Fish for the future

An update of the Department of Fisheries, Western Australia, Invertebrate and Reef Health Research and Monitoring at Cocos (Keeling) Islands

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Contents

1.0 Executive Summary

The Cocos (Keeling) Islands are an Australian External Territory located in the Indian Ocean (12° 12" S, 96° 54" E). The group is comprised of two separate coral atolls, consisting of 27 islands. The southern atoll consists of 26 islands, surrounding a shallow lagoon, two of which are inhabited with a total population of approximately 600 people. Since 2002, a Service Delivery Arrangement (SDA) has been in place between the Department of Fisheries, Western Australia (DoF) and the Commonwealth Government of Australia to manage the fish resources of the Cocos (Keeling) Islands (CKI). Research and monitoring of selected targeted invertebrates and reef health have been undertaken by DoF since 2006, with priorities determined based on a risk based framework. For further details of previous research see Bellchambers and Evans (2013).

Coral reefs play an important ecological role in forming and maintaining both the physical and ecological structure of the marine environment and provide social and economic benefits to the community. Broad level, cost effective reef monitoring programs allow scientists and managers to detect changes in the marine environment and may provide insight into the potential causes of disturbance and allow for adaptive management practices for both fish and fish habitats. The broad benthic communities of the southern atoll of CKI and, in further detail, the trends in abundance of hard (Scleractinian) coral communities are described in this report. The location of the monitoring sites and periodicity of sampling (2010 to 2014) has enabled assessment of natural impacts which have significantly influenced this geographically remote location, namely a large scale hard coral mortality event which occurred in the lagoon between 2012 and 2013.

Lambis lambis (gong gong or common spider conch) is a gastropod mollusc regarded as a delicacy by the Cocos Malay population of CKI. Although there are no recreational catch records, historical surveys indicate that *L. lambis* have been heavily fished at CKI for the last thirty years. DoF commenced surveys of *L. lambis* stocks at CKI in 2007 to assess trends in their abundance and distribution, to provide an indication of the stock status of the species. A comparison of DoF data (2007-2011 and 2014) with historical data (1992) indicates the average densities of *L. lambis* have significantly decreased and continue to decline. *L. lambis* has been identified as one of the most vulnerable species to overfishing in the CKI. This report supports these concerns with large reductions in densities recorded over an 18-year period and significant reductions occurring over the last 8 years. It is likely that fishing has played a role in the decrease in density of *L. lambis*.

Giant clams (*Tridacna sp.*) are also a popular food source for the local Cocos Malay population and anecdotal information suggests that giant clams may have been collected for use as food and ballast by passing ships. Given the status of giant clams worldwide and their inherent vulnerability to overexploitation, a comprehensive survey was conducted in 2011 to document the distribution, abundance and size frequency of giant clam populations at CKI, with a second survey conducted in 2014. A comparison of the 2011 and 2014 datasets shows a decline in abundance of giant clams in key habitats. In addition, in 2014 only one species of giant clam (*Tridacna maxima)* was identified in the survey. While in 2011 one individual *T. derasa* was identified, no *T. derasa* were observed in 2014. As in the DoF 2011 survey no

T. gigas were recorded and anecdotal reports continue to support that the species may be locally critically endangered or extinct. The current level of recreational harvest of giant clams at the CKI is unknown. Therefore estimates of catch were calculated to provide an indication of the potential recreational harvest. Estimates of recreational take indicate that catches of *T. maxima* continue to be close to maximum sustainable yield therefore extremely conservative catch limits (both recreational and commercial) should be legislated and enforced.

The benthic marine environment of CKI is influenced by many factors, both natural and anthropogenic. Although in general the outer reefs of CKI continue to be stable, significant ($p \leq 0.05$) reduction of key habitats, particularly live hard corals, have been recorded in the lagoon where a majority of the key invertebrates discussed here (*L. lambis* and giant clams) are found. In addition, both *L. lambis* and giant clams have life history traits that make them particularly vulnerable to overexploitation.

In 2015, DoF began providing compliance services at CKI to support the anticipated introduction of new recreational fishing rules. Ongoing monitoring of the key invertebrate species is required to assess the effectiveness of these measures. Continued periodic monitoring of the reef health is also required to detect and assess naturally occurring perturbations and their potential impact on fish and fish habitats. Future research needs to focus on providing monitoring and biological data to understand trends in abundance of targeted species and to assess the effectiveness of management initiatives.

2.0 Background

The Cocos (Keeling) Islands (CKI) are a remote Australian territory located in the eastern Indian Ocean, approximately 2800 km northwest of Perth and 1200 southwest of Jakarta, Indonesia (12° 10' S 96° 50' E; Figure 2.1). The CKI are comprised of 27 separate islands on two coral atolls, two of which (Home and West Islands) are inhabited by a total population of approximately 600 people (Australian Bureau of Statistics 2015). The southern, populated CKI atoll is approximately 165 km^2 and consists of 26 islands surrounding a shallow lagoon (Woodroffe *et al.* 1994, Woodroffe and McLean 1994). The smaller, un-populated North (Pulu) Keeling Island atoll is located approximately 24 km to the north of the southern atoll with a land area of 1.2 km^2 (Director of National Parks 2015). The lands and waters extending 1.5km offshore of Pulu Keeling Island have been protected as a National Park since December 1995 under the *Environmental Protection and Biodiversity Conservation Act 1999* (Director of National Parks 2015).
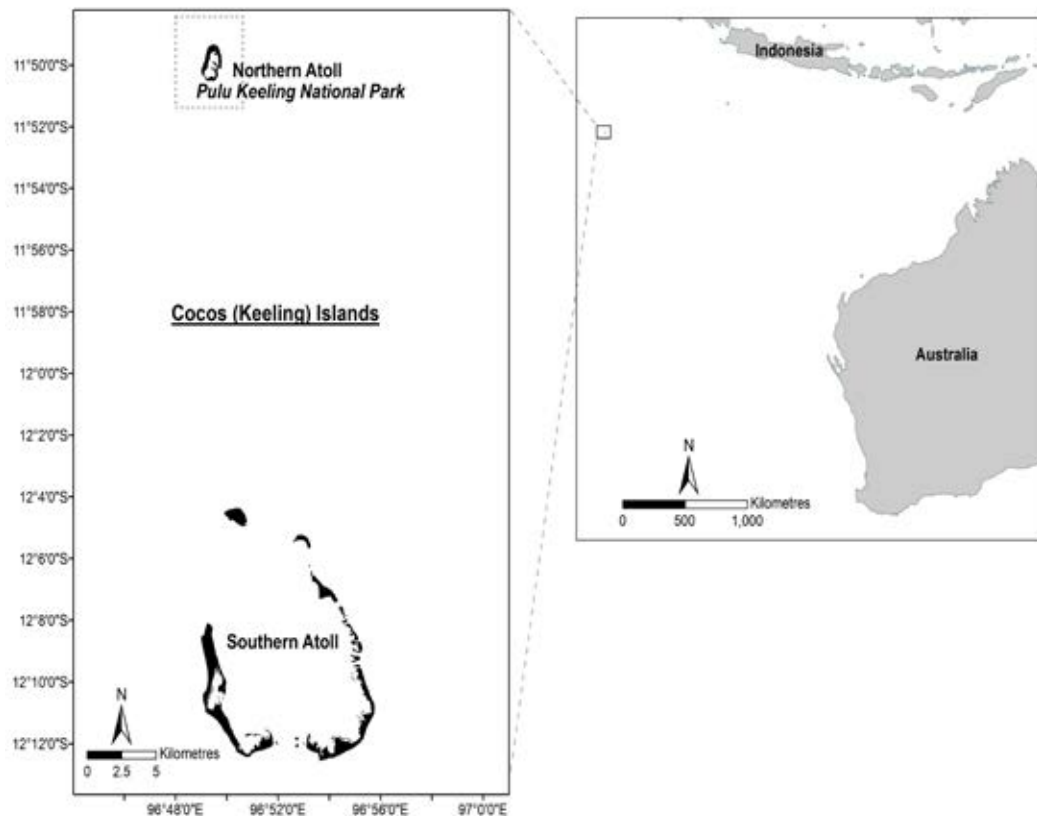
Figure 2.1   Location of the Cocos (Keeling) Islands relative to mainland Australia.

The CKI have three main aquatic habitat types: outer reef terrace (subtidal); reef flats including sandy and rocky shores (predominantly intertidal); and lagoon (predominantly subtidal) (Berry 1989). The outer reef terrace has the most abundant and diverse coral growth with up to 60 % cover (Williams 1994). The reef flats are varied and merge into the lagoon habitats in the channels between the islands (Williams 1994). Seagrass beds have developed on the inshore reef flats where sand has accumulated to depths of about 5 cm (Williams 1994). Within the lagoon, seagrass habitats are recognized as extremely important in

stabilizing soft sediment and providing nursery areas for fishes (Berry 1989). *Thalassia hemprichii* dominates the seagrass beds (Williams 1994). The seagrass may be directly (via herbivores) or indirectly (via detritivores) at the base of the food chain and therefore is important in the local ecosystem (Berry 1989). Protected embayments within the lagoon, particularly those backed by *Pemphis acidula* (small leafed mangrove) are also biologically rich and presumably important as fish nursery areas (Berry 1989).

The CKI are Australian territories, with the fish resources managed by the Department of Fisheries, Western Australia (DoF) on behalf of the Australian Commonwealth Government, under a Service Delivery Arrangement (SDA). On Home Island, there are approximately 417 Cocos Malay residents, while West Island has approximately 190 residents (West Islanders), largely comprised of Government employees, contractors, other residents and their families (Bellchambers and Evans 2013). Recreational and subsistence fishing is widespread on the atoll. The majority of harvesting of marine species is done by the Cocos Malay community, who are dependent on the local marine resources for a large part of their diet (Hender *et al.* 2001, Bellchambers and Evans 2013). The West Islanders predominantly fish for sport, targeting larger pelagic or reef-dwelling species (Hender *et al.* 2001). Only one commercial fishing license currently operates at the CKI, authorised to fish for the marine aquarium trade, with the primary target being the endemic Cocos Angelfish (*Centropyge joculator*). Commonwealth Ministerial approval has also been granted for the collection of giant clams (*T. maxima*) for aquaculture broodstock purposes. Providing for sustainable, on-going recreational and subsistence use of marine resources by people living at CKI, rather than for export of fish to other places, has been the primary fisheries management focus.

Various fish species (e.g. Lethrinids and Serranids) are caught by both the Cocos Malay and West Islanders, with an estimated 7.3-10.3 % of the standing fish stocks of the atoll harvested (Hender *et al.* 2001). Several invertebrate species are also taken, such as gong gong or common spider conch (*Lambis lambis*) and giant clams (*Tridacna sp*). The *L. lambis* is important to the local Cocos Malay population and is often collected in large numbers. There are also small tourism operators whose business in dependent on the ongoing health of the local marine environment.

DoF has undertaken work over a number of years on the development and refinement of a suite of island-specific recreational fisheries rules, including the development of specific recreational bag (catch) limits for CKI. The legislation to provide for the proposed recreational fishing rules has been developed and is with the Commonwealth Government for consideration. The proposed recreational fishing rules have been designed to develop community engagement and acceptance of the concept of sustainability and daily catch (bag) limits.

DoF research and monitoring at CKI is assessed based on risk to prioritise research and management objectives. This report summarises the research on reef health, which was highlighted as a risk due to the lack of ongoing monitoring to detect and assess natural or anthropogenic impacts influencing the fish habitats of CKI. This report also updates the research and monitoring on two targeted invertebrates (gong gong and giant clams) that were highlighted as high-risk either due to lack of knowledge and/or current/potential fishing pressure.

3.0 Status of Cocos (Keeling) Islands coral reef habitats 2010-2014

3.1 Introduction

Coral reefs play an important ecological role in forming and maintaining both the physical and ecological structure of the marine environment and provide social and economic benefits to the community (Moberg and Folke 1999). However, coral reefs are also dynamic environments that undergo cycles of natural disturbances e.g. wave impacts, salinity fluctuations, sedimentation, bacterial infection and thermal stressors (Baird and Marshall 2002, Blakeway 2005). These stresses occur at different timescales and frequency, with the capacity to recover and temporal scale of recovery dependent on a number of natural occurring variables (Hoegh-Guldberg *et al*. 2008, Burke *et al*. 2012, Gilmour *et al*. 2013). Recovery times can also be impacted by anthropogenic stresses (e.g. fishing, pollution, physical damage) and geographic isolation. Geographic isolation may reduce larval recruitment from neighbouring reefs and increase the dependence on self-recruitment (Ayre and Hughes 2000, Hughes *et al*. 2003, Ayre and Hughes 2004, Gilmour *et al*. 2013). Therefore it is important that long term reef monitoring datasets are captured, to understand which changes are natural phenomena and which are a result of anthropogenic impacts that may require management intervention (Magurran *et al*. 2010). This is particularly important in remote locations where subsistence based communities are highly reliant on the local coral reefs for sources of food and income.

The CKI are some of the most remote coral atolls in the world, with Christmas Island and Indonesia being the closest land masses at 950 and 1000 km away, respectively. Formed on the remains of ancient volcanos and part of the Vening-Meinesz seamounts (Bunce 1988) these islands have been built up by coral accumulation over thousands of years that is between 500 – 1000 metres thick (Darwin 1897, Bunce 1988, Woodroffe *et al*. 1990, Parks Australia 2004). The surrounding waters are also up to 5000 - 6000 m deep, creating a further isolation barrier (Parks Australia 2004, Hourston 2010). The CKI represent the western extension of the Western Pacific marine biogeographic region. The marine communities are predominately comprised of species from the tropical Indo-West Pacific, with low levels of West Indian ocean species and overall low endemism (Woodroffe & Berry 1994).

Previous surveys of the benthic coral reef environment at CKI by Veron (1990, 1994) identified 29 genera consisting of 99 species of hermatypic corals. Eighty seven of which were known Western Australia species, nine had never been recorded in the eastern Indian Ocean and the remaining three were thought to be endemic (Veron 1994). Although the majority of the species at CKI are known Western Australian species, the level of regular external coral recruitment and current genetic makeup of the coral species at CKI is unknown. The isolation of CKI reefs compared to other Australian reefs has also been demonstrated with Veron (1990) estimating that 94 days would be required for direct transport of coral propogules from Western Australia. In addition, there is no evidence that Christmas Island acts as a 'stepping stone' for the dispersal of coral to CKI (Richards and Hobbs 2014). The existence of Indonesian or South East Asian hard coral species on the reefs

was also reported to be low or not reliable, primarily due to the lack of knowledge of Indonesian corals communities at the time (Veron 1990). However, it was estimated that it would take 46 days (or 23 days for winter spawns) for coral propogules from Indonesia to reach CKI.

Coral loss has been reported in many locations around the world, including Australia (De'ath *et al.* 2012), the Caribbean (Eakin *et al.* 2010), Venezuela (Bastidas *et al.* 2012) and Chagos Archipelago (Riegl *et al.* 2012). The detrimental effect this loss can have on fish assemblages and communities that are reliant on these reefs for tourism and recreational, commercial and substance fishing can be severe (Turner *et al.* 2007, Cooley *et al.* 2009). Historically, CKI has experienced natural disturbances such as coral bleaching, cyclones, outbreaks of crown of thorns starfish, and die off of lagoon corals due to de-oxygenation events (Colin 1977, Woodroffe and Berry 1994, Marsh 1994, Bunce 1988, Hender *et al.* 2001, Hobbs and McDonald 2010, Hobbs and Macrae 2012, Hobbs *et al.* 2012).

Currently there is a lack of comparable long-term data on the marine habitats of the CKI. Only four major studies of coral species diversity and distribution have been undertaken at CKI between 1879 and 1994 (Richards and Hobbs 2014). This has included collections for the British Museum in 1879 (see Veron 1990), Wood-Jones in early 1900's (see Wood-Jones 1912), Gibson-Hill in the 1940's (see Wells 1950) and Veron in the 1980's (see Veron 1990). A Coral Research Atoll Bulletin was also dedicated to the CKI in 1994 (Woodroffe and Berry 1994) and is the most recent detailed description of the marine environments. A report by the Deptartment of the Environment and Heritage (2005) is the only other published report detailing a long term monitoring reef health monitoring program for CKI. The study, conducted over different years between 1997 and 2005 (see Department of the Environment and Heritage 2005) focused on the outer reefs of the southern atoll only and broadly categorised the habitats based on visual assessment as described by the survey technique used in ReefCheck© (Hill and Loder 2013).

Given the lack of ongoing long term monitoring, DoF developed and implemented a long term reef monitoring program in 2010. This program was ranked as a priority due to the remoteness, relatively low anthropogenic impacts to the reef, high level of hybridisation of marine flora and fauna (Hobbs *et al.* 2009) and high level of historical and current subsistence fishing dependent on the marine environment. This report describes the findings of the long-term reef monitoring program at CKI including;

- The broad habitats identified and the spatio-temporal trends
- Environmental influences on the CKI marine ecosystem
- Spatio-temporal hard coral diversity and abundance (in percent cover) of CKI

3.2 Methods

3.2.1 Study site

This study focused on the southern atoll comprised of a shallow water (<10m) lagoon with a variety of habitats, surrounded by a coral terrace of between 5 to 30 m deep stretching up to 1 km from the lagoon before dropping away to depths of several thousand meters (Williams

1994). There are also intertidal reefs surrounding the island land masses. CKI is exposed to strong south east trade winds nominally between April and November as well as predominately southerly swells (Bureau of Meteorology 2015).

Five monitoring sites, which are distributed throughout the lagoon and surrounding coral terrace, were chosen to represent the variety of coral reef habitats at CKI (Figure 3.1). This includes three sites on the coral terrace (North, East and West; sites 1-3) between 12 and 14 metres deep. Two sites are located within the lagoon (coral outcrop and blue hole habitats; sites 4 & 5) with both sites approximately 5 metres deep. Each monitoring site also has an in-situ temperature logger that is downloaded and serviced annually.
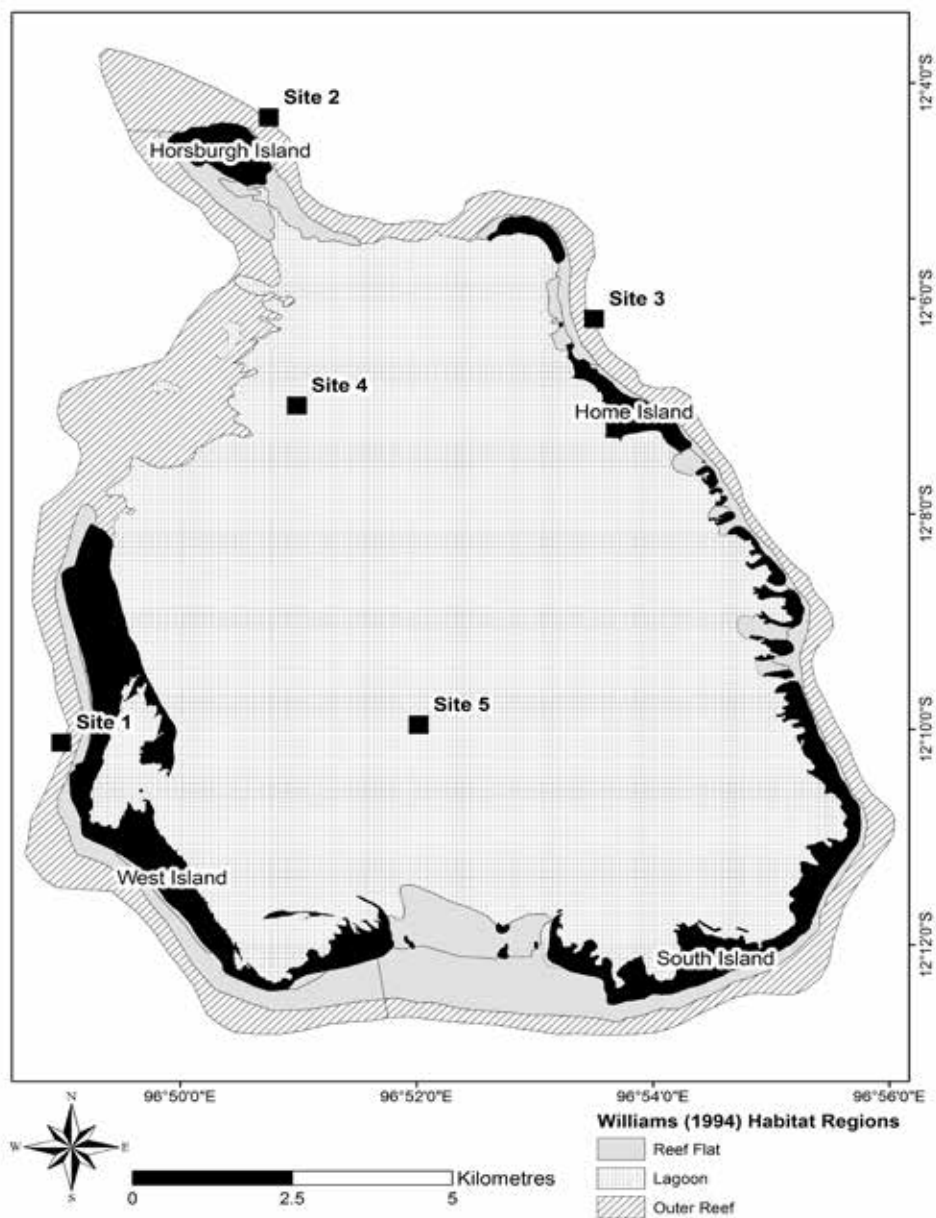

Figure 3.1 Location of reef monitoring sites at Cocos (Keeling) Islands

3.2.2 Habitat data collection and identification

The sites have been monitored annually between 2010 and 2014. Each site consists of three replicate 50 m transects spaced at least 25 m apart and permanently marked at 0, 25 and 50 m by a float attached to a small concrete mooring block. Sites were surveyed in autumn (March-May) using diver operated video (DOV) approximately 1 m above the substratum, that captured approximately a 1 m swath of video or 50 m^2 per transect. All DOV footage was downloaded and converted to Audio Video Interleave (AVI) format using Xilisoft Converter Ultimate$^©$ (Xilisoft Corporation) for analysis.

Benthic cover was estimated using the point count software TransectMeasure© (http://www.seagis.com.au). The software overlays a predetermined set of points onto video allowing for habitat discrimination and conversion to percent cover. The analysis was undertaken on 40 frames per replicate with 12 fixed points in each frame (4x3 point grid). Each point was classified into the following five broad categories; hard (Scleractinian) corals, soft corals, macro algae, abiotic and other. The abiotic category included subcategories of sand, rubble, rock, dead hard coral and relic reef. The 'other' category was used to identify benthic organisms such as sponges, seagrass, molluscs, hydroids and bryozoans. The hard corals were further categorised to genera and any injuries identified (i.e. bleaching or disease). Analysis was completed by trained hard coral analysts, with validation of previously analysed transects repeated on a regular basis to ensure accurate identification.

3.2.3 Broad habitat data analysis

The point count data for each transect was exported from TransectMeasure© into Microsoft Excel. Data was plotted to display broad habitat composition over time for all CKI sites combined. Patterns in the benthic communities were further analysed based on the living broad habitat types of hard coral, soft coral, and macro algae using PERMONOVA (Anderson 2001) in PRIMER-E v.6. From a square root transformed Bray-Curtis resemblance a two way fixed model PERMANOVA (maximum permutation = 9999 and Monte Carlo test applied) was used to test the data set with Year and Site as factors. Pairwise tests were conducted on significant results to determine the factors responsible for significant differences. Non-metric multi-dimensional scaling (nMDS) were then visualised based on a Bray-Curtis resemblance of averaged data at the site and year level to show changes in habitats.

3.2.4 Hard coral composition analysis

Mean abundance and standard error of coral cover was calculated to determine percentage cover of hard coral genera at the site and level for each year. A measure of genus richness was also calculated from the mean and standard error of genera per year at the site, zone (lagoon and outer reef) and region (all sites combined) level. To further examine changes amongst sites and years and discriminate dominant genera that may be driving patterns of change in the benthic community, PRIMER-E V6 was used to conduct a canonical analysis of principal co-ordinates (CAP) on the square root transformed mean percentage cover of dominant genera data, using a Bray-Curtis resemblance (Anderson and Willis 2003). Dominant genera were classified as hard coral genera or biotic benthos ≥5% coverage, at any

site in any year throughout the study period. Two-way PERMANOVA (maximum permutation = 9999 and Monte Carlo test applied due to low levels of permutations) were then performed on the dominant genera by selecting each individual genera from the untransformed mean percentage cover data, performing a Bray-Curtis resemblance matrix with a dummy value added. Pairwise tests were conducted on significant results to determine the sites (factors) driving changes (Anderson and Willis, 2003).

3.2.5 Seawater temperature data

HOBO Pendant® temperature/light data loggers were used to collect temperature data at each site. The temperature loggers took a measurement every 20 minutes and were downloaded and serviced annually. Data from these loggers, where available, was averaged to provide mean daily temperature at each site. Sea surface temperature (SST) data for the same period was obtained from the National Oceanic and Atmospheric Administration (NOAA) website for the Pulu Keeling National Park virtual station (www.coralreefwatch.noaa.gov/satellite/vs/australia.php#PuluKeeling_Australia) to compare with the in-situ logger data.

The thermal bleaching threshold used for this report was derived from NOAA's coral reef watch website (www.coralreefwatch.noaa.gov) that estimates that corals start to become stressed when the SST is $1^{o}C$ above the highest monthly mean temperature (Glynn and D'Croz 1990). The long-term highest monthly mean SST at CKI is $28.5^{o}C$, therefore the thermal bleaching threshold is $29.5^{o}C$. The daily mean in-situ temperature data was used to calculate cumulative heating exposure (as described by Berkelmans and Willis 1999 and Berkelmans 2002), at each site, with December -April used as 'season' and $29.5^{o}C$ as the CKI bleaching threshold. This was only possible for years when a full dataset from the in-situ temperature loggers for the 'season' was available.

3.2.6 Meteorological data

Meteorological data from the CKI Bureau of Meteorology (BoM) station at West Island Airport (www.bom.gov.au/climate/data/) and historical records from the National Library of Australia and Partners website (www.pandora.nla.gov.au) and monthly rainfall were also collated. Mean monthly wind speeds and direction were also examined to determine their impact on lagoon conditions.

3.3 Results

3.3.1 Long term trends in overall reef composition and health

Over 95% of the coral reef habitats surveyed by this study were categorised into three broad habitat classes; hard coral, soft coral and abiotic substrate (Figure 3.2). Macro algae and all individual subcategories within 'other' made up <1% of the benthos at any site and were not examined further individually due to their low abundance. At the regional level (all of CKI), soft coral was the only habitat where the percent cover was consistent across the study period with a mean coverage of 13.5%. Regional levels of hard coral cover were consistent between 2010 and 2012 ranging between 47.5% and 51.6% of the total benthic habitat cover respectively. However, a reduction of hard coral cover was observed between 2012 and 2014

(35.5% cover in 2013 and 31.6% in 2014 of the total benthic habitat cover) suggesting a reduction of ~40% of total hard coral cover at CKI between 2012 and 2014 (Figure 3.2).
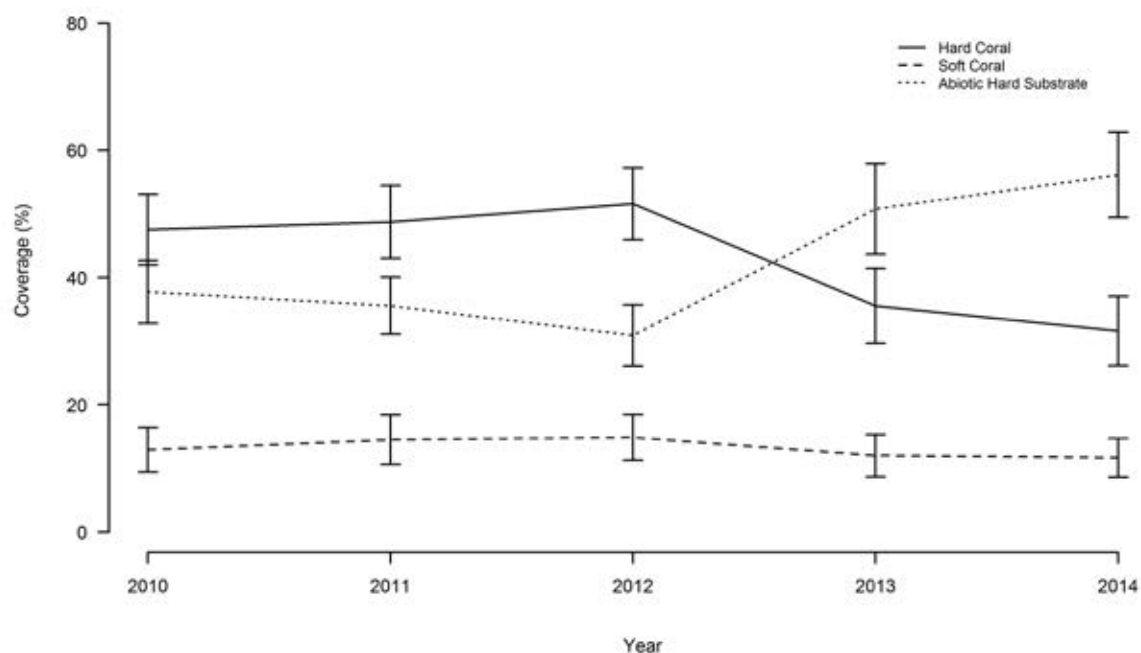
Figure 3.2 Broad habitat composition of CKI (macro algae and other <1%)

There was a significant difference in benthic habitat composition at the study sites by site and year (PERMANOVA, p≤0.0001 (MC); Table 3.1). Pairwise comparisons of factors site and years confirmed that the changes that occurred to the benthos throughout the study period (years) within each study site were not uniform ($p \leq 0.05$ (MC)). Significant differences ($p \leq 0.05$ (MC)) were observed at sites 4 and 5 (lagoon) throughout the study period. However, no significant difference ($p \geq 0.05$ (MC)) was observed at sites 1, 2 or 3 (outer reef). Site 4 showed a significant difference in live benthos recorded in 2014 from 2010, 2011 and 2012 ($p \leq 0.05$ (MC)). In addition a significant difference in site 4 was observed in 2010 – 2011 and 2010 -2012 ($p \leq 0.05$ (MC)). Site 5 showed no significant difference in live benthic cover between the years 2010 to 2012 or between year 2013 and 2014. However significant differences ($p \leq 0.05$ (MC) were observed between all years from 2010 to 2012 against both years 2013 and 2014, suggesting a severe disturbance or impact between 2012 and 2013/14 at sites 4 and 5.

Table 3.1   PERMANOVA summary of significant interactions of broad habitat compositions

Source	df	SS	MS	Pseudo-F	*p* (perm)	Unique Perms	P (MC)
Year	4	3040	760	12.132	0.0001	9930	0.0001
Site	4	25279	6319.8	100.88	0.0001	9940	0.0001
Year x Site	16	5298.5	331.16	5.2862	0.0001	9894	0.0001

The overall similarity between all sites, as shown in a non-metric multi dimensional scaling plot (Figure 3.3), stayed consistent throughout the survey period at 40%. Sites 1, 2 and 3 show little change throughout the study period with each site staying within 80% similarity over the 4 years. Before 2013 sites 1, 2 and 5 are most similar at 80% similarity. Post 2012 site 5 has a shift to only 40% similar and site 4 to 60% to their 2010-12 broad benthic composition.
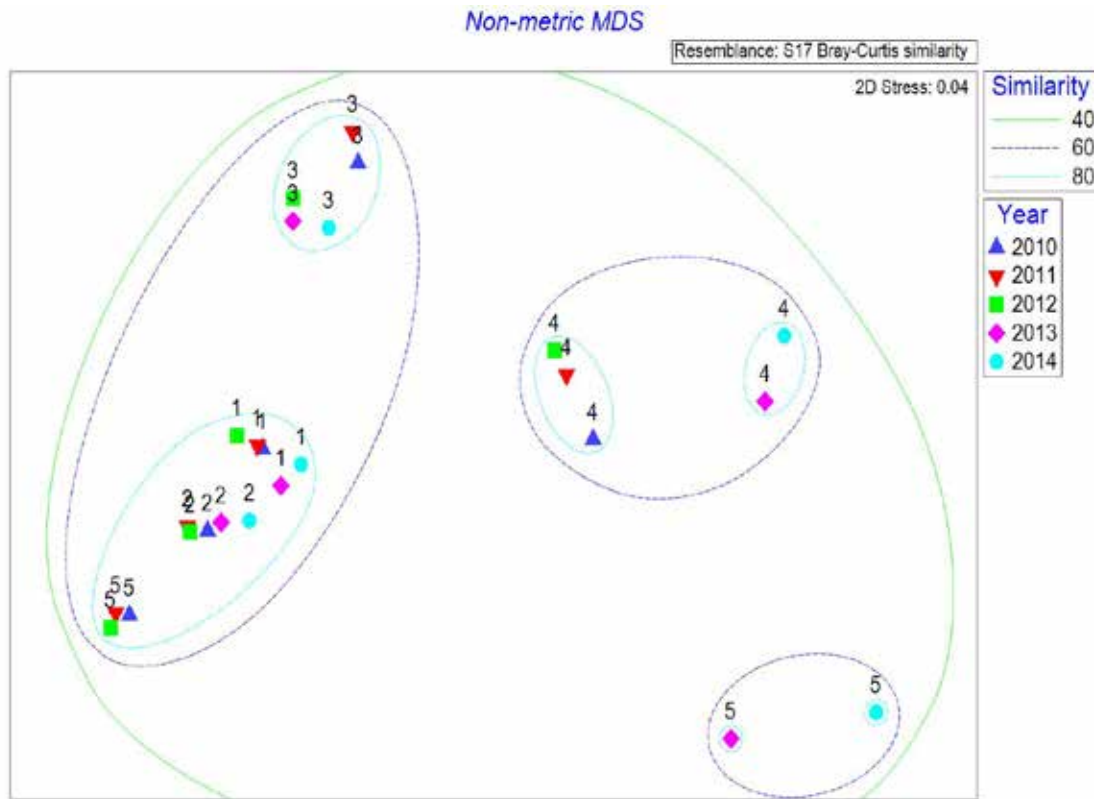
Figure 3.3 nMDS of similarity of monitoring sites at the broad benthic habitats

The observed changes predominately occurred at the lagoon sites (Figure 3.4) and were driven by a decrease in hard coral post 2012. For example, between 2012 and 2014 site 4 recorded a 50% decrease in hard coral cover (20% in 2012 to 10% in 2014), while site 5 recorded a 62% decrease in hard coral cover (76% in 2012 to 14% in 2014) (Figure 3.4). The significant differences ($p \leq 0.05$ (MC)) observed at site 4 pre 2013 are related to an increase in soft coral cover and slight decrease in abiotic substrate.
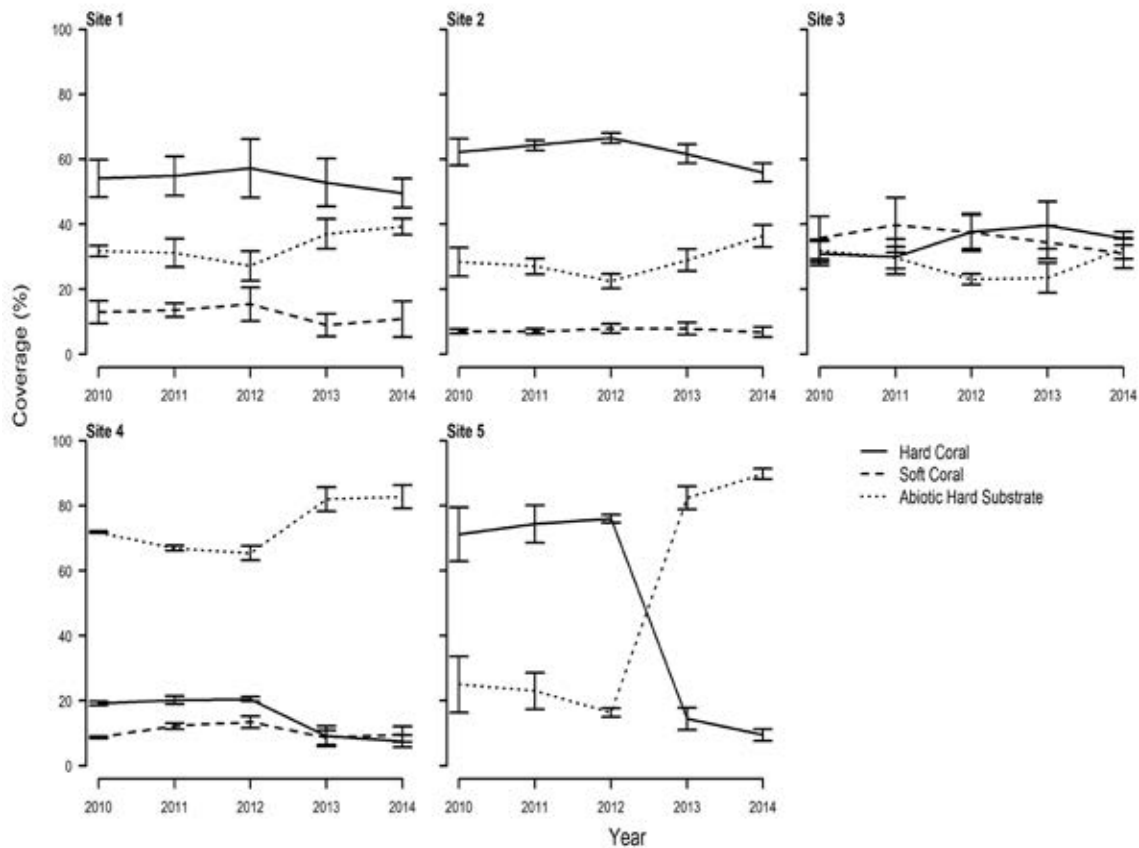
Figure 3.4 Percent coverage of dominant broad benthic habitats by site

3.3.2 Seawater temperature data

The mean daily in-situ seawater temperature at CKI ranged between 26°C and 29.5°C at sites 1 to 4 (outer reef and outer lagoon sites). Site 5, an inner lagoon site, recorded consistently higher temperatures and displayed a slight annual increase in mean daily temperatures between 2010 and 2014 (Figure 3.5). The NOAA SST recorded a single bleaching alert level 1 in April 2014, which was also detected by the in-situ loggers (Figures 3.5 and 3.6). The NOAA data recorded 2013 and 2014 as the warmest years within the study period. An anomalous cold water event was recorded by both the in-situ data and NOAA SST over a four day period from the 22nd to 26th of February 2013 (Figures 3.5 and 3.6). Seawater temperatures over this period decreased by 2 to 2.5°C at sites 1, 2, 3,and 5 and 1°C at site 4. Similar decreases in water temperature were observed several times during the study period, e.g. February 2012 (Figures 3.5 and 3.6). However, the February 2013 cold water event was the quickest temperature drop and recovery recorded during the study period.
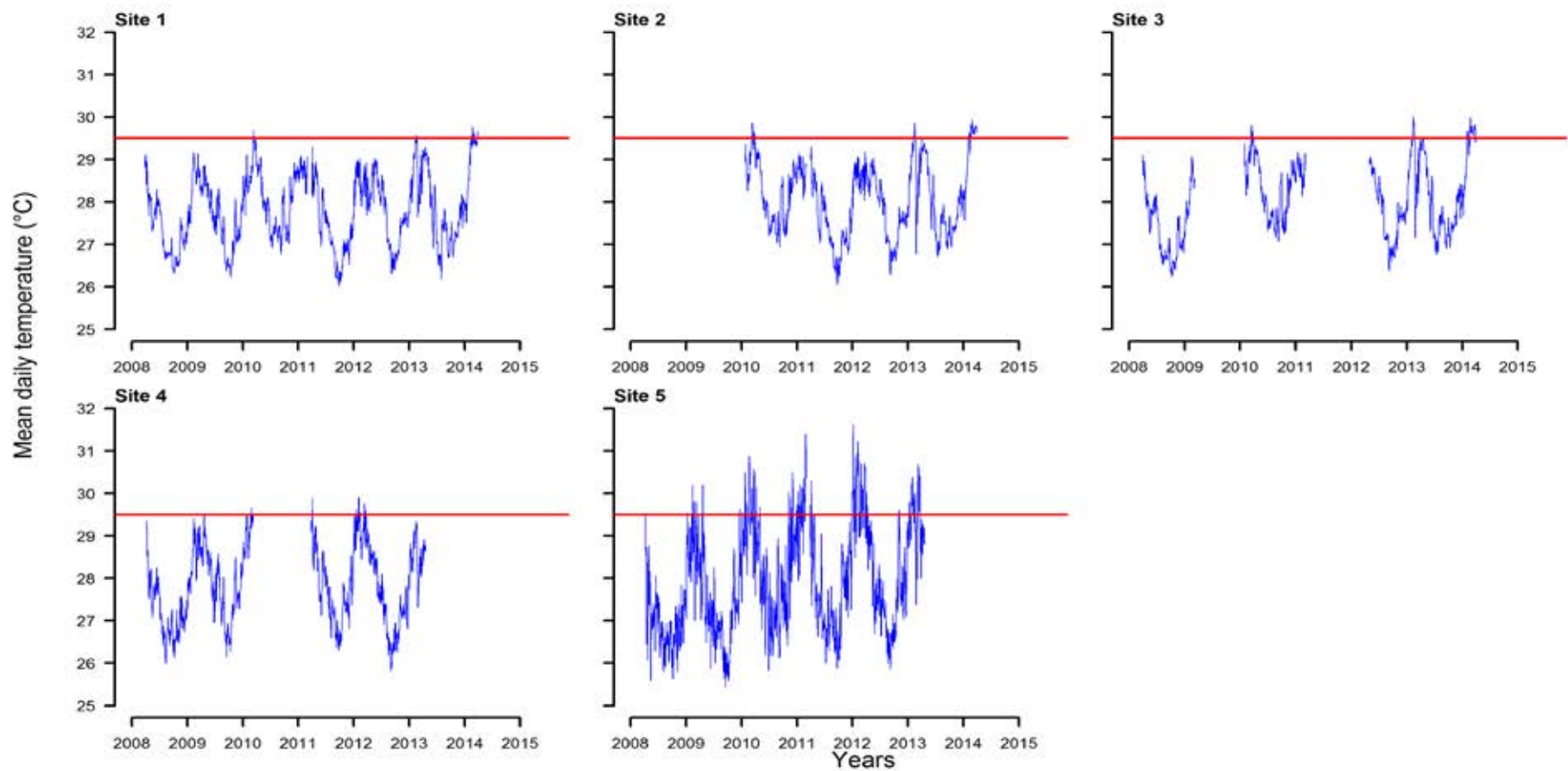
Figure 3.5 Mean daily in-situ seawater temperature data (red line indicates CKI thermal bleaching threshold, 29.5^{o}C)
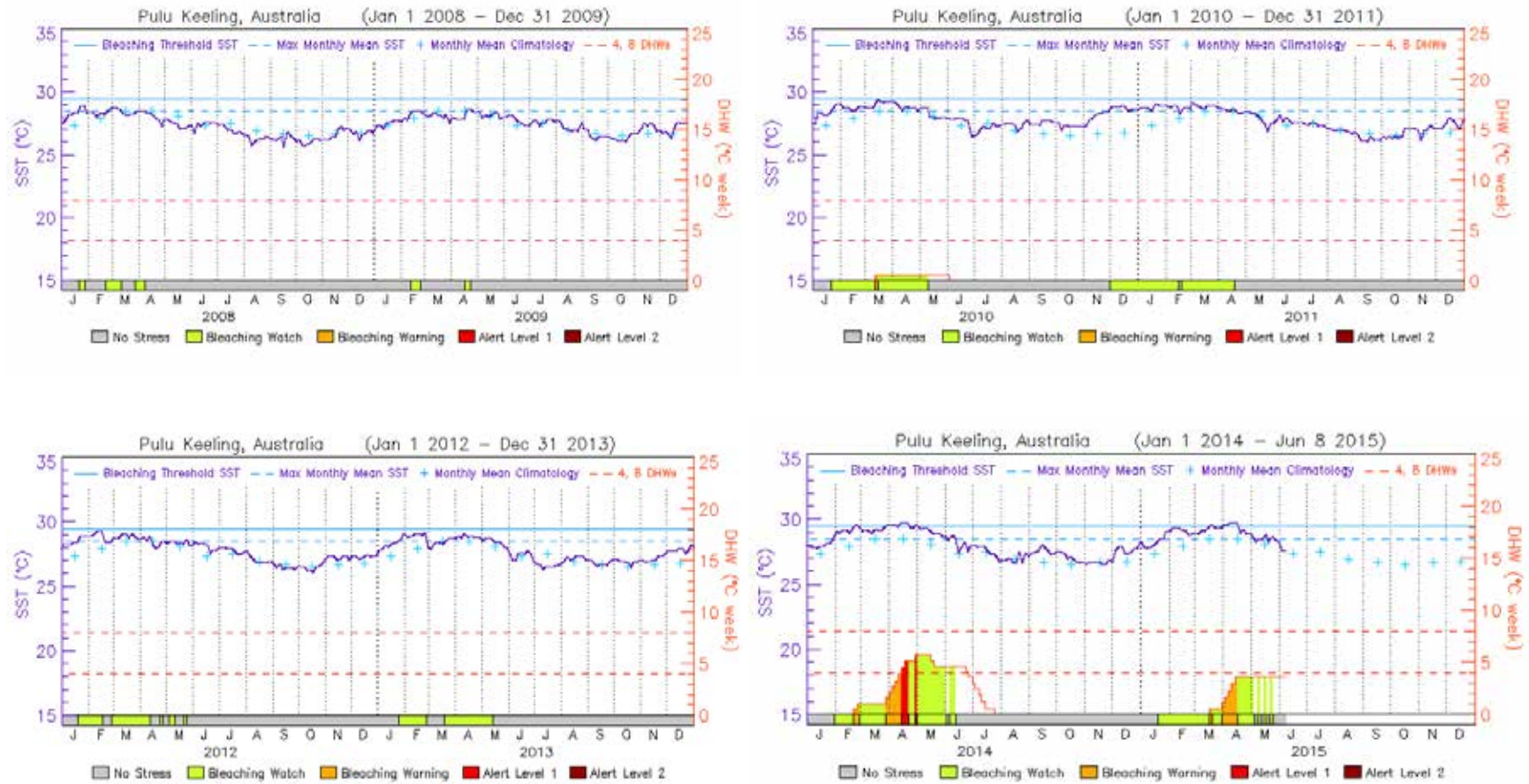
Figure 3.6 NOAA SST January 2008 – June 2015 (www.coralreefwatch.noaa.gov/satellite/vs/australia.php#PuluKeeling_Australia)

Degree heating day curves show that the outer reef sites (sites 1 to 3) had similar trends throughout the study period, where data is available (Figure 3.7). In 2012/13 season, the outer reef sites recorded 3, 14 and 17 days above the 29.5°C NOAA CKI bleaching threshold for sites 1, 2 and 3 respectively. No bleaching or mortality of hard coral was observed at these sites in the 2012/13 season. In the 2013/14 season the outer reef sites recorded the highest number of cumulative days of seawater temperature above the CKI bleaching threshold, during the study period. Sites 2 and 3 recorded 11 and 5 days respectively above 30°C in 2013/14 season with no significant loss of hard or soft coral observed (Figure 3.4). The two lagoon sites show different trends to both each other and to that of the three outer reefs sites. Where full datasets were available, the highest numbers of degree heating days observed in the lagoon were in the 2011/12 season (Figure 3.7). In this season the number of days the lagoon sites spent above the NOAA CKI 29.5°C bleaching threshold was 14 days for site 4 and 71 days for site 5. Site 5 also recorded 47 days above 30°C and 5 days above 31°C during the 2011/12 season. No large scale hard coral bleaching or mortality was observed at either site at the 2012 monitoring survey. Interestingly, site 5 recorded more days above the bleaching threshold in the 2011/12 season (Figure 3.7) when no widespread hard coral bleaching or mortality was recorded, compared with the 2012/13 season when the hard coral mortality event at this site was recorded (Figure 3.4).
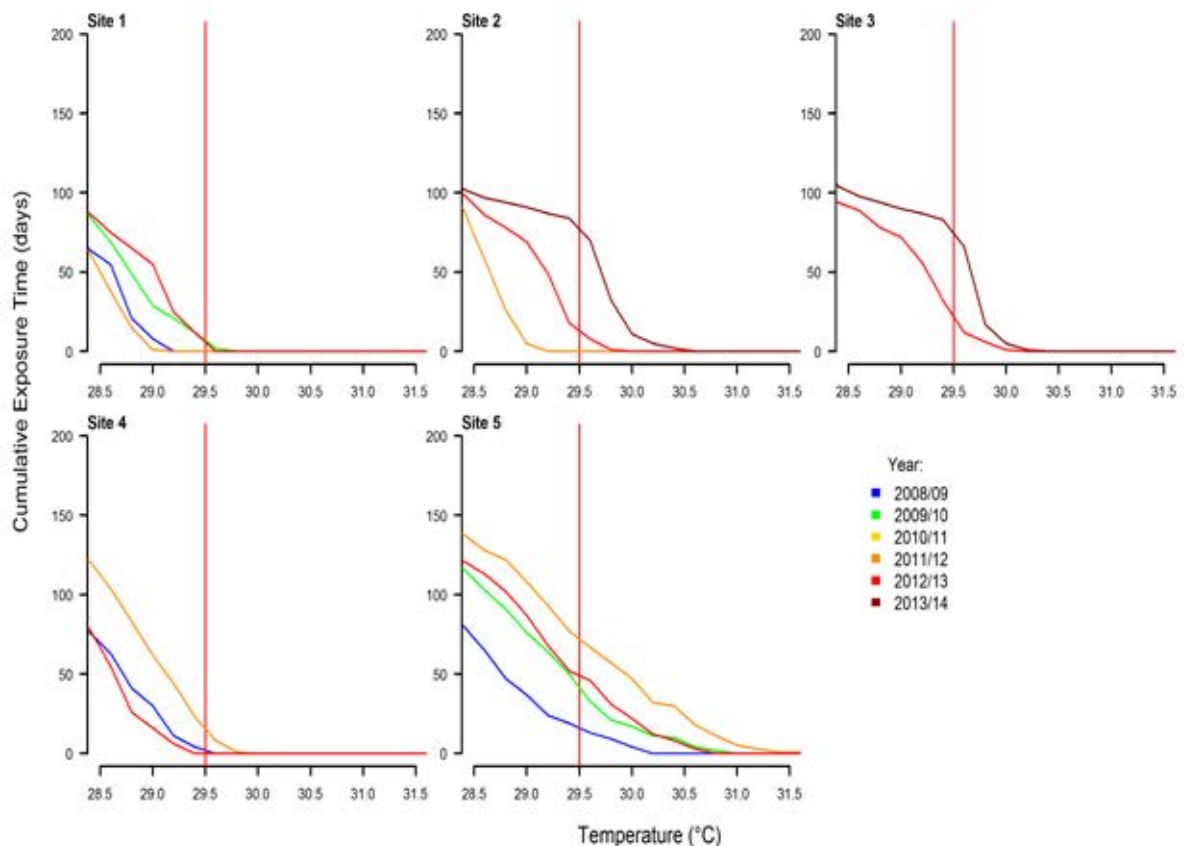
Figure 3.7 Cumulative exposure time of seawater temperatures (red line indicates CKI thermal bleaching threshold, 29.5°C)

3.3.3 Meteorological data

The daily average wind speed per month (Figure 3.8) showed no long-term low wind days throughout the reef monitoring study period (2010-2014). Average wind speeds in the 2010/11 and 2011/12 summer doldrum seasons ranged between 6 to 10 knots compared to the 2012/13 (mortality event year) and 2013/14 season, which had 10 to 13 knot mean monthly winds.
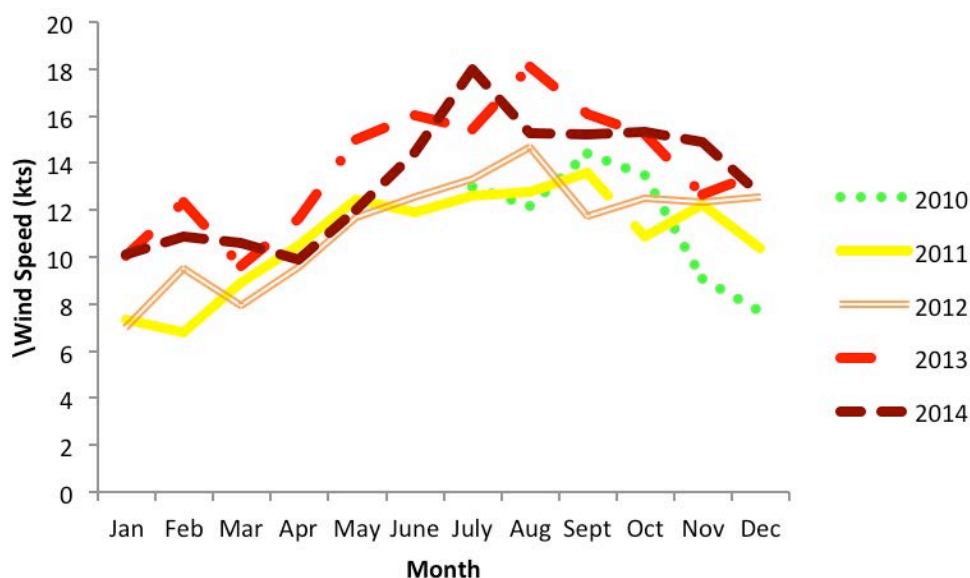
Figure 3.8   Mean 9 am wind speed (knots)

The wind direction at CKI is predominately ESE to SE, with greater than 50% of all winds coming from this direction (Table 3.2). The exception during the study period was 2012 when there was a ~10% increase in wind from the east.

Table 3.2   Annual wind direction (%)

Year	N	NNE	NE	ENE	E	ESE	SE	SSE	S	SSW	SW	WSW	W	WNW	NW	NNW	No Data
2010	2	5	3	7	16	29	26	2	4	3	0	2	2	0	1	1	1
2011	1	2	4	5	17	32	20	5	3	2	3	1	1	2	2	1	1
2012	1	2	2	6	31	29	15	3	2	2	1	1	1	2	1	1	1
2013	1	1	2	2	16	35	32	5	2	0	1	1	1	1	0	0	1
2014	2	3	3	6	19	30	25	5	2	1	0	0	0	1	1	2	0

During the doldrums, when the seawater temperature is also warmer, winds are generally from an easterly and southerly direction (Table 3.3). The summer season of 2011/12 has a slightly higher percentage of days recording winds from the north east.

Table 3.3   Doldrums (December – April) wind direction (%)

Season	N	NNE	NE	ENE	E	ESE	SE	SSE	S	SSW	SW	WSW	W	WNW	NW	NNW	No Data
2010/11	3	1	5	5	7	11	14	9	10	6	6	3	5	5	7	1	3
2011/12	1	7	9	13	13	9	13	5	3	3	2	4	9	3	5	1	2
2012/13	0	3	3	4	15	28	18	11	6	1	1	1	4	1	0	1	3
2013/14	1	3	3	9	9	16	26	11	9	1	3	0	1	4	0	1	2

Average annual rainfall at CKI during the study period was 2309 mm, with a minimum annual rainfall of 1464 mm in 2012 and maximum of 3490 mm in 2010 (Figure 3.8). No cyclone directly tracked over CKI during the study period. However, two systems passed nearby creating anomalous rainfall conditions and a third system (tropical cyclone Kate) passed CKI in December 2014, outside the study period. In 2010 tropical cyclone Anggrek was in the area and produced high rainfall during a typically low rain period (late October / early November) with 455 mm over 14 days. The low pressure system which became tropical cyclone Rusty in February 2013 was linked to the other anomalous rainfall event affecting CKI where 1000 mm was recorded for the month. The majority of this rain (845 mm) fell between the 23rd to 25th February 2013 with 416 mm in a 24 hour period on the 25th February 2013.
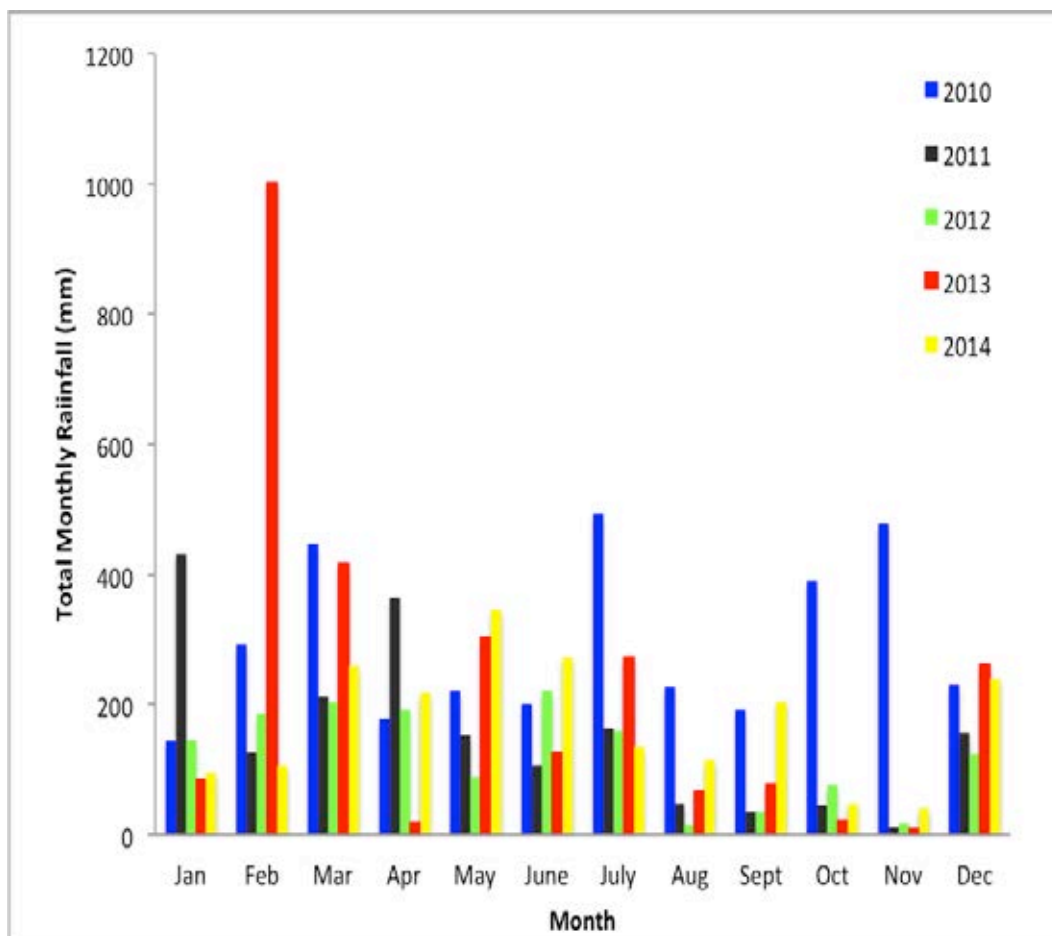
Figure 3.8   Total monthly rainfall CKI 2010 to 2014

3.3.4 Spatial and temporal hard coral composition

Twenty two hard coral genera were identified during this study (Table 3.4). However, only eight occurred at ≥5% cover at any site during the study period (Table 3.4). Three new genera not previously reported at CKI were also observed; *Coscinaraea*, *Pectinia* and *Trachyphyllia*. Although soft corals were found at most sites due to difficulties with identification they were not further classified (Figure 3.4).

Table 3.4   Hard coral genera observed at CKI during the study period, in decreasing order of abundance.

Genera >5%		Genera <5%			
1. *Acropora*	5. *Porites*	9. *Fungia*	13. *Coscinaraea*	17. *Favites*	21. *Trachyphyllia*
2. *Pavona*	6. *Favia*	10. *Astreopora*	14. *Pachyseris*	18. *Herpolitha*	22. *Turbinaria*
3. *Montipora*	7. *Pocillopora*	11. *Seriatopora*	15. *Lobophyllia*	19. *Pectinia*	
4. *Echinopora*	8. *Isopora*	12. *Stylophora*	16. *Leptoseris*	20. *Platygyra*	

At the regional level (all CKI sites) the mean number of hard coral genera recoded was 6.47 (±1.64) in 2010, with a gradual decline to 5.47 (±3.42) in 2014 (Figure 3.9). Outer reef sites had consistently higher diversity of hard coral genera than lagoon sites. Sites 1 and 2 displayed no change in genus richness from 2010 to 2014, while site 3 displayed a slight increase in the number of genera (Figure 3.9). Conversely a 60% decrease in the mean hard coral genera abundance was observed within the lagoon between 2010 to 2014, from 5.17 (±1.17) in 2010 to 2.00 (±1.10) in 2014, with the loss consistent over both lagoon sites (Figure 3.9).
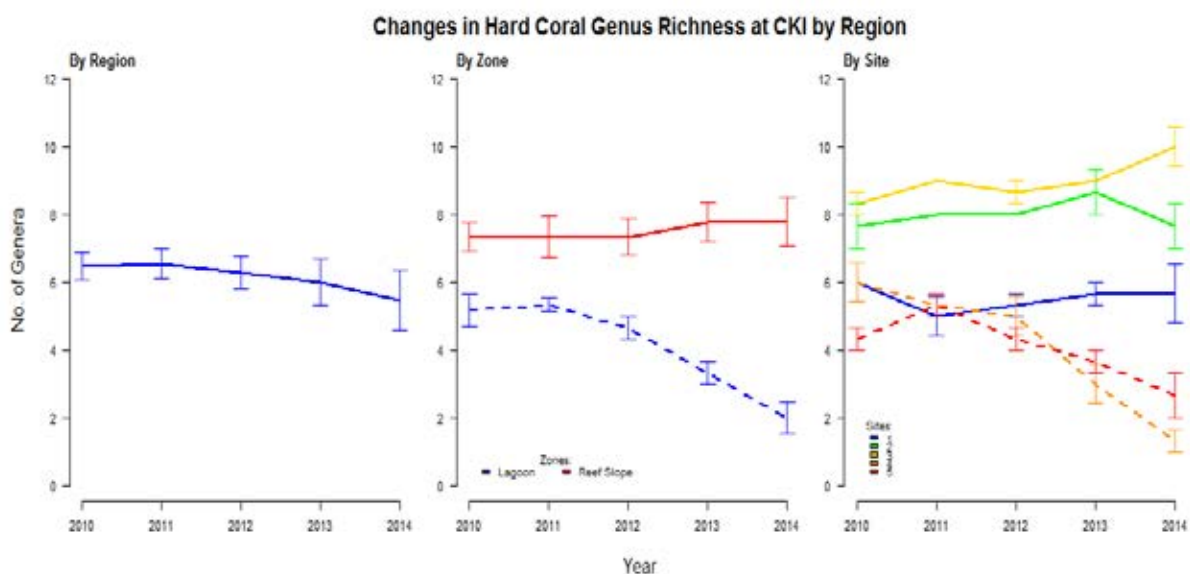
Figure 3.9   Mean hard coral genera by (a) region, (b) zone, (c) site

The most abundant (percent coral cover) and widely distributed genera of hard coral at CKI is *Acropora*, which was recorded at all sites (Figures 3.10 and 3.11, Appendix A). Similarly, *Montipora* was also observed at all of the monitoring sites (Figure 3.10, Appendix A). While both *Porites* and *Pocillopora* were also widely distributed, they were generally less abundant with a lower percent cover. *Pavona, Echinopora, Favia, Isopora* were all observed in high percent cover, however their distribution was limited to one or two sites (Figures 3.10 and 3.11, Appendix A). For example, *Echinopora* was only observed at site 5 at 34.9% ±12.2 percent cover in 2012 declining to 4.1% ±2.0 by 2014.

CAP analysis on dominant benthos (eight dominant hard coral genera and soft coral) showed that the abundance of individual genera was driving the separation amongst the CKI reef sites temporally and spatially (Figure 3.10). Sites 2 and 3 (outer reef) displayed the highest diversity of hard coral and were the most similar of all the sites, separated from the other outer reef site, site 1, by the presence of *Pavona* and soft coral (Figure 3.10). The outer reefs sites also showed little annual change (Figure 3.10). In contrast, sites 4 or 5 (lagoon sites) displayed significant ($p \leq 0.05$) separation between the sites between 2010-12 to 2013-14 (Figure 3.10). Site 4, which has the lowest percent cover of hard coral (Figure 3.4) observed significant ($p \leq 0.05$) changes from 2010 - 2012 to 2014 in percentage cover of *Isopora*, *Acropora* and *Pocillopora*. However, a significant decrease ($p \leq 0.05$) in *Pocillopora* or soft coral was not recorded at site 4. Site 5 recorded significant decreases ($p \leq 0.05$) in percent cover of all the dominant genera.
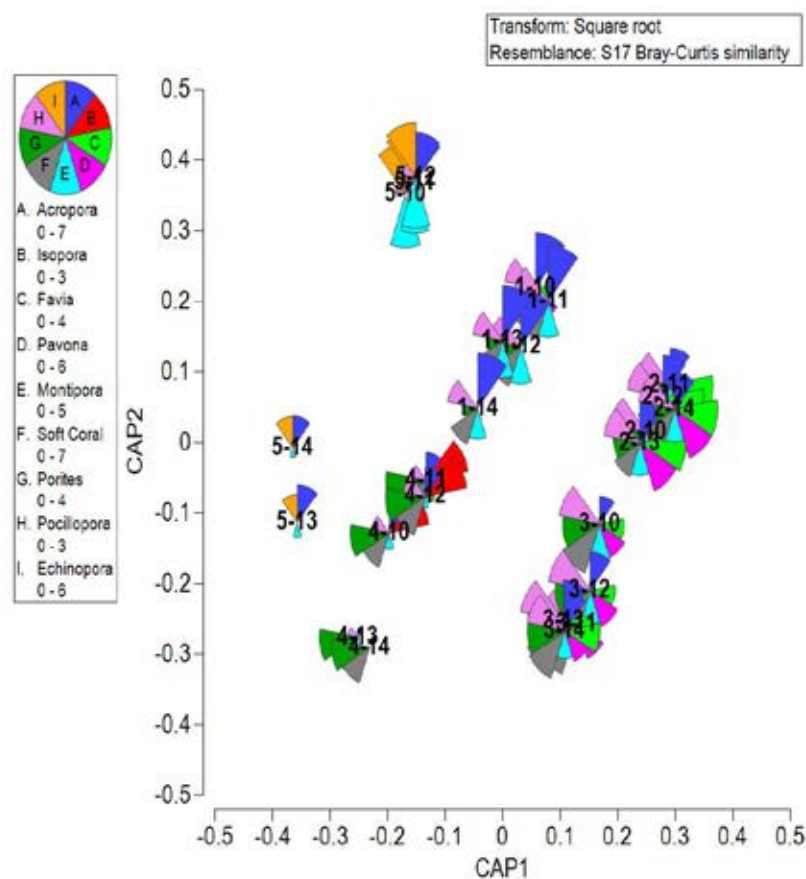
Figure 3.10   CAP analysis of hard coral genera and soft corals

At the site level, dominant hard coral displayed similar trends to the broad hard coral category with predominately no significant shift in habitat on the outer reef (sites 1 to 3) ($p \leq 0.05$) (Figure 3.11). However, hard coral genera cover at the lagoon sites (sites 4 and 5) displayed declines (Figures 3.4 and 3.11). Site 1 did not display any significant shifts ($p \leq 0.05$) in the two dominant genera (*Acropora* and *Montipora*). While site 2 showed annual fluctuations in the dominant genera (*Pavona*, *Favia* and *Acropora*) there were no significant ($p \leq 0.05$) shifts between years or during the survey period. Site 3 had no significant ($p \leq 0.05$) shift in the cover for three of the five dominant genera (*Acropora*, *Porites* or *Pavona*).

However, *Pocillopora* displayed significant seasonal fluctuations throughout the survey period, whilst *Favia*, displayed significant ($p \leq 0.05$) increases from 2011 onwards (Figure 3.11). The abundance of *Acropora* at site 3 also increased over the survey period, from 3.4% ±1.7 in 2010 to 11.2% ±3.5 in 2014 (see Appendix A for all genera mean cover between 2010 to 2014).

With the exception of site 4, the lagoon sites showed a significant ($p \leq 0.05$) decrease in all the dominant genera between 2010-12 and 2013-14 (Figure 3.11). While *Porites* did not display a significant ($p \leq 0.05$) decline at site 4, significant declines in *Isopora* and *Acropora* were observed ($p < 0.05$) with a 100% loss of *Isopora* cover in 2013 and 2014. Similarly, a severe loss of *Acropora* was observed with 4.0% ±2.1 cover in 2012 to 0.1% in 2013 and no observations in 2014. However, the most noticeable impact on coral cover was observed at site 5 (Figure 3.11). The three dominant genera (*Echinopora, Acropora and Montipora*) all recorded significant ($p \leq 0.05$) declines between 2012 and 2014, with >50% loss of coral cover observed for *Echinopora* from 34.9% (±12.2) in 2012 to 4.1% (±2.0) in 2014; *Acropora* from 21.51% (±7.7) in 2012 to 5.2% (±0.3) in 2014 and *Montipora* from 17.4% (±6.7) in 2012 to 0.1% (±0.1) in 2014.
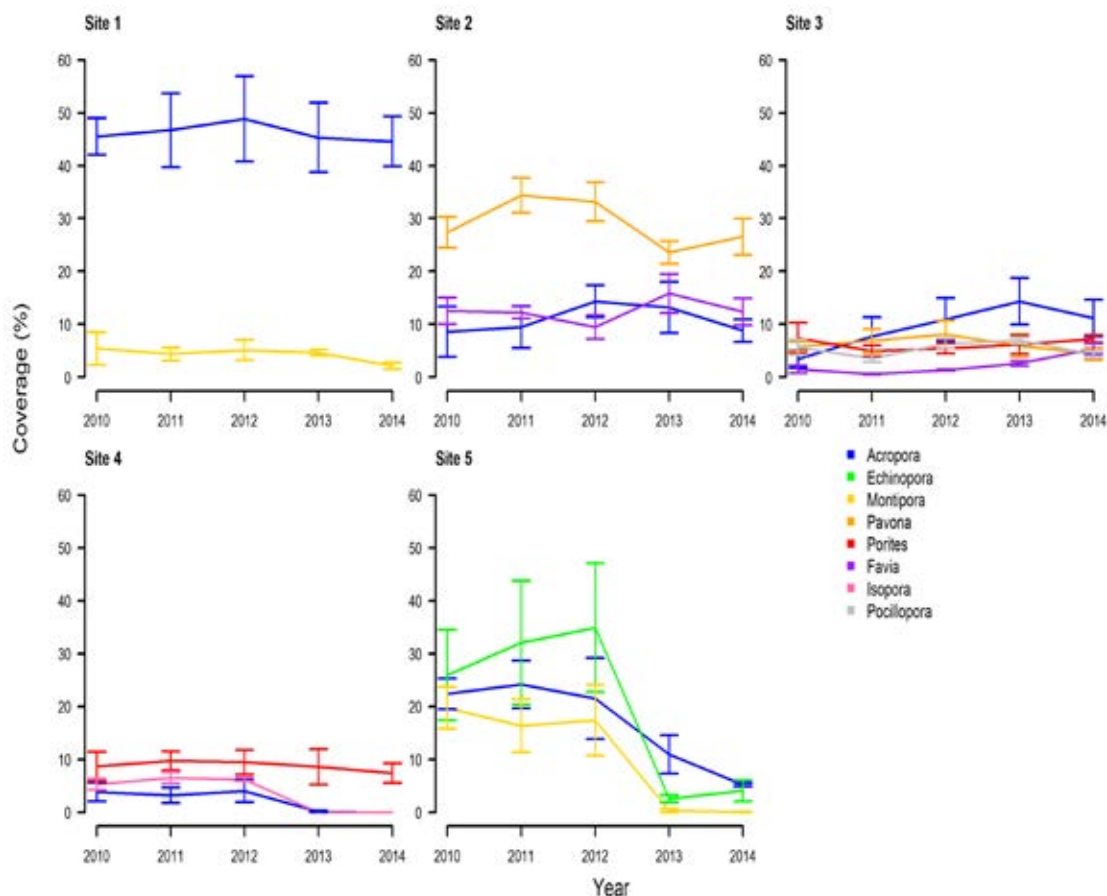
Figure 3.11   Percent cover of dominant hard coral genera by site.

3.4 Discussion

The benthic coral reef habitats of CKI are dominated by hard and soft corals, at levels typical of other remote tropical coral reefs in the Indo-Pacific region (Williams *et al.* 2013). Broad reef habitat composition between the CKI monitoring sites were similar ranging between 40 to 60% similarity, with sites 1, 2 and 5 having 80% similarity prior to 2013. The pre-2013 composition of live hard and soft coral is also comparable with previous surveys undertaken at CKI between 1997 and 2005 (Department of the Environment and Heritage 2005). However, this survey is the first to use diver operated video and point count analysis to provide a percent composition of benthic habitats at CKI. Twenty two genera were observed during the study. Previous studies have reported, 29 genera (Veron 1990) and 33 (Richards and Hobbs 2014). Given that this study was designed to assess and monitor broad level benthic community composition not to document diversity the differences in the number of genera observed in this study in relation to previous studies is comparable. Of the 22 observed hard coral genera eight genera were clearly dominant, the genus *Acropora* had the highest percent cover and *Acropora*, *Montipora*, *Porites* and *Pocillopora* had the widest distribution. Although other genera occur at multiple sites, the remaining four dominant genera appear to have a greater affiliation to specific sites that may be influenced by environmental conditions such as wave action, turbidity or thermal fluctuations.

In the period between the 2012 and 2013 surveys, a significant (~40%) decline in hard coral cover was observed at CKI. However, the decline was not uniform across CKI, with only the two shallow (~5m deep) lagoon sites significantly affected (>50% decrease). Benthic habitats at the outer reef sites (12-14m deep) remained stable at both the broad habitat and hard coral genera level, with no bleaching, disease or significant mortality observed. As warm water thermal bleaching was not highlighted as a threat during the study period, by either the NOAA SST or in-situ temperature data, it was expected that all sites would remain stable. Similarly, there was no direct impact of a cyclone or anomalous swell events nor is sedimentation likely to have impacted coral mortality due to the small land mass and lack of freshwater river or streams at CKI. In addition, dredging for the new Ruma Baru jetty did not impact coral mortality due to good environmental controls and the completion of the jetty in 2011 some two years before the lagoon mortality event (Department of Regional Australia, Local Government, Arts and Sport 2012). Similarly, pollution is unlikely to have had an impact given the remoteness of CKI and low population (Australian Bureau of Statistics 2015).

With the paucity of other long term in-situ data, such as salinity, dissolved oxygen and turbidity, the exact cause of the 2012/13 lagoon mortality can not be defined. However, the only anomalous event observed over this period was the rainfall event recorded in February 2013. Rainfall and associated reduction of salinity is recognised as a cause of coral reef mortality (Mayer 1914, Vaughan 1914), however it has received little attention compared to thermal bleaching (Berklemans *et al.* 2012). The rainfall event at CKI between the 23rd and 25th February recorded 845mm rainfall over a 72 hour period with the 165km^2 shallow water lagoon and islands potentially acting as a sink for any freshwater. The event also coincided with relatively low diurnal tide movements (0.3 to 0.8m). Therefore, it is probable that the

salinity of the lagoon would have been significantly reduced, however, the depth, spatial extent and exposure time is unknown. In addition, the temperature of the lagoon also dropped by up to 2.5^{o}C during this period, but was still within thresholds, before recovering rapidly to near 29°C over this period.

Although freshwater coral reef mortality has been reported previously, it is generally related to freshwater flood plumes from river outflows (Butler *et al.* 2013), rather than rainfall. Although not as obvious as river discharges, groundwater reservoirs have also been recorded to flow or discharge directly into oceanic waters (Burnett *et al.* 2003). This is a possibility at CKI, which has a fresh ground water lens (described in Jacobson 1976). This may have contributed to a greater reduction of salinity over a prolonged period, particularly in the inner lagoon. It is possible that a combination of these factors contributed to the mortality.

Although spatially robust the monitoring program can not quantify the full spatial extent of coral loss within the lagoon. Personal observations of the authors on subsequent surveys confirm the findings of the monitoring program, with high levels of hard coral mortality observed throughout the middle and inner lagoon. The impact on specific coral genera was not uniform within the lagoon. The genera *Porites* was the only dominant genera not significantly affected within the lagoon, with no mass mortality observed 2012 and 2014. *Acropora* and *Isopora* suffered 100% mortality at the outer lagoon site and significant reductions of *Acropora*, *Echinopora* and *Montipora* were observed in the inner lagoon. This is consistent with similar studies of hyposaline impacts on coral, with *Porites* resistant to hyposaline environments (Blakeway 2004), with other genera, particular *Acropora* being more susceptible (Blakeway 2004, Tan *et al.* 2012, Butler *et al.* 2013, Berkelmans *et al.* 2012). In addition, the authors observed other habitats, including seagrass that have suffered mortality post 2013.

Large scale loss of hard coral cover is of particular concern for remote locations such as CKI, which have limited capacity for external recruitment (Veron 1990). In addition, impacts on lagoon coral can have flow on effects to the reef front and reduce overall resilience of coral communities (Reigl *et al.* 2012). At CKI at least nine catastrophic die off events have been recorded since human colonisation in 1826 with the last reported in 1983 (Bunce 1988). Widespread loss of coral, such as *Acropora*, can have significant impacts on functionally important benthic habitats, which provide refuge and structure for fish stocks. Disturbed ecosystems also have an increased risk of disease that may spread to nearby reefs (Haapkylä *et al.* 2011). The rate of recovery of coral reefs from disturbance is dependent on many factors such as; recruitment ability, water quality, the scale of disturbance, frequency of disturbance and appropriate management (Graham *et al.* 2011). Coral reefs impacted by acute natural disturbance, as suggested at CKI, rather than chronic anthropogenic impacts such as pollution and nutrient runoff, are more likely to recover (Wakeford *et al.* 2008, Graham *et al.* 2011). For example, Butler *et al.* (2013) report an 18 year time period for full recovery from a flood event at a high latitude reef off the Great Barrier Reef. While no studies of coral recovery have been conducted at CKI it is recommended that management agencies maintain long term monitoring programs to detect changes and implement appropriate management

measures that reduce anthropogenic pressures. This is particularly important at CKI where the marine ecosystem is heavily relied upon by the local community for subsistence.

Broad level, cost effective reef monitoring programs allow scientists and managers to detect changes in the marine environment and may provide insight into the potential causes of disturbance. The long term monitoring program described here has shown that it is capable of describing the broad benthic communities and detecting trends in the benthic environment.

3.5 Recommendations

The Department of Fisheries, Western Australia, Research Division, makes the following recommendations with regards to the coral reefs of the Cocos (Keeling) Islands:

- Continue ongoing monitoring on biennial basis to monitor fish habitat structure;

- Encourage research on coral species diversity and abundance and genetics, to provide coral diversity, abundance and resilience estimates for the Cocos (Keeling) Islands;

- Support additional in-situ environmental loggers (e.g. dissolved oxygen) to monitor the impact of environmental conditions; and

- If commercial collection of coral at the Cocos (Keeling) Islands is to be considered the following recommendations are proposed;

  o No harvest of hard coral in the lagoon, to allow recovery of existing stock;

  o A precautionary approach with conservative harvest limits for the outer reef, to assist recovery of lagoon coral communities and ensure future resilience to potential natural anomalous events (e.g. thermal bleaching, disease, cyclones);

  o Before allowing collection of coral from the outer reef a comprehensive survey of abundance and distribution for the proposed harvest species is recommended.

4.0 Status of the distribution and abundance of *Lambis lambis* (gong gong) at the Cocos (Keeling) Islands

4.1 Introduction

The common spider conch *Lambis lambis* (gong gong / spider conch) is a gastropod mollusc of the strombid family. Found throughout the shallow water reefs and intertidal waters of the Indo-Pacific from the east coast of Africa to Tonga, *L. lambis* can reach a maximum shell length of 290mm (Poutiers 1998). The sexes are separate and the species sexually dimorphic, with females significantly larger than males (Poutiers 1998, Beesley *et al.* 1998). Throughout its distribution, *L. lambis* is either overexploited or showing signs of overexploitation due to extensive harvest for food or the shell craft industry (Jagadis *et al.* 2012, Bellchambers and Evans 2013, Mazo *et al.* 2013). *L. lambis* is regarded as a delicacy by the local Malay population of Cocos (Keeling) Islands (CKI) and is the target of recreational fishing pressure.

Although the distribution and general morphology of *L. lambis* has been reported globally, relatively little information exists on the biological traits, life history or fisheries of *L. lambis* (Bellchambers and Evans 2013). Most studies involving *L. lambis* draw comparisons with its close relative, the queen conch (*Strombus gigas*) (Bellchambers and Evans 2013). In the *S. gigas* fisheries even with diverse stock management regulations in place since 1970, some locations have stocks at such low abundance that reproduction is failing due to lack of encounters between males and females (Stoner and Ray-Culp 2000, Tewfik and Guzman 2003, Kramer *et al.* 2009). Similar trends and strict management regulations are now being observed in *Lambis* fisheries worldwide. Jagadis *et al.* (2012) report that in India six of the eleven species of the *Lambis* are categorised under Schedule IV of the Indian Wildlife Protection Act of 1972, meaning commercial exploitation is banned. In Singapore, *Lambis* are listed as vulnerable in the Singapore Red Data book (Davison *et al.* 2008) and research in the *L. lambis* fishery in Guiuan, Eastern Samar, indicates the stocks are also overexploited (Mazo *et al.* 2013). Similar concerns have been highlighted for *L. lambis* at CKI (Hender *et al.* 2001, Bellchambers and Evans 2013).

While recent work on habitat associations (Bellchambers *et al.* 2011), reproduction biology and size at sexual maturity of the *L. lambis* at CKI and in the Philippines (Mazo *et al.* 2013, Bellchambers and Evans 2013) increased the understanding of the species, *L. lambis* were highlighted as the invertebrate species most at risk from overexploitation during a risk assessment process. The Department of Fisheries, Western Australia (DoF) currently maintains an ongoing distribution and abundance survey of *L. lambis* at CKI. This report is an update of research report by Bellchambers and Evans (2013) and reports on the distribution and abundance of *L. lambis* at CKI from the DoF long term monitoring surveys (2007 – 2014).

4.2 Methods

4.2.1 Site selection and survey methods

Field surveys to monitor the distribution and abundance of *L. lambis* in the CKI southern atoll were commenced by DoF in 2007. The 2007 survey was a pilot study focused primarily on

habitats close to Home Island where *L. lambis* were reported to occur in high numbers and the majority of historical fishing had occurred. Historical comparisons were also made to previous studies, including Lincoln-Smith *et al.* (1993) and Hender *et al.* (2001). In 2008, an expanded survey of 67 sites was conducted to ensure greater coverage of shallow water environments (see Bellchambers and Evans 2013). The 2008 data provided a broad dataset to develop and implement a robust and cost-effective monitoring program, with 40 of the 67 sites surveyed in 2008 (Figure 4.1) being used for annual monitoring from 2009 to 2011. This survey marks the first survey in a triennial monitoring program after the initial baseline surveys, and was conducted in March 2014. Surveys were conducted by two observers on SCUBA or snorkel. The observers enter the water on a set mark, separated by ten metres and swim parallel 100 m x 2 m belt transects on a set bearing. A total count of *L. lambis* and the percent cover of broad habitat types (Table 4.1) are recorded at 25 m intervals.

Table 4.1   *L. lambis* **broad habitat categories**

Abiotic – Sand/Rubble	Sub-Massive Corals	Macroalgae
Relic (Dead) Reef	Branching Coral	Hard Macroalgae (*Acanthopora* sp.)
Massive Coral	Seagrass	Filamentous Algae

4.2.2 *L. lambis* relative abundance estimates

Total counts of *L. lambis* were converted to densities of individuals per hectare (ind/ha). Annual density (ind/ha) estimates of *L. lambis* were calculated by the mean of all 40 sites, per 25 m sampling interval (n=320), in each survey year (2008-2011, 2014). Densities by grouped general areas (A to K) (Figures 4.3 and 4.4) were calculated by averaging all 25 m intervals of sites occurring in that area (A to K). At the site level, the density of *L. lambis* (ind/ha) is calculated by averaging the four 25 m intervals between the two observers (n=8), per year.

The analyses were performed in PRIMER-E V6 with PERMONOVA add-on (Clarke & Gorley 2006, Anderson *et al.* 2008). Significant changes in density of *L. lambis* at CKI were tested using PERMANOVA (unrestricted, permutation=9999) with year as a factor, using site density data, square root transformed in a Bray-Curtis resemblance matrix, with dummy value of one (Anderson and Willis 2003). To show localised trends, *L. lambis* densities for each survey year were graphed by site, with sites grouped into general areas (A to K). Significant changes in *L. lambis* densities between years at the general area (A to K) and site level were tested for in pairwise PERMANOVA (unrestricted, permutation=9999) using the density data, square root transformed in a Bray-Curtis resemblance matrix, with two way random factor designs of year and general area and year and site, respectively (Anderson and Willis 2003).
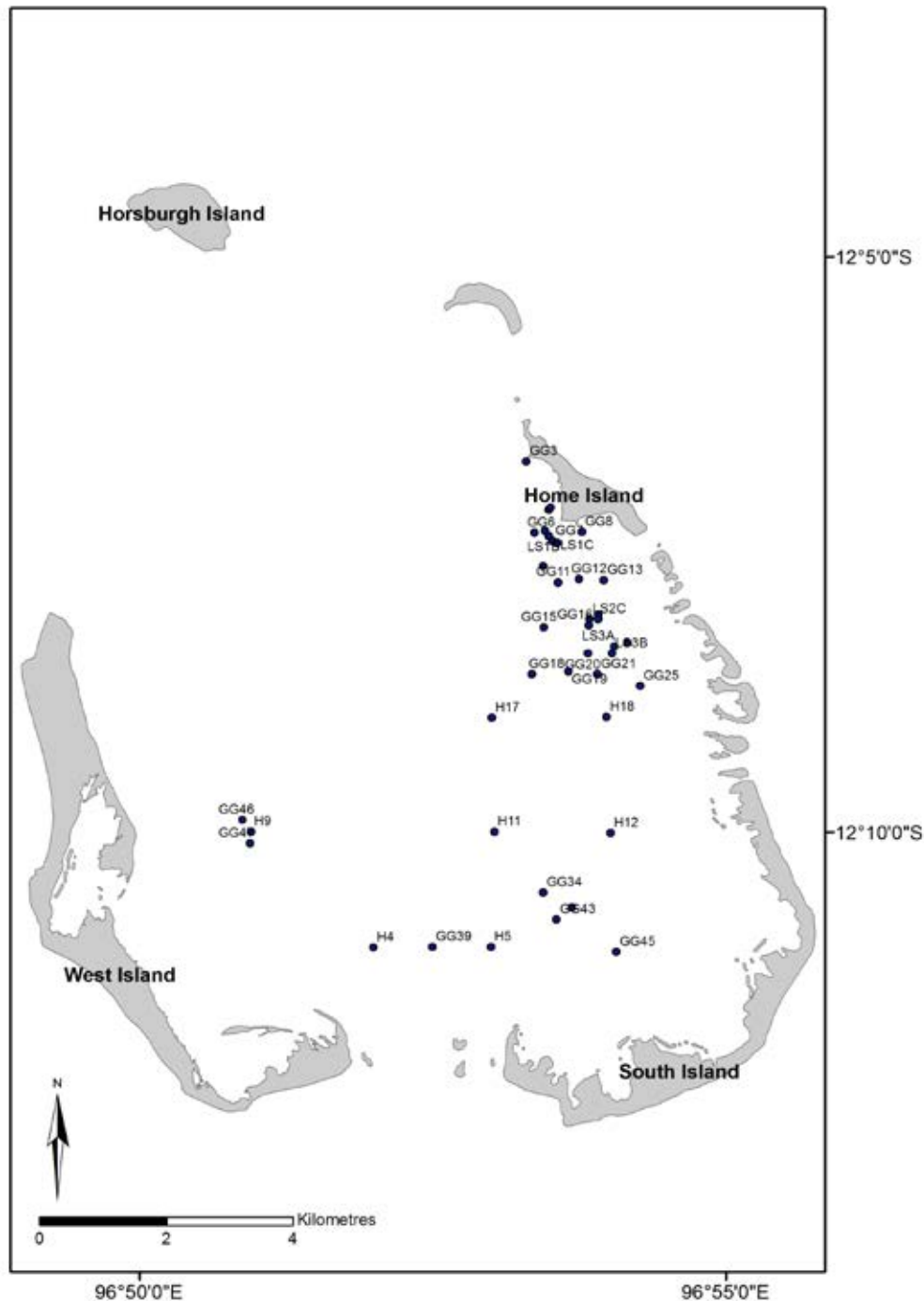
Figure 4.1 The 40 current *Lambis lambis* monitoring sites (LS = original Lincoln-Smith *et al* (1993) [9 sites]; H = original Hender *et al*, (2001) [7 sites]; GG = original DoF sites [24 sites]).

4.2.3 Historical comparisons of *L. lambis* at CKI

Comparisons of the current *L. lambis* densities to previous surveys that used a similar technique and were undertaken in 1992 (Lincoln-Smith *et al.* 1993) are made. Comparisons are made using three of the five sites surveyed by Lincoln-Smith *et al.* (1993) (sites 1, 2, and 3). Each site has three replicates (a, b and c), therefore the mean density of the replicates for

each site from the 1992 survey and subsequent DoF surveys of the same sites (2007-11, 2014) were used to estimate percentage change in *L. lambis* density between 1992 and 2014.

4.2.4 *L. lambis* survey habitat analysis

Previous studies have shown that the distribution and density of *L. lambis* at CKI is closely linked to habitat type with density positively correlated with the percentage cover of *Acanthopora sp.*, macro algae and sub-massive coral (Bellchambers *et al.* 2011, Bellchambers and Evans 2013). The habitat composition per site was calculated by averaging the percent cover per 25 m interval from both observers of the nine habitats categories (Table 4.1). An overall CKI habitat composition was measured by averaging the overall composition of each habitat category, from all sites. To examine if changes had occurred between previous surveys and 2014, PERMANOVA and pairwise PERMANOVA were used. Both tests used year and site as factors (unrestricted, permutation=9999) on untransformed percentage cover data for all survey years on a Bray-Curtis resemblance matrix with a dummy value added (Anderson and Willis 2003). Only sites that were surveyed in 2008 and 2014 (n=40, Figure 4.1) were used to test for significant differences in habitat. PERMANOVA (unrestricted, permutations = 9999) with year as a factor, was performed on the habitat categories by selecting individual habitats within the sub-sampled 2008 and 2014 data, from the untransformed percentage cover data and performing a Bray-Curtis resemblance matrix with a dummy value added.

4.3 Results

4.3.1 Relative abundance estimates

A total of 665 *L. lambis* were observed at 33 of the 40 sites surveyed in 2014. The mean density of *L. lambis* at CKI in 2014 was 415.6 ±111.7 SE ind/ha, which is significantly ($p<0.0001$) lower than previous years (Figure 4.2). A significant interaction (p=0.0057) was observed between 2009 and 2010, indicating a significant reduction in abundance between those years (Figure 4.2).
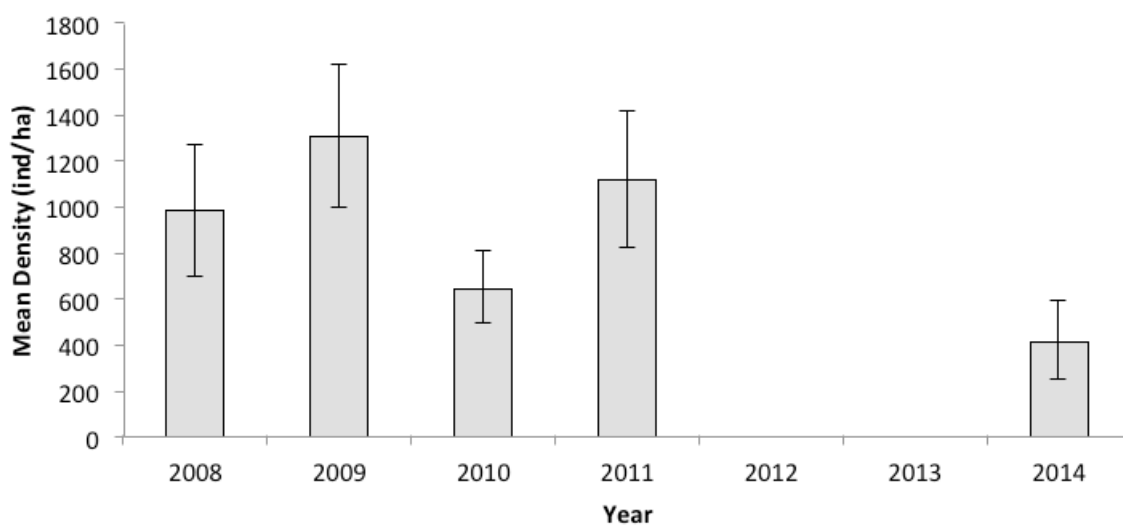
Figure 4.2 Mean densities of *L. lambis* (ind/ha ± SE) from the 40 sample sites (2008 to 2011, 2014). No surveys were conducted in 2012 and 2013.

4.3.2 *L. lambis* density by general area

In 2014, the density of *L. lambis* was the lowest recorded during the monitoring program ranging between 0 to 3625 ind/ha (Figure 4.3 and 4.4). The number of sites with a high density was also the lowest in 2014 with only five sites recording a density >1000 ind/ha (sites GG19, GG34, GG43, H12 and LS2A) compared to 11, 19, 10 and 12 sites in 2008, 2009, 2010 and 2011 respectively (Figures 4.3 and 4.4). Significant ($p<0.05$) changes in density were observed at 9 of the 11 general areas (A to K) during the monitoring program with only general areas 'A', 'G' and 'H' not displaying significant ($p<0.05$) changes in relative abundance (Figures 4.3 and 4.4).

Between 2011 and 2014 none of the 11 general areas (A to K) (Figures 4.3 and 4.4) observed increases in *L. lambis* density. Significant decreases ($p<0.05$) in *L. lambis* density were observed in general areas 'B' (1130 to 70 ind/ha), 'C' (1125 to 237.5), 'F' (2575 to 683.3 ind/ha), 'J' (993.73 to 281.25 ind/ha) and 'K' (1058.33 to 108.33 ind/ha).
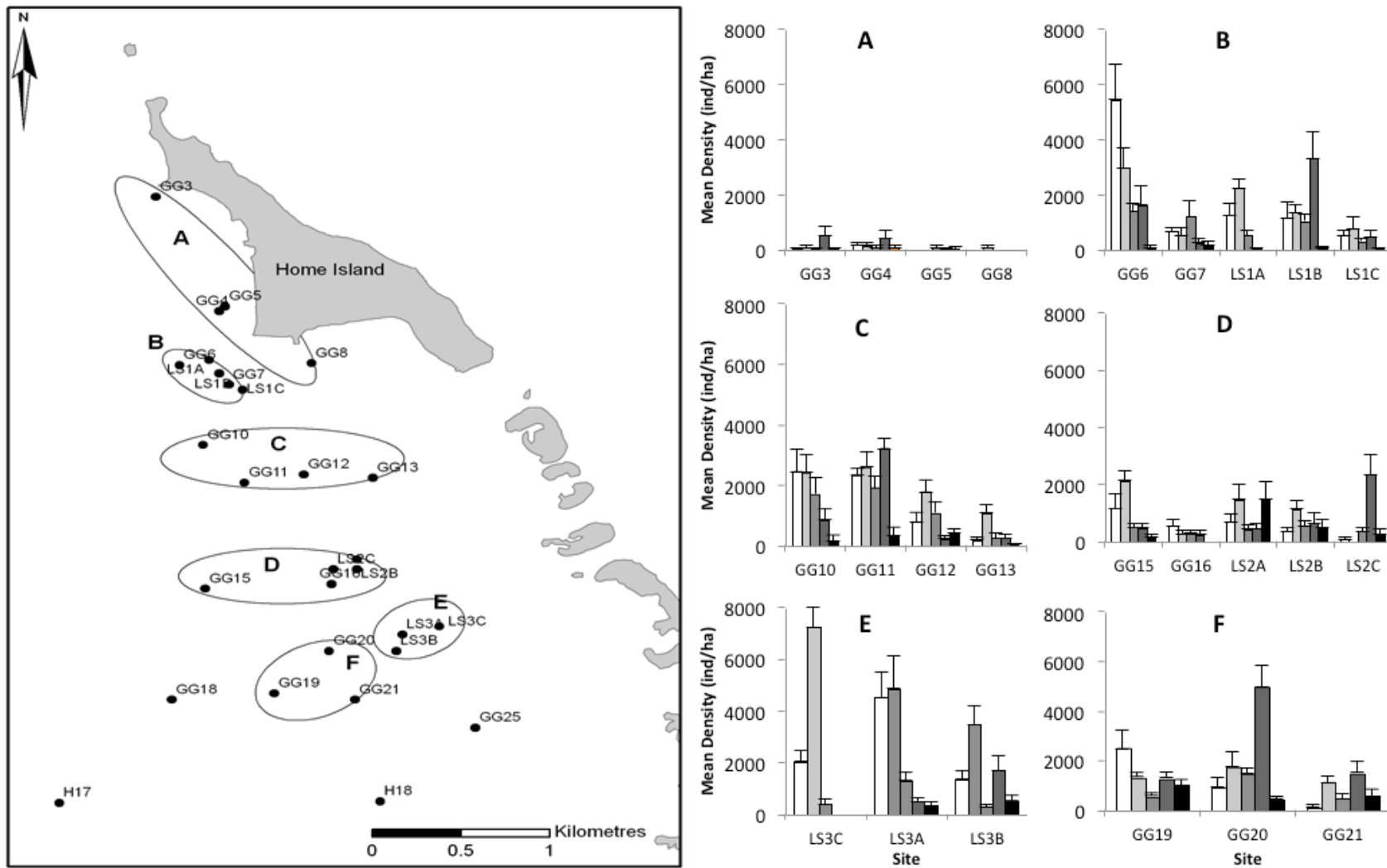
Figure 4.3   General Areas A – F with a time series of density of *L. lambis* (ind/ha) for each site.
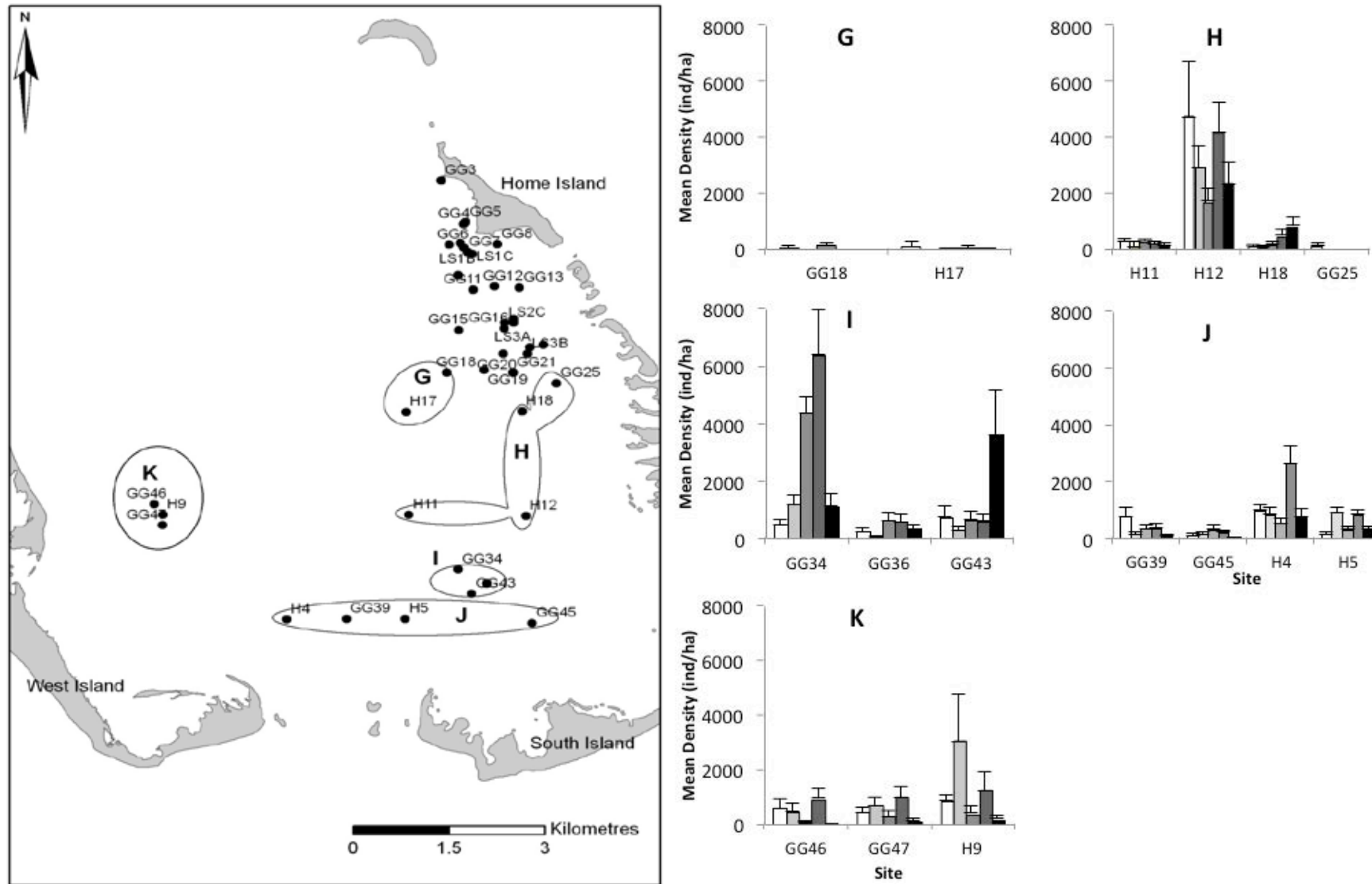
Figure 4.4 General Areas G – K with a time series of density of *L. lambis* (ind/ha) for each site

4.3.3 Historical abundance comparison of *L. lambis*

In 2014, the density of *L. lambis* at the three sites originally sampled by Lincoln-Smith *et al.* (1993) was 358.3 ±91.4 SE ind/ha, which is a 94.0% decrease from the original survey (5925.0 ±1268.8 SE ind/ha). However, the largest decline in density occurred at LS3 (10291.7 ±2384.0 SE ind/ha in 1992, decreasing 97.3% to 275.0 ±98.4 SE ind/ha in 2014 Figure 4.5). The decline in *L. lambis* at these sites is such that in 2014 the *L. lambis* population across all three sites is only 6.0% of the observed relative abundance of 1992.
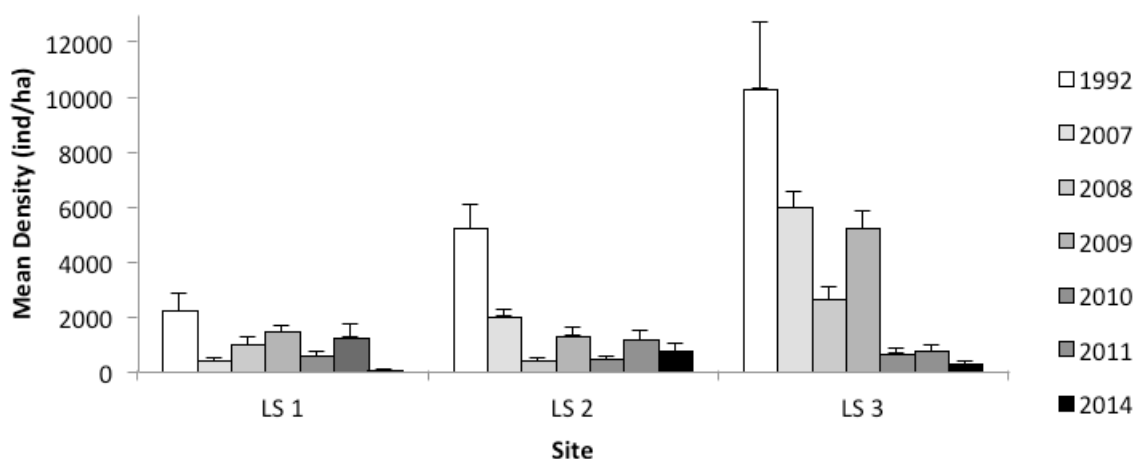
Figure 4.5 Relative abundance of *L. lambis* (ind/ha ± SE) comparisons at three locations (each containing 3 sites/replicates) sampled in 1992 (Lincoln-Smith *et al*, 1993) and DoF surveys (2007 - 2014).

4.3.4 *L. lambis* habitat

There was a significant change in habitats over the survey period (p=0.0001) and between sites in 2014 (p=0.0001). There was also a significant difference (p=0.001) between the 2008 and 2014 survey. There were significant differences (p<0.05) amongst four of the nine broad habitat types, with significant decreases in massive coral (~50%), sub-massive coral (~45%), and filamentous algae (87%). In contrast, there was a significant (p=0.0003) increase in relic (dead) reef (59%) (Table 4.2). There were no significant differences in hard macro algae (*Acanthopora sp.),* macro algae or seagrass.

Table 4.2 Mean percent cover of habitats at study sites *L. lambis*.

	Abiotic – sand/rubble	Relic (dead) reef	Massive coral	Sub-massive coral	Branching coral	Seagrass	Macro algae	Hard macro algae	Filamentous algae
2008	44.7	3.06	3.45	7.25	3.69	2.42	18.34	12.34	4.73
2014	45.75	6	1.72	4.01	3.39	1.34	25.47	11.48	0.59
p value	0.4327	0.0003	0.0129	0.0001	0.475	0.232	0.073	0.680	0.0015

4.4 Discussion

L. lambis stocks at CKI have declined 94% from levels recorded by Lincoln-Smith *et al.* (1993). Significant decreases over shorter time periods (3 years) have also been observed. Despite significant declines in abundance *L. lambis* continues to be the target of substantial fishing pressure with fishing regulations still to be legislated (Berry 1989, Bellchambers and Evans 2013).

While the density of *L. lambis* increased slightly at some sites in 2014, the majority of sites displayed decreases. In addition, the distribution of densities reported by Bellchambers and Evans (2013) are not evident. The absence of clear patterns in density may be due the low abundance of *L. lambis* at the majority of sites during the 2014 survey (only five sites had densities > 1000 ind/ha). Despite the overall decline in density, distribution of *L. lambis* was not uniform throughout the lagoon as some sites displayed significant increases in density (e.g. GG 43), which may in part be due to selective harvesting by fishers (Bellchambers and Evans 2013). However, the number of sites with significant increases in density is decreasing, which may indicate that selective fishing is no longer viable due to low overall abundances.

In addition, there has also been a significant mortality event that caused mass losses in hard coral communities particularly of the lagoon between 2012 and 2013. Habitat surveys conducted in 2008 and 2014 show a significant change in the distribution and percent cover of some habitats e.g. >50% declines in massive and sub-massive corals and filamentous algae. There was a significant increase in relic (dead) reef, while no significant change was observed in hard macro algae (*Acanthopora* sp.) and macro algae habitats. Bellchambers and Evans (2013) reported the *L. lambis* is a herbivore and detritivore that favours hard macro algae (*Acanthopora* sp.) and macro algae for food and or/shelter, which is consistent with other *L. lambis* fisheries (Mazo *et al.* 2013). It is likely that the massive and sub-massive coral habitats are used for shelter rather than food, therefore *L. lambis* are now using relic reef for this purpose. *L. lambis* are slow moving and also generally located in shallow (<2m) areas of the lagoon. While benthic structure and food are still available, the event that caused the 2012/13 widespread lagoon hard coral mortality may have also caused the mortality of *L. lambis*. Although there were no recent reports of a wide spread fish kill at CKI, any large-scale mortality of this cryptic benthic species may not have been highly visible. The timing of the proposed anomalous event in late February 2013 coincides with the spawning of *L. lambis* (Bellchambers and Evans 2013, Mazo *et al.* 2013) which may have also impacted recruitment.

Regardless of the contributing factors, significant declines in *L. lambis* were observed in 2014 that are clearly not representative of annual or seasonal variation and the abundance of *L. lambis* at CKI is now critically low. The ecological role of *L. lambis* worldwide it is also still not understood, therefore it is difficult to predict the potential impact a reduction or loss of the species would have on the wider ecosystem.

4.5 Recommendations

The Department of Fisheries, Western Australia, Research Division, makes the following recommendations with regards to the *Lambis lambis* stocks of the Cocos (Keeling) Islands:

- Reduce fishing pressure on the *L. lambis* at CKI to ensure stocks do not decrease to where recovery is not possible;

- Continue the current DoF monitoring program of *L. lambis* densities on a biennial or triennial basis;

- Increased knowledge of spawning, size at sexual maturity, habitat associations, movement and ecology of *L. lambis* at CKI is needed to inform future stock management arrangements; and

- A program to quantify recreational harvest of *L. lambis* would be useful to inform management

5.0 Abundance and distribution of giant clams at the Cocos (Keeling) Islands

5.1 Introduction

Giant clams are bivalve molluscs in the subfamily Tridacninae within the family Cardiidae (Schneider and O'Foighil 1999, WoRMS 2015). With the recent description of two new giant clam species (*Tridacna ningaloo*; Penny and Willan 2014 and *Tridacna noae*; Su *et al*. 2014) there are now twelve identified species of giant clams, ten of which are within the *Tridacna* genus; *T. crocea* [Lamarck 1819], *T. derasa* [Röding 1798], *T. gigas* [Linnaeus 1758], *T. maxima* [Röding 1798], *T. mbalavuana* [Ladd 1934], T. *ningaloo* [Penny and Willan 2014], *T. noae* [Röding 1798], *T. rosewateri* [Sirenko and Scarlato 1991], *T. squamosa* [Lamarck 1819] and *Tridacna squamosina* [Sturany 1899]) and two in the genus *Hippopus*; *Hippopus hippopus* [Linnaeus 1758] and *H. porcellanus* [Rosewater 1982]. Global distribution of giant clams varies between species, however they are generally found within the Indo-West Pacific region (Rosewater 1965, Harzhauser *et al*. 2008). Currently, the most widespread species, *T. maxima,* occurs in an area bounded by East Africa and the Red Sea to the west, southern Japan to the north and Polynesia to the east (Knop 1996). Newly described species *T. ningaloo* and *T. noae,* are morphologically similar to *T. maxima* and have similar distributions (Huelsken *et al*. 2013, Borsa *et al*. 2014, Penny & Willan 2014, Su *et al*. 2014).

Giant clams are filter feeders that host symbiotic photosynthetic dinoflagellate known as zooxanthellae within the mantle tissue (Knop 1996). When the mantle is exposed to sunlight giant clams can obtain almost 100% of their dietary requirements through the symbiotic relationship. However, the dependence on photosynthesis restricts the distribution of giant clams to shallow (< 20m) or oligotrophic waters (Braley 1989, Klumpp *et al*. 1992, Munro 1992). Giant clams are protandrous simultaneous hermaphrodites, which first reach sexual maturity as males and then develop female gonads to simultaneously function as both male and female (Nash *et al*. 1988). They are broadcast spawners and the release of sperm usually precedes the release of ova (Nash *et al*. 1988, Munro 1992). Once settled, the growth rates of giant clams are typically slow with most species reaching maturity within 5 – 7 years (Kinch and Teitelbaum 2010). The largest species of giant clam, *Tridacna gigas*, is the largest living bivalve with a shell length > 120 cm (Rosewater 1965).

Giant clams are the basis of important fisheries throughout their distribution. They are harvested in many Indo-Pacific countries to meet demand for their meat, shell and also for aquarium industries (Lucas 1994, Neo *et al*. 2015). Due to their high meat content, *Tridacna gigas* and *T. derasa* are the most sought after species, while *T. maxima* and *T. crocea* are popular in the aquarium industry due to the variations in mantle pattern and colouration (Mies *et al*. 2012). Giant clams are susceptible to overharvesting due to their biological characteristics e.g. slow growth, low reproductive success, long planktonic stage, and susceptibility to mass mortality events (Alder and Braley 1989, Penny and Willan 2014). They are also susceptible to natural and anthropogenic changes in the environment such as habitat loss, increased nutrients, pollution and overharvesting (Hoegh-Guldberg 1997, Elfwing *et al*. 2001). Overharvesting of several species of giant clam has been reported in a

number of countries e.g. the Philippines, Malaysia, Indonesia, Micronesia, Taiwan and Singapore (Lucas 1994, Tan and Zulfigar 2003, Guest *et al*. 2008, Su *et al*. 2014). As a result of these natural and anthropogenic pressures all species of giant clam, with the exception of the two recently described species, are currently listed on the International Union for Conservation of Nature Red List of Threatened Species (IUCN 2015) and on Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (UNEP WCMC 2015).

Previously, three species of giant clam have been reported at the Cocos (Keeling) Islands (CKI); *T. gigas*, *T. derasa* and *T. maxima* (Maes 1967, Berry 1989, Wells 1994). However, *T. gigas* has not been observed recently and is currently considered to be locally critically endangered or extinct (Hourston 2010, Bellchambers and Evans 2013). There is currently no commercial fishery for giant clams at CKI and the depletion of *T. gigas* appears to be a direct result of subsistence harvesting as giant clams are a popular food source for the local population (Hourston 2010, Bellchambers and Evans 2013). With the depletion of *T. gigas* stocks at CKI there was concern that fishers would further target *T. derasa* stocks which are also critically low, with only one *T. derasa* reported out of a total of 1,885 individual giant clams observed at CKI in 2011 (Bellchambers and Evans 2013).

The baseline survey by Bellchambers and Evans (2013) was the first to comprehensively document the abundance, distribution and size frequency of giant clam at CKI, with no other historical catch records for giant clams at CKI. As such the long term level of stock depletion is not quantifiable. The depletion of giant clam stocks can also be accelerated by natural impacts, such as thermal bleaching (Andréfouët *et al.* 2013) or other anthropogenic impacts, such as historical collection for ship ballast. The Bellchambers and Evans (2013) baseline survey has now been modified into a long term monitoring program to quantify changes in species abundance, distribution and size frequency of giant clams. This reports updates Bellchambers and Evans (2013) and reports on species, abundance, distribution and size frequency of giant clams at CKI from the DoF long term monitoring surveys.

5.3 Methods

5.3.1 Site selection

In 2011, DoF surveyed giant clams at 77 sites at CKI, using a random stratified sampling design (Quinn and Keough 2002, Bellchambers and Evans 2013). Sites were grouped into six habitat strata as described by Williams (1994). An additional site in 'The Rip', a no-take area, was also surveyed. The 2014 survey repeated 70 of the 78 sites that were surveyed in 2011 due to unfavourable weather (Table 5.1 and Figure 5.1).

Table 5.1   **Marine habitats and map units of the Cocos (Keeling) Islands (after Williams 1994).**
* denotes habitats which are marginally suitable but were included in the analysis

Habitat System	Habitat	Map Unit	Suitable / Not Suitable	Strata	Area (km^2)	No. Sites
Outer Reef	Coral Terrace	A	Suitable	Strata 1	21.86	14
Reef Flat	Coral and Algal Flat	B	Suitable	Strata 2	9.61	8
	Coral Flat	C	Suitable	Strata 2	0.10	1
	Aligned Coral Flat	D	Suitable	Strata 2	2.49	3
	Seagrass Flat (*Thalassia hemprichii*)	E	Not Suitable	Strata 3	0.27	1
Lagoon	Prograding Sand Sheet	F	Not Suitable	Strata 3	8.72	3
	Intertidal Sand and Silt Flat	G	Not Suitable	Strata 3	5.47	1
	Seagrass Sand and Silt Flat	H	Not Suitable	Strata 3	12.95	3
	Seagrass Bed (*Thalassodendron ciliatum*), Seagrass Bed (*Syringodium isoetifolium*) and Seagrass Flat (*Thalassia hemprichii*)	I,T and U	Not Suitable	Strata 3	2.68	1
	Coral and Algal Flat	J	Suitable*	Strata 4	17.69	9
	Blue Hole Mosaics 1, 2 and 3	K,L,M	Suitable	Strata 5	16.60	4
	Algal Covered Staghorn Rubble	N	Suitable	Strata 5	13.15	6
	Massive Coral Outcrops	O	Suitable	Strata 6	12.63	9
	Emergent Reef	P	Suitable	Strata 2	0.06	1
	Sandy Lagoon Floor with occ. Coral Outcrops	Q	Suitable	Strata 6	6.85	3
	Sand Shoal	R	Not Suitable	Strata 3	0.19	1
	Sandy Lagoon Floor with Scattered Coral Outcrop and Seagrass Beds	S	Suitable	Strata 6	2.33	1
The Rip	No-Take Area	N/A	Protected	N/A		1

5.3.2 Development of a cost effective monitoring program for giant clams

The results from the 2011 DoF baseline survey were analysed to assess the impact of reducing the number of paired belt transects from two replicates per site to one (see Bellchambers and Evans 2013). An F-test was used to assess variation in giant clam density between replicates within sites. The F-test showed that the variances were unequal ($p \leq 0.001$). Therefore, a T-test assuming unequal variance was performed to test the variance between the replicates within each site. These tests were performed on the 2011 dataset with all sites included and then again on the dataset excluding any sites on which giant clams were not observed. Both of the T-tests performed on the 2011 dataset indicated that there was no significant difference in the density between the replicates at each site (all sites $p=0.44$; sites where clams were recorded $p=0.40$).

A one way ANOVA was performed on the 2011 data to compare giant clam length measurements between the two replicate transects per site with no significant difference detected ($F_{1,1348}=2.25$, $p=0.13$).

As there were no significant differences in densities or size frequency of giant clams between replicates per site, replicate transect 'A' from 2011 was surveyed at each site in 2014.

5.3.3 Survey method

Surveys were conducted between the 14th November and 2nd December 2014 by SCUBA, snorkel or reef walks in depths ranging from intertidal to 15 m. At each site, a paired transect (50 x 2 m belt transect) was surveyed by two observers spaced approximately 5 m apart and sites were surveyed on the same bearing as the 2011 survey. The abundance of giant clams and percentage cover of broad habitat types were recorded at 10 m intervals. All giant clams were identified to species level, unless the defining morphological features were undeveloped, eroded or unable to be seen in which case it was recorded as 'unknown'. The first 20 giant clams on each transect were measured from apex to apex to the nearest millimetre using callipers.
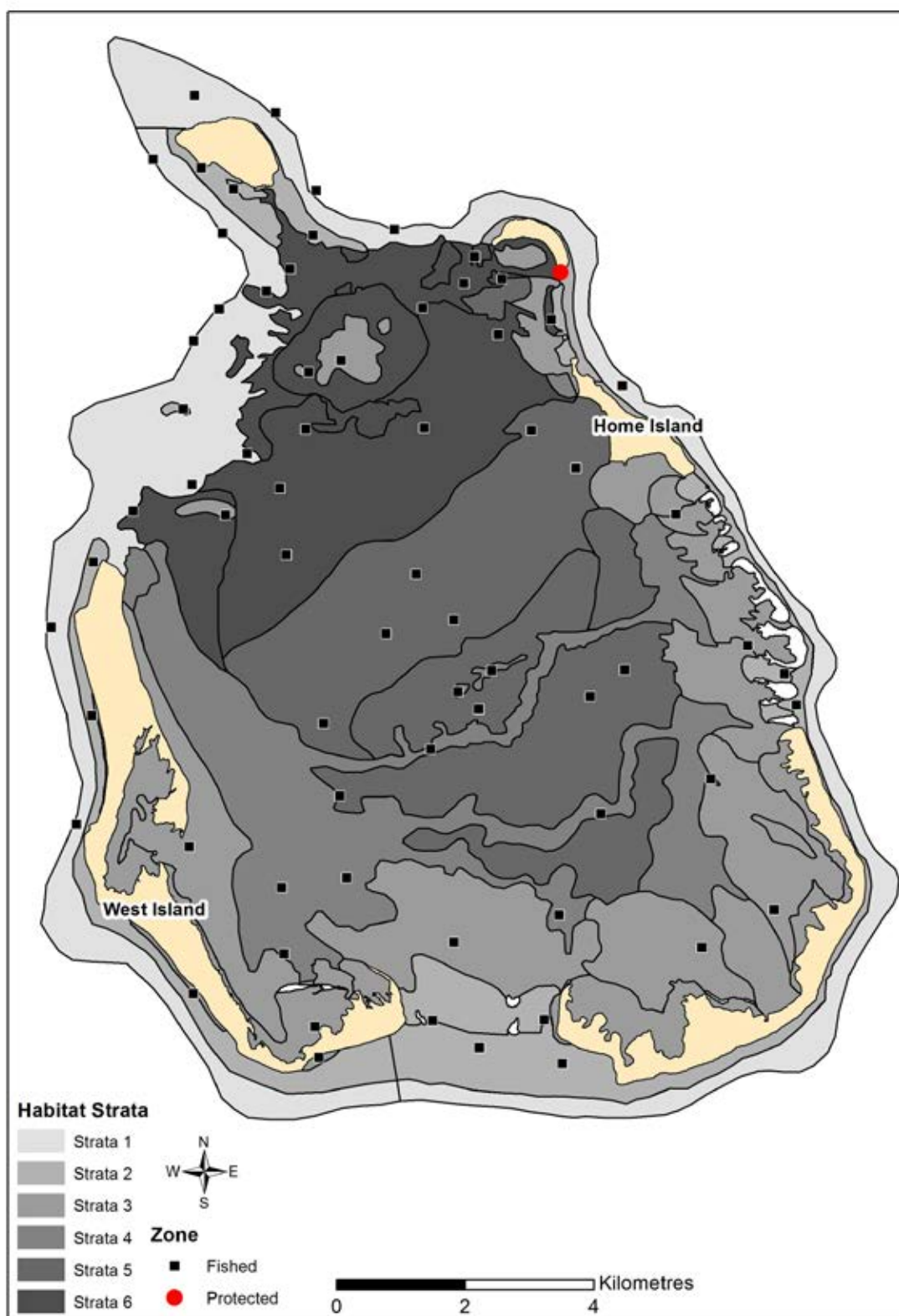
Figure 5.1 Giant clam survey sites (n=70) at the Cocos (Keeling) Islands showing DoF defined habitat strata (Table 5.1). Black squares represent survey sites in fished areas and the red circle the protected survey site.

5.3.4 Statistical analysis of giant clam density

The densities of giant clams (per m^2) surveyed in 2011 and 2014 were compared for each of the six habitat stratum and overall for CKI. Analyses were performed in PRIMER-E V6 with PERMONOVA add-on (Clarke and Gorey 2006, Anderson *et al.* 2008). 'The Rip' could not be compared between survey periods due to the small sample size (*n*=1) and was excluded from all further analysis.

Changes in the density of giant clams between 2011 and 2014, were tested using permutational analysis of variance (PERMANOVA) (unrestricted, permutation=9999) in PRIMER-E V6. PERMANOVA was undertaken using a Bray-Curtis distance similarity matrix (with dummy value of one added) on square root transformed site density data, with year as a factor. Square root transformed data was used to reduce the contribution of the most abundant densities (i.e. zero). To test for significant changes in densities of giant clams within habitat strata between years, a two way random design in PERMANOVA of mean densities of each of the six stratum and year surveyed was also conducted (Anderson *et al.*, 2008). Pairwise comparisons were used to detect any habitat strata which recorded significant changes between years.

5.3.5 Statistical analysis of the size frequency distributions of giant clams

A one-way ANOVA was used to compare the length measurements of *T. maxima* between 2011 and 2014 in the fished area (n=69) and the no-take area of 'The Rip' (n=1). Comparison between years was also conducted for three of the six habitat strata (1 [n=14], 2 [n=13] & 6 [n=13]) (Table 5.1 and Figure 5.1). The remaining strata (3, 4 and 5) were not compared between years due to low abundances.

To display the size frequency distribution of the fished *T. maxima* population the data from each site within the fished area (n=69) was pooled, grouped into 10mm size classes and plotted. This was repeated for the three strata used in analysis (1, 2 and 6) and 'The Rip'. As with Bellchambers and Evans (2013), for the purpose of this report 150mm was chosen as the size for full sexual maturity based on an average of published estimates (Green and Craig 1999, Chambers 2007, Apte *et al.* 2010).

A one-way ANOVA was used to compare length measurements between the three abundant strata (1, 2 and 6) and 'The Rip' in 2014. Tukey-Kramer post hoc analysis was performed to identify where significant (*p*<0.05) difference in means occurred between strata (McDonald, 2014).

5.3.6 Standing stock estimates

The abundance of giant clams in each habitat stratum was estimated based on the methods of Hesp *et al.* (2008). Densities recorded in transects (n=69) were assumed to conform to a delta-log normal distribution, therefore a parametric resampling analysis was used to evaluate uncertainty in abundance estimates. For this, 10 000 random values for the proportion of transects per strata that recorded individuals (non-zero densities) $P*$ and 10 000 random values for the mean of the $\log_e$ transformed non-zero densities for transects, $\overline{x}^*$ $\overline{x}^*$ were drawn

from binomial and normal distributions, respectively. The values of $\bar{x}^\bullet$ were drawn using the equation:

$$\bar{x}^\bullet = \bar{x} + SE.r$$

Where SE is the standard error of the non-zero values, and r is a random normal variate. Each value of $\bar{x}^\bullet$ was then back-transformed and corrected for bias, using the equation:

$$E_{corr} = E_{uncorr}.0.5.\sigma^2$$

Where:

E_{uncorr} is the back-transformed estimate prior to bias correction,

E_{corr} is the bias-corrected estimate flowing back-transformation and

σ^2 is the variance on the log-e transformed values.

Each of the 10 000 values of E_{corr} was then multiplied by a value of P^* to produce an estimate of mean density which was then multiplied by the total area of the stratum to obtain 10 000 estimates of the total abundance of giant clams in each stratum. The point estimate and lower and upper 95 % confidence limits for each stratum were taken as the median 2.5 and 97.5 percentiles, respectively, of the 10 000 values for total abundance. The estimates of total abundance for the six strata were then summed to allow estimation of the overall abundance of giant clams at CKI. All calculations were undertaken in Microsoft Excel.

Changes in the estimates of standing stock between 2011 and 2014 were undertaken based on the 95% confidence intervals reported in Bellchambers and Evans (2013) and mean densities observed in 2014.

5.3.7 Current harvest estimates

Census data for CKI has not been updated since 2011 (Australian Bureau of Statistics 2011) therefore the harvest estimates described in Bellchambers and Evans (2013) are still applicable. Several estimates of recreational giant clam catch were calculated assuming a fishing population of 108 people fishing once or twice a week with 5, 10 or 20 giant clams collected per trip.

5.3.8 Sustainable harvest estimates

The annual sustainable harvest of mature giant clams at CKI for 2014 was calculated using the formula (see Bellchambers and Evans 2013):

$$P_{opt} = 1 - \text{exponential} \ (-F_{opt})$$

Where:

P_{opt} is quantity (median standing stock and upper and lower 95% confidence intervals) of the mature population observed and

F_{opt} is the optimal fishing exploitation rate (Pauly 1984).

The fishing exploitation rate was calculated using the formula:

$$F_{opt} = 0.6 \ \text{x} \ M,$$

Where:

M equals natural mortality (Perry *et al.* 1999).

Green and Craig (1999) estimated the natural mortality of *T. maxima* as 0.3; however, this estimate is not based on actual data, but uses an empirical relationship between natural mortality and the mean environmental temperature.

The total instantaneous mortality rate (Z) can be used as a surrogate for natural mortality if fishing pressure is low. Total instant mortality (Z) is equal to the sum of the natural mortality (M) plus the fishing mortality (F), represented by the equation:

$$Z = F + M$$

On Ningaloo Reef in Western Australia, where giant clams are protected, total mortality (Z) has been calculated for *T. maxima* as 0.226 (Black *et al.* 2011).

Both the estimate of natural mortality (M=0.3) and total mortality (Z=0.226) were used to provide estimates of sustainable harvest of giant clams. The average of these mortality rates (0.263) was considered as a reasonable estimate of all mortality rates (Dr Anthony Hart, Principle Research Scientist, Mollusc Section, Department of Fisheries, Western Australia *pers comm.*) and was used for calculating estimates of sustainable harvest for giant clams.

5.4 Results

5.4.1 Giant clam diversity

Only *T. maxima* were conclusively identified in the 2014 surveys. With no other species of giant clams identified this report focuses only on *T. maxima*.

Of the 69 fished sites surveyed in 2014 755 individuals were identified as *T. maxima* (96%). Thirty three individuals were classified as unknown (4%), due to the shell being embedded in the substrate, shell features eroded and/or the shell having undeveloped features. An additional, 165 *T. maxima* were recorded in 'The Rip', therefore a total of 953 individuals were observed over 70 sites.

5.4.2 Average density of *T. maxima*

The average density of *T. maxima* at CKI was 0.055 clams per m^2 (SE ±0.0179, n=69) excluding the 'The Rip' or 0.066 clams per m^2 (SE ±0.0233, n=70) including 'The Rip'. Average density was not significantly different from 2011 (0.054 clams per m^2 ± 0.017 SE, $n = 69$; $p = 0.8645$) (Table 5.2).

The no-take site of 'The Rip' recorded the highest density of *T. maxima* in 2014 (0.825 clams per m^2, n=1) and 2011 (1.055 clams per m^2, n=1) (Table 5.2). Statistical comparisons for 'The Rip' were not possible due to the small sample size. Within the six fished habitat strata (Figure 5.1) strata 2 (reef flats) and strata 6 (coral outcrops) had the highest abundance of *T. maxima,* 0.112 ±0.046 SE and 0.0992 ±0.058 SE respectively (Table 5.2). Densities in the remaining habitat strata varied between 0 and 0.055 *T. maxima* per m^2. However, densities of *T. maxima* had high standard errors, which is consistent with species that have patchy

distributions. Mean densities of *T. maxima* by habitat strata were not significantly different (*p*<0.05) between 2011 and 2014 (Table 5.2)

Table 5.2   Mean density of *T. maxima* by strata for the two survey periods.

Strata	Habitat	Mean density of *T. maxima* (sq m^2 ± SE) 2011	2014	*p* value	*N*
Strata 1	Coral terrace	0.055 ± 0.028	0.070 ± 0.049	0.8845	14
Strata 2	Reef flat	0.120 ± 0.065	0.112 ± 0.046	0.4606	13
Strata 3	Seagrass/sand	0.007 ± 0.009	0.003 ± 0.005	0.3134	10
Strata 4	Coral/algal Flat	0.002 ± 0.003	0.003 ± 0.004	0.7278	9
Strata 5	Algal covered rubble	0 ± 0	0 ± 0		10
Strata 6	Coral outcrops	0.099 ± 0.044	0.099 ± 0.058	0.2466	13
Overall	Fished	0.0536 ± 0.0168	0.066 ± 0.018	0.8845	69
The Rip	Protected	1.055	0.825		1

5.4.3 Standing stock of *T. maxima*

The total standing stock of *T. maxima* in 2014 was calculated at 5,935,040 individuals for the fished area of CKI (~133.96 km^2) (Table 5.3) which overall is not substantially different to 2011 (6,916,269 individuals). However, several strata displayed changes in abundance in 2014 e.g. strata 2 (reef flats) decreased by 1,000,000, strata 6 (the lagoon coral outcrops) also decreased by ~400,000 clams, while strata 1 (outer coral terrace) increased by 383,562 individuals.

Table 5.3   Standing stock of T. maxima in fished areas of Cocos (Keeling) Islands.

Strata	Median (No.of individuals)	95% Lower Confidence Interval	95% Upper Confidence Interval	Area (km^2)
Strata 1	1,308,131	596,571	2,776,337	21.86
Strata 2	1,938,851	757,935	4,876,955	12.20
Strata 3	145,148	0	912,714	30.27
Strata 4	50,394	0	151,181	18.14
Strata 5	0	0	0	29.75
Strata 6	2,492,516	1,043,358	5,751,475	21.74
Total	5,935,040	2,397,864	14,468,662	133.96

5.4.4 Size frequency of *T. maxima*

Length measurements were recorded for 532 of the 755 *T. maxima* observed in the fished area of the CKI in 2014. Lengths of *T. maxima* in the fished area ranged from 13 to 270 mm with a median of 105 mm and a mean of 102.20mm ±2.10 SE. No significant difference between the mean lengths of *T. maxima* in the fished area between the 2011 and 2014 surveys was observed ($F_{1,1099}$ = 0.22, *p* = 0.64, n = 69). Length measurements from within the no-take area of 'The Rip' showed that the mean length of *T. maxima* was 133.67mm ±6.66 SE. No significant difference was found in the size frequency of *T. maxima* in the 'The Rip' between 2011 and 2014 ($F_{1,80}$ = 0.28, *p*=0.59, n=1). On a finer spatial scale, no significant differences

between the lengths of *T. maxima* measured in 2011 and 2014 were found within any of the three strata with enough abundance of *T. maxima* for analysis (strata 1, 2 and 6).

In 2014, 15.8% of measured *T. maxima* were fully mature (≥ 150 mm), 66.5% were sub-adult (51 - 149 mm) and 17.7% were juveniles (≤ 50 mm). The no-take area of 'The Rip' has a higher proportion of mature adults (33.3%)

T. maxima measured in the fished areas of CKI display a decline in abundance after reaching 130 mm (Fig. 5.2 and 5.3). Figure 5.2 also displays bimodal distribution due to a high proportion of *T. maxima* 20 – 60 mm and 100 – 130 mm.
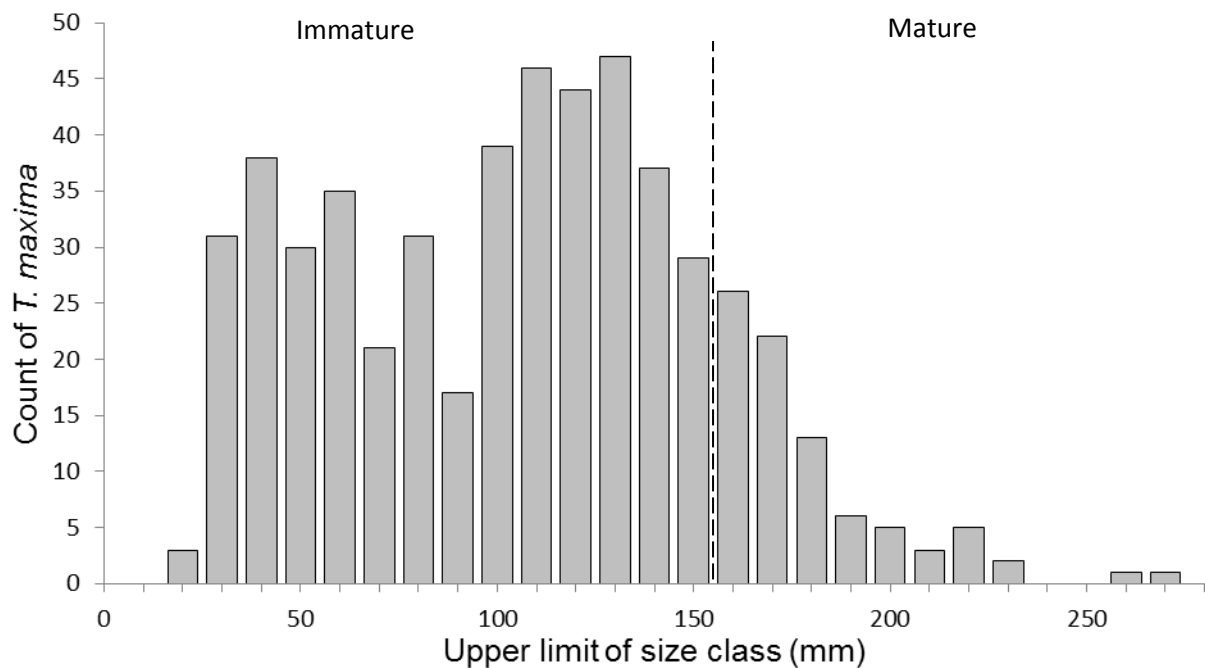
Figure 5.2 Size frequency of *T. maxima* in the fished area in 2014 (10mm size classes). Dashed line indicates the estimated size of full maturity (> 150 mm).

A significant change occurred between the mean lengths of *T. maxima* measured within the three habitat strata that observed higher abundances of giant clam and 'The Rip' in 2014 ($F_{3,564}$ = 12.87, *p*<0.0001, n=69). Tukey-Kramer (*p*<0.05) post hoc analysis showed that strata 1 (coral terrace) had significantly smaller *T. maxima* than the remaining strata (2 and 6) and 'The Rip'. In addition 'The Rip' had significantly (*p*<0.05) larger *T. maxima* than the remaining groups (strata 1, 2 and 6) (Figure 5.3). Overall in the fished areas of CKI declines in abundance begins to occur at 130 mm with few individuals surviving to 200 mm (Figure 5.3). The habitat strata 2 (reef flat) shows the most dramatic decrease of sub-adult populations of *T. maxima* with abundance decreasing after reaching 120 mm. The protected area of 'The Rip' showed a more stable adult population structure with a higher percentage of *T. maxima* surviving into the 150-200 mm size classes (Figure 5.3)
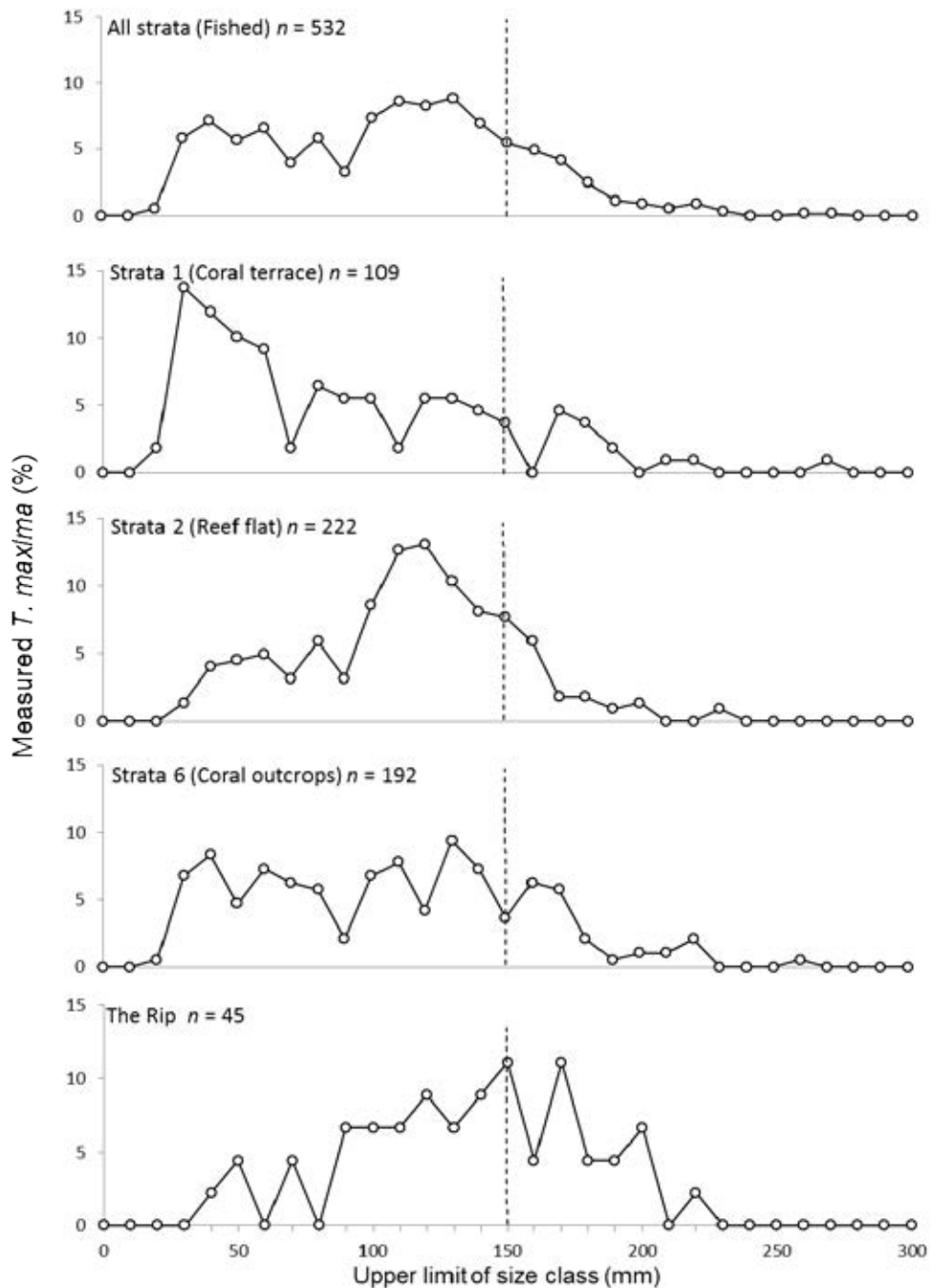
Figure 5.3 Size frequencies of *T. maxima* (10 mm size classes) measured in 2014. Dashed line indicates estimated size at full sexual maturity.

5.4.5 Estimated annual harvest of *T. maxima*

The estimated annual harvest of *T. maxima* was calculated based on a percentage of the Cocos-Malay population (108 people) who fish once or twice per week collecting five, 10 or 20 clams per trip. These are estimates only and used in the absence of any other assessment of recreational fishing pressure at CKI. The estimate ranges from 28,080 to 224,640 *T. maxima*

per year. The lower estimate is based on fishers collecting 5 *T. maxima* once a week while the upper estimate is based on fishers collecting 20 *T. maxima* twice a week (Table 5.4).

Table 5.4 Estimated total annual harvests of *T. maxima* at the Cocos (Keeling) Islands based on two factors (trips per week and clams harvested per trip)

No. of people	No. of days fished / year	No. of *T. maxima* harvested	Total annual harvest
108	52 (1/week)	5	28 080
108	52 (1/week)	10	56 160
108	52 (1/week)	20	112 320
108	104 (2/week)	5	56 160
108	104 (2/week)	10	112 320
108	104 (2/week)	20	224 640

5.4.6 Estimated optimal harvest of *T. maxima*

The optimal annual harvest of *T. maxima* (Table 5.5) was calculated using the estimated total *T. maxima* standing stock (Table 5.3) and the proportion which were recorded to be 100% sexually mature in 2014 (15.8%). Using the P_{opt} formula, the annual average optimal harvest was calculated using three different natural mortality estimates including the average (0.263). An increase in the percentage of mature adults (>150 mm) was observed between 2011 and 2014 (2011: 11.8%; 2014: 15.8%), as a result the optimal harvest has slightly increased.

Table 5.5 Estimates of optimal annual harvest of mature *T. maxima* using different natural mortality rates (*M*). Lower estimates and upper estimates are based on 95% confidence intervals.

Natural mortality (*M*)	Optimal harvest	Optimal harvest (Lower)	Optimal harvest (Upper)
0.226[1]	118,837	51,783	312,457
0.300[2]	154,375	62,370	376,342
0.263[*]	136,803	55,271	333,505

[1]Black *et al.* 2011, [2]Green and Craig 1999, [*] average of *T. maxima* estimates

5.5 Discussion

Only one species of giant clam (*Tridacna maxima*) was identified during the 2014 survey, within the fished and no-take areas of the CKI. No *T. derasa* were recorded in 2014, which reinforces serious concerns for this species at CKI (Bellchambers and Evans, 2013) and suggests any continued level of fishing is unsustainable. Similarly no *T. gigas* were recorded in 2014 and anecdotal reports suggest it may be locally extinct. It should be noted that 4% of the giant clams observed were unable to be conclusively identified to species due to their small size and/or obscured features important for identification.

The abundance, distribution and status of giant clams species varies considerably worldwide (Othman et al 2010). At CKI, the mean density of *T. maxima* in 2014 was 0.055 per m^2 ±0.0179 SE, in fished areas. This density has not changed significantly to levels reported for CKI in 2011; 0.056 ±0.012 SE (Bellchambers and Evans 2013). The density estimates for *T.*

maxima at CKI are higher than median worldwide estimates of 0.000818 individuals per m^2 and that for comparable north-eastern Indian ocean locations such as Mermaid (0.0158 individuals per m^2), Cartier (0.00218 individuals per m^2) and Ashmore Reefs (0.00383 individuals per m^2) (Rees *et al.* 2003). However, it is significantly lower than non-fished areas, such as Ningaloo Reef in Western Australia, which has densities of 0.86 ±0.41 SE individual per m^2 (Black *et al.* 2011).

No significant difference in standing stock estimates of *T. maxima* at CKI was observed between 2011 and 2014. Calculating the standing stock by habitat strata allows for greater discrimination of giant clam densities, particularly when complimented by catch and effort data. Interestingly, although not significant, the two lagoon habitat strata, 2 (reef flats) and 6 (coral outcrops) observed a lower median number of individuals in 2014 compared to 2011. This suggests potential impacts from the same event that caused wide-spread hard coral mortality at CKI between 2012 and 2013 (section 3 of this report). However, no significant impact was observed on the *T. maxima* populations, which were likely protected from severe impact by the propensity of the species to favour the outer reefs and outer lagoon habitat of CKI.

The size structure of the *T. maxima* population at CKI did not change significantly between 2011 and 2014. In 2014, the *T. maxima* population of CKI was comprised of 17.7% juveniles (<50 mm) and 66.5% of sub adults (50-150 mm), which indicates regular recruitment and transition to the next life stage occurs. The lowest percentage of the population in 2014 was in the fully mature population, ≥150 mm (15.8%). This remains lower than reported in other parts of the Indian Ocean where fishing pressure is low (Apte *et al.* 2010; Black *et al.* 2011). Ensuring adequate survival of sexually mature individuals is essential for ensuring ongoing recruitment, particularly as giant clams require relatively high densities to ensure successful spawning (Munro 1992, Kinch and Teitelbaum 2010).

The no-take area of 'The Rip' had higher mean density and mature individuals of *T. maxima* than any of the fished strata. Dramatic declines in abundance are not evident at 'the Rip' until the clams reach 200 mm indicating the natural mortality is driving this decline. However, in the fished areas of CKI the decreases in abundance occur at or before 100% maturity (150 mm) with few individuals surviving to 170 mm. This suggests that harvesting of *T. maxima* is likely extensive and collection of individuals is undertaken before they reach sexual maturity. However, the estimate of standing stock and the similarity of the observed densities between 2011 and 2014, indicate that exploitation of the fishery has not changed. Any relationships to changes in abundance of *T. maxima* due to fishing pressure can not be further quantified due to the lack of catch and effort data for CKI.

There is currently no legal minimum size limit for *T. maxima* at the CKI. The minimum legal size published in international literature varies but typically corresponds with the size at sexual maturity (150 mm) (Green and Craig 1999, Gilbert *et al.* 2006, Chambers 2007, Apte *et al.* 2010). The estimated optimal annual harvest increased slightly from what was reported in Bellchambers and Evans (2013). However, the 95% confidence intervals expanded slightly and, as such, using the lower optimal harvest value of the average of natural mortality estimates (55,271 individuals) the optimal harvest of *T. maxima* has decreased slightly from 2011. Based on this estimate of optimal harvest, 108 fishers catching 10 clams once a week

would exceed the sustainable catch annually by 889 *T. maxima*. It must be noted that the estimates provided are for recreational fishing only. At present there is no commercial fishing for giant clams at CKI, with the exception of a small number taken for aquaculture broodstock purposes.

The life history characteristics and accessibility of giant clams make them particularly vulnerable to overfishing. Giant clams are slow growing, have high mortality, and as protandrous hermaphrodites exhibit a selective mode of spawning that is dependent on adult density. As a result, populations can become unsustainable when densities decline below a critical level (Lucas 1988). The geographical isolation of the CKI highlights the possibility that the *T. maxima* population is almost entirely self-recruiting (Hourston 2010) with the potential for genetically divergence from other populations. The recent description of *T. noae* and *T. ningaloo*, which are morphologically very similar to *T. maxima* and with similar widespread distributions (Su *et al.* 2014), also introduces the risk of under and/or overestimation of all these species at CKI. Natural impacts, such as thermal bleaching and reduced oxygenation of the lagoon have also historically impacted the CKI lagoon (Bunce 1988, Hobbs and McDonald 2010, Hobbs and Macrae 2012). The recreational / subsistence fishing pressures on the CKI giant clam population increase the susceptibility to overexploitation as evidenced by the critical decline of *T. derasa* and potential localised extinction of *T. gigas*.

5.6 Recommendations

The Department of Fisheries Western Australia, Research Division, makes the following recommendations with regards to ongoing management of the giant clam stocks of the Cocos (Keeling) Islands:

- Complete protection for giant clam species *T. derasa* and *T. gigas*

- A sustainable harvest level for *T. maxima* is possible within the following parameters:
    - The number of *T. maxima* taken per fisher per day should be limited;
    - Minimum size limits of 150 mm shell length of *T. maxima*

- Continue the current DoF monitoring program of giant clam density on a triennial basis;

- Encourage further research on genetic connectivity and identification of giant clams at CKI; and

- A program to quantify recreational harvest of giant clams be implemented.

6.0 Acknowledgements

7.0 References

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Appendix A

Hard Coral Genera	Site	2010 Mean Cover (%)	2010 St Error	2011 Mean Cover (%)	2011 St Error	2012 Mean Cover (%)	2012 St Error	2013 Mean Cover (%)	2013 St Error	2014 Mean Cover (%)	2014 St Error
Acropora	1	45.5	3.5	46.7	7.0	48.8	8.0	45.3	6.6	44.6	4.7
	2	8.6	4.8	9.5	4.0	14.3	3.0	13.2	4.8	8.8	2.1
	3	3.4	1.7	7.7	3.6	10.9	4.0	14.4	4.4	11.2	3.5
	4	3.8	1.8	3.2	1.5	4.0	2.1	0.1	0.1	0.0	0.0
	5	22.4	2.9	24.2	4.5	21.5	7.7	11.0	3.6	5.2	0.3
	Mean CKI	16.7	4.4	18.2	4.6	19.9	4.6	16.8	4.4	14.0	4.3
Pavona	1	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	2	27.3	2.9	34.4	3.3	33.2	3.7	23.6	2.1	26.6	3.4
	3	5.8	0.9	6.9	2.2	8.1	2.7	5.9	2.2	4.4	1.2
	4	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	6.6	2.9	8.3	3.6	8.2	3.5	5.9	2.5	6.2	2.8
Montipora	1	5.4	3.1	4.4	1.3	5.1	1.9	4.7	0.6	2.2	0.6
	2	2.8	0.5	3.6	0.6	2.9	0.7	2.7	0.5	2.8	0.4
	3	4.9	0.6	1.9	0.2	3.1	0.6	1.8	0.6	1.7	0.8
	4	0.3	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.0	0.0
	5	19.7	3.9	16.3	5.0	17.4	6.7	0.3	0.3	0.1	0.1
	Mean CKI	6.6	2.0	5.3	1.8	5.7	2.0	1.9	0.5	1.3	0.4
Echninopora	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	25.9	8.5	32.1	11.7	34.9	12.2	2.5	0.7	4.1	2.0
	Mean CKI	5.2	3.1	6.4	4.0	7.0	4.3	0.5	0.3	0.8	0.5
Porites	1	0.5	0.3	0.1	0.1	0.5	0.3	0.4	0.3	0.2	0.1
	2	4.7	1.4	1.0	0.4	1.2	0.6	1.6	0.4	0.6	0.3
	3	7.4	2.8	5.0	1.0	5.5	1.0	6.1	1.8	7.2	0.6
	4	8.6	2.7	9.8	1.8	9.5	2.3	8.6	3.4	7.4	1.9
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	4.2	1.2	3.2	1.1	3.3	1.1	3.3	1.1	3.1	1.0
Favia	1	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	12.6	2.5	12.2	1.1	9.5	2.3	15.8	3.7	12.4	2.6
	3	1.5	0.7	0.6	0.1	1.3	0.1	2.6	0.4	5.3	1.0
	4	0.0	0.0	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	2.8	1.4	2.6	1.3	2.2	1.1	3.7	1.8	3.5	1.4
Pocillopora	1	1.9	0.1	3.2	0.5	2.1	0.5	1.7	0.3	1.8	0.4
	2	3.6	0.9	2.6	0.1	3.5	0.3	3.6	0.3	3.7	0.3
	3	5.6	1.4	3.6	0.7	6.4	1.0	6.9	0.6	4.3	0.6
	4	0.2	0.0	0.2	0.1	0.5	0.2	0.1	0.1	0.0	0.0
	5	0.1	0.1	0.1	0.1	0.2	0.2	0.0	0.0	0.0	0.0
	Mean CKI	2.3	0.6	1.9	0.4	2.5	0.6	2.5	0.7	2.0	0.5

Hard Coral Genera	Site	2010		2011		2012		2013		2014	
		Mean Cover (%)	St Error	Mean Cover (%)	St Error	Mean Cover (%)	St Error	Mean Cover (%)	St Error	Mean Cover (%)	St Error
Isopora	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	5.3	1.0	6.5	1.0	6.2	0.4	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	1.1	0.6	1.3	0.7	1.2	0.7	0.0	0.0	0.0	0.0
Fungia	1	0.1	0.1	0.0	0.0	0.1	0.1	0.1	0.1	0.1	0.1
	2	0.0	0.0	0.1	0.1	0.0	0.0	0.2	0.0	0.0	0.0
	3	0.1	0.1	0.1	0.1	0.0	0.0	0.2	0.1	0.1	0.1
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	3.0	0.7	1.2	0.3	2.1	0.6	0.6	0.3	0.1	0.1
	Mean CKI	0.6	0.3	0.3	0.1	0.4	0.2	0.2	0.1	0.1	0.0
Astreopora	1	0.0	0.0	0.0	0.0	0.4	0.3	0.4	0.3	0.1	0.1
	2	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.1	0.1
	3	0.8	0.5	1.2	0.3	1.3	0.5	0.7	0.5	0.6	0.2
	4	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.2	0.1	0.2	0.1	0.4	0.2	0.2	0.1	0.2	0.1
Seriatopora	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.8	0.8	0.0	0.0	0.0	0.0	0.1	0.1
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.2	0.2	0.0	0.0	0.0	0.0	0.0	0.0
Stylophora	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.1	0.1	0.4	0.2	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.3	0.3	0.1	0.1	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Coscinaraea	1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.2	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pachyseris	1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.2
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Hard Coral Genera	Site	2010		2011		2012		2013		2014	
		Mean Cover (%)	St Error	Mean Cover (%)	St Error	Mean Cover (%)	St Error	Mean Cover (%)	St Error	Mean Cover (%)	St Error
Leptoseris	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lobophyllia	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.1	0.1	0.1	0.1	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Favites	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Herpolith	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pectinia	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Platygyra	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trachyphyllia	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Turbinaria	1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mean CKI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0