11-2019

Invertebrate and Reef Health Research and Monitoring at Cocos (Keeling) Islands – 2019 Update

N Konzewitsch

S. N. Evans

Recommended Citation

Konzewitsch, N. and Evans, S.N. 2019. Invertebrate and Reef Health Research and Monitoring at Cocos (Keeling) Islands – 2019 Update. Fisheries Research Report No. 303 Department of Primary Industries and Regional Development, Western Australia. 41pp.

Department of
Primary Industries and Regional Development

GOVERNMENT OF WESTERN AUSTRALIA

Fisheries Research Report No. 303

Invertebrate and Reef Health Research and Monitoring at Cocos (Keeling) Islands – 2019 Update

Konzewitsch, N. and Evans, S.N.

November 2019

Correct citation:

Konzewitsch, N. and Evans, S.N. 2019. Invertebrate and Reef Health Research and Monitoring at Cocos (Keeling) Islands – 2019 Update. Fisheries Research Report No. 303 Department of Primary Industries and Regional Development, Western Australia. 41pp.

Enquiries:

WA Fisheries and Marine Research Laboratories, PO Box 20, North Beach, WA 6920
Tel: +61 8 9203 0111
Email: library@fish.wa.gov.au
Website: www.fish.wa.gov.au

A complete list of Fisheries Research Reports is available online at **www.fish.wa.gov.au**

Department of Primary Industries and Regional Development
Gordon Stephenson House
140 William Street
PERTH WA 6000
Telephone: (08) 6551 4444
Website: dpird.wa.gov.au
ABN: 18 951 343 745

ISSN: 1035-4549 (Print) ISBN: 978-1-921258-36-7 (Print)
ISSN: 2202-5758 (Online) ISBN: 978-1-921258-37-4 (Online)

Contents

1.0 Format of Report

The purpose of this report is to provide an update of the marine invertebrate and reef health resources at the Cocos (Keeling) Islands as reported in the Fisheries Research Report (FRR) 272 (Evans *et al.* 2016). As an update of observed results and trends, this report should be considered an addendum to FRR 272. Unless specifically updated in this report, the general introduction, survey and analysis methodology for each section are detailed in FRR 272.

This report updates the research and monitoring programs reported in Evans *et al.* (2016) with additional data collected between 2015 and 2018. This timeframe aligns with the Service Delivery Arrangement between the Department of Primary Industries and Regional Development (Western Australia) and the Australian Commonwealth Government (represented by the Department of Infrastructure, Transport, Cities and Regional Development). The section numbering in this report follows that of the FRR 272 in the Fisheries Research Report Series which is available online at:

http://www.fish.wa.gov.au/Documents/research_reports/frr272.pdf

2.0 Background

The fish resources of the Cocos (Keeling) Islands (CKI) are managed by the Department of Primary Industries and Regional Development, Western Australia (DPIRD) on behalf of the Australian Commonwealth Government under a Service Delivery Arrangement (SDA) which commenced in 2002. On the 1st July 2017, as a part of widespread Machinery of Government changes in the Western Australian public sector, the former Department of Fisheries was merged with the former Department of Agriculture and Food, Department of Regional Development, and the Regional Development Commissions. The merge and name change to DPIRD has not resulted in significant changes to the SDA for aquatic science, management and compliance at the Indian Ocean Territories. All the relevant acts and agreements remain in place and DPIRD's aquatic science and assessments at CKI, and the wider Indian Ocean Territories, continues to be risk based.

3.0 Status of the Cocos (Keeling) Islands coral reef habitats 2010-2017

3.2 Methods

The five long term reef monitoring sites implemented by DPIRD in 2010, and surveyed annually until 2014, were resurveyed in April 2016 and February 2017. In 2016, a sixth monitoring site (site 6) was established in the marine protected area of the Pulu Keeling National Park (PKNP) (Figure 3.1). PKNP is 15nm north of the southern atoll and the monitoring site was implemented and surveyed under exemption from Parks Australia (Exemption CINP_2015_10). Site 6 is located on the western reef slope of Pulu Keeling in 12-14m water depth, comparable with the reef slope sites at the southern atoll (Figure 3.1). Survey methods at site 6 are the same as the existing sites, which consist of three replicate 50m transects separated by a 25m interval and sampled by diver operated video. Site 6 was surveyed in 2016 and 2017 and is reported as a separate 'region' and 'zone' to the sites at the southern atoll.

Recently published taxonomic research on scleractinian corals has described the reclassification of a number of hard corals that are known to occur at CKI (see Richards & Hobbs 2014, Hoeksema & Cairns 2019). Of particular relevance to CKI is the taxonomic reclassification of species within the genera *Favia* (Budd *et al.* 2012, Huang *et al.* 2014) and *Fungia* (Gittenberger *et al.* 2011, Benzoni *et al.* 2012, Oku *et al.* 2017) into several separate genera. Analysis of the imagery from the DPIRD CKI reef health monitoring program is identified to genera level only. Without re-analysis of the imagery from all previous surveys, it is not possible to separate the recently accepted nomenclature for the 2016 and 2017 data. We believe the impact of the reclassifications will have little consequence on the results reported in this addendum. Therefore, in this addendum, the naming convention for hard coral genera will remain consistent with that of the original report. An update of the scleractinian nomenclature will be assessed in future reports.

Unless otherwise stated, all error margins in the results are 1 standard error (SE).

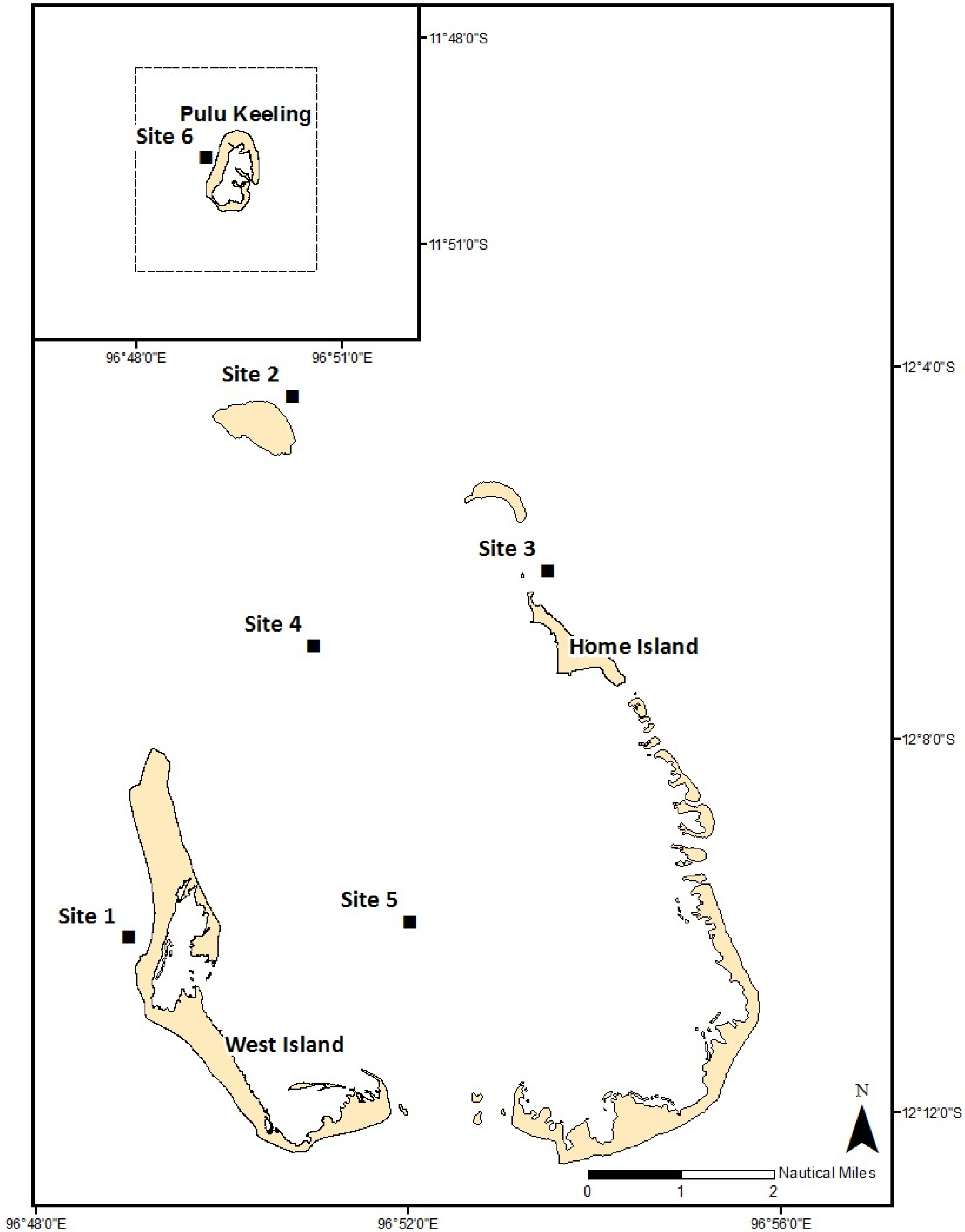

Figure 3.1 DPIRD long term benthic monitoring sites, including an inset of the Pulu Keeling National Park (dashed line).

3.3 Results

3.3.1 Long term trends in overall reef composition and health

The composition of the coral reef communities of CKI consist primarily of three broad classes; hard coral, soft coral and abiotic substrate (Figure 3.2). Macro algae and all other subcategories comprise less than 5% of the benthos at any one site and were not examined further due to their overall low abundance at the CKI reef monitoring sites. At the CKI regional level, soft coral at the southern atoll has shown a slight decrease from an average cover of 13.5% (between 2010 to 2014) to 10.2% in 2016 and 9.9% in 2017. The CKI southern atoll regional hard coral cover for 2016 and 2017 shows limited recovery from the decline observed between 2012 and 2014 (Figure 3.2). A further 4.5% decline was observed between 2014 and 2016 (from 31.6% to 27.1%) before a slight recovery in 2017 to an estimated overall percentage cover of hard coral cover of 29.1% (Figure 3.2).

Within the PKNP region (site 6), soft coral is the dominant biota with 30.0% cover in 2016 and 42.6% in 2017. Hard coral cover at PKNP was 19.3% in 2016 and 15.3% in 2017. Regionally, the overall level of hard coral cover at PKNP is lower than the CKI southern atoll, not accounting for the unbalanced design for comparison (Figure 3.2). In 2016 and 2017 there remained no observations of crown-of-thorns starfish (*Acanthaster planci*) at any of the monitoring sites.

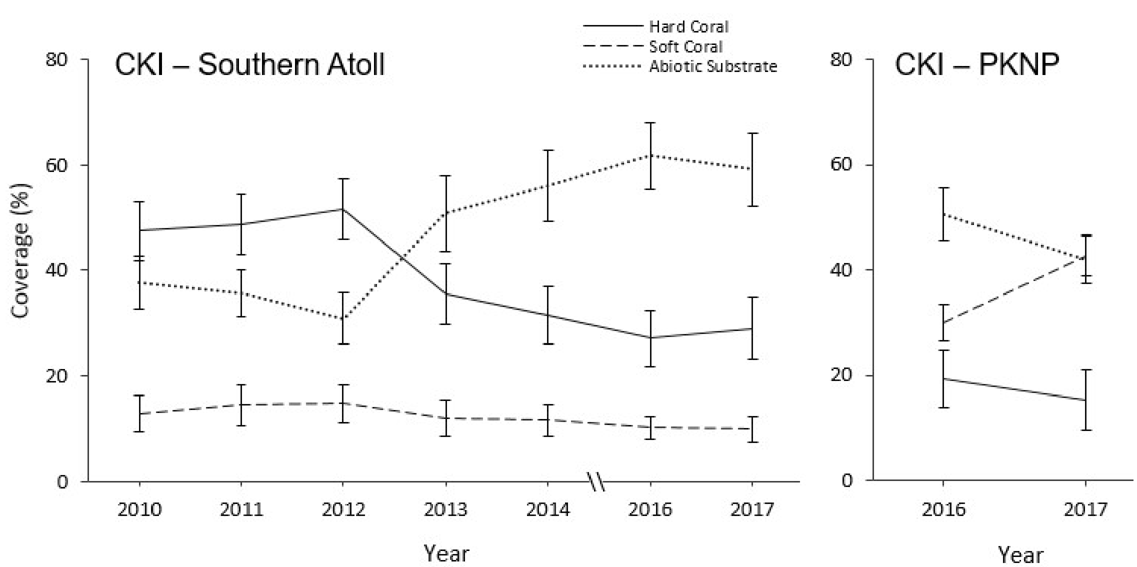

Figure 3.2 Broad habitat composition divided into the southern atoll (5 sites) and the protected PKNP (1 site).

The overall similarity within and between all sites post 2014, as shown in a non-metric multi-dimensional scaling plot (Figure 3.3a & b), no longer remains consistent at 40%, with site 5 shifting to less than 40% similarity in 2016 and 2017. Site 2 was the only site with no observed change throughout the entire study period (2010 to 2017) remaining within 80% similarity over the eight-year period. Site 1 shifted from 80% similarity to 40% in 2016 and 2017 while site 3 shifted from 80% similarity to 60% in the same period. Site 4 shifted from 80% similarity to 60% similarity in 2013 then remained 80% similar between 2013 and 2017. Site 6 (PKNP) showed an overall similarity of 40% to the southern atoll sites. The strength and direction of the changes in the benthic communities between years are shown by the length and direction of the temporal vectors overlaid on the nMDS plot (Figure 3.3b).

a)

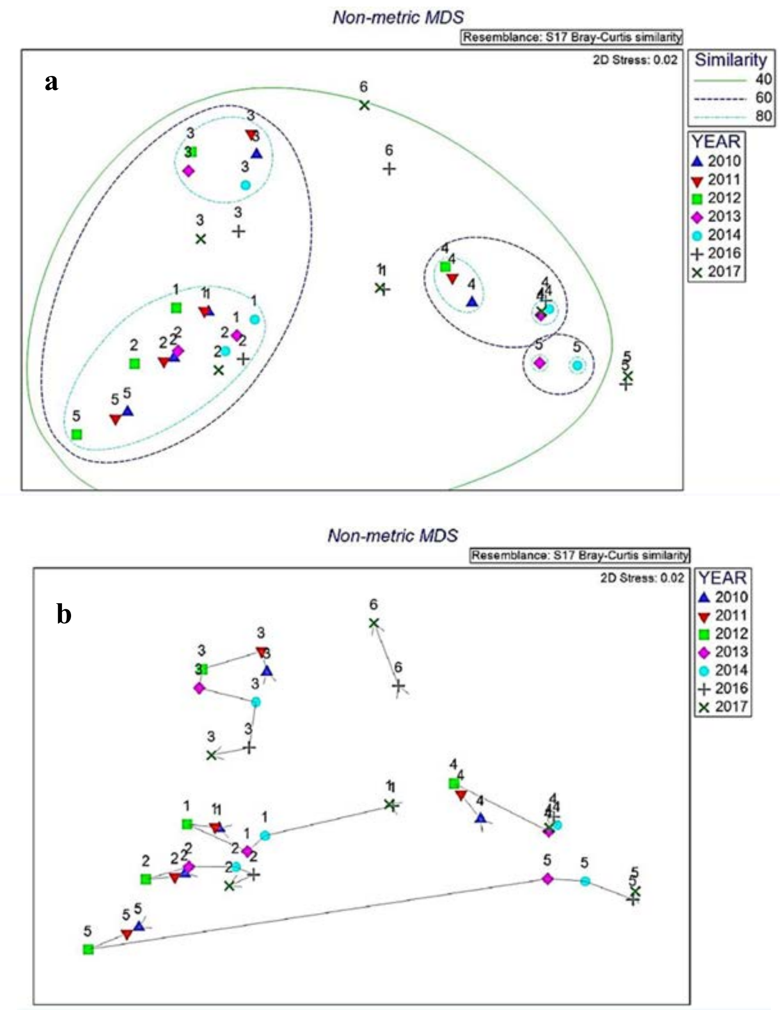

Figure 3.3 Non-metric multi-dimensional scaling plot (nMDS) of similarity of the broad benthic habitats for CKI southern atoll (sites 1 -5) and PKNP (site 6) overlaid with; (**a**) resemblance levels depicting similarities of 40%, 60% and 80%, and, (**b**) time trajectories depicting the change in the broad benthic habitats at each site between years.

The observed changes in the dominant benthos at the CKI southern atoll 2014 varied between sites. The three outer reef sites (sites 1, 2 and 3) showed different trends in hard coral cover with site 1 recording a significant ($p \leq 0.05$ (MC)) 38% decrease between 2014 and 2017 (49.5% in 2014, 30.5% in 2016 and 30.9% in 2017), site 2 remained stable (55.9% in 2014 to 57.1% in 2017) while site 3 showed an increase from 35% in 2014 to 46% in 2017 (Figure 3.4). Hard coral at the lagoon sites (sites 4 and 5) also displayed varied trends with site 4 relatively stable at 7.5% hard coral cover in 2014 and 9% in 2017, while site 5 decreased significantly ($p \leq 0.05$ (MC)) from 9.5% in 2014 to 2.2% in 2017 (Figure 3.4). The 2016 surveys were undertaken between the 23rd and 30th April and visual and analysed observation of the reef showed little signs of bleaching at any site.

Surveys of the PKNP site (site 6) commenced in 2016 and show a lower cover of hard coral compared to the CKI southern atoll with 19% and 15% coral cover observed in 2016 and 2017 respectively (Figure 3.4).

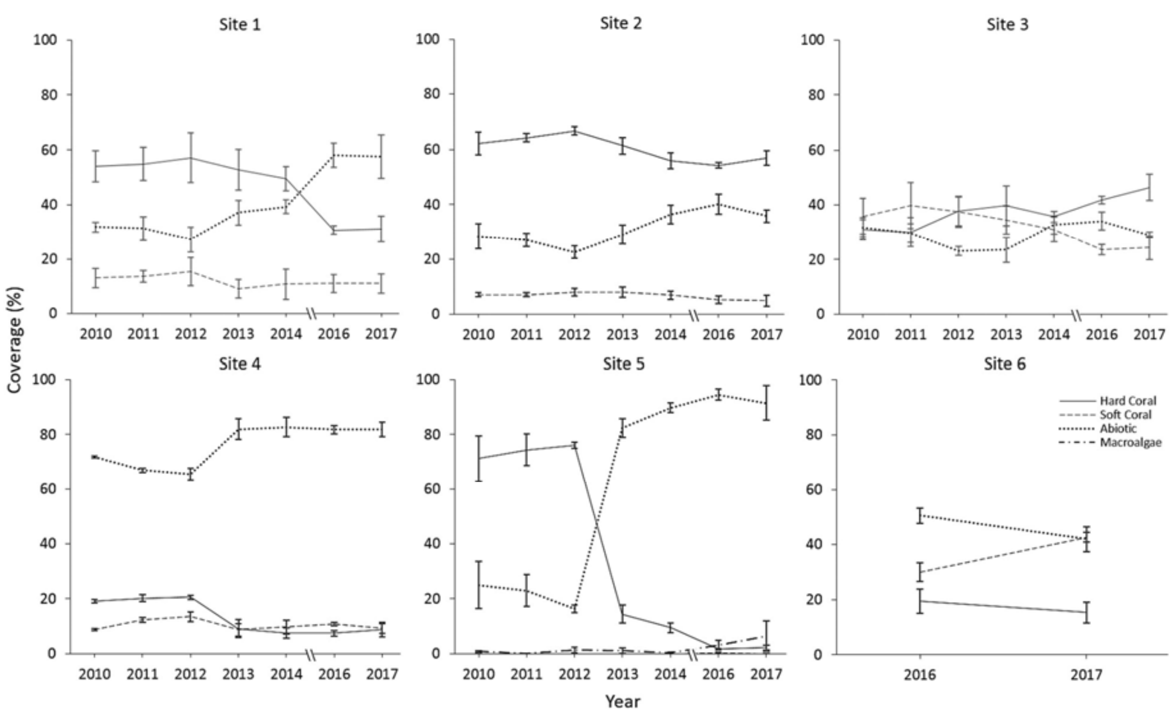

Figure 3.4 Percent cover of dominant broad benthos at the CKI southern atoll (sites 1 – 5) and PKNP (site 6).

3.3.2 Seawater temperature data

The mean daily in-situ seawater temperature at the CKI southern atoll sites between January 2015 and December 2017 ranged between 24.6°C and 31.7°C with both observations recorded at site 5 (Figure 3.5). While this site recorded the highest daily mean temperature, it was on average one of the coolest sites, with a mean temperature of 27.8°C. Site 2 had a slightly cooler mean temperature of 27.7°C while site 3 was the warmest site with a mean of 28.5°C.

The NOAA Coral Reef Watch issued a bleaching warning for CKI from March to April in 2015 and three level 1 bleaching alerts for March and April in 2016 (NOAA Satellite and Information Services, 2019; Figure 3.6). These anomalies were also detected in the in-situ data with sites 3, 4 and 5 recording maximum mean daily temperatures of 30.0°C, 30.5°C and 31.3°C respectively in March 2015 (no data was available for sites 1 and 2 during this period). Similarly, in 2016 all five sites recorded maximum daily mean temperatures in excess of 30°C in February, March and April. The highest daily mean temperature throughout the entire study period (2010 to 2017) was also recorded during this period at site 5 (31.7°C in February 2016). Site 1 recorded the highest outer reef temperature of 30.5°C in March 2016 (Figure 3.5). Limited in-situ temperature data was available for PKNP between 2016 and 2017.

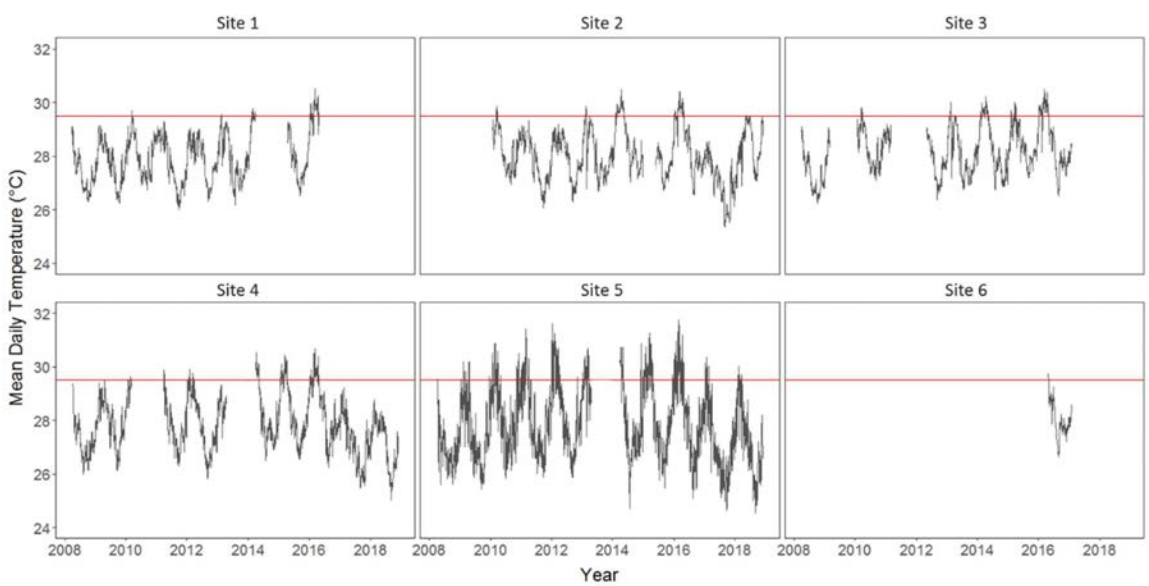

Figure 3.5 Mean daily in-situ seawater temperature data (red line indicates thermal bleaching threshold, 29.5°C).

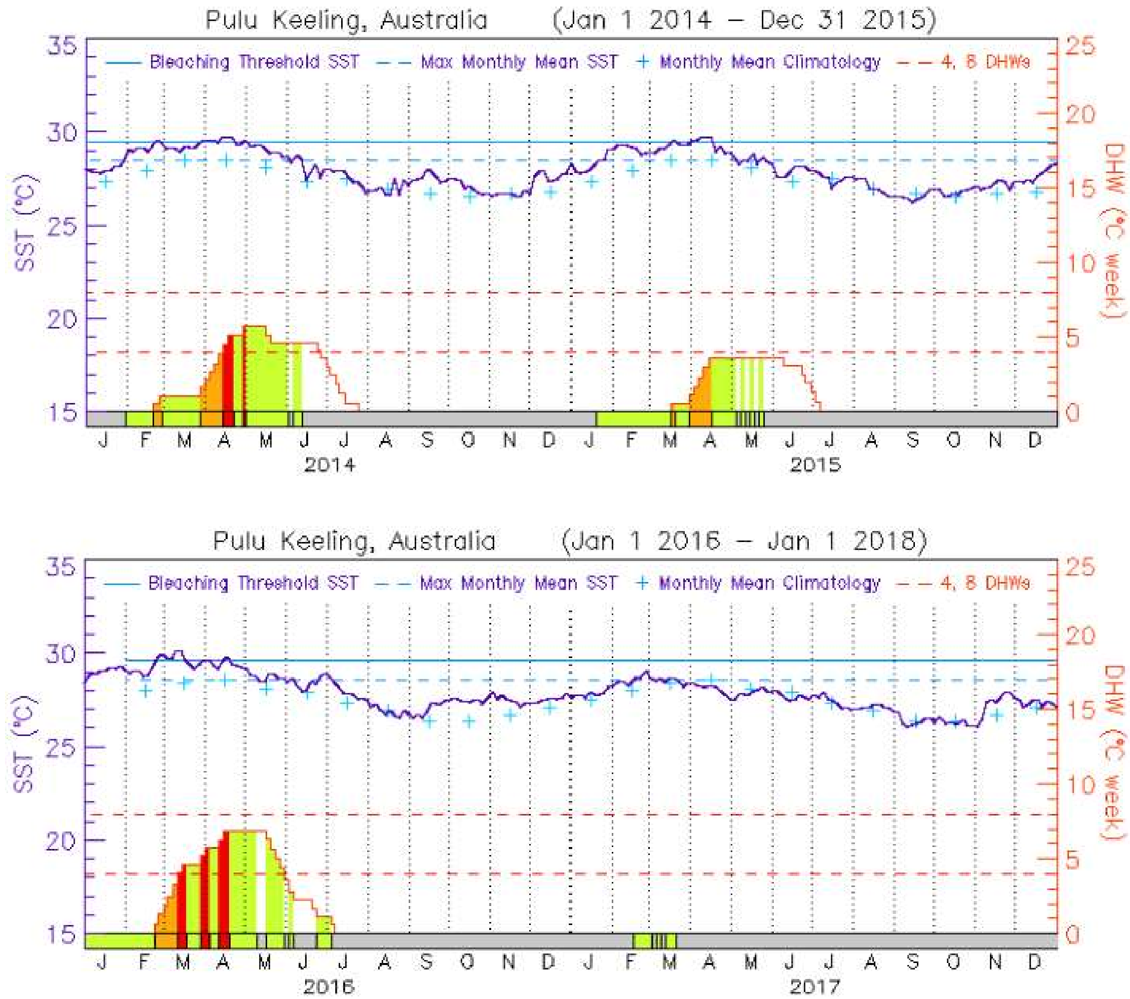

Figure 3.6 NOAA SST January 2014 – December 2017 (www.coralreefwatch.noaa.gov/satellite/vs/australia.php#PuluKeeling_Australia)

Prior to the 2014/2015 season, the highest number of cumulative days of observed seawater temperature above the CKI bleaching threshold on the outer reef sites (sites 1, 2, and 3) was recorded at sites 2 and 3 during the 2013/2014 season (80 and 79 days respectively) (Figure 3.7). In 2014/15, the outer reef showed 24 days above the CKI bleaching threshold (data available for site 3 only). In 2015/16, sites 1, 2 and 3 recorded 76, 81 and 105 days above the CKI bleaching threshold respectively. The observation of 105 days of daily mean temperature above the threshold is the highest at any site throughout the entire study period (2010 – 2017). There were no observed days above the bleaching threshold in 2016/17, although data is limited (Figure 3.7).

The degree heating day curves show that site 5, in the inner lagoon, is the only site exposed to daily mean temperatures above 31°C (Figure 3.7). In 2014/15, sites 4 and 5 recorded 70 and 78 days above the bleaching threshold respectively while site 5 recorded three days above 31°C. sites 4 and 5 also recorded a high number of cumulative heating days in 2015/16 with 87 and 81 days above the CKI bleaching threshold and 20 and 51 days above 30°C respectively. In 2015/16, site 5 also recorded 28 days above 30.5°C, 15 days above 31°C and three days above 31.5°C. Only site 5 recorded cumulative exposure time above the CKI bleaching threshold in 2016/17 (13 days) (Figure 3.7).

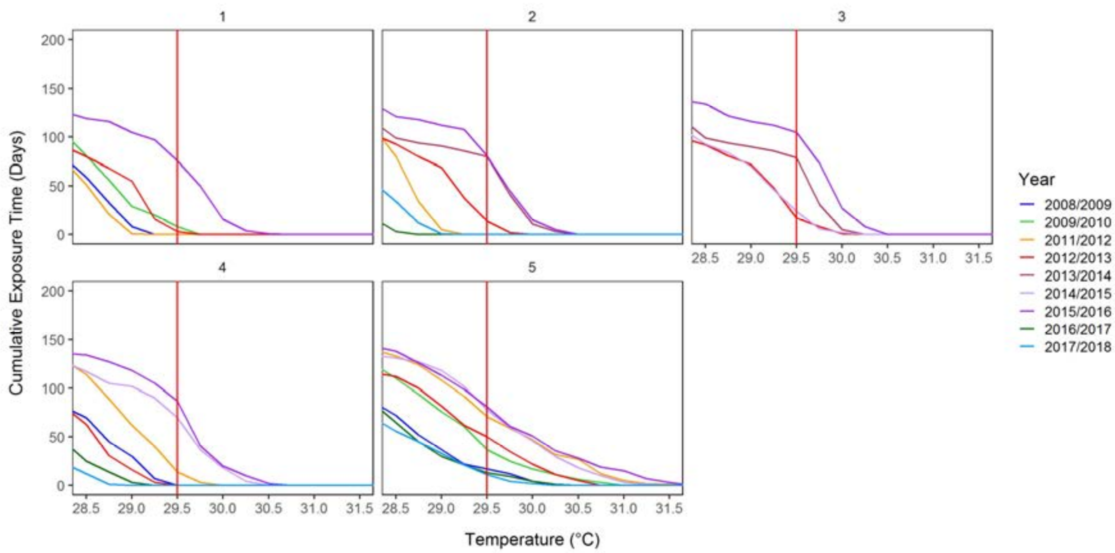

Figure 3.7 Cumulative exposure time of seawater temperature at each southern atoll site (red vertical line indicates CKI thermal bleaching threshold of 29.5°C). Note: incomplete seasonal datasets were omitted; therefore, data is not available for all sites for all years.

3.3.3 Meteorological data

In late December 2014 Tropical Cyclone (TC) Kate passed to the north of CKI as a Category 1 system. The system then intensified to a Category 4 severe TC as it moved to the west, away from CKI. TC Kate produced wind gusts of 55 knots (102 km/h) and 107.6 mm of rain on CKI resulting in flooding on West and Home Islands (BoM, 2019). Gilmour *et al.* (2019) also reported significant wave energy events (wave height greater than 4m) at CKI with >30 days recorded in 2014 and approximately 5 to 10 in 2015.

3.3.4 Spatial and temporal hard coral composition

Sixteen hard coral genera have been identified during the monitoring program (Appendix A) with twelve genera identified across both regions (CKI southern atoll and PKNP) during the 2017 survey (Table 3.4). The four genera that were not recorded in 2017 were *Favites, Herpolitha, Lobophyllia* and *Pachyseris* (Appendix A). Only five genera occurred at ≥5% cover at any one site in the CKI southern atoll and only one occurred at ≥5% cover at the PKNP site (Table 3.4). In 2017, *Acropora* remained the most abundant genera overall, however, it was no longer the most widely distributed. Instead, *Porites* had the widest distribution being detected at all six sites, mostly in low abundance. *Montipora* had the highest percent cover at the PKNP site.

Table 3.4 Hard coral genera observed in 2017, in decreasing order of abundance.

Genera ≥ 5%		Genera < 5%		
Southern atoll		**Southern atoll**		
1. *Acropora*	4. *Porites*	6. *Montipora*	9. *Fungia*	12. *Leptastrea*
2. *Favia*	5. *Pocillopora*	7. *Echinopora*	10. *Goniastrea*	
3. *Pavona*		8. *Astreopora*	11. *Isopora*	
PKNP		**PKNP**		
1. *Montipora*		2. *Acropora*	5. *Goniastrea*	8. *Echinopora*
		3. *Pocillopora*	6. *Leptastrea*	9. *Astreopora*
		4. *Porites*	7. *Favia*	

Post 2014 surveys show that there is a declining trend in generic richness at the CKI southern atoll, driven by declines in the outer reef region (Figure 3.9). Generic richness of the lagoon region ceased to decline in 2016 and increased in 2017, however the lagoon region remains much less diverse than the outer reef slope. Generic richness of the PKNP region remained stable and relatively high during the study period with a mean number of genera of 6.66 ± 0.33 and 6.33 ± 0.33 in 2016 and 2017 respectively.

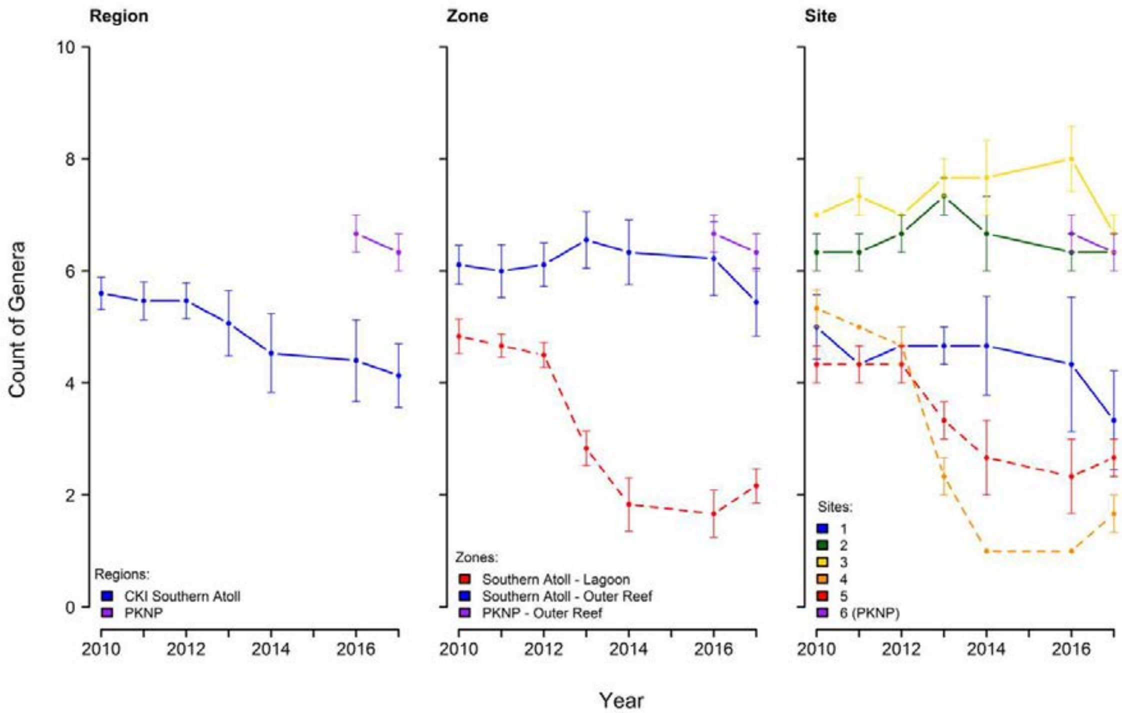

Figure 3.9 Mean generic richness of hard coral by region, zone and site. Dashed line indicates lagoonal sites.

Since 2014, the abundant genera of hard coral at sites 2, 3 and 4 has remained relatively stable (Figure 3.10). The most notable change occurred at site 1 with a significant decrease in *Acropora* (p=0.015(MC)) from 44.57% (± 4.72) to 24.52% (± 2.31) between 2014 and 2016 (Figure 3.10 & Appendix A). The dominant morphology of *Acropora* at site 1 in 2014 was tabulate (97.8%). A significant loss of *Acropora* (p=0.02(MC)) was also observed at site 5 between 2014 and 2016 with a decrease from 5.17% (± 0.32) to 0.89% (± 0.01) (Figure 3.10 & Appendix A). No significant changes in hard coral genera were detected at PKNP (site 6) between 2016 and 2017 (Figure 3.10).

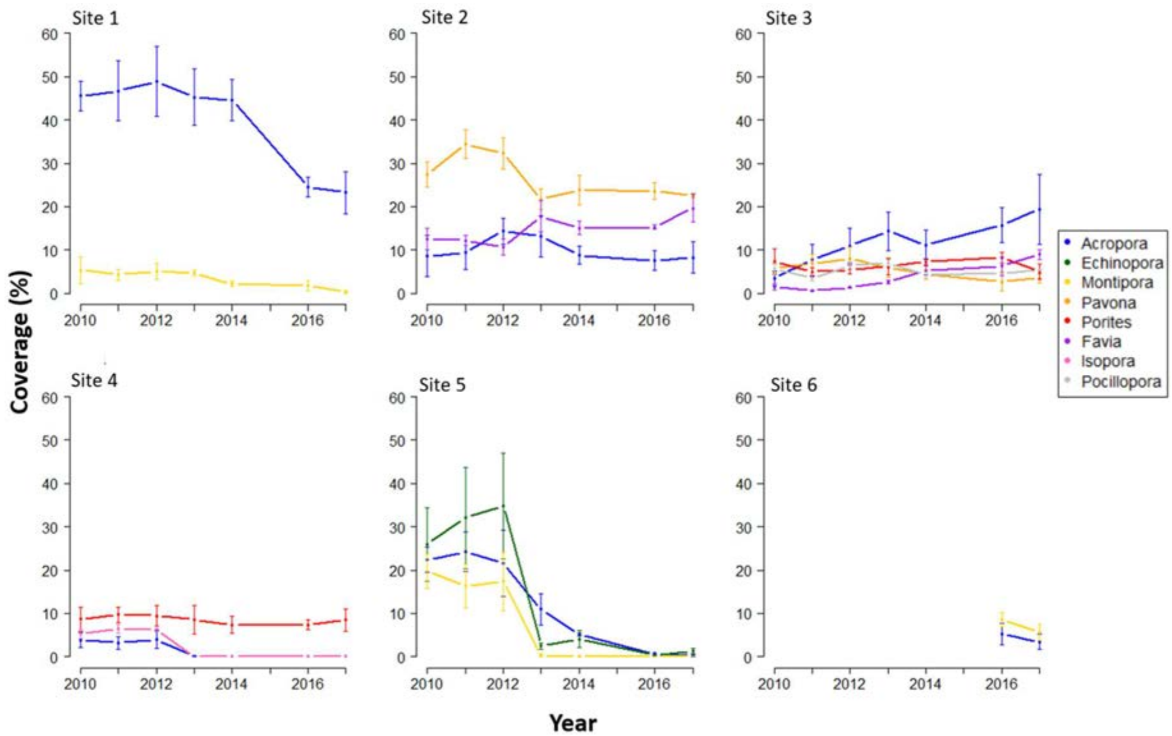

Figure 3.10 Percent cover of dominant hard coral genera (> 5% cover) by site

3.4 Discussion

The 2012/2013 lagoon mortality event resulted in a significant reduction (-43.5%) in hard coral cover and a corresponding overall increase (+52.3%) in abiotic substrate (Evans *et al.* 2016). However, since 2014, there has not been a significant change to the overall hard coral cover at the CKI southern atoll. Although encouraging given the widespread regional bleaching in the Indo-Pacific region in 2016 (Gilmour *et al.* 2019), it also demonstrates that recovery from the 2012/13 lagoon mortality event is not yet evident. However, given the level of impact in 2012/13, significant recovery would not yet be expected (Butler *et al*. 2013). While hard coral cover at the CKI southern atoll has yet to recover, neither macroalgae or soft coral has established in its place with both groups stable throughout the study period. In 2017, four previously observed hard coral genera were not detected at CKI, however these have historically occurred in very low abundances at the monitoring sites (Appendix A) and are likely to be below the level for consistent detection using benthic diver operated video. There is an overall declining trend in hard coral generic richness at the CKI southern atoll sites, which is also primarily driven by the large-scale mortality event within the lagoon (see Evans *et al*. 2016).

The monitoring site at PKNP, established in 2016, provides an important link to examine anthropogenic impacts on the CKI reef systems, with the benthic environment of PKNP completely protected and the island uninhabited. This report describes the percent cover of hard corals at PKNP as 17.3% and soft corals at 36.3% (mean 2016 and 2017), with a generic richness comparable to the outer slope sites at the CKI southern atoll. Parks Australia (2005) observed a similar benthic composition in 2002 to 2004 (~15-25% hard and ~40-45% soft corals), which suggests a level of long term stability for the PKNP reefs. A comparison of the three CKI southern atoll outer reef sites to the PKNP site shows PKNP has higher cover of soft coral (mean 2016 to 2017; PKNP=36.3%, CKI Southern atoll=13.3%) and lower cover of hard coral (mean 2016 to 2017; PKNP=17.3%, CKI Southern atoll=43.5%). The assemblage of hard coral genera at the PKNP site is most similar to Site 1, which would be expected given the western slope orientation of both sites. Both sites are dominated by *Acropora* and *Montipora*, however PKNP has a lower percent cover of *Acropora* (2017: site 1 = 23.3% & site 6 (PKNP) = 3.4%). This suggests that the PKNP site may be more prone to swell impact than the southern atoll sites which may restrict the growth of large tabulate *Acropora* which occur at site 1.

Thermal bleaching has been reported to have occurred at CKI in relation to strong El Niño events in 1983 (Veron 1990) and 1998 (Goreau *et al.* 2000). However, the extent of these bleaching events was not quantified and to date there are no historic quantitative assessments of widespread thermal coral bleaching at CKI. Between 2014 and 2016 one of the most severe El Niño periods was recorded causing a pan-tropical marine heatwave referred to as the third global bleaching event since 1980 (Hughes *et al.* 2017, Zhang *et al.* 2017). This event caused extensive coral bleaching throughout the Indian Ocean during the austral summer of 2015/16 including widespread coral mortality at Christmas Island, ~950km east of CKI (Gilmour *et al.* 2019). However, the impact of the 2016 global bleaching event on the Eastern Indian Ocean varied with widespread mortality was recorded at some coral reef systems (Christmas Island, Scott Reef) and little to no bleaching at others (Rowley Shoals, Ningaloo) despite extended periods of heat stress (Gilmour *et al*. 2019).

Although higher seawater temperatures were observed at CKI between 2014 and 2016, this monitoring program showed that the hard corals at CKI did not display widespread mortality in relation to the 2015/16 global bleaching event. The reason for this observed resilience is not known, but it may be due to previous exposure to high or varied temperatures, genomic variation within species or a high proportion of less susceptible species (Gilmour *et al.* 2019). With the identification of coral communities that can withstand increased thermal pressure described as a global research priority (Hughes *et al.* 2017), further research on the thermal tolerance of CKI hard coral may be warranted. Some localised hard coral mortality was observed post 2014 at one southern atoll outer reef site (site 1). The loss was primarily driven by a significant decrease ($p<0.05$) in *Acropora* between 2014 and 2016. Despite experiencing similar water temperatures, *Acropora* did not decline at the other two southern atoll outer reef sites (sites 2 and 3). Given that pre-2016 over 95% of *Acropora* at site 1 was of tabulate morphology, the hard coral cover loss at this site was most likely caused by more localised hydrodynamic forces, such a large swell event which hit CKI in late 2014 likely as a result of TC Kate (Gilmour *et al*. 2019).

The results from this monitoring program indicate that the reefs of CKI were not significantly impacted by the third global bleaching event between 2014 and 2016 and localised hard coral loss was most likely driven by hydrodynamic forces. The results also show that there has been no significant recovery in the cover or generic richness of hard corals after the impact from the large scale mortality event that occurred in the CKI lagoon between 2012 and 2013 (Evans *et al.* 2016). The rate of recovery for hard coral at CKI is currently unknown (Evans *et al.* 2016), however, this report indicates that recovery from these localised mortality events within the lagoon may be decadal. This timeframe is in line with other reported localised mortality events in systems such as the Great Barrier Reef (Butler *et al.* 2013). Long recovery times coupled with additional localised disturbances and a likely increase in the frequency of thermal bleaching events (Smale *et al*. 2019) all work together to increase the risk to remote systems such as CKI from disease (Haapkyla *et al.* 2011) and decreased recruitment ability (Graham *et al.* 2011). These factors can have an overall negative effect on important benthic habitats and their ability to provide the required food source, refuge and structure to maintain sustainable fish resources. This is of particular significance to remote ecosystems such as CKI where some of the marine resources may have limited external recruitment and are heavily relied on by the local community for subsistence.

The data collected from long term cost effective benthic monitoring programs enable scientists and managers to detect shifts in the marine environment and provide insight into the potential causes of change. In turn, this allows for more informed assessment of aquatic resources with consideration to any observed habitat changes. Access to this type of quantitative data allows management agencies to make informed decisions on the appropriate management measures that will ensure the sustainability of fish and aquatic resources in relation to the health of the broader ecosystem.

3.5 Recommendations

The Department of Primary Industries and Regional Development, Western Australia, Aquatic Science and Assessment Branch makes the following recommendations with regards to the coral reefs of the Cocos (Keeling) Islands:

- Continue ongoing monitoring on a regular basis to measure coral reef health and fish habitat structure;
- Encourage scientific research to increase knowledge on the diversity, resilience, abundance and genetics of corals;
- Support in-situ environmental loggers (e.g. temperature, 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:
    - No harvest of hard coral in the lagoon, to allow for recovery of existing stock;
    - 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);
    - Before coral collection from the outer reef is undertaken, 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

4.2 Methods

The second iteration of the triennial *Lambis lambis* monitoring program was conducted at CKI in April 2017. All 40 long term monitoring sites were resurveyed in 2017. The survey techniques and analysis methodology were the same as that described in Evans *et al.* (2016). Unless stated otherwise all error margins are 1 standard error (SE).

4.3 Results

4.3.1 Relative abundance estimates

A total of 3279 *Lambis lambis* were observed in 2017 resulting in an estimated mean density of 2049.4 ± 522.0 SE individuals per hectare (ind/ha). This mean density is the highest observed during the DPIRD monitoring program and significantly (p<0.05) higher than four of the five previous surveys (Figure 4.1). The increase in density in 2017 signals a level of recovery in the relative abundance of *L. lambis* at CKI following the lowest observed levels in 2014 (415.6 ± 111.7 SE ind/ha). Although this is an encouraging result, the mean density in 2017 was not significantly higher than that recorded by DPIRD in 2009 (Figure 4.1).

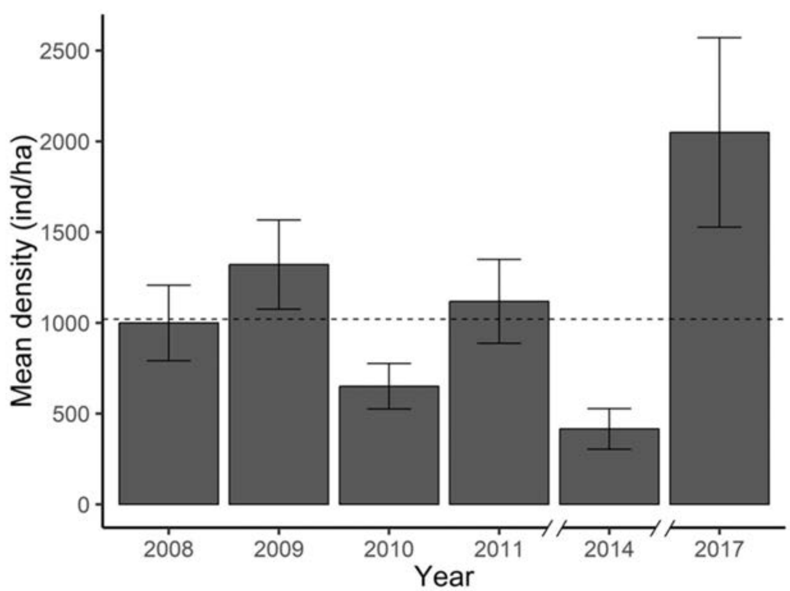

Figure 4.1 Mean densities of *L. lambis* (ind/ha) from the 40 sampling sites (2008 to 2011, 2014, 2017). Dashed line indicates the mean density for the baseline period (2008-11).

4.3.2 *L. lambis* density by general area

Lambis lambis were present at 36 of the 40 sites surveyed in 2017 which is an increase from the 2014 survey where they were present at 32 sites (Figures 4.2 & 4.3). In 2017, there were 18 high density sites (>1000 ind/ha) which is a marked increase from the four reported in 2014 but comparable to 2009 (19 sites). The four sites that did not record any *L. lambis* in 2017 were all in the northern extent of the survey area, within close proximity (<500 m) to Home Island.

Changes in *L. lambis* density between 2017 and previous years were not uniform. A direct comparison between 2017 and 2014 shows that six of the eleven regions had a significant ($p<0.001$) increase in density (regions B, C, D, E, F and G) (Figure 4.2 and 4.3). However, when comparing regional densities in 2017 to all other previous years (2008 to 2011), only two regions (F and G) showed a significant ($p<0.05$) increase. An increase in density was observed in regions H and I, however when compared to previous years this was not statistically significant. This result is driven by the non-uniform distribution of density across the sites within regions e.g. density increases were driven by a large increase at a single site (region H: site H18, region I: site GG34) while other sites within the region experienced a decrease in density (region H: H12, region I: GG43; Figure 4.3). When combined, the two high-density sites (H18 and GG34) within regions H and I contributed 32.9% (1079 individuals) of the abundance of *L. lambis* throughout the entire survey area in 2017. Furthermore, the density at site GG34 (region I) was 18,100 ± 2644 SE ind/ha, which is the highest density recorded during the DPIRD monitoring program (Figures 4.2 & 4.3).

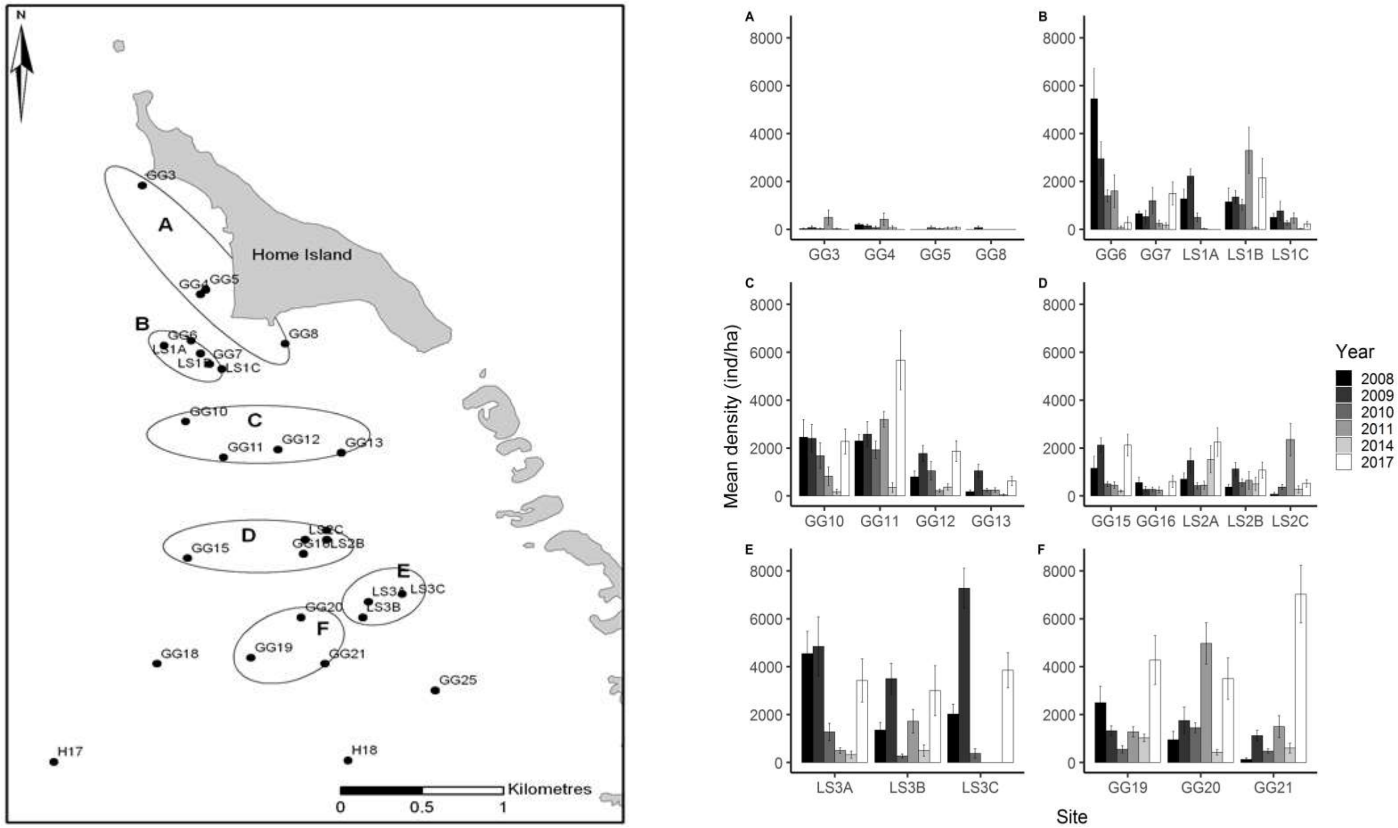

Figure 4.2 Mean density (ind/ha) of *L. lambis* at monitoring sites in Regions A-F.

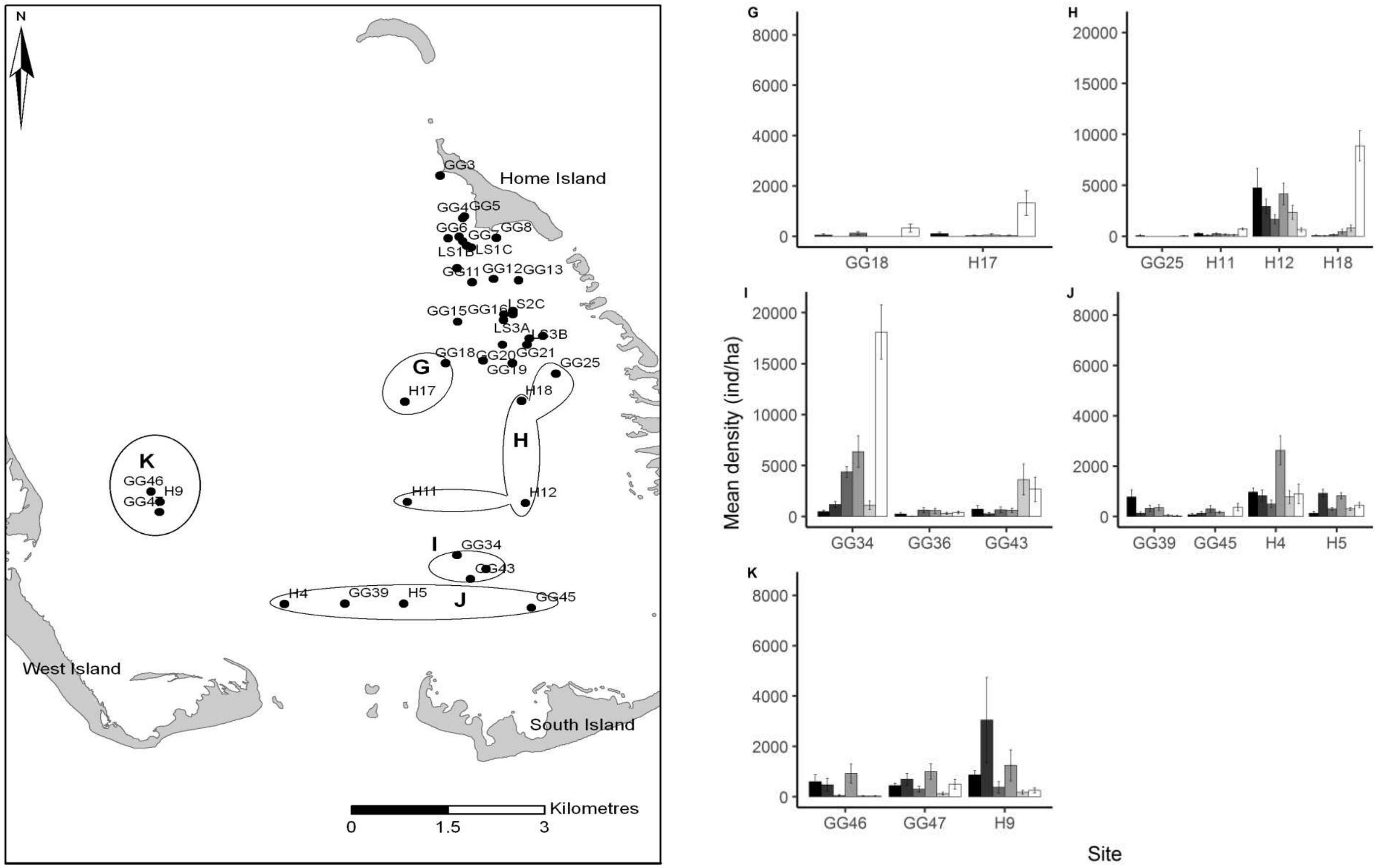

Figure 4.3 Mean density (ind/ha) of *L. lambis* at monitoring sites in Regions G-K (Note differing y-axis values for regions H & I)

4.3.3 Historical abundance comparison of *L. lambis*

In 2017, the mean density of *L. lambis* from the nine sites (three replicates at three 'locations') originally sampled by Lincoln-Smith *et al.* (1993) was 1888.9 ± 232.1 ind/ha. This represents a significant increase from the mean density observed at these sites in 2014 (358.3 ± 91.4 ind/ha, p<0.0001), however, it does not represent an increase from the DPIRD baseline surveys between 2008 and 2011 and still represents a significant (p<0.0001) decrease of 68.1% from the density observed in 1992 (5925.0 ± 1393.4 ind/ha: Lincoln-Smith *et al.* 1993) (Figure 4.4).

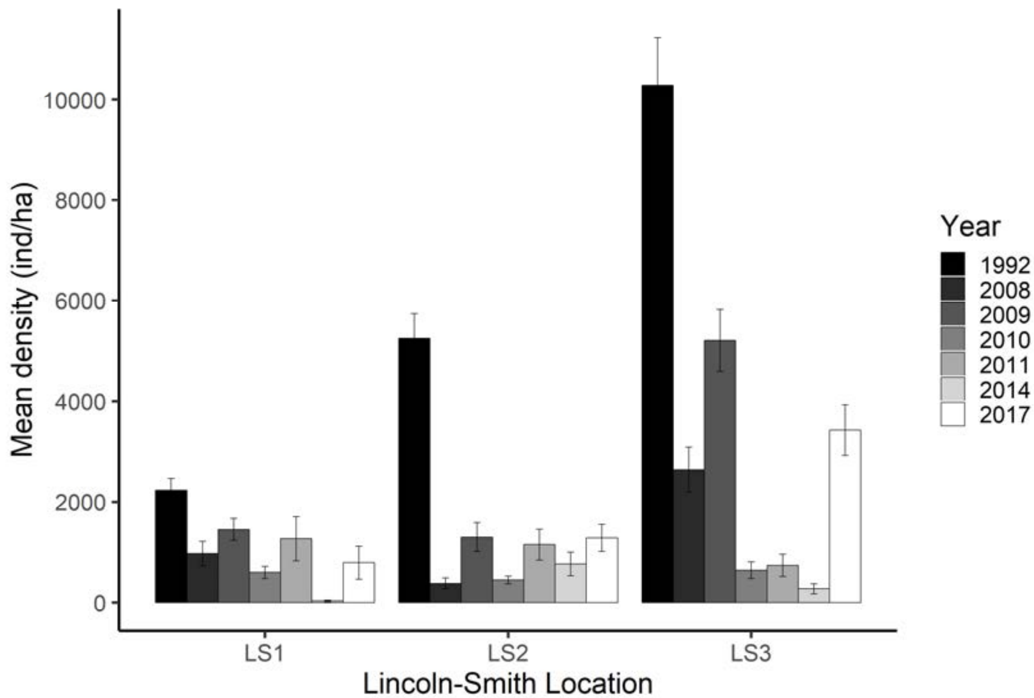

Figure 4.4 Mean density of *L. lambis* (ind/ha ± SE) at three locations (each containing three sites) sampled in 1992 (Lincoln-Smith *et al.* 1993) and DPIRD surveys (2008 - 2017).

4.3.4 *L. lambis* habitat

A significant change in overall benthic habitats was observed in 2017 when compared to both the 2008 and 2014 surveys (p=0.02 and p=0.04 respectively). Comparison within each habitat category found that these changes where driven by shifts in four of the nine broad classes from 2008 and one class from 2014 (Table 4.1). A significant increase in the abiotic and relic reef categories was recorded between 2008 and 2017 (p<0.05) but not between 2014 and 2017 (Table 4.1). A significant decline in massive and submassive corals was observed between 2008 and 2014 but not between 2014 and 2017 (Table 4.1). Filamentous algae recorded a significant increase to 2% from the decline observed between 2008 (4.7%) and 2014 (0.6%) (Table 4.1).

Table 4.1 Mean percent cover of broad habitats at *L. lambis* study sites. Asterisks indicate level of significance of previous survey year (2008 or 2014) when compared to the 2017 values (* $p \leq 0.05$, ** $p \leq 0.01$, *** $p \leq 0.001$).

Survey Year	Abiotic	Relic reef	Massive coral	Submassive coral	Branching coral	Seagrass	Macroalgae	Hard Macroalgae	Filamentous algae
2008	**44.7***	**3.1****	**3.5***	**7.3****	3.7	2.4	18.3	12.3	4.7
2014	45.8	6.0	1.7	4.0	3.4	1.3	25.5	11.5	**0.6*****
2017	53.3	5.8	1.2	3.1	2.2	1.1	22.1	9.0	2.0

4.4 Discussion

In 2017, *Lambis lambis* stocks at CKI showed a significant recovery from the historically low levels reported in 2014 (Evans *et al.* 2016). Although encouraging, the overall relative abundance in 2017 was not significantly higher than that reported in the 2009 DPIRD baseline survey and therefore only represents a recovery of the stock to levels that have been previously reported as relatively low (Bellchambers and Evans 2013). A direct comparison to the sites surveyed by Lincoln-Smith *et al.* (1993) highlights this by showing a 68% decline in *L. lambis* stocks at CKI in 2017 from the levels observed in 1992.

The non-uniform increase in density between 2014 and 2017 of *L. lambis* throughout survey area further supports the aggregating and patchy distribution of this species and the theory of rotational harvesting practices occurring within the *L. lambis* fishery at CKI (Bellchambers and Evans 2013, Evans *et al*. 2016). It is also probable that post- 2014 fishers have targeted more accessible and historically more abundant fishing grounds in the north of the lagoon after a period of low abundance. This may have resulted in reduced fishing activity throughout the south-eastern section of the lagoon (e.g. regions F, G, H and I) which would have contributed to these regions displaying a higher level of recovery than other areas (e.g. regions A, B, C, D and E).

It is unlikely that the practice of rotational fishing alone would have caused the observed increase in relative abundance between 2014 and 2017. There are several other possible factors that may have resulted in an increase in the relative abundance of *L. lambis.* Firstly, there may have been an overall reduction in fishing pressure. Although there is no recreational catch and effort data available for CKI, anecdotal information suggests that fishing effort was reduced in response to a science and education program conducted by DPIRD to increase community stewardship of the resource (Home Island Seniors Group, pers. comm., 2018).

As there were no significant changes observed in the preferred habitat of *L. lambis* (submassive coral, macroalgae and hard macroalgae) (Bellchambers *et al.* 2011, Bellchambers and Evans 2013) between 2014 and 2017, it is unlikely that the increase in relative abundance was related to a shift in habitat composition. However, it is plausible that other environmental factors observed during the monitoring period may have contributed. For example, an anomalous rainfall event in February 2013 caused a large scale hard coral mortality event in the CKI lagoon (Evans *et al.* 2016). This event is likely to have coincided with the spawning of *L. lambis* in that year (Bellchambers and Evans 2013, Mazo *et al.* 2013) which may have had cascading benefits for larval *L. lambis* through factors such as the temporary movement of predators out of the CKI lagoon. Ultimately, in the absence of fishing regulations, it is most likely that the observed increase in *L. lambis* stocks was due to a combination of factors such as an increase in community stewardship and environmental conditions favourable to *L. lambis* recruitment.

4.5 Recommendations

The Department of Primary Industries and Regional Development, Western Australia, Aquatic Science and Assessment Branch makes the following recommendations with regards to the *Lambis lambis* stocks of Cocos (Keeling) Islands:

- Manage the fishing pressure on *L. lambis* and promote community stewardship to ensure sustainable fishing practises
- Continue the current DPIRD monitoring program
- Support a program to quantify the catch and effort of *L. lambis* to inform management arrangements
- Encourage scientific research on the effect of environmental conditions on the recruitment success of *L. lambis*

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

5.2 Methods

The third DPIRD giant clam (*Tridacna sp.*) monitoring survey at CKI was conducted between the 24th November and 4th December 2018. Survey and analysis methods were the same as the previous survey in 2014 (see Evans *et al.* 2016). However, due to inclement weather, only 76 of the 78 monitoring sites were able to be surveyed, an increase of six sites on the 2014 survey (n = 70). In this addendum, the 2018 density and size frequency results are based on all sites surveyed in that year (i.e. n = 76). However, for temporal comparisons of giant clam density and size between survey years, the data for all years was subset to the 70 sites surveyed in 2014. An additional kernel density plot of size frequency is included in this addendum for a temporal comparison of *T. maxima* size frequency at the protected area of ‘The Rip’ (n = 1) and the fished sites within strata 2 (coral flats, n = 13). This plot was developed in R (R Core Team, 2018) using the ggplot2 package (Wickham, 2016). Unless stated otherwise all error margins are 1 standard error (SE).

5.3 Results

5.3.1 Giant clam diversity

In 2018, *Tridacna maxima* were the only giant clam species conclusively identified by underwater visual census, therefore this report focuses only on *T. maxima*. The results of 16S rDNA genotyping by the Western Australian Museum of eleven giant clam samples collected from two different sites at CKI in 2014 confirmed the identification of the collected species as *T. maxima* (Lisa Kirkendale, *pers. comm.*, 2016). Although this does not preclude the presence of other giant clam species at CKI that are morphologically similar to *T. maxima* (e.g. *T. noae*), these results support the continued classification of all giant clams in this report as *T. maxima*. Of the 75 fished sites surveyed in 2018, 978 individuals were identified as *T. maxima*. An additional 441 *T. maxima* were recorded at the protected site ‘The Rip’, resulting in a total of 1419 individual *T. maxima* observed over 76 sites.

5.3.2 Average density of *T. maxima*

The mean density of *T. maxima* in the fished area at CKI was 0.065 ± 0.016 SE clams per m^2 (n = 75) which has not changed significantly (p=0.75) between the three survey years (2011, 2014, 2018) when comparing the subset data (n = 69). The observed density within the protected area of ‘The Rip’ was 2.205 clams per m^2 (n = 1) in 2018 which is more than double that recorded in 2014 (0.825 clams per m^2) and 2011 (1.055 clams per m^2). Statistical comparisons for the protected area were not possible due to the small sample size. In 2018, between the six fished habitat strata, one - coral terrace, two - reef flats and six - coral outcrops, contained the highest overall densities ranging from 0.127 ± 0.044 SE (strata 2) to 0.089 ± 0.044 SE (strata 1) clams per m^2 (Figure 5.1, Table 5.1). Densities in the remaining habitat strata were all below 0.002 clams per m^2. Strata 1 contained the site with the highest density in the fished area (site 5: 0.815 clams per m^2).

Based on the 69 sites surveyed in 2011, 2014 and 2018 there was a significant ($p<0.05$) difference in density between survey years in strata 1 with the mean density of this strata increasing to 0.089 (±0.044) in 2018 from 0.055 (±0.028) and 0.070 (±0.049) in 2011 and 2014, respectively. There was no significant difference ($p>0.05$) in density between years within the five other strata.

Table 5.1 Mean density of *T. maxima* by strata for 2014 and 2018 including significance from pairwise comparison (n = 69 per year).

Strata	**Habitat**	**Mean density of *T. maxima* (m^2 ± SE)**			
		2014	**2018**	***p* value**	***n***
Strata 1	Coral terrace	0.070 ± 0.049	0.089 ± 0.044	0.04	14
Strata 2	Reef flat	0.112 ± 0.046	0.127 ± 0.044	0.89	13
Strata 3	Seagrass/sand	0.003 ± 0.002	0.002 ± 0.002	0.90	10
Strata 4	Coral/algal flat	0.003 ± 0.003	0.002 ± 0.002	1	9
Strata 5	Algal covered rubble	0 ± 0	0.001 ± 0.001	1	10
Strata 6	Coral outcrops	0.099 ± 0.058	0.115 ± 0.038	0.31	13
Overall	Fished	0.055 ± 0.014	0.066 ± 0.016	0.13	69
The Rip	Protected	0.825	2.205	NA	1

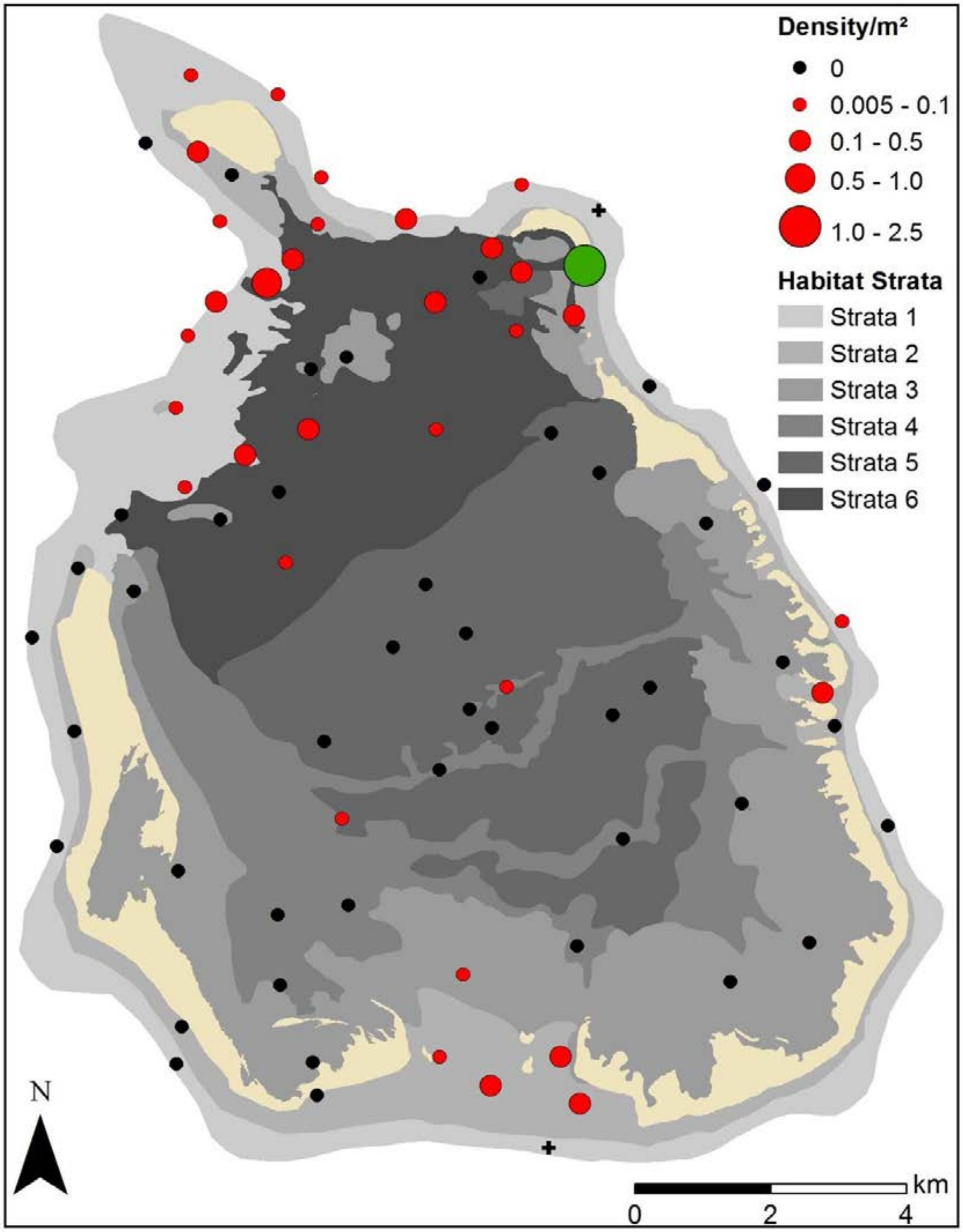

Figure 5.1 Survey sites categorised by the density of *T. maxima* observed in the 2018 survey. Unsurveyed sites in 2018 are represented with an '**+**'. The protected area of 'The Rip' is shaded green.

5.3.3 Standing stock of *T. maxima*

The total standing stock of *T. maxima* in 2018 was calculated at 6,500,885 individuals for the fished area of CKI (~133.96 km^2) (Table 5.2) which is slightly higher than the standing stock estimate for 2014 (5,935,040) but lower than that of 2011 (6,916,269). The increase in overall standing stock from the previous survey appears to have been driven by the abundance in strata 1 (coral terrace) which is estimated to have increased by approximately 500,000 individuals since 2014.

Table 5.2 2018 standing stock of *T. maxima* in fished areas of CKI.

Strata	Median (No. of individuals)	95% Lower Confidence Interval	95% Upper Confidence Interval	Area (km^2)
Strata 1	1,800,625	789,444	3,897,096	21.86
Strata 2	2,089,856	768,812	5,433,498	12.20
Strata 3	75,681	0	227,044	30.27
Strata 4	36,283	0	108,850	18.14
Strata 5	14,876	0	44,629	29.75
Strata 6	2,483,564	1,152,143	4,901,770	21.74
Total	6,500,885	2,710,399	14,612,887	133.96

5.3.4 Size frequency of *T. maxima*

Length measurements were recorded for 699 of the 978 *T. maxima* observed in the fished area of the CKI in 2018. Lengths of *T. maxima* in the fished area ranged from 8 to 226 mm with a median of 94 mm and a mean of 96.9 mm ± 1.77 SE. Overall, there was no significant difference between the size of measured *T. maxima* in the fished area in 2018 to that observed in 2011 or 2014 ($F_{2,1788} = 2.38$, p=0.09). Length measurements from the no-take area ('The Rip') showed the mean length of *T. maxima* was 142.2 mm ± 5.8 SE with a median of 150.0 mm. In 2018, 13.3% of *T. maxima* in the fished area were classed as fully mature (≥150 mm), 68.7% as sub-adult (51 - 149 mm) and 18.0% as juveniles (≤ 50 mm) (Figure 5.2). Overall in 2018, the decline in frequency within fished areas begins to occur at 130 mm with a low proportion of individuals larger than 200 mm (Figure 5.2).

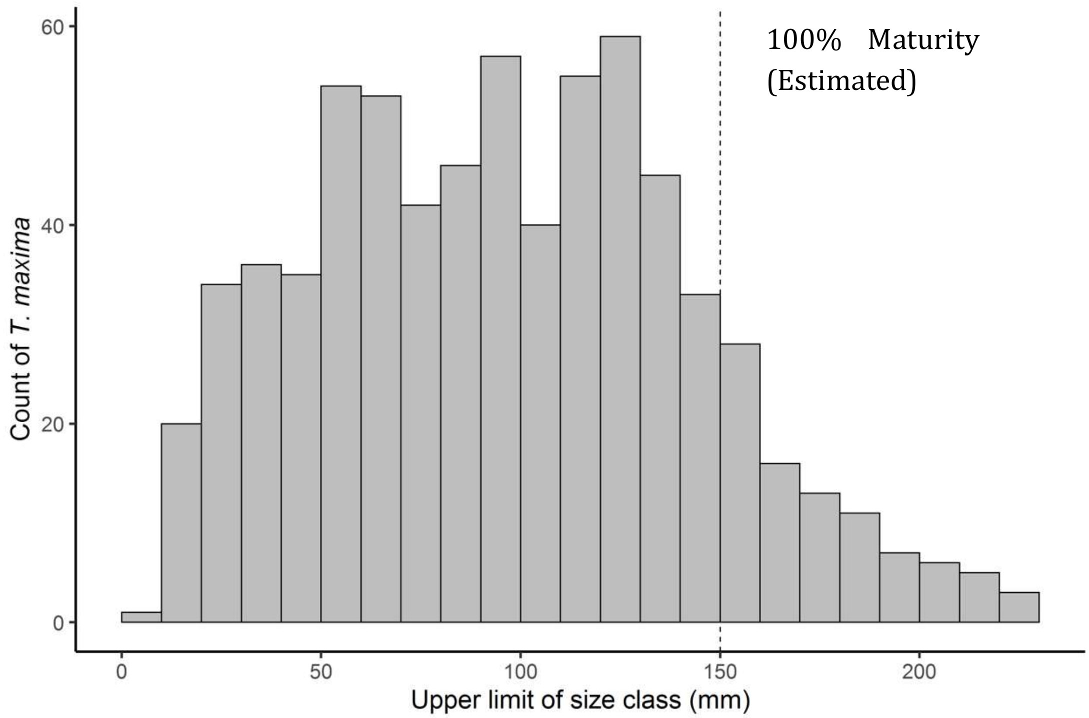

Figure 5.2 Size frequency of *T. maxima* for all sites in the fished area in 2018 (10mm size classes; n = 699). Dashed line indicates the estimated size of full maturity (>150 mm).

No significant difference was found in the size frequency of protected *T. maxima* between 2018, 2011 or 2014 ($F_{2,129}$ = 1.2, p=0.29). The protected area ('The Rip') had a much higher proportion of mature adults (52.0%) and significantly larger *T. maxima* than the fished area ($F_{1,747}$ = 44.2, p<0.0001). ANOVA showed that within the fished area in 2018 there was a significant (p<0.0001) difference in the size of *T. maxima* between the three strata with enough individuals for analysis (strata 1, 2 and 6). Tukey-Kramer post hoc analysis showed that this was driven by strata 1 (coral terrace) having significantly (p<0.0001) smaller *T. maxima* than strata 2 and 6. Post hoc analysis also showed that there was a significant (p=0.02) decrease in the size of *T. maxima* in strata 1 between 2011 and 2018. In 2018, as with previous surveys, the most accessible area (strata 2: reef flat) continues to display a large decrease in the sub-adult population of *T. maxima* with size frequency decreasing sharply after 130 mm (Figure 5.3). By comparison, the protected site of 'The Rip', which is also on a reef flat, only displays a decrease in size frequency after individuals reach full maturity (>150 mm) (Figure 5.3). Temporal comparison of size measurements within 'The Rip' shows bimodal distribution in 2018 with a small peak around 50mm and a second larger peak around 150mm, which differs from the unimodal trend observed in the 2011 and 2014 surveys (Figure 5.3).

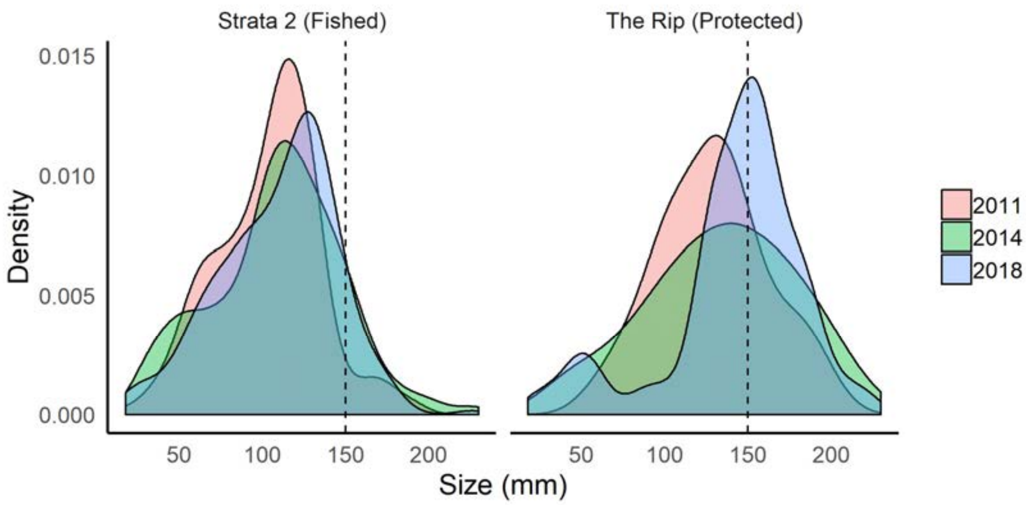

Figure 5.3 Kernel density plots of size measurements from *T. maxima* in fished sites within strata 2 and the protected area of 'The Rip' for each survey year. Dashed line indicates the estimated size of full maturity (> 150 mm).

5.3.5 Estimated annual harvest of *T. maxima*

There was no significant change in the population of the CKI between the 2011 and 2016 censuses (Australian Bureau of Statistics, 2019) thus the harvest estimates used are those described in Evans *et al.* (2016).

5.3.6 Estimated optimal harvest of *T. maxima*

The optimal annual harvest of *T. maxima* was calculated using the estimated total *T. maxima* standing stock (Table 5.2) and the proportion which were recorded to be 100% sexually mature in 2018 (13.3%). Using the P_{opt} formula, the annual average optimal harvest was calculated using the natural mortality estimate for *T. maxima* of 0.300 (Green and Craig, 1999). Evans *et al.* (2016) also used an estimate of natural mortality by Black *et al.* (2011) from a population at Ningaloo Reef, Western Australia which has now been found to be *Tridacna noae* (Johnson *et al.* 2016) and as such has been removed from this addendum. A decrease in the percentage of mature adults (>150 mm) was observed between 2014 and 2018 (2014: 15.8%; 2018: 13.3%) and as a result the optimal harvest has decreased to 142,428 individuals in 2018 from 154,375 individuals in 2014 despite the increase in standing stock. Consequently, the upper (320,154) and lower (59,382) optimal harvest estimates also decreased.

5.4 Discussion

As with the 2014 survey, only one species of giant clam (*T. maxima*) was identified in 2018. The continued absence of *Tridacna derasa* and *Tridacna gigas* supports the concern for the future of these species at CKI (Lincoln-Smith *et al.* 1993, Hender *et al.* 2001, Bellchambers *et al.* 2013, Evans *et al.* 2016). The decline of these giant clam species has been observed globally with greater than 50% of natural populations recently listed as severely depleted, locally extinct or data deficient (Neo *et al.* 2017). The minimum density of giant clams required to maintain a reproductive and genetically robust population is not currently known (Neo *et al.* 2017), however the lack of any observations of *T. gigas* or *T. derasa* within these spatially robust surveys in either 2014 and 2018, suggests serious concern for these two species at CKI.

The overall mean density of *T. maxima* in 2018 increased slightly but has not changed significantly to that observed in 2014 and correspondingly the standing stock estimates increased slightly between surveys. The increase in abundance was driven by a higher mean density in strata 1 (coral terrace) compared to previous surveys. The density within the other two most abundant strata (reef flats and coral outcrops) remained stable between 2014 and 2018 after recording a decrease in 2014 which may have been attributed to the proximity of these strata to a lagoon mortality event in 2012/13 (Evans *et al.* 2016). The size of *T. maxima* measured within strata 1 (coral terrace) in 2018 were significantly smaller than the other two abundant strata (reef flats and coral outcrops) and also compared to previous surveys. This, along with the observed increase in density in this stratum suggests a higher level of recruitment into this habitat type. Giant clams at CKI are traditionally harvested by reef walking and snorkelling (Lincoln-Smith *et al.* 1993) and there is no evidence to suggest that this practice has changed. As such, the depth of the coral terrace may provide refugia for giant clams while the proximity to oceanic water may safeguard against mortality events that have historically affected waters within the lagoon.

As with previous surveys, the size frequency of *T. maxima* in the fished area, and most noticeably in the accessible strata 2 (reef flat), continues to show a sharp decline in individuals after 130 mm, which indicates that *T. maxima* is being collected before estimated full maturity (150 mm). 'The Rip' showed a significantly higher mean density and higher proportion of mature individuals than any of the fished strata in 2018. The decline in size frequency is not evident at this non-fished site until *T. maxima* reach full maturity (150 mm) indicating that natural mortality is most likely the driving factor.

Overall the population of *T. maxima* at CKI is relatively stable with increased recruitment on the reef slopes counter balancing a decline in density observed within some areas of the lagoon and reef flats in 2014 (Evans *et al.* 2016). However, there is some concern for the population of *T. maxima* at CKI; the slow growing and high mortality life history characteristics coupled with the ease of accessibility to fishers make the species particularly vulnerable to overfishing (Alder and Braley 1989, Neo *et al*. 2017). It is probable that the decline of *T. gigas* and *T. derasa* at CKI will result in higher fishing pressure for the remaining species and that the population may experience reduced fecundity due to the practice of harvesting before full maturity. Despite *T. maxima* being the most widespread of the giant clams and one of the tridacnine species of least concern globally, they are also vulnerable to large localised declines (Adessi 2001, Barott *et al.* 2010, Ramah *et al.* 2019) and mass mortality events from natural impacts such as thermal bleaching and reduced oxygenation (Adjeroud *et al.* 2001, Andrefouet *et al.* 2013). With the discovery of *Tridacna noae* at Christmas Island (Neo & Low 2018) there is also the possibility that this recent taxonomically resurrected species may occur at CKI. The shell of *T. noae* is morphologically very similar to *T. maxima* and a difference in mantle pattern is the most reliable method for discerning between the species visually. The taxonomic resurrection of *T. noae* has caused issues for the management of *T. maxima* in some countries as past surveys of *T. maxima* are likely to be overestimates and fishers do not differentiate between the two species (Neo *et al.* 2017). Genetic analysis of individuals at CKI revealed all to be *T. maxima* however the collection of these individuals was limited.

5.5 Recommendations

The Department of Primary Industries and Regional Development, Western Australia, Aquatic Science and Assessment Branch makes the following recommendations with regard to the *Tridacna sp.* stocks of Cocos (Keeling) Islands:

- Complete protection for the giant clam species *Tridacna derasa* and *Tridacna gigas*
- A sustainable harvest level for *T. maxima* is possible within the following parameters:
  - The number of *T. maxima* per fisher per day should be limited;
  - Minimum size limits of 150 mm shell length of *T. maxima*
- Develop educational material to assist recreational fishers with the identification of *Tridacna sp.* and promote community stewardship
- Continue the current DPIRD monitoring program
- Encourage further research on the genetic connectivity and identification of *Tridacna sp.*
- Support a program to quantify the catch and effort of giant clams to inform management arrangements

6.0 Acknowledgments

The authors would like to thank the Cocos (Keeling) Islands community for their assistance, local knowledge and support of DPIRD scientific research. We are grateful to Parks Australia, particularly Ismail MacRae and Trish Flores from Parks Australia CKI, for their logistical support of these research and monitoring programs.

We acknowledge consultant Cathie Page for her expertise in coral identification. We thank our colleagues Joshua Dornan, Jason How, Keyley Hogan-West, Cordelia Moore and Jamin Brown for their invaluable assistance in the field, laboratory or office. We also thank Lynda Bellchambers and Paula Kalinowski for their reviews of this manuscript.

This work is funded by a service delivery arrangement between the Department of Primary Industries and Regional Development, Western Australia and the Australia Commonwealth Department of Infrastructure, Transport, Cities and Regional Development.

7.0 References

Addessi, L., 2001. Giant clam bleaching in the lagoon of Takapoto atoll (French Polynesia). *Coral reefs*, 19(3): 220.

Adjeroud, M., Andréfouët, S. & Payri, C., 2001. Mass mortality of macrobenthic communities in the lagoon of Hikueru atoll (French Polynesia). *Coral Reefs*, 19(3): 287-291.

Alder, J. & Braley, R.D. 1989. Serious mortality in population of giant clams on reefs surrounding Lizard Island, Great Barrier Reef. *Australian Journal of Marine and Freshwater Research*. 40(2): 205-213.

Andréfouët, S., Van Wynsberge, S., Gaertner-Mazouni, N., Menkes, C., Gilbert, A. & Remoissenet, G., 2013. Climate variability and massive mortalities challenge giant clam conservation and management efforts in French Polynesia atolls. *Biological Conservation*, 160: 190-199.

Australian Bureau of Statistics. 2019. 2016 Census. Accessed at https://www.abs.gov.au/websitedbs/censushome.nsf/home/2016 on 2019-06-16

Barott, K.L., Caselle, J.E., Dinsdale, E.A., Friedlander, A.M., Maragos, J.E., Obura, D., Rohwer, F.L., Sandin, S.A., Smith, J.E. & Zgliczynski, B., 2010. The lagoon at Caroline/Millennium Atoll, Republic of Kiribati: natural history of a nearly pristine ecosystem. *PLoS One*, 5(6), p.e10950.

Bellchambers, L.M., Meeuwig, J.J., Evans, S.N. & Legendre, P., 2011. Modelling habitat associations of the common spider conch in the Cocos (Keeling) Islands. *Marine Ecology Progress Series*, 432: 83-90.

Bellchambers, L.M. & Evans, S.N., 2013. A Summary of the Department of Fisheries, Western Australia Invertebrate Research at Cocos (Keeling) Islands, 2006-2011. Fisheries Research Division, Western Australian Fisheries and Marine Research Laboratories.

Benzoni, F., Arrigoni, R., Stefani, F., Reijnen, B., Montano, S. & Hoeksema, B., 2012. Phylogenetic position and taxonomy of *Cycloseris explanulata* and *C. wellsi* (Scleractinia: Fungiidae): lost mushroom corals find their way home. *Contributions to Zoology,* 81(3): 125-146.

BoM (Bureau of Meteorology). 2019. Tropical Cyclone Kate. Accessed at http://www.bom.gov.au/cyclone/history/kate14.shtml on 16-05-2019.

Black, R., Johnson, M.S., Prince, J., Brearley, A. & Bond, T., 2011. Evidence of large, local variations in recruitment and mortality in the small giant clam, Tridacna maxima, at Ningaloo Marine Park, Western Australia. *Marine and Freshwater Research*, 62(11): 1318-1326.

Budd, A.F., Fukami, H., Smith, N.D. & Knowlton, N. 2012. Taxonomic classification of the reef coral family Mussidae (Cnidaria: Anthozoa: Scleractinia), *Zoological Journal of the Linnean Society,* 166(3): 465–529.

Butler, I.R., Sommer, B., Zann, M., Zhao, J.X., & Pandolfi, J.M. 2013. The impacts of flooding on the high-latitude, terrigenoclastic influenced coral reefs of Hervey Bay, Queensland, Australia. *Coral reefs,* 32(4): 1149-1163.

Evans, S.N., Konzewitsch, N. & Bellchambers, L.M., 2016. An Update of the Department of Fisheries, Western Australia Invertebrate and Reef Health Research and Monitoring at Cocos (Keeling) Islands. Fisheries Research Division, Western Australian Fisheries and Marine Research Laboratories.

Gilmour, J.P., Cook, K.L., Ryan, N.M., Puotinen, M.L., Green, R.H., Shedrawi, G., Hobbs, J.P., Thomson, D.P., Babcock, R.C., Buckee, J., Foster, T, *et al.*, 2019. The state of Western Australia's coral reefs. *Coral Reefs* 38(4):651-667.

Gittenberger, A., Reijnen, B.T. & Hoeksema, B.W., 2011. A molecularly based phylogeny reconstruction of mushroom corals (Scleractinia Fungiidae) with taxonomic consequences and evolutionary implications for life history traits. *Contributions to Zoology*, 80(2): 107-132.

Goreau, T., McClanahan, T., Hayes, R. & Strong, A.L., 2000. Conservation of coral reefs after the 1998 global bleaching event. *Conservation Biology*, 14(1): 5-15.

Graham, N.A.J, Nash, K.L., & Kool, J.T. 2011. Coral reef recovery dynamics in a changing world. *Coral Reefs* 30(2): 283-294

Green, A. & Craig, P., 1999. Population size and structure of giant clams at Rose Atoll, an important refuge in the Samoan Archipelago. *Coral Reefs*, 18(3):205-211.

Haapkyla, J., Unsworth, R.K., Flavell, M., Bourne, D.G., Schaffelke, B. & Willis, B.L. 2011. Seasonal rainfall and runoff promote coral disease on an inshore reef. *PloS One* 6(2): e16893

Hender, J., McDonald, C.A. & Gilligan, J.J., 2001. Baseline surveys of the marine environments and stock size estimates of marine resources of the south Cocos (Keeling) atoll (0–15m), eastern Indian Ocean. *Marine Resources of the Cocos Atoll, Indian Ocean* 69 pp.

Hoeksema, B. W. & Cairns, S. 2019. World List of Scleractinia. Accessed at http://www.marinespecies.org/scleractinia on 2019-05-13

Huang, D., Benzoni, F., Fukami, H., Knowlton, N., Smith, N.D. & Budd, A.F., 2014. Taxonomic classification of the reef coral families Merulinidae, Montastraeidae, and Diploastraeidae (Cnidaria: Anthozoa: Scleractinia). *Zoological Journal of the Linnean Society*, 171(2): 277-355.

Hughes, T.P., Barnes, M.L., Bellwood, D.R., Cinner, J.E., Cumming, G.S., Jackson, J.B., Kleypas, J., Van De Leemput, I.A., Lough, J.M., Morrison, T.H. & Palumbi, S.R., 2017. Coral reefs in the Anthropocene. *Nature*, 546(7656): 82.

Johnson, M.S., Prince, J., Brearley, A., Rosser, N.L. & Black, R., 2016. Is *Tridacna maxima* (Bivalvia: Tridacnidae) at Ningaloo Reef, Western Australia?. *Molluscan Research*, 36(4): 264-270.

Lincoln-Smith, M.P., Underwood, A.J., Smith, A.K. & Hawes, P.M.H. 1993. A study of the impact of harvesting marine invertebrate and fish on the marine ecosystems of Cocos (Keeling) Islands, Indian Ocean. The Institute of Marine Ecology, The University of Sydney, Sydney.

Mazo, A.M., Germano, B.P. & Ilano, A.S. 2013. Spawning period and size at sexual maturity of spider conch *Lambis lambis* (L. 1758) (Gastropoda: Strombidae) in selected reef areas of the Visayas, central Phillipines. *Silliman Journal* 54:1

Neo, M.L. & Low, J.K., 2018. First observations of *Tridacna noae* (Röding, 1798) (Bivalvia: Heterodonta: Cardiidae) in Christmas Island (Indian Ocean). *Marine Biodiversity*, 48(4): 2183-2185.

Neo, M.L., Wabnitz, C.C., Braley, R.D., Heslinga, G.A., Fauvelot, C., Van Wynsberge, S., Andrefouet, S., Waters, C., Tan, A.S.H., Gomez, E.D. & Costello, M.J., 2017. Giant clams (Bivalvia: Cardiidae: Tridacninae): a comprehensive update of species and their distribution, current threats and conservation status. *Oceanography and Marine Biology: An Annual Review*, 55: 87-388.

NOAA Satellite and Information Services, 2019. Coral Reef Watch. Accessed at https://coralreefwatch.noaa.gov/satellite/index.php on 2019-06-16

Oku, Y., Naruse, T. & Fukami, H., 2017. Morpho-molecular evidence for polymorphism in the mushroom coral Cycloseris hexagonalis (Scleractinia: Fungiidae), with a new phylogenetic position and the establishment of a new genus for the species. *Zoological science*, 34(3): 242-252.

Parks Australia. 2005. Status of the coral reefs at the Cocos (Keeling) Islands. Commonwealth of Australia. Accessed at https://www.environment.gov.au/system/files/resources/22f846c7-0b13-4dc6-880a-a59f42bbb5f8/files/reef-report.pdf on 2019-06-16

R Core Team. 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ramah, S., Taleb-Hossenkhan, N., Todd, P.A., Neo, M.L. & Bhagooli, R., 2019. Drastic decline in giant clams (Bivalvia: Tridacninae) around Mauritius Island, Western Indian Ocean: implications for conservation and management. *Marine Biodiversity*, 49(2): 815-823.

Richards, Z. T. & Hobbs, J. P. A. 2014. The status of hard coral diversity at Christmas Island and Cocos (Keeling) Islands. *Raffles Bulletin of Zoology*, 30: 376–398.

Smale, D.A., Wernberg, T., Oliver, E.C.J., Thomsen, M., Harvey, B.P., Straub, S.C., *et al.* 2019. Marine heatwaves threaten global biodiversity and the provision of ecosystem services. *Nature Climate Change.* 9(4):306–12.

Veron, J.E.N., 1990. Re-examination of the reef corals of Cocos (Keeling) Atoll. *Records of the Western Australian Museum*, 14(4): 553-581.

Wickham, H. 2016. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York.

Zhang, N. Feng, M. Hendon, H.H. Hobday, A.J. & Zinke, J., 2017. Opposite polarities of ENSO drive distinct patterns of coral bleaching potentials in the southeast Indian Ocean. *Scientific Reports,* 7(1): 2443

8.0 Appendix

Appendix A		2016		2017	
Hard Coral Genera	**Site**	**Mean Cover (%)**	Std Error	**Mean Cover (%)**	Std Error
Acropora	1	**24.5**	2.3	**23.3**	5.0
	2	**7.5**	2.3	**8.3**	3.6
	3	**15.9**	4.1	**19.6**	8.3
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.9**	0.0	**0.9**	0.0
	Mean (Southern Atoll)	**9.8**	1.7	**10.4**	3.4
	6 (PKNP)	**5.2**	2.4	**3.4**	1.7
Favia	1	**0.0**	0.0	**0.0**	0.0
	2	**15.3**	0.6	**19.7**	3.1
	3	**6.3**	2.2	**9.0**	1.1
	4	**0.0**	0.0	**0.4**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**4.3**	0.5	**5.8**	0.8
	6 (PKNP)	**0.2**	0.0	**0.9**	0.0
Pavona	1	**0.0**	0.0	**0.0**	0.0
	2	**23.6**	2.0	**22.5**	0.5
	3	**4.0**	2.9	**3.5**	1.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**5.5**	1.0	**5.2**	0.3
	6 (PKNP)	**0.4**	0.0	**0.0**	0.0
Porites	1	**1.4**	0.3	**0.2**	0.0
	2	**0.9**	0.3	**2.0**	0.0
	3	**8.4**	1.3	**5.0**	1.8
	4	**7.4**	1.1	**8.5**	2.6
	5	**0.2**	0.0	**0.4**	0.0
	Mean (Southern Atoll)	**3.7**	0.6	**3.2**	0.9
	6 (PKNP)	**1.3**	0.7	**1.3**	0.5
Pocillopora	1	**1.8**	0.2	**2.1**	0.6
	2	**4.1**	0.8	**4.4**	0.2
	3	**4.7**	0.2	**5.4**	0.1
	4	**0.0**	0.0	**0.2**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**2.1**	0.2	**2.4**	0.2
	6 (PKNP)	**2.5**	1.0	**2.4**	0.2
Montipora	1	**2.5**	1.2	**0.9**	0.0
	2	**1.0**	0.3	**0.9**	0.2
	3	**2.2**	1.0	**2.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.2**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**1.2**	0.5	**0.8**	0.0
	6 (PKNP)	**8.6**	1.7	**5.7**	1.8
Echinopora	1	**0.0**	0.0	**0.0**	0.0
	2	**0.0**	0.0	**0.0**	0.0
	3	**0.0**	0.0	**0.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.6**	0.1	**1.3**	0.7
	Mean (Southern Atoll)	**0.1**	0.0	**0.3**	0.1
	6 (PKNP)	**0.0**	0.0	**0.4**	0.0
Astreopora	1	**0.4**	0.0	**0.4**	0.0
	2	**0.0**	0.0	**0.0**	0.0
	3	**0.7**	0.2	**0.4**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**0.2**	0.0	**0.2**	0.0
	6 (PKNP)	**0.7**	0.0	**0.2**	0.0

Appendix A *(cont)*		2016		2017	
Hard Coral Genera	**Site**	**Mean Cover (%)**	Std Error	**Mean Cover (%)**	Std Error
Fungia	1	**0.2**	0.0	**0.4**	0.0
	2	**0.0**	0.0	**0.0**	0.0
	3	**0.2**	0.0	**0.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.7**	0.0	**0.2**	0.0
	Mean (Southern Atoll)	**0.2**	0.0	**0.1**	0.0
	6 (PKNP)	**0.4**	0.0	**0.0**	0.0
Goniastrea	1	**0.4**	0.0	**0.0**	0.0
	2	**0.0**	0.0	**0.5**	0.3
	3	**0.6**	0.3	**0.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**0.2**	0.1	**0.1**	0.1
	6 (PKNP)	**0.5**	0.1	**0.7**	0.2
Isopora	1	**0.0**	0.0	**0.0**	0.0
	2	**0.0**	0.0	**0.4**	0.0
	3	**0.0**	0.0	**0.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**0.0**	0.0	**0.1**	0.0
	6 (PKNP)	**0.0**	0.0	**0.0**	0.0
Leptastrea	1	**0.0**	0.0	**0.0**	0.0
	2	**0.0**	0.0	**0.0**	0.0
	3	**0.2**	0.0	**0.2**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**0.0**	0.0	**0.0**	0.0
	6 (PKNP)	**0.7**	0.0	**1.1**	0.0
Favites	1	**0.0**	0.0	**0.0**	0.0
	2	**0.2**	0.0	**0.0**	0.0
	3	**0.0**	0.0	**0.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**0.0**	0.0	**0.0**	0.0
	6 (PKNP)	**0.0**	0.0	**0.0**	0.0
Lobophyllia	1	**0.0**	0.0	**0.0**	0.0
	2	**0.2**	0.0	**0.0**	0.0
	3	**0.0**	0.0	**0.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**0.0**	0.0	**0.0**	0.0
	6 (PKNP)	**0.0**	0.0	**0.0**	0.0
Herpolitha	1	**0.0**	0.0	**0.0**	0.0
	2	**0.0**	0.0	**0.0**	0.0
	3	**0.0**	0.0	**0.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**0.0**	0.0	**0.0**	0.0
	6 (PKNP)	**0.0**	0.0	**0.0**	0.0
Pachyseris	1	**0.0**	0.0	**0.0**	0.0
	2	**0.0**	0.0	**0.0**	0.0
	3	**0.0**	0.0	**0.0**	0.0
	4	**0.0**	0.0	**0.0**	0.0
	5	**0.0**	0.0	**0.0**	0.0
	Mean (Southern Atoll)	**0.0**	0.0	**0.0**	0.0
	6 (PKNP)	**0.0**	0.0	**0.0**	0.0