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A STUDY OF CORALLIVORES OF BAWE AND CHUMBE ISLANDS, ZANZIBAR

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ABSTRACT

Coral reefs are considered keystone ecosystems due to their socioeconomic, ecological, and educational value. In Zanzibar, Tanzania, reefs provide income and a protein source for large portions of the population. Fish are reliant upon reefs as grounds for feeding, breeding, and nursing. Reefs are related to mangrove and seagrass ecosystems; the three ecosystems are interconnected in their functions of protecting biodiversity of coastal organisms, coastal erosion, and improving water quality. Coral reefs are also subject to degradation by both anthropogenic and natural causes. Predation is one example of natural degradation of coral. Six species of corallivores were studied to compare their population dynamics on Bawe and Chumbe Reefs, Zanzibar. Transect and

quadrat analyses were performed in order to determine the relative density of each corallivore species. The Line Intercept Transect Method was employed in order to determine substrate distribution. Data was analyzed using 21 transects (1,260 m^2) inside the MPA and 14 transects (840 m^2) outside of the MPA. Triggerfish had a significantly higher mean outside the Chumbe MA compared to both Bawe and within the MPA. Butterflyfish had a significantly higher observed mean within the MPA compared to outside the MPA, which had a higher mean than Bawe. There was a positive correlation between the percent of live coral and number of butterflyfish. There was a significantly higher percentage of live coral cover at the Chumbe Island reef than Bawe Island. There was a significantly higher number of both *Drupella* and *Coralliophila* snails at Bawe Island than inside the Chumbe MPA. This suggests that the benefits obtained from the implementation of the MPA and removal of COTS from the Chumbe Island Reef may have aided in the increased live coral cover, but has not significantly impacted the corallivore fish species which reside there. This study contributes to the limited information of corallivore densities, especially in the Zanzibar archipelago and will provide a baseline for future studies on coral predation.

INTRODUCTION

A coral polyp is the small, fleshy protrusions from the coral colony. Polyps are linked to form coral colonies by a common gastrovascular system (Muller-Parker and D'Elia, 1997). Coral is a symbiotic relationship between zooxanthellae and calcareous algae. The zoozanthellae receives protection from the coral, and works inside three layers of tissue (Figure 1). Inside the tissue layers is a nutrient rich environment in which the zooxanthellae produces photosynthetic products for its host; however, the water surrounding the polyp is comparatively very nutrient poor. Reef systems have equal levels of photosynthesis and respiration, which is known as autotrophism (Viles and Spencer, 1995).

Corals reproduce both sexually and asexually. Sexual reproduction must occur for a new settlement of coral to occur, and settlement must occur on stable rocky surfaces. Coral growth continues by asexual reproduction, which means that colonies of coral are clones of the same polyp (Viles and Spencer, 1995).

Coral reefs are a significant resource in Tanzania as they comprise two-thirds of the country's coastline (Wagner, 2004). Over 150 species of coral have been recorded within Tanzania reefs (Muhando, 2000). Coral reefs in Tanzania are being degraded by anthropogenic and natural factors (Bergman and Öhman, 2001; Mohammed et al., 1999) the most serious being the bleaching event in 1998 which caused coral mortality ranging from 10-90% (Mohammed et al., 1999). This bleaching event has lead to increased monitoring of coral reefs in Tanzania and the surrounding islands. The islands Pemba and Unguja comprise the country of Zanzibar, which is located east of mainland of Tanzania

(Figure 3). These two large islands and other small islands in the archipelago are surrounded by fringing reefs (Bergman and Öhman, 2001).

Coral reefs are considered keystone ecosystems due to their socioeconomic, ecological, and educational value (Muhando, 2000). Coral reefs are regarded as keystone ecosystems because they provide ecological services, which extend beyond their surface area (Wagner, 2004).

Socioeconomic Values

The ocean and coral reefs provide income for much of the Zanzibar population, along with historical and religious sites, which are significant to the local culture. Tanzanian coastal fisheries are sustained by coral reefs as over 30% of fish are harvested on or adjacent to the coral reef environment (Muhando, 2000). In the past decade, Zanzibar has also seen a spike in tourism; the ocean and coral reefs are also a major tourist attraction (Wagner, 2004).

Artisanal fishing has provided an economic foundation for most countries in the Southwest Indian Ocean Region (Mapunda, 1983), especially island nations. Most of the boats used are small, and are unable to reach the Exclusive Economic Zone (EEZ) (Figure 2) which extends over 200km offshore (Jiddawi, 2010). There are currently 12 boats in Zanzibar licensed to fish in the EEZ, which contains highly valued larger migratory fish (Jiddawi and Ohman, 2002).

There are many stakeholders within the fishing sector in the Zanzibar economy. Fishing is linked to boatbuilding, fuel and ice suppliers, tourism, hotels, and directly to fishmongers and fishermen (Jiddawi, 2010). The ocean directly supplies 25% of the population of Zanzibar with employment (Jiddawi, 2010), and 2.2-10.4% of the national

GDP of Zanzibar is derived from marine fisheries (Jiddawi and Ohman, 2002). Fish

supplies 90% of the animal protein for the Zanzibar population (Wagner, 2004); the

average person consumes 17kg of fish a year. Due to growing population, and an increase

in demand for a cheap protein, there has been an increase in fishermen, and as a result, an

overexploitation of artisanal fishing (Mapunda, 1983; Jiddawi, 2010).

Seaweed farming is also another form of income for many Zanzibaris, and is as

another form of income can reduce the fishing pressure on coral reefs. Farming of wild

seaweed has occurred since the 1930s, and commercial production began in the early

1990s. Seaweed farming takes place in the intertidal zone and currently 90% of all

seaweed farmers in Tanzania are female (McClanahan, 2000). This is attributable to the

fact that seaweed farming is considered a "safe" profession that occurs close to shore

(Jiddawi, 2010). Seaweed farmers often pull up natural seagrasses to plant seaweed.

Seagrasses stabilize sediments of the intertidal zone, which is one negative side effect of

seaweed farming (McClanahan, 2000). Without seagrasses, sediment is able to drift into

reefs, decreasing the amount of sunlight available for photosynthesis, and making the

reefs less productive. Seaweed farming also lowers beneficial bacteria production and

decreases diversity and abundance of other intertidal organisms (Wagner, 2004).

Tanzania, especially Zanzibar, has seen a spike in tourism since the country was

opened to a free market system in 1984 after being a socialist nation for about 20 years.

Tourism has stimulated the local economy, decreased some pressure on jobs in the

fishing industry, and led to an increase in the standard of living. However, tourism has

also caused an overuse and misuse of natural resources (Bergman and Öhman, 2001;

Howell, 2010). Tourism has caused a spike in construction of hotels in both urban and

rural areas with beach access. This congestion, and the lack of sewage treatment can cause pollution and increased sedimentation rates by soil and beach erosion. The ocean and coral reefs are major tourist attractions, increased boat traffic and poor handling of the local flora and fauna can destruct the local ecosystem (Howell, 2010).

Ecological Values

Extensive research has been performed on coal reefs and their links to adjacent habitats. Coral reefs are linked to mangrove and seagrass bed habitats in such a way that one cannot be studied without considering the others (McClanahan, 2000).

Mangroves and seagrass beds are the ecosystems in which many marine species feed, breed, and nurse. Seagrasses are an important source of food for many gastropods, bivalves, polychaetes, and vertebrates, including the coral-eating parrotfish (McClanahan, 2000). Both of these ecosystems also protect the shore from coastal erosion. Roots hold sediment and sand in place, instead of washing it to sea (McClanahan, 2000). Mangroves are also a natural sewage filter; improving water quality before chemical or contaminants reach the vulnerable coral reefs. The reefs in turn protect the mangroves and seagrass beds from harsh wave action, reducing the force on their way towards shore. This is another way of protecting the shore from erosion (*A Guide to the Wise Use and Protection of Our Resources*, 2005).

Educational Values

Coral reefs are a great example of "demonstrating biological and ecological complexity to students" (Muhando, 2000). Coral reefs are a useful vector for teaching students about the ecology of different ecosystems. There is also extensive research done within coral reefs due to the immense biodiversity located in and around them; but also

because this biodiversity is being threatened by many different sources. Biodiversity can generally be defined as a variety and variability among living organisms and the ecological complexes in which they occur (Howell, 2010). Biodiversity is important because ecosystems with high biodiversity are able to support a variety of species. An ecosystem is described as biodiverse if it has: complex topography, efficient biological recycling, high retention of nutrients, and a variety of behavioral patterns (Muhando, 2000).

Coral Degradation

Coral reefs are subject to many forms of degradation, both anthropogenic and natural. Anthropogenic factors include destructive fishing techniques, overexploitation, and pollution (including eutrophication and increased sediment in terrestrial runoff). Natural causes include natural warming and cooling cycles, coral bleaching, and predation. It is difficult to detach these issues that cause coral degradation and categorize them as solely anthropogenic or natural as most issues are connected and one issue may lead to another.

Subsistence fishing is a common practice in the Zanzibar Archipelago and has been for centuries. However, some traditional gears used are considered destructive to coral reefs including: explosives, *kigumi (dragnets), juya* (beach seines), and poisons (McClanahan, 2000; Muhando, 2000). Dynamite has been used in Tanzania for over 4 decades (Jiddawi and Ohman, 2002); each blast instantly kills all fish, other living organisms, and destroys reef habitat (Wagner, 2004). Widespread use of explosives occurred during the 1980's and 90's (Wagner, 2004); the practice is now illegal in Zanzibar (Jiddawi and Ohman, 2002). *Juya* or beach seines are large nets that are dragged

across the coral surface, which is harmful both to coral and other organisms that attach themselves to the coral substrate (McClanahan, 2000). Dragnets, or *kigumi*, are similar to beach seines as they also abrade the coral surface with chains used to weight the net. Fishermen are also known to hit coral heads with sticks in order to scare fish from hiding (Wagner, 2004). Dragnets are a legal fishing gear (Jiddawi and Ohman, 2002). Poison derived from the *Euphorbia* plant kills fish instantaneously and is the most commonly used poison (McClanahan, 2000). Poisons indiscriminately kill adult fish along with fish larvae and juveniles (Jiddawi and Ohman, 2002), which has a major effect on fish recruitment.

In general, overexploitation can have major effects on coral health. Economic growth tends to be slower than population growth, which may lead to employment issues in the fishing sector such as: too many fishermen and not enough fish and catching immature fish which leads to issues in recruitment. The reef is also directly impacted by human traffic on the coral at low tide while harvesting octopus, mollusks, and shells (McClanahan, 2000). Coral polyps are damaged or killed by trampling and coral heads are broken in order to obtain invertebrates from the substrate.

Agricultural malpractice can also result in soil destabilization and increased sediment in terrestrial runoff (McClanahan, 2000). The term agricultural malpractice describes many actions that lead to a toss of topsoil, which is transmitted to the ocean. These actions include: forest clear-cutting for agrarian areas, establishment of poor irrigation systems, slash and burn and deep tillage agriculture (Muller-Parker and D'Elia, 1997). This increase in sediment can lead to increased turbidity of coastal waters, which

leads to decreased light available for photosynthesis. This can lead to decreased productivity of both reefs (Muhando, 2000) and seagrass beds.

Chemical and sewage pollution can lead to eutrophication of coastal habitats. Corals survive best in a low-nutrient environment; therefore eutrophication has the ability to destroy fringing reefs (McClanahan, 2000). Zanzibar currently does not have wastewater treatment, which is a major threat to corals near densely populated areas like Stonetown (Strömberg, 2000). Untreated sewage contains high levels of nutrients; which is emptied directly into the ocean. Chemical fertilizers contain high concentrations of nutrients, and if they are not absorbed by plants or soil, are transported in irrigation runoff to the ocean (Muller-Parker and D'Elia, 1997).

Coral bleaching occurs when coral becomes stressed by sedimentation, pollution, disease, but most commonly an increase in water temperature or UV radiation (McClanahan, 2000). Coral bleaching is visible as a white skeleton, as the zooxanthellae have been expelled or voluntarily have left their protective coral host. The zooxanthellae give the coral color and photosynthetic products (McClanahan, 2000; Wilkinson, 1998). As their source of food leaves, the coral polyp often dies with the absence of their endosymbiotic algae. El Nino Southern Oscillation (ENSO) is a natural global phenomenon, which refers to cycles in wind systems and ocean currents, which direct cool and warm ocean waters. The famous 1998 increase in sea-surface temperatures caused bleaching events around the world (McClanahan, 2000; Wilkinson, 1998). In the Indian Ocean, coral bleaching was greater in intensity than all previous bleaching reports from the region (McClanahan, 2000).

Interestingly, coral cover on reefs surrounding Chumbe Island decreased from

51.9% to 27.5% and on Bawe Island there was a slight increase from 53% in 1997 to 57.7% between 1997 and 1999 (Mohammed et al., 1999). These values are lower than most areas in the Southwest Indian Ocean as Chumbe and Bawe reefs are in shallower waters and experienced less stress due to night cooling patterns (Mohammed et al., 1999). However, 80-95% of *Acropora* species bleached on Chumbe, which is much higher than other sites (Wilkinson, 1998).

Coral Predation

Corals have many natural predators including several species of fish, snails, sponges, and sea urchins (Nelson, 2007). Comprehensively, 114 species of vertebrates (including 11 families of osteichthyan fish) at least occasionally consume live coral (Rotjan and Lewis, 2008). Corallivores can be can be classified into three categories of feeding strategies: browsers, scrapers and excavators. Excavators remove live coral tissue along with large chunks of the underlying skeleton, scrapers remove only a small portion of the skeleton while feeing upon live coral, and browsers remove live coral polyps, algae, and other detritus without damaging the coral (Francini-Filho et al., 2008; Rotjan and Lewis, 2008). The corals most commonly grazed coral species include: *Acropora*, *Pocillopora*, *Montipora,* and *Porites*. Corallivores have been reported to only prey upon 18 of the 111 described coral genera (Rotjan and Lewis, 2008).

Acanthaster planci, or the Crown-of-Thorns-Starfish (COTS), preys upon coral by expelling its' stomach onto the coral. The COTS will cover as much coral as possible with it's stomach and secrete a digestive enzyme which melts coral waxes, the main energy stores of the coral polyp. Once digestion is complete, the COTS absorbs the digested material, and retracts its stomach, leaving a signature white "feeding scar"

(Nelson, 2007). COTS are considered browsers as they feed on coral polyps without damaging the coral skeleton (Rotjan and Lewis, 2008). COTS densities are usually around 5-20 individuals per km^2 (McClanahan, 2000), which are sustainable levels for reef health. COTS were previously associated with preserving coral diversity as they preferentially consume *Acropora* coral species. *Acorpora* is a fast-growing coral (Rotjan and Lewis, 2008) and if preyed upon, it may allow for establishment of slower-growing corals, increasing the biodiversity of the reef.

COTS "outbreaks" (when populations reach the above stated sustainable density), can occur as COTS are extremely fertile; Adult COTS release millions of eggs developing to larvae, larvae develop into juveniles, and juveniles mature at 18-24 months and begin feeding on live coral (McClanahan, 2000). Due to the efficiency of COTS feeding mechanism, "outbreaks" can cause both localized and widespread coral polyp tissue loss. COTS are prone to outbreaks due to many compounding factors including: the COTS large size, flexibility, high fecundity, and life-history characteristics (McClanahan, 2000). Removal of natural predators such as the giant triton (*Charonia tritonis*), starry pufferfish (*Arothron hispidus*), two species of triggerfish (*Balistoides viridescens,* and *Pseudoblasistes flavimarginatus*), and humphead Maori wrasse (*Cheilinus undulates*) may also contribute to COTS outbreaks (Nelson, 2007).

In recent years, COTS outbreaks have become more widespread, and are lasting for longer periods of time (Muhando and Lanshammar, 2008). In Zanzibar, the Institute of Marine Science is monitoring the impact of COTS by researching long-term coral cover. There are several permanent transects in use within the Zanzibar archipelago including the following islands: Chumbe, Misali, and Bawe. In some areas in Zanzibar,

COTS removal programs are in place as a method to avoid a severe decrease in live coral cover. A continuous removal program began at Chumbe Island Coral Park Ltd. (CHICOP) in 2004 (Muhando and Lanshammar, 2008), and a one-time removal occurred within Misali Island Conservation Area (Condos, 2006).

Triggerfish (*Balistidae*) are facultative corallivores; the species have other food sources besides coral. Triggerfish also feed on benthic algae, detritus, small mollusks, and crustaceans. Triggerfish have the ability to remove skeletal tissue as certain species are scrapers and others are excavators. Rotjan and Lewis (2008) found that triggerfish prey upon coral at a rate of 1.87 bites/min specializing on *Pocillopora* and *Pavona* coral species. Though triggerfish are scrapers or excavators, because they are facultative corallivores, they have less of an impact on coral health and growth rates as only a portion of their diet consists of coral polyps. Triggerfish are considered a keystone predator of sea urchins. However, because the species is more aggressive and territorial, it is more likely to enter baited traps (McClanahan, 2000). Overexploitation triggerfish leads to a predation release of grazing sea urchins, which will drastically increase sea urchin population. On heavily fished reefs, this will cause sea urchin populations to colonize coral colonies and erect algae. Algae is able to recover from predation, but coral is not, leading to a decrease in biodiversity.

Most butterflyfish (*Chaetodontidae*) are omnivorous (53 species), but 14 species are obligate corallivores (Rotjan, and Lewis, 2008). Those that eat coral are browsers, which remove individual coral polyps, without damage the coral skeleton (Rotjan, and Lewis, 2008). This suggests that butterflyfish have a lesser impact on coral health and growth rates. Butterflyfish are obligate corallivores and are therefore considered to be an

indicator species of coral reef health as fish are motile and emigrate from deteriorating coral reefs as reefs degrade. One species of butterflyfish, *Chaetodon trifascialis* has coevolved with it's preferred prey, *Acropora* coral. The butterflyfish is a specialist, which lives as a solitary individual in permanent territories, which it defends against other intruding butterflyfish and other corallivores. A decrease in the *Chaetodon trifascialis* population is associated with a decrease in *Acropora* coral cover and health (Reese, 1981). This also suggests that a decrease in the *Chaetodon trifascialis* population suggests a decrease in biodiversity of the reef.

Parrotfishes (*Scaridae*) are omnivorous fish that use their fused beak-like jaws to consume algae, detritus, and live coral (Francini-Filho et al., 2008). Parrotfish are either scrapers or excavators, and can produce large amounts of sediment on reefs, especially when their population densities are high. A study in Panama found that a population of *Scarus iserti* with a density of 1fish per m^2 generated 0.5 kg $CaCO_3$ m^2 per year (Glynn, year). The effects of degradation by the parrotfish's predation is a growing concern in the fish's contribution to coral fitness and health (Addison and Tindal, 1990; Francini-Filho et al., 2008). However, it should be noted that few species of parrotfish have high rates of erosion. The composition of detritus, coral, or $CaCO_3$ skeleton ingested differs depending on fish age, feeding strategy, and species. Only one of 18 scarid species in the Great Barrier Reef, three of ten species in the Red Sea, and one of six species in the Caribbean have what are considered high rates of erosion (Glynn, year).

The two most common corallivore snails are *Coralliophila* species and *Drupella* species. *Drupella* is motile and prefers to consume *Acropora, Montipora, Seriatopora,* and *Pocillopora* species of coral (McClanahan, 2000; Rotjan and Lewis, 2008).

Aggregations of *Drupella* are responsible for a 35% decrease in live coral cover over 10

years in Japan and a 86% decrease in live coral cover in Australia over a ten-year period

(Rotjan and Lewis, 2008). It is not known definitively what causes *Drupella*

aggregations, but studies have produced many hypotheses. One study suggests that

Drupella aggregations occur following coral mortality events like bleaching and

hurricanes. This suggests *Drupella* aggregations are associated with dying corals, which

prompt the aggregations to feed. (Baird,1999). Another study in Masoala found that high

aggregations are found in areas where predatory fish are overexploited (McClanahan,

2000).

In contrast, *Coralliophila* is sedentary, and prefers crevices in massive *Porites*.

Their effect on coral is considered to be low (McClanahan, 2000), but it is hypothesized

that they may drain energy resources, by extracting photosynthetic products produced by

zooxanthellae (Beeden et al., 2008). Similarly, corallivory by *Coralliophila* is known to

contribute to phase shifts from coral to algal dominated reefs (Rotjan and Lewis, 2008).

Management Strategies

In order to decrease the threat of overexploitation and destructive fishing methods,

local and national governments have passed legislation in order to preserve reefs and

preserve marine biodiversity (Muhando, 2000). Marine Protected Areas (MPAs) like

Chumbe Island Coral Park (CHICOP), and Mafia Island Marine Park have zoned small

no-take areas. Conservation Areas like the Pemba Channel Conservation Area (PECCA),

which includes Misali Island Conservation Area (MICA) limit or forbid the use of certain

fishing techniques. PECCA is governed by 34 fishing communities, which establish rules

for the times when gear use is permitted (*A Guide to the Wise Use and Protection of Our*

Resources, 2005).

Data on benthic and fish populations was collected on Bawe and surrounding islands in 1989, and collected again in 1991. Included in the study was Mnemba Island, which was sold to a private entity, and now includes a 200m exclusion zone. Over the two-year period there was a decrease in population density and number of species found at unprotected sites, and an increase at Mnemba Island (McClanahan, 2000). There is much evidence that fish populations are higher in MPAs than in heavily fished areas. It has also been proven that the "spillover effect" increases fish catches adjacent to MPAs through immigration of adult fish from MPAs, but can also be attributed to the spillover of fish eggs and larvae (McClanahan, 2000).

A study performed in Kenya on the relationship between fishing intensity and corallivore snail population size found that over an eight-year period ending in 1995, snail populations increased in all reefs surveyed. However, the greatest increases were in reefs with heavy fishing and the Marine Protected Areas surveyed had the lowest abundance of snails (McClanahan, 2000).

Corallivores play a part in the natural degradation of coral reefs. This study will provide a baseline study comparing the corallivore ecology at two reefs. A comparison of two reefs, unprotected (Bawe) and protected by an MPA (Chumbe), will allow the comparison of the relative intensity of corallivore predation with varying fishing pressure and human use. This study may provide information on the influence of the removal of a major predator, the COTS, from the reef ecosystem, and the effect on other coral predators.

STUDY AREA

The coastal waters of Tanzania range from 4°S-10°S and consist of mainly fringing and patch reefs. The Zanzibar archipelago, less than 50m off the coast of Tanzania contains two large islands, Pemba and Unguja, but also includes many smaller islands, which lie just offshore the larger two. Bawe Island is two nautical miles east and Chumbe Island is seven nautical miles south west of Stonetown, the largest city on Unguja (Figure 4) (Strömberg, 2000).

Chumbe Island Coral Park Ltd. (CHICOP) closed the fringing reef west of Chumbe Island in 1992, and the reef became a marine protected area (MPA) in 1994. The MPA is about 1,300m long and 300m wide (0.39km^2), and marked with buoys. Fishing within the MPA is strictly banned and monitored by rangers, however, the east side of Chumbe Island and areas surrounding the MPA are open to fishing. An ecotourism project on the island houses seven "eco-bungalows" for tourists, which feature a solar-powered lighting and heated water, rainwater catchment system, composting toilets, and a greywater filtration system. The philosophy at Chumbe is to interact with the environment in a way that does not compromise the natural resources. One of the qualifications of an eco-tourism project is an education program that teaches tourists, students, and the local community about local ecosystems and the benefits of ecotourism.

There is also a private resort on Bawe Island. Tourists are transported to the island frequently from Stonetown, by boat, which takes 30 minutes to an hour depending on weather. Bawe Island does not advertize sustainable use of natural resources. For example, unlike Chumbe, tourists are not taught about the benefits of the coral reef

ecosystem and are more likely to remove organisms from the substrate, or damage coral by stepping on it.

METHODOLOGY

Data was collected from Bawe Island over the dates of October 9-13th and Chumbe Island over the dates of November 5-24th (Figure 4). Data was obtained by snorkeling and skin diving along transects set out on both Bawe and Chumbe fringing reefs. Data was collected on a dive slate, and transferred to computer at the end of collection. The exact location of each transect was identified by GPS (Table 1 and 2), and transferred to a map (Figure 5) at the end of each data collection

Predator Assessment

Ten 30m by two meter belt transects (Figure 6) were determined on reefs surrounding Bawe Island and 35 30m by two meter belt transects were determined on reefs surrounding Chumbe Island. Of the 35 transects, 21 were within the MPA, and 14 were outside the MPA. This was done using a 30m tape measure, and transects were chosen randomly. First, the tape was laid on the coral substrate. Next, fish families known to prey upon corals were counted if they were observed within one meter of either side of the tape. These families included: the parrotfish, butterflyfish, and triggerfish. The density of total predator species, and relative density of each species were determined at the culmination of the study. Crown of Thorn Starfish were also assessed within these transects when present.

Quadrat analyses were used to determine the density of corallivore snails. A half meter by half meter or 0.25m^2 quadrat was placed along the transect every five meters, alternating sides (Figure 6).

Substrate Assessment

The substrate was determined using the Line Intercept Transect method. The substrate observed for each meter of the 30m transect was recorded. The substrate labels were determined before the transect was surveyed. Live coral describes coral colonies where the majority of the coral was live coral polyps. Dead Coral describes coral which intact, yet was reduced to its' skeleton and had been covered with encrusting or coralline algae. Bleached coral also falls within the category of dead coral. Rubble describes broken coral and other detritus. Sand describes sandy substrate with minimal detritus or coral growth.

RESULTS

Predator Assessment

At the Bawe reef, triggerfish make up 5% of the corallivore population, with an average of 0.8 fish/m^2, and a density of 0.007 fish/m^2. At Chumbe Island, triggerfish make up 8% of the corallivore population, with an average of 1.9 fish/m^2 and a density of 0.016 fish/m^2. The average triggerfish observed at Chumbe is significantly higher (t=7.8, df=75, p<0.05) than the average observed at Bawe Island. Within the Chumbe MPA, triggerfish make up 6% of the corallivore population, with an average of 1.4 fish/m^2, and a density of 0.012 fish/m^2 (Figure 7). Outside the MPA at Chumbe Island, triggerfish make up 12% of the corallivore population, with an average of 2.7 fish/m^2 and a density

of 0.022 fish/m^2 (Figure 8). There was significant effect of the site (F=46.1, df=2,476, p<0.0001) in the number of triggerfish observed, such that a significantly higher number of triggerfish were observed outside the Chumbe MPA compared to both Bawe (q=4.1) and inside the Chumbe MPA (q=4.2). Triggerfish observed at Bawe and inside the Chumbe MPA did not vary significantly (q=1.4).

At the Bawe reef, butterflyfish make up 34% of the corallivore population, with an average of 5.6 fish/m^2, and a density of 0.04 fish/m^2. At Chumbe Island, butterflyfish make up 50% of the corallivore population, with an average of 12.0 fish/m^2, and a density of 0.10 fish/m^2. The average number of butterflyfish observed at Chumbe is significantly higher (t=7.9, df=475, p<0.05) than the average observed at Bawe Island. Within the Chumbe MPA, butterflyfish make up 52% of the corallivore population, with an average of 12.8 fish/m^2, and a density of 0.10 fish/m^2. Outside the MPA at Chumbe Island, triggerfish make up 46% of the corallivore population, with an average of 10.7 fish/m^2 and a density of 0.09 fish/m^2 (Figure 8). There was significant effect of the site (F=46.1, df=2,476, p<0.0001) in the number of butterflyfish observed, such that a significantly higher number of butterflyfish were observed within the Chumbe MPA than outside the MPA (q=5.6). Similarly, a significantly higher number of butterflyfish were observed outside the Chumbe MPA than at Bawe (q=8.89).

Figure 11 depicts the correlation between the average percent live coral cover per transect (60m^2), and the corresponding number of butterflyfish found in that transect.
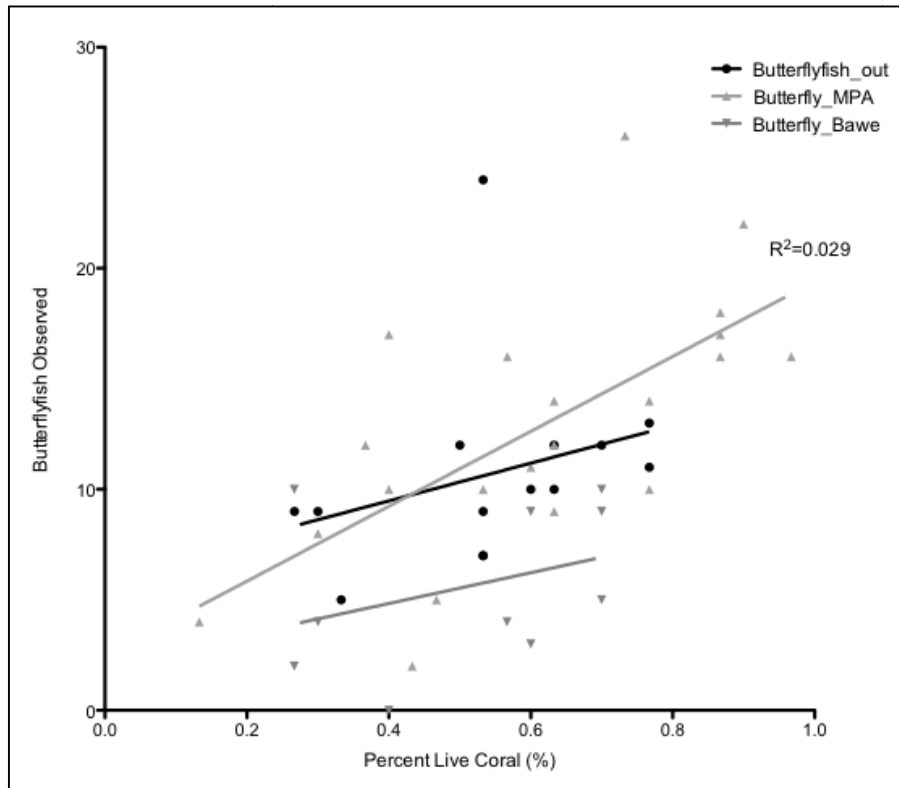
Figure 11. Relationship Between Butterflyfish Observed and Percent Live Coral Substrate. Butterflyfish observed increased significantly at a rate of $y=17.0\pm4.5x+2.5\pm2.9$ with an increase in percent coral cover (F=14.2, df=(1,19), p=0.0013, R^2=0.029) within the Chumbe MPA. At Both the Bawe reef (F=1.1, df=(1,8), p=0.32, R^2=0.12) and outside the Chumbe MPA (F=1.22, df=(1,12), p=0.29, R^2=0.092), butterflyfish tended to increase with an increase in percent live coral cover, but the relationship is not significant.

At the Bawe reef, parrotfish make up 61% of the corallivore population, with an average of 10.2 fish/m^2, and a density of 0.085 fish/m^2. At Chumbe Island, parrotfish make up 42% of the corallivore population, with an average of 10.2 fish/m^2 and a density of 0.085 fish/m^2. The average number of parrotfish observed at Chumbe is not significantly different (t=0.06, df=460, p<0.05) than the average observed at Bawe Island. Within the Chumbe MPA, parrotfish make up 42% of the corallivore population, with an average of 10.3 fish/m^2, and a density of 0.086 fish/m^2. Outside the MPA at Chumbe Island, parrotfish make up 43% of the corallivore population, with an average of

10.1 fish/m^2 and a density of 0.083 fish/m^2 (Figure 8). There was not a significant effect

of the site (F=0.0795, df=2,457, p=0.9236) in the number of parrotfish observed at any of

the sites.

      Figure 12 and figure 13 depict the correlation between the average percent

"degraded substrate" and live coral cover, respectively, per transect and the

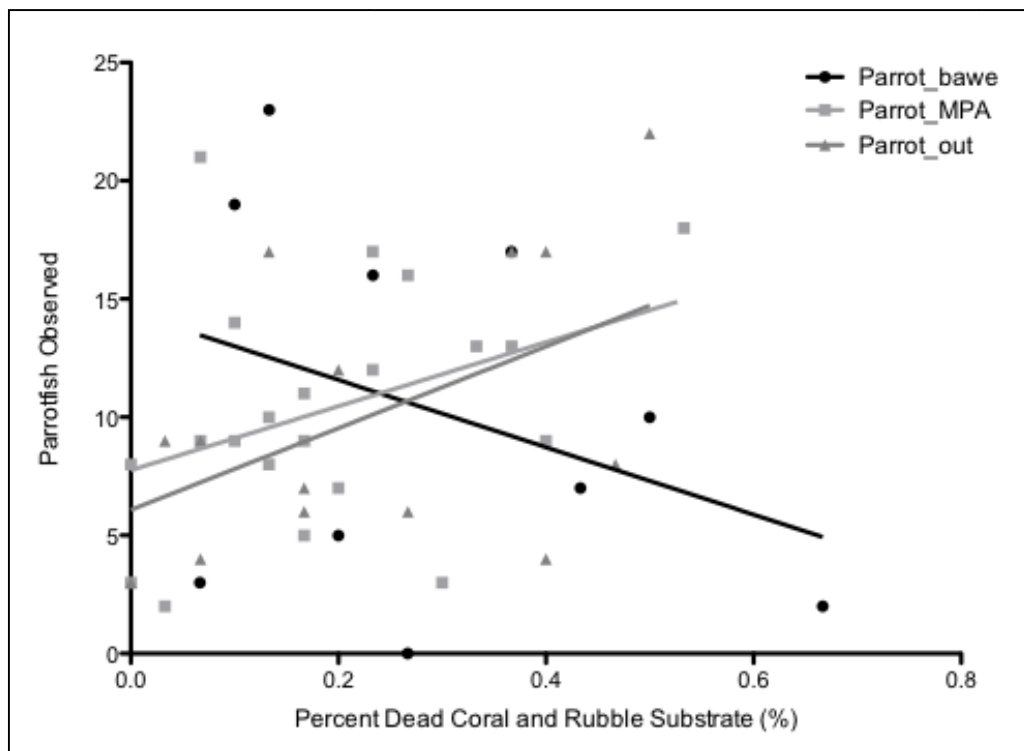corresponding number of parrotfish found in that transect.

Figure 12. Relationship Between Parrotfish Observed and Degraded Substrate. Within the
Chumbe MPA (F=3.0, df=(1,19), p=0.10, R^2=0.137) and outside the Chumbe MPA
(F=3.76, df=(1,12), p=0.08, R^2=0.23) there tended to be an increase in parrotfish
observed with an increase in degraded substrate, but the relationship was not significant.
At Bawe reef (F=1.1, df=(1,8), p=0.33, R^2=0.12), there was a decrease in parrotfish with
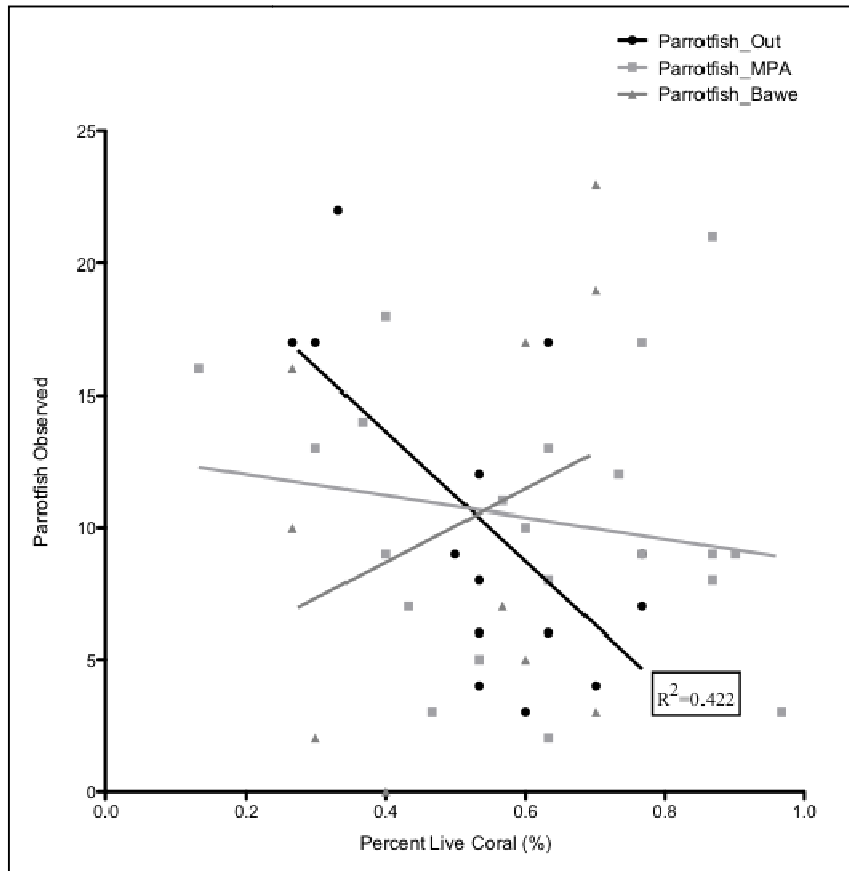an increase in degraded substrate, but the relationship was not significant.

Figure 12. Relationship Between Parrotfish Observed and Live Coral Cover.
The number of parrotfish outside the Chumbe MPA decreased significantly (F=8.75, df=(1,14), p=0.012, R^2=0.422) at a rate of y=-24.48 ± 8.27x+23.42 ± 4.682 with an increase in live coral cover. Parrotfish within the MPA decreased (F=0.63, df=(1,21), p=0.438, R^2=0.03) and parrotfish at Bawe increased (F=0.88, df=(1,10), p=0.375, R^2=0.099) with an increase in live coral cover, but the relationship was not significant.

At the Bawe reef, *Drupella* species make up 63% of the snail corallivore population, with an average of 5.9 snails/m^2, and a density of 1.5 snails/m^2. At Chumbe Island, *Drupella* species make up 41% of the corallivore population, with an average of 1.6 snails/m^2 and a density of 0.44 snails/m^2. Within the Chumbe MPA, *Drupella* species make up 36% of the corallivore population, with an average of 1.1 fish/m^2, and a density of 0.28 snails/m^2. Outside the MPA at Chumbe Island, *Drupella* species make up 46% of the corallivore population, with an average of 3.2 snails/m^2 and a density of 0.68 snails/m^2. There was significant effect of the site (F=182.7, df=2,177, p<0.0001) in the

number of *Drupella* snails observed per m^2, such that a significantly higher number of *Drupella* snails were observed at the Bawe reef than outside the MPA (q=24.45). Similarly, a significantly higher number of butterflyfish were observed outside the Chumbe MPA than within the Chumbe MPA (q=7.67).

At the Bawe reef, *Corallophila* species make up 37% of the snail corallivore population, with an average of 3.4 snails/m^2, and a density of 0.85 snails/m^2. At Chumbe Island, *Corallophila* species make up 59% of the corallivore population, with an average of 2.4 snails/m^2 and a density of 0.62 snails/m^2. Within the Chumbe MPA, *Corallophila* species make up 64% of the snail corallivore population, with an average of 2.0 snails/m^2, and a density of 0.50 snails/m^2. Outside the MPA at Chumbe Island, *Corallophila* species make up 54% of the snail corallivore population, with an average of 2.7 snails/m^2 and a density of 0.80 snails/m^2. There was significant effect of the site (F=4.05, df=2,178, p<0.0001) in the number of *Corallophila* snails observed per m^2, such that a significantly higher number of *Corallophila* snails were observed at the Bawe reef compared within the Chumbe MPA (q=3.612). *Corallophila* snails observed at Bawe and outside the Chumbe MPA did not vary significantly (q=0.549), nor did outside the MPA to inside the MPA (q=3.295).

Substrate Assessment

At Bawe reef, substrate distribution was observed for each transect (n=297): a total of 55 (18%) points were identified as sand, 35 (12%) points were identified as rubble, 54 (18%) points were identified as dead coral, and 153 (51%) points were identified as live coral (Figure 9).

At the Chumbe reef, substrate distribution was observed for each transect (n=1050): a total of 219 (21%) points were identified as sand, 64 (6%) points were identified as rubble, 153 (15%) points were identified as dead coral, and 614 (58%) points were identified as live coral (Figure 9). Within the Chumbe MPA (n=633) a total of 128 (20%) points were identified as sand, 30 (5%) points were identified as rubble, 90 (14%) points were identified as dead coral, and 385 (61%) points were identified as live coral. Outside of the Chumbe MPA (n=417) a total of 91 (22%) points were identified as sand, 34 (8%) points were identified as rubble, 63 (15%) points were identified as dead coral, and 229 (55%) points were identified as live coral.

A Chi-square analysis determined that there a significant difference exists in the average amount of rubble, dead coral, and live coral observed at Bawe, within the Chumbe MPA, and outside the Chumbe MPA (χ^2=19.3, df=4, p=0.0007). Rubble and dead coral were combined to "degraded substrate"; a Chi-square analysis showed that there is still a significant difference between "degraded substrate" and live coral. A t-test of the % live coral cover between the Bawe and Chumbe Reefs showed that Chume has a significantly higher percent of live coral (t=15.62, df=34, p<0.0001).

DISCUSSION

Predator Assessment

Interestingly, no COTS were observed within the transects at Bawe Island. One COTS was observed, however, it was not observed within a transect. Previous research on COTS at various islands in the Zanzibar archipelago recorded COTS within transects at Bawe, at a density of 0.06 per m^2 (Mohammed et al., 1999). There are not, nor in the

past have there been, any removal programs in place at Bawe Island. Possible explanation

for this population decrease are: natural population fluctuations, the fact that the previous

study occurred immediately following the mass bleaching even in 1998. There may have

been an unnaturally high density of COTS because a reproductive event caused the

population to mature at the same time. Since, the population may have died and failed to

have their progeny reproduce on the Bawe reef. The mass-bleaching event of 1998 was

caused by natural ENSO cycles characterized by an increase in Sea Surface Temperature

(Wilkinson, 1998). COTS spawn when water is highest (Nelson, 2007), so a natural or

induced spawning could have occurred after an increase in sea water temperature in 1998,

causing high densities of COTS in some areas in 1999. As no COTS were observed

within transects, and only one was observed outside the study area, it is safe assume that

the COTS population on the Bawe reef is currently sustainable.

There were also no COTS observed at the Chumbe Island reef. One COTS was

observed on the reef, but it was not observed within a transect. Another COTS was

observed after it had been removed from the reef by a Chumbe ranger. There is a COTS

removal program at Chumbe Island which was instated in 2004 (Muhando and

Lanshammar, 2008). From the establishment of the removal program until 2008 over

3,000 COTS have been removed from the MPA (Muhando and Lanshammar, 2008), and

are continuously being removed from the MPA.

At the Bawe reef, triggerfish make up 5% of the corallivore population, with an

average of 0.8 fish/m^2, and a density of 0.007 fish/m^2. At Chumbe Island, triggerfish

make up 8% of the corallivore population, with an average of 1.9 fish/m^2 and a density of

0.016 fish/m^2 (Figure 7, Table 3.1). A t-test determined the average triggerfish observed

at Chumbe is significantly higher (t=7.8, df=75, p<0.05) than the average observed at Bawe Island. Bawe Island reef has higher fishing pressure than Chumbe Island. As triggerfish tend to be aggressive and territorial they are more likely to enter baited traps (McClanahan, 2000). Triggerfish are sometimes used as an indicator species for overfishing on reefs because they have this nature (McClanahan, 2000). The average number of triggerfish observed at Chumbe may be higher because there is less fishing pressure in the MPA, providing an area where triggerfish are less likely to enter dema traps.

Within the Chumbe MPA, triggerfish make up 6% of the corallivore population, with an average of 1.4 fish/m^2, and a density of 0.012 fish/m^2. Outside the MPA at Chumbe Island, triggerfish make up 12% of the corallivore population, with an average of 2.7 fish/m^2 and a density of 0.022 fish/m^2 (Figure 8, Table 3.1). A One Way ANOVA test determined there was significant effect of the site (F=6.7, df=2,74, p=0.002) in the number of triggerfish observed, such that a Tukey's Multiple Comparison Test determined that a significantly higher number of triggerfish were observed outside the Chumbe MPA compared to both Bawe (q=4.1) and inside the Chumbe MPA (q=4.2). Triggerfish observed at Bawe and inside the Chumbe MPA did not vary significantly (q=1.4).

It is unexpected that on average, more triggerfish are observed outside the Chumbe Island MPA than inside the MPA. Research has proved that the "spillover effect" of an MPA leads to an increase in fish catches in surrounding areas (McClanahan, 2000). It could be possible that some of the triggerfish are being forced out of the MPA due to competition by triggerfish or other fish species which occupy the same niche.

Additionally, three dema traps were qualitatively observed juxtaposed to three of the transects surveyed outside of the MPA. One of the traps contained a triggerfish, and the rest were empty. It is perplexing that with obvious fishing pressure, the triggerfish population is still higher outside the Chumbe MPA. A follow-up study may be warranted in order to investigate the cause of lower mean triggerfish observed within the MPA.

Triggerfish seem to have the least amount of potential impact on coral reef regeneration as a study has shown that they are facultative corallivores, which means only part of their diet is coral, supported by the fact their gut contains less than 1% coral (Rotjan and Lewis, 2008). The fact that triggerfish don't prey upon coral exclusively may be the reason why they were observed comparatively less within the transects which were on exclusively coral reef habitat (within the Chumbe MPA). However, a healthy triggerfish population is a vital resource to the reef ecosystem by regulating the sea urchin population.

At the Bawe reef, butterflyfish make up 34% of the corallivore population, with an average of 5.6 fish/m^2, and a density of 0.04 fish/m^2. At Chumbe Island, butterflyfish make up 50% of the corallivore population, with an average of 12.0 fish/m^2, and a density of 0.10 fish/m^2 (Figure 7, Table 3.2). A t-test determined the average number of butterflyfish observed at Chumbe is significantly higher (t=7.9, df=475, p<0.05) than the average observed at Bawe Island. Butterflyfish are known to be indicator species of coral reef health as fish are motile and emigrate from deteriorating coral reefs as reefs degrade. The majority of butterflyfish species are associated with *Acropora* species of coral. Though no quantitative data was collected on the percent cover of *Acropora* species we can assume that as there are more butterflyfish at Chumbe Island, the reef has a higher

percent of live coral cover and better overall coral health.

Within the Chumbe MPA, butterflyfish make up 52% of the corallivore population, with an average of 12.8 fish/m^2, and a density of 0.10 fish/m^2. Outside the MPA at Chumbe Island, triggerfish make up 46% of the corallivore population, with an average of 10.7 fish/m^2 and a density of 0.09 fish/m^2 (Figure 8, Table 3.2). A One Way ANOVA determined there was a significant effect of the site (F=46.1, df=2,476, p<0.0001) in the number of butterflyfish observed, such that a Tukey's Multiple Comparison test determined that a significantly higher number of butterflyfish were observed within the Chumbe MPA than outside the MPA (q=5.6). Similarly, a significantly higher number of butterflyfish were observed outside the Chumbe MPA than at Bawe (q=8.89). Similar to the comparison the Bawe and Chumbe Reefs, though no quantitative data was collected on the percent cover of *Acropora* species we can assume that as there are more butterflyfish within the Chumbe MPA, the reef has a higher percent of live coral cover and better overall coral health.

Figure 11 depicts the correlation between the average percent live coral cover per transect, and the corresponding number of butterflyfish found in that transect. This positive correlation (F=14.2, df=(1,19), p=0.0013, R^2=0.029) supports the hypothesis that butterflyfish are associated with live healthy coral. At both the Bawe reef and outside the Chumbe MPA, the correlation is also positive, but not significant.

Butterflyfish tend to be territorial; therefore polyps are removed in high densities within these territories. This may deplete colony resources, decrease growth, and reproductive rates. However, some studies have shown that frequent removal of polyps by obligate butterflyfish corallivores does not negatively impact their host (Rotjan and

Lewis, 2008). However, because butterflyfish are browsers and only remove individual coral polyps they have less impact on coral regeneration than parrotfish as the polyps only need to focus on asexual reproduction of polyps instead of both sexual and asexual reproduction of coral. Oren et al. (1997), found that single-polyp, linear, and small ($\leq 2cm^2$) tissue-only lesions have best rates of regeneration (Rotjan and Lewis, 2008). Corallivory by large populations of butterflyfish are not considered to be a threat to coral cover degradation or coral heath because their feeding strategy causes little harm to coral. Figure 11 supports the theory that large butterfly populations are not a threat to coral, as large populations are associated with a higher percentage of live coral.

At the Bawe reef, parrotfish make up 61% of the corallivore population, with an average of 10.2 fish/m^2, and a density of 0.085 fish/m^2. At Chumbe Island, parrotfish make up 42% of the corallivore population, with an average of 10.2 fish/m^2 and a density of 0.085 fish/m^2 (Figure 7, Table 3.3). A t-test determined the average number of parrotfish observed at Chumbe is not significantly different (t=0.06, df=460, p<0.05) than the average observed at Bawe Island. The relatively high relative percent of parrotfish observed at the Bawe reef compared to the relative percent observed at Chumbe is most likely attributed to the significantly larger population of butterflyfish observed at the Chumbe reef than the Bawe reef.

Qualitative observation found that the vast majority of parrotfish on the Bawe reef were juveniles (Observation made by fish color) and on Chumbe there seemed to be a more even distribution of age. Along with an increase in size, parrotfish tend to become brilliantly colored with age, so this distinction between reefs was easily made. Parrotfish are large fish with marketable value in the fisheries industry. Therefore, we would expect

the average parrotfish observed to be higher at Chumbe as the mature fish are being

caught at the Bawe reef. Fewer mature parrotfish at Bawe may lead to an issue with

recruitment if fishing continues to occur, especially at an unsustainable level. The

difference in size and age of fish observed at the two reefs may affect the rate at which

coral is consumed as juvenile fish consume less coral and more algae when feeding

(Bruggemann et al., 1996). However, juvenile parrotfish tend to live and feed in schools,

which may cause localized pressure for colonies where they preside.

Within the Chumbe MPA, parrotfish make up 42% of the corallivore population,

with an average of 10.3 fish/m^2, and a density of 0.086 fish/m^2. Outside the MPA at

Chumbe Island, parrotfish make up 43% of the corallivore population, with an average of

10.1 fish/m^2 and a density of 0.083 fish/m^2 (Figure 8, Table 3.3). A One Way ANOVA

test determined there was not a significant effect of the site (F=0.0795, df=2,457,

p=0.9236) in the number of parrotfish observed at any of the sites. For all sites sampled,

the average parrotfish observed and parrotfish density is not significantly different. This

could potentially have an impact on the reef at Bawe if the parrotfish population is

feeding at unsustainable levels on a reef with lower live coral cover.

Parrotfish have greater potential to decrease growth rates owing to their feeding

strategy. While feeding, parrotfish also preferentially remove polyps with reproductive

potential, which indirectly causes neighboring polyps to devote energy to regeneration

instead of reproduction (Rotjan and Lewis, 2008). Studies have shown that large

populations of parrotfish negatively impact coral fitness and health (Rotjan and Lewis,

2008) and are instrumental in breaking down reefs as a bioerosion contributor

(Bruggemann et al., 1996; Glynn, year). However, not all species of parrotfish are

obligate corallivores; most species that consume coral are omnivores that also consume algae and other detritus. Studies have proven that a reduction in herbivorous fish can lead to a phase shift from coral dominated reefs to algal dominated reefs (Rotjan and Lewis, 2008; Miller and Hay, 1998). This proves that some parrotfish may be helping decrease the risk of a phase shift if they are consuming algae.In order to determine if parrotfish are associated with a certain substrate, a regression analysis was employed.

Figure 12 depicts the correlation between the average percent "degraded substrate" per transect and the corresponding number of parrotfish found in that transect. Though the trends in Figure 12 are not statistically significant, this interaction suggests that at the Chumbe reef, the parrotfish may be omnivorous and preying primarily on dead coral and the algae covering it. Similarly, Figure 13 depicts the correlation between the average percent live coral cover per transect and the corresponding number of parrotfish found in that transect. Figure 13 confirms that especially outside the MPA, parrotfish are associated with degraded substrate due to the significantly negative correlation between parrotfish and live coral cover. Similarly, Figure 13 confirms that parrotfish at the Bawe reef are associated with consuming live coral as the parrotfish population has a negative correlation with "degraded substrate" and positive correlation with live coral cover.

At the Bawe reef, *Drupella* and *Corallophila* species make up 63% and 37% of the snail corallivore population, with an average of 5.9 and 3.4 snails/m^2, and a density of 1.5 and 0.85 snails/m^2. At Chumbe Island, *Drupella* species make up 41% of the corallivore population, with an average of 1.6 snails/m^2 and a density of 0.44 fish/m^2 (Table 3.4 and Table 3.5) A One Way ANOVA determine there was significant effect of the site in the number of both *Drupella* snails (F=182.7, df=2,177, p<0.0001) and

Corallophila snails (F=4.05, df=2,178, p<0.0001) observed per m^2, such that a significantly higher number of *Drupella* snails were observed at the Bawe reef than outside the MPA (q=24.45). Similarly, a significantly higher number of *Drupella* snails were observed outside the Chumbe MPA than within the Chumbe MPA (q=7.67), however, *Corallophila* snails observed at Bawe and outside the Chumbe MPA did not vary significantly (q=0.549).

Both of these species affect coral species differently. *Drupella* species are more destructive than *Coralliophila* species as they are motile, and consume coral as they move. It could also be that the large population of *Drupella* species is caused by the lower percentage of live coral. *Drupella* snails tend to be associated with coral mortality events or dying corals. Fishing also tends to lead to an increase in *Drupella* snails as fishing of predatory fish leads to a release in snail predatory pressure. Therefore, it is not surprising that the Bawe reef has the largest *Drupella* population. It is concerning that the highest densities are most prevalent on the Bawe reef, as the snails are able to lead to degradation on reef and Bawe already has a lower percent coral cover than the reef at Chumbe Island.

Carrying capacities for *Drupella* and *Coralliophila* are unknown, so it is unknown if the densities found on the reefs are sustainable for coral growth. However, we do know that invertebrate grazing scars, or those which are circular or square, have the lowest recovery rate (~16% regenerated tissue over 90 days) (Rotjan and Lewis, 2008). Therefore, we would want to have the lowest populations of corallivorous snails possible to ensure regeneration of coral polyps. The reef within the Chumbe MPA has the most sustainable levels of snail corallivores followed by the reef outside the Chumbe MPA.

A study mentioned earlier also reported that *Drupella* was found consuming a branching species of *Porites* when in heavily fished reefs, but consuming *Acropora* within MPA sites. As *Acropora* is uncommon on heavily fished reefs, this suggests that *Drupella* is a coral generalist and is capable of switching feeding preference to other branching species (McClanahan, 2000). Though this study did not document the species of coral within the quadrats, unofficial observation discovered that the density of *Acropora* was lower at the Bawe Island reef, and higher at the Chumbe Island reef. When available, *Drupella* would prey upon it's preferred species, *Acropora*. However, other branching species were more prevalent than *Acropora*, and *Drupella* was often found preying upon these species especially in areas where *Acropora* was unhealthy (bleached or dying). We cannot assume that the density of *Acropora* was low due to predation by *Drupella* as *Acropora* is a preferred prey species of COTS and some species of vertebrate corallivores.

We also know that *Acropora* cover has been declining at Bawe Reef since about 1999, just after the 1998 coral bleaching event (Muhando and Lanshammar, 2008). Unlike other areas in the region, the reef at Chumbe Island has a higher *Acropora* cover than other reefs in the region. This is often attributed to the COTS removal program. Also, as the MPA allows fish to mature, predatory fish may keep the populations of corallivorous snails low. Snails have the potential to cause widespread damage, especially to *Acropora* species of coral. Since mature fish are removed from the ecosystem at the Bawe reef, a release in fishing pressure may allow populations of *Drupella* snail to significantly decrease the *Acropora* cover especially after a mortality event like the 1998 bleaching event.

Coralliophila snails are sessile obligate corallivores, and are often found in high concentrations after abiotic stressors like hurricanes (Rotjan and Lewis, 2008). These snails are associated with the living margin of corals; consuming polyps using enzymes and not damaging the coral skeleton. This corallivore tends to be a significant predator after abiotic stressors, and like *Drupella* is limited in population size by fish predation. On the reefs surveyed, Bawe had a significantly higher mean *Coralliophila* abundance when compared to inside the Chumbe MPA. The means of *Coralliophila* are more similar to each other than the means of *Drupella* observed on the reefs.

Though there is no stated value for an unhealthy density of invertebrate corallivores, we may want to consider the effect that removal of mollusk corallivores has on the local ecosystem. A study by Miller found that removal of *Coralliophila abbreviata* led to significantly more live *Acropora palmate*, than at colonies where snails were not removed. Colonies with intact *Coralliophila* lost tissue at a rate of 3cm^2 per day. Should there be an abiotic stressor like a hurricane, a removal program may want to be considered in order to assist in the regeneration of live coral cover.

In a healthy coral reef situation, the amount of coal consumed by corallivores seems unlikely to negatively affect coral reef ecosystems (Rotjan and Lewis, 2008). A study in the Caribbean suggests that parrotfish contribute to facilitating coral recruitment, growth, and fecundity and should not be actively removed from reef habitats (Mumby, 2009). However, in unequal proportions, these corallivores have the potential to decrease growth rates of coral. It may be possible that on Chumbe Reef where parrotfish are subject to decreased fishing pressure, they may have a larger affect on coral recruitment, growth, and fecundity. Currently, conservation models stress only the negative affects of

the COTS; it is possible that the model needs to consider corallivore vertebrates, especially parrotfish, as they have the potential to affect coral growth more than other corallivores species. In the future, carrying capacity of corallivores species to be researched in order to have a better picture of reef conservation and regeneration (Rotjan and Lewis, 2008).

Substrate Assessment

At Bawe reef, substrate distribution was observed for each transect (n=297): a total of 55 (18%) points were identified as sand, 35 (12%) points were identified as rubble, 54 (18%) points were identified as dead coral, and 153 (51%) points were identified as live coral (Figure 9). It is important to note that Bawe reef is a popular reef for both commercial fishing and tourism. On one day of data collection, 31 individuals were counted walking on coral in the shallow lagoon. Upon speaking with our boatman, we found that they were fishermen collecting fish and octopus by spear and many other mollusks and crustaceans which inhabit the intertidal zone. A diving company boat was observed on a different data collection day preparing for a dive near the Bawe reef.

At the Chumbe reef, substrate distribution was observed for each transect (n=1050): a total of 219 (21%) points were identified as sand, 64 (6%) points were identified as rubble, 153 (15%) points were identified as dead coral, and 614 (58%) points were identified as live coral (Figure 9). The dynamics of fishing are interesting at Chumbe Island. As seen in Figure 5, the MPA only includes the western side of the island. Outside the MPA, fishing boats were observed every day working on all other reefs surrounding the island. Fishermen and women were observed on many days during low tide collecting invertebrates along a sandbar to the north. In this area the fishing

technique *kigumi*,was also observed several times; divers will scare fish into a net with sticks. Dema traps were also set next to three of the transects surveyed on the south end of the island, which may have significantly affected results. There is obviously heavy fishing pressure on the reefs and seagrass beds surrounding the Chumbe Island MPA.

Within the Chumbe MPA (n=633) a total of 128 (20%) points were identified as sand, 30 (5%) points were identified as rubble, 90 (14%) points were identified as dead coral, and 385 (61%) points were identified as live coral (Figure 10). Outside of the Chumbe MPA (n=417) a total of 91 (22%) points were identified as sand, 34 (8%) points were identified as rubble, 63 (15%) points were identified as dead coral, and 229 (55%) points were identified as live coral (Figure 10). Figure 10 shows that inside the Chumbe MPA, there is a slightly higher percent of live coral than outside the Chumbe MPA, which has a slightly higher percent of live coral cover than the Bawe reef.

A Chi-square analysis determined that a significant difference exists in the average amount of rubble, dead coral, and live coral observed at Bawe, within the Chumbe MPA, and outside the Chumbe MPA (χ^2=19.3, df=4, p=0.0007). Rubble and dead coral are combined to be classified as "degraded" substrate (Nelson, 2007); a Chi-square analysis showed that there is still a significant difference between "degraded substrate" and live coral (χ^2=13.94, df=2, p=0.0009).

Research has been performed previously on the coral cover of the Bawe Reef. The reef had about 60% total coral cover in the 1990s, but has since decreased to between 40-50% cover. *Acropora* was affected significantly by the 1998 bleaching event and has decreased to 1-2% cover at present (Muhando and Lanshammar, 2008). The live coral

cover (51%) observed in this study is lower than it has been in the past, but is consistent with current research for coral cover post coral bleaching and COTS outbreaks.

In the 1990's Chumbe had 50-60% live coral cover, similar to Bawe. Chumbe suffered more than most reefs in the Zanzibar archipellago from the 1998 bleaching event, with significant decreases in all species of live coral cover. However, since the bleaching event, live coral cover has increased to around 65%, with a significantly higher percent of live *Acropora* cover (Muhando and Lanshammar, 2008). It should be noted that the MPA at Chumbe Island was instated in 1994, and the increase in live *Acropora* has been attributed to the MPA and COTS removal program (Muhando and Lanshammar, 2008; Muhando, 2000). The live coral cover (58% for Chumbe as a whole and 61% within the MPA) observed in this study is a bit lower than the most reported numbers from Chumbe. However, in conversing with staff and rangers at Chumbe, there has been a recent bleaching event in April 2010, which may have caused the percent live coral to be a bit lower than the expected.

Possible Sources of Error

There are many possible sources of human error that may have occurred. There was a learning curve in identification of fish and snail species which may have skewed data significantly. There is a species of crab which looks very similar to the Drupella snail, which may have caused higher levels of *Drupella* to be reported.

Transects were placed in areas where a significant portion of live reef was seen in order to be certain that territorial fish (triggerfish/butterflyfish) would be included in the fish counts. Transects were observed each day so that one was perpendicular to shore and another was parallel. This was done in order to eliminate bias in coral cover and fish

counts based on the depth and reef topography. Transect methods chosen may have also skewed results.

RECOMMENDATIONS

In order to improve the methodology of this study, the species of coral should be determined. For a repeat study, this would increase the amount of discussion that could occur about results from this study in comparison to previous data collected by other researchers. The Line Intercept Transect method was useful, but could have more viable results if I could compare the species of coral found within the transects. For example, instead of making an unofficial statement about the species of coral that *Drupella* was associated with, I could provide a quantitative measure of the species of coral. Qualitative results would also be helpful when comparing the percent coral cover of the study areas together. *Acropora* is a very sensitive species of coral, and qualitative results would provide insight into the corallivores that prey upon it.

Also, in order to determine the rates of bioerosion on each reef and the effect that each fish family has on coral health, species identification of each fish family needs to occur. The intensity of feeding depends on feeding strategy and whether the fish species is a generalist or specialist, the only way an accurate prediction of the effect each fish family has on coral health is by determining the effect of each species present in the reef according to it's species population size and feeding strategy. A study similar to that conducted by Bruggemann et al. (1996) would be beneficial in determining the rates of bioerosion for each corallivore genus.

This study could be improved if there were, in general, more transects surveyed in order to increase the validity of my data. There were much fewer transects surveyed at the Bawe reef than the Chumbe reef. More transects at the Bawe reef would increase the validity of the data there. Also, from personal observation there is more tourism on the east side of Bawe Island, which is the area I surveyed. It would also increase the validity of my data to survey the west coast reef, as that side may see more fishing pressure which may affect the substrate due to fisherman trampling coral, and a possible decrease in reef fish. If more transects were surveyed it may be possible to split Bawe into two sites as was done in this study with Chumbe Island. This comparison would allow for interesting comparisons about the effect of fishing and MPAs on corallivore populations.

It would also be beneficial to include other reefs in this survey. For example, many studies have mentioned have mentioned that the Bawe reef is situated is impacted less from the sewage of Stowntown than the Changuu and Chapwani reefs (Wagner, 2004; Mohammed et al., 1999). Adding one of these two sites would allow the researcher to infer the impact of Zanzibar sewage on corallivory.

This study found a larger triggerfish population outside the Chumbe MPA than inside the MPA, which was unexpected due to the nature of triggerfish to enter dema traps. A follow-up study may be warranted in order to investigate the cause of lower mean triggerfish observed within the MPA.

Chumbe Island Coral Park (CHICOP) has an education program that is funded by eco-tourism to the island. Every year about 350 primary and secondary students and 50 teachers are brought to the island to learn about coral reef, intertidal, and coral rag ecosystems, waste management, biodiversity, and eco-tourism. This is the largest

environmental education program in the region, but yet there are many schools and students who do no have the opportunity to attend due to lack of funds. I would suggest that this program be expanded with financial assistance by the Department of Education in order to include all schools, regardless of financial situation, increased access to Chumbe Island fieldtrips. If environmental education of the local natural resources was mandated less anthropogenic damage to coral reefs would occur.

CHICOP also works with the Department of Fisheries to improve the education of local fishermen. Each year there are meetings with the local community to discuss the Chumbe reef's protected status and other reefs around Zanzibar. This aspect of the education program can be expanded to educate local fishermen about the effects of overfishing and destructive fishing methods. In recent years the concept of Integrated Coastal Management (ICM) has been implemented with success in East Africa and Tanzania. ICM involves all stakeholders, in planning and on-going management. The implementation of ICM along with a fortified adult-education program would allow local stakeholders to make educated policies regarding the fisheries and coral reefs of Zanzibar.

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APPENDIX
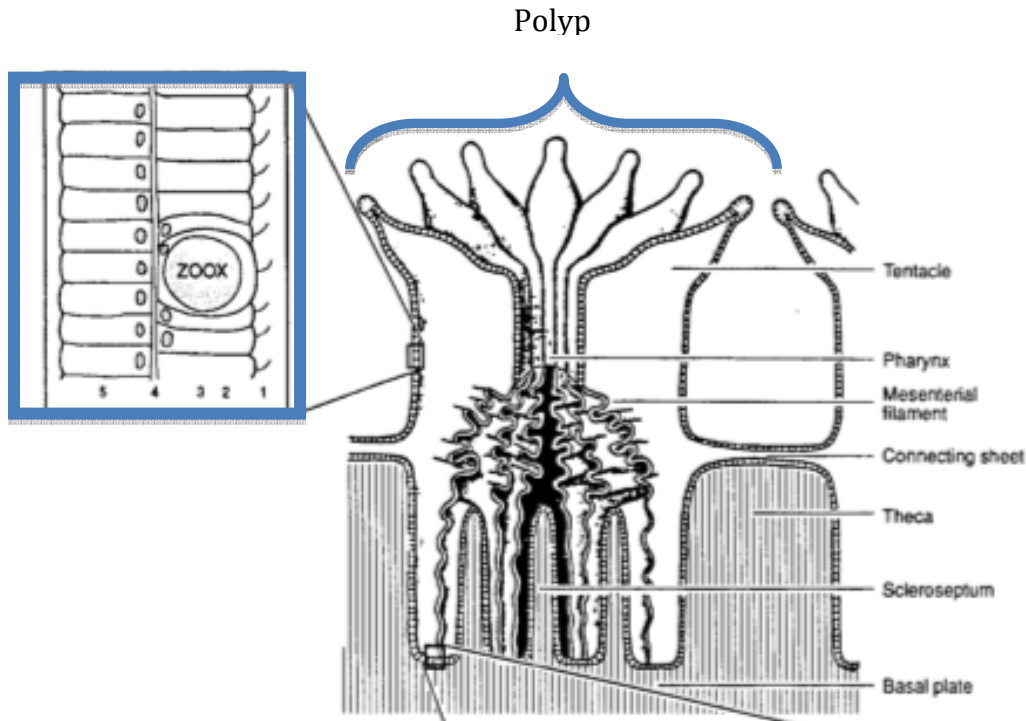
Polyp

Figure 1. Coral Polyp Anatomy.
The coral polyp is labeled in order to show how it protrudes from the colony and how each of the individual polyps are connected. The box shows a magnified diagram of the relationship of the zooxanthellae (labeled 'zoox') with the three surrounding layers of tissue (Muller-Parker and D'Elia, 1997).
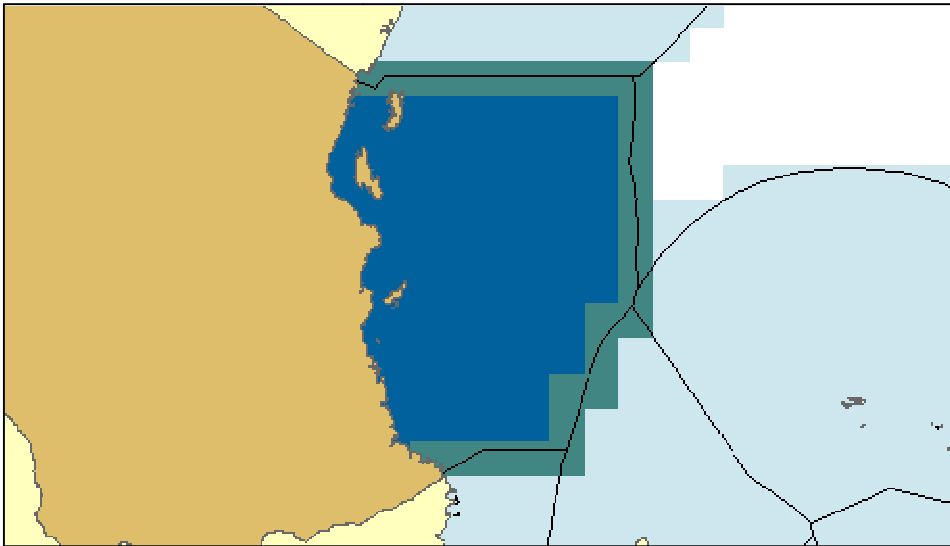
Figure 2. Exclusive Economic Zone (EEZ) of Tanzania.
The area of Tanzania's EEZ (241,541km^2) is depicted by the lines in this figure. Fishermen in Tanzania are not able to utilize the Tanzania's EEZ as most boats are not equipped with motors. The scarcity of motors leads to overexploitation of benthic coral reef fish, and the economy is unable to benefit from the more valuable pelagic fisheries income (http://www.seaaroundus.org/eez/834.aspx).

Figure 3: Islands Unguja and Pemba in relationship to the Tanzanian coast.
The two large islands are surrounded by fringing reefs as are many small islands which
make up the Zanzibar archipelago. Many small islands lie on the western shores of
Unguja and Pemba including Bawe and Chumbe Islands on Unguja
(http//:thebesttraveldestinations.com).
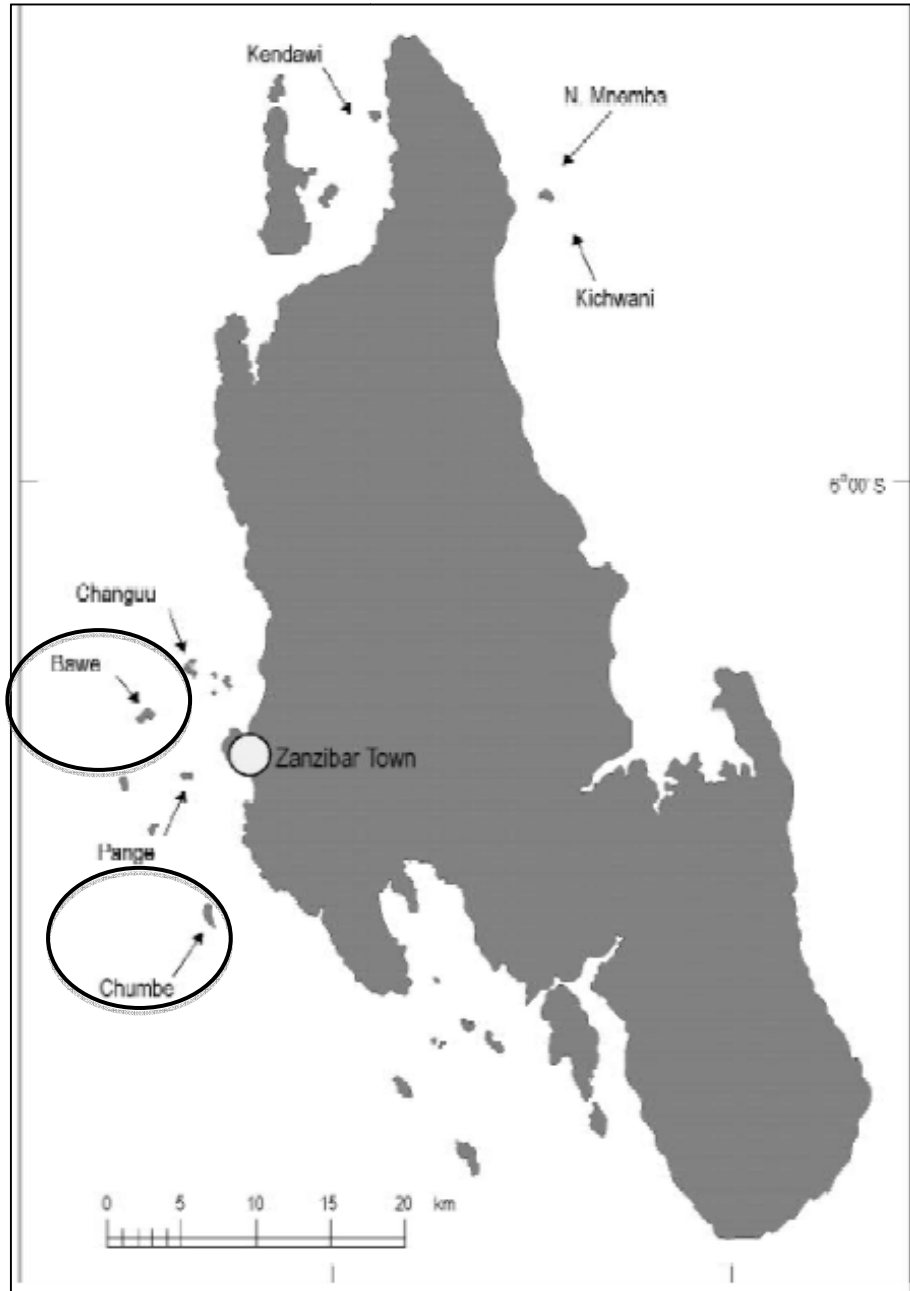
Figure 4. Study Sites: Bawe and Chumbe Islands
Both Bawe and Chumbe are circled and labeled to the west of Unguja. The proximity of
the islands to the urban center of Zanzibar Town (Stonetown) is visible in this map
(Bergman and Öhman, 2001).
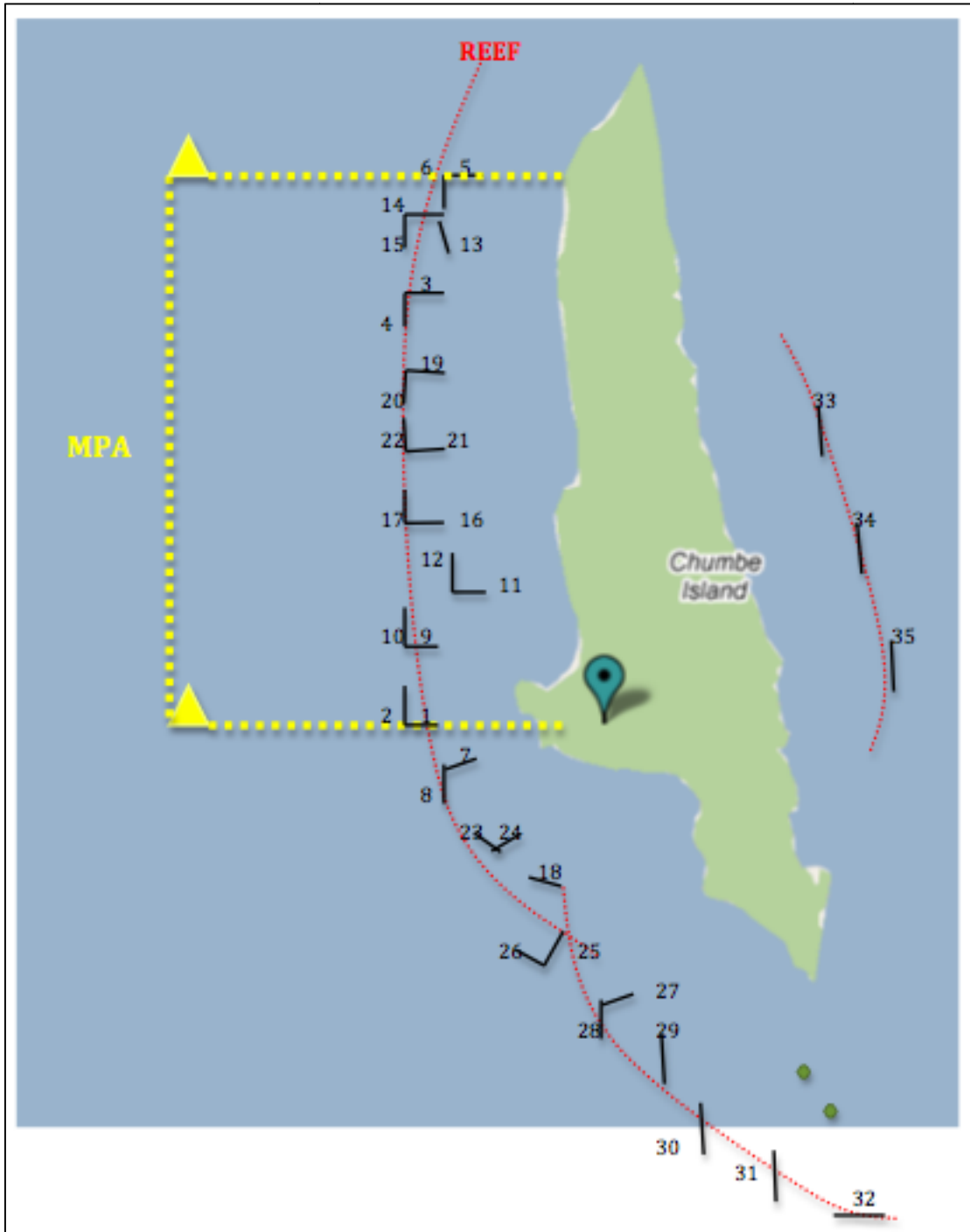
Figure 5. Transects on Chumbe Island. This figure shows the approximate location of the transects surveyed at on the Chumbe Island reef.
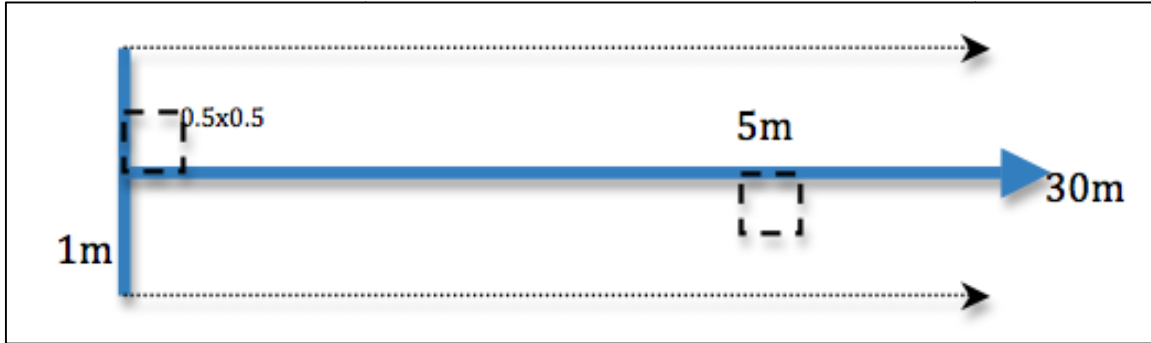
Figure 6. Diagram of Transect. A tape measure was used to determine transects on Bawe and Chumbe Island Reefs. A 30m tape was laid, and fish corallivores observed within one meter of either side of the transect were counted. Quadrats were sampled using a string with an area of 0.25m2. The string was laid on the substrate to observe the snails within the quadrat. The next quadrat was placed on the opposite side of the tape, 5m down the transect.
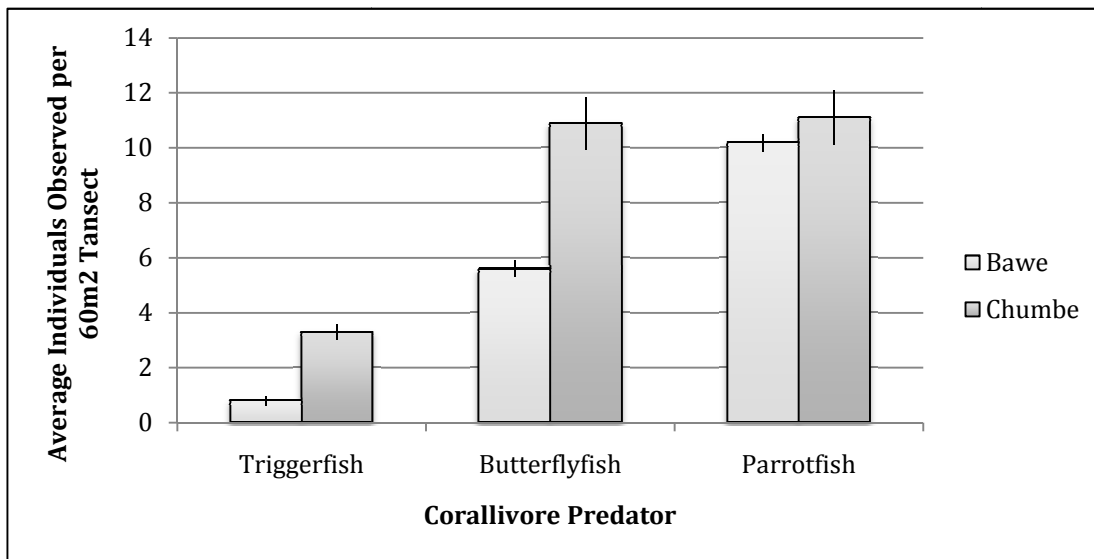
Figure 7. Average ($\pm$ SE) Corallivore Fish Observed per 60m^2 Transect. An average of 2.0 triggerfish (t=7.8, df=75, p<0.05) and 12.0 butterflyfish (t=7.9, df=844, p<0.05) were observed per transect (60m^2) at the Chumbe reef, which is significantly higher than the average observed at Bawe island reef. An average of 10.2 parrotfish (t=0.060, df=460, p=0.48) were observed per transect (60m^2) at Chumbe Island Reef, which is not significantly different from average observed at Bawe Island reef.
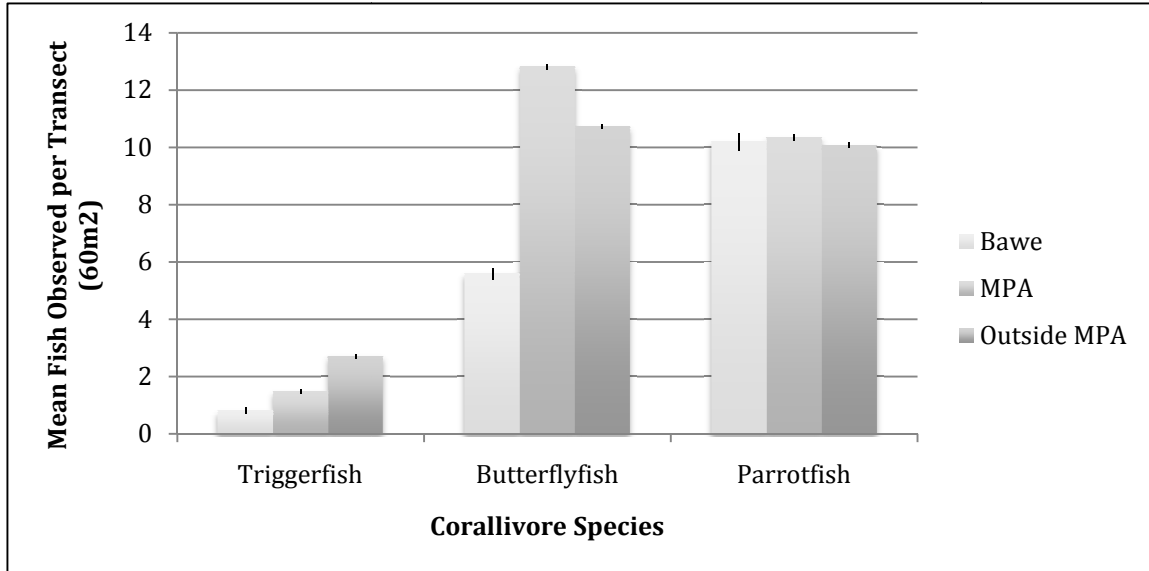
Figure 8. Average (± SE) Corallivore Fish Observed per 60m² Transect. There was a significant effect of the site (F=46.1, df=(2,476), p<0.0001) on the number of triggerfish observed, such that a significantly higher number of triggerfish were observed outside the Chumbe MPA compared to both Bawe (q=4.1) and inside the Chumbe MPA (q=4.2). There was significant effect of the site (F=46.1, df=(2,476), p<0.0001) in the number of butterflyfish observed for all sites, but not a significant effect of the site (F=0.0795, df=(2,457), p=0.9236) in the number of parrotfish observed at any of the sites.
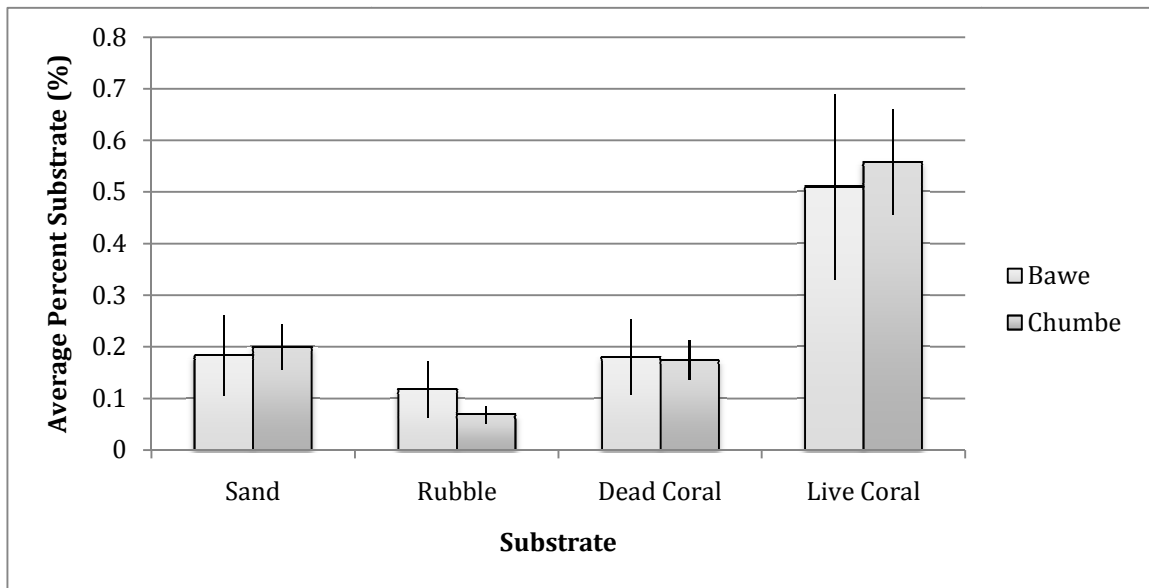
Figure 9. Average (± SE) Percent Substrate. At Bawe and Chumbe Island reefs, substrate distribution was observed for each transect. Sand was identified 18% and 21%, 12% and 6% substrate identified as rubble, 18% and 15% substrate identified as dead coral, and 51% and 58% of substrate identified as live coral for Bawe and Chumbe respectively.
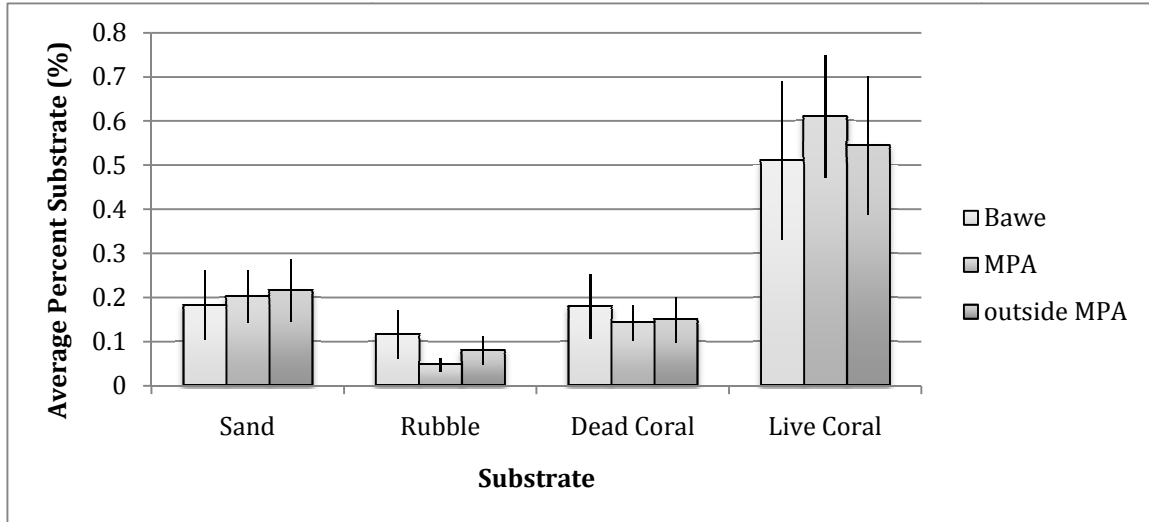
Figure 10. Average (± SE) Percent Substrate. A Chi-square analysis determined that a significant difference exists in the average amount of rubble, dead coral, and live coral observed at Bawe, within the Chumbe MPA, and outside the Chumbe MPA (χ^2=19.3, df=4, p=0.0007). Rubble and dead coral are combined to be classified as "degraded" substrate (Nelson, 2007); a Chi-square analysis showed that there is still a significant difference between "degraded substrate" and live coral.

Table 1. Bawe Island GPS Coordinates

Transect #	Direction	GPS_cord
1	50° NE	S°6, 9.426 E39°, 8.092
2	80° NE	S6°, 9.435 E39°, 8.006
3	60° NE	S6°, 9.367 E39°, 8.062
4	100° SE	S6°, 9.364 E39°, 8.070
5	140° SE	S6°, 8.720 E39°, 8.167
6	220° SW	S6°, 8.727 E39°, 8.167
7	50° NE	S6°, 9.435 E39°, 7.992
8	100° SE	S6°, 9.438 E39°, 7.985
9	50° NE	S6° 9'14.64 E39°, 8'08.12
10	80° NE	S6° 9.278 E39°, 8.116

Table 2. Chumbe Island GPS Waypoints

Transect#	GPS Waypoint	Tansect #	GPS Waypoint
1	204	18	223
2	204	19	219
3	206	20	220
4	206	21	221
5	207	22	221
6	207	23	225
7	224	24	225
8	224	25	227
9	211	26	227

10	212	27	229
11	213	28	229
12	214	29	230
13	215	30	231
14	216	31	232
15	216	32	233
16	217	33	236
17	218	34	235
		35	234

Table 3 Corallivore Data

3.1Triggerfish

Site	Number	Percent	Mean	Density
Bawe	8	5	0.8	0.007
Chumbe	69	8	1.9	0.016
Inside MPA	31	6	1.4	0.012
Outside MPA	38	12	2.7	0.022

3.2 Butterflyfish

Site	Number	Percent	Mean	Density
Bawe	56	34	5.6	0.04
Chumbe	419	50	12.0	0.10
Inside MPA	269	52	12.8	0.10
Outside MPA	150	46	10.7	0.09

3.3 Parrotfish

Site	Number	Percent	Mean	Density
Bawe	102	61	10.2	0.085
Chumbe	375	42	10.2	0.085
Inside MPA	217	42	10.3	0.086
Outside MPA	141	43	10.1	0.083

3.4 *Drupella*

Site	Number	Percent	Mean	Density
Bawe	88	63	5.9	1.5
Chumbe	92	41	1.6	0.44
Inside MPA	35	36	1.1	0.28
Outside MPA	57	46	3.2	0.68

3.5 *Corallophila*

Site	Number	Percent	Mean	Density
Bawe	51	37	3.4	0.85
Chumbe	130	59	2.4	0.62
Inside MPA	63	64	2.0	0.50
Outside MPA	67	54	2.7	0.80