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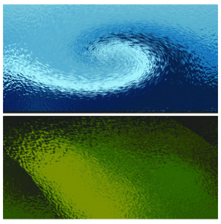

REVIEW ARTICLE

Biodiversity and bio-evaluation methods in transitional waters: a theoretical challenge

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Abstract

1 - This paper addresses the question of the weaknesses of the methodologies developed based on the analysis of the composition of benthic invertebrate communities in transitional waters.

2 - Benthic communities are an important element of the bio-evaluation methodologies suggested for the Ecological Quality status of the European transitional and coastal waters in the context of the Water Framework Directive. It is argued that the assessment of Ecological Quality status requires both fundamental and applied science.

3 - The lack of performance of many biotic indices under varying and highly fluctuating environmental conditions may well be related to weakness in theory supporting marine biodiversity.

4 - We propose some possible approaches for future development in the field of biotic indices and marine biodiversity theory.

Introduction

The European Water Framework Directive (WFD) has been designed to protect and restore aquatic environments including those of the coastal and transitional (primarily estuaries and lagoons) waters. This directive, as well as the European Marine Strategy, provides a legal background and thus a chance for the protection of marine biodiversity (Borja 2005, 2006). The WFD provides a framework for the evaluation of the "Ecological Quality" (EcoQ) status of aquatic ecosystems. The assessment of water bodies' EcoQ is based on several quality elements including benthic invertebrate fauna. Each of these quality elements must provide a separate evaluation of the water body's EcoQ. The final classification of the status of water body into one of the five EcoQ classes (High, Good, Moderate, Poor, or Bad) is defined as the lowest EcoQ class determined by the different quality elements following the rule of the "one out, all out". Therefore, assessing a "Moderate" (or worse) EcoQ status with a benthos-based biotic index will automatically set the Ecological Quality status of the water body to "Moderate" (or worse). Such a result implies that restoration measures will have to be taken in order to reach the "Good" EcoQ status in 2015. Unreliable assessment could either prevent required protection and restoration measures to be taken or, conversely, induce a waste of both human and financial resources and lead to inefficiency. Consequently, this matter deserves much attention from researchers because it transcends the field of applied science and affects that of management and policy-making.

Applications of Biotic Indices to transitional waters: a theoretical challenge

In Europe, the development of benthic invertebrates-based bio-evaluation methods has been considerably stimulated since the publication of the European WFD. The development and use of Biotic Indices (BIs) has also triggered considerable scientific debate (*e.g.* Diaz et al., 2004) but this is mainly focused on computational problems rather than fundamental debates on the validity of the approaches (see Borja et al., 2003; Simboura, 2004; Ruellet and Dauvin, 2008). Every bio-evaluation method includes some measure of a biodiversity aspect of the benthic macrofauna communities. The most popular BIs that have been proposed are based on Pearson and Rosenberg (1978)'s model (known as P-R model), which describes the response of soft-sediment subtidal benthic macrofauna towards increasing inputs of organic matter. This model is a spatial development of the "ecosystem re-setting" concept, as expressed by Eugene Pleasants Odum back in the 50s. The dominant approach in Europe consisted in classifying taxa according to their level of tolerance/sensitivity to pollution following the works of Glémarec and Hily (1981) and Grall and Glémarec (1997). This classification of species, or groups of species, is either fixed and determined through an extensive review of literature data and expert judgement (Borja et al., 2000; Simboura and Zenetos, 2002; Dauvin and Ruellet, 2007) or ecosystem-specific and defined according to the α-diversity of stations at which the species occur (Rosenberg et al, 2004). The level of tolerance/sensitivity is based on the theory of demographic strategies, which considers that *r*-strategists are favoured in unpredictable, unstable, environments whereas *k*-strategists are favoured in stable, predictable, environments (Pianka, 1970). It also presumes that *r*-strategists appear as early colonizers before being out-competed by *k*-strategists. Consequently, these BIs assume that perturbed environments are characterized by unstable conditions which lead to the disappearance of the benthic fauna and of the secondary succession process, as described by Pearson and Rosenberg (1978). This most probably holds true for benthic communities experiencing catastrophic events such as episodic dystrophic crises (Rosenberg et al., 2002) or oil spills (Dauvin, 2000). However, it fails to explain the observed responses of benthic fauna subjected to chronic disturbances. Theoretically, there is no argument upon which to conclude any instability in these systems: regardless of their demographic strategy, species can either withstand any pollution through adaptation of their life-traits, or disappear. Any species may react differently according to the type and intensity of disturbance (Bustos-Baez and Frid, 2003) and its biological traits. As an example, the typical estuarine amphipod *Corophium volutator* proved to be sensitive to metal contamination in the Fal estuary (Warwick, 2001) whereas Norkko et al. (2006) described the opportunistic behaviour of this species in experimentally defaunated sediments. The reaction of benthic organisms to the large set of disturbances affecting coastal and transitional water bottoms (Ellis et al., 2000) is thus complex and still unpredictable so that the P-R model may not always be applied.

Transitional ecosystems such as estuaries and lagoons are characterised by variable salinity, temperature and oxygen availability and muddy bottoms which are generally organic-matter enriched. Moreover, in areas subjected to tides, tidal flats may constitute the largest part of the ecosystems. Benthic species inhabiting these ecosystems are considered by most BIs as opportunist or tolerant and constitute species-poor benthic communities. Consequently, these BIs systematically classify these benthic communities as corresponding to degraded environmental conditions and classify these ecosystems as "Moderate", "Poor" or "Bad" according to the WFD (*e.g.* Labrune et al., 2005; Quintino et al., 2006; Chainho et al., 2007; Zettler et al., 2007, Blanchet et al., 2008). This problem is, however, partly caused by the lack of appropriate reference conditions against which results of BIs can be compared (Dayton et al., 1998). Such works clearly demonstrated that reference conditions must be defined on the basis of the benthic habitat including at least the type of sediment, the salinity regime, the intertidal or subtidal level, the presence/absence of recognized engineer species (*e.g.* seagrasses, mussel beds, oyster reefs) and the biogeographic province or sector. The definition of appropriate reference conditions is a key problem for the application of BIs in ecosystems such as transitional waters and semi-enclosed ecosystems but also one of the main difficulties. Finding reference conditions corresponding to natural environmental conditions in European transitional waters is indeed a particularly difficult task when considering the level of disturbance in these systems. Estuaries and lagoons receive higher amounts of pollutants from their catchment areas and present higher water residence times than the nearby coastal zone. Moreover, many pollutants tend to be adsorbed on particulate organic matter that compose their muddy bottoms. There are thus confounding effects in many estuaries and lagoons between the level of pollution and both salinity and sediment organic content which do not always allow to discriminate results in community patterns caused by natural variability or by anthropogenic impact. Consequently, results obtained with the use of most BIs may be falsely considered as consistent with the pollution level of an estuary (*e.g.* Dauvin et al., 2006).

These observations lead Elliott and Quintino (2007) to define the "Estuarine Quality Paradox": studies conducted in estuaries and lagoons indeed challenge the widely accepted P-R model and the common concept that high biodiversity ensures good ecological functioning (Loreau et al., 2001). Estuaries and lagoons may thus be considered as exceptions to the above concept in that they function successfully although supporting a low (bio)diversity (Elliott and Quintino, 2007).

It is thus probable that most commonly used BIs have been based on false or incomplete paradigms (Elliott and Quintino, 2007), at least for the transitional ecosystems. Consequently, attempts to inter-calibrate these indices, though required by the WFD, may be misleading (Borja et al., 2006; Simboura and Reizopoulou, 2008) and thus that the development and testing of new indices with existing methods should be fostered. Marine ecologists are consistently pressed by stakeholders, environmental managers and policy-makers to deliver operational methodologies that allow classification of water bodies into the five EcoQ classes, as defined by the WFD. The main problem is that, for the time being, there are no alternative theories to complement or replace the P-R model. Therefore, our understanding of: (1) the response of benthic organisms to disturbance, and (2) the control of macrofaunal biodiversity in transitional waters, may well suffer by theoretical gaps.

Some proposals for future research

Fundamental studies on the patterns and processes associated to transitional water benthic communities must be pursued. As an example, Petersen (1913, 1918) and Thorson (1958) defined the *Macoma balthica* community, which occurs in the estuaries of Northern Europe. This community was also identified in the Atlantic region of Southern France and Spain where it is called the *Cerastoderma edule – Scrobicularia plana* community (Bachelet, 1979; Borja et al., 2004). *M. balthica* indeed reaches its southern limit near the Gironde estuary (Hummel et al., 1998; 2000). Apart from their ecological preference, the level of ecological similarity between these communities has not been estimated. Ecological similarities have to be calculated by using different biodiversity measures. The use of higher taxonomic levels (taxonomic sufficiency) (Warwick, 1988 ; Arvanitidis et al., 2009) might be tested, as an alternative, to identify common patterns between estuarine ecosystems and to cope with species biogeographic distribution patterns. A complementary approach might include the biological trait analysis (BTA) in benthic assemblages (Bremner et al., 2006; Bremner, 2008; Marchini et al., 2008). Indeed, BTA provides a description of the functioning of the benthic community that may be closely related to ecological functioning (Bremner et al., 2006) as suggested by Elliott and Quintino (2007).

Although most BIs rely on species × abundance matrix to assess the EcoQ status of benthic communities, the study of biomass patterns may also provide additional information. This parameter is usually neglected because it is time-consuming. However, recent works conducted in transitional waters showed that there is a real potential of biomass-based indicators (Reizopoulou and Nicolaidou, 2007; Sabetta et al., 2007; Lavesque et al., 2009).

The above complementary measures could thus be tested and used to define benchmark levels (state and associated natural variability) describing the Biological Integrity of benthic communities occurring in given natural environmental conditions (intertidal/subtidal; poly-, meso-, oligo- or hypersaline conditions; level of turbidity and presence/absence of a maximum turbidity zone in estuaries) of transitional water habitats. These reference conditions must be obtained in the least impacted transitional waters in Europe. The Biological Integrity concept and its use for ecological quality assessment has been successfully developed in North America (*e.g.* Weisberg et al., 1997; Llanso et al., 2002a,b; Ranasinghe et al., 2002). This concept leading to a multi-metric, habitat-specific index has yet to be included in the European legislation. Though the M-AMBI and BQI provided the possibility to integrate habitat-specific references (Rosenberg et al., 2004; Muxika et al., 2006), the multi-metric (or multivariate) indices that have been proposed in Europe involve only a very few metrics (*e.g.* 3 metrics are used in M-AMBI (Muxika et al., 2006), 4 metrics in the Danish (DKI) and UK approaches, 3 metrics in ISI (see Borja et al., 2007), and 3 in the Portuguese approach (Teixeira et al., 2008)) whereas the B-IBI approach uses many more (4 to 7) metrics (Weisberg et al., 1997). We think that the use of such a low number of metrics does not allow for a sufficient description of the biological integrity of a benthic community (Lavesque et al., 2009). If benthic organisms display different patterns of response according to the type and level of perturbation, they will indeed behave in a complex manner. Detecting such complex patterns will require a larger set of variables that may be analysed using multivariate methods. The Marine Strategy Framework Directive has already taken the first steps towards this direction by examining the possibilities of the use of multi-metrics in the new indices to be used for the estimation and management of the good EcoQs.

Conclusion

Benthic communities have been widely accepted by the scientific community as a high potential Ecological Quality indicator. However, the current state of the understanding of the fundamental benthic invertebrate biodiversity patterns and functioning in transitional waters in response to perturbation may be limited to allow for an operational set of ecological quality indicators before 2015. The concept of the Biological Integrity measures emerges as one of the potential concepts which should be implemented in the forthcoming European legislation in order to obtain comprehensive biodiversity patterns and complement and integrate current measures.

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