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The Impact of Sea Surface Temperature on Outbreaks of Acanthaster planci on the Great Barrier Reef

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Abstract

The causes of increasing outbreaks of *Acanthaster planci* on the Great Barrier Reef have been a point of hot debate in recent years. It is unknown whether the increased success is due to nutrient runoff, salinity levels, or a decrease in predation, among other possibilities. In this paper I argue that the primary influence on outbreak status is sea surface temperature. From existing literature, I demonstrate that sea surface temperature in the Great Barrier Reef has increased by 0.4 °C per year over the past three decades. I attempt to tie this increase with an increase in frequency of *A. planci* outbreaks on a selection of reefs throughout the Great Barrier Reef region. Due to the development of *A. planci,* specifically the fact that it takes them between 2 and 3 years to reach full maturity, I examined the potential relationship between an outbreak and the sea surface temperature 1 and 2 years before the event. Through my exploration of the data and my subsequent data analysis, it is clear that there are no statistically significant results when comparing the three classifications of outbreak (active, incipient, and recovering) and not outbreaking populations with temperature at each of the three time relationships. However, when I considered the three stages of outbreak to be “affected” and those not outbreaking to be “unaffected”, I found a statistically significant relationship. This finding has important implications when looking at the temperature changes that have been predicted for the Great Barrier Reef region due to global climate change. If the water temperature continues to increase, *A. planci* will more often be living within their optimal temperature range and will be more successful, continue to have major outbreaks that devastate the reef ecosystem, and eventually destroy it all together.

Introduction

Coral Reefs Under Stress

Coral reefs are some of the most biologically diverse and productive ecosystems on earth. A healthy reef buzzes with activity, sounds, and colors and is populated by dense aggregations of fish and invertebrates. A coral reef can contain thousands of distinct species of marine plants and animals.

Healthy reef systems are dominated by reef-building corals, which have a role analogous to trees in a forest; generating the physical framework of the reef and providing resources for thousands of associated plants and animals (Bruno, 2010). Coral reefs are composed mainly of scleractinian corals (Stoddart, 1969). Most scleractinians, and particularly those that build reefs, are colonial, anemone-like animals that shelter microscopic algae and produce skeletal structures composed primarily of calcium carbonate (Spalding, 2001). The structure built up by corals over thousands of years provides complex refuges in which animals can hide from predators. Corals derive energy and nutrients through a symbiotic relationship with photosynthetic algae as well as through suspension feeding (Cowen, 1988). When corals die, the abundance of reef fish quickly decreases, mainly due to the lack of places for larval fish to inhabit as they leave the open water and settle on the reef where they will spend their adult lives (Kleypas and Gattuso, 2006). The loss of a reef can be catastrophic for the community and ecosystem that is built around it.

Although corals are found throughout the worlds oceans, reef-building corals are confined to waters that exhibit a specific few characteristics: the water must be warm, rather clear, nutrient-poor, and saline, though exact ranges for each of these variables differs by species (Haynes, 2001). Physiologically and behaviorally, corals have evolved to take advantage of this unique environment and thrive.

Today, very few coral reefs remain in pristine condition. Corals and fishes are much less plentiful than they were only a few decades ago (Jackson, 1997). This decline is due in part to the numerous stressors that coral reefs are exposed to in the modern environment, some of which have been tied to the global climate changes that are currently occurring. Several factors are conspiring to put tremendous stress on coral reef systems, including water temperature, storm intensity, and ocean acidity.

Warming seas pose a serious risk to the world's coral reef ecosystems. Corals are extremely sensitive to changes in temperature (Doney et al., 2012). Rising sea surface temperatures will affect every aspect of such an ecosystem. Temperature is a key environmental factor controlling the distribution and diversity of marine life. It is a critical component of reef building and has a major impact on the rate of coral reef growth (Fraser and Currie, 1996). All animals and plants have temperature limits and when these are reached, natural processes such as photosynthesis and reproduction can fail to function properly. Atmospheric temperatures, lack of cloud cover, and freshwater run-off all contribute to rising sea surface temperatures.

Increases in water temperatures following the trend that has been seen in recent years (an increase of approximately 0.4°C per year) have in the past and will continue to cause events of mass coral bleaching. Bleaching occurs when coral polyps, stressed by heat or ultraviolet radiation, expel the symbiotic algae (zooxanthellae) that live within their calcium carbonate tissues and provide corals with most necessary food and oxygen (Hoegh-Guldberg, 1999). When the algae are expelled, the coral appears white (Brown and Ogden, 1993). Corals can recover from short periods of bleaching, but as the length and severity of the stress increases with temperature, the ability of the coral to bounce back from such events dramatically decreases (Muller-Parker and D'Elia, 1997).

The Great Barrier Reef is the largest coral reef in the world. It is about 1,250 miles long and located off the northeastern coast of Australia (Pulsford, 1996). The temperature gradient along the Great Barrier Reef has shifted substantially over the last century and will continue to do so. The average annual sea surface temperatures on the Great Barrier Reef are projected to continue to warm over the coming century; there is predicted to be an increase of 0.5°C by 2020, 1.2°C by 2050, and, 3°C warmer than the average of all annual sea surface temperatures from 1961 to 1990, which is 25.9 °C (Lough, 2007). Whatever climate scenario is used, all projections are outside the observed Great Barrier Reef sea surface temperature climate range up to 1990 by the year 2035 (ibid). The two warmest five-year average sea surface temperatures in the last 400 years have been recorded in the last three decades (Jackson, 1992). Analysis of coral cores in centuries-old corals suggests that current temperatures are warmer now than any time in the last three centuries. According to predictions, by 2035 the average sea surface temperature will be consistently warmer than any previously recorded (Corrège, 2006).

The Crown of Thorns

Acanthaster planci, more commonly known as the crown of thorns sea star, is found throughout the Indian and Pacific Oceans, including on the Great Barrier Reef. Their bodies can grow to one meter in diameter, and are covered with sharp poisonous spines that serve as a defense against potential predators (Chesher, 1969). As can be seen in the photograph in Figure 1, *A. planci* has between thirteen and sixteen arms that extend out radially from a central body. When threatened, individuals can autotomize arms if necessary. Up to 60% of individuals in a population can have missing arms. This can be a useful indication of predation patterns (Nishida and Lucas, 1988). They vary in color, with their spines generally having a different color from the rest of their body (Birkeland and Lucus, 1990). Interestingly, they have different color patterns depending on their environment. They are purple and blue in the Maldives and Thailand, red and grey in the Great Barrier Reef, and green and red in Hawaii.

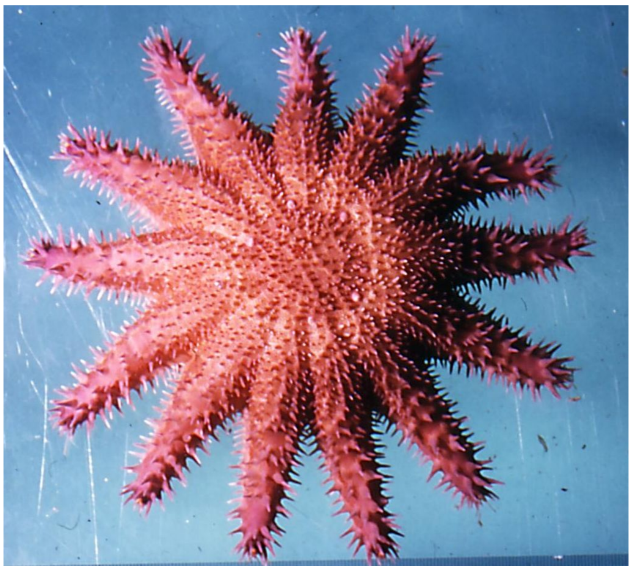


Figure 1. A fully mature *A. Planci* (Wikipedia).

A. planci can take over coral reefs quickly due to their ability to spawn millions of eggs a year. Once fertilized, eggs develop into larvae in 24 hours, which then disperse throughout the water column. A month after hatching, the larvae go through metamorphosis to become juvenile sea stars with five arms. They continue growing arms for the next several months. Then, two years after birth, *A. planci* are fully mature and begin to reproduce (Johnson, Sutton, Olson, and Giddins, 1991).

A. planci reproduce sexually through broadcast spawning from December to April on the Great Barrier Reef. Peak spawning activity has been recorded during December in the central Great Barrier Reef when water temperature is approximately 28 °C. The exact timing of spawning is not predictable and spawning has been observed during the day and at night, and at various stages of the tidal and lunar cycles (Babcock and Mundy, 1998). The starfish release eggs and sperm into the water through pores on the top of their central disc. The large volumes of sperm released by male starfish are the primary cause of the high rates of fertilization derived from individuals that might not be very close together. Under natural conditions, this means that fertilization rates can be high even though individuals may be spaced widely apart. This ability allows the geographic range of the species to expand and change distribution patterns quickly to effectively capitalize on the most suitable environments. (Hoey and Chin, 2004).

Reproductive output is also related to the availability of food. When there is little food available, adult females can reabsorb their body wall and skeletal tissues, which reduces their size, lifespan, and overall reproductive capacity. However, when food is plentiful, females are able to take advantage of the favorable conditions and produce more eggs (Stump, 1993). When the eggs are fertilized, they develop into larvae that spend three weeks drifting as planktonic organisms in ocean currents (Keesing and Lucas, 1992). The larvae that develop have small hairs, called cilia, which propel them through the water and produce local feeding microcurrents that trap the plankton on which they feed (Hoegh-Guldberg, 1994). When they are about two millimeters across, the larvae settle on shallow reefs. They live among rocks and rubble on the reef and are almost invisible until they are about six months old. As can be seen in Figure 2, they eventually become juvenile starfish, which feed on coralline algae (Benzie, 1999).

Figure 2. Juvenile *Acanthaster planci* (GBRMPA, 2004).

A. planci first breeds when it is two years old and breeds for a total of five to seven years thereafter. Each female can produce up to 60 million eggs during a single spawning season (Lucas, 1982). The starfish tend to gather together to spawn which increases the chance of fertilizing the eggs. Indeed, fertilization rates in the field for *A. planci* are the highest for any invertebrate. Therefore, a small population of *A. planci* can produce a very large number of offspring (Yokochi and Ogura, 1987).

There are three specific age classes for *A. planci*: juvenile (younger than 18 months), sub-adult (18-21 months), and adult (older than 21 months). The complete life cycle of *A. planci* can be seen in Figure 3. Development is rapid for juveniles, at a rate of up to 16.7 millimeters per month, and slows with the transition from sub-adult to adult. Development steadies at about four millimeters per month once full maturity is reached (Stump, 1996).

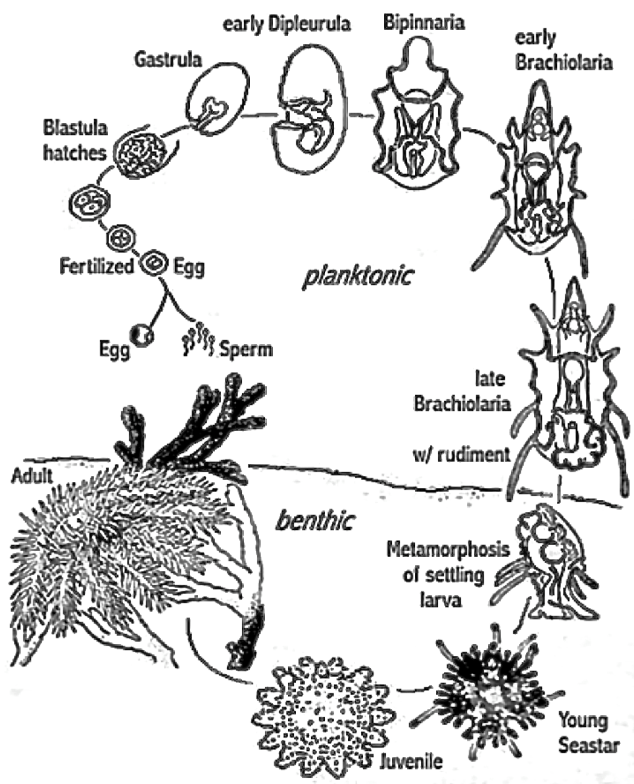


Figure 3. The life cycle of *A. planci* (Hart and Russ, 1996).

Unlike any other species of sea star, *A. planci* prey directly on live corals, rather than the organisms residing within them. They spend about half their time feeding. Individuals larger than 40 centimeters have been known to feed during the day while individuals smaller than 20 centimeters usually feed at night (Lucas, 1982). When there are few *A. planci* in a reef area, they are very cryptic, tending to hide in the reef and under corals during the day.

The choice of diets for *A. planci* follows principles of optimal foraging theory, by which organisms forage so as to optimize their energy intake over time (Ormond, Bradbury, and Bainbridge, 1990). Pratchett (2007) found that *A. planci* exhibited strong and consistent hierarchy of feeding preferences. On the Great Barrier Reef, *Acropora* was found to be the preferred genus of corals. The most readily eaten coral species was *Acropora hyacinthus*, followed by *A. gemmifera, A. nasuta, A. formosa*, *Stylophora pistillata, Montipora undata*, and *Pocillopora damicornis.* These are all crucial to the structure and compositon of the Great Barrier Reef. Most corals were eventually consumed, showing that when food is limited *A. planci* is likely to eat virtually all coral species present, causing extreme devastation to coral reef ecosystems (Pratchett, 2007). These feeding preferences are important because they can have major impacts on the geographical distribution of *A. planci,* as well as the composition of the ecosystem at various population densities.

In some cases, *A. planci* may only eat a portion of the entire coral colony. As a result, the impacted reef can recover quite rapidly from low levels of predation by *A.*

planci (De'ath and Moran, 1998). Although most studies agree that *A. planci* is incredibly harmful to the reefs that it inhabits, it has been shown that at low enough population densities *A. planci* can be beneficial. Through consumption, they prevent one species of coral from monopolizing the reef area and therefore increase the diversity of the reef as a whole. Some reefs seem to support small populations of *A. planci* for many years, with only a small reduction in coral cover. Reefs with one *A. planci* per 50 square meters had much more coral biodiversity than reefs that had one per 50 square kilometers (Porter, 1972).

Scientists estimate that a healthy coral reef with about 40-50% coral cover can support about 20-30 *A. planci* per hectare (ten square kilometers). But when the species is present in higher densities, there is intense competition for food and a wider variety of corals will be eaten, leading to devastation of the reef ecosystem (Done, 1999).

Generally, an outbreak is a rapid and dramatic increase in the abundance of an organism to a level above which the available resources can sustain that is completely independent of changes in the resource base. For *A. planci,* this means a population exceeding 200 sea stars per hectare (Glynn, 1974). During an outbreak, as can be seen in Figure 4, reefs can be populated to the point where individual sea stars are layered on top of each other. They can eat so much that they can kill most of the living coral in that part of the reef, reducing hard coral cover to less than 1% (Cameron, Endean, and DeVantier, 1991). Reefs affected in this way can take many years to recover, if they ever do. Results from fine-scale surveys indicate that coral cover of more than 10% is needed for juvenile starfish to survive and grow (Walbran and Henderson, 1992).

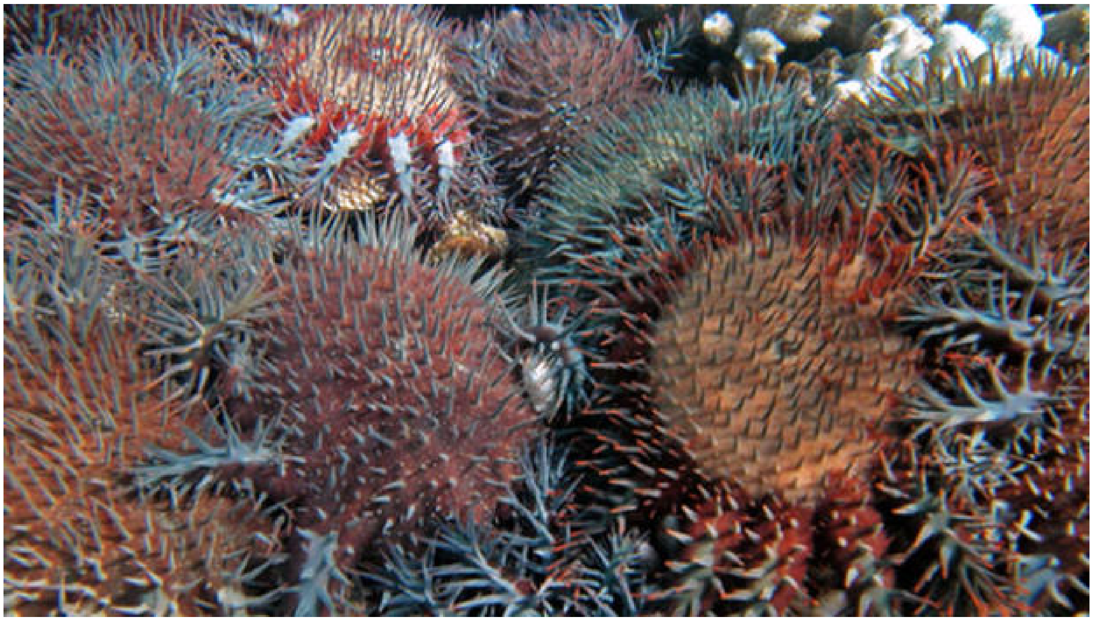


Figure 4. An active outbreak of *A. planci* (Fabricius, 2012).

Since the invention of SCUBA equipment in the mid-20th century (when reefs first started to be seriously monitored and recorded), cycles of *A. planci* outbreaks on the Great Barrier Reef have been observed (Van Woesik and Done, 1997). There have been three noted major outbreaks (1962 to 1976, 1979 to 1991, and 1993 to 2004 respectively), though likely there were others before records were reliably kept (Uthicke, Schaffelke, and Byrne, 2009). These outbreaks have followed the same general pattern, starting in the northern section of the Great Barrier Reef near Cairns and progressing southward over the following ten to 15 years. It is thought that the north to south spread of the outbreak is due to *A. planci* larvae being transported from one reef to another by the south flowing East Australian Current (Moran, Bradbury, and Reichelt, 1988).

The absence of historical information on the timing of outbreaks prior to 1960 makes it difficult to assess whether the pattern and intensity of outbreaks has changed. It is also difficult to determine if the length of time between outbreaks is changing, which would affect the extent of coral recovery between outbreaks (Engelhardt et al., 2001). This lack of information has lead to a debate among scientists about the history and nature of outbreaks in this region. Some scientists, such as Brodie et al. (2005) believe outbreaks are a result of human actions and have been occurring since the 1960's. They suggest that enhanced nutrient supply is critical for enhanced *A. planci* larval development, noting that nutrient discharges from rivers have increased at least four-fold in the central Great Barrier Reef over the last several decades. Others have studied skeletal elements of *A. planci* on impacted reefs and believe the outbreaks are a cyclical part of the marine ecosystem and have occurred periodically for over 7000 years (Keesing et al., 1992).

Though a great deal is known about *A. planci* and the general coral reef ecosystem in which they take part, there is no certain explanation for why these outbreaks have taken place. There are four main hypotheses regarding the reason for outbreaks of *A. planci* in recent years. The first is the natural event hypothesis, which states that the recorded series of *A. planci* outbreaks are a part of the natural ecological cycle of the reef ecosystem. It claims that *A. planci* have a number of modifications and traits due to their life history that cause large fluctuations in populations and make them predisposed to have periodic outbreaks. These include high fecundity (lifetime production of 12-24 million eggs by females), an extended larval dispersal phase (that enables migration from their natal reef through water currents), a long lifespan (more than eight years which allows multiple breeding cycles), a large stomach (allowing feeding on coral, a high energy food source), and a rapid growth pattern (allowing them to reach reproductive maturity within two years of settling on a reef). At certain times when temperature, salinity, and other environmental conditions become favorable, the naturally high reproductive capacity of the *A. planci* allows them to capitalize on these conditions and produce especially large numbers of juveniles that results in an outbreak (Babcock and Mundy, 1993).

The second hypothesis, known as the larval survival hypothesis, is based on the idea that a small increase in the survival rate of *A. planci* larvae can result in a massive influx of juvenile starfish on reefs that are further along in the current flow. This could be due to the fact that higher levels of nutrients from coastal areas cause coastal waters to become nutrient rich, resulting in a phytoplankton bloom that provides more food for *A. planci* larvae. Larval survival rates increase and subsequently cause outbreaks to develop (Moran, 1998). This hypothesis falls short because it does not seem to explain why outbreaks are intermittent and scattered throughout the reef area.

According to the third hypothesis, the predator removal hypothesis, fishing and shell collecting by humans have led to smaller populations of the key predators of *A. planci*. These predators include the giant triton snail, humphead Maori wrasse, sweetlip emperor, and starry puffer fish. This lack of predators allow for the starfish population to increase beyond natural and sustainable levels (Pearson and Endean, 1969). Juvenile *A. planci* are more prone to predation as they have less developed spines and external support. Field studies have shown that juvenile starfish are preyed upon by a variety of benthic organisms including crabs, shrimp, and fish (Keesing and Halford, 1992).

The final hypothesis, which could be considered an offshoot of the natural event hypothesis and is the focus of this paper, says that rising sea surface temperatures in the Great Barrier Reef have allowed *A. planci* to flourish and have lead to an increase in the frequency of outbreaks in both time and location.

In this paper, I hypothesize that the rise in sea surface temperatures due to global warming has caused outbreaks of *A. planci* on the Great Barrier Reef. Given the current pattern of increase of sea surface temperature, these outbreaks may cause the destruction of the entire Great Barrier Reef system in the near future.

Methods and Materials

Data on *A. planci* outbreaks was collected from surveys by independent scientists or indirectly gathered and compiled by the Australian Institute of Marine Science (AIMS). For the primary outbreak data, AIMS uses the Long Term Reef Monitoring Program, which is a team of trained divers and researchers that takes surveys through underwater visual censuses and records corals and other benthic organisms (Wilkinson, 2004). Data collected this way captures the natural variability of coral and fish populations and documents the effects of disturbances like *A. planci,* cyclones, and bleaching events (Baker, Glynn, and Riegl, 2008). Sixty-four reefs were surveyed in 11 different sectors of the Great Barrier Reef, from Cape Grenville in the north to Capricorn Bunker in the south (see Figure 5). These areas are surveyed every year using the manta tow technique. Every three years, another 117 reefs are surveyed.

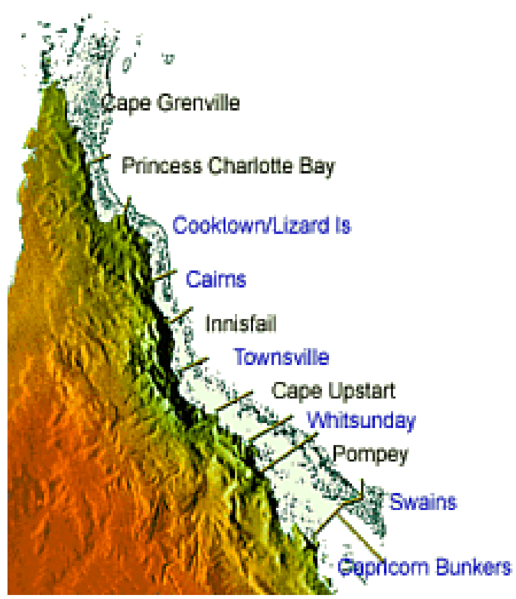


Figure 5. A map of the Great Barrier Reef region. Sectors labeled in blue have permanent survey sites. Those labeled in black are sampled less regularly (AIMS, 2009).

Observations began in January 1982 and continue today. There are two survey techniques commonly used: the manta tow and the free diving transect exploration. Using the manta tow technique, summarized in Figure 6, a person is towed several meters behind a boat for a short distance to observe and record marine resources. It is considered effective for community-based coral reef monitoring. The manta board is attached to a motorboat with several meters of rope, which has buoys placed at distances of six and 12 meters from the board. A snorkeler grips the board and is towed for approximately two minutes, at the end of which the boat pauses to allow the surveyor to record observed data. The coverage of bottom features may be recorded on a percentage scale or on a scale of one to five, where a score of five indicates the greatest cover and zero is used for absence. However, a rating system such as this is problematic because observers could place a disproportionately large number of values in the middle category, creating significant observer bias in the final reported data.

For each manta tow (that lasts two minutes), the number and size of *A. planci* was recorded, along with the percentage cover of live coral, bleached, dead coral, soft coral, sand and rubble. Weather conditions (wind strength, cloud cover, and tide) are noted. The composition of the reef and coral species present is documented (Porter et al., 2005). Broad-scale characterization of coral reefs often makes use of several methods including manta tows and free diving surveys. For the manta tow technique, the survey is conducted by being pulled behind a research vessel (Lucas, 1990). Percent cover of living hard and soft coral and dead hard coral is calculated from the manta tow results by representing each cover category by the mid-point of its range.

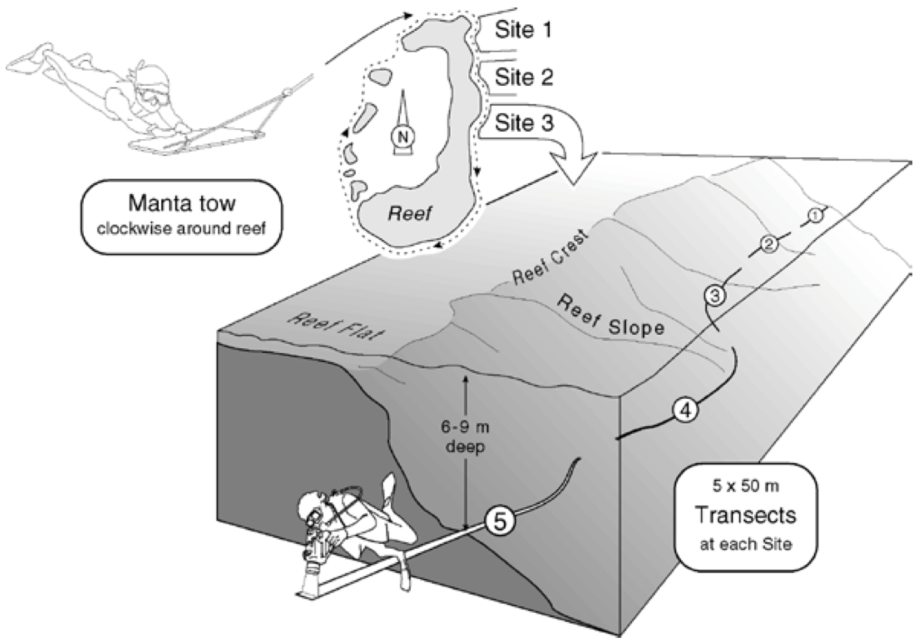


Figure 6. A schematic diagram of a manta tow survey on a coral reef (AIMS, 2009).

Coral cover, the number of *A. planci* per reef and the average number of *A. planci* per tow are used to assess the outbreak status of each reef. There are four categories: active outbreak (AO), incipient outbreak (IO), recovering (RE), or no recent outbreak (NO). An outbreak is referred to as incipient when the density of individuals on a reef is at the point that coral damage is likely. This occurs when there are 0.22 adults per manta tow or greater than 30 adults per hectare. Adults are considered to be at least 26 centimeters in diameter or greater than three years old. An outbreak is considered to be active when the density of adults is greater than one adult per manta tow, and adults are greater than 15 centimeters in diameter. Once *A. planci* densities exceed this threshold, the population will begin to consume coral faster than it can grow and is considered to be an outbreak population (Pratchett, 2010). The size of the individuals is important because it is an indication, in addition to population density, of how successful a population is on a reef. A reef is considered recovered from an outbreak when coral cover on the reef has reached or exceeded 80% of its previous maximum measured value (Done, 1988).

An alternative to the manta tow technique is a simple transect exploration. This method is similar to the manta tow, except that observers are not towed behind a boat. Free divers and snorkelers lay down plastic or metal squares and take survey of what organism lie within the one square meter area. This allows a degree of independence, which permits very shallow areas to be catalogued safely. Since the observer can get much closer to the substrate, it is possible to record a more detailed account of reef features such as coral and algae. Surveys are conducted along profiles, which tend to run either parallel or perpendicular to the reef. Surveys are directed using a compass bearing.

Survey teams of two to four divers are used for safety reasons and to delegate survey responsibilities. Transects and manta tow methods may use the same recording scales. Transects, denoted by quadrats laid down on the reef are used extensively for sampling in all branches of Ecology and many approaches are available. For coral reef assessment, quadrats usually have a minimum area of one square meter and are divided into a uniform grid of smaller segments. Data for sea surface temperature was recorded from research vessels and monitoring stations set at points around the entire region.

For this project, reefs were selected that had temperature data (also from AIMS) and a corresponding historical record of outbreaks. In some cases, this was just one data point per reef. At several points in the region there are clusters of smaller reefs (within 0.25 of a longitudinal degree) from a point of temperature monitoring. These were combined and used the temperature value of the nearest monitoring site. Once all these were obtained, they were recorded along with their most recent outbreak data point and the status of the reef at that time.

To test the hypothesis that sea surface temperature is correlated to instances of outbreaks, I used a 1-way ANOVA to compare the temperature of reefs in each of the different outbreak categories. Then, as an alternative way to explore the results, the data was recast so that the three categories at some stage of outbreak (active, incipient, and recovering) were all considered "affected reefs" and the reefs that were not impacted by *A. planci* were considered "unaffected reefs". A separate 1-way ANOVA was used to explore the data this way for each of the three time variations between outbreak and temperature data (none, delayed by one year, and delayed by two years).

Results

When there was no time delay, the highest average temperature was seen for the reefs in recovery, followed by the reefs with an incipient outbreak and those with an active outbreak. The lowest temperatures by far were those that had never had an outbreak. Results of the ANOVA test indicated that when all four categories were used, there was no significant correlation between temperature and outbreak status. (P=0.8905, df=3).

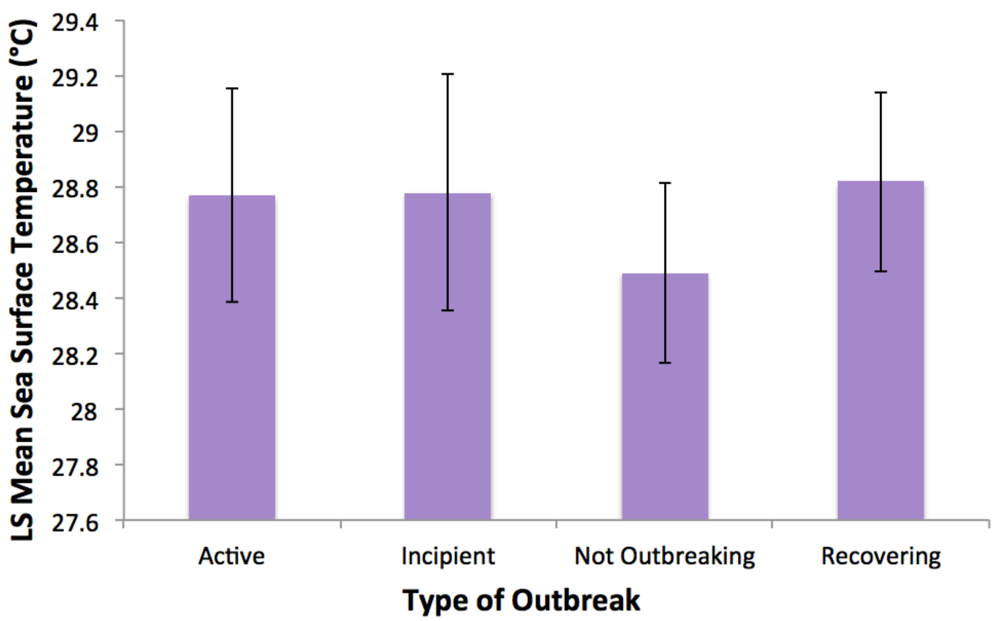
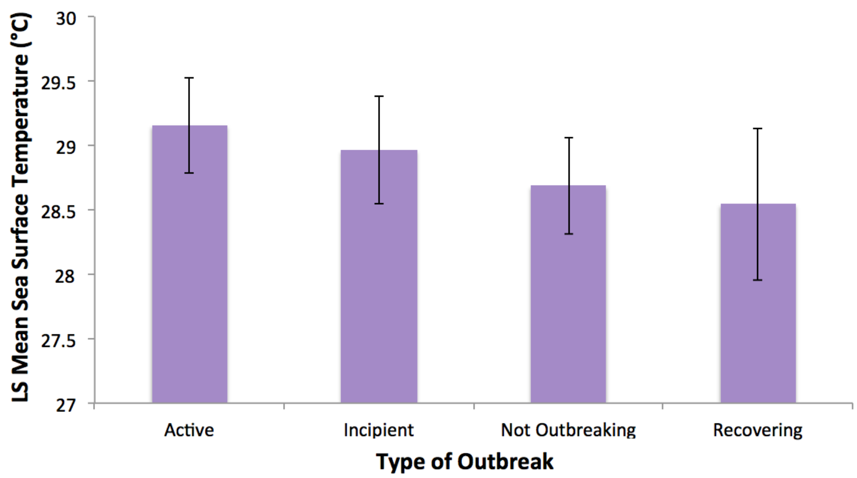


Figure 7. The mean temperature for the four classifications of outbreak status: Active, Incipient, None, and Recovering (n=3).

Due to the life history of *A. planci,* specifically the fact that they reach full maturity in two years, a temperature comparison was done for the same reefs at time points one and two years before the outbreak status was measured. Results of these comparisons can be seen in Figure 8. When using the four categories, for the data comparing outbreak status to the sea surface temperatures delayed by two years, there was no data for incipient outbreaks in these reefs. Again, using the four outbreak categories, there was no significant correlation between temperature and outbreak status for the data that was collected a year before the outbreak data (P=0.7582, df=3) or two years before the outbreak data (P=0.2225, df=3).

A)



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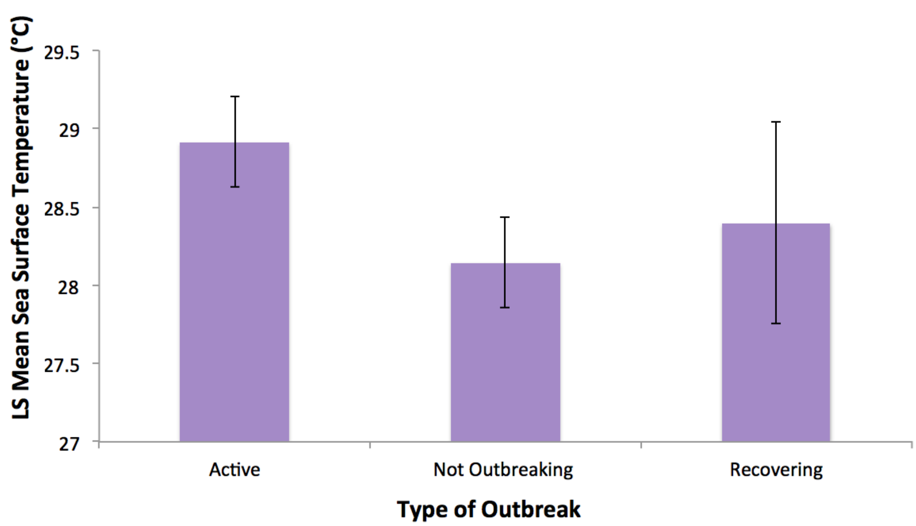


Figure 8. The mean temperature and reefs at the different stages of outbreak: A) one year and B) two years before the outbreak data was taken.

However, for the two-category analysis, there was a statistically significant correlation between sea surface temperature and outbreak status (P=0.009740, df=1). As shown in Figure 9, the affected reefs had a mean temperature of 28.8°C, while the unaffected reefs had a temperature of 28.5°C.

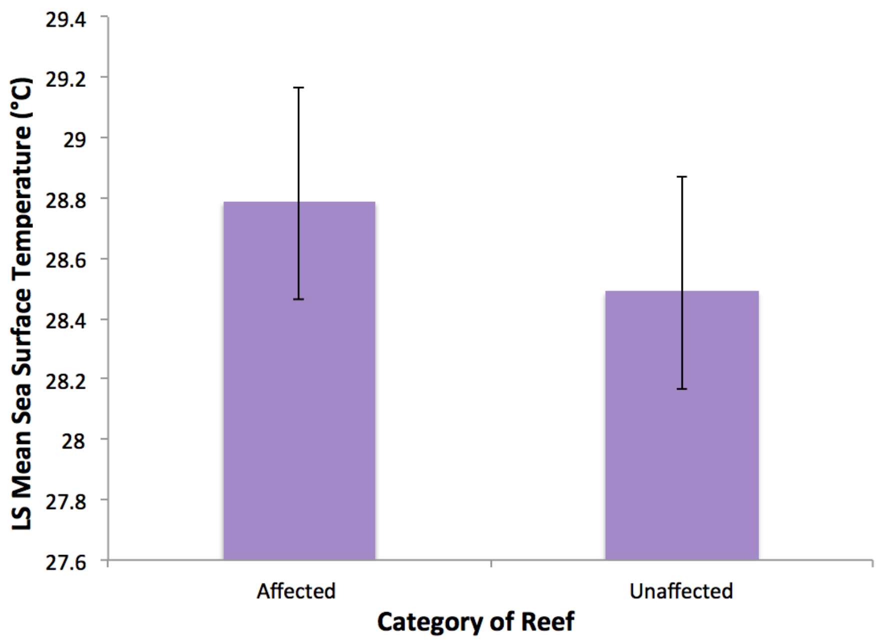
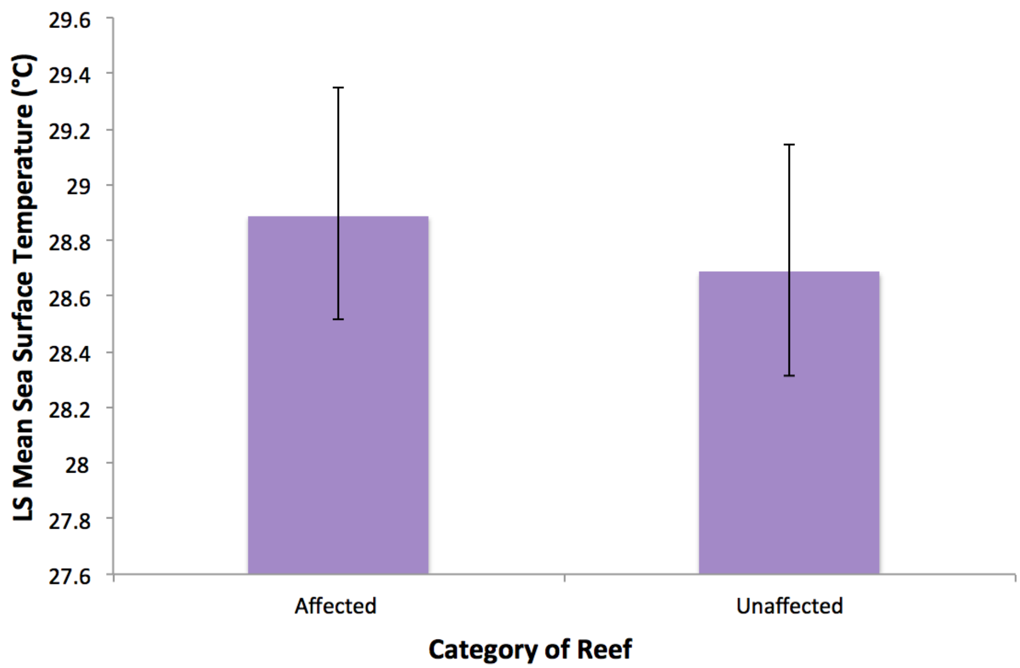


Figure 9. A comparison of affected (active, incipient, and recovering) reefs and unaffected reefs with no delay between collection of temperature data and *A. planci* outbreak surveys.

Even using the two-category approach, as can be seen in Figure 10, there is not a statistically significant relationship between affected and unaffected reefs delayed by one year (P=0.633668, df=1) and two years (P= 0.458363, df=1). This is likely because there is no real role that such a small variation in sea surface temperature makes in the development process of *A. planci* at the one and two year points after fertilization.

A)



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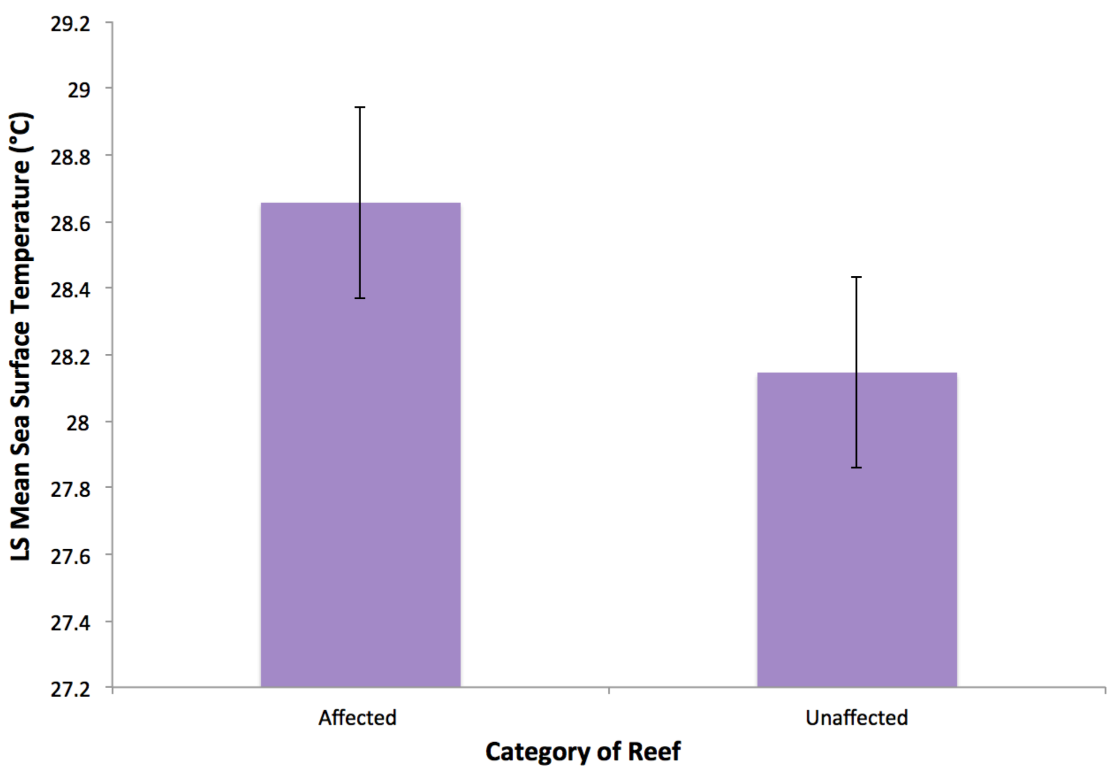

Figure 10. A comparison of affected (active, incipient, and recovering) reefs and unaffected reefs with A) one year and B) two years between collection of temperature data and *A. planci* outbreak surveys.

Discussion

The main purpose of this paper is to explore the idea that changes in sea surface temperature on the Great Barrier Reef have a direct impact on the frequency and locations of *A. planci* outbreaks in the region, and therefore the health of the subject reef systems.

Within the temperature range of 28 to 31°C, it is known that *A. planci* spawn with increased success. Therefore, I hypothesized that there would be more instances of *A. planci* outbreaks in the locations and times when temperatures fall within this range for *A. planci* reproduction. Additionally, due to the fact that it takes *A. planci* two years to reach full maturity, I thought it was possible that sea surface temperature during key stages of development (at one and two years following fertilization) could also be impacting outbreak patterns.

For the data taken the same year as the outbreak surveys, the highest average temperature was for the reefs in recovery. For those with a one and two-year lag, the highest average temperature was for those with an active outbreak. This could mean that, due to the development pattern of *A. planci,* a higher temperature during key development periods has a positive impact on the ability for populations on certain reefs to outbreak. The lowest average temperature for the same-year and the two-year lag was in the reefs with no outbreak. For the one-year lag, the lowest temperature was for reefs in recovery. Through my analysis of temperature data as it relates to outbreak status, I determined that there is no significant correlation between sea surface temperature and the instances of outbreaks of *A. planci* on the Great Barrier Reef, when looking at the four reported categories separately.

Due to this conclusion, I decided to explore the data in a different way. I considered the reefs with outbreak events (whether they be active, incipient, or recovering) all to be "affected reefs" and all reefs without an outbreak reported to be "unaffected". I thought this was a more meaningful representation of the data and provided a more statistically valuable result.

Regarding the data analysis looking at the two distinct categories (affected and unaffected reefs), my hypothesis was confirmed. There was a significant result when comparing the outbreak and temperature from the same year. At 28.5°C, the unaffected reefs were consistently 0.3°C colder than affected reefs which are of 28.8°C. This is slightly less than the average annual increase in temperature that has been measured. The peak temperature for *A. planci* metabolic functioning is 30°C (Yamaguchi, 1974). As the water gets warmer and approach this value, the populations are able to function more efficiently and reproduce more effectively, and therefore take over a reef more quickly. A 0.3°C difference appears to be enough to allow that shift.

Other studies support my hypothesis, and have also found that there is a correlation between sea surface temperature and the prevalence of *A. planci* outbreaks. In a lab experiment, Yamaguchi (1974) found that *A. planci* functions ideally when sea surface temperatures are between 28°C and 31°C, and trouble with oxygen uptake when the water temperature around them is raised above 31°C. Glynn (1975) found that temperatures above a certain level (that varies by species) cause stress to corals and their symbiotic zooxanthellae. Eventually the zooxanthellae are lost and the coral is bleached, and the colony subsequently recovers or dies. Death of a coral settlement would mean the absence of a food source for *A. planci.* Incidents such as these have been known to occur during El Niño events.

A major issue with this study was the inconsistency in or complete lack of data for certain regions. To combat this I would first spend time organizing a more thorough survey of the Great Barrier Reef regarding sea surface temperature and *A. planci* populations. Along with these surveys, research could examine the role of salinity levels, nutrient runoff patterns (via rainfall levels), and predator populations as they relate to outbreaks of *A. planci.* Maybe then we would have a more complete picture of why outbreaks have occurred and how they can be curtailed in the future. Another issue with this study was the very small sample size. More reefs with data in this regard could impact the results and provide a different outcome, or at least make them more reliable. Possibly, the sea surface temperature has an impact on the presence of a phytoplankton bloom, which provides food for the sea stars and induces an outbreak. Also, it could be related to rainfall. Nutrient runoff from humans could be washed into the ocean and provide food for phytoplankton, leading to a bloom and then an outbreak.

My findings indicate a dire future for the Great Barrier Reef. As it stands now, the average sea surface temperature in the warmer months (October through March) is 27 °C and in the cooler months (April through September) is 24 °C (GBRMPA, 2013). This means, given the historical increase in water temperature of 0.4 °C per year, that within 3 years, summertime temperatures will reach the optimal range for *A. planci* outbreak acceleration. By 2030, temperatures throughout the entire Great Barrier Reef will be ideal for *A. planci* propagation, and the Great Barrier Reef will suffer significantly, potentially being completely overrun by these menacing creatures.

To fight this seemingly inevitable development, there are some measures that can be taken. One is to develop an effective poison that can be input into the water system and attacks only the *A. planci* in the area. Another could be the addition of more individuals of the key predators of *A. planci* into the region. There are many possible ways to combat these disastrous consequences, but something needs to be done, and soon, before the entire Great Barrier Reef is gone forever.

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