

Variations in a shallow rocky reef fish community at different spatial scales in the western Mediterranean Sea

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ABSTRACT: Fishing grounds off the northern coast of Mallorca Island (western Mediterranean Sea) are subject to very different fishing efforts as a result of both commercial and recreational activities. We analysed the variability at different spatial scales of the fishable community associated with shallow littoral rocky bottoms. Underwater visual censuses (UVC) were carried out in 3 localities that have different fishing pressure. Three sites were sampled within each locality, and nine 50 m long UVC transects were completed within each site. Multivariate correlograms showed that the transects were completely independent statistical units even when located less than 75 m apart. The spatial variability of the fish assemblage is remarkable. A redundancy analysis (RDA) revealed highly significant differences between the species relative abundance of the different localities. The bottom structure and rugosity were not detected to have any effect on the fish assemblage. The variability of the bottom structure at the below-locality level shows that these tests were powerful enough and that the effects of bottom structure were not confused with the effects of fishing pressure. Localities with lower fishing pressure showed significantly higher species richness (8.4 ± 0.3 and 8.9 ± 0.3 species 250 m^{-2}) and abundance (54.6 ± 4.4 and 65.9 ± 4.1 ind. 250 m^{-2}) than localities with higher fishing pressure (6.7 ± 0.3 species 250 m^{-2} and 41.1 ± 4.5 ind. 250 m^{-2}). *Serranus cabrilla* and *Symphodus tinca* were significantly less abundant in the localities with higher fishing pressure. These results suggest that differences in fishing pressure may generate strong gradients of faunistic change even at relatively small spatial scales (5 to 10 km), and that the observed pattern emulates what would normally be expected to happen after protection measures are established in marine protected areas.

KEY WORDS: Spatial variations · Fishing pressure · Nested sampling design · Visual censuses · Vulnerable fish species · NW Mediterranean Sea

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INTRODUCTION

Numerous studies over the last decades have analysed the factors affecting fish communities. The effects of protecting marine areas have been studied extensively and the studies have shown the benefits and limits of management policies against overfishing (Bell 1983, Roberts & Polunin 1991, Rakitin & Kramer

1996, Pet-Soede et al. 2001, Gell & Roberts 2003, Russ & Alcalá 2004). The so-called reserve effects on fish communities have been extensively described and analysed in the Mediterranean Sea (Harmelin 1987, García-Rubies & Zabala 1990, Francour 1994, Dufour et al. 1995, Reñones et al. 1997, La Mesa & Vacchi 1999, Macpherson et al. 2002, Lipej et al. 2003). Species-specific effects are also well known, especially in

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the cases of vulnerable species (Harmelin et al. 1995, Coll et al. 1999, Reñones et al. 1999, Lenfant et al. 2003).

Obviously, the degree of protection is not the only factor influencing fish assemblages. Substrate characteristics (Luckhurst & Luckhurst 1978, Friedlander & Parrish 1998, García-Charton & Pérez-Ruzafa 1998), wave exposure (Gust et al. 2001, Micheli et al. 2005) and recruitment (García-Rubies & Macpherson 1995, Macpherson et al. 1997) are also influential.

The processes related to these types of factors often imply spatially structured outcomes, which makes fish community characteristics scale-dependent (García-Charton & Pérez-Ruzafa 1999). Examples of scale-mediated processes are experimentally-induced disturbances in fish assemblages (Syms & Jones 1999), exposure to pollution (Guidetti et al. 2003) and of course, the effect of protection (García-Charton & Pérez-Ruzafa 1999). In particular, one of the main sources of uncertainty when interpreting the results of field experiments that compare areas with different protection levels arises from spatial and temporal variability. This topic has recently been revised by García-Charton & Pérez-Ruzafa (1999), who made an overview of the different processes implicated, and demonstrated the existence of a spatial structure in a number of ecological descriptors (e.g. total abundance measured as the number of individuals, biomass and species richness), as well as in some species-specific patterns. Multi-scale nested sampling designs make it

possible to divide the variability observed at several spatial or temporal levels, and interpret relationships that are influenced by a variety of processes (Hewitt et al. 2002). These multi-scale designs have recently been adopted for studying fish communities (Reñones et al. 1999, Gust et al. 2001, Guidetti et al. 2003).

In this paper, we describe and analyse the spatial variations of the fishable community on the northern coast of Mallorca Island (western Mediterranean). This area has significant differences in fishing pressure caused by both commercial and recreational fishing activities. The differences in fishing pressure can result in changes in the fishable community, which are expected to be scale-dependent. Therefore, scale-dependent spatial differences (spatial autocorrelation), and the putative effects of bottom morphology were evaluated using a multi-scale experimental design in order to identify spatial patterns of the fishable community related to differences in fishing pressure.

MATERIALS AND METHODS

Study area. The study was conducted at 3 localities on the northern coast of Mallorca Island (Fig. 1). The localities were: Cavall Bernat (CB), Cala Extremer (CE) and Punta de la Sal (PS). All localities have very similar rocky bottoms with a slight slope (<20%). They are located at depths ranging between 5 and 15 m. At these depths, *Posidonia oceanica* seabeds are almost non-existent. These seabeds begin at a 15 m depth and continue to between 25 and 35 m where maerl beds (calcified red seaweed) start to appear. The 3 localities are similarly exposed to the predominant weather. Waves come from the N, NNE and NE 47% of the time (Source Ministerio de Fomento, Puertos del Estado, www.puertos.es [Ministry responsible for public works]). Waves from these directions clearly cause the worst effects in the area studied. If we only consider the days with waves higher than 2 m, then waves come from N, NNE and NE 74% of the time.

Fishing pressure in the area. Both commercial and recreational fishing are practised in the area. Five commercial fishing boats dedicated to trammel net fishing, operate for between 100 and 150 d in the area during spring, summer and autumn (4 from PP [Port de Pollença] and 1 from PA [Port d'Alcúida], Fig. 1, top right panel) (source DG Pesca, Govern de les Illes Balears [Department of Fisheries, Balearic

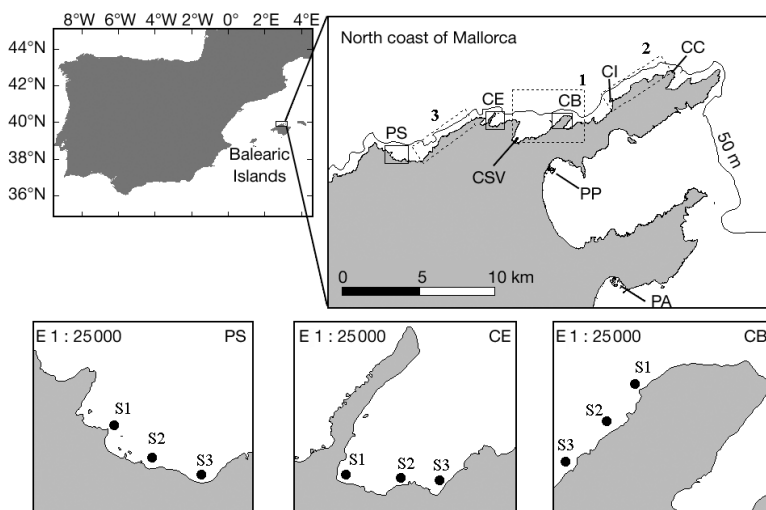


Fig. 1. Map of the study area indicating where study localities are situated (CB: Cavall Bernat; CE: Cala Extremer; PS: Punta de la Sal) and the sites within each locality (S1, S2, S3; in the lower panels). Cala Sant Vicenç (CSV) is the nearest urban area to the localities and Port de Pollença (PP) and Port d'Alcúida (PA) are the nearest safe ports. Dotted rectangles (labelled 1, 2 and 3) in the main panel indicate the fishing grounds both for commercial and recreational fisheries. CC: Cape of Catalunya; CI: Colomer Island

Islands Government] unpubl. data). Three more boats from PP occasionally fish in the area for an average of 30 d. In terms of commercial value, the most important species in the area is *Palinurus elephas*, which is primarily exploited at a depth of 50 to 100 m (Goñi et al. 2003). In the depth range of 5 to 35 m, *Scorpaena scrofa* is the most fished target species of the commercial fleet. The fisheries that focus on these 2 species also yield a commercially valuable by-catch (Table 1). Although official sales registers of the only fish wharf in Mallorca provide the only available information about the catch made by the trammel net fishery in the area, a large amount of the catch does not enter into the classical infrastructures of bulk marketing, such as fish wharfs. Thus, these statistics underestimate the abundances of the target and by-catch species caught.

Comparing the study sites, we see that CB is completely inside the *Scorpaena scrofa* fishing grounds of the local fleet (Rectangle 1, Fig. 1, top right panel) and CE is located relatively close, only 1 km away. Neither CE nor PS is fished by the commercial fleet. About a total of 4 km of trammel net is laid per fishing day in these grounds (data from the commercial boats involved).

Angling is the most popular recreational fishing method in Mallorca (Morales-Nin et al. 2005). For recreational fishing boats based in PP (100 boats), the most frequented fishing ground is the area from Cape de Catalunya (CC) to Colomer Island (CI), which is visited on 30% of the fishing days (Rectangle 2, Fig. 1). Recreational fishing boats only occasionally reach Rectangle 1 and CB. However, 18 recreational fishing boats are based in Cala Sant Vicenç (CSV). In addition, an average of 10 to 12 boats are towed there by car during spring and summer.

During the survey, 40% of the recreational fishing boat owners operating from CSV were interviewed. Based on their responses, we deduced that the number of fishing days per boat during summer ranged from 20 to 50 d. The preferred fishing grounds were CB and another area situated between CE and PS (Rectangle 3, Fig. 1), although some minor effort was

Table 1. Biomass (kg) and nominal values (€) of the main species captured during 2002 by 5 commercial boats fishing with trammel nets in the studied area (source: DG Pesca, Govern de les Illes Balears [Department of Fisheries, Balearic Islands Government] unpubl. data). The species are ordered according to economic importance. The relative importance of recreational fishing catches in relation to fishing method (1: angling, 2: spearfishing) are also indicated (Morales-Nin et al. 2005)

	Commercial fishing kg	€	Recreational fishing Percentage	Fishing method
Demersal fish species				
<i>Scorpaena scrofa</i>	747.64	8538.05	1	2
<i>Dentex dentex</i>	525.20	7610.15	0.03	1,2
<i>Sparus pagrus</i>	487.70	6569.32	0.02	1,2
<i>Lophius</i> spp.	474.60	3322.20		
<i>Scorpaena pocus</i>	623.80	3181.38	0.5	2
<i>Epinephelus marginatus</i>	86.70	1462.63	6	2
Rajidae	540.40	1291.56	0.01	1
<i>Diplodus sargus</i>	221.70	1192.75	7	1,2
<i>Spondyliosoma cantharus</i>	165.00	1173.15	0.2	1,2
<i>Diplodus vulgaris</i>	170.40	1128.05	6	1,2
<i>Trachinus</i> spp.	117.70	922.77	9	1
<i>Conger conger</i>	322.39	773.74	0.05	1,2
<i>Pagellus erythrinus</i>	145.60	751.30	2.8	1
<i>Sciaena umbra</i>	44.40	694.86	1	2
<i>Muraena helena</i>	178.60	682.25	3	1,2
<i>Mullus surmuletus</i>	87.20	533.66	6.5	1,2
<i>Serranus scriba</i>	81.80	359.92	9	1
<i>Symphodus tinca/Labrus</i> spp.	165.10	348.36	4	1,2
<i>Scyliorhinus</i> spp.	293.30	325.56	0.1	1
<i>Zeus faber</i>	18.80	203.42		
<i>Sparus aurata</i>	19.80	165.33	2	1,2
<i>Diplodus annularis</i>	49.80	154.38	8.5	1,2
<i>Phycis phycis</i>	51.40	143.41		
<i>Mustelus</i> spp.	18.90	95.45		
<i>Mycteroperca rubra</i>	8.00	84.00	0.2	2
<i>Sarpa salpa</i>	51.40	77.10	4	1,2
<i>Epinephelus caninus</i>	7.70	68.84	0.1	2
<i>Chelidonichthys</i> spp.	12.60	66.15		
<i>Oblada melanura</i>	15.60	41.34	5	1,2
<i>Serranus cabrilla</i>	6.40	19.84	10	1
Mugilidae	22.10	19.45	2.5	1,2
<i>Diplodus puntazzo</i>	3.60	6.62	0.3	1,2
Others	19.70	122.26	3.69	
Pelagic fish species	532.00	4096.26		
Decapod crustaceans				
<i>Palinurus elephas</i>	1566.70	66616.08		
Others	21.00	312.54		
Cephalopods				
<i>Sepia officinalis</i>	508.10	4034.31		
<i>Octopus vulgaris</i>	174.10	409.14	7.5	1,2
Others	81.80	1250.00		

exerted in CE and PS. CB was the most frequented fishing ground; it was visited on 50 to 85% of the fishing days. Only 2 boats went fishing in CE (5 to 15% of the fishing days) and 1 in PS (10% of the fishing days). During training and sampling periods (25 calm days), fishing activities were observed in CB every day, while no fishing activities were observed in

CE, PS, and Rectangle 3. *Serranus cabrilla* is the main target species for recreational fishing in the area (Table 1).

Sampling method. The fish abundance was estimated by underwater visual censuses (UVC). Using this method, we can detect spatial patterns of fish communities in relation to fishing pressure by comparing areas that differ greatly in fishing intensity (Pet-Soede et al. 2001). Moreover, it is more efficient than traditional sampling techniques (e.g. trawling, trammel nets, hook-and-line) when dealing with heterogeneous habitats (de Girolano & Mazzoldi 2001), as is the case of the rocky bottoms considered here. Censuses were always carried out between 10:00 and 16:00 h, the water temperature was almost constant (22 to 23°C), and the minimum horizontal underwater visibility was at least 20 m in all transects. The UVC were carried out along 50 × 5 m strip transects between a depth of 8 and 12 m. Sampling was performed by 2 divers. One diver swam slowly about 1 m above the bottom extending a 50 m tape measure and simultaneously recording all individuals of vulnerable species detected swimming within the transect. The second diver examined all the holes and crevices within the transect using a torch, recording individuals of mimetic species or individuals displaying cryptic behaviour. All individuals, except those belonging to the genus *Mugil*, were identified to the species level.

As some species can be found either swimming or hidden, the records made by the 2 observers were compared. If there were 2 individuals of the same species and size in the same position, they were considered to be the same fish and therefore only 1 individual was counted.

UVC were restricted to species that are usually fished (vulnerable species). Species were included in this group based on data from the fishing authorities (DG Pesca, Govern de les Illes Balears [Department of Fisheries, Balearic Islands Government] unpubl. data) and Morales-Nin et al. (2005) (Table 1).

Study design. A nested sampling design was adopted to analyse the spatial variations that occur in the fish assemblages (Kingsford & Battershill 1998). The sampling structure was defined in 3 sampling spatial scales. The largest scale was represented by 3 localities (L), situated more than 3 miles apart from each other (CB, CE and PS in Fig. 1, top right panel). The intermediate scale was represented by 3 sites (S) located at random (75 to 175 m apart) within each of the localities (S1, S2 and S3 in Fig. 1, bottom panels), and the smallest scale was represented by 9 transects (T) located randomly within each site. This represents a total of 81 transects with 27 UVC conducted in each locality from 14 August to 30 September in the summer of 2002.

Bottom morphology. Data on bottom morphology was also recorded for each transect using 2 different, complementary measuring techniques. The bottom structure was defined as the percentage of the total surface of the transect covered by each of the 4 rocky bottom types that were defined *a priori*: (1) rocky bottom without boulders (RBWB), (2) large-sized boulders (LSB) ($\varnothing > 2$ m), (3) medium-sized boulders (MSB) ($2 \text{ m} > \varnothing > 1$ m), and (4) small-sized boulders (SSB) ($\varnothing < 1$ m). Moreover, the substrate rugosity index (SR) (Luckhurst & Luckhurst 1978), was defined by a scale of increasing rugosity from 1 to 4, taking into account the intermediate values: (1) without vertical variations higher than 1 m and with very few holes; (2) vertical variations between 1 and 2 m, and with few holes or crevices; (3) vertical variations higher than 2 m with holes and crevices occupying at least 25% of the total transect length; and (4) very high vertical variations with holes and crevices occupying more than 25% of the total transect. We preferred this semi-quantitative methodology to the quantitative methods for assessing topography. Quantitative methods such as the 'chain-and-tape' (Risk 1972) are more precise when dealing with bottoms with very few or even no holes, crevices and overhangs. However, these surface features are of particular interest and are very abundant in the localities we surveyed and quantitative methodologies are clearly limited when quantifying them (McCormick 1994).

Length and biomass estimates. The total length of individuals was estimated to the nearest 2 cm. To ensure 2 cm precision, 67 individual plastic fish models with different shapes and lengths ranging from 4 to 44 cm were distributed at random over a 5 m deep sea floor and their length was estimated. In order that the divers did not mechanically learn the sizes of the particular test fish, the procedure was repeated on different days, randomly rearranging the plastic fish in every trial, until the observed and actual lengths were not significantly different at high probability levels ($p \geq 0.7$) (Bell et al. 1985). The observed and actual length distributions were compared using a 1-sample Kolmogorov-Smirnov test (Siegel & Castellan 1988). After the training period, the lengths estimated by the divers were compared (*t*-test) and significant differences were never observed.

To analyse the community metrics (i.e. indices referring to all the species combined), the mean length and mean maximum length were determined for each transect:

$$\text{Mean length} = \sum L/n$$

where L is the length of an individual and n is the total number of individuals.

$$\text{Mean maximum length} = \sum(L_{\max j} \times n_j) / n$$

where $L_{\max j}$ is the maximum length obtained by species j , and n_j is the number of individuals of species j (Nicholson & Jennings 2004).

Weight values were determined for each individual of the censused species by applying published weight-length (W–L) relationships (Morey et al. 2003 for almost all the species; Stergiou & Moutopoulos 2002 for *Oblada melanura*, and Petrakis & Stergiou 1995 for *Spondyliosoma cantharus*). We preferred to use W–L relationships obtained in the Balearic Islands in order to eliminate possible biases due to geographic variations. Mugilidae and *Mycteroperca rubra* biomass values were not estimated because the former was not classified at the species level and there was no W–L relationship available for the latter.

Data analysis. Bottom morphology: The bottom structure was measured according to the 4 percentage of cover variables and subjected to a principal coordinates analysis (PcoA, using the between-transect chord distance as a dissimilarity estimate). The 3 first principal axes were retained (explaining 98.2% of the variability). To test for between-locality and within-locality differences, we used a nested MANOVA on the 3 first principal axes, and a nested ANOVA on the rugosity index. The MANOVA and ANOVA were carried out using CANOCO 4.5 and Statistica 6.0, respectively.

Geostatistical procedures: The aim here is to evaluate the minimum distance between samples to assure that samples taken farther away are independent replicates. The spatial pattern of the fish assemblage composed of vulnerable species was described using spatial multivariate correlograms (Legendre & Legendre 1998) with a MATLAB routine. The rationale of the multivariate extension of the spatial correlogram (Mantel correlogram) implies calculating the Mantel statistic, z . The Mantel statistic measures the closeness between 2 matrices. In this case, the 2 matrices are the pair-wise geographical distance matrix (X) and the pair-wise faunistic (dis)similarity matrix (Y). From the possible faunistic (dis)similarity measures, we chose to use the Bray-Curtis distance (Bray & Curtis 1957). The Mantel correlogram is made by plotting a number of Mantel z , one for each of the distance classes considered. In our case, we are interested in the small-scale spatial patterns. Consequently, we only considered the first 450 m and this distance was divided into 6 lags of 75 m. For distance Class 1, all pairs located between 0 and 75 m were recoded as ones in the X_1 matrix and the other pairs as zeros (z_1 was the corresponding Mantel z). The process was repeated for each of the 6 distance classes considered. The significance of each z_i ($i = 1$ to 6) was tested using an iterative permutation test by randomly permuting rows and columns of the

species abundance matrix (Manly 1991). This procedure implies multiple testing, since a test is performed on each of the lags considered. Therefore, we applied a progressive Bonferroni correction (Legendre & Legendre 1998).

In addition, we completed a second multivariate correlogram to discern the spatial patterns in species composition imposed by the bottom structure. These patterns are not affected by the relevant environmental variables. In short, a PCoA was carried out on the Bray-Curtis similarity distance matrix, and the axes extracted were considered as response variables in a redundancy analysis. The variables describing the bottom structure were considered as covariables. Therefore, the sample scores within the resulting ordination space parallel the residuals from a more conventional univariate regression model, and the pair-wise Euclidean distances within this space estimate the faunistic similarity that is not explained by the environmental variables considered.

Statistical analyses: To carry out univariate analyses, we analysed the species richness, abundance, biomass, mean length and mean maximum length using a nested model $Y_{ijk} = \text{mean} + L_j + S(L)_{ij} + T_{ijk}$ (Underwood 1997, Kingsford & Battershill 1998, García-Charton & Pérez-Ruzafa 1999), where Y_{ijk} is each individual value (k) of the dependent variable in any site (j) in a given locality (i). In this ANOVA design, locality was a fixed factor (L), site factor (S) was random and nested within locality, and the transects (T) were the replicates (the error term in the model). This allowed us to discriminate between the 3 spatial scales studied and also to test for significant differences between Localities and Sites. The species-specific mean length of species with more than 20 individuals in each locality were compared using a 1-way ANOVA. Species richness, abundance, biomass and length parameters were tested for assumptions of normality and homogeneity of variance before applying parametric tests. When these assumptions were not met, the data were log- or square-root-transformed (Underwood 1981).

To study the multivariate descriptors, the species abundances were analysed using Redundancy Analysis (RDA). The RDA model considered is the multivariate extension of the hierarchical (nested) analysis of variance (when species turnover is not very large, ter Braak & Smilauer 2002). To evaluate the significance of the axis, the multivariate approaches usually use permutation-based tests instead of comparing the observed F -ratio against a theoretical distribution (Manly 1991). The model used for each of the strata as well as the permutation procedure in each case are detailed in Table 2. In addition to testing the variability explained at different sampling scales, it is also possible to test the effects of specific environmental vari-

ables. Only species with an abundance greater than 3 individuals were included in the analysis. Mugilidae was not considered as the individuals were not classified to the species level.

Non-linear multidimensional scaling (MDS) (Legendre & Legendre 1998) was also completed at site level (pooling the data of the 9 transects from a site; i.e. 9 sites and 14 species) in order to compare the RDA ordi-

Table 2. Model description (variables and co-variables) and the permutation type used in the 4 tests conducted. In all cases, the response variable was the abundance of the 14 most abundant vulnerable fish species (Table 3), the analyses completed were Redundancy Analyses, and the number of permutations was 1999

	Aim	Variable	Co-variable	Permutation type
Test 1	Effects of the substrate rugosity	Rugosity index	Site	Unrestricted permutation
Test 2	Effects of the bottom structure	4 cover percentage bottom types	Site	Unrestricted permutation
Test 3	Between-locality differences	Locality (category)	None	Sites permuted. Transects within sites not permuted
Test 4	Between-site differences	Sites (category)	Locality (category)	Sites permuted. Transects within sites not permuted

Table 3. Total number of individuals (n) and mean total length \pm SE for species and localities and mean maximum total length \pm SE according to localities (see Fig. 1 legend for explanations of locality abbreviations). None of the between-locality differences are significant, except for *Mullus surmuletus* (CE > [PS = CB])

Species	CB		CE		PS	
	n	Mean TL	n	Mean TL	n	Mean TL
Muraenidae						
<i>Muraena helena</i>	1	44.0	2	50.0 \pm 10.0		
Gadidae						
<i>Phycis phycis</i>					1	12.0
Serranidae						
<i>Epinephelus costae</i>			2	14 \pm 2.0		
<i>Epinephelus marginatus</i> ^a	16	16.6 \pm 1.5	9	21.8 \pm 1.8	18	21.9 \pm 1.0
<i>Mycteroperca rubra</i>					1	20.0
<i>Serranus cabrilla</i> ^a	24	7.3 \pm 0.8	68	8.2 \pm 0.3	147	8.5 \pm 0.3
<i>Serranus scriba</i> ^a	104	13.1 \pm 0.3	129	13.0 \pm 0.3	112	13.9 \pm 0.4
Sciaenidae						
<i>Sciaena umbra</i> ^a	1	26.0	4	25.0 \pm 1.0	5	24.0 \pm 0.6
Mullidae						
<i>Mullus surmuletus</i> ^a	69	10.3 \pm 0.3	73	12.9 \pm 0.3	144	10.7 \pm 0.3
Sparidae						
<i>Dentex dentex</i> ^a	2	32.0 \pm 2.0			7	34.0 \pm 2.6
<i>Diplodus puntazzo</i> ^a	14	17.3 \pm 0.8	21	13.9 \pm 0.8	13	16.2 \pm 1.4
<i>Diplodus sargus</i> ^a	39	16.7 \pm 0.7	80	14.3 \pm 0.7	60	16.4 \pm 0.5
<i>Diplodus vulgaris</i> ^a	350	14.2 \pm 0.2	301	12.3 \pm 0.3	368	12.5 \pm 0.3
<i>Oblada melanura</i> ^a	69	14.0 \pm 0.4	125	11.8 \pm 0.4	124	13.0 \pm 0.3
<i>Spondyliosoma cantharus</i> ^a	2	15.0 \pm 1.0	6	11.3 \pm 1.7	1	14.0
Labridae						
<i>Labrus merula</i> ^a			10	16.0 \pm 1.8	9	17.3 \pm 1.9
<i>Labrus viridis</i> ^a	3	10.0 \pm 0.0	11	12.4 \pm 1.1	3	10.7 \pm 1.3
<i>Symphodus tinca</i> ^a	95	11.9 \pm 0.6	310	8.5 \pm 0.3	217	11.3 \pm 0.4
Mugilidae			6	39.0 \pm 0.8	20	27.0 \pm 0.5
Scorpaenidae						
<i>Scorpaena porcus</i>	1	14.0	1	22.0	1	12.0
<i>Scorpaena scrofa</i>			1	10.0		
Mean total length	790	13.3 \pm 0.4	1159	11.7 \pm 0.4	1251	12.3 \pm 0.5
Mean maximum total length	790	17.4 \pm 0.6	1159	16.8 \pm 0.5	1251	17.1 \pm 0.5

^aSpecies included in the multivariate analyses

nation bi-plot with the one obtained using an unconstrained ordination method.

RESULTS

Species assemblage

The abundance and mean length of the vulnerable species are presented in Table 3. All species were affected by 2 or more fishing gears. The most abundant species in descending order were *Diplodus vulgaris*, *Symphodus tinca*, *S. scriba*, *Oblada melanura*, *Mullus surmuletus*, *Serranus cabrilla* and *Diplodus sargus*, which accounted for more than 90% of the total number of individuals censused. The largest individuals were *Muraena helena*, *Dentex dentex* and *Mugil* sp. and the smallest individuals were *D. sargus* and *S. tinca*.

Geostatistical procedures

Multivariate correlograms showed that there is no significant autocorrelation between samples at any of the lag distances considered. The same result was obtained with the raw data (i.e. Bray-Curtis dissimilarity) and after the data were adjusted for the effects due to bottom structure (Fig. 2). These results support the idea that transects were fully independent statistical units even when situated less than 75 m apart.

Bottom morphology

The bottom structure of the different localities is presented in Table 4. Transects at CB exhibited larger cover percentages of rock (RBWB) and hence, the bot-

Table 4. Bottom morphology of the sampling localities. Values represent mean cover (% \pm SE). RBWB: rocky bottom without boulders; LSB: large-sized boulders ($\varnothing > 2$ m); MSB: medium-sized boulders ($2 \text{ m} > \varnothing > 1$ m); SSB: small-sized boulders ($\varnothing < 1$ m). The SR values represent the mean (\pm SE) of the substrate rugosity index for each area. CB: Cavall Bernat; CE: Cala Extremer; PS: Punta de la Sal

Substrate type	Locality		
	CB	CE	PS
RBWB	62.4 \pm 4.5	7.5 \pm 3.9	0.0 \pm 0.0
LSB	20.1 \pm 4.0	47.5 \pm 5.0	54.7 \pm 5.6
MSB	10.4 \pm 1.5	27.5 \pm 3.1	31.2 \pm 3.7
SSB	7.1 \pm 1.5	17.5 \pm 3.3	14.1 \pm 2.6
SR	2.6 \pm 0.2	2.0 \pm 0.1	2.3 \pm 0.1

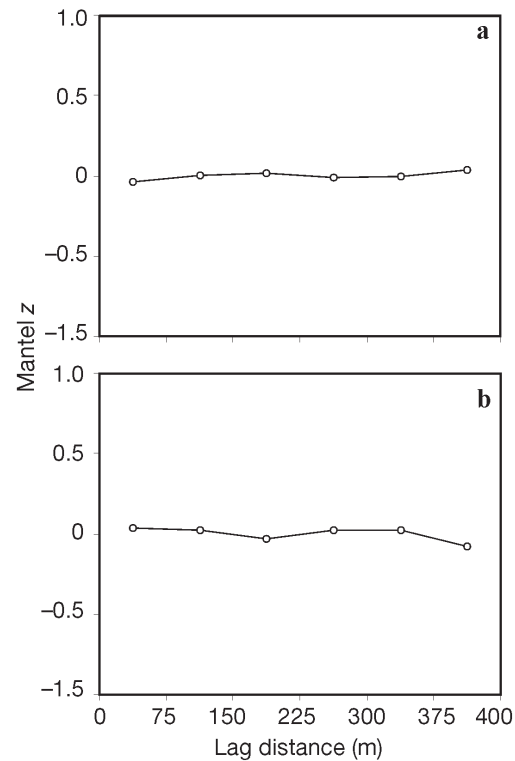


Fig. 2. All-directional spatial correlogram for the faunistic (dis)similarity measured by Bray-Curtis distance before (a) and after (b) adjusting for the possible effects of the bottom structure, based on 81 samples and the 14 most abundant vulnerable species (Table 3). Note that the negative Mantel z statistic denotes between-sample similarity (due to the recoding procedure; i.e. 1 for the samples within a specific distance class, 0 otherwise)

tom structure was found to be significantly different between localities (nested MANOVA results: Variance explained = 45.1%, $F = 32.059$, $p = 0.03$). Note, however, that the residual variability was very high (54.9%). Specifically, 44% corresponded to between-transect within-site variability and 10.9% to between-site within-locality variability. In contrast, the analysis of the rugosity index suggested that between-locality differences were non-significant (nested ANOVA results: Variance explained = 40.5%, $F = 1.24$, $p = 0.35$).

Bottom morphology was found to be unrelated to species composition at the within-site level (Table 5). Neither the transformed cover percentages of the 4 categories nor the rugosity index explained a significant percentage of the variability in species relative abundance. Note that these tests were performed considering the sites as co-variables. Therefore, we were testing the relationship between bottom structure and species assemblage at the within-site level (between-transects). Similarly, bottom structure and species

Table 5. Results of multivariate analyses (RDA). Variance explained by different explanatory variables. The values in brackets refer to the variance explained by all the variables in the model (i.e. the values for Tests 3 and 4 add up to 100%). *F*-ratios and probability values after 9999 permutations are also indicated

	Aim	Variance explained (%)	<i>F</i> -ratio	<i>p</i>
Test 1	Effects of the substrate rugosity	1.2 (1.5)	1.090	0.2985
Test 2	Effects of the bottom structure	4.4 (5.6)	1.390	0.1590
Test 3	Between locality differences	13.1 (58.3)	5.891	0.0040
Test 4	Between site differences	9.4 (41.7)	1.453	0.0035

composition were found to be unrelated at the between-site (i.e. within-locality) level, even in the case of a nested analysis (sites were permuted while within-site transects were not; variance explained = 1.2%; *F* = 0.959; *p* = 0.74).

Spatial structure

The total number of vulnerable species censused was 16 for CB and 18 for CE and PS. In CE and PS, the mean number of species (8.4 ± 0.3 and 8.9 ± 0.3 species 250 m^{-2} , respectively) and the abundance (54.6 ± 4.4 and 65.9 ± 4.1 ind. 250 m^{-2} , respectively) were significantly higher than in CB (6.7 ± 0.3 species 250 m^{-2} and 41.1 ± 4.5 ind. 250 m^{-2}), but non-significant differences were found in biomass (Fig. 3 and Table 6), mean length and mean maximum length (Table 3).

The results of the multivariate analyses obtained by decomposing the variability between the 3 levels considered show that Locality accounts for 13.1%, and Site for 9.4%. The remaining 77.5% is the residual variability (i.e. related to the transects). The relatively large (and highly significant, Table 5) percentage expressing the variability between Localities is worth noting. The ordination bi-plot corresponding to a RDA where the explanatory variable is the Locality is shown in Fig. 4a. The first axis is the only one that describes a significant pattern related to between-locality differences. The second axis was tested and found to be non-significant (*F* = 2.295, *p* = 0.08). These results point to the existence of faunistic dissimilarity, where CB and PS were the localities that were most different, and CE was located at an intermediate position. This point was confirmed with the results from non-metric MDS (Fig. 4b). *Serranus cabrilla* showed the largest correlation with this gradient (according to species results shown in Fig. 5). Other species contributing to this gradient in a lower stand were *Symphodus tinca*, *Labrus merula*, *Sciaena umbra*, *Diplodus sargus*, *Oblada melanura* and *Mul-*

lus surmuletus. The remaining species (i.e. *Epinephelus marginatus*, *Labrus viridis* and *Dentex dentex*) did not contribute to the configuration of this pattern.

In contrast to the results from analyses of species richness, total abundance and biomass (Table 6), the differences between sites were also found to be significant (Table 5).

At the species level, *Serranus cabrilla* showed significant differences both in abundance and biomass with lower values in CB.

Symphodus tinca also had significantly lower values in CB but only for abundance (Figs. 6 & 7).

Mean length was non-significant for all species except for *Mullus surmuletus* which had significantly higher values in CE (Table 3).

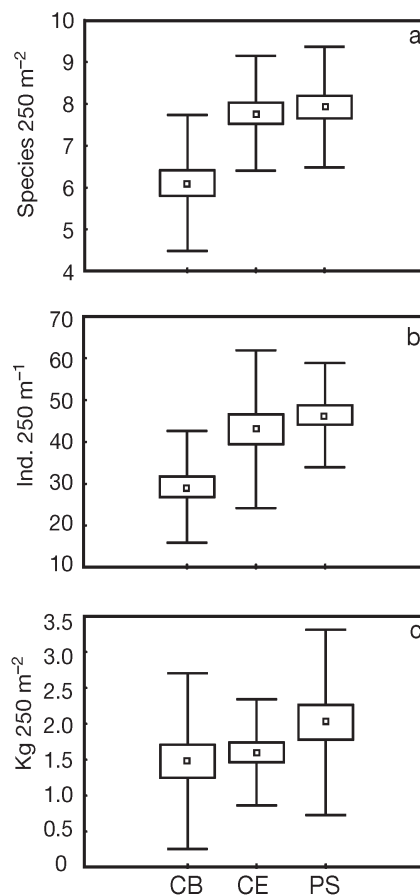


Fig. 3. Variations in ecological parameters, (a) number of vulnerable species, (b) abundance in number of individuals, and (c) biomass in kilograms, in the 3 localities studied. CB: Cavall Bernat; CE: Cala Extremer; PS: Punta de la Sal. Square: mean, rectangle: \pm SE, line: \pm SD

Table 6. Results of nested ANOVAs. Abundance (ind. 250 m⁻²), *S* (number of vulnerable species), biomass (g 250 m⁻²). Mean squares (MS), *F*-values and SNK post hoc comparisons are indicated. ns: non-significant differences, ***p* < 0.01, ****p* < 0.001, ns: non significant

Source of variation	df	Abundance				<i>S</i>				Biomass		
		MS	<i>F</i>	<i>p</i>	SNK	MS	<i>F</i>	<i>P</i>	SNK	MS	<i>F</i>	<i>p</i>
Locality	2	2225	17.77	**	(CE = PS) > CB	27.42	40.38	***	(CE = PS) > CB	225	2.59	ns
Site(loc.)	6	125	0.53	ns		0.63	0.29	ns		87	0.58	ns
Error	72	238				2.34				148		

DISCUSSION

All 21 vulnerable species surveyed are common in shallow rocky bottoms throughout the NW Mediterranean, both in marine reserves (Bell 1983, García-Rubies & Zabala 1990, La Mesa & Vacchi 1999) and in unprotected areas (Moranta et al. 1997, García-Chariton & Pérez-Ruzafa 1998). Almost all the species surveyed were less abundant (individuals and biomass) in north Mallorca than in protected areas throughout the NW Mediterranean. Some notable examples are *Sciaena umbra*, *Diplodus sargus* (Harmelin 1987, García-Rubies & Zabala 1990, Macpherson et al. 2002, García-Chariton et al. 2004), *D. vulgaris* (Macpherson et al. 2002, García-Chariton et al. 2004), *Serranus cabrilla* (Harmelin et al. 1995, Macpherson et al. 2002) and *Epinephelus marginatus* (Coll et al. 1999, Reñones et al. 1999, Macpherson et al. 2002, García-Chariton et al. 2004). In contrast, the abundance values reported here are comparable with those from other unprotected areas of the NW Mediterranean (García-Rubies & Zabala 1990, Harmelin et al. 1995, Moranta et al. 1997, García-Chariton & Pérez-Ruzafa 1998, Macpherson et al. 2002, García-Chariton et al. 2004). *Scorpaena scrofa*, which is the main target species of commercial fishing in the area at the depth range studied, was only censused in 1 transect. However, visual censuses clearly underestimate the abundance of this species due to its cryptic and mimetic behaviour. Other Scorpaenidae species have been found to be undetectable when performing visual censuses, although a posterior rotenone treatment in the censused transects revealed a moderate presence of the species (Willis 2001).

The results presented here indicate that the spatial structure of fish assemblages at a small scale is less evident than expected.

The most surprising result was the absence of spatial autocorrelation even between samples (transects) taken less than 75 m apart. This means that the faunistic dissimilarity between any pair of transects (at the within-locality scale) is independent of the between-transect distance. We attribute this result to (1) the noticeable horizontal mobility of the species considered (e.g. the sparids and *Mullus surmuletus*, Harmelin 1987); (2) the gregarious behaviour of some species of

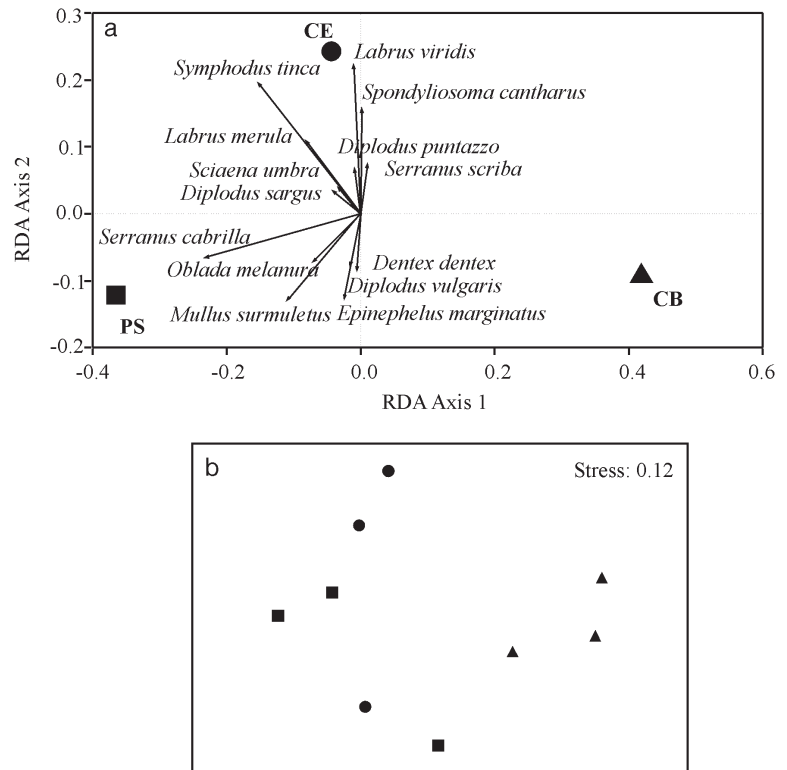


Fig. 4. (a) Redundancy analysis (RDA) and (b) non-linear multidimensional scaling (MDS) ordination diagrams. In the RDA bi-plot, the contribution of fish species to the differences between localities are also represented. The angle between arrows (species, Table 3) denotes the degree of correlation. Locality symbols (▲, ●, ■) can be projected perpendicularly onto the line overlaying the arrow of a particular species. These projected points fall within the predicted increase of the abundance of a particular species in the 3 localities. The MDS analysis corroborates the results obtained with the RDA. CB: Cavall Bernat; CE: Cala Extremes; PS: Punta de la Sal

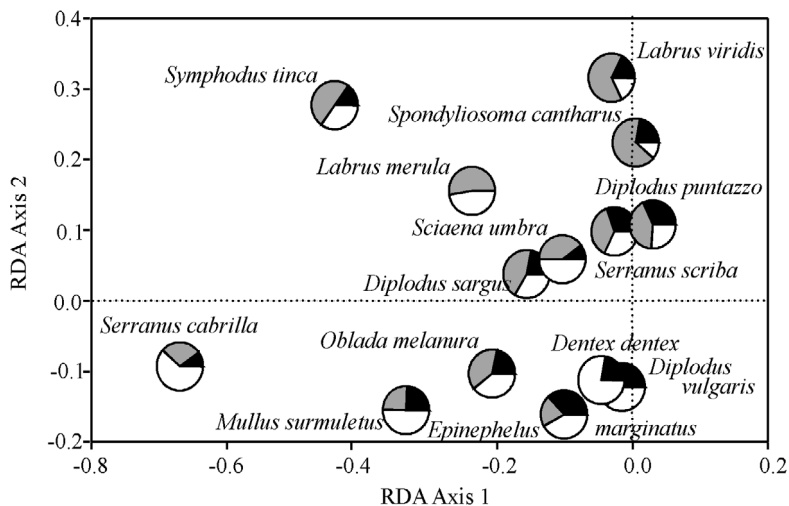


Fig. 5. Pie charts indicating the relative abundance of species at the 3 localities on the same ordination bi-plot shown in Fig. 4a (black: Cavall Bernat; grey: Cala Extremer; white: Punta de la Sal)

the family Sparidae (Quignard & Pras 1986, Corbera et al. 1996) and *Symphodus tinca* (Quignard & Pras 1986), that generates variability at the lowest spatial scale (between transects located close to each other); and (3) the very similar environmental set-up of all the samples. To deal with the last point, the localities and sampling time were *a priori* selected for their homogeneous orientation, wave exposure, water temperature, weather, depth, slope and underwater visibility. Habitat structure was fixed to some extent when we limited the sampling to rocky bottom areas. Variations in the bottom morphology were considered to be important factors affecting the spatial variability of fish assemblages (Luckhurst & Luckhurst 1978, Friedlander & Parrish 1998, García-Char-ton & Pérez-Ruzafa 1998, 2001); however, our results show significant between-locality differences for bottom structure but not for rugosity. This is due to the large RBWB coverage in one of the localities (CB). We suggest that, although the structures forming the bottom were not the same, the resulting habitat was very similar in all localities in terms of number of holes, crevices and vertical variations (i.e. rugosity). The following points sup-

port this assessment: (1) the bottom structure and rugosity were not detected to have any effect on the fish assemblage (at any of the below-locality scales, Table 5), and (2) the community descriptors (mean species number, mean total abundance and mean total biomass) did not show any significant variations at the within-locality level (Table 6). The variability of the bottom structure at the below-locality level (54.9%) shows that these tests were powerful enough and that the effects of bottom structure were not confused with the effects of fishing pressure. The high variability of bottom structure at the within-locality level and the fact that no significant differences in the community descriptors were found at this level supports the conclusion that, for the species studied, the differences found in bottom structure were not important enough to cause a relevant effect. Poor correlations between

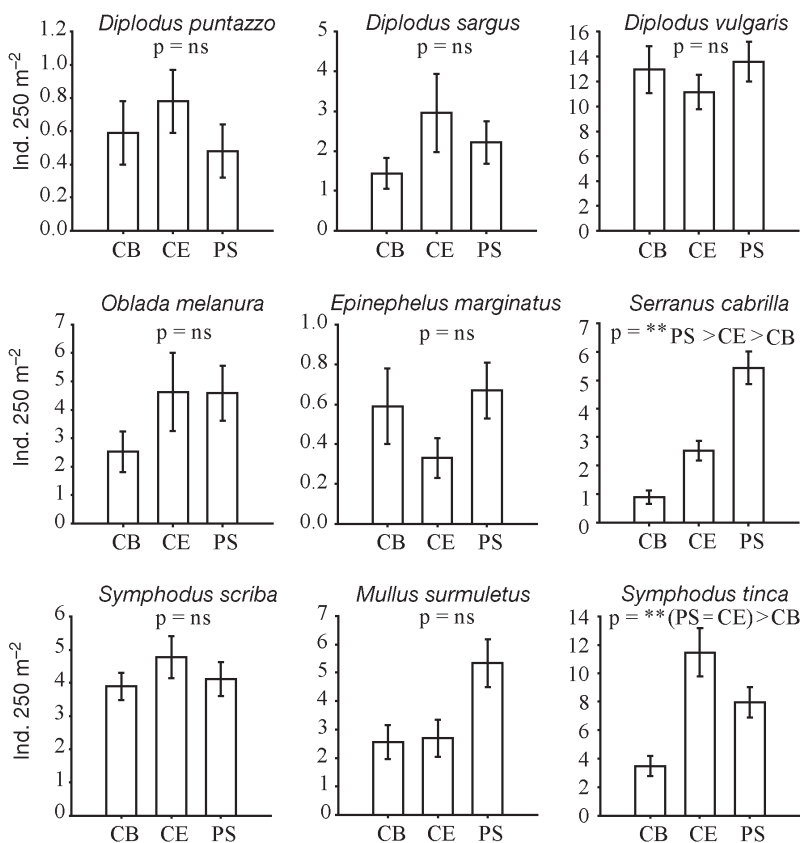


Fig. 6. Mean number of individuals (\pm SE) for the 9 most abundant vulnerable species. CB: Cavall Bernat; CE: Cala Extremer; PS: Punta de la Sal. **p < 0.001, ns: non significant

topography and fish abundances do not necessarily mean that this variable does not play an important role in species densities, especially when inadequate methods are applied (McCormick 1994). However, in our case, the method used is able to discern surface features of biological interest, such as shelter sites (McCormick 1994).

We paid special attention to wave exposure as it has been reported to be an important factor in structuring marine communities. This effect is especially evident when comparing exposed and sheltered habitats for both invertebrate benthic and fish assemblages (Gust et al. 2001, Micheli et al. 2005). The predominant waves to the north of Mallorca move in N, NNE and NE directions, which are also the highest waves. The 3 localities share a very similar N to NE orientation and hence, a complete exposure to the predominant waves in the area, which assures that wave exposure is not a differentiating factor for the 3 localities studied.

The vulnerable species studied have a high commercial value for the trammel net fishery, and they are also the main target species of recreational fishing in Mallorcan rocky bottoms (Morales-Nin et al. 2005). Fishing activity in the studied area peaks in summer when commercial fishing effort and recreational and spear fishing reach their maximum off the Island (Morales-Nin et al. 2005). CB is the most visited locality for recreational boats leaving from CSV and it also supports the largest trammel net fishery (within a depth range of 35 to 5 m). Moreover, recreational boats from PP that reach the northern coast predominantly fish in the area next to CB. Consequently, it could be expected that the resulting differences in fishing pressure would cause some changes in the fish assemblages.

The results obtained are in accordance with this scenario. Significant differences were found between localities in terms of the mean number of species and abundance, with higher values in the lower fishing pressure localities (CE and PS, Fig. 3 and Table 6).

Multivariate analyses (RDA) demonstrated highly significant differences between localities (Table 5). Moreover, MDS corroborated the large faunistic differences between CB and the other 2 localities. The abundance of vulnerable species was lower in the most fished locality (CB) (Fig. 4).

At the species level, some clear examples were found. *Serranus cabrilla* showed significantly higher values in both abundance and biomass (Figs. 6 & 7),

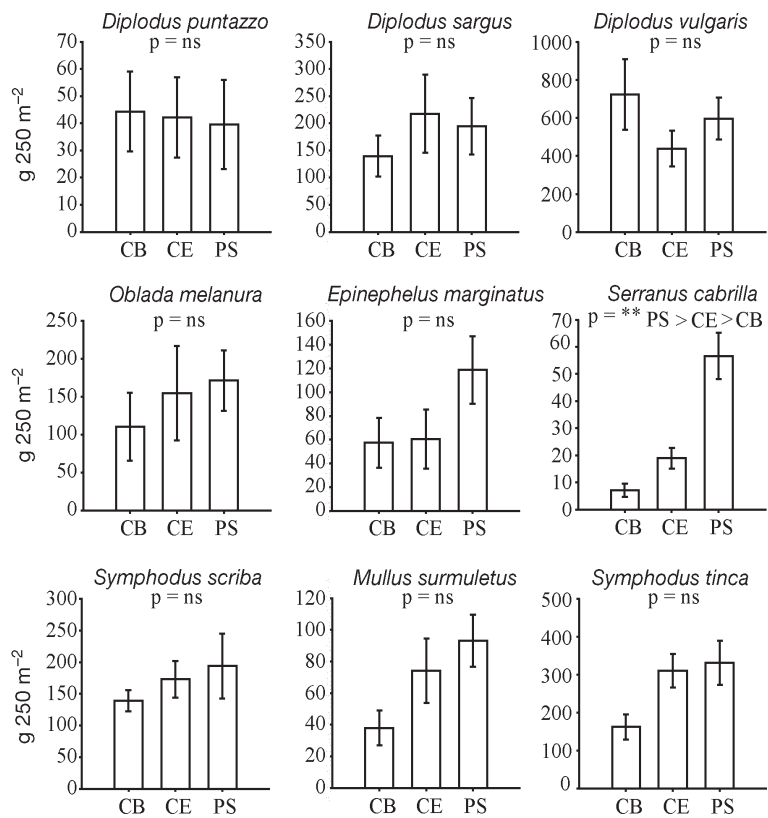


Fig. 7. Mean biomass (g ± SE) for the 9 most abundant species. CB: Cavall Bernat; CE: Cala Extremer; PS: Punta de la Sal. **p < 0.001, ns: non significant

and larger although non-significant mean length values in the lower fishing pressure localities. This species is considered to be a good indicator of angling pressure (Perez-Barroso et al. 1993) and is a major component of on boat recreational fishing in Mallorca (Morales-Nin et al. 2005). Our results are in accordance with those found in Carry-le-Rouet marine reserve (NW Mediterranean, Harmelin et al. 1995), where higher abundance and larger individuals were found inside the protected area. Moreover, to the north of Mallorca, not only were the values significantly higher in the localities with lower fishing pressure, but also this species seems to be positively affected by the longer distances from both recreational and commercial fisheries of PS. *Symphodus tinca* is another species commonly fished off Mallorca (Morales-Nin et al. 2005) that showed significant differences, but in this case only in abundance, although mean biomass was also higher in the localities with lower fishing pressure. *Mullus surmuletus*, which is a species mainly fished by trammel net and which has a high commercial value, reached larger although non-significant biomass values in the lower fishing pressure localities. Moreover, this species was the only

one which had significant differences in mean length values, with larger individuals in CE. The same trend in biomass, and higher although non-significant mean length values, in the lower fishing pressure localities were observed for *Epinephelus marginatus*, which showed a significant response to the protection measures established in the Mediterranean (García-Rubies & Zabala 1990, Coll et al. 1999, La Mesa & Vacchi 1999, Reñones et al. 1999). Generally, the sparid species follow the opposite trend (Table 3); however, only *Diplodus puntazzo* and *Oblada melanura* had the largest individuals in the higher fishing pressure locality. *D. sargus*, *O. melanura* and *Serranus scriba* had higher abundance and biomass values in the lower fishing pressure localities. Only *D. puntazzo* and *D. vulgaris* followed a different pattern, with the highest biomass recorded in the highest fishing pressure locality. However, results concerning *D. vulgaris* should be interpreted with caution due to its gregarious behaviour (Corbera et al. 1996) and horizontal mobility (Harmelin 1987). These 2 characteristics can cause biases when performing UVC (Harmelin-Vivien et al. 1985, Kulbicki 1998). *D. vulgaris* is the dominant species in terms of biomass in all localities (Fig. 7).

The mean length and mean maximum length of all species collectively, were not smaller at the sites supporting larger fishing pressure. However, these community descriptors depend heavily on the relative abundance of species and the faunistic composition. In this sense, no specific trends were detected due to the replacement of the most important species in relative abundance and length (mean and maximum) between localities. Moreover, these community metrics are not good short-term indicators of fishing pressure (Nicholson & Jennings 2004).

To conclude, the present study provided results that clearly support the assumption that the assemblages of vulnerable species in open fishing areas respond negatively to fishing pressure. This effect is evident even at a relatively small spatial scale (5 to 10 km). Moreover, the observed pattern in terms of a fishing pressure trend emulated what normally happens after marine protected areas are established.

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