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Linking Ecological and Socio-economic System Analysis – a Methodological Approach based on Ecological Risk

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Linking
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based on Ecological Risk

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Abstract

The presented study applies the DPSIR approach to the analysis of two environmental issues in the North Sea. The first case-study is eutrophication of coastal waters associated with anthropogenic nutrient emissions in the North Sea catchment area. The second (emerging) issue is offshore wind power generation, driven by the need of mitigating climate change (by cutting greenhouse gases emissions) and reduce energy dependence on third countries. The presented integrated analysis focuses on methodological aspects related to the application of the DPSIR scoping framework and evaluation of its suitability as decision support system (DSS) in coastal management. Moreover, in the context of decision-making under uncertainty, the issue of assessing potential massive changes in ecosystem functioning is approached through the development of the ecological risk concept. The results are discussed under two perspectives: (1) in terms of management possibilities for the single case-studies as well as (2) for evaluating the deployed methodology.

For the case study eutrophication, some options for nutrient emission reduction are presented and evaluated with respect to their economic, ecologic and social effects. It is shown that nutrient emission reduction within the catchments does not correspond to equal reduction of primary production in the coastal zone. Similarly, emission reduction show non-linear relationships with nutrient emission abatement costs (in the catchment) and risk reduction resulting from emission decrease in the coastal zone. In the light of international management of river and sea basins there is a need for a spatially integrated emission-reduction approach, focusing on communication, negotiation and burden sharing.

In the case of the emerging issue of marine-use for offshore wind power generation, the analysis aims at scoping possible aspects needing management. The ecological consequences assessed in this study are limited to effects on the supporting services during wind farm construction phases, while recognising that a more comprehensive analysis of the effects throughout the whole life-time of a project is needed. Model runs show that that offshore wind farm construction can considerably affect ecosystem integrity in the short term (e.g. primary production). When appraising possible offshore wind farm construction scenarios in terms of ecological risk and cost-benefit analysis, perceptions about the risks connected with climate change play a decisive role in assessing trade-offs of different options. The socio-economic valuation of such large-scale energy projects (do they present net benefits or net costs for society as a whole?) depends much on the value given to mitigation of both climate change and marine ecological risks.

This study concludes that the DPSIR approach offers a helpful conceptual framework for providing valuable knowledge to decision-makers. However, for a successful application of the DPSIR scoping framework, scientists (and practitioners) require not only adequate analytical tools but also original team-work for integrating their research into interdisciplinary science. The concept of ecological risk can inform decision-making by addressing uncertainties otherwise not

appraised under the current praxis of project valuation (e.g. cost-benefit analysis), which only includes valuation of marginal ecosystem changes. In this context, the definition of trade-offs (e.g. acceptable levels of risk for society as a whole), calls for 'good' governance based on broader participation. The ecological risk concept presented in this study is a first attempt to address a complex issue. Future research should aim at incorporating a broader set of ecosystem integrity indicators into ecological risk assessment. In addition to this, it would be worth to test non-linear approaches for the operationalisation of ecological risk. Finally, the suitability of the ecological risk concept for communication in the frame of decision-making and negotiation procedures should be evaluated.

Zusammenfassung

In der hier vorgelegten Arbeit wird der DPSIR Ansatz zur Analyse zweier Umweltfragen in der Nordsee angewandt. In der ersten Fallstudie wird die Eutrophierung resultierend aus anthropogenen Nährstoffemissionen in den Einzugsgebieten der Nordsee betrachtet. Die zweite Fallstudie befasst sich mit der aktuell gewordenen Offshore-Windenergieerzeugung, deren Bedeutung sowohl vor dem Hintergrund der durch den Klimawandel gewünschten Reduktion von Treibhausgasemissionen (GHGs) als auch in der Reduzierung der Abhängikeit von Drittländern in Fragen der Energieversorgung zu sehen ist. Die Analyse konzentriert sich auf methodologische Aspekte hinsichtlich der Anwendbarkeit des DPSIR Ansatz und Prüfung seiner Tauglichkeit als Entscheidungshilfe (DSS) im Küstenmanagement. Darüberhinaus wird im Zusammenhang mit Entscheidungsfindungsprozessen unter Unsicherheiten die Frage möglicher, weitreichender Änderungen der Ökosystemfunktionen unter Entwicklung des Konzeptes des ökologischen Risikos betrachtet. Die Ergebnisse werden unter zwei Gesichtspunkten betrachtet: (1) bezüglich der Managementmöglichkeiten in Einzelfallstudien und (2) bezüglich der Bewertung der angewandten Methodologie.

Für die Fallstudie Eutrophierung werden einige Handlungsoptionen zur Reduzierung von Nährstoffemissionen aufgezeigt und hinsichtlich ihrer ökonomischen, ökologischen und sozialen Wirkung bewertet. Es zeigt sich, dass die Reduzierung der Nährstoffemissionen innerhalb der Einzugsgebiete nicht mit einer vergleichbaren Reduzierung der Primärproduktion im Küstengebiet korrespondiert. In ähnlicherweise sind die Reduzierung von Emissionen und die damit verbundenen Implementierungskosten im Einzugsgebiet nicht linear mit dem durch die Verringerung der Emissionen in den Küstenzonen reduzierten Risiko verbunden. Angesichts des internationalen Managements von Flüssen und Seebecken besteht eine zwingende Notwendigkeit räumlich integrierter Emissionsreduzierung mit Fokus auf Kommunikation, Verhandlung und Lastverteilung.

Für die Fallstudie der Offshore-Windenergieerzeugung zielt die Analyse auf die Hervorhebung möglicher Aspekte, die weiterer Betrachtung und Management bedürfen. Die in der Studie bewerteten ökologischen Folgen werden in ihrer Auswirkung auf Unterstützungsdienstleistung (supporting services) während der Bauphase der Windparks beschränkt, wobei eingeräumt wird, dass eine eingehende Betrachtung während des gesamten Lebenszyklus notwendig ist. Modellsimulationen zeigen, dass der Einfluss, den der Bau von Windparks auf die ökologische Integrität hat, auf kurze Sicht beträchtlich sein kann (z.B. Primärproduktion). Bei der Abwägung verschiedener Szenarien für Offshore-Windparks bezüglich des ökologischen Risikos und der Kosten-Nutzenanalyse ist die Wahrnehmung des mit dem Klimawandel verbundenen Risikos bei der Festlegung von Trade-Offs für die verschiedenen Handlungsoptionen von ausschlaggebender Bedeutung. Der sozioökonomische Wert derartig großangelegter Energieprojekte (sind sie für die Gesellschaft als Ganzes als Nettonutzen oder als Nettokosten zu bewerten?) hängt sehr

von der Bedeutung ab, die Klimawandel und Reduzierung des maritimen ökologischen Risikos beigemessen wird.

Diese Studie kommt zu dem Schluss, dass DSPIR ein hilfreiches konzeptionelles Bezugsystem ist, welches Entscheidungsträgern wertvolle Erkenntnisse liefert. Jedoch sollten Wissenschaftler (und Fachleute) berücksichtigen, dass die erfolgreiche Anwendung hinsichtlich der Verwendung geeigneter Analysewerkzeuge und der Integration interdisziplinärer Arbeit in Teams keine leichte Aufgabe darstellt. Das Konzept des ökologischen Risikos kann für die Entscheidungsträger eine wertvolle Hilfe bei der Beurteilung von Unsicherheiten und Risikofaktoren sein, die bei der üblichen Praxis der Projektbewertung (z.B. Kosten-Nutzenanalyse), die nur marginale Veränderungen im Ökosystem betrachtet, unberücksichtigt blieben. Die Definition von Trade-Offs (z.B.: die Definition akzeptabler Risken für die Gesellschaft als Ganzes) macht eine möglichst breite Beteiligung notwendig. Das in dieser Arbeit vorgestellte Konzept des ökologischen Risikos stellt einen ersten Ansatz zur Betrachtung komplexer Vorgänge dar. Zukünftige Forschungsvorhaben sollten eine größere Zahl an Indikatoren für die Ökosystemintegrität bei der Ermittlung des ökologischen Risikos in Betracht ziehen. Darüberhinaus wäre es wünschenswert, verschiedene nicht lineare Ansätze für die ökologische Riskoabschätzung anzuwenden. Schließlich sollte die Eignung des Konzepts des ökologischen Riskos als Grundlage im Rahmen von realen Entscheidungsfindungsprozessen überprüft werden.

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1 Introduction

During the last decades interdisciplinary research has been increasingly applied to environmental issues, based on the relatively recent realisation that humankind depends on its natural environment and human action can significantly influence ecological development and equilibrium (Bossel, 1996). The need of integrated research has been especially emphasised for coastal areas, which are characterised by extreme variability, highly diverse environments and multifunctionality (von Bodungen and Turner, 2001; Bowen and Riley, 2003; Kremer, 2004; LOICZ, 2005). Both natural and (increasingly) anthropogenic processes are discontinuous and highly dynamic in coastal areas: the complexity and variability of interwoven natural and human systems results in multiple causal chains, thus calling for cross-sectoral, integrated analysis (Pentreath, 2000; Antunes and Santos, 1999; Elliott, 2002). Moreover the coexistence of multiple, sometimes conflicting, uses in coastal areas, requires the aggregation of detailed sectoral information (including stakeholder perspectives) into an holistic 'bigger picture' of complex environmental issues, if science is to be an aid in decision-making (IOC, 2006; Karageorgis et al., 2006). Up to date, much effort has been made for integrating the traditionally separated sectors of natural sciences, social sciences and economics, resulting in a number of different integrated approaches for scoping analysis in coastal areas (e.g. Olsen et al., 1997; Holling, 1978). Among the instruments developed for addressing cause-effect relationships relating the human and natural systems, the DPSIR (Drivers-Pressure-State-Impact-Response) scoping framework (Caspersen, 1999), originally conceived as Pressure-State-Response (OECD, 1994), has been established as 'the main non-mandatory organising framework' in major works on real policy issues (e.g. at the EU level, Karageorgis et al., 2006).

Aim of this study is to apply and further develop methodological aspects related to the application of the DPSIR to coastal zone issues. In the following, the DPSIR scoping framework, together with its limitations will be shortly introduced (section 1.1), in order to provide the background for the main hypothesis to be tested in this work, exposed in section 1.2.

1.1 The DPSIR approach

The DPSIR approach handles complex humankind-ecosystem interactions by linking human systems –driven by ethics and values, economic considerations and social structure– to ecological systems –driven by laws of physics (Bossel, 1996). The application of the DPSIR allows the production of policy-relevant findings by organising the analysis along a logical sequence, which links observable or measurable natural phenomena to the social and economic causes of changes. It is then possible to recognise intervention opportunities (e.g Karageorgis et al., 2006; Bidone and Lacerda, 2004). According to the DPSIR framework, there is a chain of causal links from 'driving forces' (anthropogenic socio-economic activities) over 'pressure' upon the environment (use and pollution) to environmental 'state' and 'impacts' on ecology and society, finally leading to political 'response'. The DPSIR interdisciplinary conceptual map allows the analysis of the whole management cycle of a specific environmental issue (Karageorgis et al., 2006), and may lead to an enlargement of the geographical scope of the studies in order to encompass physical units such as river or sea basins, thus overcoming political and institutional boundaries.

Given a selected environmental issue and a study area, the analysis along the DP-S-I-R sequence can be simplified into four main steps (see figure 1.1, which indicates that it is -theoretically- possible to start the analysis at any point):

1. determine which past (*hindcasting*), present or future human activities (*forecasting*) and needs are the *drivers* of environmental changes by exercising anthropogenic *pressure* on the ecosystem (e.g. emissions, land use)
2. assess to which extent each driver/pressure contributes to changing the ecosystem by means of selected indicators that are representative for the ecosystem *state*;
3. assess how those changes can *impact* natural systems and eventually human welfare;
4. develop management strategies and governance structures (*response*) for reducing or preventing undesired impacts.

For each part of the DPSIR-chain a set of indicators, based on several criteria, can be used for describing relevant problem areas of the environment and for assessing the effects of policy measures. Qualitative and quantitative scenario studies can be used for assessing drivers and pressures; quantitative information of the actual state of the environment can be derived from monitoring of indicators, while modelling of indicator-related processes leads to the possible ecological future state and trends (Luiten, 1999).

A most delicate aspect in the DPSIR framework is the assessment of impacts. If we accept that the link between the natural and human dimension is represented by 'ecosystem services' (as shown in figure 1.1), i.e. those ecosystem processes and functions that make human existence possible, easier and more enjoyable (Millennium Ecosystem Assessment (2005); de Groot (1992); Fisher et al. (2007b), then impacts on human well-being should be measured in terms of reduced benefits associated with changes in provision of ecosystem services (section 1.1.1).

Yet socio-economic evaluation of the benefits connected with ecosystem services, as reflected in the economic concept of total economic value (TEV), does not fully reflect the value of the ecosystem as a whole (Turner et al., 2003b; Fisher et al., 2007a). A certain amount of ecosystem structure and processing capability is required to generate any service flow.

1.1.1 The ecosystem service approach

Existence and well-being of humankind is associated with availability of natural resources and related to structures, processes, and existence of ecosystems (ecosystem services). The human system depends on the ecosystem in particular for its essential processes of resource renewal: food production, waste absorption and air as well as water purification. Those essential natural processes of human life support cannot be replaced by technical progress on a significant scale

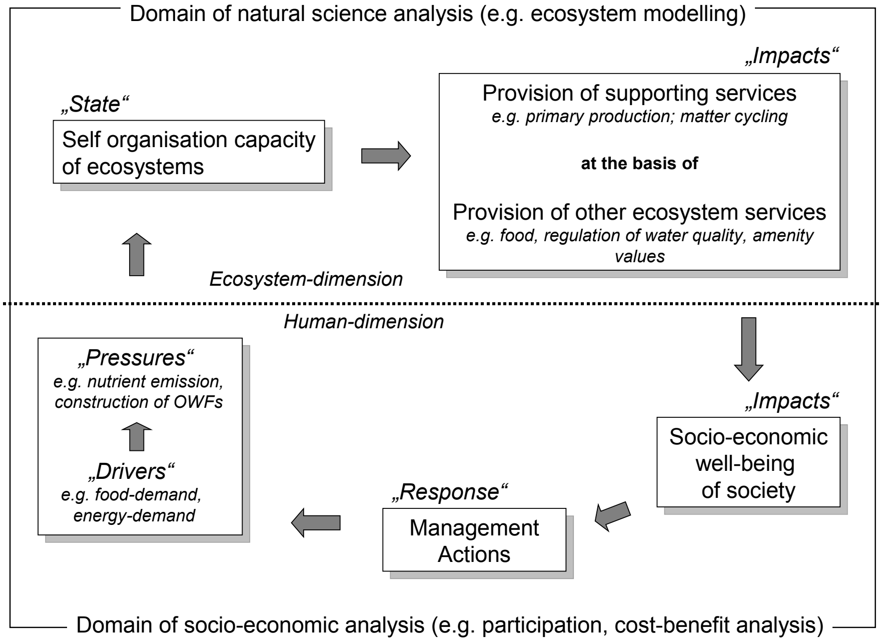


Figure 1.1: DPSIR scoping framework and deployed analytical instruments. The upper part represents the analytical field domain of natural sciences, while the lower part represents the aspects to be analysed by deploying socio-economic methods (OWFs=offshore wind farms).

(Bossel, 1996). 'The term ecosystem services is multi-faceted, as it refers to direct and indirect contributions to human welfare, as well as the underlying processes and structures of the ecosystems' (Fisher et al., 2007a). According to Fisher et al. (2007a), 'ecosystem services are the aspects of ecosystem consumed and/or utilised to produce human well-being'. A schematic representation of different kinds of ecosystem services is shown in table 1.1, where the different categories of ecosystem services are presented, according to the Millennium Ecosystem Assessment (2005) classification. A peculiarity of this classification is that the category *'supporting services'* (e.g. nutrient cycling, primary production) includes the processes at the basis of ecosystem functioning and necessary for the production of all other ecosystem services; *'provisioning'*, *'regulating'* and *'cultural'* services represent the direct contributions to human welfare (Fisher et al., 2007a). Fisher et al. (2007a) classify ecosystem services in intermediate and final services, the former being of indirect use, the latter of direct use to humankind. Food consumption or sheltering are examples of direct uses of ecosystem services, while life-supporting processes and conditions, such as nutrient cycling or pollination, are examples of indirect uses. Intermediate ecosystem services (associated with indirect use) can usually be related to more than a single direct use (Fisher et al., 2007a): in the case of marine nutrient cycling (intermediate service), the direct uses associated (final services) would be water quality or fish abundance. When coming to the economic evaluation of benefits associated with ecosystem services, there is a distinction to be made between *'service'* and *'benefit'*: a benefit takes place when a service is used, while a service is what is provided. In other words, once the service exists, the benefit exists only if the service is used. As a consequence of this, the same service can be associated with different ben-

Table 1.1: Ecosystem services as classified by the Millennium Ecosystem Assessment (2005).

Ecosystem services		
Supporting services (*life-supporting functions*) e.g. primary production, production of atmospheric oxygen, nutrient cycling, soil formation ...		
Provisioning services e.g. production of basic goods (food, fibers, ...)	Regulating services e.g. air and water quality regulation (absorption of wastes, climate regulation, ...)	Cultural services e.g. 'spiritual' goods (amenity, recreation, ...)

efits, in dependence of the final use: Fish abundance can be a benefit to humankind both in form of amenity (colourful coral reef fishes) or in terms of food. Moreover, complex and non-linear interactions between ecosystem structures as well as feed-back loops among nested, sequential and parallel ecosystem processes, result in overlapping of intermediate and final services when assessing economic benefits (e.g. water quality would be an intermediate service if one would be interested in the final service fish production) (Fisher et al., 2007a). According to Fisher et al. (2007a), recognising the multiple aspects of ecosystem services (structure, intermediate and final services) is crucial in evaluating benefits: the authors warn that, in order to avoid double counting, either intermediate or final services should be accounted for in economic evaluation of ecosystem services.

Economic valuation consists in attributing a monetary value to the main function-based benefits provided by an ecosystem. The aggregation of those function-based values (excluding double counting) provided by a given ecosystem is called *'total economic value of the ecosystem'* (*TEV*, Turner et al., 2003b). Monetary valuation of TEV, which includes market (e.g. fish production) and non-market benefits (e.g. philanthropic values) obtained from ecosystem services, is most meaningful for the case of marginal changes in provision (Turner et al., 2003b). Economists expect that marginal values of services decline as provision increases and vice versa (offer-demand theory), however, below some critical 'provision threshold', there may be no meaningful value to be attributed to ecosystem services (Turner et al., 2003b).

The reason for this is that the estimation of TEV (Fisher et al., 2007b; Turner et al., 2003b) does not take into account that, below some 'critical threshold', the supporting mechanisms at the basis of ecosystem functioning may fail, thus abruptly interrupting the provision of those life-supporting conditions that humankind takes for granted. Below critical thresholds, valuation may be precluded because of scientific complexity and consequent ignorance about the welfare impacts of severe ecosystem degradation or collapse (Turner et al., 2003b). The economic value (loss of benefits) of such massive changes cannot be assessed with the same methods used for assessing the consequences of marginal changes. In this context, the risk of massive disruption of service provision is linked to some minimum configuration of ecosystem structure and processes, which is required for ecosystem functioning. This means that the *'total (eco)system value' (TSV)* also includes some 'primary value', i.e. continued (or 'healthy') functioning, which is not captured in economic evaluation of ecosystems (TEV, see figure 1.2) nor in society's demand for services (Fisher et al., 2007b). Under these considerations, TEV is an underestimation of TSV (Turner et al., 2003b).

Supporting services can be interpreted in terms of 'primary' or 'insurance' value (Turner et al., 2004; Barkmann and Marggraf, 2004), i.e. as the 'minimum asset' of processes (and structures) needed for ecosystem self-maintenance in time, growth and adaptive capacity at the basis of

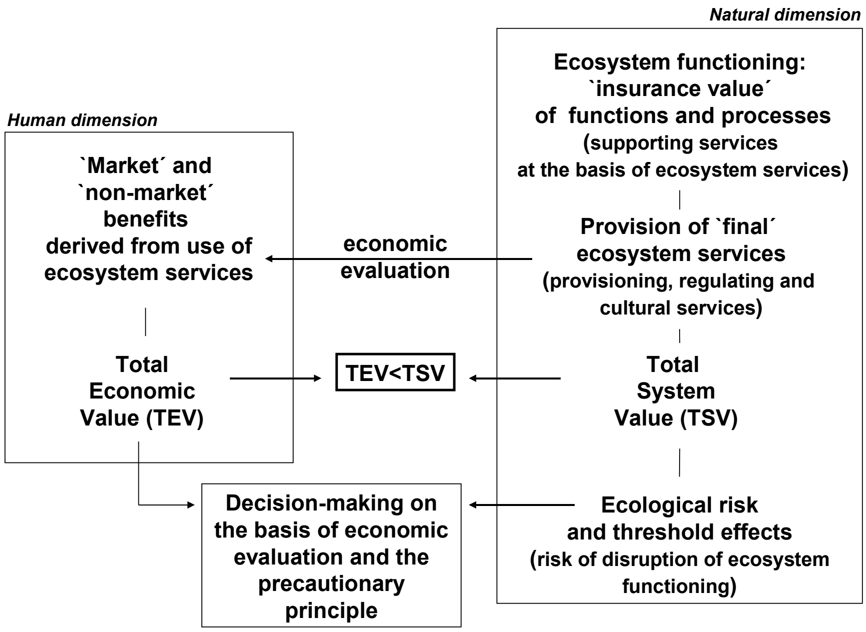


Figure 1.2: Total Economic Value of the ecosystem (TEV) vs. Total Ecosystem Value (TSV). For explanation see text.

ecosystem functioning. This implies a value of supporting services 'per se', independently of their indirect or direct uses, as, would they be disrupted, no other final services at all would be available. In other words, the ability of ecosystem to provide supporting services can be used as a proxy for self-organisation capacity, i.e. for ecosystem 'integrity' and 'healthy' functioning (see section 2.3).

1.1.2 Risk and decision-making

Applied within the DPSIR framework, the monetary economic valuation of ecosystem goods and services can only provide a good indication of social welfare impacts (gains and losses) under certain conditions and in selective contexts, i.e. welfare impacts can be assessed for society only if changes in ecosystem state are marginal, in the sense that they do not approach critical ecosystem thresholds. This means that monetary economic valuation of the ecosystem is assessed under the implicit assumption that collapse of ecosystems can be excluded 'a priori'. Under this constraint, the values derived through the appraisal process will be underestimates of the full TSV (Turner et al., 2003b), as the economic analysis of impacts is not suited to assess the losses associated with total destruction of whole life-support systems, the value of which is not commensurate with monetary values and/or is infinitely high (Nunneri et al., in press). However, uncertainty surrounding the understanding of complex ecosystem functioning does not allow estimating critical thresholds: under uncertainty, potentially every change –even minimal– in functioning of the ecosystem can lead to threshold breaching. In other words, if one could –theoretically– keep the supporting services –and the 'demand' for them– *exactly* at the current

levels, there would be presumably no risk that a major disruption of ecosystem functioning takes place.
In this context, it can be argued that economic analysis should be augmented by ‘ecological risk’ analysis (i.e. the risk of major disruption of ecosystem services), which has its foundations in ecological science and ethical, political and socio-cultural disciplines and knowledge, in order to bridge the gap between TEV appraisal and TSV (see figure 1.2), when informing stakeholders and decision-makers (Nunneri et al., in press).

In the context of decision-making and environmental management (see section 2.4), the appraisal of alternative costs and benefits is affected by uncertainty about system boundaries and processes. Moreover, the effects of scale and thresholds underline the case for a precautionary approach (EC, 2000a).
Sustainability requires each generation to maintain the self-organising systems that provide the context for all human activity and therefore possess ‘primary’ value. If intergenerational equity is accepted, current generations have a precautionary duty of preserving critical ecological resources that are necessary for the provision of future generations and there is a need to develop a ‘risk norm’ to assist in the organisation of political, legal and economic regulatory institutions (Nunneri et al., in press).
In this context, conserving ecosystem processes at the basis of self organisation and with them supporting services, can be taken as the insurance that ecosystems will continue existing in time and so will their provision of services. Ecological risk is assessed based on changes in integrity (self-organisation) processes due to human intervention and can be taken as a measure of the risk that, by those changes, a threshold is exceeded, thus leading to major disruption of ecosystem functioning (see section 2.3).

The key point connecting society (and the expression of its will and needs through policy) to environmental issues goes through numerous stages (Parsons, 1995) and involves values and perceptions. Risk acceptance is a result of different social and psychological factors, which may lead to risk amplification or ignorance (Kasperson et al., 1988; Renn, 1998). Acceptance of some risk level stems from people’s vision(s) of the world. For example, distinct conceptions of nature imply contrasting assessments of (coastal) ecosystem vulnerability to changes (Levy et al., 2000) and thereby different interpretations of the precautionary principle and willingness to pay for risk prevention (see section 3.1.2). The inclination to environmental risk or its refusal (risk aversion) in a society will determine the level of acceptable risk and thereby policy and law implementation, along with the willingness to pay for (and commit in) risk prevention (see section 3.1.2).

1.2 Aim of the study

The aim of this study is to apply and further develop methodological aspects related to the operationalisation of the DPSIR framework. In particular the focus is set on the development of an ecological risk concept, i.e. a methodology for assessing risk of major disruption of ecosystem services related with ecological changes, in order to include risk assessment into Decision Support Systems (DSS) for decision-makers.

The analysis has a twofold goal:

1. to test the suitability of the DPSIR as decision support system (DSS) in coastal areas, by applying it to two distinct case studies in the North Sea; and
2. to develop and test the suitability of 'ecological risk' assessment, based on modelling and indicators, as part of impact assessment evaluation under uncertain critical thresholds.

The DPSIR approach has been applied to two case studies in the North Sea coastal area, under the hypothesis that it is a valuable instrument for decision support in coastal areas, i.e. that delivers essential information for decision-making. The chosen case studies are eutrophication and the construction of offshore wind parks in the North Sea coastal waters (section 2.5, page 21). Those two issues have been focused upon within two three-year research projects, which offered the frame for the present study. Eutrophication has been one of the main topics analysed under the auspices of the 'EUROCAT' (EUROpean CATchment, catchment changes and their impact on the coast) Project, funded within the EU 5^{th} Research Framework Programme (2001-2004, Project No. EVK1-CT-2000-00044, see Salomons, 2004 for major details). Offshore wind has been the main focus of the BMBF-funded project 'Zukunft Küste-Coastal Futures' (2004-2007, Project No. FKZ03F0404A, see Kannen, 2004 for major details). Offshore wind and eutrophication have been chosen for this study, as they present different challenges to the analysts. The issue of eutrophication represents a 'mature' problem in the sense that a regulatory policy framework has already been elaborated and implemented with some degree of success (de Jong, 2006). The 'industrialisation of the sea', embodied by the construction of offshore wind parks represents an emerging issue, where both the analysis of ecological effects and the adoption of a regulatory framework are still being developed (Elliott, 2002; Köller et al., 2006). In this context the main research questions to be answered are:

- Is the DPSIR a valuable instrument as a decision support system (DSS) in coastal zones?
- What are the strengths and limitations of the DPSIR approach?

The focus of the study is further set upon ecological changes and their evaluation in terms of ecological risk. The concept of ecological risk (section 2.3, page 16) has been developed for addressing the evaluation of ecosystem changes (see 1.1.1) under the uncertainties surrounding ecosystem complexity and the impossibility to determine critical thresholds. The ecological risk of major disruption of ecosystem services due to collapse of ecosystems is tightly connected with ecosystem total value and 'insurance' value of ecosystem structures and processes. The research questions to be answered in this context are concerned with the definition and operationalisation of the ecological risk concept, in particular:

- How is ecological risk defined and operationalised?
- Which parameters used for assessing ecological risk are suitable indicators for describing the overall ecosystem value (TSV) independently of the issue?
- What are the weaknesses and strengths of the ecological risk concept?
- In what aspects can the ecological risk methodology be improved?
- What are possible future uses of the ecological risk concept?

Chapter 2 briefly reports the main features of the DPSIR approach. In this chapter the analytical instruments deployed in this study at each stage of the DPSIR scoping framework are introduced. Chapter 3 reports the analysis of the two considered issues, eutrophication and offshore wind farm construction (section 3.1 and section 3.2 respectively), structured along the DPSIR conceptual chain. Section 3.3 in chapter 3 deals with the cumulative ecological risk assessed for both uses of the North Sea jointly. Chapter 4 discusses the main findings reported in chapter 3 in the light of their relevance for integrated management of the North Sea. Moreover, the challenges posed by the operationalisation of the ecological risk concept within the two case-studies are examined more closely. Chapter 5 concludes by answering the research questions formulated in this section.

2 Materials and Methods

Following the DPSIR scheme, integrated assessment (IA) can start by acknowledging the urgency of some environmental issue or simply by scoping what drivers and pressure may cause major environmental impacts in the future. The main step at this stage is the identification of causal-chains linking drivers and pressures to changes in the ecosystem state. As current pressures and drivers might be different from the future ones, the use of socio-economic scenarios (as described in section 2.1) can help to consider a broad range of possibilities, thus rehearsing the (governance) ability to cope with arising environmental issues. The resulting environmental state, as well as the environmental effects of possible alternative management options can be assessed by means of modelling and selected indicators (see section 2.2.1), which measure the magnitude of changes with respect to the current (or desired) ecosystem state. At this stage also the proposed ecological risk assessment should take place (see section 2.3), in order to be factored into socio-economic appraisal of possible management options. Finally, a choice among possible alternative responses to the problem is made, based on their overall effects upon human welfare (see section 2.4). As the processes of anthropogenic origin causing environmental problems may originate in a much wider area (e.g. catchment) than the coastal region itself, the whole scoping exercise and especially the elaboration of possible responses should foresee the involvement of multiple, in some cases international, stakeholders (Nunneri et al., 2005). The above mentioned analytical instruments deployed at the different stages of the DPSIR scoping analysis in the framework of this study will be described in more detail in the following sections.

2.1 Drivers and pressures: scenarios

Scenario studies aim to cope with the uncertainty which inevitably surrounds how the future will unfold. The use of scenarios can shed light on what human needs and actions are likely to generate future environmental problems and requirements for management (Ledoux et al., 2002). Scenarios are ad-hoc constructed futures, each representing a distinct possible world, therefore each internally consistent and plausible. A scenario is not a forecast, because it does not attach probabilities to a particular outcome (Lorenzoni et al., 2000). Scenario analysis has been increasingly applied to different disciplines in a variety of ways and there is a multiplicity of scenario typologies (for a review see EEA, 2001; van Notten et al., 2003). Descriptive forecasting scenarios, among which baseline, (reference, or non-intervention) scenarios, describe possible futures thus allowing to imaginatively explore the possible impacts of certain development patterns (Burt and van der Heijden, 2003). But scenarios can be also used for testing management options with respect to their effects upon the environment or for hindcasting, i.e. for exploring how to achieve a desirable (normative) future situation (van Notten et al., 2003). Scenarios can include 'unexpected' changes (including those of human priorities and values), which can bring about drivers and pressures different from the current ones and therewith impacts which would not take place under the current trends. Moreover, values and priorities result in different human perceptions of nature vulnerability to changes (van Asselt and Rotmans, 1996) and thereby in different attitudes towards risk (risk-seeking or risk aversive) and the resulting choice of management strategies (see section 3.1.2) for an example of scenario-dependent risk attitudes and application of the precautionary principle). The use of scenarios for scoping relevant emerging issues is one way of mitigating the problems of the time-lag lying between the initiation of scientific analysis and the use of scientific knowledge (findings) in decision-making and management (Nunneri et al., 2005). Through scenario-based description of alternative futures it is possible to elaborate a 'robust' management strategy, which will reveal the most advantages under different possible developments (Ledoux et al., 2002). Moreover scenario assessment is an appropriate tool for integrating social and natural sciences because it facilitates analysis of interwoven processes through the combination of descriptive (storylines) and quantitative assessments ('guesstimates' essential for modelling). In this context, scenarios are an essential component of integrated assessment studies (see Bertrand et al., 1999; Alcamo, 2001; Rotmans et al., 2001, for scenario methodology and examples).

In this study, scenario assessment has been used as a starting point for depicting socio-economic drivers and pressures potentially causing changes in the state of the North Sea coastal ecosystem. Scenarios are described by means of qualitative assumptions, the 'storylines', and quantitative assumptions, which are the starting point for modelling ecosystem changes and assessing ecological risk.

2.2 State: modelling and indicators

Once assessed possible pressures on the environment associated with different scenarios (e.g. the amount of nutrient emissions), resulting changes upon the marine ecosystem can be best assessed through a validated model, which has been proven to mirror the ecosystem characteristics and behaviour in the study area. For the North Sea the choice was to use the ecosystem model ERSEM (European Regional Seas Ecosystem Model, Baretta et al., 1995). ERSEM has been used to simulate (nutrient reduction) scenarios in a number of cases, both for the North Sea (Lenhart and Pohlmann, 1997; Lenhart, 2001) as well as for the Continental Coastal Shelf region (Heath et al., 2002). Selected ERSEM output parameters have been used to indicate the ecosystem state. In the following a brief description of the model is given first, followed by the selected indicators used for assessing ecosystem 'functioning'.

2.2.1 The ecosystem model ERSEM

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 model forcings are irradiance and temperature, suspended particulate matter (SPM) concentration, hydrodynamical information for advection and diffusion, data on atmospheric nutrient input to the North Sea as well as inorganic and organic river load data. The model consists in different connected 'modules', that describe the dynamic interaction between physical, chemical and biological processes and therewith the transformation of ca. 60 state variables. The biological and biochemical modules can be combined with different hydrodynamic models, one-dimensional (1D), two-dimensional (2D or box model) and three-dimensional (3D), which subdivide the North Sea in different ways.

A box model version of ERSEM has been used for assessing the ecosystem changes within this study. It mirrors the complexity of biological and bio-geochemical processes (Moll and Radach, 2003) within the pelagic and benthic system, while the physical features (e.g. hydrodynamics, light irradiance) are covered in a simplified but realistic form (Lenhart and Pohlmann, 1997). The box model covers an area of 577,620 km^2 and a total volume of 51,047 km^3. 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, which represents the central and northern North Sea with sufficient resolution, is more finely resolved in the coastal area (Lenhart, 2001).

The macro-nutrients nitrate, ammonium, phosphate, and silicate as well as carbon are used as state variables. The ecosystem is represented as follows:

a) in the pelagic environment
    - primary producers: phytoplankton organisms, divided into diatoms, which use Si for building up their cell walls, and non-diatoms (including flagellates, picophytoplankton and so called inedible, e.g. Phaeocystis)
    - consumers: zooplankton and zoobenthos and
    - decomposers: bacteria feeding on organic excretions and dissolved organic matter

b) in the benthic environment
    - consumers (e.g. filter feeding organisms)
    - decomposers responsible for re-mineralisation of sedimented organic matter (detritus)

The energy exchange between the pelagic and benthic ecosystem occurs through enrichment (sedimentation and active uptake through filter feeders) and loss (re-mineralisation and release) processes. Sediment dynamics is a key element for matter exchange within the pelagic and benthic system. Those processes allow representing the nutrient storage into the sediment. In addition, ERSEM is able to simulate aerobic, anaerobic and sulphate reduction conditions within the sediment.

Among the input variables, light irradiance is the most relevant variable for primary production, which is modulated by SPM concentration (causing light attenuation) and temperature, which -within temperate ranges- controls the speed of biological processes (e.g. nutrient uptake and primary production). Especially in the case of eutrophication studies, the incorporation of different pathways for nutrient cycling through the trophic net, which act on different time scales, indicates the overall effects on the ecosystem related to nutrient reduction. A lower availability of nutrient therefore not only results in a depressed primary production but also structural changes within the ecosystem, e.g. the increasing importance of the microbial loop (Lenhart, 2001). Moreover the complex benthic module allows modelling the slow processes in the benthos, including buffering effects (e.g. related to the continuous input of organic loads due to eutrophication). In Figure 2.1 the main variables considered in the model are reported.

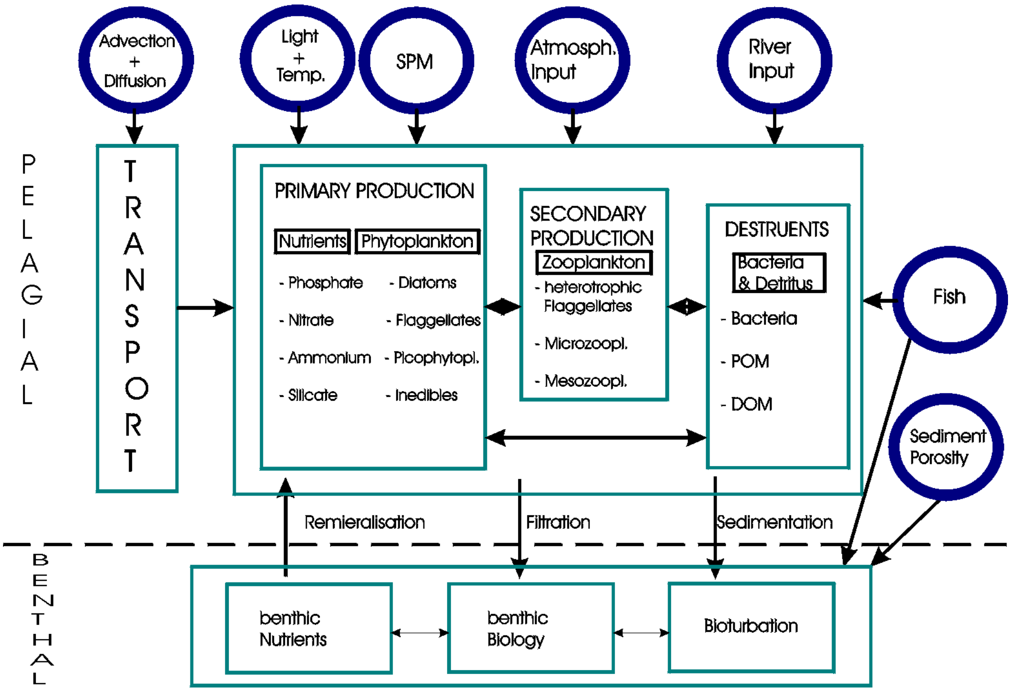


Figure 2.1: Schematic overview of the ecosystem model ERSEM. The rectangles represent modules which are connected with inter-compartmental fluxes (arrows). The circles indicate the forcings (Lenhart, 2001).

When using the ERSEM box-model version (figure 2.2), one should be aware of its peculiarities. To represent the ecosystem dynamics in the coastal region with its highly variable conditions, relevant information about morphology and 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 SPM 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 and Pohlmann (1997) and Lenhart (2001). The ERSEM model simulates benthic ecosystem processes (benthos module) in an accurate manner, being able to account for slow re-mineralisation processes taking place into the sediment. This allows for including time-lags between processes occurring in the water column and those occurring in the sediment (re-mineralisation and fixation). The sediment module dynamically define aerobic, anaerobic and sulphidic sediment layers within the first 30 cm of sediment, based on organic matter availability bioturbation effects.

According to Radach and Moll (2006) ERSEM is one of the best validated ecosystem models for the North Sea. This is due to the fact that in the first phase of the ERSEM project the simulation results were validated against the data from the UK North Sea Project, which has been carried out in 1988/89 in the southern North Sea (Howarth et al., 1994). The second major validation was based on data for nutrients and chlorophyll from the ICES database, which were aggregated into a 1° by 1° squares for the whole North Sea (Howarth et al., 1994). However, the higher trophic levels are not validated with the same data intensity as nutrients and chlorophyll or primary production. This is due to (1) the limited availability of data (compared to nutrient data)

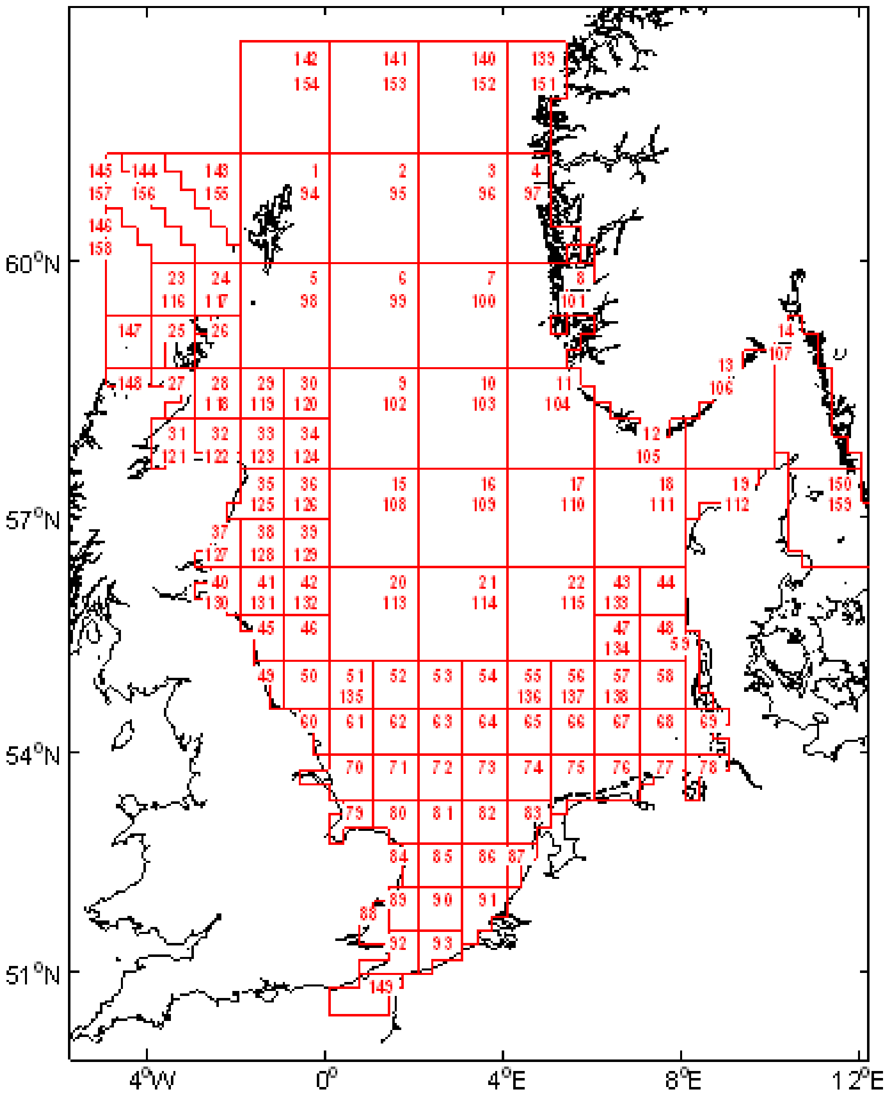


Figure 2.2: Overview of the ERSEM boxes. While the coastal waters are modelled by using a single box for the whole water depth, in more offshore areas a surface and a deeper box are considered, i.e. of the two numbers in the box the former represents the surface box, the latter the deeper box (Lenhart, 2001).

and (2) the intrinsic nature of ERSEM, which is designed to reproduce nutrient flows through the ecosystem. The simulations are related to functional groups which represent certain transitions within the mass flow. The focus is not set on the simulation of single species but on certain functional groups, which represent certain key species. For instance diatoms represent the transition of inorganic silicate to organic silicate via primary production. In this context the comparability with data is limited, i.e. there is not a complete match between data and simulation results (lenhart pers. comm.).

There are two major issues to be considered when applying the model ERSEM to the issues considered in this study:

1. the box model simplifies existing horizontal gradients (e.g. nutrient concentrations are higher along the coast an decrease with increasing distance from shore) by assuming constant concentrations in each box; gradients are reflected by box dimensions (and numbers): a major number of smaller boxes in coastal areas allows to roughly approximate coast-offshore gradients

2. the 2-dimensional model ignores vertical gradients (e.g. concentration of SPM in the water column), by assuming an average concentration in each box.

Although in principle it would be possible to reduce boxes dimensions and to adopt a 3-D model for taking into account vertical gradients, the time for running such a model would increase considerably. Especially for the simulation of the different years for the offshore wind farm scenarios the computing time would not have been feasible within the available time frame (Lenhart, pers. comm.). Moreover, due to the lack of detailed information about the vertical distribution of increased SPM concentrations during the construction phase, the deployment of the vertical resolution from a 3-d model could not include more detailed SPM distribution over the water column.

2.2.2 Selected integrity indicators

Indicators are deployed to assess the state of a complex indicandum (i.e. the object about which information is needed), otherwise not directly measurable (Baumann, 2001). Indicating the ability of ecosystem to function, and thereby to provide supporting services, in terms of 'integrity' or 'health', results from an anthropogenic vision, based primarily on human concerns and interests (Bossel, 1999). The concepts of 'integrity' and 'health' are not 'monolithic' (De Leo and Levin, 1997), i.e. they are subject to a broad range of interpretations (for a short overview see Burkhard and Müller, 2007; De Leo and Levin, 1997; Westra, 1996). The concept of integrity used for this study defines integrity based on self-organisation capacity of ecosystems, i.e. a system is integer if it is able to create structures and gradients through energy conversion processes (self-organisation capacity) and thereby, 'to function' in time (Gunderson and Holling, 2002; Barkmann, 2002). This concept of integrity is strongly linked with sustainability in the sense that an integer ecosystem will be able to provide services in the long-term (Müller et al., 2000).

Ecosystem development is associated with energy conversion processes, e.g. the transformation of 'high quality' energy (exergy, see Jørgensen, 2000) into non-convertible energy fractions (entropy) or stored within biomass, detritus or information. 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. Barkmann (2002) mentions four factors relevant for ecosystem self-organising capacity: (1) availability of usable energy, (2) availability of material substrate (nutrients, water, carbon), (3) availability of biological information and (4) the level of already existing self-organisation. A gradient of usable energy (exergy) drives energy conversion processes which use material substrate for creating biocenotic structures, based on existing biological information (i.e. existing biocenotic communities). Based on ecosystem theory, energy

and matter balances are key-variables for maintaining ecosystem diversity, i.e. (genetic) information, which is essential for coping with changes (resilience, Beaumont et al. e.g. 2007). In particular Baumann (2001) and Barkmann (2002) mention, as a measure of the key-aspects of self-organisation the use of available energy (exergy) and substrates as well as biotic and abiotic heterogeneity, the following processes: exergy capture, cycling of elements, storage capacity and matter losses as well as heterogeneity (diversity). The term exergy defines potential of doing work (exergy appears in physics as energy, matter and information): (ecosystem) structures are created through exergy degradation (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. The ability of an ecosystem to develop also depends on availability of material substrate (diversity of abiotic structures) and limiting nutrients. The storage capacity (here assessed within the sediment) and the matter balance (here assessed within the water column) are therefore essential processes governing the exchange rate of the different pools, and the possibility of temporarily dampening or buffering external inputs; matter losses outside the system reduce the capacity of primary and secondary production, which are essential life-supporting processes. Storage capacity and (re)cycling of limiting substances are essential for guaranteeing biological diversity (organism diversity and genetic diversity). Diversity is, in turn, one main aspect of organisation and complexity of ecosystems: biodiversity depends also on abiotic diversity, e.g. gradients (Barkmann, 2002). The extent to which ecological systems are able to (re)utilise limiting substances depends on the heterogeneity and the biotic diversity of the system; cycling of nutrients can be taken as a measure of ecosystem efficiency. These fundamental ecosystem processes are at the basis of ecosystem provision of services.
The delineation of clear-cut boundaries between self-organisation processes and ecosystem services, is not feasible, due to the overlapping nature and complexity of many ecosystem processes. In this study ecosystem integrity processes are indicated by ecosystem supporting services. The integrity processes, ecosystem supporting services and selected ERSEM parameters for indication are reported in table 2.1. In the following a brief explanation of those parameters is given.

- Exergy capture, i.e. the use of available nutrients and light for synthesis of organic material is indicated by primary production (expressed in g C $m^{-2}y^{-1}$), which represents the starting point of the food-web.
- Turnover of winter nutrients (expressed in times y^{-1}) gives a measure of the effectiveness of the ecosystem in cycling nutrients through the system. The turnover of nutrients is calculated on the basis of the total available nutrients (winter nutrient content) and the total uptake by phytoplankton during the year. In other words, it gives the ratio of the used and the available nutrients and thereby the cycling.
- Heterogeneity is described as the average ratio of abundance of diatoms and non-diatoms in spring-summer (i.e. for the time-span April to August).
- Storage (expressed in mmol N $m^{-3}y^{-1}$) within the system is related to material accumulated in the sediment during one year. It is calculated as the sum of all organic nutrient inputs into the sediment from the water column (sinking of dead organic material plus the particles taken out of the water by active filtering of filter feeders) minus the release of remineralised inorganic nutrients (DIN) into the water column. Positive values of storage reflect accumulation in the sediment, while negative values reflect net release out of the sediment.

Table 2.1: Use of ERSEM parameters for indicating ecosystem integrity processes. Please note that all indicators are expressed for one-year time as cumulative values, except for the ratio diatoms/non-diatoms, which is the average during the months April to August.

Parameters and Indicators of Ecosystem Integrity			
ERSEM parameters	Metrics of ERSEM parameters	Integrity process	Integrity indicator
primary production	g C $m^{-2}y^{-1}$	exergy capture	primary production
N uptake by phytoplankton winter DIN content	mmol N $m^{-3}y^{-1}$ mmol N m^{-3}	cycling of nutrients	turnover of winter nutrients = N uptake by phytoplankton/ winter DIN content
diatoms/non-diatoms average (April-August)	dimensionless number	heterogeneity	diatoms/non-diatoms average (April-August)
org. sediment input (sinking) org. sediment input (filterfeeders) inorg. sediment output	mmol N $m^{-3}y^{-1}$ mmol N $m^{-3}y^{-1}$ mmol N$m^{-3}y^{-1}$	storage capacity (sediment)	nutrient stored in the sediment = org. sediment input (sinking) + org. sediment input (filter- feeders) - inorg. sediment output
box input organic box input inorganic box output organic box output inorganic	mmol N $m^{-3}y^{-1}$ mmol N $m^{-3}y^{-1}$ mmol N $m^{-3}y^{-1}$ mmol N $m^{-3}y^{-1}$	matter balance (water column)	nutrient transport in adjacent areas = box input (org. + inorg.) - box output (org. + inorg.)

- Matter balance (expressed in mmol N $m^{-3}y^{-1}$) in the water column, is calculated as the difference between advective transport of N (organic and inorganic) into the considered box(es) and out of it during one-year time. Negative matter balance values correspond to net matter losses (source), positive values to net matter accumulation (sink) with respect to the adjacent environment.

2.3 Impact: ecological risk

Although there is a broad range of definitions of risk, both in science and in public understanding, a common feature of all risk concepts is 'the distinction between reality and possibility' (Renn, 1998). 'Risk' denotes the possibility that undesirable outcomes may occur as a result of natural hazards or human action: in this sense risk is bound to the choice among different alternatives, which are connected with both win and loss chances (Breckling and Müller, 2000). Therefore risk assessment implies the need of making causal connections between actions and their effects, although under constraining uncertainties (Renn, 1995). The principal role of risk concepts is that they offer a way to cope with changes and uncertainties (Valverde, 2007). Risk can be seen as a notion combining both descriptive and normative aspects: the descriptive part of risk implies the analysis of cause-effect relationships, while the normative aspect sees risk as a 'social construct' based on social values and preferences (e.g. Potthast, 2004; Renn, 1995). Risk has been technically defined as 'negative expected utility' (Benjamin and Cornell, 1970), i.e. the product of the probability of an undesirable future event multiplied by the caused losses of benefits ('utility'), usually expressed as 'monetary cost'. The 'probability of occurrence' of an undesired event and the related 'losses' are the key variables necessary for quantifying risk assessment based on the technical definition. However, some authors recognise the necessity to include social factors into risk assessment; those analysts may prefer quantification procedures other than assigning probabilities (for an analysis of risk concepts between probabilism and contextualism see Thompson and Dean, 2007).

Based on the concept of ecosystem services and the benefits associated to their uses (section 1.1.1), the term *'ecological risk'* within this study denotes *an estimate of the risk of major disruption in the provision of ecosystem services expected/desired by human societies*.
The concept of ecological risk is connected with the one of 'environmental security', which can be defined as 'the public safety from environmental dangers' and includes some pro-active maintenance of natural assets needed by humans (Müller et al., 2007). In this context, the possible collapse of an ecosystem is a major environmental danger to be 'feared', and ecosystem integrity (provision of life-supporting services) can be seen as having 'insurance' value against the risk of ecosystem collapse.

If one accepts what has been highlighted in section 1.1.1 about the infinite value that should be attributed to life-supporting functions (supporting services of ecosystems), then it is evident that ecological risk, cannot be assessed using the technical approach, as the estimated monetary losses connected with major ecosystem service disruption should be infinite. Furthermore, under uncertainty surrounding overlapping cause-effect chains and non-linearity of ecosystem processes, the estimation of the probability of exceeding critical thresholds, and therefore the anticipation of a decision-outcome and its occurrence probability is also not precisely quantifiable (Dequech, 2003; Renn, 1998).

For those reasons, ecological risk is a complex, not directly approachable phenomenon (indicandum), which can be described through selected indicators. Based on its definition, ecological risk is related to ecosystem integrity. The starting assumption of this study is, therefore, that ecological risk can be assessed as a function of the integrity indicators reported in table 2.1:

$$EcRisk = f(ecosystem\ integrity) \tag{2.1}$$

In this study some first attempts to assess ecological risk by further specifying the relationship expressed in eq. 2.1 are reported. In this context, two fundamental steps are used for condensing information expressed by integrity indicators into a single, dimensionless parameter expressing ecological risk:

1. because the integrity indicators are different in both their measurement units and range of values, they are normalised into dimensionless numbers within a selected range (between 0 and 100) in order to aid comparability. Although ecosystem processes and functions are usually non-linear, the normalisation of indicator values is carried out –in the first approximation– in a linear way by applying the following equation:

$$I'_n = \frac{I_n - min(I_n)}{max(I_n) - min(I_n)} * (newmax(I_n) - newmin(I_n)) + newmin(I_n) \tag{2.2}$$

where:
I'_n= normalised value of the n^{th} indicator
I_n = current value of the n^{th} indicator
$min(I_n)(max(I_n))$ = current minimum (maximum) value of the n^{th} indicator in the considered set of values

$newmin(I_n)(newmax(I_n))$ = chosen miminum (maximum) value for the normalisation procedure. The minimum value for the normalisation is set to 0, the maximum to 100;

2. ecological risk is then assessed as the 'average difference' of the indicator values between a certain situation and a reference state, by applying the following equation:

$$ER_y = \frac{\sum_{n=1}^{n=M} |I'_{n*} - I'_{ny}|}{M} \tag{2.3}$$

where:
ER_y= ecological risk in the year y
I'_{n*} = normalised reference value of the n^{th} indicator
I'_{ny} = normalised value of the n^{th} indicator in the year y
M = selected number of indicators (in this case M=5).

By applying equation 2.3 ecological risk is assessed in a linear way. This means that equal marginal 'distance' increment give the same contribution to the overall risk appraisal.

2.4 Response: alternatives, trade-offs and the role of society

Decision-making for environmental management (see figure 2.3) can be generally divided into three phases:

1. the discovery of an issue (in some cases controversial) and the related definition of the problem;
2. the placement of the problem in the political agenda and thereby the assessment of possible alternative actions and selection of one of those, based on some criteria; and
3. the management phase, i.e the implementation and evaluation of policy effectiveness.

Once selected a response strategy, the appraisal of success and effectiveness in tackling the problem during the implementation phase completes the management cycle, which can start again, if the effects of management are not considered satisfactory. On the other side, the management phase is often characterised (after some time-lag, which can vary from issue to issue) by a decrease in interest, due, for instance, to the emergence of other issues, which substitute for the 'managed' one in the political agenda (Parsons, 1995). The emergence of an issue (and thereby commitment in solving it) can be triggered by disparate circumstances and disappear soon after even without having reached some real improvement of the situation, or periodically re-emerge in cycles of attention (de Jong, 2006; Parsons, 1995). In practice policy making is much less linear than illustrated in figure 2.3. It is influenced by a complex mix of beliefs, history and culture, trust and accountability concerns as well as by power networks involving multiple actors: the public, governmental organisations, business, the media and the science establishment (Parsons, 1995).

In the ideal rational approach to the solution of environmental concerns, the ecological, social and economic effects of alternative response strategies need to be all 'quantified' in order to be factored into decision-making for environmental management.
'Any choice among alternatives implies that the chosen option is considered more 'valuable'

than the discarded ones, therefore decision-making implies (implicit or explicit) valuation of ecosystem benefits' (Costanza, 2000). Under the caveats already discussed, monetary valuation of marginal change in ecosystem service provision can be carried out based on different techniques –not free from limitations and criticisms (e.g. Chee, 2004; de Groot et al., 2002; Ledoux and Turner, 2002; Turner, 2006) and then considered jointly with costs and benefits of different nature (e.g. costs of alternative implementation) and used for identifying option trade-offs (e.g. Brown et al., 2001). Ledoux and Turner (2002) summarise the main methodological approaches for strategic socio-economic management option appraisal as follows:

- Cost-effectiveness analysis, to be applied when there are definite targets in order to determine which option achieves the target at lowest cost;
- Cost-benefit analysis, which values all costs and benefits in monetary terms and can be used when targets have to be defined within the assessment exercise (no pre-definite goals)
- Multi-criteria analysis, deployed to compare options based on a set of relevant aspects, which can include both quantitative and qualitative aspects (i.e. expressed in different measurement units), and are given different weight for the decision.

All these methodologies, although generally deployed in decision-making, do not explicitly deal with the issue of potential irreversible changes, i.e. with risk and uncertainties. In this context the application of the precautionary principle (Harremoes et al., 2002), which aims at minimising losses in advance of scientific proof of adverse effects, should be incorporated into the decision-making procedure (Turner, 2006; Ledoux and Turner, 2002; Chee, 2004). The conventional economic viewpoint recommends the application of cost-benefit or risk-benefit analysis and highlights the fact that the precautionary principle is not a panacea and will itself incur social costs (e.g. Turner et al., 2003a). Co-evolving environmental and socio-economic systems produce feedback effects which might cause impacts thresholds to be breached, causing ecosystems to flip from no apparent impact to significant impact with only seemingly marginal changes (e.g. Köhn, 1997). The adoption of precautionary behaviour in the face of uncertainty is reinforced by ethical reasoning, which warns against the creation of uncompensated costs for future generations (Turner, pers. comm.).

However, as decisions are taken in the present, the rights of future generation are often underrepresented and the agreement about –and the implementation of– management measures is often hindered by the (non-)action of powerful stakeholders. Lack of consensus during the decision-making process or protectionist attitude towards private interests can result in major (societal) 'non-commitment' or opposition. In particular, the main factors hindering implementation of management strategies are to be found in:

- conflicting perception of the issue at stake by different stakeholders (including risk acceptance or risk aversion positions)
- interest protection (in relation with distribution of potential economic win and losses)
- power distribution among interested parties (e.g. lobbyism)

For these reasons, management responses resulting from broader stakeholder agreement are more likely to be implemented than imposed solutions. Stakeholder dialogue is meant to mitigate stakeholder conflicts, either in a top-down or in a bottom-up approach, depending on the kind of

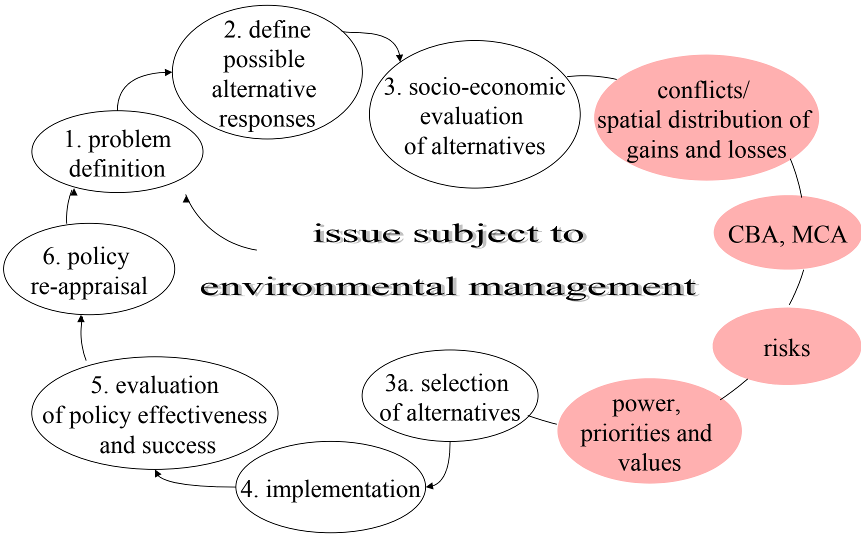


Figure 2.3: Decision-making procedure. The steps to be undertaken in environmental management are schematically represented in a clockwise sequence. Decision-making for environmental management starts by problem definition, includes evaluation of possible alternatives (represented in points 3-3a, where CBA=cost-benefit analysis, MCA=multi-criteria analysis), followed by the implementation of the selected ones and finally by evaluation of success of management possibly leading to iterative improvement through starting the whole procedure again (but including acquired knowledge). The effects of culture, history, beliefs as well as policy-networks are not addressed in their complexity in this simplified scheme.

society, needs and awareness of the general public (Nunneri et al., 2005). Stakeholder participation can take different forms, in dependence of the socio-cultural and economic context: varying from a minimum involvement (stakeholder 'information'), to a maximum, i.e. full participation in decision making. The responsibility of institutions and governments involved in integrated assessments is to take into account different needs and perceptions of stakeholders and to develop compensation mechanisms as well as win-win synergies in order to facilitate cooperation among the involved parties. As some of the activities impacting the coastal zone can originate in regions far away from the coast, possible responses should be tailored to fit different cultural, political and economic possibilities of the involved stakeholders, by making compensation mechanisms available on a broader, in some cases international, domain in order to achieve wide-ranging commitment in implementation.

2.5 The North Sea

The North Sea is a large semi-enclosed sea on the continental shelf of north-west Europe; its boundaries are the coastlines of England, Scotland, Norway, Sweden, Denmark, Germany, the Netherlands, Belgium and France, and imaginary lines delimiting the western approaches to the Channel, the northern Atlantic between Scotland and Norway, and the Baltic in the Danish Straits (Walday and Kroglund, 2002). The total North Sea area is about 750000 km^2, with a catchment area of ca. 850000 km^2. Average water depth is 90 m, ranging up to 725 m near the Scandinavian coast (Skagerrak), while the Southern North Sea has very shallow coastal waters. Atlantic water enters the North Sea mainly from the north, and moves eastwards following an anticlockwise circulation. Figure 2.4 shows the general water circulation patterns in the North Sea; 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, which flows along the coastline as continental coastal current, and finally adds to the Norwegian Trench outflow back into the Atlantic. In the Skagerrak, the North Sea water mixes with less saline water from the Baltic, and is transported north along the Norwegian west coast. The residence time of North Sea Water is about 3 years (Sündermann et al., 2001). The seabed is predominantly sandy; high mud contents are found in deeper areas and in coastal areas under riverine influence. The coastlines offer a large variety of habitats, ranging from mountainous and rocky, often dissected by deep fjords, in Scotland and Norway, cliffs, pebble beaches, estuaries, sand and mud flats in northern England and sandy beaches characterised by dunes, estuaries and tidal inlets from the Channel to the Danish west coast (Walday and Kroglund, 2002).

From an ecological perspective, the North Sea has an international role as nursery area for fish and is of global relevance due to its (migratory) bird population (Walday and Kroglund, 2002). In particular the Wadden Sea, the coastal fringe extending in the Southern North Sea from the Netherlands to Denmark, is the largest unbroken stretch of mud flats in the world, mostly sheltered by barrier islands. Due to its shallow waters and the net inflow of organic matter (through the main discharging rivers), the Wadden Sea has a high primary productivity, which is at the basis of its function as a nursery area for fish and shrimp of the North Sea as well as a central feeding ground and recovery area for migratory birds. Since 1985 most parts of the Wadden Sea is designated as National Park and Biosphere Reserve by a trilateral agreement of The Netherlands, Germany and Denmark (Colijn and Reise, 2001).

The North Sea and the Wadden Sea are intensely exploited by riparian European countries for a wide range of resources. These include fish, marine sands and gravels, oil. It is also a major shipping route, serving world ports like Hamburg and Rotterdam, and oil and gas terminals linked to offshore rigs by pipelines. Especially along the coast and in the islands tourism is the dominating activity, along the Wadden Sea coast agriculture dominates the mainland (Colijn and Reise, 2001). Those uses also represent the main pressures upon the North Sea marine environment, with contamination, overfishing and eutrophication as as the main consequences (Turner, 2005).

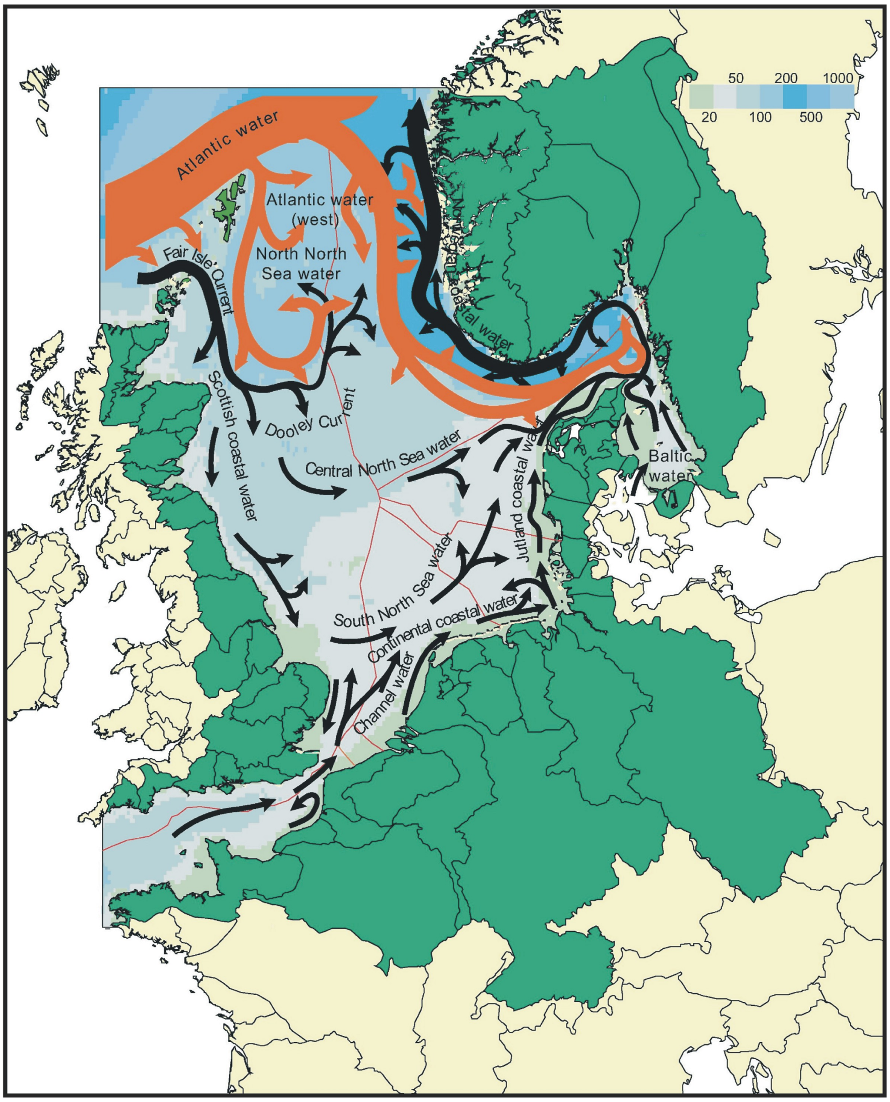


Figure 2.4: Schematic diagram of the general circulation in the North Sea (after Turrell 1992). In dark green the North Sea catchment area.

In the 1970's adverse public reaction over the poor state of coastal/marine environment triggered a series of EC Directives, with the combined aim of restoring lost quality states and creating favourable conditions and context for future sustainable utilisation of coastal and marine resources (Nunneri et al., in press). Ledoux et al. (2005) summarised the evolution of the North Sea policy-making regime, which represents an ongoing search for the most appropriate (i.e. efficient, effective and fair) mechanisms for controlling and monitoring pollution within the aquatic environment (Ledoux et al., 2005; de Jong, 2006). In addition to many EU directives relevant for the North Sea (e.g. the Birds and Habitat Directive, the Nitrate Directive, the Water Framework Directive), there are currently a number of international agreements and legislative frameworks in place for protecting and improving the North Sea environment by managing excessive anthropogenic pressures (e.g. OSPARCOM, the North Sea Conference, the Trilateral Wadden Sea cooperation). The established strategy relies significantly on a combination of technology-based emission standards and quality/risk standards buttressed by a growing recognition, given the uncertainties, of the need for a precautionary (risk averse) approach. The precautionary principle has been explicitly stated in international environmental agreements such as the 1990 Ministerial Declaration on the North Sea and Maastricht Treaty of the European Union (Nunneri et al., in press).

In this study the focus is set first upon an 'historical' issue for the North Sea, namely eutrophication and then upon the possible effects of a new utilisation of the North Sea area for power generation by means of offshore wind farms. In the following sections the two issues are briefly described.

2.5.1 Case-study eutrophication

Eutrophication results from nutrient enrichment, primarily nitrogen and phosphorus. In the North Sea, due to the prevailing circulation patterns, eutrophication affects mainly the coastal zones, in particular the UK estuaries, the Dutch Coast and the German Bight (de Jong, 2006; Colijn and Reise, 2001). The coastal waters of the Southern North Sea receive large quantities of nutrients of anthropogenic origin via rivers and the atmosphere (i.e. from agricultural run-off, sewage systems and gas emissions). Major rivers that drain into the Southern North Sea include the Elbe, the Weser, the Ems, the Rhine and Meuse, the Scheldt, the Thames, and the Humber.

Eutrophication is apparent in the coastal zone in the form of undesirable qualitative changes in the structure and functioning of the planktonic ecosystem, and can be observed in the form of occasional accumulation of foam on beaches. National and international regulations on sewage treatment facilities and farming practises aiming at the reduction of nutrient supply to the coastal sea have already been implemented in the various countries which border the North Sea (for an historical perspective of eutrophication see de Jong, 2006). Although ecological quality objectives for a non-disturbed ecological state have been agreed upon, the scientific knowledge needed for estimating nutrient reduction levels leading to those objectives is still lacking (de Jong, 2006, page 229 and following).

In the North Sea marine/coastal areas, the effects of anthropogenic, biochemical and abiotic controls upon eutrophication, together with their consequences on single species and entire foodwebs have been monitored over several decades (Bot et al., 1996; Colijn, 1992, 1998). The increased primary production can result in shifts of algae species composition; moreover, the sinking of dead organic material to the bottom and the associated bacterial degradation can lead to oxygen depletion bottom waters, thus causing the death of sessile benthic organisms. The historical development of the eutrophication status within the German Bight –an area sensitive to elevated nutrient inputs (Colijn et al., 2002a)– can be derived from the Helgoland road data, a dataset covering 40 years of observations. Radach et al. (1990), Radach (1998), Hickel et al. (1993) have related the increased flagellate concentrations in the water to the enhanced nutrient concentrations. This implies a more common occurrence of toxic algae and a regular blooming of the nuisance algae Phaeocystis, leading to beach foams and increased organic carbon deposition to sediments (Lancelot et al., 1982). In spite of a significant reduction in phosphate emissions, flagellate abundance does not seem to decrease, while Phaeocystis appears to be inefficiently grazed by zooplankton (Jickells, 1998). The modification of phytoplankton succession might reflect changed N/P ratios due to eutrophication as well as alternating hydrographic regimes, possibly triggered by the North Atlantic Oscillation (NAO) (Wirtz and Wiltshire, 2005). A study by Philippart et al. 2007 has shown that long-term variations in limiting nutrients (phosphate and silicon) were weakly correlated with biomass and more strongly with community structures of phytoplankton, macrozoobenthos and estuarine birds. The authors affirm that nutrient enrichment and subsequent nutrient reduction are at least partly responsible for the observed trends in these communities.

In general, anthropogenic nutrient emissions into the North Sea increased until the mid 80s (Brockmann et al., 2003). To protect the marine environment, the Second North Sea Conference in 1987 agreed to reduce anthropogenic nutrient inputs into the North Sea to 50% of the 1985 level by 1995. This decision was endorsed in the Action Plan by the OSPAR Ministerial Meeting, which elaborated reduction measures (OSPAR, 2003b). The OSPAR targets for nutrient reduction have been achieved for P but not N: the reduction of inputs in the North Sea can be assessed at about 20% reduction, compared to the 1985 levels (OSPAR, 2003b). In 2003, Denmark was the only Contracting Party having achieved the overall reduction in nutrient inputs by 50% compared to the input levels in 1985 (OSPAR, 2006). According to OSPAR 2006 atmospheric deposition of nitrogen is estimated to amount to one third of all nitrogen inputs in the Greater North Sea. Although riverine and direct inputs of nitrogen have reduced substantially between 1990 and 2001 by about 10% and 30% respectively, there has been no similar reduction in the total amount of nitrogen deposited from the atmosphere; there was also a substantial reduction (ca. 33%) in direct discharges of phosphorus, but no statistically confirmed conclusions could be reached on riverine inputs. For smaller areas of the Greater North Sea, the situation with regard to trends in waterborne inputs varies considerably with some significant upward trends in riverine inputs and/or direct discharges of nitrogen (Channel, Belgian and Dutch coast, Norwegian West coast) and of phosphorus (Channel, UK East coast, Skagerrak, Norwegian West coast), which 'still need to be confirmed and their reasons established' (OSPAR, 2006; ICES Advisory Committee on Ecosystems, 2006).

From OSPAR's point of view 'it is difficult to ascertain the effects of nutrient emission reduction s in the sea and of the incidence of eutrophication effects, and whether the reduction target of 50% is sufficient to progress towards the Strategy's objective. OSPAR will need to address these questions on the basis of model-based assessments' (OSPAR, 2006).
In a recent report of the ICES workshop about time-series data related to eutrophication (ICES Advisory Committee on Ecosystems, 2006), the invited experts agreed upon the difficulties in

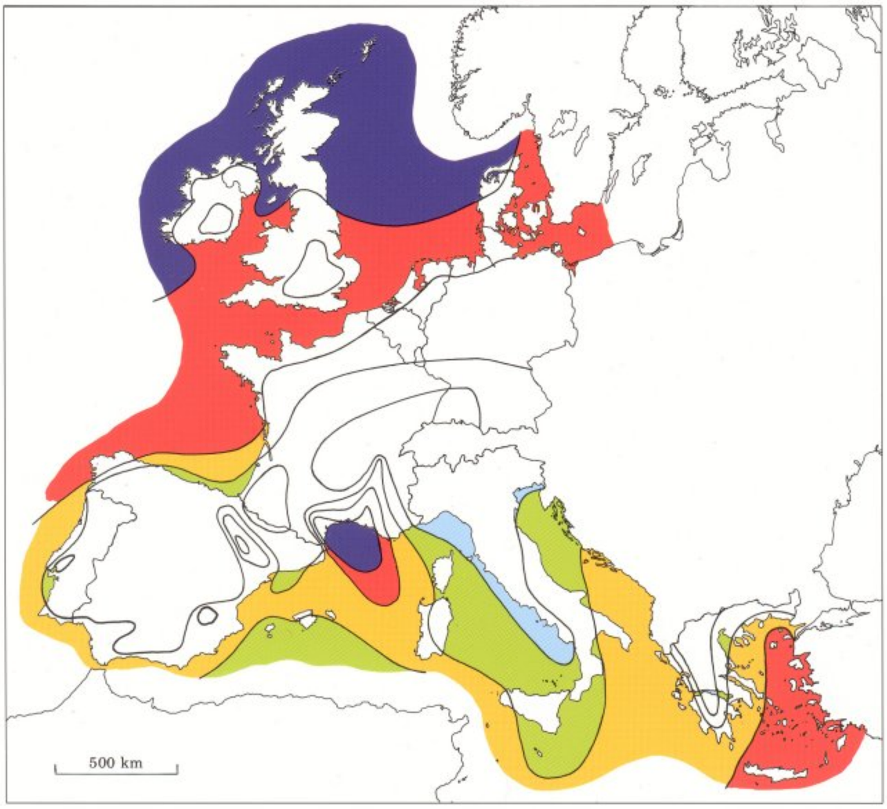
identifying threshold levels in the driving forces creating ecological impacts. Among these, the complexity surrounding determination of a single threshold level or combination of parameter levels that can be generally useful for indicating or predicting ecological impacts has been highlighted.

In this study, the DPSIR scoping framework is applied to eutrophication in the coastal waters influenced by Elbe river and to the international area of Southern North Sea with the aim of finding possible strategies for nutrient reduction and to apply the concept of ecological risk to different nutrient reduction targets. The detailed analysis is set out in chapter 3.

2.5.2 Case-study offshore wind farms

Wind energy is in many aspects among the most commercially advanced renewables, with steadily increasing installed capacity worldwide during the last decade. The construction of offshore wind farms can be seen as a new and relative extensive use of coastal and offshore sea areas. Installing wind turbines offshore has a number of advantages compared to onshore development. Onshore, difficulties in transporting large components, limited area for accomodation of large projects and opposition due to various siting issues (e.g. visual impacts and noise), can limit the dimensions and number of wind farms. Offshore locations can take advantage of the high capacity of marine shipping and handling equipment, which far exceeds the lifting requirements for multi-megawatt wind turbines. In addition, the winds tend to blow faster and smoother at sea than on land, yielding more electricity generation per square meter of swept rotor area (IEA, 2006). Larger onshore wind farms tend to be in remote areas, so electricity must be transmitted over long power lines to inhabited centres, while offshore wind farms can be closer to coastal cities thus simplifying some transmission issues, yet far enough away to reduce visual and noise impacts (IEA, 2006). Good wind resource, proximity to load centres, and expansion of development areas are some of the reasons why development of offshore wind energy is moving forward. By the close of 2005, there were 804 MW of offshore wind power operating in Denmark, the Netherlands, Sweden, and United Kingdom (IEA, 2006). However, initial high investment costs, sub-sea cables to shore, difficult access to the turbines resulting in higher maintenance cost and harsher environmental conditions due to salt water and additional loads from waves and ice are more severe at sea. Those pose both a challenge to technology development and a need of financial support.

In the North Sea area wind resources are quite large (see fig. 2.5), and therefore construction of large wind farms is judged to be cost-effective for achieving the EU Kyoto commitments (EC, 2006) and the targets for renewable energy formulated in the RES-directive (EC, 2001). The EU sees in (offshore) wind one of the most promising technologies for pushing renewable energy growth (EC, 2004). According to Söker et al. (2000), Europe has the technical potential to cover its entire demand for power with offshore wind energy alone, but in order to attain the goals for climate protection agreed in international protocols, wind energy must be developed as 'a matter of urgency'. Countries in the Southern North Sea have in particular large projects for offshore wind farms in their relatively shallow coastal waters or exclusive economic zones (EEZ). In the context of such an emerging sector, societal perceptions and power structures play a key-role in the definition and implemention of successful legislative frameworks for offshore wind development.



Wind resources over open sea (more than 10 km offshore) for five standard heights										
	10 m		25 m		50 m		100 m		200 m	
	$m s^{-1}$	Wm^{-2}	$m s^{-1}$	Wm^{-2}	$m s^{-1}$	Wm^{-2}	$m s^{-1}$	Wm^{-2}	$m s^{-1}$	Wm^{-2}
	> 8.0	> 600	> 8.5	> 700	> 9.0	> 800	> 10.0	> 1100	> 11.0	> 1500
	7.0-8.0	350-600	7.5-8.5	450-700	8.0-9.0	600-800	8.5-10.0	650-1100	9.5-11.0	900-1500
	6.0-7.0	250-300	6.5-7.5	300-450	7.0-8.0	400-600	7.5- 8.5	450- 650	8.0- 9.5	600- 900
	4.5-6.0	100-250	5.0-6.5	150-300	5.5-7.0	200-400	6.0- 7.5	250- 450	6.5- 8.0	300- 600
	< 4.5	< 100	< 5.0	< 150	< 5.5	< 200	< 6.0	< 250	< 6.5	< 300

Figure 2.5: Offshore wind speeds across Europe (Troen and Petersen, 1989).

For this case study the DPSIR approach is applied in order to scope the policy networks behind offshore wind development and the impact connected with the realisation of planned projects. In some sense those aspects are already, in the DPSIR terminology, a socio-political ‘response’ to an environmental issue, namely climate change. In the light of possible future catastrophic events, the need of reducing greenhouse gas emissions for decreasing the risks associated with global warming requires giving priority to renewable energy in the political agenda. However, there are new environmental concerns connected with offshore wind construction in relation with the effects of exploration, construction, and operation upon the marine habitat and organisms (Elliott, 2002). Not only direct effect upon the ecosystem could cause considerable impacts, but also changed morphology, hydrology or even indirect changes produced upon these by reduced wind velocities have been mentioned (e.g. Köller et al., 2006). In this sense the response to an existing environmental issue is the starting point for a new application of the DPSIR, as it involves changing the environment and therefore possibly triggering new impacts.

In this study some key-aspects of the development of offshore wind farm scenarios are analysed;

in particular, the concept of ecological risk is applied for assessing impacts during the construction phase only. Moreover, offshore wind represents a new use of the sea on top of other uses, so the cumulative effects of eutrophication and construction of offshore wind farms are assessed in terms of ecological risk.

3 Analysis

In this chapter the analysis of the two case-studies is reported along the DPSIR scoping framework. The issue of eutrophication is examined in section 3.1 and the construction of offshore wind farms in section 3.2.
Figure 3.1 summarises the essential considerations taken into account for this study and reviewed in chapters 1 and 2.
The application of the DPSIR scoping framework for supporting decision-making cannot ignore the gap existing between TEV and TSV, when evaluating potential loss of environmental services and goods, in this context, the concept of ecological risk has been developed as an alternative to critical load and safe minimum standards approaches under uncertainties (see sections 3.1.2, 3.2.3 and 3.3). Incorporating risk appraisal into decision-making allows policy-makers to consider additional information, when assessing management option trade-offs (e.g. in sections 3.1.2 and 3.2.4). Scenarios are used for scoping possible future states of the 'world' and the related changes in selected environmental variables in order to determine 'preferable' development paths or management strategies (see sections 3.1.1 and 3.2.2). When envisioning the future, development-choices need to take into consideration a multiplicity of interests represented by various actors and stakeholder (e.g. section 3.2.1), who may play a relevant role by exercising their influence and power. Appraisal of gains and losses at different spatial scales allows designing compensation mechanisms and thereby achieve broader commitment in management implementation (see section 3.1.3). In the context of decision-making stakeholder dialogue cannot be ignored, with successful policies often being the result of negotiation among different parties.

3.1 Eutrophication

'After half a century of failing attempts to develop yardsticks to judge human impacts on the ecosystems, or instruments for predicting ecosystem developments, to be used by politicians and administrators' (de Jong, 2006, page 285) there is a need to make uncertainties explicit, as well as to actively involve stakeholders into decision making (Köhn, 1997). De Jong (2006) affirms, in his historical perspective about eutrophication, that the OSPAR target of 50% reduction of N and P in the North Sea would have never been adopted, had additional stakeholders been admitted into the policy-making process. The analysis reported in this section shows how scenarios and modelling can be deployed to address relationships between reduction scenarios and ecosystem state. It also identifies ways in which participatory approaches can enrich decision-making and assessment of alternatives and how uncertainties can be addressed by adopting the concept of ecological risk.

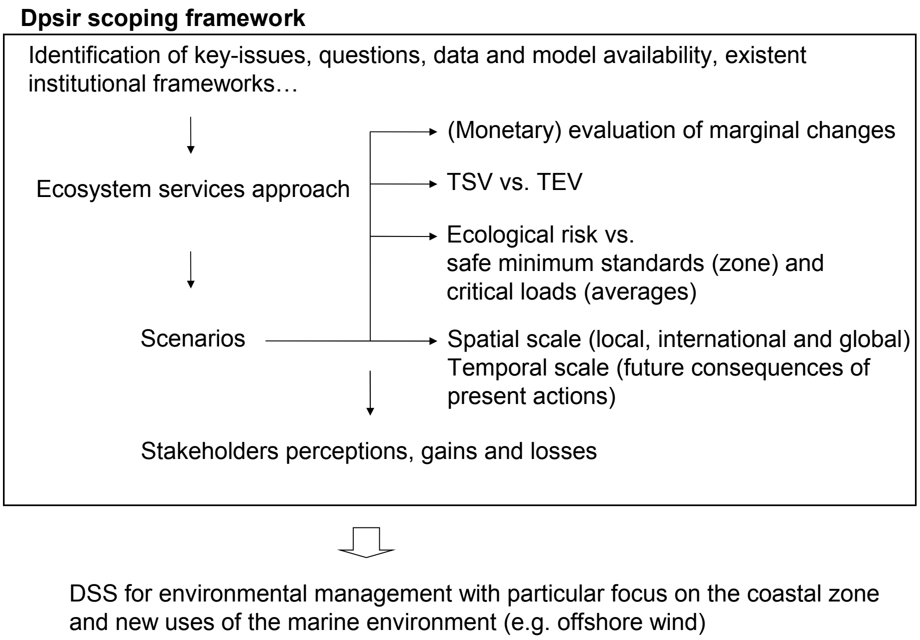


Figure 3.1: Application of the DPSIR as Decision Support System.

This section includes three papers already published in ISI journals[1].

The paper reported in section 3.1.1 analyses alternatives for nutrient emission reduction in the Elbe River Catchment with the ultimate aim of reducing eutrophication of the German Bight coastal waters. Based on scenarios and modelling, this paper shows how different emission reduction measures can be evaluated with respect to their ecological, economic and social consequences by means of a multi criteria analysis. Moreover, the basis for the ecological risk assessment methodology has been set within this study.

Section 3.1.2 reports a paper dealing with the international issue of eutrophication in the Southern North Sea. It shows how the ecological risk concept can be used for assessing ecological benefits (in term of ecological risk reduction) associated with nutrient reduction scenarios and how the combined assessment of ecological risk and reduction measure implementation costs can be used for informing international decision-making and as support for negotiation in environmental matters.

The paper reported in section 3.1.3 is focused again on the Elbe River Catchment and deals with view and perceptions of stakeholders about possible nutrient emission reduction measures. This paper shows how perceptions of an issue may vary across stakeholders and touches the issue of ‘cryptic’ decision-making: transparency, participatory approaches and adequate burden sharing agreements are perceived as necessary for ensuring commitment in legislation implementation.

[1] Please note that the original papers have been slightly modified in order to fit the structure of this document.

3.1.1 Driver/Pressures: eutrophication scenarios and their use

The study reported in this section[2] shows the complete DPSIR assessment cycle from scenario study to modelling of possible ecological changes and finally to evaluation of those changes by means of a multi-criteria analysis. It also deals with a pragmatic approach researchers and practitioners often need to adopt when dealing with environmental issues: namely that of combining different analytical tools (modelling, participation and policy-relevant socio-economic evaluation of alternatives). It shows how different elements of the analysis can be combined into an interdisciplinary consistent study. The contribution of this paper within the present work is related to cause-effect scoping of relationships between drivers, pressures and environmental state. The paper mainly deals with the relationships between drivers and pressures, linking human action with ecological changes. The definition of socio-economic scenarios is the basis for 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.

This study concentrates on the Elbe catchment and its coastal waters. The overall goal is to address the eutrophication issue by analysing the physical unit catchment-coastal zone. Inland anthropogenic activities (e.g. land use and anthropogenically induced matter fluxes) that influence (through river systems) the ecological quality and the socio-economic service functions of adjacent coastal zones are identified, and possible reduction measures and their environmental, economic and social effects are assessed.

Within the DPSIR scoping framework, scenario assessment is the starting point for the analysis: 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 and aggregated impact indicators corresponding to the demand of the WFD (Janssen et al., 2001). In order to achieve 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 state) 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?

[2]This section reports an excerpt of the study published in Regional Environmental Change (2005, vol. 5, n. 2-3: 54-81) as: J. Hofmann, H. Behrendt, A. Gilbert, R. Janssen, A. Kannen, J. Kappenberg, H. Lenhart, W. Lise, C. Nunneri and W. Windhorst, 'Catchment–coastal zone interaction based upon scenario and model analysis: Elbe and the German Bight case study'.

After a brief description of the Elbe catchment, estuary and coastal zone, as well as of the models used for this study, the analysis focuses first on the catchment and, by scenario assessment on possible reduction strategies and their effects in the coastal zone.

Brief description of the Elbe catchment-coastal zone

The Elbe River Basin (148,268 km^2) covers large parts of two central European countries: about two-thirds of the catchment area belongs to Germany and one-third to the Czech Republic (about 27% of Germany and 63% of the Czech Republic territory belong to the Elbe catchment). The catchment stretches through different geographical regions, from middle mountain ranges to large flatlands and lowlands, including areas of interregional importance for plants and animals otherwise endangered (fig. 3.2). The total population of the catchment is about 24 millions, with a mean density of 167 persons/km^2 (58% of the Czech citizens and 22.9% of the German citizens live in the Elbe catchment). About 61% of the catchment is used for agriculture; forests cover 29% and 6% is urban area (data from CORINE land cover, see fig. 3.3). Different economic activities such as agriculture, drinking water supply, industry and tourism often represent conflicting interests. The changes occurred after the German reunification in 1989/1990 resulted in regionally differentiated socio-economic patterns throughout the catchment. Due to anthropogenic activities a large number of different substances both human-made and of natural origin enter the river and are transported to the North Sea. Due to the prevailing circulation patterns in the North Sea, the river Elbe waters are received in the south-eastern corner of the North Sea, namely the German Bight (the coastal zone definition is based primarily on the boundary conditions of the ERSEM model, see figure 3.4).

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 (see figure 2.4, on page 22), which has an eastward and a northward component in the German Bight. 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 (e.g. Weser and 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 to 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. 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,

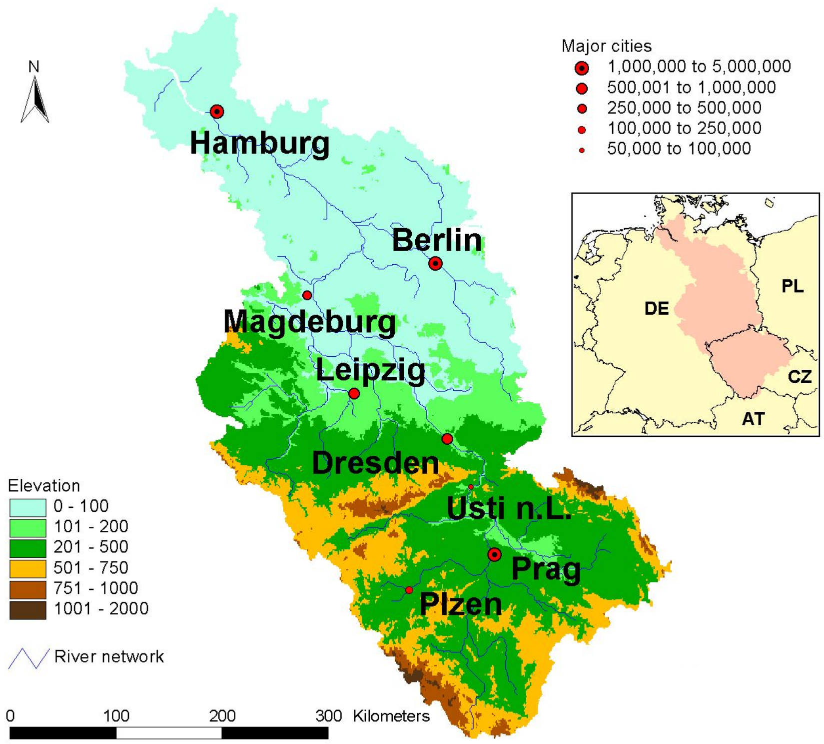


Figure 3.2: The Elbe river basin (Hofmann et al., 2005).

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; van Beusekom et al., 2001). Meteorological and hydrographic conditions can 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; Niermann, 1990).

The Wadden Sea area of Schleswig-Holstein is the main impact area for the coastal investigations of the Elbe case study and can generally be characterised as a rural area with a low population density (80 inhabitants per km^2 for Nordfriesland and 94 for Dithmarschen, compared to 229.4 inhabitants per km^2 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. Eutrophication might represent a threat, menacing biodiversity and auto regulation of the ecosystem on the one side, having potentially negative effects on economy (tourism, fishery) on the other side. 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

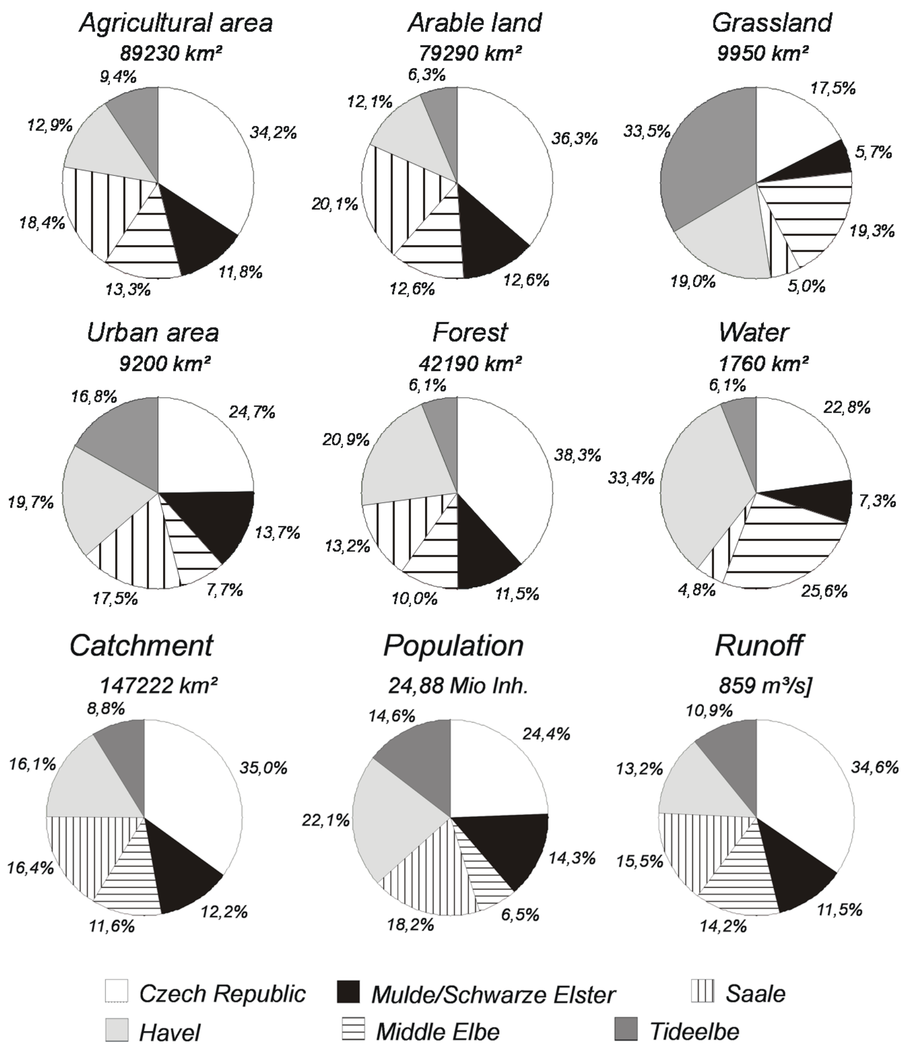


Figure 3.3: Top: Land use in the different regions of the Elbe river basin (the German part is regionalised according to the WFD coordination regions); Bottom: Share of the German part (WFD coordination regions) and the Czech part with regard to catchment, population and mean runoff (1993-1997) within the Elbe river basin (Hofmann et al., 2005).

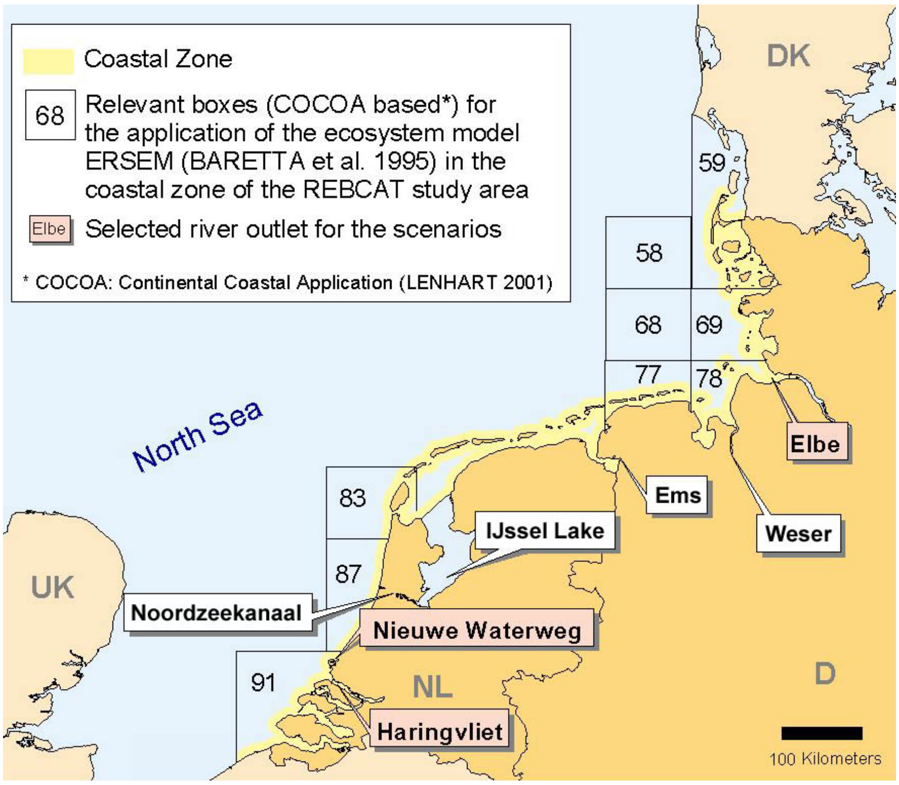


Figure 3.4: The Elbe river coastal zone as represented by the ERSEM model boxes of ERSEM. Note that the integrated Elbe box comprises the boxes 58, 59, 68, 69, 77 and 78. The boxes 83, 87 and 91 are related to the Rhine and were not considered in this study, but for the study reported in section 3.1.2 (Hofmann et al., 2005).

coastal defence and nature conservation, tourism, agriculture, wind farms (onshore and offshore), fisheries, shipping and, to a limited degree, mariculture (Kannen et al., 2000).

Methodological issue: combining different available analytical instruments

Under the DPSIR framework, the assumption is 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 and Hofmann, 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

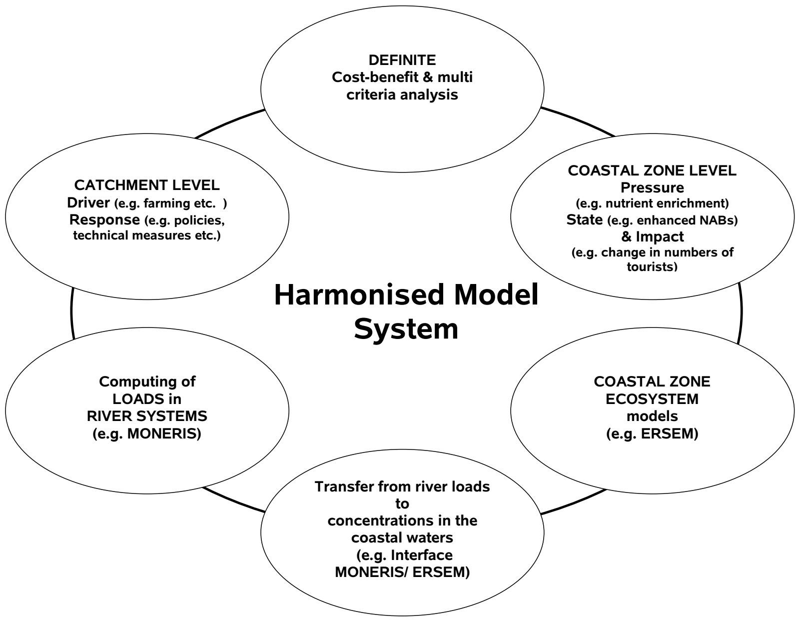


Figure 3.5: Harmonised approach to the modelling and evaluation of eutrophication issues within the Elbe catchment and coastal zone (after Hofmann et al., 2005, slightly modified.

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, cost-benefit or multi-criteria analysis), among a selected pool of measures.

Application of the DPSIR framework The conceptual approach used in EUROCAT to operationalise the DPSIR framework links scenarios, indicators and modelling (figure 3.5). In principle, all elements of the DPSIR approach, namely drivers, pressures, state, impact and response, need to be described by relevant indicators, which can be calculated by models (for forecasting possible future changes) and/or measured for monitoring. The set-up of a regional Policy Advisory Board (PAB) for the Elbe provides a suitable involvement of stakeholders, i.e. users and managers (see section 3.1.3).

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., 2002b; 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., 2002b). Table 3.1 shows the set of potential indicators defined by EUROCAT team members. The assessment approach 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 5^{th} International Conference on the Protection of the North Sea in Bergen in 2002 (North Sea Conference, 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.

For assessing the effects of socio-economic scenarios in the coastal zone, the following parameters derived from ERSEM 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 can be considered for assessing the state of the coastal zone (table 3.1). In addition to those, also the integrity indicators reported in table 2.1 are assessed. The selection of those parameters for assessing ht estate of the coastal zone 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. (2005) assumed that the ecological integrity according to the definition chosen by Barkmann and Windhorst (2000) and Barkmann et al. (2001), has the potential to serve as an integrating approach coupling structures and processes of ecosystems (see section 2.3).

Modelling tools and information exchange This assessment presents an holistic strategy encompassing 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 (see figure 3.5), the actual flow of information and the model in use is illustrated in figure 3.6. In order to address the related questions, the following modelling tools are used:

Table 3.1: Selected primary indicators for eutrophication in the coastal zone (Hofmann et al., 2005).

Indicator	Metrics
NO_3 concentration, NH_4 concentration, NO_2 concentration	$\mu mol\, l^{-1}$ in winter (December-January; time of lowest pelagic biological net production)
PO_4 concentration	$\mu mol\, l^{-1}$ in winter (December-January; time of lowest pelagic biological net production)
N/P ratio	–
Chlorophyll-a concentration	$\mu g\, l^{-1}$, mean summer (3 summer months)
Chlorophyll-a concentration	$\mu g\, l^{-1}$, spring bloom average (1 month around peak)
Timing of spring bloom	Week in the year, $\pm$ 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	mgl^{-1}, bottom water in summer
Diatom/non-diatom ratio (phytoplankton species composition)	Biomass ratio in growing season (March-September)
Phaeocystis: length of bloom	Length of period with $> 10^6\, cells\, l^{-1}$
Net primary production	$gC\, m^{-2}\, year^{-1}$
Export of organic material (flux)	$gC\, m^{-3}\, day^{-1}$

- 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),
- CENER for calculating cost-effective nutrient emission reductions by measure implementation in the catchment (Lise and van der Veeren, 2002; Lise, 2003), and
- a multi-criteria analysis (MCA) 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. For this study, the calculated nutrient loads by MONERIS modified by the transfer function are applied to the ERSEM modelling, the daily nutrient loads are used as calculated by Lenhart and Pätsch (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. Furthermore, coastal state indicators are extracted from ERSEM (see figure 3.6).

Scenario assessment

The starting point of analysis is scenario assessment. 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, for the catchment emissions, and by ERSEM, for the ecological effects in the coastal zone. Following the DPSIR approach, the next level of analysis focuses on several essential fields of social life of

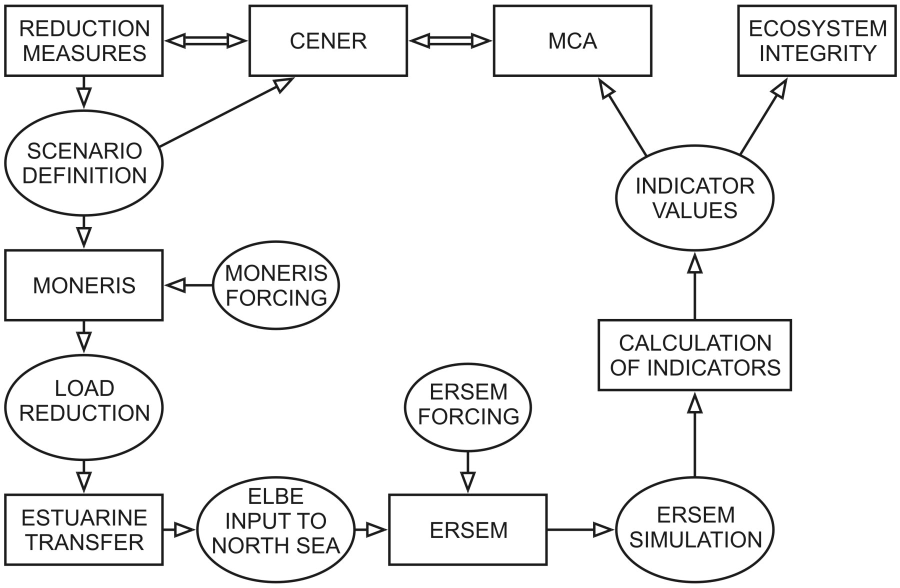


Figure 3.6: Model integration and information exchange (Hofmann et al., 2005).

the socio-economic system (drivers) which, in turn, depend on scenario assumptions (values and priorities). Scenarios are the framework for evaluating different management measures aiming at nutrient emission reduction in the Elbe catchment. Particularly, the goals for nutrient reduction will be different under different scenarios, due to the different priorities (maximum under the third scenario, and minimum under the first one). The three scenarios, briefly outlined below, represent, two extremes and a middle future, which might also be seen as the most likely one, named Strong EU. 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 sociocultural situation, namely (1) market liberalisation and the related consumerism and globalisation; (2) central leadership of the EU, and (3) as a counter-trend to globalisation, a trend towards regionalised and self-sufficient 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 and Hofmann, 2002). The storylines are briefly reported below.

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.

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.

Deep Green scenario (DG) angered by BSE crises, food contamination scandals (such as the nitrophene 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 regionalised 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 scenarios are described along six dimensions (Governance, Lifestyle, Social values, Relevance of the EU, Economy and Environment), which are inter-dependent (Nunneri and Hofmann, 2002). 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). Different attitudes can be found throughout the scenarios with respect to interactions between marginal costs of ecosystem conservation and ecological risks (see section 3.1.2). A high level of self-organising capacity, e.g. high ecosystem integrity, minimises the risk that the ecological system fails to provide the minimum level of natural resources needed by human societies. It is additionally assumed that, with an increasing use of ecosystem services, socio-economic risks decrease as the resource availability increases. In parallel, however, the ecosystem integrity is decreasing as well, causing increasing ecological risks (Windhorst et al., 2005). The increasing occurrence of anoxic zones can be taken as an example for ecological risks in coastal zones (Rachor and Albrecht, 1983; Niermann, 1990). Risk reduction requires the reduced use of ecosystem services, and thus the implementation of management measures. 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., 2005). The scenario settings described above are in agreement with those used in other projects like GLOWA Elbe[3] 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.

[3]Global Change in the Hydrological Cycle, http:// www.glowa.org/eng/elbe.

Modelling nutrient emissions in the river catchment

The model MONERIS was developed for the estimation of nutrient inputs via various point and diffuse pathways in German rivers with catchments larger than 500 km^2. 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. A detailed description of MONERIS is given by Behrendt et al. (2000), a brief overview of the model is given in the following. While the inputs from municipal wastewater treatment plants (WWTPs) and from industry are directly discharged into the rivers (point sources), 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 (diffuse sources). Among them one of the most relevant is agricultural land-use, assessed under the entry 'tile drainage'. 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 figure 3.7) are possible for the catchment areas now covered:

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

For the diffuse inputs, the model takes into account various transformation, loss and retention processes, by means of different sub-models, which are validated by comparing the results with independent datasets. 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 test the effect of management measures. Since the Elbe is a transboundary river, the compilation of a harmonised database, of statistical data about e.g. population, WWTPs and agricultural activities, was an essential task (Hofmann et al., 2005). Based on the availability of monitoring data, the time-series 1983-1987, 1993-1997 and 1998-2001 for nutrient emissions in the Elbe catchment were modelled.

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. Based on 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. 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,

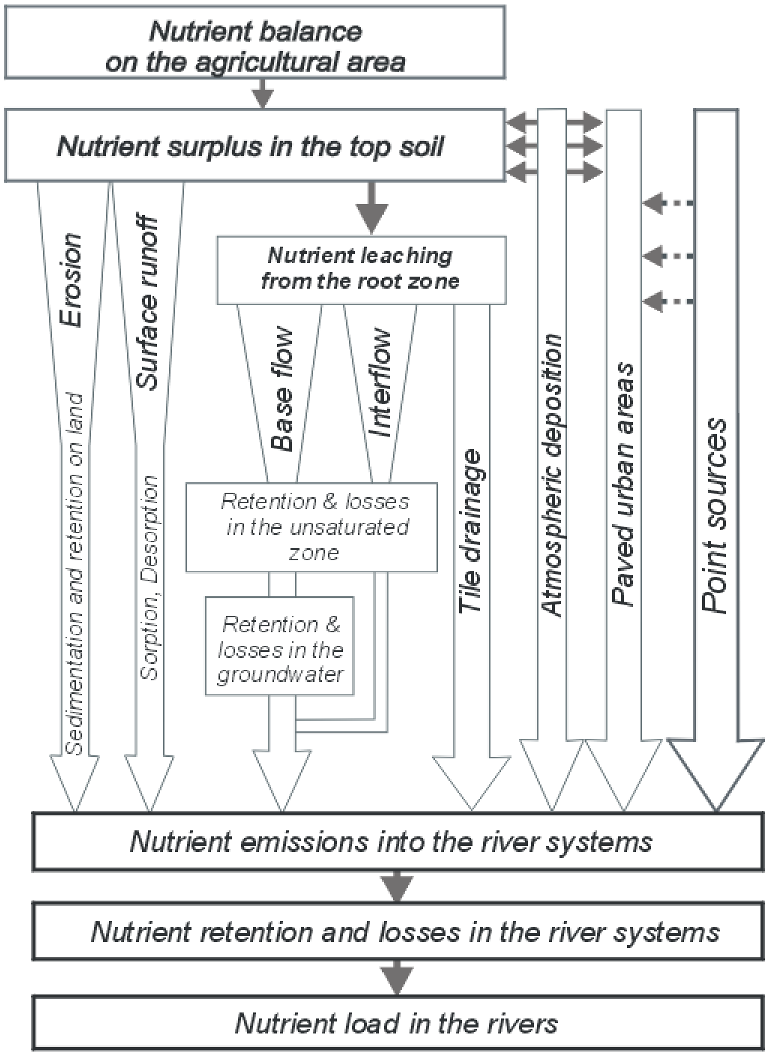


Figure 3.7: Pathways and processes considered in the model MONERIS (Behrendt et al., 2000).

macrophytes and fish) in inland waters are related to nutrient concentrations and not to nutrient emissions or loadings. Knowing the background levels is important to define reference concentrations against which evaluate target levels under human influence. Based 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. The proportion of emissions attributable to agriculture could not be quantified in absence of a background level. 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. 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 kg N ha^{-1} $year^{-1}$ of nitrogen emissions to the air over nitrogen deposition under background conditions, which applies to all regions.
- 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 assumptions, and using MONERIS, it is possible to calculate both nutrient loadings and concentration with background conditions for individual catchments. It has to be borne in mind that the calculated concentrations include a retention factor dependent on the hydrological and morphological conditions in the water bodies, therefore the calculated background values represent an upper limit of expected nutrient concentrations under background conditions. The reduction of emissions achieved since 1983 will be presented as the background context for the issue of further reduction.

Nutrient emission reduction during the last two decades Various measures can directly or indirectly reduce nutrient surplus. Measures can be different in nature (e.g. taxes, emission-quotas or technical requirements) and can apply on different levels (e.g. on EC, national, regional or business level). During the last years the policy focus has been successfully set on reducing emission from point sources, thus obtaining a large reduction in P-emissions (60% in the period 1985-1995) mainly due to the use of phosphate-free detergents and to improvement of WWTPs. On the other hand, until 1999 the emission of N to the North Sea has only been reduced by 35% (BMU, 2002a). In more recent time the focus for reduction has shifted to diffuse sources, particularly regarding N-reduction. Figure 3.8 shows the total N-emission reduction and the reduction in diffuse sources during the periods 1983-1987, 1993-1997 and 1998-2000, which are reported in detail, but only for the German part of the Elbe basin, in tables 3.2 and 3.3. The results in tables 3.2 and 3.3 refer to the German part of the Elbe catchment, as published recently

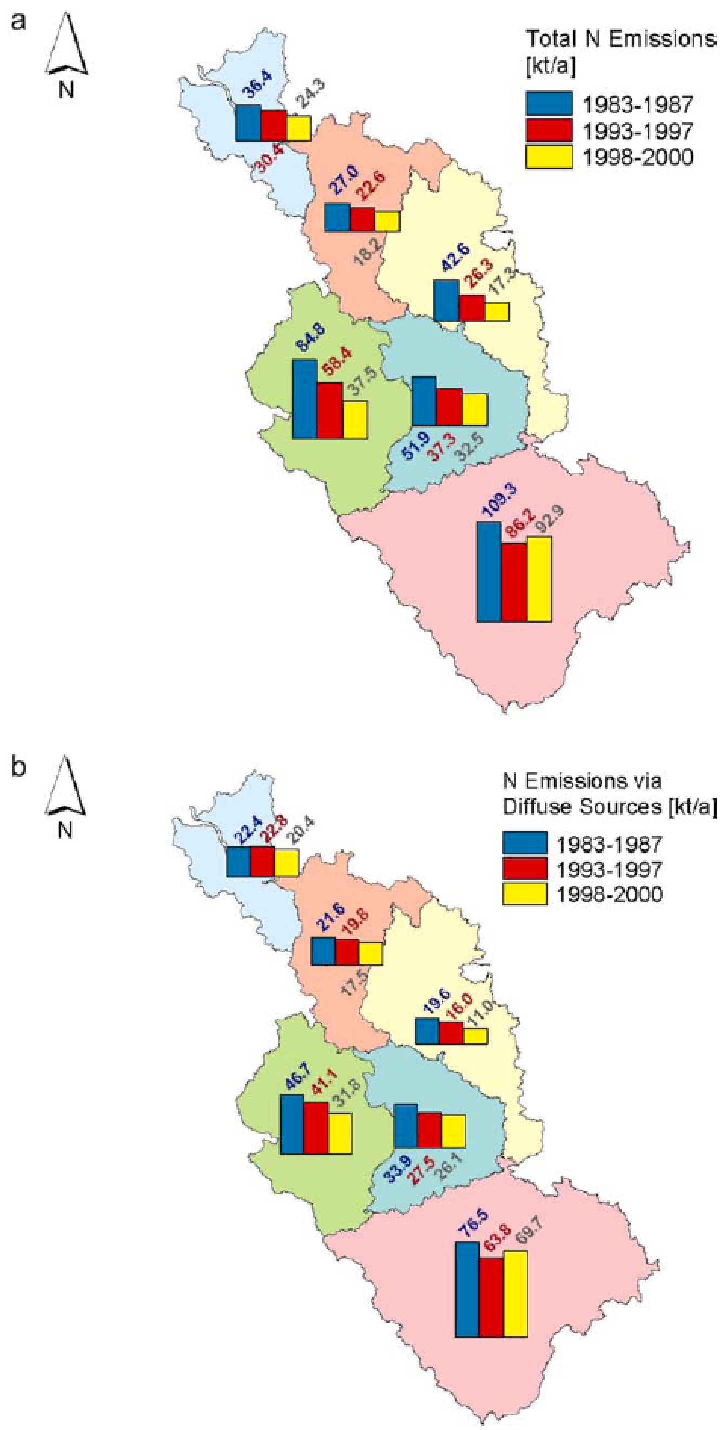


Figure 3.8: Total N emissions (a) and N emissions via diffuse sources (b) in the Elbe catchment as calculated with MONERIS for different regions and different time periods. The columns indicate the periods 1983-1987 (left column), 1993-1997 (middle column) and 1998-2000 (right column). The German part of the catchment is subdivided into the coordination regions foreseen for the implementation Water Framework Directive, while the Czech part is shown as a whole (Nunneri and Hofmann, 2005).

Table 3.2: 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 (data Behrendt et al., 2003; published in Hofmann et al., 2005).

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	
Sum diffuse sources	($t\,year^{-1}$)	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
Sum point sources	($t\,year^{-1}$)	1,255	2,544	12,563	-90
	(%)	22.7	34.7	68.8	
Sum all sources	($t\,year^{-1}$)	5,523	7,341	18,254	-70
	(%)	100.0	100.0	100.0	

by Behrendt et al. (2003).
Total phosphorus emissions into the Elbe river basin were about 5.53 kt P year^{-1} in the period 1998- 2000 (table 3.2). Compared with the period 1983-1987, the phosphorus emissions were reduced by about 12.7 kt P year^{-1} or 70%. Therefore, 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 (table 3.3). 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 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

Table 3.3: 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 (data Behrendt et al., 2003, published in Hofmann et al., 2005.

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	
Sum diffuse sources	($t\,year^{-1}$)	81,000	92,890	134,580	40
	(%)	79.2	69.3	58.3	
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
Sum point sources	($t\,year^{-}1$)	21,290	41,240	96,090	-78
	(%)	20.8	30.7	43.7	
Sum all sources	($t\,year^{-1}$)	102,290	134,130	230,670	56
	(%)	100.0	100.0	100.0	

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 contributors, even though the decrease is 36 and 51% respectively. The main part of all diffuse emissions is caused by agriculture (figure 3.9).
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 regionalisation of nutrient surpluses shows that the P surplus is in general 2-4 kg P ha^{-1} year^{-1}; only some areas in the tidal Elbe show values inferior to 2 kg P ha^{-1} year^{-1}. The N surplus is in general 40-60 kg N ha^{-1} year^{-1}, with higher values in the tidal Elbe of about 80-100 kg N ha^{-1} year^{-1}, and even up to 120 kg N ha^{-1} year^{-1}. Caused by the political changes in 1989/1990, the reunification of

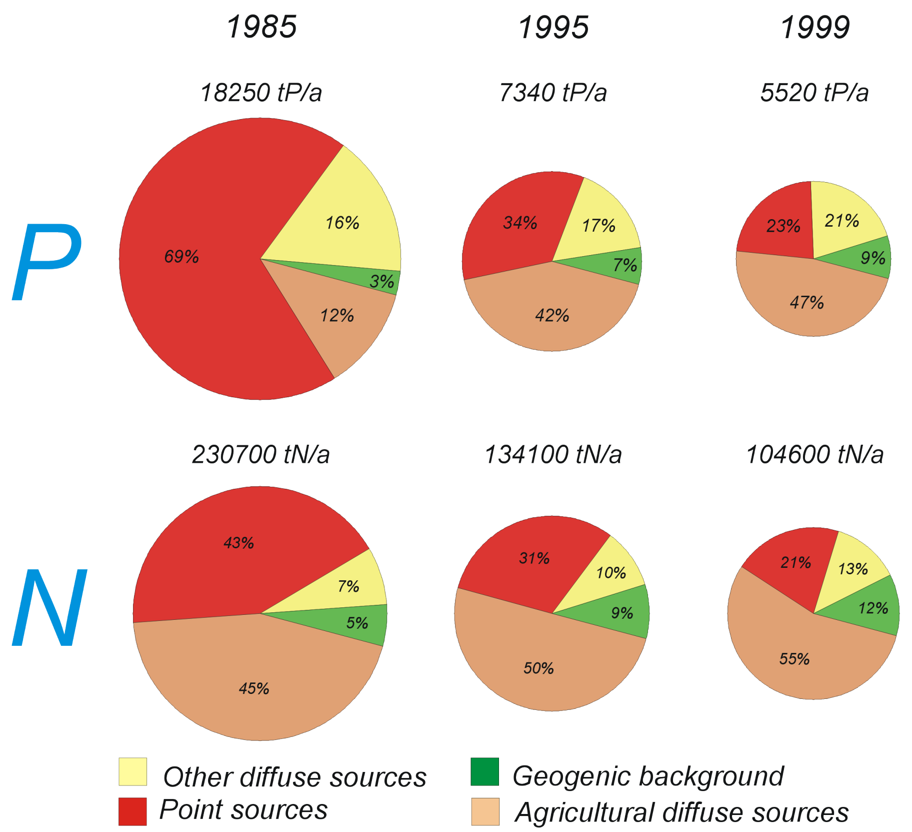


Figure 3.9: Causes of phosphorus and nitrogen inputs in the German part of the Elbe catchment for the years 1985, 1995 and 1999 –periods 1983-1987, 1993-1997 and 1998-2000 (Behrendt et al., 2002a).

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 kg N ha^{-1} $year^{-1}$. 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%.

Future emission reduction in the Elbe basin 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[4]. 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,

[4] Dr. H. Gömann, homepage http://www.faa-bonn.de.

Table 3.4: Nitrogen inputs via various pathway. Explanation of abbreviations in Fig. 3.10 (Hofmann et al., 2005).

Abbreviation	Explanation
1985	Mean value of the total N emissions during the time period 1983-1987
1995	Mean value of the total N emissions during the time period 1993-1997
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)
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)
Lib steady state	Status of N in groundwater at steady state which corresponds to after 2050 continued
Lib + medium measures	Lib 2025 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

- 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, figure 3.10 show different reduction possibilities for total nitrogen emissions (for explanation of abbreviations see table 3.4).

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 figure3.10. 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 3.5. 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 applica-

Table 3.5: 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 (see Gömann et al., 2003; Global Change in the Hydrological Cycle http://www.glowa. org/eng/elbe); source:Hofmann et al., 2005).

Scenario	Measure
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 to 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^3ha^{-1} paved urban for all areas where this was not already $23m^3ha^{-1}$ 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 $23m^3ha^{-1}$ 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 $346km^2$) are additional to the surface waters used for retention; 0.5% of agricultural area (according to Corine $450km^2$) is transferred to retention areas

tion of ERSEM. The values given in table 3.6 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.

Linking MONERIS and ERSEM by a transfer function When following the nutrient destiny from their sources through the river system and finally to the coastal waters, the dynamic model ERSEM needs to be linked with the steady-state model MONERIS. This 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

Table 3.6: 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 (Hofmann et al., 2005).

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

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}) and (4) total phosphorus (TP, t P year^{-1}). ERSEM needs as riverine 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 (t N day^{-1}), (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}) and (7) silicate (t Si day^{-1}).

The fine-tuning of the interface between MONERIS and ERSEM was done stepwise through (1)the definition/calculation of transfer functions for nutrients between MONERIS and ERSEM; (2)the artificial resolution of year cycles (5-year means) in order to derive seasonality (monthly values); and (3) data processing of MONERIS data output to ERSEM data input (more details are found in the published version, Hofmann et al., 2005).

Effects of simulated river load reductions in the coastal waters

For the simulation of the response in the coastal zone caused by changing nutrient loads from the river management strategies, the ERSEM boxes 58 and 59, 68 and 69, and 77 and 78 were chosen for an integrated Elbe box (figure 3.4). This coastal area is nearly identical with the OSPAR regions O-II-3D of the greater North Sea. In this section, the results of the ERSEM simulation for the Elbe region 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 figure 3.11, the DIP concentration is presented for the different scenarios, for the aggregated ‘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 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

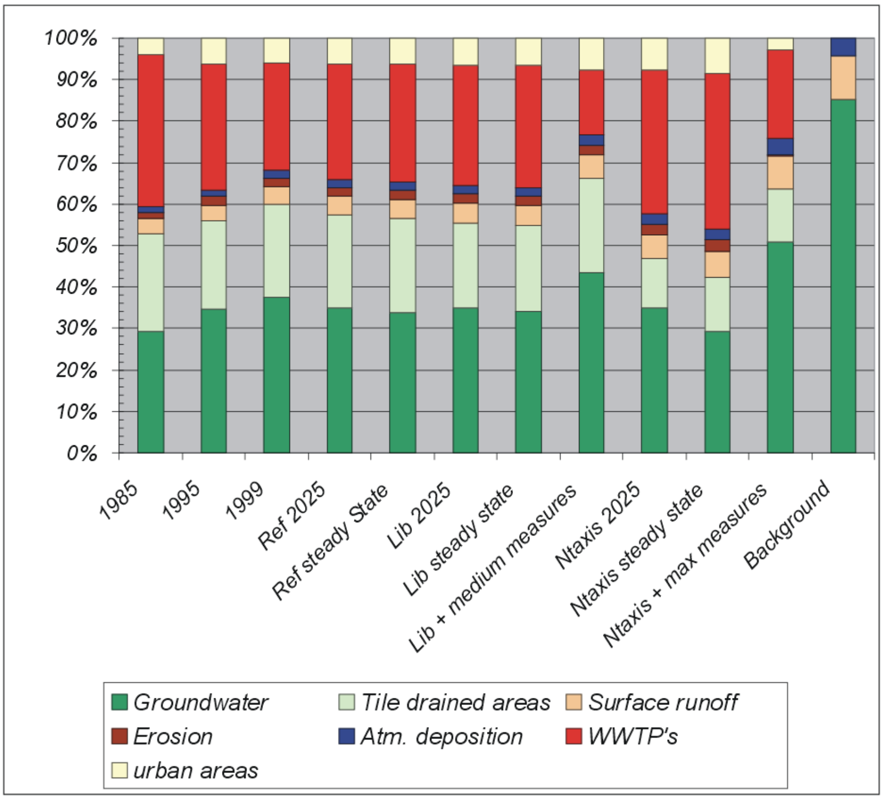


Figure 3.10: Percentage contribution of various pathways in relation to total nitrogen emissions (t N year^{-1}) in the Elbe catchment under different future agricultural policies development (for explanation, see table 3.4). For comparison the time series 1985, 1995, 1999 and the natural background are reported (Behrendt et al., 2002a).

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-series for the aggregated Elbe box (figure 3.12) 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 (figure 3.13). The time series in box 78 show a decrease in the level of the second peak in the spring bloom (figure 3.14). 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 (figure 3.15). In contrast, only the diatom time series for the pristine condition scenario in box 78 (figure 3.16) shows sporadically higher values if compared with all other scenarios, and

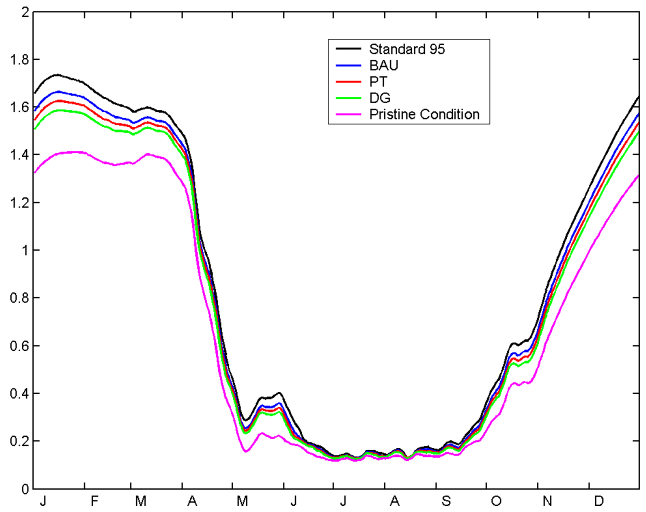


Figure 3.11: Modelled time-series of DIP concentrations (mmol P m3) related to the aggregated Elbe-box for different scenarios (BAU business as usual, PT policy targets, DG deep green) compared to the standard run 1995 and the pristine conditions (Hofmann et al., 2005).

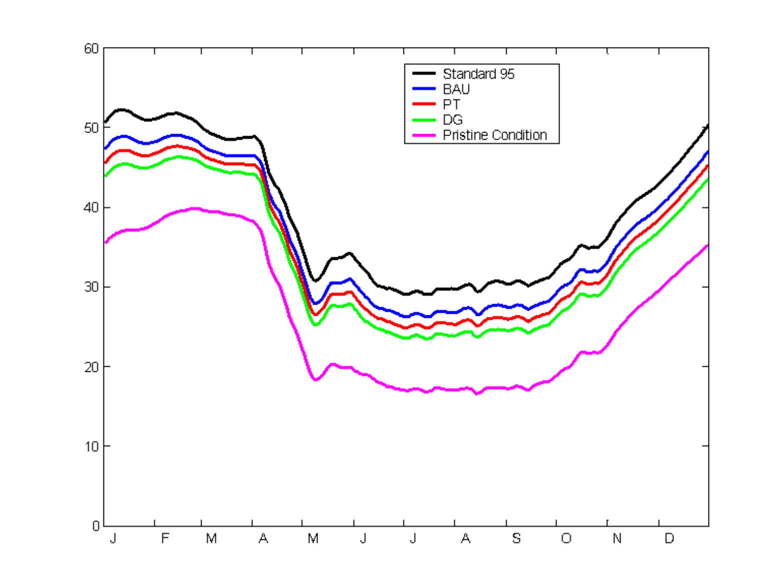


Figure 3.12: Modelled time-series of DIN concentrations (mmol N m^{-3}) related to the aggregated Elbe-box for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions (Hofmann et al., 2005)

Table 3.7: Key parameters related to the standard run and the selected nutrient reduction scenarios, analysed for the aggregated Elbe box (Hofmann et al., 2005).

Key parameters (Elbe Box)	Std. 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	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

with the standard run. This shows that increased production due to eutrophication was mainly based on flagellate production. 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 3.7).

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 loads have on a specific geographic area (OSPAR, 1998). In table 3.7, 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^{st} January and to end on the 31^{st} March. The mean winter DIN/DIP ratio is calculated as (NO_3+NH_4)/PO_4, 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 nutrient and the silicate concentration, whereas the latter was not effected by anthropogenic changes. Based on scenario runs, therefore, the reduced N and P loads will be assessed 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 was 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 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 has not caused an overall increase in the algae biomass, but only 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 ecosystem integrity When interpreting ERSEM results, 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 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 calculated integrity indicator values for the Elbe box mirror that nearly all indicators are sensitive to reduced nutrient loads from the Elbe, but to a different extent (the indicator values are reported for both N and P in table 3.9). 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 (Windhorst et al., 2005). The storage function of the coastal ecosystem changes in relative terms (i.e. its normalised values) more than the other indicators. This confirms that this indicator could reveal essential information about the functioning of the coastal ecosystem. The overall change of the ecological state 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. (2002b) 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 (for an appraisal in terms of ecological risk see section 3.1.2).

Ranking of policy options by applying a multi criteria analysis

In the following, an example of the use of multi criteria analysis (MCA) for evaluating management alternatives for nutrient reduction is given. Please note that these reduction percentages differ from the values in table 3.6, as the values of Lenhart and Pätsch (2001) has been used. 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 3.6. Also, the reference year is 1985, and not 1995. The CENER model (for a detailed description refer to Lise and van der Veeren, 2002; Lise, 2003) is used to calculate cost-effective joint N and P emission reduction policies in the Elbe river basin which achieves a desired reduction in the load to the German Bight. The costs calculated by the CENER model are used as an input to the MCA analysis.

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 the coastal areas, in

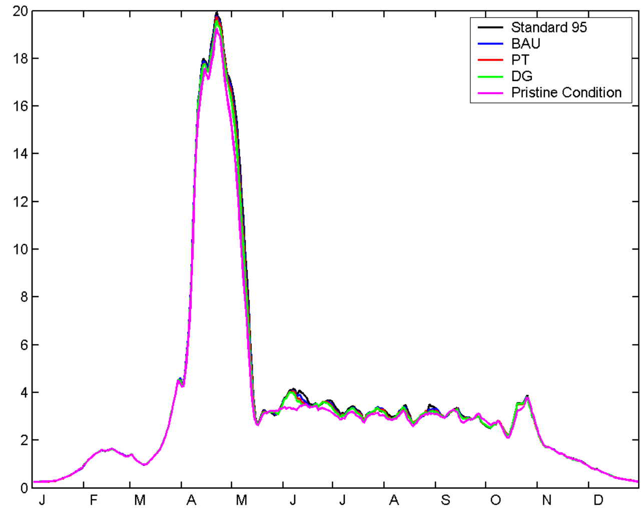


Figure 3.13: Modelled time-series of chlorophyll-a (mg chl m^{-3}) related to the aggregated Elbe box for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions (Hofmann et al., 2005).

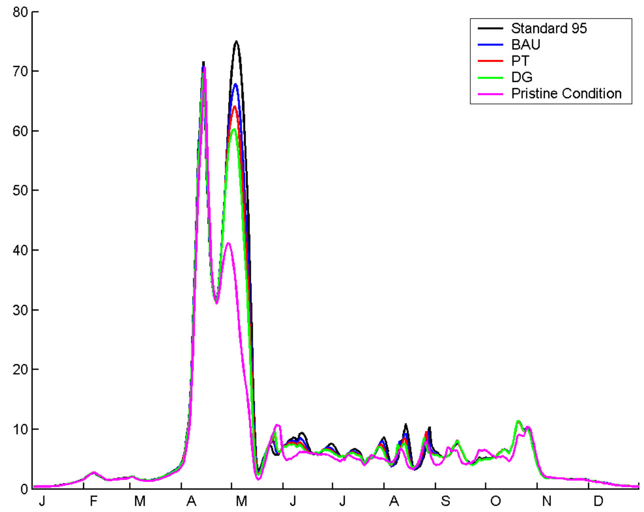


Figure 3.14: Modelled time-series of chlorophyll-a (mg chl m^{-3}) related to input box 78 for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions (Hofmann et al., 2005).

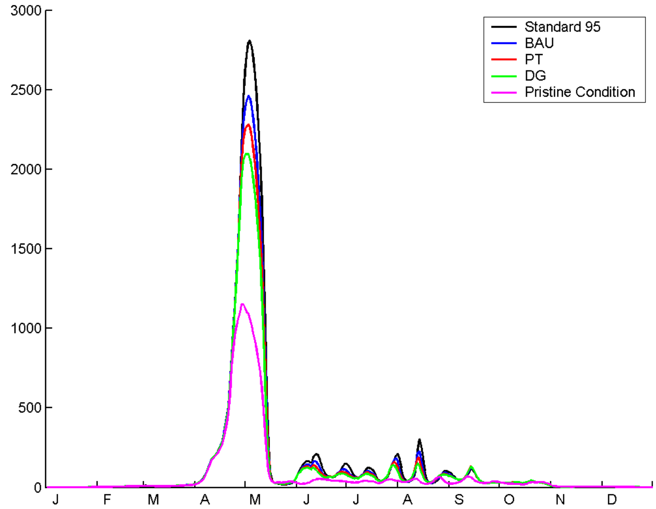


Figure 3.15: Modelled time-series of flagellate concentration (mg C m^{-3}) related to input box 78 for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions (Hofmann et al., 2005).

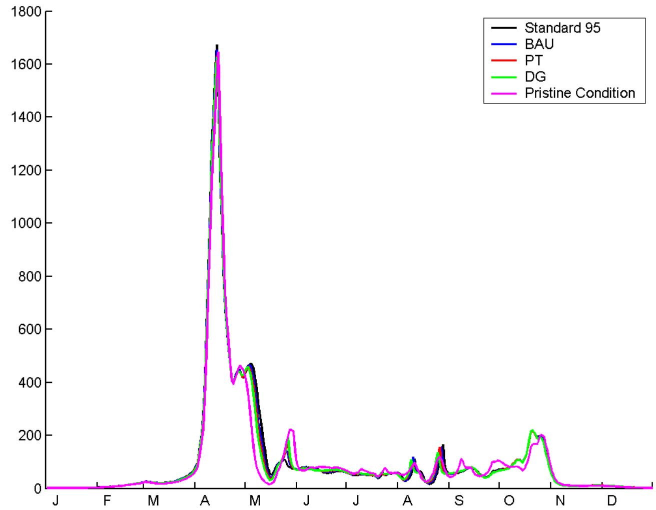


Figure 3.16: Modelled time-series of diatom concentration (mg C m^{-3}) related to input box 78 for different scenarios (BAU, PT, DG) compared to the standard run 1995 and the pristine conditions (Hofmann et al., 2005).

Table 3.8: Reduction in load, catchment area devoted to nutrient retention, costs of emission reduction at diffuse and point sources and wetland/dam construction in the Elbe river basin (Hofmann et al., 2005).

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 used for 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 (additional dams and wetlands)	0	203	0	915	0
Total costs	399	857	1,815	1,809	5,951

order to suggest a ranking of the considered possibilities. 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 enhanced integrity indicator values, thus reflecting lower risks of major ecosystem services. 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. This multiplicity of different costs and benefits associated with alternatives (e.g. monetary values, flows, qualitative assessments) calls for a multi-criteria decision-aid for assessing trade-offs. Under a MCA alternatives can be ranked based on performance scores (qualitative and quantitative effects of alternatives) and weights given to each effect (i.e. how important this aspect would be to society)[5]

In this study, the three scenarios are used as a basis for performing a MCA. In the BAU, no additional measures are taken, while the future evolves autonomously. The BAU scenario has been taken 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. 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 3.8:

- 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.

[5] A more detailed description of the method is given in the published version (Hofmann et al., 2005).

Table 3.9: Effects reduction measures in the Elbe basin as well as in the German Bight and used weights for the MCA (Hofmann et al., 2005).

								Weight level	
Policy options	C/B	Unit	BAU	MRHR	MR	SRHR	SR	1	2
Social								0.333	
Recreational amenity	B	Million Euro	5	10.5	10.5	15.3	15.3		0.111
Wetland amenity	B	Million Euro	0	39.8	0	128.8	0		0.222
Costal unemployment	C	%	7.4	6.9	6.9	7.2	7.2		0.333
Sectoral transition		-/+	++	+	-	0	–		0.333
Economic								0.333	
Cost (total)	C	Million Euro	399	857	1,815	1,809	5,951		0.600
Fish catch		-/+	++	+	+	0	0		0.175
Coastal tourism		-/++	0	+	+	++	++		0.225
Environmental								0.333	
Primary production	C	$mmol\,m^{-3}\,year^{-1}$	271	263	263	254	254		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

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 values of the variables considered for determining the effects of the considered measures to be included in the MCA are assessed in different ways: the total costs associated with measure implementation are calculated with the CENER model, coastal environmental effects in the coastal zone (integrity processes) are based on calculations with ERSEM, while for the social effects 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 derived from expert judgement.
table 3.8 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 Euros 252 million to Euros 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 3.9, while the single effects within a category are given different weights, as in column ‘weight level 2’).

Figure 3.17: Ranking of selected alternatives according to three evaluation criteria, economic, environmental and social, which represent also categories of effects (see table 3.9). The ranking reflects the condition of equal weight given to the three evaluation criteria. Under this condition, the strong reduction high-retention alternative (SRHR) is the highest ranked alternative. Social criteria contribute the most to the total score of SRHR, while economic criteria contribute the most to the total score of BAU (Hofmann et al., 2005).

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, which are shown in the last two columns of table 3.9. The information of table 3.8 is used as input for the MCA to rank the five policy alternatives, and the result is presented graphically in figure 3.17.

This figure can be used to examine the rankings of the alternatives according to each group of criteria separately. Here, it can be seen that the ranking of the alternatives for the social criteria corresponds to the overall ranking. Ranking for the economic impacts results in BAU being the best alternative, whereas ranking for the environmental impacts results in BAU as the worst alternative.

The outcome of the multi-criteria analysis shows that, be giving equal importance to economic social and ecological effects, 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 show a less advantage trade-off.

3.1.2 State/Impacts: ecological risk for international management

The paper presented in this section[6] introduces the concept of ecological risk as an alternative to safe minimum standards (SMS) and critical loads (CL) under uncertainty. In this paper *reduced* ecological risk is regarded as a non-monetary benefit of reduction scenarios. The socio-economic appraisal of different reduction scenarios takes ecological risk and reduction-measure implementation costs into account. The use of ecological risk instead of a damage function quantified in monetary terms, offers an alternative to cost-benefit analysis, as ecological risk reduction, for all the reasons exposed in sections 1.1.1 and 2.3 cannot be given a monetary value. Moreover, this study deals in more detail with scenarios as value-worlds, addressing possible different interpretation of the precautionary principle affecting policy choices. The spatial focus of this paper is set upon the Southern North Sea, and it applies to the issue of ecosystem regional and supra-regional changes for international decision-making in the context of common goals and large-scale environmental issues.

Given the extensive spatial scale of the North Sea region, a significant variation in ecological processes can be expected. This in turn makes the formulation of region-wide threshold values/safe minimum standards for selected parameters (indicators) a complex and less meaningful task. The objective of this study is to contribute to a more integrated analytical approach, which can encompass both small scale and larger scale functioning and possible cumulative interactions in the coastal zone. The key indicator in the proposed approach is ‘ecological risk’ (section 2.3). A deterioration in system integrity and therefore an increase in ecological risk results in a reduction in the ecosystem provision of services and goods available for human society. Given the manifold uncertainties affecting the definition of safe minimum standards (SMS) (i.e. definition of marginal impacts, regional versus global changes, acceptance and affordability of costs for implementing precautionary approaches, use versus abuse of ecological services and functions, Crowards, 1996) the question to be answered is: ‘what level of ecological risk/integrity do ‘we’ (society) want to achieve at what price?’(Barkmann and Marggraf, 2004, page70). This

[6]A slightly modified version of the study reported in this section is in press in Ecological Indicators (2006, doi:10.1016/j.ecolind.2006.09.002) as: C.Nunneri, W.Windhorst, R.K.Turner and H. Lenhart ‘Nutrient emission reduction scenarios in the North Sea: a cost-effectiveness and ecosystem integrity analysis’.

study provides an analysis of different eutrophication reduction scenarios in terms of abatement implementation costs and reductions in ecological risk. While it is not the role of science to make decisions, i.e. to select which alternative is most socially desirable, this paper does offer a basis for a decision support system which can better inform the choice making process. The uncertainties surrounding the understanding of the ecosystem functioning call for a precautionary approach, in which the current generation has the precautionary duty to develop a 'risk norm' to assist in the organisation of political, legal and economic regulatory institutions. In practice, the application of the precautionary principle is surrounded by ambiguity and subject to negotiations among parties representing different interests. If, in principle, the approach requires (1) a foreseeable and harmful damage impact, (2) a cause and effect knowledge function and (3) a policy response (Turner and Hartzell, 2004), in practice, one or more of the components are missing or are under specified. Given these caveats, the analysis presented in this paper offers scientific information suitable as a basis for stakeholder participation and negotiation in risk reduction dialogues. The proposed approach is based on the concept of ecosystem integrity (Barkmann and Windhorst, 2000; Barkmann, 2000). It offers an alternative approach to the one based on regional/local threshold-values of single parameters (e.g. as used within the OSPAR Comprehensive Procedure, (OSPAR, 2003a). The focus on the aggregated indicator 'ecological risk' allows the comparison of different regional ecosystems. The ecological risk notion, derived from the concept of ecosystem integrity, is portrayed in relative terms between two extremes. It is assumed that in pristine conditions (no human impacts) there is maximum provision of services and goods and in this sense zero ecological risk. A minimum provision state (maximum ecological risk) is then calibrated in terms of the conditions present in 1995 and with reference to the prevailing high eutrophication levels. Due to its definition, modelling and computation (changing, ceteris paribus, the nutrient loads to the North Sea), the 'ecological risk' indicator is restricted to eutrophication contexts and does not refer to some 'overall' pollution/resource degradation risk for the North Sea area.

Study area, catchments and coastal zones

The three river basins of the Rhine, Elbe and Humber are differentiated first by their relative sizes. The Elbe catchment is about 80% of the size of the Rhine's area, while the Humber is only 13% of the Rhine's area. The total area of the three catchments (35,7810 km^2) represents about 50% of the whole North Sea catchment. Population density, extent of urbanisation and land-use patterns also vary across the candidate catchments (see table 3.10). This range of socio-economic drivers and environmental pressures (drivers and pressures are defined in terms of the DPSIR scheme, EEA, 1999) creates conditions, which result in different fluxes and loads of nutrients and also means that abatement strategies will not necessarily be uniform.

The catchments, coastal zones and regional sea to open ocean form a continuum in biophysical and geochemical terms. While the riverine nutrient loadings have a direct influence on the marine coastal areas, these areas themselves are an integral part of the North Sea ecosystem which is affected by the prevailing open sea conditions extending out into the North Atlantic Ocean. Eutrophication in the North Sea Marine ecosystems is sensitive to a variety of external forcing. Physical controls, such as hydrodynamics including stratification, solar energy input or temperature are often dominant (Berlamont et al., 1996). Due to the general residual anti-clockwise

Table 3.10: Key-features of the three regional catchments (data by Behrendt, 2004 and Cave et al., 2003; published in Nunneri et al., in press).

Main characteristics of considered river catchments				
Feature	Unit	Elbe	Humber	Rhine
Basin area	km^2	148,27 0	24,240	185,300
Main crossed countries		Czech Republic, Germany	United Kingdom	Germany, Netherlands
Sub-basins	Number	185	6	423
Length of river	km	1090	690	1320
Mean discharge	m^3/s	708	250	2388
Total population	1000 inh.	24,611	13,668	57,256
Population density	inh/km^2	166	564	309
Urbanised area	% of catchment	5.9	12.3	7.9
Agricultural area	% of catchment	61.4	72.	8 51.8
Arable land	% of catchment	54.7	43.8	35.6
Pasture	% of catchment	6.8	29	16.2
Forest	% of catchment	30.5	13.7	37.2
Connections to WWTPs	% of households	71.4	79	92.4

circulation pattern (Lee (1980), see figure 2.4), the water masses entering the northern North Sea from the Atlantic turn eastward in the central North Sea and in general do not influence the continental coastal areas (Lenhart and Pohlmann, 1997). The southern North Sea is influenced by the inflow of the English Channel and the freshwater influx from the continental rivers. Along with the freshwater influx the river input causes excessive supply of nutrients(Brockmann et al., 1990) into the coastal regions. As a result of this continuous supply of nutrients the highest primary production occurs in the southern North Sea (Joint and Pomroy, 1993; Skogen and Moll, 2000).

Scenario definition

The precautionary principle (pp) essentially reduces to a requirement that no significant deterioration of the environment should occur unless the benefits to society associated with the deterioration heavily outweigh the costs of the deterioration. But under conditions of uncertainty and multiple stakeholders (often with conflicting interests) that typify catchment-coastal contexts, the precautionary (uncertainty/ risk reduction versus cost of implementation) trade-off is by and large a matter of judgement and negotiation. A number of positions can be determined depending on the prevailing worldview/interest, the degree of scientific uncertainty present (related among other factors to the type of polluting substances and loads involved) and the existing institutional arrangements (e.g. Royal Society, 1997; Renn, 1998; O'Riordan et al., 2001). We have identified three stylised future scenarios: a low reduction scenario (LOW-RED), a medium reduction scenario (MID-RED), and a high reduction scenario (HIGH-RED). Simplifying matters greatly, the first scenario typifies local industrial and developer stakeholder positions, which accord economic growth a priority status (weak sustainability). The second scenario is one in which environmentalist views gain ground in society and in the political agenda (strong sustainability). In this future polluters may over comply with clean up regulations and standards for a mix of reasons including the exploitation of 'win–win' resource efficiency measure, green marketing gains and the wider socio-political attitudinal changes. Finally, the third scenario is

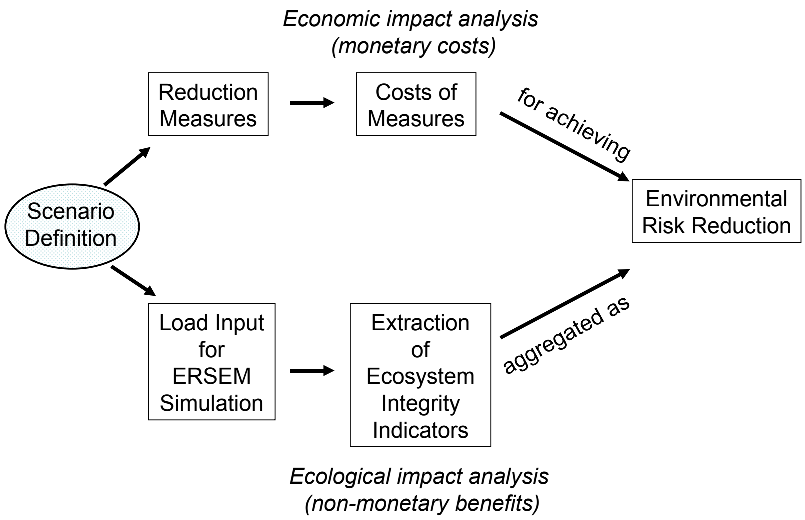


Figure 3.18: Methodological scheme used for the study reported in this section. starting from scenario assessment the two paths of analysis (economic and ecological) merge into the assessment of ecological risk reduction (benefits) and the costs necessary to achieve them (Nunneri et al., in press).

underpinned by long run environmentalist perspectives (very strong sustainability). Scenarios are the starting point for both the economic (upper part fig. 3.18) and the ecological impact analysis (lower part fig. 3.18) by providing:

- the framework for assessing feasible reduction measures for reducing nutrient emissions at the catchment level;
- reduction of nutrient loads to the coastal waters (in percent compared with 1995 loads) as input for the assessment of the ecological consequences of emission reductions upon the North Sea ecosystem through modelling with ERSEM.

We assume that different worldviews represented in the scenarios will result in different interpretation and application of the precautionary principle, given the prevailing environmental quality standards and targets policy -see figure 3.19. Monitoring would require complementary indicators of significant environmental changes. Figure 3.19 theoretically illustrates the case where regulators are coping with loadings comprising natural substances and in contexts in which, while scientific data is available, knowledge of the full implications of ecosystem change is incomplete. In such circumstances the HIGH-RED position would involve the re-establishment of pristine environmental conditions, typical of pre-industrial times (strict precautionary principle [spp]). The MID-RED position would be a reduction in loading to a level determined by the best available abatement technology, even where such technologies had only been demonstrated in restrictive conditions (technology forcing), e.g. laboratories. The weak sustainability position

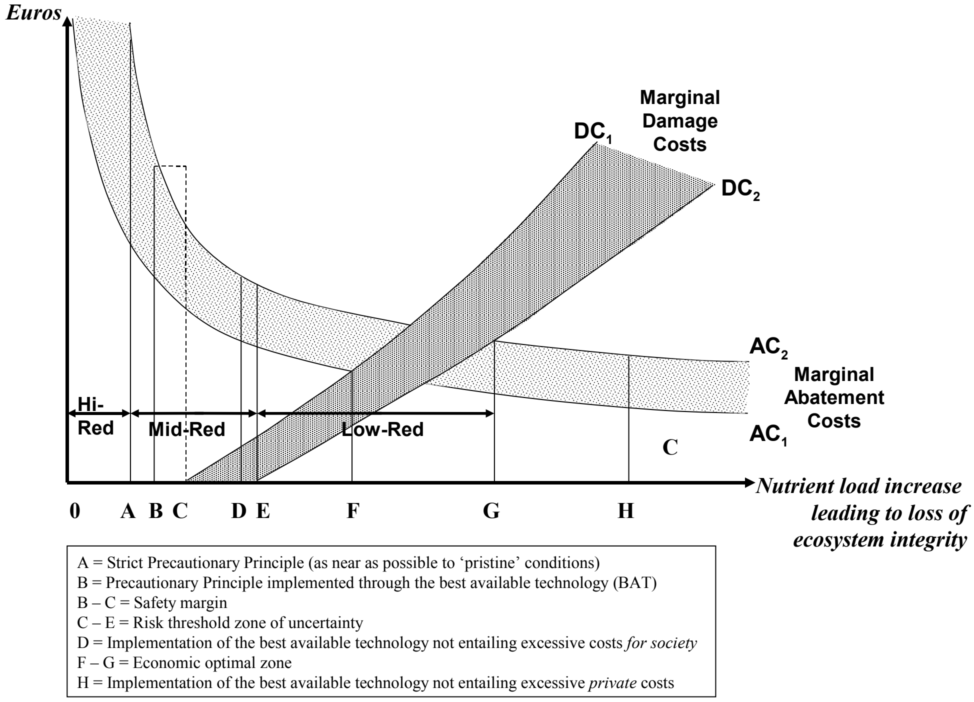


Figure 3.19: Theoretical representation of different management positions based on economic considerations and different interpretations of the precautionary principle (assuming that all cost can be expressed in monetary terms). Marginal abatement costs (ranging between AC1 and AC2) and marginal damage costs to the environment (ranging between DC1 and DC2) are shown: for further explanation about different precautionary approaches (and not) see text (Nunneri et al., in press).

typically argued for by industrial stakeholders would be represented by the zone between the two BATNEEC (Best Available Technology Not Entailing Excessive Cost) positions and encompassing economic efficiency optima. The 'excessive costs' could be quantified in terms of private financial costs alone (position H in figure 3.19); or in full social (economic) cost terms including all known externalities (position D in figure 3.19). In our approach, we use the 'ecological risk' indicator instead of a damage cost function quantified in monetary terms. We then compare the different eutrophication 'ecological risk' positions with their consequent cost implications for society, in terms of the costs of alternative nutrient abatement measures necessary to achieve a given nutrient loading and risk situation.

Nutrient fluxes reduction under each scenario

The defined scenarios enter the ecosystem model in form of changes of river nutrient loads (lower path in fig. 3.18). They are treated as differences between a standard run, which simulates the year 1995 as realistically as possible, compared to the simulations applying the reduction scenarios for the rivers, but leaving the rest of the forcing the same as in the standard run. Finally, the results of the ERSEM model runs are condensed into ecosystem integrity indicators. These indicators allow an overview of the vital changes related to the scenarios, based on a reduced number of parameters which nevertheless represent the complexity of the ecosystem. The ERSEM incorporation of different pathways for nutrient cycling through the trophic net, which act on different time scales, indicate the overall effects on the ecosystem related to nutrient reduction.

The indicator values calculated by ERSEM react in different ways and to a different extent to reduced nutrient loads from the rivers. A linear nutrient reduction may result in non-linear effects, thus highlighting the need to analyse the overall functioning of the ecosystem (ecosystem integrity and ecological risk). The integrity indicators (see table 2.1 on page 16) are normalised following the procedure explained in section 2.3. Scenario formulations imply a given nutrient reduction target relative to 1995 baseline conditions and a package of enabling measures and financial cost burden (the time span of the scenarios is 25 years into the future, starting from 2001). For the ERSEM model, LOW-RED implies a 20% reduction, MID-RED a 40% reduction, and HIGH-RED a 60% reduction, compared to the year 1995. The nutrient input reductions to the North Sea are modelled in 10% reduction steps, starting with a 20% reduction with respect to the year 1995 and covering all steps between 1995 and pristine values, which are assumed to be 10% of the 1995 loads (Behrendt, personal communication, 2004).

Economic impact assessment: reduction measures and costs

Once the scenarios have been formulated, the economic analysis (upper path in figure 3.18) appraises the range of feasible abatement options, as well as the monetary costs of measure implementation. The distribution of the gains and losses across multiple stakeholders and other social issues (including ethical concerns about intrinsic value in nature and social equity) are not dealt with in this study (see Turner et al., 2003b). We assume here that different abatement measures are acceptable under each scenario (different societal willingness to pay in order to reduce risks). For each scenario and catchment a specific set of measures for nutrient emission reduction was considered as reported in table 3.11.

Those measures have been assessed according to scenario priorities (economic growth, strong sustainability and very strong sustainability) and taking into account regional specificities and needs. It is assumed that the acceptance of ecological risk decreases from LOW-RED to HIGH-RED, while the willingness to pay for environmental conservation increases (see figure 3.18). The selected measures used for this study, as well as their implementation costs are assessed in detail by Cave et al. 2003; Coombes et al. 2004; Lise 2003. Our economic analysis includes the description of different scenario measures in terms of (1) their effects upon nutrient fluxes to the North Sea (inputs for the ERSEM model) and (2) their implementation costs. In our analysis, there is only an approximate correspondence between nutrient abatement brought about by measures in the catchment (see table 3.11) and nutrient fluxes used as inputs for the ERSEM model. In reality, some of the modelled reductions in nutrient loading have already been achieved. Our purpose here is to demonstrate the utility of an integrated economic cost and risk assessment method, rather than to perform an actual and comprehensive policy analysis.

Table 3.11: Overview of the main reduction measures for the three river catchments and the resulting emission reductions (Nunneri et al., in press).

Scenario	Catchment	Achievement of measures	Measures	Reduction(%) [a]	
				N	P
LOW-RED	Humber	Current emission reduction trends/levels are maintained	300 ha add. wetlands due to management realignment	11	31
	Rhine		No additional measures	20	32
	Elbe		No additional measures	-2	25
MID-RED	Humber	Reduction of inputs from the catchment(point and diffuse sources), implementation of the nitrate directive (good agricultural practice) and of the urban waste water directive	20% reduction of riverine loads (point sources + nitrate directive implementation), 1321 ha additional wetland due to management realignment	19	41
	Rhine		Farm measures, WWTP update, tile drainage reduction up to 10% of arable land	50	65
	Elbe		Farm measures, WWTP update, tile drainage reduction up to 10% of arable land	50	65
HIGH-RED	Humber	Over-compliance with environmental directives and standards	50% red in point sources, Nitrate Directive implementation, 7400 ha of add. wetlands	27	42
	Rhine		Farm measures, WWTP update, tile drainage reduction up to 20% of arable land	70	75
	Elbe		Farm measures, WWTP update, tile drainage reduction up to 20% of arable land	70	75

[a] Please note that reduction percentages of N and P refer to 1995 (ERSEM basis year) for the Humber and to 1985 (basis year for OSPAR targets) for the Rhine and Elbe catchments.

Implementation costs of measures The nutrient abatement direct costs for the Humber, Rhine and Elbe catchments are adapted from Cave et al. (2003); Coombes et al. (2004); Lise et al. (2004). The abatement measures vary across the catchments ranging from long-term capital-intensive schemes such as sea walls to changing land use/farm regime practices. For the Humber the scheme costs are standardised to the financial year 2001–2002 using GDP-deflators as recommended by HM Treasury. Capital costs are annualised over a 25 year period at an annuity rate of 3.5% and costs are discounted at the same rate over a 25 year period (Cave et al., 2003). For the Elbe and Rhine catchment, the capital costs are computed in 2000-prices under the assumption that they occur during the first year (no discounting is applied) (Lise, 2003). For this study they are standardised to the year 2001 by using the GDP deflator for the Euro Area as reported by the International Monetary Fund (2005, page 213). The resulting costs for each scenario (expressed in million of Euros) are reported in table 3.12.

Table 3.12: Costs (million of Euros in 2001 prices) for implementing nutrient reduction measures under different scenarios (data from Cave et al., 2003, Lise, 2003, Lise et al., 2004; published in Nunneri et al., in press).

Catchment	Measure costs under the three reduction scenarios								
	Defences/wetland creation			Reduction from diffuse sources			Reduction from point sources		
	LOW RED	MID RED	HIGH RED	LOW RED	MID RED	HIGH RED	LOW RED	MID RED	HIGH RED
Humber	144	153	251	0	120	120	0	166	415
Rhine	0	207	937	492	492	545	344	348	456
Elbe	0	208	937	151	222	468	258	447	447

Ecological benefits: ecological risk reduction

Ecological risk represents the end-point of the integrated analysis, by merging the economic cost and the ecological impact analysis (see figure 3.18. The decrease in ecological risk (environmental benefit) is the result of the nutrient reduction brought about by the implementation of measures (with given economic costs) in the catchment area. While the single selected indicators give an insight into the specific processes at the core of ecosystem integrity, and therefore are, even in the more handy concept of integrity ('amoeba visualisation') not of immediate use to managers and decision-makers, the aggregated indication of 'ecological risk' allows a synthesis of all indicators that may be useful for comprehensive decision-making purposes. The ecological risk notion used in this study is derived from the concept of ecosystem integrity (see section 2.3) ecological risk gives a relative measure of the ecosystem failure in providing the level of natural resources supporting human societies. Although we do not make use of typical risk analysis instruments such as probability theory, exposure, hazard, etc. (see Hill et al., 2000), the US US-EPA (1998), definition of ecological risk: 'the probability that adverse ecological effects occur as a result of exposure to one or more stressors' still fits the way the term is used here, if we allow for arbitrarily setting the 0 and 100 probability levels. The aggregated indicator 'ecological risk' offers an ecological perspective for dealing with safe-minimum-standards threshold values, by assuming, as a first approximation, that the level of risk in 1995 is maximum and in pristine conditions minimum. This offers a baseline against which other nutrient reduction strategies can be assessed (Crowards, 1996; Barkmann and Marggraf, 2004). For each scenario the corresponding level of ecological risk is computed in a relative way: i.e. assuming a maximum capacity of providing ecosystem services under pristine conditions (no human induced stressors impacting the ecosystem), which corresponds to an ecological risk value of zero, and a maximum ecological risk value of 100 for the situation in the year 1995 (high eutrophication levels due to maximum levels of stressors, i.e. anthropogenic nutrient emissions).

Appraisal of nutrient reduction scenarios

Three target situations for dealing with eutrophication in the selected North Sea coastal areas will be described in terms of resulting different degrees of ecological risk for the coastal waters and the costs of achieving them.

In order to indicate the influence of the selected indicators on the self-organising capacity and the relative impact caused by the different reduction scenarios it is necessary to transform the values of the indicators modelled by ERSEM into relative numbers using the formula (2.2). The two extremes have been chosen for each indicator as the absolute values for the year 1995 (assumed to correspond to a 100% nutrient emission level), the scenarios, and the 10% emission level assumed as conditions in the absence of human impact – the so-called 'pristine conditions' – (Behrendt, personal communication, 2004). The scenario reductions imply the reduction of both N and P (in equal degree), although the integrity and ecological risk analysis has been carried out separately for N and P. As Nitrogen inputs are considered of major importance (and since phosphorous inputs have been considerably reduced during the last decades), we focus our attention upon nitrogen reduction. In figure 3.20, the ecosystem integrity is indicated through its main five processes (via selected indicators) for the three coastal zones. The eutrophicated state is indicated by highest primary production and extremely positive N sediment budget (N retained in the sediment) for all the three case studies. The tendency to reduce primary production as a consequence of reduced N inputs into the system is clear throughout the three different regional ecosystems. In the same way, the tendency to increase the diatom/ non-diatom ratio (i.e. increase the trends towards a low nutrients input situation more similar to historical long term records), to increase the cycling of winter nutrients (the system needs to re-use available nutrients, if the inputs of new nutrients is reduced) and to minimise matter losses (i.e. to increase the quantity of matter retained by the system) and nutrient retained in the sediment, are also observable in the case studies. The sediments represent a buffer for the system, being a sink for overabundant nutrients and matter and a source in scarcity times. In all case studies the sediment budget (N in the sediment) tends to decrease in pristine conditions.
In figure 3.21 the relative change in the five processes triggered by changing the inputs of nitrogen into the coastal ecosystem is presented for the Rhine coastal zone. The spiral pattern of the processes shows that the system can cope with different boundary conditions (N availability) by changing the relative role of different processes. In particular two counteracting trends are to be seen for different processes: while the primary production and the amount of N stored in sediments decrease towards their minimum pristine value, the turnover of winter nutrients, the matter balance and the diatom/non-diatom ratio increase from their minimum level in 1995 towards a maximum level in pristine conditions The ecological risk related to the coastal waters influenced by the three water systems is shown in figure 3.22 for different reductions in nutrient availability. In general a reduction of N inputs by 40% with respect to 1995 levels would considerably reduce the ecological risk across all three coastal areas. In particular, the risk reduction for the Humber presents the highest risk reduction for the LOW-RED with respect to the 1995 situation (about 26% risk reduction). The Rhine and Elbe coastal zones get lower risk reductions for the LOW-RED scenario, while the reduction in risk is very similar throughout the catchments for the MID-RED scenario. The Humber presents the highest, and the Rhine the lowest risk reduction under the HIGH-RED scenario.

Concluding remarks

The OSPAR targets for nutrient reduction have been achieved for P but not N: the reduction of inputs in the North Sea can be assessed at about 20% reduction, compared to the 1985 levels (OSPAR, 2003b). We have shown the costs of achieving different nutrient reduction levels and

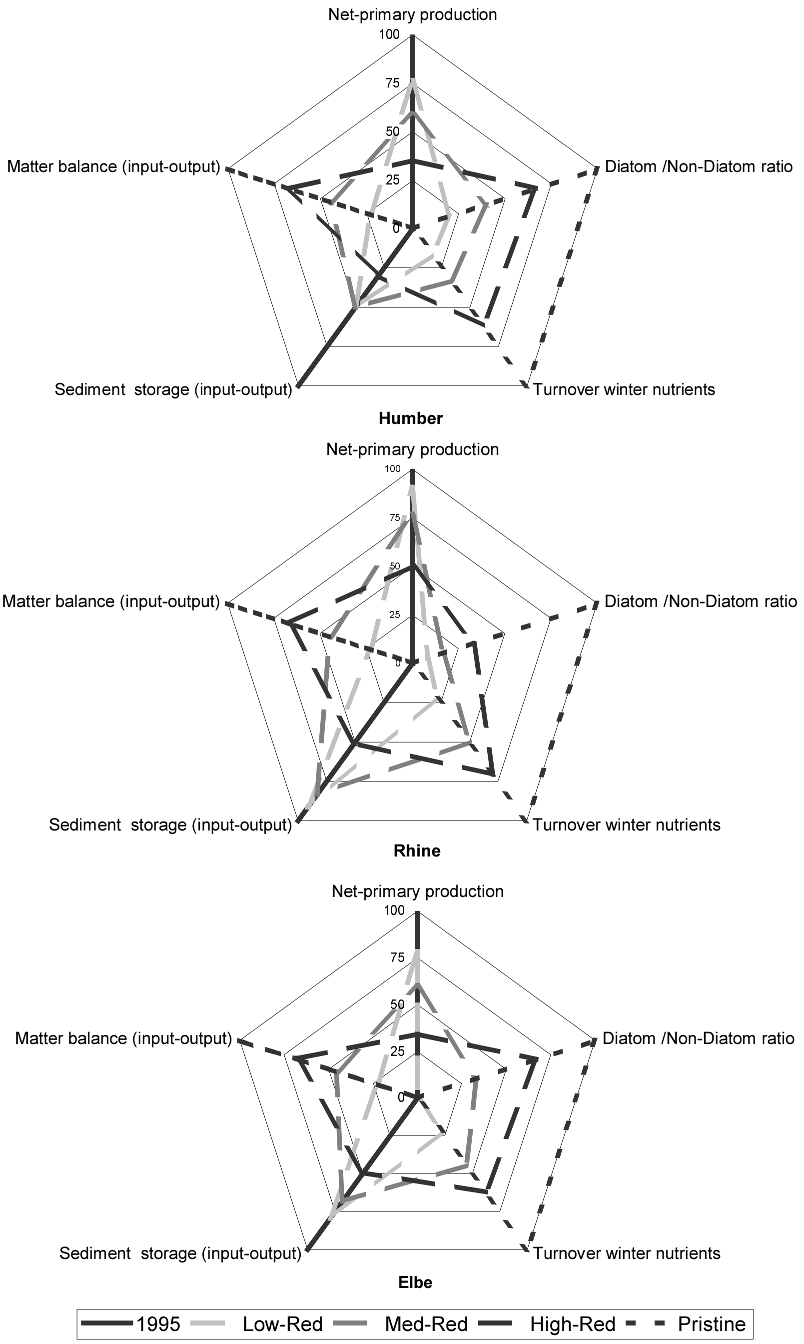


Figure 3.20: Ecosystem integrity indicators (normalised values, for nitrogen only) for three coastal ecosystems (Humber, Rhine and Elbe coastal zones) under the three considered scenarios. Indicator values are reported in relation to the reference conditions in 1995 and the assumed pristine conditions (10% emission compared to 1995 levels). The normalised values range between 0, centre of the graph, and 100, outer line (Nunneri et al., in press).

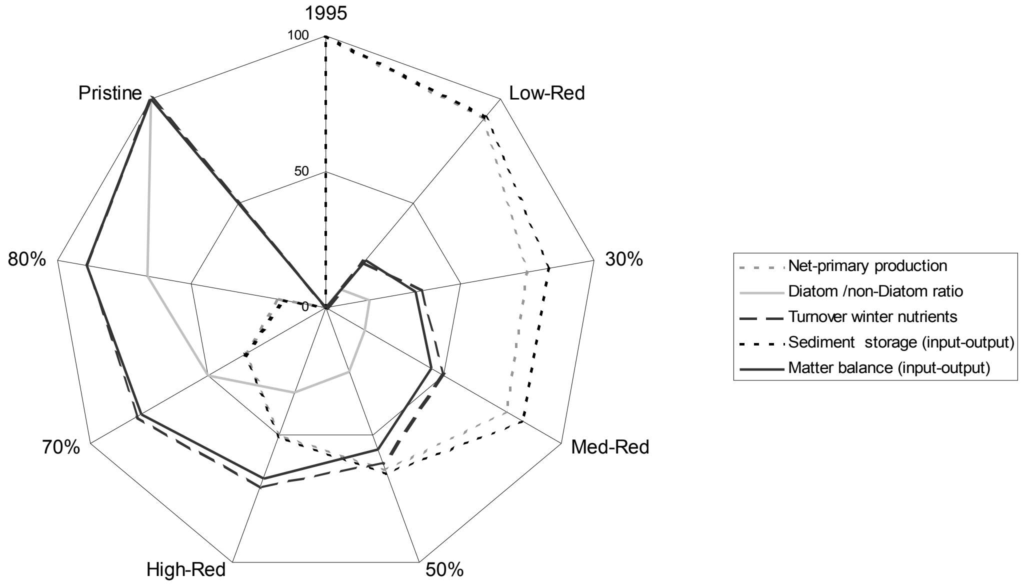


Figure 3.21: Rhine coastal area: Integrity processes (represented by normalised indicator values) and their relative changes related to Nitrogen input reductions. N reductions are in 10% steps, starting from a 20% reduction compared to 1995 levels (Nunneri et al., in press).

their ecological potentials for three catchments and their coastal areas in the North Sea. Through the concept of ecological integrity and ecological risk it has been shown that regional coastal areas can react in different ways to nutrient reduction. In particular, there are regions in which a 'maximum' intervention (a reduction up to 60%) is justified by a considerable further decrease in ecological risk. This applies in particular to the Elbe coastal zone. On the other hand, the costs of abatement measures do not increase linearly with risk reduction (see table 3.13).

The relatively low-cost measures implemented under the LOW-RED scenario bring about the highest reduction for the Humber coastal zone, while lower reduction in risk occurs in the Elbe and Rhine coastal areas. In the Humber, the abatement costs increase threefold under the MID-RED scenario, and by about fivefold for achieving the HIGH-RED reduction targets. In the Elbe, the costs double for implementing the MID-RED scenario measures and increase about five times for achieving the HIGH-RED reduction targets. The costs for the Rhine catchment are in absolute terms the highest (due also to catchment extension), but show a less steep increase throughout the scenarios (the costs under the HIGH-RED scenario being about the double of the costs under the LOW-RED scenario). In practical terms, the costs per capita (given the

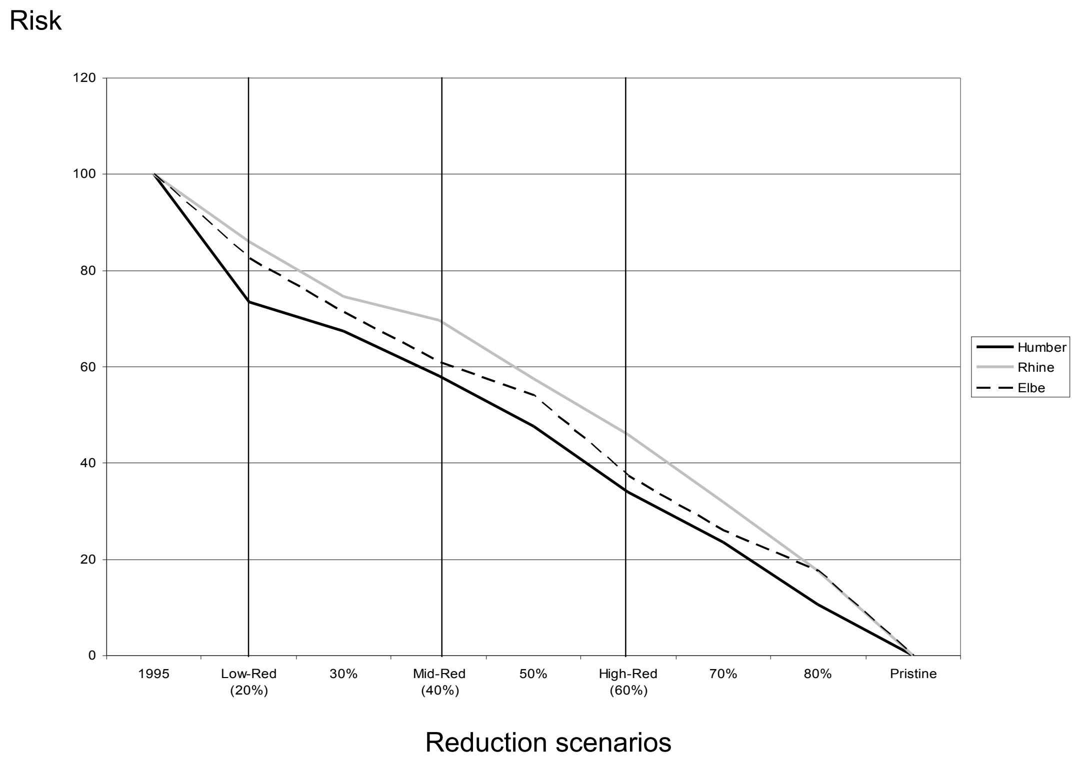


Figure 3.22: Ecological risk values for the three coastal areas are shown for 10% reduction steps (starting by a 20% reduction and including the scenario reductions). At the extreme left the 100% ecological risk in the reference conditions (1995) and at the extreme right the 0% risk in pristine conditions (Nunneri et al., in press).

actual population, see table 3.10) would be similar for all three catchments for the LOW-RED scenario, but highest for the Elbe catchment and lowest for the Rhine under the HIGH-RED scenario (due to different population densities). The trade-offs between further risk reduction and increasing costs of measure implementation should be carefully assessed: in the light of a joint reduction of ecological risk in the North Sea, a burden-sharing policy for reduction measures looks a promising way forward.

Table 3.13: Overview of abatement costs and ecological risk reduction for each scenario and catchment as well as for the three catchments altogether –costs are presented as both average costs and total costs– (Nunneri et al., in press).

Catchment	Indicator	Scenarios		
		LOW-RED	MID-RED	HIGH-RED
Humber	Economic costs (million €)	144	439	786
	Economic costs/inh. (€/cap)	11	32	58
	Ecological risk (0;100)	73	58	34
Rhine	Economic costs (million €)	836	1047	1937
	Economic costs/inh. (€/cap)	15	18	34
	Ecological risk (0;100)	86	70	46
Elbe	Economic costs (million €)	409	878	1852
	Economic costs/inh. (€/cap)	17	36	75
	Ecological risk (0;100)	83	61	37
All three catchments	Total economic costs/inh. (€/cap)	42	86	167
	Average economic costs/inh. (€/cap)	14	29	56
	Average ecological risk (0;100)	81	63	39

3.1.3 Response: distribution of gains and losses

The paper reported in this section[7] shows how participation of stakeholders and experts can enrich integrated analysis by addressing governance issues related to nutrient reduction strategies. The spatial focus is set upon the Elbe river catchment. The river transboundary nature (two thirds of the catchment belong to Germany and one third to the Czech Republic), allows also to address international issues such as 'mutual help' and compensation for implementing possible reduction strategies.

The overall goal underlying this study is the coupling of Integrated River Basin Management (IRBM) and Integrated Coastal Zone Management (ICZM), as targeted in the Water Framework Directive (WFD). Thus the first step towards is the identification of inland anthropogenic activities (e.g. land use and anthropogenically induced matter fluxes) influencing (through river systems) the ecological quality and the socio-economic service functions of adjacent coastal zones. The Advisory Board (AB) set up for the Elbe case study involved governmental and non-governmental institutions having some interest in the issue of river basin management, either because of their activities in the study area or because of their interest or involvement in the issue of nutrient reduction, although not directly related with the area. The participatory approach ongoing for the Elbe catchment represents a first attempt to determine the present perceptions of selected groups of interest, especially regarding concern about environmental issues, (use-)conflicts in the catchment area, and the measures considered feasible for nutrient emission reduction. Particularly, the early scoping of stakes and conflicting interests regarding the issue of nutrient emission reduction can inform future decision-making, thus facilitating the way towards win-win solutions and integrated management. It is worth noticing, however, that the current perceptions as reported in this paper might be different in the future, thus emphasising

[7]The study reported in this section has been published in slightly modified form in Estuarine, Coastal and Shelf Science (2005, vol.62: 521–537) as: C. Nunneri and J. Hofmann 'A participatory approach for Integrated River Basin Management in the Elbe catchment'.

the importance of incorporating socio-economic scenarios in integrated projects.

Participatory approaches in environmental issues

A review of participatory processes is given by van Asselt and Rijkens-Klomp 2002, page 168. The authors ‘propose to define participatory methods as methods to structure group processes in which non-experts play an active role in order to articulate their knowledge’.
They describe different forms of participation according to the goal of application. In this context the procedure used for this study can be seen as a part of a ‘decision-support process’, i.e. a way to enrich assessment. van den Hove 2000, on page 458 gives ‘many justifications for such calls for participatory approaches to environmental problems’, which are based on the peculiar characteristics of environmental issues: ‘complexity, uncertainty, large temporal and spatial scales, and irreversibility’. The author emphasises above all the relation of the physical characteristics of environmental issues with the social characteristics, such as transversality and cross-sectoral involvement of lifestyle and societal production patterns. Such characteristics are the basis for participatory processes aiming to tackle the problem and proposing possible solutions taking into account possible impacts, the spatial distribution of costs and benefits, the possibly resulting conflicts. In this case, the participatory process does not address involved people directly, but the collected information originates from respondents representing the interests of the organisations they belong to.

The interviews

As a tool for obtaining an insight about the AB members’ perceptions and suggestions a series of in-depth, semi-structured interviews were used, recorded on a tape and finally transcribed. The typewritten text of the answers was sent to the respondents, in order to have possible misunderstandings corrected. Anonymity of the interviewees was assured: in the following the respondents, identified with the institution they belong to, will be referred to as Ex (E1, E2, . . . , E9, where ‘E’ stands for ‘Elbe’). The choice of the interview protocol and interview questions depended on the one hand, on the dimension of the catchment, on the other hand, on the characteristics of many institutions represented in the AB (that focus on various issues and comprise different fields, rather than specifically working on the Elbe catchment). In this context face-to-face interviews were chosen as a suitable and still not costly means. The open-end questions presented to the interviewees addressed the following main points:

1. General views and perception of the environmental problems in the study area (is nutrient emission perceived as a main issue?);
2. Evaluation of the present institutional framework;
3. Proposals for developing management measures for reducing nutrient emissions;
4. Criteria for choosing among measures (how to evaluate measures?).

Respondents

An initial selection of the institutions to be included in the AB was carried out by the scientists involved in the project, based on previous knowledge of the scientists and on declared (and expected) field of interest of institutions. Since there was no pre-existing group from which to draw a sub-group, previous experience and cooperation of the scientists with institutions played a major role (and represent a major bias) in the choice. The pre-selected institutions were then invited by a formal letter to join the AB. The aim was to have both representatives of the coast and of the catchment, as the location was expected to influence the institutions' interests. Although the intent was that of balancing governmental, private institutions and NGOs, the initial bias towards governmental institutions was even increased during the process of building the AB through formal invitation. In fact governmental institutions were much more willing to cooperate than other kind of institutions. In the end most of the respondents (seven out of nine) spoke on behalf of the government or governmental operating agencies, such as ministries, thus representing those having the power to inform, shape and implement the decision-making process at the national and the catchment level. Only a minority of the interviewees revealed themselves to be key-stakeholders, i.e. directly affected by the decision-making process. Not all the contacted institutions reacted to the invitation letter, in some cases personal contacts revealed themselves to be very useful, especially for cooperation with Czech institutions. Furthermore, a major issue of bias in the interview results is the over-representation of German members of the AB, due to scarce participation of the Czech institutions. Two sub-groups of respondents were identified: the first comprising institutions expected to focus their interests and activities on the catchment, the second one on the coastal zone. This paper deals with the nine institutions belonging to the former group. The presented results, far from embodying the 'general public perceptions', represent the points of view of (mainly political) institutions and organisations operating at an international or national scale in the Elbe catchment and provide different perspectives than those of the scientists involved in the project. This paper represents the first stage of an interdisciplinary approach, aiming at a comprehensive qualitative issue analysis, which is at the basis of integrated management. More interviews are planned with institutions operating locally in the coastal zone and having to deal with the consequences of eutrophication

Results

The contribution of the respondents has revealed itself much more as a set of guidelines for integrated management, rather than a concrete guide for applying reduction measures. Nevertheless, the AB members' judgement about feasibility, political relevance and criteria for evaluating a number of measures can be used as a basis for a future Multi Criteria Analysis (MCA), with the aim of determining a ranking of management options, following the suggested criteria. Although it was expected that the location of the organisation would also shape its interests, the first result of the interviews (Table 3.14) was that the majority of the institutions located in the hinterland declared to be interested in the coastal zone as well as in the catchment itself. Table 3.14 gives an overview of the fields of interests of the respondent institutions.

The field of interest of E1 is changing from a past catchment-oriented activity, leaving completely out of its range of action the coastal zone (downstream of the weir Geesthacht) as well

Table 3.14: Schematic representation of the interviewed institutions (Nunneri and Hofmann, 2005).

Kind of institution	Area of interest		
	Only catchment (hinterland)	Catchment and coast	Others
International / intergovernmental	E1 (past)	E1 (future)	
National governmental	E6 E7	E2 E3 E4 E5	E3
National private (associations)		E8	E8
National NGOs		E9	

as the river Elbe tributaries, to a future catchment-coast continuum area of interest. Institution El is planning to reshape its scope for including the coast in its policy planning activities in the future in order to realise the WFD target of coordinating ICZM and IRBM. Institution E1 is the only real 'region-based' respondent. In the case of interviews E3 and E8, the area of interest includes coast and catchment because activities are focused on particular issues (the 'other interests'), independently of the location. Those institutions aim to protect some particular right or a particular sector related to particular groups of citizens and can be seen as 'issue-based' respondents. The institution members of the AB are classified according to their area of interest, whether catchment, coast, both catchment and coast or other interests.

Some introductory questions were aimed at obtaining an overview of the perceived pollution problems in the North Sea as well as in the catchment and a first understanding of the possible related conflicts, then reduction measure possibilities were addressed in more detail.

The issue of highest concern Every interviewee, due to the activity of the organisation they represented, perceives problems with a different priority scale. Nutrient pollution does represent an issue and was voluntarily mentioned by all the respondents, although not all showed the same concern about it. On the one hand, one respondent noted 'The problem of major concern is the emission and transport of nutrients by the Elbe River. This problem has led to construction of Waste Water Treatment Plants (WWTPs) in the new Bundesländer and, with some delay in time, in the Czech Republic. Also the emissions from chemical industry diminished during the last years. This was not much due to measures, but mainly to ceasing or decreasing production. In general there has been a shift of the problem of emissions from point to diffuse sources. The agricultural sector is the target for future emission reduction measures' (E1). On the other hand, another respondent did not even mention the nutrient problem: 'The expansion and density of inhabited areas near the water bodies and the resulting pollution of anthropogenically produced substances (softeners, medicaments, cosmetics), as well as the alteration of the natural structures of the water bodies (morphodynamics) are the issues of major concern' (E4). Another response was that: 'The pollution deriving from the catchment is for sure the largest, long-term problem' in the coastal waters according to E9, although it is not clear, whether the term pollution here is used as a general term also including nutrient enrichment. Some of the remaining respondents, who all mentioned the nutrient issue, focused their highest concern on: 'The emission of dangerous substances as defined in the list of the WFD' (E2). 'Anoxia in the tidal Elbe, nutrients and pollutants such as mercury, hexaclorbenzene and organic stannous compounds in the catchment'

(E5). ‘Conflicts in land-use, such as losses of agricultural areas by building and pavement activities or compensation measures for nature protection’ (E8). Although no pattern is to be found in those answers depending on the kind of institution, the diversity of answers clearly reflects different expertise and interests, thus allowing for a first insight into the possibility of evolving conflicts. The two respondents who did not explicitly mention the nutrient issue focused much more on anthropogenic influx on the inland and coastal waters (emission of chemical compounds with potential negative effects on aquatic life) and on the problem of infrastructure and morphodynamics, also in relation with ecological issues (e.g. salmon fishes mobility). In this context it is worth to mention that anoxia in the tide Elbe is an issue related to algal blooms and consequent oxygen depletion and, therefore, also strongly related to nutrient emissions in the catchment. Among the other respondents, concern about the nutrient issue does not always occupy the first position. On the one hand, this might be a result of the already improved environmental situation (decreased nutrient emission) if compared to the 1980s (E2) and, on the other hand, it can be due to the implicit difficulties in addressing diffuse sources for further reduction of nutrient flow to the coastal zone. The long residence time of N in groundwater (15-30 years) has been suggested as the reason why, even with effective measures, improved environmental conditions will be visible in no earlier than 15 years (E2, E5, E6). The actual concentrations are still consequences of the past load. The reasons mentioned might support the opinion that the situation will improve ‘on its own’ (due to the management measures already applied in the catchment), even though with a certain time lag, thus not requiring pro-active action. The nutrient issue seems locally to be less important than other issues, such as anthropogenic or dangerous substances. This can be strongly linked to the high uncertainty related to the potential effects of chemical substances of recent synthesis if compared to the relatively ‘well known’ effects of nutrient enrichment.

Identifying present and future conflicts A variety of conflicts in the Elbe catchment have been highlighted. They can be mainly related to the problem of nutrient emissions, but are also of general nature. With regard to nutrient emissions, the first mentioned conflict is that between environmental protection and agriculture (E1, E2, E3, E6, E7),‘which accounts for the main source of diffuse nutrient pollution’ (E2). However interview E8 highlighted that the (relative) greater role played by agriculture during the last years is a consequence of very high emission reductions in other sectors: the nutrient emission from agriculture has increased-in percent-because emissions from agriculture has been reduced to a minor extent if compared with point source emissions (such as WWTPs). ‘The success of reduction measures is not only caused by improved waste water treatment technology, but also by optimised fertiliser application on arable land. In the period between 1985-1995 there was a 35% reduction of nutrient surplus on arable land’ (E8). Fishery in the catchment as well as in the coastal area is related to water quality. For the Elbe river ‘fishery does not play an important role upstream of Geesthacht’ although ‘It is a target to reestablish fishery as an economic component of medium enterprises (94 fish species are present in the Elbe river)’(E5). Although this statement does not much relate to eutrophication, but rather to chemical pollution, it connects pollution in general and environmental quality, embodied by environmental quality supporting ‘eatable’ fish fauna in the river waters. This topic introduces the relation between environmental standards and the economy. In general, the regulation or mitigation of the activities having pollution as an externality has ‘negative’ effects on economic growth (E1, E5), i.e. the costs for reaching environmental targets (E6, E7). The costs for high environmental protection will be reflected in higher prices (or taxes) and in the end will be borne by citizens and consumers of agricultural products, such that a following conflict, which no interviewee expressed explicitly, would be that of public acceptance and public goals, desires and perceptions, possibly in disagreement with practice and policies. Regarding this point, particularly 'Within the Czech Republic there are conflicting interests: on the one hand, there is a wide interest in improvement and, on the other hand, the further achievements and their costs are not recognised by the people' (E6). The different economic situation in the Czech Republic and Germany, as highlighted by E6, will shape acceptance of (stringent) environmental targets and willingness to pay for improved environmental quality, which are both likely to be higher for the German than for the Czech citizens. An economic issue related to 'nutrient pollution' is the guarantee of further existence of agriculture as an economic sector. The goals for environmental protection and the related costs should be shared in such a way that in the process farmers can still make some gains from their activity, and do not bear alone the complete costs for nutrient reduction (E3, E8). 'The main target is the reduction of diffuse pollution, but in the same time agriculture should subsist as an economic branch' (E3). Other conflicts involve the use of nature and environmental protection. Some examples could be water use conflicting with water protection and recreation conflicting with protected areas (E1, E8). Agriculture is also affected by water pollution (e.g. refuse and wastes): the agricultural areas will not be usable in the aftermath of river flooding (e.g. recent Elbe flooding in August 2002) (E8). All the presented conflicting interests co-exist in the area of the Elbe catchment. An integrated management of the catchment is based on procedures for finding compromises among the different interests and harmonise them (E3), such integrated processes are based on communication among the different groups of interests, requiring cooperation of the existing institutions, NGOs, groups of interests and the general public. Unfortunately this is a further field of contention. The involvement of the general public is often seen as problematic and represented as strongly influenced by NGOs' activities, especially press releases and information campaigns. Still there is no determination in involving the general public in a standard (and voluntary) procedure for decision-making: 'In the future the WFD, Art. 14, calls for informing the public, although no direct participation is explicitly mentioned' (E5). In some international institutions NGOs can directly exert political pressure, among which 'first of all the NGOs (e.g. WWF) that work in OSPAR and HELCOM commissions have to be mentioned'(E2). Other NGOs use the press for exerting indirect pressure through the general public. In a few summarising words the present decision-making process seems to be 'pressure and reaction based', e.g. strongly directed by the activity of lobbies (E2), rather than a 'compromise process' (E3) based on dialogue and communication among the different groups of interests and the general public.

Legislative sectoral frameworks and role of institutions The WFD opens a new dimension for European Water Policy. It sets new horizons for international cooperation, presenting new challenges for the International River Basin Commissions that will have to coordinate international management plans. In the case of the Elbe River, the coordination plan also involves the Czech Republic, thus also acting as preparatory for the EU membership. The Czech Republic has already transposed the essential objectives of the WFD into national legislation: in 2001 the Parliament (E7) approved the new Water Act. In Germany the WFD has been transposed into national law, with the 'Novelle des Wasserschutzgesetz', which came into force on the 25^{th} June 2002. The interviews carried out for the Elbe catchment attributed the present legislative framework, and especially the WFD, a good potential. The holistic approach to water management links resources to uses, terrestrial environment to aquatic environment and finally water bodies to development and human health. According to the Water Framework Directive, Member States have not only to prevent the deterioration of the status of all bodies of surface water, but also to protect, enhance and restore all bodies of surface water, except for artificial and heavily modified water bodies. The WFD defines five classes of ecological status: 'bad', 'insufficient', 'moderate', 'good' and 'high', the goal is to reach good surface water status by the end of 2015. It has however been highlighted that lacking clear definitions might hinder the implementation of the legislative framework (E8), and that the challenge is rather ambitious (E1, E2). It should, therefore, be reached step by step (E3). Moreover, its achievement might be deferred for the future member countries, which includes the Czech Republic (E6, E7). There was also hint to various clauses (E9), which would allow for delay in the implementation procedure. Although the interviewees did not explicitly mention the articles of the WFD they hinted to, this might refer to Art. 4, which states that provided no further deterioration in the status of the affected water bodies occurs, under some given conditions (e.g. the completion of the improvements within the timescale would be 'disproportionately expensive'), the deadlines can be extended to a maximum of two further updates of the river basin management plan (see Art. 13), 'except in cases where the natural conditions are such that the objectives cannot be achieved within this period' (EC, 2000b), this means that there still exists an open deadline. Another open question is what 'disproportionately expensive' might mean for different countries (different economic conditions as well as socio-cultural values), and to what extent international cooperation can deal with (and solve) the problems of excessive costs for achieving an 'international standard' rather than many different 'national' standards. More closely on the issue of nutrient emission reduction, two sectors are the most relevant: WWTPs and agriculture. 'In the Elbe catchment 239 WWTPs have been improved or constructed: every WWTP with more that 20000 equivalent inhabitants has at least biological treatment of its wastewater' (E5). In Germany the general situation is good and most of the interviewees see the further improvement of the WWTP as too costly (E1, E5), although the efficiency of smallest WWTPs still can be improved by introducing tertiary treatment (E5). 'In the Czech Republic WWTP connection remains an issue. Through the transposition of the Wastewater Treatment Plant Directive 91/271/EEC into a strategic plan, in the Czech Republic, all communities with more than 2000 inhabitants will be equipped with a sewage treatment plant by 2010' (E7). As far as farming and agriculture are concerned, as a result of the Common Agricultural Policy change in 1992, farming intensity has already been reduced, and further changes are expected through the redesign of the CAP. The regulation EEC 2078/92 on environmentally sound agriculture (not to be confused with 'organic') production methods and the following EEC 1257/1999, obliging Member States to offer funding programs for farmers opting for environmentally sound agriculture, opened the way to 'process integrable incentive' (E8), namely incentives that are most acceptable for the farmers, being part of the productive process and giving them a greater role in care and sustainability of the landscape, without depriving them of their cultural/ historical role.

In 2002 an amendment of the Federal Nature Conservation Act was passed in Germany, redefining the relationships between nature conservation, agriculture and forestry: for the first time rules of the agricultural good professional practice were drawn up. These rules are in no way to be interpreted too literally, since the farmer still faces dilemmas that are not solvable through a series of pre-given solutions (E8). 'The good professional practice (GPP) is a framework of rules, which can not be seen as a fix law. Depending on climatic or spatial conditions and characteristics and on vegetation and culture requirements, the farmers have to act in their sole discretion. The GPP represents a framework containing also legal obligations (e.g. machines for spreading pesticides have to pass regular controls, -called TÜV), but farmers have finally to act in their sole discretion' (E8). Although according to interview E7 in the Czech Republic the use of fertiliser is lower than in some member countries, the whole republic 'will be declared a sensitive area in the sense of the Nitrate Directive' (E6). In the Czech Republic the consumers still pay great attention to low-price products, therefore expensive nutrient reduction techniques are likely not to be implementable without some subsidisation (E6). The respondents stressed the general lack of cooperation, harmonisation (including that of timetable implementation) and synergetic action of the various existing conventions and institutions dealing with environmental issues. 'Too many institutions work simultaneously on the same topics [without communication among each other, Authors' note]. The coordination must be improved' (E3). A better agreement and for instance the approaches of the WFD and the OSPAR Strategy to Combat Eutrophication are similar. However the former aims to achieve 'good ecological status' by 2015 for river basin districts, while OSPAR's target is 'no occurrence of eutrophication in marine waters' by 2010. Moreover, even within the same country, sometimes the fields of work are such that different governmental offices are not able to cooperate and 'the direct concurrence of different governmental offices for water issue and nature protection does not allow to find a common strategy' (E4). Sometimes governmental offices have to deal with contradictory tasks: 'in the Czech Republic unfortunately the Ministry of Agriculture, which is responsible for the main polluter [agriculture, Authors' note] has also the competence for water protection' (E6). The International Commission for the Protection of the Elbe (IKSE/MKOL) is according to many interviewees the most significant institution for international cooperation in the Elbe catchment (E1, E5, E6, and E7). Founded in 1990 with an agreement between Germany, the Czech Republic and the European Commission, the IKSE has been successful in coordinating international action, for instance in promoting and facilitating the spatial division of costs and benefits for the construction and improvement of waste water treatment plants in the Czech Republic (E5). The IKSE represents a strong basis for the implementation of WFD tasks and also for future international cooperation (E1, E5, E6, E7). In the past the main weakness of the IKSE was that 'The IKSE activities focused mainly on the river itself and neglected the tributaries and the river basin' (E5). In the future the focus will be broaden, in order to include the entire basin and the coastal zone (E1). To summarise the main points, tasks and pertinences of different institutions are often overlapped or lack coordination. Extra work is needed because the flow of information between institutions is too slow (E3). Regarding the legislative framework, as far as the WFD implementation is concerned, the institutions still lack in cooperation and synergetic action, suggesting that competences still have to be distributed in a consistent way. Great expectations are set on the IKSE and its re-structuring as one of the main institutions for implementing the WFD. According to the respondents, as far as the nutrient reduction issue is concerned, the WWTP quality gap between Germany and Czech Republic is being levelled up, and agriculture should be targeted for further nutrient reduction.

How to achieve further reduction of nutrient emissions? The majority of the interviewees agreed that measures for reducing nutrient emissions should be mainly applied to diffuse sources, identified with agriculture. Table 3.15 summarises the perceptions of the respondents about feasibility and specificity of some given nutrient reduction measures. Please note that the number of responses (frequencies) in Table 3.15 do not add up to nine in most cases. This is due to the fact that many interviewees just express their opinion about some measures and not about others. Of those, only the ones who stated their refusal to answer are reported in the column 'no judgement'. In general technical measures are welcomed by most of the interviewees, who expect them to be rather effective. Economic incentives are perceived as successful if they are also acceptable for the farmers, which, according to E8, would be the case, if they follow the concept of 'eco-efficiency' and 'integrability in production' (see further). Stricter limit values and regulations are not very welcomed, voluntary agreements and practices are favoured. Lacking controls would not ensure that the limits are respected and, fertilisers' prices should be very high (more than 200%) in order to reduce the use of fertilisers. Moreover, the yield production is so much higher by using fertilisers, that possibly higher prices would lead to black market and commerce, rather than reduced use (E8). The opinions about distribution and re-distribution of subsidies (e.g. decoupling) were different, but some sceptical answers can also be related to particular (hidden) interests or misunderstanding, e.g. decoupling of subvention perceived simply as 'less' subvention. Connected with this measure, the reduction of farming and livestock density (especially of cows) in some specific regions (e.g. Vechta) is seen as a way to reduce nutrient emissions (E2), since 'the surplus of fertilisers for arable land is about 14 kgN/ha, while for farming about 110kgN/ha' (E8). However, since the market demand of meat, at least in Germany, is not satisfied by the local production (E8), this reduction of farming should be coupled with a decrease in demand, if self-supply should be a target. There are numerous studies by Isermann highlighting the (positive) effect of reduced meat demand on reduction of nutrient emissions (e.g. Isermann and Isermann, 2000). The majority of the respondents are dubious about the use of wetlands for nutrient retention. The pilot studies about nutrient retention in wetlands (e.g. Dehnhardt and Meyerhoff, 2002) are usually not known. While the role of wetlands is well recognised with regard to habitat and species enrichment, many of the respondents reacted sceptically at the proposal of using wetlands for nutrient retention and reduction of nutrient flow to the coastal zone. This highlighted, on the one side, the need for more in-depth research and especially dissemination of knowledge, and, on the other side, some hesitation to accept a measure characterised by high uncertainty in terms of its effectiveness. Although the respondents hesitated to fill additional measures directly into the tables given in the questionnaire, in their talks a significant number of additional points were touched, although mention of concrete measures was nearly absent in the interviews. Regarding the suggestion of specific measures the majority of the answers showed evidence of non commitment. The given general statements can be seen as guidelines for nutrient reduction measures and were probably due to inadequate knowledge in this field. In general there was a claim for clear targets (E9, E3), in order to make regulations (WFD) understandable and implementable. The use of the precautionary principle approach was suggested as a way for handling unknown effects (E4, E6), both of measures and of pollutants. In Table 3.15 general the need of disseminating information was highlighted (E7, E8), involving the general public, NGOs and affected interest groups in transparent decision-making processes (E7, E8, E9). The following statement is an example of answers that were given while talking about reduction measures. 'The effects of each single measure [for reduction] should be described properly (also concerning the effects on individuals) also in combination with other measures. Once the relationships cause-effects have been described it is possible to think of scenarios (Decision Support System). An essential role is played by concerned/involved/interested people (stakeholder)... a transparent and correct [decision- making] procedure is more important than [using] perfect measures' (E9). In general, the measures for reaching environmental targets, independently of their specific nature, should undergo a decision-process that after E1 needs consensus and unanimity among

Table 3.15: Frequencies of stated preferences for the given nutrient reduction measures in the Elbe catchment (Nunneri and Hofmann, 2005).

Measure	No judgment	Feasibility		Specificity		Effectiveness	
		High/ medium	Low	High	Low	High/ medium	Low
Technical measures for the agriculture	0	6	2	8	0	7	0
Economic incentives	0	5	2	6	1	6	1
Stricter limit values and regulations	1	1	5	4	1	4	1
Nutrient-elimination WWTP	2	3	1	4	0	3	1
Less subventions for 'traditional' agriculture	1	4	1	5	1	5	1
Higher subventions for env. friendly agriculture	1	5	0	3	0	3	1
Decoupling of subvention/production	0	3	3	5	1	4	1
Increase of fertiliser prices	0	1	3	5	0	5	0
Wetland restoration	1	3	4	5	2	5	1

the different groups representing different interests, but other respondents expressed the need of compromises and possibly political intervention for reaching solutions (E3).

All the interviewees stressed the need of voluntary agreements among polluters in order to define reachable deadlines and reduction targets. Such agreements find higher acceptance among polluters and present the highest possibility of success (e.g. E6). An example of a voluntary agreement for phosphate emission reduction among the Czech government and the detergent producers was given in interview E6: 'The soft legislation agreement between the ministry of environment and the producers of washing powders is a good example. Thus the producers of washing powders have a sufficient –ten years– period (since 1995) to change their technology and from 2005 non-phosphate washing powders will be on the market. If this technology change should be more rapid by enforcements, then the higher product prices would lead to the opposite effect since people would start to import illegal products'. Another reason supporting voluntary agreements is the general lack of enforcement and controls of law implementation, in Germany as well as in the Czech Republic (E2, E3, E7). Despite the respondents' awareness that lacking controls are the weakest point in the implementation of regulations, some measures introducing new controls or requiring more specific controls were proposed during the interviews. 'The legislative framework is sufficient. It is necessary to apply and enforce it. Random controls should be carried out more often. The problem is the lack of personnel. Voluntary agreements are effective if it is possible to enforce compliance.' (E2) Among the proposals for reducing agriculture nutrient emissions, also some that would be based on increased controls were mentioned, such as introducing a compulsory nutrient balance for farmers (E1, E9) and regular controls of final quality of agricultural goods (E7). 'The policy of emission trade should be avoided since governments can free themselves from their ecological responsibility' (E2). The proposed additional measures for reducing nutrient emissions are listed below:

- increase of extensive agriculture (E4);
- apply regional specific solutions (E3), each region should deal with specific problems, there is no 'general solution';
- regulation of agriculture near water bodies through land-easement and buffer zones (E9);
- reduction of drainage systems (E9);

- training and counselling of farmers (all interviews);
- application of good agricultural professional praxis (in particular E3, E5, E7);
- acceptable and integrable agricultural subventions (E8), concept of integrating environmental protection in agriculture production;
- application of the BAT (best available technology) (E7);
- N-min method (E5), consisting in analysing the soils in order to decide whether they are still rich enough for agriculture or if there is a need for fertilisers (nitrogen).

Some interviewees have mentioned good professional practice as the solution for nutrient emission reduction. On the other hand, some disagreed on this point (E8), highlighting that this regulation is no cookbook, and still farmers face dilemmas, for which there is no '(best) given solution'. In this context, training and counselling of farmers turned out to be a major point. All the interviewees stressed the necessity and the need of a dialogue with the farmers, in order not only to disseminate the actual state of knowledge (regarding BAT, as well as 'good professional practice'), but also in order to understand and elaborate their own needs. Certainly communication weaknesses between policy-makers and key-stakeholders (farmers) and resulting acceptance of policies (or opposition to them) are two of the major points responsible for failure of measures for environmental friendly agriculture. Cudlínová et al. (1999) give an example of inefficiency of subsidies for environmental-friendly agriculture in the Czech Republic and ascribe it to ignorance of the social factor ('who' are the recipients, what are the production patterns) in the region. Examples of highly acceptable subventions are those called by E8 'integrable subvention'. According to E8 these subventions must, on the one side, undergo the principle of 'ecoefficiency, which aims to combine ecological and economic efficiency (economic incentives)' and, on the other side, 'Governmental incentives are incentives of minor importance if they can not be integrated in the production process. Examples of acceptable incentives are for instance offering advice for free about water-protection or environmental programmes, e.g. for using fallow land as buffer zones near small water bodies. This possibility is foreseen by the EU, but has not found [political, Authors' note] acceptance in Germany up to now' (E8) E8 brought up the issue of communication between decision-makers and key-stakeholder (the farmers). Strongly related to this point are:

1. Refusal of political targets (e.g. the 50% reduction of nutrient emission with respect to the reference year 1985). Those targets usually do not find the acceptance of the farmers, since they are neither comprehensible nor logical for them.
2. Possible non-cooperation of the farmers for the implementation of the WFD, due to the fact that they have not be involved in the preliminary studies or definitions for the implementation of the WFD, and will be then reluctant to accept the results of this work.

As an example for failure of communication and recalcitrance of farmers E8 mentioned the construction of the ICE Railway in Germany: as the Deutsche Bahn (German Railways) tried to buy land from the farmers for constructing the new railway route, a general refusal of selling made impossible to proceed any further. A private company took over the negotiation and carried out a survey asking the farmers how they would see their future activity. Based on the responses (ceasing activity, expansion, other planned changes), it was possible to organise area exchange among farmers and buy (few) areas in order to realise the Railway project.

Costs of reduction This issue has already been mentioned. Many interviewees agreed that costs and benefits should be (re-)distributed in such a way that there are no net losers or winners (e.g. E3, E5, E8). In this context 'farmers' income depends on their use of arable land, therefore if they contribute to nature protection they need financial compensation' (E8). Three general principles were mentioned:

- the polluter pays principle (E4, E9, E2): those who are responsible for pollution have to bear the costs of pollution reduction;
- the solidarity principle (E4, E5): the costs of reduction must be divided in such a way that are bearable for all parts (e.g. construction of WWTP in the Czech Republic with economic help from Germany), especially applicable when the most well-off country holds the benefits;
- the compensation principle (E3, E6): the economic losses due to application of reduction measures should be compensated.

Financial difficulties should not hinder (as far as possible) environmental improvement: 'The money should be not used in a specific country, but for a specific issue' (E2), thus allowing for a win-win solution. A further criterion for justifying expenditures is that 'the relationship cost/reduction should be clear, in order to decide for which goal we are spending money and which successes are likely to take place' (E8). Fulfilling this criterion in the case of a multi-causal effect as eutrophication, seems not to be possible, rather, there should be a precautionary principle orientation. Nevertheless this criterion represents an important feature for the acceptance of (and the willingness to pay for) measures. The issue of 'who is going to pay for what' is a major point in the present society. Although all the respondents are animated by a theoretical approach based on social equity and justice, the financial burden sharing in the implementation of environmental measures is a delicate issue containing in itself the potential for triggering conflicts.

Criteria for choosing among measures The respondents were asked to evaluate given criteria for selecting measures. The results are reported in figure 3.23. The given criteria can be sorted into two groups, the first one (from the left to 'low uncertainty in the reduction'), in which the respondents considered the criteria relevant or indispensable, and the second one (from 'low uncertainty in the reduction' to the right), in which there is more uncertainty about the importance of the criteria. Nutrient reduction specificity would be (as expected) the first criterion together with political and economical acceptance, followed by technical feasibility. Public acceptance and ecological side effects are also ranked very important and absolute costs play a minor role if compared with cost-effectiveness. Despite this, absolute costs are indispensable for one and relevant for five out of nine interviewees, which suggests that there is a threshold, over which absolute costs play a significant role in the choice (or the refusal) of a measure.

Relatively less importance was given to low uncertainty in the reduction specificity, i.e., in simple words, how large is the range in which the reduction obtained is to be expected by applying a certain measure. Although this might be related to the understanding of the concept of uncertainty, this does not agree with the general refusal of wetlands as a measure for reduction, which was justified by reference to the scarcity of available knowledge regarding the possible role of wetlands for nutrient reduction. A general interpretation can be that the criteria of the group

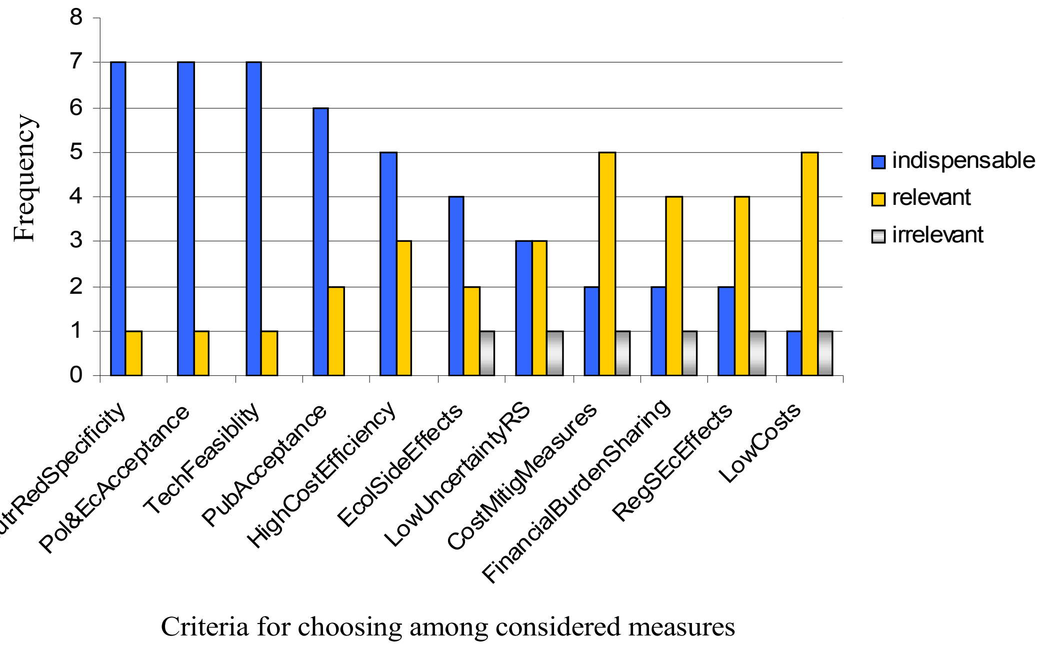


Figure 3.23: Ranked criteria to be used for evaluating measures. Please note that one respondent refused to answer this question, therefore the maximal frequency is eight. Not all the remaining respondents did rank all criteria, such that in some cases the total frequencies do not add up to eight (Nunneri and Hofmann, 2005).

on the right will increase in importance depending on their magnitude, for instance very high costs are a very valid reason for not implementing a given measure, even though it is efficient. The criteria of the other group can be seen as ‘present perception criteria’ or even as ‘politically correct answers’. In order to test the robustness of these evaluations of criteria, the results of the MCA carried out using these data will be shown to the respondents and they will be given the possibility of re-thinking their evaluations based on the results. Interviewees E4 and E8 added to the proposed criteria the clear cause-effect relationship of measures, as an indispensable criterion, also related to costs of measures (whoever pays has to know that they are not paying in vain).

Outlook for international decision-making

If the target of nutrient emission reduction in the Elbe catchment (seen as ‘one large source’ of nutrients for the North Sea) should be reached, the economic and socio-cultural differences between the Czech Republic and Germany cannot be ignored in the process of international decision-making. Based on the interviews (with no pretension of completeness), the main perceived difference between the Czech Republic and Germany is connected with the current economic situation. Although 'formally' the issue of nitrate emission reduction has been assessed as a priority in both countries, the Czech institutions mentioned 'economic growth' as the top priority in their country. In the Czech Republic further economic development is a constraining factor for the implementation of reduction strategies: the public (and therewith the political) willingness to pay for environmental goods is still lower than in Germany. This implies that economic support should be made available, if the final goal is that of implementing reduction strategies and not that of a formal agreement on common objectives. Both the Czech and German institutions addressed the issue of cost sharing, although in different ways. While the Czech institutions expressed their confidence in the future role of EU subventions (e.g. subventions for environmental-friendly agriculture), the German institutions agreed that funds should be made available for dealing with a specific issue rather than for a selected region or country. Such a solution would allow inter-regional and international distribution depending on the real needs (i.e. facilitate flow of money from Germany to the Czech Republic). In the case of EU subventions, as mentioned before, their effectiveness upon emission reduction is doubtful, if no further dialogue with the key-stakeholders –the farmers and the main industrial polluters (e.g. the washing powder producers in the Czech part of the Elbe)– takes place. On the other hand, an ineffective institutional framework, (as currently perceived) in both Germany and the Czech Republic (conflicting interests and lack of coordination), may hamper the sharing of issue-specific funds. In this context, the IKSE in its current restructuring has the potential to gain a central role for coordinating actions and acting as a catalyst for communication among different institutions (at the national and international level). Particularly, the IKSE could support the development of a 'forum' for participatory decision-making. This procedure would allow the setting of (achievable) goals and, in the ideal case, would involve all interested parties, including 'polluters' and the general public. In this way, once the priority issues and conflicting interests are addressed, a cost sharing based on socially acceptable conditions could be achieved.

3.2 Offshore wind parks

'Success in managing the marine environment is only as good as the ability to predict the consequences of human activities' (Elliott, 2002). In the case of new uses of the sea, the DPSIR scoping framework provides the first possibility to shed light on possible cause-effect chains and thereby offers the first input for management improvement. Offshore wind represents a new use of the marine area and the study of ecological impacts is still in his fledging state. In this section, with no pretension of completeness, some key-aspects related with the development of an offshore wind sector and thereby with installations of wind turbines in the sea are analysed along the DPSIR scoping framework.

The core of this section bases on three papers submitted for publication, a brief overview of this section is given in the following.

Section 3.2.1 reports a policy analysis study submitted for publication in Regional Environmental Change. This study deals with the socio-economic factors than can play a role in promoting or hindering the development of offshore wind farms. Based on a comparison between two North

Sea countries, the UK and Germany, some main aspects (e.g. regulatory framework, power distribution and financing issues) are analysed with respect to their ‘efficiency’ in promoting offshore wind. The paper includes insights from interviewed key-actors in the offshore sector. In particular the comparison between Germany and the UK allows for distinguishing how similar needs and goals may result in different frameworks and ultimately actions due to different socio-economic aspects (culture, values and approaches).

Section 3.2.2 briefly reports five scenarios assessed for future development of the German North Sea uses (including or not offshore wind farm construction). Those scenarios are based on social perceptions and needs and result ultimately in construction of offshore wind farms to a different extent. Installation of wind turbines in the sea, is taken as a pressure for assessing ecological risk in the marine environment.

The impact analysis reported in section 3.2.3 has been submitted for publication in Gaia. The study analyses the ecological changes associated with the construction of offshore wind parks in the German exclusive economic zone in terms of ‘ecological risk’. The short-time effects of wind park construction upon the ecosystem are simulated with ERSEM under the assumption that changes of SPM concentration due to installation of offshore turbines, temporary affect the ecosystem integrity and thereby the ability of the ecosystem to provide (supporting) services.

Finally section 3.2.4 reports an assessment of the German offshore wind farm construction scenarios in terms of their costs (capital investments and operation and maintenance) and their benefits (assessed in terms of greenhouse gases emission reduction). Those aspects, assessed in monetary terms, are factored into a cost-benefit analysis (CBA). Some considerations related with conventional energy production and ecological risk associated with the considered offshore wind scenarios conclude this section.

3.2.1 Drivers: the need of renewable energies

Besides major environmental issues of global interest (climate change, sustainable use of resources and minimisation of toxic/dangerous wastes and emissions), issues of political and economic nature are behind the recent political engagement in favour of renewables, namely independence from third countries and thereby security of supply and price stability (Jansen and Uyterlinde, 2004; Horvath, 2006). In the context of the EU Kyoto commitments (EC, 2006) and the predetermined goals for renewable energy formulated in the RES-E directive (EC, 2001), the EU sees in (offshore) wind one of the most promising technologies for pushing renewable energy growth (EC, 2001). Wind energy is in many aspects among the most technically and commercially advanced of all renewables, with steadily increasing installed capacity worldwide during the last decade (Foxon et al., 2003). However, as discussed below there have been problems of durability of components in the harsh marine environment. The cumulative wind power capacity operating in the EU (EU-25) exceeded 48,000 MW in 2006. About 43% of the installed capacity belonged to Germany alone (more than 20,000 MW), 4,1% to UK (EWEA, 2006). Yet, if the success of renewable policies for offshore wind power has to be measured in terms of offshore installed capacity (Butler and Neuhoff, 2004), it is apparent that the UK have been more successful than Germany —currently ca. 300 MW are installed in the UK vs. ca 7 MW in Germany (Lönker, 2005).

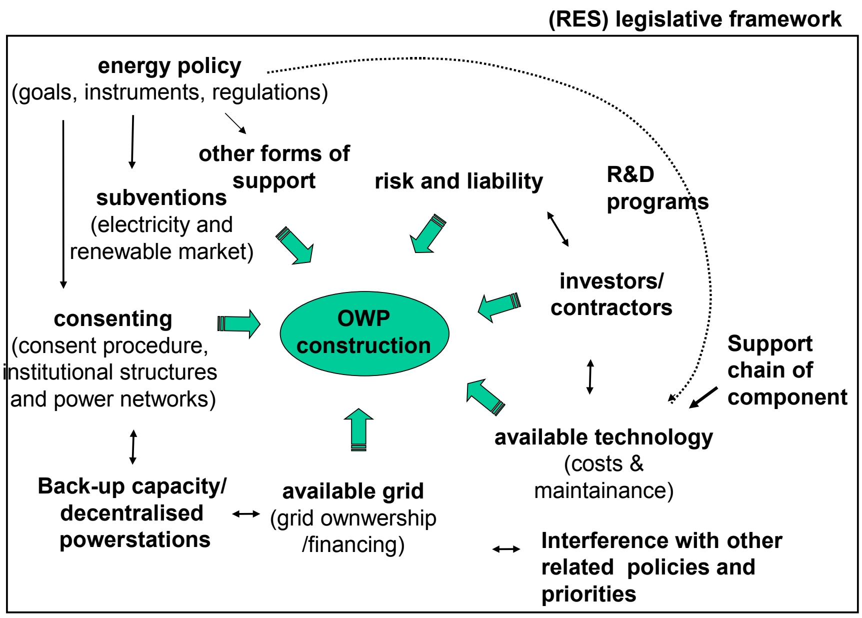


Figure 3.24: Schematic view of factors shaping the development of offshore wind farms.

This paper [8] focuses on a comparison of the offshore wind energy development in the United Kingdom and Germany and aims at highlighting what aspects of the complex context surrounding the development of offshore wind parks (see 3.24) play a key role in facilitating (or hindering) offshore wind energy development and how. The analysis is based on literature review and interviews with selected key-actors (23 in total) including British (15) and German actors (8), who will be anonymous in the following analysis. The interviews were carried out in the context of two main events: (1) the German business visit to the East of England organised by the POWER project and (2) the BWEA Offshore Wind Conference organised in Glasgow; both events took place in October 2006. Figure 3.24 shows the main aspects playing a role in the development of offshore wind projects. The construction of offshore wind parks is embedded into a general policy and legislation framework. Renewable energy is seen as a way of reducing emissions causing climate change (in the amount reported in the Kyoto agreements) and was formalised for the EU-Member countries in the RES-E directive. Broadly speaking, all the aspects reported on the left side are directly influenced by government commitment. At the national level governments are responsible for setting goals and choosing instruments and regulations for achieving them, this includes the realisation of planning and consent procedures, the guarantee of participatory structures and availability of financial supports for new technologies. On the right side of figure 3.24 the other main aspects and actors involved are reported, namely investors, manufactures, developers and contractors, as well as utility companies, who need to asses project-viability based on technological know-how, costs, risks and revenues.

[8] The study reported in this section has been submitted for publication in REC as: Nunneri, C., Licht-Eggert, K., Kannen, A., Tovey, N.K., and Turner, R.K., Governance and offshore wind parks: a comparative analysis for Germany and the UK.

The assumption underlying this study is that policy networks and power associated to key actors must be different in the two countries, in order to explain these developments (van Waarden, 1992). A detailed analysis of the interwoven interaction exposed in figure 3.24 would not be possible in the available time-frame, therefore we concentrate on testing the following hypothesis:

1. In the UK the Government has been more committed and set a ‘more efficient’ framework for the development of offshore wind parks (e.g. faster consent procedure than in Germany, clearer regulation and responsibilities);
2. The offshore wind business is financially more attractive in the UK than in Germany, in particular:
   a) The capital investments/MW installed capacity are in the UK lower than in Germany/ the financial support through Government and legislative frameworks is in the UK higher than in Germany.
   b) UK Round 1 pilot projects have been realised faster because they present less (economic) risk than larger German projects to be realised far away from the shore.
3. Offshore is in the UK an exclusive business of larger companies, which have available knowledge and funds to easily go through the consent procedure and the realisation phase, while in Germany large companies have rather opposed offshore wind development

In the following, after a brief introduction about the current state of offshore wind projects in the two countries, the main findings about the issues noted above are discussed.

Current state of offshore wind projects

The long term strategy of the offshore wind energy development differs in total capacity and time-scale between the two countries being the target of German Government 20-25 GW by 2030 (BMU, 2002b,c) and the one of the UK Government 5.4-7.2 GW total installed capacity by 2008-2009 (BWEA, 2005). Given those targets, the current achievements of both countries are very modest if compared to those goals.
Although policies for promoting renewable energy started much earlier in the UK and Germany, both countries set new frameworks in 2000, the UK by promulgating the Utility Act, of which the Renewables Obligation is part, and Germany by issuing the Renewable Energy Source Act. Since then (over the intervening six years), real progress has been very different for the two countries (figure 3.25). The short term objectives for offshore wind farm installations, up to 2010 are very similar (between 3 and 3.6 GW for the UK and 3 GW installed capacity for Germany), but progress towards those targets has been implemented with different speeds in the two countries.

In both countries a similar number of projects received approval. In Germany fourteen projects with total capacity of 5 GW ((Lönker, 2005; Kannen et al., 2007) and in UK thirteen projects for a total of 1.2 GW have been consented between 2000 and 2006. Of those, in the UK, four ‘nearshore’ projects with a total capacity of ca. 300 MW are already generating for the grid in the UK ((BWEA, 2005), whilst in Germany only two single turbines stand in the 12-mile zone (Lönker, 2005; BWE, 2007). Moreover, in the UK further projects for a total of 582 MW are in

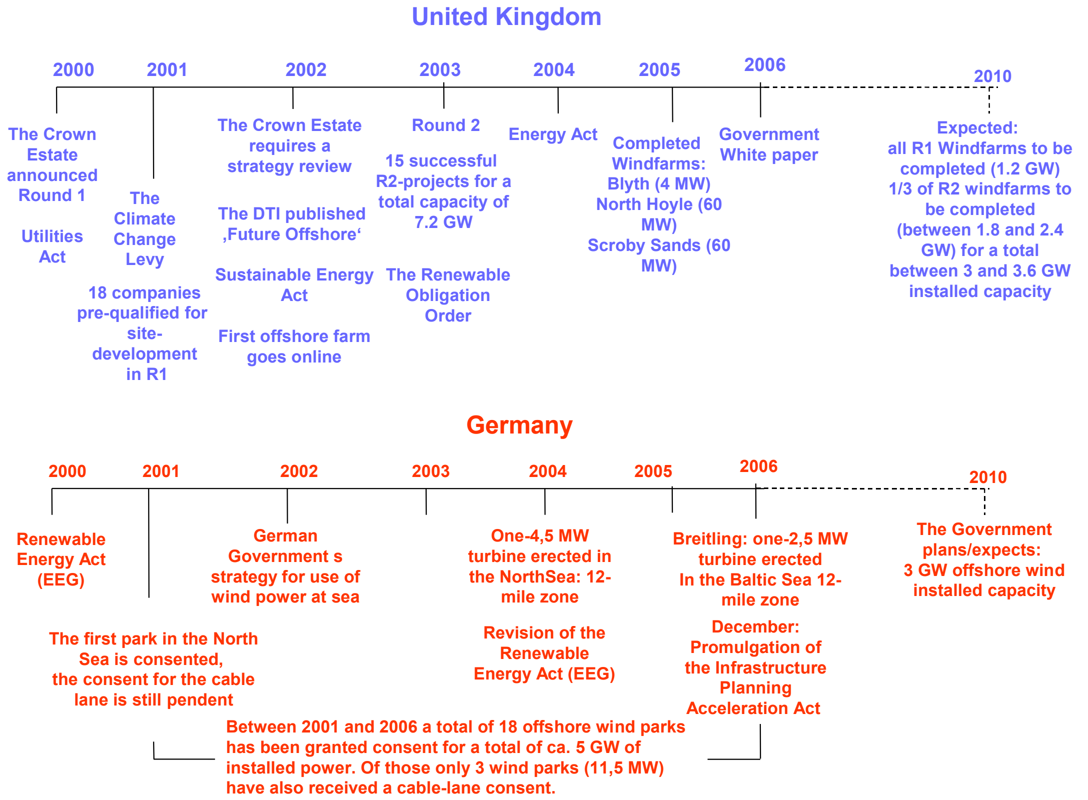


Figure 3.25: Main steps towards the realisation of offshore wind projects in the time-frame 2000-2006.

construction and a further 2088 MW have received approval (BWEA, 2007). The ‘consented’ projects in the German North Sea have a maximum total capacity (depending on the capacity of the installed turbines) of about 32 GW (in addition about 1 GW is consented for the Baltic Sea, where projects are less numerous). Of those projects in the North Sea only three received ‘full consent’ (i.e. including cable line-consents) in February 2007.
The UK Round one projects (both realised and consented) consist of wind farms with a maximum of 30 turbines located near the coast (3 to 9.5 km) in relatively shallow waters. In Germany, the first –from BSH, the German Federal Maritime and Hydrographic Agency – consented projects allow for a maximum of 80 turbines to be erected up to 90 km offshore and in water depth of up to 35 metres. However, most of these projects still need cable consent.

Commitments of Governments

The national goals for greenhouse gases emission reductions and renewables are shown against the European targets in figure 3.26. EU leaders set a firm target of cutting 20% of the EU’s greenhouse gas emissions by 2020 and will be willing to put this goal up to 30% if the US, China and India make similar commitments. EU leaders also set a binding overall goal of 20%

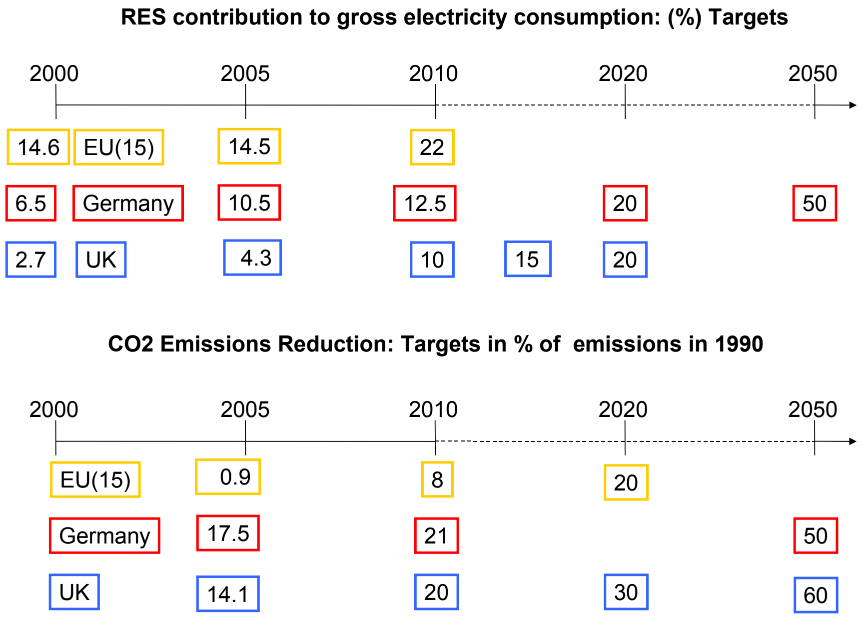


Figure 3.26: Main declared goals for CO_2 emission reduction and renewable energy on behalf of the EU, the United Kingdom and Germany.

for renewable energy sources by 2020, compared to the present 6.5% (EC, 2007). According to Blowers and Hincliff 2003 the significant CO2 emission reduction achieved by Germany and the UK are the results of changes other than implemented climate actions and both countries are unlikely to meet the Kyoto Protocol targets in the years to come unless real action is taken. According to available information, in 2005 Germany had reached about 70% and the UK about 30% of their respective targets for renewable energy (EUROSTAT (data from 2007)). In this context, behind the faster development of offshore wind farms in the UK there could be the 'urgency' of progressing towards 2010 targets. On the other hand, Germany might have taken time to optimise development, as it is further on the road towards meeting its objectives. If this perspective is accepted, than it is comprehensible that in Germany a more precautionary approach giving priority to nature protection in the coastal zone has been adopted (see further, Kruppa (2007).

The key German law providing incentives for implementing renewable energies is the 'Gesetz für den Vorrang Erneurbarer Energien', 'Renewable energy Sources Act' (EEG 2000/2004). The EEG obliges grid operators to prioritise the purchase of renewable electricity. Moreover, the grid operators are obliged to extend the grid in order to facilitate the input of renewable electricity, as long as the costs are 'bearable'. This means that responsibility and costs for feeding renewables into the grid are born in the first place by the grid operators. To avoid market distortions, the extra costs incurred by individual grid operators from purchasing green electricity (and paying for it at a price higher than the market price, due to the feed-in tariff) are shared nationwide and passed on at a uniform rate to consumers. In this a way that the state has no extra financial burden (Wüstenhagen and Bilharz, 2006). A guaranteed minimum purchase price (feed-in tariff), based on costs, technology and location, is guaranteed for the long term (20 years) in order to help the economic viability of renewable energy projects. For offshore wind the feed-in tariff is currently 9.1 €Ct/kWh for those farms starting production until 2010 and will decrease in time for those projects realised later on.

The UK Government's policy about offshore renewable energy was set in the 2003 White Paper and reviewed recently (DTI, 2006; HMSO, 2006). The Utilities Act 2000 gives the Secretary of State the power to require electricity suppliers to supply a certain proportion of their total sales in the United Kingdom from electricity generated from renewable sources. The UK financial support scheme for renewables is the Renewables Obligation (RO), enforced by the RO Order (2002), which requires all licensed electricity suppliers to obtain an increasing proportion of electricity from 'eligible' renewable source. In the UK it is the Regional Electricity Supply Companies (who in some cases are also the Distribution Network Operator) that purchase renewable electricity from the wind farms. The achievement of this production of renewable energy is demonstrated by possession of Renewables Obligation Certificates (ROCs), which are released by the Office of Gas and Electricity Market (Ofgem). At the end of each RO-period (each year in April), suppliers who have a shortage of ROCs compared with the due amount have to pay a penalty (buy out). The buy out fund will then be re-distributed among those suppliers holding ROCs. A Renewables Obligation Order is issued annually detailing the precise level of the obligation for the coming year-long period of obligation and the level of the buy-out price.

As exposed, both countries have set goals and put in place policy instruments for renewable energies development. This means that economic incentives and political legitimation of investments into the generation of renewables as well in offshore wind farms are provided. The question is how those instruments are implemented in practice and if their use is sufficient for promoting offshore wind energy.

Legal framework for offshore wind farms Both countries have realised a legal framework for the erection of wind farms. The English Energy Act 2004 establishes a Renewable Energy Zone (REZ), adjacent to the UK's territorial waters, within which renewable energy installations can be erected. The British REZ corresponds in this sense to the German Exclusive Economic Zone (EEZ). The Act enables The Crown Estate to award licences for wind farm sites and other renewable energy projects in the REZ by extending the requirement for consent under section 36 of the Electricity Act 1989. The Act facilitates a streamlining of the consents process for projects within the REZ and also in inshore waters by providing for navigation matters within section 36. Under section 36, navigation rights can be extinguished in order to accommodate a wind farm project (DTI, 2005). This, however, can be done only in territorial waters, provided that the navigation route is not of international relevance. The legislation introduces two new features – a safety zone scheme and a statutory scheme for the decommissioning of offshore renewable energy installations and related electric lines (DTI/MCEU, 2004). The UK-procedure requires an approved decommissioning program to be in place before construction begins (Energy Act, Chapter 3, Part 2). The development of offshore wind is organised in Rounds of bids announced by the Crown Estate for site allocation.

The German 'Seeaufgabengesetz' (Federal Maritime Responsibilities Act), implemented by the 'Seeanlagenverordnung' (Marine Facilities Ordinance), which is based on the UN Law of the

Sea (UNCLOS), requires that a wind farm project can be denied approval if: (a) it does impair the safety and efficiency of navigation, and (b) it is detrimental to the marine environment (BSH, 2005). Art. 3 of the Marine Facilities Ordinance, provides that an approval is a non-discretionary administrative act, i.e. in the absence of both of the above reasons for refusal applicants have a legal claim to approval (BSH, 2005). There are, however, possibilities to accommodate projects that demonstrate negative effects under point (a) or (b), provided that (a) negative environmental effects can be limited (e.g. by prescribing precise construction times) or (b) the risk of shipping collisions is judged minimal or acceptable. Decommissioning guidelines are of general nature in Germany and only financial coverage for decommissioning must be guaranteed at the time of application (BSH, 2005). The German legislation gives thereby clear priority to navigation, although mitigation measures can be considered. In the UK the possibility of giving consent in spite of proof of adverse effects is foreseen provided that: (1) there is an imperative overriding public interest (2) there is no alternative solution and (3) compensating habitats are provided for the loss of the site (CEFAS, 2004). In the situation where the BSH has received several applications for the same site, the application which first met all requirements for approval (i.e. all documents needed are available to the approval authority) is decided first (Art. 5 of the Marine Facilities Ordinance). In this context, it is not possible for applicants to block an area just by submitting an incomplete application, as the BSH does examine multiple applications for the same site at the same time as long as none is complete (Kruppa, 2007; DENA, 2007). Nonetheless, once documentation is completed, the German procedure applies a first come first served approach, thus resulting in project-planners competing against time for sites and revenues. Once obtained consent, the developers are currently given time up to 2011 for beginning construction (as a consequence of the Infrastructure Planning Acceleration Act 2006). Cooperation among different planners is a wish of the regulator (as a German regulator affirmed), although this strategy has not been implemented so far by any developer, and there is no regulatory facilitations in this direction. Under the current framework, cooperation possibilities only foresee joining cable lines of neighbouring farms in one single bundle to the mainland.

Consent Procedure In both countries wind park developers need to go through a number of requirements before they are granted consent and can start construction. UK developers need to wait for a round of bid from the Crown Estate (Round 1 and Round 2 have already taken place), and need to receive an agreement for lease on behalf of the Crown Estate, before they can submit applications to the Department for Trade and Industry (DTI). German developers can submit applications at any time, and processing takes place accordingly on behalf of the authorising body, the Bundesamt für Seeschifffahrt und Hydrographie (BSH, Federal Maritime and Hydrographic Agency) for installations in the the EEZ and the 'Länder' governments for installations in the territorial waters. A schematic representation of the two consenting procedures is reported in Figure 3.27 and 3.28. The advantage of the UK 'Round' strategy is that experience form one round can be used for addressing emerging problems in following rounds, thus enabling the Government to change the institutional and regulatory framework before opening another round of bids. This allows the government to direct planning in the Sea, by choosing suitable areas for offshore installations. According to the EU Strategic Environmental Assessment (SEA) Directive (2001/42/EU), which requires the identification and mitigation of likely significant environmental effects of certain plans and programmes, the Department for Trade and Industry (DTI) commissioned a SEA ahead of the Round 2 offshore wind farm leasing announcement, which provided three lease sites for wind farm development in Round 2 (the North West, Greater Wash, and Thames Estuary). And as a result of this analysis and stakeholder participation a coastal strip between 8 and 13 km from shore has been excluded from possible construction sites. In the case of Germany, area pre-selection is realised in two ways: on the one hand, by not paying feed-in tariffs to projects located in protected areas and, on the other hand, by the designation, by the BSH, at the end of 2005 of three suitable areas (Nördlich Borkum in the North Sea, Kriegers Flak and Westlich Adlergrund in the Baltic Sea) where faster licensing procedures will apply (BMU, 2007). Those areas have been declared particularly suitable for offshore wind energy on the base of shipping security and nature conservation. In the course of an approval procedure, location in one of those areas will be regarded as an additional positive aspect in the appraisal report. Contrary to the procedure in the UK, area designation in Germany does not prevent applicants from applying for different sites. In other words, it is not a planning instrument in the hand of the government (Kruppa, 2007). Although only named as such in the UK, the designation of suitable areas is akin to a Strategic Environment Assessment (SEA), which gives both authorities and applicants the opportunity to identify at an early stage possible impacts and mitigation strategies (EC, 2005), thus clearing the requirements for the applicants' Environmental Impact Assessments (EIA). Interviewed stakeholders, in particular developers and experts from both countries complained about the duration of the consent procedure. The shared general perception is that requirements for grid connection and consent procedure are far too slow. Nonetheless UK developers need about 3 years in total for receiving notification of acceptance: the DTI tries to keep the consent procedure at one year and two years is about the time needed for planning and submitting application (interviewed UK-experts). During Round 1, the quickest consent under the Electricity Act –in the REZ) took five and half months, the longest took over 13 months which made an average of 9 months for Electricity Act applications; it was slightly longer for Transport and Works Act applications (within the territorial waters), which took on average 12 months to process (interviewed UK regulator). For Round two, however, this time-frame has already been overtook, as the longest consenting time has been 18 months. The German BSH takes 2 to 3 years for consenting a project (Lönker 2005, interviewed German regulator), the cable consent for territorial waters can take up to another 2 years . In total developers need a time up to six years for obtaining approval notification: the extreme case being characterised by scarce transparency and stepwise requirements of new documents on behalf of the BSH (interviewed German developers). In addition to this, BSH consent is not 'full consent', as cable consent in territorial waters is of pertinence of the federal states and a juridically separated procedure. As an example, the three projects receiving cable consent in 2007 had already received BSH consent as early as 2004. This means that cable consent alone 'delayed' the final approval of two years (see further).

'All inclusive' consent in UK, 'step by step' in Germany A delicate aspect for the German procedure is the consent on land and within the territorial waters (up to the 12 nautical mile limit), which is of pertinence of the federal states, while the State is responsible for approving construction of the wind farm and cable lane in the EEZ until the territorial water limit. This juristic aspect related to the German federalist political structure, affects cables and grid connection. This results in cable lines needing two permits: one from the federal state –within the 12 nautical mile limit– the other from the BSH –covering the reach from the 12 miles to the wind farm location. Consent granted by the BSH for installations in the EEZ (including cable) is not

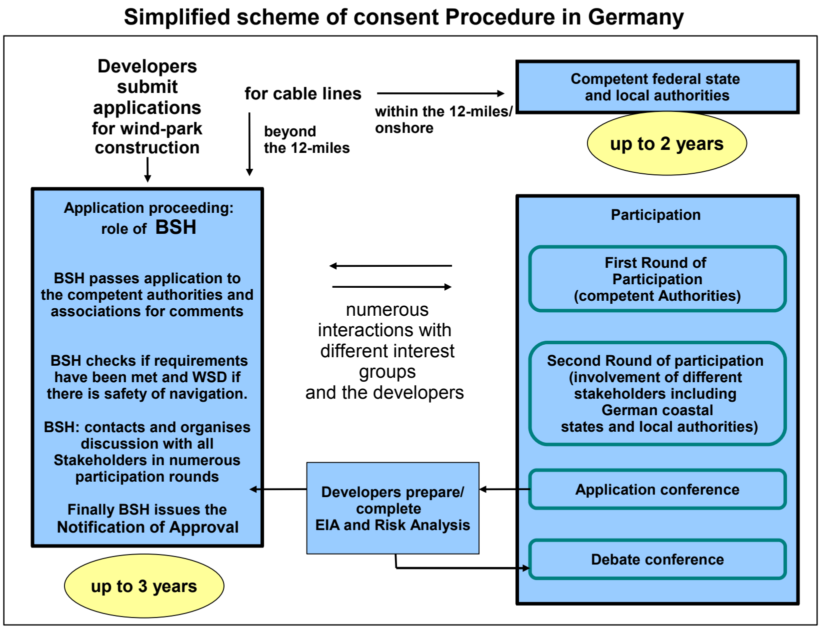


Figure 3.27: Schematic representation of the main stages involved in the consent procedures in Germany. An estimation of the duration of different stages is given, based on the interviews with regulators.

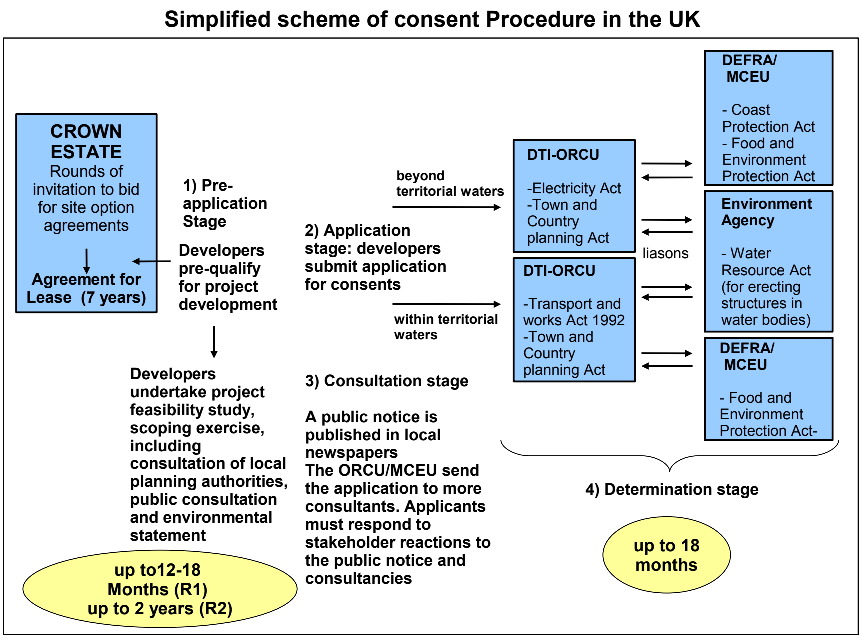


Figure 3.28: Schematic representation of the main stages involved in the consent procedures in the United Kingdom. An estimation of the duration of different stages is given, based on the interviews with regulators.

legally binding for approval procedures involving installations on land and in the territorial sea. This means that, while consented UK projects are ‘ready to go’, the majority of the so called ‘consented’ German projects only have received consent for wind farm construction, but not for the complete cable connection. Complete cable line consent is pending for most of the Projects already consented in the German EEZ (Lönker, 2005), three projects received full consent early in 2007. The German legislation does not foresee an ‘all inclusive consent’, thus obliging regulators to bridge normative gaps. A German regulator pointed out two major issues related to cable consent: (1) no environmental impact assessment (EIA) is required for cable line consenting in the territorial waters, although these are the most vulnerable areas in the North Sea, in addition to that (2) no stakeholder dialogue is foreseen. Furthermore, the cable needs to cross the national park protected area, thus requiring a special permit (Lönker, 2005). Although by law no EIA is required for the cable lane by law, in practice the authorities require the developers to provide the documents needed for an EIA and proof the impacts on the environment. Obtaining cable consent in territorial waters can last years because up to seven different authorities, which oft lack consensus, hold responsibilities. According to Lönker (2007) the BSH should take over coordination and supervision of this process. This would also imply taking decision-power way from some federal agencies. However, this is a delicate aspect involving the political federal structure of the country and that may face major opposition in practice. A German developer added that during cable-consent procedures legal actions are common, even after having reached informal consensus about a cable route, thus highlighting that power games and interest conflicts play a major role in ‘paying back’ to developers the ‘disruption’ of other activities through the planned project. In this context, the BSH has developed a strategy to cope with insufficient regulation, namely that of delaying consent for the offshore cable line until consent by the local and federal authorities is given. This allows the regulator to require changes, if the cable line in territorial waters should be judged ‘sub-optimal’ (as the same interviewed regulator said). This aspect, requiring the regulator to supply to legislative gaps, is a major-bottleneck in the German consent procedure.

In the UK, the Secretary of State is responsible for all development consents for offshore renewable energy proposals of above 1 MW capacity in waters around England and Wales (except those applications under the Transport and Works Act 1992 in Welsh waters) as these projects are outside the jurisdiction of local planning authorities (LPAs), which does not extend below the low water mark. For onshore works related with offshore wind farms, developers have the option of (1) asking the Government for consent called ’deemed planning’ (where the government gives consent but the LPAs are involved as consultees) or (2) separately seek local authority approval under the Town and Country Planning Act. In addition to this, a Port Authority may need to supply a River Works Licence for any work taking place within it jurisdiction. Only land-connection and transformer buildings comes under the jurisdiction of the District Council who must respond within 8 weeks. Developers have the right to appeal to DCLG – Department for Communities and Local Government– and the Secretary of State can over-rule a rejection of planning consent by a Local Authority. Local authorities can force a public inquiry in respect of onshore works where they object to offshore wind farm projects. For example the London Array offshore farm requested consent for the onshore works under the Town and Country Planning Act from the local authority. This request was rejected by the local authority. The developer can only overturn the local authority’s decision through a public inquiry. The result of the public enquiry, largely dependent on the inspector’s report, is awaited (interviewed UK experts).

Power Games: who is given the power to oppose, hinder or delay offshore wind development? Both in Germany and in the UK there are 'powerful' and 'less powerful' stakeholders. In Germany the approval procedure is organised in two rounds of consultation. Stakeholders in the first round can oppose consenting. These stakeholders are national authorities and industries with a national remit in the marine environment who cannot prevent offshore wind farms from construction but can suggest alternative locations for proposed offshore wind farms or restrict their size (because of conflicting interests). All of them represent interests in general marine issues (environment, waterways, defence, fishery) or major users of the marine areas like pipeline and mining industry and telecommunication (Gee et al., 2006; Licht-Eggert and Gee, 2006). Of those, one of the most powerful stakeholders are the 'Waterways and Shipping Directorates' (WSD) due to the strong position given to safety of navigation by the UN Law of the Sea (UNCLOS). A second round of consultation includes national and regional interest groups and local stakeholders, including district administrations. Second round stakeholders only have limited influence on the consent procedure. Although they cannot prevent wind parks from being constructed in the selected location, they can ensure that safety and environmental aspects are taken into account. Additionally, there is a requirement to inform the public, which mostly takes place through notices placed in national and local newspapers (Kannen et al., 2007). If, on the one side, German regional and local stakeholders are unable to influence the consent process directly, on the other side they can, by giving or withholding consent for the cable connection to the mainland, considerably delay projects. In fact wind farm construction cannot begin until a cable route has been agreed upon and all planning regulations have been taken into account (Knight, 2006; Kannen et al., 2007).

In the UK stakeholders are involved officially in the planning of offshore wind farms in two ways. The first is the SEA in order to select suitable areas ahead of a round of bidding. The second is the evaluation of the Environmental statement submitted with the application form, upon which they are asked their opinion by DTI and DEFRA. Stakeholders that are obliged to be involved in the process (called 'statutory consultees') have the power to oppose consenting. The statutory consultees are governmental agencies, local authorities, national parks authorities and other nature conservation bodies or organisations protecting cultural and historical interests as well as the regional development agencies (BWEA, 2002). Although, in principle, any stakeholder can object to offshore wind farm schemes regardless of their locations, whether in territorial waters or in the REZ and the secretary of state will evaluate objections. However, in order to avoid their projects being be turned down, developers consult the major statutory consultees while they prepare their environmental statement. The Ministry of Defence (MoD) plays a major role during pre-application consultation (interviewed UK expert). By 2004 almost half of proposed wind farms were ruled out by the MoD even before they had applied for planning permissions (Toke, 2007). As the DTI would consult the MoD about their concerns, developers prefer to contact the MoD before handing in their application, in order to avoid the project to be turned down. The consultation, however, can turn into an iterative discussion for optimal site choice, which may delay pre-application studies (which are location-dependent) and eventually project submission. This is especially true if other statutory consultees raise objections that are not compatible with the MoD ones. In Britain there is a ban of offshore schemes within a relatively large 74 kilometre radius , while in Germany wind farms are banned within 5 kilometres of air defence radar installations (Toke, 2005b). According to one UK interviewed expert this issue is underestimated in Germany and problems will arise as soon as the first farms become operational. British developers need to complete an EIA (including consultation) before handing in their application, while the German developers can perform an EIA following the standards given by the BSH, but may need to refine their EIA as a consequence of the second application conference, where stakeholder discuss the peculiarities of the site.

There is, in general, a substantial difference in the public opposition to offshore wind farms. In the UK they are seen as a valuable alternative to onshore turbines, whereas in Germany the view is they possibly spoil the horizon and the seaside amenity (despite being planned at least 30 km off the coastline). The German stakeholders have been found to be in favour of offshore wind development, although under some restrictions, while the general public is rather split into two groups, those who are in favour of offshore wind development, and those who are afraid of visual effects (for more details see Licht-Eggert et al., 2007; Gee et al., 2006; Licht-Eggert and Gee, 2006). This means that the general public attitude towards the visual impact of offshore wind farms (much more visible in the UK than they will be in Germany) is different in the two countries and much more in favour of offshore wind in the UK. Nevertheless, offshore wind farms in the UK have encountered objections from various interest groups, including local fisherman, landscape protection interests and the Royal Society for the Protection of Birds (RSPB), which objected ca. 10% of onshore wind farms and one (out of 18) of the Round 1 offshore schemes (Shell Flat in the Irish Sea). As a result of pressures from the RSPB and other interest groups the DTI agreed that in future there will be an ‘exclusion zone’ for offshore wind farms of 8 to 13 kilometres from the shore (Toke, 2005b). According to Toke 2005b a possible legal action of the RSPB in EU courts, would mean that some of the round 2 projects may be subject to considerable planning delays.

To sum up, in both countries selected stakeholders are legally involved in the decision-making procedure for consenting wind farms, and in both countries there are priorities to be respected. Though the UK is a maritime nation dependant on shipping, there is provision for designating exclusion zones for shipping under Section 36, and defence issues are seen as critical. In Germany, on the other hand, shipping is given highest priority, and this may be as a result of the somewhat limited coastline and access by shipping to ports on Germany’s North Sea coast. In Germany, the federal structure of the country allows local and regional authorities to play a major role in consenting works in territorial waters, thus giving local stakeholders the power to delay construction, while in the UK depending on the size of the project, and for offshore starting from 1 MW, jurisdiction for territorial waters can also be passed onto the Secretary of State.

Economic aspects of offshore wind project development

The main actors involved in cost-benefit appraisal of the offshore wind projects are, besides developers, manufacturers and engineering contractors, the stakeholders involved in the electricity market (including generating companies, grid owners and operators, suppliers and consumers). As the differences in the electricity market can affect the offshore business, an overview of the electricity market structure in the two countries is given in the following.

The electricity Market An electricity supply system consists generally of five main parts: (1) generation of electricity (2) transmission –at a high voltage– over large distances (3) distribution

–at a lower voltage– over short distance to end-consumers (4) supply to households (the economic interface of contracting with the end consumers and (5) consumption of electricity. In the following the main similarities and differences in the electricity markets of Germany and the UK will be shortly highlighted.

In both countries the electricity market is dominated by a small number of large and in some cases vertically integrated Generating companies. Namely a share of about 90% of the total power generation is owned by seven large companies in the UK and four in Germany .

In UK only in Scotland is there real vertical integration – and even then since April 1^{st} 2005 transmission is not part of integration. In Germany the transmission network is operated and owned by companies which are owned by the main four generators . Therefore, these companies are able to disrupt the transmission market to their advantage (Lönker, 2007). In the UK, since April 1st 2005 with the introduction of the British Electricity Transmission and Trading Arrangements (BETTA, see (Ofgem/DTI, 2005)), there has been a single Great Britain System Operator (the National Grid Company) for the whole grid of England, Wales, and Scotland, although the three companies, i.e. National Grid, Scottish Power, and Scottish and Southern remain the Transmission Network Operators in their respective historic regional areas. After BETTA, connection to the system and use of the transmission network is transparent and governed by charges which are the same for all generators. A peculiarity of the British system is that in the North of Scotland the connection charges are relatively high, while in the South West of England the charges are negative. Many of the best locations for renewable generation in the UK are in the North of Scotland and the higher connection charges in this area might make the development of renewables in this area less attractive. However, the UK Government instigated a review in late 2005 to consider reduced connection charges for Renewable Generators in this region. In both countries the distribution network companies and the suppliers are either owned by the major companies, or are smaller enterprises (especially in Germany in large number, ca. 900). In both countries there are governmental institutions, which oversee the market, Ofgem (The Office of Gas and Electricity Market), in the UK and the Federal Network Agency, (instituted in 2005) in Germany, ensure non-discriminatory access and efficient use-of-system charges. The consumers in the UK are free to choose their provider based on transparent price comparisons, while in Germany the possible change of supplier is not as trivial and transparency is limited. In the UK transparent price policy is an incentive for suppliers to reduce generation costs, and as they pass renewable energy generation costs on to consumers, also to reduce those costs; in Germany extra costs for renewal energy generation are spread uniformly upon all consumers.

The most striking difference among the market structures in two countries seems to be in the grid operation and ownership. The companies owning the transmission network in Germany are daughter companies of the main generators strongly involved in the nuclear and conventional energy sector. They can defend their market share by hampering the development of renewable energies as well as hindering free competition in the German electricity market (e.g. Lönker, 2006c). In spite of their obligation to prioritise renewable energy and extend the grid for feeding it in, as stated in the German Renewable Source Electricity Act (EEG, 2000/2004), those companies may ignore their obligation. An example is given for E.ON that has being ignoring for years their obligation to extend the grid in Schleswig-Holstein (Lönker, 2005). One German wind farm developer said that the German transmission grid should be property of the state, another one mentioned a strategy of transmission net owners to put forward high connection costs in order to keep free capacity for feeding in the grid with their own conventional power: once the grid capacity is fully used, the network operator cannot feed any new (renewable) electricity into the network. If German large generating companies have been protecting their monopoly through their network ownership, the UK ones protect their interests under the Renewables Obligation through the control over the produced quota of renewable energy, in order to protect their investments (see following sections).

Financial aspects

Due to steel price increases (foundations), copper price increases (cables) and increasing global wind turbine demand (especially driven by the USA market, and emerging interest in wind energy in China) resulting in higher prices, offshore wind is currently proving more expensive than anticipated (DTI, 2006; UK and German experts). In this context, under the current market situation, capital costs of offshore wind projects have increased rather than decreasing during the last years (DTI, 2006). Lönker (2006c) depicts the current situation in Germany as at a 'now or never' crossroad. He mentions that in Germany minimum capital costs for construction are currently estimated around 2000 €/kW, investments for the test field in Borkum are estimated to be around 2900 €/kW. According to DTI 2006 the capital costs for the UK projects range currently up to 1800 £/kW (ca. € 2600). According to those estimations, it is reasonable to assume that similar capital investments would be needed at present in the two countries. However, for UK projects realised in Round 1 the capital investments have been lower (e.g. in the case of Scroby Sands about 1900 €/kW in current prices). Greater distances from German shores can result in connection costs representing a high percentage of required investments, up to 25% of initial capital, as an interviewed developer affirmed. For distances ranging between 30 and 100 km from shore, cable prices of about 500000 €/km quickly add up to high sums. Operation and maintenance (O&M) costs, as well as accessibility for repair may bring about higher costs in the deeper waters of the German EEZ, although tide and wave conditions can also play a significant role in shallow waters (e.g. in the UK at Scroby Sands, only 3 km from shore, wave heights limit access to only 60% of time).

The German offshore farms in deeper waters are more likely to use tripods as foundations. The costs of foundations increases with their depths, in Germany it is possible that foundation-costs make up to 50% of the whole investments. For this reason there is a need to produce greater electricity output per foundations, and the 5 MW technology is planned to be installed (Lönker, 2005), in order to increase the project's rate of return (the produced electricity being a function of the cube of the installed capacity). The 5 MW turbines are still in their test phase (e.g. Repower and Multibrid are being installed in the Beatrice project in Scotland), and neither the manufacturers nor the owners and developers are willing to bear higher failure risks by installing unproven technology (as two interviewed manufactures affirmed). While the 5 MW technology is especially developed for the offshore market and operate in the harsher marine environment, turbine manufacturers tend to resist taking the higher risk of failures in the harsh environment of the smaller turbines and prefer to sell them in the onshore markets in other parts of the world (Lönker 2006b; interviewed experts). On the contrary, the UK developers used for existing Round 1 farms proven technology (e.g. 2-3,5 MW) and in spite of this, there have been noteable failures in the relatively small offshore turbines in the Middelgrunden Array in Copenhagen and

Scroby Sands through lack of attention to protect the transformers and gearboxes respectively for the marine environment. A new legislation piece aiming at speeding up the construction of offshore wind parks in Germany is the Infrastructure Planning Acceleration Act which entered into force at the end of 2006, and which obliges the Network operators to take over the costs for connecting offshore farms to the onshore grid, while connection costs can be spread over the four grid companies (and passed on to the end-consumers at a uniform rate). This implies lower costs for the wind farm developers as the costs associated for the grid connection is transferred from them to the network operators for the projects beginning construction before 2011. Moreover, several offshore wind farms can be connected in a more efficient way minimising the number of single connections to the mainland (BMU 2007).

Supporting schemes Toke (2005a) affirms that the nature of the policy instrumentation can much more influence wind power deployment than the resource base for wind power. Indeed, according to the Risø European Offshore Wind Atlas (Troen and Petersen, 1989) the wind resource in the German EEZ is of the same magnitude as in England and Wales (e.g. at 100 m height, speeds of 8.5-10 m/s), while it is higher in Scotland (speeds >10 m/s). The UK, applies a market-oriented quantity-fix scheme (with tradable certificates), while Germany applies an 'institutionalised' price-fix (feed-in) scheme. Many authors have highlighted the advantages and disadvantages of such mechanisms (Bower, 2003; Wüstenhagen and Bilharz, 2006; Toke, 2007; Tovey, 2005) . Criticism from insiders (developers and experts in each country) affects both financial support schemes, although each scheme does reflect the historic-political culture of each country (Wüstenhagen and Bilharz, 2006) and therefore there is no real willingness to radically change the mechanism, in order not to loose investors' confidence. In the UK, the Renewables Obligation (RO) is dominated by the larger companies in both onshore and offshore schemes. It can be difficult for very small operators to access the financial rewards of the RO scheme as all renewable electricity must be sold through a supplier, and the suppliers are reluctant to deal with small quantities of such electricity. In particular, with regard to offshore wind, the Renewables Obligation facilitates large companies able to finance larger offshore projects out of their balance sheet, and to influence the whole renewable market with their investments strategies. Moreover, subject to the market, developers need to take into account long term fluctuations and instabilities of returns. On the other side, the German feed-in does not allow prediction of how many investors will enter the market and does not seem to offer enough support for beginning construction at the current time: while in the UK the current payment for wind energy can be as high as 13 Ct/kWh when all factors connected with the ROCs are taken into account , the German feed-in tariff is currently 9.1 €Ct/kWh and will start its progressive reduction –by 2% per annum– in 2008 (Lönker, 2005). The year in which the project will start operation will determine the amount of the feed-in tariff, which will be constant for ca. 12 years and decrease after that for another eight years to 6.9 €Ct/kWh, for a total of ca. 20 year-financial support. The former German Government originally applied this progressive decrease in order to motivate investors to quickly start projects. The main companies, as E.On affirmed in 2006, have not felt such offshore wind revenue arrangements to be appealing under the feed-in tariff (Lönker, 2006b). Already in 2003 Kooijman et al. (2003) affirmed that the incentives for offshore wind energy in Germany were insufficient to make it successfully compete with other energy sources in the liberalised market. Only recently the German government undertook two fundamental steps for making the offshore business more attractive: (1) accepted to commit financially in the realisation of the test field in Borkum and (2) promulgated the Infrastructure Planning Acceleration Act with the aim of relieving developers and investors from part of their capital costs. In the UK, the Renewables Obligation has a technology-unspecific character: but in order to support offshore wind the UK-Government has made available capital grants covering up to 10% of Round 1 project costs (BWEA, 2006; Lönker, 2006b; Toke, 2007). Capital grants are also planned for grid connection for Round 2 projects (Toke, 2007). By these means the UK government supports 'selected' electricity technology (i.e. onshore and offshore wind farms, through extra grants) in the belief that they can provide the cheapest electricity (Mitchell and Connor, 2004). Wind power developers need to sell their Renewables Obligation Certificates (ROCs) to electricity suppliers, in return for long term contracts at acceptable prices for their electricity, if they are to attract investment (Toke, 2005b). Bankers want long-term guarantees, not a years' supply of Renewables Obligation Certificates. This puts the control of the renewable energy market into the hands of the large electricity suppliers. There are fears that as the proportion of renewable energy rises towards government targets, major electricity suppliers will pursue their interests by restricting their investments in renewable energy in order to protect the prices that their existing investments receive (Toke, 2005b). The risk involved in the RO is large for developers: (1) Price risk (generators do not know what they will be paid beyond the (short-term) contract); (2) Volume risk (generators do not know if they will be able to sell their generation in the future, certainly once the current 10% target for 2010 is met); (3) Market risk (generation value varies according to market rules) (Mitchell and Connor, 2004). In Germany the game is different, but the winner is the same: the developers are often small and medium enterprises (SMEs) that develop a project and take it through the consent procedure for selling it as 'consented' to larger companies that will realise it. The original developer will, in some cases, aspire to some operational role. However, those SMEs would not be able to realise the construction themselves, due to unavailable starting capital and expertise covering all fields. Again, those companies that can purchase 'consented' projects are the four German electricity market oligopolists. In some cases, the question is whether those big companies purchase the projects in order to realise them or just to be sure that they will not be realised, as an interviewed German developer dared to express. An exception has been the Butendiek wind farm, which was originally born as a 'citizen' wind park. In December 2006 the Irish company, Airtricity, entered the project as a strategic partner for the future planning and construction.

Risk and the role of larger companies

This section looks briefly at the risks related to marine construction, including both ecological and financial risk. Developers are mainly interested in analysing the economic risk of their projects, while it is usually the role of stakeholders and regulators to ensure that ecological risks possibly associated with new uses of the sea areas are taken into consideration. Ecological risk, when leading to additional technical requirements requested by stakeholders and regulators can considerably affect investment costs (e.g. in Germany, the exclusion of parks constructed in protected areas from feed-in schemes is the main aspect that obliges construction in deeper offshore waters). Risk in relation to investments in renewable energy projects can be described by the negative impact which uncertain future events may have on the financial value of a project/investment. Cleijne and Ruijgrok (2004) mention three forms of risk to be taken into account in renewable projects: regulatory risk (change in public support/policy), market and operational risk and technological risk. Technological risks associated with offshore wind parks strongly depend on the new type of turbines (larger than onshore and operating in harsh environmental conditions) and the little experience with the logistics of installations and operation. Especially in Germany the 5 MW technology, which would make investments profitable, is still in its test-phase and therefore connected with high risk. The interviewed experts mention uncertainties about investment subsidies, raw materials and supply chain market (discussed in section 4) as the most relevant barriers for project development, associated with planning risks and affecting project returns; moreover grid-connection feasibility, due to either structure and age of the grid (UK) or power games of grid owning companies and grid extension needs (Germany), may considerably affect investment climate. In principle, any delay represents a cost for the developers and the investors. In general, the economic risk of delayed construction due to weather conditions is spread over different companies under the UK multi-contracting scheme, which replaced the engineering procurement contract, which was judged too risky (EPC, see Rakar (2006), interviewed experts). In Germany there is little or no experience with such large projects and still different kinds of possible contracts, also including both 'turn-key' possibilities, whereby the trend is to move to multi-contracting as in the UK. Under the current renewable support schemes both in the UK and in Germany large generator companies dominate the market, although in different ways. Under the UK Renewables Obligation larger companies protect their investments by ensuring that the Renewables Obligation Certificate market does not crash due to achievement of the set quota. In other words, the obligation to produce renewable energy makes investment profitable as long as the quota is not reached. In Germany the large generator companies, who are the owner and operator (through daughter companies) of the transmission network, are powerful stakeholders that protect their own interest in the conventional energy field, by hampering feed-in of renewables. The German oligopolists represent strong economic interests and have engaged in lobby work towards the government in more than one occasion (e.g. Lönker 2006a; Neue Energie 2007; Nikionok-Ehrlich 2007. Last year Eon declared there would be no possible realisation of offshore wind parks until 2010 and beyond, without an increase of the feed-in tariffs, as future investment in renewables in Germany will only be worth doing in the context of a political support for a broader renewable energy mix not including nuclear energy in the longer run (Lönker, 2006b). The current situation sees Eon and Vattenfall more open with regard to offshore wind (Lönker, 2006a). The two companies will start a pilot project in Borkum co-financed by the German government, where twelve 5 MW turbines will be installed 45 km offshore the Borkum Island, foreseeing a capital investment of about 175 million euros. This is no surprise as those companies own the grid in the coastal zone of the North and Baltic Sea and could take advantage of their situation by prioritising grid connection of their own projects (Lönker, 2006c,a, 2007). Eon currently plans to install 500 MW in the German EEZ. The accusation levelled against the company is that their investments in the offshore wind sector only have the purpose of hindering construction or at least controlling its pace and thereby the effects on the market, in order to protect their own investments in conventional energy (Lönker, 2006a, 2007).

The issue of grid connection is relevant in both countries, although in different ways. In general, electricity from renewable sources, decentralised and intermittent in production, needs adaptation of the grid structure, which has been constructed for the transmission of centrally produced electricity from large conventional power plants. The offshore issues related to the grid are grid extension, renewal (especially in the UK, where the infrastructures are old) and balance of produced/required electricity. In both countries offshore plans to would require many kilometres of new high voltage transmission lines to overcome the dispersed nature of the turbine sites and –especially in the UK where there are North- South constraints ((Hansen and Skinner, 2005; DENA, 2005). Currently, the demand for grid connection is so high in the UK that some projects may need to wait 15 to 20 years to be able to connect. In addition, according to some interviewed UK experts, the existing grid is 'old' and will need updating. To overcome these issues, for example, Airtricity will use the existing connection of the recently decommissioned (December 31st 2006) nuclear power plant Sizewell A for feeding electricity produced in Greater Gabbard offshore wind farm into the grid (Hill, 2006). In Germany, the issue of high grid connection costs has been up to now one of the main aspects affecting project profitability, and leading to delayed beginning of construction. The Infrastructure Planning Acceleration Act obliges the network operators (belonging to the main large generating companies) to take over the costs for connecting offshore farms to the onshore grid. This reduces both costs and uncertainty of connection and implies, for example for Butendiek offshore wind farm, a cost decrease of about 20% and therefore profitability even under the current feed-in tariff of 9.1 cent/kWh (Butendiek, 2006). In this sense, the Infrastructure Planning Acceleration Act reduces the power of large German network operator by obliging them to take over connection costs and planning for the wind parks that will start construction until 2011.

Concluding remarks

This paper has concentrated on some of the issues involved in the construction of offshore wind parks in both countries and it represents a first analysis of the current situation that will need to be deepened in order to describe more subtle interaction mechanisms among stakeholders and governmental institutions. With regard to the main hypothesis expressed at the beginning of the paper we can affirm that:

1. In the UK the Government has been more committed and set a 'more efficient' framework for the development of offshore wind parks, in particular:
   - On average the consent procedure is faster in the UK than in Germany (up to 18 months for notification on behalf of the British DTI vs. up to 3 years on behalf of the German BSH). In Germany spread competences at the federal/local level considerably delay cable consent and thereby beginning of construction. In the UK the streamlining of consent through the DTI allows for clear regulation and responsibilities (shared with the other involved governmental departments, DEFRA, DfT and MoD –as consultee), while in Germany the BSH, which is responsible for both construction and cable permit in the EEZ, should take over responsibility in coordinating the permits and action required ant the local level for the permits in territorial waters. This would, however, require a re-organisation of competencies and the withdrawal of 'consenting power' from some federal and local authorities. However, one aspect is most efficiently dealt with in Germany: the participation process, which calls different stakeholders around a table (Licht-Eggert et al., 2007); on the contrary in the UK is the developer, or the authorising bodies, who need to contact stakeholders one by one, thus requiring long iterative processes.

- In Germany there are two separated consent procedure for cable lines (within and beyond the territorial water limit) causes interest conflicts leading to delays in approving cable routes across the territorial waters. As those cables often need to cross the Wadden Sea National Park, a special permit must be issued from the local authorities (Länder).
- The UK Government acted faster in delineating suitable areas for offshore (end of 2002) whereas in Germany this happened at the end of 2005. The UK concentrated developments in those areas, thus allowing faster EIAs, based on the SEA already carried out. In Germany development in those areas is welcome, but other areas are not excluded, thus giving more freedom of choice to developers, but requiring major effort on behalf of the BSH for approving projects in other areas.
- The UK Government recognised early the need of making capital grants available for supporting offshore projects, while the German government only recently got involved in co-financing the offshore test field in Borkum and in December 2006 through the Infrastructure Planning Acceleration Act, decided to relieve developers from part of their financial burden by obliging the grid companies to take over costs (and planning) of the grid extension up to some offshore area suitable for ‘plug in‘ of the consented wind farms.

2. The offshore wind business has been financially more attractive in the UK than in Germany.

   - Although based on the available data it has not been possible to demonstrate in an unquestioned way that capital investments/MW installed capacity are in the UK lower than in Germany (so that it is reasonable to assume that, in principle, they are not) the capital grants made available by the government and the revenues obtained from the Renewables Obligation mechanism (currently in the UK about 13 €Ct/kWh against 9.1 €Ct/kWh in Germany), have made investments in the UK more attractive than in Germany. In deeper German waters connection and foundation-costs will play an important role in capital investments. Thus making developers require installation of 5MW turbine technologies –not proven yet– in order to enhance the possible revenues. The German Government has recently recognised the need of economic incentives on top of the EEG feed-in tariff and, by promulgating the Infrastructure Planning Acceleration Act, has introduced an extra financial support to planned projects, thus making them profitable (through saved connection costs) even under the current feed-in tariff. Moreover, an update of the EEG is planned in 2008, and possible adjustments to the feed-in tariff can be made based the one-year experience with the Infrastructure Planning Acceleration Act.
   - UK Round 1 projects are in general smaller that German pilot projects (30 turbines vs. 80) and planned in nearshore areas (3 to 9 km from the shore). In contrast, larger German projects to be realised far away from the shore, would have implied higher connection costs and higher foundation-costs, which would have been compensated by the installation of a 5MW-technology. The technological risk associated with installation of unproven technology would have been too high for German developers.

3. Offshore in the UK is a business controlled by larger companies, which have available knowledge and funds to easily go through the consent procedure and the realisation phase.

> Under the Renewables Obligation regime, those companies can protect their investments by taking care that the market does not crash. Although under a certain degree of risk, is their interest to produce as much renewable energy to have a share in the buy out fund but not as much to reach the set quota. In Germany large companies have been opposing offshore wind development in order to protect their investments in conventional energy. The main generating companies, owners of the grid infrastructure, have much economic power and in this sense enforcement of the EEG has been in some cases unsuccessful. According to many experts, the strategy of the two main companies potentially involved in the offshore sector is that of controlling offshore wind development in such a way that their own profit can be maximised. This was meant to hamper development during the last years. Recently, however, those companies started to show more interest in the offshore wind sector.

To conclude, the comparison between the two different countries in the North Sea allowed to highlight how similar targets can result in different legislative frameworks and those in turn can be more or less efficiently implemented based on different historical and political settings, as well as cultural values and power distribution among the involved key stakeholders. The UK government has, through the Round strategy, been able to improve the regulatory regime by trial and error, while Germany has adopted from the very beginning a more precautionary approach. The German approach has, however, resulted in (late-recognised) higher costs and risks put on the developers. In the future it will be possible to verify, if the more lengthy preparation in Germany will also result in an expanding sector.

3.2.2 Pressures: Scenarios for offshore wind farm development

Five scenarios for offshore wind development have been originally assessed in the Coastal Futures project. They describe different world views and priorities that will lead to different degrees of realisation of offshore wind projects in the German Exclusive Economic Zone (EEZ). Those scenarios can be quantified in terms of installed capacity (and therefore required area) in the German EEZ for three time horizons 2010, 2030 and 2055. The five scenarios stem from five broader scenario families which contain a total of 13 scenarios, therefore they are named with a letter (indicating the belonging to the original family, and a letter, distinguishing among scenarios belonging to the same family, for more details see Burkhard, 2006). The five scenarios selected for analysing the consequences of changes in the marine environment are (sorted from no installed capacity to maximum installed capacity):

- the North Sea as recreation area with dominating soft tourism (D1),
- the North sea as shipping area (E1),
- the North sea as natural area with maintenance of the existing state of nature protection (A2),
- the North sea as energy park where the main part of public energy needs are covered by renewable sources (B1), and
- the North sea as industrial area with living and working offshore (C1).

The installed capacity in the time-horizon 2010 range from ca. 930 MW in scenario E1 to ca. 2400 MW throughout the scenarios. Scenario D1 foresees no installed offshore wind parks in the North Sea; E1 foresees a low offshore wind power capacity installed (ca. 2329 MW in 2030 and 15000 MW in 2055); scenario A2 a medium offshore wind power capacity installed (ca. 15000 MW in 2030 and 55000 MW in 2055) and scenario B1 and C1 foreseen both a high offshore wind power capacity installed (25000 MW in 2030 and 90000 MW in 2055). In short, D1 gives priority to tourism and 'natural-looking environment' and does not allow wind turbines to 'spoil' the horizon; E1 gives priority to good-transport and thereby allows a limited construction of offshore wind parks, which is compatible with the enlarged shipping lane-network. Scenario A2 sees the enlargement of protected areas along the German coast, but does not disregard the installation of offshore wind parks. Finally, scenario B1 sees Germany having a leading role in energy production, and scenario C1 foresees the 'industrialisation' of the North Sea (including energy production), whereby resource use is intensive but sustainable through closed production cycles.
The scenarios assess the construction –and expansion– over time of different wind farms, assuming that already consented wind parks will be built in the near future and, according to scenario priorities, other projects (including not even planned farms) will be built in the following years. In the following assessments of the consequences of offshore wind farm construction (sections 3.2.3, 3.2.4, and 3.3) are assessed for the two 'extreme' scenarios E1 ('the North Sea as shipping area') and B1 ('the North Sea as energy park where the main part of public energy needs are covered by renewable sources'), foreseeing respectively a minimum (but higher than zero) and maximum capacity installed. Those scenarios are implemented on a year by year basis: each year construction takes place in different locations until the foreseen capacity has been installed. Table 3.16 shows the construction phases assumed to take place within the scenarios for some selected years, including the name of the constructed wind parks, the number of installed turbines and the boxes where construction takes place. Please note that construction can take place either in box 58, outside it, or in as well as outside it. In the considered scenarios, construction is assumed to begin in 2007: scenarios were assessed during 2004-2005, and at that time this seemed a reasonable assumption.

3.2.3 State/Impact: ecological risk

In the light of political targets for GHGs emissions and renewable energy (see section 3.2.1, page 90), the construction of offshore wind parks represents a chance to supply energy in a more sustainable way. However, it is still not clear which risks may be brought about by the construction of offshore wind parks, if and how they may negatively affect the functionality of marine ecosystems. There are a number of studies that have analysed possible effects of offshore wind park construction and operation upon the marine ecosystem, and especially upon birds, fishes and mammals (e.g. Köller et al., 2006; Erich, 2005; Hoffmann et al., 2000). In this study [9], the ecological risk associated with the construction of offshore wind parks in the German EEZ is analysed. In contrast, this study focuses on ecosystem services and ecosystem functionality rather than on effects upon single species (belonging to the upper levels of the food-web). The

[9]The study reported in this section has been submitted for publication in Gaia as: C. Nunneri, H.-J. Lenhart, B. Burkhard and W. Windhorst, 'Ecological Risk as a tool for evaluating the effects of offshore wind power installations in the North Sea'.

Table 3.16: Construction of wind parks in the two considered scenarios E1 and B1. For each scenario the realised projects and the interested ERSEM boxes (in brackets the deeper boxes), as well as the number of installed turbines is reported.

Year	Scenario	Construction of	ERSEM-Box	Turbines (nr.)
2007	E1	Borkum West and Borkum Riffgrund West	67	92
	B1	Borkum West, Borkum Riffgrund West and Butendiek	58, 67	172
2008	E1	Butendiek, Borkum Riffgrund	58, 67	115
	B1	Sandbank 24, Borkum Riffgrund and Amrumbank West	47(134), 57(138), 67, 68	237
2011	E1	Sandbank24, Borkum Riffgrund	47(134), 57(138), 67	122
	B1	Dan Tysk, Nordergruende, Borkum West, Borkum Riffgrund West, Borkum Riffgrund	58, 67, 78	700
2012	E1	Amrumbank West	68	80
	B1	/	/	/
2013	E1	/	/	/
	B1	Noerdlicher Grund, Sandbank 24, Nordsee Ost, North Sea Windpower	47(134), 57(138), 67, 68	1281
2015	E1	Nordsee Ost and North Sea Windpower	67, 68	128
	B1	Offshore Helgoland, Riffgatt, Dan Tysk	58, 68, 76	304

goal of of the analysis is twofold: it aims to test (1) the applicability of the overall ecological-risk methodology and (2) the suitability of the selected parameters for integrity and risk indication.

In this study the focus is set first on one selected area in the North Sea, namely box 58 of the ERSEM model (figure 3.29). This means that we assess the impacts deriving from construction in different areas in the North Sea for box 58 only. The box volume is 98.37 km^3; the box area 4257 km^2, the mean water depth 23 m. We chose box 58 as in this area two of the planned and already consented wind parks in the German exclusive economic zone (EEZ) are located, namely Butendiek and Dan Tysk. This allows distinguishing direct effects of construction (i.e. local effect in the area where construction takes place) from indirect effects (i.e. effects of construction in adjacent areas). The modelled scenarios imply construction phases –and therefore increases of SPM concentration –either in box 58 or in other boxes around it and further away from it as well as both in box 58 and around it (details in section 3.2.2, see table 3.16). The selected integrity indicators assess the consequences of wind park construction in terms of ecosystem changes in the water column and the sediments. Only effects upon the provision of supporting services (e.g. limited primary production due to increased turbidity) have been considered so far.
The extension, location and construction in time of offshore wind parks in the EEZ is assessed through scenarios, which are based on different use-strategies for both coastal and marine areas. The effects of the construction of offshore wind parks upon the ecosystem (disturbance) was simulated with ERSEM. Figure 3.30 shows the role of ecosystem modelling in the analysis. The key input parameter for assessing the changes in the ecosystem is the increased SPM concentration associated with offshore wind park construction in the German EEZ. Being the higher SPM concentration in the marine water related to the construction phase of the offshore wind parks, this analysis is restricted to short-term changes. Other indication parameters and models can be

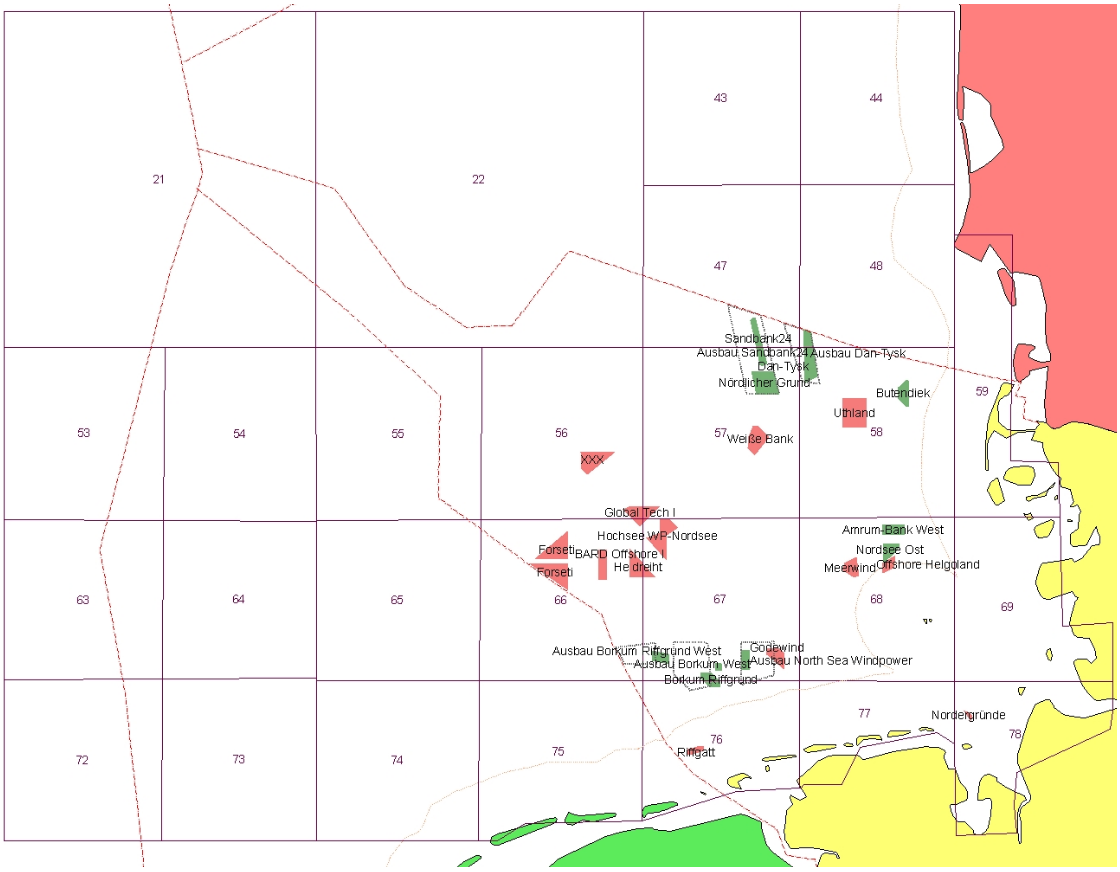


Figure 3.29: ERSEM Boxes and planned wind farms in the German exclusive economic zone (EEZ) (source: Burkhard, ÖZK Kiel).

used for assessing long-term impacts, but this goes beyond the scoping nature of this paper.

Modelled scenarios: enhanced suspended particulate matter (SPM) during construction

In brief, the considered scenarios (details in section 3.2.2, table 3.16, page 107) range between two extremes, represented by scenario E1 and scenario B1.
These scenarios are implemented on a year by year basis: each year construction takes place in different locations until the foreseen capacity has been installed. Table 3.16 (page 107) shows the construction phases assumed to take place within the scenarios for some selected years, including the name of the constructed wind parks, the number of installed turbines and the boxes were construction takes place. Please note that construction can take place either in box 58, outside it, or in as well as outside it.
The scenarios enter the ERSEM model in the form of increased suspended particulate matter (SPM) concentration as a result of the construction phase. In the natural background concentration for SPM a strong gradient occurs, reaching from 0.5 mg/l in the central North Sea up to 5 mg/l in the coastal areas (Puls and Sündermann, 1990; Pohlmann and Puls, 1994). In addition

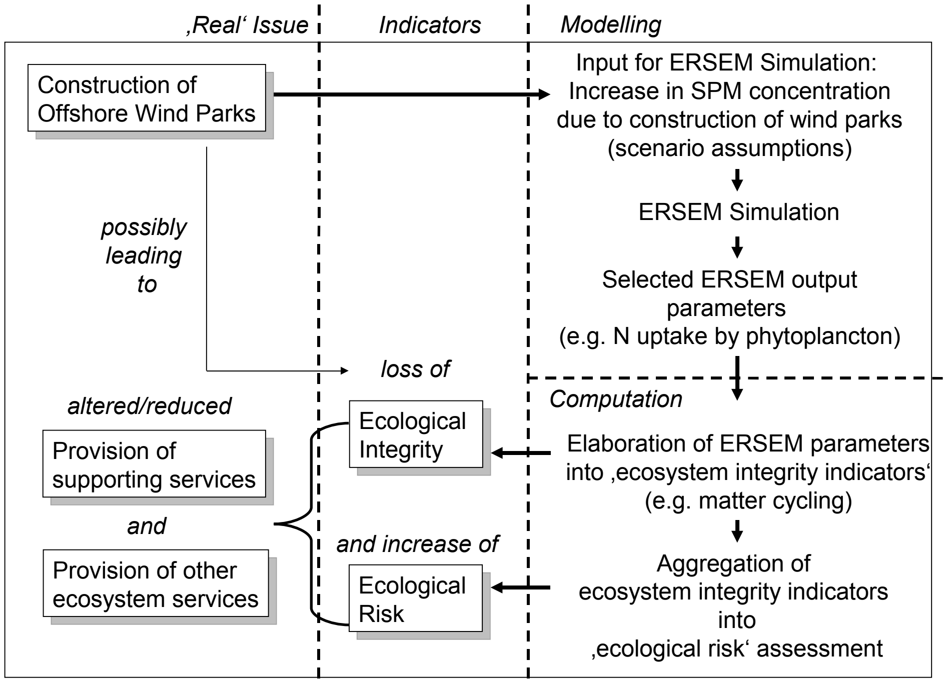


Figure 3.30: Role of modelling for assessing ecosystem functioning by selected indicators. Observable issues (left) are translated into scenarios and enter the model in the form of assumptions (e.g. increased SPM concentrations). The ERSEM simulation (right upper side)) provides output parameters that can be used for computing integrity indicators (right lower side) and ultimately assessing ecological integrity and ecological risk (middle).

strong seasonal variations occur, with the highest values in winter because of a strong storm-induced resuspension of SPM from the bottom into the water column. The mean yearly value of SPM in box 58 is about 3 mg/l. This relatively high value are due to its position near the coast. In realistic conditions an increased SPM concentration appears only in the vicinity of the pile. Nevertheless, in order to show the general effect of increased SPM concentration on the ecosystem, an increase in SPM level of 2 g/m^3 (used as a threshold level in a study by the Danish Hydrographical Institute, DHI, 1999) on top of natural background SPM concentrations is assumed during construction phases. The higher concentration is applied for the whole ERSEM box in which the wind park is constructed, independently of the dimension of the offshore wind park, i.e. the number of installation is not related to the SPM increase.

Analysis and ecological risk assessment

The integrity indicators derived from ERSEM parameters are normalised following the procedure described in section 2.3. In terms of non-normalised indicator values, it is possible, in some cases, to relate the modelled results to measurements or available ‘average’ values.
The yearly primary production for the German Bight exhibit high values of 261 g C m^{-2} y^{-1}

for the years 1988/89 from measurements (Joint and Pomroy, 1993) and 240 g C m^{-2} y^{-1} from a synthesis work (van Beusekom and Diehl-Christiansen, 1994). The comparable model results from ERSEM range between $247^{-2}64$ g C m^{-2} y^{-1} (Lenhart, 1999), while the value for the standard simulation for box 58 is 303 g C m^{-2} y^{-1}. This relatively high value is nevertheless within the variability of simulated primary production values for the German Bight, which range from 197 to 340 g C m^{-2} y^{-1} (ASMO, 1997). The value of this indicator ranges in the scenario simulation between a minimum of 14.8 g C m^{-2} y^{-1} in the extreme assumption of a 12-month construction in box 58 (thus reducing light penetration and thereby primary production) to a maximum of 385 g C m^{-2} y^{-1}, occurring in scenario B1 for the year 2013 in which construction occurs in a number of boxes surrounding box 58 (namely boxes 47+134, 57+138, 67 and 68) and therefore the nutrients that are not utilised in these boxes are transported into box 58.

The calculated turnover in the standard run is 5.8 for box 58 and 6.8 for the German Bight. This is rather low in comparison to the Dutch coastal region with turnover rates around 14 (Lenhart, 1999; for a more detailed analysis about the Dutch Wadden Sea see Philippart et al., 2007). Indicator values range in the scenarios between a minimum of 0.2 in the extreme assumption of a 12-month construction and a maximum of 6.8 for box 58, the highest value being found in the case of scenario B1, in year 2013 (as above).

The diatom/non diatom ratio for box 58 is 0.37, which is on the lower end in comparison with simulated diatom/non-diatom ratios which range from 0.33 up to 0.86 for the German Bight (ASMO, 1997). In the frame of the eutrophication problem, an increase of diatoms would be brought about by reduction measures, thus making possible to connect higher ration with lower degree of eutrophication. This tendency could be backed up by the reduction scenarios by different ecosystem models (ASMO, 1997). In the case of offshore wind, the natural diatom and flagellate bloom succession is extremely altered due to highly reduced light penetration (high SPM concentration) during construction. In addition, the nutrients which are not utilised during the spring bloom or during the summer period are aggregated in the water column and lay the basis for higher primary production –in comparison to the standard run– after the construction time is over. In this way the natural seasonality is distorted. There is no clear pattern left between diatoms and non-diatoms. The instantaneous local condition of light and nutrients determine the proportion in the production between both algae group.

For the indicators ‘storage’ and ‘matter balance’ there are no measurements available that can easily be compared with the modelled data. Especially the transport of organic matter can only be aggregated in a consistent manner from ecosystem models. For inorganic nutrient budgets assumptions can be made for the transport in and out of the German Bight, when combining nutrient measurements with advective transport rates. However, the poor information on the organic sediment input and how much of this is released again in inorganic form, forces to adopt the fluxes for this pelagic-sediment interface as a closure for the budget (Beddig et al., 1997). Matter balance from the transport of inorganic and organic material values range between a minimum of -33.8 in the case of 12-month construction and a maximum of 9.1 for scenario B1 in year 2008 (construction in boxes 47+134, 57+138, 67 and 68, neighbouring box 58). In the first case a high amount of the nutrients in box 58 could not be utilised because of light limitation and therefore is transported out of the box. In scenario B1 in 2008 these nutrients that are not utilised in the box or boxes where construction occurs enter box 58, thus increasing import of inorganic nutrients into the box. At the same time the import of organic material is reduced, nevertheless leading to a small positive balance between the transport in and out of the box. Storage values in box 58 vary from a minimum of -26.0 in the case of 12-month construction to 13.9 in year

2012 for scenario B1 with no construction. In the first case there is hardly any production of organic material because of light limitation, resulting in a poor input of organic material in the sediment; the sediment releases previously accumulated material, and the storage indicator value is therefore negative. In the case of no construction the usual seasonal blooming cycle is kept, where organic material stored in the sediment during the summer is slowly remineralised and released as inorganic nutrients towards the end of the year (to achieve the high winter nutrient concentrations). When construction takes place in box 58, the input of organic material into the sediment takes place later in the year; this results in delayed re-mineralisation of part of the stored material.

As the changes in ecosystem integrity are similar for the two analysed scenarios, in the following the effects of construction on box 58 are presented for scenario B1 only, as in this scenario the highest capacity is installed in the North Sea EEZ (high offshore wind park construction activity). In figure 3.31, the processes of ecosystem integrity are represented through the normalised integrity indicator values in a relative scale ranging from 0 to 100. This representation, in a radar diagram (the so-called amoeba), allows recognising at a glance changes between a maximum and a minimum value assumed in the modelled scenarios (and used as extremes for the normalisation).

In particular, spatial occurrence of construction phases (inside box 58 or in other boxes) considerably affects the impacts on the selected area (box 58): direct effects, i.e. the cases in which construction takes place in box 58, are different from indirect effects, i.e. the cases in which construction takes place outside box 58. Construction in box 58 takes place in the years 2007, 2011 and 2015. In those years all indicators decrease with respect to the reference year 2006, during which no construction takes place (either in box 58 or elsewhere). Primary production in box 58 is light-limited due to increased turbidity. The turnover of winter nutrients is low, due to low phytoplankton DIN uptake; N not taken up from phytoplankton for photosynthesis is transported outside box 58, which results in low matter balance (i.e. high transport of –principally inorganic– N out of the box). When construction takes place in neighbouring boxes, i.e. outside box 58, as it is the case in years 2008, 2009 and 2013, high indicator values for all integrity processes can be observed. In particular: high primary production in box 58 takes place in this case not only because turbidity does not negatively affect light penetration, but also because inorganic nutrients cannot be used in the neighbouring areas, where construction takes place (due to light limitation brought about by high turbidity) and are transported into box 58, where they can be taken up. This is also shown by high values of the integrity indicator matter balance (high transport into box 58). Turnover (defined as phytoplankton uptake divided by winter DIN) is high, due to high phytoplankton uptake for photosynthesis. A special case is that of heterogeneity values, which are quite high throughout the years, independently of where construction takes place (within or outside box 58). The diatom non diatom ratio used for indicating heterogeneity stays in a relative constant range, thus highlighting that this indicator is not sensitive to changes in the case of offshore wind construction (for the reasons explained above).

When considering the aggregated Elbe box (fig. 3.32), the behaviour of the indicators is determined by the fact that construction of wind farms is assumed to take place at different time in different ERSEM boxes and also foresees parallel construction in more than one box.

A pattern, which can clearly be seen, is the reduction of primary production in years where construction takes place in a box within the aggregated Elbe box, thus resulting in increased SPM concentrations within the Elbe box. This takes place in the years 2007 (box 58), 2008 (box 68),

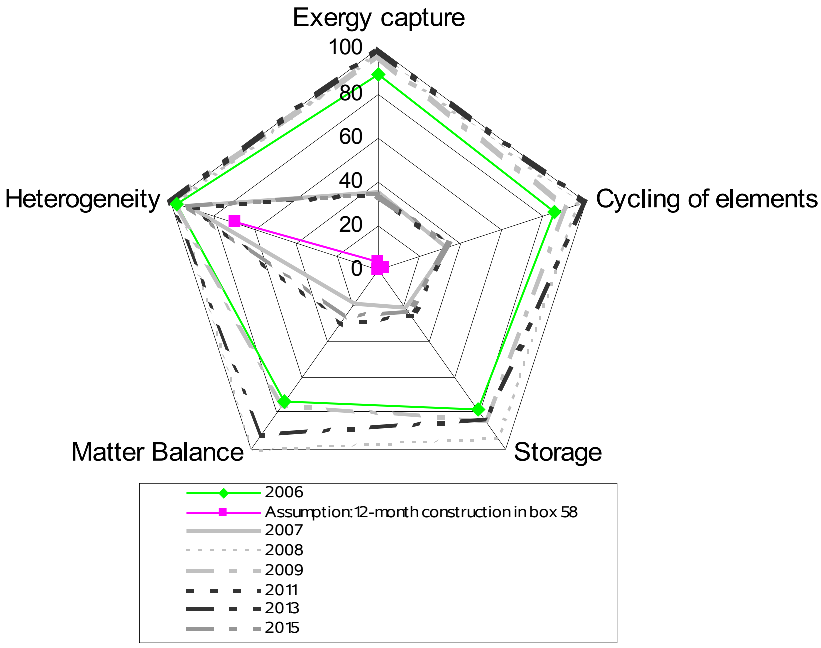


Figure 3.31: Integrity indicator changes in ERSEM box 58 under scenario B1. The normalised integrity indicator values are shown for selected years. Lowest indicator values are in the middle of the graph (normalised values closer to 0), while highest values at the border (normalised values closer to 100). Direct and indirect effects on box 58 can be distinguished (for explanation see text).

2011 (box 58 and 78), 2013 (box 68) and 2015 (box 58 and 68). In those years, according to a reduced primary production, also other related fluxes are reduced, e.g. organic input to the sediment, in similarity to the findings within eutrophication scenarios. In particular the indicator values in year 2015 are very similar to those of the maximum disturbance, defined as a 12-month construction occurring in box 58. This occurs because boxes 58 and 68 together cover about 65% of the volume of the considered area, thus imposing a light limitation to a consistent part of the aggregated Elbe box.

In the case of offshore wind, assessing the risk against some maximum and minimum reference conditions has proven a complex task. In the following we present an assessment of ecological risk in box 58, based on the assumption that high integrity indicator values are associated with low environmental risk and vice versa. As both direct effects (i.e. enhanced light limitation of primary production caused by construction in box 58) and indirect effects (additional input of nutrients and organic matter from adjacent boxes in the case of construction taking place in adjacent boxes) play a role, indicator values are not necessarily 'minima' in the 'undisturbed situation' and 'maxima' in the 12-month construction in box 58. In this sense, changes of integrity indicator values towards 100 (the maximum normalised value assumed by integrity indicators)

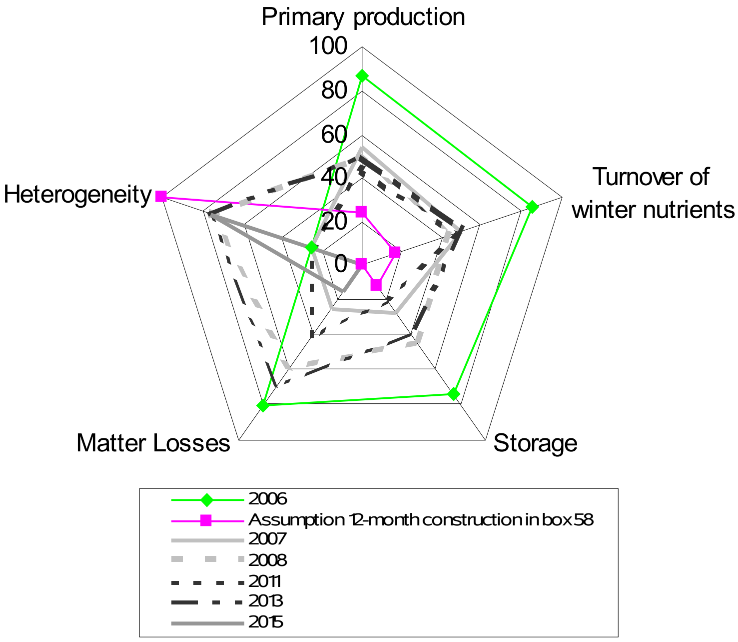


Figure 3.32: Integrity indicator values for the aggregated Elbe-box. The indicator values for selected years in which offshore wind park construction takes place in scenario B1 are shown.

are assumed to lead to lower risk, while changes towards 0 (the minimum normalised value assumed by integrity indicators) are considered as increased risk.
Figure 3.33 shows the ecological risk in box 58 associated with the two considered construction scenarios. A risk increase corresponds to direct effects associated with construction in box 58 (e.g. years 2007, 2011, 2015 for scenario B1), while risk decrease corresponds to indirect effects due to construction in neighbouring boxes (e.g. years 2008, 2009, 2013 for scenario B1). Nevertheless, the selected area is not separated from the adjacent waters and it would be quite simplistic to accept such a risk evaluation. The effects of construction are compensated every year; this means that, in average, the system recovers after construction ceases, thus leading to negligible average long-term effects.

In figure 3.34 the ecological risk assessment for the offshore wind construction scenarios in the aggregated Elbe box is reported up to 2015. It is clearly to be seen that risk arises in the years where construction takes place, while it is zero in other years. The maximum risk, beside the one associated with the 12-month construction in box 58, is associated with scenario B1 for the year 2015, due to construction in a large area in the Elbe box. For scenario E1 the maximum risk occurs in year 2012, and is associated with construction in box 68 only. By comparing the risks analysed for areas with different extension, it can be seen that enlarging the considered area may result in compensated or enhanced effects of construction, thus highlighting the need

Figure 3.33: Ecological risk for wind farm scenarios in box 58. Construction years present an increase in risk, with respect to the actual situation (2006) with no construction.

of considering the whole of the interested ares (in this case the German EEZ).

Focusing the spatial resolution on box 58, has allowed distinguishing direct and indirect effects of construction upon a selected area, although the main findings are not extendable to the whole German exclusive economic zone. In other words, as it has been seen by examining the effects of construction upon the aggregated Elbe box, effects can be 'compensated' by neighbouring boxes, thus leading to low ecological risk associated to offshore wind construction, as long as the affected area is not a considerable percentage of the considered area. In further studies, the effects upon a larger area, including either the German EEZ or the southern North Sea could be tested.

3.2.4 Response: cost-benefit analysis

Cost-benefit analysis (CBA) evaluates alternative options (e.g. policies, projects or scenarios) based on the consequences assessed in terms of gain and loss in economic welfare for society as a whole. A detailed explanation of CBA is given in Pearce et al. (2006). Briefly, within a CBA, the economic appraisal of some alternative is realised by measuring costs and benefits in monetary terms, whereby the challenge is both the identification of different cost and benefit items and their monetary assessment. Within a CBA the net present value (NPV) of an action is assessed , i.e. the sum of costs and benefits, discounted when they take place in the future (see

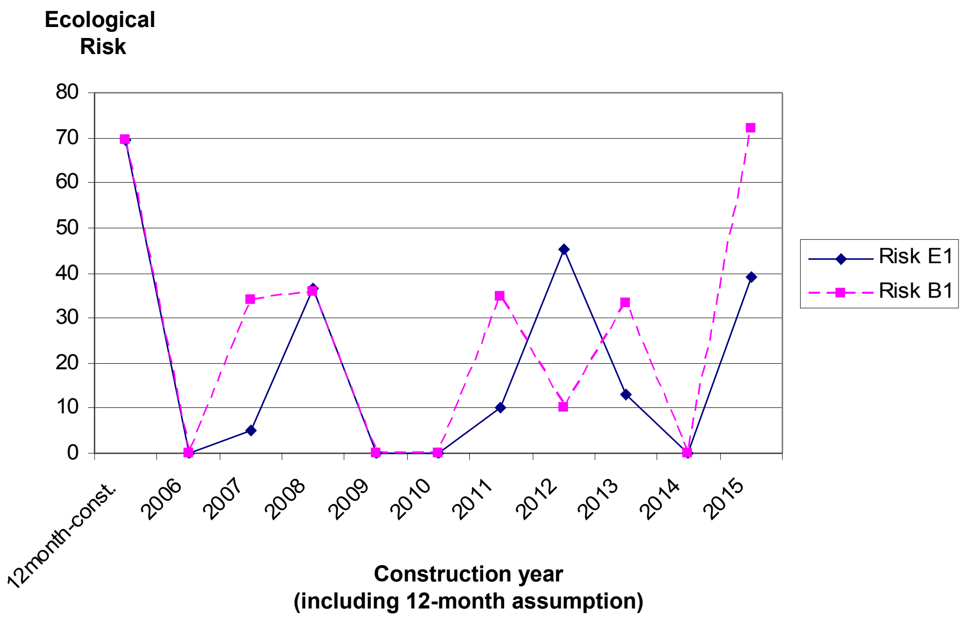


Figure 3.34: Ecological risk calculated with respect to minimum and maximum values of indicators throughout the offshore wind construction scenarios. Construction years present an increase in risk, with respect to the actual situation (2006) with no construction.

Pearce et al., 2006, page 183 and following), over the life-time of the considered project. As a decision-rule, an action will be accepted if its NPV is positive, i.e. total benefits outweigh its total costs. However, from the perspective of single actors, a project valuable for society as a whole might imply net losses, thus requiring compensation mechanisms.
In the following, CBA is applied to the case study offshore wind in two different contexts, for comparing:

a) two scenarios for offshore wind installations in terms of socio-economic costs and benefits (for the time-frame 2007-2055) and

b) a coal-fired power plant and an offshore wind farm with the same nominal capacity, 3000 MW, assuming they will have both a 28-year life-time (2007-2035).

Within the CBA those alternatives are compared to each other in terms of their NPV for society as a whole and not from the perspective of (e.g.) wind farm planners or investors.

Assessment of costs and benefits In considering power generation projects there are different types of costs and benefits to be taken into account:

- the initial costs associated with the construction of the plants or turbine installations (capital investments),
- the costs of operating and maintaining the power plants (O&M costs),
- the costs of coal in the case of a coal-fired power plant,
- the costs associated with greenhouse gas emissions (in the case of a coal-fired plant) or, alternatively, the benefits associated with emission reduction[10],
- the benefits of electricity generation (measurable in terms of electricity market price).

Decommissioning costs are neglected in the analysis[11]. Table 3.17 gives an overview of the used parameters. All costs and benefits are in 2006 prices. A constant discount rate of 3.5% is used for costs and benefits that will occur in the future (i.e. all running costs and yearly benefits associated with electricity production, as well as capital investments of future offshore installations). For offshore wind farms, the minimum capital investments are assessed based on Lönker (2006c), similar estimates are reported by Jeske and von Hirschhausen (2005), while the maximum are derived from the investments planned for the offshore test field in Borkum (Stiftung Offshore Windenergie, 2006); the minimum and maximum capital investment costs for a coal-fired power plant are assessed by Parkinson, 2001[12].
Operation and maintenance costs are assessed based on Parkinson (2001) for both power generation kinds. The minimum and maximum costs related to coal firing (capital costs and fuel costs) are reported by Heinzow et al. (2005) and Neumann (2007) respectively.
The marginal benefits (in NPV) associated with reduced CO_2 emissions are assessed by Tol (2005) as the average estimate among selected peer-reviewed studies. The author mentions, however, a much higher average estimate when including grey-literature (ca. € 67/t CO_2), ca. € 11.5/t CO_2 when considering only peer-reviewed studies with a pure 'rate to time preference' of 3% and about € 36/t CO_2 (ca. 51 \$) for studies with a 1% 'pure rate to time preference'. The average value throughout all considered peer-reviewed studies (€ 36/t CO_2) will be used for the CBA, although the influence of lower estimates will be tested in a sensitivity analysis. The emission saved are assessed about 0.8 t CO_2/MWh (WindGuard, 2004), but they can range up to 1 t CO_2/MWh, (Tovey, pers. comm.; Parkinson, 2001). According to Hirschberg and Dones (2004) capacity factors of, respectively, 0.43 and 0.47 are used for wind and coal generation. This means that the plants are assumed to produce electricity at the nominal capacity for, respectively, 43% and 47% of the time. For assessing the benefits associated with electricity production the assumption will be that electricity is sold at the current market price in Germany (55 €/MWh, Kiel Stadtwerke, pers. comm.).

Offshore wind scenario comparison For each of the two analysed scenario, namely B1 and E1, wind farm development has been assessed in terms of location and installed capacity for each year. The assumption is that a farm will be operational on the year following the construction

[10]Please note that in the comparison of the two different power- generation alternatives either the costs of emissions or the benefits of emission reduction should be considered in order to avoid double counting.

[11]In the case of offshore wind, towers are assumed to be repowered with new generators at the end of their life-time.

[12]Those costs assessed in 2001£have been transformed in Euros by applying a conversion rate of Eur/GBP=0.62 as suggested by the German Bank Sparkasse, while the inflation rates have been assessed by Apollo Real Estate, NAI Apollo, at: 1.3 (2002), 1 (2003), 1.8 (2004), 1.9 (2005) and 1.7 (2006): an average inflation rate of 1.5 has been used for computing 2006 prices).

Table 3.17: Assessment of the main variables included in the cost-benefit analysis.

Technology	**Costs**			**Benefits**	
	Capital inv.	**O&M costs**	**Fuel costs**	**Electricity revenues**	**Saved CO_2 emissions**
	(k€/MW)	(€/MWh)	(€/MWh)	(€/MWh)	(€/tCO_2)
Offshore Wind	2000-2916	13.9-17.4	–	55[a]	36 [b]
Coal-fired power	1480-1741	12.2	4.4-10	55	–

[a]The difference between the market price of 55€/MWh and the feed-in tariff of 91€/MWh is considered in the sensitivity analysis as an extra cost, see text.
[b]This value is already a 'present value', i.e. discounted over time, will therefore not be discounted in the CBA.

year (see section 3.2.2, table 3.16). In a first approximation, the feed-in tariff will be neglected, while in a second time the feed-in will be considered as an extra-cost for those projects beginning operation within 2010 only[13]. The CBA has been carried out for different combinations of the values assessed in table 3.16. The results are the following:

1. assuming maximum capital and O&M costs scenario B1 would have in the long term a higher NPV than scenario E1 (€ 19 billions vs. € 3 billions), both scenarios would show a positive NPV towards the end of the considered period (see fig. 3.35, table 3.18).
2. assuming minimum capital and O&M costs scenario B1 will have a positive NPV since 2032, while scenario E1 from 2042; in 2055 scenario B1 shows a considerably higher NPV than scenario B1 (€ 71 billions vs. € 13 billions).

At some point in time, if scenario construction would stop (see figure 3.35), either the one scenario or the other would have a positive NPV; in both cases scenario E1 would have in the long run a NPV inferior than scenario B1.

Those results hold also including the extra cost associated with the feed-in tariff. The maximum feed-in tariff for offshore wind (91 €/MWh) represents, from the perspective of society, a cost (ca. 36 €/MWh) on top of the market price (ca. 55 €/MWh). However, being in both scenarios the capacity installed by 2010 a minimum percentage of the total installed capacity (3% in E1 and 6% in B1), the inclusion of the feed-in tariff, does not considerably affect the end-results of the CBA (see table 3.18). Much more decisive effects are shown when decreasing the benefits associated with CO_2 emission reduction to the lowest average values assessed by Tol (2005). In this case, for maximum costs scenario E1 will have higher NPV that scenario B1. Nevertheless, both scenarios will be ineffective under this condition, as both will show negative NPV values (see table 3.18). When considering maximum costs, a minimum value of € 29/t CO_2 is required for scenario B1 to reach positive values (scenario E1 would still show a negative NPV).

Under the considered conditions, it can be said that scenario B1 would be preferable to scenario E1 within the considered time-frame. In the examined cases, E1 has a NPV comparable to B1 only when considering no feed-in and minimum costs as well as benefits associated with CO_2

[13]The German feed-in law foresees that offshore wind farms operating before 2010 will receive a fixed price for the produced electricity (feed-in tariff) for a maximum of 20-years. For farms starting operation in the German EEZ, the feed-in tariff depends on the year in which they go on-line and on numerous other factors (EEG, §7). The assumption for the sensitivity analysis is that only the projects constructed until 2010 will sell their electricity at the maximum feed-in tariff value (9.1 €Ct/kWh) for 20 years.

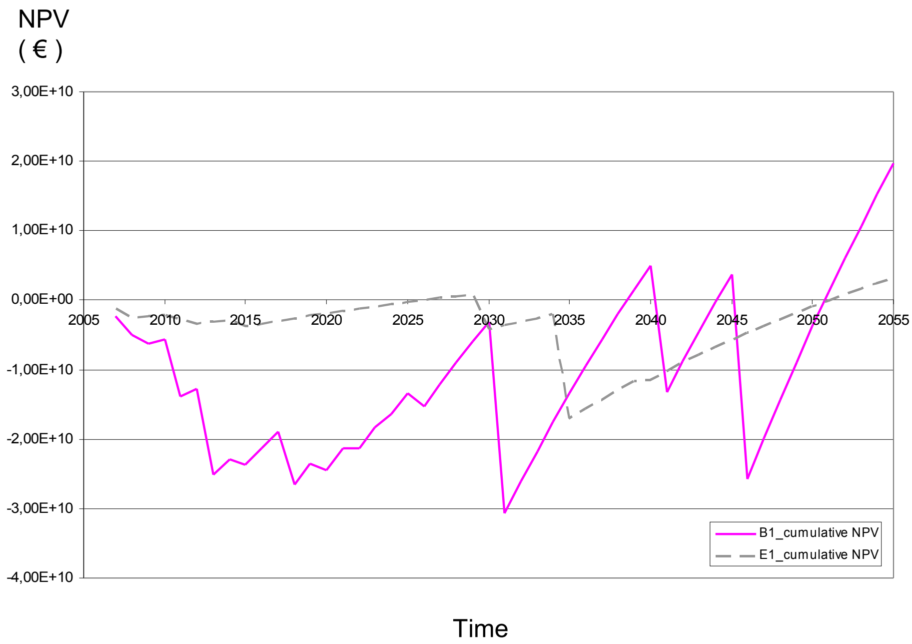


Figure 3.35: Net present value (NPV) of the considered scenarios B1 –maximum installed capacity– and E1 –minimum installed capacity. Assumptions: maximum construction and O&M costs. Please note that a positive NPV in a given year means that benefits up to the considered year outweigh costs in the same time-frame. Step by step construction (i.e. high capital investments) is the cause of NPV negative peaks.

Table 3.18: CBA-sensitivity to CO_2 reduction benefits and feed-in tariff for offshore wind scenarios.

benefits €$/tCO_2$	feed-in	costs	NPV_{E1} in 2055 (€)	NPV_{B1} in 2055 (€)	$NPV_{E1} > 0$ (year)	$NPV_{B1} > 0$ (year)
36	yes	min	$1.2 \cdot 10^{10}$	$6.7 \cdot 10^{10}$	2043	2033
		max	$1.5 \cdot 10^{09}$	$1.5 \cdot 10^{10}$	2054	2052
	no	min	$1.3 \cdot 10^{10}$	$7.1 \cdot 10^{10}$	2042	2032
		max	$3 \cdot 10^{09}$	$1.9 \cdot 10^{10}$	2052	2051
11.5	yes	min	$2.8 \cdot 10^{09}$	$2.1 \cdot 10^{10}$	2051	2050
		max	$7.3 \cdot 10^{09}$	$-3.1 \cdot 10^{10}$	–	–
	no	min	$2.1 \cdot 10^{10}$	$2.5 \cdot 10^{10}$	2049	2049
		max	$-5.8 \cdot 10^{09}$	$-2.7 \cdot 10^{10}$	–	–

reduction.
However, when appraising the effects of offshore wind farm construction together with the connected risks, as expressed in section 3.2.3, a trade-off between risks and benefits has to be carried out. Under uncertainty, and although modelling has been carried out only for selected years, it is reasonable to affirm that scenario E1 presents less ecological risk (lower risk and more spread in time) than scenario B1. If this risk could be quantified in terms of expressed preferences or willingness to pay, it would be possible to factor this aspect into a multi criteria analysis. This would allow to express trade-offs between the considered scenarios when including also potential ecological risks associated with new developments.

The assessed scenarios, although realistic in the sense that they foresee a step-by-step offshore wind development, do not offer a linear overview of the NPV of a single farm. For this reason the following CBA, which compares the NPV of wind and if coal-fired power generation, has been performed.

Offshore wind vs. coal-fired power In this section a CBA is used for comparing two different alternatives for electricity generation. Two possibilities of producing electricity are appraised in terms of their NPV: a coal-fired power plant and an offshore wind farm. The assumption underlying this study is that the two power plants have the same nominal capacity (3000 MW), are constructed in 2007 and will be operational from 2008. An equal life-time of 28 years is assumed for both projects. The choice of the installed capacity is based on the original goal set by the BMU (2002b) for installed offshore wind capacity by the year 2010.
In the considered time-frame, using the minimum and maximum estimates reported in table 3.16, the two projects have been evaluated for three cases:

1. assuming for both projects minimum cost-estimates; the generation of offshore wind has a positive NPV already in 2016, while coal-generation in 2019, whereas since 2013 offshore wind NPV is higher than coal NPV. The NPV at the end of the considered life-time is respectively, € 11 billions for the offshore wind project and € 4 billions for the coal-fired power station;
2. assuming maximum estimates for both projects; the generation of offshore wind has a positive NPV from 2021, while coal-generation from 2025. The NPV at the end of the considered life-time is respectively, € 8 billions for the offshore wind project and € 2 billions for the coal-fired power station;
3. assuming minimum estimates for coal-firing and maximum for offshore wind; the offshore wind project has a positive NPV from 2021, while coal-generation from 2019. Wind would be more effective than coal-power from 2023. The NPV at the end of the considered life-time is respectively, € 8 billions for the offshore wind project and € 4 billions for the coal-fired power station (see figure 3.36).

By including the extra costs associated with the feed-in tariff for offshore wind generation during the first 20 years of operation, the two alternatives would be much closer in terms of NPV (although wind would have a slightly higher NPV) except for the case in which minimum costs are assumed by coal generation and maximum by wind, in which coal-firing would be more valuable than wind generation (€ 4 billions vs. € 2 billions, see table 3.19).

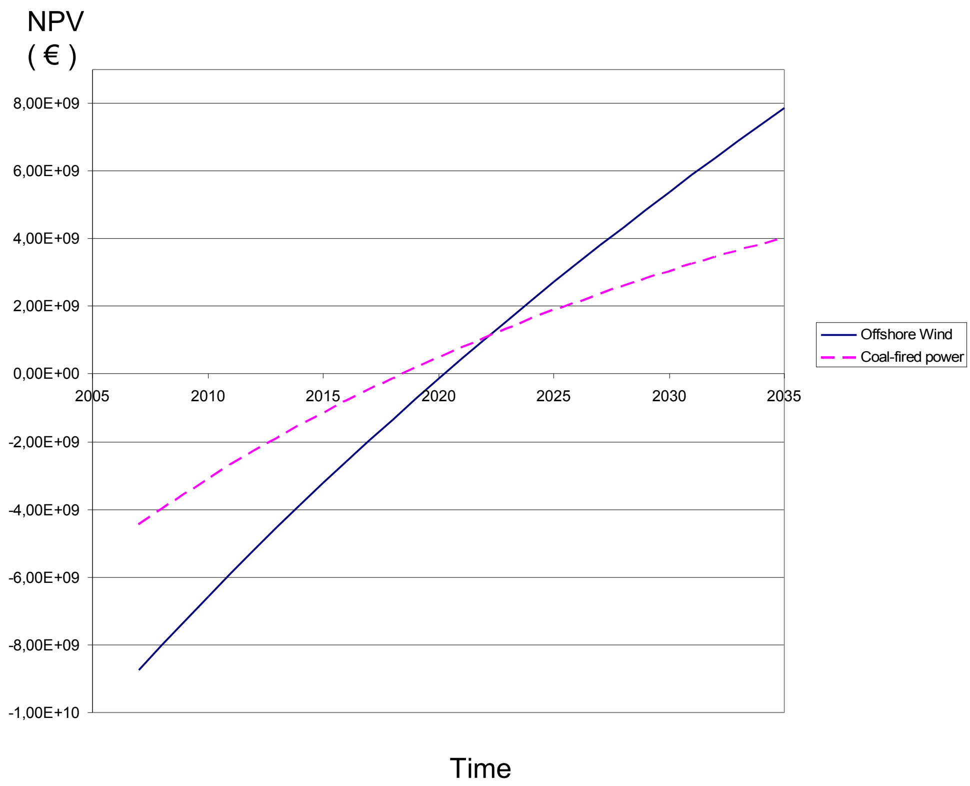


Figure 3.36: Net present value of a coal-firing power plant and a offshore wind farm with the same nominal capacity and the same life-time, assuming minimum costs for coal-fired power and maximum for offshore wind among those reported in table 3.17.

As the benefits assessed for CO_2 emission reduction play a considerable role in determining the net-present value of the projects, and Tol (2005) reports a large variability of estimates, further calculations throughout the three considered constellations have been carried out in order to determine the influence of estimated CO_2 reduction benefits. The results are reported in table 3.19.

In the case not considering the feed-in extra costs a minimum benefit value associated with emission reduction of € 22 /t CO_2 is required for wind generation to show higher NPV than coal-firing generation. Under some sub-constellation of the case 'no feed-in' benefits can be considerably lower (e.g. € 8 /t CO_2 under the assumption foreseeing minimum costs). Including the feed-in tariff would bring the minimum required benefit estimates up to € 45/t CO_2. Benefits of CO_2 reduction lower that those mentioned would lead to coal-fired power to be more effective than offshore wind generation (within the considered range of values). As the consequences of climate change are uncertain and estimates of benefits deriving from emission reduction vary considerably in the existing literature (see Tol, 2005), evaluation of renewable energy projects in the light of their contribution to climate change mitigation will largely depend on the value society attributes to climate change risks and prevention. However, if benefits would be assessed

Table 3.19: CBA-sensitivity to CO_2 reduction benefits: coal-firing vs. offshore wind. Under different assumptions offshore wind will be more valuable than coal within a certain time span ('year swap' is the year in which NPV of the wind project becomes higher than NPV of the coal-firing project) and given a minimum estimate of the benefits associated with CO_2 emission reduction. The only exception within the considered options will be in the case including a feed-in tariff for a constellation foreseeing minimum cost-estimates for coal and maximum for wind: the NPV of coal will stay higher than that of wind during the considered time-interval. Only benefits higher than 45€/tCO_2 would change this result.

Assumptions		NPV_w	NPV_c	swap	$NPV_w >0$	$NPV_c >0$	min. benefits
feed-in	capital inv., O&M	(€)	(€)	(year)	(year)	(year)	(€/t CO_2)
	coalmin-windmax	$7.87{\cdot}10^9$	$4.03{\cdot}10^9$	2023	2021	2019	22
no	coalmax-windmax	$7.87{\cdot}10^9$	$1.94{\cdot}10^9$	2018	2021	2025	12
	coalmin-windmin	$1.13{\cdot}10^{10}$	$4.03{\cdot}10^9$	2013	2016	2019	8
	coalmin-windmax	$2.09{\cdot}10^9$	$4.03{\cdot}10^9$	none	2031	2019	45
yes	coalmax-windmax	$2.09{\cdot}10^9$	$1.94{\cdot}10^9$	2035	2031	2025	36
	coalmin-windmin	$5.53{\cdot}10^{10}$	$4.03{\cdot}10^9$	2031	2024	2019	42

at the minimum combined mean value reported by Tol (2005) of about 11 €/tCO_2 (ca. 16 $), under the current assumptions coal-firing would be more efficient than offshore wind, except for the case of considering minimum costs for offshore wind.

3.3 Cumulative effects

The target of this paper[14] is to determine 'how healthy' the ecosystem is under the pressures analysed in the eutrophication and offshore wind case-studies, i.e. how well the ecosystem can carry out its functions under different conditions. Particular emphasis is given to (1) the comparison of the effects of each of these different uses upon the North Sea ecosystem in terms of ecosystem integrity and ecological risk; and (2) a test of the ecological risk suitability for assessing cumulative or combined effects of eutrophication scenarios and offshore wind construction scenarios.

Scenario combination

Scenario assessment has been based on world views and priorities (storylines), which will result in pressures upon the marine environment (quantified by guesstimates). Those pressures are the quantitative aspects linking socio-economic scenarios to modelling, by providing the assessment of input variables to be used in the model. In the case of eutrophication, the pressures impacting the marine ecosystem are nutrient emissions, therefore under each scenario a different degree of emission reduction is considered, based on scenario priorities, risk perceptions and interpretation of the precautionary principle (see sections 3.1.1 and 3.1.2):

[14] The study reported in this section has been submitted for publication in Landscape Online as Nunneri, C., Lenhart, H.-J., Burkhard, B., Colijn, F., Müller, F. and Windhorst, W. 'Ecological Risk for assessing effects of different activities: an example including eutrophication and offshore wind in the North Sea'

- a low reduction scenario (LOW-RED, BAU), where nutrient inputs into the North Sea are reduced of 20% with respect to 1995 levels;
- a medium reduction scenario (MED-RED, PT), where nutrient inputs into the North Sea are reduced of 40% with respect to 1995 levels; and
- a high reduction scenario (HIGH-RED, DG), where nutrient inputs into the North Sea are reduced of 60% with respect to 1995 levels.

The time horizon of those scenarios is up to 2025. In brief, economic growth drives society (weak sustainability) under the first scenario, a political commitment to strong sustainability dominates the second scenario and finally long run environmentalist perspectives (very strong sustainability) prevail in the third scenario. In the case study 'offshore wind' the pressures resulting from socio-economic scenarios driven by human needs and perceptions are the construction of offshore wind farms and the resulting temporary increase in suspended particulate matter (SPM) in the marine waters. Three levels of total installed capacity in 2055 (the required area being proportional to the installed capacity) have been quantified based on scenario assumptions (see section 3.2.2):

- under the North sea as shipping area (E1) scenario, ca. 2329 MW installed in 2030 and 15000 MW in 2055;
- under the North sea as natural area (A2) scenario, ca. 15000 MW installed in 2030 and 55000 MW in 2055; and
- under the North sea as energy park (B1) scenario, ca. 25000 MW installed in 2030 and 90000 MW in 2055.

In short, E1 gives priority to good-transport, thereby allowing limited construction of offshore wind parks, which is compatible with the enlarged shipping lane-network, scenario A2 prioritises nature conservation although allowing some offshore wind farm installations, whereas scenario B1 sees Germany having a leading role in energy production. Different world views are behind those scenarios (for a detailed description see Burkhard (2006). The storylines formulated to underpin the scenarios, describe society values and priorities playing a major role in socio-economic development. According to the prevailing attitudes and priorities, scenarios belonging to the two case studies can be combined into 'eutrophication reduction and offshore wind construction' scenarios. This has been done based on two main aspects: globalisation degree (governance) and societal values. This procedure excludes some combinations. Figure 3.37 shows the scenarios which describes comparable world-views, plotted against two axis: globalisation (regional vs. global) and social values (individualism vs. community). Based on the pattern shown in figure 3.37, three 'eutrophication reduction and offshore wind construction' scenarios can be considered: (1) low nutrient reduction being associated with low offshore installed capacity (LOW-RED-E1); (2) middle nutrient reduction being associated with high installed capacity (MID-RED-B1) and (3) high nutrient reduction being associated with middle installed capacity (HIGH-RED-A2). We choose here to analyse the effects of the second combined scenario (MID-RED-B1) in order to compare it to the effects obtained for each 'single issue' scenarios and test whether from the single scenarios would have been possible to assess ecological risk of the combined scenario without the aid of modelling. It is worth to stress that, while eutrophication scenarios represent changes in the long run, the construction scenarios are assessed on a

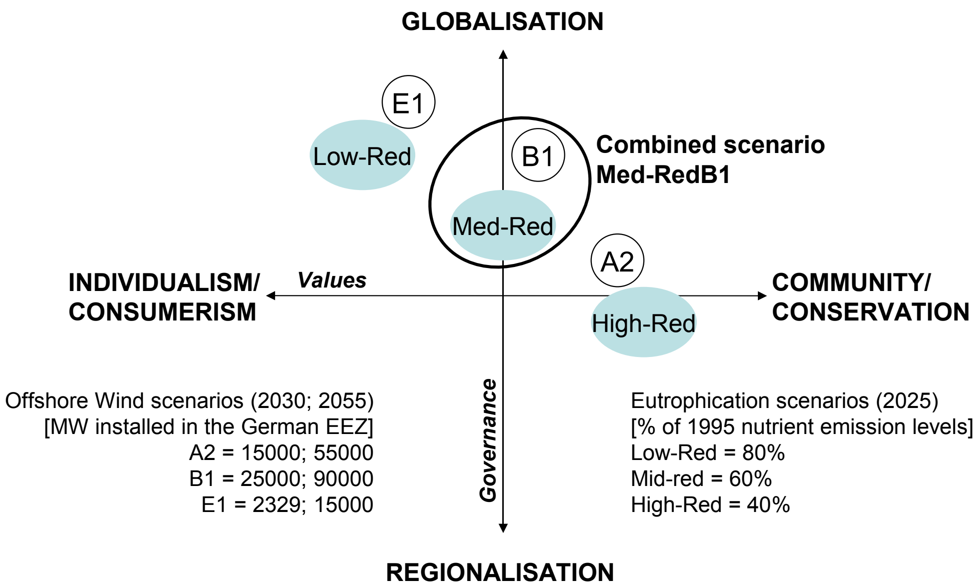


Figure 3.37: Combining eutrophication and offshore wind scenarios. Scenarios dealing with different issues can be combined based on the assumptions underlying the storylines (world visions, perceptions and values, for more details see text). Scenarios presenting similar socio-economic characteristics are closed to each other when represented on a graph having governance and social values as orthogonal axis.

year by year basis, in which every year construction takes place in different areas depending on which project is assumed to be realised (Nunneri et al., in press).

Ecological integrity and risk assessment

The two case studies are described by means of the same indicators, nevertheless the effects upon ecosystem integrity and the assessment of ecological risk have to be interpreted differently. The results for the eutrophication scenarios reflect a system which has been exposed to continuous reduced nutrient supply by the rivers and has adopted a new steady state (long-term perspective) (Hofmann et al., 2005; Nunneri et al., in press). On the contrary, the offshore wind-park scenarios relate primarily to the construction phase (higher suspended matter concentration) and therefore represent a short-term disturbance.
In table 3.20 the minimum and maximum scenario values are reported compared with reference

Table 3.20: Comparison among minimum and maximum indicator values throughout the scenarios and available literature data or ERSEM standard modelling runs. Please note that the minimum and maximum values as listed in the table are the minimum and maximum found among all scenarios for a single case study (i.e. eutrophication, offshore wind or the combined scenario) and the values reported in a column may not correspond to a single scenario (in the case of eutrophication) or construction year (in the case of offshore wind). The reference values are either from existing literature or from ERSEM standard runs.

	Scenarios								
	Eutrophication		Offshore wind		Combined		Absolute		
Indicators	Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.	References[a]
Net primary production	234	266	214.1	317.6	166.3	252.6	166.3	317.6	240-261 266^{EB} 303^{EBM}
Turnover of winter nutrients	3.7	4.6	3.1	4.5	2.6	3.9	2.6	4.6	3.1^{EB} 6.8^{GB}
Sediment storage	-7.4	7.1	-3.5	10.6	-3.6	9.7	-7.4	10.6	6.9^{EB}
Matter bilance	-47.4	-21.7	-55.4	-44.9	-42.6	-34.1	-55.4	-21.7	-47^{EB}
Diatom/non-diatom ratio	0.37	0.4	0.46	0.5	0.38	0.43	0.37	0.5	0.33-0.86 0.47^{EB} 0.37^{EBM}

[a] Data from Joint and Pomroy 1993; van Beusekom and Diehl-Christiansen, 1994; ASMO, 1997. The values containing EB are values obtained from the ERSEM standard run for the Elbe box, those containing GB are obtained from the ERSEM standard run for the German Bight and those containing EBM are data obtained for the Elbe-box through combined ERSEM and MONERIS modelling (where MONERIS riverine transport data are used as input for ERSEM modelling).

values. As it can be seen throughout the scenarios the selected parameters may vary considerably.

Especially noticeable is the lowest primary production (166 g C m^{-2} y^{-1}) occurring in the combined scenario Med-Red-B1. This value occurs under scenario B1 for the year 2015, where construction within the Elbe box, takes place in box 58 and 68. The extreme low primary production (if compared with reference values) results from the combination of lower nutrient concentrations (due to the 40% reduction assumed in the Med-Red scenario) and light limitation (due to construction) in the Elbe box. These two boxes, which represent the deeper part of the aggregated Elbe box, cover about 65% of the total volume of the Elbe box. The strong effect of this construction constellation involving both box 58 and 68 can already be seen in the minimum value for primary production in offshore wind scenario B1 (214 g C m^{-2} y^{-1}), in the combined scenario nutrient reduction further diminish primary production. The maximum primary production (317 g C m^{-2} y^{-1}) occurs for offshore wind scenario E1 in year 2011. In this year construction takes place only in boxes outside the Elbe box and adjacent to it. This means that nutrients which cannot be used in those boxes due to light limitation are transported into the Elbe box, where they are taken up and result in enhanced primary production.

In the case of eutrophication scenario reductions imply the reduction of both N and P (in equal degree), while the integrity and ecological risk analysis presented here only focuses on the nitrogen related fluxes within the reduction scenario runs in comparison to the standard run. The eutroph-

icated state is indicated by highest primary production and extremely positive N sediment budget (N retained in the sediment). The tendency to reduce primary production as a consequence of reduced river inputs into the system is clear throughout the scenarios (see section 3.1.2). The sediments represent a buffer for the system, being a sink for overabundant nutrients and matter and a source in scarcity times. Since less organic material is produced as a consequence of the river input reduction, a smaller amount reaches the sediment, but the budget between organic sediment input and remineralised inorganic flow back into the water column remains positive even under pristine condition. In the same way, the tendency to increase the cycling of nutrients in relation to the winter content (the system needs to re-use available nutrients, if the inputs of new nutrients is reduced) and to minimise matter losses (i.e. to increase the quantity of matter retained by the system) are observable. The values of diatoms/non-diatom ratio are spread between 0.4 for the standard run and 0.37 for the pristine condition. Basically this implies that the group of diatoms is slowly increasing with reduced river inputs, which is in agreement with the documented increased flagellate concentrations in the water to the enhanced nutrient concentrations (Radach et al., 1990; Radach, 1998; Hickel et al., 1993).
The risk associated with the three scenarios shows values ranging between a maximum of 86 (being the theoretical maximum value of risk 100) for the Low-Red scenario, and 29 for the High-red scenario (being the risk associated with pristine conditions set equal to 0). An analysis of ecological risk associated with eutrophication scenarios including the socio-economic aspects is reported in section 3.1.2.

In the case of offshore wind farm construction, the situation without wind parks is assumed to present a minimum risk level, while a whole year construction in box 58 offers some of the highest alterations of the indicator values in the system (3.2.3). Due to modelling conditions, however, as the ERSEM model is calibrated for the year 1995, the construction of offshore wind parks is 'superimposed' to the 1995 nutrient level, thus implicitly implying a scenario 'maximum eutrophication level-selected offshore wind construction scenario (B1 or E1)'. An increase in SPM concentration of 2 g/m^3 (used as a threshold level in a study by the Danish Hydrographical Institute, DHI, 1999) on top of natural background SPM concentrations is assumed during construction phases from Mai to September, with a fading off phase back to background concentration in October. The higher concentration is applied for the whole ERSEM box in which the wind farm is constructed, independently of the dimension of the offshore wind project, i.e. the number of installation was not related to the SPM increase.
The behaviour of the indicators is determined by the fact that construction of wind farms is assumed to take place at different time in different ERSEM boxes and also foresees parallel construction in more than one box.
A pattern, which can clearly be seen is the reduction of primary production in years where construction takes place in a box within the aggregated Elbe box, thus resulting in increased SPM concentrations within the Elbe box. In those years, according to a reduced primary production, also other related fluxes are reduced, e.g. organic input to the sediment, in similarity to the findings within eutrophication scenarios. In particular the indicator values in year 2015 are very similar to those of the maximum disturbance, defined as a 12-month construction occurring in box 58. This occurs because boxes 58 and 68 together cover about 65% of the volume of the considered area, thus imposing a light limitation to a consistent part of the aggregated Elbe box. Ecological risk assessment for the offshore wind construction scenarios is reported up to 2015 (see figures 3.33 and 3.34, pages 114 and 115). It is clearly to be seen that risk arises in the years

where construction takes place, while it is near zero in other years. For the Elbe box (page 115), the maximum risk, beside the one associated with the 12-month construction in box 58, having a value of 69, is associated with scenario B1 for the year 2015, value of 72, due to construction in a large area in the Elbe box. For scenario E1 a maximum risk of 45 occurs in year 2012, and is associated with construction in box 68 only.

The results for the combined scenario Med-RedB1, are shown in the form of ecological risk only. The 'cumulative' risk associated with the combined scenario has been assessed in two ways: (a) based on single scenarios through normalising the values for both eutrophication scenarios and offshore wind construction scenarios, and (b) based on the indicator values obtained by modelling the combined scenario Med-RedB1. The aim was that of testing whether the cumulative effects under the combined scenario could be extrapolated from the results previously obtained from the offshore wind and eutrophication scenarios separately. The assumption underlying this test was that ecological risk of Med-RedB1 would be obtainable by subtracting the 1995 risk from B1 (it was implicitly included, being ERSEM calibrated for 1995) and adding to the result the risk of Med-Red. The indicators were computed by ERSEM modelling, under the assumption that the nutrient reduction scenario had taken place already and construction would take place in a '40% less nutrient rich environment' than in 1995. In figure 3.38 the ecological risk assessment for some scenarios is reported. In this graph the risk has been assessed by normalising each indicator considering as maximum and minimum values the absolute maximum and minimum among all values assumed by each indicator throughout the two offshore wind scenarios, the three eutrophication scenarios as well as the 1995 and the pristine conditions for eutrophication. The indicator values of the combined scenario have not been included in the choice of the minimum and maximum as they were assumed to be comprised in the range of the values obtained in the single scenarios. This was not the case, as in three cases the values of the combined scenario are clearly lower or higher than the values obtained for the scenarios within the single case studies; this is observable for primary production, showing extremely low values in 2011 (211 $^{-2}y^{-1}$) and 2015 (166 g $C/m^{-2}y^{-1}$) and for turnover in 2015 (a minimum value of 2.6). In this context the combined scenario shows an inferior primary production if compared with the one obtained in 2015 for scenario B1 only (214 g $C/m^{-2}y^{-1}$), a less negative matter balance than in scenario B1 alone (reflecting the lower availability of organic and inorganic matter) and a lower diatoms non-diatoms ratio (0.3 vs. 0.49 in the combined scenario). The combined scenario shows the lowest primary production throughout the considered data set, thus highlighting effects resulting from the combination of lower nutrient availability and light limitation due to construction. While the effects of reduction and construction upon primary production are preserved when the boxes are aggregated into the Elbe Box, there is a local phenomenon which is weakened by this procedure. This has to do with the effect that a reduced primary production in a box related to construction work implies a lower uptake of nutrients. These nutrients that are not utilised are transported into the neighbouring box and cause an increase of primary production in that box slightly over the level of the standard run. In the combination of boxes with lower production and slightly increased one the later signal gets lost. Therefore there is no scenario for the aggregated Elbe box where primary production is above the standard run. The ecological risk values assessed from the modelled data was not obtainable from the single assessment (see figure 3.38, not even in the case of a normalisation procedure including the values of the combined modelled scenario. In general the extrapolated risk was an overestimate of the risk assessed from modelled data.

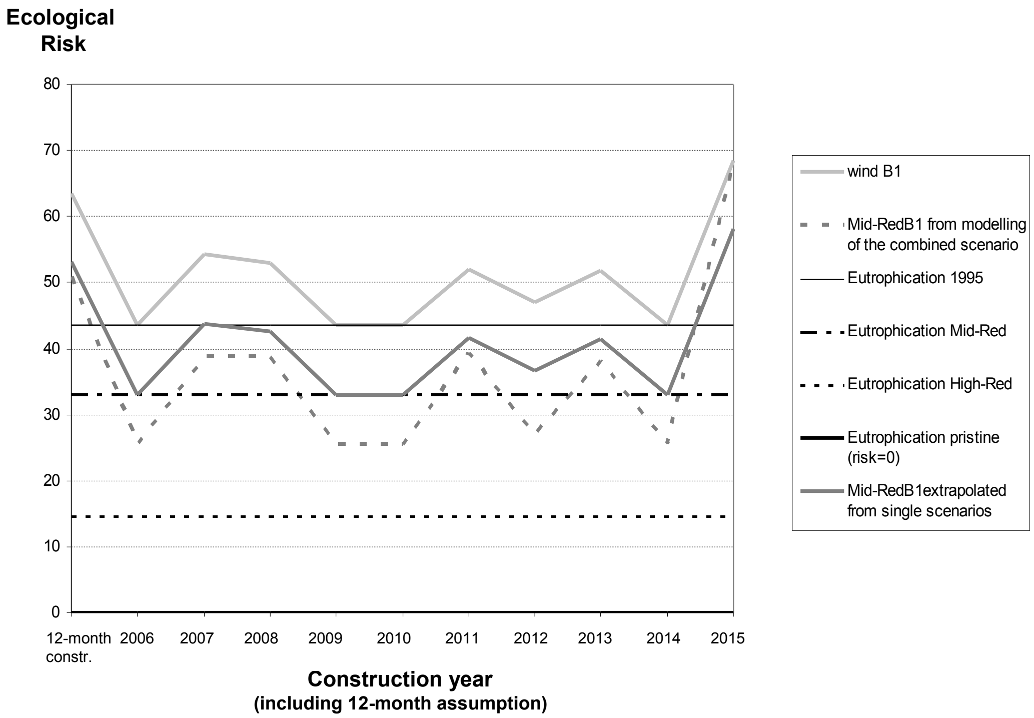


Figure 3.38: Ecological risk calculated by normalising indicator values with respect to minimum and maximum values of indicators throughout all eutrophication and offshore wind scenarios, but excluding the indicator values of the combined scenario. For explanation see text.

Concluding remarks

As shown in the results, the adopted methodology based on ecological integrity and risk makes use of scenarios for assessing the impacts related to different issues. In the case of eutrophication, the modelled reductions have been considered in a time-frame of 25 years, thus being the presented results the end-effect of nutrient reduction strategies. In the case of offshore wind annual construction impact is the main considered pressure and not the ultimate effect upon the ecosystem due to the presence and operation of turbines in the sea. With respect to primary production, the effects brought about by the offshore wind construction scenarios in the Elbe box are much more significant than those resulting from a reduction of nutrient emission corresponding to pristine conditions (234 g C $m^{-2}y^{-1}$). The primary production under the combined scenario is even lower than the one under scenario B1. This shows that cumulative changes may bring about unforeseeable effects with respect to the effects assessed under single scenarios. Through the selected indicators it is possible to show ecosystem process changes taking place as a consequence of changes in human pressures and relative to selected reference conditions. Those changes are always relative in the sense that they allow assessing (integrity and) risk in comparison with some reference situation(s), which is either desirable (pristine condition) or known (current state, 1995 eutrophication level). While in the case of eutrophication the delineation of an 'undisturbed' situation and of an 'high' or 'maximum' risk situation has been relatively straightforward, the choice of reference situations for an emerging issue such as wind farm construction has posed different challenges. Among them the role of the area selected for analysis, which may considerably change reference values and thereby findings. When evaluating if the risk of offshore wind construction is higher or lower of those brought about by eutrophication, the two assessment must be comparable. This means that they need to be calculated with respect to the same reference points. The results of the assessment for the same reference situations is given in figure 3.38, where the risk associated with different scenarios is shown. Being both the assessment of integrity and risk based upon normalised indicator value, a change of reference conditions may considerably change risk appraisal, as it is the case for offshore wind if one compares figures 3.34 (page 115) and 3.38(page 127). It is crucial to distinguish between features brought about by modelling constraints and risk patterns. By looking at figure 3.38 it would be tempting to say that the risk brought about by offshore wind construction is higher than that brought about by eutrophication. But this effect is due to modelling conditions, which superimpose risk to the 1995 (maximum) eutrophication level. In this sense, it is not so surprising that ecological risk related to offshore wind is higher than this related to eutrophication in 1995. The question is then how to compute cumulative risk due to combined actions, or, which is the same, how to assess the effects of one single use independently of other uses. The hypothesis that ecological risk assessed for different issues can be simply added or subtracted to assess cumulative effects has been tested and rejected. It was not possible to obtain the ecological risk levels for scenario Med-RedB1 by subtracting the 1995 risk from the offshore wind scenario B1 and adding the value obtained to the Med-Red scenario, as it can be seen in figure 3.38. This even after having taken the same references conditions, including also the combined scenario. The conclusion is that non-linear effects prevail when considering changes jointly (e.g. nutrient reduction and offshore wind construction). These effects are not obtainable as a sum of the single actions. This poses new challenges to the risk methodology with respect to if and how it can be improved for depicting cumulative effects and stresses the need of modelling tools for exploring different use combinations.

4 Discussion of Results

This chapter draws on and expands material already published or currently being reviewed (Hofmann et al., 2005; Nunneri et al., in press; Nunneri and Hofmann, 2005; Nunneri et al., 2007a,b,c). The main findings reported in the previous chapter are discussed, in particular addressing the aspects relevant for management. Moreover, the cardinal issues emerged through the application and development of the concept of ecological risk within the two case studies are examined. Some considerations of relevant aspects connecting ecological risk and DPSIR for management, namely the issues of risk acceptance, governance and participation, conclude this chapter.

4.1 DPSIR in the North Sea, aspects relevant for management

The DPSIR approach has been applied to two case studies in the North Sea, which present a different level of 'management maturity'. The issue of eutrophication is, even if not completely understood in its multiple causes, well documented and there is a policy framework in place to reduce nutrient emissions to the coastal waters via the river systems. The issue of offshore wind farm construction is an emerging use of the sea, and thereby still lacking comprehensive regulation and understanding of its possible impacts on the marine ecosystem. In the following the main findings reported in chapter 3 for the two case studies are discussed.

4.1.1 Eutrophication

The application of the DPSIR scoping framework to analyse the interaction of activities in the catchment, and their effects on eutrophication in the coastal waters, offered a valuable aid for structuring a complex analysis. Following the DPSIR causal-chain, nutrient emission sources in the river catchments discharging in the North Sea have been addressed by possible emission reduction measures. In particular, the Elbe basin (section 3.1.1) and the Rhine and Humber basins have been studied in more detail (section 3.1.2).

The uncertainty affecting future socio-economic conditions has been analysed through scenario assessment. The presented scenarios (sections 3.1.1 and 3.1.2) have been associated with different policy contexts as well as environmental objectives, and interpretation of the precautionary principle and thereby risk attitudes (e.g. in section 3.1.2). Scenario priorities towards economic growth, sustainability or nature conservation (very strong sustainability) have been used as a starting point for assessing feasible reduction measures. Both the changes induced by such management measures (i.e. reduction in fluxes of nutrients) on the coastal ecosystem and the costs of implementation have been assessed in the analysis.

The Elbe river catchment has been taken as an example for analysing the main relevant drivers in the catchment and the effects of reduction measures on the coastal waters. In the following section the main findings related to the Elbe river are discussed.

The Elbe river catchment and coastal zone as an example

In case of the Elbe river basin, the most important outcome of the DPSIR application has been the definition of scenarios and the application of models to evaluate measures in the catchment in order to meet different environmental standards in the coastal zone of the German Bight (section 3.1.1). The application of the ecological risk concept has been used for comparing the economic and ecologic effects of nutrient emission reduction in three catchment-coastal areas (section 3.1.2). Finally stakeholder involvement has enabled to focus on societal aspects relevant for successful reduction strategies (section 3.1.3).

The scenario storylines have been used to assess the pressures impacting on the coastal zone through the river system (by applying the MONERIS model) and the consequent changes and impacts on the ecosystem state (by applying the ERSEM model).

From the modellers' point of view, a key-aspect of the integration of existing analytical tools for carrying out the analysis along the DPSIR scoping framework, has been the linkage of models originally developed for sectoral analysis. In practice, the challenge was linking the steady-state meso-scale catchment model MONERIS with the dynamic ecosystem model ERSEM by means of a transfer function (see page 49). Through the MONERIS model, the input of nutrients from the catchment to the coastal zone has been described for different point sources and diffuse pathways, thus allowing a test of the efficiency of management measures acting upon different aspects of nutrient inputs. The application of the model MONERIS identifies the diffuse pathways of nutrient emissions into the river systems, and the separation of the total emissions and loads into various sources. As a result, wastewater management and agriculture could be highlighted as the main drivers of nutrient emissions. Agriculture has been addressed as the sector that should next be tackled by reduction measures, if the target is that of a further reduction of the nutrient emissions (especially of nitrogen, see sections 3.1.1 and 3.1.3). Hence, food and energy demand as well as population density patterns (wastewater issues) can be accounted as the principal drivers with regard to eutrophication.

The high residence time of nutrients (15-30 years) in groundwater is a problem in the Elbe catchment and can provide a good political excuse for no action or minimal interest in elaborating a reduction strategy within the river catchment (sections 3.1.1 and 3.1.3).

Moreover, according to Lenhart (2001), a reduction by 50% in riverine nutrient load for N and P cannot be linearly transferred to a similar reduction in primary production in the coastal waters. A 50%reduction of the organic and inorganic river loads, would result in a maximum primary production reduction of about 20% in the coastal zone (Hofmann et al., 2005).

The impact of reduced nutrient loads has been assessed for a broader area influenced by the Elbe river (the aggregated Elbe box, see figure 3.4, page 35). A difference in the resulting time series for the scenarios of DIN and DIP was found. Whereas DIP concentrations under different reduction scenarios show strongest differences in winter, DIN concentrations show clearly distinct patterns all over the year for different scenarios (see figures 3.12 and 3.11 pages 52 and 52). 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 current high nitrogen load of the Elbe River. While the corresponding reaction of the biological parameters in the aggregated Elbe box was very low, box 78 (the coastal area directly influenced by the river Elbe plume) showed a clear reaction to reduced nutrient input. For box 78, diatom and flagellate abundance expresses a clear decrease in the eutrophic state of the coastal zone (see figure 3.16, page 56). However, it should 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. The evaluation of reduction scenarios on the status of coastal waters needs a reference state, and indicators which can describe ecosystem functions and ecosystem integrity. In the sense of the WFD, the estimation of natural background concentrations has been used as a baseline to assess ecological integrity of the coastal ecosystem by means of self-organising processes (see table 3.9, page 58 and figure 3.20, page 69). Nearly all integrity indicators are sensitive to reduced nutrient loads from the Elbe, but to a different extent. In relative terms, the storage function of the coastal ecosystem shows higher changes than the other indicators throughout the scenarios. This indicator could reveal essential information about the functioning of the coastal ecosystem. In general, the integrity of the ecosystem is increasing with decreasing riverine nutrient loads. However, the 'most integer' state, i.e. the one under 'pristine condition', would not be reached even under the highest reduction (60% assumed in the deep green scenario).

Possible nutrient reduction strategies have been appraised by a multi-criteria analysis (MCA). Five policy alternatives (no additional measures=BAU scenario, MR and MRHR=PT scenario, SR and SRHR=DG scenario, see page 54) some including high-retention possibilities in the catchment (e.g. wetlands) have been compared based on their economic, social and ecological effects. The effects of those alternatives are assessed under the BAU scenario, but they are comparable to the formulated scenarios. The outcome of the multi-criteria analysis shows that, under the assumptions of equal relevance across the economic, social and environmental effects (see section 3.1.1, page 54) the strong-reduction alternative with the possibility to construct high nutrient retention basins and dams in the catchment (SRHR) is the highest ranked management option. However, three alternatives are quite competitive, namely BAU (due to low implementation costs), MRHR and SRHR (due to relative high socio-ecological benefits), while the options without high retention in the catchment show a less advantageous trade-off (see 3.17, on page 59).

The management options appraised through the MCA are assessed based on both scientific knowledge and restricted expert consultation. In other words, they included mainly the 'rational perspective' of what measures are possible, but little information about measure feasibility or acceptance among the affected stakeholders. A number of affected stakeholders have been involved in appraisal of management measures. The interviewed stakeholders gave a varied insight into policies, policy acceptance and feasibility of different measures for nutrient reduction, as well as possible conflicts within the catchment of the River Elbe, thereby underpinning the existence of factors other than rational choice (e.g. transparency and strategic communication), which would affect measure implementation and acceptance (see section 3.1.3).

A number of issues have been mentioned in the interviews, allowing for a multi-perspective point of view. In general, there is a wider range of quality issues associated with riverine and estuarine waters and not all stakeholders consider eutrophication as the most burning one. Agriculture is the sector that experiences the maximum pressures for nutrient reduction, however, farmers are rarely, if at all, involved in the discussion of reduction targets and in the appraisal of reduction measures. In general, successful reduction strategies should find a way of achieving voluntary (or at least committed) agricultural cooperation, by helping farmers to bear the costs of reduction. For instance, subventions for environmental friendly agriculture, which are indicated as the most 'reasonable' measures for combating catchment driven nutrient enrichment of the coastal waters, should be 'reasonable to apply' on behalf of the farmers' point of view, because only in this way they will be accepted and integrated in production patterns. This calls for more participative and effective communication and exchange of information about the interested and affected parties. Under the current institutional framework and governance structure, however, there is an urgent need of improved communication between policy-makers and key-stakeholders, in order to achieve win-win solutions. The actual lack of communication will represent an obstacle to environmental policies, if policy makers involve key-stakeholders only in a passive role, as actors of an already written screenplay. An in-depth comprehension of the production patterns and an 'empathic' reasoning would allow for successful cooperation towards nutrient reduction. In this context, not only further education of farmers plays an important role, but also 'further education' of decision-makers will be desirable. Within the Elbe catchment still much has to be done for achieving synergetic action in handling environmental problems. The dissemination of knowledge is stagnating not only in the direction flowing from science towards policy (in which actually much has been done during the last years), but much more among different institutions, even within the same country, and between policy networks and the general or the affected public.

International issues

The issue of eutrophication is, by its transboundary nature, of international importance. In the context of an holistic management, an analysis of economic and anthropogenic activities, causing undesired impacts on the environment, should not only be interdisciplinary, but also spatially integrated. This means that the study area needs to be extended to units that are physically and geographically recognisable (such as river and sea basins), rather than being confined within political boundaries. Applied within a sea basin, as the North Sea, in practice two 'directions' need to be taken into account for international management: the one is away from the coastline, accounting for the duality coastal-catchment states, the other is along the coastline, considering a multiplicity of coastal states. In the light of the OSPAR international targets, as well as for implementing any European directive concerning marine water quality and pollution (e.g. the Water Framework Directive, the Nitrate Directive, etc.), a better understanding of catchment–coastal zone interactions is indispensable, as well as appraisal of the role of single catchments in the light of the overall contributions to quality objectives.

Those two spatial aspects of international management related to the eutrophication issue have been analysed in sections 3.1.2 and 3.1.3.

Financial burden sharing for the implementation of environmental measures is a delicate issue containing in itself the potential for triggering conflicts. The willingness to pay for achieving better water (and environmental) quality is a matter of values and perceptions, as shown in section 3.1.2 (figure 3.19, page 64). Measures that are judged as 'disproportionately expensive' in one country can be implemented in another (due to both different economic conditions as well as socio-cultural values). In the case of common goods, there should be a way for international cooperation to deal with (and solve) the problems of excessive costs for achieving a single 'international standard' rather than many different 'national' standards. Many interviewees agreed that costs and benefits should be (re-)distributed in such a way that there are no net losers or winners. The application of the polluter pays principle (those who are responsible for pollution have to bear the costs of pollution reduction) should be supported by compensation principle and the solidarity principle, i.e. the costs of reduction must be divided in such a way that they are bearable for all parts. For example, if the target of nutrient emission reduction has to be achieved in the Elbe catchment (seen as 'one large source' of nutrients for the North Sea), the economic and socio-cultural differences between the Czech Republic and Germany cannot be ignored in the process of international management. For instance, the construction of WWTPs in the Czech Republic has been carried out with economic help from Germany, as this country would enjoy the 'ultimate' benefit of 'cleaner' coastal waters; this kind of burden sharing is especially applicable when the most well-off country holds the benefits. Financial difficulties should not hinder (as far as possible) environmental improvement: if money is available for a specific issue, this should allow for optimal allocation of funds among the involved parties, thus enabling win-win solutions. On the other hand, an ineffective institutional framework, (as currently perceived from the interviewees) in both Germany and the Czech Republic (interest conflicts exacerbated by lack of communication and coordination), may hamper the sharing of issue-specific funds. In this context, some institutions (e.g. the IKSE, the international commission for the protection of the Elbe river) are seen to have the potential to take over a central role for coordinating actions and acting as a catalyst for communication at the national and international level.

Economic issues, however, are not the only concern of stakeholders when they decide to commit in environmental management. Expenditures should be justified by a clear relationship cost/environmental improvement, in order to facilitate stakeholder commitment also at international level. Fulfilling this criterion in the case of a multi-causal effect as eutrophication (and most environmental issues), seems not to be possible, rather, there should be a precautionary principle orientation.

For international management 'along the coastline', the contribution of three river catchments to eutrophication of the southern North Sea has been appraised in terms of ecological risk and economic analysis of reduction measure implementation (see section 3.1.2). The effects of local reduction through single river catchment in their respective coastal zones have been compared in the light of international action for reaching a common goal. Through the concepts of ecosystem integrity and ecological risk has been possible to show that similar emission reduction in the catchments do not necessarily result in equal ecological risk reduction in the respective coastal waters (see figure 3.22, page 71). Moreover, through the economic analysis associated with the implementation of reduction measures, it was evident that reduction of nutrient emission in the Rhine catchment (the largest considered) would be in terms of costs pro capita, the most cost-effective reduction strategy (see table 3.13, page 72). The used approach could potentially be deployed for supporting international decision-making by extending the analysis to the North Sea as a whole. This would facilitate an evaluation of cumulative/synergetic effects vs. local effects, thus allowing for burden-sharing and compensation strategies.

The central findings for integrated management of eutrophication in the North Sea can be summarised as follows. The analysis in the Elbe and Rhine river catchments has highlighted diffuse sources as one of the main target of further nutrient reduction measures (sections 3.1.1 and 3.1.3); in particular among considered emission reduction measures those including wetland retention have been ranked high within a multi-criteria evaluation. However, when considering the effects of reduced nutrient emission in different coastal zones it has been found that achievement of (e.g. OSPAR) emission reduction targets do not necessarily results in a similar reduction of ecological risk (e.g. primary production) in the coastal zones (e.g. section 3.1.2, table 3.13, page 72). Moreover, economic costs of measure implementation do not vary linearly with ecological risk reduction, therefore requiring the assessment of trade-offs as well as spatial burden sharing among interested parties. These complex aspects, involving risk acceptance and willingness to pay for risk prevention, demand more transparent and participative decision-making approaches. International eutrophication reduction in the North Sea relies on reduction measure implementation within river catchments, thereby requiring more inclusive and spatially integrated participatory approaches. Further analysis should consider the effects of combined reduction of all river catchment discharging in the North Sea, in order to assess the possible contribution of each river system towards OSPAR reduction targets. Such an assessment, augmented by socio-economic analysys, would allow a North-Sea-wide management process.

4.1.2 Offshore wind

In the case of offshore wind, the starting point of the analysis has been to observe what drivers are the cause of its development and what aspects may hinder it, through the action or non-action of the main involved actors (section 3.2). In a similar way as carried out for eutrophication, scenarios for the construction of offshore wind farms have been assessed in order to quantify the resulting pressures (section 3.2.2) and impacts on the ecological state (section 3.2.3). Finally, a cost benefit analysis has been carried out for comparing the considered scenarios as well as offshore wind and coal-firing electricity generation (section 3.2.4).

Drivers for offshore wind farm development have been analysed in detail in section 3.2.1. Besides the extent of already achieved targets for both renewable energy and greenhouse gases emission reduction, some other aspects play a major role in facilitating or hindering the development of the offshore wind sector. The comparison between two different countries in the North Sea, the UK and Germany, highlighted how similar targets can result in different legislative frameworks and those in turn can be more or less efficiently implemented based on different historical and political settings, as well as cultural values and power distribution among the involved key-actors. The analysed aspects revealed that the factors influencing policy implementation range from economic aspects (the nature and extent of subsidies, currently larger in the UK than in Germany) to the government attitude towards engineering works in the North Sea (precautionary or more risk-seeking) and finally to political and power structures within each country. The UK government set out better conditions by realising a faster consent procedure and making capital grants available, in such a way that a quick realisation of the first (nearshore) projects took place, thus allowing for an appraisal of their ecological and economic effects and a refinement of both legislation and consent procedure. The German government, on the contrary, has adopted from the beginning a more precautionary attitude, requiring offshore wind farms to be constructed further away from the coast, in order to ensure both environmental protection of the sensitive Wadden Sea coastal area and minimal opposition (e.g. due to visual impacts). However, the German government has not recognised early enough the economic burden that this decision put on developers, thus making investments too risky. The result has been that large energy companies, which should have invested in renewable projects, have not moved until recently. Those companies, who are obliged to invest in renewables in the UK and free to do so in Germany, have preferred –in Germany– to protect their conventional investments. Only recently the German government took decisive action: the state will financially support an offshore test-field for the 5 MW technology (Borkum) and a new legislative act for the realisation of a 'power socket' in the sea has been promulgated. In the future it will be possible to verify, if the more lengthy preparation in Germany will result in a delayed but nevertheless expanding sector.

The analysis of governance issues surrounding the development of offshore wind farms in the North Sea, in particular the regulatory framework in place, can be interpreted as a 'response' to other environmental issues of global concern, namely global warming and climate change. Action undertaken in response to one environmental issue (climate change) [1] by starting a new activity, which itself may change the environment and cause new impacts, opens a new DPSIR-loop for analysis.

Further analysis has focused on offshore wind farm project realisation in the German exclusive economic zone (EEZ). The assessment of socio-economic scenarios, based on societal needs, values and perceptions (section 3.2.2) has allowed the quantification of different plausible pressures in the German EEZ, in terms of offshore wind installed capacity over a 50 year-time frame. Based on those quantified pressures both a risk analysis (section 3.2.3) and a cost benefit analysis (section 3.2.4) have been carried out.

Given the quantification of offshore installations in each scenario, the assessment of impacts should include –in the optimal case– all phases of the project life-time, i.e. construction, operation and decommissioning. Offshore wind farm projects may affect the marine environment in manifold ways, such as through sediment resuspension, noise, magnetic fields, change of hydrographical and structural patterns. Various modelling tools can be deployed for scoping and assessing the effects of construction and operation upon the ecosystem in general and in particular on fish and top predators, including mammals and birds. In this study the assessment has been limited to supporting services, as indicated by the ERSEM model.

The concept of ecological integrity and ecological risk has first been tested upon a selected area of the North Sea (ERSEM box 58) and then applied to a larger area (the ERSEM Elbe box, see section 3.2.3) for assessing short-term changes in the ecosystem. Those changes are due to increased suspended matter concentration occurring during the construction of offshore wind parks and affecting ecosystem primary production by means of light limitation. Therefore, the underlying assumption is that construction of offshore wind parks will, in the short-term, reduce ecosystem integrity and increase ecological risk, i.e. the risk of disruption of ecosystem functioning. Through modelling it has been seen that effects upon the ecosystem are compensated for each year (Lenhart, pers. comm.), therefore only the time frame 2006-2015 has been considered

[1] Current lifestyle requires energy production and consumptions(D) which results in increased GHGs emissions(P) altering the climate (S) and possibly leading to floods and other catastrophes (I), one of the possible responses to this loop is to produce renewable energies which reduce GHGs emissions(R).

as an example for assessing ecological risk during the construction-phase. In the case of offshore wind farm construction scenarios, when limiting the evaluation to a sub-area of the whole exclusive economic zone corresponding to a single ERSEM box (box 58 in this case), there is a need to distinguish among internally and externally driven changes (direct and indirect effects, section 3.2.3). It has been highlighted that construction in areas neighbouring ERSEM box 58, which was selected for the analysis, may lead to risk reduction in this box (indirect effects), if compared with situations in which construction takes place directly in the box (direct effects). This is mainly due to the fact that integrity processes are hindered by 'in situ' construction (light limitation) but can be enhanced by construction in a neighbouring area through e.g. transport of nutrients and thereby increase primary production in the considered box.

For this reason a larger area suitable for an 'overall' appraisal of risk has been examined by considering a multiple 'coastal box' (the Elbe box). The effects in the Elbe box have been analysed for two reasons: (1) to investigate whether indirect effects observed when considering box 58 only are levelled out in the case of multiple box aggregation (2) to be able to compare eutrophication and offshore wind scenario effects in the same area. When analysing the effects of scenario construction in the aggregated Elbe box, local effects of construction are compensated for within the aggregated Elbe box (figures 3.31 and 3.32, pages 112 and 113). With respect to primary production, the effects brought about by offshore wind construction scenarios in the Elbe box are much larger than those associated with nutrient emission reduction scenarios, i.e. the indicator values range in wider intervals (see section 3.3, table 3.20, page 124).

Ecological risk arises in the years where construction takes place, while it approaches zero in other years. In figures 3.31 and 3.32 (page 112 and 113 respectively) the ecological risk assessment for the offshore wind construction scenarios is reported up to 2015. In the case of the Elbe box, the maximum risk, beside the one associated with the 12-month construction taking place in box 58, is associated with scenario B1 in year 2015, due to construction in an area extending over 50% of the considered Elbe box. It is apparent that scenario B1 shows a higher number of risk peaks than scenario E1.

By means of a CBA the two scenarios for minimum and maximum offshore wind farm construction into North Sea German EEZ have been analysed in term of their net present value (NPV). Costs and benefits are affected by large uncertainties due to (1) lack of experience of construction and maintenance in deeper offshore water (as it will be the case for the German projects) and (2) environmental benefits of greenhouse gases emission reduction in the light of climate change and global warming (see table 3.17, page 117). The NPV of the scenarios can vary consistently depending on the assessed costs and benefits (section 3.2.4).

As it has been shown in table 3.18 (page 118), the net present value of offshore wind projects, may vary in dependence of real costs and assessed CO_2 reduction benefits in the light of climate change mitigation. Scenario E1 (less installed capacity) would be preferable to B1 only in the case of minimum capital investment and O&M costs as well as benefits associated with CO_2 emission reduction and the extra costs of the feed-in tariff. However, in dependence on the value given to protection of the marine environment, scenario E1 might be preferable to B1, if conservation of marine ecosystems would be valued more than mitigating climate change.

When comparing a hypothetical coal-fired power plant with an offshore wind farm having the same nominal capacity, it is evident that the offshore wind project in the light of climate change is economically a more efficient alternative to coal-firing (see figure 3.36, page 120). This result is valid under the considered cost and benefit estimates and depends to some extent on the worries about climate change consequences and thereby on the willingness of minimising them. This result only holds for estimates of benefits deriving from emission reduction lower than a net present value of CO_2 reduction of 22 €/t CO_2 (or 45 €/t CO_2, if considering the feed-in tariff) which are, according to Tol (2005) still within 'reasonable' existing estimates (see table 3.19, page 121). This means that, if willingness to pay for climate change mitigation would be lower than 45 €/t CO_2 (since there is a feed-in tariff in place in Germany) offshore wind projects would be –under some circumstances– less economic than coal-firing. However this assessment does not encompass in any form a monetary value for the possible ecological risk arising from offshore wind project development in the marine environment. The trade-off between ecological risk minimisation and climate change risk minimisation may considerably change those results.

The main results of the scoping analysis for the case-study offshore winds can be briefly summarised as follows. A comparison between the UK and Germany has stressed the different approach of the two governments towards offshore wind development targets and highlighted that the German approach has resulted in higher costs and encountered opposition of the large energy companies, which has delayed development until recently (section 3.2.1). Modelling of possible short-term consequences of offshore wind farm construction has been used for comparing two offshore wind farm construction scenarios among each other and with eutrophication scenarios. It has been shown that the changes brought about by offshore wind farm construction can be locally very significant, but are of limited duration[2], and the system is able to recover after each construction phase. The effects of massive construction were not, however, included in the considered 'step by step' scenarios, although it has been shown that simultaneous construction in a large area can significantly affect ecosystem integrity. In terms of costs and benefits, offshore wind can be very profitable under the constraint –in the 'worst case' (considering highest costs for wind farm projects and minimal of coal-firing)– that society as a whole recognises a minimum benefit associated with CO_2 emission reduction of 45 €/t CO_2.

In general, it can be said that the lack of knowledge about and experience with the issue of offshore wind installations has resulted in an incomplete application of the DPSIR conceptual framework, in the sense that it highlighted some relevant aspects while others have not been considered. The main obstacle to a more complete analysis has been the lack of adequate sectoral modelling tools for describing the effects of offshore wind construction upon different compartments of the ecosystem: research about broader effects upon the environment and the marine ecosystem is still ongoing: within the Coastal Futures project the effects upon birds, fish and benthos are currently being analysed. For this reason, and based on an already validated model, ERSEM, only short-term ecological issues arising from construction have been considered so far: a further development would require analysing the long-term impacts associated with structural changes as well as operation and maintenance of offshore wind parks (e.g. changed wind patterns and fetch, noise effects on mammals, vertical structures as barriers for bird migration, habitat loss –for birds– or habitat creation –for benthos, etc.) The effects of decommissioning should also be taken into account (although they might be similar to the ones brought about by construction). A first attempt to couple different available models is being carried out within the Coastal Futures project. This would allow a more comprehensive application of the DPSIR

[2]In Germany legislation recognises this issue by prescribing construction in some periods of the year

approach, considering more complex offshore wind farm-ecosystem interactions. The development of a single assessment (including various ecological effects) is, however, not so advanced that an integrated analysis as shown for the case-study eutrophication could be carried out at this stage.

4.1.3 Cumulative effects of eutrophication and offshore wind

In the context of management of the North Sea as a whole, there is a need to evaluate the effects of different uses both in comparative terms (what is better?) and in terms of joint pressures impacting upon the marine environment. In section 3.3 the joint effects brought about by the two considered uses of the North Sea have been analysed. It has been seen that offshore wind farm construction can considerably affect primary production (see table 3.20, page 124). This effect would be much higher under offshore wind farm construction scenarios than under nutrient reduction scenarios, although limited in time (during construction), and not reflecting on the following year. As it has been shown for the combined scenario, when diminishing the availability of nutrients by (e.g.) 40%, the risk associated with offshore construction can also be further reduced with respect to the situation in which offshore wind farms are constructed 'on top' of 1995 nutrient levels (see figure 3.38, page 127). This shows a potential for limiting the ecological risk brought about by offshore wind, by reducing nutrient emissions into the coastal waters. However, it must be said that ecological risk has been assessed by deploying parameters originally used for the case study eutrophication and thereby further risk reduction under reduced nutrient availability can be a result of the chosen indicators, which are 'more' suitable for appraisal of changes related to nutrient reduction scenarios than to changes in other environmental parameters.

4.2 Ecological risk

This section discusses some main points regarding the operationalisation and limitations of the ecological risk concept, which have been faced during this study.

4.2.1 Operationalisation of ecological risk

'Ecological risk' has been defined as the risk of major disruption in the provision of ecosystem services expected/desired by human societies and operationally defined by equation 2.3. There have been different trials to compute ecological risk, before being able to formalise a procedure valid for both case studies, and still equation 2.3 (page 18) might not be suitable for application to other case studies. In accordance with the technical definition of risk, it seemed relevant that risk values are non-negative. For this reason, the absolute value sign in equation 2.3 has been introduced in order to avoid negative values that may arise when indicator values in reference conditions are smaller than scenario values. In the first approximations the sign of the difference between indicators have been set manually according to expert (the scientists working in the project) judgement about reference conditions. This approach worked for eutrophication but resulted in a confused normative approach leading sometimes to negative risks in the case of offshore wind construction. The approach uses two linear elaborations of the integrity indicators, the normalisation (equation 2.2) and the risk assessment (equation 2.3). There are some main aspects that would be worth analysing in a more in depth study about ecological risk methodology.

The first is the application of non-linear assessment procedures. This could foresee both normalisation and risk as non-linear relationships or one of the two. However, a combined non-linear approach for both normalisation and risk assessment should be carefully tested as it might hide or enhance some effects. A non-linear normalisation and risk appraisal could be done in a systematic way as a sort of sensitivity analysis, by adopting different non-linear equations and therefore determining a range of 'possible' ecological risk associated to a certain ecological state, or based on some hypothesised non-linear relationship, derived from ecosystem theory. In the case of non-linear ecological risk assessment ecological states (i.e. indicator values) far apart from chosen reference situations would possibly result in higher ecological risk values and would therefore implicitly force a more precautionary approach in the case of larger changes, whereas they could result in a minimisation of minimal changes.

The second relevant aspect for the computation procedure of ecological risk, is that, in the first approximation, no tolerance intervals are assumed around (reference) values of integrity indicators. This means, for example, that close indicator values may be pulled apart by the linear normalisation. In the ecological risk assessment then they would contribute to ecological risk in base of their 'distance' among each other and from the reference value (equation 2.3, page 18). In other words, through this procedure the distance among different indicator values (and thereby the value of ecological risk) might not always reflect a 'real' issue. Tolerance levels can be incorporated in the assessment by in-depth analysis of time-series (with respect to reference conditions) and including expert judgement (with regard to single indicator values). The inclusion of tolerance intervals accounting for 'natural variability' could significantly change the estimation of risk.

Unfortunately in the available time frame it has not been possible to try out any of the mentioned alternative approaches for ecological risk assessment.

4.2.2 Parameters for Indication

The analysis of this study has been limited to two environmental issues in the North Sea (eutrophication and offshore wind farm construction) and ERSEM model parameters as available indicators. In general, the selected integrity indicators offer a good overview of the changes related to supporting services within case studies, in some cases they have been of difficult interpretation or not sensitive enough. In particular:

- for changes related to the eutrophication scenarios, the sediment budget (storage of nutrients in the sediment) indicator has shown strong changes (sections 3.1.1 and 3.1.2) that highlighted the need for further analysis of sediment dynamics.

- for changes related to the offshore wind scenarios: heterogeneity indication (diatom/non-diatom ratio) is not sensitive to changes in SPM (section 3.2.3), which calls for assessment of diversity on the basis of a larger number of parameters.

These difficulties can be solved, in the case of the storage indicator only by a deeper understanding of the system in general, i.e. by further field-studies and model validation; for the case of heterogeneity, the indication can be assessed on a broader basis, in order to make this indicator more sensitive to system changes. This could foresee biodiversity indicators, which includes more species in addition to the diatoms/non-diatoms ratio (e.g. secondary production and upper levels of the food-web) . The issues of effects on higher trophic network levels, including more functional groups, are currently under investigation (Garthe, Opiz, pers. comm.) and could be included in future ecological risk estimates. Abiotic heterogeneity resulting from the erection of new structures in the marine environment should also be included, as it plays a role as new substrate for benthos growth and thereby for biomass production and filtering processes. Further studies are needed to test if (under its current definition) and how the concept of ecological risk can include other parameters, potentially indicating ecological integrity and risk independently of a special environmental issue.

In the light of the assessment of ecological risk and under the caveat, that reference situation(s) do not always offer a single interpretation-key (i.e. should they be taken as an 'optimum' or as a 'preferential direction'?), indicators available in modelling or monitoring time-series offer the advantage that indicator values resulting from modelling can be compared to historical or time-series 'average values'. However some 'process' indicators are not directly measurable (e.g. turnover or matter exchange between the sediment and the water column). The choice of 'ecological quality indicators' is an ongoing discussion within the OSPAR activities (e.g. for eutrophication problem areas). The (ERSEM) simulation in the southern North Sea coastal area can contribute to this discussion by presenting the reaction or responsiveness of different parameters in relation to the scenarios applied. Process parameters derived from fluxes, rather than from state variables could open new insight by providing information, which can be related to ecosystem theory, but is unlikely to be measurable (Lenhart, pers. comm.).

From a descriptive point of view, the choice of reference conditions is fundamental for the assessment of ecological risk. The incomplete understanding of the ecosystem complexity does not allow defining reference conditions that represent the ecosystem in its optimal state, i.e. to select optimal values for integrity indicators (maximum integrity). Under uncertainty, selection of a reference state is also a delicate matter, which should be based on broader –at least on behalf of the scientific community– consensus. Once selected a reference condition, the determination of risk thresholds and standards is tightly connected with risk acceptance and cannot be set based on the sole discretion of the analysts. This aspect is central to decision-making and will be discussed in section 4.3.

4.2.3 Assessment of cumulative impacts

The concept of ecological risk has been tested for assessing the effects of joint uses of the North Sea, namely, the use of assimilating capacity (through nutrient discharge, represented by eutrophication) and resource use (represented by offshore wind park construction). While the deployed methodology is suitable for addressing relative changes both in the short (offshore wind construction, see section 3.2.3, page 106) and long term (eutrophication, see section 3.1.2, page 67 and following), it cannot be used for assessing cumulative risk resulting from scenario combination on the basis of single modelled issues. In other words ecological risk assessment cannot be used for extrapolating the risk related to a 'more complex' situation based on previously examined 'simpler' ones (section 3.3). Non-linear effects prevail in the response of ecosystem to complex changes. Those non-linear effects are evident in the risk associated with the combined scenario considering a 40% nutrient reduction and a highest installed wind capacity offshore (see figure 3.38, page 127). The procedure for ecological risk appraisal is based on modelling, which offers a solid basis for appraisal of non-linear ecosystem changes, which are not captured in the ecological risk indicator alone. Although the assessment of ecological risk allows for interregional comparisons and comparisons of different issues, thus offering a platform for broader discussion among experts and decision-makers, it cannot replace an in-depth assessment based on interpretation of single indicators and the deployment of validated modelling tools. This means, in practice, that no ecological risk assessment can be made in absence of reliable modelling tools.

4.3 Acceptable risk levels

Throughout this study ecological risk has been assessed for different areas and issues. However, nowhere has an attempt been made to answer the question 'how safe is safe enough?'. This section deals briefly with legitimation of risk standards, thereby considering why, in this case, science does not have 'the right to determine the division-line between tolerable and intolerable risk levels' (Renn, 2001).

According to Renn (1998) scientific analysis cannot determine acceptable risk levels unless '(1) there is a threshold of exposure between zero and some risk or (2) the benefit associated with each risk level is identical', such that the choice of the minimum risk option is obvious. In the case of environmental trade-offs involving major risks (ecological risk), the uncertainty surrounding the determination of thresholds in the case of ecosystem functioning (discussed in chapter 1) does not allow to satisfy the first criterion expressed above. Costs and benefits clearly vary across alternatives (e.g. in both the considered case studies), thus not allowing to satisfy the second constraint either. This context makes the choice of 'safety standards' a complex task, when dealing with ecosystem functioning and connected risks.

The importance of values and perceptions has been mentioned many times in this study (e.g. sections 1.1.2, 2.4, 3.1.2). The way in which causalities (and resulting risks) are approached is influenced by world-views (Valverde, 2007; McDaniels et al., 1995) and so is risk acceptance. There are multiple dimensions influencing risk attitudes: voluntariness of risk exposure (as counter-posed to imposition), knowledge or familiarity with the risk as well as past experiences (catastrophic or reassuring), nature (risk amplification or minimisation) and reliability of channels of risk communication (e.g. governments, media or experts), reversibility of consequences, availability of compensation for risk exposure, advantages connected with risk exposure and risks and gains associated with any other alternative (Thompson and Dean, 2007; Fischhoff, 2007; Kasperson et al., 1988). The 'intuitive understanding of risk is multidimensional, i.e it cannot be reduced to the product of probabilities and consequences' (Renn, 1998). Rather, it is influenced by social and cultural contexts (Slovic et al., 2004; Rippl, 2002, e.g.) and cannot result in risk acceptance levels of 'universal validity and legitimisation' (Renn, 1998). On the contrary, the technical definition, measuring risks in terms of 'expected values' (the product of probability and damage), assumes society's indifference to low-damages/high-probability risk and high-damages/low-probability risk, given that they produce the same expected value. According to this approach to risk, public response to risk should be uniform, which is not observed in reality (Kasperson et al., 1988).

In general, it can be found that 'people are risk averse if the potential losses are high and risk prone if the potential gains are high'(Renn, 1998); risk minimisation is not the only criterion for individual and societal decisions: risk levels in practice are not accepted in absolute terms (Fischhoff, 2007). Rather risk acceptance is evaluated in the context of the broader range of consequences (also positive) of a certain alternative: 'an alternative has a societal acceptable level of risk if its benefits outweigh its risks for any member of society' (Fischhoff, 2007).

In environmental management, the socio-economic approach towards risk-acceptance foresees the examination of trade-offs between present costs (e.g. the implementation of options), future benefits and possible future costs (connected with risk). While the costs do usually take place at present and the benefits as well as the costs possibly associated with risk will take place later in time, risks and benefits should be discounted over time (Renn, 1998), if they *were* assessed in monetary terms. In practice, in the case of ecological risk in its current form, i.e. not entailing a damage function and a probability, it is not possible to include this aspect in an economic cost-benefit analysis, although trade-offs could be assessed based on participatory approaches. Social preferences and values may be explored through elicitation of willingness to pay (WTP) or indirectly through choice experiment studies and factored into MCA appraisals.

As risks and benefits are spread over society as a whole it does not seem reasonable that the determination of acceptable levels takes place in governmental agencies or scientific elites 'detached' from the general public. This approach would not recognise the existence of trade-offs. Democratic decision-making needs to involve those affected by decisions, in order to guarantee respect of different values, lifestyles and 'cognitive patterns' (Rippl, 2002; Thompson and Dean, 2007). In this context, risk communication is a key-aspect to be addressed. The cultural theory approach[3] can be used for taking into account a number of (partly conflicting) attitudes and values that can be reflected in discrepant risk-tolerance levels. By recognising the co-existence of 'plural rationalities' (as counter-posed to the 'single' 'rational actor paradigm', implicitly assumed behind the technical approach to risk), cultural theory may offer a way to communicate risk across a multiplicity of individuals that have different views and priorities, thus helping in finding the way towards (democratic) risk-management (Adams, 2002).

Renn (2001) points out that, although complex and delicate in its realisation, participation cannot be replaced by expert judgement for determining risk acceptance. The author stresses the fact that 'experts do not represent the scope of values and interpretations that characterise the horizon of legitimate values within the affected population'. If the role of the scientific community is

[3]Cultural theory identifies four groups of individuals (the egalitarians, the fatalists, the hierarchists and the individualists), who have characteristic worldviews and thereby different behavioural patterns (e.g. Rippl, 2002; Langford et al., 2000).

that of explain the nature (and likelihood, when possible) of hazards, decision-makers need to establish a legitimate approach for risk decision-making under scientific uncertainty and plural values. In this context, scientific risk analysis is indispensable for informing decision-making, and has a special duty of disclaiming false knowledge, nevertheless, it cannot replace the general public in the definition of acceptable risk (Fischhoff, 2007).

For all the above reasons, a number of steps that may seem relevant to science at a first glance have not been undertaken in the risk appraisal procedure, although the possibilities have been envisaged. Among them, no threshold for risk appraisal has been set. This step would have been a subjective normative approach based on personal judgement. Moreover, it would have come on top of the main (but inevitable) normative approach in ecological risk assessment, which has been the determination of reference conditions. However, reference conditions were, at least in the case of eutrophication, legitimated by the existing legislative framework, namely the Water Framework Directive requirement of an approach for the definition of pristine undisturbed environment, and a general agreement of the broader scientific community in defining most parts of the North Sea coastal waters in the early 90s as eutrophicated.

Although originally planned, there was no possibility to verify what level of safety (ecological risk) society is willing to achieve (accept) under different alternative options for offshore wind development scenarios (see Nunneri, 2006). In the context of further research including more deliberative approaches, the potential of the aggregated indicator ecological risk for facilitating communication and participation in decision-making can be tested by means of dialogues with experts and/or the general public.

4.4 Integrated management, governance and participation

When dealing with environmental management, it can be found that 'this field is not short of good intentions and legal regulations' (Parsons, 1995, page 516), and the examined case studies are no exception. However, implementation and enforcement are usually insufficient or unsuccessful. In order to help designing successful policies, science for integrated management needs not only to be based on integrated models, but also to involve decision-makers and other stakeholders from the early stages, e.g. problem formulation, identification of system boundaries and determination of policy objectives and evaluation criteria (Turner, 2000). In the previous section the issue of legitimacy of risk standards and risk tolerance levels has been briefly discussed. In sections 3.1.3 and 3.2.1 it has been shown that power structures and policy networks may play a considerable role in pushing or hindering development and implementation of policies: stakeholders and the general public can shape, support or hinder management strategies through their beliefs, needs and commitment. Good governance is based on legitimation of actions, participation of those affected by decisions and responsiveness, transparency and accountability of government, the private sector and civil society to the public (Edgar et al., 2006). The DPSIR approach can be used as guidance for delivering scientific tools and concepts for integrated management of coastal seas, under the condition that the social and participatory component of the analysis is not simply ignored or replaced by 'hard' science.

A biogeophysical unit from the perspective of the natural scientist can be fragmented (depending on its extension and history) in manifold political, economic and cultural realities. This implies a varied range of values and perceptions, influencing both risk acceptance and willingness to pay for and commit in management strategies. For this reason, involvement of the main institution, private stakeholders and the general public should aim at ensuring that the provided information enables decision-making to be legitimate, democratic and transparent. There are major advantages related to good governance including –in the optimal case– full participation. Stakeholders may be depositary of knowledge components not available for the scientists (e.g. historical knowledge, insights about existing policy networks, power and communication structures), which makes participation a valuable tool to enrich analysis, especially if deployed since the very beginning of the study. Moreover, when discussing about possible management strategies and the criteria for choosing among them (based on cost, benefit and risk sharing) participation will allow to handle conflicts and consider compensation measures for those most negatively affected. The consideration of multiple perspectives is fundamental if management is understood as a learning process towards sustainable use of coastal resources (Turner, 2000).

Within any science for policy, the role of powerful actors, lobbies and the general public needs to be understood in order to allow decision-makers to design successful regulatory instruments, and undertake negotiation strategies at an early stage. As it has been shown in this study, participatory research aiming at scoping the socio-economic aspects related to perceptions of stakeholders and the general public can be accommodated within the DPSIR framework. However, it is responsibility of decision-makers, to accept the findings of participatory research and take them into account, by transparently reporting the reasons behind their choices.

5 Conclusions

In this study two environmental issues in the North Sea have been analysed along the DPSIR scoping framework. The two core aims were to test the applicability of the DPSIR approach as a decision support system for decision-making in coastal zones and develop a methodology for including assessment of uncertainties surrounding ecosystem functioning (ecological risk) into the appraisal of alternative actions (see section 1.2). In the following the main conclusions with regard to the validity of the DPSIR as a DSS are reported and the main aspects related to the development of the ecological risk concept are addressed. An outlook for future research needs concludes this chapter.

5.1 DPSIR as decision support system (DSS)

Based on the relatively recent understanding that natural resources and assimilation capacity will not indefinitely support increasing use, there is a pressing demand for science to demonstrate, quantify and predict the effects of human activities on the environment. Decision-makers need to face the challenge of managing finite resources under the expansion of human population and needs. Due to their multiple uses, coastal areas are particular prone to overexploitation of natural resources. Mismanagement leading to overexploitation and breaching of (unknown) thresholds, can ultimately result in losses of welfare or major ecological threats such as ecosystem ‘collapse’. Decision-makers are made accountable for their deliberations and need to be able to explain and justify them on request. Decision-makers need therefore comprehensive and (as far as possible) complete information dealing in an holistic way with different aspects of the issue they need to decide upon (Elliott, 2002). They require a clear understanding of interlinkages, cause-effect relationships, risks and uncertainties as well as socio-economic interests. But they also require to know how people they represent perceive the issues and to take into account the distribution of gain and losses among the affected parties.

The DPSIR approach has been applied under the assumption that it is a valuable instrument for informing decision-makers (i.e. it can be deployed for providing the information mentioned above) for issues in the coastal zone. If ‘science has the role to explain, demonstrate and illustrate to environmental managers, politicians, and workers in other disciplines, the complexity of the system, the linking between components, the knock-on effects of activities and the responses at different levels of the system’ (Elliott, 2002), then the DPSIR allows to structure analysis in order to provide this kind of information.

Moreover, based on DPSIR analysis it can be indicated at what point of the causal chain intervention could be realised at all or most effectively (Brouwer et al., 2003), by addressing spatial scale issues and multiple stakeholder perceptions interests and knowledge (e.g. sections 3.1.1 and 3.1.3).

Within this study, the main obstacle to the application of the DPSIR scoping framework to the issue of offshore wind farm has been the lack of fully quantified and predictive models for assessing the multiple effects connected with the realisation and operation of offshore wind farms. This is, however, not a limit of the DPSIR approach in itself. Rather, the analysis along the DPSIR has allowed to highlight the need of more detailed knowledge and –above all– the needed interlinkages of single sectoral studies and models. Within the Zukunft-Küste- Coastal Future project ongoing research is addressing the issue of joining different sectoral appraisal of ecological impacts (e.g. modelling effects of wind parks on fishes, birds, mammals) into a more comprehensive integrated assessment (Garthe, Opiz and Lenhart, pers. comm.). However, given that most of the modelling tools are still in development, management in the case of such an emerging issue will need to rely in the early phase much more on qualitative relationships and expert judgement. The precautionary approach adopted by most of the North Sea states, which started offshore wind sector development by consenting small installations, much recognises this aspect.

Based on the experience gathered during the research projects that have framed this study, some of the main advantages of the DPSIR approach in the light of its role as decision support system can be highlighted. The use of the DPSIR scoping framework:

- allows structuring a complex problem along a simplified structure which can facilitate the scoping of an emerging issue (e.g. offshore) but can also efficiently include complexity at more mature stages of knowledge (e.g. eutrophication);
- through the linking of complementary information from different disciplines, offers an holistic overview of the system to managers, by linking economic and social considerations with evaluation of environmental changes and risks (see schematic approaches in figures 1.1, 3.18 and 3.30 as examples for this study);
- deals with an issue and its causes and effects, therefore taking implicitly into account different scales (i.e. it can show decision-makers that a local impact has causes at the national, international or global scale, thus pointing out possible intervention strategies or limitations, as discussed in section 4.1.1);
- allows the application of diverse established analytical tools and modelling, thus offering the possibility to organise already available knowledge and analytical instruments into more informative integrated information (for the case study eutrophication see as an example figure 3.5);
- provides the structure for early recognition of potential overlapping of different disciplines and models, thereby allowing scientists with different backgrounds to carry on hand in hand research (e.g. scheme reported in figure 3.6);
- allows to structure integrated and transdisciplinary research thus achieving a more complex level of analysis, which would not be obtainable by simply aggregating sectoral results at the end of the studies (integrated assessment is more than the sum of sectoral assessments, e.g. model integration for the Elbe case study, see page 49);

- provides the setting for scoping future development patterns based on assumptions (scenarios) and the application of quantitative models, thus allowing to test robust management strategies and development alternatives (for both case-studies scenarios have been the starting point of the analysis, see chapter 3);
- if transparently applied, allows indicating the limitation of models and available scientific knowledge, thus highlighting the need for further research and uncertainties that may affect alternative appraisal;
- allows scientists and decision-makers a broader and mutual understanding (has a capacity-building effect);
- represents a semantic field for communication and thereby opens a way for interregional and international management of transboundary issues; in the case of eutrophication the same analytical approach has been applied to more than one catchment in Europe, thus allowing to aggregate information at an international level (e.g. for the southern North Sea) as a basis for negotiation and synergetic action (e.g. section 3.1.2);
- can be made understandable for the general public and thereby allows participation, thus facilitating 'good' governance;
- finally, it is compatible with other integrated approaches such as the one used by the Millennium Assessment, the SWOT approach, Gottret and White's institutional approach or Bossel's systemic framework (Stoll-Kleemann et al., 2006).

Some of the DPSIR strengths are, however, at the same time its limitations. These need to be explicitly addressed if the DPSIR approach has to be applied successfully for informing decision-making. The main limitations of the DPSIR scoping framework are:

- its simple and linear structure, if blindly followed, may result in oversimplified or incomplete assessment (ignorance of relevant aspects, e.g. multiple causalities, multiple effects);
- being a scoping framework and not a computational model, the DPSIR does not provide, in itself, the analytical tools for carrying out single steps of analysis; The choice and development of such instruments are –still– responsibility of the scientists involved (e.g. including participation, choosing geographic and temporal scale, choosing analytical tools);
- in the case of an emerging issue, there might be insufficient available knowledge to allow a complete and detailed analysis along the DPSIR approach; in those cases sectoral analysis need to be completed first, or at least during the integrated analysis (which is usually a challenging and in some case formidable task): in the case of the offshore-wind case study, limited knowledge and modelling instruments have not allowed to reach a higher level of complexity including multiple phases of development and multiple effects;
- the successful application of the DPSIR (and any other integrated assessment framework) requires tight team-work to realise a common understanding for organising and linking sectoral science in such a way that the outcome provides more information than single sectoral assessments; this iterative and delicate task is usually not realisable within a three-year time, which is the average time of project financing.

Scientists –and participating stakeholders– need to put their effort into connecting analytical tools along the DPSIR conceptual framework, rather than thinking of applying it to some issue as a black-box model. In other words, application of the DPSIR approach does not, alone, guarantee 'good science', but it can enable it through a logical sequence of analytical stages.

5.2 Ecological risk: where do we go from here?

The concept of 'ecological risk' has been developed to address a gap existing between the economic evaluation of ecosystem services and their essential role as life-support system, which should be given an infinitely high value well beyond the marginal welfare changes that economic valuation appraises (see sections 1.1.1 and 2.4). Ecological risk has been defined as an estimate of the risk of major disruption in the provision of ecosystem services expected/desired by human societies (section 2.3). In this study a first step towards ecological risk assessment has been carried out by assessing the ecological risk associated with eutrophication scenarios and offshore wind farm construction scenarios.

The concept has been operationalised during the elaboration within the case studies: after some trials (see section 4.2.1, page 138) ecological risk has been assessed by applying equation 2.3. This equation takes as input the values of integrity indicators normalised by equation 2.2 (page 17). The used approach foresees two linear elaboration of the indicator values, the normalisation procedure (eq. 2.2) and the risk assessment (eq. 2.3).

Ecosystem behaviour is characterised by non-linear patterns. However, in the first approximation the idea has been that such non-linearity is already included in the indicator values. Nevertheless, it is possible to assess ecological risk through non-linear procedures, as it is envisaged in section 4.2.1 (page 138). Another key-point for the assessment of ecological risk, is that of adopting reference conditions which are as broadly as possible accepted as 'optimal' or 'undisturbed state'. The most sensitive point in the assessment of ecological risk is the choice of reference conditions. From a descriptive point of view, the choice of reference conditions is tightly related to integrity concepts and indicator choice and values. The incomplete understanding of the ecosystem complexity does not allow defining reference conditions that represent the ecosystem in its optimal state (maximum integrity), i.e. to consider 'optimal' reference values for integrity indicators. A way to choose reference situations can be that of taking into consideration existing legislation and agreements[1], or to assess them within a participatory approach.

As discussed in section 4.3, the assessment of acceptable risk levels should be based on participation and not be the task of a single scientist nor of the scientific community alone.

The aggregated indicator ecological risk has been assessed based on ERSEM model parameters as indicators for ecological integrity processes (supporting services). In this sense the assessment has been limited from the beginning to available ERSEM parameters (reported in table 2.1, page 16). As the ERSEM model has originally been developed for eutrophication studies, those indicators are, in some cases, not sensitive enough to changes –e.g. diatoms/non diatoms ratio in the case of offshore wind construction effects– or may over react –e.g. to high nutrient decrease in the case of eutrophication pristine conditions (see section 4.2.2). As a general rule, being the ecological risk indicator an integrated indicator, i.e. summarising in one value a number

[1] in this context, existing legislation is assumed to reflect the perceptions and needs of the 'majority' and thereby used as a 'proxy for general perceptions'

of variables, it should be assessed on a broader set of parameters for each integrity process, i.e. information (in form of genetic variability, e.g. biodiversity), energy and material substrate availability (Barkmann 2000,Barkmann 2002, page 251).

This is especially the case if ecological risk should be used for filling the gap between TEV ans TSV (section 1.1.1, see figure 1.2), i.e. in order to assess the overall functioning of the ecosystem, thus offering additional information to the economic appraisal of marginal ecosystem changes. The main aspects of ecological self-organisation have been indicated up to now based on a few ERSEM parameters. However, the processes and indicators selected for assessing ecosystem thermodynamic efficiency (e.g. exergy capture, nutrient cycling, storage in the sediment as well as exchange in the water column) and complexity (heterogeneity), are not the only possible for appreciating the two mentioned key-aspects. An improvement would be the incorporation of indicators from various models having different focuses. A broader indication based on different methods (e.g. different modelling tools, expert judgement) and including different aspects of each 'self-organisation pillar' would 'stabilise' the indication of integrity and thereby that of ecological risk. For instance heterogeneity indication could include biological diversity of species belonging to higher food-web levels (e.g. benthal organisms, fish, mammals and birds); in addition to this, as abiotic diversity can enhance biological diversity by providing gradients and niche for its development, some indication of abiotic diversity (e.g. abiotic features such as types of substrate) should be included in the indication of heterogeneity.
Further enlargement of the indication pool could include appraisals of other processes and aspects of energy use (e.g. entropy production), complexity (e.g. species abundance divided by functional groups), storage of energy and matter (e.g. carbon or energy storage in biomass) and efficiency (e.g. respiration losses per biomass unit). The inclusion of different parameters originating from different analytical processes for indicating the core aspects of self-organisation would considerably stabilise indication, thereby compensating for limitations of single specific modelling tools. The assessment of ecological risk based on a wider range of processes and indicators would allow assessing current organisation and complexity of the ecosystem, and thereby ecological risk, independently of any issue. The highly integrative nature of this analytical step, however, will require tight cooperation in order to integrate available modelling tools and existing knowledge.
Although the need of assessing ecological risk based on a larger number of indicators is recognised, ecological risk is not immune from the dilemma affecting any other indicator-based approach, namely that of keeping the number of selected indicators low enough to allow operationalisation and overview and high enough for representing complexity.

Through its application within the selected case studies some main advantages and disadvantages of the ecological risk concept have been highlighted. Among the strengths of ecological risk are:

- it addresses uncertainties otherwise ignored in appraisal of alternatives for many environmental issues, thus offering additional information when dealing with choice among alternative actions (trade-off assessment);
- it can account for changes with respect to selected reference situations, thus representing minimal to large changes, which have both –under uncertainty– the potential to lead to threshold breaching;

- it ‘integrates’ different indicator values into a single aggregated indicator, which should in no way replace detailed analysis, but can be used for a general overview of the ecological state and thereby for broader stakeholder negotiation about acceptable levels of risk;
- it can be used for assessing cumulative risks, provided that modelling tools are available to underpin the assessment (i.e. cumulative risk cannot be extrapolated from risks associated with single issues, see section 4.2.3).

Through its application, however, also some weaknesses have been highlighted:

- ecological risk assessment depends on the choice of reference conditions (i.e. ecological risk is always relative to the selected reference state), there is, in this sense, no absolute value of risk, for which absolute reference conditions would be needed;
- ecological risk assessment includes a ‘subjective’ normative aspect, which is the choice of reference conditions; this choice should be based in the future upon broader consensus (e.g starting by the scientific community);
- the linearity of the used approach considers every marginal change as having the same relevance for the overall risk appraisal; it could be argued that it would be reasonable to assume that risk increases the more the indicator values are different from those of the reference state, thus applying a non-linear approach;
- ecological risk has not yet been tested with regard to its communicability to decision-makers and stakeholders.

Some possible future uses of ecological risk could foresee its application in communication with decision-makers, stakeholders and the public, as well as its application within elicitation of state preferences for selected alternative management strategies (see section 4.3).

5.3 Outlook

In this study relevant information to support decision-making has been gathered in the light of the DPSIR approach and augmented by ecological risk considerations.

When multiple interests are at play, ‘good’ governance is the main challenge for management, i.e. the management process should be participatory, transparent and legitimate. It can be argued that participation in decision-making in the North Sea countries is still not fully inclusive, nor spatially integrated (e.g. sections 3.1.3 and 4.1.1). A more inclusive approach for the North Sea management should include not only the coastal countries, but also inland countries belonging to the North Sea catchment (see figure 2.4, page 22), and involve stakeholders since the early stages of decision-making (i.e. definition of the problem, elaboration of alternative solutions and criteria for choosing optimal strategies). The application of the DPSIR scoping framework can help in structuring future research to support more participatory decision-making and ‘good governance’. The application of the DPSIR approach would enable decision-making to consider biogeophysical units and thereby to design ‘ad hoc’ management measures, which take into account efficiency as well as distribution of economic burdens (for the countries of the North Sea catchment as a whole). In this context, the concept of ecological risk can play a role in communicating the uncertainties related to different options, when assessing impacts and trade-offs of alternatives. For instance, ecological risk could be used to underpin the policy-process of establishing thresholds (e.g. the OSPAR targets and 'acceptable' values for quality indicators) by showing what level of risk would be associated with those thresholds, i.e. ecological risk appraisals could be used during the negotiation procedure for the setting of standards. In the context of further environmental research, social preferences and values with regard to risk acceptance and trade-offs of alternatives, may also be explored through elicitation of willingness to pay (WTP) or through choice experiment studies. However, the ecological risk concept should be further developed previous to its deployment in 'real' decision-making.

The definition and operationalisation of the ecological risk concept presented in this study represent a first approach to a complex and sensitive issue, that of appraising potential disruption of essential ecosystem services. In this study ecological risk has been assessed based on available modelling tools (ERSEM) and ecosystem integrity indicators (ERSEM output parameters). Further methodological research is needed in order to:

1. assess ecological risk based on a broader basis, i.e. including a higher number of indicators for each of the self-organisation pillars (information, energy and material substrate availability);
2. test the application of non-linear approaches for both the normalisation of indicator values and the ecological risk assessment; and
3. set a broader basis for the normative definition of 'reference conditions' (e.g. through participatory approaches).

Assessment of ecological risk based on a broader pool of indicators and modelling tools would allow compensating for single model limitations, thereby potentially enabling ecosystem functioning independently of any issue. A sensitivity test of risk outcome under different non-linear approaches would consent to determine a risk-zone related to a given action, thus offering a more solid aid to decision-making. The normative definition of 'reference conditions' can be achieved on a broader basis after consultation with experts, decision-makers and practitioners, whereas risk acceptance and thereby the thresholds for –say– minimum, medium and maximum risk, can (and should) be assessed in the framework of broader participation. Choice experiments involving selected stakeholders or the general public could be used for eliciting willingness to pay for risk prevention or risk acceptance under different use-scenarios for the North Sea marine environment.

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