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Theoretical and practical aspects on benthic quality assessment according to the EU-Water Framework Directive – examples from Swedish waters

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ABSTRACT

A previously presented objective method to calculate each species sensitivity to disturbance is here slightly modified and implemented in the Benthic Quality Index (BQI) for marine benthic invertebrates. A framework for assessment of water bodies based on multi-site BQI-values is also presented, where a certain variation of BQI-values is allowed to cover the heterogeneity within each water body. The 20th percentile, using bootstrapping, from the available sites' BQI-values is compared with the status boundaries for quality assessment. The reliability of the assessment depends on the background information available for the boundary setting as well as the number of sampling sites included in the assessment. Agreement between time series of quality assessments in areas with known changes in anthropogenic disturbances is encouraging. Problems associated with water body assessment based on few or no samples, as well as multiple sampling occasions during the 6-yr WFD cycle are discussed.

1. Introduction

The escalating anthropogenic effects on coastal marine ecosystems became obvious towards the end of the last century. One of the prime factors for this severe disturbance is eutrophication leading to enhanced primary production and subsequent organic enrichment of the seabed. In stratified and enclosed coastal areas this enrichment could lead to the development of hypoxia and anoxia, and the number of dead zones in the world has doubled every decade since the 1960s (Diaz and Rosenberg, 2008).

The benthic fauna has proven to be ideal for environmental quality assessment in sedimentary habitats; most species live for several years, they are rather stationary, and by these features integrate the benthic habitat and near-bottom water quality. It has been shown in several studies that some species, often referred to as opportunists, are tolerant to organic pollution, whereas sensitive species are almost never found in polluted areas. Thus, the benthic species composition and abundance change gradually along gradients of disturbance and the community structure is a strong diagnostic tool of the environmental quality as demonstrated by the Pearson–Rosenberg model (Pearson and Rosenberg, 1978). This conceptual framework has a wide acceptance (Heip, 1995) and gives the theoretical background to most environmental quality assessments and the development of most indices used in this context (Puente and Diaz, 2008).

Managers of the environment have a wish that a universal single index number could tell the quality status. Instead scientists have provided more and more indices, and their different qualities are difficult to evaluate (Diaz et al., 2004). The European Union Water Framework Directive (WFD) states that the presence of sensitive taxa, and taxa indicative of pollution together with diversity and abundance of the benthic fauna should be used for measuring the status of the sedimentary habitat. The status of all European coastal waters should be classified as one of the categories: *High, Good, Moderate, Poor* or *Bad.* For that purpose several new multimetric indices have recently been developed and suggested useful in different European coastal areas; most of them described by Pinto et al. (2009). One index that attracted several scientists was AMBI (Borja et al., 2000), an index based on the Pearson–Rosenberg model where the species are listed into different categories based on literature data of their sensitivity and tolerance to disturbance. Another multi-metric index applied to different data sets are the Benthic Quality Index (BQI) (Rosenberg et al., 2004). The use of new indices has also expanded in the USA (e.g. Weisberg et al., 1997). Different indices correlate generally rather well to subtidal marine perturbations, but if a dominant indicator species is classified differently by different methods, the result will diverge (Labrune et al., 2006).

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The BQI is, in the Skagerrak and the Kattegat on the Swedish west coast, based on an objective classification of the sensitivity of species sampled from gradients including both heavily disturbed and rather undisturbed areas. The assumption is that sensitive species only occur in samples with high diversity, and tolerant species are to be found predominantly in samples with low diversity (Rosenberg et al., 2004). With this method, the sensitivity values are calculated from samples along gradients from heavily disturbed to rather undisturbed areas. The most tolerant species, i.e. the opportunist polychaete *Capitella capitata,* will score a number close to 1, and very sensitive species up to around 18. An accurate assessment of species sensitivity values is necessary for a qualified diagnosis and needs to be based on a large data set. The species abundance and total number of species at a station also have an influence on the BQI, but to a minor degree. For a useful status index the sensitivity values need to span over a wide range of values to increase the resolution in the assessment and to prevent low influence of the sensitivity values compared to the other components of the index. For the numerical method to derive sensitivity values a high diversity is preferred, as in the Skagerrak and the Kattegat on the Swedish west coast. In the Baltic, low salinity reduces the species number considerably. Consequently, Baltic species will score low values, within a narrow range, by this method. Here we describe the BQI-method (Rosenberg et al., 2004) with some clarifications since it has been used incorrectly in some papers (e.g. Fleischer et al., 2007). We also provide a list of species sensitivity values calculated from a large dataset from the Kattegat-Skagerrak area, and describe an alternate species classification method for the salinity stressed east coast of Sweden.

The BQI as well as most other benthic indices were developed for the use on single samples. The European WFD requires the ecological status of entire water bodies to be classified. One of the aims with this paper is therefore to present a strategy that aggregates BQI from several sites for classification of water bodies (the Swedish method). The aggregation method is general and independent of the index formulation. Data from the Swedish monitoring programmes on benthic invertebrates, both west and east coast, were used as examples to enlighten the different aspects of BQI and its use in status classification.

2. Material and methods

2.1. The Benthic Quality Index (BQI)

The Swedish Benthic Quality Index (BQI) (Rosenberg et al., 2004) is based on the ^{10}Log(number of species + 1) times the summed occurrence of species weighted by each species specific sensitivity to disturbance. In the Baltic proper several of the sensitive benthic species are mobile (e.g. the swimming amphipod *Monoporeia affinis*) with the ability to quickly re-colonize previously disturbed areas. While their re-colonization indicates improvement of the environment, the mobility also means that individuals accidentally can end up in a polluted environment. If being captured in a grab during such ''accidental'' dispersal, outliers would appear in the quality assessment. This phenomenon was apparent in time series of BQI from sites in the Baltic Proper where the oxygen concentration decreased towards zero during a number of years. The abundance dropped drastically due to anoxia, but occasional appearance of a few individuals caused peaks in the BQI, without the abundance adjustment, despite the poor environment. To reduce the influence of such situations an abundance factor was multiplied by the original BQI: $N/(N+5)$, where N is the total number of individuals per sample (0.1 m^2), and 5 corresponds to the half saturation constant. The half saturation constant defines the abundance at which the adjustment will be 0.5. The abundance factor reduces the index value considerably when there are less than ca 20 individuals per sample (0.1 m^2). At higher abundances this factor has minor influence. The parameterization of this adjustment was based on expert judgments. On the west coast of Sweden the combination of few individuals and high sensitivity values are rare and the influence of this factor is therefore minor. In contrast, the abundance factor has more impact on the BQI in several coastal areas of the Baltic Proper and parts of the Gulf of Bothnia. In low productive parts of Bothnian Bay, abundances below 20 individuals per sample (0.1 m^2) are frequent also in undisturbed areas (National and Regional monitoring data from 80 annual sampling sites in the Bothnian Bay during 1995–1997). BQI will generally be low in these areas due to the abundance factor. Thus, a half saturation constant of 5 is a compromise. Having a lower value increased the risk of falsely accepting a bad or poor environment as good, while higher values considerably reduced the range of BQI-values in the northern Baltic. The complete formulation of the BQI used for the Swedish assessment within the WFD is:

$$BQI = \left[\sum_{i=1}^{S_{classified}} \left(\frac{N_i}{N_{classified}} * Sensitivity\ value_i\right)\right] * \log_{10}(S+1) * \left(\frac{N_{total}}{N_{total}+5}\right), \quad (1)$$

where $S_{\text{classified}}$ is the number of taxa having a sensitivity value, N_{i} is the number of individuals of taxon i, $N_{\text{classified}}$ is the total number of individuals of taxa having a sensitivity value, the *Sensitivity value*$_{\text{i}}$ is the sensitivity value for taxon i, S is the total number of taxa, and N_{total} is the total number of individuals in the sample (0.1 m^2). Taxa not given a sensitivity value are excluded from the sensitivity factor but included in the total number of species and abundance factors when calculating BQI.

In the monitoring programmes in Sweden some taxonomical groups are commonly not determined to the species level. In the assessment these groups are treated as groups since species determinations only have been performed in a few programmes. This applies for Chironomidae, Oligochaeta and Ostracoda. This is a drawback since these groups are heterogenous in sensitivity. We therefore recommend an increase in the taxonomic resolution in standard monitoring for these groups, especially in the Baltic where they in some areas can have large abundances. Some taxa are not regarded as being quantitatively sampled and are excluded from the assessment; these are listed in Supplement 1.

2.2. Calculation of the sensitivity values

The species sensitivity values used on the west coast of Sweden were numerically calculated from species specific abundance distributions from a large Nordic data set from the Kattegat and Skagerrak. This dataset included samples from 5–300 m depth during the period 1969–2005. The 1920 sampling occasions were distributed among 426 stations with various degrees of disturbance. The samples were collected with 0.1 m^2 van Veen or Smith-McIntyre grabs, sieved on 1 mm sieve, and in most cases sorted at six times magnification.

Calculation of the sensitivity values required three steps. First, the ES_{50} for each sample in the dataset was calculated according to Hurlbert (1971) (see Box 1). Each individual of a species occurring in more than 20 grabs were given an ES_{50} value. The sensitivity value of a species was set to the 5th percentile of the ES_{50}-values given to all individuals of the species. Two alternative methods to calculate the 5th percentile are described in Box 1.