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Integrated coastal management from the perspective of nutrient control

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Abstract. There is an urgent need to control nutrient discharges into coastal aquatic systems because they are one of the main anthropogenic environmental impacts on coastal seas. Efficient management of nutrient fluxes in the coastal zone requires a sound scientific basis. This involves research on the dynamics of natural and social systems. In this paper some concepts and tools are discussed which are currently available in ecology and economics to preserve the aquatic environment. In ecology, knowledge of ecosystem functions and responses to human perturbations at the ecosystem level is more advanced than at the community level. Therefore, the biological integrity approach as the primary management goal is a more useful concept than the biological diversity approach.

Economics is the necessary link between the environmental issue that is caused by human activity and the deduction of an environmental policy that addresses the origin of these issues. Different approaches of selecting cost-minimal solutions and optimal levels of environmental quality, as well as different categories of environmental goods, are discussed.

Ecological and economic sciences are mainly referring to regional situations. Hence Integrated Coastal (Zone) Management (IC(Z)M) strategies are case-specific and basically local. However, the main challenge of ICZM research is to handle the variety of temporal and spatial scales of natural and social systems in order to obtain a comprehensive description of processes controlling nutrient dynamics in the coastal zone which can be used by decision-makers.

Keywords: Biological integrity; Black Sea; Environmental economics; River catchment.

Abbreviations: ICM =I ntegrated Coastal Management; ICZM =I ntegrated Coastal Zone Management.

Introduction

Marine eutrophication has emerged as one of the major environmental issues of our time (Nixon 1995). River loads of nitrogen (N) and phosphorus (P) have increased worldwide more than fourfold due to human activities in the river catchment areas (Martin et al. 1981); this has resulted in considerable eutrophication in many coastal seas (e.g. Service et al. 1996). The increased N- and P- discharges have led to elevated organic production and widespread oxygen deficiency in bottom waters and sediments of coastal seas. Negative effects of hypoxia such as a change in the structure and stock of benthic communities (Heip 1995) and large-scale fish kills are now global phenomena. Conversely, major engineering works such as river dams have cut the silicate flow. Increased N- and P-concentrations together with decreased silicon (Si) concentrations in the coastal seas led to the appearance of novel and/or noxious algal blooms along many of the worldÕs coastlines (Smayda 1990).

Coastal systems are appropriate examples for demonstrating general ecological effects of anthropogenic impacts on entire landscapes linked by an aquatic continuum from land to the sea. In this paper an example of the severe effects of altered nutrient fluxes in the coastal zone will be given, followed by discussion of ecological concepts and economic issues for coastal zone management. Finally, some generic strategies and scientific tools (ecological models and a mix of economic measures) will be outlined.

The Black Sea example

The semi-enclosed Black Sea is one of todayÕs most endangered seas as it is affected by altered river nutrient discharges. The catchment area of the Black Sea is ca. 2 million km^2, roughly 5 × its own area (Fig. 1). Ca. 162 million people live in the Black Sea basin, disposing their effluents into the inflowing rivers, of which the river Danube is by far the most important. Ca. 70 % (210 km^3 annually) of the entire discharge is contributed by the Danube, which collects effluents from 10 European countries. In the Danube river a five-fold increase in total inorganic nitrogen and a two-fold increase of phosphate load have been observed since the 1960s. Many diffuse sources are responsible for these increases such as nitrogen-fertilizers from agriculture and phosphate, mainly from point sources such as waste

Fig. 1. The Danube delta and catchment area.

from cities and industrial centres including Bucuresti, Beograd, Budapest, Wien and MŸnchen.

As a result of the increased nitrogen and phosphorus inputs, phytoplankton densities in the northwestern shelf waters of the Black Sea Ð which are less than 200 m deep Ð have increased dramatically by about one order of magnitude over the last two decades (Bodeanu 1993). Elevated phytoplankton blooms and their subsequent sedimentation on the sea floor have led to frequent hypoxic and anoxic events on the northwestern shelf. These shallow parts are important for benthic life and fish stocks in the otherwise very deep Black Sea. The entire Black Sea shelf is now prone to the occasional formation of an anoxic benthic layer (Mee 1992). Consequently, the standing stocks of zoobenthos (molluscs, crustaceans) and macroalgae, which are important breeding grounds for fish, decreased by factors of 4 and 10, respectively (Gomoiu & Tiganus 1990; Tolmazin 1985). Catches of flounder and turbot in particular decreased sharply (Caddy & Griffiths 1990). Sensitive high-value fish species have disappeared throughout the Black Sea to be replaced by large numbers of less important small omnivorous fishes such as sprat (Mee 1992). Of 26 commercially important species of fish caught in the 1960s, only six remain in significant quantities; however, this is also a result of overfishing (Caddy & Griffith 1990).The Ukrainian and Romanian fishing fleets in the Black Sea have now almost disappeared. In the Danube Delta a considerable decrease in the size of fish catches has also been observed (see Fig. 2).

Besides these well known eutrophication effects, which are similar to those described for almost all densely populated coastal areas of the world (Vollenweider et al. 1992), the regulation and damming of rivers due to the need for hydro-electric power has also had severe consequences for the coastal ecosystems. The river Danube was dammed in the early 1970s at the Yugoslavian/Romanian border, some 600 km upstream from the river mouth. This led to a decrease in silicate concentration of the Danube river of ca. 70% due to sedimentation of organic (diatoms) and inorganic (clays) particles (Ôartificial lake effectÕ, Bennekom & Salomons 1981). In the Black Sea surface waters a concomitant decrease in silicate concentration of more than 60% was observed. The resulting changes in the Si:N:P ratio of the nutrient load of Black Sea appear to be larger than would be caused by eutrophication alone, and seem to be responsible for dramatic shifts in phytoplankton species composition from diatoms (siliceous) to coccolithophores and flagellates (non-siliceous) (Humborg et al. 1997). While diatoms increased

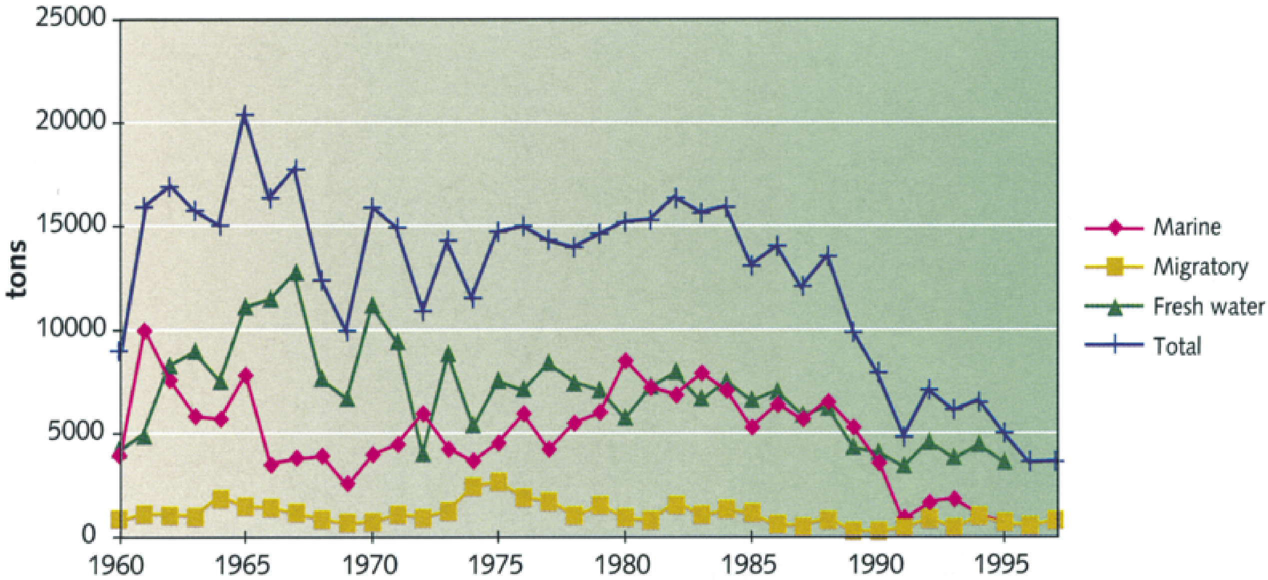

Fig. 2. Catches of main fish species in the Danube Delta.

by a factor of 2.5 blooms of non-diatoms, of which some species are facultatively toxic, e.g. *Chromulina* spec., increased by a factor of six. The possible effects on the food web are still speculative. Due to the fact that primary producers (phytoplankton) have, by far, the largest biomass in marine ecosystems (1 and 4 orders of magnitude higher than zooplankton and fish biomass, respectively) changes in their community structure will have severe impacts on the entire ecosystem structure of the coastal seas. However, the combined effects of altered nutrient discharges from the Danube, overfishing and the introduction of alien species such as ctenophores (Mee 1992) on food-web structures has not yet been investigated.

Coastal eutrophication is a worldwide phenomenon (Vollenweider et al. 1992; Nixon 1995) and more than 36 000 dams are currently in operation around the world and more are still being constructed (Goudi 1994). The changed ecosystem structure of coastal seas due to altered river nutrient discharges provides a good example for the need of new, integrated concepts for the protection of coastal environmental resources. For environmental management of coastal areas, which connect terrestrial, freshwater and marine systems, a large-scale view is required. A useful concept in this context is that of Ôbiological integrityÕ (Anon. 1990), which was developed in the early 1970s for the management of water resources. This will now be discussed with respect to nutrient control.

Ecological concepts for protecting aquatic resources

ÒEcology is more a science of case studies and statistical regularities, than a science of exceptional, general lawsÓ (Shrader-Frechette & McCoy 1995). This down-to-earth statement does have important implications for ecological management concepts. The uniqueness and complexity of ecological properties are so pervasive that it is unrealistic to expect that concepts and models will ever develop into precise ÔlawsÕ that are universally applicable for protection strategies (Shrader-Frechette & McCoy 1995). This is especially true at the community level but also to a lesser extent for the ecosystem level. Because ecology is based primarily on field observation and experiment, and most work has been done in terrestrial ecosystems followed by freshwater than marine systems, it is reasonable to say that we have more ecological knowledge about terrestrial and freshwater systems than marine systems.

Some definitions

In ecology, community perspectives are largely based on population dynamics and focus on species interactions and processes such as extinction and invasion. The community perspective focuses on the living part of the ecosystem, disregarding the abiotic and physical description of the system. In contrast, ecosystem perspectives are based on thermodynamics and focus on the dynamics of energy and materials maintained by organisms and physical transport. Either perspective can be applied at any level in the ecological hierarchy (Angermeier & Karr 1994). The terms Ôbiological diversityÕ and Ôbiological integrityÕ are corresponding expressions often used during the last two decades in concepts for environmental policy. The idea of biological integrity goes beyond the concept of biological diversity, because it includes fundamental biogeochemical processes such as primary production, respiration and nutrient cycling that generate and maintain ecosystem elements.

ÒThe capability of maintaining a balanced, adaptive community having a species composition, diversity and functional organization comparable to that of natural habitat of the regionÓ (Karr & Dudley 1981) is a widely quoted definition of Ôbiological integrityÕ and is advised as the primary directive for water policy. Various forms of this definition now provide the basis for biotic assessments of surface waters, e.g. by the US Environmental Protection Agency (Anon. 1990). As with many other environmental management concepts such definitions often have no direct operational meaning, even though that would be desirable. However, with respect to the above definition, a general question is apparent; to what extent and on which perspective, is ecology as a science purely descriptive or does it also have some predictive power? ÒAn ecosystem is a thermodynamically open system which is far from equilibriumÓ (Odum 1992). Are ecologists able to define an Ôadaptive communityÕ or a Ôfunctional organizationÕ which meet the general necessities of a sustainable ecosystem, a Ôminimum ecological demandÕ within the densely populated coastal zone, which is assumed by the integrity definition given above?

Description of ecosystem functions and responses

The relationship between aquatic biological diversity and ecosystem functions and responses to perturbations has not been quantified. Hence, there is insufficient scientific knowledge to generate overall environmental protection goals for the coastal zone such as the conservation and sustainable use of freshwater and marine resources. The effects of altered phytoplankton species composition on the coastal food web as a consequence of damming and eutrophication, as mentioned above, is unknown and cannot be predicted.

Generally speaking, for the community perspective, basic ecological concepts and theories such as community, diversity and stability are imprecise and loosely defined. Diverse communities are generally thought to bemore stable or resilient to human perturbations than less diverse ones. Only a few terrestrial field studies exist on the impacts of biological diversity on ecosystem functions such as resilience (e.g. Tilman & Downing 1994). It can also be argued that complexity does not necessarily lead to higher stability as is often stated by environmentalists. The Black Sea and the Baltic Sea, for example, have fewer species than the Mediterranean and North Sea, respectively, due to the brackish character of their water masses which allow fewer but more opportunistic species to live there. Are the Black Sea and Baltic Sea therefore less stable? Elmgren (1984) pointed out that the trophic dynamics (primary production, respiration and nutrient cycling) in the Baltic Sea do not differ from those of other coastal seas. The yield of fish per unit of primary production in the Baltic is also similar to other coastal seas (Nixon et al. 1985). Rather, the physical settings, such as water residence times and flushing of the system transporting oxygenated waters in the deep layers are, by far, the most important factors in the stability and susceptibility of a coastal marine ecosystem and therefore have to be addressed by coastal zone management strategies.

Ecologists today might be able to identify and predict ecosystem functions and responses of the environment by using a Ôthermodynamic viewÕ (ecosystem perspective), which takes the physico-chemical driving forces into account (Schindler 1990). The productivity of aquatic systems Ð an important parameter to characterize biogeochemical fluxes in a given system Ð is mainly a function of the supply of allochthonous nutrients and energy (light, organic matter) as well as the residence times of water masses. A change in productivity patterns as a result of altered nutrient river discharges can therefore be predicted. Highly productive aquatic systems in coastal areas are, for example, wetlands, deltas and estuaries with high filter capacities for nutrients due to natural removal processes (denitrification and sediment burial).

These systems also represent ecological assets of considerable value. These regions are also susceptible to man-made inputs of nutrients and organic matter due to the long residence times of water masses and thus also of nutrients and organic matter. During the remineralization process all oxygen is consumed and the system switches to anaerobic pathways of organic matter decomposition such as sulphate reduction. From the thermodynamic viewpoint, resilience, defined here as the speed with which an ecosystem returns to its former state after it has been perturbed (Begon et al. 1996) by altered nutrient inputs, might be described by the relationship between nutrient supply and residence times of water and organic matter and the resulting switches between mineralization pathways.

One may conclude that, at present, ecologists are able to estimate and predict, to a certain extent, budgets and fluxes of biogeochemically important elements. However, predictions of human impacts on variations at the species level is not yet practicable. Large ecosystems (river catchments, coastal seas) tend to have a more steady-state character than their individual components (communities), which tend to be oscillatory or chaotic. Therefore, the description of the former is more advanced and hence geochemical fluxes in the coastal zone seem to be ÔmanageableÕ. ÒIf we are serious about sustainability, we must raise our focus in management and planning to large landscapes and beyondÓ (Odum 1992). Coastal zone management should, therefore, concentrate on functional processes such as the fundamental biogeochemical cycles in the coastal zone.

The reasons for appearance or disappearance of individual species is at present beyond our knowledge. Biological integrity and not biological diversity is the more meaningful aim (Angermeier & Karr 1994) even though both are poorly, if at all, defined. This does not mean that we do not need a community perspective for the solution of practical environmental problems. A profound knowledge of the local and regional species composition and their natural history based on long-term monitoring programs is indispensable as it provides an early warning system for possible changes of the environment due to natural changes or anthropogenic impacts. Due to the fact that the extent of ecological theory is limited regarding both perspectives a Ôbottom-upÕ approach, i.e. a check of ecological theory instance by instance, and a case-specific ecological knowledge of ecosystems are essential for coastal environmental action plans (Shrader-Frechette & McCoy 1995). These reflections on ecology bring about the question: ÒWhat contributions are required from other sciences, in this case economics?Ó

Issues of environmental economics

Economics as Ôthe science of scarcityÕ, in this case the lack of a sustainable ecosystem, provides different views on environmental issues. Four such views are presented.

1. Different layers of analysis

1. Starting from the natural scientific diagnosis of a deteriorated ecosystem, economics can be applied to select a cost-minimal solution for reaching a given goal, i.e. a certain reduction of nutrients and pollution loads in a point vs. non-point discharge setting or in comparing different nutrients of which either could be the limiting factor of primary production, for example N or P in eutrophication.

2. Alternatively, environmental economics can be used to deduce Ôan optimal level of environmental qualityÕ, which is defined as Òthe degree of environmental deterioration at which the marginal cost of pollution just equals the marginal willingness to pay of the potential beneficiaries, i.e. the population concerned, for an improvement of the status of the environmentÓ (Siebert 1992). While this approach appears to be somewhat bizarre, especially to natural scientists, it is, for the economist, merely the explicit statement of one of the interdependencies between the natural environment and the anthropogenic influences on it. While (1) only requires a cost-benefit analysis in the sense of identifying the ecological effects of one unit of money invested, (2) furthermore depends on methods to define Ômarket valuesÕ, the price someone is willing to pay for non-marketable goods such as environmental quality (Costanza et al. 1997; Mitchell & Carson 1989).

3. Assuming that environmental aims have been agreed upon, either by the process described above or simply by political decisions more or less influenced by these considerations, the determination of an efficient instrumental mixture is required. Efficiency in this context is the combination of those measures that will achieve the aim at minimum costs. An instructive example is the discussion in the 1970s and 1980s regarding the advantages and disadvantages of command and control vs. market-oriented instruments as taxes or tradeable permits in the USA, which led eventually to trading arrangements for such different pollutants as NO_x, SO_2 and dust. The management of P- and N-loads by taxing the usage or setting up a tradable emissions permit program, either of which would allow for reallocation between emitters towards the most effective use of the scarce resource ecosystem and its Ôfilter capacityÕ should therefore be considered.

4. While the three levels of environmental economics mentioned above are (more or less) applicable for Western societies where functioning administrations to monitor and enforce policies exist, it is not the case in most developing countries or countries in transition. Here, an institutional analysis is required complementary to that of the desired level of environmental protection and available instruments.

Different categories of environmental goods

Economics provide a second dimension with respect to environmental issues, i.e. the differentiation between private and public goods (Coase 1960; Olson 1965). While a private asset, for example a piece of bread, can only be used by a single person or a limited

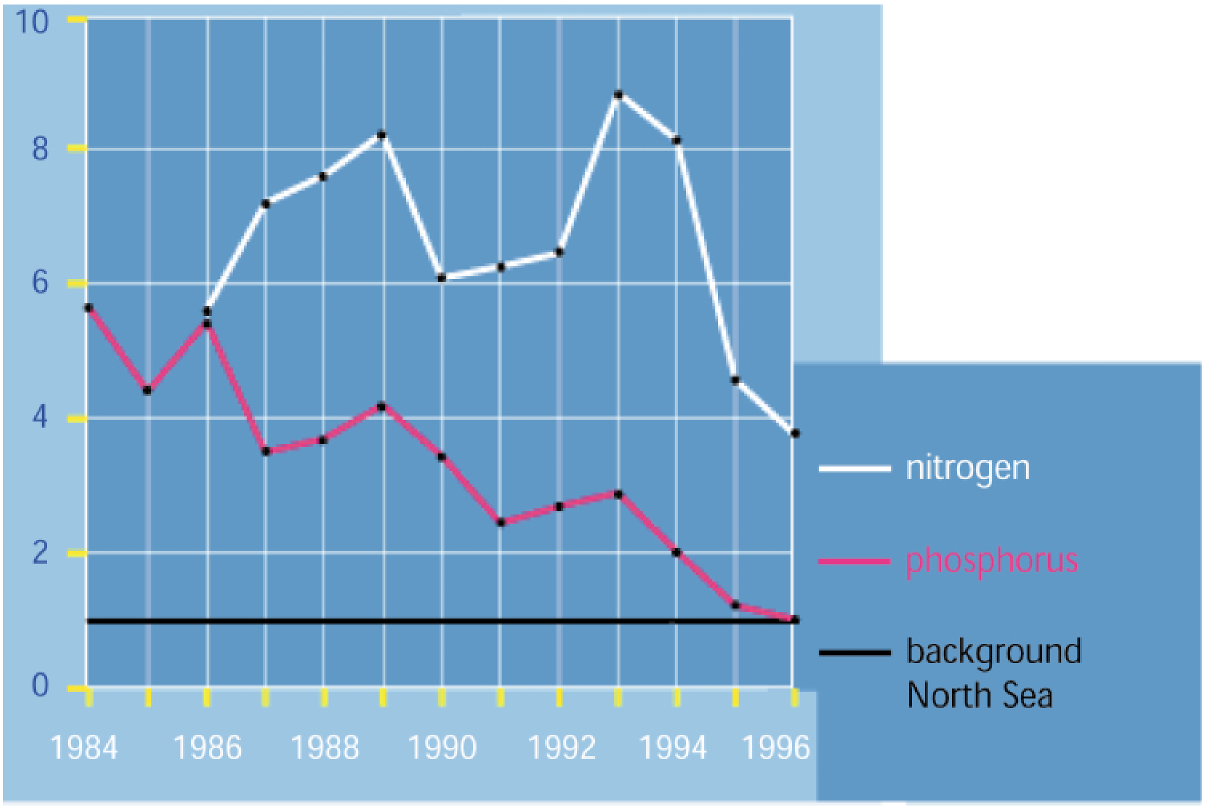

Fig. 3. Average nutrient levels in the Dutch coastal zone with respect to the North Sea background level. (From Anon. 1998.)

number of people, public goods may be used by different parties without becoming scarce. A good example is an aquifer that is used (within its regenerative capacity) for recreation, as a breeding ground for fish and as a deposit for biodegradable pollutants. If an ecosystem such as the Danube is considered a transnational public asset there are significant difficulties Ð due to the principle of national sovereignty Ð to be overcome before a feasible environmental strategy can be implemented. One of the most pronounced difficulties in this case is the asymmetry between the ÔupstreamÕ and ÔdownstreamÕ properties of a river where there is a polluter (upstream) who does not suffer the effects of environmental degradation (downstream). Secondly, differing economic wealth of the countries involved leads to significant disparities in the willingness to invest in environmental protection (Grossman & Krueger 1994). Obviously, agreements have to be reached regarding the (physical) distribution of abatement as well as the corresponding cost-sharing, before any national strategy concerning these transnational media can be promising.

That national governments behave according to these considerations can be shown by the following two examples: 1. The United Kingdom did not sign the Geneva Agreement on SO_2-reduction, since it only causes the pollution, especially in Scandinavia, but is not affected by it (upstream/downstream problem). 2. Examples for a shared financing of an environmental strategy, that still does not obey the polluter-pays-principle, are investments into the reduction of French salt inputs into the Rhine, which are only 30 % funded by France, while the balance is paid by the downstream countries, especially The Netherlands (MŠler 1991; Stršbele 1991).

It follows from these two dimensions of economic analysis that:

(a) economics is only the second step after the natural sciences have analysed the physical environmental problem, and

(b) economics is the necessary link between the environmental issue that is being caused by human action and the deduction of an environmental policy that addresses the origins. This does not only reflect the interdependencies between different levels of environmental policy within any one system, but also the institutional issues, especially between different jurisdictions.

Towards a comprehensive description of key processes

Interdepending natural and socio-economic sciences

ÒSome process is needed to decide what is the total mix of products and services that coastal areas should produce at any particular time, who should produce and pay for it, and who should benefit, and by how much. That process is integrated coastal zone managementÓ (Ehler 1993). Decision-makers obviously need more information on environmental issues in the coastal zone and it is a challenge for scientists to provide it. The overall aim of coastal zone management research is then to give a thorough description of temporal and spatial scales of natural and social systems in order to obtain a comprehensive description of environmentally important processes and feedback mechanisms.

Initially, in order to control nutrient fluxes in the coastal zone a description of natural and anthropogenic nutrient sources and sinks in different aquatic subsystems in the entire drainage areas, up to the deposition sites in the sea, is required. Secondly, temporal and spatial lags in the response of the natural system to pressures and impacts of the social systems in the coastal zone have to be identified. This leads to the following questions which are, of course, only examples, to be adopted to any specific case in question: What are the major pathways of nutrients from land to the sea and how long does it take for the nutrients to reach the deposition sites in the coastal seas? What kind of emission structure (point vs. diffusive sources) in the river catchment area is mainly responsible for eutrophication? From the economistÕs point of view, an assessment of social factors and stakeholders, as well as their respective future development, is needed. For example: How much fertilizer is, and will prospectively be, used in the drainage area? What projections of future populations and type and extent of economic activity that influence the discharge rates to coastal seas can be given? How can the limited funds be used most efficiently for the building of sewage plants? What are the administrative and legal action requirements that fulfil the obligation of a catchment wide management?

By answering these questions environmental action plans as, for example, developed in the frame of the HELCOM (Helsinki Convention for the protection of the Baltic) or of the OSPARCOM (Oslo Paris Convention for the protection of the North Sea) can certainly be improved further.

From this perspective the Ronneby declaration of 1990 adopted by the Prime Ministers of the Baltic Sea countries may be considered a good example of political action lacking scientific input (Wulff & Niemi 1992). It contains the decision to reduce the N- and P-inputs into the Baltic Sea and the North Sea by 50 % within the next 10 years. Even though this was a much desirable aim, it can only be considered a statement of political goodwill, since it neglected the fact that nutrient residence time in the soil and groundwater alone would make a 50% reduction impossible, let alone the probability of an almost total reduction of ongoing inputs (Fig. 3). It can be seen from this figure that nitrogen concentrations in Dutch waters, for instance, are still high, although nitrogen concentrations in the North Sea have been lowered since their peak in the 1970s. This is in contrast to phosphorus, which flows mainly from point sources, where a decreasing trend is obvious.

Ecological modelling approaches for management applications

A description of nutrient cycles in the coastal zone implies their modelling. Biogeochemical models of different types are available, ranging from chemical-biological coupled with circulation models to highly idealized box models. Owing to different requirements of computer power, the time scale of the problem at hand often determines which model is preferable. Assessment of the effects of changed nutrient discharges in coastal areas requires analysis on the scale of decades. Therefore, the first step for the modelling of nutrient fluxes for management applications is the establishment of simple box models, which ignore the short-term details of circulation patterns, but are suitable for long-term hind- and forecast assessment (scenarios).

The model approach should be focused on the proper description of the long-term effects of eutrophication in coastal seas, i.e. the enrichment of organic matter in the sediments, which is accompanied by hypoxia and anoxia. The parameterization of biogeochemical dynamics should include new production, accumulation of sedimented organic matter and switches in mineralization pathways as key processes. It may also include essential nutrients as nitrogen and phosphorus and other stoichiometrically linked elements such as carbon, oxygen, silicon and sulphur. A more detailed description can, for example, be found in the LOICZ biogeochemical modelling approaches (Anon. 1996) and many other models on nutrient dynamics in coastal seas (Stiegebrandt & Wulff 1987; Wulff et al. 1989; Fennel & Neumann 1996). Physical transport processes, such as import of matter from the rivers, exchange with the open sea, sedimentation and resuspension has to be described in order to reproduce the residence times of water and matter properly.

Nutrient input data for coastal sea models can be served by a GIS (geographic information system) based hydrological model developed for an entire river (Wittgren & Arheimer 1995). The main emphasis should be put on comprehensive description and visualization of all point and diffusive nutrient sources in the river catchment area to assess all nutrient emissions relevant to the coastal seas.

Furthermore, coupled physical-chemical-biological models of coastal seas give information on the spatial distribution of certain discharges in the sea (Fennel & Neumann 1996). Thus, they may help to decide which river discharges have local, regional or basin scale effects. This spatial distribution can then be applied to calculating environmental damage caused by a given discharge, which in turn can be used as the basis for international environmental cooperation and management. They may help decision-makers to take *ad hoc* decisions after ecological disasters such as oil spills or river floods carrying high amounts of nutrients and pollutants into the sea. They also elucidate the long-term effects discussed herein.

To what extent can models be predictive? For many natural scientists it is rather unusual to give forecasts, because the complexity of ecosystem properties is too pervasive. However, there is an urgent need to develop cost-effective measures countering cultural eutrophication. Ecological models, in the context of coastal zone management, are case specific and should be regarded more as an analytical and communication tool between the sciences involved. Model response depends on the external forcing and therefore the predictive potential is limited by the accuracy of forecast of the forcing. A reasonable forecast can only be estimated in cooperation with social scientists. However, ecological models can be used to test scenarios related to the effects of human activities in the catchments on different nutrient discharge patterns to coastal seas.

Efficient mix of environmental instruments

While environmental problems occur on a local, regional or global level, environmental management is essentially local, since there must be some point of origin where the pollutant enters the aquatic system. A program mixture of local measures has to be designed such that it is not only effective in the sense of technically being able to reduce the pollutant load but also be efficient in the sense of being cost minimal for any given aim. The basis for deducing this mix of measures is the description of the *status quo* of the sources, i.e. who contributes how much to the pollutant in question. This, in turn, is fundamental for the cost-benefit analysis in which different reduction scenarios for each pollutant will lead to the overall costs for any desired (quantitative) reduction aim. Eventually a cost-function can be formulated which has the reduction quantity as the variable and the associated costs as the dependent. A practical example in the context of nutrients is the calculation of costs associated with the building (or upgrading) of some filter capacity for a fertilizer factory, which drains into a river vs. a municipal sewage plant in the same area.

The argument so far has implicitly assumed that all desirable, i.e. low cost, sources of a pollutant are available for the implementation of the cost minimal strategy. What happens though, if this is not the case and if the countries involved have very different standards of living? A good example for this is the Baltic Sea, where there is a significant difference in economic prosperity, and corresponding environmental consciousness between the different countries. In order to address this problem of externalities between the countries, the political willingness and the economic ability of the parties involved need to be considered. Practically this leads to cross-border subsidies for building, for example, sewage plants. Good examples in this context are the EU or the World Bank, who sponsor environmental programs worldwide in order to maximize the ecological effects of the limited funds available, i.e. minimize the effects of the pollution.

Conclusion

For the management of altered nutrient fluxes in the coastal zone, which appear to be responsible for drastic changes in the ecosystem structure of the coastal seas, a catchment area wide approach is needed such as is also requested by the current amendment of EU water and environmental law. There exist no generalizations for ecological or environmental economic protection strategies for the coastal zone and therefore coastal zone management is basically local or regional. However, natural and social scientists have to translate their knowledge, which is based on scientific, monitoring and socio-economical data for a science and policy understanding. Based on trans-disciplinary modelling approaches they should provide sectoral and scenario analyses on different usage patterns such as extensive vs. intensive agriculture, the consequences of major engineering works and of different point sources. Alternative strategies for the use of aquatic systems as natural treatment systems for human waste and sinks for organic matter may also be taken into account as a natural service provided by aquatic ecosystems. Decreasing the emissions can certainly increase these capacities (positive feedback).

The idea of balancing ecosystems and allocating nutrient impacts in relation to the carrying capacity of different subsystems, which can be estimated through productivity/remineralization patterns, may be an overall goal for coastal zone management strategies. A comprehensive description of the different spatial-temporal scales of natural and man-made nutrient fluxes and their feedback can provide different options for decision-makers for the management, investment and regulation of nutrient fluxes in the coastal zone.

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