Research Article

Do Not Stop: The Importance of Seamless Monitoring and Enforcement in an Indonesian Marine Protected Area

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The harvesting of groupers (Serranidae) in Indonesia for the live reef food fish trade (LRFFT) has been ongoing since the late 1980s. Eight sites in Komodo National Park that included two fish spawning aggregation (FSA) sites were monitored for groupers and humphead wrasse, *Cheilinus undulatus*, from 1998 to 2003 and from 2005 to 2008 to examine temporal changes in abundance and assess the effectiveness of conservation and management efforts. Monitoring identified FSA sites for squaretail coralgrouper, *Plectropomus areolatus*, and brown-marbled grouper, *Epinephelus fuscoguttatus*. Both species formed aggregations before and during full moon from September to December, prior to lapses in monitoring (2003–2005) and in enforcement (2004–2005). Following these lapses, data reveal substantial declines in *P. areolatus* abundance and the apparent extirpation of one aggregation at one site. Other non-aggregating species targeted by the LRFFT showed similar declines at three of eight monitored sites. This paper highlights the impact of FSA fishing and the need for a seamless monitoring and enforcement protocol in areas where aggregation fishing pressure is high. Within Komodo National Park, local fishers, particularly those operating on behalf of the LRFFT, pose a serious threat to population persistence of species targeted by this trade.

1. Introduction

Groupers (Serranidae) and the Convention for International Trade in Endangered Species (CITES) Appendix II-listed humphead wrasse (*Cheilinus undulatus*) are among the most vulnerable species to overfishing, because they form temporally and spatially predictable fish spawning aggregations (hereafter, FSA) that are readily targeted by fishers [1–3]. In addition to concentrating reproductive activity to set times and locations, some FSA species have life history traits that make them especially vulnerable to over exploitation, such as hermaphroditic sexual patterns and/or delayed maturity (see [4–6]). When fishing pressure is heavy and persistent, aggregations can decline rapidly and in severe cases can be extirpated [7, 8].

Some aggregating groupers and humphead wrasse are highly sought after by local Indonesian fishers for the

Southeast Asian live reef food fish trade (LRFFT), which supplies live coral reef fishes to restaurants and seafood markets throughout Southeast Asia (e.g., [9]). Wild-caught groupers, such as the humpback grouper, *Cromileptes altivelis*, humphead wrasse, and some species of *Epinephelus* and *Plectropomus* fetch prices ranging from US\$2 to US\$35 per kg in Indonesia [10], an equivalent of 0.12%–2.14% of the total average annual income (US\$1,634 = per capita income 2006). Hence, fishing for the LRFFT is an attractive incentive for local Indonesian fishers but a bane to local coral reef ecosystems which are seeing the rapid decline in large predatory fish and overall changes to the structure of reef fish populations [11]. The primary problem with the LRFFT is the targeting of FSA to meet demand [1], which has been widely implicated in spawning aggregation loss and species population declines throughout the Indo-Pacific

(e.g., [12, 13]). Indonesia, in particular, began exporting live reef fish to Hong Kong in 1988 [14, 15] and by the 1990s was supplying over 50% of all live fish to Southeast Asian markets [12]. By the late 1990s, many regions in Indonesia experienced sharp declines in the abundance of the fish targeted by the trade, and there are currently few known remaining FSA. In 2000, an estimated 48,500 metric tonnes was exported from Indonesia, which represents a threefold increase from export volumes a decade prior [16]. Currently, the retail volume of the LRFFT within the Coral Triangle region varies from 18,000–50,000 metric tons annually and is valued at >US\$800 million [17].

Nestled in Indonesia's Lesser Sunda Islands within the Coral Triangle, Komodo National Park (KNP) features one of the world's richest and most diverse marine environments for corals and reef fish [18]. The 4000 residents living within KNP and 45,000 people living adjacent to the Park are reliant on marine resources for their daily needs, and consequently, there has been heavy local fishing pressure on the Park's coral reef fish and invertebrate populations. One of the key management objectives for KNP is to protect biodiversity and maintain spawning stocks of commercial fishes and invertebrates of local fisheries [19]. To that end, a zoning system was established for KNP in 2001, which includes regulations to ensure the long-term survival of the Park's flora and fauna, its ecosystems, and its local communities. One of the key targets of these regulations is to protect FSA within the KNP, placing all known sites within "no-take" (i.e., no fishing) zones.

In 2005, Pet et al. [2] provided a detailed analysis of reproductive seasonality and trends in abundance and individual fish length for two coaggregating species (*Plectropomus areolatus* and *Epinephelus fuscoguttatus*) at two spawning sites identified through exploratory monitoring in KNP. In their paper, the authors defined a September through February (primary) reproductive season, with greater abundances and reproductive behavior recorded during full moon periods although new moon aggregations (April to July) were also noted. Both new moon (e.g., [7, 20]) and full moon aggregation formation (e.g., [21]) have been observed elsewhere for both *E. fuscoguttatus* and *P. areolatus* although it is currently unclear whether reproduction occurs within only one or both of these two lunar periods.

Among the many attributes of the Pet et al. [2] study, were abundance counts of aggregating *P. areolatus* and *E. fuscoguttatus* from 1995 to 2003 that allow subsequent comparisons. This paper builds on Pet et al. [2] by incorporating three additional years of data (2006–2008) and an additional six sites and seven species not included in the original assessment. The important feature of the current study is that directly following the original Pet et al. [2] study, there was a 29-month lapse in monitoring. Additionally, from 2004 to 2005, enforcement was at its lowest level in a decade [22], allowing fishing to continue relatively unabated within the KNP.

The KNP monitoring program was designed by The Nature Conservancy in partnership with the Komodo National Park Authority and initially intended to (a) provide data on temporal changes to the population of economically

important fish species and (b) assess the effectiveness of conservation and management efforts in the KNP, particularly the enforcement of no-take zones. With the exception of the KNP, there are no other known long-term datasets from Indonesia on grouper or humphead wrasse FSA and only one site-based case of the immediate effects of aggregation fishing [23]. The objective of the current study was to document changes in abundance of spawning aggregations of highly vulnerable groupers and wrasse following changes in the level of enforcement within the KNP.

2. Methods

2.1. Underwater Visual Census. As reported in Pet et al., approximately 300 potential FSA sites within the KNP were surveyed between 1995 and 2000 during hundreds of exploratory and monitoring dives [2, 19]. To protect remaining fish populations from exploitation, particularly by the live reef fish trade, the names and coordinates of sites are not provided. General locations of FSA sites are shown in Figure 1 in Pet et al. [2]. Eight of these sites were selected for semimonthly monitoring and were monitored initially around full and new moons from March 1998 to February 1999 to establish the presence of FSA. Of those eight, only two sites (Sites 7 and 8) were characterized as FSA sites, and monitoring continued at these sites through 2003 and then from 2005 to 2008. Five additional sites (Site 1–Site 5) were monitored between 1998 and 2003, while three other sites (Site 6–Site 8) were monitored from 1998 to 2006. All sites were monitored over a 3-day period around full and new moon. The timing for sampling was based on preliminary data collected during peak aggregation months and lunar phases for different species [24]. From January 2008, monitoring was conducted only around the full moon. No data were collected from April 2003 to August 2005, when the lapse in monitoring occurred.

Eleven species were selected for monitoring, based on their relative abundance, known vulnerability to overfishing, and high economic value within the LRFFT. These included the known aggregation-spawning species *E. fuscoguttatus*, and *C. undulatus*, camouflage grouper, *Epinephelus polyphkadion*, *P. areolatus*, leopard coral grouper, *Plectropomus leopardus* and nonaggregation or unknown aggregation spawning species potato grouper, *Epinephelus tukula*, Malabar grouper, *Epinephelus malabaricus*, brown-spotted grouper, *Epinephelus chlorostigma*, highfin coral grouper, *Plectropomus oligacanthus*, yellow-edged lyretail, *Variola louti*, blacksaddled grouper, *Plectropomus laevis*, humpback grouper, and *C. altivelis*. Among those, only two species, *E. fuscoguttatus* and *P. areolatus*, aggregated in numbers characteristic of FSA [2]. Two species (*E. tukula* and *E. malabaricus*) were rarely observed and have not been included in analyses.

Details of underwater visual census (UVC) methods are described in Pet et al. [2, 25]. Two divers entered the water at the same GPS location, descended to a maximum depth of 30 m, and started surveys at the same spot identified *in situ*. The surveys covered a transect area of approximately 200 m over a 30-minute period. At least one of the two observers was from the Park Authority and consistently participated in the surveys over the 10-year period. Abundance counts

FIGURE 1: Number of *Plectropomus areolatus* and *Epinephelus fuscoguttatus* recorded at fish spawning aggregation sites at full moon in Komodo National Park from March 1998 to December 2009. Only the main moon phase for fish aggregations is shown in graphs. Enforcement was at its lowest in a decade from 2004 to 2005, and no data were collected between April 2003 to August 2005. Site numbers are included in parentheses (S7 = Site 7 and S8 = Site 8). Note the differences in scale on the y-axis.

and length estimations (± 3 cm total length, TL) of target species were made during a timed (30-minutes, *ca.* 200 m length) swim performed by a two-diver team. Behaviors or signs typically associated with spawning aggregations were also recorded and included color change, gravid females (i.e., swollen bellies), and territoriality. Given the relatively small size of aggregations in KNP (<100 fish), all fish were recorded and subsampling was not required. To reduce potential bias and variability among individual divers within and among surveys, divers were (re)trained 1–2 times annually, with intensive training when new monitors were introduced to the program. The same monitors from the Park Authority led the monitoring over the 10-year survey period.

2.2. Enforcement and Fishing Pressure. In 2007, three 16–20 m vessels patrolled KNP coastal waters during daylight hours. Each vessel had a speedboat to support surveillance and enforcement activities. Surveillance teams worked 10-day patrol shifts, with at least two vessels simultaneously on patrol in the Park. Active patrols totaling 499 days

(or 50 ten-day trips) were conducted between 1 February and 31 December 2007. Data were collected on all boats encountered, including the origin of fishers or dive operators, GPS coordinates and primary activities being undertaken (dive tourism and specific fishing activities). For fishers, information was collected on gear, catch composition (fish, squid, lobsters, etc.) and catch volume (kg). Data presented from 2007 are indicative of the level of enforcement being undertaken in the years 2006 to 2008.

3. Results

3.1. Fish Abundance Trends. As reported in Pet et al. [2], increased abundance and presumed spawning were found for both *P. areolatus* and *E. fuscoguttatus* from September to December (Figure 1, Supplementary Material which is available online at doi:10.1155/2011/501465). Prior to the monitoring and enforcement lapses, abundances for *P. areolatus* were consistently higher (>50 fish) at Site 8 than at Site 7 (Figures 1(a) and 1(b)), while the converse was found

for *E. fuscoguttatus* (Figure 1(c), Supplementary Materials). Following the 29-month monitoring hiatus (2003 to 2005), *P. areolatus* abundance declined by 93%, to less than 10 fish at Site 8 during peak aggregation periods (Figure 1(b)). At Site 7, where abundances of *P. areolatus* were already considered depressed, continued declines occurred between 2005 and 2007 and climaxed with a failure of aggregation formation in both 2006 and 2008 (Figure 1(a)). Paired *t*-tests on maximum annual counts found significant differences in abundances in *P. areolatus*, between the period before (1998–2003) and after (2005–2008) the lapse in enforcement ($P < 0.01$). Mean size of fish changed significantly between the two periods mentioned above (*t*-test, $P < 0.05$).

For *E. fuscoguttatus* at Site 7, a continued decline in abundance was observed between 1998 and 2003. However, following the enforcement and monitoring lapses, abundances increased to 1998 levels. Conversely, abundance at Site 8 appeared relatively unchanged (max. fish = 8) (Figure 1(d)). The size range of both *P. areolatus* and *E. fuscoguttatus* varied throughout the monitoring period with many species showing an overall decrease in mean size; however, the observed changes could not be definitively linked to fishing (Table 1, Supplementary Materials). Paired *t*-tests conducted on maximum annual counts did not find significant differences in abundances in *E. fuscoguttatus*, between the period before (1998–2003) and after (2005–2008) the lapse in enforcement ($P = 0.82$). Mean size did not change significantly between periods in *E. fuscoguttatus* ($P = 0.67$).

Among the other species monitored across the eight survey sites, initial abundances were relatively low (<20 individuals) and varied throughout the survey by species and site. At Sites 7 and 8, abundances declined for *P. leopardus*, *P. laevis*, and *P. oligacanthus*. For *P. oligacanthus* at Site 8, ca. 50 individuals were commonly observed around new moon from January 2001 to January 2003, but by late 2005, *P. oligacanthus* had declined to <20 individuals, with a further decrease to <10 individuals by 2006 (Figure 2). Overall, *P. oligacanthus* declined by 67% across all survey sites. For *P. leopardus*, declines were also noted at Sites 3 and 6. Abundances of *P. leopardus* were higher during new moon monitoring periods. At Site 8 peaks consistent with aggregation abundance levels were observed in August 1998 ($n = 84$), March 2001 ($n = 50$) and 2002 ($n = 59$), and February 2003 ($n = 46$). In contrast, by 2005, no more than six individuals of any of these species was present during any month, and, in many instances, no *P. leopardus* were observed (Figure 2). *C. undulatus* was present at all sites, with highest abundances ($n = 21$) recorded at Sites 6 and 7, particularly around new moon (Figure 2). However, by 2006, *C. undulatus* abundance at Sites 6, 7, and 8 had also declined. Increases in abundance of *C. altivelis* occurred at Sites 1 and 2 from 2003 to 2005; however, no clear trends were evident for this or any of the other target species among sites. Paired *t*-tests conducted on maximum annual counts did not find significant differences in abundances in *P. oligacanthus* ($P < 0.05$), *P. leopardus* ($P < 0.01$), *C. altivelis* ($P < 0.05$), or *C. undulatus* ($P < 0.05$), between the period before (1998–2003) and after (2005–2008) the lapse in enforcement.

TABLE 1: The mean size (cm) of three species of grouper recorded in different years in Komodo National Park. Only years with 12 months of data are shown. The number of fish measured can be found in Supplementary Materials. Asterisks indicate changes between the periods before (1998–2002) and after (2005–2007) the lapse of enforcement that were significant.

Species	1998	1999	2000	2001	2002	2005	2006	2007
<i>P. areolatus</i> *	53.2	49.3	50.0	49.9	48.4	47.7	47.6	44.9
<i>E. fuscoguttatus</i>	68.8	64.8	66.7	69.6	67.1	65.9	65.3	69.1
<i>E. polyphkadion</i>	47.8	35.6	39.4	42.5	44.6	42.1	46.0	39.0
<i>P. leopardus</i>	43.6	41.4	40.7	40.4	39.8	37.1	39.1	
<i>P. laevis</i>	78.1	86.0	97.0	97.4	85.4	66.8	74.8	
<i>P. oligacanthus</i> *	44.3	43.4	42.0	42.6	42.2	38.6	40.4	
<i>V. louti</i>	42.6	36.0	37.9	39.0	38.6	36.0	45.5	
<i>C. altivelis</i> *	42.5	42.5	43.7	45.0	44.5	40.1	36.0	
<i>C. undulatus</i>	77.6	77.7	93.3	93.6	88.6	86.8	75.4	

Species such as *P. laevis* and *V. louti* were recorded at all sites but consistently found in low abundance (<14 fish) (Figure 2). While *V. louti* or *C. altivelis* showed no lunar pattern, *P. laevis* abundance peaked around new moon. The highest numbers of *P. laevis* were recorded at Site 7 from 1998–2003 ($n = 14$), yet there was a distinct decline in this species at Sites 6, 7, and 8 by 2005 (<5 fish). Paired *t*-tests showed significant changes in maximum fish abundances changed between the period before (1998–2003) and after (2005–2008) the lapse in enforcement for *V. louti* ($P < 0.05$) but not for *P. laevis* ($P = 0.08$). Overall, for KNP, annual peak abundances (maximum number of fish observed across sites) declined or remained consistently low over 10 years, with no species achieving abundances above 20 by 2008 (Figure 3).

3.2. Enforcement and Fishing Pressure. A total of 1734 boats were recorded by patrols within KNP between 1 February and 31 December 2007 (499 patrol days and 50 patrol trips). Patrols encountered an average of 7.4 fishing boats per day in the Park (range 1–34 boats) carrying a total of 6340 fishers. The majority of fishers encountered within the no-take zones were fishing on the north and northeast corners of Komodo Island where known FSA occur and within the nearshore waters of Padar and Rinca Islands (Figure 4). The majority of fishing activities were recorded within 500 m of the coast within no-take zones (64.9%), while a relatively smaller percentage of fishing activities occurred within traditional use or pelagic zones where fishing is allowed (35.1%). Sites 7 and 8 and reefs on the northeastern side of Komodo Island were heavily targeted by fishers within the Park. Of the 1734 fishing boats recorded between February to December 2007, 709 boats (40.9%) came from villages within the Park and 1011 boats (58.3%) from villages outside the Park. Preferred gear types within KNP include hook and line (34.4%), encircling gill nets (20.7%), lift nets (“bagan”) (14.3%), and seine nets (12.1%). Hook and line is the primary gear type used by local fishers to catch groupers and humphead wrasse.

FIGURE 2: Continued.

FIGURE 2: Continued.

FIGURE 2: Graphs showing the changes in abundance of grouper and Napoleon wrasse species in Komodo National Park at sites monitored from 1998 to 2003 ($n = 5$) and from 1998 to 2006 ($n = 3$). Enforcement was at its lowest in a decade from 2004 to 2005. Note the differences in scale on the y-axis. S1–S2 (Sites 1–Sites 8) refers to different sites. Grey bars: full moon data, lines: new moon data. Only the most dominant moon phase is presented in graphs, where it was obvious from the data. Overall percentage declines for target species before and the period of reduced enforcement were *Plectropomus areolatus* (76.9%), *Ephinephelus fuscoguttatus* (6.9%), *P. leopardus* (84.3%), *P. oligacanthus* (66.9%), *Cheilinus undulatus* (62.1%), *Variola louti* (79.5%), *Cromileptis altivelis* (65.2%), *P. laevis* (33.3%), and *E. polycephalus* (46.2%).

FIGURE 3: The maximum number of fish recorded in each year (across sites) for eight species of grouper and Napoleon wrasse in Komodo National Park. Enforcement was at its lowest in a decade from 2004 to 2005. Asterisks indicate significant differences in maximum fish counts between periods 1998–2003 and 2005–2008, for individual species.

4. Discussion

The current study highlights the importance of consistent, long-term monitoring and enforcement of marine protected areas (MPAs) and illustrates the rapidity with which

unregulated fishing can cause FSA loss and/or abundance declines among targeted commercial species. Specifically, peak abundances following the lapses in monitoring and enforcement (across sites and years) declined 6.9%–84.3% ($57.9 \pm 25.1\%$, mean $\pm$ SD) overall for targeted species and $55.3 \pm 30.8\%$ (mean $\pm$ SD) for aggregating species (Figure 3).

For KNP, these impacts are particularly pronounced, since the sites affected were the only identified aggregations within the Park and among the few known remaining grouper FSA sites within Indonesia [23]. In contrast to other regional locales where grouper aggregations commonly contain 100s or 1000s of individuals (e.g., [7, 13, 23, 26, 27]), *P. areolatus* aggregations in the KNP appeared relatively depauperate even during initial monitoring. Given the current level of impact to these aggregations, a substantial recovery period, likely in the order of decades, will be needed if populations are to be restored [28]. The loss of both *P. areolatus* aggregations in KNP adds to decadal declines in reproductive stocks from unreported and unregulated commercial fishing in Indonesia that has left few reproductively viable aggregations. Following 20 years of activity by the LRFFT in Indonesia, the country still has no regulations in place that could be applied to the LRFFT or to govern the industry. Operating standards and best practices are not well documented [17].

In contrast to *P. areolatus*, *E. fuscoguttatus* FSA appeared to be less impacted by the enforcement lapse. The observed variation in the impact from fishing between *P. areolatus* and *E. fuscoguttatus* has been observed elsewhere (K. Rhodes, unpublished data). The preference among Asian diners and higher prices paid by the LRFFT for *P. areolatus* and *P. leopardus* relative to *E. fuscoguttatus* may be at least partly responsible for the observed disparity [9]. In addition to

FIGURE 4: GPS positions of boat-based fishing activities in Komodo National Park from recorded between February to December 2007. The different zones where fishing is allowed are shown as a single “use” zone.

price and preference, differences in depth distribution and feeding behavior may have also affected the vulnerability of *E. fuscoguttatus* to fishing. The species appears less prone to hook and line fishing during aggregation periods and is typically found deeper (15–40 m) than *P. areolatus* (3–15 m) (A. Muljadi, *personal observation*). Similar differences in depth preferences have been observed in these two species at sites at Manus, Papua New Guinea [29], Pohnpei [21], and Ayau Island, Indonesia (J. Wilson, *personal communication*).

In general, impacts from fishing varied among species and sites, with the greatest impacts noted among highly prized coral grouper (*Plectropomus* spp.) and *C. undulatus*, with the most pronounced impacts within FSA sites. Surveillance and enforcement data identified a number of illegal fishing activities at, or close to, grouper FSA sites throughout the year (Figure 4). The spatial pattern of fishing boats suggests that groupers were vulnerable both within aggregation sites and adjacent habitats, where home range habitats and migratory corridors likely occur for these species (e.g., [3, 21, 27]).

Enforcement records show no patrol days in 2004 and only 59 patrol days in 2005; the lowest records over a 10-year period [22, Figure 4]. During lapses in monitoring and in enforcement, no-take zones throughout KNP became more accessible to both local and outside fishers. Despite considerable efforts by the Park Authority and the non-profit organization Putri Naga Komodo to socialize and implement the zoning plan, noncompliance within no-take zones continues and is one of the greatest challenges faced by managing agencies. Enforcement data showed 64.9% of boats encountered during patrols were fishing in no-take zones. Prosecutions have been made for illegal destructive fishing methods, such as cyanide and bomb fishing, but little is done to prosecute fishers caught fishing within the no-take zones that have been in place since 2001. Dive operators report that unrestricted nighttime fishing occurs in northern Komodo near FSA sites when patrols leave the area to anchor in secluded bays. Clearly, overall enforcement is ineffective as illustrated by these reports of noncompliance in combination with *in situ* fish abundance trends. This pattern

of poor enforcement of no-take zones (marine reserves) is characteristic of many MPAs in Indonesia [30, 31] and of MPAs in general [32].

Socioeconomic theory of compliance explains that deterrence is based on the probability of detection and the certainty of penalty compared to the gains from illegal activities (or noncompliance) [33]. In addition, other factors may be important in contributing to compliance in MPAs, such as perception of legitimacy, moral and personal values of individuals, fairness in how benefits and costs are distributed, demonstration of the socioeconomic improvement of coastal communities, and availability of alternative livelihoods [29–31]. In the case of KNP, there are no penalties enforced for fishing in no-take zones, and fishers profit from selling their fish to the rapidly expanding local markets in Labuan Bajo, adjacent to the Park. Some alternative livelihoods have been provided, but these are not sufficient to make a meaningful impact to a population of 20,000 living within and surrounding KNP.

For Indonesia, the economic impacts from the loss of FSA and the impact of illegal fishing are significant though poorly understood. For example, 28,000 tourists currently visit KNP, with entrance fees alone bringing in \$420,000 per year (R. Djohani, *personal communication*). The economic value of certain large fish species, such as groupers and sharks, is substantial for KNP tourism [34] but is not fully understood by managers/policy makers and local communities. One likely driver of noncompliance is the paucity of revenue sharing from tourism activities with local communities, including entry fees. Many of the visiting dive operators are foreign-owned with outside staff and little investment in local communities. To remedy this situation, KNP may benefit from a structure similar to that developed for Raja Ampat, Eastern Indonesia. In Raja Ampat revenues generated through a tourism entrance fee system is distributed as follows: 30% of revenues goes to the government, 28% to local communities conservation initiatives within MPAs, 28% to community well-being within the Regency, and 14% to the direct administration and management of the fund. To date revenues generated from tourism in Raja Ampat have provided support for community patrols within MPAs and food supplements to clinics for mothers and infants. This model provides clear and tangible benefits to local communities and is one of a range of incentives to ensure compliance with fisheries regulations relating to destructive fishing practices.

Community comanagement of marine resources and clear mechanisms for determining territorial user rights to zones where fishing is allowed have been discussed but has not yet materialized in KNP. Given the lack of community buy-in for KNP (as demonstrated by local communities both within and surrounding the Park fishing in no-take zones), the long-term conservation of the Park will require greater empowerment and revenue sharing with communities, including participating in patrols. Currently, there are little benefits of communities “doing the right thing” and following zoning regulations relating to fishing. There are clear opportunities for greater comanagement in MPAs across Indonesia. Institutional and legal mechanisms are

available that allow the integration of a formal management framework with community-based management [35, 36]. For KNP, a clear change in the financial and governance structure is needed to promote greater conservation of the Park’s resources. In addition to these changes, more focused patrols and enforcement, with greater penalties for noncompliance, are needed. Coupled with enforcement is the need for stronger economic incentives, a reduction in illegal gains and greater penalties, and a clearer relationship between compliance and community benefits from the tourism and fisheries sectors.

Finally, it is important to appreciate that apparently even a brief interruption or lapse/reduction of enforcement of an MPA can lead to dramatic declines. In effect there were only two potential aggregation periods during which there was little to no enforcement, yet enforcement lapses allowed enough of an increase in fishing activity to markedly reduce abundance and one FSA and decimate another. This is a strong reminder that consistent vigilance and effective community buy-in to management schemes is key to fisheries management. It also appears increasingly likely that all grouper and humphead wrasse fisheries need to be stopped for an extended period in Indonesia to allow for population recovery if fisheries managers are serious about the conservation and management of these fish species. The failure to manage the trade and limit illegal fishing is endangering coral reef ecosystem function through the removal of top predators [11] and imperiling Indonesia’s economic and food security. However, any additional measures are likely to fail without greater community awareness and buy-in that includes a greatly revised fee structure, in the case of KNP, focused more on revenue sharing. In geographically remote areas where national, provincial, or regency institutions are weak or inadequately resourced, alternative mechanisms of management, including nonformal rules, may have a stronger influence on social behavioral change. In regions where enforcement is inadequate, community-managed areas may be best for achieving conservation/fisheries goals or objectives [36, 37].

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