The Official Journal of the International Association for Landscape Ecology, Chapter Germany (IALE-D)

Landscape Online

International Peer-Review Free-Access Online

Received: 11.06.2007 Received in revision: 26.02.2008 Accepted: 30.03.2008 Published: 10.07.2008

Nunneri, C., Lenhart, H.-J., Burkhard, B., Colijn, F., Müller, F. & W. Windhorst 2008
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Landscape Online 5, 1-20. **DOI:10.3097/LO.200805**

The use of *'ecological risk'* for assessing effects of human activities: an example including eutrophication and offshore wind farm construction in the North Sea

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Abstract

This paper takes the move from the uncertainty surrounding ecosystem thresholds and addresses the issue of ecosystem-state assessment by means of ecological integrity indicators and '*ecological risk*'. The concept of '*ecological risk*' gives a measure of the likelihood of ecosystem failure to provide the level of natural ecological goods and services expected/desired by human societies. As a consequence of human pressures (use of resources and discharge into the environment), ecosystem thresholds can be breached thus resulting in major threats to human health, safety and well-being. In this study we apply the concept of '*ecological risk*' to two case-studies in the German exclusive economic zone: eutrophication and construction of offshore wind farms. The effects of different future scenarios for single-uses upon ecosystem integrity are analysed as well as the effects of one combined scenario. We conclude that in the short term construction of offshore wind farms can influence some processes to a much larger degree than eutrophication, however, combined impacts deriving from eutrophication and offshore wind farm construction need a more detailed analysis. Due to non-linear ecosystem processes, effects of combined or multiple uses of marine resources in terms of '*ecological risk*', cannot be extrapolated from single-use scenarios.

Keywords

marine ecosystem, ecosystem integrity, ecosystem services, ecological indicators

1 Introduction

Natural catastrophes, such as hurricanes and flooding, tsunamis and land-slides are increasingly reported in the news. While the coming into existence of such natural events can not, or only partly (for example by combating climate change) be influenced by human action, the wide range of damages possibly brought about by those events, such as disruption of life-supporting processes and infrastructures, is often affected by human decisions and actions.

There is a large amount of literature dealing with interactions of social and ecological systems and the challenge of managing human action in such a way that allows avoiding major ecological threats such as ecosystem ‘collapse’ (among others Perrings & Pearce 1994, Scheffer et al. 2001, Folke et al. 2002, Beaumont et al. 2007). Anthropogenic change and simplification of functions and structures of the ecosystem may ultimately result in reduction of ecosystems integrity (Barkmann & Windhorst 2000, Burkhard & Müller 2005) and thereby increase ecosystem vulnerability (see Aven 2007), i.e. how prone to damage the system is. This, in turn, affects ecosystem provision of goods and services (Millennium Ecosystem Assessment 2005, De Groot 1992), upon which society depends (Haller 1990). Human actions such as increased use of ecosystem services and resources (e.g. waste and chemical discharge, coastal squeeze) reduces the ability of ecosystems to cope with unexpected changes (i.e. maintaining life-support processes) and thereby menaces human survival. The ability and speed of ecosystem recovering can play a major role in guaranteeing environmental security.

Due to numerous uncertainties surrounding the detailed knowledge of interwoven ecosystem processes and their cause-effect relationships, it is not possible to exactly determine to what extent the ecosystem can bear increasing anthropogenic pressures (resource exploitation and material discharge). The most obvious way to reduce the vulnerability of the system would seem that of ‘removing’ human pressures and letting the system recover its ability to adjust to ‘extraordinary’ events. In this context, it is necessary to provide some ‘coordinate system’ for judging the changes related to (alternative) measures intended to tackle issues connected with ecosystem uses and impacts. Measuring the deviation of the actual “ecosystem state” and the expected changes with respect to some ‘pristine’ (or other reference) conditions is of primary relevance, as there is no guarantee that removing human pressure will lead to ecosystem recovery (i.e. that its state will adjust to conditions previous to human intervention, e.g. Hughes et al. 2005, Walker et al. 2006).

Given the uncertainties surrounding ecosystem complexity and the impossibility to determine critical thresholds, as briefly reported previously, this study reports a method for addressing the evaluation of ecosystem changes brought about by human activities. The focus is set upon the magnitude of ecosystem changes, which is expressed in terms of “*ecological risk*” (Nunneri et al. 2007). In this paper we apply the ‘*ecological risk*‘ methodology for:

1. appraising the effects of different single human activities (ecosystem uses) upon the North Sea ecosystem with respect to some reference situation; and
2. testing whether the concept of ‘*ecological risk*‘ can be applied for assessing cumulative or combined effects of different uses. In particular, the hypothesis to be tested is whether ‘*ecological risk*‘ assessed for different single issues can be simply added up (or subtracted) in order to assess cumulative impacts resulting from multiple uses.

This study focuses on methodological issues; two examples of human activities in the North Sea are taken as case-studies, namely eutrophication and offshore wind energy generation. The case-studies are used in order to test the applicability of the ‘*ecological risk*‘ concept. Those different activities are chosen because they utilise different ecosystem services: nutrient discharge, resulting in eutrophication, is an example of use of ecosystem assimilation capacity, while offshore wind power generation is an example of resource exploitation. Moreover, eutrophication can be considered an ‘historical’ issue in the North Sea (for a review see De

 ISSN 1865-1542

Jong 2006), while the construction of offshore wind farms is a new use of the marine area, aiming at tackling various issues, such as greenhouse gases emission reduction (e.g. BMU 2007, Köller et al. 2006). This study assesses the changes in the *'ecological risk'* related to selected nutrient reduction and offshore wind farm construction scenarios. *'Ecological risk'* is assessed for each single case study first as individual event and then combined effects are considered. The paper is structured as follows: section 2 explains the used methodology, including some ecosystem theory background as well as a description of selected indicators based on ERSEM modelling and the procedure for *'ecological risk'* assessment; section 3 illustrates the analysis first for each single issue and then for one scenario resulting from the combination of one eutrophication and one offshore wind scenario; finally section 4 reports the main conclusions and perspectives.

2 Methodology

Aim of this study is to test the *'ecological risk'* methodology for comparing the impacts deriving from two selected human activities in the North Sea. The first step of analysis consists in assessing different levels of use-intensities (pressures) upon the ecosystem by means of scenarios. The ERSEM model (Baretta et al. 1995) is used for modelling different scenarios. Once the scenarios have been assessed in terms of the changes that they cause in the coastal areas, those changes enter the ecosystem model ERSEM, separately first (Hofmann et al. 2005, Nunneri et al. 2007) and finally in a combined form (nutrient reduction and increased SPM concentration). Different "integrity indicators" are derived from ERSEM parameters and used for assessing *'ecological risk'*. Within this study we refer to overall functioning of the ecosystem and not to specific aspects, such as single species protection. In the following the concept of *'ecological risk'* is introduced first, and then the methodological steps for the analysis are described: the assessed scenarios for eutrophication and offshore wind farm construction, ERSEM modelling and integrity indicators and finally the *'ecological risk'* assessment procedure.

2.1 From ecosystem integrity to the concept of 'ecological risk'

There is a broad pool of definitions of "ecosystem integrity", "self-organising capacity" and "ecosystem health" (e.g. Woodley et al. 1993, Westra & Lemons 1995, Crabbé et al. 2000, Barkmann 2002; Rapport 2003). Our interpretation of integrity is based on Barkmann et al. (2001). Integrity is essentially self-organisation capacity of ecosystems, i.e. their ability to create structures and gradients during ecosystem development. Development is associated with energy conversion processes where "high quality" energy (exergy, see Jørgensen 2000) is transformed into non-convertible energy fractions (entropy) or stored within biomass, detritus or information The term "exergy" defines potential of doing work, exergy appears in physics as energy, matter and information; (ecosystem) structures are created through exergy degradation (Kay 2000, Jørgensen 2000).

In practice, „an ecosystem has integrity if it retains its complexity and capacity for self-organization and sufficient diversity, within its structures and functions, to maintain ecosystem self-organizing complexity through time“ (Kay 1993). Based on ecosystem theory, energy and matter balances are key-variables for maintaining ecosystem diversity, which is essential for coping with changes (resilience, e.g. Beaumont et al., 2007). Baumann (2001) mentions some key-variables for ecosystem self-organisation that can be further subdivided into their components (Baumann 2001). Among them, exergy capture, cycling of elements, storage capacity, heterogeneity (diversity) and matter losses are important elements of ecosystem functions. Energy balance in coastal zones needs to take into account flows related to organic and/or inorganic nutrient inputs from the atmosphere or from adjacent regions as well as incoming solar radiation.

ISSN 1865-1542

The availability of material substrate (diversity of abiotic structures) and limiting nutrients for ecosystem development depends on its storage capacity and is essential as life-supporting function and thereby for guaranteeing species diversity. The storage capacity (in the sediment) and the matter balance in the water columns are thereby essential processes governing the exchange rate of the different pools, and the possibility of temporarily dampening or buffering external inputs; (re) cycling of limiting substances, especially nutrients plays in this context an important role. Matter losses reduce the capacity of primary and secondary production, which are essential life-supporting processes. One of the aspects of organisation and complexity of ecosystems is their biotic diversity (organism diversity and genetic diversity), which depends on abiotic diversity. 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 is a measure of ecosystem efficiency (Baumann 2001).

Those mentioned above are essential ecosystem processes at the basis of ecosystem provision of services. Fundamental services are especially regulation of indispensable ecological processes and life-support systems (e.g. provision of clean water and climate regulation). For this study, we define *'ecological risk'* as: a measure of the likelihood of ecosystem failure to provide the level of natural ecological goods and services expected/ desired by human societies (Nunneri et al. 2007). *'Ecological risk'* addresses the uncertainty in determining when unknown thresholds could be breached thus disrupting the provision of ecosystem functions that humankind generally takes for granted as boundary conditions for human existence (De Groot 1992, Millennium Ecosystem Assessment 2005). Given the uncertainties surrounding ecosystem knowledge and the determination of thresholds, it is impossible to determine the risk of ecosystem 'collapse' in a classical way based on probabilities and monetary estimation of possible damage (Potthast 2004, Breckling & Potthast 2004). *'Ecological risk'* is assessed for this study on the basis of ecosystem integrity and supporting services are taken as indicators quantified by modelling (ERSEM parameters, see further).

2.2 Scenarios

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 EEA 2001, van Notten et al. 2003). Scenarios used in this study are assessed in terms of 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 model input variables. The model then allows assessing the impact in terms of ecosystem integrity indicators. 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 (for more details see Nunneri et al. 2007):

- a low reduction scenario (Low-Red), where nutrient inputs into the North Sea are reduced of 20% with respect to 1995 levels;
- a medium reduction scenario (Med-Red), where nutrient inputs into the North Sea are reduced of 40% with respect to 1995 levels; and
- a high reduction scenario (High-Red), where nutrient inputs into the North Sea are reduced of 60% with respect to 1995 levels.

The time horizon of those scenarios is 2025.

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 (for a detailed description see Burkhard 2006):

- 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

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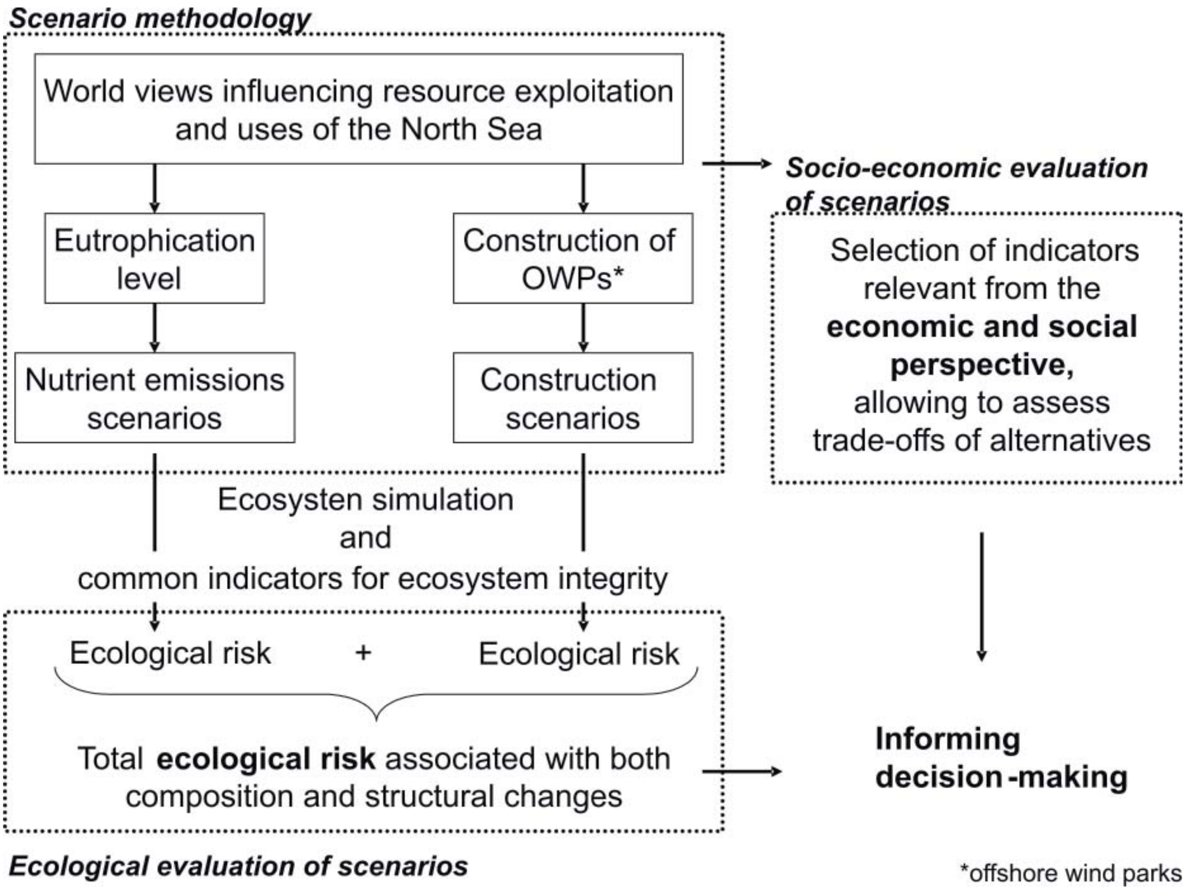

Figure 1. Theoretical approach for combining the *'ecological risk'* resulting from different issues as described through different scenarios. The assessment of risk should include the appraisal of scenario socio-economic components (in terms of costs and benefits) in order to allow the identification of trade-offs and thereby the determination of acceptable risk.

- under the North Sea as energy farm (B1) scenario, ca. 25000 MW installed in 2030 and 90000 MW in 2055.

It is worth to stress that, while eutrophication scenarios represent changes in the long run, the construction scenarios are assessed on a year by year basis, in which every year construction takes place in different areas depending on what project is assumed to be realised. The scenarios dealing with the two considered issues have been developed within different research projects and need therefore to be linked among each other in order to assess their 'combined' (cumulative) effects upon the marine ecosystem by means of ecosystem modelling. Figure 1 shows a theoretical approach for the assessment of cumulative effects, which is complemented –in the optimal case – by socio-economic appraisal and trade-off of options. The storylines formulated to underpin the scenarios describe society preferences, values and priorities playing a major role in socio-economic development. According to the prevailing socio-economic attitudes, 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: governance (globalisation degree) and societal values (individualism and consumerism vs. community and conservation). This procedure excludes some combinations. Figure 2 shows the scenarios which describe comparable world-views, plotted against two axes: globalisation (regional vs. global) and social values (individualism vs. community).

Based on the pattern shown in fig. 2, three 'eutrophication reduction and offshore wind construction' scenarios can be considered: (1) low nutrient reduction

ISSN 1865-1542

being associated with low offshore installed capacity (LowRed-E1); (2) middle nutrient reduction being associated with high installed capacity (MedRed-B1) and (3) high nutrient reduction being associated with middle installed capacity (HighRed-A2). In this paper we choose to analyse the effects of the second combined scenario (MedRed-B1) in order to compare it to the effects obtained for each 'single issue' scenario. Aim of this analysis is to test whether it is possible to assess *'ecological risk'* of the combined scenario from the ones of single scenarios (i.e. without the aid of modelling).

2.3 Ecosystem modelling, key indicators and 'ecological risk' assessment

The defined scenarios are transposed into the ERSEM model in form of changes of river nutrient loads for the case-study eutrophication, or in suspended matter concentration (SPM) for the case-study offshore wind farms. Those changes are analysed compared with a standard run, which simulates the year 1995 as realistic as possible. The simulation applies the scenarios, but leaves the rest of the forcing exactly as in the standard run.

The ecosystem model ERSEM (European Regional Seas Ecosystem Model) is used for assessing ecosystem changes resulting from each scenario. ERSEM describes the North Sea via the dynamic interaction between physical, chemical and biological processes (Baretta et al. 1995, Lenhart 2001). The model represents the biological and biogeochemical processes in a unique complexity (Moll & Radach 2003) within the pelagic and benthic system. In the box model version used for this study (see Fig. 3) the physical features are covered in a reduced but nevertheless realistic form (Lenhart & Pohlmann 1997). ERSEM carries the macronutrients, nitrate, ammonium, phosphate, and silicate as well as carbon as state variables. The ecosystem is represented by three groups within the trophic net: phytoplankton as producers, zooplankton and zoobenthos as consumers, and bacteria in both the pelagic and benthic environment as decomposers. In addition, the benthic module in ERSEM is able to simulate aerobic, anaerobic

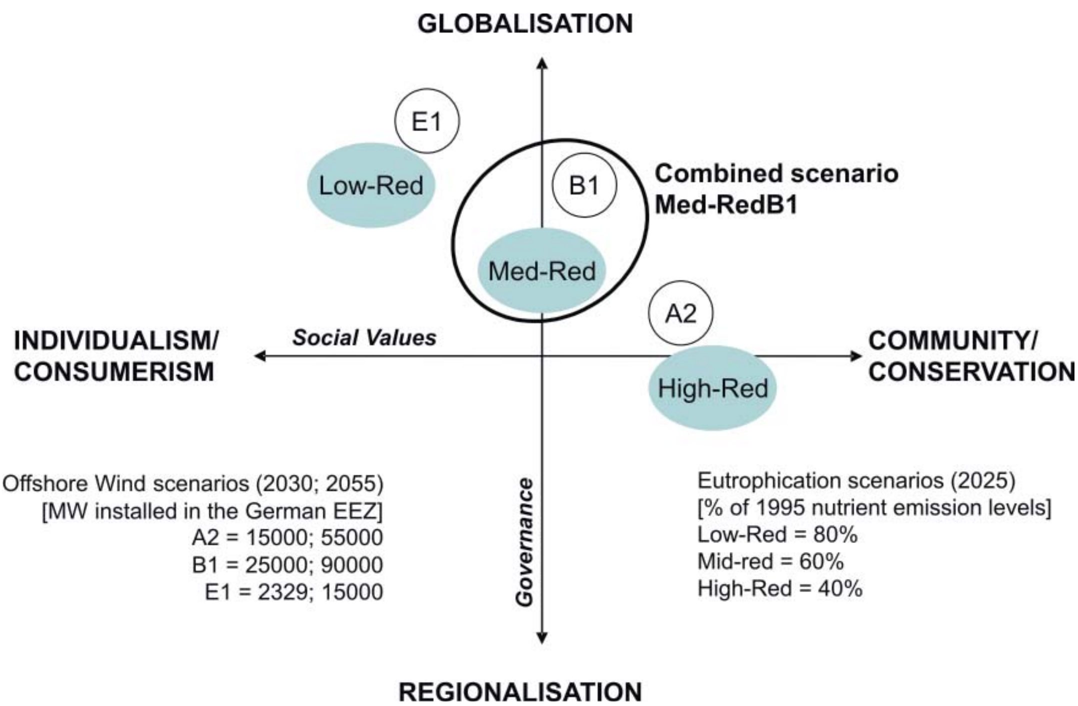

Figure 2. Scenario position with respect to governance and social values in them represented. Based on these aspects, scenarios dealing with different issues can be combined based on the assumptions underlying more general aspects regarding world visions, perceptions and values.

ISSN 1865-1542

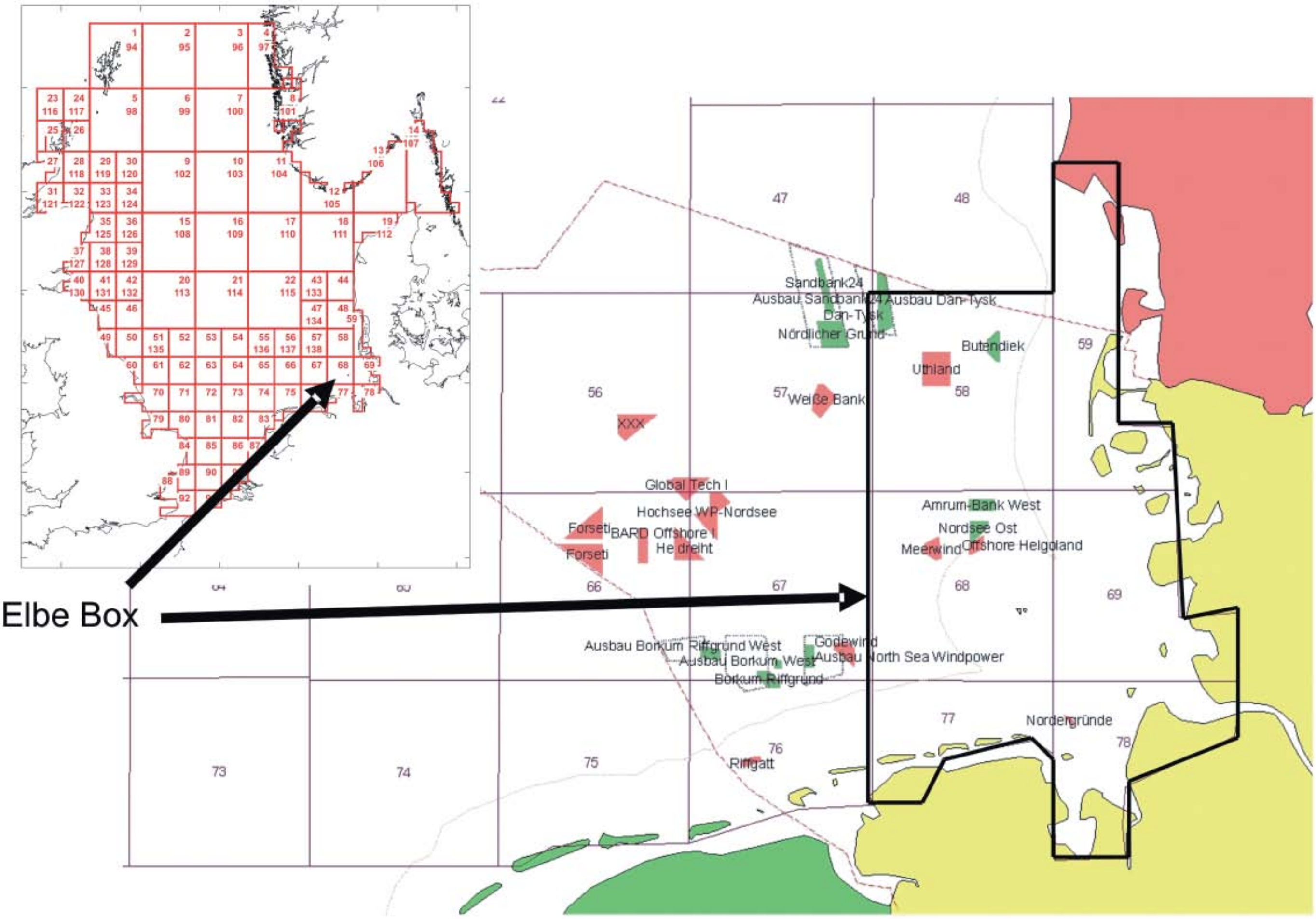

Figure 3. Study area: The Elbe-box is delimited by a bold line. In the German exclusive economic zone (EEZ) the wind farms included in the considered scenarios are shown.

and sulphate reduction conditions within the sediment. The complex benthic module allows representing the slow benthic processes, including buffering effects related to the continuous input of organic loads due to eutrophication. For this reasons ERSEM has been used to simulate reduction scenarios in a number of studies in the North Sea (OSPAR 1998; Lenhart et al. 1997, Lenhart 2001) as well as for the Continental Coastal Shelf region (Heath et al. 2002).

Based on ecosystem self-organisation, intended as the web of relationships and processes that organise the ecological system as a whole, and thereby determine ecosystem integrity and their ‘functioning’ over time (Müller in press), a subset of the parameters provided by the ERSEM model is selected for computing integrity indicators (see table 1). All selected indicators (table 1) refer to supporting services (e.g. production of biomass by photosynthesis, production of atmospheric oxygen, nutrient and matter cycling, soil formation). This choice is motivated by the key-role of supporting services, which are at the basis of the provision of other services (Millennium Ecosystem Assessment 2005) and direct benefits to humankind, such as water quality, waste assimilation or amenity. In the following a brief explanation of the processes and indicators reported in table 1 is given. For a detailed explanation of the used ecosystem integrity indicators see Nunneri (2007).

ISSN 1865-1542

Table 1. ERSEM parameters, their metrics and their elaboration into ecosystem integrity process indicators.

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

2.4 *'Ecological risk'* assessment

In order to assess the influence of the single ecological integrity indicators on the self organising capacity of the ecosystem and the *'ecological risk'*, some elaboration of the indicator values needs to be undertaken. In short, three steps are followed:

1. for each case-study a minimum and a maximum disturbance situation are needed as references in order to be able to express the changes associated with each scenario in relative terms within those two extremes (both in terms of integrity indicator and *'ecological risk'* values);
2. integrity indicator values are linearly transformed (normalised) into relative numbers within the range of values assumed under the selected minimum and maximum disturbance conditions; this allows assessing the relative changes caused by different scenarios against the reference situations (see Windhorst et al. 2005). For the sake of simplicity, we choose to represent normalised indicator values in a new interval comprised between 0 and 100;
3. *'ecological risk'* is assessed with respect to selected reference situations (minimum disturbance and maximum disturbance) representing minimum and maximum risk level (0 and 100); the aggregated indicator *'ecological risk'* is calculated by adding up and averaging the "distances" between the values of integrity indicators in a given state (scenario) and the ones in the reference situation of minimum risk (*'ecological risk'* values range also between 0 and 100). In this process each integrity indicator is given equal weight.

This procedure does not assess risk in terms of probabilities and damage functions. The extremes for the assessment ('minimum' and 'maximum' risk levels) are set based on available knowledge. Those minimums and maximums are usually not absolute, i.e. they can change when considering a different issue. In this sen-

ISSN 1865-1542

se our assessment of risk depends on reference conditions. '*Ecological risk*' offers a perspective for dealing with safe-minimum-standards and threshold-values by assuming, in the first approximation, that the level of risk in a reference situation is acceptable and the risk in another reference situation is not (i.e. social costs outweigh the benefits, Nunneri et al. 2007).

For each case study minimum and maximum disturbance situations have been chosen as references for computing *'ecological risk'*. For the case study eutrophication we considered as references for *'ecological risk'* assessment: (1) the nutrient input levels in 1995 as maximum eutrophication level (maximum risk) and (2) the 'pristine conditions' set at 10% of 1995 nutrient levels as minimum disturbance upon the marine ecosystem (minimum risk). Although eutrophication in the North Sea has reached its maximum during the eighties, the choice of 1995 as a basis year for maximum disturbance in the case study eutrophication was due to available detailed information on riverine nutrient loads by the MONERIS model (see Hofmann et al. 2005). The focus has been set on the realisation of a consistent model setup, so that the forcing for the ERSEM model run was also compiled for that year. This set-up offered the possibility for the representation of scenario reduction levels based on measures for nutrient reductions within the catchment area (Hofmann et al. 2005, Nunneri et al. 2007). The choice of "pristine conditions" as minimum disturbance derives from the assumption that no human nutrient discharge takes place (no human settlement and forest coverage of the catchment area, see Hofmann et al. 2005, Nunneri et al. 2007).

In the case of offshore wind farm construction, the current situation without wind farms (year 2006, set equal to 1995 ERSEM basis year) is assumed to present a minimum risk level, while a whole year construction in a selected area (box 58 of the ERSEM model, see further) offers one of the highest alterations of the indicator values in the system. Due to modelling conditions, however, as the ERSEM model is calibrated for the year 1995, the construction of offshore wind farms is 'superimposed' to the 1995 nutrient level, thus implicitly representing a scenario 'maximum eutrophication level and superimposed offshore wind farm construction (either offshore wind construction scenario B1 or E1)'. It can be easily noticed here that references are (1) case-study dependent and (2) arbitrarily set, i.e. there is no mandatory reason why they need to be chosen in this way and not in another. This opens a wide field about the legitimacy of those references and the need of participation in the definition of broadly accepted environmental targets (see Nunneri 2007), which is, however, beyond the scope of this paper, focussing upon methodological issues.

3 Results

The concept of *'ecological risk'* was firstly applied for evaluating eutrophication reduction scenarios in terms of their costs and risks (see Nunneri et al. 2007) and subsequently for assessing the effects of offshore wind farm construction upon a limited area in the North Sea. The spatial focus of this study is related to the German exclusive economic zone (EEZ), and in particular the area delimited by the ERSEM-model boxes 58, 59, 68, 69, 77, 78 (called in the following 'Elbebox', see Fig. 3).

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, as there are key-differences between the case studies. The results for the eutrophication scenarios reflect a system which has 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. 2007). On the contrary, the offshore wind-farm construction scenarios relate primarily to the construction phase (higher suspended matter concentration) and therefore represent a short-term disturbance. In table 2 an overview of scenario values is reported: minimum and maximum indicator values for each case-study can be compared with (1) the selected reference conditions of minimum and maximum disturbance (used for calculating minimum and maximum) *'ecological risk'* and (2) further refe-

ISSN 1865-1542

Table 2. Overview of minimum and maximum indicator values throughout the scenarios. The minimum and maximum integrity indicator values for each case-study can be compared with (1) the selected references for minimum and maximum disturbance (minimum and maximum risk) as well as (2) available literature data or ERSEM standard modelling runs. Please note that the minimum and maximum values listed in the table are the minimum and maximum found among all scenarios for a single case study (i.e. eutrophication, offshore wind farm construction or the combined scenario) and the values reported in a row may not correspond to a single scenario of the case studies.

		Indicators				
		Net-Primary Produktion [g C m^{-2} y^{-1}]	Yearly Turnover of Winter Nutrients	Sediment storage [mmol N m^{-3} y^{-1}]	Matter bilance [mmol N m^{-3} y^{-1}]	Diatom / Non-Diatom
Eutrophication scenarios	minimum value	234	3.7	-7.4	-47.4	0.37
	maximum value	266	4.6	7.1	-21.7	0.4
	minimum risk reference (pristine)	234	4.6	7.1	-21.7	0.4
	maximum risk reference (1995)	266	3.7	6.3	-47.4	0.37
Offshore wind scenarios	minimum value	214.1	3.1	-3.5	-55.4	0.46
	maximum value	317.6	4.5	10.6	-44.9	0.5
	minimum Risk Reference (2006)	303.7	4.3	6.9	-47.0	0.47
	maximum Risk Reference (12-month construction)	239.1	3.3	-1.8	-55.4	0.5
Combined scenario	minimum value	166.3	2.6	-3.6	-42.6	0.38
	maximum value	252.6	3.9	9.7	-34.1	0.43
All scenarios	absolute minimum	166.3	2.6	-7.4	-55.4	0.37
	absolute maximum	317.6	4.6	10.6	-21.7	0.5
Other Reference values for the considered indicators *		240-261 266 EBM 303 EB	3.1 EB 6.8 GB	6.9 EB	-47 EB	0.33-0.86 0.37EBM 0.47EB

* Data from Joint & 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 through combined modelling from ERSEM and MONERIS (for the riverine nutrient transport, Hofmann et al. 2005) in the Elbe box.

rence values. As it can be seen the selected parameters may vary considerably throughout the scenarios. Especially noticeable is the lowest primary production value (166 g C m^{-2} y^{-1}) occurring in the combined scenario MedRed-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 MedRed 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 primary production for scenario B1 related to offshore construction alone, which shows the lowest value within the B1 scenario (214 g C m^{-2} y^{-1}). The additional effect of nutrient reduction, further reducing primary production, is clearly to be seen in the combined scenario.

The maximum primary production (317 g C m^{-2} y^{-1}) occurs for scenario E1, without any nutrient reduction, 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.

ISSN 1865-1542

4.1 Eutrophication

Scenarios 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. In Fig. 4, ecosystem integrity is indicated through the five main integrity processes (via selected indicators) for the Elbe box. The eutrophicated 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. 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 riverine 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 minimize matter losses (i.e. to increase the quantity of matter retained by the system) are observable. The parameter diatom/non-diatom ratio is difficult to grasp from Fig. 4. Basically the values are spread between 0.4 for the standard run and 0.37 for the pristine condition. This implies that the group of

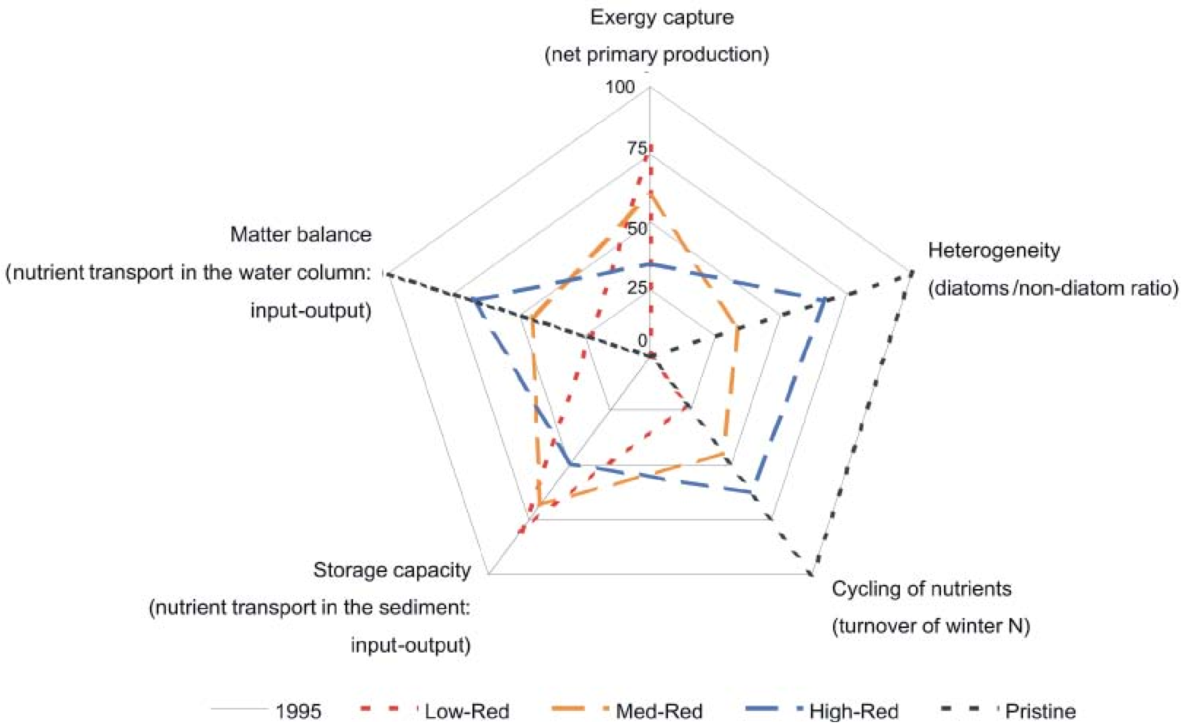

Figure 4. Integrity indicators. Normalised values for the three considered eutrophication scenarios and the two reference conditions of maximum (1995) and minimum risk (pristine). The eutrophicated state (maximum disturbance, 1995) is indicated by the highest primary production and extremely positive N sediment budget (N retained in the sediment). The tendency to reduce primary production as a consequence of reduced N inputs into the system is clearly seen. In the same way, (i) the tendency to increase the diatom/non-diatom ratio (i.e. to increase the trends towards a low nutrient input situation more similar to historical records of non-eutrophicated situations), (ii) to increase the cycling of winter nutrients (the system needs to re-use available nutrients, if the inputs of new nutrients is reduced), (iii) to minimise matter losses (i.e. to increase the quantity of matter retained by the system) and (iv) to retain nutrients in the sediment, are also observable. The sediments represent a buffer for the system, being a sink for overabundant nutrients and a source in scarcity times. The sediment budget (N in the sediment) tends to decrease in pristine conditions (minimum disturbance).

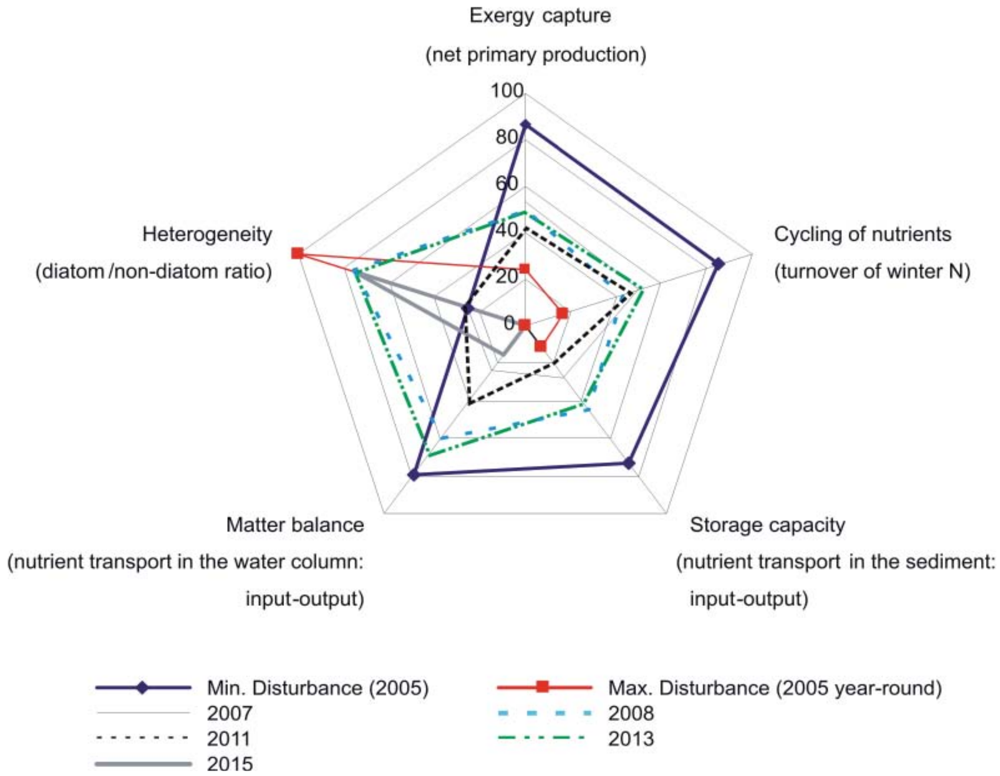

Figure 5. Integrity indicator values for selected years in which offshore wind farm construction takes place in scenario B1 (indicator values are normalised within the value set of both scenarios for offshore wind). Under construction continuing over a 12-month period (maximum disturbance) all indicator values present a decrease if compared with the minimum disturbance situation (2005), with the exception of the diatom/non-diatom ratio (heterogeneity, for more detailed explanation see text).

diatoms is slowly increasing with reduced river inputs, which is in agreement with the documented increased flagellate concentrations in the water due to the enhanced nutrient concentrations (Radach et al. 1990, Radach 1998, Hickel et al. 1993).

An analysis of *'ecological risk'* associated with eutrophication scenarios including the socio-economic aspects is reported in Nunneri et al. (2007). The risk associated with the three scenarios presents 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). In Nunneri et al. (2007) it has been shown that although in the Elbe coastal zone a 'maximum' intervention (a reduction up to 60%) is justified by a considerable decrease in *'ecological risk'*, the implementation costs do not vary linearly with risk decrease. The costs double with respect to the Low-Red scenario for implementing the Mid-Red scenario measures and increase about five times for achieving the High-Red reduction targets. In this context, trade-offs between risk reduction and reduction implementation costs need to be factored into decision-making.

4.2 Offshore wind

Throughout the offshore wind construction scenarios an increase in SPM concentration of 2 g/m3 (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 May 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 turbines was not related to the SPM increase.

ISSN 1865-1542

In figure 5 the integrity indicator values are reported for offshore wind farm construction scenario B1 and selected construction years up to 2015. The patterns recognisable in figure 5 are determined by the fact that construction of wind farms is assumed to take place at different points in time within 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. This is due to increased SPM concentrations within the Elbe box resulting from construction activity. This takes place in the years 2007 (box 58), 2008 (box 68), 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) similarly 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 light limitation to a consistent part of the aggregated Elbe box.

The indication of heterogeneity by the ratio between diatoms and non-diatom phytoplanktonic species is not an optimal indication in this case, as the natural diatom and flagellate bloom succession is extremely altered due to highly reduced light penetration (high SPM concentration) during construction phases. 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-diatom species abundance. The instantaneous local conditions of light and nutrients determine the proportion in the production between both two algae groups (Nunneri 2007).

In figure 6 the *'ecological risk'* assessment for the offshore wind farm construction scenarios 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

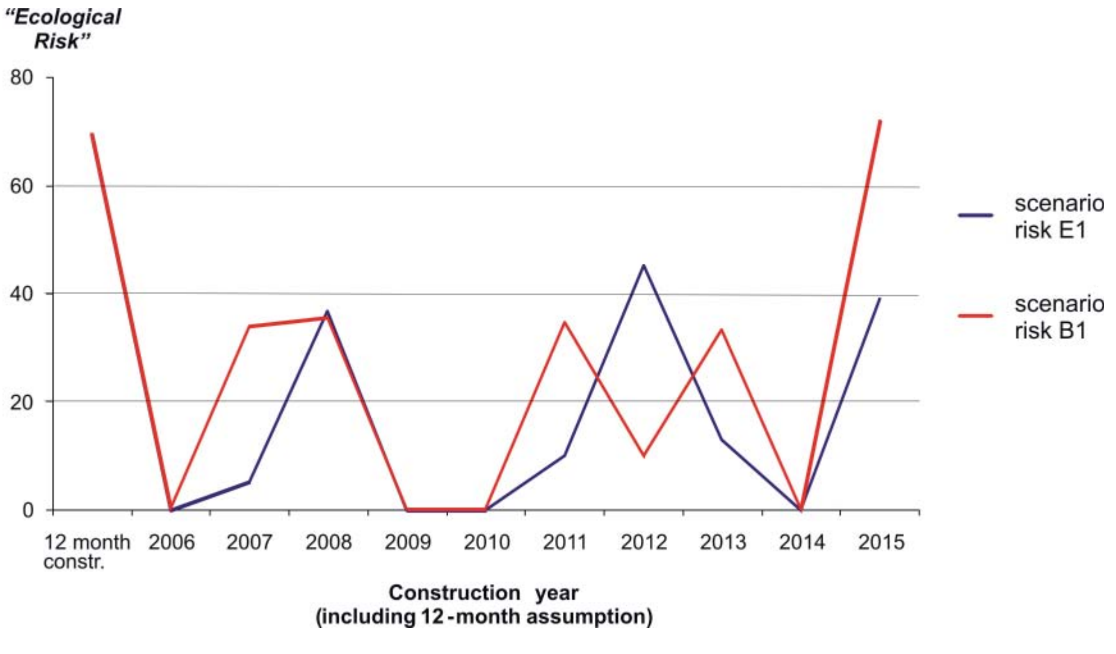

Figure 6. *'Ecological risk'* of offshore wind farm construction scenarios up to the year 2015. Years during which construction takes place present an increase in risk, with respect to the actual situation (2006) with no construction. Maximum risk levels are found in the reference situation, assuming construction to take place for a period of 12 months in box 58, a considerable area in the Elbe Box, and also for scenario B1 in year 2015, where construction takes place in an area covering about 60% of the Elbe Box.

ISSN 1865-1542

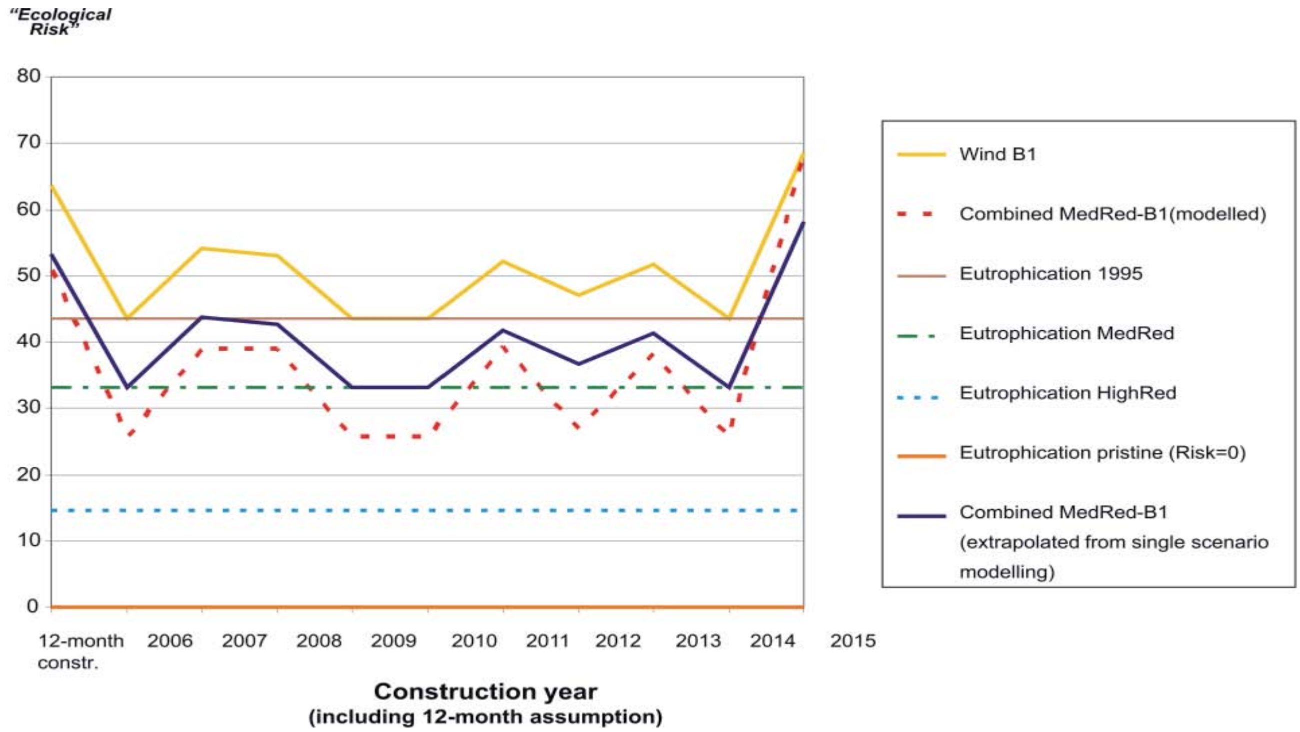

Figure 7. '*Ecological risk*' calculated with respect to minimum and maximum values of indicators throughout all scenarios, but excluding the combined scenario. For explanation see text.

construction in a large part in the Elbe box, covering about 60% of the total considered area. For scenario E1 the maximum risk occurs in year 2012, and is associated with construction in box 68 only.

4.3 Combined scenario

The results for the considered combined scenario joining eutrophication scenario Med-Red and offshore wind construction scenario B1 (MedRed-B1), 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 MedRed-B1.

The aim was that of testing whether the cumulative effects of the combined scenario on ecosystem integrity could be extrapolated from the results previously obtained separately from single offshore wind and eutrophication scenarios. The assumption underlying this test was that *ecological risk* of the combined scenario MedRed-B1 would be obtainable by subtracting the 1995 risk from the risk of scenario B1 (it was implicitly included, being ERSEM calibrated for 1995) and adding to the result the risk of scenario 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 7 the '*ecological risk*' assessment for selected scenarios from the two case-studies, including the combined scenario MedRed-B1 is reported. In this graph '*ecological risk*' has been assessed by normalizing 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

ISSN 1865-1542

of the minimum and maximum as they were assumed to be comprised in the range of the values obtained in the single scenarios. In three cases the values of the combined scenario are clearly lower or higher than the corresponding values under the considered scenarios in the two case studies. This is observable for primary production, which shows minimum values in 2011 (211 g C m^{-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 a reduced 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 (0.43 vs. 0.49 in the combined scenario). The combined scenario MedRed-B1 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 utilized are transported into the neighbouring box and cause an increase of primary production in that box slightly over the level of the standard run. When aggregating boxes with lower and boxes with slightly increased production the latter effect is compensated for. 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 fig. 7), not even in the case of a normalisation procedure carried out including the values of the combined modelled scenario. In general the extrapolated risk was an overestimate of the risk assessed from modelled data.

5 Discussion

In the light of uncertainty surrounding the exact point of ecosystem collapse, the adopted methodology based on ecological integrity and risk was used for comparing scenarios for different kinds and intensities of human uses of the marine resources. With the deployed methodology changes with respect to a reference situation, which is considered “acceptable”, are measured. There are some substantial differences between the two considered case studies: in the case of eutrophication, the modelled reduction scenarios have been considered in a 25 year-time-frame, i.e. the presented results are long-term effects of nutrient reduction strategies; in the case of offshore wind, the modelled scenarios represent the temporary (yearly) changes due to construction and not the long-term effect upon the ecosystem resulting from the presence and operation of turbines in the marine environment. The results have been spatially aggregated into one Elbe box in order to (a) compare the two case studies and (b) assessing effects of offshore wind farm construction on a relatively large scale, which represents a considerable part of the German EEZ. Considering the aggregated box, however, does not allow the appraisal of local effects of offshore wind farm construction (i.e. effects in single boxes, which are compensated for in the aggregated Elbe-box).

Selected indicators are used to show ecosystem process changes taking place as a consequence of human pressures and relative to selected reference conditions. By “relative changes”, we mean that (integrity and) risks are assessed 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 a ‘high’ or ‘maximum’ risk situation has been relatively straightforward (see also Nunneri et al., 2007 for major details), the choice of references for an emerging issue such as wind farm construction has posed new challenges (especially due to local/spatial effects of construction

ISSN 1865-1542

scenarios). When comparing single indicator values under different scenarios, it is evident that , with respect to primary production, the effects brought about by offshore wind construction scenarios in the Elbe box are much more considerable than those resulting from a reduction of nutrient emission corresponding to pristine conditions (234 g C $m^{-2}y^{-1}$). Primary production under the combined scenario is even lower than under scenario B1. This shows that cumulative changes may bring about unforeseeable effects with respect to the ones assessed under single scenarios. It is worth noticing that a reduced primary production has lead to a decreasing risk in the case of eutrophication and an increase of risk in the case of offshore wind construction scenarios. This is due, on the one side, to the choice of reference situations of minimum risk (i.e. the current situation without wind farms, which corresponds to eutrophication year 1995), and, on the other side, to the spatio-temporal complexity of the issue (where and when offshore wind farms are constructed within the scenarios).

In general, it can be said that the integrity indicators do not present a linear behaviour and might assume similar values in very dissimilar situations. Moreover, some processes can be maximised and other minimised under the maximum risk situation: while in the case study eutrophication there is some linear behaviour of indicators (from maximum to minimum risk situation), this is not the case for values associated with offshore wind farm construction (compare figures 4 and 5). Nevertheless the selected indicators allow depicting changes resulting from human pressures with respect to a chosen reference situation, thereby indicating how far changes reach from a known or acceptable situation towards a one which is assumed to represent maximum risk of collapse.

For management issues it can be relevant to evaluate the risk related with one use of the marine environment (e.g. offshore wind farm construction) against the one brought about by another use (e.g. eutrophication), i.e. the two assessments must be comparable. This means that they need to be calculated with respect to the same references. The analysis for the combined scenario has been the first attempt to represent *'ecological risk'* on a common scale for two different issues. The result is that effects of a single human use may change when observed against new reference situations, which are derived taking into account a second activity. The results of the assessment against a common reference situations is given in figure 7, where the risk associated with different scenarios is shown. Being both the assessment of integrity and risk based upon normalised indicator values, a change of reference conditions may considerably affect risk appraisal, as it is the case for offshore wind if one compares figures 6 and 7. In this case including indicator values related with eutrophication scenarios considerably changed the normalisation extremes and thereby *'ecological risk'* appraisal.

It is crucial to distinguish between features brought about by modelling or methodological constraints and risk patterns. By looking at figure 7 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 due to offshore wind farm construction to the 1995 (maximum) eutrophication level. In this sense, it is not surprising that *'ecological risk'* related to offshore wind is higher than that 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. It was not possible to obtain the *'ecological risk'* levels for the scenario Med-Red-B1 by subtracting the 1995 risk from the offshore wind scenarios and adding the value obtained to the Med-Red scenario (as it can be seen in figure 7). This happened also when considering the same references conditions, i.e. having included also the indicator values of the combined scenario for the normalisation procedure.

ISSN 1865-1542

6 Conclusions

This paper aimed at assessing the single and joint effects brought about by two different uses of the North Sea: assimilating capacity (nutrient discharge, represented by eutrophication) and resource use (represented by offshore wind farm construction). Those issues have been analysed in terms of *'ecological risk'* in a selected area of the German coastal waters. The key-findings are that changes from a eutrophicated state to a "pristine" state of the coastal waters are characterised by decrease of primary production, tendency to increase cycling of matter and reduce losses as well as shifting of phytoplankton towards higher diatoms abundance. Those are long-term changes which take place over a 25-year time-frame. Risk decreases in a rather linear way from scenario to scenario towards the pristine conditions.

The short term changes which characterise construction of offshore wind farms are more variable in dependence of spatial distribution of disturbance (i.e. location of wind farms) with respect to the selected area. In general, when construction takes place in the selected area, this results in reduced primary production, reduced sedimentation, especially of organic matter, and an alteration of the natural diatom and flagellate bloom succession. However, construction taking place outside the considered area may increase integrity indicators. Therefore also risk values show peaks in correspondence of construction within the selected area. It has been seen that, in terms of absolute indicator values, offshore wind construction can considerably affect some ecosystem functions. This was the case for primary production. Due to considerable alteration of water turbidity, primary production changes would be much more decisive under offshore wind construction scenarios than under nutrient reduction scenarios. For the combined scenario, it was shown that diminishing the availability of nutrients of, e.g. 40%, *'ecological risk'* associated with offshore construction could be also further reduced (Fig. 7). This shows a potential for limiting *'ecological risk'* brought about by offshore wind farm construction, by reducing the pressure on the ecosystem due to excessive nutrient emissions into the coastal waters. However, the *'ecological risk'* has been assessed by deploying parameters originally used for the case study eutrophication and thereby the further risk reduction under the reduction of nutrient emission can be a result of the chosen indicators, which are 'more' suitable for appraisal of changes related to nutrient reduction scenarios.

The hypothesis that *'ecological risk'* assessed for different issues can be simply added or subtracted to assess cumulative effects has been tested and rejected in this paper. While the used methodology based on scenario assumptions and the chosen modelling tool, ERSEM, is suitable for addressing relative changes both in the short and long term, it cannot be used for extrapolating the effects of multiple or combined human pressures from the assessment of single ones. Ecological risk assessment, in its current operationalisation, does not allow extrapolating the risk for a 'more complex' situation based on previously examined 'simpler' ones. The conclusion is that non-linear effects prevail when considering changes jointly (e.g. nutrient reduction and offshore wind construction): those effects are not obtainable as a sum of single effects. The new challenges for the presented risk methodology are then if and how it can be improved also for depicting cumulative effects. The need of modelling tools for exploring different use combinations has been stressed in this study.

Although the assessment of *'ecological risk'* allows for interregional comparisons (e.g. Nunneri et al. 2007) and comparisons of different issues (this study), thus offering a platform for broader discussion among experts and decision-makers, it cannot replace a broader assessment based on interpretation of single indicators. Similarly, by assuming a highest and lowest risk reference situation, this paper does not deal with the issue of risk acceptance levels. Under the current knowledge it is not possible to fix thresholds for minimum, medium and maximum risk. As many authors have affirmed, this issue is a delicate one needing participation on a broader basis in order to legitimize the setting of thresholds and acceptable risk levels under uncertainty (e.g. Kasperson et al. 1988, Renn 1998, Slovic et al.

ISSN 1865-1542

2004, Fischhoff 2007, for a brief review see Nunneri 2007). While the ERSEM model has been used over years for dealing with eutrophication issues, this is the first time it is used also for assessing effects deriving from a different use of the North Sea. The selected integrity indicators offer in general a good overview of the changes occurring in the system. However, in general, further research should aim at improving *'ecological risk'* assessment, based on a broader set of indicators, including also other parameters in addition to the ERSEM ones, in order to be suitable for assessing ecological integrity and risk potentially for any given issue. In principle for any integrity process more than one indicator should be taken into account. Furthermore, the case of combined effects of multiple uses should be analysed in a more systematic way, for instance by modelling the effects of all possible scenario combinations in order to highlight non-linear effects. Finally, the *'ecological risk'* concept should be tested with regard to its potential for communication and in a participatory context for setting acceptable reference conditions (risk levels).

7 References

Aven, T. 2007. A unified framework for risk and vulnerability analysis covering both safety and security. Reliability Engineering and System Safety 927, 45-754.

Baretta, J.W.; Ebenhöh, W. & P. Ruardij 1995. An overview over the European Regional Sea Ecosystem Model, a complex marine ecosystem model. Netherlands Journal of Sea Research 33, 233-246. doi:10.1016/0077-7579(95)90047-0

Barkmann, J. 2002. Modellierung und Indikation nachhaltiger Landschaftsentwicklung. In: EcoSys, Beiträge zur Ökosystemforschung 9.

Barkmann, J.; Baumann, R.; Meyer, U.; Müller, F. & W. Windhorst 2001. Ökologische Integrität: ökosystemare Risikovorsorge als Aufgabe eines nachhaltigen Landschaftsmanagements. GAIA 10, 97-107.

Barkmann, J. & W. Windhorst 2000. Hedging our bets: the utility of ecological integrity. In: Joergensen, S.E. & Müller, F. (eds.), Handbook of Ecosystem Theories and Management. Lewis Publishers, Boca Raton (FL, USA), 497-517.

Beaumont, N.J.; Austen, M.C.; Atkins, J.P.; Burdon, D.; Degraer, S.; Dentinho, T.P.; Derous, S.; Holm, P.; Horton, T.; van Ierland, E.; Marboe, A.H.; Starkey, D.J.; Townsend, M. & T. Zarzycki 2007. Identification, definition and quantification of goods and services provided by marine biodiversity: Implications for the ecosystem approach. Marine Pollution Bulletin 54, 253-265. doi:10.1016/j.marpolbul.2006.12.003

BMU 2007. Offshore Wind power Deployment in Germany. Report, Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit, Berlin.

Breckling, B. & T. Potthast 2004. Der ökologische Schadensbegriff - eine Einführung. In: T. Pottharst (ed.): Ökologische Schäden, begriffliche, methodologische und ethische Aspekte, Theorie in der Ökologie, Bd. 10, 1-15.

Burkhard, B. 2006. Nordsee 2055 Zukunftsszenarien für die Küste. In: EcoSys, Beiträge zur Ökosystemforschung 46, 70-89.

Burkhard, B. & F. Müller 2006. Von der norddeutschen Kulturlandschaft zu den Grenzen der Ökumene - Modellierung des Wasser- und Stoffhaushaltes auf verschiedenen Skalen. In: Kulke, E., Monheim, H. & K. Wessel (eds.): GrenzWerte - wissenschaftliche Abhandlungen des 55. Deutschen Geographentages Trier 2005, 383-392.

De Groot, R. 1992. Functions of Nature, Wolters-Noordhoff.

De Jong, F. 2006. Marine Eutrophication in Perspective. Springer.

EEA, European Environment Agency 1998. Environmental Risk Assessment, Approaches, Experiences and Information Sources. Written by Robyn Fairman, Carl D. Mead & W. Peter Williams, MARC, King's college London for the EEA, Copenhagen.

EEA, European Environment Agency 2001. Scenarios as tools for international environmental assessments. Technical report, European Environment Agency.

Fischhoff, B. 2007. Acceptable risk: a conceptual proposal. Published online. URL http://www.pierce-law.edu/risk/vol5/winter/Fischhof.htm. (Date: 03.04.2007).

Folke, C.; Carpenter, S.; Elmqvist, T.; Gunderson, L.; Holling, C.S. & B. Walker 2002. Resilience and Sustainable Development: Building Adaptive Capacity in a World of Transformations. Ambio 5, 437-440. doi:10.1639/0044-7447-(2002)031[0437:RASDBA]2.0.CO;2

Haller, M. 1990. Risiko Management und Risiko-Dialog. In: M. Schütz (ed.): Risiko Management und Risiko-Dialog, in Risiko und Wagnis. Die Herausforderung der Industriellen Welt Vol.1, Gerling Akademie, 229-256.

Heath, M.R.; Edwards, A.C.; Pätsch, J. & W.R. Turrell 2002. Modelling the behaviour of nutrients in the coastal waters of Scotland. Fisheries Research Services Marine Laboratory Aberdeen, Scottish Executive Central Research Unit Contract.

Hickel, W.; Mangelsdorf, P. & J. Berg 1993. The human impact in the German Bight: Eutrophication during three decades (1962 - 1991). Helgoländer Meeresuntersuchungen 47, 243-263. doi:10.1007/BF02367167

Hughes, T.P.; Bellwood, D.R.; Folke, C.; Steneck, R.S. & J. Wilson 2005. New paradigms for supporting the resilience of marine ecosystems. TRENDS in Ecology and Evolution, 20, 380-386. doi:10.1016/j.tree.2005.03.022

Joint, I. & A. Pomroy, 1993. Phytoplankton biomass and production in the Southern North Sea. Marine Ecological Progress Series 99, 169-182. doi:10.3354/meps099169

Kasperson, R.E.; Renn, O.; Slovic, P.; Brown, H.S.; Emel, J.; Goble, R.; Kasperson, J.X. & S. Ratick 1988. The social amplification of risk: a conceptual framework. Risk Analysis 8, 177-187. doi:10.1111/j.1539-6924.1988.tb01168.x

Kay, J.J. 1993. On the nature of ecological integrity: some closing comments. In: Woodley, S., Kay, J. & G. Francis (eds.): Ecological integrity and the management of ecosystems. University of Waterloo and Canadian Park service, Ottawa, 201-214.

Köller, J.; Köppel, J. & Peters, W. 2006. Offshore wind energy research on environmental impacts. Springer, Berlin.

Lenhart, H.-J. 2001. Effects of River Nutrient Load Reduction on the Eutrophication of the North Sea, Simulated with the Ecosystem Model ERSEM. In: Kröncke, I. & Türkay, M. & J. Sündermann (eds.): Burning issues of North Sea ecology, Proceedings of the 14th international Senckenberg Conference North Sea 2000, Senckenbergiana maritima 31, 299-311.

Lenhart, H.-J. & T. Pohlmann, 1997. The ICES-boxes approach in relation to results of a North Sea circulation model. Tellus 49A, 139-160.

Lenhart, H.J.; Radach, G. & P. Ruardij 1997. The effects of river input on the ecosystem dynamics in the continental coastal zone of the North Sea using ERSEM. Journal of Sea Research 38, 249-274. doi:10.1016/S1385-1101(97)00049-X

Millennium Ecosystem Assessment (2005). Ecosystems and Human Well-being: Synthesis report. Washington D.C.

Moll, A. & G. Radach 2003. Review of three-dimensional ecological modelling related to the North Sea shelf system - Part 1: Models and their results. Progress in Oceanography 57, 175-217. doi:10.1016/S0079-6611(03)00067-3

Müller, F. 2005. Indicating ecosystem and landscape organisation, Ecological Indicators 5, 280-294. doi:10.1016/j.ecolind.2005.03.017

Müller, F.; Hoffmann-Kroll, R. & H. Wiggering 2000. Indicating ecosystem integrity - theoretical concepts and environmental requirements. Ecological Modelling 130, 13-23. doi:10.1016/S0304-3800-(00)00210-6

Nunneri, C.; Windhorst, W.; Turner, R.K. & H. Lenhart 2007. Nutrient emission reduction scenarios in the North Sea: an abatement cost and ecosystem integrity analysis. Ecological Indicators 7, 776-792. doi:10.1016/j.ecolind.2006.09.002

Nunneri, C. 2007. Linking Ecological and Socio-economic System Analysis – a Methodological Approach based on Ecological Risk. Berichte aus dem Forsch.- und Technologiezentrum Westküste der Universität Kiel, 45, 181, Büsum.

OSPAR 1998. Report of the ASMO Modelling Workshop on Eutrophication Issues, 5-8 November 1996, The Hague, The Netherlands, 86 pages. OSPAR, 2003. Bremen Statement. URL http://www.ospar.org/eng/html/md/Bremen_statement_2003.htm (Date: 02.05.2007).

Perrings, C. & D. Pearce 1994. Threshold effects and incentives for the conservation of biodiversity. Journal Environmental and Resource Economics, 4, 13-28. doi:10.1007/BF00691930

Potthast, T. 2004. Conceptual, epistemological, and ethical perspectives on 'ecological damage' with regard to genetically modified organisms. In: Risk Hazard Damage – speculation of criteria to assess environmental impact of genetically modified organisms. In: Breckling, B. & R. Verhoeven (eds.): Naturschutz und biologische Vielfalt 1, Bundesamt für Naturschutz, Bonn, 245-256.

Radach, G. 1998. Quantification of long-term changes in the German Bight using an ecological development index. ICES Journal of Marine Science 55, 587-599. doi:10.1006/jmsc.1998.0403

Radach, G.; Berg, J. & E. Hagmeier 1990. Long-term changes of annual cycles of meteorological, hydrographic, nutrient, and phytoplankton time series at Helgoland and at FV Elbe 1 in the German Bight. Continental Shelf Research 10, 305-328. doi:10.1016/0278-4343(90)90054-P

Scheffer, M.; Carpenter, S.; Foley, J.A.; Folke, C. & Walker 2001. Catastrophic shifts in ecosystems. Nature 413, 591-596.

Slovic, P.; Finucane, M.; Peters, E. & D. MacGregor 2004. Risk as analysis and risk as feelings: some thoughts about affect, reason, risk and rationality. Risk analysis, 24, 311-322. doi:10.1111/j.0272-4332.2004.00433.x

Van Beusekom, J. & S. Diehl-Christiansen 1994. A synthsis of phyto- and zooplankton dynamics in the North Sea environment Godalming: WWF - World Wide Fund For Nature, 148 pp.

van Notten, P.; Rotmans, J.; van Asselt, M. B. A. & D.S. Rothman 2003. An updated scenario typology. Futures 35, 423-433. doi:10.1016/S0016-3287-(02)00090-3

Walker, B.H., Gunderson, L.H., Kinzig, A.P., Folke, C., Carpenter, S.R. & L. Schultz 2006. A handful of heuristics and some prepositions for understanding resilience in social-ecological systems. Ecology and Society 11 (1): 13 (online), URL http//www.ecologyandsociety.org/vol11/iss1/art13/. (Date: 02.05.2008)

Windhorst, W.; Colijn, F.; Kabuta, S.; Laane, R. & H.-J. Lenhart 2005. Defining a good ecological status of coastal waters - a case study for the Elbe plume. In: J. Vermaat, L. Bouwer, W. Salomons & R. Turner, (eds.): Managing European coasts: past, present and future, Springer, 59-73.