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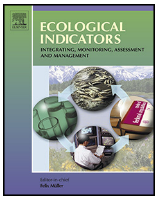

Assessment of the ecological quality (EcoQ) of the Venice lagoon using the structure and biodiversity of the meiofaunal assemblages

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ABSTRACT

Transitional Environments (TEs) have been deeply modified to meet human requirements, and for this reason are currently ranked among the most endangered aquatic ecosystems. The Adriatic basin hosts a large number of TEs of which the Lagoon of Venice is the largest one, but information on its meiofauna are very dated or focused to localized areas. The present study is the first to document the spatial distribution of meiofauna in the whole Venice lagoon. Furthermore, the health status of the TE of Venice has been assessed by means of several faunal parameters (richness, diversity indices, structure of the entire meiofaunal assemblage and only rare *taxa*). All the univariate meiofaunal parameters (i.e. richness, diversity indices, Ne:Co ratio) were consistent in highlighting the worst ecological quality of the Porto Marghera district. Instead, the structure of the entire meiofaunal assemblage as well as that of rare *taxa* seemed to detect variations not directly related to pollution and likely due to the different hydrodynamic conditions of Lido and Malamocco inlets. On the basis of our results, we have also critically discussed the usefulness of the various faunal parameters in the monitoring assessment of the TEs.

1. Introduction

Meiofauna are the most diversified component of the marine biota: as many as 24 of the 35 animal *phyla* have representatives that live in meiofauna. They play an important role in benthic food webs, not only as consumers, but also because they feed on detritus, diatoms and algae, and prey on other small metazoans (see Zeppilli et al., 2015 and references therein). Meiofauna are also the most abundant benthic group in the marine realm and their function seems to be much more complex than previously supposed, and requires further investigations to clarify their importance in the marine systems (see Balsamo et al., 2010 for review). Due to the short generation time, the high sensitivity to any environmental change and the lack of pelagic larval dispersion, meiofauna represent a promising tool for environmental monitoring assessment (Sandulli and de Nicola-Giudici, 1990; Pusceddu et al., 2007; Semprucci and Balsamo, 2012). Furthermore, meiofaunal organisms may display a rapid response to natural environmental alterations or anthropogenic pressure and can integrate information based on the analysis of the macrobenthic compartment (Balsamo et al., 2012). The assessment of the ecological quality status (EQS) of aquatic ecosystems, since the Water Framework Directive (WFD, 2000/60/EC), is one of the major objectives of applied aquatic ecology in Europe. In line with this Directive, a variety of indices and approaches for assessing the EcoQ (Ecological Quality) has been discussed, but the majority of them are focused on macrofauna (e.g. Borja et al., 2000; Simboura and Zenetos, 2002) and, only in few cases, on meiofauna (Pusceddu et al., 2007; Moreno et al., 2011; Semprucci et al., 2014, 2015a,b).

The wide range of physical and biotic conditions has made transitional environments (TEs) interesting habitats for studies of the distribution, assemblage structure and habitat preferences of many meiofaunal organisms. TEs have been deeply modified to meet human requirements and are currently ranked among the most endangered aquatic ecosystems (Airoldi and Beck, 2007). The Adriatic basin hosts a large number of TEs of which the Lagoon of Venice is the largest one. This lagoon is affected by a variety of inorganic and organic pollutants (Pusceddu et al., 2007). For instance, Venice and Mestre cities represent an important source of municipal wastewater discharges. Porto Marghera is one of the most disturbed industrial areas in Italy and Foraminifera revealed from moderate to strong impact of trace elements (see Coccioni et al., 2009 for details). Due to the shallowness of the water column, the low water exchange and high organic matter productivity, sediments of

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Venice represent the main sinks for many toxic substances. Here, dredging operations and fishing of clams often re-suspend and mix sediments leading to a redistribution of the pollutants along with both benthic and pelagic organisms (Fabbrocini et al., 2005). In addition, illegal dumping, agricultural drainage and even atmospheric deposition seem to influence the ecological quality of the area (Pusceddu et al., 2007; Coccioni et al., 2009).

Many studies on meiofauna have been carried out in Italian TEs (Colangelo and Ceccherelli, 1994; Villano and Warwick, 1995; Fiordelmondo et al., 2003; Fabbrocini et al., 2005; Pusceddu et al., 2007; Cibic et al., 2012; Frontalini et al., 2014; Semprucci et al., 2014). In particular, in the northern Italian sector, some information are available from the Po Delta lagoon (Sacca di Goro) (Colangelo and Ceccherelli, 1994), the 'Valli di Comacchio' complex (Guerrini et al., 1998), the Palude Della Rosa at Lagoon of Venice (Villano and Warwick, 1995) and the Marano lagoon (Cibic et al., 2012). However, they are generally dated and focused on circumscribed areas. Thus, the present study may offer a notable advance in the knowledge on the meiofauna inhabiting the TE systems because it documents for the first time their spatial distribution in the whole Venice lagoon. Furthermore, the health status of the TE of Venice is assessed and all the meiofaunal parameters used are critically discussed for the evaluation of their usefulness in the monitoring of the TEs.

2. Material and methods

2.1. Study area

The lagoon of Venice is the largest wetland in the Mediterranean Basin, located along the north-eastern Adriatic coast, with a surface area of ~550 km^2 and an average depth of 1.5 m (Fig. 1). The entire lagoon area is represented by land (8%), including Venice itself and many smaller islands, water (67%), and sandbanks (25%). The lagoon is connected to the Adriatic Sea by three inlets: Lido, Malamocco and Chioggia. The semidiurnal tidal cycle exchanges about 50% of the lagoon water with the sea during spring tides, and this is further reduced to 25% during neap tides (Silvestri et al., 2000). Salinity varies between 34.4–34.9‰ at high tide and 32.8–33.6‰ at low tide (Marcello, 1967; Albani and Serandrei Barbero, 1982). The water dynamics have relevant effects at the inlets and within the main channels and poor close to the mainland. Natural and artificial channels of varying depths, salt marshes, mud flats and small estuaries determine the complex morphology and hydrodynamics of the lagoon (Coccioni et al., 2009). The sediments of the lagoon are primarily composed of clayey silts in the tidal flats, and sands to silty sands in the main channels, and close to the entrances of the inlets (Albani et al., 1991; Basu and Molinari, 1994). Albani et al. (1995) suggested a very limited mobility of bottom sediment within the lagoon. The contamination of the lagoon waters and sediments began in about 1920 when the first industrial district of Porto Marghera was built that was one of the most important industrial areas in Italy until the 1970s (Apitz et al., 2007). Despite the closure of many factories, the overall pollution impact from Porto Marghera is considerable and from moderate to strong levels of trace elements (Hg, Zn, Pb and Cu) were still detectable (see Coccioni et al., 2009 for review).

2.2. Sampling routine

Meiofaunal assemblages were studied at the lagoon of Venice during summer 2004 (from 20 July to 9 September 2004). Sediment samples were taken at 21 sites. They were sub-divided in five main zones for their different level of anthropogenic impact: Zone 1 (Sts. 1, 2, 3, 4, 5 and V50), Zone 2 (Sts. 9, 10, 11 and 13), Zone 3 (Sts. 23, 26, 27, 32 and 92), Zone 4 (Sts. 52 and 54), and Zone 5 (Sts. 25, 25B, 72 and 78) (Fig. 1). In detail, Zone 1: Unpolluted, but with a Poor Water Exchange (UPWE); Zone 2: Polluted, Airport surrounding (PA); Zone 3: Polluted, industrial district Marghera (PM); Zones 4 and 5: Unpolluted and with a Good Water Exchange (UGWE).

At each site, sediments were collected by means of a box-corer (40 cm × 40 cm width and 20 cm in height), sub-sampled with Plexiglas corers (diameter: 26 mm; height: 50 mm), and preserved in buffered (Borax) formalin (4% formaldehyde) in filtered tap water (Danovaro et al., 2004).

2.3. Meiofaunal analyses

For meiofaunal extraction, sediment samples were sieved through a 500 μm mesh, and a 45 μm mesh was used to retain the smallest organisms (Semprucci et al., 2013a). The fraction remaining on the latter sieve was re-suspended in water, followed by settlement in Ludox AM (McIntyre and Warwick, 1984). Meiofauna were counted and classified to higher *taxon* under stereomicroscope, after staining with Rose Bengal (0.5 $g\,l^{-1}$) (Danovaro et al., 2004). The density (n. of individuals 10 cm^{-2}), *taxon* richness, Shannon-diversity (Shannon and Weaver, 1949) and Pielou-evenness (Pielou, 1969) (both $\log_2$) of the assemblages were then calculated. The rare *taxa* were defined as the *taxa* that represented <1% of the total abundance of all investigated samples (Bianchelli et al., 2010). As suggested by Bianchelli et al. (2010), the general dominance of nematodes and copepods in the meiobenthic assemblages may mask changes in the relative contributions of other *taxa*. When statistical analysis is restricted to rare meiofaunal *taxa*, the differences tested between the habitats may be more evident. EQS was assessed using the number (richness) of meiofaunal *taxa* as a factor (Danovaro et al., 2004, modified according to WFD classes). In order to evaluate the possible effects of the human impact on the meiofaunal assemblage, the total number of nematode and copepod individuals were computed in the Ne:Co ratio that was also analysed according to Raffaelli and Mason (1981). The hypothesis was that the divergent auto-ecological characteristics of the two groups (the extreme tolerance of nematodes and the high sensitivity of copepods) might detect the occurrence of pollution.

2.4. Statistical analysis

Statistical analyses were performed using SPSS Statistics v. 21 and PRIMER v. 5 programs. Difference in mean values of the univariate measures was tested by one-way ANOVA with Tukey's comparison test ($p < 0.05$). Prior to analysis, the normality and homoscedasticity assumptions were checked using the Kolmogorov–Smirnov and Levene's tests, respectively. When required, the data were $\log(1+x)$ transformed.

The multivariate relationships between the entire meiofaunal assemblages and rare *taxa* were analysed by non-metric multidimensional scaling (nMDS) using the Bray–Curtis similarity measure (fourth root-transformed data). A SIMPER test (cut-off of 90%) was used to determine the contribution of each *taxon* to the total dissimilarity (Clarke and Gorley, 2001; Clarke and Warwick, 2001).

3. Results

All examined samples were composed of silty muddy sediments, on average 40% of clay and 60% of silt.

Total meiofaunal abundance ranged from 77.4 ind. 10 cm^{-2} (Zone 3 at St. 92) to 3083.9 ind. 10 cm^{-2} (Zone 1 at St. V50). The Zones 2 and 1 displayed the highest abundance values, while the Zone 5 the lowest ones (Table 1).

Meiofaunal assemblages appeared well represented, with a total of 12 *taxa*: platyhelminthes, nematodes, kinorhynchs, rotifers,