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Italian marine reserve effectiveness: Does enforcement matter?

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ABSTRACT

Marine protected areas (MPAs) have become popular tools worldwide for ecosystem conservation and fishery management. Fish assemblages can benefit from protection provided by MPAs, especially those that include fully no-take reserves. Fish response to protection can thus be used to evaluate the effectiveness of marine reserves. Most target fish are high-level predators and their overfishing may affect entire communities through trophic cascades. In the Mediterranean rocky sublittoral, marine reserves may allow fish predators of sea urchins to recover and thus whole communities to be restored from coralline barrens to macroalgae. Such direct and indirect reserve effects, however, are likely to be related to the enforcement implemented. In Italy, many MPAs that include no-take reserves have been declared, but little effort has been spent to enforce them. This is a worldwide phenomenon (although more common in some regions than others) that may cause MPAs and reserves to fail to meet their targets. We found that 3 of 15 Italian marine reserves investigated had adequate enforcement, and that patterns of recovery of target fish were related to enforcement. No responses were detected when all reserves were analyzed as a whole, suggesting enforcement as an important factor to be considered in future studies

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particularly to avoid that positive ecological responses in properly managed reserves can be masked by neutral/negative results in paper parks. Positive responses were observed for large piscivores (e.g. dusky groupers) and sea urchin predators at reserves where enforcement was effective. Those reserves with low or null enforcement did not differ from fished areas.

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1. Introduction

Generally speaking, marine protected areas (MPAs) refer to portions of the coastline and/or sea where human activities, especially fishing, are restricted or banned (Agardy et al., 2003). As fish assemblages usually include many species targeted by fishing, they are primarily expected to benefit from protection within MPAs, especially those having no-take reserves (Dayton et al., 1995; Micheli et al., 2004; McClanahan et al., 2007). The evaluation of these benefits, in terms of increase in density and size of target fishes (Mosquera et al., 2000; Côté et al., 2001; Halpern, 2003; Micheli et al., 2004; Claudet et al., 2006; Guidetti and Sala, 2007), can be useful to assess the ecological effectiveness of reserves. Moreover, most target fishes are high-level predators and their functional extinction may cause community-wide changes (Sala et al., 1998; Jackson et al., 2001). Protection from fishing, therefore, may directly restore populations of target fishes and indirectly drive whole communities towards an unfished state (Sala et al., 1998; Shears and Babcock, 2002; Micheli et al., 2004; Bevilacqua et al., 2006; Guidetti, 2006). We use hereafter the term ‘ecological effectiveness’ of marine reserves to define the responses to protection from fishing encompassing direct and indirect effects.

Marine reserve studies have undoubtedly improved our understanding of the unfished state of ecosystems and target populations (Shears and Babcock, 2002; Guidetti and Sala, 2007). It is common wisdom, however, that a number of reserves do not meet their potential ecological objectives and that negative/neutral results in reserve studies are mostly underreported in the literature (Halpern and Warner, 2002). Failing reserves are attributable to causes like inappropriate design (Sala et al., 2002) or ineffective enforcement (Mora et al., 2006), which may be overlooked if negative/neutral results are not taken into account.

It is becoming increasingly recognized that a large proportion of marine reserves around the world receive ineffective enforcement. These are the so-called ‘paper parks’, where protection occurs only in theory (Mora et al., 2006). In such cases the use of proper sampling designs suggested by many authors to properly investigate reserve effectiveness (CIESM, 1999; Guidetti, 2002), e.g. by comparing replicated ‘reserve vs fished’ sites, is useless. In fact, the comparison ‘reserve vs fished’ sites makes sense only if protection really occurs. The scant information in many published studies about compliance and enforcement at the reserves investigated makes the interpretation of results uncertain. A major effort, therefore, is needed to make inferences about reserve effectiveness, paying special attention whenever data from both well-enforced and paper

reserves are pooled to extract general patterns (e.g. in meta-analyses). Pooling data from enforced reserves and from paper parks carries the risk of incorrectly downplaying the importance of reserves because neutral results from paper parks could mask the positive responses of well-enforced reserves.

In the Mediterranean Sea there has been a rush in recent years to establish MPAs and reserves (Juanes, 2001). In Italy there are currently 25 MPAs formally established (with more than 20 in the process of becoming established), ranging in size from 120 to more than 50,000 hectares in total surface area. Italian MPAs include one or more no-take/no-access zones (hereafter called ‘reserves’ in the text and formally defined as ‘A zones’ according to Italian law), surrounded by buffer zones (defined as ‘B and C zones’, where restrictions to human uses, including fishing, become progressively more lax) (Villa et al., 2002).

Previous studies that investigated fish response to protection within Italian marine reserves showed (1) positive effects (Vacchi et al., 1998; Guidetti et al., 2005; Guidetti, 2006) or neutral results (Tunesi et al., 2006) on fish density and size, and (2) no obvious patterns in terms of community shifts (Sala et al., 1998; Guidetti et al., 2005; Micheli et al., 2005; Guidetti, 2006; Guidetti and Sala, 2007). As regards the community shift, two target sea breams, i.e. *Diplodus sargus* and *Diplodus vulgaris*, have been identified as the most effective predators of sea urchins, with the latter being the most important grazer in rocky reefs (Sala et al., 1998). When released from predation control, sea urchins may increase in density and overfeed on macroalgae, which in turn may cause the transition from macroalgal beds to barrens (Sala et al., 1998). Since the recovery of sea breams (and other predator fish) was observed within reserves, along with lower urchin density and less extended barrens (Guidetti and Sala, 2007), *Diplodus* density can be assumed to be an index of the potential of reserves to recover from barrens to algal beds or to maintain flourishing algal beds.

In spite of the increasing number of MPAs in Italy, no general evaluations have been done to assess the ecological responses to protection from fishing. A nation-wide project, named “Sistema Afrodite”, was thus started in 2002 (Greco et al., 2004), with the aim of allowing a balanced assessment of the actual effectiveness of marine reserves in the country (including potential neutral/negative results).

This paper is intended to (1) assess the effects of different levels of enforcement on the ecological effectiveness of reserves (i.e. direct and indirect effects), and (2) highlight the risk of misinterpreting analyses about the effectiveness of multiple marine reserves whenever the enforcement is not properly taken into account.

Table 1 – Size (surface area of MPAs and related A zones), year of formal establishment, level of enforcement at the time when fish sampling was done and presence of rocky substrates at the depth ranges sampled by visual census (a, b and c indicate 4–7, 12–16 and 24–30 m depth, respectively) at the 15 MPAs investigated (see Section 2)

Marine protected area	Total surface (ha)	A zone surface (ha)	Establishment (year)	Level of enforcement	Presence of rocky substrate (depth range)	Number of visual census (A zone + fished = total)
Portofino	346	18	1998	High	a, b, c	96 + 288 = 384
Cinque Terre	2726	79	1997	Medium	a, b	64 + 128 = 192
Tavolara-Capo Coda Cavallo	15,357	529	1997	Medium	a, b, c	96 + 288 = 384
Sinis-Isola Mal di Ventre	25,673	529	1997	Low	a, b	96 + 128 = 224
Capo Carbonara	8598	332	1999	Low	a, b, c	96 + 288 = 384
Isole Ventotene-Santo Stefano	2799	410	1997	Medium	a	24 + 48 = 72
Punta Campanella	1539	181	1997	Low	a	24 + 48 = 72
Capo Rizzuto	14,721	585	1991	Medium	a, b, c	96 + 96 = 192
Isole Egadi	53,992	1067	1989	Low	a, b	32 + 64 = 96
Isola di Ustica	15,951	60	1986	Medium	a, b, c	96 + 192 = 288
Isole Ciclopi	623	35	1989	Low	a, b, c	96 + 128 = 224
Porto Cesareo	16,654	173	1997	Low	a, b	64 + 128 = 192
Torre Guaceto	2227	179	1991	High	a, b	64 + 64 = 128
Isole Tremiti	1466	180	1989	Medium	a, b, c	72 + 144 = 216
Miramare	120	30	1986	High	a	16 + 32 = 48

2. Materials and methods

2.1. Sampling areas and procedures

We examined fish response to protection in 15 Italian marine reserves (Mediterranean Sea; Fig. 1) during two to four sampling campaigns (depending on the reserve) carried out at

all 15 locations (i.e. each spatial unit including reserve and nearby fished sites) between May 2002 and October 2003.

Replicated visual censuses were done at several reserves and nearby fished sites at each location. We focused on fish associated with rocky reefs because (1) rocky reefs are the most common habitat protected within the entire system of marine reserves in Italy (Boero et al., 2005); (2) previous visual census studies showed that rocky reefs host the most of fish targeted by fishing and therefore these fish assemblages more clearly respond to protection from fishing than others (Francour, 1994). Fish assemblages in fished areas were sampled outside the MPAs or within the 'B or C zones' when no alternatives were available (e.g. at MPAs entirely encompassing small islands far away from the mainland). The use of buffer zones to contrast no-take reserves is supported by recent studies that suggest the ineffectiveness of partial closures for target fish species (Denny and Babcock, 2004). All of the reserves investigated had the same level of formal protection (fully no-take), in contrast to fished conditions. The MPAs and reserves investigated, the year of formal establishment, and the level of enforcement during the period when fish were sampled, are all reported in Table 1.

We assessed densities and size of fish in natural rocky reefs, except at Miramare where fish were sampled in artificial habitats (i.e. jetties formed by transplanted boulders), both within the reserve and fished sites. Fish were sampled from about 5–30 m depth, depending on the distribution of rocky habitats at each location both in the reserves and fished sites. Sampling was done by visual census along transects 25-m long and 5-m wide according to the 'strip transect' method (Harmelin-Vivien et al., 1985). Overall, 3096 visual censuses were done (sampling effort at each location is reported in Table 1). Fish density was estimated by counting single specimens to a maximum of ten individuals, whereas classes of abundance (11–30, 31–50, 51–100, 101–200, 201–500, >500 individuals) were used for larger schools. Fish size

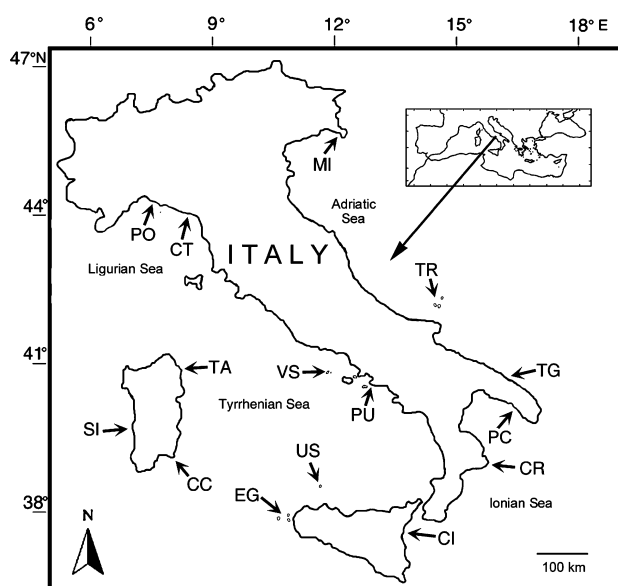


Fig. 1 – Location of the 15 MPAs studied along the coast of Italy (PO: Portofino; CT: Cinque Terre; TA: Tavolara-Capo Coda Cavallo; SI: Sinis-Isola Mal di Ventre; CC: Capo Carbonara; VS: Isole Ventotene-Santo Stefano; PU: Punta Campanella; CR: Capo Rizzuto; EG: Isole Egadi; US: Isola di Ustica; CI: Isole Ciclopi; PC: Porto Cesareo; TG: Torre Guaceto; TR: Isole Tremiti; MI: Miramare).

was assessed by classifying fishes within three size categories (i.e. small, medium, large) on the basis of the maximum total length attained by each species (Froese and Pauly, 2006). Fish biomass was evaluated using size distributions and length–weight relationships from the literature (Francour, 1990; Froese and Pauly, 2006). Early juvenile stages (settlers and recruits) were not taken into account.

Five research groups worked within the same research framework, using standardized methods in all the 15 locations investigated. As the level of personal experience may bias the results of fish visual censuses (Williams et al., 2006), meetings and an intensive training were done for all participants, to standardize the procedures and the observers' ability to collect accurate data, before sampling started.

2.2. Treatment of data

Methods derived from meta-analysis were used to examine and summarize the general response of fish to protection. As visual censuses were done at several fished and unfished sites and sampling was repeated through time, mean values were used to approximate average conditions in space and time (see Guidetti and Sala, 2007).

We examined the response to protection at species/family level (Mugilidae and Atherinidae include species difficult to identify visually at species level), at target vs non-target fish level, and at functional level (i.e. trophic groups). Fish taxa were pooled into functional groups based on their trophic position because fishing disproportionately targets species at higher trophic levels (Pauly et al., 1998), and recovery from fishing potentially includes increased abundances or biomass of high-level predators and shifts in trophic structure (Micheli et al., 2004). Each species/family was assigned to one of eight trophic groups using the available information about diet and size in the database "FishBase" (Froese and Pauly, 2006), and in Mediterranean studies (Sala, 2004; Guidetti and Sala, 2007): (1) large piscivores, (2) small piscivores, (3) invertebrate-feeders of group 1 (major predators of sea urchins), (4) invertebrate-feeders of group 2 (whose diets seldom include sea urchins), (5) small cryptobenthic carnivores, (6) detritivores, (7) planktivores and (8) herbivores (see Figs. 3 and 4 for species groupings). We split invertebrate feeders into two groups because of the major role a few fish species can have in regulating sea urchin populations and hence potentially affecting the entire benthic community (Sala et al., 1998; Guidetti, 2006). Piscivores included species feeding exclusively on fishes and species feeding on both fishes and invertebrates (Micheli et al., 2004).

We first considered in our analyses all reserves to look for possible general responses. Then, we grouped the reserves into three categories based on the level of enforcement. Categorizing enforcement at each reserve required obtaining information about (1) the frequency of illegal fishing within the reserves, and (2) the efficacy of the reserve personnel, the coast guard or other marine police forces in doing an active surveillance against illegal activities. This information was directly collected by the researchers involved in the project, and/or gathered by questioning the reserve personnel and local people. The relative enforcement categories were high (poaching very occasional if any, patrol very active and

continuous), medium (illegal fishing occurring but limited by infrequent surveillance) and low (common illegal fishing and virtually inexistent surveillance) (Table 1). Categorization was obtained by first assigning a score to surveillance and poaching for any single marine reserve in terms of percentage of days per year when there was an active surveillance (<25, 25–75, >75%, corresponding to score values of 0, 1 and 2, respectively) and events of poaching (<25, 25–75, >75%, corresponding to scores of 2, 1 and 0, respectively). Then, the product of surveillance and poaching scores was calculated and the enforcement category assigned with 0 = low, 1–2 = medium and 4 = high enforcement.

We quantified the effects of protection within reserves as the natural logarithm of the ratio between the values of the response variable (i.e. fish density and biomass) in reserves and fished conditions as response ratios, $\ln R$ (Hedges and Olkin, 1985; Micheli et al., 2004). Data were thus normalized and the response to protection examined independently of the absolute densities at each location. As estimations of average values can be affected by sampling effort, we calculated weighted means using the natural logarithm of the total area covered by the censuses from which the estimates were obtained (Mosquera et al., 2000). Positive response ratios indicate greater density and/or biomass of species or trophic groups in unfished than in fished areas, whereas negative values indicate greater values in fished areas compared to unfished areas. A ratio of zero, instead, means that densities are similar between reserves and fished conditions. Averages of the mean response ratios were considered significantly different from zero (i.e. there is a significant protection effect) when the 95% confidence limits around the mean do not overlap with zero (Micheli, 1999 and references therein). Based on the evidence that effective reserves and, more generally, areas characterized by null/low levels of exploitation can host particularly high fish biomass (Friedlander and DeMartini, 2002; McClanahan et al., 2007; Stevenson et al., 2007), we also estimated total fish biomass within the reserves and fished areas at the 15 locations investigated. We then calculated the relationship between total fish biomass within the reserves and the enforcement level.

As reported above, the transition from macroalgae to barrens can be enhanced by the removal of predators of sea urchins, i.e. *D. sargus* and *D. vulgaris* (Sala et al., 1998; Hereu et al., 2005; Guidetti, 2006). A threshold density of ~12 adult *Diplodus* fish 125 m^{-2} was found to maintain sea urchin population density under the threshold (~8–9 urchins m^{-2}) critical for triggering community shifts (Guidetti and Sala, 2007). Therefore, only those reserves where conditions (e.g. effective enforcement and compliance and/or habitat availability) are appropriate to host sufficiently dense populations of *Diplodus* may have the potential to recover from barrens back to macroalgal beds or maintain flourishing macroalgal beds. We thus evaluated this potential by assessing density of *Diplodus* in the reserves in relation to enforcement.

3. Results

Across all locations combined, total fish density was on average 1.15 times greater in reserves than in fished areas ($\ln R = 0.16 \pm 0.17$; 95% CI) (Fig. 2A). The lower the enforcement

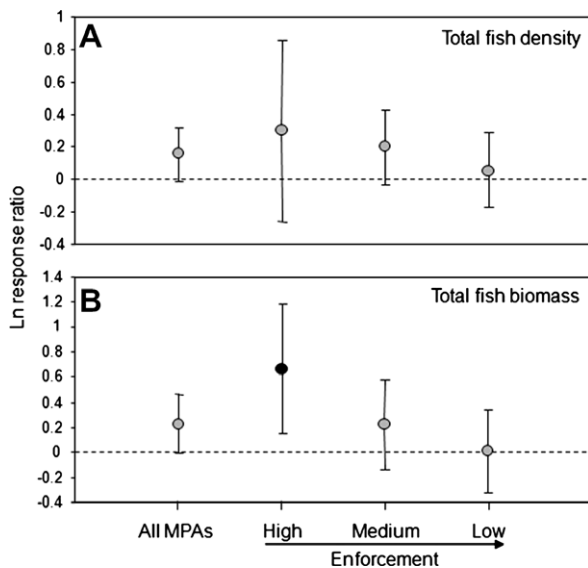


Fig. 2 – Fish response to protection, measured as the natural log ratio of (A) total fish density and (B) total fish biomass between reserves and fished areas, in all MPAs investigated and in relation to the enforcement level. Bars indicate 95% confidence intervals. Black circles: significant ratios; grey circles: non-significant ratios. See Section 2 for details on the analysis.

level, the less pronounced the differences: from 1.31 ($\ln R = 0.30 \pm 0.56$; 95% CI) to 1.06 ($\ln R = 0.06 \pm 0.23$; 95% CI) times greater in reserves than at fished sites at reserves where enforcement was high to low, respectively (Fig. 2A). In all cases, however, CI overlapped the zero values, which means that differences were not statistically significant. Especially in the case of reserves where enforcement is high, this outcome can be explained by the relatively low number of cases considered ($n = 3$). This caused large confidence intervals (see the values above) and a low probability of getting significant response ratios.

The effects of protection on fish density varied among taxa and trophic groups, and were strongly affected by the enforcement (Figs. 3 and 5). Considering all reserves, the only fish that responded positively to protection was the dusky grouper *Epinephelus marginatus*, which was on average 1.64 times more dense at reserves than at fished sites ($\ln R = 0.55 \pm 0.44$; 95% CI). At the reserves where enforcement was high, protection caused significant increases in density of nine fish taxa (all effect sizes for such taxa were >0 and CI did not overlap zero), in most cases represented by large piscivorous fishes and predators of sea urchins (i.e. *D. sargus* and *D. vulgaris*) (Fig. 3). At the reserves where enforcement was medium, only two fish taxa, i.e. *E. marginatus* and *Labrus merula* (their effect sizes were >0 and CI did not overlap zero), positively responded to protection (Fig. 3). No effects on fish density were detected at reserves where enforcement was low (all CI overlapped zero; Fig. 3). Large predatory fishes, predators of sea urchins, and herbivores as groups responded significantly to protection in well-enforced reserves (all effect sizes were >0 and CI did not overlap zero), while no responses were de-

tected at reserves characterized by medium or low enforcement, or when data from all the reserves investigated were pooled (all CI overlapped zero; Fig. 5).

Well-enforced reserves had ~ 2.4 times greater density of all target species combined than fished sites ($\ln R = 0.89 \pm 0.74$; 95% CI), whereas no significant effects were found for all non-target species combined ($\ln R = 0.28 \pm 0.64$; 95% CI). No effects of protection on fish density of both target and non-target species combined emerged, instead, when data were pooled for all reserves, or considering reserves characterized by medium or low enforcement (all CI overlapped zero).

Protection effects on total fish biomass were mostly significant and positive at well-enforced reserves ($\ln R = 0.66 \pm 0.51$; 95% CI; Fig. 2B). Only *Dicentrarchus labrax*, *E. marginatus* and *L. merula* were found to respond positively to protection when considering all reserves (their effect sizes were >0 and CI did not overlap zero). At the reserves where enforcement was high, protection caused significant increases in biomass of 18 fish taxa (with effect sizes >0 and CI not overlapping zero; Fig. 4), in most cases represented by target fish or impacted as by-catch. At the reserves with medium enforcement, again *E. marginatus* and *L. merula* were the only fish that significantly and positively responded to protection (again their effect sizes were >0 and CI did not overlap zero), whereas no effects were evident at reserves with low enforcement (CI overlapped zero; Fig. 4). As regards the functional groups, large predator fish, small piscivores, all invertebrate feeders, and herbivores responded significantly to protection in well-enforced reserves (all effect sizes were >0 and CI did not overlap zero), whereas no significant responses (CI overlapped zero) were detected at reserves with low enforcement (Fig. 5). At all reserves pooled and at those with medium enforcement only, small piscivores displayed a general and positive response to protection ($\ln R = 0.24 \pm 0.19$; 95% CI; Fig. 5).

Response ratios showed that well-enforced reserves had on average ~ 2.65 times greater fish biomass of all target species combined than fished sites ($\ln R = 1.06 \pm 0.71$; 95% CI), while no significant differences were found considering biomass values of all non-target fishes combined ($\ln R = 0.30 \pm 0.31$; 95% CI). No effects, moreover, emerged on biomass of all target and non-target species combined taking into account all reserves, as well as reserves with medium or low enforcement (all CI overlapped zero).

Average values of total fish biomass were highly variable among the study locations, which is likely to be due to local factors not considered here (e.g. productivity or habitat features). Biomass of fish ranged from ~ 34 to 187 and from ~ 16 to 161 g m^{-2} in the reserves and in fished conditions, respectively. Fish biomass was higher within the reserves than in fished conditions in 10 locations out of 15 (although in some cases the difference was small). Total fish biomass within the reserves was positively related with the level of enforcement ($r = 0.66$, $p = 0.007$, $n = 15$), although reserves and fished sites at Tavolara (characterized by medium enforcement) showed the highest values of fish biomass. Conversely, no relationship was found in fish biomass at fished sites having attributed to these latter the same enforcement level typical of each near reserves ($r = 0.27$, $p = 0.322$, $n = 15$), thus

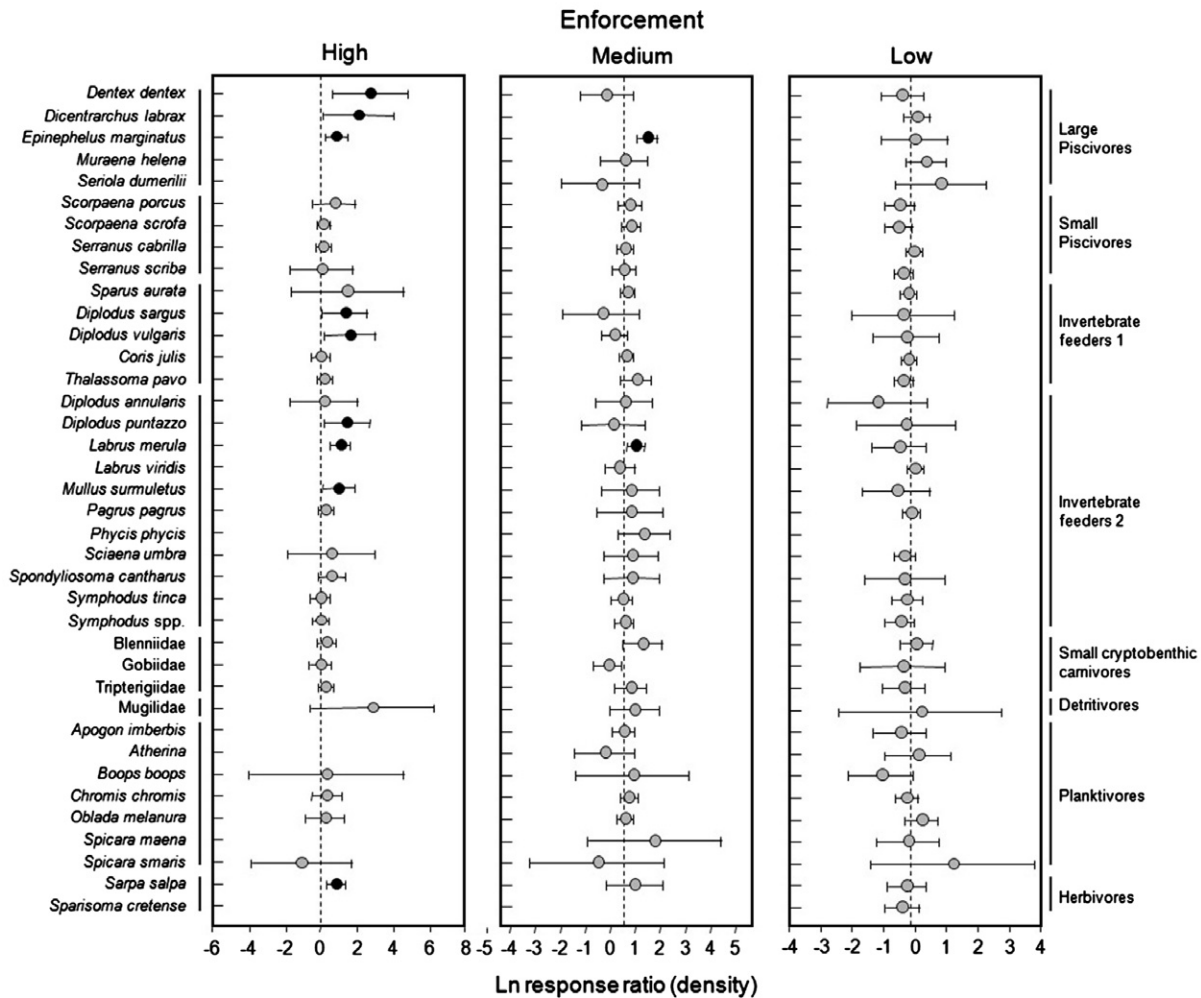


Fig. 3 – Fish species response to protection, measured as the natural log ratio of density between reserves and fished areas, in relation to the enforcement level. Bars indicate 95% confidence intervals. Black circles: significant ratios; grey circles: non-significant ratios. See Section 2 for details on the analysis.

providing suggestive evidence of the role that enforcement exerts in enhancing average fish biomass values relative to local conditions that may change from place to place.

As a general rule, finally, the density of *Diplodus* fish was above the threshold (~ 12 individuals per 125 m^{-2}) required to control sea urchin populations only at the reserves where enforcement is high. The threshold was not achieved when all reserves were considered (Fig. 6). Looking at each reserve, the density of *Diplodus* fish was above the threshold at all three well-enforced reserves (i.e. Miramare, Torre Guaceto and Portofino) and at Tavolara. The maximum value was observed at Torre Guaceto and the minimum at Ustica.

4. Discussion

This study evaluated the consequences of different levels of enforcement on the ecological effectiveness of 15 Italian marine reserves on fish, in terms of direct effects on target fish and their potential of indirectly influencing entire rocky reef communities (Guidetti and Sala, 2007 and references therein).

In addition, we demonstrated the importance of taking into account the enforcement at the reserves studied when procedures are used to summarize or generalize the effectiveness of multiple reserves (e.g. in meta-analysis studies and reviews).

Many target fish species clearly responded to protection in well-enforced marine reserves, similar to what has been observed in other Mediterranean reserves (e.g. Harmelin et al., 1995; Garcia-Charton et al., 2004; Claudet et al., 2006; Guidetti and Sala, 2007). However, in the present study the response of fish was to some extent variable among reserves, which could also be attributed to their different reserve age (Gerber et al., 2002; Dufour et al., 2007; Guidetti and Sala, 2007).

Protection effects were evident for large predators like the dusky grouper *E. marginatus*, a species that is included in the IUCN red list as endangered and at risk of dramatic reduction (see <http://www.iucnredlist.org>). Fish predators of sea urchins also clearly responded to protection. The critical threshold of ~ 12 adult individuals per 125 m^{-2} (Guidetti and Sala, 2007) was exceeded only at the three well-enforced reserves (Mira-

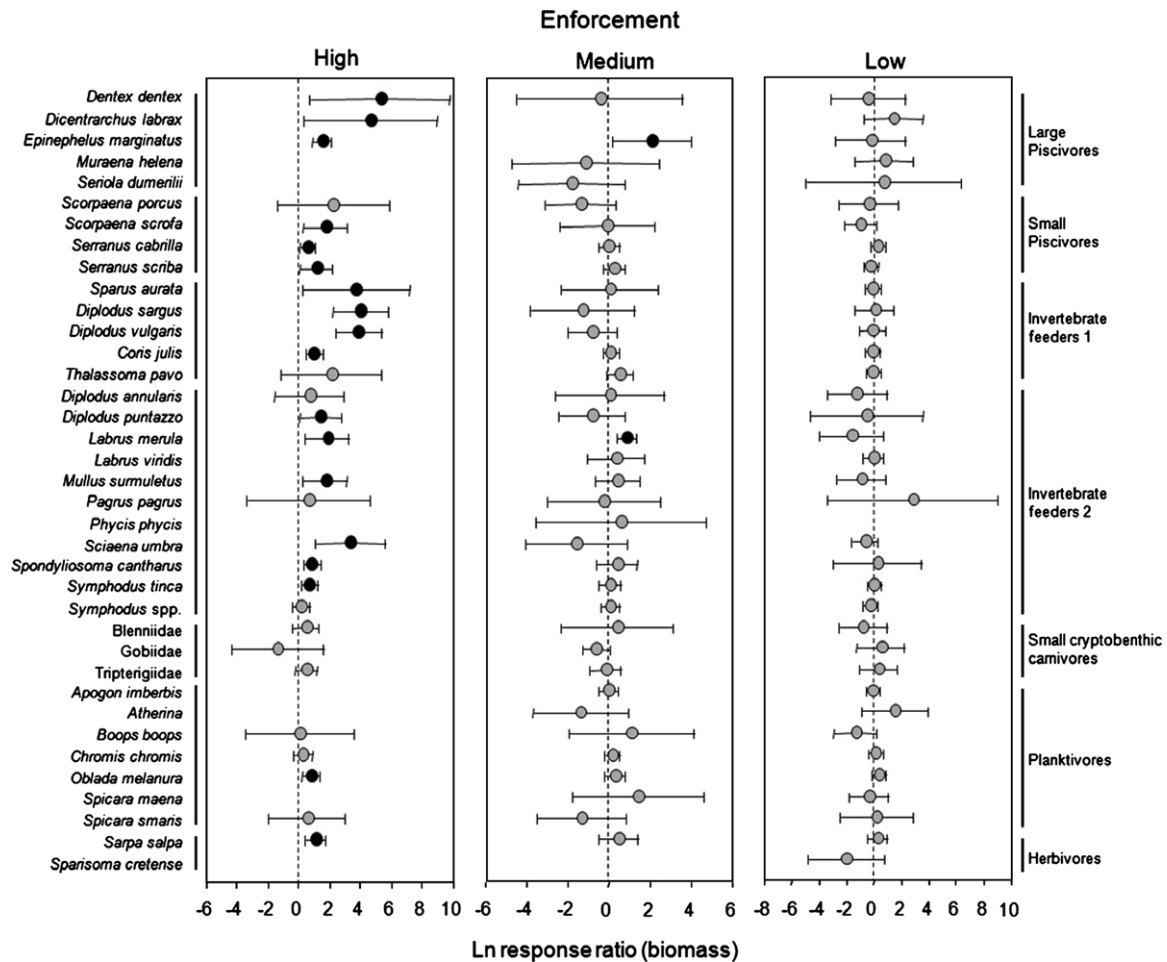


Fig. 4 – Fish species response to protection, measured as the natural log ratio of biomass between reserves and fished areas, in relation to the enforcement level. Bars indicate 95% confidence intervals. Black circles: significant ratios; grey circles: non-significant ratios. See Section 2 for details on the analysis.

mare, Torre Guaceto, Portofino) and at Tavolara (characterized by medium enforcement). This ecological threshold, therefore, seems to be attainable only where enforcement is high or where local fishing pressure is not very strong. At Ustica Island marine reserve where enforcement has been effective for many years, *Diplodus* fish continued to have low densities as had historically been the case. This was attributed to the paucity of habitats suitable for juvenile stages around the island (Vacchi et al., 1998). The Ustica case suggests that the recovery of functionally relevant species potentially affecting the whole communities does not always occur, sometimes regardless of good enforcement, and proper information about fishing pressure and local ecological conditions are needed to elucidate why these unpredicted responses can occur. Herbivorous fishes (mostly *Sarpa salpa*) also displayed slightly greater density and biomass at the reserves than in fished conditions, which is not consistent with results obtained by Guidetti and Sala (2007). Such a discrepancy could be due to the differences in local densities of piscivorous fish predators and, to some extent, to different local fishing traditions (e.g. the use of specific gears that may impact *S. salpa* as target or by-catch).

Rocky reef communities strongly impacted by fishing may thus show extirpation or functional extinction of fish species that have important ecological roles (e.g. predators or herbivores), with consequences on ecosystem functioning and services (Holmund and Hammer, 1999; Worm et al., 2006). MPAs or reserves that fail to increase fish densities and sizes may also face economic losses, e.g. in terms of decreased attractiveness of seascapes deprived of large charismatic fish for recreational divers or lower incomes from fishing undertaken in proximity to the reserves' boundaries. A crucial point, therefore, is that the enforcement and good compliance are fundamental pre-requisites for fish populations to replenish (Guidetti and Sala, 2007), spillover of adult fish to occur (Roberts et al., 2001), other community-wide effects to be felt (e.g. trophic cascades or barren-algal transitions; Sala et al., 1998) and economic initiatives to be activated (Holmund and Hammer, 1999).

With the exception of Tavolara (characterized by medium enforcement and relatively low fishing impact), a positive relationship was found between the level of enforcement and the total fish biomass. Particularly high values were found at the Miramare reserve, very small in size and where

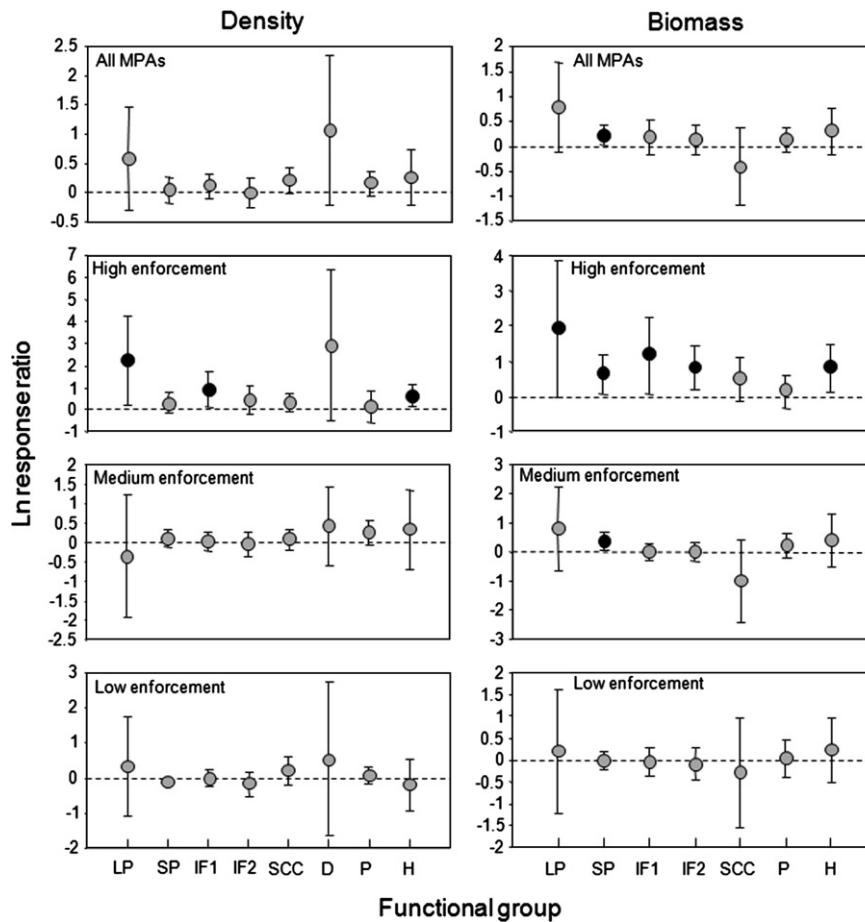


Fig. 5 – Functional group response to protection, measured as the natural log ratio of density and biomass between reserves and fished areas, in all MPAs investigated and in relation to the enforcement level. Bars indicate 95% confidence intervals. Black circles: significant ratios; grey circles: non-significant ratios. See Section 2 for details on the analysis.

artificial reefs fall on muddy sand at less than 10 m depth. It would be interesting to ascertain whether such large biomass is actually supported by local productivity/turnover or whether fish tend to concentrate within this small but undisturbed reserve, while needing external subsidies to persist (Stevenson et al., 2007).

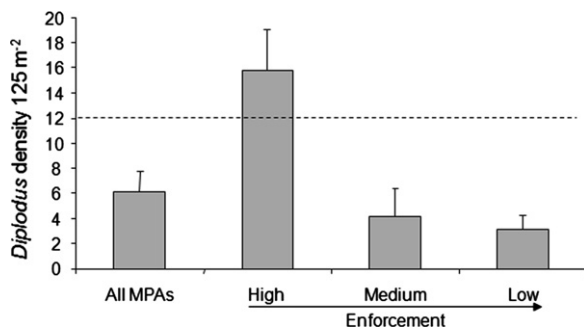


Fig. 6 – *Diplodus* density (*D. sargus* plus *D. vulgaris*) in all MPAs investigated and in relation to the enforcement level. The dotted line indicates the ecological threshold of *Diplodus* density potentially responsible to keep sea urchin density low. See Section 2 for details.

Our analyses included a relatively large number of reserves. This allowed us to offer a more balanced picture of the effectiveness of a national system of MPAs (and related reserves) and to show that scant enforcement made a proportion of reserves fail in meeting their objectives. However, it is admittedly not easy to formally assess the level of enforcement (Jameson et al., 2000; Mora et al., 2006) because achieving compliance within reserves may involve different approaches, from drastic or top-down rule imposition (and therefore repression of illegal activities) to gradual education and awareness creation through a soft glove approach (Salm et al., 2000). The development of protocols and proper metrics to monitor and assess enforcement at many reserves is thus not an easy task, but certainly deserves major attention in the future.

This study also stressed the need to carefully consider the enforcement when analyzing data from multiple reserves by pooling data to provide generalizations. Clear effects of protection, in fact, would not have emerged if reserves were not analyzed in relation to the enforcement. When data from all reserves were pooled, no general differences were found in the patterns of abundance, biomass of fish species, or trophic structure of assemblages between fished areas and reserves. 'Blind' assessments of the effectiveness of multiple reserves

thus actually carry the risk that positive effects from enforced reserves can be masked by neutral/negative results from reserves characterized by scant enforcement. Consequently, it makes no sense to invest in sound sampling designs to test for protection effects without assessing enforcement levels. Such 'blind assessments' could, in fact, lead to the conclusion that marine reserves are ecologically ineffective. This could in turn induce stakeholders and policy makers to dismiss MPAs and reserves as worthwhile conservation/management tools, since the social conflicts generated by their creation would not be balanced by any apparent benefit.

Effects of protection on fish species and trophic groups were not detected at unenforced reserves (i.e. paper parks). In some terrestrial tropical regions, however, even paper parks were found to be successful in mitigating some human impacts (Rodríguez and Rodríguez-Clark, 2001). Our data, in contrast, showed that no evidence of positive effects of such paper reserves (see also Bearzi, 2007). Our impression is that at sea, the concept of restricting human activities by marine area (Russ and Zeller, 2003) has not gained cultural acceptance (Clark, 1981). Local communities opposing marine reserves, in addition, often exert a strong pressure on local policy-makers. This generally results in scant effort by local maritime police-forces and/or reserve personnel patrolling the reserves, and, therefore in continued or even increased illegal fishing within the paper parks. In Italy, MPAs are usually underfunded and understaffed, which strongly affects their governance and, in turn, their ecological effectiveness. However, those Italian marine reserves that were able to find funds for improving staff and surveillance showed the most significant ecological responses. This seems a crucial point, since the investment in enforcement may provide the greatest return on maintaining the ecological benefits of the reserve to the fishery (Byers and Noonburg, 2007). Inadequate public involvement and communication/education in the process of MPA development (e.g. selection, planning and management) are also important issues that in Italy have been often neglected in the past. Public and stakeholder involvement has been limited in most of the 15 reserves studied. This lack of community participation undoubtedly leads to numerous conflicts and disapproval by locals about the establishment of marine reserves, does not increase the perceived legitimacy of decisions, and lowers compliance with restrictions (Friedlander et al., 2003).

All the above issues suggest the need for a new strategy for MPAs at national level, where major efforts and funds are invested in informing the public and promoting participation in the decision-making process. Instead of decreeing even more paper parks, Italy should concentrate on enforcing regulations within the existing MPAs and equipping them with surveillance personnel devoted to this task.

In conclusion, this study provides evidence that reserves established along the coasts of Italy can be ecologically effective, provided that compliance is good and enforcement, where needed, is effective. Well-enforced reserves not only can meet ecological and socio-economical objectives, they can also help promote creation of new reserves (Agardy et al., 2003; Friedlander et al., 2003). A better understanding of how compliance and enforcement can affect ecological outcomes can help resource managers and policy-makers de-

sign better MPAs in response to the specific management problems needed to be solved or purposes to be served and make well-informed decisions regarding the MPAs and reserves already established. As well, articulating clear objectives for MPAs can help convert the vehement public opposition usually encountered at the time of MPA establishment (especially by skeptical users like fishermen) into broader acceptance and better chances of success for MPAs.

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