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The use of biotopes in assessing the environmental quality of tidal estuaries in Europe

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A R T I C L E I N F O

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A B S T R A C T

In Europe, the Water Framework Directive (WFD) (European Commission 2000) – and the recently proposed EU Marine Strategy Directive – have established a framework for the protection of groundwater, inland surface waters, estuarine (transitional) waters and coastal waters. TheWFD has several objectives: to prevent water ecosystem deterioration, to protect and to enhance the status of water resources but the most important aspect is to achieve a 'Good Ecological Status' (GES) for all waters, by 2015. In essence, the WFD requires a water body to be compared against a reference condition and then its ecological status designated – if the water body does not meet good or high ecological status, i.e. it is in moderate, poor or bad ecological status, then remedial measures have to be taken (e.g. pollution has to be removed). Many indices were developed from benthic work and are often thought fit for purpose. Based on the successional model proposed by Pearson and Rosenberg (1978), most of these indices were effectively established for soft sediment benthos. However, those developed in the framework of the WFD were derived from work on the subtidal. They are difficult to use in the intertidal and in transitional waters. As they were derived from work on organic pollution, there is no or little evident link with chemical and physical pollution. Ecomorphology brings together a biological approach and a sedimentological approach to estuarine ecology. It considers the use of the biotope and related concepts (biocenosis, bio-facies, ecotone, habitat…) as a basis to a novel approach to environmental quality assessment. It addresses the problem of the estuarine quality paradox in recognising the role of nutrients and organic matter in biogeochemical cycles. The discussion shows the complementarity of biotopes with the Sato-Umi and the ecohydrolgy approaches.

1. Introduction

In Europe, the Water Framework Directive (WFD) (European Commission, 2000) – and the recently proposed EU Marine Strategy Directive – have established a framework for the protection of groundwater, inland surface waters, estuarine (referred to as "transitional" in the text of the directive) waters and coastal waters. As highlighted by Borja (Borja and Heinrich, 2005; Borja, 2005), it has several objectives: to prevent water ecosystem deterioration, to protect and to enhance the status of water resources but the most important aspect is to achieve a 'Good Ecological Status' (GES) for all waters, by 2015. In essence, the WFD requires a water body to be compared against a reference condition and then its ecological status designated – if the water body does not meet good or high ecological status, i.e. it is in moderate, poor or bad ecological status, then remedial measures have to be taken (e.g. pollution has to be removed). The WFD ecological status is defined in relation to the health of 5 biological elements in coastal and transitional waters of which 3 are benthic (the benthic macrofauna, macroalgae and the angiosperms such as sea grasses and salt marshes) – the others are phytoplankton and fishes (the latter is only assessed in transitional waters). The WFD centres on the influence of hydromorphology in affecting the biota although the chemical status of the water body is also assessed. The reference condition relates to what is expected for an area and is defined according to one of four ways: by choosing similar but unimpacted areas (i.e. a physical control similar to the test area but without human influences), by extrapolation (i.e. assessing what the area was like at some previous time), by deriving predictive models (i.e. predicting the benthic community of an area based on the physical characteristics – see below) and lastly, by using expert judgement.

Quantitative indices were developed in the framework of the WFD. Most of them were developed from benthic work. They are often thought to be fit to purpose. Based on the successional model proposed by Pearson and Rosenberg (1978), most of these indices were effectively established for soft sediment benthos (Dauer, 1993; Ducrotoy, 1998; Reiss and Kroncke, 2005; Fano et al., 2003;

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Dauvin et al., 2007; Zettler et al., 2007). In a review carried out by Diaz et al. (2004), 32 amongst the 64 indices considered were dealing exclusively with macrobenthic communities and many of these indices relate directly to organic enrichment. The most widely used in tidal estuaries are as follows: AMBI: Azti Marine Biotic Index (Borja et al., 2000); BENTIX: Biological Benthic Index (Simboura and Zenetos, 2002) was simplified from AMBI with only two categories of species; BQI: Benthic Quality Index (Rosenberg et al., 2004) relies on the calculation of the tolerance value of each species using ES50 which represents the probability of the number of species in a theoretical sample of 50 individuals (rarefaction); BOPA: Benthic Opportunistic Polychaetes/Amphipods ratio (Dauvin et al., 2007) was first proposed by Gesteira and Dauvin (2000) to compare frequencies of opportunistic polychaetes to amphipods, considered as sensitive to pollution (except Jassa spp.) such as metals, hydrocarbons, organic matter (Dauvin, et al., 1993; Gesteira and Dauvin, 2000); IB et I2EC : Indice Biotique et Indice d'Evaluation de l'Endofaune Côtière (Glemarec and Grall, 2000; Grall and Glemarec, 1997). All these indices were derived from work on the subtidal. They are difficult to use in the intertidal and in transitional waters. As they were derived from work on organic pollution, there is no evident link with chemical and physical pollution, except for BOPA. The synergy between pollutants is not well understood, for example with physical disturbances or the sedimentary dynamics (Rosenberg et al., 2004) in the case of dredging or in mobile sands. Most importantly, indices based on biodiversity cannot reflect estuarine communities functioning because the estuarine fauna and flora do not show recovery to maintain a full *k*-strategist complement; large individuals (both fauna and flora) are not present. In tidal estuaries, there is a naturally lower biomass/abundance ratio and higher abundance/species richness ratio, and the trophic system is dominated by organic/detritus-responsive invertebrates and nutrient reflecting algae. Most of these indices are not able to cope with the naturally low diverse areas in estuaries and other transitional waters. Elliott and Quintino (2007) have emphasized the difficulties and have produced discussions about the "estuarine quality paradox", which calls attention to the similarities between normal estuarine benthic fauna and flora and those subjected to anthropogenic stress. This type of anomaly has lead to refinements of many of the indices used for defining ecological status but without success.

The aim of this paper is to introduce the work conducted on biotopes in the framework of the ENCORA European Concerted Action in "ecomorphology". Then, biotopes are compared to indices in their ability to reflect quality in naturally organic matter enriched environments. The discussion opens to the ecosystem approach to environmental quality and shows its complementarity with the Sato-Umi (Yanagi, 2007) and the ecohydrolgy concepts (Wolanski, 2007).

2. The eco-morphological approach: bio-facies or biotopes

2.1. The bio-sedimentary approach

The bio-sedimentary approach is part of the eco-morphological methodology proposed by the ENCORA European Concerted Action. It can be applied to the study of changes in coastal biotopes (for a full presentation and discussion of biotopes, see Ducrotoy, 1998; Olenin and Ducrotoy, 2006) at selected sites. The aim of the method is to assess the nature and the scale of the changes that affect the geomorphology and ecology of coastal habitats, tidal estuaries in particular, in response to natural and human induced disturbances, including the global climate change and sea level rise. The universal potential of the research protocol was emphasised by Ducrotoy (1989, 1998) and Olenin and Ducrotoy, (2006). The approach deals with sedimentary processes and how to interpret them in the context of ecologically sensitive areas. It is based on the definition of bio-facies or biotopes.

Biotopes have been used extensively during the last decade as tools for managers in relation to the classification of coastal zones and marine areas. Connor (1995a, b, 2004) described marine benthic biotopes using a large-scale multivariate analysis of the faunal community types and the environmental characteristics using TWINSPAN (Two-Way Indicator Species Analysis). Inspired by this work, Olenin and Ducrotoy (2006) have extended the use of the concept to research in functional ecology and possible applications in the framework of the WFD. In actual fact, biotope may be viewed not only as a structural unit convenient for mapping a coastal zone but also a sub-unit of the ecosystem emphasizing its own processes. These processes will change according to the biotope. Thus, once their biological characteristics have been taken into account, biotopes differ not only in their structure but also in their functions, which they perform in coastal marine ecosystems: production, storage and distribution of organic material; reproduction of biological resources; modification of bottom sediments, etc. As ecosystems are considered as cybernetic and self-controlling, biotopes reconciliate the divisive controversy between the population-community view (networks of interacting populations) and the process-function approach (biotic and abiotic components). Because, in their extended definition, biotopes can be considered as functional units of a coastal marine ecosystem, they can be used as indicators of change due to various pressures, including human impacts. The concept of the biotope (or bio-facies) further helps to determine the type and number of measurements, frequence, and type of data set required. It allows to selecting observations amongst many possibilities. Following such a methodology, a benthic biotope index was recently developed for classifying habitats in the Sado estuary in Portugal (Caeiro et al., 2005). The index was initially derived from benthic composition and structure (TWINSPAN) but discriminant analysis was used to combine benthic community metrics. A subset of the physical and chemical parameters allowed the authors to discriminate seven biotopes. The Benthic Biotope Index B*I*bio was used for predicting biotopes at selected stations after the data was divided into a prediction and validation subset.

2.2. The spatial dimension of benthic biotopes

Diaz et al. (2004) indicated the necessity for (and recent advances in) benthic mapping techniques and discussed cost-effective ways of obtaining information needed by managers but also of linking the physical and biological aspects. A summary of the bio-sedimentary approach follows. It requires a good knowledge of the benthic system and, at the very least, the dominant organisms in each habitat type.

2.2.1. Zoneography

Firstly the estuary is characterized according to its main geomorphological features such as shingle and sand dunes, bars, channels, swell-surges, shell beds, high production areas, animal banks (*Pygospio* sp. *Sabellaria* sp.), sand ridges and ripple-marks, wind erosion areas. Remote-sensing and aerial pictures give valuable information as functional ensembles arise (Dupont, 1981, 1983). Sedimentary dynamics parameters enable distinguishing between shore-bars, the outer pseudo-delta, strands and mud-flats and ebbing tide currents. Dynamical features are deduced from sediment grain size analysis (AFNOR or other standard). This zoneography leads to the establishment of a morpho-sedimentay units chart, based on geo-morphological assemblages, dynamical limits and other sediments characteristics (carbonate, organic