

The suitability of the marine biotic index (AMBI) to new impact sources along European coasts

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Abstract

In recent years, several benthic biotic indices have been proposed to be used as ecological indicators in estuarine and coastal waters. One such indicator, the AZTI Marine Biotic Index (AMBI), was designed to establish the ecological quality of European coasts. The index examined the response of soft-bottom benthic communities to natural and man-induced disturbances in coastal and estuarine environments. It has been successfully applied to different geographical areas and under different impact sources, with increasing user numbers in European marine waters (Baltic, North Sea, Atlantic and Mediterranean). The AMBI has been used also for the determination of the ecological quality status (EcoQ) within the context of the European *Water Framework Directive* (WFD).

In this contribution, 38 different applications including six new case studies (hypoxia processes, sand extraction, oil platform impacts, engineering works, dredging and fish aquaculture) are presented. The results show the response of the benthic communities to different disturbance sources in a simple way. Those communities act as ecological indicators of the 'health' of the system, indicating clearly the gradient associated with the disturbance.

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1. Introduction

Recently, several benthic biotic indices have been proposed as ecological indicators in estuarine and

coastal waters (Hily, 1984; Washington, 1984; Rygg, 1985; Majeed, 1987; Codling and Ashley, 1992; Dauer, 1993; Engle et al., 1994; Grall and Glémarec, 1997; Weisberg et al., 1997; Roberts et al., 1998; Van dolah et al., 1999; Smith and Rule, 2001; and Eaton, 2001), to determine natural and man-induced impacts. One such indicator, the AZTI Marine Biotic Index (AMBI), which was developed by Borja et al. (2000) has been applied successfully to different geographi-

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cal areas and under different impact sources (Borja et al., 2003a,b, 2004a), with increasing user numbers within Europe (Table 1).

The AMBI was designed primarily to establish the ecological quality of European coastal and estuarine waters by examining the response of soft-bottom benthic communities to natural and man-induced disturbances in the environment. Hence, the AMBI offers a ‘disturbance or pollution classification’ of a

site, representing the benthic community ‘health’ (*sensu* Grall and Glémarec, 1997). Secondly, it has been used for the determination of the ecological quality status (EcoQ) within the context of the European *Water Framework Directive* (WFD) (Borja et al., 2003b, 2004a,b). The ultimate aim of the WFD is to achieve, by 2015, a good EcoQ within all the European waters, by the elimination of priority hazardous substances, and contribute to achieving

Table 1

Different impact sources and geographical areas for which AMBI has been applied, in recent years

Impact sources	Locations (countries)	Seas	Author
Various sources along UK	(United Kingdom)	Atlantic	A. Miles, A. Prior (p.c., 2003)
Outfall and harbour	Brittany (France)	Atlantic	Borja et al. (2003a)
Engineering works (dyke)	Basque Country (Spain)	Atlantic	Borja et al. (2000, 2003a)
Sewerage works	Basque Country (Spain)	Atlantic	Borja et al. (2000, 2003a)
Harbour construction	Basque Country (Spain)	Atlantic	This contribution
Submarine outfall	Basque Country (Spain)	Atlantic	Borja et al. (2000, 2003b)
Harbour and river inputs	Basque Country (Spain)	Atlantic	Muxika et al. (2003)
Various sources	Tejo estuary (Portugal)	Atlantic	M.J. Gaudencio (p.c., 2003)
Eutrophy	Mondego estuary (Portugal)	Atlantic	Salas et al. (2004)
River inputs	Guadalquivir (Spain)	Atlantic	AZTI (unpublished data)
Heavy metals	Huelva (Spain)	Atlantic	Borja et al. (2003a)
Estuarine inputs	Cádiz (Spain)	Atlantic	A. Rodríguez-Martín (p.c., 2003)
Various sources	(Morocco)	Atlantic	H. Bazairi (p.c., 2003)
Various sources	Latvia	Baltic	V. Jermakovs (p.c., 2004)
Anoxia-hypoxia	Sweden	Baltic	This contribution
Dredging mud disposal	Sweden	Baltic	S. Smith (p.c., 2003)
Various sources along Sweden	Sweden	Baltic	M. Blomqvist (p.c., 2003)
Various sources in a lagoon	Smir (Morocco)	Mediterranean	A. Chaouti (p.c., 2003)
Dredging in harbour	Ceuta (Spain)	Mediterranean	This contribution
Diffuse pollution (mines, agriculture, ...)	Almería and Murcia (Spain)	Mediterranean	Borja et al. (2003a)
Aquaculture cages	Murcia, Valencia (Spain)	Mediterranean	AZTI (unpublished data)
Mining debris	Mar Menor (Spain)	Mediterranean	L. Marín (p.c., 2004)
Submarine outfall	Catalonia (Spain)	Mediterranean	M.J. Cardell (p.c., 2003)
Marina	Catalonia (Spain)	Mediterranean	S. Pinedo (p.c., 2003)
Wastewater discharge in a lagoon	(France)	Mediterranean	G. Reimonenq (p.c., 2003)
Inputs to a coastal lagoon	Adriatic Sea (Italy)	Mediterranean	Casselli et al. (2003)
Various sources	Adriatic Sea (Italy)	Mediterranean	Forni and Occhipinti Ambrogi (2003)
Stagnation and industrial and urban pollution	Port of Trieste (Italy)	Mediterranean	Solís-Weiss et al. (2004)
Submarine outfall	Gulf of Trieste (Italy)	Mediterranean	Solís-Weiss (p.c., 2004)
Various sources	Adriatic Sea (Italy)	Mediterranean	R. Simonini (p.c., 2004)
Submarine outfall	Saronikos Gulf (Greece)	Mediterranean	Borja et al. (2003a)
Aquaculture cages	Three locations (Greece)	Mediterranean	This contribution
River inputs	Thames (United Kingdom)	North Sea	M. Davison (p.c., 2002)
Oil-based drilling muds (oil platforms)	Eleven locations (United Kingdom)	North Sea	This contribution
Impacts on sandy shores	(Netherlands)	North Sea	S. Mulder (p.c., 2003)
Ester-based drilling muds (oil platforms)	North Sea (Netherlands)	North Sea	Borja et al. (2003a)
Re-opening of a brackish lake to sea influence	Veerse Meer (Netherlands)	North Sea	V. Escaravage (p.c., 2004)
Sand extraction	Belgium	North Sea	Bonne et al. (2003); this contribution

Key: p.c., personal communication.

Table 2

Summary of the AMBI values and their equivalences (modified from Borja et al., 2000)

Biotic coefficient	Dominating ecological group	Benthic community health	Site disturbance classification	Ecological status
0.0 < AMBI ≤ 0.2	I	Normal	Undisturbed	High status
0.2 < AMBI ≤ 1.2		Impoverished		
1.2 < AMBI ≤ 3.3	III	Unbalanced	Slightly disturbed	Good status
3.3 < AMBI ≤ 4.3		Transitional to pollution	Moderately disturbed	Moderate status
4.3 < AMBI ≤ 5.0	IV–V	Polluted		Poor status
5.0 < AMBI ≤ 5.5		Transitional to heavy pollution	Heavily disturbed	
5.5 < AMBI ≤ 6.0	V	Heavy polluted		Bad status
6.0 < AMBI ≤ 7.0	Azoic	Azoic	Extremely disturbed	

The last column shows the proposed equivalent ecological status for the application of the WFD (Borja et al., 2003b).

concentrations in the marine environment near background values for naturally occurring substances. EcoQ is established on the basis of physico-chemical and biological variables (Borja et al., 2004a). In coastal and estuarine waters, benthic community measures are especially important, as they integrate impacts over a wide period of time. In the benthic EcoQ determination, three parameters are proposed by the WFD: diversity, species abundance and the presence/absence of indicator and stress-sensitive species. The latter is represented by the AMBI (Borja et al., 2003b, 2004a,b).

The AMBI is based upon ecological models, such as those of Glémarec and Hily (1981) and Hily (1984) (for the development of the model, see Borja et al., 2000; for additional details, see Borja et al., 2004b). The most novel contribution of AMBI has been a formula (1) to allow the derivation of a series of continuous values (firstly called the ‘Biotic Coefficient’, in Borja et al. (2000), and then AMBI, as outlined above), based upon the proportions of five ecological groups (EG) to which the benthic species are allocated:

$$\begin{aligned} \text{AMBI} = & [(0 \times \% \text{EG I}) + (1.5 \times \% \text{EG II}) \\ & + (3 \times \% \text{EG III}) + (4.5 \times \% \text{EG IV}) \\ & + (6 \times \% \text{EG V})] / 100 \end{aligned} \quad (1)$$

with EG I being the disturbance-sensitive species, EG II the disturbance-indifferent species, EG III the disturbance-tolerant species, EG IV the second-order opportunistic species and EG V the first-order opportunistic species (see Borja et al., 2000).

Several thresholds have been established over the scale of the AMBI, based upon proportions amongst

the various ecological groups (see Fig. 2, in Borja et al., 2000). These thresholds (Table 2) are coincident with the benthic community health proposed by Grall and Glémarec (1997) (based upon Reish, 1959, Bellan, 1967 and Pearson and Rosenberg, 1976).

The AMBI is expected to be used to calculate the EcoQ, but it would be only a part of a set of measures and indices in the WFD, such as diversity, richness, etc. The thresholds used for site pollution classification are not the same as the thresholds proposed for the EcoQ, to accomplish WFD specifications (Borja et al., 2003b, 2004b) (Table 2).

The increasing use of this tool requires integration of different applications and results obtained by various authors (see, also, Table 1). Hence, the main objectives of this contribution are to explore: (i) the suitability of the AMBI to its use in the Atlantic, Baltic, North Sea and Mediterranean European coasts; and (ii) its usefulness in relation to different new impact sources, not explored previously, i.e. sand extraction, hypoxia processes, oil platform impacts, dredging and fish aquaculture.

2. Methods

Six different ‘case studies’ along the European coast are compared. These case studies are based on data from the authors and other published sources representing a variety of pollution types, environmental problems and geographic settings.

Case study 1 is located in the Gullmarsfjord (Swedish west coast; see Fig. 1). This fjord has a maximum depth of 118 m. The bottom water is usually renewed with oxygen-rich water, each spring.

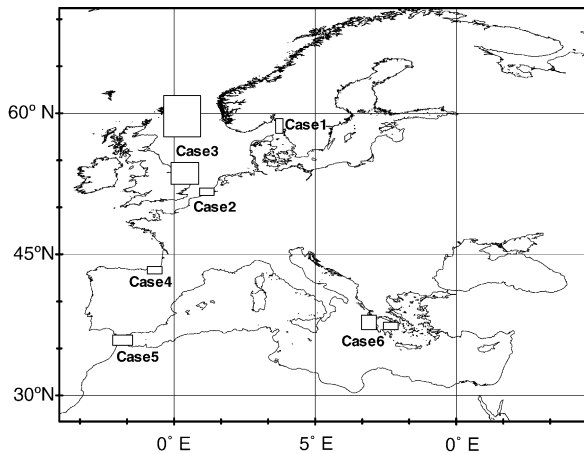


Fig. 1. Location of the six case studies, within the context of an European framework. Note that there are two sites for Case 3 (northern and central North Sea) and two sites for Case 6 (western and eastern Greece).

In spring 1997, this water renewal did not occur and the fauna were: more or less unaffected at 60 and 75 m water depth (oxygen saturation > 15%); significantly reduced at 85 and 95 m (saturation < 10%); and eliminated below about 100 m water depth (Rosenberg et al., 2002). In spring 1998, the fjord was re-oxygenated and the succession of benthic fauna was studied at five sampling stations and over a 2-year period by Rosenberg et al. (2002).

The Kwintebank (Case study 2) is an intensively exploited sandbank, located in the southern North Sea (Belgian coast; see Fig. 1). Three stations (1, 6 and 9) were sampled for macrobenthos during three periods (in the late 1970s, late 1990s and 2001, respectively), within the framework of different projects (Bonne et al., 2003). These data were used to assess the impact of sand extraction on subtidal sandbanks. In the late 1970s about 310,000 m³ yr⁻¹ sand was extracted on the Kwintebank, whereas the extracted volume amounted to about 1,360,000 m³ yr⁻¹ in the late 1990s, and 1,700,000 m³ in 2001 (Fund for Sand Extraction, FPS Economy, S.M.E.s, Self-employed and Energy, Brussels, Belgium). The sand extraction intensity, in the late 1990s, was very high at Stations 1 and 6 (56,000 and 92,000 m³ 0.5 km⁻² yr⁻¹, respectively); and low at Station 9 (4,000 m³ 0.5 km⁻² yr⁻¹) (Bonne and Vincx, 2003).

Case study 3 is located in the central and northern North Sea (Fig. 1), where numerous oilfields are

installed. Biological and physico-chemical data were obtained from the Marine Environmental Surveys Database on the UKCS-UK Benthos, provided by the UK Offshore Operators Association (UKOOA). From this database, the 11 piles sampled in 1988 were selected for this contribution; five of these are situated in the northern North Sea (Beryl A, Beryl B, Buchan, Miller and Thistle), whilst six are located in the southern/central North Sea (Audrey, Barque, Cleeton, Cilpper, Ravenspurn and Sole). Moreover, complementary biological and chemical information was obtained from Davies et al. (1984), Shimmield et al. (2000), and Breuer et al. (1999, 2004).

Case study 4 is situated on the Spanish Basque coast (Fig. 1). For the construction of a new dyke in Bilbao harbour, a large area of the seabed was dredged. These works commenced in 1993. An intensive dredging period extended from 1995 to 1997, changing the bathymetry of the area from -20 to -30 m, in some places. As a result, all of the benthic fauna disappeared (V. Valencia, AZTI, personal communication, 2003). The works finished in 1999. The zone was monitored before the construction (from 1989 to 1993) and three surveys were carried out (in 2000, 2001 and 2003) to monitor the impact of the new structure.

The immediate recolonisation following a harbour dredging in 1999 (Case study 5) was studied in the Spanish Mediterranean (Ceuta, North Africa; see Fig. 1). The impact on the benthic communities was studied using a before-after control impacted (BACI) approach, by Guerra-García et al. (2003). These authors undertook six samplings, at dredged and control locations: before dredging and after 3, 15, 30, 90 and 180 days. Community structure data and MDS ordination were used to analyse the results.

The impact of cage culturing of fish on benthic communities was investigated at three commercial fish farms with different types of sediment (ranging from silt to coarse sand) at 20–30 m water depth in Cephalonia, Ithaki and Sounion (Greece; Fig. 1), by Karakassis et al. (2000) (Case study 6). A transect of stations and a control station were sampled near each farm, for macrofauna and geochemical variables in July and October 1995 and April 1996.

Most of the selected studies have, in common, the aim to explain the effect of different impact sources on soft-bottom communities; this is based upon the study

of structural parameters, such as abundance, biomass, richness, diversity, evenness, abundance-biomass comparison (ABC) curves (Warwick, 1986; Clarke, 1990), or multivariate methods. The latter include: clustering (Sokal and Sneath, 1963); multi-dimensional scaling (MDS) (Kruskal and Wish, 1977; Schiffman et al., 1981; Field et al., 1982); principal component analysis (PCA) (Kendall, 1975; Jolliffe, 1986); or correspondence analysis (CA) (Hill, 1974; Fielding, 1992). Based upon the abundance of individuals, as provided by the authors in the above-mentioned papers, the corresponding Biotic Coefficient (AMBI), *sensu* Borja et al. (2000), was calculated using a freeware program available on www.azti.es, which includes the EG of more than 2,700 taxa, updated continuously. Whenever the species composition per replicate was available, the AMBI was calculated for each of the replicates, then averaged for the entire station, as recommended by Borja et al. (2004b). These values have been used to illustrate, in a simple format: (i) spatial disturbance or pollution gradients; (ii) the evolution of the effect of disturbance or pollution on the communities; (iii) and the sensitivity of AMBI to different impact sources. The assessment was undertaken according to the classification listed in Table 2.

In this contribution, the term ‘disturbed’ has been used with the same meaning as ‘polluted’ (which was used in the original contribution of Borja et al., 2000). Hence, ‘unpolluted’, ‘slightly polluted’, etc., are presented here as ‘undisturbed’, ‘slightly disturbed’, etc. The use of ‘disturbed’ is recommended when the impact source is natural (e.g. the inner part of an estuary, with high levels of natural stress and unlikely to be classified as undisturbed), mechanical (e.g. high exposure or dynamics, with changing characteristics in the substrata) or physical (e.g. dredging or engineering works). For comparison, the use of ‘polluted’ is recommended when the impact source is chemical.

3. Results

3.1. Case study 1

Although the mean abundance was only available for the dominant species in Rosenberg et al. (2002),

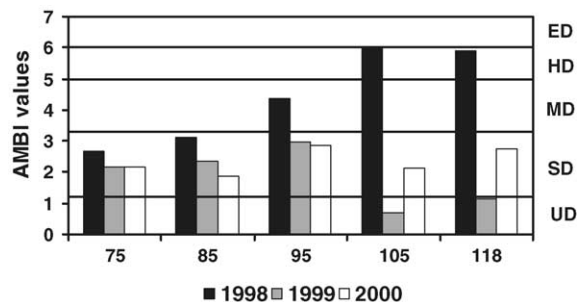


Fig. 2. AMBI values in 1998, 1999 and 2000, for each sampling station in Gullmarsfjord (note that the label for each sampling station coincides with the water depth). Key: UD = undisturbed; SD = slightly disturbed; MD = moderately disturbed; HD = heavily disturbed; and ED = extremely disturbed.

the AMBI values calculated with those data (based throughout upon more than 80% of the total abundance, for each station) show a clear increasing gradient with water depth, during 1998 (Fig. 2). The sampling stations at 75 and 85 m water depths were classified as slightly disturbed (AMBI values < 3.3) and dominated by the EG III (47–91% of total abundance), which represents the tolerant species. The station at 95 m water depth was classified as moderately disturbed (AMBI values near 4.5), with the EG III and V (first-order opportunistic) species being co-dominant, at 54 and 46% of the total abundance, respectively. The 105 and 118 m water depth stations were classified as heavily disturbed (AMBI values near 6), with the EG V species representing more than 95% of the community abundance.

In 1999 and 2000, a clear recovery was detected by the AMBI at the 95, 105 and 118 m water depth stations; these were classified as slightly disturbed or undisturbed. The EG III became dominant and the EG V disappeared within these communities. At the 75 and 85 m water depth stations a recovery was also detected, but was not so important (Fig. 2). The EG I (sensitive) species became more important as a consequence of a decrease in the percentage of EG IV (second-order opportunistic) species.

3.2. Case study 2

The three stations are classified as undisturbed or slightly disturbed, during all the sampling periods

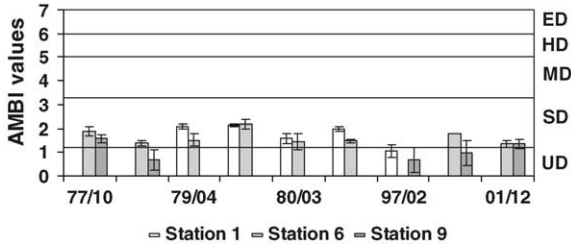


Fig. 3. AMBI values, with standard error, for Stations 1, 6 and 9 on the Kwintebank during different campaigns (yy/mm on x-axis).

(Fig. 3). A two-way ANOVA was carried out to compare the AMBI values between periods of low (late 1970s and 1980s) and high sand extraction intensity (late 1990s and 2001s) and between stations (with Stations 1 and 6 being heavily exploited and Station 9 only sporadically exploited). No significant

differences were found among the stations ($d.f. = 2$; $F = 3.31$; $p = 0.050$), whereas the AMBI was significantly lower during the periods of high sand extraction intensity ($d.f. = 1$; $F = 6.58$; $p = 0.013$).

3.3. Case study 3

AMBI values show a clear decreasing gradient, as one moves away from the stations located near the platform wells in all the studied cases (Fig. 4) and in the prevailing current direction: (i) from 0 to 100 m, AMBI values lie between 5 and 6 (heavily polluted) and the benthic community is dominated by first-order opportunistic species; (ii) from 100 to 500 m, the AMBI values lie between 3.3 and 5 (moderately polluted), except at Audrey (Fig. 4a), with increasing dominance of EG IV and III, and the presence of I and II; and (iii) from 500 to 1,000 m, the AMBI values are

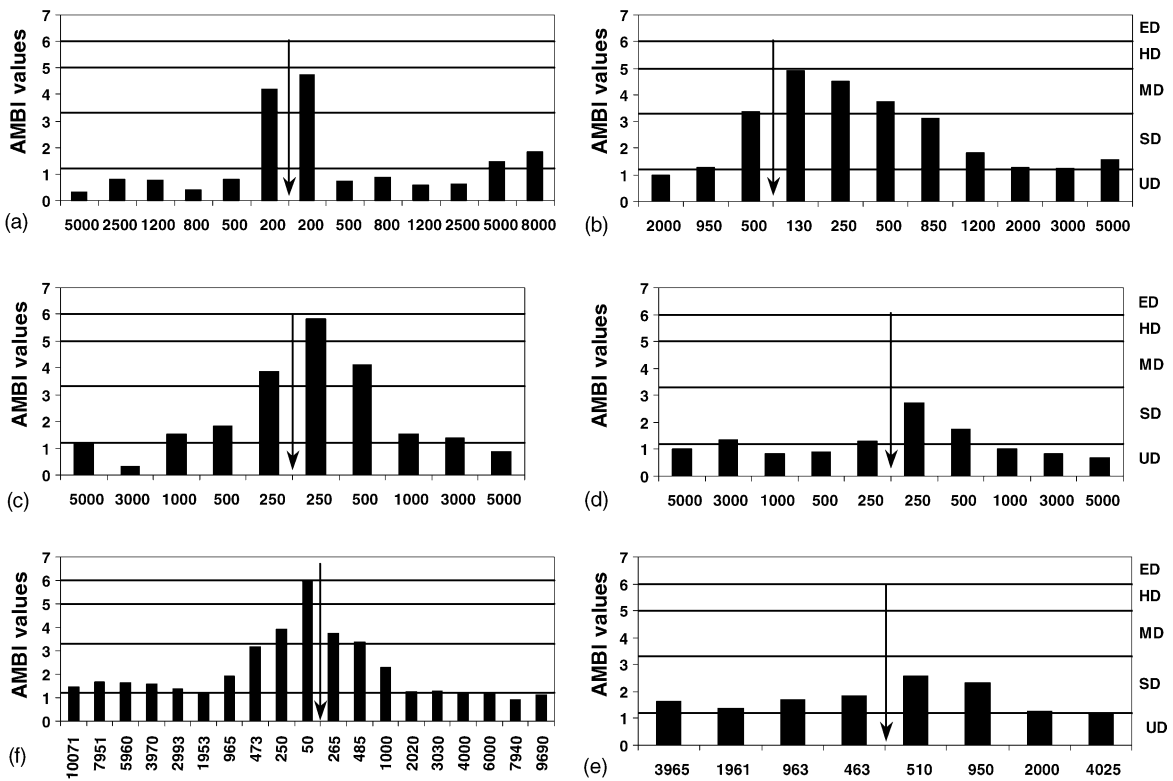


Fig. 4. AMBI values for each sampling station at the Audrey (a), Beryl A (b), Beryl B (c and d) and Thistle (e and f) oil platforms. The stations are labelled with the distance along the transect from the platform, which is located by an arrow. The transect orientations are: (a) and (e) 135°, (b) 168°, (c) 180°, (d) 90° and (f) 45°, relative to North. Key: UD = undisturbed; SD = slightly disturbed; MD = moderately disturbed; HD = heavily disturbed; and ED = extremely disturbed.

normally <1.2 (unpolluted), with EG I and II dominating. This gradient depends upon the regional prevailing current direction. Hence, the regression between the distance (from 0 to 1,200 m) and AMBI relating to stations in the prevailing current direction is: $AMBI = -0.004 \times \text{distance} + 5.354$, with the correlation being strong and highly significant ($F = 168.31$; $p = 0.000$; $r = 0.928$). At greater distances, there was only a weak correlation ($F = 0.389$; $p = 0.000$; $r = 0.150$).

Conversely, a strong and highly significant correlation was found between the total hydrocarbons in the sediment and the AMBI values, when data were available (Beryl A and Beryl B), following a logarithmic model ($F = 157.02$; $p = 0.000$; $r = 0.914$) (Fig. 5a). Hence, at the furthest stations, sensitive species are dominant in all cases. Approaching the oil platforms, they are progressively substituted by indifferent, tolerant and second- and first-order opportunistic species. These changes are related to the high hydrocarbon values in the sediment. Likewise, correlations between grain size and AMBI values, together with those between organic matter and AMBI, were only moderate ($p = 0.000$; $r < 0.50$).

3.4. Case study 4

Before the engineering works commenced (1989–1993) in Bilbao harbour, the AMBI values were very similar (except in 1990, with an AMBI = 2.1), with the area being classified as undisturbed or slightly disturbed and the EG I being dominant (Fig. 6). When intensive dredging finished in 1997, the area became totally azoic (AMBI = 7) as a result of the elimination of a surficial sediment layer of about 10 m. Over recent years, the area has improved in terms of its classification, but with AMBI values higher than previously, i.e. nearer to 3 (except in 2001, AMBI = 2.3); these represent a moderately or slightly disturbed situation.

3.5. Case study 5

At the dredged location, the proportion of first-order opportunistic species (EG V) increased within 3 days after dredging (from 30 to 60%; Fig. 7b). Between 15 and 90 days after dredging, the second-order opportunistic species (EG IV) largely dominated

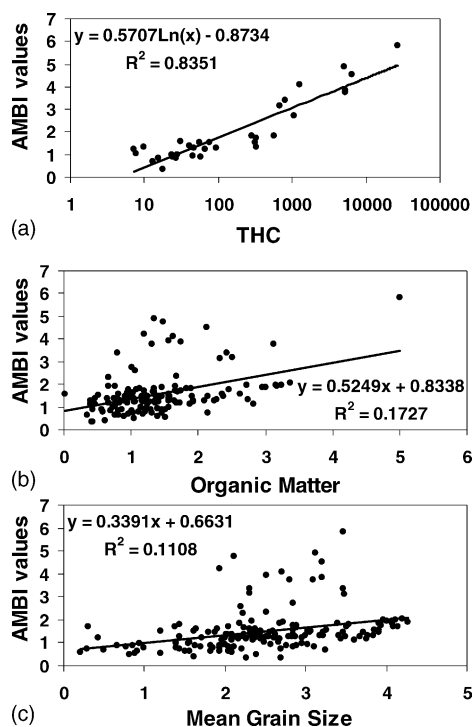


Fig. 5. Regressions between (a) AMBI values and total hydrocarbons (THC) measured in mg kg^{-1} , (b) percentage of organic matter content in sediment and (c) mean grain diameter, in Φ units.

(90%). Only after 180 days had the proportion returned to the situation before dredging (Fig. 7b). Likewise, in the control location there was a decrease in the sensitive species abundance (EG I), after 3 days (Fig. 7a). The most important change occurred after 15 days, when the second-order opportunistic (EG IV) species increased in their proportion to 75%, decreasing subsequently (Fig. 7a).

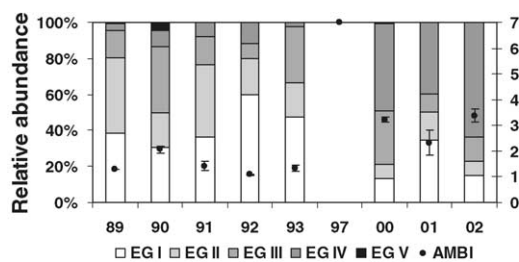


Fig. 6. Relative abundance of each ecological group for each replicate and average AMBI values, with standard error bars for each of the sampling occasions: 89, 90, 91, ... represent the sampling year.

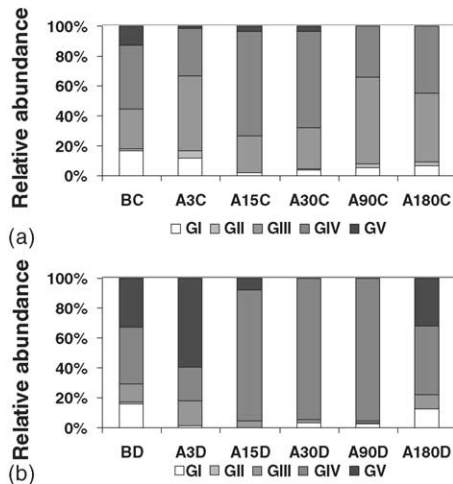


Fig. 7. Relative abundances of the five ecological groups (EG I–EG V), during each sampling (key: B = before dredging, A3 = 3 days after dredging; A15 = 15 days after dredging; etc.; C = control station; and D = dredged station) at the control station (a) and the dredged station (b).

This pattern is reflected in the AMBI values (Fig. 8): hence, the dredged location increased from 4.05 to 5.07 (moderately disturbed to heavily disturbed) after 3 days, decreasing to 4.54 after 15 days, then fluctuated around 4.3 (always moderately disturbed). Likewise, the control location improved within 3 days after dredging; it worsened after 15 days (reaching 4.09 values), then improved.

3.6. Case study 6

A comparison of the derived regression lines was carried out between the sites (Fig. 9). The AMBI was used as dependent variable, with the distance from the cages as independent variable. The regression was

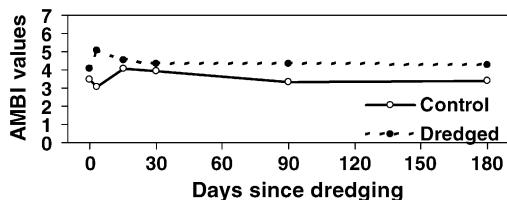


Fig. 8. Evolution of AMBI values throughout the BACI study, at the control station and the dredged station. Note: 0 day corresponds to the sampling undertaken before dredging commenced.

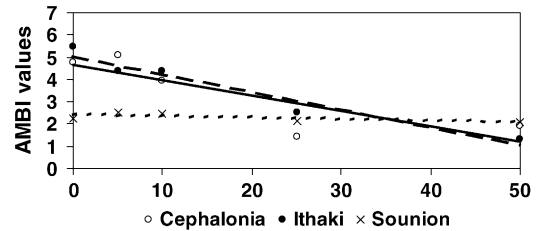


Fig. 9. Regression lines between AMBI values and the distance from the aquaculture cages, for each of the sites. Key: (—) Cephalonia; (---) Ithaki; (· · ·) Sounion.

highly significant for the model ($F = 10.38$; d.f. = 5; $p = 0.002$). No significant differences were detected between the Cephalonia and Ithaki regression lines, but significant differences were detected between Cephalonia and Sounion and between Ithaki and Sounion, both in terms of the intercept and the slope.

Hence, a clear gradient is detected by the AMBI, in Cephalonia and Ithaki. The AMBI values decrease, as the distance from the cages increases. In Sounion, the AMBI is a little bit higher below the cages than at the control site, but the gradient is not very clear. Under the Cephalonia and Ithaki cages, benthic communities should be considered as heavily disturbed: being: (a) moderately disturbed at a distance of between 5 and 10 m; (b) slightly disturbed at a distance of 25 m; and (c) undisturbed at 50 m.

4. Discussion

Even though the impact sources studied in this contribution are different, the effects can be grouped into three classes: (i) oxygen demand (Case 1: dissolved oxygen depletion); (ii) physical disturbance (Case 2: sand extraction; Case 4: engineering works; and Case 5: dredging activities); and (iii) increasing organic matter and associated pollutants (Case 3: drill mud dumping; and Case 6: aquaculture).

In Case study 1, 1 year after the anoxic episode, the deepest communities were still dominated by opportunistic species and were classified as heavily disturbed, the station at 95 m water depth was moderately disturbed and the shallowest stations were slightly disturbed. Two years after the anoxic episode, the benthic community had already recovered, with the area being classified as slightly disturbed or undis-

turbed. Such a rapid improvement was probably possible because the only impact was a punctual anoxic episode, rather than a chronic impact. This conclusion agrees with that of Rosenberg et al. (2002), i.e. following multivariate analysis, that the benthic communities at all depths more or less returned to the same faunal composition as during pre-disturbed conditions, with such a return being slowest at the deepest stations. These researchers stated that the benthic fauna succession in this fjord followed the Pearson–Rosenberg successional model, on which the AMBI is based (Borja et al., 2000, 2004b).

There are no major changes in the AMBI values in Case study 2, where all the stations are classified as undisturbed or slightly disturbed. Increasing sand extraction intensity, from the late 1970s onwards, did not result in an increase in the AMBI. Rather, it was lower during the period of high sand extraction intensity, than in the period of low extraction. The lower AMBI results from a decrease in the EG III. Moreover, the AMBI did not differ between stations characterised by different sand extraction intensities. Therefore, in this case study, the AMBI is not a good indicator for detecting the impacts of sand extraction. This conclusion is in agreement with that of Bonne et al. (2003), who did not detect an increased abundance of opportunistic species as a result of the extractions.

Elsewhere, it has been shown that the usefulness of the AMBI is not only limited to organic pollution assessment, but also reflects disturbances, for example, by hydrocarbons, engineering works and harbour dredging (see Table 1). Conversely, the reworking of ‘organic-poor’ sediments of subtidal sandbanks (Vanosmael et al., 1979, 1982) does not appear to favour typical opportunistic macrobenthic species. The competitive ability of the species classified as opportunistic species in the AMBI is probably not advantageous in organically poor and naturally–physically stressed environments, such as offshore subtidal sandbanks or the inner part of estuaries (as outlined by Borja et al., 2004b). Pearson and Rosenberg (1978) pointed out that the use of any indicator schemes must be accompanied by a detailed knowledge of both the abundance and range of species in the area concerned, together with complementary community structure information (as outlined by Borja et al. (2004b)).

Case study 4 is related also with physical disturbance. Before the engineering works commenced, the area was classified as unpolluted or slightly polluted. The pollutants carried by the Nervión river did not cause any detectable effect, due to dilution (Gorostiaga et al., 2004). Following the dredging period, a recovery was expected in the AMBI, from 7 to values close to those before the works. No data were available from 1997 to 1999, but from 2000 to 2003 AMBI values lie between 2 and 3.5; this indicates partial recovery in the macrozoobenthic community. The monitoring should be continued, to confirm this trend and ensure that the impact of the works, on the benthic community, was only transient. However, it is not expected to reach those values before the works due to the enclosure, which: (i) slows water renewal; (ii) causes important pollutant retention; and (iii) increases organic matter levels, as observed in other harbours (Muxika et al., 2003).

The pattern of recovery following dredging in Ceuta, as detected by the AMBI, is more evident than through the use of other tools (total abundance, diversity, evenness and Margalef’s index), as shown by Guerra-García et al. (2003). By means of AMBI, it is possible to deduce two different impact effects: (i) in the dredged area, there is an immediate effect due to the physical disturbance (detected within 3 days), with a posterior gradual recolonisation (occurring in the same period as mentioned by Dornie et al. (2003a)); and (ii) an effect on the ‘near-control’ area (due probably to the deposition of suspended materials, after several days), as detected by the AMBI after 15 days. The general pattern is similar to that shown by: (a) Sánchez-Moyano et al. (2004), in dredging works in the south of Spain; and (b) Case study 4, where the recovery is nearly total after 3 months.

Oil platforms can produce several environmental impacts, in response to the platform itself and to the discharge of drilling muds and cuttings (Frascari et al., 1992): (i) physical impacts, such as the generation of turbulence, erosion, changes in grain size, etc.; and (ii) biological, such as community changes and pollutant incorporation. For example, the amount of diesel oil discharged (associated with drill cuttings used in drilling operations), in 1981, into UK continental shelf waters, was estimated as 7,000 t (Davies et al., 1984). Drilling chemicals discharged in the same area up to 1989 were 39,902 t yr⁻¹ (Breuer et al., 2004).

In Case study 3, the highest AMBI values (therefore, the highest disturbance) are reached near the oil platforms, in all situations. Clear gradients are shown in all directions, but preferential currents are shown by the smoothest gradients (Fig. 4). The impact of the oil platforms reached up to 500–1,000 m, as detected by AMBI (in the northern North Sea, some of the stations located at 10,000 m from the platform are still slightly disturbed). The same pattern has been detected by Davies et al. (1984), for the same oilfields, using community structure parameters: likewise, by Borja et al. (2003a,b) using AMBI on ester-based muds in the Dutch area of the North Sea. Davies et al. (1984) detected oil concentrations between 1,000 and 10,000 times the background levels, within 250 m of the platforms; this explains the high correlation obtained, in this contribution, between the AMBI values and the total hydrocarbons. The pattern of distribution of the pollutants (together with its impact on the benthic community, detected by means of the AMBI) coincides with the axis of the most persistent current, often producing an ellipsoidal distribution (Davies et al., 1984). Further, Shimmield et al. (2000) found high disturbances in cores obtained from sediments retrieved at a distance of 65 m from the drilling cutting piles. Higher depletion of interstitial dissolved oxygen concentrations were found in these cores, in comparison with those obtained at 165 m and 300 m; likewise, higher heavy metal concentrations in the superficial layer and higher Ba concentrations.

The same distributional impact pattern is shown in Case study 6, in which AMBI values decrease as the distance from the cages increases. All the stations are moderately or heavily polluted, up to 25 m from the edge of the cages (the same pattern has been detected in fish-farms elsewhere (Mazzola et al., 2000)). At Sounion, this gradient is not detected; this can be explained by the high current velocities at this site, which are 6.3 cm s^{-1} (compared with only 3.5 and 2.8 cm s^{-1} at Cephalonia and Ithaki, respectively). Hence, high current velocities could spread the pollutants (organic carbon and nitrogen content of the sediment), avoiding a localised impact on the bed near the farm. This interpretation agrees with the conclusions of Karakassis et al. (2000), except that these authors detect also an impact on the macrofaunal community at Sounion. However, the faunal composition of Sounion differed from that at the other sites: for

example, *Capitella* cf. *capitata* dominated the macrofauna over distances of up to 10 m at Cephalonia and Ithaki, whereas the dominant species at Sounion was *Protodorvillea kefersteini*. This particular study concluded that ‘impacts of fish farming on benthos in the Mediterranean vary considerably depending on site characteristics’ (Karakassis et al., 2000). These conclusions, which are similar to those using the AMBI, were obtained using ABC curves and MDS ordination plot approaches (Karakassis et al., 2000).

Benthic communities, affected by different impact sources, react similarly; essentially, they change from sensitive groups (I and II) to lower successional stages (the opportunistic IV and V groups). The percentage of each EG in the community depends upon: (i) the intensity of the impact; (ii) the duration of the impact; and (iii) the distance from the source (as shown in this contribution). In the case of sand extraction, the community behaves in a different way, not detected by the AMBI as disturbance in this contribution. As mentioned by Rosenberg et al. (2002), benthic communities show great resilience and elasticity. Hence, the resilience of these communities following the cessation of the impact source is dependent upon the species composition, their different life-cycles, reproduction periods and larval dispersal patterns (Rosenberg et al., 2002). However, the AMBI appears to be independent of these controlling factors in most cases, because the case studies shown in this contribution, together with others presented previously (Borja et al., 2000, 2003a; Gorostiaga et al., 2004; Salas et al., 2004), include different settings, species and time-scales. The proportion between the different EGs appears to control the final result.

Conversely, elasticity (rapid recovery) is promoted by the presence of undisturbed communities in the vicinity of a particular site, as demonstrated when local impacts are produced (such as local hypoxia, dredging works, dumping, etc.). On this basis, Dornie et al. (2003b) suggest that physical and biological recovery rates are mediated by a combination of physical, chemical and biological factors; these, in turn, differ in their relative importance in different habitats. Hence, the AMBI could integrate, simply and usefully, these different factors into an unique number; this incorporates an equilibrium between the five EGs, connecting with the classical ecological theories on disturbance

models and recovery of impacted, or stressed, communities (Bellan, 1967; McArthur and Wilson, 1967; Pianka, 1970; Pearson and Rosenberg, 1978; Gray, 1979). This feature explains how the AMBI is able to respond successfully to very different environmental impact sources, including: drill cutting discharges; submarine outfalls; harbour and dyke construction; heavy metal inputs; eutrophication processes; diffuse pollutant inputs; recovery in polluted systems, under the impact of sewerage schemes; dredging processes; mud disposal; and oil spills (see Table 1, together with the associated references). That is why, combined with other metrics, it can also be a successful tool in implementing the WFD (Borja et al., 2004a,b).

5. Conclusions

This contribution has demonstrated the usefulness of the AMBI in detecting different and new impact sources and disturbance gradients. The results obtained with the AMBI are comparable with those obtained using other methods and parameters (including univariate and multivariate statistical analyses); as such, they are appropriate to what physico-chemical data show. The AMBI values provide a single and clear way (useful in terms of environmental advice) to establish the ecological quality of soft-bottom benthos, which is complementary to the above-mentioned methods.

The AMBI is appropriate to use for all European coastal environments, as it is independent of longitude and latitude, because it is based upon general ecological principles and paradigms. In this contribution, it has been used for the Atlantic Ocean region (Baltic Sea, North Sea and Bay of Biscay), as well as in the Mediterranean Sea (Gibraltar Strait and Greece). Results from 38 case studies, from all the European seas, have been collated. The AMBI reacts in the same way to different disturbance sources, e.g. anoxic episodes, hydrocarbon pollution, engineering works, dredging or fish farming cages.

The AMBI is useful to compare the ecological quality of the soft-bottom benthos in BACI experiments, environmental impact studies and in determining the ecological status (as in WFD).

However, the AMBI has not been shown to be useful in naturally-stressed and poor communities,

e.g. high hydrodynamic energy areas, subtidal sandbanks, and the inner parts of the estuaries, etc.

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References

- Bellan, G., 1967. Pollution et peuplements benthiques sur substrat meuble dans la région de Marseille. 1 Partie. Le secteur de Cortiu. Rev. Int. Océanogr. Med. VI–VII, 53–87.
- Bonne, W., Vincx, M., 2003. The impact of marine sand extraction on benthic copepod communities of a subtidal sandbank. In: Benthic Copepod Communities in Relation to Natural and Anthropogenic Influences in the North Sea, Chapter III. Ph.D. Thesis, Ghent University, Biology Department, Marine Biology Section, Belgium, pp. 165–202.
- Bonne, W., Rekecki, A., Vincx, M., 2003. Impact assessment of sand extraction on subtidal sandbanks using macrobenthos. In: Benthic copepod communities in relation to natural and anthropogenic influences in the North Sea, Chapter IV. Ph.D. Thesis, Ghent University, Biology Department, Marine Biology Section, Belgium, pp. 207–226.
- Borja, Á., Franco, J., Pérez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. Mar. Poll. Bull. 40, 1100–1114.
- Borja, Á., Muxika, I., Franco, J., 2003a. The application of a Marine Biotic Index to different impact sources affecting soft-bottom benthic communities along European coasts. Mar. Poll. Bull. 46, 835–845.

- Borja, Á., Franco, J., Muxika, I., 2003b. Classification tools for marine ecological quality assessment: the usefulness of macrobenthic communities in an area affected by a submarine outfall. ICES CM2003/Session J-02, Tallin, Estonia, 24–28 September 2003.
- Borja, Á., Franco, J., Valencia, V., Bald, J., Muxika, I., Belzunce, M.J., Solaun, O., 2004a. Implementation of the European Water Framework Directive from the Basque country (northern Spain): a methodological approach (viewpoint). *Mar. Poll. Bull.* 48, 209–218.
- Borja, Á., Franco, J., Muxika, I., 2004b. The biotic indices and the Water Framework Directive: the required consensus in the new benthic monitoring tools (correspondence). *Mar. Poll. Bull.* 48, 405–408.
- Breuer, E., Howe, J.A., Shimmield, G.B., Cummings, D., Carroll, J., 1999. Contaminant Leaching from Drill Cuttings Piles of the Northern and Central North Sea: A review. Centre for Coastal & Marine Sciences and Scottish Association for Marine Science, Dunstaffnage Marine Laboratory, Scotland, UK, 49 pp.
- Breuer, E., Stevenson, A.G., Howe, J.A., Carroll, J., Shimmield, G.B., 2004. Drill cutting accumulations in the Northern and Central North Sea: a review of environmental interactions and chemical fate. *Mar. Poll. Bull.* 48, 12–25.
- Casselli, C., Ponti, M., Abbiati, M., 2003. Valutazione della qualità ambientale della laguna costiera Pialassa Baiona attraverso lo studio dei suoi popolamenti bentonici. In: XIII Congresso Società Italiana de Ecología, Como, Villa Olmo, 8–10 Settembre 2003, poster.
- Clarke, K.R., 1990. Comparisons of dominance curves. *J. Exp. Mar. Biol. Ecol.* 138, 143–157.
- Codling, I.D., Ashley, S.J., 1992. Development of a biotic index for the assessment of pollution status of marine benthic communities. Final Report to SNIFFER and NRA, NR 3102/1.
- Dauer, D.M., 1993. Biological criteria, environmental health and estuarine macrobenthic community structure. *Mar. Poll. Bull.* 26, 249–257.
- Davies, J.M., Addy, J.M., Blackman, R.A., Blanchard, J.R., Ferbrache, J.E., Moore, D.C., Somerville, H.J., Whitehead, A., Wilkinson, T., 1984. Environmental effects of the use of oil-based drilling muds in the North Sea. *Mar. Poll. Bull.* 15, 363–371.
- Dernie, K.M., Kaiser, M.J., Richardson, E.A., Warwick, R.M., 2003a. Recovery of soft sediment communities and habitats following physical disturbance. *J. Exp. Mar. Biol. Ecol.* 285–286, 415–434.
- Dernie, K.M., Kaiser, M.J., Warwick, R.M., 2003b. Recovery dates of benthic communities following physical disturbance. *J. Anim. Ecol.* 72, 1043–1056.
- Eaton, L., 2001. Development and validation of biocriteria using benthic macroinvertebrates for North Carolina estuarine waters. *Mar. Poll. Bull.* 42, 23–30.
- Engle, V.D., Summers, J.K., Gaston, G.R., 1994. A benthic index of environmental condition of Gulf of Mexico estuaries. *Estuaries* 17, 372–384.
- Field, J.G., Clarke, K.R., Warwick, R.M., 1982. A practical strategy for analysing multispecies distribution patterns. *Mar. Ecol. Prog. Ser.* 8, 37–52.
- Fielding, A., 1992. Axiomatic approaches to scoring ordered classifications. Department of Economics, University of Birmingham. Discussion Paper 92-06, 30 pp.
- Forni, G., Occhipinti Ambrogi, A., 2003. Applicazione del coefficiente biotico (Borja et al., 2000) alla comunità macrobentonica del Nord Adriatico. Meeting of the Italian Society of Marine Biology, Tunisia, June 2003.
- Frascari, F., Rosso, G., Bortoluzzi, G., Barbanti, A., Bonvicini, A.M., Pagliai, R., Crema, A., Castelli, M., Mauri, R., Zunarelli, E., Orlando, D., Prevedelli, L., Ceffa, Ratti, S., 1992. Environmental impact of water based drilling muds and cuttings in a northern Adriatic Sea site. *Bull. Inst. Océanogr., Monaco*, no special 11, 305–324.
- Glémarec, M., Hily, C., 1981. Perturbations apportées à la macrofaune benthique de la baie de Concarneau par les effluents urbains et portuaires. *Acta Oecol.* 2, 139–150.
- Gorostiaga, J.M., Borja, Á., Díez, I., Francés, G., Pagola-Carte, S., Sáiz-Salinas, J.I., 2004. Recovery of benthic communities in polluted systems. In: Borja, Á., Collins, M. (Eds.). *Oceanography and Marine Environment of the Basque Country*, Elsevier, Amsterdam. Elsevier Oceanography Series no. 70, pp. 549–578.
- Grall, J., Glémarec, M., 1997. Using biotic indices to Estimate Macrobenthic Community Perturbations in the Bay of Brest. *Est. Coast. Shelf Sci.* 44, 43–53.
- Gray, J.S., 1979. Pollution-induced changes in populations. *Phil. Trans. Royal Soc., London Ser. B* 286, 545–561.
- Guerra-García, J.M., Corzo, J., García-Gómez, J.C., 2003. Short-term benthic recolonization after dredging in the harbour of Ceuta, North Africa. *P.S.Z.N.: Mar. Ecol.* 24, 217–229.
- Hill, M.O., 1974. Correspondence analysis: a neglected multivariate method. *Appl. Stat.* 23, 340–354.
- Hily, C., 1984. Variabilité de la macrofaune benthique dans les milieux hypertrophiques de la Rade de Brest. Thèse de Doctorat d'Etat, Univ. Bretagne Occidentale. vol. 1, 359 pp.; vol. 2, 337 pp.
- Jolliffe, I.T., 1986. *Principal component analysis*. Springer-Verlag, New York, 271 pp.
- Karakassis, I., Tsapakis, M., Hatziyanni, E., Papadopoulou, K.N., Plaiti, W., 2000. Impact of cage farming of fish on the seabed in three Mediterranean coastal areas. *ICES J. Mar. Sci.* 57, 1462–1471.
- Kendall, M.G., 1975. *Multivariate analysis*. Charles Griffin & Company Ltd, London, 210 pp.
- Kruskal, J.B., Wish, M., 1977. *Multidimensional Scaling*. Sage Publications, Beverly Hills, California, 96 pp.
- McArthur, R.H., Wilson, E.O., 1967. *The theory of island biogeography*. Princeton University Press, Princeton, USA, 224 pp.
- Majeed, S.A., 1987. Organic Matter and Biotic Indices on the Beaches of North Brittany. *Mar. Poll. Bull.* 18, 490–495.
- Mazzola, A., Mirto, S., la Rosa, T., Fabiano, M., Danovaro, R., 2000. Fish-farming effects on benthic community structure in coastal sediments: analysis of meiofaunal recovery. *ICES J. Mar. Sci.* 57, 1454–1461.
- Muxika, I., Borja, Á., Franco, J., 2003. The use of a biotic index (AMBI) to identify spatial and temporal impact gradients on benthic communities in an estuarine area. ICES CM2003/Session J-01.

- Pearson, T., Rosenberg, R., 1976. A comparative study of the effects on the marine environment of wastes from cellulose industries in Scotland and Sweden. *Am. Biol.* 5, 77–79.
- Pearson, T., Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr. Mar. Biol. Ann. Rev.* 16, 229–311.
- Pianka, E.R., 1970. On r- and K-selection. *Am. Nat.* 104 (940), 592–597.
- Reish, D.J., 1959. An Ecological Study of Pollution in Los Angeles—Long Beach Harbors, California. Allan Hancock Foundation Publications. Occasional Paper 22, 117 pp.
- Roberts, R.D., Gregory, M.R., Foster, B.A., 1998. Developing an efficient macrofauna monitoring index from an impact study—a dredge spoil example. *Mar. Poll. Bull.* 36, 231–235.
- Rosenberg, R., Agrenius, S., Hellman, B., Nilsson, H.C., Norling, K., 2002. Recovery of marine benthic habitats and fauna in a Swedish fjord following improved oxygen conditions. *Mar. Ecol. Prog. Ser.* 234, 43–53.
- Rygg, B., 1985. Distribution of species along pollution-induced diversity gradients in benthic communities in Norwegian fjords. *Mar. Poll. Bull.* 16 (12), 469–474.
- Salas, F., Neto, J.M., Borja, A., Marques, J.C., 2004. Evaluation of the applicability of a marine biotic index to characterise the status of estuarine ecosystems: the case of Mondego estuary (Portugal). *Ecol. Indic.* 4, 215–225.
- Sánchez-Moyano, J.E., Estacio, F.J., García-Adiego, E.M., García-Gómez, J.C., 2004. Dredging impact on the benthic community of an unaltered inlet in southern Spain. *Helgol. Mar. Res.* 58, 32–39.
- Schiffman, S.S., Reynolds, M.L., Young, F.W., 1981. Introduction to Multidimensional Scaling: Theory, Methods and Applications. Academic Press, New York, 440 pp.
- Shimmield, G.B., Breuer, E., Cummings, D.G., Peppe, O., Shimmield, T., 2000. Contaminant leaching from drill cuttings piles of the northern and central North Sea: field results from the Beryl “A” cuttings pile. Scottish Association for Marine Science, 28 pp.
- Smith, S.D.A., Rule, M.J., 2001. The effects of dredge-spoil dumping on a shallow water soft-sediment community in the Solitary Islands Marine Park, NSW, Australia. *Mar. Poll. Bull.* 42, 1040–1048.
- Sokal, R.R., Sneath, P.H.A., 1963. Principals of Numerical Taxonomy. Freeman and Co, San Francisco, CA, USA, 339 pp.
- Solís-Weiss, V., Aleffi, F., Bettoso, N., Rossin, P., Orel, G., Fondamiani, S., 2004. Effects of industrial and urban pollution on the benthic macrofauna in the Bay of Muggia (industrial port of Trieste, Italy). *Sci. Tot. Env.* 328, 247–263.
- Van dolah, R.F., Hyland, J.L., Holland, A.F., Rosen, J.S., Snoots, T.R., 1999. A benthic index of biological integrity for assessing habitat quality in estuaries of the southeastern U.S.A. *Mar. Environ. Res.* 48, 269–283.
- Vanosmael, C., Heip, C., Vincx, M., Claeys, D., Rappé, G., Braeckman, A., Van Gansbeke, D., 1979. De invloed van zandwinning op de bodemfauna voor de Belgische kust. Ministerie van volksgezondheid en leefmilieu, Brussels, 50 pp.
- Vanosmael, C., Willems, K.A., Claeys, D., Vincx, M., Heip, C., 1982. Macrobenthos of a sublittoral sandbank in the Southern Bight of the north Sea. *J. Mar. Biol. Assoc. UK* 62, 521–534.
- Warwick, R.M., 1986. A new method for detecting pollution effects on marine macrobenthic communities. *Mar. Biol.* 92, 557–562.
- Washington, H.G., 1984. Diversity, biotic and similarity indices: a review with special relevance to aquatic ecosystems. *Water Res.* 18, 653–694.
- Weisberg, S.B., Ranasinghe, J.A., Schaffner, L.C., Díaz, R.J., Dauer, D.M., Frithsen, J.B., 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. *Estuaries* 20, 149–158.