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Effects of the edible sea urchin, *Paracentrotus lividus*, fishery along the Apulian rocky coast (SE Italy, Mediterranean Sea)

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Abstract

The effects of the sea urchin *Paracentrotus lividus* (Lamarck) fishery, and possible indirect influences on the co-occurring (but not harvested) sea urchin *Arbacia lixula* (L.), were investigated in shallow rocky reefs in southern Apulia (SE Italy, Mediterranean Sea). Density, size (test diameter), biomass and size-frequencies of both species were assessed in summer 2001 at one location heavily impacted by *P. lividus* fishery (recreational and commercial), and at three controls. Average sea urchin density (pooling *P. lividus* and *A. lixula*), and density of *P. lividus* did not significantly differ between the impacted location and the controls, while average density of *A. lixula* was significantly greater at the location where *P. lividus* was fished. The average size of *P. lividus* was significantly lower at the exploited location than at the controls, whereas that of *A. lixula* did not differ. Total sea urchin biomass (pooling *P. lividus* and *A. lixula*) did not differ between the exploited location and the controls. The average biomasses of *P. lividus* and of *A. lixula*, conversely, were lower and greater, respectively, at the fished location than at the control sites. In some cases, distribution patterns of the investigated variables (i.e., density, size and biomass) were affected by fishery at the spatial scale of a few metres (i.e., between replicates). At the controls, *P. lividus* larger than 4 cm (the threshold size for commercialisation) accounted for about one third of the populations surveyed, whereas at the fished location specimens of this size were rare. Size-frequency distributions of *A. lixula* were fairly similar at the fished and control locations. Although appropriate experimentation would be needed to confirm these patterns, the present study provides suggestive evidence that intense *P. lividus* harvesting may cause reduction in average size and biomass of this echinoid because of the selective harvesting of largest specimens. Densities of *P. lividus* at fished and control areas, instead, did not change, which suggests that *P. lividus* populations subject to fishing could have the potential to recover by appropriate management policies (e.g., catch quotas). Indirect effects of the *P. lividus* harvesting on the co-occurring *A. lixula* suggest a possible competitive dominance by *P. lividus* in the absence of selective fishing on this latter species, and a sort of compensation when it is fished. Implications for management of *P. lividus* exploitation, and for possible ecological consequences of the sea urchin fishery, are also discussed.
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Keywords: Sea urchins; Fishery; *Paracentrotus lividus*; *Arbacia lixula*; Mediterranean Sea

1. Introduction

Data on intertidal and shallow subtidal invertebrates of commercial importance are generally scant (Hobday et al., 2001). Many marine invertebrates, in the past, were not considered to be liable of overfishing

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due to their biological features (e.g., comparatively fast growth rates, high fecundity and wide-scale larval dispersal; see Tegner et al., 1996 and references therein). Several studies, however, showed that increased pressure on near shore invertebrate species might lead to overfishing in many regions of the world (Tegner, 1993; Pfister and Bradbury, 1996).

Several sea urchin species are intensively harvested in many regions for the delicacy of their gonads (Lawrence, 2001). In past decades, sea urchin fishery was generally practised by a few people using artisanal methods (Boudouresque, 1987) and the product was sold locally. More recently, however, mainly due to the increased demand from Japanese markets (Yokota, 2002), this fishery has expanded in new regions and has undergone considerable modernisation (Perry et al., 2002). The consequent overfishing of some of commercial sea urchins (Régis, 1988; FAO, 1996a; Kalvass and Hendrix, 1997; Lesser and Walker, 1998) led to the enforcement of specific management policies (e.g., catch quotas, rotational fishery, aquaculture) to allow stock restoration and/or mitigation of the impact on natural populations (San Martin, 1987; Botsford et al., 1999; Yokota et al., 2002).

Tortonese (1965) described aspects of the distribution, morphology and biology of the edible sea urchin *P. lividus* (Lamarck) and of the often co-occurring non-edible *Arbacia lixula* (L.) in shallow subtidal rocky reefs along the coasts of the Mediterranean Sea, where *P. lividus* is intensively collected (Le Direac'h et al., 1987; Boudouresque and Verlaque, 2001; Yokota et al., 2002). In particular, along the Apulian rocky shores (SE Italy), *P. lividus* is mainly harvested in shallow rocky reefs, easily accessible from land and close to small harbours. The laws regulating *P. lividus* fisheries are rarely enforced, so that authorised fishermen and poachers (by means of SCUBA), as well as occasional collectors, freely exploit local populations.

Many of the available reports on *P. lividus* fishery concern the French coast and focus on the availability, depletion and management of natural stocks in terms of economic resource (see Boudouresque, 1987). The ecological effects of this fishery directly on populations of *P. lividus*, and indirectly on the co-occurring *A. lixula*, are poorly investigated. Sea urchin grazing, in addition, may cause important changes in the structure of epibenthic assemblages in several coastal habitats worldwide (Dayton, 1985; Elner and Vadas, 1990;

Estes and Duggins, 1995; Sala et al., 1998; Balch and Scheibling, 2000). In particular, in Southern Apulia, sea urchin grazing has been hypothesised to maintain barrens in rocky substrates primarily impacted by the destructive fishery of the endolithic date-mussel, *Lithophaga lithophaga* (Fanelli et al., 1994; Fraschetti et al., 2001; Guidetti et al., 2003). This suggests that sea urchins exert a role in affecting distribution patterns of benthic assemblages, and dynamics of recolonisation in rocky substrates (Boero et al., 1993; Fanelli et al., 1999). There are, however, no data about a possible different role of the two sea urchins in maintaining barrens, which would involve an interaction between date-mussel and sea urchin fisheries. From this perspective, only few studies considered the indirect effects that variations in sea urchin populations (e.g., due to fishery) could determine on hard bottom benthic communities (McClanahan and Sala, 1997; Benedetti-Cecchi et al., 1998). All the above issues support the statement by Emlet (2002) who suggested that, despite the huge amount of literature about the ecological role of sea urchins, there are so far few studies about fisheries ecology of echinoids.

This study, therefore, is aimed at providing information on: (1) the effects of *P. lividus* harvesting on average density, size and biomass, and population structure of this target species, and (2) the indirect effects on the co-occurring *A. lixula* in shallow sublittoral Mediterranean rocky reefs.

2. Materials and methods

2.1. Study area

Surveys were carried out in summer 2001 at four locations along the southern Apulian coast (SE Italy; Fig. 1): Porto Selvaggio, Torre del Serpe, Torre Minervino, and Ciolo. Porto Selvaggio (indicated as F in figures and tables) is subject to intense sea urchin fishing (mainly during summer) since it is a tourist site very easily accessible from land. The control locations (Torre del Serpe, Torre Minervino and Ciolo: indicated as C1, C2 and C3, respectively), instead, are inaccessible from land, and not affected (except sporadically) by sea urchin harvesting. The shallow sublittoral habitats at all studied locations are generally calcareous plateaus alternated with sub-vertical walls.
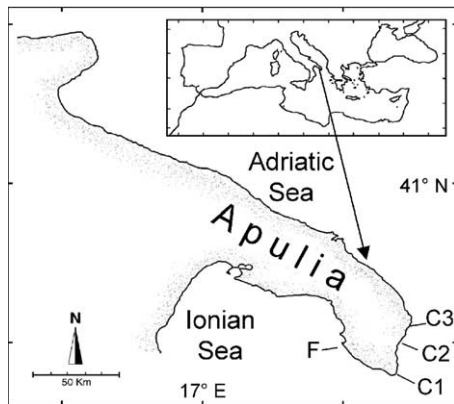
Fig. 1. Map of the studied area. Fished location: Porto Selvaggio (F); controls: Ciolo (C1); Torre Minervino (C2); Torre del Serpe (C3).

From the surface to about 3–5 m depth, the rocky substrate is generally covered by macroalgae (chiefly articulated Corallinaceae, Dictyotales and *Cystoseira* spp.) and sessile invertebrates (e.g., sponges, hydroids, bryozoans) alternated with barrens. From 5–8 to about 12–15 m depth there is a gently sloped plateau covered by photophilic algae with numerous medium–large boulders. Overall, therefore, there are no macroscopic differences among locations in terms of rocky substrate, physical heterogeneity and macroalgal cover.

2.2. Sampling design and data collection

Sea urchin density was sampled at three sites (randomly selected at about 100–300 m apart from each other) within each of the four studied locations (one exploited and three controls), in rocky reef habitats at about 5–6 m depth, where the impact of sea urchin fishery is maximum (Catoira Gòmez, 1992). All counts were made approximately around 12 a.m. At each site, sea urchins were counted in 10 replicate quadrats. *P. lividus* and *A. lixula* were counted in different series of quadrats to avoid dependence of data. A random sample of individuals >1 cm in diameter (test without spines; $n = 115$ for *P. lividus*; $n = 40$ for *A. lixula*) within quadrats from each location was also measured by means of a vernier calliper. This method is suitable for assessing the fraction of sea urchins >1 cm test diameter (Turon et al., 1995; Sala and Zabala, 1996). Average biomass per square meter was estimated based

on size/wet weight relationships reported by Ros and Niell (1981) for both species. Data about diameters of the two sea urchins were grouped into size classes of 1 cm to construct frequency distributions (Sala and Zabala, 1996).

2.3. Statistical treatment of data

Asymmetrical analyses of variance (see Underwood, 1993, 1994; Glasby, 1997 for details) were used to test for differences in the average abundance, size, and biomass of sea urchins among the fished and the three control locations. In the analyses, the 'impact vs controls' (F vs Cs) term was considered as fixed, while the term 'among controls' was random. Besides the effects of fishing, differences in average density were assessed at two spatial scales (with sites random and nested within locations), whereas differences in average size and biomass were tested only at the spatial scale of locations. Data were first analysed considering sites from both fished and control locations as simply nested within locations (for average density), or considering the four locations as simply random (for average size and biomass). A second analysis for each investigated variable was then done only on data associated with the controls. The variance associated with the exploited location was then calculated by subtracting the sums of squares of the second analysis from those of the first. In the analyses, the overall residual has been partitioned into the residual of the location impacted by sea urchin fishery, and that of controls. A 2-tailed *F*-test was then used to test whether the impact affected the density, size, and biomass of sea urchins at the scale of quadrats. When this test was significant, these terms (i.e., Res(F) and Res(Cs)) were used as denominators to test for sites at the fished location (S(F)) and at the control locations (S(Cs)), respectively. The impact at the scale of sites between the fished location and the controls was considered significant whenever either S(F) or S(Cs) was significant and the other was not. Whenever these terms were both significant or not, a 2-tailed *F*-test was done to compare spatial patterns at the scale of sites between the fished location and the controls. Prior to analysis, the homogeneity of variance was tested by Cochran's test and, when necessary, data were appropriately transformed. If transformations did not produce homogeneous variances, ANOVA was

used, nevertheless, on untransformed data after setting $\alpha = 0.01$ in order to compensate for the increased likelihood of Type I error (Underwood, 1997). The analyses were performed using the GMAV5 software package (University of Sydney, Australia).

3. Results

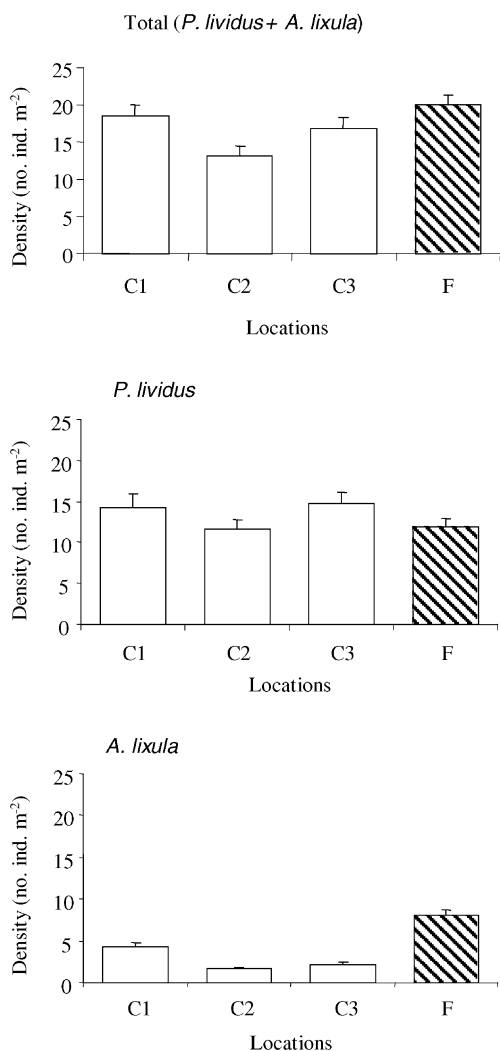Pooled densities of *P. lividus* and *A. lixula*, and of *P. lividus* alone (Fig. 2; Table 1) varied significantly

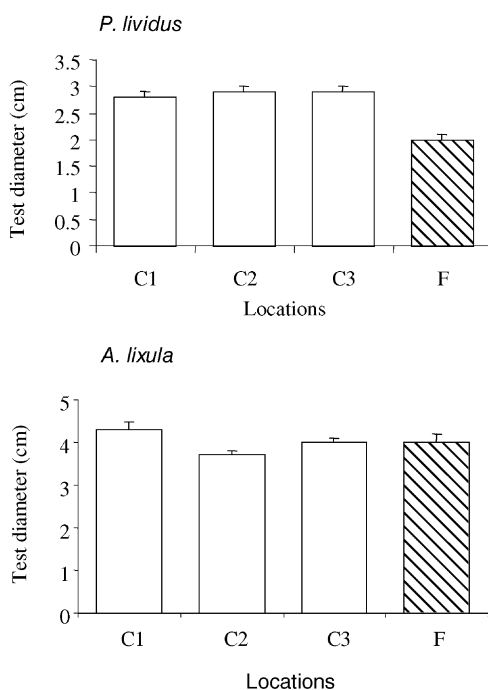Fig. 2. Mean density (±S.E.) of sea urchins at the fished location (F) and three controls (C1, C2, C3).

Fig. 3. Mean test diameter (±S.E.) of sea urchins >1 cm at the fished location (F) and three controls (C1, C2, C3).

among sites at the controls, but not at the fished location. Mean density of *A. lixula* was significantly greater at the fished location than at the controls (Fig. 2; Table 1), and variability at the scale of a few metres (i.e., among quadrats) significantly differed between fished and control locations, as shown by the significant 2-tailed *F*-test.

Average size of *P. lividus* was significantly lower at the fished location than at the controls (Fig. 3; Table 2). In addition, the significant 2-tailed *F*-test revealed that there were also differences in the variability in size at the scale of quadrats. Average size of *A. lixula* did not significantly differ between fished location and the controls (Fig. 3; Table 2), but the fishery of *P. lividus* appeared to influence the spatial distribution of *A. lixula* of different size at the spatial scale of quadrats (2-tailed *F*-test significant).

Total sea urchin biomass (*P. lividus* and *A. lixula* pooled) did not show significant differences between the fished location and the controls (Fig. 4; Table 3). Variability in the distribution patterns of total biomass, however, differed between quadrats (2-tailed *F*-test

Table 1

Asymmetrical ANOVAs comparing density (no. of individuals per square meter) of sea urchins at three sites within one location impacted by *P. lividus* fishery (F) and three controls (Cs)

Source	d.f.	Whole density		*P. lividus*		*A. lixula*	
		MS	*F*	MS	*F*	MS	*F*
Location = L	3	263.00		244.09		258.10	
F vs Cs	1	327.40	1.42 ns	55.22	0.58 ns	664.22	12.17*
Cs	2	230.80	1.36 ns	94.43	0.73 ns	55.03	6.56*
Sites (L) = S(L)							
S (L)	8	133.90	2.32*	100.42	1.93 ns	7.35	1.13 ns
S (F)	2	28.30	0.49 ns	12.03	0.23 ns	4.23	0.37 ns
S (Cs)	6	169.10	2.93*	129.89	2.50*	8.39	1.74 ns
Residual	108	57.63		51.96		6.50	
Residual (F)	27	52.72		33.62		11.53	
Residual (Cs)	81	59.26		58.08		4.82	
2-Tailed *F*-test							
Replicates			1.12 ns		1.73 ns		2.39**
Sites			–		–		1.98 ns
Cochran's test			ns		ns		ns
Transform			Nil		Nil		Nil

ns; $P > 0.05$.
* $P < 0.05$.
** $P < 0.01$.

significant) between the fished location and the controls. Opposite patterns have been shown considering each of the two species in relation to harvesting. The biomass of the target sea urchin *P. lividus*, in fact, was

Table 2

Asymmetrical ANOVAs comparing size (test diameter without spines) of sea urchins at one location impacted by *P. lividus* fishery (F) and three controls (Cs)

Source	d.f.	*P. lividus*		d.f.	*A. lixula*	
		MS	*F*		MS	*F*
Location = L	3	23.54		3	1.86	
F vs Cs	1	70.50	59.25**	1	0.01	0.01 ns
Cs	2	0.05	0.04 ns	2	2.79	3.55 ns
Residual	456	1.19		156	1.03	
Residual (F)	114	0.56		39	1.76	
Residual (Cs)	342	1.40		117	0.79	
2-Tailed *F*-test						
Replicates			2.47**			2.24**
Cochran's test			*			*
Transform			Nil			Nil

ns; $P > 0.05$.
* $P < 0.05$.
** $P < 0.01$.

three to seven-fold lower at the fished location than at the controls (Fig. 4), and its spatial distribution was different also among quadrats (as indicated by the significant 2-tailed *F*-test) (Table 3). Conversely, the average biomass of *A. lixula* was two- to five-fold greater at the fished than at the control locations (Fig. 4; Table 3).

The *P. lividus* test size-frequency distribution was similar at the three control locations: the mode was formed by sea urchins from 2 to 3 cm, and individuals larger than 4 cm (threshold size for commercialisation) accounted for about one-third of the populations at the three Cs (Fig. 5). At the fished location, instead, the modal class was formed by sea urchins from 1 to 2 cm in diameter, and specimens larger than 4 cm were rare. Size-distribution, moreover, clearly showed a far more evident decline of *P. lividus* in relation to increasing size at the fished location (Fig. 5). As regards *A. lixula*, test diameter frequency distributions were substantially comparable at the fished and control locations, with the modal class represented by individuals from 4 to 5 cm (Fig. 5). At the fished location, nonetheless, a second mode constituted by sea urchins from 2 to 3 cm in diameter, and some individuals >6 cm, were also recorded (Fig. 5).

Table 3
Asymmetrical ANOVAs comparing biomass ($g\,ww\,m^{-2}$) of sea urchins at one location impacted by *P. lividus* fishery (F) and three controls (Cs)

Source	d.f.	Whole biomass		*P. lividus*		*A. lixula*	
		MS	F	MS	F	MS	F
Location = L	3	1.29		118.54		138147.1	
F vs Cs	1	1.25	0.95 ns	241.77	14.66**	364125.8	54.49**
Cs	2	1.31	4.10*	56.92	2.70 ns	25157.74	3.76 ns
Residual	36	0.26		16.49		6682.23	
Residual (F)	9	0.09		2.81		9541.87	
Residual (Cs)	27	0.32		21.05		5729.02	
2-Tailed *F*-test							
Replicates			3.59*		7.49**		1.66 ns
Cochran's test			ns		ns		*
Transform			$x + 1$		$x + 1$		Nil

ns; $P > 0.05$.
  * $P < 0.05$.
  ** $P < 0.01$.

4. Discussion

4.1. Direct and indirect effects of sea urchin fishery, and ecological implications

In shallow Mediterranean rocky reefs, intense (and unregulated) exploitation reduces the mean size and biomass of *P. lividus*. In addition, at the location where *P. lividus* is harvested, the co-occurring non-edible sea urchin *A. lixula* showed significantly greater average density and biomass.

Confirming previous reports from other areas of the Mediterranean (Gras, 1987; Le Direac'h, 1987; Régis, 1988), the size of *P. lividus* was smaller at the fished location compared with controls. Gras (1987) observed a dramatic rarefaction of large-sized *P. lividus* after 2 years of intense fishing in some populations near Marseille (France), stressing that an excessive exploitation may cause deterioration of the economic value of the resource. In the present study, harvesting of *P. lividus* reflected also in population structures from fished and control locations. Since humans selectively collect the largest sea urchins (usually >4 cm), large-sized *P. lividus* were rare at the exploited locations. This fundamentally distinguishes humans from other (chiefly fish) predators, usually feeding upon *P. lividus* of intermediate size (around 2–4 cm; Sala and Zabala, 1996). *P. lividus* size-distributions in the presence of intense fish predation are usually bimodal, as juveniles (<2 cm) remain sheltered and specimens larger than 4–5 cm tend to progressively escape from predation, as generally happens for other sea urchin species in other regions (see Scheibling, 1996 for a review). Population structures like those observed in southern Apulia, instead, show that human harvesting may cause decrease in large-sized *P. lividus*, truncating the age structure of harvested populations.

Gras (1987) and Le Direac'h (1987) invoked intense harvesting to explain a marked reduction in average densities of *P. lividus* along the Mediterranean French coast. In the present study, instead, mean density of *P. lividus* appeared to be unaltered by the fishery. Whether this discrepancy is due to actual differences or to an artefact due to the implementation of different methods for measuring sea urchin density remains unknown. Previous reports chiefly perceived the decline of *P. lividus* populations in terms of depletion of the economic resource rather than of ecological consequences on harvested populations, possibly focusing mainly on sea urchins of commercial size. The high reproductive potential of *P. lividus*, moreover, could explain why average density did not change between fished and unfished locations. Specimens become sexually mature at a far smaller size than the commercial one (~4 cm) (Lozano et al., 1995; Boudouresque and Verlaque, 2001), warranting the maintenance of

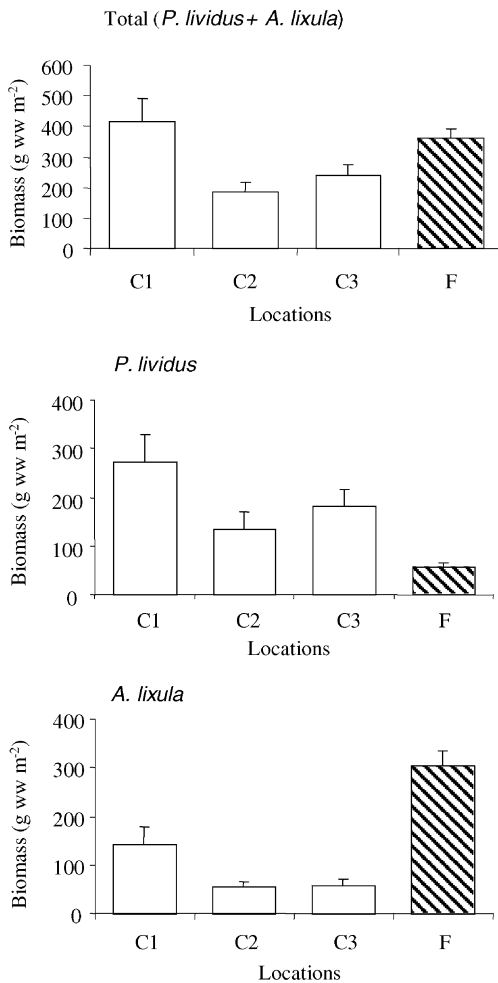## Total (*P. lividus* + *A. lixula*)

P. lividus

A. lixula

Fig. 4. Mean biomass (±S.E.) of sea urchins at the fished location (F) and three controls (C1, C2, C3).

populations (at least in terms of numbers) in areas characterised by intense exploitation.

Observations on average biomass of *P. lividus*, which was lower at the fished location than at controls, substantially reflect the fact that sea urchins did not differ in density, whereas they showed lower average size at the location subject to fishing. This outcome is consistent with previous studies that reported the decrease in *P. lividus* catches in the zones subject to intense exploitation (Boudouresque, 1987).

The *P. lividus* fishery has also possible indirect effects on the non-edible sea urchin *A. lixula*. The significantly greater density and biomass of *A. lixula* at

the location subject to intense harvesting of *P. lividus* suggests that *A. lixula* could benefit from this fishery. Although this is still debated by various authors (Benedetti-Cecchi et al., 1998; Boudouresque and Verlaque, 2001), the two species may partially share the same habitat and resources in the rocky sublittoral. This suggests that they could compete for space and food, but appropriate experiments should be planned to test for specific mechanisms of interaction between the two species. McClanahan and Shafir (1990) reported that, at low predation level, competitive dominance of the sea urchin *Echinometra mathaei* in Kenyan reef lagoons could reduce the abundance of subordinate sea-urchin species. When the competitive dominant is preferentially predated, instead, the abundance of the other sea urchins tends to increase. These patterns (McClanahan and Shafir, 1990; present study) seem to be consistent with the compensation mechanisms that occur when dominant competitors are negatively affected by some factors (an anthropogenic impact in our case) allowing competitive inferiors to increase in abundance due to competitive release (Micheli et al., 1999). From this perspective, sea urchin fishing was also hypothesised to shift the ratio of the commercially exploited *Strongylocentrotus franciscanus* to the smaller (and not harvested) *S. purpuratus* along the northwestern coast of America (Tegner and Dayton, 1977), a pattern confirmed by recent field observations (Dayton, pers. commun.). Our results and literature data, thus, support the hypothesis that shifts in relative species abundance, due to competitive interactions, can be mediated by (human, in our case) predation (Paine, 1966), and suggest to include competition and compensation mechanisms in ecosystem fisheries models (McClanahan and Sala, 1997).

The pattern observed for *A. lixula* (greater numbers and biomass at the fished location) may also have important ecological implications that probably deserve further investigation. Some authors, in fact, reported that *P. lividus* chiefly feeds upon fleshy erect algae and drifting detritus, while *A. lixula* preferentially grazes on encrusting coralline algae (Verlaque and Nédelec, 1983; Frantzis et al., 1988; Boudouresque and Verlaque, 2001). Benedetti-Cecchi et al. (1998) and Bulleri et al. (1999), conversely, provided evidence that the two species have less different ecological roles and diets than previously thought. They also showed that the grazing of *A. lixula* (in the absence
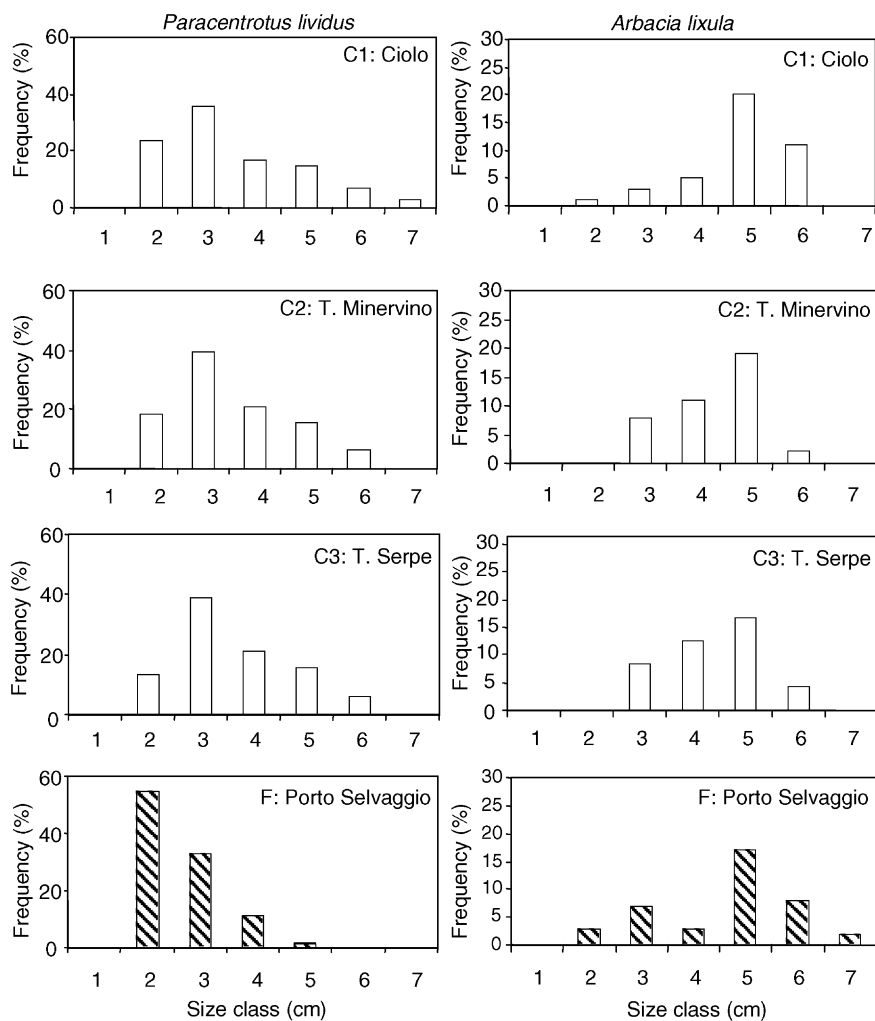
Fig. 5. Sea urchins >1 cm test size-frequency from the fished location (F) and three controls (C1, C2, C3). Size classes: 1, 0–1 cm; 2, 1–2 cm; 3, 2–3 cm; 4, 3–4 cm; 5, 4–5 cm; 6, 5–6 cm; 7, 6–7 cm.

of *P. lividus*) may be more effective than that of *P. lividus* in lowering the cover of several algal groups. All the above issues suggest the existence of complex interactions between the two sea urchins. Whatever the differences between the two species in terms of role or magnitude of effects, any change in their relative abundances (e.g., due to harvesting) could alter the structure of shallow subtidal benthic assemblages. This implies that sea urchin fishery could indirectly interfere with the natural distribution and dynamics of shallow epibenthic assemblages (e.g., transition between vegetated rocky reefs and coralline barrens)

(see Sala et al., 1998 for a review). In the specific case of southern Apulian rocky coasts, the sea urchin fishery could have important implications for the recovery of natural assemblages after desertification caused by date-mussel fishery (Fanelli et al., 1994), the increase of *A. lixula*, in the absence of *P. lividus*, leading to higher grazing pressures.

Many processes affect the size and viability of sea urchins populations, such as natural predation, competition, recruitment, early mortality, bacterial and parasite infestations, and food availability (Jangoux, 1987; McClanahan and Shafir, 1990; Turon et al., 1995; Sala

and Zabala, 1996; Lòpez et al., 1998). This study suggests that human predation may also influence patterns of distribution and population structure of sea urchins. Fisheries, as widely demonstrated, affect not only the target species, but also may have indirect consequences on non-target species, triggering complex processes whose effects, cascading up or down through the different trophic levels, could have the potential to shape benthic assemblages and alter biological processes (Steneck, 1998; Pinnegar et al., 2000; Jackson et al., 2001).

4.2. Implications for management

The fact that the size of *P. lividus* was smaller at the fished location than at the controls, whereas average densities of this sea urchin did not change, suggests that sea urchin harvesting, at least in the studied region, could not cause detrimental effects strictly considering the consequences on *P. lividus* populations, if the fishery was done in limited areas. Appropriate management strategies (e.g., the observance of the minimum fishable size, catch quotas, and rotational fishery) could be sufficient to manage the resource, avoiding overfishing, decline, and crisis of this fishery (Perry et al., 2002).

Indirect effects of this fishery, like changes in populations of other sea urchins like *A. lixula*, and other possible alterations to the environment (e.g. to sessile assemblages), however, once reliably assessed, would require other management measures to satisfy markets (e.g. sea urchin farming; Yokota et al., 2002 and papers therein), and on the other hand to prevent or mitigate any environmental alteration related to the *P. lividus* fishery. Bulleri et al. (1999) suggested that *P. lividus* should be exploited primarily in the areas where *A. lixula* is also present, since they evaluated that the two species may exert a comparable ecological role. In this case, potential impacts of *P. lividus* fisheries on other components of benthic assemblages could be mitigated by grazing of non-exploited species. Although the same authors recommended caution before embarking in any such management, since the assessment of the relative importance of *A. lixula* and *P. lividus* still requires experimental studies to be understood, there are also evidences that the two species could not be ecologically equivalent. This involves that harvesting of *P. lividus* should require careful

management to prevent possible population outbreaks and overgrazing of rocky substrates by *A. lixula*.

In conclusion, the results of this study represent a preliminary assessment of possible (direct and indirect) effects of *P. lividus* fishery in the Mediterranean. We have thus provided suggestive evidence of its effects comparing one location subject to *P. lividus* fishery and multiple controls. In such impact assessments, however, it is often difficult to find 'perfect' controls in the 'real' world, as theoretically required. Regardless of this problem, anyway, appropriate experimentation is still needed to fully elucidate the effects of sea urchin fishery. The global demand of sea urchin roes is increasing (Pfister and Bradbury, 1996; Perry et al., 2002; Yokota, 2002), which leads to predict a possible increase of sea urchins fishery before biological and ecological effects can be properly understood. This would be the typical frame where management actions fail due to uncertainty. For these reasons, a precautionary approach such as that proposed worldwide to all fisheries (recommending that fisheries should have effective management plan for monitoring and assessing the related effects; FAO, 1996b) should be employed also in the case of sea urchin harvesting.

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