PERGAMON

Continental Shelf Research 20 (2000) 1655–1686

CONTINENTAL SHELF RESEARCH

Importance of temporal and spatial scales in applying biological and physical process knowledge in coastal management, an example for the Ems estuary

Victor N. de Jonge*

National Institute for Coastal and Marine Management, P.O. Box 207, 9750 AE Haren, The Netherlands

Received 10 June 1999; received in revised form 8 December 1999; accepted 5 January 2000

Abstract

Managing intertidal mud flats requires a knowledge about the physical and biological structure of these systems, their physical and ecological roles and the development and influence of socio-economic activities on these systems. In this paper it is shown how, against the background of the strong natural variation, the effects of dredging and anthropogenic eutrophication can be simplified, represented and applied for management purposes. Freshwater (nutrient) discharges differently but linearly affect the annual primary production in the three (Lower, Middle and Dollard) estuarine reaches. The differences in effect along the estuarine axis are primarily caused by the turbidity gradient. Channel dredging affects the slope of the logarithmic turbidity gradient between the turbidity maximum with relatively stable concentrations in suspended matter, and the North Sea near the barrier islands. The direct consequence of a varying slope of the gradient is that the most seaward situated reaches of the estuarine system are more strongly influenced by channel dredging than the upstream reaches. The relative effects of both channel dredging and eutrophication become more manifest in the lower reaches of the estuary than in the upstream parts. Based on the correlations found, the expectation is that mean annual river discharges lower than ca. 35 $m^3 s^{-1}$ will turn the entire estuarine system to full heterotrophy. Results from experiments studying stabilization and destabilization by organisms are helpful to conceptual development, but are not helpful to any decision maker because the spatial scale of these systems usually does not fit the needs of an environmental manager who usually has to consider entire systems. Considering the aspect of scales results in the conclusion that at the process level (primary production process) the time

* Corresponding author. Tel.: + 31-50-5331359; fax: + 31-50-5340772.
E-mail address: v.n.djonge@rikz.rws.minvenw.nl (V.N. de Jonge).

PII: S0278-4343(00)00042-X

scale in the Ems estuary is mainly determined by the seasonal cycle and much less by factors like wind and tide. From the management perspective and considering seasonality, the longer term effects of both anthropogenic eutrophication and channel dredging can be best judged based on an annual time scale. Weather conditions and human activities both impact either large parts of the estuary (wind resuspension, eutrophication) or the entire area (dredging), something that is due to the long tidal travelling distance in comparison to the length of the estuary. This results in the necessity to consider large areas when judging the environmental impact of the above-mentioned parameters. The available information about the socio-economic subsystem and the natural subsystem can be integrated in instruments like management knowledge systems or decision support systems to help us to structure discussions in prioritizing between policy options.

Keywords: Temporal scales; Spatial scales; Biological processes; Physical processes; Coastal management

1. Introduction

1.1. Study area

The Wadden Sea is a shallow inshore area along the east coast of the North Sea with a surface of roughly 6000 km^2 and extending from the Netherlands to Denmark. The Dutch part of the Wadden Sea with an area of approximately 3000 km^2 is situated between the North Sea and the mainland by dikes. The Wadden Sea is intersected by a number of estuaries of which the Ems, Weser and Elbe are the most important.

The strong dynamics in the area are caused by freshwater run off (salinity changes, inputs of weathering products as minerals and nutrients), vertical and horizontal tides (oscillating water currents, mixing and transport of dissolved substances and particles) and seasonal cycle resulting in varying day lengths, wind (drift currents and resuspension and deposition resulting in variation in turbidity), cloud formation (variation in daily irradiation) and temperature (variation in biological activity).

Some of the substances that are carried to estuarine areas are derived from human activity which may positively (the nutrients phosphorus and nitrogen) or negatively (contaminants like heavy metals, PCBs, TBT, HCH etcetera) influence the functioning of the various organisms.

Mainly, the estuary of the River Ems will be used as an example in this paper because of the extensive information available (see e.g. Baretta and Ruardij, 1988; de Jonge, 1992a, b). Still, it is believed that the results have general applicability. The Ems estuary is located at the border between the Netherlands and Germany (Fig. 1) and is situated in an agricultural area. The estuary connects the northeastern part of the Netherlands and the northwestern part of Germany with the North Sea and forms an important navigation route for sea-going vessels and river ships. There are three important harbours: Eemshaven, Delfzijl and Emden.

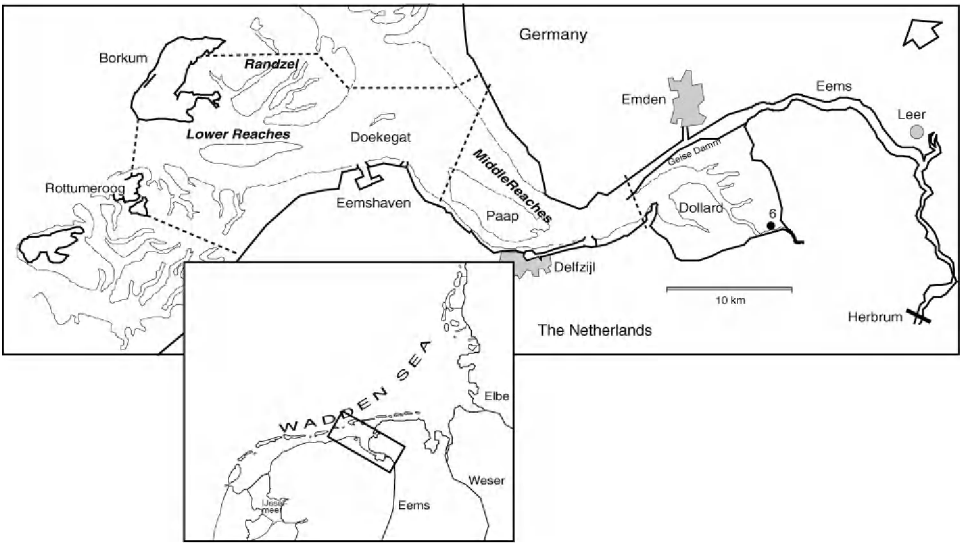

Fig. 1. Map of the Ems estuary.

1.2. The abiotic environment

The river Ems has a length of 330 km. It drains an area of 12 650 km^2 and has a long-term average discharge (1941–1971) of approximately 120 m^3 s^{-1} (based on data by Hinrich, 1974). The weir near Herbrum (Fig. 1), some 100 km from the barrier islands Borkum and Rottumeroog, separates the river Ems from marine influences and can be regarded as the head of the estuary. The estuary covers approximately 500 km^2 of which nearly 50% is intertidal flats. An exception is formed by the Dollard of which 80% is covered by intertidal flats. The distribution of the sediment composition is given in Fig. 2 and indicates that the estuary contains areas varying from very muddy to very sandy. The tidal prism between the barrier islands is approximately 10^9 m^3. The mean tidal range, an over-time-varying parameter (Führböter, 1989), increases from 2.3 m near the barrier islands to 3.2 m at Emden (de Jonge, 1983).

The complicated geomorphology and strong tidal currents result in a complex pattern of water currents with a remarkable residual current pattern (de Jonge, 1992a, b), steep average gradient in salinity and nutrients and characteristic longitudinal distribution of suspended matter (Fig. 3). Although it is very well known that contaminants may strongly contribute to the disfunctioning of estuarine ecosystems this factor cannot be included in the present paper because of lack of useful information.

1.3. Biological characteristics and relevant ecological processes

1.3.1. Biotic characteristics

The estuary is relatively rich in habitats (Fig. 4). Available characteristics are temporarily available small and very wind-sensitive (unstable) beds of the blue mussel

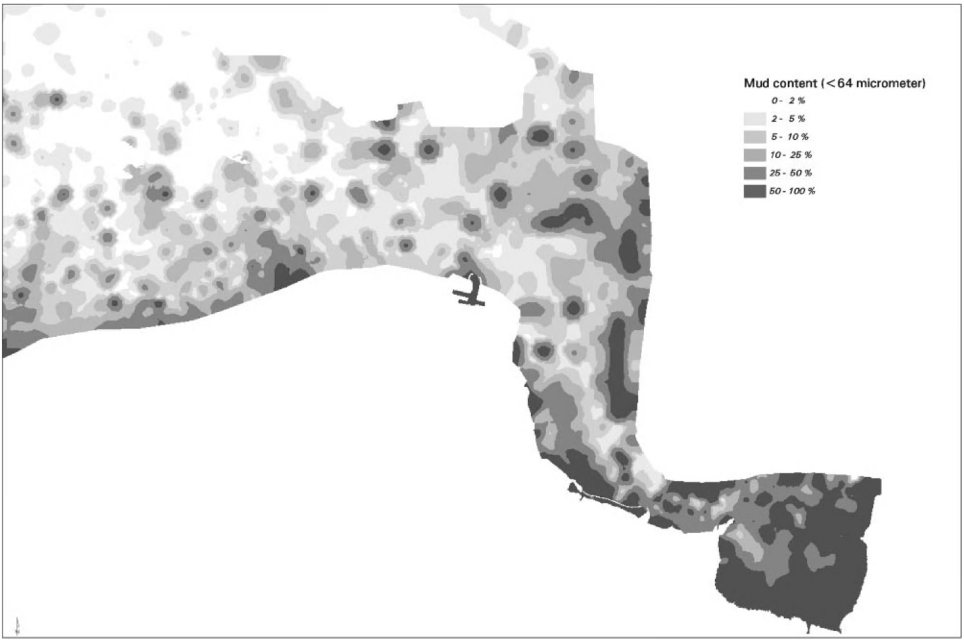

Fig. 2. Map of the sediment distribution in the Ems estuary.

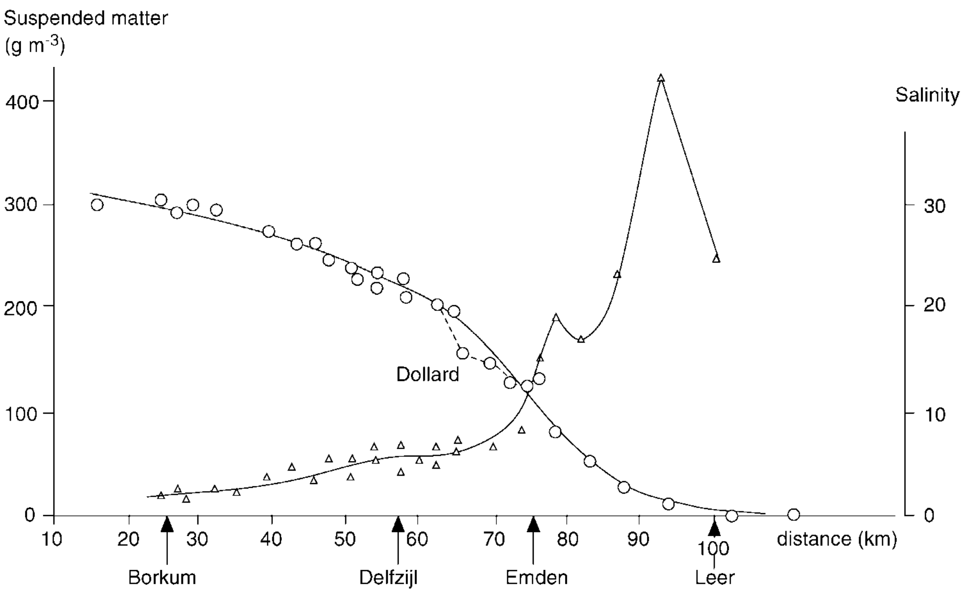

Fig. 3. Longitudinal distribution of salinity and suspended matter in the Ems estuary.

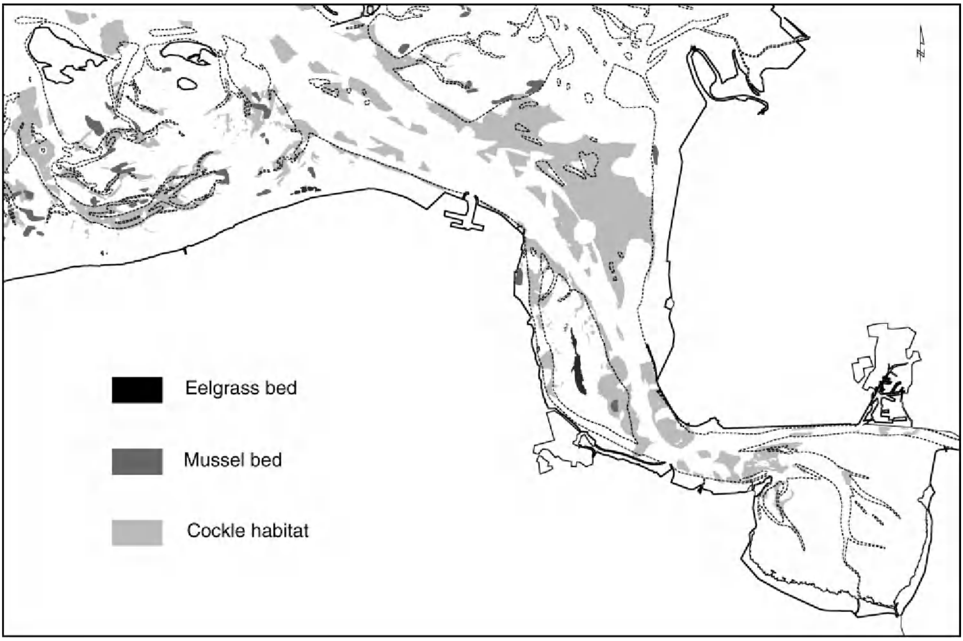

Fig. 4. Map of eelgrass bed in 1997 and available beds of blue mussel in 1990 in the Ems estuary at a background of a habitat map for edible cockle.

(*Mytilus edulis*) along small creeks on the Paap in the central part of the estuary and on the Randzel in the Lower Reaches. Suitable habitats are available for the edible cockle (*Cerastoderma edule*) and on the Paap intertidal flat (Fig. 1) a recently developed intertidal bed of eelgrass (*Zostera marina*) of over 1 km^2 (Fig. 4; de Jonge et al., 2000) is present. Another visually recognizable component is the microphytobenthos, mainly consisting of benthic diatoms, that temporarily gives the intertidal sediments its brown colour.

The production of organic carbon in the Ems estuary is realized by mainly microalgae. These algae consist of microphytobenthos living on and in the intertidal sand and mud flats and phytoplankton that lives in the water column. The microphytobenthos mainly consists of benthic diatoms. The phytoplankton consists of two categories: strict phytoplankton species composed of benthic diatoms and flagellates that both live in the water column during their whole life. The phytoplankton categories are temporarily supplied with tychoplankton mainly consisting of resuspended benthic diatoms. As will be shown later (cf. de Jonge and van Beusekom, 1995) the resuspended microphytobenthos is present in the water column in changing amounts due to the wind conditions. This phenomenon has consequences for estuaries that may, from the perspective of the functioning of ecosystems, be considered as wind dominated.

The primary production (P^B) is a function of a number of factors and can in a very general way be written as

$$P^B = f_1(\mathrm{Chl} - a), f_2(\mathrm{nutrient}), f_3(\mathrm{light}), f_4(\mathrm{temperature}), f_5(\mathrm{respiration}), f_6(\mathrm{grazing}) \ldots .$$

For more detailed information the reader is referred to specific literature on modelling efforts (Kremer and Nixon, 1978; Platt et al., 1977; Platt, 1981; Baretta and Ruardij, 1988; Platt et al., 1990, 1991; Cloern, 1999).

1.3.2. Role of nutrients

A characteristic longitudinal distribution of the nutrient concentrations in the estuary is given in Fig. 5 and is derived from a recent study by van Beusekom and de Jonge (1994). Apart from the light conditions (see below), the available nutrients determine the actual growth and even partly the species composition of the microalgal assemblages. On a seasonal basis the nutrient concentrations may vary considerably to values as low as near the detection limit. The interannual variation may also be large as can be seen from data presented in Fig. 6 and Figs 4.2 to 4.5 in de Jong et al. (2000).

1.3.3. Role of light

The light conditions in the estuary are not stable but vary due to a number of natural as well as human-induced factors. This leads to a mean gradient changing

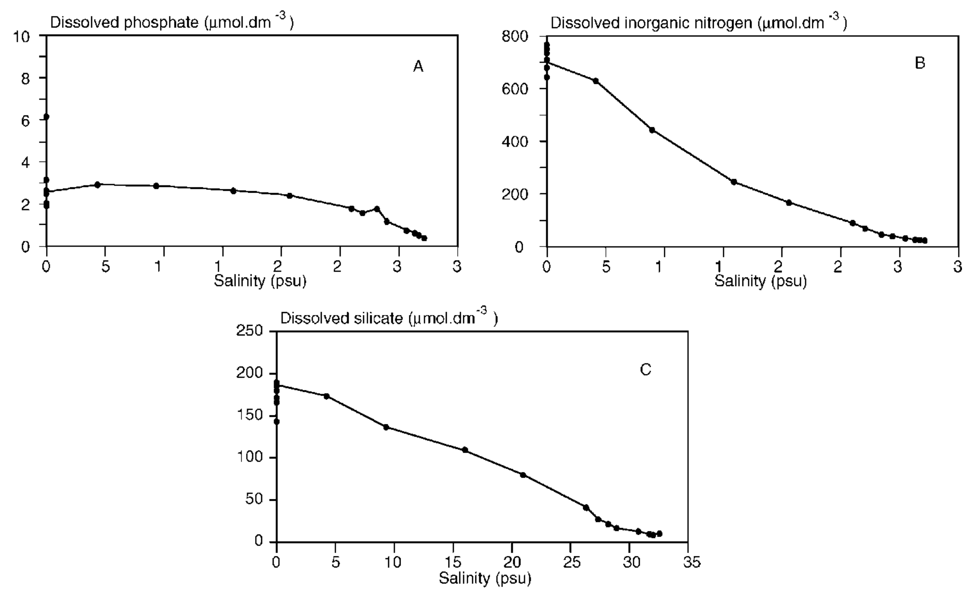

Fig. 5. Typical concentration gradients of phosphate, nitrogen and silicate in autumn 1992.

slowly over time of suspended matter concentrations and consequently in turbidity which in turn governs the primary production.

The concentrations of suspended matter can be easily converted into light-attenuation coefficients (k) by the equation published by Colijn (1982) for this area:

$$k = E_0 + 0.04\mathrm{SM},$$

where, $E_0 (= 0.5)$ is the background light-attenuation coefficient (m^{-1}) and SM the concentration of suspended matter ($g\,m^{-3}$).

From this and the relation between light and primary production it follows that the primary production will change inversely proportional to the changes in the light-attenuation coefficient.

1.4. Relevant human activities

Some dynamics in the Wadden Sea and the Ems estuary are directly caused by human activities. The estuary receives nutrient-rich water from the drainage basin which, depending on river discharge, may result in enhanced primary production (Baretta and Ruardij, 1988; de Jonge and Essink, 1991). Some recent annual amounts of the inputs to the Ems estuary are presented in Fig. 6 (de Jonge et al., 1998).

The main channels in the estuary are heavily dredged for navigation purposes resulting in remarkable changes in the mean concentrations of suspended matter (de Jonge, 1983). The dredged amounts vary from year to year, but an appropriate long-term figure of its amount is over $6 \times 10^6\,m^3$ per annum (Mulder, 1998). In addition to the channel dredging, of which the sludge is partly brought on land and partly is dumped into the area itself, the area is also used as a dumping site for harbour sludge. The sedimentological and morphological developments of the area due to a number of processes of which one is dredging and another is gas extraction have been described by Mulder (1998).

1.5. Managing coastal areas

Carrying out human activities in an area that has an important natural function can easily lead to conflicts. Managing tidal flat estuaries therefore requires a knowledge about both the natural function of these environments and knowledge about the potential of these areas to human use based on the assessment of the systems ‘resilience’. The problem, however, is that the term ‘resilience’ is not yet a real operational one (cf. Costanza and Mageau, 1999).

A knowledge about structure and functioning and the temporal fluctuations within the natural system have to be based on solid scientific information about the physical and biological structure, the physical and ecological functioning and operating geo-chemical processes.

On top of this, the effects of the development and influence of socio-economic activities and the effects of polluting substances have to be assessed.

Based on this information, policy makers and decision makers have to judge which activities can be permitted and which cannot. To enable this task, it is of utmost

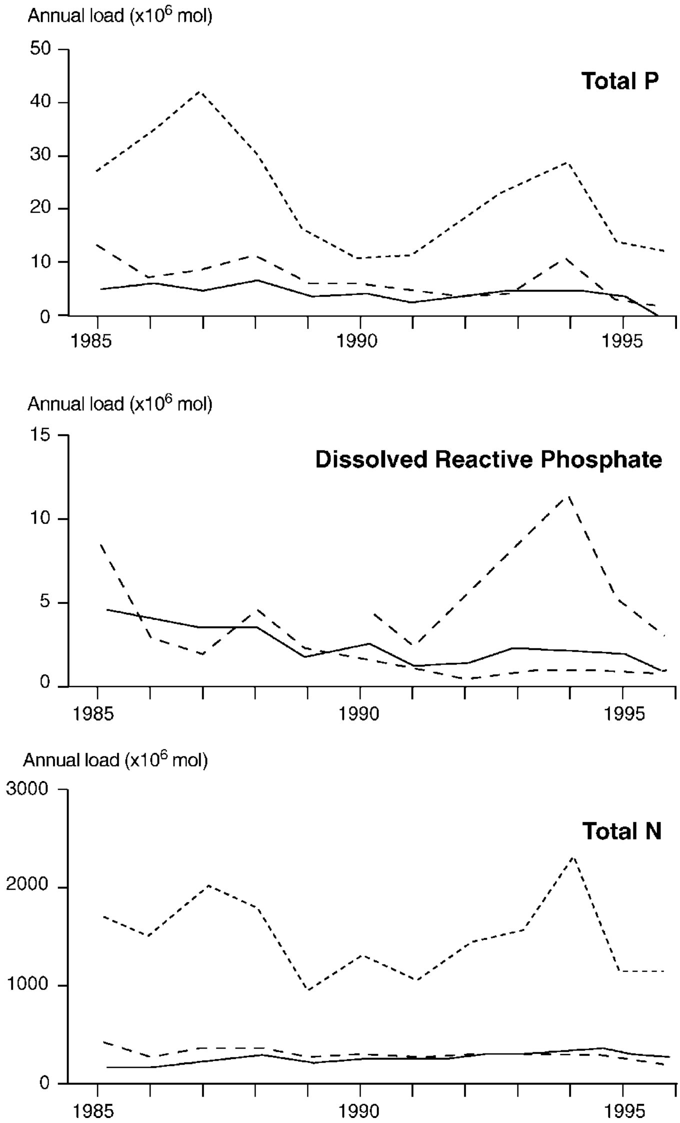

Fig. 6. Interannual variation in nutrient loads from river Ems (- - -), Westerwoldsche Aa (——) and Eems canal (- - -) near Delfzyijl (Fig. 1).

importance that instruments are available for allowing rational decision-making. These decisions have to be based on the application of proper criteria in the developed instruments reflecting proper interests and reflecting the proper scales in time and space.

The application of scientific knowledge further requires information at the proper integration level which includes proper scaling in time and space. This is important because management measures in a tidal area may, depending on transport characteristics such as flushing time and tidal travelling distance, impact the entire system as will be illustrated below.

The available information about structure and functioning of the socio-economic subsystem and the natural subsystem may be integrated in management knowledge systems, policy support systems or decision support systems (DSS) for allowing us to prioritize properly the different policy options given.

Modification of scientific knowledge into information that can be used by managers, policy makers and decision makers is an important item in this paper. Special attention will be given to the role of suspended matter (turbidity) and nutrient loads as factors in regulating the primary production. Mainly previously published information will be used.

The applicability of some simplified relationships will be demonstrated. The need for presenting properly scaled information to policy makers and environmental managers will be explained and demonstrated.

2. Natural variation and anthropogenically induced variation in suspended matter and primary production

When focussing on all the (physical, chemical and biological) processes operating in the area, an estuary may be defined as "an area with material import from the river and the sea, where transformation of material takes place, where retention of material occurs and from which export occurs to the sea and the atmosphere". Key words are import, transformation, retention and export of material or compounds (Fig. 7). Based on processes some concepts will be selected that in a simple way present the effects of human activities on the natural environment.

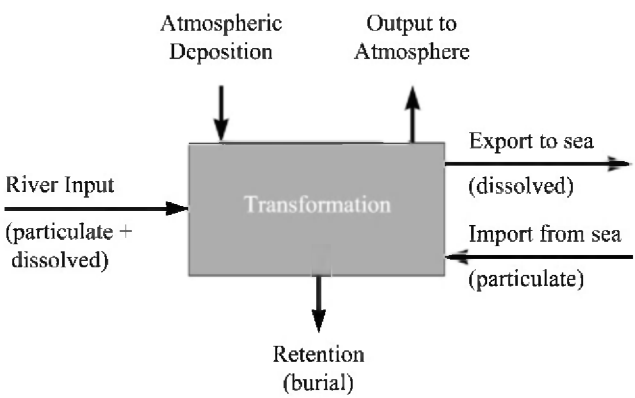

Fig. 7. General diagrammatic representation of fluxes and processes operating in an estuary.

2.1. Natural variation

2.1.1. Influence of wind and tide on suspended matter (mud)

It is well known that in the main channels under normal weather conditions the tide-induced currents cause differences in concentrations of suspended matter (Postma, 1967). This signal usually shows a lunar cycle due to changes in tidal range and maximal current velocities. The variations in concentrations of (mainly) fine suspended material and aggregates may be due to local erosion or to a flux of material from elsewhere. The natural variation in the concentrations of suspended matter in tidal flat estuaries is, however, not only caused by the tide but also by wind-induced waves. During submersion of the intertidal flats the effect of the tidal currents on the reworking of sediments (Postma, 1967) is strongly amplified by wind-induced waves (e.g. de Jonge and van Beusekom, 1995). When currents are strong enough, the resuspended material is transported from the tidal flats into neighbouring channels. A part of the suspended material is dispersed by the complex mixing process of the water (Zimmerman, 1976a, b; de Jonge, 1992a, b; de Swart et al., 1997), but at the same time is also subject to estuarine accumulation processes (Postma, 1967).

On the longer time scale (period of years) the above-mentioned processes govern the transport and accumulation of fine suspended matter in the upstream direction to intertidal flats and salt marshes. This accumulation can be deduced from calculations on accretion (Reenders and van der Meulen, 1972). It approximates 1400×10^3 t of fine sediment (mud) annually (de Jonge, 1995).

However, on the short time scale (period shorter than a year) there is near equilibrium between the sediment resuspension from the intertidal flats and the sediment deposition on the intertidal flats (Fig. 8; cf. de Jonge, 1983).

2.1.1.1. Time series per reach. For the lower part of the Ems estuary as well as the Dollard the concentrations of fine suspended material can be described as a function

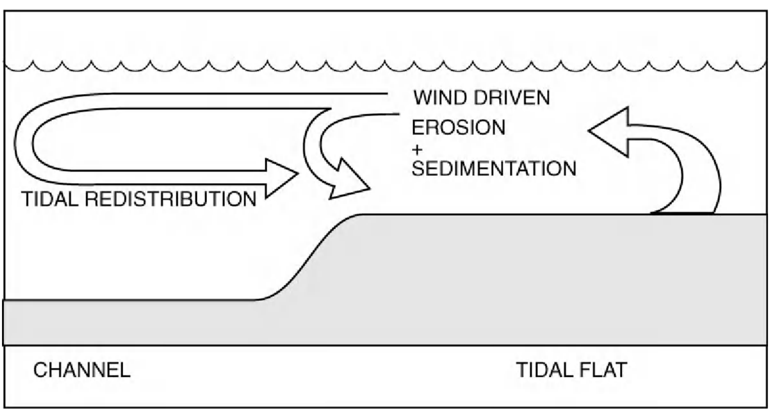

Fig. 8. Diagram illustrating the importance of resuspension by wind waves and redistribution by tidal currents.

Table 1
Equations used to calculate resuspended mud concentrations (y, g m^{-3}) as measured in the main channels from effective wind speed (x, m s^{-1}) at high tide and to calculate the resuspended fraction of microphytobenthos (y, $[B_w/B_w + B_s]$) in the main channels from effective wind speed (x, m s^{-1}) at high tide. Here B_w is the resuspended fraction in the water column, B_s is the fraction in the top 0.5 cm layer of the sediment and in which $(B_w + B_s)$ is the total amount. The effective wind speed is the mean wind speed during three high water periods preceding sampling of the main estuary channel

Parameter/subarea	Regression function	Significance level (p)	Explained variance (r^2)	Calculated mean annual value (g m^{-3})
Mud				
Lower Reaches	$y = 4.98x + 7.68$	$0.001 < p < 0.01$	0.572	46.9
Dollard	$y = 27.25x + 28.51$	$0.001 < p < 0.01$	0.646	178.2
Fraction of resuspended benthic diatoms				
Lower Reaches and Dollard together	$y = 5.41x - 10.77$	$p < 0.001$	0.893	0.32 (L.R.) 0.19 (Dollard)

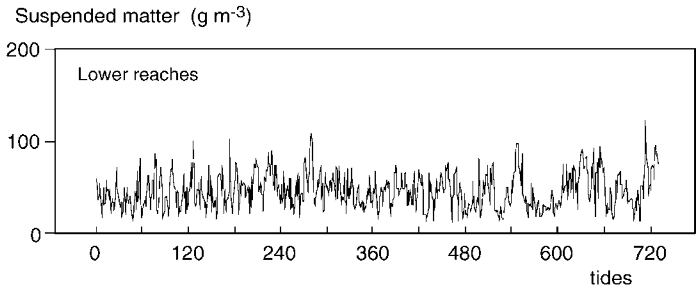

Fig. 9. Variations in fine suspended matter in the main tidal channel of the Lower Reaches in the Ems estuary due to resuspension by wind waves and transport to the main channels by the tidal currents.

of the ‘effective wind speed’ (mean wind speed during three high water periods preceding sampling of the main estuary channel). The results of the linear regression analysis, reported before (de Jonge and van Beusekom, 1995), are summarized in Table 1. The result of the strongly varying ‘effective wind speed’ from tide to tide, thus during a very short time scale, is a strongly varying concentration of fine suspended matter (mud) in the main tidal channels over the seasons (de Jonge, 1995). A presentation of this for the Lower Reaches of the Ems estuary is given in Fig. 9.

When interested in a signal caused by human activity, the complex variation pattern in Fig. 9 should be known because it represents the ‘natural variability’ the natural noise behind which the anthropogenic effect or human-induced signal (like the effects of dredging the nearby shipping route) should be resolved.

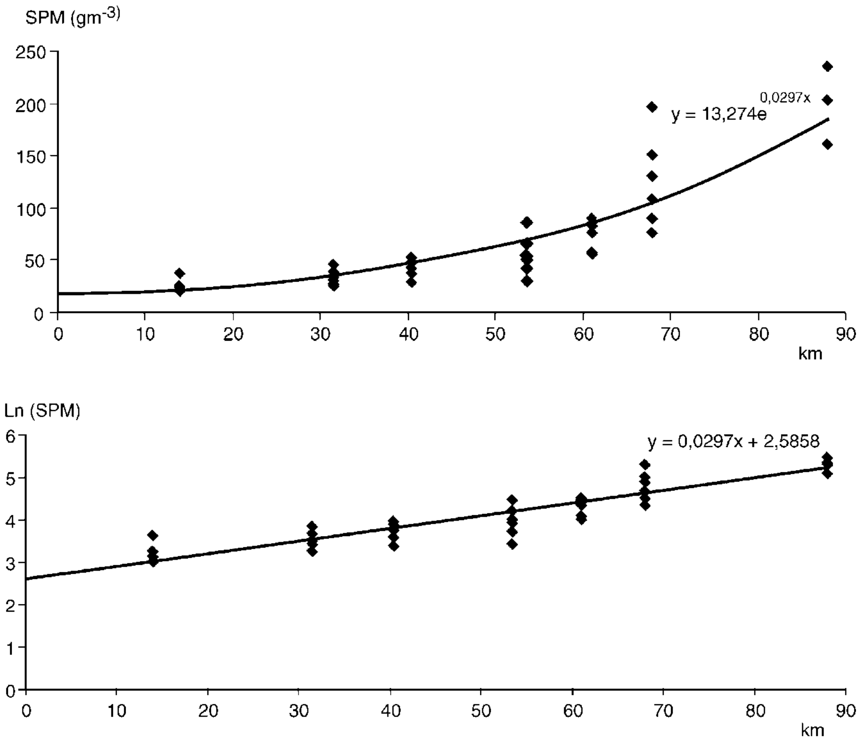

Fig. 10. Mean concentration gradient in suspended matter between turbidity maximum and the sea for the years 1971, 1972, 1975, 1976, 1978, 1979 before (upper) and after transformation (lower).

2.1.1.2. Annual estuarine concentration gradients. Apart from local time series the estuarine gradient in mud concentrations needs to be considered. Importantly, this gradient may change from year to year. The changes in the gradients, represented by the ranges in values per reach in Fig. 10, give a good impression of the interannual variations in suspended matter over the entire estuary axis from the sea to the river. The curve representing the average gradient for the years 1971–1979 can be described by the function $y = 13.274e^{0.0297x}$ (Fig. 10 upper panel). A Ln transformation results in the equation $y = 0.0297x + 2.5858$ (Fig. 10 lower panel). The application of this relationship is given in Section 2.2. under the titles 'Channel dredging', 'Nutrient discharge' and 'Integration of effects of eutrophication and dredging'.

2.1.2. Wind and tide as driving forces in suspending benthic diatoms

There is a lot of evidence that benthic diatoms living in and on the sediments are able to stabilize the surface layers of these sediments (de Jonge and van den Bergs, 1987; Vos et al., 1988; Paterson, 1989, 1997; Sutherland et al., 1998) due to the production of exopolysaccharides (Staats, 1999). There is also evidence that macrobenthos organisms, epibenthos organisms and birds by their activity contribute to the destabilization of these surface layers or enhance the turbulence close to the bottom (Daborn et al., 1993; Austen et al., 1999, this paper).

2.1.2.1. Laboratory experiments. The stabilization potential by natural populations of benthic diatoms was tested in some experiments in the early 1980s as conducted by de Jonge and van den Bergs (1987) and Delgado et al. (1991). The system used by de Jonge and van den Bergs basically consisted of a cylindrical tank with two rotating concentric perspex cylinders that generated a near-laminar waterflow at low angular velocities. Sediment including benthic diatoms was placed in a cylindrical tray and the water column sampled by a syringe through a septum in the bottom of the system (see, for details, de Jonge and van den Bergs, 1987).

A further series of two sets of experiments with a monoculture of *Navicula salinarum* have also been carried out but by using clean Merck sand as sediment. The sediment was inoculated with a monoculture of *Navicula salinarum*. The inoculations were derived from a stock culture of known density that was further grown in the experimental system to a final density of 3×10^6 cells cm^{-2}. These densities were reached at days 4, 10 and 17. Clean sand was used as a reference.

The results of the first set of experiments are illustrated by the different slopes of the curves in Fig. 11 for angular velocities higher than 1.6 rad s^{-1} above which the current velocity became a rough turbulent flow (cf. de Jonge and van den Bergs, 1987). The effect of the growth period of 4, 10 and 17 d on the sediment stability is also clear. The rough turbulent flow strongly enhanced the resuspension of sediment and benthic diatoms. However, the longer the growth period before the experiments the better the stabilization of the sediment by the benthic diatoms and/or their products.

In the second set of experiments the same benthic diatom species (*Navicula salinarum*) was now each time grown in the system for 10 d to reach the final density of 3×10^6 cells cm^{-2}. The night before the experiments were carried out, individuals of a faunal organism were brought into the system in realistic numbers m^{-2}. The species used were: *Crangon crangon*, *Corophium volutator*, *Cerastoderma edule* and *Macoma balthica*. The result of this set of observations was completely different from the previous one (Fig. 12). Compared to the reference in which no faunal organism was added to the system, *Corophium volutator* showed the least and the epibenthic species *Crangon crangon* the strongest destabilization effect now expressed as the percentage of the diatom population that was found present in suspension.

2.1.2.2. Field observations. Benthic diatoms reach the water column simultaneously with the fine sediment fraction, mainly consisting of sediment aggregates and kept together by organic matter. Just as for the mud-fraction (Table 1) also this process can be described as a function of 'effective wind speed' in which weighting was carried out by sampling several intertidal stations per area and by carrying out integrated water sampling in the main channel over the entire area under consideration (Lower Reaches, Middle Reaches and Dollard).

The result of the differences in wind speed in Lower Reaches and Dollard is that on average in the former usually ca. 30% and in the latter ca. 20% of the microphytobenthos is present in the water column.

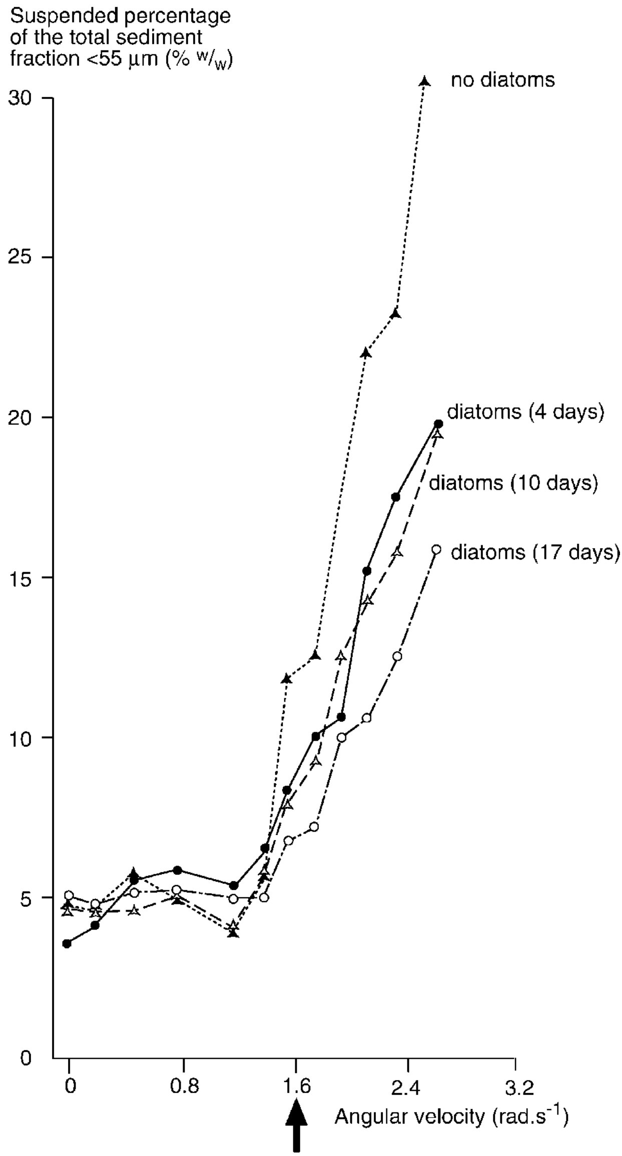

Fig. 11. Relation between angular velocity and the suspended fraction of sediment (particles < 55 µm) for a sediment without diatoms and for sediments containing the species *Navicula salinarum* at a density of 3×10^6 cm^{-2} grown there for 4, 10 and 17 d, respectively.

2.1.3. Natural variations in river discharge and estuarine primary production

In an unaffected system the nutrient input by the river and consequently the primary production mainly depends on natural weathering and leaching from terrestrial sediments under the influence of precipitation. In a densely populated area, or an

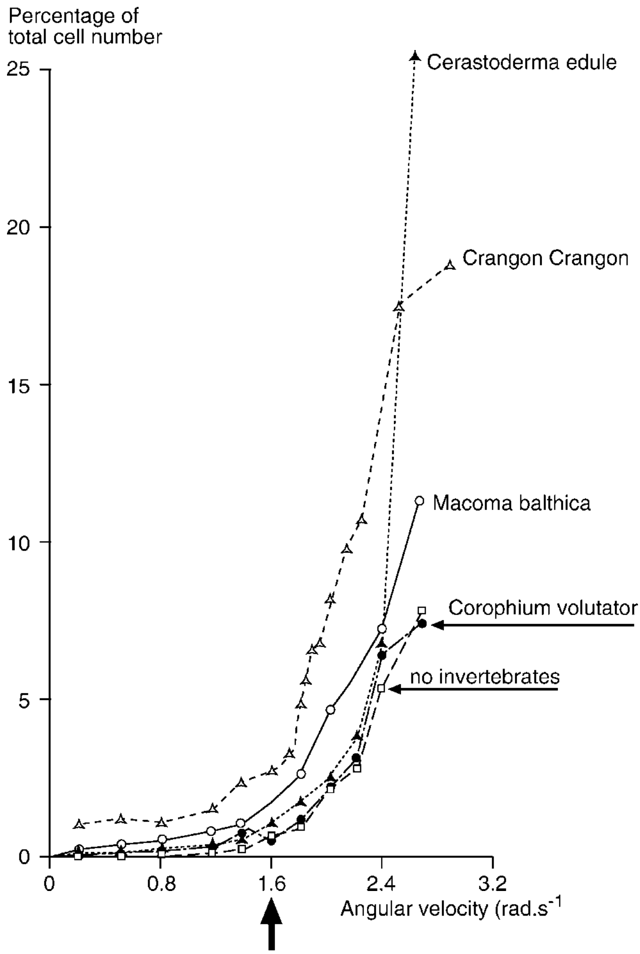

Fig. 12. Relation between angular velocity and the suspended fraction of the total cell number of the diatom species *Navicula salinarum* for a sediment containing the diatom at a cell density of 3×10^6 cm^{-2} and grown there for 10 d and sediments containing diatoms together with: 10 individuals of *Crangon crangon*; 210 individuals of *Corophium volutator*; 19 individuals of *Cerastoderma edule*; 19 individuals of *Macoma balthica*, respectively.

area where intensive agricultural activities are carried out, the nutrient load consists of a natural component and an anthropogenic component. Such an anthropogenic contribution to eutrophication may result in undesired symptoms as presented in Fig. 13.

In the past (de Jonge, 1990, 1997; de Jonge and Essink, 1991; de Jonge et al., 1996) good correlations between annual primary production and nutrient loads have been demonstrated for e.g. the western Dutch Wadden Sea. A good correlation was also obtained between the mean annual concentrations of chlorophyll-*a* in the western

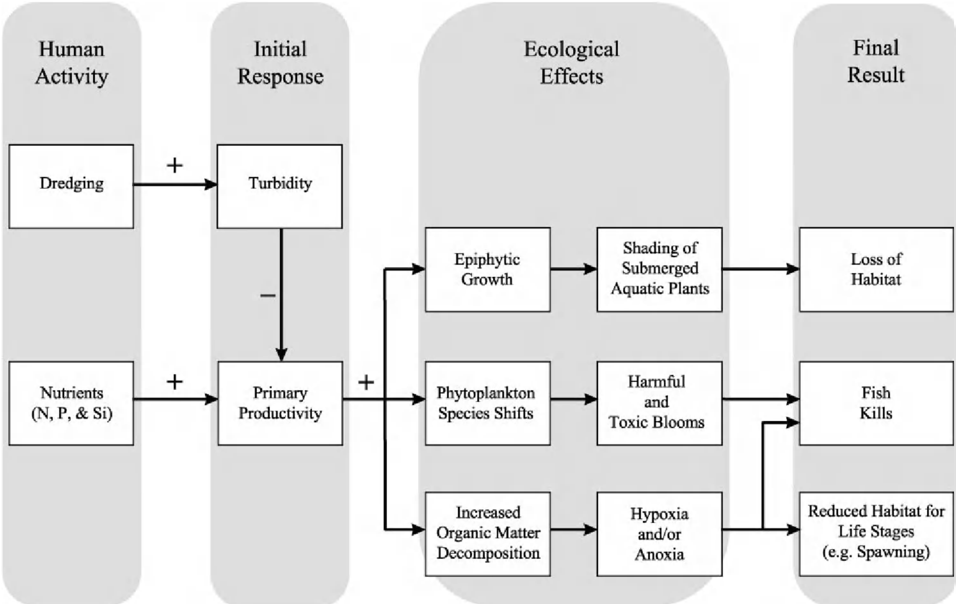

Fig. 13. Symptoms of dredging and eutrophication.

Dutch Wadden Sea and nutrient loads from Lake IJsselmeer (de Jonge et al., 1996). Moreover, Gieskes and Schaub (1990) and Schaub and Gieskes (1991) found a weak but significant correlation between chlorophyll-*a* concentrations in the Dutch coastal zone of the North Sea and the freshwater discharge from the rivers Rhine and Meuse. Finally, also a good correlation was found between annual primary production in the different reaches of the Ems estuary as a function of the discharge of fresh water, implicitly representing the general differences in nutrient loads (de Jonge and Essink, 1991).

For the Ems estuary, data on primary production of phytoplankton (including resuspended benthic diatoms) are available for the year 1972 and the period 1976–1980. The available primary production data for phytoplankton showed an increase since 1972 (de Jonge and Essink, 1991). However, the trend over time, tested by Spearman's Rank Correlation Test, was statistically not significant for any of the three reaches distinghuished (Lower Reaches, Middle Reaches and Dollard). The increased primary production was thought to be caused by the nutrient discharges from the rivers Ems and Westerwoldsche Aa for which no detailed loading data were available. Therefore, and assuming that nutrient concentrations did not vary strongly between years, and despite the strong changes in light regime due to dredging (see above), the annual primary production was related to the mean annual freshwater discharges. The correlation was statistically significant for the Lower Reaches and the Middle Reaches, but not for the Dollard (Table 2, Fig. 14). Interestingly, the slope of the regressions increases from the upper reaches in seaward direction indicating that during the investigation period the response to nutrient intervention increases in

Table 2
Correlation coefficient (r), significance level (p) and regression functions of primary producer parameters (y) on freshwater discharge parameters (x) for three reaches of the Ems estuary

Parameter/subarea	Regression function	Significance level (p)	Correlation coefficient (r)
Lower Reaches	$y = 3.4x - 64.3$	$0.025 < p < 0.05$	0.757
Middle Reaches	$y = 1.6x - 57.9$	$0.01 < p < 0.05$	0.843
Dollard	$y = 0.9x - 22.8$	n.s.	0.629

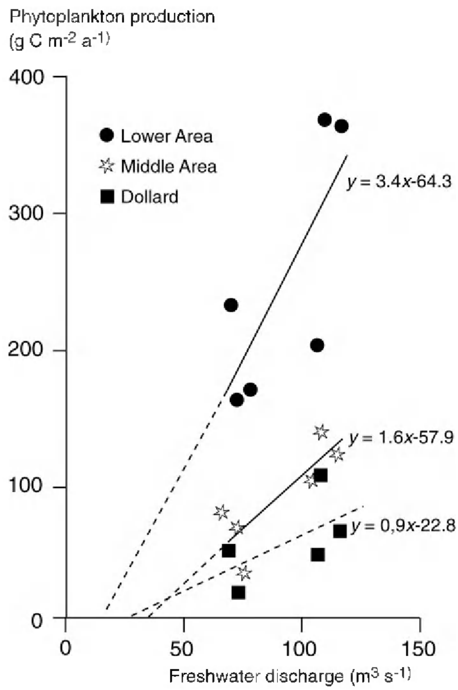

Fig. 14. Annual primary production in the water column of the three main areas (Lower Reaches, Middle Reaches and Dollard) of the Ems estuary plotted as a function of the sum of the freshwater discharge of the two main rivers. The entirely broken line indicates the Dollard area for which the correlation coefficient was not statistically significant. See also Table 2 and for details de Jonge and Essink (1991).

seaward direction. These results indicate that in the Middle Reaches and the Lower Reaches the annual primary production is, at least partly, governed by nutrient loads, a conclusion which does not hold for the very turbid Dollard. This fact is interesting because it suggests that the signal from changes in nutrient loads is nearly exclusively measurable in the low-turbid seaward situated reaches but not in the lower river close to the source. It further means that a large part of this estuary was possibly not nutrient limited during the period of the available primary production data.

2.2. Anthropogenically induced variation

2.2.1. Channel dredging

Channel dredging in the Ems estuary is responsible for changing levels in mean annual suspended matter concentrations and consequently turbidity (de Jonge, 1983). The mean annual concentration of suspended matter increased by a factor ranging from 1.3 to 2.1 between 1954 and the period 1970–1979. The fluctuations in the mean annual concentrations of suspended matter showed a statistically significant correlation with the distance dredged annually and not with the volume dredged annually. This indicated that the relation depended more on how 'extensive' rather than on how 'intensive' the dredging was.

It was hypothesized that the fluctuations in concentrations of suspended matter downstream the turbidity maximum were caused by the intensified erosion and sedimentation cycle that is initiated after the local natural equilibrium between channel morphology and current pattern is disturbed resulting in an increased time propagation of the tide which resulted in a decrease between high water and high water slack (de Jonge, 1983). This explanation was supported by the morphological changes that occurred in part of the study area between 1975 and 1979. For more detailed information the reader is referred to de Jonge (1983).

From the measurements presented so far it is clear that interannual changes in the suspended matter concentrations within the turbidity maximum are small compared to the relative changes in the other estuarine parts. It means that the total available mass of mud in the turbidity maximum is more or less constant and that the actual spatial distribution in the water and on the shoals may be dependent on factors such as river discharge (Postma, 1981) and e.g. variations in the tidal range under the influence of the lunar cycle.

When the above-given statistically significant correlations between mean annual suspended matter concentrations and the distance dredged are compared with the mean gradient presented in Fig. 10, it emerges that the main effect of dredging is presumably a change in the slope of the logarithmic gradient in suspended matter concentrations, and thus the light extinction coefficient (K_z), between the turbidity maximum and the barrier islands (Fig. 15). This might be related to an increased temporal tidal propagation and not a further landward intrusion of the tide due to the weir near Herbrum (see Fig. 1 for location of the weir).

The direct consequence of a change in the slope of the gradient is that dredging more heavily affects the most seaward situated reaches of the estuarine system than the more upstream reaches. This simplification of the complex reality is attractive because it can directly be applied in coastal management models.

2.2.2. Nutrient discharge

The variation in nutrient discharges is caused by the natural variation in the water discharge of the river and by the human-induced eutrophication. Of main importance in the eutrophication are agriculture and urbanization. The longitudinal distribution in concentrations of nutrients is relevant for the present paper. An example derived from van Beusekom and de Jonge (1994) is given in Fig. 5. Based on field

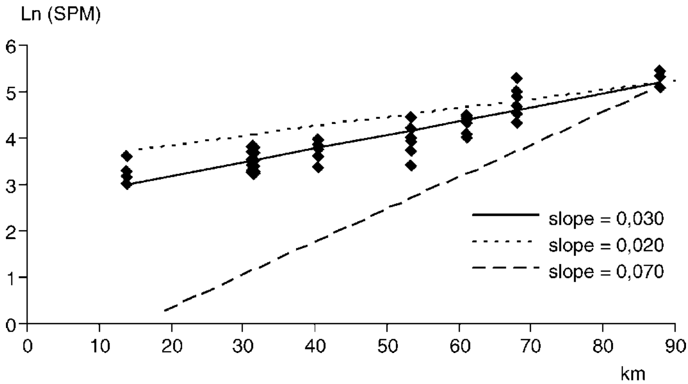

Fig. 15. Illustration of realistic effects of annual channel dredging on the slope in the turbidity gradient (Ln[SPM]) between the turbidity maximum and the sea.

measurements a statistically significant correlation was established between freshwater discharge and primary production in the different reaches of the Ems estuary (de Jonge and Essink, 1991). However, recently de Jonge et al. (1998) also found a statistically significant correlation for the period 1985–1996 between freshwater discharge from the main rivers and annual loads of nitrogen and phosphorus indicating the correctness of the assumption.

2.2.3. *Integrating effects of eutrophication and dredging*

As indicated above, the turbidity gradient must have been a factor determining the mean slope in the relations presented in Fig. 14. The response of the algae in different subareas not only depends on the actual nutrient load but also on e.g. the turbidity. It can be speculated here that under changing light conditions in the estuary also the response may change over the three subareas (cf. DeGroodt and de Jonge, 1990). This means that it can be hypothesized here that the slope of the different curves per reach is subject to changes in both nutrient discharges as well as in turbidity. It can be argued that the slope in the curve of Fig. 14 for the Dollard (given) and the upper reaches of the main stem of the estuary (no data available) will be more or less stable. Because no data are available for this part of the main stem, the Dollard will be used to illustrate the effects of this high-turbidity region. Then depending on the turbidity conditions the slope of the middle reaches and the lower reaches may change significantly as e.g. indicated by the spreading in the plotted values around the regression lines (cf. also Fig. 13). The response to increased or decreased nutrient loads will be a higher or lower annual primary production according to the function given (cf. also Fig. 13).

The combined effects of observed changes in turbidity and freshwater discharges (nutrient loads) on the primary production are given in Fig. 16. Here the effect of changes in turbidity due to channel dredging is given by the equation $Y = 37.02\ e^{0.0164X}$ which is based on the data in de Jonge (1983). The effects of changes in freshwater discharges, resulting in varied nutrient supply, were already presented in Fig. 14.

In Fig. 16 the annual scale as well as a geographic (reach) scale have been used to integrate the total effect of turbidity and nutrient supply under a very wide range of conditions. These conditions range from no channel dredging to dredging the system over its entire length (100 km). For the combination of eutrophication and freshwater discharge it ranges from nearly no river discharge to 240 $m^3 s^{-1}$, a value that during short periods can easily be reached but not easily during an entire year.

From Fig. 16 it is quite clear that, due to the combination of river discharge and channel dredging, the primary production of phytoplankton in the Lower Reaches (Fig. 16; top panel) is varying much stronger than in the Dollard (Fig. 16; bottom panel). It clearly shows the impact of interannual variations in human activities (channel dredging resulting in additional turbidity) in combination with the natural variation in freshwater discharge containing a natural as well as a over-time-varying human-induced eutrophication component.

It is clear that apart from the situation described above also other factors may operate as either stimuli or stress factors influencing the primary production. It is, however, not the aim of this paper to fully discuss all these aspects but merely to present some simplified relationships that may be applicable for policy preparation and decision-making.

3. Relevant scales in time and space

3.1. Factors determining appropriate scales

Determination of the proper time and spatial scales and integration levels requires a certain knowledge level on the natural variation in parameter values relevant to human intervention as well as on the magnitude of the signal caused by that intervention.

3.1.1. Scales in time

3.1.1.1. Natural parameters. The natural parameters that contribute to temporal scales are the tide and the wind (daily scale), the lunar cycle in the tide (monthly scale) and the season (annual scale).

Changes in wind conditions are responsible for short-term changes in the erosion–sedimentation cycle connected to intertidal flats. Due to the strong short-term variations of wind speed this process consequently represents a scale whose proper size has to be expressed in terms of days. All these factors are not human induced and have either some periodicity (tide-related parameters) or less or even no periodicity (wind-related parameters). It is remarkable that despite the obviously strong influence of the tidal cycle and given the parameters used here and the correlations presented so far, the seasonal cycle seems to overrule all the scale sizes of other parameters. Thus, focusing on the temporal scale results in the conclusion that, when talking about

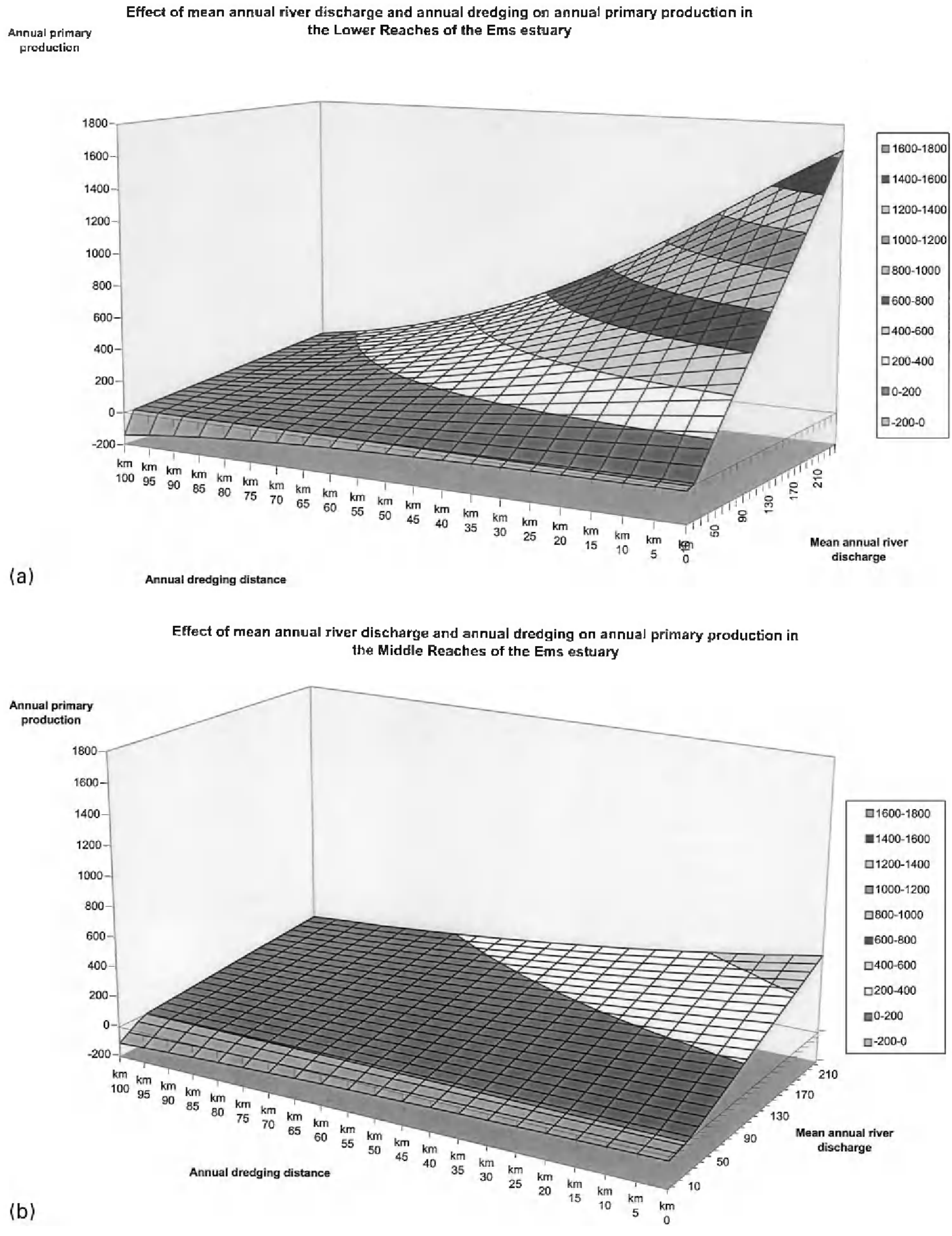

Fig. 16. Integrative effect of variation in mean annual freshwater discharge and annual channel dredging on the annual primary production in Lower Reaches (upper panel), Middle Reaches (central panel) and Dollard (lower panel).

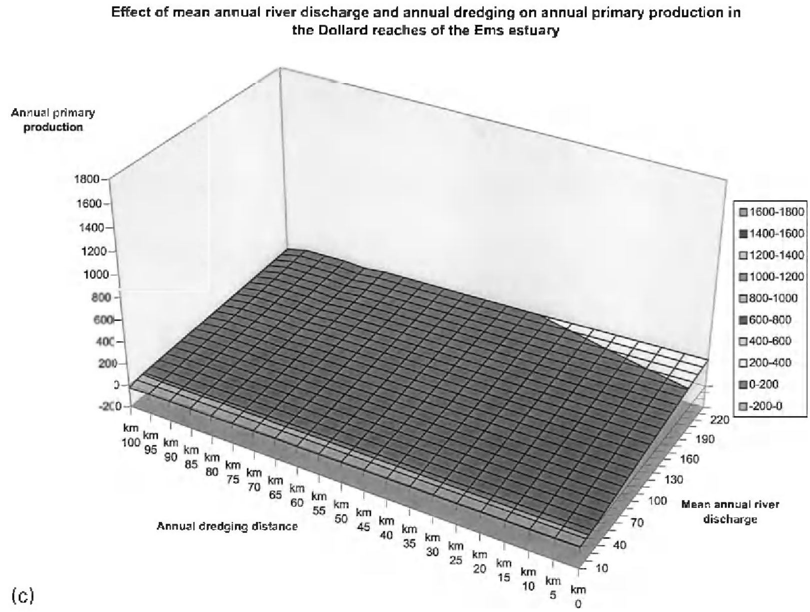

Fig. 16. (*Continued.*)

processes related to biota, the seasonal cycle seems to dominate. This is certainly not true for the spatial scale (see below).

3.1.2. *Scales in space*

3.1.2.1. Laboratory experiments. Reproducible experimental observations done in the laboratory to demonstrate the effect of the sediment stabilization or destabilization by the various species cannot properly represent or even mimic the natural field situation. This is due to the fact that under field conditions usually a, mozaic pattern varying in size, occurs representing sediment-stabilizing as well as sediment-destabilizing entities. The precise response of the natural system to tidal currents and wind-induced waves thus very much depends on the composition, distribution and density of these stabilizing and destabilizing operators in the field.

3.1.2.2. Field observations. Channel dredging may impact large parts of the estuary due to the introduction of geomorphological deformations. This (see above) presumably results in an increase in the erosion–sedimentation cycle accompanied by a temporary increase in suspended matter (turbidity) which influences the primary production potential. The area that can be influenced directly by a fixed dredger is, over a full tidal cycle, twice the tidal travelling distance of the local water mass (Fig. 17). Consequently, in the Ems estuary an area with a length varying between 24 and 34 km may be affected. This is roughly half the entire estuary between the barrier

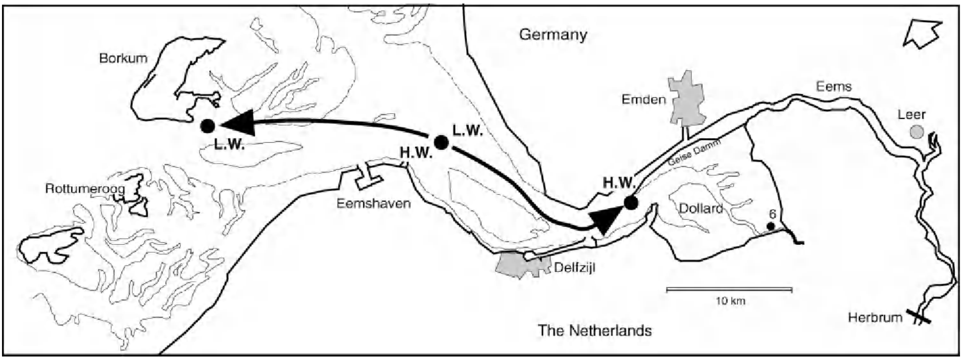

Fig. 17. The stretch influenced by a fixed dredger in the Ems estuary.

islands and the lower part of the river Ems with a length of ca. 45 km. Given the fact that the length of the area under consideration is less than four times the tidal travelling distance and only twice the stretch that is directly influenced by any dredger, this means that, when considering the primary effects of channel dredging, it does not make sense to consider an area smaller than the total length of the estuary.

When considering the biotic part of the natural system it is surprisingly remarkable that, despite the recognized complexity, for the field situation a very good correlation between wind conditions and the relative amount of resuspended benthic diatoms was found. This indicates that the physical perturbations are more important than the above-mentioned biological effects caused by the presence and the growth of organisms.

The spatial scale of this wind-induced process is quite different from what has been written above with reference to its time scale. Float experiments showed that the tidal travelling distance when starting from the high tidal flats is approximately 8 km while the travelling distance close to the main channel easily reaches that of floats released in the main channel which is approximately 12–15 km. This means that in terms of space, areas have to be considered covering at least the tentative selected reaches as defined before (Lower Reaches, Middle Reaches and Dollard).

Thus, when focussing on the spatial scale it can be argued that, contrary to the scale in time, the tide-related parameters such as flushing time and tidal travelling distance, determine the spatial scale thereby completely overruling all other parameters that contribute to the spatial scale.

In the next section the importance of the different scales as well as the effects of using it will be illustrated.

3.2. Examples from the Ems estuary

3.2.1. Wind-induced waves and tidal currents

Approximately 25% of the total algal production of the entire estuary (Table 3) is caused by resuspended microphytobenthos. The spatial redistribution of the primary

Table 3
Mean relative contribution to the total annual primary production of different algal categories in different subareas of the Ems estuary over the period 1976 – 1980 (values based on Colijn, 1983; Colijn and de Jonge, 1984; de Jonge, 1995)

Parameter	Contribution to the annual primary production (%)		
	Lower Reaches	Middle Reaches	Upper Reaches (Dollard)
Water column			
Real phytoplankton	57	59	2
Resuspended microphytobenthos	25	23	26
Intertidal flats			
Microphytobenthos	18	18	72

production products is caused by many factors of which the wind and tide play a predominant role (de Jonge, 1995). It has been shown that wind resuspension of microphytobenthos is essential in feeding the strongly developing copepod population in the water column in early spring (de Jonge and van Beusekom, 1992) when circa 50% of the available algal food consists of resuspended microphytobenthos. Without this source the low food availability in real phytoplankton during spring may lead to a slower development in the copepod population and consequently to lower food availability to higher trophic levels than under normal average conditions. This effect is expected to be much stronger in the Dollard where the phytoplankton abundance is very low as compared with the lower reaches (Table 3). It also means that any decrease in the wind field and any human intervention in terms of land reclamation will lead to diminished food supply to the pelagical system which at that moment seems heavily dependent on the food supply from the intertidal flats.

In terms of scales this means that a relevant ecological time scale for resuspension processes may vary from one year (overall mean approach) to a period as short as one or two months (temporal ecological effect of a phenomenon). Because the wind affects the entire estuarine system, the relevant spatial scale necessarily covers large estuarine reaches as defined before (Lower Reaches, Middle Reaches and Dollard).

3.2.2. Channel dredging

The effect of changes in turbidity due to dredging is stronger in seaward direction given the fact (see above) that mainly the slope of the suspended matter gradient changes due to dredging (Fig. 15).

For e.g. in the year 1977 the mean annual concentration in suspended matter in solely the lower reaches, due to the governing wind conditions, was calculated as 46.9 g m^{-3} (Fig. 9). This concentration corresponds to a k-value of 2.4 m^{-1} and includes the effect of channel dredging over a distance of 40.2 km (de Jonge, 1983). The increase in the mean annual concentration of suspended matter for the entire estuary over that in 1954 was 68.3 (1977)/45.8 (1954) = ca. 1.5. Under near natural conditions

(1954) the local concentrations in suspended matter in solely the lower reaches of the estuary may have been close to 46.9/1.5 = ca. 30 (1954) g m^{-3} which corresponds to a k-value of ca. 1.7 m^{-1}. Application of the equation given in Fig. 15 with a slope of 0.035 results in the same average concentration for the entire estuary, but in a mean concentration of 24 g m^{-3} or a k-value of 1.5 for solely the lower reaches.

In 1978 the mean annual concentration of suspended matter due to channel dredging was 96.6 g m^{-3}. The increase in suspended matter compared to 1954 was 96.6 (1978)/45.8 (1954) = ca. 2.1. Thus, the concentrations in 1978 in solely the lower reaches must have been clearly over (cf. effect of differences in slopes in Fig. 15) 2.1×30 (1954) = 63 (1978) g m^{-3} corresponding to a k-value of over 3.0 m^{-1}. This k-value must have led to a potential reduction in primary production of roughly 40% assuming further unchanged environmental conditions (see Fig. 16).

An ecological consequence of dredging activities is that during conditions of increased background turbidity, the contribution of resuspended microphytobenthos in feeding the pelagial may also increase.

In terms of scales the changes indicated here show that the relevant scale is basically the entire estuary because the entire estuarine suspended matter gradient is affected by channel dredging.

3.2.3. *Nutrient loads*

As discussed above, addition of nutrients stimulates the primary production of a system. The available data in de Jonge et al. (1998) suggest that the relation between primary production and river discharge also holds for total phosphorus and dissolved reactive phosphate as X-value when (in the absence of primary production data) the chlorophyll-*a* concentrations are considered, a parameter which is crucial in the process of primary production. Thus, river discharge may indeed be read as a surrogate for discharge of nutrients. These findings are in agreement with those obtained for other areas (Gieskes and Schaub, 1990; Schaub and Gieskes, 1991; Harding et al., 1992).

The differences of the slopes in Fig. 14 illustrate that the primary effect of eutrophication is strongest in the most seaward reach of the estuary. This corresponds to the relatively high turbidity in the upper reaches.

Interestingly, the regression lines in Fig. 14 indicate that in the Ems estuary, the Dollard included, during mean annual river discharges lower than 21–35 m^{-3} s^{-1}, the annual primary production in the different reaches becomes near zero due to which the pelagial system seems to turn to full heterotrophy. Or in other words, to sustain a net primary production in the water column under the present eutrophic conditions a river discharge higher than ca. 35 m^{-3} s^{-1} seems to be profitable.

From a scale point of view the presented results show that the best temporal scale is the annual one while from a spatial point of view there is a gradually changing response from the lower river in seaward direction. It seems that the previously mentioned tripartition of the area in Lower Reaches, Middle Reaches and Dollard is a suitable spatial scale.

4. Operationalization of usable concepts

One of the 'duties' of researchers operating in the field of the applied science might be to offer suitable relationships, information and instruments that are usable as a rational basis to 'decision-making' in coastal zone management. Presentation of the simplest possible relationships between relevant parameters as presented in this paper is one of these ways. These relationships, however, have to be based on solid conceptual knowledge about the functioning of estuarine and coastal systems.

Managing estuarine and other coastal systems does not mean that the decision maker has to know the exact outcome of a certain policy option. This means that the application of very complex ecosystem models is questionable because it has been widely accepted that prediction of future developments is beyond the scope of our possibilities and that we have to accept that all these predictions are basically 'explorations'. Thus, for management purposes it is often enough to know what the natural variability is and to explore the main direction of change due to the various policy options that represent desired socio-economical states.

When returning to the element level (cycling and fluxes of C, N and P) and considering the natural variability as well as the effects of human activities due to dredging and anthropogenic nutrient loads and considering the most appropriate scales in time and space we may end up with the integrative result as given in Fig. 16.

Although the simple curves on which this figure is based may look attractive to decision makers, a lot of details are necessarily missing. Among these missing items are the natural short-term (weekly period) variation in primary production and the process governing chlorophyll-*a* concentrations per reach and the influence of freshwater discharge on the changes in the turbidity gradient (Postma, 1981). Another missing factor is e.g. the response of algae to river discharge driven salinity changes and therefore to salinity stress. Further missing information is the possible short-term variations and transport in suspended matter content due to channel dredging and also due to the dumping of harbour sludge (Mulder, 1998).

A direct consequence of the application of the simple concepts in Fig. 16 is that a combination of increased turbidity and eutrophication can easily lead to a situation where the bulk of the primary production will be realized in the coastal area instead of in the estuary. This is particularly important from the functional point of view. When the production is mainly realized in the area in front of the estuary, the estuary will turn into a system that is mainly heterotrophic instead of being an area which can be subdivided in to a heterotrophic upper and an autotrophic lower part.

The step to be followed now is further completion of the simple equations used to describe the relationships between changes in turbidity, nutrient input and annual phytoplankton production per subarea. This can be done in many ways such as by application of available ecosystem models (Baretta and Ruardij, 1988; Brinkman, 1993) or by developing specific effect-oriented models. This action may result in some extension of the simple equations per subarea that describe the cumulative effect of dredging and eutrophication on the primary production process.

Another aspect not incorporated in the present paper is to explicitize the effect of a change in e.g. intertidal area due to land reclamation or sea-level rise. Also, this aspect may be important because it has been demonstrated that in spring, intertidal areas play an important role in feeding the pelagical system (de Jonge and van Beusekom, 1992).

5. Application of scientific concepts in integrated coastal zone management

In the Netherlands, with over 370 inhabitants km^{-2}, the sectoral coastal management has changed into an integral one in which the natural components and the human-driven components of the system have been integrated. Moreover, also the principle of 'sustainability', as introduced by the World Commission on Environment and Development, the Brundtland Commission, (WCED, 1987) was adopted.

To structure the 'integral system', a computer-based policy or decision supporting instrument has been developed. This computer model package is based on a modelling environment developed by Engelen (1988) and Engelen et al. (1995) and is the result of cooperation between a large number of market parties in the Land Water Environment Information Technology Programme (Research Institute for Knowledge Systems/RIKS, INFRAM and three governmental services (National Institute for Coastal and Marine Management/ RIKZ, Directory 'Noord Nederland' and Directory 'Noord Holland' all part of Rijkswaterstaat, Dutch Ministry of Transport, Public Works and Water Management).

In this instrument elements from dynamic carbon (including nitrogen and phosphorus) flow models (cf. Baretta and Ruardij, 1988; Brinkman, 1993) have been integrated with elements from the human-driven part of the system that are relevant to the natural processes and that can be modelled in a dynamic way. An example of such an attempt is the policy supporting instrument WADBOS (Wadden Sea Policy Supporting System or Wadden Sea DSS). This system covers the entire Dutch Wadden Sea (the Ems estuary included) and consists of an ecological subsystem and a socio-economic subsystem. The system represents the Wadden Sea area as an integral holistic system in which the physical, ecological and socio-economic aspects are tightly coupled in space and time.

In the socio-economic subsystem, processes and parameters relevant to human activity and relevant to the ecosystem have been modelled so that relevant output from the human-driven part of the system (e.g. boats spilling oil, releasing TBT and producing NO_x) can affect elements from the ecological subsystem.

The subsystems are connected to each other at two different scales: the macro-level which covers the modelling descriptions for the entire Wadden Sea and a micro-level consisting of two different spatial levels namely the 12 defined spatial compartments and a very fine spatial grid consisting of cells of 25 ha (500×500 m) each.

The different subsystems consist of connected 'Model Building Blocks' representing either very simple straightforward relations between two parameters or complex dynamic models on their own.

To make the WADBOS DSS instrument suitable to decision-making, four different main panels have been defined by means of which from different perspectives (administrative, economical or ecological) parameters and some variables in the system can be changed. These main panels are:

(1) *The physical, ecological and socio-economic system* (*system diagram panel*): This system contains relevant socio-economic aspects like extraction (sand, shells and gas), fisheries (edible cockle, mussel, shrimp), transport and recreation (boating). The ecosystem contains processes like water transport (including emissions) and geomorphological developments and species like harbour seal, oyster catcher and eider duck, but at the lower trophic level also functional groups like filter feeders (edible cockle, blue mussel) grazing microalgae and finally also microalgae and detritus.

(2) *The policy options and measures* (*policy panel*): This panel offers to set several policy options like closing areas for fishery, changing infrastructural measures, setting quota and subsidies to activities.

(3) *The external influences beyond our influence* (*scenario panel*): This contains factors such as meteorological conditions (wind, precipitation, irradiation, temperature) but also freshwater run off, development of welfare and economical developments.

(4) *The preferred state of the system* (*criteria panel*): Important criteria here are changes in habitats, natural dynamics, safety against flooding, development in resources (natural goods and species), economy, landscape and tax development. The resulting state of the system after the parameters and state variables have been changed can be visualized by the temporal and spatial changes in indicator species and compounds. The available model is not a black box but completely open and transparent and suitable to change all the available model descriptions if required.

The WADBOS DSS enables policy makers and analysts to explore effects due to autonomic developments as well as the additional effects due to the policy options chosen. The chosen transparent physical structure, the possibility to couple data bases, to store detailed model results and the display of maps with spatial results strongly facilitate the communication between policy makers and stakeholders. Unfortunately, the system has not yet been described for the scientific community so that a suitable reference is not yet available.

It is the aim to make the present results also applicable to the Ems estuary in the above-described DSS.

There is a fundamental difference between the preparation of policies and decision-making. The process of decision-making belongs to the political arena and is a normative one. This means that the basis for any decision as defining criteria and setting values is not the task of the scientist but the privilege of the political arena and is consequently also not the subject of this study. The steps in the decision-making process are given in Fig. 18 and show that the process needs iteration during which the state of the system, after setting a new policy, can be optimzed until consensus is reached among the stakeholders involved.

The current basis for decision-making is more and more based on instruments containing rational information and tools suitable to rationally weigh the outcomes of several policy options e.g. by application of multicriteria analysis. Despite this, during the final stage of the decision-making process these instruments often do not play a prominent role any longer. It means either shortcomings in the instruments

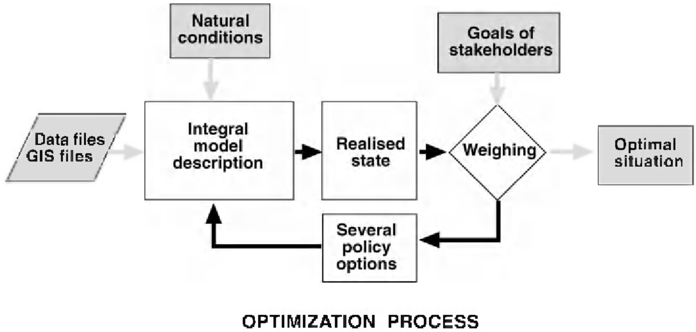

Fig. 18. Necessary iteration in the process of optimizing the 'desired state' from exploring policy options during the 'decision process' in which a Policy Support System or Decision Support System may play an important rational role.

developed or the human desire for taking the final decisions on an emotional basis instead of a rational one.

To make the conceptual knowledge about the effects of human activities to the natural system applicable, several steps are required. The first one is to define criteria and values to judge the suitability of different policy options, not necessarily in financial terms but in any suitable unity showing the impact of human interventions on the ecosystem. A certain investment, accompanied by human activity and some 'environmental effect', leads to a certain benefit expressed in monetary units and employment. An ecologist might be happy when the effect of the human activity can be expressed in a criterion represented by a species and where the value is the number of individuals available under the pressure of the human activity. A chemist may be interested in a specific compound and its concentration which is responsible for a certain effect at the species level. The connection between the three is a certain concentration (value) of a compound (criterion) derived from a certain human activity which resulted in an increase or decrease in the numbers (value) of a certain species (criterion).

The question is, however, whether a certain information level is suitable to a policy maker or a politician. They possibly will easily label the above-given scientific information as expert information which in that form is unsuitable to base decisions upon at the level of the political arena. Lack in scientific knowledge of decision makers requires simplification as well as upscaling of scientific results to a level that is appealing to them. To meet these requirements, upscaling might be necessary until a level (e.g. vegetation or plant communities or landscape) that is understandable because of the connection with emotional experience.

References

Austen, I., Andersen, T.J., Edelvang, K., 1999. The influence of benthic diatoms and invertebrates on the erodibility of an intertidal mudflat, the Danish Wadden Sea. Estuarine, Coastal and Shelf Science 49, 99–111.

Baretta, J., Ruardij, P., 1988. Tidal Flat Estuaries, simulation and analysis of the Ems estuary. Ecological Studies, Vol. 71. Springer, Berlin, p. 353.
van Beusekom, J.E.E., de Jonge, V.N., 1994. The role of suspended matter in the distribution of dissolved phosphate, iron and aluminium in the Ems estuary. Netherland Journal of Aquatic Ecology 28, 383–395.
Brinkman, A.G., 1993. Biological processes in the EcoWasp ecosystem model. Institute for Forestry and Nature Reserve/ IBN-DLO Report 93/6, p. 111.
Cloern, J.E., 1999. The relative importance of light and nutrient limitation of phytoplankton growth: a simple index of coastal ecosystem sensitivity to nutrient enrichment. Aquatic Ecology 33, 3–16.
Colijn, F., 1982. Light absorption in the waters of the Ems-Dollard estuary and its consequences for the growth of phytoplankton and microphytobenthos. Netherland Journal of Sea Research 15, 196–216.
Colijn, F., 1983. Primary production in the Ems-Dollard estuary. Ph.D. Thesis, Groningen, The Netherlands, p. 123.
Colijn, F., de Jonge, V.N., 1984. Primary production of microphytobenthos in the Ems-Dollard estuary. Marine Ecology Progress Series 14, 185–196.
Costanza, R., Mageau, M., 1999. What is a healthy ecosystem? Aquatic Ecology 33, 105–115.
Daborn, G.T., Amos, C.L., Brylinsky, M., Christian, H., Drapeau, G., Faas, R.W., Grant, J., Long, B., Paterson, D.M., Perillo, G.M.E., Piccolo, M.C., 1993. An ecological cascade effect: migratory birds affect stability of intertidal sediments. Limnology and Oceanography 38, 225–231.
DeGroodt, E.G., de Jonge, V.N., 1990. Effects of changes in turbidity and phos phate influx on the ecosystem of the Ems estuary as obtained by a computer simulation study, In: McLusky, D.S., de Jonge, V.N., Pomfret, J., (Eds.), North Sea – Estuaries Interactions Hydrobiologia 195, 39–47.
Delgado, M., de Jonge, V.N., Peletier, H., 1991. Experiments on resuspension of natural microphytobenthos populations. Marine Biology 108, 321–328.
Engelen, G., 1988. The theory of self-organization and modelling complex urban systems. European Journal of Operational Research 37, 42–570.
Engelen, G., White, R., Uljee, I., Drazan, P., 1995. Using cellular automata for integrated modelling of socio-environmental systems. Environmental Monitoring Assessment 34, 203–214.
Führböter, A., 1989. Changes of the tidal water levels at the German North Sea coast. Helgoländer Meeresunters 43, 325–332.
Gieskes, W.W.C., Schaub, B.E.M., 1990. Correlation of the seasonal and annual variation of phytoplankton biomass in Dutch coastal waters of the North Sea with Rhine river discharge. Coastal Estuarine Studies 36, 311–320.
Harding, Jr., L.W., Leffler, M., Mackiernan, G.E., 1992. Dissolved Oxygen in the Chesapeake Bay: a Scientific Consensus. A Maryland Sea Grant Publication, College Park, MD, USA, p. 18.
Hinrich, H., 1974. Schwebstoffgehalt, Gebietsniederschlag, Abfluß und Schwebstofffracht der Ems bei Rheine und Versen in den Jahren 1965 bis 1971. Deutsche Kewässerkundliche Mitteilungen 18, 85–95.
de Jong, F., Bakker, J.F., van Berkel, C., Marencic, H., Dankers, N., Dahl, K., Gätje, C., Potel, P. (Eds.), 2000. Quality status report of the international Wadden Sea, period 1985–1996. Common Wadden Sea Secretariat, in press.
de Jonge, V.N., 1983. Relations between annual dredging activities, suspended matter concentrations and the development of the tidal regime in the Ems estuary. Canadian Journal of Fisheries and Aquatic Science 40 (Suppl. 1), 289–300.
de Jonge, V.N., 1990. Response of the Dutch Wadden Sea ecosystem to phosphorus discharges from the River Rhine. In: McLusky, D.S., de Jonge, V.N., Pomfret, J. (Eds.), North Sea – Estuaries Interactions Hydrobiologia 195, 49–62.
de Jonge, V.N., 1992a. Tidal flow and residual flow in the Ems estuary. Estuarine, Coastal and Shelf Science 34, 1–22.
de Jonge, V.N., 1992b. Physical processes and dynamics of microphytobenthos in the Ems estuary (The Netherlands). Ph.D. Thesis, University of Groningen, p. 176.
de Jonge, V.N., 1995. Wind driven tidal and annual gross transports of mud and microphytobenthos in the Ems estuary, and its importance for the ecosystem. In: Dyer, K.R., D'Elia, C.F. (Eds.), Changes in Fluxes in Estuaries, pp. 29–40.

de Jonge, V.N., 1997. High remaining productivity in the Dutch western Wadden Sea despite decreasing nutrient inputs from riverine sources. Marine Pollution Bulletin 34, 427–436.

de Jonge, V.N., Bakker, J.F., van Stralen, M.R., 1996. Possible change in the contribution of the river Rhine and the North Sea to the eutrophic status of the western Dutch Wadden Sea. Netherland Journal of Aquatic Ecology 30, 27–39.

de Jonge, V.N., van den Bergs, J., 1987. Experiments on the resuspension of estuarine sediments containing benthic diatoms. Estuarine Coastal and Shelf Science 24, 725–740.

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. Netherland Journal of Sea Research 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. Limnology and Oceanography 40, 766–778.

de Jonge, V.N., Bijkerk, R., Dijkhuizen, D., Peletier, H., 1998. Primaire productie in een troebel system. In: Essink, K., Esselink, P. (Eds.), Het Eems-Dollard estuarium: interacties tussen menselijke beïnvloeding en natuurlijke dynamiek. Rapport RIKZ-98.020, pp. 63–76.

de Jonge, V.N., Essink, K., 1991. Long-term changes in nutrient loads and primary and secondary production in the Dutch Wadden Sea. In: Elliott, M., Ducrotoy, J.P. (Eds.), Estuaries and coasts: Spatial and Temporal Intercomparisons. International Symposium Series, Olsen & Olsen, Fredensborg, Denmark, pp. 307–316.

de Jonge, V.N., de Jong, D.J., van Katwijk, M.M., 2000. Policy plans and management measures to restore eelgrass (*Zostera marina* L.) in the Dutch Wadden Sea, Helgoland Mar. Res., in press.

Kremer, J.H., Nixon, S., 1978. A Coastal Marine Ecosystem: Simulation and Analysis. Springer, Berlin, p. 217.

Mulder, H.P.J., 1998. Geomorfologische en hydrologische ontwikkelingen. In: Essink, K., Esselink, P. (Eds.), Het Eems-Dollard estuarium: interacties tussen menselijke beïnvloeding en natuurlijke dynamiek. Rapport RIKZ-98.020, pp. 25–45.

Paterson, D.M., 1989. Short-term changes in the erodibility of intertidal cohesive sediments related to migratory behavior of epipelic diatoms. Limnology and Oceanography 34, 223–234.

Paterson, D.M., 1997. Biological mediation of sediment erodibility: ecology and physical dynamics. In: Burt, N., Parker, R., Watts, J. (Eds.), Cohesive Sediments. Wiley, New York, pp. 215–229.

Platt, T. (Ed.), 1981. Physiological bases of phytoplankton ecology. Canadian Fisheries and Aquatic Science (Ottawa) 210, p. 346.

Platt, T., Bird, D.F., Sathyendranath, S., 1991. Critical depth and marine primary production. Proceedings of the Royal Society of London, Series B 246, 205–217.

Platt, T., Denman, K.L., Jassby, A.D., 1977. Modelling the productivity of phytoplankton. In: Goldberg, E.D. (Ed.), The Sea: Ideas and Observations on Progress in the Study of the Seas, Vol. VI. Wiley, New York.

Platt, T., Sathyendranath, S., Ravindran, P., 1990. Primary production by phytoplankton: analytical solutions for daily rates per unit of water surface. Proceedings of the Royal Society of London, Series B 241, 101–111.

Postma, H., 1967. Sediment transport and sedimentation in the estuarine environment. In: Lauff, G.H. (Ed.), Estuaries. American Association in Advance Science Publication no. 83, Washington, pp. 158–179.

Postma, H., 1981. Exchange of materials between the North Sea and the Wadden Sea. Marine Geology 40, 199–215.

Reenders, R., van der Meulen, D.H., 1972. De ontwikkeling van de Dollard over de periode 1952–1969/'70. Rijkswater- staat, Direktie Groningen, afd. Studiedienst, nota 72.1, 23pp. and 14 appendices.

Schaub, B.E.M., Gieskes, W.W.C., 1991. Trends in eutrophication of Dutch coastal waters: the relation beween Rhine river discharge and chlorophyll-*a* concentrations. In: Elliott, M., Ducrotoy, J.-P. (Eds.), Estuaries and Coasts: Spatial and Temporal Intercomparisons, ECSA 19 Symposium. International Symposium Series, Olsen & Olsen, Fredensborg, Denmark, p. 85.

Staats, N., 1999. Exopolysaccharide production by marine benthic diatoms. Ph.D. Thesis, University of Amsterdam, p. 97.

Sutherland, T.F., Grant, J., Amos, C.L., 1998. The effect of carbohydrate production by the diatom *Nitzschia curvilineata* on the erodibility of sediment. Limnology and Oceanography 43, 65–72.

de Swart, H.E., de Jonge, V.N., Vosbeek, M., 1997. Application of the tidal random walk model to calculate water dispersion coefficients in the Ems estuary. Estuarine, Coastal and Shelf Science 45, 123–133.

Vos, P.C., de Boer, P.L., Misdorp, R., 1988. Sediment stabilisation by benthic diatoms in intertidal sandy shoals; qualitative and quantitative observations. In: de Boer, P.L., van Gelder, A., Nio, S.D.F. (Eds.), Tide-influenced Sedimentary Environments and Facies. Reidel, Dordrecht, pp. 511–526.

WCED (World Commission on Environment and Development), 1987. Our Common Future. Oxford University Press, Oxford, UK.

Zimmerman, J.T.F., 1976a. Mixing and flushing of tidal embayments in the western Dutch Wadden Sea. Part I: distribution of salinity and calculation of mixing time scales. Netherland Journal of Sea Research 10, 149–191.

Zimmerman, J.T.F., 1976b. Mixing and flushing of tidal embayments in the western Dutch Wadden Sea, part II: analysis of mixing processes. Netherland Tournal of Sea Research 10, 397–439.