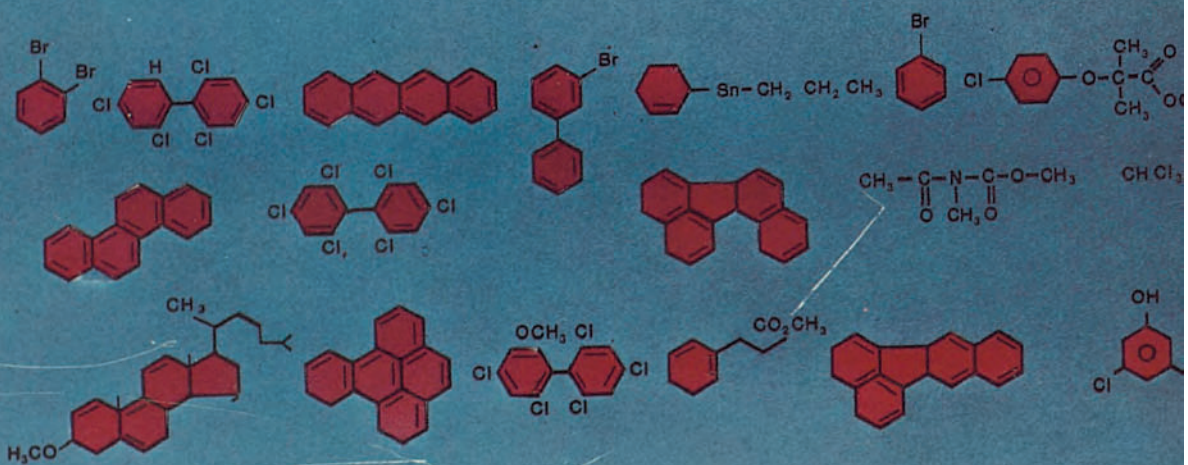
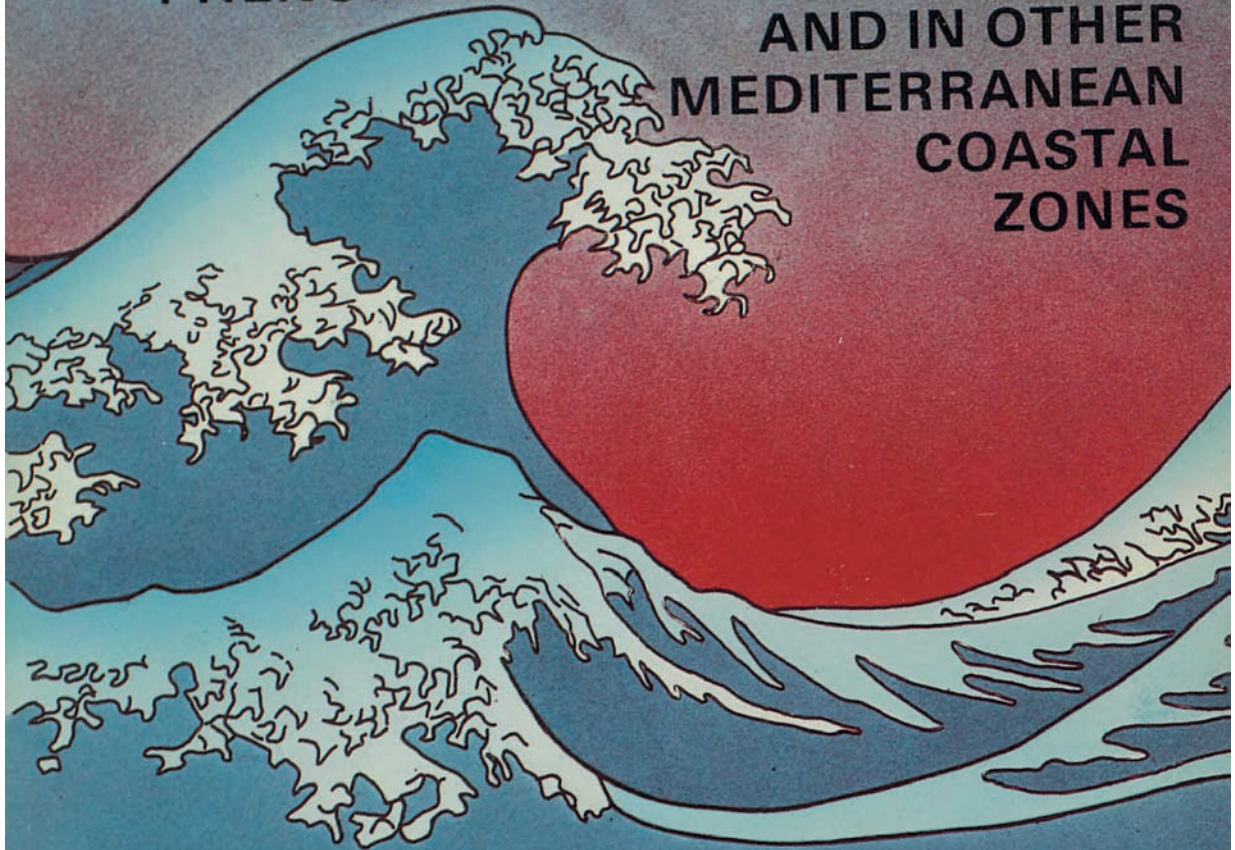


Commission of the European Communities

Water Pollution Research Reports

EUTROPHICATION-RELATED PHENOMENA IN THE ADRIATIC SEA AND IN OTHER MEDITERRANEAN COASTAL ZONES



EUTROPHICATION ALONG THE FRENCH COASTS

A. MÉNESGUEN
IFREMER - Centre de Brest
B.P. 70, 29280 Plouzané, France

Summary

*The French coasts exhibit two kinds of eutrophication; the proliferation of green macroalgae (*Ulva* sp.) affects the very shallow Mediterranean lagoons as well as a lot of widely open beaches in Brittany, whereas phytoplanktonic mass blooms are to be found in front of the large estuaries and in some Atlantic embayments. Both these types of eutrophication seem to have increased during the last twenty years, probably in relation with the increase of the nutrients loadings from agriculture and sewage. However, the severity of eutrophication is not only related to the absolute amount of loadings. The local residence time of the waters, deduced from hydrodynamics, is a crucial parameter; numerical models have highlighted the role of tidal residual circulation. Reduction of eutrophication requires a precise knowledge of the in-situ limiting nutrient. If green macroalgae are clearly nitrogen limited, phytoplankton may be either phosphorus or nitrogen limited. Ecological numerical models may help to quantify the effects of various strategies of lowering the loadings of nutrients.*

1. AN OVERVIEW OF THE FRENCH COASTAL EUTROPHICATION PROBLEM

Since the seventies, various and gradually more numerous problems of coastal eutrophication have been reported along the French coasts. During summer 1978, a red tide of *Gonyaulax polygramma* and *G. spinifera* (Lassus *et al.*, 1980) invaded the Bay of Seine and was the first example of an exceptional phytoplankton bloom reported by the French newspapers. Simultaneously, a second type of coastal eutrophication began to cause severe problems for the recreational use of some sandy beaches along the coasts of Brittany.

During spring, rapid vegetative multiplication of free floating Ulvae produces a huge algal biomass which accumulates and decays during summer, causing unpleasant odours. Therefore, at least two kinds of coastal blooms can be described: the "macroalgal bloom" and the "phytoplanktonic bloom".

1.1. The proliferations of green macroalgae

1.1.1. Spatial distribution

Figure 1 presents the volumes of green macroalgae collected during spring and summer by the local administrations (CEVA 1989). Despite the discrepancy between the real biomass production and the collected samples, two kinds of sites seem to be involved in macroalgae mass blooms. The first one is the classical lagoon site, with high residence times due to the static containment by the borders of the lagoon. This is the case for the Mediterranean coastal lagoons, especially near Montpellier - Palavas (Languedoc), and for the Arcachon inlet, along the Atlantic coast. Whereas the latter is primarily invaded by green macroalgae of genus *Enteromorpha* (Ribes, 1988), the former are subject to intense proliferation of *Ulva sp.* More surprising is the second type of eutrophicated sites. Situated along the macrotidal coast of Brittany, a lot of bays exhibit huge accumulations of *Ulva sp.* especially on sandy beaches (Piriou, 1986). These accumulations look paradoxical in such open sites, with important tidal range and strong eddy diffusion. The seasonal growth pattern of these green macroalgae is similar in open or semi-enclosed sites. Starting in April, owing to the increasing available light, the growth is very rapid and stops in June, when self shading and nutrient depletion create strong limitation (fig. 2). The summer phase depends on the type of site. Whereas the tidal mixing ensures continuous reaeration of the water within open sites, the sedimentation of large quantities of algae: 3-4 kg.wet weight.m⁻² (Pierre, 1990) induces a sudden decay of biomass in Mediterranean lagoons, with frequent anoxia and sulphur cycle triggering. Reducing conditions near the bottom allow massive release of phosphorus from the muddy sediment of these lagoons. Along with the large amounts of remineralized nitrogen, these nutrient pulses can induce subsequent phytoplanktonic blooms. The autumn brings favourable growth conditions again, but the autumnal *Ulva* bloom is never so marked as the spring one, probably because of the senescent status of the algae. Normal winter conditions cause fragmentation and decaying of the thalli.

1.1.2. Pluriannual trend

A reliable appraisal of the pluriannual trend of macroalgal blooms cannot be drawn today from the existing data. Only five evaluations of the total biomass lying in the southern part of the Bay of St-Brieuc have been conducted since 1986 by IFREMER/CEVA (Piriou, 1989) using a stratified sampling technique coupled with aerial photogrammetry. Therefore, the only available data on a pluriannual basis are the volumes of stranded algae collected at tourist sites by local authorities. On the Brittany coast, these data (CEVA, 1989) show an apparent increase until 1984, with a subsequent stabilization (fig. 3). 1988 shows an increase again, corresponding to a cleaning cost of 2.5 million francs.

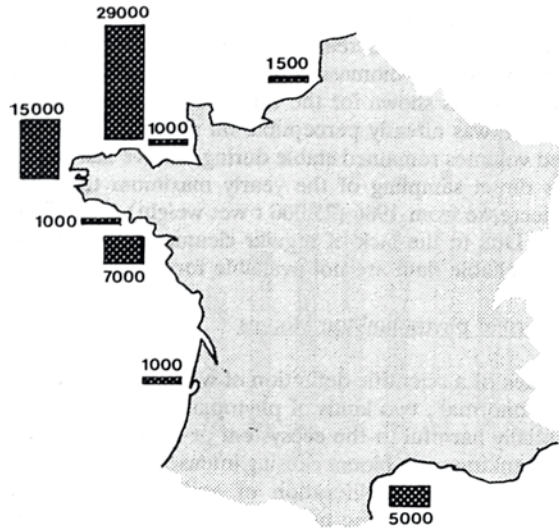


Fig. 1 :Collected volumes (m³) of green macroalgae during 1987. (CEVA 1989).

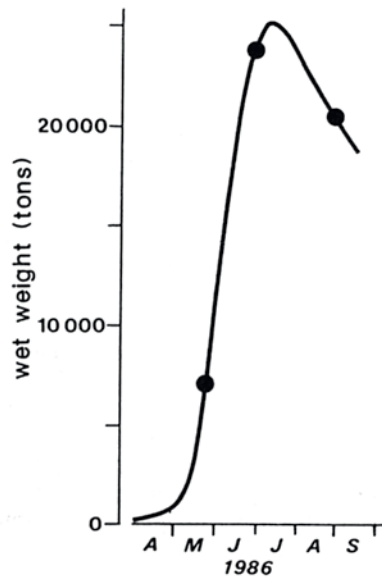


Fig. 2 :Temporal evolution of the total Ulva biomass on the beaches of the southern bay of St. Brieuç (Brittany, France) during 1986. (Piriou 1989).

The increasing impact on the recreational use of many sandy beaches undoubtedly reveals an increase in the eutrophication process, but no realistic trend about the real biomass production can be drawn from this "touristic point of view". As shown for the Bay of St-Brieuc (Piriou *et al.*, 1989), *Ulva* proliferation was already perceptible on aerial photographs from 1952. The collected volumes remained stable during the five last years (about 6,000 m³), whereas direct sampling of the yearly maximum total biomass indicates a strong decrease from 1986 (25,000 t wet weight) to 1988 (11,450 t) and 1989 (5,400 t). Due to the lack of regular cleaning up in the lagoons (Arcachon, Palavas), reliable data are not available for those sites.

1.2. The abnormal phytoplankton blooms

In spite of the lack of a scientific definition of when a phytoplankton bloom may be considered as "abnormal", two kinds of phytoplankton events will be reported here as being potentially harmful to the ecosystem or to the human consumer. The first is due to phytoplankton mass bloom causing intense coloration of the sea, the second one being related to the proliferation of toxic species. There is of course a considerable overlap between the two groups. Some toxic species (*Dinophysis sp.*) can be harmful at very low concentrations.

1.2.1. Spatial distribution

Figure 4 from Belin *et al.* (1989) shows the location of the sites of reported discoloured waters from 1975 to 1988. The main occurrences are to be found in large estuaries (Loire, Seine), in semi-enclosed basins (Berre, Arcachon, Vilaine, Douarnenez) in harbours or along water mass discontinuities (Celtic Sea).

1.2.2. Pluriannual trend

As for green macroalgae, a strong bias may alter conclusions drawn from the numbers of reported discoloured waters. Are these blooms really more numerous today than twenty years ago, or are we only getting the results of a better monitoring strategy?

Despite this uncertainty, some interesting points can be noted in figure 5. Apart from 1978, the exceptional year, a global increase can be observed until 1984, followed by a weak decrease. The increase in the discolouration of waters is mainly due to the proliferation of dinoflagellates (*Noctiluca scintillans*, *Prorocentrum micans*, *Prorocentrum minimum*, *Gonyaulax sp.*, *Gyrodinium aureolum*, *Gymnodinium sp.*), which corroborates the observed shift from diatom to dinoflagellate dominance during the last twenty years in the North Sea (Radach and Berg, 1986).

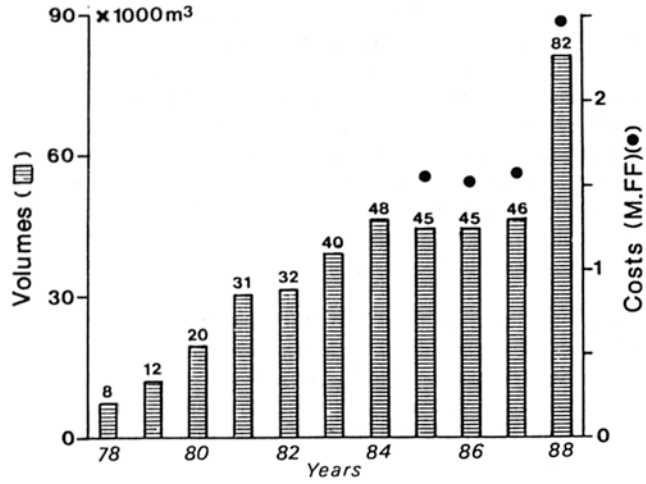


Fig. 3: Recent evolution of yearly collected volumes of Ulvae in Brittany with their costs. (CEVA 1989).

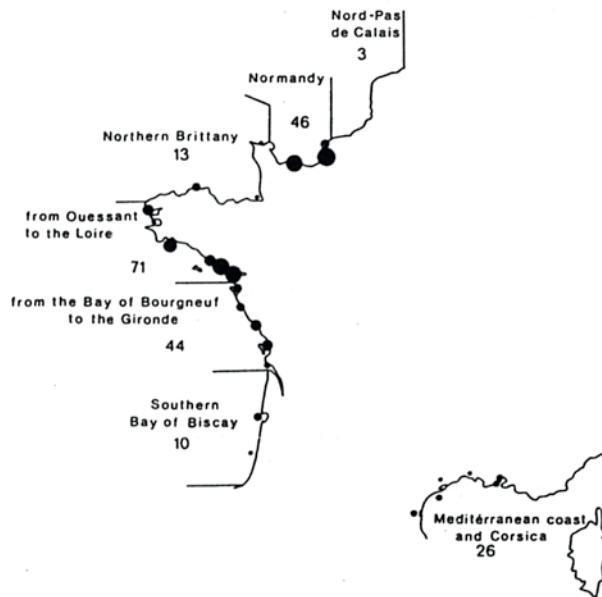


Fig. 4: Cumulated occurrences of discoloured waters from 1975 till 1988. (Belin et al. 1989).

As far as the three "French" toxic species are concerned, a slightly different pattern is observed (fig. 6). Whereas reported cases of Gyrodinium blooms seem to be stable during the last fifteen years, the counts of Dinophysis blooms dramatically increase during the eighties. This is a typical case of bias introduced by the sampling procedure. In 1983 only, numerous cases of human diarrhoea shellfish poisoning (DSP) were ascertained to be caused by Dinophysis sp. and a reliable monitoring was only recently introduced. The PSP species Alexandrium minutum was detected for the first time in 1988. However, it is probably too early to determine any reliable trend for the occurrences of toxic blooms along the French coasts.

Apart from direct toxicity, phytoplankton mass blooms may produce indirect toxicity by causing anoxia of the bottom waters. As opposed to the Skagerrak or the Baltic Sea (Rosenberg, 1985), the French coast does not exhibit serious recurrent problems of anoxia. Apart from the fish kill in Vilaine Bay in July 1982 (Merceron, 1987), no catastrophic event due to phytoplankton-related anoxia has been reported. However, as measurements of oxygen concentrations showed during 1984 in Vilaine Bay (Merceron 1987), recurrent hypoxia can affect the bottom layer every year without any recorded lethal effects (fig. 7). Long term effects of such recurrent anoxia are however well established (Rosenberg, 1985).

2. TWO IMPORTANT CAUSES OF COASTAL EUTROPHICATION

Vollenweider's determination (1968) of the status of a lake from its residence time and its phosphorus loading has emphasized the crucial roles of the hydrodynamics and the nutrient inputs. Some conclusions can be drawn from studies in coastal areas in France on both these controlling factors of eutrophication in coastal waters.

2.1. The role of hydrodynamics

As pointed out previously, eutrophication problems in Brittany look somewhat puzzling because they occur in bays which are exposed to the open sea and subject to a high tidal range (between 4 and 8 m). Refined hydrodynamical studies are required to find the key of this paradox, and numerical models have proved to be very effective in estimating residual currents, which are hardly measurable in-situ.

The first example concerns the Bay of St-Brieuc, the southern part of which exhibits an important Ulva bloom. If particles are released at low tide at various places, their simulated drift during 12h25' (i.e. to the next low tide) will bring them rather far from their departure location, except in some places, especially in the southeastern part of the bay (fig. 8). In other words, the tidal residual drift depends on the location in the bay. The barycentric presentation of these residual velocities (Orbi and Salomon 1988) allows the calculation of the synthetic map of lagrangian residual currents presented in figure 9 (Ménèsguen and Salomon 1988). Marine regions with invisible vectors, in the southern and southeastern parts, are characterized by

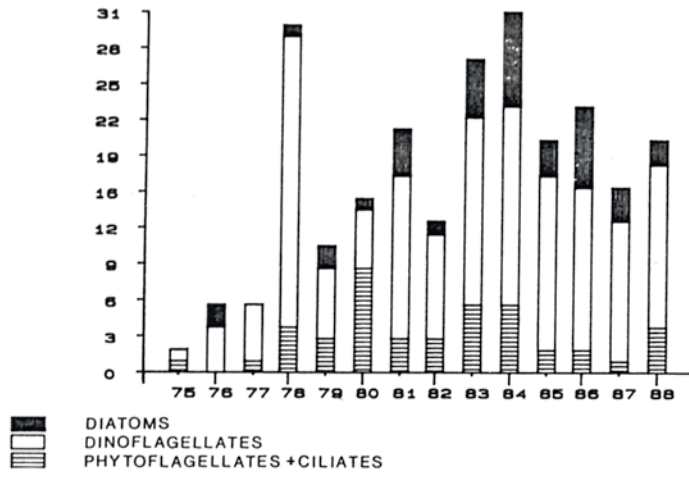


Fig. 5: Evolution of yearly occurrences of discoloured waters. (Belin et al. 1989).

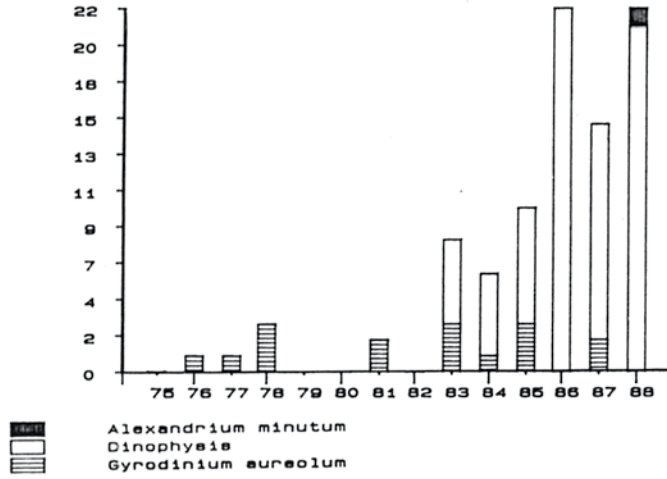


Fig. 6: Evolution of yearly occurrences of toxic blooms. (Belin et al. 1989).

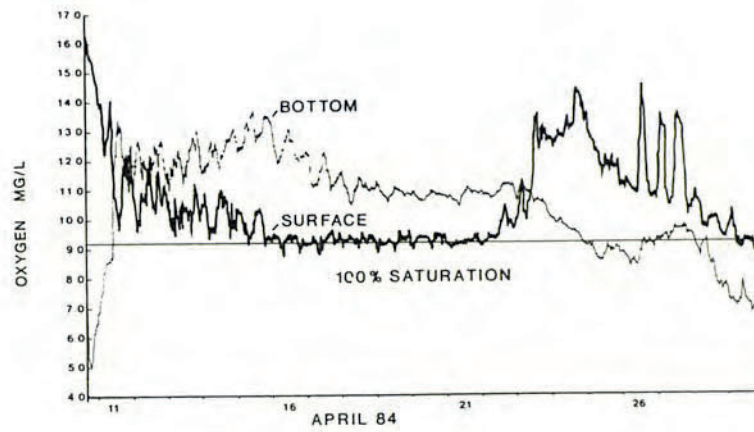


Fig. 7: Continuous records of dissolved oxygen in surface and bottom waters in the Bay of Vilaine during April 1984. (Merceron 1987).

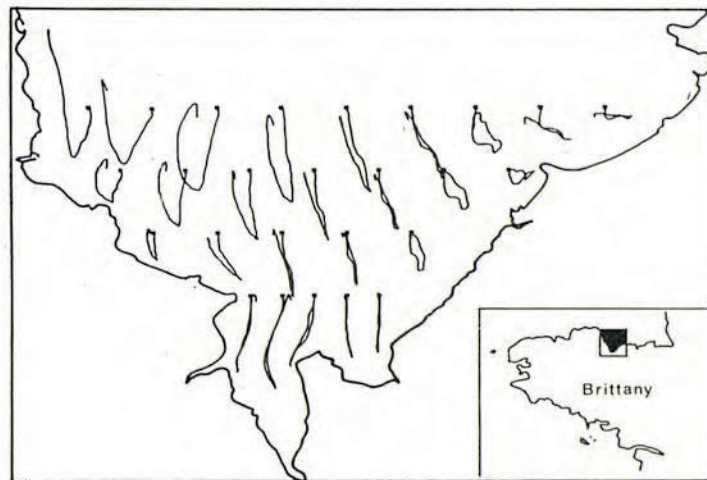


Fig. 8: Simulated drift of particles from one low tide to the next one in the southern Bay of St. Brieuc.

vanishing residual drift, i.e. without any lateral washing out from one tidal period to the next. The water body oscillates quasi-purely on itself. The locations of stranded and suspended algae at low tide correspond to shallow areas without tidal residual drift.

The second example corroborates this but for phytoplanktonic eutrophication this time. Even on semi-open sites, the interaction of tide with the local geomorphology can produce dynamical trapping of water bodies. Derived from a 3D-hydrodynamical model (Salomon and Lazure 1988), the computed tidal residual drift on the coastal zone from the Loire estuary up to the Gulf of Morbihan (fig. 10) clearly shows a large area of weak residual currents in the Vilaine Bay, corresponding exactly to the location of intense phytoplanktonic blooms, leading to recurrent hypoxia and to the catastrophic episode of July 1982.

This last anoxia was due to the conjunction of abnormal enriched freshwater discharge by the Vilaine river with calm and sunny weather. Due to the lack of residual flow out of the bay, a stable stratification occurred and prevented significant reaeration of the bottom layer. Concurrently, the enrichment triggered an intense phytoplankton development, quickly inducing a total self-shading in the bottom layer. Respiration along with decaying of phytoplankton biomass completely depleted bottom oxygen within a few days (Merceron 1987).

2.2. The role of nutrients (nitrogen and phosphorus)

As opposed to fresh waters, where phosphorus has largely been considered as the limiting nutrient, marine waters are often claimed to be nitrogen-limited (Ryther and Dunstan 1971). Unfortunately, the status of the coastal waters seems not to be so clearcut. The interface between the inland waters and the high sea can exhibit various sensitivities to nutrients, depending on the dispersive capacities of the coastal area relative to the terrestrial inputs and the buffering capacity of the sediments. As an illustration of the complexity of determining the respective role of nitrogen and phosphorus, two contrasting cases will be presented here.

In the case of Ulvae mass blooms, no direct positive correlation can be found between the annual nitrate loadings in the bays along the coast of Brittany (fig. 11 a) and the yearly collected volumes of algae (fig. 11 b). As explained previously, not only the absolute flux of nutrients entering the coastal zone has to be taken into account, but also the residence time of this enrichment in the coastal ecosystem. This is the reason why embayments as the Bay of St-Brieuc, with six times lower inputs than the Bay of Brest, can produce at least ten times more biomass of Ulvae. This example illustrates the false result which can be obtained (i.e. Ulva blooms are not related to nitrogenous loadings) when a correlation is sought between the phenomenon and only one of its causative factors, neglecting another of major importance. In mathematical terms, this is related to the well-known difference between absolute and conditional correlations. As a matter of fact, the growth of green macroalgae is strongly dependant on nitrogen availability, as early mentioned by Letts and Richards in 1911 and duly verified in-situ (see next section).

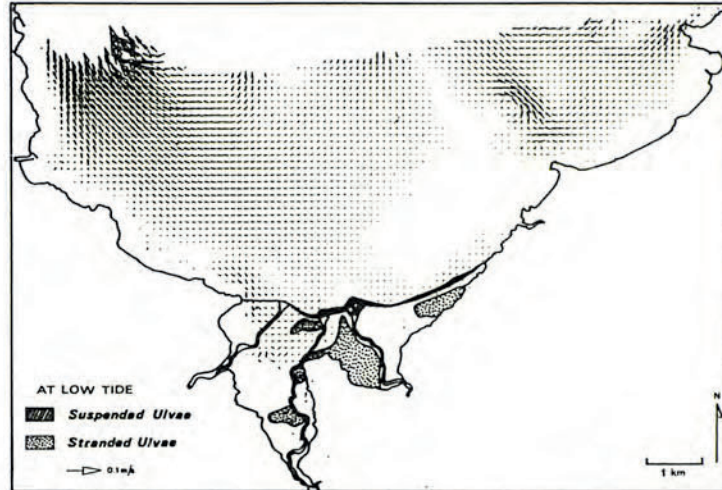


Fig. 9: Map of computed lagrangian residual currents in the southern Bay of St. Brieuc with location of *Ulva* accumulations. (Ménèsquen and Salomon 1988).

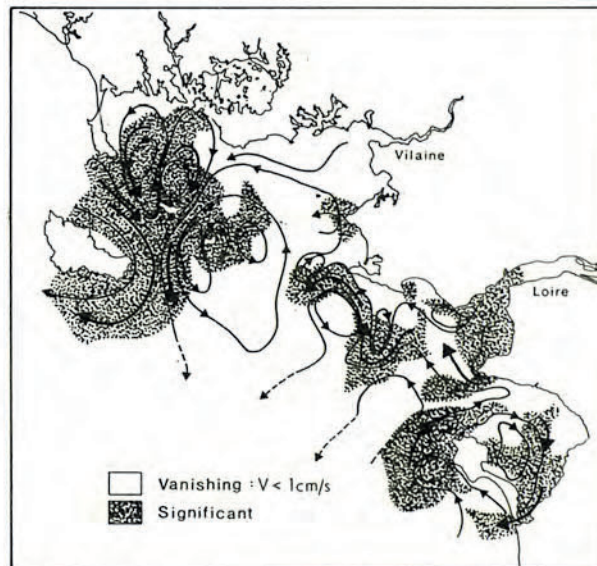


Fig. 10: Computed tidal residual drift in the Loire-Vilaine coastal zone. (Salomon and Lazure 1988).

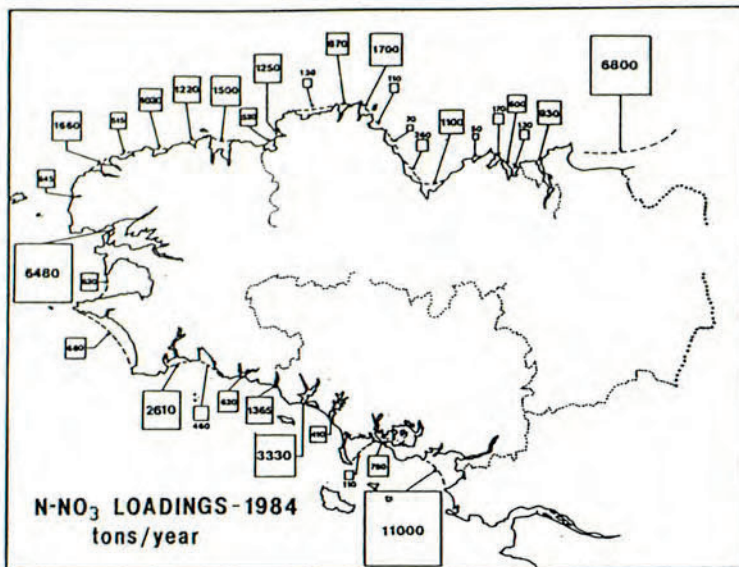


Fig. 11.a : Gross nitrate loadings along the coasts of Brittany during 1984. (Piriou 1986).

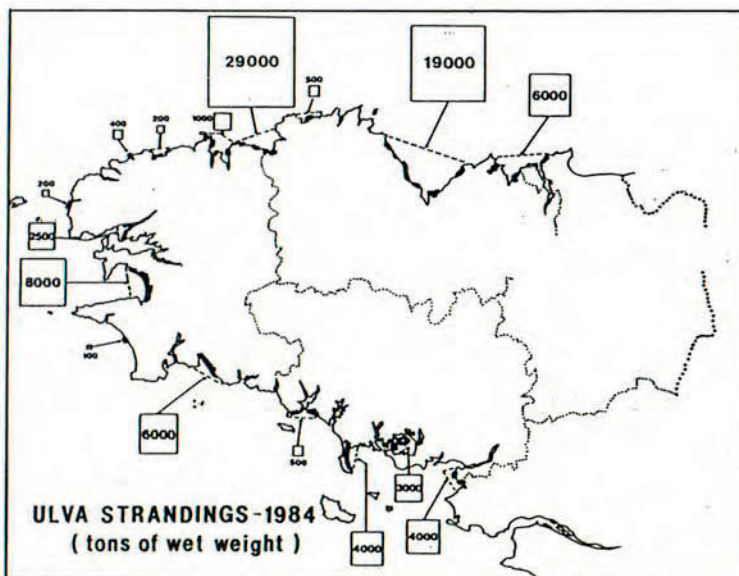


Fig. 11.b : Collected volumes of Ulvae along the coasts of Brittany during 1984. (Piriou 1986).

The case of phytoplanktonic eutrophication in the Bay of Vilaine provides roughly speaking the opposite situation. The quantities of nitrogenous fertilizers used over the drainage basin of the Vilaine river (fig. 12), as well as the estimated annual nitrogen loadings in the estuary (fig. 13 a) show a significant increasing trend during the seventies. Phosphorus loadings (fig. 13 b) vary more randomly. Growing eutrophication problems in the bay seem therefore to be related to the increasing nitrogen inputs, whereas bioassays revealed that phosphorus was the effective limiting factor during early summer (see next section).

No reliable a priori relationship can be found nowadays between the total loading of a nutrient and the risk of eutrophication in coastal waters. Refinement of the Vollenweider's diagram for lakes is required for coastal waters, taking into account the shift from phosphorus to nitrogen limitation and the bifurcation between macroalgal and phytoplanktonic production.

3. SCIENTIFIC REMEDIES

While a lot of mechanisms related to coastal eutrophication are qualitatively described, they are not yet quantitatively understood. For example, why does one particular species suddenly overcome the competing species, what is the fate of benthic nutrients?

Nevertheless, some tools are available which can help from now in the development of remedial strategies. The first practical question to be answered must be: "What is the limiting nutrient?", and the second: "To what extent is it useful to reduce the loading of this limiting nutrient?"

3.1. The detection of the limiting nutrient

Since the work of Droop (1968) and Caperon (1968), evidence has grown up that algae (macroalgae as well as phytoplankton) can store nutrients and that their growth rate is related to the level of their internal storage, not to the instantaneous concentration of nutrients in the ambient water. The limiting nutrient at a precise moment will therefore be the one which is most depleted in the algal tissue. For instance, several annual cycles of measurement of nitrogen content of *Ulva* in the Bay of St-Brieuc (Dion 1988) reveal a recurrent intense nitrogen depletion beginning in May and ending in autumn (fig. 14 a), whereas the phosphorus content remains at its upper level all the year long (fig. 14 b). Nitrogen has therefore been considered to be the limiting factor in *Ulva* mass blooms along the coast of Brittany. This is in agreement with the results of Sfriso *et al.* (1989) in a completely different ecosystem, the Lagoon of Venice.

Apart from the seasonal evolution of cell quotas, the popular technique of bioassays has been successfully employed for detecting the limiting factor of phytoplankton growth. Queguiner (1988) showed that without addition of phosphorus, any

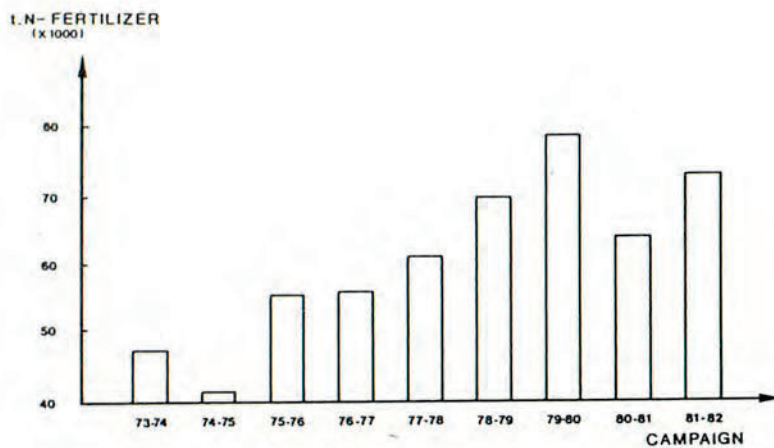


Fig. 12 : Evolution of the consumption of nitrogenous fertilizers in the drainage basin of the river Vilaine. (in Merceron 1987).

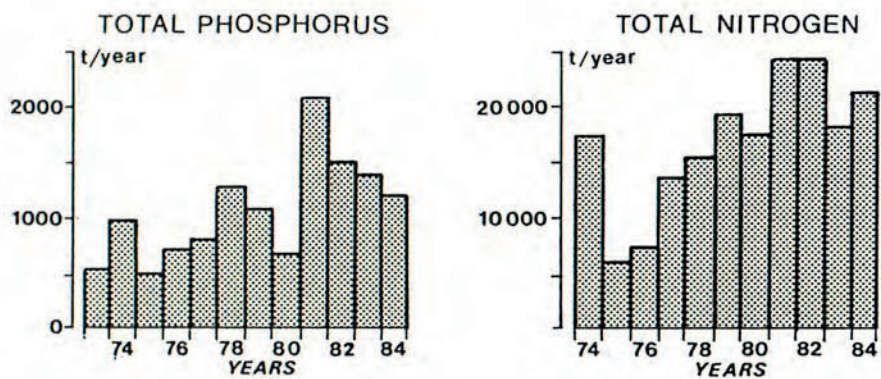


Fig. 13 : Evolution of the loadings from the Vilaine estuary. 13.a : total phosphorus, 13.b : total nitrogen. (Queguiner 1988).

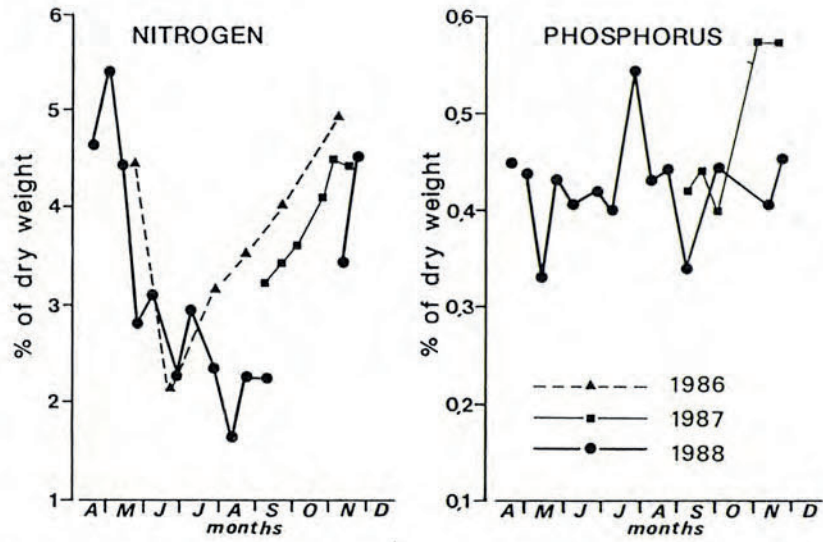


Fig. 14 : Seasonal evolution of the chemical composition of Ulva in the southern Bay of St-Brieuc. 14.a : nitrogen content, 14.b : phosphorus content. (Dion 1988).

enrichment of water from the Bay of Vilaine would be unsuccessful in enhancing primary production during spring and summer (fig. 15). Phosphorus was therefore considered as the limiting factor in that embayment during the critical season.

3.2. Forecasting of the effects of remedial strategies

The highly dynamic, frequently nonlinear behaviour of the coastal ecosystems is not in favour of straightforward evaluations of the response to various clean-up strategies. Both physical and biological time-scales have to be considered together, and only numerical models are able to cope with such different processes.

It must be stated again that no good model can be built without a sound knowledge about the processes, and that the best model is not necessarily the biggest, but the most comprehensive one. For instance, no specific modelling is required for non-limiting elements, whereas all the main processes affecting the budget of the limiting element are necessary. As an illustration of this operational point of view of ecological modelling, the structure of a simple model of *Ulva* mass blooms (Ménèsquen, 1990) on open, tidal sites is presented on figure 16 a (biological submodel) and figure 16 b (hydrodynamical box-model). The aim of this work was to evaluate the respective effect of nitrogen and phosphorus on the macroalgal proliferation.

Owing to the massive dominance of *Ulvae* in the shallow waters under study, no phytoplankton, nor herbivorous predators were taken into account, because of their negligible importance in the surf zone. The growth of algae was related to temperature, light, and internal nutrient status. The global ecological model, calibrated on data from the Bay of St-Brieuc, was then used to produce simulations with different reducing ratios of the N- and P-loadings. Clearly, the maximum *Ulva* biomass is controlled first by N-loading (fig. 17). Moreover, P-loading becomes totally neutral if N-loading is reduced to one third of its actual value. In spite of incompleteness of this model (the transient buffering of nutrient by the beach has not been taken into account), recommendations for N- rather than P- regulation in the loadings can be made. Extrapolation to other kinds of ecosystems (lagoons) must be done with caution, and of course no case for phytoplanktonic eutrophication control can be made from this macroalgal model.

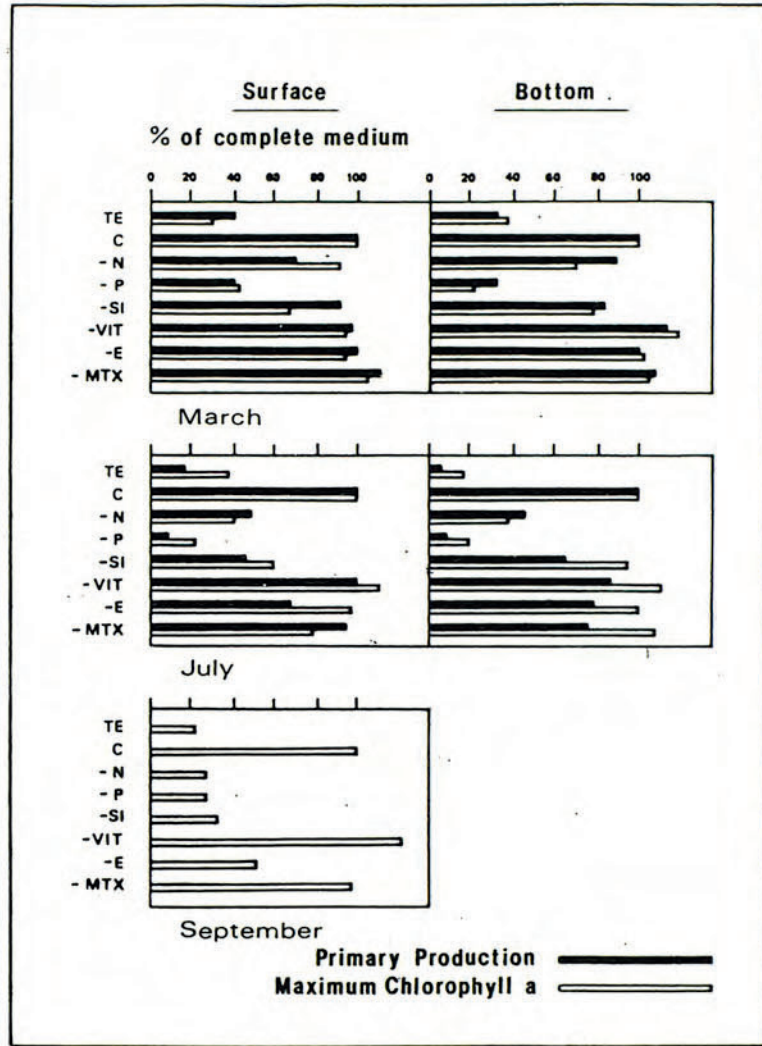


Fig. 15 : Bioassays in the Bay of Vilaine. (Queguiner 1988).

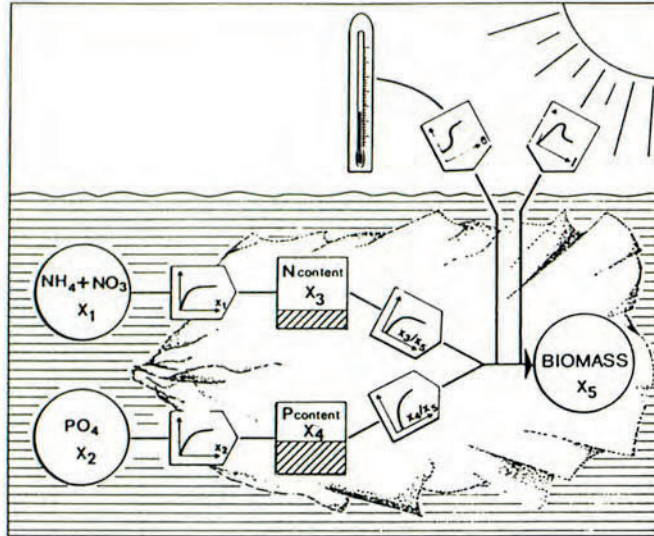


Fig. 16 : Flow diagram of the numerical model of Ulva proliferation. 16.a : biological submodel.

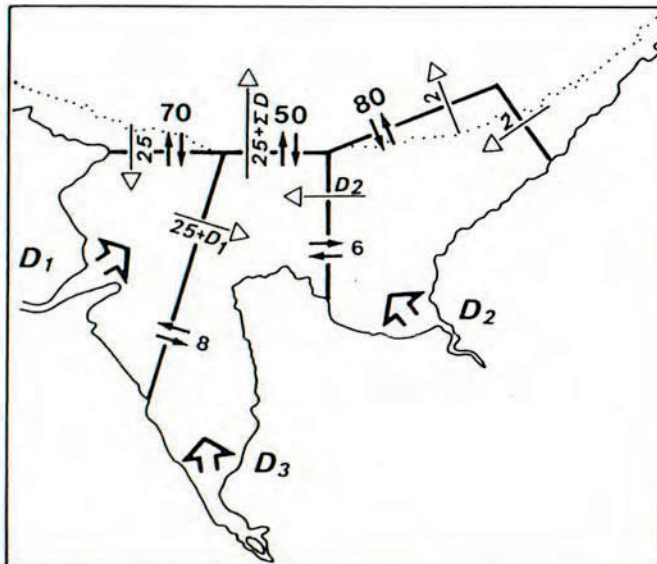


Fig. 16.b : Hydrodynamical submodel. (Menesguen 1990).

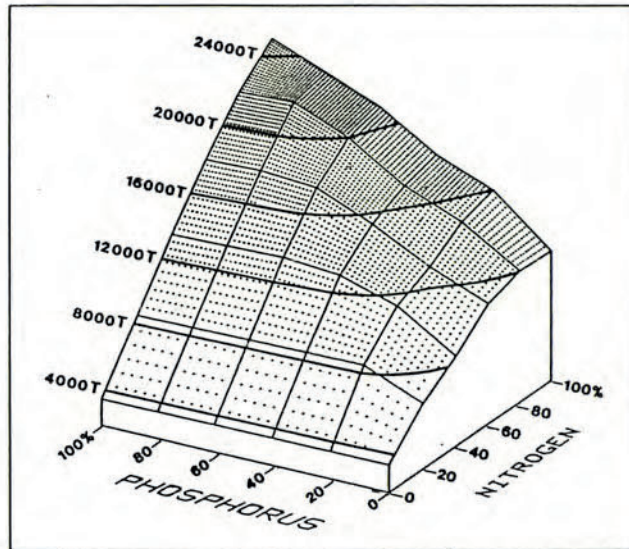


Fig. 17 : Computed response surface of the summer maximum biomass of *Ulva* to nitrogen and/or phosphorus loading reductions in the southern Bay of St-Brieuc. (Ménèsquen 1990).

BIBLIOGRAPHY

- BELIN, C., BERTHOME, J.P. and LASSUS, P. (1989). Dinoflagellés toxiques et phénomènes d'eaux colorées sur les côtes françaises: évolution et tendances entre 1975 et 1988. *Hydroécol. Appl.* 1/2, 3-17.
- CAPERON, J. (1968). Population growth response of *Isochrysis galbana* to nitrate variation at limiting concentrations. *Ecology* 49, 866-872.
- CEVA (1989). Synthèse des enquêtes sur les échouages d'algues en Bretagne depuis 1978.
- DION, P. (1988). Etude de certains aspects écophysologiques des marées vertes. Report CEVA/R37 (IFREMER's grant Nr. 87 2 430 443 DERO/EL).
- DROOP, M.R. (1968). Vitamin B12 and marine ecology. IV. The kinetics of uptake, growth and inhibition in *Monochrysis lutheri*. *J. Mar. Biol. Assoc. U.K.* 48, 689-733.
- LASSUS, P., MAGGI, P. and BESSINETON, C. (1980). Les phénomènes d'eaux colorées de la baie de Seine en 1978. *Science et Pêche, Bull. Inst. Pêches Marit.* 298, 1-28.
- LETTS, E.A. and RICHARDS, E.M. (1911). Report on green seaweeds and especially *Ulva latissima* in relation to the pollution of the waters in which they occur. Royal Commission on Sewage Disposal. 7th Report - HMSO - London, Appendix III, Section II.
- MENESGUEN, A. (1990). La modélisation des "marées vertes" littorales et ses applications. *La Houille Blanche* 3/4, 1990, 237-242.
- MENESGUEN, A. and SALOMON, J.C. (1988). Eutrophication modelling as a tool for fighting against *Ulva* coastal mass blooms. in: *Computer Modelling in Ocean Engineering*, Schrefler B.A. & O.C. Zienkiewicz, eds., Proc. Intern. Conf. held in Venice, 19-23 sept. 1988, Balkema, Rotterdam, p. 443-450.
- MERCERON, M. (1987). Mortalités de poissons en Baie de Vilaine (juillet 1982). Causes, mécanismes, propositions d'action. Report IFREMER/DERO-87. 14-EL, 99 p.
- ORBI, A. and SALOMON, J.C. (1988). Dynamique de marée dans le golfe normand-breton. *Oceanol. Acta* 11 (1), 55-64.
- PIERRE, D. (1990). La prolifération des algues dans les étangs palavasiens (Hérault). Description et moyens de récolte envisageables. Note CEMAGREF/Aménagements littoraux et aquaculture, Montpellier, 4 p.

- PIRIOU, J.Y. (1986). Les marées vertes sur le littoral breton. Bilan 1985. Report IFREMER/DERO-86. 29-EL, 79 p.
- PIRIOU, J.Y. (1989). Biomasses d'ulves en Baie de Saint-Brieuc. Report IFREMER/DERO-89. 26-EL, 21 p.
- PIRIOU, J.Y., MENESGUEN, A. and SALOMON, J.C. (1989). Les marées vertes à ulves: conditions nécessaires, évolution et comparaison de sites. Paper presented at the 19th ESCA Symp. on "Milieux estuariens et littoraux", Caen (France), sept. 1989, 13 p.
- QUEGUINER, B. (1988). Synthèse opérationnelle du programme pluriannuel 1983-1988 de la Commission Quadripartite chargée de l'Aménagement Halieutique et de la Protection Hydrobiologique de la Baie de Quiberon-Vilaine. Report University of Western Brittany, Brest, 34 p.
- RADACH, G. and BERG, J. (1986). Trends in den Konzentrationen der Nährstoffe and des Phytoplanktons in der Helgoländer Bucht (Helgoland Reede Daten). Ber. Biol. Anst. Helgol., n° 2, 63 p.
- RIBES, E. (1988). Contribution à l'étude de la prolifération des algues vertes dans le bassin d'Arcachon. Report of the Institute of Marine Biology, Arcachon (France), 31 p.
- ROSENBERG, R. (1985). Eutrophication - the future marine coastal nuisance? Mar. Poll. Bull. 16 (6), 227-231.
- RYTHER, J.H. and DUNSTAN, W.M. (1971). Nitrogen, phosphorus and eutrophication in the coastal marine environment. Science 171, 1008-1013.
- SALOMON, J.C. and LAZURE, P. (1988). Etude par modèle mathématique de quelques aspects de la circulation marine entre Quiberon et Noirmoutier. Report IFREMER/DERO-88. 26-EL, 104 p.
- SFRISO, A., PAVONI, B. and MARCOMINI, A. (1989). Macroalgae and phytoplankton standing crops in the central Venice Lagoon: primary production and nutrient balance. The Science of the Total Environment 80, 139-159.
- VOLLENWEIDER, R.A. (1968). The scientific basis of lake and stream eutrophication, with particular reference to phosphorus and nitrogen as factors in eutrophication. Tech. Report to OCDE, Paris, DAS/CSI/68-27, 182 p.

EUTROPHICATION-RELATED PHENOMENA IN THE ADRIATIC SEA
AND YUGOSLAV COASTAL REGION

D. DEGOBBIS

"Ruder Boskovic" Institute, Center for Marine Research
52210 Rovinj, Yugoslavia

Summary

Environmental problems in the Yugoslav Adriatic coastal region have arisen from: 1) local anthropogenic nutrient inputs and 2) eutrophication processes at a regional level (in the northern Adriatic). Dinoflagellate blooms ("red tides") occurred during summer in the Pula (Istria peninsula) and Split harbors (central Dalmatian coast), but to date have been non-toxic. In some other embayments (e.g., Rijeka Bay, Kastela Bay, Sibenik Bay) increased phytoplankton biomass and primary production can also be related to urbanization and human activities. Undesirable eutrophication-related phenomena (e.g. near-anoxic conditions in the bottom layer in 1977, 1988, and 1989) occurred in the Istrian coastal area primarily as a consequence of eutrophication processes controlled by Po River nutrient inputs and water dynamics in the open northern Adriatic. Local eutrophication can be eliminated by disposing wastewaters into the open coastal sea at an appropriate depth using long submarine outfalls. To date disposal facilities have been completed only in a few urban centers (e.g., Piran, Rabac, Hvar, Dubrovnik). Region wide eutrophication can only be controlled by moderating the anthropogenic nutrient load in the Po River watershed.

1. INTRODUCTION

Ecologically speaking, and in the broadest sense, eutrophication means a substantially increased biomass beyond the prevailing conditions in a given ecosystem, due to an unusually rich supply of nutrients in the euphotic layer. Unpolluted river discharges and upwelling induce natural eutrophication in marine environment. Sewage and manure discharges, as well as land drainage from agriculture areas are responsible for anthropogenic (man-made) eutrophication. Since the physiological process is the same, it is difficult in some cases to estimate the relative contributions of different mechanisms (e.g., natural vs. anthropogenic nutrient discharges of a polluted river). However, while naturally slower eutrophication processes generally allow the ecosystem to adjust to increased trophic conditions, rapid anthropogenic influences often led to ecosystem disequilibria and to stressed environments with possible harm to living resources.

Periodic increased "ingressions" of Levantine intermediate waters from the Ionian Sea, approximately each 9 years (1), represented a mechanism of natural eutrophication of central and southern Adriatic regions.

During such an event the nutrient inputs to the Adriatic Sea were approximately as large as the yearly contributions of the heavily polluted Po River, one of the major Mediterranean rivers, which discharges to the northern Adriatic Sea (2). Increased sardine catch (within 20%) followed "ingression" events with three year delays (necessary for fish growth). In the last decade more drastic changes were observed in the open central Adriatic waters. In these waters primary production rates increased from about $60 \text{ g m}^{-2} \text{ y}^{-1}$ of organic carbon in the period 1963-1980 to about $100 \text{ g m}^{-2} \text{ y}^{-1}$ in 1983-1985 (3,4). At the same time the sardine catch per unit effort increased from about 2500 to 5000 kg y^{-1} (3). It was assumed that these changes resulted from a combined influence of eutrophied northern Adriatic waters and natural eutrophication mechanisms.

Anthropogenic eutrophication was identified in several Adriatic and Mediterranean coastal regions (5). Along the Yugoslav Adriatic coast eutrophication is significant within the harbor area of larger urban or industrial centers, and apparently related to inadequate wastewater disposal. In addition, combined natural and anthropogenic eutrophication at a regional level in the northern Adriatic periodically affected the Yugoslav coastal ecosystem.

In this paper anthropogenic eutrophication-related phenomena observed in Yugoslav coastal waters are described, with some comments on their reduction or elimination.

2. LOCAL EUTROPHICATION PHENOMENA

Eutrophication problems in the Yugoslav coastal sea (Fig. 1) are less marked than along the western Adriatic coast of Italy. First, the Yugoslav coast is much less densely populated and industrialized (Tab. 1). Along the coast there are only 10 cities with populations exceeding 10,000, including two (Rijeka and Split) exceeding 200,000. Freshwater discharge rate to the eastern Adriatic (mainly due to diffused karstic groundwater sources) is several times smaller than along the western coast (Tab. 1).

TABLE 1. Data relevant to anthropogenic eutrophication along the Yugoslav and Italian Adriatic coasts.

PARAMETER	YUGOSLAV COAST	ITALIAN COAST
Shoreline length, including islands (km)	6116	1259
Resident population (10^6)	1.4	4.1
Inhabitants per km shoreline length	228	3257
Tourist overnights (10^6)	70	120
River and karstic groundwater discharge ($10^6 \text{ m}^3 \text{ y}^{-1}$)	22,000	73,000
Industrial wastewater discharge ($10^6 \text{ m}^3 \text{ y}^{-1}$)	500	1,500
Ship wastewater discharge ($10^6 \text{ m}^3 \text{ y}^{-1}$)	2	8

Modified from (6).

Second, although several times higher than in seawater, orthophosphate concentrations in karstic waters is generally much lower than in polluted alpine rivers on the western Adriatic coast (Tab. 2). The very high N/P ratio of karstic waters (Tab. 2) is unfavorable for phytoplankton growth. In some heavily polluted karstic streams (e.g., the Rječina River; Tab. 2) the orthophosphate content is greatly increased, and the N/P ratio reduced.

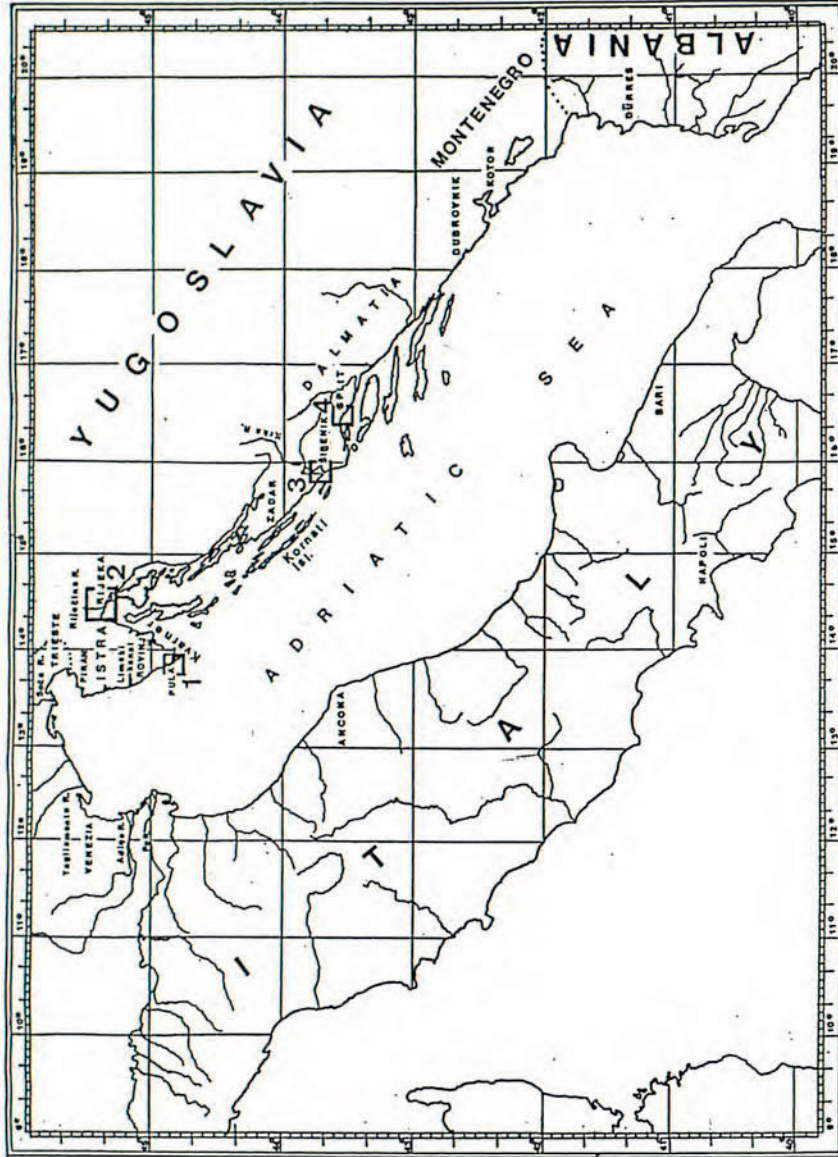


FIG. 1. The Adriatic Sea and some eutrophied areas along the Yugoslav coast: 1 - Pula harbor, 2 - Rijeka and Bakar bays, 3 - Sibenik Bay, 4 - Kastela Bay with Split harbor (Vranjic).

Third, the Yugoslav coastal sea can be much deeper than the Italian coastal sea, up to 100 m within a few hundreds meters from the shore. The water dynamics is significant, particularly in the channels between the shore and islands (11). Thus, dispersion of introduced nutrients is favored and their impact on the marine ecosystem reduced.

TABLE 2. Nitrate (NO₃), orthophosphate (PO₄) and orthosilicate (SiO₄) average concentrations (μmol dm⁻³), and total inorganic nitrogen (TIN)-orthophosphate ratio in seawater and freshwaters of the Adriatic area. Freshwater source mean flow rates(Q) in m³ s⁻¹.

	Q	c(NO ₃)	c(PO ₄)	c(SiO ₄)	$\frac{c(\text{TIN})}{c(\text{PO}_4)}$
<u>Seawater</u>					
Northern Adriatic (7)		1.1	0.08	4.1	27
Central Adriatic (8)		0.3	0.03	2.2	28
<u>Karstic freshwater sources</u>					
Tagliamento River, NE Italy (9)	97	63	0.2	-	456
Soca (Isonzo) River, NE Italy (9)	204	58	0.3	35	208
Limski kanal groundwaters, Rovinj area (9)	-	110	0.8	68	142
Rjecina River, Rijeka area (8)	15	31	1.1	18	32
Rijeka Bay groundwaters (8)	70	35	0.4	25	88
Bakar Bay groundwaters, Rijeka Bay area (8)	-	39	0.3	17	154
Krka River, Sibenik area (10)	49	18	0.1	24	187
<u>Alpine rivers</u>					
Adige River, N Italy (9)	212	68	1.7	99	46
Po River, N Italy (9)	1585	147	4.6	120	37

Anthropogenic eutrophication-related phenomena have been reported in some semi-enclosed embayments with reduced water exchange, ranging from "red tide" events and near bottom anoxic conditions to relatively small but significant increases of phytoplankton primary production or bloom frequency (4,5,12-15,17,21).

Red tides

Localized "red tides", usually non toxic dinoflagellate blooms occurred regularly during the seventies in the Pula harbor (Istria Peninsula; zone 1 in Fig. 1) and several times during the eighties in Split harbor (central Dalmatian coast; within zone 4 in Fig. 1), clearly related to inadequate disposal of urban and industrial wastewaters in these semi-enclosed areas.

In the Pula harbor nutrient concentrations were greatly increased, particularly orthophosphate, compared with the coastal waters off Pula (Tab. 3). This was due to inadequate sewage and industrial wastewater disposal. Late spring blooms were dominated by *Prorocentrum micans* (up to 2·10⁶ cells dm⁻³), whereas in summer *Gonyaulax polyhedra* dominated (12). Among about 20 other dinoflagellate species *Gymnodinium adriaticus* and *Noctiluca miliaris* were the more frequent. Chlorophyll *a* concentrations up to 120 μg dm⁻³ were measured during the blooms. Cell extracts did not revealed significant toxicity effects on guinea pigs. Subsurface oxygen renewal was adequate in this shallow harbor (<10 m), since bottom oxygen saturation in the bloom periods did not decrease below 70% (12). Mortality