



ELSEVIER

Available online at www.sciencedirect.com

SCIENCE @ DIRECT®

Marine Environmental Research xxx (2004) xxx–xxx

 MARINE
 ENVIRONMENTAL
 RESEARCH

www.elsevier.com/locate/marenvrev

Evaluating the effects of protection on fish predators and sea urchins in shallow artificial rocky habitats: a case study in the northern Adriatic Sea

P. Guidetti ^{*}, S. Bussotti, F. Boero

Laboratory of Zoology and Marine Biology, Dipartimento di Biologia, DiSTeBA, CoNISMa, University of Lecce, Via Provincial Lecce-Monteroni, 73100 Lecce, Italy

Received 12 January 2004; received in revised form 15 May 2004; accepted 21 May 2004

Abstract

Man-made defence structures (e.g., breakwaters, jetties) are becoming common features of marine coastal landscapes all around the world. The ecology of assemblages of species associated with such artificial structures is, however, poorly known. In this study, we evaluated the density and size of fish predators of echinoids (i.e., *Diplodus sargus*, *Diplodus vulgaris*, *Sparus aurata*), and the density of sea urchins (i.e., *Paracentrotus lividus*) at defence structures (i.e., breakwaters) inside and outside the marine protected area of Miramare (northern Adriatic Sea) in order to: (1) assess possible differences in fish predator density and size between protected and fished breakwaters; (2) assess whether fish predation may have the potential to affect sea urchin density in artificial rocky habitats. Surveys were carried out at four random times over a period of two years. Total density, and density of medium- and large-sized individuals of the three predatory fishes were generally greater at the protected than at the fished breakwaters, whereas no differences were detected in the density of small-sized individuals. Density of the sea urchin *P. lividus* did not show any difference between protected and fished breakwaters. The results of this study suggest that: (1) protection may significantly affect predatory fishes in artificial rocky habitats; (2) differences in predatory fish density, and size may be unrelated with the density of the sea urchin *P. lividus*; (3) protected artificial structures such as breakwaters, originally planned for other purposes, could represent a potential tool for fish population recovery and enhancement of local fisheries.

© 2004 Elsevier Ltd. All rights reserved.

^{*} Corresponding author. Tel.: +39-832-298853/320853; fax: +39-832-298626/320626.

E-mail address: paolo.guidetti@unile.it (P. Guidetti).

Keywords: Artificial habitats; Fishery impact; Marine protected areas; Visual census; Asymmetric ANOVA; Adriatic Sea

1. Introduction

Man-made structures are becoming very common in shallow coastal waters all around the world. Some artificial reefs are specifically planned for enhancing fish productivity and/or mitigating the destructive effects of illegal trawling (Bohnsack, Johnson, & Ambrose, 1991; Relini, Relini, Torchia, & Palandri, 2002; Wilson, Leung, & Kennish, 2002). Most of the man-made structures built in marine coastal areas (e.g., breakwaters, jetties), however, are aimed at a wide array of other purposes, such as protection of coastal maritime, industrial and urban developments and activities against erosion and wave action (Bulleri & Chapman, in press; Chapman & Bulleri, 2003; CIESM, 2002; Glasby & Connell, 1999). Coastal defences are expected to increase in the near future, as human populations are increasing on coastal zones worldwide (Connell, 2000, & references therein). These artificial structures generally introduce novel hard-bottom habitats into shallow coastal waters usually characterised by soft bottoms. Several authors thus stressed the urgent need for better understanding the potential impact of coastal defences, and their associated assemblages, on natural environments (Chapman & Bulleri, 2003; Glasby & Connell, 1999). Most of the available information on the assemblages associated with defence structures concerns intertidal and subtidal epibiota, and their interactions with some physical and biological factors (Bacchiocchi & Airoidi, 2003; Bulleri & Chapman, in press; Chapman & Bulleri, 2003; Coleman & Connell, 2001; Connell, 2000, 2001a, 2001b; Dethier, Mc Donald, & Strathmann, 2003; Glasby, 1999). Scant and spatially limited information, instead, is available about associated fish assemblages (Guidetti, 2004; Rilov & Benayahu, 1998). No attempts, to our knowledge, have been made to evaluate whether defence structures could be included in management programs aimed at enhancing local fisheries. This latter issue may be especially important in those regions that have experienced high levels of fishing pressure, and/or where natural rocky substrates are lacking.

There is an increasing body of scientific evidence suggesting that fishing may alter marine benthic ecosystems (Dayton, Thrush, Agardy, & Hofman, 1995; Dayton, Thrush, & Coleman, 2002; Jackson et al., 2001; Micheli et al., 2001; Sala, Boudouresque, & Harmelin-Vivien, 1998; Steneck & Carlton, 2001). Fishing, in fact, has the potential to directly affect target species (e.g., in terms of changes in density, biomass, and size), and, on the other hand, indirectly influence the structure of whole assemblages and ecosystem functioning (Jennings & Kaiser, 1998; Shears & Babcock, 2002, 2003; Steneck, 1998) mainly via top-down perturbations (see Micheli et al., 2001 for a review). The bulk of marine species targeted by fisheries, in fact, are high-level predatory fishes (Myers & Worm, 2003, & references), whose removal

from natural systems may cause dramatic changes in the rest of the community through the so-called “trophic cascades” (Paine, 1980; Witman & Dayton, 2001).

In shallow sublittoral rocky reefs in the Mediterranean, sea urchins, namely *Paracentrotus lividus* and *Arbacia lixula*, may heavily influence marine benthic communities, by affecting the transition between macroalgal beds and barrens (Bulleri, Benedetti-Cecchi, & Cinelli, 1999; Guidetti, Fraschetti, Terlizzi, & Boero, 2003; Guidetti, Terlizzi, & Boero, 2004; Sala et al., 1998). Although some marine invertebrates have been reported as potential predators of sea urchins (see Guidetti, in press, and references therein; Sala, 1997; Sala et al., 1998), recent studies have provided evidence that only a few fish species may actually be considered efficient sea urchin predators in the Mediterranean (Guidetti, in press; Sala, 1997). They include the sparids *Diplodus sargus*, *Diplodus vulgaris*, *Sparus aurata* (that prey upon adult and juvenile sea urchins), and the labrids *Coris julis* and *Thalassoma pavo* (that are predators of juvenile sea urchins only). Some of these fishes are of great economic importance (mainly *S. aurata* and the two *Diplodus*; Harmelin, Bachet, & Garcia, 1995; Jouvenel & Pollard, 2001), which means that a decrease in their abundance due to fishing could have consequences on their preys (i.e., sea urchins), and in turn on the entire rocky reef benthic communities. From this perspective, there is no evidence about the existence of trophic cascades in shallow artificial rocky habitats.

Several authors have stressed the efficacy of marine reserves as a management tool in the framework of both general conservation policies and programs aimed at enhancing local fisheries (CIESM, 1999; Gell & Roberts, 2003; Halpern, 2003). There are, however, no data about the possible inclusion of artificial defence structures, originally planned for other uses (e.g., breakwaters), in management programs of coastal fisheries, although it is well known they attract local professional and recreational fishermen. The inclusion of defence structures in management programs could be particularly important in those regions, such as the central and northern Adriatic Italian coasts, where the natural foreshore is mostly sandy, and artificial structures constitute most of the available rocky substrates (Bacchiocchi & Airoidi, 2003). The present study, carried out at the MPA of Miramare and adjacent areas (northern Adriatic Sea), where rocky substrates are mostly artificial (i.e., breakwaters), thus offers a chance to improve the knowledge on the ecology of shallow artificial rocky habitats, and to assess whether protection may affect the associated marine assemblages.

Specifically, we asked the following questions: Are abundance and size of predatory fishes larger at protected than at fished breakwaters? Is the abundance of sea urchins smaller at protected than at fished breakwaters?

2. Materials and methods

2.1. Study area

This study was done at the MPA of Miramare and adjacent areas (Northern Adriatic Sea, NE Italy; Fig. 1). The Miramare MPA is a relatively small reserve

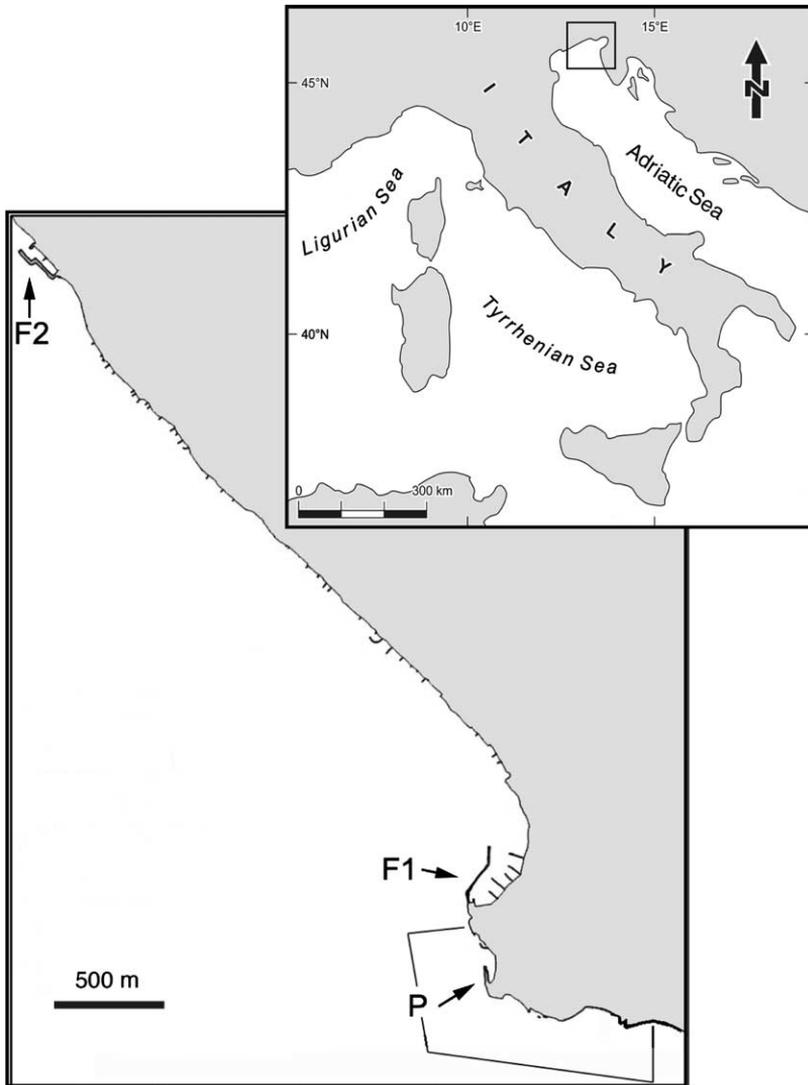


Fig. 1. Study area and location of the protected (P) and fished (F1, F2) breakwaters at the marine reserve of Miramare.

(about 121 ha) established in 1986, where enforcement of protection is successful and illegal poaching negligible.

Within the MPA, the foreshore is formed by both natural and artificial rocky substrates, these latter being represented by external breakwaters (made of transplanted boulders with their longer axes ranging from 1 to 3 m), running parallel to the coast, with internal seawalls, that provide shelter from onshore winds to small boats.

In this study, samplings were done at the single protected (hereafter P) breakwater within the MPA, and at two fished breakwaters (F1 and F2) outside the reserve (Fig. 1). Both fished breakwaters are located north of the protected one (i.e., there is spatial segregation) as appropriate fished breakwaters were not available southwards. Breakwaters, which were sampled on four occasions over the two study years (from spring 2002 to late summer 2003), have similar general features (e.g., in terms of wave exposure) and extend from the water surface down to about 5–8 m depth over muddy sand. Local macrobenthic assemblages did not differ in relation to the protection level. In particular, the cover of erect (e.g., Dictyotales, *Corallina* sp., *Halimeda tuna*, *Padina pavonica*) and encrusting macroalgae (e.g., *Peyssonnelia* spp.), which may be important for the survival of juvenile sea urchins, did not differ between the protected and the fished breakwaters (Elia, 2003).

2.2. Data collection and sampling design

In the study area, fish predators of sea urchins are represented only by the sparids *D. sargus*, *D. vulgaris* and *S. aurata*, since the labrids *C. julis* and *T. pavo* are absent (Castellarin, Visintin, & Odorico, 2001; Guidetti, Verginella, Viva, Odorico, & Bero, submitted). Densities of the three predatory fishes were estimated by visual census at 4–7 m depth, along 25 m long and 5 m wide transects (according to the ‘strip transect’ method; Harmelin-Vivien et al., 1985). Four transects (used as replicates), located a few meters from each other (the sampled breakwaters are not long enough for an appropriate randomisation of transects; see further details below), were performed at each breakwater in each sampling time. Each fish recorded was assigned to one of three size classes, i.e., small, medium, and large, corresponding to the lower, intermediate and upper 33% range of sizes reported in the literature for each species (Fischer, Bauchot, & Schneider, 1987). Juvenile stages (i.e., settlers and recruits) were excluded from the study, since their numerical contribution may greatly influence mean values, while having no predatory effect on sea urchins.

Within the same depth range (i.e., 4–7 m), counts of sea urchins were performed along 10 m long and 2 m wide transects. Three replicated transects were done at each of the three breakwaters at each sampling time. Only *P. lividus* was included in the present study, because *A. lixula* was generally found at shallower depth, and *Sphaerechinus granularis* was too rare to be considered ecologically relevant in influencing benthic assemblages. The method used here for assessing echinoids is suitable for evaluating the adult fraction (with test diameter >1 cm) of sea urchin populations (Sala & Zabala, 1996).

2.3. Statistical treatment of data

We used asymmetrical analyses of variance (ANOVA; GMAV5 software package, University of Sydney, Australia) to test for differences between protected and fished breakwaters in the variables investigated. Asymmetrical (ACI) designs, and their mechanics and potential for detecting spatio-temporal changes are discussed in Glasby (1997) and Underwood (1994). The term ‘Protected vs. Fished breakwaters’

(hereafter ‘P vs. Fs’) was fixed, whereas the term ‘between Fs’ was random. Due to the fact that breakwaters (both P and Fs) were not long enough to enable the sampling of different transects at each of the four sampling times (i.e., there has been overlap among sampling areas from one to another sampling time), independent tests were made for each time of sampling. With regard to fish size data, analyses (i.e., comparison P vs. Fs) were carried out pooling the data collected in the four sampling times. Predatory effects of fish on sea urchins (which are related with predator size; Guidetti, in press), in fact, are expected to cumulate through time, and to be independent of short-time fluctuations of predatory fishes. Prior to the analyses, data were tested for homogeneity of variances by means of Cochran’s test and, whenever necessary, they were appropriately transformed (Underwood, 1997).

3. Results

In *T1* and *T2* the average density of *D. sargus* was significantly higher at the protected breakwater than at the fished ones, while no statistical differences were detected in the remaining sampling times, i.e., *T3* and *T4* (Fig. 2; Table 1). As far as the size is concerned, medium- and, even more, large-sized specimens were significantly more abundant at the protected than at the fished breakwaters, whereas no differences were detected for small-sized *D. sargus* (Fig. 3; Table 2).

D. vulgaris was recorded only at the protected breakwater during the first sampling time (i.e., *T1*). This fish was significantly more abundant at the protected than at the fished breakwaters in *T2* and *T4*, whereas no difference between the protected and the fished breakwaters was observed in *T3* (Fig. 4; Table 3). *D. vulgaris* showed the same patterns of spatial distribution related to size as observed for *D. sargus*. Medium- and large-sized individuals, in fact, were more abundant at the protected than the fished breakwaters, while small *D. vulgaris* did not display any difference related to the level of protection (Fig. 5; Table 4).

S. aurata was found exclusively at the protected breakwater during *T1*, *T2* and *T4*, and it was significantly more abundant at the protected than at the fished breakwaters in *T3* (Fig. 6; Table 5). Small-sized *S. aurata* were absent at both protected and fished breakwaters, large-sized individuals were exclusively censused at the protected breakwater, and medium-sized individuals showed significantly greater density at the protected than at the fished breakwaters (Fig. 7; Table 6).

Even though no formal tests are allowed, due to potential temporal dependence of data, inspection of the graphs reveals large temporal variability in the density of the two species of *Diplodus*, both at the protected and at the fished breakwaters (Figs. 2 and 4). In contrast, the density of *S. aurata* was fairly constant from one sampling time to another (Fig. 6).

The density of the sea urchin, *P. lividus*, did not differ between protected and fished breakwaters, at each of the four sampling times. A significant variability between the fished breakwaters was observed only in *T3* (Fig. 8; Table 7). The inspection of the graph revealed a remarkable variability in time at each of the three breakwaters investigated (Fig. 8).

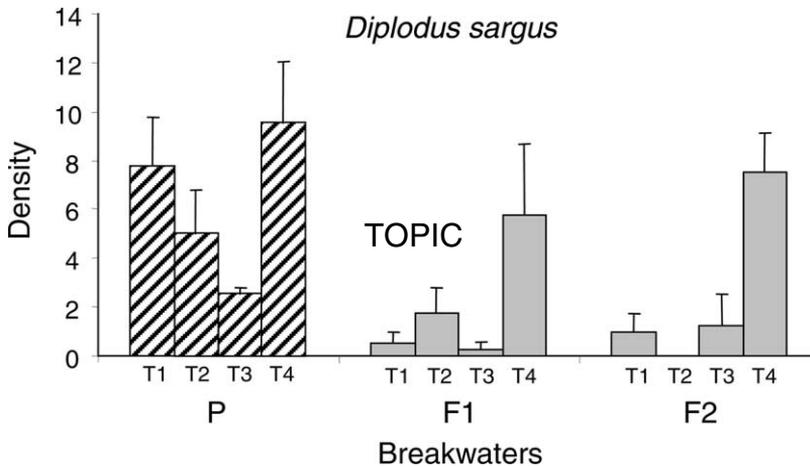


Fig. 2. Average density (\pm SE; number of individuals 125 m^{-2}) of the predatory fish *D. sargus* at the

Table 1

Asymmetrical analyses of variance comparing density of the predatory fish *D. sargus* at each of the four sampling times ($T1$, $T2$, $T3$, $T4$), at one protected (P) and two fished breakwaters (Fs)

Source of variation	d.f.	$T1$		$T2$		$T3$		$T4$	
		MS	F	MS	F	MS	F	MS	F
Location (L)	2	3.55		25.75		5.08		14.08	
P vs. Fs	1	7.04	24.28**	45.38	8.37*	8.17	3.58n.s.	22.05	0.99n.s.
Fs	1	0.06	0.21n.s.	6.12	1.13n.s.	2.00	0.88n.s.	6.12	0.27n.s.
Residual	9	0.29		5.42		2.28		22.30	
Cochran's test			n.s.		n.s.		n.s.		n.s.
Transformation			Sqrt		none		none		none

Significance levels: n.s. = $p > 0.05$; * = $p < 0.05$; ** = $p < 0.01$.

4. Discussion and conclusions

The present study carried out at the Miramare MPA and adjacent areas suggests that protection may lead to high density and large size of predatory fishes associated with coastal defence structures.

Several authors have reported that abundance, size and/or biomass of many commercial fishes (often predators) are generally greater within MPAs than in fished areas, both in the Mediterranean basin (Francour, 1991; García-Rubies & Zabala, 1990; García-Charton et al., 2004; Harmelin et al., 1995; Vacchi, Bussotti, Guidetti, & La Mesa, 1998) and elsewhere in the world (Halpern & Warner, 2002; Polunin & Roberts, 1997; Willis, Millar, & Babcock, 2003). The species of fish investigated in the present study, namely *D. sargus*, *D. vulgaris* and *S. aurata*, are of great commercial value, and are targeted by many kinds of fisheries in the Mediterranean Sea

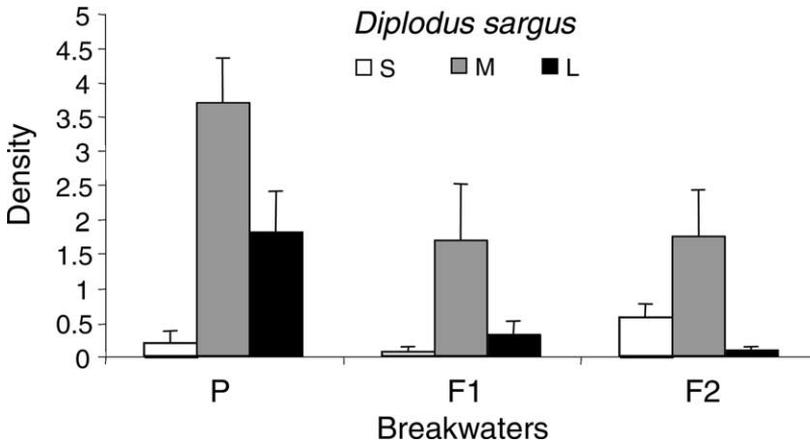


Fig. 3. Average density (\pm SE; number of individuals 125 m^{-2}) of *D. sargus* at the protected (P) and fished (F1, F2) breakwaters (data of the four sampling times cumulated), in relation to size (S: small; M: medium; L: large).

Table 2

Asymmetrical analyses of variance comparing density of the predatory fish *D. sargus* in relation to the size class (see Section 2), at one protected (P) and two fished breakwaters (Fs)

Source of variation	d.f.	Small		Medium		Large	
		MS	F	MS	F	MS	F
Location (L)	2	1.08		20.68		14.33	
P vs. Fs	1	0.17	0.16n.s.	41.34	4.85*	28.17	13.67**
Fs	1	2.00	1.83n.s.	0.03	0.004n.s.	0.50	0.24n.s.
Residual	45	1.09		8.53		2.06	
Cochran's test			*		n.s.		*
Transformation			none		none		none

Significance levels: n.s. = $p > 0.05$; * = $p < 0.05$; ** = $p < 0.01$.

(e.g., spearfishing; Harmelin et al., 1995). The above issues and the outcomes of this study thus suggest that fishing is likely to be one of the most important factors affecting their populations in Mediterranean coastal waters, and that the positive effects of protection (in the form of greater density and/or size) may be also evident at breakwater structures.

Increases in density and size of target fish species have been already observed at artificial reefs constructed with the aim of enhancing fishery resources (Relini et al., 2002; Wilson et al., 2002). The difference between coastal defences and artificial reefs resides, however, not only in their purpose. Coastal defences, in fact, are usually built near the shore and, in most cases, they emerge from the water. Artificial reefs, conversely, are placed off-shore and rest on bottoms far deeper than the water surface (usually more than 10 m depth). As several littoral fishes in the Mediterra-

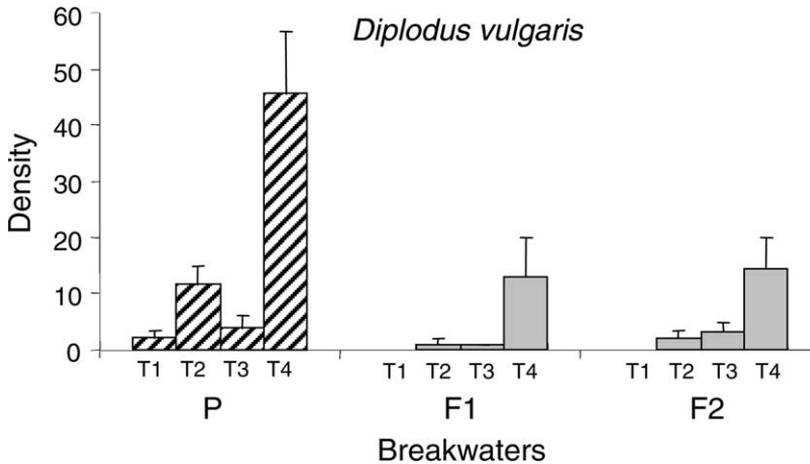


Fig. 4. Average density (\pm SE; number of individuals 125 m^{-2}) of the predatory fish *D. vulgaris* at the protected (P) and fished (F1, F2) breakwaters, in each of the four sampling times (T1, T2, T3 and T4).

Table 3

Asymmetrical analyses of variance comparing density of the predatory fish *D. vulgaris* at the sampling times T2, T3 and T4, at one protected (P) and two fished breakwaters (Fs)

Source of variation	d.f.	T2		T3		T4	
		MS	F	MS	F	MS	F
Location (L)	2	123.58		10.33		1346.34	
P vs. Fs	1	241.05	12.94**	8.17	0.96n.s.	2688.17	10.31**
Fs	1	6.12	0.33n.s.	12.50	1.48n.s.	4.50	0.02n.s.
Residual	9	18.63		8.47		260.77	
Cochran's test			n.s.		n.s.		n.s.
Transformation			none		none		none

Significance levels: n.s. = $p > 0.05$; * = $p < 0.05$; ** = $p < 0.01$.

mean settle in very shallow waters (especially sparids; García-Rubies & Macpherson, 1995), coastal defences seem to constitute far more suitable structures than traditional artificial reefs for settlement/recruitment of such fishes (see Guidetti, 2004). Coastal defences, therefore, might play the traditionally perceived role of artificial reefs in providing proper habitats for adult fish and, in the meantime, favour their recruitment.

D. sargus, *D. vulgaris* and *S. aurata* are the most important predators of small and adult sea urchins in the Mediterranean Sea (Guidetti, in press; Sala, 1997), and are the only predatory fishes of echinoids present in the study area. Due to the fact that such predatory fishes were generally more abundant and larger at the protected breakwater, we would have expected lower sea urchin density over rocky substrates at the protected breakwater, as a consequence of higher fish predation rates. We

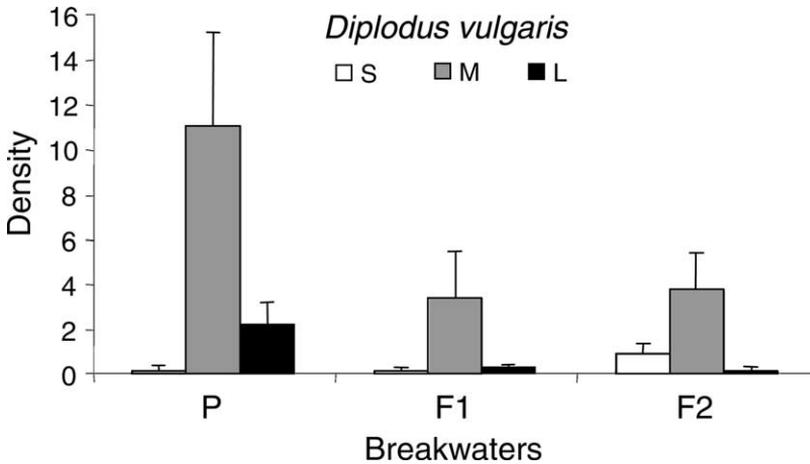


Fig. 5. Average density (\pm SE; number of individuals 125 m^{-2}) of *D. vulgaris* at the protected (P) and fished (F1, F2) breakwaters (data of the four sampling times cumulated), in relation to size (S: small; M: medium; L: large).

Table 4

Asymmetrical analyses of variance comparing density of the predatory fish *D. vulgaris* in relation to the size class (see Section 2), at one protected (P) and two fished breakwaters (Fs)

Source of variation	d.f.	Small		Medium		Large	
		MS	F	MS	F	MS	F
Location (L)	2	3.00		295.58		20.02	
P vs. Fs	1	1.50	1.37n.s.	590.05	4.36*	44.01	8.16**
Fs	1	4.50	4.13n.s.	1.12	0.008n.s.	0.03	0.006n.s.
Residual	45	1.09		135.44		5.39	
Cochran's test			*		*		**
Transformation			none		none		none

Significance levels: n.s. = $p > 0.05$; * = $p < 0.05$; ** = $p < 0.01$.

observed, instead, that the density of *P. lividus* did not differ between breakwaters characterised by significant differences in density/size of predatory fishes. Breakwaters in the study area, however, could be regarded as small rocky islands lying over muddy sand, i.e., the most common substrate in the study area. Predatory fishes could primarily feed upon other invertebrates associated with muddy-sandy substrates in the vicinity of the artificial structures, in contrast to natural rocky habitats where they are known to prey upon sea urchins (Sala & Zabala, 1996). Possible changes in the feeding habitats of predatory fishes at these peculiar artificial habitats could thus be elucidated with more specific studies (e.g., gut content analyses).

Due to the local and correlative nature of our study, we cannot conclude whether top-down processes may be more or less important in artificial rocky habitats than in

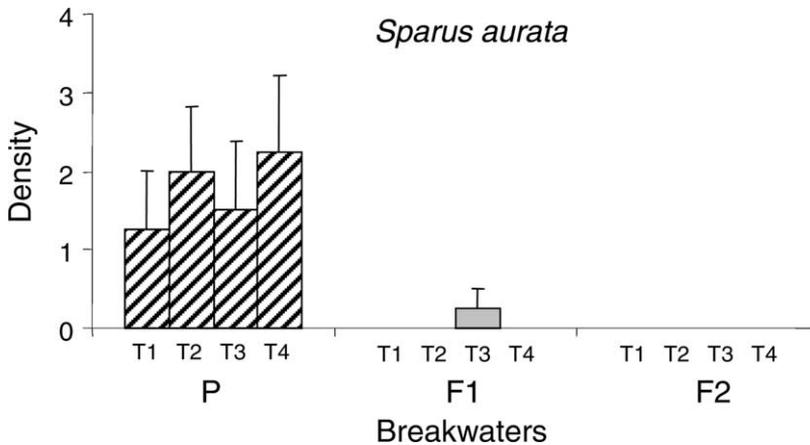


Fig. 6. Average density (\pm SE; number of individuals 125 m^{-2}) of the predatory fish *S. aurata* at the protected (P) and fished (F1, F2) breakwaters, in each of the four sampling times (T1, T2, T3 and T4).

Table 5

Asymmetrical analyses of variance comparing density of the predatory fish *S. aurata* at the sampling time T3, at one protected (P) and two fished breakwaters (Fs)

Source of variation	d.f.	T3	
		MS	F
Location (L)	2	0.62	
P vs. Fs	1	1.17	6.50*
Fs	1	0.06	0.33n.s.
Residual	9	0.18	
Cochran's test			n.s.
Transformation			Sqrt

Significance levels: n.s. = $p > 0.05$; * = $p < 0.05$; ** = $p < 0.01$.

natural reefs in the Mediterranean. The understanding of why in some cases (or places) predation effects reverberate down the entire community, whereas in other cases (or places) they do not, is nevertheless an intriguing challenge which could involve the assessment of possible thresholds in predator density/biomass for consequent effects on their preys. From this perspective, a meta-analysis from several Mediterranean sites revealed that negative correlations between densities of predatory fishes and sea urchins exist only when combined densities of *Diplodus* (i.e., *D. sargus* and *D. vulgaris*) exceed ≈ 15 individuals 100 m^2 (Guidetti and Sala, unpublished data). At Miramare, only in one sampling time this threshold has been clearly exceeded, while on average the *Diplodus* density was around the above mentioned threshold value. This could involve that density of predatory fish (and then the intensity of predation on sea urchins) is not usually high enough to cause significant effects on *P. lividus* populations. It is worth noting, moreover, that the values of *Diplodus* density observed at Miramare are approximately in the middle of the range

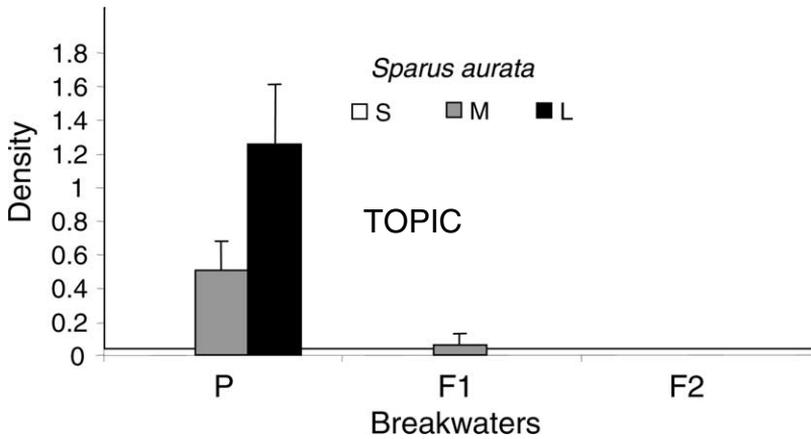


Fig. 7. Average density (\pm SE; number of individuals 125 m^{-2}) of *S. aurata* at the protected (P) and fished (F1, F2) breakwaters (data of the four sampling times cumulated), in relation to size (S: small; M: medium; L: large).

Table 6

Asymmetrical analyses of variance comparing density of the predatory fish *S. aurata* in relation to the size class (see Section 2), at one protected (P) and two fished breakwaters (Fs)

Source of variation	d.f.	Medium	
		MS	F
Location (L)	2	1.19	
P vs. Fs	1	2.34	11.7**
Fs	1	0.03	0.70n.s.
Residual	45	0.20	
Cochran's test			**
Transformation			none

Significance levels: n.s. = $p > 0.05$; * = $p < 0.05$; ** = $p < 0.01$.

of values reported from other Mediterranean MPAs (ranging from less than 1 to more than 50 individuals per 100 m^{-2} ; Guidetti and Sala, unpublished data), which suggests that, on the whole, artificial rocky substrates could allow fish population recovery in a way similar to natural rocky habitats.

Before drawing any conclusion about predation rates on sea urchins in artificial defence structures, more detailed studies involving, e.g., data about population structure and biomass of sea urchins, along with appropriate experimental work, should be done. Nevertheless, the present study provides some clues about the ecology of artificial defence structures originally constructed for other purposes than fish population recovery, that could have implications for management. Similar structures, in fact, are becoming an important component of the current transformation of coastal landscapes worldwide (Bacchiocchi & Airoidi, 2003; Chapman & Bulleri, 2003; Glasby & Connell, 1999), and their proliferation is occurring at high rates also in several

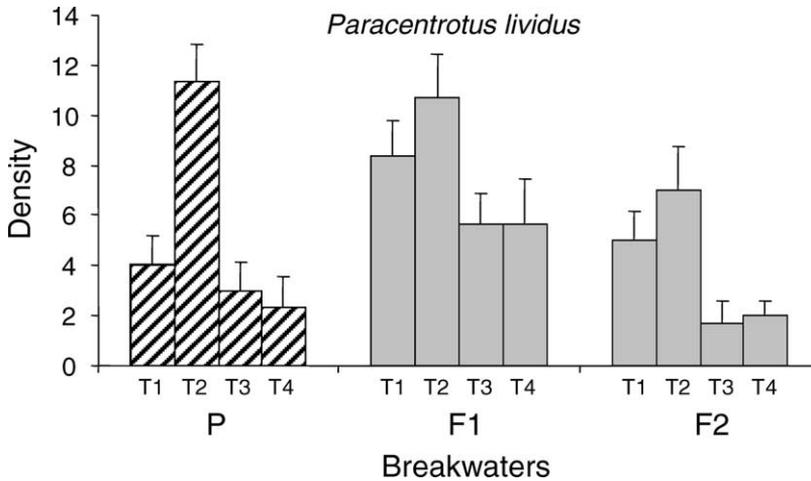


Fig. 8. Average density (\pm SE; number of individuals 20 m⁻²) of the sea urchin *P. lividus* at the protected (P) and fished (F1, F2) breakwaters, in each of the four sampling times (T1, T2, T3 and T4).

Table 7

Asymmetrical analyses of variance comparing density of the sea urchin *P. lividus* at each of the four sampling times (T1, T2, T3, T4), at one protected (P) and two fished breakwaters (Fs)

Source of variation	d.f.	T1		T2		T3		T4	
		MS	F	MS	F	MS	F	MS	F
Location (L)	2	15.44		16.33		12.44		12.33	
P vs. Fs	1	14.22	2.97n.s.	12.50	1.52n.s.	0.89	0.04n.s.	4.51	0.15n.s.
Fs	1	16.67	3.48n.s.	20.17	2.45n.s.	24.00	6.76*	20.16	0.69n.s.
Residual	6	4.78		8.22		3.55		29.33	
Cochran's test			n.s.		n.s.		n.s.		n.s.
Transformation			None		none		none		none

Significance levels: n.s. = $p > 0.05$; * = $p < 0.05$; ** = $p < 0.01$.

coastal zones of the Mediterranean, especially in those areas where rocky foreshores and/or natural shelters for boaters are absent. We should thus improve as much as possible the knowledge about their ecology and their overall impact, also in order to include them within management programs. Although the available body of scientific evidence should be obviously widened to other regions, and probably deserves long term studies and proper experiments, the results of this investigation suggest that appropriate management and protection of defence structures could promote population recovery of fish, with potential benefits for local adjacent fisheries.

Acknowledgements

This study was partially supported by the 'Sistema Afrodite' and 'Venere' projects funded by ICRAM and CoNISMa, respectively. Many thanks are due to F. Bulleri

and T.J. Willis for their critical reading of an early version of the ms, to two anonymous referees for their constructive criticism, to L. Verginella for her help in collecting data of fish, and the staff of the Miramare natural marine reserve for the field assistance.

References

- Bacchiocchi, F., & Airoidi, L. (2003). Distribution and dynamics of epibiota on hard structures for coastal protection. *Estuarine, Coastal and Shelf Science*, *56*, 1157–1166.
- Bohnsack, J. A., Johnson, D. L., & Ambrose, R. F. (1991). Ecology of artificial reef habitats and fishes. In W. Jr., Seaman & L. M. Sprague (Eds.), *Artificial habitats for marine and freshwater fisheries* (pp. 61–107). San Diego: Academic Press.
- Bulleri, F., Benedetti-Cecchi, L., & Cinelli, F. (1999). Grazing by the sea urchins *Arbacia lixula* L. and *Paracentrotus lividus* Lam. in the Northwest Mediterranean. *Journal of Experimental Marine Biology and Ecology*, *241*, 81–95.
- Bulleri, F., & Chapman, M. G. (in press). Intertidal assemblages on artificial and natural structures in marinas on the west coast of Italy. *Marine Biology*.
- Castellarin, C., Visintin, G., & Odorico, R. (2001). L'ittiofauna della riserva naturale marina di Miramare (Golfo di Trieste, Alto Adriatico). *Annales, Series Historia naturalis*, *11*, 207–216.
- Chapman, M. G., & Bulleri, F. (2003). Intertidal seawalls – new features of landscape in intertidal environments. *Landscape Urban Planning*, *62*, 159–172.
- CIESM (1999). Scientific design and monitoring of Mediterranean marine protected areas. CIESM Workshop Series, no. 8, Monaco, 64pp.
- CIESM (2002). Erosion littorale en Méditerranée occidentale: dynamique, diagnostic et remèdes. CIESM Workshop Series, no. 18, Monaco, 104pp.
- Coleman, M. A., & Connell, S. D. (2001). Weak effects of epibiota on the abundances of fishes associated with pier pilings in Sydney Harbour. *Environmental Biology of Fishes*, *61*, 231–239.
- Connell, S. D. (2000). Floating pontoons create novel habitats for subtidal epibiota. *Journal of Experimental Marine Biology and Ecology*, *247*, 183–194.
- Connell, S. D. (2001a). Urban structures as marine habitats: an experimental comparison of the composition and abundance of subtidal epibiota among pilings, pontoons and rocky reefs. *Marine Environmental Research*, *52*, 115–125.
- Connell, S. D. (2001b). Predatory fish do not always affect the early development of epibiotic assemblages. *Journal of Experimental Marine Biology and Ecology*, *260*, 1–12.
- Dayton, P. K., Thrush, S. F., Agardy, T. M., & Hofman, R. J. (1995). Environmental effects of marine fishing. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *5*, 205–232.
- Dayton, P. K., Thrush, S., & Coleman, F. (2002). *Ecological effects of fishing in marine ecosystems of the United States* (pp. 45). Arlington, Virginia: Pew ocean Commission.
- Dethier, M. N., Mc Donald, K., & Strathmann, R. R. (2003). Mc Colonization and connectivity of habitat patches for coastal marine species distant from source populations. *Conservation Biology*, *17*, 1024–1035.
- Elia, M. (2003). Analisi di popolamenti bentonici del subtidale di barriere artificiali dell'area marine protetta di Miramare (Trieste). Thesis, University of Lecce, 94 pp.
- Fischer, W., Bauchot, M. L., & Schneider, M. (1987). Fiches FAO d'identification des espèces pour les besoins de la pêche. Méditerranée et Mer Noire. Zone 37. II. Vertébrés (pp.761–1530). FAO, Rome.
- Francour, P. (1991). The effect of protection level on a coastal fish community at Scandola, Corsica. *Revue Ecologie (Terre Vie)*, *46*, 65–81.
- García-Charton, J. A., Pérez-Ruzafa, Á, Sánchez-Jerez, P., Bayle-Sempere, J. T., Reñones, O., & Moreno, D. (2004). Multi-scale spatial heterogeneity, habitat structure, and the effect of marine reserves on Western Mediterranean rocky reef fish assemblages. *Marine Biology*, *144*, 161–182.

- García-Rubies, A., & Macpherson, E. (1995). Substrate use and temporal pattern of recruitment in juvenile fishes of the Mediterranean littoral. *Marine Biology*, *124*, 35–42.
- García-Rubies, A., & Zabala, M. (1990). Effects of total fishing prohibition on the rocky fish assemblages of Medes Islands marine reserve (NW Mediterranean). *Scientia Marina*, *54*, 317–328.
- Gell, F. R., & Roberts, C. M. (2003). Benefits beyond boundaries: the fishery effects of marine reserves. *Trends in Ecology and Evolution*, *18*, 448–455.
- Glasby, T. M. (1997). Analysing data from post-impact studies using asymmetrical analyses of variance: a case study of epibiota on marinas. *Australian Journal of Ecology*, *22*, 448–459.
- Glasby, T. M. (1999). Effects of shading on subtidal epibiotic assemblages. *Journal of Experimental Marine Biology and Ecology*, *234*, 275–290.
- Glasby, T. M., & Connell, S. D. (1999). Urban structures as marine habitats. *Ambio*, *28*, 595–598.
- Guidetti, P., Fraschetti, S., Terlizzi, A., & Boero, F. (2003). Distribution patterns of sea urchins and barrens in shallow Mediterranean rocky reefs impacted by the illegal fishery of the rock-boring mollusc *Lithophaga lithophaga*. *Marine Biology*, *143*, 1135–1142.
- Guidetti, P., Terlizzi, A., & Boero, F. (2004). Effects of the edible sea urchin, *Paracentrotus lividus*, fishery along the Apulian rocky coasts (SE Italy, Mediterranean Sea). *Fisheries Research*, *66*, 287–297.
- Guidetti, P. (2004). Fish assemblages associated with coastal defence structures in southwestern Italy (Mediterranean Sea). *Journal of the Marine Biological Association of the United Kingdom*, *84*, 669–670.
- Guidetti, P. (in press). Consumers of sea urchins (*Paracentrotus lividus* and *Arbacia lixula*) in shallow Mediterranean rocky reefs. *Helgoland Marine Research*.
- Guidetti, P., Verginella, L., Viva, C., Odorico, R., & Boero, F. (submitted). Effects of protection on fish assemblages, and comparison of two visual census techniques in shallow artificial rocky habitats: a case study at the marine protected area of Miramare (northern Adriatic Sea). *Journal of the Marine Biological Association of the United Kingdom*.
- Halpern, B. S. (2003). The impact of marine reserves: do reserves work and does reserve size matter? *Ecological Applications*, *13*, 117–137.
- Halpern, B. S., & Warner, R. R. (2002). Marine reserves have rapid and lasting effects. *Ecology Letters*, *5*, 361–366.
- Harmelin-Vivien, M. L., Harmelin, J. G., Chauvet, C., Duval, C., Galzin, R., Lejeune, P., Barnabe, G., Blanc, F., Chevalier, R., Duclerc, J., & Lasserre, G. (1985). Evaluation des peuplements et populations de poissons. *Méthodes et problèmes. Revue Ecologie (Terre Vie)*, *40*, 467–539.
- Harmelin, J. G., Bachel, F., & Garcia, F. (1995). Mediterranean marine reserves: fish indices as tests of protection efficiency. *PSZN: Marine Ecology*, *16*, 233–250.
- Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J., & Warner, R. R. (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science*, *293*, 629–638.
- Jennings, S., & Kaiser, M. J. (1998). The effects of fishing on marine ecosystems. *Advances in Marine Biology*, *34*, 201–352.
- Jouvenel, J. Y., & Pollard, D. A. (2001). Some effects of marine reserve protection on the population structure of two spearfishing target-fish species, *Dicentrarchus labrax* (Moronidae) and *Sparus aurata* (Sparidae), in shallow inshore waters, along a rocky coast in the northwestern Mediterranean Sea. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *11*, 1–9.
- Micheli, F., Polis, G. A., Boersma, P. D., Hixon, M. A., Norse, E. A., Snelgrove, P. V. R., & Soule, M. E. (2001). In M. E. Soule' & G. H. Orians (Eds.), *Conservation biology: research priorities for the next decade* (2nd ed., pp. 31–57). Washington, DC: Island Press.
- Myers, R. A., & Worm, B. (2003). Rapid worldwide depletion of predatory fish communities. *Nature*, *423*, 280–283.
- Paine, R. T. (1980). Food webs, linkage, interaction strength, and community infrastructure. *Journal of Animal Ecology*, *49*, 667–685.
- Polunin, N. V. C., & Roberts, C. M. (1997). Greater biomass and value of target coral-reef fishes in two small Caribbean marine reserves. *Marine Ecology Progress Series*, *100*, 167–176.

- Relini, G., Relini, M., Torchia, G., & Palandri, G. (2002). Ten years of censuses of fish fauna on the Loano artificial reef. *ICES Journal of Marine Science*, 59, 132–137.
- Rilov, G., & Benayahu, Y. (1998). Vertical artificial structures as an alternative habitats for coral reef fishes in disturbed environments. *Marine Environmental Research*, 45, 431–451.
- Sala, E. (1997). Fish predators and scavengers of the sea urchin *Paracentrotus lividus* in protected areas of the north-western Mediterranean Sea. *Marine Biology*, 129, 531–539.
- Sala, E., & Zabala, M. (1996). Fish predation and the structure of the sea urchin *Paracentrotus lividus* populations in the NW Mediterranean. *Marine Ecology Progress Series*, 140, 71–81.
- Sala, E., Boudouresque, C. F., & Harmelin-Vivien, M. (1998). Fishing, and the structure of algal assemblages: evaluation of an old but untested paradigm, trophic cascades. *Oikos*, 82, 425–439.
- Shears, N. T., & Babcock, R. C. (2002). Marine reserves demonstrate top-down control of community structure on temperate reefs. *Oecologia*, 132, 131–142.
- Shears, N. T., & Babcock, R. C. (2003). Continuing trophic cascade effects after 25 years of no-take marine reserve protection. *Marine Ecology Progress Series*, 246, 1–16.
- Steneck, R. S. (1998). Human influences on coastal ecosystems: does overfishing create trophic cascades? *Trends in Ecology and Evolution*, 13, 429–430.
- Steneck, R. S., & Carlton, J. T. (2001). Human alterations of marine communities. Students beware. In M. D. Bertness, S. D. Gaines, & M. E. Hay (Eds.), *Marine community ecology* (pp. 445–468). Sunderland, USA: Sinauer Associates, Inc.
- Underwood, A. J. (1994). On beyond BACI: sampling designs that might reliably detect environmental disturbances. *Ecological Applications*, 4, 3–15.
- Underwood, A. J. (1997). *Experiments in ecology: their logic design and interpretation using analysis of variance*. Cambridge: University Press.
- Vacchi, M., Bussotti, S., Guidetti, P., & La Mesa, G. (1998). Study on the coastal fish assemblage in Ustica Island's Marine Reserve (Southern Tyrrhenian Sea). *Italian Journal of Zoology*, 65, 281–286.
- Willis, T. J., Millar, R. B., & Babcock, R. (2003). Protection of exploited fish in temperate regions: high density and biomass of snapper *Pagrus auratus* (Sparidae) in northern New Zealand marine reserves. *Journal of Applied Ecology*, 40, 214–227.
- Wilson, K. D. P., Leung, A. W. Y., & Kennish, R. (2002). Restoration of Hong Kong fisheries through deployment of artificial reefs in marine protected areas. *ICES Journal of Marine Science*, 59, 157–163.
- Witman, J. D., & Dayton, P. K. (2001). Rocky subtidal communities. In M. D. Bertness, S. D. Gaines, & M. E. Hay (Eds.), *Marine community ecology* (pp. 339–366). Sunderland, USA: Sinauer Associates, Inc.