Can private management compensate the ineffective marine reserves in China?

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REPORT

Can private management compensate the ineffective marine reserves in China?

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Abstract Marine reserves (MRs) have emerged as a preferred method to protect coral reefs from overfishing and human disturbance. However, due to ineffective enforcement by governments, many MRs have been reduced to mere “paper parks” which fail to achieve conservation goals. This is especially true in countries such as China where compliance is low and resources dedicated to enforcement may be scarce. Privately managed marine reserves (PMMRs) may be effective in areas where government enforcement is lacking. To determine if PMMRs are a viable alternative strategy to protect coral reefs, we surveyed and compared fish assemblages and coral coverage in national MRs in Sanya, China to areas of reef privately leased to and managed by dive operators and hospitality industries. We found higher fish abundances and fish sizes in PMMR sites than in MR sites. However, while PMMRs are protected from fishing, other human impacts such as marine debris and illegal coral collection were evident in most tourist sites. Despite protection, long-term monitoring data of PMMRs revealed that in recent years, fish abundances have slightly recovered but species richness has not, indicating the need for a more comprehensive coral reef management plan. We strongly recommend coupling PMMRs with expertise supported regulations as an alternative coral reef management strategy in China.

Keywords Coral Reefs · Marine protected areas · Privately managed reserves · South China Sea

Hui Huang and Colin Kuo-Chang Wen have contributed equally to this work.

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INTRODUCTION

Marine protected areas (MPAs) or marine reserves (MRs) have been recognized as an efficient means of maintaining biodiversity and reducing human disturbances on coral reefs (Sale et al. 2011). Increasing human activities, especially overfishing, have threatened coral reefs around the world. Such exploitative activities target large fish species (such as Serranidae, Lutjanidae) and benthic herbivorous invertebrates (such as sea urchins) leading to a decline in populations (Hughes et al. 2007). With the removal of top predators and herbivorous species, trophic cascades occur consequently affecting the balance of corals and macroalgae (Mumby et al. 2007). Prohibition of fishing and collection of benthic species inside MRs could limit reef damage from destructive fishing methods, and maintain the demographic structure of fishery target species as well as the health of coral reef ecosystems (Halpern and Warner 2002). In addition, MRs support surrounding fisheries by increasing recruitment to those populations via larval export and spill over of adult fishes (Harrison et al. 2012). Although many MRs have demonstrated higher biodiversity and abundances of target species, some reserves fail to help reefs recover from coral degradation, with fish populations remaining low despite long periods of protection (Reviews in Russ and Alcala 1999; Alcala and Russ 2006). Additionally, global- or large-scale regional stressors such as climate change, global warming diminish coral reefs beyond the reserves boundary (Allison et al. 1998). Understanding the circumstances and possible reasons behind unsuccessful reserves is the first step toward designing better conservation strategies.

Many factors can contribute to poor resilience and failed recovery of marine organisms in MRs (Sala et al. 2002). Improper location or improper coverage of a reserve might

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prevent populations from recovering (Miller and Russ 2014). For example, some reefs might not connect to surrounding reefs via current and not receive a consistent supply of larval fishes which limits replenishment (Van Herwerden et al. 2009), while some coral reefs might be replenished by these adjacent reefs within a short period of time (Almany et al. 2007). Unsuitable environments such as loose substrata and degraded habitats can also limit the recruitment and settlement of fish and coral larvae (Fox 2004; Feary et al. 2007; Shima et al. 2008). Even if settlement is possible in suboptimal habitats, subsequent survivorship of settled fishes and corals may be relatively low due to constant disturbance (Munday 2001). A critical principal in selecting marine reserves should be the identification of reefs in adequate condition with efficient replenishment and connectivity (Almany et al. 2009). However due to financial constraints, it is generally not practical to examine replenishment and connectivity among reefs prior to designing each marine reserve (Mora et al. 2006), even for well-known and well-managed reefs on the Great Barrier Reef in Australia (McCook et al. 2010) and the Coral Triangle in South Asia (Weeks et al. 2014).

The establishment of a successful marine reserve requires systematic criteria and a design plan that suits the purpose of the reserve, whether that be for maintaining biodiversity, sustaining fisheries or both (Margules and Pressey 2000; Rodrigues et al. 2004). Even with proper planning, poor management and inadequate enforcement fail to increase the diversity and abundance of fishes (Rogers and Beets 2002; Mora et al. 2006). This problem is especially prevalent in Asian countries, for example, China where there is a lack of data on species abundances and diversity within MRs (Mora et al. 2006). In addition, perturbations from human activities, such as land and coastal development, as well as the impacts from pollution and sedimentation should be taken into account in the design and management of MRs. While the above problems may be circumvented with proper design of MRs, damaged reef habitats may take decades to recover and consistent external sources of fish and coral larvae cannot be guaranteed over time and relied upon to help a reserve recover (Edgar et al. 2014). Improving management and enforcement on the other hand by limiting the damage in the first place is, in comparison, a relatively achievable way to improve marine reserves in a short period of time.

Marine reserves management which involves local shareholders or communities in the implementation and enforcement of the reserve has been successful in many countries. For instance, locally managed marine areas (LMMAs) improve livelihoods as well as conserve ecosystems, especially common in South Pacific Islands, such as Melanesia and Polynesia (Govan et al. 2009). Another example is community-based MPAs (CB-MPAs or community-based resource management) which were managed and developed by local government, local non-government organization or local neighbors to meet their requirements for either fisheries or tourism purposes (e.g., Johannes 2002). The CB-MPAs are commonly established in countries with more decentralized government countries, such as Philippines, Indonesia, and Kenya (e.g., Horigue et al. 2012). All these marine resources managed by local stakeholders and the community (we will use general term—locally managed MPAs later in this article) were much more successful and effective than MPAs managed by the central government (Claudet et al. 2006; Pollnac et al. 2010). By considering the socioeconomic needs of local communities and engaging them early in the planning process, locally managed MPAs can be designed to strike a balance between conservation and community development (Claudet and Guidetti 2010). Ideally, locally managed MPAs would lead to sustainable coral reef usage, resulting in higher fish abundance and coral cover, and subsequently higher fish catches for the local community (Claudet et al. 2006). Although several Locally managed MPAs have shown higher fish abundances than government reserves (Cinner et al. 2009), involving local stakeholders in reserve management is still difficult in many cases as conservation may not be part of their agenda (Jones 2007; Pita et al. 2011). In such cases, privately managed marine reserves (PMMRs) may be a viable alternative to protect the marine environment.

Initially, PMMRs or entrepreneurial MRs were used to describe areas that were privately owned or leased by hotels and the dive tourism business, with access to those areas being made available only to the paying customer (Colwell 1998). With an economic incentive to maintain the marine environment in pristine conditions, these PMMRs could effectively protect the marine environment from destructive fishing and damage from anchoring or divers (Colwell 1998). Meanwhile, an increasing number of terrestrial and marine areas have been purchased or leased to private owners with the assistance of NGOs, and have become successful private reserves (Groves et al. 2000). Irrespective of whether these PMMRs are driven by economic incentives or encouragement from NGOs, they have become successful in regions where government or local communities have no intention of protecting the natural resources or ecosystems. Successful case studies can be found in Malaysia (Teh et al. 2008), Vietnam (Svensson et al. 2009), Indonesia (Bottema and Bush 2012), and Honduras (Colwell 1998). Such PMMRs and some locally managed MPAs are not government managed, but have reported higher success than many government-managed reserves (Qiu et al. 2009). PMMRs would gain potential benefits from tourism by enhancing enforcement and compliance (Christie and White 2007), whereas locally managed MPAs do not necessarily receive their benefits due to voluntary local enforcement behaviors and have community priorities other than tourism (Cinner et al. 2009). Although some privatized marine resource might lead to diminished environments without well-trained or well-educated stakeholders (e.g., Cabral and Aliño 2011), PMMRs could still be a potential alternative management strategy in places that have limited government compliance and enforcement, such as China.

Coral reefs cover a mere 10 % of China's total coastline (excluding isles in the South China Sea), but play an important economic role as a source of fish, and lately, as a source of revenue through the growing tourism industry (Hutchings and Wu 1987; Fiege et al. 1994; Zhao et al. 2012). As in many countries, overexploitation of fisheries and a lack of alternative livelihoods have led to Malthusian overfishing in southern China (Pauly et al. 1989), where fishermen turn to destructive fishing methods in order to attain sufficient catches, further undermining the integrity of coral reef ecosystems (Hughes et al. 2012; Zhao et al. 2012). Since the 1990s, substantial numbers of national marine reserves have been established to protect the marine resources along China's coasts as well as the offshore coral reefs near southern China (Liu et al. 2003). Despite the establishment of 4000 ha of MRs and PMMRs in China since 1990, coral coverage has declined by 80 % in the past 20 years (Hughes et al. 2012). In Hainan Province, coral coverage has declined from 85 % to 10–20 % both inside and outside marine reserves due to a lack of efficient enforcement and management (Zhang et al. 2006). The emerging dive tourism industry presents an alternative strategy MR management (Ma et al. 2013). In China, areas of the coastline which include reef within national MRs can be leased to beachfront hotels and dive operators for their exclusive use. Tourism operators discourage fishing in their leased areas by cordoning off the reef with buoys and by chasing away encroaching fishing boats. This in essence creates PMMRs in the form of small privately owned marine areas managed and enforced by the tourism operators.

Irrespective of whether it is through PMMRs or locally managed MPAs, having healthy coral reefs with high biodiversity is the key to attracting tourism, and thus creating jobs for the local community and income for private enterprises through scuba diving tours, boat tours, hotels, and peripheral services such as restaurants and transport services (Fabinyi 2008; Svensson et al. 2009). However, tourism without adequate management can also bring damage to the reefs through direct damage or indirect pollution (Hawkins and Roberts 1993). For example, intensive numbers of inexperienced divers without buoyancy control can kick corals and physically damage them (Barker and Roberts 2004). Increased sewage discharge from high numbers of tourists on float pontoons has been linked to decreased coral cover and increased prevalence of coral disease (Lamb and Willis 2011). In addition, land or lease owners might adopt destructive fishing methods in privately managed areas permanently damaging reefs which lead to "Malthusian overfishing" from Pauly (1997), especially in poor countries of Southeast Asia (Béné et al. 2010; Pomeroy 2012). These small privately managed marine reserves might easily lead to political or socioeconomic marginalization of the community from the government and scientists (Teh et al. 2013). Therefore, the overall negative influence of the private tourism industry and PMMRs on coral reefs and reef fishes are not always known, especially in areas where tourism is intense (but see Albuquerque et al. 2014). More studies on PMMRs for the purpose of tourism are needed to estimate the overall net influence on corals and reef fishes (Reviews in Sciberras et al. 2013).

In this study, we aim to investigate whether PMMRs could be a successful alternative to national marine reserves for protecting coral reefs in China where rarely studied (Reviews in Sciberras et al. 2013). To achieve this, we compare sites with and without private management both inside and outside MRs. First, we quantify and compare the strength of enforcement between sites by examining (1) response time to fishing boat encroachment, (2) observed frequency of fishing activity (illegal and legal), and (3) observed tourism activities. Second, we compare fish and benthos abundance and diversity at those sites and relate that to the observed level of enforcement, to evaluate the effectiveness of protection effort. Last, to determine if MRs and PMMRs benefit fish communities over the long term, we analyze monitoring data from the past 10 years for fishes in the same region to identify whether populations within reserves have declined, remained steady, or recovered.

MATERIALS AND METHODS

Description of study area

Sanya Bay is located on the southernmost tip of tropical China and is next to the South China Sea, which had an extensive coral reef area with rich populations of coral reef fishes. However, as a consequence of rapid coastal development in Sanya City and high demand for seafood and marine souvenirs, the coral reefs in Sanya Bay have degraded considerably. In order to protect this delicate coral reef region, the Sanya Coral Reef National Nature Reserve which includes coastal and marine areas of about 8500 hectares was established in September 1990 by the

State Oceanic Administration (SOA, People's Republic of China) after consultation with marine experts. This Sanya marine reserve covered three separate regions: (1) Ximaozhou and the Dongmaozhou Islands region, (2) Luhuitou and the Xiaodonghai Dadonghai coastal region, and (3) Yalong Bay coastal region (Fig. 1). Each region is divided into zones: a central zone, buffer zone, and an experimental zone. All human activities are banned in the central zone, and in the experimental zone, tourism is allowed but fishing is banned. However, no clear or precise boundaries were documented and no management criteria, regulations, or policy relating to these zones was declared by the government or agencies until 2014 (www.sycoral.com.cn). Although Sanya marine reserve was supported by both the State Oceanic Administration, People's Republic of China, and the Hainan Provincial Oceanic and Fishery Department, which runs the Sanya Coral Reef National Nature Reserve, almost no enforcement or regular patrolling was conducted in this region.

Four privately managed reserves were selected in this study. They were managed by three different companies: Xidao Amusement Park, Yalong Bay Mangrove Tree Resort Sanya, and Park Hyatt Sanya Sunny Bay Resort. Xidao Amusement Park has been based in Ximaozhou since 1999 (Fig. 1, Site B), and has various tourist activities including snorkeling, scuba diving, banana boat rides, and other water sports. They also have a fishing platform and a seafood restaurant for tourists. They claim they have 3000 tourists per day on average. Xidao Amusement Park covers the most land and ocean area compared to the other privately managed reserves. Xidao Amusement Park, however, also has land-use conflicts with local villagers. Yalong Bay Mangrove Tree Resort Sanya based in Yalong Bay (Fig. 1, Site H) offers a restaurant, scuba diving, and a floating platform next to a reef for snorkeling access. The floating platform was built around 2000 and is only available to customers from the resort. The floating platform was not licensed by the provincial governments until recently, and has been available on and off sporadically in the past few decades. The Park Hyatt Sanya Sunny Bay Resort, which is based in Taiyangwan (Fig. 1, Site G) reserves an isolated area of Sunny Bay only for the customers of the resort. This company also has a lease within Baifuwan (Fig. 1, Site F), an area which they use in collaboration with a local diving shop. The lease length of each privately managed reserve should be around every

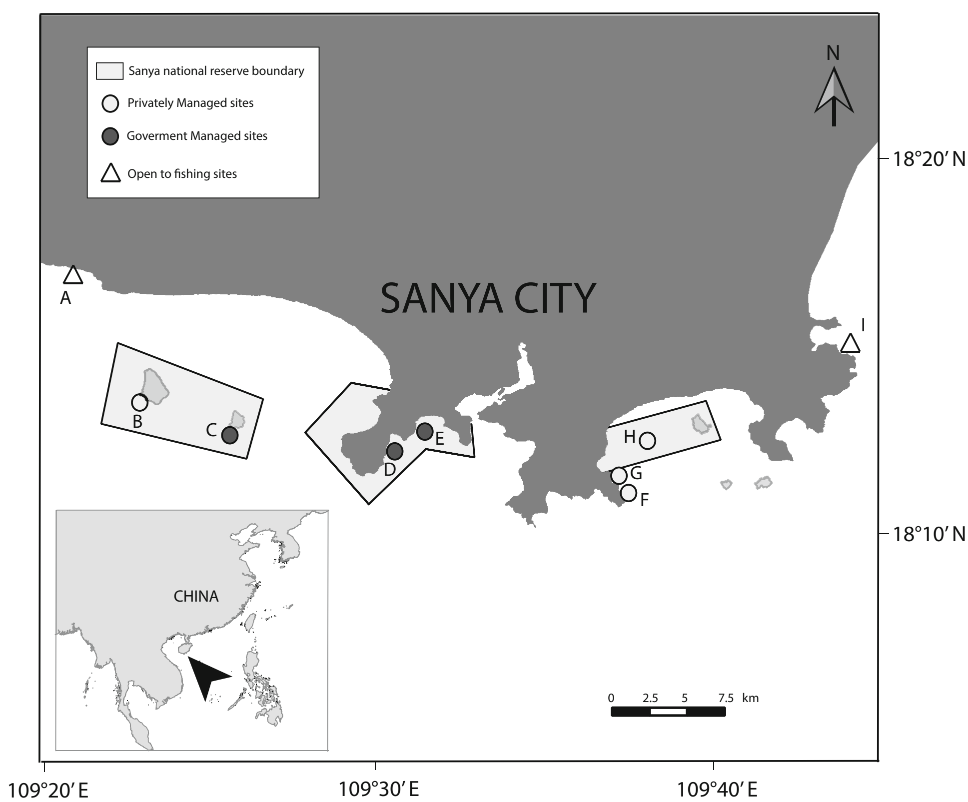

Fig. 1 Sampling sites for benthos, fishes, and fishing activity around Sanya City, Hainan, China. *Dark circles* indicate sites without local enforcement. *Light circles* indicate the sites with private management. Areas marked with a *line* indicate areas protected by local government. The sites were *A* Hongtang; *B* Ximaozhou (private tourism area); *C* Dongmaozhou; *D* Xiaodonghai; *E* Dadonghai; *F* Baifuwan (private resort); *G* Taiyangwan (private resort); *H* Yalong Bay (Xidao, private tourism area); and *I* outer Tielugang

5–10 years according to the companies, however, the exact information could not be confirmed with the provincial government.

Study design and data gathering

To examine the efficacy of MRs and PMMRs at Sanya Bay, Hainan Island, China, we employed a two-factorial 2 × 2 orthogonal sampling design (reserve and private-managed status; Sciberras et al. 2013) to compare (1) fishing activities, (2) tourism practices, and (3) fish and benthos communities (Fig. 1; 18°12′N, 109°30′E). In 2014, we surveyed nine sites that covered different combinations of reserve status and management types which were marine reserve with private management, marine reserve without private management, nonmarine reserve with private management, and nonmarine reserve without private management. Long-term data (2006–2013) on tourist numbers, fishes, and corals in the same sites from Huang et al. (unpublished data) and the Chinese government website (www.systats.gov.cn) were analyzed for trends between the tourism industry and the condition of the coral reef ecosystem over these years. The fish survey method was the same between this study and Huang et al. (unpublished data), and the coral data was standardized from both a point-count transect method and a recent CPCe method.

Measurement of illegal fishing and marine reserve enforcement

To quantify and understand the difference between marine reserves with and without private management, we recorded the response time (minutes) of the dive operator or national coastguard when we entered the reserves on a local traditional fishing boat. Response time was defined as the time between our entry into the reserve and the time the dive operator or coastguard began to approach us. Due to budget limitations, one visit for each site was recorded. Meanwhile, we also recorded the number of fishermen in the area to represent fishing activity at each site. Fishing activities inside national reserves (illegal) and outside national reserves (legal) were recorded by observations made from shore with binoculars.

Benthic and fish surveys

Most coral reef areas around Sanya are shallow (<9 m depth) and narrow (100 m from shore). Therefore, the reef was surveyed at two depths (2–3 m and 6–8 m) and at each depth, three-replicate 50-m transects were performed. The benthos was photographed 25 times along each transect using evenly spaced quadrats 50 cm × 50 cm in size. Each image was analyzed by CPCe V3.6 with 20 random points (Kohler and Gill 2006). Corals were identified to family level following English et al. (1994). The macroalgae data included macroalgae on the rock, dead corals, and all other substrata, except turf algae. Other benthic biota were identified to order level, and abiota were given general categories such as sand, rubble, and dead coral.

Fish assemblages were recorded using underwater visual censuses (UVCs) along the same transects as the benthic surveys. For each individual fish observed, we recorded the species and its body size class (<5 cm, 5–9 cm, 10–19 cm, 20–30 cm, and >30 cm total length, TL). Due to high turbidity, only fishes within a 2-m-wide belt transect were counted and some species were only identified to genus level for analysis.

Local fishery and tourism practices and pressures

The intensive tourist activities and negligent behaviors have caused a lot of damage to the reefs. To understand the impact of fishing and tourism on the local coral reef fishes and environment, we interviewed the local fishermen to understand the fish species they harvested and we conducted observations on tourism activities. For fishing activities, we recorded the species caught, numbers of fish harvested, fishing tools used, fishing location, and frequency. We also recorded whether fishermen used harvested fishes for their own consumption or sold them to local markets and tourists as well as fishing method. We compared data on harvested fishes with our underwater visual census data. For illegal fishing and tourism activities, we recorded the numbers of fisherman and tourism in each survey sites to represent the pressure of illegal fishing and tourism. We also visited the tourist center and tourist park in the government-managed sites (sites C & D) and privately managed sites (sites G & F) to observe and record tourist behaviors.

Statistical analyses

Fishing, tourism activities, and coral reef fishes among different managements

Individual numbers of fisherman and tourist among different survey sites were used to represent the fishing and tourist pressure. In a univariate analysis of count data, N-mixture models were used to examine differences in (1) fishing and tourism activity, (2) fish abundances and species richness, and (3) coral coverage between four treatments (MR with private management, MR without private management, nMR with private management, and nMR without private management). N-mixture models (Poisson, negative binomial, zero-inflated Poisson and zero-inflated negative binomial) were used due to the highly disturbed data with many zeros in this study (Joseph et al. 2009). This approach was taken as no prior assumption of homogeneity was necessary. Factors were examined in a nonlinear regression model framework using maximum likelihood to estimate parameters. A log-link function was chosen as data not normally distributed. The null model (no factor) and alternative models (combination of protection status and privately managed factors) were compared with Akaike's information criterion (AIC; Symonds and Moussalli 2011). AIC corrected (AICc) was used in this study due to the small number of replicates and many zeros in our data. The model with the greatest Akaike weight was selected for the best goodness of fit. N-mixture models and AIC analyses were conducted in R language (v. 3.1.1; R Core Team 2014).

Coral reef fish and benthos composition among different managements

A multivariate analysis using PERMANOVA (permutational multivariate analysis of variance; Anderson 2001) was used to test for differences in (1) fish assemblages and (2) coral coverage, among reserve status with/without private management. A two-factor design—reserve status (with and without private management) was selected in PERMANOVA. Homogeneity of the multivariate variance was verified for the two model terms using PERMDISP (permutational analysis of multivariate dispersions, $p > 0.05$; Anderson 2001). No assumption of normality is required for this nonparametric multivariate analysis (Anderson and Millar 2004). Fish abundance data $\log(x + 1)$ transformed and coral data were fourth-root transformed. The Bray–Curtis coefficient was chosen to construct the matrix. Type I (sequential) sums of squares and Monte Carlo randomization were used to meet the assumptions of PERMANOVA and generate a probability distribution to calculate a p value for each factor. A similarity percentage (SIMPER) analysis was used to evaluate the contributions of fishes and corals to the difference among areas with different reserve statuses. Patterns of variation in fishes and corals among areas with different reserve statuses were visualized using Principal Coordinates Analysis (PCO) to highlight the differences identified in the PERMANOVA results (PCO; Anderson and Willis 2003).

RESULTS

Coral reefs around Sanya city seemed to have varied levels of human disturbance. Underwater visibility is around 1 to 3 meters during our survey period. There was massive city development along the coast with many hotels and apartments being constructed in the past few years. The sedimentation from river and land run-offs were common through our survey, except the farthest site A and site I.

Fishing, tourism activities, and coral reef fishes among different managements

Fishing boats and fishermen were virtually absent in PMMRs, indicating effective enforcement (Fig. 2). In PMMRs, boats belonging to dive operators approached soon after we arrived with a traditional fishing boat (in 36 ± 35 SD minutes at sites with MR status and in 17 ± 15 SD minutes in nMR status). In the reefs with private management, almost all of our approaches in a traditional fishing boat were encountered by the local tourism operator in less than 120 min. No enforcement from either the coast guard or the government was observed in MR sites during our entire survey period (2 h). Fishing boats and fishermen were absent in all but one site under private management (Fig. 2a, b; Table 1). Some tourists (including recreational

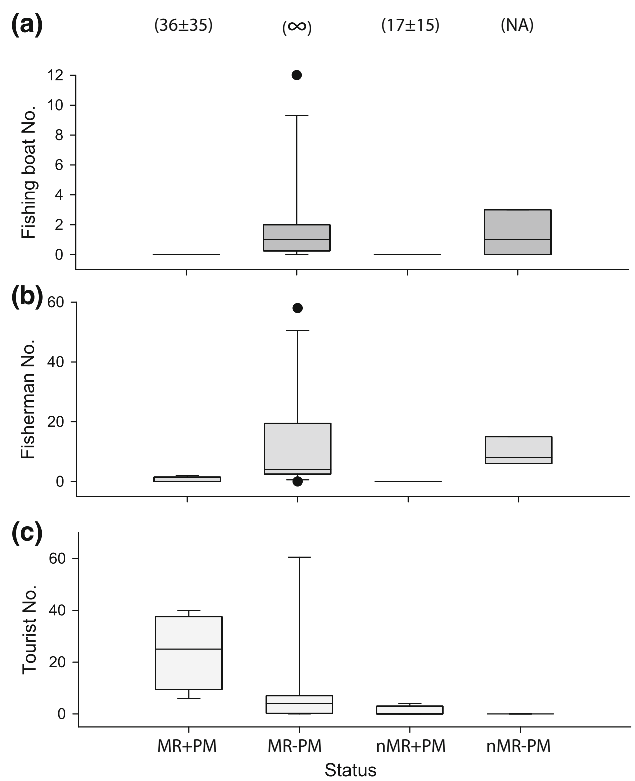

Fig. 2 Box plot of the fishing and tourism activities among four treatments. MR means marine reserves; PM means private management; symbol + or − means with or without PM. The *numbers* on the top of each category indicate the average response time of enforcement, either from government or local operators. Infinity symbol (∞) indicates no response was detected after the whole sampling period (approximately 2 h). NA indicates no response was expected due to no management being applied in nonmarine reserves without private management

Table 1 Results of N-mixture models for fishing/tourist pressure and coral reef fish assemblage among factors—marine status, private management (PM), and depth. The best goodness-of- fit formula was chosen according to the Akaike weight (highest)

Formula	Model	AICc	Akaike Weight
Fishing boat ~ PM	GLNB	63.72	0.783
Fisherman ~ PM	GLNB	122.00	0.761
Tourist ~ Status	GLNB	125.20	0.497
Fish species number ~ null	GLM	176.08	0.374
Fish abundance ~ Status × PM	GLM	1291.42	1
Fish largest size ~ PM	GLNB	789.15	0.527
Live coral ~ Status	GLNB	179.95	0.318
Macroalgae ~ null	GLNB	179.95	0.544

scuba divers) were observed in MRs regardless of private management status (Fig. 2c; Table 1). No tourists were observed in nMRs without private management status, even though the reefs are in a relatively good condition. Overall, all privately managed sites were enforced well, but local provincial government-managed sites were not. However, privately managed sites also had much more tourists than others.

Coral reef fish and benthos composition among different managements

In total, 26 fish families, 75 fish species, and 3573 individuals were recorded in nine sample sites in Sanya Bay. On average, across sites with different reserve statuses and private management statuses, we found 8.0(±2.8 SD) fish species and 68.7(±64.6 SD) individuals. Most fishes observed in this study were small, below 30-cm TL, with the exception of the elongated cornetfish (*Fistularia commersonii*). Large individuals were virtually absent. The number of coral reef fish species was similar regardless of MR status and PMMRs status (Fig. 3a; Table 1). On the other hand, fish abundances were higher in PMMRs (Fig. 3b; Table 1). Private management status showed a strong relationship with the largest fish in both MRs and nMRs (Fig. 3c; Table 1). This was consistent with the low fishing activity observed at these sites. The composition of coral reef fishes was significantly different between sites, with the differences being driven by private management status and depth, but not by MR status ($p < 0.01$; Table 2). The presence of larger fish species such as *Fistularia commersonii* and *Cheilinus chlorourus* in privately managed sites were the main drivers in the difference between sites with and without private management (Fig. 4). Sites without private management showed higher numbers of smaller fish species such as Apogonidae (*Ostorhinchus nanus* and *Archamia fucata*) and Blennidae (*Meiacanthus sp.*). A clownfish (*Amphiprion clarkii*) commonly collected for aquaria, was also more common in sites with private management. In contrast, coral and macroalgae cover did not show any correlation with either reserve status or private management status (Fig. 5; Table 1). The coral covers were much higher in privately managed marine reserve and nonmarine reserves without private management. The macroalgae showed the opposite trend to coral cover. Both

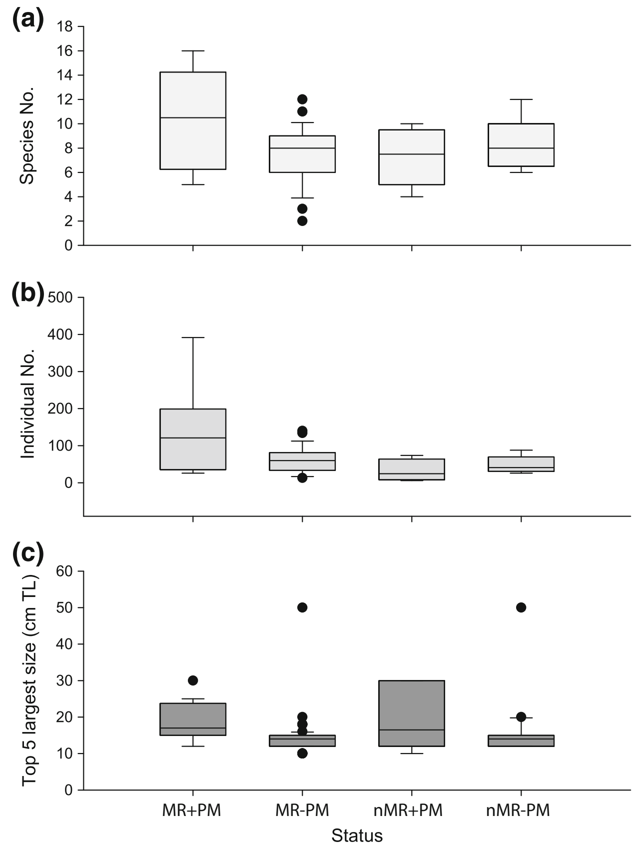

Fig. 3 Box plot of coral reef fish species numbers, abundances, (individual *numbers*) and top five largest individuals for the sampling sites. MR means marine reserves; PM means private management; symbol + or − means with or without PM

Table 2 Results of PERMANOVA on coral reef fishes with three-factor analysis

Source	df	SS	MS	Pseudo-F	P(perm)
Status	1	1092.3	1092.3	2.1358	0.071
Private management (PM)	1	1688.1	1688.1	3.3007	0.005
Depth	1	2379.3	2379.3	4.6523	0.002
Status × PM	1	1670.1	1670.1	3.2655	0.005
Status × depth	1	1352.6	1352.6	2.6448	0.027
PM × depth	1	1097	1097	2.145	0.056
Status × PM × Depth	1	1277.7	1277.7	2.4982	0.028
Residual	28	14 320	511.43		
Total	35	24 877			

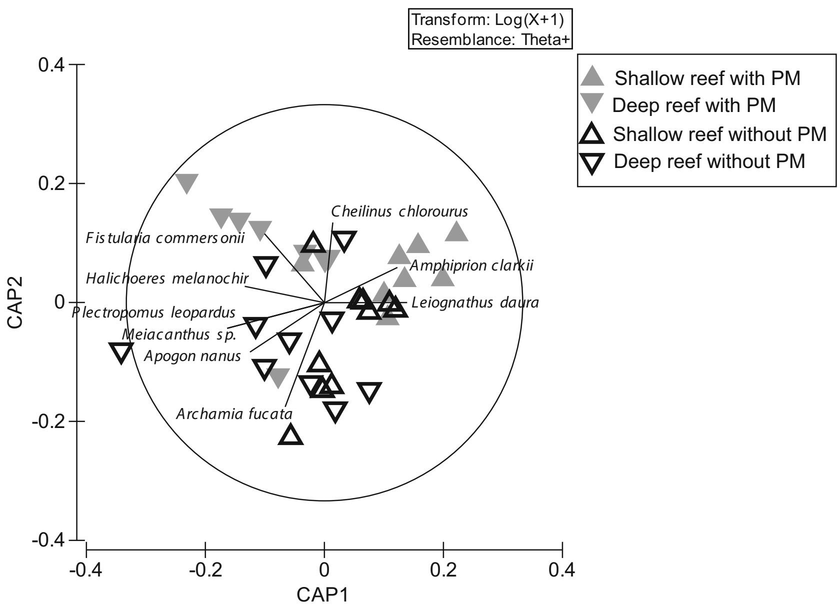

Fig. 4 CAP plot of major fish species that drove the differences among depths and sites

management status did not affect the coral and macroalage growth in this region.

Long-term tourist versus fishes

We compared coral reef fish data along with the tourist numbers in Sanya from the past 10 years (Fig. 6). Fish abundances and species numbers did not change much between 2005 and 2009. A clear recovery of both abundances and species numbers occurred after 2010, however, a dramatic decline was recorded between 2011 and 2014, with the exception of fish abundances at sites with private management, which remained constant. Meanwhile tourist numbers increased gradually between 2005 and 2009 (400 to 600 million), before doubling in the next 3 years from 2010 to 2012 (660 to 1100 million). The general peak season for tourism is Winter (October to January) due to large amounts of tourists from northern China avoiding the cold weather. Tourism revenue increased accordingly from US$13.8 billion to US$29.2 billion in the same time period. Local Chinese tourists accounted for the increased volume, while the number of foreign tourists remained constant (around 50 million) since 2007.

We also examined the recorded fish species from 2006, 2009, 2010, and this study in 2014 with 20 local fishermen surveys (Huang unpublished data; Table S1). We found that 64 of 136 species disappeared from 2006 to 2014. Seven of these species used to be relatively abundant (>1 per 20 m^2), but we did not observe them in our 2014 data. In addition, most catch of the local fishermen are small species, such as damselfish (Pomacentridae), wrasse (Labridae), and rabbitfish (Siganidae), and can only support the local fish market. These

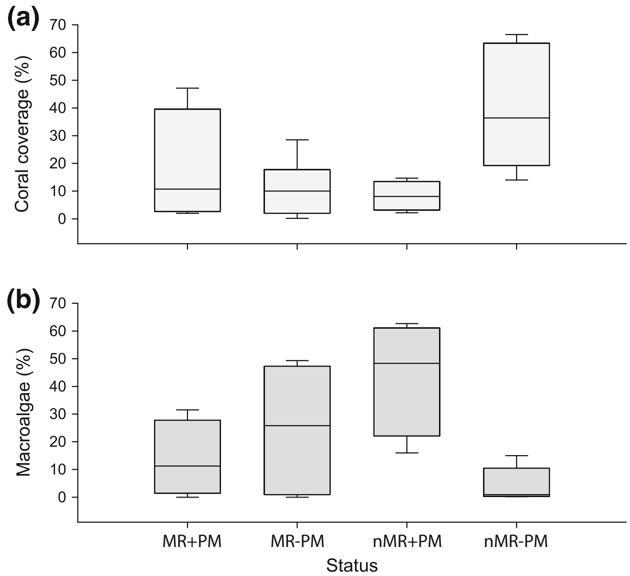

Fig. 5 Box plot of coral and macroalgae coverage on reefs around Sanya encompassing four different types of protection

absent species were *Chaetodon octofasciatus*, *C. lunulatus* (Chaetodontidae), *Diademichthys lineatus* (Gobiesocidae), *Halichoeres hortulanus* (Labridae), *Oxymonacanthus longirostris* (Monacanthidae), *Plectroglyphidodon dickii* (Pomacentridae), and *Sphyraena flavicauda* (Sphyraenidae). Most of them are coral-associated species, either corallivores such as Chaetodontidae and *O. longirostris* or coral inhabitants such as Labridae. In contrast, a number of species remained abundant across these years, such as *Pempheris oualensis* (Pempheridae), and many other damselfishes: *Abudefduf sexfasciatus*, *Dascyllus reticulatus*, *Pomacentrus coelestis*, *P. bankanensis*, and *Stegastes obreptus*. In addition, *Siganus fuscescens* (Siganidae) were common during all of the sampling years, and were even plentiful in fishermen's catches observed in 2014.

Tourist behaviors and consequence environmental damage

From our observations of tourism activities, we found several impacts from intensive tourism (Fig. 7). Debris from diving and snorkeling, especially littered mouthpieces were very commonly observed underwater at the diving sites (Fig. 7a). We also observed several tourists collecting live coral as souvenirs outside of marine reserves to take back to their families in their hometowns; they said they did not know if collecting live coral was illegal or permitted (Fig. 7b). In two privately managed tourist centers, tourists were offered the opportunity to take photos with many captive sea turtles that had been caught by fishermen. Tourists were also offered the option to release a sea turtle for approximately $150 USD, which was promoted as a way of achieving good karma for those who adopt Buddhist beliefs (Fig. 7c). In addition, these tourist centers were selling illegal harvested corals and marine creatures as souvenirs, including ornaments made from endangered species such as giant clam, hawksbill turtles, and green turtles. Many of them were marketed as locally supported souvenirs but were believed to be imported from southeast Asia (Fig. 7d).

Management of national marine reserves and privately managed areas

Our visits and discussions with several officers and experts in Sanya highlighted that many factors could cause a huge difference in enforcement between government-managed and privately managed reserves. First, local provincial government park rangers receive a lower salary (approximately $150 USD per month) compared to employees in privately managed reserves that promote enforcement (approximately $200-400 USD per month). No rewards or encouragement are given to government park rangers if illegal fishing is found, so this in combination with their lower salary means park rangers would prefer to stay and rest in the office instead of patrolling the MR to penalize illegal fishing activities. Secondly, government-managed reserve officers are only offered a limited budget for fuel for the enforcement boat. This would limit the enforcement frequency, especially when the areas are much larger than privately managed reserves. Lastly, local villagers said that most of the park rangers are locals too. Park rangers and local fishermen might have known each other for a long time, making rangers less willing to penalize the fishermen. This is another possible reason why enforcement is low in government-managed reserves.

DISCUSSION

For marine reserves to be effective, enforcement and compliance are critical (McCook et al. 2010; Arias 2015). Locally managed MPAs, such as LMMAs or CB-MPAs which involve local stakeholders in the planning process for the reserve and its subsequent management, have been reported to show higher levels of compliance than some government-managed marine reserves (Pollnac et al. 2010; Weeks et al. 2014). However, both government-managed marine reserves and locally managed MPAs in China have been shown to be widely ineffective due to insufficient government enforcement and lack of conservation awareness from local residents (Hughes et al. 2012). Our study suggests that PMMRs are a possible solution to preserve coral reefs suffering from overfishing in southern China and similar countries. The tourism industry's extensive

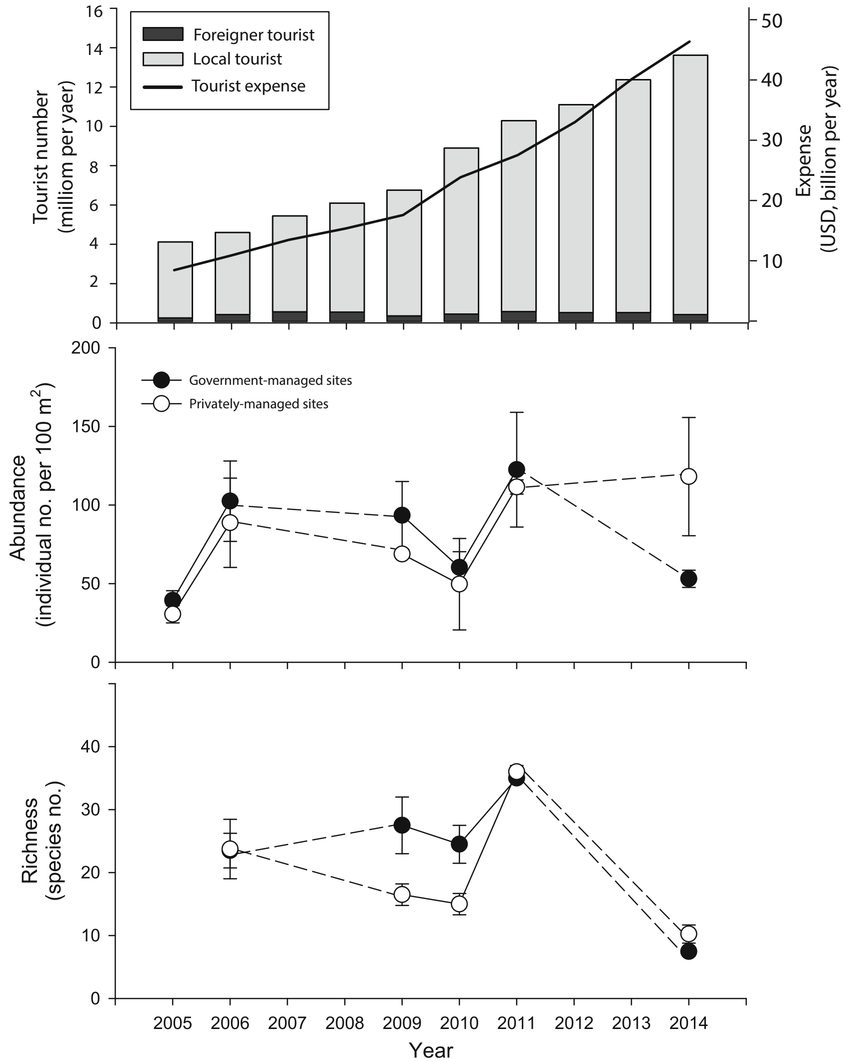

Fig. 6 Tourism trends (*number* and income), fish abundances, and fish species numbers in the Sanya region in the past 10 years (2005–2014). The tourism information was modified from a local government source (http://gov.sanyatour.com/tongji.asp). Government-managed sites in long-term data are from site C, D, E; privately managed sites are from site B, H

operations and daily presence on the reefs could provide constant, effective enforcement of marine reserves.

Coral reef fish and benthos composition among different managements

Fish abundances and species richness were not consistently higher in sites under private management. This might be a result of damaged coral reefs or natural variation between the coral reefs around Sanya. Sedimentation from coastal development around Sanya is a major human disturbance to coral reefs (Li et al. 2013). Sewage discharge and run-off from aquaculture farms leads to eutrophication, resulting in algal blooms on the reefs (Tytlyanov et al. 2014). Even with protection from fishing, such conditions present a challenge for reef fishes that can be difficult to recover from. Hence, beyond effective private management, a comprehensive land–ocean management plan is necessary to improve environmental conditions in order to preserve the coral reef ecosystem (Alvarez-Romero et al. 2011). We also found significantly higher abundances of clownfish (*A. clarkii*) in PMMRs than other sites. This might indicate a high catch rate of this popular marine aquarium species outside of management areas. A similarly high catch rate of clownfish was found on many coral reefs after the popular movie “Finding Nemo” (Jones et al. 2008). However, we also heard the SCUBA instructors speared fishes in their own operated tourism diving sites, the marginalizing fishing might also happen in these privately managed sites (Pauly 1997). In addition, comparing current fish species

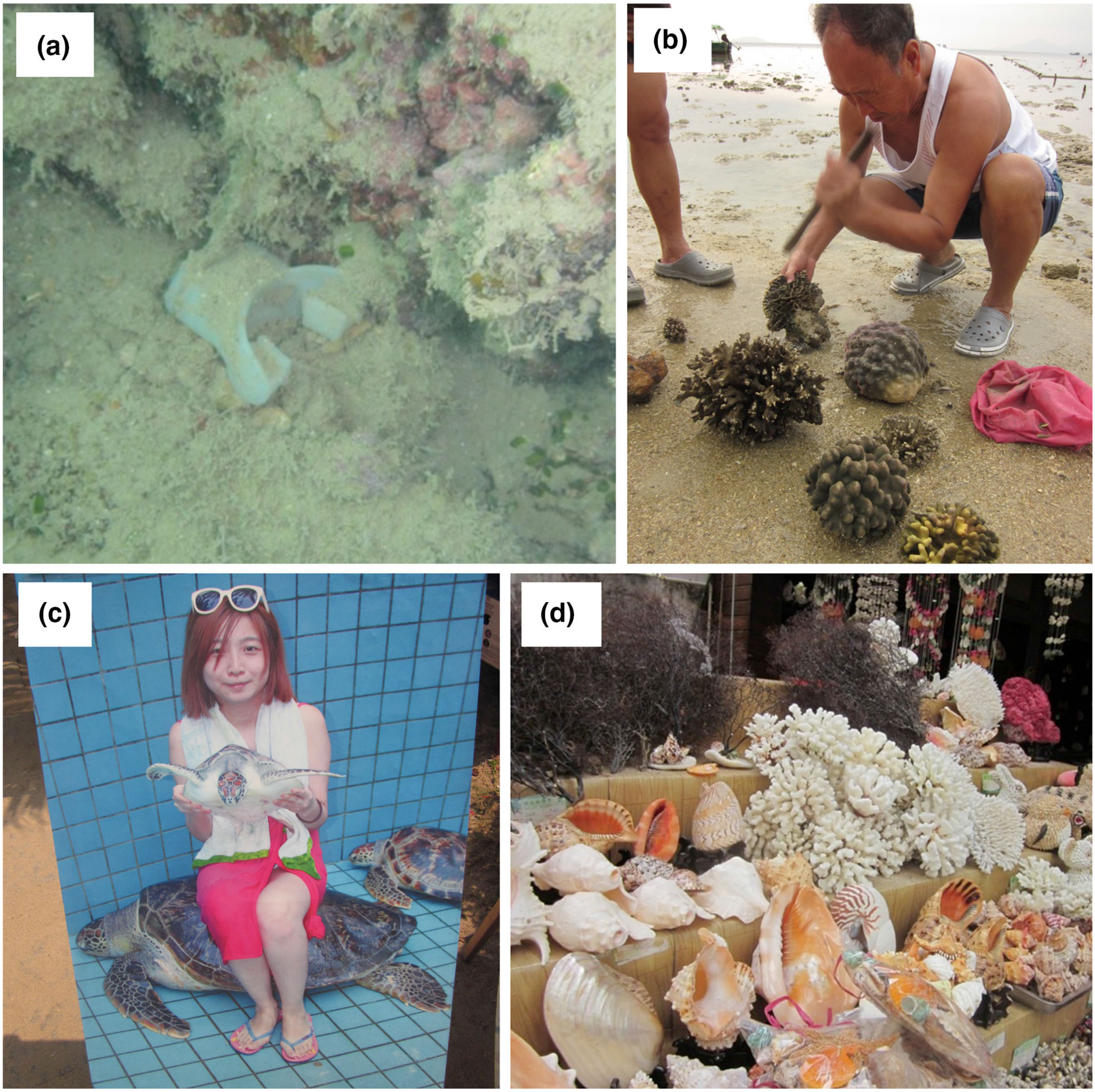

Fig. 7 **a** The littered mouthpieces were common underwater at the diving sites. **b** Tourists collected live corals outside of reserves. **c** Tourists can take a photo with a sea turtle. **d** Private shop sells illegal corals and marine creatures as souvenirs

with historical data from the past 10 years shows that many coral-associated species such as corallivores and coral inhabitants seem to have undergone local extinction. This local extinction of coral reef fishes might correlate with the huge coral loss (Wilson et al. 2008), especially *Acropora* coral which used to be abundant, but is now rarely seen in this region. Further evidence is required to draw concrete conclusions about this relationship.

Tourism pressure in privately management reserves

While PMMRs may protect the reef from fishing activity, such areas are under pressure from other human activities such as tourism. Tourists and scuba divers were observed in high densities at pontoons in the PMMRs. During our surveys, over 80 divers were recorded in 100 square-meters of reef at one time, and another about 200 snorkelers observed on a yacht, awaiting disembarkation for a snorkel tour. Marine debris such as snorkel mouthpieces and food packaging were frequently observed at these tourist hotspots (Fig. 7a). Although high visitation on tourism platforms has been linked to coral disease (Lamb and Willis 2011), this is not the case here as we did not observe any notable coral disease in our sampling sites. However, no comprehensive coral disease survey has been undertaken around the Sanya region yet. A future coral disease survey done by experts might find different results or relationships between coral disease and intensive tourism. We found that other human impacts, such as sedimentation, physical damage, marine debris, and overharvesting of corals were of greater concern in Sanya. Catastrophic physical damage was also observed from the collecting of Margilus coral snail (*Magilus antiquus*) from coral reefs for its shell. Enormous coral reefs around Sanya were destroyed by these Margilus coral snail collectors. Only limited studies have documented the impact of sedimentation as well as the consequent outbreak of benthic filter feeders in high turbidity water such as tube worms and sea cucumbers (Huang et al. 2009; Li et al. 2012; Huang et al. 2014; Zhang et al. 2014). Further studies about the influences of sedimentation, land-sourced pollutants, and large-scale stressor such as global warming and ocean acidification, as well as the impacts of human behavior on corals and other marine creatures around Sanya are needed to evaluate the damage to coral reefs from human activities in China.

Couple of PMMRs in our study were lacking in terms of environmentally sound practices. Selling souvenir made of shells and corals sends the wrong message and indirectly encourages tourists to illegally collect corals and shells from the reef and beaches. Many studies have also suggested to legislate the regulation of trading marine curio and souvenir to protect endangered species (e.g., Dias et al. 2011). However, many of the marine products from giant clams, hawksbill turtles, and green turtles in these PMMRs were already protected under Chinese regulation in Sanya. Hence, while we advocate PMMRs as an effective alternative to MRs in China, an environmental education programs are needed to be instituted for the PMMRS and MRs and fines be imposed on violations (Pasquini et al. 2011).

The potential for the tourism industry to be nonexploitative means that it can be a more economical and sustainable industry than commercial fishing over the long term. The tourism industry creates jobs through tour operators, hotels, restaurants, and transport services (Svensson et al. 2009). The recreational diving industry can also generate substantial revenue through the implementation of user fees (Depondt and Green 2006) and there has been a demonstrated willingness of divers to pay for entrance into protected areas (Arin and Kramer 2002). The long-term economic value of marine animals as a nonconsumptive resource through tourism is larger than if harvested. For example, in Palau, shark diving is responsible for \$2.7 million USD in annual revenue, as compared to \$10 800 USD if the sharks have been fished for consumption (Vianna et al. 2012).

CONCLUSIONS

Marine reserves have been broadly proven to increase the abundance and diversity of fishes, and improve the health of coral reefs when regulations are enforced. However, ensuring that enforcement occurs is still problematic without support from local stakeholders, especially when government management is insufficient. Our study indicates that the tourism industry could potentially enforce protection of the marine environment where the government has failed. However, we did not have any biological data before establishment of private management on these sites, and the concrete effectiveness of PMMRs needs further Before-After-Control-Intervention design study in the future. Nevertheless, without conservation awareness and regulations, the tourism industry could also exert other pressures on coral reefs. The government and NGOs have critical roles to play in establishing sustainable management protocols in order to achieve a balance between conservation objectives and profit for private operators. This study is the first to document the effectiveness of PMMRs in China. With rapid privatization of natural spaces in China and the demarcation of land and coastal areas for development, privately managed reserves could potentially become an important tool to conserve biodiversity both on land and in the water in the near future.

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