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Seasonal variability of benthic indices: An approach to test the applicability of different indices for ecosystem quality assessment

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Abstract

In recent studies, several benthic biological indices were developed or used to assess the ecological quality status of marine environments. In the present study the seasonal variability of several univariate and multimetric indices was studied on a monthly scale (September 2000 until May 2002) in different areas of the North Sea such as the German Bight, the Oyster Ground and the Dogger Bank. The stations were chosen to reflect a gradient in the hydrographic regime, temperature and organic matter supply. The seasonal variability was highest for the univariate indices such as the Shannon–Wiener Index and the Hurlbert Index. Thus, due to sensitivity to recruitment the corresponding ecological status ranged from 'good' to 'poor' depending on the season. For the multimetric indices such as the AMBI or the BQI the seasonal variability and the corresponding ecological status were low. The results are discussed concerning possible consequences for ecological quality assessment especially related to the Water Framework Directive (WFD).

Keywords: Indicator species; AMBI; BQI; Diversity indices; Macrofauna; Water Framework Directive (WFD); North sea

1. Introduction

Due to the increasing impact of human activities on the marine ecosystem in the last decades the need for quality assessment and monitoring of marine systems has become increasingly important. The European Water Framework Directive (WFD), which came into force in December 2000, emphasises the assessment and achievement of the ecological quality status of coastal and estuarine waters. Also other international initiatives and agreements draw attention to the need for the assessment of the quality of marine environments, e.g., the Ecological Quality Objectives (EcoQO) concept developed by OSPAR (Frid and Hall, 2001; Painting et al., 2004; Rogers and Greenaway, 2005). The assessments of the ecological status will comprise physico-chemical and hydromorphological characteristics as well as different biological compartments of the ecosystem (e.g. plankton, benthos, fish).

The benthic fauna is an important component in marine ecosystems, playing a vital role in nutrient cycling, detrital decomposition and as a food source for higher trophic levels. Due to the relatively sessile habit and, thus, the incapability to avoid unfavourable conditions, macrobenthic species are sensitive indicators of changes in the marine environment caused by natural or anthropogenic disturbances. Since benthic species are relatively long-lived they integrate water and sediment quality conditions with time and, thus, indicate temporal as well as chronic disturbances.

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Effects of these disturbances include changes in diversity, biomass, abundance of stress tolerant or sensitive benthic species, and the trophic or functional structure of the benthic community (Pearson and Rosenberg, 1978; Warwick and Uncles, 1980; Warwick, 1986; Warwick and Clarke, 1994; Kaiser et al., 2000; Grall and Chauvaud, 2002). Thus, a variety of indices are available, which measure the status of ecological conditions and trends in succession of marine benthic systems. Univariate diversity indices such as the Shannon–Wiener index were the most commonly used indices in the past. In more recent studies multimetric indices were developed to get a more sensible tool for the assessment of ecological quality in a benthic ecosystem. Based on the model of Pearson and Rosenberg (1978), many of these indices used indicator species or ecological groups of species according to their sensitivity to stress, such as the Benthic Index (BI) Grall and Glémarec, 1997), the biotic index (BENTIX) (Simboura and Zenetos, 2002) and the Azti Marine Biotic Index (AMBI) (Borja et al., 2000) or used a combination of univariate and multimetric indices such as the benthic index of biotic integrity (B-IBI) (Weisberg et al., 1997) and the Ecological Quality Ratio (EQR) (Borja et al., 2003b). However, most of these indices have been designed to differentiate anthropogenic impacted sites from undisturbed reference sites (Van Dolah et al., 1999; Borja et al., 2003a; Muxika et al., 2005), but univariate as well as multimetric indices respond to any disturbance, be it natural or man-induced (Wilson and Jeffrey, 1994). For the assessment of a general ecological quality status of marine environments as well as for the indication of reference conditions, the 'natural' variability of the indices on different temporal and spatial scales has to be assessed and taken into account (Vincent et al., 2002).

In the present study macrofauna communities were sampled in three different areas of the southern North Sea on a monthly scale from September 2000 until May 2002 in order to assess the seasonal variability of the benthic fauna. Previous results showed that the seasonal fluctuations in a marine environment result in changes of abundance, diversity and community structure of benthic communities (Reiss and Kröncke, 2005).

The objectives of this study were to (i) compare different univariate and multimetric indices used for quality assessment purposes with regard to their variability on a seasonal scale and (ii) whether this seasonal variability differs under different environmental conditions.

2. Material and methods

A total of 16 monthly sampling cruises were carried out at three stations in the southern German Bight (station GB5), the Oyster Ground (station OG7) and at the north-eastern Dogger Bank (DG9) from September 2000 to May 2002 (North Sea, Germany; Fig. 1). The stations reflect a gradient in the hydrographic regime, temperature and organic matter supply. Details of the study sites and their macrofaunal communities are given in Kröncke and Rachor (1992), Kröncke et al. (2004),

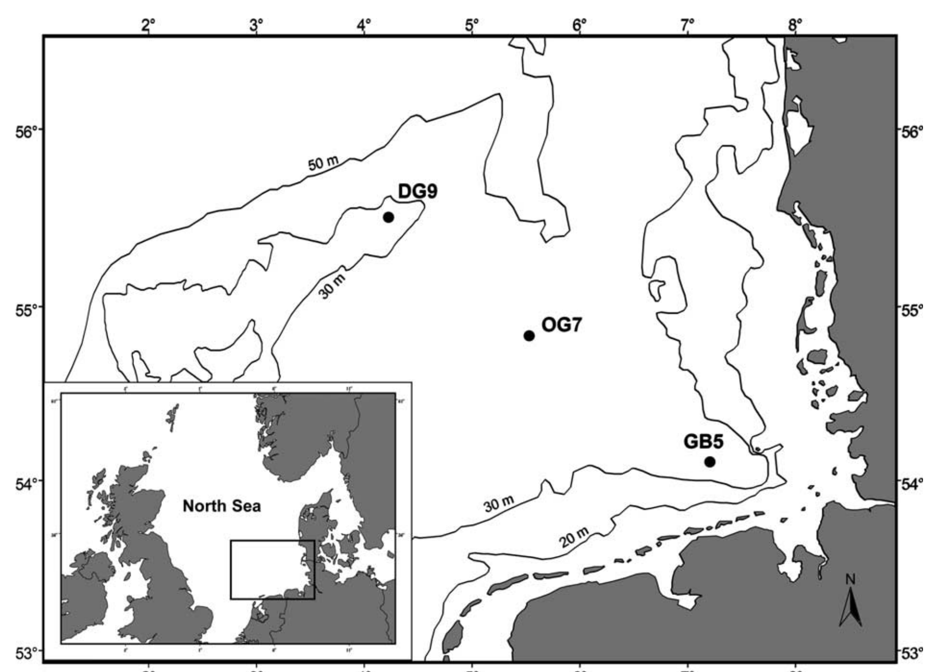

Fig. 1. Area of investigation in the North Sea with sampling sites.

Reiss and Kröncke (2004, 2005) and Reiss et al. (submitted).

2.1. Sampling and sample treatment

Samples were generally obtained with the RV 'Senckenberg'. In May 2001/02 and January 2001/02 samples were taken with the RV 'Gauss' and the FRV 'Walther Herwig III', respectively. The macrofauna was sampled with a 0.1 m^2 Van Veen grab during daylight only. Whenever possible, five replicates were taken at each station. The samples were sieved over 0.5 mm mesh size and fixed in 4% buffered formalin. In the laboratory the samples were additionally sieved over 1 mm mesh size. Since most monitoring and quality assessment studies are based on 1 mm mesh size, only the data of the 1 mm fraction were used for the present study.

2.2. Data analyses

2.2.1. Univariate indices

A variety of diversity indices have been used in benthic ecology to assess the environmental quality and the effect of disturbances on benthic communities. In the present study, calculations of four diversity indices were carried out: the Shannon–Wiener Index, the Hurlbert Index, the taxonomic diversity and the taxonomic distinctness. The computer software PRIMER (Clarke and Warwick, 1994) was used for data analyses.

The Shannon–Wiener Index (H') is the diversity index most commonly used in benthic ecology. This index incorporates species richness as well as equitability of the community. In this study, the Shannon diversity was calculated using the logarithm for a base 2. This index is dependent on sample size. In contrast, the Hurlbert Index (ESn) is less dependent on sample size and is based on the rarefaction technique of Sanders (1968) and was modified by Hurlbert (1971). In this index the expected number of species (ES) is calculated among certain number of individuals, e.g., of 100 individuals (ES100) as used in the present study.

The taxonomic diversity (Δ) and the taxonomic distinctness (Δ^*) are indices based on the taxonomic spread of species. The indices assess average taxonomic separation of all pairs of individuals in a sample. The taxonomic diversity (Δ) is empirically related to the Shannon–Wiener index but includes an additional component of taxonomic separation, whereas the taxonomic distinctness (Δ^*) is a measure purely of taxonomic distinctness (Warwick and Clarke, 1995; Clarke and Warwick, 1999). Similar to the Hurlbert Index, both indices appear to be less influenced by sample size than other diversity indices.

2.2.2. Multimetric indices

In the present study, the AZTI Marine Biotic Index (AMBI) and the related Ecological Quality Ratio (EQR) both developed by Borja et al. (2000, 2003b), as well as the Benthic Quality Index (BQI) by Rosenberg et al. (2004) were tested.

In the AZTI Marine Biotic Index (AMBI) benthic species were assigned to five ecological groups ranging from sensitive species (group I) to species highly tolerant to stress (group V). A Biotic Coefficient can be calculated based upon the percentage of each ecological group within each sample:

$$\mathrm{AMBI} = \frac{1}{100} \times (0 \times \%\mathrm{EG_I} + 1.5 \times \%\mathrm{EG_{II}} + 3 \times \%\mathrm{EG_{III}} + 4.5 \times \%\mathrm{EG_{IV}} + 6 \times \%\mathrm{EG_V})$$

where EG gives the percentage of the total numerical abundance in the sample for each of the five ecological groups ($\mathrm{EG_I}$ to $\mathrm{EG_V}$). Thus, the AMBI can derive continuous values from 0 (unpolluted) to 6 (heavily polluted) being 7 when the sediment is azoic (Borja et al., 2000). A list that includes >2700 benthic species and their assignment to the ecological groups as well as the AMBI© program for calculations of the AMBI are available on the web page: http://www.azti.es.

The Ecological Quality Ratio (EQR) combines the Biotic Coefficient, the Shannon–Wiener Index and the species number to one cumulative index ranging between 0 (bad ecological quality) and 1 (high ecological quality) according to the requirements of the EU Water Framework Directive (Borja et al., 2003b).

In the Benthic Quality Index (BQI) of Rosenberg et al. (2004) the Hurlbert Index (here ES50) was used to categorise benthic species according to their sensitivity against disturbance. They assumed that tolerant species are mainly found in disturbed environments and, thus, mainly occur at stations with low ES50, whereas sensitive species mainly occur at stations with high ES50. Based on this conclusion a species tolerance level ($\mathrm{ES50}_{0.05}$) was calculated, which reveals the minimum ES50 value for 5% of each macrofauna population. On this basis, the following benthic quality index was proposed:

$$\mathrm{BQI} = \left(\sum_{i=1}^{n}\left(\frac{A_i}{\mathrm{tot}A} \times \mathrm{ES50}_{0.05i}\right)\right) \times {}^{10}\log(S+1)$$

where A is the mean relative abundance of species i and S is the number of species at the station. The BQI normally varied between 0 (bad ecological quality) and 20 (high ecological quality) with a total of five stages of classification. The limits of the classification by Rosenberg et al. (2004) varied according to the water depth. Thus, we used the classification by Rosenberg et al. for the stations deeper than 20 m.

3. Results

The macrofauna communities found at the stations GB5, OG7 and DG9 belong to the *Nucula nitidosa* community (GB5), the *Amphiura filiformis* community (OG7) and the *Bathyporeia-Fabulina* community (DG9), respectively (Salzwedel et al., 1985; Kröncke and Rachor, 1992; Wieking and Kröncke, 2003; Kröncke et al., 2004; Reiss and Kröncke, 2005). The most abundant macrofauna found at each station is listed in Table 1, additionally to the assignments to ecological groups (AMBI) and tolerance levels $ES50_{0.05}$ (BQI).

The seasonal patterns of mean abundance and species number at each station are shown in Fig. 2. At each station the species number increased during summer and was lowest during winter and spring (Fig. 2b). Mean abundance showed a similar pattern but the differences between summer and winter were much more distinct in the German Bight (GB5) than at the Dogger Bank (DG9) (Fig. 2a). For further details of seasonal changes of the macrofauna communities see Reiss and Kröncke (2005).

3.1. Univariate indices

3.1.1. Shannon–Wiener index

At station GB5 in the southern German Bight the Shannon–Wiener index ranged from 1.5 to 3.8 (Fig. 3a). Thus, the corresponding ecological status according to Molvaer et al. (1997) varied between 'poor' in summer and 'good' in autumn to spring (Table 2). At station OG7 in the Oyster Ground the Shannon Index varied between 2.2 and 4.6, which corresponds to an ecological status from 'moderate' in winter to 'high' in autumn. At station DG9 at the Dogger Bank the variability was less pronounced. The Shannon Index ranged from 3.2 (October 2001) to 4.3 (September) and, thus, the ecological status varied between 'good' and 'high' (Table 2).

3.1.2. Hurlbert Index (ES100)

The seasonal patterns of the Hurlbert Index were very similar to those of the Shannon Index (Fig. 3b). The variability was highest at station GB5 ranging from 11 (August 2001) to 24 (September 2001) corresponding to an ecological status between 'poor' and 'good' (Mol-vaer et al., 1997). At station DG9 the seasonal variability was

Table 1
Most abundant macrofauna species at each station and their assignment to ecological groups according to the AMBI (Borja et al., 2000) and their tolerance level ($ES50_{0.05}$) according to the BQI (Rosenberg et al., 2004) (x—no value available)

Station	Taxon	Mean abundance/m^2	Ecological group (AMBI)	$ES50_{0.05}$
GB5	*Phoronis muelleri*	3635	II	9.1
	Ophiura albida	466	II	9.4
	Nucula nitidosa	330	I	7.9
	Owenia fusiformis	290	II	7.4
	Spiophanes bombyx	246	III	11.1
	Echinocardium cordatum	137	I	9.3
	Lanice conchilega	114	II	x
	Scalibregma inflatum	91	III	7.5
	Tellimya ferruginosa	84	II	9.6
	Amphiura brachiata	67	II	x
OG7	*Amphiura filiformis*	2364	II	9.5
	Echinocardium cordatum	462	I	9.3
	Pholoe baltica	172	I	8.2
	Corbula gibba	139	IV	4.7
	Nucula nitidosa	137	I	7.9
	Phoronis muelleri	87	II	9.1
	Harpinia antennaria	72	I	12.5
	Nemertea	65	III	9.4
	Mysella bidentata	62	III	7.5
	Spiophanes bombyx	55	III	11.1
DG9	*Bathyporeia elegans*	849	I	x
	Bathyporeia nana	185	I	x
	Echinocardium cordatum	181	I	9.3
	Spiophanes bombyx	162	III	11.1
	Amphiura brachiata	155	II	x
	Phoronis muelleri	87	II	9.1
	Urothoe poseidonis	74	I	x
	Chaetozone sp. F-group	72	IV	10.2
	Nemertea	66	III	9.4
	Pontocrates arenarius	60	II	x

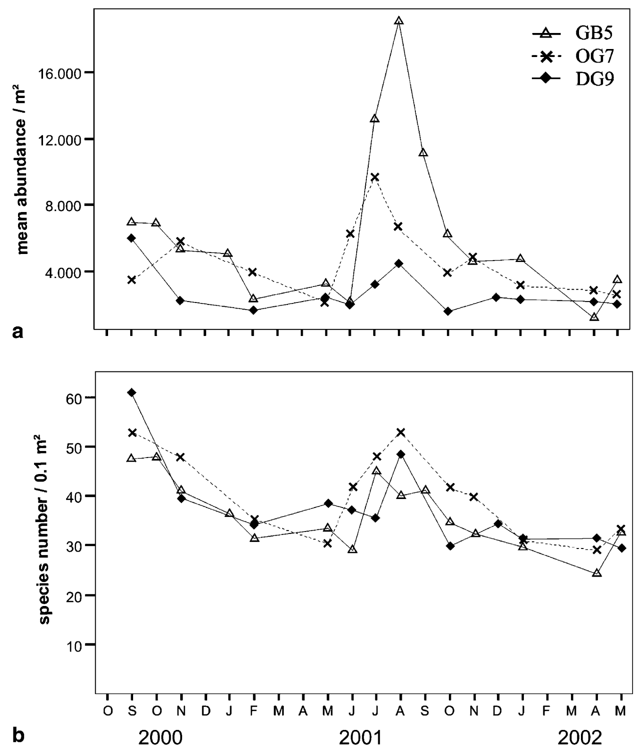

Fig. 2. Mean abundance (a) and mean species number (b) for the three stations during the study period.

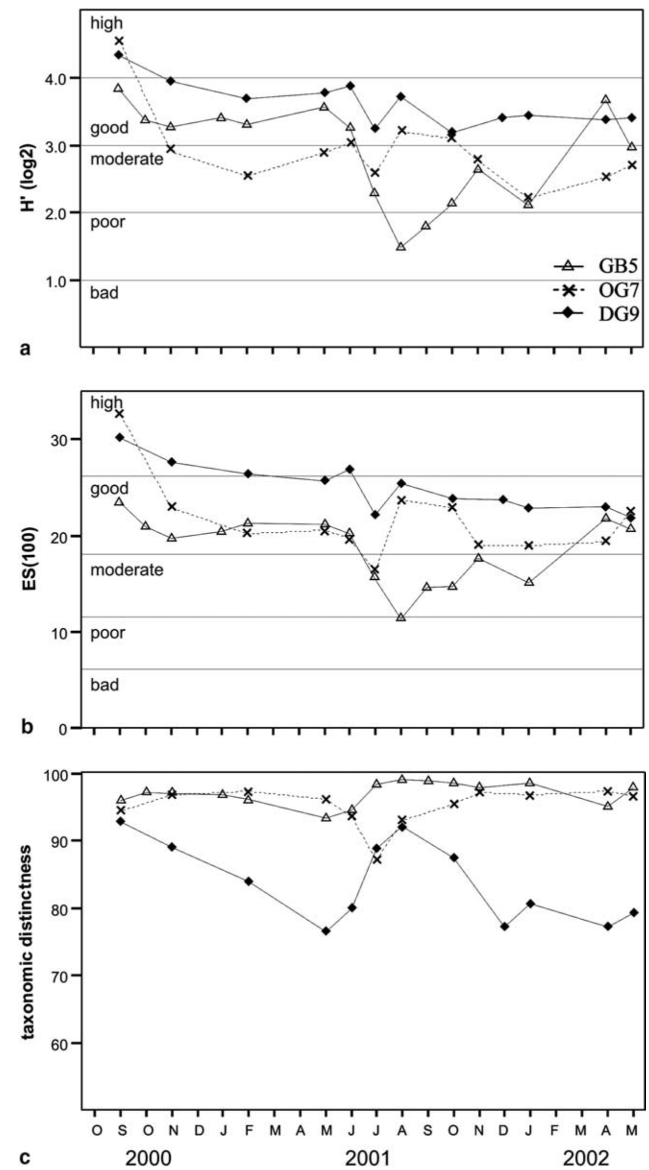

Fig. 3. Mean Shannon–Wiener Index (a), Hurlbert Index ES100 (b) and taxonomic distinctness (c) for the three stations during the study period. The classification of the ecological status referred to Molvaer et al. (1997).

lowest ranging between 22 (July 2001) and 30 (September 2000) and an ecological status between 'good' and 'high' (Table 2).

However, the high variability of the Hurlbert and the Shannon indices and the corresponding ecological status at station GB5 in the German Bight was mainly driven by the increase in abundance of the macrofauna due to recruitment in summer. This is confirmed by the significant correlation ($p < 0.01$) of both indices and the mean abundance in contrast to no significant correlation between the indices and the mean species number ($p > 0.05$) (Table 3). In contrast, at the Dogger Bank station DG9 this relationship appeared vice versa with significant correlations of the indices with species number ($p < 0.01$), but none for abundance ($p > 0.05$) (Table 3).

3.1.3. Taxonomic diversity and taxonomic distinctness

The seasonal pattern of the taxonomic diversity was the same as found for the Shannon–Wiener and the Hurlbert indices (not shown).

In contrast, the seasonal pattern of taxonomic distinctness was totally different to those of the Shannon–Wiener and the Hurlbert indices as well as the pattern of taxonomic diversity (Fig. 3c), because the highest variability was found at the Dogger Bank (DG9) and the lowest in the German Bight (GB5).

At station GB5 the taxonomic distinctness remained rather constant varying between 93 (May 2001) and 99 (August 2001). In contrast, at the Dogger Bank station DG9 the index varied strongly between 76 (May 2001) and 93 (September 2000/01). At the station OG7 in the Oyster Ground taxonomic distinctness revealed a rather constant value of 95, with the exception of July 2001, when the index dropped to 87 (Fig. 3c). For the taxonomical distinctness no classification of ecological status was available.

3.2. Multimetric indices

3.2.1. AMBI

Fig. 4a shows the seasonal pattern of the AMBI. At all stations the values were rather constant over the

Table 2
Ecological status at each station, the seasonal range of the ecological status and the constancy of the assignment of the status revealed with the different univariate and multimetric indices

Station	Index	Mean ecological status	Range of ecological status	Constancy (%)
GB5	Shannon–Wiener	Moderate	Poor–good	53
	Hurlbert (ES100)	Good	Poor–good	60
	Taxonomic distinctness	–	–	–
	AMBI	Good	Good	100%
	BQI	Good	Moderate–good	73
	EQR	Moderate	Poor–moderate	53%
OG7	Shannon–Wiener	Moderate	Moderate–high	67
	Hurlbert (ES100)	Good	Moderate–high	83
	Taxonomic distinctness	–	–	–
	AMBI	Good	Good–high	92
	BQI	Good	Moderate–good	92
	EQR	Moderate	Poor–moderate	58
DG9	Shannon–Wiener	Good	Good–high	92
	Hurlbert (ES100)	Good	Good–high	67
	Taxonomic distinctness	–	–	–
	AMBI	High	Good–high	92%
	BQI	–	–	–
	EQR	Moderate	Moderate–good	83

Table 3
Correlation between Shannon–Wiener Index, Hurlbert Index and the mean abundance and species number

		Abundance	Species number
GB5	H'	−0.624**	−0.004
	ES(100)	−0.605**	0.064
OG7	H'	−0.227	0.409**
	ES(100)	−0.433**	0.272*
DG9	H'	0.258	0.697**
	ES(100)	0.168	0.688**

Significance levels indicated as $p < 0.05$*, $p < 0.01$**.

whole study period. Thus, the corresponding ecological status remained stable as well, with 100% constancy (proportion of samples having the same ecological status) at station GB5 and 92% at station OG7 and DG9 (Table 2). Nevertheless, the proportion of the ecological groups at each station changed slightly during the study period (Fig. 5). For example in the German Bight (GB5), the abundance of group III species ('disturbance tolerant') increased during spring, but abundance of group II species ('disturbance-indifferent') was highest in summer. However, changes in abundance due to recruitment were mainly evident for the dominant species (ecological groups) of each community and, thus, the ecological status remained rather constant.

3.2.2. BQI

The tolerance levels ($ES50_{0.05}$) for the calculation of the BQI were only available for a part of the species found at our study sites. However, at station GB5 about 91% of the individuals and 42% of the species and at station OG7 about 92% of the individuals and 46% of the species could be included. In general, the same ecological status at station GB5 and OG7 was found as for the AMBI, which was classified as 'good'. However, the seasonal differences were slightly higher compared to the AMBI caused by an increase during summer at each station.

Due to the fact that at the Dogger Bank station DG9 only 33% of the individuals and 40% of the species could be included in the analyses, the results seem not to be valid for further interpretation (Fig. 4c).

3.2.3. EQR

The EQR corresponds to an ecological status about one category below the status revealed by the AMBI and the BQI. Thus, all stations were classified as 'moderate', compared to 'good' (GB5 and OG7) and 'high' (DG9) revealed with the AMBI (Table 2). The seasonal pattern of the EQR is shown in Fig. 4b. The constancy of ecological classification was low at station GB5 with 53% and at station OG7 with 58%, but higher at station DG9 with 83% (Table 2).

4. Discussion

The aim of the present study was to evaluate the 'natural' seasonal variability of several indices used or developed for quality assessment strategies according to the Water Framework Directive (WFD). To our knowledge no disturbances caused by anthropogenic impact such as dumping, dredging, etc., occurred at the stations sampled during this study. Also severe natural disturbances known to occur occasionally during a seasonal cycle,

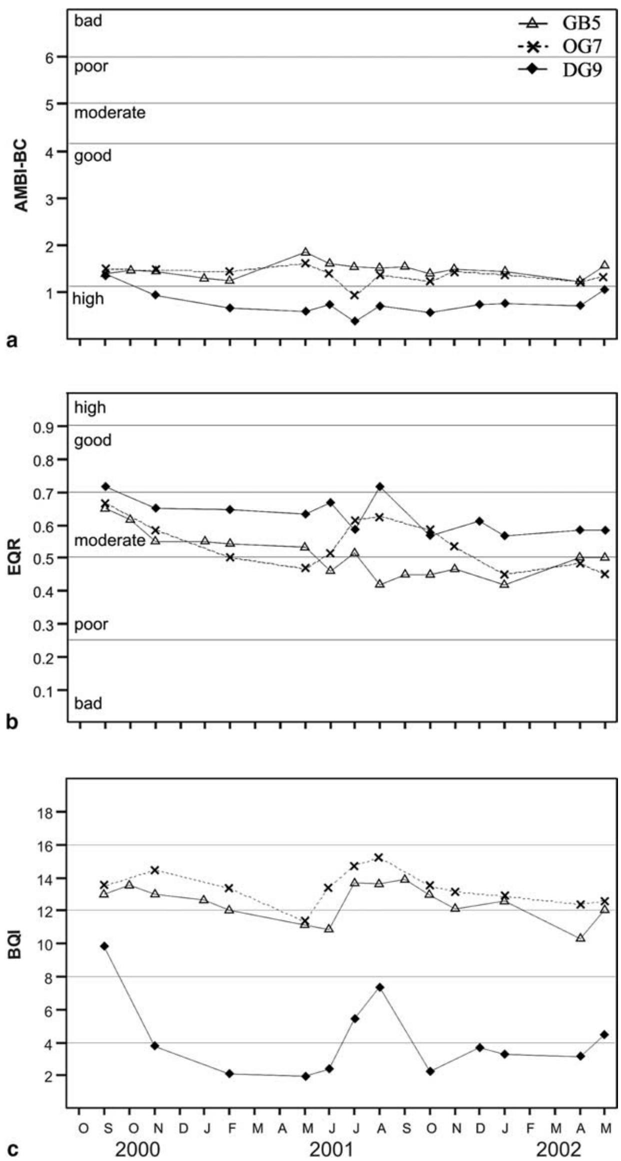

Fig. 4. Mean AMBI (a), EQR (b) and BQI (c) for the three stations during the study period.

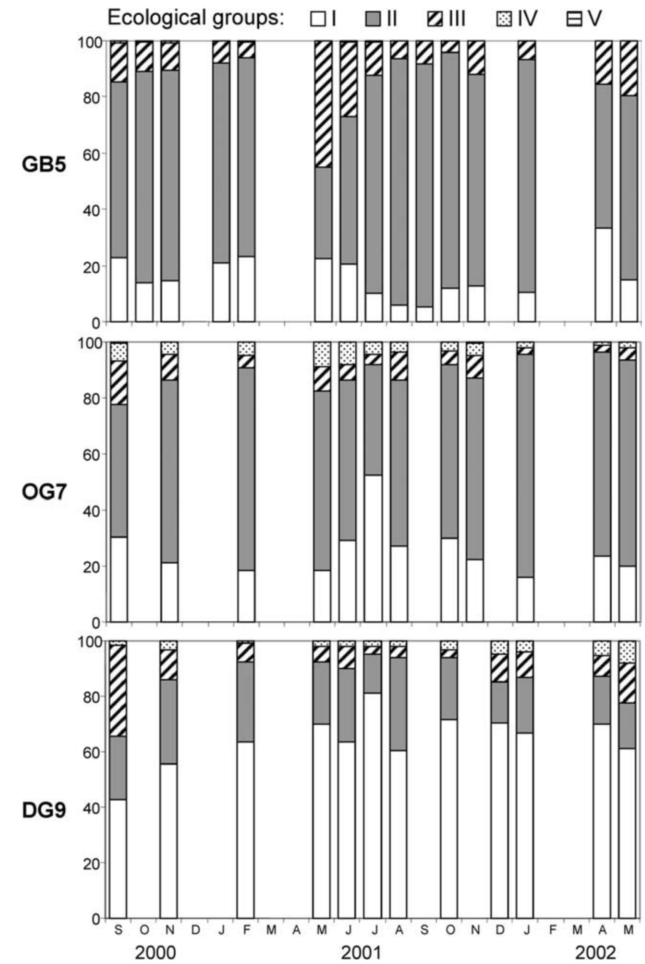

Fig. 5. Relative abundance of ecological groups according to the AMBI: (I) disturbance-sensitive, (II) disturbance-indifferent, (III) disturbance-tolerant, (IV) second-order opportunistic and (V) first-order opportunistic (Borja et al., 2000).

such as oxygen deficiency in summer or extreme cold winters were not observed during the study period (Reiss and Kröncke, 2005). Thus, we assume that changes in the macrofauna communities reflected the 'natural' seasonal variability.

Nonetheless, fishing activities affect benthic communities (Craeymeersch, 1994; van Santbrink and Bergman, 1994; Kaiser and Spencer, 1996; Rumohr and Kujawski, 2000) and in the whole southern North Sea beam trawling and, to a lesser extent, otter trawling are common (Rijnsdorp et al., 1998). Since no detailed spatial data on fishing effort were available, fishing impact cannot be excluded as a source of disturbance at our stations, which might have had an effect on the variability of the indices.

4.1. Seasonal variability of univariate indices

Most benthic indices developed in recent studies (Grall and Glémarec, 1997; Weisberg et al., 1997; Borja et al., 2000; Simboura and Zenetos, 2002; Rosenberg et al., 2004) have been based on the model of Pearson and Rosenberg (1978). This model states that macrofauna communities along a gradient of increasing organic enrichment (disturbance) change in diversity, abundance and species composition according to their tolerance against the disturbance. Many of these indices have been tested successfully to detect anthropogenic disturbances such as dredging, dumping, engineering works, sewerage plans, gravel extraction, etc. (Muxika et al., 2005). However, most biological indices respond on any kind of disturbances whether caused by anthropogenic impact or natural processes (Wilson and Jeffrey, 1994).

Our results show that the seasonal variability was highest for the univariate indices such as the Shannon–Wiener Index and the Hurlbert Index. In contrast, multimetric indices such as the AMBI or the BQI seem to be less influenced by seasonal changes of the macrofauna communities. These results coincide with the findings of Salas et al. (2004) who showed that the AMBI varies only slightly in time in an estuarine ecosystem.

If using the Shannon or the Hurlbert Index the macrofauna at station GB5 in the German Bight was classified as 'poor' ecological status in summer and 'good' status in autumn and winter. Thus, the ecological status fluctuated over a range of three classification categories (out of 5) depending on the time of the year, when sampling took place. In the German Bight (GB5) and in the Oyster Ground (OG7) the high seasonal variability of the indices was primarily caused by the increase in abundance of a few species due to recruitment and growth in summer (Reiss and Kröncke, 2005). In contrast, at the Dogger Bank (DG9) seasonal variability in abundance was smallest, because the relatively moderate increase of species numbers during summer influenced the diversity indices most (Table 3). Thus, the seasonal variability of the diversity indices was generally lower at station DG9 compared to stations OG7 and GB5. Therefore, especially in areas where seasonal changes of the macrofauna communities go along with a drastic increase of abundance, which is the case for many coastal areas in temperate regions (Künitzer, 1990; Niermann, 1990; Bosselmann, 1991; Albertelli et al., 2001), seasonal variability of diversity indices and the ecological status will be high.

4.2. Seasonal variability of multimetric indices

Recently, multimetric indices have been developed to get a more sensible tool for the assessment of ecological quality of benthic ecosystems. In contrast to the univariate indices the multimetric indices incorporate the ecological preferences of benthic species. Thus, most of these multimetric indices use ecological groups of species according to their sensitivity to stress, such as the Benthic Index (BI) of Grall and Glémarec (1997) for the French coast, the biotic index (BENTIX) of Simboura and Zenetos (2002) for the Mediterranean or the benthic index of biotic integrity (B-IBI) of Weisberg et al. (1997) for estuaries of the south-eastern USA as well as the indices used for this study, the Azti Marine Biotic Index (AMBI) of Borja et al. (2000, 2003b) and the Biotic Quality Index (BQI) of Rosenberg et al. (2004).

These indices (AMBI, BQI) were chosen since they are based on two different approaches of ecological grouping (see below) and since they were appropriate for the same biogeographic region as our study site in the North Sea.

The seasonal variability of both indices, the AMBI and the BQI, was generally low (Fig. 4; Table 2). Although the composition of ecological groups, in the case of AMBI, changed seasonally within each community (Fig. 5), these seasonal changes were mainly restricted to changes in abundance of the dominant species, which belonged to the dominant ecological groups. However, a slight increase in ecological status during summer was observed for both indices. Multimetric indices, which classify macrofauna species into ecological groups, seem to be a promising approach for ecological quality assessment in order to avoid drawbacks due to the seasonal variability of the benthic communities. However, the classification of benthic species into ecological groups or as indicators of stress is a crucial point in this approach, which has lead to intense debates (Borja, 2004; Simboura, 2004; Dauvin, 2005). The ecological grouping used for the AMBI was based on literature and personal experience of the scientists involved. In contrast, the BQI uses a more objective way of ecological grouping by calculating species tolerance levels based on the Hurlbert Index (ES50). But in turn, that makes the BQI more dependent on the data set used for the calculation of the tolerance levels. Since the BQI was developed and verified in coastal areas of the Skagerrak and Kategatt (Rosenberg et al., 2004), many species of the North Sea are not included. Thus, an enhancement of this method with North Sea data sets and further testing is necessary.

In order to fulfil all requirements for the ecological quality assessment proposed in the Water Framework Directive a recently published guidance document stated that 'methods combining composition, abundance and sensitivity may be the most promising' (Vincent et al., 2002). Thus, cumulative indices such as the Ecological Quality Ratio (EQR) proposed by Borja et al. (2003b) seem to accomplish these requirements, as they combine the Shannon–Wiener Index, the species richness and the AMBI in one cumulative index. Our results show that the seasonal variability of the EQR is approximately in the same range as the AMBI or the BQI, but the ecological status was generally one to two categories below the other indices (Fig. 4; Table 2). This might be caused by the Shannon–Wiener Index, which is strongly influenced by several parameters such as sample size, identification procedure and sampling methodology, which affect also the species richness, another component of the EQR. Furthermore, diversity indices and especially species richness are habitat type dependent, which means that different ranges of values may appear in different habitat types.

A further cumulative index that is almost exclusively based on diversity indices, is the Norwegian classification tool (Molvaer et al., 1997), which combines the Shannon–Wiener Index ($\log_2$), the Hurlbert Index (HS100) and the Total Organic Carbon (TOC) content of the sediment. Since, the seasonal variability especially of the univariate indices can be remarkably high, users of cumulative indices such as the EQR and the

Norwegian classification tool have to take this weakness into account if assessing the ecological quality status or biological reference values.

5. Application

The Water Framework Directive, which is the main initiative for the development of biological indices for ecosystem quality assessment in European waters, is restricted to coastal marine environments (Vincent et al., 2002). But the stations investigated in the present study were situated in offshore areas in the southern North Sea. Reiss and Kröncke (2005) showed that the seasonal variability in abundance, diversity and community structure of the endobenthos was mainly caused by recruitment in spring and summer. It was found to be highest in the German Bight, where seasonal fluctuations in environmental parameters were also higher than in the Oyster Ground (OG7) and at the Dogger Bank (DG9). Apart from the dominant role of recruitment, the seasonal variability of the benthic communities seems to be a result of synergistic effects of numerous factors such as food availability and limitation, water temperature, predation and hydrodynamical stress. Since, in near-shore coastal areas seasonal fluctuations of environmental and faunal parameters will be higher than in offshore areas, the seasonal variability of indices will also be higher.

In conclusion, for the assessment of the ecological status of marine benthic environments univariate indices such as the Shannon–Wiener index or the Hurlbert Index are not appropriate, if the seasonal variability is high. Also cumulative indices including univariate indices, such as the EQR and the Norwegian classification tool, have to be used carefully as an ecological classification device.

The use of these indices will be problematic if the benthic ecological quality has to be assessed and no contemporary reference areas are available or different data sets from different seasons were compiled.

Multimetric indices based on general life history traits of the benthic fauna seem to be less influenced by the seasonal variability of the macrofauna. This is supposed to be the case not only for the indices tested in the present study (AMBI, BQI), but also for other multimetric indices using the same approach such as the BENTIX, BI, etc. (Grall and Glémarec, 1997; Simboura and Zenetos, 2002).

However, our results show that the seasonal variability differs between marine regions under different environmental conditions. Thus, the choice of the adequate index, which is essential for the assessment of the ecological quality of marine regions, might depend on the research or monitoring topic, as well as on the study area.

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