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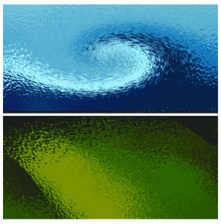

MONOGRAPH

Eutrophication in transitional waters: an overview

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Abstract

Despite their ecological and economic importance, Transitional Waters (TWs) have fallen behind all the other water categories in respect to the implementation of the Water Framework Directive (WFD) and further, pose some interpretation problems in other existing Water Directives. The problems faced by TWs concerning the implementation of WFD are mainly related to their characteristics, including: the high spatio-temporal variability, the fast response time to perturbations, their high productivity, the fact that primary production in these systems is normally not dominated by phytoplankton, their high socio-economic importance with a long historical tradition, and the strong anthropogenic pressures which exists in these systems. The present review attempts to bring together the main elements characterizing TWs and to identify the current understanding of the process of eutrophication and the problems this raises in establishing reference conditions, in view of the need of implementing the WFD. It is thus necessary to reach agreement on working definitions of TWs that can form the basis for the development of methodologies permitting the establishment of reference conditions; gain an understanding of the processes of eutrophication and the drivers and pressures that play a major role in their evolution; and investigate the use of indicators susceptible of accurately reflecting the ecological quality status of these types of water, as required by the WFD.

Keywords: Eutrophication assessment, environmental indicators, lagoon ecosystems, estuaries, Water Framework Directive.

1. Introduction

Eutrophication of inland, transitional, coastal and marine waters has been considered as the major threat to the health and integrity of aquatic ecosystems during the last decades. In a recent report by the European Commission (2007), it was stated that a large percentage of the European water bodies are at risk of failing to reach Good Ecological Status by 2015 and eutrophication related drivers are among the most important factors causing pressure and impacts on these water bodies.

Eutrophication was defined by Nixon (1995) as "*an increase in the rate of supply of organic matter to an ecosystem*". This definition puts emphasis on eutrophication as a process. In the EU water related policies eutrophication has been defined as: "*The enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the water balance of organisms present in the water and to the quality of the water concerned*" (European Commission, 1991). This implies that eutrophication is defined as a process with focus on nutrients and their effects on primary production, rather than on external sources of organic matter that tries to link cause and effect. In any case, eutrophication should deal with the processes as well as the associated effects of nutrient enrichment, thus specifically emphasizing anthropogenic eutrophication (Andersen *et al.*, 2007).

Despite their ecological and economic importance, Transitional Waters (TWs) have fallen behind all the other water categories in respect to the implementation of the Water Framework Directive and, further, pose some interpretation problems in other existing Water Directives. According to the Water Framework Directive (WFD) TWs are defined as ''*bodies of surface water in the vicinity of river mouths which are partially saline in character as a result of their proximity to coastal waters but which are substantially influenced by freshwater flows*'' (European Commission, 2000). This definition of transitional waters is equivalent to the concept of an estuary, which is the term scientifically used. However, classical definitions of estuaries exclude coastal systems such as lagoons, deltas and sounds (Day *et al.* 1989). In a recent paper McLusky and Elliot (2007) point out the main physiographic forms to be included under the term TWs as the following:

- *Classical estuary, tidally dominated at the seaward part; salinity notably reduced by freshwater inputs; riverine dominance inward*
- *Fjord, land freshwater seepage or markedly seasonal riverine inputs; limited tidal influence; stratified; long narrow, glacially eroded sea inlet, step side, sill at mouth*
- *Lentic non-tidal lagoon, limited exchange with the coastal area through a restricted mouth; separated from sea by sand or shingle banks, bars, coral, etc. shallow area, tidal range ≤ 50 cm*
- *Lentic microtidal lagoon, as above but with tidal range ≥ 50 cm*
- *Ria, drowned river valley, some freshwater inputs; limited exchange*
- *Fjard, glacially carved embayment, sea inlet, smaller then fjord; limited freshwater inputs*
- *River mouth, river outlet as well-defined physiographic coastal feature*
- *Delta low energy, characteristically shaped, sediment dominated, river mouth area; estuary outflow*
- *Coastal freshwater/ brackish, water plume outflow of estuary or lagoon, notably diluted salinity*

With this exhaustive list, the authors conclude that by considering that the term "TWs" encompass all these habitats[1], legislators in all countries may easily identify TWs.

The problems faced by TWs concerning the implementation of WFD are mainly related to their characteristics, including:

- The high spatio-temporal variability of the physico-chemical characteristics, hydromorphology and biota.

- The fast response time to perturbations due to their shallowness, as well as important and highly variable influences of freshwater (river discharges) and marine inputs (tides, winds, storms).

- Their high productivity (300-500 g C m^{-2} y^{-1}), which make them naturally eutrophic (Nixon, 1995). This makes it difficult to distinguish

[1] Perhaps what is missing in our opinion are Atlantic lagoons, e.g. Bassin d'Arcachon and Ria Formosa which are often meso-tidal. (e.g. Arcachon – average tide = 3 m)

between natural and cultural eutrophication.

- The fact that primary production in these systems is normally not dominated by phytoplankton (except in deep estuaries) such that simple Vollenweider type relationships do not apply (Nixon *et al.*, 2001). In estuaries and coastal lagoons primary production is primarily carried out by angiosperms, epiphytic algae, drift and attached macroalgae and epibenthic microalgae.
- Their high socio-economic importance with a long historical tradition that has created a synergy between TWs and humans. Some TW systems would not exist if not regulated by man (Carrada, 2007).
- The strong anthropogenic pressures that exist in these systems (Aliaume *et al.*, 2007) where even salinity, which is typically natural induced stressor, can be regulated by humans and induce major changes.

1.1. Estuaries

The most widely quoted definition of an estuary in the scientific literature is given by Pritchard (1967): *an estuary is a semi-enclosed coastal body of water which has free connection with the open sea and within which sea water is measurably diluted with fresh water derived from land drainage*. In an attempt to address the limitations of this definition, Fairbridge (1980) gave a more comprehensive definition of an estuary: *an estuary is an inlet of the sea reaching into a river valley as far as the upper limit of tidal rise, usually being divisible into three sectors; (a) a marine or lower estuary, in free connection with the open sea; (b) a middle estuary subject to strong salt and fresh water mixing; and (c) an upper or fluvial estuary, characterized by fresh water but subject to daily tidal action. The limits between these sectors are variable, and subject to constant changes in the river discharge*. This definition also excludes some coastal systems such as lagoons, deltas, sounds and non-tidal estuaries (Day *et al.*, 1989); therefore it is more convenient to speak about "estuarine systems" as a wide concept including classical estuaries and other coastal systems, from lagoons receiving very little continental drainage to deltas receiving large freshwater inputs.

Estuaries have been classified using geomorphic, hydrographical and ecological criteria, but the ecological ones have not been widely used. The geomorphic classification of Pritchard (1952) considers four types of estuaries: (a) "coastal plain", which are the classical estuaries formed by the marine transgression into fluvial valleys; (b) "lagoon-type, bar-built", which correspond to coastal lagoons and intermittent estuaries with restricted connexion to the sea; (c) "fjord-type", which are present in high latitudes and were formed by the marine transgression into glacial valleys; and (d) "tectonically produced estuaries", which are formed by land sinking due to tectonic faults. Another geomorphic classification was proposed by Davies (1973), considering the different types of estuarine systems as points along a continuum ranging from systems dominated by marine processes and with little freshwater inputs (coastal lagoons) to systems dominated by fluvial processes and with large freshwater inputs (deltas), being the classical estuaries in the middle of this geomorphic gradient.

Regarding the hydrological classifications of estuaries, Pritchard (1955) proposed three types according to their degree of stratification: (a) highly stratified or salt wedge estuary, which is present in microtidal coasts and mostly associated to a delta; (b) partially mixed estuary, where mixing due to tides dominates over fluvial processes and is mainly present in mesotidal coasts; and (c) well mixed estuary, where tidal mixing is totally dominant and is usually present in macrotidal coasts. A more dynamic and continuous hydrological classification was proposed by Hansen and Rattray (1966), which is the most widely used one. This is a two-dimensional classification based on two non-dimensional parameters, one related to circulation features (x axis) and the other related to stratification features (y axis). These authors concluded that there are four main types of estuaries: (a) type 1 or well mixed estuary, where the net flow is towards the sea at all depths and the salt flux is diffusive; (b) type 2 or partially mixed estuary, where a upper layer with a net flow towards the sea and a lower layer with a net flow towards the continent, and the salt flux is both diffusive and advective; (c) type 3 or fjord-type, where circulation is like in type 2 but the salt flux of the lower layer is mostly advective; and (d) type 4 or highly stratified estuary, with a strong stratification and a weak advective salt flux.

1.2. Coastal lagoons

Coastal lagoons are shallow coastal aquatic environments with limited exchange with the adjacent sea (Tagliapietra and Volpi Ghirardini, 2006; McLusky and Elliot, 2007). Similar definitions have been proposed by Barnes (1995) and Kjerfve (1994), and have been included in the CORINE technical guide (Bossard *et al.*, 2000), identifying coastal lagoons as "*a shallow coastal water body separated from the ocean by a barrier, connected at least intermittently to the ocean by one or more restricted inlets, and usually oriented shore-parallel*".

Coastal lagoons can be classified considering the degree of exchange with the adjacent sea into choked, restricted and leaky (Kjerfve, 1994), whereas based on geomorphology a system is called a coastal lagoon when the width of the connections with the adjacent sea at high tide are less than 20% of the barrier length (Bird, 1994), see Section 3.

The tidal regime allows to divide coastal lagoons into non-tidal (tide <0.5 m), microtidal (average tide 0.5 -1.0 m), mesotidal (1.0 -4.0) and macrotidal (> 4 m). Non-tidal lagoons are locally called "coastal ponds in Italy and Southern France, where they are for the most part choked or restricted. In the Southern European Arc, Mediterranean coastal lagoons are non-tidal or microtidal, whilst along the Atlantic coast of Portugal and France they are mesotidal.

The main factors responsible of origins and evolution of coastal lagoons are watershed and coastal geomorphology, tectonics, sediment supply from rivers and adjacent marine coastal zones, tidal range and hydrodynamics (Brambati, 1988; Bird, 1994). Briefly, the coastline typology is a key factor in lagoon formation. Low-lying coastal areas are suitable for the formation of littoral barriers allowing tidal expansion and ingression, whilst steep rocky shores provide strong resistance to marine ingression during transgression periods. Pristine lagoons originated during the Holocene sea level rise, when coastal flooding was accompanied by the accretion of littoral barriers. Tectonic processes also contributed to lagoon formation and nowadays control their persistence, e.g. through steady state conditions depending on opposing factors such as emergence, accretion and subsidence. Littoral arrows and barriers that enclose lagoons are formed from the equilibrium between sediment deposition and erosion, and depend to a large extent on coastal hydrodynamics (Barnes, 1980; Brambati, 1988; Kjefvre, 1994). Along microtidal low coasts, the formation of sedimentary barrier islands and littoral arrows is the main driving force of coastal lagoon formation, and, even with a small amplitude, tides are responsible for the sea-inlet persistence and hydrodynamics. Among others, climatic and meteorological factors such as wind and storminess have important impacts on lagoon morphodynamics.

Sediment characteristics and dynamics are associated with riverine transport, tidal regime and climatic factors. Usually, in microtidal lagoons gradients of sediment granulometry occur with a dominance of the finest fraction in the inner part, and coarse particles close to the sea inlets. Accordingly, water depth increases towards the sea-side. In non-tidal lagoons, depositional processes dominate both in the inner and outer zones, with an increasing depth in the central part of the system.

1.3. The transitional waters quality paradox

Even though the original term described in Elliot and Quintino (2007) refers to estuaries, it can be easily extended to all transitional water bodies. It is well-known that all these systems are naturally stressed because their inherent variability in terms of physico-chemical characteristics, e.g. salinity, temperature, concentration of inorganic and organic nutrient, oxygen, etc. Therefore, in these systems biota have become adapted to cope with “natural” stress. Concerning eutrophication, it is also possible to consider these systems as naturally eutrophic when compared with pristine freshwater or marine systems (NAS, 2000). Thus, in terms of nutrients pressures, it becomes difficult to distinguish between natural and anthropogenic eutrophication. This difficulty has been called the Estuarine Quality Paradox, but here we will term it Transitional Waters Quality Paradox since all TWs systems are under the influence the same type of pressures.

Due to the above-mentioned reasons, techniques to assess eutrophication as well as indicators developed to evaluate the stress typical of freshwater or marine ecosystems should be carefully evaluated before being applied to transitional waters environments (Viaroli *et al.*, 2004; Giordani *et al.*, submitted).

1.4. Aims of this review

This review was prepared following a request from DG-Environment where it has been observed that there is a major unresolved problem for all EU Water Directives (WFD, Nitrates, Urban Waste Water Treatment –UWWT- Directive) concerning transitional water systems (TWs) and their assessment.

In the Common Implementation Strategy (CIS) of the WFD a Guidance Document (GD) on eutrophication assessment in the context of European Water Policies has been drafted. This document includes: a general conceptual framework of the eutrophication process; consideration of the different EU water policies and International Conventions that address eutrophication and harmonization of the assessment and monitoring requirements based on the WFD; overview and harmonization of eutrophication assessment methodologies (river, lakes, transitional, coastal and marine waters); and case studies. However, also in this document TWs were poorly represented. Therefore, it was decided to develop a complementary document dealing specifically with eutrophication in transitional waters taking into consideration the special characteristics of these systems. The present article thus represents a general summary overviewing eutrophication in transitional waters and a first attempt in identifying the various problems involved with an ecological quality assessment of this water category. The document does not intend to be exhaustive and cover all aspects of TWs eutrophication, but it has been produced with the idea of circulating it among a wide audience of TWs experts to stimulate discussion in order to arrive at a general consensus on the methodology to propose for this water category. Furthermore, a second objective of this document is to provide a focus for an eventual contribution to the Guidance Document of Eutrophication concerning TWs.

2. Characterization of transitional waters

The Water Framework Directive (WFD)

Article 5 establishes the need for Member States to characterize their River Basin Districts. This includes 1) identification of surface water bodies' types and definition of biological and chemical reference conditions (natural baseline) for those types and 2) identification of the significant anthropogenic pressures and impacts on the surface water status.

This initial characterisation is the basis for designing monitoring networks in terms of which water bodies to include in the different monitoring programmes and which variables (i.e. physico-chemical and biological elements/ parameters) to monitor. The Directive in its Annexes II and V provide further clarification, principles and criteria identification of water bodies' types, establishment of references conditions, and identification of pressures and assessment of their impacts on the aquatic ecosystems.

2.1. Delineation of Transitional Waters

A correct identification of water categories is extremely important, as these are the basis of the Directive's requirements (i.e. identification water body types, establishment of reference conditions monitoring and assessment of quality status). Yet, the definition of transitional waters leaves room for subjective interpretation and thus may pose difficulties for the comparison of the Member States assessment systems and may jeopardise the achievement of the environmental objectives of the Directive (see page 2).

Within the Coast Working Group (WG) of the WFD CIS, Member States have discussed this definition and agreed on some criteria for identification of TW (CIS, 2003), which are as follow:

- *boundaries between transitional waters, freshwaters and coastal waters must be ecologically relevant;*
- *"...in the vicinity of a river mouth" meaning close to the end of a river where it mixes with coastal waters;*
- *"...partly saline in character" meaning that the salinity is generally lower than in the adjacent coastal water;*
- *"...substantially influenced by freshwater flow" meaning that there is a change to salinity or flow;*
- *if riverine dynamics occur in a plume outside the coastline because of high and strong freshwater discharge, the transitional water may extend into the sea area;*
- *for the purposes of the Directive, the main difference between transitional and coastal waters is the inclusion of the abundance and composition of fish fauna in the list of biological quality elements for the classification assessment of transitional waters.*

Member States have further agreed on criteria for defining of the seaward and freshwater boundaries of TWs. To define the seaward boundary, four methods are proposed: 1) use of boundaries defined under other European and national legislation such as the UWWT Directive; 2) salinity gradient; 3) physiographic features, and 4) modelling (CIS, 2003). For each case the method that results in the most ecological relevant definition of the transitional water should be selected.

To define the freshwater boundary of TWs two main methods are proposed: the position of fresh/salt water boundary (the WFD defines freshwater as less than 0.5 salinity) or the tidal limit of estuaries (CIS, 2003).

In spite of these common agreed principles in support of the identification of TWs there still remains room for the misuse of the term particularly in the Baltic area where, for the moment, each Baltic country has adopted a slightly different approach. In practice most decided not to designate any TWs.

Recently, McLusky and Elliot (2007), have defined the main physiographic forms under the category of TWs (see this article's introduction for definitions) with consideration to the relevant terminology in international legislation.

In the same article it is concluded that any possible repercussions – legal, administrative or environmental- of the different approaches to delineation of TWs will only become apparent at a later stage of the implementation of the WFD. It is envisaged that because of differences in sensitivity of coastal and TW ecosystems to pressures, an erroneous/different approach to identification of water category may lead to different degrees of protection and management from the Member States, i.e. a non-harmonized implementation of the Directive.

Further, delineation of TWs has not been made with consistent and coherent criteria by some Member State. In some cases typical estuaries have been allocated to coastal waters, and coastal lagoons (of the same type) have been allotted erratically to coastal, transitional or inland waters. This situation should be corrected to achieve the objectives established in the WFD.

2.2. Typology in TWs

The basic unit to which the environmental objectives of the Directive apply is the water body. This is a discrete and significant unit with uniform typology (i.e. similar physical attributes and biological patterns) and quality status. The Directive gives no indication of the minimum size of TWs to be identified as separate water bodies but a water body is seen as the endpoint of a hierarchical approach that starts with the delineation of surface water categories, followed by division into types, sub-division of types considering significant natural physical features, possibly further division according to differences in status or extent of protected areas. Thus, identification of water bodies is intrinsically linked to the typology, establishment of reference conditions, analysis of pressures and assessment of quality status.

The WFD proposes two systems for differentiation of TWs in types, "system A" or "system B"(Table 1). "System A" starts with allocation of TWs to the appropriate ecoregion, followed by division into types considering annual salinity and mean tidal range for which fixed boundaries are set. "System B" allows for more flexibility; types must be differentiated using "the values for the obligatory descriptors and such descriptors, or combinations of descriptors, as are required to ensure that type specific biological reference conditions can be reliably derived". The only condition is that this system should achieve at least the same degree of differentiation as would be achieved using "system A"

Salinity is both an impact (of hydrological alteration) and a factor of natural variability in TWs. In many cases the typology has been based on the present salinity regime rather than the regime under original (reference) conditions. This is a misinterpretation of the criteria for establishing TWs types.

Types should group sites where the biological patterns are similar in the natural baseline conditions, in order to enable the detection of the effects of human disturbance. The types derived as prescribed in the WFD use physical, chemical and morphologically attributes as a proxy of the biological patterns and to be meaningful need to be validated using biological data. These will be significant only if the variability of the biological parameters is smaller within types than between types. It is also expected that types can be different depending on the biological parameters chosen.

The grouping of TWs into types is a critical issue considering that these are very complex ecosystems characterised by a natural high spatial and temporal variability. In addition these ecosystems are affected by severe and diverse anthropogenic pressures and present important alterations to their natural

Table 1. Typology system A and System B for TW (WFD, 2000/60/EC).

System A	
Fixed Typology	Descriptors
Ecoregion	The following as identified on Map B in Annex XI: Baltic sea; Barents Sea; Norwegian Sea; North Sea; North Atlantic Ocean; Mediterranean Sea.
Type	Based on mean annual salinity < 0,5 ‰ Freshwater 0,5 to < 5 ‰ Oligohaline 5 to < 18 ‰ Mesohaline 18 to < 30 ‰ Polyhaline 30 to < 40 ‰ Euhaline Based on mean tidal range < 2 m microtidal 2 to 4 m mesotidal > 4 m macrotidal
System B	
Alternative Characterisation	Physical and chemical factors that determine the characteristics of transitional water and hence biological population structure and composition.
Obligatory factors	Latitude, longitude, tidal range, salinity.
Optional Factors	Depth, current velocity, wave exposure, residence time, mean water temperature, mixing characteristics, turbidity, mean substratum composition, shape, water temperature range.

biological communities, which further complicate the partition of TWs in ecological meaningful types.

There are two main TWs types of functional biological- hydromophological significance: estuaries (river mouth) and coastal lagoons. These two main types can subsequently be divided in sub-types according to salinity, tidal range and geomorphology.

2.3 Reference conditions

The main purpose of grouping TWs into types is to enable type specific reference conditions to be defined, which in turn are used as the anchor of the classification system. According to the WFD water bodies need to be assessed for their ecological quality and classified using a system of five quality classes (high, good, moderate, poor, and bad) on the basis of an Ecological Quality Ratio (EQR), which is defined as the ratio between reference and observed values of the relevant biological quality elements (Table 2).

In the Directive, high ecological status is defined as 'slight' or 'minor' deviation from the reference conditions of a surface water body type, while the good status is defined as 'small' deviation. Both the status of biological quality elements and the status of the supporting hydro-morphological (such as depth variation and structure of the intertidal zone) and physico-chemical elements (such as salinity or nutrient conditions) must be in 'high' status, if a water body is to be classified as 'high', and likewise must be 'good' status if it is to be classified as 'good'.

In the guidance developed within the CIS it is suggested, considering the expected variability of type specific reference conditions, to be more practical to consider that high status is equal to reference conditions (CIS, 2003).

The Directive specifies that reference

Table 2. Biological quality elements and metrics required for the classification and assessment of the high, good, and moderate ecological quality status of transitional surface waters according to the normative definitions described in the Annex V of the WFD.

Quality element	Metrics
Phytoplankton	Taxonomic composition, abundance, biomass, plankton blooms
Aquatic flora	Taxonomic composition, abundance, macroalgal cover as a proxy for biomass
Benthic invertebrates	Taxonomic composition, abundance, diversity, sensitive taxa (e.g. sensitive vs. insensitive species of organisms)
Fish	Taxonomic composition, abundance

conditions can either be spatially based, i.e. defined by collecting biological information from water bodies, which are (almost) in natural baseline conditions (sites with minor anthropogenic impacts), or derived by modelling, or by combination of both. If reference conditions are to be defined using modelling, predictive models or hind-casting using historical, paleoecological, and other available data can be applied.

For spatially based type-specific biological reference conditions, Member States need to develop a reference network for each surface water body type. The network shall contain a sufficient number of sites of high status to provide a sufficient level of confidence about the values for the reference conditions.

In many countries there may be no reference sites available or data, particularly biological data, are insufficient to carry out statistical analysis or validate models. In this case, expert opinion may be the only possibility to define reference conditions.

Where it is not possible to establish reliable type-specific reference conditions for a quality element in a surface water body type due to the high degree of natural variability associated to the element (not just as a result of seasonal variations), then the element may be excluded from the assessment of ecological status for the surface water type.

A comparable approach should be pursued for heavily modified (HMWB) and artificial water bodies (AWB). The reference conditions of these water bodies mainly depend on the hydromorphological changes necessary to maintain the specified uses listed in the WFD Article 4(3)(a). Maximum ecological potential (MEP) equal to the reference conditions for HMWB and AWB is intended to describe the best approximation to a natural aquatic ecosystem that could be achieved given the hydromorphological characteristics that cannot be changed without significant adverse effects on the specified use or the wider environment. Accordingly, the MEP biological conditions should reflect, as far as possible, the biological conditions associated with the closest comparable natural water body type at reference conditions, given the MEP hydromorphological and associated physico-chemical conditions (CIS, 2003).

Again, there is a methodological problem to establish reference conditions if the salinity criteria is not correctly used for establishing the typology. Reference conditions of TWs may reflect the original salinity regime, not the present one when is altered by hydrological modifications. This situation is quite common in many coastal lagoons. For example, after a severe anoxic crisis, a second connection with the sea was opened in Sacca di Goro in 1992 (Viaroli *et al.*, 2001). This fact modified the hydrodynamics in the lagoon as shown in Marinov *et al.* (2006).

3. Susceptibility to eutrophication

As stated by Cloern (2001), the early

development of eutrophication studies addressed mainly deep and large lakes. Nutrient loadings and responses of phytoplankton communities were considered as the main factors in regulating primary productivity, biomass build up and subsequent microbial decomposition and oxygen depletion (Vollenweider and Kerekes, 1982). A similar limnological approach was also applied later in studying coastal marine waters and estuaries (see for example Vollenweider *et al.*, 1992). However, great differences were found among systems, with recognition of system-specific attributes that influence the sensitivity of coastal marine ecosystems to nutrient enrichment. Among others, shallow depth is responsible of key features such as water turbidity, benthic vegetation, sediment biogeochemistry and nutrient stoichiometry which are of paramount importance in determining community assets and ecological successions.

New paradigms were also postulated, eutrophication being assumed to be a result of combined nutrient and organic matter enrichment (Nixon, 1995). Moreover, coastal areas are under pressure from a multiplicity of stressors, which can amplify stress effects through interaction of nutrients with hydrology alterations, overexploitation of fish stocks and aquaculture, biological invasions, toxic contamination and climatic changes (Cloern, 2001; Crossland *et al.*, 2005).

The main aspects we will consider in this Section are geomorphology and morphometry, hydrology and hydrodynamics and meteo-climatic conditions. Other aspects related to nutrient loads and ecosystem functioning such as primary production, top-down processes and biogeochemical buffers which also can change the way systems respond to nutrient enrichment will be treated in Section 4.

3.1. Morphology

Morphology of transitional water ecosystems controls internal hydrodynamics and exchanges with the adjacent sea, which in turn are responsible of water retention and flushing. Therefore, the adverse effects of eutrophication should be less detrimental in ecosystems with a higher flushing time. However, this relationship is not linear and depends upon a number of other factors, e.g. sedimentary biogeochemical buffers, which are site-specific.

Coastline development and the width of the littoral zone generally increase the carrying capacity of the water body by enhancing the buffering ability. Coastline development is also one of the factors determining wind fetch and controlling resuspension and mixing processes.

The morphology of transitional water bodies is defined though a number of direct and indirect morphometric parameters. Direct parameters are surface area, volume, mean depth, maximum depth, and section area. Indirect parameters are derived mainly from ratios of direct parameters such as (Håkanson *et al.*, 2006): exposure (Ex), dynamic ratio (DR), form factor and volume development (Vd), which are explained in Table 3 and Figure 1. Volume development, in particular, describes the form of a coastal area and gives an idea of the dilution of watershed-derived nutrients.

An important indirect parameters is the water residence time τ_{rd}=V/Q, namely the ratio of the total volume (V) to fresh water flow (Q). The ratio between the total volume and fresh water flow plus the exchanged flow (R) with the open sea gives the total water exchange time τ_{ex}=V/(Q+R). These values can be obtained applying the LOICZ methodology based on simple water and salinity mass balances (Gordon *et al.*, 1996). A parallel approach has been proposed by Håkanson *et al.*(2006) to calculate hydraulic retention time based on morphometric parameters, considering that actual water exchange

Table 3. Main indirect morphometric parameters.

Parameter	Name	Definition	Explanation
Ex	Exposure	$Ex = A_t / A$	A_t = cross sectional area (m^2) A = surface water area (m^2)
Vd	volume development	$Vd = 3D_m / D_{max}$	D_m = mean depth (m) D_{max} = maximum depth (m)
DR	Dynamic ratio	$DR = (A / D_m)^{1/2}$	A (km^2), D_m (m)

normally varies over time and space (both in the horizontal and vertical directions, e.g., above and beneath the thermocline). When the exposure (Ex) lies between 0.002 and 1.3, it is possible to calculate the surface (τ_{rd}^{s}) and deep-water (τ_{rd}^{d}) residence times from morphometric parameters using the following empirical equations (Håkanson *et al.*, 2006):

$$\tau_{rd}^{s} = -4.33 \cdot \sqrt{Ex} + 3.49$$
$$\tau_{rd}^{d} = -2.51 - 138 \cdot \log A_t + 269 \cdot \log Vd$$

where A_t is the section area in km^2 and τ is the residence time in days.

Håkanson *et al.* (2006) also proposed a simple expression to define a sensitivity index (SI, dimensionless) from the exposure (Ex, dimensionless) and the dynamic ratio (DR, dimensionless)

If DR ≥ 0.25, then SI = $\sqrt{\frac{DR/0.25}{Ex}}$

If DR < 0.25, then SI = $\sqrt{\frac{0.25/DR}{Ex}}$

Based on SI, one can conclude that very enclosed and very shallow TW bodies tend to be more sensitive to nutrient loading than deep waters bodies and in turn more susceptible to eutrophication.

Contrarily to lakes, measurements of morphometric parameters in TWs require boundaries between the transitional area and the open water be identified. The topographical bottleneck (Fig. 1) of TWs is the line where the exposure of the coast from winds and waves from the open sea is minimized (Pilesjö *et al.*, 1991). In the case of choked coastal lagoons such a line is intuitively easy to identify, whilst in the case of leaky lagoons and, even more, of river mouths, identification of the boundary may require testing the location of the borderline, until the minimum value of exposure (Ex) is obtained (Pilesjö, *et al.*, 1991). Ferreira *et al.* (2006) developed a semi-quantitative methodology that allows the division of estuaries and inshore coastal waters (e.g. coastal lagoons, embayments, rias, etc.) taking into account natural criteria (morphological and salinity based classifications) and the human dimension (pressure and state classification).

Once the transitional water body is appropriately delimited, morphometric parameters can be determined which are relevant to the quantification of internal fluxes (e.g. mean depth and water surface area), to mass-balance calculations (e.g. the coastal volume), and to regulation of the water exchange between the coast and the sea (e.g. the section area, the exposure and the filter factor).

In the case of coastal lagoons, there are other morphological parameters that become relevant in determining the susceptibility to eutrophication and which can be estimated from limnology. The ratio of watershed area to lagoon area is clearly one of the most important physical attributes responsible for eutrophication levels. A practical example can be found in the comparison of the behaviour of the Curonian and Vistula lagoons. Both

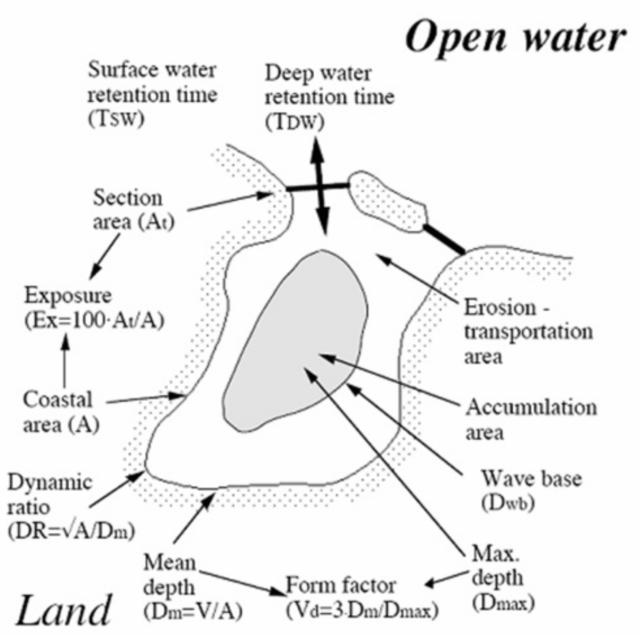


Figure 1. Illustration of key coastal parameters that may be determined from digitized bathymetric maps using a GIS (Geographical Information Systems) from Pilesjö, *et al.* (1991) and Håkanson et al. , (2006).

water bodies share similar geology, climatic conditions and morphology, but display very different levels of eutrophication since the flow of the Vistula River has been artificially diverted from the lagoon (Chubarenko *et al.*, 2001).The catchment area to river flow ratio is also an important factor considering the dilution of the nutrient loads in the freshwater inputs

Mean depth and hypsographic curves can be used for assessing the water volume to sediment surface ratio, which can be used as measure of the influence of benthic processes on the whole ecosystem metabolism (NAS, 2000). Moreover, the mean depth is a measure of the tendency of the water mass to undergo stratification, and can also be used to evaluate the development of shallow areas (e.g. the littoral zone). In transitional waters with low salinity and significantly long residence time the mean depth could also be used to predict the tendency to undergo regime shifts as is done for shallow or deep lakes (Scheffer 1998; Carpenter 2003).

3.2. Hydrology and hydrodynamics

Estuarine and coastal lagoon dynamics are controlled by hydrological factors which operate at several temporal scales, spanning diel, seasonal and annual (Paerl *et al.*, 2006; Arhonditsis *et al.*, 2007). Mixing and transport processes in TW systems are driven by tide, river flow and wind (Nagy, 2003). Among others, river flow could be regarded as a main factor that controls physical and chemical characteristics in the TW ecosystems (Borsuk *et al.*, 2004). For example, it is possible that reduced freshwater discharges associated with drought conditions will favour slower growing taxa, whereas in periods of higher river flows tolerant species to low salinity will be favoured (Arhonditsis *et al.*, 2007). Overall, freshwater inputs contribute to eutrophication process, while exchanges with adjacent coastal waters can prevent eutrophication through nutrient export and dilution. The combination of freshwater and marine water inputs affects water column stratification, which in turn is assumed to enhance the susceptibility to eutrophication, favouring phytoplankton growth in the photic zone and anoxia in the bottom waters (Howarth *et al.* 2000).

In case of river mouths, the lower density of river flow entering coastal waters produces a plume. The dynamics of the plume will depend on initial outflow conditions, subsequent mixing with marine water and existing alongshore marine currents. Riverine plumes transport large amounts of sediment, which settle and are deposited at the delta front. Rivers that carry large amounts of fresh water into relatively calm coastal seas produce stable and highly stratified river plumes, which tend to favour the development of eutrophication.

The relative contribution of hydrological factors to hydrodynamics depends on topographical characteristics of TWs ecosystems. In macrotidal systems, tides control surface water retention time. In microtidal systems, tides also play an important role, mainly affecting the dynamics of salinity, suspended particulate matter and nutrients.

The transport and mixing in a water body is normally described by the general equations of continuity and motion (Tritton, 1988), however, these are not possible to solve without introducing simplifying assumptions. The solution of these equations provides the velocity field obtained by the balance of forces acting on the water: inertia, pressure, gravity, rotational (Coriolis) and friction forces. In theory, it may be possible to distinguish driving processes from mixing processes. In practice, this is often impossible. Surface water mixing causes a change in boundary conditions, which causes water exchange, and so on. In many TWs systems the flow is governed by a balance between the bottom friction and the pressure gradient which may be barotropic, due to a water level or baroclinic gradient, which depends on density gradient. In any case, circulation patterns tend to be more complex and variable in TW systems than in fresh or marine water systems.

3.3. Meteo-climatic conditions

In non-tidal or microtidal systems, meteo-climatic factors, especially wind and precipitation are interlinked with hydrological factors and play a dominant role. In addition, solar radiation and temperature are also related to eutrophication susceptibility. Therefore, the geographic location of TW ecosystems becomes a driving factor for eutrophication processes. Solar radiation and temperature are usually greater at lower latitudes, and influence the rate of nutrient cycling and the length of the growing season. Temperature in particular affects biological process rates, and is a driver for evaporation. Evaporation intensity can be significant at lower latitudes, causing increase in salinity and nutrient concentrations especially in water basins with high residence time. However, the increase in salinity could also improve the eustatic exchange between the hypersaline semi-enclosed lagoons and the sea, while in the case of oligohaline environments the effect could be exactly the opposite. In the Baltic Sea climatic or geographical indices appeared to be of primary importance in describing the variation of chlorophyll-a levels (Gasiūnaitė *et al.*, 2005).

Precipitation should also be considered as a factor increasing the susceptibility to eutrophication though several mechanisms, runoff from the land leading in turn to decreased salinity and increased nitrogen and phosphorus (the latter mostly from sediment transport) loadings from agricultural land. A typical example is the Curonian lagoon (Lithuania), where precipitation accounts for 5-7% of total non-endogenous nitrogen inputs. In case of nitrogen limited systems the atmospheric deposition of nitrogen could also cause shifts in nutrient limitation

patterns.
Wind speed is a driving force, preventing or disrupting stratification as well as producing resuspension episodes, which increase water-sediment exchange rates. Wind can affect the distribution and growth of macrophyte assemblages (Marinov *et al.*, 2007) and, in some cases, effectively control the development of cyanobacteria blooms by shifting the phytoplankton community towards diatom dominance under the boreal oligohaline conditions (Pilkaityte and Razinkovas, 2006). Wind speed and fetch also influence resuspension, the latter in combination with morphology and wind rose. Resuspension processes in transitional waters are much less investigated and their effects on the susceptibility to eutrophication are not so clear. Floderus and Håkanson (1989) demonstrated the important role of resuspension on the fate of ephemeral mud blankets in the Kattegat area. This process might enhance nitrogen cycling and foster decline in water quality. Recent studies on resuspension in the Curonian lagoon (Razinkovas *et al.*, in preparation), also pointed to the possible enhancement of eutrophication effects by shifting N:P ratio in the water column to further nitrogen limitation during the nitrogen fixing cyanobacteria bloom. However, other evidence of resuspension effects on benthic-pelagic fluxes in the Kattegat (Christiansen *et al.*, 1997) points towards a decrease of dissolved reactive phosphate and inorganic N fluxes from sediment after resuspension. Turbidity could limit the development of the macrophyte belt, thus effectively fostering phytoplankton blooms in the case of highly eutrophic phytoplankton dominated-water bodies, by diminishing the role of submersed macrophytes in the nutrient turnover.
Finally, climatic variation can make a TW system more or less susceptible to eutrophication and climate change needs to be carefully evaluated for each TW body (Blenckner, 2004; Eisenreich, 2005).

3.4. Turbidity and light availability

Turbidity of the water column is a main factor that determines the light availability for primary production, especially in estuaries (Cloern, 1999). Therefore, turbidity should be considered in conjunction with the nutrient inputs to understand their impact on the ecology of the TW and its associated water quality. Cloern (1999) developed a combined indicator based on nutrients and light resources to assess the response of estuaries when nutrient enrichment occurs. This index is based on a model of phytoplankton growth, which includes the combined effects of photosynthetic efficiency, light availability, temperature, photo-adaptation and nutrient availability. Based on this model and data for a specific transitional water system it is possible to assess whether phytoplankton growth rates are more sensitive to changes in light or to nutrients. It is clear that priority to nutrient reduction should be given to TW systems classified as nutrient sensitive, whereas other type of measures would be more effective when light is the limiting factor.
Turbidity can be caused by suspended sediment particles. Soil erosion in the catchments and habitat destruction of floodplains and of the wetlands in the periphery of the water bodies have contributed to increased sediment loadings to TW systems. For example the Humber estuary, a major TW system in Great Britain, draining 28 % of England (Jickells *et al.*, 2000), has lost 90 % of its intertidal mudflats in the last 3,000 years. At present it has become a simplified turbid estuary with 100,000 tonnes of silt moving in and out every tide. Many estuaries such as e.g. the Gironde estuary in SW France are characterised by a maximum turbidity zone at intermediate salinity levels where mineralization exceeds primary production due to extreme light limitation. Another source of turbidity is

biological due to planktonic microalgae. Increased inorganic nutrient concentrations in the water column enhance phytoplankton growth, a phenomenon described as bottom-up control. The micro-algae will continue to increase under such circumstances until they become light-limited. In contrast, grazing by natural or cultured populations (aquaculture) of filter-feeding organisms such as bivalves may keep the population densities under control and maintain them at lower densities, a phenomenon which is called top-down control. In addition, the harvesting of these filter feeders from the TW represents an export of N and P elements, which may counterbalance increased nutrient inputs from the watershed (Ferreira *et al.*, 2007a).

Several of the microtidal or virtually non-tidal estuaries on the Mediterranean coast are stratified water bodies with a salt wedge (Ibáñez *et al.* 1997). The lower saltier water is isolated and has become anoxic in some cases, due to the accumulation of organic matter and the lack of vertical mixing. Because of very different metabolic pathways prevailing under oxic versus anoxic conditions, such systems show particular behaviour with respect to nutrient over-enrichment. Stratification of the water column has also been observed in shallower lagoons due to friction of water flow by floating dense macro-algal communities.

4. Nutrient enrichment and ecosystem responses

4.1. Introduction

Agriculture and urban activities are major sources of phosphorus and nitrogen to aquatic ecosystems, whilst atmospheric deposition contributes mainly as a source of N. These non-point inputs of nutrients derive from activities dispersed over wide farmland areas and are variable in time due to effects of weather (Vitousek *et al.*, 1997; Carpenter *et al.*, 1998). In the second half of 20th century, the net addition of P to croplands was approximately 400 million tons (Carpenter *et al.*, 1998). Up to 20% of the accumulated P has been released and transported to superficial water bodies. On a global scale the nitrogen cycle has become largely modified by the introduction of artificial nitrogen fertilizers produced by the chemical industry since the 1920's and this phenomenon has accelerated after the Second World War. Currently about 80 million tons of chemical N fertiliser is produced per year, which is almost the same amount as fixed by natural processes, i.e. 110-130 millions of tons per year, mainly biological nitrogen fixation (Vitousek *et al.*, 1997). An almost equivalent amount is also recycled through disposal of livestock manure on farmlands. The non-point sources of P and N are difficult to control. Overall, agriculture is a main source of N. For example in the Bassin d'Arcachon and in the Po plain agriculture represent up to 60% of the total N loading to coastal waters (Palmeri *et al.*, 2005; De Wit *et al.*, 2005).

Due to strong reduction of the use of phosphorus in detergent and improved waste-water treatment, in most places the loading of P form point sources to the TW has decreased, whilst N loading has further increased (Howarth and Marino, 2006). Concomitantly with a decrease of P this has often resulted to increased N:P input ratios that nowadays often strongly deviate from the Redfield ratio (N:P = 16:1 mol/mol). In some cases damming or water regulation in the tributaries has reduced the input of Si to TW. The changing nutrient ratios may have adverse impacts on the community structures and this aspect deserves to receive emphasis in managing the water quality of TW especially with respect to setting ecological quality objectives.

Phosphorus has the additional complication of its strong tendency to interact with sediment processes and a long-term history of P loading may be reflected by high P pools in the sediment that will deliver P to the water column for a very long period after discontinuation of the P overloading to the TWs. This aspect needs further study in the future, also considering effects of climatic changes on the hydrologic regime. Persistent drought followed by transient heavy rainfall could modify amount and timing of P delivery through erosion and transport of particulate P. Preliminary data on the Po River shows evidence that up to 50% of the annual P loading can be delivered in less than two months by a few flood episodes. This P load is composed mainly of particulate P, whose fate has to be clarified, e.g. by analysing P speciation.

Changes in primary production are direct responses to the increasing nutrient availability. The resulting biomass build-up then fuels microbial decomposition processes, eventually leading to oxygen depletion and nutrient recycling, inducing feedback loops within the system (Viaroli *et al.*, 2008). Allochthonous organic carbon can accumulate and be processed within the system giving rise to microbial decomposition, thus indirectly supporting primary production through nutrient mineralization and recycling.

However, contemporary conceptual models point out that coastal eutrophication is not a simple function of organic matter inputs and transformations, but rather is a complex suite of interactions among system hydrogeomorphology, nutrient enrichment, internal biogeochemical processes, primary producer and animal communities (Cloern, 2001). This means that each transitional water body has site-specific particularities, which will determine its ecological response and at the end determine its susceptibility to nutrient over-enrichment.

Site specificities become critical in transitional water ecosystems, where the transition from the continental to the marine domains determines a steep gradient of sedimentological, hydrological and biological conditions and induces a natural, often wide, variability of community structure and processes. Site specificities for TW have to take these aspects into consideration in the assessment of ecological status and in the definition of reference conditions as it is required by WFD. Furthermore, shallow depth and geomorphological heterogeneity favour the coexistence of taxonomically and functionally diverse plant communities, including rooted phanerogams, slow-growing and perennial macroalgae, fast growing and ephemeral macroalgae, and benthic, epiphytic and pelagic microalgae (Nielsen *et al.*, 2004). Thus, a great variability of primary production processes is naturally associated to communities with diverse specific growth and production rates (Sand-Jensen and Nielsen, 2004).

TWs are a natural receptacle for runoff from their catchments and therefore show a natural tendency for nutrient accumulation and eutrophication. The tendency to accumulate nutrients within the TWs is partly counterbalanced by the mixing with seawater, which combined with tidal and wind-driven flushing and direct water outflow results often in export of nutrients to the coastal sea. However, under certain circumstances, i.e. during periods of high primary productivity and low dissolved inorganic nutrients in the TWs, the trend may be reversed and the TWs may also function as a sink for nutrients coming from the sea. The LOICZ modelling approach provides a simple method to determine the mass balances of water, salts (conservative components) and the non-conservative elements C, N, P and Si. This approach quickly detects whether the transitional water body is a source or a sink for these elements (Giordani *et al.*, 2005).

When considering nutrient over-enrichment in aquatic ecosystems, in addition to the total quantities it is also important to consider their stoichiometric ratios. The prediction that changing stoichiometric relationships will induce directional changes in the community composition of primary producer communities is rooted in resource ratio theory (Tilman, 1982). Hence, the element ratios of the nutrient inputs from the catchment flowing into the TWs gives a good indication if the communities are being forced in a direction as predicted by the stoichiometric ecology theory (Hessen *et al.*, 2004). However, it needs to be considered that the actual element ratios of the nutrients within the TWs are modified by a myriad of biogeochemical processes and that interaction with the benthos is of particular importance. Local and regional changes have also increased the input of atmospheric deposition in the TW. Hence, changing ratios of DON (Dissolved Organic Nitrogen)/DIN (Dissolved Inorganic Nitrogen) will induce community changes.

Finally, the problem of nutrient over-enrichment in TW needs to be considered in perspective of global changes as predicted by IPCC. Particularly, the increase of temperature and pCO_2 will have strong impacts on the TW communities that interact with the nutrient over-enrichment in ways that are still poorly studied (Eisenreich, 2005).

4.2. Eutrophication and primary production in TWs: a bottom-up effect

The shallow depth of most transitional waters implies that benthic communities and benthic biogeochemical processes are major drivers of nutrient cycling in TWs. Submerged aquatic vegetation – SAV (phanerogams and macroalgae) and microphytobenthos (MPB) are primary producers that will respond to nutrient over-enrichment in direct and indirect ways and control the overall primary production (McGlathery *et al.*, 2004; Sundbäck and McGlathery, 2005). Therefore, levels and patterns of primary production should depend on the dominant components within the benthic community. A combination of hydrological and geomorphologic factors could greatly influence the primary production extent, leading to the dominance of phytoplankton and macroalgae in choked and restricted lagoons and benthic phanerogams in leaky ecosystems (Knoppers, 1994).

The sediments also encompass a sequence of oxic and anoxic habitats where bacterial processes strongly interact with the sediment chemistry and the N, P and Si biogeochemical cycling may be strongly modified by the nutrient over-enrichment. The nature of the sediment has a strong impact on the microbial process rates in conjunction with the hydrodynamic conditions. Muddy and silty sediments are cohesive and have characteristic redox profiles where the exchange of chemical species between the water column and the sediment is mainly determined by molecular diffusion. In contrast, sandy sediments are non-cohesive and permeable. In these sediments, the interstitial water movements increase the transport rates of oxygen and other solutes including DOM by orders of magnitude. This results in increased microbial process rates and such sediments are therefore very efficient bioreactors for the degradation of OM. As a result, these sediments show much more heterogenic patterns for geochemical parameters and it is often difficult to recognise the classical redox sequence. In addition, the exchange between water column and sediment is also strongly influenced by benthic fauna that contributes to bioturbation and bio-irrigation of the sediment.

The primary productivity of coastal marine ecosystems can be assessed using the classification criteria proposed by Nixon (1995), which cover the productivity range of near-shore coastal waters (Table 4). Under the conditions occurring in temperate regions, the trophic status ranking by primary production fully conforms to that based on nutrient concentrations – e.g. total phosphorus concentrations – and coastal waters are typically classified as meso- or eutrophic (Richardson and Jørgensen, 1996). Similar patterns have been reported by Borum (1996) for several coastal systems dominated by phytoplankton, for which the dependence of phytoplankton production on nitrogen loadings can be modelled with a logarithmic function.

In pristine transitional waters, primary production depends mainly on benthic vegetation and microphytobenthos (MPB), whilst the phytoplankton contribution is far less important (Sand-Jensen and Nielsen, 2004; Sundbäck and McGlathery, 2005). Increasing nutrient loadings can induce persistent and radical changes in abundance or productivity of one or more components of the community, leading to a dominance of one or a few of them (Sand-Jensen and Borum, 1991; Hauxwell and Valiela, 2004). Under these circumstances, shift in community structure and in production timing rather than changes in the total annual production can occur (Duarte, 1995; Valiela *et al.*, 1997; Raven and Taylor, 2003). In parallel, sediments will become increasingly heterotrophic, with an increased efflux of nutrients, which generate a positive feedback loop accelerating eutrophication through the increased internal loading (Eyre and Ferguson, 2002; Viaroli *et al.*, 2008). In other words, above certain thresholds, the community response to increasing loadings becomes non-linear. An example of the dramatic final outcome of eutrophication processes is given by the lagoon of Venice, where from 1980 to 1990 the annual gross production of ephemeral macroalgae (mainly *Ulva* and *Gracilaria* species) attained 1500-5000 g m^{-2} y^{-1} as dry mass (Sfriso and Marcomini, 1996; Sfriso and Facca, 2007) which is equivalent to 500-1500 g C m^{-2} y^{-1}; with the production bulk concentrated in less than four months, from April to late June. Similar patterns were also modelled (De Vries *et al.*, 1996) or measured (Nienhuis, 1992; Viaroli *et al.*, 2005) for other lagoons in different climatic regions in Europe (Table 5). Overall, at present the trophic condition of lagoons can be classified, according to the Nixon's scheme (Table 4) as ranging from eutrophic to hypertrophic. The resulting NEM, which is a proxy of the primary production rates, can be considered as a response to the nutrient loadings, and definitely a measure of the net ecosystem production (Fig. 2).

Table 4. Classification scheme of trophic status based on primary production (Nixon, 1995) and its relationship with nutrient fixed boundaries (OECD, 1968).

Trophic status	g C m^{-2} y^{-1}	mg P m^{-3}
Oligotrophic	≤ 100	≤ 10
Mesotrophic	100-300	10-35
Eutrophic	300-500	35-100
Hypertrophic	≥ 500	≥ 100

Experimental studies have demonstrated that abundance and distribution of the various primary producer forms are affected by a large variability, and that simple predictive relationships cannot be used for very shallow coastal lagoons (Taylor *et al.*, 1995; Nixon *et al.*, 2001). Although boundaries are often overlapped, broad relationships can be found between nitrogen loading and combinations of different primary producer components and between DIN (Dissolved Inorganic Nitrogen) and the net ecosystem metabolism (see Viaroli *et al.*, 2008 and references therein).

4.3. Eutrophication and primary production in phytoplankton dominated TWs

A first response of nutrient over-enrichment is the stimulation of primary production resulting in vigorous growth. Among phytoplankton some specific groups are enhanced with respect to others. These changes

Table 5. Average annual carbon production (g C m^{-2} y^{-1}) in transitional waters ecosystems of different European regions. SAV: submerged aquatic vegetation – mainly *Zostera* species, MA: macralgae on soft substrates – mainly *Ulva* species; MPB: microphytobenthos, PHY: phytoplankton.

Lagoon	SAV	MA	MPB	PHY	TOTAL
Grevelingen	170	0	30	170	370
Meere	300	60	50	0	410
Wadden Sea	10	15	90	190	310
Veerse Meer	10	110	90	240	450
Venice 1980-1990	0	500-1500	0	0	500-1500
Sacca di Goro 1997	0	650	0	nd	650

in phytoplanktonic community structures may change the light penetration features of the water column, the food web structure in the pelagic and the biogeochemical processes in the TW in general.

When the hydrodynamics are not so strong and at low salinities (< 9 psu) filamentous cyanobacteria are stimulated by nutrient over-enrichment (e.g., *Planktothrix agardhii*, *Aphanezomenon flos-aquae* and *Anabaena* sp.) and may become dominant particularly during summer. This is the case for many of the coastal lagoons and microtidal estuaries on the Baltic coast (Pilkaitytë, 2007), although examples of year-round dominance of filamentous cyanobacteria in TW at lower salinities do also exist in Southern Europe (Chomérata *et al.*, 2007). Some species are capable of nitrogen fixation and are therefore particularly well adapted to areas that have suffered high phosphate loadings. Many of these filamentous cyanobacteria comprise toxic strains, and occurrence of these toxic strains strongly jeopardises the recreational and fishing exploitation of these systems. Nevertheless, the mechanisms which determine whether toxic or non-toxic strains proliferate remains elusive and requires further study including using novel molecular ecology approaches to explore different strains.

In Europe, the filamentous species have not been reported to bloom at higher salinities (> 9 psu) and increasing the salinity of the TW by manipulating the tidal inlet has been used as a strategy to combat cyanobacterial blooms. However, some sub-tropical strains seem to be physiologically capable of coping with higher salinities (Moisander *et al.*, 2002)

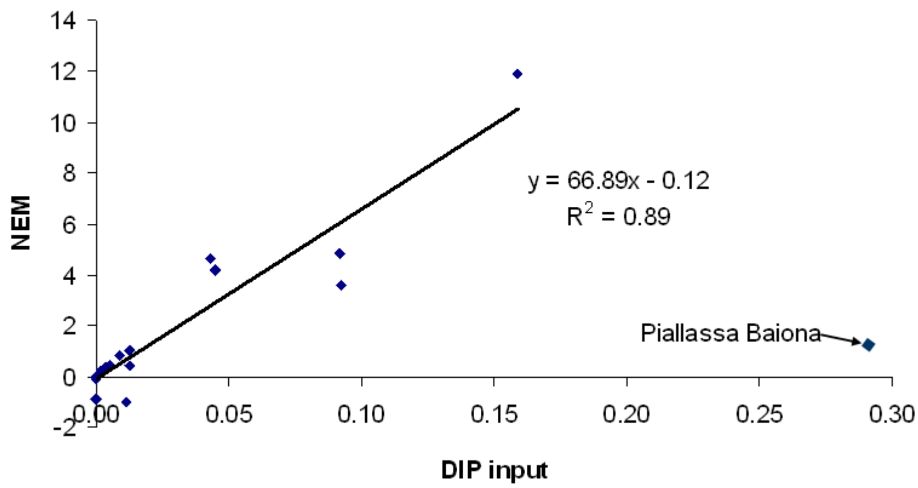


Figure 2. Dependence of Net Ecosystem Metabolism (NEM, mol C m^{-2} y^{-1}) on the dissolved inorganic phosphorus loading (mmol m^{-2} y^{-1}) in 22 transitional water ecosystems of the Italian coast (Giordani *et al.*, 2005).

and climate change may, therefore, interact with nutrient over-enrichment to stimulate the year-round occurrence of cyanobacterial blooms perhaps even at higher salinities. Small single cell cyanobacteria do naturally exist in Mediterranean TW at different salinities where *Synechocystis/ Synechococcus* sp. form part of the picoplankton community (< 2 µm in diameter). These picocyanobacteria are also a characteristic part of oceanic communities at low nutrient levels (reference conditions) and are *per se* not indicative of nutrient over-enrichment (Bec, 2006). Among the species that are capable to form high density blooms are the small size chlorophytes (0.6 – 4 µm, i.e. picophytoplankton or small nanophytoplankton) in hypereutrophic Valle di Comacchio and "Etang de l'Or" where the community is dominated by picoeukaryotes (Sorokin *et al.*, 1996; Andreoli *et al.*, 1998; Bec, 2006).

Changing N:P ratios may have a striking influence on the phytoplankton community structure in TW. Hence, the combination of high phosphorus, low salinities and N limitation may favour the N_2 fixing filamentous cyanobacteria as mentioned above. In this respect, the sediment can represent a source of phosphorus (see below). Particularly, when the TW has been submitted to high phosphorus loadings during many decades. Decreasing N to Si ratios in TW and coastal seas increase the competitiveness of dinoflagellates with respect to diatoms. The former comprise several toxic species that may form harmful algal blooms. Another problem that needs to be taken into account is that some species, but not all, are able to use organic nitrogen compounds. Organic nitrogen inputs into TW originate from freshwater inputs through the tributaries, but also through atmospheric deposition. Hence the ratio of organic to inorganic N may have a profound effect on the phytoplankton community structure.

In highly turbid estuaries, the phytoplankton community shows a typical transect in relation to the maximum turbidity zone (MTZ). Above stream of the MTZ, the phytoplankton communities are often dominated by diatoms (Lemaire *et al.*, 2002). The MTZ itself is a zone where primary productivity is virtually non-existent due to light limitation and it represents the zone of intense degradation. In spring a phytoplankton bloom may occur downstream of the MTZ on recycled nutrients comprising diatoms, dinoflagellates and cryptophytes (Lemaire *et al.*, 2002). Eutrophication of such an estuary may result in increased phytoplankton concentration in the upstream part of the MTZ. As a result, hypoxia becomes more pronounced in the MTZ and in some cases (e.g., Loire estuary) a completely anoxic MTZ persists for part of the year.

4.4. Changes in primary producer community in shallow TWs: succession and shift in benthic communities

In shallow oligotrophic and clear TWs, perennial phanerogams form extensive meadows of few species. In the temperate bioclimatic zone, e.g. in the Northern Adriatic, the dominant species are *Zostera marina* and *Zostera noltii*, often associated with *Cymodocea nodosa*. In choked lagoons under riverine influence, species of the genus *Ruppia* often become dominant. *Ruppia* is a freshwater macrophyte with a pronounced marine tolerance, which has a world-wide distribution and often forms extensive and productive meadows in shallow transitional waters. *Zostera noltii* also occurs in the Mediterranean bioclimatic zone, associated with *Ruppia cirrhosa* and *R. mediterranea*. Under oligotrophic conditions, sessile and perennial macroalgae like *Valonia* and *Acetabularia* are often found. At increasing nutrient loadings, opportunistic and ephemeral seaweeds such as *Gracilaria, Ulva* and *Cladophora* along with cyanobacteria become dominant, inducing degraded conditions.

In choked or restricted lagoons with soft-muddy sediments, microphytobenthic (MPB) communities are favoured by wind-driven resuspension, which can favour out-competition of other components of the benthic vegetation community (Sundbäck and McGlathery, 2004).

Conceptual representations and a qualitative assessment of relationships among primary producers and their succession is summarised in Table 6. In nutrient poor and well-flushed ecosystems, rooted phanerogams dominate until the moment that they are limited by light penetration (depth effect) or by turbidity and shading by floating vegetation. Increasing loading rates support the development of epiphytic microalgae and filamentous macroalgae, whilst high loaded water masses become dominated by phytoplankton. The community evolution along eutrophication gradients has also been investigated using a conceptual model, characterised by at least three successional phases dominated by seagrasses, macroalgae and phytoplankton (Schramm, 1999). Sea grass species are assumed to take advantage of nutrient available in pore-water, whilst nutrient loadings to the water column favour phytoplankton and epiphytic algae to develop until rooted phanerogams are displaced and substituted by either floating ephemeral macroalgae or phytoplankton communities (Borum, 1996; Hemminga, 1998). In turbulent or turbid waters, picoplankton and cyanobacteria, and/or MPB tend to dominate since they benefit from frequent sediment resuspension. In coastal lagoons, different communities of primary producers can coexist at different degrees of development/dominance. The phytoplankton component attains a certain level only in the open water mass, where it can be supported by water exchanges with the adjacent sea and freshwater nutrient loadings. In polluted waters, small-sized cyanobacteria, nano- and picoplankton can often prevail, with an increase in the heterotrophs to autotrophs ratio (Coppola *et al.*, 2007).

Table 6. Conceptual representation of the succession of aquatic vegetation along an increasing eutrophication gradient according to 1: Nienhuis (1992), 2: Valiela *et al.* (1997) and Dahlgreen and Kautsky (2004); 3: Schramm (1999); Viaroli *et al.* (2008)

Succession phases and conditions (pristine → altered)				Ref
phanerogams	phanerogams+epiphytes	macroalgae+phytoplankton		1
seagrasses		macroalgae	phytoplankton	2
perennial benthic macro-phytes	macrophytes+ fast grow-ing epiphytes	free floating macroalgae+phytoplankton	phytoplankton	3
perennial benthic macro-phytes	macrophytes+ fast grow-ing epiphytes	free floating macroalgae+phytoplankton	Phytoplankton Picoplankton/cyanobac-teria	4

In phanerogams meadows, nutrient over-enrichment particularly stimulates the epiphyte and fast growing macroalgae communities. Epiphyte communities comprise microalgae (often diatoms) and cyanobacteria together with their accompanying bacterial flora that occur on the leaves and stems of phanerogamms and on the thalli of macroalgae. Excessive epiphyte growth is detrimental for the growth of their hosts due to shading. This may result in collapse of the macrophyte stands. In general, while nutrient slight nutrient over-enrichment results in an immediate increase of the growth of the seagrasses, subsequent reactions in the ecosystem may create unfavourable conditions among which the excessive epiphyte growth is an example. It appears that the seagrass vegetation has evolved to occupy a niche of nutrient limited environments in the TW. This is, for example, shown by the recurrent association of seagrasses with nitrogen fixing bacteria (Welsh, 2000). Hence it may be concluded that a seagrass meadow characterised by less vigorous plants living in association with nitrogen fixing bacteria is a better guarantee for prolonged survival for this vegetation compared to a more eutrophied site where plants appear more vigorous but at the same time are exposed to risks of disappearance (De Wit *et al.*, 2001). Another feature of seagrasses is that their photosynthesis rates are often limited by the inorganic carbon availability reflecting that these species were derived from land plants by secondary evolution. Hence, it may be expected that increased CO_2 concentrations as predicted by IPCC may stimulate seagrasses in the future and partly off-set other negative impacts on seagrass meadows characteristic of the last decades (Palacios and Zimmerman, 2007).

Benthic vegetation communities are to a certain extent self-regulated, mainly through their capacity to control nutrient concentrations and to maintain oxic conditions. In healthy seagrass meadows, production and respiration rates are usually well balanced with smooth fluctuations. Oxygen is also released in the root system through radial oxygen loss (ROL), which sustains oxidation of reduced sediments allowing seagrass survival in a hostile environment (Hemminga, 1998; Pedersen *et al.*, 1998). In the rhizosphere, sulphate reduction rates can attain some 100 mmol m^{-2} d^{-1} with the production of toxic sulphides for macrophytes (Isaksen and Finster, 1996; Welsh *et al.*, 1996; Holmer and Nielsen, 1997). When the ROL is active sulphides are oxidised by bacterial processes, as well as being controlled by reactions with iron and oxygen. When nutrient availability increases, the increased productivity and biomass of epiphytes and phytoplankton, and later in the succession the macroalgal coverage, threaten the capacity of seagrasses to control the redox status and sulphide concentrations in the sediments (Heijs *et al.*, 2000; Azzoni *et al.*, 2001). Under increasingly low redox and high sulphide concentrations a feedback loop in the root-sediment interactions can be established, which reduces root and rhizhome elongation, inhibits cellular respiration and often has lethal effects (Touchette and Burkholder, 2000; Koch and Erskine, 2001; Calleja *et al.*, 2007). The inhibitory to lethal effects of sulphides further limit the degree of oxygen release to the sediments, thus enhancing sulphide accumulation. Overall, anoxia and sulphide production favour seagrass displacement. Iron buffers along with oxygen production and transport to the rhizosphere could provide a feedback link between sediment and seagrasses, controlling the fate of benthic vegetation. For example in carbonate rich sediments, iron can be adsorbed and retained by carbonate particles, lowering sulphide buffering. Under these circumstances, dissolved sulphides released into pore-waters are toxic to vegetation at very low concentrations, whilst in carbonate-poor sediments the toxicity threshold increases by two orders of magnitudes (Holmer *et al.*, 2003; Calleja *et al.*, 2007).

4.4. Macroalgal blooms

Macroalgal blooms have been identified as the most severe response of vegetation communities to increasing nutrient loadings (Valiela *et al.*, 1997; Raffaelli *et al.*, 1998; Raven and Taylor, 2003). Opportunistic *r*-selected seaweeds such as *Gracilaria, Ulva* and *Cladophora* spp. along with cyanobacteria indicate the degraded-eutrophic state, which corresponds to high nutrients conditions. Macroalgae with a sheet-like poorly differentiated thallus and an annual life cycle (i.e. ESG II, according Orfanidis *et al.* 2003) are particularly stimulated at high N loading even after P reduction (high N/P ratio). Species like *Ulva rigida* may attain biomass densities above 10 kg of fresh weights m^{-2}, before they collapse. Floating foliose and filamentous macroalgae are often transported by currents. Drifted biomass can be stranded and deposited at the sediment surface where it causes alterations of biogeochemical processes. In microtidal TWs with calmer waters the macroalgal blooms may contribute to a physical stratification of the water column, with oxygen oversaturation in the surface layers and hypoxia or anoxia in the lower strata (Krause-Jensen *et al.*, 1999; Brush and Nixon, 2003).

In shallow environments dominated by macroalgae, oxygen concentrations undergo much wider fluctuations than in seagrass meadows (Viaroli *et al.*, 2001; Viaroli and Christian, 2003 and references therein). The abnormal O_2 production is usually accompanied by retention of labile organic matter (OM) within the water mass and at the sediment surface. The sudden collapse of such an algal bloom under calm weather conditions and favoured by high temperatures may result in a dystrophic crisis where the whole water column becomes anoxic (Izzo and Hull, 1991; Castel *et al.*, 1996; Viaroli *et al.*, 1996; de Wit *et al.*, 2001). Under such circumstances sulphate-reducing bacteria produce hydrogen sulphide, which is a highly toxic compound in the water column. The anoxic dystrophic crises is followed by a phase of re-oxidation of hydrogen sulphide either by chemotrophic bacteria creating characteristic white waters (e;g. Sacca di Goro and part of Etang de Thau

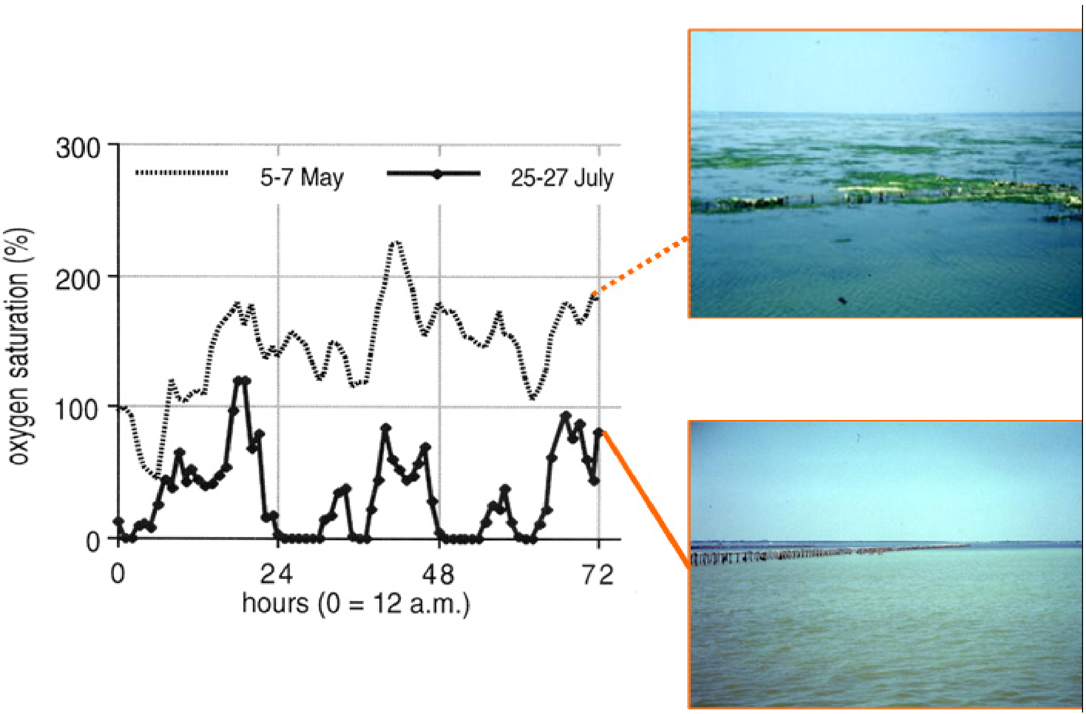


Figure 3. Typical oxygen (% saturation) trends during maximum macroalgal growth (5-7 May 1992) and in the pre-dystrophic phase (25-27 July 1992). Modified from Viaroli & Christian (2003).

in 2003 and 2006) or by phototrophic sulphur bacteria resulting in red waters (during the 1970's- early 1990's in Etang de Prévost (Caumette *et al.*, 1996). A typical oxygen pattern in macroalgal mats is shown in Figure 3.

4.5. Organic matter (OM) degradation: Effects of increased OM inputs related to increased primary production (PP) and increased OM loadings from the catchments

In deep estuaries, a part of the phytoplankton production is consumed in the pelagic food web resulting in transfer of OM to higher trophic levels concomitant with remineralisation of a major portion of that OM. An effective microbial loop enhances the remineralisation and turnover of nutrients, while it reduces the transfer of organic matter to higher trophic levels. Therefore, the structure of the pelagic food web is very important and in case of a strong top-down control the consumers can keep the phytoplankton densities under control, thus contributing to high water transparencies. Benthic filter feeders also contribute to the grazing of phytoplankton showing an important interaction between the pelagic and the benthos. In this respect, bivalves are of particular importance in TWs. In many cases, the native oyster species (*Ostrea edulis*) has been massively collected by coastal human populations since historic times in TWs. Hence, *O. edulis* was often among the first species to become completely depleted in the TWs, with a likely negative impact on water column transparencies (Lotze *et al.*, 2006). Nowadays, a large part of the filter-feeding in TWs is artificial, i.e. a result of aquaculture often using introduced species. This artificial top-down effect can probably be considered as a positive contribution to maintain water quality and needs to be taken into consideration for the management strategies in view of the WFD. This subject poses some conceptual problems when defining reference conditions for TWs and needs further study for appropriate assessments. Changes in food web structure can also be driven by bottom-up effects as induced by changes in community structure of the phytoplankton. Particularly, the efficiency of grazing on filamentous cyanobacteria is low in TWs and a cyanobacterial population may, by reducing the grazing pressure, introduce a positive feedback loop enhancing its own proliferation.

In shallow TW dominated by benthic macroalgae, a major part of the primary production is mineralised within the sediment. Sediments are spatially structured environments comprising oxic and anoxic habitats where bacterial degradation processes closely interact with the sediment geochemistry. The reduced products originated from the anaerobic microbial degradation processes, i.e., NH_4^+, and particularly Mn^{2+}, and HS^- are noxious for most of the water column biota and their efflux into the water column is therefore undesirable. A high input of HS^- and H_2S into the water column can also trigger a dystrophic crisis (see above). Under healthy conditions, these reduced compounds are oxidised in the superficial layers of the cohesive sediments that remain oxic, or by the mixing of oxic waters with anoxic waters in non-cohesive sediments. Such balanced conditions are typical for a healthy ecosystem and are often encountered in lagoons and estuaries where phanerogames persistently structure the benthic communities.

Inorganic free sulphides (HS^- and H_2S) interact with the rest of the sediment microbiology and geochemistry (De Wit *et al.*, 2001). HS^- and H_2S can be oxidised, as mentioned earlier, in the superficial layers by sulphur-oxidising bacteria that consume oxygen; however, in addition specialised sulphur-oxidising bacteria can oxidise these compounds in a chemical reaction with nitrate as the electron acceptor and others in a photosynthetic reaction using light. HS^- and H_2S can also be sequestered in the sediment by reaction with iron forming iron sulphide (FeS) and pyrite (FeS_2). However, iron is also important for sequestering phosphates by a reaction that occurs in the oxic part. Hence, through redox conversion, sulphide and phosphate may compete for binding with iron and as a result under highly reducing conditions with a high amount of HS^- and H_2S in the sediment, the phosphate can be liberated from the iron complexes and flux as ortho-phosphate into the water column (Deborde *et al.*, 2008). Such a phenomenon will enhance the eutrophication of the water column that might induce a shift in the primary producer communities (see Fig. 3). Hence, excessive reduction of sediments followed by a dystrophic crisis may induce dangerous positive feedback loops that stabilise the system in an undesired state characterised by a high level of eutrophication, loss of phanerogam species (see subsection 4.3) and frequently occurring dystrophic crises. In this respect, systems that are poor in iron are more sensitive towards this phenomenon compared to iron rich systems. In general for understanding the water quality it is necessary to gain some insight into the functioning of the sediments. In some systems that are particularly rich in calcium and poor in iron, phosphate may also be bound with calcium as apatite (McClathery *et al.*, 2001). In conclusion, the cycling of phosphorus is determined to a large extent through its interactions with the sediment and the benthic biota. The phosphate in the sediment may represent an internal source that can be transferred to the water column. Similarly as in lakes, TW that have been subjected to a long history of phosphate over-loadings, may take a very long time to fully recover to the reference conditions due to slow and continuing release of phosphate from the sediment to the water column. Nevertheless, in this respect precise information is often missing and it is, therefore, most important to evaluate the sedimentary P pools in TW and consider if these pools may delay ecosystem recovery to the reference states as defined in WFD.

Increased nitrate-loading to the TW can, in principle, give rise to enhanced denitrification. Denitrification is a process that occurs under anoxic conditions and results in conversion of NO_3^- via N_2O to N_2. This process can be very useful for N abatement, although the N_2O that is also produced during this process is a powerful and therefore undesirable green-house gas. Moreover, the denitrification process is in competition with another dissimilatory reduction pathway, i.e. dissimilative reduction of nitrate into ammonium (DRNA). DRNA conserves N for the ecosystem and this process is stimulated with respect to denitrification when OM loadings are high. Therefore a highly eutrophied environment is often less favourable for denitrification. Denitrification occurs in the sediment in anoxic layers, where nitrate that originates from the water column of the TW or is produced by nitrification of NH_4^+ into NO_3^-, hence the processes of nitrification and denitrification can be strongly coupled. The study of these processes and their interaction in TW were the subject of the EU project NICE (http://www2.dmu.dk/LakeandEstuarineEcology/NICE). The picture has become more complicated recently, since the discovery of anaerobic oxidation of ammonium (ANAMMOX). ANAMMOX also results in production of N_2 and can be considered as an alternative pathway for biological N abatement in aquatic ecosystems (Dalsgaard and Thamdrup, 2002), although its importance in TW remains to be assessed in detail.

In hyperautotrophic-dystrophic lagoons, the vegetation community does not respond solely to external stressors, but can also amplify responses through biomass build up, organic matter (OM) accumulation within the system and its decomposition (Nedergaard *et al.*, 2002; Banta *et al.*, 2004). Decomposition processes are regulated not only by OM quantity but depend also on its quality, which may control critical steps in the seasonal evolution of oxygen availability. In turn, OM quality and its recalcitrance depends on types, growth rates, life cycles and elemental and macromolecular composition of benthic vegetation (Enriquez *et al.*, 1993). Comparative studies demonstrate that under summer conditions *Ulva* sp. decomposes at an almost constant rate of 3% per day, which is three times greater than that of *Zostera marina* (Buchsbaum *et al.*, 1991). Similar differences have been found when comparing the degradation of the macroalga *Monostroma obscurum* and the seagrass *Zostera noltii* (Bourgues *et al.*, 1996). Decomposition modes and rates not only influence the extent of the oxygen deficit and sulphide release, but also strongly modify nitrogen and phosphorus pathways and fate (Amtoft Neubauer *et al.*, 2004; Lomstein *et al.*, 2006).

4.6 Sedimentary biogeochemical factors as a tool for assessing buffering capacity and vulnerability towards eutrophication

Granulometry, bulk density, dry matter, water content and porosity, provide basic information on sediment composition and are important determinants of diffusive properties and permeability and therefore of exchanges of oxygen and nutrients with the water column. Sediment structure and composition control the capacity to retain inorganic nutrients and toxic compounds, e.g. sulphides.

Based on sediment granulometry and their functioning in the hydrodynamic context of the transitional water ecosystem, two major sediment types can be distinguished, i.e. cohesive and non-cohesive. Muddy and silty sediments are cohesive and sandy sediments are non-cohesive and permeable. The interstitial water in cohesive sediments is maintained in place by different interaction forces. Under natural conditions, there is virtually no lateral movement or percolation in cohesive sediments. For intertidal flats when such cohesive sediments emerge during ebb tide these remain fully saturated with water and do not dry out during ebb tides. In contrast, in non-cohesive sandy sediments the water is attracted much more loosely. Under quiet hydrodynamic conditions, this force is sufficient to keep the interstitial water in place, while under many natural conditions the hydrodynamic forcing induces water movement through the sediment comprising percolation, lateral flow and upward welling. Intertidal non-cohesive sediments loose a large part of their interstitial water during ebb tides.

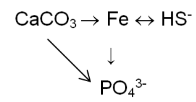
In cohesive sediments, the transport of solutes is determined by molecular diffusion, which is an efficient process at the µm scale, but very inefficient for longer distances. The mineralisation of organic matter in the sediment creates a large oxygen demand. While the oxygen delivery is limited by molecular diffusion, the sediment of cohesive sediments becomes anoxic at very shallow depth (i.e. typically below 0.5 – 2 mm depth). In non-cohesive sediments, the interstitial water movements increase the transport rates of oxygen and other solutes including DOM by orders of magnitude. This results in increased microbial process rates and such sediments are therefore very efficient bioreactors for the degradation of OM, by mixing reactants and diluting inhibitory end-products.

The sulphide produced in the anoxic sediment zones by sulphate reducing bacteria can partly be oxidized by aerobic processes the intensity of which are strongly determined by the physical characteristics and hydrodynamics of the system. The remaining sulphide strongly interacts with the biogeochemical processes. In coastal areas, the biogeochemical interactions between iron and sulphide play a major role in regulating sulphur speciation and retention of phosphate in the Fe-bound pool (Howarth and Stewart, 1992; Fenchel *et al.*, 1998). Via a suite of redox reactions, sedimentary reactive iron immobilises sulphides as highly insoluble FeS and FeS_2 . Therefore, the extent of sulphide release is regulated by the sedimentary sulphide/iron-monosulphide/pyrite system, which represents a potent mechanism for the removal of toxic hydrogen sulphide (de Wit *et al.*, 2001; Rickard and Morse, 2005 and references therein). The sedimentary buffering capacity is not a simple function of the reactive iron concentration, since not all of the iron is available to react with sulphide; benthic fauna and microbial processes play also an important role in the sedimentary sulphur cycle (Meysman and Middelburg, 2005; Rickard and Morse, 2005).

Factors influencing iron availability have a great influence on iron-based buffers (Chambers *et al.*, 2001; Rozan *et al.*, 2002). The biogeochemical reactions of iron and sulphide control also sedimentary phosphorus cycling (Golterman, 1995; Giordani *et al.*, 1996; Heijs *et al.*, 2000; de Wit *et al.*, 2001; Rozan *et al.*, 2002).

The calcium/carbonate/phosphate and the iron hydroxide/phosphate/sulphide systems have been proposed to interact, through feedback loops (Fig. 4.). Carbonates control both Fe and PO_4^{3-} mobility, iron controls PO_4^{3-}, whilst Fe and HS^- exert a reciprocal control through the formation of insoluble iron sulphides. Assuming similar iron, phosphates and sulphide concentrations, when $CaCO_3$ is low, Fe and PO_4^{3-} are potentially soluble, attaining high concentrations in the pore water, whilst sulphides are kept low by iron trapping. At high $CaCO_3$ concentrations, Fe and PO_4^{3-} are potentially insoluble, due to adsorption onto carbonates, thus allowing a potential sulphide release into the pore water.

McFadden *et al.* (2007) proposed to assess coastal vulnerability as a difference between impacts and effects of adaptation at the ecosystem level. Among other indicators, the adaptation of the ecosystem could be assessed as the potential buffering capacity. Single variables give only a partial view of the ecosystem buffering capacity or adaptability to changes/impacts. Tentatively, one can propose to integrate the simple, easily measured state variables mentioned above, into a synthetic index to provide a rapid assessment of the potential vulnerability level (PLV) for the identification of sites sensitive to eutrophication (Viaroli *et al.*, 2004). This index combines the water retention time (WRT) in the aquatic ecosystem with five sedimentary variables, namely granulometry and sedimentary OM, carbonates, reactive iron and Acid Volatile Sulphide (AVS). These state variables have been chosen because they represent the potential capacity of the ecosystem to react against specific pollutants and other forcing factors. Granulometry and sedimentary OM provide basic information on sediment composition and are important determinants of exchanges of oxygen and nutrients with the water column. The sedimentary carbonate content can be used to assess the



$CaCO_3$	Fe	HS^-	PO_4^{3-}
-	+	-	+
+	-	+	-

Figure 4. Relationships between sedimentary geochemical variables that account for potential buffering capacity. When $CaCO_3$ is low or absent (-), Fe is present and abundant (+) and available for reacting with HS^- which becomes low (-); therefore both $CaCO_3$ and Fe cannot react with PO_4^{3-} which can potentially increase (+). For details see De Wit *et al.* (2001) and Chambers *et al.* (2001) and references therein.

capacity of the sediment to retain phosphate through the calcium/carbonate/phosphate system. Reactive iron provides an indication of the sediment capacity to buffer against sulphides, whilst the AVS to reactive iron ratio is an index of the buffer saturation. WRT is the ratio of the recipient volume to the water flux through the system. Ecosystems with low WRT undergo rapid flushing, whilst a high WRT is an index of potential water stagnation. In turn, water stagnation can increase the retention of pollutants within the system and can induce a risk of water oxygen depletion. WRT can be easily estimated with water mass balances (Gordon *et al.*, 1996). Since oxygen availability is critical in the management of coastal systems, hydrological and sedimentary data should be integrated with an assessment of oxygen metabolism and the amplitude of its variation with time (see also the next sections).

5. Indirect effects of nutrient enrichment in TW

5.1. Oxygen deficiency and dystrophic crises

Among the most compelling evidence of the negative consequences of eutrophication are recurrent and persistent periods of hypoxia and anoxia (Diaz and Rosenberg, 1995). Oxygen deficiency may be the most deleterious effect in coastal and transitional waters.

A reduction in dissolved oxygen is one of the most important effects of eutrophication on aquatic organisms (Breitburg, 2002). Although hypoxic and anoxic conditions may naturally exist in aquatic systems, their occurrence both in frequency and extent of anoxic events in coastal and transitional waters appears to be increasing due to human activities, leading to faunal impoverishment and mass mortality (Diaz, 2001). Over the last 15 to 20 years the number of coastal ecosystems subjected to seasonal hypoxia has spread rapidly. It is suggested that the main cause for this is due to the input of excess nutrients to the system (Diaz, 2001). Global warming may accelerate these effects and enlarge the affected areas.

Hypoxia impoverishes biological communities resulting in a reduction in species diversity (Bachelet *et al.*, 2000), while anoxia, can result in the release of H_2S from the sediment, causing extensive death of organisms. Even short-lived anoxic events can cause mortality of fish and benthic fauna (Stachowitsch, 1992). Some parts of Mediterranean lagoons become completely depleted in macrofauna, macroalgal and phanerogam species, especially during summer or early autumn (Viaroli *et al.*, 1996; Koutsoubas *et al.*, 2000; Reizopoulou and Nicolaidou, 2004). This situation has also been reported in microtidal salt wedge estuaries like the Ebro (Ibáñez *et al.* 1995). Changes in fish communities and crustaceans in response to hypoxia and anoxia can also render these organisms more susceptible to fishing pressure (Breitburg, 2002).

The decomposition of large amounts of macroalgae contribute significant quantities of dead organic material to the sediments and this often leads to oxygen depletion known as “dystrophic crises” (Sfriso *et al.*, 1992; Viaroli *et al.*, 1996). Another important factor for inducing low oxygen concentrations (hypoxia) in bottom waters is the degree of water renewal. The shallow Mediterranean lagoons with limited water exchange can be very vulnerable in anoxic events during calm and warm periods when stratification occurs, while the tide-dominated systems where tidal mixing reduces the potential for stratification, are less affected.

In most of transitional water ecosystems (excluding deep estuaries), due to the limited depth, most of the processes regulating trophic status are restricted to the benthic system, where sediment-water interactions play a prominent role (Castel *et al.*, 1996). Under these conditions, relationships between production and respiration processes rather than production itself are predictive of trophic status and or trophic evolution.

5.2. Trophic cascade effects

As has been mentioned earlier, transitional waters are particularly important as they are among the most biologically productive areas, heavily exploited by humans. The changes in nutrient loading and species composition are undoubtedly affecting the productivity and sustainability of these systems; therefore it is important to improve our understanding of how an increase in nutrients interacts on the food web, in order to effectively manage transitional water systems.

Many studies illustrate the importance of the interactions of nutrient inputs and food webs in lakes (Carpenter *et al.*, 1985; Rudstam *et al.*, 1993); however for coastal and transitional ecosystems the cascading effects of nutrient addition on the higher trophic levels are much less understood, partly due to the more open nature of these systems and to the more complex trophic structure (Posey *et al.*, 2002).

Nutrient enrichment increases primary production (Nixon, 1995; Pitta *et al.*, 1998) and the cascading effects to the food chain may be variable. In marine pelagic systems nutrient additions generally lead to an increase in phytoplankton biomass. Herbivore biomass is controlled by carnivores; however, there is a weak coupling between phytoplankton and herbivores (Micheli, 1999). An increase in plant productivity may result in an abundance and productivity increase of herbivores; however, predation may mask these responses (Sarda *et al.*, 1996; Pitta *et al.*, 1998).

As discussed above, eutrophication in shallow estuaries and lagoons leads to a replacement of sea-grasses by opportunistic macroalgae, as the dominant benthic autotrophs (Valiela *et al.*, 1997; Raffaelli *et al.*, 1998). The replacement of sea-grass meadows by opportunistic macroalgae may give origin to a new trophic structure (Dolbeth *et al.*, 2003; Cardoso *et al.*, 2004). Eutrophication may also affect herbivores, through harmful algal blooms, since many sources of nutrients, as sewage and animal wastes, agricultural and other fertilizer runoffs, as well as aquaculture, contribute to stimulate these blooms (Anderson *et al.*, 2002).The changes in primary producers can affect the production of other trophic levels (Beukema *et al.*, 2002). An increase of snail mortality caused by consumption of nutrient-induced cyano-bacterial blooms, was observed in an estuarine habitat (Armitage and Fong, 2004). With increasing organic enrichment, abundance and growth rates of certain benthic species also increase (Tsutsumi 1990), while changes in the benthic trophic structure, such as loss of burrowing-taxa have been observed (Weston 1990). In the Mondego estuary due to huge productivity of macroalgae, the sea-grass meadows have been severely reduced, causing impoverishment of the whole ecosystem in terms of macrofaunal abundance, biomass and species richness with consequent decrease of secondary production (Lillebo *et al.*, 2007).

Nutrient supply may also affect the upper trophic levels, such as fish and wading bird populations (Cabral *et al.*, 1999); however very few studies have examined top-down and bottom-up control simultaneously (Posey, 1995). Some studies showed that predator exclusion effect was strong for infauna but not for microalgal biomass (Sarda *et al.*, 1998). Improved understanding of consumer-resource dynamics, particularly in such productive systems, would aid management of important upper trophic level fisheries (Micheli, 1999).

5.3. Changes in zoobenthic community

The increased organic load to the sediment leads to changes in the composition and structure of the benthic communities. High levels of organic material cause a decrease in species diversity and biomass of benthic communities. The larger long-lived species are the first to disappear and the communities are dominated by smaller short-lived opportunistic species (Pearson and Rosenberg, 1978).

Lagoons and estuaries are dynamic and complex ecosystems characterised by frequent fluctuations in environmental parameters on a seasonal or even on a daily basis. In this sense, transitional waters are considered as naturally stressed environments. This natural instability discourages the settlement of many organisms, thus, these ecosystems present generally a low number of species and low diversity, which clearly reflect the dominance of certain highly selective species, living under natural stress, that are adapted in these non-strictly marine environments. The number of species and biodiversity are habitat type dependent and also vary due to natural stress reflecting environmental fluctuations (e.g. salinity gradient). Indeed, Reizopoulou and Nicolaidou (2004) found a strong negative correlation between benthic diversity and confinement (sensu Guelorget and Perthuisot, 1983), as instability of environmental conditions increases with increasing isolation from the sea (Fig. 5).

Temporal variations in benthic communities associated with eutrophication phenomena in coastal lagoons have been the subject of numerous studies (Lardicci *et al.*, 1997; Tagliapietra *et al.*, 1998, Dolbeth *et al.*, 2003). According to the above authors, increased organic disturbance results in changes of species number and species composition, increase of opportunistic species, an increase of densities, and in a decline of suspension feeders and carnivores in favour of sub-surface deposit feeders. Deposit feeders play an important role in nutrient regeneration and may significantly contribute to the nutrient balance of transitional waters.

The sedentary infauna is more affected than vagile organisms, such as crustacea and fish, which are able to migrate. Recovery is generally rapid as organisms soon recolonize the affected areas (Fig. 6).

The development of indicators as a tool for ecological status assessment and for the protection of biodiversity in transitional systems has been encouraged due to the

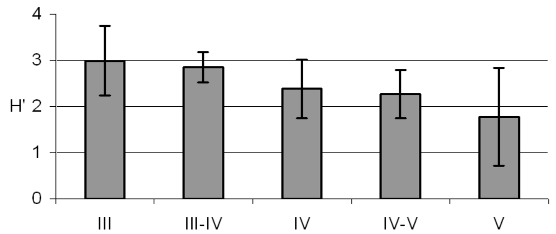


Figure 5. Benthic diversity against confinement in six Mediterranean lagoons (Reizopoulou & Nicolaidou, 2004).

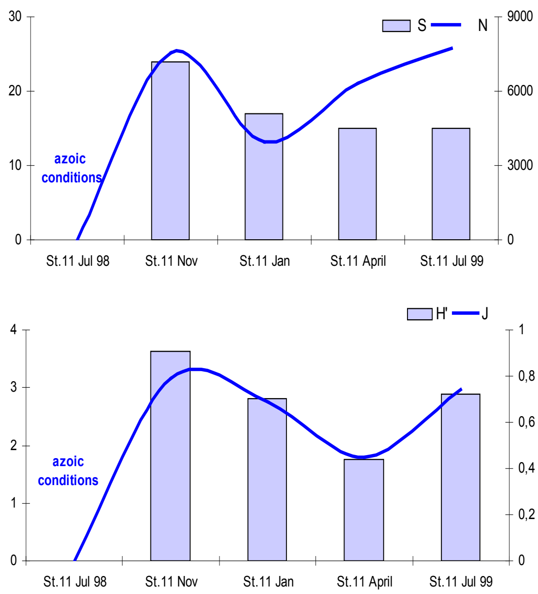


Figure 6. Annual variation of zoobenthic descriptors (N=no of individuals; S=no of species) in a coastal Mediterranean lagoon. The figure shows the recovery of benthic communities after a dystrophic crisis (Reizopoulou and Krasakopoulou, 2000).

implementation of WFD. However, since transitional waters are systems with strong environmental gradients, in order to assess the ecological status, restrictions such as the dependency of the species richness and diversity indices on these natural gradients need to be taken into account. Taxonomic free indices such as body size, abundance distribution among functional groups, functional diversity and productivity can prove to be more effective and relevant for these ecosystems (Mouillot *et al.*, 2006).

5.4. Changes in fish community

Seagrass beds are necessary to sustain carrying capacity of transitional waters, since they are an essential fish habitat. In the shallow lagoons and estuaries, beds of eelgrass comprise an important fish and shellfish habitat.

Eutrophication mainly affects fish communities by changing the general qualities of their recruitment habitats. The degradation of the quality and quantity of eelgrass systems by nutrient loading strongly affects juvenile fish communities and causes a dramatic decline of fish abundance. Seagrasses support juvenile fish offering protection and food resource.

As has been mentioned earlier, hypoxic conditions impoverish biological communities resulting to a reduction in species diversity. Changes in fish communities and crustaceans in response to hypoxia and anoxia can also render these organisms more susceptible to fishing pressure (Breitburg, 2002). Anoxic conditions and release of hydrogen sulphide can cause massive mortality of fish and shellfish communities.

6. Setting up indicators and class boundaries in TW

The general objective of this Section is to review the main indicators that have been recently proposed to assess eutrophication and in a more complete sense, the ecological status in TWs and the definition of class boundaries proposed for each methodology. However, due to the limited amount of space and time available, this review cannot be considered as exhaustive. In addition, as the WFD approaches implementation deadlines, new methodologies are being developed, validated and intercalibrated within Member States. This is particularly true for TW where there is a considerable delay in comparison with other aquatic ecosystems.

As has already been indicated in the Introduction, since these systems are naturally stressed - the TW Quality Paradox-, the extension of the application of indicators developed for other ecosystems is not straightforward (Blanchet *et al.*, 2007; Dauvin, 2007).

A note of concern should be added here: It appears that WFD has been the catalyst for the development of new indicators and in the last years a considerable number of methodologies, indices metrics and evaluation tools have appeared, which complicates the task of selecting suitable methodologies. Unfortunately, it seems to have gone unnoticed that before embarking in the development of another indicator, existing ones should be reviewed. Finally, even though one methodology could be slightly superior to another, socio-economic considerations could make it economically non-viable or administratively unachievable, e.g. small administrations or consultant companies do not have an easy access to expert taxonomic identification.

6.1. Eutrophication status indicators

There is a very large amount of literature concerning methodologies for assessing the eutrophication in aquatic systems. These can be broadly divided into three categories (Nobre et *al.*, 2005): screening methods, model-based, and mixed approaches (Ferreira *et al.*, 2007b).

Screening methods have been created to provide an assessment of eutrophication status based on few diagnostic physical and biogeochemical variables. Typical examples are the OSPAR common procedure on eutrophication assessment (OSPAR, 2003) and the United States National Estuarine

Eutrophication Assessment (NEEA) (Bricker *et al.*, 1999). Concerning transitional water systems, several screening methodologies have been proposed that include estuaries (Bricker *et al.*, 2003a), Fjords (Stigenbrandt, 2001) and coastal lagoons (Souchu *et al.*, 2000), see Box 6.1.

Transitional water quality and sensitivity to nutrient and OM loadings from watershed have been assessed mainly with water chemistry and phytoplankton indicators, basically referring to the trophic reference system proposed by Vollenweider *et al.* (1992) and Nixon (1995). Furthermore, Vollenweider *et al.* (1998) proposed a trophic index (TRIX) that integrates chlorophyll-a, oxygen saturation, total nitrogen and total phosphorus to characterise the trophic state of coastal marine waters, which is nowadays largely applied to coastal lagoons. TRIX is based on the assumption that eutrophication processes are mainly reflected by changes in the phytoplankton community, which is certainly not the case for shallow coastal lagoons and estuaries (excluding deep estuaries) where both MPB and benthic vegetation are the main components of the primary producer community.

Most often, the issues to be analysed are complex and cannot be resolved by considering only simple variables and linear relationships (De Wit *et. al*. 2001). Nevertheless, one can identify a set of basic variables that are indicative of ecosystem properties and functions, which can be easily measured and can be applied for classification and assessment of sensitivity to external stressors (Viaroli and Christian, 2003; Viaroli *et al.*, 2004). Among these the authors considered morphometric parameters, hydrological variables, sediment characteristics and biological elements. Overall, most of these descriptors have

Box 6.1. Methodologies for eutrophication assessment

IFREMER has developed a classification scheme for French Mediterranean lagoons (Souchu *et al.*, 2000). The scheme is based on the identification of physical, chemical and biological potential indicators of eutrophication in the various compartments of the lagoon ecosystem: benthic, phytoplankton, macrophytes, macrofauna, sediments and water, see Table.

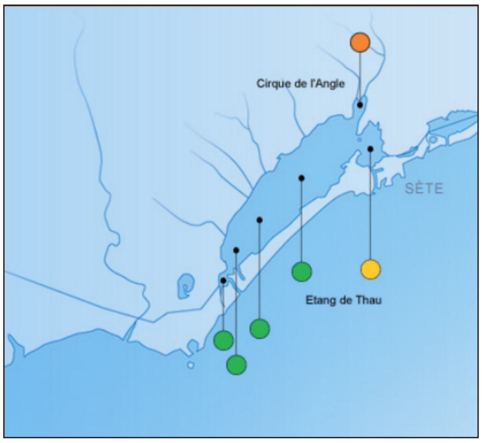
Phytoplankton	Macrophytes	Macrobenthos	Sediment	Water column
Nr. Cells < 2 μm Nr. Cells > 2 μm	Climax species biomass Species diversity	Species richness Population density	Organic matter Total N Total P	Dissolved oxygen Turbidity SRP N–NO2 N–NO3 N–NH4 Chlorophyll a Chlorophylla/phaeo Total N Total P

The methodology allows the classification of a lagoon into five eutrophication levels, formalized by five different colours from blue, signifying no eutrophication to red, signifying high eutrophication, the same colour scheme as used in the Water Framework Directive (WFD). Results of the exercise allow immediate identification of the compartment in which degradation has developed. The compartment with the lowest classification results set the final evaluation of the general ecosystem‘s state. Boundaries for the various components are determined by taking of account the states of the lagoons obtained starting from the diagnosis of eutrophication. For example, concerning the water column, the following table summarizes the elements considered as well as their range of values considering the whole year.

Selected variables of water column for defining the eutrophication status (annual period)

YEAR			BLUE		GREEN		YELLOW		ORANGE		RED
Δ%O2 SAT		0		20		30		40		50	
TUR	NTU	0		10		20		30		40	
PO_4^{3-}	μM	0		0.3		1		1.5		4	
diN	μM	0		15		20		40		60	
NO_2 –N	μM	0		0.5		1		5		10	
NO_3 –N	μM	0		7		10		20		30	
NH_4 –N	μM	0		7		10		20		30	
Chl- a	$\mu g l^{-1}$	0		5		7		10		30	
Chl- a+ phaeo	$\mu g l^{-1}$	0		7		10		15		40	
TN	μM	0		50		75		10 0		12 0	
TP	μM	0		1		2		5		8	

This methodology is currently applied by CEPRALMAR to the Languedoc_Roussillon French region (http://rsl.cepralmar.com/bulletin.html)



Application Ifremer eutrophication classification scheme to Thau lagoon for 2005 (from: http://rsl.cepralmar.com/sites/c07/2005.html).

A hybrid appr4oach for eutrophication assessment called ASSETS (ASSETS stands for Assessment of Estuarine Trophic Status, http://www.eutro.org) has also been developed jointly by National Centers for Coastal Ocean Science (NOAA) and the Portuguese Institute of Marine Research (IMAR), see Bricker *et al.*, (2003b), Nobre *et al.* (2005); Ferreira *et al.* (2006) and (2007a). The methodology includes quantitative and semi-quantitative components to link Pressure-State-Response (PSR) indicators. It consists of (Bricker *et al.*, 2003b): a heuristic index of pressure (Overall Human Influence), a symptoms-based evaluation of state (Overall Eutrophic conditions), and an indicator of management response (Definition of future outlook).

- Overall Human influence (OHI): This is based on a simple conservative mass balance for dissolved inorganic nitrogen (DIN) assuming steady-state conditions, i.e. d[DIN]/dt =0 and taking into account the nutrient inputs from the watershed (human influence) and the exchange with the open sea.
- Overall Eutrophic conditions: Six state parameters have been selected by the authors: Chlorophyll-a, ephiphytes, macroalgae, dissolved oxygen, submerged aquatic vegetation and nuisance-toxic blooms, which its corresponding values and limit concentrations.
- Definition of future outlook: They consider the susceptibility to euthrophication based on the capacity of the system to dilute and/or flush nutrients, and demographic projections. However, the authors point out that this would need more research considering other factors such as trends in agricultural practices, WWTPs, activities in the system, e.g. aquaculture

been implemented for deep to relatively deep aquatic systems, therefore they have to be further calibrated and validated for shallow coastal lagoons and estuaries using the “weight-of-evidence” approach. In order to bring together information from multiple indicators, metrics that allow integration or combination of multiple variables will also greatly improve the capacity of representing ecological status or sensitivity to a given stressor (Viaroli *et al.*, 2004).

The mean depth is an indicator of the development of the water-sediment interface with respect to the water volume. The freshwater discharge to recipient volumes ratio could be either a measure of freshwater flushing or a proxy of sensitivity to pressures from watershed (see Section 3). According to Vollenweider (1992) the combination of hydrological variables with loading estimates gives an assessment of lagoon susceptibility to eutrophication. However, this approach is valid only for deep aquatic ecosystems dominated by phytoplankton communities. Recently, water budgets and nutrient loadings have been widely used for assessing the net ecosystem metabolism (NEM) of coastal lagoons with a wide array of primary producer communities (Giordani *et al.*, 2005; 2008a). Basically, NEM is calculated with the LOICZ biogeochemical model from P loadings; thus NEM gives a measure of the trophic status and of its dependency on nutrient delivery from watershed. A version that considers also the sediment component in the LOICZ models has been recently developed (Giordani *et al.*, 2008).

The susceptibility to eutrophication is not a simple function of nutrient loadings, but also depends on sedimentary processes that are mainly controlled by a suite of sedimentological and geochemical variables. Granulometry and sedimentary organic matter provide basic information on sediment composition and are important determinants of exchanges of oxygen and nutrients across the water-sediment interface. The sedimentary carbonate content can be used to assess the capacity of the sediment to retain phosphate through the calcium/carbonate/phosphate system (Golterman, 1995; Rozan *et al.*, 2002). Reactive iron provides an indication of the sediment capacity to buffer against sulphides and phosphate (De Wit el al., 2001). Yet, the above biogeochemical variables are also related to benthic vegetation and, to some extent, the macroalgae to phanerogam ratio could be a proxy of the dominance of these key biological elements.

A metric to assess production to respiration ratios/relationships in shallow aquatic environments has been presented by Rizzo et al. (1996) - the Benthic Trophic Status Index (BTSI) – and Viaroli and Christian (2003) - Trophic Oxygen Status Index (TOSI). Both are designed to provide classification of benthic systems relative to their potential for heterotrophic and photoautotrophic activities as measured through hourly oxygen uptake in the dark (CR, community respiration) and production or uptake at light saturation (NCPmax, net community production). The regulation of primary production and respiration processes in transitional water ecosystems with different groups of primary producers can be better highlighted with the graphical representation of the TOSI (Fig. 7). Here, one can recognise that macroalgae can greatly amplify production and respiration processes, thus inducing perturbations and within-system instability.

Sedimentary variables can be integrated with water quality using simple metrics, namely the lagoon quality index (LWQI), which integrates oxygen saturation, dissolved inorganic phosphorus (DIP) and nitrogen (DIN), phytoplankton chlorophyll-a, macroalgal and seagrass coverage. LQWI (Giordani *et al.*, 2008b) is a modified version of the WQI (McClelland, 1974), in which utility functions and weight criteria are used to transform measured variables into quality

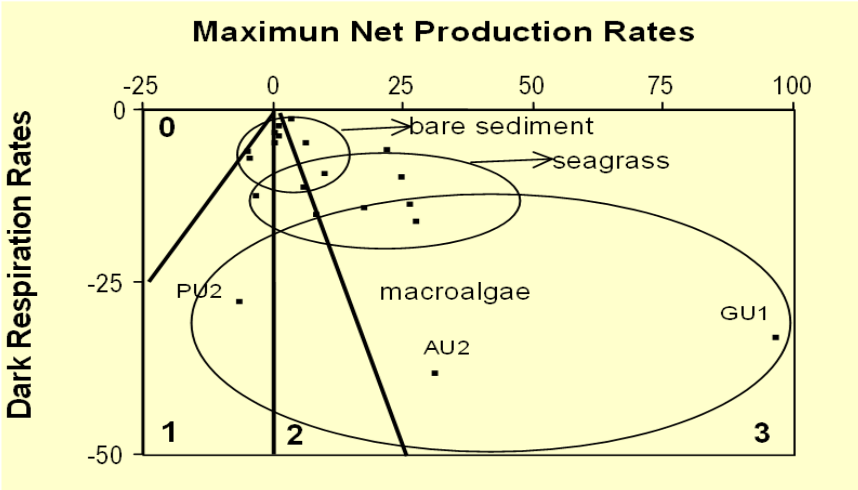


Figure 7. Relationships between net community production at light saturation (NCPmax, mmol O_2 m^{-2} h^{-1}) and dark community respiration (CR, mmol O_2 m^{-2} h^{-1}) in Mediterranean lagoons with bare sediments, dominated by macroalgae or seagrasses. The extent of oxygen production and consumption reveals a symptom of disturbance within the ecosystem. A large excess of oxygen production is associated to the retention of a large amount of organic matter (Viaroli *et al.*, 1999).

scores.

With the increase of computer power, model-based methodologies have been developed. Normally, they are based on a hydrodynamic model that incorporates a biogeochemical model that considers the dynamics of organic and inorganic nutrients (Lancelot *et al.*, 1997; Skogen *et al.*, 2004). Normally, such models are site specific; therefore they are not generally applicable. However, they are useful tools to analyze the environmental responses to changes in pressures as well as to provide environmental managers with an approximate idea of the time constants of their system. In addition, they may be used for scenario analysis of environmental options (Marinov *et al.*, 2007; Aliaume *et al.*, 2007 and references therein) as well as, when coupled with socio-economic models, decision support systems (Loubersac *et al.*, 2007; Mocenni *et al.*, 2008).

Hybrid or mixed approaches try to combine the screening methods with simplified model-based approaches in order to develop general tools that have the advantages of both approaches in terms of applicability and predictive power. For TW several approaches of this type have been developed. Nobre *et al.* (2005) combined the ASSETS (Assessment of Estuarine Trophic Status) screening model (see Box 6.1) with an ecological model to analyze the results from several scenarios or for defining homogeneous water bodies in estuaries (Ferreira *et al.*, 2006b). A similar approach was followed by Giordani *et al.* (2008b) for Sacca di Goro, were a modified version of the Water Quality Index was coupled with a biogeochemical model and interfaced with a Decision Support System (Mocenni *et al.*, 2008).

6.2. Assessing TW ecological status

The WFD has developed the concept of Ecological Quality Status, which is based on the evaluation of the biological quality of different compartments in the ecosystem. Specifically, for TW these are: phytoplankton, macrophytes, benthos and fish. In addition for transitional (and coastal) waters the WFD identifies five general chemical and physicochemical elements supporting the biological quality elements: transparency, thermal conditions, oxygenation conditions, salinity and nutrient condition.

6.2.1. *Phytoplankton*

Transitional waters show great internal patchiness and heterogeneity which can either amplify or bias the stressor effects when considering indicators that have large spatio-temporal variability such as phytoplankton. For these reasons, in addition to historical approaches based on Chlorophyll-a, there has been little development of phytoplankton indicators for these systems until recently where the MED-GIG working group has starting to develop some common approaches.

Recently, Mouillot et al. (2006) have proposed alternative approaches to taxonomic-based indices based on body size or size spectra. However, for phytoplankton, even though there is evidence of the variation of phytoplankton size structure as a function of environmental conditions, there is no methodological standardization yet for its application as an ecological status classification (Mouillot *et al.*, 2006).

6.2.2. *Macrophytes*

Pristine TW systems are considered as dominated by extensive meadows of seagrass species, which are assumed to take advantage of nutrient supply from sediment. An increasing nutrient input is thought to favour, in a first phase, phytoplankton and/or epiphytic micro- macroalgae as well as opportunistic ephemeral macroalgae that coexist with seagrasses. In a second step a regime shift can occur where seagrass is replaced by opportunistic macroalgae such as *Ulva* sp, *Enteromorpha* sp, etc. It has been established (Valiela *et al.*, 1997; Taylor *et al.*, 1999; a.o.) that the primary cause of these shifts and succession in the macrophyte community are nutrients added to water, mainly nitrogen and phosphorus.

With this conceptual schema in mind, several indicators have been developed. Some of them are based on information about species composition and try to distinguish between those tolerant to adverse environmental conditions from those that are less tolerant (Orfanidis *et al.*, 2001, 2003). However, as has been argued by Arévalo *et al.* (2007), in some cases these indices are not able to detect intermediate pollution levels. Other proposed approaches still under development (Scalan *et al.*, 2007), are based on the biomass and coverage of opportunistic, macroalgae see Box 6.2.

6.2.3. *Benthic invertebrates*

Benthic organisms are affected by the environmental conditions of sediments as well as the water column. From a general point of view, the main advantages of monitoring benthic organisms are the following (Bilyard, 1987; Dauer, 1993; Warwick, 1993; Weisberg *et al.*, 1997; Paul *et al.*, 2001):

- they are sedentary and have relatively long life-spans, thus they can respond to local environmental conditions and give integrated responses to water and sediment quality variations
- they respond to anthropogenic and natural stress (Paerson and Rosenberg, 1978) and they are sensitive to different kind of pollutants accumulating in the sediments
- they are characterised by different life cycles, trophic roles and different stress tolerance
- some of them are economically relevant

For these reasons they have been used for assessing the biological quality of the marine environment (USEPA, 1990, Elliot, 1994). Several benthic indices and indicators have been developed during the last years (Dauer *et al.*, 2000; Llanso *et al.*, 2002a and 2002b). At a European water policy level macrobenthos indicators as AMBI (Biotic Index), proposed by Borja *et al.* (2000), the factorial method combining AMBI with Shannon diversity and

Box 6.2. Macrophytes indices

Recently, the Ecological Evaluation Index (EEI) has been proposed by Orfanidis *et al.* (2001, 2003). The EEI is a number ranging from 2 to 10, representing the ecological quality status of transitional and coastal waters in five classes, from high to bad. Marine benthic macrophytes (seaweeds, seagrasses) are used as bio-indicators of ecosystem shifts, from the pristine state with late-successional species (Ecological State Group I, ESG I) to the degraded state with opportunistic species (Ecological State Group II, ESG II).
The first group comprises genera with a thick or calcareous thallus, low growth rates and long life cycles (perennials) whereas the second group includes filamentous genera with high growth rates and short life cycles (annuals). Seagrasses were included in the first group, whereas Cyanophyceae and species with a coarsely branched thallus were included in the second group. The evaluation of ecological status into five categories from high to bad includes a cross comparison in a matrix of the ESG and a numerical scoring system. The model can allow comparisons, ranking and setting of priorities at regional and national levels fulfilling the requirements of the WFD.

Ecological State Group (ESG) classification matrix for seaweed and seagrasses. The matrix scores the mean abundance (%) of ESGs to determine the ecological status of transitional and coastal waters, after Orfanidis *et al.* (2001) and (2003)

Mean abundance(%) of ESG II			
>60	**Bad** (2)	**Low** (4)	**Moderate** (6)
>30- 60	**Low** (4)	**Moderate** (6)	**God** (8)
0- 30	**Moderate** (6)	**Good** (8)	**High** (10)
	0- 30	>30- 60	>60
	Mean abundance (%) of ESG I		

A different approach has been proposed by Scalan *et al.* (2007), in this case the idea is to monitor opportunistic macroalgal blooms, as Ulva and Enteromorpha, typical of degraded transitional water systems. The authors have proposed a preliminary classification as well as discussing several methods for assessing spatial cover.

Proposed classification for opportunistic macroalgal as a function of biomass and spatial coverage (Scanlan *et al.*, 2007)

Biomass					
>3000 gww m^{-2}	**Moderate**	**Poor**	**Bad**	**Bad**	**Bad**
>1000 to 3000 gww m^{-2}	**Moderate**	**Moderate/Poor**	**Poor**	**Bad**	**Bad**
500 to <1000 gww m^{-2}	**Good**	**Good/Moderate**	**Moderate**	**Poor**	**Poor**
100 to <500 gww m^{-2}	**High/Good**	**Good**	**Good**	**Good/Moderate**	**Moderate/Poor**
<100 gww m^{-2}	**High**	**Good**	**Good**	**Good/Moderate**	**Moderate**
	<5%	>5 to 15	>15 to 25%	>25 to 75%	>75 to 100%
	%cover				

Species Richness known as M-AMBI (Borja *et al.*, 2003, 2004a; Bald *et al.*, 2005; Muxica *et al.*, 2006) and BENTIX (Simboura and Zenetos, 2002) are considered as promising indicators to assess the ecological status of European waters (see box 6.3). They were developed as indicators for marine coastal waters, but their flexibility allows their application to transitional waters.

However, these indicators are based classifying the species according to their response to organic pollution (i.e., the tolerance of various levels of dissolved oxygen) and as discussed above TW are already naturally stressed, therefore, it is not clear to what extent these indicators are able to distinguish between natural and anthropogenic stress (Elliot and Quintino, 2007). Moreover, taxonomic-based indicators are time consuming to analyse and need trained personnel and when compared for coastal lagoons they produce different results (Austoni, 2007).

Other indices that use taxonomic-free attributes such as body-size, abundance distribution among functional groups, functional diversity and productivity could prove to be more effective and relevant for these ecosystems (Mouillot *et al.*, 2006). A new method developed especially for coastal lagoons is a biomass size structure index (Index of Size Distribution - ISD) (Reizopoulou and Nicolaidou, 2007).

Further indicators of either state or health of benthic communities have been implemented by the IOC Study Group on Benthic Indicators (http://www.ioc.unesco.org/benthicindicators), among which relevant synoptic information on benthic faunal condition (e.g. measures of community composition), controlling natural abiotic factors (e.g. sediment organic matter), and levels of contaminants were tested world-wide (Hyland *et al.*, 2000). This approach has also been applied to coastal lagoons using organic carbon as a tracer of stressors against the benthic community (Magni and Tagliapietra, pers. comm.).

6.2.4. *Fish*

Unfortunately, there is little information available concerning fish and fish indicators developed for transitional water systems. Several possible indicators are discussed in Mouillot *et al.* (2006). Functional diversity, defined as the value and range of functional traits of the organisms present in a given ecosystem (Diaz and Cabido, 2001) being one promising approach. The idea is that when environmental constraints increase, functional redundancy or similarity between fish assemblages should increase and, therefore, functional biodiversity would decrease (Mouillot *et al.*, 2006). A measure of fish biomass also suggested seems too problematical to produce any relevant indicator of ecological quality. Nonetheless, the authors propose experiments based on cages to measure mortality and growth of juveniles. Even so, it seems that there is still a considerable amount of work to be performed before producing a relevant indicator.

6.2.5. *Chemical and physico-chemical supporting elements*

The definition of chemical and physico-chemical supporting elements boundaries in relation to the WFD, in contrast to previous approaches, is related to the effects of these chemical and physico-chemical elements on ecosystem health. Therefore, it is necessary to link cause and effect (Devlin *et al.*, 2007).

Also in this case one should consider the high spatio-temporal variability in these systems (see Fig. 8) and monitoring should aim at high frequency data. For example, it has been observed several times when measuring oxygen concentrations during summer that anoxic conditions may occur at night (Icely *et al.*, 2007) which are not observed with traditional monitoring programs.

Concerning oxygen, a recent classification for estuaries has been proposed by Best *et*

Box 6.3. AMBI, BENTIX and ISD

Examples of indices based on diversity values of benthic species for marine coastal and transitional waters are:

- The AZTI marine biotic index (AMBI) (Borja *et al.*, 2000);
- The BENTIX index (Simboura and Zenetos, 2002).
- The ISD index (Reizopoulou & Nicolaidou, 2007).

AMBI has been designed to establish the ecological quality of European coasts, investigating the response of soft-bottom macrobenthic communities to changes in water quality.

Most of the concepts developed within AMBI are based upon previous proposals (Bellan, 1967; Pearson and Rosenberg, 1978; Gray, 1979; Salen-Picard, 1983; Dauer, 1993; Weisberg *et al.*, 1997; Roberts *et al.*, 1998). The first step consists in the classification of the species into five ecological groups (EG) following Glemarec and Hily (1981), Hily (1984), Glemarec (1986) and Grall and Glemarec (1997):

EG I: Species very sensitive;
EG II: Species indifferent;
EG III: Species tolerant;
EG IV: Second order opportunistic species;
EG V: First order opportunistic species.

Then AMBI is calculated as follows:

$$AMBI = \frac{0 \cdot \%EGI + 1.5 \cdot \%EGII + 3 \cdot \%EGIII + 4.5 \cdot \%EGIV + 6 \cdot \%EGV}{100}$$

This allows a continuous classification with values from 0 to 6 (Hily, 1984; Hily *et al.*, 1986; Majeed, 1987). The value 7 is used to indicate that the sediment is azoic.

Classification of site pollution according to Borja *et al.* (2000).

Classification	AMBI	Ecological Quality Status (ECoQ) WFD
Unpolluted	$0.0 < AMBI \leq 1.2$	High
Slightly polluted	$1.2 < AMBI \leq 3.3$	Good
Meanly polluted	$3.3 < AMBI \leq 5.0$	Moderate
Heavily polluted	$5.0 < AMBI \leq 6.0$	Poor
Extremely polluted	Azoic	Bad

A program for calculating AMBI is available free of charge at http://www.azti.es/, along with a continuously updated list of species and their corresponding EG values, currently encompassing over 3400 taxa. This approach was originally developed by Borja *et al.* (2000) and tested on approximately 30 stations along the Basque coastline (North of Spain). As new species have been added its applicability has increased. Presently, it has been successfully applied to different geographical areas and under different impact sources, with increasing user numbers in European marine waters (Baltic, North Sea, Atlantic, Norwegian Sea and Mediterranean, all in Europe, but also in Hong Kong, Uruguay and Brazil). AMBI has also been used for the determination of ecological quality status (EcoQ) in the context of the European Water Framework Directive (WFD).

Although AMBI is particularly useful in detecting time and spatial impact gradients, its robustness could be reduced when only a very low number of taxa (1 to 3) and/or individuals are found in a sample. The same

The same could occur when studying low-salinity locations (e.g. the very inner part of estuaries), naturally-stressed locations (e.g. naturally organic matter enriched bottoms), or some special impacts (e.g. sand extraction, some locations under dredged sediment dumping, or physical impact). In the above mentioned particular cases Borja *et al.* (2004b) recommend the use of AMBI, together with other metrics, in order to obtain a more comprehensive view of the benthic community, also recommending a more detailed analysis and discussion of the results.

The BENTIX Index is a newly developed tool, based on macrozoobenthos of soft substrata, to assess the ecological quality status according to the requirements of the WFD. The zoobenthic species are classified into three ecological groups (EGs) and assigned a score from 1 to 3 according to their response to organic pollution (i.e., tolerance to various levels of dissolved oxygen):

EG I: includes species sensitive to disturbance in general; this group also includes the species indifferent to disturbance.

EG II: includes species tolerant to disturbance or stress, whose populations may respond to enrichment or other source of pollution by an increase of densities (slightly unbalanced situations). Also in this group are included second-order opportunistic species or late successional colonisers with r-strategy;

EG III: includes first order opportunistic species (strongly unbalanced situations), pioneers, colonizers, and species tolerant to hypoxia.

The value is calculated in a similar way as the AMBI index as:

$$BENTIX = \frac{6 \cdot \%EGI + 2 \cdot \%EGII + \%EGIII}{100}$$

or simply Bentix = (6 X %GS + 2 X %GT)/100, merging the groups so that GT represents all tolerant taxa and GS all sensitive taxa.

BENTIX produces a series of continuous values from 2 to 6, being zero when the sediment is azoic (all groups zero). Numerical values between 2 and zero are nonexistent in the scale because if EG I is zero the BENTIX =2 (Simboura and Zenetos, 2002). The values are the reverse to those of AMBI, which assigns the worse status (azoic) to a value of 7. Based on these values Simboura and Zenetos (2002) proposed to define accordingly the values concerning the ecological classification of the Water Framework Directive as indicated in the above classification. The boundaries between classes were set keeping equal distances among classes limited only by the two extremes of the scale (2-6) and were tested using data from various sites with known environmental pressures. The BENTIX index applies to all types of marine soft bottom benthic data.

Classification of EcoQ according to range of BENTIX (Simboura and Zenetos, 2002). Lower limits apply in physically stressed muds.

Classification	Bentix	Ecological Quality Status (ECoQ) WFD
Normal/Pristine	$4\text{-}4.5 \leq$ BENTIX < 6	High
Slightly polluted, transitional	$3\text{-}3.5 \leq$ BENTIX < 4.5	Good
Moderately polluted	$2.5 \leq$ BENTIX < 3.5	Moderate
Heavily polluted	$2 \leq$ BENTIX < 2.5	Poor
Azoic	0	Bad

The BENTIX index was validated with data from Greek marine ecosystems and appears to work successfully (different ecological quality classes corresponding to different stress). Results were independent of mesh size used, but were misleading when based on semi qualitative data from dredges. Further development of this type of environmental tool requires the consensus of scientists in the assignation of species to a particular ecological group. The Geographical Intercalibration Groups (GIGs) for Mediterranean Member States have provided an update on the intercalibration exercise including BENTIX as a tool index to test. Information on its application as well as the species list arranged by ecological groups (EG) can be found at: http://www.hcmr.gr/english_site/services/env_aspects/bentix.html.
Specifically for muds a modified scale for BENTIX is proposed, applying to naturally stressed biotopes, originally set as the muddy bottom habitats (Simboura and Zenetos, 2002) defined according to the Folk sediment nomenclature system (Folk 1974) as sediments of which the fine particles (silt and clay) account for 80% and over (Simboura & Reizopoulou, 2008). Occhipinti *et al.* (2005) tested different approaches for Quality Assessment using the Benthic Community in the Northern Adriatic Sea (Italy) with AMBI and the BENTIX. The contrasting quality levels that emerge from the two indices appear mainly due to the different assignment of the species to the ecological groups, but also to the different number and weight of the ecological groups in the formula calculation. Similar results were found by Austoni (2007) when comparing both indices.

The ISD index is an alternative taxonomic-free method, developed for lagoons, based on the distribution of individuals of benthic communities on biomass size classes. The skewness of the distribution was used as a measure of disturbance. The ISD was applied in three coastal lagoons with different levels of disturbance and classified the lagoons in accordance with abiotic factors and pressure information (Reizopoulou & Nicolaidou, 2007).

Classification	**ISD**	**Ecological Quality Status (ECoQ) WFD**
Normal/Pristine	$-1 \leq ISD < 1$	High
Slightly polluted transitional	$1 \leq ISD < 2$	Good
Moderately polluted	$2 \leq ISD < 3$	**Moderate**
Heavily polluted	$3 \leq ISD < 4$	**Poor**
Azoic	Azoic conditions	**Bad**

ISD was intercalibrated with the other benthic classification metrics in coastal lagoons in Greece (Simboura & Reizopoulou, in press). Results showed that the biotic indices assessment in coastal lagoons may be taken into account only in combination with another functional or taxonomic free index, which is not based on species' ecological theory, such as ISD index.

al. (2007), see Box 6.4. Similarly, values for winter nutrient concentration (DIN), potential primary production and dissolved oxygen levels, following the cause-effect-disturbance scheme adopted by OSPAR, have been proposed for coastal and transitional waters in UK by (Devlin *et al.*, 2007), see Box 6.4. The difference between coastal and transitional waters can be considered as being a correction to apply to the limit values for salinity.

6.3. Can ecosystems thresholds be class boundaries in WFD?

An important concept that has been recognised during the last decades (Scheffer

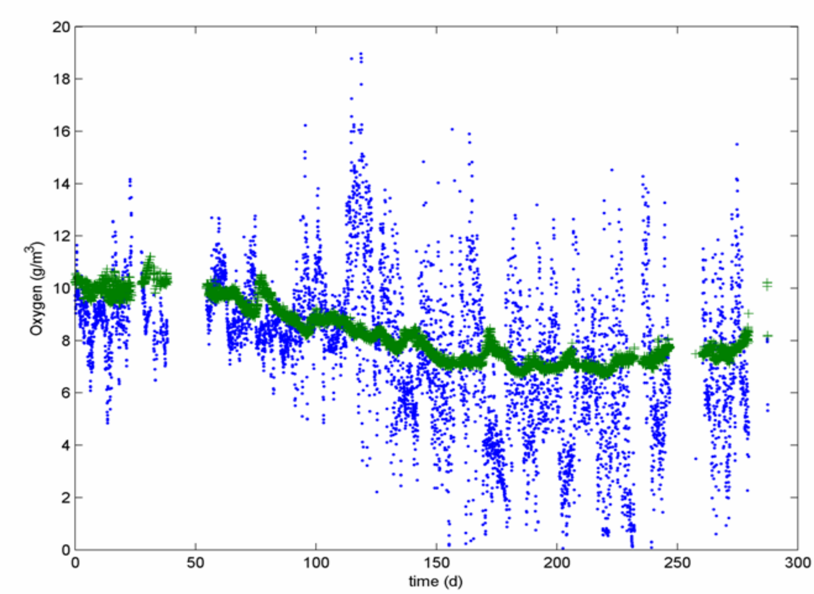


Figure 8. Hourly measurements of oxygen (blue points) and oxygen saturation concentrations (green +) calculated using hourly temperature and salinity measurements in Sacca di Goro during 1999 (Zaldívar *et al.*, 2008).

et al., 2001; De Wit *et al.*, 2001) is the existence of thresholds (a critical value of a pressure beyond which a state indicator shifts to a different regime). Even though the concept of thresholds has been embedded in ecological risk assessment for a long time (Suter, 1993), starting from the dose-response curves to a contaminant, only in the last years has the concept that "a gradual change in pressure would provoke a gradual change in the ecosystem" been modified by the realisation that this gradual change may be interrupted by a sudden and drastic effect in the ecosystem. It has been suggested that the existence of such thresholds be used as a conceptual framework for the development of strategies for sustainable management of natural resources (Mudarian, 2001; Huggett, 2005).

Following Matias *et al.* (2006) it is possible to distinguish between two different frameworks: when analysing the response of an ecosystem to changes in an external factor (control variable) in a continuous way, or when recording an abrupt change in a time series of an indicator of ecosystem status. In the first case, bifurcation theory (e.g. Dingjun *et al.*, 1997) has been developed to classify possible quantitative changes that a dynamical system may experience when one or several parameters are changed. In the second case time series analysis has developed a set of detection methods in different fields, e.g. econometrics, ecotoxicology, oceanography, statistics, etc. These methods may be divided between statistical methods, parametric and non-parametric, e.g. Qian *et al.*(2003), Li and Hunt (2004), model-based , e.g. Cox (1987), Klepper and Bedaux (1997) or time series analysis e.g. detection of abrupt changes in some characteristic property of the series, e.g. Basseville and Nikiforov (1993) and Zeileis *et al.* (2003), amongst others. For a complete review on the statistical methods for identification of thresholds, the reader is referred to Andersen *et al.* (2006).

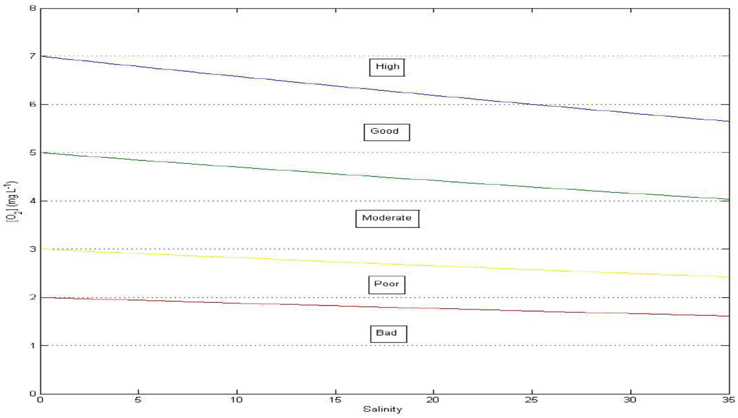
A general database on thresholds or regime shifts in ecological and social-ecological system can be found at: http://www.resalliance.org. There are several examples of regime shifts or thresholds being exceeded in TW systems. Typical examples are regime shifts in the macrophyte community due to

Box 6.4. Supporting elements in Transitional waters

Proposed WFD class limits for dissolved oxygen (DO) concentrations (mg L^{-1}) at 5%ile, i.e. they should not be below the value 95% of the time (Best *et al.*, 2007).

Fresh water 5%ile	Marine water 5%ile	WFD status
≥7.0	≥5.7	High
≥5.0<7	≥4.0<5.7	Good
≥3.0<5	≥2.4<4.0	Moderate
≥2.0<3.0	≥1.6<2.4	Poor
<2.0	<1.6	Bad

Variation of the oxygen class limits as a function of salinity (Best *et al.*, 2007).



Scheme proposed by Devlin *et al.* (2007) in relation with supporting elements (DIN and DO) and potential primary production:

	Index 1	Index 2	Index 3	Classification
Attribute	Nutrient concentration	Production	Undesirable disturbance	
Description/Units	Mean winter DIN (μM)	Potential primary production (g C m^{-2} y^{-1})	DO (mg L^{-1})	
	≤12 μM	-	-	High
	≤18 μM			Good
	≥30 μM	< 300 gC m^{-2} y^{-1}	-	Good
	≥30 μM	> 300 gC m^{-2} y^{-1}	> 5 mg L^{-1}	Moderate
	≥30 μM	> 300 gC m^{-2} y^{-1}	≤ 5 mg L^{-1}	Poor
	≥30 μM	> 300 gC m^{-2} y^{-1}	≤ 2 mg L^{-1}	Bad

The interested reader in methods for calculating PPP (potential primary production) is refereed to the original paper and references therein. However, the methods refer to phytoplankton which, as already explained when discussing macrophytes, is not in all cases the most important biological compartment in assessing total primary production. The selected values for primary production are based on the Nixon (1995) classification, see Table 5.1, whereas DIN levels are based on CSTT recommendations (1994, 1997). The reference conditions for UK coastal; and transitional waters were also defined (Devlin *et al.*, 2007) as: 20μM for transitional waters (salinity = 25) and 13μM for coastal waters (salinity = 32).

nutrient increase (Viaroli *et al.*, 2008), in benthic communities with hypoxia due to increase of organic matter (Conley *et al.*, 2007), in caged fish farms in terms of organic matter inputs (Holmer *et al.*, 2007), etc.

6.3.1. *Managing under thresholds*

The WFD defines five categories of water quality: High, Good, Moderate, Poor and Bad. However, the most important boundary is between moderate and good. In this case restoration measures have to be taken into account, this implies economic considerations. Assuming that our system responds in a non-linear fashion, then assessing the five categories should take into account that when a threshold point is reached enormous restoration measures would be necessary, or in some cases no remedial actions would be possible. This assumes that the boundaries should be fixed at a value of pressure and ecosystem status as measured by relevant biological metric that gives enough guarantee that the ecosystem is not close to the threshold value for the metric, which in case of strong non-linear response could imply passing from High to Bad as for example in the case of seagrass extinction.

In an ecosystem with thresholds one should define the boundaries between the classes according to the type of response (see Fig. 9). In these circumstances, the common practice to divide the system's indicator in equal segments probably does not hold. In addition, in case of a sharp threshold, the intermediate categories: Moderate and Poor could be difficult to distinguish and probably a temporal dimension should be added during the evaluation to establish the line between Moderate and Poor. Furthermore, the point of non-return should be assessed taking into account also socio-economic considerations.

7. Monitoring

According to the WFD monitoring networks have to be designed "so as to provide a coherent and comprehensive overview of ecological and chemical status within each river basin and shall permit classification of water bodies into five classes consistent with the normative definitions for specific water categories".

The Eutrophication Guidance (interim document; CIS, 2005) includes a chapter on monitoring aiming to specify further issues

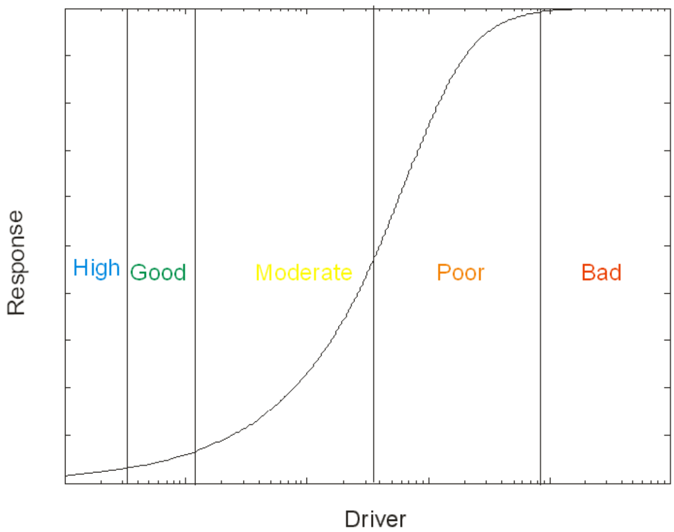


Figure 9. WFD classification under non-linear ecosystem response.

relevant to the assessment of eutrophication and which are only dealt with in general terms in the CIS Guidance on Monitoring (CIS, 2003). As stated in this document (CIS 2003) the exercise is done considering monitoring firmly based on the methodological concept of the WFD.

Thus, here the focus will be only on specific characteristics of TW waters that need to be taken into consideration when designing the monitoring networks for the purposes of the WFD when the main impact is eutrophication.

The WFD prescribes three different monitoring programmes: operational, surveillance and investigative, each with its own objectives and for which different quality elements and sampling strategies are necessary.

Operational monitoring will be carried out for all those water bodies identified as being at risk of failing their environmental objectives (i.e. achievement of good ecological status or good ecological potential, or no deterioration of status). Additionally, also water bodies that have been assessed as eutrophic under other policies will be part of operational monitoring. In this case, these water bodies will be already part of a "sensitive area/water body", or a "polluted water" or a "problem area", respectively, under the Urban Waste Water Treatment Directive, Nitrates Directive and OSPAR Strategy to Combat Eutrophication (in waters of overlapping jurisdiction).

Surveillance monitoring must be carried out in a sufficient number of surface water bodies to provide an assessment of the overall surface water status within each catchment or sub-catchments within the river basin district.

Investigative monitoring shall be carried out when the reasons for failure of the environment objectives are unknown and to ascertain the magnitude and impacts of accidental pollution.

The Directive lays down general technical specifications to ensure the quality/comparability of the monitoring results namely the minimum sampling frequencies of different quality elements (Table 7). The suggested minimum frequencies are applicable to both surveillance and operational monitoring and are generally lower than currently applied in some countries. However, it anticipates that changes to these frequencies may be necessary and chosen so as to achieve an acceptable level of confidence and precision of the monitoring results. More frequent monitoring will most likely be necessary in many cases to achieve a reliable assessment of the status of the relevant quality element, but also less frequent monitoring is justified when based on technical knowledge and expert judgment. Member States are also able to target their monitoring to particular times of year to take into account variability due to seasonal factors.

Monitoring is required over a period of a year once every 4 years for the Nitrates Directive, and the sensitivity of waters in general needs to be reviewed every 4 years for the Urban Waste Water Treatment Directive. The review

Table 7. Minimum sampling frequencies for the biological, hydromorphologica and physico-chemical elements for operational monitoring of transitional waters.

Quality Element	Minimum frequency
Biological	
Phytoplankton	6 months
Other aquatic flora	3 year
Macro invertebrates	3 years
Fish	3 years
Hydromorphological	
Morphology	6 years
Physico-Chemical	
Thermal Conditions	3 months
Oxygenation	3 months
Salinity	3 months
Nutrient Status	3 months
Other Pollutants	3 months
Priority Substances	1 month

does not explicitly require monitoring, though undoubtedly information from monitoring would be invaluable in the assessment. For the Nitrates Directive a minimum of monthly samples for nitrates is required; this compares with once every 3 months (for nutrient status) for the WFD. The OSPAR eutrophication monitoring programme gives different sampling frequencies for problem, potential problem and non-problem areas. For HELCOM there are two main monitoring frequencies recommended: frequent and highly frequent. Frequent sampling ranges from once or twice per year to 6 to 12 times per year depending on purpose and parameter. Some high frequency stations are sampled up to 26 times/year or even more often. For the MEDPOL eutrophication monitoring strategy, the optimal sampling frequency should be chosen by each country according to the parameter variability in the affected area, and with the objective of detecting a change in concentration over a selected period (e.g. 10 years).

In the relevant CIS groups it has been further recognised and agreed that the spatial and temporal monitoring requirements for Eutrophication critical variables for specific water types need to be reconsidered to be able to capture the necessary seasonality in e.g. flow dependency in nutrients and of nutrient loads, chlorophyll and oxygen. This is particularly important with regard to the TWs because of their complexity characterised by a natural high spatial and temporal variability and high productivity.

Each Transitional Water body has special morphological and hydrological characteristics responsible for its highly variable spatial and temporal physico-

Table 8. Monitoring priorities as a function of the different pressures, susceptibilities and states (adapted from Ferreira *et al.*, 2007).

			Monitoring Priority		
Pressure	**Susceptibility**	**State**	**Surveillance**	**Operational**	**Investigative**
High	High	High	Required		High
High	High	Good	Required	High	
High	High	M/P/B	Required	High	
High	Low	High	Required		
High	Low	Good	Required	Medium	
High	Low	M/P/B	Required	Low	
Moderate	High	High	Required		
Moderate	High	Good	Required	High	
Moderate	High	M/P/B	Required	High	Medium
Moderate	Low	High	Required		
Moderate	Low	Good	Required	Medium	
Moderate	Low	M/P/B	Required	Low	Medium
Low	High	High	Required		
Low	High	Good	Required	Medium	
Low	High	M/P/B	Required	Medium	Medium
Low	Low	High	Required		
Low	Low	Good	Required		
Low	Low	M/P/B	Required	High	High

chemical and biological features. These characteristics need to be considered for selection of monitoring stations, biological quality elements (BQE) and supporting quality elements (SQE) to be measured, and for sampling time/ frequency. Recently, Ferreira *et al.* (2007c) have developed a decision tree for selection and prioritization on the type of monitoring programmes, which has been summarized in Table 8.

7.1. Selection of monitoring stations and sampling frequency

The minimum number of monitoring stations for transitional water systems should agree with the number of water bodies previously defined. For these systems it is difficult to prove that two separated water bodies have similar susceptibility and anthropogenic pressures.

In order to meet the requirements of multiple Directives that normally apply to these systems, e.g. Water Framework, Nitrates, UWWT, Habitats Directive, amongst others, an integrated approach should be taken to optimize multiple requirements. In addition, the high natural variability and heterogeneity of TW systems is not likely to be captured with the minimum sampling frequencies indicated in the WFD (see Table 7).

Concerning priority substances, a Chemical monitoring Guidance Document (GD) for surface waters has been recently developed and an interim version is available at the Commission database (Communication and Information Resource Centre Administration) CIRCA (http.//circa.europa.eu/Public/irc/env/wfd/library?1=/framework_directive/chemical_monitoring). This interim version will be amended according to the outcome from the current negotiations and readings of the WFD Daughter Directive on Environmental Quality Standards. The document has been prepared by a drafting group of Member States and chaired by Germany.

In this case, the measured contaminant concentrations will be compared with the defined Environmental Quality Standards (EQS) for compliance checking.

Concerning specifically, transitional waters, there are only two references. The first one mentions the need of reaching an agreement between monitoring approaches, strategies and design that is coherent for both the WFD and Marine Strategy policy. The second one refers to the needs of adapting the sampling frequency established in the WFD for these types of systems.

In a study carried out by IFREMER (Souchu *et al.*, 2000) for French Mediterranean lagoons and confirmed afterwards by Austoni (2007) for other coastal lagoons, different results could be obtained as a function of the sampling frequency and sampling season. Therefore, it is clear that for TW surveillance monitoring will require higher frequencies than the minimum established by the WFD as well as careful considerations on the monitoring time not only seasonal but also concerning the diel (24 hours) variability considering tidal influences. Taking into account these points and the fact that for waters deemed to be at risk from eutrophication the sampling frequency should be higher according to the Nitrates Directive, a proposed frequency for monitoring adapted from Ferreira *et al.* (2007c) is shown in Table 9 for surveillance monitoring. These frequencies, however, should be complemented with some hourly monitoring for assessing the specific variability of a TW water body.

In addition, to ensure that the overall status of a water body is represented by the measures obtained the following aspects (Guidance Document 7, CIS 2003) should be taken into account:

- For shallow water systems salinity and temperature stratification should be assessed (e.g. in Sacca di Goro, Italy, stratification has been observed even at 0.5 m depths, Carafa *et al.*, 2007) before deciding the number of samples to be collected.

Table 9. Proposed surveillance monitoring sampling frequencies for transitional waters (adapted from Ferreira *et al.*, 2007).

Quality Element	Proposed frequency
Biological	
Phytoplankton: - Biomass and abundance - Species composition	 Monthly 6 months
Other aquatic flora	Seasonal
Macro invertebrates	6 months
Fish	Seasonal
Hydromorphological	
Morphology	Site dependent
Physico-Chemical	
Thermal Conditions	monthly
Oxygenation	monthly
Salinity	monthly
Nutrient Status	monthly
Other Pollutants	seasonal
Priority Substances	monthly

- In TW systems with macrophytes as the main primary producer, the water may become anoxic at night and thus daily measured dissolved oxygen concentrations may not be representative. In addition DIP may increase by several orders during anoxic crises (Viaroli *et al.*, 2006).
- In TW systems water concentrations are affected by fresh water flow discharges and by tidal fluctuations, values at high and low tide would provide an impression of the temporal variability. Agrochemical pollutants show a clear temporal distribution with peaks in spring and summer (Carafa *et al.*, 2006), others have only one short peak during the year (e.g. molinate pollution, Zaldívar *et al.*, 2003), whereas no clear seasonality could be expected from Persistent Organic Pollutants (POPs) such as polycyclic aromatic compounds (PAHs).

The development of integrated monitoring networks that combine already existing stations at different levels, e.g. local, regional and national, should be a Member State priority. This is more evident for the case of Transitional Waters where a clear definition of responsibilities is still far behind those for inland and marine waters. The influences from watershed and the adjacent sea should be considered when developing a monitoring network since both will contribute to the final ecological status of the TW system.

7.2. Selection of quality elements/ parameters to be measured

At the quality element level there are many similarities between the different policies, particularly for the biological and physicochemical quality elements that are considered to be symptomatic of eutrophication. However, there are some differences in terms of the recommended measured parameters indicative of the quality elements. More significantly surveillance and operational monitoring for the Water Framework Directive requires the monitoring for hydromorphological quality elements: there is no such explicit requirement in the other relevant policy drivers even though some of these elements are included as supporting environmental factors in the conceptual framework for eutrophication.

Concerning TW, the most exhaustive approach developed for assessing eutrophication (Souchu *et al.*, 2000, see Box 6.1) considers all biological quality elements (see Table 9) except fish, including physico-chemical parameters. However, the monitoring of hydromorphological quality elements is not considered. This is an important aspect that must receive adequate attention in TW systems since hydromorphological quality elements will define the behaviour of the water body in terms of eutrophication.

7.3. Emerging monitoring techniques

The WFD does not elaborate on the methods and techniques to be used for monitoring BQEs or SQEs. Therefore, the development of new monitoring tools able to provide the necessary information for assessing the ecological status of European water bodies at lower costs is a challenging and important task.

The SWIFT-WFD EU funded project (SSPI-CT-2003-502492) developed the concept of a toolbox for WFD monitoring as the strategy to follow when it is necessary to use a range of tools to obtain a better representation of water quality. In this context emerging monitoring techniques may complement existing "traditional" techniques until proper intercalibration is performed

According to Allan *et al.* (2006) monitoring techniques may broadly be divided in chemical and biological, the difference being whether an organism is used or not. Table

Table 10. Emerging monitoring techniques and their applicability (modified from Allan *et al.*, 2006).

Technique	Function	Type of monitoring		
		Surveillance	Operational	Investigative
Biomarkers	Linking chemistry and ecology	x	x	x
	Hot spots identification		x	x
Biological early warning systems	Early warning general and toxicity assessment	x	x	x
	Water quality (e.g. alevin fish cages)	x		x
Bioassays/Biosensors	Cause/effects relationships		x	x
	Screening of toxic compounds	x		x
Passive sampling devices	Long term changes and trends	x		
	Pollutants screening			x
	Pollutants loads	x		
Chemical/ electrochemical sensors	On site mapping		x	x
	Rapid measurements of chemical concentrations		x	x
Immunoassays	*Same as above*		x	x

10 (Allan *et al.*, 2006) summarizes some of these emerging techniques and indicates their possible application according to the monitoring requirements in the WFD. For examples of applications and related literature the reader is referred to the original paper.

8. The DSPIR approach to transitional water systems

8.1 Introduction

The ecosystem approach to management integrates social and natural sciences and includes mankind and human activities as an integral part of the ecosystem with the objective of sustainable exploitation and development of ecosystem resources, services, goods and uses. The OECD Driver Pressure State Impact Response (DPSIR) framework of 1993 includes a core set of indicators for environmental performance reviews, including eutrophication issues. The DPSIR approach is a useful tool that brings together natural science, social science and economics in one framework for adaptive management. It is an example of a System Approach Framework (SAF) that considers human activities as an integral part of the ecosystem. The DPSIR framework can be utilized to analyze individual issues and enables adaptive management, because it is an iterative process.

IMPACTS in the DPSIR framework include SOCIO-ECONOMIC, ECOLOGICAL and ENVIRONMENTAL IMPACTS. The WFD also demands impact assessment through a set of indicators or "Quality Elements". We use the DPSIR framework for the analysis of a core set of indicators, and also consider WFD quality elements within the context of eutrophication in TW environments. The classical DPSIR framework has been expanded to include WFD quality elements and to show how ecological impacts are linked to the socio-economic impacts. This approach thus identifies which ecosystem goods and services are impaired by eutrophication and how human socio-economic activities are affected.

The aims of this Section are:

1. to review and analyse applications of the Driver- Pressure-State-Impact-Response (DPSIR) framework to eutrophication in TW systems; and,
2. to further develop the DPSIR framework in the context of the WFD.

The definitions adopted for eutrophication and Transitional Waters in this Section come from the UWWT Directive and from the WFD, respectively (see Sections 1 and 2).

8.2 The Driver-Pressure-State-Impact-Response (DPSIR) framework and Eutrophication

Within the DPSIR framework, DRIVERS are: socio-economic activities, such as urban development; PRESSURES are the resulting environmental stressors, such as increase nutrient runoff; STATE are the environmental conditions and their metrics, such as dissolved oxygen and nutrient concentrations; IMPACTS are both ecological, such as loss of biodiversity, and socio-economic impacts, such as loss of fishing jobs; RESPONSES are societal reactions to environmental and ecological degradation, such as new management criteria, new infrastructure investment in water treatment or new policies and legislation.

The DPSIR framework was first used in 1993 with a "Core Set of Indicators for Environmental Performance" proposed by the Organisation for Economic Cooperation and Development (OECD, 1993) and it was applied to Eutrophication in both inland and marine waters. The main PRESSURE was identified as the emission of Nitrogen (N) and Phosphorus (P) into water and soil resulting from the use of fertilizers, waste water discharges and livestock density. The metrics for the environmental conditions or STATE were identified as B.O.D. (Biochemical Oxygen Demand), D.O. (Dissolved Oxygen), as well as Nitrogen (N) and Phosphorus (P) in water. The societal RESPONSE was to increase the percentage of the population linked to sewage treatment, including both biological and chemical treatment, charge for water treatment and increase the market share of phosphate-free detergents. Interestingly, none of these responses addressed the PRESSURES that have been identified from fertilizer use or livestock (non point sources). However, the European Community recognized this problem and introduced the Nitrate Directive, a societal response to the agricultural driver. An assessment of the effectiveness of the Nitrate and other Directives was carried out by the European Environment Agency in the late 1990's (EEA, 1998). This assessment showed that considerable progress had been made on phosphate emissions attributable to the UWWT Directive (European Commission, 1991), however no real progress had been made on the Nitrate emissions. This highlighted the need for a more comprehensive water policy. The introduction of the Water Framework Directive itself can therefore be regarded in part as a societal RESPONSE to the DRIVERS of eutrophication.

In 2001, the EEA published a report that included a DPSIR framework for eutrophication in coastal, rather than transitional water. This is shown in Figure 10.

The main driving forces again included agriculture and sewage effluent from treatment plants but "new" DRIVERS are identified, such as industry and traffic. Nevertheless, the RESPONSES are still limited to water treatment, now including industrial effluent. Measures to protect public health from algal toxins are also included. However, no specific response is proposed for addressing the agricultural drivers.

In 2003 the EEA published a Topic report on Europe's water: "An indicator-based assessment". The DPSIR framework was used extensively in this report to assess aquatic ecological quality, which is central to the WFD. The STATE includes the Chemical and Biological Quality Elements. One of the IMPACTS mentioned is "Transitional water of less than good quality". The societal

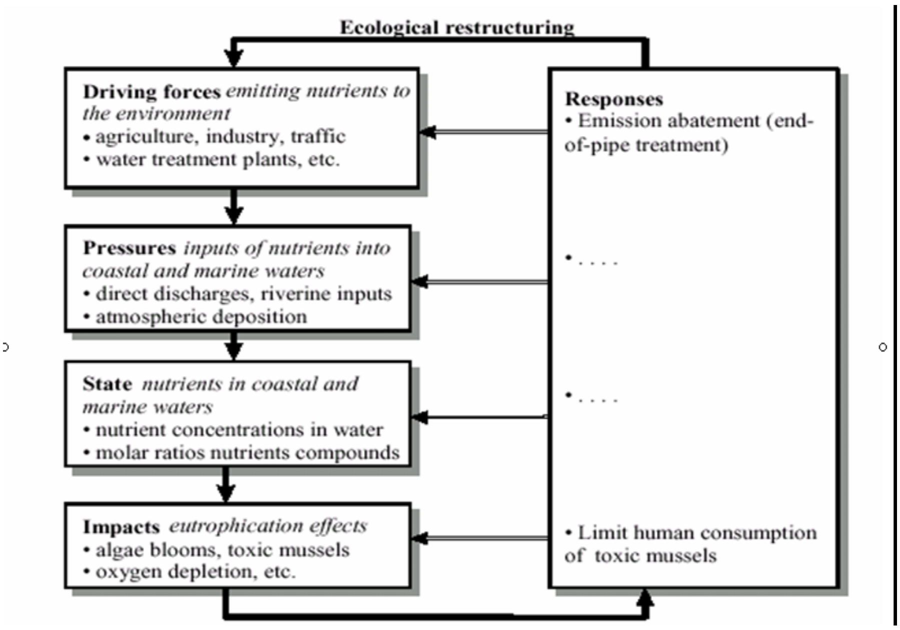


Figure 10. The EEA DPSIR framework applied to eutrophication in coastal waters (source EEA).

RESPONSE is the implementation of the WFD, which requires the integration within the programme of measures of RESPONSES contemplating the main DRIVERS. The DPSIR framework is also used in this report for assessing eutrophication and pollution from organic matter. The usual DRIVERS are identified as "industry, agriculture livestock density and fertiliser use, households and waste water treatment". The identified IMPACTS are "exceedance of standards for drinking water and bathing waters, Secchi depth in lakes, low oxygen in bottom layers of marine waters, and harmful phytoplankton in coastal waters" Some of these may be better as STATE descriptors, such as Secchi depth, and low oxygen. Exceedance of standards for drinking water and bathing waters is a SOCIO-ECONOMIC IMPACT, because it may be necessary to introduce expensive drinking water treatment, for example. There may also be a loss of tourism revenues because of poor bathing water quality. The occurrence of harmful phytoplankton is an ECOLOGICAL IMPACT.

In 2004, the OECD published an update of Key Environmental Indicators. However, this only contemplates inland waters and the PRESSURES and conditions (STATE) as well as RESPONSEs remain the same. Still no response is proposed to address pressures resulting from agricultural drivers.

8.3 The DPSIR and the WFD

The WFD does not refer to the DPSIR framework approach. However, the WFD

Box 8.1. Examples of socio-economic aspects of the WFD and ecosystem services

Public information and involvement in decision making

…to ensure the participation of the general public including users of water in the establishment and updating of river basin management plans, it is necessary to provide proper information of planned measures and to report on progress with their implementation with a view to the involvement of the general public before final decisions on the necessary measures are adopted.

Economic instruments

…the use of economic instruments by Member States may be appropriate as part of a programme of measures. The principle of recovery of the costs of water services, including environmental and resource costs associated with damage or negative impact on the aquatic environment should be taken into account in accordance with, in particular, the polluter-pays principle.

…an economic analysis of water services based on long-term forecasts of supply and demand for water in the river basin district will be necessary for this purpose. Economic and social development of the Community as a whole and the balanced development of its regions as well as the potential benefits and costs of action or lack of action.

…by 2010 MS will ensure…an adequate contribution of the different water uses, disaggregated into at least industry, households and agriculture, to the recovery of the costs of water services, based on the economic analysis conducted according to Annex III and taking account of the polluter pays principle.

Ecosystem services

…protection of water status within river basins will provide economic benefits by contributing towards the protection of fish populations, including coastal fish.

does link the natural system to the socio-economic system. In particular the DPSIR approach is relevant to the WFD because this Directive requires public information and involvement in decision-making as well as the use of economic instruments. It also links the benefits of protecting water status with economic benefits such as the protection of fisheries, including coastal fish. The WFD therefore recognizes important ecosystem services of transitional waters, such as their role as nursery grounds for juvenile fish. See box 8.1 for relevant extracts of the WFD focusing on socio-economic aspects and ecosystem services.

Within the context of the WFD, the STATE corresponds to the metrics of the ecological status. These include hydromorphological and physico-chemical elements supporting the biological quality elements.

The hydromorphological elements, such as depth variation, qualitative and quantitative structure of the bed substrate, structure of the intertidal zone, tidal regime, freshwater flow and wave exposure, are particularly important with respect to eutrophication because they may alter the residence time and flushing of the transition zone. Nevertheless, human activities such as dredging that may be used to maintain the hydrodynamics of the transition zone may also have a side-effect of releasing more nutrients that are stored in the sediment into the water column. When the sediment is anoxic, the dissolved oxygen in the water column may also be depressed while the release of some nutrients, such as phosphate increased.

The physico-chemical elements supporting the biological quality elements include transparency, thermal conditions, oxygenation conditions, salinity and nutrient conditions. All these are important in understanding eutrophication. Transparency affects all photosynthetic organisms, while temperature and salinity are master environmental variables in transition waters controlling the structure and function of estuarine environments and the ecology of these ecosystems. With respect to the metrics of these physico-chemical elements supporting the biological quality elements, some are evident, such as temperature range for thermal conditions. However, two of the physico-chemical elements supporting the biological quality elements are oxygenation conditions and nutrient conditions. These should be considered carefully with respect to the eutrophication of transition waters because of lack of agreement on the threshold criteria and wide variability. The usual metric for oxygenation condition is dissolved oxygen (DO) concentration. This is particularly applied when considering whether an environment is hypoxic or anoxic. Unfortunately there is no clear consensus about the thresholds that define hypoxic and anoxic conditions. The definitions given in Box 8.2 have been used by various authors.

Hypoxic conditions may be episodic, annual, periodic or persistent and these are distinguished by the occurrence and the duration of time that the hypoxic conditions persist. Temperature and salinity are the main environmental variables that control the dissolved oxygen solubility. Temperature and salinity therefore exert some control on another physico-chemical element supporting the biological quality elements, the oxygen condition. The oxygen condition of a transitional water is fundamental to the ecosystem metabolism of the estuary, since all aerobic organisms require oxygen for their metabolisms. Should the environment become depleted in oxygen, this may cause mass mortalities of most oxic species and their replacement by species tolerant of hypoxic or anoxic conditions, in the extreme case the whole

Box 8.2. Definitions of anoxia and hypoxia (List presented by N. Rabalais, SCOR WG 128, IMBER-LOICZ Continental Margins Science Conference, Shanghai Sept. 2007).

Hypoxia < 100% O_2 saturation
Hypoxia < 5 mg l^{-1}
Hypoxia < 3 mg l^{-1}
Hypoxia < 2 mg l^{-1}
Hypoxia < 2 ml l^{-1}
Hypoxia < 0.5 ml l^{-1}
Hypoxia ~ 30% saturation
Hypoxia < 63 mM
Anoxia < 0.05 ml l^{-1}
Anoxia = 0 ml l^{-1} or mg l^{-1} or 0%

ecosystem may switch to an anaerobic metabolism.

8.4. The use of the DPSIR approach for eutrophication in transitional waters in the context of the WFD

The DPSIR approach is very useful when applied to eutrophication in transitional waters. Some adaptations are necessary to use the DPSIR framework in the context of the WFD. There is some confusion with respect to the use and applicability of WFD terms and their attribution to STATE and IMPACT. Examples of the physico-chemical elements and the phytoplankton biological quality element (BQE) are shown in Tables 11 and 12. In this table, the metric of STATE is a measurable or quantifiable variable used to assess the STATUS of each quality element. Similar Tables can be drawn up for the remaining BQE: aquatic plants, benthos and fish.

8.5 The evolving concept of eutrophication

The concept of eutrophication in the 1990's was limited to a simple "cause-effect" relationship. The main causes or DRIVERS were considered to be sewage, animal wastes, agricultural fertilizers and detergents. These increased the PRESSURE of nutrient and organic matter loading of the transitional water. The effect could be felt as a change in STATE, such as decreased oxygen or an ecological IMPACT, such an algal bloom.

In 2001, Cloern published a paper on "our evolving conceptual model of the coastal eutrophication problem" much of which is applicable to transitional waters. In this paper, Cloern lists as "Direct responses" to eutrophication: chlorophyll; primary production; macroalgal biomass; sedimentation of organic carbon; system metabolism; phytoplankton community; Si:N; N:P; oxygen and Harmful Algal Blooms (HABs). Furthermore, Cloern (2001) lists as "Indirect responses" to eutrophication: benthic biomass; pelagic biomass; vascular plants; habitat diversity; water transparency; organic carbon in sediments; sediment biogeochemistry; bottom-water oxygen; seasonal cycles; mortality and biodiversity. Not all these are relevant in the context of the WFD. Some are more relevant to other Directives such as "Habitat Diversity" and the Habitat Directive.

Cloern's paper was published in 2001 and global change has continued to shape our concepts of eutrophication. As society responds to DRIVERS some of these become less important, as has been the case for detergents and phosphate, because of successful policy intervention. However other "new" DRIVERS may become increasingly important, and should be considered in the DPSIR framework. Changing lifestyles, global change, changing land use as well as port and shipping activities have led to "new" DRIVERS of eutrophication in addition to agriculture, industry and domestic sewage.

Several DRIVERS result from changing lifestyles. These include population growth,

Table 11. WFD terminology in the context of the DPSIR framework applied to eutrophication of transitional waters (STATE, metrics and IMPACT examples for the physico-chemical elements).

Physico-chemical elements supporting the biological quality elements	
Transparency	
Environmental STATE	Transparency of the water The opposite variable, turbidity, is often used
Metric of STATE	Secchi disk depth, Total suspended solids
Environmental IMPACT	Increase or decrease in transparency
Relevance to eutrophication of transitional waters	High: Photosynthetic micro-organisms, such as phytoplankton, and other plants need light to photosynthesise. Phytoplankton blooms may also decrease water transparency. This is particularly problematic in physico-chemical elements in naturally turbid transitional waters that may reach a turbidity maximum.
Thermal condition	
Environmental STATE	Temperature range of the water
Metric of STATE	Water temperature
Environmental IMPACT	Increase or decrease of temperature
Relevance to eutrophication of transitional waters	High: Oxygen solubility decreases with increasing temperature. Microbial metabolism, including oxygen demand increases with temperature. Transitional waters may have a thermal front
Oxygenation condition	
Environmental STATE	Range of dissolved oxygen concentrations and saturation
Metric of STATE	Dissolved oxygen concentration and percentage saturation measurements should be taken at dawn for early diagnosis of the onset of problems
Environmental IMPACT	Changes in dissolved oxygen concentration and percentage saturation
Relevance to eutrophication of transitional waters	High: Eutrophic systems may be hypoxic or even anoxic (see box 2.2 above). There maybe nigh time or seasonal oxygen hypoxia. In daylight hours, supersaturation of oxygen may be observed.
Salinity	
Environmental STATE	Range of salinity
Metric of STATE	Salinity is derived from conductivity measurements
Environmental IMPACT	Change in salinity This is particularly problematic in transitional waters, especially tidal transitional waters
Relevance to eutrophication of transitional waters	High: Oxygen solubility decreases with increasing salinity

though not important in the present European scenario, but also changing demographics. This takes account of both intra-European and extra European immigration. It also includes increasing urbanization of the coastal zone both for permanent residents as well as tourism, which cause seasonal PRESSURES. Changes in wealth and diet that follow the

Table 12. WFD terminology in the context of the DPSIR framework applied to eutrophication of transitional waters (STATE, metrics and IMPACT examples for phytoplankton biological quality element)

Biological quality element Phytoplankton	
Phytoplankton: biomass, abundance and composition	
Environmental STATE	Biomass, abundance and composition are described individually in the following tables
Metric of STATE	
Environmental IMPACT	These maybe positive, when there is an improvement in biological status, or negative if there is a deterioration
Metric of IMPACT	Change in Ecological Quality Ratio Increase in EQR reflects a positive improvement Decrease in EQR reflects a deterioration
Relevance to eutrophication of transitional waters	High: Phytoplankton distribution is affected by the density structure of water, which may cause particular problems of interpretation in transitional waters where phytoplankton cells may accumulate naturally at fronts between fresh and saline water. In tidal transitional waters, this is further complicated. Nutrient enrichment may stimulate algal growth and lead to the occurrence of algal blooms The species composition may be altered by unbalanced nutrient ratios favouring nuisance species, such as some cyanobacteria, or *Phaeocystis* and even toxic species such as *Alexandrium* and some cyanobacteria.
Phytoplankton Biomass	
Environmental STATE	Chlorophyll-a as a proxy of phytoplankton biomass
Metric of STATE	Chlorophyll-a concentrations, in particular the 90th percentiles The methodology (molecular absorption spectroscopy or fluorimetry or HPLC) should be specified, as should the solvent for extraction (methanol or acetone) because these may give different values.
Environmental IMPACT	Changes in chlorophyll-a concentration 90th percentiles
Relevance to eutrophication of transitional waters	High: Chlorophyll-a is relatively easy to monitor and is a useful proxy for phytoplankton biomass. The occurrence of repeated phytoplankton blooms over a growing season rather than one or two seasonal blooms (eg spring and autumn) maybe an indicator of persistent nutrient enrichment.
Biological quality element Phytoplankton	
Phytoplankton Abundance	
Environmental STATE	The number of phytoplankton cells is a measure of abundance. However, there are differences of several orders of magnitude in the size of pico/nano and microphytoplankton.

Metric of STATE	Total cell counts of micro-phytoplankton or including nano and pico-plankton. Different thresholds are applicable depending on the size classes that are counted Different techniques of microscopy are appropriate for the different size classes. The usual problems of counting cells of colonial forms prevail as does the choice of fixative.
Environmental IMPACT	Changes in phytoplankton abundance.
Relevance to eutrophication of transitional waters	High: Large changes in phytoplankton abundance over spatio-temporal scales are typical in transitional waters.
Phytoplankton Composition	
Environmental STATE	Phytoplankton communities may be characterised by high species diversity
Metric of STATE	Cell counts of a single taxon or a small number of dominant taxa
Environmental IMPACT	Changes in Phytoplankton community composition, in particular the frequent occurrence of blooms of dominant species that may be nuisance or harmful species
Relevance to eutrophication of transitional waters	High species diversity may be characteristic of phytoplankton communities. Increased nutrient availability and departures from the molar ratios may favour blooms of cyanobacteria, which can fix N, or dinoflagellates that do not require silicon or Phaeocystis. The bloom species maybe a nuisance, such as Phaeocystis, "trophic dead ends, harmful (HABs) or even toxic

accession of new Member States should also be considered under lifestyle changes. Examples of PRESSURES that result from these DRIVERS relevant to eutrophication of transitional waters include domestic sewage and detergents as well as increased animal manure as diets become richer in meat products. These are shown in Table 13.

Several DRIVERS result from global and historical changes. Arguably some of the demographic changes included in "lifestyles" could be included under global and historical change, as they are the "human dimension" of global change. However, as they have already been discussed above, they will not be included in this section. The following DRIVERS are grouped under global and historical change: the cost of fossil fuels, the increase in agriculture of biofuel crops (considered under agricultural drivers), the burning of fossil fuels, forest burning, land clearing and climate change. The resulting PRESSURES from global and historical changes relevant to eutrophication of transitional waters include increases in Nitrogen (NO_3^-, ,NO_2^-,NH_4^+, NO, N_2O) as well as changing temperatures, rainfall and storm events. These are shown in Table 14.

The history of the transitional water should also be considered because systems may have a long-term "memory". Decreased PRESSURES such as decreases in nutrient inputs may not be immediately effective, because nutrients stored in the sediment may be released for several decades. An example of this problem is the phosphorus concentrations in the Baltic transitional waters.

Several DRIVERS result from changes in land

Table 13. Example of drivers and pressures that are relevant to eutrophication and that result from changing lifestyles

Drivers	**Pressures**
Population growth or increase due to immigration	Inadequate UWWT Increase sewage and detergents
Urbanization of the coastal zone	Clearing coastal vegetation
Tourist developments and seasonal fluctuations in population	Reclaiming wetlands
Increased wealth, increase consumption of food	Fertilizer use and surplus Animal manure
Changes in diet, increase consumption of animal protein	Increase use of fuels and exhaust gases

Table 14. Examples of drivers and pressures relevant to eutrophication resulting from global and historical change.

Drivers	**Pressures**
Cost of fossil fuels and increase of biofuel crops	Increase fertilizer use
Burning of fossil fuels Forest burning for land clearing	Perturbations of the biogeochemocal cycles, increase (CO^2, NO^3, NO, NH^3, N^2O)
Climate change	Increase temperature Changes in rainfall and hydrological cycle
History of the system	Release of nutrients from sediments

use. These include land clearing, reclamation of wetlands, and damming. The resulting PRESSURES relevant to eutrophication of transitional waters include loss of riparian vegetation, loss of denitrification capacity, retention of Si. These are shown in Table 15.

Several DRIVERS should be considered under the heading of port and shipping activities. These are likely to be stimulated by maritime policy. They include conversion or reclamation of wetlands, dredging and morphological changes. The resulting PRESSURES include loss of riparian vegetation, loss of denitrification capacity, release of N and P, as well as changes in residence time and flushing. These are shown in Table 16.

Other "new" DRIVERS include the increase in golf development, tourism and aquaculture. Golf can be considered with agricultural drivers as can aquaculture. Tourism should be considered under urban development drivers. The PRESSURES associated with golf development are similar to those of intensive agriculture. They include use of fertilisers, wetland drainage, loss of riparian vegetation, irrigation, damming, and groundwater extraction. The STATE metrics that are associated with these are water salinity, water transparency, dissolved oxygen concentration, nutrient concentration and molar ratio, chlorophyll a and phytoplankton cell counts. All these

Table 15. Examples of drivers and pressures relevant to eutrophication resulting from changes in land use.

Drivers	Pressures
Land clearing Reclamation of wetlands Damming of rivers	Loss of riparian vegetation Decrease of denitrification capacity Increased retention of Si Changes in flushing and residence time

Table 16. Examples of drivers and pressures relevant to eutrophication resulting from port and shipping activities.

Drivers	Pressures
Land clearing Reclamation of wetlands Dredging Morphological change	Loss of riparian vegetation Decrease of denitrification capacity Release of N and P from sediment Changes in flushing and residence time

are relevant to the WFD. Other changes in STATE not considered under the WFD might include sediment quantity and quality, with respect to organic matter, mineral composition and redox potential.

Aquaculture is another “new” DRIVER of eutrophication in transitional waters. However, there are many types of aquaculture and these give rise to different PRESSURES. Several examples are given here. Fish aquaculture can be in cages or in ponds. In the case of ponds, if these have been reclaimed from the wetland, there is an associated PRESSURE due to loss of denitrification by the wetland. The ponds may be sterilized with bleach to remove juveniles and larval forms of other species and then neutralized with thiosulphate. These practices disrupt the biogeochemistry of the ponds and represent a significant PRESSURE as they affect the nutrient cycles. In both pond and cage culture, the fish are usually fed and the excess food represents an increase in the organic matter inputs to the transitional water. Even if the feeding practice is improved to minimize this pressure, the faeces of the fish represent a PRESSURE increasing the organic matter inputs to the transitional water. There are also other types of aquaculture, such as bivalve farming. This may be of species such as oyster or mussels or species that are farmed out of the sediment, or species such as clams that are farmed in the sediment. In the case of bivalves, since these are filter feeders, they may graze the phytoplankton and organic matter and mitigate the eutrophication effects in the transitional water. The reduced cell count of phytoplankton and lower chlorophyll a concentration will be registered as an improvement in STATE. However, there are also associated PRESSURES, for example if oyster rafts are shading seagrasses and causing an IMPACT on this Biological Quality Element. In the case of infauna, such as clams, these are also filter feeders and graze the phytoplankton so that there is an improvement in STATE. However, there are also PRESSURES due to this type of aquaculture that arise especially at the time of harvesting, when the sediment is disturbed. This releases nutrients and lowers the oxygen condition of the water since the sediments may be anoxic. Hand harvesting with a hoe-like tool is less disruptive than machine harvesting with a plough-like tool or suction, which has the same IMPACT as dredging.

The amount of sediment disturbance should be minimal to minimize the PRESSURE and IMPACT. The clam beds may be converted seagrass meadows or dug-out saltmarshes. The systematic removal of other competing species, such as low value cockles will IMPACT the biodiversity. Aquaculture, when Best Management Practices are followed can have a positive effect. The aquaculture industry can be a sentinel of water quality, as poor quality water decreases yields and increases mortality.

Tourism should be considered under urban development drivers. The PRESSURES associated with tourism development are similar to those of urbanization. They include increased sewage discharges and water use or extraction. In particular, there may be a strong seasonality to the pressures. Municipalities may see their population multiply by a factor of 20 or more at the height of the tourist season. Nevertheless, local infrastructure such as UWWT plants maybe dimensioned to deal with the resident population. The STATE metrics that are associated with these are water salinity, water transparency, dissolved oxygen concentration, nutrient concentration and molar ratio, chlorophyll a and phytoplankton cell counts. All these are relevant to the WFD.

8.6. Ecosystem services, Ecological Impacts and Economic Impacts

It was shown in the subsection 8.3 above, that the WFD includes concepts of public information, public participation in decision-making, socio-economic considerations and ecosystem services. For this to be successful, it is important that the public should be clearly informed about how ecological impacts can lead to the loss of ecosystem services and be linked to socio-economic impacts. One way of achieving this is to "expand" the DPSIR framework to clearly demonstrate the links between ECOLOGICAL IMPACTS and SOCIO-ECONOMIC IMPACTS. An example of this is shown in Table 17.

8.7 Some examples of the application of the DPSIR approach to transitional waters or in the context of the WFD

Table 18 below summarizes some of the studies that have applied the DPSIR approach to transitional (or coastal) waters. Some of the studies have considered the DPSIR approach in the context of the WFD.

8.8. Future and long-term protection of transitional waters and their resources

The WFD promotes sustainable water use based on a long-term protection of available water resources. In order to achieve this, it is important to make a critical examination of the gaps for the application of the DPSIR approach.

With respect to DRIVERS, these need to be constantly updated to incorporate rapid changes. An example of this is how fluctuations in the price of oil may stimulate the agriculture of biofuel crops and the use of fertilizers.

With respect to PRESSURES: it is necessary to quantify and consider "difficult" aspects such as loss of denitrifying wetlands and atmospheric deposition. For example, there maybe little benefit in addressing the watershed if a significant part of the problem is coming from the airshed.

With respect to STATE: it is important to test and intercalibrate the metrics for the physico-chemical supporting quality elements and the Biological Quality Elements. It is also necessary to carryout integrative assessment. This will be particularly challenging in transitional waters that are typified by rapidly fluctuating environmental conditions.

With respect to IMPACT: both economic impact and ecological impacts should be considered together to clearly demonstrate

Table 17. Example of the expansion of the DPSIR framework to demonstrate the links between ecological and socio-economic impacts through ecosystem services.

ECOLOGICAL IMPACT	Decrease EQR of seagrass, part of “other plants” BQE
ECOSYSTEM SERVICE	Nursery for juvenile fish of commercial value
ECONOMIC IMPACT	Loss of fishing revenue due to both decreased weight of landings and decrease value of catch
SOCIAL IMPACT	Loss of jobs in fishing industry, fishermen and related jobs such as fish processing and fish
VALUATION	This can now be assessed from the losses stated above

Table 18. Application of the DPSIR approach to transitional and coastal waters.

Aliaume *et al*. 2007	DPSIR in S. European lagoons
Bidone and Lacerda 2003	DPSIR, coastal bay, Brazil
Borja *et al*. 2006	How to determine PRESSURES under WFD
Bricker *et al*. 2003b	Eutophication in transitional waters, USA
Fassio *et al*. 2005	DPSIR Nitrate and WF Directive
Ferreira *et al*. 2007	Monitoring coastal and transitional waters
Pirrone *et al*. 2005	DPSIR Po catchment
Rovira and Pardo 2006	Eutophication marine and coastal environments
Scheren *et al*. 2003	Eutophication, lagoon , WAfrica
Trombino *et al*. 2003	Eutrophication, Po catchment

the value of ecosystem services.

With respect to RESPONSE: these should address all the DRIVERS, not only some of them. It may be futile to build more advanced UWWT plants when the agricultural practices in the watershed are responsible for the major PRESSURES relevant to the eutrophication of transitional waters.

8.9. Final remarks

The DPSIR framework approach is a useful tool that can be applied for the adaptive management of eutrophication in transitional waters. However, it needs to be adapted to fit the specific context of the Water framework Directive. The Drivers and Pressures should be frequently updated to reflect changes in the market, (e.g. the price of fuel), lifestyles and global change. The use of the quality elements of the WFD in the State and Impacts is often confusing. Impacts should include the environmental impact and ecological impact, as well as the economic and social impacts. These should be clearly linked so that correct valuations of ecosystem goods and services can be realised. The Responses should address all relevant Drivers and resulting Pressures.

Conclusions

In the Common Implementation Strategy of the WFD a Guidance Document (GD) on eutrophication assessment in the context of European Water Policies has been drafted. This document includes a: General conceptual framework of the eutrophication process; consideration of the different EU water policies and International Conventions that address eutrophication and harmonization of the assessment and monitoring requirements based on the WFD; overview and harmonization of eutrophication assessment methodologies (river, lakes, transitional, coastal and marine waters); and case studies. However, also in this document Transitional Waters were poorly represented. As a consequence the methodology to delineate TWs and typology has not been uniformly applied by Member States and this has given rise to inconsistencies throughout the EU. Current practices of TWs delineation and TWs typology, which constitute the first steps to water body identification which in turn constitutes the basic unit to which the ecological status of the WFD apply, may jeopardize the aims of the Directive for the protection and restoration of these systems through the incorrect allocation of water category. Moreover, the methodology as currently applied gives rise to severe problems for designing cross-national intercalibration exercises in the different geographic regions and creates confusion among the scientific community working on estuarine systems (McLusky and Elliot, 2007).

The present document attempts to bring together the main elements characterizing Transitional Waters (TW) and to identify the current understanding of the process of eutrophication and the problems this raises in establishing reference conditions for these waters, in view of the need of implementing the WFD.

It is thus necessary to reach agreement on working definitions of TW that can form the basis for the development of methodologies that permit the establishment of reference conditions; gain an understanding of the processes of eutrophication and the drivers and pressures that play a major role in its evolution in TW; investigate the use of indicators susceptible of accurately reflecting the ecological quality status of these types of water, as required by the WFD.

The following main conclusions can be summarised from this document:

1: Assessing susceptibility of a transitional water body to eutrophication requires not only a precise knowledge of all the exchanges between the water body and the adjacent systems (atmosphere, land, sea), occurring naturally or as a consequence of human activities, but also a profound understanding of all factors that affect the water body functioning and dynamics. Moreover, TW systems are naturally stressed because their inherent variability in terms of physico-chemical characteristics, e.g. salinity, temperature, concentration of inorganic and organic nutrient, oxygen, etc. Therefore, in these systems biota has become adapted to cope with "natural" stress. Considering eutrophication further, for the reasons described above, it is also possible to view these systems as naturally eutrophic when compared with pristine freshwater or marine systems. If we regard these systems in terms of nutrients alone there is no significant difference between natural and anthropogenic pressures with respect to their effects, and it thus becomes difficult to distinguish anthropogenic or cultural eutrophication.

2. Transitional waters are a natural receptacle for runoff from their catchments and therefore show a natural tendency for nutrient accumulation and eutrophication.

The fate of the nutrients in a transitional water ecosystem is, however, determined by a myriad of interactions related to the ecology and the biogeochemistry of these systems. This means that each transitional water body has site-specific particularities, which will determine its ecological response and in the end determine its susceptibility to nutrient over-enrichment. Site specificities for TW have always existed throughout history and, therefore, the definition of reference conditions required by WFD needs to take these into consideration.

3. The shallow depth of most transitional waters implies that benthic communities and benthic biogeochemical processes are major drivers of nutrient cycling in transitional waters. Aquatic phanerogamms (seagrasses, pondweeds), macroalgae and microalgae (microphytobenthos) are benthic primary producers that will respond to nutrient over-enrichment in direct and indirect ways. The sediments also comprise a sequence of oxic and anoxic habitats where bacterial processes strongly interact with the sediment chemistry and the N, P and Si biogeochemical cycling may be strongly modified by nutrient over-enrichment. The nature of the sediment has a strong impact on the microbial process rates in conjunction with the hydrodynamic conditions. In addition, the exchange between water column and sediment is also strongly influenced by benthic fauna that contributes to the bioturbation and bio-irrigation of the sediment. These processes are extremely complex and operate over a wide range of time scales requiring careful understanding in order to assess trends in water quality and ecological status.

4. Coastal lagoons and estuaries are under growing anthropogenic pressure, inducing an increase in their level of eutrophication, which may compromise their functions and derived goods (e.g. fisheries, aquaculture products) and services (e.g. waste degradation). Human forcing, when superimposed on the high natural variability of transitional waters, implies dramatic changes in ecosystem resources. Loss of biodiversity, at the levels of ecosystems, species and genes, is of concern not just because of the important intrinsic value of nature, but also because it results in a decline in 'ecosystem services' which these natural systems provide. Even before an ecosystem becomes totally degraded, much of its ecological and economic value has been lost.

5. There is a very large amount of literature concerning methodologies for assessing eutrophication in aquatic systems in general. These can be broadly divided into three categories: screening methods, model-based, and mixed approaches. Screening methods have been created to provide an assessment of eutrophication status based on few diagnostic physical and biogeochemical variables. Typical examples are the OSPAR common procedure on eutrophication assessment (OSPAR, 2003) and the United States National Estuarine Eutrophication Assessment (NEEA). With the increase of computer power, model-based methodologies have been developed. Normally, they are based on a hydrodynamic model that incorporates a biogeochemical model that considers the dynamics of organic and inorganic nutrients. Normally, such models are site specific; therefore they are not generally applicable. However, they are useful tools to analyze the environmental responses to changes in pressures as well as to provide environmental managers with an approximate idea of the time constants of their system. Hybrid or mixed approaches try to combine the screening methods with simplified model-based approaches in order to develop general tools that have the advantages of both approaches in terms of applicability and predictive power. For TW several approaches of this type have been developed.

6. However, the present review of the main indicators that have recently been proposed to

assess eutrophication and in a more complete sense, the ecological status in TWs and the definition of class boundaries proposed for each methodology reveals that, compared to other aquatic ecosystems (e.g. lakes, river, coastal etc) the considerable delay in implementation of intercalibration exercises can be attributed to the lack of a wide availability of established indicators due to problems of the very high variability of TW, and at present, insufficient validation across a substantial range of TWs environments to more extensively generalise their applicability.

7. A significant number of methodologies and related indicators of eutrophication are, however, being developed and tested on specific TW sites. These include a methodology for French Mediterranean lagoons, allowing the classification into five eutrophication levels; a methodology to link Pressure-State-Response (PSR) indicators, consisting of a heuristic index of pressure, a symptoms-based evaluation of state, and an indicator of management response; as well as some others However, many of these indicators are based classifying the species according to their response to organic pollution (i.e., the tolerance of various levels of dissolved oxygen) and as discussed above TW are already naturally stressed, therefore, it is not clear to what extent these indicators are able to distinguish between natural and anthropogenic stress. Moreover, taxonomic-based indicators are time consuming and need trained personnel and when compared for coastal lagoons they produce different results.

A further concept is that of thresholds. This has been embedded in ecological risk assessment for a long time, starting from the dose-response curves to a contaminant. However, only in the last years has the concept that "a gradual change in pressure would provoke a gradual change in the ecosystem" been modified by the realisation that this gradual change may be interrupted by a sudden and drastic effect in the ecosystem. There are several examples of regime shifts or thresholds being exceeded in TW systems. Typical examples are regime shifts in the macrophyte community due to nutrient increase, in benthic communities with hypoxia due to increase of organic matter.

8. The WFD states that monitoring networks have to be designed "so as to provide a coherent and comprehensive overview of ecological and chemical status within each river basin and shall permit classification of water bodies into five classes consistent with the normative definitions for specific water categories". The Directive stipulates three different monitoring programmes operational, surveillance and investigative each with its own objectives and for which different quality elements (QE) and sampling strategies are necessary, and further lays down general technical specifications to ensure the quality/ comparability of the monitoring results namely the minimum sampling frequencies of different quality elements. In the case of TW bodies each has special morphological and hydrological characteristics responsible for its highly variable spatial and temporal physico-chemical and biological features. These characteristics need to be considered for selection of monitoring stations, biological quality elements (BQE) and supporting quality elements (SQE) to be measured, and for sampling time/ frequency.

9. The WFD promotes sustainable water use based on a long-term protection of available water resources. Although the WFD does not refer to the DPSIR framework approach, the WFD does link the natural system to the socio-economic system. In particular the DPSIR approach is relevant to the WFD because this Directive requires public information and involvement in decision-making as well as the use of economic instruments. It also links the benefits of protecting water status with economic benefits such as the protection of

fisheries, including coastal fish. The WFD therefore recognizes important ecosystem services of Transitional Waters, such as their role as nursery grounds for juvenile fish. The use of the DPSIR framework has been shown to be a powerful tool in assessing pressures on transitional waters and providing insight on the cost-benefit impact of regulatory processes on the long-term trend of water quality and ecosystem status resulting from different policy drivers.

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