

An underwater photograph of a coral reef. In the foreground, there is a large, branching, orange-brown coral structure. To its left, a large, flat, brownish coral plate is visible. In the background, a diver in a blue wetsuit is swimming. The water is clear and blue.

The effect of a small-scale Marine Protected Area on the coral reef fish community assemblage in the Philippines

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Abstract

Global overexploitation of coral reef fisheries is having detrimental effects on the long term sustainability of reef ecosystems. With coral reef ecosystem services in the Philippines being dominated by reef fisheries, marine protected areas (MPA) are an increasingly implemented protection technique against the impacts of overfishing. The beneficial effects of MPA to coral reef fish assemblages remains in question. This study aims to determine the effects of a small scale MPA on reef fish communities over a six year period, through comparison to surrounding areas susceptible to fishing practices. Abundance and fish length data collected through Underwater Visual Census methods was analysed utilising both univariate and multivariate approaches to target species-specific and community level effects. Analysis showed the fish community inside the MPA to have a lower variability across the study period than outside the MPA, in addition to higher mean fish abundances, species richness and diversity. Abundance at both MPA inter-boundary locations increased from 2013-2018 with a significant Year effect. In contrast biomass and individual weight of commercially important fish; Scaridae and Lutjanidae, decreased across the study period. The results indicate the MPA is providing some spill over to surrounding areas, buffering against the effects of overfishing, and slowly increasing reef fish abundances. However, annual variances in abundances appear to be due to non-anthropogenic stressors impacting reef fish habitats (e.g. coral damaging storms). Overfishing outside the MPA has led to juvenile dominated populations of commercially exploited fish. Management techniques must be addressed to ensure juveniles reach sexual maturity as the MPA effect alone does not appear sufficient.

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Abbreviations

CCC:	Coral Cay Conservation
ES:	Ecosystem Services
TEV:	Total Economic Value
TL:	Total Length (cm)
CPUE:	Catch Per Unit Effort (kg/effort-1)
MPA:	Marine Protected Area
ha:	Hectare (1 ha = 0.01km ²)
UVC:	Underwater Visual Census
SST:	Sea Surface Temperature (°C)
LRFT:	Live Reef Food Fish Trade
W:	Individual Weight (g)
SRCC:	Spearman's Rank Correlation Coefficient
<i>N</i> :	Abundance
<i>S</i> :	Species richness
<i>H'</i> :	Shannon-Wiener Diversity Index
<i>J'</i> :	Pielou's Measure of Evenness
SE:	Standard Error
nMDS:	nonmetric multidimensional scaling
<i>r_s</i> :	Spearman's Correlation Coefficient
HC:	Hard Coral
EAM:	Epilithic Algal Matrix
SCUBA:	Self-Contained Underwater Breathing Apparatus

Method terminology:

Transect: 100m section of reef surveyed through UVC techniques. 6 permanent transects present on the Napantao reef – 3 inside and 3 outside the MPA boundary.

Replicate: An independent 20m section of a transect, used as a sample for data analysis. 4 replicates make up 1 transect (5m gaps between each replicate).

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Introduction

During the past 25 years the majority of coastal fish resources have been overexploited (Lauck *et al.*, 1998; Castilla, 2000), raising doubts about the long-term sustainability of certain fisheries (Pauly *et al.*, 2002; Murray *et al.*, 2004). The fragile ecosystems associated with coral reefs (Graham *et al.*, 2006) are under constant threat from the impacts of fisheries (Hughes *et al.*, 2003; Mumby *et al.*, 2006; Bozec *et al.*, 2016). Coral reef fisheries are considered small-scale, non-target and multi-gear. Artisanal in nature, technologically poor fishing gears are utilised by their respective fishers (Teh, Teh and Sumaila, 2013), and have been shown to carry a strong impact on coral reefs (McClanahan, Graham and Darling, 2014). In current times these negative impacts are only exacerbated by ever increasing human population trends (Cohen, 2003). In 2000, it was estimated that 15-35% of the 6.2 million fishers throughout the global coral reef regions operate on a small-scale (Sadovy, 2005). Further highlighted in 2008, was the fact that small-scale fishers account for 90% of fishers worldwide (González *et al.*, 2008; Teh, Teh and Sumaila, 2013). In 2013, it was approximated by the UN Global Atlas (2013) that 3.5 billion people lived within 200km of the coast; Southeast Asia exhibited the highest fisher population size of 3.35 million, with the greatest contributors being Indonesia and the Philippines (1.7 million and 912,000, respectively) (Teh, Teh and Sumaila, 2013). These figures represent the global overexploitation of coral reef fisheries, and the immense pressure put on coral reef ecosystems (Teh *et al.*, 2007). With coral reef ecosystem services (ES) in the Philippines having an estimated annual total economic value (TEV) of 4 billion US\$; primarily contributed to by reef fisheries (Tamayo, Anticamara and Acosta-Michlik, 2018), sustainability is essential for the future prosperity of coral reef ecosystems.

Fish form an important part of coral reef ecosystems; occupying a range of functional groups (e.g. herbivore, coralivore, invertivore) (Bellwood *et al.*, 2004) and hence playing key roles in reef maintenance (e.g. the role of Parrotfish in preventing algal blooms (Mumby, 2009)). Coral reef fish also contribute a large proportion of the reefs standing biomass, promoting local biodiversity through niche occupation (Dallison, 2015; Aguilar-Medrano and Calderon-Aguilera, 2016). Overfishing on coral reefs can have severe consequences including trophic cascades and reducing reef resilience to change; both

of which are detrimental to long-term sustainable exploitation (Cinner *et al.*, 2009). A healthy reef, with optimal abundances of fish, invertebrates and other coral reef associated organisms is able to resist phase shifts, regenerate and retain critical function (Bellwood *et al.*, 2004). Coral reef fisheries will often use size selective fishing gear, generally targeting and hence removing the larger individuals first (Jackson *et al.*, 2001). Therefore reefs dominated by sexually immature juvenile fish show a strong indication of overfishing (Froese, 2004). The total length (TL) of many fish species is related to their ability to spawn and thus, their life-stage (Froese and Binohlan, 2000). The removal of juvenile fish from coral reefs prior to reaching sexual maturity is likely to result in a reduction in population size as mortality surpasses recruitment. Therefore, surveys of species-specific lengths and the length at first maturity form an integral part of fisheries management.

Fishery overexploitation can lead to a decrease in catch per unit effort (CPUE) and a resulting drop in fishery productivity. As has been seen in the Philippines this often drives fishers to employ Malthusian fishing practices (Pet-Soede and Erdmann, 1998); often utilising dynamite, cyanide, drop rocks and other destructive fishing techniques in an attempt to increase their catch (Saila, Kocic and McManus, 1993; Steneck, 2009). Such practices result in the removal of macro-benthic organisms and the loss of the complex 3-dimensional reef structure that many coral reef associated fish depend upon for habitat, feeding and reproduction (Saila, Kocic and McManus, 1993; Sumaila *et al.*, 2000). 55% of coral reefs globally are considered threatened and another 33% highly threatened due to overfishing and destructive fishing practices, mitigating such stressors is a necessity for local and national governments (Cinner and McClanahan, 2006; Teh *et al.*, 2007; McClanahan, Hicks and Darling, 2008; Kronen *et al.*, 2010). Mounting near (e.g. overfishing) and far field (e.g. ocean acidification and temperature induced coral bleaching (Hoegh-Guldberg *et al.*, 2007; McClanahan *et al.*, 2007)) stressors are pushing the world's reefs towards a point of no return, consequently calling for new management approaches to preserve coral reef functioning (Hughes *et al.*, 2017).

The implementation of marine protected areas is increasingly considered in coastal areas as a means to preserve vagile fauna and habitat from the effects of fishing (Francour *et al.*, 2001; Halpern, 2003) and is one of the most commonly used management techniques in the protection of coral reefs (Burke *et al.*,

2011). They are loosely defined as marine areas actively managed for conservation. The level of protection can range from a complete restriction of activity (fishing and recreation likewise, e.g. no-take zones), to areas of limited fishing activities and/or restrictions on particular fishing gear, for the ecological benefit of the protected area (Burke *et al.*, 2011). When managed effectively MPAs can help achieve conservation and resource management targets simultaneously and so are considered instrumental to sustainable ocean utilisation (Pauly *et al.*, 2002).

However, the degree to which such local conservation techniques can improve reef resilience and biomass remains under question (MacNeil *et al.*, 2015). MPA establishment and maintenance has resulted in a range of outcomes, varied in success with respect to their respective coral reef fish communities. The importance of using a fully protected MPA (No-take zone) over a more moderately protected approach has been highlighted in order to achieve conservation goals (Thiault *et al.*, 2019; Topor *et al.*, 2019). No-take zone MPAs are also more effective as a fishery and conservation tool for organisms that have relatively sedentary adult life stages and exhibit larval dispersion, enabling biomass exportation to the surrounding areas (MPA spill over) (Sladek Nowlis and Roberts, 1999; Chiappone and Sullivan Sealey, 2000). The beneficial properties of MPAs have primarily been observed through increases in mean size and abundance of reef fish (for reviews see: Roberts and Polunin, 1991; Bohnsack, 1998; Halpern, 2003). However, MPA effects may be diverse in direction and magnitude (Halpern and Warner, 2002), and so depend upon factors such as species and time with respect to the MPAs establishment (Mosquera *et al.*, 2000). Previous studies have shown biological responses (abundance, density, biomass, average size and diversity of organisms) to either consistently increase within the MPA over time (Russ and Alcala, 1996), show little change over time (Denny and Babcock, 2004), or an initial rise before then falling back to original levels (Dufour, Jouvenel and Galzin, 1995). Responses were also variable between fish taxa, generally with fishery exploited (Côté, Mosqueira and Reynolds, 2001) and large bodied species (irrespective of fishery status) (Mosquera *et al.*, 2000; Claudet *et al.*, 2006) showing a more positive response to protection. Studies have also shown non-target species to show either no response or negatively respond which they have suggested could be due to increased pressure from predators (McClanahan *et al.*, 1999).

Fish assemblages are studied for the effects of MPAs using a range of metrics. Pelletier *et al.* (2005) reviewed the numerous metrics used with the most frequent being abundance, biomass, diversity indices and mean size. However, Pelletier *et al.* (2005) concluded that many of these often gave insignificant results (species richness in particular was not sensitive to MPA status). The use of indicator species as a proxy for reef health has become a more frequent practice in recent times (Crosby and Reese, 1996). Such species should be abundant throughout the study area and easy to sample (Linton and Warner, 2003). The use of a variety of indicator species to study effects to fish assemblages could provide a more detailed result (Kremen, 1992). Consequently, this study adopts both univariate and multivariate analysis approaches, including the use of Chaetodontidae as an indicator species (Crosby and Reese, 1996) in order to assess the effects of a small-scale MPA on the coral reef fish community over a six year period.

Aims and objectives

This study aims to investigate the spatial differences and temporal changes in the coral reef associated fish community at the Napantao reef, Philippines, inside and outside of the MPA boundary from 2013 to 2018. Both community and species-specific analysis was conducted to aid in the identification of the potential underlying causes for the changes observed, and to comment on the efficacy of the MPA.

Objectively, this was determined through:

- Univariate analysis of coral reef fish and Chaetodontidae abundance and diversity
- Multivariate analysis of coral reef fish community composition
- Univariate analysis of commercially important fish abundance, individual weight, mean biomass and composition of size classes
- Benthic substratum analysis of reef hard coral cover

Methodology

Study Site

Sogod Bay

Sogod Bay, located in southern Leyte, the Philippines (Fig 1), currently contains 25 MPAs established between 1993 and 2016, including the Napantao MPA. They cover an area of approximately 196 hectares (ha), however this figure is likely a lower estimate as the size of several are not known due to the unavailability of accurate GPS coordinates.

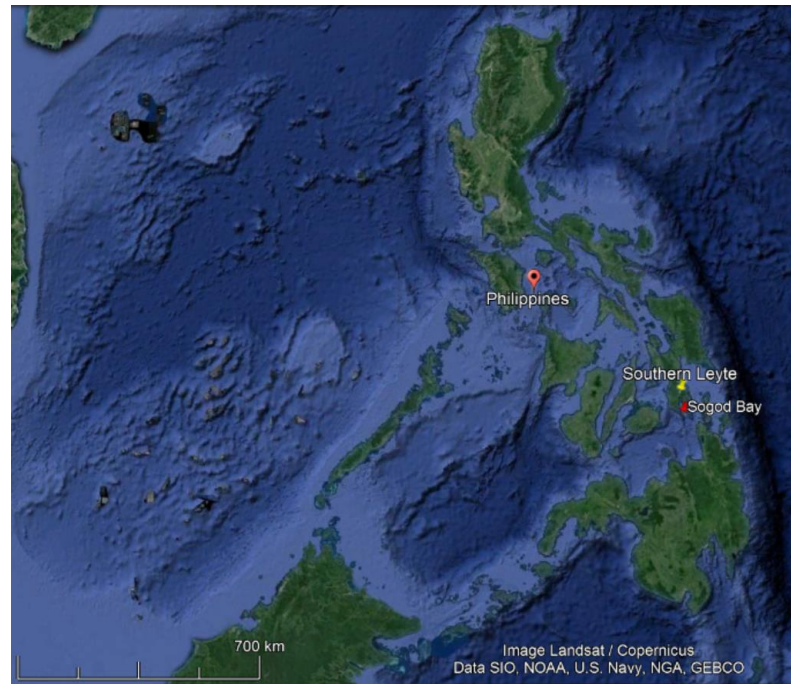


Figure 1. Satellite image of the Philippines. Southern Leyte and within that Sogod Bay, have been labelled. A scale bar is included. Image obtained from Google Earth.

The coral reefs of southern Leyte remain some of the least disturbed habitats of the Philippines. Sogod Bay is an important fishing ground, rich in a number of commercial species such as tuna, herring and anchovies. As a result it has been identified by the Fisheries Sector Program of the Department of Agriculture as one of the country's ten largest bays in need of assessment and management (Calumpong *et al.*, 1994). The Bay is also an important feeding ground for megafauna (e.g. Pilot whales, whale sharks and manta rays).

Sogod Bay is characterised by naturally limited mangrove areas, narrow fringing coral reefs and limited seagrass beds (Calumpong *et al.*, 1994). Depths in the bay reach a maximum of approximately 1.4km in the central channel.

Napantao Marine Protected Area

The site for this study is the Napantao marine protected area; located in the waters of the barangay (Philippines smallest administrative district) of Napantao, within Sogod Bay, on the western coastline of Panoan Island. Figure 2 shows a satellite image of the study area. Including the Napantao MPA and survey transect sites, marked using GPS co-ordinates (see appendix A1).



Figure 2. Satellite imagery of the Napantao marine protected area, Sogod Bay, Southern Leyte, Philippines. Shaded yellow area marks the MPA. Red pins mark the locations of the 6 permanent survey transect stakes (marking middle of transect); 3 inside the MPA boundary and 3 outside. The red lines are scaled to represent the 100m transect paths surveyors would take (general north-south direction, whilst maintaining 12m survey depth). Coral Cay Conservation (CCC) field base is labelled. A scale bar and northward heading arrow are included. See appendix A1 for MPA boundary and transect stake GPS co-ordinates.

The Napantao MPA was established in 1996 and covers 5 hectares. The area is a no-take zone, whereby all fishing practices are prohibited inside the MPA boundary and a ban on destructive fishing practices

(Pet-Soede and Erdmann,

1998) in the region is in

place. The Napantao MPA

has a relatively good

infrastructure compared to

other MPAs in Sogod Bay.

This includes a part-time

(night shift primarily)

Table 1. Summary of the Napantao marine protected area established infrastructure (Barangay Napantao, southern Leyte, Philippines.)

MPA Infrastructure	Present?	Comments
Trained Bantay Dagats	✓	Environmental & enforcement training
Permanent Bantay Dagat presence	✗	Part-time (primarily night shifts)
MPA guardhouse	✓	Bantay Dagat not always present
Demarcation buoys	✓	Not placed at exact boundaries
Patrol Boat	✓	Not always available
Bantay Dagat gear (flashlight, fins, mask, etc.)	✓	Available but rarely used
User fees collected	✓	Regularly collected

Bantay Dagat (MPA supervisor; job to ensure MPA bylaws are enforced) with access to a patrol boat and guardhouse. The MPA boundary is also marked with demarcation buoys (include strobe lights at night) (Table 1).

The barangay is currently home to around 500 people of which about 50 are fishers (estimated from most recent CCC records and local staff knowledge). Another 10-15 fishermen fish in the area but live outside the barangay. In 2013, the population was estimated to be around 300 with only ~ 25 fishers (Dallison and Ferguson, 2017). Only ~ 10 fishermen fish the Napantao waters every day. Hook and line fishing is seasonal. The boats used by local fishers are mostly non-motorised. Fishing takes place in waters surrounding the MPA on a regular basis. Levels of poaching inside the MPA boundary on average are described as medium ($>1/\text{month}$, $<1/\text{week}$) (pers. Comm. Bantay Dagat).

CCC international and local staff consider the overall anthropogenic impacts to the MPA as ‘low’. Due to a perceived low commercial and artisanal fishing impact in addition to low sewage and industrial pollution.

This region is characterised by calm waters and little precipitation from roughly September to June; a period known as *Amihan*, governed by the northeast trade winds. From June to September, the trade winds switch to south-westerlies which leads to increased levels of precipitation and storm surges with hot, humid weather, termed the monsoon season or *Habagat*.

Data Collection

The data used in this study was provided by Coral Cay Conservation (CCC). The data was collected over a six year period from 2013 to 2018. The data consisted of coral reef fish abundance, commercially important fish size class (total length, cm) frequency and benthic substratum data.

Biophysical Survey

MPA assessments are conducted using an enhanced Reef Check method. The Reef Check methodology developed in the 1990s is widely recognised, and used to survey coral reefs around the world (Hodgson, 1999). It is a mode of underwater visual census (UVC) that upon completion of a skill development

course (SCUBA diving, underwater survey methods and faunal identification) allows civilians and scientists alike to proficiently conduct surveys, with reliable results (Léopold *et al.*, 2009). An augmented reef check methodology is used at the Napantao MPA due to a high biodiversity within Sogod Bay. This includes the recording of additional specific fish species (e.g. *Chaetodon* sp.).

The Napantao MPA is surveyed biannually around March and October, with surveys during the study period taking place on average at 10:07 UTC +8 (local time). Sea surface temperature (SST) at the time of surveying was on average $28.2^{\circ}\text{C} \pm 0.15^{\circ}\text{C}$. At 10m depth, closer to the surveying depth (~12m), the average temperature was $27.6^{\circ}\text{C} \pm 0.15^{\circ}\text{C}$. The horizontal visibility on the reef during March and October surveys was $15.9\text{m} \pm 0.9\text{m}$ and $17.9\text{m} \pm 0.9\text{m}$ respectively. Surveys are conducted along 6 permanent transect sites; 3 located inside the MPA boundary and 3 outside (Fig 2). The transect sites are marked by a number of metal stakes permanently positioned on the reef (front cover image), indicating the middle, and endpoints of each transect. These remain on the reef throughout the years and are positioned continuously at a depth of around 12m where topography allows. Each transect is 100 metres long and divided into 4 replicates of 20 metres each. Between each replicate is a 5m gap where no data is recorded. This survey set up allows for robust statistical analysis (Hodgson, 1999) as each replicate is independent.

Underwater Visual Census (UVC) for the Assessment of Fishes

Fish diversity and abundance data was collected using UVC methodologies (Thresher and Gunn, 1986; Samoilys and Carlos, 1992). Selected fish families and species (appendix B1) recognised as reef indicators of fishing pressure, Live Reef Food Fish Trade (LRFFT) and the ecological status of the reef are recorded. A number of Chaetodontidae (Butterflyfish) species (appendix B2) are recorded as they provide an accepted method of indicating the ecological status of the coral reef (Crosby and Reese, 1996). Chaetodontidae are split into two main groups (obligate and facultative coralivores) dependent on their primary feeding type. The total length (TL) (cm) of three commercially important fish families (Grouper (Serranidae), Parrotfish (Scaridae) and Snapper (Lutjanidae)) was recorded to quantify biomass (kg).

Fish data is recorded by two SCUBA divers swimming side by side along a 100m 'belt' transect. Target species abundance is recorded within an imaginary 5x5x5m box along four 20m replicates. Between each 20m replicate, the divers will wait, and not record any fish for a 5m interval, this separates the total 100m transect into four 20m independent replicates (Figure 3). Each replicate sums to a total of 500m³ with total transect area surveyed of 2,000m³.



Figure 3. An augmented method that Coral Cay Conservation utilises to carry out MPA assessments that is adapted from a standardised Reef Check method. This schematic demonstrates the Underwater Visual Census (UVC) for the assessment of fishes. Displayed here are two of the normal four replicates that make up a 100m transect; replicates separated by 5m gaps. Two divers swim along the transect; one either side of the line. For the first 20m, they will record the specified target species within an imaginary 5x5m box, sizing those that are commercially important. Once located within the 5m gap, no fish are recorded, the divers wait 1 minute before proceeding with the next replicate, to ensure they are independent.

Fish Biomass Calculations

During UVC fish surveys the individual size of commercially important fish were characterised through total length (TL (cm)) estimations (surveyors must pass (100%) a UVC fish length estimation practical test; using 10cm ranged categories), and subsequently placed into appropriate size classes: 0-10cm; 11-20cm; 21-30cm; 31-40cm; 41-50cm; 51-60cm; >60cm. Length-weight relationships (LWR) (Equation 1) were utilised to calculate the mass, ((W) g) of each individual and from this the biomass (kg) present over a replicate. The individual weight of species in each sample can vary considerably. Therefore, specific steps were taken within this study when assessing the W of individuals in each commercially important fish family in accordance with Froese and Pauly, (2000) and Froese, Thorson and Reyes, (2014).

The weight (W, g) of each size class of family was derived through the use of length-weight allometric coefficients (appendix B3) (a and b equation 1) calculated by Dallison and Ferguson, (2017). The relationship between total length (TL (cm)) and weight (W, g) for fish species is expressed by equation (1):

$$W = aL^b$$

Where, a = the condition factor (Hilborn and Walters, 1992), b = species specific growth parameter (Quinn and Deriso, 1999).

Point Intercept Transect (PIT) for the Benthic Substrate Composition

In addition to fish abundance, the substratum along the transect lines was recorded using a point intercept methodology. Along the transect line, benthic organisms and substrate types were recorded at 0.5m intervals (Figure 4). To minimize bias, a plumb line was dropped at each designated 0.5m point and the substrate type was recorded. Every replicate (20m) contained 40 benthic points.

In this study hard coral forms the primary benthic focus. See appendix B4 for recorded hard corals and full benthic category list.

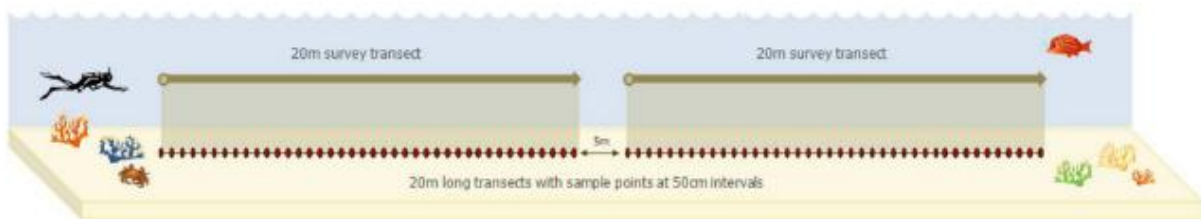


Figure 4. An augmented method that Coral Cay Conservation utilises to carry out MPA assessments that is adapted from a standardised Reef Check method. This schematic demonstrates the Point Intercept Transect (PIT) for the Benthic Substrate Composition. Displayed here are two of the normal four replicates that make up a 100m transect; replicates separated by 5m gaps. A diver swims along the transect line for the first 20m, they will record the specified target abiotic substrates and biotic benthic organisms using a plumb-line every 50cm (indicated by marks along the line). Once located within the 5m gap, no data is recorded, to ensure replicates are independent.

Data Analysis

Data manipulation and calculation of arithmetic means was conducted using Microsoft Excel 2013. A combination of SigmaPlot 13 and Primer V6 were used for statistical and graphical purposes.

Biophysical survey data was tested for significant temporal correlations ($P < 0.05$) across the six year study period from 2013 to 2018 using the Spearman's Rank Correlation Coefficient (SRCC).

Each independent 20m replicate (n= 288) was analysed over a period from 2013 to 2018 with each annual data collection resulting in n= 48 replicates recorded over two separate monthly periods to

remove the effects of seasonality changes conferred by coral communities. Community assemblages present on the coral reef and throughout the sample period were described using abundance (N), species richness (S) and the Shannon-Weiner diversity index (H') (Shannon, 1948) with the complementary Pielou's measure of species evenness (J') (Pielou, 1966). S represents the number of different species found in an ecological community. H' is used to characterize species diversity based on abundance and evenness and represents the degree of uncertainty relating to an individual's species identity (Morris *et al.*, 2014). J' represents species evenness and is constrained between 0 and 1. A community closer to 0 is less even and indicates the presence of a dominant species. Values for S , H' and J' were calculated through the Primer V6 univariate statistical package.

Due to the large size of the raw dataset, the arithmetic means ($\pm$ standard error, SE) of fish N , S , H' and commercial fish biomass were determined by the amalgamation of the 24 replicates from inside the MPA each year and outside the MPA. This produced $n = 2$ data sets for each year sampled; inside ($n = 1$) and outside ($n = 1$) of the MPA boundaries.

The data was tested for normality and homogeneity of variance using the Shapiro-Wilks (Shapiro and Wilk, 1965) and Brown-Forsythe (Brown and Forsythe, 1974) tests respectively. Data demonstrating to be non-parametric ($P > 0.05$) was transformed (square root) to give a normal distribution and homogeneity of variances. The two-way ANOVA test was used to test for statistically significant ($P < 0.05$) Year, Location and Year $\times$ Location effects on abundance (N), species richness (S) and diversity (H') of fish and target Chaetodontidae across the years 2013 to 2018, both inside and outside the MPA. Chaetodontidae abundance was also tested for Year, Feeding Type and Year $\times$ Feeding Type effects.

Coral reef fish community compositions were analysed through multivariate comparisons, examined by means of nonmetric multidimensional scaling (nMDS) plots based on the Bray-Curtis (Bray and Curtis, 1957) similarity matrix with a $\text{Log}(X+1)$ data transformation. Each data point represents the community at a single replicate. The closer the data points are to each other on the nMDS plots, the more similar the fish community compositions were between the corresponding replicates. A stress value; determined using the Kruskal Wallis formula, indicates the nMDS plots 'goodness of fit' in

condensing the multivariate dataset to a 2D plot. Stress values and fit: <0.1 = good, <0.2 = OK but other methods required to cross check observational interpretation, >0.3 = unreliable. Analysis of variance was used to test for differences between the fish communities inside and outside the MPA by using the ANOSIM routine in Primer V6 (Clarke and Warwick, 1994). Determined using the Primer V6 SIMPER routine, the Bray-Curtis similarity matrix was used to analyse the contribution of each species to the dissimilarity between the communities inside and outside the MPA and the similarities within each location.

Commercially important fish data was analysed using two-way ANOVA's to test for statistically significant Year, Location and Year x Location effects on abundance (N), individual weight (W (g)) and mean biomass ($\text{kg}/500\text{m}^3$). Commercial fish size class population percentage composition graphs were used to further support the weight and biomass results.

The percentage proportion of hard coral substratum cover was analysed using a two-way ANOVA, to test for statistically significant Year, Location and Year x Location effects. This was used to provide potential insight into the underlying causes of changes to the coral reef fish communities.

Following the two-way ANOVA, a Holm-Sidak Post Hoc method was used in some cases for further pairwise comparisons with a significant result showing $P < 0.05$.

Results

Biophysical Assessment

Over the six year study period from 2013 – 2018, SRCC tests showed that there was no statistically significant correlations in SST (SRCC, $r_s = -0.006$, $P = 0.96$) or water temperature at 10m (SRCC, $r_s = 0.05$, $P = 0.71$). The MPAs distance to the nearest population centre and freshwater river input was $1.4 \pm 0.09\text{km}$ and $0.9 \pm 0.07\text{km}$ respectively. Major coral damaging storms were recorded during October 2013, December 2014 and November 2017.

Coral Reef Community Analysis

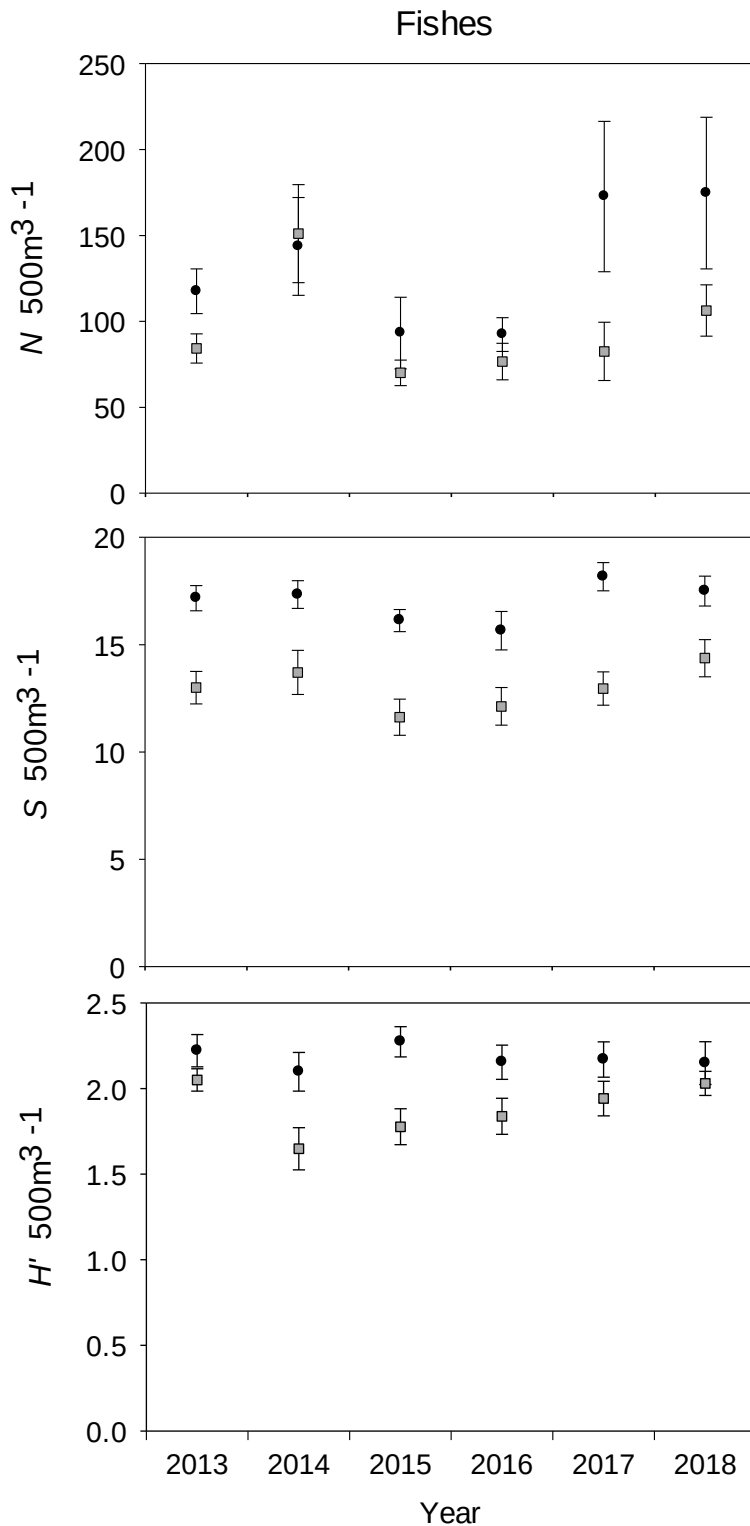
Coral Reef Associated Fish

Over the six year study period from 2013 to 2018 a total of 32,669 fish were recorded from 26 Families and one Superorder. Fusiliers (Caesionidae) showed the highest abundance across the study period, being recorded 10352 times whilst rays (Batoidea), a superorder were observed the least (5 times with 4 individual's in 2014). Fusiliers appeared to dominate inside the MPA across all survey years except 2016 where Cardinalfish (Apogonidae) had the highest abundance (15 ± 6.53). Cardinalfish also had the highest abundance outside of the MPA on 4 out of 6 years whereby fusiliers dominated outside the MPA in 2014 and 2015. See appendix C1 for species yearly maximum abundances inside and outside the MPA.

The mean abundance of coral reef associated fishes inside the Napantao MPA was lowest in 2016 ($N = 92.3 \pm 20.8$) and greatest in 2018 ($N = 174.7 \pm 44.1$), representing an increase in average abundance of 89.1% (Fig 5). Inside the MPA fish abundance increased by 48.6% between 2013 and 2018. Outside of the MPA saw an 26.2% increase in abundance over the study period. The greatest mean fish abundance outside the MPA was observed in 2014 and the lowest in 2015 ($N = 157.9 \pm 28.4$ and $N = 70.1 \pm 7.4$) with a 53.6% decrease (Fig 5). Spatially N was $43.6\% \pm 16.1\%$ higher inside the MPA from 2013-2018. The total fish abundance was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 12.108$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 3.635$, $p < 0.05$). No significant Location x Year interaction effect was found (ANOVA, $F_{5,275} = 1.318$, $p = 0.256$), however 2017 showed a significant spatial difference in N (Holm-Sidak, Diff means = 3.79, $t = 3.38$, $P < 0.001$). The Year effect was due to differences in N between 2014 and 2015 (Holm-Sidak, Diff means = 2.58, $t = 3.25$, $P < 0.05$), and 2014 and 2016 (Holm-Sidak, Diff means = 2.39, $t = 3.00$, $P < 0.05$).

Both inside and outside the MPA saw an overall increase in species richness (S) across the study period; 1.9% and 10.6% respectively (Fig 5). S was highest in 2017 inside the MPA ($S = 18.2 \pm 0.7$). Spatially, S was $31.4\% \pm 2.9\%$ higher inside the MPA over the study period. Species richness was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 80.937$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 2.638$, $p < 0.05$), however there was no significant Location x Year interaction effect

(ANOVA, $F_{5,275} = 0.478$, $p = 0.793$). The relatively higher F statistic for Location suggests it was the primary underlying cause of variance in the means.



Diversity (H') was highest in 2015 inside the MPA, also showing a high degree of evenness ($H' = 2.3 \pm 0.1$ and $J' = 0.8 \pm 0.03$). H' decreased across the study period both inside and outside the MPA by 3.2% and 1% respectively. Spatially H' was $16.4\% \pm 3.9\%$ higher inside the MPA from 2013-2018. Diversity was significantly affected by the independent effect of Location (ANOVA, $F_{1,275} = 19.853$, $p < 0.001$), however the Year (ANOVA, $F_{5,275} = 1.638$, $p = 0.150$) and Location x Year interaction effects (ANOVA, $F_{5,275} = 1.172$, $p = 0.323$) were not significant.

Figure 5. Mean abundance (N), species richness (S) and Shannon-Wiener diversity index (H') per 500m³ for coral reef associated fishes over a six year period from 2013 to 2018 at the Napantao marine protected area (MPA), southern Leyte, Philippines. The data is represented in two sets with the standard error given (SE); inside the MPA boundaries (Black filled circle); and outside the MPA boundaries (Grey filled square).

Fish Community Composition

nMDS plots (Fig 6) grouped replicates from the whole coral reef associated fish abundance data set across the study period, dependent upon their similarity in community composition (Fig 6a). This initially appeared to show differences in the community compositions inside and outside the MPA (Fig 6a). The community compositions recorded outside of the MPA were more variable over the study period. However, Figure 6a has a stress level of 0.22, indicating the model has an 'OK' fit to the multivariate dataset. Therefore, further statistical testing was required to confirm the differences in community compositions at inter-boundary locations were significant. ANOSIM tests confirmed the community compositions of coral reef associated fish at the Napantao Reef from 2013-2018 to be significantly different within and outside the MPA (ANOSIM, global $r = 0.184$, $P < 0.05$).

Figures 6b and c used average abundance values for coral reef fish on the Napantao reef. Values were averaged over 8 and 24 replicates, to give average values per transect / year / location (Fig 6b) and per year / location (Fig 6c) respectively. This use of averages does not allow for robust statistical ANOSIM analysis. However figures 6b and c both clearly show distinctive community composition groupings relating to their inter-boundary locations and so supports the results of figure 6a. Areas of 80% similarity in community composition are marked by the red line, which in figure 6c clearly defines the two groups (inside and outside the MPA). Both figures (6b and 6c) show a wider distribution of data points outside the MPA, suggesting the coral reef fish community composition was more variable over the study period than it was inside the MPA.

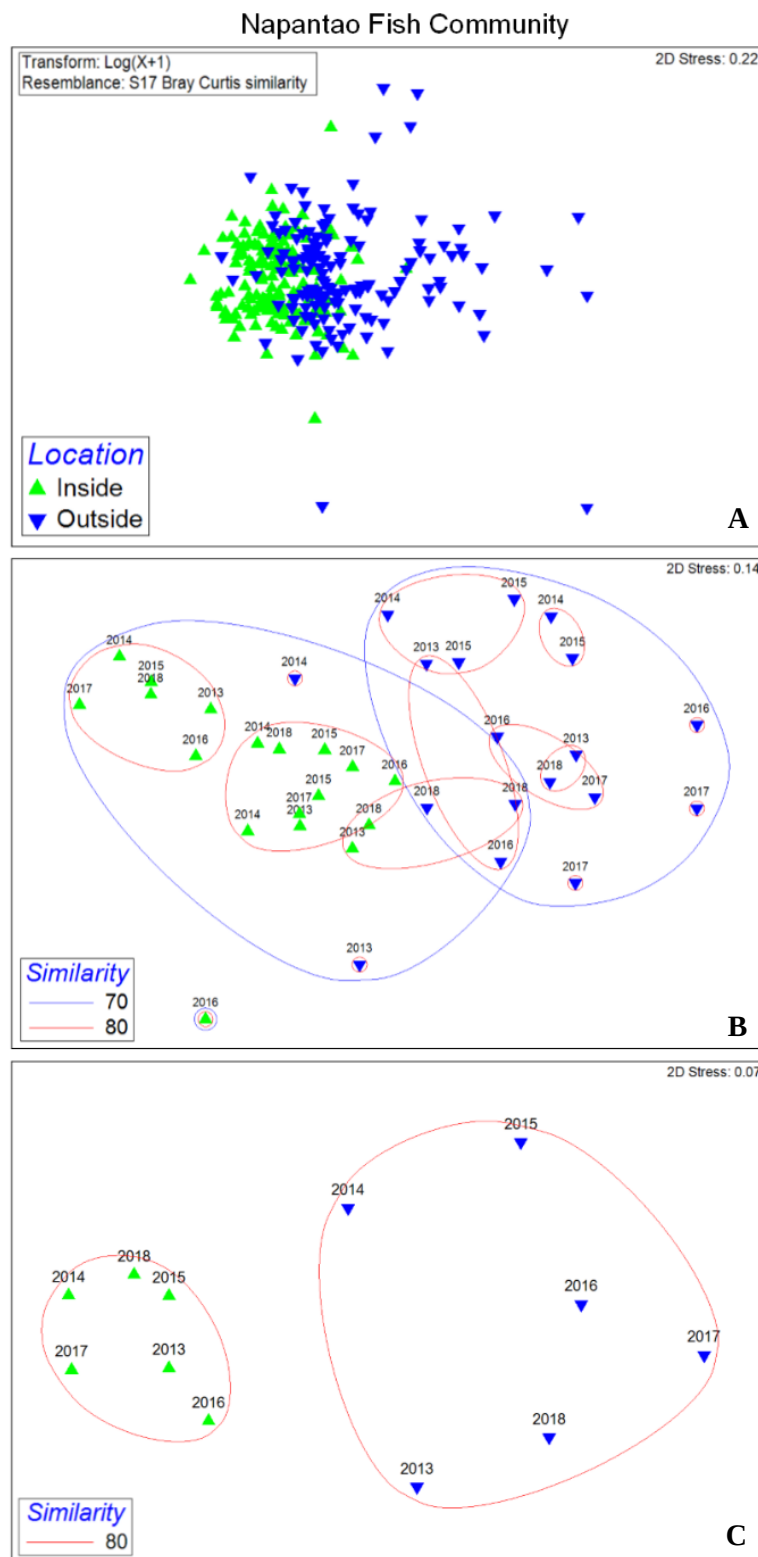


Figure 6. 2D multidimensional scaling (MDS) plots of the coral reef fish community at the Napantao marine protected area (MPA), southern Leyte, Philippines from 2013-2018. Data transformed via Log(X+1) function and subjected to Bray Curtis similarity matrix. (A) Created using entire fish abundance dataset from 2013-2018. Figure legend shows data point's location; inside or outside the Napantao MPA. (B and C) Created using fish abundances, averaged per transect / year / location (8 samples per point) and per year / location (24 samples per point) respectively. Figures B and C include similarity area perimeter lines, grouping points of certain resemblance levels (shown in figure legends). All plots include a stress value.

In order to identify which species were causing the differences in the community compositions inside and outside the MPA, a SIMPER analysis was conducted. This first showed the average dissimilarity between the communities observed at each inter-boundary location from 2013-2018 to be 51.36%. The species actively contributing the most to this dissimilarity (discriminating species) were the fusiliers (Caesionidae) and cardinalfish (Apogonidae), contributing 8.72 and 7.81% respectively. From 2013-2018 fusiliers had an average abundance per replicate (20m) inside and outside the MPA of 48.9 ± 11.7 and 23.3 ± 5.4 respectively. In contrast cardinalfish had average abundances per replicate inside the MPA of 13.1 ± 1.8 and outside of 24.7 ± 3.6 .

This SIMPER analysis was also used to identify the species with the highest contribution to the similarities within each of the inter-boundary locations and hence the “dominant” species at each location. The average similarity between the community compositions of replicates from inside the MPA over the study period was 57.47%. The top three species with the greatest contributions to this similarity were the angelfish (Pomacanthidae), parrotfish (Scaridae) and eastern triangle butterflyfish (*Chaetodon baronessa*). They contributed 12.25, 11.82 and 10.68% respectively and in turn can be considered “dominant” species within the Napantao MPA boundaries. Outside the MPA, the coral reef fish community shared a lower average similarity of 49.45% between the replicates. In this case the top three contributing species were the eastern triangle butterflyfish (*Chaetodon baronessa*), parrotfish (Scaridae) and cardinalfish (Apogonidae). Respective contributions (14.09, 12.17 and 11.53%). For species yearly maximum abundances inside and outside the MPA see appendix C1.

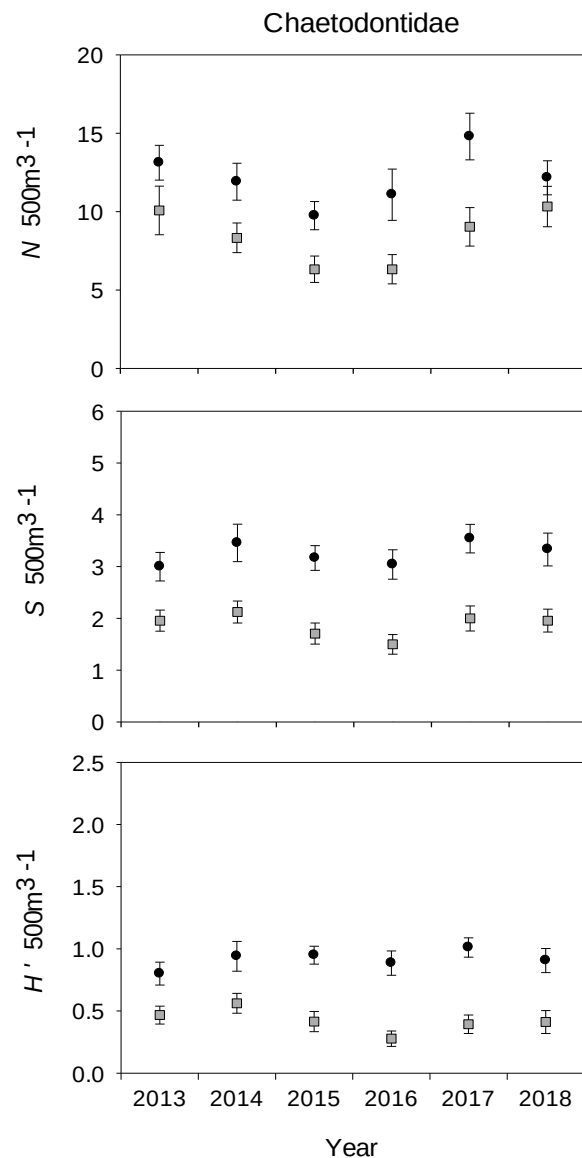
Chaetodontidae

Across all years of the study period at both inter-boundary locations on the Napantao Reef, the eastern triangle butterflyfish (*Chaetodon baronessa*) was the dominant *Chaetodon sp.* with the highest average abundance (*N*). *Chaetodon baronessa* abundance was highest in 2018 outside the MPA (7.96 ± 0.92) and lowest during 2015 inside the MPA (4.46 ± 0.42).

Chaetodontidae abundance (*N*) (Fig 7) was highest during 2017 inside the MPA ($N = 14.79 \pm 1.49$). From 2013 to 2018, *N* inside the MPA decreased by 7.3% but in contrast increased by 2.5% outside the

MPA. Spatially, Chaetodontidae abundance was $47.3\% \pm 8.7\%$ higher inside the MPA than outside. (Fig 7). Chaetodontidae abundance was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 33.282$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 3.577$, $p < 0.05$). No significant Location x Year interaction effect was found (ANOVA, $F_{5,275} = 0.516$, $p = 0.764$).

Chaetodontidae S and H' both appeared to plateau across the study period at both inter-boundary locations (Fig 7). S showed no change between 2013 and 2018 outside the MPA, and only an 11.1% increase inside. H' increased by 13.1% inside the MPA and decreased by 11.9% outside over the study period. Spatially, S and H' were on average $75.2\% \pm 7.2\%$ and $127.3\% \pm 23.2\%$ respectively higher



inside the MPA than outside (Fig 7).

Chaetodontidae species richness was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 82.668$, $p < 0.001$). However, no significant Year (ANOVA, $F_{5,275} = 1.470$, $p = 0.200$) or Location x Year interaction effects were found (ANOVA, $F_{5,275} = 0.498$, $p = 0.778$).

Similarly Chaetodontidae diversity was only significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 81.634$, $p < 0.001$). Year (ANOVA, $F_{5,275} = 0.844$, $p = 0.519$) and Location x Year interaction effects were not significant (ANOVA, $F_{5,275} = 1.209$, $p = 0.305$).

Figure 7. Mean abundance (N), species richness (S) and Shannon-Wiener diversity index (H') per 500m^3 for Chaetodontidae over a six year period from 2013 to 2018 at the Napantao marine protected area (MPA), southern Leyte, Philippines. The data is represented in two sets with the standard error given (SE); inside the MPA boundaries (Black filled circle); and outside the MPA boundaries (Grey filled square).

Obligate coralivore *Chaetodon sp.* total abundance was on average $1477.3\% \pm 386.3\%$ higher than facultative *Chaetodon sp.* abundance and $390.9\% \pm 93.8\%$ higher than other *Chaetodon sp.* across both inter-boundary locations from 2013 to 2018 (Fig 8). Spatially, obligate *Chaetodon sp.* abundance was $20.2\% \pm 7.1\%$ higher inside the MPA than outside over the study period. Chaetodontidae abundance (N) was significantly affected by the independent effects of Feeding Type (ANOVA, $F_{1,550} = 983.054$, $p < 0.001$) and Year (ANOVA, $F_{5,550} = 4.440$, $p < 0.001$), in addition to significant interaction effect between the variables Feeding Type x Year (ANOVA, $F_{5,550} = 3.821$, $p < 0.05$).

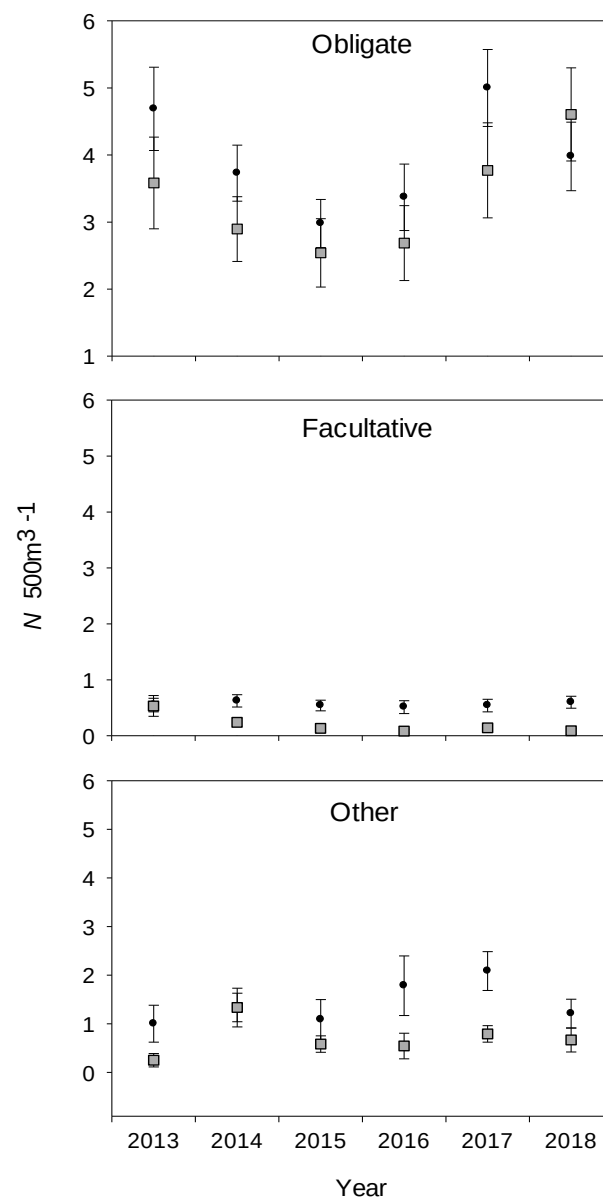


Figure 8. The arithmetic mean abundance (N) (500m^3) of Chaetodontidae obligate coralivores; facultative coralivores; and other Chaetodontids whose feeding modes were not analysed, taken over a six year period from 2013 to 2018 at the Napantao marine protected area (MPA), southern Leyte, Philippines. The data is represented in two sets with the standard error given (SE); inside the MPA boundaries (Black filled circle); and outside the MPA boundaries (Grey filled square).

Commercially Important Fish

Scaridae (parrotfish) was the most abundant commercial species over the study period and was recorded in the greatest abundance (N) in 2018, inside the MPA ($N = 11.5 \pm 1.4$) (Fig 9). Scaridae experienced an increase in N over the study period of 17.4% inside the MPA, but a decrease outside the MPA of 2%. Spatially, Scaridae abundance was $68.5\% \pm 38.5\%$ higher inside the MPA than outside from 2013-2018. Scaridae abundance was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 22.487$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 4.593$, $p < 0.001$), in addition to an interaction effect, Location x Year (ANOVA, $F_{5,275} = 2.742$, $p < 0.05$).

Serranidae (grouper) abundance inside the MPA increased by 64.5% from 2013 to 2018 where it was greatest ($N = 6.4 \pm 1.6$). Outside the MPA saw a 21.4% increase over the study period (Fig 9). Spatially, Serranidae abundance was on average $216.6\% \pm 31.2\%$ greater inside the MPA than outside. Serranidae abundance was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 98.533$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 3.501$, $p < 0.05$), however no significant Location x Year interaction was found (ANOVA, $F_{5,275} = 1.140$, $p = 0.340$).

In contrast, Lutjanidae (snapper) saw decreases in abundance of 54.9% and 48.9% inside and outside the MPA respectively from 2013 to 2018 (Fig 9). However, Lutjanidae abundance plateaued across the study period within a range of $N = 1.8$ so these changes were actually relatively minor. Spatially, Lutjanidae abundance was $136.3\% \pm 74.3\%$ higher inside the MPA than outside. Lutjanidae abundance was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 11.886$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 2.890$, $p < 0.05$), however no significant Location x Year interaction was found (ANOVA, $F_{5,275} = 1.199$, $p = 0.310$).

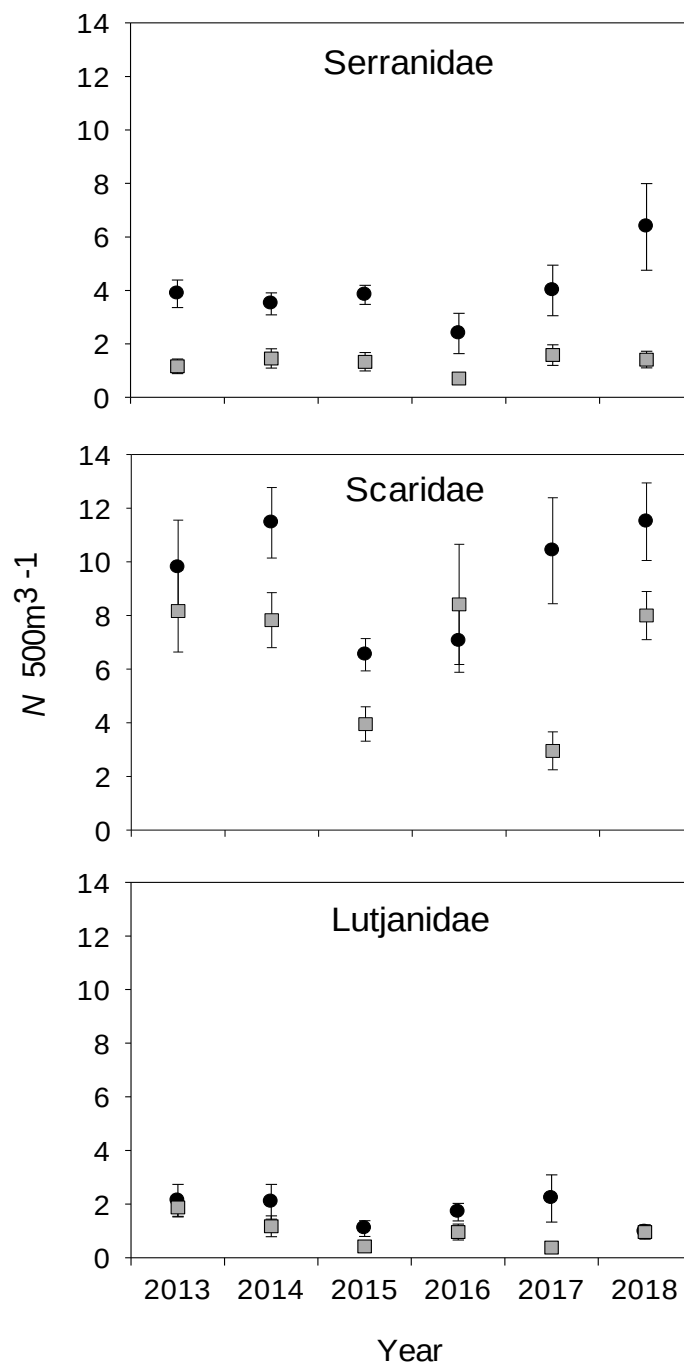


Figure 9. The arithmetic mean abundance (N) ($500m^3$) for commercially important fishes: Grouper (Serranidae); Parrotfish (Scaridae); and Snapper (Lutjanidae) over a six year period from 2013 to 2018 at the Napantao marine protected area (MPA), southern Leyte, Philippines. The data is represented in two sets with the standard error given (SE); inside the MPA boundaries (Black filled circle); and outside the MPA boundaries (Grey filled square).

Lutjanidae had the largest average individual W (g) of the commercially important fishes across the study period ($W = 176.3 \pm 25.3$ g). Lutjanidae W was greatest in 2014, inside the MPA ($W = 327.55 \pm 35.25$ g) (Fig 10). Lutjanidae W both inside and outside the MPA dropped respectively by 20.6% and 59.4% from 2013 to 2018. Spatially, Lutjanidae individual weights were on average $32.3\% \pm 26.3\%$ lower inside the MPA than outside from 2013 to 2018. Lutjanidae individual weight was significantly affected by the independent effects of Year (ANOVA, $F_{5,368} = 7.643$, $p < 0.001$) and the interaction between Location x Year (ANOVA, $F_{5,368} = 2.979$, $p < 0.05$), however the independent effect of Location was not significant (ANOVA, $F_{1,368} = 2.658$, $p = 0.104$). Lutjanidae biomass was greatest inside the MPA in 2014 (0.68 ± 0.27 kg/500m³⁻¹) (Fig 10). Both inter-boundary locations saw a decrease in biomass over the study period of 64.2% and 79.2% respectively inside and outside the MPA boundary. Spatially, Lutjanidae biomass was $136.6\% \pm 35.8\%$ higher inside the MPA from 2013-2018. Lutjanidae biomass was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 11.395$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 4.119$, $p < 0.001$), however no significant Location x Year interaction was found (ANOVA, $F_{5,275} = 0.997$, $p = 0.420$).

Serranidae had the lowest average W of the commercial fish; with the lowest value occurring outside the MPA in 2017 ($W = 21.30 \pm 4.28$ g) (Fig 10). Outside the MPA Serranidae experienced a 48.0% decrease in W over the study period, however in contrast, Serranidae W inside the MPA increased by 78.2%. Spatially, Serranidae W was $12.7\% \pm 28.2\%$ higher inside the MPA than outside on average over the study period. Similarly to Lutjanidae, Serranidae individual weight was significantly affected by the independent effects of Year (ANOVA, $F_{5,745} = 9.010$, $p < 0.001$) and the interaction between Location x Year (ANOVA, $F_{5,745} = 4.161$, $p < 0.001$), however the independent effect of Location was not significant (ANOVA, $F_{1,745} = 0.185$, $p = 0.667$). Serranidae, also had the smallest average biomass with the lowest average value recorded outside the MPA boundary in 2016 (0.02 ± 0.01 kg/500m³⁻¹) (Fig 10). From 2013 to 2018 Serranidae biomass decreased by 36.8% outside the MPA but in contrast increased by 105% inside the MPA. Spatially, Serranidae biomass was $230.0\% \pm 38.3\%$ higher inside the MPA than outside from 2013-2018. Serranidae biomass was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 45.124$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 6.181$,

$p < 0.001$), however no significant Location x Year interaction was found (ANOVA, $F_{5,275} = 1.934$, $p = 0.089$).

Scaridae W decreased over the study period inside and outside the MPA boundary by 31.8 and 56.5% respectively (Fig 10). On a spatial scale Scaridae W was on average $43.9\% \pm 33.4\%$ higher inside the MPA boundary than outside from 2013-2018. Scaridae individual weight was significantly affected by the independent effects of Location (ANOVA, $F_{1,2287} = 6.200$, $p < 0.05$) and Year (ANOVA, $F_{5,2287} = 37.228$, $p < 0.001$), in addition to an interaction effect between Location x Year (ANOVA, $F_{5,2287} = 10.687$, $p < 0.001$). Scaridae represented the greatest average biomass on the Napantao reef from 2013 to 2018 ($0.87 \pm 0.12 \text{ kg}/500\text{m}^3$). However, Scaridae saw a decrease in biomass inside and outside the MPA of 19.9% and 57.4% respectively from 2013-2018. Spatially, Scaridae biomass was $137.9\% \pm 81.6\%$ higher inside the MPA than outside across the study period. Scaridae biomass was significantly affected by the independent effects of Location (ANOVA, $F_{1,275} = 15.454$, $p < 0.001$) and Year (ANOVA, $F_{5,275} = 7.130$, $p < 0.001$), however no significant Location x Year interaction was found (ANOVA, $F_{5,275} = 2.166$, $p = 0.058$).

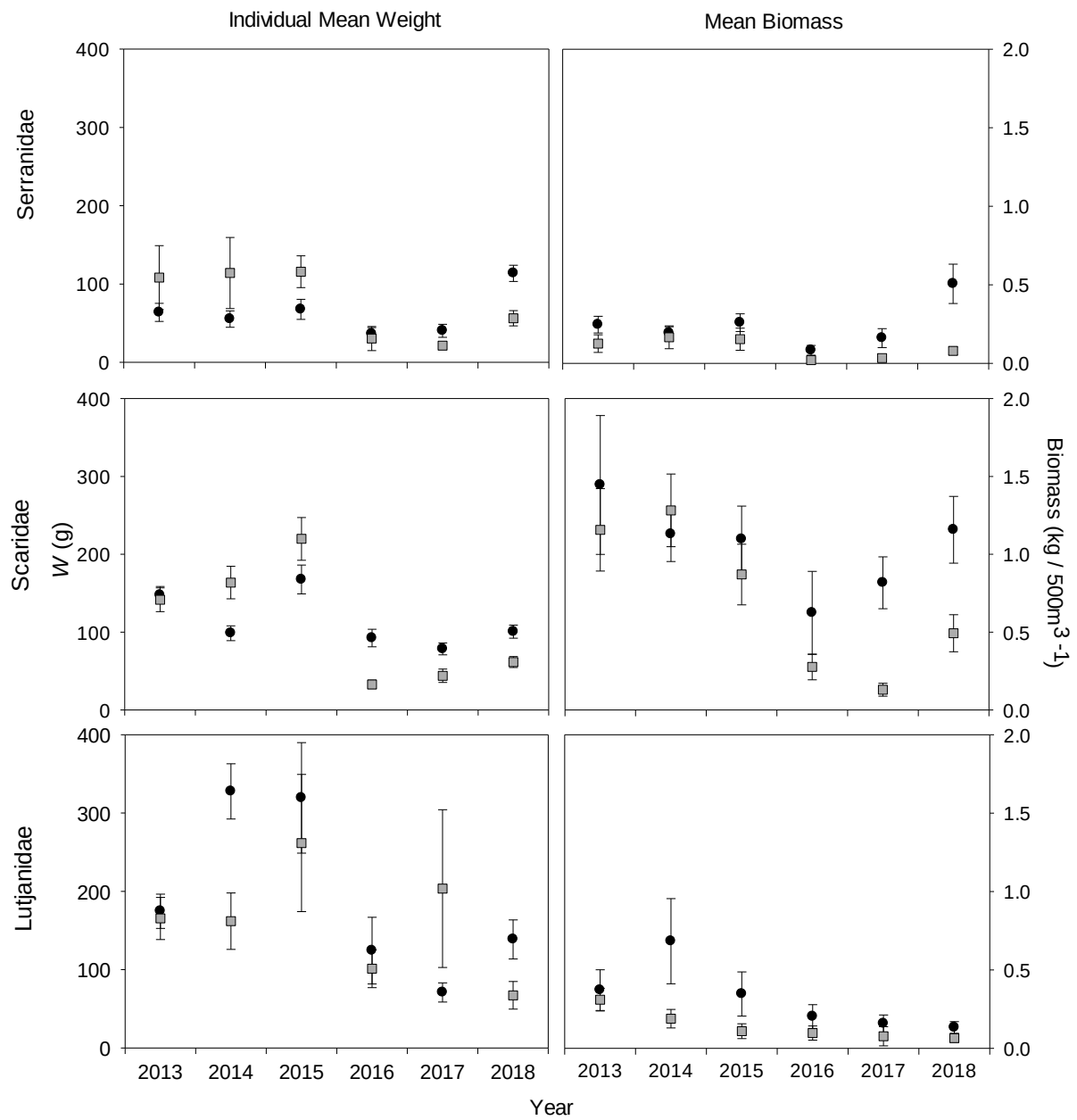


Figure 10. The individual mean weight (g) (weights of all individuals per year / number of individuals at each location) and the mean biomass (kg/500m³) (total biomass per year / number replicates per year at each location (24)) for commercially important fishes: Grouper (Serranidae); Parrotfish (Scaridae); and Snapper (Lutjanidae) over a six year period from 2013 to 2018 at the Napantao marine protected area (MPA), southern Leyte, Philippines. The data is represented in two sets with the standard error given (SE); inside the MPA boundaries (Black filled circle); and outside the MPA boundaries (Grey filled square).

The size structure compositions of the commercially important fish recorded each year were dependent on their family (Fig 11). Serranidae inside the MPA boundary was largely dominated by individuals in the size class 11-20cm in 2013 and 2018. By 2018, the proportion of Serranidae in this size class rose from 59.1% to 72.5%. Between 2013 and 2018 however, inside the MPA, Serranidae of class 0-10cm dominated, rising to 58.2% in 2016 before falling to their lowest in 2018 at 16.3%. Outside the MPA, the Serranidae population was observed to be increasing in size (TL cm) from 2013 to 2015 as the proportion of 0-10cm class decreased from 46.4% to 31.3% respectively, whilst size class 21-30 increased from 3.6% to 37.5% over this time. Following this the outside MPA population reverted back to a smaller size class dominated Serranidae community.

Scaridae populations inside the MPA were dominated by size class 11-20cm across all years of the study period with the exception of 2014 where size 0-10cm was highest (44.7%). From 2013 to 2018, the proportion of Scaridae inside the MPA boundary in size class 21-30cm decreased from 34.9% to 20.7% respectively. Outside of the MPA, the proportion of Scaridae in the 0-10cm size class increased over the study period from 27.6% in 2013 to 45.3% in 2018. This corresponded with a decrease across all other size classes.

For Lutjanidae the population composition was made up of larger size class individuals in 2013 (21-30cm), which was then reduced by 2017 where smaller individuals (0-10cm) were more abundant. Inside the MPA, in 2013, Lutjanidae was composed of only 3.9% 0-10cm fish, however by 2017 this increased to 35.8%. In 2018, this dropped back down and size class 21-30cm increased to 43.5%, equal to its level in 2013. Outside the MPA boundary, Lutjanidae of size class 11-20cm made up the highest percentage population composition for the majority of years across the study period, with the exception of 2015 and 2016. There was a decrease in the size class 11-20cm from 2013 to 2018 (64.4% to 52.2% respectively). Between 2013 and 2018, only the 0-10cm size class increased overall (0% to 34.8% respectively).

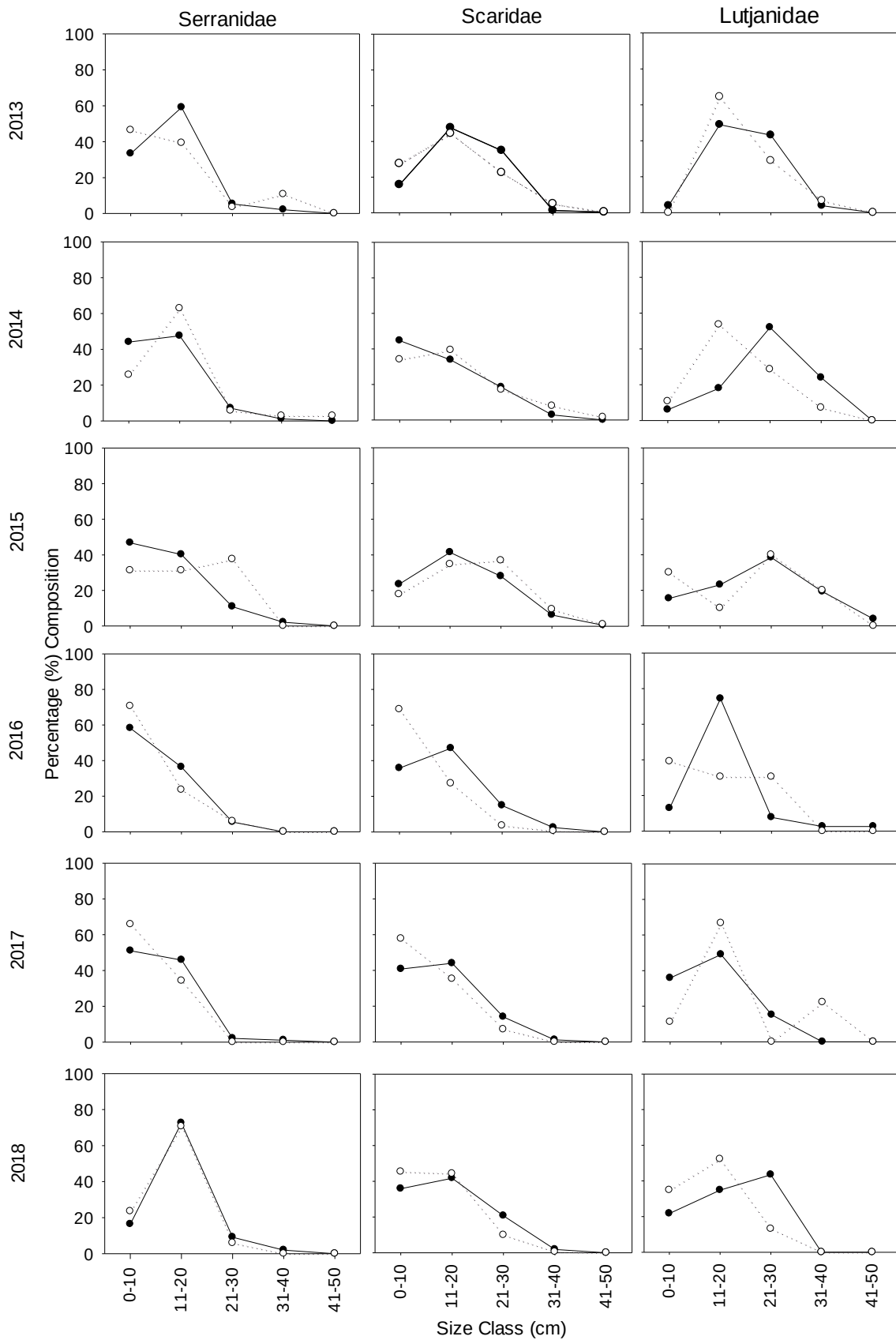


Figure 11. The percentage composition (%) of fish length size classes (cm) for commercially important fishes: Grouper (Serranidae); Parrotfish (Scaridae); and Snapper (Lutjanidae) over a six year period from 2013 to 2018 at the Napantao marine protected area (MPA), southern Leyte, Philippines. The data is represented in two sets; inside the MPA boundaries (Black filled circle); and outside the MPA boundaries (Grey filled square).

Benthic Hard Coral Cover

Hard coral (HC) has the highest percentage cover over the study period for both inter-boundary locations when compared to the other substrate categories recorded (Appendix B4) during surveys from 2013-2018. Inside the MPA this is followed by rock and outside by sand.

The percentage HC cover was highest in 2017 for both inter-boundary locations ($53.4 \pm 3.1\%$ inside MPA and $42.9 \pm 4.8\%$ outside) (Fig 12). HC cover inside the MPA boundary has decreased by 4% from 2013 to 2018. In contrast, outside the MPA, cover has increased by 10.1%. Spatially, the Napantao reef inside the MPA had $11.2\% \pm 1.9\%$ more HC cover than outside from 2013-2018. The percentage HC cover was significantly affected by the independent effects of Location (ANOVA, $F_{1,252} = 21.025$, $p < 0.001$) and Year (ANOVA, $F_{5,252} = 2.955$, $p < 0.05$), however no significant Location x Year interaction was found (ANOVA, $F_{5,252} = 0.716$, $p = 0.612$).

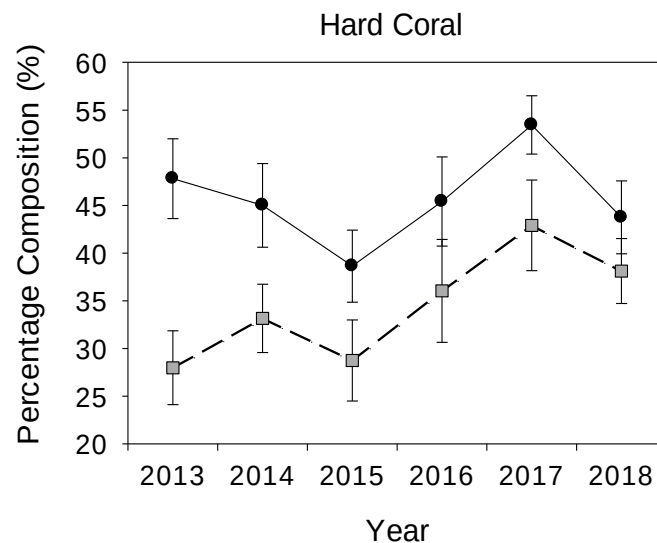


Figure 12. The percentage composition (%) of Hard Coral cover over a six year period from 2013 to 2018 at the Napantao marine protected area (MPA), southern Leyte, Philippines. The data is represented in two sets; inside the MPA boundaries (Black filled circle); and outside the MPA boundaries (Grey filled square).

Discussion

Coral Reef Associated Fish Community

The Napantao reef saw an overall increase in mean fish abundance from 2013 to 2018. A significant Year effect on fish abundance indicated that varying levels of local stressors to the reef between the years were impacting the respective abundances. Increases in abundance outside the MPA over the study period, likely due to a spill-over effect from the MPA (Halpern, Lester and Kellner, 2009), indicate the MPA is to some extent successfully increasing local fishery production. The decrease in reef fish abundance from 2014 to 2015 both inside and outside the MPA may be attributed to the major coral damaging storms that occurred in late 2014. Decreases in hard coral cover on the Napantao reef were recorded from 2014 to 2015, hence a reduction in coral reef habitat likely led to the immigration of reef fish to less affected areas (Halford *et al.*, 2004) away from the relatively shallow survey sites susceptible to damage via increased wave action. A significant Location effect on fish abundance in addition to a higher average abundance inside the MPA from 2013-2018 suggests the MPA is successfully providing protection from direct local fishing pressures. However, there have been reports from local staff working at the CCC base of illegal fishing inside the MPA at night. Poaching is considered low due to the night time placement of Bantay Dagats at the MPA, but this cannot be considered conclusive and unrecorded fishing pressures may hinder the MPAs success (Russ and Alcala, 2004). Overfishing outside the MPA could indirectly negatively impact the fish abundance on a larger spatial range (e.g. inside MPA) than just the directly affected area through changes to specific species populations (Priede *et al.*, 2011; Godbold *et al.*, 2013). However, the abundance results inside the MPA suggest otherwise in this case.

Species richness and diversity were both higher inside the MPA than outside from 2013-2018 and were significantly affected by an independent Location effect, suggesting a positive MPA effect. Greater H' values often represent a healthier reef system; higher diversity typically improves a reefs resilience to anthropogenic and non-anthropogenic stressors (e.g. overfishing and acidification) (Chapin *et al.*, 2000). Therefore, the area of reef inside the MPA boundary can be considered healthier and more resilient to stress. The insignificant Year effect on diversity suggests that the coral reef fish community

diversity on the Napantao reef plateaued across the study period. This may suggest that the benthic coral habitat has remained optimal for reef fish biodiversity during the study period. If a significant decrease in HC cover were observed we would have expected a drop in diversity (Pratchett *et al.*, 2011). This result may support that of Pelletier *et al.* (2005), whereby species richness was not sensitive to MPA status. The use of a pre-set fish community to be recorded during surveys instead of a continuous dataset is also likely to have influenced the Year effect.

The coral reef fish community compositions at the Napantao Reef were significantly different between inter-boundary locations. The community outside the MPA had a higher variability from 2013-2018, indicating a more stable population inside the MPA. The primary discriminating species behind the differences in communities belonged to the Caesionidae and Apogonidae families. This was reflected in their dominance of the reef fish abundance: Caesionidae had the highest recorded abundance inside the MPA boundaries across all years except 2016, and outside the MPA in 2014/15. Where Caesionidae did not have the highest abundance, Apogonidae took its place. Caesionidae predominately exhibited schooling behaviour on the Napantao reef during survey times, likely contributing to this observed dominance in abundance. Surveys always took place during the day, on average at 10:07 UTC +8, therefore this schooling behaviour is indicative of their diurnal planktivorous feeding modes (Carpenter, 1988). Caesionidae populations are often considered to be under threat from artisanal fishing practices, hence their absence would suggest intense fishing (Campbell and Pardede, 2006). However, artisanal fishing at the Napantao MPA is recorded as low so their dominance inside the MPA was expected. Apogonidae; primarily dominating in abundance outside the MPA were most often observed during surveys to be hiding amongst branching corals. This observed behaviour is supported by their nocturnal feeding habits on coral reefs and the fact they tend to rest during the day (Marnane and Bellwood, 2002). They are believed to exhibit a high degree of fidelity to their resting places and often form large aggregations. Gardiner and Jones (2005) determined these aggregations to be dependent upon the coral benthic substratum, presenting results indicating Apogonidae's preference to branching corals, specifically *Porites cylindrica* (a non-Acropora branching coral) (Edinger and Risk, 2000). Apogonidae dominance at both inter-boundary locations can be attributed to hard coral representing the highest %

cover of benthic substrate at the Napantao reef across the study period. Apogonidae dominance from 2016-2018 outside the MPA could be related to the increase in HC cover observed over this period. Actual Apogonidae abundances may potentially be higher than recorded by surveyors due to difficulties experienced in locating all individuals when hidden amongst branching corals. Hard corals (including branching coral morphology forms) often form a complex 3-dimensional structure, and reefs with high proportions of this coral cover such as the Napantao reef can be considered ‘complex’; a state beneficial to a wide range of coral reef associated fish species (Saila, Kocic and McManus, 1993; Gardiner and Jones, 2005).

Chaetodontidae

Chaetodontidae, N , S and H' were significantly affected by the independent effects of Location, with all displaying higher values inside the MPA than outside across the study period. This further indicates the MPA is providing a buffer zone against local stressors to reef fish. The insignificant Year effects on S and H' were expected as only 7 *Chaetodon* sp. are recorded during surveys, all of which are commonly seen during dives on the Napantao reef. For further context, in 2018 over four roaming dives inside the Napantao MPA at a depth range between 0-14m, 25 *Chaetodon* sp. were recorded. Therefore, surveys recording a larger number of *Chaetodon* sp. are required to reliably determine the Year effects on species richness and diversity. *C. baronessa* was the most abundant Chaetodontidae species across the study period at both inter-boundary locations and SIMPER analysis results deemed *C. baronessa* to be a dominant species on the Napantao reef at both inter-boundary locations. This was reflected in the significant effect of Feeding type on Chaetodontidae abundances; *C. baronessa* is an obligate corallivore (Pratchett *et al.*, 2008); a species dependent upon coral as its primary food source, often constituting >80% of its diet (Cole and Pratchett, 2011). Its abundance is often used as a proxy for coral reef health due to its highly selective prey preference. In abundance, it will feed almost exclusively on the fast growing *Acropora* coral (Edinger and Risk, 2000); *Acropora hyacinthus* (Berumen, Pratchett and McCormick, 2005). Obligate *Chaetodon* sp. were significantly higher in abundance across the study period than facultative *Chaetodon* sp. both inside and outside the MPA, which indicates Napantao reef has sufficient hard coral cover to sustain a substantial obligate Chaetodontidae population (Pratchett,

Wilson and Baird, 2006). HC cover increased outside the MPA from 2015-2018. The significant Year and Feeding type x Year effects on Chaetodontidae abundance in addition to the increase in abundance outside the MPA from 2015 to 2018 provides further evidence that HC cover contributes to their success.

Commercially Important Fish

Scaridae, Serranidae and Lutjanidae are all vitally important to the small scale fishery present at the Napantao reef, through both their high commodity and subsistence values to local fishers. The number of local fishermen/women has approximately doubled between 2013 and 2018, therefore it is essential to understand the population's abundance, biomass and size structure at both inter-boundary MPA locations for the continued conservation of these commercially exploited fish.

Serranidae

Serranidae abundance was significantly affected by Year. Abundance increased in 2018 inside the MPA but mainly plateaued across the study period for all previous years at both inter-boundary locations. The significant independent effects of Location are shown by a higher average abundance inside the MPA. The Napantao MPA successfully maintained a fairly constant Serranidae density over the study period and abundance inside the MPA is now moving in a positive direction. In addition, no declines in Serranidae density outside the MPA were observed. This could indicate a spill-over effect from the MPA (McClanahan and Mangi, 2000). However, Serranidae have been shown to be mobile fish; many species of which have large home ranges (e.g. *Plectropomus leopardus*) reaching 1.8ha and travelling up to 2km (Samoilys, 1997b, 1997a; Zeller, 1997). This presents issues of reduced observability to surveyors and increases the probability of repeated counts of the same individuals.

Serranidae had the lowest average *W* and biomass of the commercial fish. Both were significantly affected by the independent effects of Year. The increase in individual Serranidae *W* and mean biomass in 2018 away from the plateau indicates a larger and hence more mature population is present in 2018. This is further supported by the size class compositions which showed 2018 inside and outside the MPA to have the highest proportion of Serranidae within the 11-20 and 21-30cm size classes, in addition to

the lowest proportion of 0-10cm fish. Consequently, the Napantao reef across both inter-boundary locations is hosting a more mature Serranidae population in 2018 than 2013 which promotes their survivorship (Unsworth *et al.*, 2007). Species-specific methodology was not undertaken, however it can be assumed that this current, larger (TL (cm)) population will have a greater proportion of sexually mature adults (Rhodes and Tupper, 2007). Serranidae biomass was higher inside the MPA than outside from 2013-2018 and was significantly affected by Location; evidence of the fishing effects outside the MPA. The results suggest that the MPA has successfully provided protection for juvenile Serranidae to reach maturity and maintain a constant level of biomass within local fishing areas through overspill (McClanahan and Mangi, 2000). This is important for the future prosperity of this fish family; considering their slow growth and tendency to aggregate increases their vulnerability to unsustainable removal by overfishing (Colin, 1992; Sadovy and Colin, 1995; Unsworth *et al.*, 2007).

Scaridae

Scaridae had the highest mean abundance across the study period compared to Serranidae and Lutjanidae. The abundance was significantly affected by Year, Location and Year x Location. Abundance inside the MPA was on average higher than outside and has increased annually from 2015-2018, indicating a positive MPA effect. However, Scaridae abundance outside the MPA showed no clear trends and is likely effected by the varying fishing pressures on any given year (Campbell and Pardede, 2006).

Scaridae represented the greatest biomass over the study period both inside and outside the MPA, however the biomass appears to be primarily dominated by smaller individuals, supported by the relatively low *W* values. Scaridae *W* and biomass, were both significantly affected by Year, and both decreased over the study period. In addition the proportion of Scaridae of size class 0-10cm is greater in 2018 than 2013, whilst larger size classes saw a reduction. This indicates that from 2013-2018 Scaridae at the Napantao reef shifted towards a more sexually immature and small species dominated population. Most likely from overfishing due to their high social and commercial value to local fishers. The significant interaction effect of Location x Year on *W*, indicates larger individuals are removed

from the reef soon after maturity and potentially leaving the MPA, however the possibility of unrecorded poaching from inside the MPA cannot be dismissed.

Juvenile Scaridae inhabit the epilithic algal matrix (EAM), predated on cryptobenthic invertebrates (Welsh, Goatley and Bellwood, 2013). Welsh, Goatley and Bellwood (2013) reported a rapidly increasing somatic relationship with home-ranges until individuals reach 10 – 15cm (TL) at which point, this relationship breaks down. The rapid increase is thought to be prey and resource availability driven. Napantao reef has been shown to have a higher invertebrate abundance than other coral reefs in Sogod Bay (Seraphim *et al.*, 2016). However, a population dominated by smaller individuals utilising a niche resource, may force the increase of home ranges. This may suggest why a large proportion of 0-10cm individuals are observed outside the MPA. This increase in home range and cosmopolitan activity increases their susceptibility to surveys, potentially resulting in multiple count errors.

Lutjanidae

Lutjanidae had the lowest average abundance of the commercial fish from 2013-2018 and was significantly affected by the independent effects of Year and Location. The abundance plateaued across the study period at both inter-boundary locations, however, was higher inside the MPA than outside. The MPA appeared to provide some protection from fishing pressure, however has not succeeded in improving fishery production.

Lutjanidae *W* and biomass were significantly affected by the effect of Year; both decreasing from 2013-2018. The decreasing biomass may be a result of increased agnostic behaviour in adult Lutjanidae and increased home-ranges (Nanami and Yamada, 2008). Hence reducing observability to surveyors. The reduction in *W* highlights a need for improved conservation of Lutjanidae, especially inside the MPA to ensure individuals are able to reach sexual maturity and continue supporting the local fishery. Initially, both the *W* and size class distribution appear highly varied across the study period. Dallison and Ferguson (2017) suggested this may be due to the non-species specific methodology. However, the size class contribution results for Lutjanidae at both inter-boundary locations may be interpreted to show a population experiencing an approximate 3 year cycle whereby individuals transition from small juveniles to larger adults which are then removed from the reef either through fishing practices or

increased home ranges. This trend seems to have occurred twice over the six year study period. However, Lutjanidae are least active during the day when surveys took place and are most active during feeding periods at dusk and dawn (Kamukuru and Mgaya, 2004). Consequently, a longer study period using a species-specific methodology that takes into account behavioural patterns would be required to confirm this potential cycle.

Limitations

Although providing a contextualised review of coral reef fish community abundance and diversity, the methods utilised have their limitations. Assessing the fish community's full structure is difficult when utilising SCUBA protocols for UVC transects; they involve a small soak time therefore not accounting for semi-diurnal changes in species-specific behaviour. Surveys are conducted during the day but fish activity is often minimal between 09:00 and 16:00 (Chapman and Kramer, 2000).

Data handling for this study incorporated seasonality, aiming to limit impacts of spawning migrations (Sale *et al.*, 2010), ontogenetic shifts, habitat selectivity (Mcafee and Morgan, 1996) and home range relocations (Chapman and Kramer, 2000). Therefore, the true impacts on abundance and diversity cannot be holistically accounted, so may serve as justification for variations over the study period and between inter-boundary locations.

The limitations are accepted due to diurnal fishing in the area, and coral reef biodiversity (Bellwood, Hoey and Hughes, 2012), in addition to CCC protocols in place. The citizen science structure of CCC makes it unfeasible to address the above limitations.

Conclusions

The results of this study indicate the Napantao MPA to be maintaining coral reef fish abundance and providing an over-spill effect to areas outside the MPA boundary. Reef fish abundances inside the MPA appear to be increasing in recent years; future studies will show if this positive outlook continues. However, the MPA has not met its target of increased fishery production as the effects of overfishing outside the MPA outweigh the spill-over of mature Serranidae, Scaridae and Lutjanidae from the MPA. Consequently decreases primarily in Scaridae and Lutjanidae average biomass and individual weight

over the study period have resulted in a more sexually immature population. The implementation of species-specific size catch restrictions in areas outside the MPA no-take boundary may increase the proportions of exploited fish that reach a stage of sexual maturity, hence ensuring the long-term sustainability of the fishery. The MPA is host to a more diverse and temporally less variable community, indicating a healthier and more complex reef, more resilient to local stressors than the surrounding unprotected areas. I believe that a significant proportion of community variability outside that of fishing pressure at the Napantao reef is due habitat effecting stressors (e.g. coral damaging storms) and inter/intra species-specific competition.

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Appendix A

Table A1. Table containing the GPS co-ordinates for the Napantao MPA boundary corner points, and the permanent stakes on the reef marking transect mid points used by Coral Cay Conservation surveyors during Underwater Visual Census (UVC) SCUBA surveys of the Napantao reef.

MPA Boundary Corner Points	Latitude	Longitude
North East	10° 3' 57.20"N	125° 8' 8.50"E
North West	10° 3' 57.40"N	125° 8' 5.20"E
South East	10° 3' 41.40"N	125° 8' 5.20"E
South West	10° 3' 41.60"N	125° 8' 1.90"E
Transect	Latitude	Longitude
Inside 1	10° 3' 45.22"N	125° 8' 2.99"E
Inside 2	10° 3' 53.11"N	125° 8' 5.86"E
Inside 3	10° 3' 54.62"N	125° 8' 4.79"E
Outside 1	10° 4' 1.95"N	125° 8' 3.13"E
Outside 2	10° 4' 8.28"N	125° 8' 0.42"E
Outside 3	10° 4' 13.42"N	125° 7' 56.84"E

Appendix B

Table B1. Table containing the target fishes Coral Cay Conservation surveyors recorded during underwater visual census (UVC) SCUBA transects (Napantao Reef, Philippines, 2013-2018).

Common Name	Latin Name
Angelfish	Pomacanthidae
Barracuda	Sphyrnidae
Butterflyfish*	Chaetodontidae
Cardinalfish	Apogonidae
(Damselfish)	(Pomacentridae)
Anemonefish	<i>Amphiprion sp.</i>
Sergeant Damselfish	Pomacentridae
Emperor	Lethrinidae
Fusilier	Caesionidae
Goatfish	Mullidae
Groupers	Serranidae
Jack/Trevally	Carangidae
Lionfish	Scorpaenidae
Lizardfish	Synodontidae
Moorish Idol	<i>Zanclus cornutus</i>
Moray Eel	Muraenidae
Parrotfish	Scaridae
Pufferfish	Tetraodontidae
Rabbitfish	Siganidae
Ray	Rajiformes
Sandperch	Pinguipedidae
Snapper	Lutjanidae
Spinecheeks	Nemipteridae
Squirrelfish/Soldierfish	Holocentridae
Surgeonfish	Acanthuridae
Unicornfish	<i>Naso sp.</i>
Sweetlips	Haemulidae
Toby	Tetraodontidae
Triggerfish	Balistidae
(Wrasse)	(Labridae)
Crescent Wrasse	<i>Thalassoma lunare</i>
Humphead Wrasse	<i>Cheilinus undulatus</i>
Red Breasted Wrasse	<i>Cheilinus fasciatus</i>

Table B2. Table containing the species-specific Butterflyfish (Chaetodontidae) that Coral Cay Conservation surveyors recorded during underwater visual census (UVC) SCUBA transects (Napantao Reef, Philippines, 2013-2018). Also includes their individual feeding types: Obligate or Facultative coralivores.

Common Name	Latin Name	Feeding Type
Eastern Triangle Butterflyfish	<i>Chaetodon baronessa</i>	Obligate
Redfin Butterflyfish	<i>Chaetodon trifasciatus</i>	Obligate
Vagabond Butterflyfish	<i>Chaetodon vagabundus</i>	Facultative
Klein's Butterflyfish	<i>Chaetodon kleinii</i>	Facultative
Raccoon Butterflyfish	<i>Chaetodon lunula</i>	Facultative
Long-Nosed Butterflyfish	<i>Forcipiger flavissimus</i>	Facultative
Humphead Bannerfish	<i>Heniochus varius</i>	Facultative
Other Butterflyfish	<i>Chaetodon sp.</i>	Other

Table B3. Table containing the Length-Weight allometric coefficients used to determine the weight (g) of sized (TL, (cm)) Grouper, Snapper and Scaridae from 2013 to 2018 at the Napantao reef, Philippines. Coefficients calculated by Dallison and Ferguson 2017.

Genera	Common Name	Allometric Coefficients	
		a	b
Serranidae	Grouper	0.0145	3.0225
Lutjanidae	Snapper	0.0147	3.0312
Scaridae	Parrotfish	0.0152	3.0398

Table B4. Table containing the target benthic substrates Coral Cay Conservation surveyor's record during point-intercept (PIT) SCUBA transects (Napantao Reef, Philippines, 2013-2018).

Substrates		
Soft Coral	Sponge	Recently Killed Coral
Rock	Silt/mud	Rubble
Sand	Nutrient indicator algae	Other*
Hard Coral Lifeforms:		
Acropora branching	Acropora encrusting	Acropora submassive
Acropora digitate	Acropora tabulate	Non-Acropora branching
Non-Acropora encrusting	Non-Acropora foliose	Non-Acropora submassive
Non-Acropora mushroom	Heliopora (blue coral)	Millepora (fire coral)
Tubipora (organ-pipe coral)		
Other:		
Anenome	Corallimorph	Halimeda
Tunicate	Zoanthid	Gorgonian
Hydroids		

Appendix C

Table C1. Table containing the maximum abundances observed during a single 20m replicate, of five dominant coral reef fish recorded by Coral Cay Conservation surveyors at the Napantao Reef, Philippines from 2013 to 2018 inside (In) and outside (Out) the MPA boundaries.

Common Name (Latin name)	2013		2014		2015		2016		2017		2018	
	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
Fusilier (Caesionidae)	150	50	600	580	500	98	130	50	1000	21	811	85
Cardinalfish (Apogonidae)	101	55	71	83	30	44	125	128	89	340	60	232
Easter Triangle Butterflyfish (<i>Chaetodon</i> <i>baronessa</i>)	17	19	10	12	8	16	16	13	21	17	13	21
Parrotfish (Scaridae)	43	36	30	22	13	10	27	55	52	16	26	17
Angelfish (Pomacanthidae)	17	14	17	28	14	13	18	18	31	17	19	16