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Effects of mariculture on macrobenthic assemblages in a western mediterranean site

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ABSTRACT

The effects of solid organic wastes from a marine fish farm on sediment was tested using macrobenthic fauna as biological indicators. Impact on benthic fauna was evaluated in the vicinity of a fish farm in the Tyrrhenian Sea (Western Mediterranean) between July 2001 and October 2002. Changes in benthic community structure were investigated using multivariate, distributional and univariate analyses (diversity indices, AMBI and M-AMBI). The results showed sharp disturbance of assemblages under the cages and no effects in the area more than 25 m from the cages. Sediment alterations were related to an increase in farmed biomass and its wastes, as well as to low current speed that allowed accumulation of organic matter on the sea floor. It was possible to follow the ecological succession from slightly altered assemblages to heavily polluted ones in the very short period of a single fish fattening cycle (15 months).

1. Introduction

Aquaculture is developing all over the world and in Italy has developed rapidly in the last 20 years, reaching 0.4% of total world fish production (FAO, 2007). This expansion is due to an increase in world demand for fish products impossible to meet by fishing. The demand for fish in the European Union exceeds production by European fisheries and aquaculture (Zenetos et al., 2003). This has lead to the introduction of new technologies for rearing fish in coastal waters, new feed compositions, and research into the biology of farmed species (Read and Fernandes, 2003). The increment in reared fish has had environmental impacts such as organic enrichment, chemical pollution associated with medication and antifouling products and the genetic impacts of escapees (Pearson and Black, 2001). Organic enrichment is due to wastes in dissolved and particulate form: most of the carbon fed to fish is converted into product but a considerable amount of uneaten food and faeces settles out as sediment (Hall et al., 1990; Karakassis et al., 2000; Klaoudatos et al., 2006). This sediment provides a substrate for bacterial growth, creating polluted zones characterised by low oxygen values, mainly in sediment, high concentrations of nutrients in the water column (organic nitrogen, organic carbon and phosphorus) and in sediment (organic nitrogen, organic carbon, phosphorus and sulphur) (Brown et al., 1987). Anaerobic mineralization of organic matter in sediment causes release of toxic compounds such as ammonia, sulphides and methane (Thamdrup and Canfield, 2000).

There has been much research into the impacts of organic enrichment (Gowen and Bradbury, 1987; Holmer, 1991; Wu et al., 1994; Tsutsumi, 1995; Karakassis et al., 1998; Yokoyama, 2002; Carroll et al., 2003). All studies reported modification of the biochemistry of the sea floor but the degree of modification depended on species reared, feed used, water depth, hydrodynamic features and sea floor type (sand, mud ...); in other words on the carrying capacity of the area (Karakassis et al., 1998; Pearson and Black, 2001). In Mediterranean fish farms internal and local impacts were detected but no regional ones (Karakassis et al., 1998; Klaoudatos et al., 2006; Maldonado et al., 2005; Aguado-Gimenez et al., 2007).

Soft bottom benthic assemblages can be affected by environmental perturbations and there is extensive literature on the relationship between them and anthropogenic sediment changes (Pearson and Rosemberg, 1978; Gray, 1981; Warwick and Clarke, 1991; Simboura et al., 1995; Morrisey et al., 2000). The macrofauna living in fish farm sediment eventually shows a prevalence of small and tolerant species (Brown et al., 1987; Karakassis et al., 1999; Brooks and Mahnken, 2003; Macleod et al., 2004; Heilskov et al., 2006), in line with the empirical succession model of Pearson and Rosemberg (1978) which predicts reduced mean body size, shallower distribution of fauna in sediment and impoverished functional community structure with increasing organic load (Heilskov et al., 2006). Otherwise a moderate, even continuous, loading of organic matter increases benthic fauna diversity or abundance (Nickell et al., 2003). Some invertebrates of benthic assemblages are useful indicators of environmental alterations in soft bottom monitoring programmes. By virtue of their large numbers, diversity of feeding habits and ecophysiological adaptation strategy, polychaetes have been shown to be good indicators and are used in coastal studies for monitoring environmental disturbance (Pocklington and Wells, 1992; Solis-Weiss et al., 2004; Tomassetti and Porrello, 2005; Giangrande et al., 2005). Other invertebrates useful for monitoring studies due to their ecological

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importance, sedentary life style and ease of collection include molluscs, mainly bivalves and gastropods (Feldstein et al., 2003).

In some Italian marine fish farms the impact on sediment was evaluated using various indicators: bacteria and meiofauna (Mazzola et al., 1999; Mazzola et al., 2000; Mirto et al., 2000; Mirto et al., 2002; La Rosa et al., 2001; La Rosa et al., 2004), chemical parameters (Ceccarelli and Di Bitetto, 1996; Santulli et al., 2003; La Rosa et al., 2004; Porrello et al., 2005) and stable indicators such as carbon and nitrogen isotopes (Sarà et al., 2004). Other studies investigated the impact of an Italian fish farm on a bottlenose dolphin population (Díaz López et al., 2005), degradation of *Posidonia oceanica* meadows caused by mariculture (Cancemi et al., 2003) and the effects of tuna farming on macrobenthic fauna (Santulli et al., 2003). There have been few studies on the environmental impact of Italian marine fish farms on benthic assemblages.

Here we investigated the effects of a marine fish farm on macrobenthic assemblages. The aims were to evaluate spatial and temporal changes in macrobenthic assemblages during a complete fattening cycle and identify the best ecological index of structural changes; we also studied relationships between certain chemical indicators and macrobenthic assemblages.

2. Materials and methods

The fish farm was 2500 m east of Porto Ercole (Tuscany, Italy) in the Tyrrhenian Sea (Mediterranean Sea) (Fig. 1). It consisted of two submersible cages (G1 and G2) with a capacity of 2000 m^3 each, producing 180 tons $year^{-1}$ of European seabass *Dicentrarchus labrax* (Linnaeus, 1758) and meagre *Argyrosomus regius* (Asso, 1801). The fish were fed with industrial feed (46–52% protein, 16–22% fat, 8.5–9% ash, 0.6–1.9% fibre, 1.2–1.5% total phosphorus). The daily rations of up to 5 meals per day were 0–1.7% of the total biomass reared. During the study period the biomass in the cages ranged from zero (July 2001) to 43 tons (October 2002) in G1 (seabass) and from 12 (December 2001) to 30 tons (May 2002) in G2 (meagre).

The location of the fish farm offshore can be classified as CW-M3 (Coastal Water-Sedimentary Shallow) according EU Water Framework Directive 2000/60; depth ranged from 30 to 32 m; salinity was 38 PTU; tide range was a maximum of 20 cm; sea floor was mainly muddy with 80–90% silt-clay component; the dominant sea current direction was E-NE/W-SW with a mean velocity of 5.8 cm/s at −27 m (3.6 SD), 6.8 cm/s (4.4 SD) at −14 m and 8.8 cm/s (5.8 SD) at −3 m. More detailed hydrodynamic and biogeochemical features of the site are described in Porrello et al. (2005).

Sampling was carried out in July 2001, December 2001, May 2002 and October 2002 according to rearing cycle. Eleven stations were fixed: one beneath each cage and the other nine around them to a maximum distance of 150 m (Fig. 1).

Sediment was collected around the cages with a Van Veen grab (0.1 m^2, 20 l), equipped with screen doors; at the stations beneath the cages sediment was collected by divers with sampling boxes (35 × 35 × 20 cm, 24 l) built to obtain a similar quantity of sediment to the Van Veen grab. Sediments were washed through a 1.0 mm mesh screen. Residues were fixed in 10% buffered formalin. In the laboratory, biological samples were preserved in 70% ethyl alcohol, identified to the lowest possible taxon and counted. The biological quantitative data obtained were used to calculate several indices: Shannon–Wiener diversity (Shannon and Weaver, 1949); AMBI and M-AMBI. The AMBI index (Borja et al., 2000) was derived using the AMBI software (http://www.azti.es), after following guidelines from Borja and Muxika (2005). M-AMBI (Borja et al., 2004; Muxika et al., 2007) was calculated using the same software, taking the control stations values of diversity, AMBI and richness as reference conditions. This method has been intercalibrated among different Member States in Europe (Borja et al., 2007) and compared with other methods elsewhere (Borja et al., 2008). Affinities between stations based on macrofauna data were established using multivariate methods such as non-metric multidimensional scaling (nMDS) analysis with Primer 6.0 software (Software package from Plymouth Marine Laboratory, UK; Clarke and Warwick, 2001). MDS analysis was applied to a dissimilarity matrix calculated from the species–sample data matrix. To test the ordination, the Kruskal stress coefficient was calculated (Kruskal and Wish, 1978). The Bray–Curtis coefficient (Bray and Curtis, 1957) was used to calculate dissimilarity matrix relationships between species.

Taxonomic diversity index, delta, were calculated from macrofauna abundance, using Primer 6 software. Simper analysis, performed with the Primer 6.0 software, was carried out to assess the role and importance of single species, or groups of species for separation of stations.

To test the relationship between geochemical and biological variables the Spearman rank correlation was calculated.

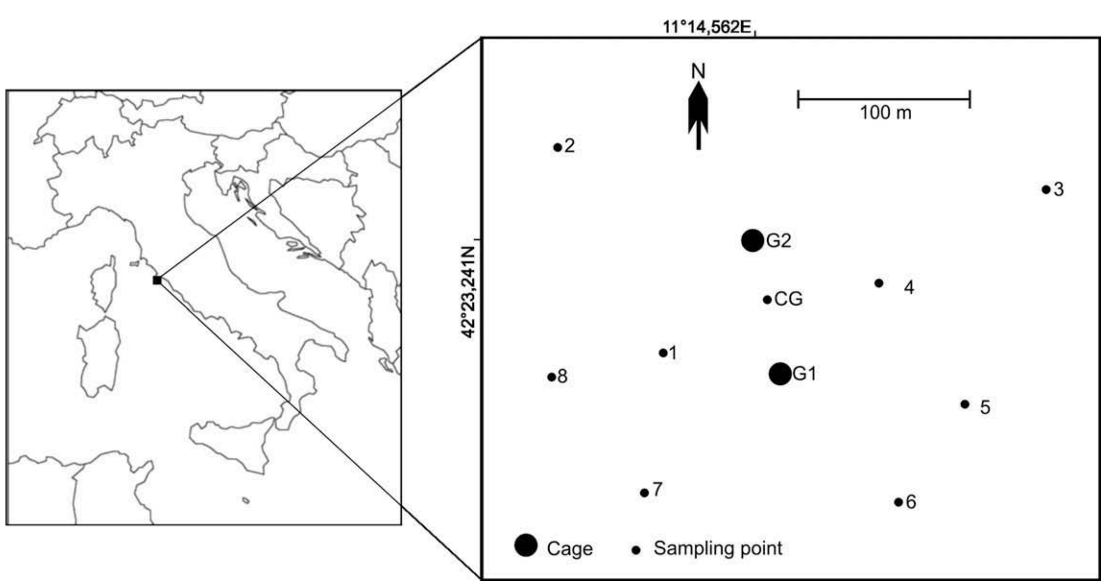

Fig. 1. Locations of marine fish farm and sampling stations.