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Effectiveness of five small Philippines' coral reef reserves for fish populations depends on site-specific factors, particularly enforcement history

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ABSTRACT

Community-managed, no-take marine reserves are increasingly promoted as a simple, precautionary measure to conserve biodiversity and sustain coral reef fisheries. However, we need to demonstrate the effects of such reserves to those affected by the loss of potential fishing grounds and the wider scientific community. We surveyed changes in fish communities in five small marine reserves in the central Philippines and three distant Control sites over seven years. We conducted underwater visual censuses of 53 fish families within the reserve (Inside), with a kilometre of the boundary (Outside) and at Control sites. We found significant differences between fish communities Inside and Outside the reserve only at the two sites with strictest compliance with fishing prohibition, while there were significant differences to distant Control sites in all cases. The strongest responses to reserve protection were found in predatory fishes (groupers and breams) and in butterflyfish. Other abundant fish families showed weak effects of protection. For all taxa analysed, we found significant effects of reserve Site and Site × Treatment interactions. The detection of fish responses to reserves is complicated by potential spillover effects, site-specific factors, particularly compliance, and the difficulty of identifying appropriate control areas.

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1. Introduction

Coral reefs harbour high levels of biodiversity and provide vital economic services for millions of people in coastal communities in developing countries, particularly in southeast Asia (Connell, 1978; Cesar et al., 2003). Despite comprising less than 0.5% of the ocean floor, coral reefs harbour almost a third of all marine fish species (Paulay, 1997). The economic value of goods and services provided by coral reefs, including fisheries, tourism and coastal protection, has been estimated at almost US$30 billion per annum (Cesar et al., 2003). However, coral reefs face diverse anthropogenic threats including overexploitation of resources, habitat destruction, pollution and climate change (Wilkinson, 2004). Coral reef fishes are

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now widely overexploited and their habitat destroyed by destructive fishing methods such as blast fishing (see Jackson et al., 2001; Pauly et al., 2002; Wilkinson, 2004). These fisheries comprise multiple species, gears and landing sites taken by subsistence, artisanal and, increasingly, commercial fishers (Munro and Williams, 1985; McManus, 1997). They are therefore difficult to manage using conventional fisheries controls.

No-take marine reserves, areas where all forms of resource extraction are permanently banned, are now widely advocated as a precautionary, pragmatic management tool for protecting marine biodiversity, habitat, fisheries and ecosystem services (e.g. NCEAS, 2001; Palumbi, 2002; United Nations, 2002). Advocacy for marine reserves is based on both theoretical studies modelling reserve effects (see review by Gerber et al., 2003) and empirical studies (see reviews of Russ, 2002; Gell and Roberts, 2003).

Recent meta-analyses demonstrate that marine reserves have positive effects on many organisms, particularly those targeted by fishing (reviews by Roberts and Polunin, 1991; Mosquera et al., 2000; Côté et al., 2001; Halpern, 2003). Furthermore, some studies have demonstrated that marine reserves can have benefits beyond their boundaries through spillover of adult fish (Russ et al., 2003; Kaunda-Arara and Rose, 2004). However, the efficacy of marine reserves in the face of large-scale, diffuse threats such as eutrophication and climate change remains unproven and is likely to be more limited (Agardy et al., 2003).

Demonstration of the effects of marine reserves may be significantly biased by the use of single spatial comparisons among reserves of different ages (Edgar et al., 2004). If the selection of reserve sites is non-random because of socio-political factors, it may be impossible to separate natural spatial variation from reserve effects purely through spatial comparisons (Edgar et al., 2004; Russ et al., 2005). Similarly, random spatial effects cannot be assessed in studies of single reserve sites through time (Gell and Roberts, 2003). Thus meta-analyses based on such studies may also be systematically biased (Mosquera et al., 2000; Côté et al., 2001; Halpern, 2003). However, Russ et al. (2005) have recently used data from the Philippines to demonstrate that the magnitude of recovery of large predatory fishes in reserves was similar whether measured through spatial or temporal comparisons but such studies are rare (but also see Hawkins et al., 2006).

Quantifying both spatial and temporal variation in the same study (i.e. looking at changes at multiple sites over time) provides a powerful test of the hypothesised effects of marine reserves although such studies are expensive and subject to confounding biological and physical factors. However, to convince often-sceptical resource users, fisheries managers and policymakers we need to be able to demonstrate that reserves work as we intend them to. In particular, where marine reserves are managed by local communities, it is imperative to demonstrate the effects of reserves in order to maintain engagement by the community.

Our objective in this study was to rigorously measure the impacts of establishing five reserves in coastal coral reef habitats in northern Bohol, central Philippines, on the abundance of a suite of fishes. These reserves spanned three municipalities (the spatial unit responsible for declaring and enacting reserve legislation) over a distance of approximately 40 km. The reserves were designated as completely no-take (locally referred to as *sanktwaryo*), which also, significantly for this region, protects them from habitat destruction caused by methods such as blast fishing. We monitored these reserves, together with three control sites over seven years. We sampled within and just outside each reserve to investigate spillover effects while comparisons with distant control or reference sites were used to partition natural temporal fluctuations. Our hypotheses were: (1) fish abundance, particularly for targeted species, would increase within reserves, (2) fish abundance would also increase just outside reserves but to a lesser degree and with a time lag, (3) fish abundance would be lowest and show no real temporal trends at distant control sites.

2. Methods

2.1. Study sites

Five reserve sites in three municipalities were monitored in north-west Bohol, central Philippines (Fig. 1, Table 1). The monitoring programme was implemented through a local non-governmental organisation, Project Seahorse Foundation for Marine Conservation, in cooperation with local communities.

All reserves were established by small island communities lying off the coast of Bohol, except for Asinan which is a coastal community on the mainland of Bohol. Reserves were rectangular in shape, ranged from 10.5 to 66 ha in size and had their nearest edge 0–1000 m from shore (Table 1). In addition, three distant control sites were also established: Putik, Pandao and Canlangi (Fig. 1). These control sites are potentially confounded by factors such as initial state, exploitation history and local biogeography, but were selected as the most comparable sites for the different reserves.

The enforcement and compliance history and current practices of each reserve were determined through informal interviews with community representatives. Enforcement commencement date was determined by a combination of when a guardhouse was erected, and when the local community began to enforce “no-take” of any sort. In many areas communities confused reserve protection with preventing use of illegal gear because the latter is such a problem in this region (Table 1). This effectively differentiated between when the habitat was protected through prevention of illegal and/or destructive fishing methods such as dynamite fishing, and when fish populations were protected from any form of exploitation. In 2004, all the reserves were rated using the Coastal Conservation Environment Foundation (CCEF) national reserve rating system to determine their effectiveness (White et al., 2006).

2.2. Monitoring design

In total we had eight Sites for the study: five reserves and three distant controls. We defined three categories of Treatment: (1) Inside, within the marine reserve, (2) Outside, within 1000 m of the reserve boundary, (3) Control, distant reference sites open to fishing. Our design was not orthogonal because Site and Treatment were identical for distant Control sites.

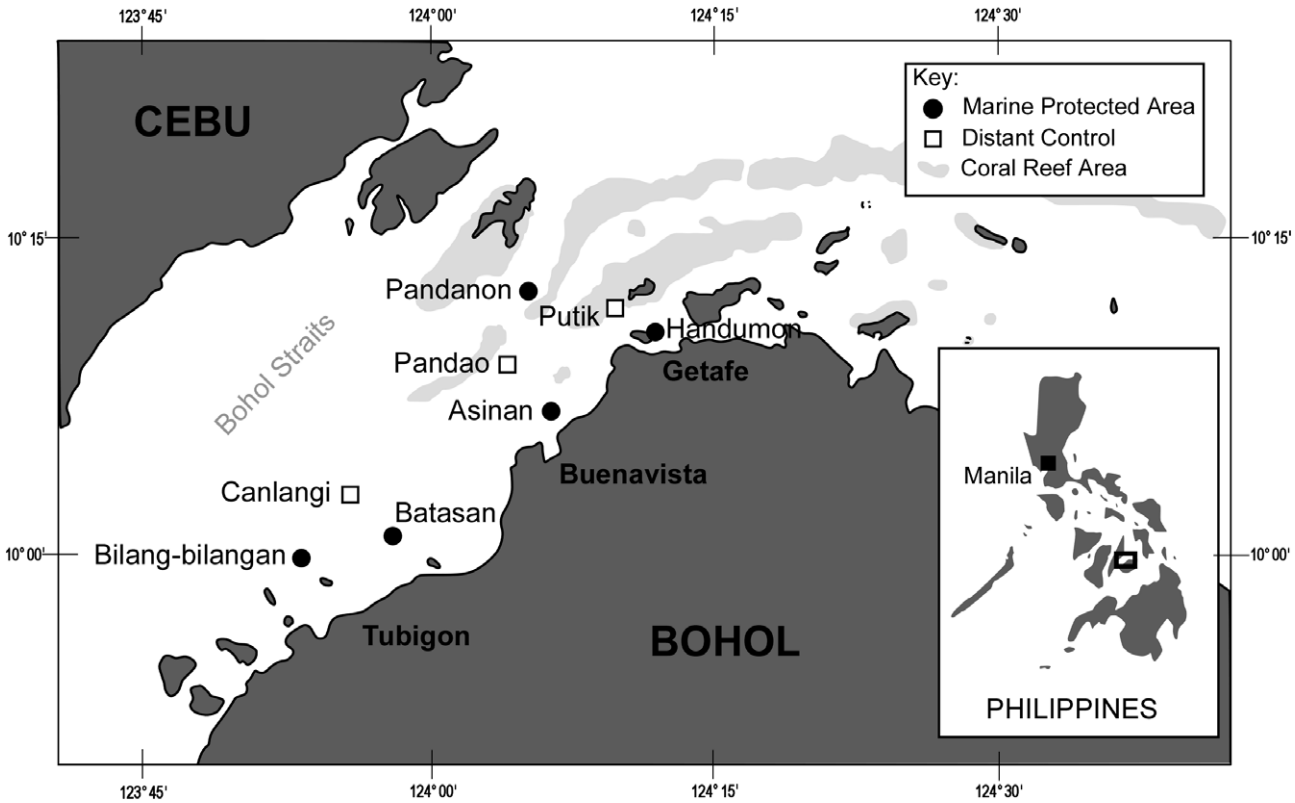

Fig. 1 – Map of Bohol, central Philippines, indicating location of reserve sites, Control sites and Municipalities.

Table 1 – Characteristics, compliance history, and management of five marine reserves in northern Bohol (see Fig. 1)

Site name	Handumon	Pandanon	Asinan	Bilang-bilangan	Batasan
Municipality	Getafe	Getafe	Buenavista	Tubigon	Tubigon
Lat/long coordinates	10°10.3 N 124°10.6 E	10°11.1 N 124°4.6 E	10°6.4 N 124°6.4 E	09°59.4 N 123°52.7 E	10°0.7 N 123°59.0 E
Size (ha)	50	20	66	10.5	21
Distance from nearest shore (m)	0	0	~1000	0	~300
Year proposed[a]	1995	1999	1999	1999	1997
Reserve enforcement commenced	1995	2004	2000	1999	1999
Level of protection	Strict[b]	Poor[c]	Good[d]	Poor	Strict[e]
Reserve ranking[f]	V	II	IV	II	IV
Monitoring commenced	1998	1998	1998	1998	1999
Marker buoys	Lost a number of times	No	Yes	No	Yes
Marker posts	Yes	No	No	Yes	Yes
Guardhouse	Yes	Yes (2004)	No	No	Yes
Guards	Yes	Yes (2004)	No	No	Yes
Daytime patrols	Yes	No	Yes	No	Yes
Night time patrols	Yes	No	Yes	No	Yes
Issues, concerns and lapses in management	Fishers from other island sometimes encroach	Blast fishing; conflict/lack of cohesion in management	Encroachment	Fishers from other island sometimes encroach	

a Year proposed refers to the start of any form of habitat protection whether local or formalised through government ordinance.
b Illegal fishing just outside reserve widespread; occasional fishing encroachment into the reserve.
c Although full enforcement only commenced early in 2004, use of illegal fishing methods had been prevented since 1999.
d Enforcement patrols prior to 2000 were only against illegal fishing methods; legal fishing methods were allowed in the sanctuary until 2000.
e Strictly enforced since 1999, though prevention of illegal fishing started in 1998.
f Coastal Conservation Environment Foundation ranking: I = initiated, II = established, III = enforced, IV = sustained, V = institutionalized (White et al., 2006).

We first conducted manta tows (English et al., 1997) at each Site and Treatment (Inside/Outside reserve) to determine the extent of the coral reef area. Tows were conducted along the reef crest (1–5 m) and the reef slope (7–11 m) and the GPS coordinates of broad habitat types were recorded. Fixed 50 × 5 m belt transects were then established using iron stakes and rope for each Site and Treatment. Transects were located haphazardly in areas established as coral dominated by the manta tows and large enough to accommodate a 50 m length transect. In each site two deep (4–10 m) and two shallow transects (<4 m) were set up, though the full design was not attained until 2001. Underwater visual census methods (English et al., 1997; Samoilys, 1997) were used to record (1) fish abundance and size, (2) benthic habitat, though only data from the former are presented here.

Surveys began in 1998 and was conducted bi-annually to coincide with the dry season (February–May) and the wet season (August–November). We here report data up to the end of 2004, giving us 14 potential sampling periods in total although the actual number varied across the reserves.

2.3. *Fish surveys*

Surveys were conducted by one trained biologist and one trained volunteer using standard UVC procedures over 15 min for each 50 × 5 m transect (Samoilys, 1997; Samoilys and Carlos, 2000). Highly mobile fish were counted first before those that are territorial and less mobile, to maximise accuracy (Samoilys and Carlos, 2000). Fishes that crossed the other side of the transect were noted and later discussed with the buddy to avoid duplication in counting. We censused fish at the family level because of difficulties in accurately identifying all species, particularly with changing observers among seasons and years. Such aggregation at the family level is common in marine reserve studies, not least because the majority of species within a family share similar life history (e.g. Samoilys and Carlos, 2000; Alcala et al., 2005). We surveyed a total of 53 families ranging from highly valued food fish, to fish of little food value (Table 2).

2.4. *Data manipulations*

We assigned fish families to one of six functional groups based on trophic level, fishery value and abundance/distribution, i.e. representation at most sites at most time periods (Hiatt and Strasburg, 1960; Hobson, 1968; Goldman and Talbot, 1976; Heemstra and Randall, 1993; Samoilys and Carlos, 2000; Alcala et al., 2005) (Table 2). Virtually everything is eaten in this region of the Philippines with the exception of a few fish families regarded as "worthless". We also specifically looked at families suggested as indicators of reef health – butterflyfish (family Chaetodontidae) and angelfish (family Pomacanthidae) (e.g. Hourigan et al., 1988; Crosby and Reese, 1996). These families also form a significant component of the aquarium fishery (Wood, 2001).

Data from 1998 Wet, 1998 Dry and 2000 Dry (periods 1, 2 5) are presented graphically but samples were insufficient for quantitative analyses. Preliminary analyses found that depth and season did not contribute significantly to variation among sites (non-parametric MANOVAs and repeated-measures ANOVAs), so we pooled all data within one calendar year to increase our replication and ability to detect effects of protection. As with most similar data sets our whole community data were not multivariate normal and therefore did not meet the assumptions for parametric analyses (Anderson, 2001). However, individual fish family count data (n) were generally from over-dispersed negative binomial distributions and therefore could be $\log_{10}(n + 1)$ transformed to satisfy the assumptions of parametric analyses, although back-calculated data are presented graphically.

2.5. *Multivariate analyses*

Similarities among fish communities within Sites (matrix of abundance by family) were calculated using the Bray-Curtis similarity index on the fourth-root transformed abundance data by family. Non-metric multidimensional scaling (nMDS) was then applied to the similarity matrix to produce a two-dimensional representation of community similarities using the program PRIMER v.5 (Clarke and Gorley, 2001). While nMDS is primarily a pattern seeking tool and any groupings cannot be easily interpreted in terms of explanatory variables, we preferred its use to other multivariate techniques such as canonical correspondence analysis because the former does not rely on any particular assumptions about the underlying structure of the data. We undertook these analyses on data sets from Inside and Outside each reserve together with its appropriate Control site. We ran the analyses with the full data set and with only those families contributing more than 5% or 10% of total abundance – the patterns produced were qualitatively similar and so the full data set was retained for presentation. We then used non-parametric multivariate analysis of variance (npMANOVA) in order to estimate the statistical probabilities associated with differences between Years and Sites (Anderson, 2001; McArdle and Anderson, 2001). Ten thousand permutations were undertaken using the program PERMANOVA with factors Location and Year followed by pairwise comparisons between locations (Anderson, 2005).

2.6. *Univariate analyses*

To investigate general patterns across all reserves we analysed our abundance data using general linear models (GLMs) to look for differences between Inside and Outside treatments over time. Repeated-measures analyses of variance (rmANOVAs) (Zar, 1999) were run with Year as the fixed within-subjects factor, Site (reserve) and Treatment (Inside vs. Outside reserve) as fixed among-subjects factors and Transect as the repeatedly measured subject. Our Control sites could not be included in these analyses because our design was not orthogonal. However, we included Controls when we performed rmANOVAs for each reserve separately to investigate whether the patterns were different between reserves and Controls. We conducted these rmANOVAs with Year as the fixed within-subjects factor and Treatment (Inside, Outside or Control) as a fixed among-subjects factor. We used *post-hoc* Tukey tests to determine overall differences among Treatments. We also used a Bonferroni-adjusted value of $\alpha = 0.01$ to control for multiple comparisons in these latter analyses.

Table 2 – Fish families censused in this study

Functional group	Family	
Top-trophic level Piscivores; large fish; high value food fish	Haemulidae (Sweetlips) Lethrinidae (Emperors) Lutjanidae (Snappers)	Nemipteridae (Breams) Serranidae (Groupers)
Mid-trophic level Omnivores/herbivores; medium-sized fish; medium value food fish	Acanthuridae (Surgeonfish) Balistidae (Triggerfish) Holocentridae (Soldierfish) Labridae (Wrasses)	Mullidae (Goatfish) Scaridae (Parrotfish) Siganidae (Rabbitfish)
Low-trophic level Planktivores/omnivores; small fish; no or little commercial value, except some species occur in the aquarium fishery	Apogonidae (Cardinalfish) Blenniidae (Blennies) Caesionidae (Fusiliers) Cirrithidae (Hawkfish) Gerreidae (Silver biddies)	Gobiidae (Gobies) Pinguipididae (Sandperch) Pomacentridae (Damselfish) Scorpaenidae (Scorpionfish) Synodontidae (Lizardfish)
Pelagics Piscivores; large fish; generally high value food fish	Carangidae (Trevally) Ephippididae (Batfish) Sphyraenidae (Barracudas)	
Coral health indicator species Majority are valuable in aquarium fishery	Chaetodontidae (Butterflyfish) Pomacanthidae (Angelfish)	
Others Variety of families from diverse trophic groups but rare or patchy in distribution	Aulostomidae (Trumpetfish) Callionymidae (Dragonets) Centriscidae (Shrimpfish) Chandidae (Milkfish) Diodontidae (Porcupinefish) Monacanthidae (Leatherjackets) Echeneidae (Remoras) Engraulidae (Anchovies) Fistularidae (Flutemouths) Gobiesocidae (Clingfish) Hemiramphidae (Halfbeaks) Leiognathidae (Ponyfish) Malacanthidae (Sand Tilefish)	Microdesmidae (Wormfish) Mugilidae (Mullets) Muraenidae (Moray Eels) Ophichythidae (Snake Eels) Ostraciidae (Boxfish) Platycephalidae (Flatheads) Plotosidae (Eeltail Catfish) Priacanthidae (Bigeyes) Sillaginidae (Smelt-Whitings) Syngnathidae (Pipefish) Tetraodontidae (Pufferfish) Terapontidae (Grunters) Zanclidae (Moorish Idol)

Families were categorised into functional groups based on four criteria: trophic level, fishery type/value, abundance/distribution and representation in previous studies.

We chose our response variables on the basis of previous work (see above), abundance and importance to human community (e.g. Samoilys and Carlos, 2000; Russ et al., 2005): these variables were: the abundance of top-trophic level fish and the two most abundant top-trophic families, Serranidae and Nemipteridae; the two dominant mid-trophic families, Labridae and Scaridae; the dominant low-trophic family Pomacentridae and the dominant coral health indicator family Chaetodontidae. We performed all our univariate analyses using the program SPSS v.11 (SPSS, 2003).

Finally, to examine broader patterns of responses to protection over the time period of the study, and for comparison with meta-analyses of reserve effects on fish communities (Mosquera et al., 2000; Côté et al., 2001), we calculated the natural logarithm of the Response Ratio (ln RR) for 24 fish families over the last three years of sample data i.e. 2002–2004. The Response Ratio is defined as mean abundance Inside reserve/mean abundance Outside reserve. As the variation in our sampling effort among sites was relatively small we did not weight the effect sizes and used arithmetic means. Effect sizes were considered to be significantly different from zero when the confidence interval did not include zero (Côté et al., 2001).

3. Results

3.1. *Compliance*

Compliance level varied considerably among the five reserves, with no-fishing only well enforced and respected in three: Handumon, Batasan and Asinan whereas the other two reserves had little or no protection (Table 1). Years of compliance also differed among reserves: in 2004 Handumon had been protected for nine years, Batasan for five years and Asinan for four years. In these two younger reserves, protection from illegal and destructive fishing methods was enforced one to two years prior to reserve establishment (Table 1). In contrast, protection at Pandanon reserve only began in 2004, though again, illegal fishing methods had been excluded for five years at this proposed reserve site.

3.2. *Fish community patterns*

When similarities among fish communities were analysed using nMDS we found clear separation among Treatments (Inside, Outside or Control) at all reserve sites (Fig. 2). Separation between Inside and Outside locations was greatest at Batasan

and Handumon, the two sites with strongest levels of protection and greatest age since protection was enacted (Fig. 2). Asinan, a site with good protection but a shorter history of protection, and Bilang-bilangan, a site with poor protection showed some separation between Inside and Outside fish communities while transects were completely intermixed at Pandanon (Fig. 2) where enforcement only began in 2004 (Table 1). Control fish communities were generally more similar to Outside communities for all reserve sites except Pandanon (Control site Pandao) which showed strong separation (Fig. 2).

These patterns were confirmed by the results of npMANOVA (Table 3). The factors Treatment, Year and the Treatment × Year interaction were all highly significant except for the Treatment × Year interaction at Handumon. Pairwise comparisons showed that Inside and Outside communities were significantly different from each other at Handumon

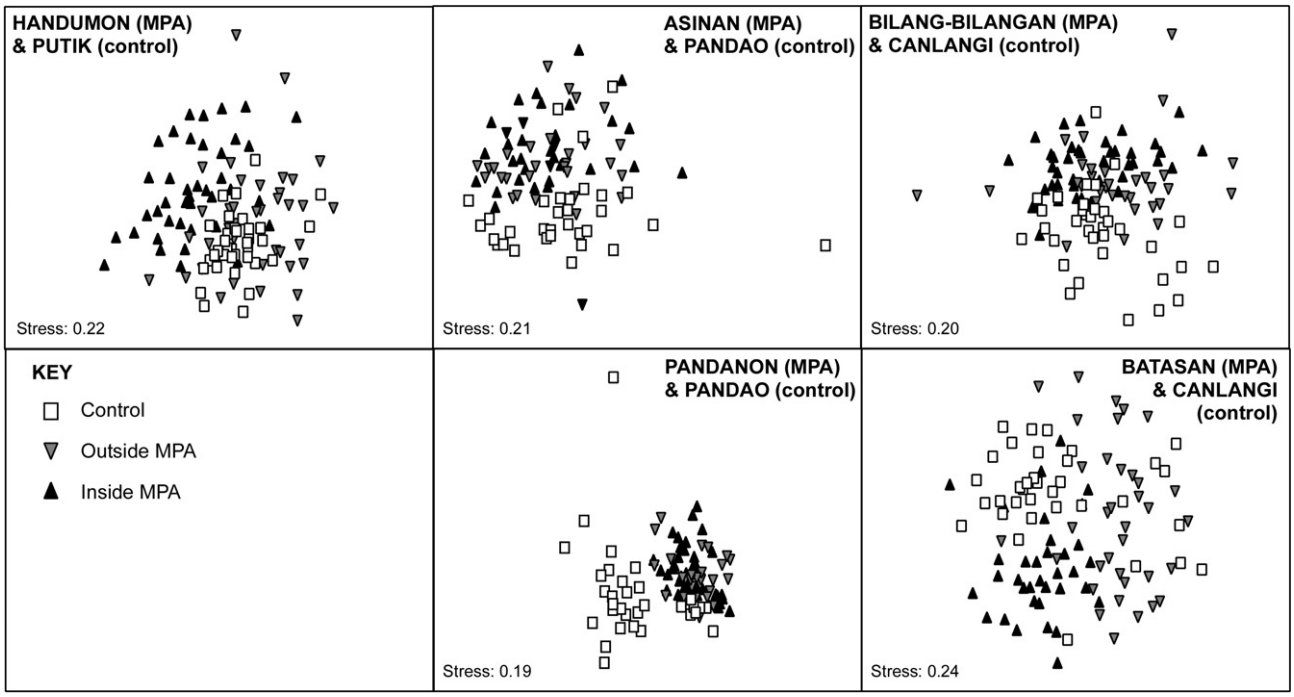

Fig. 2 – Non-metric multidimensional scaling plot of fish communities inside and outside five reserves in the central Philippines, each with associated Control site.

Table 3 – Results of npMANOVA and associated pairwise comparisons for fish communities from five reserves and associated Control sites calculated using 10,000 permutations

Reserve site and control	Source of variation	*F* statistic and degrees of freedom	Significance	Pairwise comparisons
Handumon and Putik	Treatment	$F_{2,45} = 7.91$	**<0.001**	Inside vs. Outside; $p <$ **0.001**
	Year	$F_{4,45} = 3.65$	**<0.001**	Inside vs. Control; $p <$ **0.001**
	Treatment × Year	$F_{8,45} = 0.92$	0.658	Outside vs. Control; $p <$ **0.001**
Asinan and Pandao	Treatment	$F_{2,36} = 5.64$	**<0.001**	Inside vs. Outside; $p = 0.068$
	Year	$F_{3,36} = 3.45$	**<0.001**	Inside vs. Control; $p =$ **0.001**
	Treatment × Year	$F_{6,36} = 1.86$	**0.001**	Outside vs. Control; $p =$ **0.001**
Pandanon and Pandao	Treatment	$F_{2,36} = 6.41$	**<0.001**	Inside vs. Outside; $p = 0.192$
	Year	$F_{3,36} = 3.45$	**<0.001**	Inside vs. Control; $p <$ **0.001**
	Treatment × Year	$F_{6,36} = 1.76$	**0.004**	Outside vs. Control; $p <$ **0.001**
Bilang-bilangan and Canlangi	Treatment	$F_{2,36} = 5.09$	**<0.001**	Inside vs. Outside; $p = 0.079$
	Year	$F_{3,36} = 3.89$	**<0.001**	Inside vs. Control; $p <$ **0.001**
	Treatment × Year	$F_{6,36} = 1.54$	**0.025**	Outside vs. Control; $p =$ **0.002**
Batasan and Canlangi	Treatment	$F_{2,45} = 10.80$	**<0.001**	Inside vs. Outside; $p <$ **0.001**
	Year	$F_{4,45} = 3.91$	**<0.001**	Inside vs. Control; $p <$ **0.001**
	Treatment × Year	$F_{8,45} = 1.82$	**<0.001**	Outside vs. Control; $p <$ **0.001**

Significant *p* values are shown in bold.

and Batasan only while both Inside and Outside communities were significantly different from Control fish communities at all reserves.

Average total fish density across all transects and sampling periods for each Treatment (Inside/Outside reserves and Controls) varied greatly, with coefficients of variation ranging from 42% to 172%. The most numerous fish families were the Pomacentridae (damselfish), Apogonidae (cardinalfish) and Labridae (wrasses), which on average made up 32%, 19% and 15%, respectively, of the total count per transect.

3.3. *Response of top-trophic level fishes*

Average densities of top-trophic level fishes were generally very low, with distributions strongly skewed left, with a long upper tail ranging from 0 to 206 fish 250 m^{-2} (mean = 12 ± 22 sd; median = 6). Mean densities ranged from 5 (±4 sd) fish 250 m^{-2} at Putik Control to 36 (±40 sd) fish per 250 m^2 at Bilang-bilangan reserve. On average three families accounted for the majority of top-trophic level fishes: Serranidae (46%), Nemipteridae (24%) and Lutjanidae (14%), with only small numbers of Haemulidae (3%) and Lethrinidae (1%).

Overall, we found that the response of top-trophic level fish to protection varied significantly among sites and over time ($F_{3,90} = 7.87$, $p < 0.001$ for Year; $F_{4,34} = 3.89$, $p = 0.010$ for Site; Fig. 3). The Year × Treatment interaction ($F_{3,90} = 3.33$, $p = 0.035$) showed that there was a significant effect of reserve protection while the Site × Treatment interaction ($F_{4,34} = 2.85$, $p = 0.038$) indicated the response was site-specific. At Handumon, the longest established reserve, there was no change in abundance of top-trophic level fish over time but there were significantly more Inside the reserve than Outside or at the Control site (Table 4). Similarly, at Asinan, there were more top-trophic level fish Inside the reserve although the Year × Treatment interaction was not significant (Table 4). At Batasan and Pandanon abundance of top-trophic level fish increased significantly Inside the reserve compared to other Treatments while at Bilang-bilangan there was a significant increase both Inside and Outside the reserve, although the latter was greater (Table 4).

Despite being the most abundant top-trophic level fish family, the Serranidae did not show any significant overall effect of reserve protection although there were significant differences in abundance between Sites ($F_{4,30} = 2.49$, $p < 0.001$). Only at Batasan was there a highly significant increase in serranids over time Inside the reserve compared to other Treatments (similar to that shown for all top-trophic fish in Fig. 3), while the significant Year × Treatment interaction at Asinan and its Control, Pandao, reflected the large numbers of serranids at the Control site in 2003 and 2004 (Table 4).

Similarly, there was no overall effect of reserve protection over time on the abundance of Nemipteridae (non-significant Year × Treatment interaction) despite significant differences among Years, Sites and Treatments ($F_{3,90} = 17.02$, $p < 0.001$ for Year; $F_{4,30} = 4.71$, $p = 0.005$ for Site; $F_{1,30} = 8.17$, $p = 0.008$ for Treatment; $F_{4,30} = 4.90$, $p = 0.004$ for Site × Treatment). Again, at Batasan, there was a highly significant increase in nemipterids over time Inside the reserve compared to other Treatments (Table 4). Furthermore, abundance of nemipterids was higher Inside reserves than other Treatments at three of the five sites (Asinan, Batasan and Bilang-bilangan) (Table 4).

3.4. *Response of mid-trophic level fishes*

Densities of mid-trophic level fishes were considerably higher than the top-trophic level group, and again were strongly skewed left, with a long upper tail ranging from 0 to 860 fish 250 m^{-2}, with a mean of 92 (±102 sd) fish 250 m^{-2} across all sites. The Labridae and the Scaridae made up the majority of fish in this group, accounting for on average 72% and 19%, respectively, of all mid-trophic level fishes recorded. Overall, the other families were relatively uncommon: Mullidae (4%), Siganidae (2%), Holocentridae (<1%), Acanthuridae (<1%) and Balistidae (<1%).

Mid-trophic level fishes did not show clear reserve effects with non-significant Year × Treatment interactions for both labrids and scarids ($p > 0.10$ in both cases; Figs. 4 and 5). Labrids did show significant differences in abundance among Years and Sites as well as a significant Site × Treatment interaction ($F_{3,90} = 23.18$, $p < 0.001$ for Year; $F_{4,30} = 21.83$, $p < 0.001$ for Site; $F_{4,30} = 2.95$, $p = 0.036$ for Site × Treatment; Fig. 4). Batasan was the only site at which there appeared to be evidence of increased abundance of labrids over time relative to the Control site (Table 4). Overall, at Pandanon there was higher abundance of labrids both Inside and Outside the reserve relative to the Control site, Putik. There was a large recruitment of labrids at Pandao Control in 2003–2004 (Fig. 4).

Scarids showed similar patterns to labrids with significant differences among Years, Sites and Site × Treatment interaction ($F_{3,90} = 8.04$, $p < 0.001$ for Year; $F_{4,30} = 2.14$, $p = 0.002$ for Site; $F_{4,30} = 3.28$, $p = 0.024$ for Site × Treatment) but no consistent response to protection among the different reserves (Fig. 5, Table 4). For individual reserves, Batasan showed increasing abundance Inside the reserve over time compared with other Treatments but this was not significant at our reduced α (Table 4).

3.5. *Response of other fishes*

Pomacentrids were the dominant low-trophic level fishes and reached very high abundances at some sites: over 1500 fish 250 m^{-2} in one case. As with the mid-trophic level families, there were strong Year, Site and Site × Treatment effects ($F_{3,90} = 15.68$, $p < 0.001$ for Year; $F_{4,30} = 35.52$, $p < 0.001$ for Site; $F_{4,30} = 5.82$, $p < 0.001$ for Site × Treatment) but no significant Year × Treatment interaction (Fig. 6). There were significant differences in pomacentrid abundance between Inside, Outside and Control locations at two reserve sites: Handumon and Pandanon, though densities were only higher Inside the reserve at Handumon (Fig. 6, Table 4).

Chaetodontids were the dominant coral health indicator family with approximately an order of magnitude greater abundance than pomacanthids. Across all reserves there were significant Treatment effects ($F_{1,30} = 5.04$, $p = 0.032$) but there was no significant Year × Treatment interaction (Fig. 7). Similar to other families there were strongly significant effects of Year, Site, Site × Treatment interaction as well a Year × Site interaction ($F_{3,90} = 21.44$, $p < 0.001$ for Year; $F_{4,30} = 10.56$, $p < 0.001$ for Site; $F_{4,30} = 4.04$, $p = 0.010$ for

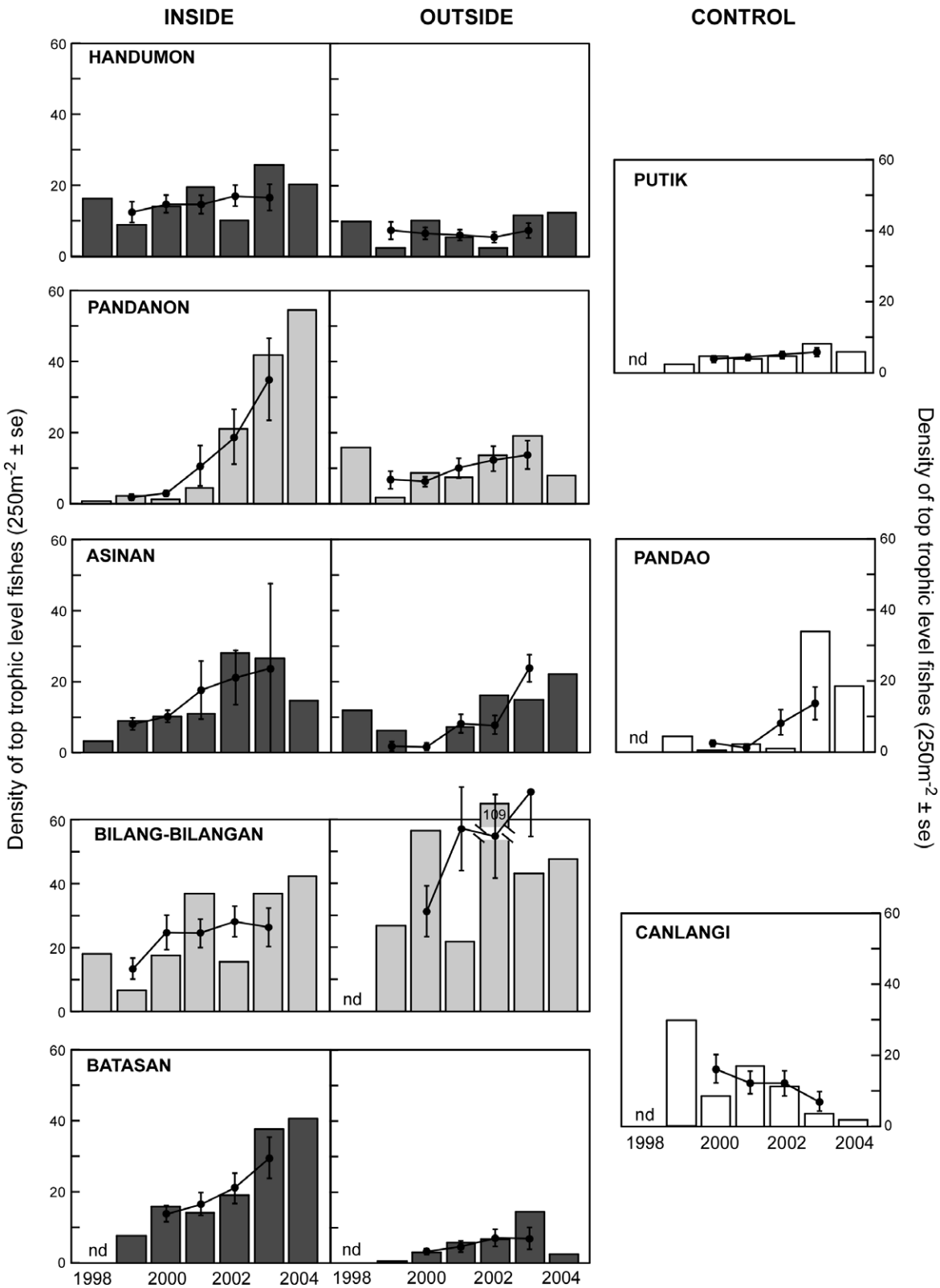

Fig. 3 – Abundance of top trophic level fishes inside and outside five reserves in the central Philippines and three associated Control sites. Mean abundance for each year is shown by bars (omitting error estimates for clarity) while the three-year running mean is shown by the superimposed line. Darker shades indicate greater levels of protection. nd = no data.

Site × Treatment; $F_{12,90} = 2.86$, $p = 0.003$ for Year × Site). At both Batasan and Bilang-bilangan chaetodontids were significantly more abundant Inside the reserves (Fig. 7, Table 4).

As expected, there was a range of responses to reserve protection among fish families (Fig. 8). The majority of families with greater numbers inside the reserves were those known

Table 4 – Summarized results of individual repeated-measures ANOVAs for each Reserve and associated Control

Fish taxon	Source of variation		Handumon and Putik		Pandanon and Putik		Asinan and Pandao		Batasan and Canlangi		Bilang-bilangan and Canlangi	
			F	Sig.	*F*	Sig.	*F*	Sig.	*F*	Sig.	*F*	Sig.
Top-trophic level	Between-subjects	Year	3.64	0.037	7.64	**<0.001**	9.86	**0.001**	0.88	0.500	0.47	0.801
		Year × Treatment	0.70	0.615	3.89	**<0.001**	2.48	0.066	5.50	**<0.001**	3.04	**0.004**
	Within-subjects	Intercept	572.88	**<0.001**	206.52	**<0.001**	4980.38	**<0.001**	390.40	**<0.001**	1655.38	**<0.001**
		Treatment	12.68	**0.002**	0.92	0.419	81.88	**<0.001**	16.02	**0.001**	32.30	**<0.001**
Family Serranidae	Between-subjects	Year	3.25	0.014	3.78	**0.006**	39.57	**<0.001**	10.32	**<0.001**	6.34	0.027
		Year × Treatment	1.63	0.129	0.79	0.636	6.70	**<0.001**	37.99	**<0.001**	6.75	0.011
	Within-subjects	Intercept	116.92	**<0.001**	53.40	**<0.001**	217.45	**<0.001**	811.93	**<0.001**	1368.17	**<0.001**
		Treatment	5.22	0.031	1.85	0.213	0.39	0.688	0.12	0.892	3.82	0.058
Family Nemipteridae	Between-subjects	Year	2.16	0.075	3.17	0.016	6.07	**<0.001**	5.96	**<0.001**	1.28	0.286
		Year × Treatment	1.11	0.379	2.40	0.022	1.43	0.197	3.76	**0.001**	1.78	0.089
	Within-subjects	Intercept	180.95	**<0.001**	120.58	**<0.001**	207.70	**<0.001**	379.60	**<0.001**	99.30	**<0.001**
		Treatment	1.15	0.359	2.19	0.168	25.08	**<0.001**	123.47	**<0.001**	12.35	**0.002**
Family Labridae	Between-subjects	Year	8.63	**<0.001**	6.87	**<0.001**	13.25	**<0.001**	9.78	**<0.001**	3.09	0.017
		Year × Treatment	1.34	0.238	0.67	0.742	3.04	**0.005**	7.53	**<0.001**	2.64	0.012
	Within-subjects	Intercept	444.70	**<0.001**	2627.31	**<0.001**	1073.30	**<0.001**	11006.48	**<0.001**	5207.95	**<0.001**
		Treatment	7.016	0.015	30.66	**<0.001**	1.58	0.259	15.47	**<0.001**	3.13	0.088
Family Scaridae	Between-subjects	Year	6.06	**0.007**	10.55	**<0.001**	9.69	**<0.001**	39.18	**<0.001**	11.90	**<0.001**
		Year × Treatment	0.88	0.506	1.70	0.110	0.48	0.893	3.88	0.018	1.20	0.313
	Within-subjects	Intercept	116.23	**<0.001**	157.41	**<0.001**	405.54	**<0.001**	1121.94	**<0.001**	530.05	**<0.001**
		Treatment	4.45	0.045	0.14	0.872	4.32	0.049	15.05	**0.001**	9.32	**0.005**
Family Pomacentridae	Between-subjects	Year	8.21	**<0.001**	8.70	**0.003**	39.57	**<0.001**	10.32	**0.001**	6.34	0.027
		Year × Treatment	1.82	0.084	2.21	0.111	6.70	**<0.001**	37.99	**<0.001**	6.75	0.011
	Within-subjects	Intercept	283.72	**<0.001**	862.51	**<0.001**	217.45	**<0.001**	811.93	**<0.001**	1368.17	**<0.001**
		Treatment	13.93	**0.002**	46.05	**<0.001**	0.39	0.688	0.12	0.892	3.82	0.058
Family Chaetodontidae	Between-subjects	Year	4.78	**0.001**	2.70	0.032	17.60	**<0.001**	3.45	**0.010**	1.41	0.238
		Year × Treatment	2.30	0.028	1.55	0.154	0.944	0.503	2.56	0.015	2.44	0.019
	Within-subjects	Intercept	147.75	**<0.001**	84.73	**<0.001**	246.62	**<0.001**	626.85	**<0.001**	613.86	**<0.001**
		Treatment	3.18	0.090	2.76	0.116	1.95	0.198	13.20	**0.002**	17.27	**0.001**

Significant differences are shown in bold using a Bonferroni corrected $\alpha = 0.01$ to control for multiple comparisons within each rmANOVA.

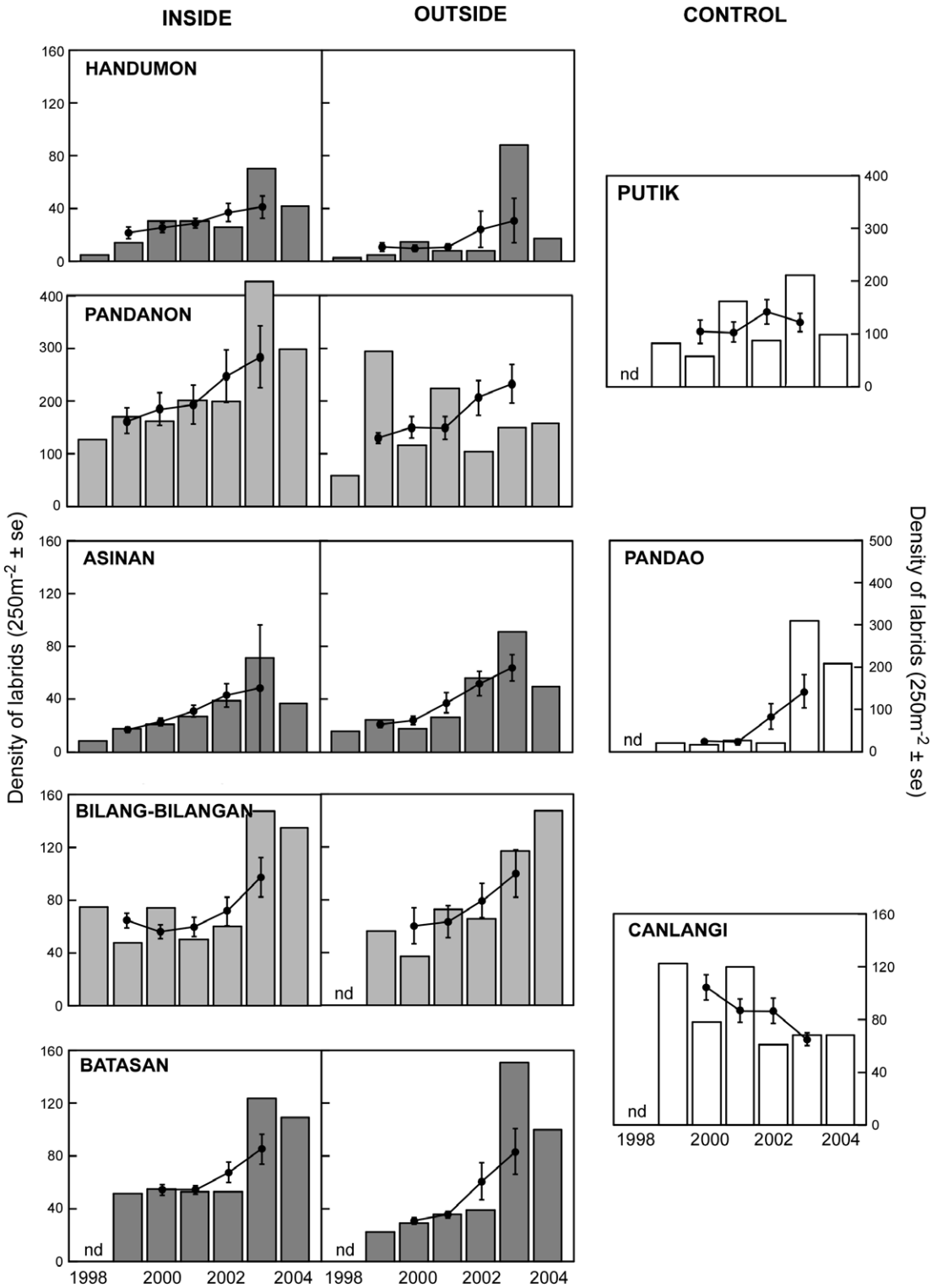

Fig. 4 – Abundance of family Labridae inside and outside five reserves in the central Philippines and three associated Control sites. Mean abundance for each year is shown by bars (omitting error estimates for clarity) while the three-year running mean is shown by the superimposed line. Darker shades indicate greater levels of protection. nd = no data. Note different scale on y-axis for Pandanon reserve and Pandao Control.

to be larger and/or targeted by fishers such as acanthurids, haemulids, nemipterids, serranids and siganids, of which serranids and haemulids were significantly so (Fig. 8). The cryptic families, or those less-preferred by fishers, such as apogonids, blennids, caesionids and syngnathids were more abundant outside the reserves, the former two significantly so.

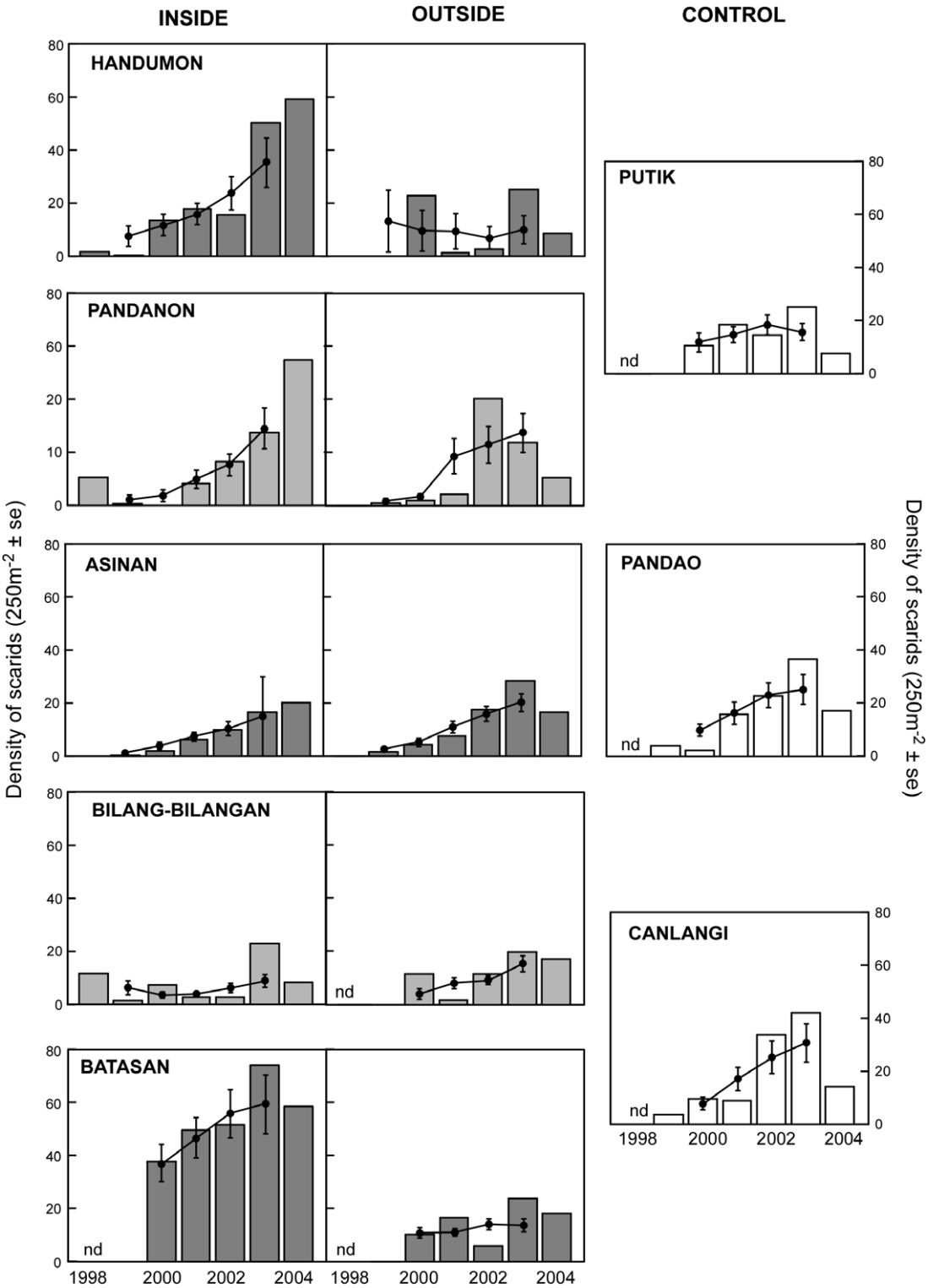

Fig. 5 – Abundance of family Scaridae inside and outside five reserves in the central Philippines and three associated Control sites. Mean abundance for each year is shown by bars (omitting error estimates for clarity) while the three-year running mean is shown by the superimposed line. Darker shades indicate greater levels of protection. nd = no data.

4. Discussion

Our study provides evidence of the beneficial effects of reserves for fish communities, including species taken by local fisheries, but also shows that effects vary considerably among reserves. Significant positive effects appeared to be dependent on good levels of enforcement and/or compliance. However, even at reserves with poor compliance, abundance of

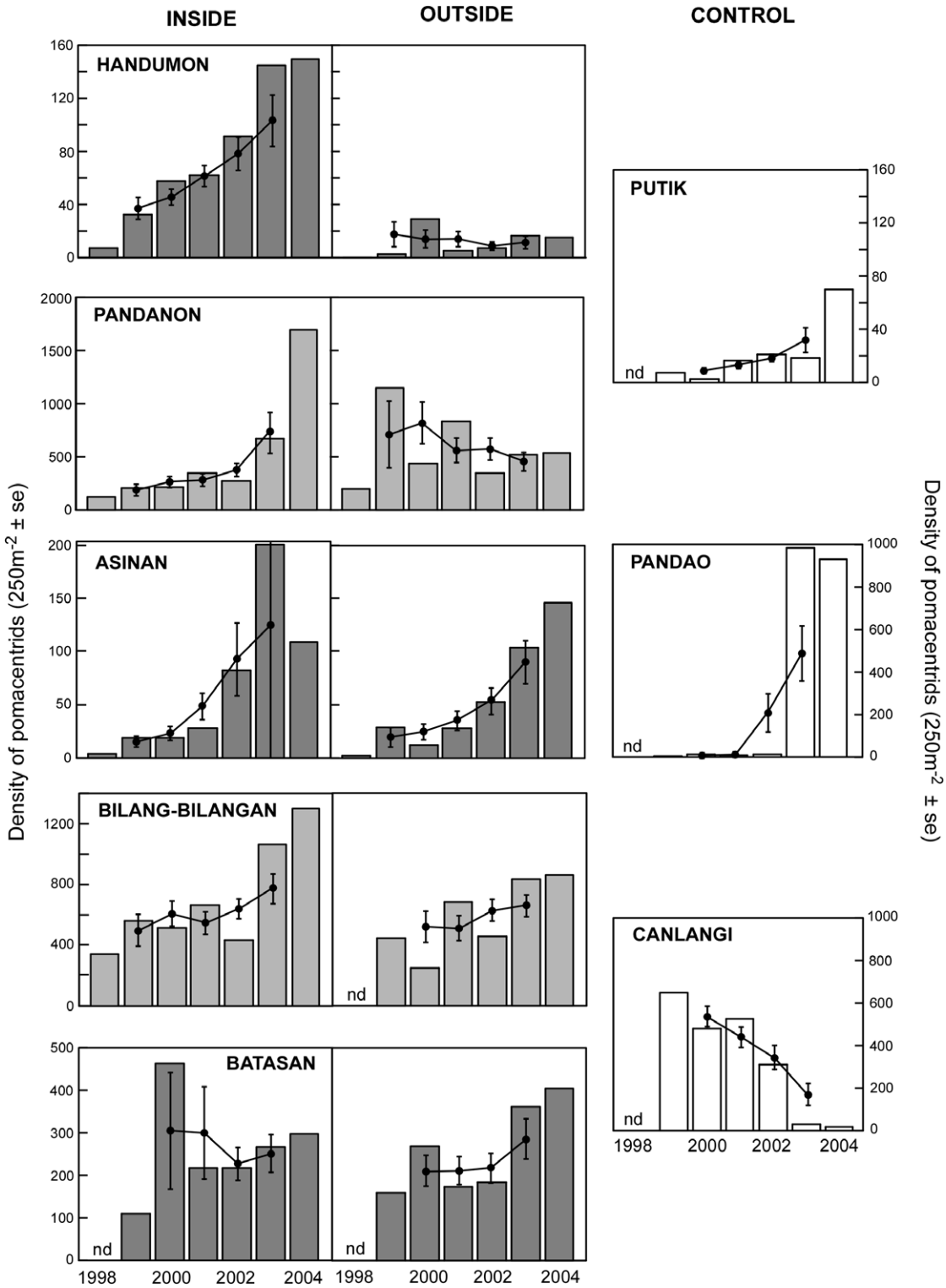

Fig. 6 – Abundance of family Pomacentridae inside and outside five reserves in the central Philippines and three associated Control sites. Mean abundance for each year is shown by bars (omitting error estimates for clarity) while the three-year running mean is shown by the superimposed line. Darker shades indicate greater levels of protection. nd = no data. Note different scales on y-axis among sites.

most fish taxa remained the same or greater than at Control sites, even if these differences were not statistically significant. This suggests reserve status confers an overall benefit to fish communities in this central region of the Philippines, where fish numbers are generally declining if outside such reserves. Reserve effects were most apparent with top-trophic level fishes and coral health indicator species, suggesting that both cessation of direct fishing mortality and protection of

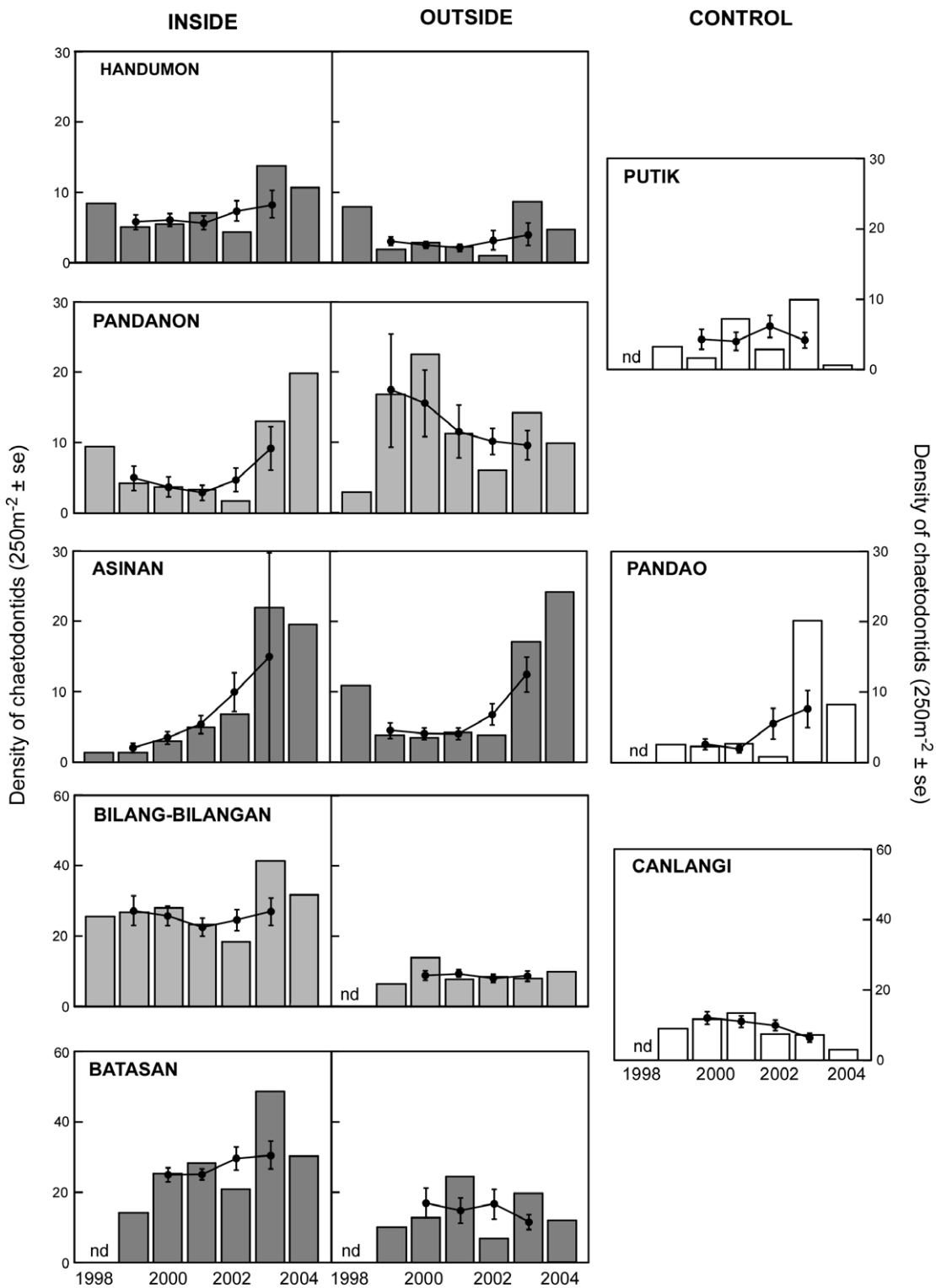

Fig. 7 – Abundance of family Chaetodontidae inside and outside five reserves in the central Philippines and three associated Control sites. Mean abundance for each year is shown by bars (omitting error estimates for clarity) while the three-year running mean is shown by the superimposed line. Darker shades indicate greater levels of protection. nd = no data. Note different scale on y-axis for Asinan and Bilang-bilangan reserves and Canlangi Control.

habitat from destructive practices were important to recovery of fish communities.

Enforcement of fishing bans within reserves is clearly very important. While it might seem axiomatic that reserves will only work when fishing is prevented, few reserve studies present data on enforcement and compliance, with one notable exception: Walmsley and White (2003) ranked measures of enforcement and management at four reserves in the Philip-

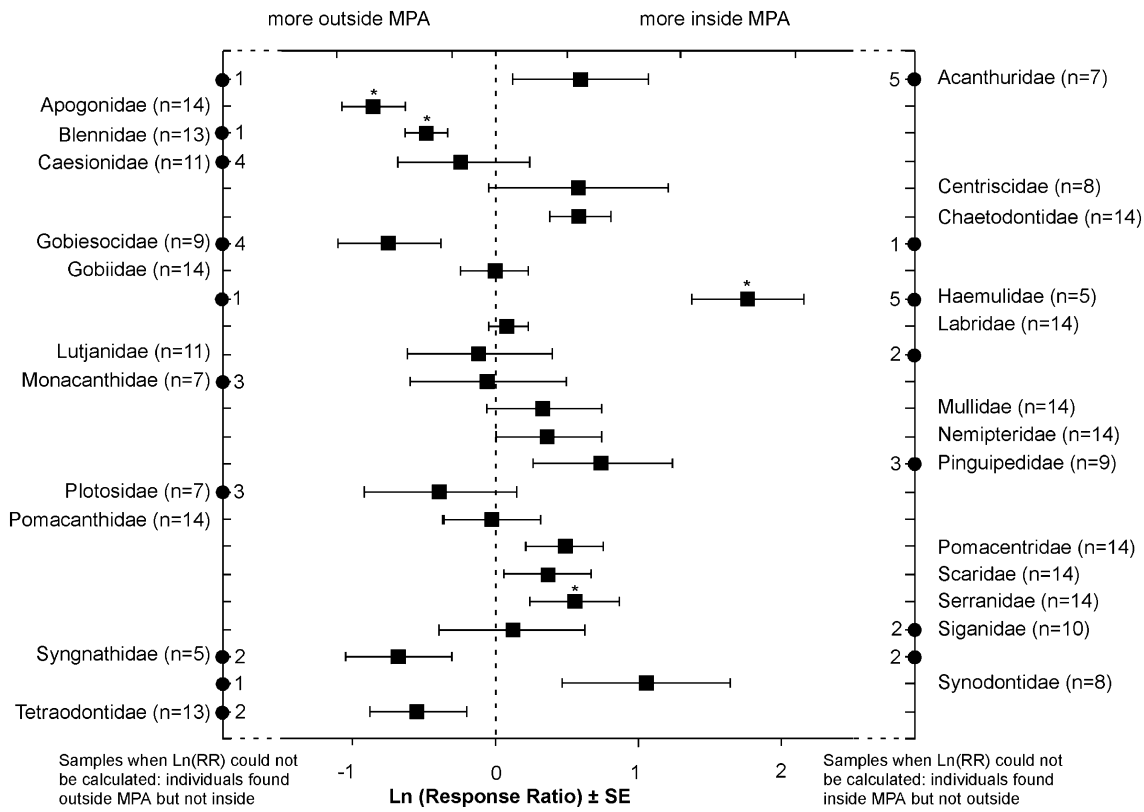

Fig. 8 – Ln Response Ratios (Inside: Outside) for abundance of twenty-four families of fishes from five reserves from the central Philippines for 2002–2004. Ratios significantly different from zero are indicated with an asterisk.

pines over a 19 year period, and found it to be significantly correlated with increases in fish abundance. Similarly, an absence of 'no-take' reserve effects on a predatory and highly targeted serranid on the Great Barrier Reef in Australia was attributed to poaching (Russ, 2002). Our study included reserves assessed contemporaneously across a range of qualitative enforcement levels. Only at the two reserves with best enforcement (Handumon and Batasan) did we find significant differences between fish communities Inside and Outside the reserve. At Asinan, a site with good enforcement but occasional encroachment and slightly later commencement, we found some evidence that differences in fish communities were developing.

Enforcement of fishing bans and compliance with these bans are dependent on many factors. The qualitative system that we used to rank our marine reserves provides a systematic basis for assessing the contribution of these factors (White et al., 2006). Marine reserves rated highly will have legal ordinances, management plans, secure funding, biophysical monitoring and educational feedback while poorer performing reserves will lack community acceptance, adequate planning and appropriate funding (White et al., 2006).

Village communities confirmed poor enforcement as a possible reason for the lack of positive change at certain reserve sites. This involvement of local communities in interpreting the monitoring datasets is known to improve management of marine sanctuaries (Verheij et al., 2004), and continues to be a strong component of our reserve monitoring programme in Bohol. Detailed measures of enforcement, compliance and community participation in these reserves in Bohol are now underway to examine these relationships further (Pajaro, unpublished data). When proposing new reserves, adequate provision (funding, logistics, institutional support) must be made for ongoing enforcement whether by local communities or fisheries management agencies.

Despite the small size (<1 km^2) of the study reserves, we still detected positive effects on some locally important species, supporting findings that reserve size does not significantly affect proportional changes in fish abundance i.e. the percentage increase following reserve establishment (Halpern, 2003). Small reserves are potentially easier to establish and enforce and will have less impact in the initial stages on economically disadvantaged subsistence fishers often with limited vessel range (Sanchirio and Wilen, 2001; Carter, 2003).

Predatory fish (serranids and nemipterids) responded strongly to well-enforced reserves in our study, although there was also a significant site effect perhaps related to larger-scale habitat differences. Serranids showed significant positive reserve effects at inshore Handumon and offshore Batasan, while nemipterids were more abundant and showed large increases in abundance only at two offshore reserves, Asinan and Batasan. Increases in abundance or biomass of predatory fish to protection from fishing within coral reef reserves have been reported around the world including Australia, Brazil, Egypt, Kenya, New Caledonia, the Philippines and Tanzania (Samoilys, 1988; Watson and Ormond, 1994;

Wantiez et al., 1997; Russ and Alcala, 1998; McClanahan et al., 1999; Williamson et al., 2004; Ashworth and Ormond, 2005; Floeter et al., 2006). These fish are usually highly prized for food but are also largest in body size, which seems to be an important, independent predictor of response to protection (Mosquera et al., 2000; Jennings, 2001; Russ, 2002). The serranids also represent some of the more vulnerable reef fishes due to their life history strategies (Sadovy and Vincent, 2002), making this taxon an important indicator to monitor long term. Monitoring taxa at different trophic levels is especially important for assessing fishing impacts, since fishing reduces abundance of higher trophic levels before lower trophic levels (Jennings and Lock, 1996; Pauly et al., 1998).

Increased abundance of labrids and scarids both Inside and Outside the reserves appeared to be a general pattern, but these could not be distinguished from abundance increases at the Controls, indicating broad scale changes in the abundance of these taxa. A significant positive reserve effect on the pomacentrids was only seen at Handumon while at other reserve sites they tended to be more abundant both Inside and Outside the reserve than at Control sites. Our data for these three families were similar to results of a meta-analysis of reserve effects in 12 studies (Mosquera et al., 2000), wherein all three families were significant more abundant within reserves, though in the case of the labrids, only when fished species were considered (Mosquera et al., 2000). The Labridae are a large and diverse family ranging from the larger fishery species (e.g. *Cheilinus* spp.) to the small non-target species (e.g. *Halichoeres* spp.), and therefore reserve effects are likely to be masked by grouping these species together. Other studies have shown different patterns; for example pomacentrids and some small labrids were more abundant outside marine parks in Kenya (McClanahan et al., 1999). Local fishing practices and habitat may determine the direction of response to reserves of these lower trophic groups.

One of the strongest positive responses to reserves in our study was shown by the chaetodontids, which have been proposed as important indicators of human disturbance on coral reefs because many species have a close association with living coral and benthic habitat heterogeneity (Reese, 1977; Hourigan et al., 1988; Crosby and Reese, 1996; Bozec et al., 2005). Indeed, White (1988) demonstrated that densities of chaetodontids in the Philippines were significantly related to hard coral cover across multiple sites. In Bohol, the increase in these fishes is likely to be linked to increases in coral cover or retention of reef topography, presumably due to protection from blast fishing. The proximity of Outside sites to reserves (within visual range) may have provided indirect protection from destructive fishing techniques, particularly for well-enforced reserves with guards. Elsewhere chaetodontid responses to reserves have not been consistent. While some studies showed increased abundance (Wantiez et al., 1997; Öhman et al., 1997), others have shown no effect (McClanahan and Arthur, 2001; Williamson et al., 2004) or site-specific responses (Russ and Alcala, 1998). These differences may be due to the confounding effects of fishing and coral destruction by particular types of fishing gear. Whilst we recommend that chaetodontids are monitored in reserve studies, we recognize that they may not be of general utility in all situations (Öhman et al., 1998).

We found negative responses to reserves for 10 families of fishes although only two were significantly so. Many of these families are small and/or cryptic and it is difficult to determine whether the effects that we observed were real responses to changed conditions within the reserve (e.g. increased predator abundance) or artefacts such as increased visibility in the structurally less-complex habitats outside the reserve (see Ackerman and Bellwood, 2000). Theoretically, some families and species might be expected to experience negative effects from reserves (Jennings, 2001). Importantly, it is known that standard UVCs are not suitable for detecting differences in small and cryptic taxa (Samoilys and Carlos, 2000).

Our study demonstrates the value in monitoring reserve effects at more than one reserve site, and the mix of results highlights the complexity of processes involved, from natural fluctuations to taxon-specific responses to protection. Site effects were significant in all cases and Site × Treatment interactions in all but one of our analyses. The many possible sources of inter-site differences include physicochemical conditions, local and regional biogeography, disturbance history and anthropogenic effects. This makes the selection of appropriate control sites in studies such as ours very problematic. Usually we do not know the original state of putative control areas nor their exploitation history. Nevertheless, it is still imperative to try and match reserve areas with exploited areas in order to partition community variance between different factors (Gell and Roberts, 2003). By having a number of reserve sites together with distant Control sites in our study we were able to tease apart some of the factors contributing to inter-site differences with some confidence.

Demonstration of spillover is of critical importance in the debate over the effectiveness of reserves as a conservation or fisheries management option (Ward et al., 2001; Gell and Roberts, 2003). Our study provides some evidence for spillover at well-enforced reserves, where the increase in abundance of serranids (at Batasan), nemipterids (at Asinan) and labrids (at Batasan and Handumon) Outside was seen to mirror the increase Inside the reserve, but at a lower rate, without similar increases at the Controls. However, the power to detect spillover is low when there are such high levels of spatial variation as we found. The best evidence for spillover requires considerable effort through following individuals tagged within the reserve or sampling intensively across the reserve boundary to some considerable distance (e.g. Russ et al., 2003; Kaunda-Arara and Rose, 2004; Ashworth and Ormond, 2005). Our study can inform choice of species and sampling protocols when designing an intensive experiment to demonstrate spillover. The benefit in monitoring areas just outside reserves for detecting localised fisheries effects is clear. Evidence for spillover is a positive message to feed back to communities.

In conclusion, we have demonstrated empirically the value of small reserves for fish communities, particularly taxa taken in local fisheries, in the central Philippines, conditional on good levels of enforcement and compliance. Evidence comes from monitoring fishes at the family level only which lends support to similar community-based monitoring programmes (e.g. Uychiaoco et al., 2000). As reserves have broad scale support across stakeholders in the region (Martin-Smith

et al., 2004), we urge their continued establishment with appropriate support from management agencies, donors, policymakers and the communities that depend on them.

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