

**A Summary of
The Department of Fisheries,
Western Australia
Invertebrate Research at
Cocos (Keeling) Islands
2006 – 2011**

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Correct citation:

Bellchambers, L. M. and Evans, S. N. (2013). A Summary of The Department of Fisheries, western Australia Invertebrate Research at Cocos (Keeling) Islands 2006 – 2011. Fisheries Research Report No. 239. Department of Fisheries, Western Australia. 72p.

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1.0 Executive Summary

The Cocos (Keeling) Islands are 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. Management of the Cocos (Keeling) Islands' fish resources is conducted by the Department of Fisheries Western Australia (DoF), on behalf of the Commonwealth Government, under a Service Delivery Agreement (SDA). Between 2006 and 2011 DoF has conducted annual risk assessments to focus research and management objectives for the Cocos (Keeling) Islands. This report summarises the research on three invertebrate groups (holothurians, *Lambis lambis* and giant clams) that were highlighted as high-risk, either due to lack of knowledge and/or current/potential fishing pressure. The key findings of the research projects are discussed below.

Currently there is no commercial or recreational (including subsistence) fishing for holothurians on the Cocos (Keeling) Islands. However, recent interest in developing a commercial fishery for holothurians at the Cocos (Keeling) Islands resulted in a survey to provide baseline data on the previously unfished local holothurian populations. Fourteen species of holothurians were recorded during the survey with five species having relatively high abundance, the most abundant species being *Holothuria atra*. A total of 20,556 holothurians were counted however 97% of these were considered to be of low commercial value. The high and medium value species found in this survey were all in extremely low abundances, with restricted distributions. Several species displayed distinct habitat preferences; *H. atra* was associated with sand dominated habitats, *Actinopyga mauritiana* was associated with relic reefs and soft corals, while *Holothuria fuscopunctata* and *Stichopus herrmanni* were both associated with reef flats. The densities recorded in this study represent the natural abundance and distribution of holothurian populations at this atoll. Given the low numbers of commercially important species it is highly unlikely that a commercial fishery would be economically viable at the Cocos (Keeling) Islands and any benefit may be outweighed by the ecological benefits of maintaining the natural holothurian population.

Lambis lambis (or gong gong) is a gastropod mollusc regarded as a delicacy by the Cocos Malay population of the Cocos (Keeling) Islands. Although there are no recreational catch records, historical surveys indicate that *L. lambis* have been heavily fished at the Cocos (Keeling) Islands in the last thirty years. DoF have conducted surveys of *L. lambis* stocks annually since 2007 to assess trends in their abundance and distribution to provide an indication of the sustainability of recreational fishing for the species. A comparison of DoF data (2007 – 2011) with historical data (1992) shows average densities of *L. lambis* have decreased significantly. *L. lambis* has been identified as one of the most vulnerable species to overfishing in the Cocos (Keeling) Islands. This report supports these concerns with large reductions in densities recorded over a 15-year period and significant reductions occurring over the last 5 years. It is likely that overfishing has played a role in the decrease in density of *L. lambis*.

Giant clams (*Tridacna spp*) are a popular food source for the local Cocos Malay population and anecdotal information suggests that giant clams may have been collected for some time for use as food and ballast by passing ships. Given the status of giant clams worldwide and their inherent vulnerability to overexploitation, DoF conducted a comprehensive survey in 2011 to document the distribution, abundance and size frequency of giant clam populations at the Cocos (Keeling) Islands. Only two species of giant clams (*Tridacna maxima* and *Tridacna derasa*) were identified in the survey. The population was dominated by *T. maxima* with only

one *T. derasa* recorded. No *T. gigas* were recorded during this survey and anecdotal reports suggest the species may be locally extinct. The current recreational harvest of giant clams at the Cocos (Keeling) Islands is unknown. Therefore, broad estimates were calculated to provide some understanding of the potential scale of recreational harvest. Estimates of recreational take indicate that catches of *T. maxima* are close to maximum sustainable yield therefore, extremely conservative limits should be set on any additional take (recreational or commercial).

Of the invertebrate groups surveyed by the three projects in this report only holothurians are unharmed therefore only their densities represent a natural population. Both *L. lambis* and giant clams are the targets of significant recreational fishing pressure. Both species also have life history traits that make them particularly vulnerable to overexploitation. There is an urgent need for the implementation and enforcement of fisheries management to ensure the sustainability of stocks of targeted invertebrate species. 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 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 Cocos (Keeling) Islands are comprised of 27 separate islands of two coral atolls, of which two (Home and West Islands) are inhabited by a total population of approximately 608 people (Australian Bureau of Statistics, 2011 Census). The main Cocos (Keeling) atoll is approximately 165 km² and consists of 26 islands surrounding a shallow lagoon (Woodroffe *et al.* 1994, Woodroffe and McLean 1994). The smaller North Keeling Island atoll is located approximately 24 km to the north of the main Cocos (Keeling) atoll and is approximately 2 km² (Woodroffe *et al.* 1994, Woodroffe and McLean 1994).

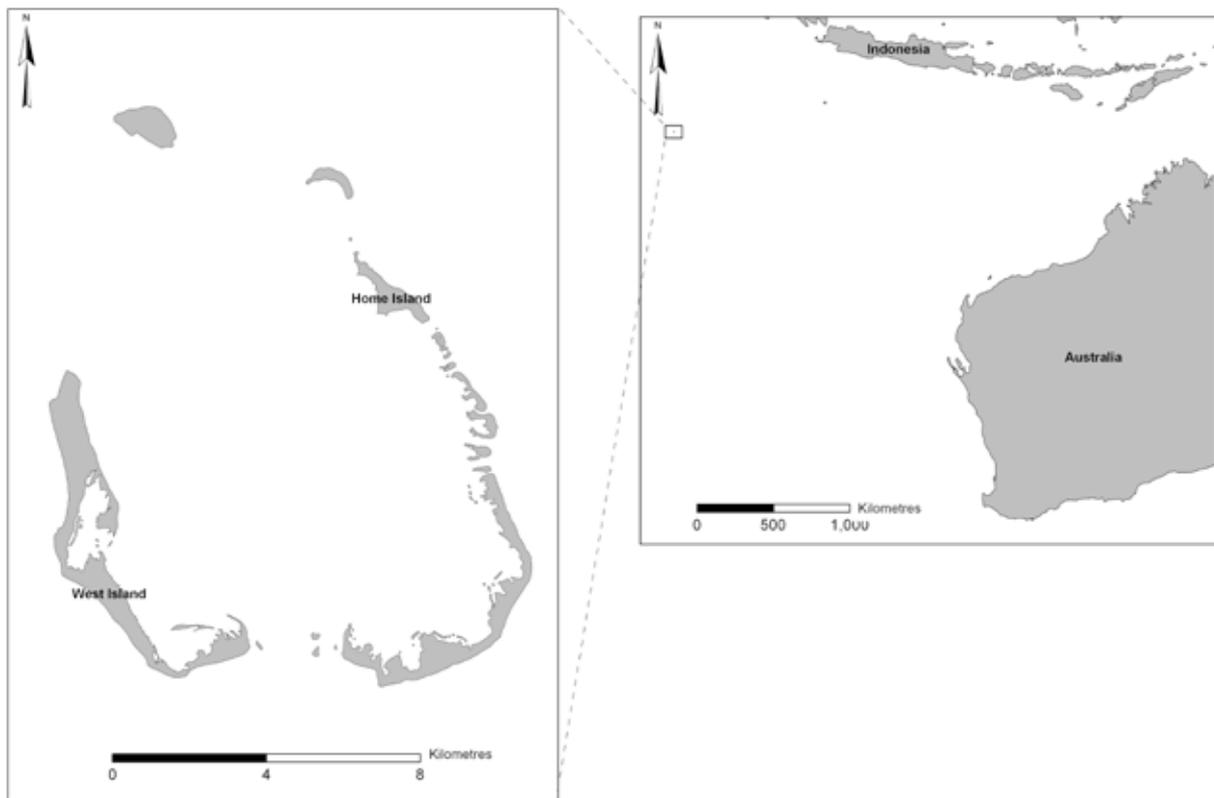


Figure 2.1. Location of the Cocos (Keeling) Islands relative to mainland Australia.

The Cocos (Keeling) Islands 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 is probably directly (via herbivores) or indirectly (via detritivores) at the base of the local food chain and is therefore 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 important as fish nursery areas (Berry 1989).

The Cocos (Keeling) Islands 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 Agreement (SDA). On Home Island, there are approximately 417 Cocos Malay residents, while around 190 Australian mainland-based workers occupy West Island. Both cultures on the atoll undertake fishing activities. 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). The Australian mainland-based 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 Cocos (Keeling) Islands, targeting fish for the marine aquarium trade, with the primary target being the endemic Cocos Angelfish (*Centropyge jocularis*). In 2000, DoF sought expressions of interest in developing new fisheries and the only invertebrate fishery highlighted as a possibility was holothurians, all other fishing is recreational. Various fish species (e.g. Lethrinids and Serranids) are caught by both the Cocos Malay and Australian mainland-based 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 (*Lambis lambis*) and giant clams (*Tridacna sp.*). The common spider shell or gong gong (*L. lambis*) is important to the local Cocos Malay population and is often collected in large numbers.

Currently, there are no legislated Island-specific fishing rules at the Cocos (Keeling) Islands although recreational fishing guidelines were introduced by DoF in 2006. The guidelines were designed to develop community engagement and acceptance of the concept of sustainability and daily limits and are not presently enforced. However, it is not known if the guidelines will ensure that fishing is restricted to sustainable levels. Currently, no Fisheries compliance officers are based on the islands, although. Between 2006 and 2011 DoF has conducted annual risk assessments to focus research and management objectives. This report summarises the subsequent research on three invertebrate groups (holothurians, 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 Abundance and Distribution of Holothurians

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Related Publication: Bellchambers L.M., Meeuwig J. J., Evans S. N. and Legendre P. 2011. Modelling habitat associations of 14 species of holothurians from an unfished coral atoll: implications for fisheries management. *Aquatic Biology* 14: 57-66

3.1 Introduction

Effective management of the fish resources of the Cocos (Keeling) Islands, such as the various holothurian species, is important given the valuable ecological role these organisms play in the marine environment. Studies of coral reef holothurians found they play an important ecological role in nutrient cycling and bioturbation processes (disturbance of sediment by burrowing and feeding) in marine benthic communities (Uthicke 1999, Skewes *et al.* 2002, Mangion *et al.* 2004, Uthicke *et al.* 2004b, Purcell 2010). Modeling has showed that a mixed population of *Holothuria atra* and *Stichopus chloronotus* has the potential to rework 4600 kg of sediment (dry weight) per year (Uthicke 1999). Often the productivity of a habitat, particularly in coral reef systems, is a result of carbon fixation and nutrient recycling provided by benthic micro-algae (Uthicke and Klumpp 1998, Uthicke *et al.* 2004b). Coral reefs are regarded as one of the most productive marine ecosystems (Uthicke and Klumpp 1998) and are estimated to provide \$375 billion (USD) worth of goods and services (Costanza *et al.* 1997, Skewes *et al.* 2002). Previous studies have suggested that holothurians contribute to the oxygenation of the upper sediment levels and play a role in bioturbation that in turn may increase the productivity of benthic micro-algae (Uthicke and Klumpp 1998, Uthicke 1999, Uthicke and Karez 1999, Uthicke *et al.* 2004b, Uthicke *et al.* 2009), providing increased food resources for other species. The ecological consequences of removing holothurians by fishing are unknown; however, given their important functions in their environment, it is likely that removal of these animals may reduce the overall productivity of coral reefs (Uthicke *et al.* 2004b, Friedman *et al.* 2010).

Holothurians are often amongst the most valuable resources for small tropical island communities (Kinch *et al.* 2008, Purcell 2010). Several species (e.g. *Actinopyga mauritiana* and *Holothuria atra*) are highly prized by subsistence fishers as a direct food source (Kinch *et al.* 2008) and by commercial operators (e.g. *Holothuria fuscogilva*, *H. whitmaei* [formerly *H. nobilis*] and *H. scabra*) due to the high demand from Asian markets (Conand and Byrne 1993, Choo 2008, Purcell 2010). Holothurians are particularly susceptible to overharvesting with most fisheries severely over exploited (Conand 2004, Toral-Granda 2006, Friedman *et al.* 2010, Purcell 2010). Conand (2004) identified 42 species under population stress as a result of commercial exploitation. Similarly, an international review of 28 countries found that 21 countries, including Australia, had over exploited holothurians stocks, five countries had declining stocks and Malaysia had one species close to extinction. Of the 28 countries reviewed only Cuba had apparently stable stocks (Toral-Granda 2006). Holothurian fisheries are prone to over-exploitation because there is often little or no scientific data on stock size before or after fishing begins and consequently, fisheries are inadequately managed.

Currently there is no recreational or commercial harvest of holothurians at the Cocos (Keeling) Islands. Previous studies estimated the standing stock of holothurians to be 92,770,199 ($\pm 14,564,921$ [95 % C.I.]) and this likely represents the population in its natural state (Hender *et al.* 2001). The Australian Fisheries Management Authority (AFMA) indicated that previous

surveys were inadequate for assessing holothurian stocks because the reef flats and outer reef slopes of the Cocos (Keeling) Islands had not been surveyed (Anon 2002). Therefore, a comprehensive assessment of the abundance and commercial viability of holothurians stocks at the Cocos (Keeling) Islands is required to make an informed decision about the sustainability and viability of a future fishery.

3.1.1 Objectives

The overall goal of this research was to assess the stocks of holothurians at the Cocos (Keeling) Islands, in order to make an assessment of the feasibility of commercial fishing for holothurians. Specifically, this comprised of three objectives:

1. Updated and more comprehensive estimates and maps of the distribution and densities of holothurians and associated habitats;
2. An updated indication of the status of stocks, including a comparison with previous studies and other regions of the world; and
3. Management recommendations for the regulation of developmental fisheries.

3.2 Methods

3.2.1 Survey methodology

Field surveys were undertaken by a four person team between the 8 – 25th of May 2006. The study was limited to depths of less than 29 metres (due to limits of diving) and covered all habitats including the intertidal reef flats, shallow lagoon habitats and the outer reef slopes. Sampling was conducted systematically across the entire atoll by dividing the atoll into one minute of latitude and longitude blocks and then randomly selecting a site within each of these blocks (Figure 3.1). This equated to a sampling density of one site per 1.8 km². Our survey sites also included sites previously surveyed by Berry (1989) and Hender *et al.* (2001). A total of 79 sites were sampled.

This survey used similar rapid assessment techniques to those used in holothurian surveys at Warrior Reef, Torres Strait (Skewes *et al.* 2000), Timor Sea MOU Box (Skewes *et al.* 1999) and Milne Bay Province, PNG (Skewes *et al.* 2002). Either SCUBA, snorkelling or reef walks were used depending on the water depth at each site. Sites were located using a hand held GPS. Once located, two divers entered the water and swam adjacent 100 metre x 4 metre belt transects spaced 10 metres apart on a predetermined bearing. Every 10 metres along the transect the abundance of holothurian species were recorded and a visual assessment was made of the percentage cover of the dominate substrate and habitat types.

Paired t-tests indicated that there were no significant differences between the observations on the two replicate transects at each site, so the abundance of holothurian species and percentage cover of each habitat were averaged across the two transects to give the overall density of each holothurian species and the average habitat percentage cover for each site.

3.2.2 Habitat categories

In this study habitat is categorised in two different ways: fine-scale and broad-scale. The fine-scale habitat data was collected during the current survey using the methods described above. The fine-scale habitat categories used in the current study are: sand, rubble (limestone

rubble), seagrass, macroalgae, massive coral (corals that are solid and similar in shape in all directions), submassive coral (corals less than, or not quite, solid and similar in shape in all directions), plate coral (or laminar, forming a tier), foliose coral (forming a whorl), soft coral (coral without exoskeleton), branching coral (forms branches), branching coral (dead), reef flat (limestone platform) and relic reef (old eroded limestone reef). Fine-scale habitat categories were based on those used by previous studies to allow for comparisons (i.e. Williams 1994, Hender *et al.* 2001). However, for estimates of holothurian standing biomass habitat categories that could be quantified across the entire lagoon were required. Therefore, each of the sites surveyed during the current study were also allocated a broad-scale habitat category, adapted from Williams (1994).

3.2.3 Standing biomass

Standing biomass was estimated by calculating the average density of holothurians per 400 m² for each broad-scale habitat type (Williams 1994) converting individuals ha⁻¹ and then multiplying by the hectares of each habitat type. This was done for all holothurians and then for each economic category (high, medium, low, unknown; Purcell 2010).

3.3 Results

2.3.1 Holothurians of the Cocos (Keeling) Islands

Fourteen species of holothurians belonging to seven genera were recorded on the transects (Table 3.1) with four additional species identified outside the surveyed transects. A total of 20,556 holothurians were identified and counted, with an average density of 3,252 individuals ha⁻¹; 97 % of these are considered of low commercial value. The most abundant species was *Holothuria atra* which had a mean density of 2,983 individuals ha⁻¹ (\pm 455 SE). Of the fourteen species recorded during the survey, only four species had relatively high abundance albeit significantly lower than *H. atra*. *Holothuria fuscopunctata* was the second most abundant species with a mean density of 100 individuals ha⁻¹ (\pm 32.7 SE) followed by *Synapta maculata* (64.4 individuals ha⁻¹; \pm 32.6 SE), *Holothuria edulis* (53.6 individuals ha⁻¹; \pm 17.1 SE), and *Stichopus chloronotus* (36.2 individuals ha⁻¹; \pm 11.0 SE). The remaining holothurian species recorded during the survey had sporadic distributions and low abundances. No holothurians were observed at 15 of the 79 sites surveyed.

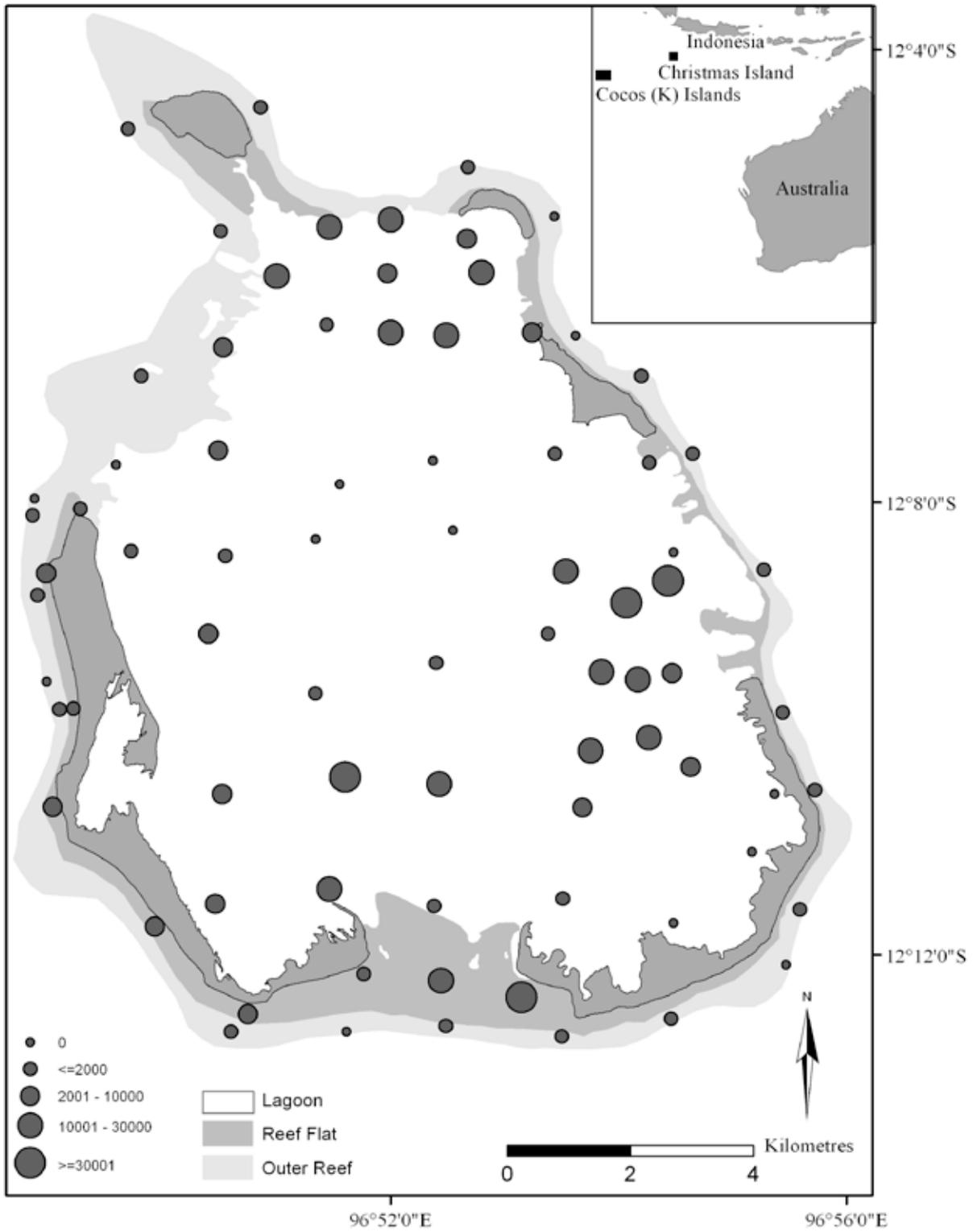


Figure 3.1. Location of Cocos (Keeling) Islands displaying sampling sites and abundance of, holothurians.

Table 3.1. Species, abundance and commercial value of holothurians surveyed at the Cocos (Keeling) Islands. Commercial values of holothurian species are from Purcell (2010). Species marked with * were present at the survey sites but were outside the area encompassed by the 100 m x 4 m transects.

| Species Name | Common Name | Commercial Value | Mean number individuals ha ⁻¹ | SE |
|---|----------------------|------------------|--|-------|
| <i>Actinopyga mauritiana</i> (KUY) | Surf redfish | Medium | 3.6 | 1.6 |
| <i>Actinopyga miliaris</i> (KUQ) | Hairy Blackfish | Medium | 0.63 | 0.5 |
| <i>Bohadschia argus</i> (KUW) | Leopardfish | Low | 7.4 | 2.5 |
| <i>Chiridota rigida</i> (CRI) | | Unknown | 0.16 | 0.2 |
| <i>Holothuria atra</i> (HFA) | Lolly fish | Low | 2 983 | 455.4 |
| <i>Holothuria coluber</i> (HHW) | Snake fish | Low | 0.16 | 0.2 |
| <i>Holothuria edulis</i> (HFE) | Pink fish | Low | 53.6 | 17.1 |
| <i>Holothuria fuscopunctata</i> (HOZ) | Elephant Trunkfish | Low | 100.2 | 32.7 |
| <i>Holothuria fuscogilva</i> (HFN) | White teatfish | High | 0.32 | 0.2 |
| <i>Pearsonothuria graeffei</i> (EHV) | Flowerfish | Low | 1.1 | 0.8 |
| <i>Stichopus chloronotus</i> (JCC) | Greenfish | Low | 36.2 | 11.0 |
| <i>Stichopus herrmanni</i> (JCV) | Curryfish | Medium | 0.63 | 0.3 |
| <i>Thelenota ananas</i> | Prickly redfish | Medium | 0.63 | 32.6 |
| <i>Synapta maculata</i> (RSF) | Spotted sea cucumber | Unknown | 64.4 | 0.3 |
| <i>Actinopyga echinites</i> * | Deep water redfish | Medium | | |
| <i>Bohadschia marmorata</i> * | Chalkfish | Low | | |
| <i>Holothuria scabra</i> * | Sandfish | High | | |
| <i>Holothuria whitmaei</i> * (formerly <i>H. nobilis</i>) | Black teatfish | High | | |

3.3.2 Habitats of Cocos (Keeling) Islands and distributions of holothurians

The habitat in the lagoon was primarily sand with seagrass beds along the eastern and southern edges (Figure 3.2). There was a central patch of live coral that consisted predominately of foliose and branching coral. The highest macro-algae abundance occurred in the southeast corner of the lagoon while the outer reef slopes were dominated by hard and soft corals consisting primarily of massive and branching morphologies. Plate morphologies were mainly found on the western reef slopes of the atoll. The majority of corals in the sand habitats of the lagoon were of massive morphology.

The most abundant species *H. atra* was primarily located inside the lagoon in sand habitats. *H. fuscopunctata* occurred on the outer reef slopes of the Cocos (Keeling) Islands in reef areas dominated by hard and soft corals with only a few individuals recorded within the lagoon. *S. maculata* had an isolated distribution within the lagoon being concentrated in the sand-dominated habitats of the southeast corner of the lagoon. While *H. edulis* was also found in sand habitats it was most abundant at the northern end of the lagoon. The remaining holothurian species recorded during this survey had sporadic distributions with low abundances (Table 3.1).

3.3.3 Standing biomass of holothurians

Standing abundance, weighted for mean density of holothurians per broad-scale habitat type and the areal extent of each habitat was estimated at 44.8 million holothurians (with 95 % lower and upper confidence limits of 9.9 million and 63.3 million) or approximately 3,340 individuals ha⁻¹ (Table 2.2). Of these, 96.6 % are considered to be of low economic value (mostly *H. atra*) an additional 3.3 % are of unknown value while only 0.12 % are medium or high value. The most important habitat for holothurians in the lagoon was coral and algal flats. This habitat was relatively common (13.2 %) and supported the highest abundances of holothurians, i.e. 39 % of the standing abundance of holothurians (Table 3.2). Prograding sand sheets comprised only 6.5 % of the area but had the second highest density of holothurians, supporting 16 % of all holothurians (Table 3.2). Species of high economic value were found in only two habitats outside the lagoon: outer reef coral terrace and coral and algal flats associated with reef flat. The outer reef coral terrace was relatively extensive (16.3 %) but had low densities of holothurians including economically important species. The coral and algal flats associated with the reef flat were moderately extensive (7.2 % of the atoll) with higher densities of holothurians, but economically-important species were an order of magnitude lower in abundance than on the outer reef terrace.

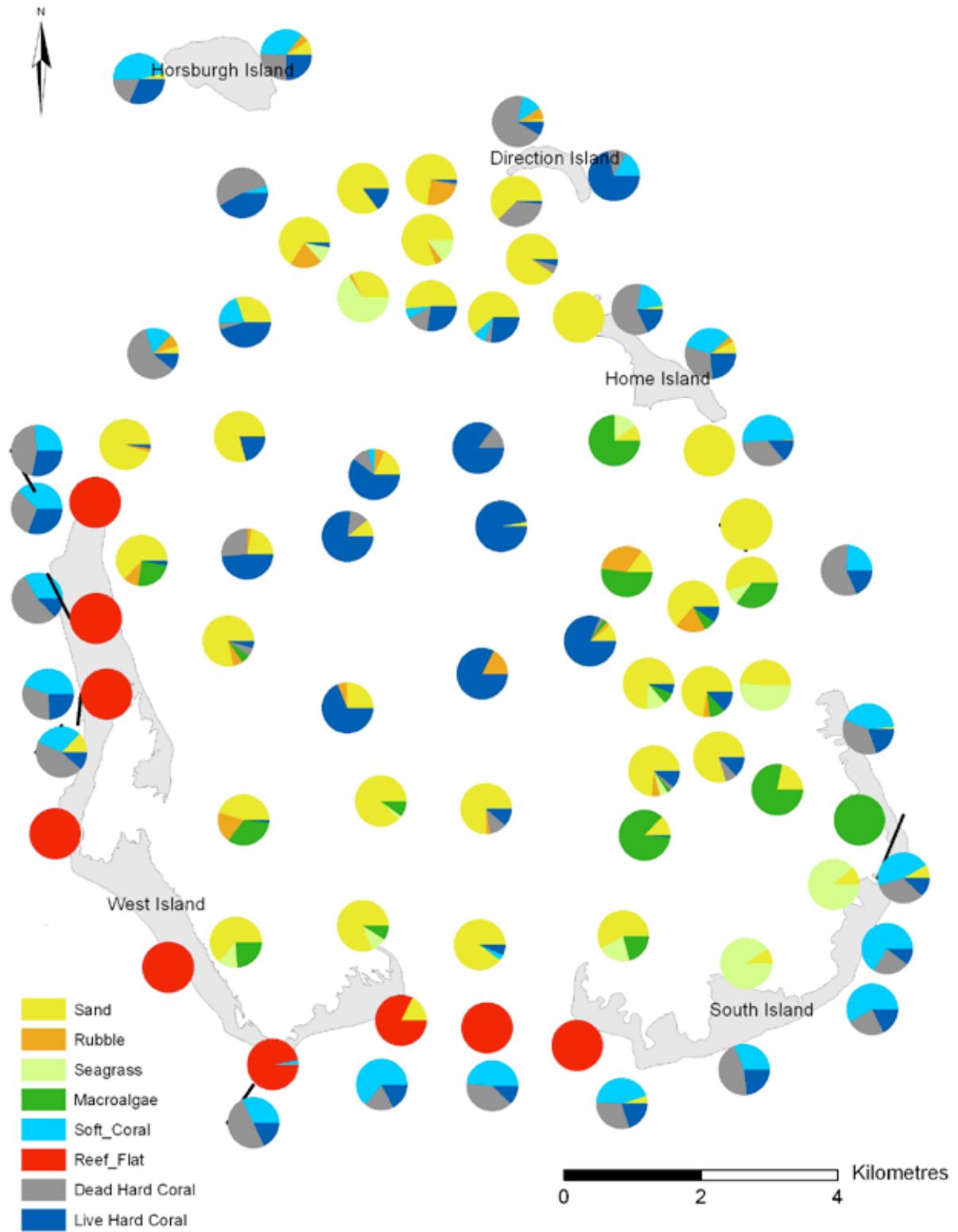


Figure 3.2. Benthic habitats at Cocos (Keeling) Islands

Table 3.2. Standing biomass of holothurians (individuals ha⁻¹) at Cocos (Keeling) Islands by habitat and commercial value. Habitat categories are based on Williams (1994) and commercial values are based on Purcell (2010).

| Outer Reef Flat | Habitats | Area (ha) | % of total | All Holothurians | | | | | | Holothurians by Commercial Value | | | |
|-----------------|---------------------------------------|---------------|------------|------------------|------------------|----------------------|------------------------------|-------------------------------|-------------------|----------------------------------|---------------|-------------------|------------------|
| | | | | No. / ha (Lower) | No. / ha (Upper) | Total No. by habitat | Total No. by Habitat (Lower) | Total No. by Habitat (Higher) | High | Medium | Low | Unknown | |
| | Coral Terrace | 2,186 | 16.3 | 146 | 37 | 200 | 318,034 | 81,258 | 436,422 | 1,093 | 24,044 | 289,619 | 3,279 |
| Reef Flat | Coral and algal flat | 961 | 7.2 | 3,633 | 153 | 5,374 | 3,491,852 | 146,868 | 5,164,345 | 1,335 | 8,009 | 3,482,509 | 0 |
| Lagoon | Prograding sand sheet | 872 | 6.5 | 8,454 | 1,596 | 11,883 | 7,375,966 | 1,392,750 | 10,367,574 | 0 | 3,635 | 7,312,349 | 59,982 |
| | Seagrass sand and silt flat | 1,295 | 9.7 | 847 | 0 | 1,310 | 1,097,045 | 0 | 1,697,104 | 0 | 0 | 1,097,045 | 0 |
| | Coral and algal flat | 1,769 | 13.2 | 9,825 | 3,522 | 12,977 | 17,377,046 | 6,228,859 | 22,951,140 | 0 | 2,456 | 17,320,548 | 54,042 |
| | Blue hole mosaic 1 | 315 | 2.3 | 7,156 | 0 | 12,528 | 2,252,272 | 0 | 3,942,841 | 0 | 0 | 2,252,272 | 0 |
| | Blue hole mosaic 2 | 914 | 6.8 | 5,808 | 0 | 9,211 | 5,308,893 | 0 | 8,418,528 | 0 | 3,808 | 3,960,724 | 1,344,361 |
| | Blue hole mosaic 3 | 431 | 3.2 | 38 | 38 | 38 | 16,181 | 16,181 | 16,181 | 0 | 0 | 16,181 | 0 |
| | Algal covered staghorn rubble | 1,315 | 9.8 | 22 | 0 | 43 | 28,767 | 0 | 56,957 | 0 | 0 | 28,767 | 0 |
| | Massive coral outcrops | 1,263 | 9.4 | 3,877 | 1,496 | 5,067 | 4,896,167 | 1,889,852 | 6,399,324 | 0 | 2,255 | 4,884,891 | 9,021 |
| | Sandy lagoon floor with coral | 685 | 5.1 | 3,825 | 214 | 5,630 | 2,618,872 | 146,707 | 3,854,955 | 0 | 4,279 | 2,612,454 | 2,140 |
| | Seagrass bed (T. ciliatum) with coral | 81 | 0.6 | 388 | 388 | 388 | 31,419 | 31,419 | 31,419 | 0 | 0 | 31,419 | 0 |
| Totals: | | 13,410 | | | | | 44,812,514 | 9,933,893 | 63,336,791 | 2,428 | 48,487 | 43,288,775 | 1,472,825 |

3.4 Discussion

The current study identified a total of fourteen species of holothurians at the sites surveyed, with a further four species identified outside the survey sites. A previous survey of holothurians at Cocos (Keeling) Islands identified thirty-four species of holothurians (Marsh 1994). However, it is difficult to compare the results of the two studies. Marsh (1994) focused on identifying species diversity by searching discrete areas, while the current study focused on quantifying the abundance and distribution of holothurians across the entire atoll using a spatially stratified survey. While Marsh (1994) conducted a qualitative survey some observations on the abundance of nine species were made. Marsh (1994) stated that *H. atra* was the most common species; *H. nobilis* and *T. ananas* were the most valuable species, but they were not abundant; *A. echinites* and *A. mauritiana* were common to abundant on reef flats and *B. marmorata*, *H. scabra* and *B. argus* were also common. With the exception of *A. mauritiana*, which was not found on the current survey, these statements are reflected in the results of our survey. In contrast, Hender *et al.* (2001) identified only eight species of holothurians at Cocos (Keeling) Islands. The higher number of species identified in the current survey, compared to Hender *et al.* (2001), may reflect the increased spatial coverage of the present survey which included reef slopes and reef flat habitats.

In the current study, an average total density of 3,252 individuals ha⁻¹ was recorded. This is approximately half the density recorded by Hender *et al.* (2001) of 7512 individuals ha⁻¹. The difference in the density of holothurians between the two surveys may be due to the habitats surveyed or survey techniques used. Hender *et al.* (2001) did not sample the reef flats or crests, which may have resulted in overestimating the abundance of holothurians across the atoll. The two studies also used different sampling techniques; Hender *et al.* (2001) surveyed holothurians species within 2 m x 2 m quadrats, while the current study used 100 m x 4 m belt transects. Scaling-up observations of densities from relatively small areas (i.e. quadrants) to densities on larger scales (e.g. individuals ha⁻¹) may result in overestimations (Andrew and Mapstone 1987). Previous studies have also suggested that cryptic or sheltering behaviour is common in holothurians and may lead to biases in abundance estimates (Hammond *et al.* 1985, Shiell and Knott 2008). However, as the current study used similar methods (surface or diver based observations) and surveys were conducted throughout the day, the contrasts in density estimates may also reflect the natural variability in the abundance of holothurians at the Cocos (Keeling) Islands.

In previous surveys of the Cocos (Keeling) Islands, *Bohadschia argus* and *Chiridota rigidia* formed the majority of the remaining holothurians in the lagoon (Hender *et al.* 2001). However, in the current survey *S. maculata*, *H. edulis*, and *S. chloronotus* formed the majority of the holothurian species occurring in the lagoon. This may be a true reflection of the variability of the holothurian populations at the Cocos (Keeling) Islands, but may also be the result of the increased number of survey sites compared to previous surveys. Incorrect identification of species in previous surveys may also account for some of the differences seen in the dominant species recorded between surveys.

Observed densities in this study of several holothurian species are higher than in the Indian Ocean, i.e. *H. atra* (Skewes *et al.* 2002, PNG; Aumeeruddy *et al.* 2005, Seychelles; Dissanayake and Stefansson 2010, Sri Lanka), *A. mauritiana* (Skewes *et al.* 2002, PNG; National Fisheries Authority 2007, PNG) and *H. fuscopunctata* (Skewes *et al.* 2002, PNG; Aumeeruddy *et al.* 2005, Seychelles; National Fisheries Authority 2007, PNG) (see Appendix A). However, several species recorded in the current survey had lower densities than those observed in other studies

Holothuria whitmaei (Skewes *et al.* 2002, PNG; National Fisheries Authority 2007, PNG), *A. echinites* (Aumeeruddy *et al.* 2005; Dissanayake and Stefansson 2010, Sri Lanka). Despite these differences in densities, given the current lack of subsistence or commercial fisheries targeting holothurians, the holothurians population at Cocos (Keeling) Islands represents an unexploited stock in its natural state.

The distribution and abundance of holothurians at the Cocos (Keeling) Islands is closely linked with the distribution and type of benthic habitats. Previous authors have reported that holothurians exhibit distinct habitat preferences (Uthicke and Benzie 2000, Shiell 2004, Conand 2008, Kinch *et al.* 2008). For example, populations of *H. nobilis* in northern Western Australia showed distinct preferences for outer-reef zones, specifically the reef flat and reef crest (Shiell 2004). Other studies have suggested that *H. atra* has no recognizable pattern of distribution (Massin and Doumenc 1986, Shiell 2004); however, in this study, *H. atra* was primarily found inside the lagoon in sand-dominated habitats. The distribution of this species may be due to its relatively unselective feeding habits (Uthicke and Karez 1999). In contrast, *H. fuscopunctata* and *S. chloronotus* both displayed a preference for reef flats; *H. fuscopunctata* occurred primarily on the outer reef slopes with only a few individuals recorded within the lagoon, while *S. chloronotus* occurred throughout the central region of the atoll. Previous studies have suggested that *Stichopus* spp. appear to select particular sediment types (Uthicke and Karez 1999), and the patchy distribution of *S. chloronotus* within the lagoon may be a result of its feeding preferences. In this study, while there appeared to be a strong association between habitat and the abundance and distribution of holothurians, other factors may also be contributing. Environmental drivers, such as sediment grain size and organic content, water depth and flow, may also be important drivers of holothurian distributions (Conand and Chardy 1985, Hammond *et al.* 1985, Uthicke and Karez 1999). Similarly, larval dispersion (Massin and Doumen 1986), recruitment (Purcell 2010) and behaviour (Shiell and Knott 2008) may also affect distribution patterns.

The abundance of the various holothurian species may also be a result of their reproductive characteristics. *H. atra* has been reported to have an extended period of reproduction (Uthicke 1996), and therefore, recruitment success is likely to be high. This may explain the dominance of the species in the Cocos (Keeling) lagoon. The relatively limited abundance (and distribution) of other species may be due to their lower reproductive output. For example, *H. nobilis*, appears to only have a short reproductive period (Uthicke 1996). In addition, *H. atra*, *H. edulis*, and *S. chloronotus* have been reported to reproduce asexually (via transverse fission) (Uthicke 1996) while other species (e.g. *H. nobilis*) have not been observed to reproduce asexually. This may explain why *H. atra*, *H. edulis* and *S. chloronotus* dominate holothurian populations at the Cocos (Keeling) Islands. It is thought that asexual reproduction may be the main method of population maintenance and growth in many holothurian populations (Uthicke 1996). However, there is currently no information about the reproductive strategies of the various holothurian species found at the Cocos (Keeling) Islands.

Despite observed patterns between holothurians and habitat, their ecology and population dynamics remain poorly understood and knowledge of the effects of reduced holothurian densities on the ecology of reefs and reef flats is limited. In a rare ecological study, abundance of the sea star, *Proteaster nodosus* increased following a decrease in holothurian numbers due to overfishing (Tomascki *et al.* 1997). Ecologically, holothurians are an important component of soft bottom communities (Conand 2008) and play an important role in benthic recycling (Uthicke and Klumpp 1998, Uthicke 2001, Mangion *et al.* 2004). Decreases in holothurian

numbers through overfishing may therefore have a major impact on reef ecology and resilience (Uthicke 2004, Uthicke *et al.* 2009, Friedman *et al.* 2010).

Many holothurian fisheries, both tropical and temperate, are in varying stages of overexploitation (Conand 2008, Purcell 2010, Friedman *et al.* 2010) with limited or no scientific data available on stock size before or after fishing. Holothurians display particular characteristics that make them vulnerable to recruitment overfishing including sessile, shallow water, and patchy distributions. In addition, many holothurian fisheries are based in developing countries where local communities are dependent on the income from, but lack the resources to manage, these relatively dynamic fisheries (Kinch *et al.* 2008, Friedman *et al.* 2010). A significant number of holothurian species in developed countries such as Australia are also overexploited (Kinch *et al.* 2008). The sandfish (*H. scabra*) fishery on Warrior Reef, Torres Strait, was closed in 1998 following several years of high fishing mortality (Skewes *et al.* 1999). The black teatfish (*H. whitmaei*, formerly *H. nobilis*) fishery off the east coast of Queensland was closed in 1999 due to overfishing, with stocks on fished reefs reduced to less than 25 % of that observed in areas closed to fishing (Uthicke and Benzie 2000). The sandfish fishery on the east coast of Australia at Hervey Bay was also closed in 2000 due to a severe stock decline. Fishery-independent surveys have occurred since the closures of these fisheries, and to date there is little evidence of recovery (Skewes *et al.* 1999, Skewes *et al.* 2000, Uthicke *et al.* 2004b). The conclusion from these Australian fisheries suggests that holothurian abundance can be severely impacted by overfishing and that recovery following such exploitation may be slow.

3.5 Management Recommendations

The holothurian populations at the Cocos (Keeling) Islands are at risk of overexploitation should a fishery be established. Currently, species of low commercial value (96 % of total abundance) such as *H. atra* are dominant while species with high commercial value (e.g. *H. fuscogilva*) are relatively rare (< 0.1 % total abundance). As the population is unfished the low abundance particularly of high value species represents natural levels. However, there are few (if any) studies of unfished populations for comparison. High value species like *H. fuscogilva* and *H. whitmaei* with their low abundances, patchy distributions and prevalence in easily-accessible locations (eg. reef flats) would be highly vulnerable to over exploitation. This is further compounded by the isolation of the atoll which may result in limited larval input from external holothurian populations (Hender *et al.* 2001, Uthicke *et al.* 2004a). As many of the high value species are not known to reproduce asexually, a decrease in population numbers of these species from fishing activities may inhibit the population's ability to be self-sustaining (Uthicke *et al.* 2009). Ultimately, this could lead to localised extinction of these species at the Cocos (Keeling) Islands, which may have flow-on effects on the functioning and resilience of the atoll's reef communities. Therefore it is strongly recommended that no commercial harvesting of holothurians occur at the Cocos (Keeling) Islands.

4.0 Distribution, abundance and reproductive biology of *Lambis lambis* (gong gong) at the Cocos (Keeling) Islands

Bellchambers, L.M., Pember, M.B. and Evans, S.N.

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

4.1 Introduction

Lambis lambis (gong gong) is a member of the strombid family found throughout the Indo Pacific from Tonga to the east coast of Africa, where it reaches a maximum shell length of 29 cm (Poutiers 1998). The sexes are separate and the species is sexually dimorphic, with females significantly larger than males (Beesely *et al.* 1998). *L. lambis* is regarded as a delicacy by the Cocos Malay population of the Cocos (Keeling) Islands and is the target of intense recreational fishing pressure. It is not known how long *L. lambis* have been harvested, however, surveys indicate that collection only became popular in the last two decades (Lincoln-Smith *et al.* 1993). The fishery is currently unregulated, although fishing guidelines have been introduced, including a daily limit of 20 *L. lambis* per fisher.

While a number of papers have reported on the distribution and general morphology of *L. lambis*, there is little information on the biology, life history or fisheries of *L. lambis*. Therefore, information has been drawn from its close relative, the queen conch (*Strombus gigas*), which has been severely overfished in the Caribbean. In the Caribbean, *S. gigas* has been fished since pre-Columbian times (Stoner 1997, Schapira *et al.* 2009). Commercial fisheries have only developed in the last few decades (Theile 2003), and while it is primarily harvested for its meat, its shell also has important commercial value (Berg and Glazer 1995, Tewfik and Guzman 2003). Due to the overharvesting throughout the tropical and subtropical waters of the Caribbean, diverse stock management regulations have been in place since the 1970s. In some locations, the densities of *S. gigas* are so low that reproduction is failing due to lack of encounters between males and females, known as the “Allee Effect” (Stoner and Ray-Culp 2000, Tewfik and Guzman 2003, Kramer *et al.* 2009). The fisheries in Bermuda and Florida were completely closed in 1978 and 1986, respectively, and have not been reopened (Theile 2001). The conservation status of *S. gigas* is such that its international trade is controlled through a CITES Appendix II listing and an injunction to halt trade was in place in 2003 and 2004 (Acosta 2006).

S. gigas is thought to live for eight years, reaching sexual maturity at three to five years (Appeldoorn 1988), which is signified by the development of a flared lip, which may be analogous to the development of flared spines in *L. lambis*. The similarities between *S. gigas* and *L. lambis* also include probable late maturation, shallow water habitat, slow movement, and tendency to aggregate in shallow water for spawning. However, in contrast to other strombids that are harvested, there is not a single scientific paper on the fisheries biology of *Lambis* spp. Therefore, research into the biology and fishery for this species at the Cocos (Keeling) Islands has been highlighted as a priority in all recent studies (Lincoln-Smith *et al.* 1993, Hender *et al.* 2001). It is considered to be the invertebrate species currently most at risk from overexploitation, based on current fishing pressure and the experience of related species in other parts of the world.

4.1.1 Objectives

The overall goal of this research was to assess the stocks of *L. lambis* at the Cocos (Keeling) Islands to make an assessment of the sustainability of recreational fishing for *L. lambis* at the atoll.

The project was divided into two areas with the following objectives:

1. Ecology, distribution and abundance:
 - a) To assess the abundance and distribution of *L. lambis* and associated habitats in the Cocos (Keeling) Islands lagoon;
 - b) To compare the current distribution and abundance of *L. lambis* with historical levels (Lincoln-Smith *et al.* 1993); and
 - c) To develop an on-going monitoring program to assess *L. lambis* stocks and assess the effectiveness of any management initiatives.
4. Reproductive biology:
 - a) To determine the timing and duration of spawning period of *L. lambis*.

4.2 Methods

Field surveys by DoF to determine the distribution and abundance of *L. lambis* in the Cocos (Keeling) lagoon were conducted annually between 2007 – 2011. The 2007 survey was a pilot study focused primarily on habitats close to Home Island where *L. lambis* are found in high numbers and the majority of historical fishing has occurred (Mohammad Chongkin, Parks Australia, pers. comm.). Where possible, sites surveyed by previous studies were also included (i.e. Lincoln-Smith *et al.* 1993, Hender *et al.* 2001) to allow historical comparisons. In 2008 an expanded survey of 67 sites was conducted which included all of the 2007 sites plus a number of additional sites to ensure a greater coverage of the suitable shallow water environments as described by Williams (1994) and local fishers. The 2008 data provided a baseline to design and implement a long-term monitoring program, with 41 of the 67 sites surveyed in 2008 used for ongoing monitoring (2009 – 2011) (Figure 4.1).

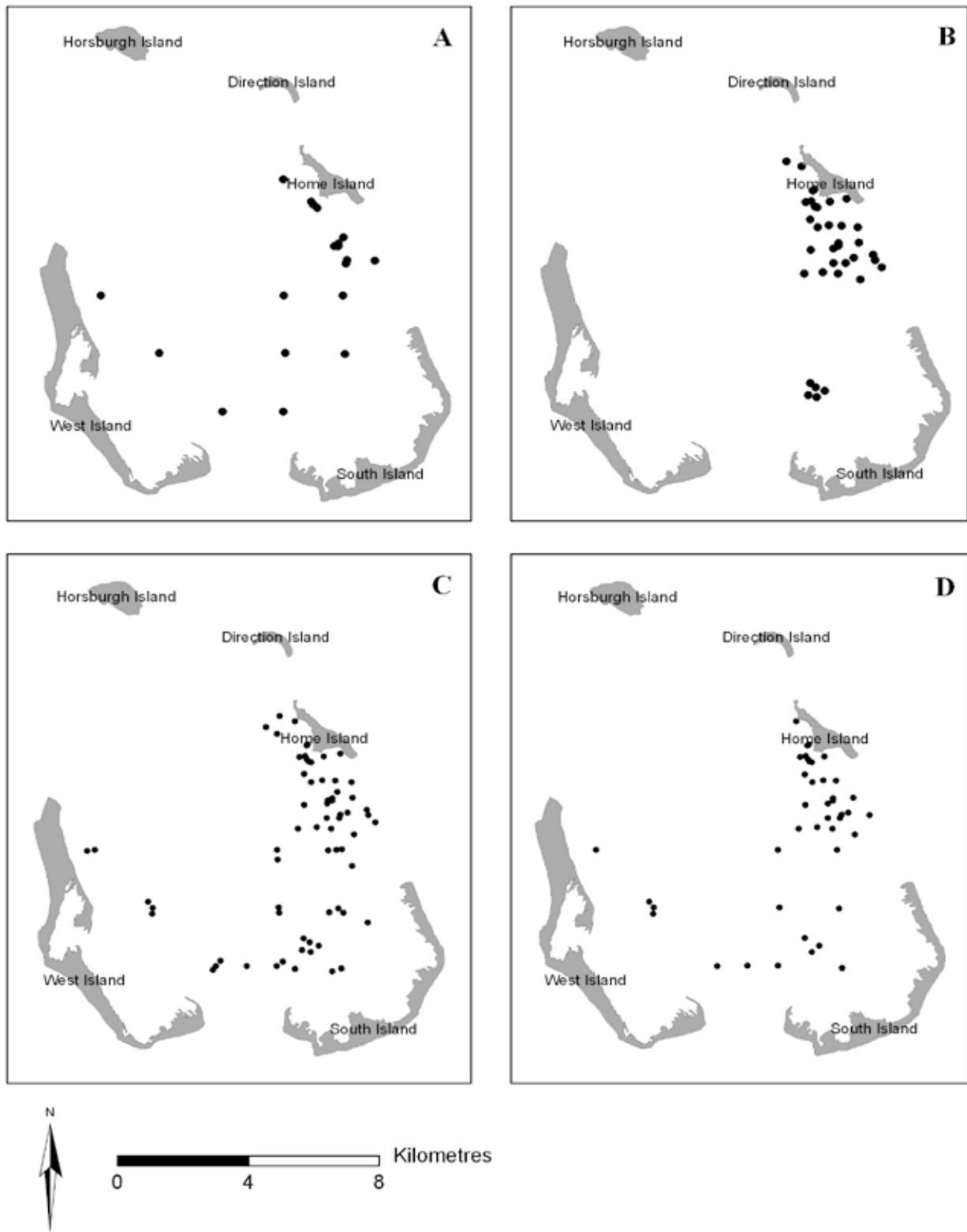


Figure 4.1. Location of *L. lambis* survey sites for each survey year. (A) sites surveyed by previous studies that were repeated by DoF; (B) sites surveyed in 2007; (C) sites surveyed in 2008; (D) sites surveyed from 2009 to 2011.

4.2.1 Ecology, distribution and abundance

Each of the sampling sites were conducted using SCUBA or snorkel along two parallel 100 m x 2 m belt transects, spaced approximately 10 m apart. The total number of *L. lambis* and the percent cover of broad habitat types were also recorded at 25 m intervals along the transect. Means and standard deviations of density and percent habitat cover were calculated for each site based on the two 100 m transects.

Constrained Analysis of Principal Coordinates (CAP) was used to explore how sites were discriminated from one another spatially, based on the composition of benthic habitats (Anderson and Willis 2003). As year was shown to have no effect on the composition of habitats, the habitat information for a single year (2008) was used in the analysis as the greatest number of sites was sampled in that year. The mean densities of *L. lambis* in 2008 were depicted using a bubble plot. The habitat variables that contributed the greatest to the discrimination of sites in the CAP analysis were overlaid on the plot to explore the relationship between the composition of benthic habitats and the abundance and distribution of *L. lambis*.

4.2.2 Historical comparisons of *L. lambis* abundance

While other studies have quantified the abundance of *L. lambis* as part of general benthic surveys (eg. Hender *et al.* 2001) in this study historical comparison was made between Lincoln-Smith (1993) and the DoF surveys (2007-2011) as the studies used comparable survey techniques and sites. Comparisons were made using data from three of the Lincoln-Smith (1993) sites (1, 2 and 3) with each site containing three replicates so the mean density of the replicates was used.

The relationship between the means and standard deviations for the densities of *L. lambis* showed that, prior to analysis of variance (ANOVA), densities should be log₁₀ (n+1) transformed (Clarke and Gorley 2001). A two-way ANOVA was used to test for the effect of location and sites sampled by Lincoln-Smith *et al.* (1993) as well as sampling year. Multiple comparisons of means were carried out using Tukey HSD posthoc tests.

4.2.3 Reproductive biology of *L. lambis*

In 2008, DoF commenced a collection program in conjunction with the Cocos (Keeling) Islands Youth Council. Monthly samples of approximately ten *L. lambis* were collected from two areas in the lagoon. One collection area was located on the eastern side of the lagoon close to Home Island and the other area was in the south eastern area of the lagoon. At each site, *L. lambis* were collected randomly in respect to size, shell thickness and sex to ensure samples were representative of the stock. Monthly samples were obtained between April 2008 and March 2009, with the exception of August and November 2008 where samples were not collected due to bad weather.

The collected *L. lambis* were preserved in 10 % buffered formalin and stored in 70% ethanol until dissected. To aid fixation of gonad material, two holes were drilled into the shell towards the posterior end. The majority of epiphytic growth was removed from the shell and the whole weight (shell and tissue) of each animal was recorded to the nearest 0.1 g. Prior to dissection, a number of shell measurements were recorded (to the nearest 0.1 mm), including shell length (SL; base of siphonal canal to tip of the shell spire) and siphonal lip thickness (SLT; taken from the centre edge of the stromboid arch, 2 – 4 mm in from the edge).

To investigate the relationship between shell development and sexual maturity, the development of the flared shell and shell spines (i.e. open vs closed) was also recorded. In addition, as *L. lambis* has been described as having sexually dimorphic shell morphologies, a record was also kept of the length and orientation of shell spines, i.e. pointing dorsally or curved posteriorly.

Animals were removed from the shell using a vice and diamond saw, and whole tissue weight was recorded to the nearest 0.1 g. Each animal was assigned a sex, determined from the presence of verge (males) or egg groove (females), and maturity-stage was recorded based on the macroscopic appearance of gonads. Maturity stage was based on gonad colour, appearance (i.e. granular) and the proportion contributed by gonadal material to total posterior tissue.

Testes and ovaries were dissected and weighed to the nearest 0.01 g. Histological preparations were made of a sub-sample of gonads from each month to support macroscopic observations. Tissue from the mid-region of the gonad, which had already been fixed in 10 % formalin, was dehydrated in an ascending series of ethanol concentrations, embedded in paraffin wax and cut transversely into 6 µm sections. To aid in differentiating cell types, both Mallory's trichrome and hematoxylin and eosin were used to stain sections from each individual.

The gonadosomatic index (GSI) of each individual was calculated based on the equation:

$$\text{GSI} = \text{W1}/(\text{W2}-\text{W1}) \times 100$$

where W1 = wet weight of the gonad and W2 = wet tissue weight of whole animal.

The few animals with very small gonads, which were assumed to be immature, were not included in the calculation of mean monthly GSI values. A combination of the trends exhibited by the mean monthly GSIs of males and females and the monthly proportions of each maturity stage were then used to determine the time of peak spawning and the duration of the spawning period of *L. lambis*.

Water temperature data for a number of sites around the Cocos (Keeling) Islands are also collected by DoF as part of a long-term coral monitoring project. Data collected at a site in the south of the lagoon are presented here, as this site is closest to where *L. lambis* were collected and water temperatures at this site presumably reflect those over much of the southern shallow area (<10m) of the lagoon. Water temperatures were logged hourly with a HOBO Pendant temperature/light data logger (Onset Computer Corporation) and daily minimum and maximum values were used to calculate mean monthly minima and maxima.

4.3 Results

4.3.1 Ecology, distribution and abundance of *L. lambis*

The data from the 41 long-term monitoring sites (2008 – 2011) show that the distribution and abundance of *L. lambis* at Cocos (Keeling) Islands can vary greatly both within and between sites, as well as over time (Figure 3.2a and 3.2b). There is no trend in *L.lambis* densities observed between 2008 to 2011 with some sites showing small variations (see Figure 3.2a, i.e. GG10, GG11) while others are more dramatic (see Figure 4.2a; i.e. GG34, LS3C where in 2011 no *L. lambis* were recorded). The distribution of *L. lambis* is also not uniform throughout the lagoon. *L. lambis* densities in the south east of the lagoon (Figure 4.2b; Regions H, I and J) are comparatively low compared to the north eastern of the lagoon (Figure 4.2a; Regions B to F). The exception to this observation is Region A (Figure 4.2a) which is closest to Home

Island where the majority of fishers reside.

Using only the 2008 survey data *L. lambis* was recorded at 51 of the 67 sites surveyed, with densities ranging from 0 to 109 individuals per 200m². The mean density of *L. lambis*, for the 67 sites surveyed in 2008, was 13.5 individuals \pm 2.9 per 200m² (Figure 4.1c). It is important to note that this is not an estimate of total standing stock merely a mean density for the specific sites/areas surveyed. As reported by Bellchambers *et al.* (2011) *L. lambis* densities are driven by habitat associations. For example, sites with moderate to high levels of hard macroalgae (*Acanthopora* sp.), macroalgae and/or submassive corals have high densities of *L. lambis* while sites dominated by seagrass, filamentous algae, branching coral or sand have low densities of *L. lambis*. Therefore, without detailed habitat maps of the Cocos (Keeling) Islands estimates of standing stock are inaccurate.

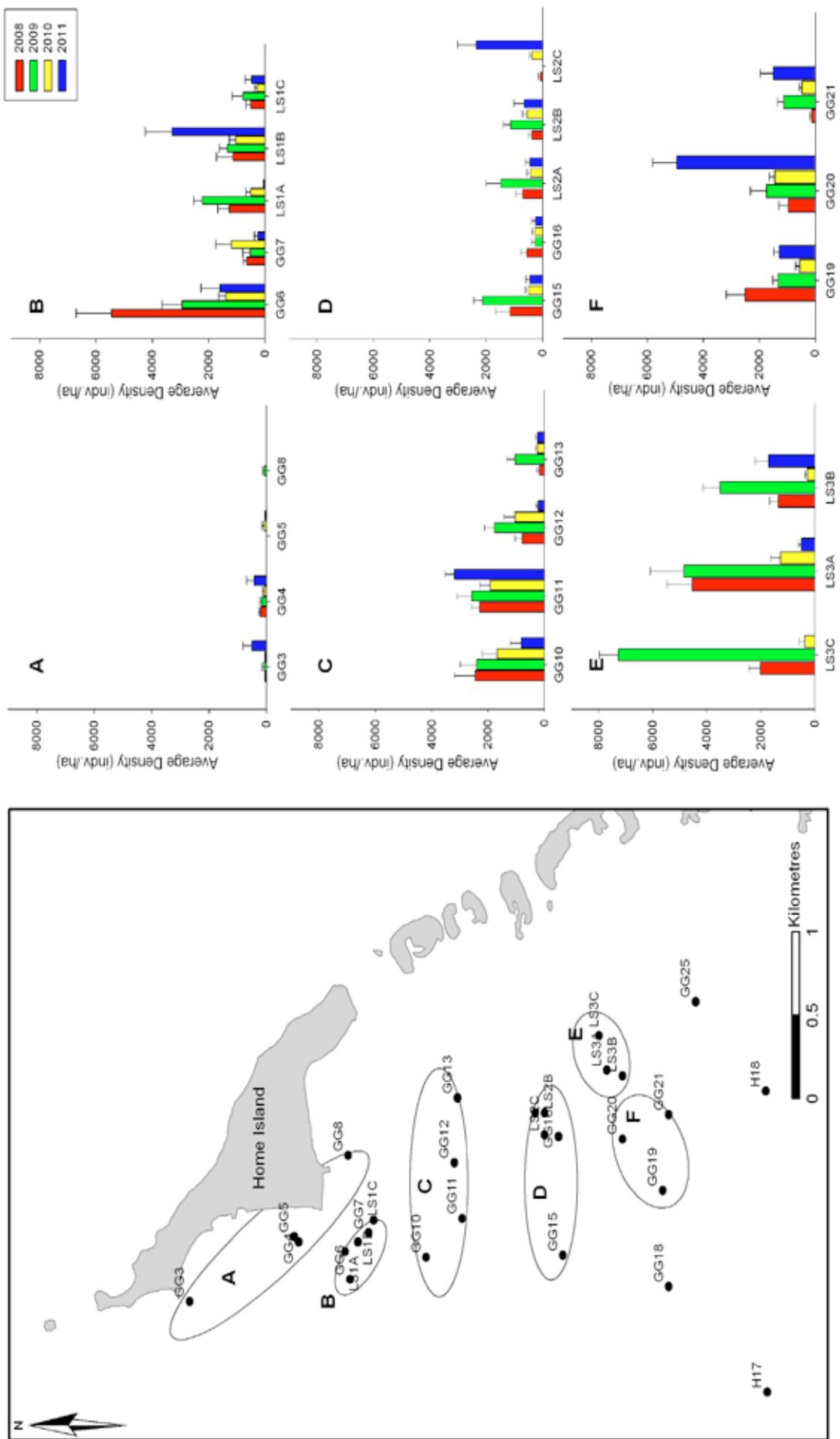


Figure 4.2a. Sites sampled in six regions (A – F) of the lagoon and the average densities of *Lambis lambis* (individuals ha⁻¹) recorded between 2007 and 2011.

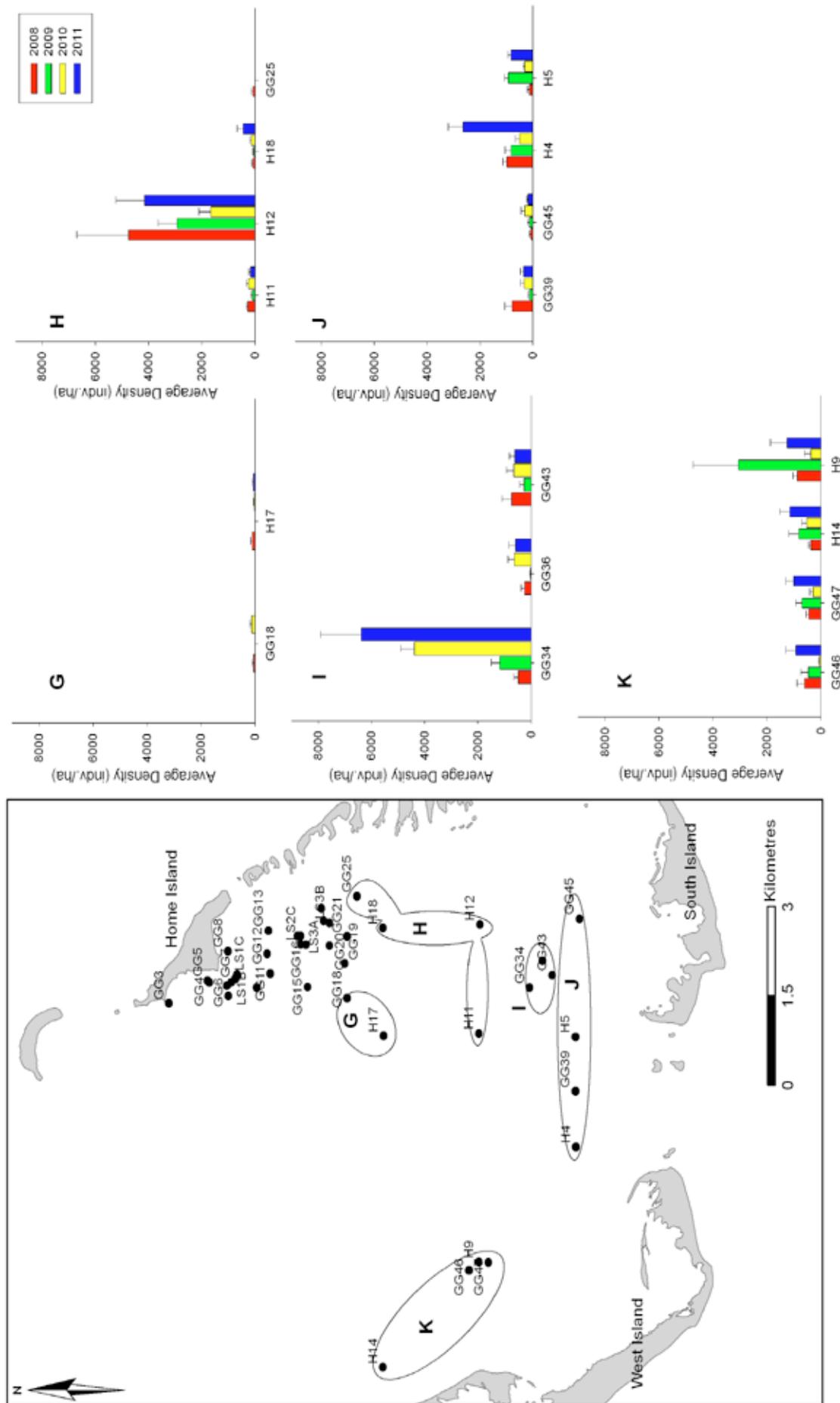


Figure 4.2b. Sites sampled in an additional six regions (G – L) of the lagoon and the average densities of *Lambis lambis* (individuals ha⁻¹) recorded between 2007 and 2011.

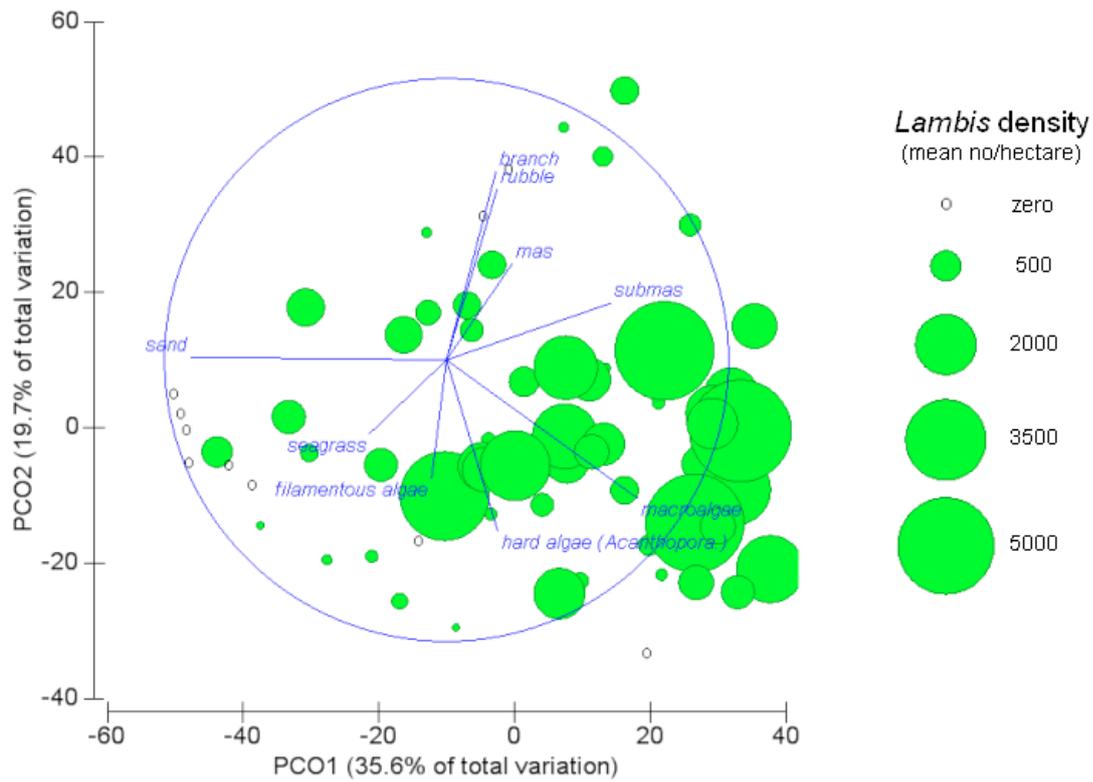


Figure 4.3. Principal coordinates ordination of benthic habitat at 2008 sites. Mean densities of *Lambis lambis* (individuals ha⁻¹) at each site are depicted by the relative size of green circles. Sites where *L. lambis* was absent are denoted by zeros.

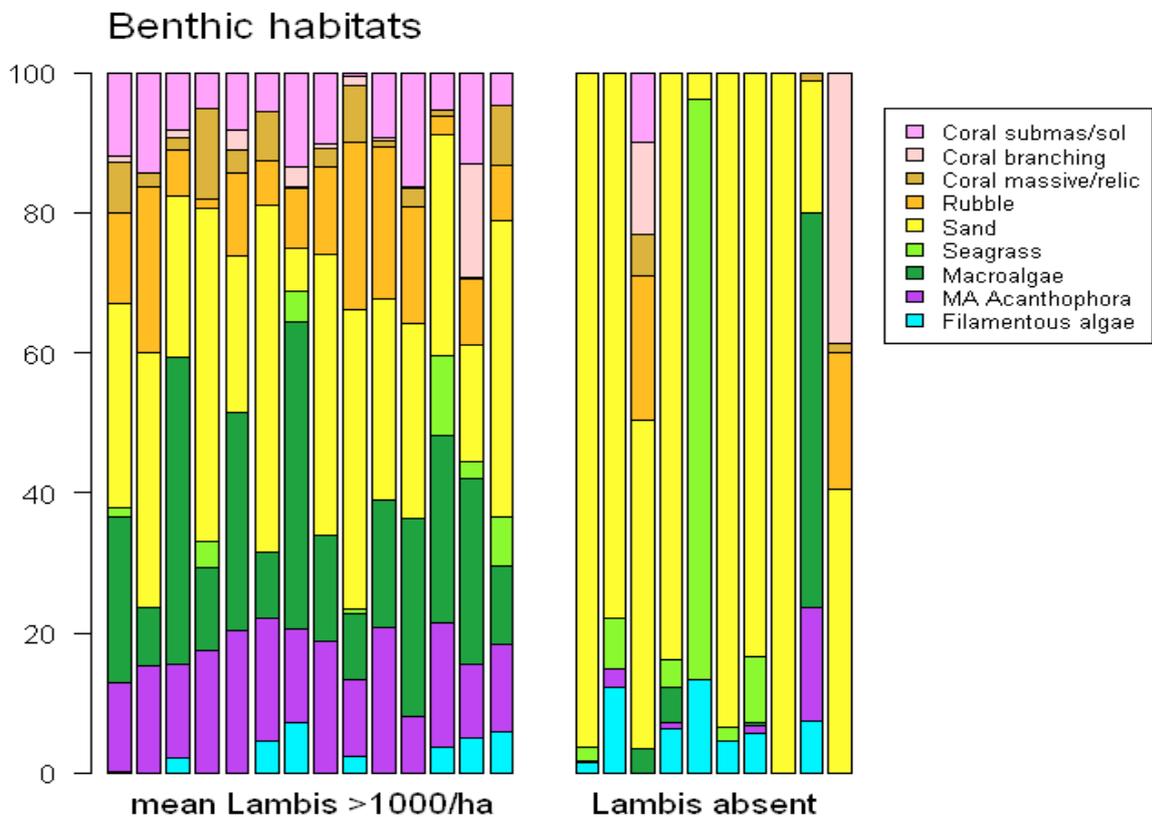


Figure 4.4. Mean percentage contribution of benthic habitats occurring at sites with high densities of *Lambis lambis* in 2008 (> 1000 individuals ha⁻¹) and sites where *L. lambis* was absent.

4.3.2 Historical comparison of *L. lambis* abundance at Cocos (Keeling) Islands

Surveys conducted at three of the locations surveyed by Lincoln-Smith *et al.* (1993) (actual survey was conducted in November 1992) and repeated annually by DoF (2007 – 2011) show a decline in density of *L. lambis*. ANOVA indicated that there was a significant difference in the density of *L. lambis* both between sites ($p < 0.001$) and between years ($p < 0.001$). Tukey HSD posthoc test indicated that location three was significantly different from location one and two ($p < 0.001$) and that 1992 was significantly different from all later years ($p < 0.001$). This illustrates that there has been a significant decrease in the density of *L. lambis* at these sites over the past 15 years (Figure 4.5).

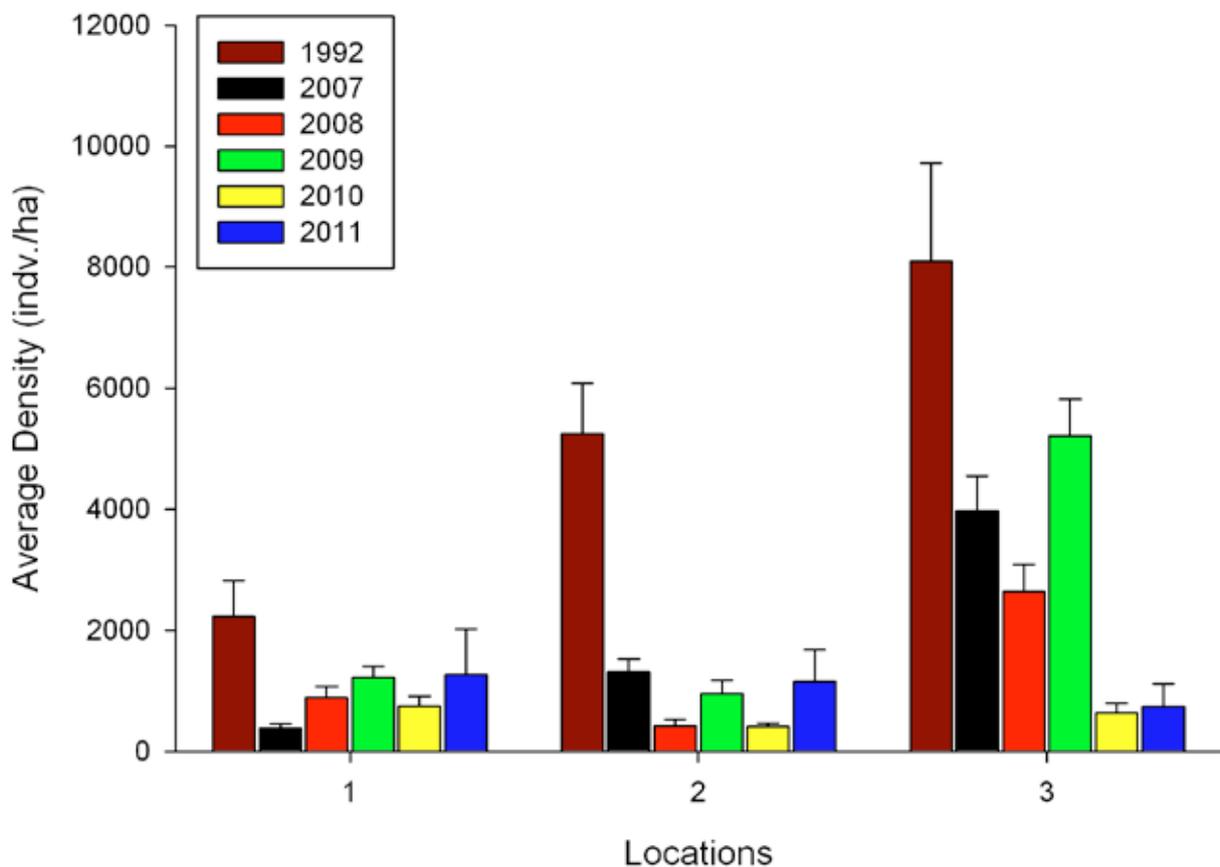


Figure 4.5. Comparison of densities (individuals ha^{-1}) of *L. lambis* at three locations sampled by Lincoln-Smith *et al.* (1993) (survey conducted in November 1992) and DoF surveys (2007 – 2011).

4.3.3 Reproductive biology

4.3.3.1 Biological data

Observations made of the development of *L. lambis* at the Cocos (Keeling) Islands have demonstrated that the smaller and presumably younger individuals that lack a flared lip and well developed spines (Figure 4.6) are cryptic and are not typically encountered on the surface of the benthos. As a result, the catch of local fishers is typically comprised of larger individuals that have a fully formed shell with a flared lip and well developed spines.

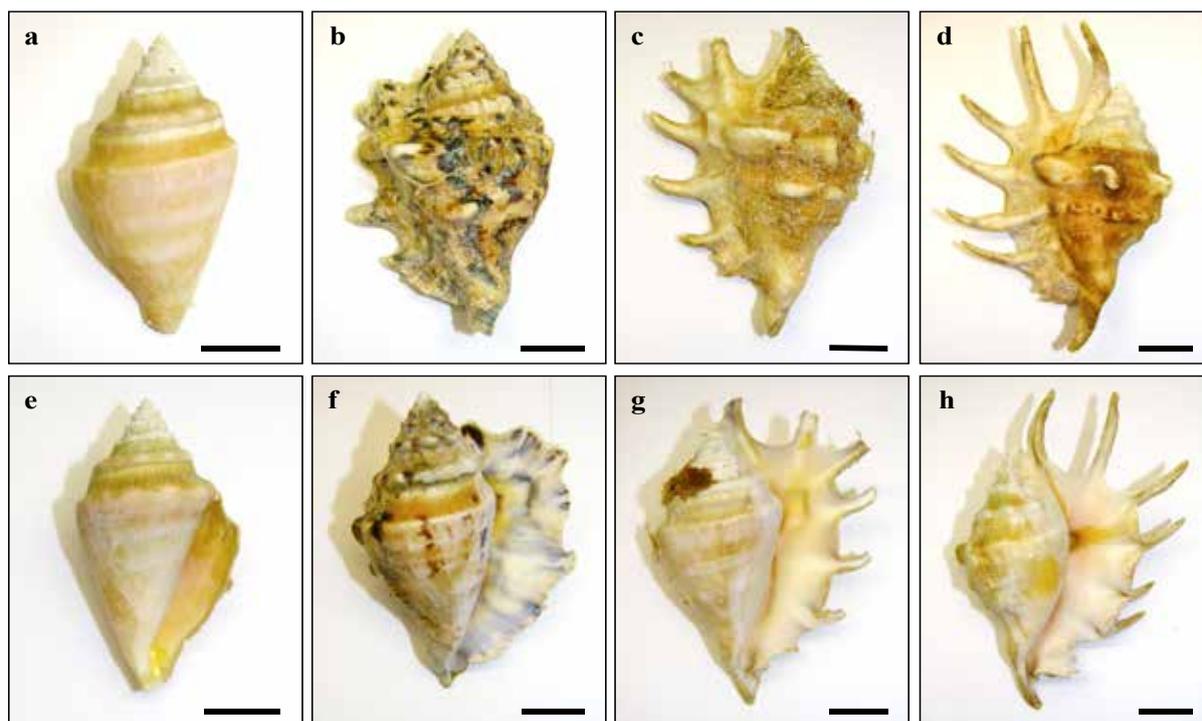


Figure 4.6. Dorsal (a – d) and ventral (e – h) views of *L. lambis* illustrating the stages in development of the flared lip as individuals reach maturity. (a, e) juvenile, with no flared lip; (b, f) development of flared lip; (c, g) further development and thickening of the flared lip, spines are still open; (d, h) adult shell with flared lip and closed spines. Scale bar = 20 mm.

A total of 190 *L. lambis* were collected for dissection and the majority of individuals had fully developed shells with partially or completely closed spines. The ratio of males to females for the total sample was close to parity at 1:0.9. An attempt was made to categorize each individual on the basis of shell spine orientation, i.e. pointing dorsally or curved posteriorly. The percentage of dorsally or posteriorly orientated spines in males was 38 % and 55 %, respectively, while females were 52 % and 42 %, respectively.

The average shell length, whole weight and tissue weight of female *L. lambis* was greater than that of males (Figure 4.7). The shell lengths of males and females ranged from 69.2 to 103.7 mm and 75.8 to 118.9 mm, respectively. The mean total weight of males and females was c. 125 and 182 g, respectively. Overall, the average individual whole weight was 152 g and the relationship between whole weight (WW) and tissue weight (TW) was:

$$TW = 0.042 \times WW^{1.207} \quad (R^2 = 0.74, n = 188).$$

In contrast to shell length, the distribution of shell lip thickness measurements recorded for males did not differ substantially to that of females with both sexes having a mode around 2.0 mm (Figure 4.7d). While the distributions of shell length, whole weight and tissue weight approached normality, the lip thickness distribution was strongly skewed to the left with small numbers of individuals having lip thickness measurements in excess of 6.0 mm. There was no relationship between shell length and lip thickness for either males or females.

The size and condition of the gonads of *L. lambis* varied over the sampling period. As the gonads of individuals developed, gonadal material contributed a greater percentage of the total posterior tissue, compared to gastrointestinal tissue (Figure 4.8). At an advanced stage of maturity, the

gonads of female *L. lambis* tended to fill the entire posterior shell space (Figure 4.8b). A small number of individuals had very small gonads that could not easily be dissected and weighed. These individuals were assumed to be immature or at a very early stage of development and were not included in the calculation of mean monthly GSI values. Each of these individuals had open spines and a thin (< 1.0 mm) shell lip thickness.

The mean monthly GSIs of female *L. lambis* reached a minimum of 6.0 in May and remained at a similar level before increasing to 12.9 in September. Values peaked in December at 13.9 and then progressively declined (Figure 4.9a). GSI values for males were considerably lower than those for females and increased progressively throughout the sampling period reaching a peak of 5.3 in February (Figure 4.9a). The trends observed in the GSIs for females and males followed similar patterns in the two sampling regions, i.e. Home and South Islands (Figure 4.9b, c).

Virtually all female *L. lambis* collected between April and July had resting ovaries, whereas the majority between September and January had developing ovaries (Figure 4.10). Female *L. lambis* with mature ovaries were observed between October and February, and those with spent or post-spawning ovaries were most common in March and persisted until May (Figure 4.10).

The histological sections made of resting gonads demonstrated that the majority of the tissue was non-gametogenic (Figure 4.11). In females, vitellogenic oocytes began to be observed in developing ovaries and were the dominant cell type in mature ovaries. Post-spawning ovaries were characterised by an unorganised structure, the presence of atretic oocytes and varying amounts of non-gametogenic tissue (Figure 4.11c). Tissue in the gonads of mature male *L. lambis* could be differentiated into regions of sperm storage and regions of production, where the sperm of multiple development types could be observed (Figure 4.11d).

3.3.2.2 Water temperatures

The mean monthly maximum water temperature declined to a minimum of 27.1 °C in August and remained low through to October before rising to a high of 29.7 °C in February (Figure 4.12). The mean monthly minimum water temperatures exhibited similar annual trends declining to a low of 25.9 °C in October and peaking at 28.4 °C in February (Figure 4.12).

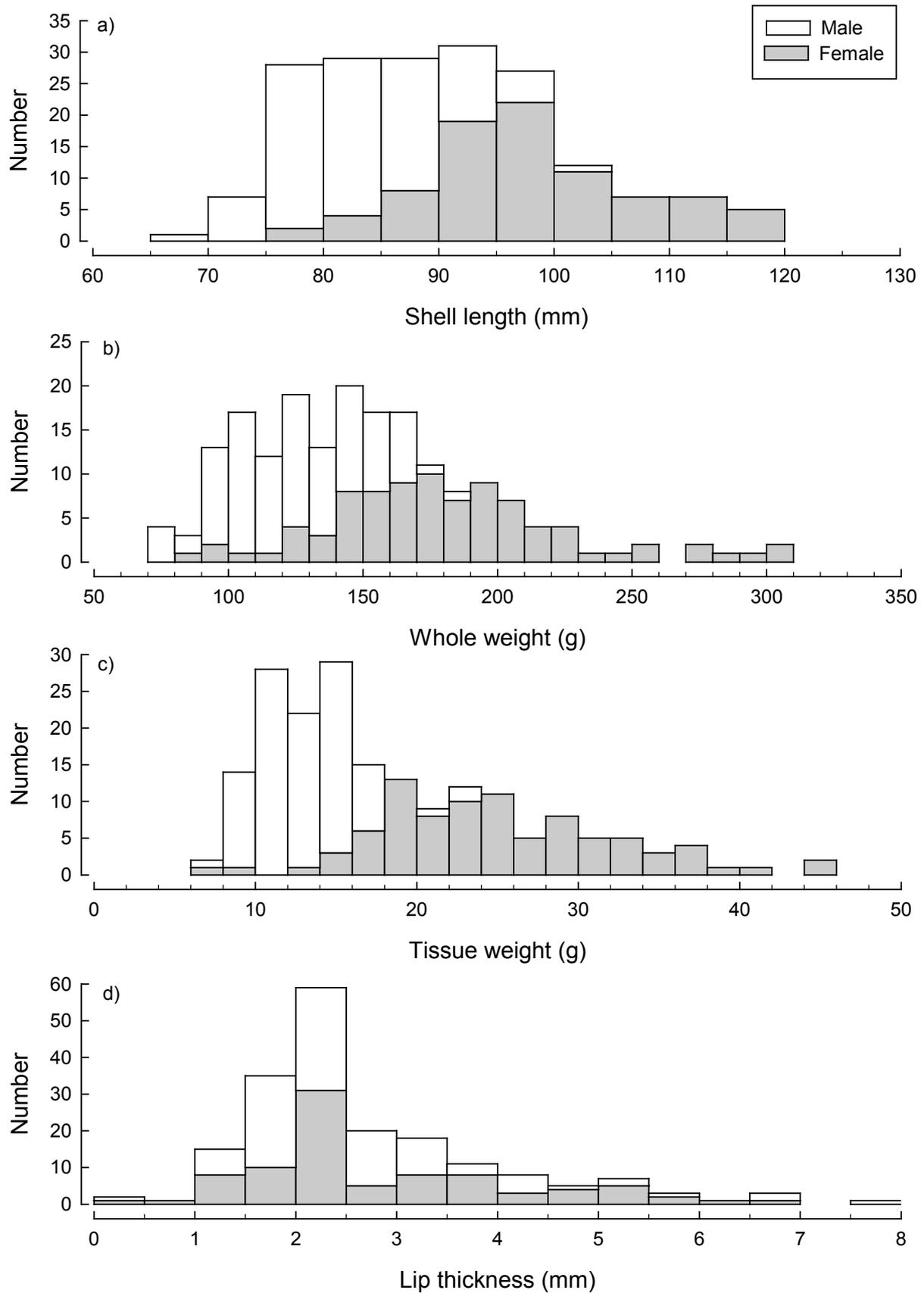


Figure 4.7. Distributions of (a) shell length, (b) whole weight, (c) tissue weight and (d) shell thickness of male (white) and female (grey) *L. lambis*.



Figure 4.8. Longitudinal sections through the shell of two female *L. lambis* showing the macroscopic appearance (a) developing (January) and (b) mature (December) ovaries. g: gastrointestinal tissue; o: ovarian tissue; s: shell space.

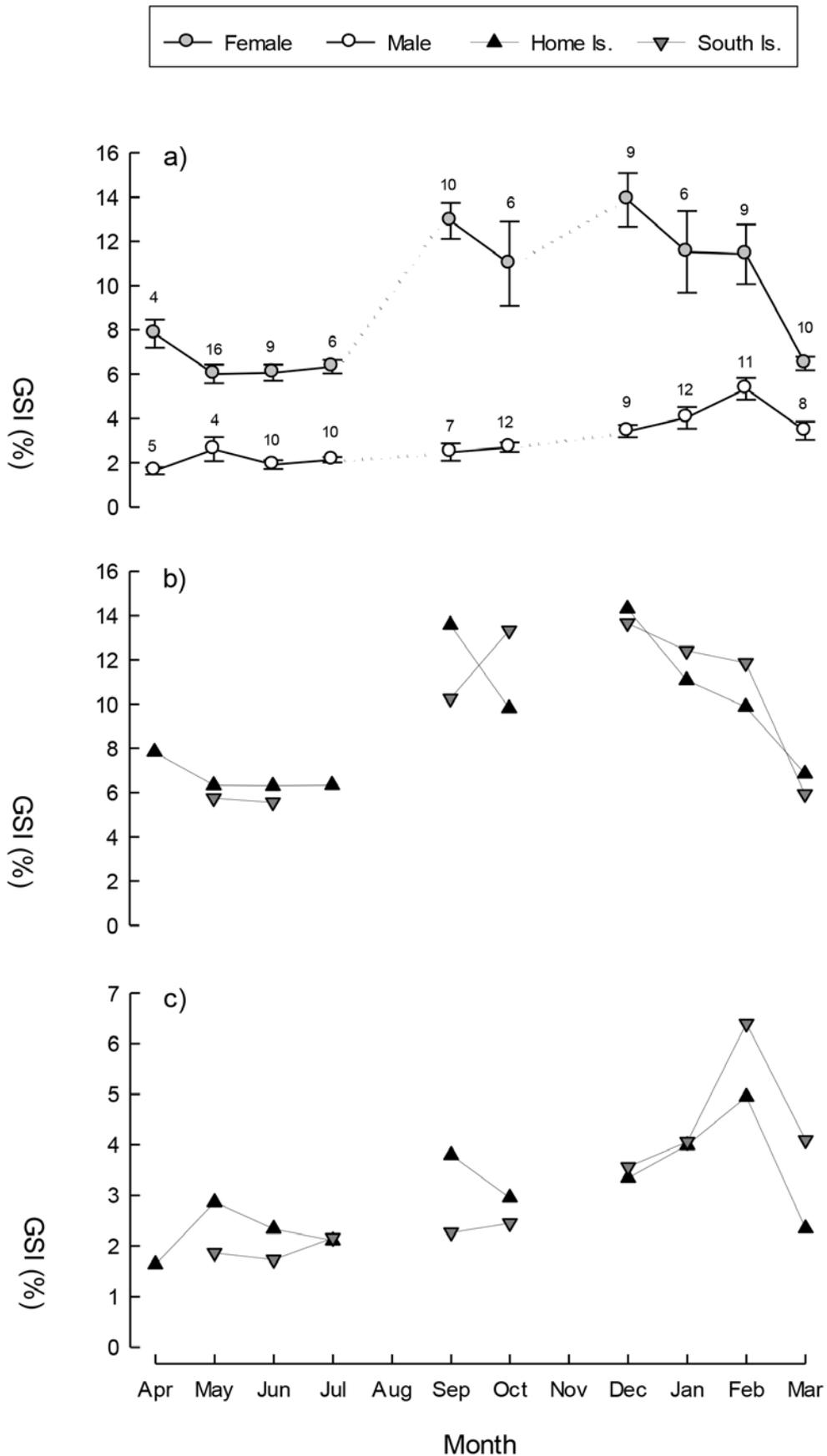


Figure 4.9. Mean monthly GSI values for *L. lambis* (a) males and females, areas combined; (b) females from Home Island and South Island and (c) males from the same locations. Numbers close to symbols indicate sample sizes for each location.

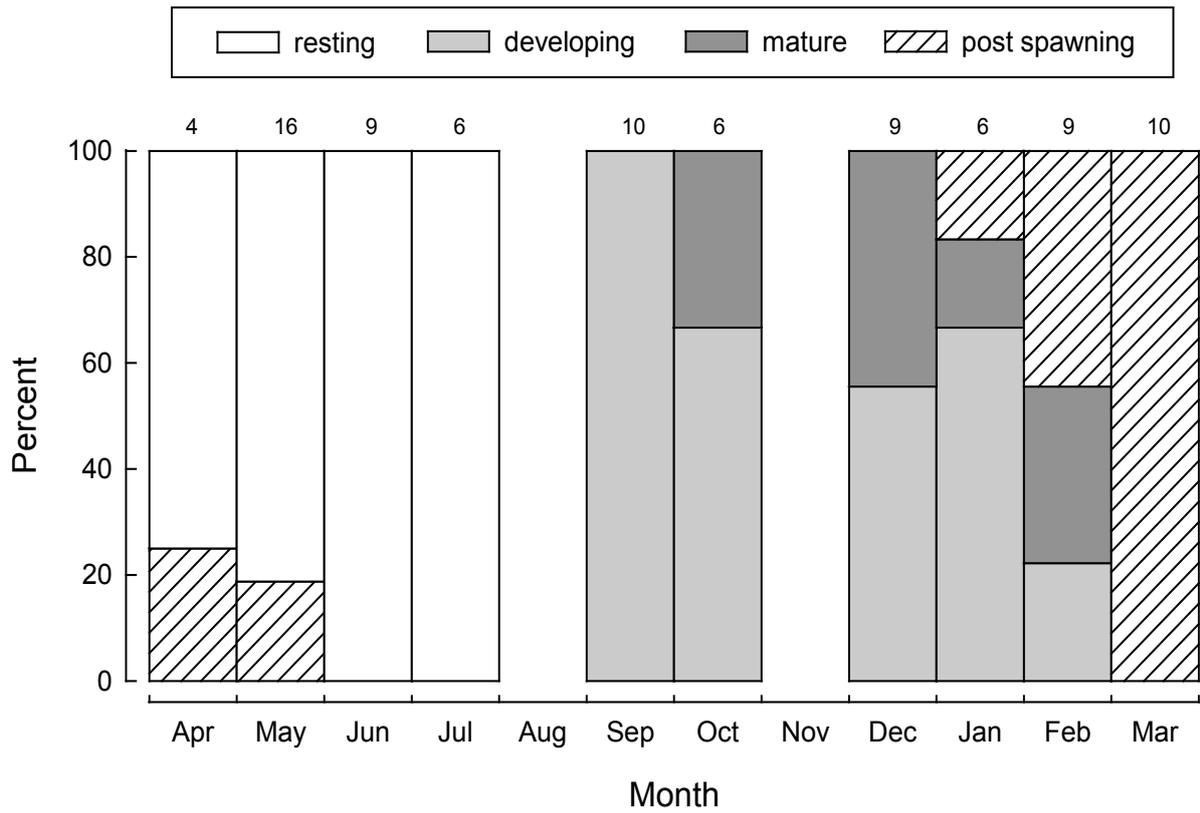


Figure 4.10. Monthly percentage frequencies of occurrence of sequential gonadal maturity stages in female *L. lambis*. Numbers above bars indicate sample sizes for each month.

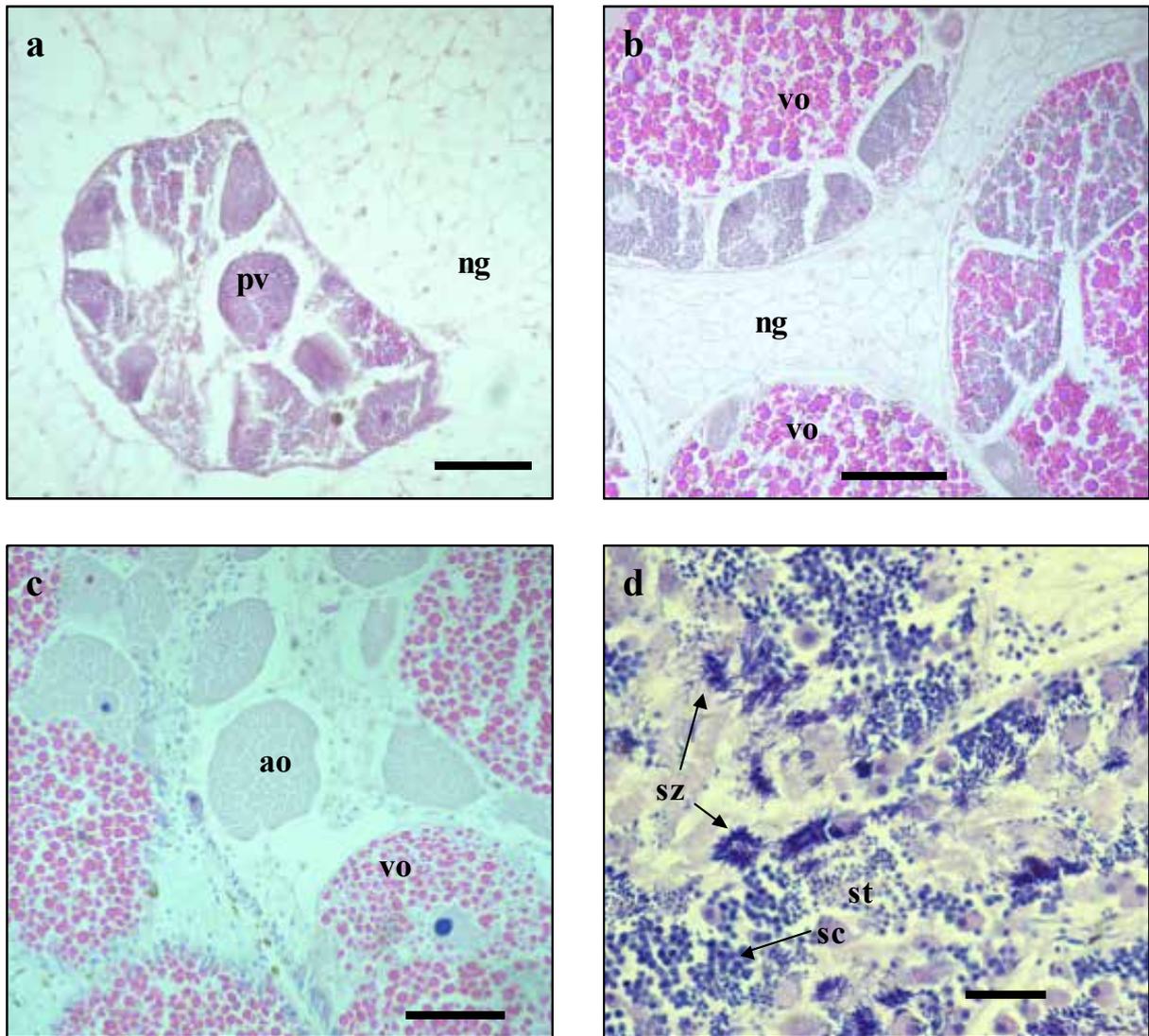


Figure 4.11. Histological sections showing selected stages in the development of the (a – c) ovary and (d) testis of *L. lambis*. (a) resting ovary showing predominance of non-gametogenic tissue, (b) developing ovary, (c) post-spawning ovary with unorganised structure, (d) testis showing sperm cells at a range of development stages. ng: non-gametogenic tissue; pv: pre-vitellogenic oocytes; vo: vitellogenic oocytes; ao: atretic oocytes; sc: spermatocytes; st: spermatids; sz: spermatozoa. Scale bars (a – c) 100 μ m, (d) 50 μ m.

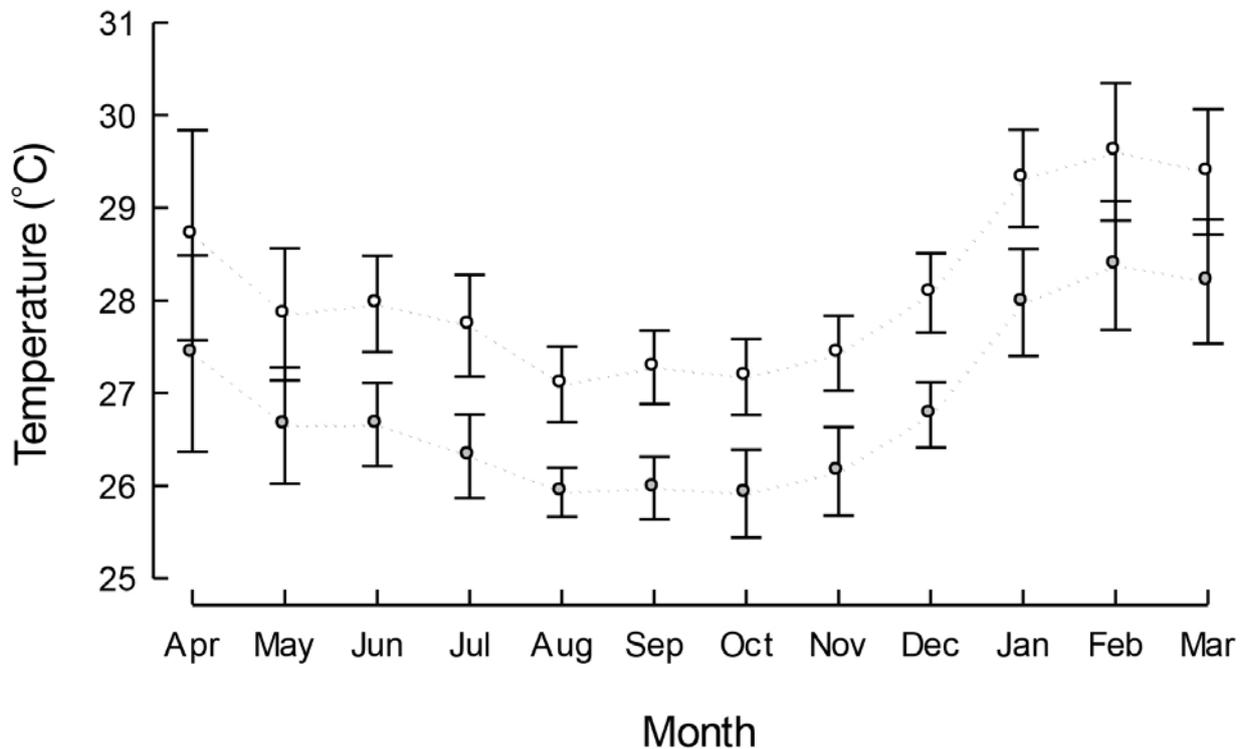


Figure 4.12. Mean (± 1 SD) monthly maximum (open circles) and minimum (shaded circles) temperatures recorded in the central southern region of the Cocos (Keeling) lagoon between April 2008 and March 2009.

3.4 Discussion

4.4.1 Ecology, distribution and abundance

Intensive fishing for *L. lambis* has been occurring at the Cocos (Keeling) Islands since at least the early 1980s (Berry 1989, Lincoln-Smith *et al.* 1993). From the earliest recorded survey (Lincoln-Smith 1993) average densities in the lagoon were approximately 4150 individuals ha^{-1} . *L. lambis* have decreased significantly since 1992 with the average density of *L. lambis* in 2007 and 2008 dropping 53 % and 77 % of 1992 levels respectively. Although surveys conducted since 2007 show some variation in densities the overall density of *L. lambis* from 2007 to 2011 is still significantly lower than surveys conducted by Lincoln-Smith *et al.* (1993).

From 2008 to 2011 a minimum of 41 sites, including three Lincoln-Smith *et al.* (1993) sites, have been surveyed annually. While some sites show a slight increase in *L. lambis* densities, the majority display a significant decrease. On average, there was a 50 % reduction in densities between 2007 and 2008. It is likely that overfishing has played a role in the decrease in density of *L. lambis* as benthic habitats have remained stable during this time. The survey sites were also relatively close together, with the majority of sites showing substantial decreases and only some showing slight increases, making it unlikely that reductions in densities were due to movement of individuals. It appears that a significant increase in density is followed by a significant decrease (i.e. GG12, GG13, LS3C, LS3A and LS3B) which may indicate selective harvesting by fishers. It is unlikely decreases in density were due to recruitment failure as the population is likely to be composed of individuals from several generations. However, the combined effects of heavy fishing pressure combined with sporadic or failed recruitment cannot be discounted.

The distribution of *L. lambis* was not uniform throughout the lagoon, but was concentrated in particular areas and showed strong habitat association (see Bellchambers *et al.* 2011). The highest densities of *L. lambis* were observed 1 – 2 km from Home Island in the eastern part of the lagoon where the habitat usually consisted of sand, macroalgae and hard algae. The lowest densities occurred in shallow, sand and seagrass dominated areas. The centre of the lagoon, which is mainly coral-dominated, is unsuitable *L. lambis* habitat. It appears that *L. lambis* are associated with macroalgae and hard macroalgae for shelter and/or for food. Most of the *L. lambis* were at least partially concealed by the macroalgae and could only be located by touch. *L. lambis* were also observed clustered around the edges of *Porites* bombies, where they were inconspicuous.

Large changes in *L. lambis* densities have been reported in the south east of the lagoon that appear to be related to habitat changes rather than overfishing. This area of the lagoon was historically popular for *L. lambis* fishing until around 2003 (Mohammad Chongkin, Parks Australia, *pers. comm.*). These sites were surveyed in 2008 – 2011 and densities were low (< 1500 individuals ha⁻¹). The region was previously covered in seagrass/macroalgae and is now mainly sand (Mohammad Chongkin, Parks Australia, *pers. comm.*) and intense fishing is no longer focused in this area. The reason for the change in habitat is unclear but could be related to El Niño events (temperature stress) and anoxia from coral spawn trapped in the lagoon. Widespread coral mortality was reported in 1869, 1962 and 1983 (Bunce 1988) and presumably other benthic communities were also affected. The gradual infilling of the lagoon with sand and silt (Bunce 1988) may also have contributed.

Large areas in the south west and west of the lagoon are not presently fished for *L. lambis*. As *L. lambis* are collected by wading, not free diving or SCUBA as with other strombids (Tewfik and Guzman 2003), it was thought that these might be areas that contain unexploited populations, particularly in deeper water. Deep-water refuges have been found in other populations, such as *Stombus luhuanus* on the south coast of Papua New Guinea where overfishing has occurred in shallow water, but deep-water populations have not been affected (Poiner and Catterall 1988). However, our results suggest that densities of *L. lambis* in the unfished areas were low due to unsuitable habitat, i.e. sand and seagrass, while deeper water areas (> 2 m) generally displayed low densities of *L. lambis* due to the lack of hard macroalgae. These findings correlate with previous descriptions of the marine habitats of the Cocos (Keeling) Islands that indicate the majority of macroalgae habitats are located in the shallow eastern portion of the lagoon (Williams 1994). However, while it appears that there are no refuge areas for adult *L. lambis* in the lagoon juveniles are believed to live embedded in the sand until sexually mature thus only the adult stage is harvested (Hender *et al.* 2001).

The ecological role of *L. lambis* in the Cocos (Keeling) lagoon is unknown, but is likely to be similar to *S. gigas*. Like *S. gigas*, *L. lambis* tends to inhabit sandy or rubble sea floors that support the growth of seagrass and algae (although *L. lambis* tends to be less associated with seagrass). *S. gigas* is a herbivore and a detritivore that feeds on dead or detrital seagrass, epiphytes and macroalgae and strongly affects the overall structure of the detrital community and the associated macrofaunal community (Stoner *et al.* 1995). A previous study suggested that *L. lambis* might feed on fine red algae rather than on seagrass (Younge as cited in Abbott 1961), therefore a negative correlation with seagrass appears reasonable. Preliminary observations of gastrointestinal contents of *L. lambis* individuals from the Cocos (Keeling) Islands during this study revealed large amounts of sediment that is consistent with a high detrital contribution to diet. Loss or substantial decreases of *L. lambis* could have significant effects on the benthic substrate, associated fauna and water quality at the Cocos (Keeling) Islands, especially since parts of the lagoon are not well flushed.

4.4.3 Reproductive biology

The majority of *L. lambis* collected were sexually mature adults, which suggests that the catch of local fishers is also likely to be comprised primarily of adults, as the samples were collected in the same way as fishing occurs. *L. lambis* with small gonads were assumed to be immature; these individuals also had very thin shell lips and spines that had not yet closed, which is considered to be the juvenile shell form of strombids. Many strombids undergo morphological changes with the onset of sexual maturity and remain cryptic up until this time (Poiner and Catterall 1988).

L. lambis at the Cocos (Keeling) Islands was found to be gonochoristic, i.e. the sexes are separate. No evidence was observed for the existence of abnormal or imposex females as has been noted for a number of other strombids, including *Strombus canarium* in Malaysia (Cob *et al.* 2008). While there was considerable overlap in the shell sizes and weights of each sex, the smallest adults were males and the largest were females. The tendency for adult females to be larger than males is typical of conch species (e.g. Reed 1993). Therefore, if fishers are selecting for size when collecting *L. lambis*, the sex ratio of the catch may be biased, i.e. females may be targeted if larger individuals are preferred. However, based on samples collected during this study, the sex ratio of the adult population at Cocos Keeling is close to parity.

Previous literature suggests that the spines of female *L. lambis* are normally dorsally orientated, whereas those of males tend to curve posteriorly (e.g. Beesely *et al.* 1998). Despite the sexual shell dimorphism exhibited by *L. lambis*, orientation of the spines alone was not a good indicator of sex, and an attempt to assign sex of individuals based on this characteristic was only successful in approximately 50 % of cases. Combining morphological traits, including shell size, may be more successful in assigning sex if genitalia are not visible.

As most strombids exhibit determinate growth, their shell ceases to increase in length once individuals reach a certain size or age or attain maturity (e.g. Poiner and Catterall 1988). The distribution of shell lengths recorded for *L. lambis* suggest that this may also be the case for *L. lambis*, which means that there is little relationship between the shell length and age of adults. Therefore, only shell length distributions of juveniles and sub-adults are likely to inform population demographics, if they can be sampled (Cob *et al.* 2009).

While shell length of *L. lambis* may not be a reliable indicator of age, it appears that there may be a relationship between shell lip thickness and age. However, determining the relationship between age and shell lip-thickness can be difficult. Previous studies have used recaptures of tagged individuals (Appeldoorn 1988, Poiner and Catterall 1988), and establishing such a relationship for *L. lambis* would require a dedicated tagging study over a number of years.

Other methods of aging gastropods rely on seasonal patterns of growth leaving evidence in the hard parts, such as opercula (e.g. Ilano *et al.* 2004, Miranda *et al.* 2008). A preliminary study of *L. lambis* suggests that these zones can be observed and that the number of zones varies among individuals (between 1 and greater than 10). Preliminary analysis suggests a positive relationship between number of opercular zones and the shell lip-thickness. Therefore, counts of opercular zones may provide a method of aging *L. lambis* if the periodicity of zone formation can be validated. In other gastropods, aging has been validated through analysis of marginal increments (e.g. Ilano *et al.* 2004, Miranda *et al.* 2008) or stable isotopes within the growth bands (Bigatti *et al.* 2007). Another issue that requires research before opercula growth bands can be used to age *L. lambis* is the possible loss and subsequent regrowth of the operculum, which may result in the age of some individuals being underestimated.

The presence of females of *L. lambis* with mature ovaries in each month between October and February implies that this species spawns between late-spring and late-summer. The mean monthly GSIs of females were greatest around December suggesting that peak spawning may occur over the summer period. However, male GSIs peak sharply in February and this may reflect a period close to peak mating. The discrepancy in timing of peak GSI values between males and females may suggest that *L. lambis* females spawn multiple batches of eggs over the season.

The conclusion that reproductive activity peaks toward late-summer is consistent with the fact actively spawning females were observed depositing egg masses at numerous shallow water sites during a field trip in March 2009. As no field trips have coincided with the summer months, egg deposition over this time can only be assumed. The egg masses observed in the field resembled those described previously for this species (Risbec 1932). Observation of *L. lambis* in captivity suggests that egg deposition follows mating by approximately two weeks and hatching of veligers occurs two to three days later (Hamel and Mercier 2006).

Although *L. lambis* appears to have a protracted spawning period, reproductive activity is clearly seasonal and peaks with the period of maximum water temperature within the lagoon in late summer. A similar seasonal pattern in reproductive activity has been demonstrated in many gastropods (e.g. Aranda *et al.* 2003).

While this study is based on small sample sizes, the consistency in the reproductive data between the two widely separated sites suggests the conclusions drawn are likely to be representative of the population as a whole. It also demonstrates that spawning is unlikely to be confined to a single small area of the lagoon. At this stage, the spatial extent and specific habitats involved in spawning are unknown. Similarly, the exact nature and extent of nursery habitats important for juvenile *L. lambis* are unknown.

4.5 Management Recommendations

Lambis lambis populations at the Cocos (Keeling) Islands have been significantly reduced by fishing. It appears that *L. lambis* have only been heavily fished from the lagoon in the last thirty years and that collections were not made before 1949 (Lincoln-Smith *et al.* 1989). *L. lambis* has been identified as one of the most vulnerable species to overfishing in the Cocos (Keeling) Islands (Berry 1989, Lincoln Smith *et al.* 1993). This report supports these concerns with large reductions in densities recorded over a 15-year period and significant reductions occurring over the last 5 years (period of DoF surveys).

A similar species, *S. gigas*, has been severely overfished and densities throughout the Caribbean are very low despite a range of management measures. *L. lambis* is vulnerable to overfishing because, like *S. gigas*, it lives in shallow water, is slow moving and is likely to have late maturation and slow growth. Importantly, the population of *L. lambis* at the Cocos (Keeling) Islands is isolated, meaning it is highly unlikely that there is any external recruitment from elsewhere. Therefore, management measures urgently need to be introduced to promote sustainable fishing and protect *L. lambis* stocks.

Island-specific recreational fishing rules are currently being developed for legislation, including a bag limit for *L. lambis*, however, at this stage there is no minimum size limit on *L. lambis* proposed. While it would appear that only mature *L. lambis* are harvested due to the cryptic nature of immature individuals, further research on the relationship between size at sexual

maturity and shell morphology is required to ensure that a sufficient number of individuals are reaching maturity and spawning before being captured by the recreational fishery.

Along with the introduction of bag and size limits, seasonal and/or spatial closures can be a potentially valuable fisheries management tool. Seasonal closures can protect spawning stock at high densities, provide shelter for older specimens that have a higher reproductive output or are more fecund than younger individuals and be an important source of larvae and new recruits to exploited areas (spillover effect). Seasonal closures for several months to protect individuals during peak reproductive periods are in place for most Caribbean countries (Theile 2001) and may be an effective management measure to prevent further decline in *L. lambis* stocks at Cocos (Keeling) Islands. However, the effectiveness of seasonal closures depends on spatial closures protecting critical spawning sites, nursery grounds and favoured *L. lambis* habitats. These areas need to be identified for *L. lambis* at the Cocos (Keeling) Islands before seasonal or spatial closures can be considered as part of any management strategy.

5.0 Abundance and distribution of giant clams

Evans, S.N., Bellchambers, L.M., and Webster, F.

5.1 Introduction

Giant clams are bivalve molluscs in the family Tridacnidae. There are currently ten recognized living species, eight species of the genus *Tridacna*: *Tridacna costata*, *T. crocea* [Lamarck 1819], *T. derasa* [Röding 1798], *T. gigas* [Linnaeus 1758], *T. maxima* [Röding 1798], *T. rosewateri* [Sirenko and Scarlato 1991], *T. squamosa* [Lamarck, 1819], and *T. teveroa* [Lucas, Ledua and Braley 1990]) and two of the genus *Hippopus* (*Hippopus hippopus* [Linnaeus 1758] and *H. porcellanus* [Rosewater 1982]). While the global distribution of giant clams varies between species, they generally occur throughout the Indo-Pacific region (Rosewater 1965), with *T. maxima* having the widest distribution stretching from East Africa and the Red Sea to Polynesia (Knop 1996).

Giant clams are filter feeders (Klumpp *et al.* 1992) although they also have photosynthetic dinoflagellate (*Symbiodinium microadriaticum*) commonly known as zooxanthellae, in their mantle tissue (Norton *et al.* 1992, Knop 1996). The zooxanthellae supply essential metabolic products such as phosphates and nitrates to giant clams through phototrophic pathways (Rosewater 1965, 1996). Giant clams are heavily reliant on their zooxanthellae and can obtain almost 100 % of their dietary requirements from this symbiotic relationship (Braley 1989, Munro 1992). Giant clams are restricted to oligotrophic or shallow (< 20 m) clear waters with adequate light for photosynthesis and are generally found on coral reefs (Yonge 1981, Blidberg *et al.* 1999). Previous surveys at various locations worldwide suggest that adults of most species of giant clams survive in a range of habitats in tropical waters (Alcala 1986, Braley 1987, 1989, Munro, 1988). However, a degree of selective exclusion occurs in the early life history stages where the presence of suitable substrate for attachment of the juvenile clam or spat is critical (Munro 1992).

Giant clams are protandrous hermaphrodites and become simultaneous hermaphrodites as they grow. This means they first reach sexual maturity as males and then later develop ovaries which function simultaneously with the testes, i.e. they produce both eggs and sperm. The timing of spawning appears to depend on location, with giant clams in the central tropics displaying no evidence of seasonal reproduction (Beckvar 1981, Munro and Gwyther 1981). At the northern and southern limits of distribution, seasonal spawning is evident with *T. gigas*, *T. crocea* and *H. hippopus* spawning in summer on the Great Barrier Reef (Braley 1984, Nash *et al.* 1988). Giant clams are broadcast spawners, and the normal spawning sequence is for sperm to be produced first, followed by egg production (Nash *et al.* 1988, Munro 1992). However, not all giant clams release eggs, as egg release in wild populations has been observed less frequently than sperm release (Braley 1984). Gamete release acts as a trigger for nearby giant clams' eggs to spawn which ensures the fertilisation of eggs and leads to clumping of individuals.

Growth rates after settlement are usually slow and vary amongst species, with most species able to reproduce at around 5 – 7 years of age (Kinch and Teitelbaum 2010). Due to their reproductive strategy it is essential that giant clam populations are maintained at relatively high densities to promote successful spawning and fertilisation of eggs or stocks will become unsustainable (Munro 1992, Kinch and Teitelbaum 2010).

Giant clams are the basis of important fisheries in many Indo-Pacific countries where they

are harvested for local consumption and to supply international markets (Rosewater 1965, Pearson 1977, Munro 1989). They are also harvested for the aquarium and ornamental trades (Othman *et al.* 2010). There are few reliable statistics on the total harvest of giant clams in the South Pacific Region and while subsistence harvests are relatively low, giant clam stocks have declined dramatically in many countries as a result of commercial exploitation for their meat and shells (Munro 1992, Wells 1997). Overharvesting has greatly reduced wild stocks and local extinctions of several species have been reported in the Philippines, Indonesia, Micronesia (Lucas 1994), Malaysia (Tan and Zulfigar 2003), and Singapore (Guest *et al.* 2008). Giant clam stocks worldwide have also been reduced due to natural and anthropogenic changes in the environment such as bleaching (Addessi 2001, Vinoth *et al.* 2012) habitat loss, increased nutrients (Hoegh-Guldberg 1997) and pollution (Elfwing *et al.* 2001).

In response to declining stocks worldwide each species of giant clam has been assigned a conservation status under the 2007 World Conservation Union's Red List of Threatened Species (<http://www.iucnredlist.org>). All tridacnids are also CITES listed, meaning that a permit is required for international trade of live specimens, meat or shells (Othman *et al.* 2010).

Three species of giant clam have been reported to occur at Cocos (Keeling) Islands, *T. derasa*, *T. gigas* and *T. maxima* (Berry 1989, Wells 1994). However, *T. gigas* may be locally extinct as several surveys have not found any live individuals. Dead valves have been found on the beaches, however, suggesting the species did occur in relative abundance (Berry 1989, Wells 1994). To date no comprehensive surveys have been conducted to specifically document the distribution and abundance of giant clams at the islands. Previous information on the distribution and abundance of giant clams has been collected as a part of general benthic (Hender *et al.* 2001) or taxonomic (Wells 1994) surveys.

There is currently no commercial fishery for giant clams at the Cocos (Keeling) Islands. The clams are harvested for local consumption primarily by the Cocos Malay population and are also collected as broodstock for a land-based aquaculture facility. DoF has undertaken comprehensive community consultations on the development of a set of Island-specific recreational fishing guidelines for the Cocos (Keeling) Islands. These new arrangements include a recreational bag limit of 10 *Tridacna* spp. per person per day, with no collection of *T. gigas* permitted (Department of Fisheries 2010).

Although giant clams are a popular food source for the local Cocos Malay population, and anecdotal information suggests that giant clams may have been collected for some time for use as food and ballast by passing ships, there are no current or historical catch records. Therefore, DoF has no baseline information to assess trends in historical and current levels of exploitation or giant clam distribution, abundance and size frequency. Given the status of giant clams worldwide and their inherent vulnerability to overexploitation DoF conducted a comprehensive survey in November 2011 to document the distribution, abundance and size frequency of giant clam populations at the islands.

5.1.1 Objectives

1. Collect baseline data on the distribution and abundance of the giant clam population;
2. Collect data on the size structure of the giant clam population; and
3. Provide management advice to ensure the sustainability of giant clams resources.

5.2 Methods

5.2.1 Site selection

Surveys were undertaken at the southern atoll of the Cocos (Keeling) Islands between November 21st and December 4th 2011 using a stratified random sampling design (Quinn and Keough 2002). Strata were based on the habitat classifications of Williams (1994), with similar habitats inside the lagoon merged, i.e. blue hole mosaics (Table 5.1). Initially, 54 survey sites were identified based on what was considered to be an achievable sampling program. The sites were randomly distributed within the 18 habitats (see Table 5.1) using ArcGIS's[©] ArcMap[©] random points generator. Each habitat was sampled at least once with the total number of sites in each habitat weighted by total habitat area. During the survey an additional 23 sites were also surveyed at randomly selected sites in areas that had 'suitable' giant clam habitat. These additional sites were selected using a combination of previously surveyed sites (i.e. Hender *et al.* 2001) and additional sites selected by ArcMap[©] random point generator. Suitable habitats were comprised of hard substrate, such as coral outcrop, coral terraces and/or coral flats as defined by Williams (1994). Unsuitable areas contained predominantly soft or transitional substrate such as seagrass flats and prograding sand sheets (see Table 5.1). Therefore, a total of 77 randomly selected sites were surveyed (Figure 5.1).

In addition, one targeted site was surveyed in the area colloquially known as 'The Rip' (see Figure 5.1), which is a protected (no-take) area. This site is discussed separately throughout the report and not included in any analyses as it was not randomly selected and is located in a no-take area.

Table 5.1. Marine habitats and map units of the Cocos (Keeling) Islands as classified by 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 | 21 |
| 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 (<i>Thalassia hemprichii</i>) | 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 (<i>Thalassodendron ciliatum</i>), Seagrass Bed (<i>Syringodium isoetifolium</i>) and Seagrass Flat (<i>Thalassia hemprichii</i>) | 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 |
| | Mixed Seagrass and Algal Bed | V | Suitable* | Strata 4 | 0.46 | 1 |

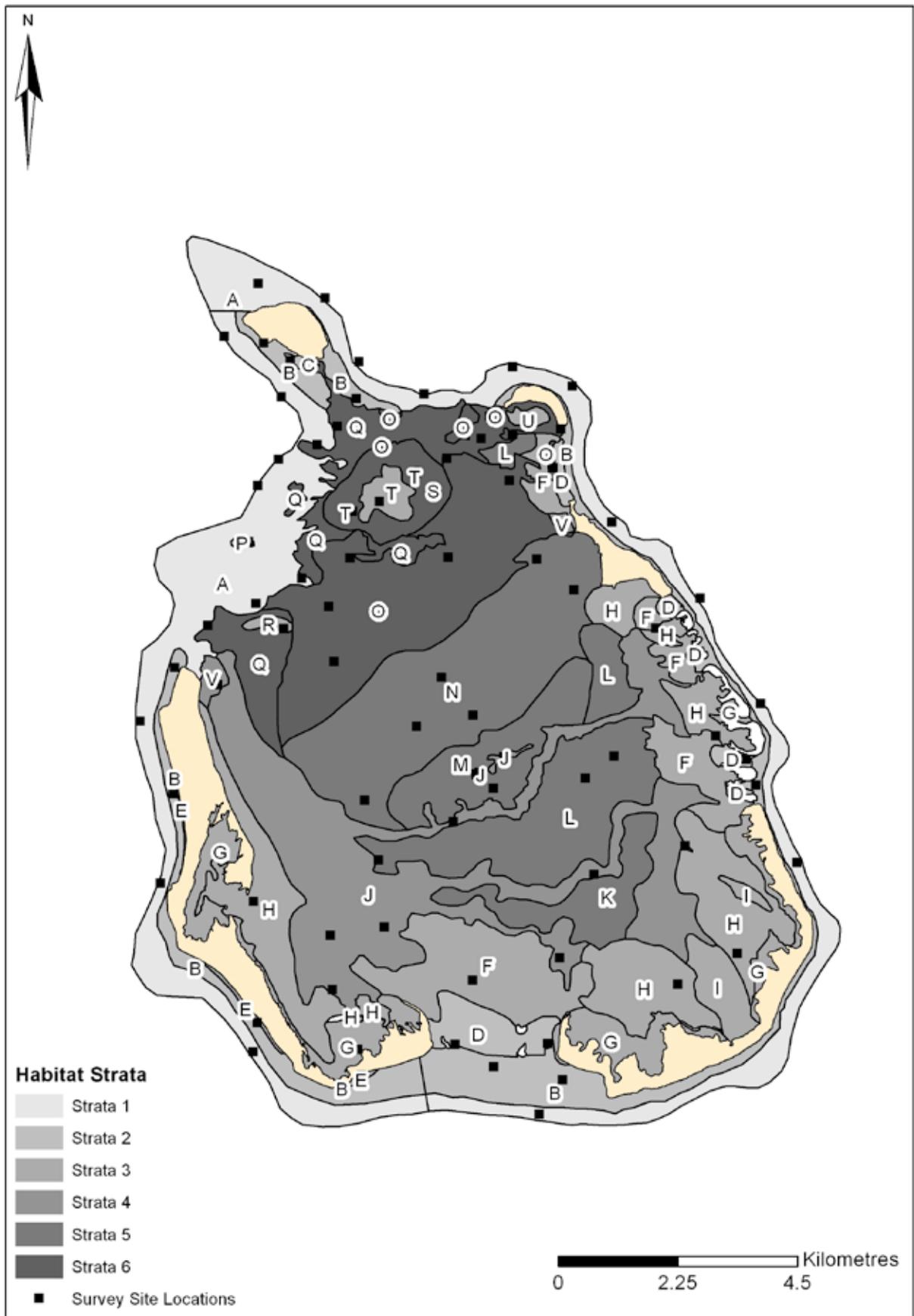


Figure 5.1. Giant clam survey sites at the Cocos (Keeling) Islands with Williams (1994) map units and habitat strata (Table 5.1).

5.2.2 Survey method

Surveys were undertaken using SCUBA, snorkelling or reef walks depending on the water depth at each site which ranged from intertidal to 15 m. At each site two sets of paired transects were surveyed with each set of transects spaced at least 30 m apart. Each paired set was surveyed by two divers conducting adjacent 50 x 2 m belt transects (spaced approximately 5 m apart) swimming on a predetermined random bearing. Giant clam abundance and a visual assessment of the percentage cover of broad habitat types were recorded at 10 m intervals. Giant clams were identified to species level based on shell morphology, as outlined in Knop (1996) instead of mantle colour which can be difficult to discern in-situ (Dr Shirley Slack-Smith, Western Australian Museum, *pers comm.*). Individuals that could not be conclusively identified due to the shell being embedded in the substrate, eroded and/or with undeveloped features were classified as “unknown”. The first 20 giant clams on each divers’ belt transect were measured to the nearest mm, from apex to apex.

5.2.3 Historical records of giant clam distribution and abundance

Prior to the current study there have been no comprehensive surveys to quantify the distribution and abundance of giant clams at the Cocos (Keeling) Islands. However, two previous studies recorded giant clams as a component of general benthic surveys (Hender *et al.* 2001, Hobbs 2008 – unpublished data) and a targeted pilot survey was undertaken by DoF in 2010. The studies vary in the techniques and intensity of sampling, as well as the habitats surveyed. Due to biases caused by the different survey approaches data from previous surveys can only provide an indication of abundance and distribution, not a direct quantitative comparison.

5.2.4 Statistical analysis

5.2.4.1 Average densities and standing stock of giant clams

An unpaired t-test was used to test for differences in average densities of giant clams between the paired belt transects. As no significant differences were observed the densities of the paired transects (within a set) were combined and analysed as an average per 100 m². As the two sets of transects conducted per site were spaced at least 30 m apart they were used as replicates doubling the number of transects surveyed from 77 to 154. Densities for these 154 transects were then calculated to report the average density per habitat unit. Densities for the protected area (‘The Rip’) were also calculated and are reported separately.

Similar habitat units were merged to form six strata ensuring each stratum contained a minimum of 20 transects (each site has 2 transects) (Table 5.1). The abundance of giant clams in each stratum was estimated based on the methods outlined in Hesp *et al.* (2008). Densities recorded in transects 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, \bar{x}^* were drawn from binomial and normal distributions, respectively. The values of \bar{x}^* were drawn using the equation:

$$\bar{x}^* = \bar{x} + SE \cdot r \quad 1$$

Where SE is the standard error of the non-zero values, and r is a random normal variate.

Each value of \bar{x}^* was then back-transformed and corrected for bias, using the equation:

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

Where:

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

E_{corr} is the bias-corrected estimate following 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 Cocos (Keeling) Islands. All calculations were undertaken in Microsoft Excel.

5.2.4.2 Size frequency distributions of giant clams

To assess size frequency distribution of the giant clam population data from each site was pooled and grouped into 10 mm categories. The size at sexual maturity of *T. maxima* was used as a proxy for all giant clams due to the dominance of this species at the Cocos (Keeling) Islands. The size *T. maxima* reach sexual maturity as males is approximately 60 mm with 50 % of both males and females sexually mature at 100 mm. While there are a number of published values of the size at which 100 % of the population reach sexual maturity (Green and Craig 1999, Chambers 2007, Apte *et al.* 2010) for the purpose of this report 150 mm was chosen as this value is the average of the published estimates.

5.2.4.3 Estimates of current harvest of giant clams

As there is no data on the recreational catch of giant clams at the Cocos (Keeling) Islands, an estimate was calculated based on several assumptions relating to fishers who potentially target giant clams. These estimates are provided in the absence of historical catch data to provide an indication of current fishing pressure and are in no way a substitute for actual catch data from catch returns, logbooks or recreational surveys.

As giant clams are harvested for local consumption primarily by the Cocos Malay population estimates of recreational catch used only this portion of the population as potential fishers. The 2011 census estimated the total Cocos Malay population as 417 people (Australian Bureau of Statistics 2011). Assuming that of the Cocos Malay population men who are aged between 15 and 54 years are the active fishers then the total potential active fishers is approximately 108 people. Several estimates of annual recreational giant clam catch were calculated assuming fishing occurred once or twice a week with five, 10 or 20 giant clams collected per trip.

5.2.4.4 Estimates of sustainable harvest of giant clams

The annual sustainable harvest of mature giant clams at the Cocos (Keeling) Islands was calculated using the formula:

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

Where:

P_{opt} is quantity of the mature population and

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

The fishing exploitation rate was calculated using the formula:

$$F_{opt} = 0.6 \times M,$$

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.

5.3 Results

5.3.1 Historical comparisons of giant clam densities

The average density of giant clams (all species) per m² were calculated from data collected in 2001 (Hender *et al.* 2001), 2008 (Hobbs 2008 unpublished data), 2010 and 2011 (DoF) (see Appendix B, Table 1). While there are similarities between the historical surveys in terms of which habitats support high densities of giant clams (i.e. Map unit A and D) the biases introduced by the differences in methodology between surveys preclude any further analysis and comparison of these results.

5.3.2 Species composition of the giant clam population

Of the 77 sites or 154 transects surveyed only two species of giant clam were identified, *Tridacna maxima* and *T. derasa*. A total of 1885 giant clams were counted with *T. maxima* comprising 91.51 % of the population (1,725 individuals) while only one *T. derasa* was identified. Another 159 (8.45 %) giant clams could not be conclusively identified due to the shell being embedded in the substrate, shell features eroded and/or the shell having undeveloped features. There were also 542 giant clams recorded in 'The Rip': *T. maxima* (89.85 %, n = 487), 'unknown' (9.96 %, n = 54) and *T. derasa* (0.0018 %, n = 1). Due to the low abundance of *T. derasa* recorded in this survey the remainder of this report discusses only the abundance and distribution of the giant clam *T. maxima* at the Cocos (Keeling) Islands.

5.3.3 Average density and total standing stock of *T. maxima*

The highest density of *T. maxima* was recorded in the no-take area ‘The Rip’ (1.2175 clams per m², SE ± 0.1625, n = 2) (Figure 5.2). Outside the no-take area Aligned Coral Flats (Map Unit D) displayed the highest abundance of *T. maxima* (0.2083 clams per m², SE ± 0.0592, n = 6). In the remaining habitats *T. maxima* densities varied between 0 – 0.2083 clams per m² with the highest densities consistently recorded in habitats classified as suitable (i.e. Map Units D, B, O and Q) due to the presence of hard substrate (Figure 5.2). Densities of *T. maxima* in most habitats had a high standard error which is consistent with species that have patchy distributions.

The total average density of *T. maxima* at the Cocos (Keeling) Islands is 0.0560 clams per m² (SE ± 0.0107, n = 154) excluding ‘The Rip’ or 0.0709 clams per m² (SE ± 0.0149, n = 156) including data from ‘The Rip’. Using only suitable giant clam habitat (see Table 5.1) the density of *T. maxima* increases slightly to 0.0638 clams per m² (SE ± 0.0121, n = 134) excluding ‘The Rip’ or 0.0808 clams per m² (SE ± 0.0170, n = 136) including ‘The Rip’.

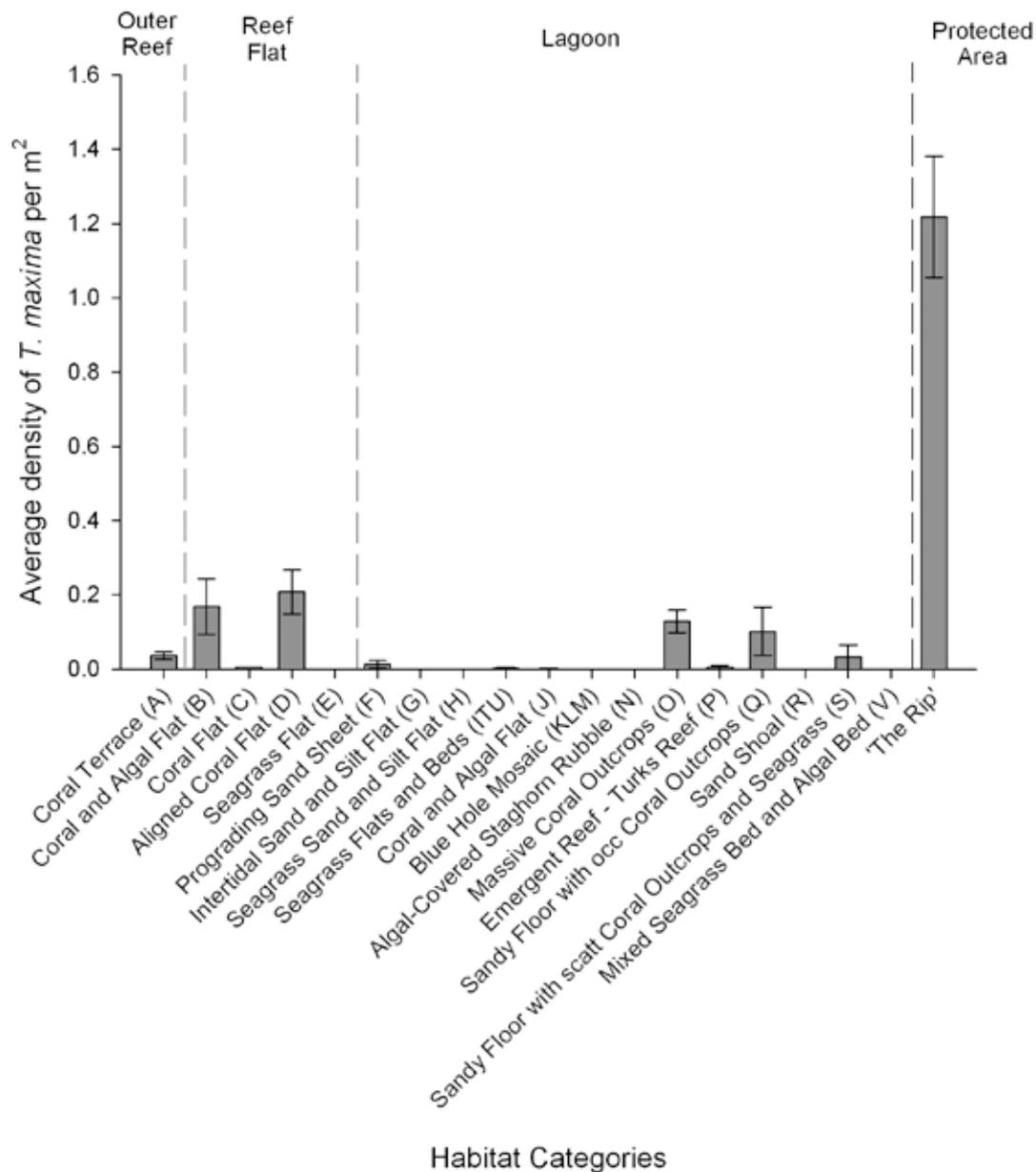


Figure 5.2. Average *T. maxima* density per m² based on habitat units of Williams (1994).

The median total standing stock of *T. maxima* was also calculated for each of the habitat strata (see Table 5.1) with upper and lower 95 % confidence intervals (Table 5.2). The sum of all six strata was calculated to provide a total standing stock of 6.92 million *T. maxima* in the non-protected area (~133.96 km²).

Table 5.2. Standing stock of *T. maxima* in non-protected areas of Cocos (Keeling) Islands.

| Strata | Median (No of Individuals) | 95% Lower Confidence Interval | 95% Upper Confidence Interval | Area (km ²) |
|----------|----------------------------|-------------------------------|-------------------------------|-------------------------|
| Strata 1 | 829,745 | 503,960 | 1,337,674 | 21.86 |
| Strata 2 | 2,983,503 | 1,280,847 | 7,004,151 | 12.20 |
| Strata 3 | 145,175 | 0 | 912,714 | 30.27 |
| Strata 4 | 26,777 | 6,149 | 60,313 | 18.14 |
| Strata 5 | 0 | 0 | 0 | 29.75 |
| Strata 6 | 2,931,069 | 1,692,303 | 4,994,919 | 21.74 |
| Total | 6,916,269 | 3,483,259 | 14,309,771 | 133.96 |

5.3.4 Size structure of the *T. maxima* population

Of the 1,725 *T. maxima* recorded (excluding the 487 individuals in ‘The Rip’) 1,159 were measured. The median length of all *T. maxima* was 107 mm with the smallest *T. maxima* recorded being 12 mm and the largest 273 mm (Figure 5.3).

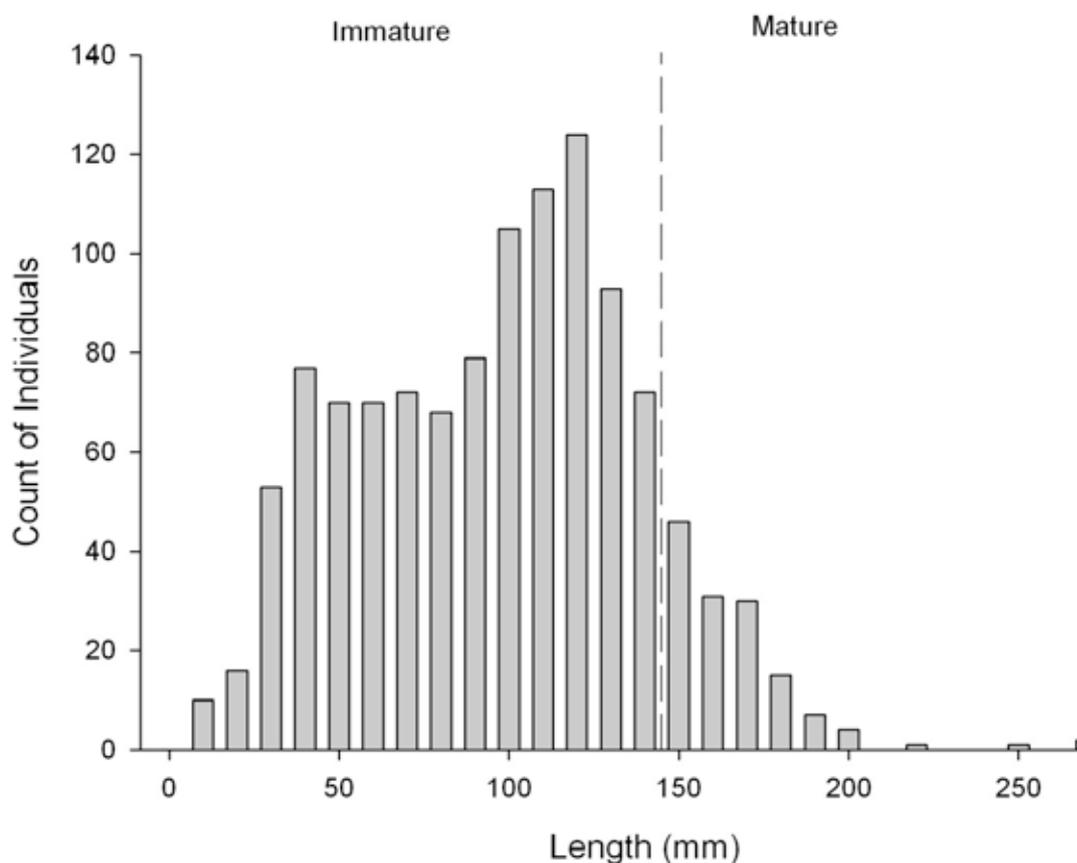


Figure 5.3. Size frequency of *T. maxima* (size classes 10 mm). Dashed line indicates size at maturity (immature < 150 mm and mature individuals ≥ 150 mm).

The majority of the *T. maxima* population is between 50 – 150 mm shell length (68.68 %), 11.82 % are 100 % mature (≥ 150 mm), 19.50 % are juvenile (≤ 50 mm) and few individuals are larger than 200 mm (Figure 5.3). Due to the cryptic nature of the smaller clams it is likely that the number of *T. maxima* ≤ 30 mm is under-estimated. There is a dramatic decline in the number of individuals > 120 mm. This decline is also evident in Figure 5.4 which shows size frequency of *T. maxima* in the habitat strata for which ≥ 50 individuals were measured.

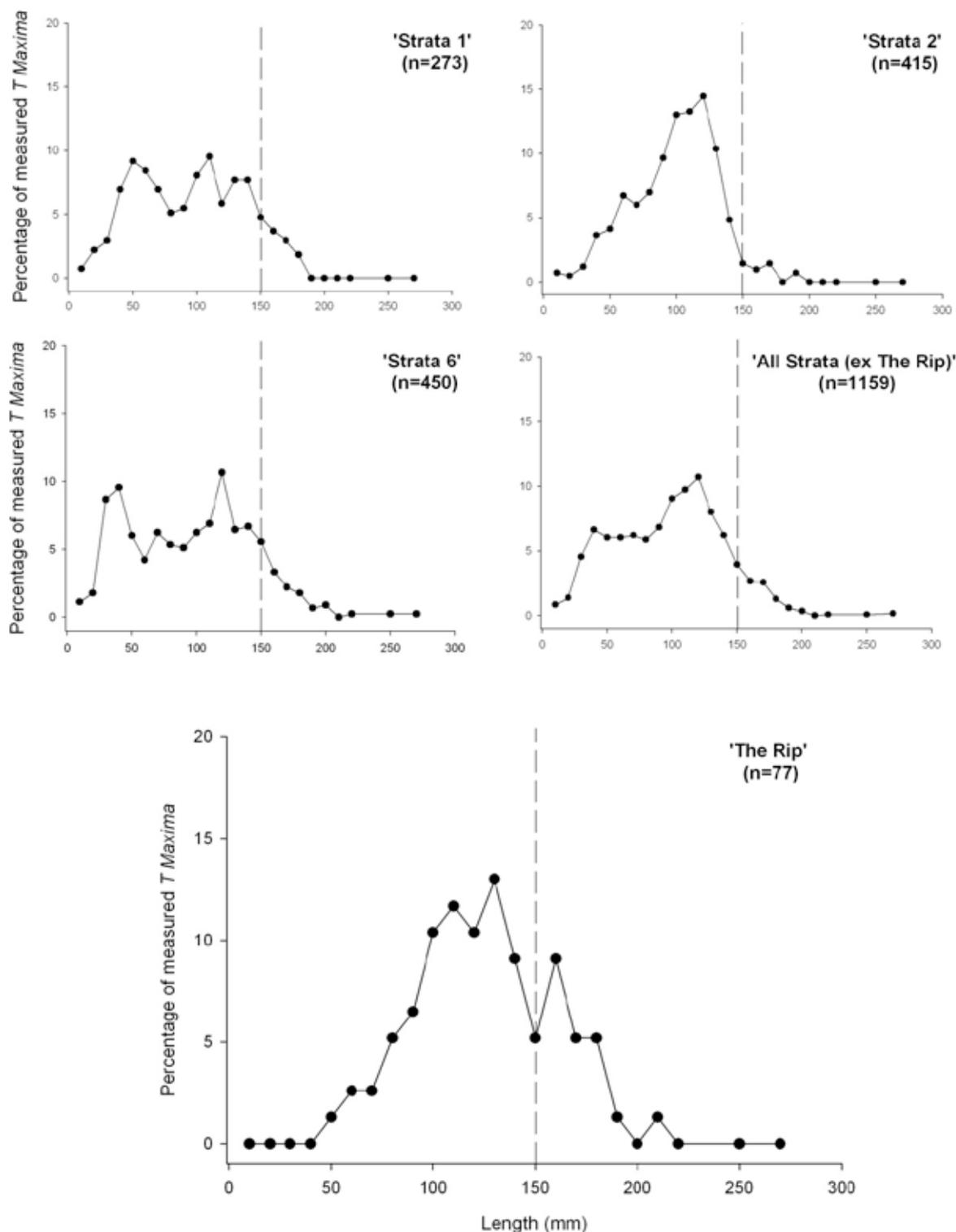


Figure 5.4. Size frequency of *T. maxima* per stratum (10 mm categories). Dashed line represents estimated size at 100 % sexual maturity.

Stratum 2, which includes the shallow water reef flats, has the sharpest decline in abundance of *T. maxima* > 120 mm (Figure 5.4). Strata 1 and 6, which include the deep water habitats of the outer reef and lagoon, respectively, display a decline in abundance of *T. maxima* > 150 mm. The protected area ‘The Rip’ (which is also shallow water reef flat habitat) is potentially providing greater refuge for *T. maxima* to survive to sexual maturity (150 mm) with 27.27 % of the population \geq 150 mm compared with non-protected areas (11.82 %).

5.3.5 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 are used in the absence of any other assessment of recreational fishing pressure at Cocos (Keeling) Islands. Assuming fishers collected only five clams once per week the annual harvest was estimated to be 28,080 *T. maxima* (Table 4.3). If fishing occurred twice per week and fishers collected 20 per trip the estimate would be 224,640 *T. maxima*.

Table 5.3. Estimated total annual harvest of *T. maxima* collected at the Cocos (Keeling) Islands based on one or two trips per year and collecting between five and 20 clams per trip.

| No. people | No. of days fished /year | No. 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 |

5.3.6 Estimated optimal annual harvest of *T. maxima*

The optimal annual harvest of *T. maxima* (Table 5.5) was calculated using the estimated *T. maxima* population size (Table 4.2) and the proportion which are 100 % sexually mature (11.82 %). Using the P_{opt} formula, the annual average optimal harvest was calculated using three different natural mortality estimates including the average of all mortality rates (0.263).

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 | 103,666 | 52,210 | 214,485 |
| 0.300 ² | 134,667 | 67,883 | 278,626 |
| 0.263 * | 119,339 | 60,103 | 246,912 |

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

5.4 Discussion

Only two species of giant clams (*Tridacna maxima* and *Tridacna derasa*) were identified in this survey. The population was dominated by *T. maxima* with only one *T. derasa* recorded

(an additional *T. derasa* was found in ‘The Rip’). No *T. gigas* were recorded. In addition, approximately 9 % of giant clams observed could not be conclusively identified to species due to their small size and/or obscured features important for identification. The low abundance of *T. derasa* observed at the Cocos (Keeling) Islands indicates that any fishing of this species is unsustainable. No *T. gigas* were recorded during this survey and anecdotal reports suggest the species may be locally extinct.

Prior to this survey there had not been a comprehensive survey conducted specifically to assess giant clam abundance and distribution. The historical information that is available was collected as a subset of general benthic or pilot surveys. Differences in sampling methodology and intensity, as well as the habitats surveyed, means the data from previous surveys cannot be statistically compared with our survey. Observational comparisons of our survey data and Hender *et al.* (2001) suggest that giant clam abundance may have remained relatively stable over a period of 10 years. However, caution should be used when comparing the two datasets as there are a number of discrepancies. Firstly, Hender *et al.* (2001) sampled only a small portion of the potential giant clam habitats and may have underestimated densities as it was not a dedicated survey and more cryptic individuals may have been overlooked (Gilbert *et al.* 2006). Secondly, Hender *et al.* (2001) used small 2 x 2 m quadrats which may result in inaccuracies due to scaling up from relatively small areas (Andrew and Mapstone 1987). Finally, Hender *et al.* (2001) did not identify giant clams to species level therefore the difference in abundance between 2001 and 2011 cannot be calculated as the data may contain more than one species.

An international review of the abundance, distribution and status of all giant clam species showed densities vary considerably between countries (Appendix B, Table 2). Typically the density of giant clam species ranges between 0.001 – 0.00001 individuals per m² with densities lower in many countries due to a long history of intensive fishing pressure and habitat destruction (Othman *et al.* 2010). Based on the results of our survey, the density of *T. maxima* at Cocos (Keeling) Islands is 0.056 ± 0.012 S.E. individuals per m². This is higher than both the median worldwide abundance for *T. maxima* of 0.000818 individuals per m² (see Othman *et al.* 2010; Appendix B, Table 2) and that for other fished areas in the north eastern Indian Ocean such as Mermaid (0.0158 individuals per m²), Cartier (0.00218 per m²) and Ashmore Reefs (0.00383 individuals per m²) (Rees *et al.* 2003; Appendix B, Table 2). However, it is significantly lower than the density at Ningaloo Reef in Western Australia (0.86 ± 0.41 S.E. individuals per m²) where recreational and commercial fishing for all giant clams is prohibited (Black *et al.* 2011).

T. maxima densities in ‘The Rip’ were approximately six-times higher than other areas sampled at the Cocos (Keeling) Islands, in fact densities in ‘The Rip’ are similar to other protected areas like Ningaloo in Western Australia. This suggests that protection from exploitation through the use of protected or no-take areas may provide some refuge for clam populations. In comparison with other locations the densities of *T. maxima* at the Cocos (Keeling) Islands suggests that at the current harvest rates this species is not overexploited. However, in the absence of historical baseline data caution must be used as it is likely that the population of giant clams has been significantly reduced from its natural state by fishing over an extended period with harvesting for local consumption being recorded for over 60 years (Gibson-Hill 1949). While there is no catch data, anecdotal and published reports suggest that *T. derasa* and *T. gigas* have been harvested in the past at the Cocos (Keeling) Islands (Berry 1989) and the current extremely low densities of these species suggest that they have been over exploited.

Only three of the six habitat strata had densities capable of supporting some level of harvesting (Strata 1, 2 and 6). Each of these strata has different levels of accessibility with the reef flat environments in Stratum 2 being the most accessible while the outer reef slopes of Stratum 1 are the least accessible. Calculating the standing stock by habitat stratum allows greater discrimination of giant clam densities and provides managers with the capacity to implement a habitat-specific total allowable catch, particularly when coupled with detailed catch and effort data, to ensure the sustainability of stocks.

The *T. maxima* population at the Cocos (Keeling) Islands is comprised of 19.5 % juveniles (< 50 mm) and 68.68 % sub-adults (50 – 150 mm) which indicates that regular recruitment and transition to the next life stage is occurring. The lower percentage of juveniles observed may be due in part to their cryptic nature. However, only 11.82 % of the population was 100 % mature (i.e. ≥ 150 mm). This is lower than reported in other parts of the Indian Ocean where fishing pressure is low (Apte *et al.* 2010, Black *et al.* 2011). The percentage of mature giant clams can vary greatly as it is related to recruitment, mortality and harvest rates (Gilbert *et al.* 2006). Ensuring adequate survival of sexually mature individuals is essential for ongoing recruitment, particularly as giant clams require relatively high densities to ensure successful spawning (Munro 1992, Kinch and Teitelbaum 2012).

Our data suggest that there is a dramatic decrease in the abundance of *T. maxima* at or before the size of 100 % sexual maturity (150 mm). This is particularly evident in Strata 2, which displays a dramatic decline in abundance in all size classes above 120 mm well before full sexual maturity. Stratum 2 is comprised of easily accessible shallow-water reef flat habitats and therefore, may be subjected to higher levels of fishing pressure. In comparison, *T. maxima* populations in the protected area ‘The Rip’ also decline around 150 mm but do not display a dramatic decline until 180 mm (with few clams reaching 200 mm) which indicates that natural mortality is driving the decline in abundance of giant clams > 180 mm. However, the lack of comparable historical survey data and catch information make it difficult to assess how long the population has been harvested at this size which may have severe impacts for future recruitment.

There is currently no minimum legal size for giant clams at Cocos (Keeling) Islands. Setting a minimum legal size that corresponds to size at maturity is frequently used as a fisheries management tool so reproduction can occur at least once before harvesting (Chambers 2007, Gilbert *et al.* 2012). The minimum legal size published in the international literature varies but typically corresponds with the size at maturity which is on average 150 mm (Green and Craig 1999, Gilbert *et al.* 2006, Chambers 2007, Apte *et al.* 2010). In Australia giant clams are protected in Queensland and the Northern Territory (Queensland Government 2012, Northern Territory Government 2012).

The current annual harvest of giant clams at the Cocos (Keeling) Islands is unknown. Broad estimates were calculated to provide some understanding of the potential current harvest. Based on the most conservative values for optimal harvest of *T. maxima* (60 103 individuals) it appears that if fishers collect the bag limit once a week the annual catch is sustainable. However, if the assumed fishing population fished twice a week the harvest rate would exceed the sustainable level. It is important to note that the estimates provided above are only for recreational fishing (i.e. there is no commercial catch) and do not account for the unique life history characteristics of this species.

The life history characteristics and accessibility of giant clams make them particularly vulnerable to over-harvesting. Giant clams are protandrous hermaphrodites and become simultaneous

hermaphrodites as they grow. Therefore, if larger individuals are removed from the population there may be an insufficient number of egg-producing individuals to ensure successful recruitment. In addition, as giant clams require high densities to promote spawning and successful egg fertilization reduced densities may also lead to failed recruitment (Munro 1992).

Giant clams populations at the Cocos (Keeling) Islands also have several additional pressures that make them more susceptible to overexploitation. Since human colonisation in 1826 at least nine catastrophic die-off events have been recorded due to increased water temperature and/or anoxia from coral spawning events. In 1983, 100 % mortality of corals, algae and fishes in the lower lagoon was recorded (Bunce 1988, Hobbs and McDonald 2010, Hobbs and Macrae 2012). In addition, the Cocos (Keeling) Islands are isolated with little or no larval input from outside areas making them vulnerable to local extinctions (Hourston 2010). The larval period for *T. maxima* and *T. derasa* is 8 – 9 days (Benzie and Williams 1997) and it is estimated that the connectivity time for passive ocean currents to the nearest landmass and associated reefs is around 80 days (Condie *et al.* 2005). Therefore, it is likely that the giant clam populations at the Cocos (Keeling) Islands are almost exclusively self-recruiting with limited outside capacity for replenishment and extremely vulnerable to over-exploitation as evidenced by the potential local extinction of *T. gigas*.

5.5 Management Recommendations

The Department of Fisheries Western Australia makes the following recommendations with regard to the harvest of giant clams at the Cocos (Keeling) Islands:

- Complete protection for giant clam species *T. derasa* and *T. gigas*;
- Complete protection for giant clam species *T. derasa* and *T. gigas*;
- Only collection of *T. maxima* should be permitted, with a conservative daily bag limit;
- To assist recreational fishers with identification of clam species educational material needs to be developed;
- Minimum size limit of 150 mm of *T. maxima* is introduced, consistent with size restrictions in other countries;
- As estimates of recreational take indicate that *T. maxima* is close to maximum sustainable yield, extremely conservative limits should be set on any additional take (recreational or commercial);
- A program to quantify the recreational harvest of giant clams is implemented;
- A monitoring program is conducted every second year to refine density estimates and assess stock trends of the giant clam population; and
- Management guidelines are reassessed based on monitoring data.

6.0 Acknowledgements

The authors thank the Cocos (Keeling) Islands community for their assistance, local knowledge and co-operation. We are grateful to Parks Australia Cocos (Keeling) Islands particularly Ismail MacRae, Mohammad Chongkin, Trish Flores and Robert Green for their assistance over many years. We extend our appreciation to Hamidah Abedin and the Cocos Island Youth Council for sample collections and Gordon Thomson (Murdoch University) for histological preparations.

We thank our colleagues Fiona Webster, Dovid Clarke, Nick Konzewitsch, Ben Rome, Josh Dornan, Partick Hayes, Karina Inostroza, Alex Hesp and Kendra Travaille for their invaluable assistance in the field, laboratory and office.

This work was funded by the Australian Commonwealth Department of Regional Australia, Local Government, Art and Sport.

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Appendix A

Table 1. Densities (individuals ha⁻¹) of commercially important holothurians in the Indian and Pacific Ocean regions.

| Species | Mean Density | Maximum Density | Location | Reference |
|---------------------------------|--------------|-----------------|-------------------------|------------------------------|
| <i>Holothuria atra</i> | 2983.54 | 28 525 | Cocos Atoll | This Study |
| | 7 183.15 | 44 000 | Cocos Atoll | Hender <i>et al.</i> (2001) |
| | 16.73 | | MOU74 Box N W Australia | Skewes <i>et al.</i> (1999) |
| | 545 | 7 270 | New Caledonia | Conand (1989) |
| | | 520 000 | Marshall Islands | Lawrence (1980) |
| | 4 870.6 | 14 600 | Laing Island, PNG | Massin and Doumen (1986) |
| <i>Bohadschia argus</i> | 7.44 | 175 | Cocos Atoll | This study |
| | 131.88 | 2000 | Cocos Atoll | Hender <i>et al.</i> (2001) |
| | 1.1 | | MOU74 Box N W Australia | Skewes <i>et al.</i> (1999) |
| | 24.4 | 212.2 | Laing Island, PNG | Massin and Doumen (1986) |
| <i>Stichopus chloronotus</i> | 36.23 | 1 013 | Cocos Atoll | This Study |
| | 35.88 | 750 | Cocos Atoll | Hender <i>et al.</i> (2001) |
| | 1.67 | | MOU74 Box N W Australia | Skewes <i>et al.</i> (1999) |
| | | 4 258 | PNG | Lokani (1991) |
| <i>Thelenota ananas</i> | 0.63 | 13 | Cocos Atoll | This study |
| | 12.57 | 250 | Cocos Atoll | Hender <i>et al.</i> (2001) |
| | 0.06 | | MOU74 Box N W Australia | Skewes <i>et al.</i> (1999) |
| | 17.5 | 31.4 | Niue | Dalzell <i>et al.</i> (1993) |
| | 16.8 | 39.8 | Tonga | Preston and Lokanni (1990) |
| <i>Actinopyga echinities</i> | 2.27 | 250 | Cocos Atoll | Hender <i>et al.</i> (2001) |
| | 241.3 | 743.8 | Laing Island, PNG | Massin and Doumen (1986) |
| <i>Bohadschia graeffei</i> | 1.11 | 50 | Cocos Atoll | This Study |
| | 6.8 | 250 | Cocos Atoll | Hender <i>et al.</i> (2001) |
| | 0.29 | | MOU74 Box N W Australia | Skewes <i>et al.</i> (1999) |
| <i>Holothuria edulis</i> | 53.64 | 1400 | Cocos Atoll | This Study |
| | 140.37 | 3500 | Cocos Atoll | Hender <i>et al.</i> (2001) |
| | 5.35 | | MOU74 Box N W Australia | Skewes <i>et al.</i> (1999) |
| <i>Holothuria fuscopunctata</i> | 99.21 | 7838 | Cocos Atoll | This Study |
| | 22 | 105.9 | Tonga | Preston and Lokanni (1990) |
| <i>Actinopyga mauritiana</i> | 3.64 | 288 | Cocos Atoll | This Study |
| | | 304 | Papua New Guinea | Lokani (1991) |
| | | 120000 | Marshall Islands | Lawrence (1980) |
| <i>Stichopus maculata</i> | 64.4 | 5088 | Cocos Atoll | This Study |
| <i>Holothuria nobilis</i> | 0.32 | 25 | Cocos Atoll | This Study |
| | 13 | 84 | New Caledonia | Conand (1989) |
| | | 275 | Papua New Guinea | Lokani (1991) |
| | 18.7 | 40.3 | Tonga | Preston and Lokanni (1990) |
| | 9.4 | 37.5 | Laing Island, PNG | Massin and Doumen (1986) |

| | | | | |
|----------------------------|------|-------|---------------|----------------------------|
| <i>Chiridota rigidia</i> | 0.16 | 13 | Cocos Atoll | This Study |
| <i>Holothuria coluber</i> | 0.16 | 13 | Cocos Atoll | This Study |
| <i>Actinopyga miliaris</i> | 0.63 | 50 | Cocos Atoll | This Study |
| | 512 | 5970 | New Caledonia | Conand (1989) |
| | | 78900 | Fiji | Preston and Lokanni (1990) |
| <i>Stichopus hermanni</i> | 0.63 | 50 | Cocos Atoll | This Study |

Appendix B

Table 1. Historical giant clams surveys of the Cocos (Keeling) Islands (mean density, SE, n) and the corresponding Williams (1994) map units and suitable and not suitable habitat.

| Map Unit | Suitable / Not Suitable | Habitat Area (km ²) | Hender 2001 | Hobbs 2008 | DoF 2010 | DoF 2011 |
|------------------------|-------------------------|---------------------------------|-----------------------|-----------------------|-----------------------|------------------------|
| A | Suitable | 21.86 | 0.046 (± 0.014, n=19) | 0.047 (± 0.019, n=17) | 0.045 (± 0.017, n=7) | 0.046 (± 0.012, n=42) |
| B | Suitable | 9.61 | 0.025 (N/A, n=1) | N/S | 0.457 (± 0.324, n=2) | 0.174 (± 0.074, n=16) |
| C | Suitable | 0.10 | N/S | N/S | N/S | 0.005 (± 0.000, n=2) |
| D | Suitable | 2.49 | 0.200 (± 0.050, n=2) | N/S | 0.292 (± 0.207, n=2) | 0.208 (± 0.059, n=6) |
| E | Not Suitable | 0.27 | N/S | N/S | N/S | 0.000 (± 0.000, n=2) |
| F | Not Suitable | 8.72 | 0.000 (± 0.000, n=3) | N/S | N/S | 0.013 (± 0.011, n=6) |
| G | Not Suitable | 5.47 | N/S | N/S | N/S | 0.000 (± 0.000, n=2) |
| H | Not Suitable | 12.95 | 0.000 (± 0.000, n=4) | N/S | N/S | 0.000 (± 0.000, n=6) |
| I, T and U | Not Suitable | 2.68 | N/S | N/S | N/S | 0.003 (± 0.003, n=2) |
| J | Suitable* | 17.69 | 0.003 (± 0.003, n=5) | N/S | N/S | 0.002 (± 0.001, n=18) |
| K, L, M | Suitable | 16.60 | 0.008 (± 0.008, n=5) | N/S | N/S | 0.000 (± 0.000, n=8) |
| N | Suitable | 13.15 | 0.019 (± 0.012, n=4) | N/S | N/S | 0.000 (± 0.000, n=12) |
| O | Suitable | 12.63 | 0.131 (± 0.080, n=4) | N/S | 0.282 (± 0.200, n=2) | 0.146 (± 0.032, n=18) |
| P | Suitable | 0.06 | N/S | N/S | N/S | 0.018 (± 0.008, n=2) |
| Q | Suitable | 6.85 | 0.000 (± 0.000, n=2) | N/S | N/S | 0.105 (± 0.067, n=6) |
| R | Not Suitable | 0.19 | N/S | N/S | N/S | 0.000 (± 0.000, n=2) |
| S | Suitable | 2.33 | N/S | N/S | N/S | 0.033 (± 0.033, n=2) |
| V | Suitable* | 0.46 | N/S | N/S | N/S | 0.000 (± 0.000, n=2) |
| Total Suitable Habitat | | | 0.046 (± 0.012, n=42) | 0.047 (± 0.019, n=17) | 0.183 (± 0.051, n=13) | 0.070 (± 0.012, n=134) |
| Total All Habitat | | | 0.040 (± 0.011, n=49) | 0.047 (± 0.019, n=17) | 0.183 (± 0.051, n=13) | 0.061 (± 0.011, n=154) |

Table 2. Summary of giant clam distribution and density (m²) data adapted from (Othman *et al.* 2010)

| Location | Area (m ²) | <i>T. costata</i> | <i>T. crocea</i> | <i>T. derasa</i> | <i>T. gigas</i> | <i>T. maxima</i> | <i>T. squamosa</i> | <i>H. hippopus</i> | <i>H. porcellanus</i> | Reference |
|--------------------------------------|------------------------|-------------------|------------------|------------------|-----------------|------------------|--------------------|--------------------|-----------------------|----------------------------|
| Australia | | | | | | | | | | |
| The Great Barrier Reef | | | | | | | | | | Braley 1987 |
| The Great Barrier Reef - North | | | 0.00029 | 0.00078 | | | | | | Braley 1987 |
| The Great Barrier Reef - South | | | 0.00059 | 0.0006 | | | | | | Braley 1987 |
| Mermaid Reef | | | 0.000588 | 0.0158 | 0.000415 | | | | | Rees <i>et al.</i> 2003 |
| Cartier Reef | | | 0.00218 | 0.0151 | | | | | | Rees <i>et al.</i> 2003 |
| Ashmore Reef | | | 0.000612 | 0.00383 | 0.0114 | | | | | Rees <i>et al.</i> 2003 |
| Caroline Islands | | | | | | | | | | |
| Palau | 110 | 0.139 | 0.00545 | 0.00182 | 0.00545 | 0.00636 | 0.00364 | | | Hardy and Hardy 1969 |
| Egypt and Jordan | | | | | | | | | | |
| Gulf of Aqaba | | | | 0.0047 | 0.0031 | | | | | Roa-Quiaoit 2005 |
| Northern Red Sea and Gulf of Aqaba | 12,800 | 0.0009 | | | | | | | | Richter <i>et al.</i> 2008 |
| Malaysia | | | | | | | | | | |
| Palau Tioman | 2620 | 0.00992 | 0.0538 | 0.0252 | | | | | | Tan <i>et al.</i> 1998 |
| New Zealand (Tokelau Islands) | | | | | | | | | | |
| Fakaofu Atoll | | | 0.175 | 0.000082 | | | | | | Braley 1989 |
| Nukunonu Atoll | | | 0.686 | 0.000041 | | | | | | Braley 1989 |
| Atafu | | | 0.0841 | | | | | | | Braley 1989 |
| Papua New Guinea | | | | | | | | | | |
| Milne Bay Province | | 0.00149 | 0.000034 | 0.000082 | 0.000179 | 0.000137 | | | | Kinch 2002 |
| The Philippines | | | | | | | | | | |
| Central Visayas | 530,000 | 0.000163 | 0.00008 | 0.0000667 | | | | | | Alcala 1986 |
| Western Visayas | 70,000 | 0.000229 | 0.0003 | 0.000131 | | | | | | Alcala 1986 |
| Cagayan, Sulu Sea | 56,500 | 0.00181 | 0.00255 | 0.000124 | | | | | | Alcala 1986 |
| Cagayancillo Island | 65,000 | | 0.000016 | 0.000047 | 0.000078 | | | | | Juinio <i>et al.</i> 1989 |
| Palawan Regions | 210,000 | 0.329 | 0.0000381 | 0.000267 | 0.000271 | 0.000138 | | | | Alcala 1986 |

| Location | Area (m ²) | <i>T. costata</i> | <i>T. crocea</i> | <i>T. derasa</i> | <i>T. gigas</i> | <i>T. maxima</i> | <i>T. squamosa</i> | <i>H. hippopus</i> | <i>H. porcellanus</i> | Reference |
|-----------------------------|------------------------|-------------------|------------------|------------------|-----------------|------------------|--------------------|--------------------|-----------------------|----------------------------------|
| Western Pangasinan | 530,000 | 0.000074 | | | | 0.000011 | 0.000032 | | | Juinio <i>et al.</i> 1989 |
| Polillo | 210,000 | 0.034 | 0.000029 | 0.000001 | | 0.000533 | 0.0007 | 0.000024 | | Juinio <i>et al.</i> 1989 |
| Zambales | 104,000 | 0.000183 | | | | 0.000067 | 0.00001 | | | Juinio <i>et al.</i> 1989 |
| Albay | 145,000 | 0.000821 | | | | 0.000703 | 0.000414 | | | Juinio <i>et al.</i> 1989 |
| Sorsogon | 148,000 | 0.000311 | | | | 0.000818 | 0.000027 | | | Juinio <i>et al.</i> 1989 |
| Calatagan | 111,000 | 0.000126 | | | | 0.000108 | 0.000261 | | | Juinio <i>et al.</i> 1989 |
| Lubang Island | 149,000 | 0.000564 | | | | 0.000148 | 0.000134 | 0.000007 | | Juinio <i>et al.</i> 1989 |
| Ambil Island | 250,000 | 0.000268 | 0.000036 | 0.0000004 | 0.0000004 | 0.000448 | 0.000328 | 0.000008 | | Juinio <i>et al.</i> 1989 |
| Apo Reef | 88,000 | 0.000295 | 0.000011 | | | 0.000943 | 0.000011 | | | Juinio <i>et al.</i> 1989 |
| Puerto Galera | 146,000 | 0.000027 | | | | 0.000096 | 0.000096 | | | Juinio <i>et al.</i> 1989 |
| Northeast Negros | 29,000 | 0.000069 | | | | 0.000034 | 0.000034 | | | Juinio <i>et al.</i> 1989 |
| Ei Nido | 255,000 | 0.0011 | | | | 0.00009 | 0.00049 | 0.000047 | 0.000004 | Juinio <i>et al.</i> 1989 |
| Inaguan-Aborlan | 45,000 | | 0.000069 | | | 0.000022 | 0.000022 | | | Juinio <i>et al.</i> 1989 |
| Camiguin | 213,000 | 0.000113 | | | | 0.00031 | 0.000155 | | | Juinio <i>et al.</i> 1989 |
| Republic of Kiribati | | | | | | | | | | |
| Aberama Atoll | 28,600,000 | | | | 0.00023 | | 0.0000115 | 0.000351 | | Munro 1988 |
| Abiang Atoll | 89,900,000 | | | | 0.0000549 | | 0.0000592 | 0.000221 | | Munro 1988 |
| Maiana Atoll | 28,000,000 | | | | 0.0000768 | | 0.0000921 | 0.0000571 | | Munro 1988 |
| Tarawa Atoll | 29,600,000 | | | | 0.0000189 | | 0.0000264 | 0.0000169 | | Munro 1988 |
| Samoan Archipelago | 505,000 | | | | | 0.00565 | | | | Green and Craig 1999 |
| Singapore | | | | | | | | | | |
| Southern Islands | 9,760 | 0.000724 | | | | 0.000102 | 0.00155 | | | Guest <i>et al.</i> 2008 |
| Thailand | | | | | | | | | | |
| Lee-Pae Island | 6,400 | 0.244 | | | | 0.0629 | | | | Chantrapornsy <i>et al.</i> 1996 |
| Vanuatu | | | | | | | | | | |
| Port Anatom | | | | | | 0.0016 | | | | Zann and Ayling 1988 |

| Location | Area (m ²) | <i>T. costata</i> | <i>T. crocea</i> | <i>T. derasa</i> | <i>T. gigas</i> | <i>T. maxima</i> | <i>T. squamosa</i> | <i>H. hippopus</i> | <i>H. porcellanus</i> | Reference |
|---------------|------------------------|-------------------|------------------|------------------|-----------------|------------------|--------------------|--------------------|-----------------------|----------------------|
| Inyeug Island | | | | | | 0.005 | | 0.001 | | Zann and Ayling 1988 |
| Port Patrick | | | | | | 0.0016 | | | | Zann and Ayling 1988 |
| Lakariata | | | | | | 0.0005 | | | | Zann and Ayling 1988 |
| Lelepa | | | 0.0003 | | | 0.0003 | | 0.0003 | | Zann and Ayling 1988 |
| Cook's Reff | | | | | | 0.001 | | 0.0025 | | Zann and Ayling 1988 |
| Pentecost | | | | | | 0.0006 | | 0.0009 | | Zann and Ayling 1988 |
| Loltong Bay | | | | | | 0.002 | | | | Zann and Ayling 1988 |
| Lesalav Bay | | | | | | 0.0009 | | 0.0001 | | Zann and Ayling 1988 |
| Reef Island | | | | | | 0.0013 | | 0.0023 | | Zann and Ayling 1988 |
| Hog Bay | | | | | | 0.0002 | | 0.0002 | | Zann and Ayling 1988 |

