



# Evidence for spillover of reef fishes from a no-take marine reserve: An evaluation using the before-after control-impact (BACI) approach

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## ABSTRACT

No-take marine reserves (NTRs) may promote recovery of exploited populations within their boundaries and spillover of fishes to adjacent fishing grounds, thus potentially benefiting the local fisheries. Although some studies have measured spillover by examining gradients of fish abundance and body size across reserve boundaries, there are no such studies to date including information from before reserve establishment, thus seriously limiting interpretation of results. We measured reef fish spillover from a NTR (Itacolomis Reef, Eastern Brazil) by estimating biomass and body size across the reserve boundary before (2001) and after (2002–2005) initiation of protection. Replicate sites were sampled inside and outside the reserve, with unprotected sites included in three distance categories from the reserve boundary: 0–400, 400–800 and 800–1200 m. This latter category generally surpasses the scales over which spillover is expected to influence patterns of fish abundance outside reserves (generally <500 m), particularly for relatively sedentary fishes, thus acting as a control for the reserve effect. Habitat measurements were undertaken at the same sites, from 2003 on. Biomass of *Scarus trispinosus*, a major fishery resource and the dominant species in terms of biomass (37.4% of total biomass), was lower inside the reserve area before its establishment. During this same period, no individuals of two primary target species, *Mycteroperca bonaci* and *Sparisoma axillare*, were recorded inside the reserve. Coral cover was consistently lower inside the reserve from 2003 on. Biomass and body size of *M. bonaci*, as well as biomass of *Ocyurus chrysurus*, increased continuously inside the reserve after its establishment, with no similar increases recorded in control sites. Evidences of spillover (i.e. higher biomass inside the reserve and in unprotected sites closer to its boundary) were obtained for *M. bonaci*, *O. chrysurus* and *S. trispinosus*, although this pattern was only marginally significant for *O. chrysurus*. Despite these positive signs, recovery and spillover of *S. trispinosus* were probably inhibited by increased poaching from 2003 on. Our results indicate that the NTR at Itacolomis Reef was established *a priori* on poor quality habitats. Thus, future spatial comparisons between protected and unprotected sites would underestimate changes due to protection. These findings highlight the importance of baseline information and continued monitoring for adequately understanding the effects of NTRs, as well as the need of closer collaboration between natural and social scientists in order to effectively protect high-quality habitats in the long term while accounting for the socio-economic needs of local fishing communities.

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## 1. Introduction

No-take marine reserves (i.e. areas of the marine environment fully protected from fishing and all other types of exploitation) are recognized as an important tool for reef fisheries management (Bohnsack, 1998; Roberts and Hawkins, 2000). This recognition is largely based on the overfished status of many reef fish stocks

worldwide, and also on the difficulties in implementing conventional management strategies, such as catch and effort restrictions, for managing such complex and multi-species fisheries (Roberts and Polunin, 1993; Russ and Alcala, 1996). No-take reserves may promote the recovery of critical spawning stock biomass within their boundaries (Halpern and Warner, 2002) and benefit adjacent fishing grounds via exportation of biomass (Gell and Roberts, 2003). Despite mounting positive evidences (e.g. Roberts et al., 2001; Russ et al., 2004) the usefulness of no-take marine reserves as a fisheries management tool over large spatial and temporal scales is still controversial (Willis et al., 2003; Hilborn et al., 2004; Sale et al., 2005).

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Exportation of biomass from no-take reserves may occur through two mechanisms: net emigration of juveniles and adults (spillover effect) and net export of pelagic eggs and larvae (recruitment effect) (Gell and Roberts, 2003). The spillover effect is expected to operate over small spatial scales (from tens to a few hundred meters, depending on species' mobility and habitat connectivity), while the recruitment effect is expected to operate more diffusely and over broad spatial scales (generally tens of kilometers, depending on the dispersal capability of the pelagic larvae and patterns of ocean currents) (Gell and Roberts, 2003). Fisheries benefits derived from the spillover effect may be minimal when compared to the benefits expected from the recruitment effect (Abesamis et al., 2006). However, most demonstrations of marine reserves functioning available to date focused on the spillover effect, due to the inherent difficulties in evaluating the recruitment effect (Gell and Roberts, 2003; Sale et al., 2005). Despite its lower contribution to the local fisheries, demonstrations of benefits from spillover are important for obtaining support for reserves from local fishing communities (Russ and Alcala, 1996; Sale et al., 2005; Abesamis et al., 2006).

Marine reserve theory predicts that fishes will tend to move from inside reserves, where fish abundance is expected to be higher due to protection, to outside reserves, where fish abundance is expected to be lower due to fishing pressure (Rakitin and Kramer, 1996; Chapman and Kramer, 1999). Such directional movements are likely to be produced by increased density-dependent competitive interactions inside reserves (Abesamis and Russ, 2005). Spatial patterns in abundance of non-target fishes should not be directly affected by reserves, although indirect cascading effects may occur (e.g. increased predation risk) (Micheli et al., 2004). Finally, reserves may affect fish body size in a similar way they affect fish abundance, with larger individuals expected to occur in the center of reserves and smaller individuals beyond their boundaries (Rakitin and Kramer, 1996; Chapman and Kramer, 1999).

Most studies on the effects of marine reserves have performed only spatial comparisons of reserve and fished sites at one time (Halpern and Warner, 2002; Halpern, 2003). Temporal comparisons incorporating samples obtained before and after reserve establishment, with concomitant monitoring of control sites ("before-after control-impact", BACI), is plausibly a more powerful sampling design (Osenberg and Schmitt, 1996; Russ, 2002; Halpern et al.,

2004; but see Underwood, 1996). Evidence supporting the spillover effect is even more equivocal. Several studies have measured gradients of fish abundance and body size or catch per unit of effort across or near reserve boundaries (Rakitin and Kramer, 1996; McClanahan and Kaunda-Arara, 1996; Russ and Alcala, 1996; Russ et al., 2004). Other studies have used tag-recapture techniques to demonstrate fish movement from reserve to non-reserve areas (e.g. Holland et al., 1993; Zeller and Russ, 1998; Johnson et al., 1999). In a few cases the contribution of reserves to the absolute yields of the nearby fishery was demonstrated (Alcala et al., 2005; Abesamis et al., 2006). No studies to date have monitored gradients of fish abundance and body size across reserve boundaries starting from before reserve establishment (Russ, 2002).

Extractive reserves are co-managed multiple-use protected areas in Brazil established by the initiative of local communities with support from the Federal Protected Areas Agency (formerly IBAMA, now ICMBio). The Marine Extractive Reserve of Corumbau (MERC) was the first Extractive Reserve to encompass coral reefs, and also the first one to make use of no-take reserves as a fishery management tool (Moura et al., 2007). In this study we evaluated the effects of a no-take reserve established within the MERC by addressing three main questions: (1) Are target fish populations recovering inside the reserve? (2) Are there evidences of fish spillover from the reserve to the surrounding unprotected reefs? (3) Is protection from fishing leading to the occurrence of larger fishes inside the reserve and in sites closer to its boundary? Fish assemblages were surveyed yearly before (2001) and after (2002–2005) the reserve establishment, both inside and outside the reserve. Our findings indicate that baseline information and long-term monitoring efforts are critical for adequately understanding the effects of no-take marine reserves.

## 2. Material and methods

### 2.1. Study area

The MERC is located in the northern portion of the Abrolhos Bank, Eastern Brazil (Fig. 1), covering 930 km<sup>2</sup> of nearshore habitats and coralline reefs. Since its legal establishment in 2000 the MERC is co-managed by a deliberative council in which fishermen occupy 50% plus one additional chair. Exploitation of marine resources

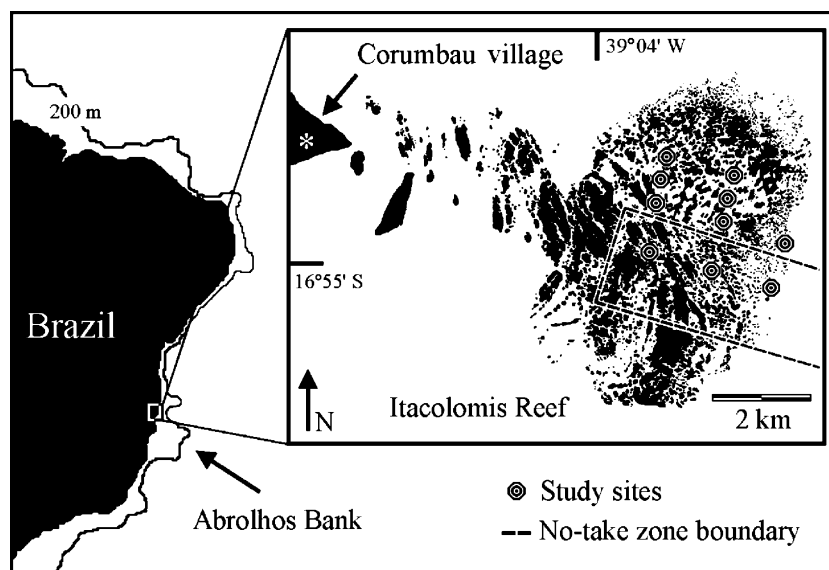


Fig. 1. Map showing Itacolomis Reef and study sites. The main fishing village near Itacolomis Reef (Corumbau) is highlighted.

is only allowed for locals with use rules (e.g. zoning and gear restrictions) defined by the Deliberative Council and endorsed by the ICMBio. Handlining, spearfishing and various types of nets are allowed, while destructive fishing practices (e.g. drive-nets above reefs and collections for aquarium trade) are prohibited. The parrotfish (family Scaridae) fishery was banned from the entire MERC during the summer of 2002 through an informal agreement among local fishermen.

The possibility of using no-take reserves as a fishery management tool was debated since 1999, even before the legal establishment of the MERC. No-take reserves embedded within the multiple-use Extractive Reserve were envisaged as a mean to restore fish populations and benefit adjacent fishing grounds through spillover. The Management Plan, approved in November 2001, created several no-take zones, the main one (~10 km<sup>2</sup>) covering about 20% of the largest reef complex within the MERC (Itacolomis Reef). This reserve extends from the central portion of Itacolomis Reef to the eastern limit of the MERC. Fishing pressure decreased sharply soon after the reserve establishment, but increased from 2003 on. In 2004 the buoys marking the reserve boundaries were destroyed by the sea, contributing to a further decrease in compliance. The management history of the MERC is described in detail by Moura et al. (2007).

## 2.2. Sampling design and field measurements

Sampling was undertaken inside (3 sites) and outside the reserve (7 sites), with unprotected sites included in three zones from the northern reserve boundary: 0–400 (3 sites), 400–800 (2 sites) and 800–1200 m (2 sites) (Fig. 1). Since emigration of fishes from reserves is expected to influence patterns of fish abundance outside reserves generally on scales <500 m (Roberts and Polunin, 1991; Russ, 2002), the 800–1200 m zone was considered as a control to the reserve effect. However, it is important to note that large reserves (such as the one at Itacolomis Reef) could probably export fish to unprotected zones beyond 500 m, especially highly mobile carnivores such as the species from family Carangidae (Russ et al., 2004). Thus, results obtained for these species must be interpreted with caution. Each site was about 300 m in diameter and composed of one to three interconnected reef pinnacles. Surveys were always done in the summer (January–March), thus avoiding seasonal artifacts.

Reef structures in the Abrolhos region display a characteristic form of mushroom-shaped pinnacles, sometimes fused together and forming large platforms (Leão and Kikuchi, 2001). Two habitats were sampled within each site: tops (horizontal inclination; 2–6 m depth) and walls (vertical inclination; 3–15 m depth). Fish counts were made using a nested stationary visual census technique in which individuals ≤10 cm total length (TL) are counted in a 2 m radius and individuals >10 cm TL are counted in a 4 m radius (Minte-Vera et al., in press). Individuals ≤10 cm TL were recorded in two different size categories: ≤2 and 2–10 cm, while individuals >10 cm TL were recorded in four size categories: 10–20, 20–30, 30–40 and >40 cm. Individuals ≤2 cm TL for small-bodied species (≤25 cm maximum TL) and individuals ≤10 cm TL for large-bodied species (>25 cm maximum TL) were not included in the analysis in order to reduce errors (Bellwood and Alcalá, 1988). Extensive training in fish size estimation was undertaken prior to sampling with the use of fish models to minimize operator variability (Samoily, 1997). Fifteen to twenty samples were obtained per habitat per site per year, totaling 1557 samples along the entire study period.

Habitat measurements were undertaken from 2003 to 2005 at the same sites where fish assemblages were surveyed. Benthic cover was estimated using two distinct methods, one for each habitat. Point-intercept lines (10 m length; 100 points) were

haphazardly laid on pinnacle tops, and groups of four quadrats (50 cm × 50 cm; 25 intercepts) equally distributed within a 10 m line were haphazardly laid on pinnacle's walls. Each group of quadrats was considered as a single sample. Organisms immediately below each point were recorded and classified as follows: algal turf, crustose calcareous algae, fleshy algae, live coral, octocoral, sea urchin and zoanthid. Topographic complexity on pinnacle's tops was estimated with the deployment of a 10 m chain following all contours and crevices of the bottom. Surface length relative to linear chain length was used as an index of complexity (Luckhurst and Luckhurst, 1978). Eight benthic cover samples and four complexity samples were obtained per site per year.

## 2.3. Data analysis

Species were grouped as targeted and non-targeted by fisheries. Target species were further subdivided into three trophic categories (small carnivores, large carnivores and large herbivores; Table 1). Quantitative analyses were performed for each category and for the most abundant target species (>1% of total biomass). Despite their relatively high abundance, the three species within family Acanthuridae and *Haemulon aurolineatum* (see Table 1) were not analyzed separately because they are rarely captured in the MERC. The eight species retained for analysis belong to five primary target reef fish families (Carangidae, Haemulidae, Lutjanidae, Scaridae and Serranidae) and represented more than 60% of total fish biomass recorded in this study (Table 1).

Fish counts were converted to biomass using length–weight relationships (Froese and Pauly, 2006). Estimates were calculated by multiplying the weight from the midpoint of each size class by the number of fish per size category, and then summing size categories (cf. McClanahan and Kaunda-Arara, 1996). When length–weight information was not available, parameters from similarly sized congeners were used. Counts of benthic organisms were converted to percentages.

Two-way analysis of variance (ANOVA) was used to evaluate spatial and temporal variation in fish biomass and habitat characteristics. Reef zones (no-take plus the three unprotected zones) and years (2001–2005 for fish biomass and 2003–2005 for habitat characteristics) were the two main factors in the ANOVA. Additional one-way ANOVA were performed to clarify spatial patterns in fish biomass according to reef zones within each year. Normality and homocedasticity were improved by converting fish biomass to log<sub>10</sub> (x+1) and benthic cover percentages to arcsin √x. Student–Newman–Keuls (SNK) multiple comparisons of means were performed as *post hoc* tests. Size distributions of fishes between reef zones and years were compared using the Kolmogorov–Smirnov two-sample test (Zar, 1999). This latter analysis was performed only for the main target species within each trophic group (namely *Mycteroperca bonaci*, *Ocyurus chrysurus* and *Scarus trispinosus*).

Changes in the structure of fish assemblages over time were evaluated using the Bray–Curtis similarity index (Bray and Curtis, 1957). Similarities in fish biomass were calculated for all pairs of protected and unprotected sites within each year, and then correlated to the duration of protection. Both target and non-target species were included in order to account for possible trophic cascade effects (Micheli et al., 2004).

Canonical correspondence analysis (CCA; ter Braak, 1996) was used to evaluate the influence of habitat characteristics and distance from the reserve boundary on the structure of target reef fish assemblages. Only data from 2003 to 2005 were used in the CCA because no habitat measurements were made before this period. A forward selection procedure was employed to select the three most important independent variables affecting fish assemblages.

**Table 1**  
Reef fish species recorded at Itacolomis Reef from 2001 to 2005

Family/species	OF	TC	%	Family/species	OF	TC	%
Synodontidae				Chaetodontidae			
<i>Synodus intermedius</i>	N	–	0.9	<i>Chaetodon sedentarius</i>	N	–	<0.1
Muraenidae				<i>Chaetodon striatus</i>	N	–	0.3
<i>Gymnothorax funebris</i>	T	SC	<0.1	Pomacanthidae			
<i>Gymnothorax moringa</i>	T	SC	<0.1	<i>Holacanthus ciliaris</i>	N	–	0.2
<i>Gymnothorax vicinus</i>	T	SC	<0.1	<i>Pomacanthus arcuatus</i>	N	–	3.5
Holocentridae				<i>Pomacanthus paru</i>	N	–	4.8
<i>Holocentrus ascensionis</i>	N	–	<0.1	Pomacentridae			
Scorpaenidae				<i>Abudefduf saxatilis</i>	N	–	1.8
<i>Scorpaena brasiliensis</i>	N	–	<0.1	<i>Chromis multilineata</i>	N	–	<0.1
<i>Scorpaena plumieri</i>	N	–	<0.1	<i>Microspathodon chrysurus</i>	N	–	0.6
Sphyraenidae				<i>Stegastes pictus</i>	N	–	<0.1
<i>Sphyraena barracuda</i>	T	LC	0.3	<i>Stegastes</i> spp. <sup>a</sup>	N	–	4.5
Serranidae				Ephippidae			
<i>Cephalopholis fulva</i>	T	SC	<0.1	<i>Chaetodipterus faber</i>	T	SC	0.2
<i>Epinephelus adscensionis</i>	T	LC	<0.1	Labridae			
<i>Epinephelus itajara</i>	T	LC	<0.1	<i>Bodianus rufus</i>	T	SC	<0.1
<i>Epinephelus morio</i>	T	LC	<0.1	<i>Halichoeres brasiliensis</i>	T	SC	0.3
<i>Mycteroperca bonaci</i>	T	LC	1.7	<i>Halichoeres penrosei</i>	N	–	<0.1
<i>Serranus flaviventris</i>	N	–	0.2	<i>Halichoeres poeyi</i>	N	–	0.3
Cirrhitidae				Scaridae			
<i>Amblycirrhitus pinos</i>	N	–	<0.1	<i>Cryptotomus roseus</i>	N	–	<0.1
Carangidae				<i>Scarus trispinosus</i>	T	LH	37.4
<i>Carangoides bartholomaei</i>	T	LC	0.5	<i>Scarus zelindae</i>	T	LH	<0.1
<i>Carangoides crysos</i>	T	LC	1.5	<i>Sparisoma amplum</i>	T	LH	2.9
<i>Carangoides rubber</i>	T	LC	<0.1	<i>Sparisoma axillare</i>	T	LH	3.1
<i>Caranx latus</i>	T	LC	0.1	<i>Sparisoma frondosum</i>	T	LH	0.4
<i>Pseudocaranx dentex</i>	N	–	0.3	Labrisomidae			
Lutjanidae				<i>Malacoctenus</i> sp.	N	–	<0.1
<i>Lutjanus jocu</i>	T	LC	0.4	Gobiidae			
<i>Lutjanus synagris</i>	T	LC	0.2	<i>Coryphopterus</i> spp. <sup>b</sup>	N	–	0.3
<i>Lutjanus alexandrei</i>	T	LC	<0.1	<i>Elacatinus Figaro</i>	N	–	<0.1
<i>Ocyurus chrysurus</i>	T	SC	1.5	Bleniidae			
Haemulidae				<i>Ophioblennius atlanticus</i>	N	–	<0.1
<i>Anisotremus moricandi</i>	T	SC	<0.1	<i>Parablennius marmoratus</i>	N	–	<0.1
<i>Anisotremus surinamensis</i>	T	SC	1.1	<i>Scartella cf. cristata</i>	N	–	<0.1
<i>Anisotremus virginicus</i>	T	SC	1.1	Acanthuridae			
<i>Haemulon aurolineatum</i>	T	SC	2.3	<i>Acanthurus bahianus</i>	T	LH	2.2
<i>Haemulon parra</i>	T	SC	0.8	<i>Acanthurus chirurgus</i>	T	LH	11.5
<i>Haemulon plumieri</i>	T	SC	0.9	<i>Acanthurus coeruleus</i>	T	LH	11.2
Sparidae				Tetraodontidae			
<i>Calamus pennatula</i>	T	SC	<0.1	<i>Sphoeroides spengleri</i>	N	–	<0.1
Mullidae				Diodontidae			
<i>Pseudupeneus maculatus</i>	T	SC	0.2	<i>Diodon hystrix</i>	N	–	0.2

Occurrence in fisheries (OF): T, target and N, non-target. Trophic category (TC): LH, large herbivores; LC, large carnivores and SC, small carnivores. Percentage of total biomass, %.

<sup>a</sup> Data pooled for *Stegastes fuscus* and *S. variabilis*.

<sup>b</sup> Data pooled for *Coryphopterus dicrus*, *C. glaucofraenum* and *C. thrinx*.

Absolute values of distance from the reserve boundary were used instead of categories.

### 3. Results

#### 3.1. Fish assemblages

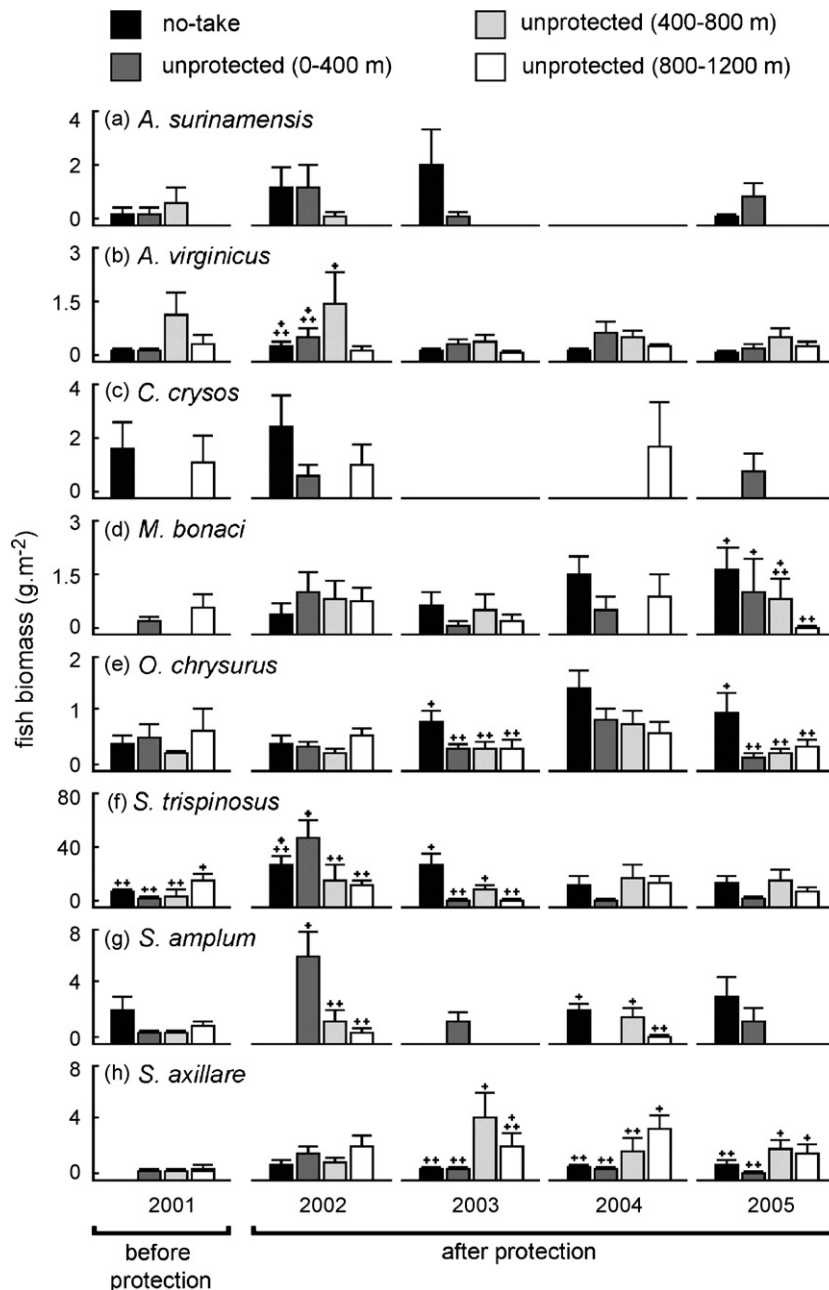
Predominant target fish species in terms of biomass (>1% of total biomass) are shown in Table 1. Large herbivores dominated fish assemblages (83.6% of total target fish biomass), followed by small carnivores (10.4%) and large carnivores (6.0%). A single herbivore species, the greenbeak parrotfish *Scarus trispinosus*, accounted for the highest fish biomass (37.4% of total fish biomass and 45.6% of total target fish biomass).

Before the reserve's establishment (2001) there were significant differences in biomass between reef zones for *S. trispinosus*, large herbivores and non-target fishes. In all cases, highest biomass values were recorded in reefs farther (400–1200 m) from the reserve boundary (Fig. 2). In addition, no individuals of the black grouper *Mycteroperca bonaci* and the gray parrotfish *Sparisoma axillare* were

recorded inside the area selected for the reserve in 2001. During this same period, a relatively higher proportion of large (>40 cm TL) *M. bonaci* individuals was recorded in the 800–1200 m zone (Fig. 4a), while larger *S. trispinosus* individuals were recorded inside the area selected for the reserve and in reefs nearer to its boundary (0–400 m) (Fig. 4b). Size distributions of the yellowtail snapper *Ocyurus chrysurus* were relatively homogeneous, except for a higher proportion of small individuals (2–10 cm TL) in the 400–800 m zone (Fig. 4c).

Significant temporal variation in fish biomass was recorded for all but two species (the grunts *Anisotremus surinamensis* and *A. virginicus*). The interaction between reef zone and year was not significant for *A. surinamensis*, *A. virginicus*, small carnivores and non-target fishes, indicating that spatial variation of these species/groups was consistent through time (Table 2). Biomass of *A. surinamensis* was highest inside the reserve and in the 0–800 m zone (Fig. 2a); biomass of *A. virginicus* was highest in the 400–800 m zone (Fig. 2b); biomass of small carnivores was highest in the 0–800 m zone, particularly between 2002 and 2004 (Fig. 3a; Table 3); biomass of non-target fishes in the 400–1200 m





**Fig. 2.** Biomass (mean + 95% confidence limits) of target fish species across the no-take reserve boundary along 5 years. Homogeneous groups within each year, as determined by Student–Newman–Keuls (SNK) *post hoc* comparisons, are linked by equal number of signals.

zone, with highest values recorded between 2001 and 2003 (Fig. 3e; Table 3).

All other species/groups showed significant or marginally significant ( $p < 0.075$ ) interactions between reef zone and year, indicating different temporal trajectories within each zone (Table 2). Biomass of the most abundant high mobile piscivore, the blue runner *Carangoides crysos*, was highest in 2001 and 2002, particularly inside the reserve and in the 800–1200 m zone, declining sharply from 2003 on (Fig. 2c). Biomass of *M. bonaci* and *O. chrysurus* increased along the entire study period inside the reserve, showing more variable responses in unprotected reefs. By the end of the study biomass of these two species was lower in control sites (800–1200 m) than before the reserve establishment (Fig. 2d and e).

Biomass of *S. trispinosus* increased sharply inside the reserve and in unprotected sites closer to its boundary (0–400 m) soon after

initiation of protection (2002), declining in the unprotected zone from 2003 on, and also in the no-take zone from 2004 on (Fig. 2f). Biomass of the parrotfish *Sparisoma amplum* varied greatly during the study period, with highest values recorded inside the reserve in 2005 and in the 0–400 m zone in 2002 (Fig. 2g). Biomass of *S. axillare* increased both inside and outside the reserve, but such increase was more pronounced in reefs farther (400–1200 m) from the reserve boundary (Fig. 2h). Biomass of large carnivores showed no clear temporal trend, although by the end of the study it was relatively higher inside the reserve than in unprotected reefs (Fig. 3b). Patterns of large herbivores and target fishes as a group mirrored that of *S. trispinosus*, due to the dominance of this species within these groups (Fig. 3c and d).

A spatial pattern indicative of spillover (i.e. higher biomass inside the reserve and in unprotected reefs closer to its boundary)

**Table 2**

Two-way analyses of variance (ANOVA) testing the effect of reef zone (no-take and the three unprotected zones) and year (2001 to 2005) on reef fish biomass

	Reef zone		Year		Reef zone × year	
	F	p	F	p	F	p
<b>Species</b>						
<i>A. surinamensis</i>	3.11	0.025	1.59	0.174	0.98	0.460
<i>A. virginicus</i>	8.02	<0.001	1.37	0.239	0.60	0.842
<i>C. crysos</i>	2.71	0.043	3.76	0.004	1.72	0.055
<i>M. bonaci</i>	1.73	0.156	2.66	0.031	1.98	0.022
<i>O. chrysurus</i>	3.69	0.011	8.92	<0.001	1.63	0.074
<i>S. amplum</i>	3.42	0.016	3.80	0.004	6.57	<0.001
<i>S. axillare</i>	10.99	<0.001	9.21	<0.001	3.25	<0.001
<i>S. trispinosus</i>	2.33	0.072	7.13	<0.001	4.70	<0.001
<b>Trophic groups</b>						
Large herbivores	2.15	0.091	12.67	<0.001	6.44	<0.001
Large carnivores	3.50	0.014	6.36	<0.001	1.78	0.045
Small carnivores	4.38	0.004	14.30	<0.001	1.28	0.220
<b>Totals</b>						
Target fishes	1.66	0.173	16.32	<0.001	4.91	<0.001
Non-target fishes	21.41	<0.001	3.01	0.017	1.05	0.395

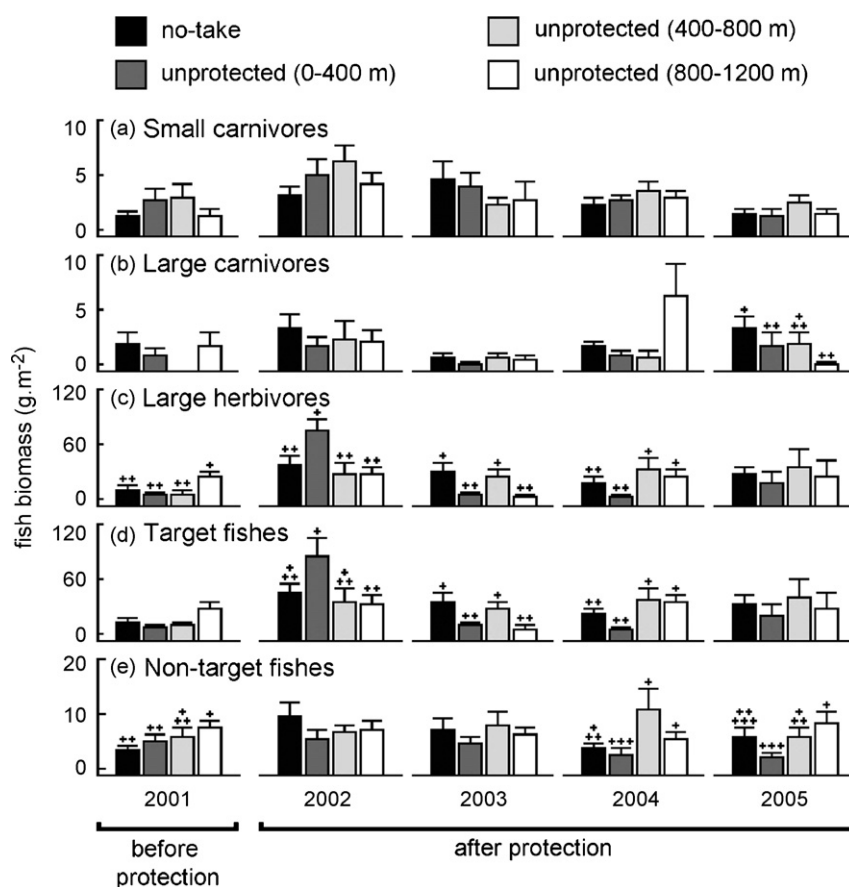
was recorded for *M. bonaci* in 2005 (Fig. 2d) and for *S. trispinosus* in 2002, although in this latter case biomass in the 0–400 m zone was even higher than inside the reserve (Fig. 2f). There was also a clear decline in biomass of *O. chrysurus* from inside to outside the reserve in 2004, but this pattern was only marginally significant ( $p < 0.075$ ) (Fig. 2e). Evidences of spillover were always associated with years in which highest biomass values of these species were recorded inside the reserve (Fig. 2d–f).

Body size of *M. bonaci* increased inside the reserve during the entire study period, showing more variable responses in unprotected reefs. From 2003 on, larger *M. bonaci* individuals were consistently recorded inside the reserve than elsewhere (Fig. 4a). Body size of *S. trispinosus* was higher inside the reserve from 2001 to 2004, but declined in the last year of the study (2005), when larger individuals were recorded in the 0–400 m zone (Fig. 4b). Body size of *O. chrysurus* increased inside the reserve from 2001 to 2004, but a similar increase was recorded in the 400–800 m zone (Fig. 4c).

Similarity in fish assemblages' structure between protected and unprotected sites decreased through time. This trend was observed for each zone separately, and also for all unprotected sites pooled. However, the relationship was statistically significant only when calculations were made considering the sites within the 0–400 m zone ( $r^2 = 0.12$ ;  $p = 0.015$ ) and all unprotected sites pooled ( $r^2 = 0.09$ ;  $p = 0.001$ ).

### 3.2. Habitat characteristics

Turf algae (40.4%) was the dominant benthic cover type considering all sites and sampling periods, followed by fleshy algae (18.4%), live corals (9.5%), crustose calcareous algae (9.1%), zoanths (7.8%), octocorals (1.1%) and sea urchins (0.5%; mainly *Echinometra lucunter* and *Lytechinus variegatus*). There were no significant differences between reef zones only for benthic complexity and for fleshy and turf algae cover. Crustose calcareous algae, sea urchins and zoanthid cover were relatively higher inside the reserve, while live coral and octocoral cover were higher in the 400–800 m and the 0–400 m zones, respectively (Fig. 5). Significant



**Fig. 3.** Biomass (mean + 95% confidence limits) of target and non-target fishes across the no-take reserve boundary along 5 years. Homogeneous groups within each year, as determined by Student–Newman–Keuls (SNK) *post hoc* comparisons, are linked by equal number of signals.

**Table 3**  
Significant differences in fish biomass according to reef zones and years, as determined by Student–Newman–Keuls (SNK) *post hoc* comparisons (groups arranged in decreasing order of fish biomass)

	SNK	
	Reef zone	Year
Species		
<i>A. surinamensis</i>	NT = UP1 = UP2 > UP1 = UP2 = UP3	ns
<i>A. virginicus</i>	UP2 > UP1 = UP3 = NT	ns
<i>C. crysos</i>	NT = UP3 = UP1 > UP3 = UP1 = UP2	02 = 01 > 01 = 05 = 04 = 03
<i>M. bonaci</i>	ns	04 = 05 = 02 = 03 > 05 = 02 = 03 = 01
<i>O. chrysurus</i>	NT > UP3 = UP1 = UP2	04 > 03 = 02 = 05 = 01
<i>S. amplum</i>	UP1 = NT = UP2 > NT = UP2 = UP3	02 = 04 = 01 = 05 > 05 = 03
<i>S. axillare</i>	UP3 = UP2 > UP1 = NT	04 = 02 = 03 = 05 > 01
<i>S. trispinosus</i>	ns	02 > 03 = 05 = 04 = 01
Trophic groups		
Large herbivores	ns	02 > 04 = 03 = 05 > 03 = 05 = 01
Large carnivores	NT = UP3 > UP3 = UP1 = UP2	02 = 04 = 05 > 01 = 03
Small carnivores	UP2 = UP1 > UP1 = UP3 = NT	02 = 04 = 03 > 01 = 05
Totals		
Total target fishes	ns	02 > 04 > 03 = 05 > 05 = 01
Total non-target fishes	UP2 = UP3 > NT > UP1	02 = 03 = 01 > 03 = 01 = 05 = 04

Reef zones: NT, no-take and UP, unprotected. Distance categories from the no-take zone boundary: 1, 0–400 m, 2, 400–800 m and 3, 800–1200 m.

interannual variability was recorded only for crustose calcareous algae, which declined slightly in 2004 and increased again in 2005. The interaction between reef zone and year was significant only for turf algae cover (Table 4).

### 3.3. Influence of habitat characteristics and distance from the no-take zone boundary on reef fish assemblages

Results from the CCA showed that depth, fleshy algae cover and distance from the reserve boundary were, in decreasing order, the main predictors of reef fish assemblage structure. The first two axes of the CCA explained 96% of the relationship between explanatory variables and fish assemblages' structure. Sites closer to the reserve boundary were also relatively deeper. Four species (*A. surinamensis*, *M. bonaci*, *O. chrysurus* and *S. amplum*) were strongly associated with deeper protected sites and intermediate levels of fleshy algae cover; *Scarus trispinosus* with middle-depth protected sites and low fleshy algae cover; *Carangoides crysos* with middle-depth unprotected sites dominated by fleshy algae; *Anisotremus virginicus* and *S. axillare* with shallow unprotected sites and middle to high fleshy algae cover (Fig. 6).

## 4. Discussion

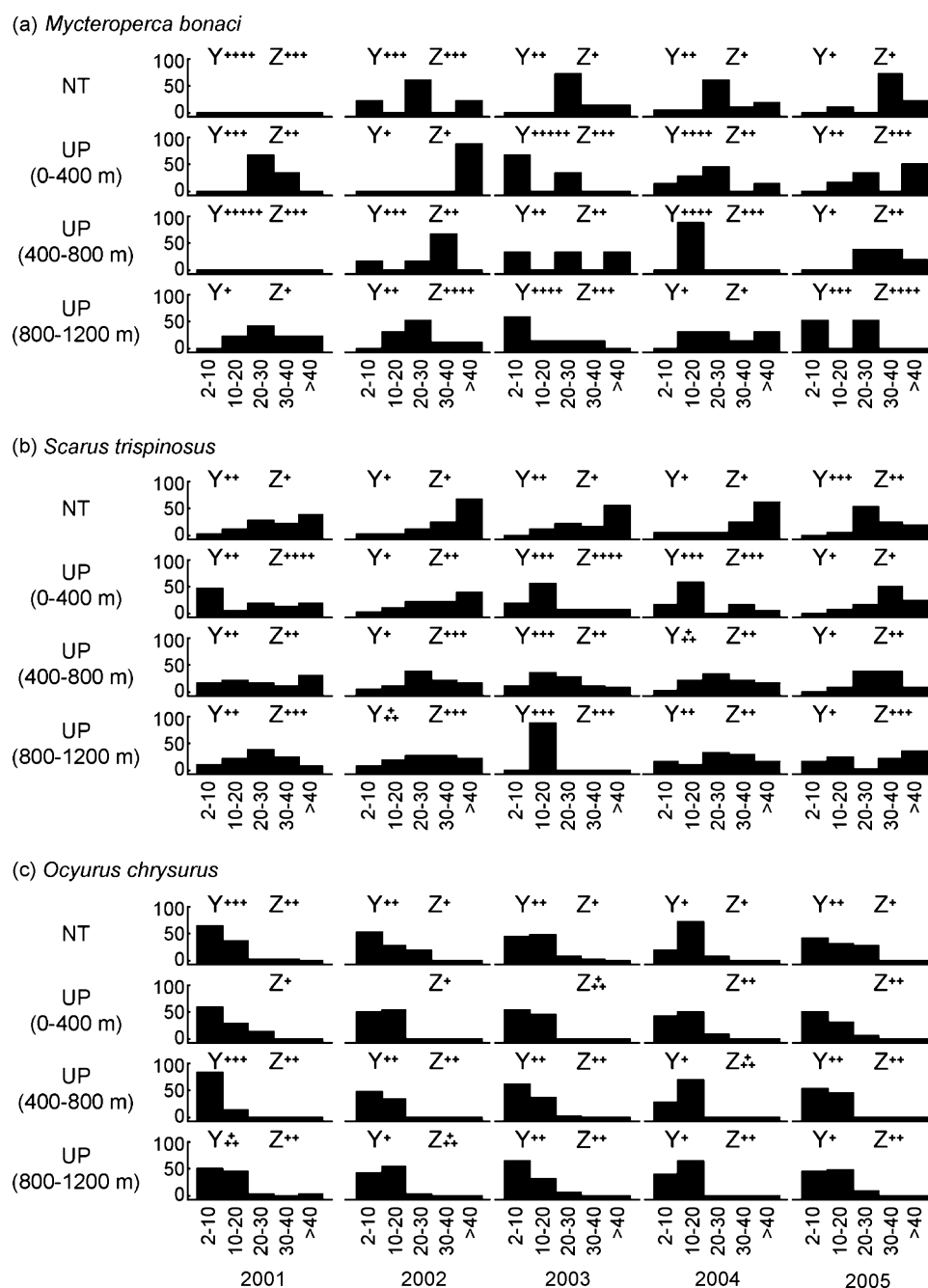
Despite the increased interest in evaluating and improving the effectiveness of no-take marine reserves, there are very few studies using powerful sampling designs (Russ, 2002; Halpern et al., 2004), thus generally leading to skepticism over the usefulness of reserves as a fishery management tool (Gell and Roberts, 2003). Examining gradients of fish biomass and body size across reserve boundaries and comparing information from before and after reserve establishment are important steps towards a more comprehensive understanding on the role of reserves (Russ, 2002).

Results from the present study indicate that the reserve at Itacolomis Reef was established *a priori* on poor quality habitats, even after an extensive negotiation process involving local fishermen, scientists, NGOs and the Federal Protected Areas Agency. Biomass of large herbivorous fishes, particularly of the dominant species in terms of biomass, *S. trispinosus*, was lower inside the reserve area before its establishment. In addition, no individuals of *M. bonaci* and *S. axillare* were recorded inside the reserve during this same period. Finally, habitat integrity was consistently lower inside the reserve,

as indicated by the lower coral and octocoral cover (Luckhurst and Luckhurst, 1978; Bell and Galzin, 1984). Although not quantitatively estimated few changes in benthic cover were noticed since the beginning of the study, suggesting that this poor habitat condition was already in place before the reserve establishment. Without these informations, future spatial comparisons of the no-take versus the unprotected zones would be misleading, with an underestimated assessment of change in fish biomass due to protection. In fact, socio-political bias in the process of establishing marine reserves may be the rule, rather than the exception (Edgar et al., 2004), thus seriously limiting the interpretation of results from studies performing only spatial comparisons of reserve and fished sites at one time (e.g. Halpern and Warner, 2002; Floeter et al., 2005).

Given that the results from the baseline fish survey (2001) were presented to the local fishing community before the final decision on the no-take zone establishment was made, it is clear that fishermen avoided loosing access to some of their traditional fishing spots. This idea is well corroborated by their first proposal, in which only a 1 km<sup>2</sup> portion of shallow reef was considered to be protected. This proposal changed to a large belt extending from the shoreline to the eastern limit of the MERC only after extensive discussions between all stakeholders were made. Following this second proposal, some fishermen that use small non-motorized boats expressed their concerns in protecting coastal reefs, as they are unable to explore outer reefs in periods of rough sea conditions, thus leading to the exclusion of coastal reefs from the final reserve proposal (see Fig. 1 for the final arrangement). Adopting an optimized selection procedure based on sound socio-economic and ecological advice is a critical step for establishing effective reserves (Roberts et al., 2003; Chuenpagdee and Jentoft, 2007), and encouraging local fishermen to protect some of their fishing grounds is an important component of this process.

Our results add to the growing body of empirical evidence showing that no-take reserves promote build-up of biomass of commercially important species within their boundaries and also spillover of fishes to nearby unprotected areas (Halpern and Warner, 2002; Gell and Roberts, 2003). Similarity between protected and unprotected sites declined linearly with increasing duration of protection (see Micheli et al., 2004 for similar findings), indicating that fish assemblages inside the reserve became progressively more different from those outside the reserve. Biomass and body size of *M. bonaci* increased continuously inside the reserve fol-



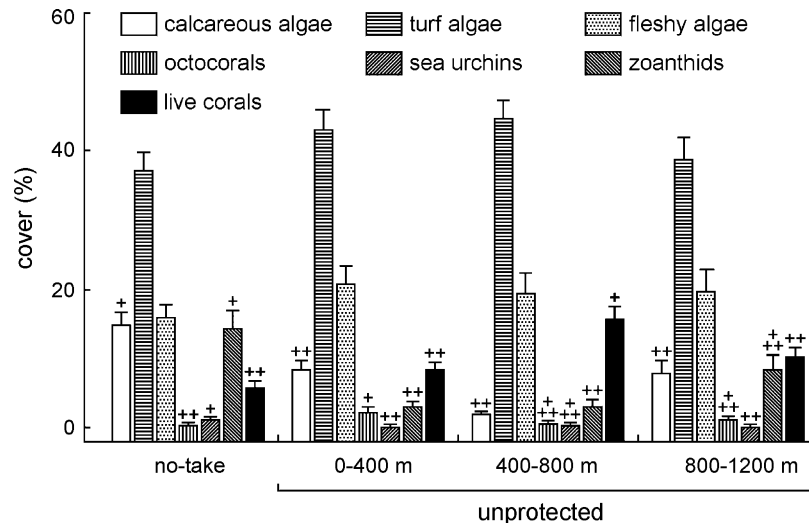
**Fig. 4.** Size frequency distributions of three primary target fish species across the no-take reserve boundary along 5 years. Unprotected reefs are arranged in three distance categories from the reserve boundary (0–400, 400–800 and 800–1200 m). Significant differences in size distribution were determined using pairwise Kolmogorov–Smirnov tests. Homogeneous groups within each year (between reef zones, Z) and within each zone (between years, Y) are linked by equal number of signals.

lowing protection, with no comparable increases being recorded in unprotected reefs. Biomass of *O. chrysurus* showed a ~3.5-fold increase inside the reserve after initiation of protection, and again no similar trend was observed outside the reserve. By the end of the study (2005) biomass of these two species was lower in control sites (800–1200 m) than before the reserve establishment, strengthening the idea that increases recorded inside the reserve were due to protection. Spatial patterns of change in biomass of *M. bonaci* and *O. chrysurus* over time was consistent with the hypothesis that as biomass increased inside the reserve fishes tended to move from inside to outside the reserve, although this pattern was only marginally significant ( $p < 0.075$ ) for *O. chrysurus*. Despite the aforementioned increases, biomass of commercially

important fishes at Itacolomis Reef is still much lower than that in older and better enforced no-take reserves within the Abrolhos Bank (Francini-Filho and Moura, in press). This was expected since many species of reef fishes are long lived and recovery inside reserves may take decades to develop fully (Russ and Alcala, 2004).

Exploitation of four species classified as “non-target” in this study (*Elacatinus figaro*, *Pomacanthus arcuatus*, *P. paru* and *Holacanthus ciliaris*) ceased after the aquarium trade was banned from the entire MERC in 2000. Although biomass of these species is expected to increase over time, both inside and outside the no-take zone, there were no evidences of recovery for the non-target fish category (see Fig. 3e), as well as for each species separately (not shown). This





**Fig. 5.** Benthic cover across the no-take reserve boundary. Homogeneous groups for each benthic cover category (between zones comparisons), as determined by Student–Newman–Keuls (SNK) *post hoc* comparisons, are linked by equal number of signals.

**Table 4**

Two-way analysis of variance (ANOVA) testing the effect of reef zone (no-take and the three unprotected zones) and year (2001–2005) on habitat characteristics

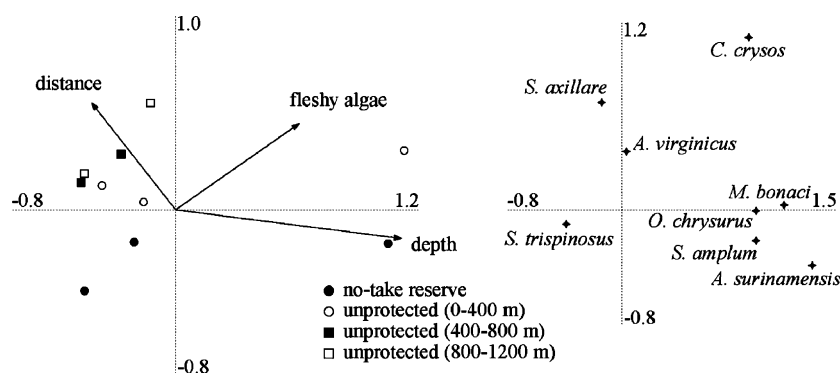
Category	Reef zone		Year		Reef zone × year	
	F	p	F	p	F	p
Calcareous algae	11.28	<0.001	6.35	0.002	1.04	0.394
Complexity	2.09	0.103	0.66	0.515	0.971	0.447
Fleshy algae	0.69	0.553	2.18	0.115	1.36	0.230
Live corals	10.12	<0.001	0.03	0.962	0.99	0.429
Octocorals	4.09	0.007	0.15	0.856	0.79	0.576
Turf algae	1.63	0.181	1.02	0.360	3.05	0.006
Sea urchins	2.83	0.039	1.28	0.277	0.79	0.571
Zoanthids	4.83	0.002	1.50	0.225	0.25	0.955

lack of response may be explained by the fact that exploitation for the aquarium trade was concentrated on coastal reefs and estuarine creeks far from the study area within Itacolomis Reef.

Recovery and spillover of *S. trispinosus* were apparently inhibited by poaching inside the reserve, which tended to increase along the study period. Biomass of this species increased sharply inside the reserve and also in unprotected reefs closer (0–400 m) to its boundary between 2001 and 2002, soon after the reserve establishment and banning of the parrotfish fishery from the MERC. This increase was followed by an abrupt decline in unprotected reefs from 2003 on, after local fishermen decided to re-open the par-

rotfish fishery, and also by a decline inside the reserve from 2004 on (Fig. 2f), after a clear increase in poaching levels by spearfishers. Although poaching levels were not systematically estimated, several instances of illegal fishing inside the no-take zone were witnessed by the authors, always from 2004 on and during the summer (December–April), when most spearfishing activity is concentrated. Informations gathered during informal conversations with local fishermen also support the idea of increased poaching from 2004 on.

Results from the present study, which integrates a broader long-term monitoring program within the MERC (Moura et al., 2007), were shown yearly to locals. Despite being aware of the positive reserve effects, there was a strong resistance from some fishers, particularly spearfishers, to the idea of banning the parrotfish fishery and stop fishing inside the reserve. These results indicate that fishermen knowledge on marine reserves' effectiveness alone does not guarantee compliance. Difficulties for storing and commercializing catches, dependency on a few middlemen and increasingly opportunities for commercializing species such as parrotfishes (some of the few large-bodied fishes still present in relative abundance on the reefs) are important factors affecting fishermen compliance in the long term (Moura et al., 2007). Russ and Alcala (1999) report on the collapse of protection in one out of two small marine reserves in the Philippines (Apo and Sumilon) with similar management histories, pointing to the influence of local politi-



**Fig. 6.** Canonical correspondence analysis (CCA) plot showing: (a) relationship between independent variables (arrows) and monitored sites, and (b) distribution of eight target reef fish species in the two-dimensional ordination space.

cians, limited community engagement and the distant location of supporting institutions as the main factors impeding protective management in Sumilon. Broader community engagement, continued dissemination of results from monitoring programs and fair trade opportunities are possible solutions for increasing fishermen's compliance, although they cannot substitute the role and duties of government in establishing, managing and supporting an effective surveillance regime (Pomeroy and Berkes, 1997; Glaser and Oliveira, 2004).

The capacity of coral reefs to support goods and services on which human welfare depends is seriously compromised (Hughes et al., 2003; Bellwood et al., 2004). There are strong evidences that co-managed protected areas, in which local fishing communities are empowered to play a relevant role, represent an effective solution for managing human activities and for replicating the no-take marine reserves' experience over broad spatial scales (Alcala and Russ, 2006). In Brazil, there are at least five Marine Extractive Reserves already established country-wide, with increasing opportunities for establishing new ones. If adequate support is provided, these multiple-use co-managed protected areas may become a strong marine zoning instrument and a way to establish new no-take reserves.

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