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Marine Pollution Bulletin 53 (2006) 5–19

MARINE POLLUTION BULLETIN

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Viewpoint

Marine monitoring: Its shortcomings and mismatch with the EU Water Framework Directive's objectives

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Abstract

The main goal of the EU Water Framework Directive (WFD) is to achieve good ecological status across European surface waters by 2015 and as such, it offers the opportunity and thus the challenge to improve the protection of our coastal systems. It is the main example for Europe's increasing desire to conserve aquatic ecosystems. Ironically, since *c*. 1975 the increasing adoption of EU directives has been accompanied by a decreasing interest of, for example, the Dutch government to assess the quality of its coastal and marine ecosystems. The surveillance and monitoring started in NL in 1971 has declined since the 1980s resulting in a 35% reduction of sampling stations. Given this and interruptions the remaining data series is considered to be insufficient for purposes other than trend analysis and compliance. The Dutch marine managers have apparently chosen a minimal (cost-effective) approach despite the WFD implicitly requiring the incorporation of the system's 'ecological complexity' in indices used to evaluate the ecological status of highly variable systems such as transitional and coastal waters. These indices should include both the community structure and system functioning and to make this really cost-effective a new monitoring strategy is required with a tailor-made programme. Since the adoption of the WFD in 2000 and the launching of the European Marine Strategy in 2002 (and the recently proposed Marine Framework Directive) we suggest reviewing national monitoring programmes in order to integrate water quality monitoring and biological monitoring and change from 'station oriented monitoring' to 'basin or system oriented monitoring' in combination with specific 'cause–effect' studies for highly dynamic coastal systems. Progress will be made if the collected information is integrated and aggregated in valuable tools such as structure- and functioning-oriented computer simulation models and Decision Support Systems. The development of ecological indices integrating community structure and system functioning, such as in Ecological Network Analysis, are proposed to meet a cost-effective approach at the national level and full assessment of the ecosystem status at the EU level. The WFD offers the opportunity to re-consider and re-invest in environmental research and monitoring. Using examples from the Netherlands and, to a lesser extent, the United Kingdom, the present paper therefore reviews marine monitoring and marine environmental research in combination and in the light of such major policy initiatives such as the WFD.

Keywords: Water Framework Directive; Ecological Network Analysis; Monitoring; Estuaries; Coasts; Ecological complexity; Indicators; Indices

1. Introduction

Monitoring de facto implies the rigorous sampling and re-sampling of an area, a habitat or a biological, physical or chemical component for a well-defined purpose and against a well-defined end-point (e.g. McLusky and Elliott, 2004). That aim may be the detection of a trend or the non-compliance with a threshold, standard, trigger value or baseline, thus leading to a well-defined (and agreed in advance) policy action. In contrast, surveillance can be a series of regular spatial and temporal analyses where the purposes and trends are defined a posteriori. The European Union Water Framework Directive (WFD; EC, 2000) makes a further distinction in that surveillance monitoring

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doi:10.1016/j.marpolbul.2005.11.026

is distinguished from operational monitoring, which is related to the compliance with standards, discharge limits, etc., and investigative (or diagnostic) monitoring, which is required to determine why an area is not complying with these standards or has changed as the result of human activities. The latter will therefore interrogate the cause of any spatial or temporal change in the component under consideration in cause and effect studies.

The growth of the EU has produced an increase in the harmonisation of the environmental legislation of the member states through the adoption of EU Directives advising and dictating how the member states may or have to act. The WFD is the most important water directive to date and has a main aim: "*prevents further deterioration and protects and enhances the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystem*". It distinguishes two complementary ways of reaching the goals: the optimization of the physical habitat-providing conditions and the further improvement of water quality. If a departure from reference conditions, defined as Good Ecological Status (GES), is detected then member states will be required to implement management measures and hence they will require sufficient monitoring systems in order to check if/when GES is met. Ecological status is '*an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters, classified in accordance with Annex V*'. The WFD will be required to work intimately with the Habitats and Species Directive (McLusky and Elliott, 2004) and also the recently adopted EU Marine Strategy (EC, 2005a) and recently proposed EU Marine Framework Directive (EC, 2005b).

The WFD's priority is to protect the ecological community structure, but also to protect functioning, defined as rate processes between species under specific environmental conditions. The protection of both species and their activities within the entire river basin district requires a holistic approach or, failing that, a focus on elements acting as surrogates for this holistic view, hence an indicator approach. However, as yet we (a) do not yet have sufficiently robust or well-explained indicators, and (b) do not know what the best or most suitable surrogate is in these highly variable coastal and estuarine (transitional waters) environments. Are they, for example, the dominant species, key link species or characteristic species? Table 1 lists the habitat-providing, morphological and physico-chemical conditions and the biological elements involved in the required monitoring, our perception of what the creators of the Directive had in mind and what has to be monitored in practice, as proposed by the WFD. The Directive contains weaknesses in that it assumes sufficient knowledge of community structure in relation to its quality. It focuses on selected groups of species (Table 1) without addressing why other groups (micro organisms, meiofauna, zooplankton) are not included and therefore it assumes that the existing system indicators are sufficient to judge the system's status. The absence of suitable system indicators is, however, one of the main problems of many existing monitoring programmes. Therefore it is surprising if an umbrella directive such as the WFD is implemented to focus on particular species, namely either nuisance species or species indicative of a well-understood environmental response, particular species groups or landscape elements as listed in Table 1.

During the early stages of implementation of the WFD, the EU launched the Marine Strategy (EC, 2005a) whose aim is "*to promote sustainable use of the seas and to conserve marine ecosystems*". This was adopted as a first step towards a proposed Marine Framework Directive (MFD, EC, 2005b) because marine biodiversity is under significant anthropogenic pressures such as physical engineering, physical and chemical pollution and the introduction of invasive species. As with the WFD, a Marine Framework Directive will require areas to be tested against reference

Table 1
List of environmental elements, means and symptoms and parameters to be measured during the monitoring programs as required by the EU Water Framework Directive

Quality elements	Means and symptoms	Parameter and frequency
Morphology		
Geomorphology of system	Mean for realizing optimal habitat providing conditions	Soundings once per six years
Physico-chemical conditions		
Temperature, oxygen, salinity, nutrients, pH	Means for realizing optimal habitat providing conditions	Water sampling once per three months
Other pollutants		Sampling once per three months
Priority substances		Sampling once per month
Biology		
Phytoplankton	Toxic blooms	Species composition and countings twice a year
Macro-algae	Algal mats	Species composition and countings every three years
Macrophytes	Loss of habitat	Surface area of eelgrass beds and saltmarshes every three years
Macrobenthos	Anoxia	Species composition and countings every three years
Fish (transitional waters)	Migration barrier	Species composition and countings every three years

conditions although for the latter these are proposed as Good Environmental Status rather than Good Ecological Status. This will require:

1. Improvement of implementation of old and new legislation in an integrated way.
2. Coherent marine policy based on an ecosystem-based approach.
3. Improvement of knowledge on the quality status of European seas and the procedures and methodologies to assess this information.

In general the formulated goals of the WFD (and a resultant MFD) are clear but the strategy of how to evaluate the system's quality is not. The WFD (EC, 2000) has created new opportunities but also uncertainties (Elliott et al., 1999). For example, there are misunderstandings of text interpretation and missing elements to be monitored (zooplankton, microbial food web, benthic microalgae). The legal text has to be implemented and interpreted by scientists and managers but with some leeway because of subsidiarity, for example managers could apply for derogation in order to reduce the monitoring. However, this would not be following the spirit of the Directive and may lead to a delay in implementing management measures.

Member states are required to achieve GES by 2015 following the assessment of the quality of River Basin Districts. Given this, the assessment has to consider the quality of the community structure in our coastal waters and transitional water bodies as well as the interrelationships between all species and thus the functioning of the system. At this early stage of implementation there is a strong call for alternative paradigms and strategies to be followed (de Jonge et al., 2003). The tools (appropriate monitoring, manuals, protocols and measures) necessary for achieving the WFD aims are only now being developed (e.g. Rosenberg et al., 2004) and implicitly rely on a good knowledge of the ecosystem functioning under certain stressors. The challenging aspect of the WFD is its holistic or integral approach and so, using examples from the Netherlands and the United Kingdom, below we describe important knowledge gaps by confronting the status of current monitoring with the main goal of the WFD. We further analyse alternatives for monitoring transitional and coastal water bodies so that the main goal of the WFD (the attainment of good ecological status and its inherent assumption of the effective protection of ecosystem structure and functioning) can be fulfilled.

2. Historical aspects of monitoring

2.1. General development

Monitoring has developed over time from a focus on a single factor measured at a restricted number of stations to complex programmes covering many areas and many physical and chemical parameters and biological species. In our countries, this included developments in basic water quality parameters until the 1990s, the development of investigative monitoring which was then followed by programme reductions because of sharper and restricted targets (trend and compliance monitoring) and the political requirement to be cost-effective.

The first compounds monitored in the Dutch marine system were nutrients in 1973 as these were responsible both for a greatly increased production and partly the observed reduction in species diversity. Due to accidental discharges, for example with copper sulphate and dieldrin in 1965 (Roskam, 1966; Koeman, 1971), persistent contaminants were added to the monitoring programme. The programmes started as surveillance monitoring as there was no generally accepted threshold values for true compliance monitoring. Marine protection and monitoring then became recognized more widely in the early 1970s when several important agreements, conventions and directives were adopted, for example IMO, 1948; ICES, 1964; RAMSAR, 1971; Oslo Convention, 1972; Paris Convention, 1974; HELCOM, 1974, London Dumping Convention, 1974, Barcelona Convention, 1976; REMPEC, 1976; IPCC, 1988 and OSPAR, 1992 (see also Ducrotoy and Elliott, 1997). Pollution related to inputs at the tidal limits was the main drive for harmonised monitoring of estuaries, especially in order for governments to fulfil their obligations by reporting polluting load estimates to the relevant bodies (e.g. OSPAR in the NE Atlantic). These inputs were calculated either, as in NL, based on measurements in the estuarine salinity gradient from where the apparent influx was calculated by applying the approach of Officer (1979), or, as in the UK, based on the loadings as the product of measured concentrations at the tidal limit of estuaries and the riverine flow rates. Since the mid-1970s compliance monitoring and trend monitoring has been recognized as important instruments in both NL and the UK. They have, however, constantly been accompanied in NL by reductions in the monitoring programmes (Table 2) and regular changes in monitoring intensity in the UK. Annual quality assessments started in 1993 with reports on the water quality of the Dutch state waters followed by the first 'Quality Status Reports' of the North Sea (NSTF, 1993) and the Wadden Sea (de Jonge et al., 1993a,b). In the UK, the National Marine Monitoring Plan was commenced in the late 1980s by establishing estuarine and coastal stations for chemical, benthic and biological effects measurements (http://www.jncc.gov.uk/pdf/nmmp-2ndreport.pdf).

2.2. Water quality monitoring until early 1970s

The Dutch physical water monitoring in river systems started in 1901 because of the need to quantify the water discharges of the river Rhine for safety reasons and thus to support policy, plans and related decisions. A good example is the engineering or re-configuring of the Dutch coastline by the Deltaplan (this plan was to separate all

Table 2
Change in number of sampling stations for water quality monitoring in Dutch coastal waters from 1964 onwards

Year/Stations	North Sea	Wadden Sea	Ems estuary	Schelde estuary	Oosterschelde	Total stations	European Union adopted directive
1964				11		ca 87	
1971		5	*30*			ca 111	
1973		*50–60*				*ca 166*	
1975							75/440 Nitrate
1976		25	20				76/160 Bathing waters 76/464 Dangerous subst
1977							77/795 Waste water
1978							78/659 Fresh water fish
1979							79/409 Wild birds 79/923 Shellfish
1981		11	11	12	*14*	124	
1975–1982	*76*						
1983–1987	23				10		
1988–1993	26	10	6	*15–18*	11	71	92/43 Habitat 00/60 WFD 01/42 Strategic Env Ass

Compiled based on Heesen (1995). As a comparison the adoption of some relevant EU directives in the 1970s is also indicated.

the estuaries in the south-west of NL from the sea, except the Schelde), the closure of the Wadden Sea and in between the two regions the existence of a mixture of natural dunes and artificial seawalls. The monitoring of basic water quality parameters (BOD, oxygen, faecal coliforms) in the rivers started in 1950 at four stations along the river Meuse. Systematic monitoring of the water quality in coastal areas started because of the increasingly bad quality of many aquatic systems in the 1960s. At the start of the programme the authorities were interested in the distribution and the levels of compounds along the Dutch coast and in the Wadden Sea. Expansion of the monitoring activities were triggered by accidents with large scale consequences such as the first major oil spill—the *Torry Canyon* in March 1967, which covered England's south-west coast with 119,000 tons of crude oil. This exposed the poor preparedness for clean-up and the poor scientific knowledge behind oil spill responses, for example the use of detergents which proved to be as harmful as the oil itself.

In the 1960s the production and use of xenobiotic compounds, detergents and artificial fertilizers exceeded the assimilative capacity of the natural freshwater systems. These systems changed rapidly from having a high biodiversity to only a low number of often opportunistic species. Estuarine and eventually coastal ecosystems tended to follow the same pattern. While freshwaters were increasingly monitored for biological effects, estuaries were not, although they also received high loads of pollutants such as the high loads of organic matter from the Dutch north-east leading to hypoxia in the Ems estuary (BOEDE, 1985). The monitoring of coastal waters would have been necessary, since they behave differently due to their size and physical characteristics. In the UK, prior to the implementation in 1985 of the 1974 Control of Pollution Act, few estuaries and no coastlines were designated as controlled waters, i.e. licensing of discharges, and the accompanying compliance monitoring, was not carried out in these areas. Similarly, prior to the Dutch Pollution of Surface Waters Act (NG, 1970) the main requirements for industrial estates along the coast were only two:

(1) to discharge to coastal waters and
(2) to prevent any interference of electrical equipment with broadcasting activities.

2.3. Start of investigative monitoring in the early 1970s

Localised investigative monitoring studies were also started to accompany the large-scale assessments. For example, in 1973 the project Biological Study of Potato Waste Effluent was started because of plans to utilize the Ems estuary (Fig. 1) with its 100 km^2 shallow Dollard basin for the degradation of organic effluent; this became the BOEDE project (Biological Research Ems Dollard Estu-

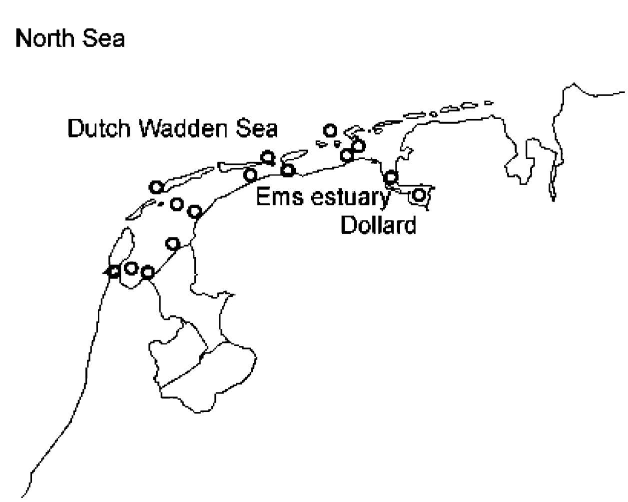

Fig. 1. Current distribution of sampling stations of the Dutch Monitoring Program in the Dutch Wadden Sea and the Ems estuary.

ary). The effluent, equivalent to 20×10^6 inhabitants, was derived from the local potato flour and strawboard industries (BOEDE, 1985) and the project was meant to support national policy-making based on local expertise. In this area the scientific research, advice and monitoring developed together over 12 years which led to an enormous local scientific expertise culminating in a dynamic process-oriented ecosystem model (Baretta and Ruardij, 1988). The project results were used by the ministries of 'Planning & Environment', 'Transport, Public Works and Water Management', 'Agriculture Nature Conservation and Food Quality' and 'Economic Affairs'. They made it possible to gather insight into the ecological structure and functioning of the area. This project assessed a local discharge point for organic matter as well as the distribution of sludge from the harbour of Delfzijl, the effect of channel maintenance dredging in the Ems estuary (de Jonge, 1983) and the effects of nutrients on the estuarine production (Baretta and Ruardij, 1988; de Jonge and Essink, 1991). The results finally supported the closure of any major effluent discharges to the Ems estuary or other receiving water bodies in NL. The ecosystem model produced by BOEDE was similar in concept to the UK GEMBASE (General Ecosystem Model of the Bristol Channel and Severn Estuary) developed in the late 1970s (Radford and Joint, 1980; Radford, 1981). Although the latter was an academic exercise, it achieved importance when a tidal power barrage was proposed across the Severn estuary. The BOEDE model was then used as a second check on its outcome (Baretta and Ruardij, 1988). Despite the benefits of the combination of local monitoring and scientific research, the BOEDE project was terminated because of financial reasons in 1985 and the local monitoring network further reduced from 30 stations in 1970 to 6 in 1988 (see below and Table 2).

2.4. Budget driven programme reductions

Because the Dutch monitoring programme was considered by the National Institute for Coastal and Marine Management/RIKZ (Directorate General Rijkswaterstaat, Ministry of Transport, Public Works and Water Management) as being costly and 'data rich but information poor', a major and budget-reduction driven revision was started in 1992 by a task team. The sampling effort was reduced in extent and intensity on the basis that no statistically significant information loss would result from the revision. The sampling stations were reduced 35% from 400 to 260; the use of monitoring as surveillance was considered as not being of value and the use of monitoring data for purposes (e.g. investigative) other than compliance monitoring and trend monitoring was discussed but not accepted as being of importance. As an example, the present monitoring network for water quality in the Wadden Sea is presented in Fig. 1. After the programme reduction, the most important conclusion by Laane et al. (1996) was that '*since the information needs were now well established more policy related information could be gained for less money*'. Ironically, at that time the European Commission started preparing for the WFD resulting in new requirements for monitoring. Consequently, as the result of the reorganisation of the Dutch monitoring, new developments on ecological indicators were based on less-available information (Kabuta and Laane, 2004).

The budget-driven reduction of the monitoring programme has indirectly been stimulated by the way authorities interpreted their tasks. It has been due to carrying out the required duties in a very strict and rigorous way: analysing the available data sets mainly statistically thus without sufficient interpretation within a wider environmental context and with large gaps (zooplankton, micro organisms, phytoplankton, microphytobenthos) in knowledge. This in turn is due to a disinterest of many governments in the ecological functioning of our coastal systems as shown by reductions in budgets for this sort of research. This is one of our main worries but at the same time it is also the challenge for implementing the WFD (see below).

Given the limited resources made available for Dutch and British monitoring, a hierarchy and prioritisation of monitoring appeared to be developing in the statutory bodies. Most importantly are the obligations resulting from international agreements related to the North Sea Task Force and OSPAR, and in NL on agreements in the International Scheldt Commission, the Ems-Dollard treaty (International Ems Commission) and the Tri-lateral Monitoring Programme in the Wadden Sea (co-operation between Denmark, Germany and NL). Second, operational monitoring as required by national legislation such as for land-based and vessel discharges, including sewage, dredging and industrial waste. Third, wider surveillance monitoring and, lastly, special investigations and diagnostic monitoring. This prioritisation led to large scale, and therefore of concern, changes in the monitoring programmes. Table 2 illustrates the changes in the number of Dutch water quality monitoring stations since 1964 and reflects the Dutch government's decreased attention for the quality of aquatic ecosystems. In contrast, in the same period, the European Commission increasingly showed more interest in the loss of biodiversity and habitats and there has been an accompanying move to the adoption of the Ecosystem Approach (cf. the global Convention on Biological Diversity, http://www.biodiv.org). The attitude changed from the 'need to know' how it works (e.g. environmental projects running in the north east and in the south west of the NL) into an administrative one of 'need to do' following obligations due to international agreements (see above). The large-scale reductions in the Dutch monitoring programme were justified on the basis that they still provided sufficient coverage. This, according to Laane et al. (1996), included the objective selection of concentration gradients, representative concentration distributions over large areas, future problem areas, operational aspects, available discharge points, laboratory capacity and quality, and international directives. In contrast, and driven by intense political discussions on the

Schelde, the number of monitoring stations there was slightly increased (Table 2). This example and that from the north-east of NL suggest that developments in the monitoring programmes respond to current pressures rather than assuming a responsibility to safeguard the environment.

Although of course monitoring programmes need to be (re)evaluated regularly (van Zeijl et al., 2001) and their goals reconsidered (Elliott and de Jonge, 1996), these are seldom scheduled and even more seldom carried out. Reconsideration of any monitoring programme is often not a formal step defined in the policy life cycle.

2.5. Changing goals

The increased demand by authorities for more cost-effective operations also caused other changes such as from an initial general surveillance and investigative monitoring in NL to operational monitoring such as compliance and trend monitoring (Laane et al., 1996). On the one hand this led to clarification and better specification of the previously more generally and thus less well-formulated targets. On the other hand this type of monitoring is so specific that it strongly reduces the multiple use of data, such as feeding simulation models by time series or by generating values to set model boundary conditions (see the example below related to Fig. 3). Furthermore, the current monitoring also provides a restricted insight in the condition of the system because of its inflexibility. This is due to the rigid distribution of monitoring stations and the reduced possibility to gather additional data from other localities and dates.

2.6. Cost-effectiveness and the degrees of freedom provided by the European Directives

EU Directives give the framework but the member states have the freedom of determining the method of implementation (so-called *subsidiarity*) and hence there is latitude for deciding the strategy for WFD monitoring. The monitoring in the Directive explicitly requires the structural elements of phytoplankton community, macroalgae, macrophytes, benthos and, for transitional waters, fishes and it implicitly includes functional aspects such as primary production. There is, however, no explicit requirement to monitor other groups or trophic levels, no matter how important, where the main material fluxes occur such as the zooplankton, or the microbial primary producers. These are important in carbon production or the microbial food web and also in their role determining the species structure. The zooplankton grazers link primary producers to the appealing or charismatic system predator components (usually birds and fish), and thus those of greatest relevance to managers, policy makers and the public. Hence, as there is no obligation to monitor the basis of the food web sufficiently there is also no opportunity to explain any bottom-up governed change in e.g. the abundance of grazers and carnivores. Thus, authorities may be concentrating more on monitoring the easier structural components rather than the context given by structure and functioning of the system.

3. Quality aspects of monitoring programme

3.1. Appropriateness of time series

The quality of time series data is usually lacking because it has many missing months' data or even missing years (e.g. Fig. 2A and B) thus allowing only a sub-optimal analysis of the patterns. The third example (Fig. 2C) shows the change from a chlorophyll-*a* time series to reactive silicate data. The simultaneous monitoring of the two, however, would have been more valuable because of the link between phytoplankton growth and the availability of reactive silicate.

3.2. Appropriateness of sampling station density

There are many more failures and omissions in available data sets, for example, in the Ems estuary only three stations are left for judging the water quality of the system (Fig. 3). These three stations do not sufficiently cover the entire salinity gradient thus missing information about the mixing between fresh and sea water; hence it would be important to have a more detailed picture about its distribution in the estuary. Moreover, as shown by Fig. 3, the distribution of ortho-phosphate in the estuary is so complex such that three stations would not provide a good overview of any development. Based on the current monitoring, no interpretation of the concentration gradient is possible nor the calculation of the influxes using the method of Officer (1979). The actual gradients are usually even more complex than in this example (van Beusekom and de Jonge, 1994). The National Marine Monitoring Plan (NMMP) in the UK similarly uses only 3 salinity-stratified stations within each estuary together with intermediate and coastal stations so while it gives good large-scale (country-wide) coverage, the resolution within any single estuary-coastal area is poor (details at http://www.jncc.gov.uk/pdf/nmmp2ndreport.pdf). This lack is particularly important given the high spatial heterogeneity in estuaries (McLusky and Elliott, 2004).

4. Consistency of monitoring programme

4.1. Current programmes

The UK NMMP and the Dutch biological monitoring programme (BIOMON) have been operational since 1990. The NMMP aims to link community assessment (benthos) to other biological effects, such as indications of detoxification mechanisms and oyster-embryo assays, and the causes of change such as sediment contaminant analyses. These inter-linked determinands should then give an overall indication of the health of UK estuaries and

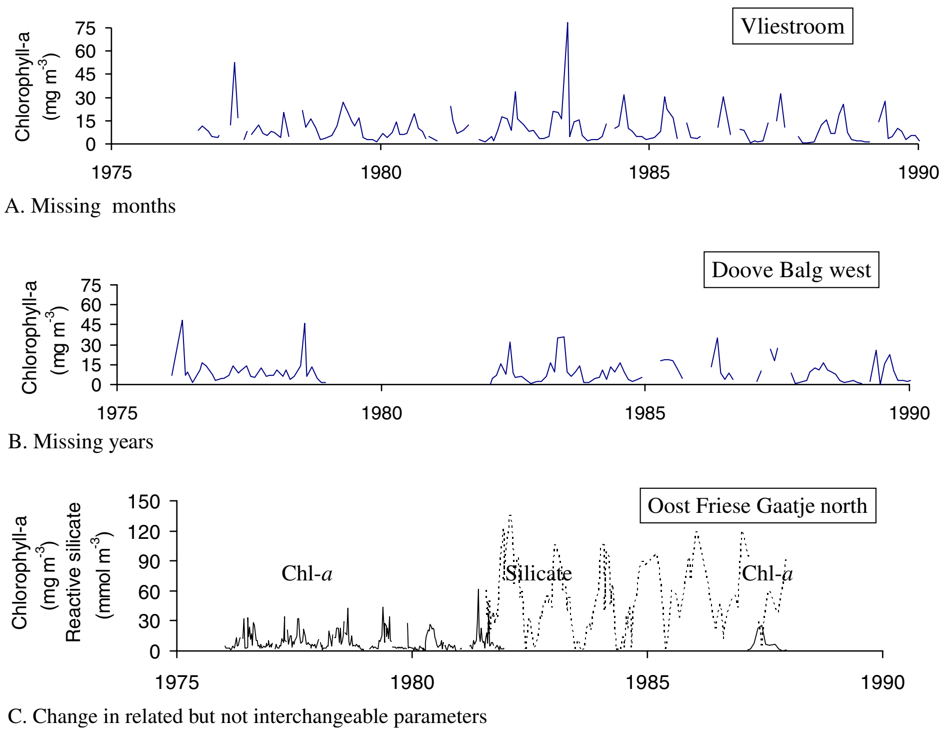

Fig. 2. Time series for chlorophyll-*a* and reactive silicate as generated by the Dutch Monitoring Program. At site "Vliestroom" the chlorophyll-*a* series is frequently interrupted for up to several months (A). At site "Doove Balg west" the chlorophyll-*a* time series is interrupted for many years followed by periods of many missing months (B). At site "Oost Friese Gaatje north" an exclusive switch from chlorophyll-*a* to reactive silicate occurred, followed by combined measurement of the two parameters for one year after which the monitoring series stopped. Data from Rijkswaterstaat.

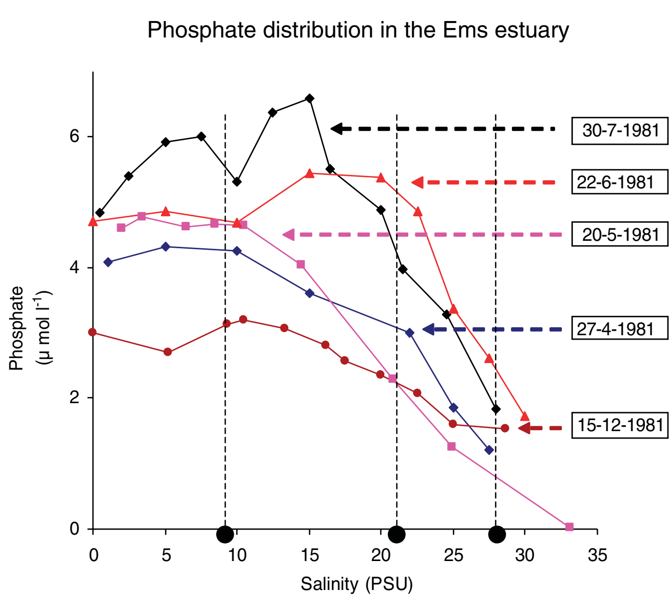

Fig. 3. Ortho-phosphate distribution as a function of salinity (redrawn from de Jonge and Villerius, 1989). The approximate position of the three present monitoring sampling stations is indicated by the black circles on the *x*-axis and the dashed lines. Values collected at these three points cannot represent the proper gradient in substances over the estuary. Consequently the apparent river influx (load) cannot properly be inferred from only these three points.

coasts. Similarly, species and organism groups are monitored in all Dutch brackish and marine waters (Table 3; Yland, 1995). However, in construction and execution, the monitoring approach of the Dutch coastal systems is internally inconsistent. It represents a transect- and a station-oriented approach in the Wadden Sea and a region-oriented approach in the Delta area in the south-western part of NL (Fig. 4).

There are further large differences in the biota monitored between the different systems (Table 3). Some groups or parameters are not monitored (bacteria, primary production, zooplankton, fishes) whereas other groups are monitored for only some systems (microphytobenthos, meiobenthos, birds, sea mammals). In other cases, groups are monitored in nearly all coastal systems and the North Sea (macrozoobenthos, macrophytes) whereas most groups responsible for the majority of the material fluxes, organisms smaller than 1 mm, are missing in this approach. However, as in the WFD, the components which are supposed to integrate changes over time and which are temporally and spatially more stable have emphasis, especially the macrozoobenthos, whereas the more variable and less-integrative components are not monitored as intensively. Indeed if any of the biological elements required for monitoring is shown to have such a high variability that its use for determining Good Ecological Status is compromised then the element can be discounted. Although macrozoobenthos is included for its integrative properties, it also shows a large interannual variation. These aspects make it difficult to reach conclusions about the quality,

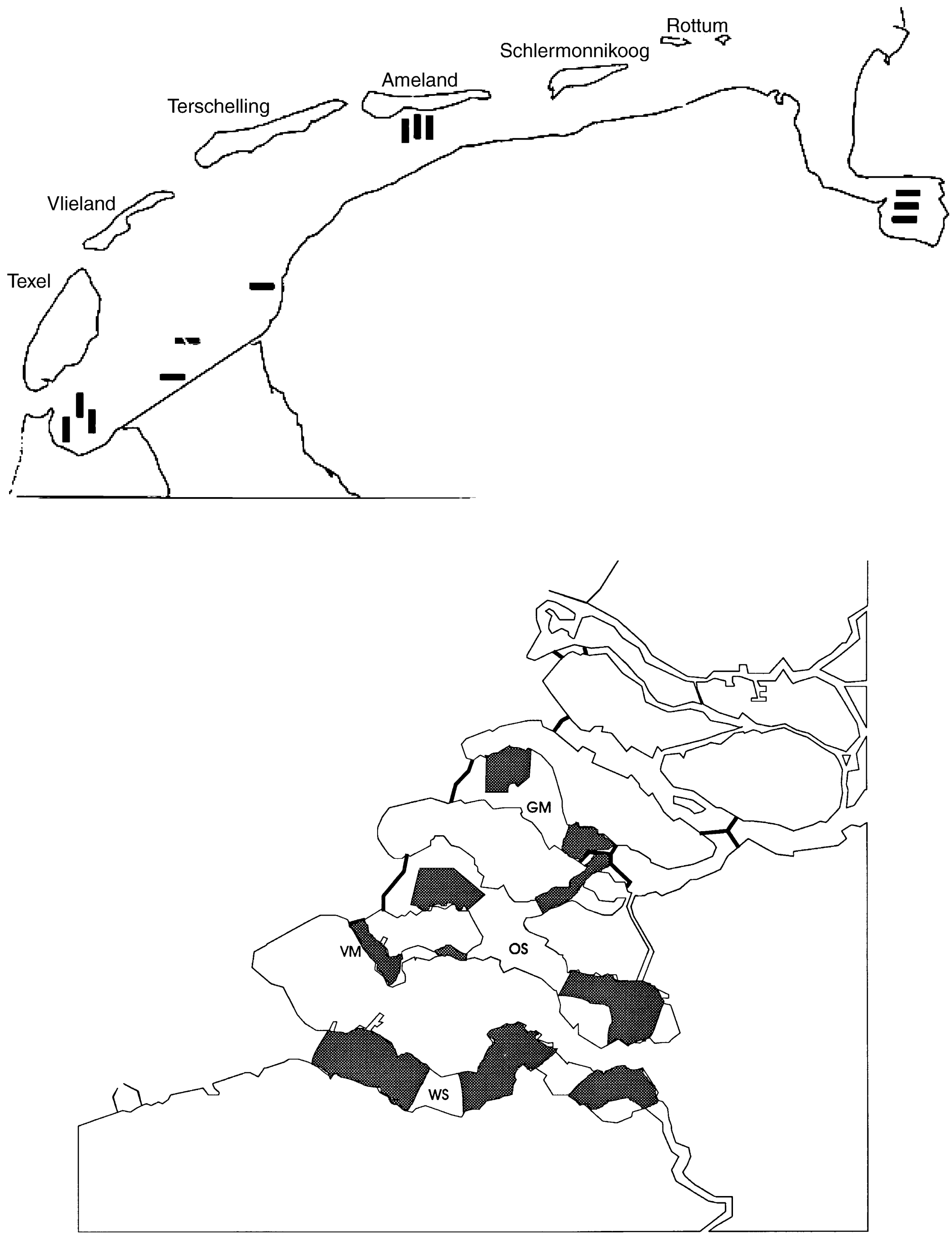

Fig. 4. The transect (lines) and station (dots) approach in the Wadden Sea (upper panel) compared to the region approach in the south-western part of The Netherlands (lower panel).

condition or health of an area such as the Wadden Sea. Hence is it suggested here that we should devote more effort on biological systems analysis in combination with all the abiotic conditions before we can decide on suitable species-based system indicators.

4.2. Requirements of the WFD

The WFD requires monitoring a mixture of structure elements (species), structure-oriented elements (salt marshes, sea grasses) and process-oriented parameters (chlorophyll-*a* as a surrogate for primary production). The answers derived from the measurements, however, are not based on a thorough understanding of the structure and the functioning of the available food web. There is a sufficiently good understanding of the qualitative aspects of the local food webs and the variation in it but, unfortunately, there is no sufficiently coherent understanding of the functioning of the aquatic systems on which to base rational management measures.

Table 3
Groups of organisms as monitored in eight different coastal systems in The Netherlands in 1993

Groups of organisms	Monitored Dutch water systems							
	NS	WS	ED	LG	OS	W	LV	VD
Phytoplankton	Black	Black	Black	Black	Black	Black	Black	Black
Micro zooplankton	Black	White	White	White	White	White	White	White
Primary production	Grey	Grey	Grey	Grey	Grey	Grey	Grey	Grey
Macro zoobenthos	White	White	White	White	White	White	White	White
- littoral	White	Black	Black	White	Black	Black	White	White
- sublittoral	Black	Black	White	Black	Black	Black	Black	Black
Macrophytes	White	White	White	White	White	White	White	White
- salt marshes	White	Black	Black	White	Black	Black	White	Black
- seagrasses	White	Black	Black	Black	Black	White	Black	White
Microphytobenthos	White	White	Black	White	Black	White	White	White
Meiobenthos	Black	White	White	White	White	White	White	Black
Hard substrate	White	White	White	Black	Black	Black	Black	White
Zooplankton	Grey	Grey	Grey	Grey	Grey	Grey	Grey	Grey
Breeding birds	White	White	White	Black	Black	Black	Black	Black
Water fowl	White	White	White	Black	Black	Black	Black	Black
Sea birds	Black	Black	White	White	White	White	White	Black
Sea mammals	Black	Black	White	White	White	White	White	Black
Fishes	Grey	Grey	White	White	Grey	Grey	White	Grey
Bacteria (degraders)	M	M	M	M	M	M	M	M

Compiled based on Yland, 1995. Black: in operation, Grey: under consideration for execution, White: missing, M: missing and not considered, NS: North Sea, WS: Wadden Sea, ED: Ems estuary, LG: Lake Grevelingen, OS: Oosterschelde, W: Westerschelde, LV: Lake Veere, VD: Voordelta.

The consequence of the choices made in the past—mainly focussing on a selection of structure elements, with only an indirect link to functioning and with the perceived aim of reducing as much as possible the cost of the monitoring programme—is that the EU member states are not able to meet the requirements as formulated by the OSPAR Commission (1993) in terms of Ecological Quality Objectives (EcoQ) "*An overall expression of the structure and function of the marine ecosystem taking into account the biological community and natural physiographic, geographic and climatic factors as well as physical and chemical conditions including those resulting from human activities*" (OSPAR Commission, 1993) nor those by the WFD (EC, 2000). It is acknowledged here that there is insufficient funding to measure and monitor everything and so there is the need to achieve cost-effective monitoring and thus to rely on surrogates for the detecting of change. This in turn implies that one still has to search for the most suitable system indices requiring a thorough analysis of the ecological food web of our systems over several years. Implicitly the EU acknowledges the problems of judging the system's biological and ecological quality and has devolved this to the member states although there are Common Implementation Strategies to ensure a comparable approach. Despite this, there is a dilemma in needing to judge quality and measure anthropogenic change in highly variable systems. Intuitively, scientists know and recognise good or poor quality, such as a reduction of benthic biomass or removal of eelgrass beds or biogenic reefs, but the challenge is to express these as legally-defendable definitions of anthropogenically-induced change. One questions whether the present monitoring can only detect large statistically significant change and unusual occurrences or whether it has the ability, in these variable estuarine and coastal systems, to measure and detect subtle change.

It has long been clear that the available operational evaluation techniques such as the use of a set of species and other quality indicators fail in judging the ecological quality of the system (de Bruin et al., 1992; van der Windt, 1995). Thus, we need to find other ways to judge the system's condition. As is acknowledged by the WFD, albeit as a last resort, there will be the increasing need to resort to 'expert-judgement' in the absence of a fully-quantitative approach, (i.e. does the estuary look and function as an estuary based on a rapid and skilled-eye assessment). However, as indicated here, whether for the Wadden Sea or the UK waters, the available time series, the spatial distribution of the sampling stations for water quality and biology and the selected system aspects measured are usually insufficient to describe the system's quality.

4.3. Harmonisation of national monitoring, WFD and other EU directives

The WFD, together with the EU Habitats and Species Directive (HSD), which is aimed at protecting representative habitats and vulnerable species, requires a coherent monitoring programme, which meets the prerequisite for the necessary judgement of the system's condition which in turn is meant to contribute to the protection of the systems' structure and functioning. Here we consider that the WFD should firstly optimise the habitat providing conditions and secondly assess the water quality. The third step is to assess the quality of the structure, the species composition, and its functioning, e.g. carbon fluxes within and through the food web, and find the most suitable surrogates for indicating the system quality. This will require new system indices and could, for example, be achieved by the further development and application of Ecological Network Analysis (see below). The final step is then to judge whether or not the system can be considered to be in a good condition (as Good Ecological Status for the WFD, Favourable Conservation Status for the HSD or Good Environmental Status for the proposed Marine Framework Directive (EC, 2005b)). A main difficulty is that the WFD appears to have been developed from a freshwater perspective and then extended to the more variable estuaries and coasts, and the HSD from a terrestrial basis before being extended to waters thus causing difficulties in the approach. One questions whether the monitoring proposed for the implementation of the WFD is aimed at fulfilling the Directive rather than ensuring good ecological status.

At present we have several tools to evaluate the system's status (cf. Kabuta and Laane, 2004), e.g. quantitative methods for description and assessment of ecosystems: the AMOEBA approach (ten Brink and Hosper, 1989; ten Brink et al., 1991) or Nature Target Types (NTT; Bal et al., 1995). The latter concept is derived from terrestrial systems and focuses on the combination of aspects such as *rarity*, *declining presence* and *dependency on the Dutch environment*. The AMOEBA is a structure approach, which indicates the relative abundance of species that are assumed to be 'key species' although the central role of these species is not proven. This approach needs to be further developed in order to link species and their functioning within the system. The development of indicators has increased in recent years (e.g. Rogers and Greenaway, 2005; Aubry and Elliott, in press) and other numerical approaches include the recently developed benthic indices (AMBI: Borja et al., 2003; BQI: Rosenberg et al., 2004) which are based on sediment organic enrichment paradigms. At the other end of the spectrum we can monitor the presence and network of biotopes (e.g. see http://www.ukmarinesac.org.uk; Connor et al., 2004) as an indication of the good functioning of water bodies. Either we do not sufficiently know what the chosen indicators stand for (as in AMOEBA) or we do not even know the place or position of the majority of the species within the approach chosen (as in NTT). Hence we have to search for better and more appropriate system indicators to assess the condition of highly variable systems such as estuaries and coasts. However, while aiming for such an ideal, we realistically acknowledge that with the available resources, the final outcome is likely to be a compromise.

4.4. Future developments and improvements to monitoring

The inherent spatial and temporal complexity and variability of coastal ecosystems will always present problems for meaningful monitoring. This holds for the larger (and less problematical) elements as well as the small pelagic and microbial components. It can be argued that for coastal waters there is a mismatch between can be meaningfully monitored and what the WFD requires hence we have to search for other and better indices than are presently available which are based on a better understanding of the relation between structure and functioning of these coastal ecosystems.

The frequency of monitoring is a more intractable problem in that although it is detailed in the WFD, that frequency may have little biological relevance and in particular the proposed minimal frequency of twice a year (phytoplankton) or once per three years (macrozoobenthos, fish, macrophytes, macroalgae) is inconsistent in terms of natural spatial and temporal variability, management actions or decision making. For example, the chances of missing toxic or noxious algal blooms, considered here to be the reason for including this component in the WFD monitoring, is very high under a biannual sampling regime. Similarly, a triennial assessment of the fish community in transitional waters will miss important features due to large inter- and intra- annual variation. In addition, the spatial intensity is not given and so again left to the judgement of marine scientists in competition with the financial parsimony of managers. These dilemmas require to be resolved for the final implementation of the WFD.

It is necessary to incorporate into monitoring the complex behaviour of the ecosystem's species structure and functioning and thus also the full recognition of related existing dynamics (e.g. May, 1976). Systems may behave apparently predictably or may behave apparently chaotic (unpredictable). Both aspects are important as has been demonstrated theoretically (Huisman and Weissing, 2001) and experimentally (Roelke et al., 2003; Becks et al., 2005) that systems may express both types of behaviour due to existence or absence of e.g. 'external disturbances' (e.g. input of nutrients by changing freshwater inputs or changes in light). Together with this is the responsibility to maintain biodiversity by preventing large areas from the effects of human disturbance (EU Habitats & Species Directive; WFD; Gorshkov et al., 2004).

4.5. Operational aspects of future harmonised monitoring

It is suggested here that simple monitoring programmes may be insufficient to assess the system's quality. Given the aims of the EU 'key directives' (WFD, HSD), the possibility of a Marine Framework Directive, and also OSPAR agreements, in general we need to integrate water quality monitoring and biological monitoring with applied system research (see below). Applied system research will come into play during the investigative (diagnostic) monitoring indicated by the WFD in cases of failure of compliance with an agreed standard or quality threshold (i.e. that an area is at less than Good Ecological Status). The precursors to this—surveillance and operational monitoring—will be covered within the United Kingdom respectively by the monitoring carried out under the NMMP and the monitoring required by licence compliance testing. Licence compliance includes authorisation, consents, permits and licences for discharges, environmental impact assessments and resource exploitation (aggregates, dredging, etc.) (McLusky and Elliott, 2004). Given that statutory bodies in the UK, such as the environmental protection agencies, are operating under financial restrictions, they have already indicated that the monitoring budget cannot be increased, especially not for operational monitoring. Consequently, monitoring tasks will increasingly be passed to operators (companies, industries, contractors) under the 'polluter pays principle' and thus unfortunately be carried out strictly locally and in isolation from integration at the ecosystem level.

There is yet another dilemma in trying to reconcile the scientist's desire to fundamentally understand the system, especially the bottom-up processes. In contrast, the manager/policy-maker's concerns are mainly regarding the high level (top–down) responses, hence their focus on the charismatic species such as birds, fishes and mammals. The monitoring to date pays little attention to the bottom-up processes even though the components with the largest material fluxes are the primary producers in the water column and on the intertidal flats, the organisms involved in the microbial food web in water and sediment, the meiofauna, the microzooplankton and the macrozooplankton. Given the natural dynamics and the implicit requirements of the WFD, however, these phenomena require to be included as part of interrogating both the structure (species presence) and what they do (functioning). We will make progress if we integrate and aggregate the collected information in attractive 'tools' such as computer simulation models and Decision Support Systems. Physical and chemical models are used to good effect by environmental protection agencies, consultancies and other NGO's especially as a visual representation of problems and solutions whereas biological or ecological models have been little used in decision support or making. Thus if we restore our previous monitoring efforts and combine this with 'cause–effect' studies and ecological modelling, we reach the stage of 'multiple data use'. One can argue that this ultimately will lead to the best possible insights in how the system is structured and how it works. However, the dilemma is that monitoring of ecosystem health has to satisfy different purposes, e.g. health and quality, overexploitation of areas and components, and habitat loss (Elliott and Cutts, 2004).

4.6. The way ahead

The move towards a functional and ecosystem approach is in line with the goals of the WFD, HSD, the EU Marine Strategy (and the proposed Marine Framework Directive (EC, 2005a,b) and ICZM strategy and the desires of the habitat protection part of international conventions (such as Annex V of OSPAR). All of these have the common aim to protect our ecosystems by obtaining and evaluating data in order to judge system structure and its functioning. However, this relies on the ability and the will to modify and extend the present monitoring practices and to move from 'station-oriented monitoring' to 'basin or system-oriented monitoring' in combination with specific 'cause–effect' studies for these highly dynamic coastal systems.

There are several possibilities for the future development of monitoring:

(1) Current monitoring of environmental quality is maintained and a structural approach such as the AMOEBA approach (ten Brink et al., 1991) is used to evaluate the system's condition in the hope that we achieve environmental sustainability while recognising that the coastal and estuarine systems are highly variable and have already significantly changed due to human activities (cf. de Jonge et al., 1993a,b). In essence, there is the need to compare change against reference conditions, yet, especially in the developed countries, pristine, unchanged conditions rarely exist. The latter have been documented only anecdotally which means that there is no suitable natural reference situation available on which to rely. The only certainty is that in the previous system differed strongly from the present one.

(2) The 'habitat approach' is followed in which physical (mean depth, surface area, current velocity, wave length, tidal range, etc.) and physico-chemical properties and values (salinity, temperature, nutrient levels, etc.) are combined and subdivided in several classes representing potential habitat entities (de Jong, 1999). This should be analogous to the definition and determination of reference conditions according to typologies in the WFD. It may be more cost-effective and efficient to monitor the causes of change rather than the change itself but often the cause of the change observed is unclear and is therefore not 'monitorable'.

The potential habitat entities can be established with the available composition of species in the water column and sediments (e.g. benthic macrofauna as by Ysebaert et al., 2002 using logistic regression analysis). In doing this, realistic habitat entities can be distinguished, e.g. the biotope approach (Connor et al., 2004), in which the condition assessment will bring together the above initiatives (WFD, Habitats Directive, Marine Strategy, etc.). Apart from cataloguing habitats, the biotope approach of the UK and the Dutch HABIMAP method also allow effects of human activities and measurable habitat properties such as situation and extent to be tested or explored. This means that the method can also be used in a retrospective way, and tested and applied in decision making. The use of this habitat concept has not found a broad acceptance among those involved in European policy preparation. This may be due to the difficulty in describing adequately the ecological communities in the coastal systems because of the highly dynamic nature of these systems. There are also differences in e.g. the time scale of life cycles and succession in associations or communities of microalgae, macroalgae and that of benthos in either deep or shallow waters.

(3) A broader functional ecosystem-wide view is taken with the further development and application of functional modelling systems such as Ecological Network Analysis (ENA) (Fig. 5; Ulanowicz, 1980, 1986, 1997; Baird et al., 2004; Belgrano et al., 2005). ENA can be defined as a meta-analysis of foodwebs/ecosystems, which analyses the direct and indirect trophic effects between species or functional groups in a quantitative way, and quantifies competitions and mutualisms, and energy/material transfers (carbon, phosphorous, nitrogen, energy, information) and losses between and from trophic levels; it identifies cycles, quantifies material throughput through cycles of various lengths, quantifies recycling, and describes the organisation and resilience as whole system indices using information theory. The method allows the calculation of different system aspects spanning from the species to the community and ecosystem level. At the ecosystem level, the combination of structure and function are integrated in whole system indices which may be less sensitive to small scale changes at the species level and more functional than community structures indicated by species lists. The ENA method of ecosystem analysis investigates the system properties in a 'snapshot' which means that insights into temporal changes have to be obtained by repetitive analysis of the ecosystem over various time steps. The approach can by used in combination with dynamic system modelling, using its outcomes as input data and providing new starting points for dynamic models.

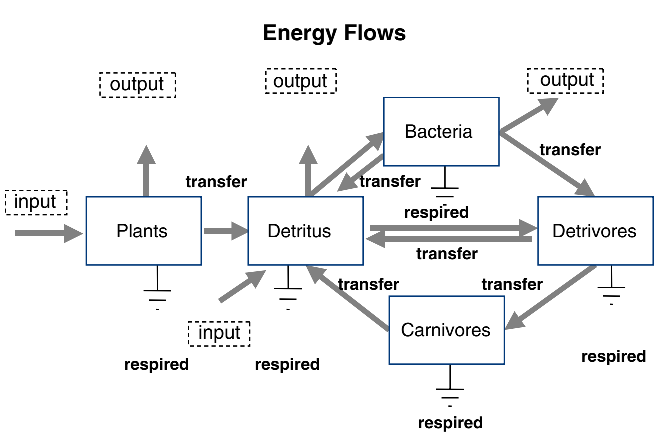

Fig. 5. Diagram of a hypothetical food web structured at the trophic level and indicating the information needed for any thorough analysis to calculate the required system indicators. The boxes represent (mean) biomass of the distinguished groups of organisms and the arrows the fluxes per unit of time.

Examples of practical applications of ENA to highlight environmental impacts include e.g. the evaluation of estuarine conditions before and after the construction of a dam (Baird and Heymans, 1996), of nitrogen overloading in an estuary (Christian and Thomas, 2003), and of changes in freshwater inflow into estuaries (Scharler and Baird, 2005). Although ENA has been used as an academic approach for over 2 decades, it has not yet been adopted as a management tool. This may be because of its complexity or the fact that it requires either a large amount of data which may not be available or, failing that, relies on extrapolations from other systems which thus reduce its ability to detect site-specific change. In addition, at present it is questioned whether we know sufficient about the species in a system to functionally link all the elements. Although it is not possible to fully resolve all trophic interactions between all species in an ecosystem, the insights gained into ecosystem functioning using this approach are nevertheless valuable. Further development is required to detect the key factors and hopefully also the key species acting on ecosystem stability and resilience. Perhaps, eventually, techniques such as ENA may be applicable as a part of a management tool and decision support system. As suggested by Baird et al. (2004), there is the need to carry out further pilot studies to derive the required functional indices as operational tools and to clarify the different elements to be assessed in surveillance, operational and investigative monitoring.

The alternatives presented above indicate that in order to fulfil all management demands for coastal and transitional waters there needs to be a comprehensive approach offering several components (Table 4). This pragmatic approach fits within the recommendations and definitions of the WFD but extends the approach to continue to provide greater understanding of the estuarine and coastal system. Despite the competing demands for understanding change to ecosystems, and despite the inertia in the monitoring/assessment system, the present monitoring requires to be made more rational, coherent, rigorous and streamlined but especially more comprehensive. Additional and more sophisticated approaches and new techniques are required together with an increase in monitoring spatially and temporally (cf. also Kabuta and Laane, 2004) until we reach the system approach. We accept that not all aspects can be monitored at all areas and at all times and that we have to fulfil all competing aims. In practice, this means a change from an ecoplot (small area) and station approach to one covering an entire tidal basin or estuary. This could be reached by reconsidering the present monitoring network and the placing and number of stations. More effective remote techniques such as airborne and satellite survey should be used but with adequate groundtruthing by conventional techniques.

Table 4
The components of effective monitoring and assessment programmes

Components of monitoring	Features	Suggested body responsible
Wide-scale low level approach of surveillance	of all areas based on the potential for change, e.g. the number of anthropogenic stressors, followed by low-effort level (expert judgement) biological assessments	regional statutory environment agency or municipality, (with out-sourcing of the field and laboratory work to a specialised laboratory)
Process-orientated approach	for the understanding of compliance with licences in areas where there is permitted human activity which has potential to change the system	under the 'polluter-pays principle' to be paid for by developers/industries; carried out by independent specialists
Functional approach in investigative and diagnostic monitoring	hypothesis-driven approach, in order to determine causes of non-compliance and also to provide further fundamental knowledge	carried out by independent specialists, in dedicated research laboratories
Selection of case-study areas	in which the functioning is studied in relation to the structure to provide fundamental knowledge of the processes in affected and non-affected systems	carried out by dedicated research laboratories with input from the statutory agencies

4.7. Merging harmonisation and 'cost-effectiveness'

The above suggestions require that the management of monitoring is also addressed. Many countries, especially those such as the UK and NL which have a long history of estuarine and coastal management, have a fragmented and diverse monitoring portfolio. This requires to be remedied, for example by creating one large administrative 'model' from which the role, tasks and responsibilities of different organisations may emerge (Fig. 6). For example, the NL has changed the structure within the different ministries and policy making has been strictly divided from environmental management. However, evaluation and interpretation of monitoring data is still part of the duties of the managing part of the responsible organisation (in NL the Directorate General Rijkswaterstaat as part of the Ministry of Transport, Public Works and Water Management). As the Ministry is down-sizing by delegating some of their original tasks to other bodies, whether regional or private, and thus reducing the number of civil servants, this may be an appropriate time to review the way in which monitoring is organised. This is particularly necessary as, in countries such as NL and the UK, there is the declared aim that WFD implementation will be cost-neutral, i.e. the costs of any new monitoring either has to be borne by developers and other stakeholders or paid for by a reduction in existing duties. It is of note that environmental protection agencies, such as those in NL, Sweden, the US and the UK, have reduced or even lost their in-house field and laboratory science capabilities which have been passed to external,

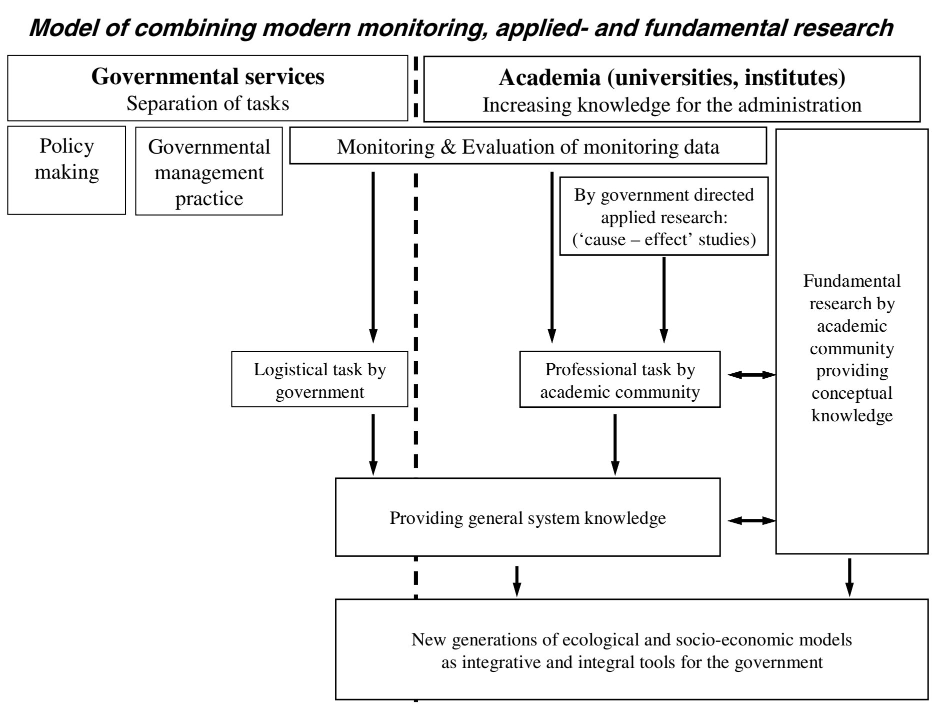

Fig. 6. A diagram illustrating the responsibilities of the government and the academic community. Indicated are the different proposed responsibilities and tasks for the governmental services and the academic community to optimise the effectiveness and the efficiency of policy supporting monitoring and research.

independent bodies such as research laboratories; the latter may be within academia or industry. While statutory bodies such as ministries and agencies, for legal reasons, have to retain overall control of statutory monitoring, the execution, interpretation and evaluation of those programmes and their results should be carried out by dedicated laboratories, whether in universities or elsewhere. This model was used previously in Sweden where the three regional marine monitoring programmes were placed with university marine laboratories (Professor Rosenberg, pers. comm.). The monitoring would thus incorporate an academic, hypothesis-driven, rigorous approach and it would ensure that the results are disseminated and thus are more widely available (thus overcoming the criticisms made earlier, see Elliott and de Jonge, 1996). This new approach has the major advantage compared to the past in that it will stimulate both the quality of the data interpretation and 'concept development' in fundamental research.

Finally, to allow a more cost-effective monitoring, recent suggestions in NL and the UK indicate a move towards making publicly-funded data publicly available. If this proposal is adopted then this would enable a wider database to be used for ecosystem modelling as well as for the initial primary purpose of ecosystem assessment and compliance with standards.

5. Conclusions

1. Monitoring programmes have to be carried out in a consistent, well-defined and rigorous way with regard to parameters, spatial extent and sampling frequency.
2. Species lists alone cannot judge the biological quality in highly dynamic and anthropogenically impacted coastal systems.
3. Implementation of ecosystem-wide approaches such as the WFD require indices covering the entire biological system, especially its functioning as well as its species composition.
4. The responsibilities for planning, sampling, chemical analysis, statistical analysis, evaluation and wider interpretation should involve the wider scientific community including both governmental services and independent academia.

Acknowledgements

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

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