Alternatives to taxonomic-based approaches to assess changes in transitional water communities

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Abstract:
1. Transitional waters, described as critical transition zones because of their position at terrestrial, freshwater and marine interfaces, provide essential goods and services to the biosphere including human populations. These ecotones face increasing human influence mainly due to population density increase in coastal areas.
2. Transitional water bodies have, to date, received little attention in the development of ecological status indicators; this is a critical deficiency when trying to meet the Water Framework Directive objective of all significant water bodies achieving good ecological status by the year 2015.
3. In order to assess changes in transitional water communities many taxonomic-based indicators have already been proposed but there are a number of concerns for their use such as taxonomic classification difficulties, their unsuitability for multi-site comparisons and their inconsistent relationships with disturbance or stress.
4. Alternative methods based on body size, abundance distribution among functional groups, functional diversity and productivity descriptors are proposed. These methods offer the opportunity to compare sites with different taxonomic compositions and allow derivation of indicators related to ecological status of communities under scrutiny.
5. Finally, the suitability of these taxonomic-free descriptors to provide relevant information for each of the four main biotic compartments in coastal lagoons is discussed. The use of biomass distribution among functional groups for fish, benthos and macrophyte and to use body-size distribution for benthos and plankton is proposed

Introduction

The Water Framework Directive (WFD, 2000/60/EC) provides an exciting challenge to scientists and environmental managers: all significant water bodies must achieve a good ecological status based upon physicochemical, hydro-morphological, chemical and biological criteria by the year 2015. Behind the vague notion of good ecological status many fundamental aspects remain unresolved (Basset and Abbiati, 2004). What are the relevant ecological indicators of a good status? Which spatio-temporal scale must be defined to carry out the sampling programme? How to classify the different status? Thus, in order to address the WFD purpose, the question is not yet whether the water bodies are in a good ecological status, but how to define a good ecological status. Even more modestly, how to assess changes in water bodies ecological status?

Among water bodies, transitional waters have been defined as Critical Transition Zones (CTZ) due to their position at terrestrial, freshwater and marine interfaces (Levin *et al.*, 2001). These zones provide essential ecosystem services such as shoreline protection, water quality improvement, fisheries resources, habitat and food for migratory and resident animals and recreational areas for human populations (Levin *et al.*, 2001). These CTZ are also very productive ecosystems (Knoppers, 1994), but are under severe stress due to human activities and climatic change inducing sea-level rise. Human impacts are mainly due to permanent and seasonal population density increases, aquaculture, fisheries, agriculture and industry. The increasing pressure inevitably leads to the expansion of environmental crises such as anoxia, over-exploitation of resources, destruction of habitat, eutrophication and pollutant contamination from the land-use watershed (Crooks and Turner, 1999).

Despite of their obvious ecological and economical interests, transitional water bodies have received little attention about development of ecological status indicators. This is partly due to the inherent and extreme spatio-temporal variability observed in such ecosystems. Basset *et al.* (2001) point out that the daily range of nutrient concentration can reach the level of annual range in some extreme cases. This is the consequence of low inertia to external events due to shallowness of water masses coupled with an interface position between marine and watershed water bodies leading to continuous perturbations such as tidal variations, winds, storms and chaotic freshwater discharges. For instance wind events are able to cause significant suspension of bottom materials injecting dissolved and particulate benthic materials into the water column leading to large increases in nutrient levels (Lawrence *et al.*, 2004). Thus monitoring nutrient concentrations or abiotic elements becomes a difficult task in such systems in order to assess an ecological status (Fano *et al.*, 2003). As a consequence physico-chemical parameters are supposed to provide supplementary information to the WFD but are not supposed to define ecological status on their own.

As an alternative to abiotic indicators, biocriteria have been developed and successfully applied in aquatic surveys (e.g. Lougheed and Chow-Fraser, 2002). However, most of them were implemented for continental (GarciaCriado *et al.*, 1999; Nichols *et al.*, 2000) or marine water bodies (Borja *et al.*, 2000; Mouillot *et al.*, 2002; Simboura and Zenetos, 2002). When biotic indicators were proposed for lagoon or estuarine ecosystems they were mainly based on indicator species, taxon richness and diversity indices (Gibson *et al.*, 2000; Fano *et al.*, 2003; Panayotidis *et al.*, 2004); but these taxonomic-based indicators raise several problems. First of all, in addition to being time-consuming, identification of individuals to the species level is not an easy task for many groups due to the size of specimens or to the presence of sibling or cryptic species (Knowlton, 1993), and many taxonomic errors may occur in ecological

assessment studies (e.g. Sheppard, 1998). Moreover, the observed taxonomic richness is an underestimation of the true taxonomic richness (real number of species living on the studied site) and the error involved will depend on sampling effort (Gaston, 1996), and is rarely consistent among sites. Finally, the specific diversity or richness is difficult to relate to an ecosystem disturbance level because the diversity-disturbance relationship remains controversial and its shape is often unimodal with the highest diversity being observed for medium levels of disturbance or stress (Huston, 1979; Mackey and Currie, 2001). For these reasons, community richness does not seem to be a useful indicator of disturbance in applied ecology (Drobner *et al.*, 1998) because a low richness can be either related to a high stress level with very few species able to survive in such demanding conditions or either related to low stress and high productivity levels with only the best competitors able to persist (Connell, 1978; Huston, 1979). Furthermore, monitoring ecological status of transitional waters not only at the regional but also at the continental scale (e.g.combining in the same evaluation lagoons or estuaries from Portugal to Greece) inevitably leads to a new problem associated with taxonomic-based indicators. Species and lineages inhabiting transitional water bodies are likely to be different among locations to be compared (Basset *et al.*, this issue). As a consequence ecological indices based on indicator taxon will be influenced by anthropogenic constraints but also by the distance between sites. For instance a species supposed to reveal a good ecological status in Greece may be absent in Spain or Portugal because this species' geographic range is likely to be limited and not because the ecological status of transitional water body masses is different in the latter two countries.

Thus, even if the WFD does not require the use of similar indicators in all the countries, some intercalibration studies are necessary among sites with different taxonomic compositions. The severe flaws related to the use of richness or indicator species in order to assess ecological status of transitional water body masses highlight the need for more integrated community-based indicators. Simpson (1949) suggested that the number of species only represents one aspect of species diversity, noting that the diversity of a given community also depends on the quantitative equilibrium between species, termed evenness. To take into account the community structure, numerous diversity or evenness indices have been proposed such as the Shannon-Weaver (1949) index and its associated evenness or the Simpson (1949) index. These diversity indices are classically used in ecology to assess the environmental impacts on ecosystems (e.g. Pires *et al.*, 2000); however, some authors criticized the limited ability of these indices to discriminate between communities or to detect changes within a given community (Cao *et al.*, 1996; Pires *et al.*, 2000; Orfanidis *et al.*, 2001). For instance the Shannon diversity index appeared inadequate for impact assessment since it cannot discriminate natural stress from anthropogenic impact (Reizopoulou *et al.*, 1996; Reizopoulou and Nicolaidou, 2004). Moreover, these classical species diversity indices do not take into account functional differences between species though some authors pointed out the necessity of including these differences between species to estimate a diversity related to changes in environmental conditions or influencing ecosystem processes (Diaz and Cabido, 2001; Mouillot *et al.*, 2005). Also, the inefficiency of the available multivariate techniques applied in transitional waters has been attributed to the fact that they were making use only of the existing information at the species level without taking into account their phylogenetic/taxonomic relationships (Arvanitidis *et al.*, 2005a, b). Thus, the most important question is not whether a proposed statistic summarizing a community satisfies some theoretical criterion, but whether it allows useful distinctions, with a functional meaning, between ecosystems to be made with field data.

Regarding the limitations listed above and induced by the taxonomic-based approach, some alternative CTZ systemic indicators have to be developed in order to address the main Water Framework Directive issues. The aim of this viewpoint is to propose an overview of alternative methodologies thus overcoming problems related to taxonomic ecological indicators for four main biotic components in coastal lagoons: fish, benthos, macrophyte and plankton.

Body size or size spectra

Body size is one of the most important traits of an organism determining, to a large extent, the type and the strength of ecological interactions to which individuals are subjected (De Roos *et al.*, 2003). More precisely, physical structure of aquatic habitats has often been cited as a driver of body-size distributions in communities (Robson *et al.*, 2005) such as discontinuities in body-size distributions explained by habitat architecture (Holling, 1992). In addition, individual development could be density dependent and influenced by changes in environmental conditions (De Roos *et al.*, 2003). Body size is generally easy to measure and amenable to intercalibration procedures, it is comparable across taxa, guilds and sites, and as a community feature, is expected to vary on disturbance gradients, according to energetic and ecological constraints (Basset *et al.*, 2004).

This taxonomic-free indicator was successfully used for some biotic components of transitional water body masses but is still under scrutiny for others. The most promising application comes certainly from the benthos; biomass profile of benthic communities has been already used to investigate community structure in coastal lagoons (Reizopoulou *et al.*, 1996; Lardicci and Rossi, 1998; Basset *et al.*, 2004); changes of benthic community biomass under disturbed conditions are well documented in benthic ecology (Pearson and Rosenberg 1978; Warwick, 1986), since small-bodied invertebrates may characterize environments with high instability, and small body size could be a consequence of environmental/anthropogenic pressures imposed on organisms. As an illustrative example, a biometric index (Index of Size Distribution - ISD) was recently developed and applied as a new tool for quantifying the degree of disturbance in coastal lagoons (Reizopoulou and Nicolaidou, 2005). This kind of indices may highlight alterations in benthic ecosystems since size variability of benthic organisms may decrease along pollution gradients (Reizopoulou and Nicolaidou, 2005).

Robust indicators describing the community structure such as size spectra in fish communities are also promising but are not yet well developed (Rochet and Trenkel, 2003), but size spectra in fish communities could be a suitable indicator allowing discrimination in coastal lagoon environments. In a different context, it is shown that the slope of the size spectrum decreases quasi-linearly with fishing mortality and that the curvature could help to detect ecosystem overexploitation (Shin and Cury, 2004).

Plankton size structure is also a common taxon-independent tool for the study of community and ecosystem structure of aquatic environments (Rodríguez *et al.*, 1987; Chisholm, 1992; Quinones, 1994; Cavender-Bares *et al.*, 2001). Size fractions (Sieburth, 1979) and size spectra, by grouping cells into logarithmic size classes (Rodríguez *et al.*, 1998; 2002), are conventional descriptors of phytoplankton size structure. Various structural abiotic environmental factors were found to explain phytoplankton size structure; they include growth factors (Helbling *et al.*, 1991), vertical water dynamics (Rodriguez *et al.*, 2001, Serra *et al.*, 2003), water depth (Gaedke, 1992) and trophic state (Glover *et al.*, 1985; Tamigneaux *et al.*, 1999; Yew-Hoong Gin *et al.*, 2000; Maranon *et al.*, 2001; Iriarte and Gonzalez, 2004).

Although there is evidence of changing phytoplankton size structure with respect to shifts in trophic conditions (Maranon *et al.*, 2001; Iriarte and Gonzalez, 2004), no methodology exists using phytoplankton biometric measures as a means for discriminating among trophic levels. In coastal marine areas, body size spectra showed common patterns of variation according to major environmental forcing factors, independently of the taxonomic composition of phytoplankton (Sabetta *et al.*, 2005). Furthermore, for the purposes of the Water Framework Directive, new techniques such as flow cytometry allow rapid, reliable and reproducible data acquisition on phytoplankton size structure (Vaquer *et al.*, 1996; Rutten *et al.*, 2005). The regularity of phytoplankton size-abundance distributions can be translated into useful size-based descriptors of community or ecosystem structure and incorporated into environmental monitoring and conservation programmes for the purposes of the WFD, even though at this stage a methodological standardization of descriptors for ecological status classifications is lacking.

Macrophyte size spectra descriptors seem much less promising than other biotic components, as most of lagoon macrophytes consist of seagrasses and/or free-living macroalgae, which are not size bounded. As a consequence, macrophyte size structure at the community level reflects more physical ecosystem attributes such as depth or mechanistic constraints (fragmentation) than disturbances. However, at the population level Hackney and Durako (2004) demonstrate the sensivity of size-frequency distributions of the seagrass *Thalassia testudinum* morphometric characteristics to differences in environment factors in coastal areas. The challenge would be to determine a macrophyte species with a high morphometric plasticity to allow discrimination of coastal lagoon environments at the Mediterranean level.

In order to apply body size–related descriptors to monitoring programmes it is important to clarify if they are just phenomenological descriptors, indirectly related to ecosystem ecological status, or if they are emergent properties of communities. Several authors have expressed concerns about this issue (Leaper *et al.*, 2001; Robson *et al.*, 2005), even though the question is no longer whether habitat architecture and environmental variables matters for body size distribution within communities, but how it matters and which are underlying mechanisms and assembly rules. The recent development of metabolic theory (Brown *et al.*, 2004) confers the theoretical background to body size related descriptors of ecological status of ecosystems, i.e. ecosystem health, by relating body size constraints on patch selection and giving up behaviour to interspecific coexistence into a metabolic theory of coexistence (Basset, 1995). Accordingly, a tendency of body size to be positively related to population energy use has been observed (e.g. Maurer and Brown, 1988) and a relative invariance of size spectra compared to taxonomic composition was also detected in many different guilds of both macrobenthos (Bourassa and Morin, 1995; Strayer, 1986; Basset *et al.*, 2004) and plankton (Sprules and Munawar, 1986; Sabetta *et al.,* 2005). Therefore, the ecological relevance of body size has already received strong experimental support; however, further research on transitional aquatic ecosystems is needed to test the relevance of the metabolic theory of coexistence (Basset, 1995) on the assembly rules responsible of the body size spectra shape at different spatial and temporal scales in such complex ecosystems.

Biomass distribution among functional groups

Many studies have highlighted the complexity of ecological systems and their fundamental unpredictability due to multiple interactions (Huisman and Weissing, 2001). One way to overcome this problem is a simplification of communities through partitioning of species into a variety of guilds, functional groups or functional types (Blondel, 2003; Jauffret and Lavorel,

2003; Mathieson *et al.*, 2000; Simberloff and Dayan, 1991). Until now the guild concept has been more widely used than that of functional groups for animals. However the guild concept refers exclusively to the mechanisms of resource sharing by species in a competitive context. Within the framework of biodiversity and ecosystem functioning, a wide range of functions and responses are required from organisms and the 'functional groups' partitioning seems more appropriate than the 'guilds' one.

This approach seems particularly relevant for macroorganisms such as fish, benthos and macrophytes because these organisms can provide functional or life history traits allowing a classification of species into homogeneous functional units as a preliminary step. Species classification into functional groups is already well tested for fish (Dumay *et al.*, 2004; Nagelkerken and van der Velde, 2004) and benthos (Pearson, 2001; Gerino *et al.*, 2003) but not so commonly for macrophytes (Steneck and Dethier, 1994) and plankton. Based on functional form typology, Littler and Littler (1980) showed that species within a functional group "behave" similarly along stress-disturbance gradients regardless of their taxonomic or biogeographic features. Thus, algal-dominated communities, when examined at the functional group level, appear to be much more temporally stable and predictable than when examined at the species level. For plankton, a promising methodology is based on phytoplankton photopigments (Paerl *et al.*, 2003) that can be a surrogate for phytoplankton functional groups such as chlorophytes, cryptophytes, cyanobacteria, diatoms and dinoflagellates.

Shannon and Cury (2004) introduced an indicator of interaction strength (IS) which quantifies the effect that a change in biomass of one group has on abundance of other groups in pelagic fish communities. They also introduced an indicator of interaction strength (IS) which quantifies the effect that a change in biomass of one group has on abundance of other groups. Nevertheless, these indices are more related to interactions than to environmental influences and do not correspond to the aim of the Water Framework Directive. A second example deals with the distribution of fish biomass along a gradient of trophic level which can be considered as a criterion for a functional classification because the 'trophic level', as a simple descriptor of fish species position in a food web, is certainly a key component to study the functional role of the fish compartment in an ecosystem (Cruz-Escalona *et al.*, 2000; Hajisamae *et al.*, 2004). Based on these considerations, Sosa Lopez *et al.* (2005) developed a useful indicator (BDAtroph) that reflects changes in the trophic structure beyond simple changes of the mean trophic level. This indicator depends on information taken from FishBase 'ecology table' (Pauly and Christensen, 1995; Froese and Pauly, 2005) consisting of trophic level data for each fish species based on both the diet composition (trophDC) and food items (trophFI). The fish biomass was then arranged along trophDC intervals in order to build the general shape of the cumulated relative biomass (cumulated relative BDAtroph) smoothed by weighted least squares (McLain, 1972). The Kolmogorov-Smirnov test (Zar, 1997) was performed between two samples in order to assess differences in the general shape of the cumulated relative biomass along the trophic levels (BDAtroph). The apparent BDAtroph consistency across short-term periods suggests that this indicator could be considered as a useful quantitative ecological indicator when long-term comparisons between fish communities are performed (Sosa Lopez *et al.*, 2005). The several shapes of the cumulated relative BDAtroph deserve to be evaluated more deeply in order to link them to specific processes such as top-down regulation of community dynamics contrasted with bottom-up factors affecting plant or animal prey availability. However, the proposed indicator could provide ecologists with the opportunity to address a variety of questions on changes and differences in fish community structure, and their relationships with environmental and human influences.

The recently developed Ecological Evaluation Index EEI (Orfanidis *et al.*, 2001; Panayotidis *et al.*, 2004) is based on the classification of macrophyte species in two Ecological State Groups, based on morphological and life strategy traits (Littler and Littler, 1980). The ESG I, characterizing pristine conditions, comprised the thick leathery, the articulate upright calcareous and the crustose calcareous species, most of them being K-selected species. ESG II, characterizing disturbed conditions, grouped the foliose, the filamentous and the coarsely branched upright species together, most of them being R-selected species. Sampling sites are classified in one of five Ecological Status Classes (bad, low, moderate, good, high) after a cross-comparison of the mean abundance value of the ESG I and II, on a matrix (Figure 1). Various applications of the index on macrophyte communities of the Aegean, as well as comparisons with other indices showed that it is an efficient tool for revealing disturbed conditions (Spatharis *et al.*, 2003; Panayotidis *et al.*, 2004). However, its efficiency remains to be assessed for other European coastal areas as well.

Thus biomass distribution among functional groups can lead to the development of useful indicators within the Water Framework Directive context to help assess differences in coastal lagoon status. These indicators would themselves be taxonomic-free but would require considerable biological knowledge about the constituent species and their functional attributes in order to classify them into functional groups. Such an approach is already tried and tested for fish, benthos and macrophyte, and deserves to be implemented for plankton (Table 1).

Functional diversity

'Functional diversity', which was initially defined as the value and range of functional traits of the organisms present in a given ecosystem (Diaz and Cabido, 2001), has been measured in different ways capturing different facets of this diversity such as the variation in the functional attributes of species (Walker *et al.*, 1999; Petchey and Gaston, 2002; Mason *et al.*, 2003), the complexity of food webs (Hulot *et al.*, 2000), the number of plant functional groups present (Diaz and Cabido, 2001) and the regularity of abundance distribution in functional trait space (Mouillot *et al.*, 2005). Thus a more general definition of functional diversity is clearly needed embracing these different components. Mason *et al.* (2005) recently redefined functional diversity as a measure (or group of measures) of the distribution of the species and abundance of a community in functional attribute space that represents: (a) the amount of functional attribute space filled by species in the community (functional richness) (b) the evenness of abundance distribution in filled niche space (functional evenness) and (c) the degree to which abundance distribution in niche space maximizes divergence in functional attributes within the community (functional divergence).

It is now generally accepted that functional diversity, rather than species diversity per se, is the key for ecosystem processes such as productivity, stability, resilience or invasibility (Hulot *et al.*, 2000; Naeem and Wright, 2003; Petchey *et al.*, 2004; Roscher *et al.*, 2004; Waldbusser *et al.*, 2004). Most of experiments relating functional diversity to ecosystem processes were carried out on terrestrial plants, but two recent studies underline the positive effect of benthic functional diversity on organic matter processes in sediments (Waldbusser *et al.*, 2004; Mermillod-Blondin *et al.*, 2005), but has yet to address other biotic components such as fish, macrophytes and plankton. Within the context of the Water Framework Directive, further investigation of the influence of environmental factors on functional diversity is needed in order to evaluate its ability to be a relevant indicator of ecological status.

Very few studies have dealt with this issue but Bremner *et al.* (2003) using benthic invertebrates, identified some consistent patterns of functional diversity across scales and Bady *et al.* (2005) demonstrated that functional diversity had a greater accuracy than species richness with less sampling effort. As an application Fano *et al.* (2003) developed the ecofunctional quality index (EQI) for the evaluation of environmental quality in lagoons using a set of data embracing primary productivity, seaweed and seagrass biomasses, structure and productivity of the benthic community, taxonomic diversity of macrozoobenthos, and finally, trophic complexity, expressed as macrozoobenthic functional diversity. However this index requires a lot of information on different biotic components and must be tested on a large set of coastal lagoons with various taxonomic compositions.

From a theoretical point of view, we can suspect that functional diversity decreases with increasing environmental constraints or stress (Figure 2). When environmental constraints increase, coexisting species are more likely to be similar to one another because environmental conditions (i.e. abiotic properties of the habitat) act as a filter allowing only a narrow spectrum of species to survive. The species that make it through the environmental filters are likely to share many biological/ecological characteristics through the niche filtering concept (Franzen, 2004; Statzner *et al.*, 2004). More precisely, environmental factors could limit the presence of certain functional traits at certain sites and thus decrease functional diversity of local communities in sites under environmental pressure such as confined parts in lagoons (Figure 2). For instance, increasing organic disturbance in coastal lagoons has been shown to lead to a decrease of macrobenthic size in three Mediterranean lagoons (Reizopoulou and Nicolaidou, 2005): small-bodied specimens were mainly represented by tolerant and opportunistic deposit feeders, while the larger-bodied specimens comprised mostly filter feeding bivalves, carnivorous polychaetes, etc.

The hypothesis that there is decreasing functional diversity along a disturbance gradient could be tested on the four biotic components. The issue that functional traits allow functional diversity estimation, however, must be resolved first, although this appears the case for benthic invertebrates (Bremner *et al.*, 2003; Fano *et al.*, 2003; Waldbusser *et al.*, 2004; Bady *et al.*, 2005; Mermillod-Blondin *et al.*, 2005) and for fishes (Sibbing and Nagelkerke, 2001; Dumay *et al.*, 2004; Nagelkerken and van der Velde, 2004) but still unresolved for macrophyte and plankton. There has been a recent effort to produce a functional classification of freshwater phytoplankton associations based on commonly shared adaptive features (Reynolds *et al.*, 2002) and to associate them with trophic states (Padisak *et al.*, 2003). Additionally, phytoplankton functional attributes such as cell volume, growth rate, and the ratio between minimum quotas of total N and total P have also been investigated (Seip and Reynolds, 1995).

Once the functional attributes are estimated for each species, many functional diversity indices can be applied providing either an overall diversity index including all functional traits (Petchey and Gaston, 2002) or a univariate diversity index for each trait (Mason *et al.*, 2003, 2005; Mouillot *et al.*, 2005). These indices are not taxonomic based since species identity and its taxonomy is never included in the analysis allowing comparison between distant sites. Nevertheless more theoretical and empirical studies are waited before considering functional diversity as a tool to assess changes in coastal lagoon ecological status.

Biomass or productivity measures

Biomass has often been used as a surrogate for productivity of primary producers, which is the most commonly cited ecosystem property in ecological studies (e.g. Loreau, 2000). Some statistical methodologies, which do not involve species identification but are based on chlorophyll a and nutrient concentrations in the water column, have been proposed for water quality assessment (Kitsiou and Karydis, 1998). As an application, the Joint Research Centre of the European Union (Druon *et al.*, 2004) is developing a risk index (EUTRISK) as an indicator of European coastal areas sensitive to eutrophication. EUTRISK is based on observations of phytoplankton biomass in the top layer of water column and determined as chlorophyll abundance by remote sensing. Nevertheless these methods contribute to assessment of eutrophication levels but not to assessment of biological status of water body masses as stated in the WFD. This is more the task of the "Eutrophication activity" WFD working group that has not yet been fullfiled.

Biomass density *per se* does not appear to be a reliable parameter for distinguishing between different ecological status of macrophyte communities. Indeed, along disturbance gradient such as eutrophication, the amount of total primary production is roughly maintained (Schramm, 1999), and the shift concerns progressive regression of slow-growing perennials and final replacement by fast-growing ephemeral forms; suggesting that an indicator of biomass turn-over rates, combining standing stocks and productivity, is a promising taxonomic-free approach.

In the classic model of benthic ecology of Pearson and Rosenberg (1978), benthic variables such as macrofaunal abundance or biomass along an organic enrichment gradient are expected initially to increase to a certain level and then to decrease. Thus, this non-monotonic indicator is not suitable to assess changes in benthic communities. Under disturbed conditions, the larger, long-lived species are the first to be eliminated and the communities are dominated by smaller, short-lived opportunistic species. A useful indicator could estimate the biomass turn-over between these two sets of species.

A measure of fish biomass or density per surface area is still very problematic to obtain in coastal lagoons because the sampling methodology is far from homogeneous among regions. An active capture method, such as a drawnet, or a passive method using 'capetchade' or trammel net, can be used to give fish biomass or density. With the former methodology it is possible to sample the entire water column from the bottom to the surface and a certain area but the largest individuals and some species (Mugilidae) are almost impossible to catch, introducing a bias in the results. The passive method causes selectivity problems, i.e. some species are more easily caught than others, and thus the samples can be biased. In addition, there is the difficulty to relate an extremely variable indicator such as the biomass/density of fish to a certain ecological status. A most promising tool is certainly to carry out experiments based on cages to measure mortality and growth of juveniles. This method may give insights into the ability of coastal lagoon environments to provide suitable conditions for juvenil fish to survive and growth, a fundamental service provided by coastal ecosystems (Baldo and Drake, 2002).

Conclusion

Transitional waters, which provide essential goods and services to the biosphere, are facing increasing human influence mainly due to population density increases on coastal areas. Thus, within the European Water Framework Directive context, indicators of ecological status in order to classify different sites are required. Monitoring nutrient concentrations or abiotic

elements is a very difficult task in such environment because, due to the shallowness of lagoons combined with the interface between marine and watershed water bodies that leads to continuous perturbations, extreme spatio-temporal variability was observed causing inconsistencies in impact assessment studies (Basset *et al.*, 2001). As an alternative to abiotic indicators, biocriteria have been developed, mainly related to indicator species or taxonomic description of the communities (Gibson *et al.*, 2000; Panayotidis *et al.*, 2004). However the current study highlights some concerns about the extensive use of such taxonomic-based indicators to classify water body masses. Individual classification into taxonomic units remains time-consuming and problematic for some groups such as plankton. Moreover, taxonomic richness is often assumed to represent a second order (unimpodal) relationship with ecosystem productivity or disturbance negating its usefulness as a relevant indicator (Drobner *et al.*, 1998). In addition, most of species and taxon are not present in all the sites being compared. As a consequence we cannot relate presence-absence of some taxon to a given ecological status in multi site comparisons. Alternative descriptors of communities, based on taxonomic-free attributes: body size, abundance distribution among functional groups, functional diversity and productivity are proposed. Such methods overcome many problems related to taxonomic indicators and deserve to be applied in extensive comparative studies; e.g. biomass distribution among functional groups for fish, benthos and macrophyte and the use of body size distribution for benthos and plankton. Functional diversity has potential to define ecological status of communities for every kind of organism but more theoretical and conceptual work is needed.

Acknowledgments

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Table 1. Suitability of four taxonomic free descriptors of four biotic compartments to assess changes in coastal lagoons (S: suitability with tools already implemented and tested, D: suitable but with more development needed, x: not suitable)

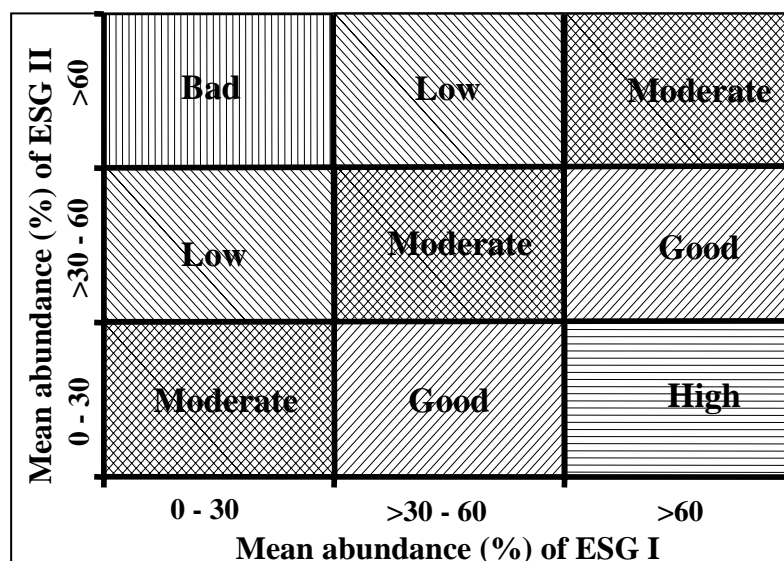| Descriptor	Biotic compartment			
	Fish	Benthos	Macrophyte	Plankton
Body size distributions	D	S	x	S
Biomass distribution among functional groups	S	S	S	D
Functional diversity	D	S	D	D
Productivity	x	x	D	D

Figure 1. Matrix for the comparison of the mean abundances of macrophytes (% cover) classified in Ecological State Groups I and II.
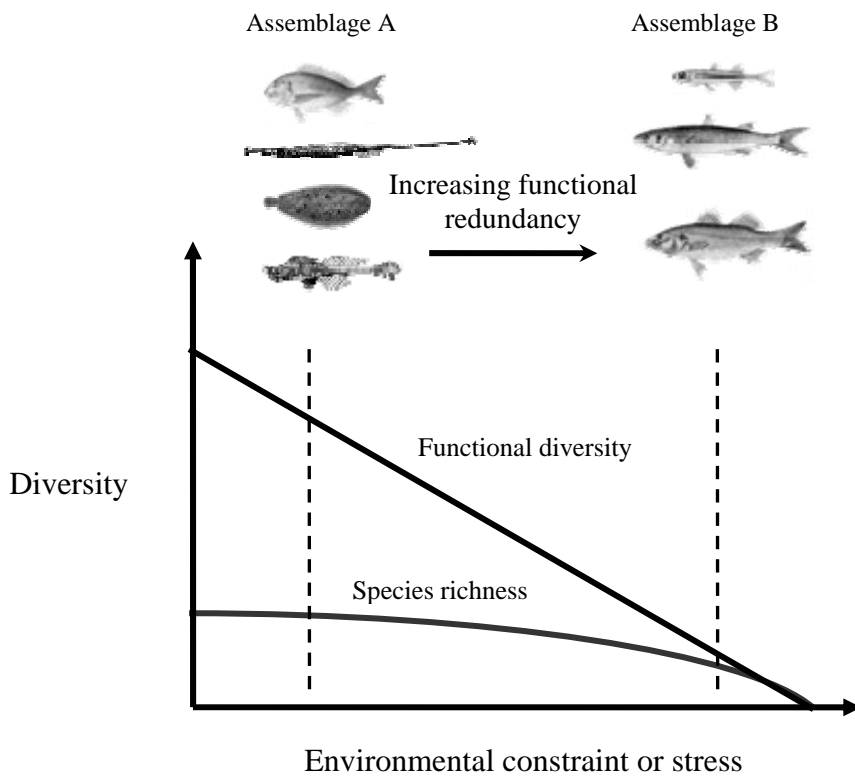
Figure 2. Theoretical relation between functional diversity, species richness and environmental constraints in fish assemblages. When environmental constraints increase functional redundancy or similarity increases in assemblages through niche filtering (between A and B), and thus functional diversity would decrease at a higher rate than species richness.