marine migrants (2), diadromous species (3), estuarine species (9), marine stragglers (3) and freshwater species (6). Compared to the checklists of de Selys-Longchamps (38 sp., 1842) and Poll (40 sp., 1945, 1947) more than 15 species had disappeared from the lower Zeeschelde in 1994. The anadromous fishes recorded by de Selys-Longchamps (1842): sea lamprey (*Petromyzon marinus*), allis shad (*Alosa alosa*), twaite shad (*Alosa fallax*), sturgeon (*Acipenser sturio*), shelly (*Coregonus lavaretus*) and the Atlantic salmon (*Salmo salar*) had all, except for twaite shad, already disappeared in 1945 (Poll, 1945). In 1991 the river lamprey was the only anadromous species persisting in the lower Zeeschelde (Van Damme *et al.*, 1994). The status of anadromous fish populations remained problematic until recently (Maes *et al.*, 1996, 1998a, 1999; Peeters *et al.*, 1998, 1999). Overall ten diadromous species have been recorded, some abundantly (eel, smelt, thinlip mullet) but others are rare (river lamprey and sea trout). Species richness shows year by year variations, but a dominance of marine migrants was always observed. The marine migrant individuals contributed 44.6% to the total catch between 1995 and 2008 with herring, flounder and sole as the most abundant species. The contribution of estuarine individuals to the total catch is 8.8%, 0.3% for marine stragglers, 27.1% for diadromous species and 19% for the freshwater species. We therefore consider the mesohaline zone as an essentially a habitat for most estuarine species, diadromous species and marine migrants (Fig. 2.11). The vast majority of species recorded consisted of juveniles. When combining all our survey results (1995-2008) the guild distribution shows similarities with other European estuaries, e.g. the Elbe (Elliott & Dewailly, 1995) and the Gironde (Lobry *et al.*, 2003). Cabral *et al.* (2001) observed a dominance of marine migrants, marine stragglers and estuarine species in the polyhaline zone of the Tagus and Selleslagh *et al.* (2009) observed a dominance of marine migrants and estuarine species in three eastern English Channel macrotidal estuaries (Canche, Authie and Somme). The same authors found also a dominance of marine migrants and estuarine species in 15 other French estuaries, whereby the freshwater group showed the highest variation ranging from 0 to 37% of species richness. No overall seasonal effect was observed although for some species a seasonal pattern was present. Sprat and herring are known to be winter migrants (Maes *et al.*, 1998a) however, herring is now more abundant in autumn compared to a previous winter accent. A similar observation was made by Childs *et al.* (2008) for the estuarine spotted grunter (*Pomadasys commersonnii*) in South Africa. The gradual increase in densities of flounder could be an indication of global warming as described by Thiel *et al.* (2003). However, the seasonal pattern of this species is complex and not only influenced by temperature. There is an effect of inter-annual variations in recruitment (Thiel & Potter, 2001) and the availability and abundance of food can also disrupt a seasonal pattern. Observed seasonal patterns can be the result of behavioural responses to changes in predation risk and are probably linked to a size-related behaviour (Maes *et al.*, 1998a). The main predators in the mesohaline zone are freshwater species, e.g. pike-perch and perch. Other predators are rarely caught, e.g. juvenile seabass, twaite shad and smelt are occasionally passing through. Large numbers of species enter the estuary to avoid predation and remain there for a short or longer period depending on water quality and food availability. Turbidity may be a driving force for fish migration into the estuary as those fish are attracted by the plume in the sea (Maes *et al.*, 1998a). We embrace the hypothesis mentioned by several authors that although some species can be considered as estuarine dependent, a large number of the individuals concerned use the mesohaline zone of an estuary on a facultative or opportunistic basis (Power & Atrill, 2003; Maes *et al.*, 2004b; Guelinckx, 2008). Indeed several fish species show variable migration patterns which could be the result of habitat selection (Morris, 2003).

We are aware that the fish assemblages in the different salinity zones are also affected by physical habitat characteristics. Supralitoral zones (tidal marshes and flood systems) are most susceptible to human pressure. The loss of mudflats (dyke reinforcements) combined with dense ship transport and a very dynamic tide, enhance the erosion of tidal marshes (Van Braeckel *et al*., 2006). These mudflats and marshes are important for fish since they serve as feeding places and shelter for many species (Cattrijsse & Hampel, 2006; Stevens, 2006). Tidal marshes possess critical biological, chemical and ecological functions (Desmond *et al.*, 2000; Mathieson *et al.*, 2000; Stevens, 2006; Maris, 2008). In addition they contribute to flood defence by dissipating wave energy, thereby reducing erosion (Dixon *et al.*, 1998). McLusky *et al.* (1992) comment on the historic loss of inter-tidal habitat and saltmarshes and estimated that the fish population in the Forth estuary was reduced by 66% as a consequence of those losses. Primary and secondary production of inter-tidal salt marshes is playing a fundamental role in the feeding of 0-group individuals of different species (Lyndon *et al.*, 2002; Hampel *et al.*, 2003; Colclough *et al.*, 2005; Cattrijsse & Hampel, 2006; Stevens, 2006; Alcaraz, 2008). The inter-tidal creek habitat accommodates juvenile fishes during the day, while larger specimens visit the creek by night, resulting in a reduction of space and energy competition (Shenker & Dean, 1979). Colclough *et al.* (2005) demonstrated that the restored inter-tidal saltmarshes in the Thames and Blackwater estuaries were utilised extensively by juvenile fishes and species preferences for particular microhabitats were even observed. A decrease of habitat diversity in the freshwater zone is also reflected by impoverished fish diversity (e.g. Jansen *et al.*, 2000; Sindilariu *et al.*, 2006). There are clear differences in juvenile responses to environmental heterogeneity (Grenouillet *et al.*, 2000). This has a direct effect on species richness (Belliard *et al.*, 1999; Schiemer, 2000) and may affect the functional structure of the fish community. Juvenile fish will benefit from structured habitats and avoid substrates lacking any suitable shelter. The creation of shallow inter-tidal habitats will enhance the restoration of the fish community as these new habitats can be used as nursery and spawning places, shelter and resting areas, as well as feeding grounds (Simoens *et al.*, 2007). An improved creek system can also increase the residence time of fry, thereby enlarging the tidal marsh capacity. Pas *et al.* (1998) observed that Tielrodebroek, a flood controlling area of about 90 ha nearby the mouth of the River Durme, functions as a spawning and nursery area for a number of freshwater species. A nursery is described as a habitat that contributes proportionally more to the adult population, than the average of other habitats in which the juveniles occur (Beck *et al.*, 2001, 2003). These habitat needs are further elaborated in chapter 4.

5 Conclusions

The fish richness increased over the years 1991 to 2007 in the different salinity zones of the Zeeschelde. A similar observation has been recorded in many of the industrialised countries because of restoration and conservation efforts (Lotze *et al.*, 2006). The gradual increase in oxygen concentration in the different salinity zones of the Zeeschelde estuary has a positive impact on the species richness and confirms the model developed by Maes *et al*. (2007, 2008). A longitudinal shift in fish assemblages, numbers and species richness is mainly explained by the salinity gradient. This allowed us to define estuarine zones for different estuarine fish guilds. However, present fish communities do not reflect the assemblages recorded a century ago (see Chapter 3). The estuary and its tidal tributaries have been heavily influenced by anthropogenic pressures such as land claim, harbour expansion, dredging activities, embankments and urbanisation (Van Braeckel *et al*., 2006). The restoration of a natural sustainable fish assemblage will be enhanced by the creation of floodplains as spawning and nursery areas. Protection of the tidal marshes in all zones should be implemented in order to reduce further loss of habitat. Seasonal patterns are complex which is due in part to a suite of opportunistic behaviour and partly because of external natural variation and human impacts. The most abundant species in the estuary are tolerant species that support poor water quality. Some species are restricted to one zone while others frequent the whole estuary. Flounder and eel are the only diadromous species found in all zones. Freshwater eurytopic species with a high tolerance to harsh conditions are also present in all surveyed zones. The results presented add to the information needed to understand estuarine dependence. It is therefore essential to continue to monitor the fish assemblages in the estuary, as only long-term surveys will help to understand the influence of episodic and long-term events (Able, 2005). Such monitoring will allow a more detailed analysis concerning the environmental and biotic relations within the different zones of the estuary and the factors which in a positive or negative way impact its functioning.

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Chapter 3

A reference list of fish species for a heavily modified transitional water as defined by the Water Framework Directive: the Zeeschelde estuary (Belgium)

Jan Breine, Maarten Stevens, Erika Van den Bergh & Joachim Maes

Abstract

A crucial step in the development of a fish-based index for the ecological assessment of water bodies as provided by the European Water Framework Directive is the development of a fish reference. This reference consists of a fish assemblage present in pristine water bodies of the same category. Based on historically reported fish survey data of the Zeeschelde estuary and its tributaries under tidal influence (Belgium), presence/absence reference lists were compiled for different salinity zones. These historical lists were then adjusted using information from recent catches. Inclusion of fish species in the reference lists depended on their natural geographical distribution and ecological demands. Fish species are attributed to guilds (functional groups) and therefore these reference lists contain guild specific information for the different zones within the estuary and its tidal tributaries. The reference corresponds with an ecological status that is referred to as Good or Maximal Ecological Potential (GEP/MEP).

Keywords: ecological potential, estuary, fish, reference list, Zeeschelde, Water Framework Directive, transitional water

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1 Introduction

All transitional waters in Flanders have been identified as heavily modified water bodies as their nature has changed fundamentally as a result of physical anthropogenic alterations. According to Article 4(3) of the European Water Framework Directive (WFD) the principal environmental objective for heavily modified water bodies (HMWB) and artificial water bodies is to obtain a “good ecological potential” (GEP) and “good surface water chemical status” instead of a “good ecological status” as required for natural systems. Similarly, the reference situation in HMWB is referred to as “maximal ecological potential” (MEP) instead of a “pristine status” (EU Water Framework Directive, 2000). According to WFD the MEP biological conditions should reflect, as far as possible, the biological conditions associated with the closest comparable natural pristine water body, given the MEP hydromorphological and associated physico-chemical conditions. Borja and Elliott (2007) consider the MEP as the reference conditions for HMWB. For a HMWB to be classified as attaining GEP status there must be no more than slight changes in the values of the relevant biological quality elements as compared to their values at MEP. The biological potential can be defined once the hydromorphological and physical chemical potentials are described. The different paths of the decision procedure are illustrated in figure 3.1.

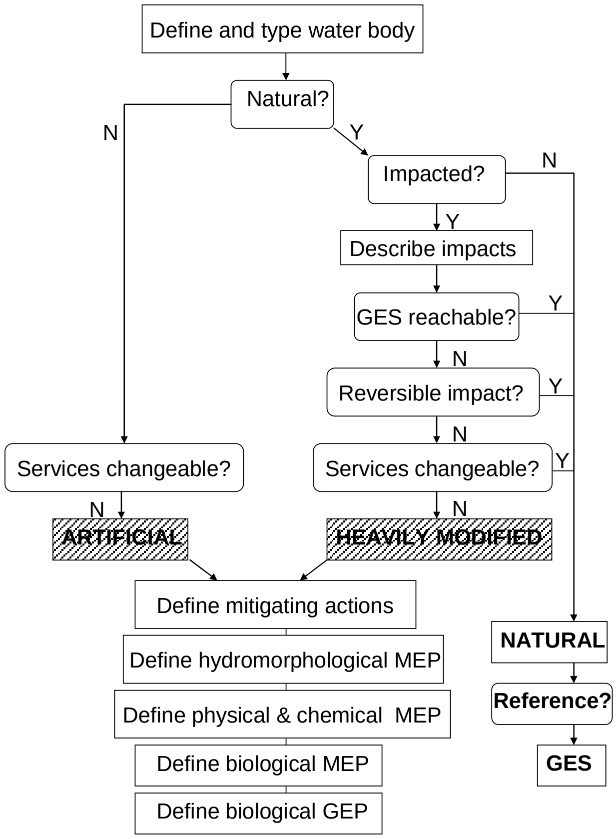

Figure 3.1: Flow diagram: guidelines to describe MEP/GEP adapted from a report of the Dutch Ministry of Transport, Public Works and Water Management (RIZA, 2006). MEP: Maximum Ecological Potential, GEP: Good Ecological Potential and GES: Good Ecological Status.

During an international workshop on the WFD and hydromorphology held in Prague 2005 it was concluded that these biological MEP/GEP conditions can also be defined from the actual status (Kampa & Kranz, 2005). A key difference in this approach is that the GEP is derived directly from the effect of mitigation measures and not indirectly from the specification and prediction of biological quality elements at MEP (Kampa & Laaser, 2009). For the benthos in the Westerschelde, the part of the Schelde estuary that is situated in The Netherlands, Escaravage *et al*. (2004) suggest that when a reference based on historically pristine conditions is absent, the maximum ecological potential has to be based on knowledge of the ecosystem functioning. This concept is further elaborated by Van den Bergh *et al.* (2005) using a scale dependent approach. In particular Escaravage *et al.* (2004) defined MEP/GEP at an ecosystem scale, an ecotope scale and a macrobenthic community scale. For the Zeeschelde, the Belgian part of the Schelde estuary, Brys *et al.* (2005) applied a similar hierarchical approach to define MEP/GEP conditions for macrobenthic invertebrates and macrophytes on tidal marshes. In addition and according to the Common Implementation Strategy (CIS, 2003a, b) they established the hydromorphological conditions required for these MEP/GEP conditions, but not for fish. For fish we take the habitat needs described in chapter 4 as the MEP/GEP conditions. In that chapter habitat needs in estuaries at a fish guild level are described ensuring a Good Ecological Status. In this chapter we compile a species list for fish that should occur in the Schelde estuary when it reaches GEP or MEP condition. This list will serve to calculate threshold scores for candidate metrics in the process of the development of a fish-based index for the Zeeschelde estuary (Chapter 8).

2 Material and methods

The study area is the Schelde estuary with special interest for the Belgian part, called Zeeschelde, and its tributaries under tidal influence. Jager and Kranenbarg (2004) defined the reference for the Westerschelde, the Dutch part of the estuary to which we add the reference list for the Belgian part of the estuary.

We defined five different zones based on the Venice system (1959, Fig. 3.2): the polyhaline and mesohaline part of River Schelde, the oligohaline part of River Schelde including the River Rupel, the freshwater part of Rivers Schelde and Durme and the freshwater tributaries under tidal influence (Rivers Dijle, Zenne, Nete, Grote Nete, Kleine Nete). Like the estuary all tidal tributaries are heavily modified.

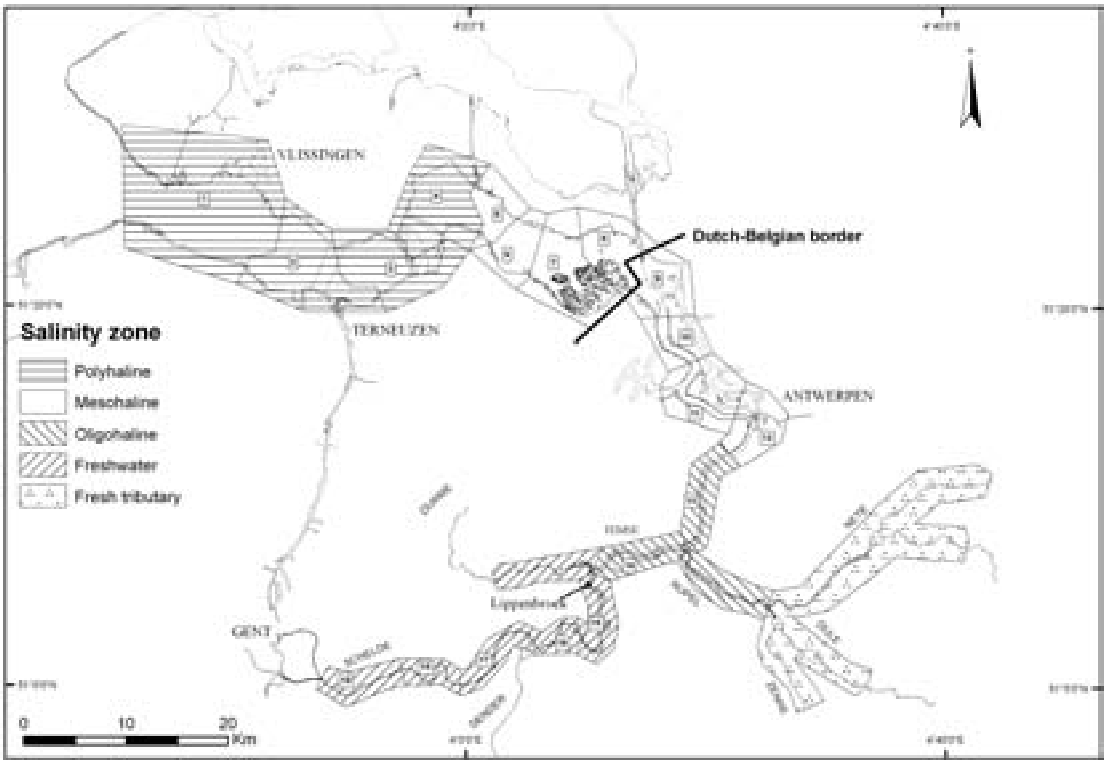

Figure 3.2: Salinity zones and Omes segments (numbers, Hoffmann & Meire, 1997) in the Schelde.

Next we compiled historical records of fish that occurred in each zone of the Zeeschelde in the period 1842 till 1947. This list was then adjusted to a MEP/GEP reference list based on data from recent sampling programmes using fyke nets (1995-2007) and the cooling-water intake screens at the Doel power plant, situated in the mesohaline part of the Schelde estuary (1991-2007). As an additional resource, we used information from peer-reviewed and grey literature reporting on non regular samplings campaigns (Table B, annex). All fish species were assigned to functional groups or guilds according to Elliott *et al*. (2007) and Franco *et al*. (2008) according to their particular niche within their area of interest. First a historical list was made. A species was included in the MEP/GEP lists if historical data indicate its presence in a particular salinity zone or if its habitat needs correspond with the habitat potentials of that particular zone (Breine *et al*., 2001, 2007). In addition, the catch frequency was considered and species that are no more or rarely caught (<5% catch frequency defined by expert judgment) are retained only in the MEP list (Fig. 3.3). Applying other threshold percentages, 1 and 10% respectively, gave only a different result for Crucian carp (*Carassius carassius*) and viviparous blenny (*Zoarces viviparus*); with the 10% threshold these species would only be a MEP species in the freshwater and mesohaline zone respectively. Eurytopic species, i.e.

fishes that are able to tolerate a wide range of conditions, and species tolerant to extreme conditions (e.g. low oxygen concentration) are placed in both lists. The GEP list differs since it should reflect a small anthropogenic impact. These historical MEP/GEP fish record lists were then adjusted following the criteria stipulated by Ramm (1990). We applied three conditions to omit some species from both the MEP and the GEP list even if they previously occurred in a particular zone: 1) they are locally or regionally extirpated, 2) their presence in a particular zone is not an indication of good status (potential) and 3) the zone is not their preferred habitat.

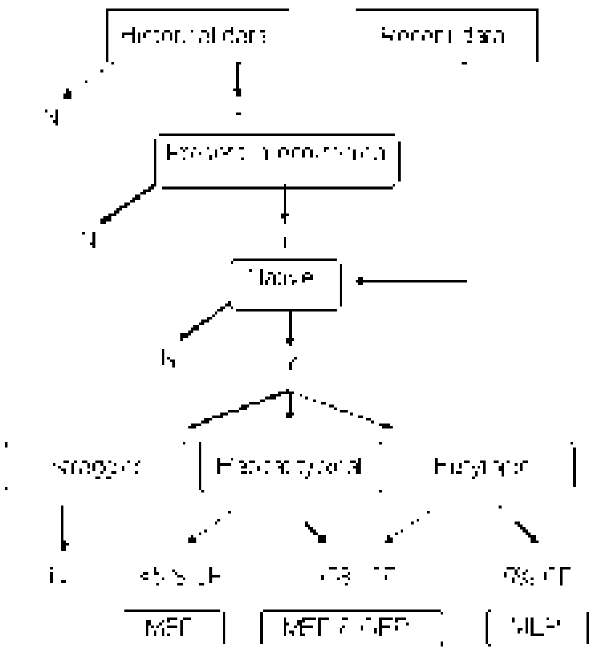

Figure 3.3: Decision tree used to allocate fish species to Maximum Ecological Potential (MEP) and Good Ecological Potential (GEP) list. At each level the answer yes or no indicates the path along the tree. Finally the attribution to the MEP or GEP depends on the catch frequency (CF). The eco-region considered is the North-East Atlantic eco-region.

Stragglers or occasional visitors in a salinity zone are not listed either since they do not depend on the estuary to complete their life cycle (Elliott *et al.*, 2007). Nevertheless some observations are interesting e.g. the snake pipefish (*Entelurus aequoreus*) was quite rare in the Zeeschelde but is now captured more frequently at Doel. de Selys-Longchamps (1842) and Poll (1947) stated that the greater weaver (*Trachinus draco*) was common, in contrast with Poll (1945) where it was considered as an irregular guest. This species was never caught in recent surveys in the estuary. All exotic species are omitted since they are indicators of disturbance (Karr, 1981), with the exception of pike-perch (*Sander lucioperca*) since this species can be considered as naturalised and has a high demand concerning oxygen concentrations (FAO, 1984). Exotic species were defined according to Verreycken *et al*. (2007). Marine species that occur in the North Sea but were never reported in the river are omitted too.

3 Results and discussion

Table C (annex) presents a presence absence reference lists for the different zones in the Zeeschelde. We structured the discussion of these lists using the ecological guild of estuarine usage (Elliott *et al*., 2007; Franco *et al*., 2008). The historical reference was based on available data from de tidal Schelde (de Selys-Longchamps, 1842 and Poll, 1945, 1947). We did not include information from archaeological studies (e.g. Van Neer & Ervynck, 1993, 1994) as anthropogenic impact in the Schelde estuary has been almost continuous since the ninth century; therefore it is scientifically impossible to trace how an unimpaired Schelde estuary would have developed.

3.1 Estuarine species

Estuarine species can complete their life cycle in the estuary. Estuarine resident species are tolerant to widely varying environmental conditions that typically characterize these transitional waters (Elliott *et al*., 2007). However, they are sensitive to the disappearance of specific estuarine habitats such as intertidal mudflats, creeks and marshes and to the accumulation of toxic substances. Therefore an estuary in MEP or in GEP status should accommodate these species. The habitat preferences for estuarine species are not fulfilled in the tributaries. According to Poll (1945, 1947), the common goby (*Pomatoschistus microps*) was quite rare in the Schelde. Common goby and sand goby (*Pomatoschistus minutus*) are at present very common (Guelinckx *et al*., 2008). The common goby is regularly found far upstream, but the freshwater is not its preferred habitat. The sand goby is less common in the freshwater part and is not kept in the freshwater lists. Transparent goby (*Aphia minuta*) is an estuarine species that should normally occur in the Schelde and is regularly caught in the mesohaline zone. This species prefers a polyhaline and mesohaline habitat (van Emmerik, 2003) and is therefore only included in the mesohaline GEP and MEP list, contrary to the list proposed by Jager and Kranenbarg (2004). Straight-nosed pipefish (*Nerophis ophidion*) was only occasionally caught in the Schelde (Poll, 1947) and has never been caught in recent surveys. This species is not retained in the Westerschelde reference list (Jager & Kranenbarg, 2004) and hence it is not considered as a GEP or MEP species. The greater pipefish (*Syngnathus acus*), Nilsson's pipefish (*Syngnathus rostellatus*) and the viviparous blenny (*Zoarces viviparus*) are estuarine resident species that in the past occurred in the Schelde (de Selys-Longchamps, 1842 and Poll, 1945, 1947). At present they are caught as far upstream as Antwerp. These species avoid freshwater (van Emmerik, 2003) and therefore are included in the mesohaline and oligohaline MEP and GEP lists only. The hooknose (*Agonus cataphractus*) is an estuarine resident species that is reported to be rare in the Schelde (Poll, 1945), which also corresponds with our catch results. Hooknose is therefore retained only in the mesohaline MEP and the polyhaline lists. Bull rout (*Myoxocephalus scorpius*) was quite common in the Schelde estuary (Poll, 1945) and is still caught from time to time. This species is included in both meso- and oligohaline GEP and MEP lists. Butterfish (*Pholis gunnellus*) is included in the reference list for the Westerschelde (Jager & Kranenbarg, 2004). Poll (1945) stated that the species was present, but it was never caught in recent samples. Therefore we exclude this species from the GEP list but included it in the mesohaline MEP list. Striped seasnail (*Liparis liparis*) used to be common in the Schelde (Poll, 1947) preferring poly and mesohaline water. Seasnail was occasionally caught in recent campaigns and is therefore a mesohaline GEP and MEP species. Both seahorse (*Hippocampus guttulatus*) and tadpole fish (*Raniceps raninus*) are absent from the lists. Seahorse was caught nearby the sea (Poll, 1945) and is stated as rare. This species prefers polyhaline water and at present is rarely caught in the Zeeschelde. The presence in the Schelde of tadpole fish has been recorded for the first time in 1943 (Poll, 1945) and this species is believed to be very rare in the estuary but more common in nearby Dutch coastal waters. Fifteen-spined stickleback (*Spinachia spinachia*) was not reported by de Selys-Longchamps (1842) or by Poll (1945). It was caught only once in Doel and it is not considered as being a GEP or MEP species.

3.2 Diadromous species

Estuaries have a crucial role as migration routes (Able, 2005). According to the season different diadromous species occur in different zones of the estuary. Absence of diadromous species is caused by human impacts, disrupting the connectivity and as a result the estuary is considered not to reach the MEP or GEP status. Thus diadromous species are, when not extirpated in the estuary or nearby estuaries, included in both lists and all zones. If all barriers, physical and chemical, would disappear these species should be able to swim all along the tributaries (see Table C, annex). The decline of sturgeon (*Acipenser sturio*), Atlantic salmon (*Salmo salar*) and allis shad (*Alosa alosa*) was already described by Poll (1945). Now they are extirpated in the Schelde basin and are not considered as GEP species. However, it is not impossible to restore their required habitat in the Schelde basin and since these species are present in some North-East Atlantic estuaries their return is possible and would indicate a MEP condition. Houting (*Coregonus oxyrhynchus*) was considered as very rare or in danger of extinction by Poll (1945, 1947). At present this species is considered to have disappeared (red list) or to be extinct (International Union for Conservation of Nature and Nature Resources: IUCN) hence it is not in our lists. In addition this species habitat area is also situated more to the north (Maitland, 2000). All the other diadromous species occur in the lists because it can be expected that they will frequent the estuary and tributaries once the habitat conditions improve (Maes *et al*., 2007). The brown trout (*Salmo trutta*) population was already declining in 1945 (Poll, 1945) and now individuals are rarely caught. However, their presence would indicate a MEP status as they are pollution intolerant species. Eel (*Anguilla anguilla*) and flounder (*Platichthys flesus*) were common in the River Schelde (de Selys-Longchamps, 1842 and Poll, 1945). Three-spined stickleback (*Gasterosteus aculeatus*) is known to be a species which is common in all types of waters in Flanders. In the mesohaline zone of the Zeeschelde three types occur (Raeymaekers *et al*., 2007) including the diadromous type. Thinlip mullet (*Liza ramado*) was previously often confounded with thicklip grey mullet (*Chelon labrosus*) a marine seasonal migrant. Poll (1945) stated that the species was abundant nearby the Belgian coast. At present specimens are recorded far upstream Antwerpen. River lamprey (*Lampetra fluviatilis*), twaite shad (*Alosa fallax*) and smelt (*Osmerus eperlanus*) are indicators of good water quality and connectivity as well as good ecological functioning of the estuary (e.g. suitable spawning locations). Sea lamprey (*Petromyzon marinus*) which is abundant according to de Selys-Longchamps (1842) is at present scarce (<5% catch frequency) and is kept in the MEP lists.

3.3 Freshwater species

The freshwater resident species can complete their life cycle in the tidal freshwater part of the estuary. They reproduce, grow up and feed in freshwater, but can also exploit the oligohaline zone evidencing their inclusion in the oligohaline MEP/GEP list too. The Zeeschelde has an important freshwater tidal zone and therefore freshwater species occupy various zones. The spatial distribution is species dependent. Some freshwater species make regular use of different zones within the estuaries, whether for seasonal migrations, nursery or feeding migrations, reproductive migrations through the estuary or the use of the estuary as a refuge (Elliott *et al*., 2007). Freshwater stragglers are considered species that occupy the mesohaline zone irregularly and only for a short time. Elliott *et al*. (2007) consider them analogous to marine stragglers but these enter the estuary from the opposite end. For the tributaries 25 freshwater species are recorded in the MEP list and 16 in the GEP list. The freshwater species ruffe (*Gymnocephalus cernuus*) is mentioned by de Selys-Longchamps (1842) but not by Poll (1945). At present this species is caught in the Zeeschelde all along its salinity gradient. Poll (1945) considers perch (*Perca fluviatilis*) to be very common in the freshwater and brackish reaches of the Zeeschelde up to Zandvliet. Recently perch is caught all over the Zeeschelde. Roach (*Rutilus rutilus*) is less abundant and is not typical for the mesohaline zone, though specimens are captured in Doel and Zandvliet. Roach is a tolerant species and its presence is justified in all GEP lists but not in the mesohaline MEP list. Bream (*Abramis brama*) and nine-spined stickleback (*Pungitius pungitus*) are typical lowland freshwater species with a tolerance for brackish water. They are opportunistic species that are caught all over the river Schelde. These species are not typical for mesohaline water and are therefore omitted from the mesohaline GEP and MEP lists since it is not its preferred habitat. Though nine spine stickleback is less common than the three-spined stickleback, it is to be found in all tributaries. As already mentioned three-spined stickleback is common in all zones. Bitterling (*Rhodeus sericeus*) is a freshwater species preferring stagnant or slow moving water with plants. Though Poll (1945) did not mention its presence in the Schelde it has been collected in different places in the Zeeschelde. Simoens *et al.* (2006) placed this species in the reference list for fresh tidal water but not for the brackish part of the Schelde. Though the species can tolerate brackish water it is not relevant to put it in the mesohaline MEP or GEP list, but it remains in the oligohaline and freshwater MEP and GEP lists. Wels catfish (*Silurus glanis*) is now frequently caught all along the tidal freshwater Schelde. Though this species can support brackish water it is kept only in the freshwater and oligohaline GEP and MEP lists since the mesohaline is not its preferred habitat (Frimodt, 1995). The weatherfish (*Misgurnus fossilis*) is now only caught in the tributaries. De Selys-Longchamps (1842) mentioned its presence in the Schelde and Poll (1942) stated that three specimens were collected in the Schelde. This species should not be present in the mesohaline zone but its presence could be indicative in the other zones. Carp (*Cyprinus carpio*) was reported by de Selys-Longchamps (1842) and Poll (1945) and is still caught in the freshwater and oligohaline zones. The species does not occur in our lists since it has an exotic origin and is tolerant to extreme conditions. Species such as white bream (*Blicca bjoerkna*), pike (*Esox lucius*) and rudd (*Scardinius erythrophthalmus*) were mentioned by Poll (1945) to be present in the Schelde. They are still caught in the Zeeschelde and even occasionally in Zandvliet (Guelinckx *et al*., 2008). These freshwater species are no part of the mesohaline fish population but can occur in the oligohaline zone. Therefore all three of them are kept in the oligohaline and freshwater GEP and MEP lists. Ide (*Leuciscus idus*) is a species that is also encountered frequently in the oligohaline zone. Ide is a rheophilic B species i.e. some stages of its life history are confined to connected backwaters (van Emmerik, 2003) with a relative high tolerance value (Breine *et al.*, 2007a). Ide is found all along the River Schelde and in most of its tributaries. However, their abundance is underestimated due to confusion with roach. Ide is considered as representative for oligohaline, freshwater and tributaries GEP and MEP lists. Crucian carp (*Carassius carassius*) is kept in the freshwater list since it is occasionally captured (>5% catch frequency) in the Zeeschelde (Simoens *et al.*, 2006). Pike-perch (*Sander lucioperca*) is an exotic freshwater species which is considered as a recent native species in the Netherlands (van Emmerik, 2003). This species can support brackish water and is quite common along the salinity gradient. Pike-perch is sensitive to temperature changes and intolerant to oxygen deficiency and can be used as an indicator for eutrophication (van Emmerik, 2003). The species prefers deeper water than provided by the tributaries and is therefore kept in the GEP lists of the main channel only. Bullhead (*Cottus gobio*) has been reported to be present over the salinity gradient (de Selys-Longchamps, 1842 and Poll, 1945, 1947) and was also recently caught in Zandvliet. This rheophilic but not obligate species lives in freshwater but can stand brackish water. Simoens *et al*. (2006) did not consider bullhead a reference species for the Schelde and its tributaries. Buysse *et al*. (2007) caught bullhead in the Nete. This intolerant species has a low range of acceptable habitats (Grandmottet, 1983) and prefers a hard substrate with gravel and stones. At present only the River Nete has a water quality that meets the demands of this species, but the morphological characteristics and substrate of the tributaries are not really optimal. We keep it as an indicator for the MEP status in the freshwater zone and tributaries. Burbot (*Lota lota*) is recently reintroduced in the upper Nete. It is possible that within time this species will be caught in the Zeeschelde since Poll (1945) mentioned that it can support mesohaline conditions although the species is not caught yet in the River Schelde. Burbot is retained in the MEP lists since it is an intolerant species. Dace (*Leuciscus leuciscus*) was not mentioned by de Selys-Longchamps (1842) and Poll (1945,

1947) and is only caught in the freshwater tributaries. Because of its rarity and ecological demands this species is included in the MEP lists for tributaries only (Turnpenny *et al*., 2004). The same reasoning applies for spined loach (*Cobitis taenia*) frequently caught in the River Nete but not found in the main channel. Bleak (*Alburnus alburnus*) is a freshwater species that is occasionally fished in the freshwater part of the main river and in the River Nete. De Selys-Longchamps (1842) mentioned its presence in the Schelde while Poll (1945, 1947) did not. According to Breine *et al.* (2007) bleak has a low pollution tolerance and is therefore only included in the freshwater and tributaries MEP lists. Stone loach (*Barbatula barbatula*) is caught in the freshwater tributaries only, where it indicates a MEP status (<5% CF). de Selys-Longchamps (1842) reported on barbel (*Barbus barbus*) and brook lamprey (*Lampetra planeri*) while Poll (1945) did not. The Zeeschelde is not their habitat. Maes *et al*. (2005) and Breine *et al*. (2007) did not include these two species in their reference lists neither. Barbel is a rheophilic A species preferring fast running water which is not typical for the Schelde tributaries. This species was not caught recently and it was decided not to retain barbel in the lists since the tributaries do not offer the required habitat demands. Brook lamprey is caught in the tributaries and therefore kept in its MEP list. Eurasian minnow (*Phoxinus phoxinus*) is an intolerant species typical for upstream water (Breine *et al*., 2004, 2007), preferring well oxygenated water and gravel substrate (Vostradovsky, 1973). Minnow has never been reported to be caught in the Zeeschelde. European chub (*Leuciscus cephalus*) and gudgeon (*Gobio gobio*) are species reported by de Selys-Longchamps (1842) but not by Poll (1945, 1947). They were caught in the freshwater tributaries (Buysse *et al*., 2007; Breine *et al*., 2007a). European chub is a rheophilic A species typical occurring in creeks and fast flowing rivers (Billard, 1997) and their presence indicates a MEP status. Belica (*Leucaspius delineatus*) is caught occasionally in the freshwater part of the Schelde but was not reported by de Selys-Longchamps (1842) and Poll (1945, 1947). Belica is a stagnophilic species that needs the presence of plants which are not really offered by the Schelde. Therefore this species is included in the tributaries list only. Tench (*Tinca tinca*) has been caught around Antwerpen but is considered a species rather belonging to standing waters and upstream the tributaries (Allen *et al*., 2002).

3.4 Marine migrants

Elliott *et al*. (2007) no longer distinguish between marine seasonal migrants and marine juvenile migrants since larval and 0+ juvenile migrations into estuaries tend to be seasonal for many marine species. But anyway estuaries in a MEP or GEP status are used by these migrants as feeding areas and refugia. Tributaries do not offer a suitable habitat for marine migrants. Herring (*Clupea harengus*) is an abundant marine juvenile species (Poll, 1945, 1947; Maes, 1997, 2001). Herring swim upstream till the oligohaline zone. Plaice (*Pleuronectes platessa*) was described by Poll (1945) as being very abundant in the Schelde, although adults were rarely caught. The species is now collected in small numbers at Doel and is retained in the mesohaline GEP and MEP lists. Sole (*Solea solea*) penetrated as juveniles quite far into the estuary (Poll, 1945). Poll (1945) mentioned also captures of numerous adults. Sole is now caught in the mesohaline and oligohaline zones and is retained in both GEP and MEP list. Juvenile of the marine species tub gurnard (*Chelidonichthys lucernus*) and whiting (*Merlangius merlangus*) have been reported in the Schelde by de Selys-Longchamps (1842) and Poll (1945, 1947). Also currently mostly juveniles are caught. The oligohaline zone is not their habitat and they are therefore retained only in the mesohaline GEP and MEP lists. At present seabass (*Dicentrarchus labrax*) is one of the most common species caught in the Schelde, which agrees with Poll (1945) who reported important quantities of juveniles. This species figures in the GEP and MEP lists of meso- and oligohaline waters. Pouting (*Trisopterus luscus*) is a marine juvenile species that was frequently observed in the Schelde (Poll, 1945, 1947) and is still captured up to Antwerpen. The species is taken into the meso- and oligohaline GEP and MEP lists. Only juveniles of brill (*Scophthalmus rhombus*) are found in the Zeeschelde. This species was not common according to Poll (1945). Consequently, it is only included in the mesohaline MEP list. Sand smelt (*Atherina presbyter*) was reported to be quite abundant in Belgian coastal waters (Poll, 1947) and is now regularly caught in the Zeeschelde. Therefore sand smelt stays in the mesohaline MEP list. Cod (*Gadus morhua*) is an uncommon seasonal migrant, of which only juveniles wander in the estuary. Cod is included in the mesohaline MEP list only. Poll (1947) reported the occasional presence of the marine juvenile migrant dab (*Limanda limanda*). In recent surveys this species is rarely caught and is therefore taken in the mesohaline MEP list only. Turbot (*Psetta maxima*) is rarely caught and if so only juveniles. Turbot is included in the Dutch list (Jager & Kranenbarg, 2004) but kept in our mesohaline MEP list only. Pollack (*Pollachius pollachius*) was described as being rare in Belgian coastal waters (Poll, 1947) and there are no records of it from de Selys-Longchamps (1842) and Poll (1945). Pollack is not collected in recent fish campaigns in the Zeeschelde and is therefore omitted from our lists. In the past sprat (*Sprattus sprattus*) entered in large numbers the estuary between January and July (de Selys-

Longchamps, 1842 and Poll, 1945, 1947). This species is still often caught and is also a reference species for the Westerschelde (Jager & Kranenbarg, 2004). It is taken into the meso- and oligohaline GEP and MEP lists. According to Poll (1947) anchovy (*Engraulis encrasicolus*) was a seasonal guest from April to August that visited the estuary in large numbers to spawn. At present they are rarely caught upstream Doel. They are retained in the mesohaline MEP and GEP lists. Thicklip grey mullet (*Chelon labrosus*) was considered as rare in the Schelde (Poll, 1947) but is occasionally caught (<5% CF) in recent surveys and is therefore included in the mesohaline MEP list. Garpike (*Belone belone*) was uncommon in the estuary (Poll, 1945). Though it was not caught recently it has a place in the mesohaline MEP list, since it is an indicator of good water quality and is also a reference species for the Westerschelde (Jager & Kranenbarg, 2004). The lumpsucker (*Cyclopterus lumpus*) was rarely caught (Poll, 1945, 1947) and this is still the case. This species is in the mesohaline MEP list. The fivebeard rockling (*Ciliata mustela*) was rarely caught in the past (Poll, 1945, 1947) but is now regularly caught in Doel. Grey gurnard (*Eutrigla gurnardus*), sting ray (*Dasyatis pastinaca*) and pilchard (*Sardina pilchardus*) were only encountered occasionally in the estuary (Poll, 1945, 1947). Of them only grey gurnard was caught haphazardly in Doel and none of the three species are withheld in the lists. Small sandeel (*Ammodytes tobianus* or *A. lancea*) was common in the Schelde estuary (Poll, 1945). This species is occasionally caught and is therefore kept in the mesohaline MEP list. Lozano's goby (*Pomatoschistus lozanoi*) is not mentioned in historical reports but is recently regularly caught in the mesohaline zone (Breine *et al*., 2001).

4 Conclusions

To assess the ecological status of heavily modified transitional waters the European Water Framework Directive requires definitions of Maximal and Good Ecological Potential (MEP/GEP) and the design of classification tools for specified biological quality elements. The hydromorphological, physical and chemical MEP/GEP are described by Brys *et al*. (2005). Their approach was also used to define the guild specific habitat needs (qualitative) for fish in the Schelde (Chapter 4). If these habitat needs are fulfilled, thanks to restoration and mitigating actions, then we consider the estuary to be in MEP condition for fish. The near fulfilment brings it in the GEP condition. Based on a literature review in combination with recent fish catch data we were able to make guild specific qualitative MEP/GEP lists for the different zones within the Zeeschelde estuary and its tidal tributaries. For each fish species the relevance of its presence in each salinity zone was examined. The geographical spreading and ecological demands were assessed and were decisive for the acceptance of a specific species within the lists. The ecological knowledge of the assessed species is available and sufficient to reduce the risk of mistakes in attribution. The lists proposed should be considered as a starting point to develop quantitative guild lists i.e. include numbers instead of presence/absence information. Attributing threshold values to these quantitative lists will allow expressing the ecological status as an ecological quality ratio (EQR) between 0 and 1. The guild approach facilitates the development of such an assessment tool. We are aware that by grouping fish into guilds particular information can be lost. On the other hand the guild approach is widely used and accepted to develop robust assessment tools for the ecological status of surface waters. Such an evaluation system normally assesses the deviation between a reference condition and the actual condition. Therefore these lists can be used to develop fish-based indices.

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Chapter 4

Ecological goals and associated habitat needs for fish in estuaries: a case study of the Zeeschelde (Belgium)

Jan Breine, Joachim Maes, Maarten Stevens, Mike Elliott & Erika Van den Bergh

Abstract

We describe habitat needs for fish populations in estuaries needed to ensure the realisation of ecological goals. We take the view that Good Ecological Status (GES), as defined by the European Union Water Framework Directive (2000/60/EEC), is obtained when those ecological goals are fulfilled. The Zeeschelde estuary is presented as a case study for the description of ecological goals, but the described approach can be applied to all North Sea estuaries. In order to make the method widely applicable we first classify fishes into guilds, relevant for the formulation of ecological goals. Next we describe guild-specific ecological goals for fish. Based on the literature, habitat requirements for specific fishes are defined and used to define habitat needs allowing a proper functioning of the estuarine ecosystem. We describe habitat needs at a guild level and indicate how these can be achieved.

Keywords: ecological goals, estuaries, fish, habitat needs, Zeeschelde

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1 Introduction

The progressive loss of estuarine areas remains a serious threat to the preservation of estuarine biotopes and the integrity of estuaries as a whole. In particular, the historical loss of intertidal areas is obvious and the associated effects are disproportionally large (Elliott & Taylor, 1989a; McLusky *et al.*, 1992; Costanza *et al*., 1997). Davidson *et al.* (1991) referred to the process of gradual loss of shallow habitats as the "estuarine squeeze". In the upper part i.e. the freshwater tidal zone, the "estuarine squeeze" has moved the high water line shoreward through land claim, building of sea defences and dock construction. The driving forces of the lower part of the "estuarine squeeze" are the extraction of sediments, the construction of barrages and sea level rise. The combined effect of these hydromorphological pressures has resulted in the loss of intertidal areas, the relative increase of deeper subtidal areas and the narrowing of the estuarine channel (Van den Bergh *et al*., 2009). A comparative analysis of 14 European estuaries showed that land claim, channel management, barrages and impoundments are the most important mechanisms resulting in disturbances of the estuarine fish communities (Cattrijsse *et al.*, 2002). These range from a loss of species diversity, through a change from an estuarine to a freshwater assemblage, to the disappearance of whole communities. Under natural conditions, erosion and sedimentation are in balance and the natural loss of a habitat at one site may be compensated by sedimentation at another site (McLusky & Elliott, 2004). Even floods or storm-surges that may destroy complete habitats are likely to be remediated in natural systems by a resulting adjustment development of the main tidal channels. Due to these natural changes in hydrogeomorphology fish communities will slowly change to a species composition more suited to the new situation. Where land claim or dredging negatively affect benthic communities, a principal food resource for estuarine fishes, those fish communities will also be affected in terms of species number, abundance and biomass (Elliott *et al*., 1998; Kennish, 2002). Reduction of the food supply and the loss of habitat reduce the value of estuaries as a nursery area and thus its carrying capacity (Thiel, 1995; Köhler & Köpcke, 1996; Drake & Arias, 1997; Colclough *et al*., 2005; Lotze *et al*., 2006; Martinho *et al*., 2007). The presence of ports, dykes and other artificial structures that stabilize the channel create an increased flushing effect and segregation of the tidal currents (Cattrijsse *et al.*, 2002). The building of docks, wharves and jetties result in a loss of intertidal area or soft sediment, although they may create an artificial hard substratum which attracts a rocky shore community and its associated fish fauna (Hostens & Hamerlynck, 1994; Pérez-Ruzafa *et al*., 2006).

To avoid further habitat loss, aquatic ecosystems are protected by binding regulations. The protection of nature, diversity of habitats, species, as well as the functioning of aquatic ecosystems is the subject of a series of international agreements and legal commitments (Apitz *et al*., 2006). For example, in Europe, environmental legislation that covers estuaries includes the European Water Framework Directive (WFD, 2000) as well as the Wild Birds and Habitats Directives (BHD, 1979, 1992). The first of these directives requires ecological quality goals to be met, whereas the latter directives require conservation goals to be met. Hence, the ecological goals require to be derived against the habitat needs of the ecological components of the system, in this case the fishes within estuaries, and they should not be in conflict with conservation goals. In addition, different competences derive from local, regional, national, multilateral and international initiatives, each with their own objectives and targets. Clearly, these commitments apply on different spatial scales covering from the regional scale (Europe, North Sea area, country) to the local scale (river basin, river, habitats). Consequently, an assessment of compliance to environmental regulations that are in vigor optimally adopts an integrated, hierarchical structure (Fig. 4.1). Under this approach, particular commitments aim at sustainable and integrated management but may focus on a different spatial level of the ecosystem and its functioning. Accordingly, objectives at each level aim at ensuring effective functioning of the ecosystem to achieve the commitments involved. The term ecological emphasizes that these are functional targets within the ecosystem, including interactions among the different fish species and between fish and their environment. Once the objectives are set, quantitative indicators or measurement endpoints have to be defined in order to measure the actual status, and to compare the ecosystem state to reference levels set by the ecological goals (Van den Bergh *et al*., 2005, 2009). Depending on the scale, indicators are either based on integrated data or represent an explicit measure of the state of the ecosystem. Therefore, any monitoring scheme should provide a wide range of data so that at each level of assessment, the necessary information can be obtained (Detenbeck & Cincotta, 2008). Restoration measures subsequently aim at restoring the processes that generate the required habitats and species populations to comply with the proposed ecological goals (Elliott *et al*., 2007a). Often the potential for restoration remains possible, since most species and functional groups persist, albeit in greatly reduced numbers (Lotze *et al*., 2006).

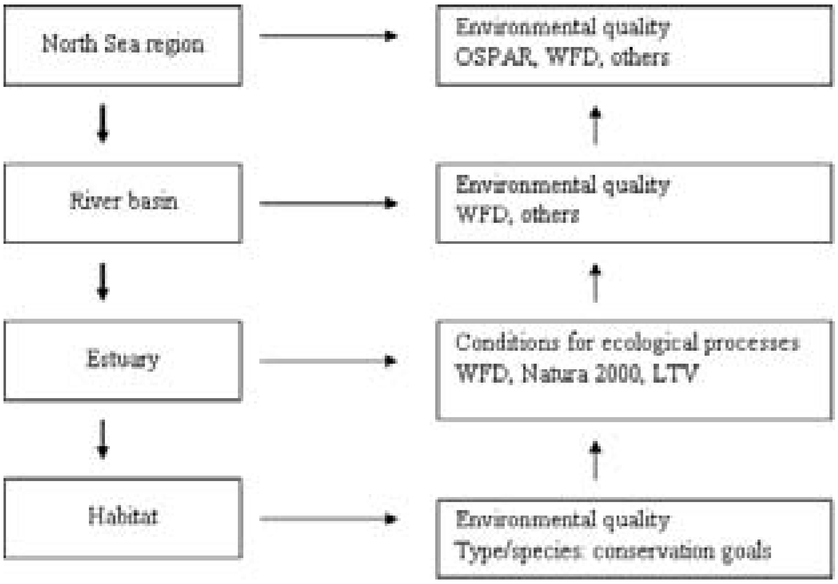

Figure 4.1: Hierarchical integration of conservation objectives depending on the spatial scale (adapted Van den Bergh *et al.*, 2009).

This study defines ecological goals for estuarine fishes using the Zeeschelde (tidal Schelde in Flanders, Belgium) as a case study and example for other North-East Atlantic estuaries. Ecological goals for estuarine fishes are defined as targets that should be reached in order to ensure a healthy and dynamic fish community in that ecosystem. As such they contribute to the reestablishment of the estuaries' autogenic processes, its organisation, vigor and resilience. Van den Bergh *et al*. (2005) describe qualitative ecological goals for 22 key indicators in the Schelde Estuary in a hierarchical way, starting from physical and chemical processes up to the level of specific habitat types and species. This study develops this approach and describes in detail appropriate ecological goals and habitat needs for specific fish guilds in estuaries in order to assure the 'good status' of fish populations as required by the Water Framework Directive. The habitat needs ensure spawning, breeding, feeding or growth to maturity. An estuary is defined as that part of a river which is under tidal influence (Fairbridge, 1980; Elliott & McLusky, 2002). Ecological status can be assessed using classification tools that were especially developed for that purpose. For example, Breine *et al*. (2007) developed a fish-based estuarine index of biotic integrity (EBI) for the mesohaline and oligohaline zones in the Schelde estuary (Chapter 7). The EBI includes attributes such as total number of fish species, percentage of smelt (*Osmerus eperlanus*) individuals, percentage of marine migrating juvenile fish and percentage of omnivorous and piscivorous fish. However, it is axiomatic that fish species of the different ecological guilds which are typically present in an estuary should be able to complete their lifecycles within or adjacent to the estuary (Jager & Kranenbarg, 2004). Hence, the guild approach can be applied to define the habitat needs to meet these ecological goals against a set of anthropogenic pressures such as pollution and morphological change. The implementation of mitigating or compensation measures to those pressures (as per Elliott *et al*., 2007a) therefore has to ensure the presence of a diversified fish community as stipulated by the WFD. In particular, the EU has stipulated that by 2015 the fish community should be comparable to that of an estuary in a good ecological status or if the water body is heavily modified which is the case for the Schelde, then the water body has to be adjudged as having Good Ecological Potential (2000/60/EEC).

This study develops these concepts by introducing the guild concept as a framework for establishing ecological goals, defining the ecological goals for the Zeeschelde estuary based on knowledge of the frequenting fish guilds and then linking these goals to specific habitat needs of the target fish fauna.

2 The ecological guild concept as a framework for defining ecological goals and habitat needs

The guild approach to categorizing estuarine fishes is used in this chapter as a concept to define goals and habitat needs. A guild or a functional group is a group of species that exploits the same class of environmental resources in a similar way (Coates *et al*., 2007; Elliott *et al*., 2007). The guild concept therefore merges biodiversity with the ecosystem functioning, as it links species to the functions or services that estuarine ecosystems are providing to them, such as the provision of food, shelter and habitat. A recent global review of the application of the guild concept to estuarine fish communities indicates that the separation of estuarine fish communities in three groups of functional guilds provides sufficient information for an assessment and that these guilds produce more information regarding the functioning of the estuarine systems than do structural indices such as taxonomic diversity (Elliott *et al*., 2007; Franco *et al*., 2008). The groups include the Estuarine Use Functional Group, the Feeding Mode Functional Group and the Reproductive Mode Functional Group. The authors defined within each of these major categories subgroups which will be referred to as guilds. The presence of the different guilds therefore indicates the particular ecological function that the estuarine ecosystem fulfils. In turn, we take the view that the conservation of each guild as an integral part of the estuarine fish community is adopted as an ecological goal such that achieving each ecological goal is assumed to indicate the good ecological status (or potential) of the estuary. This concept is globally applicable but should be adapted locally using fish sampling data and historical records to narrow down the set of ecological goals and associated habitat needs to only those guilds of the functional groups that should be present in the estuary at good ecological status or potential and, to which management plans need to be addressed. However, this approach is complicated by the fact that few fish species are confined to particular estuarine habitats or to estuaries (Craig & Crowder, 2000; Franco *et al*., 2008). Often estuarine fish populations connect to regional marine and freshwater populations (Able, 2005). Accordingly, the regional status of fish populations affects the status of those populations at finer spatial scales. Therefore, we consider it necessary to define ecological goals on four spatial scales of interest which are: the regional scale, the river basin scale, the estuary scale and the habitat scale.

3 The Zeeschelde estuary as case study

We apply the concept of ecological goals to the fish community of the Zeeschelde estuary, the Belgian part of the Schelde estuary. The ecological goals for the Zeeschelde and its tributaries under tidal influence are developed based on long term peer reviewed fish sampling data (Van Damme *et al*., 1994; Maes *et al*., 1996, 1998a,b; Peeters *et al*., 1998; Maes *et al*., 1999; Peeters *et al*., 1999; Ercken *et al*., 2002; Maes *et al*., 2003, 2004a,b, 2005; Stevens, 2006; Stevens *et al*., 2006; Cuveliers *et al*., 2007; Buysse *et al*., 2008 and Guelinckx *et al*., 2008). This data contributed to the development of a reference list (Chapter 3, Breine *et al*., 2008: Table D in annex). In this list all fish species were assigned to the ecological guilds defined by Elliott *et al*. (2007) and Franco *et al*. (2008). We consider here the GEP reference lists as it is impossible to reach the MEP status. Ecological goals (EG) and habitat needs (HN) for the Estuarine Use Functional Group were then defined (Fig. 4.2) since they comprise those of the feeding mode functional group and the reproductive mode functional group. Indeed if a sustainable population of freshwater and estuarine species is present and marine migrants and diadromous species frequent the estuary one may assume that an undisturbed trophic web is present and that the recruitment occurs normally. The species were assigned to guilds based on an extensive and critical literature review of the life strategies of the fish species combined with expert judgment. This takes into consideration ontogenetic changes, i.e. that fish can occupy different habitats during particular periods of their life history (Bulger *et al*., 1993;

Elliott & Dewailly, 1995; Schiemer & Waidbacher, 1999; Pihl *et al*., 2002; Quak, 1994; Elliott & Hemingway, 2002; van Emmerik, 2003).

For the estuarine use functional group, different guilds have been distinguished according to habitat use, which is related to the life history strategy (Elliott & Dewailly, 1995; Mathieson *et al*., 2000; Pihl *et al*., 2002; Thiel & Potter, 2001; Franco *et al*., 2008). For the marine migrants we only consider the marine juvenile migrants since estuaries are considered as essential habitats for this ecological guild.

From the literature we defined the feeding mode functional guilds for juveniles and adult life history stages based on the diet preference of each species. We used information from Batzer *et al.* (2000), Belpaire *et al.* (2000), Breine *et al.* (2001), Breine *et al.* (2004), Breine *et al.* (2007), Bruslé & Quignard (2001), De Nie (1996), Elliott & Dewailly (1995), Elliott *et al.* (2002), Gerking (1994), Gerstmeier & Romig (1998), Jager & Kranenbarg (2004), Maitland (2000), Mathieson *et al.* (2000), Muus *et al.* (1999), OVB (1988), van Emmerik (2003) and Franco *et al*. (2008) to assign species to feeding guilds (Table D in annex). We added two guilds to those defined by Franco *et al*. (2008): piscivorous and vertivores/piscivores, in order to be able to classify some freshwater species. To allocate species to a reproductive guild category we integrated information from Balon (1975, 1981), Elliott & Dewailly (1995), Belpaire *et al*. (2000), Costa *et al.* (2002), Aarts & Nienhuis (2003), van Emmerik (2003), Elliott *et al*. (2007) and Franco *et al*. (2008).

4 Ecological goals for the estuarine use functional group

Within this group we distinguish four relevant guilds for the estuary: freshwater species, estuarine species, marine migrants and diadromous species. Marine stragglers are not included since they do not depend on the estuary to complete their life cycle (Elliott *et al*., 2007).

4.1 Freshwater and estuarine species

The presence of freshwater species is restricted to the freshwater, oligohaline and mesohaline parts of the estuary (Franco *et al*., 2008). However, ecological goals for this guild specifically target the freshwater tidal part of the estuary (Fig. 4.2). For estuarine species estuaries can be regarded as essential habitats for spawning, feeding and growing, situated in the salinity range from the oligohaline to the marine (Franco *et al*., 2008).

For these guilds the following ecological goals (EG) are suggested:

EG1: On a regional and basin-wide scale a sustainable population of all reference freshwater species represented in Table D (Appendix) as well as a sustainable population of estuarine resident species (Table D) should be present.

EG2: On an estuary scale seasonal dynamics of the freshwater species and estuarine fish communities should be preserved allowing those species to move to their spawning places, nursery and feeding habitats. A nursery is defined as a habitat that compared with other habitats supports a greater contribution to the adult recruitment (Beck *et al*., 2001).

EG3: On a habitat scale, different life stages of freshwater and estuarine species should be present according to the habitat type.

4.2 Marine migrants

Marine migrants were previously defined as marine juvenile species or marine seasonal species and are a dominant guild in European estuaries (Franco *et al*., 2008; Selleslagh *et al*., 2009). They share many biological and ecological properties with estuarine species but do not spawn in the estuary. Their presence in the estuary depends on the spawning success offshore. The main difference between the marine migrants and estuarine species guilds is the time spent in the estuary. Two resources are considered of importance: space and food. The former can be subdivided into amount of area used and amount of time this area is used (extent versus duration). Based on the present knowledge of marine migrants, the following ecological goals are suggested:

EG4: On a regional and basin-wide scale marine migrants (0-group individuals) should be present in accordance to the season.

EG5: Preserving the typical seasonal sequence of marine juvenile migrants allows the full exploitation of the estuary as nursery and feeding ground and is a priority goal.

EG6: Young marine individuals should find temporary shelter and food in the different habitats.

4.3 Diadromous fish species

Diadromous fish use both marine and freshwater environments to complete their life cycle (McDowell, 1996). The (sub)tidal transition zone between rivers and oceans is a crucial habitat for diadromous fish linking spawning grounds with adult habitat. Some anadromous fish species, for instance twaite shad (*Alosa fallax*), move upstream and use the tidal freshwater area as spawning habitat (Maes *et al*., 2008). Shallow areas or vegetated habitats throughout the estuary serve as essential nurseries for 0-group anadromous and catadromous fish. Based on the present knowledge of diadromous species, the following ecological goals are suggested:

EG7: On a regional scale, endangered diadromous populations of in particular Atlantic salmon (*Salmo salar*), Atlantic sturgeon (*Acipenser sturio*), houting (*Coregonus oxyrhynchus*), allis shad (*Alosa alosa*) and European eel (*Anguilla anguilla*) (Robinet & Feunteun, 2002; ICES, 2006) should have self sustaining populations.

EG8: Basin-wide, self sustaining populations of geographically-relevant diadromous species (e.g. twaite shad (*Alosa fallax*), river lamprey (*Lampetra fluviatilis*), smelt (*Osmerus eperlanus*), thinlip mullet (*Liza ramado*) and flounder (*Platichthys flesus*) for the Schelde) should frequent the estuary.

EG9: Within the estuary and at a habitat level diadromous individuals from the > 0-group should be present.

5 Associated habitat needs for the estuarine use functional group

Estuarine species spawn only in estuaries where they complete their life cycle although they can show regular movements between the estuary and adjacent aquatic habitats (Franco *et al*., 2008). Marine migrants are defined as those species that use estuaries as a nursery area as 0-group individuals and shallow areas in the marine and brackish part of estuaries that are either turbid or vegetated may especially qualify as fish nurseries (Le Pape *et al*., 2007; Lazzari, 2008). Rijnsdorp *et al*. (1992) and Gibson (1994) hypothesize the relationship between nursery size and fish recruitment which indicates that increasing total estuarine fish nursery habitat has a positive effect on the recruitment of marine juveniles and can act as a rehabilitation measure. Estuaries are characterised by seasonal patterns in species composition that are related to species-specific life history strategies and are highly influenced by processes occurring in the sea. Complete seasonal niche partitioning of the particular estuarine ecosystems suggests optimal functioning of the fish nurseries.

The freshwater tidal area of the Schelde, including its tributaries, is dominated by representatives of eurytopic fishes, i.e. fishes that are able to tolerate a wide range of conditions and have consequently very widespread distributions (Calow, 1998). They include roach (*Rutilus rutilus*), pike (*Esox lucius*), perch (*Perca fluviatilis*) and bream (*Abramis brama*) (Chapter 2). This is the result of a lowland setting and a river system that is categorised as the bream zone (Huet, 1949). Eurytopic fish benefit from a good hydrological connection between the different components that together constitute the river corridor (channel, marshes, floodplain) and from the presence of tributaries (Pollux *et al*., 2006). Although habitat preferences often change during the course of development (Grenouillet *et al*., 2000) supratidal floodplains are considered essential habitats as they provide suitable spawning and juvenile conditions e.g. as a food source (Tockner *et al*., 2000).

Rheophilic species have all their life stages confined to lotic waters (Nobel *et al*., 2007). Rheophilic A species have a life strategy adapted to fast water and prefer the mid-channel of large rivers such as the Schelde (e.g. dace, *Leuciscus leuciscus*, Van Liefferinge *et al*., 2004). Some stages of the life history of rheophilic B species (e.g. gudgeon, *Gobio gobio* and burbot, *Lota lota*) are confined to well connected backwaters (Aarts & Nienhuis, 2005). Rheophilic B species are absent from the main channel, as these habitats are replaced by intertidal marshes although they should also occur in the tributaries.

Limnophylic species may occur but estuarine habitats are not essential. Based on habitat preferences, efforts should focus on rheophilic A and B species as well as eurytopic fishes. Based on present knowledge the following habitat needs (HN) are suggested:

5.1 Regional and basin-wide scale habitat needs: connectivity

HN1: On a regional scale fishing activities should be controlled in order to protect marine migrants and diadromous species.

Illegal fishing is one of the greatest threats to marine ecosystems (FAO, 2005). The depletion of key fish stocks can be stopped by adopting the MCG scenario (monitoring, control and surveillance) involving prevention and deterrence. The creation of marine reserves assuring protection against fishing or development is essential. If 25% of the North Sea surface would be fishery free the number of fish species present would double and its biomass would increase with 200% (Dekker *et al.*, 2009).

HN2: On a basin-wide scale, the ecological connectivity along longitudinal and transversal river gradients permits marine migrants to enter the estuary and the development of a sustainable fresh water population of rheophilic A and eurytopic fish species in the estuary.

Ecological connectivity permits unconstrained movements of diadromous fish between spawning and nursery grounds and the adult habitats (Lassalle *et al*., 2009). This includes the access to inland water systems and catchment areas which have to be (made) passable for migrating fish, even those with restricted swimming capacity, such as glass eel and three-spined stickleback (*Gasterosteus aculeatus*) or those as flounder larvae which rely on selective tidal stream transport for migration (Jager, 1999). Tributaries should be accessible since they contribute to the recruitment of migrant species (Pollux *et al*., 2006). This ecological connectivity includes an absence of physical barriers or mitigation by specialised constructions to allow fish passage. It also includes the absence of chemical barriers by ensuring a good water quality (e.g. dissolved oxygen concentration, Chapter 5) and low nitrate concentration (Tong, 2001). As an example, table 4.1 shows DO standards developed by the Thames Tideway Strategy Group (TTSG) aimed specifically at the Thames Estuary Tideway, but which have a more general application in other British transitional waters (Turnpenny *et al*., 2006).

Table 4.1: Dissolved oxygen (DO) standards proposed by the Thames Tideway Strategy Group (Turnpenny *et al*., 2006)

DO (mg l^{-1})	Return Period (yrRP, years)	Duration (# of 6 hour tides)
4	1	29
3	3	3
2	5	1
1.5	10	1

Note: the objectives apply to any continuous length of river ≥ 3 km.

Duration means that the DO must not fall below the limit for the stated number of tides.

A tide is a single ebb or flood.

The bases for these standards are:

- The one week standard (4 mg l^{-1}, 1 yrRP, >29 tides) was selected to ensure protection against chronic effects such as depression of growth and avoidance of hypoxic areas.

- The 24 h standard (3 mg l^{-1}, 3 yrRP, >3 tides) and the 6 h standard (2 mg l^{-1}, 5 yrRP, > 1 tide) were selected to provide protection to stocks.

- The lowest standard (1.5 mg l^{-1}) was included to ensure protection from mass mortalities.

However, in chapter 5 we use a modelling approach to set specific threshold values for DO in the Zeeschelde.

5.2 Estuary scale: space

HN3: The estuarine nursery size should be sufficiently large (temporal and spatial) such that it contributes significantly to the recruitment of young estuarine and marine fish populations.

The importance of estuaries for recruitment of marine species decreases upstream along the decreasing salinity gradient (Elliott *et al.*, 1990). Within a geographic area the estuary should have an undisturbed hydrographic regime assuring the presence of nursery areas such as salt and freshwater marshes with a diversified creek pattern (Rozas *et al*., 1988; Hampel *et al*., 2003, 2004). As such intraspecific and interspecific competition is prevented by abundant food resources and by the spatial and temporal segregation within the nursery areas (Martinho *et al*., 2007). Size is only one criterion, the estuary should also have an appropriate water depth and its shape should be convenient for the larvae (hydrodynamic and climatic regime). Connectivity and favourable hydrodynamic conditions (tidal transport) should allow the larvae to move to adult habitats and the physical-chemical conditions (e.g. DO, salinity, suspended matter) should not be restrictive.

HN4: At the scale of the estuary, the presence of floodplains and side waters along the tidal freshwater part of the estuary ensures the annual recruitment of freshwater eurytopic fishes.

This habitat need relates to the hypothesis that floodplains and side waters represent a critical factor in life history of eurytopic fishes through the provision of refuge and food resources (Grandmottet, 1983; Turner *et al*., 1994; Sindilariu *et al*., 2006). Pas *et al*. (1998) state that Tielrodebroek, a flood control area of about 90 ha at the mouth of the River Durme, functions as a spawning and nursery area for some freshwater species. The presence of the experimental flood control area under the influence of a controlled reduced tide (FCA-CRT) in Lippenbroek, situated in the freshwater zone of the Schelde estuary, has shown its potential as nursery and refuge area for freshwater and some diadromous species (Simoens *et al*., 2007). The presence of fish larvae of species such as Prussian carp (*Carrasius gibelio*), stone moroko

(*Pseudorasbora parva*) and three-spined stickleback (*Gasterosteus aculeatus*) indicate that spawning activities occur in Lippenbroek. Channel geometry in natural watersheds is typically meandering with a diversity of substrata (Karr & Dudley, 1981).

5.3 Habitat scale: diversity and quality

HN5: The presence of shallow, low dynamic (as opposed to high tidal energetic areas) habitats (e.g. sheltered mudflats, saltmarshes and tidal marshes with permanent pools) that provide protection and a high and continuous supply of food should be ensured for estuarine species and marine juvenile migrants (Amara *et al*., 2001; Le Pape *et al*., 2003; Gilliers *et al*., 2006).

A range of different site substrata should be present to contribute to the benthic primary production (Svensson *et al*., 2007). Although fish in estuaries are in general opportunistic (Miller & Dunn, 1980; Elliott *et al*., 2002; Breine *et al*., 2007), feeding interactions assessments reveal the existence of different diet compositions in species (Salgado *et al*., 2004; Dauvin & Desroy, 2005; Dolganova *et al*., 2008; Pasquaud *et al*., 2008). It may be assumed that the abundance and distribution of fish feeding in the estuary, and hence the carrying capacity for fishes, is related to the quantity of food available in the intertidal and subtidal areas (Able *et al*., 2005). In turn hydrographic regime, site specificity and substratum are also factors controlling fish feeding (Elliott *et al.*, 2002). Although generally small in surface, tidal mudflats are well defined as juvenile fish feeding areas (Costa & Elliott, 1991; Amara & Paul; 2003; Stevens, 2006). For estuarine species a diversity of habitats (mudflats, marshes, creeks) may lead to higher species diversity (Elliott & Hemingway, 2002). Estuarine residents often produce demersal eggs or have parental care since pelagic eggs and larvae cannot withstand the local currents and wash-out events. For example, smelt will produce its eggs in areas less liable to wash-out, ensuring its young are retained in the estuary (Kottelat, 1997; Kottelat & Freyhof, 2007) and other species migrate into the intertidal to spawn during high water levels (Edwards & Steele, 1968; Van der Veer & Bergmann, 1987). The benefits of intertidal spawning follow from the fast development rate when emerged (Taylor, 1999), due to the food availability and from temporal and spatial refuge for adult spawners and embryos (Gibson, 1982; Van der Veer & Witte, 1993). Creeks are juvenile habitat for many species (Chapter 6; Nemerson & Able, 2004) and hydrology and channel morphology influence the occurrence of nekton in tidal marsh creeks (McIvor & Rozas, 1996). The presence of tidal marshes additionally provides shelter and food for a large variety of fish species. While many larvae behave as habitat generalists, shelter and associated food availability will remain important. Sheltered microhabitats (e.g. tidepools) provide refuge from predators or competitive species (Kneib, 1987). Therefore the characteristics of mudflats and tidal marshes should be defined according to their good quality as well as their dimensions, allowing the development of a dynamic and diversified habitat. The mudflats should have an optimal sediment composition and position in the tidal frame for the maintenance of suitable prey biomass (McLusky & Elliott, 2004).

Environmental variation upstream and in the tidal tributaries will result in different species having successful recruitment at different times, a feature important for coexistence and satisfying specific demands. For eurytopic and rheophilic species the availability of sheltered diversified intertidal habitat surfaces and subtidal areas, with a diverse food supply in the freshwater estuary, are essential as nursery and feeding grounds (Karr & Dudley, 1981; Angermeier & Schlosser, 1987; Laffaille *et al*., 2004). In turn, microhabitats with low water velocities e.g. side-arms and flood zones, are essential especially for fish larvae. Varying flow regimes result in substratum sorting, erosion and deposition events which may influence the production of ecological niches and thus species diversity (Karr & Dudley, 1981). Mann (1996) gives an overview of the critical and preferred current velocities for fish larvae. Upstream, the physical conditions of the tributaries are supposed to be less severe than in the main channel, allowing the presence of macrophytes which increases the habitat structural complexity. This favours the survival of rheophilic B offspring since it provides shelter and food. In addition limnophilic species can benefit from the presence of plants. The presence of upstream situated spawning habitats such as sand beds, habitats with gravel and/or stone bottom with clear and oxygenated water and sufficient intertidal habitat are therefore required for the success of the rehabilitation programmes of diadromous populations. The interaction of temperature, surface area, stream flow and productivity influences the presence of diadromous species (Béguer *et al*., 2007) and an appropriate morphology should be present to provide shelter and food for >0-group diadromous individuals. Banks should contain varying stretches to enhance recruitment e.g. stretches with a mixture of pebbles, flooded grass, aquatic plants and tree roots. Dredging activities should not occur and mud should be absent so that spawning can occur on gravel or sand. The eggs and larvae should not be smothered by layers of fluid mud.

HN6: A good water quality is an essential requirement for fish (Huet, 1962; Mann, 1996).

For example, long term low dissolved oxygen reduces reproduction and migration particularly in a spring and summer spawning species (Landrey *et al*., 2007). Seasonal migrations of estuarine species and marine migrants can only occur if conditions in the estuary are favourable for the fish i.e. temperature (Aprahamian, 1988) and dissolved oxygen conditions should be within acceptable ranges. For example temperatures above 15ºC and DO below 5 mgl^{-1} can produce a water quality barrier to migration (Elliott & Hemingway, 2002; Maes *et al*., 2007, 2008). DO should be high enough that it does not create constraints for the larvae and juveniles (see HN2) since they are less successful at leaving or avoiding regions with low DO concentrations (Breitburg *et al*., 1999). Maes *et al*. (2007) state a threshold of 5 mgl^{-1} as the DO minimum based on criteria for US estuaries, as outlined by USEPA, and on empirical models describing the response of estuarine fish to different oxygen concentrations (Maes *et al*., 2005, 2007, 2008). Pollutants (heavy metals and organic contaminants) have a negative impact on fish growth and density (Eastwoord & Couture, 2002; Forester *et al*., 2003; Gilliers *et al*., 2006) and diversity (Courrat *et al*., 2009). Searcy *et al*. (2007) suggested a relation between a higher mortality and a slower larval and juvenile growth and hence pollution should be avoided.

As a summary figure 4.2 groups the different ecological goals and associated habitat needs.

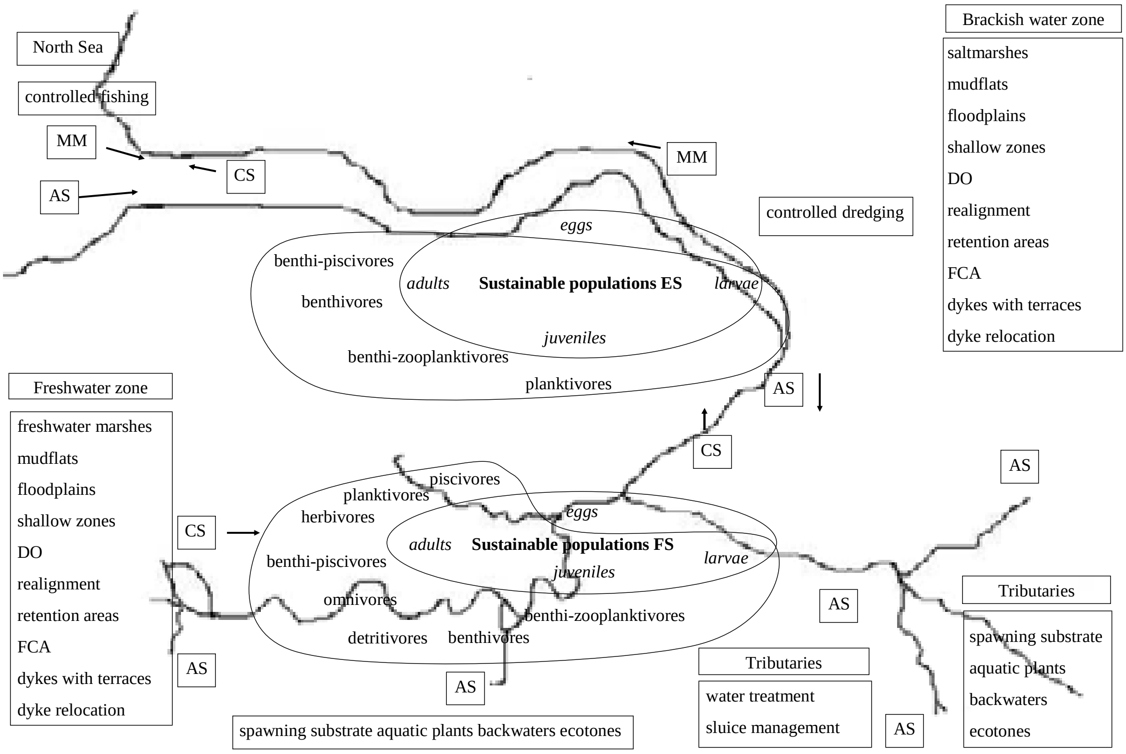

Figure 4.2 Ecological goals and associated habitat needs in the Schelde estuary.

Ellipses include the ecological goals and rectangles the associated habitat needs and the arrows indicate migration direction. MM: marine migrants, AS: anadromous species, CS: catadromous species, ES: estuarine species, FS: freshwater species, DO: dissolved oxygen, FCA: flood control area

6 Rehabilitation processes in the Zeeschelde

The primary goal for the rehabilitation of the Zeeschelde is to re-establish the estuary's autogenic processes (Van den Bergh *et al*., 2005). These authors focused on the ecological rehabilitation of the estuary whereby measures were defined for different zones. We link the proposed measures with the habitat needs defined for fish.

Adding space improves connectivity and habitat diversity

The preferred alternative to the updated Sigmaplan (see introduction) couples ecological rehabilitation and sustainable nature with flood control measures and navigation requisites (Couderé *et al*., 2005). This plan should be executed by the year 2030. It includes the creation of 1400 ha tidal wetland through managed realignment, 1100 ha tidal wetland under reduced controlled tide in flood control areas (FCA-CRT) 1500 ha of 'winter bed' for the upper reaches and 2000 ha of non tidal wetlands, 1000 of which in flood control areas (FCA-Wetland) (Fig. 4.3, Table 4.2).

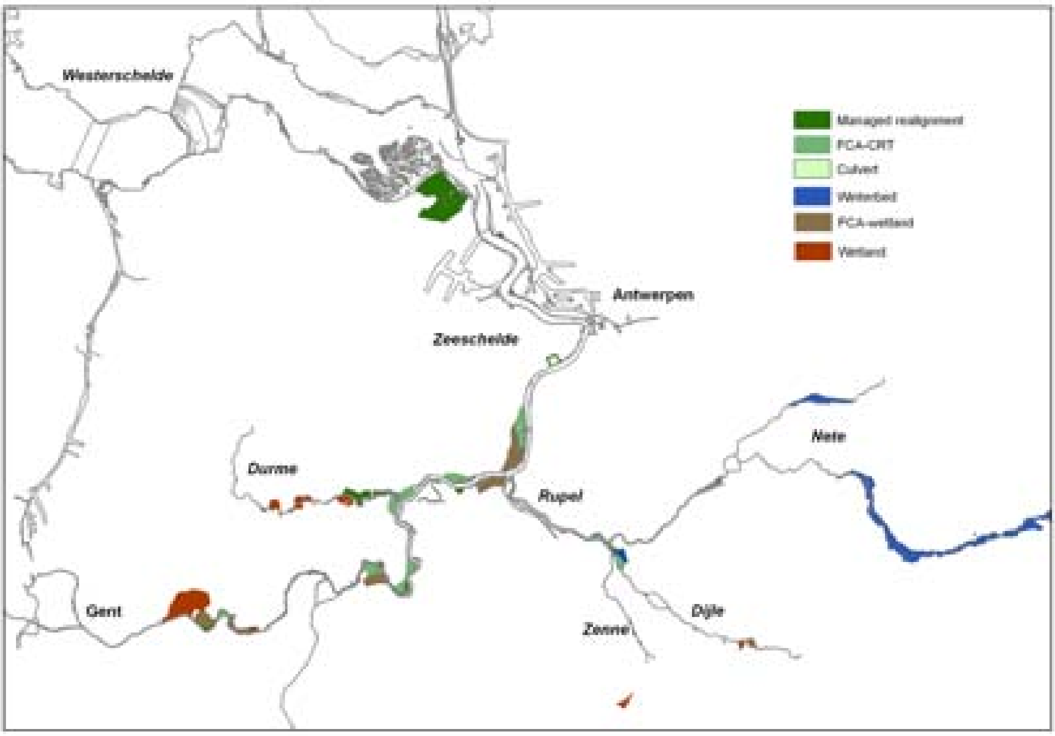

Figure 4.3: Ecological restoration measures for the preferred alternative to the updated Sigmaplan.

Table 4.2 Ecological rehabilitation measures relative for fish in the Zeeschelde.

Zone	rehabilitation	surface (ha)
mesohaline	connection	83
mesohaline	realignment	1124
oligohaline	realignment	43
oligohaline	FCA-wetland	187
oligohaline	FCA-CTR	577
freshwater	realignment	328
freshwater	FCA-wetlands	393
freshwater	FCA-CTR	281
freshwater	FCA	279
freshwater	wetland	762
tributaries	FCA-wetland	231
tributaries	FCA-CTR	89
tributaries	FCA	271
tributaries	wetland	109
tributaries	winter bed	1347

Realignment will create dynamic estuarine habitats. In chapter 6 we record the use of mudflats and tidal marshes by juvenile species. The presence of rivulets on the mudflats, the position of the marsh creeks in the tidal frame and the dimension are important characteristics that enhance the use by fish. In addition presence of permanent pools will increase the nursery function of the creeks.

Flood control areas (FCA) constitute extra storage capacity for water during storm surges during which fish from the Zeeschelde enters the floodplain. Pas *et al*. (1998) showed that Tielrodebroek, a flood control area of about 90 ha at the mouth of the River Durme, functions as a spawning and nursery area for some freshwater species that remain in the brooks. These areas are now subject to agricultural activities which can have a negative effect on the fish. The creation of a natural wetland (FCA-wetland) with permanent pools will reduce impacts from fertilisers and pesticides and enhance nutrient cycling and water retention, which in turn will increase the carrying capacity of the floodplain. Research is needed to optimise the connectivity between the floodplain and the river.

A flood control area under the influence of a controlled reduced tide (FCA-CRT) creates marshes that are less dynamic than those along the river. The realisation of creeks and permanent shallow pools constitutes an ideal spawning and nursery habitat for many fish species. Here again extra research is needed to optimise the exchange possibilities between the floodplain and the estuary.

Winter beds are for freshwater species in tributaries an essential spawning and nursery habitat. Grift (2001) observed that fish rapidly colonized newly created floodplain water bodies in the river Rhine.

Reconnected oxbow lakes have a beneficial value for the riverine fish community, since they provide a habitat that is better suitable for 0-group fish than the main stream. In addition they form important spawning and nursing areas for rheophilic species. Restoring the interaction between the river and abandoned river meanders (Oude Durme, Oude Schelde) is to be considered as beneficial for rheophilic species.

The creation of shallow tidal areas in the mesohaline zone improves oxygenation since they positively influence the surface-to-volume area (Chapter 5). Intertidal, shallow habitats are essential to reinstate physical and chemical processes and hence increase the estuary’s self-purification and filtering capacity that sustains water quality (Van den Bergh *et al.*, 2005;

Lotze *et al.*, 2006). The creation of space will reduce the current velocity and the associated effects such as erosion and turbidity and will as such also increase intertidal habitat surface. Brys *et al*. (2005) and Van Braeckel *et al*. (2006) suggest that the only sustainable way to protect tidal marshes against destructive erosion is by facilitating the creation of mudflats combined with shallow zones, although new marshes can only be created taking into consideration appropriate geomorphological conditions (Van Braeckel *et al*., 2006).

In addition by increasing the surface the carrying capacity of the estuary is enhanced. Indeed in chapter 2 we report a high abundance of marine migrants such as herring, seabass, estuarine species (Gobiidae) and diadromous fish in the mesohaline zone. Juveniles are recorded in the tidal marshes seeking shelter and food (Chapter 6). At present the tidal marshes in the mesohaline zone are not used as nurseries because there are too few permanent pools. It is therefore essential that the realignment creates areas that are permanently flooded so that these can function as a spawning place for estuarine species and as a nursery for marine migrants and diadromous species. In chapter 6 we suggest as well the importance of rivulets on the mudflats and the position of the marshes and creeks in the tidal frame for foraging fish.

Connecting pools (83 ha) will only be beneficial for fish if these can return to the main river. Research is needed to investigate the passage of marine and estuarine species through culverts.

Since 2007 species richness and fish abundance have increased in the oligohaline zone. We recorded the presence of mainly freshwater species but also diadromous, estuarine and marine species (Chapter 2). Tidal marshes in this zone are visited by juvenile freshwater and diadromous species such as eel and flounder (Chapter 6). The creation of wetland (187 ha) and estuarine nature (620 ha) will be mainly used by freshwater species and diadromous species if they can enter the wetland and flood control areas (FCA). Again care should be taken that the wetland has permanent flooded areas and that fish can occasionally migrate between the main river and the wetland. This can be done by installing tidal flap gates. However, research is needed to define optimal gate constructions. The installation of tidal flap gates in Tielrodebroek will increase its carrying capacity. In the Durme the realignment of the Bunt will also be a gain for fish. Care should also be taken for the elevation so that the created tidal mudflats and marshes can be used in an optimal way by freshwater species as spawning, nursery and feeding places and by occasionally visiting estuarine species as resting and feeding places.

Freshwater species are abundant in the freshwater zone. We recorded diadromous and occasionally estuarine species as well as marine migrants (Chapter 2). The tidal marshes are used as juvenile habitat for freshwater species and diadromous species such as flounder and eel (Chapter 6). Due to their elevation in the tidal frame and the presence of only a few permanent pools, the nursery function is not assured. As already mentioned, the realisation of an experimental FCA-CRT such as Lippenbroek in the freshwater zone, proved to be beneficial for fish since it can be incorporated as nursery and spawning places, provide shelter and resting areas and act as feeding grounds (Fig. 4.3) (Simoens *et al*., 2007). Diadromous species such as flounder (semi-catadromous) also use the Lippenbroek as a nursery. We observed that fish entered via the outlet rather than using the inlet sluices (unpublished results). Rheophilic B species benefit from this connected flood area, e.g. in 2008 we caught one burbot (*Lota lota*) in Lippenbroek (unpublished data). Occasionally marine migrants and estuarine species were caught in Lippenbroek or in the adjacent river. More research is needed to enhance or facilitate fish passage through the tidal flap gates. These results indicate that the creation of attainable wetlands (1292 ha) and estuarine nature (471 ha) will be beneficial for freshwater and diadromous fish, as well as for occasional estuarine and marine migrants. For the realisation of the latter habitat one must again take into account the elevation of marshes in the tidal frame.

Physical and chemical barriers preventing migration of diadromous species should be removed or bypassed (Stevens *et al.*, 2009). Solving fish migration problems may not primarily depend on an engineered fish passage, but rather on natural solutions such as restoration of old meanders or the creation of a nature-like bypass channels. This will avoid change of substrate which can have an effect on habitat availability of certain species (Mouton *et al*., 2007). The basic steps to create adequate fish passages in lowlands are explained comprehensively by Kroes *et al.* (2006).

To avoid strong riverine peak discharges, retention areas upstream should be created, e.g. through dike relocations. Such restoration practices are planned in the Nete basin a tidal tributary creating about 1347 ha of floodplain (Fig. 4.3). Upstream river rehabilitation and mitigation measures will improve the lateral connectivity of main channels to an ecotone complexity beneficial for fish (Angermeier & Karr, 1983; Sindilariu *et al.*, 2006). In this tributary Habitats Directive annex species such as bitterling (*Rhodeus sericeus*), burbot (*Lota lota*) and weatherfish *(Misgurnus fossilis*), as well as other freshwater and diadromous species, are recorded and will use the wetland as spawning, nursery and feeding area. The realisation of estuarine nature and wetlands in the other tributaries (Zenne and Dijle) will be beneficial for rheophilic B species.

It is important that all planned rehabilitation measures are realised in order to create sufficient diverse habitat for the fish communities in the Zeeschelde. In this phase separate plans are under environmental impact assessment (EIA). The risk exists that they will all be subject to "minor changes" so that the total picture will no longer fit. It is essential that monitoring programmes should be planned and executed in order to assess the impact of the effectuated rehabilitation actions and to define the carrying capacity of the created habitats.

7 Concluding remarks

Essential habitat requirements for estuarine fish species at three guild levels, as defined by Franco *et al*. (2008), have been defined. Some of the described ecological goals are applicable in different guilds. If the proposed ecological goals are achieved then different species representing different links within the trophic chain will be present resulting in an undisturbed and complete trophic web as it will also assure an undisturbed reproductive mode functional group.

The proposed ecological goals are only qualitative and therefore further research is needed to provide (semi-)quantitative ecological goals. In addition, the carrying capacity of each habitat type within the different estuary zones should be calculated and mechanisms presented for the recovery of the system (extended as the DPSIRR approach, Drivers, Pressures, State, Impact, Response and Recovery; Elliott *et al*., 2007a) in order to protect and maintain a good carrying capacity. Finally a fish-based evaluation system should be developed to assess the status of an estuary and to evaluate the impact of the implementation of mitigating measures (Chapters 7 and 8).

In conclusion, it is emphasised that a management structure is required which recognises the ecological goals of organisms, such as fishes, and then links those to the management responses needed to protect and maintain (and where necessary restore) an estuarine system, and those parts of its catchment and marine area, required by fishes in order to complete their life cycle.

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Chapter 5

Modelling the migration opportunities of diadromous fish along a gradient of salinity and dissolved oxygen in the Zeeschelde estuary (Belgium)

Joachim Maes, Maarten Stevens & Jan Breine

Abstract

The relationship between poor water quality and migration opportunities for fish remains poorly documented, although this knowledge is essential for a correct implementation of the EU water legislation. In this chapter, we model environmental constraints that control the movements of anadromous and catadromous fish populations that migrate through the tidal watershed of river Schelde, a heavily impacted estuary in Western Europe. Local populations of sturgeon, sea lamprey, sea trout, Atlantic salmon, houting and allis shad were essentially extirpated around 1900. For remaining populations (flounder, three-spined stickleback, twaite shad, thinlip mullet, European eel and European smelt), a data driven logistic model was parameterized. The presence or absence of fish species in samples taken between 1995 and 2004 was modelled as a function of temperature, dissolved oxygen concentration, river flow and season. Probabilities to catch individuals from all diadromous species but three-spined stickleback, increased as a function of the interaction between temperature and dissolved oxygen. The hypoxic zone situated in the freshwater tidal part of the estuary was an effective barrier for upstream migrating anadromous spawners since it blocked the entrance to historical spawning sites upstream. Similarly, habitat availability for catadromous fish was greatly reduced and restricted to lower brackish water parts of the estuary. The model was applied to infer preliminary dissolved oxygen criteria for diadromous fish, to make qualitative predictions about future changes in fish distribution given anticipated changes in water quality and to suggest necessary measures with respect to watershed management.

Keywords: fish migration, logistic model; dissolved oxygen, water pollution, freshwater tidal reach, anadromy, Schelde estuary

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1 Introduction

Worldwide, river fragmentation is primarily responsible for the decline of populations of migrating fish (Masters *et al*., 2006). In particular, many anadromous fish species, which must migrate to fresh water in order to reproduce, are endangered since they are no longer able to reach their natural spawning sites (Masters *et al*., 2006). Additionally, pollution of rivers effectively prevents upstream or downstream movements and blocks access to spawning grounds. Land use practices, habitat deterioration or removal and exploitation of commercially interesting populations, have contributed to the decline of diadromous populations.

Historically, populations of Acipenseridae, Salmonidae, Osmeridae and Clupeidae migrated into freshwater rivers along the north-eastern board of the Atlantic to spawn. Conversely, European eel *Anguilla anguilla,* as well as a number of species that are considered facultative catadromous, such as the flounder *Platichthys flesus* and the thinlip mullet *Liza ramado*, migrated from the freshwaters out to the North Atlantic to spawn. Throughout Europe, many of the anadromous populations are in decline or extirpated (De Groot 2002; Masters *et al*., 2006). Human impacts on species that migrate over considerable distances do not stop at borders. Hence, besides national programmes, a number of international initiatives has been established to halt the decline of anadromous fish species. The most important legal framework in Europe with respect to the protection of anadromous species is the European habitats directive 92/43/EEC, which lists the anadromous species under its annex 2 as species of community interest whose conservation requires special areas of conservation. In addition, *Acipenser* species as well as *Coregonus oxyrhynchus* are listed as annex IV species which need strict protection, while other anadromous species are listed in annex V as species whose taking in the wild and exploitation may be subject to management measures.

The status of the migrating fish fauna of the watershed of river Schelde basin, a medium-sized lowland river basin in West Europe, is the focus of this study. The river Schelde, with origin in France, main drainage basin in Belgium and delta in The Netherlands, is characterized by centuries of serious pollution, land claim and habitat quality deterioration. The exponent of the environmental degradation was a virtually anoxic zone during the 1970's situated just above the freshwater saltwater boundary (Van Damme *et al*., 2005). Since then, and due to efforts to better treat wastewaters, average dissolved oxygen (DO) in the river increased by about 1 mg l^{-1} per decade. Yet, the Schelde basin still has important nature values and potentials, particularly for migrating fish species. The estuary has a complete salinity gradient including extensive freshwater, brackish and salt marshes. Tides penetrate much further in the river than salt water and influence some of the major tributaries of river Schelde. It follows that, because of the absence of flow regulating constructions, unique opportunities exist for migratory fish populations in that watershed.

At present the levels of DO are increasing, which has resulted in a recovery of fish populations in the river, particularly in its estuary (Maes *et al*., 2004a,b; Van Damme *et al*., 2005). Also diadromous fish populations gradually increase in size. This recovery is well illustrated by the spatio-temporal distribution of the twaite shad *Alosa fallax* in the river Schelde (Maes *et al*., 2008). The species reoccurred in the river since 1996 and catches gradually increase in upstream direction.

The present situation and the expected improvement of migrant fish populations in the river Schelde and its basin form the central theme of this chapter. First we present up to date information of the present status and distribution of the diadromous fish fauna of the tidal watershed of the river Schelde. Next, we model water quality constraints that control the distribution and movements of adult anadromous spawners and juvenile catadromous foragers between the North Sea and upstream spawning and nursery grounds, using a generalized linear model. The models are subsequently applied to infer preliminary DO criteria for diadromous fish, to make qualitative predictions of the distribution of fish given a further improvement of water quality, and to suggest necessary measures with respect to watershed management.

2 Material and methods

2.1 Study area

River Schelde has its origin in the north of France and discharges into the North Sea near Vlissingen (The Netherlands). It is a lowland river with a total length of 355 km and a fall of about 100 m. The catchment area is approximately 21000 km^2 with a population of 10 million inhabitants (Van den Bergh *et al.,* 2005). This study focuses on the tidal part of the watershed, presented in figure 5.1. The tidal part of the river is called Westerschelde in The Netherlands and Zeeschelde in Belgium. The lower estuary (Westerschelde) is characterized by flood and ebb channels, separated by sandy or muddy intertidal areas. Due to the funnel shape of the lower estuary the maximum vertical tidal range is about 100 km upstream, in the freshwater zone (Van den Bergh *et al*., 2005). The tidal influence thus extends much further land inward than the freshwater-saltwater boundary (Fig. 5.1). As a result, an extensive freshwater region under tidal influence is present. The tidal excursion goes as far as Gent, 160 km from the river mouth, where the tide is stopped by sluices (Fig. 5.1). Also, the tributaries Durme, Rupel, Nete, Kleine Nete, Grote Nete, Dijle and Zenne are under tidal influence and are therefore considered as an integral part of the estuary (Fig. 5.1).

2.2 Fish sampling

We collected fish samples at four stations along the river Schelde (Fig. 5.1) using paired fyke nets. A fyke net is essentially a fish trap consisting of a long bag net distended by hoops, into which fish can pass, without being able to return. Paired fyke nets consist of two 7.7 m fykes between which an 11 m lead net was suspended. The first hoop of each fyke is horse-shoe shaped with a basis of 120 cm and a diameter of 80 cm. Fish can be removed on the other end of the fyke where the mesh size is 8 mm. The fishing gear was placed parallel to the river border on the tidal mudflats during low water. Fish that encounter the leader net during high water are guided into the fykes. Hence, both fish movements as well as mesh size influence the selectivity of fyke nets.

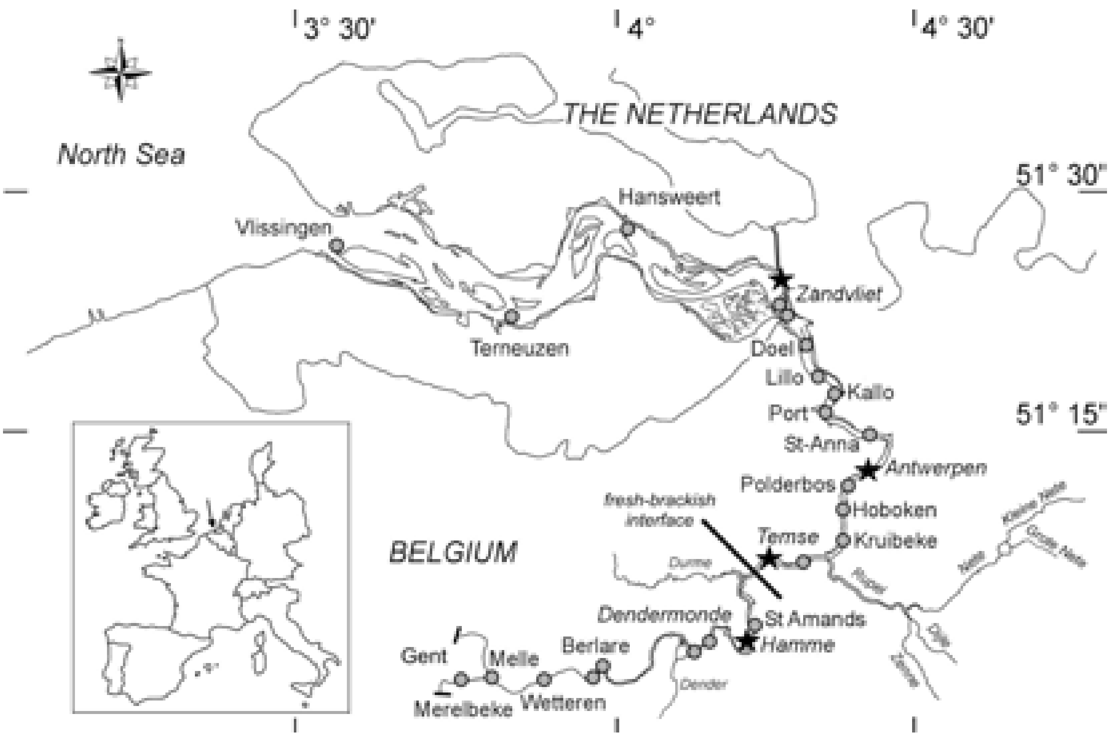

Figure 5.1: Map of the tidal part of the Schelde basin indicating the river Schelde and tributary rivers (Rupel, Nete, Dijle, Zenne, Dender, Durme). End point of the rivers represents the tidal limit. The map shows the fish sampling stations (stars) and the water quality sampling stations (circles). A flow gauge is operated by Rijkswaterstaat at station Lillo. Water quality sites and fish sampling sites are spatially not matched. To feed the statistical model, we used water quality data of the water quality stations most close to each fish sampling station.

A total of 112 fish samples was collected. The sampling design was spatially and temporally unbalanced, mostly as a result of annual changes in the allocation of research funding to fish monitoring. Spatially, most samples were taken at Zandvliet (51), 26 were taken at Antwerpen, 25 at Temse and 10 at Hamme. Samples were taken each year from 1995 till 2004 and between March - October. In 2005, 7 samples were taken, two at each site except only one for the station at Zandvliet, and these were used to validate the statistical models. All field work was done by trained fish biologists using a standardized procedure to assure the quality of the work. Fish captured were identified on site using a field guide (Nijssen & De Groot, 1987). Quality assurance of those identifications was performed by occasional cross-examination in the laboratory, especially of small sized specimens. Fish data recorded included species-specific frequencies, individual total lengths (± 1 mm) and wet weights (± 1 g).

2.3 Environmental data

Temperature, dissolved oxygen and freshwater flow were used as predictor variables in the statistical models as well as to infer spatio-temporal distribution plots of fish species. Water temperature (°C) and oxygen concentration (mg l^{-1}), both measured at the surface of the river, were derived from two different water quality databases. Data for the Westerschelde, situated in The Netherlands, were derived from Waterbase, a publicly available internet resource of the Dutch traffic and waterways ministry (Rijkswaterstaat, 2006). Data for the Zeeschelde, situated in Belgium, were downloaded from the Flemish Environmental Agency internet site (Flemish Environmental Agency, 2006). Freshwater flow rate data ($m^3 s^{-1}$) were obtained from the Dutch Rijkswaterstaat database (Rijkswaterstaat, 2006), based on a flow gauge situated in Lillo (Fig. 5.1). Flow and oxygen concentrations were measured monthly. Temperature was measured daily in The Netherlands and monthly in Belgium. All stations are shown in figure 5.1. Temperature and DO data were acquired for three sampling stations in the Westerschelde and for 19 stations in the Zeeschelde, while a flow gauge, operated by Rijkswaterstaat, is situated in Belgium. As a result, data of temperature and DO vary both monthly and spatially while for flow, only one monthly measure is available for the entire river gradient. Both agencies assure the quality of their data and QA documents can be obtained upon request.

It is important to note that fish samples and water quality samples did not match in time and space. In order to construct statistical models, we used environmental data from stations that were sampled both spatially and temporally as close as possible to each of the fish sampling stations involved. The distance between fish sampling stations and water sampling stations averaged 2.5 km and varied between 0.9 km for the fish sampling station at Hamme and 3.3 km for the station at Antwerpen. These distances are considerably lower than the tidal excursion which extends between 10 and 15 km.

2.4 Statistical models

The presence or absence of anadromous spawners and catadromous foragers was modelled using logistic regression (Hosmer & Lemeshow, 2000). Only species that occurred with sufficient frequency in the samples were retained in this analysis (Table 5.1). In the model, a binary response variable (presence / absence) was expressed as a linear combination of a set of candidate predictor variables through a logit link function. The set of continuous predicted variables included dissolved oxygen concentration, temperature, flow, the product between temperature and dissolved oxygen and the square of temperature. Temperature was entered as a second order polynomial model in order to account for a bell shaped response. Possible interactions between DO and temperature were accounted for by considering the product between these two variables. Further, one categorical predictor variable was entered in the model accounting for seasonal effects. Statistically, this variable was encoded in three binary variables assuming a value of 1 for samples taken in the designated season and zero otherwise. The complete model design is:

71 $$\text{logit } P = \log_e [P/(1-P)] = [\beta_0 0+\beta_{1J} (\text{season})]+ \beta_2 \times DO+\beta_3 \times T+\beta_4 \times F+\beta_5 \times T^2+\beta_6 \times T \times DO+\varepsilon$$

where P is the probability to capture a species in a fish trap over a 24 h period; DO represents the dissolved oxygen concentration (mg l^{-1}), F is the monthly averaged river flow (m^3 s^{-1}) and T is the ambient surface water temperature (°C). ε is the error term of the model. The model's intercept is given by the term $\beta_0+\beta_{1J}$ (season), where season represents the three binary variables spring, summer and fall each with slope β_{1J}. As a result, the categorical predictors either increase or decrease the model intercept with β_{1J} but the different slopes $\beta_{2,..,6}$ of the continuous predictor variables remain unaffected. The intercept β_0 represents the model for winter samples, for which the three binary variables are zero.

Initially, full models for each species were fitted using the maximum likelihood statistic that is available in STATISTICA 7 (Statsoft). Next, we used the procedure of best subsets in order to fit all possible models (N=63). From this list, we selected the model with the lowest Akaike's information criterion (AIC) (Johnson & Omland, 2004). This minimal adequate model was used in further model applications.

Table 5.1: Frequencies of diadromous fish species at the different sampling stations, according to figure 5.1. Species were subdivided into two groups: common species for which regression models were produced and rare species, which were only sporadically present in the fishing gear. A total of 112 samples, collected between 1995 and 2004 was broken down over 4 sampling sites. In 2005, seven samples were collected and used as validation of the regression models. These frequencies are presented between brackets.

Species	**Zandvliet**		**Antwerpen**		**Temse**		**Hamme**	
Number of fish samples	**51**		**26**		**25**		**10**	
	present	**absent**	**present**	**absent**	**present**	**absent**	**present**	**absent**
Species with regression models								
European eel	42 (0)	9 (1)	17(2)	9 (0)	4 (2)	21 (0)	1 (2)	9 (0)
European smelt	36 (1)	15 (0)	0 (0)	26 (2)	0 (0)	25 (2)	0 (0)	10 (2)
Flounder	51 (1)	0 (0)	8 (2)	18 (0)	2 (0)	23 (2)	0 (0)	10 (0)
Thinlip mullet	25 (0)	26 (1)	1 (0)	25 (2)	0 (0)	25 (2)	0 (0)	10 (2)
Three-spined stickleback	13 (0)	38 (1)	9 (1)	17 (1)	2 (1)	23 (1)	3 (1)	7 (1)
Twaite shad	22 (0)	29 (1)	1 (0)	25 (2)	0 (0)	25 (2)	0 (0)	10 (2)
Species without regression models								
Atlantic salmon	1	50	0	26	0	25	0	10
Brown trout	6	45	0	26	0	25	0	10
River lamprey	2	49	2	24	0	25	0	10
Sea lamprey	1	50	0	26	0	25	0	10

Model goodness-of-fit was evaluated using the model deviance defined as – 2 x (L_M – L_S) where L_M denotes the maximized log-likelihood value for the model of interest, and L_S is the log-likelihood for the saturated model (a saturated model has *n* parameters and fits *n* observations perfectly). Under the null hypothesis that the logistic model is true, the deviance is $\chi 2$-distributed. Inference for single parameters is based on the Wald statistic. The null hypothesis is that a single parameter β_i equals 0.

Contrary to the ordinary least squares statistic, the maximum likelihood statistic does not result in a typical R^2 value. Alternatively, the model performance was evaluated by assessing the percentage of correctly classified occurrences and non-occurrences. Hereto, we used p = 0.5 as cut-off value. The final models were additionally evaluated by comparing the model predictions with the presence and absence data of the considered fish species during the field campaign of 2005. Again, we used p = 0.5 as cut-off value, i.e. predicted probabilities >0.5 were considered as present.

2.5 Model applications

We used the minimum adequate logistic regression models in three different applications. Firstly, we used the regressions in order to predict the occurrence of anadromous and catadromous species along the entire river gradient. Probability values P can be calculated based on equation 2, where e is the natural exponent:

71 $$P = e^{\text{logit}(P)}/[1+e^{\text{logit}(P)}]$$

Monthly water quality data of 22 stations (Fig. 5.1) were entered in equation 2 in order to produce species specific spatially and temporally explicit probability plots. For simplicity, we used only environmental data for the year 2003 to demonstrate the applicability of the model. The contour plots were used to better visualize the migration opportunities of diadromous fish in the Zeeschelde.

In a second application, we tested the effect of several environmental scenarios with respect to increased oxygen concentration in the middle section of the estuary (between km 80 and km 100, Fig. 5.1). For this river section, we modified the DO input data series of 2003 by assuming either an increase in DO by 10% or by assuming minimum dissolved oxygen concentrations (5 mg l^{-1} and 6 mg l^{-1}). A threshold of 5 mg l^{-1} corresponds to a legally established DO minimum for surface waters in Flanders (Belgium) (VLAREM II, 1995).

In a recent report, the European Commission concludes that the establishment of DO criteria for fish, amongst others, is a prior research theme in order to fully implement the European Water Framework Directive (Heiskanen & Solimini, 2005). Therefore the final application of the statistical model is an attempt to infer preliminary DO criteria for migrating fish. Such minimum requirements with respect to dissolved oxygen for fish have yet to be established in Flanders, an autonomous region of Belgium competent for environmental policies (VLAREM II, 1995). In this study we applied the models in order to calculate the minimum required concentration of dissolved oxygen to yield a capture probability of at least 50%. This calculation was performed for each species separately assuming temperature and freshwater flow conditions averaged over the season of maximum occurrence in the samples for the period 1994-2005.

3 Results

3.1. General fish catch statistics

A total of 112 fish samples was taken between 1995 and 2004 at four different sampling stations in the tidal Schelde capturing 10 diadromous fish species (Table 5.1). Four anadromous species occurred irregularly in the fyke nets: two lamprey species (river lamprey *Lampetra fluviatilis* and sea lamprey *Petromyzon marinus*) and two salmonids (Atlantic salmon *Salmo salar* and sea-run brown trout *Salmo trutta*). These species were not further considered in the statistical models and applications. Thinlip mullet, European smelt (*Osmerus eperlanus*) and twaite shad returned more frequently in the samples but their distribution was limited to the brackish reaches of the estuary downstream the freshwater saltwater front situated nearby Antwerp (Fig. 5.1, Table 5.1). Flounder, eel and stickleback (*Gasterosteus aculeatus*) moved further upstream and the latter two species occurred throughout the study area (Table 5.1).

3.2 Logistic regression models

Logistic regression with the presence or absence of species in fyke nets as dependent variable and ambient oxygen concentration, temperature, river flow and season as independent predictor variables yielded statistically significant models (Table 5.2). Summary statistics for the environmental variables that were used as predictors in the regression models are given in Table 5.3. In all cases, the minimal adequate models had a lower AIC than the full models (Table 5.2). Full models had a higher percentage of correct classifications when field data were compared with model predictions, but in general the Wald statistics for individual parameters estimated when fitting a full model although the data were not significant. In contrast, minimal adequate models were statistically significant (deviance < Chi squared for a given number of degrees of freedom at a significance level of $p = 0.05$) and yielded significant regression coefficients as well.

All species but three-spined stickleback showed a significantly increasing response to the interaction of DO and temperature (Table 5.2). It follows that the capture probability in the fyke nets increased when the product of DO and temperature increased. For European eel this interaction was negatively corrected for increasing DO concentrations (Table 5.2). For smelt, the square of temperature negatively influenced the interaction effect (Table 5.2). The probability to capture twaite shad decreased significantly when freshwater flow increased (Table 5.2). The model for stickleback was different from the other models in that it was the only for which the categorical predictor (season) remained in the model as explaining variable (Table 5.2).

This fixed seasonal pattern was negatively influenced by the square of temperature, suggesting that stickleback avoided the summer warm waters of the estuary.

In general, the reduced models performed well in that the percentage of correctly classified occurrences and non-occurrences was relatively high (Table 5.2). This is illustrated in figure 5.2 which plots the field observations against modelled probabilities along a temperature and DO gradient. The model for thinlip mullet is the only exception with 3.8% of the presence correctly classified (Table 5.2). We used the models to predict the presence or absence of the considered species in fyke net observations made in the year 2005, outside the modelled period. Again, models correctly classified between 57% and 100% of the cases (Table 5.2).

Table 5.2: Logistic regression models. For each species, we present the model diagnostics, the explicit description of the minimal adequate model and the final regression parameters of the minimal adequate model. Model diagnostics are Akaike's information criterion (AIC) for the full model with all predictor variables (AIC_{FM}) and for the minimal adequate model (AIC_{MM}). For the minimal models, we present the model deviance (Dev_{MM}) together with the Chi squared statistic ($\chi^2_{p=0.05}$) for the appropriate degrees of freedom (df). The full model has 103 degrees of freedom. Recall that significant models have a Deviance < χ^2. A second set of diagnostics are the percentage of correct classifications of presence and absence and the percentage of correctly predicted occurrences of the validation data set. Final model parameters (with standard error) are presented at the bottom of the table.

Species	**$AIC_{FM,}$**	**AIC_{MM}**	**Dev_{MM} ($\chi^2_{p=0.05}$, df)**	**correct absent**	**correct present**	**correct validation**	**Minimal Model**
Flounder	114.4	106.4	102.4 (135.4, 110)	80.4%	82.0%	71.5%	logit p = $\beta_0 + \beta_6 \times T \times DO$
Stickleback	112.5	106.7	96.7 (132.1, 107)	87.1%	48.1%	85.7%	logit p = $[\beta_0 + \beta_{1J}(\text{season})] + \beta_5 \times T^2$
Thinlip mullet	117.2	112.1	108.1 (135.4, 110)	91.9%	3.8%	100%	logit p = $\beta_0 + \beta_6 \times T \times DO$
Twaite shad	80.8	77.3	71.3 (134.4, 109)	95.5%	52.2%	100%	logit p = $\beta_0 + \beta_4 \times F + \beta_6 \times T \times DO$
Eel	150.0	142.6	132.6 (134.4, 109)	60.4%	70.3%	57.1%	logit p = $\beta_0 + \beta_2 \times DO + \beta_6 \times T \times DO$
Smelt	118.3	109.7	103.7 (134.4, 109)	86.8%	58.3%	100%	logit p = $\beta_0 + \beta_5 \times T^2 + \beta_6 \times T \times DO$

Model parameters	**β_0**	**β_1 Spring**	**β_1 Summer**	**β_1 Fall**	**β_2**	**β_3**	**β_4**	**β_5**	**β_6**
Flounder	-2.688 (0.556)								0.049 (0.009)
Stickleback	0.339 (0.693)	0.803 (0.441)	0.609 (0.755)	-0.243 (0.511)				-0.008 (0.003)	
Thinlip mullet	-2.749 (0.554)								0.022 (0.006)
Twaite shad	-3.178 (1.195)						-0.013 (0.007)		0.040 (0.010)
Eel	-0.541 (0.461)				-0.269 (0.132)				0.032 (0.009)
Smelt	-2.517 (0.614)							-0.006 (0.002)	0.045 (0.010)

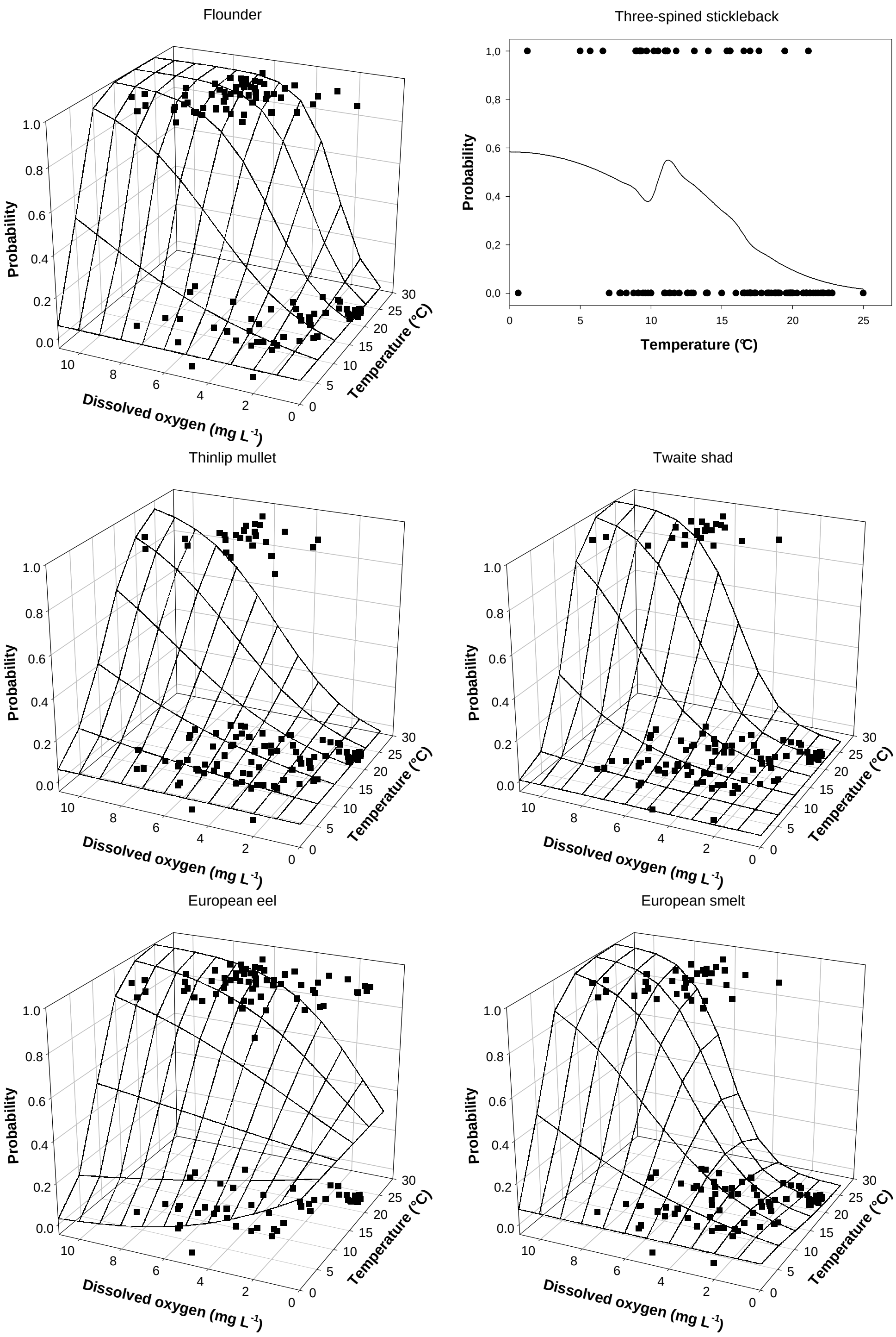

Figure 5.2: Modelled (gridded surface) versus observed (dots) fish occurrence (presence/absence) of six species as a function of temperature and dissolved oxygen. Fish occurrence is the presence or absence of fish captured in fyke nets at four sampling sites in the Zeeschelde between 1995 and 2004 (N = 112). The modelled surface is based on the minimal adequate models based on equation 1 using the fitted parameters as in Table 5.2. We assumed average river flow conditions (Table 5.3). DO was not retained as explaining predictor variable in the stickleback model so only the binary response to temperature was given.

Table 5.3: Summary statistics. Average (standard deviation) of the continuous predictor variables used in the logistic regression analysis. Sampling stations are presented in figure 5.1.

Sampling station	**Zandvliet**	**Antwerpen**	**Temse**	**Hamme**
Number of samples	**51**	**26**	**25**	**10**
Temperature (°C)	16.1 (4.6)	15.2 (5.9)	14.8 (5.9)	14.0 (5.6)
Dissolved oxygen (mg l^{-1})	5.8 (1.7)	2.7 (1.9)	2.8 (2.0)	5.4 (2.9)
River flow (m^3 s^{-1})	110.5 (60.0)	132.3 (70.4)	134.8 (71.0)	108.8 (55.8)

3.3 Model applications

3.3.1 Spatio temporal probability plots

The minimum adequate logistic regression models were used in three different applications. Firstly, we plotted capture probability distributions of the different species for the tidal part of river Schelde adopting environmental conditions for the year 2003 (Fig. 5.3). Probabilities to capture fish in 24 h samples based on fyke nets are presented in a two dimensional plane where distance to sea represents a spatial axis, and time in months a temporal axis. In agreement with the results of the model fitting procedure, spatial patterns dominate above seasonal effects due to the presence of spatial gradients of temperature and, particularly, of dissolved oxygen concentration in the estuary. The probability plots for almost all species suggest that capture probabilities are predicted to reach a minimum in the oxygen poor zone in the middle estuary while modelled probabilities are higher in the lower and upper parts of the estuary. For stickleback, a seasonal pattern emerged with maximum predicted probabilities throughout the estuary during early spring. The predicted distribution patterns based on logistic regressions with environmental variables as predictors clearly show how the low DO zone in the middle part of the tidal Schelde may interfere with the migration opportunities for fish that necessarily move between the ocean and freshwater habitats in order to successfully complete their life history.

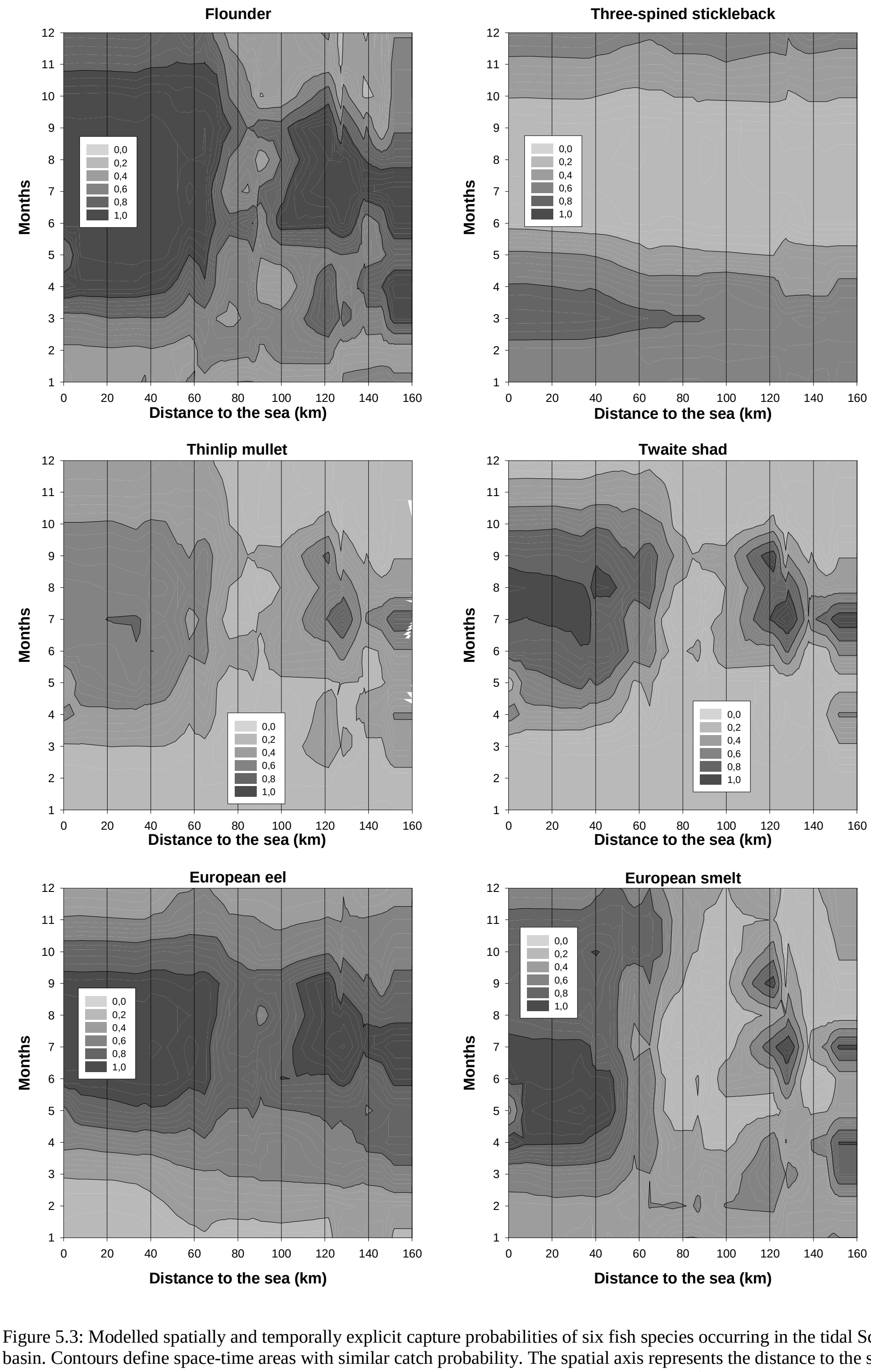

Figure 5.3: Modelled spatially and temporally explicit capture probabilities of six fish species occurring in the tidal Schelde basin. Contours define space-time areas with similar catch probability. The spatial axis represents the distance to the sea (river mouth at 0 km and most upstream area at 160 km). The temporal axis represents the months of the year (Jan. = 1, …, Dec. = 12). Probabilities vary between 0 and 1. Probabilities were calculated using minimal adequate logistic regression models according to equation 1 using parameters as in Table 5.2.

3.3.2 Environmental scenarios

In a second application, we focussed on an area with poor water quality between km 80 and km 100 where untreated waste water of the Brussels capital region reaches the Zeeschelde. We assumed three different scenarios in terms of DO relative to the 2003 situation: a 10% increase and an increase to at least 5 mg l^{-1} and 6 mg l^{-1}, respectively (Fig. 5.4). Since DO as predictor variable was retained in all models but one, mostly in the interaction term with temperature, it follows that the probability to capture fish in nets was predicted to increase if the concentration of DO increased (Fig. 5.4). This was especially evident for twaite shad and smelt. The model suggests that if water quality meets the baseline requirements, which are legally adopted by the Flemish Region, fish captures of all species are expected to increase substantially. Only for three-spined stickleback, we were unable to relate an increment in DO to increased probability of occurrence.

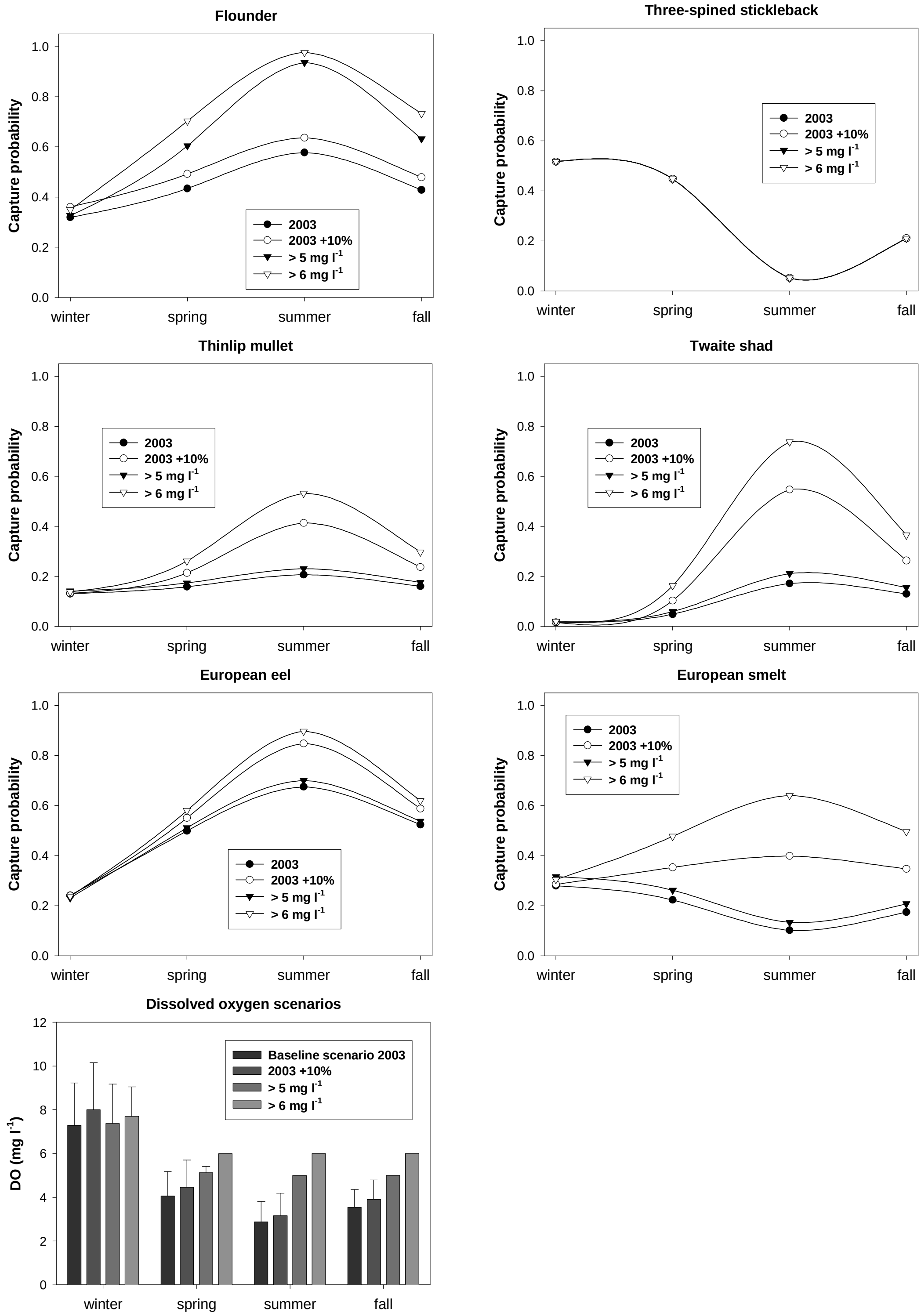

Figure 5.4: Environmental scenarios. If oxygen concentration of the Zeeschelde between km 80 and km 100 is increased by 10% or if a minimum DO concentration of 5 or 6 mg l^{-1} is imposed, capture probabilities for most diadromous species increase. Probabilities are presented as seasonal averages based on monthly calculations. The barplot compares the average seasonal DO concentration for the baseline scenario based on values for 2003 with three other scenarios. The error bars are standard deviations.

3.3.3 DO criteria

The logistic models were finally applied to infer minimum river DO concentrations at which the probability to capture a species was at least 50% during the season of peak migration (Table 5.4). Under these assumptions, thinlip mullet seemed to be the most sensitive species with a 50% probability of occurrence in the nets when DO reaches 6.2 mg l^{-1} given a temperature of 20.5°C. Smelt and twaite shad needed DO concentrations of at least >5 mg l^{-1} while there is a 50% probability to catch flounder if DO was at least 2.7 mg l^{-1}. Eel was the most tolerant species. DO was absent as variable in the stickleback regression model, so no value was derived for this species.

Table 5.4: Preliminary criteria of dissolved oxygen for migrating fish species. These criteria ($DO_{P\ =\ 0.5}$) correspond with a capture probability of 50% assuming that the other environmental predictor variables (temperature and river flow) can be substituted by their average value during the season of maximum occurrence. DO was not retained in the model for three-spined stickleback, so no concentration was defined for this species.

Species	**Season of maximum occurrence**	**Seasonal temperature (°C)**	**Seasonal river flow ($m^3\ s^{-1}$)**	**$DO_{P=0.5}$ ($mg\ l^{-1}$)**
European eel	summer	20.5		1.3
European smelt	fall	17.0		5.5
Flounder	summer	20.5		2.7
Thinlip mullet	summer	20.5		6.2
Three-spined stickleback	spring	11.6		
Twaite shad	summer	20.5	77.6	5.1

4 Discussion

Historical research by Van Damme *et al*. (1994) and by Vrielynck *et al*. (2003) suggests that at the beginning of the 20th century, ten anadromous and three catadromous species frequented the Schelde basin, albeit with different population sizes (de Selys-Longchamps 1848, 1867; Bottemanne, 1884; Poll, 1945). Sturgeon *Acipenser sturio* and allis shad *Alosa alosa* were once probably quite abundant, but extirpated during the first two decennia of the 1900's, mainly as a result of increased water pollution. Populations of sea lamprey, Atlantic salmon, sea trout and houting *Coregonus oxyrhynchus* were probably constrained by available spawning substrates in the river basin and hence, had relatively small population sizes compared to the neighbouring populations of the rivers Rhine and Meuse (De Groot, 2002). Individuals of these first three species sporadically returned in fyke net catches between 1995 and 2004. Other populations of diadromous species persisted, although some of them are now confined to the lower reaches of the estuary that are situated downstream of the freshwater saltwater front. This is the case for thinlip mullet, European smelt and twaite shad. Adults of the latter two species occur in the lower estuary but seem unable to reach upstream spawning sites. Thinlip mullet as well as flounder are catadromous but catadromy is in neither species obligate. They both spawn offshore and young of the year move upstream to estuaries and, if possible, above the tidal limit. River lamprey, European eel and three-spined stickleback were the only diadromous species with a confirmed closed life cycle in the basin (Van Damme *et al*., 1994; Maes *et al*., 1998; Maes *et al*., 2004a). Three-spined sticklebacks probably form a metapopulation with migrating forms from the *trachurus* and *semiarmatus* type and the resident forms (*leiurus* type), but they were not separated in the field. Numbers of river lamprey were highly underestimated by the fishing method used in this study since there is evidence that migrating individuals moved through the subtidal navigation channel. Sampling at a power station cooling water inlet (Maes *et al*., 2004a) confirmed the presence of upstream moving adult spawners and seaward moving juveniles.

The occurrence and spatiotemporal distribution of diadromous fish species in the Schelde basin were explained in this chapter as a function of three environmental variables. In particular, the interaction between dissolved oxygen and temperature proved to be a statistically significant predictor of fish capture probabilities. The interpretation is that under summer warm conditions in the watershed, dissolved oxygen is a limiting factor. In particular, hypoxic events in the tidal freshwater part of the estuary just above the freshwater saltwater boundary prevented upstream migration movements of both anadromous spawners and catadromous young of the year on their way to either spawning substrates or nursery areas. The relation between DO and temperature is a very important issue considering global warming due to the climate change. There is a species specific limit to acclimatisation (Pörtner & Knust, 2007). These authors showed that a mismatch between the demand for oxygen and the capacity of oxygen supply to tissues is the first mechanism to restrict whole-animal tolerance to thermal extremes. As shown by our assessment of DO criteria, tolerances were also species specific. Eel, stickleback, and flounder were the most tolerant species and their distribution shows that they are able to move through the zone of low DO. Twaite shad, smelt and mullet were much more sensitive to low DO concentrations and did not penetrate as far upstream. Clearly, this conclusion should be interpreted within the context of the limitations of this study. Observational studies like this one do not prove causality. In this study, we essentially made a statistical correlation between two datasets that were spatially and temporally unmatched. This approach is, however, not necessarily flawed. The water quality measurements that were used in this study were derived from a consistent water quality monitoring network using standardized sampling and analysis protocols. The data cover sufficiently the study area in order to capture the prevailing spatial and seasonal trends. This would have been impossible by sampling and analysing water only during fishing occasions. Therefore we claim that, by using environmental data from a database that is maintained by a governmental agency, the present statistical models can be applied to derive the spatially and temporally explicit probability plots as presented in this study, to validate the models against future water quality data or to make new predictions within the watershed.

In an application of the logistic regression models we inferred DO concentrations for which the probability of the presence of diadromous fish in diurnal fyke net samples is 50% and we proposed this data as preliminary DO criteria for fish in the watershed. Clearly, this approach lacks an experimental basis. Under the assumption that DO criteria apply during periods of peak migration, minimum values vary between 1.3 and 6.2 mg l^{-1} while the average was 4.2 mg l^{-1}. In figure 5.4, we showed that scenarios with a dissolved oxygen concentration of at least 5 and 6 mg l^{-1}, respectively, substantially increased the capture probability of fish occurrence relative to the situation as observed in 2003. Based on these results, we suggest a minimum dissolved oxygen concentration of 5 mg l^{-1} throughout the tidal part of the river basin for migratory fish (see also Maes *et al*., 2008). This minimum corresponds to the regional (Flemish) baseline water quality requirement. More field evidence on DO tolerances in estuarine fishes is presented by Möller and Scholz (1991), who used stow nets to sample fish along a DO gradient during summer and fall in the Elbe estuary (Germany). Fish were found to concentrate downstream the area of hypoxia. Species specific DO preferences were between 1.2 and 3 mg l^{-1} for eel, between 3 and 4 mg l^{-1} for flounder, between 4 and 5 mg l^{-1} for 0 group shad and >5 mg l^{-1} for smelt. Recently, Turnpenny *et al*. (2004) estimated lethality and tolerances of estuarine fishes to low DO concentrations in the Thames estuary using an experimental set up in the laboratory and in the field (see also Chapter 4). So, other than in this study, avoidance of low DO by fish has been directly tested. Based on the overall test results, a minimum DO standard of 1.5 mg l^{-1} was suggested for the tidal Thames. A number of species, among which the smelt, seemed unexpectedly tolerant to low concentrations of DO. This value is clearly lower than the 5 mg l^{-1} suggested by this study based on empirical field data. Although experimental trials can be interpreted in a straightforward manner, empirical field data have the advantage that possible fitness consequences are included. Diadromous fish species possibly tolerate low DO concentrations but may skip spawning when environmental conditions nearby the reproductive habitats are unfavourable for eggs or early life history stages. Such behaviour is common in many fishes (Jørgensen *et al*., 2006).

4.1 Applications for watershed management

Ecological rehabilitation of the diadromous fish fauna requires applicable knowledge that can be used to identify limiting factors for population recovery. Here we demonstrated that it is possible to make acceptable predictions about the future spatiotemporal distribution of migrant fishes with relatively limited information. The models that were used yield testable predictions. Empirical models of the probability of presence or absence of species rather than of fish abundance warrant straightforward interpretation and avoid inclusion of density dependent effects or recruitment variability. Predictor variables used in the models represent true ecological recourses and data of dissolved oxygen and temperature is commonly, and often freely, available in databanks.

A first essential step for river management that derives from this study is to increase the concentrations of DO in the freshwater tidal estuary of the watershed. The model results suggested that an increment of DO to a baseline concentration of 5 mg l^{-1} considerably increases the opportunity for diadromous fish species to pass the middle part of the estuary. At present, this area receives the treated municipal waste water of the Brussels capital region through the contributories Zenne and Rupel. The 1.5 million inhabitant equivalent waste water purification plant of Brussels is functioning since March 2007. Therefore, the water quality of the tidal Schelde where the river Rupel discharges into the Schelde improved consistently (Van Thuyne & Breine, 2008; 2009) and hence, the basin wide distribution of migratory fish (Stevens *et al*., 2009). The observed improved DO (see Chapter 2) is the result of a reduction in refuse.

Decreasing chemical and biological oxygen demand by the ongoing wastewater treatment programmes seems evident but is not the only solution. Estuaries are natural collectors of organic waste and the transformation of ammonia, particulate and dissolved organic matter depletes the available DO. In estuaries, aeration of water is an important source of oxygen (Van den Bergh *et al*., 2005). Aeration is more efficient in areas with a high surface to volume ratio such as marshes and flood control areas. Restoration of these habitats, although generally not essential in the life history of diadromous species, is likely a crucial measure to support fish migration. In anticipation of a population recovery, a survey of suitable spawning habitats and substrate is another requirement to support successful restoration

Migrant fish are considered as important indicators of ecosystem recovery, especially in our society, which has hardly a collective memory of migrating fish species. The return of species that were once so abundant that they were used as fertiliser would be an important milestone after decades of decline and an environmental success.

Conclusions

Two important conclusions can be derived from this study. First we can explain the presence of diadromous species in the Zeeschelde as a function of DO and water temperature. A second conclusion is that the return of diadromous species will be enhanced by implementing rehabilitation plans including the functioning of the water purification in Brussel.

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Chapter 6

Tidal marshes as habitat for juvenile fish in the Zeeschelde estuary (Belgium)

Jan Breine, Maarten Stevens, Joachim Maes, Erika Van den Bergh & Mike Elliott

Abstract

Little is known about the use by fishes of tidal marshes located in the mesohaline, oligohaline and freshwater tidal zone of an estuary. Two different fishing protocols were applied in order to assess: 1) spatial and temporal effects and 2) the influence of creek characteristics on the fish assemblages. In 2007 fish were sampled monthly from creeks in four tidal marshes, located in different salinity zones in the Zeeschelde. In 2008 nine creeks within one tidal marsh but each with different characteristics were sampled in spring, summer and autumn. Fish were caught in the creeks with winged fyke nets (1.5 cm mesh size in the cod end). For each creek we measured mouth width, level of bottom (versus mean low water level of the main river) and slope of the bank. Creek volume was calculated using cross section data, creek length and number of adjacent creek branches. In addition we recorded the presence of debris and permanent pools in both creeks and tributaries. In total we recorded 24 different fish species between 2007 and 2008 and catches were dominated by juveniles. The most abundant species was flounder. Multivariate analyses examined the variations of the fish assemblage in relation to position in the estuary and also to creek characteristics within one marsh. The influence of the salinity gradient is reflected in the different fish assemblages present in the four marshes. The highest number of fish was caught in summer. Small creeks and creeks with short flood periods are less frequented than large creeks situated lower in the tidal frame and containing permanent pools. The study emphasizes the importance of creeks in tidal marshes as habitats for juvenile fish.

Keywords: fish, tidal freshwater marsh, creek characteristics, Belgium, Zeeschelde,

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1 Introduction

Salt marshes, estuaries and coastal systems are considered as the most productive areas of the biosphere (Laffaille *et al*., 2004). They play an essential role in providing food for fishes (Kneib, 1997; Nemerson & Able, 2004; Hollingsworth & Connolly, 2006; Svensson *et al*., 2007). In addition they provide shelter from predators (Kneib, 1987; Halpin, 2000) and are important for reproduction of some fish species (Talbot & Able, 1984).

Several scientific studies about the role of tidal marshes for juvenile fishes are readily available for salt and brackish systems (e.g. Pihl *et al*., 2002; Lazzari *et al*., 2003; Nordlie, 2003; Stoner, 2003; Hampel & Cattrijsse, 2004; Hampel *et al.*, 2005; Cattrijsse & Hampel, 2006; Veiga *et al*., 2006; MacKenzie & Dionne, 2008; Madon, 2008). Tidal freshwater marshes are rare habitats, and as a result, the number of studies reporting on them is limited.

McIvor and Odum (1986) were among the first to study US freshwater marshes and found that they were used by numerous estuarine and freshwater species. Rozas and Odum (1987a) suggested that marshes containing complex, well-developed tidal creek systems are more productive for fishes than marshes with few or no tidal creeks. Odum *et al*. (1988) compared fish assemblages in freshwater marshes and marshes in the oligohaline zone. The oligohaline fish community was dominated by estuarine and marine species, while freshwater species were dominant in freshwater marshes. Rozas *et al*. (1988) assessed the importance of creeks and small rivulets, small intertidal creeks that drain the freshwater marshes. They postulated that small fish try to reach the marsh quickly via these rivulets to avoid predation. The importance of freshwater marshes in the immobilisation of excess nutrients and in the denitrification process was illustrated by Harvey and Odum (1990). Recently Kimball and Able (2007) assessed the effects of habitat restoration (i.e. *Phragmites* removal) on fish assemblages in oligohaline tidal creeks. Their results were inconsistent, indicating that restoration efforts had little effect.

In Europe freshwater marshes gained only recently in interest, but still limited to a few countries. This is due to the fact that these habitats are scarce and many of them were even in the recent past reported biologically dead (Van den Bergh *et al*., 2009). In the Schelde, Hampel *et al*. (2004) surveyed five marshes along the salinity gradient, including one tidal freshwater marsh, and recorded a small number of freshwater fish. In France, Laffaille *et al*. (2004) indicated that freshwater marshes are suitable as habitat for eel (*Anguilla anguilla*).

Worldwide overexploitation, pollution, reclamations and habitat transformation have degraded the estuaries and caused habitat loss (Lotze *et al*., 2006). This destruction has led to a growing interest in restoration, creation and conservation of tidal wetlands (e.g. Larkin *et al*., 2009), taking into consideration economical (Costanza *et al*., 1997), ecological (Hampel *et al*., 2004; McLusky & Elliott, 2004) and safety (flood-control) criteria (Zedler, 2000; Cattrijsse *et al*., 2002; Van den Bergh *et al*., 2005, 2009). In chapter 4 we described ecological goals and associated habitat needs for fish in estuaries whereby the Zeeschelde was used as a case study. A major challenge is quantifying the desirable estuarine carrying capacity for fish in the Zeeschelde including tidal marshes. Except for Hampel *et al*. (2004) no information of habitat utilisation by fish in freshwater and oligohaline marshes in the Zeeschelde estuary was found. The habitat function use of reed- and willow dominated tidal marshes for fish has not been documented (Cattrijsse & Hampel, 2006). This study has two principal objectives: 1) assess the importance of tidal marshes as a habitat for fish in the different salinity zones and 2) assess the impact of creek morphological characteristics on the fish community. In addition we compared the fish assemblage in a tidal marsh creek with the one in its adjoining mudflat and studied also the fishes in the permanent subtidal waters of that same marsh.

2 Material and methods

2.1 Study area

The river Schelde is a tidal lowland river with its origin in the northern part of France (St. Quentin), and its mouth in the North Sea near Vlissingen, The Netherlands. About half of the 355 km long River Schelde covers a tidal area of 160 km between Gent (Belgium) and the North Sea near Vlissingen (The Netherlands). The Zeeschelde (Belgium) consists of three salinity zones: mesohaline, oligohaline and freshwater zone. Four marshes were selected to assess the fish assemblages along the salinity gradient of the Zeeschelde estuary (Fig. 6.1). Schor Ouden Doel (SOD) on the left bank of the Zeeschelde has an elongated shape with 50% tidal marshes (48 ha) and 50% mudflats (46 ha). It is situated in the mesohaline zone with an average salinity of 6.1 and a diverse vegetation: reed, *Phragmites australis*), alkali bulrush (*Scirpus maritimus*) and Cough grass (*Elymus athericus*) (Fig. 6.1). (Van den Bergh *et al*., 2005). It is located nearby a dredging disposal area and sand extraction site. The tidal marsh the Notelaar (NOT) (29.3 ha) is situated in the oligohaline zone of the Schelde estuary. It is about 3.6 km long (shore side) and maximum 200 m broad. It has a relative stable vegetation including a large proportion of woody vegetation (willow and poplar). Due to the presence of trees its creeks are more sheltered than the ones in Schor Ouden Doel. It is the only tidal marsh in the oligohaline zone with permanent aquatic habitats. The Groot Schoor van Grembergen (GREM) (8.77 ha) is a tidal marsh situated in the freshwater zone of the estuary and has a stable willow and reed vegetation (Fig. 6.1). The Groot Schor van Hamme (HAM) (38.6 ha) is a tidal marsh situated in the freshwater part of the estuary, characterised by a long retention time and here too willow and poplar are present.

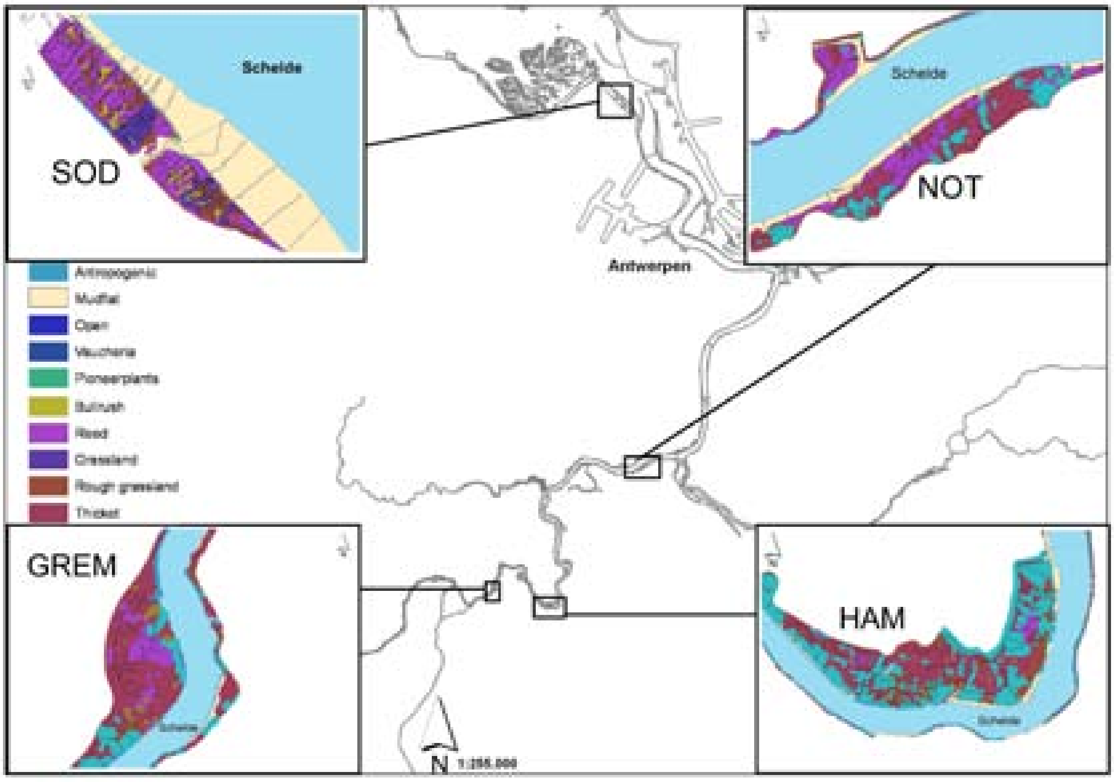

Figure 6.1: The locations of the four tidal marshes along the Zeeschelde and their vegetation cover. In the mesohaline zone SOD: Schor Ouden Doel; in the oligohaline zone NOT: Notelaar; and in the freshwater zone GREM: Groot Schoor van Grembergen and Ham: Groot Schor van Hamme.

2.2 Fish sampling

Three natural marsh creeks in the Notelaar (NOT 5, 7 & 8) and Hamme (HAM, 1, 2 & 3) and two in Grembergen (GREM, 1 & 2) were surveyed with winged fyke nets (1.5 cm mesh size in cod end) and eel fykes (1 cm mesh size in cod end). The marshes were sampled monthly between March and October 2007 to assess their importance as habitat for juvenile fishes.

The fyke entrance was orientated towards the marsh with the wings spanning the entire width of the creek. The eel fykes are smaller fykes without wings and were deployed in the smaller adjacent creeks i.e. the first branch. The fykes were set at low tide and emptied approximately 24 h later.

Winged fyke nets were deployed in an identical way in three creeks in the Schor Ouden Doel marsh (SOD, 1, 2 & 3) in March and July 2007.

In addition nine creeks were surveyed in the Notelaar (NOT 1 to 9) to assess the relationship between fish assemblages and creek characteristics, using the same protocol with winged fykes and eel fykes in spring, summer and autumn 2008.

To obtain extra information about the habitat use, permanent pools in three creeks in the Notelaar were sampled in April 2008 using electric fishing and kick sampling with a hand net (5 mm mesh size). Electric fishing was done with a 5 kW generator with an adjustable output voltage of 300 to 500 V and a pulse frequency of 480 Hz. In September 2008 three double fyke nets (type 120/8, 1.5 cm mesh size in cod end) were deployed on the mudflat at the low water level nearby one creek in the Notelaar for two successive days. At the same time seine netting (1 cm mesh sized monofilament net of 25 m by 2 m) was performed on the adjoining mudflat twice at high tide for three successive days to assess the fish assemblage near the tidal marsh.

All fish caught were identified to species level using Nijsen and De Groot (1987) as field guide. Occasional cross examination in the laboratory ensured identification quality. Fish data recorded included species-specific fish frequencies, individual total lengths (± 1 mm) and weights (g). Fish abundance in the fykes was standardised as the number of fish per fyke per day. To standardise the catches in the different marshes the fyke catches in each creek were summed up separately for winged fykes and eel fykes. Fish data were log transformed (log (x+1)) prior to statistical analysis.

2.3 Creek characteristics (Fig. 6.2)

We measured with a RTK-GPS Trimble (Real Time Kinematic-Global Positioning System, 1-2 cm precision) for each of the nine creeks in the Notelaar the creek mouth width and the elevation of the creek mouth (bottom level) relative to the mean low water level in main river. The slope of the creek bank was calculated as the ratio of the creek width and the height of the creek bank near the sampling location. Creek volume was estimated using the cross-section and the length of the main creek and adjacent creek branches (tributaries). In addition we recorded the creek slope, the amount of woody debris in the main creek on a relative scale (0-

5) (Table 6.1) and the number of permanent pools (> 10 cm of water at low tide) in the main creek and branches. These characteristics were also proposed by Allen *et al.* (2007).

Table 6.1: Relative woody debris scale

Woody debris	Relative scale
absent	0
branches alongside the creek	1
branches in the creek	2
branches in creek and tributaries	3
branches and tree stem in creek	4
branches and tree stem in creek and branches in tributaries	5

Figure 6.2: The locations of the different creeks in the Notelaar, oligohaline tidal marsh in the Zeeschelde.

2.4 Data processing and analysis

Only fyke net data are processed, the data obtained with other gears were regarded as informative. Non-metric multidimensional scaling (NMDS) ordination was performed to examine the spatial organization of the fish assemblage in marsh creeks along the salinity gradient. NMDS is a common analysis to assess fish assemblages (e.g. Chen *et al*., 2006; Mazumender *et al*., 2006). The fish community in the creeks of four marshes (Fig. 6.1) was sampled simultaneously in March and July 2007. For each creek the average abundance of each species was calculated and used as input in the NMDS analysis. Dissimilarity matrices were calculated from log (x+1) transformed fish abundance data, using Bray-Curtis distances. Only species that were caught more than once in each marsh creek were included in the analysis (16 out of 20 species). The NMDS ordination was created using random starting configurations and iterated until solutions converged. The vegan package in R 2.6.2 was used for the analysis (Oksanen *et al*., 2006). The data obtained in 2008 (NOT 1 to 9) were used to assess effects of creek variables on fish assemblage structure. Relationships between fish abundance and creek characteristics in NOT were examined by canonical correspondence analysis (CCA). CCA allows to relate the abundance of species to environmental variables (ter Braak, 1986). Again species occurring only once in the samples were omitted from the analysis. The species-abundance matrix (9 creeks x 15 species) was constructed from the average log (x+1) transformed abundance data. The matrix of creek characteristics consisted of the elevation of the creek bottom relative to the mean low water level (HeigL), the number of permanent pools (Pools), the number of branches of the main creek (Branch), the slope of the creek bank near the sample site (Slope), the volume of the creek and tributaries at high water (Volume) and the amount of debris in the creek (Debris). The constraining variables were checked for redundancy, using the variance inflation factors (VIF). CCA was performed using the vegan package version 1.8-3 in R version 2.4.0.

3 Results

3.1 Creek characteristics and environmental parameters

Table 6.2 gives the different characteristics measured in the nine creeks (Fig. 6.2) of the tidal marsh the Notelaar. Each creek is particular. The outlets of NOT2 and NOT6 are situated higher compared to the other creeks. These two creeks have also the smallest mouth (6.8 m and 8.4 m respectively). The largest creek (NOT5) has a mouth width of 44 m and is 805 m long with large branches. NOT9 differs from the other creeks because of the absence of foliage at its mouth which is 33 m wide, and the creek has six smaller branches. NOT 8 is also a large creek with a 28 m wide mouth and four large branches.

Table 6.2: Creek characteristics in the Notelaar in 2008. Debris: amount of woody debris (relative scale 0-5); HeigL: elevation of the base of the creek relative to the mean low water level in m; Pools: number of permanent pools; Branch: number of branches of the main creek; Slope: slope of the creek bank at the sampling location; Volume: volume of the main creek and branches; Crslope: creek slope in %

Creek	Debris	HeigL (m)	Pools	Branch	Slope (%)	Volume (m^3)
NOT1	2	3.16	0	3	6.2	471
NOT2	3	4.28	0	3	6.3	155
NOT3	5	3.44	1	2	11.8	348
NOT4	3	3.44	0	2	2.2	163
NOT5	1	3.28	2	5	9.0	29006
NOT6	2	4.64	1	2	14.2	206
NOT7	4	3.84	2	1	5.0	1027
NOT8	2	3.91	9	4	4.0	2466
NOT9	0	3.22	0	6	14.0	2631

Debris: the higher the score the more debris present

3.2 Fish catches

The total number of fish caught for each survey technique over the total survey period is presented in table 6.3 as catch per unit effort (CPUE).

In 2007 eight different species were collected in the creeks of Schor Ouden Doel (SOD). The most abundant species caught in the creeks was flounder (*Platichtys flesus*) (48.9% relative abundance) followed by seabass (*Dicentrarchus labrax*) (29.8%). Mostly juvenile individuals (0+) were caught, but larger seabass (> 20 cm) enter the creeks to feed on smaller fish. Occasionally larger flounder (>10 cm) was caught as well. In 2007 and 2008, 13 and 17 different species were respectively caught in the oligohaline Notelaar (NOT). In both years the most abundant species was flounder (>40% relative abundance). In the tidal marsh nearby Hamme 17 different species were caught in 2007 with eel (*Anguilla anguilla*), roach (*Rutilus rutilus*) and Prussian carp (*Carassius gibelio*) being the most abundant species (in total 63.5% relative abundance). In the Groot Schor van Hamme (HAM) the catch per unit effort was higher than in the other freshwater tidal marsh situated in Grembergen (GREM). In the Groot Schoor (GREM) 10 different species were caught in 2007 and the most abundant species were the same as in the Groot Schor van Hamme (in total 84% relative abundance).

Table 6.3: Number of individuals from each species (CPUE) collected in the different tidal marshes during the surveys 2007-2008; number of surveys between brackets. El and K occurred in permanent pools and D and S on mudflats in front of NOT.

			2007							2008					
			SOD	NOT		HAM		GREM		NOT					
Code	**Scientific name**	**EUFG**	**W(6)**	**W(24)**	**E(24)**	**W(24)**	**E(24)**	**W(16)**	**E(16)**	**W(46)**	**E(46)**	**El(3)**	**K(2)**	**D(3)**	**S(3)**
A.bra.	*Abramis brama*	Fw	0.00	0.29	0.00	0.54	0.04	0.50	0.00	0.22	0.00	0.00	0.00	59.67	5.33
A.ang.	*Anguilla anguilla*	Di	1.00	0.67	0.25	3.54	0.58	2.50	0.94	0.96	0.26	0.33	0.00	13.33	0.00
B.bjo.	*Blicca bjoerkna*	Fw	0.00	0.38	0.00	0.54	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.67
C.gib.	*Carassius gibelio*	Fw	0.00	0.00	0.00	0.13	0.00	0.06	0.00	0.09	0.00	0.00	0.00	0.00	0.00
C.har.	*Clupea harengus*	Mm	0.00	0.00	0.00	1.38	0.00	1.94	0.38	1.30	0.00	0.00	0.00	0.00	0.00
C.car.	*Cyprinus carpio*	Fw	2.00	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.00	0.00	0.00	11.33	0.00
D.lab.	*Dicentrarchus labrax*	Mm	9.33	0.00	0.00	0.00	0.00	0.00	0.00	0.67	0.00	0.00	0.00	11.67	9.33
E.luc.	*Esox lucius*	Fw	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
G.acu.	*Gasterosteus aculeatus*	Fw	0.17	0.04	0.00	0.08	0.00	0.06	0.06	0.20	0.00	16.00	13.00	1.00	0.67
G.cer.	*Gymnocephalus cernuus*	Fw	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.61	0.04	0.00	0.00	11.67	0.00
L.gib.	*Lepomis gibbosus*	Fw	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
L.idu.	*Leuciscus idus*	Fw	0.00	0.17	0.00	0.50	0.00	0.00	0.00	0.09	0.00	0.00	0.00	0.00	0.00
L.ram.	*Liza ramado*	Di	2.67	0.88	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
O.epe.	*Osmerus eperlanus*	Di	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.33	0.00
P.flu.	*Perca fluviatilis*	Fw	0.33	0.33	0.04	0.04	0.00	0.06	0.00	0.00	0.02	0.00	0.00	1.00	0.00
P.fle.	*Platichthys flesus*	Di	59.17	3.88	0.17	0.67	0.13	0.00	0.00	10.61	3.00	0.00	0.00	116.00	2.67
P.mic.	*Pomatoschistus microps*	Es	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.00	0.00	0.00	382.00	9.33
P.par.	*Pseudorasbora parva*	Fw	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.67	2.00	0.00	0.00
P.pun.	*Pungitius pungitius*	Fw	0.00	0.42	0.00	0.08	0.00	0.13	0.00	2.26	0.04	0.33	0.00	0.00	0.00
R.ser.	*Rhodeus sericeus*	Fw	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
R.rut.	*Rutilus rutilus*	Fw	0.00	1.71	0.00	1.46	0.00	0.81	0.00	0.00	0.00	0.00	0.00	10.00	1.00
S.luc.	*Sander lucioperca*	Fw	0.00	0.08	0.00	0.25	0.00	0.06	0.00	0.13	0.00	0.00	0.00	0.00	0.00
S.ery.	*Scardinius erythrophthalmus*	Fw	0.50	0.63	0.00	0.38	0.00	0.13	0.00	0.91	0.00	0.00	0.00	14.33	0.33

EUFG: Estuarine Use Functional Group (Franco *et al*., 2009): Fw: freshwater; Es: estuarine species; Di: diadromous species; Mm: marine migrant; SOD: Schor Ouden Doel; NOT: Notelaar; HAM: Groot Schor van Hamme; GREM: Groot schoor Grembergen; W: winged fyke net; E: eel fyke net; El: electric fishing; K: kick sampling; D; double fyke net; S: seine netting

Figure 6.3 illustrates the different length classes caught for the most abundant species in the oligohaline and freshwater tidal marshes (2007). The figure shows that juveniles are most abundant but that also larger specimens visit the creeks. Roach, eel, bream (*Abramis brama*) and Prussian carp are present in all tidal marshes, although eel, bream and Prussian carp are less abundant in the oligohaline tidal marsh (NOT). In general the bream (39.6% of total catch) and Prussian carp (34%) were large individuals (>20 cm). Flounder and thinlip mullet (*Liza ramado*) were not caught in the most upstream tidal marsh (GREM) and most individuals caught in the other marshes were small (<10 cm). White bream was not caught in GREM but in HAM two specimens larger than 20 cm were caught. The piscivorous pike-perch (*Sander lucioperca*) was caught in all three marshes. In the oligohaline GREM 19 individuals were caught while only 9 and 2 respectively in HAM and GREM.

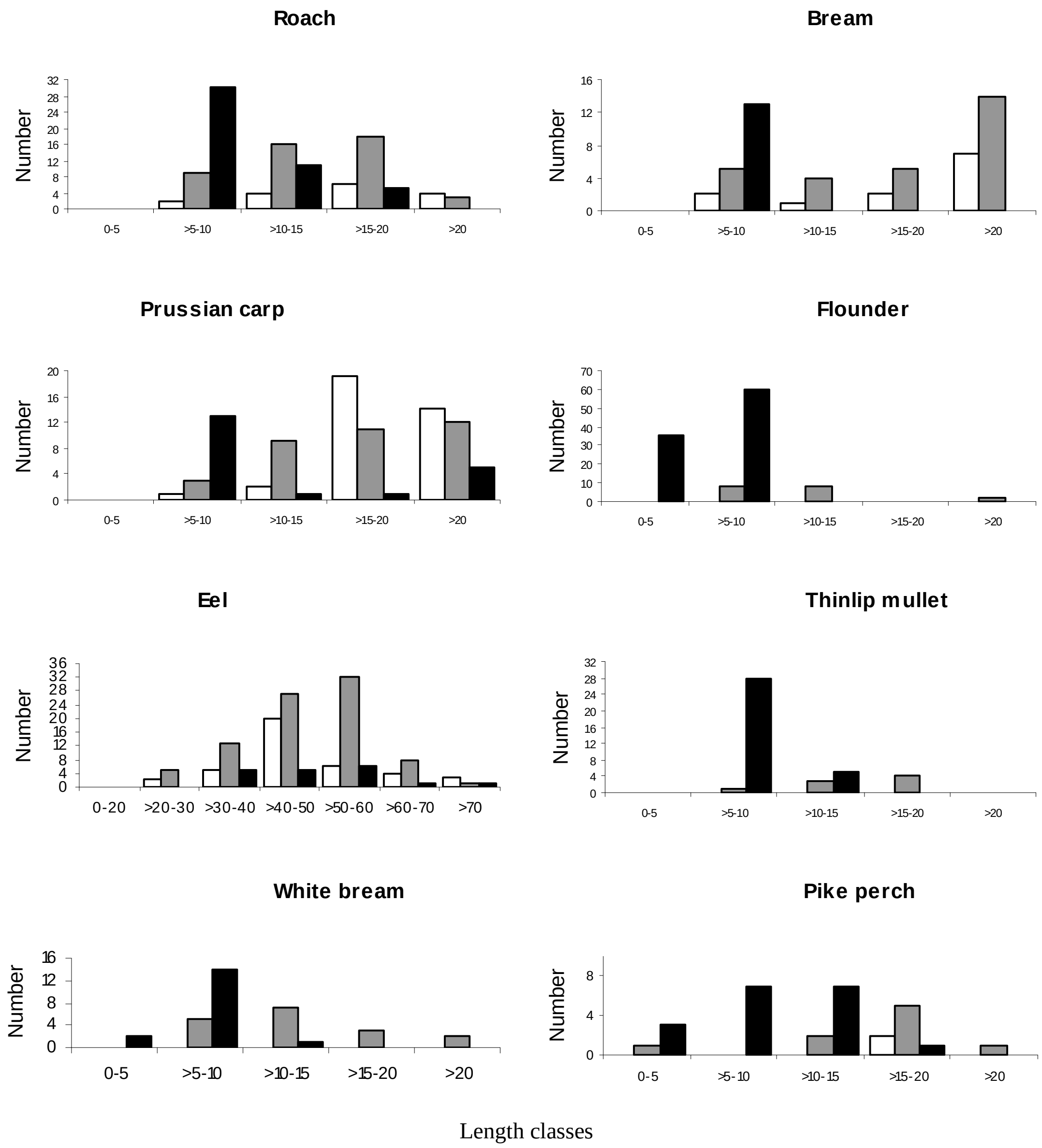

Figure 6.3: Length class frequency distribution of most abundant species caught in 2007 from the creeks of freshwater (GREM, white and HAM, grey) and oligohaline (NOT, black) tidal marshes.

3.3 Changes in the fish community along the salinity gradient

In the mesohaline Schor Ouden Doel (SOD) eight different species were caught using winged nets (n=6) (Table 6.3). Catches in March 2007 were poor with three species: flounder, seabass and three-spined stickleback (*Gasterosteus aculeatus*). In July 2007, seven species were caught (Fig. 6.4).

In the creeks of the oligohaline marsh (NOT) only bitterling (*Rhodeus sericeus*) and bream were caught in March, in July no fish were captured at all. However, the average catch results over eight month surveys (Fig. 6.5) show that in spring catch results are smaller than in summer. Data were log transformed for scaling reasons only.

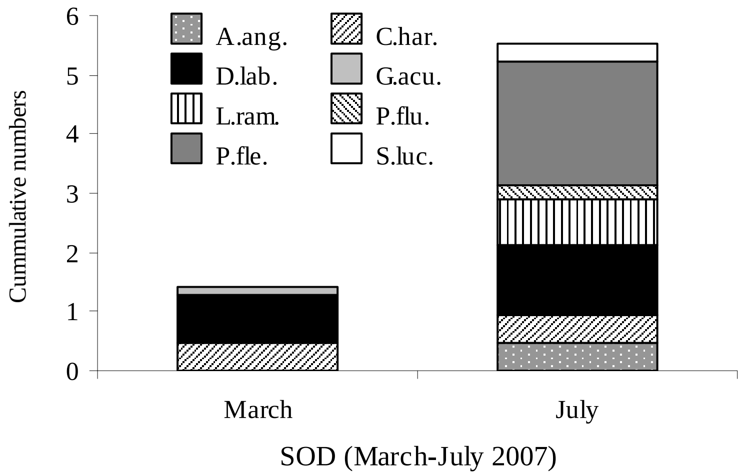

Figure 6.4: Cumulative representation of Catch Per Unit Effort (CPUE) log (x+1) transformed for each species caught in the different creeks of Schor Ouden Doel SOD (March and July 2007) (abbreviations see Table 6.3).

In the two freshwater marshes similar patterns as in NOT were observed. In the freshwater Groot Schor (HAM) eel, three-spined stickleback, bream and smelt (*Osmerus eperlanus*) were caught in March while no fish was caught in July (Fig. 6.5). In Groot Schoor (GREM) six species were caught in March with Prussian carp as most abundant species, but none in July (Fig. 6.5).

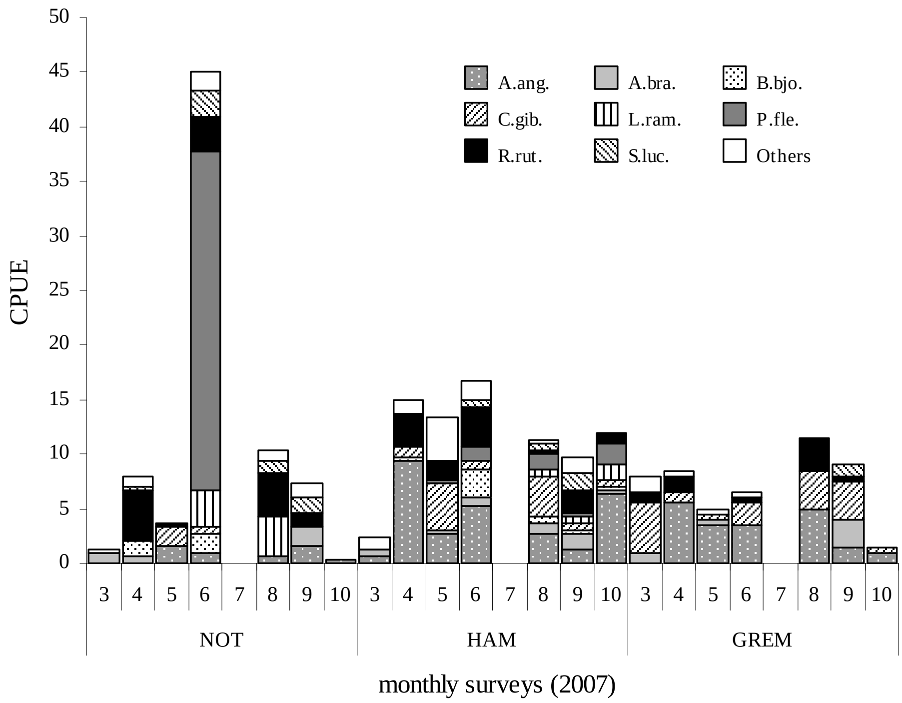

Figure 6.5: CPUE caught with winged nets in the tidal oligohaline marsh (NOT), the freshwater marshes (Groot schor, HAM and Groot Schoor, GREM) over eight months (March- October) in 2007 (abbreviations see Table 6.3). Only the 8 most abundantly recorded species have a specific pattern.

The NMDS ordination shows a clear distinction between the samples of the mesohaline marsh (SOD) the oligohaline marshes and the freshwater (NOT, HAM and GREM, Fig. 6.6). The first axis represents the salinity gradient while the second axis represents a dimension factor. Possibly the ratio marsh/mudflat surface could have an effect on the visit frequency of fish. Marsh creeks in SOD were characterized by the presence of mainly seabass, herring (*Clupea harengus*) and flounder. The latter species was also found in the oligohaline (NOT) and the most downstream freshwater (HAM, GREM) marshes. The oligohaline creeks cluster together at the lower left side of the ordination plot, while the freshwater marshes appear along an upstream gradient towards the upper left side of the plot. The fish community in the freshwater marshes was dominated by European eel, Prussian carp and roach, while percids, flounder and thinlip mullet were the characterizing species in the oligohaline creeks.

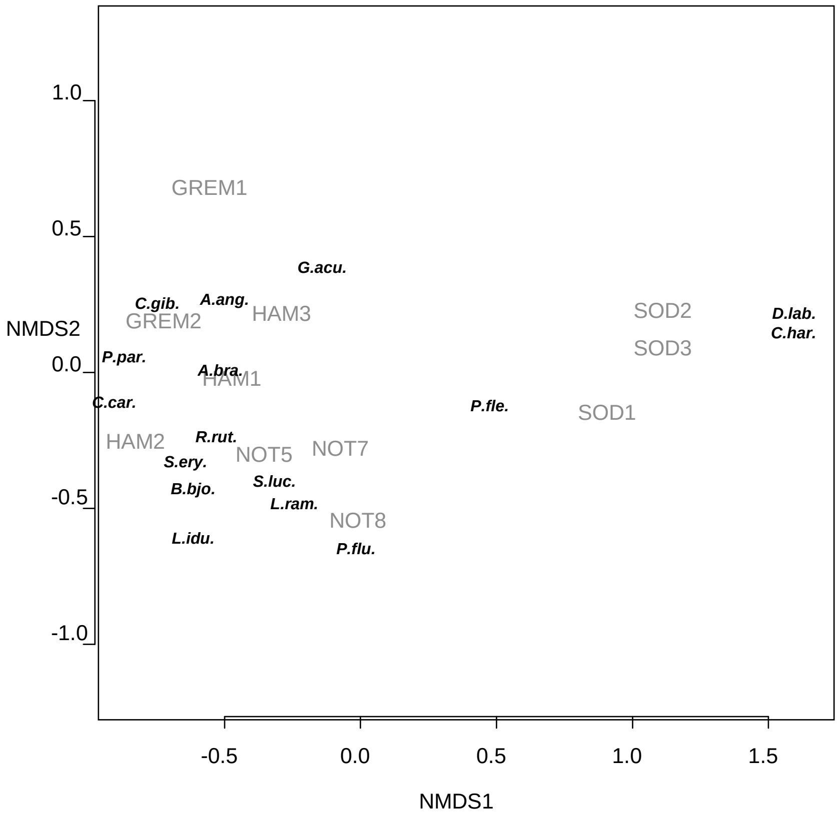

Figure 6.6: Non-metric multidimensional scaling (NMS) ordination of fish abundance data for the marsh creeks in March and July 2007. Data consist of log (x+1) transformed abundance data for 16 taxa. GREM = Groot Schoor Grembergen, HAM = groot Schor Hamme, NOT = Notelaar and SOD = Schor Ouden Doel. See table 6.3 for species abbreviations.

3.4 Species diversity related to creek characteristics in NOT

In 2008 17 different species were collected with both types of fykes in the Notelaar while 32 different species were caught in the subtidal oligohaline zone (spring and autumn catches, Guelinckx *et al*., 2008). Figure 6.7 illustrates the length classes for the most abundant species caught in 2008 in the Notelaar. The bulk of roach and bream individuals are between 5 and 10 cm long, but some larger individuals were also caught (Fig. 6.7). Herring and seabass individuals are small while some larger individuals (> 15 cm) of pike-perch were recorded. Flounder is the most abundant species, most individuals representing a length between 5 and 15 cm.

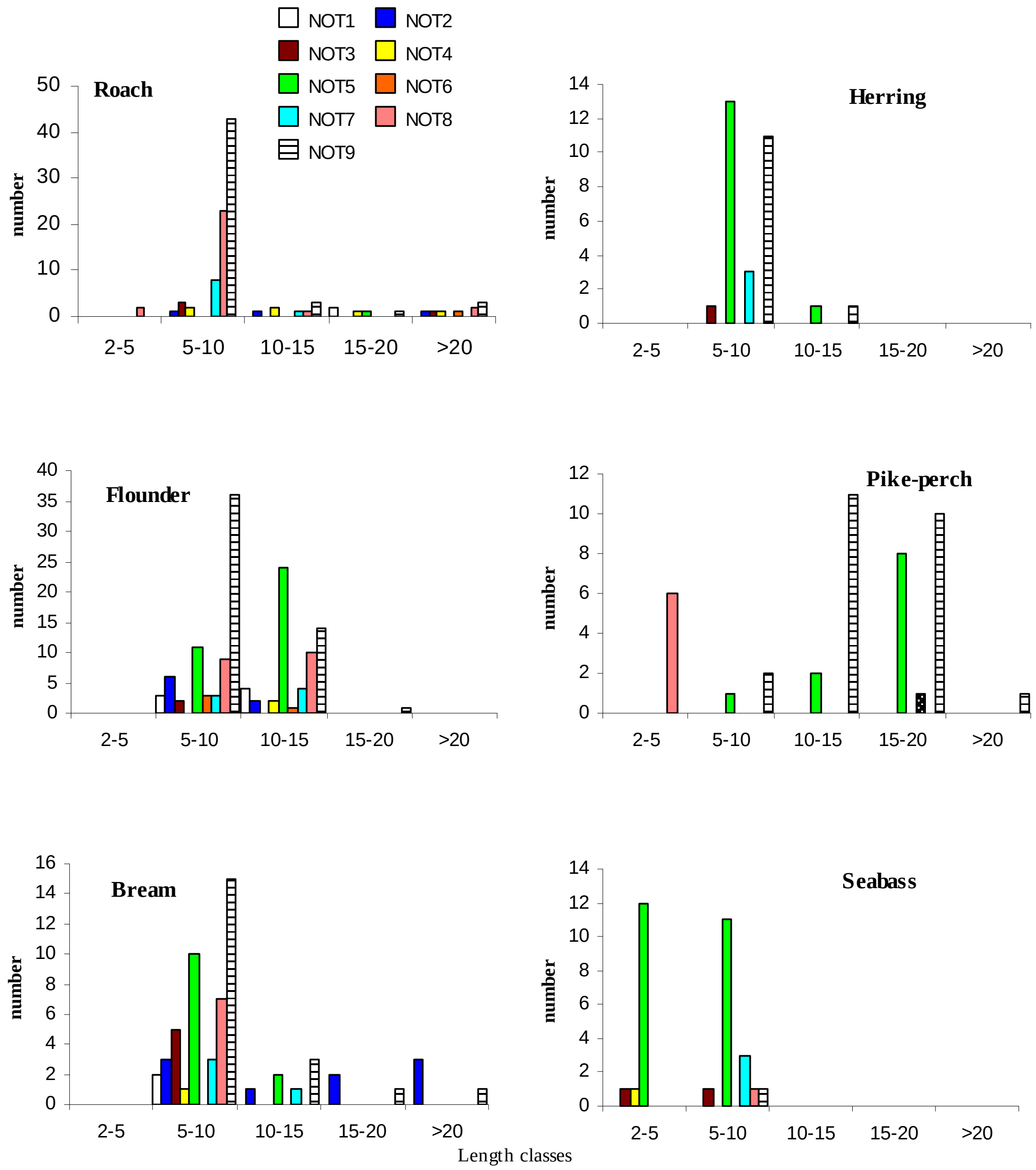

Figure 6.7: Length class frequency distribution of most abundant species caught in 2008 in nine creeks of the oligohaline tidal marsh (NOT). Lengths are expressed in cm.

The CCA plot describes the relation between the species composition in the marsh creeks and the creek characteristics in NOT (Fig. 6.8). The observed pattern seems logical but due to lack of data our results have to be considered as indicative only ($p=0.32$; $F= 1.71$; $df= 6$), although the explained fraction is high (the trace is 0.414 and the total inertia is 0.495). The eigenvalues of the first and second axis are 0.20 and 0.09 respectively. Although more environmental variables, such as the width of the creek mouth, length of the main creek and cross-section of the creek at the sampling site were measured, they were not included in the analysis since they were correlated with the creek volume. The creek volume (-0.86) and the creek bottom elevation (-0.62) correlate well with the first ordination axis, while the amount of debris (0.82) and the number of creek branches (-0.76) correlate with the second axis. Freshwater species on the one hand and marine migrant and estuarine species on the other seem to be segregated along the vector of the volume. The highest abundance of seabass, herring and common goby was found in the creek with the largest volume (NOT5). Marsh creeks that are higher in the tidal frame and contain more woody debris are situated in the upper left part of the ordination plot. Less fishes were caught in these creeks. Eel and flounder are located near the centre of the ordination, indicating they were unrelated to the variables or related to their intermediate values (Fig. 6.8). The highest number of fish was caught in NOT 5 and NOT 9 correlated respectively with volume and number of branches.

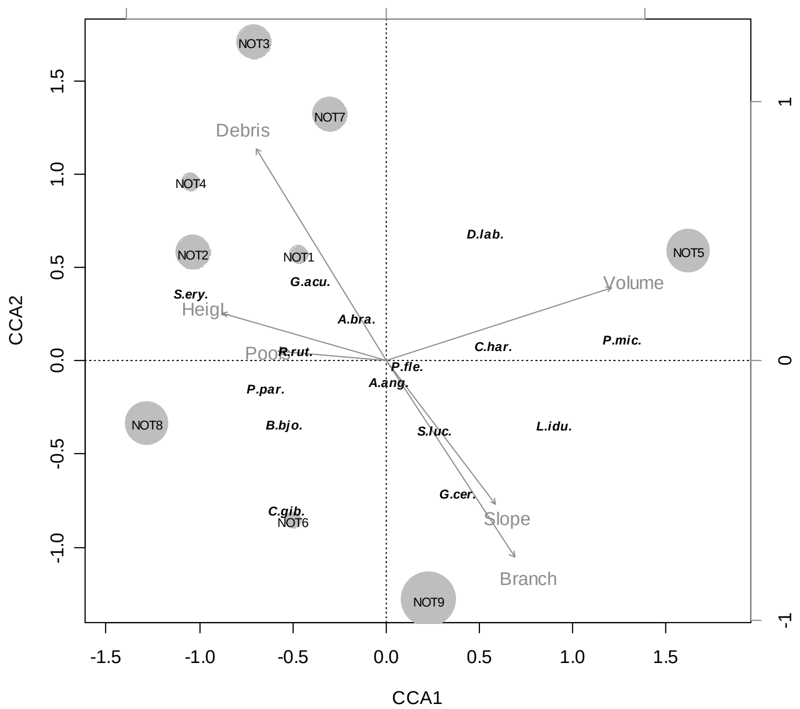

Figure 6.8: Triplot based on a CCA (eigenvalues of the first and second axis are 0.20 and 0.09 respectively) of the averaged fish abundance data in 2008 and creek characteristics of the Notelaar marsh. The diameter of each circle relates to the total number of fishes caught in that creek (abbreviations see Tables 6.2 and 6.3).

4 Discussion

In 2007 eight different species were collected in the creeks of Schor Ouden Doel, while 37 different species frequent the related subtidal mesohaline zone (Guelinckx *et al*., 2008). This indicates that only a limited portion of the fish community present in the neighbouring main channel uses the mesohaline tidal marsh. Cattrijsse *et al*. (1994) observed also that in the mesohaline zone only a small portion of estuarine inhabitants visit a nearby tidal marsh. Observation in Portugal contradicts the assertion that fewer species use tidal marshes (Vieira *et al*., 2002). Previous research (Hampel *et al*., 2004) reported only four different species in a tidal marsh neighbouring SOD: flounder (*Platichthys flesus*), seabass (*Dicentrarchus labrax*), eel (*Anguilla anguilla*) and herring (*Clupea harengus*). Unpublished results indicate that juveniles of flounder, herring, Gobiidae and thinlip mullet were caught in the tributaries of the creeks in the mesohaline marsh, confirming observations by others (e.g. Manderson *et al*., 2004) that juveniles avoid predation by swimming up into the shallow creeks (see also Hampel *et al*., 2003). The observed seasonal difference (Fig. 6.4) corresponds with Hampel *et al*. (2004) who observed a maximum catch in summer. Also Cattrijsse *et al*. (1994) recorded a maximum number of species using the creeks in a tidal brackish marsh in summer and autumn. Similar observations were made in the oligohaline tidal marsh the Notelaar (NOT). This high species richness in summer can be explained as the result of spawning activities in the sea (Martinho & Able, 2003) and an increase in salinity due to a decrease in discharge (Maris *et al*., 2008) so that marine species penetrate farther upstream the estuary (Araújo *et al*., 1999). In the freshwater tidal marsh GREM ten fish species were caught in 2007 with eel as most abundant species. Hampel *et al*. (2004) collected in the same marsh during a five months survey in 2000 only eel, carp (*Cyprinus carpio*) and spirlin (*Alburnoides bipunctatus*). The latter species is possibly a mistake in determination because it was never caught otherwise in the Zeeschelde. These authors attributed the low catches to the presence of a high amount of organic matter, also trapped in the nets, and the poor density, diversity and species richness in the tidal freshwater part of the Schelde as reported by Maes *et al*. (1997). In 1995 Maes *et al*. (1997) collected seven different species in the main channel of the tidal freshwater. In 2007 we collected 29 different species in the same tidal freshwater zone of the Zeeschelde. Since 1996 an improvement of the dissolved oxygen concentration is observed (Maris *et al*., 2008), leading to an increase of species in the whole estuary (Maes *et al*., 2007, 2008). In the Groot Schor (HAM) a higher catch per unit effort (10) was recorded compared to Groot Schoor GREM (6.3) which can partly be due to creek dimensions. In GREM the species richness is lowest.

As expected the tidal marshes studied are clearly separated along the salinity gradient (Fig. 6.6) (see also Gelwick *et al.*, 2001; Hampel *et al.*, 2004). França *et al.* (2009) assessed different estuaries along the Portuguese coast and suggested that each estuarine habitat type may contain a specific fish assemblage. The mesohaline marsh is separated from the other marshes because of the presence of the marine migrant seabass (*Dicentrarchus labrax*) and herring (*Clupea harengus*) and diadromous flounder (SOD in Fig. 6.6). The oligohaline marsh was not separated because of a typical oligohaline fish community as observed by Odum *et al.* (1988), but due to the juveniles (average length 8.0 cm) of thinlip mullet, a diadromous species (NOT in Fig. 6.6). The exotic pike-perch (*Sander lucioperca*) is a regular visitor of those NOT creeks. Unpublished results of stomach analyses reveal that pike-perch predates on juvenile flounder. Species length was on average 8.8 cm with a maximum recorded length of 19 cm. In September 2008 an increase in the marine species herring and seabass was recorded in NOT. At the same time thousands of shrimps (*Crangon crangon* and *Palaemonetes* sp.) were recorded on the adjacent mudflat (unpublished data). The salinity, measured as conductivity, was higher in September 2008 than in previous surveys (1550 compared to 786 $\mu S\ cm^{-1}$ in June, average monthly values, T. Maris pers. comm.). Probably juvenile herring and seabass followed the shrimps on which they prey. In the freshwater marshes the dominant species was eel which agrees with previous observations (Hampel *et al.*, 2004). The presence of freshwater species, e.g. rudd (*Scardinius erythrophthalmus*), bream (*Abramis brama*), Prussian carp (*Carassius gibelio*) and roach, differentiates the freshwater marshes (GREM and HAM) from the others.

Differences in fish catches between the creeks in the Notelaar (NOT 1-9) are apparent (Figs. 6.7 and 6.8). During the 2008 campaign 16 species were caught in the large wide creeks in the Notelaar and five were caught in the tributaries with eel fykes (Table 6.3). *Perca fluviatilis* was caught with eel fykes but not in the main creek. Rozas and Odum (1987a,b) related the observed differences between different creeks to the distribution of submerged aquatic vegetation. In the Notelaar there is no submerged aquatic vegetation and differences are due to creek characteristics. Most of the tributaries are higher situated in the tidal frame compared to the main creek, thus flooded later and for a shorter inundation time. One particularity is that the creeks which are most used by fish (NOT5 and NOT9) have rivulets on the mudflats (Fig.

6.2). NOT4 has also a rivulet but is less used as it is small. These rivulets flood earlier during rising tide and act as corridors for fish to reach the creeks (see also Rozas *et al*., 1988). Further research is needed to assess the importance of these rivulets.

Our analyses of the relation of creek characteristic and species composition shows that elevation of the creek mouth (HeigL) and dimension (Volume) are the two variables influencing species abundance (see also Rozas *et al*., 1988). Our results are in agreement with observations of Allen *et al*. (2007) that shallow broad creeks which fill and empty slowly support the greatest use. Such creeks offer better protection from predators than deep creeks. Creeks that are higher in the tidal frame are flooded for a shorter period and are less frequented by fish than lower creeks. According to Cattrijsse *et al*. (1994) fish migrate in and out the marsh during the first and last hour of the tidal cycle and the shorter the flood period the shorter the availability of the marsh. These higher creeks have also a steep slope profile while fish preferentially visit creeks that have gently sloping profiles (McIvor & Odum, 1988). In this last category flow is less abrupt and probably the feeding opportunities are enhanced in such creeks which have gently sloping profiles. We established the presence of different macroinvertebrates in the creeks e.g. of Gammaridae, Asellidae, Diptera larvae, Oligochaeta. The impact of debris present is not clear because most debris was encountered in the smaller creeks. We only assessed a few creek characteristics and more research is needed to define why certain creeks are more attractive and support higher fish densities than adjacent ones (e.g. Kramer & Chapman, 1999). The tidal flooding in the Zeeschelde has an asymmetric pattern (Cattrijsse *et al*., 1994) and tidal marshes are elevated in the tidal frame due to the coastal squeeze effect. Therefore during every tidal cycle the creeks fall completely dry because of their elevated position versus the mean low water level. Patches of small pools remain and some species stay there. Pioneer species such as three-spined stickleback and stone moroko (*Pseudorasbora parva*) even use these pools as a nursery (personal observations).

The complementary catches with seine netting and double fykes on the adjacent mudflat show that only part of the fish species present in the estuary use the tidal marshes (Table 6.3). Fourteen species were caught on the mudflat while only six species were caught in the same tidal cycle in the adjoining creeks (NOT 1 & 2): bream, three-pined stickleback, flounder, ruff (*Gymnocephalus cernuus*), eel and roach. This ratio is comparable with the ratio observed in the Tejo saltmarshes, Portugal (Salgado *et al*., 2004). França *et al*. (2008) evidenced that mudflats in the Tagus are feeding and nursery areas for several nektonic species. In the Ems (The Netherlands) Hiddink and Jager (2002) found substantial numbers of Nilsson's pipefish (*Syngnathus rostellatus*) seeking food on the tidal flats during high tide. Different length classes of bream were recorded, from 0+ till adults (> 20 cm), on the mudflat while in the creeks only juveniles (<10 cm) were caught. Only juvenile flounder was caught in the creek, while larger specimen (>20 cm) were collected on the mudflat. Juvenile roach visit the creek and occasionally a large individual (>20 cm) was caught. This can be an indication that juveniles of these species use the creeks as refugia and/or feeding place. These complementary catches indicate that it is necessary to develop a sampling protocol with higher resolution to assess in detail migration patterns of the fish from the head river over the tidal mudflats into the marsh. Differences between night and day catches will provide extra information about the drivers behind the observed movements (e.g. Morrison *et al*., 2002).

5 Conclusions

The most abundant fish species in the mesohaline tidal marsh were flounder and seabass while in the oligohaline tidal marsh flounder was dominant. In the tidal freshwater marshes eel, Prussian carp and roach were the most abundant species. The spatial gradient in the species distribution is linked with a salinity gradient. We did not assess the juvenile movements and therefore can not define a nursery function (Beck *et al*., 2001). However, the high number of juveniles caught suggests that the creeks are a juvenile habitat. The highest fish abundance is caught in summer (after hatching) mainly because juveniles seek then shelter in the creeks (Jin *et al*., 2007). These preliminary results indicate that the visit frequency of the creeks depends on the flood duration and creek dimensions. The lower the creek is positioned in the tidal frame, the larger its branches and more permanent its pools, the more interesting for juvenile fish. These data are important for the protection and restoration management of tidal marshes.

Our results indicate that a limited number of fish species use the tidal marshes as a feeding or shelter place. With the improvement of the water quality a change in fish assemblages takes place (see Chapter 2) combined with an increase of the use of tidal marshes. Further research is needed to assess the drivers of those migrations towards the creeks (e.g. predation avoidance), to investigate difference between night and day migrations, feeding behaviour in the creeks (e.g. stomach contents and food availability), the effect of tidal cycles (e.g. spring tide floods), the importance of microhabitats for fish larvae (e.g. shallow pools), additional geomorphological features of creeks including rivulets on the adjoining mudflats and to compare the fish assemblages on neighbouring mudflats with the species which visit the creeks.

Acknowledgements

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Chapter 7

A fish-based assessment tool for the ecological quality of the brackish Schelde estuary in Flanders (Belgium)

Jan Breine, Joachim Maes, Paul Quataert, Erika Van den Bergh, Ilse Simoens, Gerlinde Van Thuyne & Claude Belpaire

Abstract

This study presents a new approach to define an optimal combination of candidate metrics for composing a fish-based Estuarine Biotic Index (EBI). One of the key ideas was that a powerful index should simultaneously minimise two prediction errors: falsely declare the status of a site as disturbed while it is not (Type I error) and the opposite, falsely declare a disturbed site as undisturbed (Type II error). The balance between both errors is an inherent characteristic of an index and can be displayed as a curve. The area under this curve (AUC) is a measure of the misclassification rate: the smaller, the better. A stepwise approach was therefore used whereby in each step a metric resulting in the highest reduction of AUC was added to the model. Five metrics were selected and the distribution of their average was the basis to define the thresholds for the classes of the EBI. This chapter presents the fish-EBI for the brackish Schelde estuary in Flanders (Belgium). The index was calibrated against fyke net data from five sites collected during the period 1995 to 2004. These sites ranged in quality from moderately to highly impacted, i.e. classes 3 to 5 respectively. Despite the absence of the highest classes 1 (high) and 2 (good) at the sampling sites, the EBI presented can serve as an evaluation tool of the highly impacted situation in the Zeeschelde estuary as it makes a good discrimination between moderate and highly impacted sites. In addition, its definition complies with the biological status classes of the European Water Framework Directive (WFD).

Key words: brackish estuary, fish-based index of biotic integrity, Schelde, Flanders

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1 Introduction

Worldwide, estuaries suffer from ever increasing human pressure (Dennison *et al.*, 1993; Simenstad & Cordell, 2000). Representing a transition zone between land and ocean, estuaries are subject to the input of high loads of inorganic and organic compounds, leading to water quality impairment. Land claim of valuable intertidal areas for industrial development or agriculture causes further deterioration of estuarine ecosystems. Estuaries also serve many important ecological functions including nutrient transformation, in particular nitrogen and carbon. This role of estuaries has been appreciated on a global scale by Costanza *et al.* (1997) who ranked estuaries among the world's most important ecosystems in terms of ecological services provided. Besides their key role in nutrient cycling and transformation, estuaries are essential habitats in the life history of many species and in particular of fish and waterfowl. They are considered as important nurseries for the juveniles of many marine, estuarine and freshwater fishes since they promote growth and offer shelter from predators (Elliott *et al.*, 1990; Hostens & Mees, 1999; Maes, 2000; Elliott & Hemingway, 2002). Furthermore, estuaries are crucial resting areas for transient fish species, in particular diadromous fish populations, many of which are threatened and of conservation status. The assessment of the condition of estuarine ecosystems for the proper management of estuarine resources requires the collection of physical, chemical and biological data and knowledge about how these different components interact. Managers usually require such information to be translated into a simple value or index, which evaluates the current state in relation to a pristine state and which can be presented and used for further decision making. Different methods have been developed to assess the water quality and ecosystem condition of estuaries. Bioassessment methods are recently often preferred above more classic methods that rely on the measurement of physical and chemical variables, since bioassessment provides the possibility to evaluate the condition of the environment without having to measure the full complexity of the system (McLusky& Elliott, 2004). Bioassessment protocols, reflecting both short and long term effects, are relatively inexpensive and are easily performed. Estuarine bioassessment protocols use submersed aquatic vegetation (Dennison *et al.*, 1993), diatoms (Bate *et al.*, 2004), benthos (Weisberg *et al.*, 1997; Van Dolah *et al.*, 1999; Llanso *et al.*, 2002), fish (Deegan *et al.*, 1997; Harrison *et al.*, 2000; Hughes *et al.*, 2002; Whitfield & Elliott, 2002; Coates *et al.*, 2004; Harrison & Whitfield, 2004; Moy, 2004) or a combination of these taxa (Cooper *et al.*, 1994; Fairweather, 1999; Borja *et al.*, 2004). In Europe, fish-based indices are becoming important bioassessment tools since the European Union (EU) water policy recommends fish as a biological quality element to be monitored as part of the assessment of ecological status of all water bodies, except coastal waters (WFD, 2000). Fish represent one of five biological elements that should be used to assess the quality of transitional waters (estuaries, lagoon, rias, fjords, etc). In particular, data on species composition and abundance of the ichthyofauna should be used to report the ecological status of European estuaries. As a result, several fish-based indices for estuaries in Europe have been presented so far (Borja *et al.*, 2004; Coates *et al.*, 2004; Jager & Kranenbarg, 2004; Salas *et al.*, 2004). This study presents a fish-based estuarine biotic index (EBI) to assess the ecological status of the tidal brackish Schelde estuary. In addition a new approach is described that can be used to develop indices of biotic integrity in general.

2 Materials and methods

2.1 The calibration dataset

2.1.1 Description of the estuary (Fig. 7.1)

The river Schelde is a rainfall driven, lowland-river. It originates in Saint-Quentin (France) and discharges into the North Sea near Vlissingen (The Netherlands). The catchment area is 22,103 km^2, draining parts of France, Belgium and The Netherlands. The river has a total length of 355 km with a fall of about 100 m and the mean flow rate is 105 $m^3 s^{-1}$.

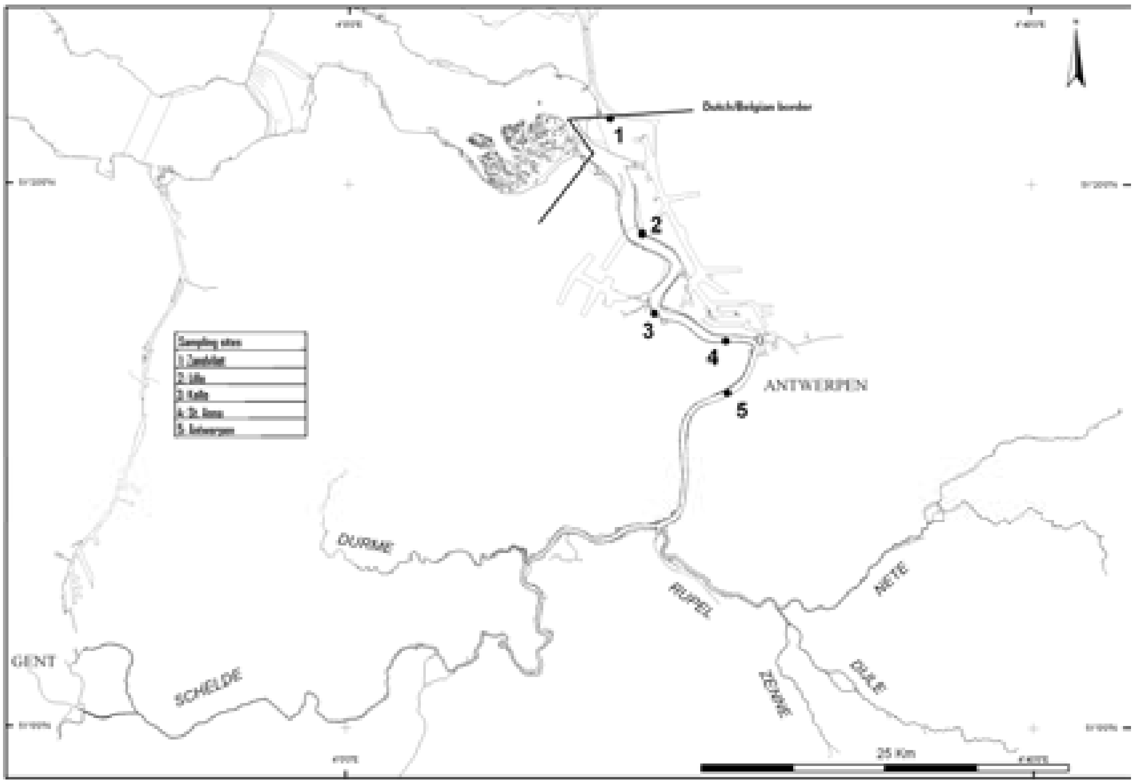

Figure 7.1: The sampled locations on the river Schelde (Belgium).

Tides with an average range of 4.5 m penetrate 160 km land inward, while the freshwater – saltwater boundary is situated about 100 km from the river mouth. The estuary consists of a lower part (Westerschelde) with multiple channels downstream the Belgian-Dutch border (between km 0 and km 58) and an upper part (Zeeschelde) with a single channel upstream the borderline (between km 58 and km 160) (Fig. 7.1). The assessed reach of the Zeeschelde is brackish with a well developed salinity gradient which is primarily determined by the magnitude of the river discharge (Baeyens *et al.*, 1998).

Approximately seven million people live in the river basin of the Schelde. The largest industrial areas are concentrated near Lille (France), Antwerpen and Gent (Belgium), and Vlissingen (The Netherlands). The river receives major discharges of industrial and domestic wastes, a substantial part of which are not treated resulting in very poor water quality in the larger part of the river, its tributaries and estuary.

2.1.2 Selection and pre-classification of the study sites (Fig. 7.1, Tables 7.1-7.3)

Between 1995 and 2004, five sites in the brackish part of the upper estuary were surveyed. The locations were selected such that they come from homogeneous segments of the river as defined in a concurrent modelling study (Hoffmann & Meire, 1997). To define the habitat quality class of each site (Hscore) a classification scheme based on a series of impact indicators was used. To identify anthropogenic activities Aubry and Elliott (2006) defined environmental indicators and grouped them into three broad indices. We applied these indicators to the Zeeschelde and selected the indicators which are most relevant for fish (Elliott *et al*., 2008c). A first important indicator was the average minimum dissolved oxygen saturation (DO%) for the years 2000-2004 obtained from Van Damme *et al.* (2005). Benthos was used as a biological indicator and scored according to Brys *et al.* (2005). Intertidal area loss (%) and land claim (%) were estimated with respect to the intertidal surface in 1960 and old maps from 1900 respectively (Van Braeckel *et al*., 2006). These two baseline years were selected based on available data. Intertidal area loss is defined as loss in the area covered by intertidal habitat (tidal marsh and mudflats). It includes both man-induced and natural variations.

Table 7.1: Classification indicators and threshold values for PSite (pressure score) to score and derive the habitat class (Hscore, Table 7.2) adapted from Aubry & Elliott (2006).

Parameter	**Score**				
	1	**2**	**3**	**4**	**5**
Minimum DO saturation monthly average (%)	>80	80 & >70	70 & >50	50 & >30	≤30
Benthos	Classification explained in Brys *et al.*, 2005				
Intertidal area loss (%)	0	<20	≥20 & <30	≥30 & <50	≥50
Land claim (%)	0	<5	≥5 & <40	≥40 & <60	≥60
Port & marina activities (absence/presence)	No				Yes
Industrial activities (degree)	Low		Moderate		High
Dredging activities (absence/presence)	No				Yes
Total score: PSite	7	8 - 14	15 - 21	22 - 28	29 - 35

Table 7.2: Conversion of classification indicator score (PSite) into Habitat score (Hscore) and appreciation.

PSite Total score	7	8 - 14	15 - 21	22 - 28	29 - 35
Hscore	5	4	3	2	1
Appreciation (quality)	Very high	High	Moderate	Poor	Bad

Table 7.3: Values and scores (in bold between brackets) for the classification indicators and the resulting habitat quality class (Hscore) for the five assessed sites (number of pooled fish surveys in italics between brackets) in the Zeeschelde estuary.

	Mesohaline									
	Zandvliet (*83*)		**Lillo (*4*)**		**Kallo (*11*)**		**St. Anna (*7*)**		**Antwerpen (*25*)**	
Minimum DO saturation monthly average (%)	60.0	(**3**)	49.3	(**4**)	39.2	(**4**)	24.0	(**4**)	22.0	(**5**)
Benthos		(**3**)		(**3**)		(**3**)		(**3**)		(**4**)
Intertidal area loss (%)	18.2	(**2**)	33.3	(**4**)	51.8	(**5**)	51.8	(**5**)	55.7	(**5**)
Land claim (%)	71.0	(**5**)	33.3	(**3**)	58.7	(**4**)	58.7	(**4**)	67.4	(**5**)
Port & marina activities (absence/presence)	No	(**1**)	No	(**1**)	No	(**1**)	No	(**1**)	Yes	(**5**)
Industrial activities (degree)	Low	(**1**)	Mod.	(**3**)	Mod.	(**3**)	Mod.	(**3**)	High	(**5**)
Dredging activities (absence/presence)	Yes	(**5**)	Yes	(**5**)	Yes	(**5**)	Yes	(**5**)	Yes	(**5**)
PSite (Total score)		20		23		24		24		34
Hscore		**3**		**4**		**4**		**4**		**5**

Land claim is the reclamation of estuarine tidal marsh to provide land for industrial development. Aerial photographs allowed to assess industrial and port or marina activities (absence/presence). Dredging activities (absence/presence) were provided by the MOW - Department of Mobility and Public Affairs, division Maritime Access. Each indicator was scored between 1 and 5 by thresholds suggested by experts (Table 7.1). These experts belong to the Research Institute for Nature and Forest (for benthos, land claim and intertidal area loss), the Biology Department of the University of Antwerpen, (for dissolved oxygen saturation values, port and marina activities, benthos) and the Provincial Fishery Commission of Antwerpen (dredging activities). Industrial activity information was obtained from the direct discharges (OC GIS- Flanders and VMM, Flemish Environmental Agency). The thresholds were based on expert judgement and logical argumentation. The main point was not to obtain an absolute expression of the quality, but to have a good ranking with respect to human impact. As a global summary, the sum of these scores (PSite) was taken and converted as Hscore (Tables 7.2 & 7.3). Table 7.3 shows that none of the sites are undisturbed as the results of the classification ranged from moderately impacted (class 3) to heavily impacted (class 5). The quality increased from Antwerpen (class 5) downstream to the Dutch border (Zandvliet, class 3). This quality trend coincided more or less with an increasing salinity. As all sampling sites are situated in the brackish part of the upper estuary, it is hypothesized that, with respect to the fish community, the impact of the salinity gradient is minor and that the human impact gradient will dominate. In chapter 2 it was shown that species richness in the different zones increased concordant with the improvement of the water quality. Changes in fish assemblages within the mesohaline zone are gradual unless human impact occurs (Greenwood, 2007).

2.1.3 Fish sampling

All field work was done by trained fish biologists using a standardized protocol. Between 1995 and 2004 fishing occurred for two successive days at each site covering all seasons. At each sampling site a pair of double fyke nets (type 120/80) were positioned at low tide and emptied the next day. Fish captured were identified on site using a field guide (Nijssen & De Groot, 1987). Quality assurance of the identifications was performed by occasional cross-examination in the laboratory. Data recorded included species-specific frequencies, individual total lengths (±1 mm) and wet weights (±1 g). After screening the data for missing values and outliers as well as pooling by averaging the catch per unit effort transformed (CPUE) data over one month, we retained 130 fishing occasions for the analysis.

2.2 Basic statistical concepts and tools for the calibration

2.2.1 Screening for responsive metrics showing a monotone relation with human pressure

Metrics which will be retained and are related with the fish assemblages have to be responsive to human pressure and disturbance. Positive metrics increase with increasing habitat quality and decrease with disturbance (e.g. total number of intolerant species) and vice-versa for negative metrics (e.g. % omnivorous species). However, a non-linear relationship can also occur in which, for instance, a metric value first increases with decreasing quality and then finally drops again (e.g. biomass). Because such complex relationships require more data to be analysed and as our data are somehow limited, we only considered monotonous ones (positive or negative).

2.2.2 The scoring system based on quintiles (as a modification of the trisection method)

In order to combine the metrics into a single score, it is necessary to re-scale (standardise) them. Therefore, all metrics were scored from 0 (low) to 1 (high) by judging each metric value with respect to a reference distribution representing the natural variation of the metric in an unimpacted ecosystem. To define five quality intervals (quintiles) of that distribution four thresholds were defined so that the score increased in steps of 0.25. For a positive metric score, 1 was given if the value was above the highest quintile, 0.75 if it was above the second highest quintile and so on until 0 if it fell below the lowest quintile. For a negative metric, the opposite held. For the reference distribution we used the best sites available (Hscore of 3). As a consequence, the scope of the index is reduced from 3 to 5, but the advantage is that the calibration process is based fully on available data.

2.2.3 The balance between type I and type II error: area under the error curve (AUC)

A biotic index is developed as a tool to discriminate between disturbed and undisturbed sites. Accordingly, two prediction errors are possible: the index can predict the site as disturbed, when in fact it is not (type I error) and, conversely, an index can declare incorrectly a disturbed site as undisturbed (type II error). The occurrence of both errors should be kept as small as possible. A type I error implies false alarms while with a type II error disturbed sites will be left undetected and untreated. In addition, type I and type II errors are interlinked in a one-to-one relation as for a given index, one cannot decrease the type I error without increasing the type II and vice versa. Increasing the type I error results in a more conservative classification of sites as disturbed. The error curve (Fig. 7.2) shows this relationship and indicates graphically the detection capacity of the index. The closer the curve is to the origin, the smaller the error and the higher the chance that there is a good balance between the type I and type II error, i.e. keeping them both small at the same time (compare index A, B and C in Fig. 7.2). This feature can therefore be expressed as the area under the error curve (AUC, max value=0.5) in which the smaller this area, the lower the overall level of misclassification and the higher the quality of the index.

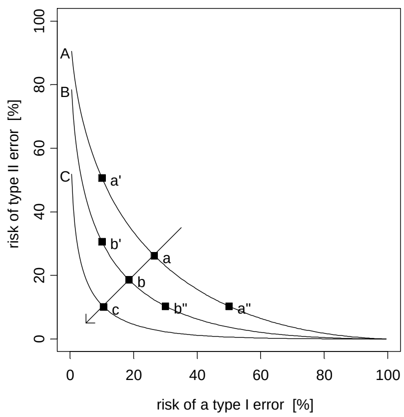

Figure 7.2: The error curve showing the trade off between type I and type II error for three hypothetical indices a, b and c. The change from a to c shows the effect of the stepwise introduction of metrics.

2.3 The strategy to calibrate the index

Figure 7.3 gives an overview of the different steps in the development of the EBI. Starting from the candidate metrics an optimal set of metrics was stepwise selected by judging their combined response with respect to the pre-classification.

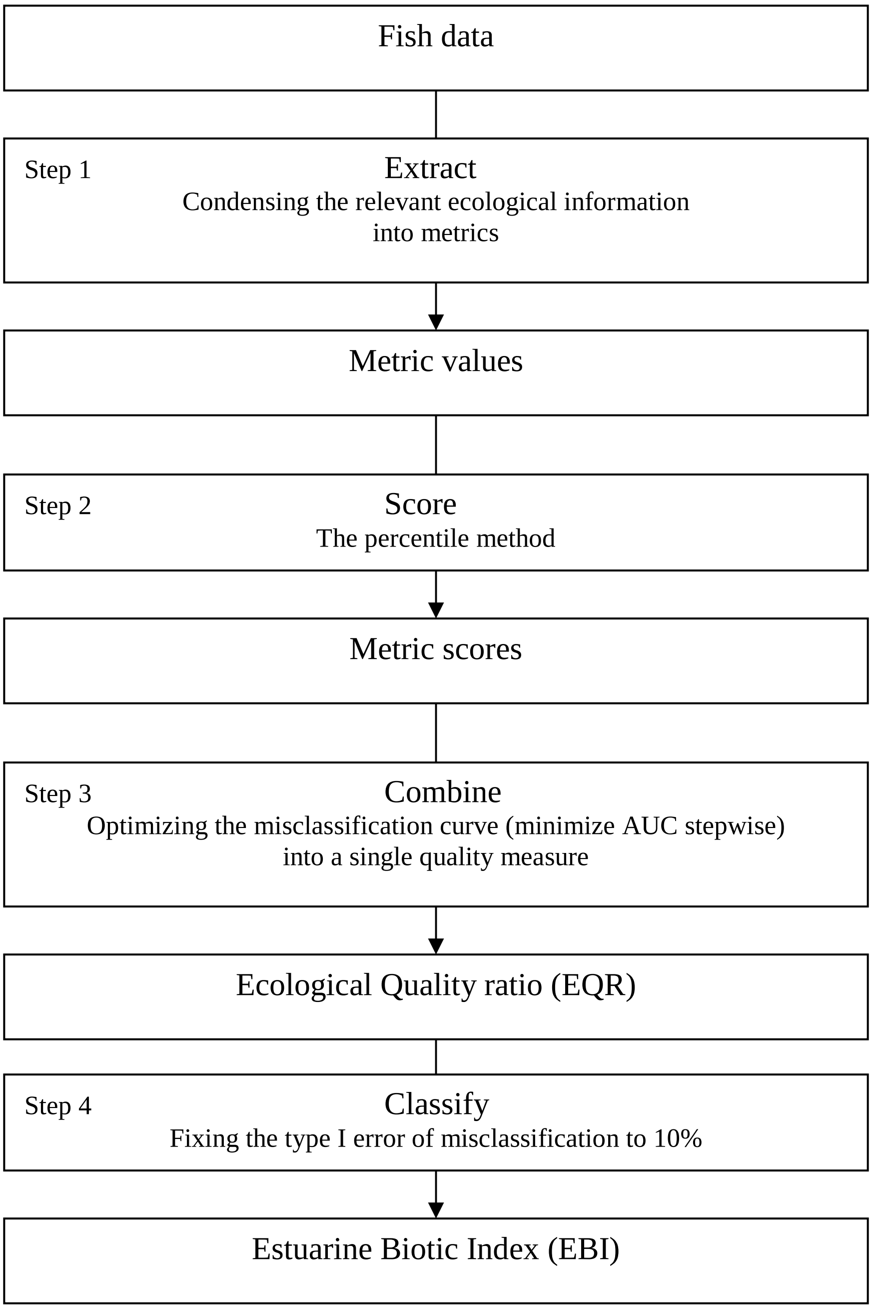

Figure 7.3: The different steps of the calibration of the Estuarine Biotic Index.

2.3.1 Composing the set of candidate metrics (Step 1, Fig. 7.3)

The first step is the selection of relevant candidate metrics based on ecological theory and empirical findings. Candidate metrics should include information about the diverse ecological functions of estuaries for fishes, such as providing spawning and nursery area and connection between the ocean and upstream zones. Using literature and expert judgement, species were categorized into different ecological guilds to delineate their preferred habitat, indicating if it concerns estuarine species and marine juvenile migrants (Elliott & Hemingway, 2002), trophic status (Elliott & Hemingway, 2002), tolerance value (the higher its value the less tolerant, Breine *et al.*, 2001), migration behaviour and feeding stratum (Table 7.4). Once assigned, species were not allowed to change between different guilds (Elliott & Dewailly, 1995) except for the ontogenetic shift of trophic status for some species. A first evaluation of these candidate metrics consisted in assessing each metric for its range (McCormick *et al.*, 2001).

Table 7.4: Fish species encountered in the Schelde estuary between 1995 and 2004 and their guilds.

Scientific name	**Habitat**	**ER**	**MJM**	**Trophic guild**	**TV**	**Stratum**
Abramis brama	Fw			OMN	1	De
Acipenser baeri	Di			INV/PISV	3	De
Agonus cataphractus	M	Y		INVV	3	Be
Alburnus alburnus	Fw			OMN	2	Pe
Alosa alosa	Di			PLAV	4.5	Pe
Alosa fallax	Di			PLAV	3.5	Pe
Ammodytes tobianus	M	Y		INVV	2	Be
Anguilla anguilla	Di			OMN	2	Be
Aphia minuta	M	Y		INVV	3	Pe
Arnoglossus laterna	M			PISV/INVV	2	Be
Atherina presbyter	M		Y	INVV/PISV	2	Pe
Belone belone	M			PISV	4	Pe
Blicca bjoerkna	Fw			OMN	2	Pe
Callionymus lyra	M			INVV	2	Be
Carassius auratus gibelio	Fw			OMN	0	Pe
Carassius carassius	Fw			OMN	2	Pe
Ciliata mustela	M			INVV	2	Be
Chelidonichthys lucernus	M		Y	PISV	3	Pe
Clupea harengus	M		Y	INVV	1	Pe
Conger conger	M			PISV/INVV	3	Be
Cottus gobio	Fw			INVV/PISV	4	Be
Cyclopterus lumpus	M			INVV	2	Be
Cyprinus carpio	Fw			OMN	2	De

Scientific name	Habitat	ER	MJM	Trophic guild	TV	Stratum
Dicentrarchus labrax	M		Y	INVV/PISV	3	De
Engraulis encrasicolus	M			INVV	1	Pe
Echiichthys vipera	M			OMN	2	Be
Esox lucius	Fw			PISV	4	De
Gadus morhua	M		Y	OMN	4	De
Gasterosteus aculeatus	Di			OMN	1	Pe
Gymnocephalus cernuus	Fw			INV	2	Be
Hippocampus ramulosus	M	Y		INVV	2	De
Hyperoplus lanceolatus	M			INVV/PISV	2	Be
Lampetra fluviatilis	Di				3	Be
Leucaspius delineatus	Fw			INVV	3	Pe
Leuciscus idus	Fw			OMN	4	Pe
Limanda limanda	M		Y	INVV	4	Be
Liparis liparis	M	Y		INVV	3	Be
Liza ramado	Di			OMN	2	Pe
Merlangius merlangus	M		Y	OMN	3	De
Misgurnus fossilis	Fw			INVV	3	Be
Mullus surmuletus	M			INVV	2.5	Be
Myoxocephalus scorpius	M	Y		OMN	2	Be
Osmerus eperlanus	Di			INSV/PISV	4	Pe
Perca fluviatilis	Fw			INSV/PISV	2	Pe
Petromyzon marinus	Di			PARA/PISV	4	De
Platichtys flesus	M	Y		INVV/PISV	2	Be
Pleuronectes platessa	M		Y	INVV	3.5	Be
Pomatoschistus lozanoi	M			INVV	3.5	Be
Pomatoschistus microps	M	Y		INVV	2	Be
Pomatoschistus minutus	M	Y		INVV	3	Be
Psetta maxima	M		Y	PISV	3	Be
Pseudorasbora parva	Fw			OMN	0	Pe
Pungitius pungitius	Fw			OMN	1	Pe
Raja clavata	M			INVV	3	Be
Rhodeus sericeus amarus	Fw			PLAV	4	Pe
Rutilus rutilus	Fw			OMN	1	Pe
Salmo salar	Di			INSV/PISV	5	Pe
Salmo trutta	Di			INSV/PISV	5	Pe
Scardinius erythrophthalmus	Fw			OMN	4	Pe
Scomberesox saurus	M			PLAV	2	Pe
Scophthalmus rhombus	M		Y	PISV	3	Be
Solea solea	M		Y	INVV	4	Be
Syngnathus acus	M	Y		INVV	2	Be
Syngnathus rostellatus	M	Y		INVV	2	Be
Trachurus trachurus	M			PISV/INVV	2	De
Trisopterus luscus	M		Y	OMN	3	De
Zoarces viviparus	M	Y		INVV	2	Be

(**Habitat**: Fw: freshwater species; Di: diadromous species; M: marine species; **ER**: Estuarine resident; MJV: Marine juvenile migrating; **TG** (Trophic guild): OMN: omnivorous; INVV: invertivorous; PISC: piscivorous; PLAV: planktivorous; INSV: insectivorous; PARA: parasites; **TV**: Tolerance value; **S** (Stratum): Be: benthic; De: demersal; Pe: pelagic).

2.3.2 Graphical screening by boxplots and scoring (Step 2) (Fig. 7.4)

The aim is to select the responsive metrics. A simple way to explore graphically the response of the candidate metrics to environmental pressures are boxplots which show how the metric distribution changes along the habitat scores. If no gradient was obvious or the distributions were not well-separated, a metric was considered to be omitted from the list although in this first screening, rules to exclude were not applied rigorously as it is possible that metric which at first sight looks less optimal, gives invaluable information in combination with other metrics. In general, also metrics that separated only between class 4 and 5 were therefore accepted at this stage. This graphical analysis was the basis of the scoring system, although no data were available from high quality sites. Therefore, as a surrogate, we took as a reference the empirical distribution of the metric at the best sites available (moderately disturbed, Hscore = 3). The consequences of this approach are discussed later (Section The range of the habitat status (Hscore)).

2.3.3 Searching for the optimal combination (subset) of metrics (Step 3) (Fig. 7.5).

The metrics were combined into a single index by taking their average, further referred to as the ecological quality ratio (EQR) having a value between 0 and 1. Again, 0 indicated the lowest and 1 the highest quality. However, including all scored metrics in the index is not appropriate as some metrics are redundant. In addition, adding many similar metrics can increase noise, such that the final result is less optimal than for a smaller set. Hence, given these disadvantages, it was necessary to find the optimal subset of metrics. To cope with this problem, a forward stepwise approach was adopted. The first variable included into the model was the metric with the smallest error (best balance between type I and type II error) as measured by the AUC. Subsequently, a metric was added which most of all decreased further the AUC. This process was repeated until all metrics were included. The result is a scree plot showing the gain by the successive introduction of metrics (Fig. 7.5). The combination of the metrics having the lowest AUC was considered as the optimal set.

2.3.4 Classify the EQR into the EBI (Step 4) (Fig. 7.6)

In order to comply with the WFD, the EQR ranging between 0 and 1 requires to be translated into a five-class system by introducing four thresholds. However, as we do not have high quality sites, we could only calibrate the system for classes 3, 4 and 5 and so only two thresholds needed to be fixed (between 3 & 4 and 4 & 5). Here, we fixed the type I error of each threshold at 10 %. To find the threshold between class 3 and 4, we observed the distribution of EQR for the habitat class 3 (Fig. 7.6) and selected the 10^{th} percentile point as threshold. A similar procedure was carried out for class 4 and 5.

2.4 Evaluation

2.4.1 The balance between the type I and the type II error

As the type I error was fixed at maximum 10%, by definition about 10% of the sites in class 3 will be classified as class 4 or 5 and approximately 10% of the sites of class 4 will be classified in class 5. For this selection it is also important to control the effect on the type II error. If these errors are too high, the index is not useful. Also it is important to observe how the 10% type I misclassification of class 3 is subdivided into small and large type I errors (classified in class 4 or 5 respectively).

2.4.2 Year and season effects

Seasonal effects on metric behaviour were assessed by comparing the different seasons between 1996 and 2001 at the stations Zandvliet (49 samples) and Antwerpen (19 samples) using a one-way analysis of variance (ANOVA).

3 Results

3.1 Candidate metrics

Sixteen candidate metrics were selected from the literature (Table 7.5) (Cooper *et al.*, 1994; Elliott & Dewailly, 1995; Deegan *et al.*, 1997; Costa & Cabral, 1999; Araújo *et al.*, 1999; Galatowitsch *et al.*, 1999; Williams & Zedler, 1999; Araújo *et al.*, 2000; Peterson *et al.*, 2000; Breine *et al.*, 2001; Gelwick *et al.*, 2001; Adriaenssens *et al.*, 2002a, 2002b; Bate *et al.*, 2002; Castillo-Rivera *et al.*, 2002; Hughes *et al.*, 2002; Thiel *et al.*, 2003; Borja *et al.*, 2004; Coates *et al.*, 2004) (Table 7.3). It was avoided to take in very similar metrics to minimize the chance of redundancy. As such total number of species without freshwater species was retained instead of total number of species as freshwater species are considered as stragglers in the mesohaline zone (Martino & Able, 2003).

Table 7.5: Candidate metrics and their predicted response to disturbances, if not generally used the source is added. (n: individuals).

Abbreviations	Candidate metrics	Response
MnsBra	Total number of species excluding freshwater species	↓
MpiFlo	% of flounder (n)	↓
MpiSme	% of smelt (n)	↓
MpiOmn	% of omnivores (n)	↑
MvaTol	Total tolerance value	↓
MpiPis	% piscivores (n)	↓
MpiErs	% of estuarine residents (n)	↓
MpiDia	% of diadromous fish (n)	↓
MpiMjm	% of marine juvenile migrating fish (n)	↓
MnsErs	Total number of estuarine resident species	↓
MvdSha	Shannon diversity H’ (Gelwick *et al.*, 2001)	↓
MvdSim	Simpson dominance index (Peterson *et al.*, 2000 (1/D ; D= $\sum p_i^2$)	↓
MnsBen	Number of benthic associated species	↓
MvdDiv	Simpson unbiased diversity index D = 1 – λ (Castillo-Rivera *et al.*, 2002)	↓
MpiExo	% of invasive fish (n)	↑
MnsDia	Number of diadromous species	↓

(↓: decrease in value; ↑: increase in value)

3.2 The calibration of the index

The boxplots (Fig. 7.4) indicated that four metrics showed no monotone response and were excluded: MpiDia, MpiErs, MvdSha and MvdSim.

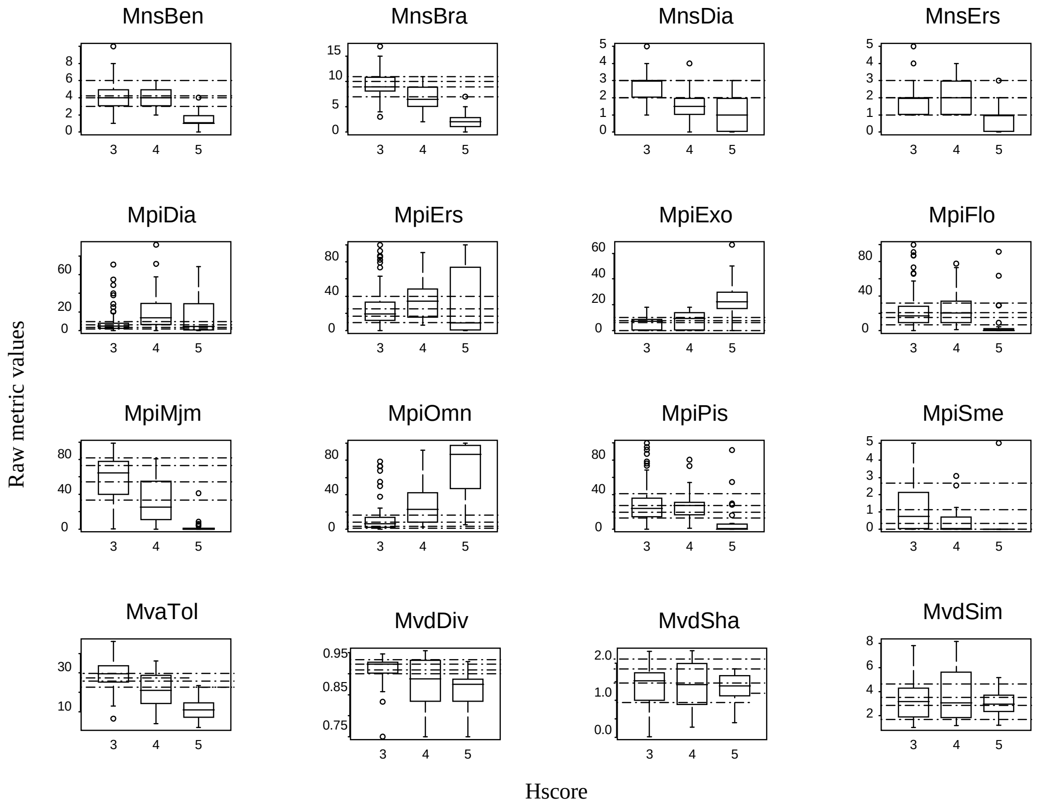

Figure 7.4: Candidate metrics as a function of the pre-classification of the habitat (Hscore) represented by boxplots + quintiles of the reference distribution to score (dotted lines) (for abbreviations of the metrics see Table 7.5).

Each interval contains an equal number of observations and some of the thresholds coincide due to zero values within these intervals.

MnsDia and MnsErs showed a rather small variation but were retained at this point because they are ecologically relevant for an estuary. Twelve metrics were accepted from which the stepwise procedure selected a final subset of five. Figure 7.5 and Table 7.6 give the thresholds to score these metrics.

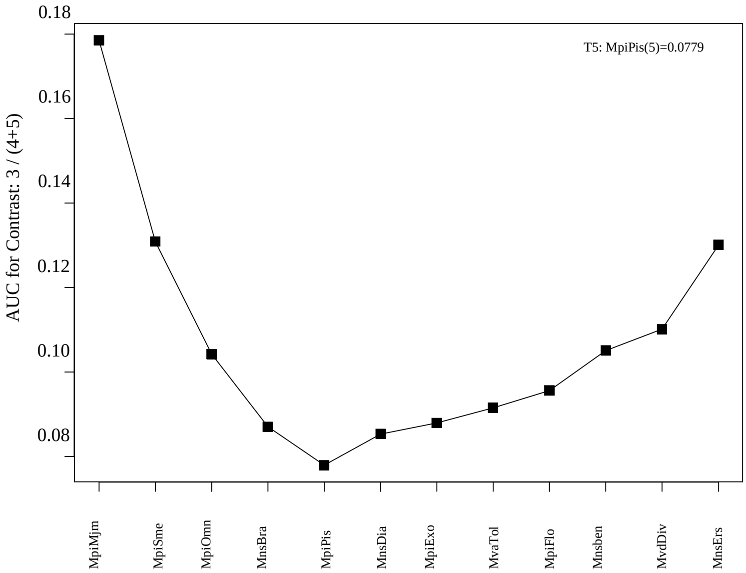

Figure 7.5: Scree plot showing the evolution of the area under the curve (AUC) by the stepwise introduction of the best performing metric (for abbreviations of the metrics, see Table 7.5).

Table 7.6: Selected metrics and threshold values (calculated as average monthly CPUE value, number of fish per fyke per day) for the fish-based Estuarine Biotic Integrity Index for the brackish Schelde estuary. (n: individuals).

Metric	**Score**				
	0	**0.25**	**0.5**	**0.75**	**1**
Species richness and composition					
% marine juvenile migrating fish (n) (MpiMjm)	≤33.0	>33.0	>54.2	>73.1	>82.0
% smelt (n) (MpiSme)	≤0.33		>0.33	>1.12	>2.68
Total number of species without freshwater species (MnsBra)	≤7	>7	>9	>10	>11
Trophic composition and habitat use					
% omnivores (n) (MpiOmn)	≥16.44	<16.44	<7.90	<3.37	<1.17
% piscivores (n) (MpiPis)	≤12.84	>12.84	>19.44	>27.23	>41.19

The boxplot in figure 7.6 shows the discriminating power of the resulting Ecological Quality Ratio (i.e. the average of these five metrics).

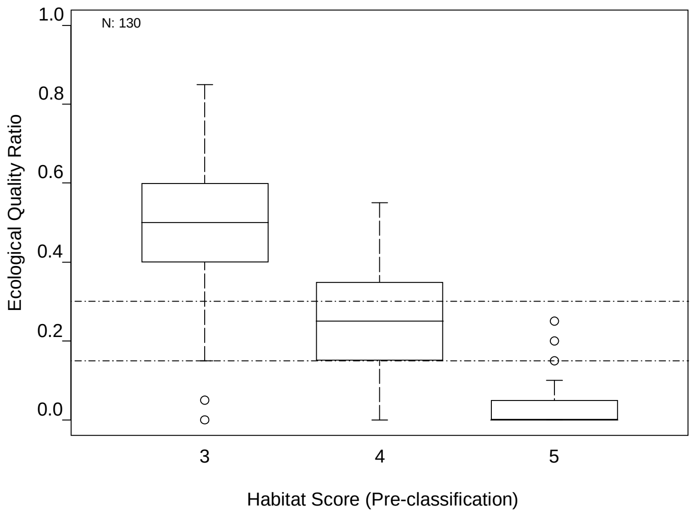

Figure 7.6: The distribution of the ecological quality ratio (EQR), i.e. the average of the scores of the 5 selected metrics, as a function of the pre-classification of the habitat (Hscore). The thresholds (dotted lines) are tuned such that the type I error is as close as possible to 10%.

The thresholds in figure 7.6, set to have approximately a type I error of 10%, are 0.30 and 0.15 (Table 7.7). The corresponding type II errors are given in Table 7.6.

Table 7.7: Estuarine Biotic Integrity Index (EBI) score ranges expressed as Ecological Quality Ratio values (EQR), their appreciation or integrity classes and colour code according to the European Water Framework Directive (WFD, 2000).

EBI score ranges	**Integrity class = WFD quality classes**	**WFD colour code**
	1 = High	Blue
	2 = Good	Green
>0.3	3 = Moderate	Yellow
>0.15	4 = Poor	Orange
≤0.15	5 = Bad	Red

3.3 Observed properties of the selected metrics (Figs.7.4 & 7.5)

3.3.1 Percentage of marine juvenile migrating individuals (MpiMjm)

The percentage of marine juvenile migrating individuals had the highest discrimination power. Deterioration in the habitat conditions of the estuary results in a decrease of the abundance of marine juveniles. The mesohaline zone is an ecocline with gradual changes in the fish assemblages and an abrupt change in these is considered as a human induced effect (Greenwood, 2007). The percentage of marine juvenile migrating individuals reflects hydrologic connectivity and marine habitat suitability. This is particularly the case when in the estuary the estuarine resident species are abundant but a low percentage of marine juvenile migrating specimens is recorded.

3.3.2 Indicator species smelt (MpiSme)

The indicator species smelt (*Osmerus eperlanus*, L.) metric is the next selected metric and its inclusion to the final model suggests that it is complementary to the MpiMjm. An indicator species should show a high degree of fidelity to a particular habitat and certain conditions (Fairweather, 1999). Smelt, was chosen as indicator because it is highly indicative in spending most of its life in the estuary.

3.3.3 Percentage of omnivorous individuals (MpiOmn)

In contrast to the other selected metrics, this trophic metric increases with disturbance. Omnivores are opportunists with a wide tolerance and so this metric increases its value with increasing disturbance.

3.3.4 Total number of species, freshwater species excluded (MnsBra)

The community structure responds to stress by a reduction in diversity, a dominance of opportunistic species and a reduction in body size (Gray, 1989). Fresh water species are excluded since they are not dependent on the mesohaline zone of the estuary (Martino & Able, 2003).

3.3.5 Percentage of piscivorous individuals (MpiPis)

Most piscivorous species are intolerant to environmental deterioration (Table 7.4) and this is reflected by a decrease in proportion of piscivores in impacted systems (Breine *et al.*, 2004). The boxplots show that this metric contributes only slightly to the discriminating power but that its contrast between class 4 and 5 probably slightly improves the index.

3.4 Metrics not included by the stepwise procedure (Fig. 7.5 versus Fig. 7.4)

The first metric not included was the number of diadromous species (MnsDia) as it showed too little variation to be discriminating. A similar argument holds for the number of estuarine species (MnsErs) which was entered as the last metric by the stepwise approach. The number of exotic species (MnsExo) on the other hand was also close to the optimum. However, a comparison with the percentage of omnivorous individuals (MpiOmn) showed a similar shape; hence the metric did not offer additional information. Another good candidate metric was the intolerance value (MvaTol) which although it showed a good response to disturbance (Fig. 7.4), was not included since it did not decrease the AUC. The remaining metrics (MpiFlo, MnsBen, MvdDiv) did not reduce the AUC, indicating that they provide no additional information.

3.5 Evaluation

3.5.1 Evaluation of the type I and II errors (Table 7.8)

For the moderate Hscore, the EBI misclassified 10% of the sites what is inherent to our approach. Only 4% were misclassified as bad (large type I error). None of the bad status locations were classified as moderate (no large type II error). However, the EBI misclassified 40 % of poor status as moderate and 20 % of bad status as poor. Although these are rather mild misclassifications (small type II error), the levels are quite high. The evaluation shows also a high unbalance between type I and II error for the poor Hscore.

Table 7.8: Estuarine Biotic Integrity Index (EBI) scores compared to predicted habitat quality status (Hscore) expressed as percentage error (M: moderate; P: poor and B: bad status).

	EBI score M	**EBI score P**	**EBI score B**
Hscore M	correct 90%	small type I 6%	large type I 4%
Hscore P	small type II 40%	correct 51%	small type I 9%
Hscore B	large type II 0%	small type II 20%	correct 80%

3.5.2 Seasonal variation

ANOVA found no significant differences in metric values between the different seasons for the assessed sites ($p > 0.05$).

4 Discussion

4.1 Sampling methodology

All samples were obtained using fyke nets, a sampling method seldom used in other European estuaries (Elliott & Dewailly, 1995), although it gives sufficient information along the salinity gradient (Chapter 2). As few studies compare fyke catches with other gears (e.g. Hinz, 1989; Thiel & Potter, 2001), we compared presence/absence data obtained with fyke nets with presence/absence data of fish impinged at cooling-water filter screens of the nuclear power plant of Doel situated in the study area (unpubl. Obs.). The data were collected in the same period between 1995 and 1998. During this period we collected the same species with both survey methods but the species richness per day per fyke net was generally higher than that obtained on the filter screens per survey. Fyke nets are relatively unselective fishing gear, catching demersal and pelagic species (Hamerlynck & Hostens, 1994) and they are also easy to install in a great variety of habitat types. Fairweather (1999) recommended for an indicator method a rapid and effective sampling; the installation must be easy to deploy and left out only for short periods. This is the case for fyke nets.

4.2 Habitat status

The present analysis relies on the objective determination of the environmental degradation and stressors in the estuary. As such, our approach is similar to Aubry & Elliott (2006) for the Humber estuary, England, although we used a more restricted set of parameters to avoid redundancy and selected indicators relevant for our situation. The intertidal area lost and land claim were identified as parameters to assess the quality of the Schelde estuary (Van den Bergh *et al*., 2005). Araújo *et al.* (2000) showed that dissolved oxygen is an important environmental factor determining the occurrence of fish species in estuaries. A depletion of dissolved oxygen usually results from an excessive organic pollution and generates a significant interference with migration routes (NRA, 1993). Dissolved oxygen therefore incorporates information about organic and inorganic components (N, P, etc…) which are not included here to avoid redundancy. We observed that the minimum yearly dissolved oxygen saturation was significantly different between the sites and was related to human impacts such as industrial activities and the presence of marina and ports (Soetaert *et al*., 2006). Dredging activities do have an impact, especially when the sediment is discharged in the same estuary.

According to Newcombe & Jensen (1996) some fish species and life stages in rivers and estuaries show ultra sensitivity to suspended sediment.

4.3 The range of habitat status (Hscore)

Table 7.3 shows that the habitat quality (Hscore) ranges from 3 to 5 (from medium to very high disturbance). In Flanders, there are no pristine or slightly impacted estuarine sites (Hscore 1 or 2), but it is still possible to derive a fish index, as the proposed method selects metrics responsive to a gradient of disturbance. We assumed, as already mentioned, that human impacts dominate over the salinity gradient. In the Zeeschelde the salinity shows seasonal variations but between 1995 and 2004 no particular trend was observed over the years (Maris *et al.*, 2008). However, in the same period water quality improved, which is reflected in an increase of species richness (Chapter 2). Greenwood (2007) states that estuaries are ecoclines forming areas of relatively slow ecological change in which fish communities change progressively unless human impact. The observed differences in fish assemblages (metric values) between the different sites are therefore assumed to be more related to human impact than to a change in salinity. In addition, the ranking of the sites in three different Hscore classes is sufficient to calibrate the index although the EBI is limited from class 3 onwards as no empirical data are available to delimit thresholds for classes 1 and 2. A second and more fundamental point is that the actual calibration does not include metrics which can become important for discriminating for higher quality (e.g. MpiDia and MniErs). In the current situation of moderate to high disturbance, this is not a major problem and the EBI will be sufficient to monitor changes, but in the future the calibration should be extended including sites of higher quality. If they cannot be found in estuaries in Flanders, similar estuaries from other countries should be studied although it is difficult to define and locate such similar estuaries. In addition, catch methods in other countries often differ, so it will be difficult to obtain comparable data.

4.4 The calibration of the index: historical or empirical approach?

The first step of the calibration consists in a selection of metrics responding to human impacts as known from theory or experience. The suite of metrics should reflect the function of the ecosystem through various aspects of the composition and abundance of the ichthyofauna. However, the interpretation in terms of quality (classes) of these metrics is not always straightforward and the characteristics of the estuary influence the value of the metrics. An adjustment of the metric criteria for habitat predictors is necessary to assess anthropogenic effects rather than natural stream characteristic effects (Breine *et al.*, 2004) although in this study this is less relevant as only one single estuary was studied. The European Water Framework Directive indicates that reference conditions can be determined according to one or a combination of four methods: a physical control area (i.e. another, similar estuary), a predictive modelling approach, hindcasting (a historical approach) and, if these are not satisfactory, an expert judgement approach (WFD, 2000). Borja *et al.* (2004), Coates *et al.* (2004) and Jager & Kranenbarg (2004) relied strongly on expert judgment to attribute the quality scores. The first development of this index (Breine *et al.*, 2001; Adriaenssens *et al.*, 2002b) defined a fish community reference based on historical data from the Schelde estuary (de Selys-Longchamps, 1842; Poll, 1945; Poll, 1947), expert knowledge (OVB, 1988; OVB, 1994) as well as on recent data (Maes *et al.*, 1997, 1998a,b, 1999, 2003). The match of this preliminary index with calibration data was low (37.1%). In contrast, the method proposed in this chapter is data driven, especially as the scoring thresholds for the metrics are the quintiles of metric distribution of the best samples. Quintiles turned out to give an optimal balance between the reduction of information (e.g. with terciles more detail is lost by the scoring) and precision (for higher percentiles, too many thresholds would be estimated from the limited data). In fact this approach is closely related to the trisection method where scoring is based on terciles.

4.5 The selected metrics

The stepwise selection method used here ensured that all metrics are complementary to each other, especially as a metric was not entered if it did not contribute to the discriminatory power. Once a parallel or correlated metric was introduced, a similar metric had no important further contribution. As an indirect consequence of this approach there was no further need for the correlation tests advocated by Hughes *et al.* (1998) and Breine *et al.* (2004). The final selection of the metrics composing the EBI corresponds quite well with the metrics of other estuarine indices in Europe (Borja *et al.*, 2004; Coates *et al.*, 2004; Jager & Kranenbarg, 2004) in that they reflect species richness, trophic composition and habitat use and hence cover a broad range of the estuarine functions (Table 7.4).

4.5.1 Percentage of marine juvenile migrating individuals (MpiMjm)

Marine juveniles use the estuary primarily as a nursery but spend much of their adult life at sea (Elliott *et al.*, 1990). According to Lutz (1975) marine juveniles may use the estuary to reduce stress and their presence in an estuary depends on the nursery and water quality conditions. Potter *et al.* (1997) also described the importance of shallow waters as a nursery for these species. Habitat preference by juvenile fishes is influenced by sediment grain size, bed roughness and presence of biogenic structure (Diaz *et al.*, 2003). They constitute the major guild in the mesohaline zone (Maes *et al*., 1998a)

4.5.2 Percentage of smelt individuals (MpiSme)

The percentage of smelt individuals is important as a signal of migration problems and also, because of its high environmental sensitivity, smelt is a good indicator for environmental pressure. Its high oxygen demand serves as an indicator to detect water quality improvement. A possible disadvantage is that the metric relies on a single species. However, because it is still broadly present in the North East Atlantic estuaries, it has the potential to serve as an indicator for a large number of estuaries (Vandelannoote *et al.*, 1998; Araújo *et al.*, 2000; Costa *et al.*, 2000).

4.5.3 Percentage of omnivorous species (MpiOmn)

Assessing feeding guilds provides a useful measure to assess the structure and functioning of estuarine fish communities (Elliott & Dewailly, 1995; Elliott *et al.*, 2002). Although Mikkelson (1993) and Deegan *et al.* (1997) concluded that human impact was not reflected in the trophic composition, in a healthy ecosystem members of different feeding guilds should be present (e.g. Coates *et al.*, 2004). An increase in omnivores is associated with an increase in human pressure. For example, Breine *et al.* (2001) and Borja *et al.* (2004) attributed a low score when the percentage of omnivorous species is very high (>80%) or extremely small (less than 1%) since both situations indicate a disturbance in the food chain. The only other metric showing an increase with disturbance (Fig. 7.4) was the number of exotic species (MnsExo) which only entered after the optimum cut off (Fig. 7.5). It was therefore not retained.

4.5.4 Total number of species, freshwater species excluded (MnsBra)

According to Tong (2001), responses to environmental stressors are reflected in fish health and their community composition and distribution. Human impacts will decrease the species richness, especially the ecosystem specific species (Breine *et al.*, 2004). This is a very simple metric and, although the species richness in an estuary is influenced by natural variability, many authors use species richness in their index (Miller *et al.*, 1988; Ramm, 1990; Cooper *et al.*, 1994; Deegan *et al.*, 1997; Hughes *et al.*, 2002; Borja *et al.*, 2004; Coates *et al.*, 2004). Fresh water species are excluded as the mesohaline zone of the estuary is not its essential habitat (Martino & Able, 2003).

4.5.5 Percentage of piscivorous species (MpiPis)

In defining piscivores, we took into account the fact that some species change their trophic membership ontogenetically. The presence of top carnivores is an indicator for a stable trophic network within an estuary (Coates *et al.*, 2004), so this metric can capture trophic disturbances. The stepwise introduction shows that the contribution of this metric is limited, although it made a contrast between class 4 and 5 (Fig. 7.4).

4.6 Evaluation

4.6.1 Evaluation of the misclassification

According to the WFD it is important that the EBI distinguishes well between disturbed and reference sites. The most important boundary is between good and moderate ecological status; any area with moderate (or worse) ecological status will require remediation to achieve good or high ecological status. Hence the power to detect heavily impacted sites should be strong. In this study, 4.0% of the moderate cases (Hscore 3) were classified as highly disturbed (large type I error) and none of the latter class were classified as moderate (large type II error). Hence, the choice of fixing the type I error at about 10% (one of the classical choices for the level of significance in statistical testing) resulted in a good discrimination. Reducing the risk of making type I error increases the risk of making a type II error. We need to achieve the optimum balance between the two types of errors. The internal validation is only an initial test of the accuracy and pertinence of the index developed and, ideally, an external dataset should be used for validation. An alternative is to divide the dataset into several parts to cross-validate but in this study the data set was too small for this purpose.

4.6.2 Seasonal variation

We did not observe seasonal impact on the metric scores as mentioned by Jager & Kranenbarg (2004). There are two possible reasons: cyclical variations in the community structure do exist within our estuary (Maes *et al.*, 1998a), but they are not reflected at the guild levels. This is in agreement with Castillo-Rivera *et al.* (2002), who observe a greater similarity in fish assemblages between seasons than between habitats, and with Breine *et al.* (2001) who did not observe important seasonal differences in fish assemblages (guild levels). The fact that the calibration is based on data scattered over the whole year represents an alternative explanation. As a consequence, seasonally dependent metrics will not give a consistent contribution to the index and will not be included by the stepwise procedure. From the point of view of fieldwork, this independence of season is an advantage, as with seasonal independence, one can collect data during the whole year. A possible disadvantage is that some sensitive and specific metrics can be missed, if they are only observable during a specific period of the year.

5 Conclusions

The proposed estuarine fish index (EBI) consists of a balanced set of metrics combining several aspects of the estuarine community such as trophic status, species richness, nursery function and presence of intolerant estuarine type species. A first internal validation proved the index to be a robust and adequate tool to distinguish heavily impacted sites from sites of moderate quality, although further validation of the EBI is necessary. A first possibility is to take new independent samples and assess the misclassification rate. Alternatively, one can apply the index in nature restoration projects and determine its reaction to expected improvements. However, as stated earlier, it is important to recognize that the index is actually limited to classes 3-5. In the future, an extension of the index will be necessary, for instance by including data from similar estuaries in Europe. However, matching of the estuaries is difficult and there is a problem to obtain comparable data. It is considered that the present study presents a new stepwise approach to calibrate a fish-based index. This approach can also be applied to other estuaries.

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Chapter 8

A zone-specific fish-based biotic index as a management tool for a temperate estuary (Zeeschelde, Belgium)

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Abstract

Fish-based indices monitor changes in surface waters and are invaluable to summarise complex information on the environment (Harrison & Whitfield, 2004). A Zone-specific fish-based multimetric Estuarine index of Biotic Integrity (Z-EBI) was developed based on a 13 year time series of fish surveys from the Zeeschelde estuary (Belgium). Sites were pre-classified using indicators of anthropogenic impact. Metrics showing a monotone response with pressure classes were selected for further analysis. Thresholds for Good Ecological Potential (GEP) were defined from zone-specific references, a modified trisection was applied for the other thresholds. The Z-EBI is defined by the average of the metric scores calculated over a one year period within each zone and translated into an Ecological Quality Ratio (EQR) to comply with the European Water Framework Directive (WFD). The indices measure fish community characteristics such as species richness and composition, species abundance and nursery function, as well as trophic functions when appropriate. As such they integrate structural and functional qualities of the estuarine fish communities. The Z-EBI performances were successfully validated for habitat degradation in the various habitat zones. Results indicate that the indices distinguish among various levels of degradation (94% matches).

Key words: biotic integrity, fish-based index, meso- and oligohaline estuary, freshwater, European Water Framework Directive, Zeeschelde,

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1 Introduction

The value of riverine fish communities as indicator of biological integrity was documented for the first time in 1913 by Forbes and Richardson. However, the first fish-based multimetric index was presented only in 1981 (Karr, 1981). This index for rivers in the Midwestern USA assesses the current status of a fish community using fish community parameters or metrics. Since that time it has been modified for worldwide use (Hughes & Oberdorff, 1999); it is now applied as a management supporting tool in many countries (Uriarte & Borja, 2009). In Europe, fish-based indices became important bio-assessment tools since the European Union (EU) water policy recommended fish as a biological quality element (WFD, 2000). It states that data on ichthyofaunal species composition and abundance should be used to report the ecological status of European estuaries. So far, various fish-based indices for European estuaries have been presented (e.g. Araújo *et al*., 2000; Salas *et al*., 2004; Breine *et al*., 2007; Martinho *et al.*, 2008a). These indices are useful decision-making tools since they evaluate changes in the state of an aquatic ecosystem as a result of management responses (Pinto *et al*., 2009).

Criteria for a basket of metrics

The metrics constituting a fish-based index, which assesses the ecological status of estuaries, have to comply with a number of criteria. First metrics must match the ecological guild concept of estuarine functional use, which assorts estuarine fish species according to their particular use of estuaries (Elliott *et al*., 2007; Franco *et al.*, 2008). Secondly, metrics should be sensitive to human impact and react unambiguously to impact changes. Thirdly, they should assess a particular ecological function of the different parts of the estuary. Various authors have described estuarine functions and how fish use the estuarine ecosystem (e.g. Courrat *et al*., 2009; Franco *et al*., 2008; Martinho *et al*., 2008b). Fourthly, metrics must be able to evaluate ecological goals. Based on a literature review Breine *et al*. (2008a) described habitat needs to realise ecological goals for fish in estuaries (Chapter 4). The metrics should evaluate whether these goals have been reached. Briefly, the ecological goals ensure that the estuarine fishes concerned reproduce, feed and grow up within a given estuary. This implies unrestricted movement between juvenile and adult feeding and spawning grounds. Finally, metrics should assess the fish communities as required by the WFD i.e. they should include information about species composition and abundance.

The lack of type-specific reference sites

One of the main constraints in the development of fish-based indices is the absence of a type-specific reference (Southerland *et al*., 2007). The reference condition is the pristine, undisturbed surface water having the same or similar abiotic characteristics as the river, lake or estuary to be assessed. Initially maximum species richness lines (MSRLs), obtained for species in reference situations, were used to derive scoring criteria (Karr, 1981; Karr *et al*., 1986). These lines related expected numbers of species within a metric to stream size at various levels of environmental quality. They were drawn by eye or generated from a 95th percentile regression (Rankin & Yoder, 1999). In the absence of a reference condition researchers developed models (Pont *et al*., 2006, 2007), used historical information (Belpaire *et al*., 2000), best available data (least-disturbed) (Harrison & Whitfield, 2006; Schmutz *et al*., 2007; Pont *et al.*, 2009) or expert judgment (Borja *et al*., 2004; Harrison & Whitfield, 2004) to define a reference state. For the meso- and oligohaline zone in the Zeeschelde estuary (Belgium) a fish-based index was developed using the best available sites for reference (Chapter 7, Breine *et al.*, 2007). Metrics were scored based on statistically derived quintiles and selected by a stepwise regression; their discriminating power was assessed towards a predefined habitat quality class. The index distinguishes between the qualifications of bad, poor and moderate sites, as no reference data were available to delimit thresholds for higher classes.

Objectives

We present an index developed in five steps, which assesses the environmental quality of the salinity zones in the Zeeschelde using fish surveys in spring, summer and autumn. The index assesses the whole Zeeschelde estuary from the tidal mesohaline zone up to the freshwater tidal area. Metrics are scored based on references developed in chapter 3. The index distinguishes between integrity classes from bad to high.

2 Material and methods

2.1 The estuary

Some 50% of the 355 km long River Schelde covers a tidal area (semidiurnal tide) of 160 km between Gent (Belgium) and the North Sea near Vlissingen (The Netherlands) (Fig. 8.1). The estuary has been typed in the salinity zones according to the Venice system (1959): an euryhaline zone (salinity of >30) between Vlissingen and Hansweert and a polyhaline tidal zone (18-30) between Hansweert and Zandvliet are situated in the Westerschelde (The Netherlands). This study focuses on the Zeeschelde (Belgium) which consists of three salinity zones: the mesohaline zone (5-18) between Zandvliet (Dutch – Belgian border) and Antwerpen, the oligohaline zone (0.5-5) between Antwerpen and Rupelmonde, including the Rupel River and the freshwater zone between Rupelmonde and Gent, including the River Durme (limnetic zone: <0.5). The freshwater discharge determines largely the salinity gradient and shows a seasonal pattern. The tidal tributaries Zenne, Nete and Dijle are an integral part of the estuary. The Zeeschelde (105 km) is characterised by a single ebb/flood channel, bordered by relatively small and narrow mudflats and marshes (28% of the total surface). Some 10 million people (477 inhabitants km^{-2}) live in the river basin (Baeyens *et al*., 1998). A wide range of human activities are concentrated in the Zeeschelde catchment, where historically urbanisation and industries developed close to the riverbanks. The large and long standing human impact has resulted in a profound environmental degradation of the estuary and is reflected in the water quality (Soetaert *et al.*, 2006; Maris *et al*., 2008), habitat loss (Van Braeckel *et al*., 2006; Elliott *et al.*, 2008a) as well as flow regime (Meire *et al*., 2005; Van Braeckel *et al*., 2006).

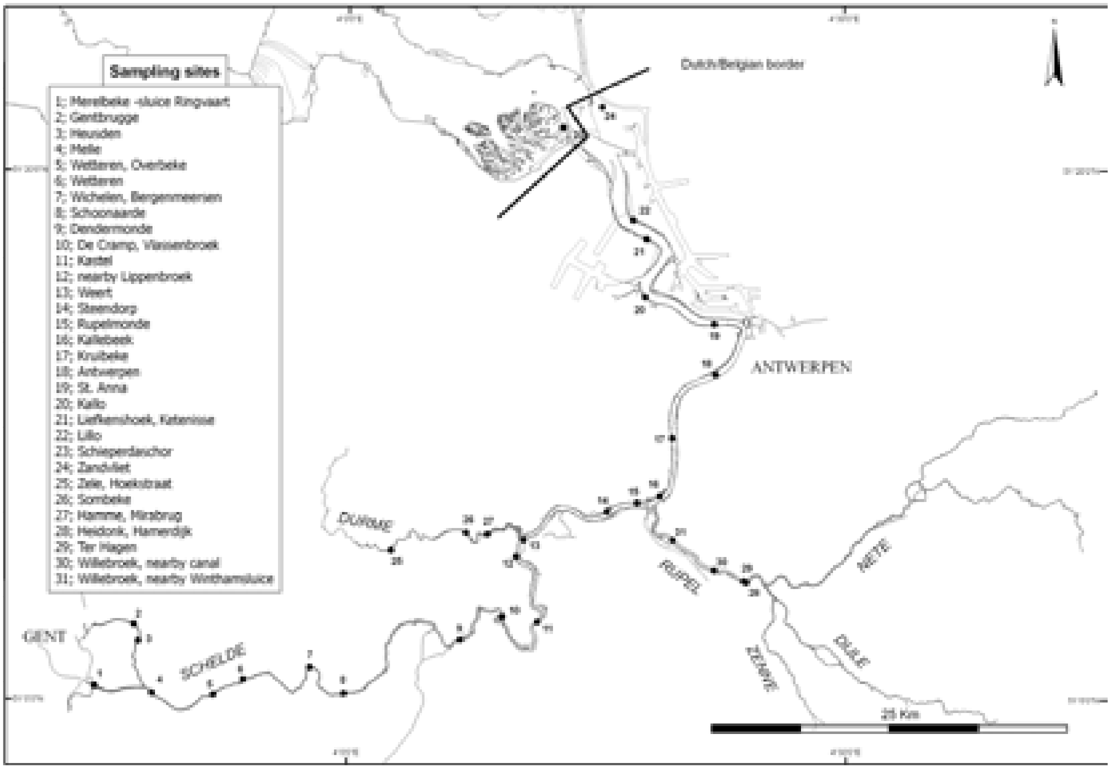

Figure 8.1: Map of the Zeeschelde basin with the sites surveyed during the period 1995-2008.

2.2 Index development

The Estuarine Index of Biotic Integrity (Z-EBI) was developed stepwise. Firstly, fish were sampled using a standardised methodology. Secondly, a habitat quality assessment was performed for each sample site. Thirdly, fish species were selected using the reference list for the various salinity zones in the Zeeschelde under the assumption of a good ecological potential (GEP), this is necessary because the Zeeschelde has been assigned the status of heavily modified water body (Chapter 3). Fourthly, each of the selected species were attributed to candidate metrics (specific to the salinity zone) based on literature review and expert judgment. In the last step, these metrics were screened for a monotone response to the anthropogenic pressure classes defined in the second step. Statistical analysis assures the final selection of the metrics. The GEP threshold for selected metrics is defined using the reference list. The other integrity classes are defined by applying trisection with the 90% values of these upper scores as in Breine *et al*. (2004).

2.2.1 Fish sampling

Fish surveys were organised at 31 different sites in the estuary over a period of 13 years (1995-2008) (Fig. 8.1). Fish data were collected by trained fish biologists using a standardised protocol (Maes *et al*., 2003). At each site, one or two double fyke nets were positioned at low tide and emptied daily; they were placed for one or two successive days. All fish caught were identified to species level on site. Occasional cross examination in the laboratory assured the quality of the fish identification. Each survey per site was standardized as number of fish per fyke per day, resulting in 1184 CPUE data retained for analysis. CPUE data were grouped per salinity zone (mesohaline, oligohaline and freshwater) and pooled per year and season. Cumulative relative percentage of recorded species with increasing sample effort in the salinity zones was calculated per year and averaged.

2.2.2 Assessment of the anthropogenic pressure in the three salinity zones

The anthropogenic impact of each site is determined using the site-specific general habitat quality class (PSite in Table 8.1). This single multimetric indicator was attributed by assessing a selection of the most relevant parameters for the Zeeschelde from the environmental integrated indicator assessment method developed by Aubry and Elliott (2006) (Harbasins project, Elliott *et al*., 2008a,b,c) (Table 8.1). A first parameter is the average minimum dissolved oxygen (DO) concentration (%) calculated for each year (1995-2008) (T. Maris, pers. comm., OMES). Benthos score is calculated annually for all zones (Brys *et al*., 2005; Speybroeck *et al*., 2008). Land claim (%) was determined with respect to the intertidal surface in 1850 (Adriaensen *et al*., 2005; Van Braeckel *et al*., 2006). Land claim is defined as the loss in area covered by intertidal habitat: tidal marshes and mudflats since 1850. The presence of marinas was assessed with aerial photographs. The degree of industrial activity (low, moderate or high) was assessed on site or judged by experts. Channel dredging activities were provided by the MOW - Department of Mobility and Public Affairs, division Maritime Access (pers. comm.). Each individual indicator was scored between 1 and 5 by thresholds based on expert judgment. The sum of these individual scores was ranked from 1 (low impact i.e. high status) to 5 (very strong impact i.e. bad status) (Table 8.1). The main goal was to obtain a ranking of human impact rather than an expression of quality. The criteria and threshold values for the habitat pre-classification (PSite) of the sites are given in Table 8.1.

Table 8.1: Classification indicators and threshold values to score and derive the habitat class pre-classification (PSite). Scores range from 1 (high status) to 5 (bad status).

	Score				
Parameter	**1**	**2**	**3**	**4**	**5**
Minimum dissolved oxygen average (%)	>80	80 & >70	70 & >50	50 & >30	≤30
Benthos (mesohaline and oligohaline zone)	Classification explained in Brys *et al*., 2005 and Speybroeck *et al*., 2008				
Land claim (%)	0	<20	≥20 & <30	≥30 & <50	≥50
Ports and marinas (absence/presence)	No				Yes
Industrial activities (degree)	Low		Moderate		High
Dredging (absence/presence)	No				Yes
PSite: total score	6	7 - 13	14 - 20	21 - 27	28 - 30
Appreciation (quality)	**High**	**Good**	**Moderate**	**Poor**	**Bad**

2.2.3 Metric selection

2.2.3.1 Candidate metrics

For each salinity zone within the Zeeschelde estuary, a list of candidate biological metrics was drawn based on ecological relevance (Table 8.2). For instance, estuarine species are considered appropriate for the mesohaline zone but not for the freshwater zone, while diadromous species inhabit all zones. Each metric should describe an estuarine function appropriate for the e.g. nursery function. In addition the European water framework criteria (Directive 2000/60/EC) should be taken into account, it states that for estuaries the species composition and abundance should be assessed. Based on their ecological demands species occurring in the GEP reference list (Chapter 3, Breine *et al*., 2008) were attributed to metrics because the GEP status is considered to be a feasible goal in the Schelde estuary. The ecological demands (reproduction, feeding behaviour, tolerance to low oxygen concentration or any appropriate special characteristic) of the selected species were defined based on the literature and personal expertise. We considered the different guilds and associated metrics as described by Franco *et al*. (2008) and added metrics relevant to the freshwater zone. Table 8.2 provides an overview of the 19 selected candidate metrics and their relevance (1 or 0) for each salinity zone. All metrics appear in two types: number of species and the relative frequency of individuals (proportion by number), except for 'total number of species' and 'Shannon-Wiener index'.

Table 8.2: Candidate metrics and their relevance (1 or 0) for each salinity zone.

Metric	**Abbreviation**	**Meso**	**Oligo**	**Fresh**
estuarine species [1,3,4,5,9,11,13,14,15,17,20, ,22,23,24,25,26]	Ers	1	1	0
diadromous species [3,5,9,11,13,14,18,19,22,23,24,25,26]	Dia	1	1	1
freshwater species [1,3,5,9,18,19,22,25,26]	Fws	0	1	1
marine migrants [1,5,9,11,13,14,18,,22,23,24,25,26]	Mms	1	0	0
intolerant species [4,20,21,22,24]	Int	1	1	1
tolerant species [4,11,13,14]	Tol	1	1	1
species sensitive to fragmentation [21]	Fra	1	1	1
species that need shelter or that are habitat sensitive [19]	Hab	1	1	1
benthic species (stratum) [3,4,11,13,14,17,19,24]	Ben	1	1	1
specialised spawners [3,19,21,25,26]	Spa	1	1	1
piscivores [1,3,7,11,19,20,21,22,24,25,26]	Pis	1	1	1
invertivores [1,3,13,19,21,22,25,26]	Inv	0	1	1
omnivores [3,7,11,13,14,19,20,21,24,25,26]	Omn	0	0	1
rheophilic (A) species [19,21]	Rha	0	0	1
rheophilic (B) species [19,21]	Rhb	0	0	1
eurytopic species [19]	Eur	0	0	1
Shannon-Wiener index [12,21,24]	Sha	1	1	1
total number of individuals[4,7,]	Ind	1	1	1
total number of species [1,2,4,6,7,8,10,11,13,14,15,16,17,18,20,21,22,24]	Tot	1	1	1

([1]Miller *et al*., 1988; [2]Cooper *et al*., 1994; [3]Elliott & Dewailly, 1995; [4]Deegan *et al*., 1997; [5]Costa & Cabral, 1999; [6]Araújo *et al*., 1999; [7]Galatowitsch *et al*., 1999; [8]Williams & Zedler, 1999; [9]Araújo *et al*., 2000; [10]Peterson *et al*., 2000; [11]Breine *et al*., 2001; [12]Gelwick *et al*., 2001; [13/14]Adriaenssens *et al*., 2002a, 2002b; [15]Bate *et al*., 2002; [16]Castillo-Rivera *et al*., 2002; [17]Hughes *et al*., 2002; [18]Thiel *et al*., 2003; [19]van Emmerik, 2003; [20]Borja *et al*., 2004; [21]Breine *et al*., 2004; [22]Coates *et al*., 2004; [23]Jager & Kranenbarg, 2004; [24]Breine *et al*., 2007; [25]Elliott *et al*., 2007; [26]Franco *et al*., 2008)

2.2.3.2 Evaluation and selection of metrics

Next, values for the candidate metrics were calculated per site where fish were caught using presence/absence data and number of individuals. Species not belonging to any reference list are omitted. Per site and zone results were pooled monthly and annually, and transformed to catch per unit effort (CPUE). In order to account for variation in sampling effort, metrics based on abundance data were measured in terms of relative abundance (see also Harrison & Whitfield, 2004).

Non-parametric analyses with metric values scouted zone effects (mesohaline, oligohaline and fresh), year effects and seasonal effects (four seasons). ANOVA was used to test for differences between metric values in the habitat (PSite) classes. If the assumptions were not met, a non-parametric Kruskal-Wallis test was applied. In order to assess the metric reaction a graphical analysis was performed by plotting box plots of the variation in candidate metric value in function of the pressure scores (PSite). This was done over the salinity zones as well as within the zones over the seasons. This visual analysis allowed the first selection of appropriate metrics for each zone, i.e. those metrics that separate different pressure classes. Only metrics with a monotone reaction where the box plots clearly differ among the PSite classes are retained. From these relevant metrics we indicated those with a high contribution to the variation (F1 or F2 $>\pm 0.55$) using a Principal Component Analysis (PCA) with the log transformed metric values (log (x+1)) (Hughes *et al*., 1998). The variation is caused by the pressures and since we assume a linear relation between the pressure and metrics PCA is used as an exploratory analysis to show the coherent metrics. From these coherent metrics we tested for redundancy by a Pearson correlation (McCormick *et al*., 2001: $r > 0.75$, $p < 0.05$). Finally we retained one metric from each pair of redundant metrics using the results of the correlation between the metric values and the PSite (Spearman rank correlation (McCormick *et al*., 2001: absolute value of r: $|r| > 0.15$, $p < 0.01$)). When needed ecological knowledge was used to take a final decision.

2.2.4 Scoring of metrics

Based on the reference list (Chapter 3) a maximum number of species for each metric in each zone is calculated representing the values expected in the Good Ecological Potential (GEP) status. In addition we calculated the maximum relative species frequency for each metric and used this ratio to define the relative individual percentage thresholds. To correct for catch failure we take the 90 percentage of the calculated GEP value as the maximum threshold value. E.g. the mesohaline reference includes 30 species in the reference; the 90 percentage of this maximum equals 27. This value is used as threshold ratio between the GEP and moderate status for the total number of species in the mesohaline zone. For the total number of individuals caught we took the average from the lowest impacted classes within each zone as the threshold values. The upper score is divided in four equal sections giving the threshold for the four status classes: GEP, moderate, poor and bad (Breine *et al*., 2004). The sum of the metric scores gives the Ecological Quality Ratio (EQR) ranging from 0 to 1. Within the EQR four integrity classes with equal intervals are defined.

2.2.5 Validation

An internal validation with data pooled per year from 1995 till 2007 (N=30) and compiled per zone consisted of a comparison between the index scores with the PSite scores (average of site values per zone per year). Allowing a class difference of one unit we assessed the over- or underscoring of the index. An external validation made use of independent fish data from 2007 and 2008 for the freshwater (N=96), the oligohaline (N=210) and the mesohaline (N=100) zones for a comparison of index scores with PSite scores.

3 Results

3.1 Fish sampling

For each salinity zone the reference species list, species collected during the different surveys and their metric attribution are represented in table E (see annex). Species not belonging to any reference list have been omitted (Chapter 3). In total 60 species were caught in the Zeeschelde estuary of which 40 qualify as reference species (Table E, see annex). In Table 8.3 the catch frequencies are given for all species caught during the different surveys, excluding winter catches. Table 8.4 compiles the distribution over time of the fishing outings used in the analysis: their distribution is uneven between the zones.

Table 8.3: Catch frequency expressed as relative percentage of catch of the species involved in the three zones (number of surveys are given between brackets) during the period 1995-2008. The figures in bold indicate that the species qualifies as a reference species for that zone.

Species	**Mesohaline (203)**	**Oligohaline (606)**	**Freshwater (227)**
Abramis brama	8.8	**19.7**	**30.0**
Alburnus alburnus	0.9	0.5	1.4
Alosa fallax	**18.6**	**1.7**	**0.3**
Ammodytes tobianus	0.0	0.2	0.0
Anguilla anguilla	**58.0**	**69.0**	**59.0**
Atherina presbyter	16.8	2.4	0.0
Blicca bjoerkna	6.2	**11.1**	**61.8**
Carassius carassius	0.0	0.8	0.7
Carassius gibelio	15.5	32.9	42.7
Chelidonichthys lucernus	**9.3**	0.2	0.0
Ciliata mustela	**14.6**	0.2	0.0
Clupea harengus	**66.8**	**42.0**	2.4
Cottus gobio	0.0	0.0	0.7
Cyclopterus lumpus	0.4	0.0	0.0
Cyprinus carpio	2.2	8.9	30.4
Dicentrarchus labrax	**75.2**	**30.8**	2.0
Echiichthys vipera	0.0	0.5	0.0
Engraulis encrasicolus	0.4	0.0	0.0
Esox lucius	3.5	**0.8**	**1.0**
Gadus morhua	16.4	0.5	0.0
Gasterosteus aculeatus	**17.7**	**23.3**	**41.3**
Gobio gobio	0.4	0.9	0.0
Gymnocephalus cernuus	**13.7**	**2.0**	**26.6**
Lampetra fluviatilis	**0.4**	**1.1**	**4.8**
Lepomis gibossus	0.9	2.7	10.2
Leucaspius delineatus	0.0	0.5	1.4
Leuciscus cephalus	0.0	0.0	0.3
Leuciscus idus	1.3	**3.8**	**4.4**
Limanda limanda	2.7	0.3	0.0
Liparis liparis	**0.4**	0.0	0.0
Liza ramado	**34.1**	**4.5**	**8.9**
Merlangius merlangus	**9.7**	0.6	0.0
Misgurnus fossilis	0.0	**0.0**	**0.0**
Mullus surmuletus	1.3	0.0	0.0
Myoxocephalus scorpius	**4.0**	0.0	0.0
Osmerus eperlanus	**43.4**	**4.5**	**4.8**
Perca fluviatilis	**29.6**	**35.3**	**56.0**
Petromyzon marinus	**0.0**	**0.0**	**0.0**
Platichthys flesus	**98.2**	**64.5**	**56.3**
Pleuronectes platessa	**16.8**	0.0	0.0
Pomatoschistus lozanoi	0.4	0.2	0.0
Pomatoschistus microps	**1.8**	**32.8**	15.0
Pomatoschistus minutus	**27.0**	**25.7**	4.4
Pomatoschistus sp.	**9.3**	**0.0**	0.0

Table 8.3: Continued.

Species	Mesohaline (203)	Oligohaline (606)	Freshwater (227)
Pseudorasbora parva	0.9	8.4	33.1
Pungitius pungitius	2.2	**2.9**	**5.5**
Rhodeus sericeus	0.9	**8**	**15.4**
Rutilus rutilus	**50.4**	**86**	**89.8**
Salmo salar	0.9	0	0
Salmo trutta	**3.5**	**0.2**	**0**
Sander lucioperca	**71.2**	**47.4**	**54.9**
Scardinius erythrophthalmus	13.3	**7.8**	**52.9**
Scophthalmus rhombus	3.1	0	0
Silurus glanis	0	**0.8**	**1**
Solea solea	**81.4**	13.7	0
Sprattus sprattus	**1.3**	**0.2**	0
Syngnathus acus	**7.5**	**1.4**	0
Syngnathus rostellatus	**0.4**	**0.2**	0
Tinca tinca	0	1.1	2.7
Trachurus trachurus	0.9	0.5	0
Trisopterus luscus	**18.6**	**0.8**	0
Zoarces viviparus	**2.2**	**0.2**	0
Total number of species	52	49	33
Number of reference species caught	29	30	21

During this period we caught 52 species in the mesohaline zone. All species from the reference list were caught at least once, except *Petromyzon marinus*. In the oligohaline zone 49 species were caught and except for *Misgurnus fossilis*, *Myoxocephalus scorpius* and *Petromyzon marinus*, all reference species have been collected at least once. In total 33 species were caught in the freshwater zone of which three species belonging to the reference were never recorded: *Misgurnus fossilis*, *Petromyzon marinus* and *Salmo trutta*. Table 8.5 presents the average cumulative relative percentage of species caught with increasing sample effort for the three zones.

Table 8.4: Number of fishing occasions per year in the three salinity zones of the Zeeschelde excluding winter data (N= 1036).

Zone	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
Mesohaline	10	0	3	8	5	0	21	5	49	1	1	2	98	0
Oligohaline	3	0	7	15	0	0	6	2	2	8	255	5	280	23
Freshwater	0	0	12	8	0	0	3	30	3	8	7	11	121	24
Total	13	0	22	31	5	0	30	37	54	17	263	18	499	47

Table 8.5 shows that on average 91% of the total species, yearly recorded annually in the freshwater zone, are caught after the second survey. In the other salinity zones more surveys are needed to obtain that result.

Table 8.5: Cumulative relative percentage of recorded species with increasing sample effort (S)

Zone \ Sample effort	S1	S2	S3	S4	S5	>S5
Mesohaline	57.8	77.2	83.7	86.5	90.8	100.0
Oligohaline	59.6	78.5	86.1	90.2	96.9	100.0
Freshwater	59.1	91.0	95.6	97.3	98.1	100.0

In all zones the total number of individuals caught is significantly correlated with the number of surveys ($p<0.001$). Averaging the CPUE data (see 2.2.4 scoring metrics) reduced this effect. For the presence/absence data the imbalance in survey data is less important since no correlation was found (FW, $p=0.069$; O, $p=0.109$ and M, $p=0.221$).

3.2 Pressure assessment of the salinity zones

The results for the pre-classification (Fig. 8.2) show that PSite scores are optimal in the mesohaline zone where impacts are moderate (63.6% of the sites have a moderate status) or strong (36.4% poor status). In the oligohaline zone PSite scores are worst with strong and very strong impacts (respectively 79.1% poor and 20.9% bad status). The habitat quality in the freshwater zone is intermediary with a range of moderate to very strong impacts (25.1% moderate, 64.3% poor and 10.5% bad status).

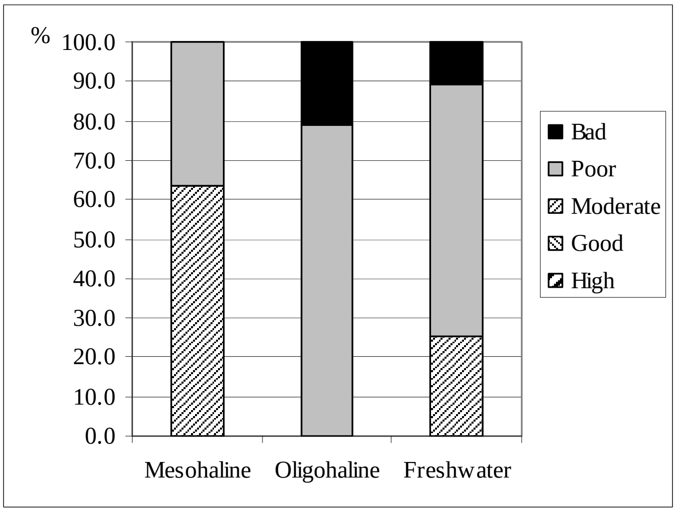

Figure 8.2: Relative occurrence (%) of habitat classes (scaled from bad to high status) in the three salinity zones assessed with PSite between 1995 and 2008 (n= 1184).

3.3 Data selection

ANOVA showed that the metric values differ significantly between the three salinity zones. Therefore we developed indices for each zone. Within each zone metric values vary significantly between years but not between seasons (ANOVA, $p > 0.1$). However, in general metric values are in winter lower compared to the other seasons. This difference is significant for ‘total number of species’ ($p < 0.01$), therefore winter data (n=99) were omitted. Only 1036 out of 1184 fish surveys are retained as only species mentioned in the reference list were used.

3.4 Selection of responsive metrics

The response to human disturbance (i.e. pre-classification) of the annual metric values was assessed in each zone (visual analysis, see also Chapter 7) and results are given in table 8.6. Only those metrics with a monotone and logical reaction to the stressors are retained for the next selection. This is based on the assumption that the metric value will change (decrease or increase) with increasing disturbance.

Table 8.6: Reaction of the values on the assessment results (↑ value increases with increasing pressure; ↓ value decreases with increasing pressure; - no reaction with pressure) in the salinity zones. Data of selected metrics are in bold. Shaded cells indicate metrics that are not selected. The maximum number of species (Mns) based on the reference list, respectively the calculated maximum proportion (%) by number of individuals (Mpi) is given in brackets. An * indicates if metric values react when data are grouped per year.

Metric	Abbreviation	Mesohaline	Oligohaline	Freshwater
Total number of species	MnsTot	**↓(30)***	-(31)*	**↓ (23)***
Total number of freshwater species	MnsFws	↓(5)	-(13)*	↓(15)*
Total number of estuarine species	MnsErs	**↓(7)***	**↓(6)**	
Total number of diadromous species	MnsDia	**↓(9)**	**↓(9)***	**↓(9)***
Total number of marine migrating species	MnsMms	**↓(10)***	**↓(4)**	
Total number of rheophilic (A) species	MnsRha			-(4)
Total number of rheophilic (B) species	MnsRhb			-(3)*
Total number of eurytopic species	MnsEur			↓(12)
Total number of specialised spawners	MnsSpa	**↓(6)***	**-(3)**	↓(8)*
Total number of piscivorous species	MnsPis	**-(14)***	**↓(15)***	-(11)*
Total number of invertivorous species	MnsInv		**-(9)**	-(6)
Total number of omnivorous species	MnsOmn			-(5)
Total number of benthic species	MnsBen	**↓(14)***	**-(12)**	↓(7)*
Total number of habitat fragmentation sensitive species	MnsFra	**↓(9)**	**-(14)***	↓(14)*
Total number of habitat sensitive species	MnsHab	**↓(16)**	**-(14)**	↓(11)*
Total number of pollution intolerant species	MnsInt	**↓(10)**	**↓(10)***	↓(8)*
Total number of pollution tolerant species	MnsTol	**-(13)**	**↑(16)**	-(15)
Shannon-Weiner	ManSha	↓	↓	↓
Total number of individuals	MnsInd	**↓***	↓	↓*
Percentage of freshwater individuals	MpiFws	**↑(16.6)**	**-(41.9)**	-(65.2)
Percentage of estuarine individuals	MpiErs	**↓(23.3)***	**↓(19.3)**	
Percentage diadromous individuals	MpiDia	**↓(30.0)***	**-(29.0)**	↓(39.1)
Percentage of marine migrating individuals	MpiMms	**↓(33.3)**	**↓(12.9)***	
Percentage of rheophilic (A) individuals	MpiRha			↓(17.4)
Percentage of rheophilic (B) individuals	MpiRhb			-(13.0)
Percentage of eurytopic individuals	MpiEur			-(52.2)
Percentage of specialised spawner individuals	MpiSpa	**↓(20.0)**	**↓(9.7)**	↓(34.8)
Percentage of piscivorous individuals	MpiPis	**↑(46.6)**	**↓(48.4)**	↓(47.8)*
Percentage of invertivorous individuals	MpiInv		**↓(29.0)**	↓(26.1)
Percentage of omnivorous individuals	MpiOmn			-(21.7)*
Percentage of benthic individuals	MpiBen	**↓(46.6)**	**-(38.7)**	↓(30.4)*
Percentage of habitat fragmentation sensitive individuals	MpiFra	**↓(30.0)**	**-(45.1)**	↓(60.8)
Percentage of habitat sensitive individuals	MpiHab	**↓(53.3)**	**↓(45.1)**	↓(47.8)
Percentage of pollution intolerant individuals	MpiInt	**↓(33.3)**	**↓(32.2)**	-(34.8)*
Percentage of pollution tolerant individuals	MpiTol	**↑(43.3)**	**↑(51.6)**	-(65.2)

A PCA with selected metrics (in bold in Table 8.6) made the next selection possible based on the factor loadings. These factor loadings are given in Table 8.7.

Table 8.7: Factor loadings (F1 and F2) and accountable variance of the first two axes from a PCA with log transformed values of the retained candidate metric (log (x+1)) values, loadings in bold are significantly contributing (abbreviation see Table 8.6). Correlation between metric and PSite is given by r the Spearman rank correlation coefficient (bold when significant at p <0.01). * Final selected metrics.

Freshwater metrics	**F1**	**F2**	**r**	**Oligohaline metrics**	**F1**	**F2**	**r**	**Mesohaline metrics**	**F1**	**F2**	**r**
*MnsTot	**-0.855**	0.067	**-0.37**	*MnsDia	0.471	0.433	**-0.17**	*MnsDia	-0.383	-0.472	0.10
MnsDia	**-0.943**	-0.134	**-0.48**	*MnsPis	**0.725**	0.135	**-0.09**	*MnsSpa	**-0.813**	0.488	0.04
MnsSpa	-0.452	**-0.815**	**-0.35**	*MnsInt	**0.619**	0.587	0.01	MnsBen	**-0.867**	-0.105	-0.03
MnsBen	**-0.868**	0.152	**-0.36**	ManSha	0.460	-0.019	0.01	*MnsHab	**-0.847**	-0.196	**-0.19**
MnsFra	**-0.941**	-0.220	**-0.42**	*MnsInd	**0.576**	-0.098	**-0.19**	MnsInd	-0.305	-0.401	0.02
*MnsInd	**-0.725**	0.229	**-0.32**	MpiSpa	0.514	**-0.758**	**-0.21**	MpiBen	0.038	0.255	-0.08
*MpiDia	**-0.797**	-0.013	**-0.49**	MpiInt	0.534	**0.595**	0.01	*MpiInt	0.195	-0.312	**-0.56**
*MpiSpa	-0.199	**-0.934**	-0.14	*MnsMms	**0.519**	-0.204	0.07	*MnsMms	-0.461	**-0.635**	**-0.22**
*MpiPis	**-0.665**	0.440	**-0.27**	*MnsErs	0.604	**-0.699**	**-0.20**	MpiErs	**-0.666**	0.620	-0.06
*MpiBen	**-0.763**	0.329	**-0.42**					MpiMms	-0.028	-0.534	**-0.62**
								MnsErs	**-0.810**	0.483	-0.03
								*MnsTot	**-0.772**	-0.498	0.08
% variance accounted for	56.9	19.9			31.7	22.4			36.0	19.0	

All freshwater metrics make a high contribution to the variance and are negatively correlated with the PSite (|r| >0.15). No significant factor loadings for the freshwater metrics were found on the other axes. Redundance (highly significant correlations (r> 0.75; p <0.05)) is observed between MnsDia and four metrics: MnsTot (r=0.752), MnsFra (r=0.922), MnsBen (r=0.727) and MpiBen (r=0.873). MnsBen and MnsFra are also redundant (r=0.761). MnsSpa is negatively correlated with several other metrics. For the oligohaline zone MnsDia, MnsSha and MpiInt do not contribute significantly to the variance. However, MnsDia is significantly negatively correlated with the human impact (PSite) and is retained because it provides important information about the connectivity of the estuary. MpiSpa is highly redundant with MnsErs (r=0.794) and although both metrics react well to pressure (|r| >0.15), MnsErs is selected because it is more relevant for this zone in the estuary. Although MnsMms performs less well (small contribution to the variance) it is kept since it is not redundant with the other metrics (r <0.36) and it provides information about marine species which are abundant in the estuary. A significant contribution to the third axis is recorded (-0.656) for the oligohaline metric MnsSha but this metric is omitted since its value is difficult to interpret. In the mesohaline group of metrics MnsDia is retained because it includes important ecological information. MnsBen and MnsHab both contribute to the variance but are redundant (r=0.795), MnsHab is better correlated with PSite (|r| >0.15). MpiInt does not make a great contribution to the variance but it reacts well with PSite; its contribution to the third axis is significant (0.761). This metric comprises fish species included in rejected metrics e.g. MnsErs and MpiErs (Table E, annex). The estuarine species metrics are relevant for the mesohaline zone but are rejected because of high redundancy with MnsSpa (r=0.95) and MnsBen (r=0.75) and an insignificant reaction with PSite ($|r|<0.15$). MpiMms does not contribute to the variance and is not significantly correlated with other metrics (not concordant). The selected metrics for the three zones are given in tables 8.8, 8.9 and 8.10. Table F (see annex) presents the selected metrics, the functions they assess and the main anthropogenic pressures.

3.5 Scoring of metrics

The 90 percentage scores and the threshold values for the good (here GEP), moderate, poor and bad status of the selected metrics for the three salinity zones are given in tables 8.8 to 8.10. For each zone the threshold values for MnsTot are given whereby an overall EQR of 0.1 (bad) is obtained independently from the other metric scores.

Table 8.8: Selected metrics, metric scores and corresponding threshold values for the freshwater zone for year data with fyke nets; the score is given in brackets (transformed to catch per day per fyke) (abbreviation see Table 8.6, Mns represents number of species and Mpi relative percentage of individuals).

Freshwater					
		Metric scores			
Metric	**90%**	**0.8**	**0.6**	**0.4**	**0.2**
MnsTot	20.7	≥15.5	<15.5≥10.4	<10.4≥5.2	<5.2
MnsInd	174	≥130	<130≥87	<87≥43	<43
MpiDia	35.2	≥26.4	<26.4≥17.6	<17.6≥8.8	<8.8
MpiSpa	31.3	≥23.5	<23.5≥15.7	<15.7≥7.8	<7.8
MpiPis	43.0	≥32.3	<32.3≥21.5	<21.5≥10.8	<10.8
MpiBen	27.4	≥20.5	<20.5≥13.7	<13.7≥6.9	<6.9

MnsTot<5.2 then EQR=0.1

Table 8.9: Selected metrics, metric scores and corresponding threshold values for the oligohaline zone for year data with fyke nets, score is given between brackets (transformed to catch per day per fyke) (abbreviation see Table 8.6, Mns represents number of species).

Oligohaline					
		Metric scores			
Metric	**90%**	**0.8**	**0.6**	**0.4**	**0.2**
MnsPis	13.5	≥10.1	<10.1≥6.8	<6.8≥3.4	<3.4
MnsInt	9.0	≥6.8	<6.8≥4.5	<4.5≥2.3	<2.3
MnsDia	8.1	≥6.1	<6.1≥4.1	<4.1≥2	<2
MnsInd	180	≥135	<135≥90	<90≥45	<45
MnsMms	3.6	≥2.7	<2.7≥1.8	<1.8≥0.9	<0.9
MnsErs	5.4	≥4.1	<4.1≥2.7	<2.7≥1.4	<1.4

MnsTot<7 then EQR=0.1

Table 8.10: Selected metrics, metric scores and corresponding threshold values for the mesohaline zone for year data with fyke nets (transformed to catch per day per fyke) (abbreviation see Table 8.6, Mns represents number of species and Mpi relative percentage of individuals).

Mesohaline					
		Metric scores			
Metric	**90%**	**0.8**	**0.6**	**0.4**	**0.2**
MnsTot	27.0	≥20.3	<20.3≥13.5	<13.5≥6.8	<6.8
MnsDia	8.1	≥6.1	<6.1≥4.1	<4.1≥2.0	<2.0
MnsSpa	5.4	≥4.1	<4.1≥2.7	<2.7≥1.4	<1.4
MnsHab	14.4	≥10.8	<10.8≥7.2	<7.2≥3.6	<3.6
MpiInt	30.0	≥22.5	<22.5≥15.0	<15.0≥7.5	<7.5
MnsMms	9.0	≥6.8	<6.8≥4.5	<4.5≥2.3	<2.3

MnsTot<6.8 then EQR=0.1

3.6 The Ecological Quality Ratio (EQR)

For a given salinity zone the sum of the individual metric scores is the index value (Z-EBI) which varies between 0 and 4.8 (6 metrics). The Ecological Quality Ratio (EQR) is calculated from the Z-EBI for which a general appreciation is given (Table 8.11). This does not mean that the index only measures one type of disturbance. It comprises all responses of the various metrics which can be illustrated by radar type diagrams (Breine *et al*., 2001; Cormier, 2003; Vasconcelos *et al*., 2007). GEP is reached at an EQR of 0.75 and MEP (maximum ecological potential) at an EQR equal to 1.

The EQR is set to 0.1 when the number of species caught in a zone over a year is less than the lower threshold.

Table 8.11: The estuarine index threshold values (Z-EBI), the Ecological Quality Ratio (EQR) and associated appreciation (integrity class).

Z-EBI	EQR	Appreciation
≥4.8	1-0.75	MEP-GEP
<4.8≥3.6	<0.75-0.50	moderate
<3.6≥2.4	<0.5-0.25	poor
<2.4	<0.25	bad

Figure 8.3 illustrates for the year 2006 the Z-EBI score (EQR) and its contributing metrics in the three salinity zones.

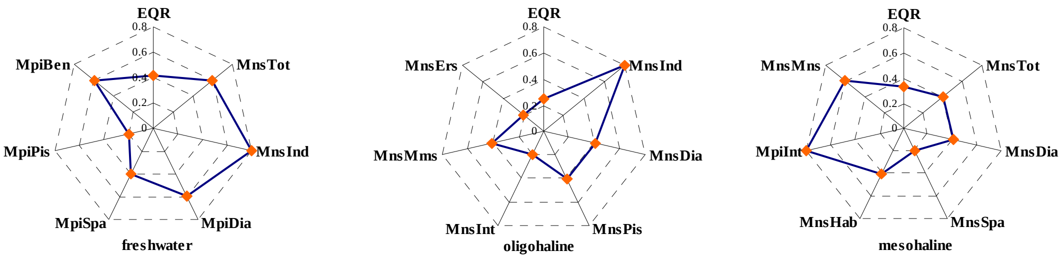

Figure 8.3: Radar plot presenting the EQR and metric scores for the three salinity zones in the Zeeschelde in 2006. For abbreviations see Table 8.6.

In all zones an equal status is recorded, but the relative contribution of the metrics differs. Diadromous score a value of 0.6 in the freshwater zone, while they reach only 0.4 in the oligohaline and mesohaline zone. This can be attributed to the large number of eel caught in the freshwater zone. In the freshwater zone the metric assessing piscivorous fish scores worse than in the oligohaline zone. The intolerant species metric performs better in the mesohaline than in the oligohaline zone.

3.7 Validation

When allowing for a one class divergence (small type I or II error), 94% of the year data classified by the EQR matched with the pressure assessment (PSite) of the sites (Table 8.12) of which 57% have a perfect match (i.e. no class difference). 100% of the EQR classification in the freshwater zone, 82% in the oligohaline and a 100% in the mesohaline zone match with the PSite classification. In the oligohaline two years were too highly scored (18.2% type II error).

Table 8.12: Pre-classification (PSite) and Ecological Quality Ratio (EQR) appreciation by year in the three salinity zones (N=30), = no error; (+) small type I error; (-): small type II error; MOD: moderate

Zone / year	Freshwater			Oligohaline			Mesohaline		
	PSite	**~**	**EQR**	**PSite**	**~**	**EQR**	**PSite**	**~**	**EQR**
1995	NA		NA	POOR	=	POOR	POOR	(+)	MOD
1997	POOR	=	POOR	BAD	=	BAD	MOD	(-)	POOR
1998	BAD	=	BAD	BAD	**(+)**	MOD	MOD	=	MOD
1999	NA		NA	NA		NA	MOD	=	MOD
2001	BAD	=	BAD	BAD	=	BAD	MOD	=	MOD
2002	POOR	(+)	MOD	BAD	=	BAD	MOD	(-)	POOR
2003	POOR	(-)	BAD	BAD	=	BAD	MOD	=	MOD
2004	POOR	=	POOR	BAD	(+)	POOR	NA		NA
2005	POOR	(+)	MOD	BAD	**(+)**	MOD	NA		NA
2006	POOR	=	POOR	BAD	(+)	POOR	MOD	(-)	POOR
2007	POOR	(+)	MOD	POOR	(+)	MOD	MOD	=	MOD
2008	POOR	=	POOR	POOR	=	POOR	NA		NA

NA: not applicable and ~ gives the relation between PSite and EQR appreciation, in bold blue more than one class difference is observed (large type II error)

The EQR scores for the independent survey data are compared with the PSite per zone and year (Table 8.13). Allowing a one class difference all zones are correctly scored but in 2008 there is a tendency to over-score the freshwater zone (16.6% Type II error).

Table 8.13: Pre-classification (PSite) and Ecological Quality Ratio (EQR) appreciation for independent fish data in the three zones in 2007 and 2008. (MOD: moderate)

Zone / year	Freshwater		Oligohaline		Mesohaline	
	PSite	**EQR**	**PSite**	**EQR**	**PSite**	**EQR**
2007	POOR	POOR	POOR	POOR	MOD	MOD
2008	POOR	MOD	POOR	POOR	MOD	MOD

4 Discussion

4.1 Pre-classification

The objective identification of environmental degradation in the various estuarine salinity zones has proven its worth (Deegan *et al*., 1997; Tong, 2001; Aubry & Elliott, 2006; Coumans *et al*., 2006; Breine *et al*., 2007; Vasconcelos *et al*., 2007; Elliott *et al*., 2008c). The purpose of the pre-classification was to quantify the anthropogenic impact in each salinity zone and to relate this to the scoring of metrics that are relevant for the estuarine fish community. The pressures (stressors) were chosen for their effects on the fish community (e.g. MEMG, 2003;

Wolter & Arlinghaus, 2003; Turnpenny *et al*., 2004; Kruk, 2007). A depletion of dissolved oxygen usually results from an excessive organic pollution and generates a significant interference with migration routes (NRA, 1993). Dissolved oxygen therefore incorporates information about organic and inorganic components (N, P, etc…) which are not included here to avoid redundancy. The reaction of the metric values to the predictor minimum dissolved oxygen average was obvious and corresponded with the findings of previous studies in the Zeeschelde estuary (Maes *et al*., 2005, 2007, 2008) as well as in other estuaries (e.g. Araújo *et al*., 2000; Turnpenny *et al*., 2006; Gutiérrez-Estrada *et al*., 2008). The loss of intertidal area, land claim, dredging, port and industrial activities all have resulted in the loss of habitat diversity and quality. Numerous are the studies which have demonstrated the positive effect of habitat diversity on fish assemblage structures (e.g. Gorman & Karr, 1978; Belliard *et al*., 1999; Chesney *et al*., 2000; Oberdorff *et al*. 2001). The effect on fish of some pressures in the estuary may be masked due to the effect of scale (zone length). Indeed dredging occurs at a local scale while fish act over a wide range of spatial and temporal scales (Habersack, 2000).

We decided to combine different pressures (chemical and physical) into a single pressure variable as already done by Breine *et al*. (2007) and Degerman *et al*. (2007). Care was taken to consider impacts acting on different scales (spatial and temporal). Combining pressures in an integrated indicator is an essential tool to assess the anthropogenic pressures on estuarine systems (Vasconcelos *et al*., 2007). It has the benefit of combining the current condition of an element of the estuary (e.g. DO) with elements representing a change over time (e.g. land claim). Finally our approach was coherent with the method developed by Aubry and Elliot (2006). However, the purpose of the pre-classification was not to assist with the metric scoring (e.g. in Breine *et al*., 2007 and chapter 7) but to select responding metrics.

4.2 Fish sampling

In chapter 7 we pleaded for a rapid and effective method to sample the estuarine fish community. Preliminary results from a gear intercalibration exercise at different estuaries in Ireland (Whyte *et al*., 2007) indicated that for species diversity, the results of fyke net catches are comparable to those obtained with other types of gears (e.g. beach seine, beam trawl, otter trawl). Within the survey period (1995-2008) seasonal fluctuations in species abundance and composition are common (Maes *et al*., 2004; Chapter 2). Especially in winter, catches of most species are reduced. We suggest a standardised sampling protocol with two double fyke nets per site, sampled daily during 48 hours at least once in each season, except in winter. We therefore decided to develop an index that integrates the data from a complete year, without winter data. Seasonal effects become irrelevant because the data were grouped into annual metric values. The physical, chemical and biological characteristics of the Zeeschelde estuary change gradually along the salinity gradient and this is reflected in the fish assemblages of the different zones (Hampel *et al*., 2004; Chapter 2). The division into the indicated salinity zones (freshwater, oligohaline and mesohaline) follows previous studies (Bayens *et al*., 1998). It corresponds to the biological segment concept, which defines a segment as a section of stream in which the fish community remains generally homogeneous over a year due to the relatively uniform nature of the physical habitat (Ramm, 1988). In the oligohaline and mesohaline zone respectively 4 and 5 surveys are needed to catch 90% of the species (See Table 8.5). These zones are probably more dynamic than the freshwater tidal zone. In the oligohaline zone less surveys were performed in 1995, 2002 and 2003 which had no impact for the internal validation (Tables 8.3 and 8.12). In the mesohaline zone there were less fishing occasion in 1997 and 2006 (Table 8.3) and EQR scores were poor in these years which could be due to a sampling effect (Table 8.12).

4.3 Selection of metrics

To accurately determine the biological integrity of the estuary, the index has to incorporate biotic responses from individuals to ecosystems (Gerritsen, 1995; Pinto *et al*., 2009). The combination of several metrics, each providing information on a biological attribute of the estuarine fish community, allows to determine the systems' overall status and condition. This emphasizes the importance of metric choice. The first step in the selection was a graphical screening method based on the initial IBI (Index of Biotic Integrity) approach (Karr, 1981). This approach has been adopted in the development of other estuarine indices (e.g. Deegan *et al*., 1997; Chapter 7). But here the graphical analysis (i.e. metric response versus PSite) was not the basis of the scoring system, since this would limit the discriminating power of the index to three classes (see Chapter 7). The assignment of fish species to the different metrics was based on previous work (Belpaire *et al*., 2000; Breine *et al*., 2001, 2004, 2007; Elliott *et al*., 2007; Franco *et al*., 2008). We limited our choice to species from the reference list (Chapter 3). Hence exotic species defined by Verreycken *et al*. (2007) were omitted. However, *Sander lucioperca* was kept in the list because of its ecological needs (e.g. high DO), migratory behaviour (Koed *et al*., 2002) and its role as a top predator. The reference list is zone specific. As a consequence stragglers for a particular zone were not included in any of its candidate metrics since they have no ecological function in a particular zone. This assured that the metrics were zone specific and relevant. The second step was a trade-off between response and redundancy. The factor loadings of the PCA with the log transformed metric values indicated low responding metrics (Hughes et al., 1989). We associated the low values of factor loadings with a small contribution to the variance. The sign association of the individual loadings also indicates consistency among metrics (Hughes *et al*., 1989). Correlation between metric values was used as a redundancy measure (Hughes *et al*., 1989; Breine *et al*., 2004). Not all categories (metrics) from the guilds were selected. Since most of the metrics have species in common the selected metrics cover all estuarine functions: spawning, nursery, feeding, connectivity and shelter as described in McLusky & Elliott (2004). Some metrics are relevant in different zones (e.g. total number of species or diadromous species), but they are comprised of different species and have different threshold values.

4.5 Properties of the selected metrics

The total number of species (MnsTot) is a typical metric in freshwater IBI's (Hughes & Oberdorff, 1999) and in estuarine IBI's (Borja *et al*., 2004; Harrison & Whitfield, 2004; Coates *et al*., 2006). Species richness decreases with increasing human impact and provides the simplest measure of species diversity (Karr *et al*., 1986; Miller *et al*., 1988; Faush *et al*., 1990; Belpaire *et al*., 2000; Breine *et al*., 2004). This metric combines a quantitative, a qualitative and a functional aspect (Jager & Kranenbarg, 2004). It was not retained in the Dutch estuarine index as data varied between season and gear (Jager & Kranenbarg, 2004). The total number of individuals (MnsInd) is used in the freshwater and oligohaline zone, but is rarely included in other estuarine indices (Coates *et al*., 2006). The number of individuals collected is also a function of the size of the sample (Magurran, 1988). In our case all catch results were obtained with the same method and recalculated to catch per unit effort (CPUE) i.e. all catches of one year in one zone are added and transformed to catch per day per fyke net. More common metrics are the percentage metrics, e.g. percentage of diadromous fishes (MpiDia). The diadromous metric gives essential information about the connectivity in the estuary and is sensitive to migration barriers (Breine *et al*., 2004). Although it is an important metric it is absent in many estuarine indices. In England this metric is embedded in a 'functional guild composition metric' and in the metric 'presence of indicator species' that provides a measure of disturbance, sensitivity to oxygen or other specific traits (Coates *et al*., 2007). Jager and Kranenbarg (2004) use in The Netherlands the diadromous species metric to evaluate the continuity of the estuary, the presence of spawning grounds and the DO conditions. For the freshwater zone the metric ‘specialised spawners’ (MpiSpa) contains species with special requirements during the reproductive phase and includes gravel spawners, phytophylic and ostracophilic spawners in accordance with Didier (1997). It detects disturbances in the spawning habitat and decreases with increasing degradation (Kestemont *et al*., 2000). For the mesohaline zone we considered species exhibiting parental care, which are also habitat sensitive species (Table E, annex). Habitat sensitive species (MnsHab) need shelter to grow up and have special demands about their habitat. This metric includes information on habitat disturbance such as canalisation and bank reinforcement. For the mesohaline zone habitat sensitive species are mainly estuarine species. As a result this metric covers the nursery function assessment (Table E, see annex). Habitat sensitive species metrics are rarely found in other IBIs, where in general the intolerant species metric is used as an alternative. Here species tolerance to oxygen depletion was used to assign species the status of either tolerant or intolerant. A habitat sensitive species metric is also found in other estuarine indices as an indicator species metric (Borja *et al*., 2004; Breine *et al*., 2007; Coates *et al*., 2007). The benthic species metric (MpiBen, freshwater zone) is a stratum metric and should not be confused with the trophic metric used by Coates *et al.* (2007). This metric assesses habitat quality and most of its species are habitat or fragment sensitive (Table E, annex). These species are vulnerable to dredging and sedimentation load. In the mesohaline zone no trophic metrics were selected since none of them seemed to react to pressure. This agrees with previous observations where trophic level metrics did not distinguish habitat quality (Mikkelson, 1993; Deegan *et al*., 1997). In other European indices feeding guilds are included since they assess the trophic structure of estuarine fish communities (see Chapter 7). However, it has been reported that fish behave as opportunistic feeders in estuaries (meso- and polyhaline zones) (Elliott & Hemingway, 2002; Elliott *et al*., 2007). In the freshwater and oligohaline zones piscivores were selected. The percentage of trophic specialists such as piscivores declines with increasing degradation (Fausch *et al*., 1990). These metrics are very sensitive to increasing pressure and are often integrated in an IBI (Hughes & Oberdorff, 1999). From the estuarine functional use group (EUFG, Table E, annex) we retained the marine migrants and the estuarine species. Marine migrants are mostly migrating juveniles frequenting the estuary for food and shelter when conditions are favourable (Franco *et al*., 2008). They remain in the estuary for short and/or seasonal periods. The marine migrant metric assesses therefore the nursery function of the estuary (Elliott *et al*., 1990). The estuarine species metric provides information about the nursery and shelter function of the estuary and is commonly used in estuarine indices (Borja *et al*., 2004; Jager & Kranenbarg, 2004; Coates *et al*., 2007). Deegan *et al*. (1997) divided this metric into nursery species and estuarine spawners to assess the nursery function of the estuary. Estuarine species may complete their life cycle in the estuary and are therefore sensitive to habitat changes and toxic substances in the estuary (Jager & Kranenbarg, 2004). They often have specific reproduction strategies to adapt to the extreme dynamic situations in estuaries.

4.6 Metric scoring

Several methodologies are applied to determine metric scoring criteria. In all methods however, reference sites play a major role (Adriaenssens *et al*., 2000a). Metric thresholds should not be based on expert judgement but rather on the evaluation of the zone-specific data (Seegert, 2000a). Minimally disturbed sites are sometimes used as a reference to select optimal metric scores and score classes are determined by dividing the total metric range in three or five equal portions (assuming a linear behaviour of the metrics) (Deegan *et al*., 1997; Hughes *et al*., 2002; Harrison & Whitfield, 2004; Breine *et al*., 2007). The absence of minimally disturbed reference sites forced us to develop another approach to determine threshold values. Various techniques have been used in order to remove subjectivity when applying the eye-fit method to define threshold values, e.g. regression analysis (Didier, 1997; Liang & Menzel, 1997; Goffaux *et al*., 2001; Breine *et al*., 2004) and area under the curve method (Breine *et al*., 2007, Chapter 7). These approaches use the pre-classification classes to define threshold values. Here a reference list for fish in the salinity zones of the Zeeschelde estuary elaborated in Breine *et al*. (2008b) was used to define threshold values. First a historical species list was compiled and this list was then adjusted with recent data obtained with various fishing gears. The alterations were made according to criteria described by Ramm (1990). Correction for the catch failures by using the 90 percentage values seems a reasonable approach. Dividing the values in equal parts is a widely applied approach for indices (Goffaux *et al*., 2001). The reference conditions were developed independently for each zone and therefore the index values can be directly compared despite differences in fish communities (Harrison & Whitfield, 2006).

4.7 Z-EBI score

Historically trisection was done on the metric values (Karr, 1981). In chapter 7 the EBI score was calibrated by fixing the type I error at 10% for each classification class. Here, we first set a minimum number of species to avoid over-scoring. The Z-EBI score is obtained by the sum of the metric scores and transformed to an EQR to comply with the WFD. It is assumed that all metrics have equal weights in terms of their contribution to the index. To facilitate the interpretation of the EQR, the final values are rated from the status bad to good ecological potential (GEP). The Z-EBI is able to discern differences in individual metrics which can determine different stressors and effects. These metrics can be presented as radar plots (Fig. 8.3). The internal validation of the calculated EQR's is to be considered as an initial test of the overall accuracy and pertinence of the developed index. To define tendencies of under- or over-estimation we allowed a one class difference between the PSite and the EQR similar to Goffaux *et al*. (2001) and Breine *et al*. (2004). Results indicate that the indices distinguish among various levels of degradation within the pre-classified sites (94% matches). In the freshwater and oligohaline zone a tendency to over-estimate the status (small type I error) is recorded while in the mesohaline zone there is a tendency to under-estimate (small type II error). The latter is probably an effect of sample effort as explained in section 4.2. An even better match was observed using independent data (100%).

Although the improved water quality is reflected in the fish assemblages (Chapter 2), the EQRs indicate that the ecological status of the Zeeschelde in the various salinity zones has not yet reached the goal set by the European Water Framework Directive. The implementation of the Updated Sigmaplan (see Introduction) is an essential step in the realisation of that goal.

5 Conclusions

The compilation of all survey data within one year in a salinity zone to calculate metric values represents a new approach to assess the ecological quality of an aquatic ecosystem. The index accounts for the seasonal variation and it covers all zones of the Zeeschelde, which is an improvement compared to the brackish index developed in chapter 7. As such we deal with the criticism of using 'snapshots' to evaluate the ecological quality of surface water; the index covers the temporal and spatial variation of the estuary. The selected metrics are relevant for each particular salinity zone, having a good discriminating power and they are easy to measure. They allow the appropriate assessment of anthropogenic impacts on fish communities. The selected metrics assess several aspects of the estuarine functions for fishes, such as foraging and nursery habitat and migration route. The historical reference is adapted to the good ecological potential status, which provides a realistically achievable goal. The indices respond also to the criteria stipulated in the European Water Framework Directive. At the same time it is a clear communication tool for environmental managers, politicians and other user groups.

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Chapter 9

General discussion: A critical view on fish assemblages and estuarine management

The overall aim of this study was to develop a tool based on fish assemblages to assess the ecological quality status of the Zeeschelde. The tool, in this case the fish index, should be relevant, simple and easily understood, scientifically justifiable, quantitative and acceptable in terms of cost. The fish index should respond to small variations in environmental stress, be insensitive to natural spatial or temporal variations and should be related to management goals. The developed approach should be transferable to other estuaries, provided that the relevant adaptations can be made. The approach should represent a substantial contribution, with regard to fish, to national and international policy instruments designed for sustainable management and conservation of the aquatic environment.

The results presented in different chapters tell us that although fish assemblages in the estuary show a complex spatial and temporal pattern it was possible to develop a fish-based index to assess the ecological quality of an estuary, in casu the Zeeschelde. The presented approach to develop indices is new and, as far as I know, unique in Europe as it encompasses three different salinity gradients including the freshwater zone. During my research I became aware of some gaps in the knowledge about the ecological functions in the estuary for fish. In this final chapter I highlight some of the gaps and how these could be assessed. Some methodological issues and how the results can be translated into management measures are discussed. Finally I question if the current implementation of the international policy instruments guarantee sustainable management and conservation for the fish communities in the Zeeschelde.

Fish habitat use in estuaries: nursery

The erroneous or incomplete attribution of functions in fish life cycles in relation to estuarine habitat types may jeopardise the effectiveness of estuarine management decisions for ecosystem functioning. While defining the functions of the estuarine habitat for fish an arising issue was that of the nursery function. It is common knowledge that for the juveniles of some marine fish species estuaries are an extension of the coastal area which they explore if conditions, especially temperature, are favourable (Maes *et al*., 2005; Guelinckx, 2008) or if specific olfactory cues are present (Vrieze & Sorenson, 2001; Strydom, 2003). Still, the estuary can play an important role as a nursery for those accidental visitors (Wilson, 2002; Martinho *et al*., 2007) as well as for diadromous, estuarine and freshwater species.

However, one of the most difficult issues in an estuary is precisely to define whether a habitat functions as a nursery (Beck *et al*., 2001; Turpie *et al*., 2002). A nursery is a habitat that compared to other habitats, supports greater contributions to adult recruitment from any combination of density, growth, survival of juveniles and movement to the adult habitats (Beck *et al*., 2001). Identifying habitats that serve as nurseries are important for fishery purposes and for conservation and management plans (Habitats Directive). Therefore I recommend a four steps sample design to assess the nursery function of an estuarine habitat.

First identify species in the estuary that have juveniles and adults in separate habitats (zone). Species concerned here include the marine migrants, estuarine and freshwater species and diadromous species.

Secondly**,** identify habitats with a high density of juveniles (calculated per unit area). The juvenile areas are structurally complex and shallow places that buffer the tidal regime and they are not supposed to be in the main channel of the estuary. Comparison analysis indicates the habitats with the highest densities which are selected for further research. We have to be aware that species density at a site is affected by sampling efficiency, natural events and human impacts. Long-term standardised surveys should determine trends in order to overcome the problems related to natural variations. A combination of methods is to be used in order to increase the catch efficiency (Whitfield & Marais, 1999): seine netting, kick sampling, electric fishing, fyke nets and other types of standing nets can catch juveniles of some species if the mesh size is small enough. Gill nets are to be avoided due to the large mortalities they cause.

Thirdly**,** investigate if the juveniles reach the adult stage. The juvenile population has to be sampled at regular intervals to assess the grow rate.

Fourth assess the movement of juveniles to the adult habitat. The movement to adult habitats must show a disjunction i.e. the habitats are separated and juveniles move from one habitat to another. To assess the movement of juveniles it is difficult to use conventional marking methods (internal or external tagging) because the fish are very small and some techniques impair the swimming ability (fin clipping). Marking with a subcutaneous injection of stained compounds could be useful. Even then it is difficult to distinguish among individuals that have migrated at different times; the sample interval and moment can bias abundance estimations. It is difficult to assess this movement especially for marine species because of their high dispersion and mortality rates (Guelinckx, 2008). Movement can be assessed using biogeochemical tracers (isotopic clock) and otolith chronologies, a method developed to assess the migration dynamics of sand goby (*P. minutus*) by Guelinckx *et al*. (2008). With this approach, assumptions concerning the habitat use of a particular species can be verified. It involves quite some effort but if needed this kind of research could be done in collaboration with several research institutes.

The proposed sample strategy is complicated, clearly requires a long enough period and involves an important effort. However, it is important because at present assumptions are made based on single-factor studies providing incomplete answers to the estuarine functions for fish, especially the nursery functions.

<u>Methodological issues</u>

The definition of reference conditions

The concept of a reference condition is considered critical for an appropriate understanding of the relationship between biological conditions and anthropogenic disturbance (Stoddard *et al*., 2006). The EU CIS (Common Implementation Strategy for the WFD) guidance 2.4 for coastal and transitional waters proposes four methods to define reference conditions: an existing reference site, historical reference, modelling or expert judgment. In Europe there are no pristine estuaries that could serve as a reference for the Schelde estuary. I combined historical data with recent data and expert judgment. From these reference lists quantitative guild lists were developed. As reference conditions show that natural variation exists, this should be considered in the scoring system and I therefore expressed our reference values as a range.

The unbiased fish-relevant pre-classification of the sites in the Zeeschelde

The pre-classification of sites according to acting pressures was used as a method to validate metrics and the index scoring ability. I realize that the approach partially captures the pressures present in the estuary and I am aware that the relative impact of each pressure on the habitat quality can be quite different, such that a simple sum of pressure scores can give misleading results. However, it is not necessary to have an absolute measure of quality of the sites involved, as for the development of an index it is crucial that the sites are well ranked with respect to each other. It is essential to get an unbiased, fish relevant pre-classification of sites in order to combine the right metrics into a performant index. A suggestion could be the use of environmental integrative indicators (EIIs) as proposed by Aubry and Elliott (2006). These authors defined environmental indicators and grouped them into three broad indices. The first environmental indicator (EII 1) is a state and impact indicator, which provides an indication of the nature in which the coastline has been modified by nearshore activities. It concentrates on the morphological change, using predominantly physical characteristics. It also gives information on climate variability, including sea level rise, in order to reflect the natural dynamism of the area. The second EII 2 is a pressure indicator which details the amount of the main activities responsible for coastal disturbance e.g. amount of dredging, fishing, oil rigs, wind farms, vessel movements etc. The third EII 3 is a state of impact indicator. It aims to give the status of the natural environment and to assess the impacts of the environmental changes represented in EII 1 and EII 2. Some component indicators relate to the quality of the water and sediments (concentration of pollutants), whereas others relate to the conditions of the habitats and their ability to maintain viable populations. Fish assemblages could be assessed in function of each EII separately after elimination of redundant and irrelevant stressors and possibly including extra system specific stressors. In the frame of an EU InterregIII project (Harbasins) environmental integrative indicators were calculated for the Schelde estuary including the Westerschelde, the Humber and the Eems (Elliott *et al*., 2008c, Figures A1-3, annex). On average the degree of change due to EII 1 impacts is the highest in the Schelde estuary. Morphological changes are higher in the Zeeschelde than in the Westerschelde (Fig. A1). The change caused by the use of resources (EII 2) is clearly higher in the Humber and stays more or less constant from the top to the mouth in both the Ems and the Humber Estuary. In the Zeeschelde this impact is less than in the Westerschelde (Fig. A2). Figure A3 indicates for the Schelde estuary a clear increase in environmental quality (EII 3) downstream from the Belgian/Dutch border towards the mouth of the estuary. Another approach could be to use other variables, such as those used in the River Habitat Survey (Raven *et al*., 2000) including scale effects to assess the importance of reach-scale versus watershed-scale variables (Frimpong *et al*., 2005). Raven *et al*. (2000) considered 7 mean habitat features for a standard site of 500 m length of river channel and its corridor extending 50 m outwards on either side. Instead of pre-classifying sites within an estuary one could also work at a higher ecosystem scale and pre-classify complete estuaries to get a range of good, moderate, poor and bad estuaries. Recently data have been recorded in different estuaries as an intercalibration exercise for the WFD. This intercalibration process aims at consistency and comparability of the classification results for the biological quality elements across member states. The essence of intercalibration must be to ensure that each assessment method's Ecological Quality Ratio (EQR) scale is calibrated against agreed benchmark conditions. This will ensure that the results of different methods are equivalent for the type of concern, despite potentially having different EQR boundary values. This work is still in progress and pre-classification of estuaries could be part of that exercise.

The use of unbalanced fish data between and within different salinity zones

The collection of field data in estuaries should provide accurate information at the most relevant spatial and temporal scale about those biota most influenced by human nature (see also Karr, 2006). In other words our data should inform about the effects of specific human impacts. Cooperation of volunteers can be helpful to complete the picture of the complex aquatic biota (see Stevens *et al*., 2009).

Biological systems are complex and variable in space and time and therefore it is very difficult to design a successful sampling design for the bioassessment. Previous IBI papers state that the IBI requires a sampling effort that collects all species in proportion to their true relative abundance (Karr, 1981; Fausch *et al*., 1990). For estuaries this is impossible even with multi-method approaches.

I suggest a standardised sampling protocol with two double fyke nets per site that are sampled daily during 48 h and at least once in all seasons except winter. Six survey sites were defined in the different salinity zones in the Zeeschelde: Zandvliet, Antwerpen, Steendorp, Kastel, Uitbergen and Overbeke. In the tributaries one site in the tidal Zenne was defined, two in the Durme and three in the Rupel and none in the Netes and Dijle. Sampling at least three times in one zone will incorporate the variability of the estuary. Less samples could increase the variance in IBI score (Hughes *et al*., 1998). Disturbed systems have essentially lost their capacity to buffer natural variation (Toth *et al*., 1982; Yoder, 1991). Hence a high variance is usually an indication of human disturbance (Karr *et al*., 1987; Fore *et al*., 1994), in this case it would be the result of too few records. Fykes are used as they sample a constant fraction of the fish population (van der Meer *et al*., 1995) and are easy to handle.

I also suggest to use the 'same' sampling team to reduce the variability and assure data quality. To improve the robustness of our data we should avoid differences in expertise and experience. All captured fish should be determined up to species level, counted, weighed and measured (total length). Subsamples should be avoided to diminish the risk of missing rare taxa. In addition some metrics may be more affected by subsamples than others because they are constituted of less species.

Impact of space and time to assess estuarine fish communities

The data used to develop our indices were obtained with a standardised protocol, using one technique i.e. fyke nets. In one salinity zone always the same sites were surveyed for both indices to avoid obscuring of fish assemblage responses (McCormick *et al*., 2001) and it allowed to elucidate different types and intensities of impacts. The sites in the different zones were also far distant to avoid that metrics express specific characteristics of an adjacent zone (observed in Maret *et al*., 1997). Both indices, EBI and Z-EBI contain metrics that provide a clear and easily interpreted signal, but there are fundamental differences due to data handling.

The EBI assesses the estuary on a **habitat (site) scale** for a particular **month**. The advantage of this approach is that it can assess a particular site of the estuary in short notice, i.e. immediately after the survey. However, it only provides a snapshot of the status. For an index it is important to assess the different processes which should occur in the estuary. When the EBI was developed diadromous fish species were nearly absent in the Zeeschelde. As a consequence the metric “diadromous species” did not react in the stepwise regression and the model did not accept to include the metric “diadromous species”. From a statistical perspective this is a logic decision but not from an ecological viewpoint. The EBI does not address situations that may be encountered in future, with diadromous species entering the estuary. As a consequence this EBI does not assess all ecological processes in the estuary. Short term occurring natural disturbances, such as flash floods, result in varying assemblages (Matthews *et al*., 1988). As a consequence the EBI value at a site can present large variations and is therefore less stable than an index incorporating data at a larger spatial scale. Compared to the zone specific estuarine index (Z-EBI) the EBI provides less information because it assesses less fish which can create high variability in the index score (Fore *et al*., 1994). Finally EBI scores were defined by fixing the type I error. According to Schrader-Frechette and Mc-Coy (1993) the type II error is a far more serious because the benefit is given to the polluter rather than the environment which is not in keeping with the spirit of the precautionary principle.

The zone-specific estuarine index (Z-EBI) provides a classification scheme that predicts and explains fish assemblages on a salinity **zone scale**, but it has links to finer scales i.e. metrics reflect also changes at a habitat level. This index assesses the fish assemblage across the whole salinity gradient of the Zeeschelde and therefore links can be made between properties downstream that are controlled by the upstream habitat. Metric values are calculated using catch per unit effort data obtained by pooling all the data recorded during one year in one zone. The index was developed using data covering large stretches reducing the influence of longitudinal natural changes (e.g. Vannote *et al*., 1980). The metric selection was based on statistical properties and ecological knowledge. This assured that the different ecological processes in the estuary are assessed (Table F, annex). The Z-EBI can rate all quality classes (bad, poor, moderate, GEP) complying with the water framework directive.

In Table 9.1 I compare the average scores obtained with EBI and Z-EBI in the mesohaline zone. Generally in case of a different appreciation, the EBI scores lower. It confirms my view that local and temporal appreciation is too sensitive to small variations which do not necessarily mean an overall negative impact on the ecosystem functioning.

Table 9.1: Comparison of EBI and Z-EBI classification in the mesohaline zone in the Zeeschelde for the period 1995-2008. Different scores are in bold.

	1995	**1997**	**1998**	**1999**	**2001**	**2002**	**2003**	**2005**	**2006**	**2007**	**2008**
EBI	mod	poor	mod	mod	**poor**	poor	**poor**	**poor**	poor	**poor**	**bad**
Z-EBI	mod	poor	mod	mod	**mod**	poor	**mod**	**mod**	poor	**mod**	**mod**

(mod: moderate)

The Z-EBI can be used for trend assessment. To reduce natural variations, such as wet or dry years, an index that assesses every three years using a moving average approach could be developed. The wide-ranging sensitivity of the Z-EBI makes it an ideal assessment tool to evaluate the rehabilitation processes or conservation activities. As the Z-EBI covers the complete EQR range I suggest using this index for assessing the ecological status of the Zeeschelde.

I am aware that the Westerschelde should have been included in this exercise. However, to my knowledge only survey data obtained with beam trawling are available. Jager and Kranenbarg (2004) measured a moderate status for the Westerschelde.

Estuarine fish-based indices have been developed in other European countries (Table G1, annex). All of them comply with WFD and are used for ecological quality assessment purposes but none assesses in exactly the same way the ecological status of an estuary (Tables

G2-6, Annex). Few metrics are common and even then the threshold values for reference scores differ. As estuaries are defined differently, most of these indices only assess the euhaline, polyhaline and mesohaline zone. In all indices an Ecological Quality Ratio (EQR) representing the difference between monitored data and reference conditions is used. References were derived using best contemporary data plus historic records and expert advice. The EQR approach is based on the assumption that anthropogenic activities alter the physical and chemical environment, impair the conditions of biota and thus degrade the functioning of the aquatic ecosystem (Solimini *et al*., 2009). Although the philosophy used to develop an index is similar, the EQR calculated by the different countries can not be compared because the final computing was too different. The main reason is that the applied survey technology and the handling of survey data to calculate the EQR differ. This is partially because sampling protocols were initially designed to assess the fish biodiversity in rivers and estuaries and not to assess the integrity. As a consequence the size of the sample effort differs between the different countries. In the UK the multi-method approach is used whereby all data within one water body of an estuary obtained within 3 years are pooled to calculate one index value every three years (Table G5). In Spain and Portugal sampling, using trawling, is undertaken every three years and the EQR value is calculated on a site level (Table G4). In The Netherlands and Germany anchor net data are used for the Eems-Dollard (Table G6). The obtained scores at the different sites are averaged to calculate the final EQR score. In Germany only month data from spring and autumn obtained with stow net catches are used. In The Netherlands the worst status of a single metric is used to determine the overall status (“one out” “all out” principle). Monitoring designs in the different countries differ in philosophy, goals and needs and vary from target sampling (e.g. salmon in Ireland or glasseel in Belgium) to multi-method biodiversity sampling (e.g. UK).

In 2002 the WFD endorsed a document called ‘Towards a guidance on establishment of the Intercalibration network and on the process of the Intercalibration exercise’ (Intercalibration Guidance). The essence of intercalibration must be to ensure that each assessment method’s Ecological Quality Ratio (EQR) scale developed in a member state of the EU is calibrated against agreed benchmark conditions. The WFD outlined three options for intercalibration. In a first option the same assessment method is used, which is based on the same metrics and the same means of identifying reference conditions. This option does not require further intercalibration of the results. At present, within the member states, the readiness to change the national survey methodology is low. The reason given is that due to the different topographies that occur in the transitional waters identified, it is not possible to use exactly the same monitoring techniques. However, research should find out if a survey methodology works better in a particular type of estuary than in another one. A second possibility is to select common metrics and set good ecological quality status boundary values after which the common metrics are applied to suitable data within each member state. Hereto the common metric boundaries are compared with those proposed by the member state (national method) and adjusted when needed. The problem is that between the member states few common metrics exists and therefore if the assessment would be limited to those metrics not all estuarine functions would be assessed (Tables G2-6). A final option is that each member state applies its national method to a common data base i.e. from an estuary with known quality status, and adapts its boundaries by comparing the different EQR values. At present only two estuaries have been assessed and collection of additional data is needed; it is at present in progress.

The adequate interpretation of the assessment results

A final question arises: "Does the signal indicated by the index reflect reality?" The estuarine index has evolved compared to the initial IBI's, which were measuring primarily effects of organic pollution (Karr & Chu, 1997). Thanks to a selection of metrics sensitive to changes in the estuarine functioning the index is sensitive to a suite of anthropogenic impacts (Table F, annex). The index can detect problems and the nature of these by observing the individual metric scores. Next to the EQR, the radar plots indicate the individual metric scores which can be related to a particular stressor. They allow also to be used as a diagnostic tool for the index.

In the freshwater zone a higher score for diadromous species is observed compared to the other zones. This is possible because in the freshwater zone the connectivity is assessed with a metric indicating the relative percentage of diadromous fish and in the other zones with the number of diadromous species. The high abundance of eel in the freshwater zone biases the results. This illustrates that we should keep on questioning the relevance of the index, the metrics and the resulting scores. In addition new stressors might appear in the estuary that have not been assessed in the actual one e.g. climate change (e.g. Duarte 2007; Pörtner & Knust, 2007), introduced species (e.g. Harrison & Whitfield, 2004), new toxic chemicals or drugs. Climate change is a disturbance that can exert short or long-term impacts on intertidal fish distributions and other aspects of their biology (Fields *et al*., 1993). The river discharge into the estuary may be affected shifting the salinity pattern (Gleick, 2003). From previous research impacts of toxic substances on individual fish are known. The skeleton can be deformed due to the presence of pesticides and heavy metals (Labat *et al*., 1977; Aldrin, 1987), skin injuries and tumours are caused by prolonged exposure to pollution and parasites attack more easily weakened fish (Girard, 1998). Therefore, we can assume that these new stressors will cause different responses in each fish species, which are still to be assessed, but on a community level they will create a disruption of evenness. I have shown in previous work that a fish-based index should be considered a complementary instrument to other ecological indices (Triest *et al*., 2001) and therefore argue for a holistic approach using different bio-assessment tools to evaluate in conjunction the status of the estuary. Each of them has its specific sensitivity at different levels of degradation and to different kinds of stressors. In addition we should be aware that according to the metric, guild or life stage of fish species involved the biotic integrity differs in scale. For example diadromous species require integrity at a much larger scale than limnophilic species found in the upstream area of the estuary. Therefore the link between fish recruitment in the sea and the presence of marine fish in the estuary needs more attention. At sea interannual variation in temperature, food availability and predation pressure induces variation in recruit numbers (Philippart *et al*., 1996) which could explain variations in catch results observed in the estuary.

The translation of results into management measures

<u>Integration in the management plans of the Zeeschelde</u>

The results show that fish assemblages in the Zeeschelde do not reach GEP. This means that management measures have to be taken. Individual metric scores deliver more specific information about problems arising (Fig. 9.1). Figure 9.1 shows that there was a problem for estuarine species (MnsErs) and intolerant species (MnsInt) in 2006. In 2007 these problems were reduced.

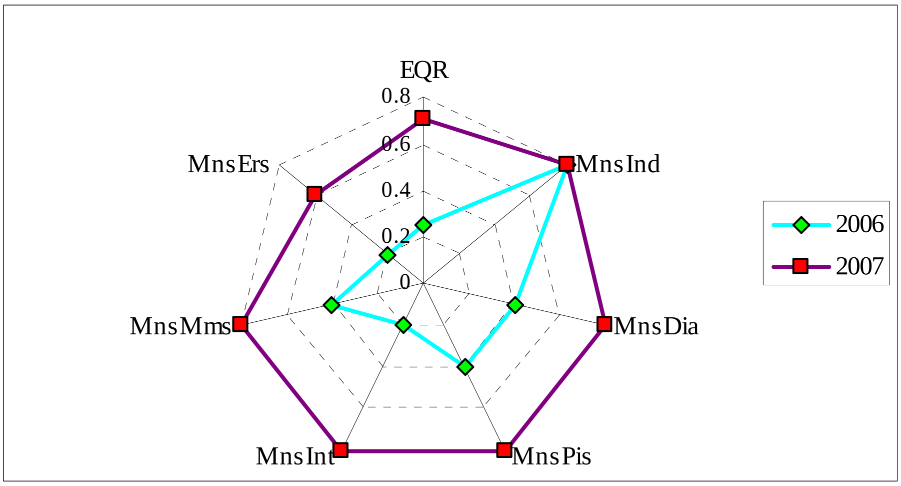

Figure 9.1: Radarplot EQR and metric scores for the oligohaline zone in 2006 and 2007. (Abbreviations see Table 8.6)

When such signals are well defined specific remediating measures can be integrated in the ecosystem management plan or management measures can be adapted likewise.

Bergerot *et al*. (2008) developed a synthetic conservation value index based on fish assemblages in rivers including the rarity, the conservation status and the species origin. This tool allows to identify restoration needs for specific sites based on a combined application of different indices. This approach could be extended to estuarine species. However, management efforts tend to concentrate in general at the scale where disturbance is perceived (Faush *et al*., 2002). This explains why Bergerot *et al.* (2008) assessed conservation needs site by site. Diadromous and marine species are less suitable for such approach, since the cause of their low rarity index value at a particular site should be assessed on a macroscale.

For morphological indicators Toffolon and Crosato (2007) selected the macroscale as the most suitable scale for management purposes, because it includes all morphological elements (intertidal areas, channels and islands) that are important for decision makers. However, Bergerot *et al*. (2008) and Toffolon and Crosato (2007) did not address the restoration process as such. The role of science is to assist in the development of options for restoration (Van den Bergh *et al*., 2005) to assess the consequences of choices (Meyer, 1997) and to adapt when and where needed in order to provide a better protection (Gray, 1999) (Fig. 9.2).

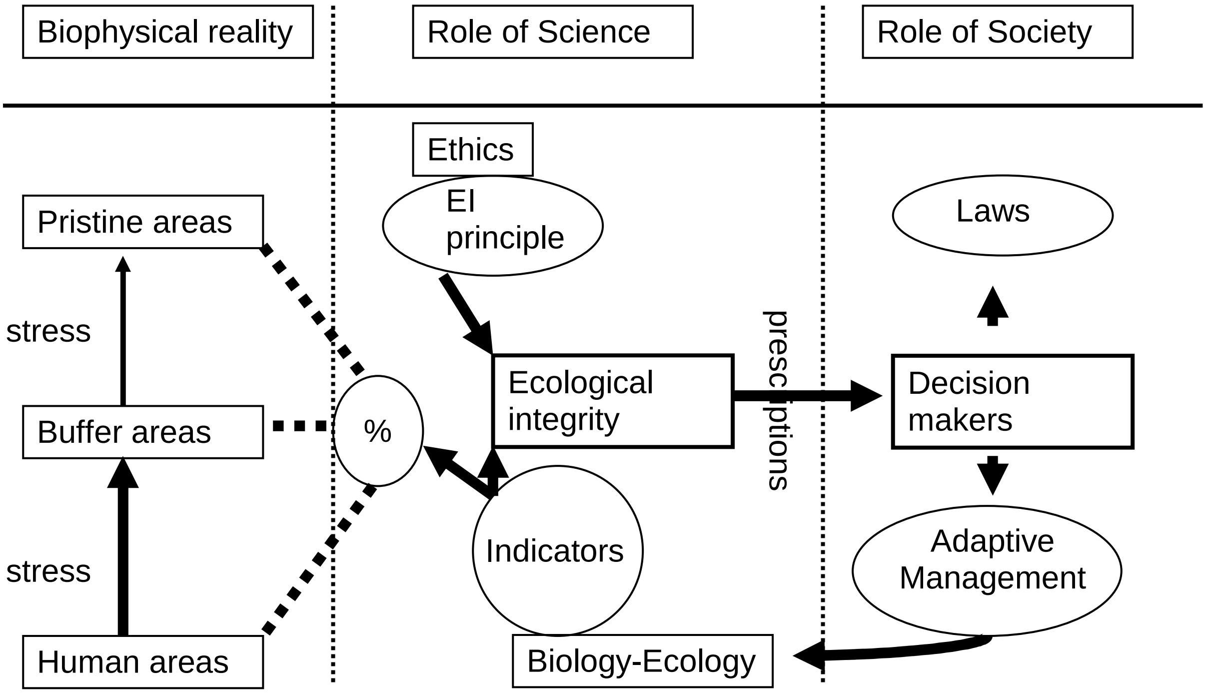

Figure 9.2: Wilderness-Normative Discourse (Adapted from Manuel-Navarrete *et al*., 2004). The percentage (e.g. surface area) of three types of biophysical realities (highest protection for pristine areas) with a particular ecological integrity (EI) is defined using biological indicators to inform the legislators and managers who execute commands, legislation, policies and regulations.

For the Zeeschelde this process was pursued along the same philosophy adding morphological, chemical and biological processes as indicators (Adriaensen *et al.* 2005; Van den Bergh *et al*. 2005). Based on a functional assessment of the estuary Adriaensen *et al.* (2005) calculated the need for 1500 ha additional tidal marshes and 500 ha of mudflats in the Zeeschelde for adequate estuarine functioning. On a lower ecosystem level they then defined more specific conservation goals for habitats and species. The optimal combination of ecological rehabilitation measures along the estuary to realise these conservation goals was proposed in a few optimised ecological restoration scenarios. After public consultation and assessment for other societal functions, the preferred scenario was designed (Ministry of the Flemish government, 2005) and adopted for realisation. A major issue was the coupling of ecological rehabilitation and the creation of sustainable nature with flood control measures, navigation requisites, port development and enhancement of the estuaries educational and recreational values. Finally it focuses on large enough scales, in time and dimensions to be appropriate for management.

The major ecological rehabilitation issues identified for the Zeeschelde were: space for processes, connectivity in the habitat network and specific habitat functions for species. The proposed rehabilitation measures include the creation of 1400 ha tidal wetland through managed realignment, 1100 ha tidal wetland under reduced controlled tide in flood control areas (FCA-CRT), 1500 ha of 'winter bed' for the upper reaches and 2000 ha of non tidal wetlands, 1000 ha of which in flood control areas (FCA-Wetland). Fish was not the major conservation issue in this plan, but was part of it.

The results indicate that the fish communities will benefit from this rehabilitation scenario (see evaluation of management measures on fish). The mudflats are important foraging grounds for young flatfishes, herring and seabass (Stevens, 2006) and the tidal marshes provide food and shelter (Chapter 6). Flood control areas with permanent pools and ditches could act as spawning and nursery areas for freshwater species as long as there is connectivity, even temporal, with the main river. The productivity and the migration possibilities in the existing and planned flood control areas should be assessed and optimisation possibilities of the design for fish should be investigated more thoroughly, because it is not unlikely that those areas have the best potentials as nurseries in the Zeeschelde. It would be a missed opportunity not to include this more specifically in the rehabilitation plans.

Pas *et al*. (1998) state that Tielrodebroek, a flood control area of about 90 ha near the confluence of the Durme with the Zeeschelde, functions as a spawning and nursery area for freshwater species following occasional inundation with river water. Simoens *et al*. (2007) presented the importance of the Lippenbroek, a 10 ha flood control area under controlled reduced tide, for juvenile fish species in the Zeeschelde. On the other hand even if a habitat covers only a small surface, it remains a nursery as long as it produces more adult recruits per unit area than other juvenile habitats (Beck *et al*., 2001). This could be an important issue when priorities must be set because of a limiting budget.

<u>Evaluation of management measures on fish</u>

Managed realignment and introduction of a controlled reduced tidal regime in a flood control area are two rehabilitation measures that add space and estuarine habitat to the river. In the Zeeschelde a few small scale restoration projects have already been realised in the frame of the preferred scenario (Fig. 9.3). In the framework of an EU InterregIII project (Harbasins) development of these sites as fish habitat was evaluated.

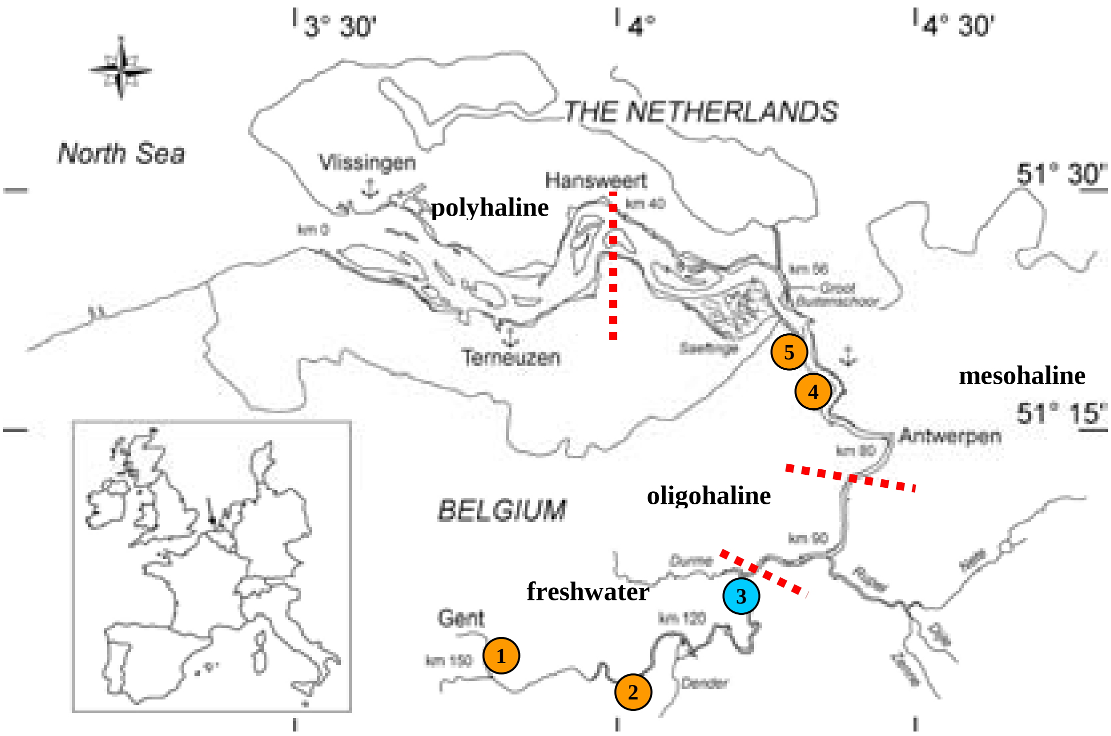

Figure 9.3: Fish sampling sites in the Zeeschelde located at restoration sites: Heusden (1), Paddenbeek (2), Lippenbroek (3), Ketenisse (4) and Paardenschor (5). Orange circles are managed realignment projects blue represents an experimental flood control area under controlled reduced tide (FCA-CRT).

The restoration sites were surveyed in March and July 2007 using the protocol described in chapters 2 (fyke nets) and 6 (winged nets). The two sites in the mesohaline zone (Paardenschor and Ketenisse) were created by realignment of dikes and levelling of previously heightened terrain. Restoration works at the Paardenschor site started in 2003 and were completed in 2004. The works at the Ketenisse site started in December 2001 and were completed in January 2003. Both sites were designed with the ‘nature by self design’ philosophy. They were levelled under MHW with a gentle slope, started as tidal mudflats and are developing towards a mosaic of mudflats and marshes. At the time of sampling the Paardenschor was 77% mudflat, Ketenisse about 40%. Results showed that both marshes were frequented most by juvenile flounder and herring. Species composition was in general in agreement to what was found on the existing marshes in the area. The Paddenbeek is a small and narrow realignment (2 ha), constructed in terraces in the freshwater zone in 2004. Apart from a badly drained mud pool and the lower mudflat terrace, it now consists of a mosaic of typical fresh tidal vegetation types: reed and willow. Although only five species were caught in this area, they were again mostly juveniles. The freshwater site near Heusden is larger and has a greater diversity of habitats including a pond, some shallow pools (< 1 m deep) and drainage channels (constructed in 2006). The impact of habitat diversity is reflected in higher species diversity (14 fish species, including flounder and thinlip mullet). Most abundant were stone moroko, roach and three-spined stickleback. The presence of 0+ and older specimens witnessed the nursery function of the pools. Pike-perch was recorded on the adjacent mudflat but was not caught in the marsh. Lippenbroek is an experimental (10 ha) flood control area under reduced tide (FCA-CRT) in the freshwater zone and is functional since March 2006. Prior to construction six 'pioneer' species (e.g. three-spined stickleback, stone moroko, Prussian carp) were recorded in the area, but also bitterling. This compares to catch results in Tielrodebroek, a functional FCA, still in agricultural use, where 10 freshwater species were collected (Pas *et al*., 1998) after temporal opening of the drainage sluices to allow river water entering the floodplain. With the introduction of the reduced tidal regime migration suitability as fish habitat increased in Lippenbroek (Simoens *et al.*, 2007). The results suggest that Lippenbroek now functions as a spawning site for freshwater species (Prussian carp, three-spined stickleback, stone moroko) and as a nursery for flounder. The higher water temperatures in the shallow water (compared to the adjacent Schelde), the relative low dynamics, the food availability and the presence of permanent pools seem to enhance the carrying capacity of FCA-CRT for fish.

These results suggest that managed realignment and introduction of a controlled reduced tidal regime in a flood control area are two rehabilitation measures that add space and estuarine habitat for fish.

Ecosystem based management

Sustainability has become an explicitly stated and mandated goal of natural resource management and the ecosystem approach has been adopted as a philosophy for managing the human uses and effects on the system (Christensen *et al*., 1996; CBD, 2004). We have tried to contribute to this kind of policy instruments in regards to fish. A major constraint to the implementation of ecosystem based management is that politically relevant scales are rather local and limited in time as opposed to the intentions of the policy instruments to be implemented such as the Habitats Directive, the Water Framework Directive and the Marine Strategy. Although there is a perception of biological degradation reflected in the enforcement of these instruments, political institutions (legislative and regulatory agencies) must balance competing values and preferences. Scientific information is merely one facet of their decision making. That is why the merits of scientifically based management plans can only be fully effective and therefore also be accepted if social, cultural, economical, health and ethical arguments are included (Davis & Slobodkin, 2004). There is a new trend appearing whereby ecologists collaborate interdisciplinary with economists and sociologists in order to create rehabilitation and conservation plans that take into consideration nature, social (e.g. willingness-to-pay) and economical interests (Costanza, 2003). Such an approach is more likely to find acceptance with the public, policy makers and environmental managers (Ducrotoy & Elliott, 2006).

The question arises if the Flemish approach to the implementation of these instruments adheres to the principles of ecosystem approach and if it will ultimately enhance the development and conservation of sustainable fish communities in the Zeeschelde.

An ecosystem approach is based on the application of appropriate scientific methodologies and focuses on levels of biological organisation, which encompass the essential structure, processes, functions and interactions among organisms and their environment. The scale of analysis and action should be determined by the problem addressed. In the Flemish situation the management approach is indeed based on the results of appropriate scientific methodologies, and at a scale fit to the addressed issue. However, when it comes to the translation into practical applications, policy and politics need to be considerated. Boundaries and issues are no longer nested on the appropriate ecosystem level but are assigned according to administrative boundaries and responsibilities. As a result different management plans overlap in space, time and issues. This approach risks being counter-productive or at least leaving gaps of unattended aspects. For the WFD the Schelde estuary was not considered as one transnational ecosystem but divided into 8 separate water bodies whereby the fresh tidal water bodies were categorised as rivers. This administrative separation into different management units might jeopardise the intended ecosystem management approach. Similarly the incomplete protection of the estuary under the habitats directive (BE2300006 ‘the estuary of Schelde and Durme between Gent and the Dutch-Belgian border’) only delimits some specific habitats in some parts of the estuary, using even a typology which is not congruent with the WFD typology. Such fragmented protection and management decisions for some specific habitats within the estuary are not beneficial for the sustainability of the complete ecosystem.

The ecosystem approach requires adaptive management to deal with the complex and dynamic nature of ecosystems and the absence of complete knowledge. Once a vision or plan has been approved and adopted politically, it is very hard to reverse or adapt it to recent new findings and insights. This brings about the danger of knowingly executing suboptimal measures, which means spending tax payers' money in a suboptimal way. Some of the habitats directive relevant species for the Zeeschelde, such as sea lamprey and twaite shad, were registered on the red list of fish species as extinct in Belgium (Vandelanootte *et al*., 1998). Therefore they were not included as species of special interest for BE 2300006. However, since that time they have been recorded from the Zeeschelde. Catches of sea lamprey are still rare but since 2003 twaite shad is often recorded in the Zeeschelde and if well managed, a sustainable population may re-establish this species once very abundant (Vrielynck *et al*., 2003). However, if it does not appear on the list of species of special interest because red lists and the designation of Natura 2000 are politically difficult to adjust to the changing situations and actual scientific knowledge, the danger will be there that specific needs for this species will not be considered in management decisions because there is no juridical need to do so.

Finally

The major issue of the preferred alternative to the updated Sigmaplan was the coupling of ecological rehabilitation and sustainable nature with flood control measures and navigation requisites.

The Z-EBI assesses the status of the Zeeschelde estuary and can be used to follow up the implementation of rehabilitation measures, as there is a link between fish assemblages and these plans. The developed approach can be applied to any other estuary.

To assess possible causes of disturbance I advocate measuring other environmental variables as well. Long-term monitoring is essential because the ability of a system to withstand a perturbation can change. In addition it can also take a while before positive results of some imposed measures become visible. This is also necessary to assess climate change effects.

Researchers should aim at implementing newly developed theories and methods in management strategies e.g. improvements in assessment tools incorporating new stressors. Climate change as well as some other stressors are not an isolated issue and therefore there is the need to create an international platform joining different programmes and initiatives around the North Sea and globally (e.g. Coastwatch (Ireland), Biodiversity Platform (Belgium), Science for Environment Policy (EC), National Oceanic and Atmospheric Administration (US), Ocean Biogeographic Information System, the International Society for Ecosystem Health (ISEH), Greenpeace and many more). The main aims of such a platform should be the collection and integration of new discoveries, the dissemination of collected knowledge (scientific and other) and new viewpoints and to act as a go-between between the policymakers, managers and researchers.

Annexes

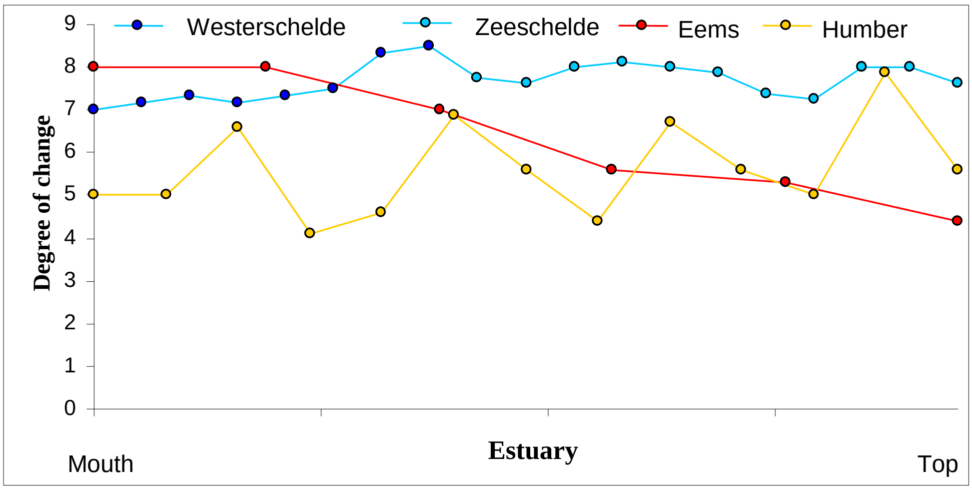

Figure A1: The evolution of the degree of change of EII 1 from the top to the mouth of the Schelde, Eems and Humber estuaries (from Elliott *et al.*, 2008c).

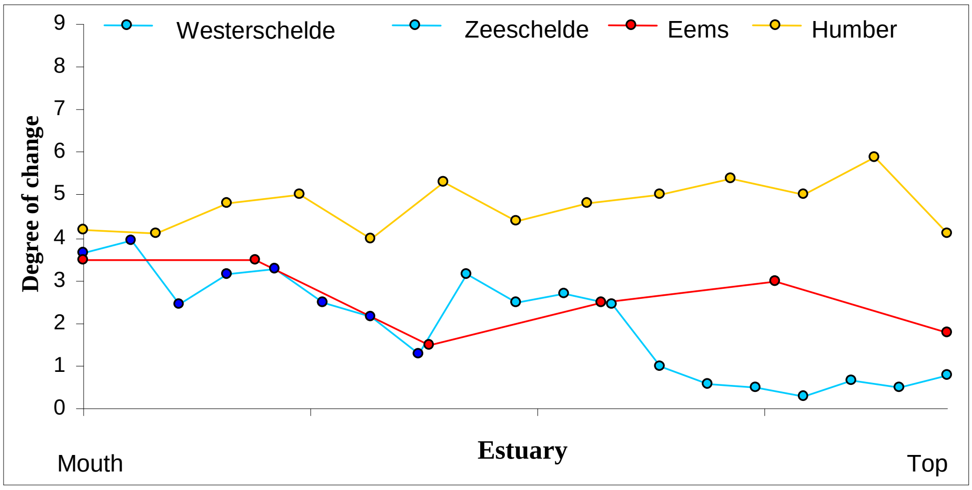

Figure A2: The evolution of the degree of change of EII 2 from the top to the mouth of the Schelde, Eems and Humber estuaries (from Elliott *et al.*, 2008c).

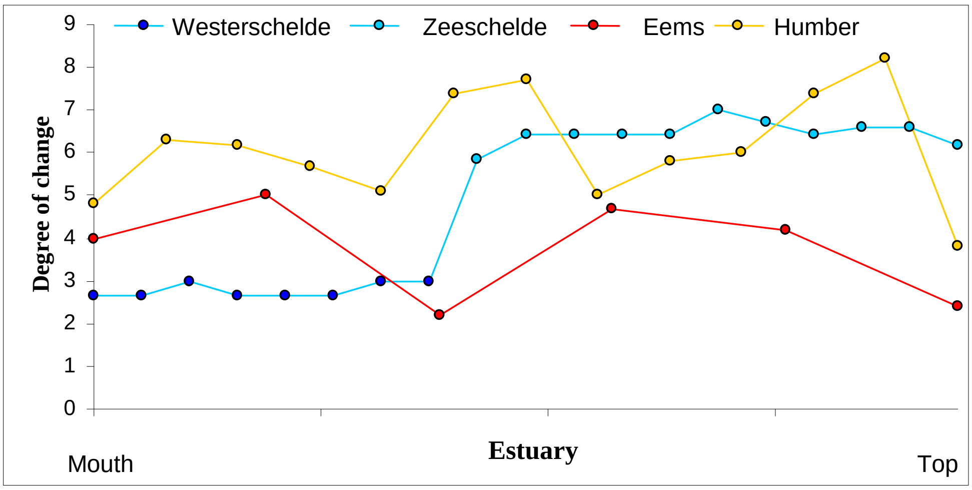

Figure A3: The evolution of the degree of change of EII 3 from the top to the mouth of the Schelde, Eems and Humber estuaries (from Elliott *et al.*, 2008c).

Table A: Catch frequency for each fish species, expressed as percentage, for the different salinity zones of the Zeeschelde between 1995 and 2008. The estuarine use guild is given between brackets (Franco *et al.*, 2008).

Abb: abbreviation; M: Mesohaline zone, O: Oligohaline zone and F: freshwater zone, with the total number of monthly catches for each zone between brackets.

Scientific name	**Abb**	**Guild**	**Common name**	**M (90)**	**O (52)**	**F (49)**
Abramis brama (Linnaeus, 1758)	A.bra.	Fw	Bream	46.7	71.2	63.3
Acipenser baeri (Brandt, 1869)	A.bae.	Di	Siberian sturgeon	6.7	0	0
Agonus cataphractus (Linnaeus, 1758)	A.cat.	Es	Hook-nose	1.1	0	0
Alburnus alburnus (Linnaeus, 1758)	A.alb.	Fw	Bleak	2.2	5.8	16.3
Alosa fallax (Lacepède, 1803)	A.fal.	Di	Twaite shad	44.4	9.6	2
Ammodytes tobianus (Linnaeus, 1758)	A.tob.	Ms	Sand-eel	3.3	0	0
Anguilla anguilla (Linnaeus, 1758)	A.ang.	Di	European eel	85.6	88.5	85.7
Aphia minuta (Linnaeus, 1758)	A.min.	Es	Transparent goby	1.1	0	0
Atherina presbyter (Risso, 1810)	A.pre.	Mm	Sand smelt	21.1	1.9	0
Blicca bjoerkna (Linnaeus, 1758)	B.bjo.	Fw	White bream	35.6	69.2	79.6
Carassius carassius (Linnaeus, 1758)	C.carr.	Fw	Crucian carp	0	9.6	2
Carrasius gibelio (Bloch, 1782)	C.gib.	Fw	Prussian carp	38.9	96.2	81.6
Chelidonichthys lucernus (Linnaeus, 1758)	C.luc.	Mm	Tub gurnard	18.9	1.9	0
Chelon labrosus (Risso, 1827)	C.lab.	Mm	Thick-lipped mullet	0	1.9	0
Ciliata mustela (Linnaeus, 1758)	C.mus.	Mm	Fivebeard rockling	15.6	0	0
Clupea harengus (Linnaeus, 1758)	C.har.	Mm	Herring	88.9	50	2
Cottus gobio (Linnaeus, 1758)	C.gob.	Fw	Bullhead	1.1	3.9	4.1
Cyclopterus lumpus (Linnaeus, 1758)	C.lum.	Mm	Lumpsucker	1.1	0	0
Cyprinus carpio (Linnaeus, 1758)	C.car.	Fw	Carp	18.9	61.5	73.5
Dicentrarchus labrax (Linnaeus, 1758)	D.lab.	Mm	Seabass	84.4	25	2
Echiichthys vipera (Cuvier, 1829)	E.vip.	Ms	Lesser weever	2.2	3.9	0
Engraulis encrasicolus (Linnaeus, 1758)	E.enc.	Mm	Anchovy	1.1	0	0
Esox lucius (Linnaeus, 1758)	E.luc.	Fw	Pike	11.1	11.5	14.3
Gadus morhua (Linnaeus, 1758)	G.mor.	Mm	Cod	25.6	1.9	0
Gasterosteus aculeatus (Linnaeus, 1758)	G.acu.	Fw	Three-spined stickleback	46.7	61.5	73.5
Gobio gobio (Linnaeus, 1758)	G.gob.	Fw	Gudgeon	0	7.7	0
Gymnocephalus cernuus (Linnaeus, 1758)	G.cer.	Fw	Ruffe	35.6	46.2	59.2
Lampetra fluviatilis (Linnaeus, 1758)	L.flu.	Di	River lamprey	5.6	11.5	16.3
Lepomis gibbosus (Linnaeus, 1758)	L.gib.	Fw	Pumpkinseed	6.7	26.9	28.6
Leucaspius delineatus (Heckel, 1843)	L.del.	Fw	Belica	0	7.7	6.1
Leuciscus cephalus (Linnaeus, 1758)	L.cep.	Fw	Chub	0	0	2
Leuciscus idus (Linnaeus, 1758)	L.ide.	Fw	Ide	7.8	32.7	22.5
Limanda limanda (Linnaeus, 1758)	L.lim.	Mm	Dab	6.7	0	0
Liparis liparis (Linnaeus, 1760)	L.lip.	Es	Sea snail	2.2	0	0
Liza ramado (Risso, 1827)	L.ram.	Di	Thinlip mullet	42.2	26.9	8.2
Merlangius merlangus (Linnaeus, 1758)	M.mer.	Mm	Whiting	20	1.9	0
Mullus surmuletus (Linnaeus, 1758)	M.sur.	Ms	Red mullet	2.2	0	0
Myoxocephalus scorpius (Linnaeus, 1758)	M.sco.	Es	Bull rout	11.1	0	0
Oncorhynchus mykiss (Walbaum, 1792)	O.myk.	Fw	Rainbow trout	1.1	0	0
Osmerus eperlanus (Linnaeus, 1758)	O.epe.	Di	Smelt	70	32.7	8.2

Table A: Continued.

Scientific name	**Abb**	**Guild**	**Common name**	**M (90)**	**O (52)**	**F (49)**
Perca fluviatilis (Linnaeus, 1758)	P.flu.	Fw	Perch	61.1	86.5	71.4
Platichthys flesus (Linnaeus, 1758)	P.fle.	Di	Flounder	98.9	61.5	61.2
Pleuronectes platessa (Linnaeus, 1758)	P.pla.	Mm	Plaice	36.7	0	0
Pomatoschistus lozanoi (de Buen, 1923)	P.loz.	Ms	Lozano's goby	1.1	1.9	0
Pomatoschistus microps (Krøyer, 1838)	P.mic.	Es	Common goby	30	48.1	20.4
Pomatoschistus minutus (Pallas, 1770)	P.min.	Es	Sand goby	54.4	40.4	12.2
Pomatoschistus sp.	P.spe.	Es	Gobiidae sp.	6.7	0	0
Psetta maxima (Linnaeus, 1758)	P.max.	Mm	Turbot	1.1	0	0
Pseudorasbora parva (Temminck & Schlegel, 1842)	P.par.	Fw	Stone moroko	13.3	59.6	77.6
Pungitius pungitius (Linnaeus, 1758)	P.pun.	Fw	Nine spine stickleback	12.2	34.6	20.4
Rhodeus sericeus (Bloch, 1782)	R.ser.	Fw	Bitterling	13.3	44.2	51
Rutilus rutilus (Linnaeus, 1758)	R.rut.	Fw	Roach	74.4	100	93.9
Salmo salar (Linnaeus, 1758)	S.sal.	Di	Salmon	2.2	0	0
Salmo trutta (Linnaeus, 1758)	S.tru.	Di	Sea trout	8.9	0	0
Sander lucioperca (Linnaeus, 1758)	S.luc.	Fw	Pike-perch	77.8	57.7	61.2
Scardinius erythrophthalmus (Linnaeus, 1758)	S.ery.	Fw	Rudd	36.7	63.5	81.6
Scophthalmus rhombus (Linnaeus, 1758)	S.rho.	Mm	Brill	7.8	0	0
Silurus glanis (Linnaeus, 1758)	S.gla.	Fw	Wels catfish	0	13.5	12.2
Solea solea (Linnaeus, 1758)	S.sol.	Mm	Sole	84.4	13.5	0
Sprattus sprattus (Linnaeus, 1758)	S.spr.	Mm	Sprat	6.7	0	0
Syngnathus acus (Linnaeus, 1758)	S.acu.	Es	Greater pipefish	17.8	9.6	0
Syngnathus rostellatus (Nilsson, 1855)	S.ros.	Es	Nilsson's pipefish	1.1	0	0
Tinca tinca (Linnaeus, 1758)	T.tin.	Fw	Tench	6.7	5.8	8.2
Trachurus trachurus (Linnaeus, 1758)	T.tra.	Ms	Scad	6.7	0	0
Trisopterus luscus (Linnaeus, 1758)	T.lus.	Mm	Pouting	34.4	7.7	0
Zoarces viviparus (Linnaeus. 1758)	Z.viv.	Es	Viviparous blenny	6.7	0	0

Di: Diadromous species; Es: Estuarine species; Fw: Freshwater species; Mm: Marine migrants (seasonal or juvenile migrants); Ms: Marine stragglers (adventitious visitors)

Table B: References used to assess the presence of fish species in the Zeeschelde and tidal tributaries, classified by salinity zone

Salinity zone / river	**Literature**
Mesohaline	de Selys-Longchamps, 1842
	Poll, 1945, 1947
	Maes *et al*., 1997
	Van Damme *et al*., 1999
	Breine *et al*., 2001 Maes *et al.*, 2001
	Adriaenssens *et al.*, 2002
	Breine *et al.*, 2007
Oligohaline	Maes *et al*., 1997
	Vrielynck *et al.*, 2003
	Breine & Van Thuyne, 2004, 2005, 2006
	Maes *et al.*, 2005
	Simoens *et al.*, 2006
	Breine *et al.*, 2007, 2007a
Freshwater	Van den Bogaerde, 1825
	Breine *et al.*, 2001
	Vrielynck *et al.*, 2003
	Breine *et al*., 2005, 2006, 2007
	Maes *et al.*, 2005
	Simoens *et al.*, 2006
Nete	Yseboodt & Meire, 1999
	Breine *et al.*, 2001
	Vrielynck *et al*., 2003
	Van Thuyne & Breine, 2003a, 2008
	Van Liefferinge *et al*., 2000, 2005
	Buysse *et al*., 2007
Dijle and Zenne	Breine *et al.*, 2001
	Vrielynck *et al*., 2003
	Van Thuyne & Breine, 2003b, 2008
	Buysse *et al*., 2007

Table C: Historical and recent presence (1) - absence (0) fish data for the Zeeschelde estuary and GEP and MEP lists for the mesohaline, oligohaline, freshwater zones and tidal estuaries. Fishes are grouped according to guilds (Elliott & Hemingway, 2002). For each data source it is indicated whether the study deals with the polyhaline (P), mesohaline (M), oligohaline (O), freshwater (F) zone or (T) tributary of the Schelde. Empty cells means no data available; italics stands for few catches or records (<5sp.); * no longer in Schelde; ° exotic species

Scientific name	**(M–O) de Selys-Longchamps, 1842**	**(M–O) Poll, 1945**	**(M–O) Poll, 1947**	**(M–O–F) Breine *et al.*, 2001**	**(T) Breine *et al.*, 2001**	**(T) Vrielynck *et al.* , 2003**	**(F) Maes *et al.*, 2005**	**(F) River Durme (Breine *et al.*, 2005-2007)**	**(O) River Rupel (Breine *et al.*, 2005-2007)**	**(F) Durme (Simoens *et al.*, 2006)**	**(O) River Rupel (Simoens *et al.*, 2006)**	**(M-0) Simoens *et al.*, 2006**	**(F) Simoens *et al.*, 2006**	**(T) Nete (Simoens *et al.*, 2006)**	**(T) Dijle & Zenne (Simoens *et al.*, 2006)**	**(T) River Nete (Buysse *et al.*, 2007) & PC**	**(T) River Dijle (Buysse *et al.*, 2007) & PC**	**(M-O) Breine *et al.*, 2007a**	**(M) surveys cooling-water Doel 1991-2007**	**(M) fyke nets surveys, 1995-2006**	**(O) fyke nets surveys, 2005-2007**	**(F) fyke nets surveys, 2005-2007**	**MEP Westerschelde**	**GEP Mesohaline zone**	**GEP Oligohaline zone**	**GEP Freshwater zone**	**GEP Tributaries**	**MEP Mesohaline zone**	**MEP Oligohaline zone**	**MEP Freshwater zone**	**MEP Tributaries**	**Guild**
Abramis brama	1	1		1	1	1	1	1	1	0	1	1	1	0	0	1	1	1	1	1	1	1	**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw
Acipenser sturio	1	*1*	*1*	1	1	1	0			0	0	0	0	0	0								**1**	**0**	**0**	**0**	**0**	**1**	**1**	**1**	**1**	Di*
Agonus cataphractus		1	1	1								0	0					1	*1*	1			**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Es
Alburnus alburnus	1	0				1		1	0	1	0	0	0	1	1	*1*	0	1			*1*		**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	**1**	Fw
Alosa alosa	1	*1*	*1*	1	1	1	0			0	0	0	0	0	0			*1*					**1**	**0**	**0**	**0**	**0**	**1**	**1**	**1**	**1**	Di*
Alosa fallax	1	1	1	1	1	1	0			1	1	1	1	1	0			1	1	1	1	1	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	Di
Ammodytes tobianus	1	1	1	1								0	0					1	1		*1*		**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Anguilla anguilla	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	Di
Aphia minuta		1	1	1								1	0					1	1				**0**	**1**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Es
Arnoglossus laterna		1	*1*	1								0	0					1					**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Atherina presbyter		0										0	0					1	1	1	1		**0**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Balistes carolinensis		0	*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Barbatula barbatula	1	0								0	1	0	1	0	0	1						1	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	Fw
Barbus barbus	1	0				*1*				0	1	0	1	1	1								**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Fw
Belone belone	1	*1*	*1*	1								1	1					1					**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Blicca bjoerkna		1				1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw

Table C: Continued.

Scientific name	**(M–O) de Selys-Longchamps, 1842**	**(M–O) Poll, 1945**	**(M–O) Poll, 1947**	**(M–O–F) Breine *et al.*, 2001**	**(T) Breine *et al.*, 2001**	**(T) Vrielynck *et al.* , 2003**	**(F) Maes *et al.*, 2005**	**(F) Durme (Breine *et al.*, 2005-2007)**	**(O) Rupel (Breine *et al.*, 2005-2007)**	**(F) Durme (Simoens *et al.*, 2006)**	**(O) River Rupel (Simoens *et al.*, 2006)**	**(M-0) Simoens *et al.*, 2006**	**(F) Simoens *et al.*, 2006**	**(T) Nete (Simoens *et al.*, 2006)**	**(T) Dijle & Zenne (Simoens *et al.*, 2006)**	**(T) River Nete (Buysse *et al.*, 2007) & PC**	**(T) River Dijle (Buysse *et al.*, 2007) & PC**	**(M-O) Breine *et al.*, 2007a**	**(M) surveys Doel 1991-2007**	**(M) fyke nets surveys, 1995-2006**	**(O) fyke nets surveys, 2005-2007**	**(F) fyke nets surveys, 2005-2007**	**MEP Westerschelde**	**GEP Mesohaline zone**	**GEP Oligohaline zone**	**GEP Freshwater zone**	**GEP Tributaries**	**MEP Mesohaline zone**	**MEP Oligohaline zone**	**MEP Freshwater zone**	**MEP Tributaries**	**Guild**
Callionymus lyra	1	*1*	*1*	1								0	0					1	*1*				**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Carassius carassius		0						*1*	0	0	1	0	1	0	0			1	*1*			1	**0**	**0**	**0**	**1**	**1**	**0**	**0**	**1**	**1**	Fw
Carassius gibelio	1							1	1	0	1	0	0					1	*1*	1	1	1	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Fw°
Chelidonichthys lucernus	1	1	1	1								0	0					1	1	1	*1*		**1**	**1**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Chelon labrosus		0	*1*									0	0								*1*		**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Ciliata mustela		*1*	*1*									1	0					1	1	1	*1*		**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Clupea harengus	1	1	1	1								1	1					1	1	1	1		**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Mm
Cobitis taenia	1	0								1	1	0	1	1	1	1							**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	**1**	Fw
Conger conger	1	*1*	*1*	1								0	0					1					**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Coregonus oxyrhynchus	1	*1*	*1*				0					0	0										**1**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Es*
Cottus gobio	1	*1*								1	0	0	0	1	1	1		1	*1*	1		1	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	**1**	Fw
Crenilabrus melops		0	*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Crystallogobius linearis			*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Ctenolabrus rupestris			*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Cyclopterus lumpus		*1*	*1*	1								0	0					1	1				**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Dasyatis pastinaca			*1*									0	0										**1**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Mm
Dicentrarchus labrax		1	1	1								0	0					1	1	1	1		**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Mm
Echiichthys vipera		1	1	1								0	0					1	1		*1*		**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Engraulis encrasicolus	1	1	1	1								1	1					1	1	1			**1**	**1**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm

Table C: Continued.

Scientific name	**(M–O) de Selys-Longchamps, 1842**	**(M–O) Poll, 1945**	**(M–O) Poll, 1947**	**(M–O–F) Breine *et al.*, 2001**	**(T) Breine *et al.*, 2001**	**(T) Vrielynck *et al.* , 2003**	**(F) Maes *et al.*, 2005**	**(F) Durme (Breine *et al.*, 2005-2007)**	**(O) Rupel (Breine *et al.*, 2005-2007)**	**(F) Durme (Simoens *et al.*, 2006)**	**(O) River Rupel (Simoens *et al.*, 2006)**	**(M-0) Simoens *et al.*, 2006**	**(F) Simoens *et al.*, 2006**	**(T) Nete (Simoens *et al.*, 2006)**	**(T) Dijle & Zenne (Simoens *et al.*, 2006)**	**(T) River Nete (Buysse *et al.*, 2007) & PC**	**(T) River Dijle (Buysse *et al.*, 2007) & PC**	**(M-O) Breine *et al.*, 2007a**	**(M) surveys Doel 1991-2007**	**(M) fyke nets surveys, 1995-2006**	**(O) fyke nets surveys, 2005-2007**	**(F) fyke nets surveys, 2005-2007**	**MEP Westerschelde**	**GEP Mesohaline zone**	**GEP Oligohaline zone**	**GEP Freshwater zone**	**GEP Tributaries**	**MEP Mesohaline zone**	**MEP Oligohaline zone**	**MEP Freshwater zone**	**MEP Tributaries**	**Guild**
Entelurus aequoreus			*1*									0	0						1				**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Esox lucius	1	1		1	1	1	0			1	1	1	1	1	1	1	0	1		1	*1*	1	**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw
Eutrigla gurnardus		*1*										0	0						*1*				**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Mm
Gadus morhua	1	*1*	*1*	1								0	0					1	1	1			**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Gasterosteus aculeatus	1	1	1	1	1	1	1					1	1	0	0	1	1	1	1	1	1	1	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	Fw/Di
Glyptocephalus cynoglossus			*1*									0	0											**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Gobio gobio	1	0				1				1	1	0	1	1	1	1	*1*					1	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	**1**	Fw
Gymnocephalus cernuus	1	0		1	1	1	1	1	1	0	1	1	1	1	0			1	1	1	1	1	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	Fw
Hippocampus guttulatus			*1*	1								0	0					1					**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Es
Hippocampus hippocampus	1	*1*										0	0						1				**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Hippoglossus hippoglossus			*1*									0	0											**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Hyperoplus lanceolatus	1	*1*		1								0	0					1					**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Lampetra fluviatilis	1	1	1	1	1	*1*	1			0	1	1	1	0	0	*1*	0	1	1	1	1		**1**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	**1**	Di
Lampetra planeri	1					1				1	0	0	0	0	1							1	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	Fw
Leucaspius delineatus								1	0	1	0	0	0	0	1			1			*1*	1	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	Fw
Leuciscus cephalus	1	0						*1*	0	0	1	0	1	0	0	1	0					1	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	Fw
Leuciscus idus		0				1	1	1	1	0	1	0	1	0	0	*1*	0	1	1	1	1		**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw
Leuciscus leuciscus	1	0								1	0	0	0	1	1	1			*1*			1	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	Fw
Limanda limanda		*1*	0	1								0	0					1	1	1			**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm

Table C: Continued.

Scientific name	**(M–O) de Selys-Longchamps, 1842**	**(M–O) Poll, 1945**	**(M–O) Poll, 1947**	**(M–O–F) Breine *et al.*, 2001**	**(T) Breine *et al.*, 2001**	**(T) Vrielynck *et al.* , 2003**	**(F) Maes *et al.*, 2005**	**(F) Durme (Breine *et al.*, 2005-2007)**	**(O) Rupel (Breine *et al.*, 2005-2007)**	**(F) Durme (Simoens *et al.*, 2006)**	**(O) River Rupel (Simoens *et al.*, 2006)**	**(M-0) Simoens *et al.*, 2006**	**(F) Simoens *et al.*, 2006**	**(T) Nete (Simoens *et al.*, 2006)**	**(T) Dijle & Zenne (Simoens *et al.*, 2006)**	**(T) River Nete (Buysse *et al.*, 2007) & PC**	**(T) River Dijle (Buysse *et al.*, 2007) & PC**	**(M-O) Breine *et al.*, 2007a**	**(M) surveys Doel 1991-2007**	**(M) fyke nets surveys, 1995-2006**	**(O) fyke nets surveys, 2005-2007**	**(F) fyke nets surveys, 2005-2007**	**MEP Westerschelde**	**GEP Mesohaline zone**	**GEP Oligohaline zone**	**GEP Freshwater zone**	**GEP Tributaries**	**MEP Mesohaline zone**	**MEP Oligohaline zone**	**MEP Freshwater zone**	**MEP Tributaries**	**Guild**
Liparis liparis		1	1	1								0	0					1	1	1			**1**	**1**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Es
Liza ramado		1	1	1	1		1			0	1	1	1	0	0			1	1	1	1	1	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	Di
Lota lota	1	0				*1*				0	0	0	0	0	0									**0**	**0**	**0**	**0**	**1**	**1**	**1**	**1**	Fw
Melanogrammus aeglefinus	1	*1*	*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Merlangius merlangus	1	*1*	1	1								0	0					1	1	1	*1*		**1**	**1**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Misgurnus fossilis	1	*1*		1	1	1				1	1	1	1	1	1	*1*		1					**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw
Mullus surmuletus			0									0	0					1	1	1			**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Myoxocephalus scorpius		1	1	1								0	0					1	1	1			**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Es
Nerophis ophidion			*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Es
Osmerus eperlanus	1	1	1	1	1	1	1	1	0	0	1	1	1	1	0	0	0	1	1	1	1	1	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	Di
Perca fluviatilis	1	1		1	1	1	1	1	1	0	1	1	1	0	0	1	1	1	1	1	1	1	**0**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	Fw
Petromyzon marinus	1	*1*	*1*	1	1		0			0	1	1	1	0	0			1					**1**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	**1**	Di
Pholis gunnellus		1	1									0	0										**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Es
Phoxinus phoxinus		0				1				1	0	0	0	1	1								**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	Fw
Platichthys flesus	1	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1		1	1	1	1	1	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	Di
Pleuronectes platessa	1	1	1	1								1	1					1	1	1			**1**	**1**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Pollachius pollachius			*1*									0	0						*1*				**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Mm
Pomatoschistus lozanoi				1								0	0					1	1	1			**0**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Pomatoschistus microps		1	1	1				1	1	1	1	1	1					1	1	1	1	1	**1**	**1**	**1**	**0**	**0**	**1**	**1**	**1**	**0**	Es

Table C: Continued.

Scientific name	**(M–O) de Selys-Longchamps, 1842**	**(M–O) Poll, 1945**	**(M–O) Poll, 1947**	**(M–O–F) Breine *et al.*, 2001**	**(T) Breine *et al.*, 2001**	**(T) Vrielynck *et al.* , 2003**	**(F) Maes *et al.*, 2005**	**(F) Durme (Breine *et al.*, 2005-2007)**	**(O) upel (Breine *et al.*, 2005-2007)**	**(F) Durme (Simoens *et al.*, 2006)**	**(O) River Rupel (Simoens *et al.*, 2006)**	**(M-0) Simoens *et al.*, 2006**	**(F) Simoens *et al.*, 2006**	**(T) Nete (Simoens *et al.*, 2006)**	**(T) Dijle & Zenne (Simoens *et al.*, 2006)**	**(T) River Nete (Buysse *et al.*, 2007) & PC**	**(T) River Dijle (Buysse *et al.*, 2007) & PC**	**(M-O) Breine *et al.*, 2007a**	**(M) surveys Doel 1991-2007**	**(M) fyke nets surveys, 1995-2006**	**(O) fyke nets surveys, 2005-2007**	**(F) fyke nets surveys, 2005-2007**	**MEP Westerschelde**	**GEP Mesohaline zone**	**GEP Oligohaline zone**	**GEP Freshwater zone**	**GEP Tributaries**	**MEP Mesohaline zone**	**MEP Oligohaline zone**	**MEP Freshwater zone**	**MEP Tributaries**	**Guild**
Pomatoschistus minutus	1	1	1	1								1	1					1	1	1	1	*1*	**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Es
Psetta maxima	1	*1*	1	1								1	1					1					**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Mm
Pungitius pungitius	1	*1*		1	1	1	1	0	1	0	1	1	1	1	0	*1*		1	1	1	1	1	**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw
Raja clavata	1	*1*	1	1								0	0					1					**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Raniceps raninus	1											0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Es
Rhinonemus cimbrius		0	*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Rhodeus sericeus		0						1	1	0	1	0	1	0	0	1	*1*	1	1	1	1	1	**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw
Rutilus rutilus	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	**0**	**1**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw
Salmo salar	1	*1*	*1*	1	1	1	0			0	0	0	0	0	0			1		1			**1**	**0**	**0**	**0**	**0**	**1**	**1**	**1**	**1**	Di*
Salmo trutta		*1*	*1*	1	1	1	0			0	0	0	0	0	0			1		1	1		**1**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	**1**	Di
Sander lucioperca		*1*		1			1					0	0					1	1	1	1	1	**0**	**1**	**1**	**1**	**0**	**0**	**0**	**0**	**0**	Fw°
Sardina pilchardus		0	*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Mm
Scardinius erythrophthalmus	1	1		1	1	1		1	1	1	1	1	1	1	1	*1*	*1*	1	1		1	1	**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw
Scomber scombrus		*1*	*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Scomberesox saurus		*1*	*1*									0	0					1					**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Scophthalmus rhombus	1	*1*	*1*	1								1	0					1		1			**1**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	Ms
Scyliorhinus canicula	1	*1*	*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Scyliorhinus stellaris		*1*	*1*									0	0											**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Silurus glanis							0	0	1	1	0	0	0	1	0	1	*1*					1	**0**	**0**	**1**	**1**	**1**	**0**	**1**	**1**	**1**	Fw

Table C: Continued.

Scientific name	**(M–O) de Selys-Longchamps, 1842**	**(M–O) Poll, 1945**	**(M–O) Poll, 1947**	**(M–O–F) Breine *et al.*, 2001**	**(T) Breine *et al.*, 2001**	**(T) Vrielynck *et al.* , 2003**	**(F) Maes *et al.*, 2005**	**(F) River Durme (Breine *et al.*, 2005-2007)**	**(O) River Rupel (Breine *et al.*, 2005-2007)**	**(F) Durme (Simoens *et al.*, 2006)**	**(O) River Rupel (Simoens *et al.*, 2006)**	**(M-0) Simoens *et al.*, 2006**	**(F) Simoens *et al.*, 2006**	**(T) Nete (Simoens *et al.*, 2006)**	**(T) Dijle & Zenne (Simoens *et al.*, 2006)**	**(T) River Nete (Buysse *et al.*, 2007) & PC**	**(T) River Dijle (Buysse *et al.*, 2007) & PC**	**(M-O) Breine *et al.*, 2007a**	**(M) surveys cooling-water Doel 1991-2007**	**(M) fyke nets surveys, 1995-2006**	**(O) fyke nets surveys, 2005-2007**	**(F) fyke nets surveys, 2005-2007**	**MEP Westerschelde**	**GEP Mesohaline zone**	**GEP Oligohaline zone**	**GEP Freshwater zone**	**GEP Tributaries**	**MEP Mesohaline zone**	**MEP Oligohaline zone**	**MEP Freshwater zone**	**MEP Tributaries**	**Guild**
Solea solea	1	1	1	1								1	1					1	1	1	1		**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Mm
Spinachia spinachia		0	0									0	0						*1*				**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Es
Sprattus sprattus	1	1	1	1								1	1					1	1	1			**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Mm
Syngnathus acus	1	1	1	1								0	0					1	1	1	1		**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Es
Syngnathus rostellatus	1	1	1	1								0	0					1	1				**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Es
Tinca tinca	1	0				1				1	1	0	1	1	1	*1*	1		*1*		*1*		**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**	**1**	Fw
Trachinus draco	1	*1*	1									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Trachurus trachurus		*1*	1	1								0	0						1	1	1		**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Trigloporus lastoviza			*1*									0	0											**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Trisopterus luscus	1	1	1	1								0	0					1	1	1	1		**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Mm
Trisopterus minutus			*1*									0	0										**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Xiphias gladius		*1*										0	0											**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	Ms
Zoarces viviparus	1	1	1	1								1	0					1	1	1	1		**1**	**1**	**1**	**0**	**0**	**1**	**1**	**0**	**0**	Es

Di: Diadromous species; Es: Estuarine species; Fw: Freshwater species; Ms: Marine stragglers (adventitious visitors); Mm: Marine migrants (seasonal or juvenile migrant)

Table D: List of fish species recorded in the Zeeschelde estuary including tidal tributaries (1991-2008) and their assignment to guilds (functional groups). Species in bold belong to the reference Maximal Ecological Potential (MEP) or Good Ecological Potential (GEP) list (Chapter 4). The sign ° indicates that the reference species was not caught in the different campaigns. **EUFG**: Estuarine use functional group; Di diadromous species, Es estuarine species, Fw freshwater species, Mm Marine migrants, Ms marine stragglers; **FMFG**: Feeding mode functional group (J) juvenile or (A) adult: B benthivores, BF benthivores piscivores, BZ benthivores zooplanktivores, De detrivores, F piscivores, H herbivores, VF vertivores piscivores, P plankivores; **RMFG**: Reproductive mode functional group; Ob oviparous with benthic eggs, Og oviparous gardeners, Op oviparous with pelagic eggs, Os oviparous shelterers, Ov oviparous with adhesive eggs, V viviparous. * exotic species.

Scientific name	Common name	EUFG	FMFG (J)	FMFG (A)	RMFG
Abramis brama	Bream	Fw	P	B	Ov
Acipenser baeri	Siberian sturgeon	Di*	B	B	Ob
Acipenser sturio	Atlantic sturgeon	Di	B	BF	Ov
Agonus cataphractus	Hook-nose	ES	B	B	Ov
Alburnus alburnus	Bleak	Fw	P	O	Ov
Alosa alosa	Allis shad	Di	P	BF	Ob
Alosa fallax	Twaite shad	Di	P	BF	Ob
Ameiurus nebulosis	Brown bullhead	Fw*	B	BF	Og
Ammodytes tobianus	Sand-eel	Es/Ms	P	P	Ob
Anguilla anguilla	Eel	Di	O	O	Op
Aphia minuta	Transparent goby	Es	P	P	Os
Atherina presbyter	Sand smelt	Mm	P	P/B	Ov
Barbatula barbatula	Stone loach	Fw		B	Og
Blicca bjoerkna	White bream	Fw	P	O	Ob
Buglossidium luteum	Solonette	Ms		B	Op
Callionymus lyra	Dragonet	Ms		B	Op
Carassius carassius	Crucian carp	Fw	BZ	O	Ov
Carassius gibelio	Gibel carp	Fw*		O	Ob
Chelidonichthys lucernus	Tub gurnard	Mm	BZ	BF	Op
Chelon labrosus	Thick-lipped mullet	Mm		De	Op
Ciliata mustela	Fivebeard rockling	Mm	BZ	B	Op
Clupea harengus	Herring	Mm	P	P	Ov
Cobitis taenia	Spined loach	Fw	B	B	Ov
Cottus gobio	Bullhead	Fw		B	Og
***Coregonus oxyrhynchus* °**	Houting	Es	BZ		Ob
Ctenopharyngodon idella	Grass carp	Fw*		H	
Cyclopterus lumpus	Lumpsucker	Mm		BZ	Og
Cyprinus carpio	Carp	Fw*		O	Ov
Dicentrarchus labrax	Seabass	Mm	BZ	BZ/BF	Op
Echiichthys vipera	Lesser weever	Ms		BF	Op
Engraulis encrasicolus	Anchovy	Mm	BF	P	Op
Entelurus aequoreus	Snake pipefish	Ms		B	Os
Esox lucius	Pike	Fw	BZ	VF	Ov
Eutrigla gurnardus	Grey gurnard	Mm		B	Op
Gadus morhua	Cod	Mm	BZ	BZ/BF/O	Op

Table D: Continued.

Scientific name	Common name	EUFG	FMFG (J)	FMFG (A)	RMFG
Gasterosteus aculeatus	Three-spined stickleback	Fw/Di	BZ	BZ	Og
Gobius niger	Black goby	Es		B/BF	Og
Gobo gobio	Gudgeon	Fw	B	B	Ov
Gymnocephalus cernuus	Ruffe	Fw	BZ	B	Ov
Hippocampus guttulatus	long-snouted seahorse	Es		B/BF	Os
Hyperoplus lanceolatus	Great sandeel	Ms		BF	Ob
Lampetra fluviatilis	River lamprey	Di	B	F	Ob
Lampetra planeri °	Brook lamprey	Fw	P		Ob
Lepomis gibbosus	Pumpkinseed	Fw*		B	Og
Leucaspius delinatus	Belica	Fw		B	Og
Leuciscus cephalus	Chub	Fw	BZ	O	Ov
Leuciscus idus	Ide	Fw	BZ	BF	Ov
Leuciscus leuciscus	Common dace	Fw	B	B	Ob
Limanda limanda	Dab	Mm	B	B/BF	Op
Liparis liparis	Sea snail	Es	B	B	Ov
Liza ramado	Thinlip mullet	Di	P/De	De/O	Op
Lota lota	Burbot	Fw	B	F	Ob
Merlangius merlangus	Whiting	Mm	B	BF	Ob
Misgurnus fossilis	Weatherfish	Fw	B	B	Ov
Mullus surmuletus	(Striped) red mullet	Ms		B	Op
Myoxocephalus scorpius	Bull rout	Es	B	BF	Og
Oncorhynchus mykiss	Rainbow trout	Fw*		O	Ob
Osmerus eperlanus	Smelt	Di	B	BF	Ob
Perca fluviatilis	Perch	Fw	B	BF	Ov
Petromyzon marinus	Sea lamprey	Di	B	F	Ob
Pholis gunnellus °	Rock gunnel	Es	B	B	Og
Phoxinus phoxinus °	Minnow	Fw	B	B	Ob
Platichthys flesus	Flounder	Mm/Di	BZ	BF	Op
Pleuronectes platessa	Plaice	Mm	B	B	Op
Pomatoschistus lozanoi	Lozano's goby	Mm/Ms	BZ	B/BZ	Og
Pomatoschistus microps	Common goby	Es	BZ	B	Og
Pomatoschistus minutus	Sand goby	Es	BZ	B	Og
Pomatoschistus pictus	Painted goby	Ms		B	Ob
Pomatoschistus sp.	Gobidae	Es	B	B	Og
Psetta maxima	Turbot	Mm		BF	Op
Pseudorasbora parva	Stone moroko	Fw*		B	Ob
Pungitius pungitius	Nine spine stickleback	Fw	BZ	B	Og
Raniceps raninus	Tadpole fish	Es		B/BF	Op
Rhodeus sericeus	Bitterling	Fw	P	BZ	
Rutilus rutilus	Roach	Fw	O	O	Ov
Salmo salar	Salmon	Di*		F	Ob
Salmo trutta	Sea trout	Di	B	BF	Ob
Sander lucioperca	Pike-perch	Fw*	BZ	BF	Og
Scardinius erythrophthalmus	Rudd	Fw	O	O	Ov

Table D: Continued.

Scientific name	Common name	EUFG	FMFG (J)	FMFG (A)	RMFG
Scomber scombrus	Atlantic mackerel	Ms		BF	Op
Scophthalmus rhombus	Brill	Mm/Ms	BZ	BF	Ob
Siluris glanis	Wels catfish	Fw	BZ	VF	Og
Solea solea	Sole	Mm	BZ	B	Op
Spinachia spinachia	Sea stickleback	Es		BZ	Og
Sprattus sprattus	Sprat	Mm	P	P	Op
Syngnathus acus	Greater pipefish	Es	BZ	B/BF	Os
Syngnathus rostellatus	Nilsson's pipefish	Es	P	BZ	Os
Tinca tinca	Tench	Fw	P	B	Ov
Trachurus trachurus	Atlantic horse mackerel	Ms		BF	Op
Trisopterus luscus	Pouting	Mm	B	B/BF	Op
Zoarces viviparus	Viviparous blenny	Es	BZ	B	V

Table E: Fish species collected in the Zeeschelde and tributaries during the surveys with fyke nets performed between 1995 and 2008. Only species from the reference list of the good ecological potential status (GEP, Breine *et al*., 2008) are withheld. **GEP** (good ecological potential): M: mesohaline zone, O: oligohaline zone, F: freshwater zone; **EUFG** (estuarine use functional group): Fw: freshwater species, Di: diadromous species, Es: estuarine species, Mm: marine migrant species; **FP**: flow preference: A: rheophilic (a), B: rheophilic (b), E: eurytopic species; **FMFG** (feeding mode functional group): P: planktonic feeders, B: benthivores, BF: feeds on benthic invertebrates and fish, F: piscivores; BZ: feeds on invertebrates and zooplankton, O: omnivorous, D: detrivores and VF: vertivores and piscivores; **RMFG** (reproductive mode functional group): V: viviparous, Op: oviparous with pelagic eggs, Ob: oviparous with benthic eggs, Ov: oviparous with adhesive eggs, Og: oviparous guarders, Os: oviparous shelterers; **RSD** (reproduction special demands, including guarders and shelterers): M in mesohaline zone, O: in oligohaline zone, F in freshwater zone; **S** (stratum adults): Pe: pelagic species, De: demersal species, Be: benthic species; **HS** (Habitat sensitivity**)**: FS: fragmentation sensitive; HS: habitat structure sensitive; **PS** (Pollution sensitive): T: tolerant species, I: intolerant species. Open cell indicates that no information was found or not relevant.

Scientific name	**GEP**	**EUFG**	**FP**	**FMFG**	**RMFG**	**RSD**	**S**	**HS**	**PS**
Abramis brama	O/F	Fw	E	B	Ov		De		T
Alosa fallax	M/O/F	Di	A	BF	Ob	F	Pe	FS	I
Anguilla anguilla	M/O/F	Di	E	O	Op		Be	FS	T
Blicca bjoerkna	O/F	Fw	E	O	Ob		De		T
Carassius carassius	F	Fw		O	Ov		Pe	HS	T
Chelidonichthys lucernus	M	Mm		BF	Op		De	HS	
Ciliata mustela	M	Mm		B	Op		Be	HS	T
Clupea harengus	M/O	Mm		P	Ov		Pe		T
Dicentrarchus labrax	M/O	Mm		BZ/BF	Op		De		T
Esox lucius	O/F	Fw	E	VF	Ov	F	De	HS/FS	I
Gadus morhua	M	Mm		BZ/BF/O	Op		De		I
Gasterosteus aculeatus	M/O/F	Fw/Di	E	BZ	Og	F	Pe	HS/FS	T
Gymnocephalus cernuus	M/O/F	Fw	E	B	Ov		Be	HS	T
Lampetra fluviatilis	M/O/F	Di	A	F	Ob	F	Be	HS/FS	I
Leuciscus idus	O/F	Fw	B	BF	Ov		Pe	FS	I
Liparis liparis	M	Es		B	Ov		Be	HS	I
Liza ramado	M/O/F	Di	B	D/O	Op		Pe	FS	I
Merlangius merlangus	M	Mm		BF	Ob		De	HS	T
Misgurnus fossilis	F	Fw		B	Ov		Be	HS	T
Myoxocephalus scorpius	M/O	Es		BF	Og		Be		T
Osmerus eperlanus	M/O/F	Di	B	BF	Ob	F	Pe	FS	I
Perca fluviatilis	M/O/F	Fw	E	BF	Ov		Pe		T
Petromyzon marinus	M/O/F	Di	A	F	Ob	M/O/F	De	HS/FS	I
Platichthys flesus	M/O/F	Di	E	BF	Op		Be	HS/FS	T
Pleuronectes platessa	M	Mm		B	Op		Be	HS	
Pomatoschistus microps	M/O	Es	B	B	Og	M/O	Be	HS	
Pomatoschistus minutus	M/O	Es	B	B	Og	M/O	Be	HS	
Pungitius pungitius	O/F	Fw	E	B	Og	F	De	HS/FS	T
Rhodeus sericeus	O/F	Fw		BZ			Be	HS/FS	T

Table E: Continued.

Scientific name	**GEP**	**EUFG**	**FP**	**FMFG**	**RMFG**	**RSD**	**S**	**HS**	**PS**
Rutilus rutilus	M/O/F	Fw	E	O	Ov		Pe		T
Salmo trutta	M/O/F	Di	A	BF	Ob		Pe	FS	I
Sander lucioperca	M/O/F	Fw	E	BF	Og		De		T
Scardinius erythrophthalmus	O/F	Fw		O	Ov		Pe	HS	T
Silurus glanis	O/F	Fw	E	VF	Og	F	Be	FS	T
Solea solea	M	Mm		B	Op		Be	HS	I
Sprattus sprattus	M/O	Mm		P	Op		Pe		
Syngnathus acus	M/O	Es		B/BF	Os	M	Be	HS	I
Syngnathus rostellatus	M/O	Es		BZ	Os	M	Be	HS	I
Trisopterus luscus	M/O	Mm		B/BF	Op		De		
Zoarces viviparus	M/O	Es		B	V	M	Be	HS	

Table F: Selected metrics for the freshwater[1] oligohaline[2] and mesohaline[3] zone in the Zeeschelde, their assessed function and main impacting anthropogenic stressors.

Metric	Abbreviations	Assesses	Main anthropogenic stressors
Total number of species[1,3]	MnsTot	diversity & richness	pollution (decrease in DO), fishery, habitat loss (quality & diversity), chemical pollution;
Total number of individuals[1,2]	MnsInd	abundance & condition	water quality (DO), fishery, habitat loss (quality & diversity), chemical pollution;
Diadromous fish (n & %)[1,2,3]	MpiDia; MnsDia	connectivity	chemical and physical barriers, fishery;
Specialised spawners (n & %)[1,3]	MpiSpa; MnsSpa	nursery function	pollution (decrease in DO), habitat loss, artificial structures, dredging; ship transport, canalisation, barriers; fragmentation;
Piscivores (n & %)[1,2]	MpiPis	feeding function & trophic structure	pollution (decrease in DO), fishery, habitat loss (quality & diversity);
Benthivores (%)[1]	MpiBen	trophic structure& substrate quality	pollution (decrease in DO), dredging, power station, physical inputs, hydrological changes;
Intolerant fish (n & %)[1,2]	MnsInt; MpiInt	water quality	pollution (decrease in DO, eutrophication),chemical pollution, untreated wastewater; land use
Marine migrants (n)[2,3]	MnsMms	feeding & nursery & shelter function	change in temperature, fishery activities, hydrological changes;
Estuarine species (n)[2]	MnsErs	spawning & nursery function; habitat quality	chemical and hormonal pollution, habitat loss, discharge manipulation (water abstraction), hydrological changes;
Habitat sensitive (n)[3]	MnsHab	natural state of habitat	habitat modifications, dredging, constructions, ship transport, canalisation, hydrological changes;

(n: number of species)

Table G1: Fish-based assessments tools for estuaries in European countries of the North-East Atlantic region

Country	Tool	Quality status boundaries High/Good	Quality status boundaries Good/Moderate
Belgium	WFD Fish Classification method	0.8	0.6
Germany	WFD Fish Classification method	0.9	0.7
The Netherlands	Species composition assessment method	0.9	0.7
France	In development		
Spain (Basque Region)	Multimetric fish and Crustacea classification scheme	0.83	0.62
Portugal	WFD Fish Classification method	0.81	0.6
Ireland & UK	WFD Fish Classification method	0.8	0.6

Table G2: Estuarine fish-index: The Netherlands for the Westerschelde and Eems-Dollard (euhaline and polyhaline) (Jager & Kranenbarg, 2004)

Metrics	Referen ce value	Fish gear	Survey frequency/ year	Assesses
Number of diadrome species	9-10	beam trawling	October	connectivity
Number of estuarine species	12-13			spawning & nursery function; habitat quality
Number of nursery species (marine juveniles)	9-10			feeding & nursery function
Number of seasonal species	5			feeding & nursery function
Abundance catadrome species	?			connectivity
Abundance estuarine species	?			spawning & nursery function; habitat quality
Abundance marine juveniles	?			feeding & nursery function
%skin ulcers on flounder	0%			condition
1-OH pyreen botgal (ng/ml)	<50			condition

Table G3: Portugal (adapted from Borja *et al.*, 2004)

Metrics	**Reference value**	**Fish gear**	**Survey frequency/ year**	**Assesses**
Species Richness (% of max. observed)	>75			diversity & richness
Pollution Indicator Species	Absence			water quality
Introduced Species	Absence			exotic species
Fishes Health (% body affected)	<5			condition
% Flat Fish (number individuals)	10-60	trawling	every 3 years	connectivity
% Omnivorous species (number individuals)	2.5-20			feeding function & trophic structure
% Piscivorous species (number individuals)	10-50			feeding function & trophic structure
Estuarine Resident Species (number)	>5			spawning & nursery function; habitat
Resident Species (%)	10-40			diversity & richness

Table G4: Spain (Basque region) (Borja *et al*., 2004)

Metrics	**Reference value**	**Fish gear**	**Survey frequency/ year**	**Assesses**
Richness	>9			diversity & richness
Introduced individuals (Fish & crustacea)	<30%			water quality
Fish health (damage, diseases...) (% affection)	<5			exotic species
Flat fish presence (%)	10-60	trawling	every 3 years	condition
Trophic composition (% omnivorous)	2.5-20			connectivity
Trophic composition (% piscivorous)	10-50			feeding function & trophic structure
Estuarine resident species number	>5			feeding function & trophic structure
Resident species (%)	10-40			diversity & richness

Table G5: Multi-method UK and Ireland (Coates *et al*., 2007)

Metrics	**Reference value**	**Fish gear**	**Survey frequency/ year**	**Assesses**
Species composition	80-100%	multi-method otter trawl fyke nets seine netting beam trawl	bi-annual or only autumn	diversity & richness
Species relative ‘abundance	80-100%			abundance & condition
Presence of indicator species	all present			connectivity
Number of taxa that make up 90% of the abundance	80-100%			richness & diversity
Number of estuarine resident taxa	80-100%			spawning & nursery function; habitat
Number of estuarine-dependent marine taxa	80-100%			spawning & nursery function; habitat
Functional guild composition	5 guilds			connectivity
Number of benthic invertebrate feeding taxa	80-100%			trophic structure & substrate quality
Number of piscivorous taxa	80-100%			feeding function & trophic structure
Feeding guild composition	4 guilds			feeding function & trophic structure

Table G6: Germany (Ems-Dollard) (Biosconsult, 2007)

Metrics	**Reference value**	**Fish gear**	**Survey frequency /year**	**Assesses**
Freshwater species	1	anchor nets	autumn	diversity & richness
Diadromous species	12			connectivity
Estuarine resident species	14			spawning & nursery function; habitat quality
Marine juvenile (nursery species)	11			spawning & nursery function; habitat quality
Marine seasonal visitors	7			feeding & nursery & shelter function

Summary

The Schelde is a lowland river originating in the northern part of France (St. Quentin), and entering the North Sea near Vlissingen, The Netherlands. The estuary covers about half of its length (355 km) as the tidal influence is stopped by sluices near Gent 160 km upstream. We focused on the Zeeschelde, the estuarine part in Flanders comprising a mesohaline, an oligohaline and a freshwater tidal zone. The Zeeschelde is subject to severe eutrophication as it receives high inputs from domestic, industrial and agricultural activities. The ecological values and nature conservation interests of the Zeeschelde are taken into consideration by a series of (inter)national policy instruments, aiming at a sustainable management and conservation of this aquatic environment. As a result several management plans apply also to the Zeeschelde or to parts of it. The most far-reaching plans are the Long Term Vision for the Schelde estuary (LTVS) and the updated Sigmaplan which combine ecological rehabilitation and sustainable habitat creation with flood control measures and navigation requisites.

Compliance with almost all national and international agreements requires monitoring of biota. In the WFD fish is one of the biotic quality elements to be used in order to assess the ecological status of transitional waters. Species composition, abundance and the proportion of disturbance-sensitive species should be quantified. Any distortion attributable to anthropogenic impact is calculated by means of the Ecological Quality Ratio (EQR), representing the difference between monitored data and reference conditions. The fish-based assessment tool that we developed was designed to comply with these criteria. In addition it can be used on a metric level to assess fish species of special interest under the Habitats Directive.

The fish assemblages in the Zeeschelde were described based on sampling results recorded over a period of 13 years. An overview was provided of the temporal and spatial variation in those assemblages along the salinity gradient in the Zeeschelde estuary (**Chapter 2**). The species richness and abundance increased over these years in the different salinity zones of the Zeeschelde. Between 1991 and 2008 a total of 71 fish species were recorded within this part of the estuary. Each salinity zone is characterised by a typical fish assemblage, although some species are shared between all three zones. The observed increase since 2007 in species richness in the freshwater and oligohaline zones coincides with a remarkable increase in dissolved oxygen.

Guild specific qualitative Maximal and Good Ecological Potential (MEP/GEP) lists were composed for the different zones within the Zeeschelde estuary and its tidal tributaries (**Chapter 3**). The geographical range and ecological demands of the detected fish species were assessed. The outcome was decisive for acceptance within these lists, which served to develop a fish-based index for the Zeeschelde.

In **chapter 4** the ecological goals and associated habitat needs were described for fish populations in estuaries. The Zeeschelde was presented as a case study for the description of ecological goals for the fish species listed in the MEP/GEP lists. In order to make the method more widely applicable we first classified fishes into guilds, relevant for the formulation of ecological goals. Next we described guild-specific ecological goals and defined habitat needs linked with a proper functioning of the estuarine ecosystem. The habitat needs ensure the completion of all lifecycle stages: spawning, breeding, feeding and growth to maturity. A hierarchical approach was adopted to define the goals and habitat needs: from a regional scale to habitat level. On a regional and basin wide scale the essential habitat need is connectivity, on an estuarine scale this is space and on a habitat scale diversity is most important. The proposed ecological goals need further quantification. However in general the rehabilitation of marshes and mudflats and the enhancement of flood control areas as fish habitats, with special attention for connectivity with the estuary, will significantly increase the carrying capacity of the Zeeschelde for most of the relevant populations. In Chapters 5 and 6 two essential habitat needs are discussed in detail.

In **chapter 5**, we modelled the environmental constraints controlling the movements of anadromous and catadromous fish populations that migrate through the tidal watershed of the river Schelde. For remaining diadromous populations (flounder, three-spined stickleback, twaite shad, thinlip mullet, European eel and European smelt) a data driven logistic model was parameterized. We modelled the presence/absence of fish species in samples taken between 1995 and 2004 as a function of temperature, dissolved oxygen, river flow and season. We demonstrated that it is possible to make acceptable predictions about the future spatiotemporal distribution of migrant fishes, even if only relatively limited information is available. An important management issue that derived from our study is that it is essential to avoid at all times DO concentrations below 5 mg l^{-1} in the freshwater and brackish tidal estuary of the watershed. Restoration of habitats such as marshes and mudflat areas will enhance aeration of the water and help to avoid severe DO drops.

The use of tidal marshes for fish and the influence of creek characteristics on the visiting fish assemblages were assessed (**Chapter 6**). As expected the influence of the salinity gradient is reflected in the different fish assemblages. We caught a high proportion of juveniles suggesting that the creeks are a juvenile habitat. The highest fish abundance was recorded in summer (after hatching) because then juveniles seek shelter in the creeks. It was also observed that the visit frequency was related to creek dimensions and inundation time. Larger creeks, lower in the tidal frame and with a more complex structure, as they include side creeks and permanent pools, are of higher interest for fish. We also observed a positive effect of rivulets on the mudflat adjoining the tidal marsh as they guide the fish towards the creeks. These observations are important for the design of tidal wetland restoration projects.

In chapters 7 and 8 different approaches to define a fish-based evaluation tool to assess the ecological quality status of an estuary (the Zeeschelde) were described. The fish index comprises metrics which are ecologically relevant variables that are sensitive to human pressures. A first step in the selection of these metrics consisted in assessing how they evolve along a pressure gradient (graphical selection).

In **chapter 7** a new concept in the index development was introduced i.e. the balance between type I (false positive) and type II (false negative) errors. The magnitude of these errors was expressed as the area under the curve (AUC). Graphical screening assured the selection of metrics responsive to anthropogenic degradation. We scored metrics by judging the metric value variation in the best available site (quintiles). A forward stepwise regression selected the metric with the best balance between the type I and type II error. Metric selection was continued until the lowest AUC was obtained. To define the EBI thresholds we fixed the maximum type I error of each integrity class threshold at 10%. It was a major concern that not all quality classes can be discriminated because of unbalanced pre-classification data. Secondly the final index had a high type II error, although we believe both types of error should be small. Therefore in the next chapter a different approach was used in order to obtain a better index.

In **chapter 8** we described the development of a Zone specific fish-based multimetric Estuarine index of Biotic Integrity (Z-EBI) based on fish surveys data from the Zeeschelde estuary (Chapter 2). Again we pre-classified sites using indicators of anthropogenic impact and selected metrics showing a monotone response with pressure classes for further analysis. Metric values were calculated using pooled annual data within one salinity zone and expressed as catch per unit effort. Metrics were selected using a Principal Component Analysis (PCA) combined with a redundancy test. We defined thresholds for the Good Ecological Potential (GEP) from salinity zone specific references developed in chapter 3. and applied a modified trisection for the other thresholds (moderate, poor and bad). The Z-EBI is defined by the average of the metric scores calculated over a one year period within each zone and translated into an Ecological Quality Ratio (EQR) to comply with the European Water Framework Directive (WFD). The indices integrate structural and functional qualities of the estuarine fish communities and can be used to assess the ecological quality of the Zeeschelde. We successfully validated the Z-EBI performances for habitat degradation in the various habitat zones. With this new index we encompass small temporal and spatial variations within the estuary. It accounts for the seasonal variation and covers the complete salinity zone, which is an improvement compared to the previous index. The developed indices are able to make the distinction between impacted and unimpacted (GEP) status.

Our results showed that the ecological status of the Zeeschelde at present varies from bad to moderate. A comparison of the average scores obtained with EBI and Z-EBI indicated that in those cases where a different appreciation appeared, the EBI scores lower. This confirms our view that local and temporal appreciations are too sensitive to small variations, which do not necessarily represent an overall negative impact on the ecosystem functioning. Implementing rehabilitation and conservation measures will improve the ecological quality status of the Zeeschelde.

At present the Z-EBI corresponds best with the demands from the different legislations and provides the most holistic information from an ecological point of view.

Samenvatting

De Schelde ontspringt in St. Quentin (Frankrijk) en mondt 355 km verder uit in de Noordzee nabij Vlissingen (Nederland). Tussen Gent en de monding is de Schelde over zowat 160 km onderhevig aan getijdewerking. In deze studie concentreerden we ons op de Zeeschelde (Belgisch estuarium) met haar drie saliniteit zones: een mesohaline zone, een oligohaline zone (inclusief de Rupel) en een zoetwater zone (inclusief de Durme, Dijle, Zenne, Grote en Kleine Nete). De Zeeschelde wordt vervuild door huishoudelijk en industrieel afval en ten gevolge van landbouwactiviteiten. Toch heeft de Zeeschelde een hoog ecologisch potentieel en een natuurwaarde die door nationale en internationale richtlijnen worden gewaarborgd. Voor het verzekeren van natuurherstel, gecombineerd met veiligheid en toegankelijkheid, werd gekozen voor het wenselijk alternatief van het geactualiseerd Sigmaplan. Als onderdeel van de studies die nagaan of aan de verschillende richtlijnen wordt voldaan, is in de meeste gevallen ook een beoordeling vereist van de status van biota. In de Kaderrichtlijn Water wordt vis vooropgesteld als een kwaliteitselement voor het beoordelen van de ecologische status van overgangswater. Een verschuiving tengevolge van menselijke activiteiten in de soortensamenstelling, abundantie en aantal gevoelige soorten wordt weergegeven in een ecologische kwaliteitsratio, die het verschil aantoont tussen de actuele en de referentietoestand. Daarom ontwikkelden we een visindex die gevoelig is voor dergelijke verschuivingen en die tevens elementen opneemt die van belang zijn voor de habitatrichtlijn.

Op basis van vangstgegevens, verzameld over 13 jaar, beschreven we de vissamenstelling in de Zeeschelde langsheen de zoutgradiënt (**Hoofdstuk 2**). In totaal vingen we voor de drie saliniteitszones 71 verschillende soorten. Elke zone was gekenmerkt door een typische visgemeenschap, die we verder onderverdeelden in gildes of ecologische groepen. De toename van het aantal soorten in de verschillende zones viel samen met een verbetering van de waterkwaliteit (opgeloste zuurstof).

Op basis van de recente en historische visstandgegevens stelden we referentielijsten samen die beantwoorden aan het Maximaal Ecologisch Potentieel (MEP) en het Goed Ecologisch Potentieel (GEP) van de drie saliniteitzones in de Zeeschelde vis (**Hoofdstuk 3**). De geografische spreiding en ecologische vereisten van elke vissoort waren bepalend om deze al dan niet in de lijst op te nemen. Deze referentielijsten werden gebruikt voor het ontwikkelen van een zone specifieke visindex voor het Zeeschelde estuarium.

We groepeerden de vissen uit de referentielijsten in gildes en expliciteerden hun ecologische doelstellingen en de ermee geassocieerde habitateisen (**Hoofdstuk 4**). De aanwezigheid van de vereiste habitatten garandeert dat de betrokken vissen hun levenscyclus kunnen voltooien. Op regionale en bekkenschaal houdt dat ondermeer ecologische connectiviteit in, op estuariene schaal is dat voornamelijk ruimte en op habitatniveau diversiteit. De bescherming en de maatregelen natuurherstel waarbij slikken, schorren en gecontroleerde overstromingsvlaktes worden gerealiseerd, verhogen de draagkracht van de Zeeschelde voor vis.

De habitateisen beschreven in hoofdstuk 4 zijn kwalitatief. Om de connectiviteit te kwantificeren modelleerden we omgevingsvariabelen die een belangrijke invloed uitoefenen op de migratie van diadrome vispopulaties in de Zeeschelde (**Hoofdstuk 5**). Zo modelleerden we de aan- en afwezigheid van migratoren in de Schelde in functie van temperatuur, opgeloste zuurstof, stroomsnelheid en seizoen. We toonden aan dat met relatief weinig informatie aanvaardbare voorspellingen konden gemaakt worden van de ruimtelijke en tijdelijke verspreiding van migrerende vissoorten. Dat in het zoetwater- en brakwatergedeelte een zuurstofconcentratie van minstens 5 mg l^{-1} een noodzakelijke habitatvereiste blijkt te zijn, is belangrijk voor het estuariumbeheer. De realisatie en bescherming van afdoende oppervlakten slikken en schorren zijn noodzakelijk om de zuurstofuitwisseling te verbeteren.

Het gebruik van schorren door vissen en het belang van kreekeigenschappen voor de bezoekende visgemeenschappen verduidelijkten we in **hoofdstuk 6**. Naargelang het zoutgehalte troffen we in de schorkreken andere visgemeenschappen aan. In alle schorkreken vingen we hoofdzakelijk juveniele exemplaren met een piek in de zomer. De positie van de kreek in het getijdevenster beïnvloedt de bezoekfrequentie van de vissen, dit is ook het geval bij aanwezigheid van een geul op het slik vóór het schor. Kreken die relatief lager liggen, breed zijn en vertakkingen hebben met permanente poelen worden het meest bezocht door vissen.

In **hoofdstuk 7** beschreven we de ontwikkeling van een op vis gebaseerd scoresysteem: de visindex (EBI). Deze visindex bevat metrieken of ecologisch relevante variabelen die gevoelig zijn voor verstoring. Een metriek die een staalnameplaats bijna altijd een zelfde status geeft als deze bepaald op basis van de omgevingsindicatoren is een goede metriek met een kleine foutenmarge. Het evenwicht tussen type I- en type II- fout kan met een curve weergegeven worden en het oppervlak onder deze lijn (AUC: area under the curve) is een maat voor de performantie van de metriek: hoe kleiner de oppervlakte hoe performanter. Met een stapsgewijze regressieanalyse selecteerden we eerst de metriek met de laagste AUC, waarna we de volgende metriek selecteerden die in combinatie met de eerste een nog kleinere AUC geeft tot uiteindelijk de AUC niet verder afnam. Finaal selecteerden we vijf metrieken en de spreiding van hun gemiddelde waarde werd gebruikt om de grenswaarden van de index te bepalen. Deze index is in staat op basis van één afvissing de kwaliteit van een staalnameplaats vast te leggen. Hij vertoont echter nog enkele tekortkomingen en daarom ontwikkelden we met een alternatieve benadering nog een andere visindex voor de Zeeschelde (**Hoofdstuk 8**).

Bij de alternatieve benadering opteerden we om voor het berekenen van de metriekwaarden alle gegevens per jaar binnen één zone te combineren. Dat impliceerde dat de resulterende index (Z-EBI) de Zeeschelde per saliniteitzone evalueert, gebaseerd op jaargegevens. Metrieken werden geselecteerd met behulp van statistische analyses, gecombineerd met ecologische achtergrondkennis. De referentielijsten werden gebruikt om grenswaarden voor elke geselecteerde metriek te bepalen. Het gemiddelde van de metriek scores berekend voor één jaar gaf de indexwaarde aan. Deze werd vertaald in een ecologische kwaliteitsratio (EQR) in overeenstemming met de Kaderrichtlijn Water. In elke zone beoordeelt de index structurele en functionele kwaliteiten en bepaalt hij de staat van de ecologische kwaliteit van de Zeeschelde. Door het gebruik van jaargegevens hielden we rekening met seizoensverschillen en door het beoordelen van een totale zone werden eveneens ruimtelijke verschillen geïntegreerd.

De indexwaarden toonden aan dat de ecologische status van de Zeeschelde naargelang de zone varieert tussen slecht en matig. Bij een vergelijking van de EBI en Z-EBI scores stelden we vast dat bij een verschil de EBI steeds lager scoorde. Dit bevestigde onze hypothese dat het gebruik van locale en tijdelijke beoordelingen te gevoelig is voor kleine veranderingen die daarenboven niet noodzakelijk een negatieve invloed hebben op het functioneren van het ecosysteem.

Momenteel beantwoordt de Z-EBI het best aan de criteria van verschillende richtlijnen en vanuit een ecologisch perspectief verschaft ze de meest holistische beoordeling.

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So Long, and Thanks for All the Fish

Douglas Adams, 1952.