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Catchment–coastal zone interaction based upon scenario and model analysis: Elbe and the German Bight case study

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Abstract This paper presents a holistic strategy on the interaction of activities in the Elbe river basin and their effects on eutrophication in the coastal waters of the German Bight. This catchment–coastal zone interaction is the main target of the EUROCAT (EUROpean CATchments, catchment changes and their impact on the coast) research project, with the Elbe being one of eight case studies. The definition of socio-economic scenarios is linked with the application of models to evaluate measures in the catchment by estimation of nutrient emissions with MONERIS (MODelling Nutrient Emissions in RIVER Systems), and their effects on coastal waters with the ecosystem model ERSEM (European Regional Seas Ecosystem Model). The cost effectiveness of reduction measures will then be evaluated by application of the CENER model (Cost-Effective Nutrient Emission Reduction) and a multi-criteria

analysis. Finally, the interpretation of ecological integrity is used as a measure to describe ecological impacts in an aggregated form.

Keywords Scenarios · Nutrients · Elbe · Water framework directive · Coastal zone

Introduction

The Elbe catchment is part of one of eight case studies within the EUROCAT project. Aim of the project is to achieve Integrated River Basin Management (IRBM), as targeted in the Water Framework Directive (WFD) 2000/60/EC (http://europa.eu.int/comm/environment/water/water-framework/index_en.html), in order to prevent and mitigate possible externalities induced in the coastal zone by considering the anthropogenic influence from the spring to the outlet within a given river basin. The new element is the holistic river district approach, which takes into account the socio-economic development in the catchment. The WFD demands a link between IRBM and Integrated Coastal Zone Management (ICZM). Implementation of the WFD will be a major issue in future debates on local as well as regional levels, which makes the EUROCAT approach relevant for future discussions concerning sustainable regional development (EEA 1999; European Community 2000).

Within the scope of EUROCAT, the Driver-Pressure-State-Impact-Response approach (DPSIR) of the European Environment Agency is the analytical framework selected to handle complex humankind–ecosystem interactions (Behrendt et al. 2000). Scenarios assess how different human activities could come into existence, thus causing an impact on the environment and potentially damaging the (coastal) ecological integrity. Furthermore, scenarios represent plausible images of the future, in which environmental risk perception and therewith interpretation of environmental risk and (coastal) vulnerability will considerably influence politi-

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cal targets. The political issues, lifestyles of the society, and social values characterising each scenario will exert pressures on the environment. Some societal drivers, such as urbanisation and food demand and their ecological impact on ecosystem integrity and function (e.g. heterogeneity), are qualitatively assessed under each scenario. A relative evaluation of the different pressure intensities under different future conditions is the basis for deriving ecosystem state indicators (e.g. species composition and ratio diatoms/flagellates for heterogeneity) and aggregated impact indicators corresponding to the demand of the WFD (Janssen et al. 2001). The Elbe case study focuses on coastal eutrophication of the North Sea, thus concentrating on analysis of nutrient emissions originating from the catchment, and their potential reduction through different management options (Hofmann et al. 2002; Nunneri and Hofmann 2002). With respect to coastal eutrophication, nitrogen and phosphorus are quantitatively the most relevant components (De Jonge et al. 2002). In order to propose a reduction of eutrophication of the North Sea based on a better inland management, the following questions are addressed:

- What must be achieved in river basins (reduction of nutrient loads of nitrogen and phosphorus) to meet standards (e.g. good ecological status) desired by the society in the coastal zone?
- What are the effects of the changes of nutrient loads in the river basin on the ecosystem in the coastal zone?

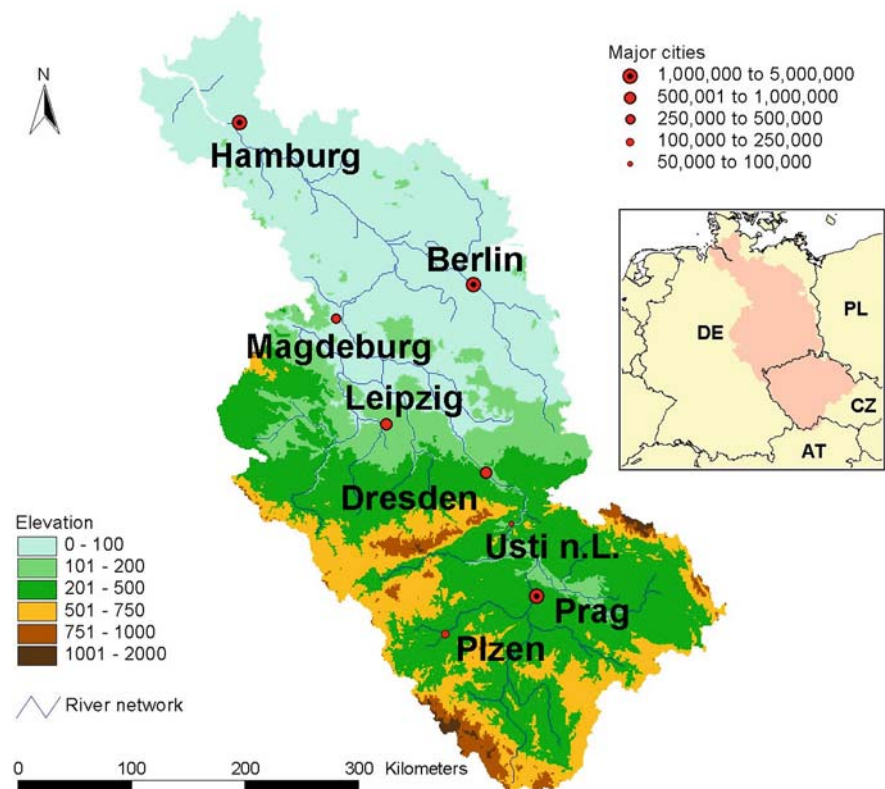
The outline of this paper is as follows. The section *Study area* provides a description of the Elbe catchment, estuary and coast, delineating the study area and focus of this paper. The section *Methodology* points out the methodology used for the study of the Elbe catchment–coast continuum. The qualitative methodology is quantified in the section *Modelling tools and information exchange* by a number of models, describing the water quality in the catchment, ecological quality by indicators at the coast, and the linkage between the catchment and the coast via a transfer function. Furthermore, the coastal indicators and the costs of taking measures and other indicators are used for a multi-criteria analysis of nutrient reduction policy options, while the coastal indicators are also used for an ecosystem integrity analysis. The result of these models are presented in the section *Results*. The final section draws conclusions.

Study area

Catchment and estuary of the Elbe River

With an area of 148,268 km² and a river length of 1,091 km, the Elbe catchment is the 12th largest river basin in Europe (Fig. 1). About two thirds of the catchment area belongs to Germany, and one third to the Czech Republic. Austria and Poland have nearly the same small shares in the catchment. Conventionally, the Elbe is divided into three parts:

Fig. 1 The Elbe river basin



- the upper Elbe (from the spring to Elbe km 96—Schloss Hirschstein) comprises the middle mountain relief with elevations of ca. 200–300 m to max. 1,600 m a.s.l., and Palaeozoic rocks as well as broad basins like the Bohemian basin consisting of soft Cretaceous rocks;
- the middle Elbe (from Elbe km 96 to Elbe km 585.9—weir Geesthacht) with a hilly landscape and dominant forming processes during the Pleistocene glaciations and deglaciations (moraines, outwash plains, melt-water drainage systems, loess deposits), and the Holocene (river meadows, floodplains, lakes);
- the lower Elbe (from Elbe km 585.9 to the North Sea, Elbe km 727.7—Cuxhaven-Kugelbake) with flat lowlands < 50 m a.s.l., and a transition to an estuary river zone downstream of Hamburg (the description of the coastal zone is given in the section *The German Bight*).

The climatic conditions are characterised by the interaction of humid maritime air masses coming from southwest to northwest directions, and dry continental air masses of easterly direction. As a result, the precipitation in the northern and western part of the north German lowland is much higher than in the eastern part with continental influence (Wit 1999). Average annual precipitation ranges from 450 mm year⁻¹ in the loess area east of the Harz mountains to more than 1,200 mm year⁻¹ in the spring region of the Elbe (Krkonoše mountains, Riesengebirge). The mean annual precipitation for the entire Elbe catchment amounts to 630 mm year⁻¹ (period 1961–1990). The average annual river discharge in the river Elbe is 717 m³ s⁻¹ (period 1977–1999), with minimum values of 461 m³ s⁻¹ in September and maximum values of 1,140 m³ s⁻¹ in April (Lenhart and Pätzsch 2001).

At present about 61% of the catchment is used for agriculture, 29% is covered by forests, and 6% is urban area. The regionalized land use for the Czech part and the WFD coordination regions in the German part of the Elbe are shown in Fig. 2. Different economic activities are concerned with nutrient emissions: agriculture, drinking water supply, industry and tourism often represent conflicting interests.

The total population of the catchment is about 24 million; 58% of the Czech citizens and 22.9% of the German population live in the catchment (Fig. 3). The average population density is 167 inhabitants km⁻², with a minimum density of 99 inhabitants km⁻² (in the middle Elbe, the area between the Saale confluence and the monitoring station of Zollenspieker) and a maximum density of 281 inhabitants km⁻² (in the tidal Elbe sector, downstream of Zollenspieker; ARGE-Elbe 2001). The largest (and most densely populated) cities are Berlin, Hamburg and Prague.

A variety of socio-economic conditions characterises the catchment. Especially due to the changing political and economic conditions after Germany's reunification in 1990, the emissions of nutrients decreased, mainly due to reduction of point source emissions for WWTPs

(implementation of effective wastewater treatment) and industry (e.g. closure of factories).

The total length of the lower Elbe to the North Sea at Cuxhaven/Kugelbake is 142 km. Near the city of Hamburg, the river divides into two branches: the Nordereibe and Südereibe, encompassing the harbour. From this point, the river forms an estuary with a width of 1.5 km downstream of Hamburg and 18 km near Cuxhaven. Because of the passing tidal wave in the North Sea, an oscillating tidal current is generated. Several tide-associated mechanisms result in the occurrence of a long residence time of any water body in this system. Thus, a given water body has to pass the same section several times, unless it reaches the open waters of the North Sea. Consequently, the residence time of water is much greater in the tidal Elbe than in the upper reaches. Between Glückstadt and Cuxhaven, the estuary becomes a mixing zone of freshwater and saltwater (Fig. 4).

With a length of 90 km, the Elbe estuary is connected to the morphological system of the Wadden Sea–German Bight. The estuarine mixing zone begins some 50 km downstream of Hamburg, which is located in the freshwater (limnic) zone. While the mouth of the inner (tidal Elbe) estuary is situated near Cuxhaven, the outer estuary stretches 30–40 km into the German Bight, without a clearly defined seaward limit. The total retention of fluvial suspended matter in the tidal Elbe estuary is estimated to be 80–85% of the total load equal to 800–860 kt year⁻¹ (kt, kiloton; Zwolsman 1994). The Elbe receives large amounts of nutrients upstream of Hamburg.

The German Bight

The river Elbe flows into the south-eastern part of the North Sea, the German Bight. Thus, also the coastal zone of the German Bight is part of the “nested” model. With respect to coastal eutrophication, the plant nutrients nitrogen, phosphorus and silicate are quantitatively the most important components. In contrast to the N and P loads, the silicate loads are minimally influenced by anthropogenic activities. The river water contains substantial amounts of these nutrients, which are the causative factor for the eutrophication process in the German Bight and the adjacent Wadden Sea (Brockmann et al. 2000; ARGE-Elbe 2001; Beusekom et al. 2001). The nutrients discharged into the coastal water cause an enhanced formation of algal biomass by initial uptake and turnover. Meteorological and hydrographic conditions lead to intense stratification, and the subsequent intensified sedimentation of organic matter can cause oxygen depletion in the bottom waters of the coastal region. In addition to this autochthonous organic matter, allochthonous riverine inputs of organic carbon contribute to the eutrophication of the coastal water (Rachor and Albrecht 1983; Hickel et al. 1989; Gerlach 1990; Niermann 1990).

Fig. 2 Land use in the different regions of the Elbe river basin (note that the German part is regionalized according to the WFD coordination regions; CR Czech Republic)

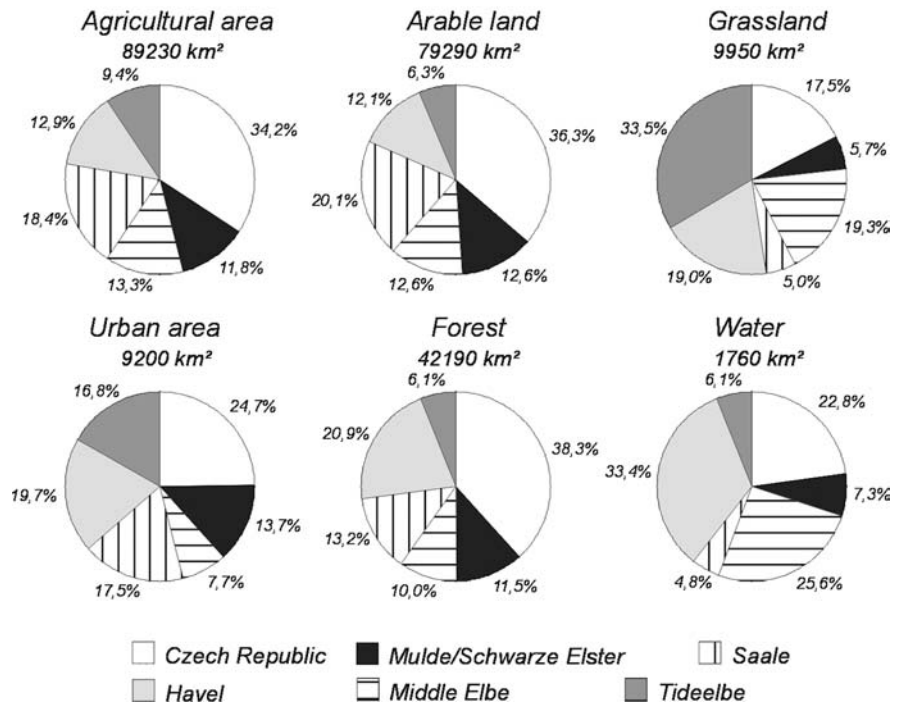
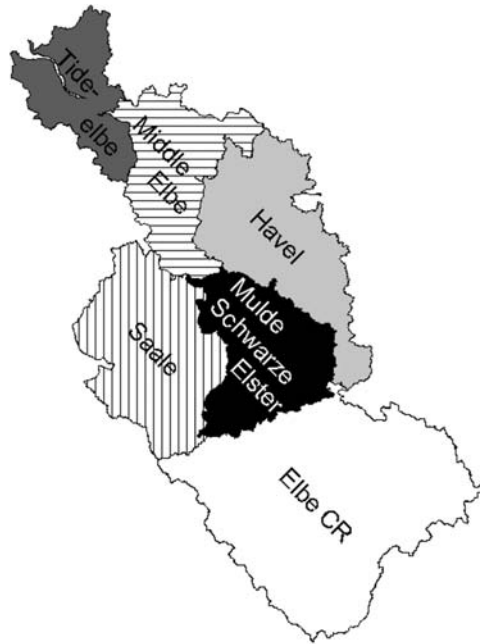


Fig. 3 Proportions of the German part (WFD coordination regions) and the Czech part with regard to catchment, population and mean runoff (1993–1997) within the Elbe river basin

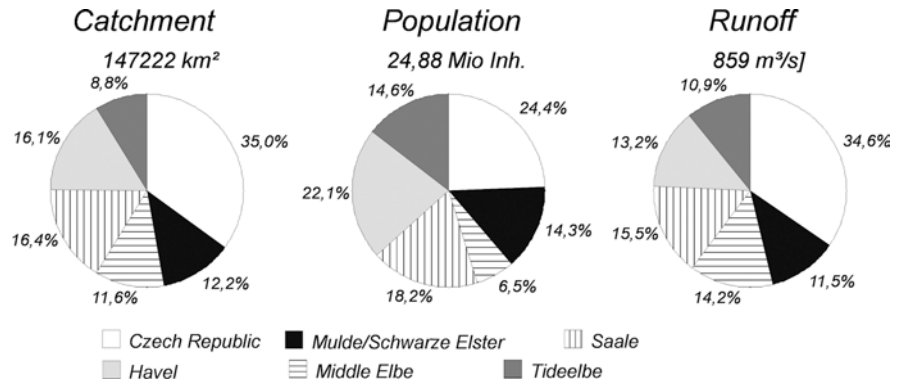
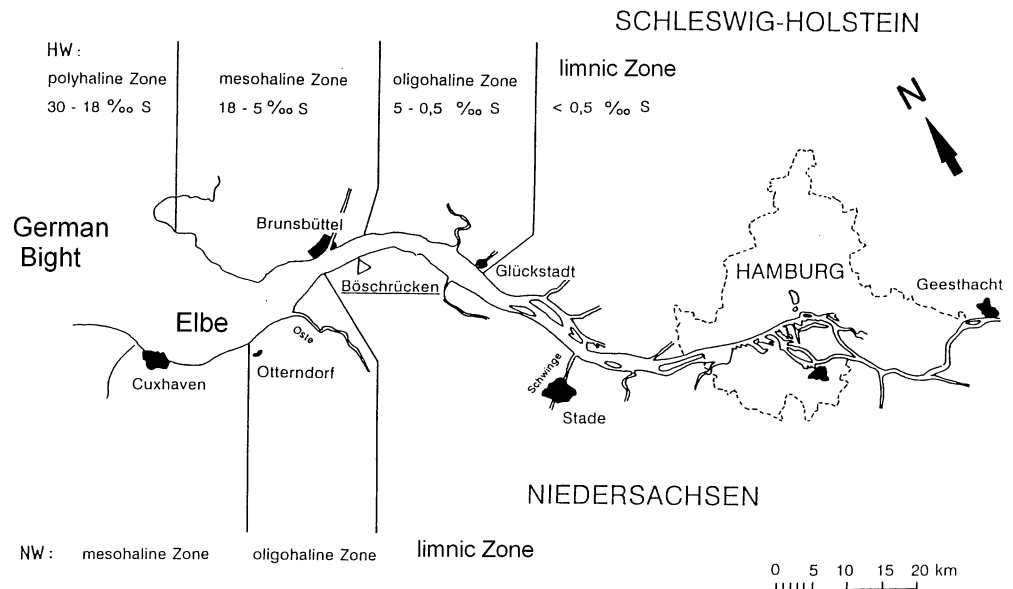


Fig. 4 The Elbe estuary and its salinity zonation (modified after Duve 1999)



The German Bight is part of the continental coastal water of the North Sea. The coastal waters which are influenced by the runoff of the river Elbe can be divided into three different water types: the marine waters of the German Bight, the more brackish northern German Wadden Sea, and the river plume of the river Elbe.

The complex current pattern of the German Bight is steered by the general counter-clockwise circulation of the North Sea (Fig. 5), which has an eastward and a northward component in the German Bight. The major inflow and outflow into and out of the North Sea occurs at the northern boundary. However, about 85% of the inflowing water masses is recirculated back into the Atlantic north of the Doggerbank, and only a small fraction (4.6%) of the northern inflowing Atlantic water masses contributes to the exchange in the southern parts of the North Sea (Lenhart and Pohlmann 1997).

The southern circulation system is triggered by the inflow through the English Channel, follows the continental coastline as continental coastal current, and finally adds to the Norwegian Trench outflow back into the Atlantic. Therefore, only the narrow band of the dominant circulation pattern along the continental coast is influenced by the mixing with the river runoff of the Rhine and Elbe, and some other smaller rivers (i.a. Weser, Ems). Due to its location at the outermost south-eastern edge of the North Sea, the German Bight has a relatively long flushing time. Calculations by Lenhart and Pohlmann (1997) resulted in a mean flushing time of 33 days with a range of 10–56 days for the period 1982–1992. This long flushing time results in a higher tendency towards sedimentation of organic material, which adds to the problem of subsequent oxygen depletion.

Due to the large catchment area of the Elbe, which is located in an industrialised and also agriculturally intensively utilised area in central Europe, a large

number of different substances of anthropogenic and natural origin enter the river and are transferred to the North Sea (e.g. Becker et al. 2002). With respect to coastal eutrophication, nitrogen and phosphorus are quantitatively the most important components. The Wadden Sea area of Schleswig-Holstein—the main impact area for the coastal investigations of the Elbe case study of EUROCAT—can generally be characterised as a rural area with a low population density (80 inhabitants per km² for Nordfriesland and 94 for Dithmarschen, compared to 229.4 inhabitants per km² in Germany and 175.9 in Schleswig-Holstein), agriculture as dominating land use (but less than 5% of the local income relates to agriculture), and tourism as dominating economic force (in some spots like the Wadden Sea islands, more than 80% of the local income is related to tourism). Land reclamation and coastal defence have historically been drivers for man's existence and survival in this area.

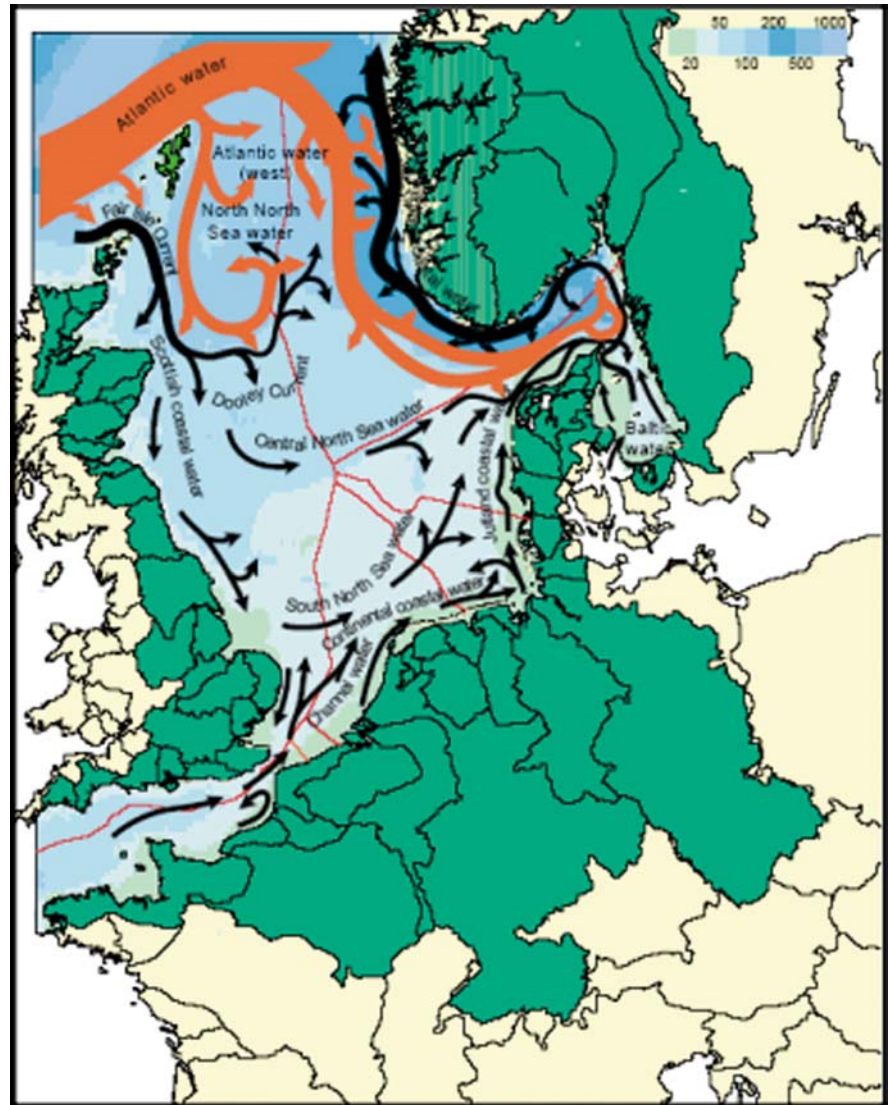
The area is characterised by a very diverse mix of human activities and pressures on- and even more offshore which influence coastal ecosystem integrity in a similar way, or even more than river inputs. Human pressures include coastal defence and nature conservation, tourism, agriculture, wind farms (onshore and offshore), fisheries, shipping and, to a limited degree, mariculture (Kannen et al. 2000). The latter is taking up momentum to become a growing and innovative industry (algae farming, recirculation plants).

Methodology

The DPSIR approach

EUROCAT aims to identify the influence of anthropogenic terrestrial land use and anthropogenic matter

Fig. 5 General circulation pattern in the North Sea (OSPAR Commission 2000). The width of *arrows* is indicative of the magnitude of volume transport (*red arrows* relatively pure Atlantic water currents, *black arrows* North Sea water and coastal water currents). The drainage system of the North Sea basin is indicated by the different contributing river catchments (*green*)



fluxes through river systems on the ecological quality and the ecological as well as socio-economic service functions of the adjacent coastal zones. The analytical framework used to describe the catchment–coast interactions in the EUROCAT case study of the Elbe is the Driver-Pressure-State-Impact-Response approach (DPSIR; Turner et al. 1998), which is also applied by the European Environment Agency (EEA), the Global Integrated Water Assessment (GIWA), and within several regional assessments of the LOICZ Basins Initiative, to which EUROCAT belongs as well. The DPSIR framework, originally established by the OECD in 1993 as PSR approach and later enhanced by the European Environment Agency, was selected as the analytical tool to handle these complex man and biosphere interactions.

The DPSIR framework is based on the assumption that the continuum catchment/coast acts as a dynamic system linking social systems (human activities and the resulting pressures) to ecological systems. Both systems are connected by feedback loops which, depending

ultimately on political choices (responses), can enhance or mitigate impacts on the environment and consequent impacts on humankind (Nunneri et al. 2002). Drivers are generally seen as socio-economic factors which cause environmental pressures and consequently lead to changes in the state of the environment. These changes can have an impact on social, economic and ecological processes and, as a result, on ecosystem functions. In order to mitigate these undesired effects, management responses or policy options can be implemented which influence the system at different levels (e.g. changing drivers, state or impact).

The focus in the Elbe case study is on the issue of eutrophication in coastal waters, and the related nutrient emissions from the catchment. The approach to combat nutrient enrichment, which forms the main causative factor for eutrophication in the German Bight, generally has to take into account the following main steps:

- determining the eutrophic status of the coastal water body;

- setting goals for water-body restoration according to the WFD, in order to reach (or maintain) a good ecological status of the coastal waters;
- analysing the relationship between loadings and impacts to coastal waters, by using process-specific indicators (e.g. productivity of algal biomass);
- determining critical value ranges related to societal conditions (quantitative, qualitative or in relative terms) for indicators; and
- identifying the most effective strategy which can be implemented for nutrient load reduction (e.g. through cost-effectiveness analysis and multi-criteria analysis), among a selected pool of measures.

The conceptual approach used in EUROCAT to operationalise the DPSIR framework links scenarios, indicators and modelling (Fig. 6). In principle, all elements of the DPSIR approach, namely drivers, pressures, state, impact and response, need to be described by relevant indicators (section *The indicator concept*) which can be calculated by models (for outlooks into the future) and/or can be measured (for monitoring).

The set-up of a regional Policy Advisory Board (PAB) for the Elbe provides a suitable involvement of users and managers (stakeholders). In the time span of 6 months numerous institutions, associations and NGOs took part in face-to-face interviews with the members of the EUROCAT team, thus giving a first insight into the use conflicts, policy background and networking in the Elbe catchment. We designed a stakeholder panel in such a way that there were 14 representatives from government at various levels, the private sector and nongovernmental organisations. These stakeholders were identified on the basis of whether they were likely to have interests in the catchment area, the coastal area, or both. The list includes multinational agencies (e.g. International Elbe Commission), national and sub-national government representatives, nongovernmental organisations and research institutions.

The indicator concept

Instead of the common procedure used by the Organisation for Economic Co-operation and Development (OECD) for driver analysis, the EUROCAT consortium adopted a slightly different nomenclature for the DPSIR framework to suit the aim of the project (Colijn et al. 2002; Rice 2003). Drivers, pressures and responses are formulated for the river catchments as well as for the coastal areas. As the focus of EUROCAT is to view the coastal zone as receptor area of catchment activities, state and impact indicators are developed only for the coastal area, and are subdivided into ecological state/impact parameters and socio-economic state/impact parameters (Colijn et al. 2002). Table 1 shows the set of potential indicators defined by EUROCAT team members.

The assessment approach used in EUROCAT is based on quantitative and qualitative assessment of indicator changes related to scenario storylines. While some management measures to mitigate negative environmental effects will be more effective than others, some will as well be more expensive than others.

To identify the societal forces which drive the amount of ecosystem services used by human activities, the EUROCAT-Elbe consortium selected six issues (drivers)—(1) food demand, (2) urbanisation, (3) energy demand, (4) mobility and transport, (5) industry and housing, and (6) nature conservation—which create pressures on ecosystems. These are consistent with the issues discussed in the progress report of the 5th International Conference on the Protection of the North Sea in Bergen in 2002 (Protection North Sea 2002). According to the EUROCAT approach, the riverine nutrient loads (nitrogen and phosphorus) are selected as forcing function or pressure indicator for the ecological change related to eutrophication in the coastal zone.

In order to link the concept of self-organising capacity of ecosystems and ecosystem integrity (section

Fig. 6 Harmonised model approach (CZ coastal zone)

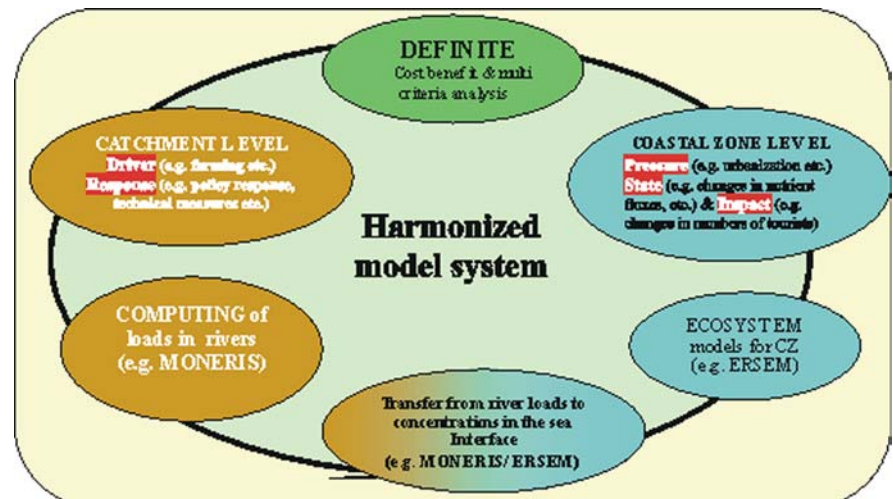


Table 1 Selected primary indicators for eutrophication in the coastal zone

Indicator	Metrics
NO ₃ concentration, NH ₄ concentration, NO ₂ concentration PO ₄ concentration	$\mu\text{ mol l}^{-1}$ in winter (December–January; time of lowest pelagic biological net production) $\mu\text{ mol l}^{-1}$ in winter (December–January; time of lowest pelagic biological net production)
N/P ratio	-
Chlorophyll-a concentration	$\mu\text{ g l}^{-1}$, mean summer (3 summer months)
Chlorophyll-a concentration	$\mu\text{ g l}^{-1}$, spring bloom average (1 month around peak)
Timing of spring bloom	Week in the year, ± 1 week around chl-a max.
Spring bloom duration	Length of period reaching 25 or 50% of max. chl-a during spring bloom
O ₂ concentration, O ₂ saturation	mg l^{-1} , bottom water in summer
Diatom/non-diatom ratio	Biomass ratio in growing season (March–September)
(phytoplankton species composition)	
<i>Phaeocystis</i> : length of bloom	Length of period with $> 10^6$ cells l^{-1}
Net primary production	$\text{g C m}^{-2} \text{ year}^{-1}$
Export of organic material (flux)	$\text{g C m}^{-3} \text{ day}^{-1}$

Ecosystem integrity) with modelling of coastal impacts, the following parameters derived from ERSEM (section *ERSEM*) simulations were selected:

1. primary production,
2. turnover rate of winter nutrients,
3. nutrient gain by the sediment,
4. diatom/non-diatom ratio, and
5. nutrient losses out of the box.

In addition, standard measures for state parameters like mean winter DIN (dissolved inorganic nitrogen) and DIP (dissolved inorganic phosphorus), winter DIN/DIP ratio, timing of spring bloom, chlorophyll-a, primary production and diatom/non-diatom ratio were calculated as well.

The approach outlined here to select indicators based on ecosystem theory is in accordance with the present discussion within international agreements like OSPAR or HELCOM, concerning the application of the ecosystem approach in environmental policy making. It also fits into the definitions for ecological quality and ecological quality objectives given by the North Sea Task Force (NSTF), which consists of experts from both OSPAR and the International Council for the Exploration of the Sea (ICES).

The Water Framework Directive (WFD) uses a list of biological and physicochemical quality elements for coastal waters (European Union 2000, see annex V, chap. 1.2.5, Water Framework Directive). Most of these elements (16 out of 19) are items which describe the state of coastal waters, and only three are targeting the functioning of the ecosystem. Therefore, Windhorst et al. (2004) assumed that the ecological integrity according to the definition chosen by Barkmann and Windhorst (2000) and Barkmann et al. (2001), and used within REBCAT, has the potential to serve as an integrating approach coupling structures and processes of ecosystems.

The use of socio-economic scenarios

Scenarios represent alternative possible futures. The use of scenarios aims to cope with the uncertainty about how the future will unfold (e.g. Bertrand et al. 1999; EEA 2001; van der Veeren 2002): a “robust” management strategy will reveal the most advantages under different possible developments (Ledoux et al. 2002). In the scenarios considered, different human activities will come into existence, thus causing (qualitative and quantitative) different impacts which will potentially result in damages to the (coastal) ecosystem integrity.

Scenario assessment starts by identifying a focal issue, or management problem (in this case, eutrophication of the North Sea coastal waters). Then, scenarios are ad-hoc constructed futures, each representing a distinct possible world, internally consistent and plausible. The three scenarios used for the Elbe catchment result from a combination of qualitative and quantitative approaches. Qualitative storylines are essential for internal consistency and make scenarios more vivid, while quantification of key variables is essential for providing data to the model simulation by MONERIS (MOdelling Nutrient Emissions in RIVER Systems), for the catchment emissions, and by ERSEM (European Regional Seas Ecosystem Model), for the ecological effects in the coastal zone. Three scenarios have been chosen (see section *Outline of the socio-economic scenarios*), each giving priority to a different field of socio-economics—(1) economic growth, (2) full implementation of the current legislation for an optimal ecosystem use in the light of sustainability, and (3) environmental protection at any cost.

The different political issues, lifestyles and social values characterising each scenario will exert pressures on the environment. Following the DPSIR approach, the next level of analysis focuses on several essential fields of social life of the socio-economic system (drivers)

which, in turn, depend on the societal value settings characterising different scenarios. The scenarios are the framework for evaluating different management measures aiming at nutrient emission reduction in the Elbe catchment (see section *Assessment for the ecosystem integrity*, Perspectives). Particularly the goals for nutrient reduction will be different under different scenarios (see sections *Nutrient emissions in the Elbe catchment and their possible reduction* and *Effects of simulated river load reductions to the coastal waters*), due to the different priorities (maximum under the third scenario, and minimum under the first one).

Modelling tools and information exchange

This assessment presents a holistic strategy on the interaction of activities in the Elbe river basin, and their effects on eutrophication in the coastal waters of the German Bight. While the DPSIR concept, as explained in the previous section, serves as the theoretical frame of the assessment (Fig. 6), the actual flow of information and the model in use is illustrated in Fig. 7.

In order to address the related questions, the following models are used:

- MONERIS for modelling nutrient emissions in the Elbe basin (Behrendt et al. 2000, 2002a),
- ERSEM for modelling the ecosystem changes in the coastal zone (Lenhart 2001),
- the CENER model to calculate cost-effective nutrient emission reductions at the coast by taking measures in the catchment (Lise and Van der Veeren 2002; Lise 2003), and
- MCA (multi-criteria analysis) for ranking policy alternatives on nutrient reduction in the catchment–coast continuum.

In order to link the catchment model MONERIS with the coastal zone (ecosystem model ERSEM), a transfer function is needed. This transfer function takes care of the changes the river load undergoes between the

last tidal-free gauge station at Neu Darchau and the actual North Sea inlet at Cuxhaven. Furthermore, to conduct a multi-criteria analysis, coastal indicators are extracted from ERSEM.

Besides the actual flow of information between the models, also the forcing needed for the model simulation is sketched in Fig. 7.

MONERIS

The model MONERIS (MODelling Nutrient Emissions in RIVER Systems) was developed for the estimation of nutrient inputs via various point and diffuse pathways in German rivers with catchments larger than 500 km². The basis for the model are data on runoff and water quality for the studied river catchments and also a geographical information system (GIS), in which digital maps as well as extensive statistical information for different administrative levels are integrated. Since a detailed description of MONERIS is given by Behrendt et al. (2000), we will just give an overview of the model as follows.

While the point inputs from municipal wastewater treatment plants (WWTPs) and from industry are directly discharged into the rivers, the diffuse entries of nutrients into the surface waters represent the sum of various pathways which have been realised over the individual components of the runoff. The distinction of these individual components is necessary because both the concentrations of materials and the processes are at least clearly distinguished from one another.

Estimates for the following specific inputs (see Fig. 8) are possible for the catchment areas now covered:

- Point sources
- Atmospheric deposition
- Erosion
- Surface runoff
- Urban areas
- Tile drainage areas
- Groundwater

In the diffuse inputs, various transformation, loss and retention processes are characterised. To quantify and forecast the nutrient inputs in relation to their cause requires knowledge of these transformation and retention processes. This is not yet possible through detailed dynamic process models, because the current state of knowledge and existing databases are limited for medium and large river basins. Therefore, existing approaches of macro-scale modelling will be complemented and modified and, if necessary, attempts will be made to derive new applicable conceptual models for the estimate of nutrient inputs via the individual diffuse pathways. The validation of these individual sub-models was performed by comparing the results with independent datasets. For example, the groundwater sub-model was validated with measured groundwater concentrations (Behrendt 2002; Behrendt et al. 2002b).

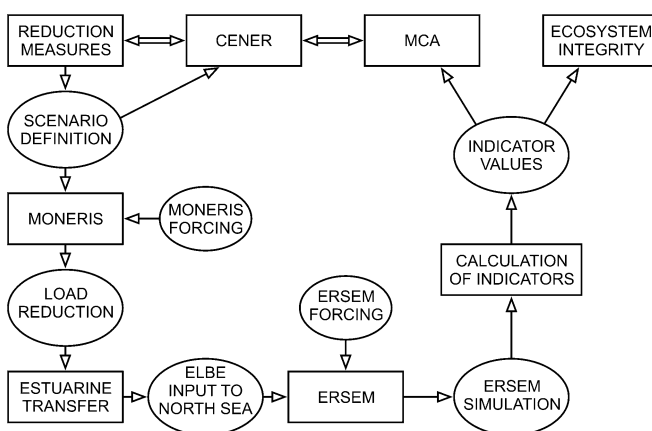


Fig. 7 Model interaction and information exchange

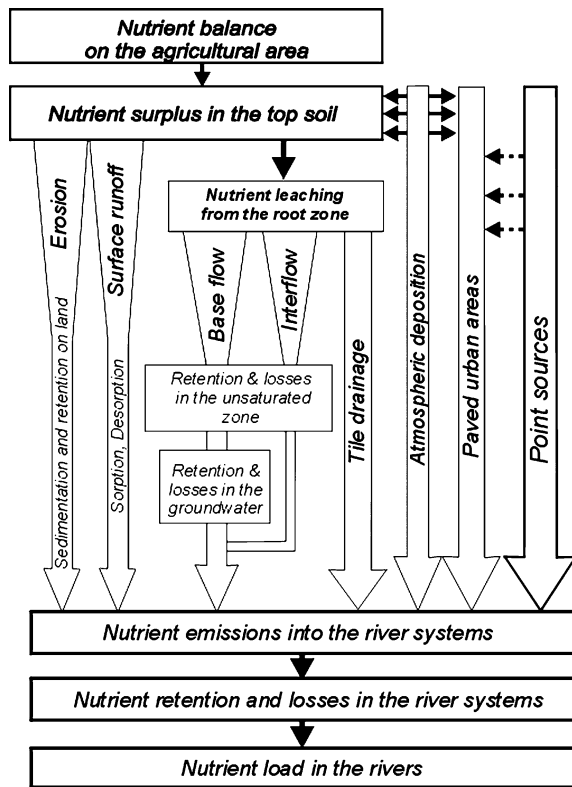


Fig. 8 Pathways and processes considered in the model MONERIS (Behrendt et al. 2000)

The final output is an estimate of annual nutrient load in the river at the outlet of the study catchment, which is equal to the emissions into the river via point and diffuse sources *minus* the estimated nutrient retention and loss within the river system.

Once MONERIS has been calibrated for a particular catchment, it can be used to develop management scenarios. For example, a manager can ask by how much would nutrient emissions into the river be reduced under a scenario of erosion control. Since the Elbe is a trans-boundary river, the compilation of a harmonised database was an essential task. Due to the cooperation and data transfer with Czech partners (Research Institute for Soil and Water Conservation, Praha, Czech Republic), the Czech part of the Elbe basin could be subdivided into 25 sub-catchments with a mean catchment size of 2,000 km². The regionalization of the German part was more detailed, comprising 160 sub-basins with a mean catchment size of 607 km². Therefore, the area-related input data, such as the GIS data of the countries for the land use, elevation, river network, administrative boundaries, hydrogeology and soil, were harmonised into a unified database for the whole Elbe basin. Statistics on population, agriculture and wastewater treatment were collected at the level of municipalities, districts or whole countries overlain with the administrative boundaries to establish thematic maps for the further analysis of the nutrient inputs and the socio-economic conditions. Through overlay of the catchment

boundaries with these data, all values were estimated directly for the catchments. Most of these data are needed as input data for the modelling of the nutrient inputs into the Elbe river system. Due to the availability of monitoring data, we decided to compute the time series 1983–1987, 1993–1997 and 1998–2001.

Linking MONERIS and ERSEM by a transfer function

The link of the dynamic model ERSEM with the steady-state model MONERIS requires an artificial resolution of year cycles (based on 5-year means) in order to derive a seasonality on the base of monthly values. This is necessary because MONERIS is balanced for a particular hydrologic period, and operates with annual average conditions for a 5-year period. The basic idea of an interface between the two models is to generate typical seasonal cycles of the ERSEM inputs from existing data, and scale them to the actual MONERIS outputs. Furthermore, since MONERIS does not give data on the subspecies of nitrogen and phosphorus, these have to be derived from the total loads of N and P, using typical ratios from the Elbe River. A further problem is that the outputs of MONERIS are upstream of the ERSEM input boxes. An analysis of longitudinal transects of the water quality variables in the Elbe estuary is used to transform the MONERIS output to the ERSEM input box.

MONERIS is a catchment model for the transport of dissolved and particulate substances by several pathways. The seaward outputs are *5-year averages of the loads in ton per year*. The substances used in EUROCAT are:

1. total nitrogen (TN, t N year⁻¹)
2. dissolved inorganic nitrogen (DIN, t N year⁻¹)
3. organic nitrogen (t N year⁻¹)
4. total phosphorus (TP, t P year⁻¹)

ERSEM is a box-model for ecological processes in coastal waters. As riverine inputs, ERSEM needs the *daily loads at the river mouths in ton per day*. The relevant substances for EUROCAT are:

1. water discharge (m³ s⁻¹)
2. TN
3. nitrate (t N day⁻¹)
4. ammonia (t N day⁻¹)
5. total phosphorus (t P day⁻¹)
6. phosphate (t P day⁻¹)
7. silicate (t Si day⁻¹)

The fine-tuning of creating the interface between MONERIS and ERSEM was done stepwise:

- definition/calculation of transfer functions for nutrients between MONERIS and ERSEM (section *Linking MONERIS and ERSEM by a transfer function*);
- artificial resolution of year cycles (5-year means) in order to derive seasonality (monthly values); and

– data processing of MONERIS data output to ERSEM data input.

From existing ERSEM input files, a 5-year period is chosen centred at the year of interest, e.g. 1995. To take care of the problem of leap-years in the averaging process, the yearly data are interpolated to a regular temporal grid of 0.002-year spacing. The yearly data are then normalised (integral of the normalised time series = 1). The five normalised yearly time series are then averaged to give a standard normalised year. From the original yearly data for very ERSEM variable, the typical scaling factors for the relation of the subspecies of nitrogen and phosphorus to the total N and P loads are then calculated. The standard normalised yearly data for the total N and P loads (on the 0.002-year grid) are then interpolated to the real days of the reference year 1995, and multiplied by a transfer factor which is derived from the analysis of longitudinal transects of total N and P in the estuary. In the final step, these (normalised) reference year data are multiplied by the MONERIS output TN and TP, giving the ERSEM input for the standard year for total nitrogen and phosphorus. Using the scaling factors for nitrate, ammonia and phosphate, the data for TN and TP are then used to generate the ERSEM input for the remaining variables.

Since there is no way to derive silicate loads from the MONERIS output, the original ERSEM input data for the Elbe are only average according to the procedure described above, as are the water discharges.

ERSEM

For the simulation of the response in the coastal zone caused by changing nutrient loads from the river management strategies, the ecosystem model ERSEM (European Regional Seas Ecosystem Model) has been applied. The ERSEM model (Baretta et al. 1995) has been used to simulate reduction scenarios in a number of cases, both for the North Sea (Lenhart et al. 1995; Ruardij and Van Raaphorst 1995; OSPAR 1998; Lenhart 2001) as well as for the continental coastal shelf region (Heath et al. 2002).

The ecosystem model ERSEM was developed to simulate the ecosystem dynamics of the North Sea. The model simulates the annual cycles of carbon, nitrogen, phosphorus and silicon in the pelagic and benthic food webs of the North Sea. The box model combines hydrodynamical and ecological processes into one model with the same resolution in space and time. The biological part of the model consists of an interlinked set of modules, describing the biological and chemical interactions between the state variables. A general description of the model is given by Baretta et al. (1995) and Lenhart (2001).

The model is forced by irradiance and temperature data, suspended matter concentration, hydrodynamical information for advection and diffusion, data on atmo-

spheric nutrient input to the North Sea as well as by inorganic and organic river load data (Fig. 9).

The model covers an area of 577,620 km² and a volume of 51,047 km³ in total. The northern and central parts of the North Sea are subdivided into 1 by 2° boxes. For resolving the horizontal gradients in the coastal areas, the spatial resolution was increased to boxes of 0.5 by 1°.

In this way, the model is finely resolved in the coastal area, but also represents the central and northern North Sea with sufficient resolution. In this study the ERSEM boxes 58 and 59, 68 and 69, and 77 and 78 were chosen for an integrated Elbe box, thus covering the Elbe estuary (Fig. 10). This coastal area is nearly identical with the OSPAR regions O-II-3D of the greater North Sea.

To represent the ecosystem dynamics in the coastal region with its highly variable conditions, the relevant information on the morphology has been provided and the transport processes have been parameterised on the scale of the box set-up of the model for the year 1995. In addition to the transport forcing, realistic forcing is provided also as time series of daily values for radiation and suspended matter concentration. For the atmospheric nitrogen input, a constant load is applied to the entire model domain. More details on the model set-up and the model forcing can be found in Lenhart et al. (1997) and Lenhart (2001). Except for the Elbe, where the calculated nutrient loads by MONERIS modified by the transfer function (see section *Linking MONERIS and ERSEM by a transfer function*) are applied, the daily nutrient loads are used as calculated by Lenhart and Pätzsch (2001).

Finally, the indicators representing changes within the ecosystem are derived from the ERSEM simulation results. These form the basis for the further assessment within the multi-criteria analysis and for the ecosystem integrity.

Ecosystem integrity

As a measure to describe ecological impacts in an aggregated form, a specific interpretation of *ecological integrity*, based on Barkmann and Windhorst (2000) and Barkmann et al. (2001), is used. Ecological integrity aims to describe the relationship between the use of ecosystem services and ecological risks endangering the capacity of ecological systems to provide these services. Operationally, ecological integrity can be defined as the guarantee that those processes at the basis of ecosystem self-organising capacity are protected and kept intact. The general concept of the chosen approach and the role of scenarios are illustrated in Fig. 11.

Based on ecosystem theory, exergy capture, cycling of elements, storage capacity, heterogeneity (diversity) and matter losses are important elements of ecosystem functions. With respect to eutrophication processes, these indicators can be modelled taking state parameters as proxies (Mueller et al. 2000; Windhorst et al. 2004),

Fig. 9 Schematic overview of the ecosystem model ERSEM. The rectangles represent modules which are connected by inter-compartmental fluxes (arrows). The circles indicate the prescribed forcing (Lenhart 2001)

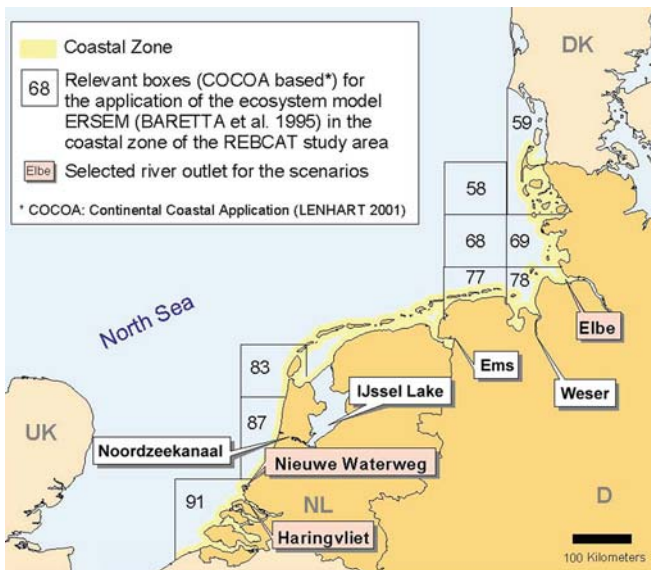
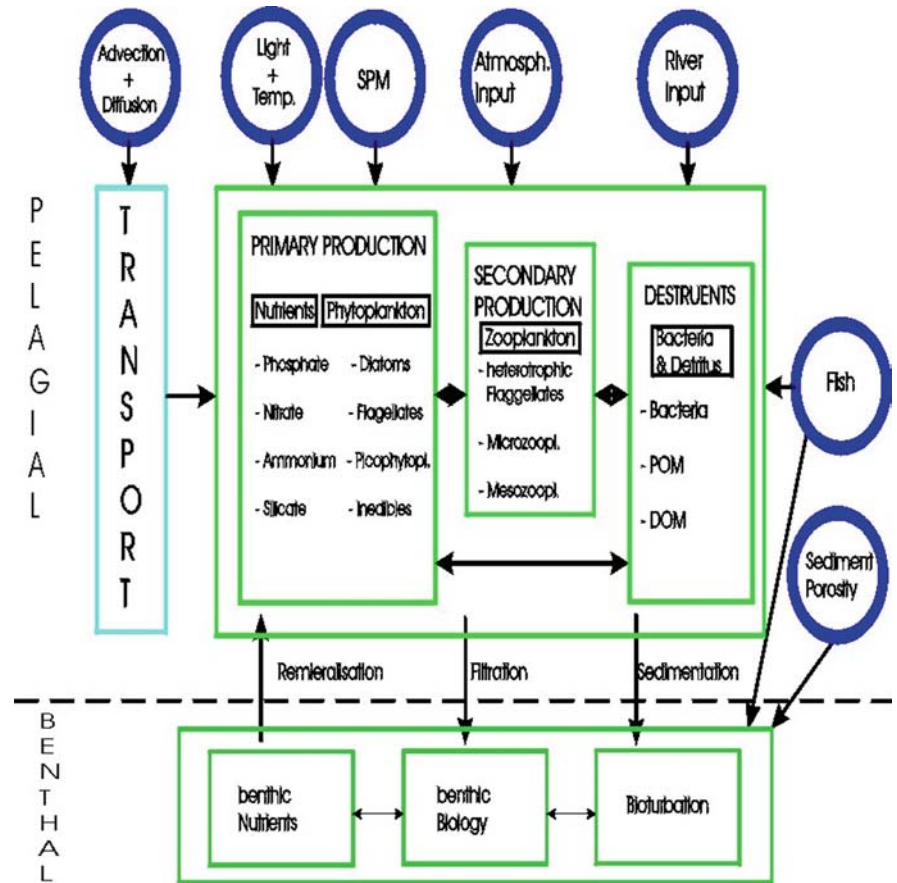


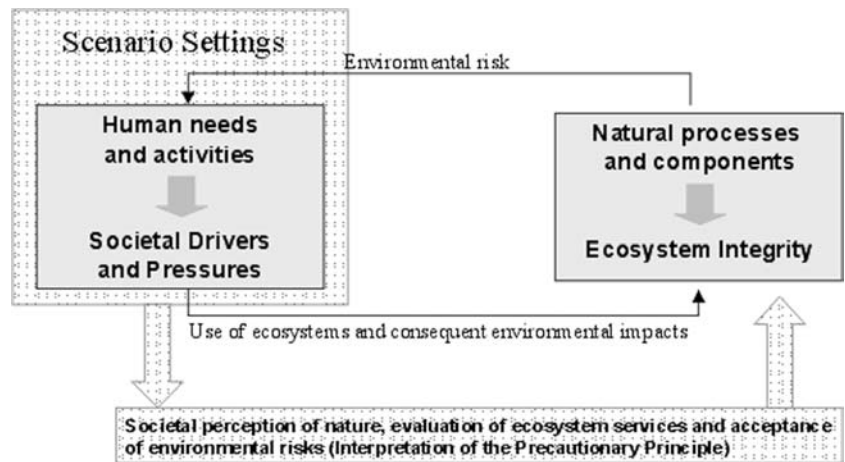
Fig. 10 The Elbe river basin and the coastal zone with the relevant model boxes of ERSEM calculations. Note that the integrated Elbe box comprises the boxes 58, 59, 68, 69, 77 and 78. The boxes 83, 87 and 91 are related to the Rhine and were not considered in this study

and aggregating them into comparable amoeba which describe the ecosystem integrity and ecological impacts on the coastal system.

The selection of “exergy capture” as an indicator for ecosystem integrity stems from the “non-equilibrium principle” as formulated by Kay (2000) and Jørgensen (2000). In coastal zones beneath solar radiation, also energy flows coupled with organic and/or inorganic nutrient inputs from the atmosphere or from adjacent regions have to be taken into account. Another important process to enhance the self-organising capacity of ecosystems is their tendency to (re)cycle limiting substances, especially nutrients. The availability of limiting nutrients and energy depends on the storage capacity, the exchange rate of the pools, and the possibility to temporarily dampen or buffer external inputs. To which extent ecological systems are able to utilise this storage capacity depends on the heterogeneity and especially on the biotic diversity of the system. Finally, matter losses reduce the capacity of primary and secondary production, which are essential functions of ecosystems.

These conceptual elements need to be linked with models in order to assess state parameters which allow the description of impact indicators within this frame-

Fig. 11 Role of scenarios in man and biosphere interactions



work. The approach outlined here to select indicators based on ecosystem theory is in accordance with the present discussion within international agreements like OSPAR or HELCOM, concerning the application of the ecosystem approach in environmental policy making.

CENER model

The CENER (Cost-Effective Nutrient Emission Reduction) model is used to calculate the cost-effective joint N and P emission reduction policy in the Elbe river basin which achieves a desired reduction in the load to the German Bight. The outcome of the CENER model is used as an input to the MCA analysis (section *Multi-criteria analysis*), namely the cost of each policy alternative. The CENER model simultaneously considers diffuse emissions from farms and point emissions from wastewater treatment plants in the WFD regions in the catchment and nutrient retention by wetlands. Besides a differentiation between nitrogen and phosphorus emissions in the model, a further differentiation is made between (technical) measures and quota restrictions to reduce diffuse nutrient emissions.

More technically, the (quadratic programming) CENER model can be written in matrix form, as follows:

$$\begin{aligned} & \text{Minimise}_{\mathbf{X}} && \mathbf{f}\mathbf{X} + \mathbf{X}^T\mathbf{H}\mathbf{X} \\ & \text{such that:} && \mathbf{A}\mathbf{X} \leq \mathbf{b} \\ & && \mathbf{L}\mathbf{B} \leq \mathbf{X} \leq \mathbf{U}\mathbf{B} \end{aligned} \quad (1)$$

Here \mathbf{X} is the vector of nutrient emission reductions, differentiating among (nitrogen and phosphorus) emission reduction at diffuse and point sources in various WFD regions, through technical measures and quota restrictions. \mathbf{X}^T is the transpose of \mathbf{X} . $\mathbf{L}\mathbf{B}$ and $\mathbf{U}\mathbf{B}$ are the lower and upper bounds of \mathbf{X} respectively. \mathbf{H} is a matrix with quadratic cost parameters; \mathbf{f} is the vector with linear cost parameters. \mathbf{A} is a matrix with inequality constraints, to guarantee that the desired load reduction is

achieved and to account for the interaction between technical measures and quota restrictions. Vector \mathbf{b} contains the upper bounds, namely the load reduction targets and the technical reduction potential (for a more detailed description of the CENER model, refer to Lise and Van der Veeren 2002 and Lise 2003).

Multi-criteria analysis (MCA)

The objective is to compare the different strategies for pollution abatement in catchments and to trade-off the costs of these strategies against the benefits to be enjoyed in adjacent coasts, in order to suggest a ranking of the considered strategies. Many of these benefits are economic ones, for example, greater income to the recreation or fisheries sectors. Other benefits may have economic implications but are not economic in themselves, for example, improved ecological quality and biological diversity, or reduced risk of adverse impacts on human health. This means that the evaluation of different abatement strategies involves the comparison of different types of effects measured in different units on different measurement scales.

All decision processes are based on a comparison of alternatives. In EUROCAT the comparison is based on a large amount of information as well as different types of information on the abatement strategies (e.g. monetary values, flows, qualitative assessments). Multi-criteria analysis (MCA) is used to manage the information on the abatement strategies, to facilitate a comparison and ranking of these strategies, and to support decision-making. There are many different types of evaluation methods (Janssen 1992). The choice of a method depends on the characteristics of the decision problem. In EUROCAT a combination of multi-criteria analysis (MCA) and graphical evaluation is used because these methods can handle decision problems with the following characteristics:

- there are multiple objectives (e.g. costs, environmental quality, social);

- these objectives are measured in different units (e.g. M€, t year⁻¹);
- there is a finite number of alternatives to compare;
- it is required to rank alternatives, reject inferior alternatives, and/or choose the best alternative.

Multi-criteria methods transform the input, performance scores and weights, to a ranking using a decision rule specific to that method. The MCA method “weighted summation” is a good candidate to use in EURO-CAT. Weighted summation is theoretically well established, can be easily explained, and is easy to use (Janssen 1992; Janssen et al. 2001). There is therefore less chance that the user will view the method as a “black box”. The formula used for weighted summation is:

$$\text{score}(a_j) = \sum_{i=1}^N w_i \times \hat{s}_{ij} \quad (2)$$

where A is the set of alternatives with a_j ($j=1..M$), C is the set of effects with c_i ($i=1..N$), s_{ij} is the score or alternative a_j for effect c_i , \hat{s}_{ij} is the standardised score of alternative a_j for effect c_i , and w_i is the weight of effect c_i .

The result is a set of rankings linked to the priorities of the various stakeholders involved (section *The DPSIR approach*) Because priorities differ, it is not expected that one strategy is best for all stakeholders. Rather than producing the best alternative, the MCA provides insight in the relation between priorities and alternatives. The results are presented to the stakeholders for feedback. This feedback has been used to improve the abatement strategies.

Results

Outline of the socio-economic scenarios

Three scenarios have been assessed, each giving particular relevance to one of three aspects which play a relevant role in the present environmental and socio-cultural situation, namely (1) market liberalisation and the related consumerism and globalisation; (2) central leadership of the EU, and (3) as a countertrend to globalisation, a trend towards regionalized and self-based economy and life. The time horizon of all scenarios spans the period 1995 (reference year) to 2025. Scenario 1 (*business as usual*) and 3 (*deep green*) pin down the outer limits of plausibility, thus delineating extreme borders, while scenario 2 (*policy targets*) illustrates a middle position, representing somehow the most likely future (Nunneri et al. 2002). The storylines are briefly reported below.

1. *Business As Usual scenario (BAU)*: the present trends towards globalisation and resource exploitation are projected into the future. Priority is given to economic growth. Unrepressed individual needs lead to increased consumerism. This, combined with the

globalisation trend, causes an increase of transportation, low-cost productive processes and free markets. People are short-term planners, who give nature exclusively an aesthetic and use value. In such a risk-inclined society, environmental targets are laxly implemented and enforced.

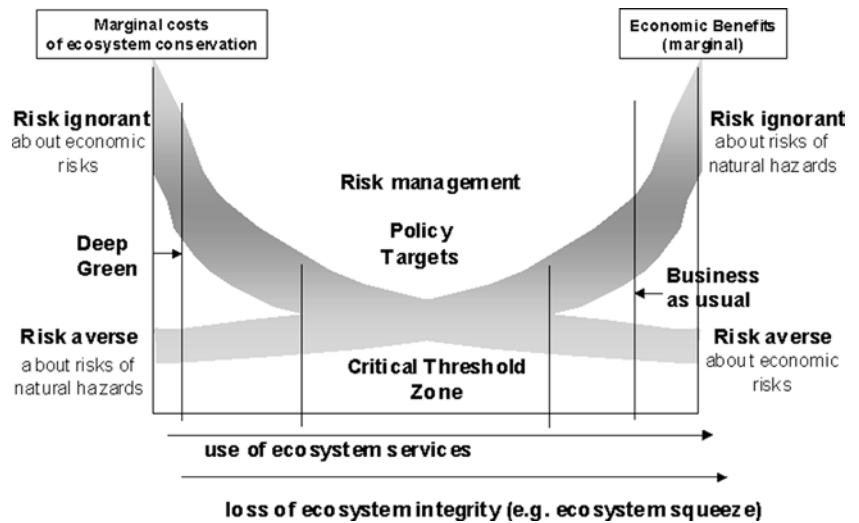
2. *Policy Target scenario (PT)*: the European Union (EU) gains leadership in economic, social and environmental policy, promoting a consistent integrative policy. The present trend towards globalisation stops (e.g. restrictions on market liberalisation, restrictions on trade). Increased awareness of environmental vulnerability leads to incremented environmental protection. The EU strongly enforces clear directives and explicit regulations in order to achieve sustainability. People are educated to be respectful of regulations, and are aware of the need for nature preservation, thus embracing the way towards sustainable development for the sake of future generations by seeking an optimum mix of economic development and protection of resources for the future.
3. *Deep Green scenario (DG)*: angered by BSE crises, food contamination scandals (such as the nitrophenol scandal in Germany in 2002), the Elbe flooding and other catastrophes, people turn spontaneously to a “greener” lifestyle, aimed at valorising and protecting the environment. This implies a change in mentality with respect to the present situation. The inversion of the globalisation trend results in regionalized life, characterised by self-supply, mutual help and communitarian values. Priority is given to environmental issues and nature conservation (over-compliance with the WFD). People are long-term, risk-averse planners, who attempt to minimise environmental risk, even at high costs.

The different political issues, lifestyles and social values characterising each scenario will exert pressures on the environment. Following the DPSIR approach, the next level of analysis focuses on several essential fields of social life of the socio-economic system (drivers) which, in turn, depend on the societal value settings characterising different scenarios.

In this context, food demand can be considered one of the most relevant drivers connected to eutrophication. In fact, given the good quality of WWTPs in the Elbe catchment (Reincke, personal communication), diffuse nutrient emissions due to agriculture at the catchment level will strongly depend on the quantity and quality of food demand (Isermann and Isermann 2001).

The position of the described scenarios with respect to interactions between marginal costs of ecosystem conservation and ecological risks is shown in Fig. 12. A high level of self-organising capacity, e.g. ecosystem integrity, is thereby thought to be beneficial as it maximises the possibilities of the ecosystem to provide ecosystem services and, in parallel, minimises the risk that the ecological system fails to provide the minimum level of natural resources needed by human societies. It is addi-

Fig. 12 Principal interactions between marginal costs of ecosystem conservation and ecological risks (Nunneri et al. 2002)



tionally assumed that, with an increasing use of ecosystem services, socio-economic risks decrease as the resource availability increases, which is in accordance with an attitude averting economic risks. In parallel, however, the ecosystem integrity is decreasing as well, causing increasing ecological risks (Windhorst et al. 2004).

As example for ecological risks in coastal zones, the increasing occurrence of anoxic zones could be taken (Niermann Rachor and Albrecht 1983; Niermann 1990). Risk aversion requires the reduced use of ecosystem services, and thus possibilities to reduce the nutrient losses caused by, for example, different land use systems in the catchments could be studied. As these possibilities are connected either with lower yields or with higher technical efforts, it is necessary to keep both economic and ecological risks as low as feasible (Windhorst et al. 2004).

The scenario settings described above are in agreement with those used in other projects like GLOWA Elbe (Global Change in the Hydrological Cycle, <http://www.glowa.org/eng/elbe>) and also reflects, although not entailing climate change, the basic ideas underlying IPCC scenarios A1 and B2 (IPCC 2000). Based on the scenarios described, different strategies for nutrient abatement are considered and evaluated.

The natural background as a “yardstick”

The general objective is to gain insight in the background or natural concentrations of nutrients in the Elbe basin, thus representing the natural reference level. With regard to natural background concentrations, we have adopted the definition of Laane (1992): “*Natural background concentrations are defined as those concentrations that could be found in the environment in the absence of any human activity*”.

Reference conditions for the various ecological components (phytoplankton, phyto- and zoobenthos, macrophytes and fish) in inland waters are related to nutrient concentrations and not to nutrient emissions or

loadings. This reference condition is important to define the optimal concentration (e.g. target levels) under human influence. Based only on this knowledge, management measures can be evaluated. Knowledge of natural background is necessary in relation to the achievement of water quality guidelines, i.e. good ecological water conditions as the basis of reference conditions, i.e. in the absence of human influence. To achieve this aim, it is not sufficient just to model nutrient emissions by considering individual pathways from diffuse and point sources. This allows only a limited analysis of the sources of these emissions; in particular, one cannot quantify the proportion of emissions attributable to agriculture. This is only possible when, in addition, the emissions for natural background conditions, i.e. the quantity of emissions independent of human influence, are modelled.

In relation to the Water Framework Directive (WFD), a “nutrient background scenario” for the study system, based on calculated background concentrations for the determination of reference conditions, is carried out. The general objective is to gain insight into the background or natural concentrations of nutrients in the Elbe basin, thus representing the natural reference level.

In the following, an attempt is made to determine realistic background emissions based on the mean annual discharge conditions for 1993–1997, and the following defined conditions.

- Nutrient inputs from point sources and urban areas are nonexistent. The same applies to inputs from drainage.
- Areas which are agricultural or urban today are considered as woodland.
- With the exception of areas subject to natural erosion (Alpine and foothills), soil input through erosion is ignored.
- There is a surplus of around $5 \text{ kg N ha}^{-1} \text{ year}^{-1}$ of nitrogen emissions to the air over nitrogen deposition under background conditions, which applies to all regions.

Table 2 Phosphorus inputs via various pathways, their contribution to the total input, and their changes for the German part of the Elbe basin during the periods 1998–2000, 1993–1997 und 1983–1987 (Behrendt et al. 2003)

Pathway	Elbe (German part), phosphorus			
	1998–2000	1993–1997	1983–1987	Change
Groundwater	(t year ⁻¹) 720	987	950	–24
	(%) 13.0	13.4	5.2	
Tile drainage	(t year ⁻¹) 159	170	150	6
	(%) 2.9	2.3	0.8	
Erosion	(t year ⁻¹) 2,112	2,189	1,481	43
	(%) 38.2	29.8	8.1	
Surface runoff	(t year ⁻¹) 130	211	100	30
	(%) 2.4	2.9	0.5	
Atmospheric deposition	(t year ⁻¹) 79	79	147	–46
	(%) 1.4	1.1	0.8	
Urban areas	(t year ⁻¹) 1,068	1,161	2,863	–63
	(%) 19.3	15.8	15.7	
<i>Sum diffuse sources</i>	(t year ⁻¹) 4,268	4,797	5,691	–25
	(%) 77.3	65.3	31.2	
Geogenic background	(t year ⁻¹) 411	411	411	0
	(%) 7.4	5.6	2.3	
Diffuse sources agriculture	(t year ⁻¹) 2,710	3,146	2,270	19
	(%) 49.1	42.9	12.4	
Municipal WWTPs	(t year ⁻¹) 1,123	2,383	10,214	–89
Direct industrial discharges	(t year ⁻¹) 132	162	2,349	–94
<i>Sum point sources</i>	(t year ⁻¹) 1,255	2,544	12,563	–90
	(%) 22.7	34.7	68.8	
<i>Sum all sources</i>	(t year ⁻¹) 5,523	7,341	18,254	–70
	(%) 100.0	100.0	100.0	

- The P concentrations in groundwater of all wetlands is the same.
- The ratio of total to dissolved phosphorus concentrations under anaerobic groundwater conditions is 1.5, instead of 2.5.

On the basis of these statements, and using a pathway-related model like MONERIS (section *MONERIS*), it is possible to calculate both nutrient loadings and concentration with background conditions for individual catchments.

For the calculated background concentrations for the determination of reference conditions in relation to the Water Framework Directive (WFD), it has to be borne in mind that the calculated concentrations have a retention factor dependent on the hydrological and morphological conditions in the water bodies. The calculated background values clearly represent an upper limit of expected nutrient concentrations under background conditions. From the estimates of nutrient emissions under background conditions, it is possible to distinguish the proportion of emissions related to human activities, namely agriculture, forestry and urban activities.

Nutrient emissions in the Elbe catchment and their possible reduction

The following results refer to the German part of the Elbe catchment, as they were published recently by Behrendt et al. (2003). The estimation of the nutrient emissions was carried out for 160 different catchment

areas covering the Elbe basin. For all catchments, the same method was applied. All calculations were done by consideration of the different flow conditions within the time periods, and for normalised conditions to detect the changes caused by human activities. The results of the calculations of the nutrient emissions into the German parts of the Elbe river basins are presented in Tables 2 and 3, and Figs. 13, 14 and 15.

The total phosphorus emissions into the Elbe river basin were about 5.53 kt P year⁻¹ in the period 1998–2000 (Fig. 13, Table 2). Compared with the period 1983–1987, the phosphorus emissions were reduced by about 12.7 kt P year⁻¹ or 70%. The target of a 50% reduction of the phosphorus loads into the North Sea was reached. Again, the decrease of phosphorus emissions is mainly caused by a 90% reduction of point sources. The decrease of diffuse phosphorus emissions was larger than for nitrogen, which is caused by a 59% reduction of the emissions from urban areas. In spite of the enormous reduction of phosphorus discharges from point sources, these sources remain the dominant pathway of phosphorus emissions, with 27% in the period 1998–2000. Among the diffuse pathways, emissions by erosion dominate and represent 26% of the total input.

Nitrogen emissions into the Elbe river basin were about 102 kt N year⁻¹ in the period 1998–2000, and thus 128 kt N year⁻¹, or 56% lower than in the period 1983–1987 (Fig. 14, Table 3). The target of the 50% reduction of nitrogen loads from Germany into the North Sea was probably achieved only within the catchment area of the Elbe River. The main cause for

Table 3 Nitrogen inputs via various pathways, their contribution to the total input, and their changes for the German part of the Elbe basin during the periods 1998–2000, 1993–1997 und 1983–1987 (Behrendt et al. 2003)

Pathway		Elbe (German part), nitrogen			
		1998–2000	1993–1997	1983–1987	Change
Groundwater	(t year ⁻¹)	38,910	48,750	60,770	-36
	(%)	38.0	36.3	26.3	
Tile drainage	(t year ⁻¹)	24,840	26,700	50,300	-51
	(%)	24.3	19.9	21.8	
Erosion	(t year ⁻¹)	3,460	3,650	2,650	31
	(%)	3.4	2.7	1.1	
Surface runoff	(t year ⁻¹)	450	600	630	-29
	(%)	0.4	0.4	0.3	
Atmospheric deposition	(t year ⁻¹)	3,970	3,060	6,550	-39
	(%)	3.9	2.3	2.8	
Urban areas	(t year ⁻¹)	9,370	10,130	13,680	-32
	(%)	9.2	7.6	5.9	
<i>Sum diffuse sources</i>	(t year ⁻¹)	<i>81,000</i>	<i>92,890</i>	<i>134,580</i>	<i>-40</i>
	(%)	<i>79.2</i>	<i>69.3</i>	<i>58.3</i>	
Geogenic background	(t year ⁻¹)	11,970	11,970	11,970	0
	(%)	11.7	8.9	5.2	
Diffuse sources agriculture	(t year ⁻¹)	55,690	67,730	102,380	-46
	(%)	54.4	50.5	44.4	
Municipal WWTPs	(t year ⁻¹)	14,980	32,230	49,340	-70
Direct industrial discharges	(t year ⁻¹)	6,310	9,000	46,760	-87
<i>Sum point sources</i>	(t year ⁻¹)	<i>21,290</i>	<i>41,240</i>	<i>96,090</i>	<i>-78</i>
	(%)	<i>20.8</i>	<i>30.7</i>	<i>43.7</i>	
<i>Sum all sources</i>	(t year ⁻¹)	<i>102,290</i>	<i>134,130</i>	<i>230,670</i>	<i>-56</i>
	(%)	<i>100.0</i>	<i>100.0</i>	<i>100.0</i>	

the decrease of the nitrogen emissions into the river systems was the large reduction of nitrogen discharges from point sources, by 78%. The estimated decrease of diffuse emissions was only about 40%. The inputs via groundwater (38%) and tile drainages (24%) are the dominant pathway in the period 1998–2000. The share of point sources in nitrogen emissions amounts to about 21%. The contributions of erosion, surface runoff and atmospheric deposition to the total nitrogen input are low and amount to about 4% only for each of these pathways.

The comparison of nutrient emissions in the period 1983–1987 and 1998–2000 shows a reduction of the total

amounts, and also distinct displacements from point sources to diffuse sources. With regard to diffuse phosphorus emissions, the pathway of erosion (+43%) and overland flow (+30%) gain more importance, while the proportions from urban areas (-63%), atmospheric deposition (-46%) and groundwater (-24%) are still decreasing. In the past years, the point sources obtained the highest reduction potential, with a strong decrease of point sources like WWTPs (-89%) and industrial inputs (-94%).

Also the nitrogen emissions reveal similar trends, with an increasing importance of erosion (+31%). Groundwater and drainages are still the main contrib-

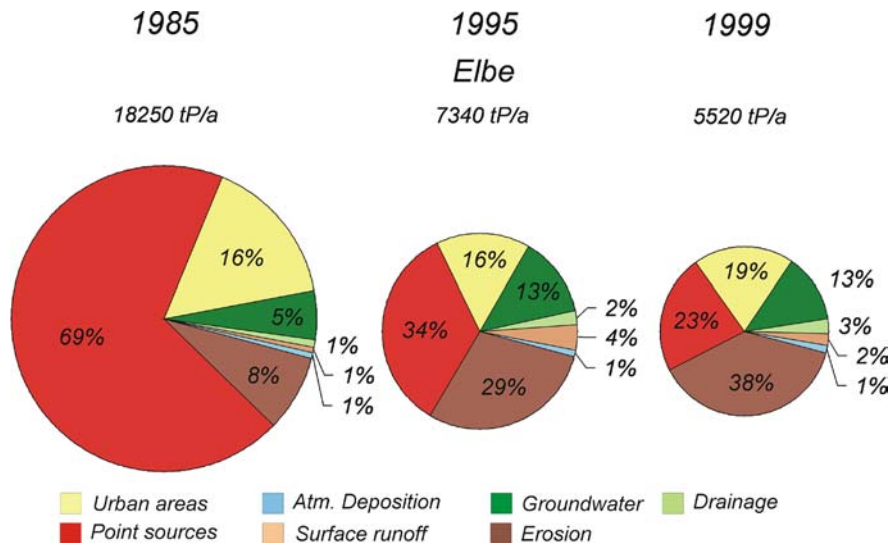
Fig. 13 Phosphorus inputs in the German part of the Elbe catchment for the years 1985, 1995 and 1999 (periods 1983–1987, 1993–1997 and 1998–2000; Behrendt et al. 2002a)

Fig. 14 Nitrogen inputs in the German part of the Elbe catchment for the years 1985, 1995 and 1999 (periods 1983–1987, 1993–1997 and 1998–2000; Behrendt et al. 2002a)

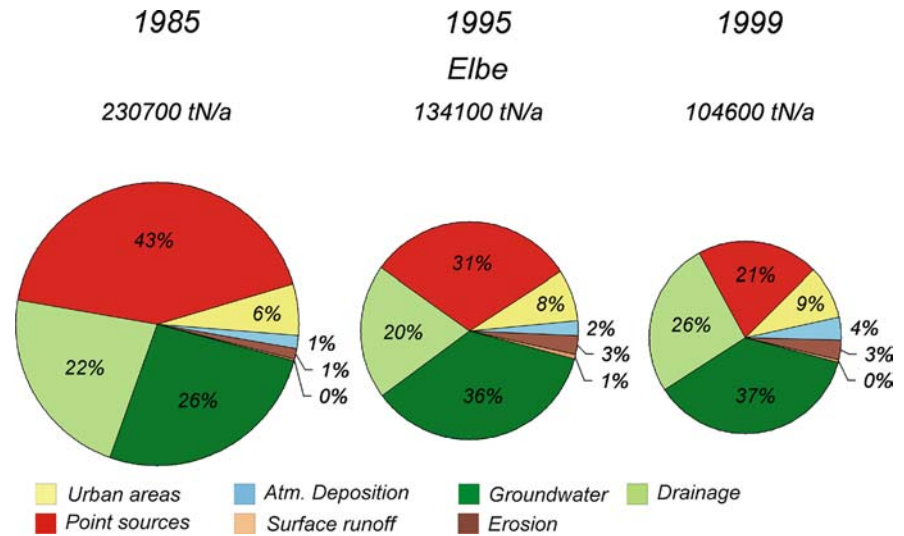
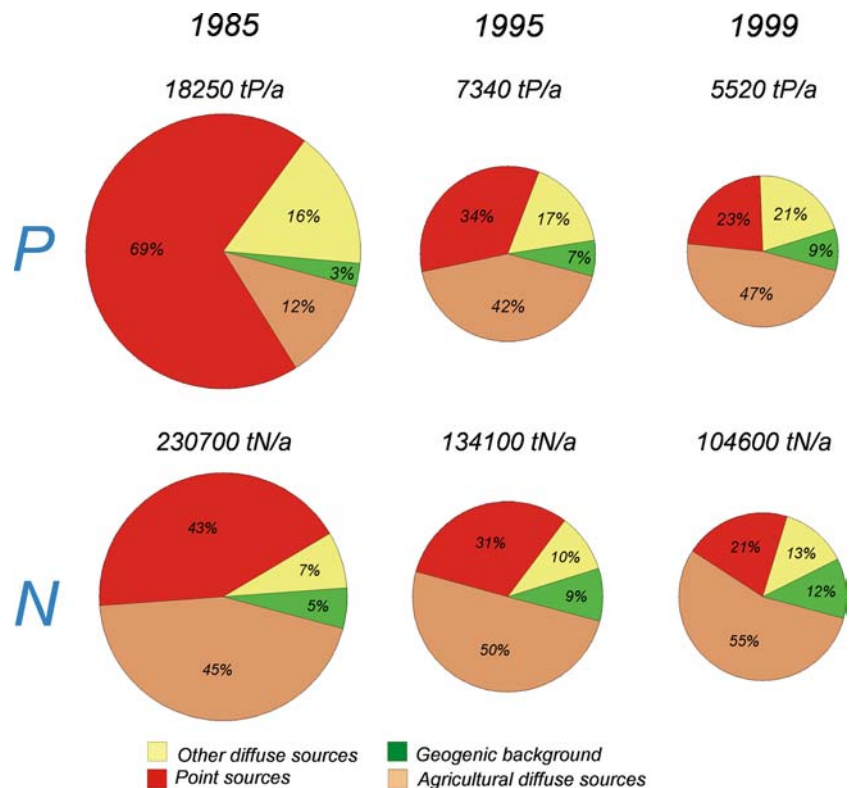


Fig. 15 Causes of phosphorus and nitrogen inputs in the German part of the Elbe catchment for the years 1985, 1995 and 1999 (periods 1983–1987, 1993–1997 and 1998–2000; Behrendt et al. 2002a)



utors, even though the decrease is -36 and -51% respectively.

The main part of all diffuse emissions is caused by agriculture (Fig. 15). Thus, the models applied have to consider in detail the agricultural activities in the catchment. The nutrient surplus in agricultural areas is one of the most important factors. The regionalization of nutrient surpluses shows that the P surplus is in general $2\text{--}4 \text{ kg P ha}^{-1} \text{ year}^{-1}$; only some areas in the tidal Elbe show lower values of $< 2 \text{ kg P ha}^{-1} \text{ year}^{-1}$.

The N surplus is in general $40\text{--}60 \text{ kg N ha}^{-1} \text{ year}^{-1}$, with higher values in the tidal Elbe of about $80\text{--}100 \text{ kg N ha}^{-1} \text{ year}^{-1}$, and even up to $120 \text{ kg N ha}^{-1} \text{ year}^{-1}$.

Caused by the political changes in 1989/1990, the reunification of Germany and structural changes in agriculture, the N surplus could be reduced during the period 1990–1993 to the level of the 1950s. Since then, the N surplus is slowly increasing again. In the past years of the last century, the level of N surplus remained constant, with values of approximately $60 \text{ kg N ha}^{-1} \text{ year}^{-1}$. The comparison of the periods 1983–1987 and 1998–2001 shows that in most parts of the Elbe basin the surplus of nitrogen could be reduced by 40–60%.

Based on both past reduction patterns and on expert knowledge, a set of measures was chosen to reduce the nutrient emissions. In cooperation with scientists of the

FAA Forschungsgesellschaft für Agrarpolitik und Agrarsoziologie e.V., Bonn (Dr. H. Gömann, homepage <http://www.faa-bonn.de>), the measures and their embedding in the agricultural policy framework for the scenarios were harmonised.

The measures for the different scenarios are as follows:

- reduction of tile drained areas,
- application of conservative tillage in agriculture to avoid soil erosion,
- introduction of P-free detergents in the Czech Republic,
- all particulate sewage from population not connected to sewers is transported to WWTPs,
- increase of storage for combined sewers,
- WWTP emissions correspond to EU wastewater guidelines,
- introduction of microfiltration in all WWTPs larger than 100,000 popequiv (population equivalent), and
- transfer of wetlands (according to CORINE land use) to effective retention areas.

According to different options of the possible future development in agricultural politics (Gömann et al. 2003), the application of MONERIS can estimate the total amounts and the contribution of various pathways within a wide range of possible nutrient reductions. As an example, Fig. 16 and Fig. 17 show different reduction possibilities for total nitrogen emissions (for explanation of abbreviations, see Table 4).

These possible reductions can be compared with past situations in three time sections (1985 = mean of the period 1983–1987, 1995 = mean of the period 1993–1997, and 1999 = mean of the period 1998–2001) and the natural background conditions (last row in Figs. 16 and 17). Since the interface between both models (MONERIS and ERSEM) allows only a certain degree of complexity, the assumptions had to be simplified to run the scenarios. Therefore, the assumptions and measures for each scenario were defined as given in Table 5.

Based on the assumed measure packages, the possible future TN and TP emissions and loads were estimated with MONERIS for the time horizon 2025. The outcome of these estimations can be used as reduction targets for the application of ERSEM. The values given in Table 6 are expressed as percentage compared to the river loads for the year 1999.

Thus, in the Elbe a 28% reduction of the load of total nitrogen can be expected if the measures of the policy target scenario (PT) are implemented. The PT scenario would be already sufficient to fulfil the OSPARCOM target of 50% reduction for the Elbe. If the measures of the green scenario were implemented, the reduction of the TN load in the Elbe can be about 36%, and the total load is then lower than 100 kt N year⁻¹.

As a next step, the effect of the simulated river load reductions to the coastal waters were computed by ERSEM (section *Effects of simulated river load reductions to the coastal waters*). Different policy options and

their costs are then analysed by applying the multi-criteria analysis (MCA, section *Ranking of policy options by applying MCA*).

Effects of simulated river load reductions to the coastal waters

In this section, the results of the ERSEM simulation for the Elbe region (Fig. 10) are presented as time series of important parameters for the aggregated Elbe box as well as for the single box 78, where the Elbe load is applied. For each scenario, selected river load reductions as described in the definitions of BAU, PT, DG and pristine conditions are applied.

In Fig. 18, the DIP concentration is presented for the different scenarios, aggregated onto the Elbe box. The time series for the scenarios BAU, PT and DG show a decrease in the winter concentrations in comparison to the time series for the standard run, depending on the degree of load reduction. With the additional reduction towards the pristine conditions, the winter concentration also reflects a further reduction. The winter concentrations are nearly kept on the level at the beginning of the year in the period between January and March. In April, with the onset of the spring bloom, the DIP concentrations drop drastically for all scenarios, resulting in a low level during June and September which all scenarios reach, no matter how strong the reduction of the river load is. One can state that for the aggregated Elbe box, there is a phosphate limitation in this period which is reached for all scenarios. In contrast, the DIN time

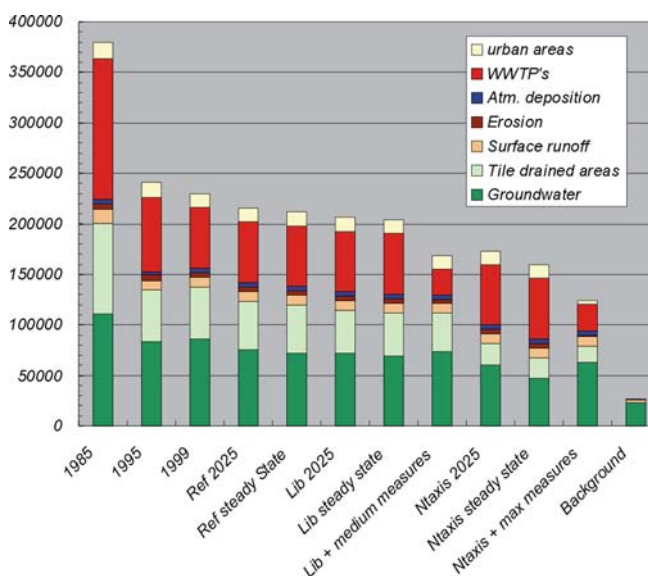


Fig. 16 Total nitrogen emissions (t N year⁻¹) via various pathways of the Elbe catchment for different options of the possible future development in agricultural politics (for explanation, see Table 4) compared to the time series 1985, 1995, 1999 and the natural background (Behrendt 2002a)

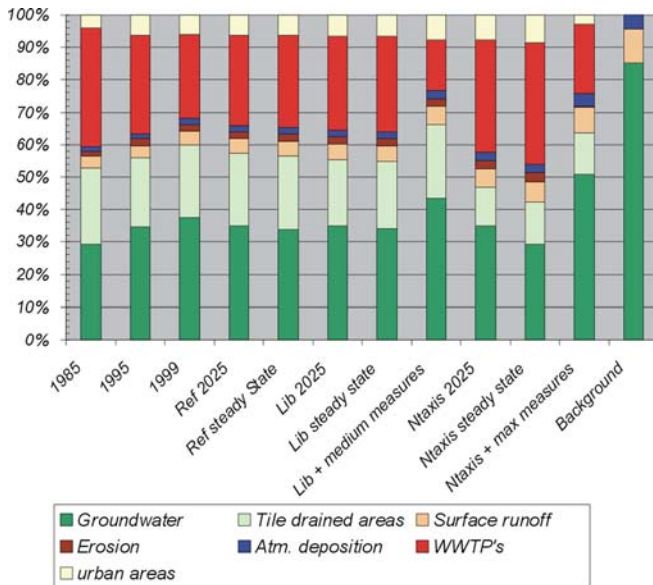


Fig. 17 Percentage of various pathways in relation to the total nitrogen emissions (t N year^{-1}) of the Elbe catchment for different options of the possible future development in agricultural politics (for explanation, see Table 4) compared to the time series 1985, 1995, 1999 and the natural background (Behrendt 2002a)

series for the aggregated Elbe box (Fig. 19) show a clear separation for the different scenarios, without any matching of the lines.

All the scenario time series give nearly parallel lines with a bigger distance towards the pristine condition time series, which results in the much stronger reduction in the applied river load. Generally, one can state that only the pristine condition time series may have reached the level where nitrogen could become limiting for

primary production. For all other scenarios, the DIN concentrations never reached a limiting level.

In the resulting chlorophyll-a concentrations, there is hardly any change towards the differences in the available nutrient concentrations within the time series for the aggregated Elbe box (Fig. 20). The time series in the input box 78 show a decrease in the level of the second peak in the spring bloom (Fig. 21). When searching for effects on the level of individual algae groups, one can see that this reduction in the spring peak is related to a decrease in the flagellate concentration in box 78 (Fig. 22).

In contrast, only the diatom time series for the pristine condition scenario in box 78 (Fig. 23) shows even sporadically higher values compared with all other scenarios, and even with the standard run. This is an interesting finding, since the increased production due to eutrophication was mainly based on flagellate production. Now the model shows that for the reduction of the Elbe load towards pristine conditions, the diatom concentration can be increased. Therefore, the reactions of both phytoplankton groups express a clear decrease in the eutrophic state of the coastal zone.

In a second step, the effects of the reduction scenarios on the coastal environment will be demonstrated on key parameters or indicators which are selected with the focus on reflecting the changes in the ecosystem analysed for the aggregated Elbe box (Table 7). The parameters taken here have already been used in the ASMO modelling workshop (OSPAR 1998); others are in discussion within the OSPAR activities in order to represent problem areas in relation to eutrophication.

The effects of nutrient reductions can first be analysed with respect to the winter concentrations of inorganic nutrients, as these show how much influence the river

Table 4 Explanation of abbreviations in Figs. 16 and 17

Abbreviation	Explanation
1985	Mean value of the total N emissions during the time period 1983–1987 (Behrendt et al. 2000)
1995	Mean value of the total N emissions during the time period 1993–1997 (Behrendt et al. 2002a)
1999	Mean value of the total N emissions during the time period 1998–2001
Ref 2025	Status of N-surplus in 2020/2025 if the recent agricultural policy will be continued (Ref=reference scenario GLOWA-Elbe; Gömann et al. 2003; Global Change in the Hydrological Cycle http://www.glowa.org/eng/elbe)
Ref steady state	Status of N in groundwater at steady state which corresponds to after 2050 continued
Lib 2025	Expected N surplus if agriculture will be driven by global market conditions (Lib = liberalization scenario GLOWA-Elbe; Gömann et al. 2003; Global Change in the Hydrological Cycle http://www.glowa.org/eng/elbe)
Lib steady state	Status of N in groundwater at steady state which corresponds to after 2050 continued
Lib + medium measures	Lib2025 and additionally: 10% reduction of tile drained area, 50% of arable land will be cultivated without plough, detergents with P will be replaced by P-free detergents in CZ, 50% storage for combined sewers (50% storage corresponds to 11.6 m^3 storage volume per ha paved urban area), all WWTPs are in agreement with the EU wastewater guideline
N taxes 2025	200% additional taxes for mineral nitrogen fertilizers
N taxes steady state	Status of N in groundwater at steady state which corresponds to after 2050 continued
N taxes + max. measures	N taxes scenario and additional the maximal variant of all scenarios will be established (caution: that is especially for point sources not identical with the results of the individual scenarios, because in SC11 and SC12 the effluent load is determined by the effluent concentration, and not by changes of the inputs to the WWTPs)
Background	Expected nutrient loads if man had never been in the catchment, and the whole catchment were covered by forest

Table 5 Assumptions and measures for MONERIS computations for different scenarios in the Elbe basin. Note that the assumptions for N surplus are based on the GLOWA-Elbe scenarios (Gömann et al. 2003; Global Change in the Hydrological Cycle <http://www.glowa.org/eng/elbe>)

Scenario	Measures
Business as usual (BAU)	The present situation regarding the drivers is continued to 2025. N surplus will be for all sub-catchments of the same level as 1999. Because of the high residence time of groundwater, the present state of nitrogen surplus is assumed to be for all sub-catchments of the same level as of 1999
Policy targets (PT)	Reduction of N surplus (cf. BAU scenario) in agriculture by 10% (implementation of the Nitrate Directive). Point sources: EU wastewater guideline is fulfilled for all WWTPs (it is assumed that this corresponds with the German values of the inhabitant-specific P emissions from WWTPs which is 0.14 g P inhabitant ⁻¹ day ⁻¹ , and the Dutch inhabitant-specific N emission of 1.7 g N inhabitant ⁻¹ day ⁻¹ . Reduction of P and N emissions from point sources in the Czech Republic by 50%. Increase of storage volume for combined sewer systems (CSS) by 50% (the mean storage volume was set to 11.5 m ³ ha ⁻¹ paved urban for all areas where this was not already 23 m ³ ha ⁻¹ paved urban area). Application of conservative tillage in agriculture on 50% of the arable land. Reduction of tile drained areas by 10%
Deep green (DG)	Reduction of N-surplus (cf. BAU scenario) in agriculture by 30% (implementation of a strong N tax). Increase of storage volume for combined sewer systems (CSS) by 100% (equals a storage volume of 23 m ³ ha ⁻¹ paved urban area). Application of conservative tillage in agriculture on 75% of the arable land. Implementation of P-free detergents in the Czech Republic. Reduction of tile drained areas by 20%. All particulate sewage from population not connected to sewers is transported to WWTPs. All wetland areas (according to Corine 346 km ²) are additional to the surface waters used for retention; 0.5% of agricultural area (according to Corine 450 km ²) is transferred to retention areas

loads have on a specific geographical area (OSPAR 1998). In Table 7, inorganic nutrients are shown in form of the *mean winter DIN concentration* in mmol N m⁻³, and the *mean winter DIP concentration* in mmol P m⁻³, which also reflects the fact that the silicate load is not reduced within the reduction scenarios. For the present analysis, the winter period is defined to start on the 1 January and to end on the 31 March.

The *mean winter DIN/DIP ratio* is calculated as (NO₃ + NH₄)/PO₄, again for the winter period (January to March). Elevated DIN/DIP ratios indicate higher potential for negative side effects, like toxic algae blooms or the growth and colony formation of *Phaeocystis*, which is a major producer of foam on the beaches.

Since the silicate loads of the rivers has not increased within the eutrophication process, the *mean winter DIN/Si ratio* and the *mean winter DIP/Si ratio* reflect the relation between the nutrients and the silicate concentration, which was not effected by anthropogenic changes. Based on the scenario runs, a reversed assess-

ment will be carried out. Here the reduced N and P loads will be judged in relation to the unaltered silicate loads for the rivers.

In order to reflect the reaction of the biological system to the changing nutrient availability, a number of parameters were chosen. First, the *timing of spring bloom (week)* represents the week of the maximum of all weekly mean chlorophyll concentrations over the year. Since this maximum chlorophyll-a occurrence represents the spring bloom within the year, the mean value for this week represents the *mean spring chl-a concentrations* in mg chl m⁻³. In addition, the standing stock of phytoplankton over the summer period from May to August

Table 6 Reduction targets for different scenarios in the Elbe basin expressed as percentage compared to the river loads from the year 1999 (e.g. the total nitrogen river load has to be reduced by 90% to reach the pristine conditions). According to Behrendt et al. (2003), the 10% level of the 1995 nutrient load of the Elbe represents background (pristine) conditions, assuming forests in the whole Elbe catchment

Scenario	Total nitrogen (%)	Total phosphorus (%)
Business as usual (BAU)	20	18
Policy targets (PT)	28	32
Deep green (DG)	36	40
Pristine conditions (PC; without population)	90	90

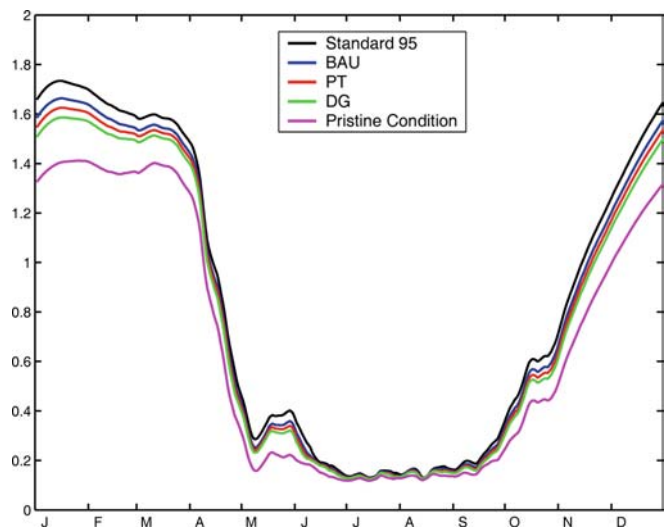


Fig. 18 Time series of DIP concentrations (mmol P m⁻³) related to the aggregated Elbe box for different scenarios (BAU business as usual, PT policy targets, DG deep green) compared to the standard run 1995 and the pristine conditions

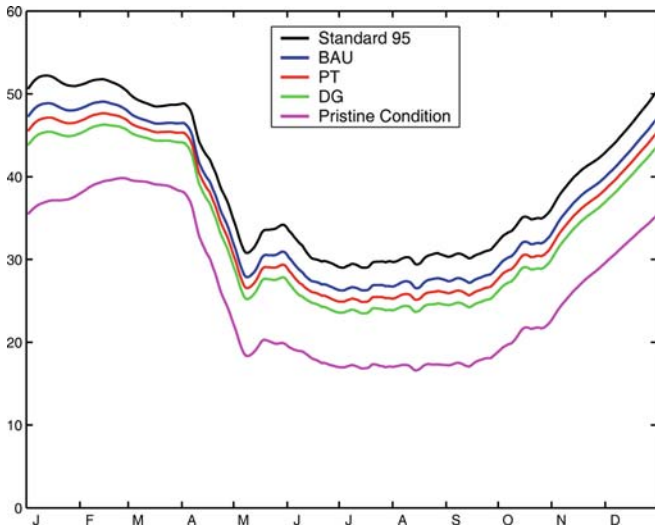


Fig. 19 Time series of DIN concentrations (mmol N m⁻³) related to the aggregated Elbe box for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions

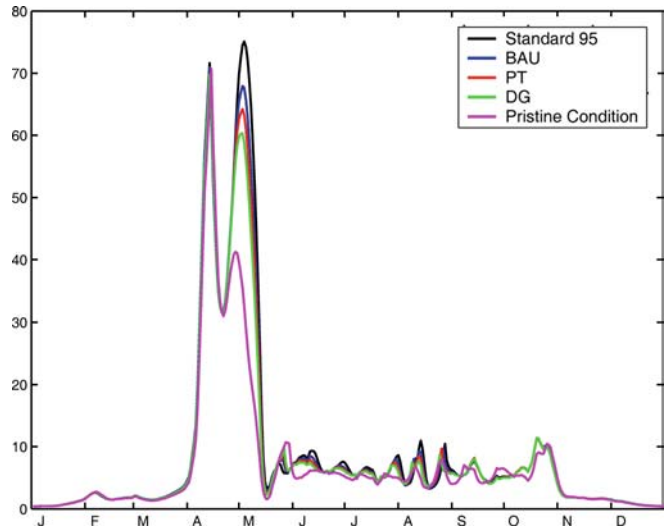


Fig. 21 Time series of chlorophyll-a (mg chl m⁻³) related to input box 78 for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions

is calculated as *mean summer chl-a concentrations* in mg chl m⁻³.

The chlorophyll-a concentration always represents the amount of living phytoplankton, not taking into account the phytoplankton which was subject to natural mortality or grazing by higher trophic levels. Therefore, the parameter *net primary production*, in g C m⁻² year⁻¹, reflects the production of all phytoplankton which has been present during the year.

The eutrophication process did not lead to an overall increase in the algae biomass, but to a major increase of flagellates, while the diatoms remained on a lower level. One indicator for a successful change in the coastal region based on reduced river nutrient loads would be a

response in a decreasing flagellate abundance. Therefore, the *diatom/non-diatom ratio* is calculated as a parameter for the whole productive period between April and September.

Assessment of the ecosystem integrity

Interpreting the data calculated with ERSEM, it should be realised that, during the reduction scenarios, only the nutrient input of the river Elbe has been reduced, while the nutrient load of the other tributaries to the North Sea has been kept constant at the 1995 level. This explains to a certain extent why even drastic reductions of

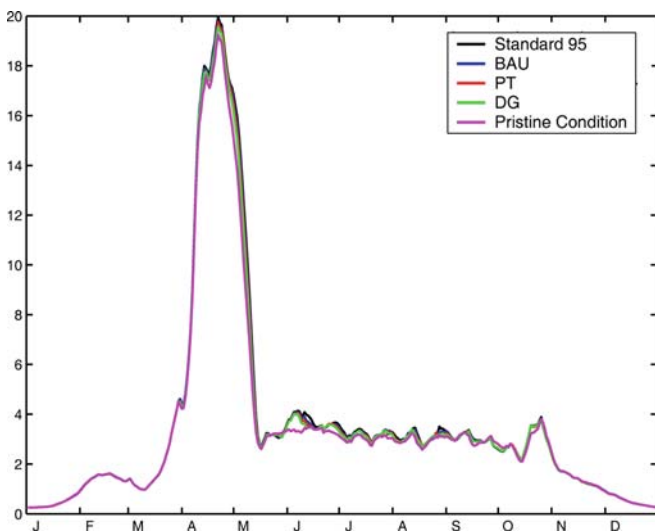


Fig. 20 Time series of chlorophyll-a (mg chl m⁻³) related to the aggregated Elbe box for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions

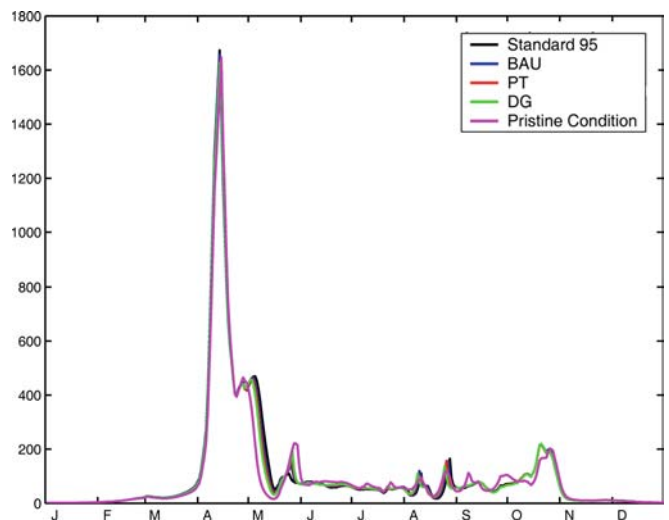


Fig. 22 Time series of flagellate concentration (mg C m⁻³) related to input box 78 for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions

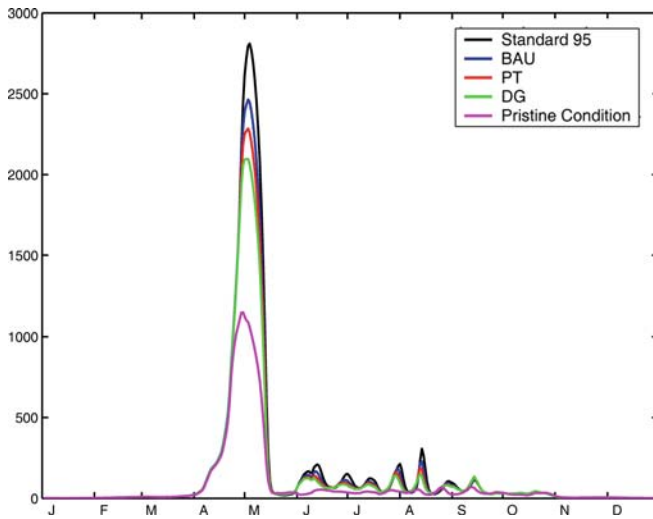


Fig. 23 Time series of diatom concentration (mg C m^{-3}) related to input box 78 for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions

the nutrient loads from the Elbe cause comparatively small changes of the ecological parameters in the Elbe box. These results hint at the need to reduce the riverine nutrient load from the other tributaries as well.

The next level of this analysis is to link the concept of self-organising capacity of ecosystems with the data and information available about the ecology in the coastal zone, as response to different levels of human impact. The calculated values from ERSEM to the Elbe box (exergy = net primary production, cycling = turnover of winter nutrients, storage = sediment input – sediment output, heterogeneity = diatom/non-diatom ratio, minimising losses = nutrient output out of the box) mirror that nearly all indicators are sensitive to reduced nutrient loads from the Elbe, but to a different extent. Obviously the interactions of the coastal ecosystem are changing from a linear nutrient reduction to non-linear effects, thus pronouncing the need to analyse the overall functioning of the ecosystem, too (Windhorst et al. 2004).

In order to indicate the influence of these indicators on the self-organising capacity and the relative impact caused by the different reduction scenarios, it is necessary to transform the calculated values into relative values (Windhorst et al. 2004). As a result, in the selected case study the storage function of the coastal ecosystem changes in relative terms more than the other indicators. This confirms the theoretical argumentation that this indicator could reveal essential information about the functioning of the coastal ecosystem. The overall change of the ecological status of the coastal zone is increasing with lower riverine nutrient loads, which goes apart with lower risks of ecological hazards. Also the results allow to indicate an overall ecological benefit, which could be achieved by economic endeavours in the catchment to reduce nutrient losses. Still, under the constraints described by Behrendt et al.

(2002c) and the selected scenarios, in this case the ecological status of the coastal zone would stay, even in the best case—the deep green scenario, far away from the assumed pristine conditions.

Ranking of policy options by applying MCA

In the following, an example of the use of MCA for evaluating management alternatives for nutrient reduction is given. Please note that these reduction percentages differ from the values in Table 6, as we follow the values of Lenhart and Pättsch (2001). In addition, the values here represent changes in the load to the German Bight, and not changes in the load to the Elbe River, as assumed in Table 6. Also, the reference year is 1985, and not 1995.

The three scenarios as formulated in the section *Outline of the socio-economic scenarios* are used as a basis for performing a MCA. In the BAU, no additional measures are taken, while the future evolves autonomously as under the BAU scenario. Then, there is still a reduction of 35% nitrogen and 48% phosphorus in the load to the German Bight in 2025, with respect to the load in 1985. In the medium reduction alternative, which compares to the PT scenario, additional effort is undertaken to reach a reduction of 50% in nitrogen and 65% phosphorus load to the German Bight. Finally, a strong-reduction alternative is considered, where many efforts are focussed on cleaning up the eutrophication problem to such an extent that the nitrogen load is reduced by 70% and the phosphorus load by 75%. This strong-reduction alternative compares to the DG scenario. There are many ways of obtaining the desired reduction in the load. The MCA investigates the plausibility of including high-retention dams and wetlands into the catchment. Such an option does not change the ultimate load reduction, but it may be more attractive from an economic, environmental and social point of view. Summarising, the MCA considers the following five policy alternatives (see also the first two rows of Table 8):

- BAU: no additional reduction measures: –35% N load, –48% P load,
- MR: medium reduction in load: –50% N load, –65% P load,
- MRHR: medium reduction in load, with high-retention possibilities in the catchment,
- SR: strong reduction in load: –70% N load, –75% P load,
- SRHR: strong reduction in load, with high-retention possibilities in the catchment.

The criteria used for the Elbe catchment are divided into three groups which represent the policy objectives of the problem: (1) economic, (2) environmental, and (3) social. The considered variables needed for the MCA are assessed using various models. The total costs are cal-

Table 7 Key parameters related to the standard run and the selected reduction scenarios, analysed for the aggregated Elbe box

Box	Standard 1995	BAU	PT	DG	Pristine condition
Mean winter DIN concentration (mmol N m ⁻³)	51.4	48.5	45.9	45.5	38.1
Mean winter DIP concentrations (mmol P m ⁻³)	1.7	1.6	1.6	1.5	1.4
Mean winter DIN/DIP ratio	30.7	30.9	29.8	29.5	27.6
Mean winter DIN/Si ratio	2.1	1.9	1.9	1.9	1.6
Mean winter DIP/Si ratio×10 ⁻²	6.9	6.6	6.5	6.3	5.7
Timing of spring bloom (weeks)	16	16	16	16	16
Mean spring chl-a concentrations (mg chl m ⁻³)	18.9	18.8	18.7	18.6	18.2
Mean summer chl-a concentrations (mg chl m ⁻³)	4.1	4.3	3.9	3.9	3.7
Net primary production (g C m ⁻² year ⁻¹)	266	259	256	253	234
Diatom/non-diatom ratio	0.37	0.37	0.38	0.38	0.4

culated with the CENER (section *Ecosystem integrity*, Table 7), coastal environmental indicators are based on calculations with ERSEM (Windhorst et al. 2004), recreational and wetland amenities are derived following Brander et al. (2003), while fish catch, tourist visits to the coast, coastal unemployment and sectoral transition are the most uncertain and are validated through expert judgement.

Table 8 shows the N and P load reduction, which is a restriction in the CENER model, the catchment area devoted to nutrient retention, and the costs of emission reduction at diffuse and point sources and wetland/dam construction in the Elbe river basin for the five alternatives, as calculated with the CENER model. The table shows that the costs are not zero in the BAU. In this alternative, 35% N load and 48% P load will be reduced by 2025 in the Elbe basin compared to the load in 1985. The three policy alternatives (BAU, MR, SR) typically have no additional wetlands and therefore no wetland costs. The costs for WWTPs increase from € 252 million to € 437 million per year, because the phosphorus will be maximally reduced, while additional nitrogen reduction through WWTPs is considered too expensive. The main cost contribution is from the reduction of diffuse agricultural emissions without high-retention possibilities. In MR and SR, on average 15 and 40% respectively of the farms need to be closed down. When high retention becomes an option, the percentage for MRHR and SRHR are 0 and 2.5% respectively. Hence, strong reduction with high retention (SRHR) appears to be a valid and achievable option.

The effects are weighed equally among the economic, environmental and social themes (each 1/3, which can be read from the column “weight level 1” in Table 9). The effects are in principle equally weighed within the three

themes. Within the social theme, both amenity values are taken together and weighed equally with coastal unemployment (1/3) and sectoral transition (1/3). The amenity value of wetlands is given a higher weight (2/6) than the amenity value of recreation (1/6), as the total value is much higher than the amenity value of coastal tourism. The weights are not proportional to costs, as there is a difference between the direct use value of coastal recreation and the indirect existence value of wetlands in the catchment. Within the economic theme, the weights on the costs (0.6) are taken much higher than the weights on fish catch (0.175) and tourist visits to the coast (0.225) to represent the importance of costs. We consider the effect of nutrient reduction on tourist visits to the coasts somewhat more important than the effect on fish catch. Within the environmental theme, primary production and biodiversity are joint N and P indicators and therefore get twice as high a weight (0.2) than the other six individual N and P indicators (0.1). The resulting weights can be derived by multiplying the weight level 1 with weight level 2, as presented in the last two columns of Table 9.

The information of Table 9 is used as input into MCA (section *CENER model*) to rank the five policy alternatives, and the result is presented graphically in Fig. 24. This figure can be used to examine the rankings of the alternatives according to each group of criteria separately. Here, one can clearly see that the ranking of the alternatives for the social criteria corresponds to the overall ranking. However, the economic impacts have BAU as the best alternative whereas the environmental impacts have BAU as the worst alternative. Figure 24 clearly shows that economic criteria have to be traded off against environmental criteria. The alternatives which are good for the economy are bad for the envi-

Table 8 Reduction in load, catchment area devoted to nutrient retention, and costs of emission reduction at diffuse and point sources and wetland/dam construction in the Elbe river basin

Policy options	BAU	MRHR	MR	SRHR	SR
Nitrogen load reduction compared to 1985 levels (%)	35	50	50	70	70
Phosphorus load reduction compared to 1985 levels (%)	48	65	65	75	75
Total catchment area devoted to nutrient retention (km ²)	0	1,791	0	8,030	0
Costs (million Euros)					
Diffuse agricultural emission reduction	147	217	1,378	457	5,515
Emissions reduction from WWTPs	252	437	437	437	437
Emission retention in additional dams and wetlands	0	203	0	915	0
Total costs	399	857	1,815	1,809	5,951

Table 9 Effects table on policy options in the Elbe basin, its effects in the German Bight and the used weights

Policy options	C/B	Unit	BAU	MRHR	MR	SRHR	SR	Weight level 1	Weight level 2
<i>Social</i>									
Recreational amenity	B	Million €	5	10.5	10.5	15.3	15.3	0.333	0.111
Wetland amenity	B	Million €	0	39.8	0	128.8	0		0.222
Coastal unemployment	C	%	7.4	6.9	6.9	7.2	7.2		0.333
Sectoral transition		-/+	++	+	-	0	--		0.333
<i>Economic</i>									
Cost (total)	C	Million €	399	857	1,815	1,809	5,951	0.333	0.600
Fish catch		-/+	++	+	+	0	0		0.175
Coastal tourism		-/+ ++	0	+	+	++	++		0.225
<i>Environmental</i>									
Primary production	C	mmol m ⁻³ year ⁻¹	271	263	263	254	254	0.333	0.200
Biodiversity	B	Scale (0–1)	0.64	0.65	0.65	0.67	0.67		0.200
Cycling N	B	year ⁻¹	3.3	3.4	3.4	3.6	3.6		0.100
Cycling P	B	year ⁻¹	6.1	6.3	6.3	6.4	6.4	0.100	
Storage capacity N	C	mmol m ⁻³ year ⁻¹	29	30	30	29	29	0.100	
Storage capacity P	C	mmol m ⁻³ year ⁻¹	1.8	1.7	1.7	1.6	1.6	0.100	
Matter losses N	C	mmol m ⁻³ year ⁻¹	991	929	929	951	951	0.100	
Matter losses P	C	mmol m ⁻³ year ⁻¹	25	24	24	23	23	0.100	

ronment, and vice versa. We conclude from this that the main trade-offs are between economic-related issues and environmental issues.

Discussion and conclusion

The political necessity to implement the European Water Framework Directive (WFD) demands a better understanding of catchment–coastal zone interactions. The main target of this paper is to elaborate on the interaction of activities in the catchment, and their effects on eutrophication in the coastal waters. In case of the Elbe river basin, the most important outcome of our study will be the definition of scenarios and the application of models to evaluate measures in the catchment in order to meet the desired standards in the coastal zone of the German Bight. The effect and cost-effectiveness of these reduction measures will be of significant importance for decision makers such as governmental institutions.

From the modellers' point of view, the linkage of steady-state meso-scale catchment models like MONERIS with dynamic ecosystem models like ERSEM requires the formulation of transfer functions. Through the MONERIS model, the input of nutrients from the catchment to the coastal zone can be described for different point and diffuse pathways, thus allowing to test the efficiency of management measures acting upon different aspects of nutrient inputs. Also the consideration of retention processes in the catchments as well as knowledge of the complex biogeophysical interaction of nutrients in coastal waters and its effects on coastal eutrophication are crucial points. Knowledge of this natural background is necessary to evaluate the effects of driving forces like economical changes in agriculture.

The uncertainty of future socio-economic conditions requires the formulation of possible changes (scenarios). The scenarios presented can describe different policy

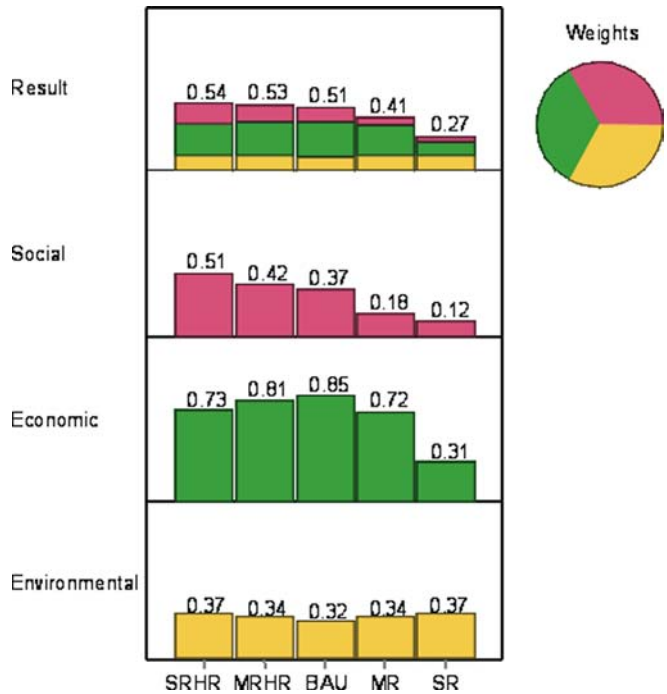


Fig. 24 Ranking of the alternatives according to the three categories. Equal weights: the *stacked bars* represent the weighted contribution of each group to these totals. It follows that if equal weight is given to the three policy objectives, then the strong-reduction high-retention alternative (SRHR) is the preferred alternative. Social criteria contribute the most to the total score of SRHR, while the economic criteria contribute the most to the total score of BAU. SR is the worst alternative. The contribution of the environmental criteria to the total score of BAU is the smallest

contexts as well as environmental objectives, and can investigate their consequences, such as the reduction in fluxes of nutrients. With regard to nutrient inputs in the Elbe and, consequently, to the coastal zone, the main relevant drivers in the catchment are the growth of living standard as well as the related density patterns, energy supply, food supply and wastewater issues. Social behaviour may affect food and energy demand as well as acceptance of policy measures (response).

The outcome of the multi-criteria analysis shows that the strong-reduction alternative with the possibility to construct high nutrient retention basins and dams in the catchment (SRHR) is the highest ranked alternative. Moreover, it can be concluded that three alternatives are quite competitive, namely BAU, MRHR and SRHR, while the options without high retention in the catchment are clearly inferior.

The scenario storylines, as pointed out in the sections *The use of socio-economic scenarios* and *Outline of the socio-economic scenarios*, have been used by the models in two ways. MONERIS and ERSEM translated the scenarios into measure packages in the catchment, and calculated the resulting reduction in the load. CENER and MCA have only one scenario as an indication for the most likely future. On top of that, five policy alternatives were compared based on the intensity of reduction and the possibility of including high-retention possibilities into the catchment. The reduction intensity compares well to the formulated scenarios, in the sense that no additional measures are taken in BAU, while MR and MRHR compare to the PT scenario, while SR and SRHR compare to the DG scenario. This difference is necessary, as decision makers cannot choose between different evolving futures, but they can choose among different reduction intensities and policy packages.

At the moment there are no options (changes in consumer behaviour) for reaching the target, as in the case of a switch to detergents without phosphorus. A price has to be paid, which can be carried by the public through a water charge. The multi-criteria and cost-effectiveness analyses indicate that the cheapest option to reduce nutrients is by constructing high-retention dams and wetlands, with maximal reduction of phosphorus at WWTPs, and where farmers economise on fertiliser use and put their animals on emission-reducing diets.

In the case of this study, it can be shown that the fulfilment of political targets (e.g. 50% reduction of nitrogen (N) and phosphorus (P) emissions, as formulated by the North Sea conference and OSPAR) will not lead necessarily to a similar reduction in primary production in the coastal zone. The targets of nutrient load reductions also need to consider the natural N/P ratio in the German Bight. According to Lenhart (2001), a reduction in the nutrient load by 50% for N and P cannot be linearly transferred to a similar reduction in primary production. With a reduction of the organic and inorganic river loads by 50%, the simulation with the ERSEM model (standard run for the year 1988) showed the maximum effect on the net primary production in a

reduction of about 20% in the coastal zone. Thus, the evaluation of reduction scenarios on the status of coastal waters needs a concept for a "sustainably used" North Sea, and indicators which can describe ecosystem functions and ecosystem integrity. In the sense of the WFD, the estimation of natural background concentrations could be used as a baseline to assess the improvement of water quality.

The application of the model MONERIS allows to distinguish the diffuse pathways of nutrient emissions into the river systems, and the separation of the total emissions and loads into the sources. As a result, wastewater management and agriculture could be identified as the main drivers of nutrient emissions. Furthermore, the model MONERIS provided estimations of the resulting nutrient loads from the rivers to the coastal zone. The impact of these loads on the coastal waters was modelled with ERSEM, and revealed a distinct difference in the resulting time series for the scenarios of DIN and DIP in the Elbe region. While the concentration of DIP shows their strongest differences in winter, the DIN concentrations follow nearly as parallel lines and keep their distance nearly all over the year.

For the DIN time series, only the pristine condition scenario may have reached the level where nitrogen could become limiting for primary production, which is especially related to the usually high nitrogen load of the Elbe River. While the corresponding reaction of the biological parameters in the aggregated box was very low, input box 78 showed a clear reaction towards the reduced nutrient input. For input box 78, the reaction of diatoms and flagellates express a clear decrease in the eutrophic state of the coastal zone. It should also be pointed out that the ERSEM model is parameterised for the simulation of eutrophic conditions, and the result observed for such low nutrient concentrations could be an overreaction of the model. In any case, it would be interesting to define the level of reduction (within a single river or a group of rivers) where the system switches from an increasing diatom concentration towards a decrease related to pristine conditions.

The use of indicators or ecological quality indicators is an ongoing discussion within the OSPAR activities in order to represent problem areas in relation to eutrophication. The simulation runs of ERSEM within EUROCAT can contribute to this discussion by presenting the reaction or responsiveness of the parameters in relation to the scenarios applied. In addition, with the parameters used for the ecological integrity assessment, numerous parameters are derived from fluxes, rather than from the state variables which are usually used. Since the state variables are mainly related to measurements, the use of fluxes could open new insight into the indicator discussion.

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