

**The Distribution and impact of Sea urchins on Coral reefs in Watamu, Kenya**

BSc with Honours in Geography at the University of Exeter

GEO3321

45 Credits

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I certify that this dissertation is entirely my own work and no part of it has been submitted for a degree or other qualification in this or another institution. I also certify that I have not collected data nor shared data with another candidate at Exeter University or elsewhere without specific authorisation.

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## **Acknowledgments**

A huge thanks to the A Rocha team in Watamu for making this project happen, in particular the marine team of Benjo, Tori, Peter and Andrew who were always around to help. Special thanks to Benjo for making this study so enjoyable and for his help in both developing the project and undertaking field work. Finally I would like to thank my supervisor Chris Perry for his advice and guidance with this project.

## **Abstract**

The distribution of sea urchin species can dictate the morphological structure and benthic characteristics of coral reefs. Sea urchins play an important role in coral reef ecosystems, however high population densities can damage reef structure and reduce biodiversity. In order to note the impact of sea urchin populations on coral reefs we examined 5 sites in a coral reef lagoon area Watamu, Kenya. Sites both inside and outside a marine park were surveyed in order to note the effect of protection on sea urchin populations. Results showed that average sea urchin density was significantly higher outside the park ( $12.69/20\text{m}^2$ ) than in protected areas ( $2.40/20\text{m}^2$ ). No significant evidence was found to show that increased sea urchin abundance affected coral cover or rugosity. Results found that unprotected reefs are predominantly echinoid grazed, have high levels of bioerosion and reduced fish stocks. We compared our results to previous studies on Kenyan coral reef lagoons and the findings suggest that marine parks are important in controlling sea urchin populations and maintaining reef productivity.

## **1. INTRODUCTION**

### **1.1 Aims and objectives**

#### **AIM:**

This project aims to investigate the abundance and distribution of sea urchins in the Watamu marine park area and their impact on coral reef characteristics.

#### **RESEARCH QUESTIONS**

The investigation aims to address the following specific research questions:

1. Does sea urchin abundance vary across sites?
2. Does sea urchin abundance reduce coral cover as a result of bioerosion?
3. Does sea urchin abundance effect the benthic composition of coral reefs?
4. Does density/abundance of sea urchins vary inside the park and out?
5. Do coral reef benthic characteristics vary within and outside the marine park?

#### **Objectives**

These research questions will be examined using the following objectives:

1. Identify 5 accessible sites in the coral reef Watamu lagoon, 3 within the marine park and 2 in unprotected areas.
2. Lay transects within these sites in order to survey coral reefs and collect data on sea urchin abundance, coral cover, reef benthic cover and rugosity.
3. Analyse data in order to note whether significant relationships exists between:
  - Sea urchin abundance and coral cover
  - Sea urchin abundance and reef rugosity
  - Sea urchin abundance and benthic community composition, in particular algae abundance
  - Sea urchin abundance and the marine park
4. Discuss the results noting any relationships between our variables and the factors that may affect these relationships.



## **1.2 Coral reefs and Bioerosion**

Coral reefs are one of the most biodiverse ecosystems in the world (Glynn 1997). Not only do they provide habitats for a huge variety of organisms, coral reefs are also important for coastal protection, tourism and fisheries (Hoegh-Guldberg, 1999). Coral reefs are mostly composed of calcium carbonate and corals are the main contributors to the calcium carbonate framework. The carbonate production and building of the reef framework is countered through erosion processes that convert this calcium carbonate in to sediments (Perry 2012). This results in the reef state being held in balance by both growth of corals and the simultaneous erosion of the reef substrate (Peryot-Clausade et al., 2000; Glynn 1997). Bioerosion can be defined as ‘the erosion of coral and coralline algal by biological agents’ and this process can have a significant impact on reef structure (Glynn, 1997; Peryot- Clausade et al., 2000). The process of bioerosion is important in reef growth and the development of reefs and some previous studies have linked elevated bioerosion rates to higher rates of carbonate production (Glynn, 1997). Other important features of bioerosion in a reef ecosystem are the creation of burrows and crevices and this increasing topographic complexity can benefit a number of reef organisms (Hutchings 1986; Glynn 1997). However where bioerosion levels are extremely high it can result in degradation of the reef skeleton and subsequent reef damage (Bak 1994; Glynn 1997, Perry et al., 2008) and therefore the reef state is held in a delicate balance. The balance of bioerosion is similar to the intermediate disturbance hypothesis, where low to intermediate levels of bioerosion can have positive impacts on the reef environment (Glynn, 1997) up until a point where levels of bioerosion increase past the threshold and begin to negatively impact coral reefs (Hereu et al., 2005). A number of bioeroders are present on coral reefs and are often categorized into internal and external eroders. The erosion by organisms on the surface of coral reefs is usually considered to be more significant than that of internal reef bioeroders (Glynn 1997). External bioeroders are mainly herbivorous fish and sea urchins (Peryot-Clausade et al., 2000) and these two grazers can be seen to be competitors on coral reefs (McClanahan et al., (1994). A study by Glynn (1997) investigated the impact of all bioerosive organisms on coral reefs. The results of this study suggested that sea urchins have the highest rates of erosion for all biologically destructive organisms.

### **1.3 Sea urchins**

Sea urchins are part of the larger echinoderm family. This family consists of irregular Echinoids that do not erode reefs, and regular urchins which are bioeroders and are also the most common sea urchins found on coral reefs (Bak, 1994). Sea urchins have a 'highly evolved jaw apparatus' (Glynn, 1997) which comprises of five self-sharpening calcium carbonate teeth. Their mouth and teeth are responsible for scraping and erosion of the hard reef substrate (Bak) and are the reason sea urchins are such key grazers on coral reefs (Jones and Andrew., 1990). As well as erosion through grazing, sea urchins are also responsible for indirectly eroding coral reefs with their spines that can weaken the reef and often expand slight cracks or crevices (Glynn, 1997). Sea urchins feed on loose sediment trapped in algal turf and also scrape material from coral surface and rubble (Carreiro-Silva and McClanahan 2001). Additionally some species of sea urchins burrow into crevices to avoid predation and trap sediment in long extended spines as it drifts by (Glynn, 1997). Sea urchin grazing on reefs is important to control the population of macroalgae, which when dominant can result in mass coral mortality (Sammarco, 1982). As sea urchin density can have such a significant effect on coral reefs through bioerosion, their population distribution and subsequent impact on coral reefs is particularly important to understand.

### **1.4 Sea urchins on coral reefs**

Previous studies have examined the impact of sea urchin populations on coral reef ecology and structure. In particular the Caribbean has been a key area for research as the natural dieback of the sea urchin *Diadema antillarum* in 1983 provided an opportunity to examine the effects of complete sea urchin removal on reef ecosystems (Lessios et al., 1984; Carpenter, 1998). Studies found that the mass removal of 95-99 % of *D. antillarum* from reefs resulted in a rise in macroalgal dominance as grazing was greatly reduced (Carpenter, 1988). Abundance of macroalgae on coral reefs can inhibit coral growth and result in an algae dominated reef (Edmunds and Carpenter 2001). Edmunds and Carpenter (2001) found that the recovery of *D. antillarum* on reefs since the mass mortality event has been correlated with a reduction in algae cover. Additionally Edmunds and Carpenter (2001) found a positive relationship between sea urchin density and juvenile coral abundance. However *D. antillarum* is also known to consume live coral as part of its diet (Carpenter 1988). So at high densities rather than assisting coral

recruitment through algae grazing, sea urchins can begin to damage live corals (Edmunds and Carpenter 2001). The ability for sea urchins to have both a positive and negative effect on coral reefs led to the suggestion by McClanahan (1995) that the relationship between these variables is not completely understood. Studies on Kenyan reefs by McClanahan and Mutere (1994) and McClanahan and Shafir (1990) have found a negative correlation between sea urchin abundance and coral cover. This led to a study by McClanahan et al., (1996) who suggested that the removal of sea urchins from coral reefs may benefit reef structure and hard coral abundance. However results from this study showed that a reduction in grazing intensity eventually led to a rise in macroalgae dominance. As well as having an effect on live coral cover sea urchins can contribute to the composition of coral communities (Sammaraco, 1982) previous studies have found the dominance of certain coral genera at varying densities of sea urchins on Kenyan coral reefs (McClanahan et al., 1999; McClanahan and Mutere 1994). These two studies noted a dominance of the genus *porites* mass at sites densely populated by sea urchins, and agree that the distribution of sea urchin species can affect the abundance of different coral genera. Sea urchin abundance can have a significant effect on coral species and coral diversity which in turn dictate reef ecology and biodiversity (McClanahan and Mutere, 1994). Sea urchin populations can also impact the structure of coral reefs. At high sea densities, bioerosion of the reef framework can result in a reduction in reef rugosity (McClanahan and Shafir, 1990). Reduced topographic complexity as a result of sea urchin abundance was found in a number of studies on Kenya coral reefs by McClanahan and Shafir, (1990), McClanahan and Mutere (1994) McClanahan (1995). These studies suggest that on reefs where sea urchin populations are dominant, extensive bioerosion can result in decreased reef rugosity.

### **1.5 Marine parks**

The research of sea urchins on Indian Ocean coral reefs has mainly been focused on the effect on marine parks in East African lagoons. McClanahan (1995:1998), McClanahan and Mutere (1994) and Carreiro-Silva and McClanahan (2001) have conducted a number of investigations along the Kenyan coast looking into sea urchin populations in relation to MPA (Marine park areas) and the level of predation at sites. They found an increase in sea urchin population density, test size and species diversity outside MPAs. Studies in the Caribbean also showed that *Diadema* densities were higher on intensely fished reefs where sea urchin predators were absent (Lewis and Wainwright, 1985). Globally, coastal areas tend to have high and growing populations (Hoegh-Guldberg

1999). This is true for Kenyan coastal regions and in these areas fishing is an important food source and income for many (Muthiga and Maina 2003). Previous research undertaken in Kenyan reefs has found that overfishing, and in particular the removal of certain key species, is having a major effect on unprotected reefs (Muthiga and Maina 2003). Both Muthiga and Maina (2003) and McClanahan (1995) highlight the importance of triggerfish in sea urchin predation and the implications when this species is removed. The removal of sea urchin predators and the resultant increase in sea urchin populations can alter the benthic community on reefs leading to changes in the substrate cover found on reefs. Muthiga and Maina (2003) and McClanahan and Shafir (1990) found that on protected reefs dominant substrate cover included hard coral and coralline algae while unprotected reefs consisted of more bare substrate or algal turf and soft coral species. This study looks to investigate if a clear relationship between sea urchin abundance and coral cover can be noted, as previously found by McClanahan and Shafir (1990). Additionally by investigating sites both inside and outside a marine park, we can note differences in both sea urchin distribution and reef composition as a result of ecosystem protection. Here we aim to investigate current sea urchin population structure, coral cover and reef rugosity also noting the effect of the marine park on these factors, albeit on a much smaller scale than previous studies. It is worth noting that the exact sites from previous studies in this area were not replicated, however the previous studies may be useful to consider as a basic comparison tool.

Our two main hypotheses' for this study is that 1) Sea urchin abundance causes a reduction in hard coral cover. 2) Sea urchin density is higher outside of the marine park. As well as this we look to investigate a number of questions on sea urchin abundance, reef composition and structure set out previously in our aims. Time constraints, environmental conditions and limited resources impacted the extent of the research; however the project provides the author with a deeper understanding of scientific studies and an insight into coral reef composition in Watamu, Kenya.

### 1.5.1 Location and characteristics

This study was undertaken in Watamu Marine Park, on the coast of Kenya. Watamu Marine Park covers an area of 10km<sup>2</sup> and has been a UNESCO and Biosphere reserve since 1968; and is therefore protected from fishing activities and is a popular attraction for tourists (Muthiga and Maina 2003). The reef found in Watamu Marine Park comprises of corals growing along a reef edge (McClanahan and Mutere, 1984) and is mostly characterized by patches of hard corals surrounded by sand and some sea grass beds. Figure 1 shows a map of the area with the marine park boundaries shown by the red line. The study sites used are marked on below and consist of 5 reefs, 2 of which are outside or on the boundary of the marine park and 3 sites that are situated inside the park. The sites are all shallow lagoon areas separated from the main sea; however the site at Uyombo (UY) is deeper than the others and may be more exposed to the sea processes. Site characteristics are explained more in the methodology.

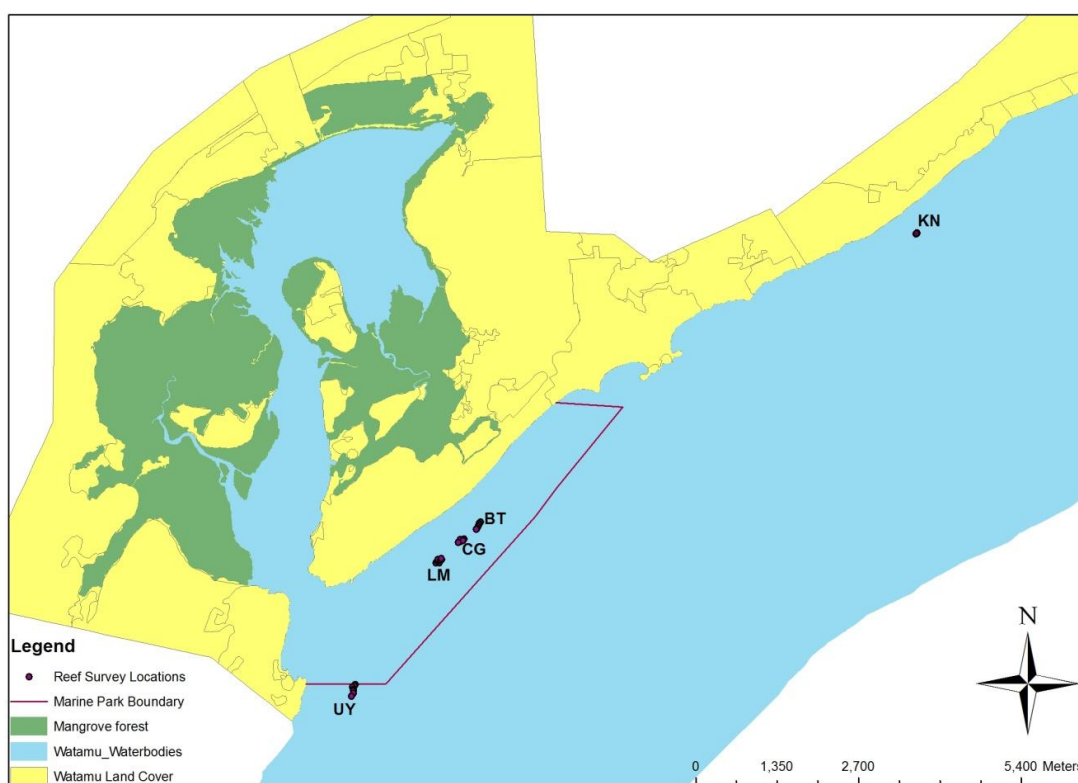


Figure 1: Shows a map of the study area in Watamu. The Watamu marine park boundary is shown by the red line. The sites surveyed in this study were provided by Arocha marine as GPS points: **CG**-Coral gardens, **BT**- Bennets, **LM**- Lambis, **KN**- Kanani and **UY**- Uyombo. Shape files were provided by coast mapping project (2013) and show Land cover, the mangrove forest and the creek characteristics.

## **2. METHODOLOGY**

### **2.1 Site selection**

In the Watamu areas five sites were surveyed, three inside the park and two others outside the MPA. At each location as many transects as possible was recorded but numbers vary due to tides and other limitations. The sites (figure 1) were chosen because of their location in the marine park and their accessibility. Coral Gardens, Bennets and Lambis were chosen as they are three similar lagoon reefs in the centre of the marine park and are therefore completely uninfluenced by activities outside the park. These three sites were shallow enough at low tide to allow surveys to be conducted using snorkels. Accessing sites outside the park was more challenging as we needed to find reef lagoon habitats similar to those inside the park yet still accessible. Uyombo was our site furthest from shore and was situated at the mouth of the tidal creek (figure 1); this site was useful as it was outside of the park and a highly disturbed and fished area (Author, *pers. comm.*). Only 6 transects were surveyed at Uyombo as firstly the depth required scuba equipment and conditions during the survey were extremely difficult with currents and strong waves. Kanani was the final site chosen outside the park and provided an enclosed lagoon reef area, also commonly fished (Author, *pers. comm.*). The reef at Kanani was well protected from the open ocean and 12 transects were recorded. It is important to note that this site was characterized by sea grass beds around patches of coral. Transects were chosen randomly with each site at areas where 10m of continuous or almost continuous coral patches could be found as was done by McClanahan and Mutere (1994). Transects were placed making sure that the same area was not covered twice and that habitats surveyed were the same (excluding large areas of sand or sea grass beds). The number of transects surveyed was limited by the tides as rough seas made surveying difficult when not at low tide.

### **2.2 Line transect method**

The line intercept transect method was used for this study as it has a number of advantages including its simplicity, reliability, and potential for monitoring temporal change (English et al., 1997). This method is rapid to deploy and allows us to calculate a number of reef characteristics such as rugosity, coral cover as well as sea urchin

numbers along the one transect. The disadvantages with the line transect method is the difficulty to standardize life form categories when noting benthic community of coral reefs (English et al., 1997). However in order to overcome this we are using Coral Point Count (CPCe) software. The CPCe method is explained further below and allows us to overcome errors that may occur in the field by analysing photographs of substrate cover using computer software. The measurements recorded for this study included; Sea urchin species density and test size, rugosity and reef benthic cover.

### **2.3 Sea urchin abundance**

Sea urchin numbers were recorded from one meter either side of the transect line. A meter long plastic pipe was used to note if the urchin was within the transect area. Two meters either side of the 10m line was surveyed resulting in the total area being 20m<sup>2</sup>. The total number of sea urchins was divided by the total area in order to calculate sea urchin density per meter squared. Sea urchin species were recorded and where they were not known, photos were taken for future identification. The size of the sea urchin was noted in order to calculate bioerosion rates. In some cases, especially with the *E. molaris*, measurements become increasingly difficult as the urchin is burrowed into crevices along the reef. The size of the sea urchins was measured using a scale/ruler; care was taken to measure body size of the sea urchin as opposed to spine lengths. The individual sea urchins were classed into size ranges; 0-20mm, 21-40mm, 41-60mm, 61-80mm, 81-100mm.

### **2.4 Rugosity**

In order to calculate rugosity of each transect a 5 meter chain was used. This was placed at the start of each transect and measured the actual bottom contour distance of the reef. The straight line distance, 5m in this case, was divided by the bottom contour distance to provide a value of topographic complexity. A 5 meter chain was used for convenience, as a 10 metre chain although covering the whole transect is much heavier and is increasingly difficult to use in a snorkelling survey. The limitation of this is that the rugosity is only calculated for half of the transect.

### **2.5 Coral cover and CPCe**

The benthic cover of each transect was recorded using photo quadrats. These were taken at four places along the transect (2m, 4m, 6m and 8m). At each 2 meters the 1m by 1m

quadrat was placed along the transect line and the substrate within the quadrat was photographed. The photos were then analysed using a computer program called Coral Point Count (CPCe). CPCe increases accuracy and efficiency of analysing large amounts of data and allows the statistical analysis of benthic cover (Kohler and Gill 2006). CPCe is especially useful in this study as it provides a method that allows identification of reef substrate cover on the computer as opposed to during the survey which can help those less experienced in species identification. Using the CPCe program we assigned 25 random points to each 1m by 1m quadrat, giving us a percentage cover for the transect out of 100%. The 25 randomly chosen points were then identified and classified using a CPCe code created specifically for the Watamu reef area. Using this code the substrate cover was identified. Once the substrate was identified each quadrat was then processed using CPCe producing an excel sheet that shows substrate characteristics for each transect and site.

## **2.6 Bioerosion rates**

Sea urchin erosion of coral reefs is a reasonably well studied process, and a clear correlation has been noted between test size and bioerosion rates (Bak, 1994). For this study we used an equation provided by the Exeter reef budget methodology (Perry et al. [www.geography.exeter.ac.uk](http://www.geography.exeter.ac.uk)) that calculates bioerosion as a result of test diameter but does not take into account species variation. The general equation for bioerosion:

$$\text{Bioerosion rate (g/urchin/day)} = 8 \cdot 10^{-5} X^{2.4537}$$

(Where X is the test diameter)

This equation was used with median urchin sizes for each category. As expected we see an increase in bioerosion rates with rising median test size. This is illustrated in figure 2 below.



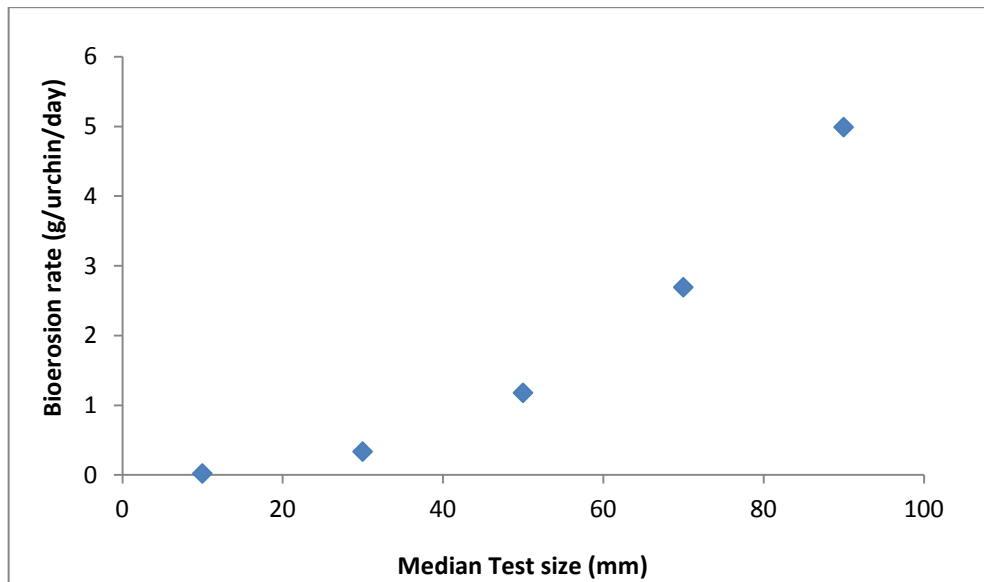


Figure 2: Graph showing the relationship between Sea urchin test size and bioerosion rates that was used to calculate the bioerosion of sea urchins at each site.

### **3. RESULTS AND ANALYSIS**

#### **3.1 Site benthic community composition**

The dominant reef benthic composition for all sites was comprised of macro and turf algae (bare substrate) followed by hard corals. The reefs at Uyombo and Lambis were found to have the highest percentage cover of turf algae at 53.17% and 44.72% respectively. As shown in figure 3.1 Bennets reef had the lowest percentage of turf algae at 24.12 % but had a higher presence of macro algae accounting for 48.39% of the substrate cover. Macroalgae cover is at its lowest at Uyombo at just 15.87% of the site. Across the sites hard coral comprises an average of 15% of the benthic cover and the standard deviation across all five sites was 5.11%. Hard coral cover is at its highest at Bennets (23.03%) and at its lowest on the transects at Lambis (9.90%). Sand and rubble percentage was recorded and it was noted that at Bennets and Kanani the discontinuous coral heads were surrounded by sand beds. Whereas at Uyombo, where 11.07% of this category was recorded, we observed this comprised of mainly rubble from broken dead coral.

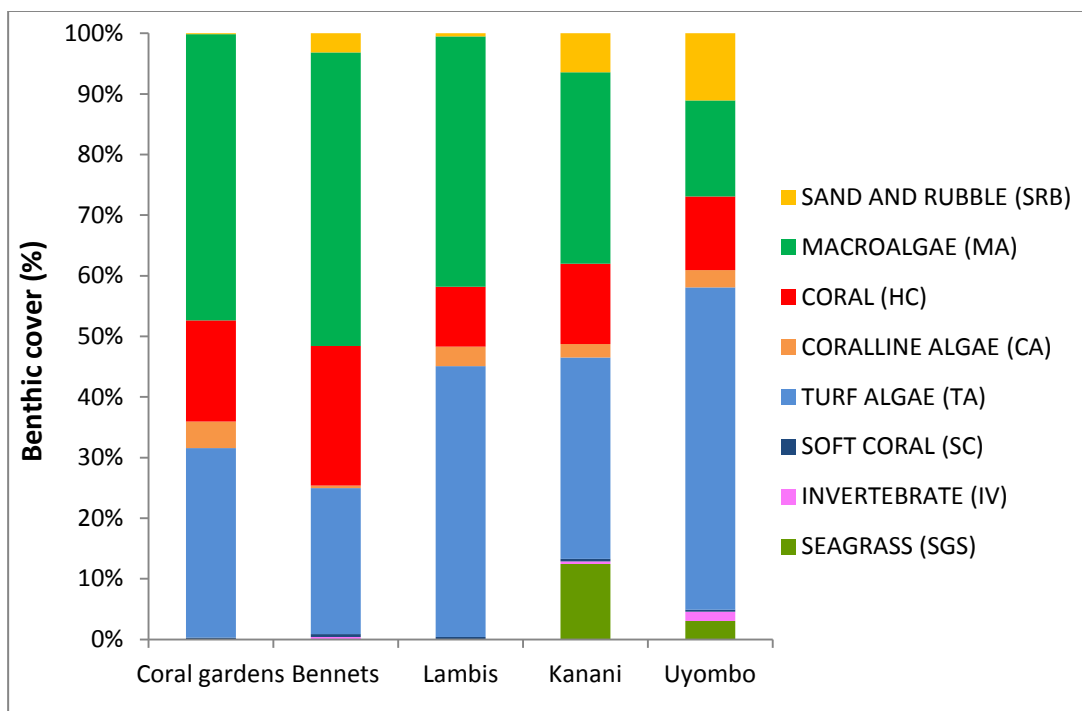


Figure 3.1: This shows the mean reef benthic cover for the transects at each site. The type of substrate cover is based on the broadest categories and displays general site composition.

Sea grass was particularly prominent at Kanani as it covered a mean percentage of 12.43% of the transects. This is not unexpected as this site featured a number of sea grass beds surrounding the reef, the only other site where sea grass was recorded was Uyombo at 3.04%. Soft coral and coralline algae deviate between sites by only 0.07% and 1.46% respectively. From figure 3.1 we can note that Bennets seems to have the smallest amount of coralline algae cover at 0.43%. The final category of invertebrates was comprised of sponges, clams, anemone, zooanthids and sea urchins. Invertebrates did not contribute to a significant percentage of the benthic cover however figure 3.1 shows that this category featured mainly in the two sites outside the park, at both Uyombo and Kanani. This was as a result of sea urchins out in the open on the substrate. This differs to the invertebrates noted at Bennets which consisted of clams (0.18%) and sponges (0.14%).

### **3.2 Coral cover**

As previously shown in figure 3.1 hard coral cover shows little variation between sites. In order to investigate this further a one way Anova analysis of coral cover by location was performed (See Appendix 3). This showed no statistical difference in hard coral percentage between sites a prob>f or p value of 0.4303 shows no significant variation. The graph below (figure 3.2) shows the coral cover across each site. The standard deviation between transects was calculated for each site, and we can note that at all sites large variations from the mean was found.

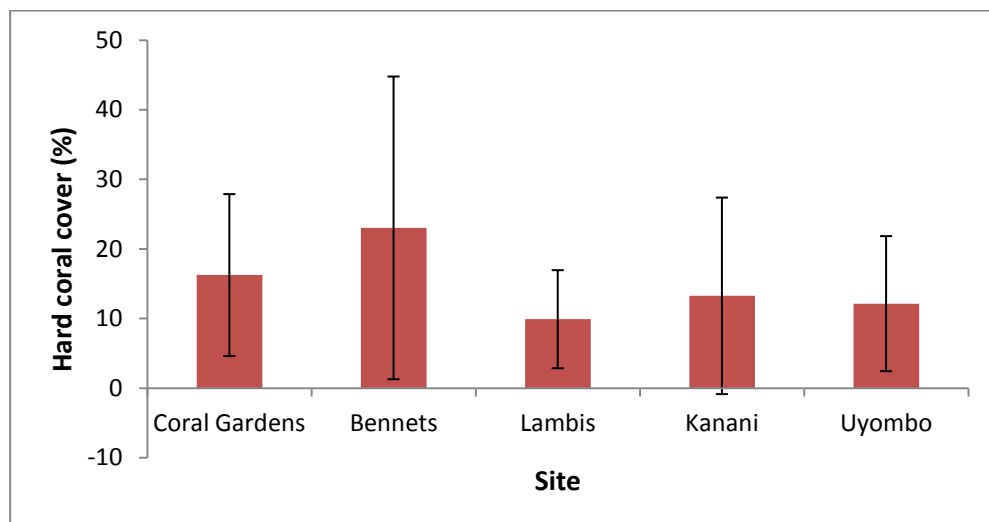


Figure 3.2: Graph showing mean percentage of hard coral cover at each site; standard deviation from the mean is shown by error bars.

Coral cover was recorded by genus except for *porites* mass which was classified to species level. An analysis of the four most dominant coral genera at each site was done (See Appendix 2 for all coral genera). As shown in figure 3.3 we can note that the four dominant coral genera in the area are *Porites* mass, *Pocillopora*, *Echinopora* and *Acropora*. Examples of these four corals are shown in figure 3.4 below. *Porites* mass was particularly common at Coral gardens at 9.42% of the total benthic cover, and Bennets at 12.86%, and was found in small amounts at Kanani (1.19%). *Pocillopora* was not recorded at Bennets or Lambis and was only found at a very small percent at Coral Gardens (0.14%). Uyombo showed the highest *Pocillopora* cover at 4.55% of the total benthic cover and also the highest percentage cover of *Acropora*. *Acropora* was found at all of the 5 sites but was only found above 5% cover at Uyombo and on all other sites was found at much lower frequencies. The reef at Bennets showed high frequency of *Echinopora* coral (7.15%) and at all other four sites it was below 2%, its lowest presence only 0.17% on the reef at Uyombo.

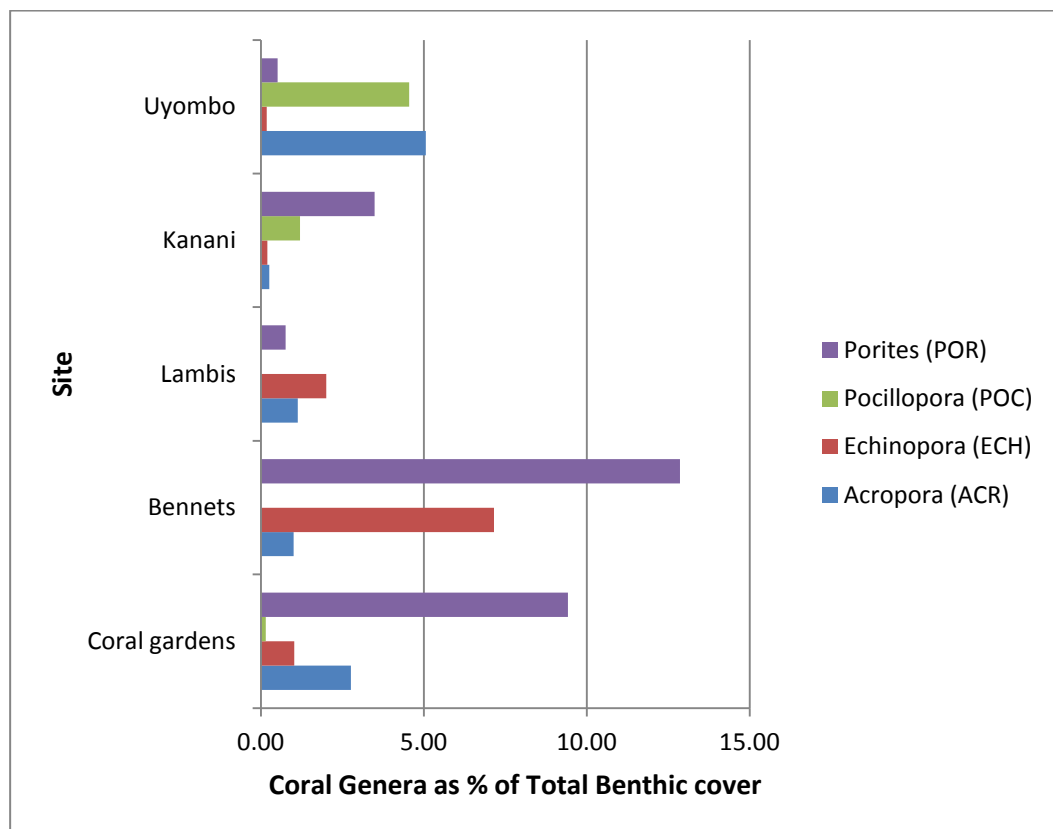


Figure 3.3: Shows the 4 most common coral genera noted in this study and their percentage cover at each site. The value of coral cover is as a percentage of the total benthic cover recorded.

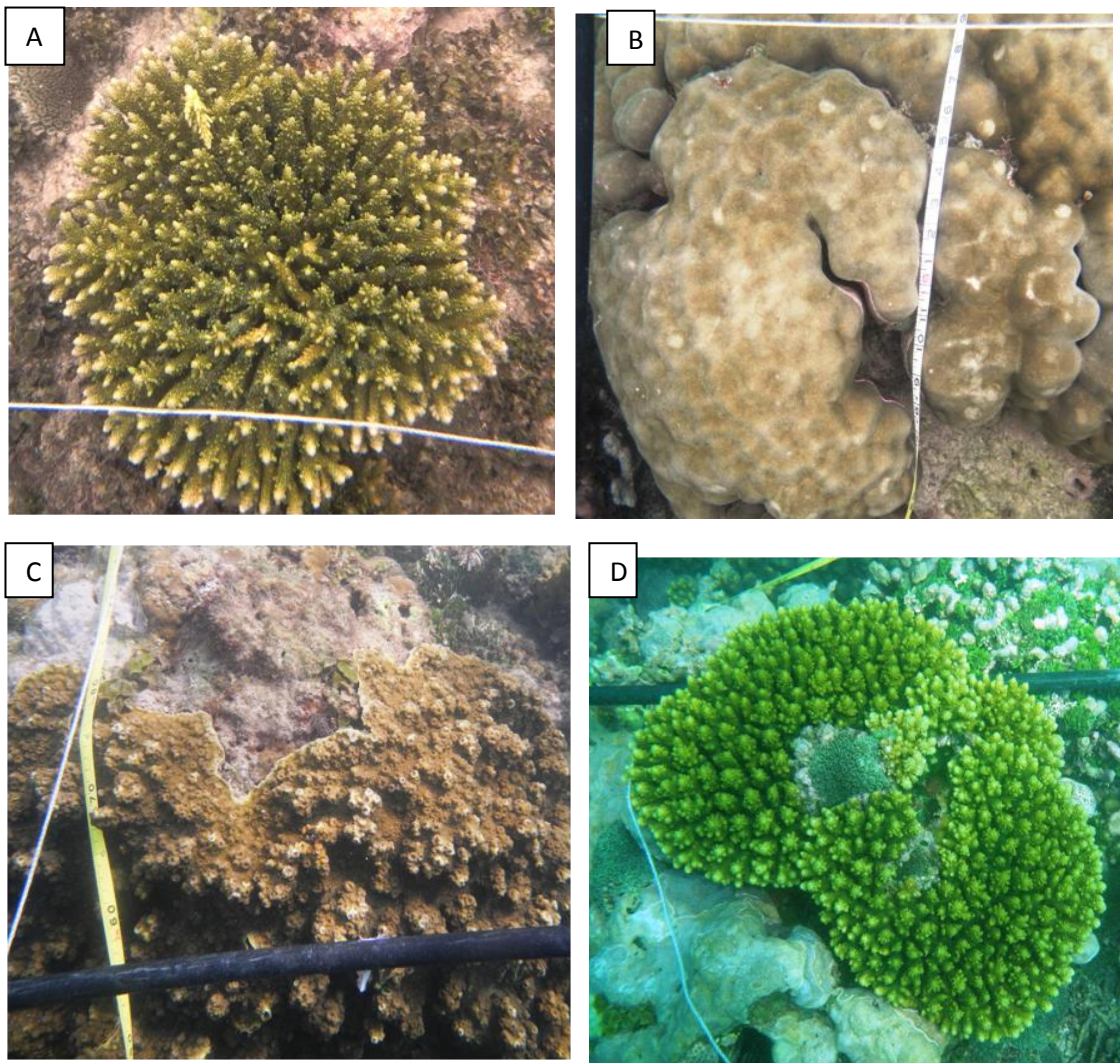


Figure 3.4: Four most dominant corals at genus level except for the species *porites* mass found on our 5 sites in Watamu: A) *Acropora*, B) *Porites* Mass C) *Echinopora* D) *Pocillipora*. (Photos:author)



### 3.3 Sea urchin species

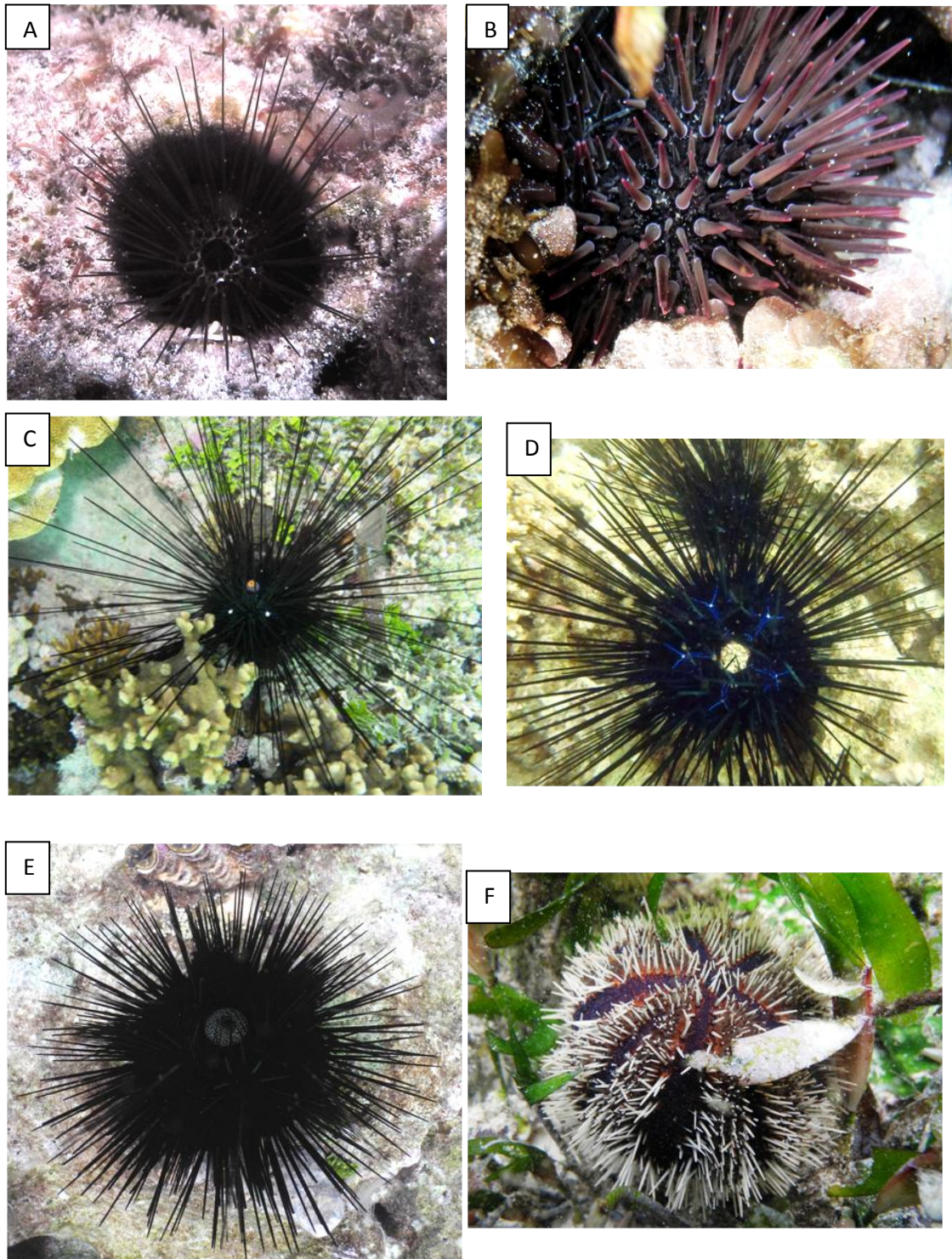


Figure 3.5: The Sea urchin species identified on the sites (excluding *E.calamaris* where only one test was noted) A) *Echinostrephus molaris* B) *Echinometra mathaei* C) *Diadema setosum* D) *Diadema savingyi* E) *Echinometre diadema* F) *Tripneustes gratilla*.  
(Photos: author except fig 2.6E provided by A rocha marine)

Figure 3.5 shows the 6 most common sea urchin species that were found on our sites in the Watamu area. Figure 3.5a shows the species *Echinostrephus molaris* which was the smallest test size found. All measured individuals of this species were less than 20mm in diameter, except for one test that was classed in the 21-40mm bracket at Uyombo. All of the sea urchins for the species *Echinostrephus molaris* were found hidden in crevices on the reef as shown in the photo from figure 3.5a. Figure 3.5b shows the species *Echinometra mathaei* which despite being reported as the dominant urchin on Kenyan reefs (McClanahan) was found rarely. The mean size as shown in table 3.1 was 10.29mm and this species was also found mainly hidden in crevices. The two species *Diadema savignyi* and *Diadema setosum* (figure 3.5 c and d) were larger than *E. molaris* and *E. mathaei* and had similar mean test sizes 44.29mm and 40mm respectively.

<b>Species</b>	<b>Total for all sites</b>	<b>Mean test size (mm)</b>
<b>Echinometra mathaei</b>	4	10.00
<b>Echinostrephus molaris</b>	68	10.29
<b>Diadema Savignyi</b>	21	44.29
<b>Diadema Setosum</b>	16	40.00

Table 3.1 shows the total number of each sea urchin species found across all study sites, and the mean test size (mm).

Figure 3.6 shows distribution of sea urchin species at each site. At Coral Gardens, Bennets and Lambis *E. molaris* is the dominant sea urchin species. The reef at Uyombo shows the dominance of the *Echinometra diadema* sea urchin which is also the largest sea urchin species found on Kenyan reefs (Carreiro-Silva and McClanahan, 2001) with a mean test size of 74.64mm. *Tripneustes gratilla* (figure 3.5f) was only found at Kanani where, as previously mentioned, sea grass beds were noted around the coral patches. The mean test for this species was 55.40mm; this species does not contribute to the bioerosion of reefs as it grazes on sea grass and has little effect on the reef composition (Silahooy, 2013). From figure 3.6 we can note that Kanani and Uyombo have a larger range of sea urchin species in comparison to the Coral gardens, Bennets and Lambis sites.

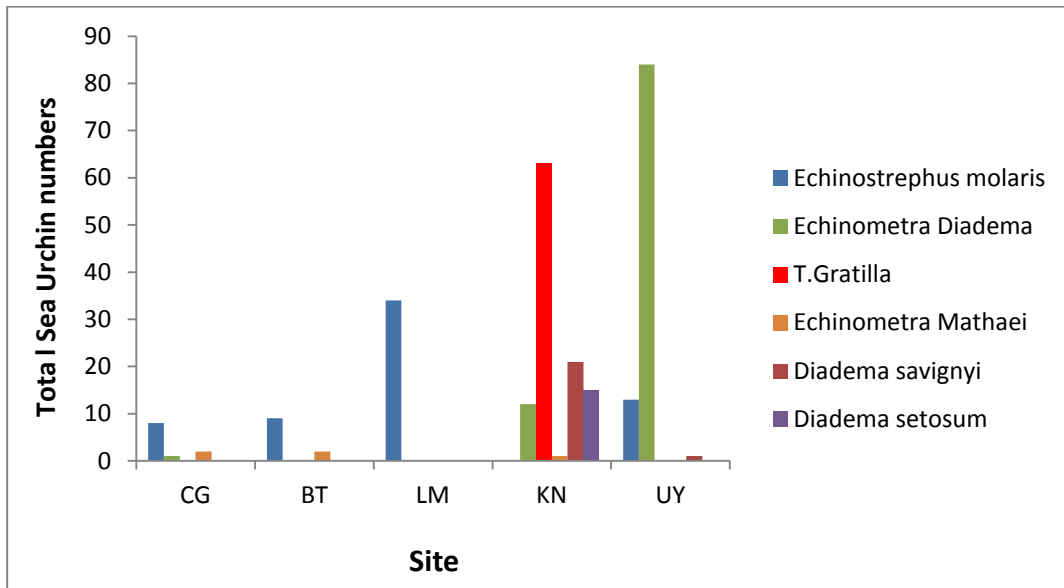


Figure 3.6: Total number of Sea urchins found at each site for each species, *E.molaris*, *E.diadema*, *T.gratilla*, *E.mathaei*, *D. savignyi*, and *D. setosum*.

### 3.4 Sea urchin abundance and reef composition

A significant difference was noted in sea urchin abundance between sites. Figure 3.7 below shows a clear variation in sea urchin density per transect.

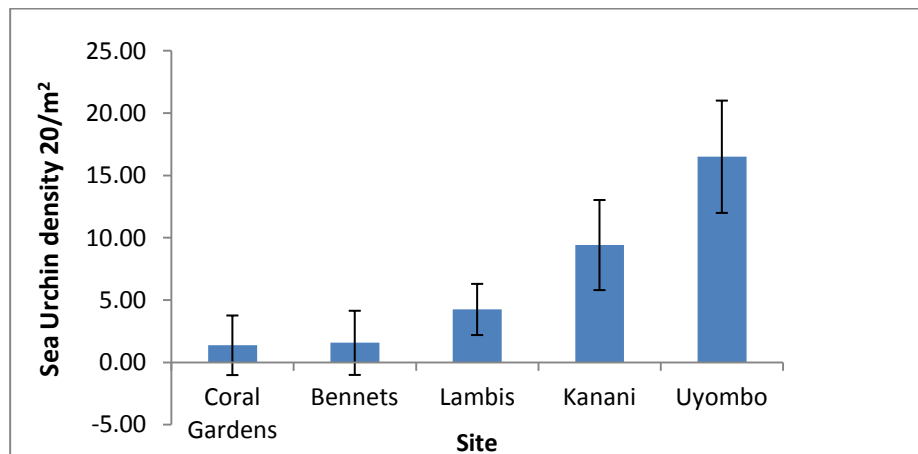


Figure 3.7: Sea urchin density/20 m<sup>2</sup> at each site. Error bars show standard deviation.

A one way Anova analysis was performed and the difference was found to be statistically significant with a prob>f (p value) of 0.0019 (See Appendix 4). As shown by figure 3.7, sea urchin density is at its highest on Uyombo at 16.50/ 20m<sup>2</sup>, followed



by Kanani with a density of  $9.42/20\text{m}^2$ . The three sites within the park have a much lower density with Lambis at  $4.25/20\text{m}^2$  and Coral Gardens and Bennets showing the least abundance with  $1.38/20\text{m}^2$  and  $1.57/20\text{m}^2$  respectively. Standard deviation is also displayed on the graph (figure 3.7); this shows the variability of our data from the mean. The standard deviation in this data set is relatively high and at Coral Gardens and Bennets this may be as a result of many transects completely absent of sea urchins. Standard deviation questions the reliability of our data and is highest at Uyombo where only 6 transects were noted. Despite this we can comfortably say that the difference in sea urchin abundance at the five sites was significant.

Our main hypothesis was that sea urchin abundance would cause sites to have less coral cover. In order to test this hypothesis a simple scatter plot was created in order to note if there was a negative linear relationship between increasing sea urchin density and coral abundance. As we know from our results above the difference in sea urchin density across sites is significant, however figure 3.8 shows that from our sites there is no significant relationship between the sea urchin density and coral cover. The scatter plot (figure 3.8) shows an  $R^2$  value of 0.0107 which suggests that sea urchin density has little effect on the coral cover.

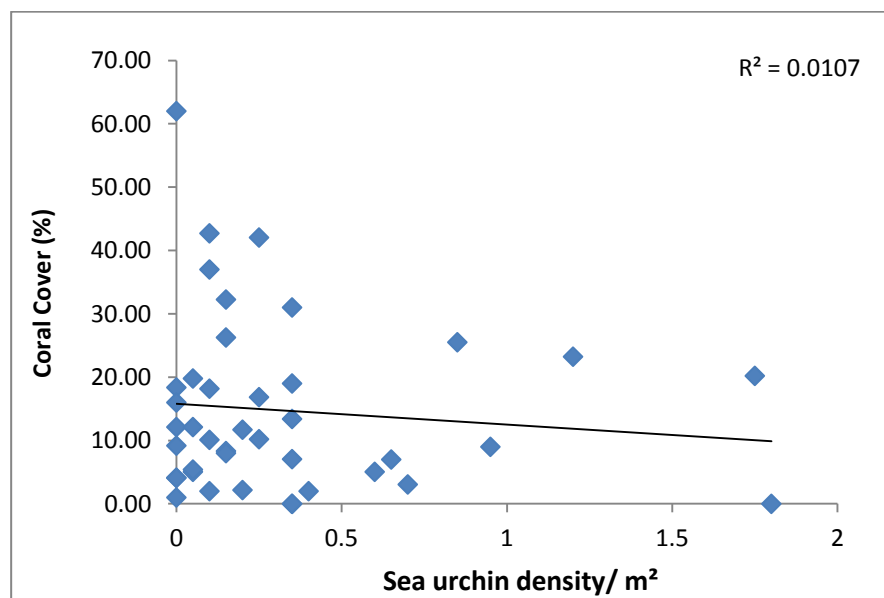


Figure 3.8: a scatter plot shows the linear relationship between sea urchin density/ $\text{m}^2$  and percentage of coral found on each transect. The  $R^2$  value is displayed on the graph.

### **3.5 Rugosity**

Rugosity was measured at each site in order to examine whether the topographic complexity and sea urchin density were related. A one way Anova analysis of rugosity showed that the values were significantly different across sites (Refer to Appendix 5). As shown in table 3.2, Kanani was the site with the highest mean rugosity at 1.57 followed by Uyombo (1.52) and Lambis (1.36). The upper and lower values with a 90% confidence level are shown and a prob>f (p) value of 0.0303 was calculated, this allows us to conclude that the difference in rugosity across sites was significant.

<b>Level</b>	<b>Number</b>	<b>Mean</b>	<b>Std Error</b>	<b>Lower 95%</b>	<b>Upper 95%</b>
Bennets	7	1.33	0.085	1.157	1.500
Coral Gardens	8	1.27	0.079	1.107	1.428
Kanani	12	1.57	0.065	1.443	1.703
Lambis	8	1.36	0.079	1.200	1.522
Uyombo	6	1.52	0.092	1.334	1.706

Table 3.2: This table shows the results from the one way Anova testing the significance of differences in Rugosity of reef across our 5 sites.

When we compare mean rugosity from table 3.2 to sea urchin density at the sites, we can note that a slight correlation seems to exist. Kanani and Uyombo show the highest mean rugosity and were the sites of highest sea urchin abundance. Lambis has the highest mean rugosity of sites inside the park and also had the highest sea urchin density of 4.25 /20m<sup>2</sup>. The reefs at Bennets and Coral Gardens had the lowest rugosity values and sea urchin density. This was plotted on to a scatter graph (figure 3.9) which shows a clear relationship between mean sea urchin density and rugosity with a regression value of 0.708. A scatter plot was created in order to note whether these two variables were related across all transects and not just for the mean values.

Figure 3.10 shows no significant relationship between rugosity and sea urchin density when we include the results from all transects. The lack of correlation on figure 3.10 suggests that the means may not be representative of the whole data set. A study with more samples may have been able to find a clearer relationship. Rugosity was also plotted against coral cover to note if an increase in rugosity was related to an increase in hard coral cover. No relationship between these two variables was found (Appendix 6).

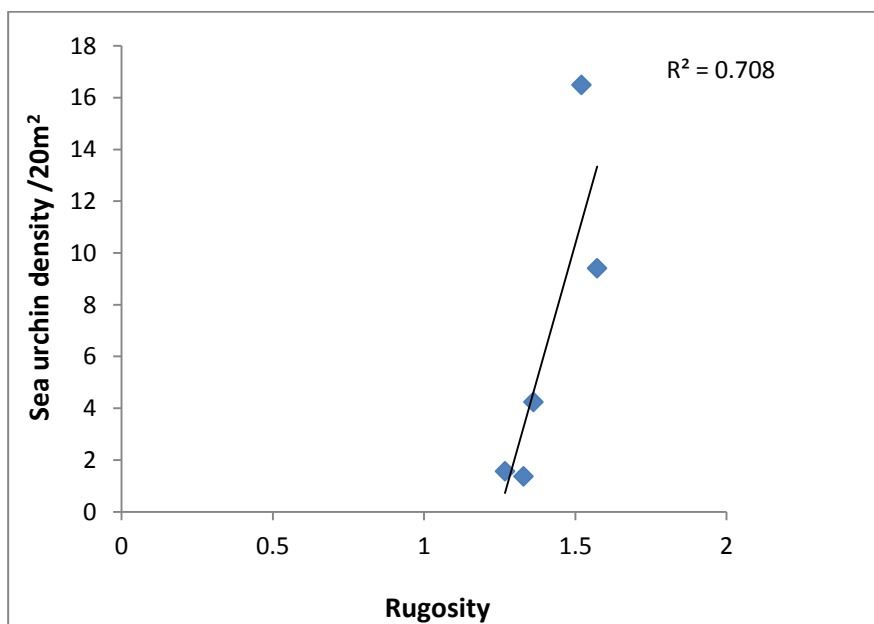


Figure 3.9: The relationship between mean rugosity and mean sea urchin density at our 5 sites. The regression value is 0.708

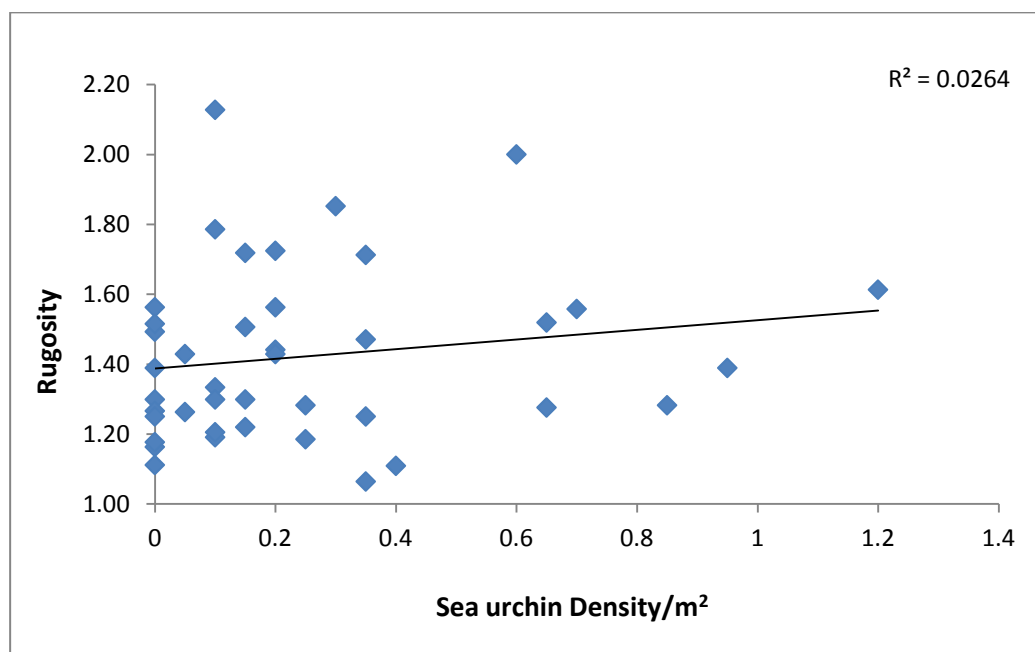


Figure 3.10: a scatter plot showing the linear relationship between sea urchin density/m<sup>2</sup> and rugosity of each transect. The  $R^2$  value is displayed on the graph.

### **3.6 Bioerosion rates**

In order to calculate bioerosion at each site we removed the sea urchin species *T.gratilla*, as this species has no effect on erosion. Bioerosion rates were calculated according to sea urchin diameters (figure 3.11) and we noted a large difference in the sea urchin size between sites. Figure 3.11 shows the sea urchin sizes at each site for all of the recorded tests. The mean sea urchin size at each site was 15.9mm at Coral Gardens, 10mm at Bennets, 10mm at Lambis, 47.6mm at Kanani, and 70.2mm at Uyombo. Differences in mean sea urchin size are related to the distribution of species as previously seen in figure 3.6. Table 3.2 shows the dominant bioerosion rates for each site calculated from total urchin numbers. Relative bioerosion was calculated to include the size of the area surveyed. Lambis and Bennets show very small relative bioerosion rates. The relative bioerosion rate at Coral Gardens is slightly larger despite not having a higher sea urchin density. This is due to the fact that one test size was measured between 60-80mm at Coral Gardens; this disproportionally increases bioerosion rates at this site. Kanani shows much higher total bioerosion; however bioerosion calculated relative to the area surveyed is 14.69 g/urchin/day. Uyombo has the highest erosion rates relative to the area surveyed at 54.95 g/urchin/day. This is a result of the large bodied *E.diadema* sea urchin being abundant.

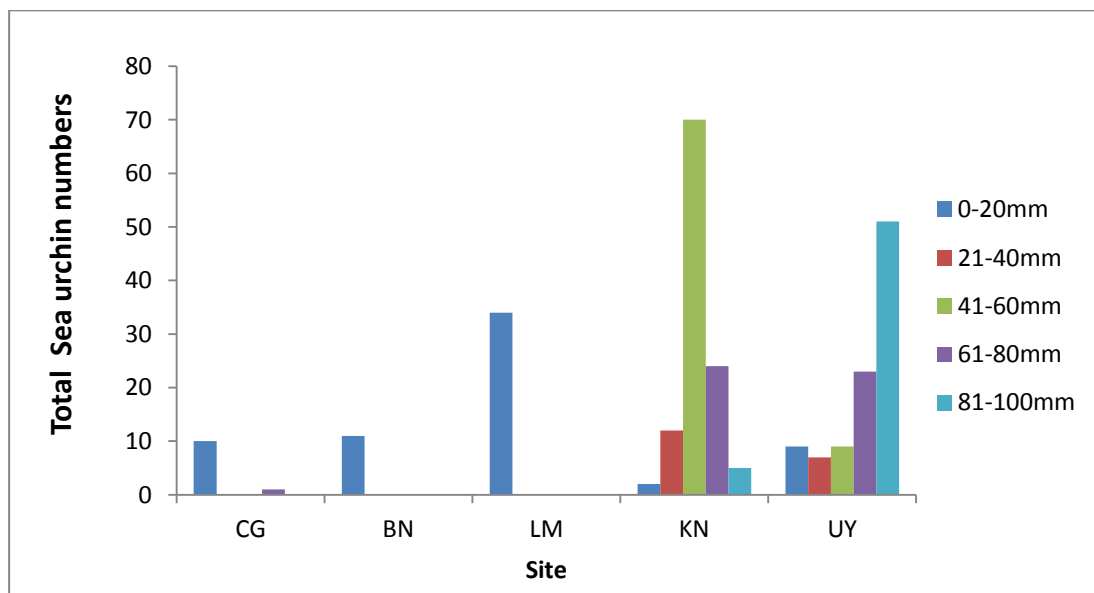


Figure 3.11: Measured test size for all sea urchins collected at each site; in size classes 0-20mm, 21-40mm, 41-60mm, 61-80mm and 81-100mm.

Site	CG	BN	LM	KN	UY
Total Bioerosion (g/urchin/day)	2.92	0.25	0.77	176.30	329.70
Transects	8	7	8	12	6
Relative Bioerosion (g/urchin/day)	0.37	0.04	0.10	14.69	54.95

Table 3.3: Total bioerosion at each site, number of transects, and relative bioerosion g/urchin/day for the size of the area surveyed.

### **3.6.1 Sea urchin abundance and Algae**

In order to investigate the relationship between sea urchin abundance and algae cover we compared sea urchin density with both turf algae and macroalgae. Macroalgae is controlled by sea urchin grazing as reported by Hutchings (1986) and Bendetti-Cecchi (1999). Whereas turf algae or bare substrate is often exposed as a result of bioerosion on reefs (McClanahan and Shafir, 1990). To observe if there was any correlation between these two variables we compared them by plotting their values on scatter graphs.

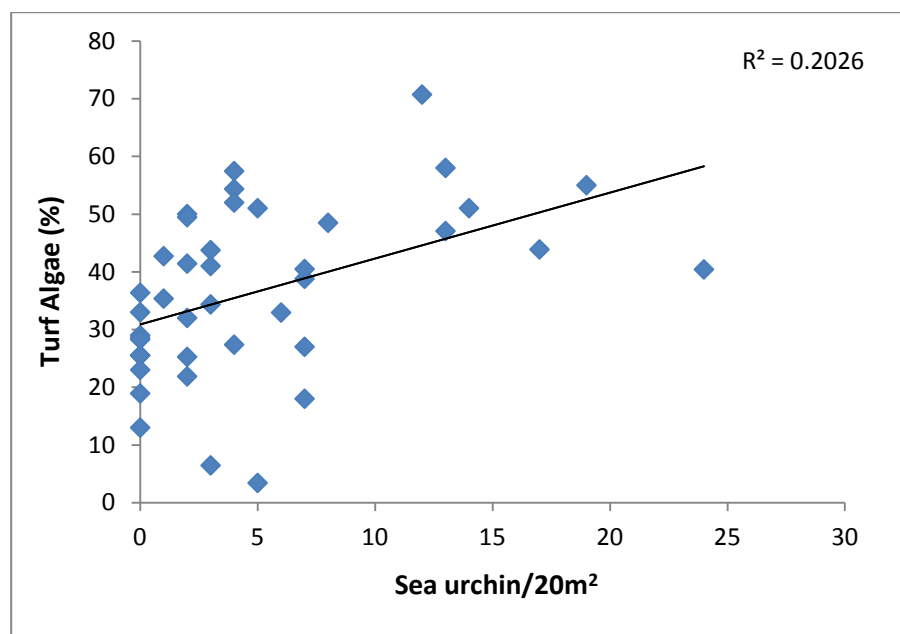


Figure 3.12: Scatter plot of sea urchin density/20m<sup>2</sup> and Turf algae cover (%) for each transect

Figure 3.12 shows a slight positive correlation between turf algae cover at each transect and sea urchin abundance. The regression is only a value of 0.2026 which implies that the relationship is not hugely significant. However, the slight positive correlation suggests that where more samples may have been taken, and potentially less limitations, a stronger relationship between these two variables may have been found

The relationship between macroalgae and sea urchin abundance was plotted into a linear regression (Figure 3.13). As you can see the figure illustrates a negative correlation, suggesting that increasing sea urchin density results in a reduction in macroalgae cover. This negative correlation would be expected, however the regression value is only 0.319, suggesting that around 30% of the difference in macroalgae cover can be attributed to sea urchin abundance.

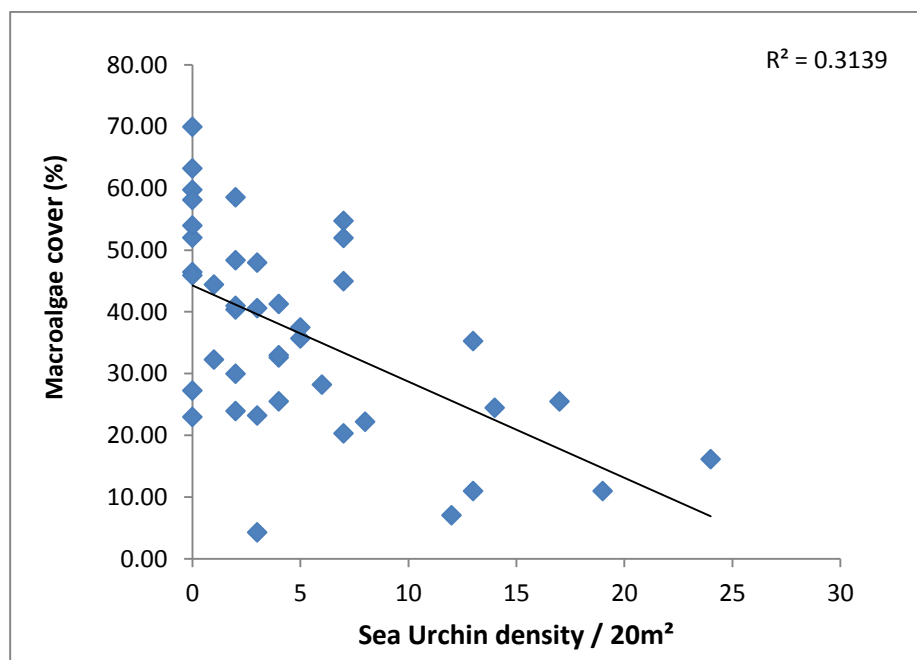


Figure 3.13: Scatter graph plotting sea urchin density/20m<sup>2</sup> against Macroalgae cover (%) for each transect.

### 3.6.2 Macroalgae species

Our results show a comparison of macroalgae species across the 5 reef sites. This comparison was produced in order to examine whether the dominance of certain algal species differs across our sites. We can note that *Halimeda*, an important calcifier, is the most abundant species found across all our sites. *Halimeda* is particularly dominant at Coral Gardens (38.85%) and Bennets (36%), decreasing across Lambis and Kanani, and contributing to only 8% of the macroalgal cover at Uyombo. *Sargassum* is found at much smaller percentages across our sites, at its highest on Kanani where the mean per transect is 6.9%. *Sargassum* is not found at all at Uyombo. *Turbinaria* is the second most abundant species of macroalgae found at Lambis at 14.6% after *Halimeda*. *Turbinaria* differs by only a few percentage across Coral Gardens, Bennets, and Uyombo, and is at its lowest at just 1% on Kanani. *Diycoia* and *Jania* were not found at all on our sites and *Padina* was found on one transect and was therefore not included in the results.

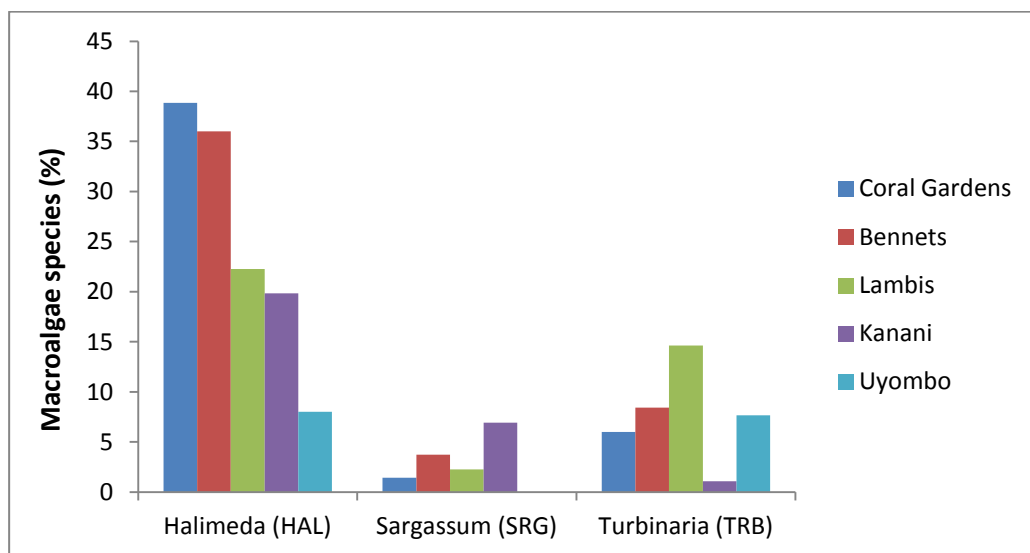


Figure 3.14: Macroalgal abundance by species at all sites. The main three abundant species were included *Halimeda*, *Sargassum* and *Turbinaria*.

### **3.7 Comparison of results within and outside the MPA**

In order to assess the effect of the marine park on our results we examined the difference in coral cover and sea urchin abundance for outside the park (2 sites) and inside (3 sites). This means we had a total of 23 transects inside and 18 outside the park. The graph (figure 3.15) shows despite large errors in the 95% confidence bars the difference in sea urchin numbers is significant. However when the effect of the MPA on coral cover was examined we noted no significant difference. In figure 3.16 we can see a slightly higher mean coral cover percentage inside the park (16.11%) compared to outside (12.89%), which would support our hypothesis that higher coral cover and lower sea urchin density would be found inside the park. However the 95% confidence bars overlap and therefore we cannot consider the difference significant.

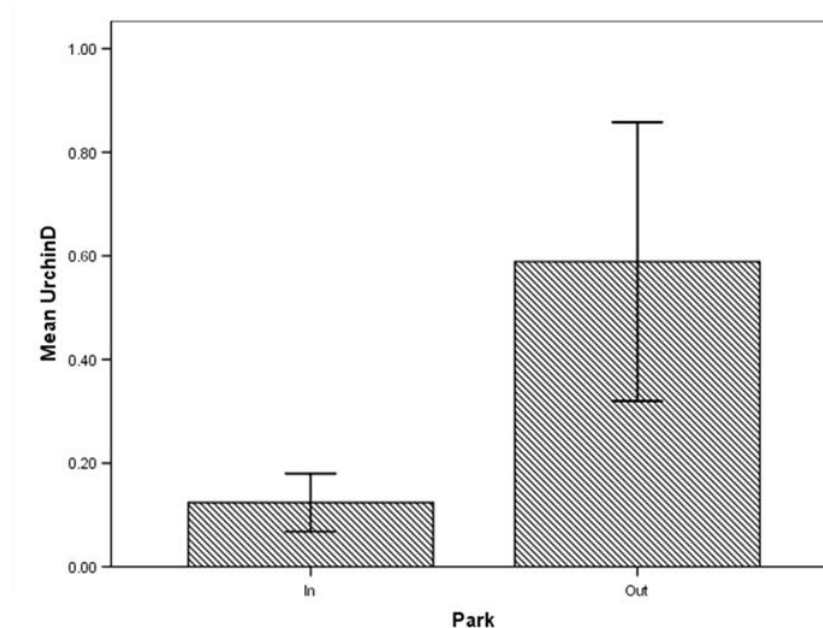


Figure 3.15: shows the mean urchin density/m<sup>2</sup> inside and outside the park. Error bars are shown to a 95% confidence interval.



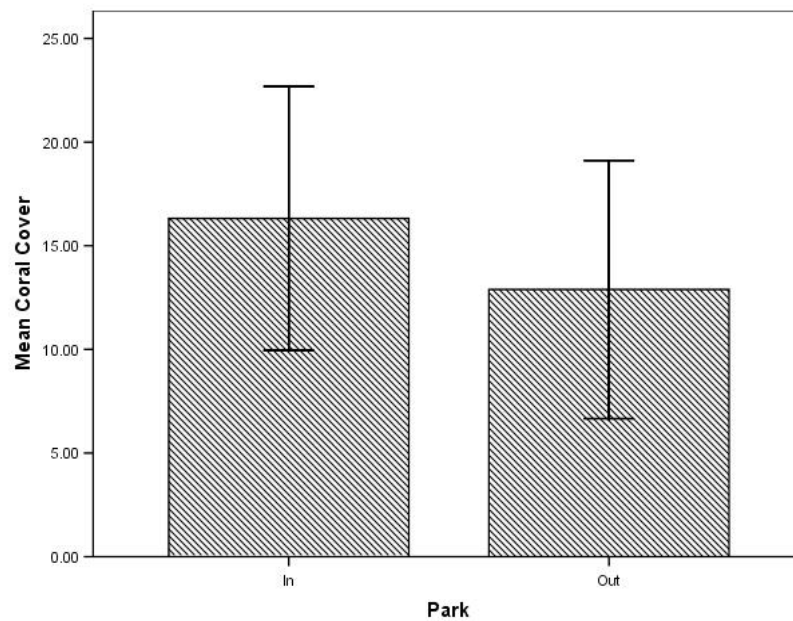


Figure 3.16 shows the mean coral cover (%) inside and outside the park. Error bars are shown to a 95% confidence interval.

### **3.7.1 Benthic cover on protected and unprotected sites**

When looking at differences in benthic cover inside and outside the MPA, we can note from figure 3.1 that Kanani and Uyombo, which are outside the park, had a higher percentage of turf algae cover at a mean of 43.20% compared to 33.37% inside the park. Additionally macroalgae percentage cover was higher (45.62%) inside the park than at the 2 sites outside (23.7%). Standard deviation is shown on the graph (figure 3.17) and shows in some cases a large variability in the data. Turf algae shows particularly high standard deviation and could bring into question the reliability of this data. Macroalgae shows higher standard deviation outside the park (11.11%), however inside the MPA results are distributed closely around the mean and show all three sites inside the park have similar percentage cover. Coralline algae is an important feature of reef ecology that promotes settlement of larvae and higher rates of accretion (Edmunds and Carpenter, 2001). Percentage cover of coralline algae is shown to be at similar levels both in and out of the MPA. Sea grass was not found at the sites within the park but this may be as a result of site selection and environmental differences rather than any impact from the marine park. Finally the graph shows that sand and rubble (defined in the methods) was higher outside the park at 8.75% as opposed to just 1.28% inside the park.

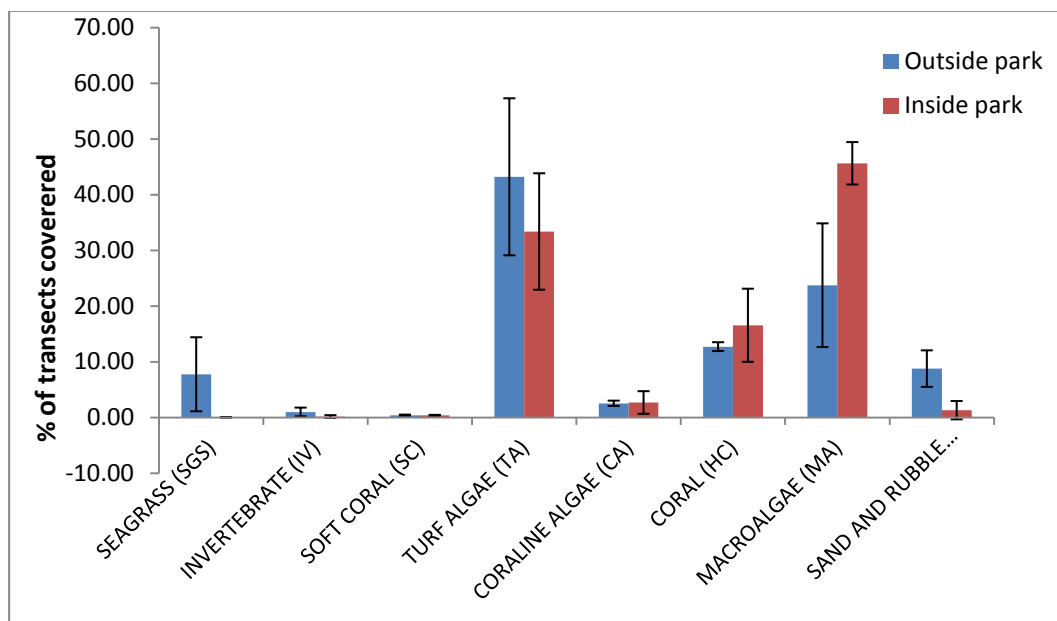


Figure 3.17: Types of benthic cover and its abundance (%) on transects outside and inside the marine park. Standard deviation for the transects is displayed as error bars.

### 3.7.2 Sea urchin species and bioerosion

Sea urchin species show an obvious difference inside the marine park compared to outside. The species *E.Molaris* was found less frequently on unprotected reefs, and even when standard deviation is included the difference is shown to be significant (Figure 3.18). *E.Mathaei* appeared both inside and outside the park but at relatively low frequencies. We can see that the dominant urchins on unprotected reefs shown in figure 3.18 are *E.diadema*, *D.setosum* and *D.savingyi*. Almost all the *E.didema* were found at Uyombo and *D.setosum* and *D.savingyi* where exclusively found at Kanani (Figure 3.6). This results in high standard deviation for all three of these species as despite both being in unprotected sites, they show almost no similarities in species distribution and bring into question the limitations of having only two protected sites in this study. Bioerosion rates as a result of sea urchin species distribution are significantly higher in our unprotected areas as was shown above in table 3.3. Sites at Kanani and Uyombo had significantly higher rates of relative bioerosion compared to our three sites within the park.

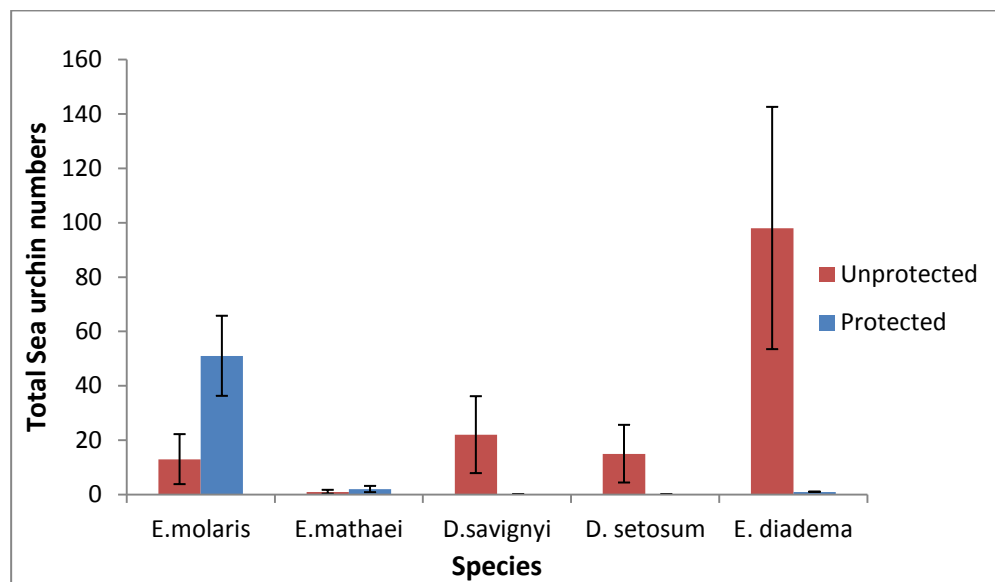


Figure 3.18: Shows the frequency of sea urchin species inside the park (protected) and outside the park (unprotected).

### **3.8 Summary of results**

	Coral Gardens	Bennets	Lambis	Kanani	Uyombo
No. of Transects	8	7	8	12	6
Mean Coral Cover (%)	16.24	23.03	9.90	13.26	12.14
Mean Rugosity	1.27	1.33	1.36	1.57	1.52
Mean Urchin Density/20m <sup>2</sup>	1.38	1.57	4.25	9.42	16.50

Table 3.4: summary of results from study including number of transects taken, coral cover (%), rugosity and sea urchin density/20m<sup>2</sup>.

## **4. Discussion**

### **4.1 Sea urchins and Coral cover**

Coral cover at the reefs investigated in this study show a very high variability between transects, and large standard deviation across sites. This makes drawing conclusions from these results difficult and inaccurate. Coral cover recorded from the 8 transects on Bennets reef had a mean coral cover of 23%, however the deviation from the mean was 21%. The uncertainty in these results may be as a result of the area chosen to lay transects, for example if areas of sandy bottom were included in the 10 meter line this could dramatically influence results. Although every effort was made to standardize habitats surveyed in some cases the discontinuous patches of coral made it near impossible to avoid all areas without corals. Our results showed that despite a significant difference in sea urchin density across the five sites coral cover was not significantly reduced by the abundance of sea urchins. Previous studies in this area by Carriero-Silva and McClanahan (2001) showed significant relationships between sea urchin numbers and coral cover. Firstly it must be considered that the number of sites and transects surveyed were not enough to provide a statistically significant relationship between sea urchins and coral cover. Secondly McClanahan and Mutere (1994) suggested that the effect of sea urchin abundance on coral reefs could be hard to separate from other environmental or biological factors. As a result this could lead us to question whether the characteristics and habitat for each site may have had an effect on this relationship. The coral cover at Uyombo may have been influenced by the tidal creek and mangrove habitat. As can be seen on figure 1 this site is positioned at the mouth of the creek and the trapping of sediments and nutrients by mangroves and sea grass beds help to provide a low nutrient system that allows coral reefs to thrive (Linton and Warner, 2003). Seagrass beds were noted surrounding coral patches at our second site outside the park, Kanani. Seagrass, much like mangroves, removes essential nutrients to avoid macroalgal blooms but provides essential nutrients and detritus that help coral growth (Harborne et al., 2006). This may help to explain why, despite higher sea urchin density and bioerosion rates at both these sites, coral cover is not significantly lower than sites within the MPA. The location of these sites and their unique habitats is a limitation when trying to make comparisons to the sites within the

MPA. However, marine parks are usually created where there is a need to protect a particularly unique environment. As a result it is often hard to compare to identical unprotected environments (Bellwood et al., 2004). Despite this there is still a need to research the sites adjacent to marine parks to understand the effect that these MPAs may have on whole ecosystems.

When comparing coral cover in relation to the marine park we did note slightly higher mean coral cover inside the park 16.11% compared to outside 12.89%, however no statistical difference was found. The results show that the dominance of coral genera varies between sites. In particular, *Porites* mass seems dominant inside the park at two of the sites. This is contrary to suggestions by McClanahan and Mutere (1994) that *Porites* mass becomes dominant on unprotected and sea urchin dominated reefs. The dominance of *Porites* mass at Coral Gardens and Bennets may be related to a mass bleaching event in 1998 which saw 50-90% coral mortality on Kenyan reefs (Lambo and Ormond, 2006). Branching corals, such as *Acropora* and *Pocillopora*, were found to have a much higher mortality rate (McClanahan et al., 2001) in contrast to *Porites* mass which has greater resistance to bleaching events (McClanahan et al., 2007). Additionally, unlike *Acropora*, *Porites* mass is not as favoured by coral eating fish, and this along with a higher resilience to warmer waters, resulted in *Porites* mass becoming a common genera on Kenyan coral reefs (Lambo and Ormond, 2006). *Acropora* is seen as a key reef building coral (Lambo and Ormond, 2006), and following the bleaching event it has been suggested that this genus has struggled to recover. On top of this, corallivory may have also limited *Acropora* recovery and it is only found at small percentages inside the park. The high percentage of *Acropora* cover at Uyombo is not expected as it usually features less on unprotected reefs where it is often collected or trampled by fishermen (Klaussen 2010; McClanahan and Mutere 1994). This may be explained by the depth of the reef at this site which would have made coral collection unlikely; additionally if fish predation does play a part in suppressing recovery of *Acropora* then outside the park reduced fish pressures may allow a higher abundance. *Echinopora* is noted at all three sites inside the park and is reported to be found on protected reefs as it is less preferred by coral eating fish according to Klaussen (2010). Reports show that recovery from the 1998 bleaching event in Watamu, Mombassa and Malindi marine parks in Kenya was 19.5% by 2001 (Lambo and Ormond, 2006). The

coral would have been expected to have recovered to a larger extent by the time of our study, but the bleaching event may have dictated the distribution of certain genera across our sites. From our results we can suggest that coral reef protection was not the only factor affecting the particular dominance of certain coral genera. Environmental factors also play a key role in coral species composition on reefs.

#### **4.2 Sea urchin grazing and algae**

Grazing from herbivores is the main factor effecting the abundance of algae and distribution of plants (Benedetti-Cecchi et al., 1999) and can also impact biogenic characteristic of habitats and therefore coral communities (Sammarco 1982; Jones and Andrew 1990). Grazers are important to remove macroalgae from reefs, providing space for juvenile coral to establish (Hutchings 1986; Jones and Andrew 1990). At Watamu the two main herbivores to consider are parrot fish and sea urchins which are competitors on reefs (McClanahan 1995). Sea urchin dominated reefs have been shown in previous studies, in both the Caribbean and Indo-Pacific to be characterized by turf algae cover (Glyn 1997; McClanahan and Shafir 1990; Edmunds and Carpenter 2001). This is where sea urchin grazing increases above a threshold and results in bioerosion of the reef substrate leaving turf algae and often reducing coral cover (Hereu et al., 2005). This is in contrast to reefs dominated by herbivorous fish that have been shown by McClanahan (1995) to result in higher calcium carbonate deposition and thus results in a more productive reef system than one dominated by sea urchins. In our study evidence of grazing was noted by differences in the benthic cover at our sites. Macroalgae was found at higher percentages across Coral Gardens, Bennets and Lambis which are all sites within the marine park. This implies that less grazers or lower intensity of grazing occurred within the marine park resulting in an increase in the abundance of fleshy algal (Benndetti-Cecchi et al., 1999). High macroalgae cover has previously been linked to sites with notably smaller sea urchin densities (Edmunds and Carpenter, 2001), and in our study correlated with sites where sea urchins populations were sparse. In order to examine whether sea urchin abundance and macroalgal expanse was directly related we correlated the two variables in a linear regression. The regression analysis provided an insignificant regression (0.3139) but it does partially explain the difference in macroalgae cover. This study did not assess fish populations

and therefore the amount of grazing attributed to fish populations cannot be quantified. High macroalgae cover on coral reefs is common where sea urchin densities are low and can lead to coral mortality as algae overgrowth inhibits coral recruitment (Sammarco 1982). At out two sites outside the park macroalgae cover was considerably less, especially at Uyombo where it was just 15.87% of the benthic cover. This difference could be attributed to sea urchin grazing at high densities, such as at Uyombo where density was 16.50 per 20m<sup>2</sup>. Turf algae cover also shows differences between sites, where Lambis and Uyombo have the higher percentages of bare substrate. This could be as a result of bioerosion, but when correlated a regression of only 0.234 was found between turf algae and sea urchin density.

Our results suggest that if we are examining dominant herbivores at each site Coral Gardens, Lambis and Bennets are all predominantly grazed by herbivorous fish as sea urchin densities were very low. At Uyombo sea urchin grazing is clearly dominant and high densities of the large bodied *E.didema* species were noted. Macroalgae cover at sites inside the park was relatively high which may partly reflect the lower sea urchin abundance. However herbivorous fish would be expected to graze macroalgae inside the park (McClanahan and Mutere 1994). It is important to consider that additional factors may be affecting grazing intensity, and without an assessment of fish in the area it is hard to suggest the reason for this. One aspect that may be important to consider is poaching from within the marine park at night noted by McClanahan (1994) and during this study (Author *pers. obs*). The illegal removal of fish, especially the larger individuals from the park, could impact grazing intensity. At sites where sea urchin density is already low the removal of finfish on top of this could lead to increased algal growth (McClanahan, 1995). However it is important to note that other factors such as waves and currents can affect fleshy algal abundance (McClanahan et al., 2001) and the population size of macroalgae may not be related solely to grazing intensity. The reef at Kanani had a relatively high sea urchin density (9.42/20m<sup>2</sup>) and the species *T.gratilla* was frequently recorded. This sea urchin species has little impact on benthic cover as it does not contribute to reef erosion (Silahooy, 2013) and the impact of even large densities is insufficient to substantially degrade the reef structure (Jones and Andrew 1990). This may explain why despite high sea urchin density at Kanani the percentage cover of bare substrate (classed as turf algae) was lower than expected. In order to see if



numbers of *T.gratilla* effected the correlation of sea urchin abundance and turf algae percentage the regression was recalculated. However even when the site at Kanani is removed no further correlation can be noted between turf algae and sea urchin density. This would imply that although relationships between sea urchin density and both macro and turf algae can be observed, the number of sites and transects may not be sufficient to provide any statistically significant correlation. Also it suggests that a number of other factors, not studied in this project, may affect grazing rates and algae cover. Our results show a clear difference in grazing characteristics at our sites. This agrees with McClanahan and Mutere (1994) who also found that protected reefs are predominantly grazed by finfish whereas outside this protection, such as at the Uyombo reefs, they become dominated by sea urchins.

Grazers can also dictate the types of macroalgae that is found to be dominant on coral reefs (McClanahan and Mutere, 1994). Our study showed that *Halimeda* was the most abundant macroalgae speices found. *Halimeda* is responsible for reef accretion and alongside coralline algae is an important calcifier on reefs (Adey, 1998). *Halimeda* was found in higher abundance at the reefs inside the marine park, namely Coral Gardens, Lambis and Bennets reef, and therefore suggests higher reef productivity and accretion at these sites. *Sargassum* has previously been found to be more abundant on unprotected reefs (McClanahan et al., 1999). However in our study we noted that abundance was not significantly different within and outside the park despite Kanani showing the highest percentage cover. *Turbinaria* is most common at Uyombo where the reef was more exposed to ocean processes than at other sites as this macroalgae species flourishes in high water turbidity (Stiger and Payri, 1999). The most commonly noted *Turbinaria* species, *Turbinaria ornate*, also shows high resistance to predation (Stiger and Payri, 1999) and therefore would be able to survive at Uyombo despite high sea urchin density and grazing pressures. Gleason (1996) showed that *Turbinaria ornata* can reduce coral recruitment on reefs and therefore may have a negative impact on reef heath where it is able to establish at high densities.

### **4.3 Bioerosion**

Bioerosion is an important process on coral reefs and benefits reef growth and accretion (1997). However where high densities of sea urchins are present grazing is suggested to effect the establishment of corals and possibly damage species, especially juveniles (Edmunds and Carpenter 2001). Bioerosion in our study varied significantly between sites, at Coral Gardens, Bennets and Lambis bioerosion was found to be between 0.04 and 0.37 g/urchin/day. This is significantly lower than at Kanani and Uyombo where relative bioerosion was 14.69 and 54.95 g/urchin/day respectively. Our bioerosion calculation only accounts for erosion by sea urchins and excludes any impact of internal bioeroders and bioerosion by fish. Additionally sea urchin erosion was calculated using body size as opposed to species specific rates. This is accurate for most species where erosion rate is dependent on size. However in the case of *Diadema savingyi* and *Diadema setosum*, which have similar test sizes (44.29 and 40mm respectively), the accuracy of this bioerosion calculation is limited, as Carriero- Silva and McClanahan (2001) found that *D. savingyi* has an erosion rate that is a third of the species *D.setosum*. This is a limitation in our calculation however it is not significant enough to discredit our bioerosion results. Bioerosion on Uyombo is notably higher than at any other site, this is due to the abundance of *E.diadema*. Despite no significant difference in coral cover at Uyombo the amount of rubble and sand recorded was highest at 11.07% and broken dead coral was frequently observed. This reinforces the results from Carriero- Silva and McClanahan (2001) who found that *E.diadema* are the most important bioeroders on unprotected reefs as despite a similar gut turnover rate to *D. savingyi*, the larger body size of the *E.diadema* results in a much higher bioerosion rate. *E.mathaei* is suggested to have high rates of bioerosion relative to their body size (Bak, 1994) and despite reports of their dominance on Kenyan reefs (McClanahan and Shafir, 1990) we found a low abundance of this species at all our sites.

Topographic complexity or rugosity is suggested to decrease where sea urchins are abundant, as a result of bioerosion (McClanahan and Shafir, 1990). However our results found a positive relationship between mean sea urchin abundance and mean rugosity at our five sites. This can be explained by the fact that high reef rugosity is beneficial to sea urchin populations providing crevices in the reef for shelter (McClanahan, 1998; Herue et al., 2005). However as discussed in our results where rugosity values are correlated with sea urchin abundance for each transect no notable correlation is found,

implying that data is spread far from the mean and the results are not significant. Our results suggest that although higher bioerosion rates were found outside the park, mean rugosity was still high at these sites suggesting that bioerosion by sea urchin is not significantly degrading reef structure. The significantly higher bioerosion rates found outside the park agree with Carreiro-Silva and McClanahan (2001) who found sea urchin erosion was 20 times higher outside MPAs. This suggests that marine parks play an important role in the distribution and abundance of sea urchins on coral reefs and therefore bioerosion rates.

#### **4.4 Marine park effect on sea urchin populations**

The main statistically significant result from our study was the difference in sea urchin abundance at sites inside and outside the marine park. As well as a difference in the sea urchin numbers between sites we also noted a differentiation in species distribution and size inside and outside the park. We suggest that the main reason for this difference can be attributed to fishing pressures outside the park and has previously been noted to be an important influence of sea urchin abundance in east African lagoons (McClanahan 1995; 1998; Carreiro-Silva and McClanahan 2001). A study on sea urchin predation conducted by McClanahan and Shafir (1990) reported that on protected reefs the survival time of sea urchins in their study was  $0.40 \pm 0.8$  days compared to  $1.68 \pm 0.07$  on unprotected reefs. The observed fish frequency also varied greatly with around 3.6 times more total fish being found on protected reefs. McClanahan and Shafir (1990) found that sea urchin densities were correlated strongly with the abundance of predators and predation intensity. Within the marine park we found the dominant sea urchin was *E.molaris* at small test sizes below 20mm. This species was dominant at all three sites within the park and all recorded sea urchin of this species were sheltered in crevices along the substrate. However outside the park we noted an obvious difference in sea urchin assemblage and a larger variety of sea urchin species was found compared to the protected reefs. Outside of the marine park the recorded size of the sea urchin was much larger, up to a mean of approximately 7 times greater on Uyombo than at the protected sites. At Uyombo the sea urchin *E.diadema* (which reached largest body sizes of 74.64mm) was predominantly found in the open on the reef substrate (Example Appendix 7). When looking at the distribution of species, almost all the *E.didema*

where found at Uyombo and *D.setosum* and *D.savingyi* where almost exclusively found at Kanani. This could suggest that the difference in species distribution outside the park was more as a result of individual site characteristics. However none of these three species were found at any significant frequency inside the park in this study or in previous work by Carreiro-Silva and McClanahan (2001). A number of other studies also found that on unprotected reefs the species diversity of sea urchins was much greater, whereas on protected reefs *E.mathaie* and *E.molaris* were dominant as a result of their small body size (McClanahan and Shafir 1990; McClanahan et al., 1999; Peyrot-Clausade et al., 2000). As a result we can conclude that outside the marine park less predation from fishing pressures results in a wider variation in sea urchin species.

An additional aspect of fishing on unprotected sites is the selective removal of larger fish by fishermen, resulting in decreasing predator size which allows sea urchin populations to increase dramatically (McClanahan 1995). This is because as predator size falls, sea urchins rapidly reach a size where they are too large to be consumed by these smaller predators (Hereu et al., 2051). The most effective predators of urchins are medium to large size fish (Sala 1997) that can halt growth of a sea urchin population by consuming the juveniles (Hereu) et al., 2005). As sea urchin sizes increase, the need for shelter becomes less important as was noted at Uyombo (Hereu et al 2005). This is important as the removal of finfish allows the population of sea urchins to expand and the subsequent competitive exclusion of herbivorous fish can seriously hinder their potential population recovery (McClanahan et al., 1996). The dominance of the *E. diadema* species at Uyombo is a hindrance to fish populations; additionally Carreiro-Silva and McClanahan (2001) suggested that *E.diadema* is the most predator tolerant species, linked to its large body size. This top down perspective looks at the importance of fish as predators on coral reefs and the extent to which individual species can control sea urchin populations. The main species responsible for sea urchin predation was the red lined triggerfish (McClanahan and Shafir, 1990). Although there are other sea urchin predators, such as wrasse, they are less effective predators and the removal of triggerfish may have a disproportionate effect on sea urchin density (McClanahan and Shafir, 1990). The loss or removal of key fish species as a result of human involvement can have cascading effects at a number of levels (McClanahan and Shafir 1990. McClanahan and Mutere (1994) found that newly protected reefs have shown

reductions in sea urchin abundance and increasing coral cover following the formation of MPAs. MPAs are therefore important tools in controlling sea urchin populations and monitoring their impact on coral reef health.

#### **4.5 Future of coral reef management**

Results suggest the need to consider that uncontrolled levels of fishing can dramatically alter coral reef ecosystems, in particular echinoid populations, altering benthic composition and biodiversity (Hereu et al., 2005; Carreiro-Silva and McClanahan, 2001). Ultimately a sustainable level of fishing needs to be achieved in order to preserve reef ecosystems and species diversity. Marine park areas begin to solve these problems as these no-take areas provide patches of preserved ecosystem. However where MPA's only cover a small area their effectiveness may be limited (Bellwood et al., 2004). There is a growing need to educate fishermen in the need for resource preservation and sustainable fishing methods (Muthaiga and Maina 2001). This is increasingly important as coastal areas are encountering both rapidly growing populations and the move away from subsistence fishing to industrial scale fisheries (Johannson, 2007). If fish are removed from a reef up to the point where a shift to an algal dominated reef occurs, the subsequent decline in coral cover could reduce fish stocks even more in the long term (McClanahan 1995). Communicating this idea to coastal populations is key to developing successful management of coral reef fisheries (Muthiga and Maina, 2003). As well as this, the enforcement of reef protection even where marine parks are established is vital. Poaching at night in Watamu Marine Park and the illegal harvesting of fish (Author *pers. obs.*) could have serious detrimental impacts and reinforces the need for improvements in reef management. Trying to control fish stocks and maintain key species is complicated and as suggested by McClanahan (1995) putting a control on stocks is unlikely to satisfy large fishing populations. The selective protection of key stone species, such as the red-lined triggerfish in our study, is increasingly difficult to implement and monitor. Furthermore the ecosystem interactions on coral reefs are complex (McClanahan, 1995) and protection of individual species could significantly affect a number of relationships within this habitat. Economic alternatives are required, especially in developing countries, to help sustain coral reefs (McClanahan et al., 1996). This has to lead to the

suggestion that sea urchins could be directly removed from reefs by local fisherman as an economic alternative. The effect of removing sea urchins from coral reefs was investigated by McClanahan et al., (1996) on Kenya coral reefs. They found during a long term study that the removal of echinoids showed benefits for fish stocks as they were no longer in competition for algal grazing. However subsequent algae abundance as a result of reduced sea urchins grazing resulted in a long term reduction in coral cover. Despite a reduction in coral cover, McClanahan et al., (1996) suggested that sea urchin removal could be useful for reef conservation if more studies could be done to note the large scale effect of this interference on the whole reef ecosystem.

## **5. Conclusion**

### **5.1 Limitations**

Our study had a number of limitations, the most important being the lack of sites surveyed. Only two sites were surveyed outside the park and three within, which means that individual site characteristics could have explained some of the variance in results. A comprehensive study should have included more sites and possibly examined reefs at increasing distance from the marine park. Our number of surveyed transects was limited and therefore finding statistical significance from this small data pool was difficult. Our understanding of coral reefs through this study is greatly limited by the absence of fish stock and predation data in our study. Collecting data on fish was not possible as a result of a lack of time and resources. Detailed data of fish abundance by McClanahan and Shafir (1990) provided a comparison tool from previous research but was not specific to our exact sites. Despite these limitations the study did show the main trends and patterns associated with sea urchin distribution and bioerosion across the Watamu lagoon. Further study where more time and resources are available could produce a more in depth study of the area including a detailed report on fishing pressures and its impact on fish populations.

Another significant limitation of our study and many previous studies on sea urchin abundance is the nocturnal behaviour of sea urchins. During this project we decided to conduct a pilot study on one of the sites, completing a night time survey in order to note any differences in sea urchin assemblage. At Coral Gardens eight transects were surveyed and we noted a much higher sea urchin density at night compared to the original survey. Sea urchin density was over three times larger and a greater range of species was found, including those not previously recorded such as *E.diadema* and *D. savingyi* (See Appendix 8 for full results). Jones and Andrew (1990) also found that *diadema* species more commonly found refuge from predation during the day and were more active during the night. The nocturnal behaviour of sea urchin species could suggest significant limitations of studies where only day time abundance is recorded. Future study of sea urchin abundance and bioerosion on reefs needs to consider the need for nocturnal surveys in order to provide a more accurate representation of sea urchin populations.

## **5.2 Conclusion**

A review of coral reef health by Bellwood et al., (2004) suggested that: “ ecological symptoms of reef collapse include: 1) A shift to echinoid dominated herbivory, 2) Reduced fish stocks 3) Destructive overgrazing and bioerosion by food limited sea urchins and 4) Reduced coral recruitment”. To some extent the points provided by Bellwood et al., (2004) can be used to assess the health of reefs in our study:

1) Our two unprotected sites had high sea urchin densities and therefore we can conclude grazing was almost exclusively echinoid dominated. This is in contrast to sites inside the park that had very low sea urchin densities that suggest grazing by fish may be more dominant.

2) As found by our results and by McClanahan and Shafir (1990), predator abundance is a controlling factor of sea urchin assemblage. Despite not recording fish abundance in this study, the distribution of sea urchin species across our sites allows us, with reasonable confidence, to assume that fish stocks outside the MPA are lower. This can also be reinforced by values of fish abundance found by McClanahan and Shafir (1990).

3) Relative bioerosion rates were significantly higher at Uyombo and Kanani than at any protected sites.

4) Mean coral cover outside the park was lower than at the reefs inside the MPA; however no statistically significant difference was found in coral cover between sites. Additionally coral recruitment, which provides a more long term method of monitoring coral reefs, was not investigated in our study.

Using these four parameters of reef health suggested from Bellwood et al., (2004), and examining our results, we can note that at sites outside the park almost all of these factors are observed. This implies that reef health outside of the marine park may be lower than that within the protected areas. Additionally the presence of weakly calcifying macroalgae species on unprotected reefs compared to *Halimeda* abundance inside the park also suggests declining reef productivity outside MPAs. These factors all suggest that reef health outside the park is declining and in the long term a reduction in coral cover may subsequently result from continued sea urchin dominance.



Our results highlight the importance of marine management in controlling the distribution of sea urchin species on coral reefs. Sea urchins have an important role on coral reefs and in a natural system their population would be controlled by predation. Fishing pressures from coastal populations disrupt this natural control on sea urchin populations. This allows high densities of sea urchins to establish which affects both the composition and productivity of reefs. The complex nature of coral reef systems makes it hard to predict the impact of disturbances on whole ecosystems (Bellwood et al., 2004). Further study is vital to try and predict coral reef responses to changing conditions in the future (Hoegh- Guldberg, 1999). Ultimately marine parks preserve key ecological functions of the reef system and may allow them to better overcome disturbances as a result of increased resilience. Small marine parks although effective on a local scale, may not be enough to preserve coral reefs from increasing natural and anthropogenic pressures. Undeniably there is a need for a better understanding of coral reef interactions and responses to disturbances, leading to the development and improvement of coral reef management across the globe.

## Appendices

### Appendix 1

Location	Transect	Total Urchins (/20m <sup>2</sup> )	Urchin Density (/m <sup>2</sup> )	Rugosity	Coral Cover (%)
Coral Gardens	1	0	0	1.16	4.12
Coral Gardens	2	0	0	1.49	12.12
Coral Gardens	3	2	0.1	1.19	18.18
Coral Gardens	4	2	0.1	1.33	42.71
Coral Gardens	5	0	0	1.18	18.37
Coral Gardens	6	1	0.05	1.43	12.12
Coral Gardens	7	0	0	1.30	9.18
Coral Gardens	8	7	0.35	1.06	13.4
Bennets	1	2	0.1	1.20	37.00
Bennets	2	7	0.35	1.47	31.00
Bennets	3	0	0	1.11	4.08
Bennets	4	0	0	1.39	16.00
Bennets	5	2	0.1	1.30	10.10
Bennets	6	0	0	1.27	1.00
Bennets	7	0	0	1.56	62.00
Lambis	1	3	0.15	1.22	8.33
Lambis	2	5	0.25	1.28	10.20
Lambis	3	1	0.05	1.26	19.79
Lambis	4	7	0.35	1.25	0.00

Lambis	5	4	0.2	1.44	11.70
Lambis	6	3	0.15	1.30	8.00
Lambis	7	4	0.2	1.43	2.17
Lambis	8	7	0.35	1.71	19.00
Kanani	1	3	0.15	1.72	32.26
Kanani	2	3	0.15	1.51	26.26
Kanani	3	5	0.25	1.72	16.84
Kanani	4	7	0.35	1.85	7.06
Kanani	5	1	0.05	1.52	5.05
Kanani	6	2	0.1	2.13	2.00
Kanani	7	1	0.05	1.25	5.41
Kanani	8	5	0.25	1.18	42.05
Kanani	9	8	0.4	1.56	2.00
Kanani	10	35	1.75	1.11	20.20
Kanani	11	7	0.35	1.79	0.00
Kanani	12	36	1.8	1.52	0.00
Uyombo	1	12	0.6	2.00	5.05
Uyombo	2	17	0.85	1.28	25.51
Uyombo	3	14	0.7	1.56	3.06
Uyombo	4	13	0.65	1.28	7.00
Uyombo	5	19	0.95	1.39	9.00
Uyombo	6	24	1.2	1.61	23.23

Appendix 1: Raw data for sea urchin density (/m<sup>2</sup> and /20m<sup>2</sup>), coral cover (%), and rugosity collected at each transect

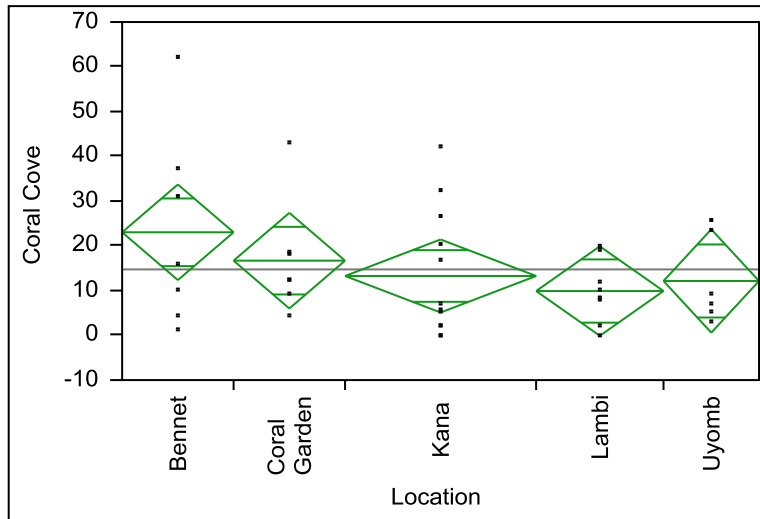
**Appendix 2 Coral genera**

Coral genera	Coral gardens	Bennets	Lambis	Kanani	Uyombo
Acanthastrea (ACA)	0.149	0.000	1.000	0.000	0.000
Acropora (ACR)	2.754	1.001	1.125	0.252	5.059
Astreopora (AST)	0.000	0.000	0.000	1.347	0.170
Coscinaraea (COS)	0.000	0.144	0.000	0.000	0.000
Cyphastrea (CYP)	0.146	0.000	0.000	0.000	0.000
Echinophyllia (EPH)	0.000	0.000	0.125	0.000	0.000
Echinopora (ECH)	1.020	7.147	2.000	0.196	0.168
Favia (FAV)	0.433	0.720	0.375	0.168	0.335
Favites (FAT)	0.144	0.000	0.250	0.000	0.000
Fungia (FUN)	0.000	0.000	0.000	0.291	0.000
Galaxea (GAL)	0.735	0.146	0.750	0.083	0.673
Gardineroseris (GAR)	0.436	0.000	0.125	0.000	0.000
Goniastrea (GON)	0.000	0.143	0.375	0.000	0.000
Hydnophora (HYD)	0.000	0.000	1.375	0.000	0.000
Lobophyllia (LOB)	0.000	0.146	0.000	0.000	0.000
Pavona (PAV)	0.149	0.146	0.000	2.678	0.000
Platygyra (PLA)	1.160	0.286	0.375	0.000	0.680
Plesiastrea (PLE)	0.000	0.289	1.000	0.084	0.000
Pocillopora (POC)	0.144	0.000	0.000	1.195	4.549
Porites (POR)	9.417	12.859	0.750	3.481	0.507
Porites Branching (PORBRA)	0.000	0.000	0.000	3.388	0.000
Psammocora (PSM)	0.000	0.000	0.000	0.098	0.000

Appendix 2: Coral genera abundance (%) for each site

### Appendix 3- Coral cover One way Anova test

#### Oneway Analysis of Coral Cover By Location



#### Oneway Anova

##### Summary of Fit

Rsquare	0.100852
Adj Rsquare	-0.00191
Root Mean Square Error	13.92603
Mean of Response	14.72875
Observations (or Sum Wgts)	40

##### Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Location	4	761.3318	190.333	0.9814	0.4303
Error	35	6787.6990	193.934		
C. Total	39	7549.0308			

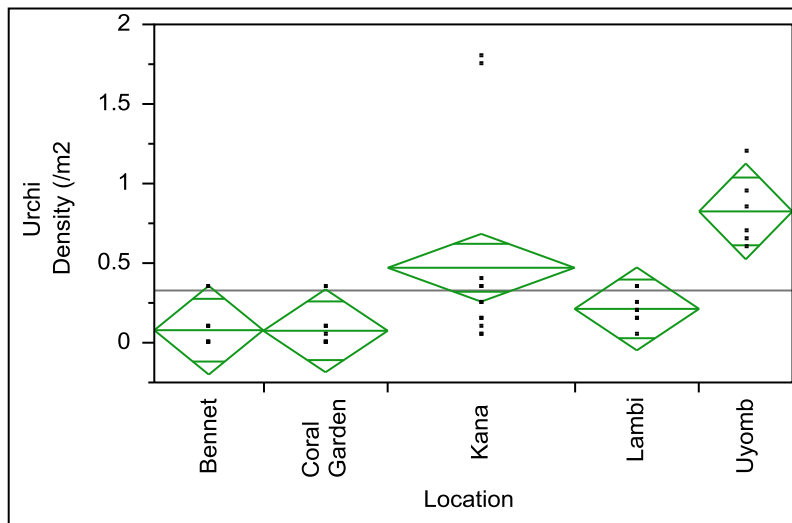
##### Means for Oneway Anova

Level	Number	Mean	Std Error	Lower 95%	Upper 95%
Bennets	7	23.0257	5.2635	12.34	33.711
Coral Gardens	7	16.6857	5.2635	6.00	27.371
Kanani	12	13.2608	4.0201	5.10	21.422
Lambis	8	9.8988	4.9236	-0.0967	19.894
Uyombo	6	12.1417	5.6853	0.60	23.683

Std Error uses a pooled estimate of error variance

## Appendix 4- Sea urchin One way Anova test

### Oneway Analysis of Urchin Density (/m2) By Location



### Oneway Anova

#### Summary of Fit

Rsquare	0.368843
Adj Rsquare	0.298714
Root Mean Square Error	0.363588
Mean of Response	0.328049
Observations (or Sum Wgts)	41

#### Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Location	4	2.7811665	0.695292	5.2595	0.0019*
Error	36	4.7590774	0.132197		
C. Total	40	7.5402439			

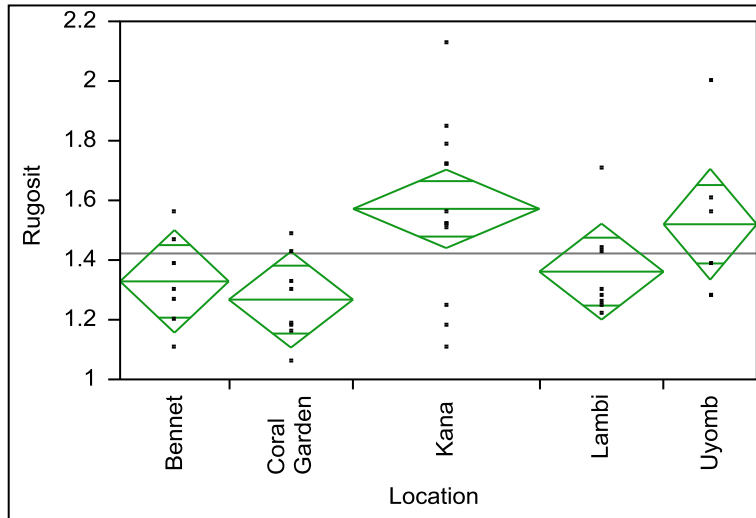
#### Means for Oneway Anova

Level	Number	Mean	Std Error	Lower 95%	Upper 95%
Bennets	7	0.078571	0.13742	-0.2001	0.3573
Coral Gardens	8	0.075000	0.12855	-0.1857	0.3357
Kanani	12	0.470833	0.10496	0.2580	0.6837
Lambis	8	0.212500	0.12855	-0.0482	0.4732
Uyombo	6	0.825000	0.14843	0.5240	1.1260

Std Error uses a pooled estimate of error variance

## Appendix 5 – Rugosity One way Anova test

### Oneway Analysis of Rugosity By Location



### Oneway Anova

#### Summary of Fit

Rsquare	0.251151
Adj Rsquare	0.167946
Root Mean Square Error	0.224406
Mean of Response	1.422195
Observations (or Sum Wgts)	41

#### Analysis of Variance

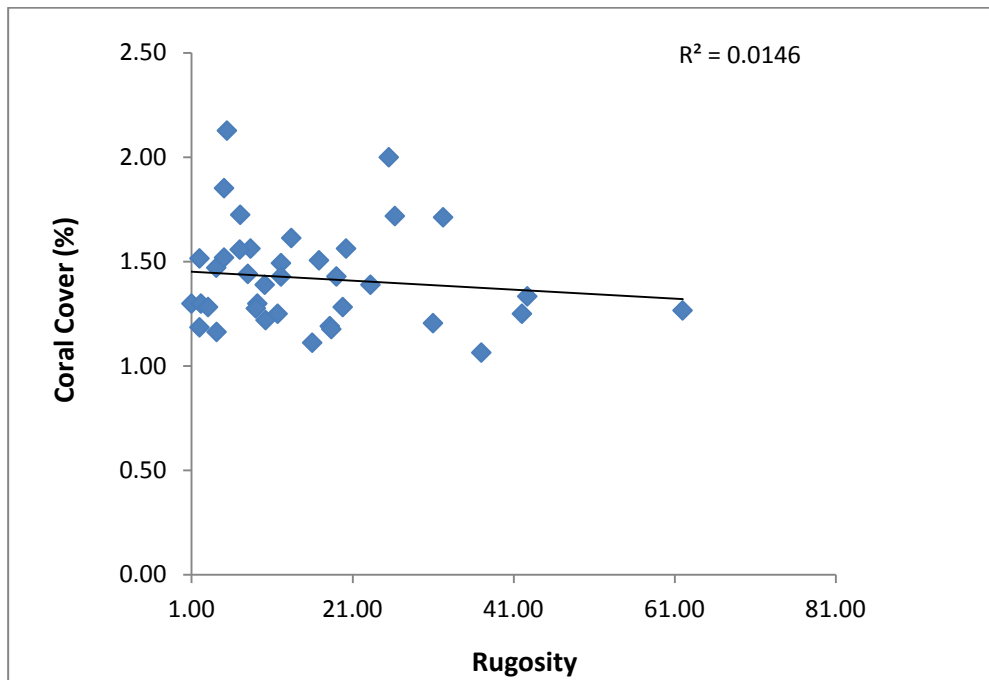
Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Location	4	0.6080126	0.152003	3.0184	0.0303*
Error	36	1.8128899	0.050358		
C. Total	40	2.4209024			

#### Means for Oneway Anova

Level	Number	Mean	Std Error	Lower 95%	Upper 95%
Bennets	7	1.32857	0.08482	1.1566	1.5006
Coral Gardens	8	1.26750	0.07934	1.1066	1.4284
Kanani	12	1.57167	0.06478	1.4403	1.7030
Lambis	8	1.36125	0.07934	1.2003	1.5222
Uyombo	6	1.52000	0.09161	1.3342	1.7058

Std Error uses a pooled estimate of error variance

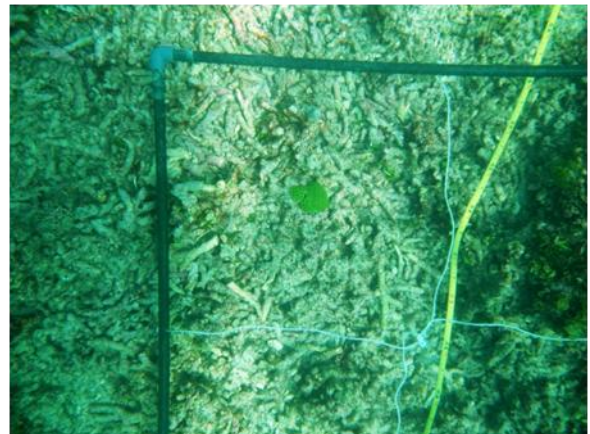
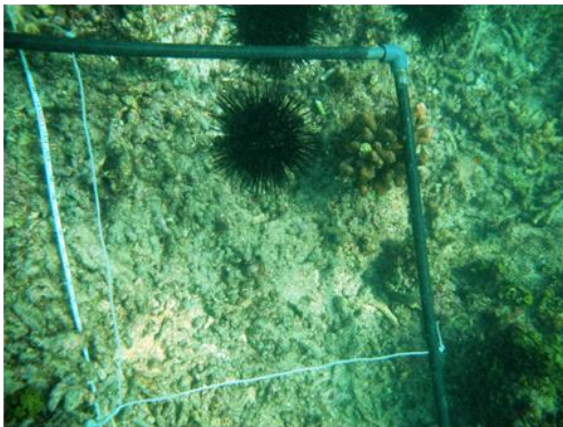
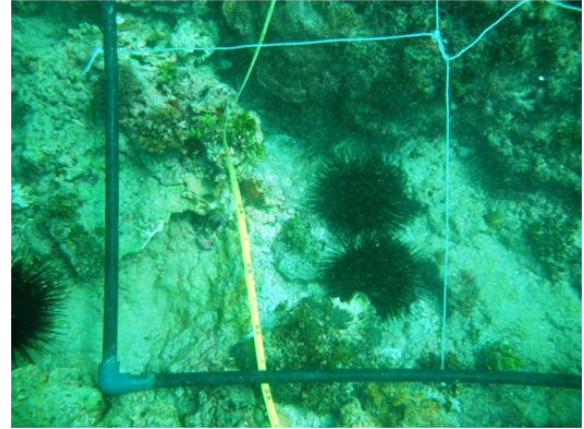
### Appendix 6- Rugosity and Coral cover relationship



Appendix 6: Linear relationship of Rugosity and coral cover (%). Regression value of 0.0146



**Appendix 7- Photos of reef characteristic at Uyombo**

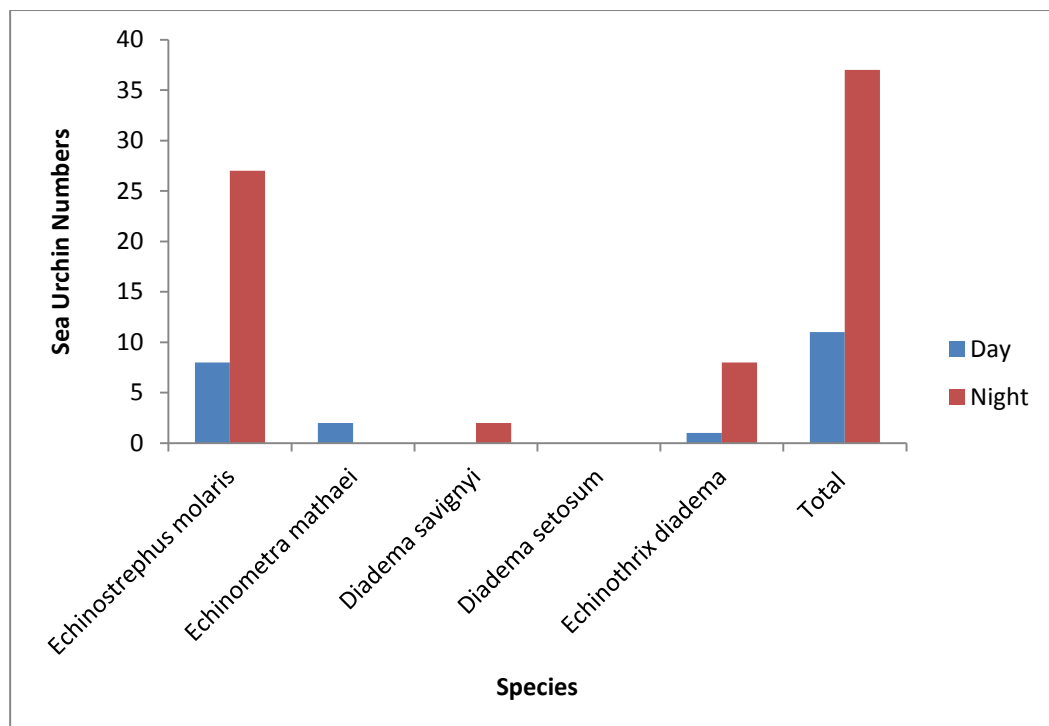


Appendix 7: Photos from Uyombo showing examples of sea urchin *E. diadema* noted on the reef substrate and rubble/ dead coral.

### Appendix 8- Results from Coral Gardens night survey

Species	Day	Night
<i>Echinostrephus molaris</i>	8	27
<i>Echinometra mathaei</i>	2	0
<i>Diadema savignyi</i>	0	2
<i>Diadema setosum</i>	0	0
<i>Echinothrix diadema</i>	1	8
<b>Total</b>	<b>11</b>	<b>37</b>

Appendix 8: Results of sea urchin species abundance at night and during the day



Appendix 8.1: Graph showing the different sea urchin species recorded at Coral Gardens when surveyed in the day (Blues) and at night (Red).

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