

SHORT COMMUNICATION

WILEY

Small no-take areas benefit hard corals more than regulation through fishing permits

Luka Seamus Wright^{1,2} 

¹Danjugan Island Marine Reserve and Sanctuaries, Philippine Reef and Rainforest Conservation Foundation Inc., Bulata, Cauayan, Negros Occidental, Philippines

²Oceans Institute, University of Western Australia, Crawley, Australia

Correspondence

Luka Seamus Wright, Oceans Institute, University of Western Australia, Crawley, WA 6009, Australia.

Email: luka.wright@research.uwa.edu.au

Abstract

1. Situated in the coral triangle, the Philippines hosts some of the world's most diverse coral reefs. Yet there are few national marine protected areas (MPAs) in place that coincide with these reefs. Municipal MPAs partially alleviate this issue but a controlled comparison of the effects of different small-scale management strategies on coral reefs is lacking.
2. Danjugan Island Marine Reserve and Sanctuaries (DIMRS) is one of these community-based MPAs that encompasses a collection of small (0.3–0.34 km²) marine reserves situated in a larger MPA where fishing is regulated through permits. The unique juxtaposition of different management strategies within DIMRS was used as a model system to test whether reserves are better for corals than limited fishing permits by comparing ecosystem indicators (coral cover, bleaching, disease, and morphological diversity).
3. Total and live hard corals were 169% and 204% more abundant inside than outside no-take zones in 2016. This distinction increased between 2002 and 2016 as a result of a more marked decrease in coral cover over time in the partially protected zones. A 70% higher coral community evenness outside the reserves further suggests that scleractinian coral communities on fished reefs are more disturbed.
4. Live coral cover within the MPAs of DIMRS in 2016 (39 ± 4%) is above the current mean for Philippine reefs (23%) and is comparable with the long-term average for reefs situated within Philippine MPAs (36%). This study suggests that exceptionally small reserves may benefit hard corals more than regulation via fishing permits.

KEYWORDS

artisanal fishing, coral conservation, disturbance, growth morphologies, local stressors, macroalgae, marine reserve design, marine reserve size, tropical benthos

This is an open access article under the terms of the [Creative Commons Attribution](https://creativecommons.org/licenses/by/4.0/) License, which permits use, distribution and reproduction in any medium, provided the original work is properly cited.

© 2022 The Author. *Aquatic Conservation: Marine and Freshwater Ecosystems* published by John Wiley & Sons Ltd.

1 | INTRODUCTION

With coral reefs that cover 12,021 km² of its coastal area (White et al., 2014), the Philippines hosts some of the most biodiverse and threatened marine ecosystems in the world (Roberts et al., 2002). Despite the presence of 24 large (>100 km²) national marine protected areas (MPAs) that cover 26,540 km² (Cabral et al., 2014), few MPAs actually overlap with coral reefs (Figure 1a). This issue is partially alleviated by 1,876 small (≤15 km²) municipal MPAs covering an additional 1,061 km² (Cabral et al., 2014). Together with national MPAs, these were estimated to protect 471 km² (4%) of the country's coral reefs (White et al., 2014), although this area has likely increased with the recent establishment of MPAs such as the municipal Cagayancillo Coastal Marine Area, the largest Philippine MPA created to date (>10,000 km²; Cabral et al., 2014). Unfortunately, a recent nationwide survey revealed that present-day Philippine coral reefs support only low (mean ± SEM, 23 ± 1%, *n* = 206) and partially declining (15% of reefs) hard coral cover (Licuanan, Robles & Reyes, 2019). Against this background, stricter municipal coral conservation strategies may be a promising solution by alleviating fishing-induced breakage (Russ et al., 2021) and changes in trophic dynamics (Mumby et al., 2007; Mumby, Hastings & Edwards, 2007; Strain et al., 2019). Hence, understanding the efficacy of small MPAs with different management strategies is of utmost importance.

Despite popular opinion to the contrary, MPA effectiveness is still widely debated (Pendleton et al., 2017). It is generally argued that

large size (>100 km²), long protection (>10 years), good enforcement, and no-take status increase the benefits of MPAs (Edgar et al., 2014). Similarly, old (>10 years), well-enforced, no-take MPAs have been suggested to have higher hard coral cover and lower macroalgal cover (Strain et al., 2019). However, none of these studies investigated MPAs in the Philippines, the country that has the most small MPAs (89% are <1 km²) in the coral triangle (White et al., 2014). A study covering 317 reefs within and adjacent to several of these municipal Philippine MPAs showed that live hard coral cover increased by 3.2% per year in MPAs and did not change outside MPAs over three decades (Magdaong et al., 2014). Importantly, this positive rate of change was independent of MPA size, enforcement status or age. Moreover, some researchers report benefits of no-take protection for Philippine coral reefs (Piquero et al., 2015; Aaron-Amper & Gulayan, 2016), whereas others claim no effect (Abesamis, Russ & Alcala, 2006; Espectado, Napata & Baylon, 2017; Russ et al., 2021) and yet others suggest idiosyncrasy (Panga et al., 2021). Evidently, as with MPAs globally (Pendleton et al., 2017), continued research on the efficacy of Philippine MPAs is required to optimise coral conservation.

One key knowledge gap regarding MPA benefits for corals in the Philippines is a comparison of no-take areas (marine reserves) with areas that allow restricted fishing. Most studies on small Philippine MPAs were traditionally concerned with their effect on fish populations (Russ & Alcala, 1996; Aurellado et al., 2009) and only recently has the focus shifted to effects on the coral habitat

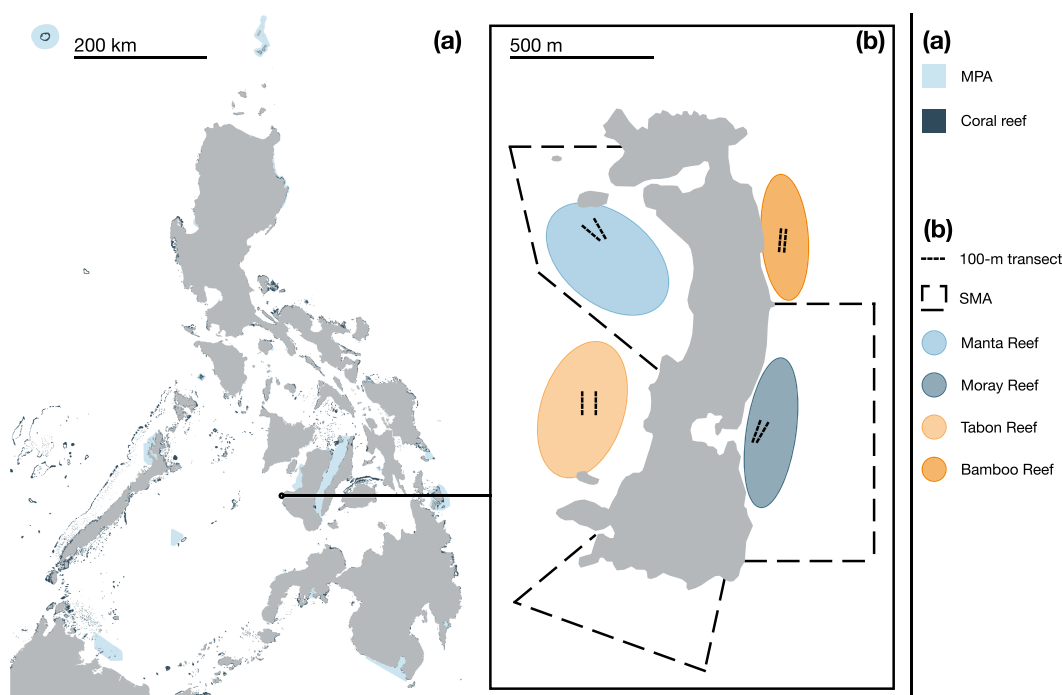


FIGURE 1 Study region. (a) Philippine coral reefs and national marine protected areas. Shapefiles for the land mass, coral reefs, and marine protected areas were sourced from [gadm.org](https://data.gadm.org/), the World Resource Institute (IMaRS et al., 2011), and the World Database on Protected Areas (UNEP-WCMC & IUCN, 2019), respectively. Map projection: WGS 84. (b) Danjungan Island in the context of the Philippines. On each side of the island, coral reefs are located inside (blue) and outside (orange) special management areas (SMAs). Ellipses mark the approximate locations of the four studied reefs and the ellipse shape indicates the reef slope. Maps are oriented north

(Magdaong et al., 2014; Piquero et al., 2015; Aaron-Amper & Gulayan, 2016; Panga et al., 2021). This is surprising considering recent evidence that the effect of hard coral cover on fish (bottom-up) is 24% greater than that of fishing (top-down) (Russ et al., 2021). Moreover, previous research has focused on comparing coral cover in marine reserves with coral cover in completely unprotected sites (Piquero et al., 2015; Aaron-Amper & Gulayan, 2016; Panga et al., 2021; Russ et al., 2021). Studies that have attempted a systematic comparison of different management strategies have not been able to control for confounding factors, and consequently report contradictory results (Magdaong et al., 2014; Strain et al., 2019). Finally, renowned marine reserves have received the highest coverage in the scientific literature, even though they might not be the best model systems. For instance, there are 54 peer-reviewed papers that mention Apo Island's reserve and only two on those surrounding Danjungan Island (Web of Science), despite the better suitability of Danjungan Island for natural experimentation (Figure 1b, cf. Russ et al., 2021) and their comparable geography, size, and coral diversity (Harborne et al., 1996).

This study aims to test the null hypothesis that contrasting municipal management strategies do not differentially influence coral ecosystem integrity, assessed via the relative cover of various coral reef benthos along with hard coral bleaching, disease and morphological diversity. This was achieved by using the unique juxtaposition of marine reserves and zones regulated through fishing permits around Danjungan Island as a model system that allowed for the natural control of potential confounders (Figure 1b).

2 | METHODS

This study was conducted on the coral reefs of Danjungan Island in the barangays of Bulata, Cauayan, Negros Occidental, Philippines (Figure 1). Situated outside any large Philippine MPA (Figure 1a), Danjungan Island is a 0.43-km² limestone island that harbours the three major tropical marine ecosystems: coral reefs, seagrass meadows and mangrove forests. In order to protect these ecosystems from anthropogenic pressures, the non-governmental organization Philippine Reef and Rainforest Conservation Foundation, Inc. (PRRCFI) acquired the island in 1994 and established an MPA surrounding it in 2000 (PRRCFI, 2004), after the initial zoning in 1996 (Harborne et al., 1996).

Danjungan Island has since developed into the successful, community-based ecotourism project Danjungan Island Marine Reserve and Sanctuaries (DIMRS), with three marine reserves, termed special management areas (SMAs), situated within the MPA (Beger et al., 2004) (Figure 1b). Artisanal fishing with gill nets, spears and hooks and lines is controlled via limited permits within the MPA and is prohibited in the SMAs (Harborne et al., 1996; Beger et al., 2004; PRRCFI, 2004). With 278 scleractinian coral species around Danjungan Island and 301 species found in the wider area (Fenner, 1997), DIMRS is likely to harbour some of the most diverse small reefs in the world (Harborne et al., 1996).

Four zones, located within and adjacent to SMA 1 on the west coast and SMA 3 on the east coast were selected within the study site (Figure 1b). For simplicity, the multiple reefs within those four areas were collectively referred to using charismatic names, inspired by the local landmarks Manta Island, Moray Lagoon, Tabon Beach, and Bamboo Bridge. Including the enclosed lagoons, SMAs 1 and 3 cover around 0.3 km² (2.89 km perimeter) and 0.34 km² (2.85 km perimeter), respectively. The unique juxtaposition of SMAs and partially protected reefs on each side of the island enabled a natural experiment, controlling for potential confounders such as proximity to human settlements on the mainland, reef slope, and wave exposure. Although Philippine monsoons generate south-western and north-eastern swells that may somewhat affect corals (Licuanan, Robles & Reyes, 2019), typhoons can seriously damage reefs (Russ et al., 2021). In this context, the steeper eastern reefs of Danjungan Island are sheltered by the bulk of Negros Occidental, whereas the west coast is exposed to the Sulu Sea. However, super typhoons that often pass near the island, such as Rai in 2021, severely impact all reefs (K. Ledesma Trebol, pers. comm.) and therefore partially resolve this east-west contrast.

Data on reef benthos along with abiotic substrata were collected on eight scuba dives between 28 and 29 December 2016. Unfortunately, additional dives were cancelled because of the approaching monsoon. Using the point intercept transect method (Hill & Wilkinson, 2004), substratum (sand, silt, rubble, etc.) and benthos (hard coral, soft coral, macroalgae, etc.) as well as scleractinian coral health (live, dead, diseased, bleached) and morphology (solitary, submassive, corymbose, etc.) were recorded at intervals of 0.5 m along four shallow (2.2–7.4 m depth) and four deep (5–11.3 m depth) 20-m transects at each site (Figure 1b). Transects from both depth categories were pooled for analysis based on statistical similarity. Total hard coral cover was calculated as the sum of live and dead coral, with the latter defined as having structural integrity but lacking polyps. For simplicity, bleached and diseased coral were amalgamated into the single category of discoloured coral.

Following data collection, the percentage cover was calculated from the 41 intercepts recorded within each 20-m transect. The percentage of scleractinian corals with a certain health or morphology characteristic was then calculated by dividing their cover by the total live coral cover. Diversity indices were obtained from hard coral growth morphology data. Besides morphological richness (i.e. the number of growth morphologies), Simpson's evenness index (*E*) (Smith & Wilson, 1996) was calculated as:

$$E = \frac{1}{D \times R},$$

where *D* is Simpson's dominance index (Simpson, 1949) and *R* is the morphological richness.

Historical data on the reef benthos surrounding Danjungan Island were sourced from four surveys carried out in 2002 and 2003 (PRRCFI, 2004). Except for Bamboo Reef, where no historical data are available, these surveys were carried out at a similar depth (6–12 m)

and in the same zones as those in 2016. Unfortunately, only means but no uncertainties around the means were reported. Therefore, historical data are presented as annual averages without confidence intervals. As a precise statistical test of change over time was consequently not possible, change was considered significant if the historical mean was outside the 95% confidence interval around the mean recorded in 2016.

Data analysis and visualization were performed in R 4.1.2 (R Core Team, 2022) within the integrated development environment RSTUDIO 2021.09.2 (RStudio Team, 2022). The effect of the categorical explanatory variable reef (factor with four levels) on the individual continuous response variables was tested with analysis of variance. The fit of gamma and Gaussian distributions to univariate data was compared with the *cdfcomp* and *gofstat* functions in the R package FITDISTRPLUS 1.1–6 (Delignette-Muller & Dutang, 2015). Depending on which fitted better, a simple linear model or a gamma generalized linear model with a logarithmic link function was built. Type-II sums-of-squares tests were performed with the analysis-of-variance *Anova* function of CAR 3.0–12 (Fox & Weisberg, 2019). *F* (Gaussian) or χ^2 (gamma) statistics and *P*-values from univariate omnibus hypothesis tests along with *t*-ratios (effect size/standard error of the effect size) and *P*-values from pairwise tests are reported in Table S1.

After testing the multivariate data for homogeneity of dispersion (Anderson, 2006), permutational multivariate analyses of variance (Anderson, 2001) and similarity percentage analyses (Clarke, 1993) were carried out. Where necessary, data were square-root transformed to avoid heterogeneity of dispersion. These steps were implemented with the *vegdist*, *betadisper*, *permutest*, *adonis*, and *simper* functions of VEGAN 2.5–7 (Oksanen et al., 2020). All multivariate analyses were based on Bray–Curtis distance matrices and run with 9,999 permutations.

Descriptive statistics were calculated with the *describeBy* function of PSYCH 2.1.9 (Revelle, 2020) and visualized with GGLOT2 3.3.5 (Wickham, 2016). In order to plot the multivariate data in two-dimensional space, non-metric multidimensional scaling (Shepard, 1962) was implemented in R with the *metaMDS* function of the VEGAN package and the standalone *veganCovEllipse* function (Oksanen et al., 2014). The map of the Philippine coral reefs (Figure 1a) and the map of the island (Figure 1b) were drawn with QGIS 3.8.3 (qgis.org) and AFFINITY DESIGNER 1.9.1 (serif.com). More detailed information on the outlined data analysis and visualization procedure can be found in the open-access repository at github.com/lukaseamus/DIMRS.

3 | RESULTS

In 2016, the coral reefs around Danjungan Island were dominated by scleractinian corals (mean \pm SEM, $39 \pm 4\%$), rubble ($19 \pm 4\%$), macroalgae ($17 \pm 3\%$), and rock ($9 \pm 1\%$). Less abundant benthos included soft corals ($3 \pm 0.9\%$), sponges ($2 \pm 0.5\%$), and other biota ($0.5 \pm 0.2\%$), such as various anemones and the giant clams *Tridacna crocea* and *Tridacna squamosa*. The four reef zones examined differed in their benthic composition (pseudo $F_{3,28} = 13.59$, $P < 0.001$, $R^2 = 59\%$). Notably, although all reefs were different from each other at the 95% confidence level, the reefs inside the SMAs were more similar to each other than to the fished sites (Figure 2a). Moreover, the variability in benthic composition was much higher for partially protected sites than for fully protected sites (Figure 2a). The abundance of scleractinian coral, macroalgae, and rubble explained 26–36%, 26–29% and 31–32% of the discrepancy in benthic composition between reefs inside and outside the SMAs (Table S2).

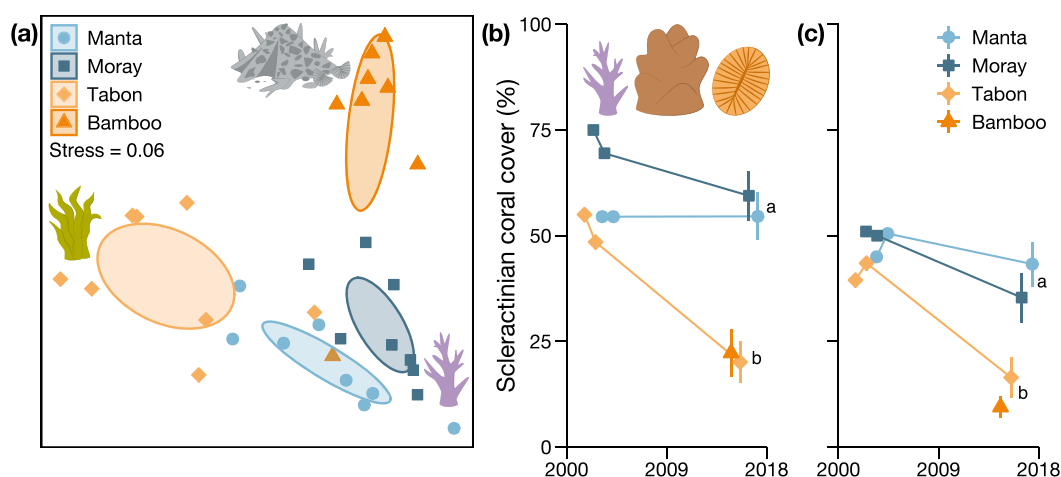


FIGURE 2 Coral reef benthos around Danjungan Island. (a) Benthic composition at fully (blue) and partially (orange) protected coral reefs in 2016. Points represent replicates for each group ($n = 8$), where distance is equivalent to dissimilarity. Ellipses are 95% confidence intervals around group centroids. Separate ellipses suggest dissimilar composition between groups at the 95% confidence level. Icons indicate the approximate locations of benthic categories that drive inter-reef differences (Table S2) in multidimensional space. (b, c) Abundance of total (b) and live (c) scleractinian coral at fully (blue) and partially (orange) protected coral reefs between 2002 and 2016. Point ranges are means \pm SEMs ($n = 8$). Letters denote groups based on similarity at the 95% confidence level. Note that all data points were x-shifted relative to one another to avoid overplotting

The total (Figure 2b) and live (Figure 2c) hard coral cover in 2016 were 169% and 204% higher within SMAs than adjacent to SMAs (Table S1). Historical data suggest that total and live hard coral cover were more similar between sites under different management in the past. From 2002 to 2016, total hard coral cover remained stable at Manta Reef and declined by 21% at Moray Reef, within SMAs (Figure 2b). In contrast, at Tabon Reef, where limited fishing is permitted, the total cover declined by 63% (Figure 2b). Similarly, live hard coral cover remained stable at Manta Reef but declined by 31% at Moray Reef and by 58% at Tabon Reef (Figure 2c). As a result of the stronger decline outside the SMAs, the discrepancy in scleractinian coral cover between fully and partially protected reefs had therefore increased since 2002. In contrast, hard coral health was seemingly unaffected by protection status, with 84% less bleached or diseased coral at sites along the east coast of Danjungan Island (Figure 3a; Table S1).

Macroalgal abundance in 2016 also seemed to be more influenced by the east–west distinction than protection status. With $40 \pm 8\%$ and $17 \pm 4\%$, the shallow western reefs of Tabon and Manta had higher macroalgal cover than the steeper eastern reefs of Moray ($8 \pm 2\%$) and Bamboo ($5 \pm 3\%$) (Figure 3b; Table S1). In fact, the prevalence of bleached or diseased coral, which was also higher at the western reefs with shallow slopes, was positively correlated with macroalgal cover (Pearson's $r_{30} = 0.72$, $P < 0.001$). However, the historical comparison revealed that macroalgal abundance increased by 194% at Tabon Reef adjacent to SMA 1 between 2002 and 2016, whereas it did not change within the SMAs (Figure 3b). Therefore, macroalgae may have benefited from the more marked decline in coral cover or putative decrease in herbivorous fish over 14 years at this reef where restricted fishing is permitted.

The reef zones examined also differed in their morphological composition (Pseudo $F_{3,28} = 5.5$, $P < 0.001$, $R^2 = 37\%$). Notably,

although the reefs of Tabon, Bamboo, and Moray each had a distinct morphological composition at the 95% confidence level, Manta Reef had a similar set of coral growth morphologies to all other reefs (Figure 4a). Manta Reef also had the highest level of morphological richness (Figure 4b). The largest discrepancy in morphological composition, that between Tabon Reef and the two eastern reefs, was explained by the distribution of corymbose (30%), submassive (22–28%) and solitary (18%) growth forms (Table S3). Accordingly, the morphological richness at Tabon Reef stood out by being 34–48% lower than at the other reefs (Figure 4b; Table S1). Despite this lack of any clear separation in morphological composition or richness between partially and fully protected reefs, Simpson's index of morphological evenness was 70% higher outside the SMAs than inside the SMAs (Figure 4c; Table S1).

4 | DISCUSSION

By comparing various ecosystem indicators within a single model system that naturally accounts for confounders, this study provides evidence that reserves are a better small-scale coral conservation strategy than fishing permits. Hard coral abundance was clearly higher within reserves (Figure 2b, c). Furthermore, the higher community evenness at fished reefs (Figure 4c) suggests that these reefs are in fact more disturbed (Dollar, 1982; Darling, McClanahan & Côté, 2013). This may be attributable to direct damage from destructive fishing methods (Russ et al., 2021), such as treading on coral during spearfishing (pers. observ.), and/or indirect effects via fishing-induced trophic cascades (Mumby et al., 2007; Strain et al., 2019) followed by a potential phase shift (Mumby, Hastings & Edwards, 2007). Indirect effects are possible given the lower abundance of herbivorous fish outside the SMAs (PRRCFI, 2004;

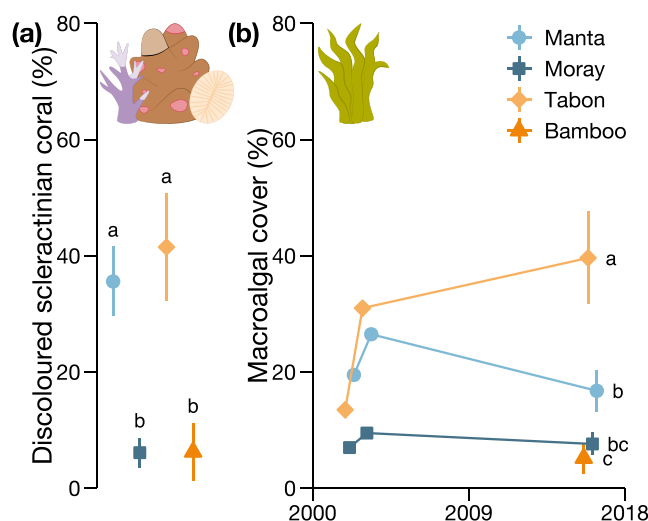


FIGURE 3 Danjungan Island coral reef variables that were not related to protection status. (a) Amount of scleractinian coral that was bleached or diseased at fully (blue) and partially (orange) protected coral reefs in 2016. (b) Abundance of macroalgae at fully (blue) and partially (orange) protected coral reefs between 2002 and 2016. Point ranges are means \pm SEMs ($n = 8$). Letters denote groups based on similarity at the 95% confidence level. Note that all data points were x-shifted relative to one another to avoid overplotting

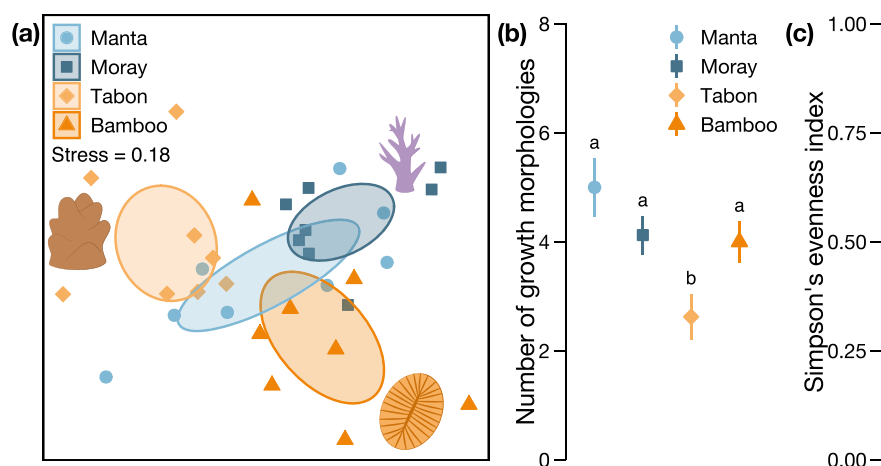


FIGURE 4 Coral reef morphology around Danjungan Island in 2016. (a) Composition of scleractinian coral morphologies at fully (blue) and partially (orange) protected coral reefs. Points represent replicates for each group ($n = 8$), where distance is equivalent to dissimilarity. Ellipses are 95% confidence intervals around group centroids. Overlapping ellipses suggest similar composition between groups at the 95% confidence level. Icons indicate the approximate locations of morphological growth forms that drive inter-reef differences (Table S3) in multidimensional space. (b, c) Morphological richness (b) and evenness (c) of scleractinian corals at fully (blue) and partially (orange) protected coral reefs. Point ranges are means \pm SEMs ($n = 8$). Letters denote groups based on similarity at the 95% confidence level

Aurellado et al., 2009), but are unlikely in light of the lack of evidence for direct competition between hard corals and macroalgae on Indo-Pacific reefs (Roff & Mumby, 2012; Russ et al., 2015). However, trophic cascades that increase corallivore abundance may alternatively reduce coral cover (Rotjan & Lewis, 2008). Selection bias during the initial zoning of reserves has also been suggested to drive dissimilarity between protected and unprotected areas (Myers & Ambrose, 2009). SMA sites were selected based on rare species, species/habitat diversity, and topographical features, but not coral cover (Harborne et al., 1996), so this probably does not explain the current differences. A comparison with historical data, collected shortly after the establishment of the SMAs, further revealed that reefs within and adjacent to marine reserves were more similar 14 years before the present research (Figure 2b, c). Therefore, although the mechanistic drivers behind this reserve effect remain uncertain, it is feasible to reject the null hypothesis and conclude that the hard corals of Danjungan Island benefit more from full protection than from regulation through fishing permits. This is in line with evidence from outside the coral triangle (Strain et al., 2019) but is a novel finding for Philippine coral reefs (Magdaong et al., 2014).

Despite these benefits for coral cover and in contrast to recent findings (Lamb et al., 2016), there was no evidence for improved coral health within the DIMRS marine reserves (Figure 4a, cf. Page et al., 2009). Although ocean warming has impacted Philippine coral reefs fairly indiscriminately in recent decades (Magdaong et al., 2014; Licuanan, Robles & Reyes, 2019; Russ et al., 2021), the reefs of Danjungan Island were relatively unaffected (pers. observ.) by the 2016 global mass bleaching event (Hughes et al., 2018), which may explain this lack of a positive reserve effect on coral health. Unfortunately, the 2020 bleaching event (Pratchett et al., 2021) seems to have had a bigger impact upon DIMRS (K. Ledesma Trebol, pers. comm.), so

future monitoring within this unique model system is essential. Interestingly, despite no influence of protection status, a positive correlation of bleached or diseased coral with macroalgal abundance was observed (Figure 3a, b, cf. Lamb et al., 2016), which could partially explain the variation of coral health around the island. Despite recent anecdotal evidence that super typhoons indiscriminately devastate the corals of Danjungan Island (K. Ledesma Trebol, pers. comm.), the exposed, seaward-facing west coast is likely to be disproportionately affected during typhoons (Russ et al., 2021). This is probably a more parsimonious reason for elevated disease- or bleaching-induced coral discoloration (Lamb et al., 2016) and macroalgal cover (Russ et al., 2015) on that side of the island. Typhoon exposure may therefore explain the 2.24 times higher macroalgal cover on the west coast ($28 \pm 5\%$) compared with the national mean of $9 \pm 1\%$ (Licuanan, Robles & Reyes, 2019), whereas that on the east coast lies below that average ($6 \pm 2\%$).

Several studies on other small Philippine MPAs reported no positive effect of no-take protection on hard corals, even when the control site was unprotected and a stronger effect than the one reported here would be expected (Abesamis, Russ & Alcala, 2006; Espectato, Napata & Baylon, 2017; Panga et al., 2021; Russ et al., 2021). This discrepancy may arise for a variety of reasons. First, Abesamis, Russ & Alcala (2006) tested for linear changes in habitat complexity across zone boundaries, but did not attempt any nonlinear modelling or categorical reserve-control comparison, which could have detected differences overlooked by the chosen analysis. Second, Espectato, Napata & Baylon (2017) report no control values and studied MPAs that were over 93% smaller than the DIMRS SMAs. Third, Panga et al. (2021) acknowledge that there were a variety of site-specific confounders, such as storms and sedimentation. Similarly, the unprotected reefs used as controls by Russ et al. (2021) experience very

different regional stressors to the protected reefs and can therefore hardly be considered controls. Through the unique juxtaposition of fully and partially protected MPA zones around Danjungan Island (Figure 1b), such biases were avoided in the present study.

Overall, the presented data show that, contrary to popular opinion (Edgar et al., 2014), even exceptionally small (0.3–0.34 km²) marine reserves can protect hard corals against local stressors (cf. Magdaong et al., 2014), and to a greater degree than alternative management strategies (cf. Strain et al., 2019). However, global and regional stressors such as climate change and storms (Panga et al., 2021; Russ et al., 2021) seem to impair a potentially regenerative effect, causing reef stasis or decline within marine reserves (Figure 2b, c), and therefore limit these benefits. Nonetheless, if full protection had not been enforced, DIMRS may well have more degraded coral reefs, as can be seen in the case of Tabon Reef (Figures 2b, c and 3b). Moreover, live coral cover within SMAs (39 ± 4%, Figure 2c) is 16% above the current mean for 206 Philippine reefs (Licuanan, Robles & Reyes, 2019) and the long-term average for 214 reefs on the Great Barrier Reef (De'ath et al., 2012) and 3% above the long-term average for 56 protected Philippine reefs (Magdaong et al., 2014) and the current mean for small MPAs on the other side of the Panay Gulf (Espectato, Napata & Baylon, 2017). In addition to being a successful community-based fisheries management project (Beger et al., 2004; Aurellado et al., 2009), DIMRS can thus also be classed as an effective coral sanctuary.

ACKNOWLEDGEMENTS

I carried out this research with personal permission from Gerry Ledesma, Kaila Ledesma Trebol, and Dave Gumban Albao. Thank you for your generosity and assistance. I am especially grateful to Ryman 'Ramram' Gonzaga Armada, Tikyo Pardillo, Carmela Ellaga and Ronald Alvarez Barcenas for helping me with the data collection. No funding was received for this project. Open access publishing facilitated by The University of Western Australia, as part of the Wiley - The University of Western Australia agreement via the Council of Australian University Librarians.

CONFLICT OF INTEREST

No conflict of interest to disclose.

DATA AVAILABILITY STATEMENT

The data and annotated R script for this study can be downloaded from the open-access repository at github.com/lukaseamus/DIMRS. I place no restrictions on data and code availability.

ORCID

Luka Seamus Wright  <https://orcid.org/0000-0002-1273-6256>

REFERENCES

Aaron-Amper, J. & Gulayan, S.J. (2016). Differences in benthic cover and fish assemblage inside and outside marine protected areas. *International Journal of Environmental and Rural Development*, 7(2), 141–146. https://doi.org/10.32115/ijerd.7.2_141

- Abesamis, R.A., Russ, G.R. & Alcala, A.C. (2006). Gradients of abundance of fish across no-take marine reserve boundaries: Evidence from Philippine coral reefs. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 16(4), 349–371. <https://doi.org/10.1002/aqc.730>
- Anderson, M.J. (2001). A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, 26(1), 32–46. <https://doi.org/10.1111/j.1442-9993.2001.01070.pp.x>
- Anderson, M.J. (2006). Distance-based tests for homogeneity of multivariate dispersions. *Biometrics*, 62(1), 245–253. <https://doi.org/10.1111/j.1541-0420.2005.00440.x>
- Aurellado, M.E.B., Hilomen, V.V., Aliño, P.M. & Lumbab, V. (2009). Movement of reef fishes across boundaries of marine protected areas in Danjungan Island, Negros Occidental, Philippines. *Philippine Agricultural Scientist*, 92(1), 66–76.
- Beger, M., Harborne, A.R., Dacles, T.P., Solandt, J.L. & Ledesma, G.L. (2004). A framework of lessons learned from community-based marine reserves and its effectiveness in guiding a new coastal management initiative in the Philippines. *Environmental Management*, 34(6), 786–801. <https://doi.org/10.1007/s00267-004-0149-z>
- Cabral, R.B., Aliño, P.M., Balingit, A.C.M., Alis, C.M., Arceo, H.O., Nañola, C.L., Jr. et al. (2014). The Philippine marine protected area (MPA) database. *Philippine Science Letters*, 7(2), 300–308.
- Clarke, K.R. (1993). Non-parametric multivariate analyses of changes in community structure. *Austral Ecology*, 18(1), 117–143. <https://doi.org/10.1111/j.1442-9993.1993.tb00438.x>
- Darling, E.S., McClanahan, T.R. & Côté, I.M. (2013). Life histories predict coral community disassembly under multiple stressors. *Global Change Biology*, 19(6), 1930–1940. <https://doi.org/10.1111/gcb.12191>
- De'ath, G., Fabricius, K.E., Sweatman, H. & Puotinen, M. (2012). The 27-year decline of coral cover on the Great Barrier Reef and its causes. *Proceedings of the National Academy of Sciences of the United States of America*, 109(44), 17995–17999. <https://doi.org/10.1073/pnas.1208909109>
- Delignette-Muller, M.L. & Dutang, C. (2015). fitdistrplus: An R package for fitting distributions. *Journal of Statistical Software*, 64(4), 1–34. <https://doi.org/10.18637/jss.v064.i04>
- Dollar, S.J. (1982). Wave stress and coral community structure in Hawaii. *Coral Reefs*, 1(2), 71–81. <https://doi.org/10.1007/BF00301688>
- Edgar, G.J., Stuart-Smith, R.D., Willis, T.J., Kininmonth, S., Baker, S.C., Banks, S. et al. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature*, 506(7487), 216–220. <https://doi.org/10.1038/nature13022>
- Espectato, L.N., Napata, R.P. & Baylon, C.C. (2017). Will small MPAs work?: The case of small-sized MPAs in Southern Iloilo, Philippines. *Ocean and Coastal Management*, 139(1), 24–32. <https://doi.org/10.1016/j.ocecoaman.2017.01.024>
- Fenner, D. (1997). *Reef corals of Danjungan area reefs, Philippines*. Bacolod: PRRCFI.
- Fox, J. & Weisberg, S. (2019). *An R companion to applied regression*. Thousand Oaks: Sage.
- Harborne, A.R., Gill, A., Raines, P. & Ridley, J. (1996). *The Danjungan Island marine reserve: CCC summary report*. London: Coral Cay Conservation.
- Hill, J. & Wilkinson, C. (2004). *Methods for ecological monitoring of coral reefs*. Townsville: Australian Institute of Marine Science.
- Hughes, T.P., Anderson, K.D., Connolly, S.R., Heron, S.F., Kerry, J.T., Lough, J.M. et al. (2018). Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. *Science*, 359(6371), 80–83. <https://doi.org/10.1126/science.aan8048>
- IMaRS, IRD, UNEP-WCMC, The WorldFish Center. & WRI. (2011). *Tropical coral reefs of the world (500-m resolution grid)*. Available at: <https://datasets.wri.org/dataset/tropical-coral-reefs-of-the-world-500-m-resolution-grid> [Accessed 13 July 2020]
- Lamb, J.B., Wenger, A.S., Devlin, M.J., Ceccarelli, D.M., Williamson, D.H. & Willis, B.L. (2016). Reserves as tools for alleviating impacts of marine

- disease. *Philosophical Transactions of the Royal Society B*, 371(1689), 20150210. <https://doi.org/10.1098/rstb.2015.0210>
- Licuanan, W.Y., Robles, R. & Reyes, M. (2019). Status and recent trends in coral reefs of the Philippines. *Marine Pollution Bulletin*, 142(1), 544–550. <https://doi.org/10.1016/j.marpolbul.2019.04.013>
- Magdaong, E.T., Fujii, M., Yamano, H., Licuanan, W.Y., Maypa, A., Campos, W.L. et al. (2014). Long-term change in coral cover and the effectiveness of marine protected areas in the Philippines: A meta-analysis. *Hydrobiologia*, 733(1), 5–17. <https://doi.org/10.1007/s10750-013-1720-5>
- Mumby, P.J., Harborne, A.R., Williams, J., Kappel, C.V., Brumbaugh, D.R., Micheli, F. et al. (2007). Trophic cascade facilitates coral recruitment in a marine reserve. *Proceedings of the National Academy of Sciences of the United States of America*, 104(20), 8362–8367. <https://doi.org/10.1073/pnas.0702602104>
- Mumby, P.J., Hastings, A. & Edwards, H.J. (2007). Thresholds and the resilience of Caribbean coral reefs. *Nature*, 450(7166), 98–101. <https://doi.org/10.1038/nature06252>
- Myers, M.R. & Ambrose, R.F. (2009). Differences in benthic cover inside and outside marine protected areas on the Great Barrier Reef: Influence of protection or disturbance history. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 19(7), 736–747. <https://doi.org/10.1002/aqc.1053>
- Oksanen, J., Blanchet, G.F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D. et al. (2014). *veganCovEllipse*. Available at: <https://github.com/jarioksa/vegan/blob/master/R/veganCovEllipse.R> [Accessed 9 Feb 2020]
- Oksanen, J., Blanchet, G.F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D. et al. (2020). *Vegan: Community ecology package*. R package, v2.5-7. Available at: <https://CRAN.R-project.org/package=vegan>
- Page, C.A., Baker, D.M., Harvell, C.D., Golbuu, Y., Raymundo, L., Neale, S.J. et al. (2009). Influence of marine reserves on coral disease prevalence. *Diseases of Aquatic Organisms*, 87(1–2), 135–150. <https://doi.org/10.3354/dao02112>
- Panga, F.M., Anticamara, J.A., Quibilan, M.C.C., Atrigenio, M.P. & Aliño, P. M. (2021). Through the boundaries: Environmental factors affecting reef benthic cover in marine protected areas in the Philippines. *Frontiers in Marine Science*, 8, 702071. <https://doi.org/10.3389/fmars.2021.702071>
- Pendleton, L.H., Ahmadi, G.N., Browman, H.I., Thurstan, R.H., Kaplan, D. M. & Bartolino, V. (2017). Debating the effectiveness of marine protected areas. *ICES Journal of Marine Science*, 75(3), 1156–1159. <https://doi.org/10.1093/icesjms/fsx154>
- Piquero, A.S., Delan, G.G., Rica, R.L.V., Corrales, C.M. & Monte, I.A. (2015). Coral lifeform structure in selected marine protected areas in Southern Cebu, Philippines. *Tropical Technology Journal*, 19(1), 1–4. <https://doi.org/10.7603/s40934-015-0002-4>
- Pratchett, M.S., Heron, S.F., Mellin, C. & Cumming, G.S. (2021). Recurrent mass-bleaching and the potential for ecosystem collapse on Australia's Great Barrier Reef. In: J.G. Canadell, R.B. Jackson (Eds.) *Ecosystem Collapse and Climate Change*. Cham: Springer, pp. 265–289.
- PRRCFI. (2004). Danjungan Island Marine Reserve and Sanctuaries (DIMRS) reef monitoring program. In: UNEP. (Ed.) *Monitoring Coral Reefs for Better Management Schemes: UNEP EAS/RCU Small Grants Fund Programme 2002–2003*. Bangkok: United Nations Environment Programme, pp. 5–31.
- R Core Team. (2022). *R: A language and environment for statistical computing*. Vienna: R Foundation for Statistical Computing.
- Revelle, W. (2020). *Psych: Procedures for psychological, psychometric, and personality research*. R package, v2.0.12. Available at: <https://CRAN.R-project.org/package=psych>
- Roberts, C.M., McClean, C.J., Veron, J.E.N., Hawkins, J.P., Allen, G.R., McAllister, D.E. et al. (2002). Marine biodiversity hotspots and conservation priorities for tropical reefs. *Science*, 295(5558), 1280–1284. <https://doi.org/10.1126/science.1067728>
- Roff, G. & Mumby, P.J. (2012). Global disparity in the resilience of coral reefs. *Trends in Ecology & Evolution*, 27(7), 404–413. <https://doi.org/10.1016/j.tree.2012.04.007>
- Rotjan, R.D. & Lewis, S.M. (2008). Impact of coral predators on tropical reefs. *Marine Ecology Progress Series*, 367(1), 73–91. <https://doi.org/10.3354/meps07531>
- RStudio Team. (2022). *RStudio: Integrated development environment for R*. Boston: RStudio, Inc.
- Russ, G.R. & Alcala, A.C. (1996). Do marine reserves export adult fish biomass? Evidence from Apo Island, central Philippines. *Marine Ecology Progress Series*, 132(1), 1–9. <https://doi.org/10.3354/meps132001>
- Russ, G.R., Questel, S.-L.A., Rizzari, J.R. & Alcala, A.C. (2015). The parrotfish–coral relationship: Refuting the ubiquity of a prevailing paradigm. *Marine Biology*, 162(10), 2029–2045. <https://doi.org/10.1007/s00227-015-2728-3>
- Russ, G.R., Rizzari, J.R., Abesamis, R.A. & Alcala, A.C. (2021). Coral cover a stronger driver of reef fish trophic biomass than fishing. *Ecological Applications*, 31(1), e02224. <https://doi.org/10.1002/eap.2224>
- Shepard, R.N. (1962). The analysis of proximities: Multidimensional scaling with an unknown distance function. I. *Psychometrika*, 27(2), 125–140. <https://doi.org/10.1007/BF02289630>
- Simpson, E.H. (1949). Measurement of diversity. *Nature*, 163(4148), 688. <https://doi.org/10.1038/163688a0>
- Smith, B. & Wilson, J.B. (1996). A consumer's guide to evenness indices. *Oikos*, 76(1), 70–82. <https://doi.org/10.2307/3545749>
- Strain, E.M.A., Edgar, G.J., Ceccarelli, D., Stuart-Smith, R.D., Hosack, G.R. & Thomson, R.J. (2019). A global assessment of the direct and indirect benefits of marine protected areas for coral reef conservation. *Diversity and Distributions*, 25(1), 9–20. <https://doi.org/10.1111/ddi.12838>
- UNEP-WCMC & IUCN. (2019). *Marine protected planet*. Available at: www.protectedplanet.net [Accessed 13 July 2020]
- White, A.T., Aliño, P.M., Cros, A., Fatan, N.A., Green, A.L., Teoh, S.J. et al. (2014). Marine protected areas in the coral triangle: Progress, issues, and options. *Coastal Management*, 42(2), 87–106. <https://doi.org/10.1080/08920753.2014.878177>
- Wickham, H. (2016). *ggplot2: Elegant graphics for data analysis*. New York: Springer.

SUPPORTING INFORMATION

Additional supporting information may be found in the online version of the article at the publisher's website.

How to cite this article: Wright, L.S. (2022). Small no-take areas benefit hard corals more than regulation through fishing permits. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 32(7), 1209–1216. <https://doi.org/10.1002/aqc.3814>