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OF INDONESIA'S BALI BARAT NATIONAL PARK, 2011–2016

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ECOLOGICAL CHANGES IN THE CORAL REEF COMMUNITIES OF INDONESIA'S BALI BARAT NATIONAL PARK, 2011–2016

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ABIGAIL ALLING², and PHILLIP DUSTAN⁴

ABSTRACT

The coral reefs of Bali Barat National Park, one of Indonesia's oldest marine protected areas, are known for their high biodiversity and excellent sport diving; however, stressors such as destructive fishing practices, elevated water temperatures, damage from anchors and careless visitors have been observed on these reefs for decades. The purpose of this study was to document and quantify changes in the fish and stony coral community structure of reefs within and outside the boundaries of Bali Barat National Park from 2011 to 2016, including its most popular dive site, Menjangan Island. The results provide further knowledge about the reefs of NW Bali and the efficacy of current management practices, and they will inform management decisions for locally managed reef stewardship programs.

Between 2011 and 2016 the reefs of NW Bali lost 44.4% of their living coral cover, declining from 36% to 20% overall cover. Mortality was principally attributed to thermal bleaching caused by persistently high sea temperatures, which peaked in January 2016 at 32.2°C, coinciding with the third documented global bleaching event. Approximately one third of all stony corals were found to be bleached or recently dead. Despite the decline in coral cover, stony coral genus richness remained unchanged, with 56 genera recorded in both years, representing a combined total of 59 distinct genera. Mean fish biomass at Menjangan Island increased, with herbivorous fish biomass quadrupling, presumably due to decreased fishing effort at the island. The abundance of fish at all sites—both inside and outside the park—more than doubled, indicating a predominance of small fish at sites where fish biomass did not correspondingly rise. Crown-of-thorns starfish (*Acanthaster planci*), not observed on transects in 2011, were found in 2016 in areas of relatively high disturbance from marine recreation and possible eutrophication from shrimp farm effluent and mainland runoff. Patterns of coral cover and damage, fish abundance and biomass, and lost fishing gear suggest that management activities inside and outside the park have reduced ecological damage. Local community conservation groups are practicing one or more of the following at Menjangan Island and in some of the locally managed conservation areas: installing moorings, removing coral predators (crown-of-thorns starfish and *Drupella* snails), reducing fishing pressure, securing live coral fragments back onto the reef and planting mangroves. These nature groups are raising community awareness about the importance of NW Bali's marine ecology to their economic and cultural wellbeing.

Keywords: Menjangan Island; ecological resilience; bleaching; community-based management

INTRODUCTION

The coral reefs of NW Bali, Indonesia are known for their high biodiversity and excellent sport diving. Among the region's most visited are the reefs surrounding Menjangan Island, within Bali Barat National Park (BBNP). The reefs of BBNP, one of Indonesia's oldest national parks, have been studied more than many other reefs in Indonesia with published observations on their marine ecology dating back

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to at least 1979 (Polunin et al., 1983). This report documents ecological changes in the community structure of reefs within and outside the boundaries of BBNP, focusing on same site ecological surveys in 2011 and 2016. We consider the influence of (a) time, i.e. 2011 vs. 2016; and (b) the interaction of time and location, i.e. how locations inside and outside the national park may have changed differently over time. Significantly, our sampling captured changes caused by the global bleaching event of 2014–2017 (NOAA, 2017; Witze, 2015). Drawing upon earlier published and unpublished observations, we suggest how management practices and biological stressors such as coral predation, nutrient pollution, fishing and thermal stress have influenced coral reef community structure.

History of Bali Barat National Park

Bali Barat National Park (BBNP) is one of 51 Indonesian national parks managed by the Ministry of Forestry, encompassing approximately 190 km², including a coastal marine protected area (MPA) of approximately 34 km² (Mahmud et al., 2015a). The area was first designated a Wildlife Reserve (*Suaka Margastawa*) in 1947, which grew in 1978 to encompass Menjangan Island, a small (<2 km²) island about 600 m off the Bali mainland (Robinson et al., 1981; Polunin et al., 1983). In 1982, the reserve was proposed for park status, including the marine area around Menjangan Island, which it officially achieved in 1995 (Mahmud et al., 2015a). Since then, BBNP has become a well-established national park and Menjangan Island's reefs the focus of various management strategies, social movements and ecological studies (Figure 1).

Since the early 2000s, park authorities have engaged with community-based groups to promote common interests in ecological wellbeing and economic opportunities. In 2001, a multi-stakeholder forum—the Coastal Care Community Communication Forum of Bali Barat National Park (*Forum Komunikasi Masyarakat Peduli Pesisir Taman Nasional Bali Barat*)—was established to co-manage the park through improved stakeholder communication and collaboration (Setiasih, 2003). The Forum, funded primarily by World Wildlife Fund (WWF) and BBNP, developed a Code of Conduct of the Park,

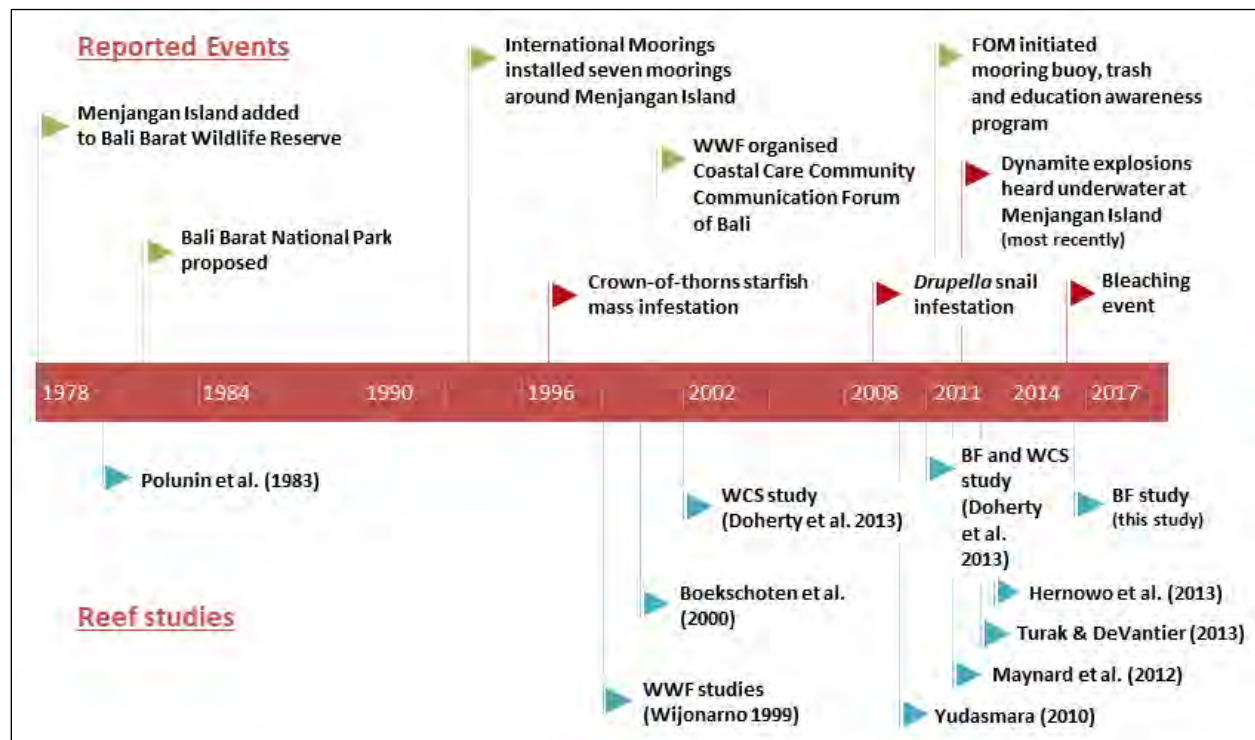


Figure 1. Timeline of significant management activities, ecological events and reef studies around Menjangan Island; references refer to the work in which the studies were described, though the publication year may differ from the study year.

organized trainings and provided a sea patrol to act against dynamite and cyanide fishing around Menjangan Island (Syarif, 2009; Doherty et al., 2013). Initially, there was broad stakeholder participation, but lack of funds and human resources led to its disintegration sometime shortly after 2008 (Yudasmara, 2010; Mahmud et al., 2015a; Doherty et al., 2013).

The group Friends of Menjangan (FOM) was launched in 2011 to engage boat drivers, park guides, resort employees, and other community stakeholders in marine conservation activities at Menjangan Island (Doherty et al. 2013). Activities of the group include mooring buoy installation and maintenance, removal of excessive coral predators including crown-of-thorns starfish (*Acanthaster planci*; COTS) and *Drupella* snails, removal of trash, and educational programs for local children. Scientific observation and periodic reef monitoring, such as this study, guide these activities and assess their effectiveness. FOM was formed as a community-based conservation movement by Biosphere Foundation (BF), Dwi Asih Sejahtera Foundation (*Yayasan Dwi Asih Sejahtera*) and BBNP, continuing these activities through the time of this writing.

Reef Stressors in NW Bali

Reef ecological threats documented in NW Bali include:

- plastic and other types of litter (Polunin et al., 1983; Setiasih, 2003; Turak and DeVantier, 2013; Doherty et al., 2013);
- outbreaks of COTS (Wijonarno, 1999; Boekschoten et al., 2000; Doherty et al., 2013) and *Drupella cornus* (Kertaharja, 2009);
- coral bleaching in 2009 (Kertaharja, 2009) and 2010 (Setiasih and Wilson, 2010);
- coral mining (Robinson et al., 1981; Polunin et al., 1983; Wijonarno, 1999);
- dynamite, cyanide and other destructive fishing methods (Robinson et al., 1981; Polunin et al., 1983; Wijonarno, 1999; Boekschoten et al., 2000; Setiasih, 2003; Doherty et al., 2013);
- snorkeling and diving practices that damage coral (Wijonarno, 1999; Boekschoten et al., 2000; Setiasih, 2003; Doherty et al., 2013);
- anchoring on coral (Wijonarno, 1999; Doherty et al., 2013);
- unclear national park regulations and zone boundaries (Wijonarno, 1999; Doherty et al., 2013);
- lack of legal basis for co-management of the park (Setiasih, 2003; Satria and Matsuda, 2004); and
- ineffective exit strategies for non-governmental organizations (NGOs) and other supporting institutions (Setiasih, 2003; Doherty et al., 2013).

Additional ecological threats to reefs in NW Bali, yet undocumented, include human population increase, lack of public garbage collection and sewage treatment services, and unchecked development—issues common to many developing islands (Burke et al., 2002; Burke et al., 2012).

This study focused on a subset of stressors, using observable field indicators to assess the condition of reefs in and around BBNP in 2016 and reef community change since 2011 (Table 1).

Table 1. Threats to NW Bali’s reef ecosystem with observed indicators and present management strategies.

Threat	Indicator	Present management at BBNP and nearby locally managed coastal areas
Accumulation of stress	Percent cover of live stony coral	Synergy of all practices described below
Thermal stress	Water temperature; Percent cover of bleached stony coral	None (considered a global threat, not amenable to local management)
Structural damage	Count of broken/upturned stony corals	Installation & maintenance of moorings; Boat driver/guide/tourist awareness campaign; Replanting of live coral fragments on some reefs outside the park; Installation of Biorock structure outside the park
Nutrient pollution related to coastal development	Percent cover of macroalgae; Cyanobacteria mats; Coral disease	Planting mangroves in some coastal areas outside the park
Coral predation by COTS	Count of COTS	Removal of COTS
Harvesting of marine organisms	Count of lost fishing gear; Fish biomass and abundance	Zoning system that regulates fishing inside the park

METHODS

Study Sites

Reefs were surveyed at ten sites, aggregated into three groups distinguished by management regime and geography (Figure 2, Table 2). They include:

- Menjangan Island National Park (Island NP)—Menjangan Island fringing reefs inside BBNP; managed through a collaboration between BBNP officials and the FOM community group; located far from villages; and separated from the Bali mainland by a 600 m-wide channel.
- Mainland National Park (Mainland NP)—mainland fringing reefs inside BBNP; managed by BBNP officials; generally located farther from villages; and adjacent to the mainland.
- Outside National Park (Outside NP)—fringing reefs outside BBNP; unregulated; within 2.5 km of villages; and both adjacent to and separated from the mainland. Sites are outside but near (0.4–2.5 km away from) coastal areas managed by community groups.

The same sites were previously sampled in 2011 by Doherty et al. (2013) and in 2002 by Wildlife Conservation Society (WCS) (reported by Doherty et al., 2013). Consistent methods were followed across all three studies. The results from Doherty et al.’s (2013) surveys in 2011 were re-aggregated to match the current study’s scheme, allowing the direct comparisons between years 2011 and 2016. The results from 2002 were aggregated and reported differently by WCS, allowing only for generalized comparisons.

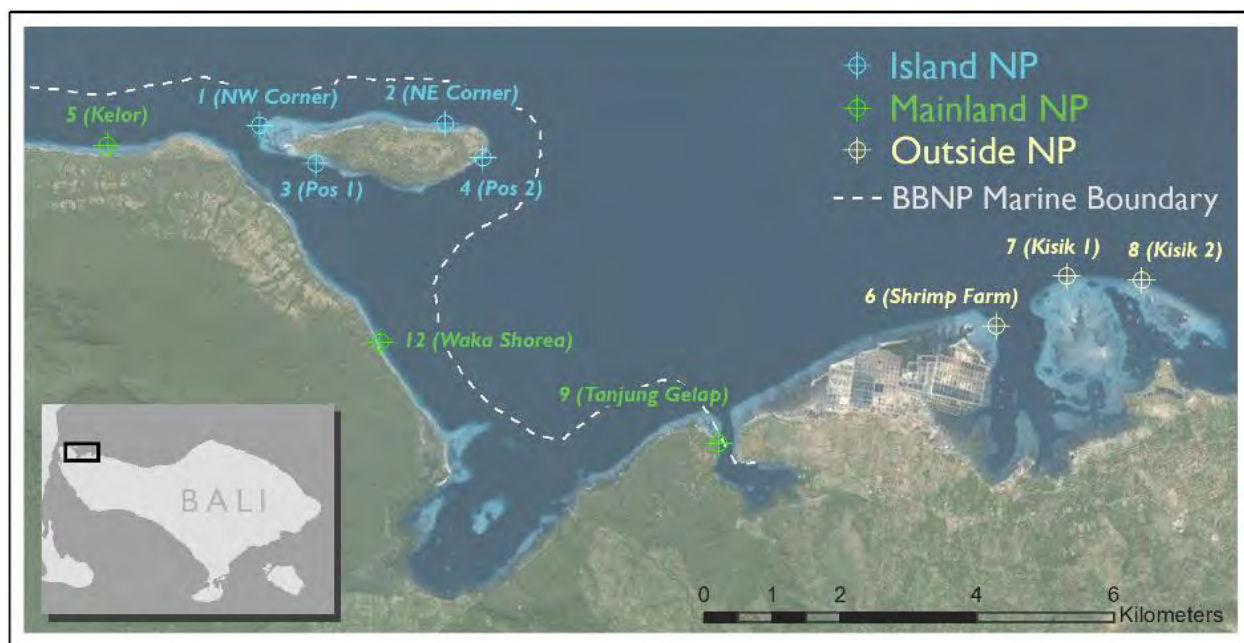


Figure 2. Sites sampled in 2011 and 2016, grouped into *Island National Park (Island NP)*, *Mainland National Park (Mainland NP)* and *Outside National Park (Outside NP)* management areas. Data: Esri (imagery); www.gadm.org (inset map boundaries); RFS|PAC™ Demonstration Project at Bali Barat National Park, <http://cees.columbia.edu/current-programs/rfs-website/rfs-home-page> (BBNP marine boundary).

Table 2. Sites and locations studied in 2011 and 2016 (geographic coordinates reported using WGS 84 datum). Colors correspond to map (Figure 2).

Management Area	Site Code	Site Name	Latitude	Longitude
Island NP	1	NW Corner	-8.09223	114.49850
Island NP	2	NE Corner	-8.09207	114.52281
Island NP	3	Pos 1	-8.09717	114.50587
Island NP	4	Pos 2	-8.09647	114.52775
Mainland NP	5	Kelor	-8.09502	114.47855
Mainland NP	9	Tanjung Gelap	-8.13412	114.55884
Mainland NP	12	Waka Shorea	-8.12083	114.51425
Outside NP	6	Shrimp Farm	-8.11876	114.59530
Outside NP	7	Kisik 1	-8.11227	114.60478
Outside NP	8	Kisik 2	-8.11282	114.61471

Field Studies

Data were collected during the NW monsoon season, in March–April 2011 and May–June 2016. At each of the ten sites, observations were accomplished along six 50 m transects, three at a depth of 2–4 m and three at 6–8 m. At Site 9, four rather than six transects overlapped across years and were used. The ten sites were grouped into the three management areas: *Island NP* with four sites; *Mainland NP* with three sites; and *Outside NP* with three sites (Figure 2; Table 2). Transects in 2016 were placed approximately, though not exactly, in the same location as those in 2011. Observations at both depths

were aggregated to the site or area level, as appropriate, to ensure a sufficient number of replicates upon which to base the statistical analysis.

Substrate Composition

The composition and coverage of benthic substrate were estimated along 50 m Point Intercept Transects (PITs) at 50 cm intervals: live stony coral cover (hereafter called coral cover), recently dead coral, bleached coral, and macroalgal cover. Live stony corals were identified to genus. The category *recently dead coral* included those that were dead but still had a recognizable polyp structure without erosion or major algal or cyanobacteria cover. Such corals had probably died within the past six months.

Coral Structural Damage

The number of broken (and yet unhealed) and overturned live colonies was counted within a 2 m x 50 m belt transect centered on each PIT. Because only the live coral colonies that were damaged were counted, the data were standardized by dividing the counts by the PIT's percent live coral cover to normalize across transects.

Crown-of-Thorns Starfish

COTS counts were recorded for each 2 m x 50 m belt transect.

Coral Genus Richness

A list of live coral genera within each 2 m x 50 m belt transect was compiled.

Fishing Gear

A count of lost and entangling fishing lines, nets and traps on the reef substrate or live corals was recorded for each 2 m x 50 m belt transect. The mean number of fishing gear pieces ($\text{no. dam}^{-2} \pm \text{SE}$; $1 \text{ dam}^2 = 100 \text{ m}^2$) at each site is presented for 2011 and 2016.

Reef Fish

An underwater visual census of all non-cryptic fishes to species level was conducted along each PIT. Fish transects were 50 m long and 5 m wide for fish greater than 10 cm total length and 2 m wide for fish smaller than 10 cm. Large fish were counted first as the transect tapes were being placed. Smaller fish were counted by subsequent surveys along each transect. Abundance per hectare and biomass per hectare were calculated for each species using published length-weight relationships, following Hoey and Bellwood (2009). Mean ($\pm \text{SE}$) reef fish biomass (kg ha^{-1}) and density (no. ha^{-1}) at each site is presented for 2011–2016, along with samples highlighting ecologically significant species and groups (herbivorous fish and predatory fish).

Herbivorous fish species from the families Acanthuridae, Ephippidae, Kyphosidae, Pomacanthidae, Scaridae, and Siganidae (Appendix 1) were assessed following IUCN guidelines (Green and Bellwood, 2009). The mean ($\pm \text{SE}$) herbivorous fish biomass (kg ha^{-1}) of these species is presented.

Seawater Temperature

Between June 2013 and May 2017 seawater temperature at Site 1, near the NW end of Menjangan Island, was recorded at 30-minute intervals using solid state thermographs attached to the base of a large *Porites lutea* colony within the area of the shallow study transects (4 m depth; Onset Computer Corp HOBO Water Temperature Pro V data logger). Loggers were changed annually. Degree Heating Weeks, a measure of thermal stress during a running 12-week period (Gleason and Strong, 1995), was calculated using 29.5°C as the threshold temperature. While NOAA has adopted slightly higher threshold for the “East Java and Bali” 5 km regional virtual station (Version 3) (<https://coralreefwatch.noaa.gov/vs/>), their calculations are based on an estimate of sea surface “skin” temperature, which is typically higher than the subsurface temperature measured in shallow water benthic environments (Smale and Wernbery, 2009). Additionally, NOAA calculations are based on nighttime temperatures derived from satellites to estimate daily mean temperature while we use the 30-minute interval raw data without application of a 1°C high-

pass filter to exclude hotspot values, both of which would tend to reduce calculations of heat stress (G. Liu, NOAA Coral Reef Watch, personal communication).

Statistical analyses

Unbalanced two-factor ANOVA with regression was used to test the effects of time (2011 and 2016, $n=2$) and the interaction of time and location (grouped by management area, $n=3$; or by site, $n=10$) on dependent variables including percent live coral cover, percent macroalgal cover, number of broken or upturned live corals, number of fishing gear pieces, overall fish biomass and abundance, herbivorous fish biomass and abundance and sub-samples of these variables. The significance test results indicate (a) whether the means of values sampled in 2011 and 2016 were different at the management area and site levels; and (b) whether the effects of time and location were independent. A significant result for (b) suggests that the direction of change over time—increase, decrease or none—varied by location. In some cases, single-factor ANOVA was used to compare the 2011 and 2016 means at isolated locations. Statistical tests were performed in Excel using the *Real Statistics Resource Pack for Excel* (<http://www.real-statistics.com/>).

RESULTS

Coral Cover

Mean coral cover across the three management areas ranged from 31.1–41.9% in 2011 and 15.0–26.3% in 2016 (Figure 3); and across sites from 24.3–58.3% in 2011 and 7.2–43.2% in 2016 (Figure 4). Coral cover decreased significantly from 2011 to 2016 in the three management areas ($F_{1,110}=32.37$, $p<0.01$) and across virtually all sites ($F_{1,96}=59.53$, $p<0.01$). The direction of change over time was the same (negative) across all management areas and sites ($F_{2,110}=0.02$, $p=0.98$, $F_{9,96}=1.31$, $p=0.24$).

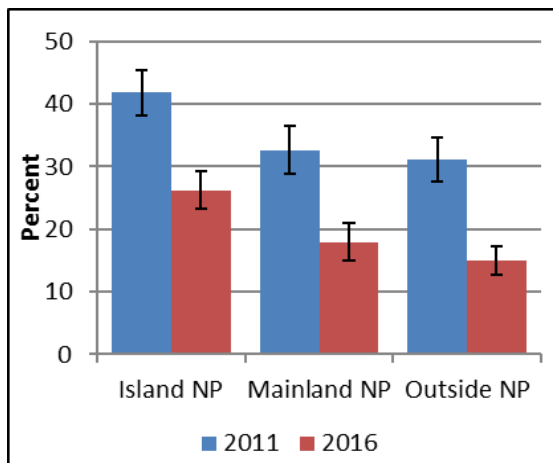


Figure 3. Mean percent live hard coral cover by management area (\pm SE).

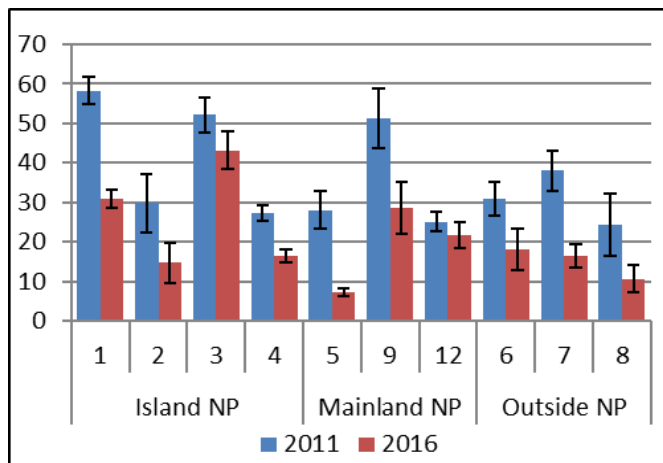


Figure 4. Mean percent live hard coral cover by site (\pm SE).

Coral Genus Richness

Though the composition changed slightly, 56 genera were recorded in both 2011 and 2016 for a total of 59 distinct genera overall (Appendix 2).

Bleached Coral Cover

No bleaching was recorded on the PITs in 2011. In 2016, the mean percentage of bleached coral cover in the three management areas ranged from 34.6% (Mainland NP) to 36.9% (Outside NP) (Figure 5) and across sites from 13.6% (Site 2) to 72.0% (Site 1) (Figure 6), which are both at Menjangan Island.

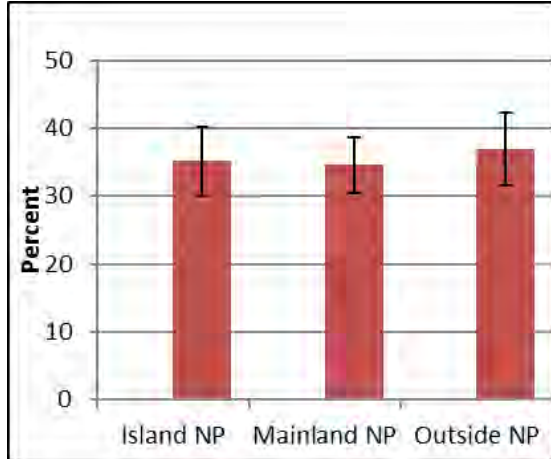


Figure 5. Mean percent bleached live coral cover by management area in 2016 (\pm SE).

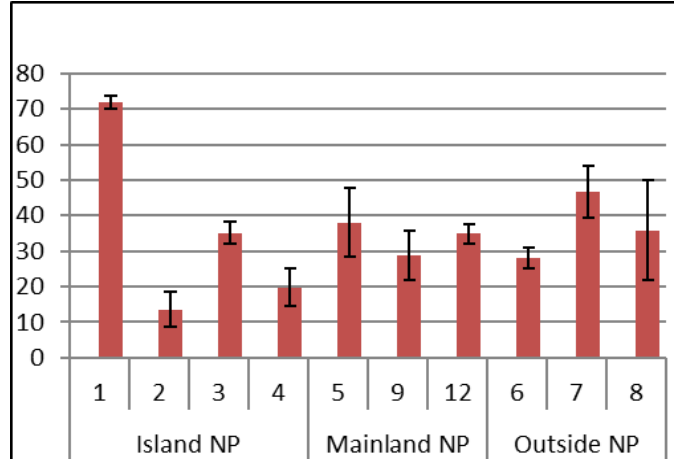


Figure 6. Mean percent bleached live coral cover by site in 2016 (\pm SE).

Structurally Damaged Live Coral

The mean number of broken and upturned live coral colonies relative to live coral cover in each management area ranged from 0.34–0.44 in 2011 and 0.76–1.45 in 2016 (Figure 7); and across sites from 0.16–0.81 in 2011 and 0.17–2.78 in 2016 (Figure 8). The proportion of damaged corals increased as coral cover decreased.

Structurally damaged live coral increased significantly in all management areas ($F_{1,110}=14.20$, $p<0.01$), and there was no significant difference in the amount of increase among management areas ($F_{2,110}=2.06$, $p=0.13$). Among sites there was a significant increase in structurally damaged colonies from 2011 to 2016 at Site 5 ($F_{1,10}=12.1$, $p<0.01$), Site 6 ($F_{1,10}=5.51$, $p=0.04$) and Site 12 ($F_{1,10}=4.97$, $p=0.05$). The other sites show no significant change.

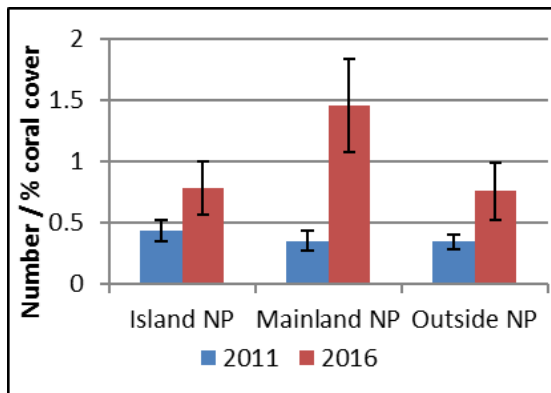


Figure 7. Mean number of structurally damaged colonies per percent live coral cover by management area (\pm SE).

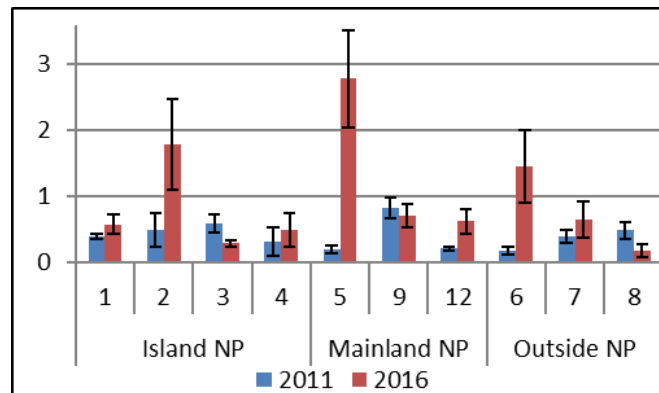


Figure 8. Mean number of structurally damaged colonies per percent live coral cover by site (\pm SE).

Macroalgal Cover

The mean macroalgal cover in management areas ranged from 1.3–3.7% in 2011 and 1.2–4.8% in 2016 (Figure 9); and across sites from 0–5.2% in 2011 and 0–13.5% in 2016 (Figure 10). There was no significant change in macroalgal cover from 2011 to 2016 ($F_{1,110}=1.57$, $p=0.21$ across management areas; $F_{1,96}=2.68$, $p=0.11$ at the site level). Despite the Island NP area decreasing in mean macroalgal cover and the other areas increasing, no significant difference was recorded in the direction of change among management areas or sites ($F_{2,110}=0.65$, $p=0.53$; $F_{9,96}=1.69$, $p=0.10$). Within the Mainland NP area, sites varied greatly in 2016 with Site 9 (13.5%) having a maximum mean macroalgal cover in 2016 while sites 5 (0.3%) and 12 (0.5%) had the lowest (Figure 10).

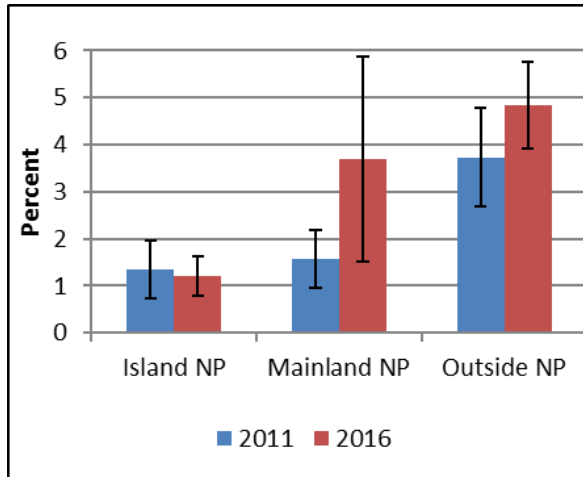


Figure 9. Percent macroalgal cover by management area (\pm SE).

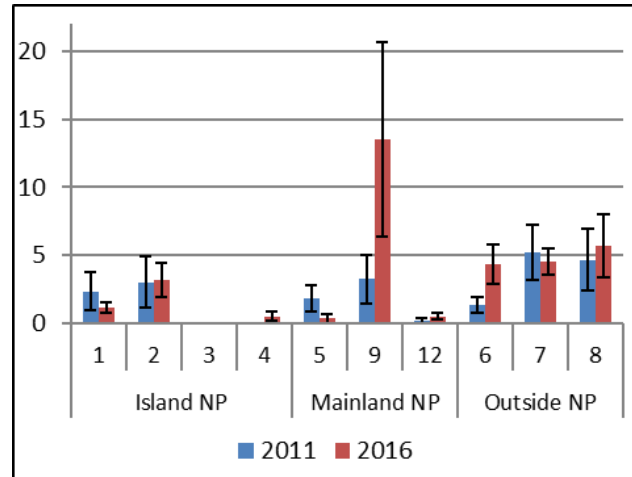


Figure 10. Percent macroalgal cover by site (\pm SE).

Crown-of-Thorns Starfish

No COTS were found on belt transects in 2011, in contrast with 2016, when they were found on some transects. The mean across sites varied from 0.00–1.17 dam^{-2} (Figure 11).

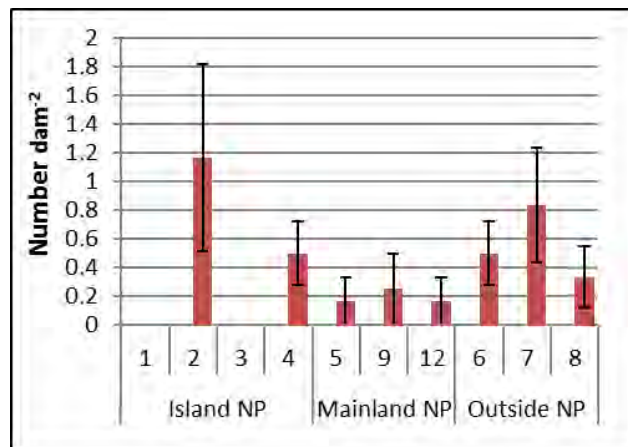


Figure 11. Mean number of COTS dam^{-2} by site in 2016 (\pm SE).

Fishing Gear

The mean number of lost fishing gear pieces in each management area ranged from 0.22–1.38 dam⁻² in 2011 and from 0.13–0.50 dam⁻² in 2016 (Figure 12); and across sites from 0.17–3.00 dam⁻² in 2011 and 0.00–0.83 dam⁻² in 2016 (Figure 13). There was a significant decrease in lost fishing gear between 2011 and 2016 ($F_{1,110}=6.84$, $p=0.01$ area level; $F_{1,96}=14.22$, $p<0.01$ site level). A significant difference in the direction of change over time among both management areas and sites was shown ($F_{2,110}=5.57$, $p<0.01$; $F_{9,96}=4.37$, $p<0.01$), with a dramatic decrease in the Island NP area, some decrease in the Mainland NP area and an increase in the Outside NP area. A decrease was observed at most sites within the national park, while an increase was observed at most sites outside of the park—the exceptions being sites 12 and 8, where there was no change detected. Sites 1, 4 and 9, all within BBNP, showed the greatest decrease in fishing gear (Figure 13).

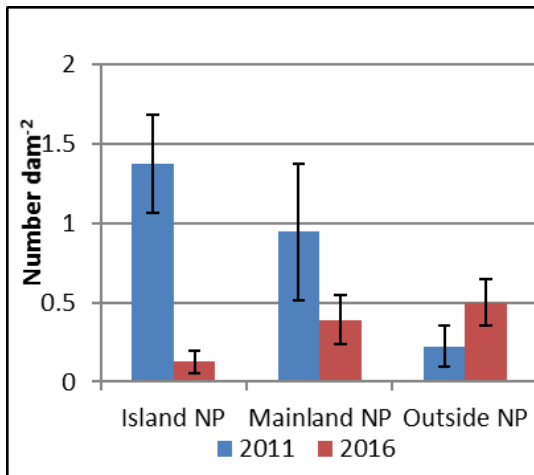


Figure 12. Mean number of lost fishing gear pieces dam⁻² by management area (\pm SE).

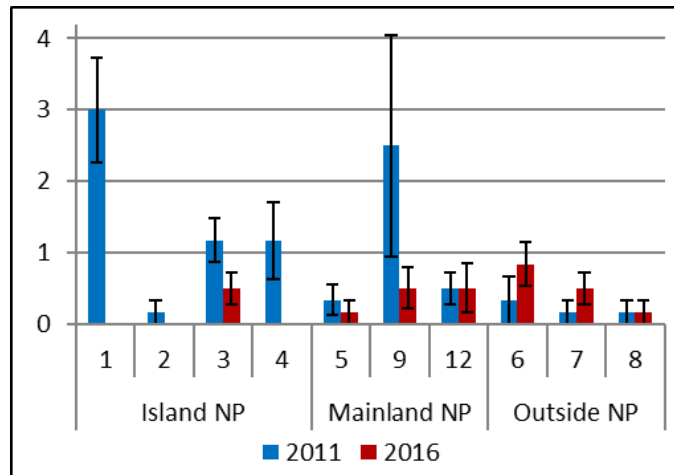


Figure 13. Mean number of lost fishing gear pieces dam⁻² by site (\pm SE).

Fish Abundance

Fish abundance increased significantly across management areas, from 12,466–41,812 ha⁻¹ in 2011 to 36,462–56,173 ha⁻¹ in 2016 ($F_{1,110}=29.19$, $p<0.01$, Figure 14); and across sites from 3,076–45,826 ha⁻¹ to

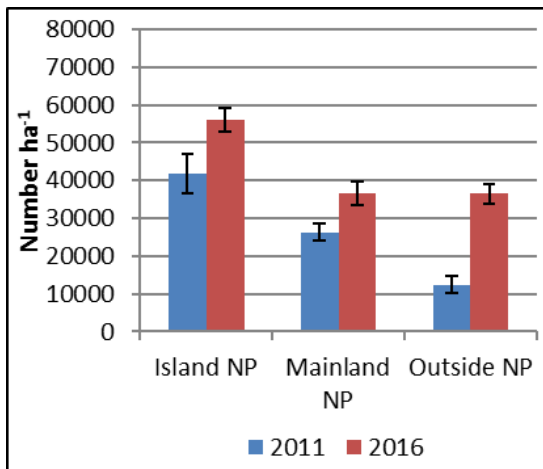


Figure 14. Mean number of fish ha⁻¹ by management area (\pm SE).

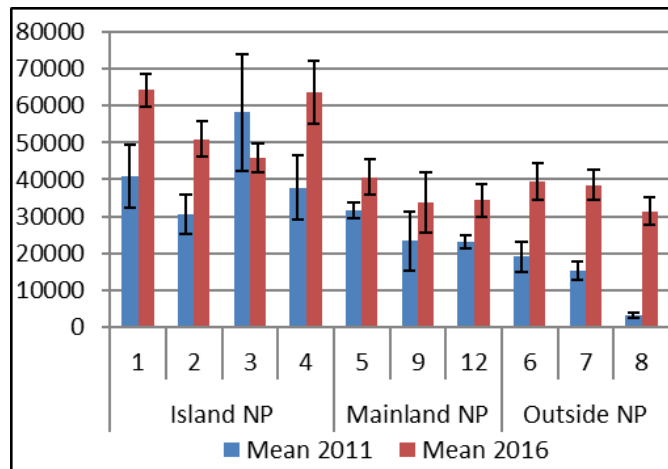


Figure 15. Mean number of fish ha⁻¹ by site (\pm SE).

31,376–64,236 ha⁻¹ ($F_{1,96}=31.50$, $p<0.01$, Figure 15). All areas had a similar increase in fish abundance with no significant difference in the direction of change among management areas or sites observed ($F_{2,110}=1.72$, $p=0.18$; $F_{9,96}=1.85$, $p=0.07$). Fish abundance outside of the BBNP area more than doubled (Figure 14) while Site 3, at Menjangan Island, yielded the only recorded drop, albeit not statistically significant.

Overall Fish Biomass

Fish biomass ranged from 247–780 kg ha⁻¹ in 2011 and 433–713 kg ha⁻¹ in 2016, (Figure 16). Mean fish biomass varied across sites from 215–987 kg ha⁻¹ in 2011 and from 313–2,396 kg ha⁻¹ in 2016 (Figure 17). Biomass changed significantly from 2011 to 2016 ($F_{1,110}=18.14$, $p<0.01$ at the area level; $F_{1,96}=27.85$, $p<0.01$ at the site level). The change among management areas and sites was significantly different ($F_{2,110}=11.67$, $p<0.01$; $F_{9,96}=5.73$, $p<0.01$) with biomass more than doubling in the Island NP and Outside NP areas while remaining constant in the Mainland NP area. A significant increase in biomass from 2011 to 2016 was found at sites 1 ($F_{1,10}=6.48$, $p=0.03$), 4 ($F_{1,10}=26.50$, $p<0.01$), 7 ($F_{1,10}=8.44$, $p=0.02$), and 8 ($F_{1,10}=5.67$, $p=0.04$), with no significant change at other sites (Figure 17).

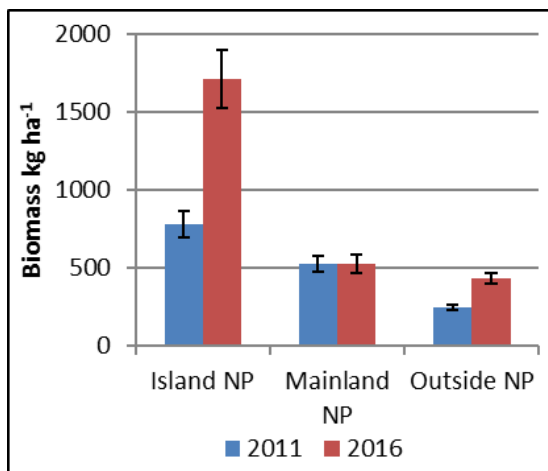


Figure 16. Mean biomass of fish by management area (kg ha⁻¹, ±SE).

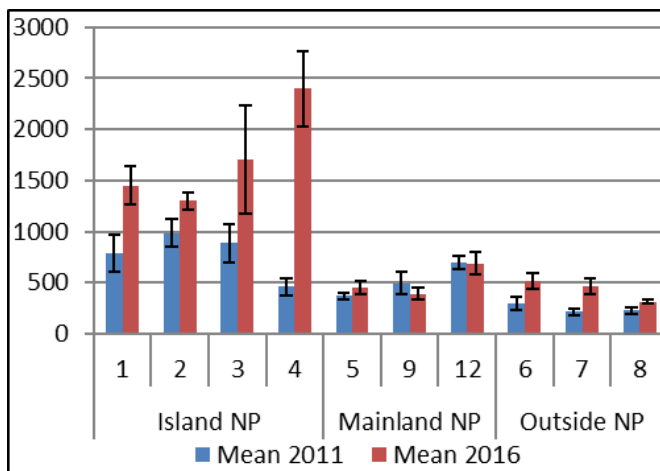


Figure 17. Mean biomass of fish by site (kg ha⁻¹, ±SE).

Biomass of Herbivorous Fish

The mean biomass of herbivores in management areas ranged from 56–138 kg ha⁻¹ in 2011 and 117–604 kg ha⁻¹ in 2016 (Figure 18). The mean biomass of herbivores at sites ranged from 32–261 kg ha⁻¹ in 2011 and 59–839 kg ha⁻¹ in 2016.

Herbivore biomass increased significantly from 2011 to 2016 ($F_{1,110}=19.25$, $p<0.01$ at the area level; $F_{1,96}=27.33$, $p<0.01$ at the site level). There was a significant difference in the change among management areas and sites ($F_{2,110}=14.29$, $p<0.01$; $F_{9,96}=4.18$, $p<0.01$), with herbivorous fish biomass more than quadrupling in the Island NP area and more than doubling in the Outside NP area, and minimal change observed at the Mainland NP area. A significant increase in herbivore biomass occurred at sites 1 ($F_{1,10}=19.27$, $p<0.01$), 2 ($F_{1,10}=22.38$, $p<0.01$), 4 ($F_{1,10}=12.68$, $p<0.01$), 5 ($F_{1,10}=5.09$, $p=0.05$) and 7 ($F_{1,10}=6.05$, $p=0.03$), while sites 9 and 12 had lower, though not significantly, herbivore biomass (Figure 19).

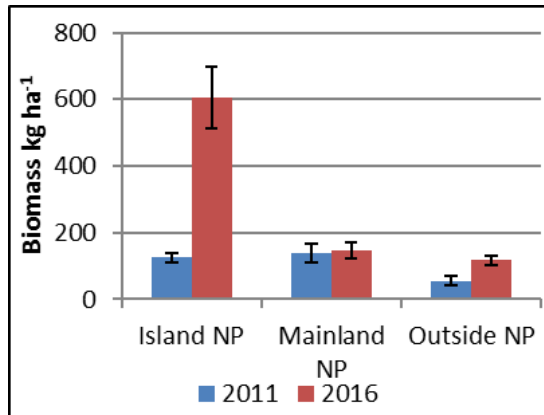


Figure 18. Mean biomass of herbivorous fish by management area ($\text{kg ha}^{-1} \pm \text{SE}$).

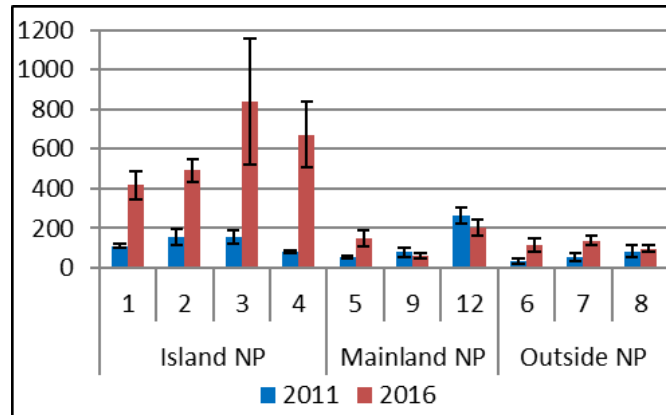


Figure 19. Mean biomass of herbivorous fish by site ($\text{kg ha}^{-1} \pm \text{SE}$).

Biomass of Predatory Fish

The mean biomass of the predatory fish families Carangidae (trevally), Lethrinidae (emperor), Lutjanidae (snapper) and Serranidae (grouper) varied across management areas from 16–82 kg ha^{-1} in 2011 and 23–102 kg ha^{-1} in 2016 (Figure 20). Mean biomass of the select predatory fish families varied across sites from 4–171 kg ha^{-1} in 2011 and 11–157 kg ha^{-1} in 2016 (Figure 21). There was no significant change from 2011 to 2016 ($F_{1,110}=0.001$, $p=0.97$ at the area level; $F_{1,96}=0.06$, $p=0.81$ at the site level). Predatory fish decreased in the Mainland NP area, contrasting with their increase in other areas, but this was not statistically significant among areas or sites ($F_{2,110}=1.66$, $p=0.20$; $F_{9,96}=0.45$, $p=0.90$).

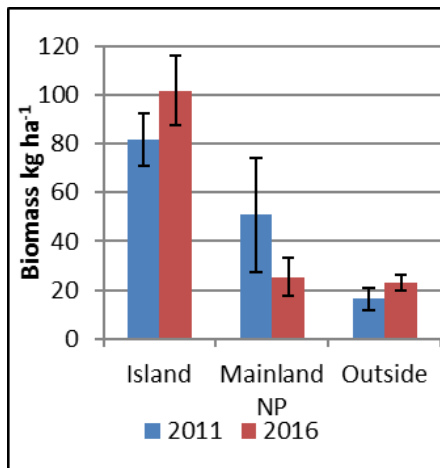


Figure 20. Mean biomass of main predatory fish families by management area ($\text{kg ha}^{-1} \pm \text{SE}$).

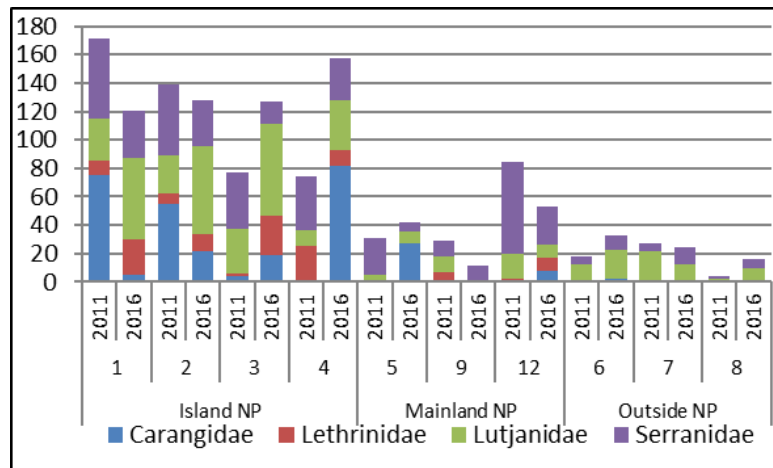


Figure 21. Mean biomass of main predatory fish families by site and family (kg ha^{-1}).

Water Temperature

Seawater temperatures at Site 1 (NW Corner) at Menjangan Island ranged from 25.6°C to 32.2°C from June 1st, 2013 through May 31st, 2017, generally following distinct seasonal trends of the Southern Hemisphere (Figure 22). Seawater temperature is coolest in July–August and warmest in November–December. However, during the El Niño of 2015–2016, water temperatures dipped almost a full degree lower than previous years in September, to 25.6°C, then began warming to 32.2°C, the maximum temperature observed over the four-year record. Degree Heating Weeks increased during this period to a

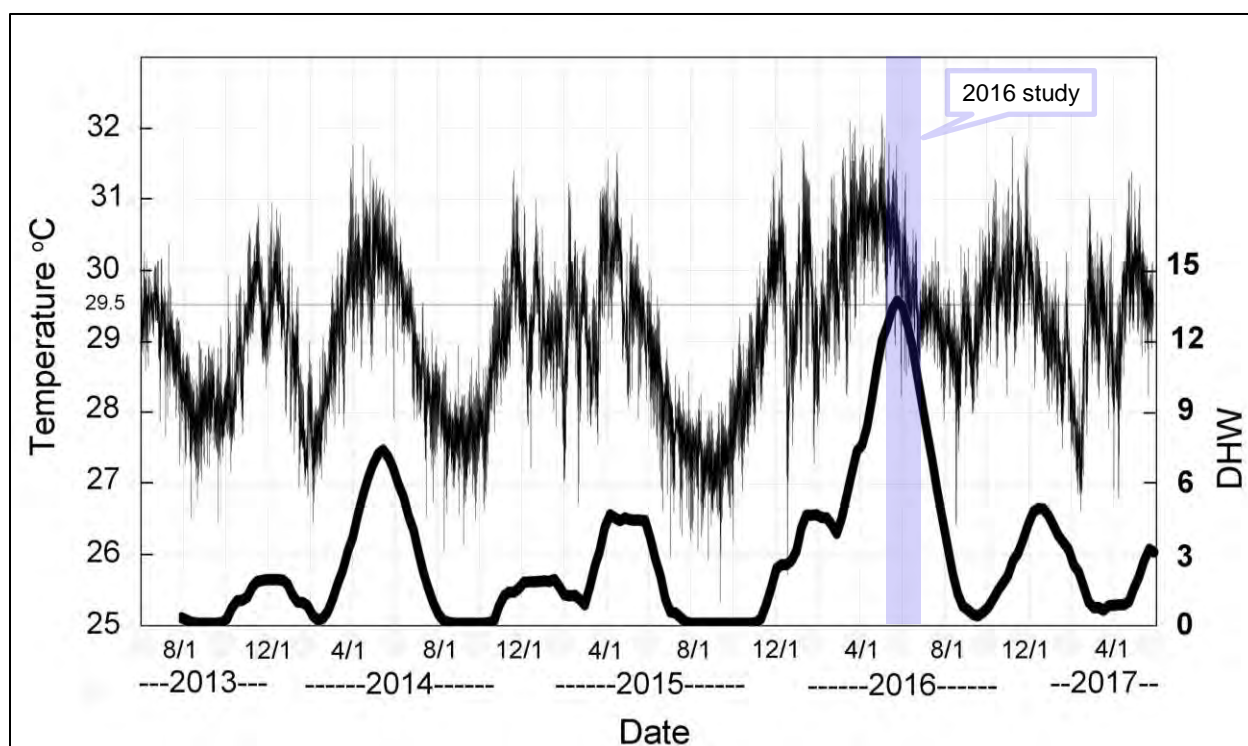


Figure 22. Temperature (thin line) and Degree Heating Weeks (DHW; thick line) at Site 1 (NW Corner), Menjangan Island, June 2013 through May 2017.

maximum of 12.6 in June 2016 with less cooling after the peak than previous years, making it the hottest period from 2013 to 2017.

The lower (thick) line of Figure 22 depicts Degree Heating Weeks (DHW), a measure of cumulative temperature stress in the previous 12-week period (Gleason and Strong, 1995). The integral of thermal heating above the threshold temperature of 29.5°C for the duration of a heating period revealed that the seasonal sum of thermal stress in 2016 was almost three times as high as previous years' (Table 3).

Table 3. Sum of degree heating weeks for seasonal periods June 2013 to May 2017 for threshold temperatures of 29.5°C.

Year	Annual DHW (sum)
2013	2.08
2014	8.89
2015	6.86
2016	21.94
2017	7.49

Summary of Results

Table 4 summarizes the change in reef indicators since 2011, with shades of red and green illustrating negative and positive ecological outcomes, respectively.

Table 4. Summary of results (Significance of change calculated with One Factor ANOVA).

Colors: ■ Significant improvement ■ Not significant improvement
■ Significant degradation ■ Not significant degradation

		Substrate and Damage					Fish Biomass		
Management Area	Site	Coral Cover	Pre-Bleaching Cover, Inferred (Recently Dead + Live Coral)	Broken Corals	Macroalgae	Fishing Gear	Overall	Predators	Herbivores
Island NP	1	Decrease p<0.01	Increase p=0.90	Increase p=0.26	Decrease p=0.43	Decrease p=0.02	Increase p=0.03	Decrease p=0.64	Increase p<0.01
	2	Decrease p=0.12	Decrease p=0.27	Increase p=0.10	Increase p=0.94	Decrease p=0.34	Increase p=0.80	Decrease p=0.85	Increase p<0.01
	3	Decrease p=0.19	Decrease p=0.64	Decrease p=0.06	None present	Decrease p=0.11	Increase p=0.18	Increase p=0.37	Increase p=0.06
	4	Decrease p<0.01	Decrease p<0.01	Increase p=0.60	Increase p=0.17	Decrease p=0.06	Increase p<0.01	Increase p=0.10	Increase p<0.01
Mainland NP	5	Decrease p<0.01	Decrease p=0.01	Increase p<0.01	Decrease p=0.18	Decrease p=0.55	Increase p=0.30	Increase p=0.74	Increase p=0.05
	9	Decrease p=0.07	Decrease p=0.61	Decrease p=0.67	Increase p=0.21	Decrease p=0.25	Decrease p=0.43	Decrease p=0.28	Decrease p=0.50
	12	Decrease p=0.43	Decrease p=0.78	Increase p=0.05	Increase p=0.26	Same p=1	Decrease p=0.95	Decrease p=0.62	Decrease p=0.32
Outside NP	6	Decrease p=0.08	Decrease p=0.20	Increase p=0.04	Increase p=0.08	Increase p=0.30	Increase p=0.05	Increase p=0.14	Increase p=0.05
	7	Decrease p<0.01	Decrease p=0.25	Increase p=0.40	Decrease p=0.77	Increase p=0.26	Increase p=0.02	Decrease p=0.82	Increase p=0.03
	8	Decrease p=0.14	Increase p=0.81	Decrease p=0.08	Increase p=0.76	Same p=1	Increase p=0.04	Increase p=0.10	Increase p=0.66

DISCUSSION

Accumulation of Stress as Reflected by Coral Cover

Between 2011 and 2016, the reefs of NW Bali lost 44.4% of their living coral cover, declining in overall cover from 36% to 20%. This value was similar to the overall mean cover of 22% observed in 2002 by WCS (Doherty et al., 2013). Other reef studies around Menjangan Island since 1996 have reported live coral cover between 1% and 74%, though methods and exact locations have varied, and reductions in cover have been attributed to a variety of factors including COTS infestations, coral mining, anchoring, damage by snorkelers and divers and mass bleaching (Table 5). Field observations in May–June of 2017 revealed wider scale mortality than captured in this study in 2016 because the thermal stress continued after transect data collection had concluded (Figure 22). A photographic temporal series of the same or similar sections of reef is included in Appendix 3 (plates 1, 2 and 3) to help visualize the magnitude of loss.

Table 5. Coral cover reported in previous studies in the Menjangan Island area.

Year	Coral Cover	Reference
1996	12–28% at 3 m 7–14% at 10 m	Wijonarno, 1999
1998	3–5% at 3 m 1–9% at 10 m	Wijonarno, 1999
2008	46%	Yudasmaro, 2010
2010	1–73.7%	Dustan et al., 2013
2011	36%	Doherty et al., 2013
2011	35%	Turak and DeVantier, 2013
2012	38%	Hernowo et al., 2013

The minimal live coral cover in 1998 was attributed to a COTS outbreak in 1997 (Wijonarno, 1999). Our pairwise comparisons of the same sites suggest that the reefs had begun to regrow following the strong COTS infestation, adding 14% between 2002 and 2011, only to be reduced again by severe thermal stress in 2016.

Despite the major decline in coral cover from 2011–2016, the presence of specific coral genera remained relatively unchanged. While a stronger comparison of coral biodiversity over time would have considered species abundance as well as presence, our genus taxonomic-level observations suggest that community structure did not collapse concomitantly with cover. Studies with stream invertebrates have suggested that taxonomic sufficiency can be achieved with even family-level identification as the number of spatial and temporal replicates may provide more information than taxonomic detail (Mueller et al., 2013). This may be especially relevant when working underwater in remote field locations where times and resources are precious.

Biodiversity at many scales has been proposed as a cornerstone of ecological resilience, a concept that broadly refers to the stability of an ecosystem and its ability to return to a pre-disturbance state following perturbation (Nyström et al., 2008). NW Bali’s history of fluctuating coral cover yet apparently stable richness of genera in recent years suggests that these reefs still possess enough resilience to recover—at least to some extent—following severe acute disturbances. Reef recovery following disturbance has been well documented at other sites. For example, a remote reef in western Australia was found to have regained 40% of its pre-bleaching cover in six years and 80% in 12 years after an 80% decline in live coral cover during the 1998 bleaching event (Smith, 2008; Gilmore, 2013). Similarly, coral cover returned to pre-bleaching levels in less than a decade in Moorea, French Polynesia, although the composition of coral genera changed (Adjeroud, 2009). A structural shift favoring massive and encrusting lifeforms was reported in Okinawa following the same bleaching event (Loya et al., 2001) while on the Great Barrier Reef the severity of thermal stress and the interval between bleaching events is shifting the community structure towards more rapidly colonizing species (Hughes et al., 2017). Both history and experiences documented elsewhere suggest that recovery of NW Bali’s reefs is possible following the decline in cover observed between 2011 and 2016 if the frequency of severe bleaching events does not increase.

Mortality from prolonged bleaching—see discussion, below—probably drove the decline in live coral cover from 2011–2016, mostly between 2015 and 2016 (Appendix 3 Plate 4). However, the magnitude of other impacts can be inferred by parsing the effects of bleaching-induced mortality. If it is assumed that coral scored as *recently dead* was living just prior to the bleaching event, the sum of the percentage of *recently dead*, *live* (not bleached) and *live bleached* coral at all sites can be used to estimate the amount of

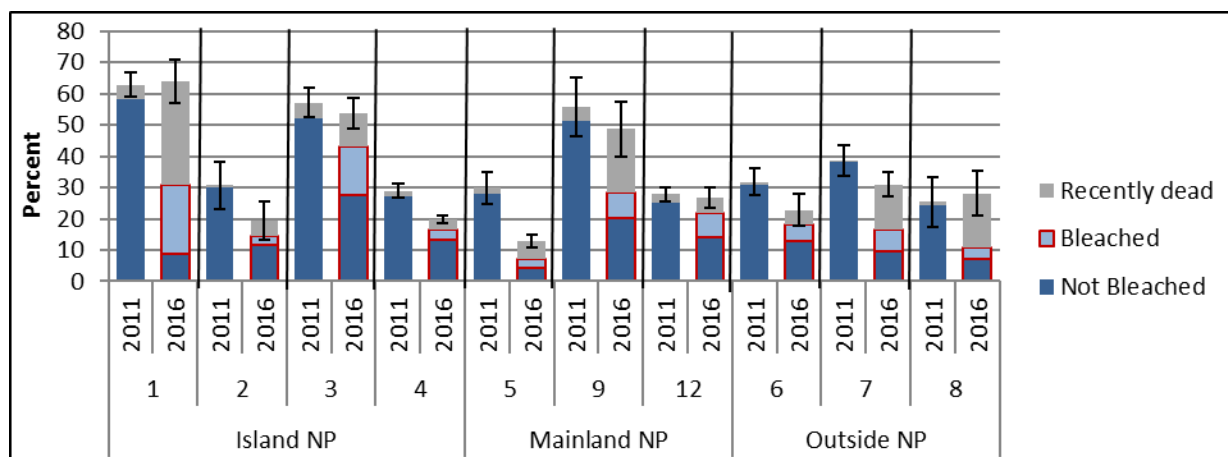


Figure 23. The sum of the mean percentage of *recently dead*, *live* (not bleached) and *live bleached* coral cover, which provides an estimate of pre-2016 bleaching live coral cover.

live coral present in early 2016 or late 2015, just prior to the bleaching event (Figure 23). Under this scenario, only Sites 4 (Island NP) and 5 (Mainland NP) show a significant decrease in cover in since 2011 (Site 4, $F_{1,10}=11.59$, $p<0.01$; Site 5, $F_{1,10}=9.31$, $p=0.012$).

This analysis suggests that other stressors, in addition to bleaching, have driven mortality since 2011 at sites 4 and 5. The area around Site 4 is the most popular destination for Menjangan Island visitors, both marine tourists and Hindu worshippers who disembark here to walk to the nearby temples. Facilities in this area include a jetty in disrepair and one of four sites on the island with public toilets. Anchoring on the reef is a common practice at this site (Appendix 3 Plate 5), as its bathymetry—shallow and rocky until a steep drop-off—is not conducive to moorings, and the jetty no longer provides a safe place to secure boats. Waders, snorkelers, and SCUBA divers are often observed standing on the reef in this area. Site 5 is known to be a popular location for local fishers, because it is far from BBNP officials' observation (see Structural Damage to Live Coral section, below).

We found very few instances of disease and suspect that diseased corals died early in the bleaching event. However, these reefs will be highly susceptible to disease in the near future, as disease outbreaks have been observed elsewhere following periods of thermal stress (Bruno et al., 2007, Harvell et al., 2007).

Thermal Stress as Reflected by Stony Coral Bleaching

We attribute most of the decrease in live coral cover since 2011 to mortality caused by the 2016 bleaching event. Abnormally elevated temperatures from February–June 2016, combined with the significantly higher amount of recently dead coral observed in 2016 compared to 2011, support this hypothesis (Figure 23). Although some sites show variation in the percentage of bleached live hard coral, the management area means are very similar, 34.6%–36.9%. Casual observations at most of these sites and others made during annual visits in May–June between 2011 and 2017 revealed noticeable bleaching in 2014 when DHW peaked at 7.0 and sparse bleaching in 2015 and 2017 when DHW values were 4.5 and 4.6 respectively (Figure 22).

Lifeform, coral taxa, bleaching history, exposure to nutrients and differences in water temperature due to water movement, sun exposure and depth can affect a coral's susceptibility to bleaching (Marshall and Baird, 2000; Brown and Dunne, 2016; Vega Thurber et al., 2014; West and Salm, 2003). The mean percentage of coral bleaching ranged from a minimum of 13.6% at Site 2 to a maximum of 72.0% at Site 1 (Figure 6). The variation between these sites can be mostly attributed to bleaching susceptibility in different coral species, genus and lifeform, history and location of the corals. For example, Site 1 had branching *Porites* as the predominant coral cover in 2011 and 2016. In 2016 at Site 1 (site of highest bleaching) the branching coral was most bleached, especially branching *Porites*. However, in 2016 hardly

any live branching coral existed at Site 2 (site of lowest bleaching), not even recently dead. It had died before the bleaching event, despite the fact that branching *Porites* dominated this site's coral cover in 2011, hence the lesser bleaching at that site. Branching coral has elsewhere has been found to be more susceptible to bleaching than massive and encrusting coral (Loya et al., 2001, Hughes et al., 2017). A further detailed study of currents, nutrients and waterflow around these reef sites may explain the variation of bleaching and threat at these sites.

Structural Damage to Live Coral

The trend of increasing broken and upturned live coral observed from 2002 through 2011 by Doherty et al. (Doherty et al., 2013) has continued through 2016 across all areas as live coral has decreased. At the Mainland NP sites, Site 5 showed the greatest increase in broken and upturned live corals, followed by Site 12. Both these sites receive relatively few tourists from BBNP, but we have seen motorized boats coming from nearby Java that anchor on the shallows of these reefs, carrying approximately 15–25 fully clothed, life-jacketed tourists. Additionally, we removed entangling monofilament nets and fishing gear from live corals on these reefs in 2014–2015, so this damage is likely caused by local fishing practices, such as careless boat anchoring and destructive fishing methods, and non-sanctioned tourism. These areas are not patrolled by BBNP officials, but there have been unsubstantiated reports of coral poaching of commercially important, rare species along the coastline near Site 5.

Outside BBNP, Site 6 showed the highest number of broken and upturned live corals. Here we witnessed local small boats anchored on the reef, carrying air compressors that supported fishers diving with hookah rigs. On at least one occasion, a diving fisher was observed to be using cyanide, though the poison is ostensibly illegal in Indonesia. Additionally, in this area we observed fishers walking out on the reef into chest-deep water and standing on corals to fish. Near sites 6 and 7 we watched fishers who threw rocks onto the reef to frighten fish into nets, and we found large, net-covered stones or “pounders” in the vicinity of Site 9.

Snorkeling and SCUBA diving activities have been correlated with reef structural damage at other tourist destinations (Lamb et al., 2014). Reef trampling has been shown not only to damage fragile and robust species, alike, but also to reduce the colonies' growth rate (Rodgers et al., 2003). Whether a result of trampling, anchoring or fishing, the broken coral fragments wash around in the surge, roll downslope and over the shelf edge where they smother and break corals and associated reef organisms on the reef wall.

Macroalgal Cover

Macroalgae and Nutrient Pollution

There was no statistically significant difference in macroalgal cover between 2011 and 2016; however, the mean macroalgal cover more than doubled at coastal sites 6 and 9. Increasing macroalgal cover can be attributed to increased nutrients in the water (Lapointe et al., 2004; De'ath and Fabricius, 2010), a decrease in herbivorous fish (Mumby et al., 2006; Adam et al., 2011; Green and Bellwood, 2009) or a combination of both (Burkepile and Hay 2006; Zaneveld et al. 2016).

Site 6 is located less than one kilometer from the effluent outlet of a shrimp farm (Figure 24). The mariculture facility opened in 1995 but was closed from 2005–2012 because of the white spot syndrome virus (I K. Nasa, Pejarakan Village resident, personal communication). The increase in macroalgae observed at Site 6 in 2016 is likely related to the shrimp farm's resumed operations since 2012.

Site 9 is located at the mouth of Banyuwedang Bay, which collects runoff from nearby Pejarakan Village and its associated watershed of approximately 11 km² (Figure 24). There are no public sewage treatment facilities for the village or anywhere else in NW Bali. The transects at Site 9, close to the jetty of The Menjangan Resort, had particularly dense macroalgal cover (35%). Since 2011, the resort has erected beach cabanas, a restaurant and bathrooms adjacent to this site. To build the facility, a section of mangrove forest was removed, and sand was trucked into the resort to fabricate a beach. Sediment from this operation and subsequent black and grey water discharge onto the reef may support the macroalgal

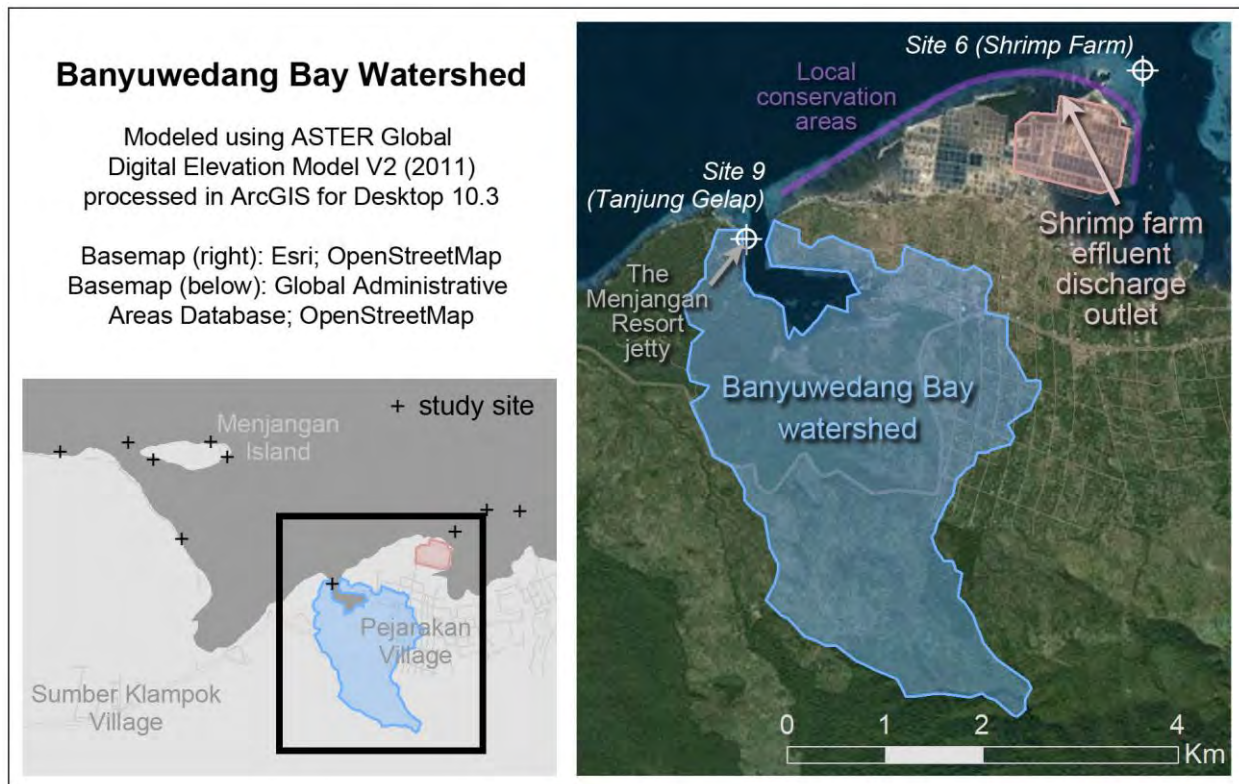


Figure 24. Detail illustrating location of sites 9 and 6 relative to the Banyuwedang Bay watershed, The Menjangan Resort jetty, local conservation areas Putri Menjangan and Alam Lestari, and a shrimp farm.

growth we observed at this site. While further study is needed to confirm whether the water in these sites contain high levels of nutrients, the increased macroalgal cover suggests eutrophication from a variety of local sources.

There are few nutrient inputs around Menjangan Island. However, patches of purple cyanobacteria 2–4 m in diameter were common inshore at Site 2, indicating nutrient loading (Goldberg, 2013). Several toilets are distributed among four locations around the island, including the temples near Site 2. The toilets here appear to drain directly onto the reef without any holding tanks or treatment.

It is especially important to monitor and manage macroalgal overgrowth after this bleaching event, when the coral is weakened and coverage much reduced, which increases the possibility of a phase shift to an algae-dominated regime. The sources of nutrients could be eliminated through wastewater management, practical on a small scale with Wastewater Gardens[®], a system that uses constructed wetlands to treat wastewater (Nelson et al., 2001).

The effects of nutrient pollution go beyond just increased macroalgal growth; nutrient loading from sewage effluent has been directly correlated with an increase in black band and white plague diseases in coral (Kaczmarek et al., 2005), and nutrient discharge has been linked to an increase in COTS populations (Brodie, 2005). Nutrient loading on reefs must be managed directly, as MPAs are not effective in protecting coral from the degradation of water quality from land-based sources of pollution (Lamb et al., 2016).

Macroalgae and Herbivorous Fish

We found no significant relationship between macroalgal cover and herbivorous fish biomass. However, though not statistically significant, Site 9 has the lowest mean biomass of herbivores as well as the greatest algal cover. In this region, local fishers target herbivorous fish for food. Herbivorous fish have been suggested to exert pressure to limit macroalgal growth (Mumby et al., 2006; Adam et al., 2011; Burkepille et al., 2010; Green and Bellwood, 2009), though the cause-and-effect directionality of this relationship has recently been challenged (Russ et al., 2015; Suchley et al., 2016). In Moorea, French

Polynesia Adam et al. (2011) found parrotfish to be especially effective in this role, but Burkepile et al. (2010) and Ceccarelli et al. (2011) suggest that generally, a diverse group of herbivores is necessary to perform different roles in algal removal. According to this view, herbivorous fish are especially important in a reef ecosystem to recover after large disturbances such as bleaching events, large storms, or COTS infestations, as they can limit the faster-growing algae and prevent a phase shift whilst the corals repopulate the reef (Adam et al., 2011; Green and Bellwood, 2009).

Crown-of-Thorns Starfish

The number of COTS observed on transects in 2016 seems low, as the authors frequently observed COTS while diving and snorkeling on reefs in the area, and the FOM team removed thousands between 2015 and 2017 (Figure 25). The numbers found on transects suggest an increase in the population since 2011, when no individuals were recorded while sampling. The belt transect method used here and in 2011 is not optimal for characterizing COTS populations; quadrats that sampled larger areas would probably have been more efficient. COTS are mobile so their numbers can change over time and, since they are nocturnal, daytime surveys are likely to miss hiding animals.

Dynamics of Menjangan Island's COTS population can be further understood using documentation from FOM, which organizes expeditions of 10 or more people to remove COTS when national park guides report frequent sightings. For each expedition, the general location and number of animals removed are recorded, so it is possible to approximately match their recorded locations with this study's numbered sites (Figure 25). On some trips, the number at each location is recorded, but on other trips the total number has been divided by the number of locations visited, giving a mean number per site. A maximum of 2,383 COTS were removed on September 15th, 2015 and a minimum of 55 on May 3rd, 2016.

The COTS removal data, while not collected with rigorous methodological consistency, offer insight into the population dynamics over a finer time scale than the transect data from 2011 and 2016. The number of COTS peaked in September 2015, but by the time of this study, in May 2016, FOM volunteers were finding fewer animals to remove.

The concept of a COTS *outbreak* is difficult to rigorously define (Baird et al., 2013). The total number of COTS removed during the two-year period from January 2015–January 2017 was about 7,500. This represents just over one percent of the approximately 700,000 animals that park rangers and local groups removed from July–September 1997 (Boekshoten, 2000). Bos et al. (2013) describe successful removal as a continuing process, re-checking cleared areas and intensifying efforts during the time just

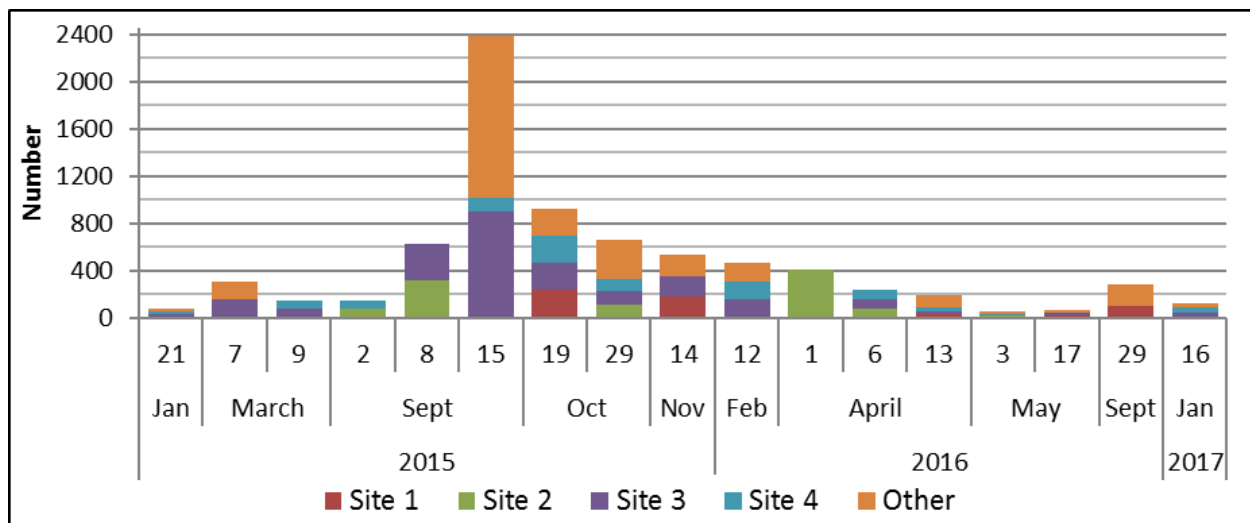


Figure 25. Count of COTS removed from Menjangan Island reef sites by Friends of Menjangan and affiliated groups (source: Friends of Menjangan).

before COTS reproduce, which in the Southern Hemisphere would mean before November. In 2016, the widespread recent coral mortality made it difficult to assess the impact of COTS, but without FOM's COTS removal program the population would be larger.

Birkeland (1990) advanced the hypothesis that COTS outbreaks on high islands may be linked to land development that releases nutrients favorable to the preferred food of COTS larvae, and it has been shown that nutrient discharge is related to increased COTS populations on the Great Barrier Reef (Brodie, 2005; Plass-Johnson et al., 2015; Fabricius et al., 2010). The nutrient loading can be local or brought by currents and upwellings, as can mass larval dispersal, enabling the population to grow (Miller et al., 2015; Wooldridge and Brodie, 2015). This is important in the case of Menjangan because presently, management actions focus on the island, yet currents may carry the larvae back and forth to and from other sites along the mainland. Just outside the national park, managers of a local reef and mangrove conservation area in Pejarakan Village—Nature Conservation Forum Putri Menjangan—have begun to remove COTS from their stretch of reef (Januarsa and Luthfi 2017). Similar efforts take place in Pemuteran Village, a popular tourist destination about 4 km east of Pejarakan along a coastline with an increasing human population.

Harvesting of Marine Organisms

Fishing within 500 m of Menjangan Island's coastline is illegal as specified in the park's Code of Conduct, though artisanal fishing is allowed in some other locations within the park (Mahmud et al., 2015b). Despite this, anecdotal and material evidence, as well as biological surveys, suggest that Menjangan Island has long been a favored fishing spot for locals as well as boats from nearby Madura and Java (Doherty et al., 2013). In 2011, signs of destructive fish bombing were obvious but have decreased since. In 2016, the significant decrease in fishing gear found on transects both at Menjangan Island and the Mainland NP sites may reflect a reduction in fishing effort inside the park. At Menjangan Island, it may also reflect effort by local dive guides and members of FOM to actively remove gear as they find it.

Further supporting the hypothesis that overall fishing effort has decreased at Menjangan Island is the substantial increase in fish biomass observed since 2011 within the Island NP area. The mean biomass of fish around Menjangan Island more than doubled from 2011–2016 while remaining relatively unchanged at the Mainland NP sites. Outside BBNP a smaller increase in biomass was observed. Since 2011, FOM, national park rangers and other local community members have increased their effort to reduce destructive fishing practices and boats from other islands targeting the area around Menjangan Island for fishing. The mainland sites, which have historically been fished more intensively by locals and are not patrolled, have shown no increase in fish biomass. Outside the park the biomass increased somewhat, but the abundance of fish increased most markedly, more than doubling at all sites. The lack of corresponding increase in biomass indicates the increase is attributable to more, smaller fish.

Upon investigation, the smaller-than-expected increase in abundance of fish around Menjangan Island—and especially at Site 3, with a decrease in the number of fish ha⁻¹—is largely attributable to the significant decrease in abundance of *Chromis viridis* observed in 2016 (Table 6). Populations of this

Table 6. Mean abundance ha⁻¹ of *Chromis viridis* in 2011 and 2016.

Site	Island NP				Mainland NP			Outside NP		
	1	2	3	4	5	9	12	6	7	8
2011	4517	7833	11000	6517	583	2825	0	1250	0	0
2016	0	0	67	0	133	950	283	0	0	0

species decreased significantly across all sites from 2011 to 2016 ($F_{1,47} = 7.22$, $p=0.01$). *C. viridis* relies on branching corals for habitat; however, the population declines observed for *C. viridis* did not match the pattern of decline observed in corals across sites, suggesting that other factors—in addition to habitat loss—drove mortality of this species. For example, Menjangan Island's Site 3 showed the greatest population decline of *C. viridis* (Table 6), yet the decline in coral cover at this site, which is dominated by branching *Porities*, was one of the smallest observed in the region (Figure 4).

C. viridis is one of the most favored species in the aquarium fish trade, an active industry in NW Bali for many years. The dramatic population decline of this species from 2011 to 2016—which cannot be explained by habitat loss, alone—suggests that this species has been heavily targeted by collectors inside and outside of the park since 2011. Only a few *C. viridis* were observed at Menjangan Island in 2016, representing a near decimation of the population since 2011 (Table 6). *C. viridis* is the most widely harvested aquarium trade fish with over 900,000 individuals being imported into the US annually, principally from the Philippines and Indonesia (Rhyne et al., 2012). Other popular aquarium fish were not found in sufficient numbers in 2011 or 2016 to make a clear comparison.

Many species of reef fish are taken for food and sold, the most prized being predatory fish: Serranidae (grouper), Lutjanidae (snapper), Lethrinidae (emperor), and Carangidae (trevally). These families do not show an increase in biomass that mirrors the increasing trend in overall fish biomass. The top-down effects of removing these predatory fish may be influencing the whole reef systems (Boaden and Kingsford, 2015). The apparent increase at Site 4 is from a school of Carangidae passing through the transects (including two large *Carnax melampyrgus*), and at Site 3 from five passing Lutjanidae (*Macolor macularis*) (Figure 21). The resident grouper population decreased, suggesting continuing fishing pressure. Only one shark was observed off-transect at Site 4 during the study period.

Herbivorous fish are also found on menus at local restaurants and in the market. Most of the fish are caught as juveniles. The biomass of these fish more than quadrupled around Menjangan Island yet showed a much smaller increase in the Outside NP area and virtually no change at the Mainland NP area—suggesting that fishing pressure at the more accessible sites may be preventing similarly large increases. In addition, it suggests that enforcement of fishing regulations around Menjangan Island may be responsible for the observed population increase.

Overall, the general increase in fish biomass and abundance is a positive sign and may suggest fish stock recovery due to reduced fishing effort at the Island NP and Outside NP sites. Certainly, the local interest in protection combined with the official national park status around the island helps. The markedly smaller increase at Mainland NP sites suggests that the fishing effort at these sites has not decreased. Well enforced no-take zones have been shown not only to benefit fish stocks but also to harbor fewer instances of coral disease and fewer COTS outbreaks (Raymundo et al., 2009).

Marine Management at Bali Barat National Park

Reef resources are central to economic, food, and environmental security of island nations. Indonesia has consistently reported the second-largest marine capture fishery production of any country, behind China, and its territorial waters harbor 16% of the world's coral reefs, second in area only to Australia (Burke et al., 2011). Unfortunately, estimates suggest that 86% of Indonesia's coral reefs face medium or higher threat levels from local human activities, and less than 15% of the country's MPAs are functionally meeting their management objectives (Burke et al., 2012).

At BBNP, activities such as fishing, tourism and business development are regulated through a system of zones. The current system, established in 2010, defines four marine use zones including the Core (*Inti*), Marine Protected (*Perlindungan Bahari*), Utilization (*Pemanfaatan*) and Traditional (*Tradisional*) zones (Mahmud et al., 2015b). Artisanal fishing is generally permitted in the Utilization and Traditional zones and prohibited in the Core and Marine Protected zones. One exception is the waters surrounding Menjangan Island, which, although included in the Utilization Zone, are off-limits to all fishing within 500 m of the island's coastline as specified in the Code of Conduct of the Park (Mahmud et al., 2015b). Despite this, several accounts describe a pattern of daily visitation of tourist boats in the

morning and fishing boats in the afternoon, which has apparently been accepted into the status quo (Andalita, 2006; Mustika et al., 2012; Mahmud et al. 2016).

Tension between fishing, Hindu religious activities and marine recreation at Menjangan Island is longstanding (Mahmud et al., 2015a; Doherty et al., 2013). In 2000, residents of surrounding villages formed the Fishermen's Group of Banyumandi (*Kelompok Nelayan Banyumandi*) in Pejarakan Village and in 2001 the Cultural Governing Body (*Badan Pengelola Adat*) of Sumber Klampok (Mahmud et al., 2015a). The prevailing sense among residents was that the park failed to provide opportunities for their participation in the park's growing tourism economy (Sunarminto, 2002; Yudasmara, 2004). Since then, a growing number of fishers have become boat drivers and tour guides for the park (Doherty et al., 2013). In turn, awareness has grown of the need to preserve the marine resources that draw tourists and bolster the local economy. Consequently, local guides and boat drivers have begun working with park authorities to stop outsiders from illegally fishing inside the park, particularly around Menjangan Island.

Marine patrols by BBNP authorities are conducted on a varying schedule; for example, throughout 2011, officials conducted a total of eight patrols and cited one person for using cyanide poison to collect ornamental fish; and in 2012, six patrols were conducted and 17 people cited for taking wildlife such as live coral, ornamental fish and octopus (Mahmud et al., 2015a). However, enforcement appears to be inconsistent throughout the park. In addition to Menjangan Island, fishing is ostensibly prohibited at Site 5—in an area known locally as Kelor—which is located inside the Core Zone adjacent to the mainland (Semedi, n.d.). We found little evidence to suggest that fishing regulations have recently been enforced at this site, as damage to live coral was pronounced, lost fishing gear was observed both in 2011 and 2016, and the abundance of *C. viridis* dropped significantly from 2011 to 2016 at that site (Table 5).

Locally Managed Marine Conservation Areas

Attention to marine resource protection has become more pronounced within Indonesia's central government, drawing international praise for leading the Coral Triangle region of Southeast Asia in designating MPAs. Indonesia has committed to declaring 200,000 km² of its territorial waters as MPAs by 2020 after achieving its prior target of 100,000 km² in 2008 (White et al., 2014). While the decision to establish BBNP's marine protected area four decades ago demonstrated great foresight, this and other studies (Botemma and Bush, 2012) suggest that management of marine resources has been most effective when surrounding communities are directly involved. Area-based conservation targets, alone, do not appear to improve protection of marine biodiversity (Edgar et al., 2014), and the magnitude of ecological benefits from management appears to depend on staff and budget capacity (Gill et al., 2017). Thus, raising awareness about marine conservation and engaging local peoples in community-wide efforts to steward their marine ecosystems is becoming an essential component to successful marine protection.

The emerging involvement of local reef stewardship groups inside and outside of BBNP is an exemplary case study. FOM was formed to allow people who were not employed by BBNP to steward the reefs of Menjangan Island (Doherty et al., 2013). The group's activities are recorded and subject to BBNP approval. In addition to BBNP and Menjangan Island there are at least three local nature-based groups in Pejarakan Village that engage in ecological restoration activities. Kelompok Alam Lestari began in 2004 to restore a mangrove forest that had been denuded by shrimp and salt farm activities (Nasa, pers. comm.). The mangrove forest, in turn, helps to remove nutrients and provide reef fish habitat. The second, Putri Menjangan, was formed in 2015 to establish a local coral reef and mangrove ecotourism and conservation area adjacent to Kelompok Alam Lestari (Januarsa and Luthfi, 2017). BF supports this group to maintain eight mooring buoys; remove COTS, *Drupella* snails and garbage from their reef; reattach live broken coral onto the reef; and engage in mangrove forest restoration through replanting. The third group, Pokmasta, maintains a small Biorock (Goreau et al., 2003) demonstration in Banyuwedang Bay that was installed in 2016 (<http://www.purprojet.com/project/pejarakan/>).

Additionally, in nearby Pemuteran Village several different groups have been using Biorock to regrow reefs over several decades, which has helped the town gain prominence for its conservation efforts. Reef Seen Divers' Resort maintains a sea turtle hatchling program and rescues adult turtles that have been maimed by fishers. The village's *Pecalang Laut*, a Balinese traditional community-based

maritime security unit funded by local businesses, enforces the ban on destructive fishing practices (Botemma and Bush, 2012). At the time of this writing, there are ongoing discussions amongst these groups to make a community-based, locally managed MPA from BBNP to Pemuteran. Included in these discussions is a recent proposal by BF to initiate a nation-wide coral reef stewardship training program and a NW Bali waste management program with BBNP and other stakeholders.

CONCLUSION

This study has documented changes in the reefs of NW Bali and revealed the complexities of reef management in a time of changing climatology, competing business and environmental concerns and a growing human population. The 44.4% decline in coral cover from 2011–2016 is attributed to a variety of local- to global-scale nested threats such as pollution, destructive fishing and tourism practices, over-exploitation of marine organisms, and most prominently, the acute thermal stress of 2014–2017. Even though the bleaching event of 2014–2017 was not over, it had severely reduced live coral cover at the time of our study.

However, despite the large loss of coral cover, the doubling of fish biomass observed around Menjangan Island suggests that no-take enforcement and collaborative management efforts between the national park and community group FOM are having a positive effect. In areas where these regulations, enforcement and management are not present, increases of similar magnitude were not observed. The efforts of FOM, and more recently Putri Menjangan, to install mooring buoys, remove COTS and the predatory gastropod *Drupella*, and replant broken coral will continue to reduce structural damage and mortality. Simultaneously, the replanting of mangrove forests, watershed reforestation, improved farming practices, and sewage treatment will decrease land-based sources of pollution. While global bleaching events are expected to become more frequent (Hughes et al., 2017), reducing local stressors such as these may enhance the ecological resilience of reefs in the face of future bleaching events (Hoegh-Gulberg et al., 2007; Hughes et al., 2003).

Membership in community-based conservation groups is growing as changes become more apparent—loss of top marine predators, demise of reef beauty and structure, complaints by tourists about garbage and poor toilet facilities, and erosion of coastlines due to mangrove forest destruction and infrastructure development. Some groups are self-funded while other are financially supported by international NGOs, local hotels and other organizations. No longer are stakeholders relying solely on the efforts of BBNP or the government to care for the marine environment and manage trash; the people of NW Bali are moving towards solving problems on their own. Towards this end, education about ocean and ecosystem health and teaching simple yet effective measures can provide working solutions to promote ecosystem resilience in the face of ever-increasing environmental pressures (Hoegh-Gulberg et al., 2007; Hughes et al., 2003).

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APPENDICES

Appendix 1

Table of selected herbivores from IUCN monitoring (Green and Bellwood, 2009).

Family (Tribe)	Common Name	Functional Group	Genera and Species
Labridae (Scarini)	Parrotfishes	Scrapers/small excavators	All <i>Scarus</i> and <i>Hipposcarus</i> All <i>Bolbometopon</i> , <i>Cetoscarus</i> and <i>Chlorurus</i> <35cm SL
		Large excavators/bioeroders	All <i>Bolbometopon</i> , <i>Cetoscarus</i> and <i>Chlorurus</i> >35cm SL
		Browsers	All <i>Calotomus</i> and <i>Leptoscarus</i> species
Acanthuridae	Surgeonfishes	Grazers/detritivores	All species except those that are planktivores (<i>A. albipectoralis</i> , <i>A. mata</i> , <i>A. nubilus</i> , <i>A. thompsoni</i> and <i>P. hepaticus</i>) or detritivores (<i>Ctenochaetus</i> spp.)
	Unicornfishes*	Browsers	All <i>N. brachycentron</i> , <i>N. elegans</i> , <i>N. lituratus</i> , <i>N. tonganus</i> and <i>N. unicornis</i>
			Juveniles (<20cm) of <i>N. annulatus</i> , <i>N. brevirostris</i> , <i>N. maculatus</i> , <i>N. mcdadei</i> , and <i>N. vlamingii</i>
Siganidae	Rabbitfishes	Browsers	<i>S. canaliculatus</i>
		Grazers/detritivores	All other species
Kyphosidae	Rudderfishes	Browsers	All
Ephippidae	Batfishes	Browsers	All
Pomacanthidae	Angelfishes*	Grazers/detritivores	All <i>Centropyge</i> species

Appendix 2

List of coral genera recorded on belt transects in 2011 and 2016.

	2011	2016		2011	2016		2011	2016
<i>Acanthastrea</i> spp.	X	X	<i>Galaxea</i> spp.	X	X	<i>Pavona</i> spp.	X	X
<i>Acropora</i> spp.	X	X	<i>Gardineroseris</i> spp.	X	X	<i>Pectinia</i> spp.	X	X
<i>Alveopora</i> spp.	X	X	<i>Goniastrea</i> spp.	X	X	<i>Physogyra</i> spp.	X	X
<i>Anacropora</i> spp.	X	X	<i>Goniopora</i> spp.	X	X	<i>Platygyra</i> spp.	X	X
<i>Astreopora</i> spp.	X	X	<i>Halomitra</i> spp.	X	X	<i>Plerogyra</i> spp.	X	X
<i>Barabattoia</i> spp.		X	<i>Heliofungia</i> spp.	X		<i>Pocillopora</i> spp.	X	X
<i>Blastomussa</i> spp.		X	<i>Herpolitha</i> spp.	X	X	<i>Podabacia</i> spp.	X	X
<i>Caulastrea</i> spp.	X	X	<i>Hydnophora</i> spp.	X	X	<i>Polyphyllia talpina</i>	X	X
<i>Coeloseris</i> spp.	X	X	<i>Leptastrea</i> spp.	X	X	<i>Porites</i> spp.	X	X
<i>Coscinaeraea</i> spp.	X	X	<i>Leptoria</i> spp.	X	X	<i>Psammocora</i> spp.	X	X
<i>Ctenactis</i> spp.	X	X	<i>Leptoseris</i> spp.	X	X	<i>Pseudosiderastrea</i> spp.	X	X
<i>Cycloseris</i> spp.		X	<i>Lobophyllia</i> spp.	X	X	<i>Sandalolitha</i> spp.	X	X
<i>Cyphastrea</i> spp.	X	X	<i>Lithophyllon</i> spp.	X		<i>Scolymia</i> spp.	X	X
<i>Diploastrea</i> spp.	X	X	<i>Merulina</i> spp.	X	X	<i>Seriatopora</i> spp.	X	X
<i>Echinophyllia</i> spp.	X	X	<i>Montastrea</i> spp.	X	X	<i>Siderastrea</i> spp.	X	X
<i>Echinopora</i> spp.	X	X	<i>Montipora</i> spp.	X	X	<i>Stylocoeniella</i> spp.	X	
<i>Euphyllia</i> spp.	X	X	<i>Mycedium</i> spp.	X	X	<i>Stylophora</i> spp.	X	X
<i>Favia</i> spp.	X	X	<i>Oulophyllia</i> spp.	X	X	<i>Symphyllia</i> spp.	X	X
<i>Favites</i> spp.	X	X	<i>Oxypora</i> spp.	X	X	<i>Turbinaria</i> spp.	X	X
<i>Fungia</i> spp.	X	X	<i>Pachyseris</i> spp.	X	X			

Appendix 3

Photographic documentation of change on NW Bali reefs, 2010–2017 (photographs by P. Dustan).
Video documentation by P. Dustan and L. Wheeler available at
<https://www.youtube.com/watch?v=yxOfLTnPSUo>.



Plate 1. Site 2 (NE Corner), Menjangan Island, ~10 m.



Plate 2. Two examples of different table *Acropora* colonies, one alive in 2011, the other dead in 2017. Note scools of *Chromis viridis* over the living colony but absent from the dead colony. Site 1 (NW Corner), Menjangan Island, ~4 m.



Plate 3. *Porites lutea* temperature monitoring station at Site 1 (NW Corner), Menjangan Island, ~4 m.

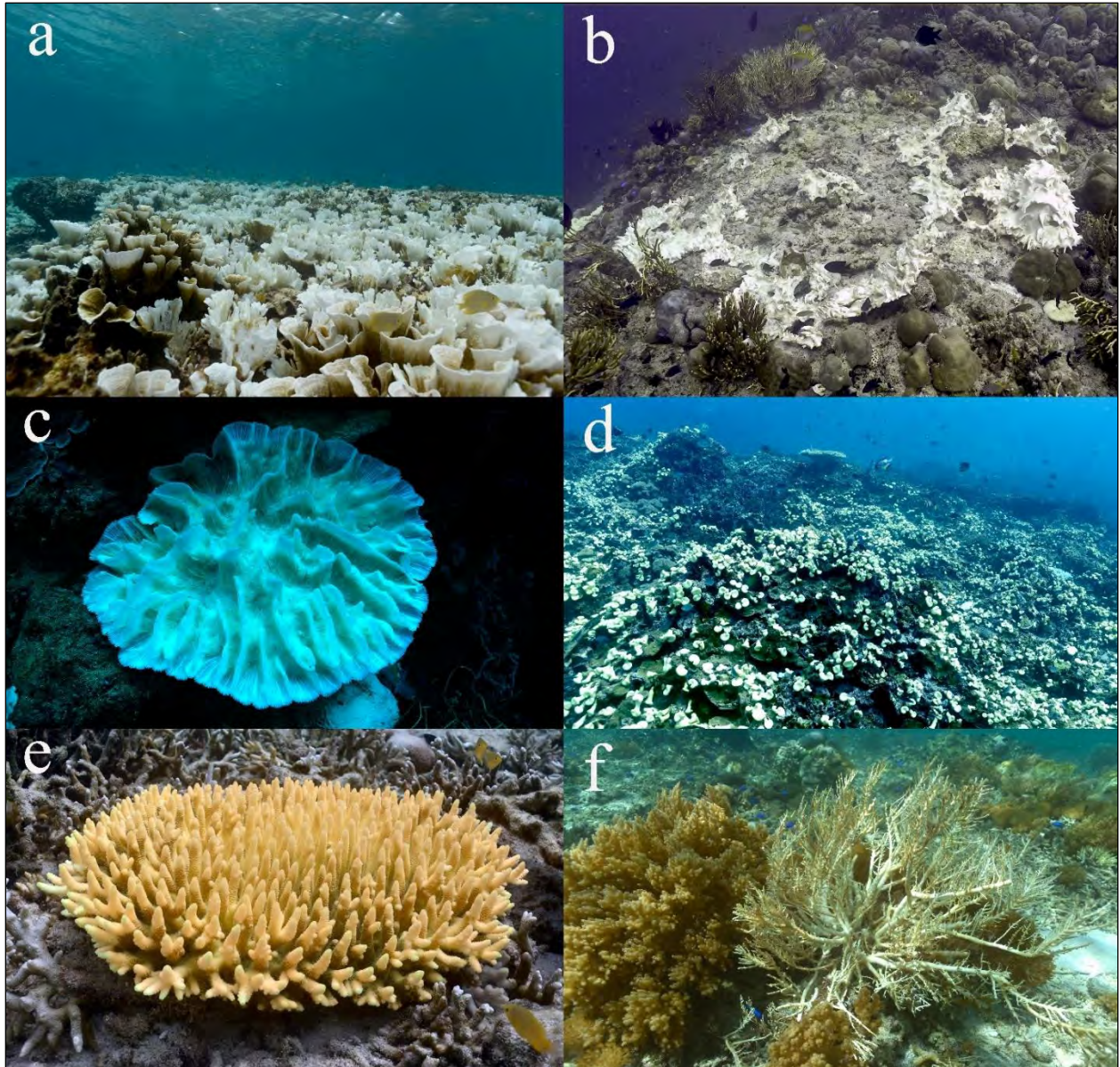


Plate 4. Bleaching stony and soft corals, Bali Barat National Park, NW Bali, 2016; (a) Reef flat monotypic stand of *Echinopora* sp., Symphony Reef (east of Site 1 [NW Corner]), Menjangan Island ~1.5 m; (b) Remains of severely bleached *Sinularia* sp., Site 2 (NE Corner), Menjangan Island, ~10 m; (c) *Pectinia* sp., Site 1 (NW Corner), Menjangan Island, ~20 m; (d) Field of bleaching/dying *Sarcophyton*, Site 2 (NE Corner), ~10 m; (e) *Acropora* sp., (east of Site 1 [NW Corner]), ~1.5 m; (f) *Nephthea* sp., Bali Straits near Site 5 (Kelor), ~4 m.



Plate 5. Local tour boats anchored at Site 4 (Pos 2), Menjangan Island.