Available online at www.sciencedirect.com

ScienceDirect

Marine Pollution Bulletin 55 (2007) 407–414

MARINE POLLUTION BULLETIN

www.elsevier.com/locate/marpolbul

Toward the application of ecological concepts in EU coastal water management ☆

Victor N. de Jonge

Institute of Estuarine and Coastal Studies, University of Hull, Hull HU6 7RX, United Kingdom

Abstract

The EU Water Framework Directive demands the protection of the functioning and the structure of our aquatic ecosystems. The defined means to realize this goal are: (1) optimization of the habitat providing conditions and (2) optimizing the water quality. The effects of the measures on the structure and functioning of the aquatic ecosystems then has to be assessed and judged. The available tool to do this is 'monitoring'. The present monitoring activities in The Netherlands cover target monitoring and trend monitoring. This is insufficient to meet the requirements of the EU. It is, given the EU demands, the ongoing budget reductions in The Netherlands and an increasing flow of unused new ecological concepts and theories (e.g. new theoretical insights related to resource competition theory, intermediate disturbance hypothesis and tools to judge the system quality like ecological network analysis) suggested to reconsider the present monitoring tasks among governmental services (final responsibility for the program and logistic support) and the academia (data analyses, data interpretation and development of concepts suitable for ecosystem modelling and tools to judge the quality of our ecosystems). This will lead to intensified co-operation between both arena's and consequently increased exchange of knowledge and ideas. Suggestions are done to extend the Dutch monitoring by surveillance monitoring and to change the focus from 'station oriented' to 'area oriented' without changing the operational aspects and its costs. The extended data sets will allow proper calibration and validation of developed dynamic ecosystem models which is not possible now. The described 'cost-effective' change in the environmental monitoring will also let biological and ecological theories play the pivotal role they should play in future integrated environmental management.

1. Introduction

Current EU Directives such as the Water Framework Directive (WFD) (EC, 2000) and the Marine Strategy Directive (MSD) (EC, 2005) are a product of Europe's increasing desire to conserve ecosystems. Based on starting points rooted in systems ecology, the WFD demands optimization of water quality and physical habitat providing conditions meant to reach a good ecological status (GES) of these ecosystems. The most common tool to judge the systems status and to describe its temporal changes is physical, chemical and biological monitoring. The starting points of the WFD imply that we (1) know the relevant physical and chemical drivers for the different systems and (2) understand how these drivers influence habitat development, biological process rates and the development of biodiversity. Actually we do not know under what physical and chemical conditions what specific type of ecosystem may develop. This is illustrated by the continuing new findings and basic discussions on the value of emerging ecological theories for biodiversity development in natural systems as Resource Competition Theory (RCT) (e.g. Tilman, 1982; Grover, 1997; Huisman and Weissing, 1999, 2001) and the Intermediate Disturbance Hypothesis (IDH) (Horn, 1975; Connell, 1978). The basic discussion about diversity flared up some 45 years ago. Hutchinson (1961) realised that the Principle of Competitive Exclusion (Gause, 1934) did not satisfactorily explain the coexistence of large numbers of plankton species and that apart from growth limiting resources more factors had to play a role to explain the species coexistence. Based on Hutchinson's

☆ The National Institute for Coastal and Marine Management/RIKZ is acknowledged for its continuous financial support and interest in the described innovative future management strategies.

E-mail address: v.n.de.jonge@planet.nl

0025-326X/$ - see front matter © 2007 Elsevier Ltd. All rights reserved.
doi:10.1016/j.marpolbul.2007.09.014

paradox of the plankton, RCT started to develop (e.g. Tilman, 1982). In a theoretical model approach Huisman and Weissing (1999, 2001) illustrated that the competition for resources among species are presumably the major forces structuring natural communities. Huisman and Weissing showed that even complex non-equilibrium dynamics can occur leading to *internally* generated species oscillations and chaos. IDH proposes that biodiversity is highest when *external* disturbance (in e.g. light or another resource or environmental factor) is neither too rare nor too frequent. This theory opposes an older, the Stability-time Hypothesis, claiming that diversity is highest in undisturbed stable ecosystems (Sanders, 1969).

The conceptual Huisman and Weissing (1999) model was experimentally tested by Roelke et al. (2003). They showed that three replicate samples under constant resource conditions all developed a different species composition and abundance. In contrast to that a second set of three replicate samples which got the same resource but now ‘pulse-wise’ all showed a comparable development in species composition and abundance. This extremely exciting experiment demonstrated that internally induced chaotic non-equilibrium dynamics might be suppressed by specific sets of external disturbances. These observations need more extensive experimental attention for a much wider range of specific environmental factors. The effects of pulse-frequency and pulse-strength should also be further investigated so that generalizations can be applied to management practice. Based on these emerging conceptual ecological developments and requirements by the EU directives an analysis will be made here to reorganize the ongoing monitoring activities in the European Union exemplified with the situation in The Netherlands. The below proposed environmental monitoring and research scheme will offer more flexibility and an increasing co-operation between academic world and governmental services and will moreover allow that insights from ecological theories can be incorporated in management tools as monitoring, modelling and evaluating the quality of natural ecosystems.

2. System drivers, functioning and structure of ecosystems

In a general way we know rather well which factors play a role as boundary conditions in the functioning of any system. Resources as light (water turbidity included) and nutrients (phosphorus, nitrogen and silicate) and the factor temperature determine the process rates. The residence time or water dilution rate determines in combination with the grazing rates whether or not an algal bloom may occur. If we are dealing with a tidal flat system also the tide and the wind driven fluxes of microphytobenthos between sediment and water (de Jonge and van Beusekom, 1995) play a crucial role in the system functioning by supplying food for the pelagic copepods (de Jonge and van Beusekom, 1992).

System expressions related to the primary producers are functioning (process) and structure (species composition or diversity). How resources (nutrients and light) regulate the primary production or the growth of algae has been subject of study for already a long time. Generally spoken we know how primary production and nutrient uptake work and we are also very well able to measure it (e.g. Strickland and Parsons, 1972; Geider and Osborne, 1992). The same holds for the secondary production which is the growth of the grazers and the carnivores (e.g. Eleftheriou and McIntyre, 2005).

Effects of variation in temperature and the water dilution rate as factors which, on top of the effects of resources, may regulate the development of species diversity need further clarification.

Another aspect which lacks knowledge is the complex interactions between species of one trophic level (e.g. resources – algae) with species of another trophic level (e.g. algae – grazers). These interactions can be described in different ways varying from generalized to very specific species–species interactions in which not only size and shape but also info-chemicals (Wolfe, 2000) may play a role. This sort of information can be applied in dynamic ecosystem models. It can also be used in analyses related to fluxes of energy or compounds like organic carbon, nitrogen and or phosphorus (Lindeman, 1942; Ulanowicz, 1980, 1986, 1997). An example of such a flux analysis is e.g. Ecological Network Analysis (ENA) (Ulanowicz, 1986). Another approach roots in ‘community ecology’ (May, 1973) and focuses on dynamical constraints which arise from less or more intensive species interactions and which may lead to destabilization of ecosystems (Pimm, 1982). An overview of a number of these approaches are presented in Belgrano et al. (2005). All aspects have one thing in common: attempting to better understand the complex relations between changing physico-chemical and physical boundary conditions and the biological response in terms of functioning and system species structure and the mutual relations between them.

3. Ecosystem theory and ecosystem management

It is generally accepted that ecosystems are more than simply assemblages of species. The biological systems adapt to environmental changes, show complex non-linear behaviour, exhibit irreversibility when looking at community structures and processes and are sometimes able to reach different or multiple stable states. System characteristics comprise (Jörgensen and Müller, 2000) the occurrence of temporal and spatial borders of a system containing elements like resources (light and nutrients), climate related aspects (temperature and system dilution rate or residence time), primary producers, grazers, carnivores and decomposers. In addition to this Jörgensen and Müller also mention a high internal system ‘connectedness’ compared to the connectedness between systems. According to these authors ecosystems are emergent in their expression, show cycling of material, self-regulation and self-organisation based on feed back loops.

The above listed characteristics and properties are illustrative for the complexity of the systems species structure and its responses. An example of how complex a specific set of phytoplankton species can respond to only one external disturbance (variation in the water temperature) is visualized in Fig. 1. Defined was a species assemblage consisting of 10 algal species with different growth rate distributions as a function of temperature. These species represented temperature specialists and temperature generalists. The growth rates all followed a simple normal distribution. The model runs were prepared by A. Procée (Theoretical Biology, University of Groningen) and carried out by application of the Huisman and Weissing model (1999). The temperature top specialist (species 1 in Fig. 1) shows a strong response within a narrow temperature range while the temperature top generalist (species 10) shows a moderate growth response over a wide temperature range. In total four model runs of 30 000 calculation steps (not necessarily representing any realistic time step like an hour or a day) were done where every 50th time step the temperature fluctuated stochastically within a range covering 2 °C (run 1) to a range covering 8 °C (run 4) around a defined mean temperature value of 20 °C. The results illustrate that the larger the range within which the temperature fluctuations occurred the more abundant the temperature generalists became in this artificial and simple phytoplankton system. The combination of this type of theoretical experiments with experimental work is absolutely exciting because it offers the possibility for testing theoretical considerations of which the findings may be used for practical ecological and management applications and even for developing policy strategies (cf. WFD and MSD). Apart from enthusiasm we also have to be critical because the results also clearly demonstrate the complexity we face when applying this ecological theory to management practice. If, like in this example, the sort of experimentally and theoretically obtained information is too specific to be applied in management and future ecosystem strategies then at least generalizations from these studies could be useful. These generalizations could be applied in tools to judge the condition or quality of our aquatic ecosystems and also in developing future conservation strategies. An example of such a generalization is e.g. the effect of pulse-frequency and pulse intensity as stimulus for a certain biodiversity development (cf. results by Roelke et al., 2003). The main challenge at this point then is how to integrate a particular theory or theories with some monitoring strategy so that collected data can be transformed in information that meets the requirements set by e.g. the EU Directives and which is suitable to environmental management (Elliott and de Jonge, 1996).

For open and dynamic shallow coastal systems it is difficult to accept that one single species could be used to indicate the systems quality or condition. The other extreme is

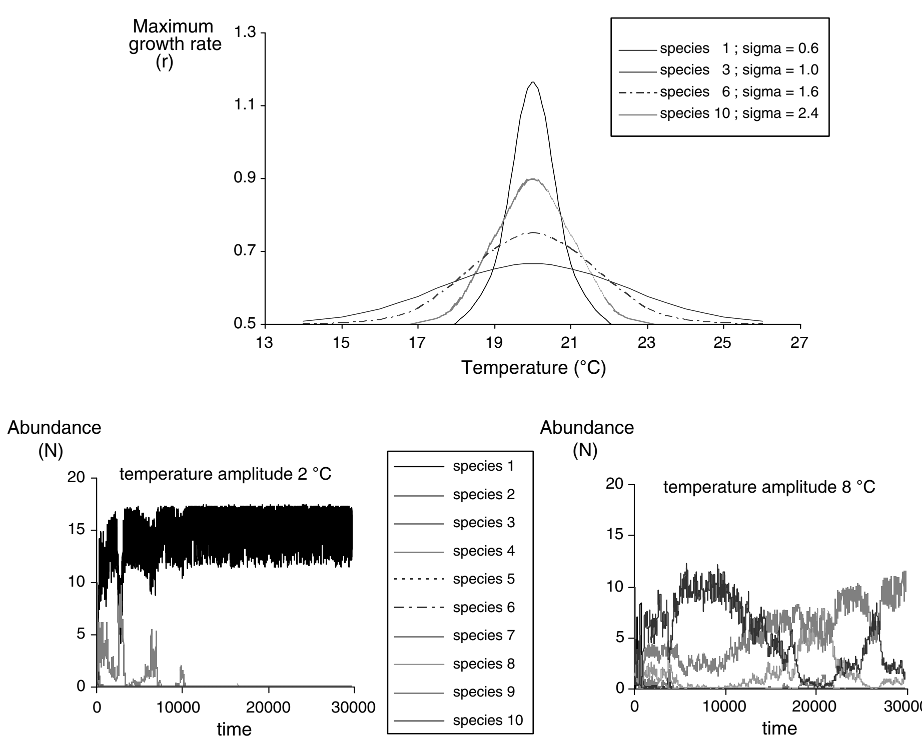

Fig. 1. A species assemblage consisting of 10 species has been defined. Among these species only the growth rates differ as a function of temperature (see examples in top part of the figure). The species with $\sigma = 0.6$ shows a strong response over a narrow temperature range and is defined as 'specialist' while the species with $\sigma = 2.4$ shows a weak response over a wide temperature range and is defined as 'generalist'. The lower part of the figure shows over time the species abundance as a function of the defined growth characteristics, internal species competition for a resource and a random variation in the external temperature at every 50th time step. The maximum variation in temperature in the left hand figure is only 2 °C while it is 8 °C in the right hand figure.

the use of a multispecies indicator as advocated by The Netherlands AMOEBA approach (cf. ten Brink et al., 1991). This last approach then gives an overview of the abundance of circa 30 species and in a way approaches a holistic view. However, comparable changes often occur in estuarine and coastal systems which make sole use of species indicators anyway questionable. Moreover, a list of species and its relative abundance is not enough to qualify its functioning. The reason for the resistance to the use of solely species oriented indices is the observed tremendous interannual variation in species abundance in our shallow coastal ecosystems which is often due to natural physical dynamics instead of anthropogenic stressors.

Attempts should be made to integrate species and their abundance with aspects representing the systems functioning. This could then culminate in the development of new and/or the application of existing 'integral ecosystem indices'. Alternatively, any 'holistic' or 'ecosystem' approach which integrates the species structure and the system functioning in any set of indices is welcomed here.

The example in Fig. 1 is illustrative for the complex response of a one trophic level system to a simple single factor variation. It illustrates the importance of understanding how and why species may respond to either natural and/or anthropogenically induced changes in boundary conditions. Integration of the primary producers with other trophic levels generates increasing complexity, not only in food web structure (e.g. range in size, shape, life cycle or life span) but also in relation to the species responses to changes in resources and mutual interaction between species. According to the WFD, this total complex of functioning structures has to be judged in preferably an integral way (EC, 2000).

An attractive first step to integrate species structure and system functioning might be a system flux analysis related approach such as Ecological Network Analysis (ENA) as developed by Ulanowicz (1980, 1986, 1997). The underlying principles of ENA are the thermodynamic laws, Lindeman's trophic analysis (Lindeman, 1942), the Finn Cycling index (Finn, 1976) and the input–output analysis (Leontief, 1951). The ENA analysis is ecosystem oriented and thus requires a 'quantified' food web and puts emphasis to all the species in the system under consideration. The attractiveness of ENA, as a step in merging system functioning and system structure, is that it may help us to judge the systems condition or quality by a set of quantitative system indicators.

Basic data needed for ENA are: biomass (B), physiological requirements (P/B), loss terms (respiration or dissipation, export, catch) and relationships between compartments (who eats what, whom and how much, Ulanowicz, 1986). These data are exactly the same data as necessary for building any dynamic ecosystem model. Information needed for the ENA food web analysis is the food web structure and compartment related data to enable the calculation of a number of indices. The main indices (cf. also Fig. 2) are: Total System Throughput (TST), Average Mutual Information (AMI) or the system complexity, Ascendency (A), Developmental Capacity (DC) and Overhead (O).

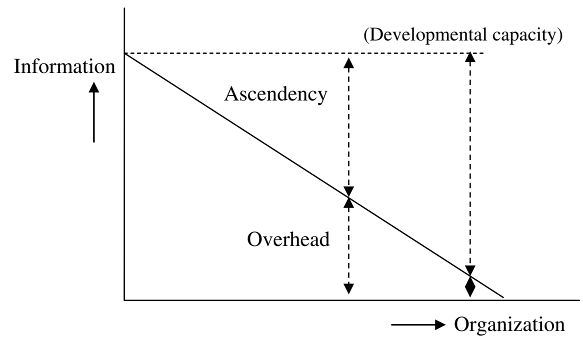

Fig. 2. Diagrammatic representation of the relations between a number of indices used in Ecological Network Analysis.

The Total System Throughput (TST) is represented by the sum of all transfers between all biotic compartments. The index is a quantification of the activity of the system. Overhead (O), an entropy term, is a measure of inefficiency of the material (carbon) flow through the food web. It is a 'disorder' term caused by the system 'dissipation' (e.g. respiration), the 'redundancy' of relations between species compartments and the 'export' from the system. System Development is the change in probability that material flows to one particular compartment instead of another compartment. The Developmental Capacity (DC) represents the 'complexity of the system' as generated by the number of combinations of the biotic compartments. The ascendency (A) is a measure of the efficiency of the material flow and can be expressed as (developmental capacity minus overhead). As shown in Fig. 2 these indices are all related to each other. At a given system specific level of Developmental Capacity (systems complexity) an increasing ascendancy corresponds to a decreasing overhead. When for example the A/DC is high then this means that thermodynamically the system is 'efficient' with only a few parallel pathways (low redundancy) in its food web structure. The Developmental Capacity or systems complexity also corresponds to another index the 'average mutual information' (AMI). Just as System Development this index represents the change in probability from unconstrained to constrained flows when weighted by the fraction this flow constitutes of the total system throughput. Scaling of the AMI by the total system throughput ($AMI \times TST$) results in an index for the systems organisation or ascendancy (A) and where $A = (AMI \times TST) = (DC - O)$.

Application of system flux analysis may be very helpful. As stated above, for a flux analysis (cf. also Fig. 3) the same information has to be collected as what is needed for conventional dynamic ecosystem modelling. The difference is that ENA is static while ecosystem modelling is dynamic. At present ENA is a system analysis based on a 'snap shot' survey. Since dynamic ecosystem modelling has got wide application in the management arena it is assumed here

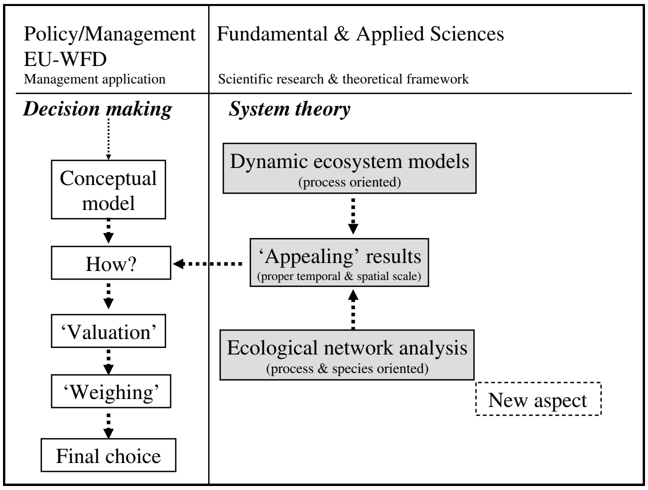

Fig. 3. Diagram relating aspects from system theory as process oriented dynamic ecosystem models and process and species oriented ecological network analysis to the decision making process.

that any extension of it by e.g. flux analysis related indices will not lead to a complete new approach but will add something new to an existing instrument or package (Fig. 3).

4. Area oriented monitoring as a strategy for merging ecosystem theory, ecosystem management and EU requirements

4.1. Present monitoring practice and the spirit of the EU water framework directive

As explained above the EU-WFD aims to protect the species structure and the functioning of aquatic systems by (1) improving the water quality and by (2) optimizing the habitat providing conditions. The effects of these measures consequently have to be judged by (3) evaluating the species structure and the functioning of the aquatic ecosystems in our transitional and coastal water bodies.

Until now the judgement of the systems quality or condition is based on mainly species oriented system indicators without a clear underlying theoretical concept (see above). Further quality judgement is based on chemical target monitoring and trend monitoring. These targets usually refer to a certain past situation which is called 'reference'. It has, however, to be concluded that the long-term ecological consequences of the use of these reference values or targets have not systematically been evaluated by extensive explorative studies. Once set these values were accepted as reliable and robust. After some time the monitoring program was even significantly reduced based on statistical analyses instead of thorough scientific ecosystem analyses (de Jonge et al., 2006). So far the functioning of the system has not played any role in judging the ecological status (see again de Jonge et al., 2006).

For biologists and ecologists it is clear that structure and functioning have to be evaluated in concert which is in principle possible when adding e.g. structure and process oriented ENA to the application of process oriented dynamic ecosystem modelling. However, the big question is 'how' since most countries aim for cost-effective monitoring which usually has to be interpreted as 'as cheap as possible'.

5. Logistic aspects of monitoring the Dutch Wadden Sea

In The Netherlands the majority of the monitoring of the water column and the intertidal flats in the Wadden Sea is following a station oriented approach and thus discrete sampling. The monitoring stations in the Wadden Sea (part of Dutch territory) are presented in Fig. 4. The area covers about 3000 km^2 and consists of at least six quite different tidal basins, the major Ems estuary (situated on the border between The Netherlands and Germany) excluded. Rijkswaterstaat (part of the Ministry of Transport, Public Works and Water Management) follows the temporal development of the water quality via 15 stations (1 per 200 km^2 or 2 per tidal basin) indicated as large circular symbols in Fig. 4. Five of these water quality monitoring stations are also used for assessing the phytoplankton species composition (Fig. 4 closed circular symbols). The total time needed to sample these 15 stations is between 42 h and 48 h (six calendar days). By ship it takes on average about 3 h to reach the next station. The operational cost of the employed ship is circa 26000 Euros per sampling period. The sampling frequency of the area varies between once and twice per month.

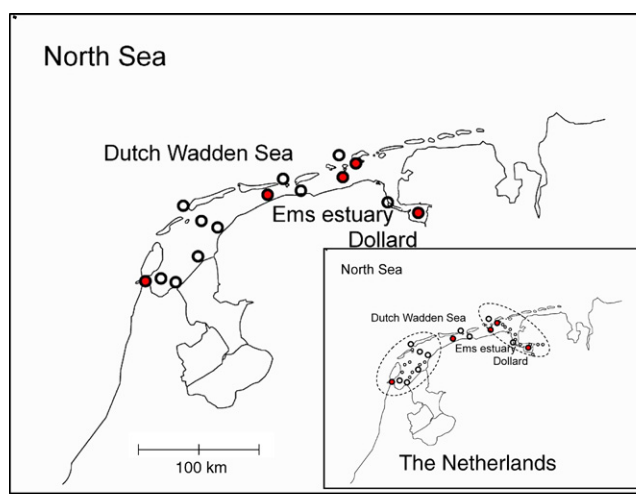

Fig. 4. Distribution of current monitoring stations in the Wadden Sea and the Ems estuary. The closed dots refer to the five stations used for water quality monitoring and phytoplankton monitoring while the large open dots refer to stations for water quality monitoring. The small open dots visualize the suggested increase in station numbers for surveillance monitoring. The additionally collected data can be used to feed, calibrate and validate process oriented dynamic ecosystem models as well as process and structure oriented tools to judge the systems biological and ecological quality or condition. Areas suggested to increase the density in monitoring stations are the western Dutch Wadden Sea and the Ems estuary.

The current monitoring program is only suited for assessing the concentration level against a given standard (target monitoring) and for assessing the trend per station (trend monitoring). The collected phytoplankton data can at present only be used to detect a harmful algal bloom once per fortnight. The data are not used to assess the quality of the ecosystem by e.g. species diversity. Although not explicitly discussed here, there is also other biological monitoring (cf. de Jonge et al., 2006) in operation with focus on macrozoobenthos and salt marshes. Additionally, other ministries have responsibility for monitoring bivalves, fishes, birds and mammals. These programs are, however, carried out in complete isolation from each other. Data files are stored independently so that integrative use of data is not easy.

Apart from the water quality monitoring the government often commits itself to what may be called 'problem oriented' research. This is project embedded research for a restricted period of time (up to ca four years) and with the aim to solve one particular management problem. On a structural basis investigations meant to increase the system knowledge and to help validate available simulation models is not carried out. However, in the policy making arena the process oriented dynamic ecosystem models are continuously applied to support the process of decision making. The available ecosystem models are usually not up to date and usually not fed with realistic up to date field data because these data are not collected under the responsibility of the Dutch government.

In The Netherlands the combination of a strong and still ongoing reduction in the number of employed civil servants and the recent paradigm of task separation (separating defining the rules or regulations, supplying the licences and control of the proper maintenance of the rules in accordance to the licences) has led to big problems in the professional functioning of these civil servant organisations. These governmental organisations are more and more dependent on 'the market' for maintaining their own professional knowledge. This market is, however, usually not equipped to fill this knowledge gap. Therefore the monitoring goals, the program and the organisations involved should be reconsidered.

6. Starting points for future monitoring

Environmental management will benefit from a long-term strategy with clear goals on the integral system (ecology and socio-economics), environmentally following a holistic or ecosystem approach, focussing on the proper temporal and spatial scales and with (biologically, ecologically and economically) a sound theoretical basis. This is usually more than we can offer or expect and therefore in a way the current monitoring activities should become less inflexible without loss of the information that is anyway collected today (de Jonge et al., 2006). The challenge here is then to anticipate in a cost-effective and also knowledge-efficient way a future which is uncertain in terms of ecosystem development, political and bureaucratic developments and requirements, demographic developments, further changes in land-use, etcetera. Against the background of the requirements of all available national and international legislation and the spirit of the EU-WFD this scene offers a future challenge which will be explained below.

In a previous paper de Jonge et al. (2006) explained how civil servant organisations can benefit from the academic community vice versa in a much more effective way then so far. That model has improved here (Fig. 5) to better clarify the role of the different parties involved. On the one hand there is, based on national and international agreements, the need for monitoring our systems while on the other hand up to date knowledge of that particular system is necessary to support any decision making process. Apart from this the monitoring process should be so flexible that the implementation of relevant exciting new findings from e.g. theoretical biology and ecology that contribute to a correct system judgement is possible. This can all be achieved by implementing the scheme of Fig. 5. In this diagram different institutions or organisations have different responsibilities. With focus on monitoring the government has the final responsibility and offers the logistical support. Contrary to now, part of the academic community takes part in the discussions on the design of the monitoring program and carries out the data interpretation. This is beneficial to the governmental services as well as the academic community because it will lead to a better understanding of 'monitoring' in the academic community

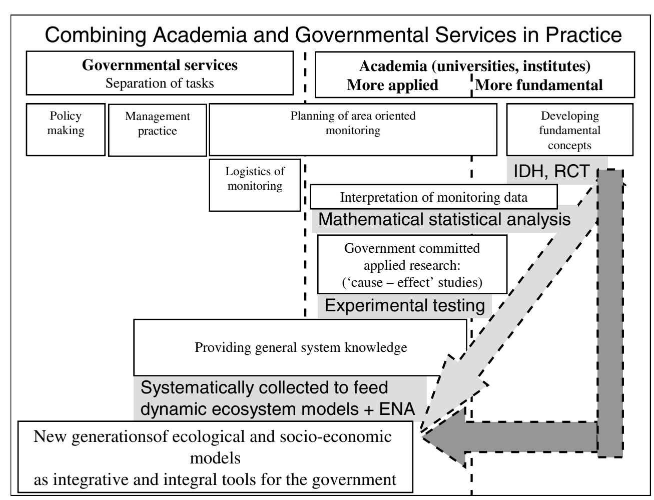

Fig. 5. Re-arrangement of some responsibilities of governmental services and academia. The government is responsible for part of the planning and the logistics of monitoring and keeps the final responsibility for the governmental monitoring. The academia gain responsibility for data analysis and further interpretation of monitoring results and 'cause-effect' studies. The building up of regional system knowledge suited for modelling exercises, the incorporation of new fundamental concepts and the judgement of the systems quality or condition becomes a shared responsibility of governmental organisations and academia. On top of this a strong additional input in the development of new 'future tools' is expected from the fundamental science (theoretical biology and ecology).

while the civil servant organisations will more directly benefit from the knowledge generated within the academia. Taking part in the data interpretation definitely will also lead to more interest in the investigations related to system monitoring, e.g. the WFD related investigative monitoring the cause-effect studies included. Ultimately the improved co-operation between both communities will even facilitate the communication on new concepts and theories from theoretical biology and ecology.

7. Cost-effective data collection

To feed ecosystem models, data on e.g. resources (light and nutrients) environmental conditions and chlorophyll-a are required. The phytoplankton species composition or the composition of available plant pigments may indicate the structure at the species level or the level of taxonomic groups and may even serve as a clue to elucidate the structure of the grazer's population. In addition to the primary producers and grazers also data on decomposers and carnivores should be present of course.

The required data for feeding ecosystem models is not available at the moment but can easily and cost-effectively be generated. Above, the time needed for monitoring the Wadden Sea, the number of stations and the costs related to ships time have been mentioned (cf. Fig. 4). Within six days the entire Wadden Sea is crossed by a (governmental) ship suitable to do any type of measurement. This ship can easily be equipped by a set of sensors to determine the temporal and spatial variation in turbidity (Optical Back Scatter/OBS, 240 readings per minute), oxygen (4 readings per minute), fluorescence (240 readings per minute) and parameters as temperature, pH and salinity (Conductivity and Temperature by a CTD). This type of environmental measurements is extremely important for defining the boundary conditions in computer simulation models, for calibration and validation purposes and is basically easy and very cheap to obtain. However, these automated readings need to be calibrated (chlorophyll-a fluorescence and turbidity) carefully for which the number of fixed stations should be increased (tripled). Increasing the number of stations is also required for a better description of the spatial variation. This increase in sampling stations can easily be executed since the navigation time is by far enough to minimally triple the number of fixed stations without any increase in the operational costs. Apart from this also the compounds necessary to let ecosystem models run should be measured afterwards. Apart from the factors mentioned above this also holds for nutrients. It also holds that while the information level is inclining, the absolute cost of it hardly increases because the measurements of all nutrient components (*ortho*-phosphate, nitrate, nitrite, ammonium and reactive silicate) in a running auto analyser take exactly 1 min per series of compounds, which is 1 min per sampled station. Further, auto analysers are like any other machinery; they have to run to prevent contamination and deterioration. Another aspect in connection to ecosystem models is formed by the necessity to sample the boundaries of the system. These are the places where water and organisms may enter or leave the system (boundary between the shallow Wadden Sea and the North Sea) and the freshwater inputs.

Table 1
Comparision between standard monitoring and the extended monitoring of the Ems estuary during part of the Dutch monitoring program

Example for water quality sampling in the Ems estuary (reality based)	Standard water quality monitoring	Extended water quality monitoring
Number of stations	4	12
Number of parameters	10	10
Number of data (algal counts included!)	140	116292
Cost of ships time	4350 €	4350 €
Cost per value per day (factor is 830)	31.0714 €	0.0374 €

A comparable approach can be chosen for the benthic part of the system but given the necessity to discuss this more principally the example for the water column may suffice here.

To be able to proceed in a cost-effective way we should not simply increase the number of monitoring stations all over the area but make choices about why and where. These choices may be based on available information for a particular tidal basin or its national or international importance. These should then be the areas to concentrate on (Fig. 4). Interesting areas in the Wadden Sea are the relatively well studied western part of the Dutch Wadden Sea (Marsdiep and Vlie tidal basin; Fig. 4) and the Ems estuary (Fig. 4) situated on the border between The Netherlands and Germany where main problems related to e.g. channel maintenance dredging (in Dutch and German territory) and river deepening (in German territory) leads to major turbidity problems in the entire estuary. An impression of the proposed increase in number of stations and its regional distribution is indicated by the small open symbols in Fig. 4.

The final bit is then the question 'what is cost-effective?' For the Ems estuary it has been calculated (Table 1) that increasing the number of monitoring stations between the weir in the river and the sea to 30 while deploying a chlorophyll-a fluorescence meter together with OBS and automatic oxygen measuring device by an optode system (PreSens, Regensburg, Germany) will increase the number of collected data by 116000 over one day. Based on only this value (CTD readings not yet included) the final price for every environmental value is then below 0.035 Euros instead of the price of over 30 Euros for the data collected during the standard monitoring activities.

8. Benefits of area oriented surveillance monitoring and changed responsibilities

The benefits of the new strategy are many (Fig. 5). The governmental services can concentrate on their new, but significantly reduced, working package compared to that in the past. Because of the new contract possibilities concerning the physical, chemical and biological monitoring (e.g. data interpretation, data handling, cause-effect studies, modelling) the institutions equipped for fundamental or more applied scientific research become stronger involved in environmental practices and problems. This means more intense contacts between the different organisations which is beneficial to both. Long-term contracts to universities and institutes, with objectively the best scientific reputation in that particular field, guarantees the necessary continuation in available 'know how' in field experience (know how to sample), analyses (especially required taxonomy qualities), data handling, mathematical and statistical analyses and interpretation of the results of it.

In summary the proposed approach and new working packages in Fig. 5 will:

1. Highly stimulate the conceptual input from the academic community.
2. Lead to significant increase in co-operation between governmental organisations and academia.
3. Lead to improved results of 'cause-effect' studies.
4. Compensate the negative effects of the reduction in the magnitude of the corps of civil servants.
5. Lead to a structural contribution in data collection for application in ecosystem models.
6. Contribute to create the required system knowledge for management purposes.

References

Belgrano, A., Scharler, U.M., Dunne, J., Ulanowicz, R.E. (Eds.), 2005. Aquatic Food Webs. An Ecosystem Approach. Oxford University Press, Oxford, p. 262.

Connell, J.H., 1978. Diversity in tropical rain forests and coral reefs. Science 199, 1302–1310.

de Jonge, V.N., van Beusekom, J.E.E., 1992. Contribution of resuspended microphytobenthos to total phytoplankton in the Ems estuary and its possible role for grazers. Neth. J. Sea Res. 30, 91–105.

de Jonge, V.N., van Beusekom, J.E.E., 1995. Wind and tide induced resuspension of sediment and microphytobenthos from tidal flats in the Ems estuary. Limnol. Oceanogr. 40, 766–778.

de Jonge, V.N., Elliott, M., Brauer, V.S., 2006. Marine monitoring: its shortcomings and mismatch with the EU Water Framework Directive's objectives. Mar. Pollut. Bull. 53, 5–19.

EC, 2000. Establishing a framework for Community action in the field of water policy. Directive 2000/60/EC of the European Parliament and of the Council. Official Journal of the European Communities L 327, 1–72.

EC, 2005. Framework for Community Action in the field of marine environmental policy (Marine Strategy Directive). Proposal for a Directive from the European Commission, Brussels, 24.10.2005, COM(2005)505 final.

Eleftheriou, A., McIntyre, A.D., 2005. Methods for the Study of Marine Benthos, third ed. Blackwell Science Ltd., Oxford, UK, 440 p.

Elliott, M., de Jonge, V.N., 1996. The need for monitoring the monitors and their monitoring. Mar. Pollut. Bull. 32, 248–249.

Finn, J.T., 1976. Measures of ecosystem structure and function derived from analysis of flows. J. Theor. Biol. 56, 363–380.

Gause, G.F., 1934. The Struggle for Existence. Williams and Wilkins, Baltimore, MD.

Geider, R.J., Osborne, B.A., 1992. Algal photo-synthesis. the measurement of algal gas exchange. In: Current Phycology, vol. 2. Chapman and Hall, London, p. 256.

Grover, J.P., 1997. Resource Competition. Chapman and Hall, London, UK.

Horn, H.S., 1975. Markovian properties of forest succession. In: Cody, M.L., Diamond, J.M. (Eds.), Ecology and Evolution of Communities. Belknap Press, Cambridge, Massachusetts, pp. 196–211.

Hutchinson, G.E., 1961. The paradox of the plankton. Am. Nat. 95, 137–145.

Huisman, J., Weissing, F.J., 1999. Biodiversity of plankton by species oscillations and chaos. Nature 402, 407–410.

Huisman, J., Weissing, F.J., 2001. Fundamental unpredictability of multispecies competition. Am. Nat. 157, 488–494.

Jörgensen, S.E., Müller, F., 2000. Handbook of Ecosystem Theories and Management. CRC Press, New York, 600 p.

Leontief, W., 1951. The Structure of the American Economy, 1919–1939. second ed. Oxford University Press, New York, 257 p.

Lindeman, R.L., 1942. The trophic-dynamic aspect of ecology. Ecology 23, 399–418.

May, R.M., 1973. Stability and Complexity in Model Ecosystems. Princeton University Press, Princeton.

Pimm, S.L., 1982. Food Webs. Chapman and Hall, London.

Roelke, D., Augustine, S., Buyukates, Y., 2003. Fundamental predictability in multispecies competition: the influence of large disturbance. Am. Nat. 162, 615–623.

Sanders, H.L., 1969. Benthic marine diversity and the stability-time hypothesis. Brookhaven Sym. Biol. 22, 71–81.

Strickland, J.D., Parsons, T., 1972. A practical manual of seawater analysis. Bull. Fish. Res. Board Can. 167, 1–311.

ten Brink, B.J.E., Hosper, S.H., Colijn, F., 1991. A quantitative method for description and assessment of ecosystems: the amoeba approach. Mar. Pollut. Bull. 23, 265–270.

Tilman, D., 1982. Resource Competition and Community Structure. Princeton University Press, Princeton, NJ.

Ulanowicz, R.E., 1980. An hypothesis on the development of natural communities. J. Theor. Biol. 85, 223–245.

Ulanowicz, R.E., 1986. Growth and Development: Ecosystems Phenomenology. Springer-Verlag, NY, 203 p.

Ulanowicz, R.E., 1997. Ecology, The Ascendent Perspective. Columbia University Press, New York.

Wolfe, G.V., 2000. The chemical defense ecology of marine unicellular plankton: constraints, mechanisms and impacts. Biol. Bull. 198, 225–244.