Ecological Indicators 19 (2012) 118–129

Contents lists available at ScienceDirect

Ecological Indicators

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

The ecological quality status of the Elbe estuary. A comparative approach on different benthic biotic indices applied to a highly modified estuary

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ARTICLE INFO

Keywords:
Sediment contamination
Water framework directive
Benthic indicators
W-statistic
BOPA
BO2A
AMBI
M-AMBI

ABSTRACT

Indices to assess the ecological status of water bodies according to the European Water Framework Directive (WFD) frequently produce widely differing results when applied to estuarine systems. Although several ecological indices have been employed to coastal environments and in estuaries in particular, there is still a lack of knowledge about their suitability for assessing the ecological status of heavily modified water bodies. Thus, we evaluated the performances of indices and fauna parameters (AMBI, M-AMBI, BOPA, BO2A, *W*-value, Shannon diversity, species richness, abundance) that have been discussed in the WFD context using data on invertebrates dwelling in two typical morphological units: the navigation channel and the river bank habitats of Elbe estuary (Germany). In addition, we tested their ability to identify several environmental factors (grain size distribution and chemical sediment contamination). All indices were able to detect major changes in macrofauna composition along the estuarine salinity gradient and were able to differentiate between navigation channel and shallow bank habitats. A strong significant correlation was found with most indices with the exceptions of the *W*-value and the BOPA with mean grain size. Almost all indices signaled poor ecological quality in the coarser fairway sediments against the finer sublitoral bank sediments. However, AMBI and BOPA showed the opposite: both indicators classified the invertebrate assemblages from the navigation channel better compare than the shallower habitats. The correlation of ecological indices and parameters with sediment contaminants and the toxicity of the sediment calculated as toxic units showed a diverse picture: all indices, except species richness and the BOPA, had a certain significant correlation with several individual sediment pollutants, however, only one index, the *W*-value, was correlated significant with the majority of chemical pollutants (Pb, Cd, Cu, Ni, Hg, Zn, β-HCH, pp′-DDD, and TBT) and the toxic units. Our results show clearly that ecological quality classification of heavily modified estuaries depends strongly on both the index and the habitat. Thus, we conclude that no index should be used on its own to estimate the ecological quality of estuaries. Further investigations and the improvement or development of such indices should place emphasis on their independence from the grain size spectrum of the sediments and on their good correlation with its pollution status.

1. Introduction

The establishment of the European Water Framework Directive (Directive 2000/60/EC of the European Parliament and of the Council) for the Community action in the field of water policy requires the definition of aims to achieve a "good" ecological quality (EcoQ) status of all water bodies including the estuaries by 2015. The first step towards this goal consists in assessing the current status of these water bodies. Benthic macrofauna plays a vital part in the assessment of the EcoQ, because they are an important component in the aquatic ecosystems and they may serve as sensitive indicators (Rosenberg et al., 2004). Therefore, the Water Framework Directive (WFD) calls for the development of tools for defining the EcoQ status of bodies of water. Several attempts have been made in the past to develop an index based system to estimate the EcoQ status which allows the translation of biological information, such as presence and abundance of macrofauna species, into five different EcoQ classes (high, good, moderate, poor, and bad): the Benthic Index (BI, Grall and Glémarec, 1997), the Azti

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doi:10.1016/j.ecolind.2011.08.007

Marine Biotic Index (AMBI Borja et al., 2000), the M-AMBI (Borja and Franco, 2004; Muxika et al., 2007), and the biotic index BENTIX (Simboura and Zenetos, 2002). These indices use lists of species where species are assigned to ecological groups according to their sensitivity to stress. The index calculation is usually performed with a relatively simple formula. However, further simplification was suggested by Dauvin and Rullet (2007): their index requires only information about the overall abundance of opportunistic polychaetes and amphipods.

Besides these recently developed indices, which were especially constructed to meet the requirements of the WFD, several community-descriptive parameters and indices exist that have been used in conjunction with the demands of the WFD. For instance species richness (e.g. Marín-Guirao et al., 2005; Simboura and Reizopoulou, 2007; Simboura and Zenetos, 2002; Dauvin et al., 2007; Borja et al., 2000), total abundance (e.g. Simboura and Reizopoulou, 2007; Dauvin et al., 2007; Borja et al., 2000), Shannon–Wiener diversity (e.g. Teixeira et al., 2007; Marín-Guirao et al., 2005; Simboura and Reizopoulou, 2007; Simboura and Zenetos, 2002; Dauvin et al., 2007; Borja et al., 2000; Labrune et al., 2006), Margalef diversity (e.g. Teixeira et al., 2007; Salas et al., 2004) and the *W*-value (e.g. Teixeira et al., 2007; Marín-Guirao et al., 2005; Salas et al., 2004), an index that evolved from abundance-biomass comparison (ABC) distribution curves (Warwick, 1986; Clarke, 1990).

WFD related studies using the AMBI on its own, or in combination with other indices have been carried out in a number of cases along the European coasts from the Baltic Sea (Muxika et al., 2005), the English channel (Dauvin et al., 2007), the Atlantic coast (Borja et al., 2000, 2003; Salas et al., 2004; Muxika et al., 2005) to the Mediterranean (Simboura and Reizopoulou, 2007; Marín-Guirao et al., 2005; Muxika et al., 2005) and some studies even report their use in estuarine systems (Borja et al., 2000, 2003; Salas et al., 2004, Muxika et al., 2005; Dauvin et al., 2007; Teixeira et al., 2007). In general, applicability of the AMBI in estuarine systems proved to be successful. The index was used to detect different anthropogenically induced changes (Borja et al., 2000), approximated the distribution of organic matter and sediment grain size (Borja et al., 2003) and detected pollution point-sources (Muxika et al., 2005).

Estuaries are the most productive and vulnerable marine coastal environments. Here, nutrient-rich freshwater mixes with highly oxygenated waters from the seas, making them one of the biologically most productive and vulnerable regions of the marine environment (Correll, 1978). In addition, estuaries themselves are specific habitats, characterized mainly by strong gradients (salinity, temperature) and by changes and fluctuations of these gradients due to the tidal regime making them unique habitats for a variety of brackish-water species. Moreover, they are also the anthropogenically most altered aquatic systems and susceptible to numerous and strong amounts of pressures. Estuaries have long been influenced by human activities like, dyke constructions, dredging, and pollution, so that today most estuarine ecosystems in industrialized regions are far from being a natural environment and can be considered as strongly disturbed ecosystems.

One of the most prominent morphological features of modified estuaries is the presence of a deep navigational channel (fairway) that was prepared in place of shallow-water areas. Large rivers and their estuaries are important routes for navigation and the increasing sizes of ships in the world's merchant fleets demands regular widening and deepening of the fairways. In the Elbe estuary navigational channel adjustments (fairway deepening) has been carried out gradually in the 20th century, starting with deepening from a primary depth of 3–4 m to 9 m in 1910. Then in 1930, the navigational channel was dredged down to 10 m, in 1962 to 11 m, from 1964 to 1969 to 12 m, and between 1974 and 1978 to 13.5 m. In 1999, a navigational depth of 14.4 m was reached. One consequence of this development was shrinking of the highly productive shallow water areas (water depth < 2 m); between 1896/1905 and 1981/1982 the Elbe estuary lost 26% of its shallow waters (Schirmer, 1994). The major requirement for an ecological index is its universal usability. In the case of the Water Framework Directive, benthic marine indices this means that they should be applicable in all marine environments ranging from undisturbed off-shore benthic communities to heavily modified water bodies like estuaries. Although, some indices have already been tested in estuarine environments (e.g. Borja et al., 2003; Salas et al., 2004; Muniz et al., 2005; Muxika et al., 2005; Puente and Diaz, 2008; Ranasinghe et al., 2009), no differentiation was made so far between different morphological structures within estuaries (navigation channel vs. shallow-water areas).

The aim of this paper is to explore: (1) the suitability of the different indices in a highly modified estuary with the typical morphological structure elements of a deep navigation channel (fairway) and shallow-water areas, and (2) their ability in identifying different environmental factors like sediment grain-size distribution and its chemical contamination.

2. Material and methods

2.1. Study site

The Elbe estuary is located at the southern coast of the North Sea and discharges the River Elbe (catchments size 148,268 km^2) into the Wadden Sea. The estuary is characterized by diurnal mesotidal conditions (the mean tidal range at the Cuxhaven tidal gage is about 3 m) and by water temperatures ranging from approximately 0 °C in the winter to 26 °C in summer. Salinity can fluctuate between 0.3 and 2.6 PSU in the inner part of the estuary (station Grauerort, river-km 660.6) and from 1.2 to 22 PSU at the mouth (station Cuxhaven, river-km 725.2) depending on season, river runoff, and tidal cycle. The estuarine water body is usually completely mixed due to tidal currents and stream flow; however, when the tide is turning stratification may occur for short intervals (Carstens et al., 2004). The transitional water body of the Elbe estuary extends from river-km 630 to km 727.7 (Office of the River Elbe Water Quality Board, Arge-Elbe, www.arge-elbe.de) and covers an area of some 500 km^2.

Anthropogenic modifications of the Elbe estuary have been going on since several centuries. Since the beginning of the 20th century, the estuary has been successively adjusted to the increasing average size of the ships in the merchant fleets to ensure the safe navigation to the port of Hamburg (Schuchardt et al., 1999); the last major fairway adjustment was carried out in 1999/2000 and the next action is already planned for. Today, a safe navigation depth of 14.4 m is maintained; annual maintenance dredging activities in the Elbe estuary move between 5 and 10 Mio. m^3 of sediment per year. Most of the dredged sediment is relocated within the estuary (Rolinski and Eichweber, 2000).

Pollution release into the river and the estuary declined considerably following the re-unification of Germany in 1990, as a result of the construction of sewage treatment plants, closure of polluting factories, and changes made in industrial production processes. The annual pollution discharge of the River Elbe has dropped from 1986 to 2007 by 29% to almost 100% in some cases, depending on the pollutant (Arge-Elbe, 2007), although historical pollutants are still present in sediments. In the estuary pollutant concentrations usually follows a declining gradient, with higher concentrations occurring close to the port of Hamburg and lower concentrations towards the mouth of the estuary. Higher concentrations are usually found also in zones of low flow velocities, where suspended particles and sediment-bound pollutants (e.g. trace metals)