Estuarine, Coastal and Shelf Science 86 (2010) 317–321

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

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

Jean-Paul Ducrotoy

Institute of Estuarine and Coastal Studies, The University of Hull, Cottingham Road, Hull HU6 7RX, United Kingdom

ARTICLE INFO

Article history:
Received 6 February 2009
Accepted 26 May 2009
Available online 11 June 2009

Keywords:
Water Framework Directive
biotope
estuary
quality paradox
Integrated Coastal Zone Management

ABSTRACT

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;

E-mail address: j-p.duc@wanadoo.fr

0272-7714/$ – see front matter © 2009 Published by Elsevier Ltd.
doi:10.1016/j.ecss.2009.05.023

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 content, etc.). Six standard units were proposed by Ducrotoy (1991): upper estuarine, sheltered estuarine, estuarine, sheltered coastal, high energy, shore-bar outer and tidal delta. This map is the base to selecting sediment sampling sites to carry out chemical analysis (i.e. metal content). The macrobenthic fauna is sampled qualitatively with a 1/50 m^2 corer through a 1 mm sieve (or semi-quantitatively) in every estuarine unit. These animals are strictly linked to the sediment and data about biota and sediment as a whole make up an entity: the bio-sedimentary facies.

2.2.2. Definition of bio-facies or biotopes

The spatial distribution of macrobenthos invertebrates in sediments is related to the physical characteristics of the substratum, particularly the hydrodynamics gradient in estuaries. Combining sediment characteristics and faunal composition leads to the concept of bio-facies (Ducrotoy, 1989). Equivalent to a biotope, a facies results from a balance between:

- Hydrodynamic parameters controlling the dynamical processes in the estuary;
- Phvsico-chemical parameters, governed by the former conditions: a) salinity is controlled by the river input and the comparative extent of the inner estuarine area, b) pollutant impact depends on the chemical's concentration, the freshwater flow and other continental inputs (runoff, etc.);
- The geomorphology of the estuary producing sheltered areas to some extent;
- The regional sedimentary characteristics governing the type of deposit;
- The regional living environment including recruitment of estuarine organisms depending on local conditions (station concept).

2.2.3. Community chart

The bio-sedimentary facies are organized along a sequence lengthwise from the marine open-waters to the higher sheltered estuary, following an ecological gradient. Along the sequences and in every estuarine unit, transects are sampled to illustrate the hydrodynamical and sedimentological gradients and to assess the benthic faunal quantitative parameters (numbers, biomass). Only then is it possible to stratify sampling and plan a quantitative programme along transects. Different methods have to be used according to the size of animals (species or stage of development) and sediment quality. The quantitative data are then subjected to multivariate analysis and cluster analysis assemblage dendrograms (centroid clustering where inter-species similarities are based on the logarithmic abundance on sites using the Euclidian measure of distance Depiereux, 1983; Green et al., 1995). The multivariate analysis of the data on intertidal species distribution puts into light faunal assemblages typical of each sedimentary facies. From the use of a standardised methodology, it is possible to generalise the communities' ecological properties.

2.3. Definition and survey of pilot-stations

A pilot-station is defined as representative of a specific community, according to the nature of the bio-facies. Such sampling sites are recommended in the framework of the WFD. They require to be defined as representative of every specific biotope, according to the nature of each bio-facies. Sampling is to be done at species level to quantify the temporal variability of populations, understand population dynamics, detect any consequences of disturbances, and identify species whose biological cycle integrates the variations of the environmental factors at different levels. Pilot-stations should be selected with a view at monitoring the quality of representative biotopes, providing hypothesis to explain possible observed changes and detecting changes at regional and local scales, on mid and long-term, the main aim being the identification of signals to predict future changes in the functioning of the considered estuary. The sampling schedule should be site-specific. Considered as minimal, bi-annual sampling should be at the times of maximum and minimum abundances in order to determine the greatest range in number and biomass.

3. Confronting indices and biotopes

Through the use of biotopes, a spatial perspective is a valuable complement to temporal data in helping to distinguish between the effects of local (often man-made) and broad scale (often natural or due to global warming) influences. It is essential biologically to assess the recruitment variation and its causes, especially its relationship with latitudinal physical gradients and this can be done on pilot-stations. It is desirable to link together long-term, wide-scale field work and localised, highly intensive field/laboratory experimental studies into factors influencing success or failure in repopulation (Souprayen and Ducrotoy 1991). Multicriteria methods consist in not only combining biotic indices (if inapplicable to some environmental conditions, combining them just add mistakes together), but rather combining multivariate approaches with pathological studies, physiological approaches or information on pressures from human activities (Ducrotoy, 1998).

Most indices were developed for soft sediment environments. They are not easily applied to coarser sediments or rocky environments. Biotopes can be indicators for mobile, coarse sediments (gravels/coarse sand) since these are increasingly subject to human activity. For instance, most offshore wind farm developments are proposed for areas of mixed and often coarse, mobile sediments. Benthic community structure and species population dynamics are variable in such environments due to natural environmental fluctuations. However, at present, there is no objective or satisfactory means of distinguishing between natural variation and an impact except in mapping biotopes and checking changes in time in their distribution. There is a requirement for the development of a full methodology based on biotopes applicable to these habitats, together with the identification of indicator species if they exist.

Ecology and biology of many species is not sufficiently known. Classifications are therefore not so robust. As for the establishment of biotopes, the calculation of indices such as AMBI should rely on sharp knowledge of the biology of all species at species level, not higher taxonomic levels. The need for consistent work in biology should be acknowledge. It does not seem a good strategy to reduce the number of ecological groups as in BENTIX. Species at their limit of their geographical distribution might show a different vulnerability to pollution. Similar doubt arises for species at the limit of their tolerance to pollution. There is a need to take into account the variability of environmental conditions in space and time (in particular seasonally) and take into account salinity, changes in temperature, UV radiations… Similarly, the approach using *r*, *K* and *T* strategies is reductionist. Considering *K* strategists as more sensitive to pollution does not take into account intermediary instances. Some *K* strategists might endure longer anoxia periods than r strategists. The use of biotopes, in considering paucispecific communities as part of a network of biotopes in an estuary, does not lead to assess the ecosystem as a whole as of bad quality. On the contrary, the biotope methodology allows determining whether an estuary is still functioning as an estuary because it uses functional symptoms as well as structural ones. In this spirit, (Elliott and Quintino, 2007) suggested that rather considering estuaries as naturally stressed areas, one should consider the definition of a subsidy rather than a stress, as a perturbation with a positive effect on the system (Costanza et al., 1993; Yanagi, 2007). The positive effect is shown by the ability of organisms which can tolerate the adverse and variable environmental conditions in an estuary to capitalise on the lack of inter-specific competition and thus achieve high population densities.

4. Towards an ecological approach: a parallel with the Sato-Umi and ecohydrology concepts

Concentrating complex information such as ecosystems quality in a single number from the calculation of an index from data collected on a single sampling site may seem illusory. One may question whether it is the best way to convey scientific information to managers who wish to get a quick and simple answer to their concerns, particularly in the framework of the WFD? In fact, because they intended to provide an over simple tool, scientists have attempted to integrate two diverging concepts. They have tried to consider structural elements (species richness, biological diversity, taxonomic distinctness), together with a functional property of ecosystems, which deals with the sensitivity and the tolerance of its constituting species to disturbances (Gray and Elliott, 2009). Problems arise from this integration. One other weakness of the approach is that the sensitivity of species has mainly been assessed against organic enrichment. The proposed indices preclude that all other conditions are uniform in time and space. However, the chemistry of estuaries is complex and the environmental parameters considered (i.e. salinity) do not change in a monotonous fashion along simple gradients (Mc Lusky and Elliott, 2004). In addition, the nature of disturbances in relation to local environmental conditions has been little considered, in particular the influence of the watershed. Background noise might prevent indices to discriminate from anthropogenic disturbances as parameters considered are mostly related to conditions which are considered as disturbances in the ecosystems in question (Gray and Elliott, 2009).

Faced to similar difficulties in Japan, scientists have attempted to propose an integrated approach to pollution and other human disturbances. The concept of Sato-Umi (Yanagi, 2007) has recently been developed in response to problems in managing nutrient enriched enclosed coastal seas such as the Seto Inland Sea. The concept attempts to reconcile the nutriment richness of certain coastal ecosystems and their so-called quality in terms of goods and services provided to humans. It is defined as "High productivity and biodiversity in the coastal area with human interactions" (Yanagi, 2007). Sato-Umi addresses the problem of the estuarine quality paradox in recognising the role of nutrients and organic matter in the biogeochemical cycles of enclosed or semi-enclosed coastal systems.

Finally, one cannot consider an estuary without considering the ecology of its watershed. The ecological health of estuaries is determined by the interaction between organisms and variations in salinity, currents, waves, suspended particulate matter, bed sediments, temperature, air exposure, hypoxia, wetland contaminants and biodiversity. Like the health of a living organism, the health of an estuary or a coastal water body, cannot be measured by one single variable, indeed a number of variables are important, all being influenced by the watershed. In essence, this is what supports the ecohydrology concept (Wolanski et al., 2006; Wolanski, 2007). It links physical, chemical and biological processes over the entire estuary for the entire food web as a function of the catchment's output and the oceanic open boundary conditions. Future research should consider integrating the biotope level into future ecohydrology models.

References

Borja, A., 2005. The European water framework directive: a challenge for nearshore, coastal and continental shelf research. Continental Shelf Research 25, 1768–1783.

Borja, A., Heinrich, H., 2005. Implementing the european water framework directive: the debate continues. Marine Pollution Bulletin 50, 486–488.

Borja, A., Pérez, V., Franco, J., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. Marine Pollution Bulletin 40, 1100–1114.

Caeiro, S., Costa, M.H., Goovaerts, P., Martins, F., 2005. Benthic biotope index for classifying habitats in the sado estuary: Portugal. Marine Environmental Research 60, 570–593.

Connor, D., 1995a. The development of a biotope classification in Great Britain and Ireland – principles and structure of classification. In: Hiscock, K. (Ed.), Classification of Benthic Marine Biotopes of the North-east Atlantic. Proceedings of a BioMar-Life Workshop Held in Cambridge. 16-18 November 1994. Joint Nature Conservation Committee, Cambridge UK, Peterborough, pp. 30–46.

Connor, D.W., 1995b. A classification system for benthic marine biotopes. In: Eleftheriou, A., Ansell, A.D., Smith, C.J. (Eds.), Biology and Ecology of Shallow Coastal Waters. Olsen & Olsen, Fredensborg, pp. 155–166.

Connor, D.W., 2004. The marine habitat classification for Britain and Ireland. Version 04. 05 JNCC, Peterborough (internet version). Available from: www.jncc.gov.ukMarineHabitatClassification.

Costanza, R., Kemp, W.M., Boynton, W.R., 1993. Predictability, scale, and biodiversity in coastal and estuarine ecosystems – implications for management. Ambio 22, 88–96.

Dauer, D.M., 1993. Biological criteria, environmental-health and estuarine macrobenthic community structure. Marine Pollution Bulletin 26, 249–257.

Dauvin, J.C., Ruellet, T., Desroy, N., Janson, A.L., 2007. The ecological quality status of the bay of seine and the seine estuary: use of biotic indices. Marine Pollution Bulletin 55, 241–257.

Dauvin, J.C., Dewarumez, J.M., Elkaim, B., Bernardo, D., Fromentin, J.M., Ibanez, F., 1993. Changes in Abra-Alba (Mollusca, Bivalvia) populations in the English-channel and North-sea, 1977–1991-relation with climatic factors. Oceanologica Acta 16, 413–422.

Depiereux, E., 1983. Utilisation critique de l'analyse en composantes principales et du cluster analysis pour la description d', chantillons d'invert, br, s benthiques en eau douce. Oikos 40.

Diaz, R.J., Solan, M., Valente, R.M., 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. Journal of Environmental Management 73, 165–181.

Ducrotoy, J.P., 1989. General methods of study of macrotidal estuaries: the biosedimentary approach. In: Macmanus, J., Elliott, M. (Eds.), Developments in Estuarine and Coastal Study Techniques. Olsen & Olsen, Fredensborg, pp. 41–52.

Ducrotoy, J.P., 1991. Baie des Veys and baie de Somme (English Channel): comparison of two macrotidal ecosystems. In: Elliott, M., Ducrotoy, J.-P. (Eds.), Estuaries and Coasts: Spatial and Temporal Inter Comparisons. Olsen & Olsen, Fredensborg, pp. 207–210.

Ducrotoy, J.P. 1998. Qualité écologique des milieux estuariens et littoraux: Application à la Manche et à la mer du Nord. MSc Thesis, Université de Caen, Agence de l'Eau Seine Normandie

Dupont, J.P. 1981. Relations entre bios et phénomènes sédimentaires intertidaux: le modèle de la Baie de Somme. Thèse de Troisième Cycle de l'Université de Haute-Normandie: 1–310.

Dupont, J.P., 1983. Les séquences biosédimentaires de la Baie de Somme. Actes du Muséum de Rouen, 1–3.

Elliott, M., Quintino, V., 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Marine Pollution Bulletin 54, 640–645.

European Commission, 2000. Water framework directive. European Communities Official Journal L327, 1–72.

Fano, E.A., Mistri, M., Rossi, R., 2003. The ecofunctional quality index (EQI): a new tool for assessing lagoonal ecosystem impairment. Estuarine Coastal and Shelf Science 56, 709–716.

Gesteira, J.L.G., Dauvin, J.C., 2000. Amphipods are good bioindicators of the impact of oil spills on soft-bottom macrobenthic communities. Marine Pollution Bulletin 40, 1017–1027.

Glemarec, M., Grall, J., 2000. Ecological and zoological groupings within marine invertebrates in relation to coastal perturbations. Bulletin de la Société Zoologique de France 125, 37–48.

Grall, J., Glemarec, M., 1997. Using biotic indices to estimate macrobenthic community perturbations in the Bay of Brest. Estuarine Coastal and Shelf Science 44, 43–53.

Gray, J.S., Elliott, M., 2009. Ecology of Marine Sediments. From Science to Management, second ed. Oxford University Press, Oxford. 225pp.

Green, M.O., Vincent, C.E., Mccave, I.N., Dickson, R.R., Rees, J.M., Pearson, N.D., 1995. Storm sediment transport – observations from the British North-sea shelf. Continental Shelf Research 15, 889–912.

Mc Lusky, D., Elliott, M., 2004. The Estuarine Ecosystem – Ecology, Threats and Management. Oxford Univerity Press, Oxford. 214pp.

Olenin, S., Ducrotoy, J.P., 2006. The concept of biotope in marine ecology and coastal management. Marine Pollution Bulletin 53, 20–29.

Pearson, T.H., Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology, Annual Review, 229–311.

Rosenberg, R., Blomqvist, M., Nilsson, H.C., Cederwall, H., Dimming, A., 2004. Marine quality assessment by use of benthic species-abundance distributions: a proposed new protocol within the European Union Water Framework Directive. Marine Pollution Bulletin 49, 728–739.

Reiss, H., Kroncke, I., 2005. Seasonal variability of infaunal community structures in three areas of the North Sea under different environmental conditions. Estuarine Coastal and Shelf Science 65, 253–274.

Simboura, N., Zenetos, A., 2002. Benthic indicators to use in ecological quality classification of Mediterranean soft bottom marine ecosystems, including a new Biotic index. The Mediterranean Marine Science 3, 77–111.

Souprayen,J., Ducrotoy, J.P., 1991. Numerical analysis of long-term intertidal sedimentary macrozoobenthic communities. In: Keegan BF. (Ed.), Space and time series data analysis in coastal benthic ecology. Commission of the European Communities, Brussels. EUR 13978, EN: 65–236.

Wolanski, E., 2007. Estuarine Ecohydrology. Elsevier, Amsterdam Boston Paris. 157pp.

Wolanski, E., Chicharo, L., Chicharo, M.A., Morais, P., 2006. An ecohydrology model of the Guadiana estuary (South Portugal). Estuarine Coastal and Shelf Science 70, 132–143.

Yanagi, T., 2007. Sato-Umi: a New Concept for Coastal Sea Management. Terrapub, Tokyo. 86pp.

Zettler, M.L., Schiedek, D., Bobertz, B., 2007. Benthic biodiversity indices versus salinity gradient in the southern Baltic Sea. Marine Pollution Bulletin 55, 258–270.