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**Thermal induced shift in abundance of
Scleractinian and Porifera of Napantao Reef,
Sogod Bay, Philippines**

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LIST OF ABBREVIATIONS

Marine Protected area	MPA
Intergovernmental Panel on Climate Change	IPCC
National Oceanic and Atmospheric Administration	NOAA
El Niño-Southern Oscillation	ENSO
Coral Cay Conservation	CCC
Air Temperature	AT
Sea Surface Temperature	SST
Water temperature at depth of 10 m	WT10

ABSTRACT

The spiralling effect of climate change is further becoming evident with increasing reports of severe and frequent bleaching of global coral reefs. Following the recent El Niño (2015-2016) further extreme bleaching events were added to records. Recent studies have suggested that sponges may take advantage of such events and come to dominate reefs. This study aims to investigate this hypothesis by examining coral and sponge abundance in a background of varying thermal stress. Furthermore, whether a Marine Protected Area (MPA) will present evidence of increased resilience of the coral reef. Data were collected from Barangay Napantao, Southern Leyte, Philippines, both inside and outside an already well established MPA, over a period of four years. Results show that sponges increase in abundance during induced thermal change. However, corals also demonstrated a capability to persist and recolonise when conditions had returned to a stabilised thermal favourable condition. In addition, the results showed that both corals and sponges overall thrived better within the MPA. Moreover, corals had a better chance of survival during stressful conditions when located inside the MPA. In conclusion, sponges tend to expand coverage on a stressed coral reef. But, corals also exhibit a level of resistance in spatial competition when conditions are favourable. Further investigation should be conducted to fully understand; (1) the importance of clades of zooxanthellae; (2) the difference, and impact, amid the unique holobiont of Scleractinian and Porifera and; (3) the importance of change in temperature contra the rate at which the change occurs.

INTRODUCTION

In 2001, the Intergovernmental Panel on Climate Change (IPCC) predicted that the following 20 years would be an increasingly stressful time for the coral reefs of our planet (Houghton *et al.*, 2001). This statement came three years after the first ever recorded global bleaching event, following the 1998 El Niño (Wilkinson and Hodgson 1999, Glynn *et al.*, 2001). They suggested that due to climate change, the intensity, frequency and impact of already occurring extreme weather phenomena would increase, whilst the general weather would likewise experience change. Possibly resulting in increased occurrence of large bleaching events (Ahmad *et al.*, 2001). According to National Oceanic and Atmospheric Administration (NOAA) another severe bleaching followed in 2010 (Heron *et al.*, 2016). The 2015-2016 El Niño brought yet another of such devastating event. Each event with greater, and more severe, impact than previously recorded (Eakin *et al.*, 2016). So why is this a matter of concern?

Coral reefs are often referred to as the rainforests of our oceans, due to the high total net value of released oxygen into the atmosphere, produced by the associated photosynthetic phytoplankton. In addition, the calcium carbonate (CaCO_3) skeleton produced by the coral polyp, provides an important architecturally complex, giving rise to an array of different habitats for numerous associated species to use for foraging, shelter and interaction. The high level of species diversity and abundance provides the surrounding human population with a reliable source of protein and income, allowing a profitable livelihood (Ahmed and Chong 2004, Cinner *et al.*, 2009). Furthermore, coral reefs provide a natural barrier, protecting coastal communities from waves and storms, hereby offering a valuable variety of important ecosystem services to mankind (Costanza *et al.*, 1997, Moberg and Folke 1999). Stating this great importance of coral reefs is argued without even considering the philosophic aspect of an essential intrinsic value.

Corals are divided into two groups, soft and hard coral. Soft corals (Alcyonacea) are ahermatypic organisms, meaning they do not build a characteristic CaCO_3 skeletal structure (Stanley and Fautin 2001), unlike hard corals (Scleractinian). Soft corals are generally found in colder and/or deeper waters (Roberts *et al.*, 2006). Whereas, in contrast, reef building hermatypic hard corals predominantly are found in warmer water, at a shallower depth. The survival of hard corals depends on the symbiotic relationship with the alga (Muscantine & Porter 1977; Falkowski *et al.*, 1984), zooxanthellae (*Symbiodinium spp.*), which inhabits the endodermic layer of the tissue of the coral polyp. It is due to this symbiosis, that the hard corals are commonly located in shallow euphotic water, at depths no deeper than 70 m, and prefer warmer water, with a species specific optimum temperature ranging from 18-30 °C (Hoegh-Guldberg 1999), with most species preferring temperatures between 23–29 °C (Cesar *et al.*, 2014). Thus, hard corals are predominantly found along the equatorial line in mostly nutrient poor water (Fig. 1).

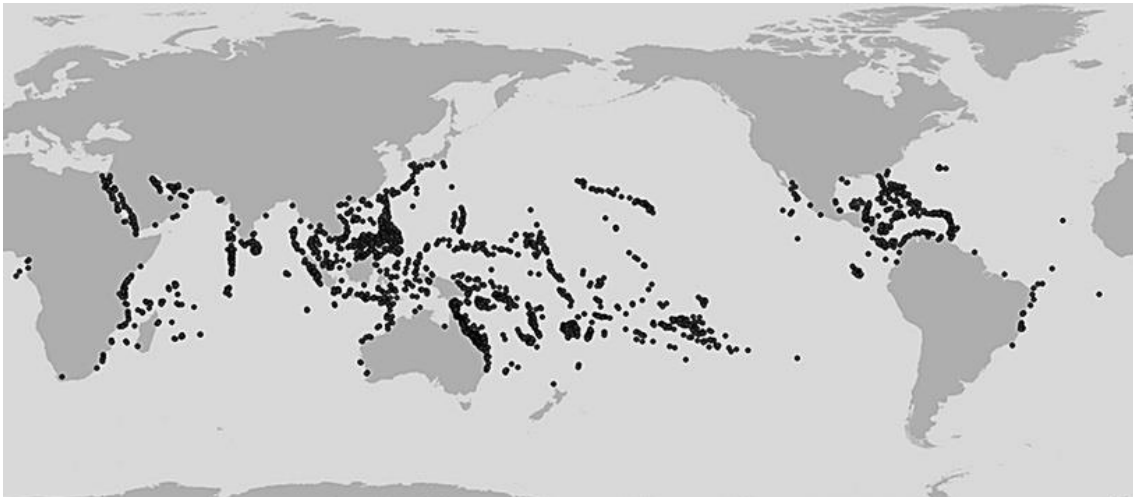


Figure 1: Worldwide distribution of coral reefs (Data source: National Ocean Service 2014).

The greatest diversity amongst hard coral is found inside the coral triangle (Malaysia, Papua New Guinea, Indonesia, Philippines, Solomon Islands, and Timor-Leste), with

605 zooxanthellate corals present in the area, a sum of 76 % of all known species (Fig. 2) (Veron *et al.*, 2009; Renema *et al.*, 2008).

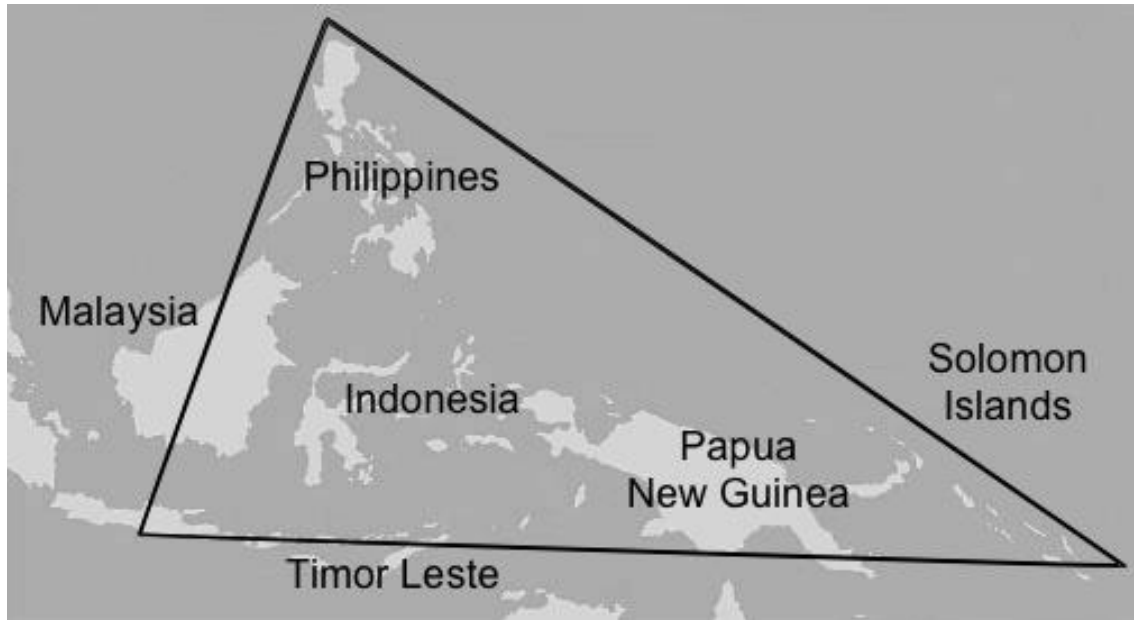


Figure 2: The Coral Triangle, scientific boundaries covering the area of Malaysia, Philippines, Solomon Islands, Papua New Guinea, Timor Leste and Indonesia, holds the largest diversity of global coral reef, with evidence of some of the oldest coral species present (Source: Veron *et al.* 2009).

A favourable temperature for hard corals can furthermore vary between reef (Maynard *et al.*, 2015), depending on their normal baseline temperature, some species exhibiting greater skills of adaptation to thermal change (Oliver and Palumbi 2009). Although, for most species, a change of more than $\pm$ one degree can instigate great stress. This level of tolerance is likely caused by the mutualistic symbiosis with the associated algae (Berkelmans and van Oppen 2006). During favourable environmental conditions the coral host and the zooxanthellae function in harmony, and the algae will provide its host with any excess glucose produced in exchange for shelter. In comparison, when the coral polyp becomes stressed, the symbiotic relationship is at risk of a break down (polyp expulsion of the algae) leading to coral bleaching. This is considered as a survival

mechanism of the polyp to overcome great stress (Fujise *et al.*, 2014). Recent research has demonstrated that the different clades and sub-clades of zooxanthellae display a variation of tolerance to heat and light stress (Stat *et al.*, 2006, Silverstein *et al.*, 2012). By expelling the already present zooxanthellae, the coral may be able to free up space for more suitable algae, thus increasing chances of colony survival (Buddemeier and Fautin 1993, Glynn 1996, Buddemeier *et al.*, 2004). If the attempt fails, the host will starve and risk death. A further explanation to why bleaching occurs, is that the zooxanthellae in reaction to increased temperatures produces reactive oxygen species (Suggett *et al.*, 2008). Leading the algae to become toxic, causing oxidative damage, to both itself and its host, which trigger the coral polyps to terminate the symbiosis. A recent study has shown that some species of corals consist of a genetic composition that can produce proteins to neutralise this reaction, and thereby prevent the need for expulsion (Levin *et al.*, 2016). Further did the study show that the algae will shift from an asexual reproduction to sexual production, increasing changes of evolutionary adaptation to the change in temperature.

Another characteristic inhabitant of the reef systems is the sponge (Porifera). This organism also takes many shapes and forms and provides extra dimension to the reef (Bergquist 1978, Wilkinson 1983). Sponges are simple organisms with no true tissue, a collaboration of many independent cells making up the multicellular sessile organism that holds a species dependent structure through a diverse combination of organic (collagenous) and inorganic components (CaCO_3 , SiO_2). (Bergquist 1978, Gosliner *et al.*, 1996). Sponges rely on a continuous intake of water through incorporated canals of the sponge walls. Here nutrients are filtered by the flagellated cells, choanocytes. The water is released through a distinct opening(s), the osculum (Castro and Huber 2008). Like hard coral, some sponges initiate a symbiotic relationship with zooxanthellae (Hill *et al.*, 1996). In comparison to the hard coral/zooxanthellae symbiosis, limited research

is available on the sponge/zooxanthellae symbiosis. Studies show that most of these symbiotic relationships, are found almost exclusively within the bioeroding family of sponges, the Clionaidae. The algae inhabit the host organism at a cellular level, and maternally passed further via sponge gemmules or larvae. The zooxanthellae appear to assist sponges in the uptake of nutrients, enhance growth, support survival, affect the bioerosion rate (Hill *et al.*, 1996), and possibly aid in reproduction. In contrast to the stress sensitive hard coral, sponges appear to be more resilient toward environmental changes (Kelmo *et al.*, 2013). Studies have divergently shown a correlation of sponges increased growth rate with increased temperature (Rützler 2002, Koopmans and Wijffels 2008). Additionally, sponges can engage symbionts capable of maintaining a photochemical efficiency under upsetting thermal conditions (Schönberg *et al.*, 2008, Carballo *et al.*, 2013).

Previous study has suggested a shift from a coral dominated reef to a sponge dominated reef following environmental change (Bell *et al.*, 2013), triggered by climate change. This was investigated by examining the presence of sponge reefs in the geological record, inspecting reported shifts from coral to sponge reefs, and lastly, how the holobiont of the sponge would react, both directly and indirectly, to change in temperature and pH. From the evidence, they proposed; sponges could be one group that would benefit from climate change, hereby resulting in a change of some reefs structures causing important alterations in the overall function of reefs. Likewise, in 2013, another study suggested that sponges would be the one organism of all reef taxa to have the potential of being resilient enough to withstand any large-scale thermal events (Kelmo *et al.*, 2013). This hypothesis was proposed after having annually monitored three contrasting reefs at four different locations during the years 1995-2011. Overall results of the period showed no significant change in the diversity of sponges. But, density had significantly increased at all sites. The assessments also showed that the event of the 1997-98 El Niño-Southern

Oscillation (ENSO) had had no measurable impact on the sponge reef assemblage, suggesting that sponges will be less negatively affected by an increase in sea surface temperature.

The general understanding is, that Marine Protected Areas (MPA) provide a sanction to marine life, and may offer a tool for coral reefs to resist and recover from disruptions, such as climate change (West and Salm 2003, Mumby and Harborne 2010, Selig *et al.*, 2012). Studies have shown that five factors play a key role in the level of efficiency of a MPA; a well enforced no take policy in an area of >100 km² kept isolated by deep water or sand for more than ten years (Edgar *et al.*, 2014, Fletcher *et al.*, 2015).

The aims of this study are to investigate the probabilities of following null hypotheses: 1) changes in the water temperature will not affect reef composition in relation to coral/sponge abundance; 2) change in temperature will not lead to a shift from coral dominated system to a sponge dominated system; And 3) a MPA will not impact the abundance of the coral and sponge during stressful times.

METHODOLOGY

Study site

The coral reef investigated is based within the waters of Sogod Bay, Barangay Napantao, Philippines (Fig. 4), located in the province of Southern Leyte, population of 300,000 (Ferguson *et al.*, 2016) in Barangay. It consists of a narrow fringed reef, with intertidal flats and beaches. The centre of the bay reaches a maximum depth of 1400 m, which creates an ideal environment for great biodiversity of both smaller organisms, on the shallower coastal slope, and mega fauna such as pilot-whales and whale sharks further off shore. The area is also known for a great biodiversity in habitats, such as mangroves,

seagrass beds and coral reefs (Ferguson *et al.*, 2016). Sogod Bay is considered an important fishing ground to the surrounding population, with the Fisheries Sector Program of the Department of Agriculture (Lancaster *et al.*, 2016) declaring the bay to continuously be assessed and managed. This has led to 25 well established MPAs, covering an area of total 292 hectares, ranging between 3.5-55 hectares, with a mean average size of 11.7 hectares (± 2.2 SE) (Lancaster *et al.*, 2016).



Figure 4: Location of Philippines, Southern Leyte, Sogod Bay with Napantao reef.

The dataset used for this study has been collected by Coral Cay Conservation (CCC) between 2013-2016 at the Napantao reef system, as described in latest Coral Cay Conservation Triennial report (Dallison and Ferguson 2017). Surveys are only conducted by volunteers that have accomplished a minimum 90 % score of accuracy in species

identification.

The data were collected at two adjacent sites, one located within the MPA, the other outside, both on same reef. The Napantao MPA was established in 1996, and covering an area of 5 hectares, and located in the waters of barangay Napantao, with surrounding population of approximately 1069 people (Lancaster *et al.*, 2016). The MPA is respected overall, with a low level of poaching inside the MPA, whilst the surrounding waters are being fished on daily basis.

Study species

This study focuses on three main groups; *Acropora spp.*, non-*Acropora spp.* and sponges. The group of *Acropora spp.* was not species specific, and refers to all *Acropora spp.* observed at the two sites (Fig. 3a). This also applies to the group of non-*Acropora spp.* (Fig. 3b) which consists of varied hard corals, such as; *Porites spp.*, *Ctenactis sp.*, *Diploastrea sp.*, *Euphyllia sp.*, *Echinopora sp.*, *Favia spp.*, *Favites spp.*, *Galaxea sp.*, *Goniopora sp.*, *Alveopora sp.*, *Herpolitha sp.*, *Lobophyllia sp.*, *Mycedium sp.*, *Montipora sp.*, *Polyphyllia sp.*, *Pachyseris sp.*, *Pectinia sp.*, *Pocillopora sp.*, *Pavona sp.*, *Seriatopora sp.*, *Turbinaria sp.*, and *Tubastrea sp.*. The last group considered is the one of sponges (Fig. 3c), including all species observed. According to previous study conducted around the neighbouring island Cebu in Canigo Channel, 33 species of sponges were found in the area. Predominantly of the suborder Haplosclerida, of the order Desmosponges (Longakit *et al.*, 2005).

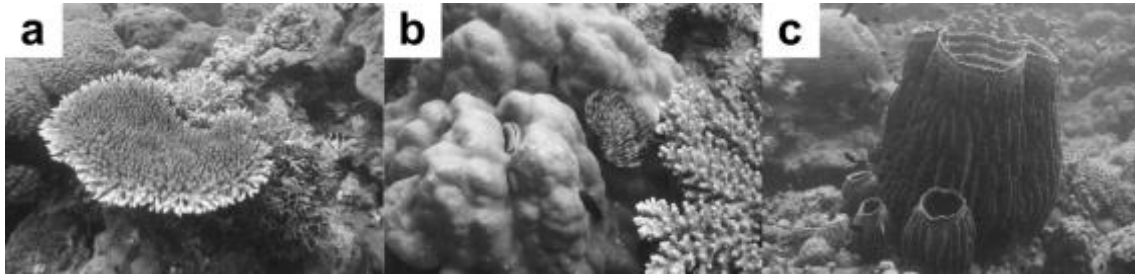


Figure 3: Examples of each group investigated in this study; (a) Branching *Acropora spp.*, delicate fast growing (b) Massive *Porites spp.*, slow growing solid colony (c) Barrel Sponge (Source: Alex Wady, Napantao Reef).

Survey Methodology

Data were collected by CCC, utilised by an augmented standardised Reef Check methodology. A transect of a length of 100 m, with 4x20 m marked replicates with 5 m gaps implemented between each section, was extended parallel with the coastal line across the coral reef. The data were collected within 24 hours, to limit environmental variation (Fig. 4).

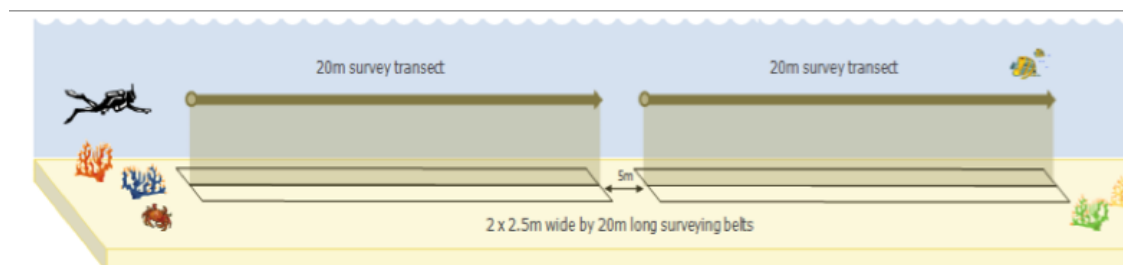


Figure 4: Setup of transect (Lancaster et al., 2016), consisting of four replicates of 20 m each, with 5 m gap along a length of 100 m.

Surveys were completed twice per year, in spring (March/April) and autumn (September/October/November). Data were collected from 12 replicates of inside and outside the MPA (n=24), by two teams of two scuba divers. Reef check request samples at depth of 6 m and 12 m, however, due to the topography (steep slope to depth of 10

m) of the site, data were only collected at 12 m. Temperatures (°C) were measured before each survey, air temperature (AT), sea surface temperature (SST), and at 10 m depth (WT10). For benthic composition, a plumb line was used at 50 cm within each of the 20 m replicates. Data were noted by a categorisation of *Acropora spp.* and non-*Acropora spp.*, with specification of structural form. If the plumb line marked any other type of substrate; such as sponge, algae, sand, rocks, rubble or recently killed coral, this was noted, without additional information. Additional data of impact on the coral reef, such as disease, coral bleaching and signs of pollution were noted by the volunteers moving in a distance of 2.5 m, one on each side of the transect, estimating the extent (% coverage) of the impact.

Statistic

All graphs were created in Microsoft® Excel for Mac, version 15.28, © 2016 Microsoft. IBM SPSS Statistics, Version 23 was used for statistical analyses. When relevant, an average with standard error was calculated. Comparing the non-parametric measures of temperatures, a post hoc analysis using two-tailed A Mann-Whitney U was used. Investigating a relation of hard coral abundance inside or outside per year or season, a two-way ANOVA was used, with risk of Type I error due to non-parametric trend of the data. When investigating a possible effect of year, site and genera on the abundance of hard coral, the data was split into the seasons, and a 3-way ANOVA was used. Pre-analysis, a normality test was run, which showed a non-normal distribution. Due to the many negatives in the data set, a transform was decided against and accepting the risk of another Type I error. To investigate the association between total of hard coral, and the *Acropora spp.* and non-*Acropora spp.* with change in water temperature at depth at 10 m a Spearman test was run on the non-parametric data. Data for sponge coverage likewise showed a non-parametric distribution and the Spearman correlation was used

to test the association of abundance of sponges per year inside and outside the MPA, the same was used when association with water temperature was investigated. Exploring the association between collectively hard coral and sponge coverage with water temperature another Spearman test was conducted. A regression was conducted on the association of level bleaching with water temperature. All data found in appendices 1-4.

RESULTS

Water temperatures at 10 m (2013 – 2016)

Year 2015 showed a lower total average of water temperatures (10 m) (27.17 °C, STDV $\pm$ 1.08) compared with 2013 (28.04 °C, STDV $\pm$ 1.09), 2014 (27.38 °C, STDV $\pm$ 1.48) and 2016 (27.90 °C, STDV $\pm$ 1.02) (Fig. 5). Highest level of variation in temperature was recorded during autumn 2015 (STDV $\pm$ 1.29). When using Mann-Whitney U to accept or reject a significant difference between each year, only the combination of 2013 and 2016, plus 2014 and 2015 showed no significant difference. All other years showed a significant difference ($P = .000-.019$). An average water temperature for all years come to total of 27.62 °C (STCV $\pm$ 1.23), with average spring (n=96) temperature at 26.76 °C (STDV $\pm$ 0.74), and autumn (n=96) 28.48 °C (STCV $\pm$ 0.99). Bringing autumn to be 1.72 °C warmer compared to spring season. A Mann-Whitney U test showed a significant difference ($U = 9.86$, ($P < .001$)).

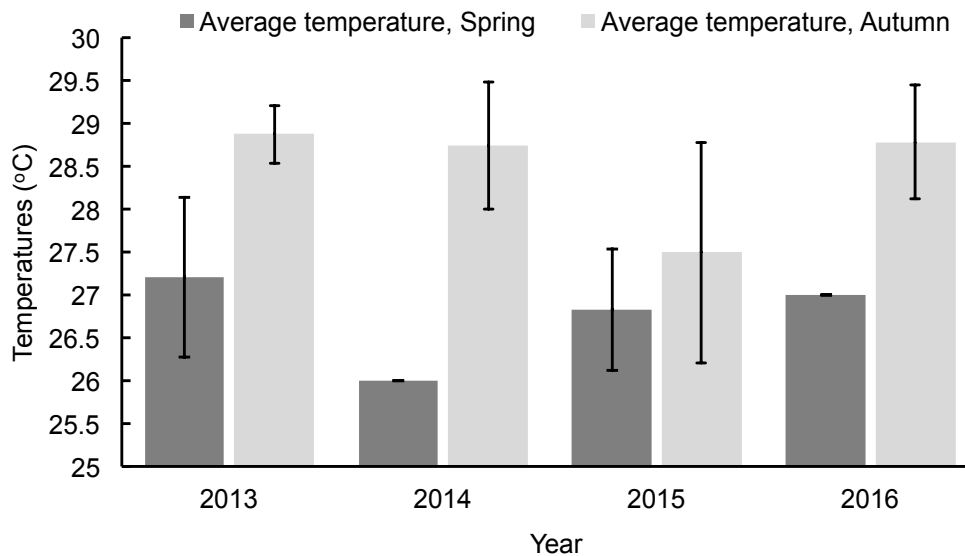


Figure 5: Average water temperatures (10 m). Temperatures were recorded between April 2013 and November 2016. Temperatures range between 25-30 °C. Autumn showing a 1.72 °C higher average temperature (28.48 °C), compared with spring (26.76 °C).

Abundance of hard coral

Investigating the impact of the MPA on abundance of hard corals, the inside and outside data was compared by spring (Fig. 6a) and autumn (Fig. 6b) per year. Spring show a total number of counts of 706 for all four years, with an average of 176.5 (STDV $\pm$ 20.57) counts per year, within the MPA. The total number of counts outside the MPA is 456, with an average of 114 (STDV $\pm$ 23.08) counts per year (Fig. 6a). During autumn, the counts inside the MPA is a total number of 741 counts for all transects over the four-year period, with an average of 185.25 (STDV $\pm$ 13.94) per season. The total counts outside the MPA add a total of 510, and an average of 127.50 (STDV $\pm$ 16.82) per season (Fig. 6b). Investigating the impact of site on the abundance a two-way ANOVA showed a significant difference ($P < .001$), whilst season and year only showed no significant difference (season $P = .324$; autumn $P = .400$)

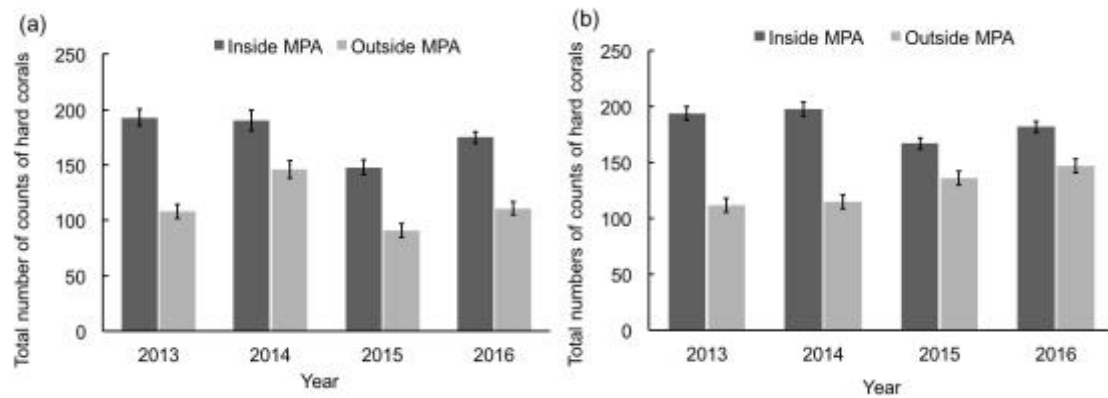


Figure 6: (a) Comparison of total counts of hard corals inside and outside the MPA during spring season, 2013-2016 (b) Equal comparison of total counts of hard corals inside and outside the MPA during autumn season, 2013-2016.

Abundance of *Acropora spp.* and non-*Acropora spp.*

At both sites, inside and outside of the MPA, the category of non-*Acropora spp.* showed the highest level of abundance in every observed year. Figure 7 illustrates that this category showed, for both seasons, a greater abundance within the MPA. *Acropora spp.* showed a similar trend, though level of abundance much lower at site outside the MPA. Result of spring (Fig. 7a) showed no significance ($P = .357$) in effect of year on abundance, whilst genera and site showed same significant effect on counts ($P < .001$). Same trend was present for autumn (Fig. 7b), year ($P = .952$), and both genera and site ($P < .001$).

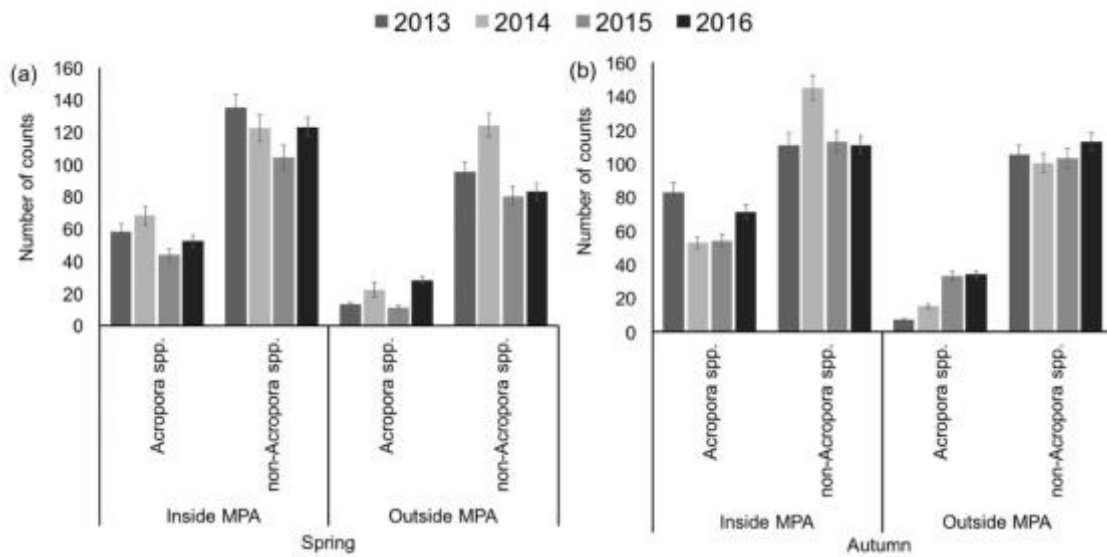


Figure 7: (a) Comparison of difference in abundance of *Acropora* spp. and non-*Acropora* spp. inside and outside the MPA area during spring, 2013-2016; (b) displaying same comparison for autumn, 2013-2016.

Hard coral coverage with change in water temperature

Overall, both *Acropora* spp. and non-*Acropora* spp. corals showed approximately same response when compared to change in water temperature at 10 m depth with 1) no consideration of whether the corals are inside or outside the MPA and 2) disregarding the change in season (Fig. 8). Testing for correlation between the genera and change in temperature the Spearman test showed no significant difference (*Acropora* $P = .168$, non-*Acropora* $P = .181$), ergo no association could be established between the coverage of the genera and change in temperature.

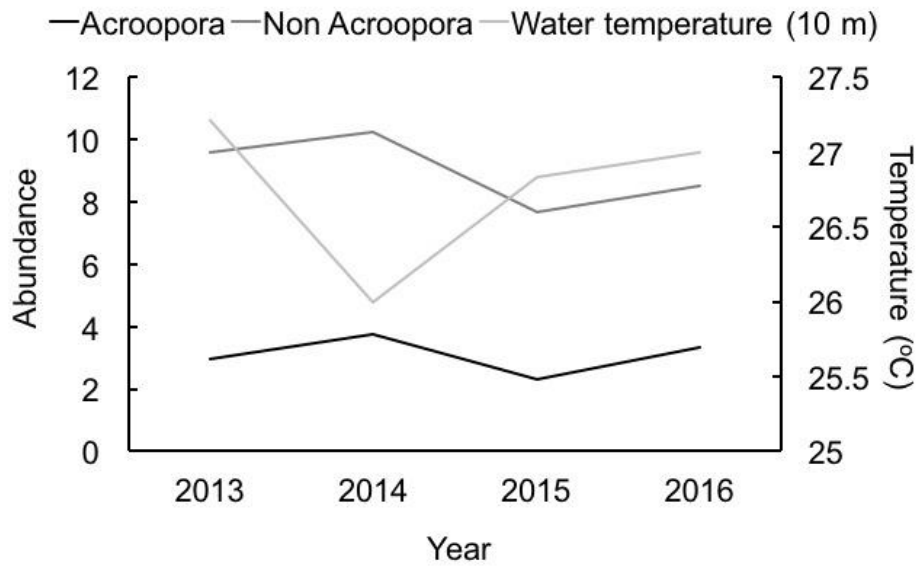


Figure 8: The correlation of *Acropora spp.* and non-*Acropora spp.* with change in water temperature at depth of 10 m.

When variables such as 1) whether the corals are present inside and outside the MPA and 2) difference in season, the graphs indicate that there is little difference in the abundance of corals depending on season. During autumn, both inside and outside the MPA *Acropora spp.* appear to recoup abundance faster than the non-*Acropora spp.* (Fig. 9). When investigating with a Spearman test, split file of location (inside or outside the MPA) and season (spring and autumn) the results showed no significant association with two groups of hard coral and water temperatures (10 m).

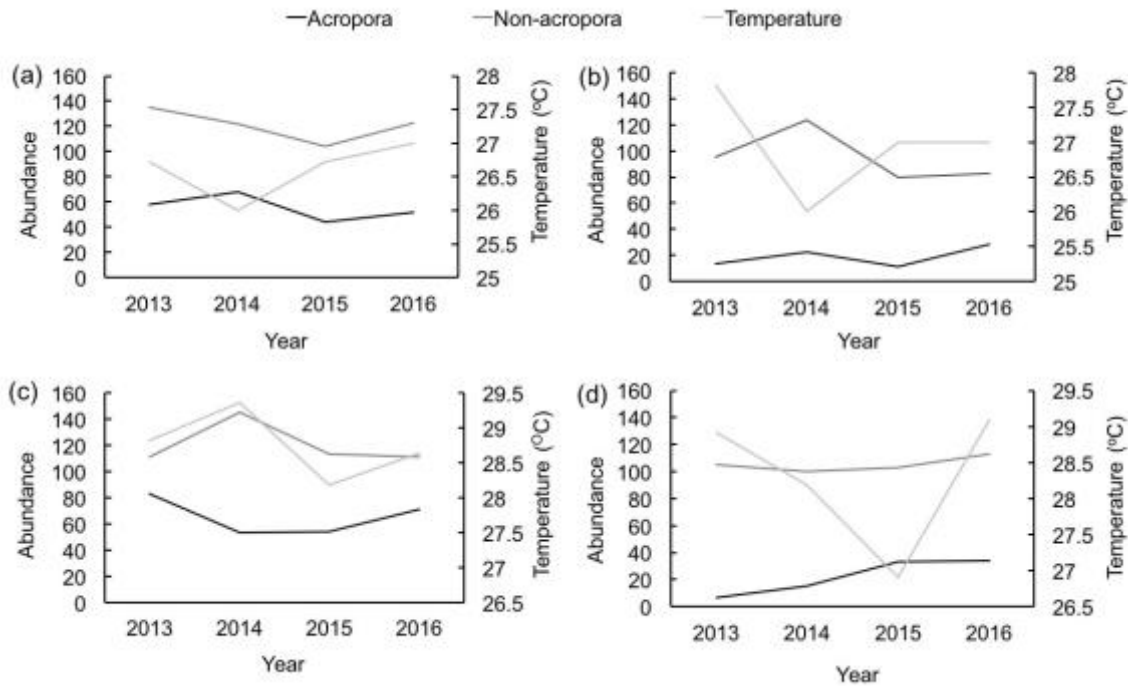


Figure 9: (a) Relation of abundance of *Acropora* spp. and non-*Acropora* spp. with change in temperatures during spring 2013–2016, inside the MPA ($P = .347$); (b) Same relation investigated for spring 2013–2016, outside the MPA ($P = .505$); (c) Relation of abundance of *Acropora* spp. and non-*Acropora* spp. with change in temperatures during autumn 2013–2016, inside the MPA ($P = .171$); (d) Same relation investigated for autumn 2013–2016, outside the MPA ($P = .732$).

Sponge coverage

The results show decline in abundance of sponges for spring outside the MPA (Fig. 10a), and autumn both inside and outside the MPA (Fig. 10b). Only spring inside the MPA show small increase. A Spearman correlation, for spring, showed no significant association between the counts of sponge coverage with year ($P = .285$) or site ($P = .537$). When investigating relations during autumn, there were no evidence of significant effect of site on abundance ($P = .548$), whilst the results for coverage per year did showed a significant association ($P < .001$).

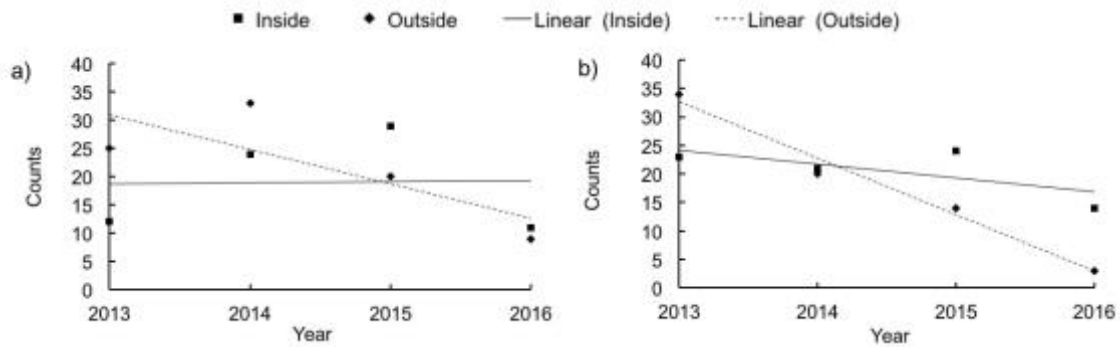


Figure 10: A comparison of the abundance of sponges inside and outside of the MPA per season; (a) spring and (b) autumn, 2013–2016.

When further investigating the data, and add a variation of temperature of water (10 m) only autumn, inside the MPA during 2014 showed a significant association ($P = .005$).

Relationship of hard corals and sponges

For both seasons, not considering the variable inside and outside MPA, the coral coverage declined rapidly during 2014 to increase in 2015 (Fig. 11). In spring 2016 (Fig. 11a), hard coral coverage had reached level as of 2013. Whilst sponge coverage show a continuous decline during 2016, reaching a lower level than of 2013. Autumn showed an even greater increase in coverage (Fig. 11b), exceeding previous levels. Hard corals show a regression in coverage during 2014 and 2015 for both seasons. Sponges show a trend of increase in same period. Using the Spearman correlation on the non-parametric data with split files of seasons, the correlation between coverage of hard coral and sponge can only be established as significant for spring of 2015 ($P = .045$) and spring 2016 ($P = .008$).

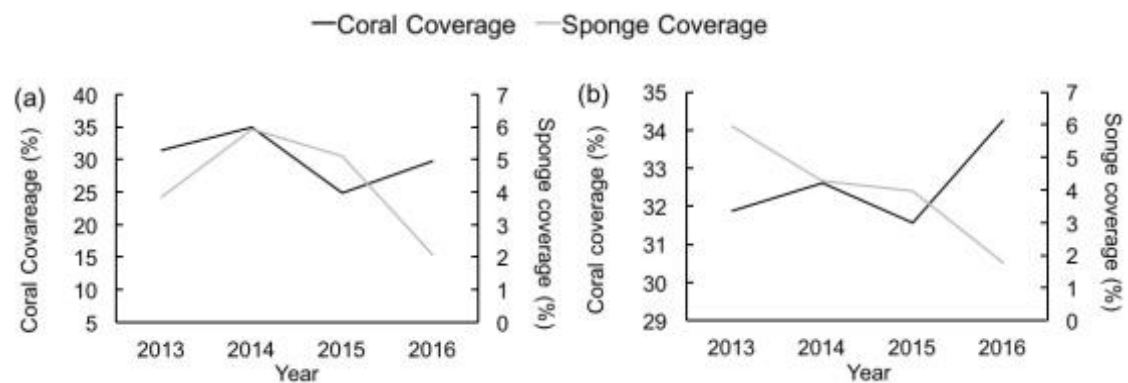


Figure 11: (a) Coverage of hard coral and sponge during spring season of the years 2013–2016 (b) Coverage of hard coral and sponge during autumn season of the years 2013–2016.

Relationship of hard coral and sponge abundance with temperature

Acropora spp. show a great variation in coverage outside the MPA for both spring and autumn (Fig. 12cd). In comparison inside the MPA a consistent abundance appears, again in both seasons (Fig. 12ab). The non-*Acropora spp.* show an almost continuous coverage independent of MPA, for both seasons. Sponges showed a greater level of consistent coverage inside the MPA. Whereas outside the MPA, a greater variance appeared, with the highest abundance during autumn 2013 and spring 2014. Lowest abundance occurs throughout 2016.

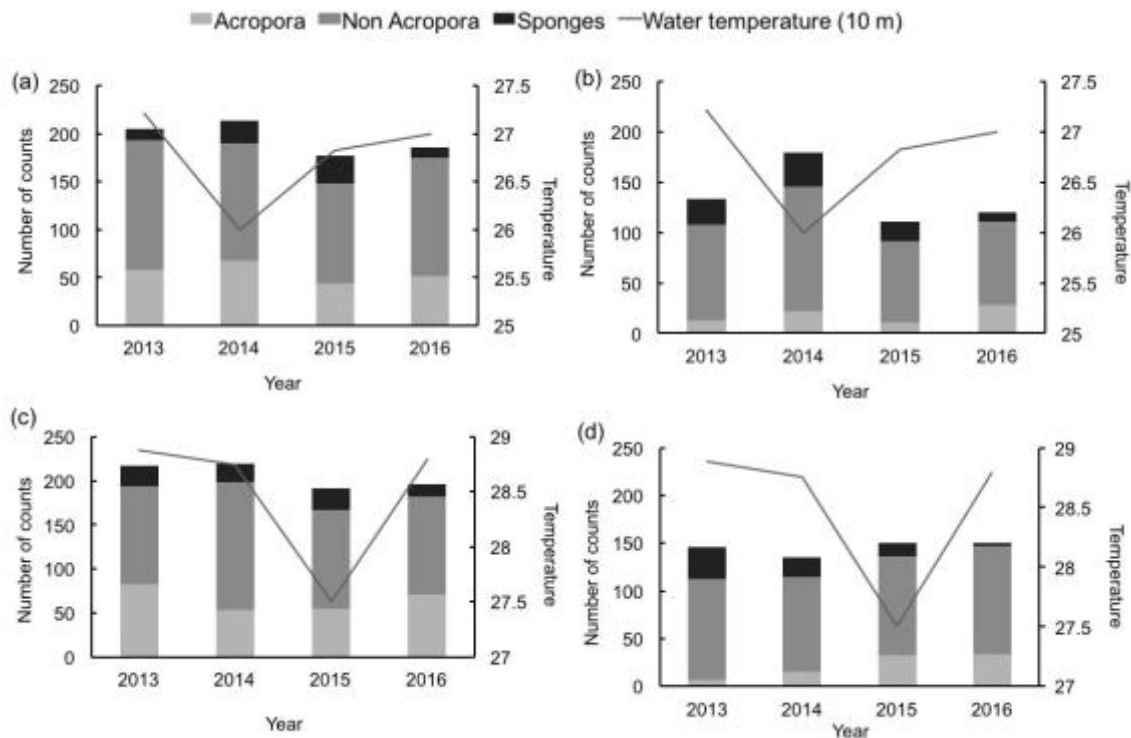


Figure 12: Correlation between abundance of *Acropora* spp., non-*Acropora* spp. and sponges with change in water temperature (10 m). (a) during spring inside the MPA, 2013–2016; (b) spring outside the MPA, 2013–2016; (c) during autumn inside the MPA, 2013–2016; (d) during autumn outside the MPA, 2013–2016.

Percentage cover of bleaching with water temperature

During spring, temperatures appear to be stable for the year 2014, 2015 and 2016 throughout the season, whilst the temperatures during autumn seem to vary for each year throughout the season. Regardless of the temperature being mostly stable for spring seasons, some level of bleaching still arises within the populations. Autumn season shows a great variability in temperature, whilst also showing a great difference in bleaching level (Fig. 13).

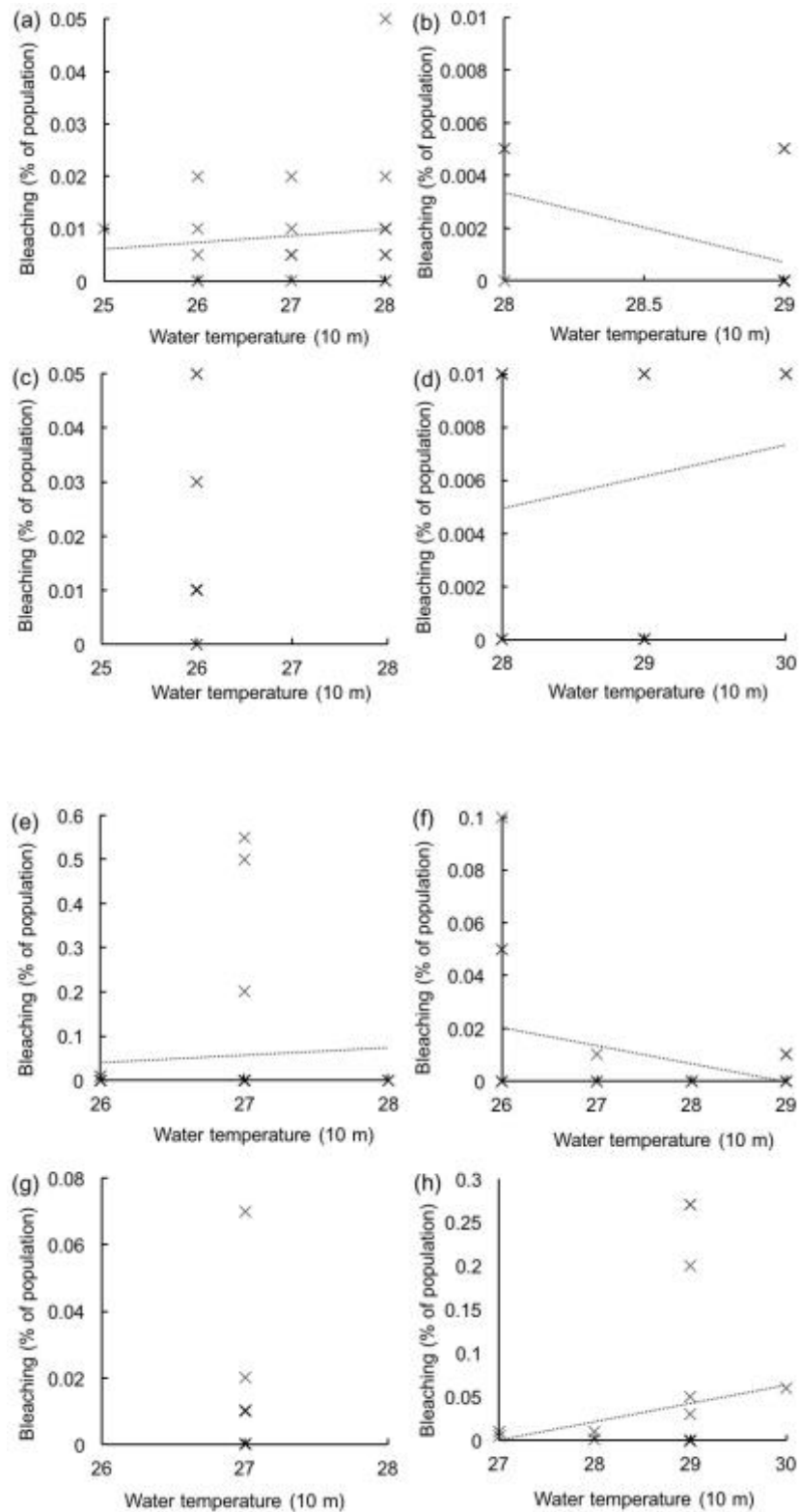


Figure 13: Bleaching (% of population) with sea water temperature at (10 m). (a) spring, 2013 (b) autumn, 2013 (c) spring, 2014 (d) autumn, 2014 (e) spring, 2015 (f) autumn, 2015 (g) spring, 2016 (h) autumn, 2016.

DISCUSSION

The aim of the study was to investigate 1) whether temperature changes would affect the reef abundance of hard coral and sponges; 2) if a coral dominated reef would shift to a sponge dominated reef following a period of thermal stress; 3) and additionally, to explore whether a MPA supports hard corals in withstanding thermal stress.

Viewing the data of water temperatures (10 m) the results show a generally stable level throughout each season, with autumn being statically significantly warmer compared to spring ($P < .001$). Year 2015 showed overall lower temperatures, when compared to the remaining three years. Further did the results show a statistically significant association between the year 2014 and 2015 (colder), and 2013 and 2016 (warmer). Comparing this result with the annual sea surface temperature (SST) publically available on the NOAA website (NOAA, 2017) (Fig. 14), the recordings for spring (March 2014 and 2015) show a corresponding pattern of cooler temperatures in the same area, when compared to the same month of previous years (Fig. 14a-d). This is in stark contrast to the temperature trends anticipated, as this was the year the El Niño was predicted to impact the area. Some studies suggest that event did postpone, and did not occur until approximately one year later (Eakin 2016, Eakin *et al.*, 2016).

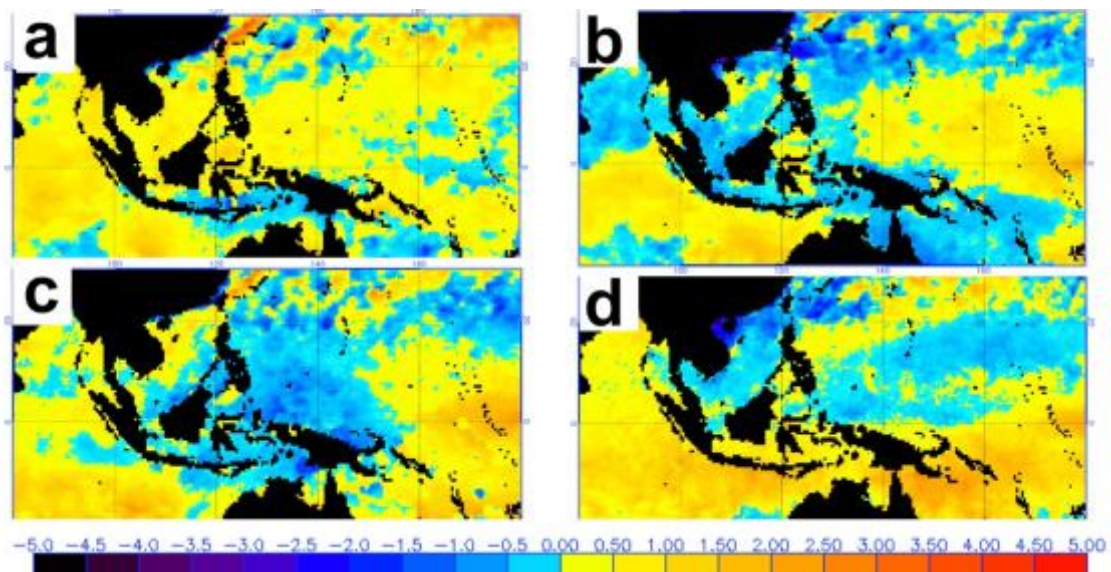


Figure 14: Map of SST of the Coral Triangle (a) March 2013 (b) March 2014 (c) March 2015 (d) March 2016 showing the drop in temperature, starting in 2014, lasting 2015, and returning to warmer temperatures during 2016. Showing the range in temperatures at nearly 2 °C (Source; NOAA, 2017).

Considering both groups of hard corals collectively the overall abundance is at statistically significantly higher level within the MPA, for both spring and autumn, with a more stable level within the MPA than outside. This would suggest that MPAs do provide some resilience for the coral to withstand any pressure that would otherwise affect the abundance via disease or bleaching and eventually die off. Which agrees with previous studies, with the addition of the area being well managed (Selig and Bruno 2010). Further was there no significant difference associated with abundance and season or years. This is surprising as the difference in temperature between the two season was at 1.72 °C. Previous research has suggested that corals become stressed when temperature change more than one degree. As the two groups of hard corals consist of two very different growth traits data were likewise treated separately to assess a possible difference in reaction to the thermal stress amongst the two groups. The *Acropora spp.* are generally considered a fast growing more sensitive coral, with branching *Acropora cervicornis* having been documented to expose a growth rate of ≈ 100 mm/year (Buddemeier and Kinzie III 1976, Fabricius *et al.*, 2004). Species of the genus *Porites spp.* have been measured to exhibit a much slower growth rate of 5-15 mm/year (Maragos 1972, Fabricius *et al.*, 2004). These results show that, for both seasons, the *Acropora spp.* is significantly higher inside the MPA. The non-*Acropora spp.* exhibit same trend, but found in higher abundance when compared to the *Acropora spp.*. Due to the previous mentioned difference in resilience and growth rate, a higher abundance of the non-*Acropora spp.* group would be expected. Interestingly, the non-*Acropora spp.*, generally considered to have a higher level of resilience compared with *Acropora spp.*, show the same overall reaction pattern, both inside and outside the MPA, as the

Acropora spp. (Gilmour *et al.*, 2013, O'Leary *et al.*, 2017). The data were then looked upon in relation to temperature, not considering the change in season – only overall temperature trends over the four-year period. When temperature falls between the years 2013 and 2014 the abundance of the two groups rises, to peak for both groups in the year 2014 at temperature 25.5 and 26 °C. When temperature suddenly rises by almost one degree between 2014 and 2015 the abundance of both groups of hard corals drastically decreases. But as the temperature levels out between 2015 and 2016 the abundance increases within both groups of hard coral. This could suggest, given the hard corals have a reasonable time to adjust, survival is more likely to occur, than when changes in temperature occur rapidly. Therefore, what could be more important to consider, when trying to predict the survival of coral reefs in the future, is the rate at which temperature changes, as an alternative to the solely focus on increase in temperatures.

Viewing the abundance of sponges, it becomes evident that overall 2016 has led to a decline both inside and outside the MPA (Dallison and Ferguson 2017). With an additional decline outside the MPA, for both seasons, over the whole period of the four years monitored. The highest sponge abundance is recorded inside the MPA during the warmer autumn period. This could suggest, in agreement with previous studies, that sponges thrive in warmer temperatures, and that the increase in temperature promotes an increase in growth rate (Koopmans and Wijffels 2008). What remain unidentified, is the decline of sponge abundance of the area, as no known factors are available to make this impact. Further did results show, when viewing the association between water temperature and sponge coverage, the only year with a significant association was the autumn of 2014, which could suggest a reaction of the sponges to the change to colder temperatures.

Comparing the data of abundance of hard coral and sponges, a trend appears which agrees with previous suggestions, a possibility of sponges taking over the reefs when

corals are stressed (Bell *et al.*, 2013, Kelmo *et al.*, 2013). As hard coral declines in abundance, sponges appear to take advantage of the opportunity, and succeed in an expansion of coverage. The results showed a significant association between the coverage of hard corals and sponges during spring of both 2015 and 2016. Perhaps this suggests the competition between the two species during a time of thermal stress. But further does the study show, as the temperatures rose to the average temperature (27.62 °C) during autumn 2015, the coral had the strength to bounce back, and returned to increase in abundance, with the sponges declining.

When viewing the bleaching of the hard corals it is evident that the bleaching increases over the years of observation. During 2013 and 2014 the range of recorded bleaching is steady between 0.00 - 0.05 presence of all transects recorded. During spring 2015 the bleaching increases to 0.06, with following to 0.1 in the autumn. When temperatures continue to reach higher levels during autumn 2016 the level of bleaching rises to highest recorded level, 0.3. This could suggest that the corals are very responsive to change in temperatures, especially when changes are radical. Further to this statement, bleaching seems to be more present during autumn than spring. Autumn is the season with highest levels and variation in temperatures. Last observation that can be taken from the bleaching of the area, is the trend during spring. Overall bleaching during spring ranges between 0.0 and 0.08, whilst autumn ranges 0.0 and 0.3.. Hence, some level of bleaching occurs, regardless temperatures being more stable during spring. This would indicate that there may be other factors present in the environment, to trigger the bleaching. As the surrounding area has a low anthropogenic influence (low population, no industrial presence), it could be suggested that the impact could come from upwelling of deeper waters, or other natural impacts. At present time, an explanation remains unfound.

In relation to the stated resilience present within most genera of sponges, one could argue that sponges present in normally warm water could have a challenge in overcoming sudden changes to a colder environment. Hence, sponges may not always be more adaptable to climate change, as climate change in some areas could produce colder temperatures. Should these views be taken into consideration, *Acropora spp.* seem to be the winner of the race, as they may suffer in abundance in an immediate reaction, but due to their ability to rapidly grow, they do have the potential to recover should conditions stabilise. As *Acropora spp.* show signs of being more sensitive when comparing to non-*Acropora spp.*, *Acropora spp.* but also, show a greater level of returning after a period where both categories have been seen to decline. This is in great relation with previous studies, that show *Acropora spp.* are, in general, a faster growing hard coral when compared to many of the species present within the Non- *Acropora spp.* group. Hence more adaptable to changes.

The results show, during periods of hard coral decline, sponges do increase in abundance. But further show that the corals are able regain coverage, once conditions have regained favourable conditions. Hence, it cannot solely be concluded that future reefs will experience a shift from the traditional coral reef composition to a sponge reef as temperatures increase. Other factors seem to be present, and affect the outcome following warmer temperatures, perhaps a healthy coral reef will have more resources to survive and withstand a pressure in competition with, for example, sponges. It would therefore be interesting to conduct further investigation of the area, possible an analysis of the composition of zooxanthellae clades present in both the sponges and corals of the reef, and likewise from other, maybe less healthy, coral reefs. Will zooxanthellae composition be different when comparing to times before the stress, and colonies of other sites. Previous studies have shown that not only several types of zooxanthellae exist, but also these varied clades provide alternate services for the host (Bellantuono *et al.*, 2011). Clade A being the zooxanthellae that show benefits when rapid growth is

needed, whereas at the other end of the scale clade D will make its host more heat tolerant (Stat *et al.*, 2013, Yuyama and Higuchi 2014). Some clades will provide the corals with heat resistance (Berkelmans and van Oppen 2006) and some with an enhanced ability of growth – could this possibly be what the coral benefit from? Shifting the clades fitting to the conditions, and not taking in zooxanthellae for a stable long-term solution (Kinzie *et al.*, 2001)? Perhaps in an alarmed situation the coral will bleach, and of such an extreme level that this is visual. Whereas when conditions have returned to the preferred temperature the corals may take up beneficial fast growing zooxanthellae over time? Some studies show an adaptation amongst the corals to withstand the increased temperature (Guest *et al.*, 2012, Ainsworth *et al.*, 2016). These are some views that genetics possibly would be able to cast light upon, and perhaps provide answers to, in relations to both corals and sponges organisms (Guzman and Conaco 2016a, 2016b). Extending from the importance of relationships amongst species, another factor is becoming more and more evident to pose almost an equally great importance; the microbial holobiont of hard coral colonies (Krediet *et al.*, 2013). This area would be another aspect to pursue further research on, but one which will present a great challenge as many factors will affect the holobiont and therefore the outcome of a survey. A fully *ex situ* modelled approach may not provide a valid result, as there may even be factors present, that we are not yet aware of.

Another approach, of more chemical character, new research show that the mucus produced by the coral for protection, by excreting dissolved organic matter (DOM), could potentially assist, or create, a sponge loop (Rix *et al.*, 2016). And so, as the corals experience an increase in light, radiation and heat, the production of mucus increase to protect colonies from harm. Hereby, unknowingly, providing extra nutrients to the water column and accidentally provide nutrients for the sponge to utilise. This could be a potential explanation to the association between the hard coral and sponge abundance during stressful times, and would explain the relation found of the reef of Napantao.

Listing these statements show how many factors influences the composition of the reefs, and without completely knowing all, and how they individually affect other relationships it will be hard to predict the future composition of the reef following the impact of climate change. In this study errors are present within the pragmatic section of not having obtained nutrients within the water column, before, during and after a thermal stress period. This could be an addition to the data collection that would possibly provide further answers to decline in sponge abundance as corals recover. This would also be possible to investigate further by measurements of pH and radiation, at the sites. It would also be interesting to have a permanent temperature device at the sites, as, of now, the temperatures are only measured whilst the surveys are conducted. Hence, the data of temperatures are more an estimate of the overall thermal pattern, than the actual state. Currents, upwelling, storms, amongst some natural phenomenon, can affect the overall thermal trends, and hereby affect the abundance of the species and bleaching pattern of the hard corals, in ways that for now remain undetected. Further errors of this study, most data proved to be non-normally distributed, which lead to decisions of either choosing a less strong non-parametric statistical analysis, or run parametric test when a risk of a Type I error to follow.

The conclusion of this study must be; from the data collected over a four-year period, some evidence suggests that temperatures will affect the abundance of species, but the coral reefs can be more resilient and recover over a period, in an otherwise healthy environment. The study also showed that sponges may be able to take advantage of a thermal unstable pattern, and a hereby stressed coral reef. But sponges can meet resistance in the competition. The results further showed the positive importance of a well managed MPAs, and its aid in the survival of our magnificent coral reefs.

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