

Evidence of trawl disturbance on mega-epibenthic communities in the Southern Tyrrhenian Sea

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ABSTRACT: Bottom trawling has direct impacts on benthic communities. These impacts are modified by the environmental context in which they occur. Communities that occur in habitats subjected to low levels of natural disturbance are considered to be the most vulnerable to bottom trawling. The present study examined the impact of otter trawl fisheries on epifaunal assemblages of the Southern Tyrrhenian Sea (Central Mediterranean Sea), across the continental shelf and down to the meso-bathyal plain. Using a long time-series of non-target epifaunal species collected by experimental otter trawl surveys, differences in assemblage structure and composition were detected among areas that experienced different levels of fishing intensity relative to Fishery Exclusion Zones. Areas that experienced the highest levels of trawling had a significantly lower abundance of crinoids and ophiuroids; the former are important structuring fauna. The epifaunal assemblage composition in the areas that had been subjected to the heaviest levels of fishing activity were characterised by a greater number of scavenging species that are resilient to the effects of fishing. Although the heavily trawled areas had higher diversity than the less heavily fished areas, this was associated with the absence of *K*-selected species such as crinoids and dominance by *r*-selected scavenging biota.

KEY WORDS: Trawling · Impact · Disturbance · Crinoids · Continental shelf · Continental slope

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INTRODUCTION

Fishing activities have led to widespread ecosystem changes in shelf seas worldwide (Jennings & Kaiser 1998, Bergman & van Santbrink 2000, Kaiser et al. 2002). The effects of bottom fisheries on benthic marine ecosystems are widely recognised and include habitat alteration, incidental mortality of non-target species, evolutionary shifts in population demographics and changes in the function and structure of ecosystems (Dayton et al. 1995, Kaiser & Spencer 1996, Auster & Langton 1999, Kaiser & de Groot 2000, Kaiser et al. 2002, Pikitch et al. 2004). A detailed understanding of the response of community components to fishing activities is necessary for effective management. Since the effects of fishing on benthic

habitat vary with the geology of the seabed and with the local natural disturbance regime, case-studies from a wider range of habitats are needed to refine model predictions of the consequences of bottom fishing disturbance (Jennings & Kaiser 1998, Hiddink et al. 2006, Kaiser et al. 2006).

The majority of studies of the impact of fishing on the seabed have been undertaken in continental shelf coarse-sediment habitats (for a comprehensive review, see Kaiser et al. 2006), whereas a much smaller proportion of studies have been undertaken in muddy sediments (Tuck et al. 1998, de Juan et al. 2007, Sánchez et al. 2007, Hinz et al. 2009). This lack of focus on mud habitats is surprising, given that muddy seabeds cover >50% of the earth's surface (Gage & Tyler 1991). In addition, the fauna that

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inhabit mud habitats fulfill important ecosystem services with respect to the cycling of organic carbon and nutrients (Widdicombe et al. 2004). Mud habitat systems sustain commercially important species, and hence are frequently subjected to intensive trawling in some areas of the world (Jennings et al. 2000). Furthermore, few studies exist of the impacts of fishing below a depth of 200 m.

In the Mediterranean Sea, the most important commercial fisheries operate over mud sediments and are characterised by bottom trawling (Caddy 1993, Farrugio et al. 1993). Otter trawls are the most common towed demersal fishing gear used in Mediterranean fisheries, targeting species that live in close proximity to the seabed. Otter trawls are towed such that the doors and groundrope remain in close contact with the seabed and thereby have an impact on the benthic habitat, particularly the epibenthic species that protrude above the sediment or that move about on the surface of the sediment (Kaiser & de Groot 2000, Hinz et al. 2009).

Previous studies in the Mediterranean basin have focused on demersal assemblages of fish and crustacea (Abelló et al. 2002, Kallianiotis et al. 2004, Ragonese et al. 2005, Dimech et al. 2008), whereas few have focused on the benthic assemblages that are associated with mud seabeds (Sanchez et al. 2000, Smith et al. 2000, de Juan et al. 2007, 2011). Some studies have attempted to relate demersal assemblages to the structuring role of macroepibenthic communities for the Mediterranean basin (Colloca et al. 2003, Massutí & Reñones 2005, Sánchez et al. 2007, Ordines & Massutí 2009), but information about these organisms is non-existent for the Southern Tyrrhenian Sea.

To date, the majority of studies designed to evaluate the impacts of bottom fishing gear have been conducted under experimental conditions. Such studies are limited in that they evaluate only the short-term effects of otter trawling activities on a small spatial scale. The predicted effects of fishing based on experimental studies need to be tested under realistic conditions and on a scale relevant to a commercial fishing ground. The present study moves in this direction by focusing on the Southern Tyrrhenian Sea bottom trawl fishery, where there remains a persistent high level of exploitation despite the wide range of management measures adopted in the distant past and more recently. These measures include limited license entry to the fishery, the seasonal closure of bottom trawl fishing (adopted on a voluntary basis at first and becoming mandatory during the last 18 yr, with variable compliance within the

study area), the cod end mesh-size regulation (often deregulated and almost never respected, particularly in the past), permanent area closures to certain fishing gear such as bottom trawls, restrictions on fishing within a certain distance from the coast and within the 50 m depth contour, engine power restrictions, minimum landing sizes, the establishment of fishery exclusion zones, the creation of artificial reefs and decommissioning or buy-back programmes to reduce fishing capacity (Reg. EC 2792/1999, Reg. EC 1543/2000, SEC(466) 2007).

The aim of the present study was to characterise and describe the main mega-epibenthic communities that inhabit mud habitats that are subject to different intensities of fishing activities along the coast of the Southern Calabrian and Sicilian Tyrrhenian Sea between depths of 14 and 650 m. Further, the study focussed on the potential long-term effects of otter trawling on the communities and evaluated their spatial and temporal variation as a response to fishing intensity. For this purpose, a quantitative historical time-series of non-target epibenthic species collected by experimental otter trawl surveys (1996–2001 and 2008–2010) was analysed in order to quantify changes in megafaunal abundance, species richness and community composition in response to temporal and spatial fluctuations of the fishing intensity in the Southern Tyrrhenian Sea.

MATERIALS AND METHODS

Study area

The study area extends over 7256 km² from Cape Suvero (Calabrian coast) to Cape San Vito (Sicilian coast), within the 700 m isobath. The survey area, part of the ongoing MEDITS trawl survey programme (MEDiterranean International Trawl Survey), has been identified since 2002 as part of the Geographical Sub-Area 10 by the General Fisheries Commission for the Mediterranean (GFCM) (MEDITS 2007) (Fig. 1). Before this date, the study area was identified as the sub-region M3b, part of the M3 region that also included the Sicily Channel (Bertrand et al. 2002). Survey frequency and intensity varied annually, such that the number of annual survey hauls was reduced from 28 to 23 from 2003 onwards.

The Southern Tyrrhenian Sea is an area with complex dynamics in terms of water exchange, and provides an important biological axis between the sub-basins of the Eastern and Western Mediterranean

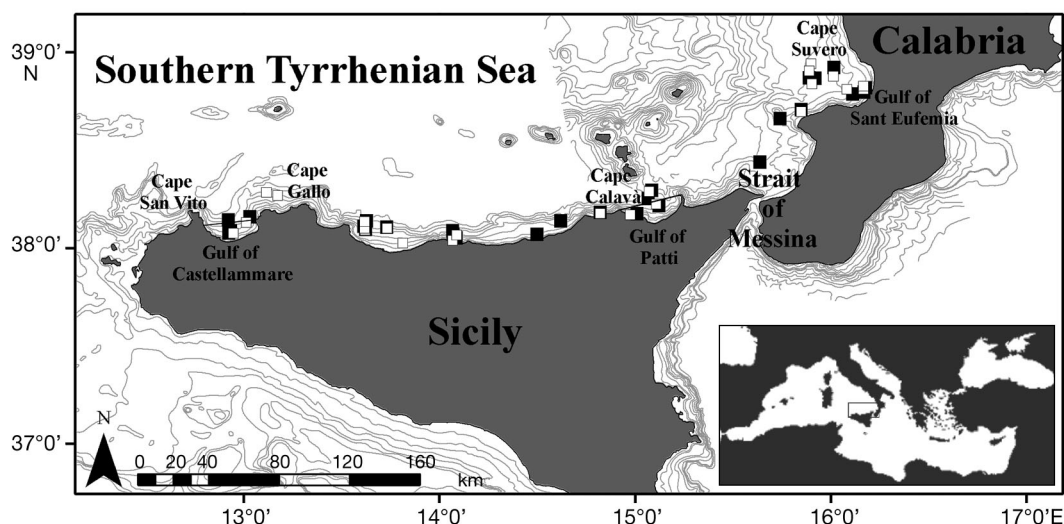


Fig. 1. Study area with the main isobaths and the sampling stations investigated during the MEDITS trawl survey. The black squares show the sites sampled from 1996, while the white squares show the sites sampled after 2002

Sea. Primary production and sedimentary regimes in the study area fluctuate seasonally due to terrigenous-continental inputs from storm related fresh-water discharge (Amore et al. 1995, Sarà et al. 1998). The continental shelf is narrow, often reduced to a coastal fringe and sometimes entirely absent, a condition that increases the dilution of the land runoff and the related nutrient supply with consequent oligotrophy of both coastal and offshore water masses (De Domenico & Pulicanò 1983, Casotti et al. 2000). Generally, the seabed is characterised by fine sand and muddy sediment deposits dominated by soft bottom communities (Pérès 1985). Two Fishery Exclusion Zones are located along the Sicilian coast, from which trawling has been banned since 1990: the Gulf of Castellammare and the Gulf of Patti (Regional Law No. 25/1990). The first gulf is subject to a 200 km² ban, and the second is entirely banned. Fishing inside the 2 gulfs is restricted to artisanal fishing that extends over the continental shelf and upper slope, employing mainly size-selective static gears such as trammel nets and set gillnets. In areas that remain open to trawling, this activity is authorised at depths >50 m according to Italian legislation. The main demersal resources that represent the bulk of the commercial landings are the 2 red shrimp species *Aristaeomorpha foliacea* (Risso, 1982) and *Aristeus antennatus* (Risso, 1816), the deep-water rose shrimp *Parapenaeus longirostris* (Lucas, 1847), the Norway lobster *Nephrops norvegicus* Linnaeus, 1758, the hake *Merluccius merluccius* (Linnaeus, 1758) and the red mullet *Mullus barbatus* Linnaeus, 1758. Despite

the diversity of small-scale fisheries that characterised the study area, otter trawlers contribute most to the production and overall landed product value (IREPA 2009).

Data collection

An annual index of fishing effort was calculated as the product of fishing capacity and fishing activity following the EU definition (Reg. EC 2091/98). Fishing capacity was measured in terms of 2 vessel characteristics; gross tonnage (GT) and engine power (kW). These 2 measures were derived by consulting the registers of the local marine offices located along the coast of the study area. Concomitantly, other information was collected such as number of vessels working in the study area, registration number, year of commissioning and type of fishing license. Data on fishing activity, expressed as the average number of fishing days per year by the vessels of each of the fleets, were derived from the IREPA database (produced in the context of the programme 'Economic observatory on the productive structures of Italian marine fisheries'). Data regarding the extent of the area fished by each of the fleets were obtained from local fishermen's associations through direct interviews. These interviews enabled the identification of the main areas where the fleets' segments were usually located and operated. Fishing effort, evaluated with the 2 parameters of fishing capacity (mean GT and mean kW), was tested using the BIOENV

procedure (Clarke & Ainsworth 1993) in order to determine which parameter had the greatest influence on the biotic communities. Data on trawlers registered and operating in the study area since 1996 were used to obtain measures of annual fishing intensity (fishing effort as *GT*-days per unit of area per year) exerted over the last 15 yr in order to observe the evolution of this index. The fishing intensity index was estimated for the depth zone between 50 and 700 m considering the main fishing areas where the otter trawlers operate, calculated using GIS software (ArcMap 9.3™, ESRI). In order to understand any changes in fishing intensity with time and area, a 2-way ANOVA was performed with year and location set as factors. However, as we were primarily interested in the response of the benthic epifaunal assemblage to variation in fishing intensity and area, time was not considered to be an explanatory variable in further analyses as time per se does not have an effect on epifaunal assemblage structure.

Epifaunal samples were collected as part of the otter trawl survey carried out in the study area in May to July from 1996 to 2001 and from 2008 to 2010, within the framework of the MEDITS project. To conform to the MEDITS sampling protocols (MEDITS 2007), the surveys were carried out according to a depth-stratified sampling design with randomly allocated hauls within each of 5 bathymetric strata: Stratum A: 10–50 m, inner shelf; B: 50–100 m, outer shelf; C: 100–200 m, shelf break; D: 200–500 m, upper slope; E: 500–800 m, upper middle slope. Samples were collected using an experimental otter trawl net (IFREMER GOC 73) (Fiorentini et al. 1999). Seawater bottom temperature was measured using a temperature probe (Minilog, Vemco) attached to the net. The present study included data from 197 separate hauls from depths that ranged from 14 to 650 m (Fig. 1).

The entire faunal component from each haul was sorted, identified, weighed and counted on board ship as routinely required by the MEDITS project. In this study we analysed only the non-target epibenthic component of the animal community incidentally caught by the net and considered as commercially unimportant species. In the context of the present study, the epibenthic species were preserved in 70% ethanol on board and were counted and identified to the lowest taxonomic level in the laboratory. Only crustaceans, echinoderms and molluscs were included in the statistical analysis.

Sediment samples were collected using a 70 l modified Van Veen grab with a sampling surface area of 0.25 m². The particle size fraction was determined according to the procedure described by Buchanan

(1984). Statistical sediment descriptors, including mean sediment grain size, sediment sorting, skewness and kurtosis were calculated and data were categorised into the following sediment fractions: sand, silt and clay (Folk & Ward 1957). Data derived for macrofauna collected by grab were not used in the present study.

Data analyses

Multivariate analysis of the community data was undertaken using Primer v.6 (Clarke & Warwick 1994a) considering only species taxonomically recognized with consistency and accuracy (crustacean, echinoderms and molluscs). The total number of individuals was standardised per km² (Density Index, *DI*) and species whose percentage abundance was less than 5% of the total dataset were removed from the analyses, as they were not consistently sampled by the trawl gear. The obtained dataset was square-root transformed prior to analysis, in order to reduce the influence of dominant species and increase the importance of less common species. A similarity matrix was computed using the Bray-Curtis similarity index (Clarke & Warwick 1994a). Similarities among sampled hauls were visualised using multi-dimensional scaling (MDS) plots. The similarity percentage (SIMPER) procedure was used to determine which species contributed most to the similarity or dissimilarity within groups defined *a priori* on the basis of the depth stratum from which they were sampled (Clarke & Warwick 1994b). The analysis of similarities (ANOSIM) routine was used (Clarke & Warwick 1994b) to test for differences in assemblage composition that occurred with depth (strata) and time (yr) to understand the spatial and temporal differences between the predefined depth strata.

Multivariate analysis was conducted on the main environmental parameters measured: depth, near-bottom temperature and sediment characteristics. All variables were normalised before analysis and a Euclidean distance matrix was constructed. The relationships between environmental variables were assessed using Pearson's correlations, and Bonferroni's adjustment was applied to counteract the problem of multiple comparisons. The influence of the measured abiotic parameters on assemblage structure was determined using the BIOENV procedure (Clarke & Ainsworth 1993).

Univariate indicators, including the number of individuals per km² (density index, *DI*), total number of species per sample (*S*), Margalef's index of species

richness (d), Shannon-Wiener index (H') and Pielou's index of evenness (J'), were calculated for each of the sampled hauls. Abundance data were used to calculate the diversity indices. The indicators were calculated for the assemblage from each *a priori* defined depth stratum. Biomass data were not available for analysis.

An ANOVA was conducted in order to detect significant differences in the univariate indicators among different strata. Prior to performing the ANOVA test, all the univariate indices were tested for normality and homogeneity of variance using Anderson-Darling and Levene's tests respectively. The 1-way ANOVA test was performed on $\log_{10}(x + 1)$ -transformed data for the density index and the total number of species, and on non-transformed data for the species richness and Shannon-Wiener indices. Pielou's evenness index was tested using the non-parametric Kruskal-Wallis test. Post hoc Tukey's tests were used to detect significant pairwise differences among the strata.

A second set of analyses was undertaken to ascertain the relationship between community composition and fishing intensity. The analyses were conducted using a reduced set of data, excluding data coming from Stratum A (10–50 m), which was not subject to fishing activities due to legal restrictions (Cacaud 2005). An ANOSIM test was used to find detailed differences in epibenthic species composition among differing fishing intensities. Considering the results obtained after the analysis of fishing intensity within the study area, the same test was performed separately inside each stratum, considering the 3 different fishing grounds as a factor.

The relationship between environmental variables and trawling intensity was analysed using Pearson's correlations, to determine whether fishing intensity was correlated with a specific environmental descriptor. The influence of fishing intensity on the composition of epibenthic communities was determined using the BIOENV procedure (Clarke & Ainsworth 1993). Effects of fishing intensity and depth strata factors were detected on biodiversity indices using a 2-way ANOVA. All data were $\log_{10}(x + 1)$ -transformed to satisfy the assumptions required by ANOVA, except mean species richness and mean Shannon-Wiener diversity.

Finally, a 1-way ANOVA was performed for species that contributed most to the percentage similarity within each stratum to uncover any differences that occurred with fishing intensity. For this purpose, fishing intensity values were grouped into 5 categories: Class 0: no fishing effort; Class 1: 1–500, Class

2: 501–700, Class 3: 701–900 and Class 4: 901–1100 tonne-days per kilometre squared per year ($t\ d\ km^{-2}\ yr^{-1}$). A significance level of 0.05 was used in all statistical tests, and post hoc analyses of means were undertaken using Tukey's test to detect significant pairwise differences among the levels of fishing intensity.

RESULTS

Fishing intensity

The BIOENV procedure indicated that GT was the best descriptor of fishing capacity in relation to epibenthic community composition and structure ($GT\ r_s$ [Spearman correlation] = 0.179 c.f. $kW\ r_s$ = 0.104). Accordingly, GT was used as a fishing capacity parameter in the evaluation of fishing intensity.

The analysis of fishing intensity across the study area identified 3 principal fishing grounds that were subject to varying degrees of fishing intensity. The western extent from Cape San Vito to Cape Gallo (a total of 554 km^2) was characterised by higher values of fishing intensity, the central area from Cape Gallo to Cape Calavà (1431 km^2), had medium values of fishing intensity and the eastern area from Cape Calavà to Cape Suvero (2625 km^2), was subjected to the lowest values of fishing intensity (Fig. 2a). The pattern of fishing intensity between each of the fishing grounds was generally constant across the historical time-series observed (1996–2010). From 2005 to 2008, fishing intensity was higher in the central area; nevertheless, the values were noticeably greater only in 2006 and 2007 compared to the western area. Fishing intensity declined continuously in the western fishing ground, and since 2007 in the central area. This same trend occurred for fishing capacity (Fig. 2b).

Fishing activity in the trawl fleet was largely stable across the study period (Fig. 2c). The number of bottom trawl vessels operating from the associated main harbours in each of the 3 fishing grounds showed a general constant trend, with some higher values registered for the central fishing ground; however, these existed only for a short period of time (Fig. 2d).

A 2-way ANOVA showed that within fishing grounds, fishing intensity remained constant with time, but that differences in fishing intensity occurred between the fishing grounds factor ($F_{2,29} = 53.32$, $p < 0.001$). As temporal variation in fishing intensity did not vary significantly, we focused our analyses on comparing the responses of the epiben-

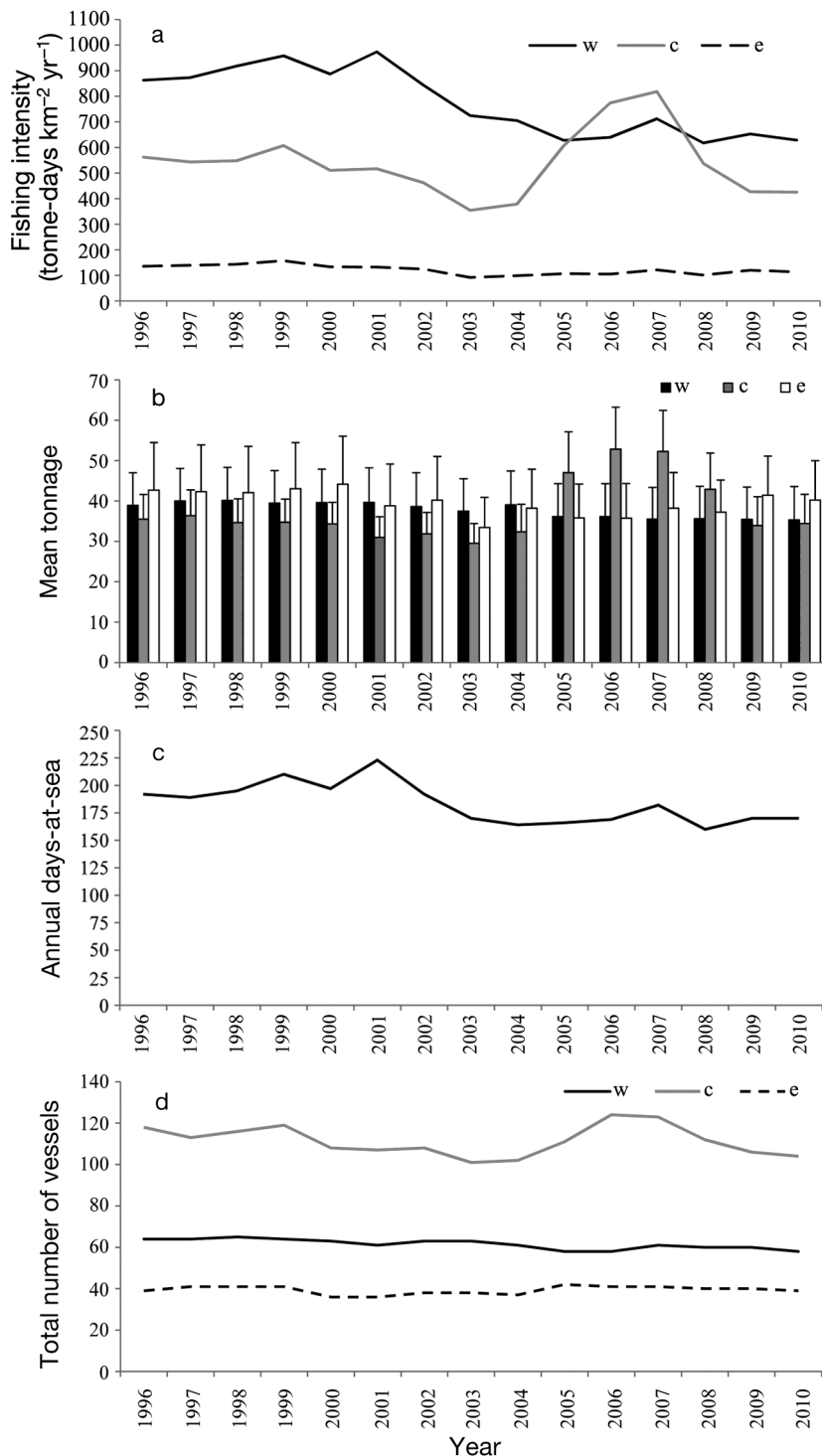


Fig. 2. (a) Fishing intensity values inside each of the 3 major fishing grounds inside the study area (fishing intensity = fishing capacity km⁻² yr⁻¹ × fishing activity km⁻² yr⁻¹ = tonne-days km⁻² yr⁻¹). (b) Mean annual values of fishing capacity inside each fishing grounds (gross mean tonnage of the vessels ± 95 % confidence limits). (c) Fishing activity in terms of annual average of days at sea for each year of investigation in the study area for the whole fleet. (d) Number of vessels operating inside the 3 different fishing grounds after considering the effective spatial fleet distribution inside the fishing area. w: western fishing ground from Cape S. Vito to Cape Gallo; c: central fishing ground from Cape Gallo to Cape Calavà; e: eastern fishing ground from Cape Calavà to Cape Suvero

thic communities to the different levels of fishing intensity experienced in each of the fishing grounds.

Epifaunal communities

A total of 88 species (43 decapods, 25 echinoderms, 13 molluscs, 6 anthozoans and 1 brachiopod) were identified in the study area over the entire period of the study (Appendix 1). In general, decapod crustaceans were the dominant taxon in terms of the number of species collected, while echinoderms were dominant in terms of density.

Multivariate analysis of the complete epibenthic dataset showed that there were significant differences in the assemblage composition among all the different depth strata (ANOSIM test $R = 0.51$, $p < 0.001$) (Table 1). The MDS plot indicated that samples from strata D and E, respectively the upper slope and upper middle slope, were strongly clustered together and separate from the other strata. The samples from depth categories A–C (continental shelf: A, inner shelf; B, outer shelf; C, shelf break) were more transitional, with a greater level of similarity among the assemblages from these 3 depth strata (Fig. 3, Table 1). There was no inter-annual variation in community structure and assemblage identity among years (ANOSIM test $R = 0.046$, $p > 0.05$). The analysis of the contribution of similarity (SIMPER) showed that only a few species contributed to the overall similarity within each group and that there were clearly dominant species, such as the asteroid *Astropecten irregularis* (Pennant, 1977) and the pagurid *Pagurus excavatus* (Herbst, 1791) (Table 2). These 2 species have wide-ranging depth distributions, but differed in their percentage contribution to community structure for each stratum. The only mollusc that was sampled consistently by the trawl gear was the bivalve *Acanthocardia*

aculeata (Linnaeus, 1758), which was found only in the outer shelf stratum. Decapod crustaceans contributed most to the similarity among sample hauls within the upper middle slope.

A summary of the mean environmental parameters evaluated in each of the 5 pre-defined depth strata are reported in Table 3. A BIOENV analysis revealed that the environmental factor that best explained the

Table 1. Pairwise tests displayed by ANOSIM procedure performed using the factor strata (all R values were significant at $p < 0.001$). Stratum A: 10–50 m, inner shelf; Stratum B: 50–100 m, outer shelf; Stratum C: 100–200 m, shelf break; Stratum D: 200–500 m, upper slope; Stratum E: 500–800 m, upper middle slope

| Strata | A | B | C | D |
|--------|-------|-------|-------|-------|
| B | 0.197 | | | |
| C | 0.305 | 0.191 | | |
| D | 0.491 | 0.407 | 0.439 | |
| E | 0.719 | 0.725 | 0.757 | 0.483 |

structuring within the epibenthic communities was depth ($r_s = 0.527$). Depth was also significantly correlated with all the environmental parameters, except the percentage of clay (Table 4). Near-bottom temperature was correlated with all the 4 granulometry descriptors analysed (mean particle size, sorting, skewness and kurtosis indices), and with percentage of sand. The mean grain size was correlated with all sediment descriptors and fractions except with the silt percentage and kurtosis index. The 4 granulometric descriptors were highly correlated with the percentages of sand and of clay. Moreover, the skewness index was correlated with the sorting index. In general, all the percentages of granulometric fractions were highly interrelated. These results showed clearly that depth was associated with variations in sediment type that changed from sand dominated sediment near the coast to finer sediment in the deeper strata. The greater percentage of the sand fraction was found mostly along the continental shelf, particularly on the coastal strata: inner and outer shelf. The percentage of clay was greater in the shelf

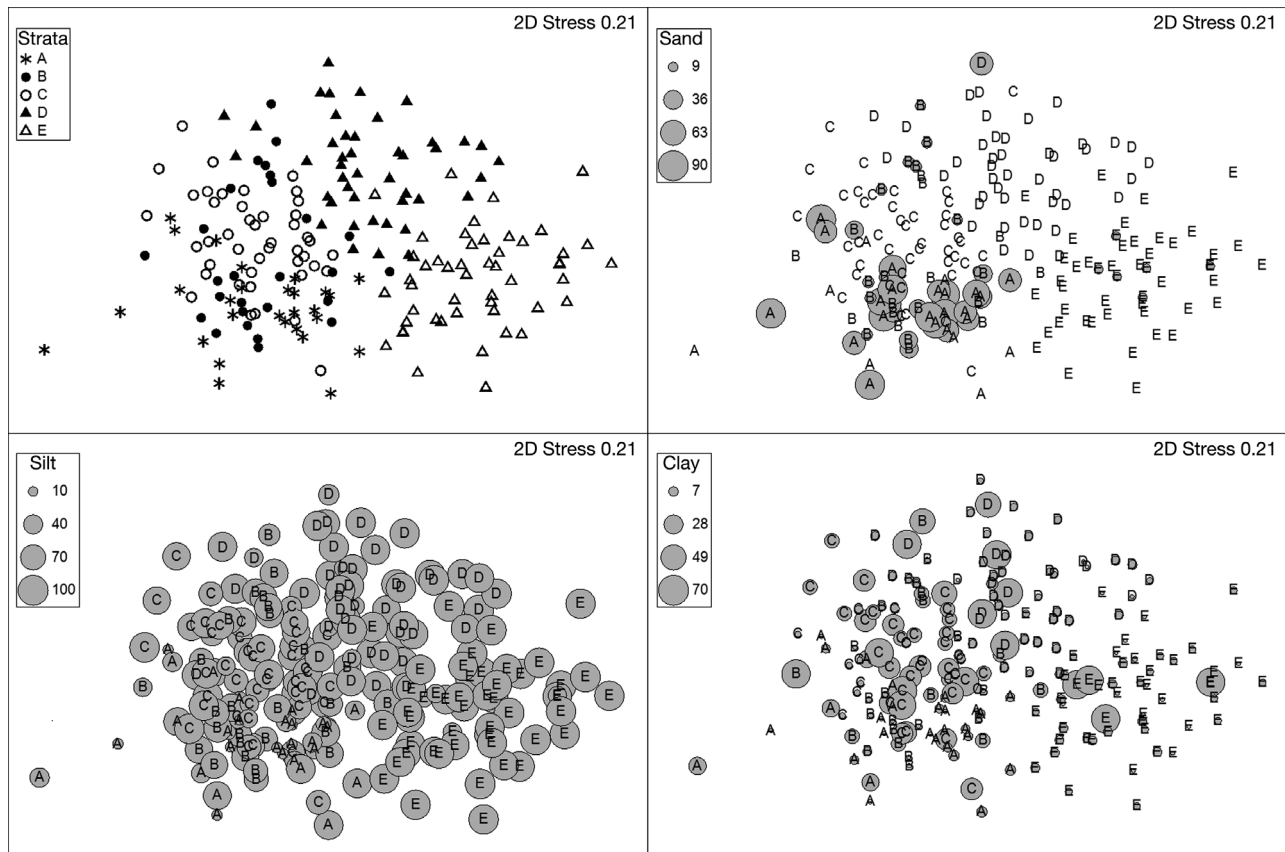


Fig. 3. MDS ordination plot for the epifaunal samples collected during the entire period of investigation based on density index. Graphical representations of scaled values of percent sand, clay and silt on the same MDS plot. Strata: A (10–50 m), inner shelf; B (50–100 m), outer shelf; C (100–200 m), shelf break; D (200–500 m), upper slope; E (500–800 m), upper middle slope

Table 2. Output from a SIMPER analysis showing those species that contributed to 90% of the similarity for sites within each of the different strata. The analysis was performed on a reduced data set (without Stratum A where fishing was not permitted). For each of these species, 1-way ANOVA test results (F) are shown that examine the effects of fishing intensity, with the associated df and p . Fishing intensity values were grouped into 5 categories: 0: no fishing intensity; 1: 1–500; 2: 501–700; 3: 701–900; 4: 901–1100 tonne-days ($t\ d\ km^{-2}\ yr^{-1}$). Post hoc multi-comparison Tukey's test results are presented; asterisks indicate different p -values: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. Contrib %: percentage contribution; Cr: Crustacea; Ec: Echinodermata; Mo: Mollusca

| | Av. density ($N\ km^{-2}$) | Contrib % | F | df | p | Tukey's test | Code |
|---------------------------------------|------------------------------|-----------|-------|------|-----------------|--------------|------|
| Stratum B (Outer shelf) | | | | | | | |
| Average similarity: 25.29 | | | | | | | |
| <i>Astropecten irregularis</i> | 275.17 | 37.89 | 0.58 | 3 | 0.635 | – | Ec |
| <i>Ophiura ophiura</i> | 65.90 | 13.15 | 4.85 | 3 | <0.05 | 0 > 1, 2*** | Ec |
| <i>Astropecten bispinosus</i> | 73.23 | 10.75 | 6.6 | 3 | <0.01 | 0 < 2, 3* | Ec |
| <i>Antedon mediterranea</i> | 141.62 | 9.31 | 4.34 | 3 | <0.05 | 1 > 2* | Ec |
| <i>Parastichopus regalis</i> | 17.53 | 5.44 | 2.71 | 3 | 0.072 | – | Ec |
| <i>Liocarcinus depurator</i> | 48.77 | 4.32 | 5.45 | 3 | <0.01 | 1 < 3** | Cr |
| <i>Medorippe lanata</i> | 35.73 | 4.15 | 3.68 | 3 | <0.05 | 1, 2 < 3* | Cr |
| <i>Tethyaster subinermis</i> | 18.39 | 2.12 | 0.7 | 3 | 0.561 | – | Ec |
| <i>Acanthocardia aculeata</i> | 10.51 | 2.09 | 1.45 | 3 | 0.259 | – | Mo |
| Stratum C (Shelf break) | | | | | | | |
| Average similarity: 30.26 | | | | | | | |
| <i>Ophiura ophiura</i> | 200.86 | 8.21 | 0.69 | 3 | 0.568 | – | Ec |
| <i>Astropecten irregularis</i> | 141.36 | 7.42 | 0.68 | 3 | 0.574 | – | Ec |
| <i>Leptometra phalangium</i> | 99.74 | 4.19 | 5.27 | 3 | <0.05 | 1 > 3* | Ec |
| <i>Parastichopus regalis</i> | 50.74 | 3.05 | 13.15 | 3 | <0.01 | 1 > 3* | Ec |
| <i>Astropecten bispinosus</i> | 117.20 | 2.76 | 3.73 | 3 | <0.01 | 0 < 2* | Ec |
| <i>Liocarcinus depurator</i> | 47.71 | 2.24 | 7.96 | 3 | <0.01 | 1 < 3* | Cr |
| Stratum D (Upper slope) | | | | | | | |
| Average similarity: 22.82 | | | | | | | |
| <i>Gracilechinus acutus</i> | 92.22 | 37.76 | 2.29 | 4 | 0.099 | – | Ec |
| <i>Astropecten irregularis</i> | 33.37 | 15.83 | 1.7 | 4 | 0.195 | – | Ec |
| <i>Macropipus tuberculatus</i> | 33.76 | 11.2 | 13.01 | 4 | <0.05 | 1 < 3* | Cr |
| <i>Munida intermedia</i> | 12.85 | 9.41 | 1.44 | 4 | 0.261 | – | Cr |
| <i>Pagurus excavatus</i> | 9.26 | 8.95 | 1.42 | 4 | 0.269 | – | Cr |
| <i>Polycheles typhlops</i> | 5.56 | 3.09 | 11.6 | 4 | <0.05 | 1 < 3* | Cr |
| <i>Tethyaster subinermis</i> | 4.14 | 2.86 | 0.45 | 4 | 0.771 | – | Ec |
| <i>Parastichopus regalis</i> | 4.70 | 2.69 | 3 | 4 | <0.05 | 0 > 2* | Ec |
| Stratum E (Upper middle slope) | | | | | | | |
| Average similarity: 29.70 | | | | | | | |
| <i>Polycheles typhlops</i> | 96.14 | 55.89 | 1.25 | 3 | 0.322 | – | Cr |
| <i>Geryon longipes</i> | 30.89 | 16.67 | 2.38 | 3 | 0.103 | – | Cr |
| <i>Pasiphaea</i> sp. | 372.04 | 12.28 | 0.25 | 3 | 0.861 | – | Cr |
| <i>Pagurus excavatus</i> | 22.96 | 8.09 | 0.92 | 3 | 0.452 | – | Cr |

break and on the upper slope, while a high silty fraction was found fairly evenly in each stratum. The highest concentrations of silt were attained on the slope strata (Fig. 3).

The mean density index and biodiversity indices for the epibenthic species differed significantly among the depth strata groups, except for Pielou's index of evenness (Tables 3 & 5). In general, the univariate parameters decreased with depth, and the higher values were found on the outer shelf. Tukey's post hoc tests showed that the mean density index and the mean total number of species for the 3 shallower strata were significantly different from

the 2 deeper strata (Fig. 4). The species richness index showed differences between the inner shelf and the upper slope and upper middle slopes, and similarly, between the shelf break and the upper slope and upper middle slopes. Species richness was highest on the outer shelf and lowest on the upper middle slope. The Shannon-Wiener index revealed differences between the 2 shallower continental shelf strata and the 2 deeper slope strata, and also between the shelf break and upper slope. The mean density index had higher values on the outer shelf. The mean total number of species decreased with depth.

A second analysis was undertaken on a reduced biotic matrix to test the effects of fishing intensity for only those depth strata that were subjected to trawling activity (51–700 m). Multivariate analysis performed on this reduced dataset, excluding the shal-

lower stratum (10–50 m), showed significant differences in the community composition among the 4 depth strata ($R = 0.53$, $p < 0.001$). An ANOSIM test revealed that significant differences in community structure occurred with fishing intensity ($R = 0.179$,

Table 3. Summary of the mean environmental parameters and univariate indicators analysed for each stratum group defined by depth and confirmed by univariate analysis of data (mean \pm 95 % confidence limits). Stratum A: 10–50 m, inner shelf; Stratum B: 50–100 m, outer shelf; Stratum C: 100–200 m, shelf break; Stratum D: 200–500 m, upper slope; Stratum E: 500–800 m, upper middle slope; w: Western, c: Central, e: Eastern fishing grounds

| | A | B | C | D | E |
|--|--------------------------|--------------------------|---------------------------|---------------------------|--------------------------|
| Number of hauls | 28 | 27 | 44 | 49 | 49 |
| Number of hauls by fishing ground | w = 4 c = 20 e = 4 | w = 8 c = 11 e = 8 | w = 12 c = 27 e = 5 | w = 7 c = 26 e = 24 | w = 8 c = 9 e = 32 |
| Environmental parameters | | | | | |
| Depth (m) | 34.0 \pm 3.5 | 71.1 \pm 4.2 | 124.1 \pm 6.8 | 352.3 \pm 17.8 | 567.0 \pm 10.5 |
| Temperature near bottom ($^{\circ}$ C) | 17.2 \pm 0.6 | 15.2 \pm 0.4 | 14.6 \pm 0.1 | 14.4 \pm 0.2 | 14.0 \pm 0.1 |
| Median particle size (μ m) | 5.1 \pm 0.5 | 6.1 \pm 0.3 | 6.8 \pm 0.2 | 6.9 \pm 0.2 | 6.8 \pm 0.2 |
| Skewness (<i>Sk</i>) | 0.2 \pm 0.1 | −0.2 \pm 0.1 | −0.2 \pm 0.0 | −0.2 \pm 0.0 | −0.1 \pm 0.0 |
| Sediment sorting (<i>s</i>) | 1.3 \pm 0.1 | 1.3 \pm 0.1 | 1.0 \pm 0.1 | 0.9 \pm 0.1 | 0.9 \pm 0.1 |
| Kurtosis (<i>Kg</i>) | 1.4 \pm 0.3 | 1.0 \pm 0.1 | 1.0 \pm 0.0 | 1.1 \pm 0.0 | 1.1 \pm 0.0 |
| Sand (%) | 50.5 \pm 13.1 | 13.3 \pm 4.9 | – | 1.2 \pm 2.3 | 1.0 \pm 0.7 |
| Silt (%) | 43.9 \pm 11.6 | 77.5 \pm 4.4 | 78.9 \pm 5.1 | 85.3 \pm 6.2 | 89.5 \pm 5.0 |
| Clay (%) | 7.00 \pm 2.7 | 9.1 \pm 4.7 | 21.1 \pm 5.1 | 13.6 \pm 6.0 | 9.5 \pm 5.1 |
| Univariate indicators | | | | | |
| Density index (<i>DI</i> , N km ^{−2}) | 789.4 \pm 314.8 | 932.0 \pm 405.5 | 807.2 \pm 214.5 | 245.7 \pm 90.5 | 172.8 \pm 38.2 |
| Total number of species | 7 \pm 1.1 | 7.8 \pm 1.2 | 6.9 \pm 1.0 | 4.7 \pm 0.7 | 4.2 \pm 0.5 |
| Species richness (<i>d</i>) | 0.9 \pm 0.1 | 1.0 \pm 0.2 | 0.9 \pm 0.1 | 0.7 \pm 0.1 | 0.6 \pm 0.1 |
| Shannon-Wiener index (<i>H'</i>) | 1.5 \pm 0.1 | 1.5 \pm 0.2 | 1.3 \pm 0.2 | 1.0 \pm 0.1 | 1.1 \pm 0.1 |
| Pielou's evenness index (<i>J'</i>) | 0.8 \pm 0.1 | 0.8 \pm 0.1 | 0.7 \pm 0.1 | 0.7 \pm 0.1 | 0.8 \pm 0.0 |

Table 4. Correlation analysis of physical factors assessed at each sampled haul (complete data set). Pearson's correlation coefficients are shown with p-values in parentheses; Bonferroni's adjustment was applied. Significant correlations at $p < 0.05$ are indicated in **bold** (df = 195). *Mz*: mean grain size; *Sk*: skewness; *s*: sediment sorting; *Kg*: kurtosis

| | Depth | Temp. near bottom ($^{\circ}$ C) | <i>Mz</i> (Φ) | <i>Sk</i> | <i>s</i> | <i>Kg</i> | Sand% | Silt% |
|---|--------------------------|-----------------------------------|--------------------------|--------------------------|--------------------------|--------------------------|--------------------------|--------------------------|
| Temperature near bottom ($^{\circ}$ C) | −0.565 (0.036) | | | | | | | |
| Mean grain size (Φ) | 0.229 (0.036) | −0.459 (0.036) | | | | | | |
| Skewness (<i>Sk</i>) | −0.229 (0.036) | 0.438 (0.036) | −0.764 (0.036) | | | | | |
| Sediment sorting (<i>s</i>) | −0.401 (0.036) | 0.407 (0.036) | −0.681 (0.036) | 0.412 (0.036) | | | | |
| <i>Kg</i> (kurtosis) | 0.466 (0.036) | −0.327 (0.036) | −0.184 (0.396) | −0.059 (2.058) | −0.062 (0.504) | | | |
| Sand (%) | −0.276 (0.036) | 0.395 (0.036) | −0.839 (0.036) | 0.631 (0.036) | 0.552 (0.036) | 0.081 (0.266) | | |
| Silt (%) | 0.34 (0.036) | −0.133 (0.068) | −0.104 (0.154) | 0.148 (1.436) | −0.047 (0.516) | 0.39 (0.036) | −0.372 (0.036) | |
| Clay (%) | −0.198 (0.216) | −0.095 (0.191) | 0.635 (0.036) | −0.542 (0.036) | −0.248 (0.036) | −0.481 (0.036) | −0.247 (0.036) | −0.803 (0.036) |

Table 5. Results of the 1-way ANOVA and Kruskal-Wallis tests on the complete dataset with Strata set as a factor, together with the results of the 2-way ANOVA test on the reduced dataset with Fishing Intensity, Strata and Interaction as factors. Post hoc multi-comparison Tukey's test results are given. Stratum A: 10–50 m, inner shelf; Stratum B: 50–100 m, outer shelf; Stratum C: 100–200 m, shelf break; Stratum D: 200–500 m, upper slope; Stratum E: 500–800 m, upper middle slope; asterisks indicate p-values: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. Log indicates that data were $\log_{10}(x + 1)$ transformed

| Complete dataset | | df | F | p | Tukey's test |
|-----------------------------------|-------------------|----|-------|-------------|--------------------------------------|
| Density index (DI) Log | Strata | 4 | 10.91 | $p < 0.001$ | A, C > D**; B > D***; A, B, C > E*** |
| Total number of species (S) Log | Strata | 4 | 18.43 | $p < 0.001$ | A, B, C > D, E*** |
| Species richness (d) | Strata | 4 | 7.35 | $p < 0.001$ | A > D**; A, C > E*; C > D* |
| Shannon-Wiener index (H') | Strata | 4 | 8.4 | $p < 0.001$ | A, B > D***; A, B > E**; C > D* |
| Pielou's evenness index (J') | | df | H | p | |
| | | 4 | 8.34 | $p = 0.080$ | – |
| Reduced dataset | | df | F | p | Tukey's test |
| Density index (DI) Log | Fishing intensity | 1 | 7.24 | $p < 0.05$ | B > D, E** |
| | Strata | 3 | 4.00 | $p < 0.05$ | |
| | Interaction | 3 | 1.92 | $p = 0.104$ | |
| Total number of species (S) Log | Fishing intensity | 1 | 7.89 | $p < 0.01$ | B > E** |
| | Strata | 3 | 3.45 | $p < 0.05$ | |
| | Interaction | 3 | 1.85 | $p = 0.142$ | |
| Species richness (d) | Fishing intensity | 1 | 6.29 | $p = 0.01$ | B > E** |
| | Strata | 3 | 3.34 | $p < 0.05$ | |
| | Interaction | 3 | 1.40 | $p = 0.245$ | |
| Shannon-Wiener index (H') | Fishing intensity | 1 | 6.44 | $p < 0.01$ | B > D*; B > E*** |
| | Strata | 3 | 4.99 | $p < 0.01$ | |
| | Interaction | 3 | 1.02 | $p = 0.365$ | |
| <i>Astropecten bispinosus</i> Log | Fishing intensity | 1 | 13.83 | $p < 0.001$ | B > D**; B > E***; C > D; C > E*** |
| | Strata | 3 | 4.72 | $p < 0.01$ | |
| | Interaction | 3 | 2.21 | $p = 0.092$ | |
| <i>Liocarcinus depurator</i> Log | Fishing intensity | 1 | 8.53 | $p < 0.01$ | B > D*; C > D, E*** |
| | Strata | 3 | 2.34 | $p < 0.05$ | |
| | Interaction | 3 | 2.65 | $p = 0.064$ | |

$p < 0.001$). Moreover, differences were detected inside the 2 shallower strata, considering the 3 fishing grounds subjected to different fishing intensity as a factor (Stratum B, $R = 0.324$, $p < 0.001$; Stratum C, $R = 0.175$, $p < 0.001$). Pairwise tests showed that differences occurred between each fishing ground and that in both strata, the differences were higher between the more and less exploited fishing ground, respectively the western and the eastern (Stratum B, $w \neq c$, $R = 0.259$; $c \neq e$, $R = 0.219$; $e \neq w$, $R = 0.338$; Stratum C, $w \neq c$, $R = 0.369$; $c \neq e$, $R = 0.268$; $e \neq w$, $R = 0.497$, $p < 0.001$). This result indicated that the benthic communities differed in relation to fishing pressure within each stratum.

A Pearson's correlation analysis showed significant correlations between 3 environmental variables and fishing intensity: mean grain size ($r = 0.263$, $p = 0.001$, $df = 167$), kurtosis index ($r = -0.334$, $p = 0.001$, $df = 167$) and percentage of sand ($r = -0.230$, $p < 0.01$, $df = 167$). Two RELATE procedures conducted to test sep-

arately the effects of depth and the effects of fishing intensity revealed that both depth ($r = 0.185$, $p < 0.001$) and fishing intensity ($r = 0.166$, $p < 0.001$) influenced community structure. In summary, depth and environmental variables were linked to the distribution of fishing with depth, which is not surprising given that fishermen are likely to target habitats that yield the highest catches of target species. Effects of fishing intensity and depth strata factors were detected on biodiversity indices using a 2-way ANOVA test ($p < 0.05$). There was no interaction between the effect of fishing intensity and depth strata (Table 5, Fig. 5).

The response of the biodiversity indices varied significantly, such that the mid depths, strata C and D, tended to show positive responses, whereas the response in the shallowest and deepest strata tended not to deviate from zero, and the values detected in Stratum B were always higher than in Stratum E. Pielou's evenness index was uniform among the 4

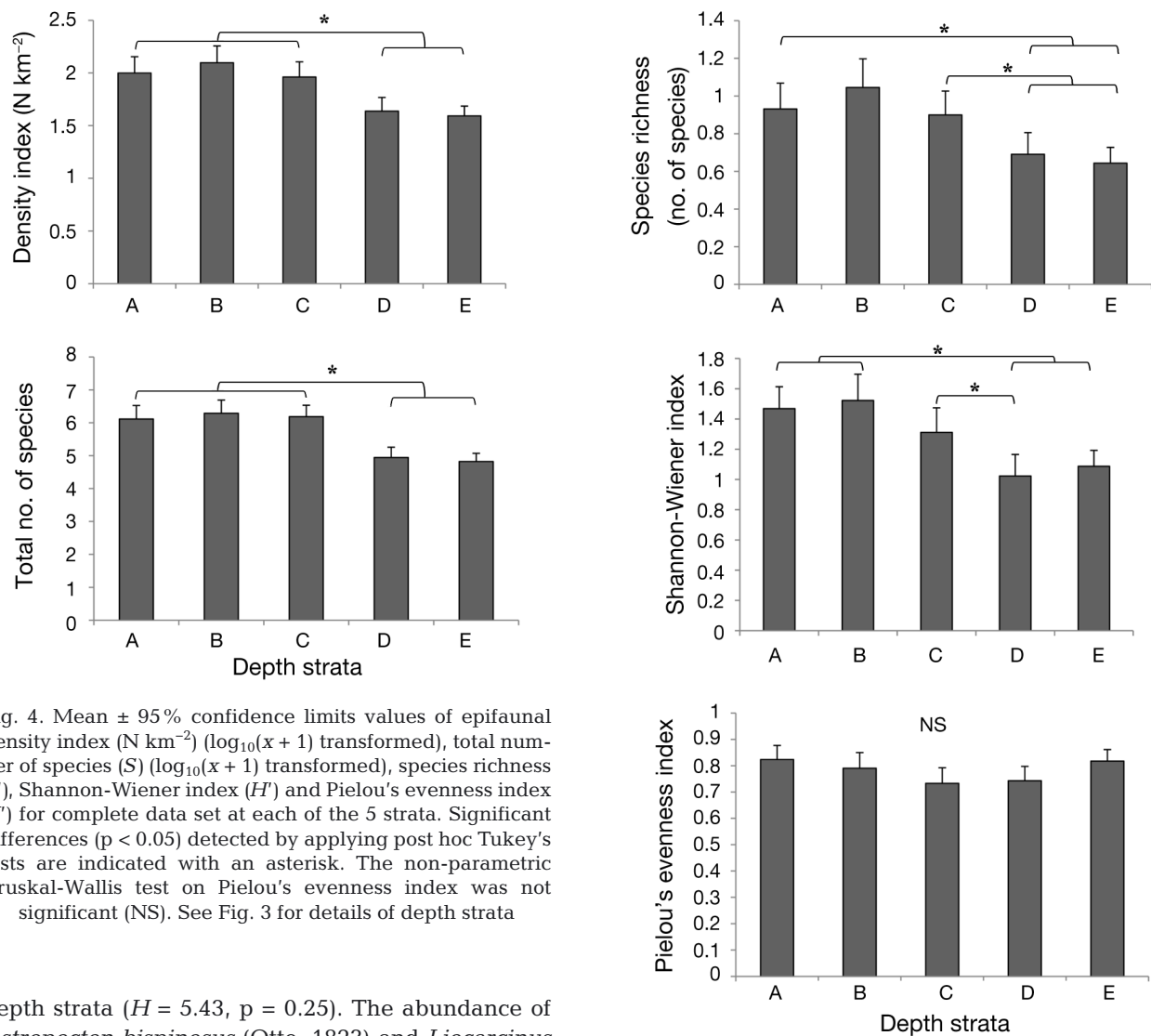


Fig. 4. Mean \pm 95 % confidence limits values of epifaunal density index ($N\ km^{-2}$) ($\log_{10}(x + 1)$ transformed), total number of species (S) ($\log_{10}(x + 1)$ transformed), species richness (d), Shannon-Wiener index (H') and Pielou's evenness index (J') for complete data set at each of the 5 strata. Significant differences ($p < 0.05$) detected by applying post hoc Tukey's tests are indicated with an asterisk. The non-parametric Kruskal-Wallis test on Pielou's evenness index was not significant (NS). See Fig. 3 for details of depth strata

depth strata ($H = 5.43$, $p = 0.25$). The abundance of *Astropecten bispinosus* (Otto, 1823) and *Liocarcinus depurator* (Linnaeus, 1758), both scavenging species, increased in all depth strata with higher levels of fishing intensity (Fig. 5).

Analysis of the abundance data showed that species considered to be indicators of disturbance increased in abundance when exposed to higher levels of trawl disturbance, whereas species considered to be sensitive to disturbance decreased. Those sites that were not subject to fishing or that were subject to a low level were characterised by a high abundance of sessile species and deposit feeders. The former, such as crinoids *Antedon mediterranea* (Lamarck, 1816) and *Leptometra phalangium* (Müller, 1841), are considered vulnerable to the effects of disturbance. The latter, such as the ophiuroid *Ophiura ophiura* (Linnaeus, 1758) and the holothurian *Parastichopus regalis* (Cuvier, 1817), live on the sediment surface (see 1-way ANOVA

results in Table 2). None of the species found at the deepest stratum (Stratum E) appeared to respond to fishing disturbance. The number of non-target species that demonstrated significant variation along a gradient of fishing intensity within each depth stratum decreased from the shallower to the deeper depth strata (Table 2).

DISCUSSION

The present study examined the structure and distribution of epibenthic assemblages across fishing grounds in the Southern Tyrrhenian Sea using a long-term dataset of the non-target species fraction caught by an otter trawl deployed as part of the MEDITS sampling programme. Depth was the domi-

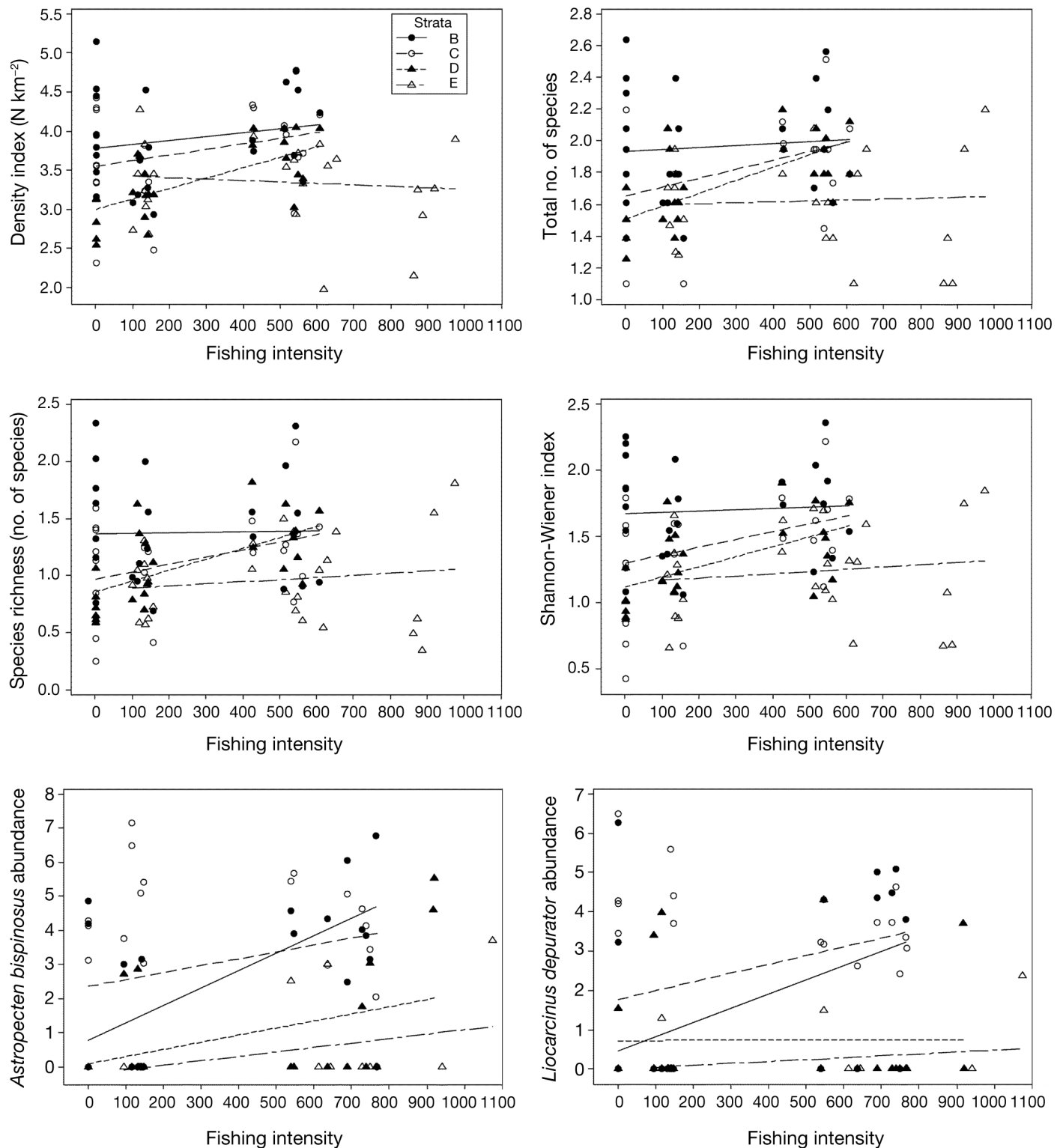


Fig. 5. Two-way ANOVA performed on means of density index ($N\ km^{-2}$) ($\log_{10}(x + 1)$ transformed), total number of species (S) ($\log_{10}(x + 1)$ transformed), species richness (d), Shannon-Wiener index (H'), *Astropecten bispinosus* abundance ($\log_{10}(x + 1)$ transformed) and *Liocarcinus depurator* abundance ($\log_{10}(x + 1)$ transformed) on reduced data set considering Fishing Intensity, Strata and Fishing Intensity \times Strata (interaction) as factors (black circle: Stratum B; white circle: Stratum C; black triangle: Stratum D; white triangle: Stratum E)

nant physical gradient across the study area (Gage & Tyler 1991), hence it was not surprising that assemblage structure differed among depth strata. The distribution of these assemblages was stable across the time-series considered.

On the continental shelf, 3 statistically different assemblages were detected: inner shelf, outer shelf and shelf break. In the context of the study area, the inner shelf is characterised by highly heterogeneous environmental conditions typical of coastal areas that are dominated by high sedimentation rate and enriched by local irregular terrigenous input (Cosentino & Giacobbe 2006, 2008). From the outer shelf to the shelf break, the change in depth was associated with a change in sediment type that influenced community structure. Generally, the shelf was characterised by a predominance of mega-epibenthic filter-feeding and suspension-feeding organisms. The shelf break is a high-energy zone characterised by upwelling phenomena as a result of the interaction between tides, storm waves and the fronts that separate the shelf and slope waters (Pinazo et al. 1996). These conditions increase the transport of organic matter and input of nutrients into the water column, thereby contributing to the structuring of a benthic assemblage that consisted mostly of epibenthic suspension feeding organisms such as the crinoid *Leptometra phalangium* (Lavaleye et al. 2002).

In contrast with previous studies undertaken in the Northern Tyrrhenian Sea (e.g. Colloca et al. 2003), 2 distinct assemblages were identified on the upper and upper middle slope. These communities were prevalently characterised by muddy sediments dominated by decapod crustaceans, as reported in the continental slope in the Central Mediterranean basin (Cartes 1993). The general decrease in biodiversity indices with increasing depth is most likely related to a reduction in food availability (Pérès 1985).

Fishing intensity is patchily distributed across the study area, with some fishing grounds fished intensively and more constantly across time, while other areas are less intensively fished or not fished at all. This pattern of fishing intensity distribution is generally related to seabed type, seabed morphology and to the occurrence of target species. Despite the recommendations of the European Common Fishery Policy for the Mediterranean basin (Multi-Annual Guidance Plans [MAGPs]), the initiative to reduce fishing effort may have only very recently affected the fleets considered in this study, in that there appears to be a general condition of overfishing of target species in the area (Lleonart & Maynou 2003).

The strong zonation patterns in the epibenthic assemblage that are related to depth made it more difficult to address the effects of trawling activity, which is also strongly depth stratified in the Mediterranean Sea (Dimech et al. 2008). For this reason, we undertook the analysis of univariate indices of the epibenthic assemblage within each depth stratum in relation to fishing intensity. The response of the epibenthic assemblages to fishing disturbance was similar in both the shelf break and upper slope. These depth strata are the most heavily exploited by trawling activity, not only due to the concentration of target species at these depths (Lleonart & Maynou 2003), but also because the fleet is legally permitted to fish in the zone extending from 50 to 1000 m. Restriction of the fleet to a limited range of depths may explain the strong correlation between fishing intensity and granulometric features, which are known to be affected by re-suspension due to fishing activities (Churchill 1989, Pilskaln et al. 1998, Palanques et al. 2001). Historically, evidence of heavy exploitation on the seabed of the continental shelf and slope across the study area has been reported by different authors (Arena & Bombace 1970, Internal Report CNR 1988) and this intensive fishing may have resulted in a change in sediment granulometry.

The main non-target species collected throughout the study area had functional traits that were typical of organisms resilient to the effects of trawling disturbance, e.g. starfish and crabs (de Juan et al. 2007 and references therein). Within each depth stratum, the abundance of mobile scavengers and opportunistic taxa was higher in areas subjected to higher levels of fishing intensity. Scavenger species that were elevated in abundance included mobile scavengers such as the starfish *Astropecten bispinosus* and the decapods *Medorippe lanata* (Linnaeus, 1767), *Polychaetes typhlops* Heller, 1862 and the 2 portunid crabs *Liocarcinus depurator* and *Macropipus tuberculatus* (Roux, 1830) (Demestre et al. 2000, Bergmann et al. 2002, Cartes et al. 2002). All these species are characterised by common features, such as their feeding habits or their ability to regenerate body tissues that become damaged by fishing (Ramsay et al. 2001). The sites that were subjected to higher values of fishing intensity are characterised by these species as a result of a 'cultivation effect'. The increase in the density and biodiversity indices with an increasing fishing intensity gradient inside the study area could be explained by the presence of fishing grounds that have been intensively trawled in recent decades, such that the epibenthic communities associated

with these locations are already in an alternative stable state (Hall 1999). The long history of exploitation of Mediterranean benthic communities may have caused selection of taxa that are less vulnerable to trawl disturbance and that are better adapted to frequent anthropogenic disturbance, making the benthic communities of continental shelf and slope locally homogeneous and dominated by opportunistic organisms such as scavengers. The shift to a more scavenger-dominated community could have important implications for overall benthic community composition, altering predator–prey relationships (Piet & Jennings 2005).

Sensitive species such as the ophiuroid *Ophiura ophiura* and the fragile crinoids *Antedon mediterranea* and *Leptometra phalangium* were only found in areas where fishing activity was the least intensive (Smith et al. 2000). In the Northern Tyrrhenian Sea, crinoid beds have an important role in terms of secondary production (Colloca et al. 2004 and references therein) and they enhance habitat heterogeneity through the development of 3-dimensional habitat (Gili & Coma 1998). The value of this habitat and the threat of intensive disturbance emphasise the importance of adopting an ecosystem approach to fisheries in order to simultaneously manage not only fish and shellfish target species, but also non-commercial organisms and habitats (Bergmann et al. 2004, Browman & Stergiou 2004, Thrush & Dayton 2010).

CONCLUSIONS

The present study provides evidence of the long-term effects of fishing in the Southern Tyrrhenian Sea on benthic communities from the shallow shelf to the deep seabed of the middle slope. Such studies were lacking for this region, but our results concur with expectations based on other publications (e.g. Kaiser et al. 2006, Dimech et al. 2008, 2012).

The apparent alteration of the epifaunal assemblage by fishing activities has emerged despite the limited quality of available fishing intensity statistics for the area (Farrugio et al. 1993, Papaconstantinou & Farrugio 2000) and despite the gaps in the time-series of scientific sampling data (Symes 1999, Cacaud 2005), which perhaps makes the findings more compelling. The present study demonstrates the link between changes in benthic communities and trends in fishing intensity, thereby providing insight into how these effects might be managed in the future.

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Appendix 1. Species list

Table A1. List of species collected and identified in the study area over the entire period of investigation. An: Anthozoa; Br: Brachiopoda; Cr: Crustacea; Ec: Echinodermata; Mo: Mollusca. Species included in the statistical analyses are in **bold**

| Species | Code | Species | Code |
|---|------|---|------|
| Acanthocardia aculeata (Linnaeus, 1758) | Mo | Macropodia longirostris (Fabricius, 1775) | Cr |
| Aegaeon lacazei (Gourret, 1887a) | Cr | Maja crispata Risso, 1827 | Cr |
| Aegaeon sp. Agassiz, 1846 | Cr | Maja goltziana d'Oliveira, 1888 | Cr |
| <i>Alcyonium palmatum</i> Pallas, 1766 | An | Marthasterias glacialis (Linnaeus, 1758) | Ec |
| <i>Alpheus glaber</i> (Olivi, 1792) | Cr | Medorippe lanata (Linnaeus, 1767) | Cr |
| Anamathia rissoana (Roux, 1828) | Cr | <i>Mimachlamys varia</i> (Linnaeus, 1758) | Mo |
| <i>Anapagurus laevis</i> (Bell, 1846) | Cr | Munida intermedia A. Milne Edwards & Bouvier, 1899 | Cr |
| <i>Anomia ephippium</i> Linnaeus, 1758 | Mo | Munida tenuimana Sars, 1872 | Cr |
| Antedon mediterranea (Lamarck, 1816) | Ec | Natica sp. Scopoli, 1777 | Mo |
| Aporrhais serresianus (Michaud, 1828) | Mo | <i>Nemausa spinipes</i> (Bell, 1835) | Cr |
| <i>Arbacia lixula</i> (Linnaeus, 1758) | Ec | Neopycnodonte cochlear (Poli, 1795) | Mo |
| Astropecten hispidus (Otto, 1823) | Ec | <i>Nepinnotheres pinnotheres</i> (Linnaeus, 1758) | Cr |
| Astropecten irregularis (Pennant, 1777) | Ec | <i>Ophioderma longicauda</i> (Bruzeliuss, 1805) | Ec |
| Astrosartus mediterraneus (Risso, 1826) | Ec | Ophiura ophiura (Linnaeus, 1758) | Ec |
| Bolinus brandaris (Linnaeus, 1758) | Cr | <i>Ova canaliferus</i> (Lamarck, 1816) | Ec |
| Bathynectes maravigna (Prestandrea, 1839) | Mo | <i>Paguristes eremita</i> (Linnaeus, 1767) | Cr |
| Calliostoma granulatum (Von Born, 1778) | Mo | Pagurus alatus Fabricius, 1775 | Cr |
| Centrostephanus longispinus (Philippi, 1845) | Ec | Pagurus excavatus (Herbst, 1791) | Cr |
| Dardanus arrosor (Herbst, 1796) | Cr | Pagurus prideaux Leach, 1815 | Cr |
| <i>Ebalia nux</i> A. Milne-Edwards, 1883 | Cr | <i>Paracentrotus lividus</i> (Lamarck, 1816) | Ec |
| Echinaster sepositus (Retzius, 1783) | Ec | Parastichopus regalis (Cuvier, 1817) | Ec |
| Eriphia verrucosa (Forskål, 1775) | Cr | <i>Paromola cuvieri</i> (Risso, 1816) | Cr |
| Ethusa mascarone (Herbst, 1785) | Cr | Pasiphaea sivado (Risso, 1816) | Cr |
| <i>Funiculina quadrangularis</i> (Pallas, 1766) | An | <i>Pecten sp.</i> O.F. Müller, 1776 | Mo |
| Galathea intermedia Liljeborg, 1851 | Cr | <i>Pennatula phosphorea</i> Linnaeus, 1758 | An |
| <i>Galeodea echinophora</i> (Linnaeus, 1758) | Mo | <i>Pennatula rubra</i> (Ellis, 1761) | An |
| Geryon longipes A. Milne-Edwards, 1882 | Cr | Pilumnus minutus De Haan, 1835 | Cr |
| Goneplax rhomboides (Linnaeus, 1758) | Cr | Pilumnus sp. Leach, 1816 | Cr |
| Gracilechinus acutus (Lamarck, 1816) | Ec | Pilumnus spinifer H. Milne Edwards, 1834 | Cr |
| <i>Gryphus vitreus</i> (Born, 1778) | Br | Polycheles typhlops Heller, 1862 | Cr |
| <i>Hacelia attenuata</i> Gray, 1840 | Ec | <i>Psammecinus microtuberculatus</i> (Blainville, 1825) | Ec |
| Holothuria tubulosa Gmelin, 1791 | Ec | <i>Pteroeides spinosum</i> (Ellis, 1764) | An |
| Holothuria poli Delle Chiaje, 1824 | Ec | <i>Scaphander lignarius</i> (Linnaeus, 1758) | Mo |
| <i>Homola barbata</i> (Fabricius, 1793) | Cr | <i>Solenocera membranacea</i> (Risso, 1816) | Cr |
| Inachus sp. Weber, 1795 | Cr | <i>Spatangus purpureus</i> O.F. Müller, 1776 | Ec |
| Latreillia elegans Roux, 1830 | Cr | <i>Sphaerechinus granularis</i> (Lamarck, 1816) | Ec |
| Leptometra phalangium (Müller, 1841) | Ec | <i>Spinolambus macrochelios</i> (Herbst, 1790) | Cr |
| Liocarcinus bolivari (Zariquiey Alvarez, 1948) | Cr | Stylocidaris affinis (Philippi, 1845) | Ec |
| Liocarcinus depurator (Linnaeus, 1758) | Cr | Tethyaster subinermis (Philippi, 1837) | Ec |
| Liocarcinus pusillus (Leach, 1816) | Cr | <i>Tethys fimbria</i> Linnaeus, 1767 | Mo |
| <i>Lophozozymus incisus</i> (H. Milne Edwards, 1834) | Cr | <i>Thia scutellata</i> (Fabricius, 1793) | Cr |
| Luidia sarsi Düben & Koren, 1845 | Ec | Trachythyone sp. Studer, 1876 | Ec |
| <i>Lunatia fusca</i> (Blainville, 1825) | Mo | <i>Virgularia mirabilis</i> (Müller, 1776) | An |
| Macropipus tuberculatus (Roux, 1830) | Cr | Xantho pilipes A. Milne-Edwards, 1867 | Cr |