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Causes, historical development, effects and future challenges of a common environmental problem: eutrophication

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Abstract

The impact of the presence of us humans as well as our activities to nature has led to over-exploitation of natural resources and to dramatic changes in land-use including the use of artificial fertilizers contributes to the deterioration of the natural environment. The population density, industrial processes and the use of fertilizers are the main causes for the eutrophication of river systems, estuaries and seas. There are several ways to determine the actual nutrient levels back to the 1950s, back to 1900 or even back to 1800. Available data indicate that the natural background concentrations of nutrients (pristine conditions or the period prior to the widespread use of artificial fertilizers and detergents) were dramatically lower than today. Available time series on chemical and biological data collected from different parts of the world show the (sometimes tremendous) increase in nutrient levels and the related productivity. The same time series, however, also show the decreases in values when measures were taken to reduce the nutrient emissions. Investigations of different systems all over the world show that nearly every system responses differently to eutrophication. Especially physical boundary conditions play an important role in the manifestation of the ultimate effect of local eutrophication. Apart from the physical boundary conditions also the transformation and retention of nutrients in estuarine and coastal systems contribute to system specific responses. Depending on all these different conditions, site specific responses with even site specific problems may occur. The challenge of this millennium is to really reach a balance between nature and mankind including its population size. A beneficial step in this discussion is assessing criteria to reduce eutrophication back to a level acceptable to both humans and nature.

Introduction

Eutrophication is a process that contains several aspects when considering the processes responsible for it. There are coastal areas that due to local natural characteristics show eutrophication phenomena as increased mineralization of organic matter and decreased oxygen concentrations (Fig. 1). Characteristics responsible for it are the geomorphology of the systems (percentage of subtidal and intertidal areas), shape of the tidal curve (rate of change in current velocities during flood and ebb), mean depth of entire system, its channels and tidal flats (Van Straaten & Kuenen, 1958; Postma, 1967). Apart from natural eutrophication also human induced eutrophication occurs due to nutrient loadings from point sources (outfall discharges of industrial plants and sewage treatment works) and of human-influenced 'diffuse sources' (run-off from any but specifically an agricultural catchment). Point source discharges are usually relatively easy controllable while diffuse and atmospheric sources are more difficult to control. The last

1. Natural conditions
   - tide associated accumulation of organic matter
   - sedimentation areas

2. Anthropogenic influence
   Input from :
   - point sources
   - diffuse sources

Figure 1. Causes of eutrophication.

ones require a change in agricultural and technical practice. It is clear that the extend of the nutrient loads is directly dependent on human activities which in turn is dependent on the development of the world's human population. Human induced eutrophication is consequently in a way related to the increase of the human population. This increase is today dramatic compared to the available historic information (Fig. 2). To date, the human population has increased over 35-fold to 6 milliard individuals. The growth rate of the population has increased from 0.0002 in the period between 0 and 1500 AD to 0.071 in the period 1950 to 2000 AD, a factor of over 350-fold. According to the present data, we will have reached the level of 8 milliard people within 30 years time.

There is in general a good qualitative understanding of the processes that contribute to eutrophication. The quantitative influence on the ecological processes and the changes in community structure are, however, still not well understood.

Nutrient inputs are indispensable for the functioning of aquatic systems. Occurrence of eutrophication effects indicate when the system cannot cope with the available internal and or external nutrient inputs. Areas with low hydrodynamic energy conditions or with other suspended matter accumulation stimulating mechanisms (lagoons, estuaries and enclosed seas) can in a relative way be considered as naturally organically enriched. Consequently, little additional material is required to make them eutrophic. In contrast, there are also naturally oligotrophic areas which drain poor upland areas and receive little organic matter.

The inputs of nutrients may lead to both observable and less obvious undesired phenomena which include turbid and foul smelling (mephitic) waters, foam on the beach, oxygen depletion accompanied by a mass mortality of animals and accompanying H_2S smell, proliferation of opportunistic macrophyte algae such as representatives of the genera *Enteromorpha* and

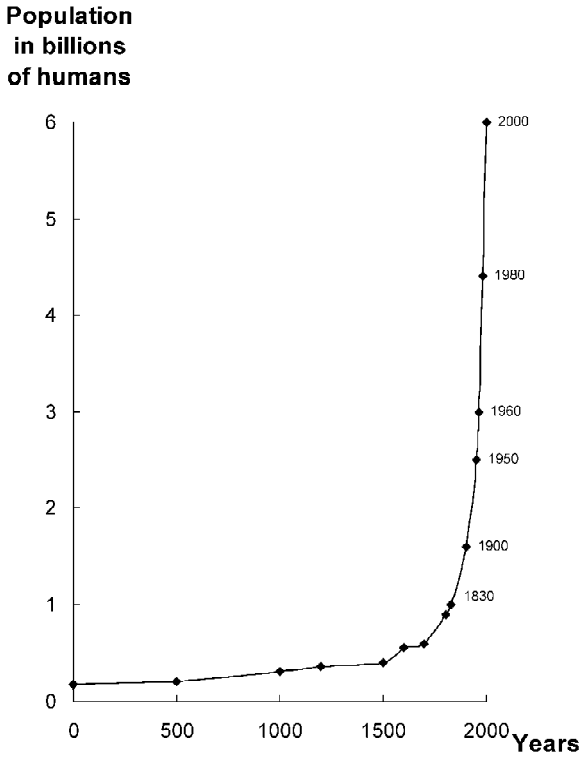

Figure 2. Increase in the world human population since era.

Ulva, the development of noxious algal blooms resulting in DSP, ASP or PSP (respectively, diarrhoeic, amnesic or paralytic shellfish poisoning) or possibly even the occurrence of the recently discovered 'phantom' dinoflagellate (*Pfiesteria piscicida*) along the east coast of the U.S. (Burkholder et al., 1992).

Since era low lying areas near the river and the coasts of the sea have been experienced to be very attractive for transport purposes and thus also for human settlement. This has resulted in a situation where these areas have been developed in densely populated and highly urbanized areas all over the world. Today, with an increasing quality of infrastructure, the human populations in these low lying areas will dramatically continue to increase, e.g. within 10 years, over 75% of the US population will live within 75 km of the coast. In developing countries, there is a continuing migration from rural, inland areas to urban, coastal areas and these trends are exacerbated by the exponential rise in populations (cf. Fig. 2) which in turn has led to large environmental problems such as eutrophication. In addition to the above, increased and more efficient agriculture has included more efficient drainage and thus the increased rate of run-off of organic matter and

nutrients which contribute to high chlorophyll and productivity in estuaries (e.g. Cloern, 1991) and estuarine areas like the Dutch Wadden Sea (de Jonge, 1990, 1997; de Jonge et al., 1996). The increased production of organic waste by agriculture and increasing human population includes excess nutrients from domestic (human-induced eutrophication), intensive arable and pastoral farming and, since the late 1960s, the widespread use of artificial fertilisers in agriculture and domestic and industrial detergents. Different land uses significantly influence the release of sediment, nitrogen and phosphorous from a catchment as was shown for the Chesapeake Bay area (Fig. 3).

The consequence of the characteristics of many coastal areas is that about 28% of the total global primary production takes place here, while the surface area of these systems covers only 8% of the earths surface (Holligan & de Boois, 1993). This means that the effects of eutrophication are most manifest in the coastal zone, including estuaries.

In this introductory paper to the ECSA symposium 2000 "Managing Eutrophication of Estuaries and Near shore Waters: a Challenge for the New Millennium", we focus on the causes and mechanisms of eutrophication as well as the consequences. We give examples varying from eutrophication caused by freshwater inflow to that caused mainly by atmospheric inputs and nutrient import from the sea instead of land and atmosphere. It will be shown that increased nutrient enrichment may lead to organic enrichment and consequently to dystrophication (the modification of bacterial activity) leading ultimately to hypoxia and anoxia.

Historical information

Nutrient enrichment of the aquatic environment has always occurred but the general problem of eutrophication' has been recognised only since the 1800s. Waste deposited in the streams and rivers either stimulated in a positive way the productivity of the aquatic systems or negatively influenced systems by, e.g., structural changes in community structure (species composition) and the occurrence of hypoxia (oxygen depletion) and anoxia (total lack of oxygen). Larger bodies of water such as estuaries, the coast or enclosed seas have a greater assimilative capacity and thus require greater amounts of organic matter to create deleterious effects. For example, C, N and P enrichment has been demonstrated in Chesapeake Bay (U.S.) (Cooper & Bruch, 1991) where enrichment of sediments could be traced back to colonial times. This increase in the carbon content of the sediments could be related to eutrophication since ca. 1800 when the local population started to increase. Andrén (1999) recently reported changes in the diatom flora of the Oder estuary (Germany/ Poland; Europe) since the early 1900s and that this could also be related to increased eutrophication of the heavily populated lower reaches of the river system and its estuary.

In Narragansett Bay (U.S.A.), pre-development nutrient inputs and productivity assessments (Table 1) (Nixon, 1997) suggest that the system presumably was nitrogen limited and that the total dissolved inorganic nitrogen (DIN) input to the system has increased 5-fold and that of dissolved inorganic phosphorous (DIP) 2-fold. During the pre-development period, most of the nutrients (77–93% of DIN input and nearly 100% of the DIP input) probably entered the bay from the ocean by the estuarine water circulation rather than from the drainage basin. This feature was also postulated for European estuaries and the Dutch Wadden Sea some 40 years ago (Postma, 1954). All of these reported values are the best possible historical values but are still approximations. The 'pre-development' primary production for U.S. coastal waters (i.e. before ca. 1800) was obtained by a functional regression of field and experimental data and estimated to have approximated circa 130 g C m^{-2} a^{-1}.

For the rivers Rhine and Ems, the shallow coastal Dutch Wadden Sea and the estuary of the river Ems (Germany, The Netherlands; western Europe), the 'natural background' inputs and concentrations have been calculated (Table 1) given the present main coastline and the present reduced volume and residence time of the connected freshwater systems (van Raaphorst & de Jonge, 2000). These concentrations very well approximate the few reliable values that were published for the early 1930s (see data in Laane et al., 1993), viz. before the large-scale introduction of artificial fertilisers and detergents and the widespread connection of sewage systems to open waters. The data in Table 1 show that there has been a very large increase in nutrient concentrations in fresh water and consequently in its inputs and concentrations in the coastal areas. In comparison to the 'natural background concentrations', the input of total nitrogen increased 12-fold for the river Ems and its estuary, 4-fold for the Baltic (Scandinavia) and 10-fold for the river Rhine. Compared to the 'pre-industrial period'

Table 1. Recent and 'prehistoric' loadings of some systems by nitrogen and phosphorus and the resultant annual primary production of these systems

System (period/ year)	N influx (mmol m^{-2} a^{-1})	P influx (mmol m^{-2} a^{-1})	Mean annual nitrogen concentration in system (μmol l^{-1})	Mean annual phosphorus concentration in system (μmol l^{-1})	Annual primary production (g C m^{-2} a^{-1})
River Rhine & river Ems 'background' situation early 1980s early 1990s			45 ± 25 (tN)	1.8 ± 0.8 (tP)	
English Channel 'background situation early 1930s early 1980s early 1990s			5.5 ± 0.5 (winter NO_3)	0.45 ± 0.05 (winter DIP)	
North Sea 'background' situation early 1930s early 1980s early 1990s			9.1 ± 3.1 (NO_3) (near coast)	0.57 ± 0.13 (DIP) (near coast)	
Dutch western Wadden Sea 'background' situation early 1980s early 1990s			13 ± 6 (tN) ca. 4 (DIN)	0.8 ± 0.3 (tP) ca. 0.3 (DIP)	<50
Ems estuary 'background' situation early 1980s early 1990s	(rivers) 315 (tN) 3850 (tN) 3850 (tN)	(rivers) 16 (tP) 90 (tP) 50 (tP)	(for salinity gradient) 10–45 (tN)	(for salinity gradient) 0.7–1.8 (tP)	
The Baltic Sea 'background' situation (ca. 1900) early 1980s	57 (tN) (rivers+AD+fix) 230 (tN) (rivers+AD+fix)	0.8 (tP) (rivers + AD) 6.7 (tP) (rivers + AD)			80–105 135
Narragansett Bay 'prehistoric' situation 'recent' (1990s)	18 – 76 (DIN) (rivers + AD) 270–330 (DIN) (+ sea input) 1445 (DIN) (rivers + AD) 1725 (DIN) (+ sea input)	∼ 1 (DIP) (rivers + AD) 61 (DIP) (+ sea input) 73 (DIP) (rivers + AD) 140 (DIP) (+ sea input)			130 290
Long Island Sound 1952 early 1980s	- 1040 (tN) (rivers + AD)	- 70 (tP) (rivers + AD)			ca. 200 300
Chesapeake Bay mid 1980s	290–2140 (DIN) 1430 (tN)	30 (DIP) 40 (tP)			400–600

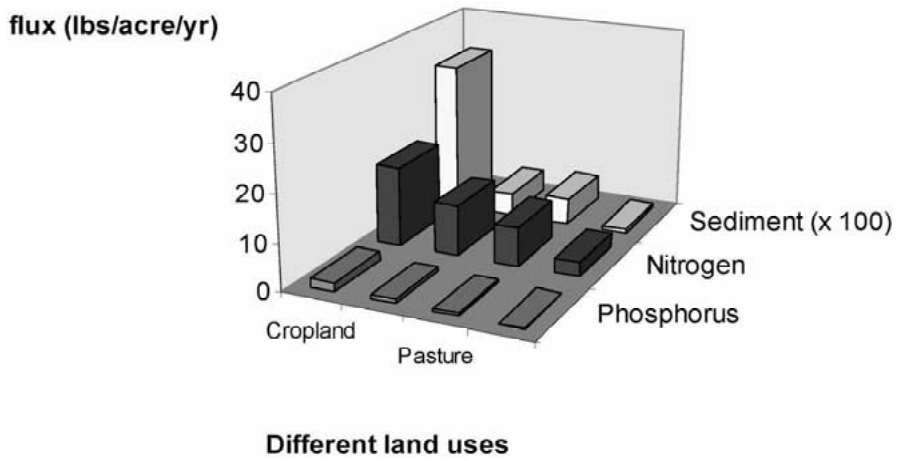

Figure 3. Release of sediment, nitrogen and phosphorus from catchment to Chesapeake Bay (modified after data from Magnien et al., 1995).

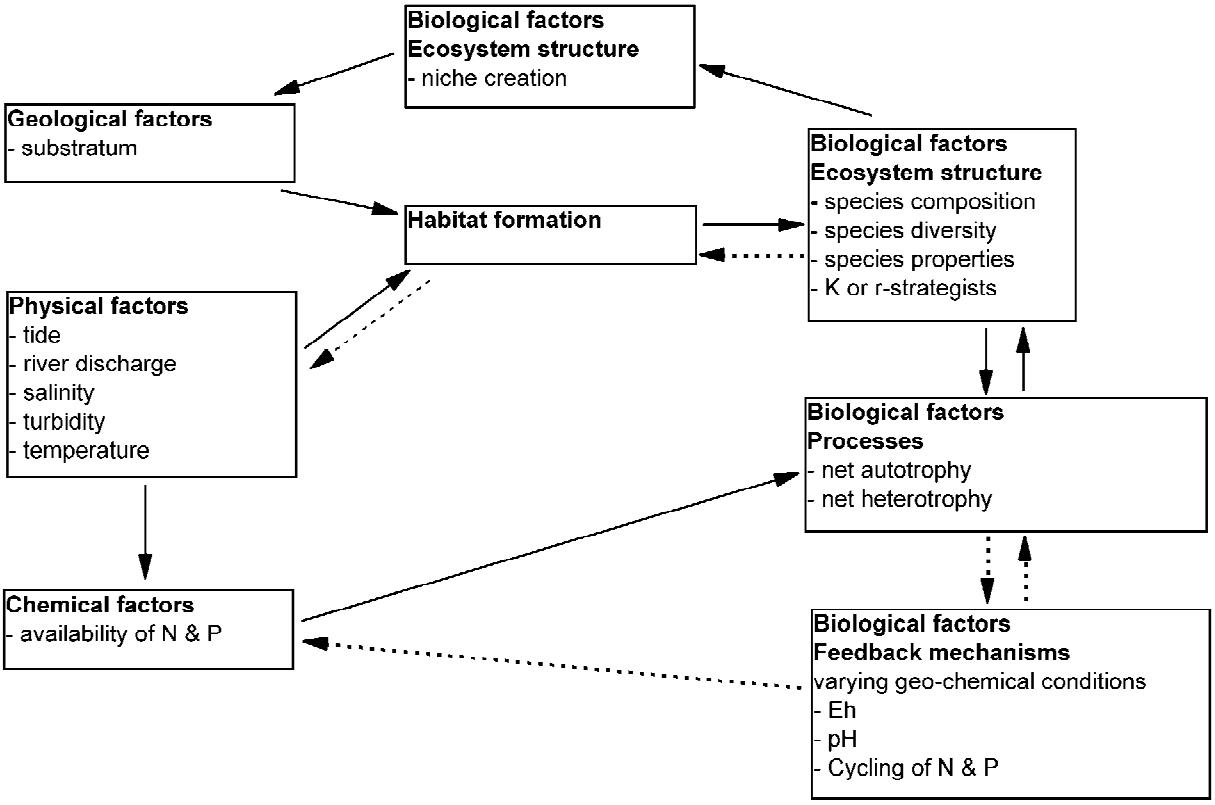

Figure 4. Operating forcing variables in the development of estuarine and marine biological communities.

the loads of total dissolved nitrogen (DIN) increased over 19-fold for Narragansett Bay (USA).

For total phosphorous (Table 1), the increase was over 8-fold for the Baltic, 6-fold for the river Ems and its estuary and 12-fold for the load of the river Rhine. Compared to the 'pre-industrial period' the phosphate (DIP) loads possibly increased over 50-fold to Narragansett Bay. The examples given above cover the development period of the now developed countries. Consequently, the future industrialisation and urbanisation of presently less-developed countries is likely to give a future similar trend.

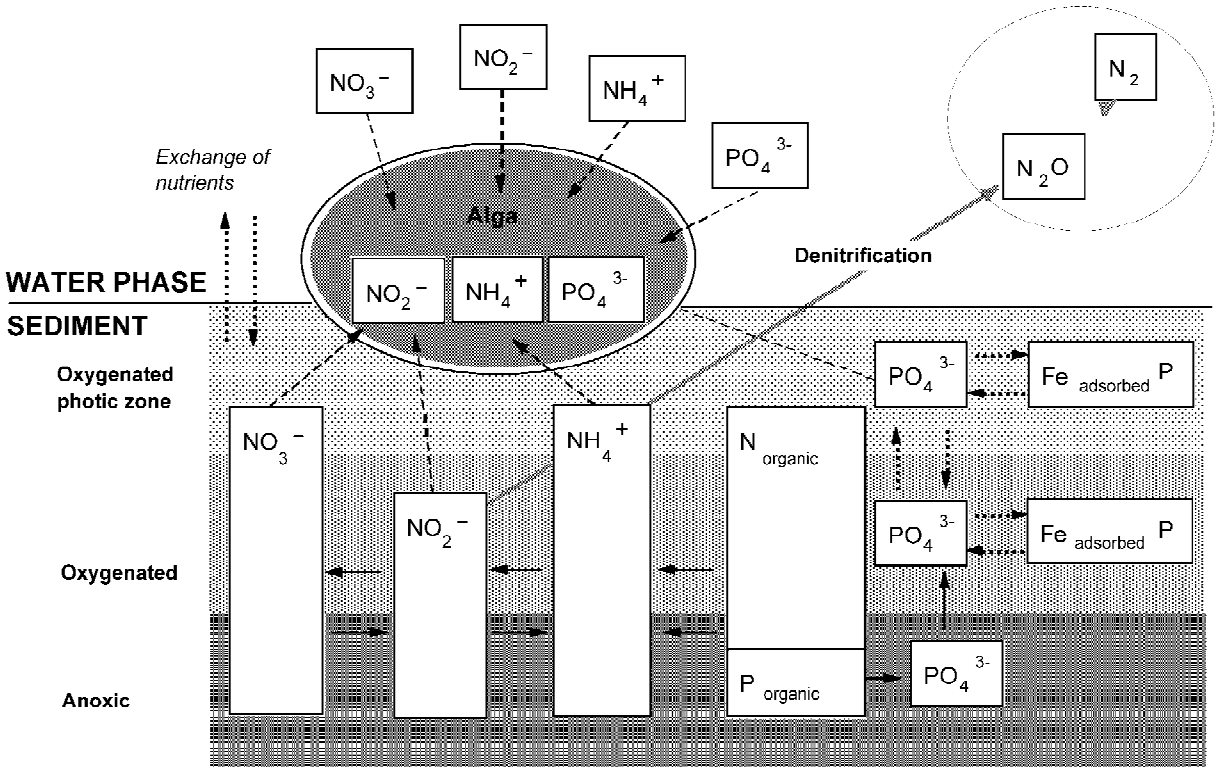

Figure 5. Schematic representation of conversion processes of nitrogen and phosphorus. Dashed lines represent assimilation, full lines represent biotic conversion (mainly microbial) and dotted lines represent geochemical equilibria. (Modified after Wiltshire, 1992 and van Beusekom & de Jonge, 1998 and references therein).

The differences in loads over the different systems are an important function of the land use (today as well as in the past). The effect of land use on nutrient and sediment release (Fig. 3) shows the absolute as well as the relative differences in these fluxes for the U.S.A. which also will be relevant for other countries with a comparable land use and application of modern equipment. The changed loads per system over time reflect the historical development in human activities and thus the changes in land use while for recent years the effect of P-reduction measures (cf. values for 1980s and 1990s for rivers Rhine and Ems in Table 1) are also notable.

Structuring elements and processes

In addition to the inputs, the mechanisms and processes which will influence the fate and effects of excess nutrient inputs will be considered. In coastal systems, eutrophication will influence not only the nutrient-related processes of the system but will also affect structural elements of the ecosystem.

Structuring elements

The physical and chemical characteristics mainly create the basic habitat conditions and niches of the marine system to be colonised with organisms. These conditions also determine colonisation rate which is dependent on the organisms' tolerances to environmental variables. Following this, the biological processes such as primary production and decomposition can modify the physico-chemical conditions while the biological inter-relationships such as reproduction, recruitment and predator–prey cycles can modify the community structure which in turn can have further consequences for the modification of the physical and chemical system in the water phase and sediment (Fig. 4). In the case of eutrophication, these processes can be summarised by the nutrient levels producing conditions for high primary production and consequent degradation possibly modifying the system through dystrophication.

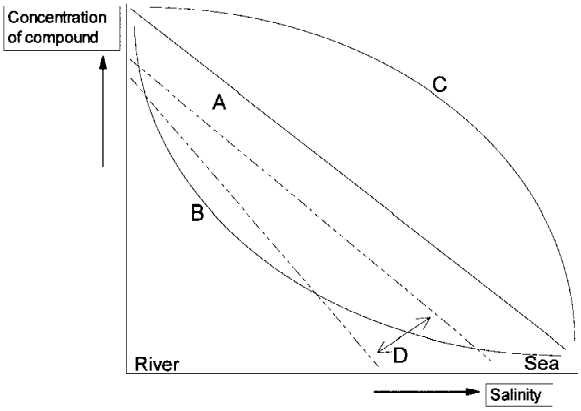

Figure 6. Typical concentration gradients between the non-tidal fresh river water and the sea. Dilution lines. (A) conservative dilution of substance within the area, (B) condition with removal of substance between the end members, (C) situation with addition of substance, (D) Illustrating different levels and slopes due changes in input and transformation rate in the system under consideration (Estuarine gradients after Officer, 1979; tidal river spikes after van Beusekom & de Jonge, 1994).

The autochthonous inputs of organic matter to the estuarine and coastal systems include reed beds, salt marshes, phytoplankton, microphytobenthos (benthic micro algae) and macro algae, whereas the allochthonous inputs are from riverine, atmospheric and marine sources, terriginous-blown material and anthropogenic inputs such as sewage. Each of these sources supplies organic matter to the system and many of the former will retain nutrients for a short period before being degraded and releasing the nutrients through the microbial loop.

Processes

As a transition area, estuaries play a filtering role for both dissolved and particulate matter, including the compounds associated to it. These estuarine areas can be considered as a reactor vessel with input, transformation, sink and output functions related to organic carbon and nutrients. Part of the transformation processes is illustrated in Figure 5 (Wiltshire, 1992; van Beusekom & de Jonge, 1998 and references therein). From the figure presented it is clear that estuaries may operate as sources and as sinks of organic matter and nutrients.

Import

The import from especially the sea is difficult to assess. This is due to the fact that the net transport of the several compounds is only a very small fraction of the total amounts moving in and out with the flood and the ebb tide. In the Ems estuary, the annual import of organic carbon was for example estimated to be only 4.5% of the annual longitudinal flux through the tidal inlet (de Jonge, 1995). Similarly, the river import to an estuary is poorly known and difficult to assess especially as in the freshwater tidal zone the mud from either marine or terrestrial origin alternately is in contact with either fresh or brackish water over a single tidal cycle. Due to the differing ionic composition of these water bodies at the freshwater–brackish water interface, sudden and extreme variations in physico-chemical conditions result in rapidly changing adsorption–desorption equilibria and consequently spiking of concentrations of several elements as dissolved Fe, dissolved Al, DIP, Eh, pH, dissolved organic carbon (Fig. 6). A possible solution for the problem of calculating the river input was presented by Officer (1979) and was based on calculating the river inputs based on estuary reaches with apparent conservative nutrient behaviour.

Transformation

Within the estuarine system, the shape of gradients in nutrients and suspended matter, and also the slopes and levels of it all vary over different time scales. General patterns are, however, observable. Important large-scale transformations take place in which biological (uptake of nutrients by primary producers, transformation by grazers and degradation by detrivores) as well as geo-chemical processes (adsorption-desorption, precipitation and dissolution) and burial of substances play a role. This produces a model of conservative behaviour and the illustrative deviation by nutrient measurements from that behaviour. In the absence of any operating process, except dilution, any concentration gradient of a substance with relatively high concentrations at the freshwater end-member and relatively low values in the sea would be a straight line (curve type 'A' in Fig. 6). When, however, significant removal of the substance occurs, due to e.g. transformations or deposition, the curve type 'B' will appear while during significant addition of substances, by point sources or local transformation processes, curve type 'C' will emerge. However, the addition or withdrawal of substances over the entire system will also change the slope of the gradient or even the level of the entire gradient ('D' type in Fig. 6).

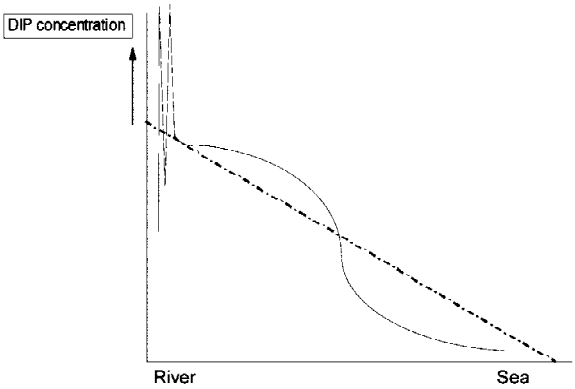

Figure 7. Typical concentration gradient of dissolved inorganic phosphate (DIP) in a turbid mesotidal estuary with a mean flushing time of ca. 30 days.

The concentration gradients of total dissolved nitrogen (DIN) and reactive silicate are often of the 'A' type (Fig. 6), while those of reactive dissolved phosphate (DIP) often experience distributions of the 'B' type as well as the 'C' type or even mixtures (Fig. 7). This is due to transformation processes (a combination of biological, geochemical and physical processes operating) and the strong reactivity of phosphate in aquatic systems. This also indicates that the behaviour of silicate, nitrogen and phosphorous differ in aquatic coastal systems. The cycle of nitrogen is almost completely determined by biological processes, that of silicate by a combination of physical processes (e.g. the physically determined dissolution of diatom frustules) and geo-chemical conditions (adsorption–desorption processes) while the phosphorous cycle is strongly determined by biological processes in combination with geo-chemical processes (adsorption–desorption processes and apatite formation). The basic pathways of the elements N, P and Si is presented in the simplified cycles of Figure 8.

The governing processes with the dominant effect depends on physical factors such as the input of nutrients, the turbidity of the system, the residence time (see below) and occurrence of stratification (Fig. 9). Systems with low turbidity are more sensitive to increased nutrient concentrations than highly turbid systems as productivity is directly governed by light extinction coefficient. With increasing natural turbidity of coastal systems, this phenomenon becomes proportionally less important. For example, in most of the North Sea estuaries which have a notable turbidity maximum zone in their upper reaches, it does not play any role in the upper reaches but does in the lower reaches cf. de Jonge, 2000). It is, however, extremely important for fjordic systems (Norway, Scotland), the Baltic Sea and very sheltered estuarine systems as found in the U.S.A. Similarly, stratification causes different physico-chemical conditions which may result in hypoxia or anoxia and corresponding processes such as hypoxia-stimulated denitrification and, due to low redox conditions, high concentrations of DIP (Fig. 5). The produced compounds may be transported to the surface layers where they will stimulate process rates.

Retention

There are two main mechanisms that stimulate retention of nutrients: denitrification (output of nitrogen to atmosphere) and burial (Fig. 9). Only the latter represents the physical retention of substances. The residence time or the flushing time of fresh water determines the degree of retention of fresh water and implicitly also determines how much of the available nutrients will be removed from the system by incorporation and storage as plant tissues, to the atmosphere (denitrification) or to the bottom sediments (by burial). All systems show, dependent on flushing time and geomorphologic characteristics, a certain removal or disappearance of nutrient compounds.

Output

Denitrification may significantly contribute to the removal of nitrogen under favourable conditions. There are indications that denitrification and nitrification are stimulated by the existence of oxygenated and hypoxic water masses (and sediment pockets) close to each other. The relative nitrogen removal by denitrification ranges between 20 and 50% of the annual input (Fig. 10 after Seitzinger, 1988 modified and supplied with recent data). This suggest a wide variation in bacterial processes enhancing the transformation of nitrogen. The presented values are, however, based on different approximations and measuring methods so that a good and close comparison of the data presented so far is difficult. Newer methods lead to higher figures.

Nutrient mass balances show remarkable differences between systems. In Table 2, values are presented for three systems which largely differ in size (Ems estuary, Chesapeake Bay and Baltic Sea). These systems are situated in western countries with relatively high population density. Important differences emerge

Table 2. A phosphorus and nitrogen budget for the Ems estuary for 1992/93 (van Beusekom & de Jonge, 1998) and for the Chesapeake Bay (Boynton et al., 1995) and the Baltic Sea (original data from Larsson et al., 1985, cf. also de Jonge et al., 1995)

Ems estuary	Phosphorus		Nitrogen	
(463×10^6 m$^{2)}$	(%)		(%)	
τ_f: 0.11 yr				
Mean K_z: 8.2–0.8	Input	Output	Input	Output
Rivers	71.4		92.3	
Point sources	~ 0			
Atmosphere	1.4	0.4	1.5	20.2
N-fixation				
Sea (dissolved)		59.3		77.1
Sea (particulate)	27.3	6.2		
Sediment (burial)		40.3		2.7
Water:				
accumulation				
Fisheries harvest	~ 0			
TOTAL	44.7 (100%)	44.7 (100%)	2362 (100%)	2278 (100%)
($\times 10^6$ mol yr^{-1})	96 mmol m^{-2} a^{-1}		5100 mmol m^{-2} a^{-1}	
Chesapeake Bay	Phosphorus		Nitrogen	
($11\,478 \times 10^6$ m^2)				
τ_f: 0.5 yr				
Mean K_z: 3.0–0.75	Input	Output	Input	Output
Rivers	43.7		59.7	
Point sources	26.4		28.3	
Atmosphere	4.8		12.0	26.0
N-fixation				
Sea (dissolved)		0		30.3
Sea (particulate)	25.1			
Sediment (burial)		96.3		34.8
Water:				
accumulation				
Fisheries harvest		3.7		8.9
TOTAL	485 (100%)	485 (100%)	10834 (100%)	10 834 (100%)
($\times 10^6$ mol yr^{-1})	42 mmol m^{-2} a^{-1}		944 mmol m^{-2} a^{-1}	
The Baltic Sea	Phosphorus		Nitrogen	
($373\,000 \times 10^6$ m^2)				
τ_f: 22 yr				
Mean K_z: 'clear'	Input	Output	Input	Output
Rivers	65.5		54	
Point sources	26.5		8	
Atmosphere	7.2		27	70–75
N-fixation			11	
Sea (dissolved)		10		10
Sea (particulate)				
Sediment (burial)		70		5–10
Water:		20		10
accumulation				
Fisheries harvest				
TOTAL	2500 (100%)	2500 (100%)	85 000 (100%)	85 000 (100%)
(10^6 mol yr^{-1})	6.7 mmol m^{-2} a^{-1}		228 mmol m^{-2} a^{-1}	

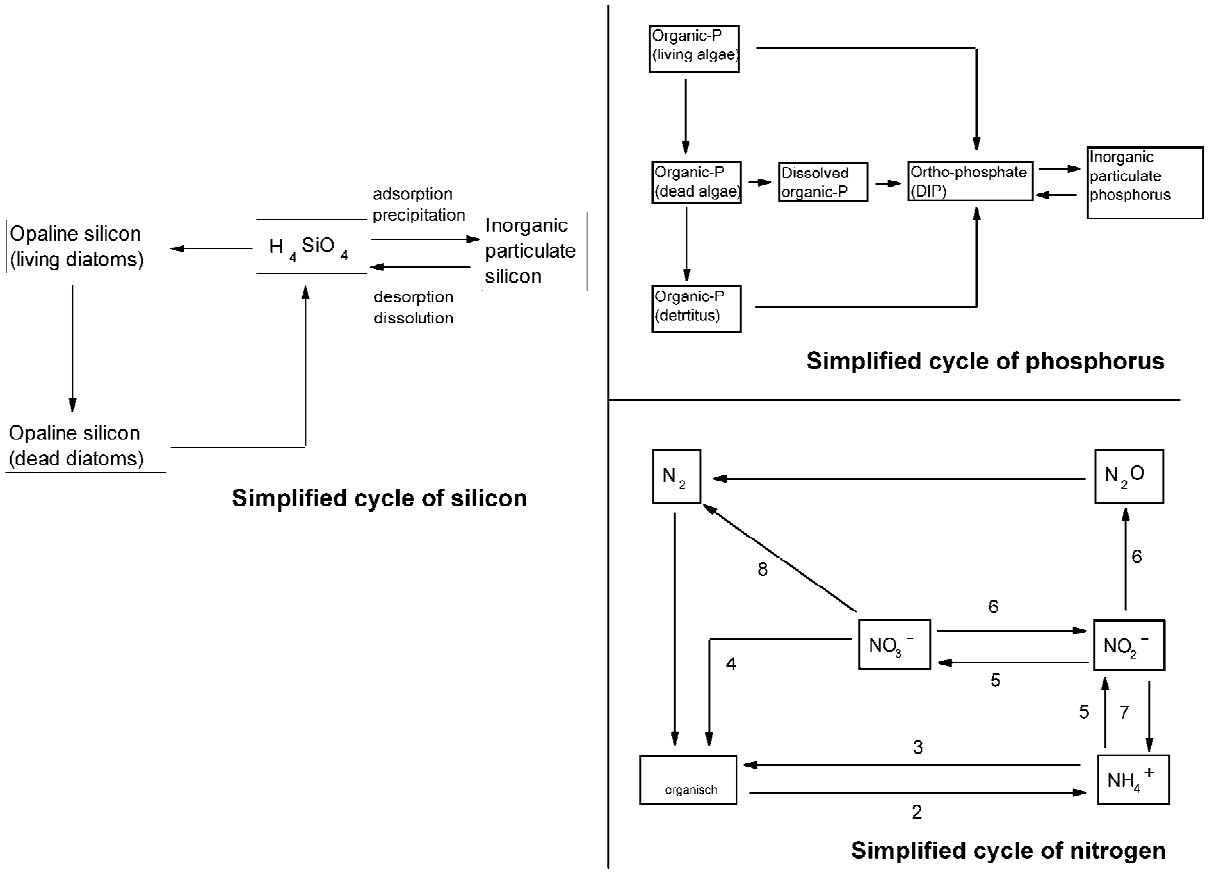

Figure 8. Cycles of N, P and Si. Explanation of the number in the N-cyclus: 1. Nitrogen fixation 2. Aerobic or anaerobic degradation or ammonification 3. Ammonium assimilation by algae 4. Nitrate assimilation by algae (at low ammonium concentrations) 5. Nitrification by chemo-autotrophic bacteria: a: $NH_4^+ + 1\ 1/2\ O_2 \rightarrow NO_2^- + 2\ H^+ + H_2O$ b: $NO_2^- + 1\ 1/2\ O_2 \rightarrow NO_3^-$ 6. Dissimilative nitrate reduction as denitrification 7. Dissimilative nitrate reduction or nitrite ammonification by a special group of bacteria (fermentative bacteria as *Clostridium* spec. and *Bacillus* spec) in strongly reduced environments as bottom sediment 8. Anaerobic ammonium oxidation (anammox) $5NH_4^+ + 3NO_3^-\ 4\ N_2 + 2\ H^+ + 9H_2O$.

when focusing on relative importance of the output of nutrients.

In the Baltic Sea and the Chesapeake Bay, both with a long flushing time, respectively, 90% and 96% of all the phosphorous influx is accumulated within the system (both sediments and water column) decreasing to only 40% for the Ems estuary due to mainly apatite formation (van Beusekom & de Jonge, 1997, 1998). For nitrogen, the differences are even greater as nearly 75% of the influx for the Baltic Sea is removed by denitrification, while in the Chesapeake Bay 35% of all the nitrogen influx is buried and for the Ems estuary most of the nitrogen influx (almost 80%) is transported to the open sea. This information shows the importance of the water column in the Baltic Sea as an accumulation compartment for nutrients.

The great differences between systems are related to the input of nutrients and to the differences in the dimensions of these systems and related factors such as the tidal range (energy) and freshwater inflow (flushing time of fresh water, residence time of fresh or seawater and turnover time of the basin water) and related important determinants as turbidity.

Despite the inherent variation, it is possible to define the most important factors in the expression of eutrophication and those that may qualitatively predict which type of system is sensitive to eutrophication (Fig. 11). The most important factors are: flushing time (f_τ), the turbidity (gradient) expressed as light extinction coefficient (k_z), and the input (external as well as internal turnover flux) and consequent concentration (gradient) of the nutrients N, P and Si. The

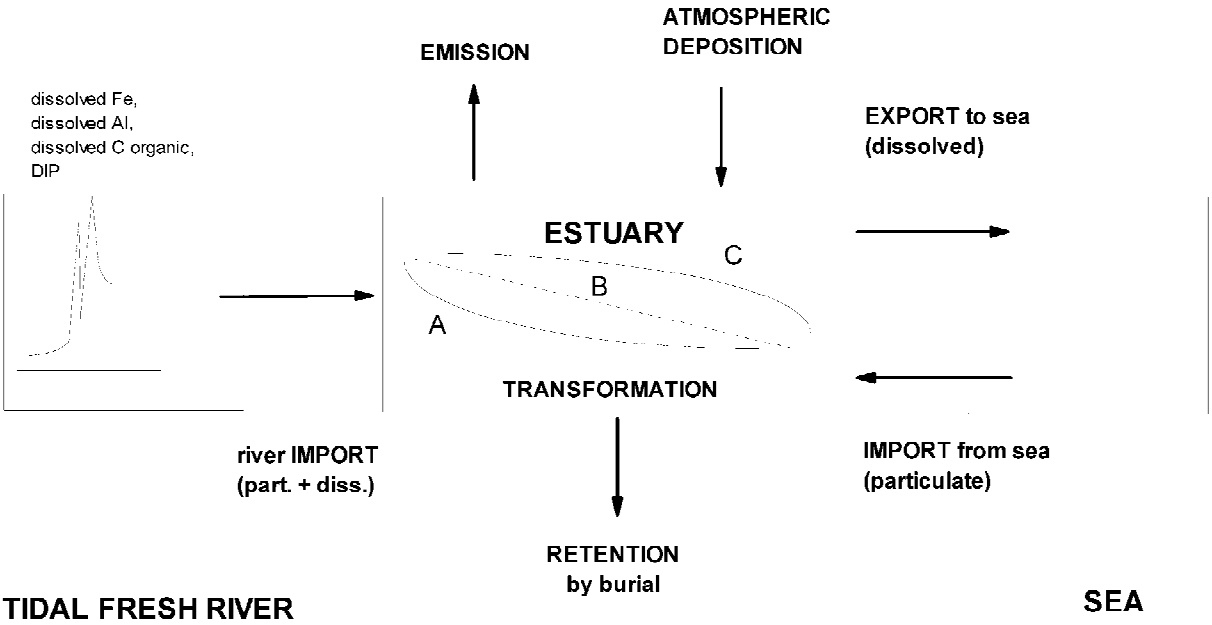

Figure 9. Diagram representing the main governing processes in an estuary.

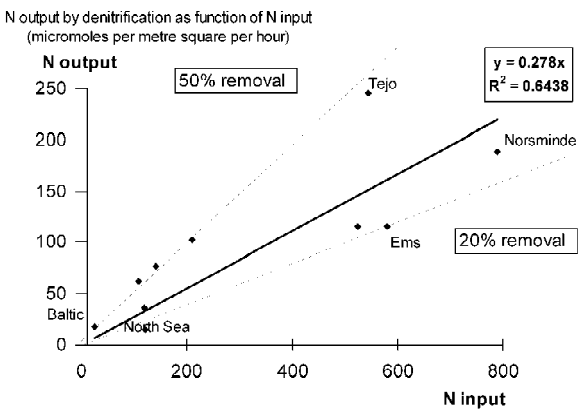

Figure 10. Scatter plot with regression line of denitrification as a function of N input of a number of systems. The systems indicated show removal between 20 and 50% of the input. Figure partly based on Nixon et al. (1996) and new data. Ems estuary (van Beusekom & de Jonge, 1998), Norsminde Fjord (Nielsen, K, L.P. Nielsen & P. Rasmussen, 1995), North Sea (Lohse et al., 1996), Wadden Sea (Kieskamp et al., 1991) and Western Scheldt (Middelburg et al., 1996).

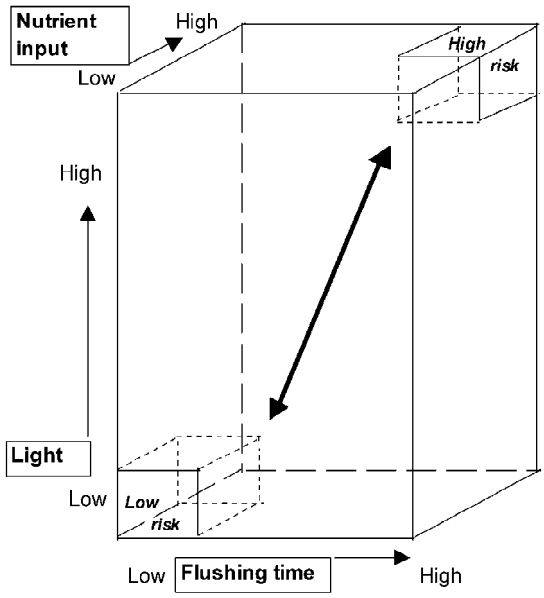

Figure 11. A 3-D classification scheme of eutrophication risk of estuaries based on flushing time, turbidity and nutrient input.

combination of mainly these three factors determines whether an estuarine system has a low or high risk of producing eutrophication symptoms.

The longer the flushing time, then the more vulnerable the system is to nutrient enrichment as the primary producers have a greater period to utilise the excess nutrients. If the flushing time is shorter, than the mean growth rate of the algae (areas with high dispersion capacity), flushing of the population will occur and thus prevent problems within the system although transport to the open sea will increase. Hence, if algal blooming does not occur within the estuary it may develop in the lowest reaches of the system or even just

outside the system in the sea. A factor not included in Figure 11 is the mean depth of the system and its mixing state (stratified or not) which, in combination with (f_τ), (k_z) and nutrient influx, may indicate the sensitivity of any system to eutrophication.

Trends and symptoms in eutrophication

Trends in eutrophication

Several trends indicate changes that have occurred in systems recently or since era. In doing this we may follow the diagram in Figure 9, thus considering input values, gradients within the estuary and output values. The symptoms following from these changed input values will be given in a separate subsection.

An example of the changes in input values are given for one of the important European rivers, the Rhine (Fig. 12) and another one for Laholm Bay in the south-western part of Sweden (Fig. 13). The river Rhine graph shows that the maximum phosphorous loads were reached in the late 1970s and early 1980s followed by a dramatic decline due to measures taken. The nitrogen loads, however, increased from the late 1950s until the late 1980s. Since then, the values dropped back to those of the 1960s.

This picture clearly shows the delay between the two components which is mainly due to the fact that phosphorous loads are better controllable than those of nitrogen because phosphorous mainly reaches the estuary from point sources while nitrogen is mainly derived from diffuse sources.

The primary effects of these long term fluctuations in loads, which were not caused by the fluctuations in river discharge, were also variations in the concentrations of these compounds in the fresh water of the river Rhine.

The biological response of the increases in the loads and concentrations were increases in the annual primary production along the coast and in the Wadden Sea, the biomass of macrozoobenthos in the western Dutch Wadden Sea, the meat content of the blue mussel and the concentrations in the western Dutch Wadden Sea (de Jonge et al., 1996).

Apart from changes in fresh water, also the concentrations of DIN and DIP in our coastal systems, the Wadden Sea included, changed over time (de Jonge & Postma, 1974; de Jonge, 1997). Even more remarkable is that the effect of the river Rhine plume was restricted to mainly the western Dutch Wadden Sea as no clear indications for eutrophication have been found for the eastern Dutch Wadden Sea (de Jonge & Essink, 1991; van Beusekom & de Jonge, this volume)

Of great concern is that also DIN and DIP (winter values) in part of the inflowing water of the North Sea, the Strait of Dover, have been increased since 1960. Whether this increase can be ascribed to human induced inputs or to natural phenomena like local up welling or the NAO (North Atlantic Oscillation) is not yet clear, but the effects are very important. Due to political and management measures, the concentrations of DIP in the inflowing fresh water has decreased substantially locally even reaching levels that are lower than the winter levels in the Strait of Dover. When applying the simple but very illustrative conservative plot of fresh water values and those of Atlantic water it becomes clear that at present the sea may be a source for phosphorous again instead of the main freshwater bodies as it must have been under pristine conditions.

The trends in loads and in concentrations of nitrogen in the coastal waters have not strongly changed yet and consequently N/P ratios have increased dramatically in Dutch coastal waters since the successful reduction measures for phosphorous. The combination of the still high nutrient concentrations and the increasing ratios have been blamed for the occurrence of nuisance blooms.

Symptoms of eutrophication

General effects

Based on the information presented so far, it may be difficult to assess the anthropogenic impact against the background of the natural variability in nutrient concentrations and related processes. This means that also the interpretation of apparent eutrophication related symptoms is sometimes very difficult.

Increase in the decomposition of organic matter may lead to hypoxia and even anoxia although these effects are strongly dependent on the turbulence of the system.

This degradation of organic material reflect a major community response, by species changes through the loss of intolerant or k-selected species and the increase of opportunistic or r-selected fauna. Loss of vascular plants is a primary effect due to which complete communities may disappear as happened in Australia and North America. Although very often the loss of the long living and slow growing species (i.e. k-selected species) is compensated by the occurrence of more op-

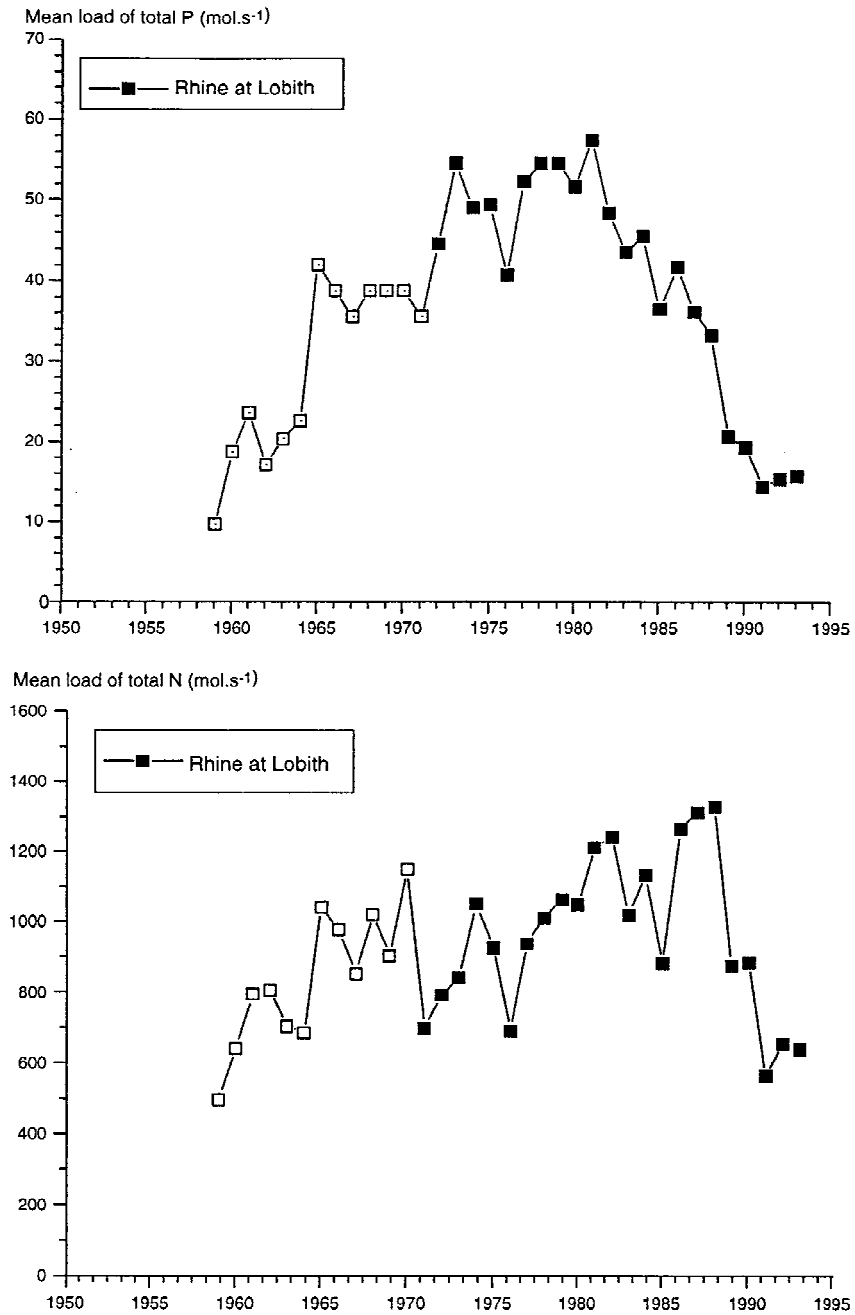

Figure 12. Development of the loads in total phosphorus and total nitrogen as measured on the border between Germany and The Netherlands (data from Rijkswaterstaat and van Raaphorst & de Jonge, 2000).

portunistic macrophytes (i.e. *r*-selected genera) such as representatives of the genera *Enteromorpha*, *Ulva*, *Chaetomorpha*, *Rhizoclonium* and *Cladophora,* this represents a deterioration of quality as these algae may form dense layers or mats associated with oxygen deficiency and high degradation rates of organic matter leading to anoxia (Raffaelli et al., 1998).

An increase in nutrient ratios may, based on differences in affinity of algae to nutrients, cause a change in the algal species composition.

The occurrence of noxious blooms of toxin-producing algae is another problem. Possibly even worse is the phenomenon of the 'phantom' algae *Burkholder et al., 1992) of which the possible re-

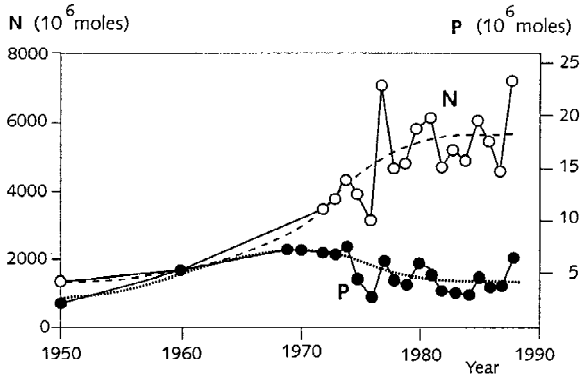

Figure 13. Nutrient loads to Laholm Bay at the south-western coast of Sweden (after Rosenberg et al., 1990) (with permission).

lation with eutrophication is yet unclear. Harmful algal blooms seem to be more common today than in the past. They can have serious effects in fish kills and neuro-toxin production (PSP, DSP, ASP) affecting animals with complex nervous systems.

The amenity value of shores is negatively influenced through foams produced by the gelatinous microalgae such as *Phaeocystis* and *Chaetoceros socialis.* The presence of the 'phantom' algae will completely destroy locally any form of recreation which will strongly affect local economy. Low turbidity systems may change from macrophyte-dominated into a phytoplankton-dominated system even with microalgal blooms affecting the integrity of the system.

In highly turbid systems where primary production is light limited, the situation usually differs from the above because benthic communities are usually poor in vascular plants like eelgrass and large brown algae. However, the effects of eutrophication from riverine sources in these systems will occur in nearby coastal area (cf. de Jonge et al., in press). Furthermore, water transport processes dictate that the effects of nutrient enrichment occur either in the lower reaches of the estuary or in the coastal area itself. Ultimately, the effect of 'habitat loss', poor water quality and a deterioration of the marine environment may negatively influence any form of human use.

Large scale effects

Symptoms of eutrophication are for instance the increase in productivity of seas as e.g. North Sea (Peeters et al., 1993) and the Wadden Sea (de Jonge et al., 1996 and references therein), the observed increase in surface algal blooms in the Dutch EEZ of the North Sea (Fig. 14), a major sudden change in plankton composition from diatoms to small flagellates (Hickel et al., 1993) and the (temporarily) large scale oxygen deficiencies in e.g. the North Sea (Zevenboom, 1993; Zevenboom et al., 1997), the Baltic, Long Island Sound and Chesepeake Bay (de Jonge et al., 1995).

Eutrophication in Florida Bay possibly produced a large scale sea grass die off (4000 ha of *Thalassia testudinum* and *Halodule wrightii* disappeared between 1987 and 1988), followed by increased phytoplankton abundance, sponge mortality and a perceived decline in fisheries (Rudnick et al., 1999). This very large change in the health of the system followed major engineering works to the Everglades area and reflected changes in the nutrient concentrations, the nutrient pool, the chlorophyll levels and turbidity. The preliminary nutrient budget for the bay assumes a large oceanic and atmospheric input of N and P to the Bay although the denitrification rates are unknown. The cause(s) of the sea grass mortality in 1987 is still unknown.

Another symptom is the dramatic increase of the permanent anoxic layer of the Baltic which rose from 18 000 to near 70 000 km^2 which is nearly the entire hypolimnion (Jonsson et al., 1990). The increase of the anoxic layer had dramatic ecological consequences. It resulted in a dramatic structural decline in population size of two important prey species (the large isopod *Saduria entomon* and the snake blenny *Lumpenus lampetraeformis*) which in turn greatly influenced the local cod populations (Elmgren, 1989). The given changes were the result of an increase in the primary production of only 30–70%. Another consequence was an increase in the density of algal mats which reduced the development of herring eggs, possibly by production of exudates. Finally, the hypoxic areas adversely affected cod egg development (Elmgren, 1989).

Sommer (1996) performed experiments with periphytic micro algae showing complex behaviour under changing ratios of Si, P and N. Relative high Si led to dominance of diatoms while relatively low Si and low N favoured the development of cyanobacteria while chlorophytes took advantage at low Si and low P. Moreover, variations in light intensity influenced the outcome.

The extensive bloom of the micro alga *Chrysochromulina* species in the late 1980s in the outer Baltic apparently developed in response to nutrient build up on the eastern North Sea and contributed to the hypoxia and eutrophic symptoms. Near the west Atlantic coast in the U.S.A. the 'phantom algae' (*Pfiesteria pis-*

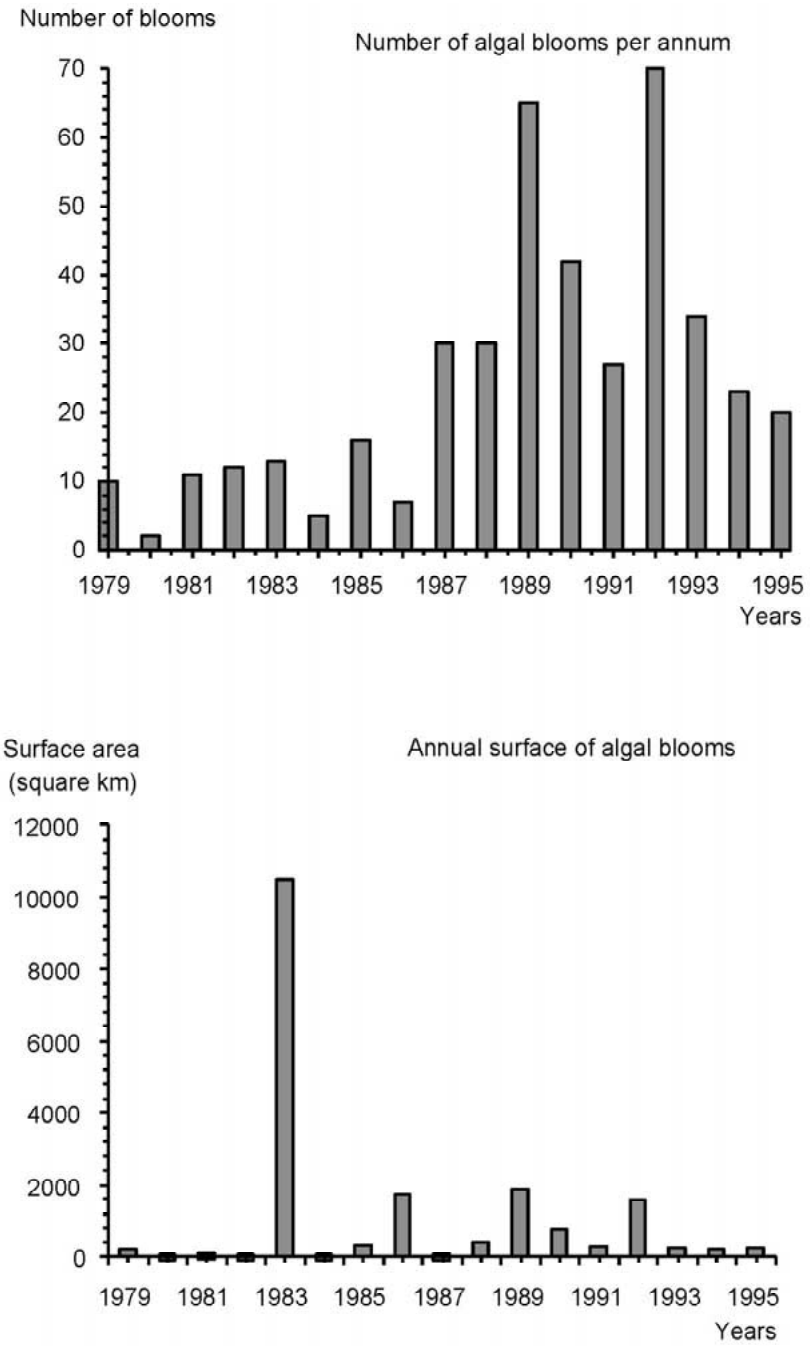

Figure 14. Surface algal blooms observed during Dutch airborne surveys over the period 1979–1995 (after Zevenboom, 1993; and Zevenboom et al., 1997) (A) frequency per annum (B) surface area in km^2 per annum.

cicida) occurred since the early 1990s (Burkholder et al., 1992).

Observed changes in primary producers generally reflected vascular plants operating as *k*-strategists under stress and replaced by more opportunistic algae like phytoplankton species (cf. Margalef, 1978). Although not discussed here, also synergistic effects where eutrophication effects are exacerbated by other pollutants are expected in urbanised and developed areas.

Local effects

Apart from large scale general effects as reported above, there are also strong local effects observable due to fish farming (Bonsdorff et al., 1997). These authors further stated that 35% and 55% of the total local nitrogen and phosphorous inputs are accounted for by local inputs leading to a strongly reduced N/P ratio. The local nutrient enrichment in the Baltic produced an increase in the primary production and an increase in turbidity which negatively affected the macrophyte populations as eelgrass (*Zostera marina*). It further stimulated the blooming of cyanobacteria. Loss of five benthic crustaceans have been observed while a gain of four was reported of which two were polychaetes (*Polydora redeki*, *Marenzelleria viridis*) new to the area. The macrozoobenthos community further showed a structural change from suspension feeders to deposit feeders.

The Peel-Harvey estuarine system (South Pacific West coast of Australia) receives a high nutrient loading, but has an additional phosphorous release during stratification-induced anoxia from the bottom sediments. This happens after a clear loading of the estuary and the subsequent development of dense populations of microphytobenthos which is responsible for the nutrient storage (McComb & Lakatelich, 1995). This release contributed to changes in macro algal community structure and increased turbidity due to algal blooms.

Creation of red-tides (noxious, toxic and nuisance micro algal blooms) in Tolo Harbour (Hong Kong) resulted from large urban nutrient inputs, a water residence time of 16–42 days and a low turbidity (cf. Figs 15 and 11) which led to dense phytoplankton populations. Diatoms decreased in abundance from 80–90% to 53% in 1982–85, dinoflagellates increased concurrently with red tides and chlorophyll-*a* levels also increased significantly. Oxygen depletion occurred due to nutrient loadings and the local development of phytoplankton in combination with reduced water exchange (long flushing time), features associated with a low-energy eutrophic environment (Hodgkiss & Yim, 1995).

In other systems, e.g. North Sea waters regular blooms of nuisance algae occurred of which it is assumed that these are related to high nutrient concentrations and high N/P ratios (Zevenboom et al., 1997). Among these species are *Phaeocystis* species, *Noctiluca scintillans*, *Gyrodinium aureolum*, *Dinophysis acuminata*, *Alexandrium tamarensis*, and several *Prorocentrum* species.

Effects at the conceptual level

A conceptual model of the main effects of eutrophication is presented in Figure 16 which gives the consequences of nutrient enrichment on primary production and for nutrient cycling, and shows the secondary effects of turbidity, changes in species composition and organic decomposition. Increased nutrient concentrations may lead to changes in both the species composition (lethal and sub lethal concentrations of nitrate and ammonium to e.g. sea grasses) and the primary production. The turbidity effect is particularly relevant to systems with a natural low turbidity because these often contain communities of macrophytes and are therefore sensitive to nutrient enrichment. Among these vulnerable species are macro algae and/or vascular aquatic plants (e.g. sea grasses) which have a restricted immersion tolerance and thus may be lost from the low turbidity systems in poor light conditions.

Future challenges

The variation in the percentage of denitrification of the total N input (Fig. 10), which might partly be caused by the application of different methods, suggests that it is necessary to reconsider the importance of this process.

The finding of anaerobic ammonium oxidation (Mulder et al., 1995) suggests that it might be worthwhile to further explore this and other unknown nitrogen transformation processes and routes in estuarine areas. In addition to these transformation processes and given the trend in decreasing nitrogen loads, it seems of increasing importance to study N fixation processes under estuarine conditions (e.g. Pearl, 1990; Pearl et al., 1994).

There are several recent and historical indications for the significant role of geochemical processes on extremely strong fluctuations in values of e.g. pH, Eh, DIP, dissolved Fe, Al and carbon in the freshwater tidal part of rivers which strongly influence particle composition (Morris et al., 1982, 1987) and hamper measuring of correct input data (van Beusekom & de Jonge, 1994). It is necessary to further unravel the interaction between fresh water, brackish water, river born mud and sea born mud during the tidal cycle and its impact on the above given concentrations.

The internal silicate input within estuarine systems from local, weathering of clay minerals should be de-

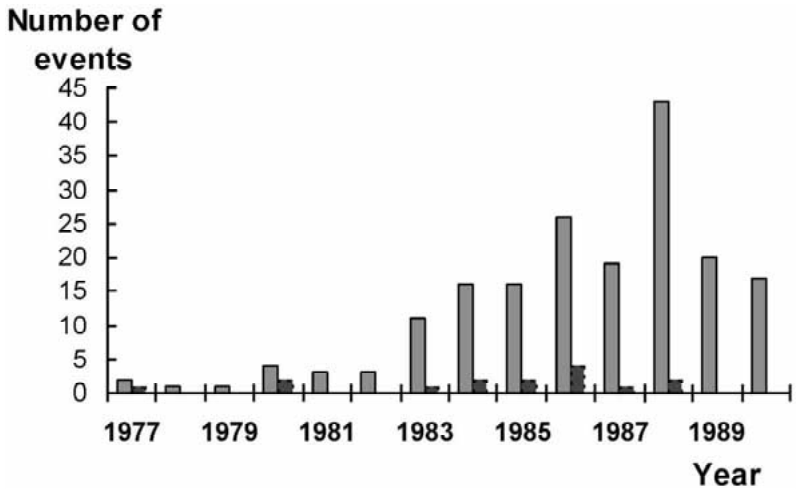

Figure 15. Red tides and associated fish kills in Tolo Harbour (after Hodgkiss & Yim, 1995; with permission).

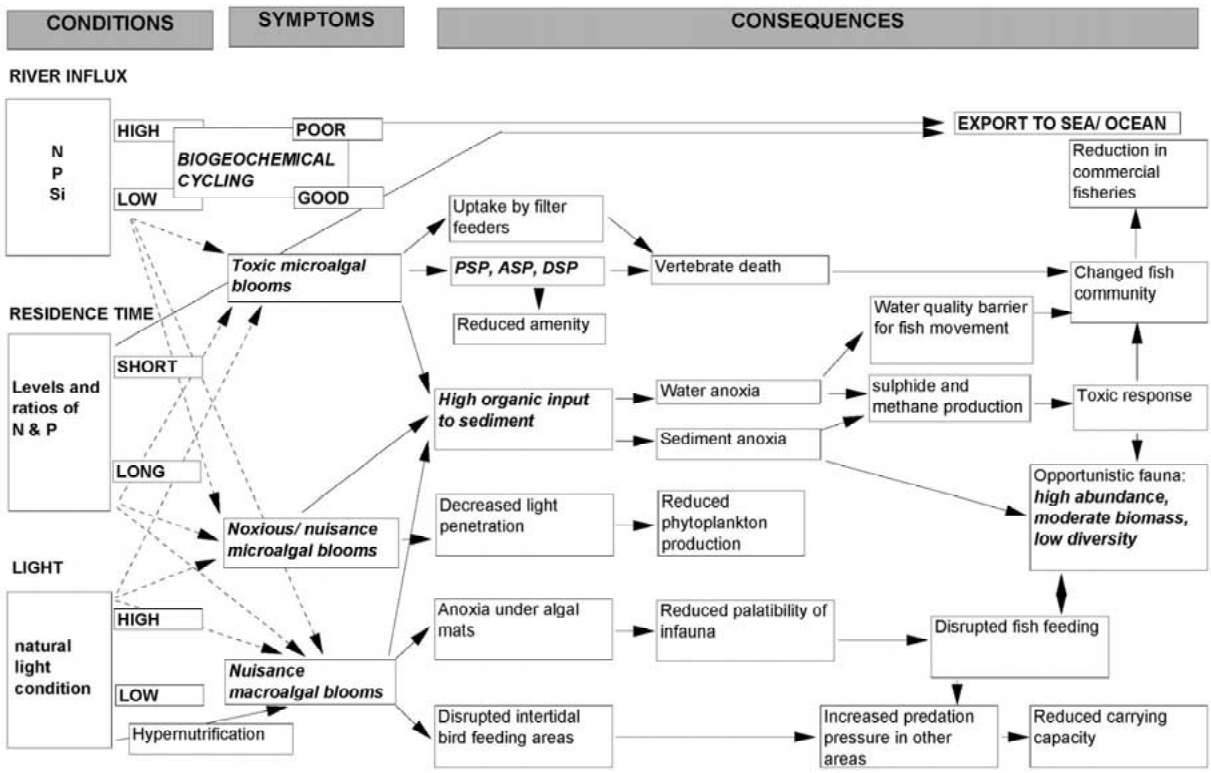

Figure 16. Overview of potential effects of nutrient enrichment in combination with turbidity conditions and residence time of water in an estuarine area.

termined. Also the role of dissolved reactive silicate in preventing the development of harmful blooms of flagellates should be investigated.

Finally, the effects of high N/P ratios should further be assessed.

Conclusions

Man has a choice of deciding how to treat natural resources, of acknowledging that organic waste (including nutrients) will continue to be produced and

that there may be decisions of whether to treat it through techniques or allow natural systems the sea included to degrade it. In essence, the biological degradation occurs whether in a treatment works or in aquatic systems.

What has to be safeguarded, however, is the integrity and sustainability of the natural system undertaking that degradation. The examples here show the very great spatial and temporal changes that may occur, under the influence of our own activities and focusing on compounds and elements that are part of the system. Unlike many other pollutants, nutrients are required for the well being of the system which has a natural ability to degrade, disperse and accumulate them. Only when this natural assimilative capacity is exceeded will problems occur.

Atmospheric input, river run off of point sources and diffuse sources will further increase the concentrations of nutrients in sea water on a global scale and while past 'hot spots' have centered on point sources, which are relatively easy to control, future controls will have to concentrate on diffuse terrestrial and aerial inputs.

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