

# **The potential of MPAs to ameliorate coral health in one of the world's diving hotspots: Koh Tao, Thailand**

Margaux Hein, School of Marine Tropical Ecology, James Cook University, 4814 QLD -Word count 6392-

## **1. Abstract**

While coral reefs are increasingly threatened worldwide, they are also increasingly used for recreational activities. Understanding the links between human activities and coral health, as well as evaluating the efficiency of marine protected areas (MPAs) as a management regime to prevent further deterioration, is critically important, particularly given the environmental and socio-economic significance of coral reefs. The aim of this study was to quantify indicators of coral health at sites inside and outside a newly rezoned MPA framework established in July 2012 in the dive-tourism hotspot of Koh Tao, Thailand. I found that the health and diversity of coral communities were not yet responding to the protected status conferred by newly zoned MPAs, but instead reflected past history of recreational use around the island. Sites characterized as past high-use sites had lower hard coral cover, lower prevalence of the branching family Acroporidae but higher prevalence of the more weedy Agariciidae, and higher prevalence of both disease and other indicators of compromised health. Sites within the newly established MPA framework are currently subjected to higher levels of environmental and anthropogenic pressures, particularly higher levels of sedimentation, and associated with this, higher prevalence of algal overgrowth, scars from *Drupella* snails, and breakage. Given the greater prevalence of these factors within protected sites, the capacity of the MPA framework to effectively prevent further deterioration of Koh Tao's reefs is unclear. Nevertheless, my study constitutes a strong baseline for future long-term evaluations of the potential of MPAs to maintain coral health and diversity on highly threatened reefs.

## 2. Introduction

The accelerating pace of coral loss worldwide (e.g. Bruno and Selig 2007; De'ath et al. 2012) and the need to understand the underlying causes of the declines are now widely recognized. Most agree that there is no single cause but rather multiple factors acting alone or in synergy, such as rising SSTs, ocean acidification, and nutrient run-off (e.g. Jackson et al. 2001, Harvell 2007, Hoegh-Guldberg et al. 2007, Burke et al. 2012). The impact of coral diseases on reef community structure and associated ecosystem services is a recent area of rising concern. First described in the Caribbean in the 1970's (reviewed in Sutherland et al. 2004), coral diseases have now been shown to affect reefs worldwide and are a growing threat to Indo-Pacific coral reefs (Harvell et al. 2007, Willis et al. 2004, Myers and Raymundo 2009, Weil et al. 2012). However, research on coral diseases is still in its infancy and further studies on the causes of spreading disease outbreaks worldwide have been consistently called for to address this significant knowledge gap (e.g. Richardson 1998, Harvell et al. 1999, 2007, Roder et al. 2013).

Links between a range of environmental factors and increasing disease prevalence suggest a role for changing climate in the rise of coral diseases. In particular, recent studies suggest a link with global warming and rising SSTs (Harvell et al. 2002, Bruno et al. 2007, Sato et al. 2009, Maynard et al. 2011, Ruiz-Moreno et al. 2012). Evidence of rising disease prevalence following bleaching events (e.g. Bruno et al. 2007, Maynard et al. 2011) and seasonal patterns of disease outbreaks (e.g. Sato et al. 2009, Haapkyla et al. 2011) further support this idea. Climate warming could have the dual effect of increasing pathogen virulence while decreasing coral host resilience (e.g. Harvell et al. 2002, 2007, Bruno et al. 2007, Maynard et al. 2001, Heron et al. 2010, Ruiz-Moreno et al. 2012). Other abiotic factors associated with climate change and escalating human population sizes, such as ocean acidification and decreased water quality, could further impede corals' resistance to infectious diseases (e.g. Harvell 2007, Hoegh-Guldberg et al. 2007, Danovaro et al. 2008, Van der Meij et al. 2010).

The role of human-related activities in coral disease dynamics is poorly understood but likely to be highly complex (Harvell 2007, 2009). Loss of ecosystem complexity through overfishing, increased nutrient-run off associated with coastal development, and activities like dredging that increase sedimentation have all been proposed as potential causes of disease outbreaks (Kaczmarzsky 2006, De'ath and Fabricius 2010, van der Mey et al. 2010, Aeby et al. 2011, Haapkyla et al. 2011, Lamb and Willis 2011, Ruiz-Moreno et al. 2012, Pollock et al. (in review)). Recently, links between coral disease and human activities in areas of high tourist visitation (Danovaro et al. 2008, Lamb and Willis 2011, Onton et al. 2011) highlight an additional cause for concern. Recreational activities on coral reefs may introduce new pathogens or further increase pollution and nitrification (e.g. sunscreen, fish feeding) (e.g. Danovaro et al. 2008, Lamb and Willis 2011). Snorkelling and diving activities may also cause breakage of coral colonies, further reducing corals' resistance to infections (e.g. Page et al. 2009, Lamb and Willis 2011, Onton et al. 2011). There is thus a crucial need to understand the links between human activities and coral health to better manage coral reefs worldwide (Lamb and Willis 2011, Onton et al. 2011).

In Thailand, 70% of annual tourism income is related to coastal marine activities (Sethapun 2000). The island of Koh Tao alone receives more than 300,000 visitors a year, of which at least 60% participate in scuba diving activities (Nichols 2013). Located in the western Gulf, this 21km<sup>2</sup> island is Thailand's most popular dive destination, and probably hosts the most number of beginner divers worldwide (Scott 2012). The island is home to more than 40 dive schools that generate nearly half of the world's PADI certifications every year (Nichols 2013). Even if some sites are visited by only half of these divers, diving pressure is still 15 times greater than the estimated carrying capacity of a coral reef for scuba diving (5000 to 6000 divers/year; Hawkins and Roberts 1997). Moreover, supporting such a flourishing scuba diving industry has required rapid coastal development at the expense of the preservation of the Island's natural resources. Related human impacts caused by deforestation for tourist accommodation, overexploitation of fishing resources and poor sewage treatment are now

putting further pressure on the Island's coral reefs (Wilkinson and Brodie 2011, Weterings 2011, Scott 2012). Koh Tao's reefs are thus highly threatened by its growing tourism industry.

To address threats related to tourism expansion on Koh Tao, a group called "Save Koh Tao" (SKT) in conjunction with the Thai Department of Marine and Coastal Resources (DMCR) defined and zoned Marine Protected Areas (MPAs) around the island in July 2012 (Marine Conservation Koh Tao 2012). The effectiveness of MPAs, however, is a much-debated topic (e.g. Jameson et al. 2002, Halpern 2003, Degnbol et al. 2006, Graham et al. 2011), and there is an urgent need to evaluate the effectiveness of this management strategy. 28% of the world's coral reefs are within protected areas but only 0.1 to 6% are thought to be effectively managed (Mora and Sale 2006, Burke et al. 2012). Many argue for caution in the use of MPAs, arguing that they are not a "magical fix" and their hypothetical benefits are often impeded by problems of compliance and governance (e.g. Jameson et al. 2002, Mora and Sale 2011, Burke et al. 2012). Although MPAs have been shown to increase the abundance, size, and diversity of exploited fish species (e.g. Lubchenko et al. 2003, Russ and Alcala 2004, Almany et al. 2013), their capacity to benefit coral communities has been questioned (e.g. Page et al. 2009, Graham et al. 2011). Some argue that reduced fishing pressure will restore complex trophic interactions needed to maintain the resilience of coral reef communities, with cascading effects down to coral populations (e.g. Mumby 2006, 2009, Raymundo et al. 2009, Selig and Bruno 2010). For example, increased herbivory within MPAs is expected to increase coral recruitment, growth and survival, and prevent shifts to algae-dominated ecosystems (e.g. Hughes and Tanner 2000, McClanahan 2008, Mumby 2009). Decreased fishing pressure may also decrease physical damage to corals by reducing the use of destructive fishing techniques (Baird et al. 2005, Graham et al. 2011). Unfortunately, very few studies have actually found an increase in coral cover within MPAs (McClanahan et al. 2007, Selig and Bruno 2010), with most studies indicating that MPAs do not alleviate declines in coral cover (McClanahan et al. 2001, Jones et al. 2004, Coelho and Manfrino 2007, Page et al. 2009, Hsieh et al. 2011).

The capacity of MPAs to mitigate coral disease is even more questionable (e.g. Coelho and Manfrino 2007, Page et al. 2009). Apart from Raymundo et al. (2009), who found decreased coral disease inside reserves, most studies indicate a general failure of MPAs to address the rising incidence of coral disease (Coelho and Manfrino 2007, McClanahan et al. 2008, Page et al. 2009). They argue that coral diseases are associated with threats, like rising ocean temperatures, which are beyond the scope of protection provided by Marine Protected Areas. Moreover, some fear that MPAs could actually promote the spread of coral diseases if they successfully increase coral host density without controlling the spread of pathogens through pollution, run-off and increased tourist visitation in a warming ocean (Bruno et al. 2007, Page et al. 2009, Lamb and Willis 2011). MPAs alone may thus not be able to mitigate disease outbreaks and coral loss worldwide. Improving our understanding of how MPAs affect the complex array of biotic and abiotic factors influencing disease prevalence in coral populations is needed to provide local managers with the information necessary to optimize the management of coral reefs.

The aim of my study is to quantify indicators of coral health at sites inside and outside the current Marine Protected Areas surrounding the Island of Koh Tao, Thailand. More specifically, I will compare coral cover and coral disease prevalence at MPA and non-MPA sites to evaluate if the coral community is responding to the new protection status or if potential differences reflect past high versus low use history. Looking at changes in hard coral cover, taxonomic diversity, and disease prevalence among protected and unprotected sites will enable me to evaluate the effectiveness of the current management scheme and identify links with specific local threats.

### 3. Methods

#### 3.1 Study sites and field surveys

Coral health surveys were conducted on the reefs of Koh Tao during early August to late September 2013. A total of 6 sites, located around the island, were surveyed. Three sites were located within the new MPA framework: Shark Island, Sairee, and Twins; and three sites were located outside the MPA framework: Ao Leuk, Hin Wong, and Sai Nuan (Figure 1). Sites were also graded and categorised as past “high use” or “low use” sites, using records of site-specific disturbances in the six-year period between 2006 and 2012 (Table 1; Figure 1). Grades were determined by the New Heaven Reef Conservation Program Director (C. Scott), who has been surveying these reefs since 2006. Sites with a “use grade”  $\geq 10$  were classified as past “high use” sites; sites with a “use grade”  $< 10$  were classified as past “low use” sites. Within the MPA framework, 2 out of 3 sites (Sairee and Twins) are past high use sites, while 2 out of 3 sites outside the MPA framework (Hin Wong and Sai Nuan) are past low use sites (Table 1; Figure 1).

Table 1. Summary of site-specific disturbances from 2006 to 2012, with corresponding “use grade” based on recreational and other human-related activities, as assessed by the New Heaven Reef Conservation Program Director (Low=1, Medium=2, High=3).

	<b>Twins</b>	<b>Sairee</b>	<b>Shark Island</b>	<b>Sai Nuan</b>	<b>Ao Leuk</b>	<b>Hin Wong</b>
<b>Sedimentation</b>	low	Medium	Low	Medium	Medium	low
<b>Diving pressure</b>	High	Medium	High	Low	High	low
<b>Snorkelling pressure</b>	Medium	Low	Low	Medium	High	High
<b>Waste water run-off</b>	Low	High	Low	High	Medium	low
<b>Boat traffic</b>	High	High	Medium	Low	High	High
<b>Overall Grade (past use)</b>	<b>10</b>	<b>11</b>	<b>8</b>	<b>9</b>	<b>13</b>	<b>9</b>
<b>MPA status (July 2012)</b>	<b>MPA</b>	<b>MPA</b>	<b>MPA</b>	<b>Non-MPA</b>	<b>Non-MPA</b>	<b>Non-MPA</b>

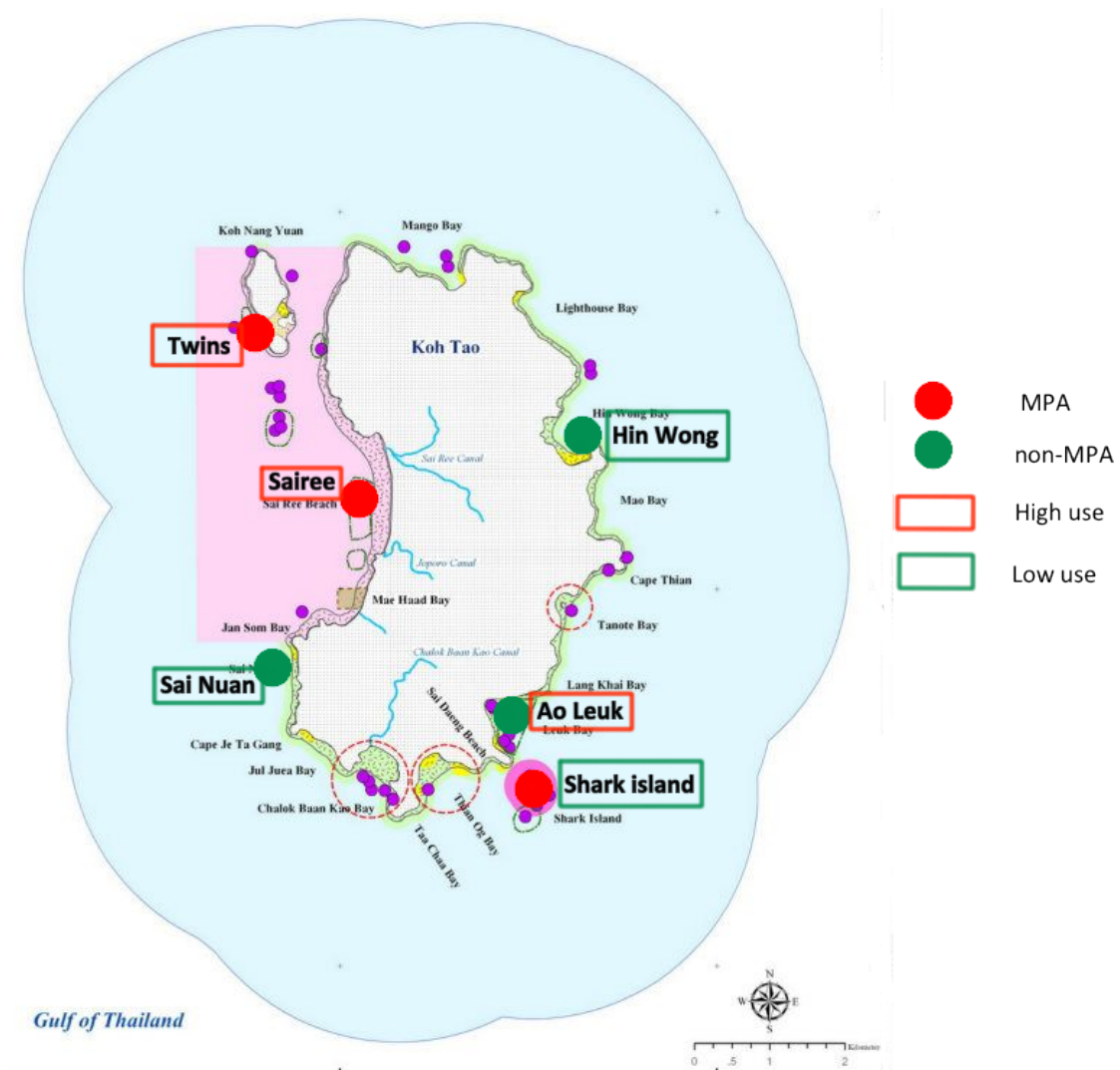


Figure 1. Map showing locations of the 6 study sites around the Island of Koh Tao, Thailand. Red dots represent sites within the MPA framework; Green dots represent sites outside the MPA framework. Red squares denote past High use sites, and Green squares denote past Low use sites.

Each of the six sites was surveyed on SCUBA along 3 randomly placed 15m x 2m belt transects. Transects were placed haphazardly at depths from 2.5 to 6m following reef contours, with a minimum distance of 5m between transects. All corals within each of the 30m<sup>2</sup> transects were recorded to the genus level and assigned to one of the following categories: 1) “healthy”, 2) diseased (black band disease, white syndromes, brown band disease, skeletal eroding band, growth anomalies, or unusual (non-focal) bleaching patterns), or 3) showing other signs of compromised health (overgrown by algae and/or sponges; predation scars from fish, *Drupella* snails and/or crown-of-thorns starfish; or broken (tips or whole fragments)). A roving diver survey was also performed at

each site, in which all colonies showing signs of disease and other indicators of compromised health were recorded. The survey area was then estimated for each site on Google Earth ® (V 7.1.2.2041).

Coral cover was estimated using the line-intercept method, by recording every coral under each transect line to the genus level, and measuring the distance it covered along the line to the nearest centimetre.

Sediment accumulation was measured at study sites by setting up 3 sediment traps at each site at a depth of 4 to 6m. Sediment traps were 10cm long PVC tubes, which were positioned approximately 5cm above substrata, and left on site for 2 to 6 weeks. Once sediment traps were recovered, sediments were filtered using Whatman® glass microfiber filters (GF/C grade, 25mm, 1.2µm), and dried. The contents of traps were then transported to Mahidol University in Bangkok and weighed to the 0.1mg.

Boat surveys were conducted at each of the 6 sites to estimate boat traffic, as well as diving and snorkelling pressure. Each site was visited twice during peak hours (10am to 2pm), giving a total of 8 hours of surveys per site. Three types of boats were differentiated: boats that passed through the sampling area but did not stop, diving boats that stopped at the sites and dropped divers, and snorkelling boats that stopped at the sites and dropped snorkelers. Snorkelling boats were further differentiated between the traditional long-tail boats carrying a maximum of 5 snorkelers and “large boats” that could carry up to more than 50 passengers.

### 3.2. Data analysis

#### 3.2.1. Coral community structure

Hard coral cover: The percent cover of hard corals was calculated for each 15m transect line by dividing the sum of all hard coral distances measured by 1500cm. I compared differences in hard coral cover between MPA and non-MPA sites, between past high-use and past low-use sites, and among sites using 1-way

analyses of variance. Data were square-root transformed to meet assumptions of normality (Shapiro-Wilks test). Tukey's (HSD) post-hoc tests were also run to identify the significance of dissimilarities among sites.

Taxonomic composition of coral assemblages: To compare trends in taxonomic composition among sites, corals were grouped into the 3 most abundant families, the Acroporidae, Agariciidae, and Poritidae, and all other families were grouped into the category Other. As for comparisons of hard coral cover, 2-way analysis of variance with fixed factors were used to compare percent cover of each coral family among sites when categorised by MPA status (Fixed factors: MPA status and coral family), and past level of recreational use (Fixed factors: use level and coral family). A one-way analysis of variance was also performed on the SKT baseline data to compare the mean percent cover of foliose and branching growth forms between past high use and low use sites.

Data collected during the study period were also compared to a six-year database collected by the Save Koh Tao group (SKT). The database contained survey data collected from 2006 to 2012, prior to the establishment of MPAs, using 100m long fixed transect lines and the "point-intercept survey" method to record coral cover and growth form diversity (% cover of branching, massive, submassive, foliose, encrusting colonies) at each site.

### 3.2.2. Coral health

Disease prevalence: Disease prevalence was calculated for each 30m<sup>2</sup> belt transect by dividing the number of diseased colonies by the total number of colonies. I then compared disease prevalence between sites differing in MPA status, past recreational use, and among sites using the non-parametric Kruskal-Wallis test because the data did not meet assumptions of normality, even after transformation. When non-parametric tests were used, multiple pairwise comparisons with Dunn's procedure and Bonferroni corrections were performed as post-hoc tests to determine the significance of dissimilarities.

Other indicators of compromised health: The density of colonies with other indicators of compromised health was estimated by dividing the total number of affected colonies found during the roving diver surveys by the Google earth® estimate of the survey area. Procedures described above for disease prevalence were repeated for the statistical analysis of other compromised health categories.

### 3.2.3. Environmental characteristics

Diving pressure: Diving pressure was calculated as the number of dive boats/ hour. Three 1-hour periods out of the 8 hours of surveys were randomly selected for each site to ensure independence of the samples and equalize sample size among all environmental pressures. Diving pressure was then compared between sites grouped according to MPA status and past use, and among sites in general using the non-parametric Kruskal-Wallis test, because data did not meet assumptions of normality, even after transformation.

Snorkelling pressure: Snorkelling pressure was calculated as the number of snorkelling boats/ hour. Big snorkelling boats were counted as equivalent to 10 small snorkelling boats. Analyses described above for diving pressure were repeated for snorkelling pressure.

Boat traffic: Boat traffic was calculated as the number of passing boats/hour, i.e. snorkelling and diving boats were not included. Data were log-transformed to meet the assumptions of normality and one-way analyses of variances were performed to compare differences in boat traffic between sites differing in MPA status and past use, and among sites.

Sedimentation: The weight of sediment in each trap was divided by the number of days the traps had been underwater to calculate sedimentation/day. Sedimentation was then compared between sites differing in MPA status and past use, and among sites using the non-parametric Kruskal-Wallis test because the data did not meet assumptions of normality, even after transformation.

#### 3.2.4. Relationship between past use intensity and coral community structure/health

A correlation matrix using Pearson's coefficient (5% significance level) was computed to identify trends in relationships among past use and coral community structure and health. Univariate data on disease prevalence, other indicators of compromised health, and coral cover were analysed using Microsoft Excel (V14.3.8) and XLSTAT (203.5.03).

Multivariate analysis: Resemblance matrices were created for both coral health variables (total prevalence of disease, algae overgrowth, *Drupella* scars, coral breakage) and environmental data (variables: sedimentation, boat traffic, diving pressure, and snorkelling pressure) based on the Bray-Curtis index of similarity. The Bray-Curtis index has been described as the most appropriate index for the description of benthic assemblages (Anderson 2001). The biological data were square-root transformed due to strong linear pairs of variables (Anderson et al. 2008; Gorley and Clarke 2008) and environmental data were normalised (z-transformed, where average = 0 and standard deviation = 1).

Similarities between coral health assemblages were illustrated using a non-metric multidimensional scaling plot (MDS). A 2-way crossed analysis of similarity (ANOSIM) was used to test differences in coral health assemblages between MPA status and past recreational use (Gorley and Clarke 2008).

In order to assess the environmental drivers of coral health, a DistLM (distance-based linear model) was performed using resemblance matrices of environmental pressures (sedimentation, boat traffic, diving pressure, and snorkelling pressure) with corresponding coral health data (variables: disease prevalence, algal overgrowth, *Drupella* scar prevalence, and coral breakage). The DistLM was performed using a stepwise selection procedure with AiCc as the selection criterion (due to low number of samples (Motulski and Christopoulos 2004)) and 9999 permutations. The model is visualised using a distance-based redundancy analysis (dbRDA) plot vectors from both the input (coral health) and

predictive (boat traffic and sedimentation) variables are overlaid to show strength and direction of Pearson correlations.

## 4. Results

### 4.1 Hard coral cover

Mean ( $\pm$ SE) hard coral cover was high ( $62 \pm 8.3\%$ ) across all sites, and did not differ significantly among sites differing in either MPA status or intensity of past recreational use (ANOVA:  $F=1.376$ ,  $df=1$ ,  $p=0.258$  and  $F=2.464$ ,  $df=1$ ,  $p=0.136$ , respectively; Figure 2). However, there was a trend towards higher hard coral cover in non-MPA sites and past low-use sites (Figure 2).

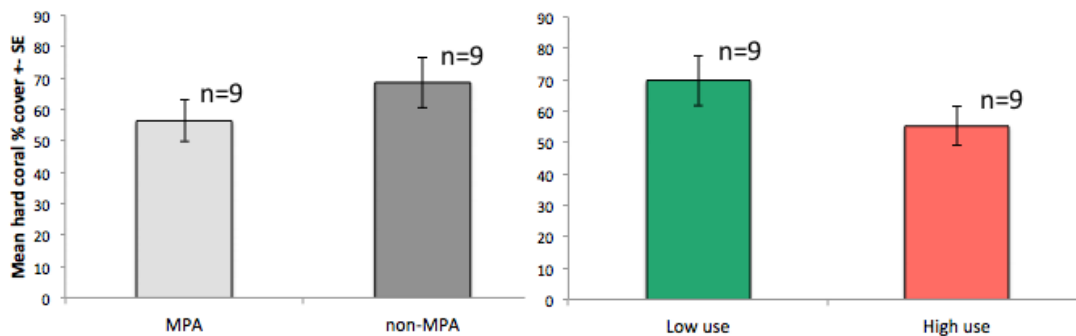


Figure 2. Comparisons of mean percent cover ( $\pm$ SE) between sites differing in MPA status (MPA vs non-MPA) and past recreational use (high use vs low use). MPA sites comprised 2 past high-use sites and 1 past low-use site; non-MPA sites comprised 2 past low-use sites and 1 past high-use site.  $n=9$  transects per MPA or use category.

Overall, mean percent cover of hard corals differed significantly among sites (ANOVA:  $F=4.416$ ,  $df=5$ ,  $p=0.016$ ; Figure 3), with mean hard coral cover twice as high at Hin Wong compared to Ao leuk (Figure 3). Both are non-MPA sites, but Hin Wong is a past low-use site while Ao leuk is a past high-use site.

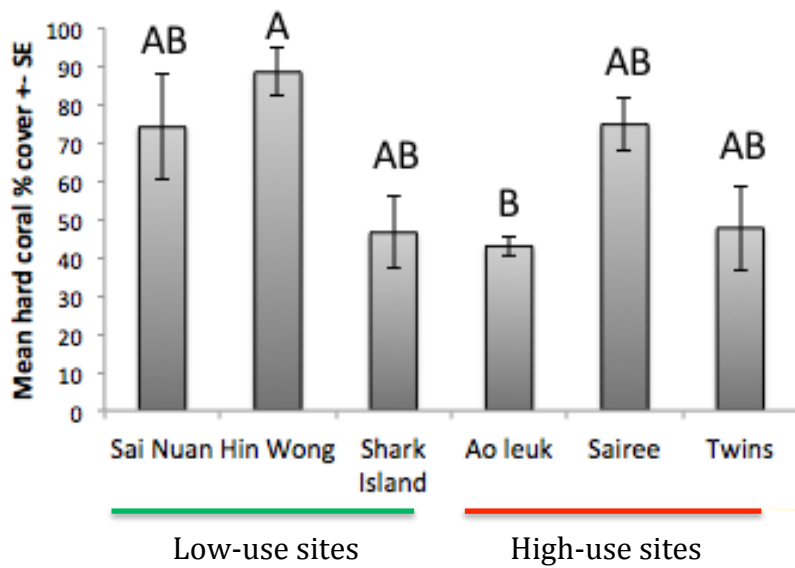


Figure 3. Comparison of mean ( $\pm$ SE) hard coral cover among the 6 study sites. Letters above histograms represent the output from a Tukey's post-hoc test. Sites with different letters have significantly different coral cover.

#### 4.2. Coral family assemblages

There was no significant difference in the percent cover of individual coral families at sites differing in either MPA status or past recreational use (Status ANOVA:  $F=0.453$ ,  $df=1$ ,  $p=0.503$ ; Past recreational use ANOVA:  $F=0.168$ ,  $df=1$ ,  $p=0.684$ ; Figure 4). Nevertheless, mean percent cover of the Acroporidae was almost two-fold greater in non-MPA compared to MPA sites, and also two-fold greater in past low use sites compared to past high-use sites (Figure 4). Conversely, percent cover of the Agariciidae was two-fold greater in past high-use sites compared to past low-use sites (Figure 4), suggesting a possible inverse relationship between the prevalence of Acroporidae and Agariciidae.

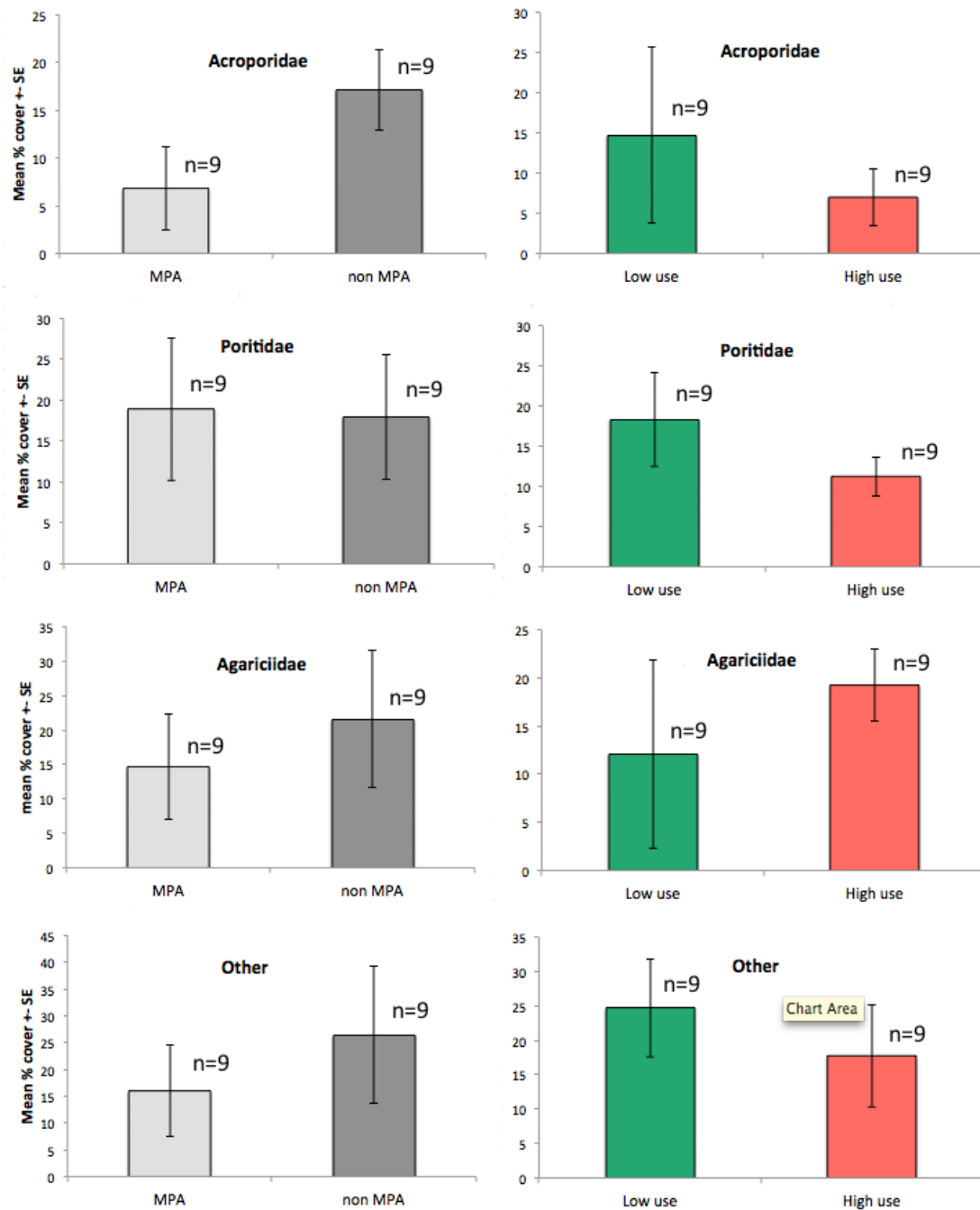


Figure 4. Comparison of mean ( $\pm$ SE) percent cover for the 4 categories of coral families between sites differing in MPA status (MPA vs non-MPA) and past recreational use (high use vs low use). MPA sites comprise 2 past high-use sites and 1 past low-use site; non-MPA sites comprise 2 past low-use sites and 1 past high-use site.  $n = 9$  transects per MPA or use category.

Furthermore, analysis of the 6-year baseline data from SKT revealed a significant inverse relationship in the abundance of corals with a foliose growth form (mostly Agariciidae) compared to those with a branching growth form (mostly

Acroporidae). Percent cover of foliose corals was more than two-fold greater than cover of branching corals at high-use sites, whereas the inverse was true at low-use sites (T-test:  $p < 0.02$  and  $p < 0.001$ , respectively; Figure 5).

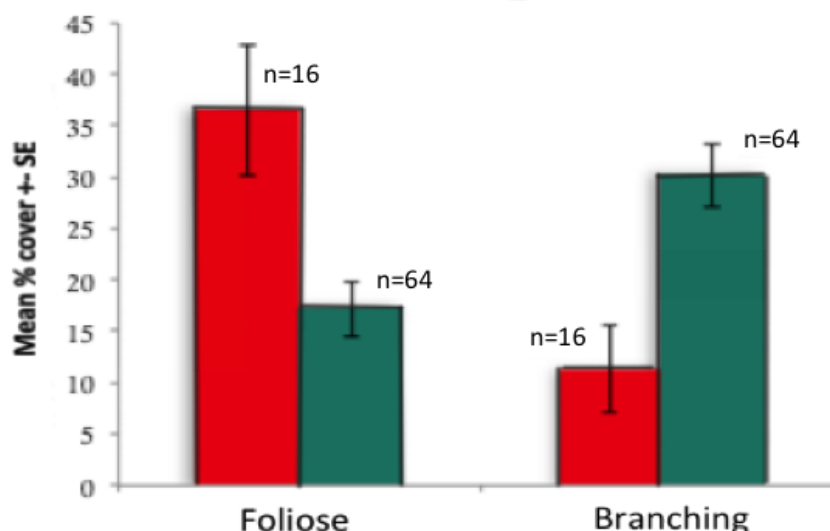


Figure 5. Comparisons of mean ( $\pm$ SE) percent cover of foliose versus branching growth forms between high use (red histograms) and low use (aqua histograms) sites, from 2006 to 2012 (data from the SKT database). n = number of transects per category

### 4.3 Coral Health

#### 4.3.1. Disease prevalence

Overall, the health status of 6373 coral colonies was recorded across all six sites. Only three types of disease were recorded in all sites combined: white syndromes, growth anomalies and skeletal eroding band disease. Overall, mean disease prevalence was low ( $0.72\% \pm 0.3\%$ ). No significant difference in disease prevalence was found between sites differing in either MPA status (Kruskal-Wallis:  $df=1$ ,  $p=0.604$ ; Figure 6) or past use (Kruskal-Wallis:  $df=1$ ,  $P=0.671$ ; Figure 6). There was a trend towards higher (by 1.7-fold) disease prevalence inside MPAs compared to outside MPAs. There was also a very weak trend towards higher (by 1.2-fold) disease prevalence at past high use sites.

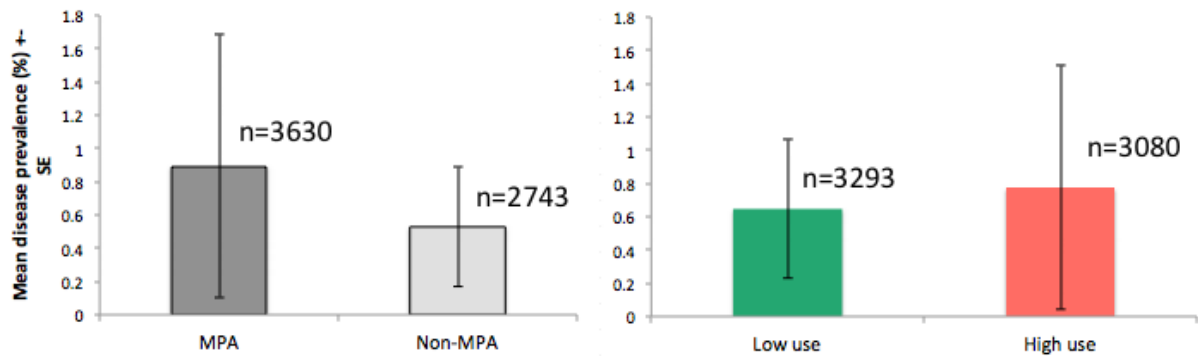


Figure 6. Comparison of mean disease prevalence ( $\pm$ SE) between sites differing in MPA status and past recreational use. MPA sites comprise 2 past high-use sites and 1 past low-use site; non-MPA sites comprise 2 past low-use sites and 1 past high-use site. Means are based on 9 transects per MPA or use category; Above each histogram, n= number of colonies surveyed per MPA or use category.

Growth anomalies were 20 times more prevalent at sites inside MPAs than outside MPAs and similarly more prevalent at past high-use sites compared to past low-use sites (Figure 7). The prevalence of white syndromes, however, was more than two-fold greater in non-MPA sites, and past low-use sites (Figure 7). Prevalence of skeletal eroding band disease was very low and only present inside the MPA framework, at a past low-use site (Shark island) (Figure 7).

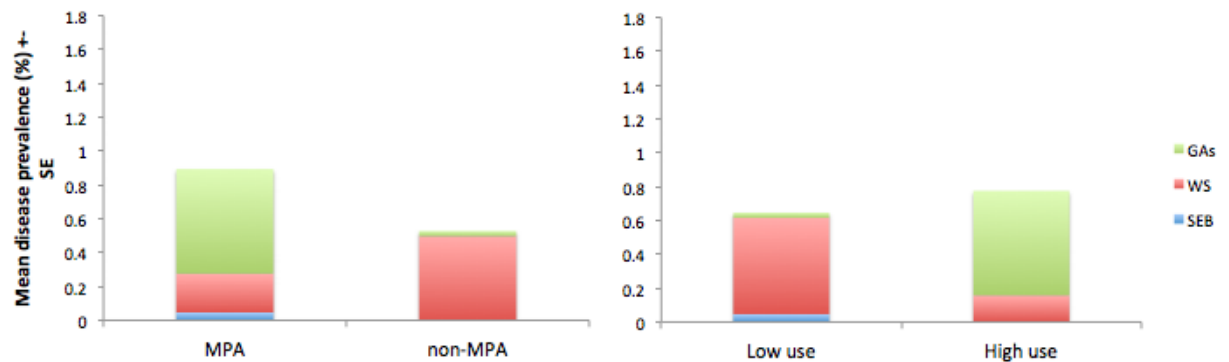


Figure 7. Comparison of mean disease prevalence for each of 3 coral diseases recorded (GAs= growth anomalies, WS= white syndromes, SEB= skeletal eroding band) between sites differing in MPA status and past recreational use.

#### 4.3.2. Other compromised health indicators

Six other indicators of compromised health were recorded: crown-of-thorns starfish predation, *Drupella* snail predation, environmentally induced bleaching (nf bleaching), algal overgrowth, sponge overgrowth, and pigmentation

response. Again, no significant difference was found between sites differing in either MPA status (Kruskal-Wallis,  $df=1$ ,  $p=0.660$ ; Figure 8) or past recreational use (Kruskal-Wallis,  $df=1$ ,  $p=0.379$ ; Figure 8). There was a trend towards higher numbers of colonies with other indicators of compromised health inside MPAs compared to non-MPAs, and at past high-use sites compared to past low-use sites.

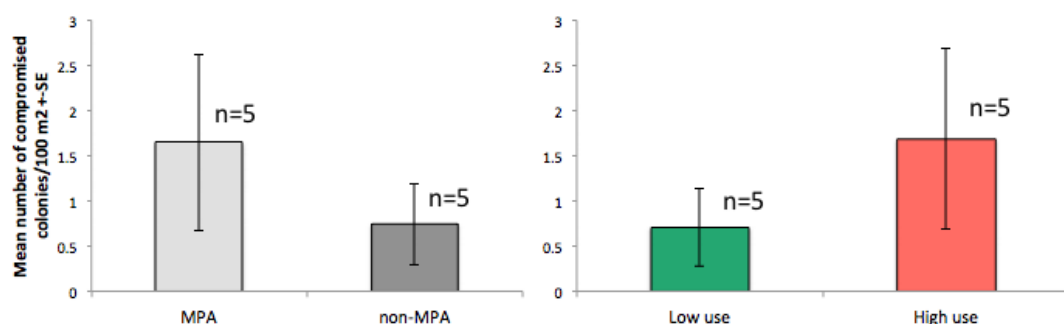


Figure 8. Comparison of the mean number of colonies with other signs of compromised health/ $m^2$  ( $\pm SE$ ) between sites differing in MPA status and past recreational use. MPA sites comprise 2 past high-use sites and 1 past low-use site; non-MPA sites comprise 2 past low-use sites and 1 past high-use site. Means are based on 2 random surveys per MPA or use category. Above each histogram,  $n$  = number of other compromised health categories recorded.

Looking more in depth at other indicators of compromised health, predation by *Drupella* snails and algae overgrowth affected 10 times more colonies than other categories, and were most prevalent in MPA and past high-use sites (Figure 9).

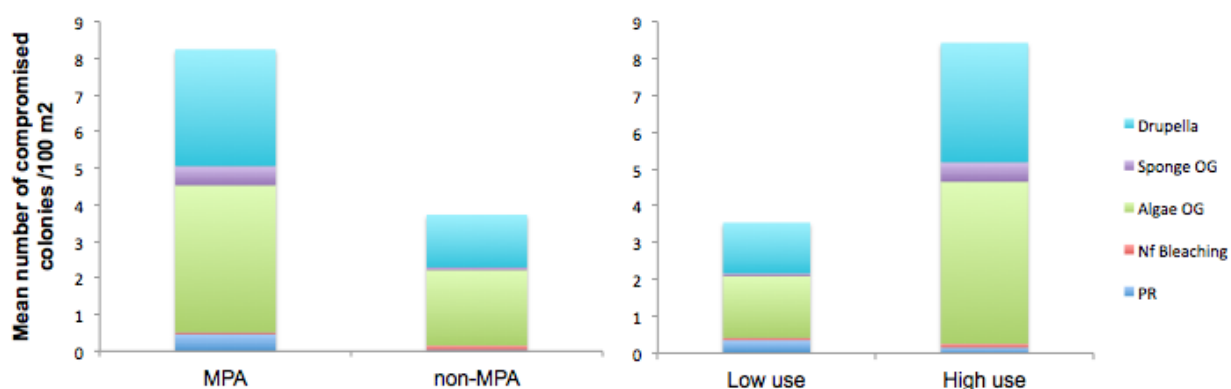


Figure 9. Comparison of the mean number of colonies/ $m^2$  in each of 5 categories of compromised health between sites differing in MPA status and past recreational use.

The non-metric multidimensional scaling plot representing coral health assemblages showed slight clustering among sites by status, with MPA sites (red and light blue symbols) tending to be distributed lower on the plot than non-MPA sites (green and dark blue symbols) (Figure 10). However, statistically, distributions of coral health assemblages did not differ significantly between sites differing in MPA status, although the 0.05 p value indicates that any similarities were marginal (ANOSIM: Global R=0.481,  $p=0.05$ ; Figure 10).

While no clustering of sites was apparent on the MDS plot in term of past recreational use, statistically, the distributions of coral health assemblages differed significantly between use levels (ANOSIM: Global R=0.556,  $p=0.001$ ; Figure 10). Overall, it appears that dissimilarities between sites differing in status and past recreational use are driven by the one non-MPA high-use site (Ao Leuk- dark blue triangles) and the one MPA low-use site (Shark Island- red diamonds), respectively (Figure 10). Algae overgrowth, *Drupella* predation, and breakage appeared to strongly affect both non-MPA, low-use sites (green triangles) and MPA high-use sites (light blue squares) (Figure 10). Disease prevalence seemed to be the driver of MPA low-use sites (red diamonds) (Figure 10).

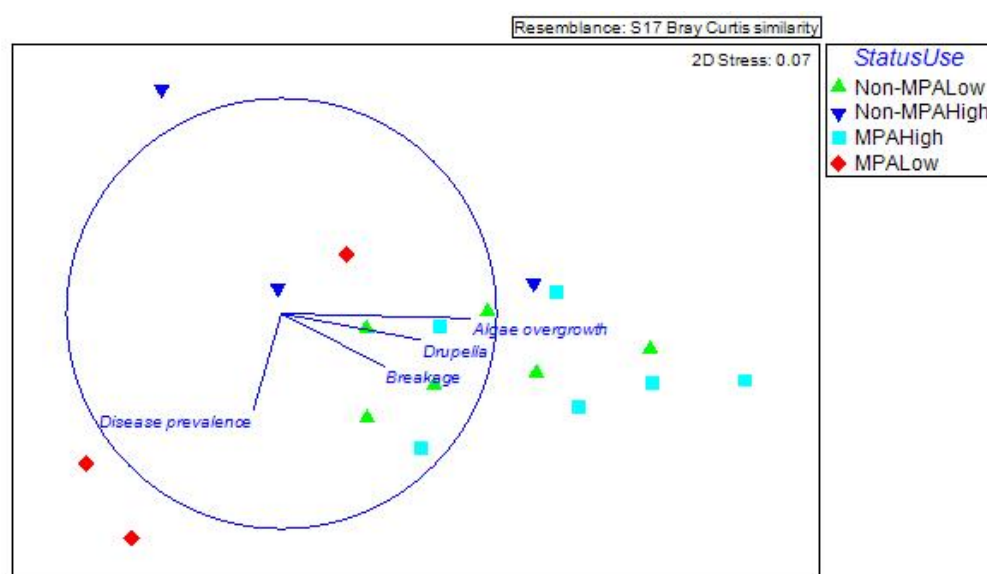


Figure 10. Non-metric multidimensional scaling analysis (MDS) plot with vectors (Pearson's correlation) representing similarities (Bray-Curtis matrix) among sites according to levels of disease and other compromised health indicators. Symbols and colours represent the combined MPA status and recreational use category for each transect. n= 3 transects/site.

#### 4.4. Environmental and local pressures

##### 4.4.1 MPA status comparison

Sedimentation was highest at MPA sites (Kruskal- Wallis:  $df=1$ ,  $p=0.043$ ; Figure 11). There was also a trend towards higher boat traffic traveling past MPA sites, but the trend was not statistically significant (ANOVA:  $F=0.102$ ;  $df=1$ ,  $p=0.753$ ; Figure 11). It is noteworthy that boat traffic was largely underestimated at one of the MPA sites (Twins) due to bad weather conditions on the days of the survey, thus mean boat traffic at MPA sites is likely higher than indicated. Diving pressure did not differ between sites differing in status, being equally high at MPA and non-MPA sites (Kruskal- Wallis:  $df=1$ ,  $p=0.306$ ; Figure 11). In contrast, snorkelling pressure was absent at MPA sites (Kruskal- Wallis:  $df=1$ ,  $p=0.003$ ; Figure 11).

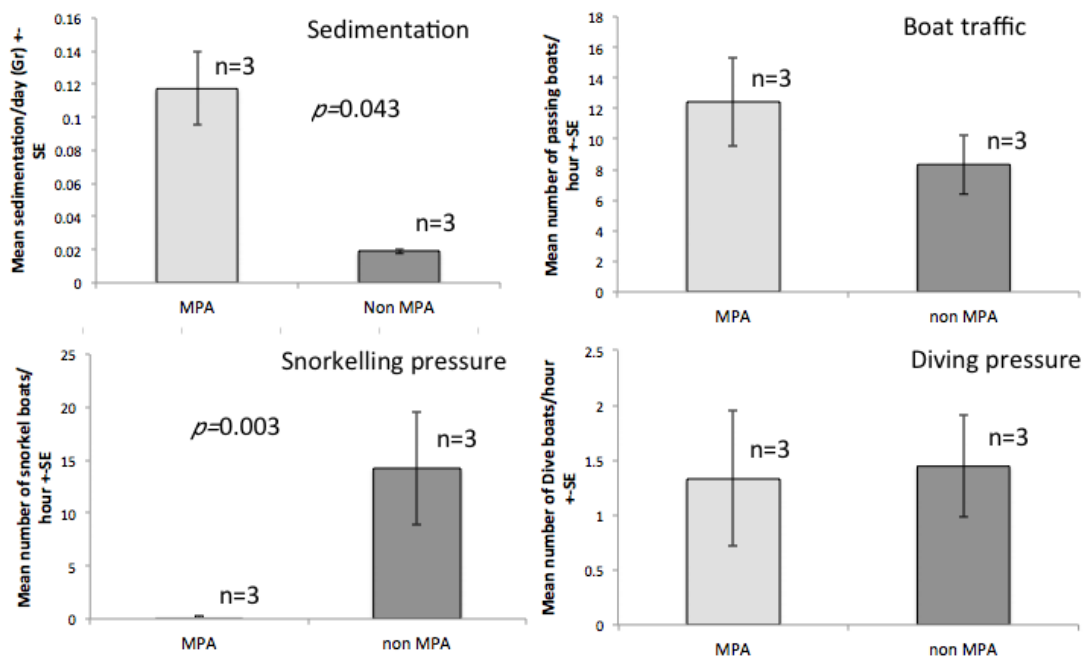


Figure 11. Comparisons of 4 indicators of environmental and anthropogenic pressure between sites differing in MPA status and past recreational use. MPA sites comprise 2 past high-use sites and 1 past low-use site; non-MPA sites comprise 2 past low-use sites and 1 past high-use site.  $n=3$  sites per Status category.

#### 4.4.2 Recreational use comparison

Statistically, none of the 4 indicators of environmental and anthropogenic pressure differed between past recreational use categories (Figure 12).

However, there was a clear trend towards higher sedimentation at past high-use sites compared to low-use sites, and also higher boat traffic at past high-use sites (Figure 12). As indicated above, boat traffic was probably underestimated at one of the high-use sites (Twins). There were also trends towards higher diving pressure at high-use sites, and higher snorkelling pressure at low-use sites (Figure 12).

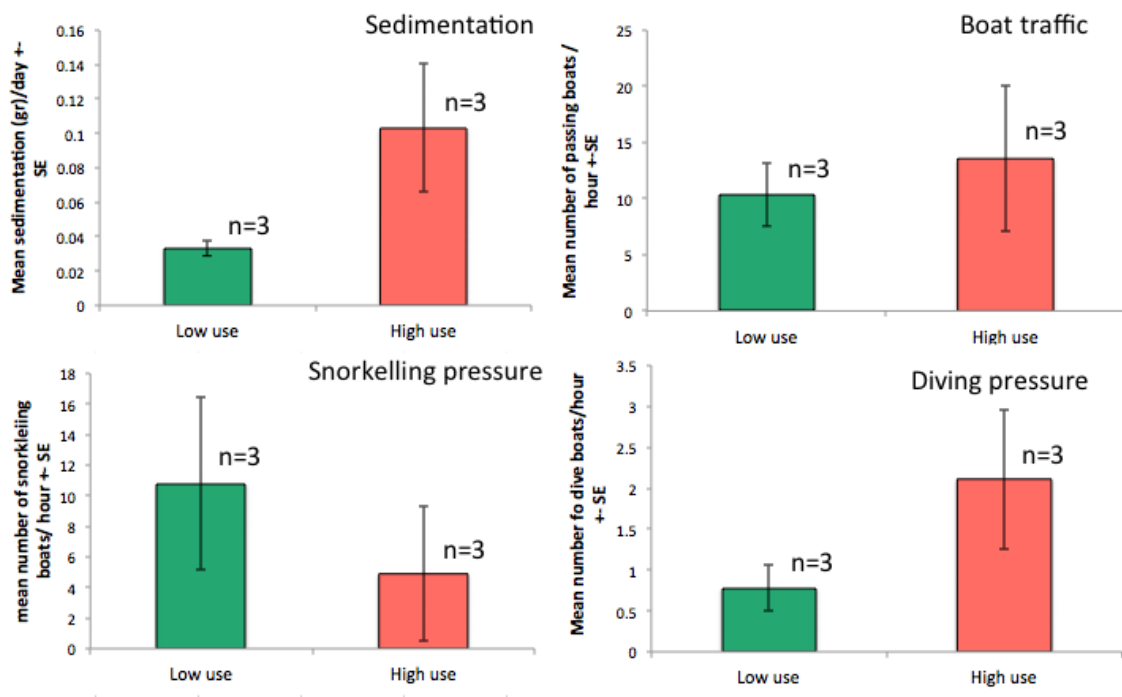


Figure 12. Comparison of 4 indicators of environmental and anthropogenic pressure between sites differing in past recreational use. n=3 sites per recreational use category

#### 4.5 Correlation matrix

Boat traffic was positively correlated with sedimentation and breakage (Figure 13), both of which were positively correlated with algae overgrowth and *Drupella* prevalence. Algae overgrowth and *Drupella* prevalence were also both positively correlated with prevalence of the foliose family Agariciidae, while sedimentation was negatively correlated with prevalence of the branching family Acroporidae (Figure 13).

There is thus a possible feedback loop, in which increased boat traffic and sedimentation decrease cover of branching corals in the family Acroporidae, providing space for weedy corals in the family Agariciidae to colonize, particularly in conditions where *Drupella* predation and algae overgrowth affect this latter family. Such a loop would lead to an overall decrease in coral diversity and health.

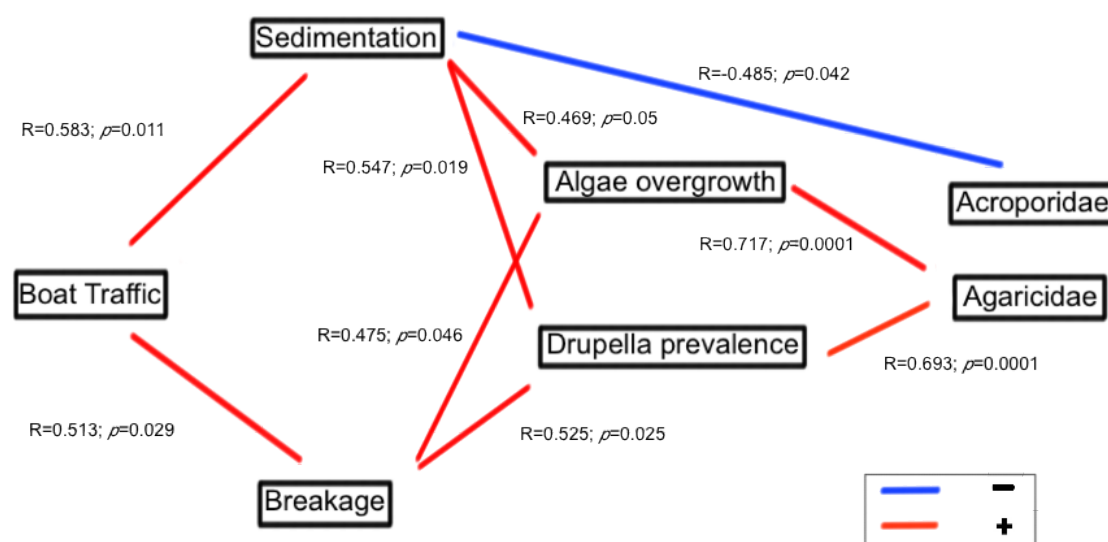


Figure 13. Sketch representing the network of significant correlations (Pearson's) among biological and environmental factors affecting the two most abundant coral families on Koh Tao reefs. Blue lines represent negative correlations; red lines represent positive correlations.

#### 4.6 DistLM

Sedimentation accounted for 10.3% of variation in coral health among study sites, whereas boat traffic accounted for only 1.7% of variation. Yet, none of these variables significantly drove patterns in coral health assemblages (Table 2). Sedimentation contributed to similarities in coral health assemblages at MPA high-use sites (light blue squares; Figure 14). Boat traffic contributed to similarities in coral health assemblages at non-MPA low-use sites (green triangles; Figure 14). According to vectors for coral health indicators (disease prevalence, algae overgrowth, *Drupella*, and breakage), all appeared to be driving coral health assemblages at MPA high-use sites (Figure 14).

Table 2. Results of distLM analysis evaluating the contributions of sedimentation and boat traffic to coral health assemblages.

Variable	SS(trace)	Pseudo-F	P	Prop.
Sedimentation	4199.3	1.8168	0.112	0.102
Boat traffic	2312.5	0.95192	0.445	0.056

AICc	R <sup>2</sup>	RSS	No. Vars
144.65	0.11991	36243	2

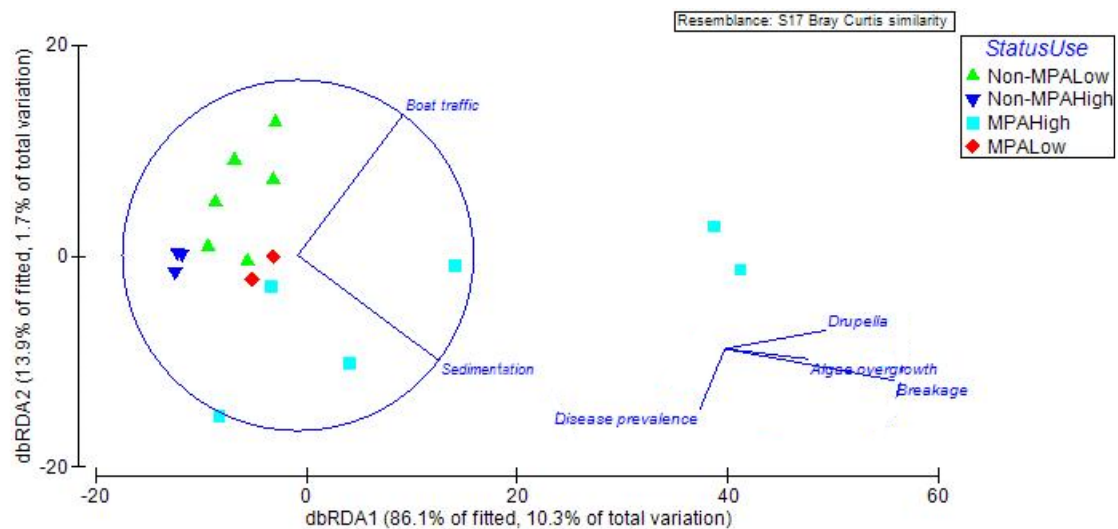


Figure 14. Distance-based redundancy analysis (dbRDA) plot visualizing similarities in coral health assemblages, with vectors from significant environmental (sedimentation) and anthropogenic (boat traffic) pressures overlaid. Vectors on the right are the original coral health indicators.

## 5. Discussion

Surveys at sites within three MPAs, one year after they had been established, revealed that coral health and community structure around Koh Tao was more reflective of past recreational use than current MPA protection status. At sites inside the MPA framework, all variables studied (hard coral cover, structure of coral family assemblages, disease prevalence, and densities of colonies showing other signs of compromised health) followed trends observed at past high use sites, reflecting past high use at 2 of the 3 MPA sites. Inversely, all variables studied at sites outside the MPA framework followed trends found at past low-use sites, reflecting past low use at 2 of the 3 non-MPA sites. Although my study does not document any positive effects of MPA status on the health and diversity of coral communities of Koh Tao, a finding consistent with most previous studies of MPAs (Coelho and Manfrino 2007, Page et al. 2009, Hsieh et al. 2011), this result is not surprising since the MPAs were rezoned and enforced only a year prior to the study. A recent global analysis suggests that the efficiency of MPAs to prevent coral loss is very dependent on the duration of protection (Selig and Bruno 2010). This study suggests that up to 15 years could be needed before positive changes are observed inside MPAs in the Indo-Pacific because of specific life history and community structure characteristics, such as the high disease susceptibility of the typically dominant coral family Acroporidae. Moreover, very little information is available about fishing pressure around the island and adequate enforcement will require full community support. Fortunately, the rezoning is an initiative from the community of Koh Tao, and examples of good conduct have already been documented, but it is likely that more than a year is needed before any effects of the new MPA framework can be observed. In this context, my study provides an important baseline for future evaluation of the capacity of these MPAs to ameliorate coral health.

### 5.1. Current state of Koh Tao's reef

Our study revealed overall high levels of mean hard coral cover and low levels of coral disease prevalence around the island of Koh Tao. Disease prevalence

ranged from 0.1 to 1.8%, which is at the lowest end of the range recorded for the Indo-Pacific (e.g. Harvell et al. 2007). Constant warm temperatures throughout the year 2013, coupled with no thermal extremes (C. Scott, personal communication), could have contributed to the overall low disease prevalence recorded during our surveys. Links between temperature anomalies and outbreaks of coral diseases are highly complex. For example, a recent study suggests that, whereas cold winters can reduce pathogen loading, mild winters maintain pathogen populations so that hot summer events then trigger disease outbreaks (Heron et al. 2010). On the other hand, warm winters can also increase coral host resistance to disease (e.g. production of antibiotics in coral mucus) (Ritchie 2006, Heron et al. 2010). I thus hypothesize that constant warm temperatures and absence of acute temperature stress throughout the year preceding my surveys meant that the corals of Koh Tao were in good condition with well-developed capacity for disease resistance.

The high coral cover and trend for higher coral cover in past low-use / current non-MPA sites, where disease prevalence tended to be lower, is at odds with a number of studies that have found increased coral disease prevalence to be associated with high host abundance (Bruno et al. 2007, Myers and Raymundo 2009, Ruiz and Moreno 2012). One hypothesis is that high coral cover reduces the distance between neighbouring corals, thereby increasing the potential for transmission of pathogens (Bruno et al. 2007). Direct competition among coral colonies may also create injuries that become entry wounds for infection (Lang and Chornesky 1990, Bruno et al. 2007). Another hypothesis is that high coral cover is associated with higher densities of corallivorous fish and invertebrates that are potential vectors of coral diseases (e.g. Bruno et al. 2007, Raymundo et al. 2009, Chong-Seng et al. 2011, Onton et al. 2011). In particular, the crown-of-thorns starfish *Acanthaster planci* and *Drupella* snails, both of which cause significant feeding injuries that provide entry wounds for pathogens, have been shown to be disease vectors (Antonius and Riegl 1998, Bruno et al. 2007, Nugues and Bak 2009, Onton et al. 2011, Nicolet et al. 2013). The high overall hard coral cover around the island may thus constitute an increased risk for future disease

outbreaks, particularly if an extreme environmental event, such as a warm thermal anomaly, reduces coral disease resistance.

Evidence is emerging that hard coral cover is a poor indicator of reef condition (e.g. Edinger and Risk 2000, Cleary et al. 2008, Darling et al. 2013). Sites with high coral cover may be composed of monospecific assemblages, and thus have low habitat complexity and low overall biodiversity (e.g. Aronson and Precht 1995, Edinger and Risk 2000). Interestingly, the prevalence of branching acroporids at Koh Tao sites was low overall, but tended to be higher at past low-use sites and non-MPA sites. Conversely, corals in the family Agariciidae, mostly constituted of *Pavona cactus*, were more abundant at past high-use sites. These results are in line with a recent study which found that competitive species like acroporids were most vulnerable to multiple stressors, and that reefs were tending towards dominance by weedy species (Darling et al. 2013). *Pavona* species were thus characterised as “survivors”, whereas *Acropora* species were characterised as “losers” (Darling et al. 2013). Branching corals are also believed to be the most vulnerable to diving impacts (Hassler and Ott 2008, Guzner et al. 2010). With the current levels of environmental and anthropogenic pressures around Koh Tao, there is a risk of a gradual shift towards higher prevalence of mono-specific stands of Agariciidae around the island (Roupheael and Inglis 2002, Guzner et al. 2010).

## 5.2. Environmental and anthropogenic pressures and links to coral health

Diving pressure was high, approximately 1.8 dive boats/ hour, at sites classified as both MPAs and non-MPAs. Even using a low estimate of 20 divers/ dive boat, this equates to 36 divers/ hour, and becomes a mean of around 150 divers/day for all 6 sites surveyed. A limit of 6000 divers/year has been suggested to be the carrying capacity for diving on coral reefs (Hawkins and Roberts 1997), a limit that would be reached within 40 days at this rate of diving/day. This study thus confirms that diving pressure on Koh Tao reefs is enormous. The impact of recreational diving is of increasing concern for coral reefs (Davis and Tisdell 1995; Barker and Roberts 2004, Krieger and Chadwick 2013). Divers directly

affect corals through physical contact, caused by touching, fin damage, dangling equipment, and sediment re-suspension. Heavily dived sites have been associated with a higher proportion of damaged colonies and lower live hard coral cover (Hassler and Ott 2008, Krieger and Chadwick 2013). Diving also has strong indirect effects on the reef. For example, increased predation by *Drupella* snails has been reported at heavily dived sites in Eliat, Israel (Guzner et al. 2010). In my study, predation by *Drupella* snails was not directly related to high diving pressure, but rather to high boat traffic, sedimentation and breakage. *Drupella* snails have been shown to decrease skeletal growth (Guzner et al. 2010), be an effective vector for the transmission of brown band disease (Nicolet et al. 2013), and more generally, to be associated with a range of diseases like white syndromes, black band disease and skeletal eroding band (Antonius and Riegl 1998, Shafir et al. 2008, Onton et al. 2011). High levels of *Drupella* predation have been documented on Koh Tao reefs, with densities reaching up to 90 individuals on a single colony after the bleaching event in 2010 (Hoeksema et al. 2013). Our observations also support the theory that *Drupella* preferentially attack corals experiencing physiological stress (e.g. Guzner et al. 2010), with *Drupella* prevalence positively correlated with sedimentation and breakage associated with high boat traffic.

Boat traffic was also associated with high algal overgrowth. Decreases in coral cover worldwide are leading to community phase shifts, with corals being replaced by macro-algal assemblages (e.g. McCook et al. 2001, Hughes et al. 2007). Algae typically proliferate after disturbances causing coral mortality and once established, they reduce space available for coral recruits and juveniles (McCook et al. 2001). Some species of macro-algae have been shown to inhibit recruitment of coral larvae and/or lead to increased mortality of coral recruits (Kuffner et al. 2006). Contact between algae and adult coral colonies can also be very deleterious to coral health (e.g. Nugues 2004, Vega Thurber et al. 2012), and facilitate the spread of coral diseases (Smith et al. 2006), as demonstrated for white plague type II in the Netherlands Antilles (Nugues et al. 2004). Contact with algae may also cause abrasions that serve as entry points for pathogens (McCook et al. 2001). More generally, algae have been shown to affect the

fecundity, growth and survival of adult colonies, and alter the microbial symbiont communities of corals (Hughes et al. 2007, Vega Thurber et al. 2012). Increases in nutrients, pollution, and sea temperatures, and decreases in herbivory have been suggested as potential factors that increase algal overgrowth on reefs (e.g. Aronson and Precht 2001, Szmant 2002, Nugues et al. 2004). In my study, algal overgrowth also increased with sedimentation. This result is in line with a recent study in which increased sedimentation increased the growth of turf algae, probably due to increased availability of nutrients from the sediments (Goatley and Bellwood 2013). Moreover, sedimentation alone can affect the health of coral communities, increasing the potential for contact with pathogens and contaminants, creating physical abrasions to coral tissues, decreasing energy available through sediment shading, and increasing energetic costs associated with sediment removal and impaired feeding (e.g. Fabricius 2005, Haapkyla et al. 2011, Erftemeijer et al. 2012).

MPAs have been advocated as a strong tool to mitigate phase-shifts from coral to algal-dominated reefs (e.g. Hughes et al. 2007). When user compliance is high, MPAs have the potential to maintain or increase herbivory, which then controls macro-algae abundance and has the potential to increase the resilience of coral communities (Hughes et al. 2007). Yet, in our study, reefs inside MPAs have high diving pressure, high boat traffic and high sedimentation. The question remains whether potential increased herbivory at these highly disturbed reefs will be enough to mitigate algal overgrowth.

### 5.3 Conclusions

This study represents a strong baseline for future studies of the capacity of MPAs to protect and possibly enhance coral health and diversity around the island of Koh Tao. So far, health and diversity of the coral communities reflect the past history of recreational use, with past high-use sites characterised by lower hard coral cover, lower prevalence of the branching coral family *Acroporidae* combined with higher prevalence of the weedy family *Agariciidae*, higher disease prevalence and higher density of colonies with other indicators of compromised

health. Past high-use sites are also subjected to higher sedimentation, boat traffic and diving pressure than past low-use sites. One challenge for the successful management of the reefs of Koh Tao is that the MPA framework mostly encompasses past high-use sites. My study revealed that past high-use sites inside MPAs were the most affected by sedimentation and displayed highest levels of algal overgrowth, predation by *Drupella*, and breakage. Moreover, with the current rate of tourism development, past low-use sites are quickly becoming high-use sites as well. The creation of MPAs that only address the issue of fishing pressure, with no further incentives to control tourism expansion, is thus likely to be insufficient to protect the island's coral reefs. Other measures, such as seasonal closures of some sites, and/or limits to the number of divers allowed at each site per year should be considered. This said, the large number of threats and degraded reef condition found in this study makes the island of Koh Tao an excellent site for future long-term evaluation of the role of MPAs in ameliorating the health of coral communities.

## 6. References

- Antonius A, Riegl B (1998) Coral diseases and *Drupella cornus* invasion in the Red Sea. *Coral Reefs* 17:48
- Aronson RB, Precht WF (1995) Landscape patterns of reef coral diversity: a test of the intermediate disturbance hypothesis. *J Exp Mar Biol Ecol* 192 : 1-14
- Aronson, R.B. & Pretch, W.F. (2001). Evolutionary paleoecology of Caribbean coral reefs. In: *Evolutionary Paleoecology: The Ecological Context of Macroevolutionary Change* (eds Allmon, D. & Bottjer, D.J.). Columbia University Press, New York, NY, pp. 171–233
- Barker NHL, Roberts CM (2004) Scuba diver behaviour and the management of diving impacts on coral reefs. *Biol Conserv* 120:481–489
- Bruno JF, Selig ER, Casey KS, Page CA, Willis BL, et al. (2007) Thermal stress and coral cover as drivers of coral disease outbreaks. *PLoS Biol* 5: 1220–1227
- Chong-Seng KM, Cole AJ, Pratchett MS, Willis BL (2011) Selective feeding by coral reef fishes on coral lesions associated with brown band and black band disease. *Coral Reefs* 30:473–481

- Cleary et al. (2008) Relating variation in species composition to environmental variables : a multi-taxon study in an Indonesian coral reef complex. *Aqua Sci* 70 :419-431
- Cole AJ, Chong Seng KM, Pratchett MS, Jones GP (2009) Coral- feeding fishes slow progression of black band disease. *Coral Reefs* 28:965
- Davis D, Tisdell C (1995) Recreational scuba-diving and carrying capacity in marine protected areas. *Ocean Coast Manage* 26:19–40
- De’ath G, Fabricius KE (2010) Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. *Ecol Appl* 10: 840–850.
- Edinger EN, Risk MJ (2000) Reef classification by coral morphology predicts coral reef conservation value. *Biol Conserv* 92: 1-13
- Fabricius KE (2005) Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis. *Mar Poll Bull* 50: 125–146.
- Gorley RN, Clarke KR (2008) PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods. PRIMER\_E: Plymouth ,UK
- Guzner B, Novplansky A, Chadwick NE (2010) Indirect impacts of recreational scuba diving: patterns of growth and predation in branching stony corals. *B Mar Sci* 86:727–742
- Haapkyla J, Unsworth RKF, Flavell M, Bourne DG, Schaffelke B, Willis BL (2011) Seasonal rainfall and runoff promote coral disease on an inshore reef. *PLoS ONE* 6(2):e16893
- Harvell CD, Mitchell CE, Ward JR, Altizer S, Dobson AP, et al. (2002) Climate Warming and Disease Risks for Terrestrial and Marine Biota. *Science* 296: 2158–2162.
- Hasler H, Ott JA (2008) Diving down the reefs? Intensive diving tourism threatens the reefs of the northern Red Sea. *Mar Pollut Bull* 56:1788–1794
- Hughes TP, Tanner JE (2000) Recruitment failure, life histories, and long-term decline of Caribbean corals. *Ecology* 81: 2250–2263
- Jameson SC, Tupper MH, Ridley JM (2002) The three screen doors: *can* marine ‘protected’ areas be effective? *Mar Poll Bull* 44: 1177–1183
- Lamb JB, Willis BL (2011) Using coral disease prevalence to assess the effects of concentrating tourism activities on offshore reefs in a tropical marine park. *Conserv Biol* 25:1044–1052

Lang JC, Chornesky EA (1990) Competition between scleractinian reef corals—A review of mechanisms and effects. In: Dubinsky Z, editor. Ecosystems of the world. Volume 25: Coral reefs. Amsterdam: Elsevier. pp. 209–252.

McClanahan TR (2008) Response of the coral reef benthos and herbivory to fishery closure management and the 1998 ENSO disturbance. *Oecologia* 155: 169–177

Motulsky H, Christopoulos A (2004) Fitting models to biological data using linear and nonlinear regression: a practical guide to curve fitting. Oxford, UK: Oxford University Press.

Mumby PJ (2009) Herbivory versus corallivory: are parrotfish good or bad for Caribbean coral reefs? *Coral Reefs* 28: 683–690

Nugues MM, Smith GW, Hooidonk RJ, Seabra MI, Bak RPM (2004) Algal contact as a trigger for coral disease. *Ecol Lett* 7: 919–923

Nugues MM, Bak RPM (2009) Brown-band syndrome on feeding scars of the crown-of-thorn starfish *Acanthaster planci*. *Coral Reefs* 28:507–510

Onton K, Page CA, Wilson SK, Neale S, Armstrong S (2011) Distribution and drivers of coral disease at Ningaloo reef, Indian Ocean. *Mar Ecol Prog Ser* 433:75–84

Page CA, Baker DM, Harvell CD, Golbuu Y, Raymundo L, Neale SJ, Rosell KB, Rypien KL, Andras JP, Willis BL (2009) Influence of marine reserves on coral disease prevalence. *Dis Aquat Org* 87:135–150

Pratchett MS, Berumen ML (2008) Inter-specific variation in distributions and diets of coral reef butterflyfishes (Teleostei: Chaetodontidae). *J Fish Biol* 73:1730–1747

Ritchie KB (2006) Regulation of microbial populations by coral surface mucus and mucus-associated bacteria. *Mar Ecol Prog Ser* 322: 1–14

Rouphael AB, Inglis GJ (2002) Increased spatial and temporal variability in coral damage caused by recreational scuba diving. *Ecol Appl* 12: 427–440

Sato Y, Bourne DG, Willis BL (2009) Dynamics of seasonal outbreaks of black band disease in an assemblage of *Montipora* species at Pelorus Island (Great Barrier Reef, Australia). *Proc R Biol Soc B* 276: 2795–2803

Shafir S, Gur O, Rinkevich B (2008) A *Drupella cornus* outbreak in the northern Gulf of Eilat and changes in coral prey. *Coral Reefs* 27:379

Szmant AM (2002). Nutrient enrichment on coral reefs: is it a major cause of coral reef decline? *Estuaries* 25: 743–766

Smith JE, Shaw M, Edwards RA, Obura D, Pantos O, et al. (2006) Indirect effects of algae on coral: algae-mediated, microbe-induced coral mortality. *Ecol Lett* 9: 835–845.

Willis BL, Page CS, Dinsdale EA (2004) Coral disease on the Great Barrier Reef. In: Rosenberg E, Loya Y (eds) *Coral disease and health*. Springer, Berlin, pp 69–104