



Response of different benthic habitats to off-shore fish cages

Rodrigo Riera^{1,*}, Mikel A. Becerro², Eva Ramos¹, Óscar Monterroso¹ & Myriam Rodríguez¹

¹CIMA SL, Canary Islands, Spain

²The bites Lab Natural Products and Agrobiological Institute (IPNA-CSIC), Canary Islands, Spain

Correspondence: R Riera, CIMA SL, Arzobispo Elias Yanes, 44, 38206 La Laguna, Tenerife, Canary Islands, Spain.

E-mail: rodrigo@cimacanarias.com

***Present address:** Department of Biodiversity, Qatar Environment and Energy Research Institute (QEERI), Doha 5825, Qatar.

Abstract

Off-shore fish farming can increase the organic load of nearby coastal marine ecosystems due to the deposition of fish food and faeces on seabeds. Seagrass meadows are particularly affected by aquaculture activities but there are few empirical data showing differential effects of the same farming activity on multiple habitat types. Here, we assessed over a 2-year period whether macrofaunal assemblages inhabiting sandy bare seabeds and *Cymodocea nodosa* meadows varied in their macrofaunal community structure to the fish farming activities. We observed high spatial and temporal variability in macrofauna composition and dynamics among seabed habitats and a limited impact of fish cages in their area of influence as compared with control areas. Seagrass meadows showed a higher abundance in macrofauna communities than sandy bare bottoms. Local marine currents could partially explain some results because of their influence on grain size composition. Differences in grain size resulted in higher abundances of the tanaid *Apseudes talpa* beneath fish cages and the absence of the sensitive amphipod *Ampelisca brevicornis*. Differences of resilience of seabeds (seagrass meadows and sandy bare bottoms) should be taken into account for environmental monitoring studies of off-shore fish cages. Our results suggest that hydrodynamics are a key factor to determine buffer areas between fish cages and seagrass meadows.

Keywords: Fish cages, buffer area, seagrass, sandy seabeds, macrofauna, Canary Islands, Atlantic Ocean

Introduction

Aquaculture is steadily supporting the demand for fish products as wild fisheries stocks decline worldwide (Zenetos, Streftaris & Larsen 2003; FAO '2007). Under this perspective, successful aquaculture could therefore be used as a tool to preserve natural systems, facilitating recovery of natural populations and conserving endangered species (Anders 1998; Ireland, Anders & Siple 2002). However, the negative consequences that aquaculture may have on natural systems are an increasing cause of environmental concern (Holmer 2010). Aquaculture techniques may change the physico-chemical environment leading to changes in macrofauna density, species richness and abundance of opportunistic organisms (Machias, Karakassis, Labropoulou, Somarakis, Papadopoulou & Papaconstantinou 2004; Edgar, Davey & Shepherd 2010). These shifts increase the risk for degradation of sensitive marine habitats such as rocky reefs, shallow macroalgae and maerl beds, or seagrass meadows (Holmer, Duarte, Heilskov, Olesen & Terrados 2003; Wilson, Blake, Berges & Maggs 2004). Worldwide efforts are underway to develop more suitable farming techniques (Troell, Halling, Neori, Chopin, Buschmann, Kautsky & Yariah 2003).

Increasing our understanding of the negative consequences of aquaculture in natural systems is critical to implement successful farming techniques (Borja, Rodríguez, Black, Bodoy, Emblow, Fernandes, Forte, Karakassis, Muxika, Nickell, Papageorgiou, Pranovi, Sevastou, Tomassetti & Angel 2009). Farm managers are interested in environmental impact as the decrease of dissolved oxygen

and hydrogen sulphide production in sediments beneath fish cages can affect the health of cultured stocks and associated husbandry practices (Black, Kiemer & Ezzi 1996). Light reduction and deposition of particulate carbon released can have profound effects on the water column (Sarà 2007) and sediment quality (Pusceddu, Frascchetti, Mirto, Holmer & Danovaro 2007), which may lead to major environmental alterations that can be particularly intense in sensitive communities such as seagrass meadows (Ruiz, Perez & Romero 2001). A number of conservation measures have been recently proposed to minimize the negative effects of aquaculture in seagrass meadows (Holmer, Black, Duarte, Marbá & Karakassis 2008). These factors include minimum distance from cages, farm-free areas or conservation of well-preserved seagrass meadows by protection figures (Pergent-Martini, Boudouresque, Pasqualini & Pergent 2006).

Data on the consequences of farming on seagrass beds are restricted to a few species. The environmental impacts of fish cages on *Posidonia oceanica* meadows has been extensively recorded in the Mediterranean Sea (Holmer *et al.* 2008), with high mortalities beneath fish cages and progressive regression on surrounding areas even after fish farming has ceased (Delgado, Ruiz, Perez, Romero & Ballesteros 1999). Other seagrasses (e.g. *Zostera marina*) have been largely affected by mussels and oyster cultures (Everett, Ruiz & Carlton 1995; Nuckles, Short, Barker & Kopp 2005). Since seagrass resistance to environmental stressors vary as a function of species (Hemminga & Duarte 2000), broad generalizations on the consequences of fish farming on non-investigated species should be made with caution. Lack of data on seagrass species offer little guarantee to maximize their protection and filling this gap should be an ecological, environmental and political priority.

Cymodocea nodosa is the most abundant seagrass species in the Canary Islands, where it can form extensive meadows on the eastern and southern coasts of the islands (Pavon-Salas, Herrera, Hernandez-Guerra & Haroun 2000). In the Canary archipelago, *C. nodosa* meadows constitute the main primary benthic producer due to the proliferation of *Diadema africanum* populations that have converted many algal dominated communities into sea-barren seabeds (Tuya, Boyra, Sanchez-Jerez, Haroun & Barbera 2004). Meadows of *C. nodosa* also harbour a remarkable infaunal biodiversity

(Brito, Núñez & San Martín 2005) that can be used as bioindicators of the environmental health of coastal areas in the Canary Islands (Riera, Monterroso, Rodríguez & Ramos 2011a; Riera, Monterroso, Rodríguez, Ramos & Sacramento 2011b; Riera, Tuya, Sacramento, Ramos, Rodríguez & Monterroso 2011c). Meadows of *C. nodosa* are legislated as 'species of interest for canarian ecosystems' (BOC 2010/112, Catálogo Canario de Especies Protegidas), i.e., *C. nodosa* is only protected within marine protected areas (MPAs). Outside MPAs there could be a conflict of interests between fish farms and *C. nodosa* meadows since both share the same environmental requirements to succeed (e.g. good water quality and water renewal). The effect of fish farming on *C. nodosa* is unknown but the diverse macrofauna present in these meadows could be used to determine the impact of farming activities on this seagrass.

Our study provides information on the effect of fish farming on seagrass meadows of *C. nodosa* and on unvegetated sandy bottoms. We will look at multiple biotic and abiotic variables to determine whether the impact of fish farming on the two soft-bottom communities differs. Specifically, we addressed the following null hypotheses: i) No differential response of fish cages occur on macroinfaunal assemblages from sandy bare bottoms and *Cymodocea nodosa* meadows; ii) no differential response occur on infaunal assemblages between wind- and leeward conditions.

Material and methods

Study site

The present study was conducted in a fish cage farm located at Barranco Hondo (NE Tenerife, coordinates 28°22'53"N/16°21'09"W) that cultures seabream (*Sparus aurata*) and European seabass (*Dicentrarchus labrax*) (Fig. 1). The fish farm consisted of one group of four fish cages with an annual production of ca. 200 T. The fish lease started to operate in June 2007, with 100% effectiveness 6 months later (December 2007). Thus, we consider 'after conditions' since December 2007. Fish were fed manually using commercial, pelleted and extruded diets with an average nutrient content (on a dry weight basis): 48% protein, 19.5% fat, 22% carbohydrates, 7.9% nitrogen and 1.08% phosphorus. Pellet size ranged from 2 to 7 mm diameter and fish were fed at an average of

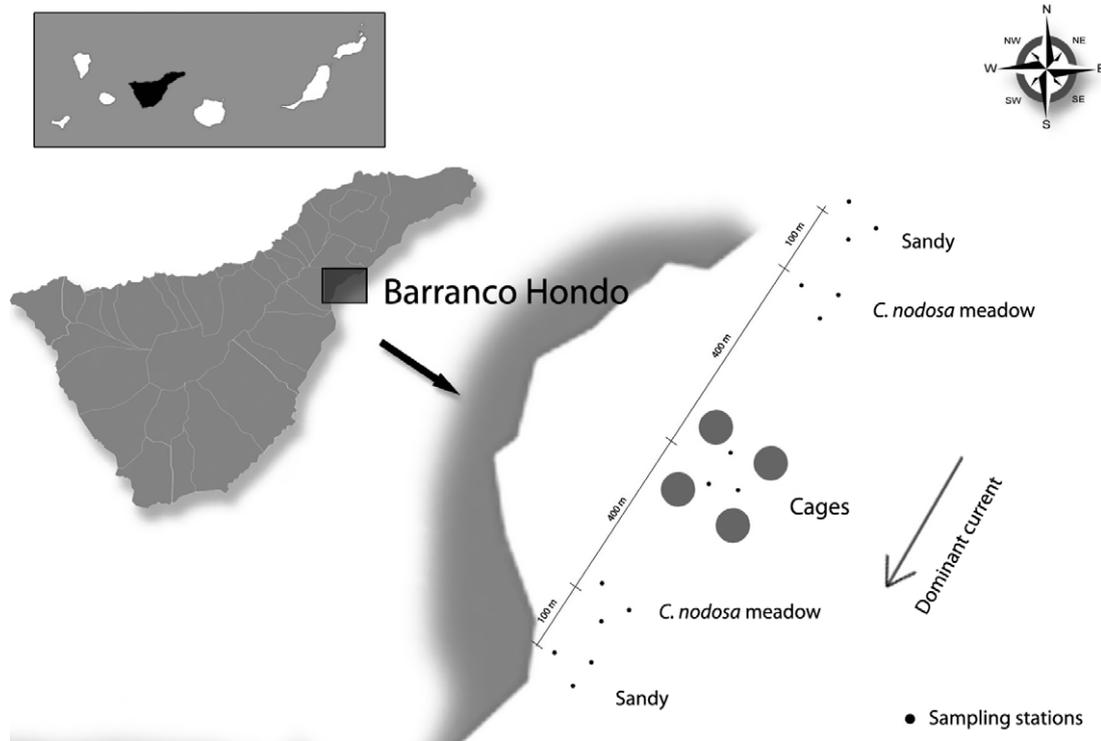


Figure 1 Map of the study area. North locations on the map correspond to windward locations and South locations to leeward locations.

ca. 2% (biomass/day), with a food conversion ratio of 1.5–1.6. Each fish cage had a diameter of 20 m and a depth of 10 m. Fish cages are installed on seabeds at 28–30 m depth and its temperature throughout the year ranged from 18 (January) to 22°C (September). Continuous currents throughout the year are present in the study area, with predominant Southwest (36.4%) and Northwest (27.8%) directions, and a mean value of 12–14 cm s^{-1} on the surface and 20–22 cm s^{-1} on the bottom layer (O. Perez, pers. comm.). The benthic habitat mainly comprised by sandy bare seabeds and *Cymodocea nodosa* meadows were located on the surroundings of the studied fish cages (350–400 m). No subtidal rocky substrates are present in the study area.

Sampling strategy

Samples were collected during 2 years (2007 and 2008) throughout six sampling campaigns (April 2007, November 2007, March 2008, June 2008, September 2008, and December 2008). We selected five sampling areas that corresponded to two habitat types (Fig. 1). The sandy bare habitat

type had an impact area located at a depth of 28 m directly beneath the fish cages ('cage'), an up current sandy bare seabed area ('sand 1') located at 1 km from the fish farm and at 20 m depth, and a down current sandy bare seabed area located at 750 m from the fish farm and at 20 m depth ('sand 2'). Two *Cymodocea nodosa* meadows were also located up ('meadow 1', 19 m depth) and down ('meadow 2', 12 m depth) the dominant underwater current at 400 m and 500 m respectively (Fig 1). The studied *C. nodosa* meadows are extensive patches (>1 km), characterized by intermediate densities (250–350 ind m^{-2}), typical of seagrass meadows from Tenerife (pers. obs.).

In each sampling area, three random stations were selected. In each station, three replicates were collected for macrofaunal quantification and identification and one additional core for the analysis of sedimentary factors (grain size, organic matter, phosphorus and nitrogen). Sediment cores (20 cm inner diameter, area: 314 cm^2) were pushed into the sediment to a depth of 25 cm. The whole sediment layer was analysed for both analyses (macrofauna and abiotic variables).

To quantify grain size composition, 100 g sediment from each sampling station was oven dried at 105°C, passed through a graded series of sieves (2 mm, 1 mm, 0.5 mm, 0.25 mm, 0.125 mm and 0.063 mm), and then weighed (Buchanan 1984). These sieves characterized seven sedimentary types that are widely used in grain-size studies (Riera, Tuya, Ramos, Rodríguez & Monterroso 2012). The method of Walkley and Black (1934) was used to determine the organic matter content in the sediment. This method is suitable for determining low concentrations of organic carbon in the water and sediments (Sawyer, McCarty & Parkin 2003). Total nitrogen (%) was determined following the Kjeldahl method (Bradstreet 1965). Total phosphorus was determined as orthophosphate by the molybdate-ascorbic acid method (Andersen 1976).

Samples were preserved in 10% seawater formaldehyde solution and sieved through a 0.5 mm sieve, systematically used in subtidal sedimentary samples in the Canary archipelago (Riera *et al.* 2011a). The supernatant was separated into high taxonomic groups and preserved in 70% ethanol. All individuals were counted and determined to the lowest possible taxonomic level by means of a binocular microscope and an optical microscope equipped with Nomarski interference contrast (Nikon Elipse-8).

Statistical analysis

Several statistical methods available in PRIMER 6 software (Plymouth, UK) (Clarke & Warwick 2001) and SYSTAT 12 (SPSS 1999) were used to analyse macrofaunal and sedimentary variables as a function of habitat ('seagrass' versus 'sandy bare bottom', excluding fish cage stations) and hydrodynamics ('windward' versus 'leeward', excluding fish cage stations). Variations remained homogeneous in sedimentary data, satisfying anova assumptions. Thus, no parametric tests (e.g. *t*-test) were needed.

We calculated Bray-Curtis similarity on square-root transformed macrofaunal abundance and used permutational multivariate analyses of variance (PERMANOVA) (Anderson 2005) to test for differences in macrofaunal assemblage structure across former factors.

Because of the large number of grain size fractions (gravels, very coarse sands, coarse sands, medium sands, fine sands, very fine sands and silt/clay) quantified in this study, we used factor analysis (FA) to look for coherent grain size groups

of variables that were correlated with one another within groups but largely independent between groups (Tabachnick & Fidell 2001). These groups of correlated variables or factors help interpret the underlying mechanisms that have created the relationship between variables. Specifically, we used a principal component analysis extraction (PCA) with a minimum eigen value of one to estimate number of factors. To facilitate interpretation, we used varimax rotation since it minimizes the number of variables that load highly on a factor and maximizes the loading variance across factors.

Results

Environmental variables

The content of silt and clay, and very fine sands were higher in sediments beneath fish cages and *Cymodocea nodosa* meadows as compared with sandy bare seabeds. Fine sands dominated beneath fish cages throughout the study period (April 2007–December 2008). Medium sands were more abundant in sandy sediments compared with the sediments beneath fish cages and *C. nodosa* meadows. The remaining grain size fractions were minimal, as well as, organic matter, total phosphorus and total nitrogen content at all sampling sites throughout the study period (Table 1).

Hydrodynamic conditions affected consistently the sedimentary composition of sampling stations depending on their location, showing significant differences between windward and leeward stations in fine-grained sedimentary fractions, silt and clay ($F = 12.232$, $P = 0.002$), very fine sands ($F = 5.101$, $P = 0.034$), fine sands ($F = 5.004$, $P = 0.036$) and medium sands ($F = 10.990$, $P = 0.003$) (Table 2). However, coarse-grained sedimentary fractions showed no consistent changes between both stations groups (wind- and leeward), i.e. coarse sands ($F = 0.746$, $P = 0.397$), very coarse sands ($F = 0.243$, $P = 0.627$) and gravels ($F = 3.395$, $P = 0.079$) (Table 2). The remaining abiotic factors not varied significantly between both groups (total phosphorus ($F = 0.647$, $P = 0.430$), organic matter ($F = 0.003$, $P = 0.958$) and total nitrogen ($F = 0.342$, $P = 0.564$)) (Table 2).

Macrofauna

A total of 18 735 individuals, belonging to 252 species, were collected along six sampling

Table 1 Sedimentary variables of sampling sites throughout the study period. 2007 and 2008 represent 'before' and 'after' conditions respectively

	Fish cage		<i>C. nodosa</i> meadow		Sandy seabed	
	2007	2008	2007	2008	2007	2008
Silt/clay (%)	14.8 ± 2.04	9.05 ± 4.83	15.25 ± 10.54	12.28 ± 5.46	6.21 ± 2.80	6.33 ± 3.41
Very fine sands (%)	24.15 ± 18.76	34.2 ± 16.64	16.95 ± 16.07	25.54 ± 10.44	6.61 ± 6.25	11.07 ± 3.54
Fine sands (%)	42.65 ± 13.51	36.51 ± 5.38	18.37 ± 6.93	25.51 ± 6.52	17.33 ± 17.99	30.73 ± 22.73
Medium sands (%)	12.35 ± 7.01	11.50 ± 7.61	14.85 ± 6.91	13.74 ± 3.58	30.33 ± 27.14	21.60 ± 8.47
Coarse sands (%)	2.8 ± 0.71	4.56 ± 3.91	13.49 ± 8.09	11.45 ± 5.05	12.48 ± 7.93	10.62 ± 7.69
Very coarse sands (%)	1.75 ± 0.21	2.24 ± 2.54	13.70 ± 10.13	9.65 ± 6.82	11.29 ± 11.82	9.09 ± 8.29
Gravels (%)	1.35 ± 0.49	1.92 ± 3.33	7.32 ± 8.92	3.55 ± 3.49	17.43 ± 19.79	10.51 ± 11.77
Total P (ppm)	4.55 ± 6.29	10.56 ± 9.68	4 ± 4.82	11.8 ± 9.58	3.9 ± 4.77	12.47 ± 11.64
OM (%)	0.71 ± 0.40	0.08 ± 0.08	0.54 ± 0.26	0.24 ± 0.20	0.37 ± 0.38	0.28 ± 0.24
Total N (ppm)	19 ± 7.07	16.37 ± 13.12	19.5 ± 6.61	17.62 ± 11.28	18 ± 2.83	13.09 ± 13.64

Table 2 One-way ANOVA of sedimentary factors considering two station groups (wind-versus leeward sites). Significant differences ($P < 0.05$) are highlighted in bold

Source	df	MS	F	P
Silt and clay	1	344.284	12.232	0.002
Very fine sands	1	568.427	5.101	0.034
Fine sands	1	1057.35	5.004	0.036
Medium sands	1	1265.85	10.990	0.003
Coarse sands	1	32.900	0.746	0.397
Very coarse sands	1	17.510	0.243	0.627
Gravels	1	403.440	3.395	0.079
Phosphorus	1	58.907	0.647	0.430
Organic matter	1	0	0.003	0.958
Nitrogen	1	38.254	0.342	0.564

campaigns (April 2007, November 2007, March 2008, June 2008, September 2008, and December 2008). Amphipods and tanaids were the most abundant taxonomic groups, with 8128 (43.38% of the overall abundance) and 7871 (42.01%) individuals respectively (Fig. 2). Species of copepods and oligochaetes were scarce and represented by single specimens.

The most abundant species were the tanaid *Apseudes talpa* (7822 ind) and the amphipod *Ampelisca brevicornis* (6710 ind) (Fig. 3). The remaining taxa were represented by less than 400 individuals and only 19 species obtained overall densities higher than 50 specimens.

The largest macrofauna densities were found in the down current *Cymodocea nodosa* meadow (mean abundance ± standard error, 348.67 ± 57.38 ind) whilst the smallest densities were obtained in down current sandy bare seabeds (60.17 ± 6.42 ind). Macrofaunal abundances in the sediments under

the fish cages were intermediate (217.39 ± 29.96 ind) and similar to the two up current locations (*C. nodosa* meadows, 245.72 ± 30.99 ind; sandy seabeds, 207.22 ± 45.33 ind).

If fish cages stations and seasonality are excluded in the statistical analysis, consistent results are found. Differences in overall macrofaunal abundance were observed between habitats (meadows versus sandy bare bottoms) ($F = 20.287$, $P = 0.0001$), however, were inconsistent between habitats with different hydrodynamic conditions (wind- versus leeward sites) ($F = 10.256$, $P = 0.002$). The most abundant species, the tanaid *Apseudes talpa* showed significant changes between wind- and leeward sites ($F = 42.24$, $P = 0.0001$), and consistent trends were observed considering habitats at different hydrodynamic conditions ($F = 0.645$, $P = 0.645$). The amphipod *Ampelisca brevicornis* showed significant differences considering separately both factors ('Habitat', $F = 71.83$, $P = 0.0001$; 'Hydrodynamics', $F = 7.199$, $P = 0.009$), as well as, the interaction between them ('Habitat' × 'Hydrodynamics', $F = 24.09$, $P = 0.0001$) (Table 3).

Macrofauna community varied significantly as a function of seabed (meadows versus sandy bare bottoms) ('Habitat', $F = 10.461$, $P = 0.001$), as well as, between windward and leeward stations ('Hydrodynamics', $F = 4.2851$, $P = 0.001$). The interaction between both variables were significant ('Habitat' × 'Hydrodynamics', $F = 4.7808$, $P = 0.001$), suggesting the influence of hydrodynamics on the macrofauna community regardless the habitat (Table 3).

In terms of species, the 17 most abundant macrofaunal taxa were used for statistical analysis

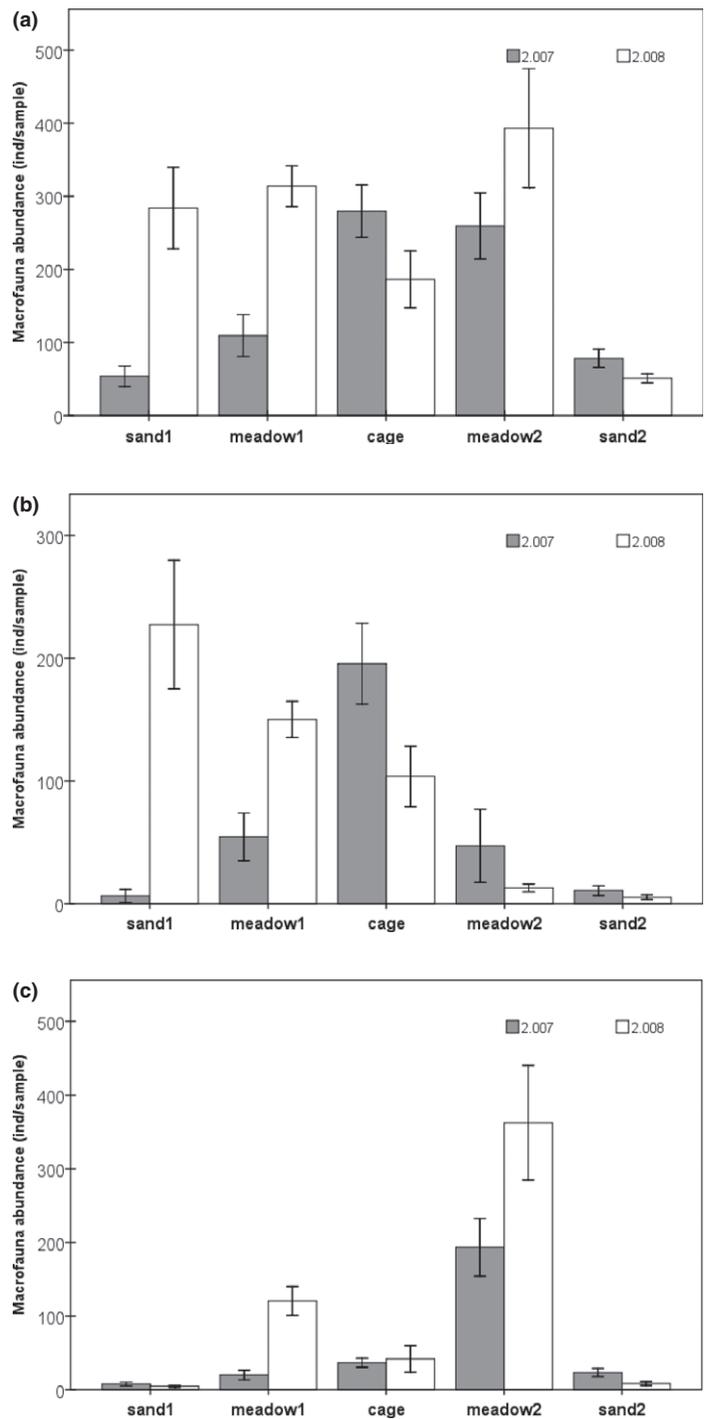


Figure 2 Macrofauna abundances at the sampling sites. (a) Overall macrofauna, (b) Tanaids, (c) Amphipods. Error bars are SE of means. 2007 and 2008 represent ‘before’ and ‘after’ conditions respectively.

and were grouped under six factor analysis (FA1 to FA6) that explained 65.6% of the total variance (Table 4). Macrofauna assemblage structure was significantly affected by sedimentary variables (Spearman’s correlation, $\rho = 0.29$, $P = 0.02$). Silt

and clay, total phosphorus and total nitrogen were the sedimentary variables that better explained the variations of macrofaunal structure throughout the study period (Spearman’s correlation, $\rho = 0.29$, $P = 0.02$).

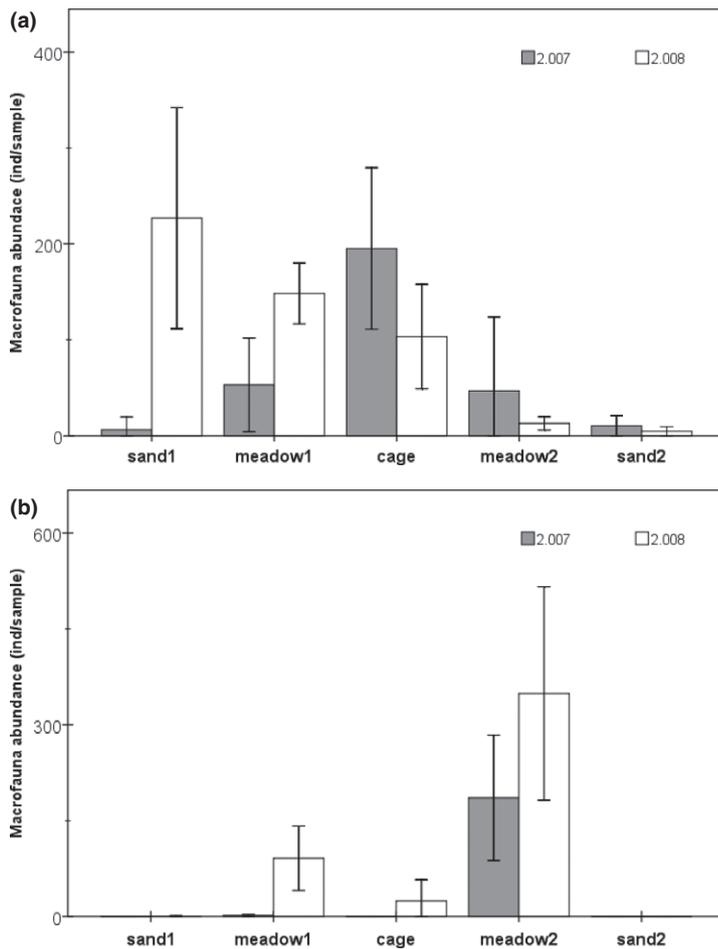


Figure 3 Abundances of (a) *Apseudes talpa*, (b) *Ampelisca brevicornis* at the sampling sites. Error bars are SE of means. 2007 and 2008 represent 'before' and 'after' conditions respectively.

Discussion

Our study suggests a differential response to fish farming activities between the unvegetated and vegetated seabeds. The most evident difference consisted of an increase in abundances in down current *C. nodosa* meadows mainly due to high abundances of the amphipod *Ampelisca brevicornis*. This area was the closest to the fish cages (400 m) downwards and its assemblage structure can be partially explained by the current intensity since no effects of organic matter dispersion from fish cages were found.

A high variability of macrofauna assemblages was observed in the study area, showing spatial and temporal differences throughout the study period among *Cymodocea nodosa* meadows, sandy bare seabeds and the sediments directly underneath of the fish cages. The environmental effects of fish cages are limited to a radius of 10 s of meters around the farm and differences among locations

could be partially explained by the continuous current. Moreover, higher macrofauna abundances were found in northwards locations, considering as a whole ('sand1' and 'meadow1'), compared with leeward places. However, this trend was not consistent in all stations, with the exception of *C. nodosa* meadows southwards that showed high macrofaunal abundances, mainly due to high densities of an amphipod.

Off-shore fish cages produce an organic load consisting of uneaten food and fish faeces, which settle beneath the cages and are spread by currents on the seabed (Tomassetti, Persia, Mercatali, Vani, Marusso & Porrello 2009). The main factors controlling the extent of benthic organic enrichment are: farm size, husbandry methods and hydrographic conditions (Harstein & Rowden 2004; Mente, Pierce, Santos & Neofitou 2006; Giles 2008). Borja *et al.* (2009) showed that most of the benthic indices examined were significantly correlated with average current speed, farm

Table 3 Results of two-way PERMANOVA testing for differences in macrofauna assemblage structure, overall macrofaunal abundance, and *Apeudes talpa* and *Ampelisca brevicornis* abundances with varying seabed ('Habitat', fixed factor) and hydrodynamics conditions ('Hydrodynamics', fixed factor). Significant differences ($P < 0.05$) are highlighted in bold. Results of pairwise comparisons between different habitats for each hydrodynamic condition are included as a result of a significant interaction between 'Habitat' and 'Hydrodynamics'.

Source of variation	Assemblage structure			Overall macrofaunal abundance			<i>Apeudes talpa</i> abundance			<i>Ampelisca brevicornis</i> abundance			
	df	MS	F	P	MS	F	P	MS	F	P	MS	F	P
Habitat (Ha) (seagrassversus sandy bare bottoms)	1	23149	10.461	0.001	638.41	20.287	0.0001	26.623	0.792	0.377	2120.1	71.83	0.0001
Hydrodynamics (H) (wind-versus leeward)	1	9482.8	4.2851	0.001	38.625	0.965	0.329	895.54	42.24	0.0001	390.35	7.199	0.009
Ha x H	1	10580	4.7808	0.001	283.63	10.256	0.002	13.687	0.645	0.425	438.31	24.09	0.0001
Residual	67	2213			27.655			21.228			18.195		

Table 4 Factors (F1–F6) obtained from analysis of the macrofauna data

Group	Species	Scores	Factor
Amphipoda	<i>Urothoe marina</i>	0.7	F1
Amphipoda	<i>Photis longicaudata</i>	0.688	F1
Mollusca	<i>Turritella brochii</i>	0.663	F1
Amphipoda	<i>Harpinia antennaria</i>	0.617	F1
Polychaeta	<i>Sigalion squamatum</i>	0.503	F1
Tanaidacea	<i>Apeudes talpa</i>	0.82	F2
Polychaeta	<i>Prionospio steenstrupii</i>	0.738	F2
Polychaeta	<i>Scoloplos (Leodamas) sp</i>	0.572	F2
Polychaeta	<i>Ditrupea arietina</i>	0.808	F3
Polychaeta	<i>Aponuphis bilineata</i>	0.797	F3
Cumacea	<i>Iphinoe canariensis</i>	0.521	F3
Amphipoda	<i>Bathyporeia elegans</i>	-0.838	F4
Amphipoda	<i>Ampelisca brevicornis</i>	-0.829	F4
Amphipoda	<i>Pariambus typicus</i>	0.849	F5
Amphipoda	<i>Pthisica marina</i>	0.668	F5
Ostracoda	<i>Cypridina mediterranea</i>	0.795	F6
Amphipoda	<i>Urothoe pulchella</i>	0.775	F6

production level and number of years of farm activity. Borja (2002) recommended the following classification: (i) 'good' sites (>30 m water depth and current speed > 15 cm s⁻¹); (ii) 'moderate' sites (15–30 m depth, 5–15 cm s⁻¹ current speed); (iii) 'bad' sites (<15 m depth, <5 cm s⁻¹ current speed). Nevertheless, Sarà, Scilipoti, Milazzo and Modica (2006) recently suggested that the relative area of influence of the impacts of fish farms increases proportionally to increasing current velocities. Moreover, they proposed that the distribution of wastes from the cages is likely to be dependent on movements at the bottom of the water column, suggesting that resuspension of organic particles (faeces and uneaten pellets) is a key factor (Cromey, Nickell & Black 2002). Kutti, Hansen, Ervik, Hoisaeter and Johannessen (2007) found that, at deep water sites, organic waste affected the benthic community on a much larger spatial scale than at shallower sites. Thus, benthos is affected by hydrodynamic conditions that govern the sediments natural assemblage capacity (MacLeod, Moltschaniwskyj, Crawford & Forbes 2007).

The relation between sedimentary variables and benthic assemblages depends on organic enrichment rates and grain size features since finer sediments tend to accumulate more organic matter than coarser ones (Gray 1981). In a former study conducted in the Canary archipelago finer sediments increased significantly beneath fish cages,

which triggered changes in meiofauna assemblage structure (Riera *et al.* 2011b). In the present study, caged sediments were dominated by the tanaid *Apseudes talpa*, a species that has been previously found beneath off-shore fish cages in the Canary Islands (Monterroso, Nuñez & Riera 2004), with preference to fine and very fine sands. Interannual variations (2007 versus 2008) might be explained by seasonal variations of this species, being more abundant in winter. The second species in importance was the amphipod *Ampelisca brevicornis*, which clearly dominated the macrofauna community structure of *C. nodosa* meadows on the southwards station. This amphipod is considered a sensitive species, only present in non-disturbed seabeds (Gomez-Gesteira & Dauvin 2000) and colonized polluted sediments after several years (Dauvin 1998). However, this species has been currently considered as tolerant to sewage input (De-la-Ossa-Carretero, Del-Pilar-Ruso, Giménez-Casalduero, Sánchez-Lizaso & Dauvin 2012), and a common macrofauna component in harbour sediments (Ingole, Sivadas, Nanajkar, Sautya & Nag 2009).

To prevent sensitive communities, e.g. seagrass meadows, from aquaculture activities a buffer area should be considered in each study area. Dumbauld, Ruesink and Rumrill (2009) showed that a buffer zone of 100 and 200 m around the edge of shellfish aquaculture installations were proposed as a management measure to fish farmers. In the present study, a minimum distance of 400 m is considered enough to be not affected by organic load (faeces and uneaten pellets) from aquaculture farms, however, the study bay was characterized by intermediate current intensity (20–22 cm s⁻¹) and the buffer zone should consider the specific hydrodynamics patterns of each location. In a conservative way, we proposed a minimum distance of 500 m in the Canary archipelago to prevent affection to *C. nodosa* meadows from off-shore fish cages.

Acknowledgments

To the remaining staff of CIMA SL (Centro de Investigaciones Medioambientales del Atlántico) for field support during sampling campaigns conducted in the present study. To the company Cultivos Marinos del Atlántico for financial support, and especially to Mr. C. Ceballos for his continuous encouragement and logistic help throughout

the study period. Last but not least, we are grateful to an anonymous referee who greatly improved the first version of the manuscript.

References

- Anders P.J. (1998) Conservation aquaculture and endangered species. *Fisheries* **23**, 28–31.
- Andersen J.M. (1976) An ignition method for determination of total phosphorus in lake sediments. *Water Research* **10**, 329–331.
- Anderson M.J. (2005) *PERMANOVA: A FORTRAN computer program for permutational multivariate analysis of variance*. University of Auckland, New Zealand, Department of Statistics.
- Black K.D., Kierner M.C.B. & Ezzi I.A. (1996) The relationships between hydrodynamics, the concentration of hydrogen sulphide produced by polluted sediments and fish health at several marine cage farms in Scotland and Ireland. *Journal of Applied Ichthyology* **12**, 15–20.
- Borja A. (2002) Los impactos ambientales de la acuicultura y la sostenibilidad de esta actividad. *Boletín Instituto Español de Oceanografía* **18**, 41–49.
- Borja A., Rodríguez J.G., Black K., Boday A., Emblow C., Fernandes T.F., Forte J., Karakassis I., Muxika I., Nickell T.D., Papageorgiou N., Pranovi F., Sevastou K., Tomassetti P. & Angel D. (2009) Assessing the suitability of a range of benthic indices in the evaluation of environmental impact of fin and shellfish aquaculture located in sites across Europe. *Aquaculture* **293**, 213–240.
- Brito M.C., Núñez J. & San Martín G. (2005) Polychaetes associated to a *Cymodocea nodosa* meadow in the Canary Islands: assemblage structure, temporal variability and vertical distribution compared to other Mediterranean seagrass meadows. *Marine Biology* **146**, 467–481.
- Buchanan J.B. (1984) Sediment analysis. In: *Methods for the study of marine benthos* (ed. by N.A. Holme & A.D. McIntyre), pp. 41–65. Blackwell Scientific Publications, Oxford.
- Clarke K.R. & Warwick R.M. (2001) *Changes in marine communities: an approach to statistical analysis and interpretation*, 2nd (ed). PRIMER-E, Plymouth, UK.
- Cromey C.J., Nickell T.D. & Black K.D. (2002) DEPOMOD – modeling the deposition and biological effects of waste solids from marine cage farms. *Aquaculture* **214**, 211–239.
- Dauvin J.C. (1998) The fine sand *Abra alba* community of the Bay of Morlaix twenty years after the Amoco Cadiz oil spill. *Marine Pollution Bulletin* **36**, 669–676.
- De-la-Ossa-Carretero J.A., Del-Pilar-Ruso Y., Giménez-Casalduero F., Sánchez-Lizaso J.L. & Dauvin J.-C. (2012) Sensitivity of amphipods to sewage pollution. *Estuarine, Coastal and Shelf Science* **96**, 129–138.

- Delgado O., Ruiz J.M., Perez M., Romero J. & Ballesteros E. (1999) Effects of fish farming on seagrass (*Posidonia oceanica*) in a Mediterranean bay: seagrass decline after organic loading cessation. *Oceanologica Acta* **22**, 105–118.
- Dumbauld B.R., Ruesink J.L. & Rumrill S.S. (2009) The ecological role of bivalve shellfish aquaculture in the estuarine environment: A review with application to oyster and clam culture in West Coast (USA) estuaries. *Aquaculture* **290**, 196–223.
- Edgar G., Davey A. & Shepherd C. (2010) Application of biotic and abiotic indicators for detecting benthic impacts of marine salmonid farming among coastal regions of Tasmania. *Aquaculture* **307**, 212–218.
- Everett R., Ruiz G. & Carlton J.T. (1995) Effect of oyster mariculture on submerged aquatic vegetation: an experimental test in a Pacific Northwest estuary. *Marine Ecology Progress Series* **125**, 205–217.
- FAO (2007) *The State of the World Fisheries and Aquaculture 2006*. Rome, Italy, Food and Agricultural Organization.
- Giles H. (2008) Using Bayesian networks to examine consistent trends in fish farm benthic impact studies. *Aquaculture* **274**, 181–195.
- Gomez-Gesteira J.L. & Dauvin J.-C. (2000) Amphipods are good indicators of the impact of oil spills on soft-bottom macrobenthic communities. *Marine Pollution Bulletin* **40**, 1017–1027.
- Gray J.S. (1981) *The Ecology of Marine Sediments*. University of Oslo, Norway, An introduction to the structure and function of benthic communities. Institute of Marine Biology and Limnology 179 pp.
- Harstein N.D. & Rowden A.A. (2004) Effect of biodeposits from mussel culture on macroinvertebrate assemblages at sites of different hydrodynamic regime. *Marine Environmental Research* **57**, 339–357.
- Hemminga M.A. & Duarte C. (2000) *Seagrass ecology*. Cambridge University Press, Cambridge, UK 292 pp.
- Holmer M. (2010) Environmental issues of fish farming in offshore waters: perspectives, concerns and research needs. *Aquaculture Environmental Interactions* **1**, 57–70.
- Holmer M., Duarte C.M., Heilskov A., Olesen B. & Terrados J. (2003) Biogeochemical conditions in sediments enriched by organic matter from net-pen fish farms in the Bolinao area Philippines. *Marine Pollution Bulletin* **44**, 685–696.
- Holmer M., Black K., Duarte C., Marbá N. & Karakassis I. (2008) *Aquaculture in the ecosystem*. Springer, Berlin 328 pp.
- Ingole B., Sivadas S., Nanajkar M., Sautya S. & Nag A. (2009) A comparative study of macrobenthic community from harbours along the central west coast of India. *Environmental Monitoring and Assessment* **154**, 135–146.
- Ireland S.C., Anders P.J. & Siple J.T. (2002) Conservation aquaculture: an adaptive approach to prevent extinction of an endangered while sturgeon population (*Acipenser transmontanus*). In: *Biology, management and protection of North American sturgeons*. (ed. by W. Vanwinkle, P. Anders, D. Dixon & D. Secor), pp. 211–222. American Fisheries Society Symposium 28, Bethesda, MD.
- Kutti T., Hansen P.K., Ervik A., Hoisaeter T. & Johannesen P. (2007) Effects of organic effluents from salmon farm on a fjord system. II. Temporal and spatial patterns in infauna community composition. *Aquaculture* **262**, 355–366.
- Machias A., Karakassis I., Labropoulou M., Somarakis S., Papadopoulou K.N. & Papaconstantinou C. (2004) Changes in wild fish assemblages after the establishment of a fish farming zone in an oligotrophic marine ecosystem. *Estuarine Coastal and Shelf Science* **60**, 771–779.
- MacLeod C., Moltschanivskyj N.A., Crawford C.M. & Forbes S.E. (2007) Biological recovery from organic enrichment: some systems cope better than others. *Marine Ecology Progress Series* **342**, 41–53.
- Mente E., Pierce G.J., Santos M.B. & Neofitou C. (2006) Effect of feed and feeding in the culture of salmonids on the marine aquatic environment: a synthesis for European aquaculture. *Aquaculture International* **14**, 499–522.
- Monterroso O., Nuñez J. & Riera R. (2004) Macrofauna de fondos blandos en las concesiones de acuicultura de la bahía de Igueste de San Andrés Tenerife. *Revista de la Academia Canaria de Ciencias* **15**, 77–86.
- Neckles H.A., Short F.T., Barker S. & Kopp B.S. (2005) Disturbance of eelgrass *Zostera marina* by commercial mussel *Mytilus edulis* harvesting in Maine: dragging impacts and habitat recovery. *Marine Ecology Progress Series* **285**, 57–73.
- Pavon-Salas N., Herrera R., Hernandez-Guerra A. & Haroun R. (2000) Distributional pattern of seagrasses in the Canary Islands (Central-East Atlantic Ocean). *Journal of Coastal Research* **16**, 329–335.
- Pergent-Martini C., Boudouresque C.F., Pasqualini V. & Pergent G. (2006) Impact of fish farming facilities on *Posidonia oceanica* meadows: a review. *Marine Ecology* **27**, 310–319.
- Pusceddu A., Fraschetti S., Mirto S., Holmer M. & Danovaro R. (2007) Effects of intensive mariculture on sediment biochemistry. *Ecological Applications* **17**, 1366–1378.
- Riera R., Monterroso O., Rodríguez M. & Ramos E. (2011a) Biotic indexes reveal the impact of harbor enlargement on benthic fauna. *Chemical Ecology* **27**, 311–326.
- Riera R., Monterroso O., Rodríguez M., Ramos E. & Sacramento A. (2011b) Six-year study of meiofaunal dynamics in fish farms in Tenerife (Canary Islands, NE Atlantic Ocean). *Aquatic Ecology* **45**, 221–229.
- Riera R., Tuya F., Sacramento A., Ramos E., Rodríguez M. & Monterroso O. (2011c) The effects of brine

- disposal on a subtidal meiofauna community. *Estuarine, Coastal and Shelf Science* **93**, 359–365.
- Riera R., Tuya F., Ramos E., Rodríguez M. & Monterroso O. (2012) Variability of macrofaunal assemblages on the surroundings of a brine disposal. *Desalination* **29**, 94–100.
- Ruiz J.M., Perez M. & Romero J. (2001) Effects of fish farm loadings on seagrass (*Posidonia oceanica*) distribution, growth and photosynthesis. *Marine Pollution Bulletin* **42**, 749–760.
- Sarà G. (2007) A meta-analysis on the ecological effects of aquaculture on the water column: dissolved nutrients. *Marine Environmental Research* **63**, 390–408.
- Sarà G., Scilipoti D., Milazzo M. & Modica A. (2006) Use of stable isotopes to investigate dispersal of waste from fish farms as a function of hydrodynamics. *Marine Ecology Progress Series* **313**, 261–270.
- Sawyer C.N., McCarty P.L. & Parkin G.F. (2003) *Chemistry for environmental engineering and science*. 5th (ed). McGraw-Hill, North America.
- SPSS (1999) *SPSS for Windows*. Standard version 10.0.1, SPSS INC., Chicago, Illinois.
- Tabachnick B.G. & Fidell L.S. (2001) *Using Multivariate Statistics, Fourth Edition*. Needham Heights, MA: Allyn and Bacon (eds).
- Tomassetti P., Persia E., Mercatali I., Vani D., Marusso V. & Porrello S. (2009) Effects of mariculture on macrobenthic assemblages in a western mediterranean site. *Marine Pollution Bulletin* **58**, 533–541.
- Troell M., Halling C., Neori A., Chopin T., Buschmann A.H., Kautsky N. & Yariah C. (2003) Integrated mariculture: asking the right questions. *Aquaculture* **226**, 69–90.
- Tuya F., Boyra A., Sanchez-Jerez P., Haroun R.J. & Barbera C. (2004) Can one species determine the structure of a rocky benthic community on a temperate rocky reef? the case of the long-spined sea urchin *Diadema antillarum*. *Hydrobiologia* **519**, 211–214.
- Walkley A. & Black J.A. (1934) An examination of the Degtjareff method for determining soil organic matter and a proposed modification of the chromic titration method. *Soil Science* **37**, 29–38.
- Wilson S., Blake C., Berges J.A. & Maggs C.A. (2004) Environmental tolerances of free-living coralline algae (Maërl): implications for European marine conservation. *Biological Conservation* **120**, 279–289.
- Zenetos A., Streftaris N. & Larsen L.H. (2003) An indicator-based approach to assessing the environmental performance of European Marine Fisheries and Aquaculture. Technical Report n 87, European Environmental Agency, Luxembourg 65 pp.