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Community-based conservation results in the recovery of reef fish spawning aggregations in the Coral Triangle

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ABSTRACT

In the Coral Triangle community-based marine protected areas (MPAs) are being established at a prolific rate. Their establishment can benefit both fisheries and biodiversity, and they provide both a socially and economically acceptable means of managing coral reefs in developing nations. However, because such MPAs are typically small (usually <0.5 km^2), they will rarely provide protection to large mobile fishes. An exception to this limitation may exist when community-based MPAs are established to protect small sites where vital processes occur, such as fish spawning aggregations (FSAs). To test the effectiveness of small (0.1–0.2 km^2) MPAs for protecting FSAs, we monitored three FSA sites where brown-marbled grouper (*Epinephelus fuscoguttatus*), camouflage grouper (*Epinephelus polyphekadion*) and squaretail coralgrouper (*Plectropomus areolatus*) aggregate to spawn. Sites were monitored during peak reproductive periods (several days prior to each new moon) between January 2005 and November 2009. All three sites are located in New Ireland Province, Papua New Guinea, and had been exploited for decades, but in 2004 two sites were protected by the establishment of community-based MPAs. The third site continued to be exploited. Over the monitoring period densities of *E. fuscoguttatus* and *E. polyphekadion* increased at both MPAs, but not at the site that remained open to fishing. At one MPA the densities of *E. polyphekadion* increased tenfold. Our findings demonstrate that community-based MPAs that are appropriately designed and adequately enforced can lead to the recovery of populations of vulnerable species that aggregate to spawn.

1. Introduction

In coral reef ecosystems community-based marine protected areas (MPAs) have proliferated in recent decades (e.g. Johannes, 2002; Alcala and Russ, 2006; Govan, 2009). Such efforts reflect the widespread understanding that in many developing nations, effective management of coral reefs is only possible with buy in and support from local communities (e.g. Alcala and Russ, 2006; Pollnac et al., 2010). Concurrent with the interest in community-based MPAs has been a growing awareness of the need to better understand the socio-political and economic factors that lead to their successful implementation (e.g. Waylen et al., 2010).

Community-based MPAs can benefit both fisheries management and biodiversity conservation (Alcala and Russ, 2006; Cinner et al., 2006), but because they are typically small (usually <0.5 km^2), even large networks of community-based MPAs may be inadequate for conserving biodiversity and ecosystems at regionally meaningful scales (Weeks et al., 2010; Mills et al., 2010). Furthermore, while empirical studies have shown that small community-based MPAs can result in greater fish density and biomass within and outside of MPA boundaries (Russ and Alcala, 2004), there remain few empirical examples of small community-based MPAs leading to recovery of large vulnerable reef fishes. This is perhaps not surprising since in most cases small MPAs will not provide adequate protection for larger-bodied mobile species (Sale et al., 2005). In the few instances where small community-based MPAs have been shown to result in the recovery of large fish species, recovery has occurred after decades of protection (e.g. Russ and Alcala, 2010).

An exception to this limitation may exist when small community-based MPAs are specifically established to protect critical processes or life stages of vulnerable species that are confined to small areas, such as fish spawning aggregations (FSAs) (Domeier and Colin, 1997). Similar to turtle nesting beaches, the locations where FSAs form are often site-specific, occur over a limited spatial scale and have a predictable temporal component, characteristics that make them well known to local fishers and extremely vulnerable

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to overfishing (Sadovy and Domeier, 2005). Such characteristics also imply that under certain scenarios small community-based MPAs may effectively protect FSAs.

In the Indo-Pacific three species of grouper, the brown-marbled grouper (*Epinephelus fuscoguttatus*), camouflage grouper (*Epinephelus polyphekadion*) and squaretail coralgrouper (*Plectropomus areolatus*) frequently co-aggregate to spawn at predictable sites and times (Johannes et al., 1999; Rhodes and Sadovy, 2002; Pet et al., 2005). These groupers make up important components of many artisanal and small-scale commercial fisheries in the Pacific (Wright and Richards, 1985; Rhodes and Tupper, 2007) and they are three of the most economically valuable species in the Southeast Asia-based Live Reef Food Fish trade (LRFFT) (Sadovy, 2005). All are susceptible to overexploitation, with *E. fuscoguttatus* and *E. polyphekadion* listed as Near Threatened (Cornish, 2004; Russell et al., 2006, respectively) and *P. areolatus* listed as Vulnerable (Thierry et al., 2008) on the 2008 IUCN Red List.

In Papua New Guinea, the most widespread threat to FSAs of these grouper is artisanal nighttime spearfishing, with free divers targeting resting groupers on the lunar days when aggregation numbers are known to peak (Hamilton et al., 2005). The LRFFT also poses a major threat, and where it has occurred it has systematically sought out and targeted FSAs, resulting in their very rapid demise (Hamilton et al., 2005; Hamilton and Matawai, 2006), a scenario that mirrors global trends (Sadovy, 2005; Scales et al., 2007).

In this paper the results of 5 years of monthly underwater monitoring data from three FSA sites in New Ireland Province, Papua New Guinea are presented. Baseline information on species composition, abundance, seasonality and status of FSAs in New Ireland were documented through local knowledge surveys in 2004, with much of this information subsequently independently verified through underwater visual census (UVC) surveys (Hamilton et al., 2004). Two FSA sites were heavily fished for subsistence, artisanal and commercial (LRFFT) purposes until 2004. Subsequently the communities that claimed traditional ownership of these two FSA sites established small no-take MPAs (0.1 km^2 and 0.2 km^2) to protect the spawning biomass of the aggregating groupers. A third FSA site, which had no form of management in place and had been heavily exploited since the 1980s (Hamilton et al., 2004), was open to continued exploitation from 2004 and served as a comparison. The aims of this paper are to (1) explore whether small community-based MPAs provide effective management of FSAs, (2) document temporal trends and spatial variability among FSAs, and (3) determine if simple monitoring protocols provide sufficient information to inform site-based conservation efforts and broader fisheries management strategies.

2. Methods

2.1. Social and environmental setting

The current study was conducted in the northern part of New Ireland Province, Papua New Guinea (PNG) (2°40′S, 150°40′E) (Fig. 1). New Ireland is located in the Bismarck Sea, which forms part of the eastern portion of the Coral Triangle (Veron et al., 2009). Although the reefs around New Ireland support very high biodiversity, they have been detrimentally affected by crown-of-thorn starfish (*Acanthaster planci*) outbreaks and coral bleaching events, and some highly valued marine resources such as sea cucumbers and groupers have been overfished (Hamilton et al., 2009). In New Ireland as throughout Papua New Guinea, the right to use land and marine resources are held by clans and tribes (Otto, 1998), which can provide a framework for community-based conservation.

2.2. Identification of FSAs considered suitable for community based conservation

A 2004 local knowledge survey that was conducted by The Nature Conservancy (TNC) identified 18 potential grouper FSAs in northern New Ireland (Hamilton et al., 2004). During the local knowledge surveys many communities expressed an interest in partnering with TNC to gain technical assistance with managing their FSAs. Given limited resources, two FSA sites were selected for long-term conservation efforts, based on the following ecological and socio-political criteria (*see* Aswani and Hamilton, 2004a and McClanahan et al., 2006 for further discussion on socio-political environments that can facilitate successful community-based conservation efforts); (1) FSA must be independently verified to support aggregations of *E. fuscoguttatus*, *E. polyphekadion* and *P. areolatus*; (2) the community with traditional ownership of the FSA must be interested in conserving the aggregation; (3) traditional ownership of the reef on which FSA occur must be reasonably well defined and undisputed; (4) the community that claims ownership of a FSA believes that community-based management is a viable option; and, (5) FSA must be located within close proximity to the community that wishes to conserve it to facilitate enforcement of management.

Eight of the 18 FSAs identified through local knowledge surveys were reported to be multi-species spawning sites and six of these sites were surveyed on SCUBA between January and July 2004 during the lunar stages when FSAs were reported to form (Hamilton et al., 2004). Out of the six sites surveyed on SCUBA four were independently verified to support spawning aggregations of *E. fuscoguttatus*, *E. polyphekadion* and *P. areolatus*. Concurrent with FSA surveys, ethnographic information on communities' attitudes towards conservation and customary ownership of FSAs were collected. Two of these four sites (Site 1 and Site 10) also met all of the other criteria outlined above, and on this basis, the communities that claim ownership of these two sites were further engaged.

In July 2004, discussions were held with the two communities that have traditional ownership of Site 1 and Site 10. In these discussions communities were informed of the importance and suitability of these sites for community-based management. Both communities were offered technical assistance with site monitoring and a no-take MPA was established around each respective FSA by consensus of each community. The MPAs ban all forms of fishing at all times of the year. These community-based MPAs are enforced through social pressure and the closures are generally well respected, although low levels of poaching by night spearfishers is known to have occurred at both MPAs between 2005 and 2009.

2.3. Site descriptions

Site 1 (MPA) and 2 (no management) are located 10 km apart at Dyual Island and Site 10 (MPA) is located within the Tigak Islands approximately 35 km from Sites 1 and 2 (Fig. 1). The three sites are all located on seaward facing reef promontories. The MPA at Site 1 is 0.2 km^2 and the MPA at Site 10 is 0.1 km^2. In order to protect these sites from further exploitation by outside entities, exact location data are not provided.

Site 1 has been fished for subsistence purposes for generations, with artisanal nighttime spearfishing regularly occurring at the site since the 1980s. Between 1997 and 2003, LRFFT operations used traps, cyanide and hook and line to capture groupers at this site, with local fishers reporting that LRRFT operations resulted in major reductions in the abundances of aggregating groupers, and the death of a local fisherman who was setting traps using surface-supplied air (e.g., hookah) (Hamilton et al., 2004). Aggregations of all three species overlap over a linear distance of approximately 400 m and between depths of 4–40 m except *P. areolatus*, which is

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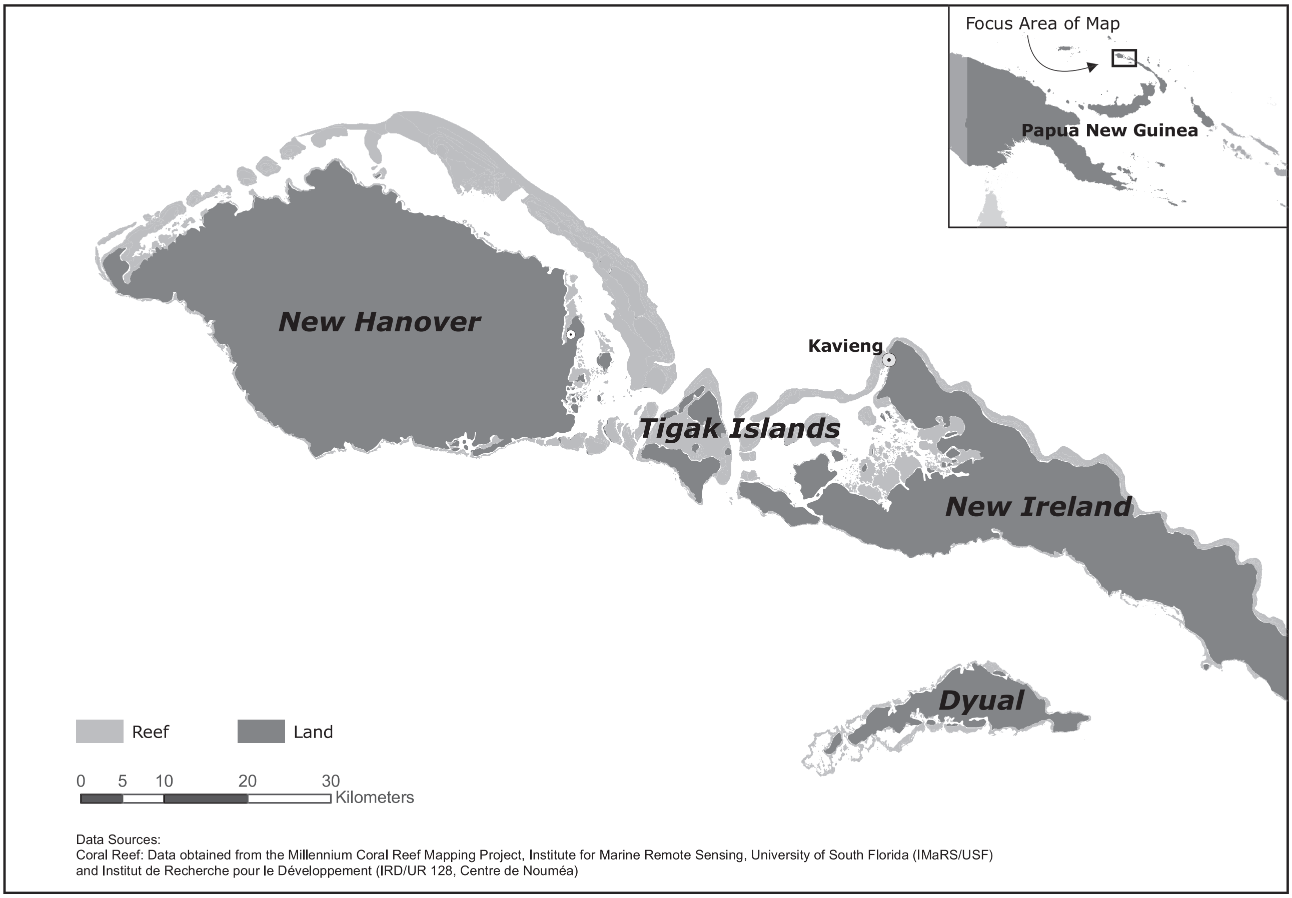

Data Sources:
Coral Reef: Data obtained from the Millennium Coral Reef Mapping Project, Institute for Marine Remote Sensing, University of South Florida (IMaRS/USF) and Institut de Recherche pour le Développement (IRD/UR 128, Centre de Nouméa)

Fig. 1. Northern New Ireland. New Hanover, Tigak Islands and Dyual Island are shown, as is the location of the provincial centre, Kavieng.

rarely sighted below 20 m. Highest densities (hereafter, 'core' area) of all three species occur around the promontory, with lower densities within a small bay that lies west and directly adjacent to the promontory.

Site 2 was historically the largest known *P. areolatus* aggregation in New Ireland. Aggregations at Site 2 occur over *ca.* 600 m of reef horizontally, with *P. areolatus* at 3–50 m depth and highest densities occurring at 3–20 m. *E. fuscoguttatus* and *E. polyphekadion* aggregate over a 200 m stretch around the promontory, with highest densities between 10 and 30 m. Artisanal fishing began at Site 2 in the 1980s when nighttime spearfishers reported maximum individual catch rates of *P. areolatus* were 50 fish h^{-1}. By the 1990s, maximum individual catch rates had dropped to *ca.* 20 fish h^{-1} (Hamilton et al., 2004).

The Site 10 location has been known locally for generations and it was frequently targeted by artisanal nighttime spearfishers prior to mid-2004. It was also fished to supply the LRFFT in 2001–2002. Patriarchal fishers report that both the number of aggregating fish and the FSA area has decreased markedly since the 1960s as a response (Hamilton et al., 2004). FSAs at Site 10 occur along *ca.* 200 m of reef horizontally and at depths of 2–40 m. *P. areolatus* aggregate at 2–20 m and west of the reef promontory. The core *E. fuscoguttatus* aggregation occurs proximate to the reef promontory, with lower numbers to the west where *P. areolatus* aggregates. *E. polyphekadion* is present throughout the site.

2.4. Monitoring

To determine densities at Sites 1 and 10, permanent transects were established between September and December 2004. At Site 1, three transects were established: (1) Transect B—100 m by 10 m wide (1000 m^2), 30 m deep, placed within the high density 'core' *E. fuscoguttatus* and *E. polyphekadion* stratum (2) Transect C—100 m by 10 m wide (1000 m^2), 15 m deep, within the bay, a low density 'non-core' stratum for all three species, (3) Transect D—50 m by 10 m wide (500 m^2), 5 m deep, placed within the core *P. areolatus* stratum, which is located on the reef flat above transect B. The recovery of the *E. fuscoguttatus* and *E. polyphekadion* FSAs at Site 1 resulted in transect D also sampling the shallow core stratum of *E. fuscoguttatus* and *E. polyphekadion* from 2006 onwards. At Site 10 two transects were established: (1) Transect A—100 m by 10 m wide (1000 m^2), 10 m deep, placed within the core *P. areolatus* stratum and the non-core *E. fuscoguttatus* stratum, and (2) Transect A1–50 m by 10 m wide (500 m^2), 30 m deep, placed within the core *E. fuscoguttatus* stratum.

Transects were used to sub-sample the FSA at Sites 1 and 10 as preliminary UVC surveys of these sites demonstrated that total counts would be confounded by the depth ranges of the three target species and fleeing behaviour of *E. fuscoguttatus*. Density estimates obtained from permanent transects can provide a useful measure of temporal trends at FSAs, and transect counts can be extrapolated to represent total abundances when FSA area is known (Pet et al., 2006). At Site 1 and 10 transects covered 18% and 14% of the total FSA areas respectively, exceeding the 10% minimum area that is recommended in order to ensure sufficient precision (Pet et al., 2006). At Site 2 surveys were conducted less frequently and no permanent transects were established due to sensitivities regarding customary ownership of this site. Instead, two divers swam the length of the entire aggregation (600 m) at a depth of 15 m and counted all *E. fuscoguttatus*, *E. polyphekadion*

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and *P. areolatus* sighted in a 10 m wide belt. This represented approximately 17% of the total FSA area at Site 2.

Between January 2005 and November 2009 monitoring of each transect was conducted on a monthly basis prior to each new moon, except when unfavourable weather or logistical problems occurred. Monitoring was conducted 2 days prior to the new moon at Site 10 and 1 day prior to the new moon at Sites 1 and 2. One of the authors (TP) monitored consistently throughout the 5-year duration of this study, accompanied by a trained community member and or a provincial fisheries officer. During monitoring, divers swam side-by-side, maintaining a position several metres above the aggregated fish. Monitors recorded the total number of each of the three species sighted within transect boundaries. Monitoring times conformed to information on peak aggregation times from local knowledge and UVC surveys conducted prior to the initiation of January 2005 surveys. Consequently, it was expected that sampling during this period would provide a 'snapshot' of the peak aggregation density. To verify that monitoring conformed to periods of peak density, intensive (daily) UVC surveys were conducted over a 5-day period at Site 1 in May and July 2009.

In addition to density estimates, at Site 1 and 10 monitors determined FSA areas for each target species annually between 2005 and 2009. Aggregation area was estimated each May at Site 1, and each July at Site 10. Aggregation areas were established by placing permanent markers at the aggregation boundaries where densities declined rapidly and neared non-reproductive values. Float lines were then attached to each marker and sent to the surface to mark aggregation boundaries with a handheld GPS (Rhodes and Sadovy, 2002) and entered into ArcGIS 9.3 (ESRI, Redlands, California, USA) for area estimation and mapping. Further refinement of this method involved calculating high density (core) and low density (non-core) stratum for each aggregating species. High-resolution Quickbird satellite imagery was used to help digitize FSA boundaries. Natural (i.e., non-aggregation) densities were calculated from 90, 500 m^2 transects surveyed outside FSAs at

Table 1
Mean densities per 1000 m^2 ± 1SE for *Epinephelus fuscoguttatus*, *Epinephelus polyphekadion* and *Plectropomus areolatus* at FSAs and outside of FSAs (data from all three FSAs combined).

Species	Mean density at FSA (n = 114)	Mean density outside of FSA (n = 90).	p
E. fuscoguttatus	3.04 ± 0.5	0.07 ± 0.04	<0.001
E. polyphekadion	2.43 ± 0.48	0.02 ± 0.02	<0.001
P. areolatus	7.34 ± 0.81	0.2 ± 0.1	<0.001

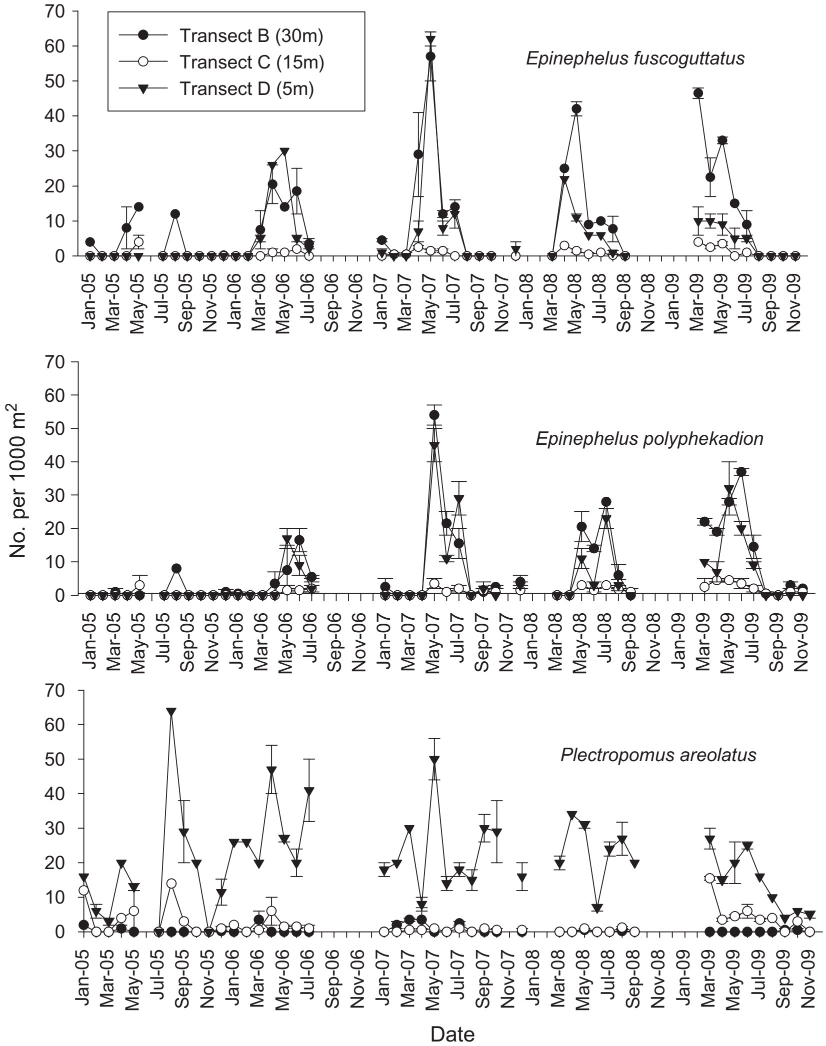

Fig. 2. Mean densities (±1SE) of *Epinephelus fuscoguttatus*, *Epinephelus polyphekadion* and *Plectropomus areolatus* on three transects at Site 1 (MPA) 1 day prior to the new moon between January 2005 and November 2009.

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10 m depths around northern New Ireland in 2006 at non-aggregation times (Hamilton et al., 2009).

2.5. Statistical analysis

Mean densities of *E. fuscoguttatus*, *E. polyphekadion* and *P. areolatus* observed within FSAs and outside of FSAs were compared using the Mann–Whitney rank sum test in SigmaStat (Systat Software, San Jose, California, USA) as data were nonparametric. We also tested linear trends in the estimated average annual density over 5 years, between 2005 and 2009, for *E. fuscoguttatus, E. polyphekadion and P. areolatus* at Sites 1, 10 and 2 using linear regression. Data were subjected to the Shapiro–Wilk test for normality and residual plots were examined to ensure that data conformed to appropriate assumptions. The linear regressions were carried out in JMP 8.0.1 (SAS Institute, Cary, North Carolina, USA).

3. Results

3.1. Long-term trends in population densities

The mean densities of *E. fuscoguttatus, E. polyphekadion* and *P. areolatus* at FSAs were significantly ($P < 0.001$) higher than the mean non-aggregated densities (Table 1). Mean density estimates taken from the 5-year survey show inter-species and inter-location variability (Figs. 2–4). Linear regressions showed significantly positive trends for *E. fuscoguttatus* (Adj. R^2 0.70, $P = 0.049$) and *E. polyphekadion* (Adj. R^2 0.90, $P = 0.008$) at Site 1. Site 10 exhibited a similar significant positive trend only for *E. polyphekadion* (Adj. R^2 0.95, $P = 0.003$). The abundance of *P. areolatus* did not exhibit a significant trend using $\alpha = 0.05$ at any site. None of the three species demonstrated a significant trend using $\alpha = 0.05$ at Site 2.

3.2. Inter-annual and lunar seasonality

There was substantial variability in the inter-annual seasonality and duration of spawning aggregations among sites and species. At Site 1, *E. fuscoguttatus* aggregated 4 or 5 months annually between March and July (Fig. 2), whereas 35 km away at Site 10 aggregations formed 3–4 months later, with highest densities from July to November (Fig. 3). Similarly, the reproductive season for *E. polyphekadion* at Site 1 varied inter-annually, occurring over 3 months in 2006–2008, and 5 months in 2009 between May and July. The highest observed densities for *E. polyphekadion* were recorded at Site 10 between July and October (Fig. 3). At all three sites *P. areolatus* formed aggregations of variable size throughout the year, however, at Sites 1 and 10 maximum *P. areolatus* densities coincided with periods when *E. fuscoguttatus* and *E. polyphekadion*

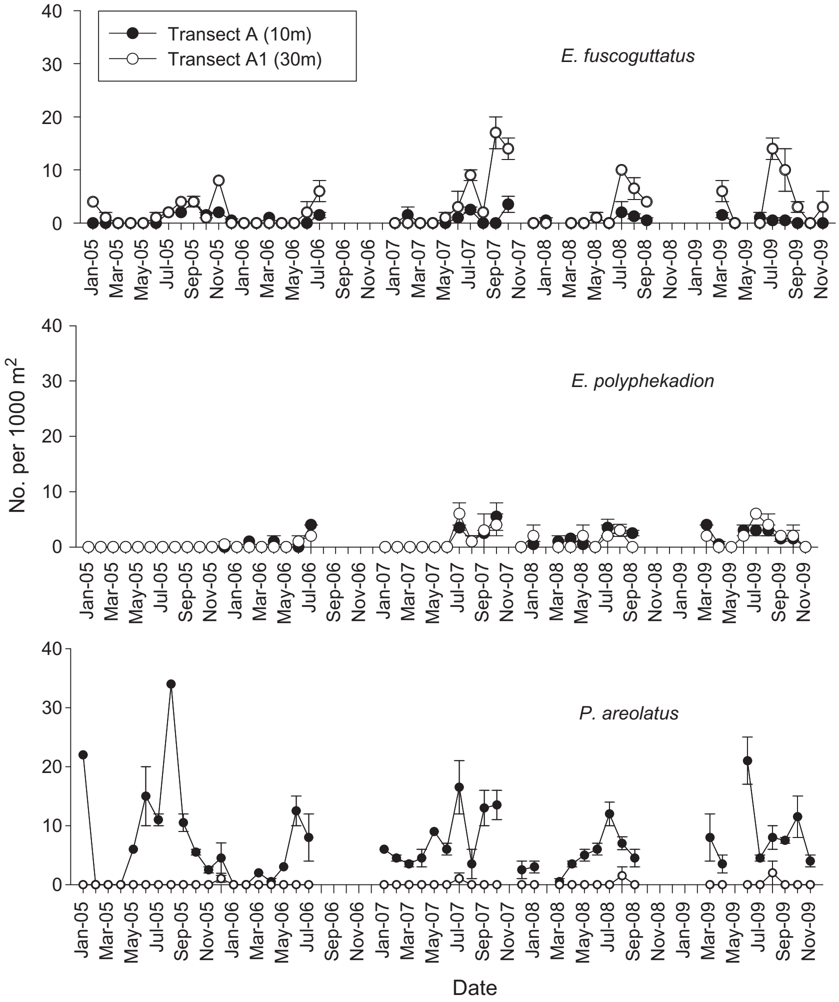

Fig. 3. Mean densities (±1SE) of *Epinephelus fuscoguttatus*, *Epinephelus polyphekadion* and *Plectropomus areolatus* on two transects at Site 10 (MPA) 2 days prior to the new moon between January 2005 and November 2009.

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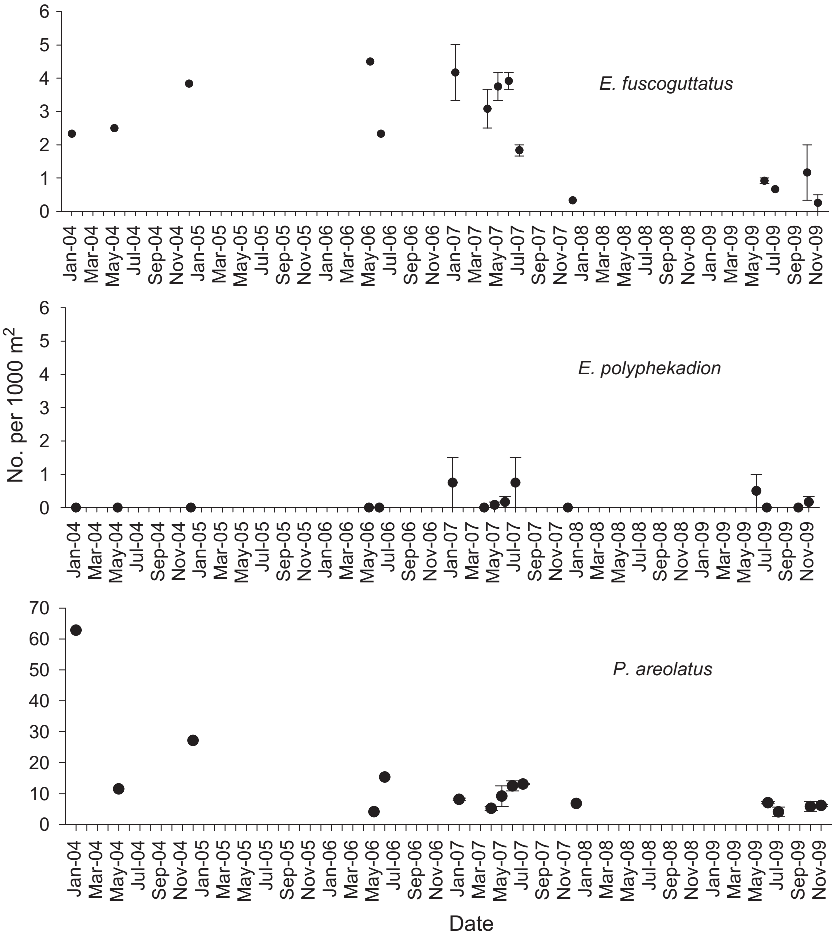

Fig. 4. Mean densities (±1SE) of *Epinephelus fuscoguttatus*, *Epinephelus polyphekadion* and *Plectropomus areolatus* during long swims covering the entire aggregation at 15 m depth at Site 2 (unprotected) 1 day prior to the new moon between January 2004 and November 2009. Note that *y*-axis scale varies between plots.

spawning aggregations formed. At Site 2, no clear seasonal patterns were discernable for *E. fuscoguttatus* and *E. polyphekadion* due to limited data (Fig. 4). Intensive surveys that were conducted at Site 1 in 2009 (Fig. 5) support local knowledge that FSAs of *E. fuscoguttatus*, *E. polyphekadion* and *P. areolatus* in New Ireland formed prior to the new moon and dispersed after presumed spawning at or around new moon (Hamilton et al., 2004).

3.3. Aggregation areas and total population size estimates

At Site 1 the area of the *E. fuscoguttatus* and *E. polyphekadion* FSA was 10,424 m^2 (core = 3184 m^2) in 2005, but expanded to 14,142 m^2 (core = 6902 m^2) during 2006–2009. This increase resulted from both species moving into shallower water around the promontory from 2006 onwards. The area of the *P. areolatus* aggregation at Site 1 was 10, 958 m^2 (core = 3718 m^2) in all years. At Site 10, the aggregation areas remained unchanged in all years, with *E. fuscoguttatus* occupying 5929 m^2 (core = 2829 m^2) and *P. areolatus* 3100 m^2 (core = 3100 m^2). *E. polyphekadion* was located throughout the site with an aggregation area of 5929 m^2. Estimates of the total number of *E. fuscoguttatus*, *E. polyphekadion* and *P. areolatus* present at Site 1 in May 2005–2009 and at Site 10 in July 2005–2009 are presented in Table 2. These estimates were calculated by extrapolating mean transects counts for each stratum using the formula: No. fish in transect(s) × total aggregation area/transect(s) area. Total estimates are the sum of the estimated abundances in each stratum (Nemeth, 2005).

4. Discussion

4.1. Effectiveness of community-based MPAs

This study presents evidence that community-based protection of FSAs using small-scale MPAs can improve reproductive populations of some large vulnerable species. At Site 1, *E. fuscoguttatus* and *E. polyphekadion* were present in low densities during preliminary surveys in 2004 (Hamilton et al., 2004) and in the first year of monitoring (2005). However between May 2005 and 2009, after the implementation of a community-based MPA, densities increased significantly. The core aggregation area of *E. fuscoguttatus* and *E. polyphekadion* also doubled over this time period, as individuals expanded their use of habitat by utilizing shallow water reefs around the promontory, which were likely areas of high fishing pressure and vulnerability prior to the area closure.

Similar to Site 1, at Site 10 densities of *E. polyphekadion* increased significantly after the implementation of a community-based MPA. Peak monthly densities of *E. fuscoguttatus* tripled after MPA establishment, although these trends were not statistically significant. *P. areolatus* densities did not change at either Sites 1 or 10 during the sampling period. At Site 2, which remained open to fishing, *E. fuscoguttatus*, *E. polyphekadion* and *P. areolatus* densities did not change.

If we assume a high degree of larval connectivity between the three FSAs, then Site 2 provides a baseline comparison which suggests that improvements in Sites 1 and 10 were due to the fisheries

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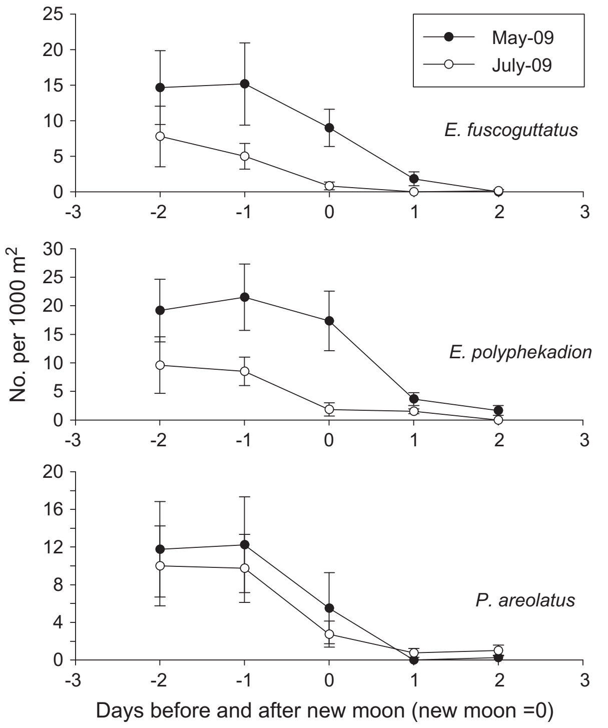

Fig. 5. Mean densities (±1SE) of *Epinephelus fuscoguttatus* and *Epinephelus polyphekadion* (Transect B) and *Plectropomus areolatus* (Transect D) at Site 1 (MPA) before, during and after new moon in May and July 2009.

closures that began in 2005 rather than natural variation in *E. fuscoguttatus* and *E. polyphekadion* abundances. This assumption appears warranted given the close proximity of the FSAs and the high degree of larval connectivity between reef fish populations (Jones et al., 2009). Our findings demonstrate that appropriately designed and enforced community-based MPAs can have a significant positive effect on the populations of vulnerable aggregating species, and lend further support to the handful of other empirical studies that have demonstrated FSAs can recover following protection (e.g. Beets and Friedlander, 1999; Nemeth, 2005).

The lack of recovery observed in *P. areolatus* at Site 1 and 10 is most likely due to poaching by nighttime spearfishers. Between 1 and 4 poaching incidents were reported at these MPAs on an annual basis from 2004 to 2009, with *P. areolatus* making up the vast majority of the catches (TP, personal observations). Nighttime spearfishers can obtain very high catches of *P. areolatus* at FSAs because this species, in contrast to the other two species, tends to reside in very shallow water at night and is not easily disturbed by the illumination of underwater flashlights (Hamilton et al., 2005). For example, a UVC and catch per unit effort study in Solomon Islands revealed that over two consecutive nights nighttime spearfishers removed 30% of a large spawning aggregation of *P. areolatus* (RH, unpublished data). Outside of FSAs the active non-cryptic foraging behaviour of *P. areolatus* coupled with the ease with which it can be approached also assists daytime spearfishers in harvesting this species (Howard Choat, personal communication). Different levels of fishing on migratory corridors to and from FSAs (Rhodes and Tupper, 2008) may have also contributed to differences in recovery among species and sites, as might regional species-specific differences in recruitment success.

Although Sites 1 and 10 are showing good recovery for certain species, densities are low relative to other monitored FSA in Melanesia (Hamilton and Matawai, 2006) and lower than reported historical abundances (Hamilton et al., 2004). Finally, late age-at-maturity in some species suggests that sufficient time for full recovery may still be some years away (Pears et al., 2007; Rhodes et al., 2011).

4.2. Temporal and spatial variability

In New Ireland *E. fuscoguttatus* and *E. polyphekadion* displayed well-defined spawning seasons, consistent with other geographies (e.g. Johannes et al., 1999; Pet et al., 2005). However, in New Ireland the spawning seasons differed markedly between sites separated by approximately 35 km of open water. At Site 1 (Dyual), a 4–5 month spawning season for *E. fuscoguttatus* and *E. polyphekadion* occurs between March and July, which is virtually identical to that of an *E. fuscoguttatus* FSA on the south coast of Manus Island (*ca.* 300 km west of Site 1) (Hamilton et al., in press). Yet at Site 10 (Tigaks), the peak season for *E. fuscoguttatus* and *E. polyphekadion* occurred between July and November. Small-scale intra-regional variation in seasonality of these species has been documented in Palau and Solomon Islands (Johannes et al., 1999; Johannes and Lam, 1999). Since no information on environmental factors such as temperature, currents or photoperiod were recorded at the studied FSA sites, it is impossible to postulate whether these differences are the result of varying environmental cues. It is likely, however, that the less clearly defined season at Site 10 relates in part to heavy fishing pressure, with Sadovy and Liu (2004) reporting weak FSA signatures in areas subjected to heavy fishing. At Site 1, the March–July season for *E. fuscoguttatus* and *E. polyphekadion* only became clear as the aggregations recovered.

In some geographies in the Indo-Pacific, such as Pohnpei, *P. areolatus* form FSAs over a shorter 5-month season (Rhodes and Tupper, 2008). However, in New Ireland FSAs of *P. areolatus* formed throughout the year, with maximum densities coinciding with the spawning seasons of *E. fuscoguttatus* and *E. polyphekadion*. Many communities in Melanesia are well aware of this monthly aggregating pattern (Hamilton et al., 2005). In Melanesia *P. areolatus* appears to display an intermediate reproductive strategy, between a "resident" spawner, typically aggregating daily and a "transient"

Table 2
Total abundance estimate for *Epinephelus fuscoguttatus*, *Epinephelus polyphekadion* and *Plectropomus areolatus* at Site 1 (MPA) and 10 (MPA) during peak aggregation periods from 2005 to 2009.

Site 1	May-2005	May-2006	May-2007	May-2008	May-2009
E. fuscoguttatus	74	163	423	185	164
E. polyphekadion	22	98	365	128	241
P. areolatus	92	111	193	124	107
Site 10	July-2005	July-2006	July-2007	July-2008	July-2009
E. fuscoguttatus	12	22	33	34	41
E. polyphekadion	0	18	28	17	26
P. areolatus	34	25	51	37	14

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spawner, which spawns only during several months of the year (Domeier and Colin, 1997).

In Melanesia, FSAs of *E. fuscoguttatus*, *E. polyphekadion* and *P. areolatus* disperse around the new moon; however, the specific lunar day on which aggregations disperse can vary slightly both within and between nearby FSAs of the same species (Johannes, 1989). The intensive UVC surveys that were conducted at Site 1 indicate that our sampling protocols captured the peak spawning seasons, with peak densities of all three species persisting until at least 1 day prior to the new moon. This finding is consistent with what we have observed through identical monitoring programs in other parts of Melanesia (RH, unpublished data), and similar to findings in Palau, where peak counts of *E. fuscoguttatus, E. polyphekadion* and *P. areolatus* at FSAs occurred between one and seven days prior to the new moon (Johannes et al., 1999).

4.3. Effectiveness of simple monitoring protocols

This study has shown that repeated monitoring of permanent transects can detect temporal trends in FSA populations, and, as such, is a useful method for establishing the annual seasonality of FSAs and informing stakeholders on changes in the status of their FSAs over time. This ecological information is highly relevant to informing both site-based conservation efforts and wider fisheries management strategies. Just as important, it is easy to collect, interpret and disseminate. Although other more technical FSA monitoring methods are available (e.g. Colin, in press), it was the simplicity and feasibility of permanent transects that appealed to communities when establishing the New Ireland monitoring program. In New Ireland the FSA monitoring program was successful in building support for ongoing FSA closures, enhancing capacity among provincial fisheries department and raising awareness on FSA and marine conservation. For example, after the monitoring commenced, both communities that claim traditional ownership of Sites 1 and 10 requested assistance in developing management plans that cover their FSA and a range of other marine habitats.

5. Conclusions

One of the guiding principles for designing MPA networks is that large areas should to be protected (>6 km^2 in diameter) in order to maximize fisheries and conservation benefits (Hastings and Botsford, 2003). However such criteria do not bode well when establishing community-based MPAs in many regions in the Coral Triangle, because the social and economic costs of implementing large protected areas make them an unrealistic and unacceptable scenario (Aswani and Hamilton, 2004b; Foale and Manele, 2004; Weeks et al., 2010).

Although small MPAs clearly have ecological limitations, this study has shown that in some cases a MPA does not need to be big to make a difference. Small community-based MPAs that were established in Papua New Guinea specifically to protect overfished FSAs of large groupers are resulting in the recovery of several aggregating species. While beyond the scope of this study, we believe that the recovery being seen is in part a reflection of our early efforts to understand the local cultural context in which a number of potential important FSAs in the region were located. By understanding which FSA sites were suitable from both an ecological and social perspective, we maximised our chances of success. This conclusion is supported by a growing body of literature that stresses that understanding local context, conservation opportunity and ecological value is critical if community-based conservation initiatives are to be successful (McClanahan et al., 2006; Game et al., 2011).

Despite these positive results, this study has also shown that small community-based MPAs may not result in the recovery of all aggregating species if enforcement is not stringent. In this study, *P. areolatus* aggregations at Sites 1 and 10 did not recover, most likely due to relatively low levels of poaching and the high vulnerability of *P. areolatus* during both aggregated and non-aggregated periods. For New Ireland FSAs other management options such as seasonal sale bans may also be beneficial. Seasonal sale bans would restrict the number of outlets where fishers could sell groupers that had been poached from protected FSAs. These bans would also offer some protection to FSAs in New Ireland that currently have no management.

We also acknowledge that the conservation of FSAs may not be sufficient to preserve aggregating species in locations where fishing pressure is very high during their non-aggregated life stages (Johannes et al., 1999). Finally, even community-based MPAs that are designed upon the best known ecological and social criteria and are rigorously enforced will fall short of addressing issues other than overfishing (e.g. effects of climate change, poor land use practices). Community-based MPAs should be viewed as one of many strategies that fit within broader ecosystem-based approaches to management.

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