Response to oxygen deficiency (depletion): Bivalve assemblages as an indicator of ecosystem instability in the northern Adriatic Sea

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Abstract: Benthic communities represent a powerful tool for the detection of natural and anthropogenic disturbances, as well as for the assessment of marine ecosystem stability. This paper shows that Bivalve assemblages could serve as excellent indicators of disturbance and ecosystem instability. The goal of this study was to compare two sets of data in order to determine the differences between two different periods belonging to Bivalve assemblage in the muddy detritic bottom of the northern Adriatic Sea in the post-anoxic period during December 1989, 1990, 1991 and quite a while later, during 2003, 2004 and 2005. Abundances of some indicator species such as *Corbula gibba*, *Modiolarca subpicta*, and *Timoclea ovata* were detected during the post-anoxic period. Recruitment in the quality of Bivalve assemblages was proved by the ecologic and biotic indexes during 2003, 2004 and 2005, during a period of relatively stable ecological conditions. Fluctuation in Bivalve diversity due to the ecological quality of the marine ecosystem in the eastern part of the northern Adriatic Sea is also discussed.

Key words:

hypoxia; Bivalve assemblages; indicator species; soft bottoms; northern Adriatic Sea

Introduction

Recent reviews and summaries have provided good introductions on how hypoxia and anoxia came to be such a large and serious problem in the aquatic ecosystem (Gray et al*.* 2002; Galloway et al*.* 2008; Rabalais & Gilbert 2008; Vaquer-Sunyer & Duarte 2008; Levin et al. 2009). The term ‘anoxia’ refers to oxygen-free conditions, while ‘hypoxia’ refers to oxygen concentration below 2,0 mL L^{-1} (Diaz & Rosenberg 1995, 2008).

Oxygen depletion events (hypoxia) and anoxia are a key threat to shallow marine coastal seas worldwide (Stachowitsch 1991; Stachowitsch et al. 2007; Vaquer-Sunyer & Duarte 2008). Hypoxia is associated with restricted water exchange in semi-enclosed hydro-geomorphology, combined with water-column stratification (Diaz & Rosenberg 2008). At the top of the list of emerging environmental challenges are so-called 'Dead zones' (UNEP 2004), which are probably a key stressor on marine ecosystems (Vaquer-Sunyer & Duarte 2008). This is the case in the northern Adriatic Sea, a zone defined as a sensitive ecosystem (Stachowitsch 1984) that encompasses the following characteristics: semi-enclosed water body; river input, mostly from the Po River; soft bottoms (<50m); high productivity; and long water residence duration (Stachowitsch 1991; Ott 1992). This area is under pressure from eutrophication and, combined with marine mucilage events (the accumulation of dissolved organic matter), this can result in oxygen depletion (Justić et al. 1993; Degobbis et al. 1999) and, consequently, benthic mortalities (Stachowitsch et al. 1991; Ott 1992). Mucilage events and benthic mortalities in this area have occurred periodically for centuries (Crema et al. 1991) but, after 1969, (Piccinetti & Manfrin 1969) have become more frequent and more widespread than ever before (Voltolina

1973; Federa et al. 1976; Stachowitsch 1984; Faganeli et al. 1985; Riedel et al. 2008).

Tolerance to hypoxia and anoxia events is a question of physiological capacity and adaptability (Hagerman 1998), which varies greatly across marine benthic organisms (Diaz & Rosenberg 1995; Vaquer-Sunyer & Duarte 2008; Richards et al. 2009). Taxonomic groups like Crustacea and Echinodermata are often absent from the most perturbed parts of gradients, while molluscs are more tolerant (Pearson & Rosenberg 1978). Hypoxia alters community composition and abundance by killing sensitive species but favouring few tolerant forms as well as decreasing recruitment and growth (Dauer, 1993; Miller et al. 2002; Stierhoff et al. 2006). The identification of molluscan species is much easier than the identification of all macrobenthic taxa to the generic or family level, which provided plausible results in a number of cases (e.g. Warwick et al. 1990; Zenetos, 1996). Therefore, using only molluscan taxocoenosis, the procedures involved in the assessment of coastal marine environments that are subjected to natural or anthropogenic-induced forcing, may be less time-consuming (Koulouri 2006).

This study deals with communities that have experienced heavy disturbance due to oxygen crisis in northern Adriatic Sea during 1989 (Zavodnik et al. 1994; Stachowitsch 1991; Degobbis et al. 2000) and the subsequent decade – a period of relatively stable ecological conditions – using solely Bivalvia assemblages as an indicator of ecosystem disturbance. The presence of a well-developed macrofauna throughout this area is described in the early identifications (*Schizaster chiajei*-community) of the benthic communities by Vatova (1949) and other authors (Gamulin-Brida 1967; Orel & Menea 1969; Orel et al. 1987; Occhipinti-Ambrogi et al. 2002). As severe oxygen depletion was not recorded in the northern Adriatic Sea until after 1989, it could be assumed that the recovery of benthos within the investigated area has been achieved. To test the hypothesis on the recovery of benthic communities, assemblage structures of Bivalvia were investigated. The purpose of this study was to investigate the fluctuations of soft-bottom Bivalve diversity due to the ecological quality of the marine ecosystem in the eastern part of the northern Adriatic Sea.

Materials and methods

The investigation was conducted in the northern Adriatic Sea (SJ 005: 45°18,4'N; 13°08,0'E) on the soft bottom, which is characterized by depth of 31m and silty-sand type sediment that is dominated by fine and very fine fractions (Travizi 1998). The samplings were performed in two different periods, the first of which was carried out between 1989 and 1991, while the second was implemented between 2003 and 2005. Samples from first set of data (1989, 1990 and 1991) were divided in two groups: group 1, from the beginning of the research until April 1990, corresponding to the period of stagnation influenced by the severe oxygen depletions in the water column; group 2, from June 1990 to December 1991, corresponding to the recovery period (Zavodnik et al. 1994) when species diversity of total macrofauna doubled. The second set of data is represented as group 3, from February 2003 to December 2005, corresponding to the period of relatively stable ecological conditions. Four grab samples collected at same station were grouped together to calculate the indices. The sampled material was sieved through a 2mm mesh during the fieldwork and preserved in a 4% buffered formalin solution. Collected Bivalves were identified in the laboratory according to Tabble (1966), Nordsieck (1969), Parenzan (1974, 1976). For classification and nomenclature, Costello et al. (2001) was used.

The Bivalve assemblage structure was analyzed with the PRIMER software package (Plymouth Marine Laboratories, UK; Clarke & Warwick 1994). Data were transformed using 4th root transformation and the Bray-Curtis similarity matrix was used to generate 2-dimensional ordination plots with the non-metric multidimensional scaling (nMDS) technique. The ANOSIM 1-way test was applied to analyse differences in species assemblage between sampling periods (Clarke & Warwick 1994). The probability value set at 0.05 Margalef's Index (d) was used to analyse species richness (Margalef 1958), Pielou's Index was used to calculate evenness (Pielou 1969) and Shannon-Wiener's Index (Shannon-Weaver 1949) was used to analyze diversity. Marine biotic indices BENTIX by Simboura & Zenetos (2002), AMBI by Borja et al. (2000) and its multivariate extension M-AMBI (Muxika et al. 2007), which are based on the tolerance of macrobenthic taxa towards pollution, were applied in order to determine the Ecological Quality Status (EcoQS) of the sampling area. For the calculation, the Add-in version of BENTIX for MS Excel 2007 (available on HCMR’s webpage: http://www.hcmr.gr/listview3.php?id=1195) and version 4.1 of AMBI and M-AMBI, with the species list updated in February 2010 (available on AZTI’s webpage: http://ambi.azti.es) was used. SIMPER analysis (in PRIMER) was used to determine the contribution of each species to the average Bray-Curtis dissimilarity between sampling periods. This method of analysis determines which species are responsible for any differences that occur.

Results

A total of 56 species, belonging to 42 genera and 25 families were identified. Bivalve assemblages were represented by 14643 specimens among which the most abundant were *Timoclea ovata* (49.9%), *Corbula gibba* (20.8%) and *Modiolarca subpicta* (12.3%). Absolute and relative abundances of the dominant Bivalve species from December 1989 to December 2005 within the investigated area are shown in Fig. 1 and Fig. 2. During the period of stagnation (12/89–4/90) Bivalve assemblages were represented by dominance of *C. gibba* (84.4%) and *T. ovata* (11.5%). After April 1990, a prominent increase was determined with respect to the number of species and the diversity as well. *Timoclea ovata* together with *M. subpicta*, *Nucula nitidosa* and *Kurtiella bidentata* were the abundant species during that period (Fig. 2).

The Cardidae family was represented by six species (*Acanthocardia deshayesii*, *A. echinata*, *Laevicardium oblongum, Parvicardium minimum, P. scabrum, Plagiocardium papillosum*) while the Veneridae by five (*Clausinella fasciata, Venerupis aurea, Tapes* (*Tapes*) *rhomboides*, *Pitar rudis, T. ovata*), the Semelidae family by four (*Abra alba, A. prismatica, A. nitida, A. tenuis*) and the Nuculidae family by three (*N. nitidosa*, *N. nucleus, N. sulcata*). Remaining families were represented by only one or two species. A list of the Bivalve species and their abundance, dominance and constancy are provided in Table 1 as three groups.

The number of species during the period of stagnation and recovery ranged from five to sixteen and the number of specimens from 196 to 1504 (Fig. 3). With consideration to the period from December 1989 to December 1991, Pielou's Index (species evenness) ranged 0.13-0.68, and Shannon-Wiener's index 0.31-2.24 (Fig. 4). The lowest evenness values were determined during the first three months, as indicated by the assemblages that were comprised of a few dominant species e.g.

(0.20; 0.13; 0.24), as well as lower diversity and evenness values (d=0.69; J’=0.13), which were found in concomitance with a population explosion of the Bivalves *C. gibba*, highly dominant during February 1990 (Fig. 2).

From December 2003 to December 2005 the number of species increased and ranged from 16 to 25, while the number of specimens decreased with a range from 56 to 210 (Fig. 3). Margalef's Index (species richness) ranged 3.17-4.95, Pielou's Index (species evenness) 0.72-0.89, and Shannon-Wiener's Index 3.05-3.75 (Fig. 4).The mean BENTIX values determined at the sampling site ranged between 2.01 and 3.06 (Fig. 5). During the stagnation and recovery period (12/89–12/91), except for samples from September 1991, the mean BENTIX results were represented with values from the range of poor conditions (2 to 2.5) while moderate conditions prevailed during the period of relatively stable ecological conditions (2/03–12/05).

The mean AMBI values varied between 0.41 (May 2003) and 4.3 (February 1990) – see Fig. 6. The highest values (4.16, 4.3 and 4.0), which are in the range of moderate conditions, were calculated during the first three samplings of the study. The lowest AMBI values, which correspond to excellent (undisturbed) conditions, were determined after the August 1990 period of research. The mean m-AMBI values determined in the sampling area ranged between 0.2 (February 1990) and 0.87 (August 2004) – see Fig. 7. The lowest values (0.2, 0.22, 0.24 and 0.4), which are in the range of poor conditions, were calculated during the period of stagnation (12/89–4/90). The remaining periods were mostly represented by higher m-AMBI values within the range of good (slightly disturbed) conditions (Fig. 7).

In the MDS plot of the whole biological matrix (Fig. 8), the first samples analysed, up to those from the furthest distances of the plot (Bray-Curtis maximum dissimilarity) from the last samples, indicated that, during the study period, the Bivalvia assemblages were subjected to continuous evolution. The analysis of the Similarity test (ANOSIM) pairwise test (Table 2) was performed in order to test the hypothesis that structural changes in the community were primarily influenced by oxygen-deficient bottom waters. The SIMPER test shows that similarity in the first group was high due to the presence of *C. gibba* (ave. similarity=70.95%). Moderate dissimilarity between groups 1 and 2 (69.72%) was mainly due to presence of the 3 Bivalve species *C. gibba* (Contrib.%=21.51), relatively low number of *T. ovata* (Contrib.%=55.11) and *Modiolarca subpicta* (Contrib.%=14.29) in group 1; between groups 2 and 3 dissimilarity (89.05) was mainly due to presence of *T. ovate* (Contrib.%=53.19), *C. gibba* (Contrib.%=16.59) and *M. subpicta* (Contrib.%=13.15) in group 2; while dissimilarity between the first (group 1) and the last (group 3) set of samples was mainly determined by high density of *C. gibba* (Contrib.%=65.02) and *T. ovata* (Contrib.%=8.34) and a total absence of *K. bidentata, Tellina distorta*, *N. nitidosa*, *Phaxas pellucidus*, *A. alba* and *Thyasira flexuosa* in the samples of group 1 (Table 3).

Discussion

Some benthic species are considered to be useful biological indicators for aquatic ecosystems (Tabatabaie & Amiri 2010; Billet et al. 1983; Hrs-Brenko 2006). The macrobenthos, comprised of mostly non-migrant inhabitants, is used in the indices that expose the ecological changes in the marine ecosystem especially in the environmentally disturbed areas. Therefore, information on benthic macrofauna provides an integrative measure for assessing and improving the ecological health of the ecosystem (Pearson & Rosenberg 1978).

The results of the post-anoxic monitoring (1989, 1990, 1991) in the northern Adriatic Sea indicated a degradation of macrofauna recovery of benthic communities (Hrs-Brenko et al. 1994, 2006; Zavodnik et al. 1994; Degobbis et al. 2000). Bivalvia assemblages in the first set of data: group 1 and group 2 (1989, 1990 and 1991) were represented with high abundance of *C. gibba*, *M. subpicta*, *N. nitidosa* and *T. ovata*. *C. gibba* is widely distributed throughout the estuaries of northern Europe and the Mediterranean; it is considered an indicator of sediment instability (Pérès & Picard 1964), organic enrichment and anoxic conditions (Diaz & Rosenberg 1995; Jensen 1990). *Corbula* is well adapted to live in unstable environments as constantly polluted milieu and in coastal and offshore areas exposed to seasonal or occasional environmental disturbances (Jensen 1988, 1990; Tomassetti et al. 1997; Žerjav Meixner 2000; Hrs-Brenko 1994, 2006) as well as in areas that have low species diversity (Graeffe 1903; Rosenberg 1972, 1973, 1977; Hrs-Brenko 1981, 2006; Hrs-Brenko et al. 1994; Theodorou 1994; Borja et al. 2000; Pruvot et al. 2000; Solis-Weiss et al. 2004).

Considering the whole period of investigation, *C. gibba* was the second most abundant species (2462 individuals) while it was an exclusive species of group 1, during the period of stagnation (12/98–4/90). During temporary environmental stress, most Bivalve species protect themselves by closing their valves tightly; for instance, *Corbula* has a special valve construction that allows hermetic shell closure (Yonge 1946; Yonge & Thompson 1976; Bonvicini-Pagliai & Serpagli 1988). Some Bivalves stretch their siphons upwards into the water column to reach waters with higher oxygen concentrations (Jørgensen 1980). In addition, Bivalves have a protective mechanism that allows them to convert to anaerobic metabolism by increasing the lactate concentration in the tissues (Žerjav Meixner 2000).

In comprehensive review on the effects of hypoxia on benthic fauna, only *Corbula* – as an opportunistic species – survived mass mortality events (Diaz & Rosenberg 1995). Although *Corbula* as an oxygen-resistant species (Rosenberg 1977; Diaz & Rosenberg 1995) and survives well for a certain period in low oxygen conditions it may diminish in prolonged hypoxia (Baden et al. 1990). Consequently, this species has the capacity to have a significant ecological impact wherever it becomes established. Recruitment of *C. gibba* varies between years depending on biotic and abiotic conditions (Giangrande et al. 1994). Muss (1973) recorded *Corbula* spat from mid-August to early January, Rosenberg (1977) from September to November, Jensen (1988) within August and September. In the northern Adriatic, after the 1989 oxygen crisis, settlement season extended through the entire year to 1990 with a peak in the summer (Hrs-Brenko 2003).

Its proliferation during December 1991 (Fig. 2) could be connected with the significant influence of freshwater during late winter and throughout spring. Furthermore, during June, the sea surface layer was stabilized and rapidly heated to usual temperature values, which coincided with the appearance of large aggregates (mucilage) in the water column (Degobbis et al. 1995). Specific dominance of *Corbula* in December might be explained by oxygen deficiency that occurred from mid-September until November, and was probably associated with the massive settling of marine snow flocks in fall (Stachowitsch et al. 1990). The pioneer species, which are dominant within particular assemblages during the early recovery period (stagnation), are usually not typical of the community in consideration (Leppäkoski 1971). However, *C. gibba* with significantly low abundance was permanently present throughout the investigated area; Vatova (1949) and Crema et al. (1991) pointed out that *C. gibba* is a typical species and highly abundant lamellibranch of the transition zone between detritic and muddy bottoms. Together with *Atrina pectinata* and *Timoclea ovata*, *C. gibba* is described as an accompanied species for *Schizaster chiajei*-community margined with *Turritella* zoocenosis (Vatova 1949; Zavodnik et al. 1994).

From June 1990, with a decrease in the number of *Corbula* specimens, *Timoclea ovata* became significantly abundant in the area together with *Modiolarca subpicta* (Table 1, Fig. 1 and 2). Species *T. ovata* is considered to be either tolerant of, or show preference for, hypoxic conditions (Wilding 2006). After a mortality event in 1983, Stachowitch (1991) recorded the successful settlement of *M. subpicta* on the tunics of ascidians and that of *Hiatella arctica* in shelters throughout newly developed multi-species clumps. Both species, *M. subpicta* and *H. arctica,* together with numerous juveniles of *Modiolula phaseolina*, *Atrina pectinata, Pecten jacobaeus, Aequipecten opercularis, Limaria hians, Anomia* sp., *Laevicardium oblongum* and *C. gibba* occurred a couple of months after the 1989 benthic catastrophe in the northern Adriatic Sea (Hrs-Brenko et al. 1994). The abundance of *N. nitidosa* increased from October 1990 and, during the 2003 sampling, was one of the dominant species together with *K. bidentata, T. distorta* and *T. ovata*. *N. nitidosa* was not noted during the period of stagnation, although it can tolerate anaerobic conditions for several days and is able to thrive in poorly aerated sediments (Sabatini & Ballerstedt 2005). Previous studies also discuss recovery after the oxygen crisis in 1989 (Zavodnik et al. 1994; Travizi 2000) but according to the diversity and abundance of Bivalve assemblages in 2003, 2004 and 2005 and the abundance of total macrofauna, it could be considered that 1990 and 1991 was a transitional recovery period, or a period of forming stability in the benthic fauna. Compared with results of previous investigations (Vatova 1949; Gamulin-Brida 1967; Orel & Mennea 1969; Orel et al.

1987) this results reveal the relatively high stability of Bivalve assemblages and indicate the resilience (stability *sensu* Holling 1973; Pimm 1984) of benthic communities.

BENTIX, AMBI and m-AMBI are the indices that were designed in order to assess the ecological quality status of the European seas based on the classification of species into several ecological groups, which represents species-level sensitivity to pollutions. Although BENTIX and AMBI are based on the same concept, there are some discrepancies (Fig. 5-7), which could be attributed to different assignment of species to the different ecological groups. The number of ecological groups varied with regard to each index as the AMBI presumed five different ecological groups (from 1 to 5, corresponding to the most sensitive to the most tolerant species respectively) while BENTIX recognised only 2 groups (sensitive and tolerant species). On the other hand, some of the Bivalve species encountered in the sampling area, such as *M. subpicta* and *T. ovata,* are considered to be sensitive species (belonging to group 1) in accordance with AMBI, whereas they were revealed as tolerant species in BENTIX. This could be regarded as the main reason for the discordance between these two indices. Since these species were represented by a great number of individuals (*T. ovata* about 3800 ind./m^2, *M. subpicta* about 1200 ind./m^2) in the investigated area, regarding these as tolerant species could be reasonable. Moreover, readjustment of the assignment of species to the correct ecological groups in order to get more appropriate results seems feasible.

These results clearly separated the three periods of investigation suggesting higher faunistic affinity in the period of stagnation (group 1: 12/89–4/90) and recovery period (group 2: 6/90–12/91) to third one (group 3: 2/03–12/05) – see Table 2, Fig. 8. However, the speed of community recovery is uncertain and depends on abiotic factors, spawning season, larvae development and dispersion, food supply, recruitment, growth and lifespan as well as survival, which is influenced by both abiotic and biotic stress agents. Recovery to a community structure that is normal to a particular habitat depends on the severity of the hypoxia (Llanso 1992), salinity, temperature, current regimes, and latitude; also, it is generally more rapid in shallow waters than in deeper waters (Diaz & Rosenberg 1995).

Ecological problems similar to those in the northern Adriatic occurred in the New York Bight – it was projected that recovery of soft-bottom macrofauna assemblages could not be expected for several years (Steimle & Radosh 1979); in the Saltkallefjord, the recovery period was eight years (Rosenberg 1976); and, in the Gulf of Trieste, the epifaunal community was projected to require more than five years without the occurrence of a stress event (Stachowitcsh 1991). Some authors highlighted the difficulties in establishing a close relationship between single environmental factors (Cabioch 1968; Warwick & Uncles 1980; Seiderer & Newell 1999; Newelle et al. 2001).

The bottom system of the northern Adriatic seems to be great enough to support restoration even after large-scale anoxias, provided the system is left undisturbed for a certain period (Ott 1992). On the other hand, the northern Adriatic causes a rapid change of oceanographic properties in the water column that, if continued for a certain period, could consequently affect the vitality of benthic organisms. Considering this investigation, the recovery of benthic macrofauna and special communities within Bivalve assemblages is a lengthy process that is characterized by a naturally disturbed soft-bottom environment and by different sources of anthropogenic impact. The Bivalve is a group that contains representatives that are among the most hypoxia-tolerant marine organisms (Diaz &

Rosenberg 1995) although certain species are particularly intolerant (Nicholson & Morton 2000). This paper establishes that Bivalve assemblages could serve as a representative indicator in evaluating ecosystem stability and the results have helped to advance our understanding of ecological succession in the marine benthos of the northern Adriatic Sea.

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References

Baden S.P., Loo L.O., Pihl L. & Rosenberg R. 1990. Effects of eutrophication on benthic communities including fish - Swedish west coast. Ambio, **19**:113-122.

Billet D.S.M., Lampitt A.L., Rice A.L. & Mantoura R.F.C. 1983. Seasonal sedimentation of phytoplankton to deep-sea benthos. Nature **302**: 520-522.

Bonvicini-Pagliai A.M. & Serpagli F. 1988. *Corbula gibba* (Olivi) as a time recorder of environmental stress. A first contribution. Rapp. Comm. int. Mer Médit., **31 (Suppl 2)**: 17.

Borja A., Franko J. & V. Pérez. 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environment. Mar. Pollut. Bull., **40**: 1100-1114.

Cabioch L. 1968. Contribution a la connaissance des peuplements benthiques de la Manche occidentale. - Cahiers de Biologie Marine. Tome IX Cahier **5**: Editions de la Station Biologique de Roscoff.

Clarke K.R. & Warwick R.W. 1994. Change in marine communities: an approach to statistical analysis and interpretation. Plymouth Marine Laboratory, 859 p.

Costello M.J., Emblow C. & White R.J. (Ed.) 2001. European register of marine species: a check-list of the marine species in Europe and a bibliography of guides to their identification. Collection Patrimoines Naturels, 50. Muséum national d'Histoire naturelle: Paris. 463 pp.

Crema R., Castelli A. & Prevedelli D. 1991. Long term eutrophication effects on macrofaunal communities in the Northern Adriatic Sea. Mar. Pollut. Bull. **22:** 503–508.

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

Degobbis D., Fonda Umani S., Franco P., Malej A., Precali R. & Smodlaka, N. 1995. Changes in the northern Adriatic ecosystem and hypertrophic appearance of gelatinous aggregates, Sci. Total. Environ. **165:** 43–58.

Degobbis D., Malej A. & Fonda Umani. S. 1999. The mucilage phenomenon in the northern Adriatic Sea. A critical review of the present scientific hypotheses. Ann. 1st. Super. Sanitá **35**: 373–381.

Degobbis D., Precali R., Ivancic I., Smodlaka N., Fux D. & Kveder S. 2000. Long-term changes in the northern Adriatic ecosystem related to anthropogenic eutrophication, Int. J. Environ. Poll., **13**: 495–533.

Diaz R.J. & Rosenberg R. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioral responses of benthic macrofauna. Oceanography and Marine Biology, An Annual Review 33: 245-303.

Diaz R.J. & Rosenberg R. 2008. Spreading dead zones and consequences for marine ecosystems. Science 321: 926-928.

Faganeli J., Avèin A., Fanuko N., Malej A., Turk, V., Tušnik P., Vriser B. & Vukovic A. 1985. Bottom layer anoxia in the central part of the gulf of Trieste in the late summer 1983. Mar. Pollut. Bull., **16**: 75-78.

Fedra K., Ölscher E.M., Scherübel C., Stachowitsch M. & Wurzian. R.S. 1976. On the ecology of a North Adriatic benthic community: distribution, standing crop and composition of the macrobenthos. Marine Biology **38**: 129–145.

Galloway J.N., Townsend A.R., Erisman J.W., Bekunda M., Cai Z., Freney J.R., Martinelli L.A., Seitzinger S.P. & Sutton. M.A. 2008. Transformation of the nitrogen cycle: Recent trends, questions, and potential solutions. Science 320: 889-892.

Gamulin-Brida H. 1967. Contribution aux recherches sur la bionomie des fonds meubles de l'Adriatique du Nord. Thalassia Jugosl., **3 (Suppl 1-6)**: 23-33.

Giangrande A., Geraci S. & Belmonte G. 1994. Life-cycle and life-history diversity in marine invertebrates and the implications in community dynamics. Oceanogr. Mar. Biol. Annu. Rev., **32**: 305-333.

Graeffe E. 1903. Uebersichte der Seettiere des Golfes von Triest, VI. Mollusca (Review of marine animals in the Gulf of Trieste. VI. Mollusca). Arb. Zool. Inst. Wien, Zool. Sta. Trieste, **14**: 89-136.

Gray J.S., Wu R.S.S. & Or Y.Y. 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. Mar. Ecol. Prog. Ser. **238**: 249-279.

Hagerman L. 1998. Physiological flexibility; a necessity for life in anoxic and sulphidic habitats. Hydrobiologia **376**: 241-254.

Holling C.S. 1973. Resilience and stability of ecological systems. A. Rev. Ecol. Syst. **4**: 1-23

Hrs-Brenko M. 1981. Population studies of *Corbula gibba* (Olivi), Bivalvia, Corbulidae, in the northern Adriatic Sea. J. Molluscan Stud., **47**: 17-24.

Hrs-Brenko M. 1998. Considerations on the 1990 and 1991 bivalve repopulations in the northern Adriatic Sea (Croatia). Period. biol.**100**(1), 59–62.

Hrs-Brenko M. 2003. The role of bivalve *Corbula gibba* (Olivi, 1792) (Corbulidae, Mollusca Bivalvia) in the recruitment of benthic communities in the northern Adriatic (in Croatian). Pomorski Zbornik, **41**: 195- 207.

Hrs-Brenko M. 2006. The basket shell, *Corbula gibba* Olivi, 1792 (Bivalve Mollusks) as a species resistant to environmental disturbances: Acta Adriat., **47 (Suppl 1)**: 49-64.

Hrs-Brenko M., Medaković D., Labura Ž. & Zahtila E. 1994. Bivalve recovery after a mass mortality in the autumn of 1989 in the northern Adriatic Sea. Period. Biol., **96 (Suppl 4)**: 455-458.

Jensen J.N. 1988. Recruitment, growth and mortality of juvenile *Corbula gibba* and *Abra alba* in the Limfjord, Denmark. Kieler Meeresforsch. Sonderh., **6**: 357-365.

Jensen J.N. 1990. Increased abundance and growth of the suspension-feeding bivalve *Corbula gibba* in a shallow part of the eutrophic Limfjord, Denmark. Neth. J. Sea Res., **27**: 101-108.

Jørgensen B.B. 1980. Seasonal oxygen depletion in the bottom waters of a Danish fjord and its effects on the benthic community. Oikos **34**: 68-76.

Justić D., Rabalais N.N., Turner R. E & Wiseman Jr.W.J. 1993. Seasonal coupling between riverborne nutrients, net productivity and hypoxia. Mar. Pollut. Bull. **26**: 184-189.

Koulouri P., Dounas C., Arvanitidis Ch., Koutsoubas D. & A. Eleftheriou. 2006. Molluscan diversity along a Mediterranean soft bottom sublittoral ecotone. Scientia Marina **70 (Suppl 4)**: 573-583.

Leppäkoski E. 1971. Benthic recolonization of the Bornholm basin (Southern Baltic) in 1969–71. Thalassia Jugosl., **7**: 171–179.

Levin L.A., Ekau W., Gooday A.J., Jorissen F., Middelburg J.J., Naqvi W., Neira C., Rabalais N.N. & Zhang J. 2009. Effects of natural and human-induced hypoxia on coastal benthos. Biogeosciences Discussion **6**: 3563-3654.

Llansó R.J. 1992. Effects of hypoxia on estuarine benthos: the lower Rappahannock River (Chesapeake Bay), a case study. Estuar Coast Shelf Sci **35**: 491–515.

Margalef R. 1958. Information theory in ecology. Gen. Syst., **3**: 36-71.

Miller D.C., Poucher S.L. & Coiro L. 2002. Determination of lethal dissolved oxygen levels for selected marine and estuarine fishes, crustaceans, and a Bivalve. Mar Biol **140**: 287-296.

Muxika I., Borja A. & Bald J. 2007. Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. Marine Pollution Bulletin, **55** (1-6), 16-29.

Muus K. 1973. Settling, growth and mortality of young bivalves in the Øresund. Ophelia, **12**: 79-116.

Newell R.C., Seiderer L.J., Robinson J.E. & Simpson N.M 2001. London Gateway Port Project benthic biological resource survey of the lower Thames Estuary, July-September 2001. Marine Ecological Surveys Limited, Cornwall.

Nicholson S. & Morton B. 2000. The Hypoxia tolerances of subtidal marine bivalves from Hong Kong. In Morton B (ed) The Marine flora and fauna of Hong Kong and Southern China V, Proceeding of the Tenth International Marine Biological Workshop: The Marine Flora and fauna of Hong Kong and Southern China, Hong Kong, Hong Kong University Press, (pp. 229239).

Nordsieck F. 1969. Die europäischen Meeresmuscheln (Bivalvia). G. Fischer Verlag, Stuttgart, 256 pp.

Occhipinti-Ambrogi A., Favruzzo M. & Savini D. 2002. Multi-annual variations of macrobenthos along the Emilia-Romagna coast (Northern Adriatic). PSZN Mar. Ecol. **23**: 307-319.

Orel G. & Mennea B. 1969. I popolamenti bentonici di alcuni tipi di fondo mobile del Golfo di Trieste. Pubbl. Staz. Zool. Napoli, **37 (Suppl 2)**: 261-276.

Orel G., Marocco A., Vio E., del Piero D. & Seta G. 1987. Sedimenti e biocenosi bentoniche tra la foce del Po ed il golfo di Trieste (Alto Adriatico). Bull. Ecol.**18**: 229-241.

Ott J. 1992. The Adriatic benthos: Problems and perspectives,. *in* Colombo, G., Ferrari, I., Ceccherelli, V. U., and Rossi, R., eds., Marine Eutrophication and Population Dynamics: Olsen & Olsen, Fredensborg, Denmark: 367-378.

Parenzan P. 1974. Carta d'indetita delle conchiglie del Mediterraneo. Bivalvia. Parte primo. Ed. Bios Taras, Taranto, 1-277.

Parenzan P. 1976. Carta d'indetita delle conchiglie del Mediterraneo. Bivalvia. Parte secondo. Ed. Bios Taras, Taranto, 283-546.

Pearson T.H. & Rosenberg R. 1978. Macrobentic succession in relation to organic enrichment and polluttion of the marine enviroment. Oceanogr. Mar. Biol., Annu. Rev., **16**: 229-311.

Pérès J.M. & Picard J. 1964. Nouveau manuel de Bionomie bentique de la mer Méditerranée. Recl. Trav. Stn. Mar. Endoume, **31(Suppl 47)**: 3-137.

Piccinetti C. & Manfrin G. 1969. Osservazioni sulla mortalita di pesci e di altri organismi verificatasi nel 1969 in Adriatico. Note Lab Biol Mar Pesca-Fano **3**: 73-92.

Pielou E. 1969. The measurement of diversity in different types of biological collections. J. Theor. Biol., **13**: 131-144.

Pimm S.L. 1984. The complexity and stability of ecosystems. Nature **307**: 321-326.

Pruvot C., Empis A. & Dhainaut-Courtois N. 2000. Présence du mollusque Bivalve *Corbula gibba* (Olivi, 1792) dans les sédiments meubles du port Est de Dunkerque (Mer du Nord). (New record of the mollusk *Corbula gibba* (Olivi, 1792) in the muddy sands of Dunkirk harbour (Nord Sea)). Bull. Soc. Zool. Fr., **125**: 75-82.

Rabalais N.N. & Gilbert. D. 2008. Distribution and consequences of hypoxia. Chapter 11, pp. 209-225 in E. R. Urban, Bjørn Sundby, P. Malanotte-Rizzoli, and Jerry Milello (eds), Watersheds, Bay, and Bounded Seas: The Science and Management of Semi-Enclosed Marine Systems. Island Press, Washington, D.C.

Richards J.G., Farrell A.P. & Brauner C.J. (eds.). 2009. Hypoxia. Fish Physiology Vol. 27, Burlington: Academic Press. 528 p.

Riedel B., Zuschin M. & Stachowitsch M. 2008. Dead zones: a future worst-case scenario for Northern Adriatic biodiversity. In: Climate warming and related changes in Mediterranean marine biota. F. Briand (ed.). CIESM Workshop Monograps **35**: 73-77.

Rosenberg R. 1972. Benthic faunal recovery in a Swedish fjord following the closure of a sulphite pulp mill. Oikos, **23**: 92-108.

Rosenberg R. 1973. Succession in benthic macrofauna in a Swedish fjord subsequent to the closure of a sulphite pulp mill. Oikos, **24**: 244-258.

Rosenberg R. 1976. Benthic faunal dynamics during succession following pollution abatement in a Swedish estuary. - Oikos **27**: 414-427.

Rosenberg R. 1977. Benthic macrofaunal dynamics, production, and dispersion in an oxygen-deficient estuary of west Sweden. J. Exp. Mar. Biol. Ecol., **26**:1 07-133.

Sabatini M. & Ballerstedt S. 2005. *Nucula nitidosa*. A bivalve mollusc. Marine LifeInformation Network: Biology and Sensitivity Key Information Sub-programme [on-line]. Plymouth: Marine Biological Association of the United Kingdom.

Seiderer L.J. & Newell R.C. 1999. Analysis of the relationship between sediment composition and benthic community structure in coastal deposits: Implications for marine aggregate dredging. ICES Journal of Marine Science **56**: 757-765.

Shannon C.E. & Weaver W. 1949. The Mathematical Theory of Communication. Urban, University of Illinois Press, 255 pp.

Simboura N & A. Zenetos 2002. Benthic indicators to use in ecological quality classification of Mediterranean soft bottoms marine ecosystems, including a new biotic index. Mediterranean Marine Science. **3/2**: 77-111.

Solis-Weiss V., Aleffi F., Bettoso N., Rossin P., Orel G. & Umani S.F. 2004. Effects of industrial and urban pollution on the benthic macrofauna in the Bay of Muggia (industrial port of Trieste, Italy). Sci. Total Environ., **328**: 247-263.

Stachowitsch M. 1984. Mass mortality in the Gulf of Trieste: The course of community destruction. P.S.Z.N.I: Mar. Ecol. **5**: 243-264.

Stachowitsch M. 1991. Anoxia in the northern Adriatic Sea: rapid death, slow recovery. In: R.V.Tyson, T.H. Pearson, Modern and ancient Continental Shelf Anoxia. Geol. Soc. Spec. Publ., **58**: 95-105.

Stachowitsch M., Fanuko N. & Richter M. 1990. Mucus Aggregates in the Adriatic Sea: An Overview of Stage and Occurrences. P.S.Z.N. I: Marine Ecology, **11 (Suppl 4)**: 327-350.

Stachowitsch M., Riedel B., Zuschin M. & Machan R. 2007. Oxygen depletion and benthic mortalities: the first in situ experimental approach to documenting an elusive phenomenon, Limnol. Oceanogr. Methods **5**: 344-352.

Steimle F.W. & Radosh D.J. 1979. Effects on the benthic invertebrate community. In Swanson, RL., and C.J. Sindermann (eds.), Oxygen depletion and associated benthic mortalities in the New York Bight, 1976, p.281-293. NOAA Prof. Pap. **11**, Natl. Oceanic Atmos. Adm., Rockville, MD.

Stierhoff K.L., Targett T.E. & Miller K.L. 2006. Ecophysiological responses of juvenile summer and winter flounder to hypoxia: experimental and modelling analyses of effects on estuarine nursery quality. Mar Ecol Prog Ser **325**: 255-266.

Tabatabaie T. & Amiri F. 2010. The impact of industrial pollution on macrobenthic fauna communities. African Journal of Environmental Science & Technology Vol. **4 (Suppl 9)**: 547-557.

Tebble N. 1966. British bivalve seashells. A handbook for identification. Trustees of the British Museum (Natural Hystory), London, 212 pp.

Theodorou A.J. 1994. The ecological state of the Elefsis Bay prior to the operation of the Athens Sea outfall. Water Sci. Technol., **30**: 161-171.

Tomassetti P., Tosti M. & De Santis A. 1997. Distribution of *Corbula gibba* (Olivi) (Mollusca, Bivalvia) in two areas of the northern Adriatic Sea related to some physical-chemical parameters. Biol. Mar. Mediterr. Vol. **4 (Suppl 1)**: 452-455.

Travizi A. 1998. Recovery of meiofauna after anoxic stress. II Spatial distribution. Period. Biol., 100, **1**: 71-79.

Travizi A. 2000. Effect of anoxic stress on density and distribution of sediment meiofauna. Period. Biol., **102**: 207-215.

UNEP United Nations Environment Programme. 2004. GEO Year Book 2003. GEO Section/UNEP, Nairobi.

Vaquer-Sunyer R. & Duarte C. M. 2008. Thresholds of hypoxia for marine biodiversity. Proceedings of the National Academy of Sciences of the United States of America 105: 15452-15457.

Vatova A. 1949. La fauna bentonica dell'alto e medio Adriatico. Nova Thalassia.,**1**: 1-100.

Voltolina D. 1973. A phytoplankton bloom in the lagoon of Venice Arch Oceanogr Limnol 18: 1-37.

Warwick R., Platt H., Clarke K., Agard J. & Gobin J. 1990. Analysis of macrobenthic and meiobenthic community structure in relation to pollution and disturbance in Hamilton harbour, Bermuda. J Exp Mar Biol Ecol **138**: 119-142.

Warwick R.M. & Uncles R. J. 1980. The distribution of benthic macrofauna associations in the Bristol Channel in relation to tidal stress. Mar. Ecol. Prog. Ser. **3**: 97-103.

Wilding T.A. 2006. The benthic impacts of the Loch Linnhe artificial reef. Hydrobiologia 555: 345 – 353

Yonge C.M. 1946. On the habits and adaptations of Aloidis (Corbula) gibba. J. Mar. Biol. Ass. U.K., **27**: 585-596.

Yonge C.M. & Thompson T.E. 1976. Living Marine Molluscs. William Collins Son and Co., Ltd., London.

Zavodnik D., Travizi A. & Jaklin A. 1994. Phytoplankton bloom consequences on benthic organisms. UNEP/FAO: Final reports on research projects dealing with eutrophication problem. MAP Tech. Repts Ser., UNEP, Athens., **78**: 91-121.

Zenetos A. 1996. Classification and interpretation of the established Mediterranean biocoenoses based on bivalvia mollusca alone: case study in an Ionian Gulf. J. mar. biol. Ass. U.K., **76(2)**: 403-416.

Žerjav Meixner V. 2000. Potrošnja kisika i ponašanje školjkaša *Corbula gibba* (Olivi, 1792) u eksperimentalnim uvjetima. (Oxygen consumption and the behaviour of shellfish *Corbula gibba* (Olivi, 1792) in the experimental conditions). Thesis, University of Zagreb, 98 pp.

Table and figure legends:

Fig. 1. Absolute abundance of dominant bivalves species from December 1989 to December 2005 in the investigated area.

Fig. 2. Relative abundance of dominant bivalves species from December 1989 to December 2005 in the investigated area.

Table 1. Average abundance dominance and constancy of Bivalves during the investigated period: Group 1 = period of stagnation (12/89-4/90). Group 2=recovery period (6/90-12/91). Group 3=resilience (stability *sensu* Holling 1973; Pimm 1984) (2/03-12/05).

Fig. 3. Number of species (S) and number of specimen (N) from December 1989 to December 2005 in the investigated area.

Fig. 4. Pieloul's index (J') and Shannon-Wiener's index (H') from December 1989 to December 2005 in the investigated area.

Fig. 5. Mean BENTIX values of the sampling site with standard deviation.

Fig. 6. Mean AMBI values of the sampling site with standard deviation.

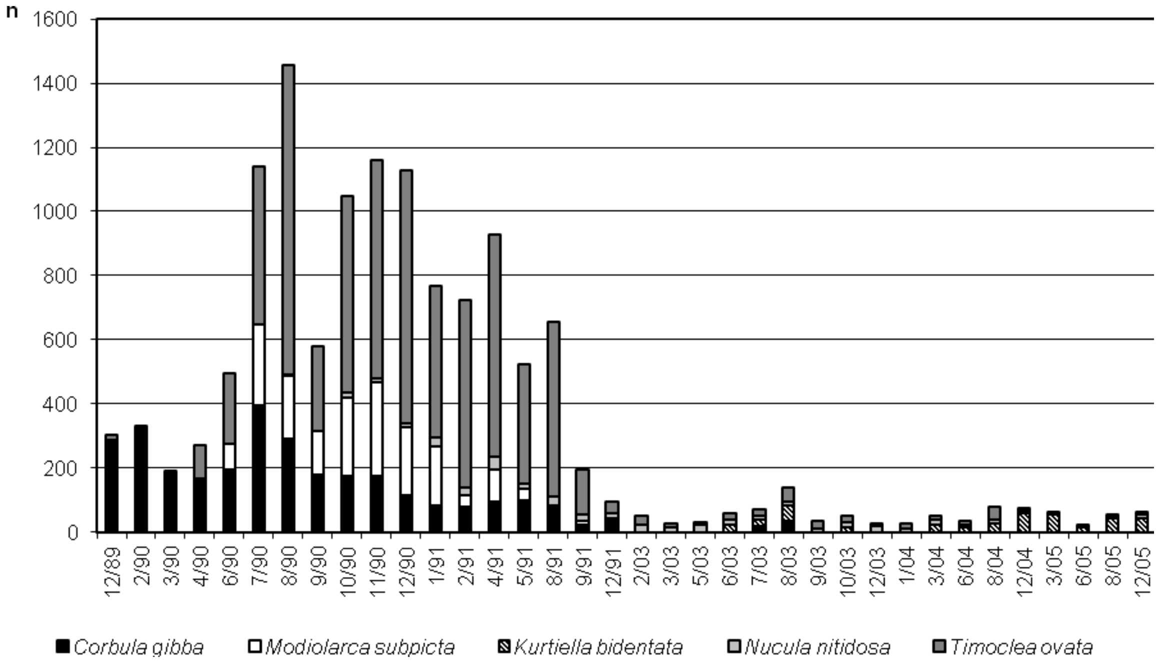
Fig. 7. Mean m-AMBI values of the sampling site with standard deviation.

Fig. 8. a) MDS diagram of the biological matrix: G1=group 1 (12/89–4/90), G2=group 2 (6/90–12/91) and G3=group 3 (2/03–12/05). Stress value=0.11. b) Hierarchical classification of three sampling groups. The affinity level (based on Bivalvia composition) was expressed as % of faunistic similarity among groups.

Table 2. Anosim pairwise test. Global R=0.975. p<0.001. Groups: 1=12/89–4/90; 2=6/90-12/91 and 3=(2/03-12/05). (* =statistical significant differences).

Table 3. Species contributions to average dissimilarity between soft bottom macrobenthos samples of group 1 (12/89-4/90). group 2 (6/90-12/91) and group 3 (2/03–12/05). Cut-off percentage 90%. Av. Abund=Average abundance; Av. Diss=Average dissimilarity; SD=Standard Deviation: Contrib%=species contribution to the average dissimilarity between groups; Cum.%=cumulative percentage.

Fig.1.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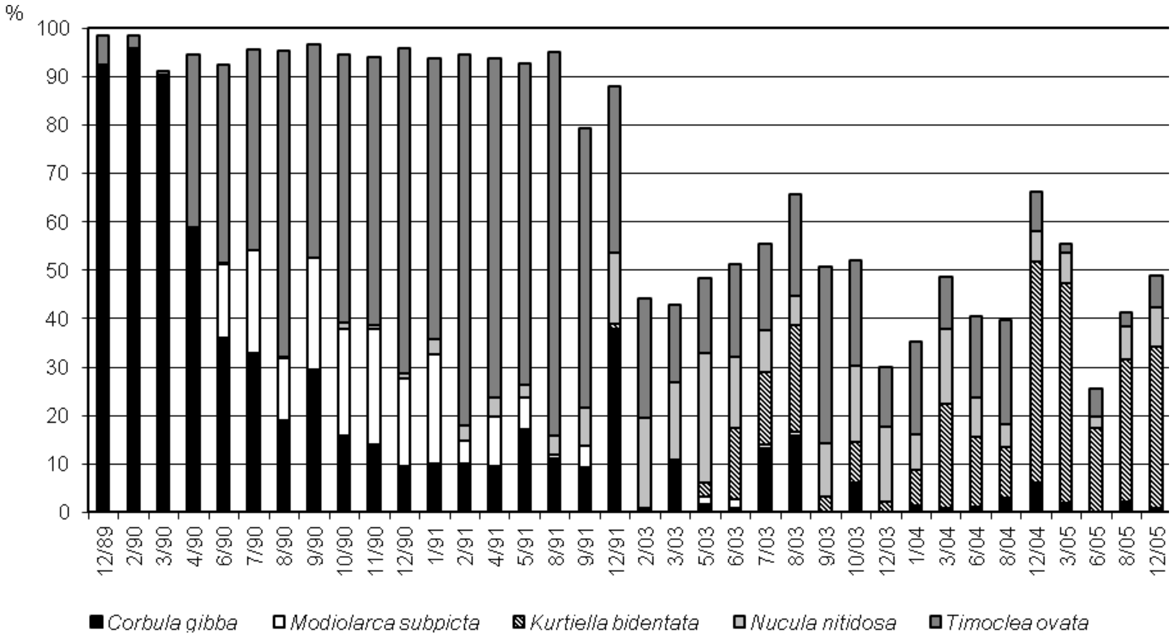
Corbula gibba
Modiolarca subpicta
Kurtiella bidentata
Nucula nitidosa
Timoclea ovata

Fig. 2.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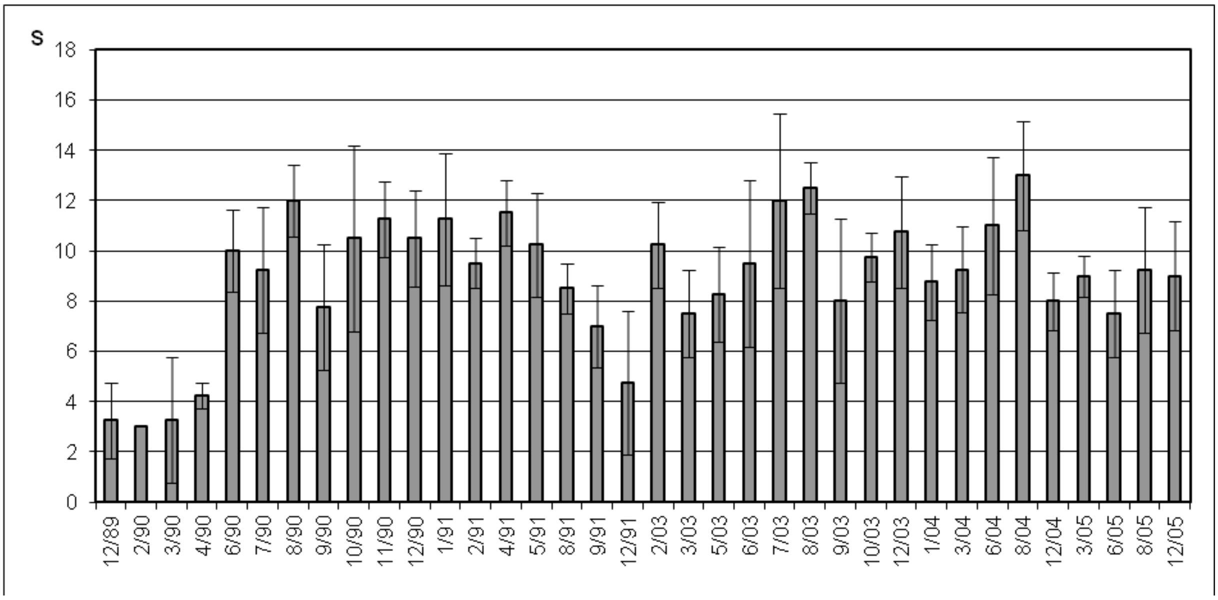
Corbula gibba
Modiolarca subpicta
Kurtiella bidentata
Nucula nitidosa
Timoclea ovata

Table 1.

	Group 1			**Group 2**			**Group 3**		
Species	Av.	D(%)	K(%)	Av.	D(%)	K(%)	Av.	D(%)	K(%)
Abra alba (W. Wood, 1802)	0.00	0.00	0.00	4.86	0.59	85.71	8.22	7.58	94.44
Abra nitida (Müller, 1776)	0.00	0.00	0.00	0.00	0.00	0.00	0.39	0.36	27.78
Abra prismatica (Montagu, 1808)	0.00	0.00	0.00	0.00	0.00	0.00	3.33	3.07	77.78
Abra tenuis (Montagu, 1803)	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.26	5.56
Abra sp.	0.00	0.00	0.00	0.79	0.10	14.29	0.22	0.20	11.11
Acanthocardia deshayesii (Payraudeau, 1826)	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.05	5.56
Acanthocardia echinata (Linnaeus, 1758)	0.75	0.27	50.00	2.64	0.32	78.57	0.06	0.05	5.56
Acanthocardia sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.15	16.67
Aequipecten opercularis (Linnaeus, 1758)	0.50	0.18	25.00	0.71	0.09	35.71	0.06	0.05	5.56
Anadara diluvii (Lamarck, 1805)	0.00	0.00	0.00	0.07	0.01	7.14	0.00	0.00	0.00
Anodontia fragilis (Philippi, 1836)	0.00	0.00	0.00	0.57	0.07	35.71	0.00	0.00	0.00
Arca noae Linnaeus, 1758	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.05	5.56
Axinulus croulinensis (Jeffreys, 1847)	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.05	5.56
Anomia ephippium Linnaeus, 1758	0.00	0.00	0.00	0.36	0.04	7.14	0.00	0.00	0.00
Atrina fragilis (Pennant, 1777)	0.25	0.09	25.00	0.50	0.06	50.00	0.06	0.05	5.56
Azorinus chamasolen (da Costa, 1778)	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.26	27.78
Cerastoderma sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.05	5.56
Clausinella fasciata (da Costa, 1778)	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.10	5.56
Corbula gibba (Olivi, 1792)	238.25	84.41	100.00	143.00	17.33	100.00	4.78	4.40	83.33
Diplodonta brocchii (Deshayes.,1852)	0.25	0.09	25.00	0.29	0.03	28.57	0.00	0.00	0.00
Ensis ensis (Linnaeus, 1758)	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.15	16.67
Gari fervensis (Gmelin, 1791)	0.00	0.00	0.00	0.00	0.00	0.00	1.61	1.48	77.78
Gastrochaena dubia (Pennant, 1777)	0.00	0.00	0.00	0.07	0.01	7.14	0.00	0.00	0.00
Hiatella arctica (Linnaeus, 1767)	0.00	0.00	0.00	3.36	0.41	85.71	0.50	0.46	22.22
Hiatella rugosa (Linnaeus, 1767)	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.10	11.11
Kurtiella bidentata (Montagu, 1803)	0.00	0.00	0.00	0.21	0.03	14.29	19.56	18.02	88.89
Laevicardium oblongum (Gmelin, 1791)	4.25	1.51	50.00	7.79	0.94	100.00	0.11	0.10	5.56
Limaria hians (Gmelin, 1791)	0.25	0.09	25.00	0.29	0.03	28.57	0.22	0.20	22.22
Lucinella divaricata (Linnaeus, 1758)	0.00	0.00	0.00	0.14	0.02	14.29	0.00	0.00	0.00
Lucinoma borealis (Linnaeus, 1758)	0.25	0.09	25.00	0.00	0.00	0.00	0.17	0.15	16.67
Lyonsia norwegica (Gmelin. 1791)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Mendicula ferruginosa (Forbes, 1844)	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.20	5.56
Modiolarca subpicta (Cantraine, 1835)	0.50	0.18	50.00	127.71	15.48	92.86	0.39	0.36	27.78
Modiolula phaseolina (Philippi, 1844)	0.00	0.00	0.00	10.50	1.27	92.86	0.00	0.00	0.00
Myrtea spinifera (Montagu, 1803)	1.00	0.35	50.00	1.14	0.14	71.43	0.61	0.56	50.00
Mysia undata (Pennant, 1777)	1.00	0.35	50.00	0.57	0.07	35.71	0.72	0.67	66.67
Nucula nitidosa Winckworth, 1930	0.00	0.00	0.00	15.57	1.89	100.00	10.89	10.04	100.00
Nucula nucleus (Linnaeus, 1758)	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.31	22.22
Nucula sulcata Bronn, 1831	0.00	0.00	0.00	0.07	0.01	7.14	0.33	0.31	22.22
Nuculana illirica Carrozza, 1987	0.00	0.00	0.00	0.00	0.00	0.00	1.22	1.13	50.00
Paphia aurea (Gmelin, 1791)	0.00	0.00	0.00	0.00	0.00	0.00	0.61	0.56	61.11
Paphia rhomboides (Pennant, 1777)	0.00	0.00	0.00	0.93	0.11	57.14	2.11	1.95	83.33
Parvicardium minimum (Philippi, 1836)	0.00	0.00	0.00	0.21	0.03	14.29	0.33	0.31	16.67
Parvicardium scabrum (Philippi, 1844)	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.26	22.22
Parvicardium sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.10	11.11
Phaxas pellucidus (Pennant, 1777)	0.00	0.00	0.00	0.36	0.04	28.57	8.50	7.83	94.44

Pitar rudis (Poli, 1795)	0.75	0.27	50.00	2.43	0.29	85.71	4.44	4.10	88.89
Plagiocardium papillosum (Poli, 1795)	0.00	0.00	0.00	5.36	0.65	85.71	0.33	0.31	22.22
Pododesmus patelliformis (Linnaeus, 1761)	1.50	0.53	25.00	0.79	0.10	28.57	0.00	0.00	0.00
Solecurtus scopula (Turton, 1822)	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.10	11.11
Spisula subtruncata (da Costa, 1778)	0.00	0.00	0.00	1.29	0.16	28.57	0.00	0.00	0.00
Tellina distorta Poli, 1791	0.00	0.00	0.00	0.00	0.00	0.00	11.89	10.96	100.00
Tellina serrata Brocchi, 1814	0.00	0.00	0.00	0.00	0.00	0.00	1.39	1.28	72.22
Tellina sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.83	0.77	16.67
Thracia papyracea (Poli, 1791)	0.25	0.09	25.00	0.07	0.01	7.14	0.11	0.10	11.11
Thracia pubescens (Pulteney, 1799)	0.00	0.00	0.00	0.29	0.03	21.43	0.06	0.05	5.56
Thracia sp.	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.20	22.22
Thyasira flexuosa (Montagu, 1803)	0.00	0.00	0.00	1.14	0.14	57.14	5.17	4.76	83.33
Timoclea ovata (Pennant, 1777)	32.50	11.51	100.00	490.93	59.51	100.00	16.67	15.36	100.00

Fig. 3.



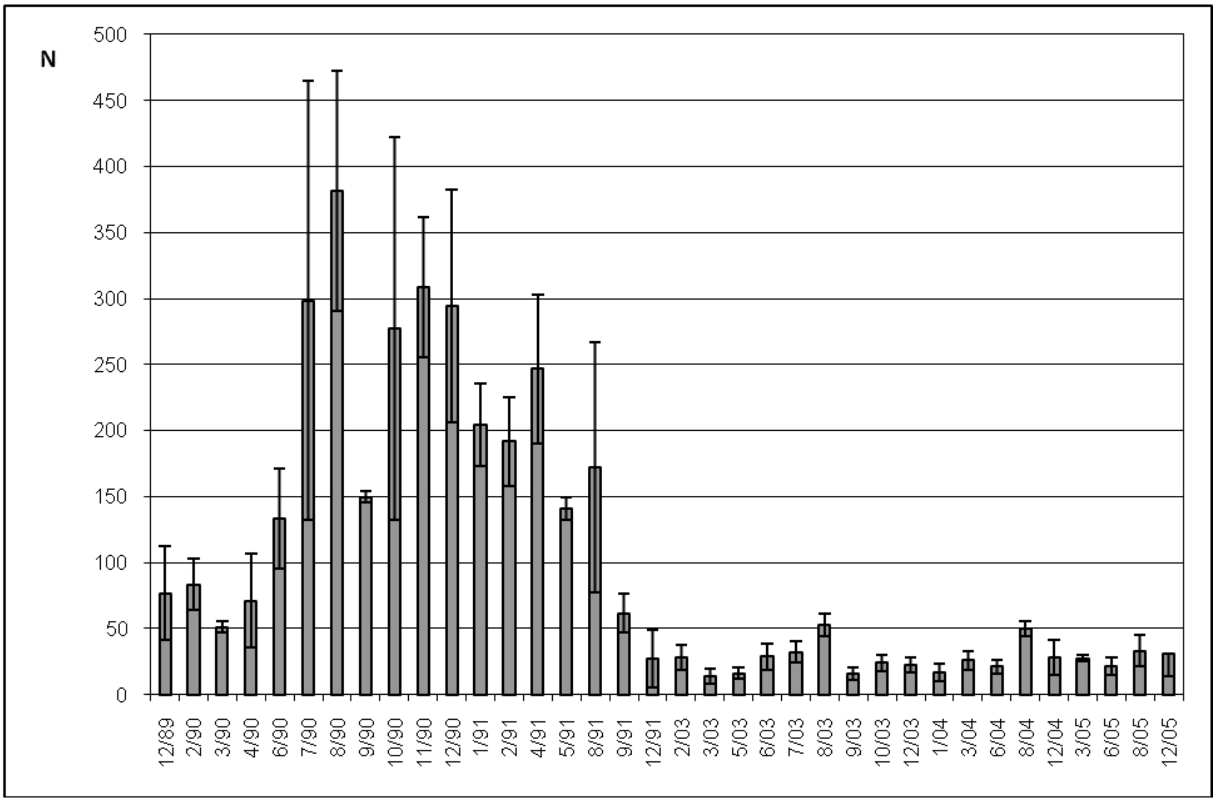

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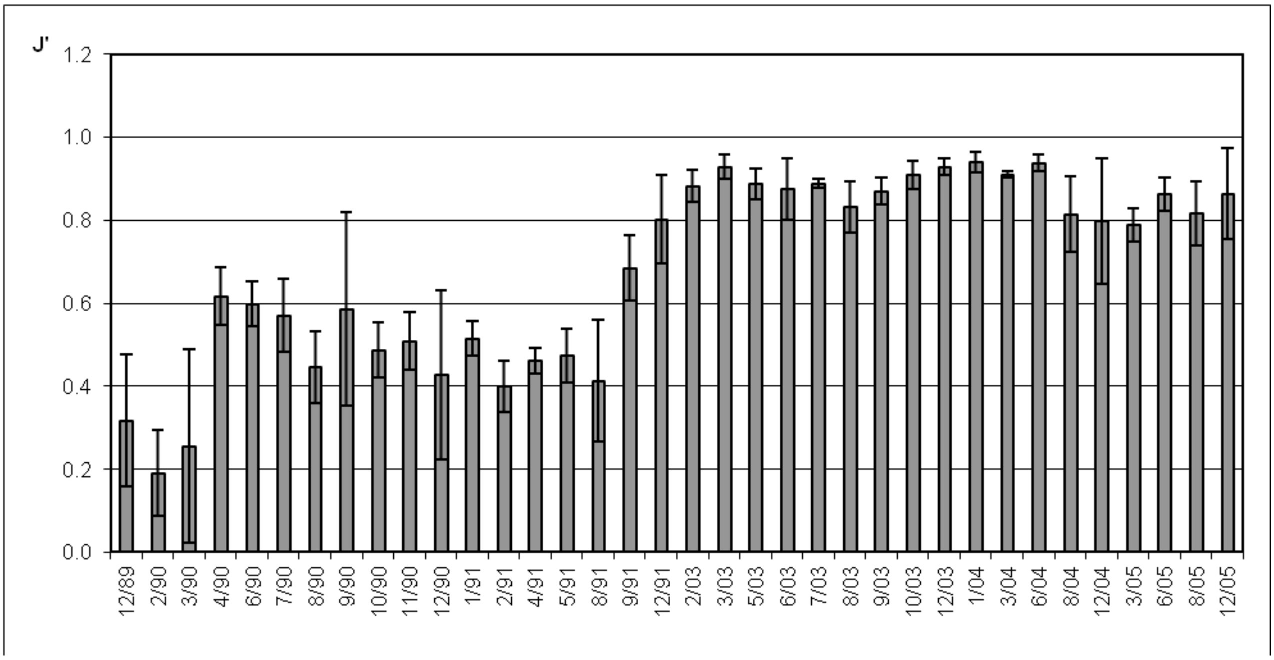
Fig. 4.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12/89
2/90
3/90
4/90
6/90
7/90
8/90
9/90
10/90
11/90
12/90
1/91
2/91
4/91
5/91
8/91
9/91
12/91
2/03
3/03
5/03
6/03
7/03
8/03
9/03
10/03
12/03
1/04
3/04
6/04
8/04
12/04
3/05
6/05
8/05
12/05

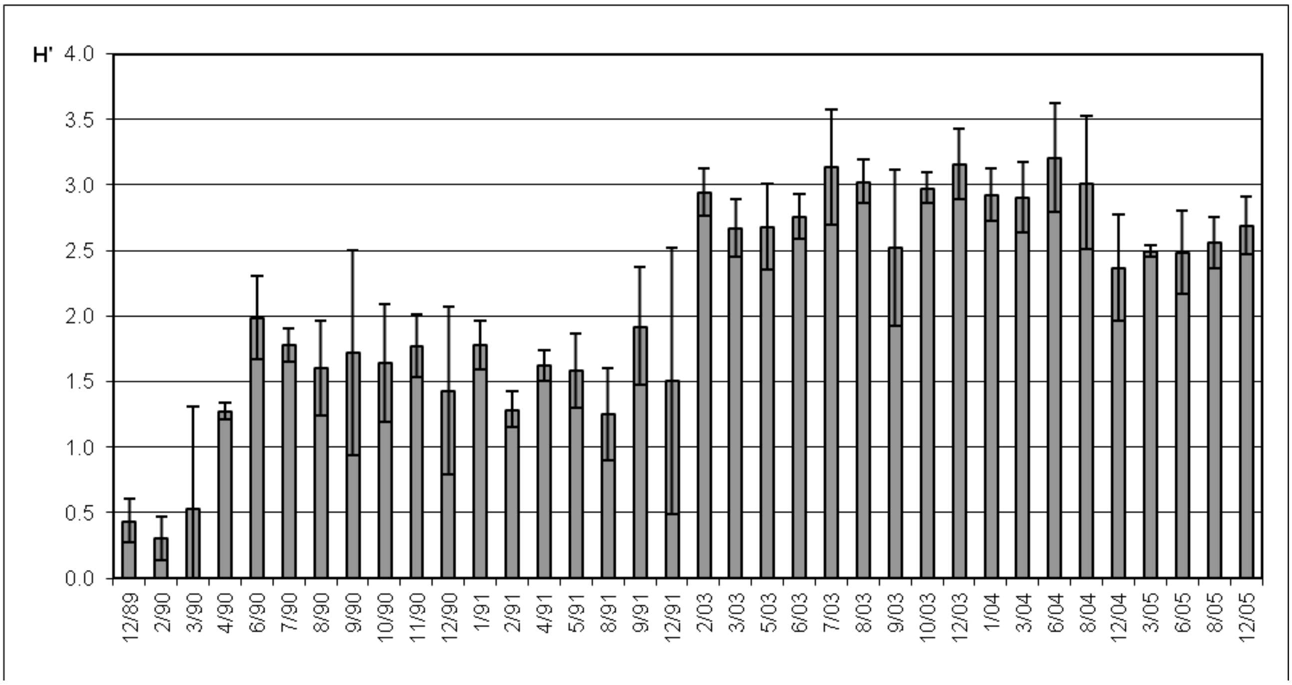
H'
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
12/89
2/90
3/90
4/90
6/90
7/90
8/90
9/90
10/90
11/90
12/90
1/91
2/91
4/91
5/91
8/91
9/91
12/91
2/03
3/03
5/03
6/03
7/03
8/03
9/03
10/03
12/03
1/04
3/04
6/04
8/04
12/04
3/05
6/05
8/05
12/05

Fig. 5.

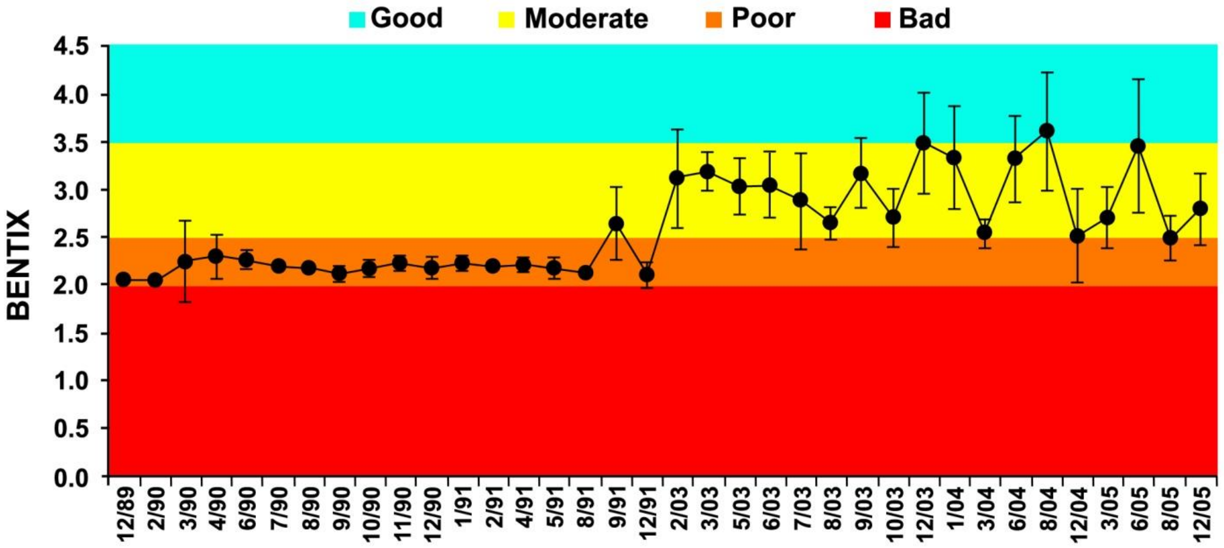
Good
Moderate
Poor
Bad
BENTIX
4.5
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
12/89
2/90
3/90
4/90
6/90
7/90
8/90
9/90
10/90
11/90
12/90
1/91
2/91
4/91
5/91
8/91
9/91
12/91
2/03
3/03
5/03
6/03
7/03
8/03
9/03
10/03
12/03
1/04
3/04
6/04
8/04
12/04
3/05
6/05
8/05
12/05

Fig. 6.

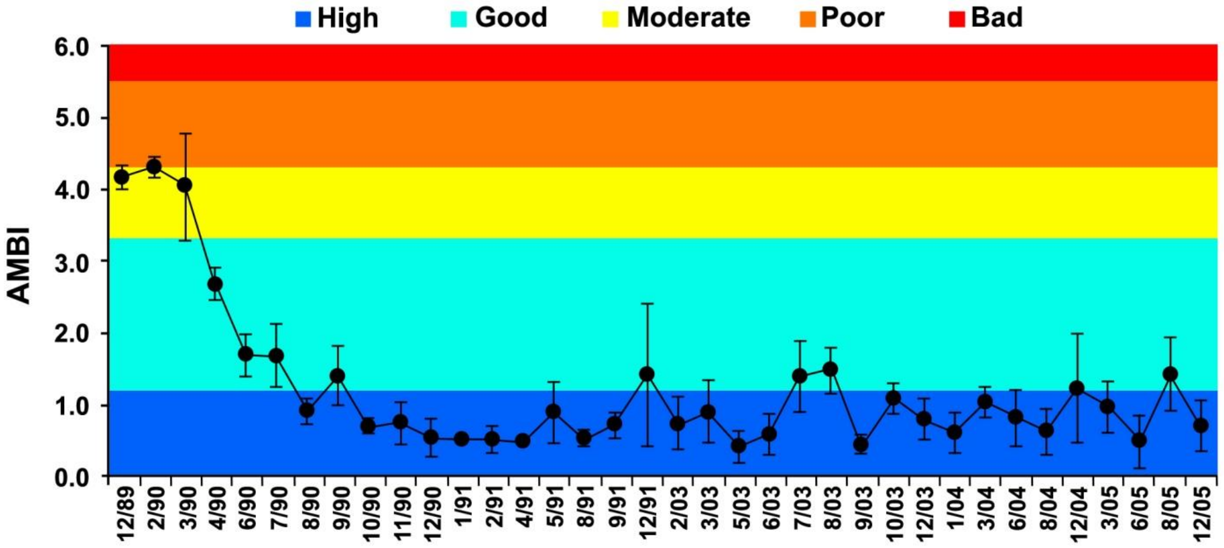
High
Good
Moderate
Poor
Bad
AMBI
6.0
5.0
4.0
3.0
2.0
1.0
0.0
12/89
2/90
3/90
4/90
6/90
7/90
8/90
9/90
10/90
11/90
12/90
1/91
2/91
4/91
5/91
8/91
9/91
12/91
2/03
3/03
5/03
6/03
7/03
8/03
9/03
10/03
12/03
1/04
3/04
6/04
8/04
12/04
3/05
6/05
8/05
12/05

Fig. 7.

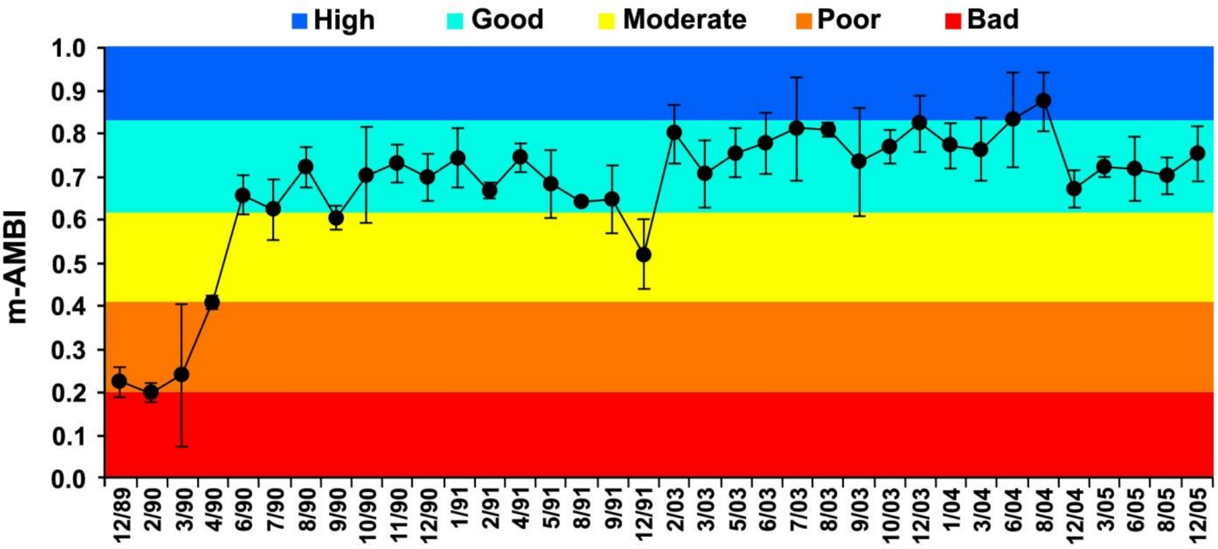

Fig. 8. a

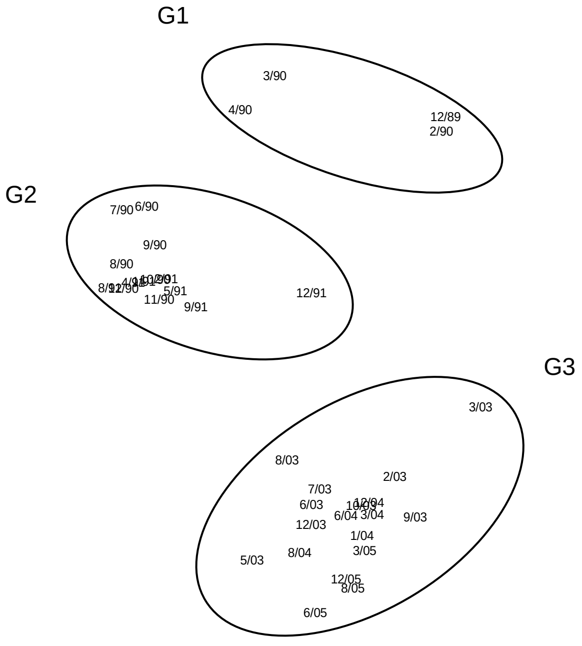

Fig. 8 b)

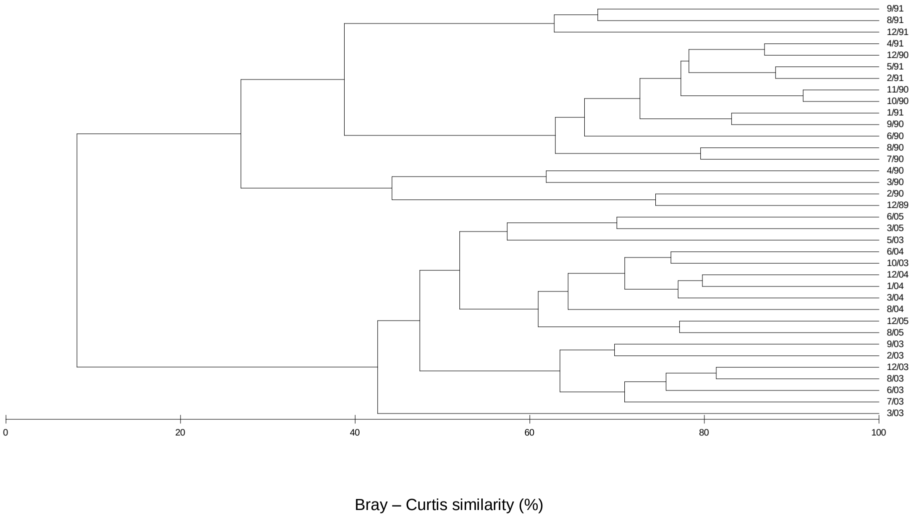

Table 2.

Groups	R-value	Significance level
1 vs 2	0.92	p< 0.001
1 vs 3	1.00*	p< 0.001
2 vs 3	0.98*	p< 0.001

Significance level = probability associated to the test

Table 3.

Similarity precentages (SIMPER)

Groups 1 and 2

Average dissimilarity =69.72

	Group 1	Group 2				
Species	Av.abund	Av.abund	Av.diss	Diss/SD	Contrib.%	Cum%
Timoclea ovata (Pennant, 1777)	32.50	490.93	38.43	2.65	55.11	55.11
Corbula gibba (Olivi, 1792)	238.25	143.00	15.00	1.02	21.51	76.63
Modiolarca subpicta (Cantraine, 1835)	0.50	127.71	9.97	1.47	14.29	90.92

Groups 1 and 3

Average dissimilarity = 92.08	Group 1	Group 3				
Species	Av.abund	Av.abund	Av.diss	Diss/SD	Contrib.%	Cum%
Corbula gibba (Olivi, 1792)	238.25	4.78	59.87	4.36	65.02	65.02
Timoclea ovata (Pennant, 1777)	32.50	16.67	7.68	0.87	8.34	73.36
Kurtiella bidentata (Montagu, 1803)	0.00	19.56	4.86	1.10	5.28	78.64
Tellina distorta Poli, 1791	0.00	11.89	3.10	2.18	3.37	82.00
Nucula nitidosa Winckworth, 1930	0.00	10.89	2.85	2.07	3.09	85.09
Phaxas pellucidus (Pennant, 1777)	0.00	8.50	2.10	0.95	2.29	87.38
Abra alba (W. Wood, 1802)	0.00	8.22	2.08	1.14	2.26	89.64
Thyasira flexuosa (Montagu, 1803)	0.00	5.17	1.29	1.20	1.40	91.05

Groups 2 and 3

Average dissimilarity = 89.05	Group 2	Group 3				
Species	Av.abund	Av.abund	Av.diss	Diss/SD	Contrib.%	Cum%
Timoclea ovata (Pennant, 1777)	490.93	16.67	47.36	3.05	53.19	53.19
Corbula gibba (Olivi, 1792)	143.00	4.78	14.77	1.86	16.59	69.78
Modiolarca subpicta (Cantraine, 1835)	127.71	0.39	11.71	1.52	13.15	82.92
Kurtiella bidentata (Montagu, 1803)	0.21	19.56	2.66	0.78	2.99	85.91
Tellina distorta Poli, 1791	0.00	11.89	1.70	1.10	1.91	87.83
Nucula nitidosa Winckworth, 1930	15.57	10.89	1.39	1.19	1.56	89.39
Phaxas pellucidus (Pennant, 1777)	0.36	8.50	1.13	0.68	1.27	90.66