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# Ashmore Reef Marine Park Environmental Assessment

Marine Section (Part 3 of 4)

Final report to Parks Australia

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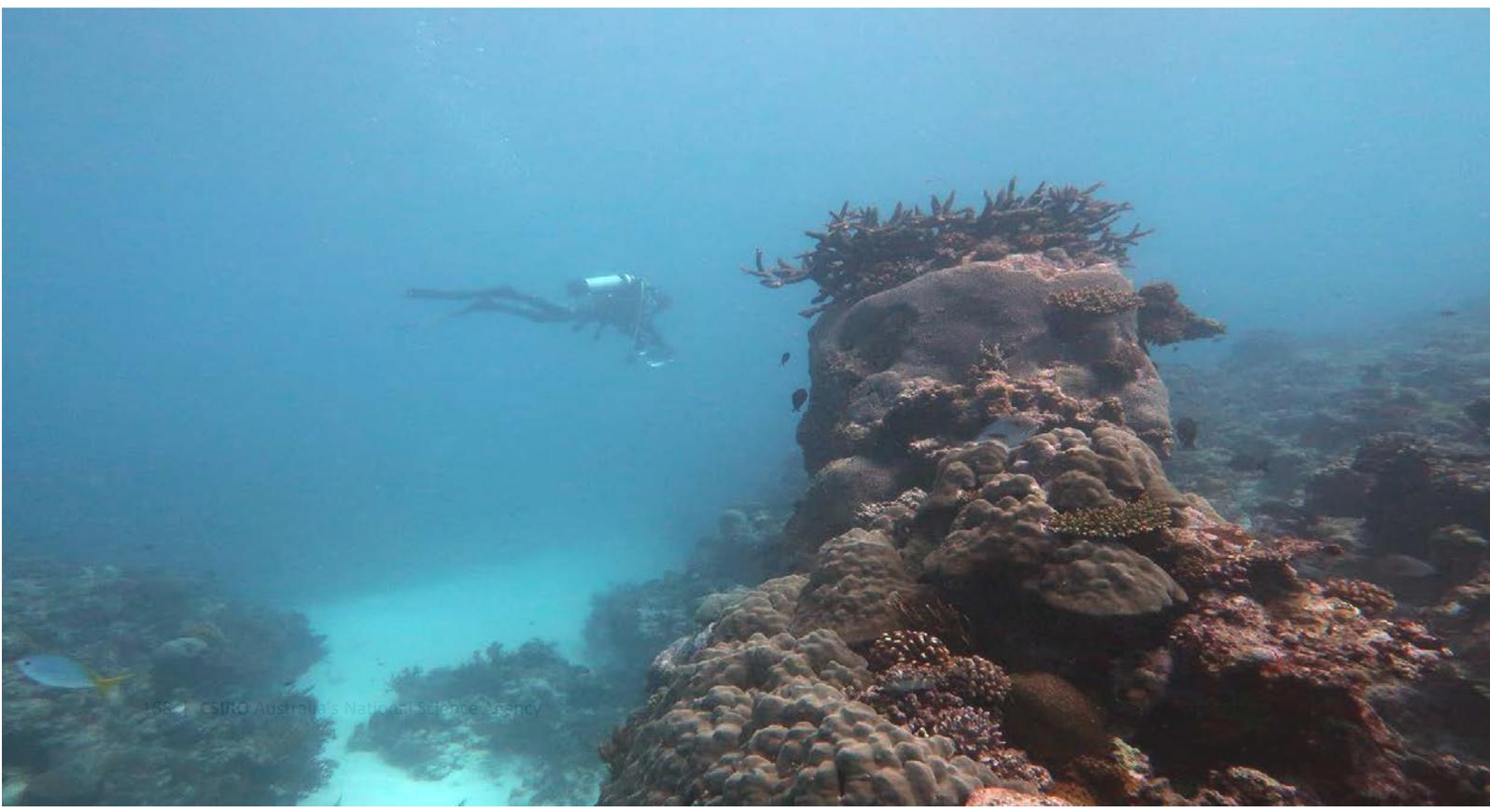
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# PART III MARINE SURVEYS



# 8 ASHMORE REEF: CORAL REEFS AND THE COMMUNITIES THEY SUPPORT: A REVIEW OF PAST SURVEYS AND 2019 FIELD SURVEY METHODS

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## 8.1 Introduction

The marine fauna of Ashmore Reef has a strong similarity to that of Indonesia, consisting of a mixture of continental and coral reef species with wide distributions throughout the Indo West Pacific (Marsh *et al.* 1993). Despite formal conservation measures due to the establishment of a marine park in the 1980s, pressure on the reef's biota from illegal fishing of shark and harvesting of trochus, holothurians and clams continued. Since 2008, there has been a permanent Australian Government presence at the reef as part of Australia's border protection measures. This presence has resulted in consistent enforcement of no-fishing zones, which is thought to have significantly reduced illegal fishing.

### 8.1.1 Marine invertebrates

Ashmore Reef has a high species richness of invertebrates, similar to the continental shelf reefs and much higher than the oceanic reefs (Edgar *et al.* 2017). Before 1998, Ashmore Reef contained high densities of holothurians, trochus and tridacnid clams (Skewes *et al.* 1999a). A total of 47 species of sea cucumber are known to occur at Ashmore Reef, which is higher than other reefs in the bioregion (Marsh *et al.* 1993). However, most quantitative surveys have recorded 12-18 species (see chapter 11) and declines in all groups have been recorded. This reduction was firstly from sustained overfishing by Indonesian fishers (Russell & Vail 1988; Berry 1993) and then during occasional periods of illegal fishing when there were gaps in the presence of enforcement vessels, thought to have been most significant in 2006 (Richards *et al.* 2009; Ceccarelli *et al.* 2013).

There has been a long history of research at Ashmore Reef, in particular biodiversity surveys and surveys on heavily exploited species such as sharks, holothurians, clams and trochus. Surveys have been recorded since 1978 when collections of echinoderms were made by the USSR R.V *Bogorov* expedition, but these reports were unpublished (Marsh *et al.* 1993). In 1986, species composition of marine invertebrates were surveyed by the Western Australian Museum as well as surveying fish, birds and vegetation (Berry 1993). The Northern Territory Museum also made surveys in 1986 and 1987 with a focus on recording the catches by Indonesian fishers such as fish, holothurians, clams and trochus. Subsequent surveys between 1999 and 2013 were undertaken on separate occasions focusing on a range of taxa and environmental variables, using a variety of methods (Skewes *et al.* 1999a; Smith *et al.* 2001; Rees *et al.* 2003; Kospartov *et al.* 2006; Ceccarelli *et al.* 2007; Richards *et al.* 2009; Heyward *et al.* 2010; Heyward *et al.* 2012; Ceccarelli *et al.* 2013; Edgar *et al.* 2017). Most of these surveys concentrated on commercially exploited species of

holothurians, clams and trochus. However, not all communities were surveyed on each occasion and methods varied making quantitative comparisons difficult. Attempts have been made to synthesise and compare data from these reports by Ceccarelli *et al.* (2011b) *et al.* (2011a) and Hosack and Lawrence (2013a) to look for historical trends in status and abundance.

In 2011, it was found that many of the abundances of commercial holothurian species were too low to detect any significant changes in the population (Ceccarelli *et al.* 2011a). Therefore, field testing of a standardised method for monitoring holothurian and trochus populations based on data and methods from all previous surveys was developed and carried out in 2013 (Ceccarelli *et al.* 2013; Hosack & Lawrence 2013a).

In our 2019 survey, holothurian counts were performed during manta tows and invertebrate transect surveys following the same methods used by Ceccarelli *et al.* (2013), to report on the abundance and current status of holothurians, trochus and clams. As per the recommendations of Hosack and Lawrence (2013a), between 12.5% and 25% of sites were located along the reef edge and 216 sites were surveyed in total for targetted invertebrates.

### 8.1.2 Coral communities

Within a regional context, Ashmore Reef contains high diversity of marine life, including hard and soft corals (Richards, Bryce & Bryce 2013), and supports distinct assemblages of benthic and pelagic communities (Commonwealth of Australia 2002). Ashmore Reef has the highest diversity of hermatypic (reef building corals) on the West Australian coast with 275 species from 14 families 51 genera recorded (Ceccarelli *et al.* 2011b).

Isolated offshore coral reefs, such as Ashmore Reef, are important as they are expected to provide critical refuges for the survival of coral reef organisms (Graham, Spalding & Sheppard 2010). However, due to their limited supply of larvae from nearby reefs, they may be more susceptible to climatic disturbances (Graham *et al.* 2006; Smith, Gilmour & Heyward 2008).

Some of the earliest surveys were undertaken in 1986 and 1987 when the Northern Territory Museum surveyed Ashmore Reef which included collecting octocorals (Marsh *et al.* 1993) and a high diversity of reef building corals was recorded in 1986 (Veron 1986). Taxonomic revisions were then made by Griffith (1997) and Ceccarelli *et al.* (2011b). Veron (1986) also documented soft corals as a prominent part of the reef community at Ashmore Reef particularly in the lagoon and on the north-west reef slope.

Since 1999, at least eight field surveys have been undertaken at Ashmore Reef to measure coral cover and composition (Skewes *et al.* 1999b; Rees *et al.* 2003; Kospartov *et al.* 2006; Richards *et al.* 2009; Heyward *et al.* 2010; Ceccarelli *et al.* 2011b; Heyward *et al.* 2012; Edgar *et al.* 2017). Benthic communities at Ashmore Reef are dominated by hard corals and turf algae (Skewes *et al.* 1999b; Heyward *et al.* 2010; Ceccarelli *et al.* 2011b). Coral cover is highly variable but is generally highest on the southern reef edge (mean 21 to 26%, range 0 to 60%) (Richards *et al.* 2009; Ceccarelli *et al.* 2013) where 196 species of hard coral have been recorded. There have been low densities of coral predators (*Drupella* spp. and *Acanthaster planci*) and low incidence of coral disease recorded at Ashmore Reef (Richards *et al.* 2009; Wilson *et al.* 2009).

Following bleaching associated with abnormally high sea surface temperatures during the austral summer in 1998 (Skewes *et al.* 1999b), 2003, 2010 (Heyward *et al.* 2010), and again in 2016/2017

(Gilmour *et al.* 2019), benthic communities at Ashmore Reef have nevertheless remained dominated by hard corals and turf algae, and sand in lagoonal areas (Skewes *et al.* 1999b; Heyward *et al.* 2010).

The historical surveys of coral at Ashmore Reef have all used different methodology making it hard to assess differences between reefs, and between different levels of protection (Edgar *et al.* 2017). In a comparable study by Ceccarelli *et al.* (2011b), hard coral cover increased from 10.2% in 2005 to 29.4% in 2009 across all habitats. This trend likely reflects the prolonged recovery following previous severe bleaching events.

The 2019 surveys used manta tows to estimate coral cover on a macroscale, while photo quadrats collected during belt transects were used to determine benthic cover (percent live hard/soft coral, other benthic biota) at a finer scale. The surveys incorporated all previous recommendations and methodologies to ensure consistency and ability to detect changes over time.

### 8.1.3 Marine vegetation

Macroalgae and seagrass are important primary producers and play an important ecological role on shallow reefs (Vroom *et al.* 2006). They contribute significantly to the productivity of the reef, are a major source of food and provide habitats for invertebrates and fish. Seagrass meadows provide food and habitat for vulnerable species including green turtles (*Chelonia mydas*) listed under the EPBC Act and dugongs (*Dugong dugon*), which are present at Ashmore Reef (Guinea 1993; Whiting & Guinea 2005a).

Previous research on seagrass has included cataloguing species and locations. However, quantitative assessments and comparative studies of seagrass have been missing (Hale & Butcher 2013). A survey in 1997 formulated the first handbook of vascular plants at Ashmore Reef and indicated that there was widespread areas of seagrass (Pike & Leach 1997). They identified five species of seagrass: *Thalassia hemprichii*, *Thalassodendron ciliatum*, *Halophilia ovalis*, *Halophila decipiens* and *Halodule pinifolia*. The presence of the first three species were later confirmed in 1998 and 2001 (Skewes *et al.* 1999b; Brown 2001). Previous studies have shown that *Thalassia hemprichii* is the dominant species at Ashmore Reef comprising over 85% of the total documented seagrass abundance at Ashmore Reef (Skewes *et al.* 1999b). Skewes *et al.* (1999b) made an initial inventory of the macroalgal assemblages at Ashmore Reef, listing nine species plus turf and crustose coralline algae and this has recently been expanded and updated by Huisman (2015, 2018).

The total area of seagrass at Ashmore Reef in 1999 was estimated to be 470 hectares (Skewes *et al.* 1999b). However, only 220 hectares of seagrass was greater than 10% cover, showing sparseness of the overall coverage of seagrass at Ashmore Reef (Brown & Skewes 2005).

In 1999, macroalgae covered over 3,000 ha at Ashmore Reef and was mainly distributed across the reef slope and crest (Skewes *et al.* 1999b). The algal community is mainly made up of turfs and crustose coralline algae with fleshy macroalgae making up less than 10% (Skewes *et al.* 1999b; Kospartov *et al.* 2006; Richards *et al.* 2009). Algae, mainly turfs and crustose coralline algae have been reported to be more abundant at Ashmore Reef compared to similar reefs in the North-west bioregion (Edgar *et al.* 2017).

Our 2019 survey included determining seagrass, macroalgal cover and diversity at 216 sites spread among each of the five major habitat types at Ashmore Reef (reef edge, deep lagoon, shallow lagoon, intertidal sand flat and intertidal reef flat) Figure 57. We also established sites within seagrass meadows suitable for long term monitoring of cover, shoot density, biomass and productivity using a set of standard protocols that are comparable with work carried out across the broader Indian Ocean region.

#### 8.1.4 Sea snakes

Ashmore Reef has long been renowned as a global hotspot for sea snake biodiversity and endemism (Guinea & Whiting 2005). However, there has been sustained declines of sea snakes recorded at Ashmore Reef (and across the North-west marine region) over recent decades. The magnitude and cause(s) of these declines are still not known but generally attributed to environmental change (Hoey & Pratchett 2017). Given its significance as a global hotspot, the sea snake fauna at Ashmore Reef has been subjected to surveys over almost 100 years, with a rich collection of population and distribution data available. However, an enigmatic decline in the sea snakes at Ashmore Reef has been detected recently with several species presumed to have become locally extinct.

Historical surveys of sea snakes have been undertaken at Ashmore Reef since 1926. Several surveys have recorded a large number of sea snakes and all 17 species from the Timor Sea have been recorded here (Minton & Heatwole 1975; Cogger 2000). Subsequent surveys in the 1990s further supported the theory that Ashmore Reef supports the greatest diversity of sea snakes in the world (Guinea & Whiting 2005; Guinea 2007).

However, surveys conducted from 2005 onwards noticed a substantial decline in sea snake diversity and numbers (Lukoschek *et al.* 2013). This decline in sea snake numbers to below the level of detection was unprecedented and unexplained. Generalist feeders such as the olive sea snake (*Aipysurus laevis*) were the last to disappear (Guinea 2013). However, in 2016, three olive sea snakes were recorded in baited camera traps set west of Ashmore Reef (Conrad Speed, pers. comm.). In 2017, a 10-day survey detected four olive sea snakes at the extreme south-east outer reef and another one in the West Island channel, and later the same year, two olive sea snakes were seen within the West Island channel (Guinea & Mason 2017). The most recent survey by the University of Tasmania in 2018 did not record any sea snakes within Ashmore Reef, while numerous specimens from three species were observed at nearby Scott, Seringapatam, Hibernia and Cartier reefs during the same survey period (Graham Edgar, pers. comm.).

Our 2019 survey used multiple survey methods such as boats surveys, reef walks and manta tows during two voyages to determine the abundance and diversity of sea snakes around Ashmore Reef and potential causes for declines.

#### 8.1.5 Turtles

Ashmore Reef is an internationally important region for foraging and nesting turtles (Whiting & Guinea 2005a; Guinea 2013; Guinea & Mason 2017). Ashmore Reef has an abundance of seagrass beds and sandy beaches providing suitable nesting conditions for green turtles (*Chelonia mydas*) which are the dominant species at Ashmore Reef and listed as vulnerable under the EPBC Act. The reefs at Ashmore are estimated to support over 10,600 green turtles (Guinea & Whiting 2005) and

studies by (Dethmers *et al.* 2006) shows that the those nesting at Ashmore Reef form a distinct management unit together with those nesting on Cartier Island.

There are also smaller numbers of loggerhead turtles (*Caretta caretta*) listed as endangered under the EPBC Act which have had a single report of nesting on West Island (Whiting & Guinea 2005a). The hawksbill turtle (*Eretmochelys imbricata*) is also reported to feed and nest around the islands of Ashmore Reef (Whiting & Guinea 2005b) and listed as vulnerable under the EPBC Act.

The seagrass and algae habitats on the reef flats are critically important to sustain Ashmore Reef's population of turtles (Brown & Skewes 2005). Any decline in these resource habitats will have a detrimental impact on turtle populations. There has also been evidence of non-native species impacting the populations of turtles through nesting interference by tropical fire ants (Guinea 2013).

Surveys of turtles have been reported in the early 1950's (Serventy 1952a, 1952b). However, there have been limited quantitative surveys on turtles at Ashmore Reef (Hale & Butcher 2013). The surveys that have been completed have been inconsistent since the early 1980's as they were on a voluntary ad hoc basis. In 1983 Australia declared a marine protected area around Ashmore Reef and a turtle management plan was a feature. Standardised assessment of turtles was only initiated in 1994 (Whiting & Guinea 2005a). While the surveys for turtles at Ashmore Reef have varied in methods and timing from year to year, in general, they have consistently reported large numbers of green turtles.

The last survey recorded at Ashmore Reef was by Guinea and Mason (2017) to assess the presence and population of turtles at Ashmore Reef. They found that foraging green and hawksbill turtles have had stable numbers, since the surveys had begun in 1994. Other surveys have included tagging nesting green and foraging sub-adult green turtles from 1987 to 2004 (Whiting & Guinea 2005b). The numbers of green turtles in other areas such as Northern Queensland have been influenced by El Niño/Southern Oscillation Events but this is yet to be established for the Ashmore Reef Green turtle population (Guinea & Mason 2017).

Our surveys were not designed to quantify turtle abundance; however, turtles observed on each 2-minute manta tow were counted and identified. Future surveys should further explore other impacts such as plastic debris, climate change (e.g. skewed sex ratios) and non-native species.

### **8.1.6 Dugongs**

Dugongs are listed among migratory marine species under the EPBC Act. The Dugongs at the Ashmore Reef represent the most isolated and probably the least known population of Australian dugongs. The only survey of dugongs at Ashmore Reef was in 1996, more than 20 years ago (Whiting 1999; Whiting 2005; Whiting & Guinea 2005a). The survey by Whiting in 1996 estimated the population at Ashmore Reef was over 100 individuals, but this was based on sighting only 8 dugongs within 6% of the reef area. Whiting (1999) reported dugongs, including calves at Ashmore Reef (12° 15'S, 123° 05'E). The abundance estimates came from an opportunistic flight. However, the logistics and purpose of the survey were unable to accommodate rigorous aerial survey techniques such as those developed by Marsh and Sinclair (1989). Between 1996 and 2005, a further 25 dugong sightings were recorded with the largest group being seven. The dugongs observed have included a mother and calf pair indicating that breeding likely occurs on the reef. Seagrass surveys in 2001 (Brown 2001; Brown & Skewes 2005) concluded that the reef could

sustain a population of 100 or more dugongs, so the 1996 estimate of this population size is plausible. However, due to the location and isolation of the region, this estimate of the dugong population remains uncertain and unsuitable as a basis for management or as a performance indicator. More recent data is mostly limited to observations of customs ships (Hale & Butcher 2013). Surveys of dugong were not undertaken as part of this study and there remains an urgent need for a standardised systematic baseline survey of the population size and structure of dugongs (adults, sub adults and calves) at Ashmore Reef. CSIRO has submitted a proposal to Parks Australia to undertake this work using small vessel and drone survey methods.

### 8.1.7 Fish and sharks

#### Fish

Ashmore Reef has been shown to be similar to oceanic reefs of the Coral Sea with some representation of 'inshore' fish communities (Edgar *et al.* 2017). Ashmore Reef has a high fish species richness, total biomass and large fish biomass (Edgar *et al.* 2017). There are over 750 fish species recorded within the Ashmore Reef Ramsar site including five species of fish listed as threatened by the IUCN (Allen 1993; Russell *et al.* 2005; Hale & Butcher 2013). However, there have been sustained declines in the abundance of several different taxa including reef fishes (Pomacentridae, Labridae, Scarinae and Siganidae), recorded at Ashmore Reef, despite documented increases in coral cover (Hoey & Pratchett 2017).

Six previous surveys of fish communities at Ashmore Reef have been conducted over a period of 32 years (Berry 1993; Skewes *et al.* 1999a; Kospartov *et al.* 2006; Richards *et al.* 2009; Speed, Cappo & Meekan 2018). However, previous studies employed different methods for surveying fish communities, with most studies focussing on comparing fish density and biomass among reefs within the Australian MOU74 region (Ashmore, Cartier, Scott and Seringapatam reefs). Previously collected quantitative data on fish and shark abundance and species composition that are considered most relevant to our current study include Skewes *et al.* (1999b), Richards *et al.* (2009), Edgar and Stuart-Smith (2018) and Speed, Cappo and Meekan (2018). Although locations and survey methodologies vary between surveys, they provide adequate capacity to understand how fish and shark communities have responded to changes in both management and habitat condition since 1998.

#### Sharks

Ashmore Reef has three species of shark listed as threatened by the IUCN (Hale & Butcher 2013). Previous surveys have recorded sharks in low densities (less than one per hectare) (Skewes *et al.* 1999a; Richards *et al.* 2009; Heyward *et al.* 2012). Despite protection status across most of the reef from 1988, illegal fishing continued at Ashmore Reef up until 2008, and this was thought to be the main cause of the low abundances of sharks recorded during previous studies (Skewes *et al.* 1999a; Richards *et al.* 2009; Speed, Cappo & Meekan 2018).

Declining populations of shark populations are of concern as there is increasing evidence for their important trophic role (Heithaus *et al.* 2010; Ruppert *et al.* 2013; Heupel *et al.* 2014; Roff *et al.* 2016). The presence of sharks have been shown to affect the food chain structure (Barley, Meekan & Meeuwig 2017). Ruppert *et al.* (2013) also found that the absence of predators may impact the ability of coral reefs to recover from impacts such as cyclones and bleaching. However, recent evidence indicates densities of grey reef sharks (*Carcharhinus amblyrhynchos*) may currently be

increasing at Ashmore Reef (Speed, Cappel & Meekan 2018). However, the rate of increase remains debatable (Guinea 2020).

## 8.2 Objectives of the 2019 marine survey

The objective of the 2019 survey was to undertake a comprehensive survey program of the coral reef communities at Ashmore Reef, with a specific focus on trochus, giant clams, holothurians, sea snakes and fishes. The percent cover of benthic groups was assessed including corals, macroalgae and seagrass across major habitats at Ashmore Reef. The survey methods used have been adapted using a combination of globally standardised Underwater Visual Census (UVC) methods (English, Wilkinson & Baker 1997).

Using standard UVC methodology (English, Wilkinson & Baker 1997; DeVantier *et al.* 1998) we assessed the abundance of fish, sharks and sea-snakes across major habitats at Ashmore Reef. Minor adaptations were made omitting secretive families such as gobies, blennies and moray eels that cannot be sampled accurately with the UVC method, reducing the number of survey divers needed.

We used survey methods and recommendations made by Hosack and Lawrence (2013a) and (Ceccarelli *et al.* 2013) to assess the abundance and current status of holothurians, trochus and clams which have been historically overfished at Ashmore Reef.

To further assess coral health at designated reef locations, quantitative fine-scale diver surveys were conducted using methodology consistent with the Australian coral bleaching task force.

## 8.3 Research methods by operations type

### 8.3.1 Habitat stratification

To maximise the likelihood of obtaining accurate estimates of the abundance of target groups, sampling was stratified into reef top and reef edge strata according to Hosack and Lawrence (2013a) and recognising five major habitat types; reef slope, reef edge, reef flat, lagoon sand and lagoon intertidal (Figure 57). In some cases, this was further subdivided (e.g. east lagoon, west lagoon, channel, reef flat and sand flat) where warranted to describe patterns of biota or to permit comparisons with previous studies. The habitat categories were adapted from Skewes *et al.* (1999b) Hosack and Lawrence (2013a) and includes the modifications to add a north and south reef edge habitat (reef edge: red dots in Figure 57) in 2013 by Ceccarelli *et al.* (2013).

## Ashmore Reef

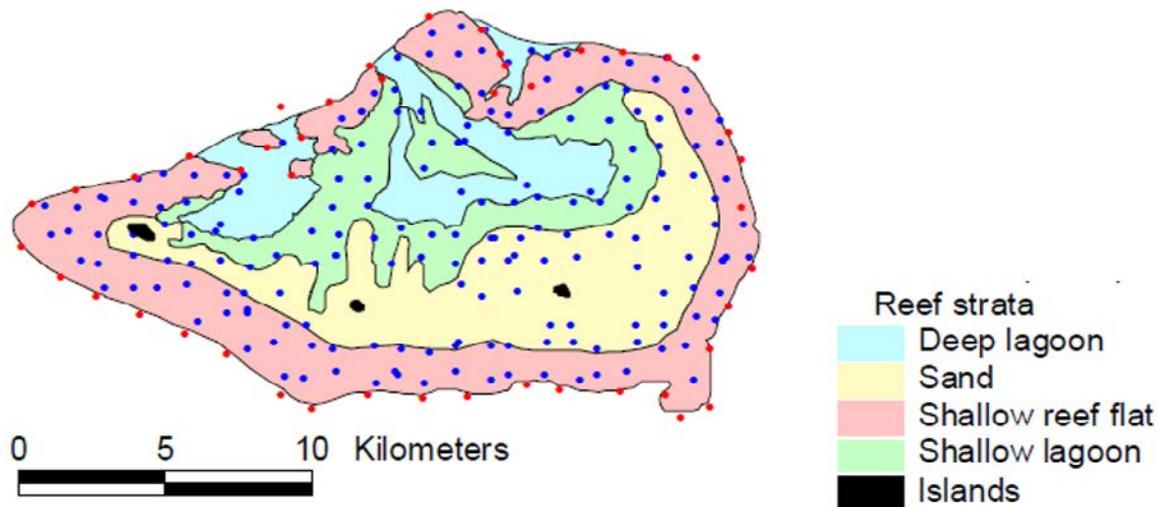


Figure 57. Distribution of habitats at Ashmore Reef adapted from Skewes et al. (1999b) and Hosack and Lawrence (2013). In addition to the habitats shown, a fifth marine habitat (reef edge: red dots) was added to the 2013 (Ceccarelli et al. 2013) and 2019 surveys.

### 8.3.2 Transect site selection

A total of 216 sites (Figure 58) were selected based on previous studies (Skewes *et al.* 1999b; Ceccarelli *et al.* 2013; Hosack & Lawrence 2013a). The detailed methodology for site selection is given in Chapter 12.

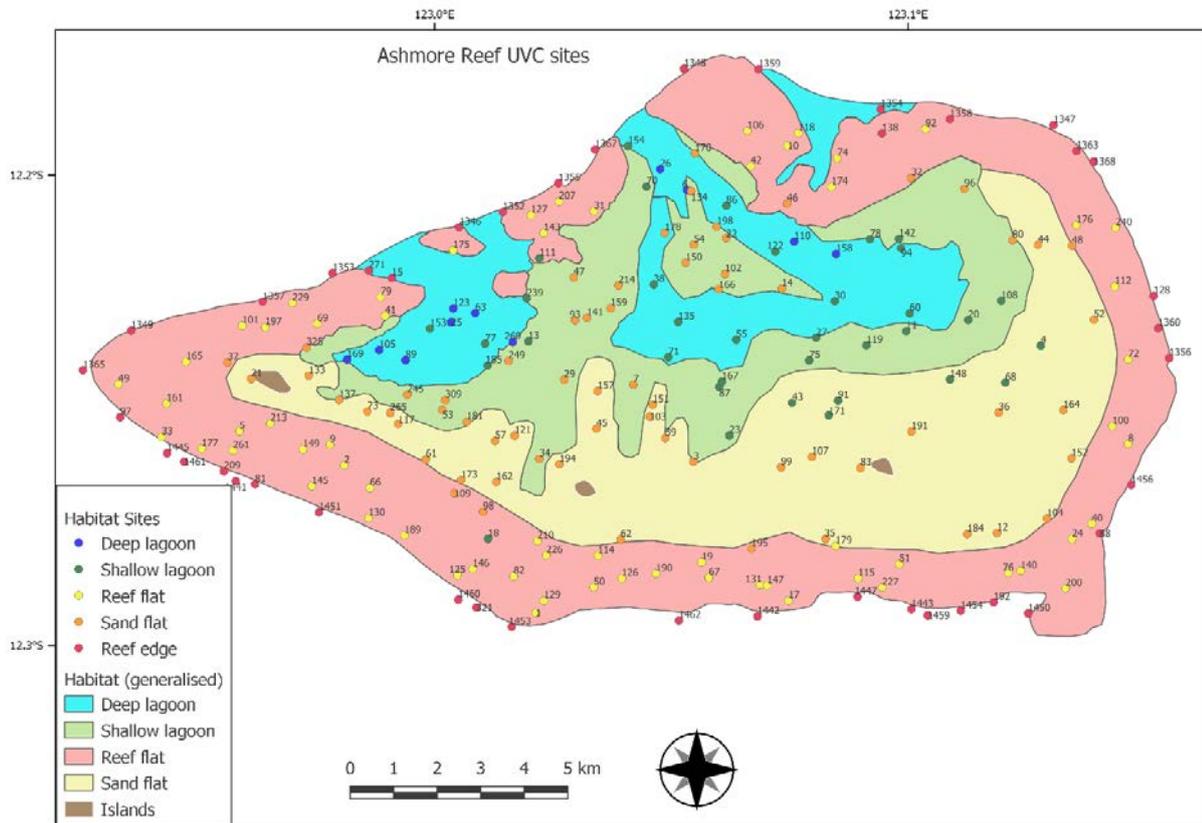


Figure 58. Location of 216 UVC sites surveyed during 2019 Ashmore Reef surveys.

### 8.3.3 Fish, invertebrate and benthic cover transect methodology

At each of 216 sites, a 50 m transect was marked out using either survey tape or a 50 m length of biodegradable cotton dispensed via a Hip Chain distance measurer ([https://www.forestry-suppliers.com/product\\_pages/products.php?mi=57081](https://www.forestry-suppliers.com/product_pages/products.php?mi=57081)). At intertidal and shallow reef flat sites, transect direction was random from the site location, while at deeper edge sites, transects were set out along the reef slope following a consistent depth contour, to ensure a depth of 9 m or less. Each transect was surveyed by two divers using either SCUBA or snorkel. The first diver laid out the transect counting all large mobile fish within a 5 m (2.5 m either side of cotton) belt transect, sharks and rays within a 10 m belt transect (5 m either side) and estimated fish size to nearest 5 cm. The second diver followed some distance behind, taking photos of the benthos to record benthic cover every 1 to 1.5 m along the 50 m transect using a Canon Powershot camera held 1 m above the seabed. When the second diver reached the end of the 50 m transect, the first diver swam back along the transect counting and estimating the size of smaller, site-attached species (mostly pomacentrids and small labrids) in a 2 m wide belt transect (1 m either side). Fish species of a cryptic, nocturnal or pelagic nature were omitted, as underwater visual surveys cannot produce reliable estimates of these taxa. The second diver followed some distance behind, recording all live echinoderms, trochus, giant clams and pearl oysters within a 2 m belt transect (1 m each side). Any of these invertebrate taxa observed outside the 2 m transect width were also noted and recorded separately. Measurements of invertebrates were limited to trochus and some

giant clam species. On 14 of the 216 sites surveyed, fish were not counted as they were surveyed by divers without sufficient fish expertise.

### 8.3.4 Manta Towing method

Manta tows were conducted along large sections of the perimeter and main lagoon edge at Ashmore Reef (Figure 59) using the methodology of Miller, Jonker and Coleman (2009). Sites were selected based on previous research using the same methods (Guinea 2007; Guinea 2013). Observations were made in 2-minute blocks of time, recording aspects of the habitat including live hard coral and live soft coral cover (percentage), recently dead coral or evidence of coral bleaching. Counts and identifications were made of crown of thorns starfish, holothurians, giant clams, sharks, sea snakes and turtles. Surveys were not designed to quantify turtle abundance, however turtles observed on each 2-minute manta tow were counted and identified. A GoPro camera was fixed to the underside of the manta board to record the habitat type covered in each section. Generally, tows followed the outer reef margin, off the crest, in about 3 to 8 m depth depending on tide and habitat type. Tows undertaken within the lagoon were made along the edge of the lagoon and attempted to cover the perimeter of any bommies situated at the edge of the lagoon.

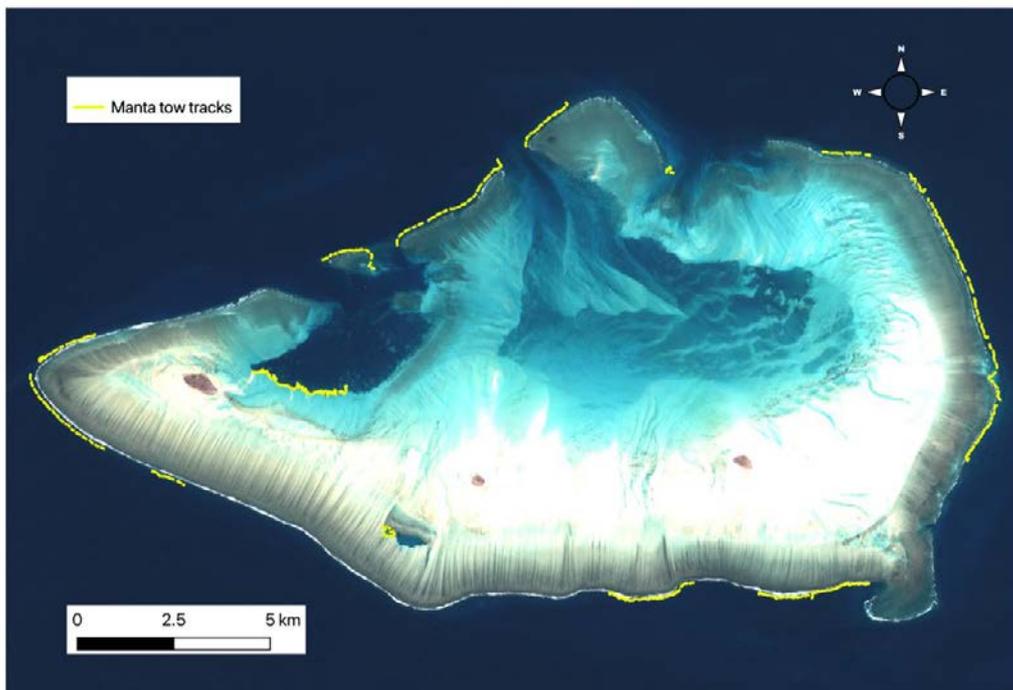


Figure 59. Manta tow tracks (yellow) undertaken during Ashmore Reef during 2019 surveys.

# 9 ASHMORE REEF: CORALS AND OTHER BENTHIC BIOTA, CORAL BLEACHING AND OTHER ASPECTS OF CORAL HEALTH

Emma Westlake, John Keesing, Ylva Olsen, Margaret Miller, Michael Haywood, Russ Babcock, Cindy Bessey, Christopher Doropoulos, Lauren Hardiman and Mark Tonks and Daniella Ceccarelli

## 9.1 Abstract

Ashmore Reef is an exposed open-ocean platform reef approximately 350 km from the Kimberley coastline. Due to its offshore location away from mainland influences and exposure to clear oceanic waters, wide tidal ranges and strong wave action, Ashmore Reef contains some of the highest coral diversity on the WA coast. Photo quadrat and manta tow surveys were conducted at 216 sites across five habitat types. Benthic communities were dominated by hard coral and algae. Coral cover was highly variable across sites and habitats. The reef edge showed the greatest proportion of coral (hard coral  $23.7\% \pm 2$  s.e., soft coral  $8.2\% \pm 2.5$ ) and sponge cover ( $1\% \pm 0.2$ ), with coral cover highest on the southern edge (mean  $32.2\%$ , range  $14.2$  to  $70\%$ ) of the atoll. Live hard coral cover was dominated by *Acropora*, *Pocillopora*, *Porites*, and Faviidae corals, with cover of each of these greatest in reef edge habitats. *Acropora* spp. accounted for half of all live hard coral cover ( $50.1\% \pm 0.6$ ) and approximately 4% of total benthic assemblages ( $3.9\% \pm 0.6$ ). No coral bleaching or disease was observed. Although 9.1% lower than 2009 surveys, coral cover was 10.1% greater than that observed during 2005 surveys. These differences are potentially due to bleaching events in 2010 and 2016/17, however may have resulted from the imperfect nature of comparisons between studies. Overall, Ashmore Reef was found to support typical coral reef benthic habitats with assemblages appearing normal and no signs of severe stress or major recent disturbance were evident.

## 9.2 Introduction

Within a regional context, the coral communities of Ashmore Reef are considered among the highest for biological diversity (Ceccarelli *et al.* 2011b). Due to its remote geographical location away from urban centres and mainland influences, and characterised by deep clear oceanic waters, wide tidal range, and exposure to strong wave action on outer slopes (Veron 1986), Ashmore Reef contains high diversity of marine life, including hard and soft corals (Richards *et al.* 2013), and supports distinct assemblages of benthic and pelagic communities (Commonwealth of Australia 2002). With 275 species from 14 families and 51 genera (Ceccarelli *et al.* 2011b) Ashmore Reef has been found to support the greatest number of reef-building coral species of any reef area on the West Australian coast (Veron 1993) and a greater abundance of soft corals and algae than similar reefs in the north-west bioregion (Edgar *et al.* 2017).

Over the last decade, numerous marine surveys have been conducted on the benthic communities of Ashmore Reef (Skewes *et al.* 1999b; Commonwealth of Australia 2002; Rees *et al.* 2003; Russell, Neil & Hilliard 2004; Richards *et al.* 2009; Heyward *et al.* 2010; Ceccarelli *et al.* 2011b). Each has found high variability in coral cover and benthic community assemblages between sites and across habitat types. Following bleaching associated with abnormally high sea surface temperatures during the austral summer in 1998 (Skewes *et al.* 1999b), 2003, 2010 (Heyward *et al.* 2010), and again in 2016/2017 (Gilmour *et al.* 2019), benthic communities at Ashmore Reef have nevertheless remained dominated by hard corals and turf algae, and sand in lagoonal areas (Skewes *et al.* 1999b; Heyward *et al.* 2010). Hard and soft corals have been observed across multiple habitat types including reef slope, reef crest and lagoonal bommies. Reef crest and slope habitats have been dominated by algal communities consisting predominantly of turf and crustose coralline algae. Reef flats have been striated with coral rubble, while sand flats were extensive and mobile, comprised of carbonate sands (Commonwealth of Australia 2002).

Low densities of coral predators (*Drupella* spp. and *Acanthaster planci*) and a low incidence of coral disease (Richards *et al.* 2009; Wilson *et al.* 2009) has been reported for Ashmore Reef. Combined with the establishment of juvenile corals following significant coral loss associated with the 1998 and 2003 coral bleaching events (Ceccarelli *et al.* 2011b; Gilmour *et al.* 2019), benthic communities at Ashmore Reef underwent rapid and sustained recovery, suggesting a high resilience of coral communities potentially due to its isolation and protection from anthropogenic disturbances (Ceccarelli *et al.* 2011b) such as pollution, nutrient runoff and habitat modification (Pratchett *et al.* 2011).

## 9.3 Methods

### 9.3.1 Habitat stratification and site selection

Benthic community assemblage types and cover were determined at 216 sites. These sites were stratified among five habitat types:

1. reef edge – upper slope at 9 m depth and not including crest surf zone
2. shallow intertidal reef flat
3. shallow intertidal lagoon sand flat
4. shallow subtidal lagoon
5. deep lagoon/bommie habitats.

Distribution of habitat types and survey sites is shown in Figure 57 and Figure 58.

### 9.3.2 Coral and other sessile animal diversity and cover

Manta tows were used to estimate coral cover on a macroscale, while photo quadrats collected during belt transects (see section 8.3.2) were used to determine benthic cover (percent live hard/soft coral, other benthic biota) at a finer scale. Post-hoc image analysis was conducted whereby a subset of 40 images were chosen at random from the photo quadrats collected for each transect. For transects that had fewer than 40 images available, all available images were analysed. Six points were overlaid on each image (3 x 2 grid) and taxa directly underneath

identified using Transect-Measure™ (SEAGIS) software. This also included abiotic substrates (sand, rubble, silt, etc.). Taxa were identified to species where possible, otherwise to the highest taxonomic level. Hard and soft corals were identified to genus and functional form described (e.g. corymbose *Acropora*, foliose non-*Acropora*, branching *Pocillopora*, etc.). Any coral bleaching or recently dead coral was also recorded using this method.

### 9.3.3 Seagrass and algal diversity and cover

Seagrass, macroalgal cover and diversity was determined at 216 sites spread among each of the five major habitat types at Ashmore Reef (reef edge, deep lagoon, shallow lagoon, intertidal sand flat and intertidal reef flat) Figure 57 using the same methods as described above for coral (see section 8.3.2). Percent macrophyte cover was estimated by analysing images overlaid with six points in the software Transect-Measure™ (SEAGIS) and the taxa underneath each point identified to the highest taxonomic resolution possible for macrophytes. Using the CATAMI classification scheme, algal assemblages were identified to genus or to highest taxonomic level where taxa could not be reliably identified (Althaus et al., 2015).

## 9.4 Results

### 9.4.1 Fine-scale benthic cover – 2019 photo quadrats

Coral cover was highly variable among sites and habitat types (Figure 60). Reef edge habitats showed the greatest proportion of hard coral ( $23.7\% \pm 2$  s.e.), soft coral ( $8.2\% \pm 2.5$ ) and sponge cover ( $1\% \pm 0.2$ ). Hard coral cover was greatest on the reef edge compared to the other four habitat types and was generally highest on the southern/southwestern reef edge (mean  $32.2\%$ , range  $14.2$  to  $70\%$ ) with cover of  $40$ - $60\%$  observed at some sites (Figure 61A). Deep lagoonal habitats showed the second highest cover of hard coral ( $11.2\% \pm 4.6$ ), predominantly within the western lagoon. Soft coral cover was highest on the north-western reef edge ( $10.5\%$ , range  $1.3$  to  $19.6\%$ ) followed by the south/southwestern reef edge (mean  $9.7\%$ , range  $0.8$  to  $32.1\%$ ) with cover of  $15$ - $35\%$  observed at these sites (Figure 61B). Benthic cover of both hard and soft coral was negligible within reef flat, sand flat and shallow lagoonal habitats. No coral bleaching or disease were observed. Although patchy, sponge distribution was greatest along the southern reef edge and deep lagoon with up to  $10\%$  cover observed at sites within these habitats (Figure 61C).

Abiotic (dead coral, rubble, sand and silt) and algal cover dominated benthic types across the five major habitats (Figure 60). Sand flat and shallow lagoonal habitats showed highest abiotic cover ( $89.6\% \pm 2.1$  and  $64.3\% \pm 5.4$  respectively), largely consisting of sand, while deep lagoon, reef edge and reef flat habitats were dominated by algae including crustose coralline, turf and macroalgae.

Seagrass cover was generally low. The highest cover was  $2.8\% \pm 0.7$  on the reef flat and was more sparsely distributed across the remaining habitat types (Figure 60).

Algal cover was greatest on the reef flat ( $49.8\% \pm 3.6$  s.e.), reef edge ( $48.3\% \pm 2.2$ ) and deep lagoon ( $41.2\% \pm 5.9$ ) (Figure 60). The high algal cover within these habitats was largely due to turf algae (mean  $32.2\%$ , range  $9.1$  to  $47.8\%$ ). While turf algae dominated algal cover in all habitat types (Figure 62), crustose coralline algal cover was greatest on the reef edge ( $5.5\% \pm 0.6$ ). Macroalgal

cover was greatest on the reef edge ( $2.7\% \pm 0.8$ ) and within the shallow lagoon ( $2.6\% \pm 0.7$ ) with 20-25% cover observed at some sites Figure 62).

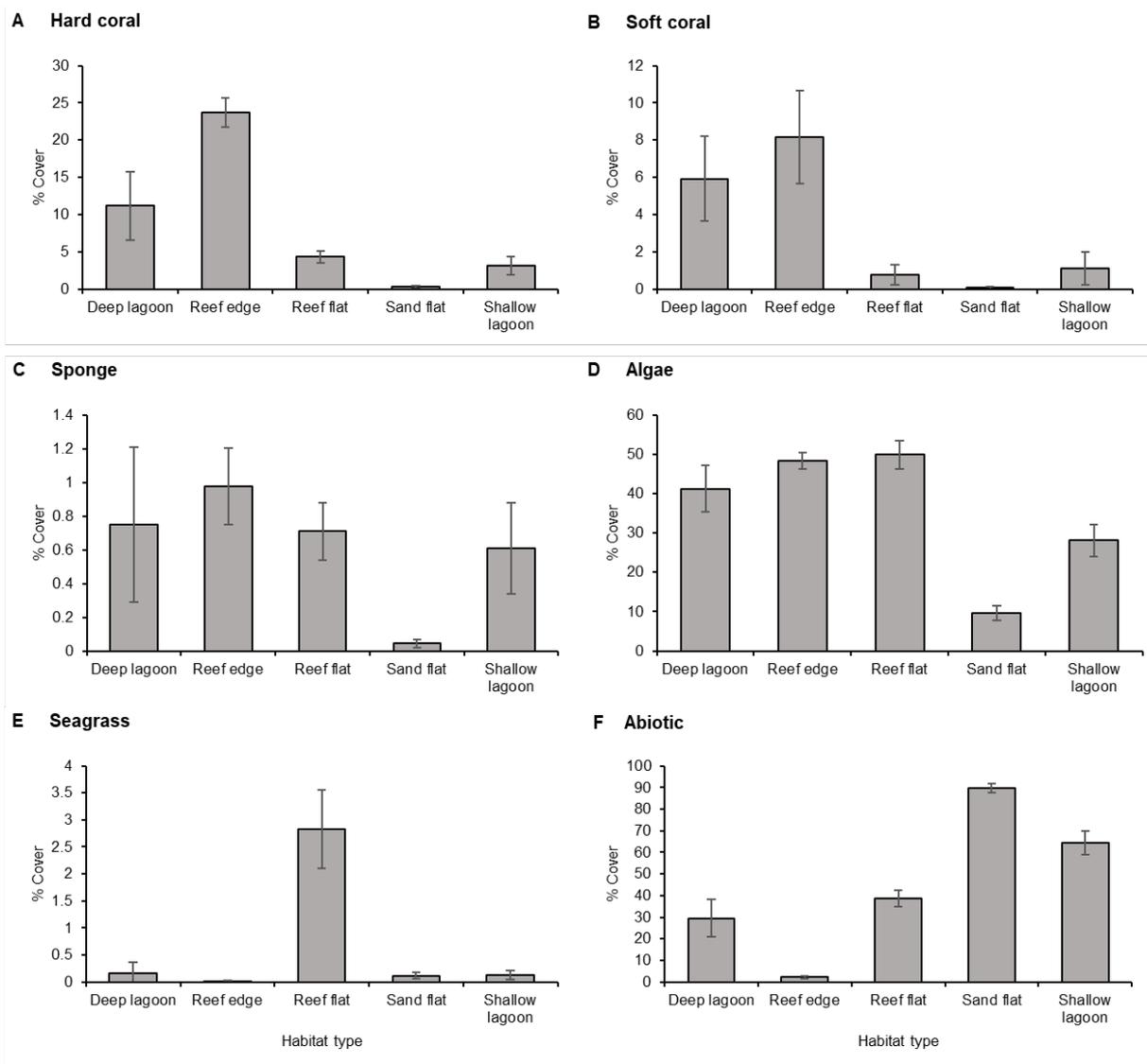
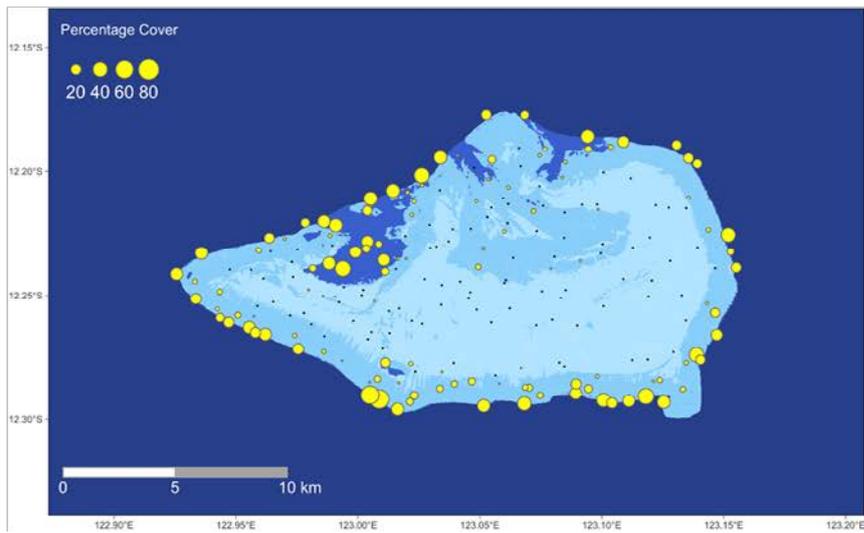
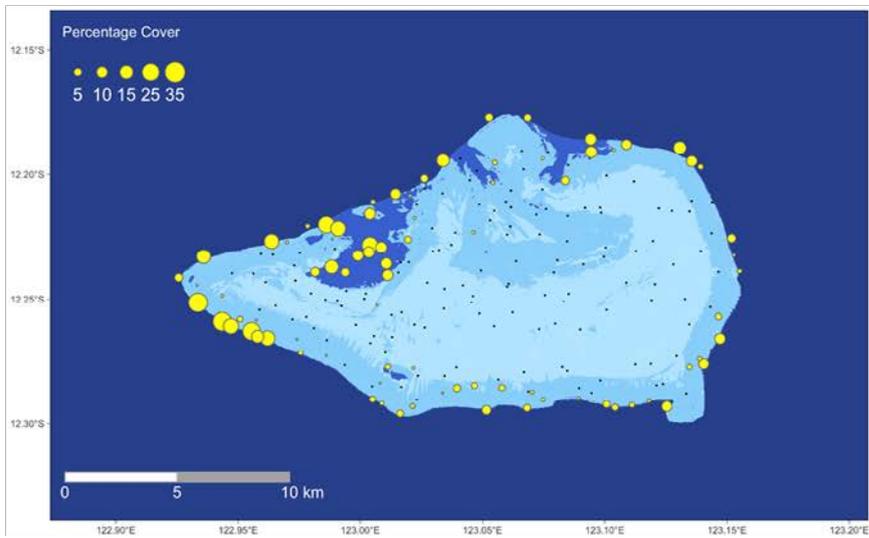


Figure 60. Percent cover of major benthic assemblages across the five habitat types at Ashmore Reef, 2019. Error bars indicate  $\pm 1$  s.e.

**A Hard coral**



**B Soft coral**



**C Sponge**

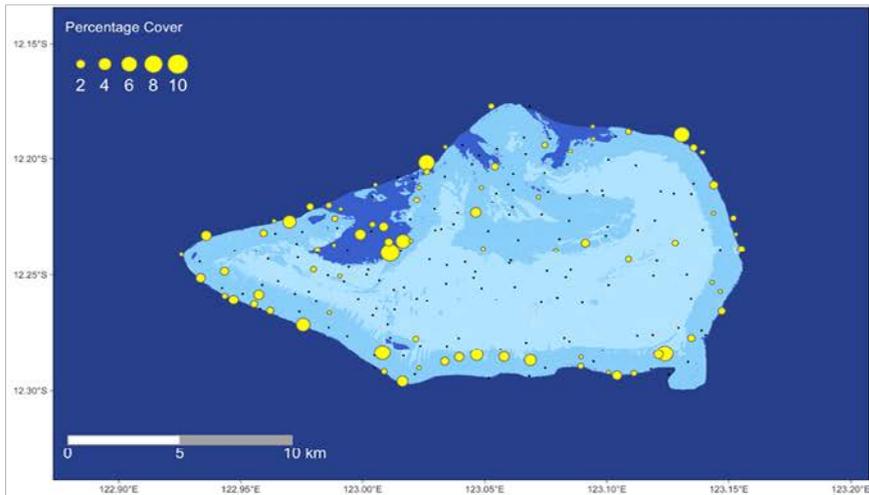
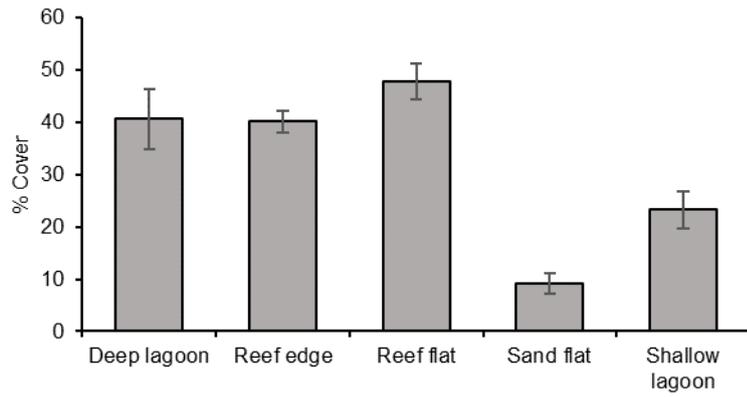
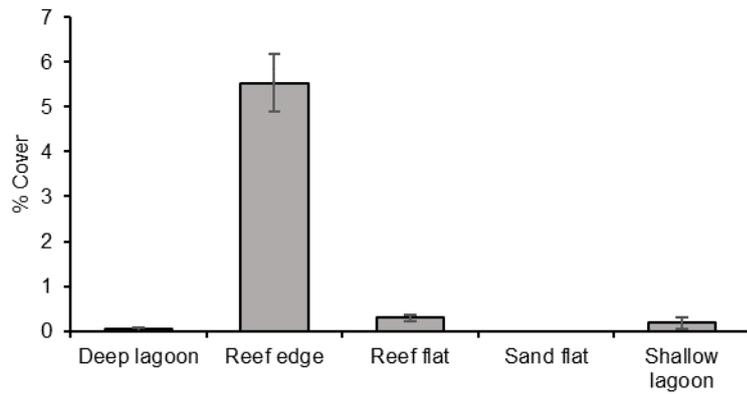


Figure 61. Percent cover of A) hard coral, B) soft coral, and C) sponges across the 216 sites surveyed at Ashmore Reef in 2019

### A Turf algae



### B Crustose coralline algae



### C Macroalgae

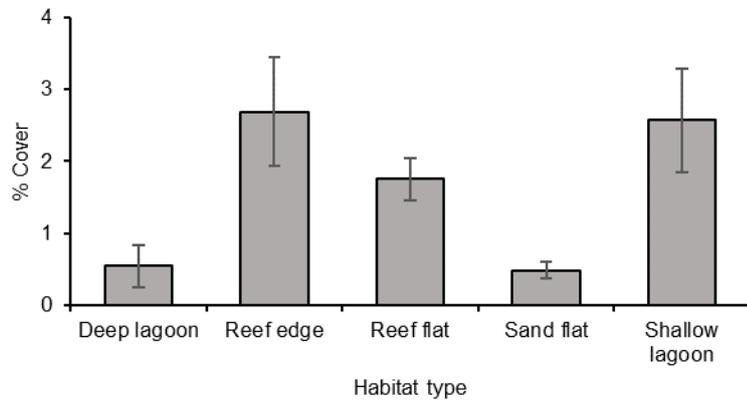


Figure 62. Percent cover of the three major algal types at Ashmore Reef: A) turf algae, B) crustose coralline algae, and C) macroalgae. Error bars indicate  $\pm 1$  s.e.

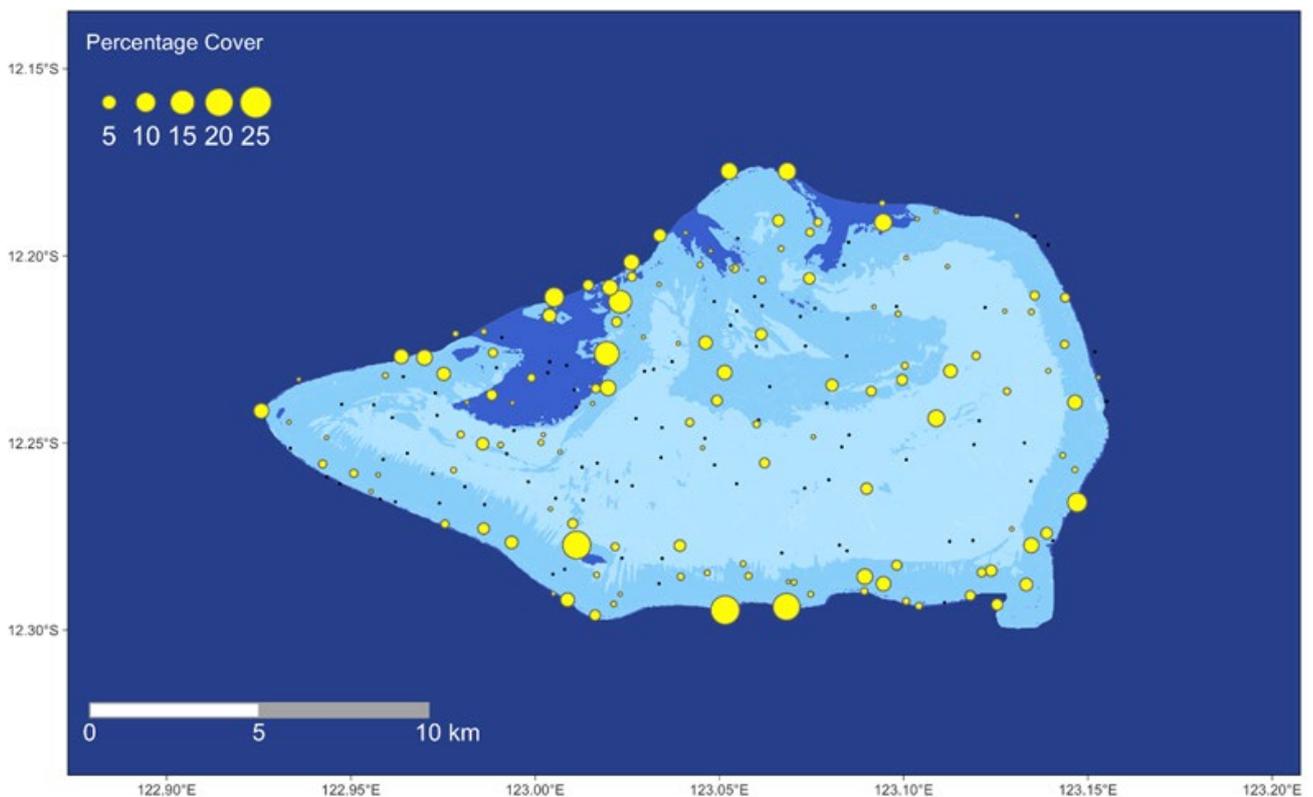


Figure 63. Percent cover of macroalgae across the 216 sites surveyed at Ashmore Reef in 2019.

*Acropora*, *Pocillopora*, *Porites*, and Faviidae corals dominated the scleractinian coral community. *Acropora* spp. accounted for half of all live hard coral cover ( $50.1\% \pm 0.6$  s.e.) and approximately 4% ( $3.9\% \pm 0.6$ ) of total benthic assemblages. *Porites* spp. comprised almost one quarter of live coral cover ( $22.1\% \pm 0.2$ ) and 1.7% ( $\pm 0.2$ ) of all benthic assemblages. Both *Pocillopora* and Faviidae spp. each accounted for approximately 10% ( $10.5\% \pm 0.1$  and  $11\% \pm 0.1$ ) of live hard coral cover.

*Acropora*, *Pocillopora*, *Porites*, and Faviidae corals were all found in the highest abundances in reef edge habitats (Figure 64). *Acropora* species accounted for almost 40% of all live hard coral cover within the reef edge ( $39.5\% \pm 0.7$ ), predominantly corymbose, bottlebrush and branching *Acropora* growth forms. Both *Pocillopora* and Faviidae coral cover was approximately four times greater on the reef edge than other habitats ( $7.2\% \pm 0.23$  and  $6.9\% \pm 0.4$ ). Hard coral cover in reef flats was dominated by *Porites* ( $7.5\% \pm 0.4$ ).

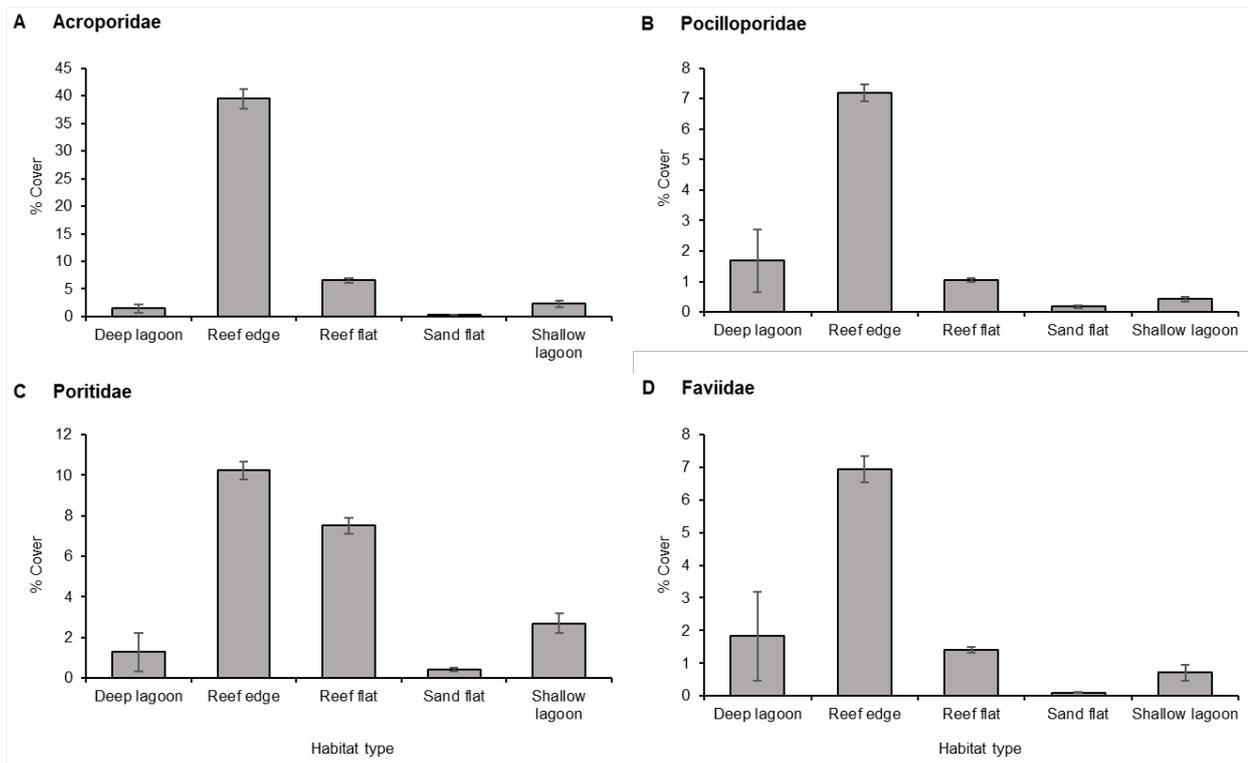


Figure 64. Percent cover of main hard coral genera contributing to overall hard coral cover across the five habitat types at Ashmore Reef: A) Acroporidae, B) Pocilloporidae, C) Poritidae, and D) Faviidae. Error bars indicate  $\pm 1$  s.e.

#### 9.4.2 Broad scale coral cover – 2019 manta tows

Manta tows were conducted along reef slope habitats around Ashmore Reef. A large proportion of the central-southern slope was not surveyed during 2019. Estimates of hard coral cover were variable and generally  $> 20\%$  except for along the north-eastern edge (Figure 65). Live hard coral cover was greatest on the southern reef edge where cover was estimated at 60-85%. Hard coral cover within the western lagoon was estimated at 20-60%. While cover of around 40-60% was observed along the south-eastern reef edge, hard coral cover on the north-eastern part of this reef edge was minimal, generally between 0-20%.

Estimates of live soft coral cover were also variable (Figure 66). Areas of higher soft coral cover were along the northern and south-western reef edges, and west lagoonal areas (4-10%). A notable patch on the southern edge was also evident (4-10%). Little to no soft corals were observed along the eastern reef edge.

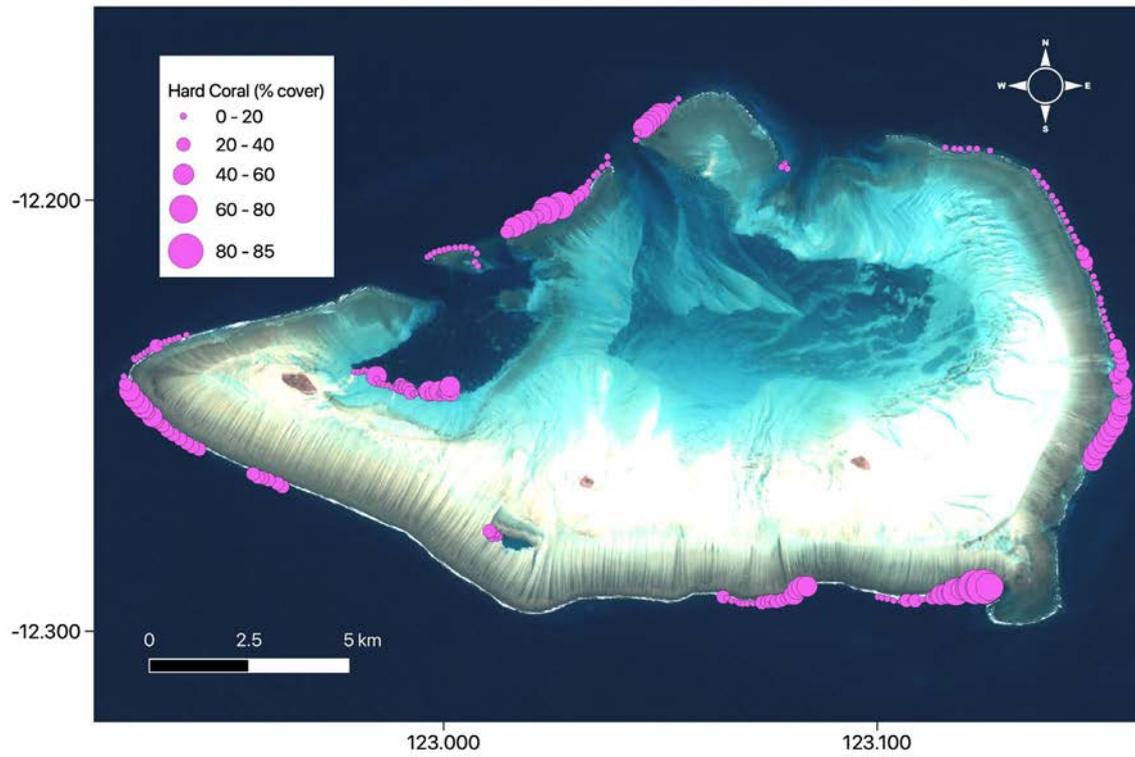


Figure 65. Estimates of percent cover of hard corals from manta tow transects at Ashmore Reef

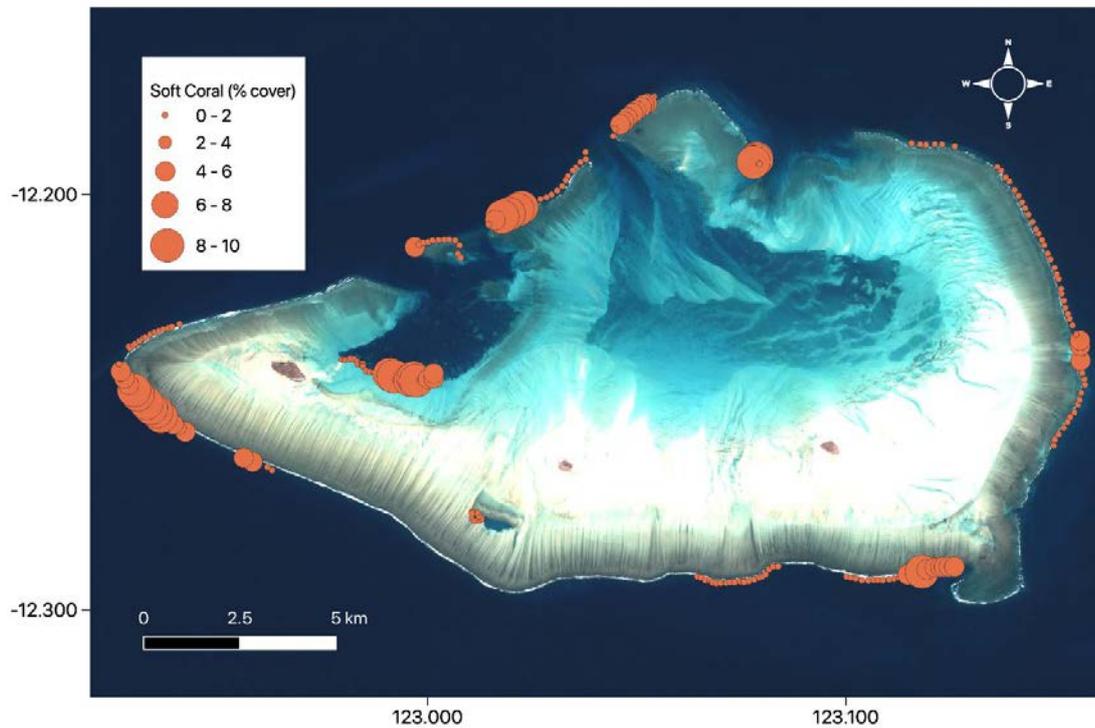


Figure 66. Estimates of percent cover of soft corals from manta tow transects at Ashmore Reef

### 9.4.3 Changes in coral cover over time

To compare live hard and soft coral cover over the last 15 years, a subsample of 2019 survey sites was chosen and compared with 2005 (Kospartov *et al.* 2006) and 2009 (Ceccarelli *et al.* 2011b) data. Latitudinal and longitudinal data from these previous surveys were used and 2019 sites were chosen and grouped based on proximity to these, resulting in six sites (Figure 67). In 2005, a total of 12 sites were sampled, while in 2009 these were merged into six sites, with three replicates at each site for both years. Sites were divided in depth ranges: 2–5 m and 6–8 m in 2019, and 3–5 m and 8–10 m in 2005 and 2009 (Table 22).

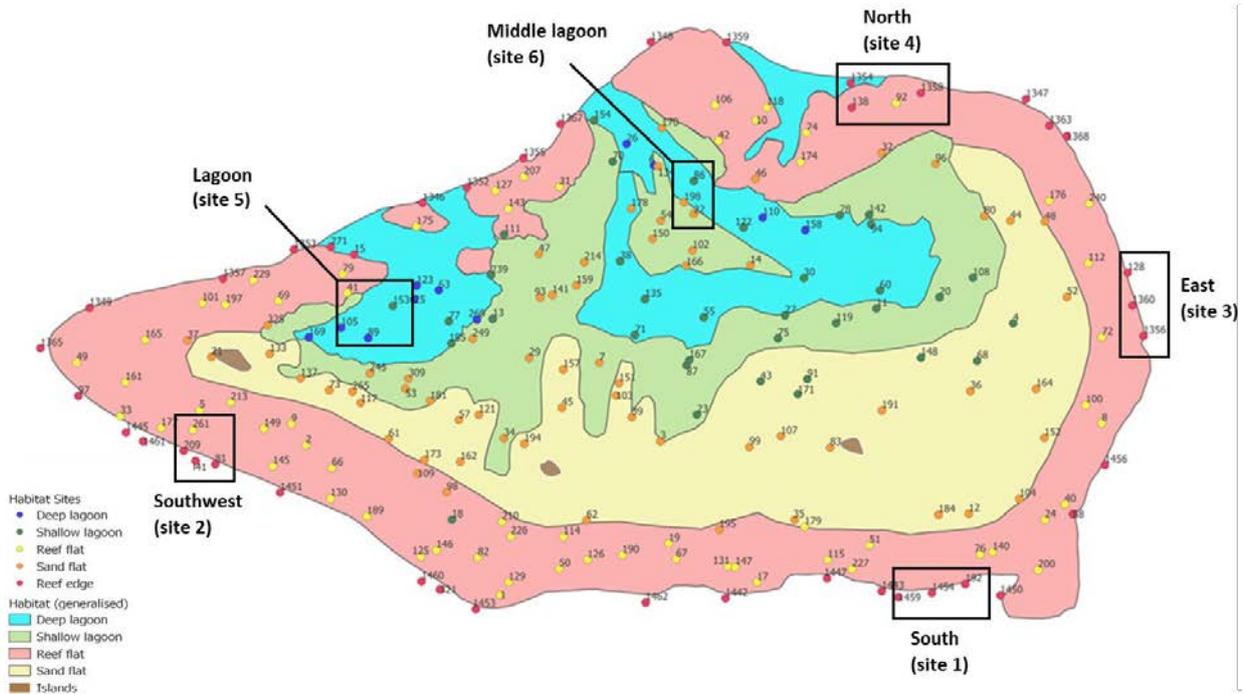


Figure 67. Map showing site groupings in 2019 for comparison with previous studies at Ashmore Reef in 2005 (Kospartov et al. 2005) and 2009 (Ceccarelli et al. 2011).

Table 22. Site groupings at Ashmore Reef in 2019 and equivalent site numbers from 2005 (Kospartov et al. 2006) and 2009 (Ceccarelli et al. 2011).

Combined site name and number	2019 site number	2019 habitat	2019 depth	2005 sites (Kospartov et al. 2005)	2009 sites (Ceccarelli et al. 2011)	2009 habitat
South (1)	192	Reef edge	8	D1, D2	1	Exposed
	1454	Reef edge	4			
	1459	Reef edge	5			
Southwest (2)	81	Reef edge	9	D3, D4	2	Exposed
	1441	Reef edge	8			
	209	Reef edge	7			
East (3)	128	Reef edge	7	D5, D6	3	Sheltered
	1360	Reef edge	3			
	1356	Reef edge	4			
North (4)	1358	Reef edge	8	D9, D10	4	Sheltered
	92	Reef flat	4			
	138	Reef edge	4.2			
Lagoon (5)	1354	Reef edge	6	D7, D8	5	Lagoonal
	41	Reef flat	3			
	105	Deep lagoon	8			
Middle lagoon (6)	89	Deep lagoon	6.5	D11, D12	6	Lagoonal
	153	Shallow lagoon	3			
	86	Shallow lagoon	4			
	198	Sand flat	7			
	22	Sand flat	4			

Live hard coral cover declined from 29.4% ( $\pm 1.8$  s.e.) in 2009 to 20.3% ( $\pm 3.4$ ) in 2019 (Figure 68). This was double that in 2005 when hard coral cover was 10.2% ( $\pm 1.5$ ). The percent of soft coral cover also declined from 8.3% ( $\pm 1.4$ ) in 2009 to 6.7% ( $\pm 1.6$ ) in 2019, however was higher than the 2005 where it averaged 4.5% ( $\pm 0.6$ ).

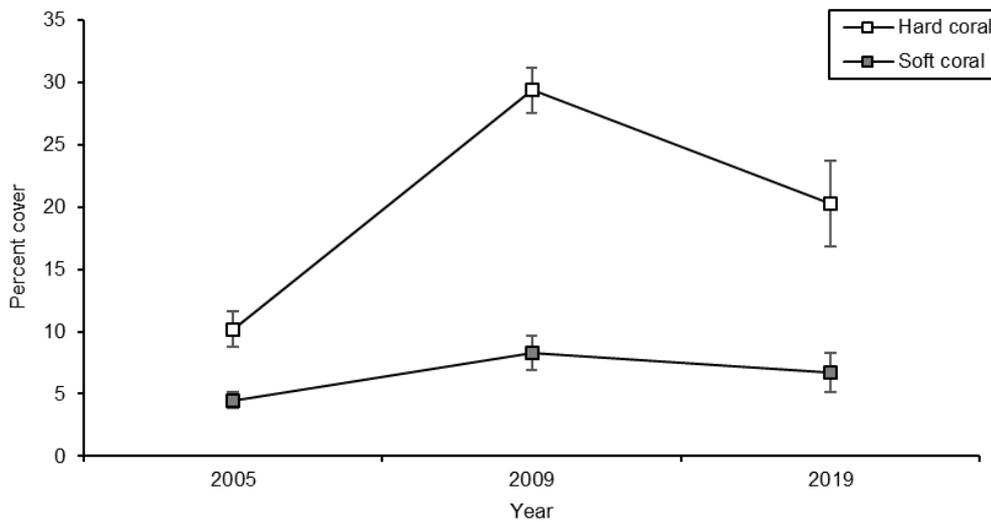


Figure 68. Change in mean percent cover of live hard and soft corals from 2005, 2009 and 2019 benthic surveys. Data from 2005 from Kospartov et al. (2006), data from 2009 from Ceccarelli et al. (2011). Error bars indicate  $\pm 1$  s.e.

Coral cover was variable among sites and years (Figure 69). In 2019, live hard coral cover was greatest at south sites across both depth ranges. In contrast, highest hard coral cover in 2005 was observed at deep and shallow North reef sites. In 2009, shallow coral cover was highest at the South site while deeper coral cover was observed at the North site.

In 2019, mean cover of live hard corals at the six comparable grouped sites ranged from 0% at the deep Middle lagoon sites to 49.6% at the southern deep reef edge site. In 2009, mean cover of hard corals at the six sites surveyed ranged from 13.6% ( $\pm 6$  s.e.) at the shallow Middle lagoon site to 41% ( $\pm 1.2$ ) at the northern deep reef slope site. Percent hard coral cover was generally greater across all sites and depth ranges in 2009 than 2005 and 2019.

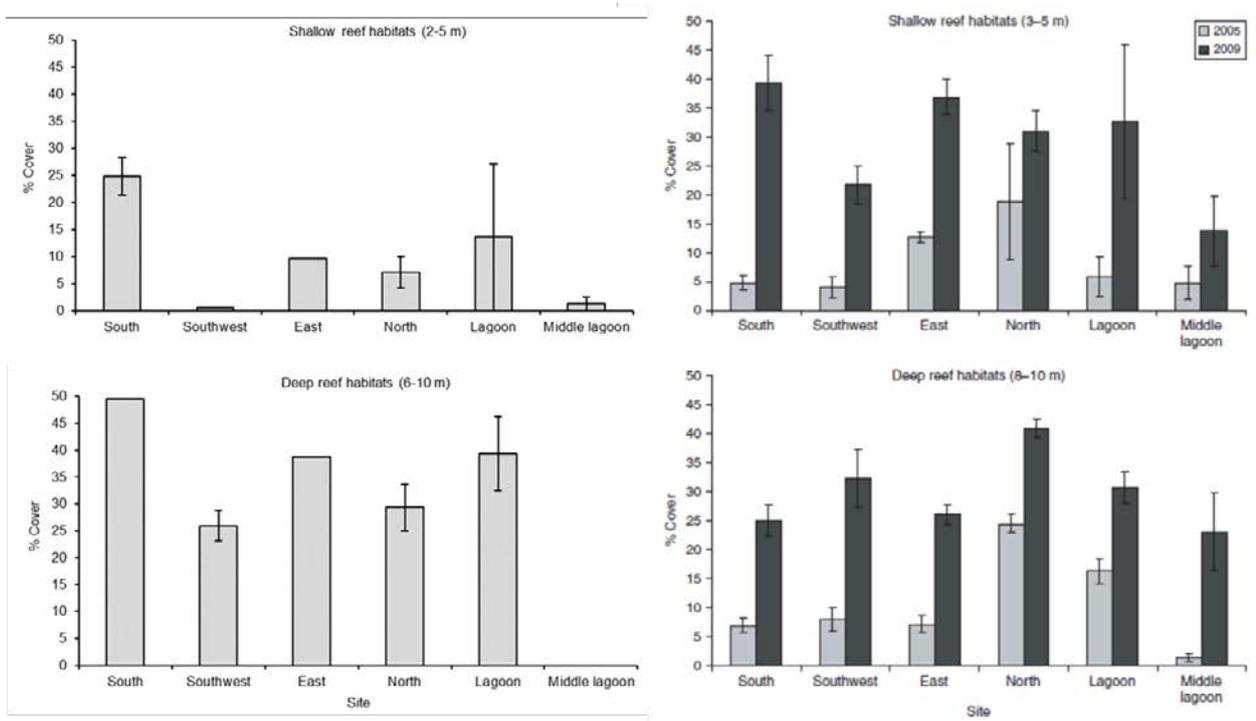


Figure 69. Temporal comparison of mean percent cover of live hard coral during 2019 benthic surveys (left panels) and 2005 (Kospartov et al. 2006) and 2009 (Ceccarelli et al. 2011) benthic surveys (right panels) for shallow (top panels) and deep (bottom panels) reef habitats. Error bars indicate  $\pm 1$  s.e. Right panels are copied from Ceccarelli et al. 2011.

## 9.5 Discussion

Visual assessment of benthic assemblages has been widely used to categorise and evaluate marine habitat composition (Sheppard *et al.* 2002; Long, Andrews & Suharsono 2004; e.g. Osborne *et al.* 2011). Specifically, percent cover of live coral is the most common measure of coral reef health and is used to document trends in the abundance of living corals, reef decline and recovery (Bruno & Selig 2007; Ceccarelli *et al.* 2011b; Sweatman, Delean & Syms 2011).

### 9.5.1 Fine and broad-scale benthic cover

In 2019, variation in coral cover and overall benthic community structure at Ashmore Reef was observed between sites and habitat type. This is consistent with previous studies (Skewes *et al.* 1999b; Rees *et al.* 2003; Kospartov *et al.* 2006; Richards *et al.* 2009; Heyward *et al.* 2010; Ceccarelli *et al.* 2011b) with such variation likely a reflection of the environmental conditions a habitat is exposed to and the tolerance and preferences of each coral and algal taxon to these.

Hard coral and algal assemblages dominated live benthic communities with highest live hard coral cover observed in southern reef edge and deep lagoonal habitats. Soft coral cover was also greatest in reef edge habitats. The reef edge, like all exposed outer slopes, is subjected to strong wave action, increasing depth, decreasing light exposure and clear oceanic waters (Alevizon *et al.* 1985; Rees *et al.* 2003). These conditions can result in higher coverage of soft corals and important reef-building hard corals including *Acropora* and *Pocillopora* that contribute to reef complexity and structure.

Abiotic (dead coral, rubble, sand and silt) and algal cover dominated benthic types at Ashmore Reef. Edgar *et al.* (2017) reported similar findings, with Ashmore Reef having more abiotic components, turf and macroalgae than neighbouring reefs. In contrast to the reef edge, reef flat habitats were dominated by algal communities and seagrass cover was highest in this habitat than any other. Due to this area being the shallowest submerged portion of the reef, and its exposure to high solar irradiance (Fulton & Bellwood 2005), algal growth and primary productivity is generally greatest in these habitats (Barnes & Devereux 1984). Abiotic benthic cover was greatest within sand flat and shallow lagoonal habitats. Such inshore habitats are generally sheltered from waves and strong currents and are characterised by little topographic relief and relatively shallow depths (Alevizon *et al.* 1985). Additionally, exposure to easterly trade winds and associated seas may contribute to the increased barrenness of the shallow eastern lagoon (Russell, Neil & Hilliard 2004) in comparison to the deeper western lagoon, where hard and soft corals are more abundant.

Manta tows and photo quadrat surveys found comparable coverage of soft corals along the north-western and south-western reef edge and deep lagoonal habitats. Although hard coral cover was greatest along the southern reef edge and deep lagoon habitats, hard coral cover varied in range between the two methods with estimates of cover generally greater using manta tows than photo quadrats. This may be due to variability in observer estimates versus more precise calculations from photo quadrat analyses or the increased coverage available using manta tow methods and accessibility to deeper depths not surveyed using the photo quadrats. In 2017, Gilmour *et al.* (2019) observed lower coral cover loss in deeper and southern areas of Ashmore Reef following the bleaching event in 2016. It's likely that these areas were not damaged to the extent of shallower sites during recent prolonged periods of heat stress and have consequently maintained higher levels of coral cover.

### 9.5.2 Changes in coral cover over time

Live hard coral cover declined between 2009 and 2019, however was still greater than observed in 2005 (Kospartov *et al.* 2006; Ceccarelli *et al.* 2011b) when coral cover was the lowest on record (Gilmour *et al.* 2019). Although low or no coral mortality was observed at Ashmore Reef following high sea surface temperatures in 1998 (Skewes *et al.* 1999b; Skewes *et al.* 2005) severe coral loss resulted from the bleaching event in 2003 (Gilmour *et al.* 2019) as was observed in neighbouring reefs (e.g. Smith, Gilmour & Heyward 2008; Gilmour *et al.* 2013; Done, Gilmour & Fisher 2015; Gilmour *et al.* 2019). Following bleaching events in 1998 and 2003, severe coral loss resulted at Ashmore Reef (Gilmour *et al.* 2019) as was observed in neighbouring reefs (e.g. Smith, Gilmour & Heyward 2008; Gilmour *et al.* 2013; Done, Gilmour & Fisher 2015; Gilmour *et al.* 2019). It's assumed the increase in coral cover from 2005-2009 was rapid (e.g. compared with Graham *et al.* 2011) but delayed, resulting from the establishment of juvenile coral colonies (Ceccarelli *et al.* 2011b). Following this increase, cover remained relatively stable before decreasing once again in 2016/2017 following mass bleaching associated with the 3<sup>rd</sup> Global Coral Bleaching Event and frequent cyclones (Gilmour *et al.* 2019). Between October 2016 and October 2017 following the 2016 bleaching event, coral cover at Ashmore Reef decreased from 36% to 24% (Gilmour *et al.* 2019), comparative to that found in 2019 (20.3% ± 3.42 s.e.). Additionally, coral cover loss was lower at southern and deeper water (> 6 m) sites than in shallower water (~ 3 m) on other parts of the reef (Gilmour *et al.* 2019). This was also evident during 2019 surveys, where shallow reef

habitats showed lower coral cover than in 2006. Edgar *et al.* (2017) characterised the assemblages at Ashmore Reef as coral-poor, recording the lowest percent live coral cover of neighbouring reefs. However, their sites at Ashmore Reef were limited to just the shallow areas near the western lagoon and channel entrance to the lagoon. It must be noted that although more sites from 2019 were used for the comparison than 2006, replication over depth zones was not consistent across all sites. Irrespective, live hard coral cover was doubled in 2019 compared with 2005, potentially suggesting the coral communities of Ashmore Reef are resilient and have the ability to rebound following disturbance.

Cover of Acroporidae, Pocilloporidae, Poritidae and Faviidae corals in 2019 was greater than other hard coral families and may reflect their potential to recover following disturbance. Coral growth form and tissue thickness have been suggested as potential influencers in coral vulnerability to damage (Loya *et al.* 2001). Coral species of the genera *Acropora* and *Pocillopora* are highly susceptible to environmental disturbance including storm damage, bleaching and corallivore outbreaks, but are often the taxa that most rapidly recolonise bare substrates (Burt, Bartholomew & Usseglio 2008) and drive early reef recovery (e.g. Doropoulos *et al.* 2015; Gouezo *et al.* 2019). In contrast, massive coral taxa such as Poritidae and Faviidae are considered among the most resistant taxa on reefs to thermal bleaching (Loya *et al.* 2001; Burt, Bartholomew & Usseglio 2008). These corals commonly suffer minor damage with tissue regeneration considered rapid, sometimes beginning almost immediately following disturbance (Brown & Suharsono 1990) and other times years after (Roff *et al.* 2014). Subsequent phase-shifts in reef fish community structure and declines in fish abundance and diversity are known to result from decreased coral cover and habitat degradation (Jones *et al.* 2004; Graham *et al.* 2006; Holbrook *et al.* 2015). However, higher cover of these corals seen at Ashmore Reef add substantially to the heterogeneity and structural complexity of the reef, in turn influencing the diversity and abundance of coral- and reef-associated marine organisms (e.g. Pereira *et al.* 2014; Darling *et al.* 2017; Komyakova, Jones & Munday 2018).

The time of reproduction and connectivity to other coral reefs influence a reef's ability to recover from disturbance (Van Oppen & Gates 2006; Done, Gilmour & Fisher 2015). Larvae of brooding corals are released several times per year and typically disperse over relatively short distances (< several kilometres), whereas that of broadcast spawning corals are produced during one or a few discrete periods and disperse far more widely (Gilmour, Speed & Babcock 2016). As a result, Ashmore Reef's geographical location may make it more vulnerable to disturbance due to delays in coral recruitment and recolonization from other reefs following stresses. However, it may in fact be this distance from mainland influences and human interference that allows rapid growth and propagation of surviving corals with enhanced protection from anthropogenic disturbances (Gilmour *et al.* 2013).

In 2019, Ashmore Reef was found to support typical coral reef benthic habitats with hard coral and algae the dominant benthic groups. Coral cover varied across locations and habitat types. The overall appearance of benthic community assemblages appeared normal and no signs of severe stress or major recent disturbance were evident. High presence of important reef-building taxa and live hard coral cover following the 2010 and 2016/2017 bleaching events suggest the coral communities of Ashmore Reef are resilient and capable of recovery following future climatic stress events.

## 9.6 Management implications and recommendations

Maintenance of diversity and monitoring cover and health of hard corals at Ashmore Reef are important objectives and we recommend that the monitoring be carried out every five years coincident with the regular invertebrate monitoring. Additional surveys should be made in the event of any large perturbation such as a cyclone or a warming event. Forecasts for the latter are made well in advance of each summer, and the additional surveys should be planned to coincide with these to document the extent and timing of any coral mortality.

# 10 ASHMORE REEF: FISH DIVERSITY AND ABUNDANCE

Damian Thomson, Daniela Ceccarelli, Michael Haywood, Russ Babcock, Christopher Doropoulos, Lauren Hardiman, Margaret Miller, Emma Westlake, Mark Tonks, Cindy Bessey, Mel Orr and John Keesing

## 10.1 Abstract

Fish and shark diversity and abundance were determined across major reef zones (refer to Figure 70) at Ashmore Reef in September 2019. A total of 22,051 finfish from 44 families and 365 species were recorded during 2019 surveys and the composition of fish and sharks was strongly influenced by reef zone. The mean density of fish observed was 2,394 individuals/ha, with highest densities observed within the reef slope (4,977/ha) and west lagoon (4,902/ha). Highest species richness of fishes was also observed within the reef slope (150 species) and west lagoon (97 species) zones, which were deeper and had a more complex coral framework than the shallow sand flats, east lagoon and reef flat areas. Reef-associated sharks were rare (mean = 1.77/ha) with a maximum density of 60 sharks/ha on the reef slope and 20 sharks/ha on the sand flat and reef flat. These densities were consistent with previous studies of sharks at Ashmore Reef and are similar to those reported by Robbins (2006) for areas of the Great Barrier Reef that are open to fishing. The fish assemblage at Ashmore Reef appears typical of Indo-Pacific coral reef fish faunas, with an emphasis on oceanic species, a strong influence of reef zone and a sustained effect of past fishing impacts

## 10.2 Introduction

Ashmore Reef, located at the far north-western edge of Australia's EEZ, was proclaimed a nature reserve in 1983 (583 km<sup>2</sup>), although traditional fishing by Indonesian fishers was permitted by way of a Memorandum of Understanding (MOU) with the Australian Government until 1988 (Reserve, 2009). After this time, a no-take area that banned fishing was declared at Ashmore Reef, although a small area in the centre of the reef within West Lagoon was exempt. Despite most of the reef being closed to fishing since 1998 illegal fishing continued and has been implicated as a likely cause of the low abundance of sharks recorded during previous studies (Skewes *et al.* 1999a; Wilson *et al.* 2009; Speed, Cappo & Meekan 2018). Since 2008, strict enforcement of the no-fishing area (due to Australian Border Force/ Australian Navy presence) is thought to have significantly reduced fishing activity at Ashmore Reef Marine Park.

Six previous surveys of fish communities at Ashmore Reef were conducted over a period of 32 years (Table 24). Previous studies employed different methods for surveying fish communities, with most studies focussing on comparing fish density and biomass among reefs within the Australian MOU74 region (Ashmore, Cartier, Scott and Seringapatam reefs). Previously collected quantitative data on fish and shark abundance and species composition that are considered most relevant to our current study include Skewes *et al.* (1999a), Richards *et al.* (2009) and Speed,

Cappo and Meekan (2018). Although locations and survey methodologies vary between surveys, they provide adequate capacity to understand how fish and shark communities have responded to changes in both management and habitat condition since 1998.

Previous surveys indicate that fish and shark diversity and abundance at Ashmore Reef has been variable since 1998:

- Skewes *et al.* (1999b) assessed fish and shark populations at 231 reef edge locations around Ashmore Reef, using a combination of stationary point count and belt transect surveys. They recorded 17 families and 104 species, with a mean density of 554/ha (fish) and < 1/ha (sharks), and concluded that despite sustained fishing, Ashmore Reef contained one of the highest biomass of fish and sharks within the Australian MOU74 region.
- Kospartov *et al.* (2006) surveyed fish abundance and composition at 12 reef flat and slope locations at Ashmore Reef in 2005. They recorded 345 fish species and higher mean densities than (Skewes *et al.* 1999b) within most large fish families, although a direct quantitative comparison was not made.
- Richards *et al.* (2009) conducted surveys in 2009 at a subset of the Kospartov *et al.* (2006) survey locations (8), and concluded that mean densities of reef fishes had declined significantly between 2005 and 2009, due largely to declines in the abundance of four non-targeted fish families: Pomacentridae (damselfishes), Labridae (wrasses), Scaridae (parrotfishes) and Siganidae (rabbitfishes).
- Edgar *et al.* (2017) and Edgar and Stuart-Smith (2018) conducted intensive fish surveys at 12 locations in the west lagoon and the north-eastern channel zones in 2017 and 2018, recording both large and cryptic fish species. We have not made detailed comparisons with these data as they were conducted only on a small part of the reef and our methods do not adequately census cryptic fish.

The aim of this study is to assess the current distribution and abundance of fish and sharks at Ashmore Reef. We aimed to provide a quantitative and spatially comprehensive picture of the distribution and abundance of fish and shark communities across Ashmore Reef over an 8-day survey period. We selected sites to include the widest possible range of habitats based on habitat maps published by Skewes *et al.* (1999b), and we aimed to locate our sites at or near those surveyed in previous surveys, to facilitate comparisons with previous survey results.

## 10.3 Methods

### 10.3.1 Reef wide fish surveys

Fish diversity, abundance and size frequency were determined at 216 sites distributed among six reef zones: reef slope (n = 37), west lagoon (n = 13), channel (n = 17), reef flat (n = 59), sand flat (n = 63) and east lagoon (n = 27) (Figure 70, SI Table. 9SI Table. 1). Reef zones used to assess spatial patterns in fish abundance and diversity differed from benthic habitat categories as initial analysis revealed fish communities within the channel zones were distinct from communities in the deep lagoon (benthic category) and warranted a separate category (see SI Table. 10 for category details). We recorded, counted and estimated the total length of all non-cryptic, diurnal and reef-associated fish and shark species. We excluded cryptic families such as gobies, blennies and moray

eels, nocturnal species such as cardinalfishes and pelagic species such as mackerel and tuna, which cannot be sampled accurately by underwater visual census (UVC).

Using standard UVC methodology (English, Wilkinson & Baker 1997), we recorded the species and length (nearest 5 cm) of individual fish along a belt transect. At each site, sharks were surveyed on 1 x 50 m x 10 m transect (500 m<sup>2</sup>), large mobile fishes along 1 x 50 m x 5 m transect (250 m<sup>2</sup>) and smaller, site-attached fish surveyed along 1 x 50 m x 2 m belt transect (100 m<sup>2</sup>). Belt transects were marked out using either survey tape or a 50 m length of biodegradable cotton dispensed via a Hip Chain distance measurer. At shallow sites (<3 m depth), transects were oriented haphazardly, while at deeper reef slope and west lagoon sites, transects were oriented along depth contours of between 5 and 9 m. Observers recorded all sharks and large mobile fish on the first pass, then returned along the same transect recording smaller site-attached species.

### **10.3.2 Diversity assessments with timed swim and video deployments**

To complement fish diversity estimates from reef-wide surveys, timed swim searches were conducted around four coral bommies located within the western lagoon reef zone (Figure 70). Bommies were situated in approximately 12 – 15 m water depth and were approximately 100 m in diameter. At each bommie, divers conducted a timed swim of between 35 and 40 minutes, recording the presence of all fish species. Observers commenced swims at the base of each bommie, progressively moving shallower while swimming in a clockwise direction around the bommie, searching as many different microhabitats as possible. The target swimming speed of observers was 10 m per minute, but actual swimming speed was governed by the diversity of fish communities encountered; i.e., observers swam faster in habitats with low fish diversity and slower in habitats with high fish diversity. An estimated distance of 300 m to 450 m (3,000 to 4,500 m<sup>2</sup>) was surveyed during each survey. To ensure fish species that actively avoid divers were recorded, video cameras were deployed for approximately 30 minutes at 8 m depth between 1600 h and 1800 h on each bommie. Video footage was reviewed, and all fish species not previously reported in UVC surveys were added to the fish species list.

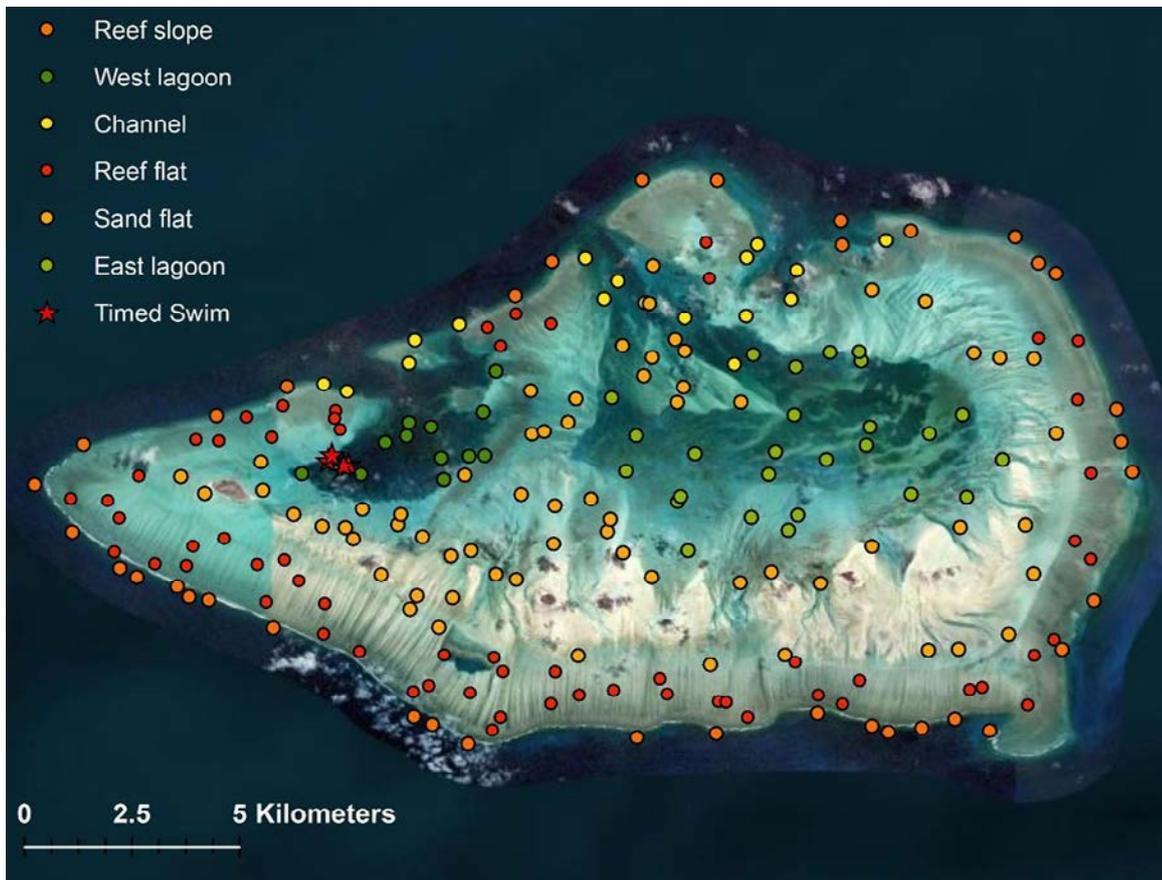


Figure 70. Map showing the location and reef zone classification of the 216 fish and shark survey sites at Ashmore Reef in September 2019. The six reef zones are shown in different colours and location of the four locations where times swims were completed are noted with a red star.

### 10.3.3 Biomass and functional group classifications

The biomass of each fish observed was estimated using length-weight relationships obtained from the scientific literature (e.g. Kulbicki, Guillemot & Amand 2005) and FishBase ([www.fishbase.org](http://www.fishbase.org)). The estimated fish lengths were converted to biomass using derived length-weight conversions ( $\text{Weight} = a \cdot \text{Length}^b$ ). Fish weights were summed for each species and transect, and then standardised for the area surveyed to give an estimate of the biomass of each fish species per hectare (i.e. values for biomass per 500 m<sup>2</sup> for the 50 m x 10 m transects, 250 m<sup>2</sup> for 50 m x 5 m transects, 100 m<sup>2</sup> for 50 m x 2 m were each multiplied to give a biomass per hectare). To enable functional group comparisons among reef zones, fish species were classified into one of seven functional groups based on their dietary composition using information derived from the literature and from FishBase (Table 23).

**Table 23. Functional group classification and diet**

<b>Classification</b>	<b>Diet</b>
Carnivore	Adults feed mainly on fish. Juveniles may feed on small fish and invertebrates such as crustaceans.
Piscivore	Feeds mainly on smaller fish
Invertivore	Feed on molluscs, sea urchins, brittle stars, polychaetes, and crabs.
Herbivore	Feeds mainly on algae
Corallivore	Feeds mainly on corals
Omnivore	Feeds mainly on fish, crustaceans, molluscs, algae or detritus
Planktivore	Feeds on plankton

#### **10.3.4 Shark abundance and composition**

Shark abundance and composition were assessed using two methods; reef-wide UVC surveys and manta-tow observations. During reef-wide UVC surveys, shark species and estimated length were recorded on 50 m x 10 m wide belt transects. Due to the extremely low abundance of sharks observed in reef-wide surveys, additional observations of sharks were made by divers using 230 two-minute manta-tow surveys. Manta tow surveys were conducted at reef slope sites, with divers recording the species of sharks sighted during each two-minute tow. Tow lengths were recorded using a handheld GPS (Garmin eTrex) and were multiplied by the estimated visibility underwater to provide an estimated survey area for each tow (m<sup>2</sup>). Densities of reef sharks were standardised to units per hectare, and estimates obtained using the two methods were visually compared using density bubble plots created in ArcMap 10.1. A comparison of the spatial distribution and density of sharks observed at Ashmore Reef in 2019 with previous years was made by comparing observed densities per hectare with those of previous surveys.

#### **10.3.5 Comparisons of fish communities with previous surveys at Ashmore Reef**

Six previous surveys of fish communities at Ashmore Reef were potentially relevant for comparison to the present survey (Table 24). These surveys were conducted over a period of 32 years and employed different methods for surveying fish communities. Of these five surveys, Skewes *et al.* (1999a) and Kospartov *et al.* (2006) were most relevant for direct comparison with the current survey, as they collected quantitative data on fish abundance and composition from similar locations. Skewes *et al.* (1999a) and Kospartov *et al.* (2006) fish surveys includes two main zones: 'reef tops' (reef flat) and 'reef edges' (reef slope). Our 2019 fish surveys included comparable reef zones, so we utilised data from these two surveys to visually assess changes in mean densities and fish lengths over the past 22 years (1998 to 2019). In situ UVC observations are potentially subject to high observer bias, so we acknowledge that our comparison of means

among years should be treated with caution. Estimates of abundance and biomass for all surveys were standardised to units per hectare.

**Table 24. Reports published from previous surveys at Ashmore Reef**

Report	Year	Sites	Fish survey methods
Berry (1993)	1986	na	Rapid species assessment; timed swim 45-60 min
Skewes et al. (1999a)	1999	231	Underwater Visual Assessment of fin fishes & reef sharks (100 m belt transects)
Kospartov et al. (2006)	2005	16	Underwater Visual Assessment fin fishes & reef sharks (50 m belt transects)
Richards et al. (2009)	2009	8	Underwater Visual Assessment fin fishes & reef sharks (50 m belt transects)
Edgar et al. (2017)	2017	12	Underwater Visual Assessment fin fishes & reef sharks (50 m belt transects) (reef life survey)
Speed et al. (2018)	2004, 2016	46, 89	Baited Remote Underwater Video assessment of sharks

### 10.3.6 Statistical analysis

Fish community structure from the UVC fish surveys in 2019 were analysed to detect differences in overall fish density and differences in the fish species composition between reef zones. We employed a combination of univariate (ANOVA) and multivariate (PERMANOVA, Principal Coordinate Ordination) techniques for testing and presenting these differences. ANOVA were conducted using R studio 3.5.1 and results presented in the main text. Multivariate analysis results were conducted using Primer V7 with results presented in the supplementary information for the sake of brevity. Temporal changes in overall fish density and common fish families were analysed by comparing mean fish densities in 1999 (Skewes *et al.* 1999a) with those in 2019. Data were analysed by univariate ANOVA using R studio 3.5.1.

## 10.4 Results

### 10.4.1 Reef-wide surveys of fish abundance, biomass and species richness

Fish populations at Ashmore Reef were surveyed at 216 sites distributed among six reef zones: reef slope (n = 37), west lagoon (n = 13), channel (n = 17), reef flat (n = 59), sand flat (n = 63) and east lagoon (n = 27) (Figure 70). A total of 22,051 finfish from 365 species and 44 families were recorded (SI Table. 11). Overall, the mean density of fish observed during this study was 2,394 individuals/ha, with significantly different density and biomass between reef zones (Table 2). The highest densities were observed on the reef slope (4,977/ha) and west lagoon (4,902/ha) (Figure 71). The east lagoon (292/ha) and sand flat areas (360/ha) featured very low densities of finfish, accounting for only 6% (1,307) of all fish observed. Surgeonfishes (Acanthuridae; 1,050/ha) and snappers (Lutjanidae, 511/ha) were the most abundant fish families, but parrotfishes (Labridae: Scarinae) and fusiliers (Caesionidae) were also observed in high abundance in several reef zones

(Figure 75). The most abundant fish species overall were the planktivorous damselfish *Pomacentrus coelestis*, the detritivorous surgeonfish *Ctenochaetus striatus* and the carnivorous snapper *Lutjanus gibbus* (SI Table. 10).

**Table 25. Test of the difference in fish density and biomass between reef zones at Ashmore Reef during 2019. Data were arcsine transformed.**

	Total Density			Total Biomass		
	Df	F value	p value	Df	F value	p value
Reef Zone	5	30.1889	< 2.8e-22 ***	5	34.0509	< 6.97e-24 ***
TOTAL	215			4	15.1	2.643e-11 ***

\*\*\* *P* value < 0.001, \*\* *p* value < 0.01, \* *p* value < 0.05

Spatial patterns in estimated fish biomass closely matched those of density (Figure 71 and Figure 72). The mean biomass of finfish recorded across all habitats of Ashmore Reef was 758 kg/ha, with the highest mean biomass observed on the reef slope (2,132 kg/ha) and west lagoon (1,230 kg/ha) and lowest biomass observed within sand flat (118 kg/ha) and east lagoon (25 kg/ha) zones (Figure 72 and Figure 74). Surgeonfishes (Acanthuridae, 208 kg/ha) and snappers (Lutjanidae, 147 kg/ha) were the two fish families with the highest biomass (Figure 4 and 5), contributing 45% of total estimated fish biomass at Ashmore Reef in 2019 (Figure 74).

#### 10.4.2 Distribution and composition of reef fish communities.

With the exception of sand flat and east lagoon sites, most sites surveyed supported a fish assemblage typical for shallow coral reef communities throughout the Indo-Pacific region. Reef slope and deeper western lagoon sites supported the highest densities of fishes, predominantly from the families Acanthuridae (surgeonfishes), Caesionidae (fusiliers), and Lutjanidae (snappers) (Figure 71, Figure 73 and Figure 75). Channel and reef flat reef zones supported moderate densities of fishes, mostly from the families Acanthuridae, Lutjanidae and Labridae:Scarinae (parrotfishes), while sand flat and east lagoon zones supported low densities of all fish families, with exception of the family Lutjanidae.

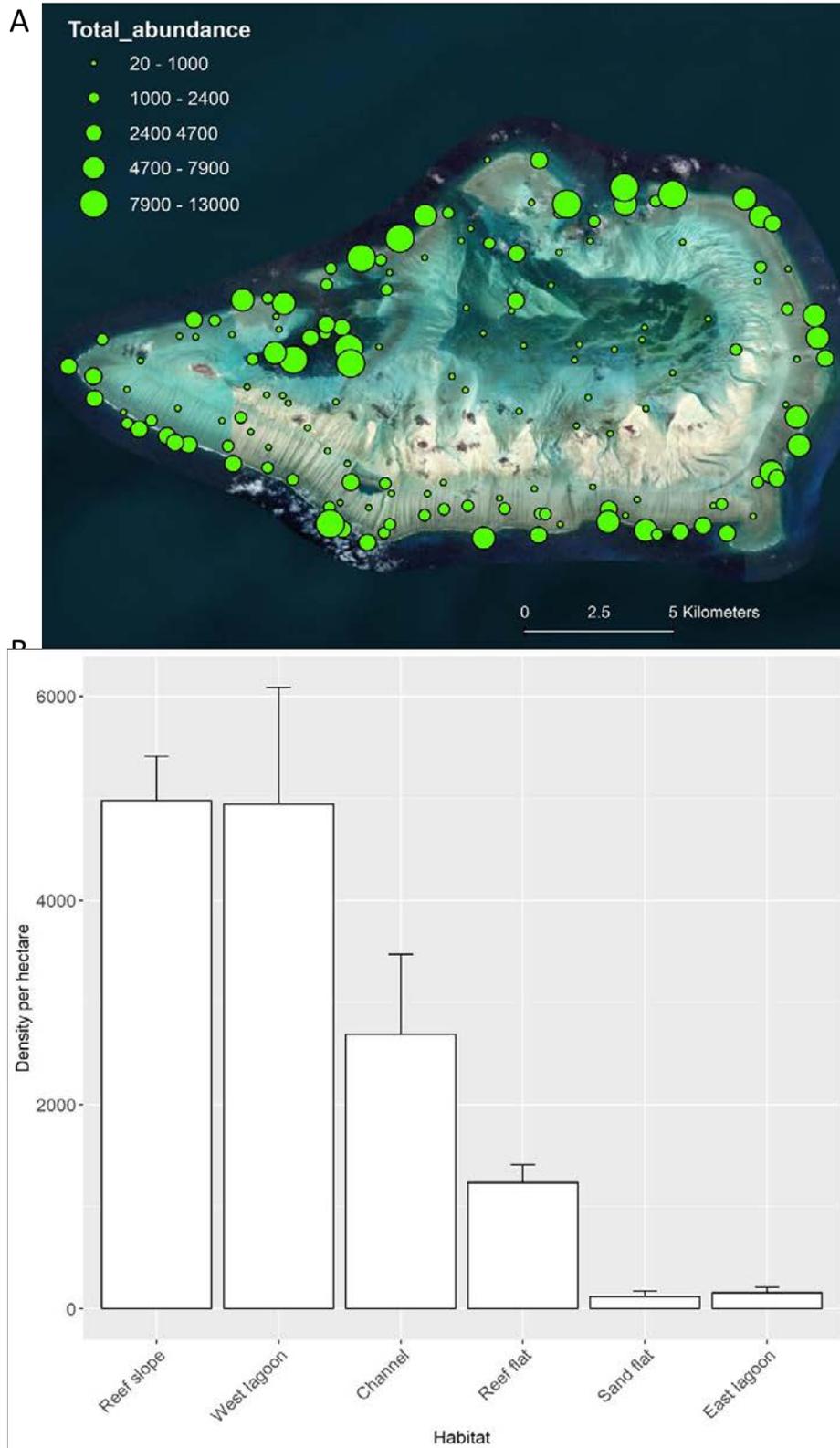


Figure 71. a) Map showing the total density of coral reef fishes observed at each of the 216 survey sites in 2019. Circle size represents the number of fish recorded per site as individuals/ha. b) Plot of mean density ( $\pm$  se) of fish observed in the six reef habitats in 2019.

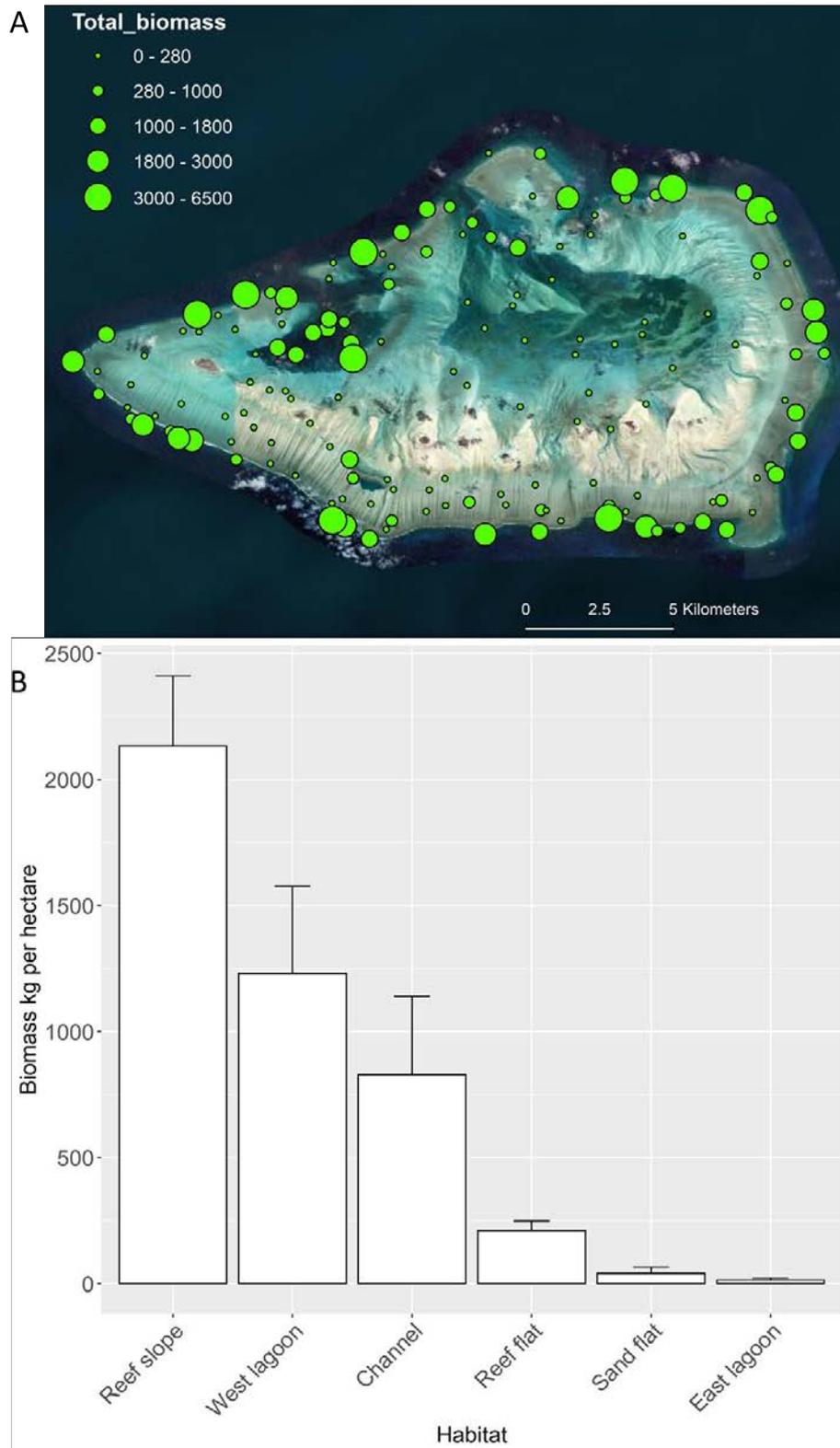


Figure 72. a) Map of biomass of coral reef fishes at Ashmore Reef at 216 survey sites in 2019. Circle size represents the number of fish recorded per site as individuals/ha. b) Mean biomass of fish (+ se) in six reef habitats.

### Density per hectare

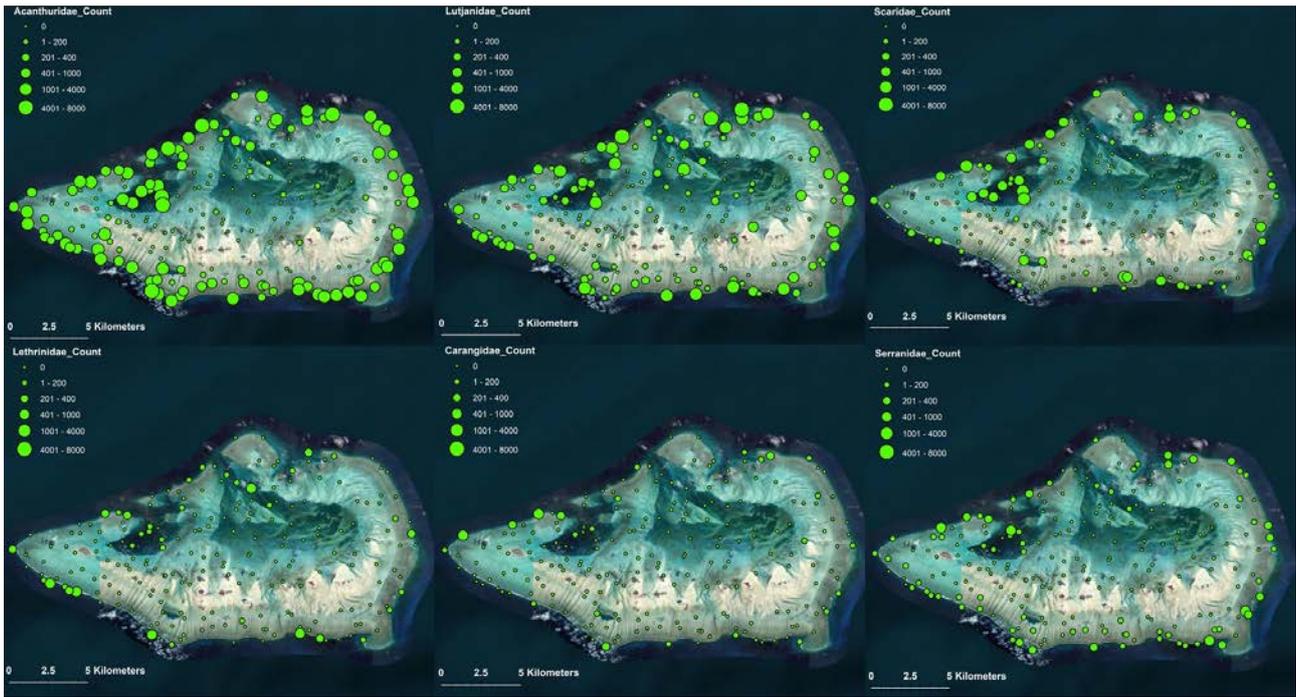


Figure 73. Bubble plots showing the total density of reef fishes observed within the six most abundance families at each of the 216 survey sites in 2019. Circle size represents the number of fish recorded per site as individuals/ha.

### Biomass kg/ha

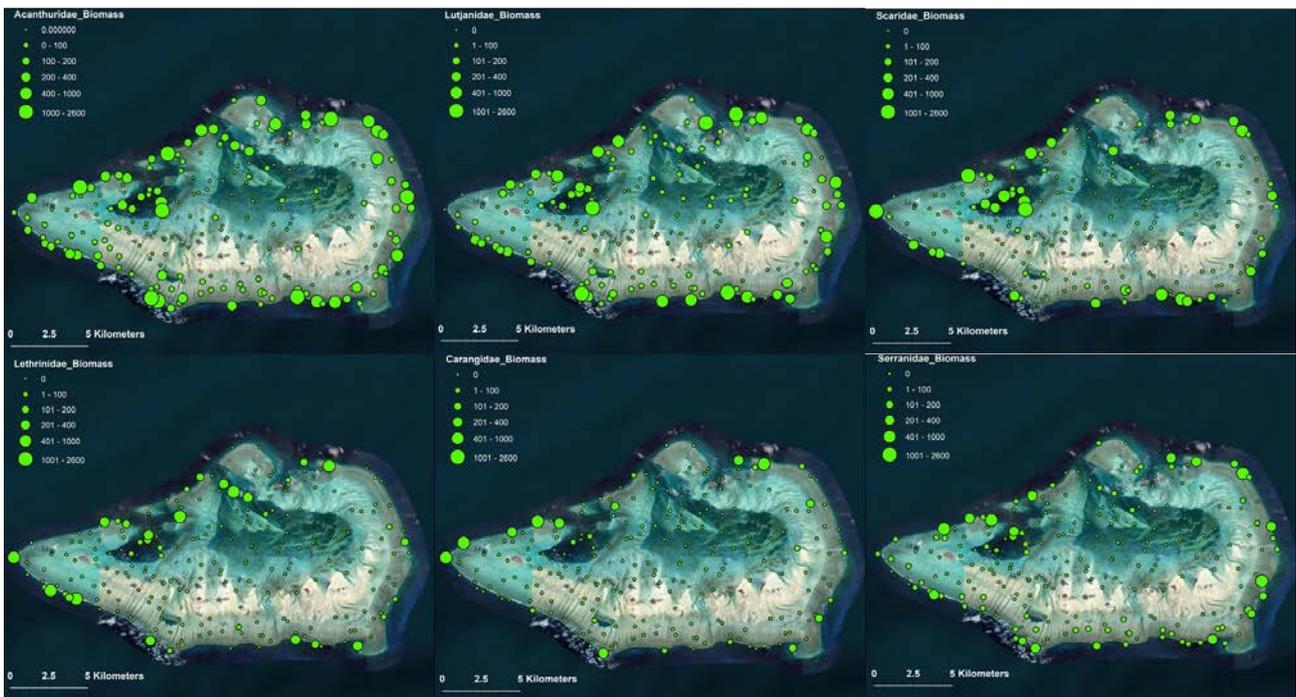


Figure 74. Bubble plots showing the biomass of reef fishes observed within the six most abundance families at each of the 216 survey sites in 2019. Circle size represents the biomass of fish recorded per site as individuals/ha.

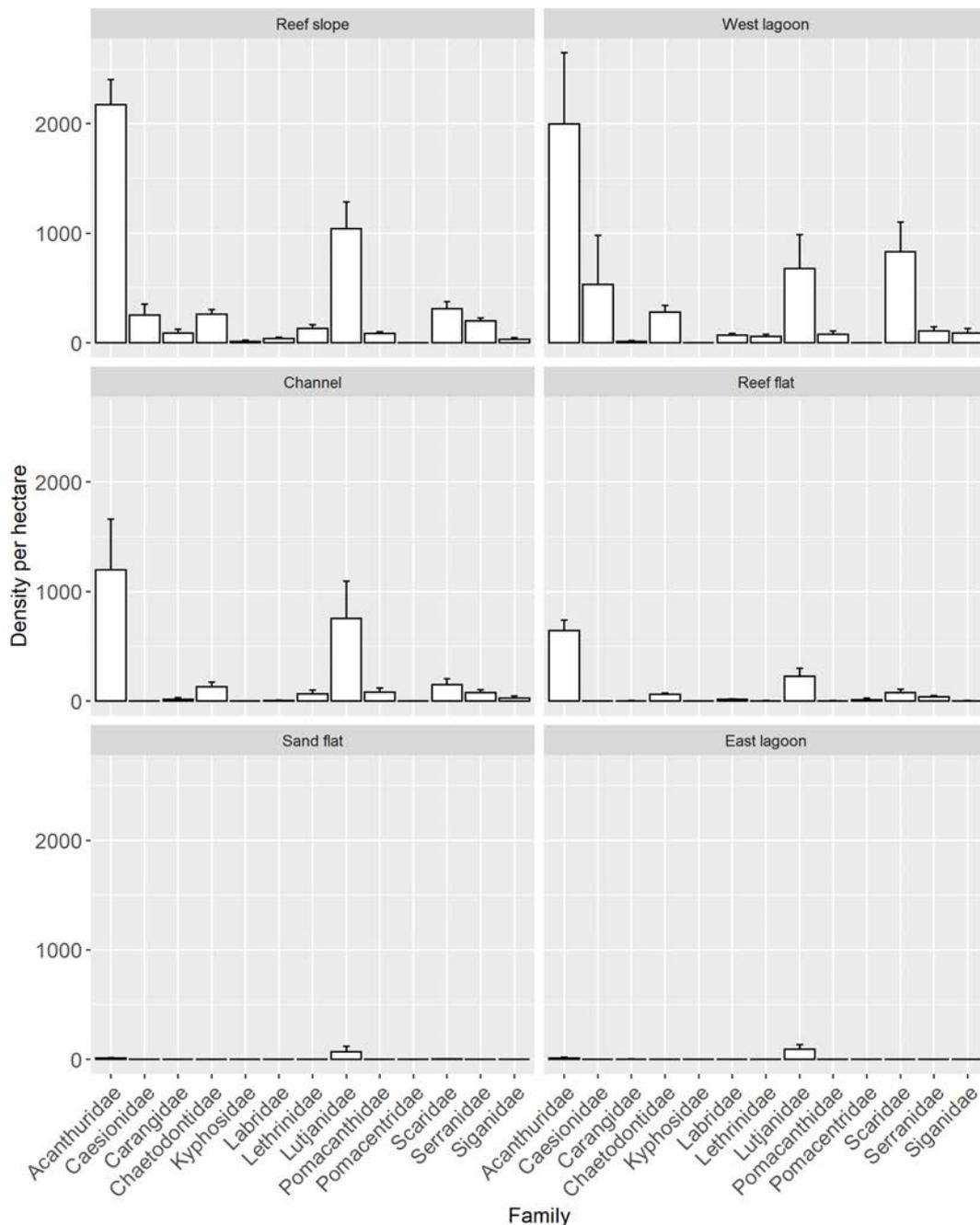


Figure 75. Mean density of the 13 most abundant families of fish within the six surveyed reef zones at Ashmore Reef.

PERMANOVA analysis indicated that the species composition of fishes differed significantly among reefs zones (SI Table. 12). The PCO plot shows that fish communities were similar within the reef slope and channel zones (95% CI overlap), however these communities were significantly different to those on the reef flat and east lagoon (Figure 76). High densities of *Ctenochaetus striatus* (lined bristletooth, surgeonfish), *Cephalopholis argus* (peacock rockcod), *Lutjanus gibbus* (paddletfish), *Acanthurus nigrofuscus* (brown surgeonfish) and *Acanthurus blochii* (ringtail surgeonfish) were typical of the reef slope and western lagoon reefs zones (Figure 76, SI Table. 12), while high densities of *Acanthurus triostegus* (convict surgeonfish), *Chrysiptera biocellata* (two-spot damselfish) and *Halichoeres trimaculatus* (threespot wrasse) were typical of reef flat zones.

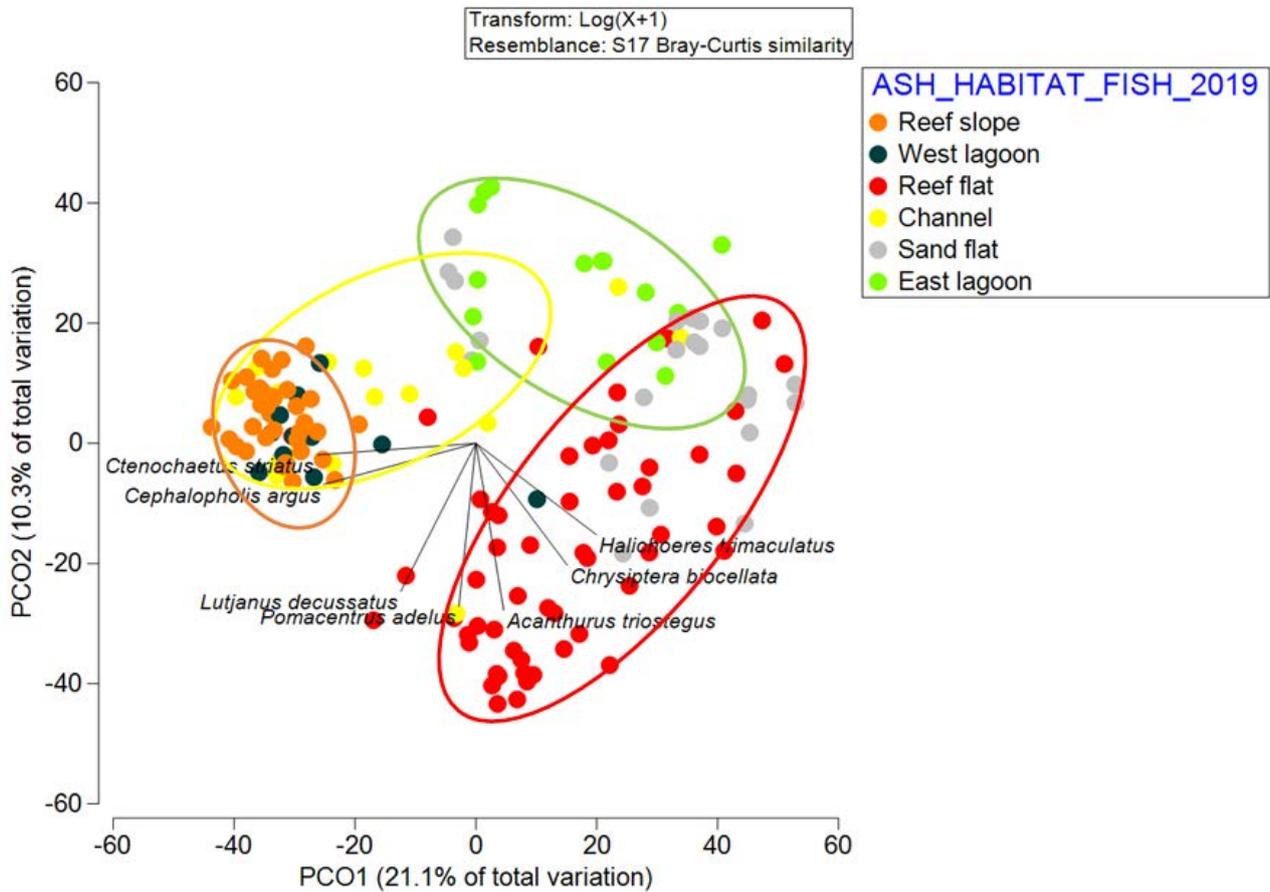


Figure 76. PCO showing the fish species composition of each reef-zone combination. Ellipses show the 95% confidence intervals associated with groups of dots representing each reef zone. The vectors represent fish species most responsible for differences between reef zones. The first two axes show percent of variation and described approximately 30% of total variation.

Community composition was strongly influenced by individual reefs zones. The PCO plot indicated there were key differences in environmental conditions important for structuring reef fish communities between reef zones. Water depth, substrate rugosity (reef complexity) and percentage cover of soft and hard corals were higher in reef slope, west lagoon and channel sites than reef flat or east lagoon sites (Figure 77). In comparison, sand flat and reef flat sites, which had lower abundance and diversity of fish families, featured a higher percentage cover of non-living substrate (ie. sand and rubble) and lower rugosity (unevenness).

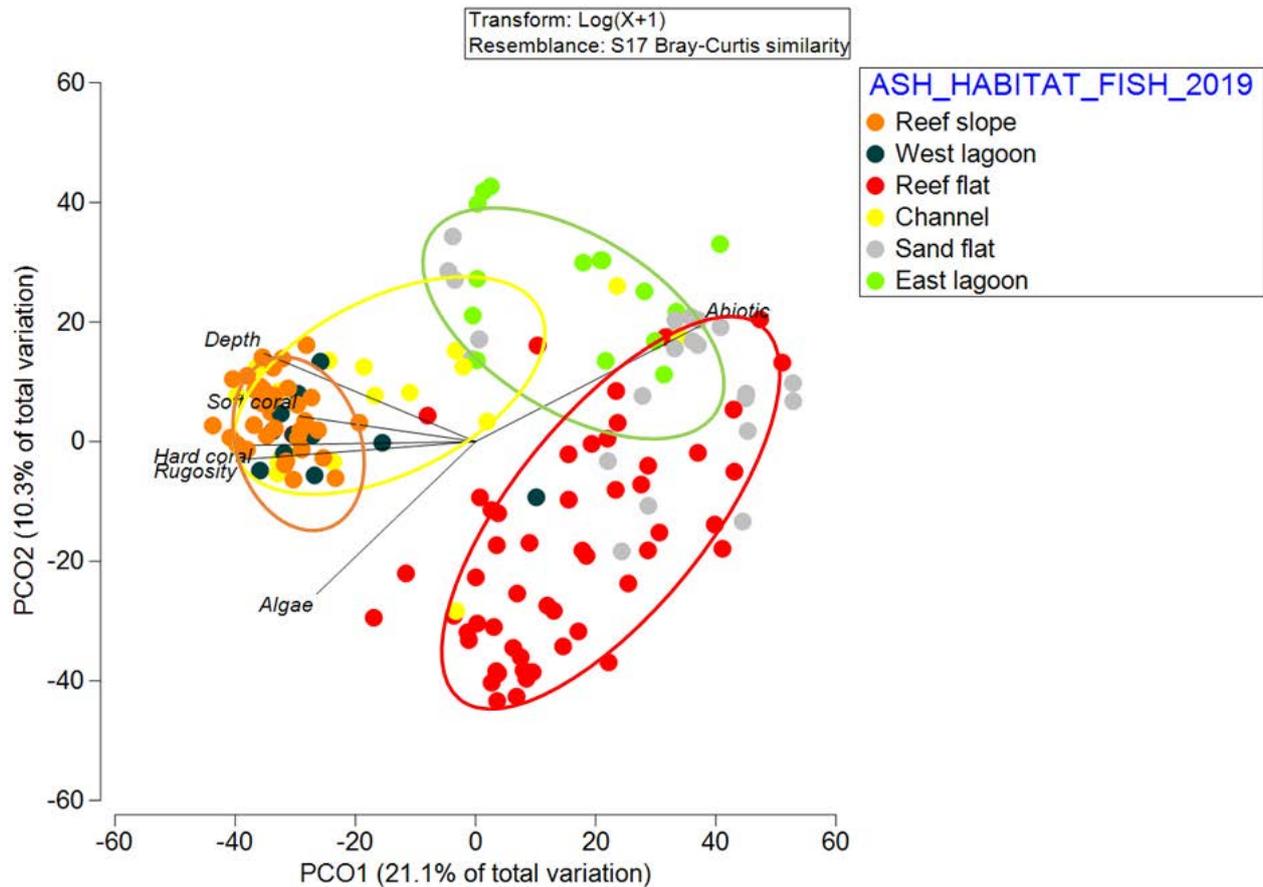


Figure 77. PCO plot showing the fish species composition of each reef-zone combination. Ellipses show the 95% confidence intervals associated with groups of dots representing each habitat combination. The vectors represent environmental variables most responsible for differences between reef zones. The first two axes show percent of variation and described approximately 30% of total variation.

Spatial patterns in fish community composition were partly reflected in the distribution of fish functional groups at Ashmore Reef. Herbivores were the dominant functional group within four of the six reef zones with mean densities of 2,450, 3,007, 1,456 and 749 individuals per hectare in reef slope, west lagoon, channel and reef flat zones, respectively (Figure 78 and Figure 79). High densities of herbivores were largely due to the high abundance of herbivorous Acanthuridae species within these four reef habitats. Piscivorous fish accounted for the second highest abundance across the same four habitats, with mean densities of 1,317, 807, 836 and 273 individuals per hectare, respectively (Figure 79). Densities of most functional groups were comparatively low within the sand flat and east lagoon zones in comparison to other zones. However, piscivore densities were approximately twice that of other functional groups, due largely to the high abundance of piscivorous Lutjanid (snappers) species such as *Lutjanus gibbus* (paddletail), *Lutjanus decussatus* (checkered snapper) and *Lutjanus bohar* (red bass) (SI Table. 11). Invertivores, omnivores and corallivores were observed in low densities in all zones (Figure 78, Figure 79).

## Functional group biomass kg/ha

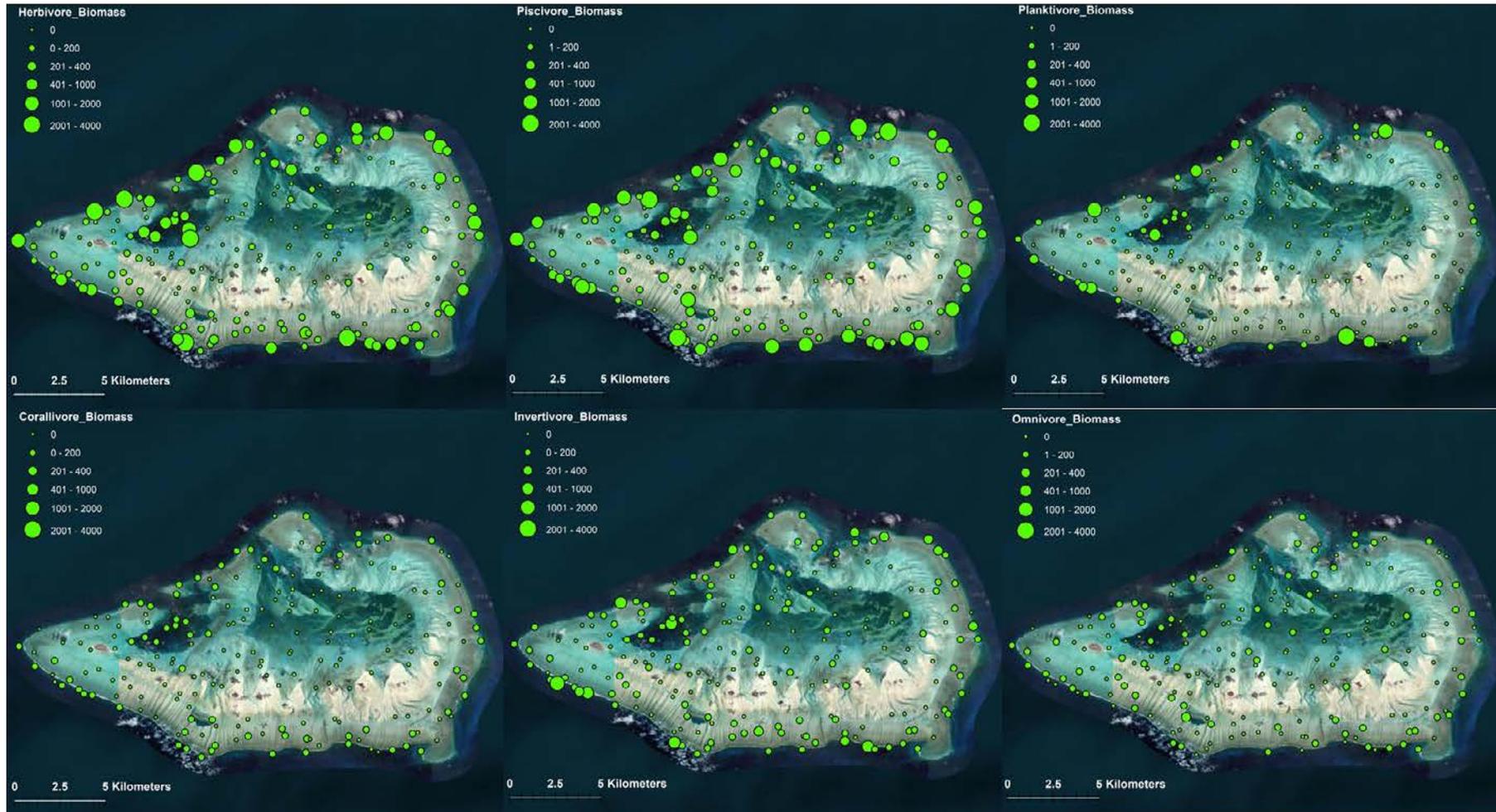


Figure 78. Bubble plots showing the biomass of functional groups of reef fishes observed within the six most abundance families at each of the 216 survey sites in 2019. Circle size represents the biomass of fish recorded per site as kg/ha.

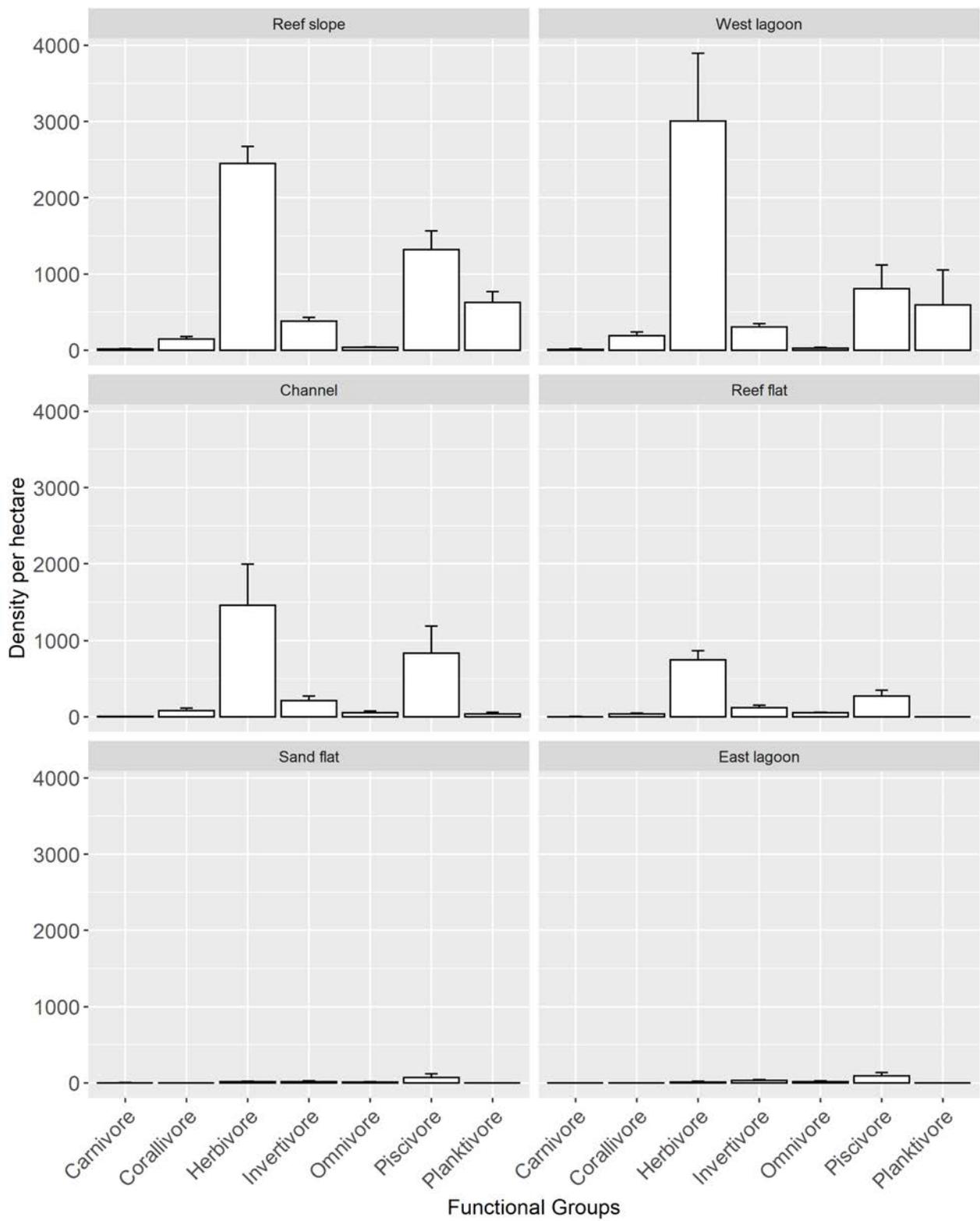


Figure 79. Mean density of the seven functional groups of fish within the six surveyed reef zones at Ashmore Reef.

### 10.4.3 Species richness of fishes

A total of 365 fish species from 44 families were recorded during this survey at Ashmore Reef. Species richness of fishes was highest on the reef slope (aggregate species richness = 150 species; mean species richness per site = 32 species) followed by west lagoon (aggregate species richness = 97 species; mean species richness = 36 species per site; Figure 80a). The east lagoon (26 species) and sand flat areas (14 species) featured comparatively low species richness, with a mean number of species recorded per survey site of less than 5 species (Figure 74c). Damselfishes (Pomacentridae) and wrasses (Labridae) were the most diverse fish families, but butterflyfish (Chaetodontidae; 30 species) and surgeonfishes (Acanthuridae) were also represented by a large number of species in reef slope and reef flat zones (SI Table. 11).

A

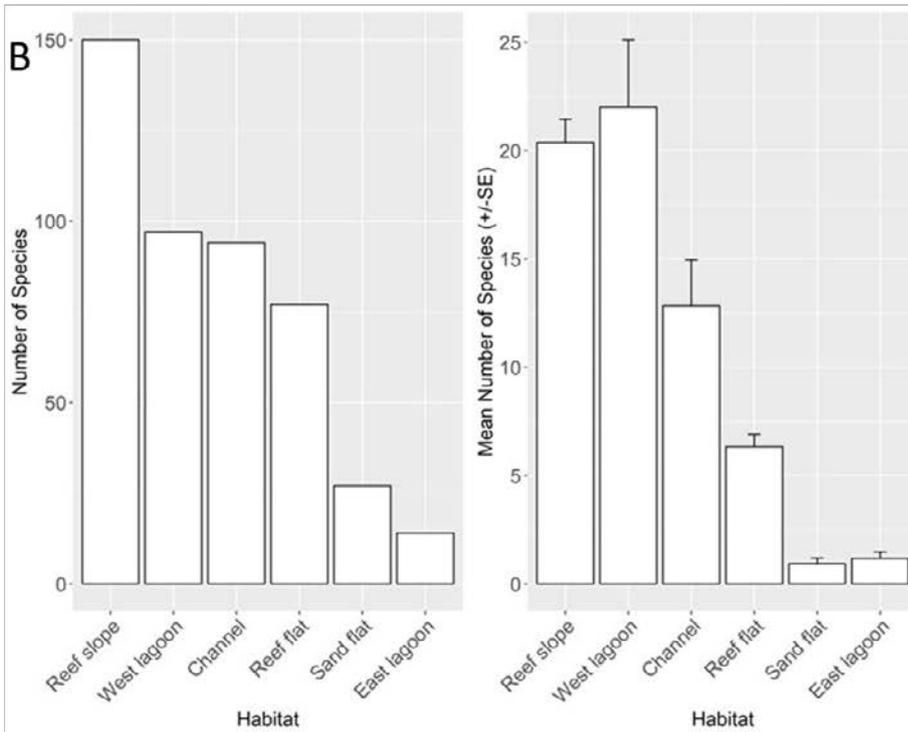
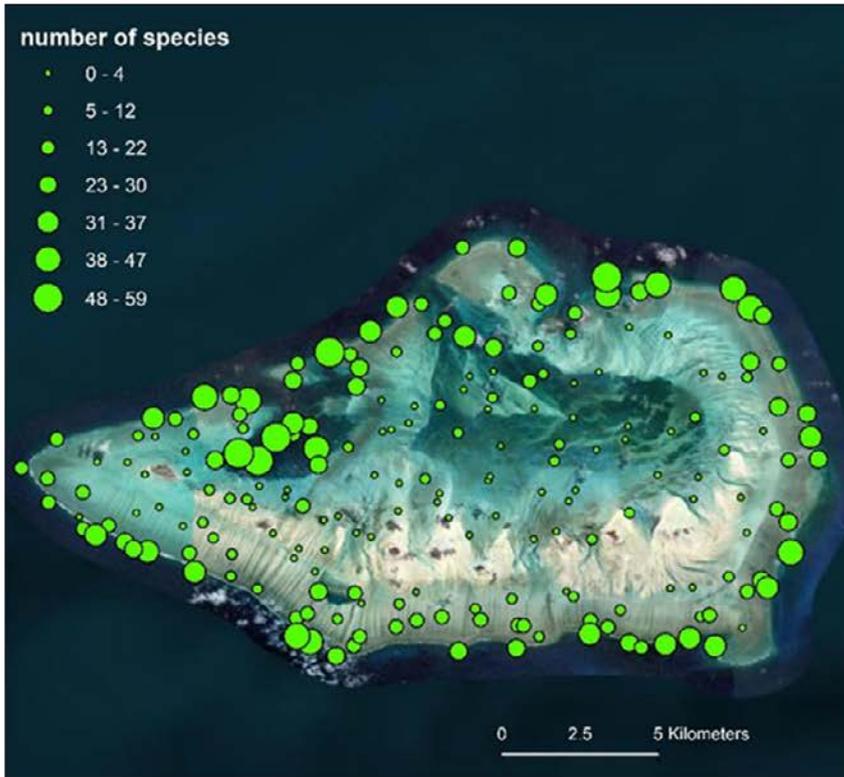


Figure 80a. Map showing the total number of fish species observed at each of the 216 survey sites in 2019. Circle size represents the number of fish species recorded per site. Figure 48b. Total number of species observed in the six reef habitats in 2019. Figure 74c Mean number of species (+se) observed in the six reef zones in 2019.

#### 10.4.4 Size frequency distributions

Visual comparison of size-frequency distributions of common fish families indicated normal distributions for most families (Figure 82). Snappers (Lutjanidae) and groupers (Serranidae) displayed the normal size-frequency distributions, with most individuals of a medium size (25-40 cm total length, TL). The size-frequency distributions of all other families were influenced by large schools of similar sized individuals. For example, large schools of *Acanthurus nigrofuscus* (dusky surgeonfish) (15 to 20 cm TL) were common on the reef slope, and these sizes were at the smaller end of the range for this family, resulting in size frequency distributions skewed toward smaller sizes (left skewed). A similar phenomenon occurred with *Acanthurus triostegus* (convict surgeonfish) on the reef flat, with large schools of similar sized individuals (10-15 cm). Visual comparison of size frequency distributions among reefs zones indicated similar distributions within families (Figure 82).

#### 10.4.5 Reef shark densities

Grey reef sharks (*Carcharhinus amblyrhynchos*) and whitetip reef sharks (*Triaenodon obesus*) were observed during UVC surveys, one additional blacktip reef shark (*Carcharhinus melanopterus*) was observed during manta-tow surveys and one tiger shark (*Galeocerdo cuvier*) was observed from the survey vessel whilst in transit. Shark observations equated to a maximum density of 60 sharks/ha on the reef slope and 20 sharks/ha in sand flat and reef flat habitats. No sharks were observed in channel, east lagoon or west lagoon reef zones (Figure 81). The distribution of sharks recorded during UVC surveys approximately matched the distribution of shark observations recorded during manta tows (Figure 81). A total of 17 sharks were observed during manta-tow surveys, with all observations occurring in the reef slope zone in proximity to where sharks were observed during UVC surveys. Density estimates obtained with manta-tow surveys (0.45 sharks/ha) were significantly lower than for UVC (1.77/ha).

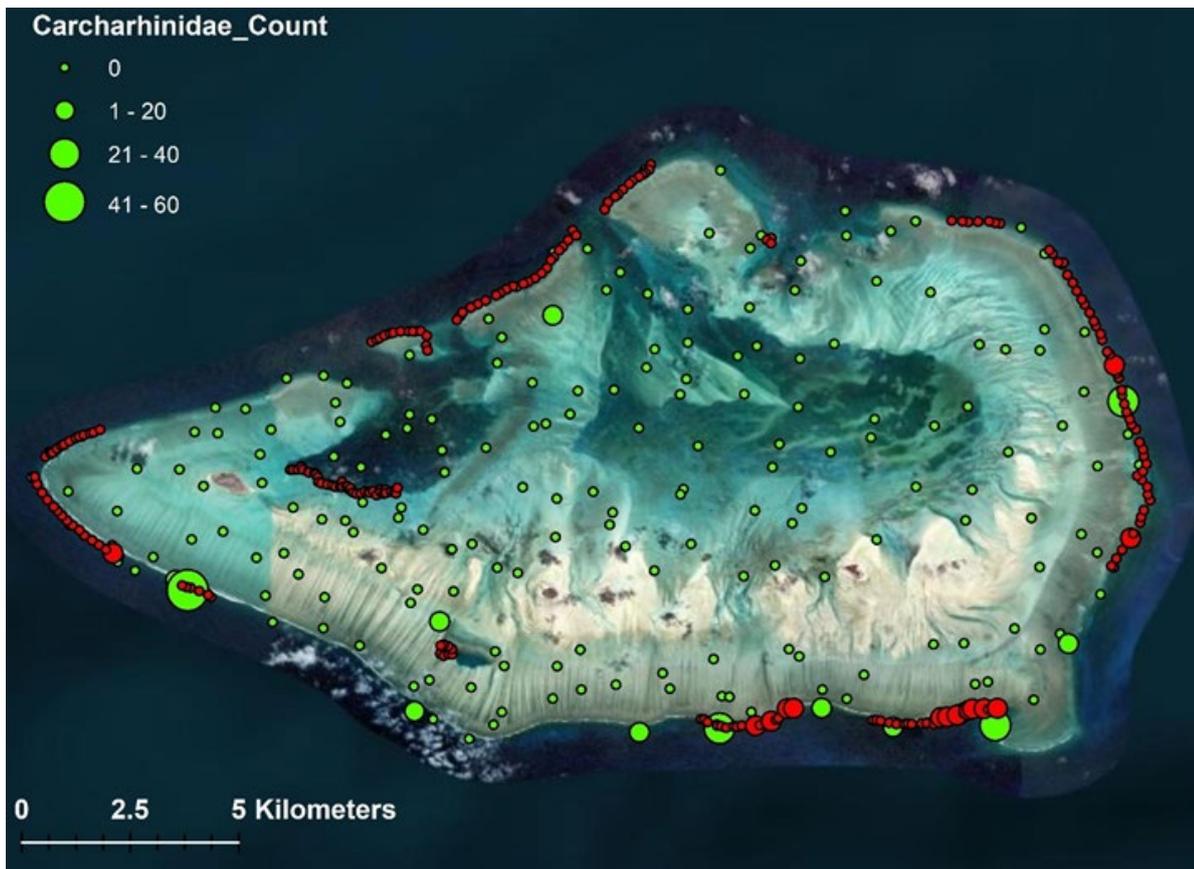


Figure 81. Map showing the density of sharks recorded during UVC surveys (green dots) and manta tow surveys (red dots) at Ashmore Reef in 2019

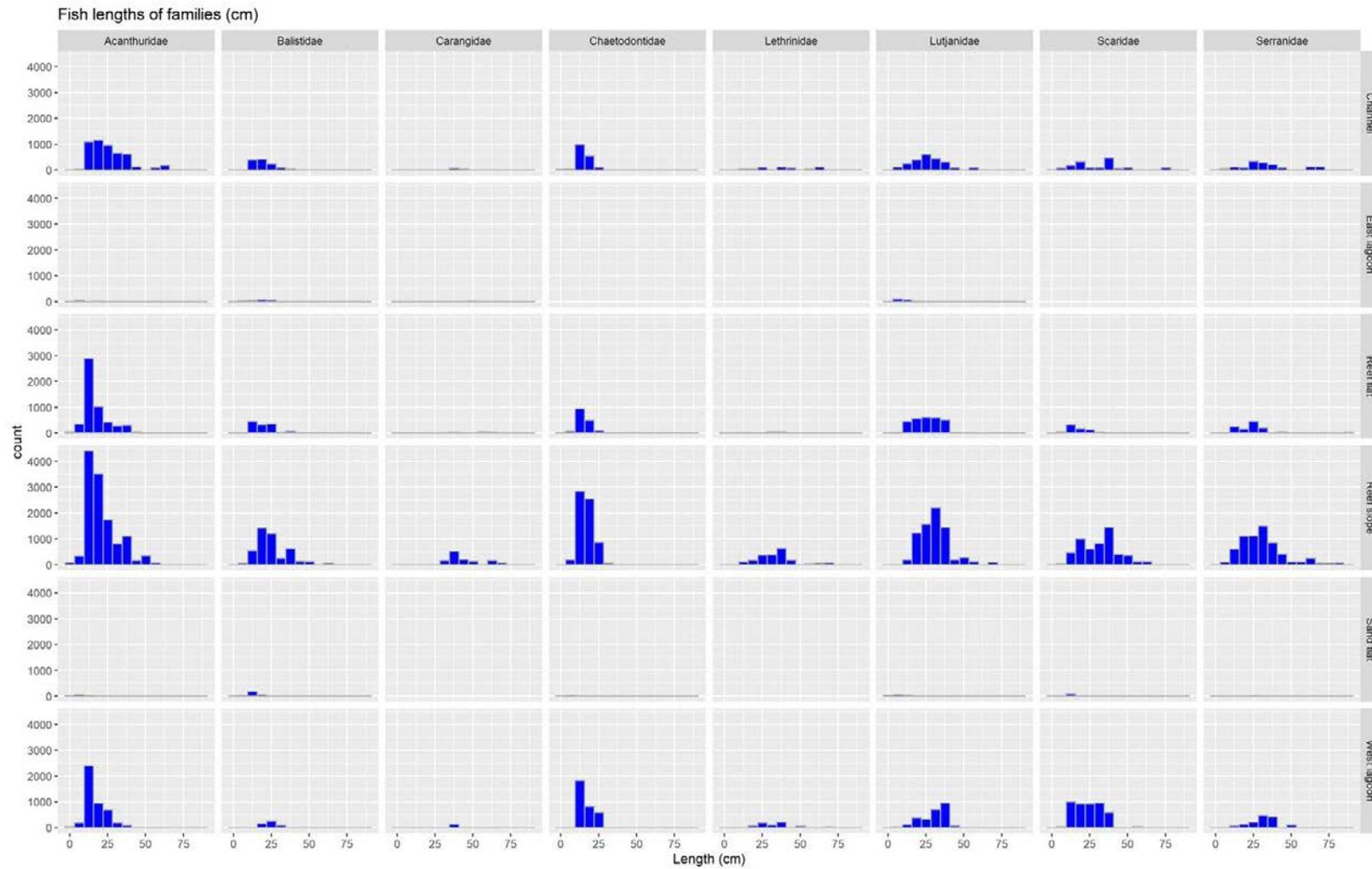


Figure 82. Size-frequency distributions of common fish families within the six reef zones at Ashmore Reef in 2019.

#### 10.4.6 Comparisons with previous surveys at Ashmore Reef

Visual comparisons of mean densities of major fish and shark families between 1998 and 2019 indicate increases in densities in some families, but not others (Figure 83).

Acanthuridae (surgeonfishes), Lutjanidae (snappers) and Serranidae (groupers) densities increased from 1998 to 2019, with Acanthuridae densities increasing by a factor of six over the 22 years (122 to 780/ha). In comparison, Carangidae (trevallies), Haemulidae (sweetlips), Lethrinidae (emperors) and Carcharinidae (whaler sharks) densities did not differ between 1998 and 2019 (Figure 83). Visual comparison of fish densities recorded in 2009 with those in 2019 indicate there was no change in the mean densities of most fish families, with the exception of Acanthuridae, which decreased from 1,300/ha (2009) to 780 per ha (2019) (Figure 83).

Comparisons of mean lengths of fish within families indicate there was no change in the mean length of fish between 1998 and 2019, with the exception of the family Acanthuridae, where mean length decreased from 28 cm to 20 cm (Figure 84). Similar to previous studies, we recorded very low estimates of shark abundance and biomass in 2019. Overall densities of reef sharks were 1.77/ha (UVC) and 0.45/ha (manta tow), which is similar to previously reported reef shark densities on the reef slopes of Ashmore Reef (Skewes *et al.* 1999a; Kospartov *et al.* 2006).

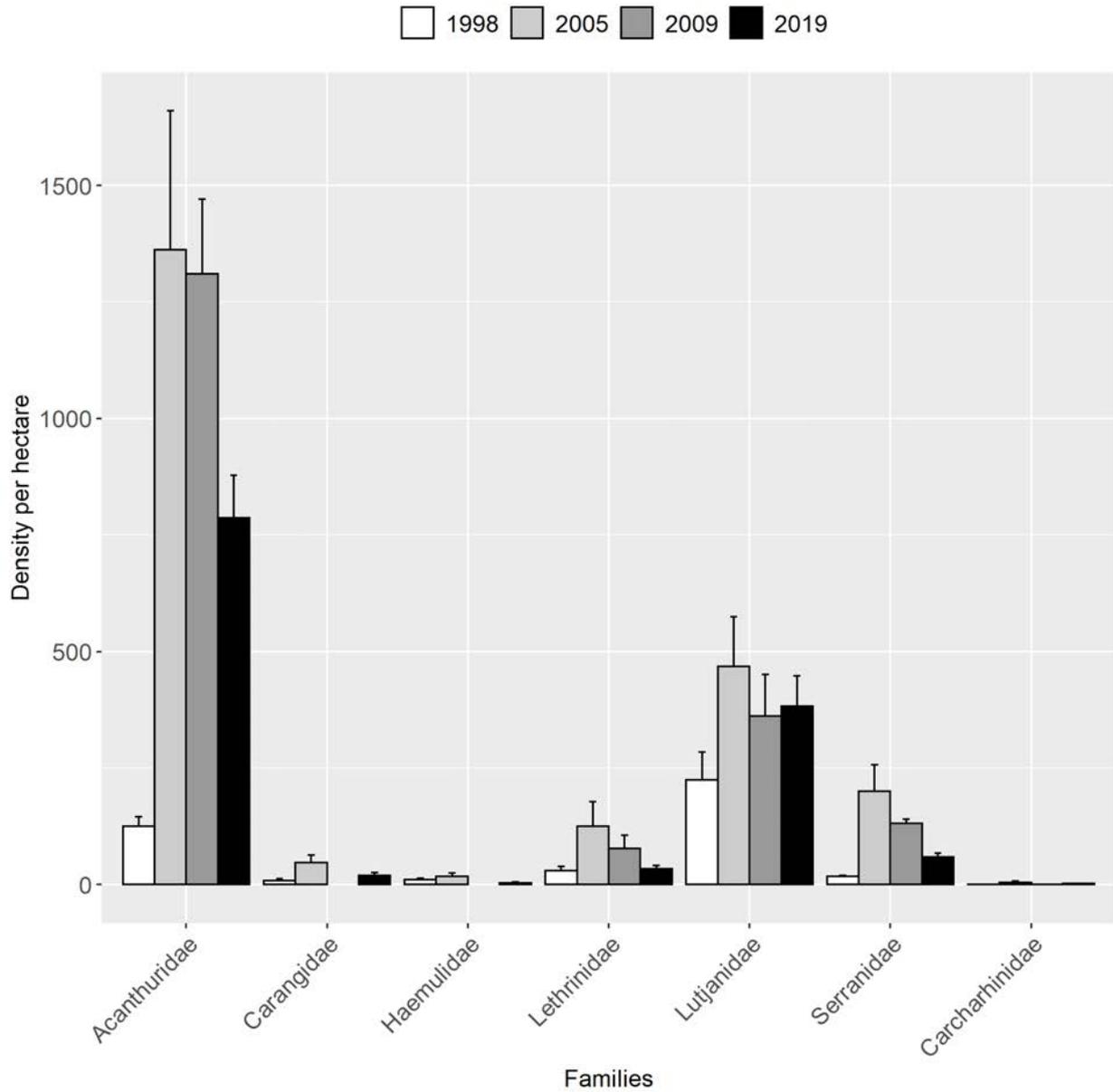


Figure 83. Mean density per ha of fish observed in 1998, 2005, 2009 and 2019 surveys in the 6 major fish families and carcharhinid sharks. SE presented for years where raw data available. Length data for some families were not available for 2009 surveys.

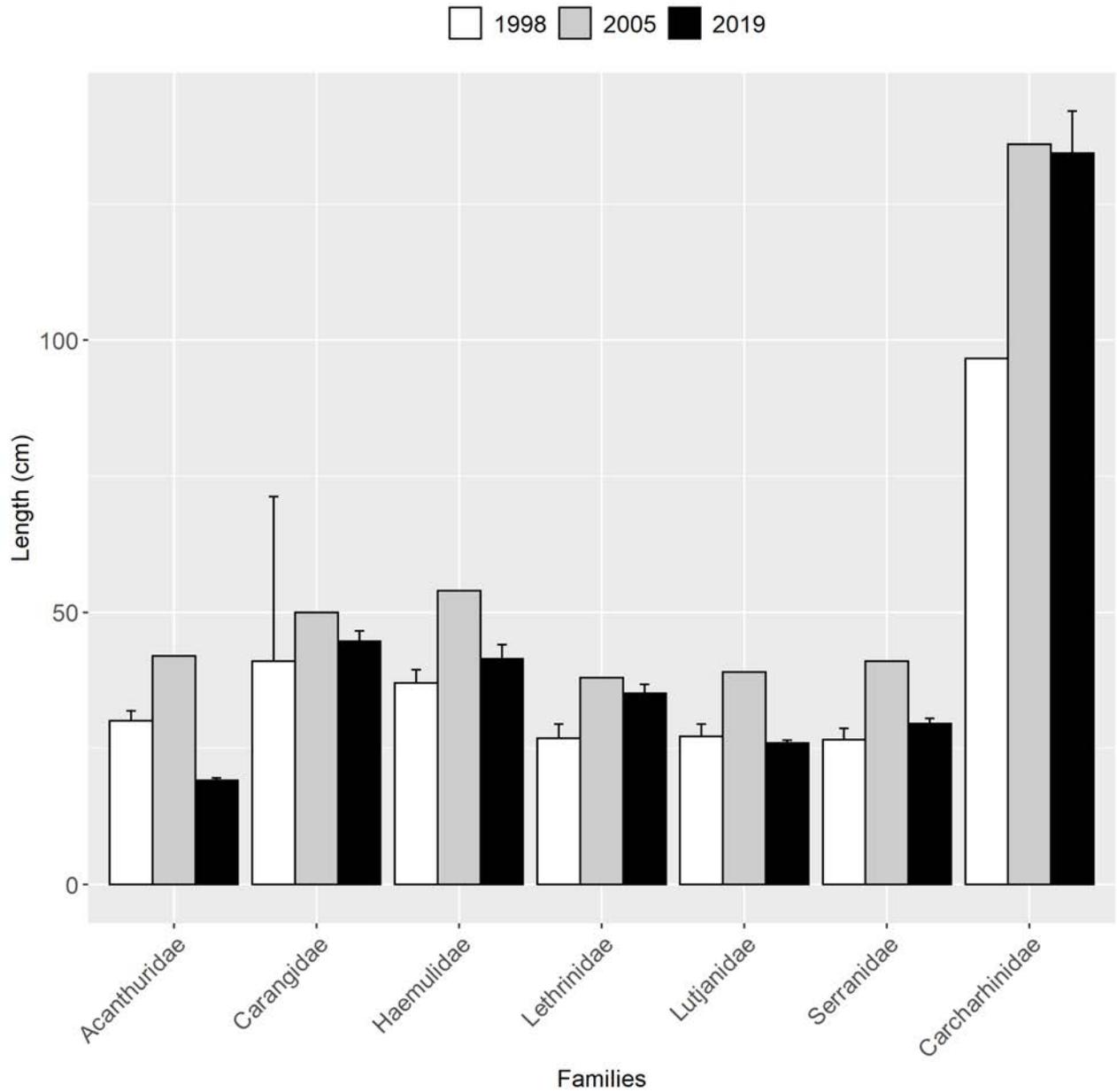


Figure 84. Mean lengths of fish observed in 1998, 2005 and 2019 surveys in the 6 major fish families and carcharhinid sharks. Standard errors are presented for years where raw data were available. Length data for families in 2009 were not available.

## 10.5 Discussion

### 10.5.1 Density and biomass

In 2019, Ashmore Reef supported high densities and biomass of fish, and an assemblage typical of shallow coral reef communities throughout the Indo-Pacific region (Allen 2008). Reef slope and deeper west lagoon sites supported the highest densities of fishes, predominantly from the families Acanthuridae, Caesionidae, and Lutjanidae, while channel and reef flat reef zones supported moderate densities of fishes, and sand flat and east lagoon zones supported low densities of fish. Differences in the density of fish families among zones were likely a reflection of preferences of the different fish species to the particular biological and physical conditions that occur at each reef zone (e.g. water depth, light penetration, wave exposure, the availability of food and shelter, and biological interactions such as predation and competition). East lagoon and sand flat zones were generally characterised by a protected lagoonal position, relatively shallow depth, low percentage coral cover and low habitat complexity, all resulting in a relatively uniform habitat. These conditions are generally favourable for a restricted number of fish species i.e. *Chrysiptera biocellata* (twospot demoiselle) and *Dischistodus perspicillatus* (white damsel), resulting in low fish diversity and overall abundance (density and biomass) in these zones. In comparison, reef slope and channel habitats were characterised by relatively high coral cover, greater water depth, higher water flow and substrate rugosity; all conditions that offer a wide range of microhabitats and therefore promote high abundance.

### 10.5.2 Species richness

As is the case for coral reef fauna in general, fish species richness at Ashmore Reef appears to be correlated to the diversity of benthic habitats. The availability of suitable habitat is a strong determinant of the structure of coral reef fish communities, as many species associate closely with the benthos (Hixon & Beets 1993). The deeper, more complex reef zones (reef slope, west lagoon and channel) had higher coral cover and rugosity, which may be a factor in supporting more diverse fish communities. These findings are supported by previous studies at Ashmore Reef, which also recorded highest fish diversity in deeper, reef edge habitats (Skewes *et al.* 1999a; Kospartov *et al.* 2006). Previously, over 750 fish species have been reported at Ashmore Reef and Cartier Islands (Allen 1993; Russell *et al.* 2005; Hale & Butcher 2013) suggesting the use of survey methods targeting cryptic and very rare species (i.e. rotenone) were used to reveal a high number of additional species not recorded in our surveys.

#### Fish functional composition

Whilst many coral reef fish species are reliant on a diversity of benthic habitats, in many cases, the benefits are mutual. Reef fish can also provide ecosystem functions that support the health and resilience of coral reefs (Green & Bellwood 2009); e.g. herbivores removing benthic algae that can impede the settlement of coral larvae. Many fish species have similar functional roles and it is often useful to group them according to their functional traits

(rather than taxonomy) to gain insights into the roles being performed and to gain an understanding of the degree of functional redundancy (Hoey & Bellwood 2009). Herbivores were the dominant functional group within four of the six reef zones in our study, due largely to the high abundance of many Acanthuridae species. Herbivores have a critical role in maintaining reef health through their role in removing turf algae and sediment (croppers and scrapers), clearing space for the settlement and growth of coralline algae and corals. While herbivores were found to dominate fish communities in most reef zones, piscivores and planktivores were also recorded in high abundance, particularly within the reef slope. Piscivores and planktivores play a major role in transferring energy from benthic invertebrates and plankton to higher trophic levels within coral reef ecosystems (Depczynski *et al.* 2007; Kramer, Bellwood & Bellwood 2013), so it is likely these functional groups are important to food-web dynamics at Ashmore Reef, as they are elsewhere.

### 10.5.3 Reef associated sharks

The four species recorded here, *Carcharhinus amblyrhynchos* (grey reef shark), *Triaenodon obesus* (whitetip reef shark) and *Carcharhinus melanopterus* (blacktip reef shark) and *Galeocerdo cuvier* (tiger shark) are considered strongly reef-associated (Barnett *et al.* 2012) and, like other shark species, are potentially involved in structuring coral reef food webs both directly through predation and indirectly through prey behaviour modification (Roff *et al.* 2016). Previous studies of sharks at Ashmore Reef have reported low densities of reef sharks similar to those recorded in this study (< 1 per ha Kospartov *et al.* 2006: 0.29 per ha Richards *et al.* 2009: < 1 per ha, Skewes *et al.* 1999a), attributed largely to sustained heavy fishing by Indonesian fishers prior to 2008. Indeed, the densities of reef sharks in this and previous studies are similar to those reported by Robbins (2006) for areas of the Great Barrier Reef that are open to fishing. Low densities of reef-sharks were noted by Skewes *et al.* (1999a) as being atypical for oceanic reefs such as Ashmore Reef, and they suggested long-line fishing may have also contributed to the low densities of sharks observed. Russell and Vail (1988) also noted that there was a general perception by Indonesian fishers that sharks were harder to catch now (1986-1988) than previously (less abundant) and that the price of sharks' fins had doubled over the last few years.

Despite the low abundance of sharks observed in 2019 surveys, recent evidence indicates densities of grey reef sharks (*Carcharhinus amblyrhynchos*) may currently be increasing at Ashmore Reef (Speed, Cappel & Meekan 2018), however the rate of increase remains debatable (Guinea 2020). It is possible that reef shark densities in our study were underestimated because sharks are predominantly nocturnal and are known to actively avoid divers in some circumstances. Differences in the densities of reef sharks obtained using UVC and manta-tow in this current study support this notion, with survey methods appearing to influence our shark density estimates, and a combination of survey methods may be more appropriate for monitoring shark densities at Ashmore Reef. Regardless of the survey methods, due to their role in influencing prey demography, habitat structure and ecosystem productivity, sharks remain a key functional group in marine ecosystems (Frisch *et al.* 2016) and continued monitoring of their densities at Ashmore Reef is recommended.

#### 10.5.4 Comparisons of fish communities with previous surveys at Ashmore Reef

Comparisons of fish densities and lengths with previous years indicate changes in some families, but not others. Acanthuridae (surgeonfish), Lutjanidae (snapper) and Serranidae (grouper) densities increased from 1998 to 2019, with Acanthuridae increasing several orders of magnitude over the 22 years (122 to 780 per ha). It is difficult to determine the mechanisms driving these potential increases. Most Acanthuridae species are herbivores and detritivores, and their abundance is strongly influenced by the availability of turf, detritus and macroalgae as a food source. 2005 survey data suggest increases in the density of the Acanthuridae may have occurred rapidly following 1998 (between 1998 and 2005), so increases may have been associated with changes in habitat condition following the 1998 global bleaching event (Goreau *et al.* 2000). Indeed, the rapid increases in the percentage cover of hard corals observed between 2005 and 2009 were also implicated as a potential driver of observed changes in the abundance of some herbivorous fishes (Richards *et al.* 2009), so it is possible that long-term increases in the abundance of Acanthuridae are a result of long-term changes in habitat condition. A complimentary decrease in the mean length of Acanthuridae observed over this same period (28 cm to 20 cm) suggest a reduction in the size of individual species, or a shift in the species composition. We did not analyse changes in species composition among years, so we are unable to determine which of these explanations is most likely. Given previous studies included different survey locations it is possible that some observer/identification bias has occurred between surveys conducted by different fish specialists, the variable nature of these previous observations is to be expected.

#### 10.6 Management implications and recommendations

Periodic monitoring of fish at Ashmore Reef is recommended and should take into account two key issues identified in this study:

- 1) Implementing targeted methods to monitor sparsely distributed populations of sharks at Ashmore Reef is recommended. For example, reef shark densities in our study may have been underestimated because sharks are predominantly nocturnal and are known to actively avoid divers in some circumstances. The use of a combination of survey methods for monitoring shark biodiversity and abundance i.e. UVC plus RUV, may therefore provide more precise estimates of reef-shark densities at Ashmore Reef.
- 2) Implementing new survey methods to obtain more precise diversity estimates is recommended. Our surveys were restricted to 8-days, and excluded cryptic families such as gobies, blennies and moray eels, nocturnal species such as cardinalfishes and pelagic species such as mackerel and tuna, which cannot be sampled accurately by underwater visual census (UVC). Previously, over 750 fish species have been reported at Ashmore Reef and Cartier Island (Allen 1993; Russell *et al.* 2005; Hale & Butcher 2013), suggesting there are potentially many cryptic and very rare species that remain undetected using visual methods. It would be highly advantageous to consider the use of eDNA sampling for future biodiversity assessments.

# 11 ASHMORE REEF: HOLOTHURIAN, OTHER ECHINODERM AND MOLLUSC ABUNDANCES

John Keesing, Daniella Ceccarelli, Christopher Doropoulos, Geoff Hosack, Michael Haywood, Russ Babcock, Damian Thomson, Lauren Hardiman, Margaret Miller, Emma Westlake, Mark Tonks and Cindy Bessey

## 11.1 Abstract

There have been nine surveys of the exploited reef holothurians, clams and trochus at Ashmore Reef since 1987. This chapter presents the findings of the June 2019 surveys and analyses of the population status and long terms trends in abundance of each species that have been fished historically by traditional Indonesian fishers and subjected to incidences of illegal fishing. Where available, data on non-exploited species is included to provide context.

Results from the 2019 surveys indicate the abundance of at least 12 holothurians and two clam species is very low. These species are *Actinopyga lecanora*, *A. mauritiana*, *A. miliaris*, *Bohadschia argus*, *B. marmorata*, *Holothuria whitmaei*, *H. fuscopunctata*, *H. fuscocrubra*, *Thelenota ananas*, *T. anax* (all  $<0.02 / 100 \text{ m}^2$ ), *H. fuscogilva* ( $0.05 / 100 \text{ m}^2$ ), *Stichopus herrmanni* ( $0.2 / 100 \text{ m}^2$ ), *Tridacna gigas* ( $<0.05 / 100 \text{ m}^2$ ) and *Hippopus hippopus* ( $0.37 / 100 \text{ m}^2$ ). These species do not appear to have recovered from overfishing in the mid to late 1980s.

At least one holothurian species, *Holothuria lessoni* is locally extinct, and others are thought to have reproductively ineffective populations. Species that are in very low abundance and were heavily fished in the past include *H. whitmaei*, *B. marmorata*, *T. ananas*, *S. herrmanni* and *Actinopyga* spp. It is likely their population levels are too low and distance between individuals too great to allow effective fertilisation rates for significant population recovery from self-seeding. Instead, it is likely that these populations are reliant on larvae dispersed from distant populations.

The continued presence of moderate densities of the asexually reproducing *H. atra*, *H. edulis* and *S. chloronotus* suggest these species will continue to increase in population size over time.

The importance of the deep-water habitat for holothurians and the possibility that some of the species with very low abundance, in particular, *H. fuscopunctata*, *H. fuscogilva*, *S. herrmanni*, *T. ananas* and *T. anax* are present in higher numbers in deep-water habitats should be determined. This is regarded as the highest priority for future work involving holothurians at Ashmore Reef. These habitats have not had any dedicated surveys since 2003. It is important that these surveys be designed and carried out using the methodology established by Hosack and Lawrence (2013a) to ensure results are comparable with the results of surveys in other habitat types made in 2013 and 2019. It is possible these deep-

water habitats have provided a “depth refuge” from fishing in the past and may have higher abundance and reproductively effective densities for some species.

There is some evidence that illegal fishing, particularly in 2006, has impacted the abundance of invertebrates that may have been in a recovery phase, especially trochus and clams. However, our analysis suggests that overexploitation of invertebrates and of holothurians in the mid-1980s reduced population levels for some species to such low levels that they have not recovered and are unlikely to do so in the near-term future. While long-term analyses show some increases and declines in abundance since 1998, these mostly reflect small changes with a high degree of variability in populations already depressed to a very low 1998 baseline (Skewes *et al.* 1999a) as a result of fishing prior to that survey. We agree with previous analyses (Hosack & Lawrence 2013a) that fluctuations in most species recorded between 1998 and 2013 were due to sampling differences, including comparisons with deep water surveys in 2001 and 2003, and the variability issue associated with surveying fundamentally rare animals with patchy distributions.

The 2019 survey of trochus at Ashmore Reef indicated abundances lower than those recorded in 2013, but similar to surveys in 2005 and 2006. The overall trend through time suggests that trochus numbers have increased since the 1998 surveys of Skewes *et al.* (1999a). There are two main issues with the assessment of trochus at Ashmore Reef. One is their apparent high degree of mobility and tendency to aggregate (Rees *et al.* 2003; Ceccarelli *et al.* 2011a), meaning they will be very patchily distributed and hence difficult to survey accurately. The second issue is that they occur in a very specific habitat (exposed, surge-affected shallow reef crest and upper slope), which may be under-represented in multipurpose surveys such as most of those made at Ashmore Reef, and weather conditions at the time of the survey will affect how accessible this habitat is. This means surveys need to be comprehensive with a larger number of sites on the reef edge considered for the next survey. Trochus appear to be biologically well-adapted to recover from overfishing. They have been shown to maintain populations with low levels of connectivity to other stocks (Berry *et al.* 2019). Trochus spawn at just two years of age (Heslinga & Hillmann 1981), and larvae have a short planktonic phase, settling after just three days close to the parent stock. A tendency to aggregate should also help facilitate high fertilisation rates, and high levels of mobility mean they can respond to changed microhabitat conditions quickly.

The giant clam *Tridacna gigas* had not been recorded at Ashmore Reef since 2006 (Ceccarelli *et al.* 2007) and were absent in the 2009 and 2013 surveys. Based on known growth rates (Munro 1993) the largest (ca. 80 cm) individuals we observed in 2019 would have been for at least 15 years old, so their reappearance is not sudden. Abundance of giant clams remains low and recovery will be slow if at all.

Our analysis suggests that density of the small, non-harvested species *Tridacna crocea* has remained the same or increased, and that *Tridacna maxima* and *T. squamosa* (combined) had increased between 1998 and 2013, with densities declining again in 2019 (Hosack *et al.* chapter 12 this study).

In contrast, the density of *Hippopus hippopus* remains much lower now than in 1998, although it is higher now than in 2006 when Ceccarelli *et al.* (2007) found low abundances

and that dead clams were equally as abundant as live ones, indicating illegal fishing (or some other agent of mortality) had taken place not long before their survey. The higher densities of *Tridacna maxima* and *T. squamosa* (combined) in 2013 and 2019 compared with those in 2005 and 2006, suggests that these species were not subject to the same heavy fishing as *H. hippopus*, and have therefore gradually increased in abundance over time. This is probably particularly the case for *T. maxima*, which was twice as abundant as *T. squamosa* in 2019. In the absence of disturbances such as coral bleaching events, the surveys we conducted should be repeated at least every five years to monitor for any changes in abundance. However, there is also a need for the “reactive” monitoring following large scale perturbations (e.g. bleaching, cyclones), suggested by Ceccarelli *et al.* (2007), to ensure effects are more readily understood and quantified. Lastly, reference sites at other reefs, impacted by fishing or to a lower level such as Rowley Shoals, Mermaid and Scott Reef, or other reef systems in the MOU74 box such as Cartier Island, Seringapatam and Hibernia reefs should be surveyed using the same methods to ensure any changes detected can be placed in the context of wider environmental change and not just fishing impacts. Sampling a wide range of reefs across anthropogenic and environmental pressure gradients increases the possibility of statistically linking temporal and spatial changes in benthic invertebrate abundance to plausible stressors.

## 11.2 Introduction

### 11.2.1 Background

Large, isolated offshore coral reefs present a challenge for conservation and management. They are difficult and expensive to survey frequently enough to observe changes in patterns of diversity and abundance in response to natural and human disturbance events, especially as climate changes. It is also difficult to enforce management regulations on such remote reefs. Ashmore Reef lies 320 km north-west of the Australian mainland and 110 km south of the Indonesian island of Roti, at the edge of the Australian continental shelf (Figure 1 -Figure 2).

Ashmore Reef Marine Park (Figure 1 and Figure 2) was established in 1983. Bilateral arrangements between Australia and Indonesia were established in 1974 under a Memorandum of Understanding (MOU74). The MOU provides limited access to areas of the Australian EEZ (the “MOU74 box”) by traditional Indonesian fishers. Very heavy rates of exploitation of holothurians and clams were reported in 1986 and 1987, and the bilateral arrangements were reviewed in 1988 as a result. Subsequently access to Ashmore Reef was greatly restricted to just a small area (within the current Recreational Use Zone), comprising mostly the deep lagoon near West Island (Commonwealth of Australia 2002). Despite formal conservation measures, pressure on the reef’s biota from illegal fishing of shark, trochus, holothurians and giant clams continued. The level of illegal fishing and the effectiveness of enforcement between 1989 and 2008 is strongly debated even today (Guinea 2020; Meekan, Cappo & Speed 2020). Field *et al.* (2009) point to a significant rise in reports and detection of illegal fishing in late 2005 and 2006. It was the reported increase in illegal fishing and lapses in enforcement presence at Ashmore Reef that prompted the surveys

conducted by Ceccarelli *et al.* (2007, 2011a) and to which those authors attribute significant declines in trochus abundance at Ashmore Reef between 2005 and 2006.

### 11.2.2 Surveys of marine invertebrates at Ashmore Reef

Ashmore Reef is regarded as having a richly diverse echinoderm and molluscan fauna (Marsh *et al.* 1993; Wells 1993) and this is in part attributed to a wide variety of habitat types and the closeness of the Indonesian archipelago and connections to the Australian and Papuan continental shelves. This contrasts with the more isolated Scott Reef and Rowley Shoals which have a less diverse fauna (Marsh *et al.* 1993; Wells 1993). The first surveys of Ashmore Reef were conducted by the Soviets in 1978, this was followed by surveys by the Northern Territory Museum and the Western Australian Museum between 1986 and 1987 (Berry 1993; Marsh *et al.* 1993). Marsh *et al.* (1993) and Wells (1993) recorded 178 species of echinoderms and 433 molluscs and compared these to the lower numbers found at Scott and Seringapatam Reefs (119 echinoderms, 279 molluscs) and Rowley Shoals (90 echinoderms, 261 molluscs).

Marine surveys of invertebrates at Ashmore Reef have subsequently concentrated on commercially exploited species of holothurians, clams and trochus, which have been historically overfished by Indonesian fishers (Russell & Vail 1988; Berry 1993; Ceccarelli *et al.* 2011a). These surveys occurred on eight previous occasions between 1987 and 2013 using a variety of methods (Russell & Vail 1988; Skewes *et al.* 1999a; Smith *et al.* 2001; Rees *et al.* 2003; Kospartov *et al.* 2006; Ceccarelli *et al.* 2007; Richards *et al.* 2009; Ceccarelli *et al.* 2013). The results of some of these studies have been synthesised and compared by Ceccarelli *et al.* (2011a) and Hosack and Lawrence (2013a,b). The latter study led to a new survey design for Ashmore Reef which was tested in 2013 and implemented for the first time in this study.

It is likely that holothurians, trochus and clams were already heavily overfished by generations of traditional fishing by Indonesians prior to the surveys by Russell and Vail (1988) and Skewes *et al.* (1999a). However, further recent declines in all groups have been recorded (Richards *et al.* 2009) and attributed to illegal fishing by Indonesian fishers particularly in 2006 (Ceccarelli *et al.* 2013). Marsh *et al.* (1993) commented on the rarity of the highly valued *Microthele* spp. (now *Holothuria whitmaei* and *H. fuscogilva*) and *Thelenota ananas* at Ashmore Reef compared with the Rowley Shoals. Other high value species *Holothuria scabra* and *Holothuria lessoni* (referred to in previous surveys as *H. aculeata* or *H. timana*) have not been recorded since at least 1987 (Russell & Vail 1988) and in the case of *H. scabra* perhaps as far back as 1978 (Marsh *et al.* 1993).

Our survey was conducted in June 2019 to assess and monitor the latest status of holothurian, clam and trochus populations at Ashmore Reef. In this chapter we also review the status and temporal trends in abundance of both harvested and non-target echinoderm and molluscan fauna at Ashmore Reef.

## 11.3 Methods

### 11.3.1 Stratification, habitat types and site selection

The abundance of non-cryptic echinoderms and molluscs were determined at 224 sites, which included the main 216 sites described in chapter 8 and seven additional sites surveyed opportunistically using the same methods. All echinoderms were surveyed apart from fully concealed types like brittlestars and buried or concealed taxa like heart urchins and some holothurians. Molluscs surveyed include clams, pearl oysters, trochus and other large conspicuous gastropods. Survey sites were stratified among three strata that incorporated reef edge as a distinct habitat and four habitat types within the interior of the reef:

1. reef edge strata – upper slope down to a depth of 9 – 12 m and not including the crest surf zone. This habitat was further stratified by northern reef edge (n = 21 sites) and southern edge sites (n = 20)
2. Reef interior stratum, which included the following habitat types:
  - shallow intertidal reef flat (n = 68)
  - shallow intertidal lagoon sand flat (n = 66)
  - shallow subtidal lagoon (n = 38)
  - deep lagoon/bommie habitats (n = 11).

The proportion of sample effort among the strata was designed to support abundance estimation across a suite of holothurian species as well as *Rochia niloticus* as per recommendations of Hosack and Lawrence (2013b). Distribution of habitat types and survey sites are shown in Figure 85 and latitudes and longitudes are found in Figure 85. The average depth surveyed in each habitat type is given in Figure 86.

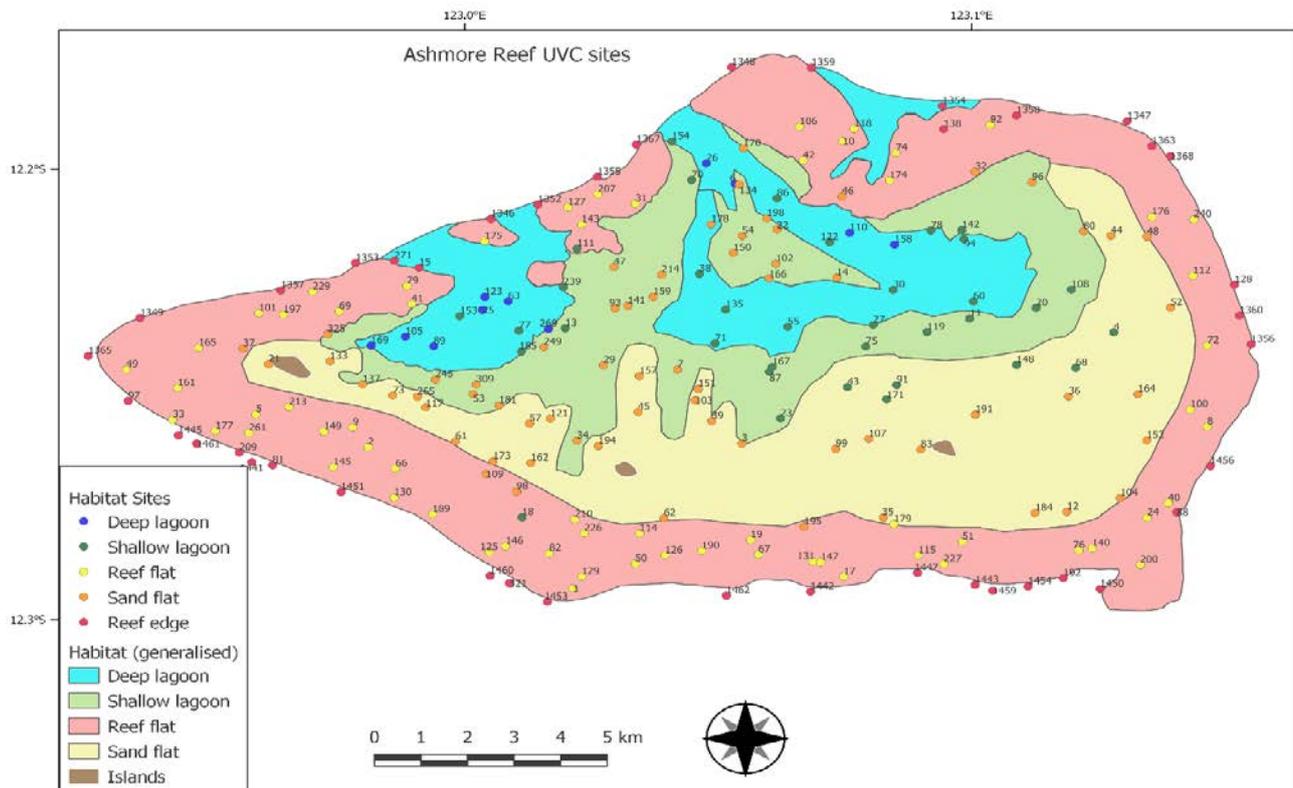


Figure 85. Map of the location of sites surveyed at Ashmore Reef in June 2019 together with habitat stratification.

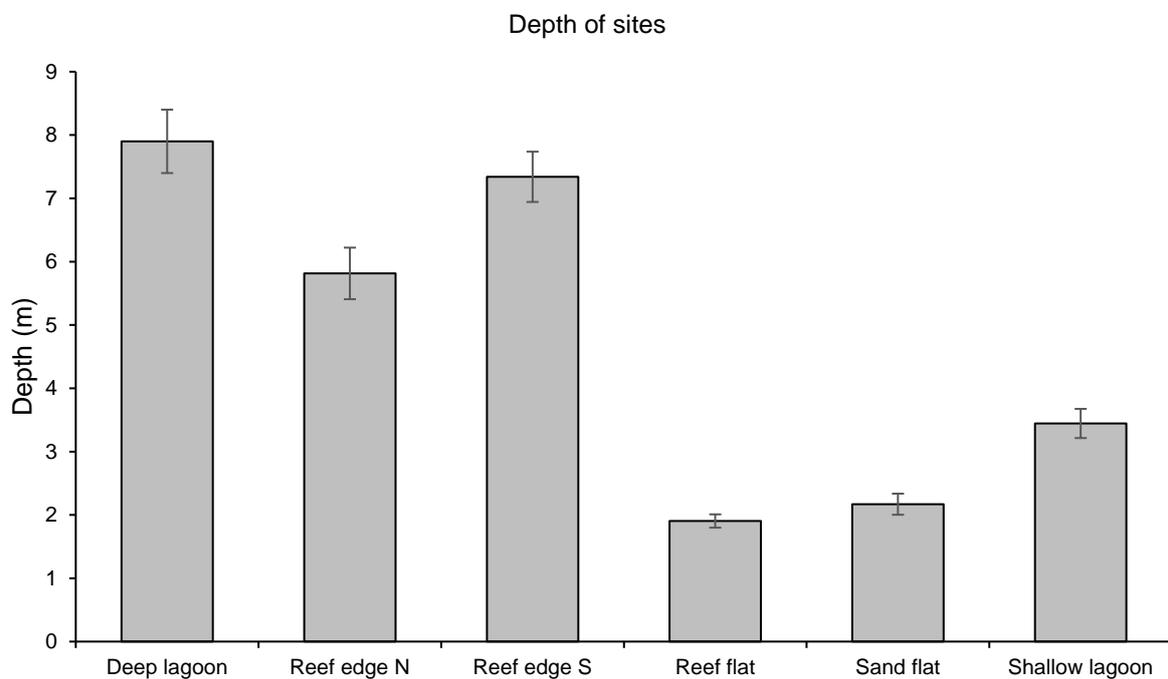


Figure 86. Mean depth of sites in each of the habitat types surveyed, at the time of survey. In general, sites on the reef flat, sand flat and shallow lagoon were surveyed between the mid- to high-tide range, as navigation is not possible at low tides. Reef edge and deep lagoon sites could be surveyed on most tides. Error bars are  $\pm 1$  standard error (s.e.).

### 11.3.2 Holothurians and other echinoderms

Holothurians and all other emergent echinoderms (starfish, sea urchins and emergent or partly emergent crinoids) were counted along 100 m<sup>2</sup> transects (50 m x 2 m). Holothurians buried in sand (dendrochirote species), or those completely hidden under dead slabs of coral, were not included in the counts. Holothurians and crinoids were also counted during the manta tow surveys; burrowing sea urchin habitat was noted during the manta tows.

All survey methodologies have limitations. Manta towing is not an effective method for surveying small starfish or small echinoids (except *Echinometra* which occupy distinctive burrows in the reef) but is effective for large starfish species such as *Linckia laevigata* and *Linckia guildingi*. While neither technique is effective for surveying brittle stars or crinoids concealed beneath dead coral slabs, partly exposed crinoids can be effectively counted by both methods.

### 11.3.3 High density aggregations of *Holothuria leucospilota*

High-density aggregations of *H. leucospilota* had previously been recorded in the eastern lagoon from 2001 to 2006 (Smith *et al.* 2001; Rees *et al.* 2003; Kospartov *et al.* 2006; Ceccarelli *et al.* 2007) with Ceccarelli *et al.* (2007) recording densities up to 190,000 ha<sup>-1</sup>. The opportunity was therefore taken to revisit these sites to check whether the high-density aggregations remained.

The sites visited were those presented in Ceccarelli *et al.* (2007) and were surveyed using a manta tow approach (Figure 89). A diver entered the water at a previously recorded waypoint of known high *H. leucospilota* densities and was towed to adjacent waypoints at a tow speed averaging 2 – 3 knots. During the tow, the diver continuously scanned the seafloor for *H. leucospilota* individuals, as well as any other holothurians, using a belt width of approximately 10 m. The depth of the seafloor ranged from 2–4 m throughout the search area.

Three tows and one spot-check were conducted. For the first tow, a diver entered the water at site L20, towed through sites L19 to L12, and towards L9 in a single continuous transect of approximately 3.5 km over a 20-minute period (Figure 89). For the second tow, a diver was dropped in the water at site L12, towed north, and in a loop back to L13 in a 2 km transect over a 20-minute period. The final tow was conducted from site L7 to L25 in a shorter 1.4 km transect, and another spot check was then conducted at site L27.

### 11.3.1 Crown-of-thorns starfish *Acanthaster planci*

Crown-of-thorns starfish are of special interest as they are an important coral predator, so in addition to the 100 m<sup>2</sup> transects, these starfish were surveyed during manta tows and observed specimens were counted. Any recently dead coral or coral scars typical of crown-of-thorns starfish feeding were also recorded.

### **11.3.2 *Linckia* spp.**

*Linckia laevigata* and *Linckia guildingi* are also of special interest as they are large, conspicuous coral reef starfish which have been counted on previous surveys. They were counted on both the manta survey and the 100 m<sup>2</sup> invertebrate transects. The smaller *Linckia multifora* was only counted on the 100 m<sup>2</sup> invertebrate transects.

### **11.3.3 Trochus**

Trochus (*Rochia niloticus* = *Tectus niloticus*) were counted and measured during invertebrate surveys at all 224 sites.

### **11.3.4 Clams**

Clams (*Tridacna* spp. and *Hippopus hippopus*.) were counted during both manta tow and on the 100 m<sup>2</sup> invertebrate transects. Clams encountered during invertebrate transects were also measured.

### **11.3.5 Pearl oysters**

Pearl oysters (*Pinctada* spp.) were counted during invertebrate surveys only as manta towing is not effective for surveying pearl oysters on reefs.

### **11.3.6 Comparison of trends between years**

Long term trend analyses of holothurian, trochus and clam densities at Ashmore Reef have been undertaken by Ceccarelli *et al.* (2011a) and Hosack and Lawrence (2013a). Both studies encountered difficulties in comparing between surveys because of different methodologies, habitat stratification and search effort. These problems were also identified by Breen (2011). Through careful and detailed analysis, Hosack and Lawrence (2013a) were able to make comparisons across most of the existing datasets, resolving the difficulties mentioned above, but with the result that predictions of species abundance for each dataset carried a large variance.

We took four approaches to examining population trends on a species by species basis. Firstly, we relied on the statistical comparisons made by Hosack and Lawrence (2013a) of the surveys between 1998 and 2013 and the 2013 vs 2019 comparison by Hosack *et al.* (chapter 12 of this study) and we compare these with the earlier long term trend analysis made by Ceccarelli *et al.* (2011a) of the 1998 to 2006 surveys. Secondly, we attempted to make qualitative comparisons of densities using the most “like with like” comparisons of historical data with our 2019 data. At the time, Breen (2011) considered the 1998 (Skewes *et al.* 1999a) surveys (Figure 87) to provide the most statistically robust dataset. Hosack and Lawrence (2013a) Hosack and Lawrence (2013a) demonstrated that the 1998 dataset stratified by habitat type (Figure 142) was the most comparable to the currently established monitoring survey methodology for Ashmore Reef designed by Hosack and Lawrence (2013a) and tested in 2013 by Ceccarelli *et al.* (2013) and used in this study in 2019. Fortunately, this means the oldest and the newest datasets can be directly compared.

Thirdly, given that by 1998, holothurians, trochus and clams had been heavily over-exploited, we wanted to look for evidence of recovery or further decline in abundances in the data from surveys that followed. To do this, surveys based on discrete spatial comparisons through time were made using data from 2006 (Ceccarelli *et al.* 2007) and 2009 (Richards *et al.* 2009). These data were compared with a subset of the 2013 and 2019 data collected close to the same locations as the 2006 and 2009 surveys (Figure 88). In doing so, we have attempted to minimise the shortcomings of surveys which used different methodologies and the consequential difficulties in comparing them identified by Breen Breen (2011) and Hosack and Lawrence (2013a). Nevertheless, these problems preclude formal statistical comparisons, so these have not been attempted. Fourthly, we examined both qualitative and quantitative information on abundance of holothurians at Ashmore Reef prior to, or during the mid-late 1980s when exploitation by Indonesian fishers is believed to have been at its highest. Most of this information comes from interviews with fishers and abundance surveys in 1986 and 1987 by Russell and Vail (1988) and frequency of observation by Marsh *et al.* (1993) in 1987. The location of sites surveyed by Russell and Vail (1988) is shown in (Figure 88).

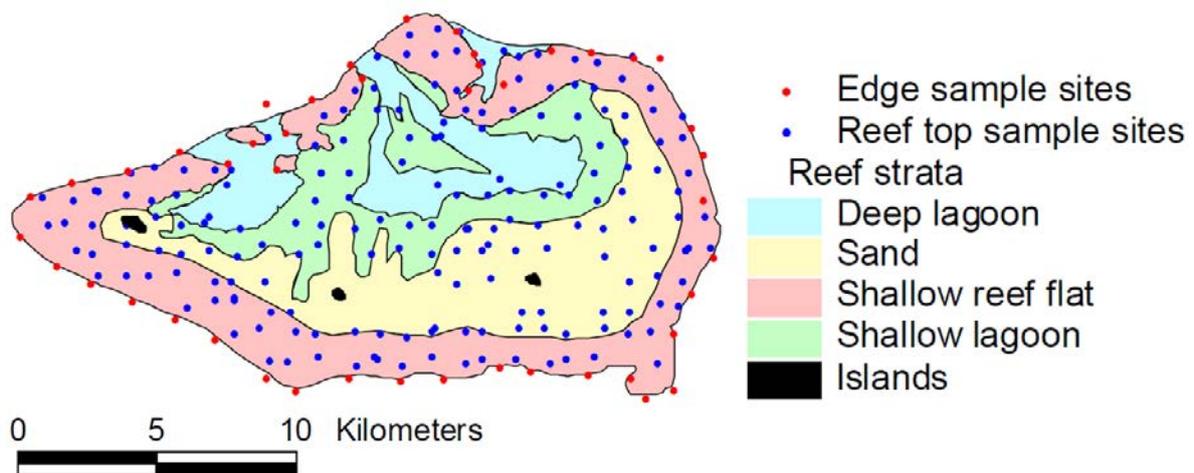


Figure 87. Map of habitat stratification of Ashmore Reef and location of sampling sites taken from figure 2 of Skewes *et al.* (1999a).

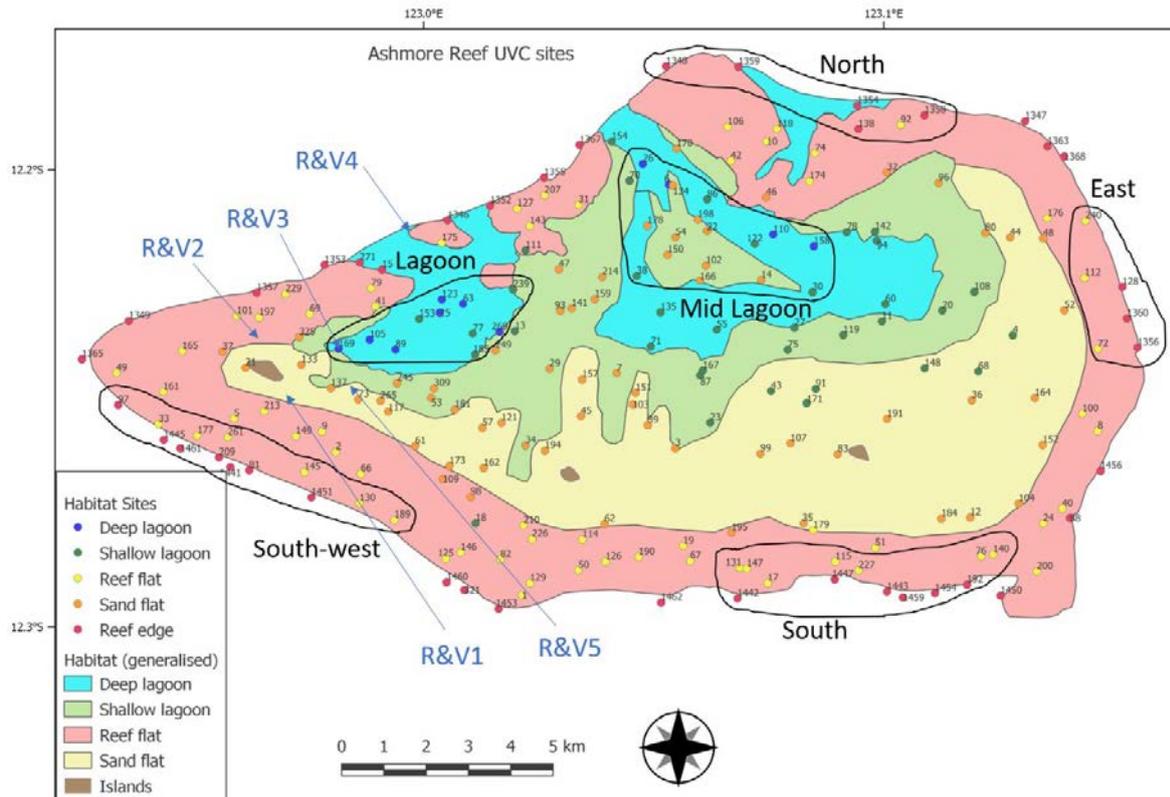


Figure 88. Map of the matchup of the 2019 survey sites (numbered) and the six location-based 2006 and 2009 survey sites used by Ceccarelli et al. (2007) and Richards et al. (2009) labelled South, North, etc. The black areas circled indicate which of the 2019 sites were used to compare with the 2006 and 2009 survey data. Five sites surveyed by Russell and Vail (1988) are indicated as R&V# in blue.

## 11.4 Results

### 11.4.1 Holothurians

#### Species diversity

A total of 18 species of holothurians were found during the 2019 survey: *Actinopyga lecanora* (reef sea cucumber), *A. mauritiana* (Mauritian sea cucumber), *A. miliaris* (military sea cucumber), *A. palauensis* (Palauan sea cucumber), *Bohadschia argus* (eyed sea cucumber or leopard fish), *Holothuria atra* (black sea cucumber), *H. coluber* (snake sea cucumber), *H. edulis* (unsavoury sea cucumber), *H. fuscogilva* (white teatfish), *H. fuscopunctata* (black-banded sea cucumber), *H. leucospilota* (stained sea cucumber), *H. rigida* (rigid sea cucumber), *H. whitmaei* (black teatfish), *Pearsonothuria graeffei* (Graeffe's sea cucumber), *Stichopus chloronotus* (green sea cucumber), *S. herrmanni* (Herrmann's sea cucumber), *Thelenota ananas* (prickly redfish), *T. anax* (royal sea cucumber). *Actinopyga palauensis* (sites 269, 353 and 358) and *Holothuria rigida* (site 49) are new records for Ashmore Reef.

Of these 18 species, two; *Thelenota anax* and *Bohadschia argus*, did not occur at any 224 survey sites (mostly 0 – 9 m deep) but they were observed in the deep lagoon habitat 10 – 20 m deep at the base of lagoon bommies. In addition, species that were rare on the 100 m<sup>2</sup> transects such as *Stichopus herrmanni*, *Thelenota ananas* and *T. anax* were more commonly

observed on the much longer two-minute manta tow transects off the reef edge in deeper water (Figure 90). On one two-minute tow around the deep hole in the reef flat known as the “Grotto” (manta tow 23 – 24, see Figure 90) we recorded five *T. ananas* which was more than was recorded in all the 224 100 m<sup>2</sup> transects.

#### High density aggregations of *Holothuria leucospilota*

In June 2019, no *H. leucospilota*, nor any other holothurian species, were detected at any the 23 sites where they had previously existed at high densities in 2005 and 2006, nor along the manta tows conducted between these sites (Table 26, Figure 89). These areas are shallow, sandy bottom near the sand cays and it is thought that these reef habitats which previously supported high densities of *H. leucospilota* have been buried in shifting sand movement rendering them unsuitable.

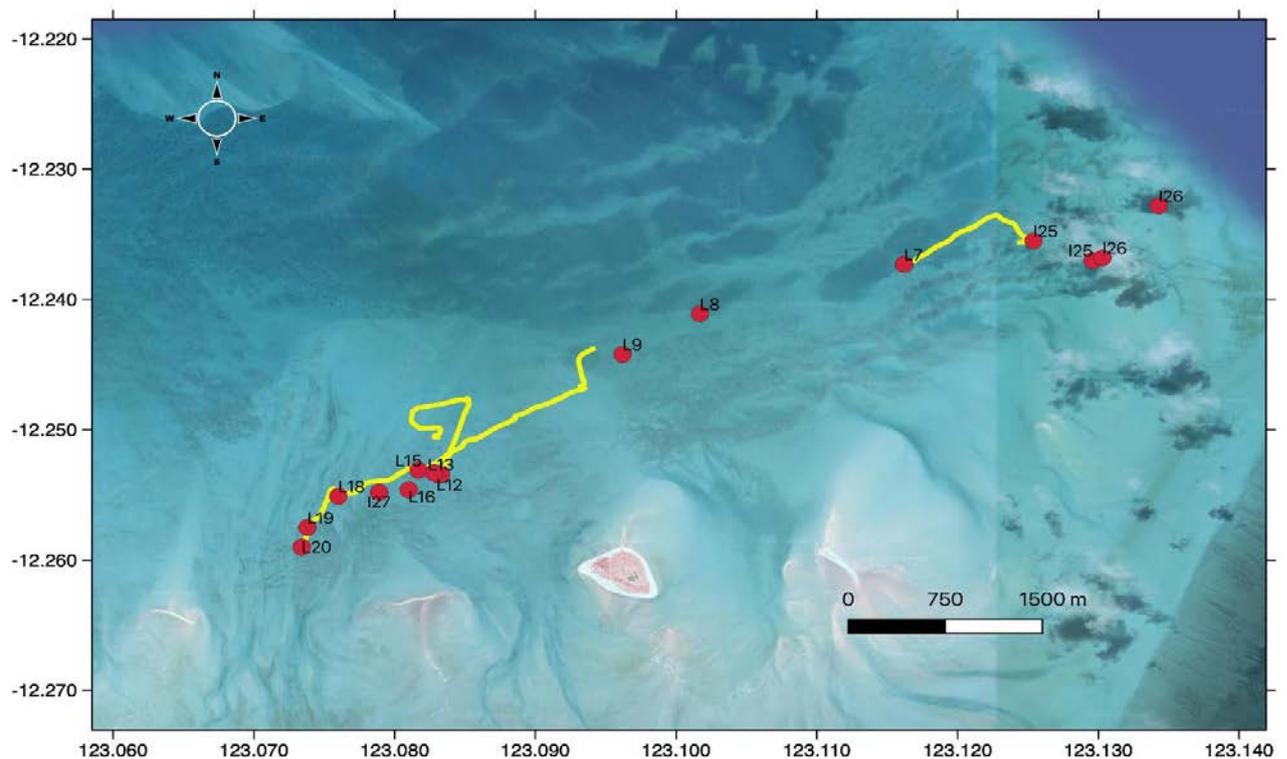


Figure 89. Manta tow tracks (yellow lines) and sites of high-density *H. leucospilota* aggregation reported in 2005 and 2006 by Ceccarelli et al. (2007) (red dots) near East Island.

Table 26. Historic and current records for previously known high density *H. leucospilota* aggregations (data for 2005 and 2006 from Ceccarelli et al. 2007).

Site	Estimated density per hectare		
	2005	2006	2019
I25	4	2	0
I26	88	6	0
I27	4,410	6,190	0
L1	present	not visited	0
L2	present	not visited	0
L3	present	not visited	0
L4	present	not visited	0
L5	present	not visited	0
L6	present	not visited	0
L7	0	0	0
L8	0	0	0
L9	0	0	0
L10	112	308	not visited
L11	112	308	not visited
L12	350	189,500	0
L13	80,000	high density	0
L14	224,333	high density	0
L15	high density	not visited	0
L16	high density	not visited	0
L17	high density	not visited	0
L18	high density	10,000	0
L19	high density	138,500	0
L20	95,588	50,000	0

### Manta tow surveys for echinoderms

Only 51 echinoderms were counted along the deep edge slopes during two-minute manta tows totalling about 26 km (Figure 90). The species observed were the starfish *Linckia guildingi* (3) and *Linckia laevigata* (8), the sea urchin *Echinothrix diadema* (2), and the holothurians *Holothuria atra* (19), *Holothuria fuscopunctata* (1), *Actinopyga palauensis* (1), *Stichopus chloronotus* (5), *Stichopus herrmanni* (5), *Thelenota ananas* (5) and *Thelenota anax* (2). Most holothurians were observed around the perimeter of the deep lagoon or in the “Grotto” between waypoints 23 and 24 (Figure 90). Large areas of *Echinometra mathaei* burrows were also seen on the exposed southern edge on the crest between waypoints 5 and 6 (Figure 90).

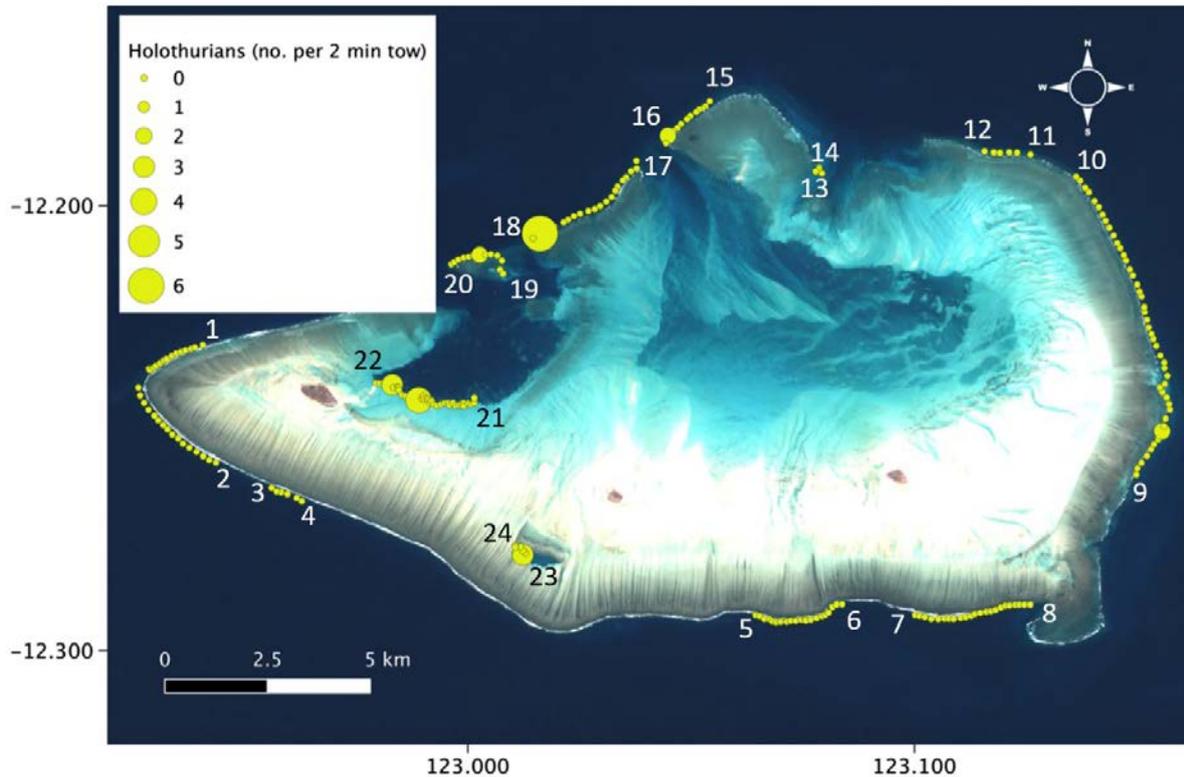


Figure 90. Map of Ashmore Reef showing the number of holothurians recorded throughout the two-minute manta tows. Each two-minute tow is shown as a yellow dot and the numbers refer to waypoints given in the supplementary information Table 13.

## Holothurian distribution and abundance

### Overall density and distribution

When all species are considered together, the highest density of holothurians were found on the reef flat (1.26/100 m<sup>2</sup>) and along the sheltered northern reef edge (1.05/100 m<sup>2</sup>), while the deep lagoon, shallow lagoon and sand flats had the lowest densities (ca. 0.3/100 m<sup>2</sup>)(Figure 91). The variability in total and individual species distributions across the 224 transects sampled (Figure 92) shows that at least this number of sites is required to gain reliable estimates. When sampling effort is lower, as it was in 2013 due to logistical constraints, even abundant species are not adequately captured (Figure 93).

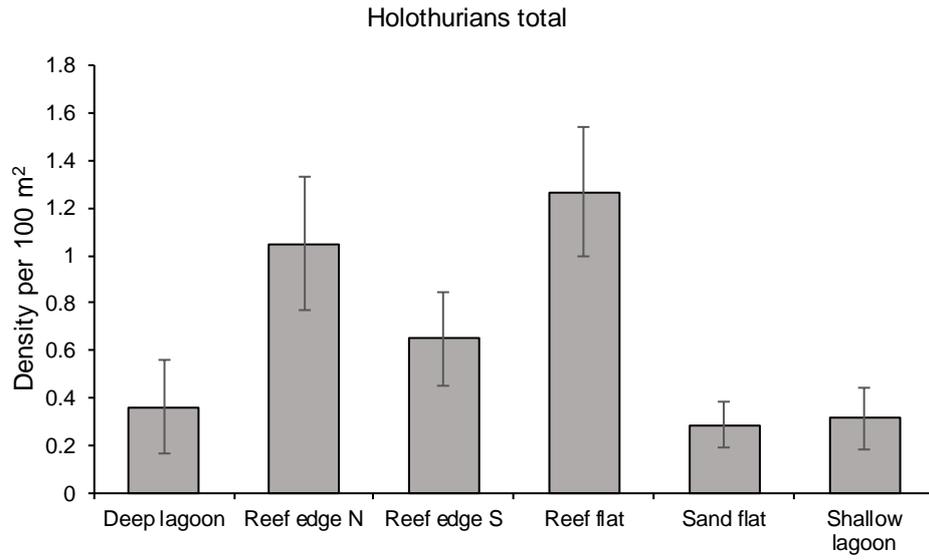


Figure 91. Mean density of holothurians (all species combined) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

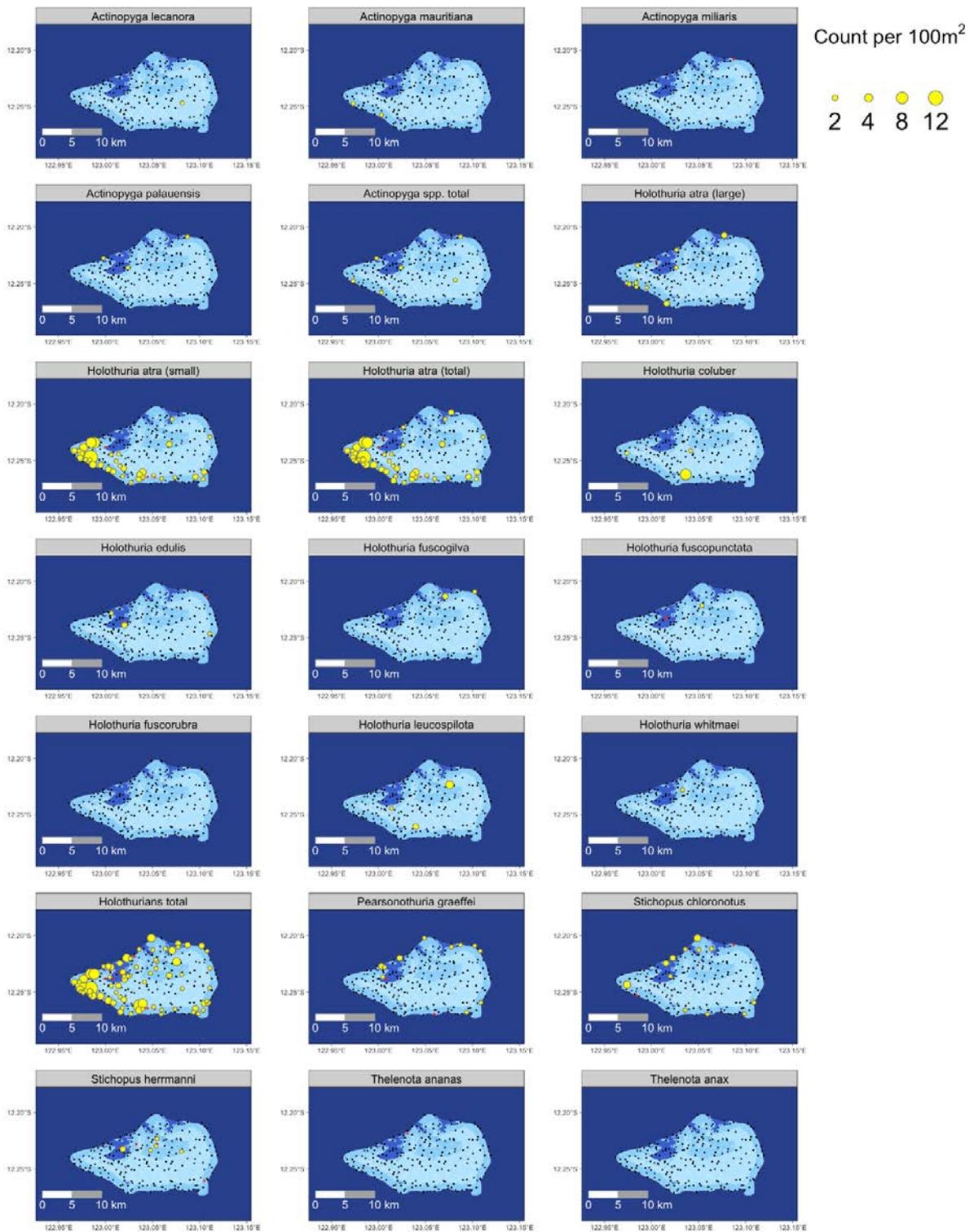


Figure 92. Distribution and density/100 m<sup>2</sup> of all holothurian species across each of the 224 transects surveyed in June 2019. The red dots indicate presence of that species at that site but that the specimen(s) were not within the area of the 100 m<sup>2</sup> quantitative survey transect.

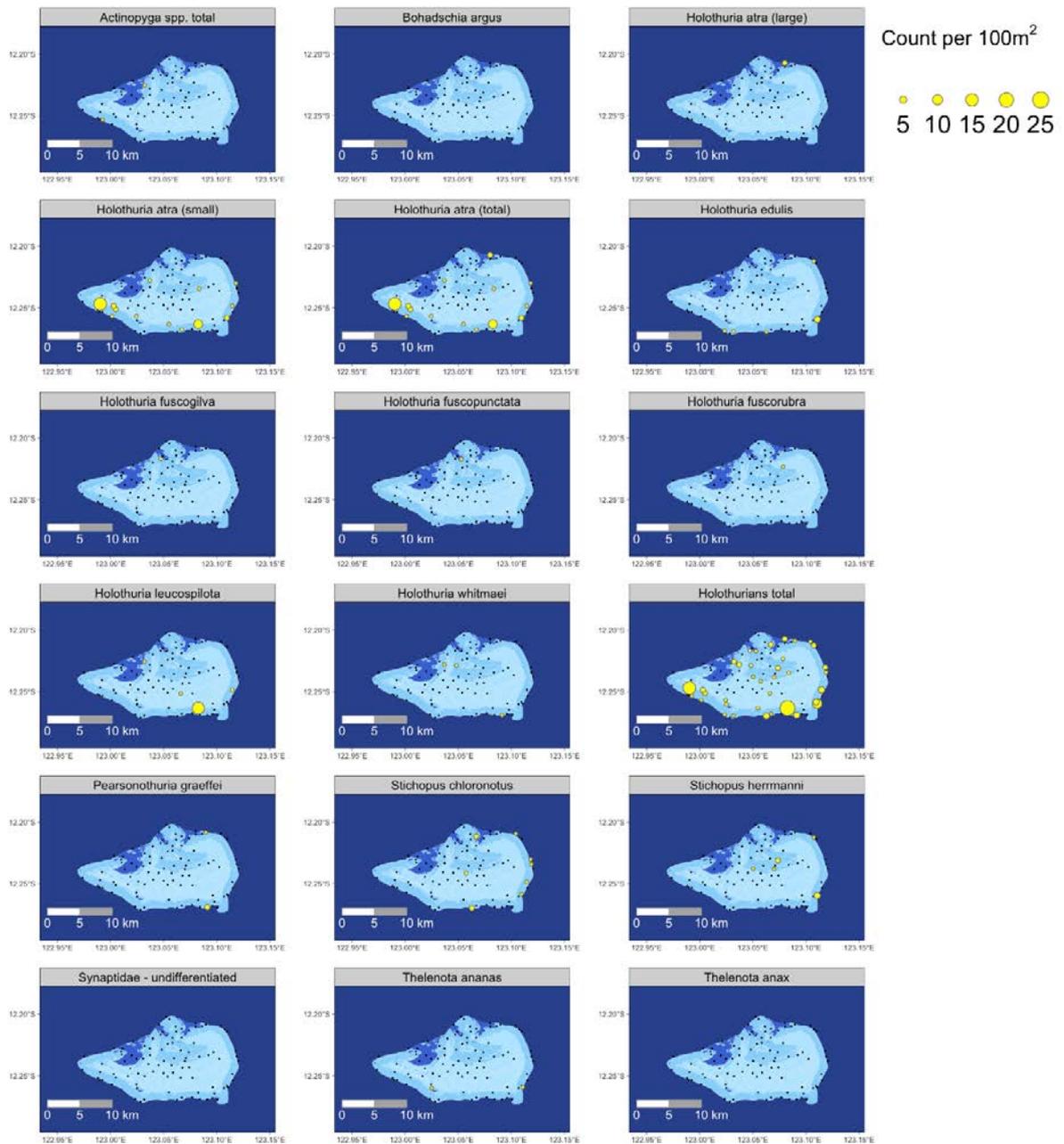


Figure 93. Distribution and density/100 m<sup>2</sup> of all holothurian species across each of the 95 transects surveyed in March 2013 Data from Ceccarelli et al. (2013) with densities converted to per 100 m<sup>2</sup> from their 80 m<sup>2</sup> transects.

### ***Holothuria atra* (black sea cucumber) density and distribution**

*Holothuria atra* is typically the most abundant sea cucumber on Indo-Pacific coral reefs (Conand 1996). There are two recognisable forms, and these were surveyed separately. This high abundance on most reefs can, in part, be explained by this species having one form which reproduces asexually (small, sand covered) as well as a sexually reproductive form (larger, shiny black, not sand covered). The latter are generally found sub-tidally at a low density and the asexual form being more abundant and occupying predominantly shallow intertidal habitats. Our results from Ashmore Reef were consistent with this pattern. Large, sexually reproducing *H. atra* were most abundant on the southern reef edge (0.25/100 m<sup>2</sup>) while the asexual form was most abundant on the reef flat (0.96/100 m<sup>2</sup>) (Figure 94).

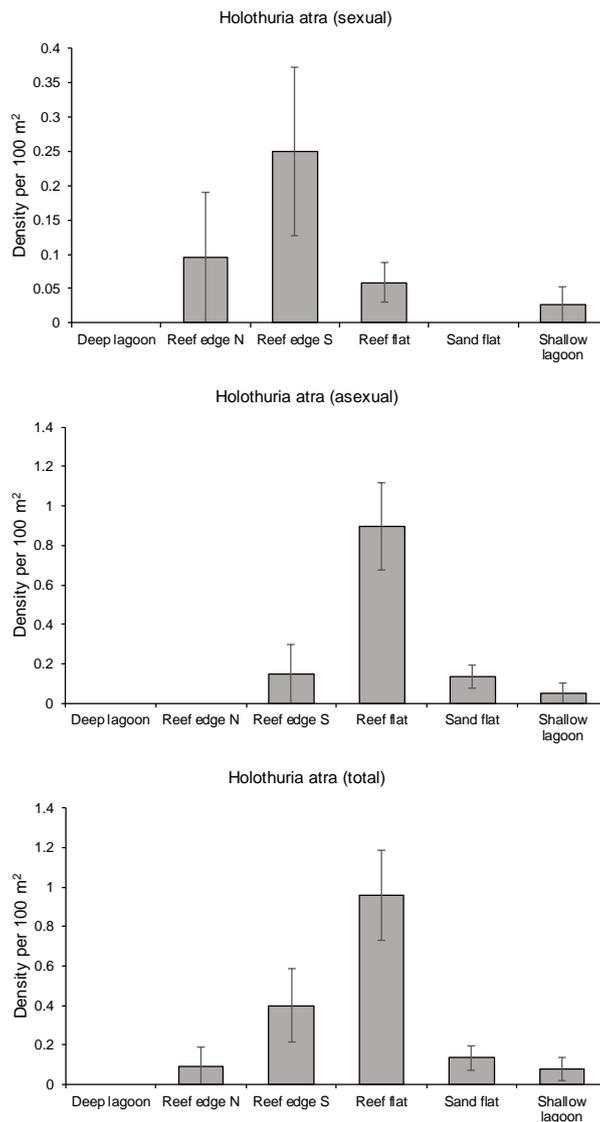


Figure 94. Density of *Holothuria atra* in each of the habitats surveyed at Ashmore Reef in June 2019. Top panel shows the large sexually reproducing form, middle panel shows the small asexually reproducing form and the bottom panel shows the combined total. Error bars are ±1 s.e.

### Historical trends in density

*Holothuria atra* was the most abundant holothurian at Ashmore Reef and despite being regarded as a low value species, often made up more than 80% of holothurian harvests from Ashmore Reef by Indonesian fishers in the mid-1980s (Russell & Vail 1988), with many thousands being taken (Figure 95).



Figure 95. Plate 2 reproduced from Russell and Vail (1988) showing holothurians (mostly *Holothuria atra*) being dried on board an Indonesian perahu at Ashmore Reef in 1986/87. Photograph: Lyle Vail.

Long term trend analyses at the whole of Ashmore Reef level have been undertaken by Ceccarelli *et al.* (2011a) and Hosack and Lawrence (2013a). Their results indicated that abundance and density may have declined between 1998 – 2003 and 2005 and recovered to 1998 levels by 2013 (Figure 96). Hosack et al. (chapter 12 in this study) found abundance had decreased between 2013 and 2019, but not significantly. Comparisons by habitat and by site level (Figure 97) indicate that in most comparisons the survey results from 2013 and 2019 were not dissimilar to results obtained in 1998. The surveys undertaken in 2019 at the South-west and North sites were appreciably greater than those in 2006 and 2009. On the Great Barrier Reef between 9% (reefs around continental islands) and 76% (offshore reefs) of *Holothuria atra* have been found to undergo fission (asexual reproduction) annually (Uthicke 1997). This alone can account for fluctuations in density between surveys since 1998 and should ensure populations recover from fishing once protected. The densities recorded on the reef flat at Ashmore Reef in 2019 are similar to those recorded from sites of similar habitat by Russell and Vail (1988) in 1987 (Figure 96).

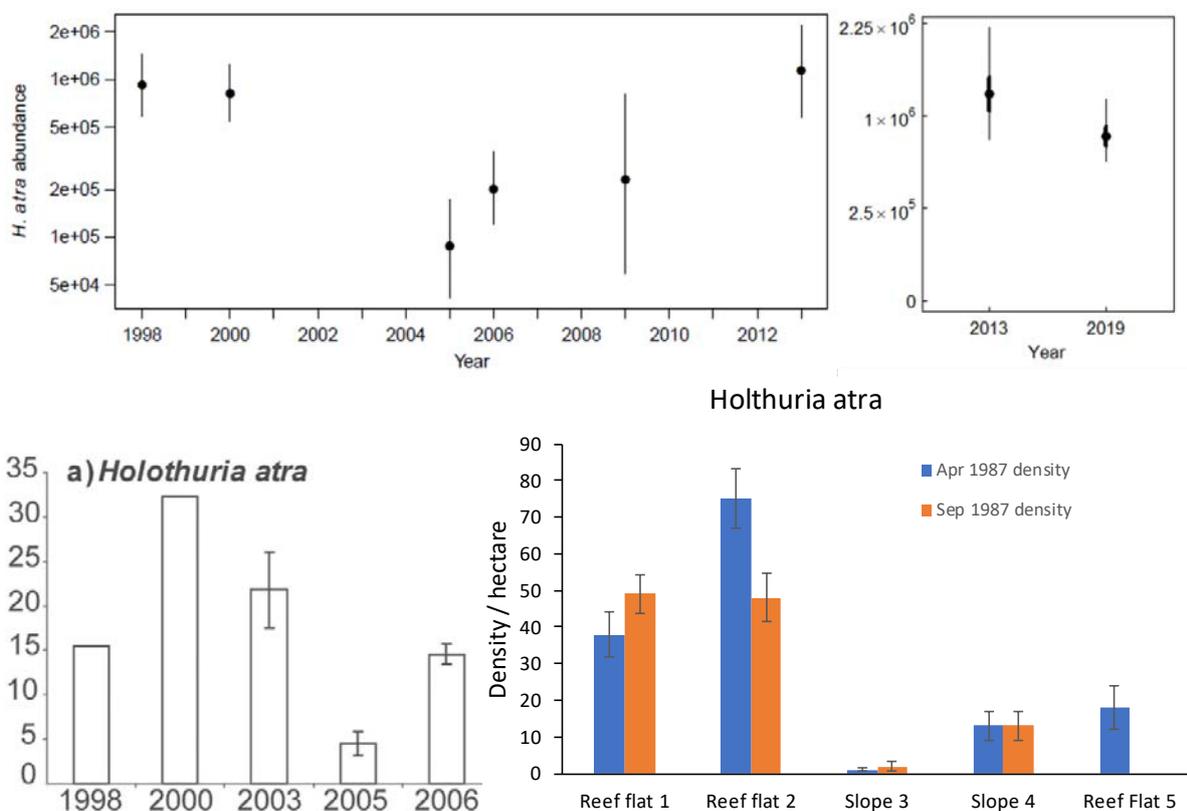
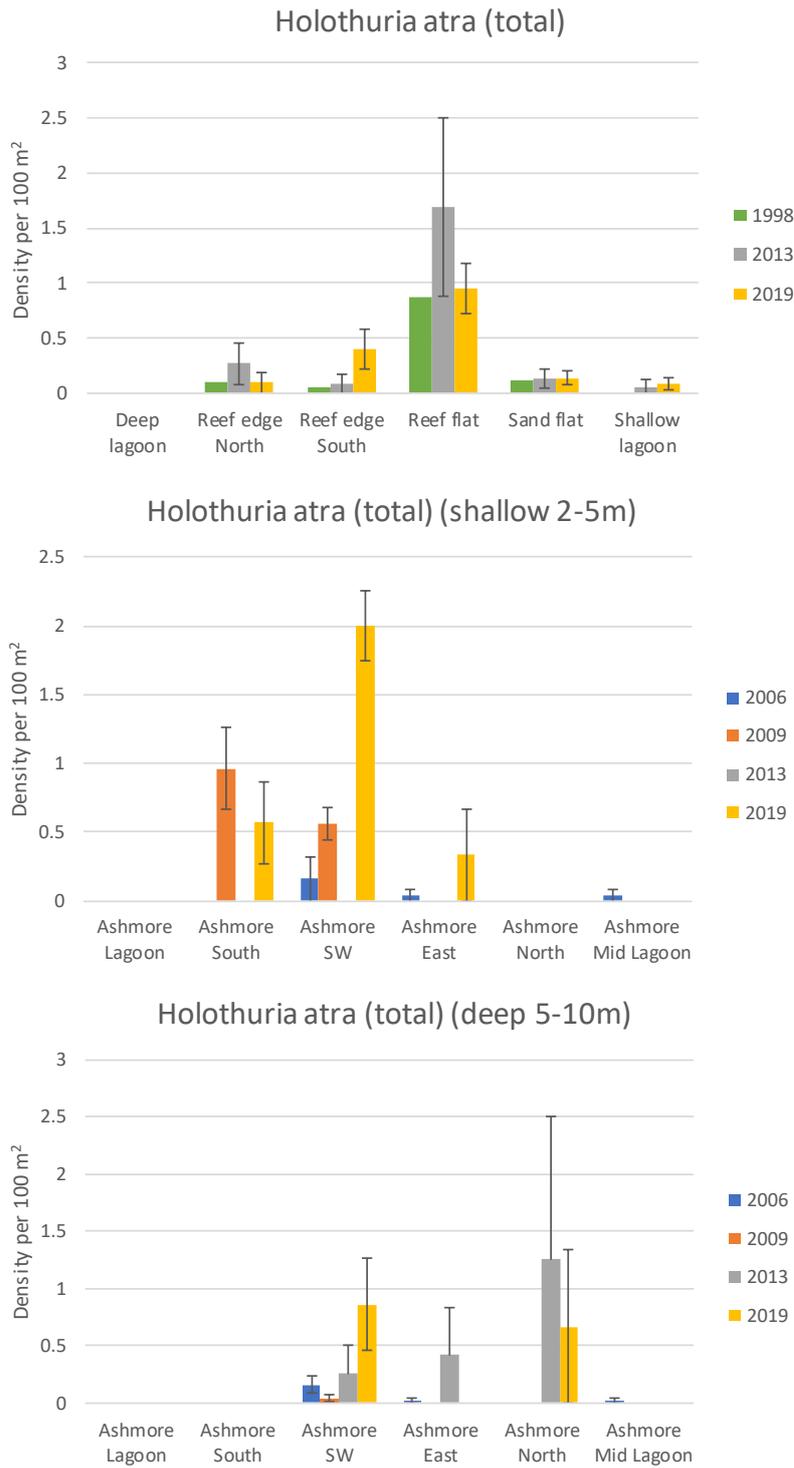


Figure 96. Long term estimates of abundance of *Holothuria atra*. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both panels. Values are medians and longest error bars are 95% confidence limits. Lower left panel is from Ceccarelli et al. (2011a) with average or mean ( $\pm 1$  s.e.) densities for the whole of Ashmore Reef expressed per hectare. Lower right panel are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.



**Figure 97.** Comparisons of historical and recent surveys of *Holothuria atra* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.

### **Other *Holothuria* spp. density and distribution**

Other than *Holothuria atra* (see above), we recorded seven other species of the *Holothuria* genus. The densities for each in each habitat are shown in Figure 98 except for *Holothuria rigida* (rigid sea cucumber) for which a single specimen located incidentally to the quantitative transects. The distribution of these density estimates across the whole reef for each species is shown in Figure 92.

*Holothuria fuscogilva* (white teatfish) and *Holothuria whitmaei* (black teatfish) are two high-value commercial species known as “teatfish”. In particular, *H. whitmaei* exhibited very low densities. Indeed, the density of 0.015/ 100 m<sup>2</sup> on the reef flat (Figure 98) was based on a single specimen, and we only observed two specimens on the reef during the entire trip. *Holothuria fuscogilva* occurred on both the reef flat and the northern reef edge, also at low densities of 0.03 – 0.05/ 100 m<sup>2</sup> (Figure 98). In general, *H. whitmaei* occurs in shallow reef flat waters and *H. fuscogilva* in deeper lagoon and back reef waters (Smith *et al.* 2001).

*Holothuria coluber* (snake sea cucumber) occurred on the reef flat (0.12/100 m<sup>2</sup>) and the sand flat (0.02/100 m<sup>2</sup>) (Figure 98).

This species anchors its posterior end under a rock and extends its body to feed, unlike many reef flat dwelling holothurians. It can quickly retract its body if disturbed and is difficult to dislodge from the reef.

*Holothuria leucospilota* (stained sea cucumber) can be extremely abundant on some reefs, but we recorded low densities of 0.11/100 m<sup>2</sup> on the reef flat and 0.05/100 m<sup>2</sup> on the sand flat (Figure 98).

*Holothuria edulis* (unsavoury sea cucumber) was also uncommon at Ashmore Reef with densities ranging from 0.01/100 m<sup>2</sup> on the reef flat to 0.05/100 m<sup>2</sup> on the sand flat and in the shallow lagoon (Figure 98).

*Holothuria fuscopunctata* (black-banded sea cucumber) occurred only on the sand flat (0.12/100 m<sup>2</sup>) although an incidental observation was also made of this species at the base of a bommie in the deep lagoon.

Neither *Holothuria scabra* (sandfish) nor *Holothuria lessoni* (lesson’s sea cucumber, also known as golden sandfish) referred to as *H. aculeata* and *H. timana* in some previous studies) were observed in our 2019 surveys.

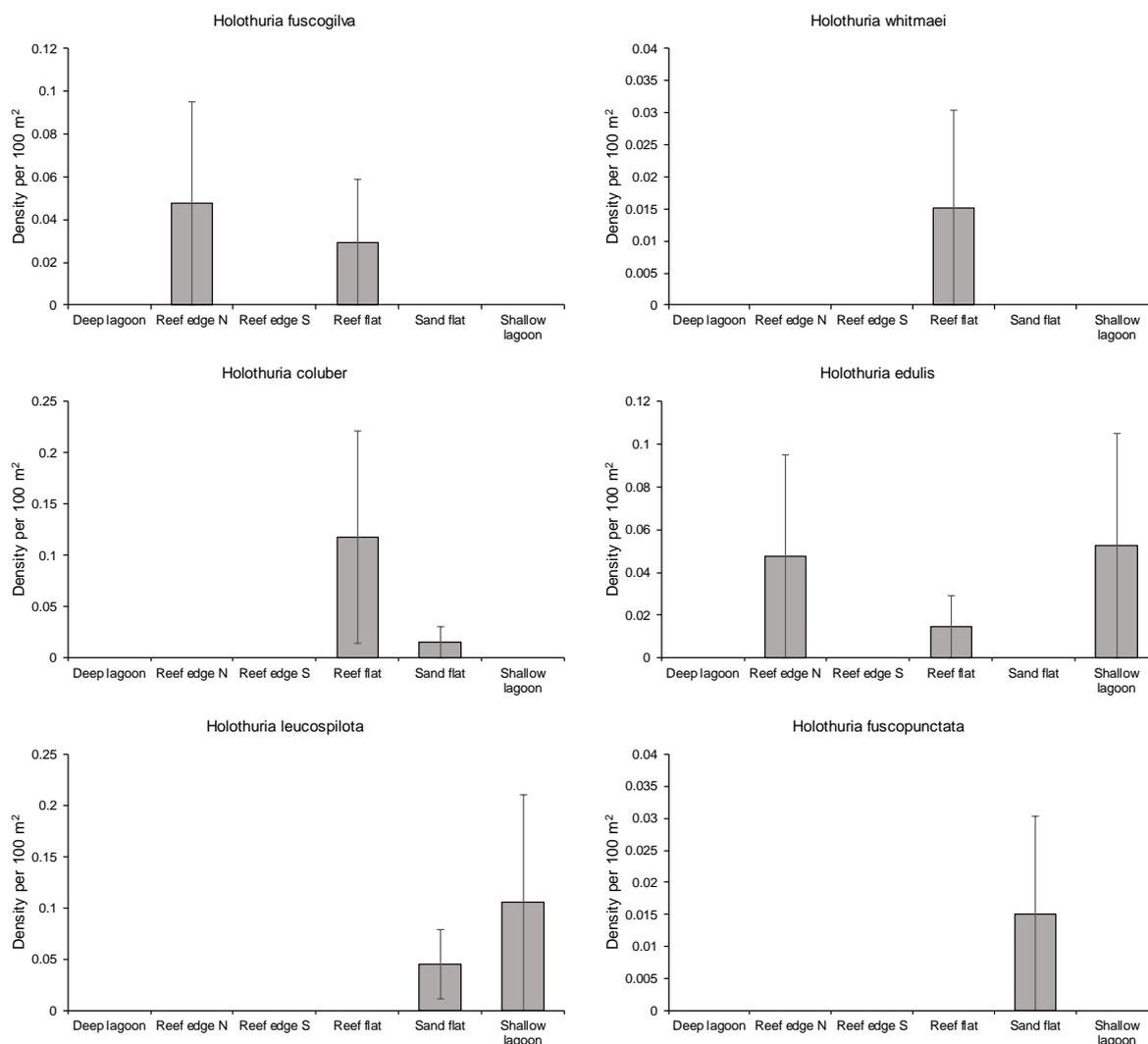


Figure 98. Mean density of holothurians (*Holothuria spp.*) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

### ***Holothuria whitmaei* (black teatfish) historical trends in density**

*Holothuria whitmaei*, referred to *Holothuria nobilis*, in older studies is one of the highest value and most heavily exploited holothurian species (Kinch *et al.* 2008) and recovery from overfishing can be negligible even after many years of protection (Uthicke, Welch & Benzie 2004). Marsh *et al.* (1993) and Berry (1993) commented on their rarity at Ashmore Reef in 1987, where they only found four individuals, compared to Rowley Shoals where they were unfished and very common. Shiell (2005) compared populations of *Holothuria whitmaei* at Ashmore Reef and other heavily fished reefs with the more pristine Mermaid and Ningaloo reefs where he found this species occurred at densities of 9 – 27/ha in suitable shallow water habitats. On the other hand, the species can be naturally uncommon. Bellchambers *et al.* (2011) found *H. whitmaei* was rare at Cocos Islands where fishing for holothurians has been historically negligible. The reports of surveys at Ashmore Reef between 1998 and 2003 all commented on the rarity of *H. whitmaei* (Skewes *et al.* 1999a; Smith *et al.* 2001; Rees *et al.* 2003). For example, Smith *et al.* (2001) only recorded three individuals. The analysis by

Ceccarelli *et al.* (2011a) shows a continued decline through 2005 and 2006 and they were also very rare in 2009 (Richards *et al.* 2009). Hosack *et al.* (chapter 12 this study) found abundance had decreased between 2013 and 2019, but not significantly. In 2013, only three were recorded and only two in 2019. The high degree of variability around estimates in all surveys from 1998 to 2019 (Figure 99 and Figure 100) suggests that no real conclusions can be drawn about their trend in abundance, except that it is not increasing. It is likely the population has declined to the extent that present distances between individuals means that the population is reproductively ineffective (Kinch *et al.* 2008; Ceccarelli *et al.* 2011a). This is the result what are termed allee effects, where populations of animals at low density are weakened by reduced overall abundance, resulting in greater vulnerability to predation and/or ineffective reproduction (see Bell, Purcell & Nash 2008; Friedman *et al.* 2011). Thus, unless there is larval recruitment from other reefs, which seems unlikely given its isolation, *H. whitmaei* is likely to become locally extinct at Ashmore Reef.

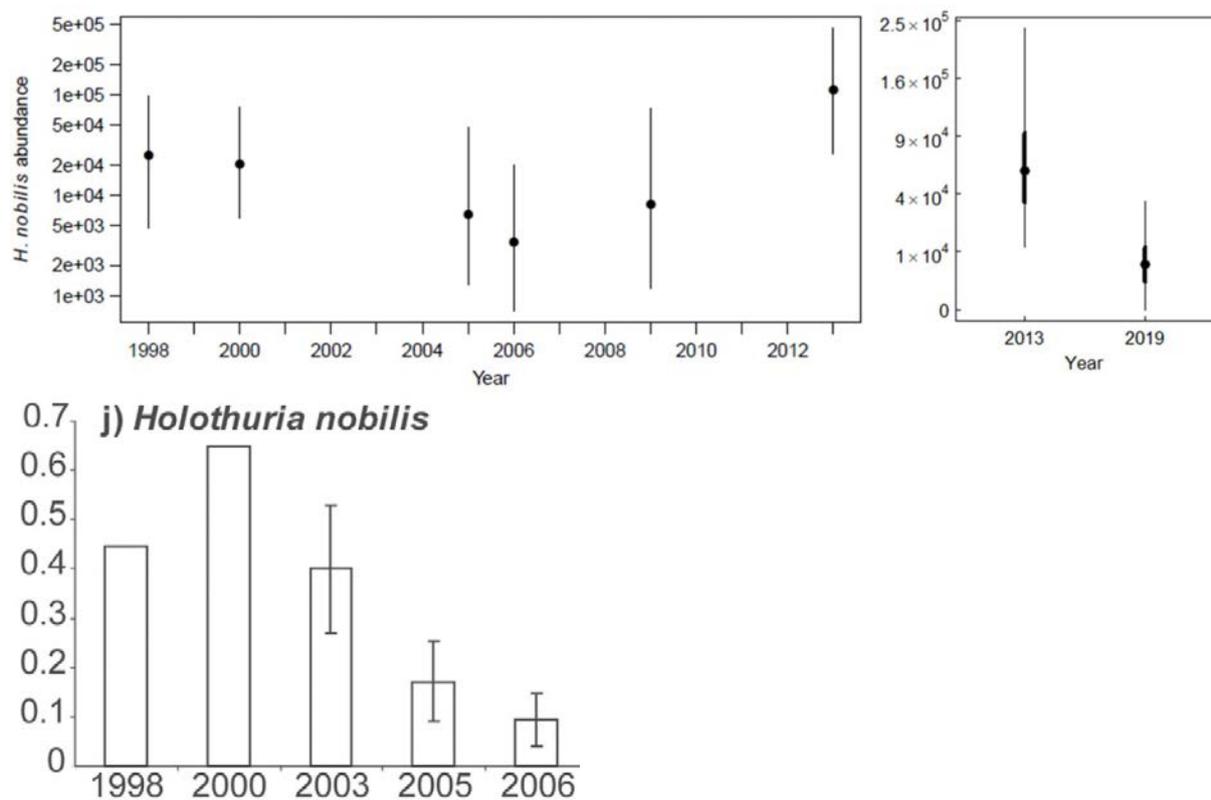
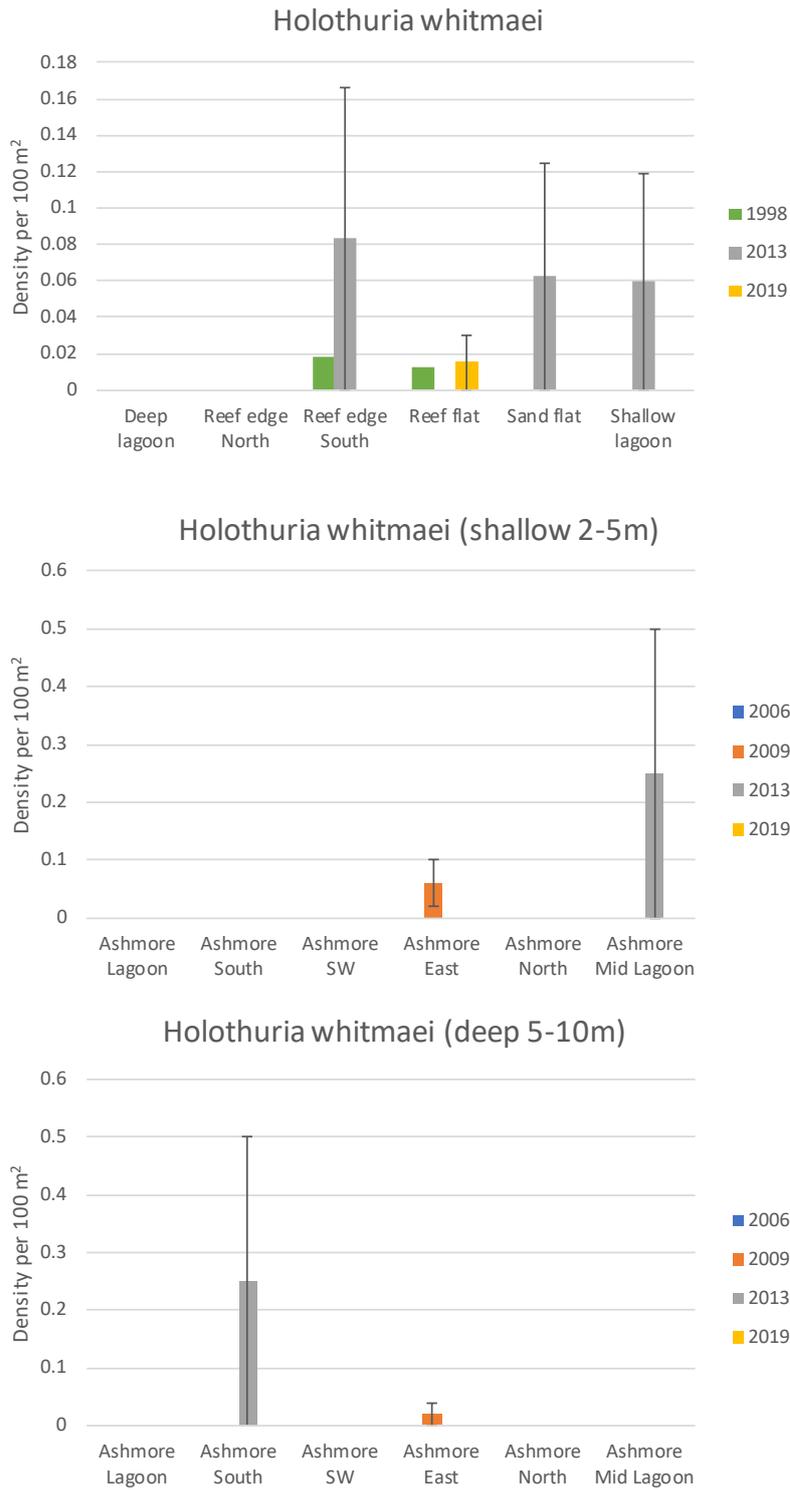


Figure 99. Long term estimates of abundance of *Holothuria whitmaei* (formerly “*nobilis*”). Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack *et al.* in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both panels. Values are medians and longest error bars are 95% confidence limits. Lower panel figure is from Ceccarelli *et al.* (2011a) with average or mean ( $\pm 1$  s.e.) densities for the whole of Ashmore Reef expressed per hectare. See these cited studies for methods.



**Figure 100. Comparisons of historical and recent surveys of *Holothuria whitmaei* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.**

### *Holothuria fuscogilva* (white teatfish) historical trends in density

Long term analysis of trends in *Holothuria fuscogilva* density by Hosack and Lawrence (2013a) indicates high variability and low abundances, revealing no trend between 1998 and 2019, although the posterior median abundance estimate was lower in 2006 compared to 1998 (Figure 101). The analysis by Ceccarelli *et al.* (2011a) also suggests a decline between 1998 and 2003 – 2006 (Figure 101). Densities in 2013 and 2019 compared to either 1998 (by habitat) or 2006 (by location) show no change (Figure 102). However, Hosack *et al.* (chapter 12 this study) found the posterior median abundance estimate had increased between the two most recent surveys in 2013 and 2019. Only one *H. fuscogilva* was recorded in 2013 and we only recorded three in 2019, however neither of these surveys covered the deep-water habitats effectively. Smith *et al.* (2001) reported that *H. fuscogilva* was relatively plentiful in deeper lagoon and channel entrances at depths greater than 15 m. It is likely that this species has not been adequately surveyed at Ashmore Reef in the past and there remains a need to add deep-water habitats at Ashmore Reef to the survey design developed by Hosack and Lawrence (2013b).

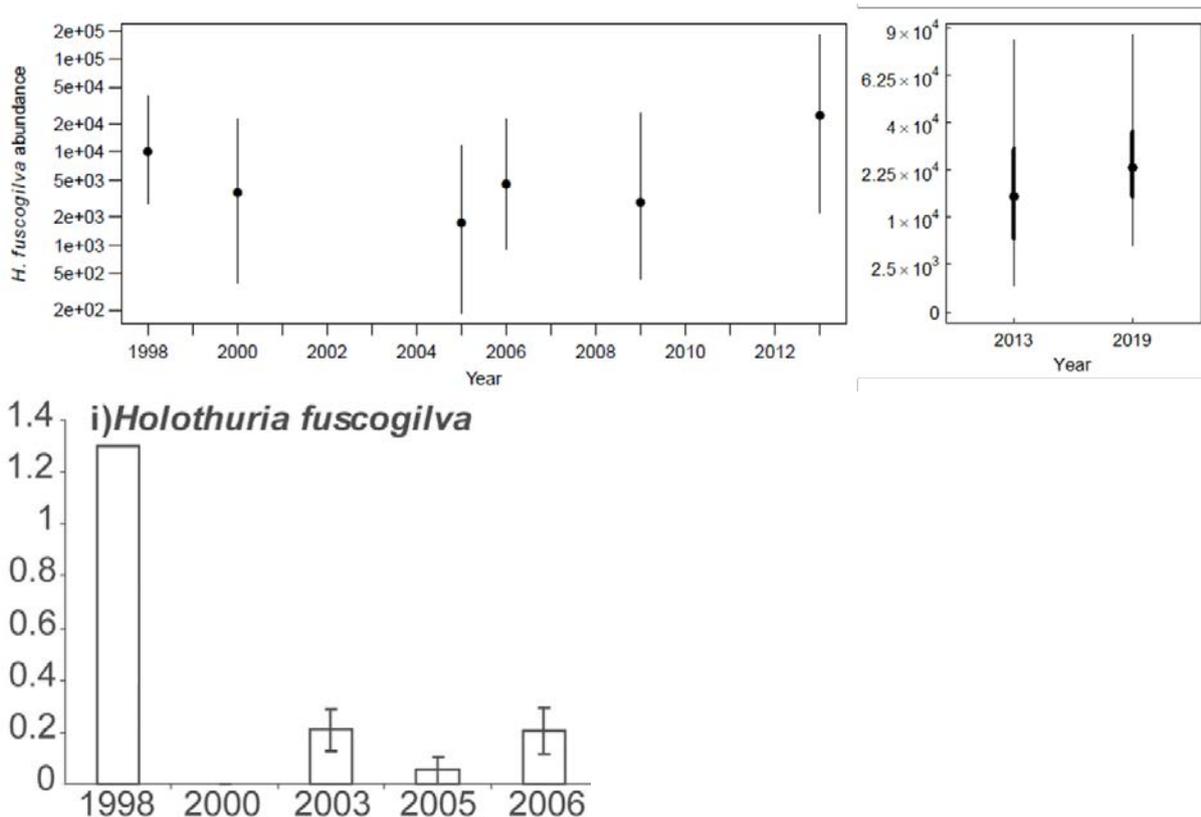
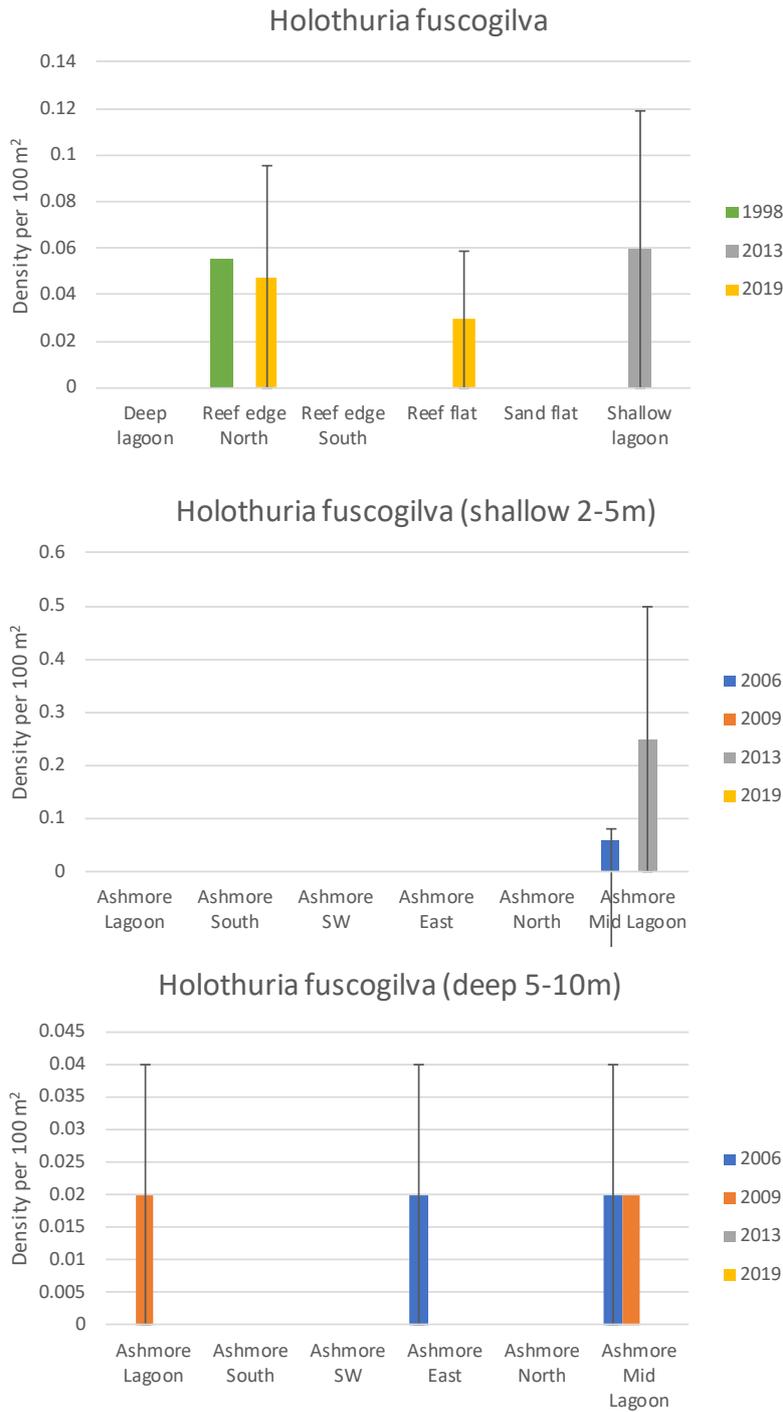


Figure 101. Long term estimates of abundance of *Holothuria fuscogilva*. Upper left panel is from Hosack and Lawrence (2013), upper right panel is from Hosack *et al.* in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both panels. Values are medians and longest error bars are 95% confidence limits. Lower panel figure is from Ceccarelli *et al.* (2011a) with average or mean ( $\pm 1$  s.e.) densities for the whole of Ashmore Reef expressed per hectare. See these cited studies for methods.



**Figure 102. Comparisons of historical and recent surveys of *Holothuria fuscogilva* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.**

***Holothuria coluber* (snake sea cucumber) historical trends in density**

The 2006 survey (Ceccarelli *et al.* 2007) was the only survey to record significant densities of *Holothuria coluber* (Figure 103, Figure 104), recording about 180 individuals in total. We only counted 9 in 2019 and they were not recorded in 2013. *Holothuria coluber* was not recorded by Russell and Vail (1988) or Skewes *et al.* (1999a).

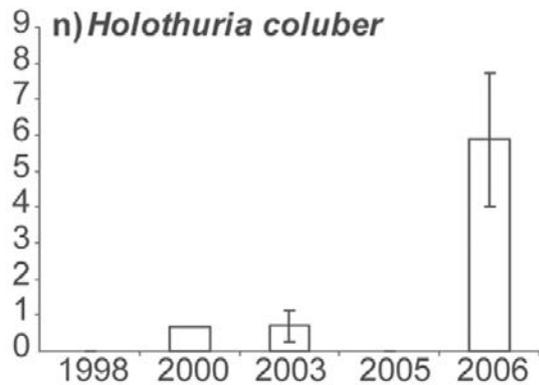


Figure 103. Long term estimates of abundance of *Holothuria coluber* at Ashmore Reef. Figure is from Ceccarelli *et al.* (2011a) with average or mean ( $\pm 1$  s.e.) densities for the whole of Ashmore Reef expressed per hectare. See cited study for methods.

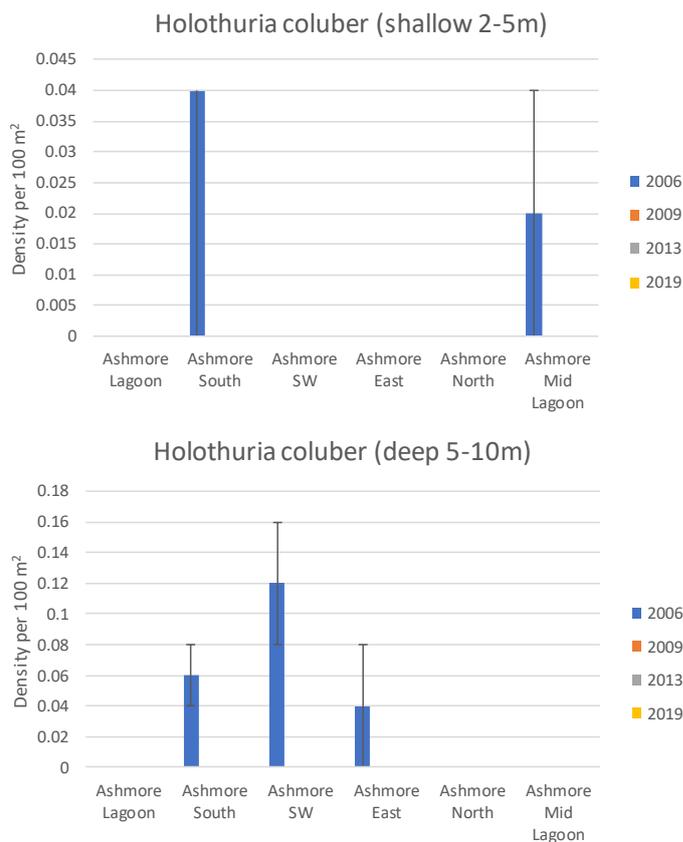


Figure 104. Comparisons of historical and recent surveys of *Holothuria coluber* at Ashmore Reef. Abundances are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.

### ***Holothuria edulis* (unsavoury sea cucumber) historical trends in density**

Long term analyses by Hosack and Lawrence (2013a) and by Ceccarelli *et al.* (2011a) suggest *Holothuria edulis* has declined in abundance between 1998 and 2009 and recovered in 2013 (Figure 105) when densities were similar to those recorded in 1998 (Figure 106). Hosack *et al.* (chapter 12 this study) found abundance had decreased between 2013 and 2019, but not significantly. Seven individuals were recorded in 2013 and six in 2019. However, some abundances (>300 at one site, Figure 105) recorded by Russell and Vail (1988) were higher than any in subsequent surveys. *Holothuria edulis*, like *H. atra*, is a species that can reproduce asexually (Uthicke 1997) so it should be more robust to overfishing than species dependent solely on sexual reproduction. About 24% of *H. edulis* undergo fission annually on the Great Barrier Reef (Uthicke 1997).

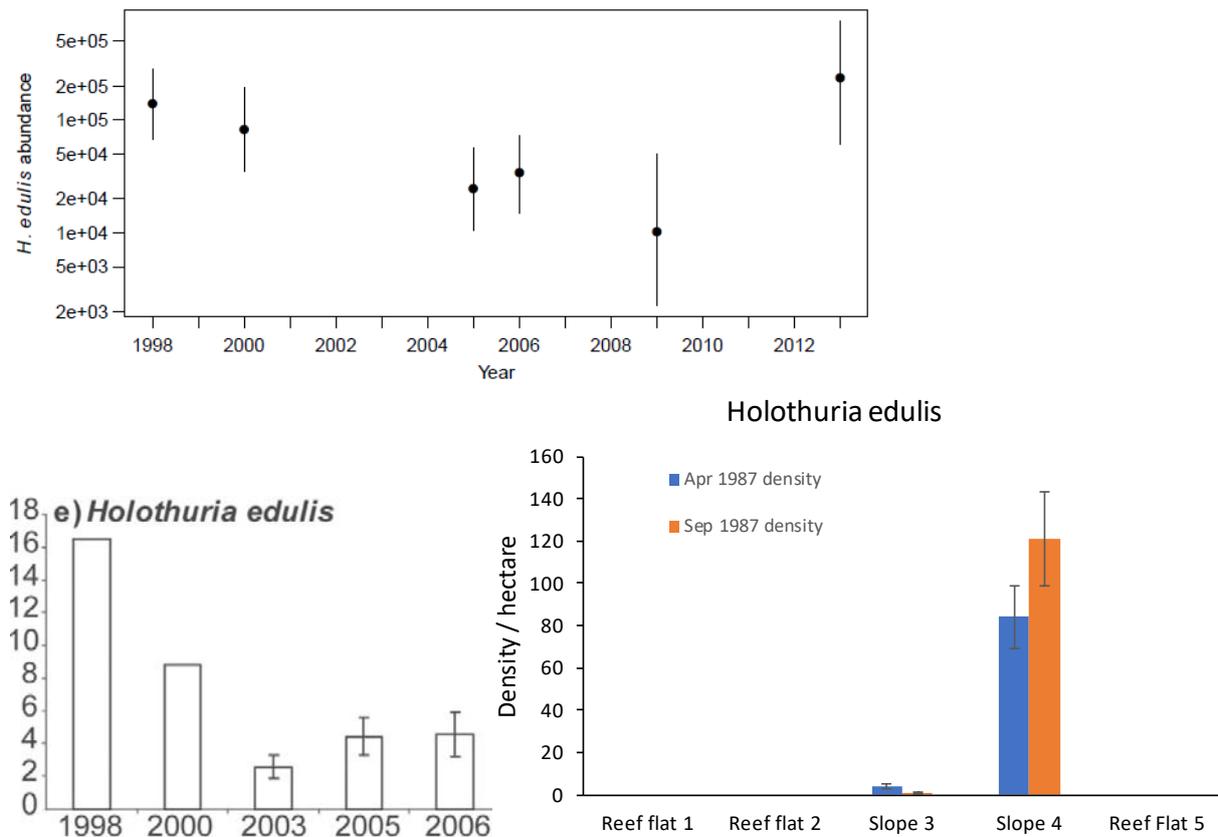


Figure 105. Long term estimates of abundance of *Holothuria edulis* at Ashmore Reef. Upper panel is from Hosack and Lawrence (2013a). These represent the total estimated population on Ashmore Reef. Values are medians and longest error bars are 95% confidence limits. Lower left panel is from Ceccarelli *et al.* (2011a) with average or mean ( $\pm 1$  s.e.) densities for the whole of Ashmore Reef expressed per hectare. Lower right panel are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.

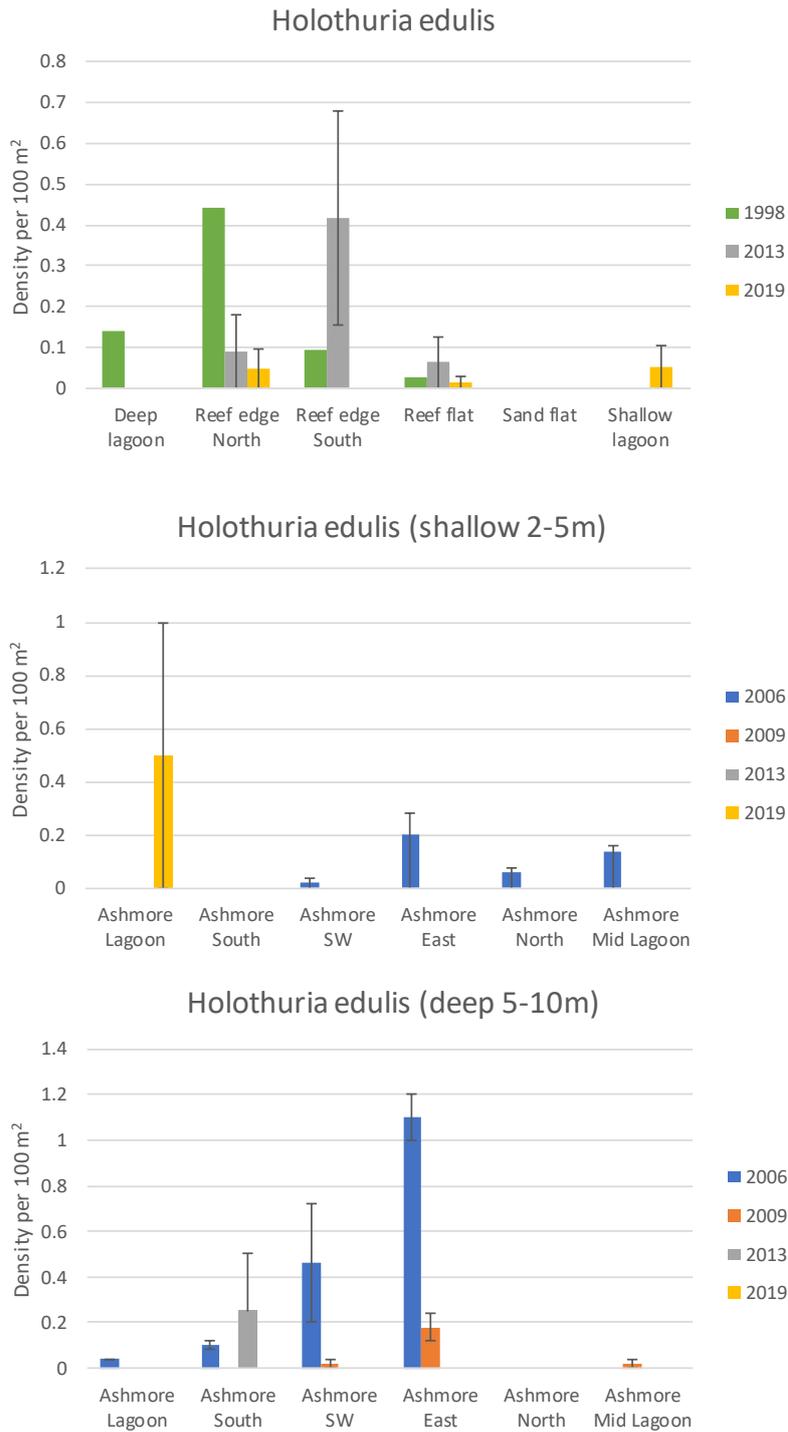


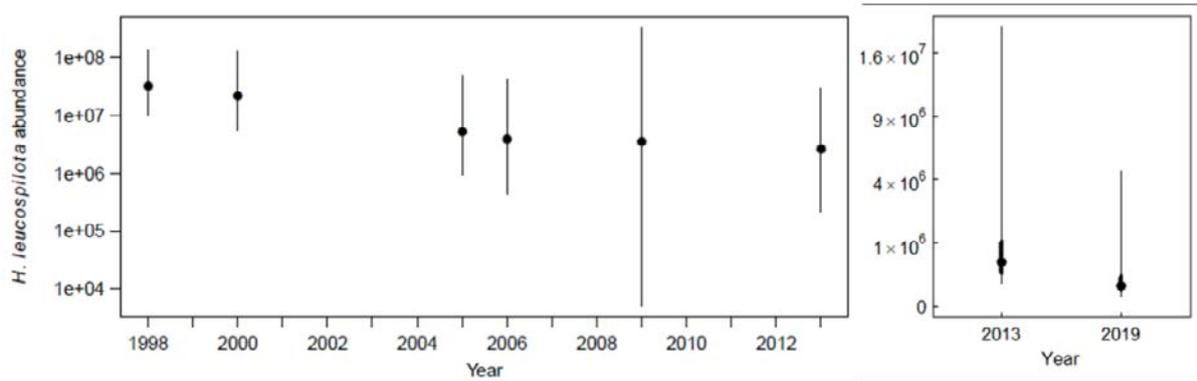
Figure 106. Comparisons of historical and recent surveys of *Holothuria edulis* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.

### ***Holothuria leucospilota* (stained sea cucumber) historical trends in density**

*Holothuria leucospilota* is harvested in some countries (Drumm & Loneragan 2005; Loneragan *et al.* 2005; Kinch *et al.* 2008), although there is no information of it being harvested at Ashmore Reef. This species is difficult to handle as it readily ejects sticky cuverian tubules when disturbed (Drumm & Loneragan 2005) which is likely to have discouraged its collection as it would result in contamination of a mixed catch bag of holothurians.

*Holothuria leucospilota* can be superabundant on reef flats and this was recorded at Ashmore Reef in 2003 (Rees *et al.* 2003) and in 2005 and 2006 (Ceccarelli *et al.* 2007) with densities up to 224,000/ha (Table 26). The large aggregations of *H. leucospilota* recorded in 2005 and 2006 near East Island were not present in 2019. The location of the 2003 aggregations was not reported by Rees *et al.* (2003). *Holothuria leucospilota* was not abundant at any of the five sites Russell and Vail (1988) surveyed quantitatively, but they did find very high densities of 1 – 2 *H. leucospilota*/m<sup>2</sup> in a spot dive (their site 6), west of West Island on the reef flat about half way between the island and the edge of the reef.

Historical analysis of abundance by Hosack and Lawrence (2013a) suggested a decline in the median abundance estimate, although the uncertainty associated with these estimates was high (Figure 107). Ceccarelli *et al.* (2011a) recorded a reduction between 1998 and 2006. However, the quantitative study sites in 2005 and 2006 did not include the aggregations encountered elsewhere on the reef in those years (Ceccarelli *et al.* 2007). The comparisons of the (Skewes *et al.* 1999a) data against 2013 and 2019 also suggest a large decline in abundance of *H. leucospilota* (Figure 108).



Holothuria leucospilota

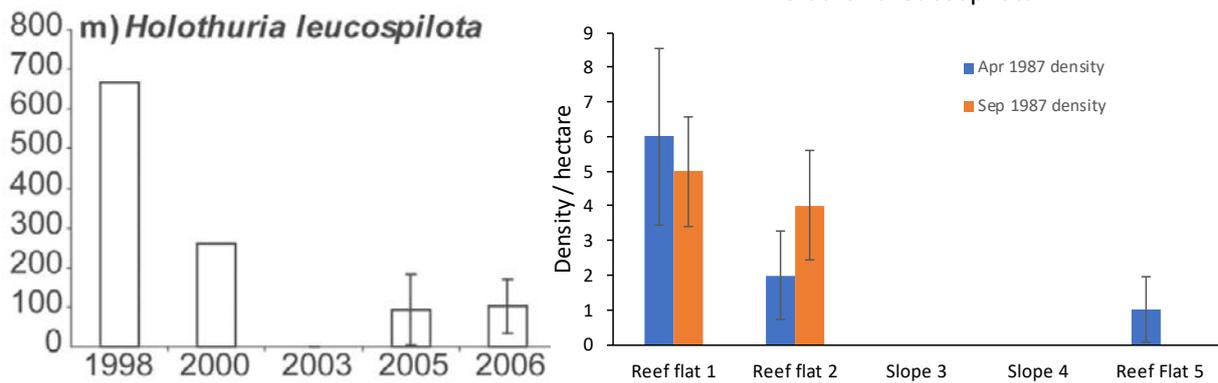


Figure 107. Long term estimates of abundance of *Holothuria leucospilota* at Ashmore Reef. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both panels. Values are medians and longest error bars are 95% confidence limits. Lower left panel is from Ceccarelli et al. (2011a) with average or mean ( $\pm 1$  s.e.) densities for the whole of Ashmore Reef expressed per hectare. Note that these data exclude the high-density aggregation sites reported by Rees et al. (2003) and Ceccarelli et al. (2007). Lower right panel are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.

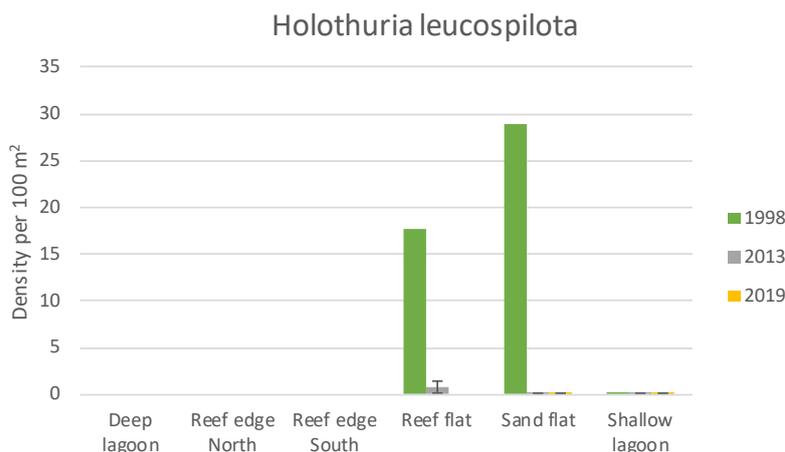


Figure 108. Comparisons of historical and recent surveys of *Holothuria leucospilota* at Ashmore Reef. Abundance is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Error bars are  $\pm 1$  s.e.

### *Holothuria fuscopunctata* (black-banded sea cucumber) historical trends in density

The long-term analysis by Hosack and Lawrence (2013a) suggests that abundance of *Holothuria fuscopunctata* (black-banded sea cucumber) has been low and relatively stable (Figure 109). Hosack et al. (chapter 12 this study) found abundance had decreased between 2013 and 2019, but not significantly. It was only recorded once in 2013 and twice in 2019, once in the deep lagoon and one in the shallower eastern lagoon. The highest densities recorded for this species were in 2003 (Rees *et al.* 2003) Figure 109) and were based on finding about 15 individuals in deep-water surveys. Thus these estimates cannot be compared easily to the majority of other surveys, although the 2006 surveys made by Ceccarelli *et al.* (2007) in the deep lagoon recorded similar densities (Figure 110). Russell and Vail (1988) recorded *H. fuscopunctata* as a very minor component of the catch by Indonesian fishers at Ashmore Reef and found it on their deep lagoon slope site 4 (Figure 88). It is possible the species has never been abundant and has had some refuge from fishing in the deep lagoon waters of Ashmore Reef. These deep lagoon waters should be resurveyed to assess the current status.

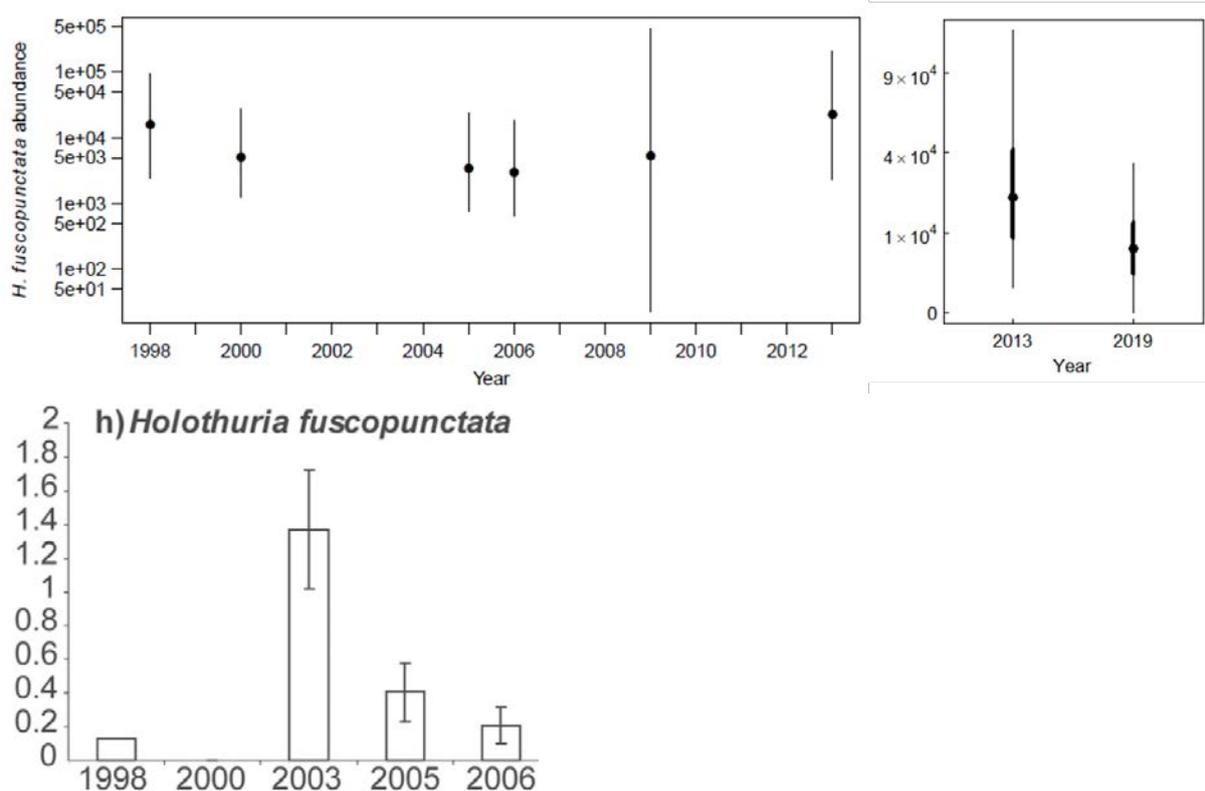
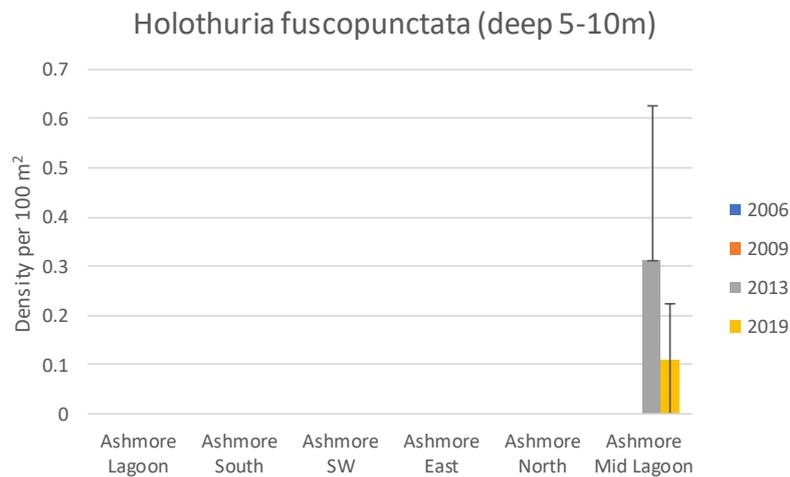
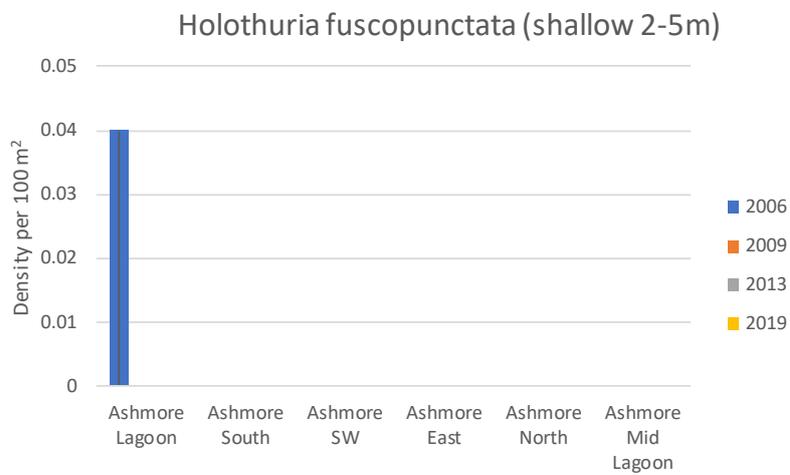
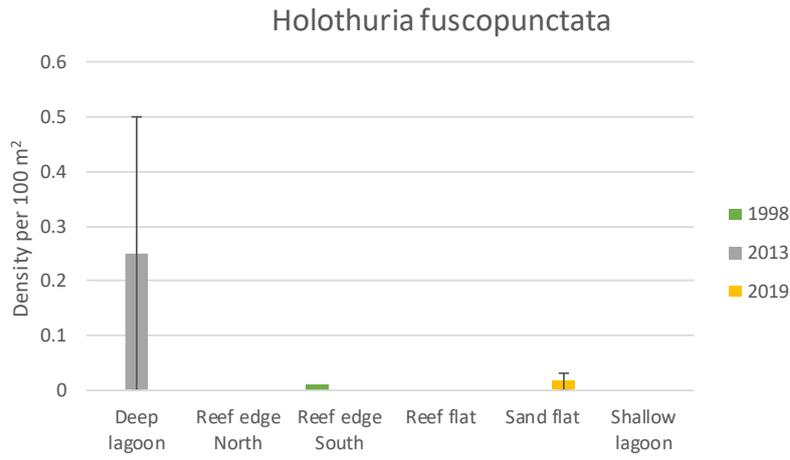


Figure 109. Long term estimates of abundance of *Holothuria fuscopunctata* at Ashmore Reef. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both panels. Values are medians and longest error bars are 95% confidence limits. Lower panel is from Ceccarelli et al. (2011a) with average or mean ( $\pm 1$  s.e.) densities for the whole of Ashmore Reef expressed per hectare.



**Figure 110. Comparisons of historical and recent surveys of *Holothuria fuscopunctata* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.**

***Holothuria scabra* (sandfish) and *Holothuria lessoni* (goldern sandish or lesson’s sea cucumber) historical trends in density**

*Holothuria lessoni* (referred to as *H. aculeata* and *H. timana* in some previous studies) has not been observed at Ashmore Reef since 1987, when Russell and Vail (1988) found 110 *H. lessoni* among the catches of holothurians on five Indonesian perahus. They recorded very few individuals in their own surveys (Figure 111). The Indonesian fishers interviewed by Russell and Vail (1988) stated that *H. lessoni* was the most valuable of all holothurian species they harvested from Ashmore Reef.

*Holothuria scabra* (sanddfish) was recorded by the Soviet survey of Ashmore Reef in 1978 (Marsh *et al.* 1993) but has not been observed since. *Holothuria scabra* is perhaps one of the most heavily exploited species of holothurian throughout the Indo-Pacific (Kinch *et al.* 2008; Friedman *et al.* 2011). Both *Holothuria scabra* and *H. lessoni* were regarded as being locally extinct by 1998 (Skewes *et al.* 1999a; Smith *et al.* 2001) although there is no evidence *H. scabra* was ever common at Ashmore Reef.

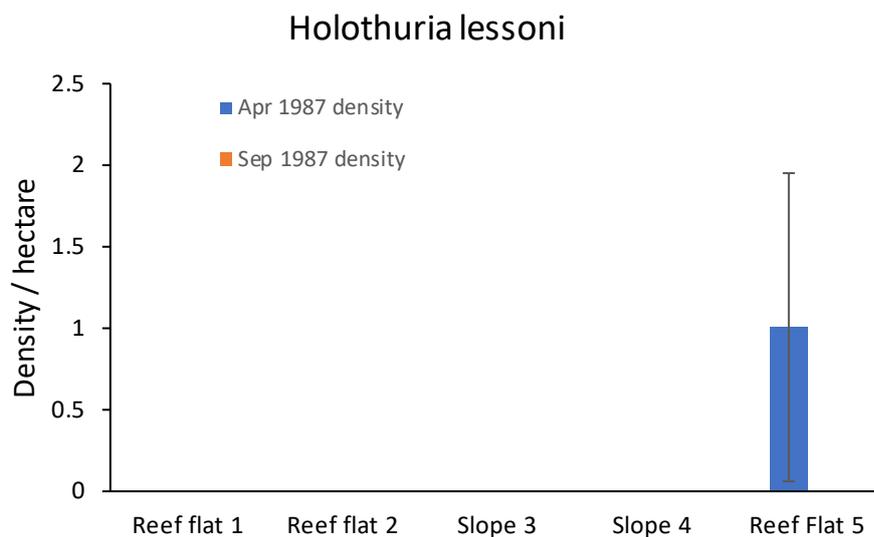


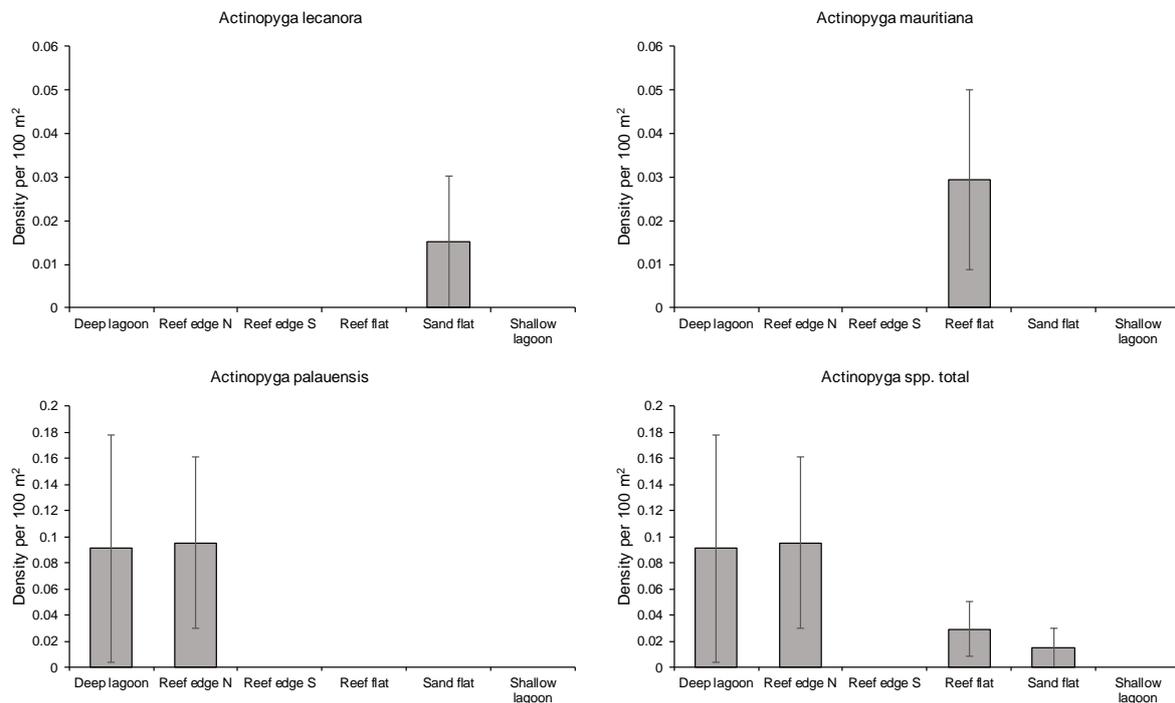
Figure 111. Mean density ( $\pm 1$  s.e.) of *Holothuria lessoni* at five sites surveyed at Ashmore Reef in 1987 by Russel and Vail (1988) See cited study for methods. Values for September 1987 are zeros.



Figure 112. Indonesian fisherman processing a large catch *Holothuria lessoni* on one of the islands at Ashmore Reef in 1986/87. Plate 12 reproduced from Russell and Vail (1988). Photograph: Lyle Vail.

### ***Actinopyga* spp. density and distribution**

We recorded four species of *Actinopyga* in June 2019 including *Actinopyga palauensis* (Palauan sea cucumber) for the first time (although it is possible that this is the species referred to as *Actinopyga obesa obesa* (plump sea cucumber) by Russell and Vail 1988). The density of each species is shown in Figure 113 and the distribution of the observations is given in Figure 92. *Actinopyga miliaris* (military sea cucumber) is not shown in Figure 113, as only one incidental record of this species was made on the northern reef edge at site 358 (Figure 92). *Actinopyga* spp. were uncommon, with densities  $<0.1/100\text{ m}^2$  in all habitats (Figure 113).



**Figure 113. Mean density of holothurians (*Actinopyga* spp.) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.**

**Actinopyga spp. historical trends in density**

Analyses of the abundance of *Actinopyga* spp. across all surveys since 1987 (Figure 114, Figure 115) show little differences and low abundances except that Hosack et al. (chapter 12 this study) found that abundance had increased significantly between the two most recent surveys in 2013 and 2019. Marsh *et al.* (1993) only recorded them at five (of 20) sites in 1987 and only found one each of both *Actinopyga mauritiana* (Mauritian sea cucumber) and the hedgehog sea cucumber *A. echinites* (Berry 1993). *Actinopyga* spp. were among the high value species targeted before catches expanded to include lower value species in the mid-1980s (Russell & Vail 1988). At the time of their 1986 and 1987 surveys, *Actinopyga* were still being taken in large numbers (Figure 116) making up about 45% of the catch of species other than *Holothuria atra* (290 *Actinopyga* of 653 non *H. atra* holothurians were counted on five perahus) (Russell and Vail 1988). The current abundance of *Actinopyga* spp. at Ashmore Reef suggest the genus has not recovered from these high levels of fishing.

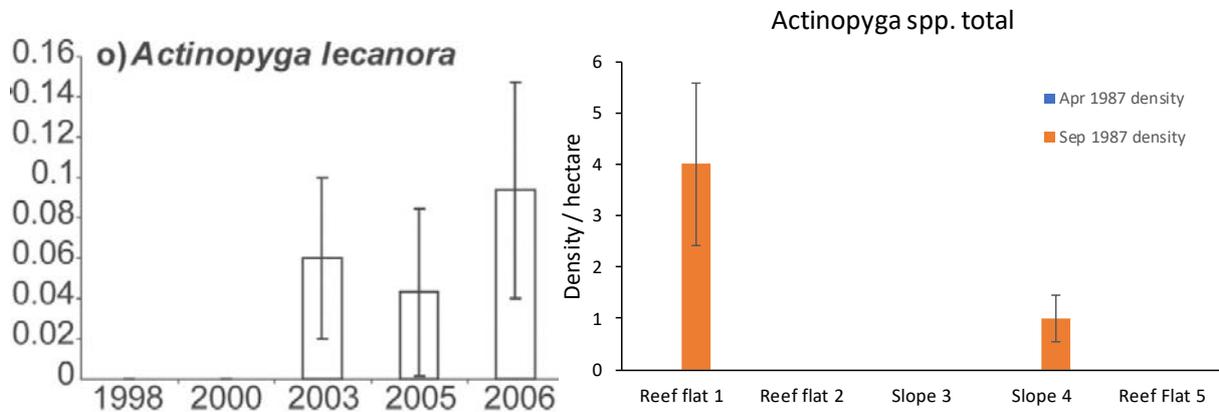
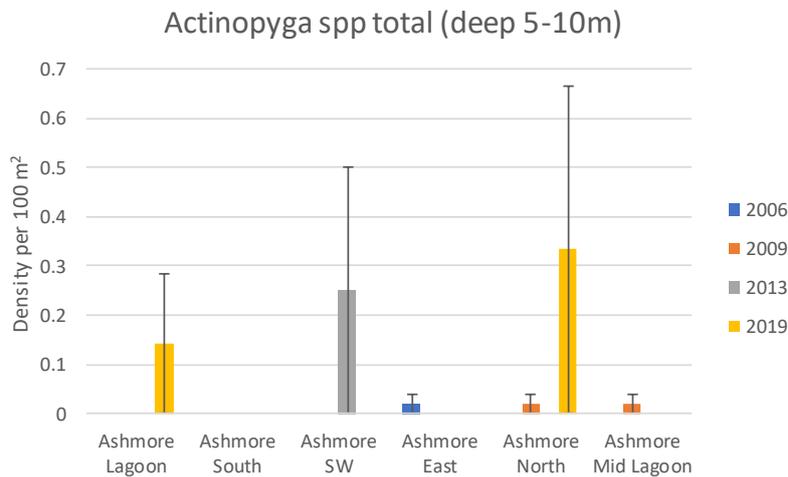
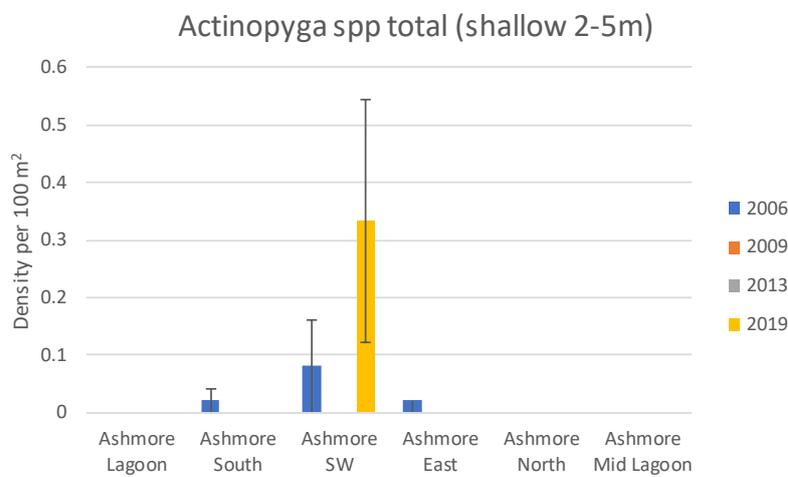
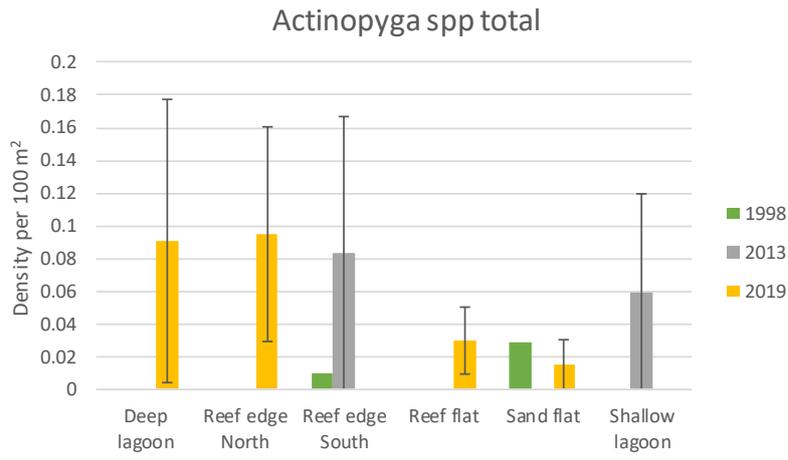


Figure 114. Long term estimates of abundance of *Actinopyga* spp at Ashmore Reef. Left figure is from Ceccarelli et al. (2011a) with mean densities for the whole of Ashmore Reef expressed per hectare ( $\pm 1$  s.e.). Right panel are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods. Values for April 1987 are zeros.



**Figure 115. Comparisons of historical and recent surveys of *Actinopyga* spp. at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e. Apparent changes in depth distribution at the same location from one survey to the next are likely to reflect a mismatch in how data was aggregated into depth categories between different surveys.**

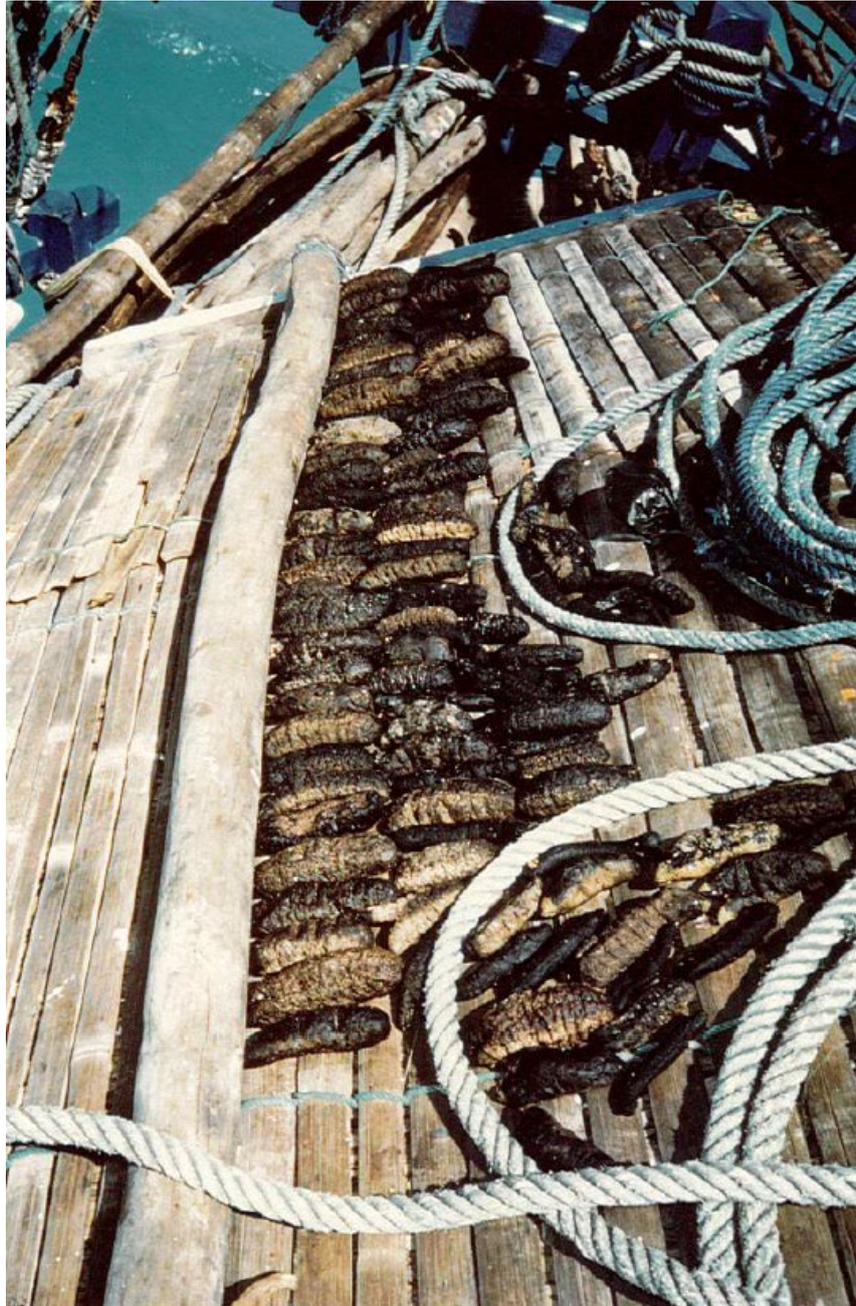


Figure 116. Plate 11 reproduced from Russell and Vail (1988) showing holothurians (mostly *Actinopyga* spp.) being dried on board an Indonesian perahu at Ashmore Reef in 1986/87. Photograph: Lyle Vail.

### ***Stichopus* spp. density and distribution**

The highest density of *Stichopus herrmanni* (reported in some previous surveys as *S. variegata* or *Holothuria variegata*) was recorded in the deep lagoon habitat (0.18/100 m<sup>2</sup>, Figure 117). It was not recorded on the reef edge. Five individuals were observed in manta tow surveys.

*Stichopus chloronotus* is one of the more common holothurians on Indo-Pacific coral reefs and we recorded it on transects in all habitats except the deep lagoon, although it was recorded on some bommies in the deep lagoon in incidental observations. Density of *S. chloronotus* was highest along the northern reef edge (0.28/100 m<sup>2</sup>, Figure 117).

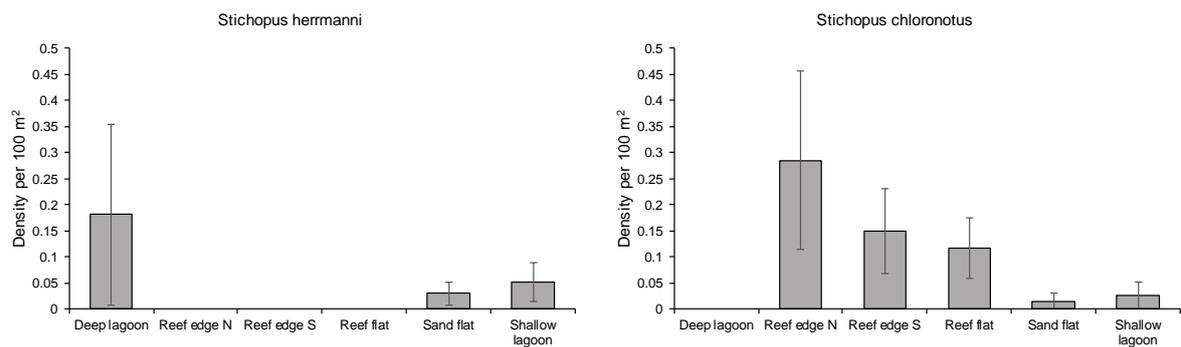


Figure 117. Mean density of holothurians (*Stichopus spp.*) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

### ***Stichopus herrmanni* (Herrmann’s seas cucumber) historical trends in density**

Most surveys between 1998 and 2019 found very few *S. herrmanni* individuals. Although mean densities of *Stichopus herrmanni* (Herrmann’s seas cucumber) in 2013 and 2019 were generally higher than in previous surveys (Figure 119) these were based on very few observed individuals, and variances were very high, precluding any firm conclusions about trends over time. Hosack et al. (chapter 12 this study) found abundance had decreased between 2013 and 2019, but not significantly. Abundances measured in 1987 by Russell Russell and Vail (1988) were considerably higher than in subsequent surveys but they surveyed very few sites. They recorded their highest abundance was on the deep lagoon slope and this further underscores the need for surveys of the deep-water habitats at Ashmore Reef.

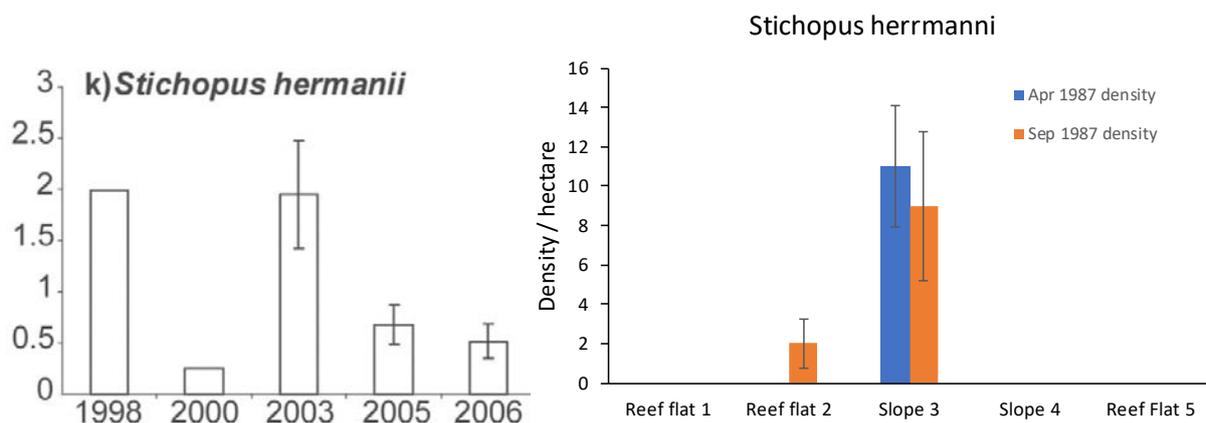


Figure 118. Long term estimates of abundance of *Stichopus herrmanni* at Ashmore Reef. Left figure is from Ceccarelli et al. (2011a) with mean densities for the whole of Ashmore Reef expressed per hectare ( $\pm 1$  s.e.). Right panel are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.

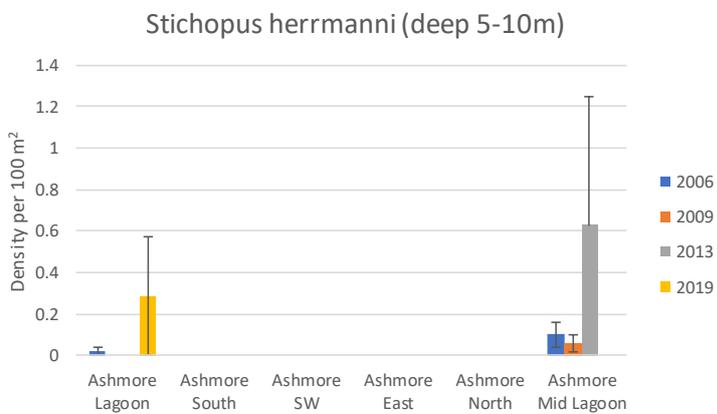
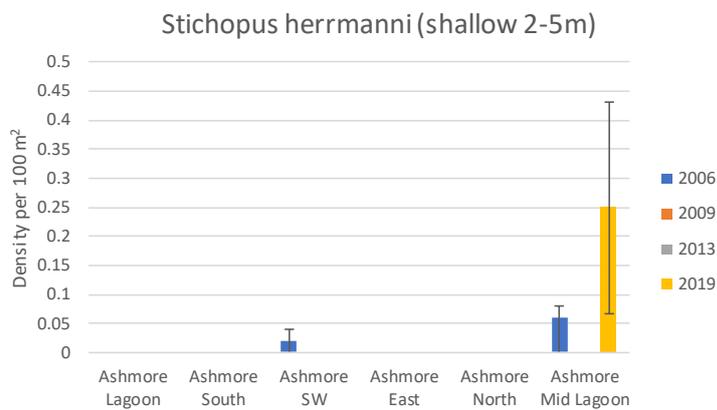
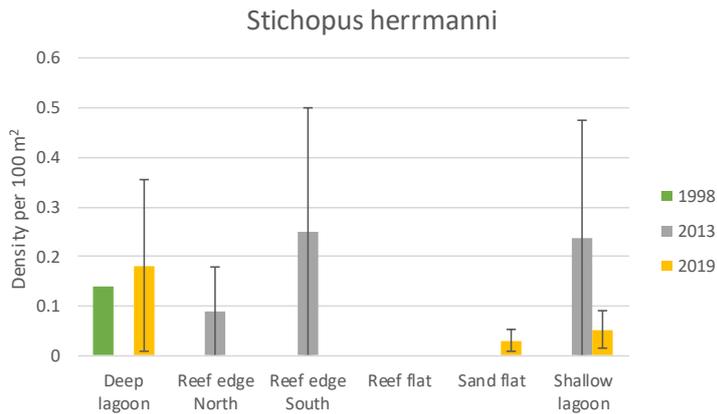


Figure 119. Comparisons of historical and recent surveys of *Stichopus herrmanni* (Herrmann’s seas cucumber) at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e. Apparent changes in depth distribution at the same location from one survey to the next are likely to reflect a mismatch in how data was aggregated into depth categories between different surveys.

### ***Stichopus chloronotus* (green sea cucumber) historical trends in density**

There is little evidence of any long-term change in the density of *Stichopus chloronotus* at Ashmore Reef. Ceccarelli *et al.* (2011a) found no change between 1998 and 2006 (Figure 120) and abundances measured by Russell and Vail at one site in 1987 were not dissimilar to subsequent surveys (Figure 120). Densities may have increased between 2006 and 2013/2019 but not between 1998 and 2013/2019 (Figure 121) suggesting site to site patchiness was responsible for these differences. Hosack *et al.* (chapter 12 this study) found abundance of *S. chloronotus* had decreased between 2013 and 2019, but not significantly.

*Stichopus chloronotus* is capable of asexual reproduction (Uthicke 1997) which may provide an enhanced capacity to recover from overfishing. The rate of asexual reproduction is highly variable by site. On the Great Barrier Reef the incidence of asexual fission on reefs near the mainland was 43% per annum and 19% on mid-shelf reefs (Uthicke 1997). Even at the lower end of this range, this would have a stabilising influence and lead to population growth of this species at Ashmore Reef.

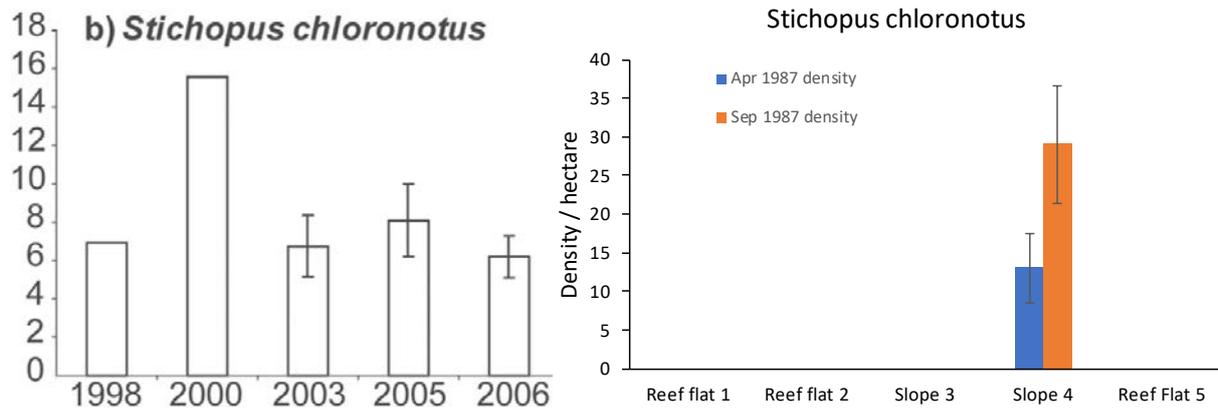
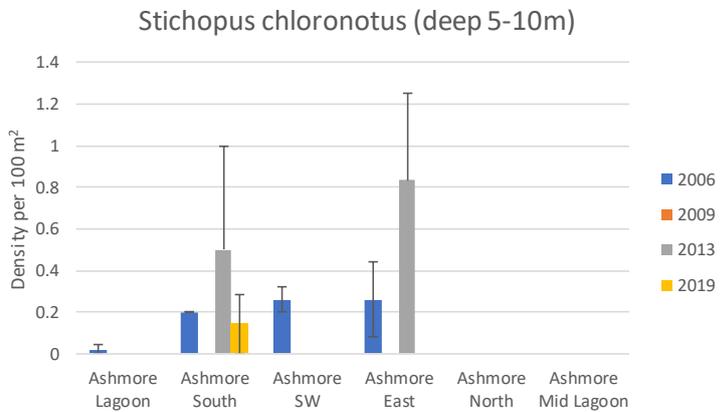
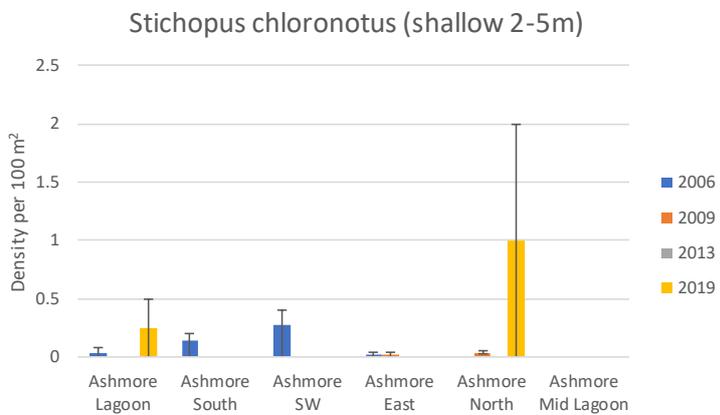
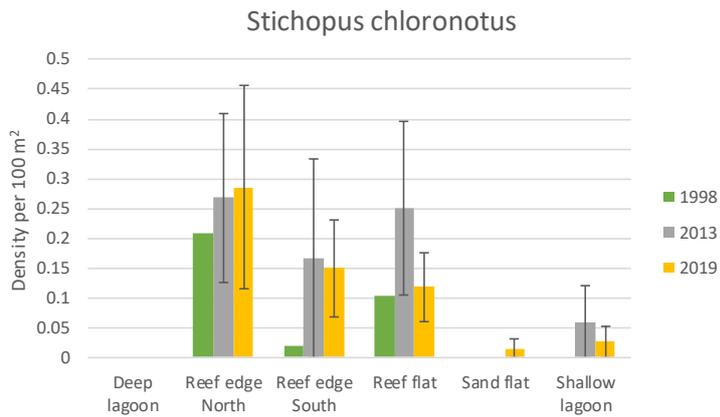


Figure 120. Long term estimates of abundance of *Stichopus chloronotus* at Ashmore Reef Left figure is from Ceccarelli *et al.* (2011a) with mean densities for the whole of Ashmore Reef expressed per hectare ( $\pm 1$  s.e.). Right panel are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.



**Figure 121. Comparisons of historical and recent surveys of *Stichopus chloronotus* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.**

### *Pearsonothuria graeffei* (Graeffe's sea cucumber) density and distribution

*Pearsonothuria graeffei* occurred on the reef edge transects and in the deep lagoon, not on the reef flat or sand flat. Its density was highest on the northern (back) reef slope (0.48/100 m<sup>2</sup>, Figure 122).

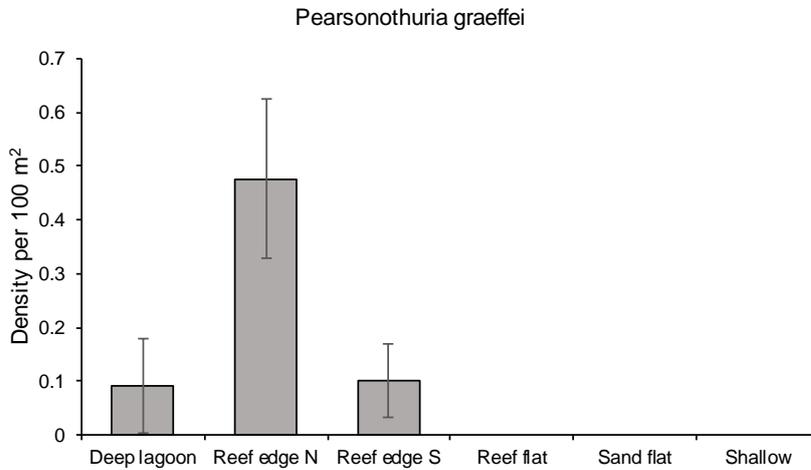


Figure 122. Mean density of the holothurian *Pearsonothuria graeffei* in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

### *Pearsonothuria graeffei* historical trends in density

There is little evidence of any long term change in density of *Pearsonothuria graeffei* between the earliest surveys in 1987 and 1998, and 2006 (Ceccarelli et al. 2011a, Figure 123). However, in our 2019 survey, densities on the northern reef edge slope were much higher (0.47/100 m<sup>2</sup> = 47/ha) than the 1998 survey which recorded ca. 5/ha (Figure 124). This was not the case for all sites when densities between 2006 and 2019 are compared (Figure 124), however the data suggests an increase in abundance of this species and Hosack et al. (chapter 12 this study) found the abundance had increased significantly between 2013 and 2019.

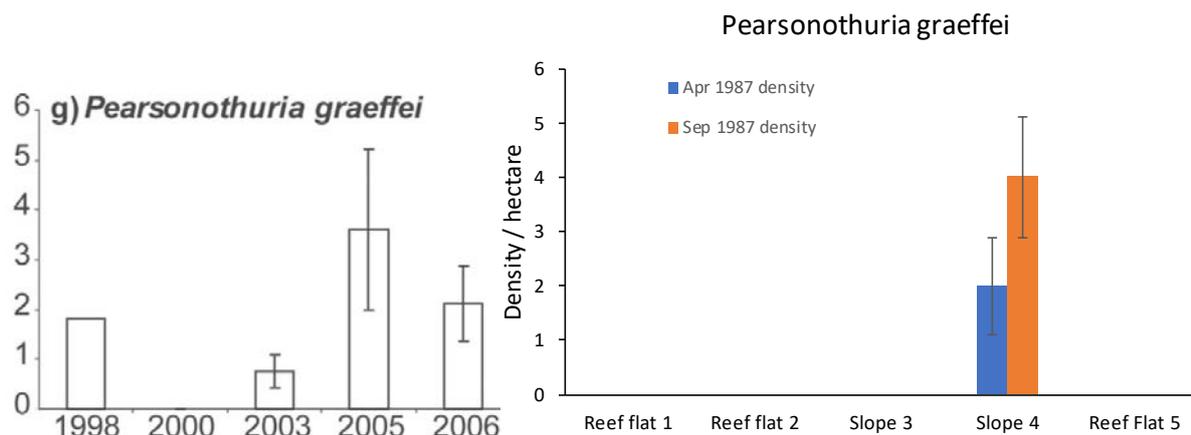


Figure 123. Long term estimates of abundance of *Pearsonothuria graeffei* at Ashmore Reef. Left figure is from Ceccarelli et al. (2011a) with mean densities for the whole of Ashmore Reef expressed per hectare ( $\pm 1$  s.e.). Right panel are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.

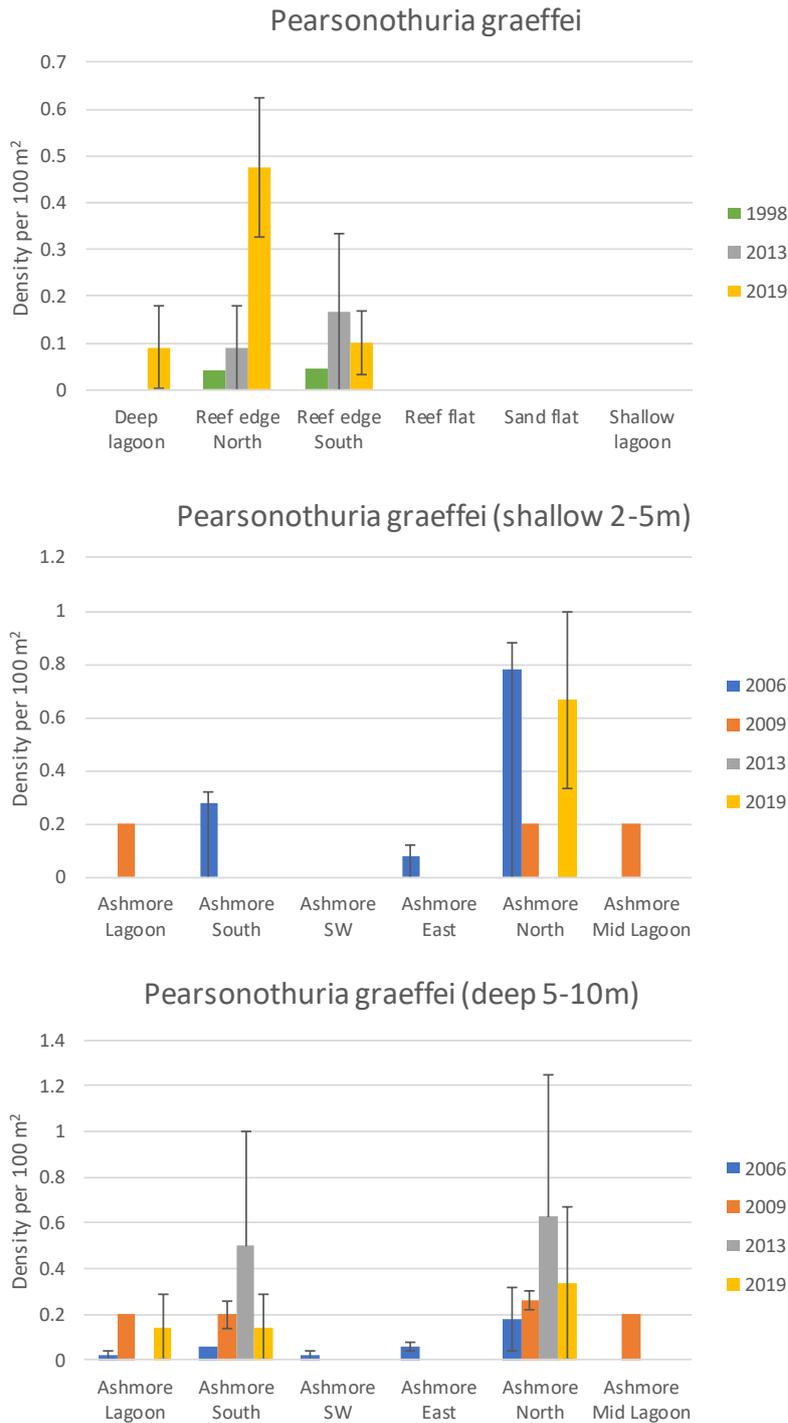


Figure 124. Comparisons of historical and recent surveys of *Pearsonothuria graeffei* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e. Apparent changes in depth distribution at the same location from one survey to the next are likely to reflect a mismatch in how data was aggregated into depth categories between different surveys.

***Bohadschia argus* (eyed sea cucumber or leopard fish) and *Bohaschia marmorata* (marbled sea cucumber) density and distribution**

*Bohadschia argus* was only recorded incidentally in the deep lagoon at the base of bommies. None were observed at any of our 224 sites. We did not observe any *B. marmorata* in 2019.

**Historical trends in density**

*Bohadschia argus* was not observed at Ashmore Reef in 2013 (Ceccarelli *et al.* 2013) and was last counted in quantitative surveys in 2009 (Richards *et al.* 2009). Ceccarelli *et al.* (2011a) found a decline in its density after 1998 (Figure 125) and our analysis (Figure 126) is in agreement. *Bohadschia marmorata* was last observed at Ashmore Reef in 2006 (Ceccarelli *et al.* 2007, Figure 125). It seems clear that both species were once more common at Ashmore Reef despite not (initially at least) being targeted by fishers. Russell and Vail (1988) recorded higher densities than other studies based on 38 *B. argus* and nine *B. marmorata* from five sites (Figure 85) and reported it as being collected by Indonesian fishers on speculation that there may be a market for it. Russell and Vail (1988) observed a large catch of 135 *B. marmorata* in one perahu that had also been taken to test the market. Marsh *et al.* (1993) found *B. argus* at five (of 20) sites and *B. marmorata* at four sites. Skewes *et al.* (1999a) recorded at least 14 individuals.

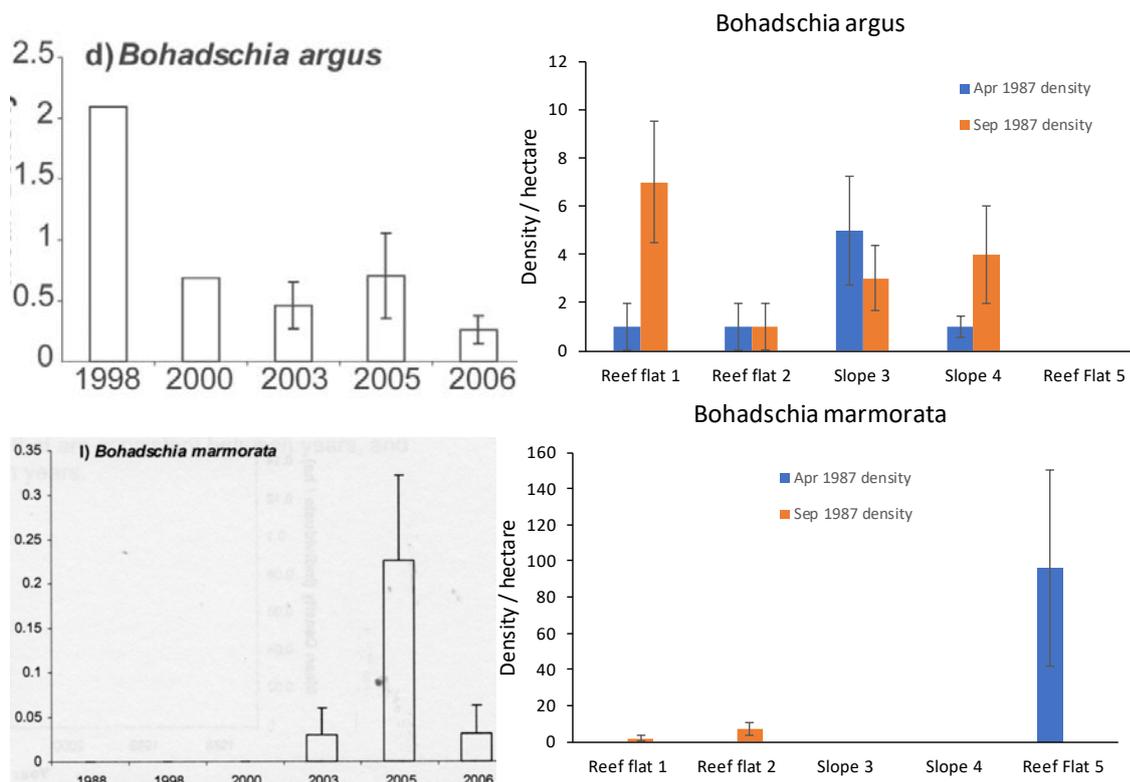


Figure 125. Long term estimates of abundance of *Bohadschia argus* and *Bohadschia marmorata* at Ashmore Reef. Upper left panel is from Ceccarelli *et al.* (2011a), lower left panel is from Ceccarelli *et al.* (2007) each with mean densities for the whole of Ashmore Reef expressed per hectare ( $\pm 1$  s.e.). Both right panels are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.

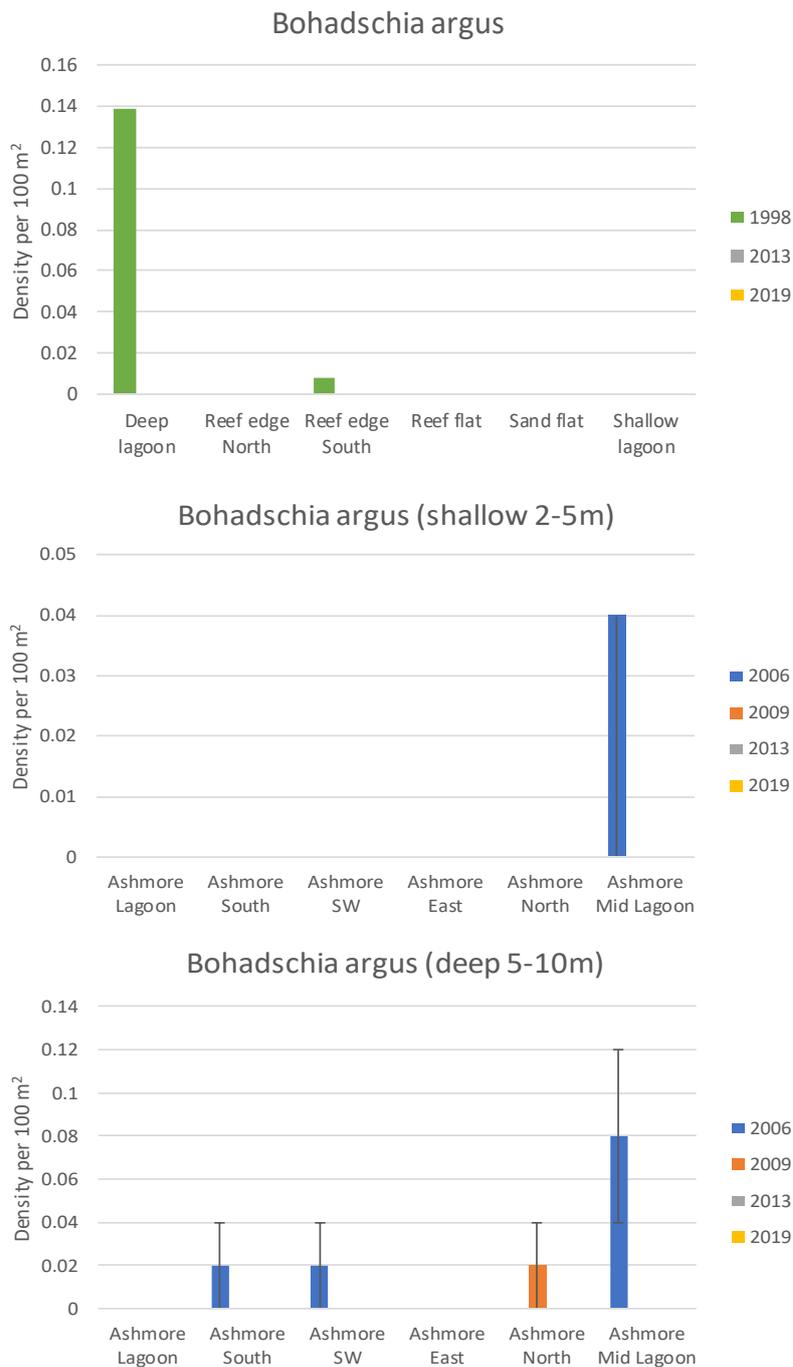


Figure 126. Comparisons of historical and recent surveys of *Bohadschia argus* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.

### ***Thelenota ananas* (prickly redfish) and *Thelenota anax* (royal sea cucumber) density and distribution**

The heavily fished commercial species *Thelenota ananas* and *Thelenota anax* were only recorded on the manta tow surveys (five *T. ananas*, two *T. anax*) and incidentally at our transect sites (one *T. ananas* on the northern reef edge, Figure 92). One each of both species were incidentally observed around the base of bommies in water > ca. 10 m in the deep lagoon at the western end of the island.

#### **Historical trends in density**

*Thelenota ananas* is one of the largest and most heavily exploited holothurians in the Indo-Pacific (Kinch *et al.* 2008) including at Ashmore Reef (Russell & Vail 1988; Marsh *et al.* 1993). *Thelenota anax* was less heavily exploited but this may be because it is less abundant and occurs in deeper water (Smith *et al.* 2001). Russell and Vail (1988) only recorded 32 *T. ananas* among 653 harvested holothurians (excluding *H. atra*) aboard 5 perahus in 1987. Their surveys at two sites with suitable habitat for *T. ananas* and *T. anax* had 15 and 17 individuals (2 – 3/ha, Figure 127) of these species respectively. Skewes *et al.* (1999a) recorded lower abundances and densities of both *Thelenota* species (Figure 128) than were recorded in 2000 and 2003 (Smith *et al.* 2001; Rees *et al.* 2003) (Figure 127). Ceccarelli *et al.* (2011a) found that by 2006 the densities of both species were at their lowest (Figure 127) and that declines from 2000/2003 may have been a result of a reduced level of surveillance of illegal fishing in 2006. However, the densities in 2005 and 2006 were similar to those in 1998 (Figure 127). The surveys in 2000 and 2003 (Smith *et al.* 2001; Rees *et al.* 2003) which recorded higher densities were the only surveys of deep water habitat where these species would be expected to be more abundant. The long-term analysis of abundance of *T. ananas* by Hosack and Lawrence (2013a) suggest abundances had recovered in 2013 and then declined again significantly in 2019 (Hosack *et al.* chapter 12 this study, Figure 127) but this needs to be regarded with caution as only 2 individuals were found in the 2013 survey, with much lower survey effort (Ceccarelli *et al.* 2013). It is important to expand the Hosack and Lawrence (2013a) design to include deep water habitats, as this is the preferred habitat for a number of important species. This will establish whether the deep-water habitat harbours significant numbers of these species. Smith *et al.* (2001) suggested the deep waters of the lagoon and back reef provided a refuge from fishing. but Indonesian fishers developed innovative methods for harvesting deep-water holothurians using a weighted spear, which may have made these refuges less effective (Russell and Vail 1988, see Figure 129).

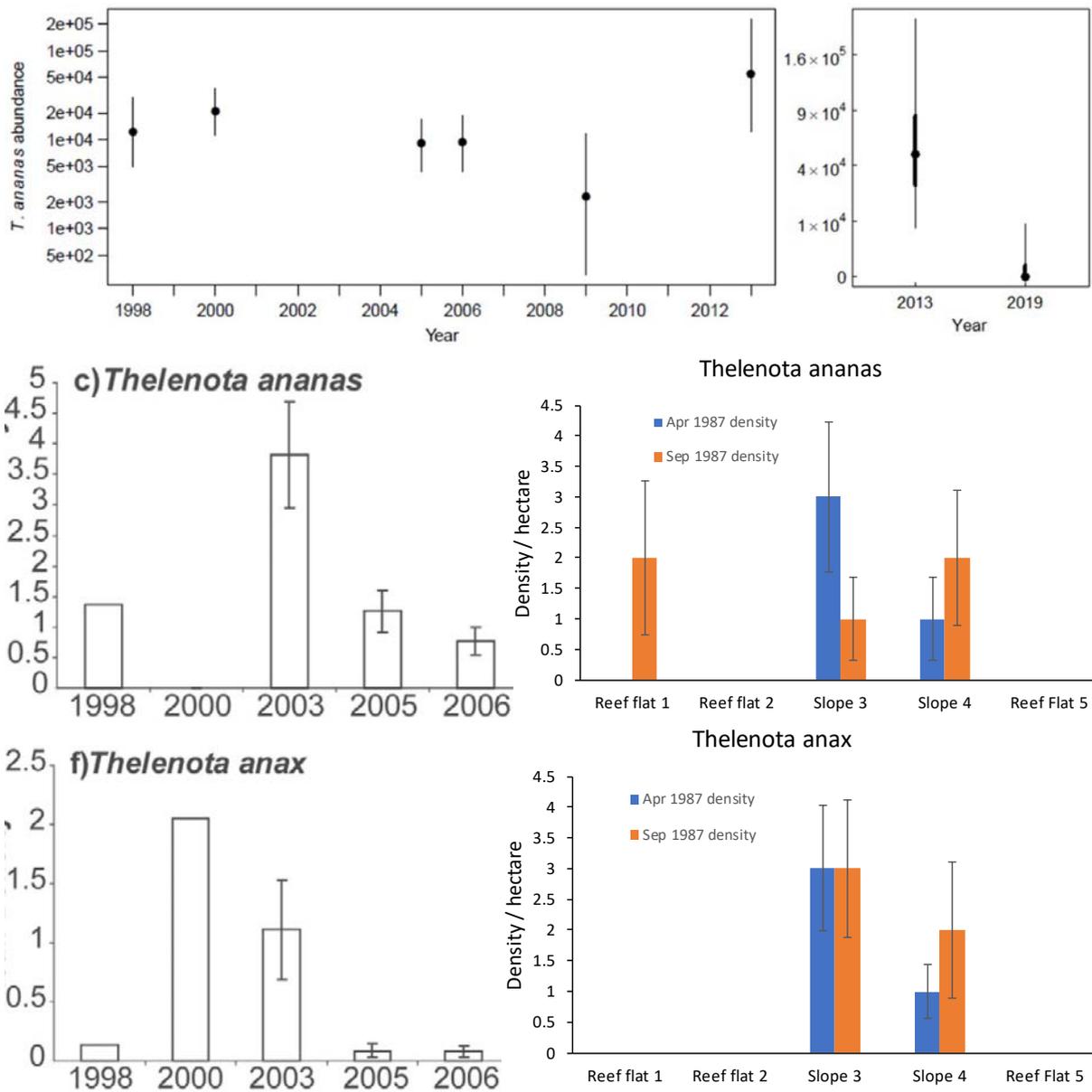
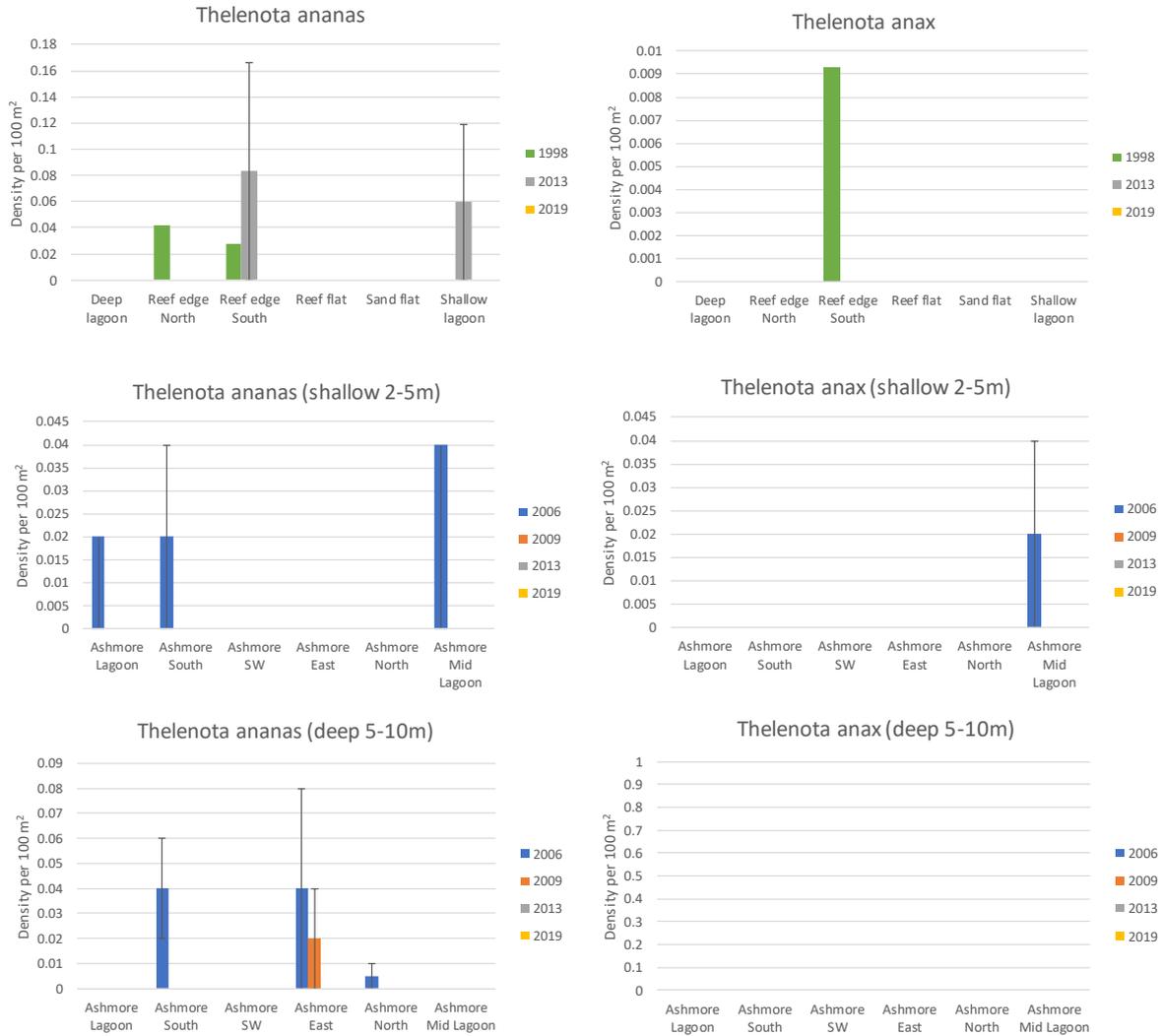


Figure 127. Long term estimates of abundance of *Thelenota ananas* and *Thelenota anax* at Ashmore Reef. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both panels. Values are medians and longest error bars are 95% confidence limits. Middle and lower left panel figures are from Ceccarelli et al. (2011a) with average or mean densities ( $\pm 1$  s.e.) expressed per hectare for the whole of Ashmore Reef. Middle and lower right panels are mean densities ( $\pm 1$  s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.



**Figure 128. Comparisons of historical and recent surveys of *Thelenota ananas* and *Thelenota anax* at Ashmore Reef. Top panels are stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are  $\pm 1$  s.e.**

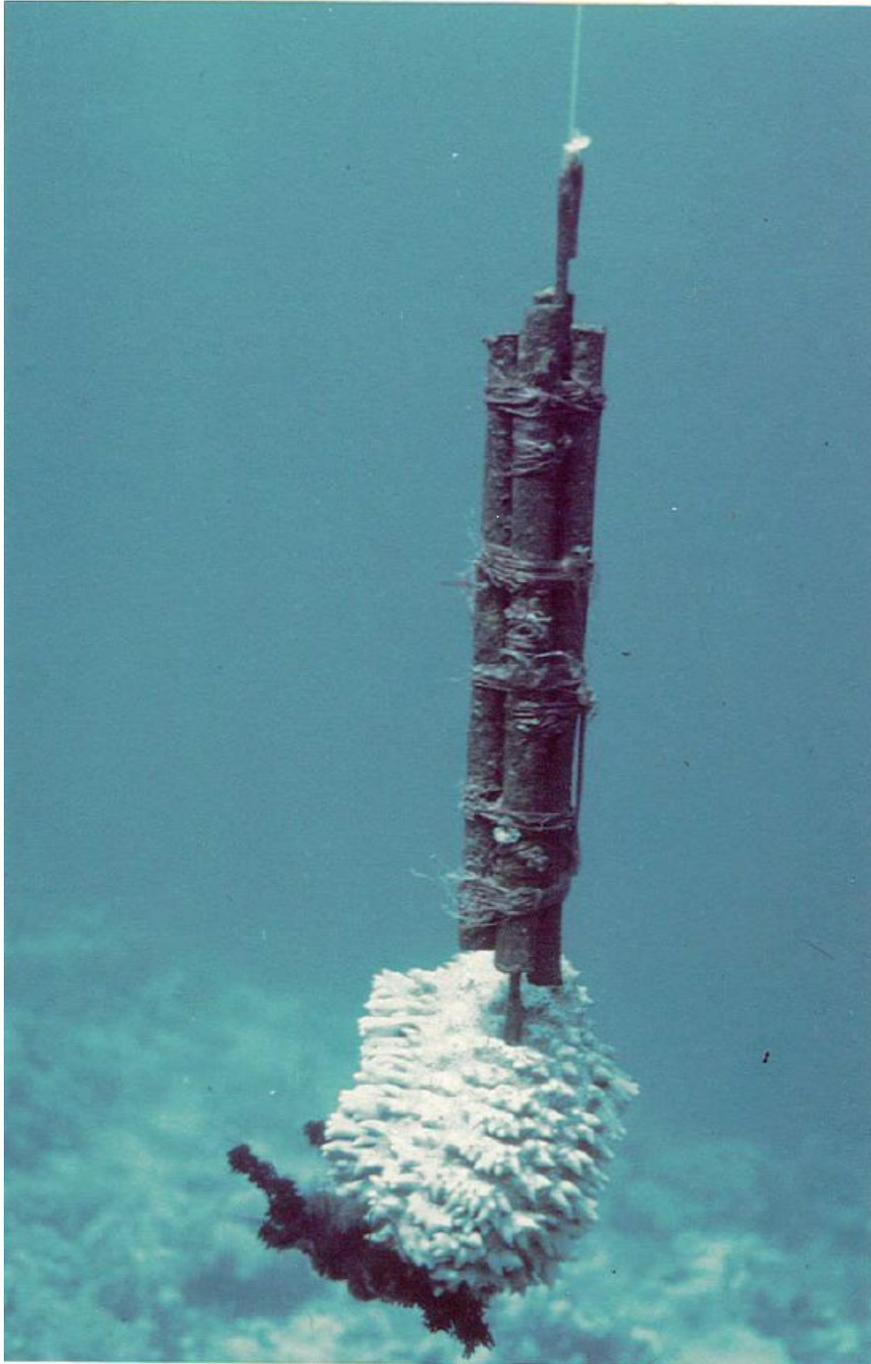


Figure 129. Weighted spear called a ladong used by Indonesian fishers at Ashmore Reef to take deep-water holothurians, in this case *Thelenota ananas*. Reproduced from plate 14 in Russell and Vail (1987). Photograph Lyle Vail.

## 11.4.2 Starfish

### Starfish diversity

A total of seven starfish species were observed at Ashmore Reef: *Acanthaster planci*, *Echinaster luzonicus*, *Fromia indica*, *Linckia guildingi*, *L. laevigata*, *L. multifora* and *Luidia maculata*.

### Crown-of-thorns starfish *Acanthaster planci*

#### *Acanthaster planci* density and distribution

A single crown of thorns starfish, 14 cm in diameter, was observed at Ashmore Reef at site 1367 on the northern edge of the reef (Figure 132). No evidence of coral damage from their feeding was detected at any other location either on the manta tows or the 100 m<sup>2</sup> invertebrate transects.

#### *Acanthaster planci* historical trends in density

Results from previous surveys suggest that crown of thorns starfish have always been rare at Ashmore Reef. Marsh *et al.* (1993) recorded it at just a single station and Richards *et al.* (2009) recorded two. It was not observed in any of the 1998, 2005, 2006 or 2013 surveys (Skewes *et al.* 1999a; Kospartov *et al.* 2006; Ceccarelli *et al.* 2007, 2013).

### Blue sea star *Linckia laevigata*

#### *Linckia laevigata* density and distribution

The blue starfish *Linckia laevigata* occurred in all habitats and was the most abundant of any invertebrate taxa we counted at Ashmore Reef. It is wide spread across the reef (Figure 132) and the highest density was on the reef flat (1.6/100 m<sup>2</sup>) (Figure 131).

#### *Linckia laevigata* historical trends in density

*Linckia laevigata* have been surveyed a number of times in concert with holothurian surveys, and the earliest data with comparable sampling structure and methods to ours were collected by Skewes *et al.* (1999a). The analysis indicates that densities may have increased between 1998 and 2013/2019 from about 1.1/100 m<sup>2</sup> to 1.6 – 2.1/100 m<sup>2</sup> in the habitats where *L. laevigata* is most common: on the reef flat and on the southern edge slope (Figure 130). We are not aware of this species ever being harvested at Ashmore Reef.

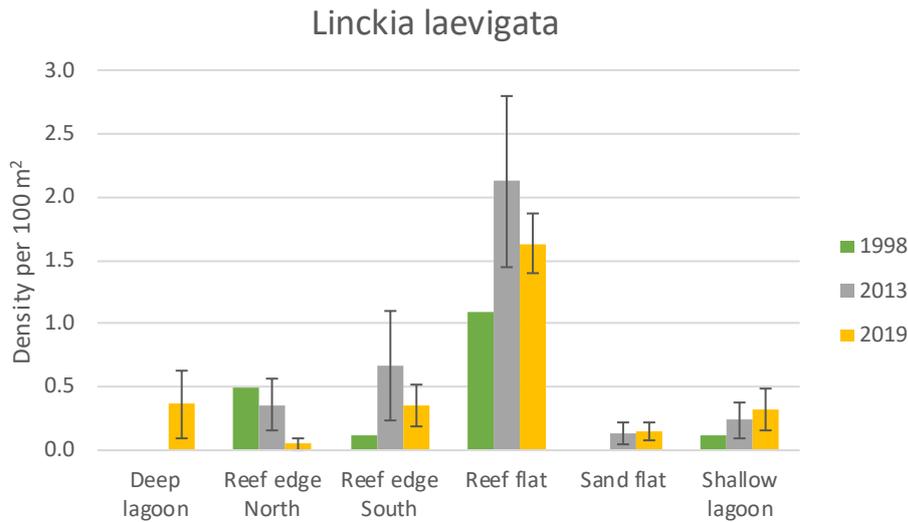


Figure 130. Comparisons of historical and recent surveys of *Linckia laevigata* at Ashmore Reef. Abundances are stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Error bars are  $\pm 1$  s.e.

## Other starfish

### Other starfish distribution and abundance

Two other species of *Linckia*, the comet sea star *L. multifora* and guilding's sea star *L. guildingi* were observed at Ashmore Reef. The former was recorded on the northern reef edge with a density of 0.05/100 m<sup>2</sup> and the latter on both the northern and southern reef edges at densities of 0.25 – 0.3/100 m<sup>2</sup> (Figure 131). *Linckia guildingi* was also observed during manta tows off the southern reef edge. Only three other species (*Luidia maculata*, *Fromia indica* – Indian starfish and *Echinaster luzonicus*), were observed mostly in the deep lagoon at densities of 0.1 – 0.2/ 100 m<sup>2</sup> (Figure 131).

### Other starfish historical trends in density

Marsh *et al.* (1993) noted that *Linckia multifora* was uncommon at Ashmore Reef. This is somewhat surprising in that it mainly reproduces asexually, enabling it to grow and maintain populations without the requirements for proximity of sexually-reproducing invertebrates (Rideout 1978).

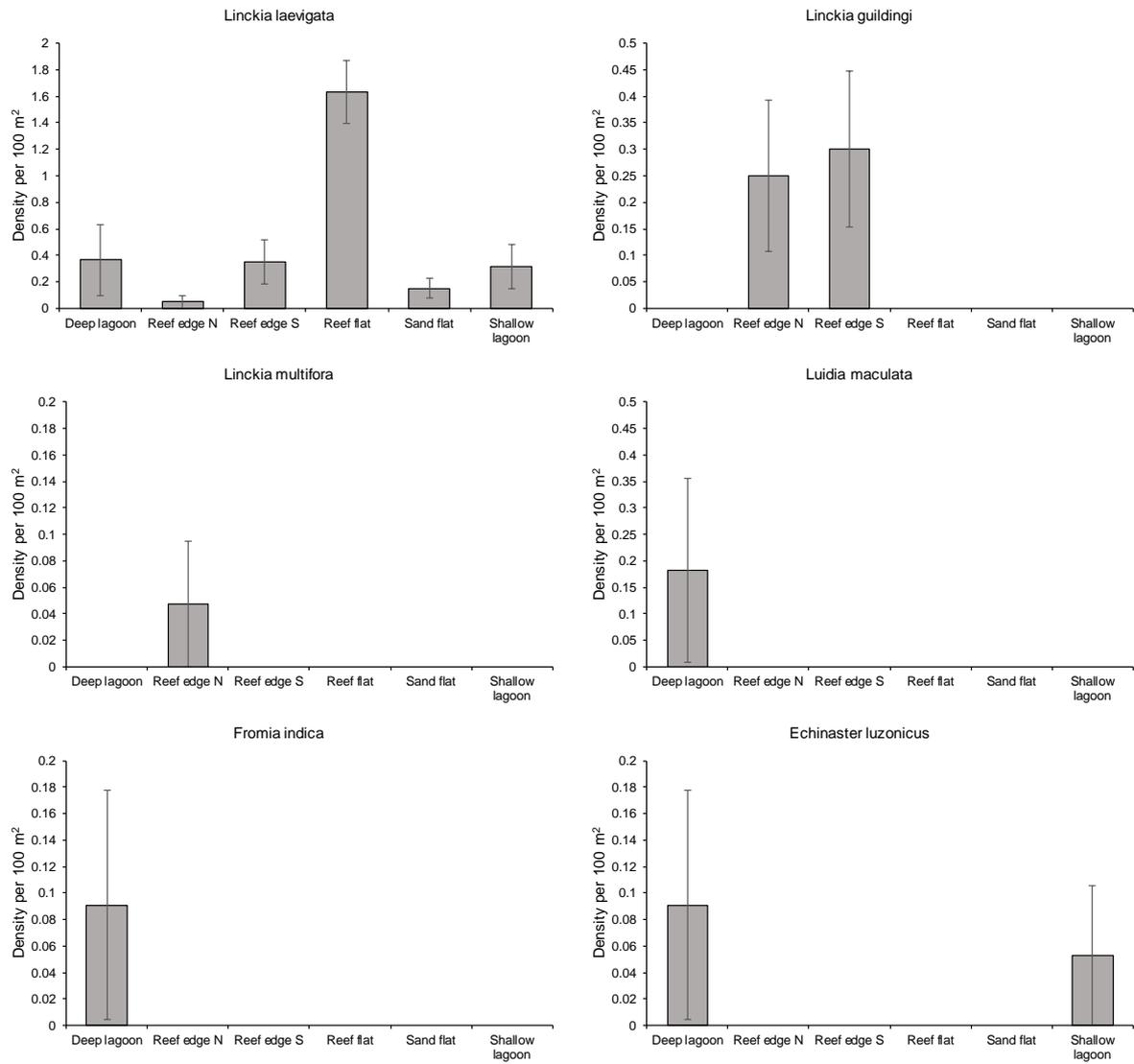


Figure 131. Mean density of starfish in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

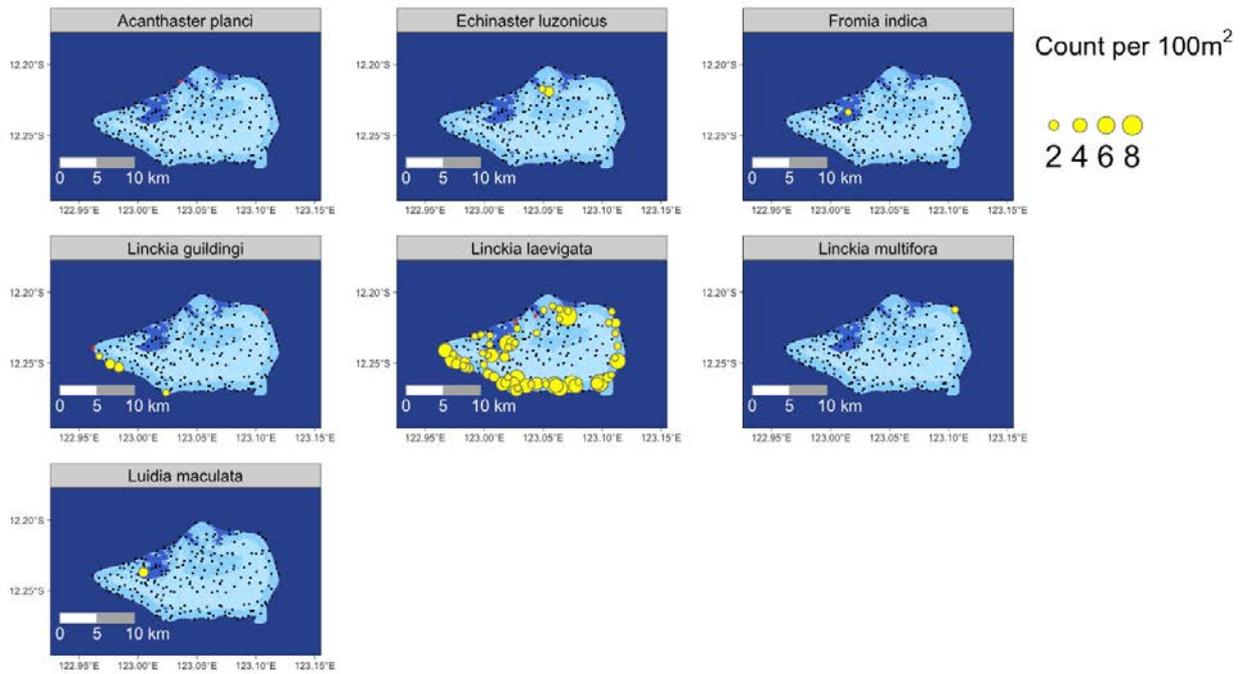


Figure 132. Distribution and density/100 m<sup>2</sup> of all starfish species across each of the 224 transects surveyed at Ashmore Reef in June 2019. The red dots indicate presence of that species at that site but that the specimen(s) were not within the area of the 100 m<sup>2</sup> quantitative survey transect.

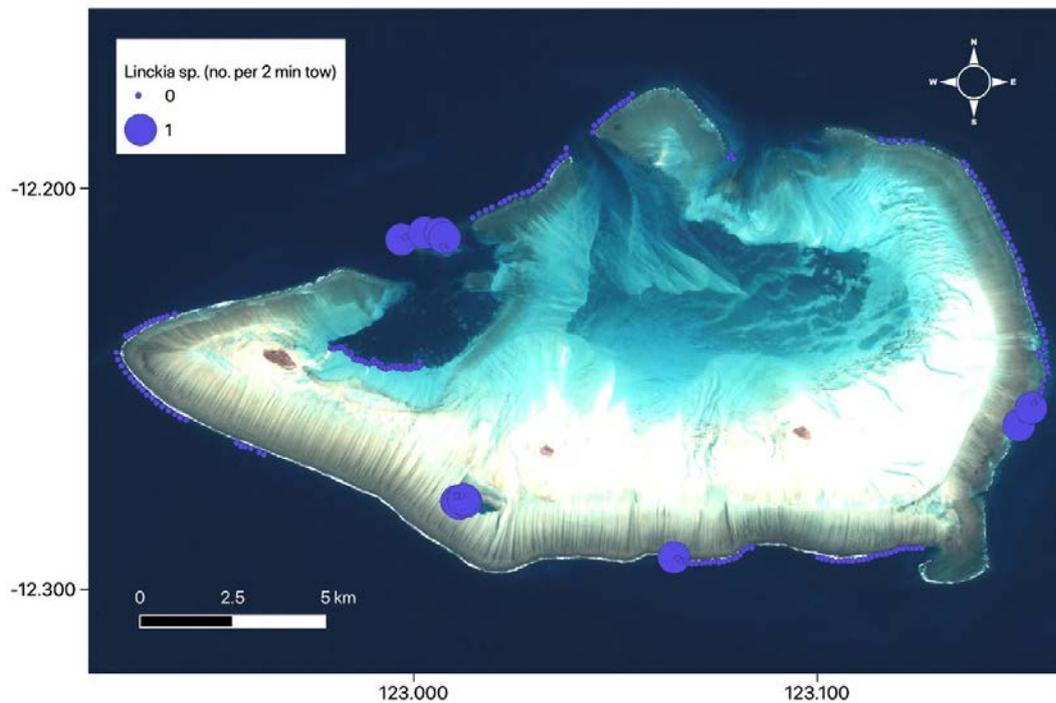


Figure 133. The combined abundance of *Linckia laevigata* and *Linckia guildingi* per each two-minute manta tow at Ashmore Reef.

### 11.4.3 Sea Urchins

#### Sea urchin density and distribution

Four species of sea urchin were recorded; the Pacific urchin *Echinometra mathaei*, the mole urchin *Echinostrephus molaris*, the crowned sea urchin *Echinothurix diadema* and the stinging sea urchin *Echinothurix calamaris*, but none were common in the habitats surveyed with densities of 0.01 to 0.14/100 m<sup>2</sup> on reef edge and reef flat habitats (Figure 134, Figure 135). We did not survey reef crest habitats to any significant extent, particularly on the exposed southern side of the reef, however manta tows of those habitats revealed some extensive areas of *Echinometra mathaei* burrows.

#### Sea urchin historical trends in density

There have not been any previous quantitative surveys of sea urchins at Ashmore Reef. Marsh *et al.* (1993) recorded 23 species of sea urchins, although eight of these were burrowing irregular urchins (sand dollars and heart urchins) and many of the other species were small sea urchins with cryptic habits not suitable for regular monitoring.

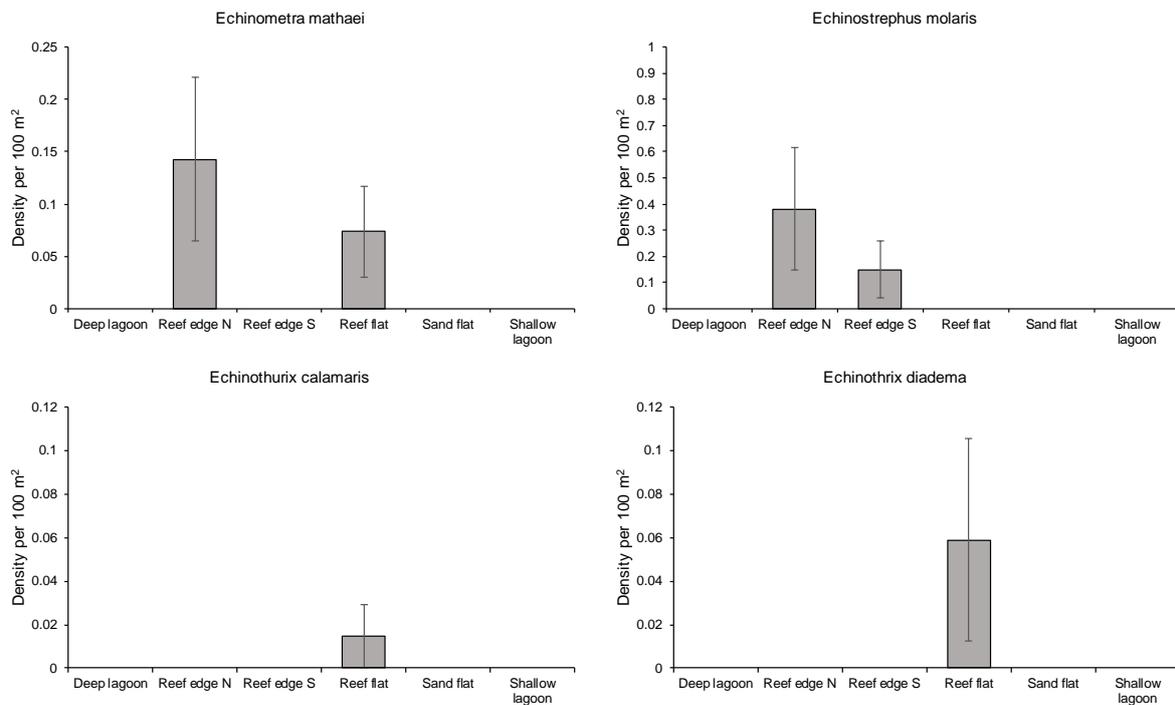


Figure 134. Mean density of sea urchins in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

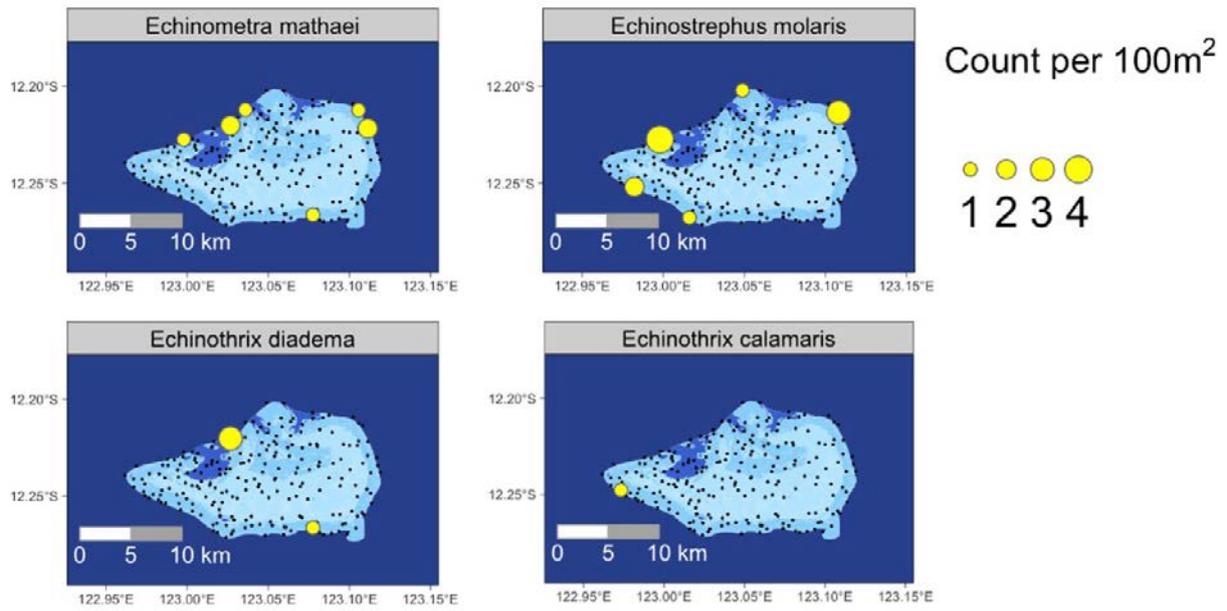


Figure 135. Distribution and density/100 m<sup>2</sup> of all sea urchin species across each of the 224 transects surveyed at Ashmore Reef in June 2019.

## 11.4.4 Crinoids

### Crinoid density and distribution

Crinoids were mostly encountered on the northern and southern reef edge slopes and in the deep lagoon, but they were uncommon, with densities in those three habitats ranging from 0.64 to 0.85/100 m<sup>2</sup> (Figure 136).

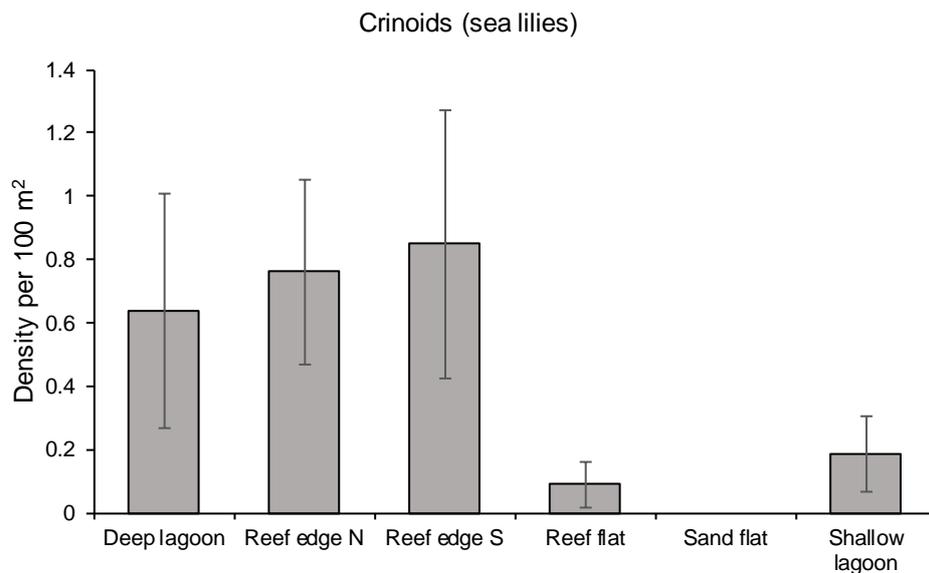


Figure 136. Mean density of crinoids in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

### Crinoid historical trends in density

Marsh *et al.* (1993) recorded 38 species of crinoids at Ashmore Reef but did not comment on their abundance. Crinoids have not been previously surveyed at Ashmore Reef. Abundances in our survey were very low relative to many coral reefs. For example, on the Great Barrier Reef back reef, front slope and deep lagoon habitats comparable to those at Ashmore Reef, Bradbury *et al.* (1987) recorded 21 – 102/100 m<sup>2</sup> which is about two orders of magnitude higher than densities at Ashmore Reef. They sampled the reef destructively and counted cryptic individuals, but this would only partly explain the large discrepancy. It is not known why crinoids are uncommon at Ashmore Reef but the same situation was observed in 2016 at Browse Island, a much smaller continental shelf reef off north-western Australia (Keesing personal observations).

### 11.4.5 Trochus *Rochia niloticus*

#### *Rochia niloticus* spatial patterns in abundance and size structure

Trochus were found in typical shallow reef edge habitat, on reef crests or upper slopes, with the highest densities (0.45/100 m<sup>2</sup>) being on the exposed southern edge (Figure 137). Only 11 *R. niloticus* were measured with size varying from 7 to 11 cm. The modal size was 10 cm (Figure 137).

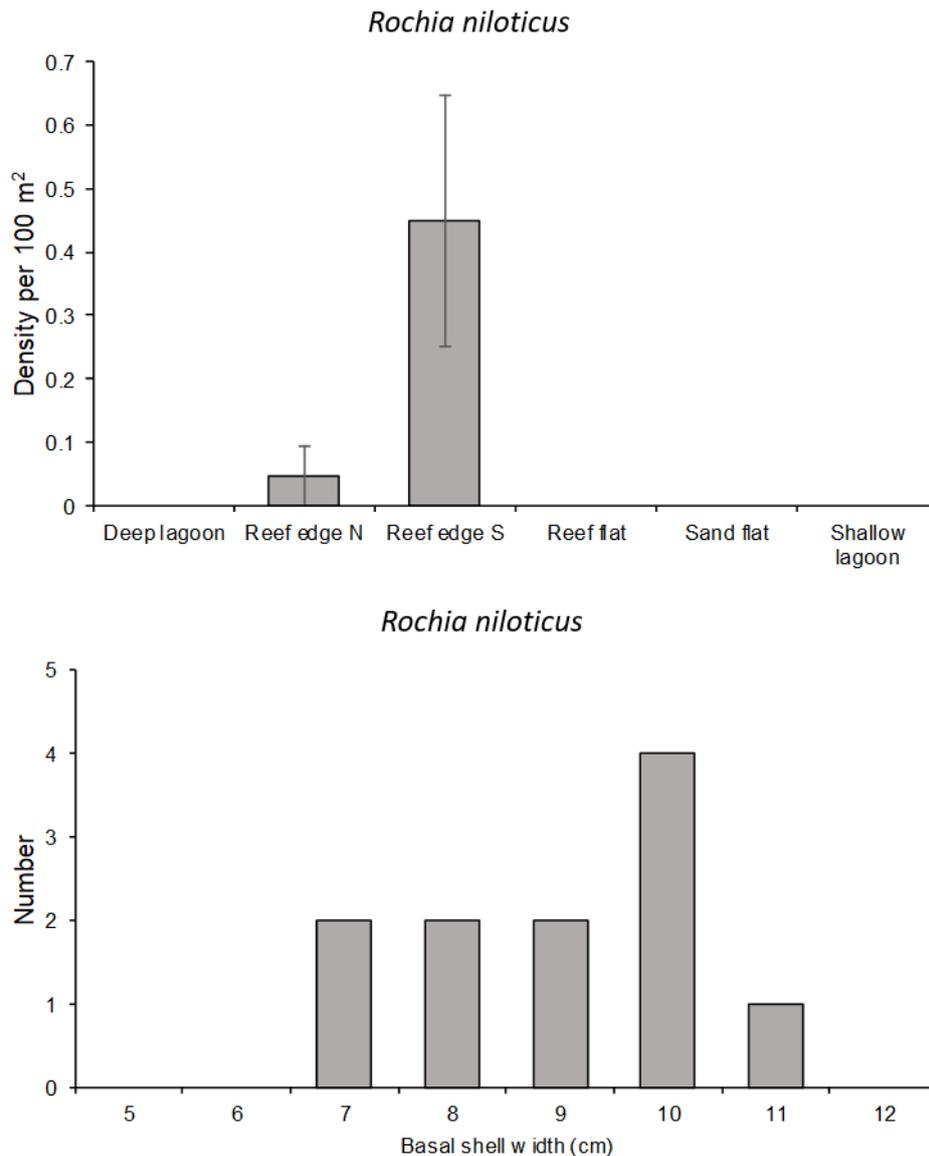


Figure 137. Upper panel is mean density of trochus in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e. Lower panel are shell measurements.

### *Rochia niloticus* historical trends density and size

Skewes *et al.* (1999a) stated that trochus populations at Ashmore Reef were severely depleted as a result of overfishing. Long term analysis of trochus abundance by Hosack and Lawrence (2013a) between 1998 and 2013 indicated numbers had increased significantly by 2005 and, except for a decline in 2009, remained high until 2013 (Figure 138). Subsequent analyses (chapter 12 of this report) indicate numbers have declined again in 2019, although there is still overlap in the 95% central credible intervals, indicating the decline was not significant (Figure 138). Ceccarelli *et al.* (2007, 2011a) determined that trochus numbers had declined between 2005 and 2006 based on surveys that used the same sites and methods. They concluded the cause was most likely illegal fishing, of which there had been reports numerous enough to prompt their 2006 survey. However, they were cautious about the comparison of their results with those of the earlier study by Rees *et al.* (2003), who up until that time had recorded the highest densities of trochus at Ashmore Reef, due to the high level of mobility of trochus and their tendency to aggregate. We continued the time series analysis undertaken up until 2006 by Ceccarelli *et al.* (2011a) by adding the data from 2013 and 2019 (Figure 138). The densities in 2013 were much higher in comparison, making the changes detected between 2005 and 2006 seem small. The 2013 data were based on a smaller number of sites of smaller transect area, recorded highly variable results, with highly aggregated trochus (16 trochus or almost 50% of all trochus counted were on one 80 m<sup>2</sup> transect). Hence the 2013 mean estimate of >250/ha on the southern reef edge had very high variability associated with it. Our counts in 2019 were much lower, just 10 trochus in total, equivalent to 45/ha on the exposed southern reef edge (Figure 138). Overall, considering the results of all surveys and overall trend through time, it is reasonable to conclude that trochus numbers have increased since the 1998 surveys of Skewes *et al.* (1999a).

Trochus are biologically well adapted to recover from localised overfishing. Their high growth rate and early maturity (2 years; Heslinga & Hillmann 1981) tendency to aggregate enhancing fertilisation rates, (Ceccarelli *et al.* 2011a), relatively long life span of 15 – 20 years (Bour, Gohin & Bouchet 1982) that permits many years of spawning potential, and a very short larval durations of non-feeding larvae, all provide an effective life history strategy to replenish population size.

Trochus at Ashmore Reef are likely to comprise a separate metapopulation based on their life history (three-day larval duration, Heslinger and Hillmann 1981). Results of molecular studies (Berry *et al.* 2019) comparing coastal populations in north-western Australia with those of offshore reefs (Scott Reef, Rowley Shoals) found that the latter were genetically isolated from each other and from the coastal populations.

The modal size for trochus in this study (10 cm) was consistent with the larger of two cohorts measured in earlier studies. Skewes *et al.* (1999a) measured 52 trochus and found two size classes with modal sizes 7.5 – 7.9 cm and 10.0 – 10.5 cm. Similarly, Rees *et al.* (2003) found two cohorts with modal sizes of 6.6 – 7.5 cm and 9.6 – 11.5 cm. Their sample size was more than 1,000 shells.

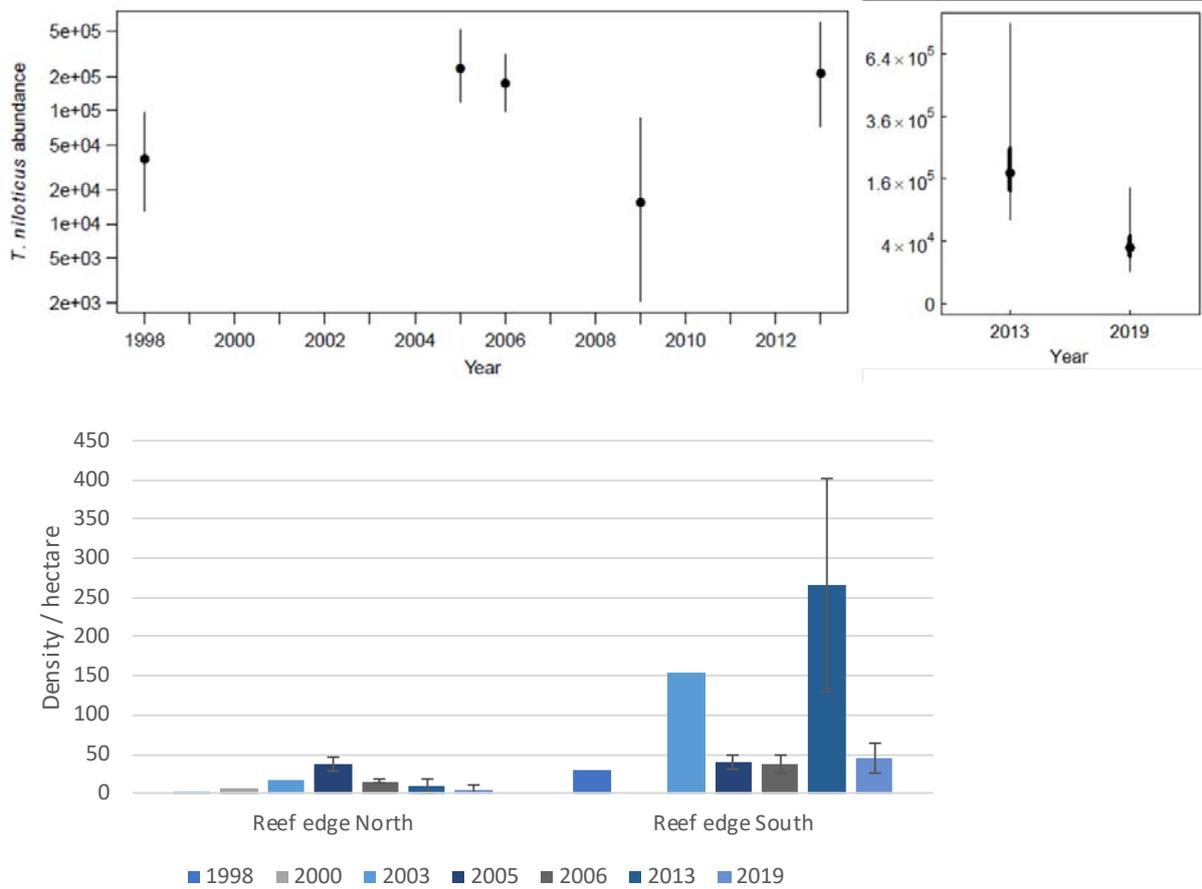


Figure 138. Long term estimates of abundance of trochus *Rochia niloticus* at Ashmore Reef. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both panels. Values are medians and longest error bars are 95% confidence limits. Lower panel is graph adapted from Ceccarelli et al. (2011a) adding data from the 2013 and 2019 surveys with densities expressed per hectare ( $\pm 1$  s.e.). See these cited studies for methods. We were unable to include the results from 2009 (Richards et al. 2009) as they could not be compared directly with the data from the other years.

## 11.4.6 Clams

### Spatial patterns in diversity and abundance of clams

#### *Horse's hoof clam Hippopus hippopus*

*Hippopus hippopus* occurred predominantly on the reef flat (0.37/ 100 m<sup>2</sup>, Figure 139) and was the second most abundant clam counted at Ashmore Reef after *Tridacna maxima*. The distribution of *Hippopus hippopus* in 2013 and 2019 is given in Figure 141 and Figure 142.

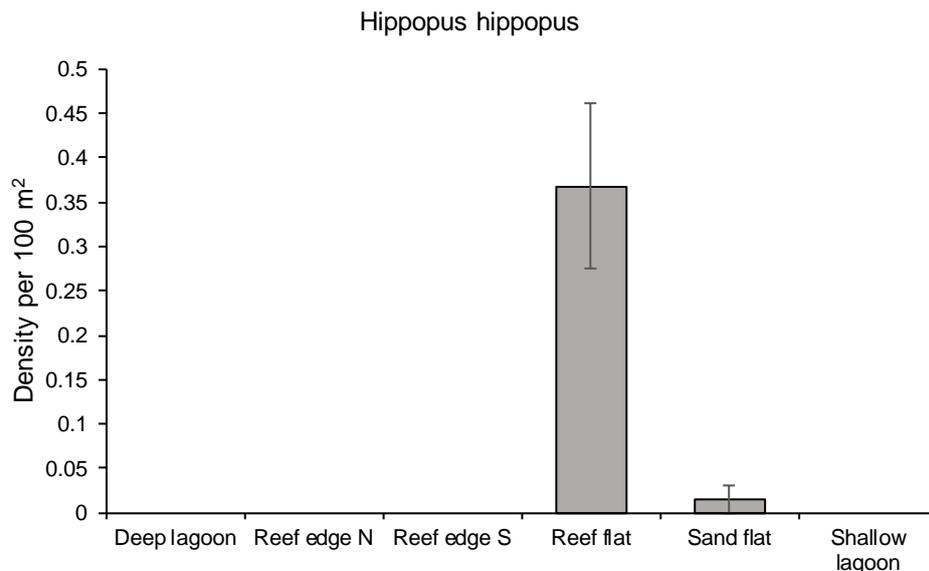


Figure 139. Mean density of the clam *Hippopus hippopus* in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

#### *Tridacna* spp. (Giant clams)

The elongate clam *Tridacna maxima* and the fluted giant clam *T. squamosa*, with which the former is sometimes confused, occur predominantly on the reef slope (Figure 140), while *Hippopus hippopus* was most abundant on the reef flat. The burrowing giant clam *Tridacna crocea* has a burrowing habit and occurred on consolidated substratum of the reef flat and the shallow part of the reef slopes (Figure 140). The blue giant clam *Tridacna gigas* and *Tridacna derasa* southern giant clam also occurred on the reef flat, but predominantly in lagoon and back reef habitats, although we saw few *T. gigas*, just three on the 224 quantitative transects and two on manta tows in the deep lagoon and the large hole in the reef top on the southern side of the reef known as the “Grotto” (Figure 143). The distribution of all clam species in 2013 and 2019 is given in Figure 141 and Figure 142.

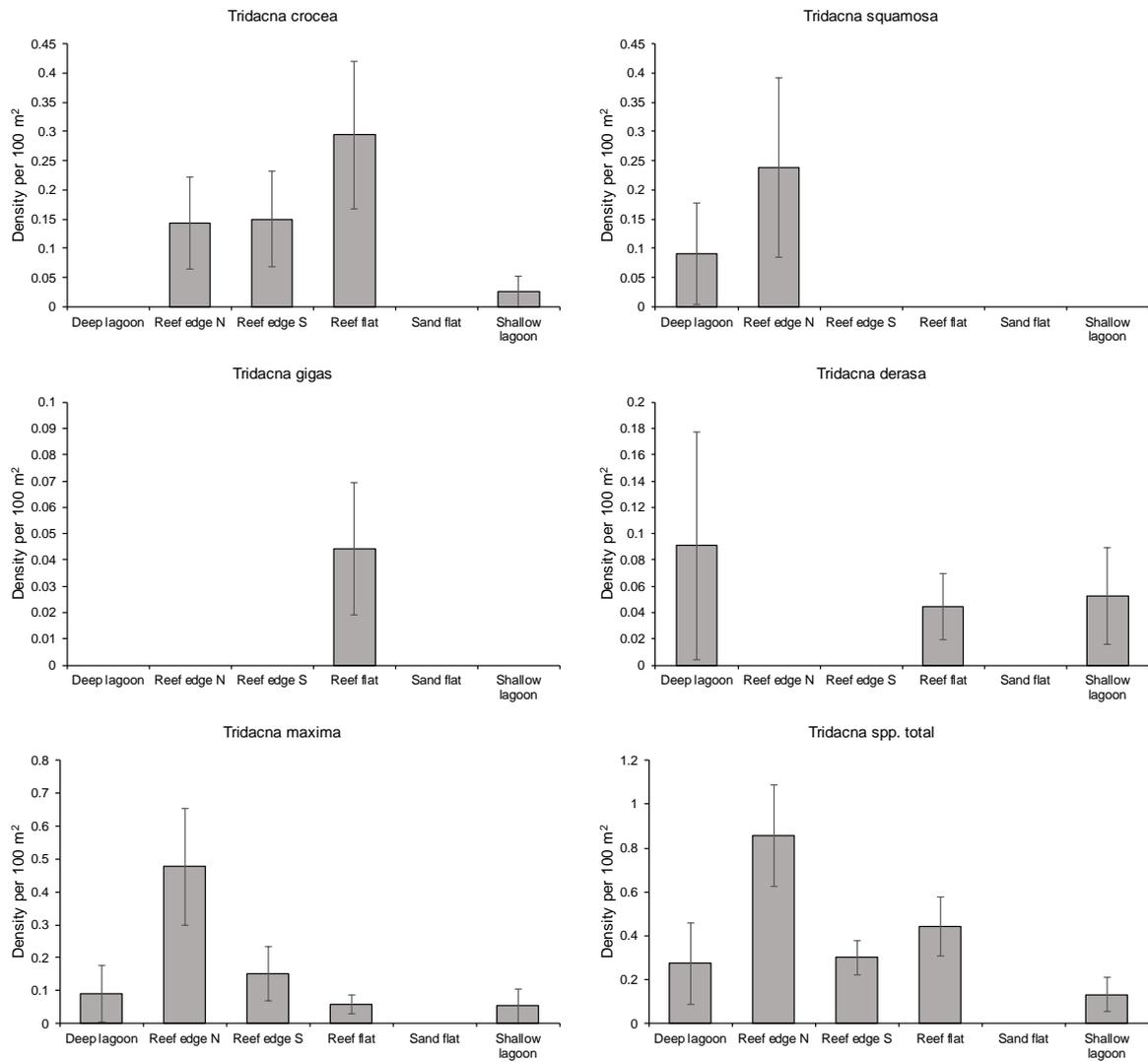


Figure 140. Mean density of tridacnid clam species and combined all species in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

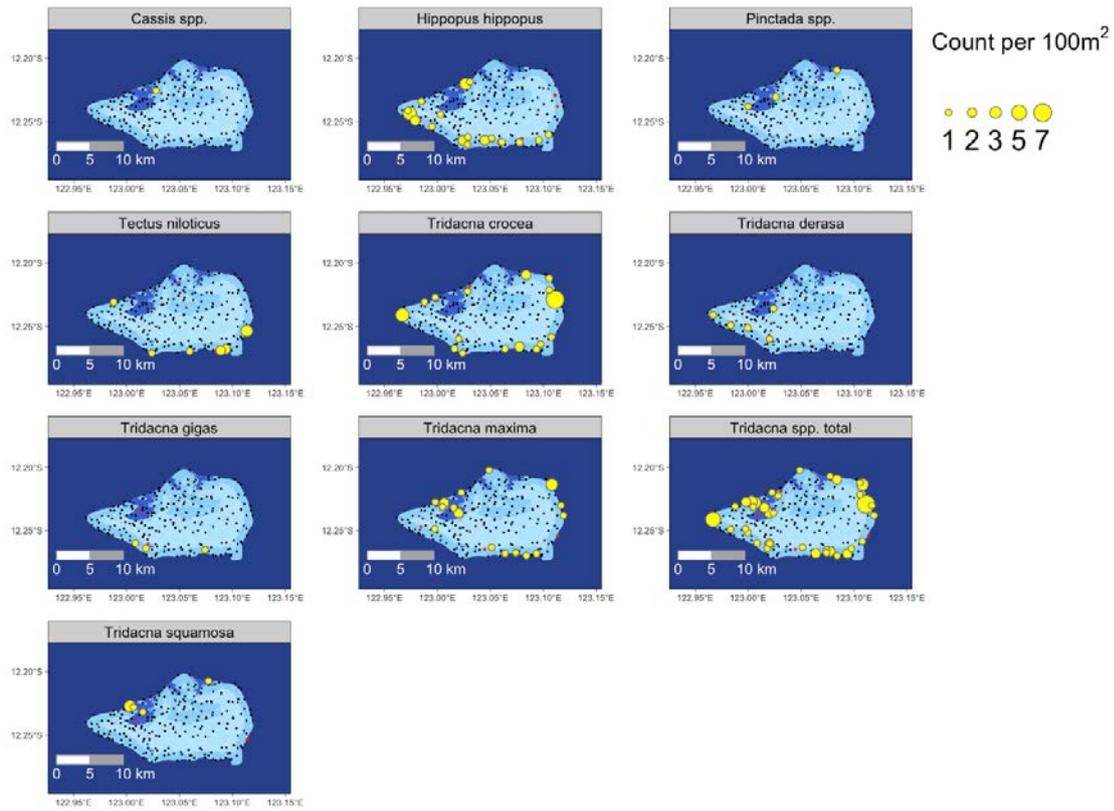


Figure 141. Distribution and density/100 m<sup>2</sup> of all mollusc species across each of the 224 transects surveyed at Ashmore Reef in June 2019. The red dots indicate presence of that species at that site but that the specimen(s) were not within the area of the 100 m<sup>2</sup> quantitative survey transect.

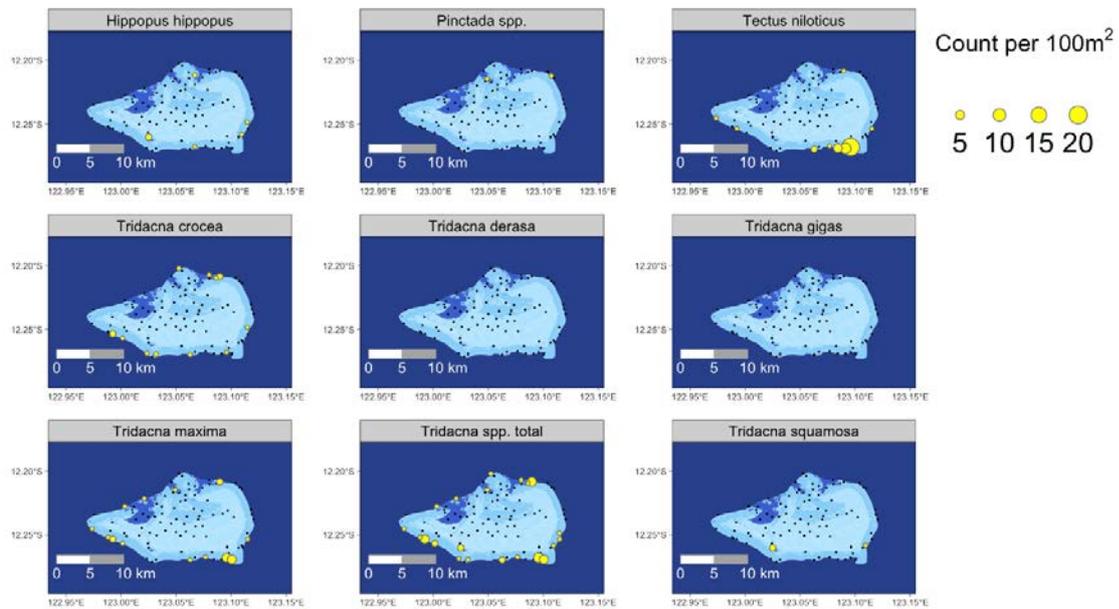


Figure 142. Distribution and density/100 m<sup>2</sup> of all mollusc species across each of the 95 transects surveyed at Ashmore Reef in March 2013. Data from Ceccarelli et al. (2013) with densities converted to per 100 m<sup>2</sup> from their 80 m<sup>2</sup> transects.

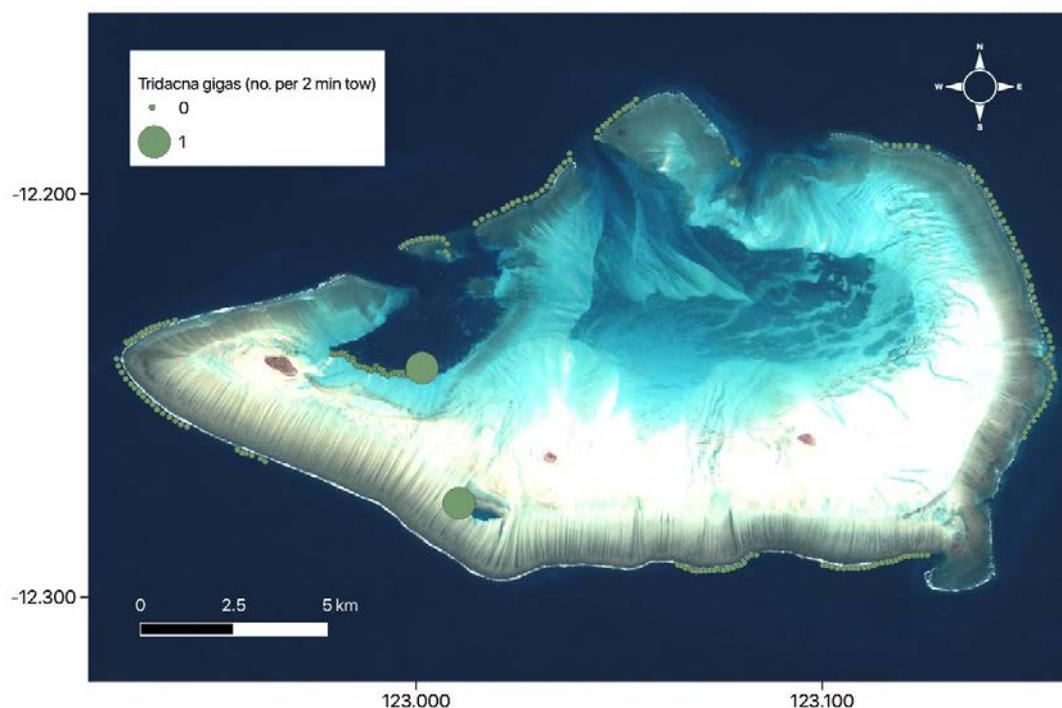


Figure 143. Map of Ashmore Reef showing the location of *Tridacna gigas* counted during the two-minute manta tows

### Clam historical trends in density

We recorded five *Tridacna gigas* which is the highest number of this species recorded at Ashmore Reef since the 1980s. It was not recorded in 2009 or 2013 (Richards *et al.* 2009; Ceccarelli *et al.* 2013) and was regarded as extremely rare in 2005 and 2006 (Ceccarelli *et al.* 2007). Russell and Vail (1988) recorded four live and 16 recently dead *T. gigas* at just four sites in 1987 indicating a very high exploitation rate by Indonesian fishers. Berry (1993) also gave a good account of clam harvest by Indonesian fishers in 1987. In a single low tide, 15 men from two perahu collected the meat from about 1,300 clams, mostly *Hippopus hippopus*, along a 2km section of the southern reef flat. This was an exploitation rate of about 6.5 clams/ha/day. This represents about 20% of the density of *H. hippopus* we measured in 2019 (Figure 139). Rather than for commerce, clam meat was mainly harvested for food for consumption by family back in port or for barter. Nevertheless, very large numbers of clams were harvested, with up to 1,000 clams observed drying on a single perahu (Russell & Vail 1988) Figure 144.

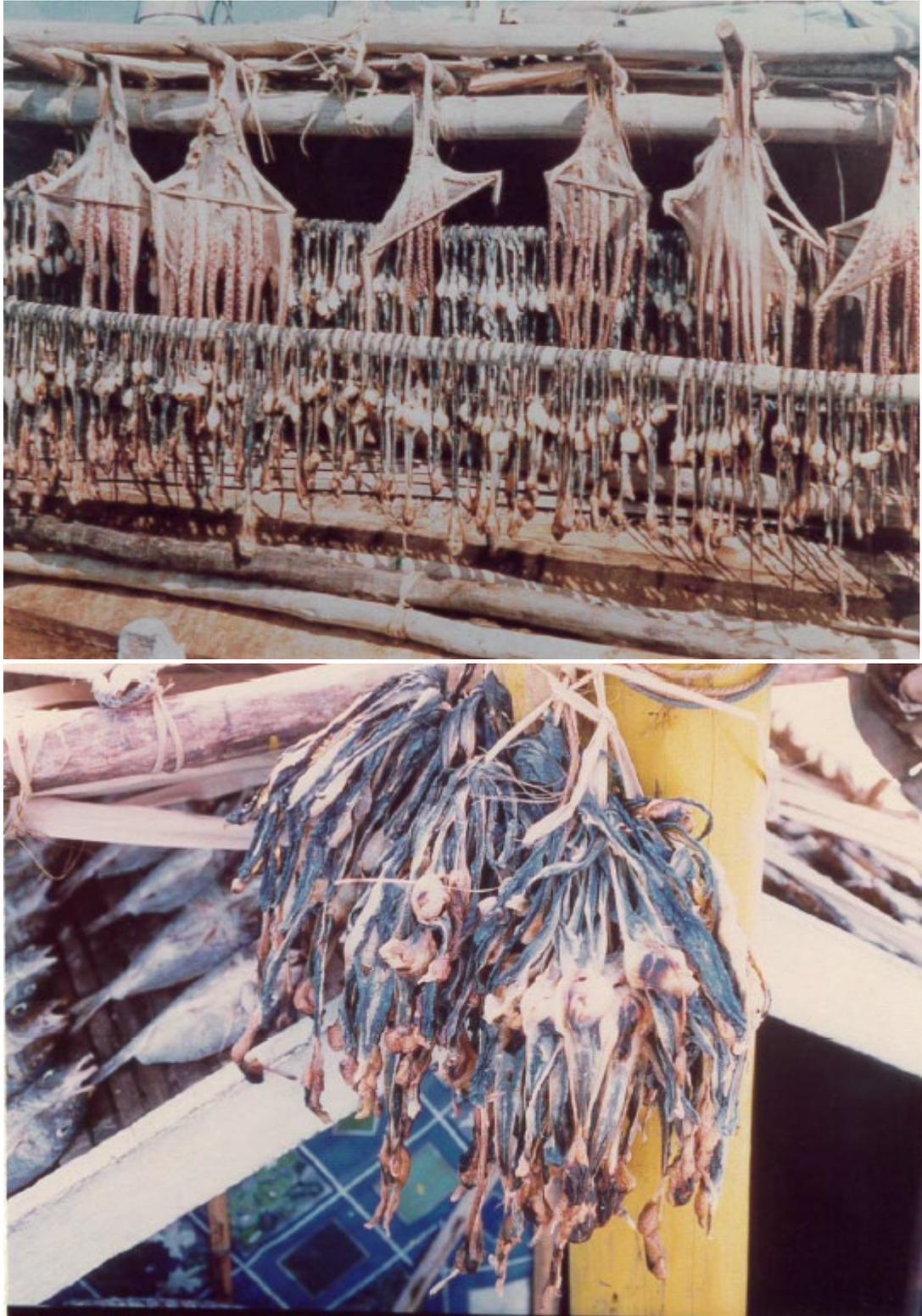


Figure 144. Plate 8 reproduced from Russell and Vail (1988) showing octopus and clam meat (upper panel) and clam meat (lower panel) being dried on board Indonesian perahus at Ashmore Reef in 1986/87. Photographs: Anne Hoggett (upper) and Barry Russell (lower).

It is difficult to compare clam densities across surveys because of the different accounting of some *Tridacna* spp. and spatial stratification measures used, but a “like with like” comparison was possible between 1998 and 2013/2019 using direct comparison of our data with those of Skewes *et al.* (1999a). In total 2013 and 2019 clam numbers were higher than or similar to those recorded in 1998 by Skewes *et al.* (1999a) (Figure 145). For *T. crocea* which was not harvested historically (Russell & Vail 1988), densities in 2013 and 2019 of 0.05 – 0.50 /100 m<sup>2</sup> are similar to those recorded by Skewes *et al.* (1999a) but the 2013 and 2019 densities of *H. hippopus* are much lower (Figure 145).

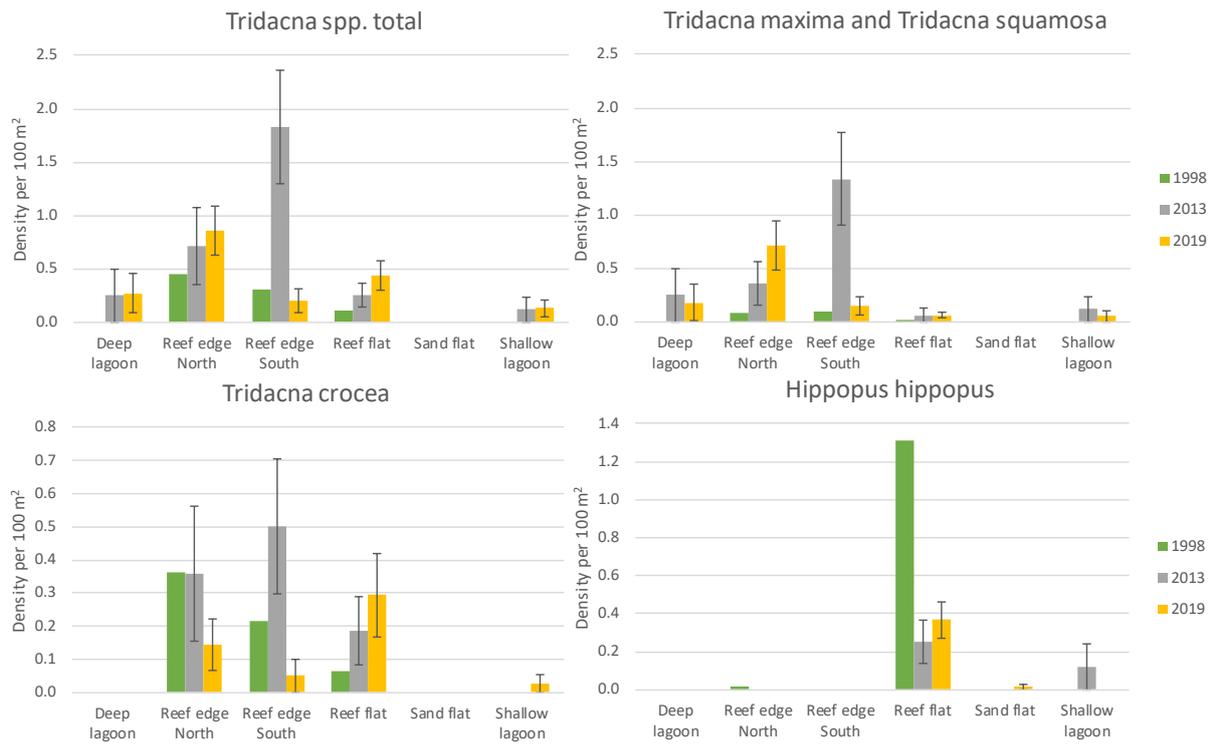


Figure 145. Comparisons of historical and recent surveys of clams at Ashmore Reef stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Error bars are ± 1 s.e.

Densities of *Hippopus hippopus*, *Tridacna maxima* and *T. squamosa* (combined) in 2013 and 2019 were much higher than those measured for these species in 2005 and 2006 by Ceccarelli *et al.* (2007), where (14–16 *Tridacna maxima* and *T. squamosa*/ha and 1–4 *H. hippopus*/ha were recorded). Ceccarelli *et al.* (2007) also found that almost half of all *H. hippopus* were recently dead, consistent with concerns that had been raised about illegal fishing at Ashmore Reef earlier that year. Hosack *et al.* (chapter 12 of this report) found that *Tridacna maxima* and *T. squamosa* had declined significantly between 2013 and 2019. Except for these species, the recent data suggest clams may be increasing in density, but differences in habitat stratification and survey methods make such comparisons difficult.

## 11.4.7 Pearl oysters and helmet shells

### Pearl oyster and helmet shell distribution and abundance

Neither pearl oysters (*Pinctada spp.*) (0.015 – 0.091/ 100 m<sup>2</sup>) nor helmet shells (*Cassis cornuta*) (0.026/100 m<sup>2</sup>) were common in the 2019 survey. *Pinctada* was recorded from just 3 sites and *Cassis* from a single site (Figure 146).

### Pearl oyster and helmet shell historical trends in density

Wells (1993) recorded *Cassis cornuta* at only one site at Ashmore Reef. Russell and Vail (1988) report that Indonesian fishers would harvest pearl oysters for their meat and for pearls, and helmet shells for the ornamental shell market. Ceccarelli *et al.* (2013) recorded densities of *Pinctada spp.* at 0.25/100 m<sup>2</sup> in the deep lagoon and 0.09/100 m<sup>2</sup> on the northern reef edge. The only species of *Pinctada* recorded at Ashmore Reef by Wells (1993) was *Pinctada margaritifera* (five stations) and this is almost certainly the same species recorded in 2013 and 2019.

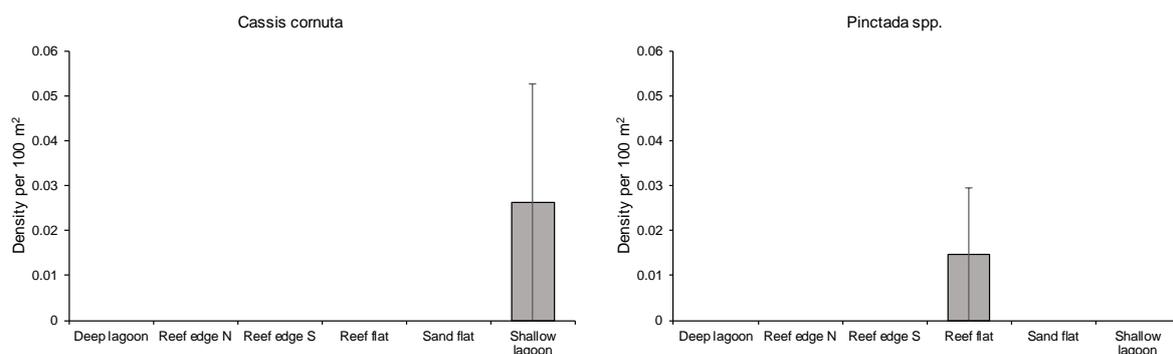


Figure 146. Mean density of helmet shells (*Cassis*) and pearl oysters (*Pinctada*) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are  $\pm 1$  s.e.

## 11.5 Discussion

### 11.5.1 Diversity

#### Overall echinoderm diversity

Ashmore Reef has a history of very high echinoderm diversity. Marsh *et al.* (1993) recorded a total 178 species of echinoderms from Ashmore and Cartier Reefs and noted the very high diversity compared to Scott Reef and Seringapatam (119) and the Rowley Shoals (90). While some of this discrepancy was attributed to a higher search effort at Ashmore Reef, it was mainly thought to arise from the Influence of both the Indonesian Archipelago fauna and the direct link of Ashmore Reef to the Australian and Papuan continental shelves.

### Holothurian diversity

The holothurian diversity at Ashmore Reef was particularly high (47 species, Marsh *et al.* 1993). This was based on three taxonomic surveys, by the Russians in 1978, and the Western Australian and Northern Territory Museums in 1986 and 1987 (Marsh *et al.* 1993). Some of those species are cryptic or burrowing and would not be expected to be found in a quantitative survey relying on visual census techniques, such as ours (we recorded 18 species in 2019, Table 27). However, there is some evidence that fishing has led to the local extinction of some species at Ashmore Reef (Smith *et al.* 2001), and more recent surveys support that view (Table 27). The most consistently recorded species since 1988 (at least 8 out of 9 surveys) were *Holothuria atra*, *Holothuria edulis*, *Holothuria argus*, *Holothuria fuscogilva*, *Holothuria leucospilota*, *Holothuria whitmaei*, *Pearsonothuria graeffei*, *Stichopus chloronotus*, *Stichopus herrmanni* and *Thelenota ananas*. Species richness in surveys that recorded actual species (not genera) only varied little, ranging between 16 and 19. However, *Holothuria lessoni* has not been recorded since 1987 (Russell & Vail 1988). This particularly high value commercial species made up between 11 and 17 % of the catch on eight Indonesian fishing vessels surveyed in 1987. *Holothuria scabra* has only been recorded by the Soviet survey in 1978 (Marsh *et al.* 1993). The species is illustrated by Russell and Vail (1988), but it is not evident that they observed it at Ashmore Reef.

Table 27. Holothurian species recorded at Ashmore Reef in quantitative surveys since 1987. Does not include extensive list of holothurians for Ashmore and Cartier Reefs published by Marsh et al. (1993). \*Smith et al. (2001) stated they found 19 species but only 12 were recorded in the report. \*\* Some surveys recorded *Actinopyga* as a generic group. \*\*\* Richards et al. (2009) show a photograph of an unidentified species which looks very similar to *Holothuria fuscobrunnea*. \*\*\*\* Russell and Vail (1988) illustrate both *Holothuria scabra* and *H. fuscobrunnea* but it is not clear if they observed these species at Ashmore Reef. Sources of data are Russell and Vail (1988); Skewes et al. (1999a); Smith et al. (2001); Rees et al. (2003); Kospartov et al. (2006); Ceccarelli et al (2007); Richards et al. (2009); Ceccarelli et al. (2013) and this study.

Species	Common name	1987	1998	2000	2003	2005	2006	2009	2013	2019
<i>Actinopyga echinites</i>	Hedgehog sea cucumber	Y	-						-	
<i>Actinopyga lecanora</i>	Reef sea cucumber	Y	-		Y	Y	Y		-	Y
<i>Actinopyga mauritiana</i>	Mauritian sea cucumber	Y	-		Y	Y	Y		-	Y
<i>Actinopyga miliaris</i>	Military sea cucumber		-			Y	Y	Y	-	Y
<i>Actinopyga obesa</i>	Plump sea cucumber	Y	-						-	
<i>Actinopyga palauensis</i>	Palauan sea cucumber		-						-	Y
<i>Actinopyga</i> scored as a group	-	-	Y	-	-	-	-	-	Y	-
<i>Bohadschia argus</i>	Eyed sea cucumber or leopardfish	Y	Y	Y	Y	Y	Y	Y		Y
<i>Bohadschia marmorata (aka vitiensis)</i>	Marbled sea cucumber	Y			Y	Y	Y			
<i>Holothuria atra</i>	Black sea cucumber	Y	Y	Y	Y	Y	Y	Y	Y	Y
<i>Holothuria coluber</i>	Snake sea cucumber			Y	Y		Y			Y
<i>Holothuria edulis</i>	Unsavoury sea cucumber	Y	Y	Y	Y	Y	Y	Y		Y
<i>Holothuria fuscogilva</i>	White teatfish	Y	Y	Y	Y	Y	Y	Y	Y	Y
<i>Holothuria fuscopunctata (aka axiologa)</i>	Black-banded sea cucumber	Y	Y		Y	Y	Y		Y	Y

<b><i>Holothuria fuscrobura</i></b>	Orange spotted sea cucumber	****						Y	***	Y	
<b><i>Holothuria impatiens</i></b>	Restless sea cucumber	Y									
<b><i>Holothuria lessoni</i> (aka <i>timana/aculeata</i>)</b>	Golden sandfish or Lesson's sea cucumber	Y									
<b><i>Holothuria leucospilota</i></b>	Stained sea cucumber	Y	Y	Y	Y	Y	Y	Y		Y	Y
<b><i>Holothuria scabra</i></b>	Sandfish	****									
<b><i>Holothuria whitmaei</i> (aka <i>nobilis</i>)</b>	Black teatfish	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
<b><i>Holthuria rigida</i></b>	Rigid sea cucumber										Y
<b><i>Pearsonothuria graeffei</i></b>	Graeffe's sea cucumber	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
<b><i>Stichopus chloronotus</i></b>	Green sea cucumber	Y	Y	y	Y	Y	Y	Y	Y	Y	Y
<b><i>Stichopus herrmanni</i> (aka <i>variegatus</i>)</b>	Herrmann's sea cucumber	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
<b><i>Thelenota ananas</i></b>	Prickly redfish	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y
<b><i>Thelenota anax</i></b>	Royal sea cucumber	Y	Y	Y	Y	Y	Y	Y			Y
<b>Total</b>		19	13**	12*	16	16	18	11	11**	18	

## Starfish diversity

We recorded seven species of starfish at Ashmore Reef: *Acanthaster planci*, *Echinaster luzonicus*, *Fromia indica*, *Linckia guildingi*, *L. laevigata*, *L. multifora* and *Luidia maculata*. This compares with 28 species known from Ashmore Reef (Marsh *et al.* 1993). Although those authors did not record *Luidia maculata*, its occurrence at Ashmore Reef is not a surprise as it is a very widespread species (Keesing 2019). What was a surprise is that we did not record any of the six very large oreasterids (*Choriaster granulastus*, *Calcita novaeguineae*, *Pentaceraster multispinis*, *P. regulus*, *P. linckii* and *P. nodosus*) recorded in Marsh *et al.* (1993) and typical of large oceanic reef systems with a wide variety of habitat types. For instance, *Pentaceraster* spp. often occur on seagrass beds, which were extensive at Ashmore Reef and were expected to host at least some species of this genera.

## Clam diversity

We observed all six species of tridacnid clams that have previously been recorded from Ashmore Reef; *Hippopus hippopus*, *Tridacna gigas*, *T. maxima*, *T. crocea*, *T. squamosa*, and *T. derasa*).

### 11.5.2 Historical exploitation of holothurians at Ashmore Reef

Macassan (Sulawesi) fishers have travelled to northern Australia to collect holothurians since the 1700s (McKnight 1976; Schwerdtner-Máñez & Ferse 2010; McKinnon *et al.* 2013), and there is evidence of Indonesian fishers harvesting holothurians from Ashmore Reef since the early 1800s (Stacey 2007). Russell and Vail (1988) interviewed a large number of Indonesian fishers who had travelled from Sulawesi and more southern islands of Indonesia (mostly Roti, Timor, Buton, Tundu and Passi Tallu) in small vessels called perahus and fished on reef flats and around the reef with the aid of small dugout canoes. These fishers indicated that the levels of fishing for holothurians recorded then had only been taking place for 3 – 6 years prior to their study (1987), and that the market for them had only been in place since about 1985. This coincides with the global expansion of the sea cucumber trade around the world in the mid-1980s triggered by two events; the Chinese opening up its economy and the removal of a range of trade barriers to doing business with China (Kinch *et al.* 2008).

This background is relevant to understanding the timing and extent of overharvesting of holothurians at Ashmore Reef. We can surmise that it began in the early- to mid- 1980s (Russell & Vail 1988) and continued until at least 1988, when restrictions were tightened by Australia to reduce the areas at Ashmore Reef that could be accessed under the MOU between Australia and Indonesia (Commonwealth of Australia 2002). Surveillance, however, was insufficient to completely restrict illegal fishing.

Evidence of the enormous scale of the harvest comes from reports from 1986, 1987 and 1988 by Russell and Vail (1988) and Berry (1993). Berry (1993) visited Ashmore Reef for 9 days in September 1986 and at least 25 perahus present throughout the entire time, fishing for holothurians, fish and clams. Russell and Vail (1988) recorded the number of perahu visits to Ashmore Reef as between two and 52 per month (between March and November, with biomodal peaks in April and September) during 1986–88, and noted that they stayed between one and 31 days per month. They found that 112 vessels, or 73% of the fleet, visited annually. In total there were about 260 visits in the two and half years of their survey (an average of 86 per year). Not all were focussed on harvesting holothurians, some targeted sharks and fish, and some harvested trochus and pearl oysters as well as holothurians. Maximum catches per vessel per trip were 700–

800 kg dry weight of holothurians, with 500 kg being more typical. Russell and Vail (1988) estimated that the harvest of holothurians from Ashmore Reef was about 120 tonnes live weight per year.

Converting this to numbers harvested and exploitation rates per hectare requires some assumptions. The average live weight of a holothurian harvested from Ashmore Reef was estimated to be 526 g based on Skewes *et al.* (2004) and being the average of 345 g for *H. atra* (which made up 80% of catch by number, Russell and Vail 1988) and the remainder being a mix of larger species like *Actinopyga* spp. (930 – 1,100 g) and *H. whitmaei* (1,500 g). Thus 120 tonnes live weight per year converts to approximately 228,000 individuals per year.

The area of the reef where most holothurians were most likely to have been taken was the reef flat (82.5 km<sup>2</sup>) and shallow lagoon (41.1 km<sup>2</sup>) which excludes islands and sand flats (61.2 km<sup>2</sup>) and the deep lagoon (31.8 km<sup>2</sup>). Thus, 120 tonnes live weight per year is equivalent to 18.4 per hectare. Over the five-year period from 1984 – 1988 it can be estimated that on average 92 holothurians per hectare were harvested. This was approximately the same density as estimated for *Holothuria atra* alone in 1998 by Skewes *et al.* (1999a).

Three earlier studies (Russell & Vail 1988; Smith *et al.* 2001; Rees *et al.* 2003); surveyed and/or reviewed the abundances published in the literature for some of the high value holothurian species on other reefs that have much lower fishing pressure. All studies found the abundances at Ashmore Reef were much lower even at the time when quantitative surveys began. Shiell (2005) contrasted populations of *Holothuria whitmaei* between Ashmore Reef and other heavily fished reefs with the more pristine Mermaid and Ningaloo Reefs off Western Australia, where he found this species occurred at densities of 9 – 27 per hectare in suitable shallow water habitats (Figure 147).

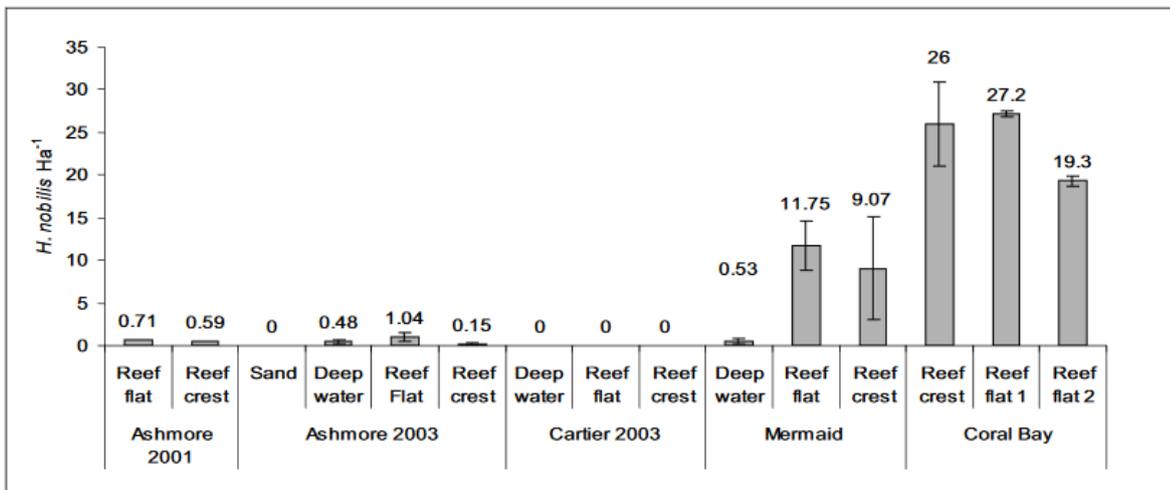


Figure 147. Comparison of densities of *Holothuria whitmaei* at the heavily fished Ashmore and Cartier Reefs with that of the less fished Mermaid Reef and the unfished Coral Bay (Ningaloo Reef) in 2004. The graph shown is figure 2 from Shiell (2005). Data from Ashmore and Cartier Reefs is taken from Smith *et al.* (2001), Smith *et al.* (2002) and Rees *et al.* (2003).

Although the possibility that some harvested holothurian species have always been rare at Ashmore Reef cannot be excluded, it is likely that by the time of the earliest surveys by Russell and Vail (1988) and Skewes *et al.* (1999a), holothurians were already heavily depleted, and that

subsequent monitoring projects such as ours are merely documenting minor fluctuations in highly spatially variable, low abundance populations of the remnant populations of some of these species.

Naturally low populations of *H. whitmaei* do exist. Bellchambers *et al.* (2011) found *H. whitmaei* was rare at Cocos Islands where they claimed fishing for holothurians was historically negligible, although Greer *et al.* (2012, 2014) suggested some fishing for holothurians had taken place historically at Cocos Islands. However, fishing for holothurians at Cocos Islands seems to have been light and their abundance (Bellchambers *et al.* 2011) contrasts that of the heavily overfished populations of spider conchs and clams, with the largest, *Tridacna gigas* being overfished to the extent it may be locally extinct (Greer *et al.* 2014).

Numerous studies have shown that serially depleted holothurian stocks rarely recover and that fisheries that deplete a stock tend to progress in space and by moving to lower value species continuing to degrade stocks (Uthicke, Schaffelke & Byrne 2009; Anderson *et al.* 2011; Friedman *et al.* 2011; Purcell *et al.* 2013; Eriksson & Byrne 2015). Interviews of Indonesian fishers at Ashmore Reef by Russell and Vail (1988) confirmed that accessible reefs closer to their home ports in Indonesia were already heavily depleted and no longer supported the catch rates that could be attained at Ashmore Reef in the 1980s.

The continued absence of *Holothuria scabra* and *H. lessoni* and the very low abundances of other species that have been heavily fished in the past (*Holothuria whitmaei*, *Bohadschia marmorata*, *Thelenota ananas*, *Stichopus herrmanni* and *Actinopyga spp.*) suggest that their population levels are too low and distance between individuals too great to allow effective fertilisation rates and to contribute significantly to recruitment from self-seeding. Instead, these population, are most likely reliant on larvae dispersed from distant populations for recovery.

This proposition is in part supported by the continued presence of the asexually reproducing form of *Holothuria atra* in relatively high numbers at Ashmore Reef, despite this species often making up 80 % of the catch by Indonesian fishers Russell and Vail (1988). Another species which was heavily harvested in the past and also reproduces asexually, *Stichopus chloronotus*, remains the second most abundant holothurian at Ashmore Reef.

Smith *et al.* (2001) and Rees *et al.* (2003) were the only studies to conduct specific surveys in the deep-water lagoon and back reef habitats down to 20m. They found that some target species, such as *Holothuria fuscogilva*, *Thelenota ananas* and *Thelenota anax*, were more common in these habitats than in shallower habitats and suggested that they may have been less vulnerable to divers in the deeper water. In our study we did not include these deep habitats beyond about 12 m, however, for another part of the study involving fish surveys on coral reef bommies at greater depths, we saw species such as *Thelenota ananas* and *Thelenota anax*, which were rare or absent on our 224 invertebrate transects. It is likely these deep-water habitats have provided a “depth refuge” from fishing in the past and may still support higher densities of holothurians.

### 11.5.3 Historical comparisons of density or abundance

There have been eight quantitative surveys of one or more of holothurians, clams and trochus at Ashmore Reef prior to our survey in 2019 (1987 by Russell & Vail 1988; Skewes *et al.* 1999a, 2000; Smith *et al.* 2001; 2003; Rees *et al.* 2003, 2005; Kospartov *et al.* 2006; Ceccarelli *et al.* 2007, 2009; Richards *et al.* 2009; 2013; Ceccarelli *et al.* 2013). The results of some of these studies have been synthesised and compared by Ceccarelli *et al.* (2011a) and Hosack and Lawrence (2013a), and recent trends in the abundance of individual species have been discussed in detail earlier in this chapter.

#### Holothurians

There are two possible interpretations of the temporal abundance patterns of holothurians observed at Ashmore Reef since the mid to late 1980s. Firstly, that the decline in abundance between 1998/2000 and 2005 – 2009 was followed by recovery to 1998 levels leading up to the 2013/2019 surveys. The alternative interpretation is that abundances have not change significantly between 1998 and 2013/2019 and that the lower levels of abundance detected for some species between 2005 and 2009 were due to sampling differences (including comparisons with deep water surveys in 2001 and 2003), and problems associated with surveying fundamentally rare animals with patchy distributions.

Of these two hypotheses, the second interpretation is more likely, and abundances of holothurians were severely depleted from unfished levels before the 1998 surveys of Skewes Skewes *et al.* (1999a). This means that the surveys since 1998 have attempted to find trends of either improvement or further declines from a very low baseline. This would have, resulted in volatility in the data, and has precluded confidently determining either a long-term decline or recovery in stocks of some species.

Both interpretations assume that the 1998 survey (Skewes *et al.* 1999a) represents a reliable baseline for that time. This assumption is probably valid given the extensive scale of the survey. However, this represents a post-depletion baseline if we accept that holothurian stocks were heavily overfished prior to 1998, and Russell and Vail (1988) present strong evidence for this. All surveys since 1998 have reflected population statuses typical of low-level abundances of species that have not recovered over long periods of time.

Several lines of evidence support this interpretation:

- Like for like comparisons between 1998 and 2013/2019 and between 2006/2009 and 2013/2019 are made (see our analysis in previous sections of this chapter), both show little change or such high variability that one cannot conclude they are different in many species.
- Two high value species, *Holothuria scabra* and *Holothuria lessoni*, the latter of which was heavily fished at least until 1987 (Russell & Vail 1988) are probably locally extinct. They have not been recorded in thousands of hours of survey time between 1998 and 2019.
- Other high value species including *Holothuria whitmaei* and *Thelenota ananas* which were also heavily fished (Russell & Vail 1988; Marsh *et al.* 1993) remain in very low numbers. We found just two *H. whitmaei* in 2019, three were seen in 2013 (Ceccarelli *et al.* 2013) and six in 2009 (Richards *et al.* 2009).

- Several other species that were common on the basis of catch surveys or reef surveys in 1986/1987 (Russell & Vail 1988) are now very rare including some unspecified *Actinopyga* species and *Bohadschia marmorata*.

In the absence of adult migration, which, for an isolated reef like Ashmore Reef can be ruled out, recruitment will rely on a combination of local reproduction (self-seeding) and larval immigration. It is unlikely that rare, widely separated individuals at Ashmore Reef can successfully reproduce, because the distances between them when spawning will be too great (e.g. Pennington 1985; Levitan 1991). Sparsely located individuals will need to move closer together for reproductive success. In addition, Ashmore Reef is remote to other reefs and thus other sources of larvae will be highly diluted by the time they reach Ashmore Reef. It is possible for long-range vagrants to arrive on the reef, but the isolation of Ashmore Reef means that these would be rare events. Additionally, potential source populations on Indonesian reefs are also heavily depleted, reducing reproductive output. These conditions both suggest that recruitment levels will be very low at Ashmore Reef.

Further, for a population to be sustained and grow, recruitment and immigration must exceed mortality and emigration. Emigration of shallow water holothurians from an isolated emergent reef like Ashmore Reef is likely to be low and fishing mortality is now probably negligible. However, for the reasons outlined above, recruitment may not exceed natural mortality of many species, which means they may gradually decline further over time until they become locally extinct. This is likely the case with *Holothuria lessoni*, and *Holothuria whitmaei* may be also succumb over time.

Ceccarelli *et al.* (2011a) suggested that there may be 12 species of holothurians at Ashmore Reef that are below the minimum population size required to be sustainable; *Actinopyga lecanora*, *A. mauritiana*, *A. miliaris*, *Bohadschia argus*, *B. vitiensis* = *marmorata*, *Holothuria whitmaei*, *H. fuscogilva*, *H. fuscopunctata*, *H. fuscocubra*, *Stichopus herrmanni*, *Thelenota ananas* and *Thelenota anax*. We agree with this list based on our analyses. These species had maximum densities of 0 – 0.02 per 100 m<sup>2</sup> in the habitat in which they were most abundant in the 2019 survey with the exception of *H. fuscogilva* (0.05 per 100 m<sup>2</sup>) and *S. herrmanni* (0.2 per 100 m<sup>2</sup>).

Two possible causes for optimism exist however, at least for some species. Firstly, there is a potential for some species to associate or aggregate, or at least thrive in some microhabitats resulting in higher local abundances. Such higher local densities may enhance fertilisation rates during spawning. There were two examples of this on our voyage. The first was that although we encountered only two *H. whitmaei* after surveying 224 sites, both animals were at the same site within metres of each other. The second was that we counted no *Thelenota ananas* on any of our 224 sites, yet 5 were observed on one two-minute manta tow. These patchy distributions and aggregation in certain microhabitats can favour enhanced fertilisation rates. The second cause for optimism is that although we did not survey the deep-water habitats down to 20 m or more in the deep western lagoon or the back reef slope, we did observe some high value species at the deep bases of lagoon bommies during underwater visual census of fishes (chapter 10). The possibility that these deep-water habitats harbour high densities of some of the depleted species such as *Thelenota ananas*, *T. anax* and *Holothuria fuscogilva* is supported by the results of the 2000 and 2003 surveys (Smith *et al.* 2001; Rees *et al.* 2003) which were the only surveys to have attempted counts in these deep-water habitats, and where they did find higher densities of these species.

Perhaps the highest priority for Ashmore Reef is to carry out a survey of the deep-water habitats to determine the population status of species that inhabit these areas. It is important that these surveys be designed and carried out consistent with the methodology established by Hosack and Lawrence (2013a) to ensure results are comparable with the results of surveys in other habitat types made in 2019.

### Trochus

The 2019 survey of trochus at Ashmore Reef indicated abundances lower (0.45/ 100 m<sup>2</sup> on southern edge) than those recorded in 2013 (ca. 2.5/ 100 m<sup>2</sup>), but similar to surveys in 2005 and 2006 (ca. 0.5/ 100 m<sup>2</sup>). The overall through time suggests that trochus numbers have increased since the 1998 surveys of Skewes *et al.* (1999a). There are two main issues with the assessment of trochus at Ashmore Reef. One is their apparent high degree of mobility and tendency to aggregate (Rees *et al.* 2003; Ceccarelli *et al.* 2011a), meaning they will be very patchily distributed and hence difficult to survey accurately. The second issue is that they occur in a very specific habitat (exposed, surge-affected shallow reef crest and upper slope), which may be under-represented in multipurpose surveys such as most of those made at Ashmore Reef, and weather conditions at the time of the survey will affect how accessible this habitat is. This means surveys need to be comprehensive with a larger number of sites on the reef edge considered for the next survey.

Trochus appear to be biologically well-adapted to recover from overfishing. They have been shown to maintain populations with low levels of connectivity to other stocks (Berry *et al.* 2019). Trochus spawn at just two years of age (Heslinga & Hillmann 1981), and larvae have a short planktonic phase, settling after just three days close to the parent stock. A tendency to aggregate should also help facilitate high fertilisation rates, and high levels of mobility mean they can respond to changed microhabitat conditions quickly.

### Clams

The giant clam *Tridacna gigas* had not been recorded at Ashmore Reef since 2006 (Ceccarelli *et al.* 2007) and were absent in the 2009 and 2013 surveys. Based on known growth rates (Munro 1993) the largest (ca. 80 cm) individuals we observed in 2019 would have been for at least 15 years old, so their reappearance is not sudden. Abundance of giant clams remains low (<0.05/ 100 m<sup>2</sup>) and recovery will be slow if at all.

Our analysis suggests that density of the small, non-harvested species *Tridacna crocea* has remained the same or increased, and that *Tridacna maxima* and *T. squamosa* (combined) had increased between 1998 and 2013, with densities declining again in 2019 (Hosack *et al.* chapter 12 this study).

In contrast, the density of *Hippopus hippopus* remains much lower now (0.37/ 100 m<sup>2</sup>) than in 1998, although it is higher now than in 2006 when Ceccarelli *et al.* (2007) found low abundances and that dead clams were equally as abundant as live ones, indicating illegal fishing (or some other agent of mortality) had taken place not long before their survey. The higher densities of *Tridacna maxima* and *T. squamosa* (combined) in 2013 and 2019 compared with those in 2005 and 2006, suggests that these species were not subject to the same heavy fishing as *H. hippopus*, and have therefore gradually increased in abundance over time. This is probably particularly the case for *T. maxima*, which was twice as abundant as *T. squamosa* in 2019.

Monitoring of clams at Ashmore Reef should be continued at least every five years. Determining the status and trend of population size of *Tridacna gigas* and *Hippopus hippopus* will be particularly important.

## 11.6 Management implications and recommendations

Results indicate that the abundance of at least 12 holothurians and two clam species at Ashmore Reef is very low and do not appear to have recovered from overfishing in the mid to late 1980s. Some species appear locally extinct, and others have reproductively ineffective populations. We have not considered the feasibility or desirability of restorative measures which would require significant logistical effort and/or radical intervention such as density manipulations within the reef during spawning time, translocation of adults or reseedling of larvae and/or juveniles. A feasibility study for this could be considered.

The importance of deep-water habitat for holothurian species that have been historically overfished should be determined. This is regarded as the highest priority research involving holothurians at Ashmore Reef. These habitats have not been subject to any dedicated surveys since 2003. It is important that these surveys be designed and carried out using the methodology established by Hosack and Lawrence (2013a), to ensure results are comparable with the results of surveys in other habitat types made in 2019.

The very high mobility of trochus and their tendency to aggregate in high numbers means surveys need to be continually comprehensive to be comparable and a larger number of sites on the reef edge should be considered for the next survey.

In the absence of disturbances such as coral bleaching events, the surveys we conducted should be repeated at least every five years to monitor for any changes in the abundance and species composition of invertebrate populations at Ashmore Reef. However, there is also a need for the “reactive” monitoring following large scale perturbations, suggested by Ceccarelli et al. (2007), to ensure their effect is more readily understood and quantified.

Reference sites at other reefs, such as those least impacted by fishing for example the Rowley Shoals, and those within the the MOU74 box where fishing occurs such as Scott Reef, should be surveyed using the same methods to ensure any changes detected can be placed in the context of wider environmental change beyond fishing impacts

# 12 ASHMORE REEF: ASHMORE REEF DESIGN AND ANALYSIS FOR BENTHIC INVERTEBRATES

Geoffrey R. Hosack, John Keesing and Margaret Miller

## 12.1 Abstract

Swim surveys targeting a defined suite of benthic invertebrate taxa were deployed at Ashmore Reef in 2013 and 2019. To increase the ability to detect temporal change and maximise sampling efficiency, the two surveys followed the same sample design by returning to the same sample set of reference sites. Both the surveys also used the same field methods. Benthic invertebrate taxa were counted along a 2 metre wide transect that extended for 50 metres. The sample design ensured broad spatial coverage and also representation of the major habitat types such as lagoon, reef flat, reef edge, intertidal and subtidal sand. A model-based approach accounted for habitat and spatial dependence while enabling comparisons of abundance across Ashmore Reef through both space and time. Very different patterns of spatial patchiness and habitat use were apparent for the investigated taxa. Although some taxa abundances appeared to increase between 2013 and 2019, there were also noticeable decreases as estimated for clams, trochus (*Rochia niloticus*) and *Thelenota ananas*. Possible causal hypotheses for declines include direct and indirect impacts from cyclones and heat stress. Other factors may include Allee effects that reduce the ability for populations to recover after a period of high exploitation, as might have occurred for *T. ananas*, or the apparent declines may arise from observation error due to very high spatial aggregation of some taxa, such as has been observed for *R. niloticus*. Similar surveys conducted at comparison reefs would be required to assess the probability that temporal change in abundance is attributable to a particular set of causal factors.

## 12.2 Introduction

Unsustainable levels of harvesting of reef top resources of holothurians, trochus and clams at Ashmore Reef in the 1980s led to stricter regulations on access and fishing (see Chapter 11 for detailed description and discussion of this). A series of surveys were later conducted to assess the population status of these resources and determine if they were showing signs of recovering from fishing. These surveys were hampered somewhat by episodes of illegal fishing (Ceccarelli *et al.* 2007) and survey design features that made abundance comparisons between surveys and trend analyses over time problematic (Breen 2011; Hosack & Lawrence 2013a, b). As a result, Hosack and Lawrence (2013a, b) undertook a detailed design and analysis study which culminated in the currently adopted Generalized Random Tessellation Stratified (GRTS) based design for invertebrate surveys at Ashmore Reef with sampling effort optimised for the known distribution of habitats and target species.

The implementation of the GRTS design permits use of both design-based (Stevens & Olsen 2004) and model-based analyses. Design based estimators must account for spatial autocorrelation that arise from shared environmental factors among neighbouring sites while the model-based approach implemented in this analysis and by Hosack and Lawrence (2013a) flexibly

accommodates spatial dependence among sites using Gaussian processes that are estimated from the observed data. The model-based approach to total abundance estimation also considers habitat information and spatial dependence that would not be addressed by the standard GRTS design-based estimator. The model-based approach thereby permits prediction of areal densities and observations to unsampled locations, which would not be possible with the design-based approach.

Here we apply these analyses to the data from the 2019 Ashmore Reef swim surveys of benthic invertebrates. This survey followed the same study design as the 2013 survey Ceccarelli *et al.* (2013). Both the 2013 and 2019 surveys used the spatially balanced design and field protocols documented in Hosack and Lawrence (2013a). The common field protocols and study design between the two surveys enabled a direct comparison of changes in abundance over the 5+ year interval between surveys and these are presented and discussed.

## 12.3 Methods

### 12.3.1 Study design

As part of the 2019 Ashmore Reef Marine Park Environmental Assessment, CSIRO implemented a study design for the estimation of benthic invertebrate abundance. The study design followed the methods and protocols documented by (Hosack & Lawrence 2013a), which used a spatially balanced stratified GRTS design (Stevens & Olsen 2004) for Ashmore Reef. The study design was first implemented in 2013 by (Ceccarelli *et al.* 2013) using field protocols derived through a consultation process with previous field researchers at Ashmore Reef as documented by (Hosack & Lawrence 2013a). A Bayesian hierarchical model was used to evaluate these 2013 data in relation to previous field sampling exercises that were conducted with heterogeneous sampling effort and study goals (Hosack & Lawrence 2013b). This analysis suggested an extension of the sampling transects from 40 m x 2 m transects to 50 m x 2 m transects. The 2019 field sampling survey followed the original study design proposed by Hosack and Lawrence (2013a) using the longer 100 m<sup>2</sup> transects. Whereas cross-year comparisons are conflated with observation error and changing field techniques for Ashmore Reef surveys prior to 2013 (Hosack & Lawrence 2013b), the carefully implemented study design by the field teams in both 2013 and 2019 permits a direct temporal comparison of abundance estimates between these two years. The common sample design means that the sample sites may be considered reference sites, or legacy sites. Returning to the same sample design each sampling year increases the ability to detect trends and temporal changes in benthic invertebrate abundance at Ashmore Reef.

### 12.3.2 Model

The analysis accounted for discrete observations, temporal abundance shifts, variable length transects, habitat variability and spatial dependence induced by unobserved covariates through the following Bayesian hierarchical model fit independently for each taxonomic unit,

$$\begin{aligned}
 y(s, t) &\sim \text{Poisson}(\lambda(s, t)o(s, t)) \\
 \log(\lambda(s, t)) &= \mu(s, t) + \omega(s) \\
 \mu(s, t) &= X(s, t)\beta \\
 \beta &\sim N(0, V) \\
 \omega(s) &= c(s, s^*, \rho, \sigma^2)^\top C(s^*, \rho, \sigma^2)^{-1}w(s^*) \\
 w(s^*) &\sim N(0, C(s^*, \rho, \sigma^2)) \\
 \rho &\sim U(a_\rho, b_\rho) \\
 \sigma^2 &\sim \text{InverseGamma}(a_\sigma, b_\sigma).
 \end{aligned}$$

The observed abundance  $y(s, t)$  of a benthic invertebrate taxon at location  $s$  and time  $t$  follows a Poisson distribution with spatially and temporally varying intensity  $\lambda(s, t)$ . The offset  $o(s, t)$ , which is also allowed to vary through space and time, accounts for variable sampling efforts caused by changing lengths of transects. The mean of the log intensity at location  $s$  and time  $t$  is given by  $x_i^\top(s, t)\beta$ , where the row vector  $x_i^\top(s, t)$  is a row of the  $n \times p$  design matrix  $X(s, t)$  and  $\beta$  is the  $p \times 1$  vector of unknown coefficients. The spatial random effects  $\omega(s)$  follow a predictive process model (Banerjee *et al.* 2008), where  $w(s^*)$  are latent random effects described further below.

The spatially and temporally varying covariates in the design matrix  $X(s, t)$  include an intercept term and binary indicator variables for year of sampling (2013 or 2019) and the habitat associated with location  $s$ . The habitat assignment was based on the habitat map developed by Skewes *et al.* (1999b) and projected to GDA94 / MGA zone 51S. The habitat categorisation followed (Hosack & Lawrence 2013a) exterior reef edge, reef flat, deep reef flat, lagoon and sand. Independent mean zero normal priors with variance 10 were placed on the coefficients  $\beta_i$ ,  $i = 1, \dots, p$ .

The 2013 and 2019 field teams implemented the GRTS survey design that specified as set of coordinates  $S^*$  with a given survey location  $s^* \in S^*$  (Hosack & Lawrence 2013a). A total of  $n = 313$  spatial locations that targeted  $m = 219$  of the GRTS survey locations were obtained from the 2013 and 2019 field campaigns. To account for un-observed spatially varying covariates, these GRTS locations were assigned spatial random effects described by a mean-zero Gaussian process with exponential spatial covariance function,  $C(s^*, \rho, \sigma^2) = \sigma^2 \exp(-\rho D)$ , where  $D$  is the  $m \times m$  Euclidean distance matrix constructed from the  $m$  sampled GRTS survey locations  $s^*$ .

In a given year, the actual sample location may vary depending on field conditions and orientation of transect. Previous field teams have noted that field conditions and vessel manoeuvring constrain the ability to precisely sample the targeted location. The actual  $n$  sample locations  $s$  were therefore assigned spatial random effects  $\omega(s)$ , a  $n \times 1$  vector, that were linear functions of the  $m \times 1$  latent spatial random effects vector  $w(s)$  for the underlying GRTS locations. The  $n \times 1$  vector  $c(s, s^*, \rho, \sigma^2)$  contained the spatial covariances between the  $n$  sampled location and the  $m$  sampled GRTS survey locations. If a sample location  $s_0$  was identical to its associated GRTS survey location  $s_0^*$  then the above linear function between  $\omega(s)$  and  $w(s^*)$  means that the spatial predictive process interpolates deterministically such that  $\omega(s_0) = w(s_0^*)$ . For the analysis, the

sampled GRTS survey location coordinates were defined by the average of the reported sampling locations. Given that there are only two sampling years using the common survey design and field method protocols, a spatio-temporal covariance function was not considered but the collection of future surveys would allow for this elaboration.

The spatial covariance prior is completed with the specification of priors for the correlation parameter  $\rho$  and the spatial variance  $\sigma^2$ . An inverse gamma prior with shape parameter  $a_\sigma = 2$  and scale parameter  $b_\sigma = 1$  was placed on the spatial variance. The correlation parameter defines the effective spatial range, which is the distance between two locations where the spatial correlation drops to 0.05 and is approximately defined by  $\rho/3$ . The uniform prior on  $\rho$  was bounded below by  $a_\rho = 2.4 \times 10^{-4}$  and above by  $b_\rho = 0.03$ . The prior constrained the effective spatial range between 100 and 12,461 metres, where the upper bound is half the maximum distance between sampled locations across both years.

### 12.3.3 Prediction

The estimated areal densities for Ashmore Reef are derived from the predictive posterior distribution of the spatio-temporal intensity,  $p(\lambda(s, t)|y)$ . The predictive posterior was estimated for the year 2019 evaluated at the centroids of grid cells, where cell sizes were equal to 4 hectares. The areal densities were calculated as number of individuals per 100 m<sup>2</sup>, which was the amount of area covered by the field transects in 2019. The estimated abundance in years 2013 and 2019 were obtained from the predictive posterior distribution of the observations for each year,  $p(y'(s, t = 2013), y'(s, t = 2019)|y(s, t = 2013), y(s, t = 2019))$ . The predictive posterior distribution considers the joint dependencies among years and grid cells for the predicted observations. The predicted abundances were scaled up by the size of the grid cells and multiplied by the number of grid cells within the Ashmore Reef sample frame to obtain a total abundance in 2013 and 2019.

### 12.3.4 Estimation

Estimation used an adaptive Metropolis-within-Gibbs sampler (Roberts & Rosenthal 2009) implemented in R package `spBayes` (Finley, Banerjee & Gelfand 2015). The unknown coefficients  $\beta$  and spatial random effects  $w(s^*)$  were initialised at zero. The initial Metropolis proposal variances were set to small values (less than 0.1) and the algorithm was run for 100,000 iterations with a target acceptance rate of 0.44. Draws from the predictive posterior distributions of the spatio-temporal intensities and observations require an  $m \times m$  matrix inversion at each iteration to obtain jointly distributed samples and so posterior realisations were subsampled. The first 40,000 iterations were considered burn in and the remaining 60,000 samples were subsampled with a thinning rate of 40 from the post burn-in samples. The Geweke diagnostic was used to evaluate convergence from the post burn-in phase. The Geweke diagnostic comparing the first 33% to the last 33% of the post burn-in phase resulted in an average number of z-scores outside  $\pm 2$  of 4.8% when applied to all unknown static parameters and latent spatial random effects. Spatial predictions therefore were for each taxon based on 1,500 sets of posterior samples from the posterior distribution  $p(\beta|y)$  and the predictive posterior distributions  $p(\lambda|y)$  and  $p(y'|y)$  described in the previous section.

## 12.4 Results

### 12.4.1 Spatial estimates

In this section, the estimated areal densities in 2019 are plotted for each taxon. Two plots are presented for each taxon. The first plot shows the estimated posterior mean areal density (number per 100m<sup>2</sup>) and the second plot shows the posterior interquartile distance of the areal density, which is a measure of dispersion or uncertainty (similar to reporting a standard deviation, but more robust).

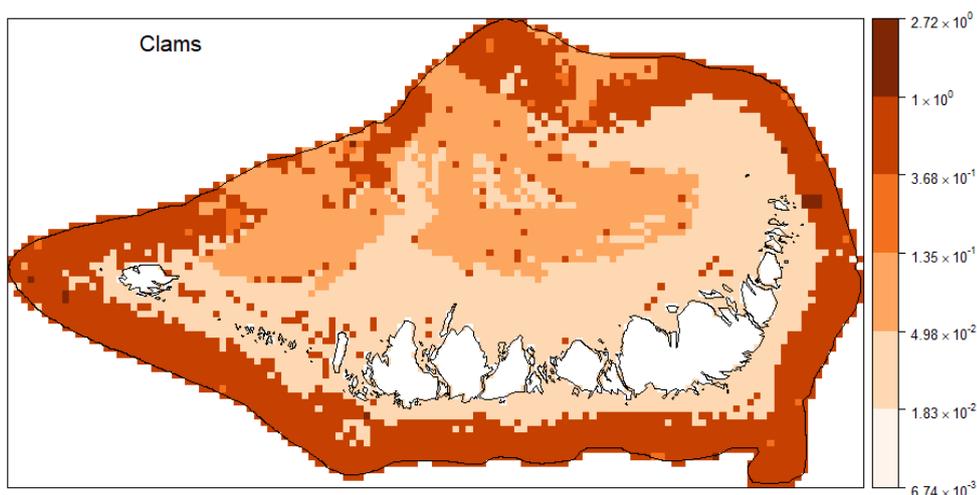


Figure 148. Posterior mean of areal density of all clam species (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

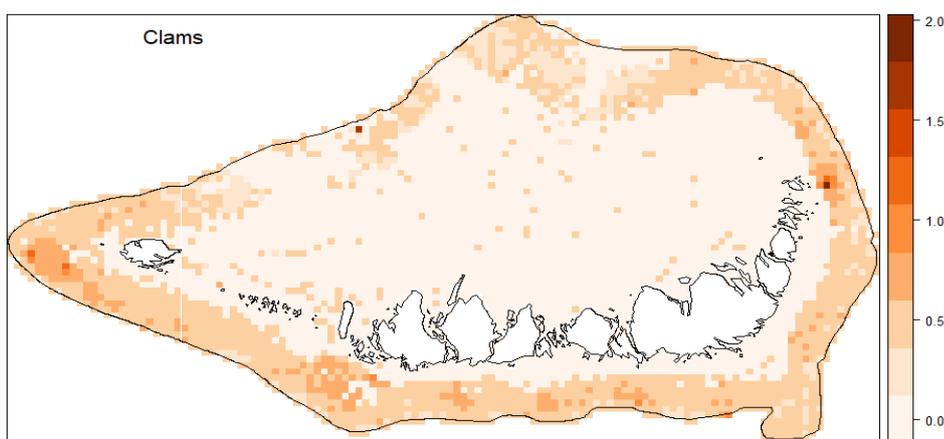


Figure 149. Posterior inter-quartile distance of areal density of all clam species (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

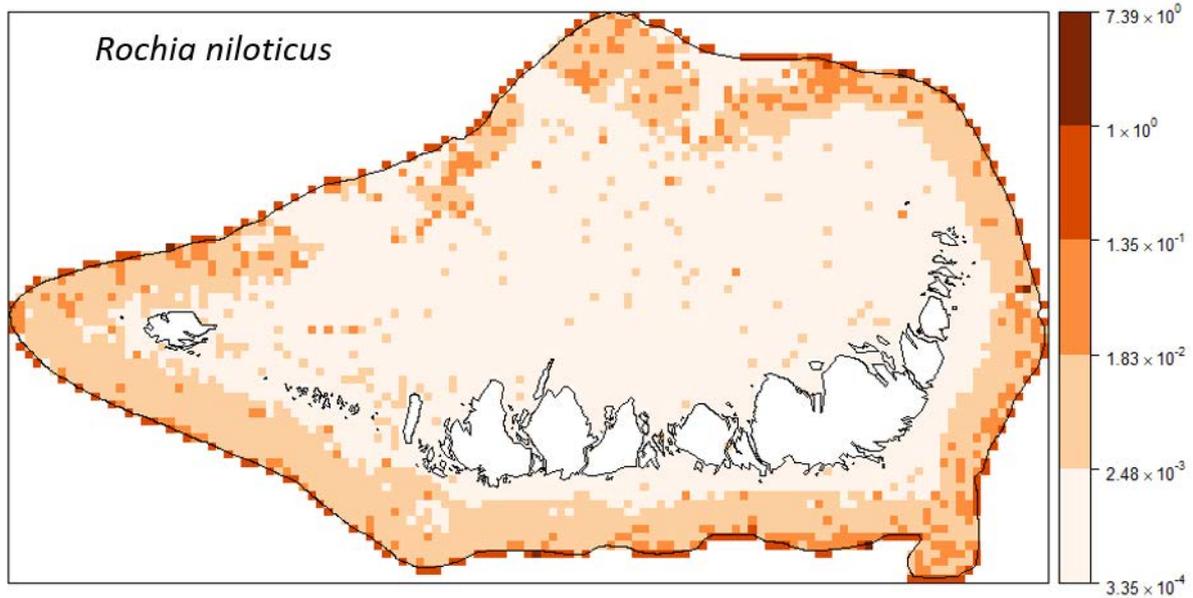


Figure 150. Posterior mean of areal density of trochus (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

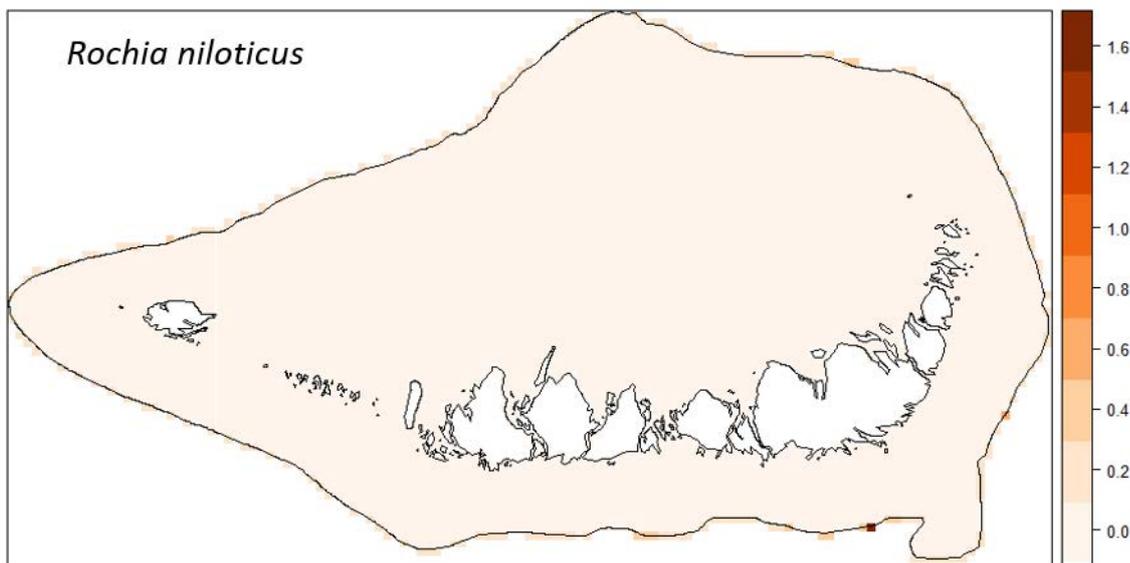


Figure 151. Posterior inter-quartile distance of areal density of trochus (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

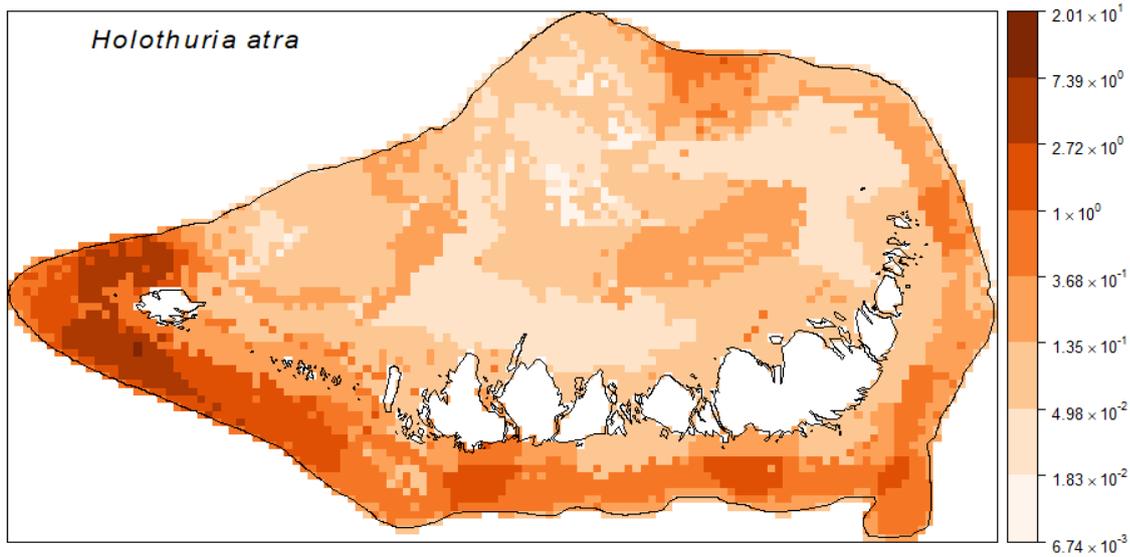


Figure 152. Posterior mean of areal density of *Holothuria atra* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

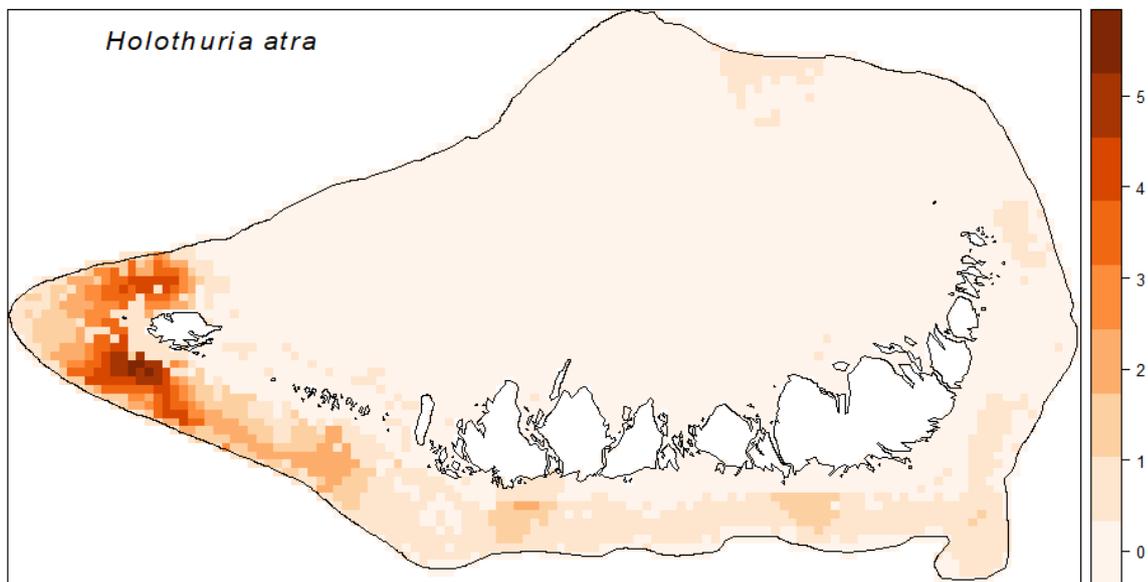


Figure 153. Posterior inter-quartile distance of areal density of *Holothuria atra* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

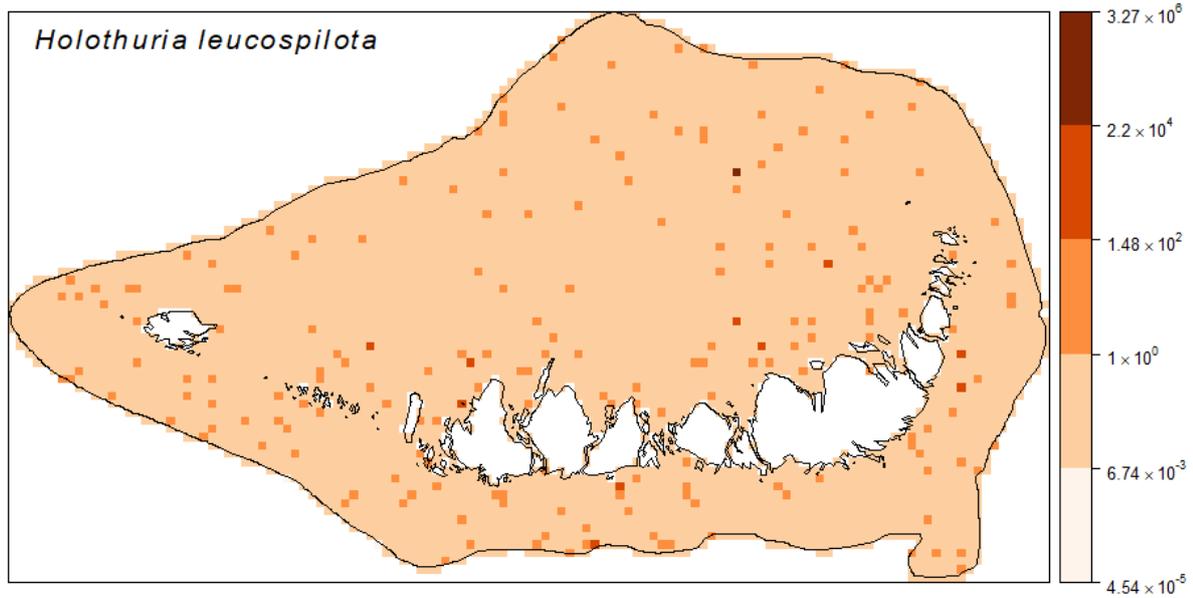


Figure 154. Posterior mean of areal density of *Holothuria leucospilota* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

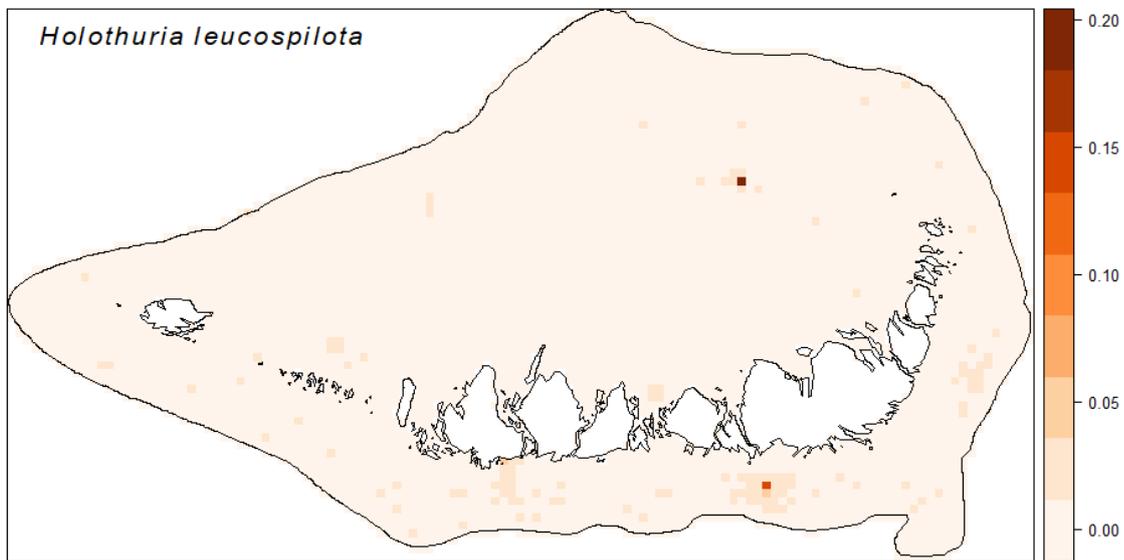


Figure 155. Posterior inter-quartile distance of areal density of *Holothuria leucospilota* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

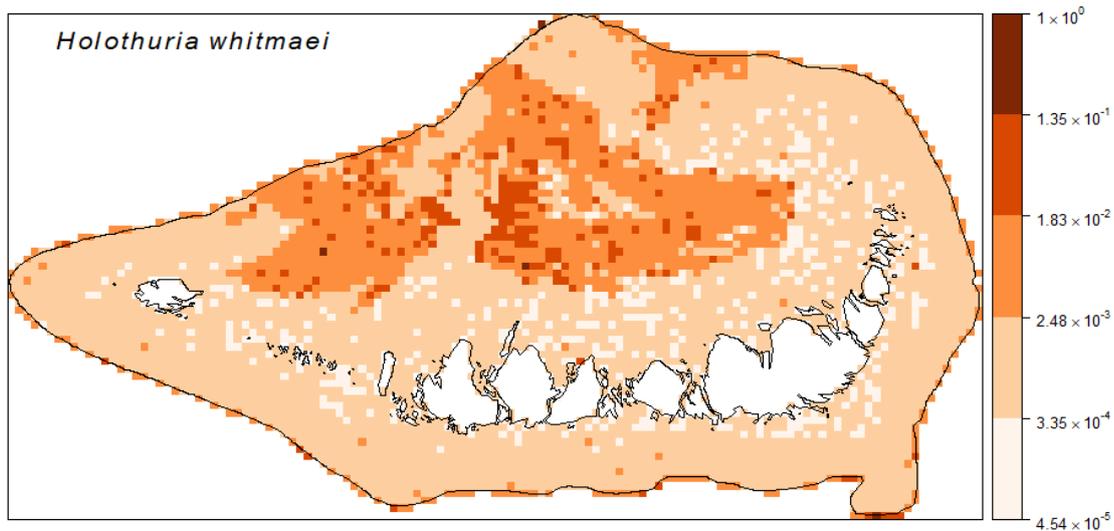


Figure 156. Posterior mean of areal density of *Holothuria whitmaei* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

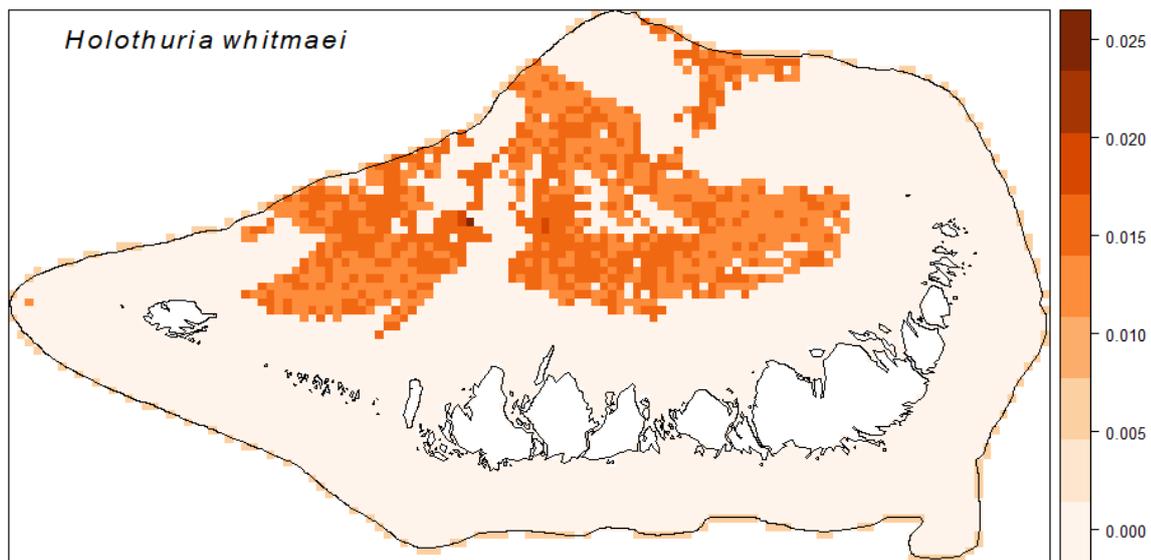


Figure 157. Posterior inter-quartile distance of areal density of *Holothuria whitmaei* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

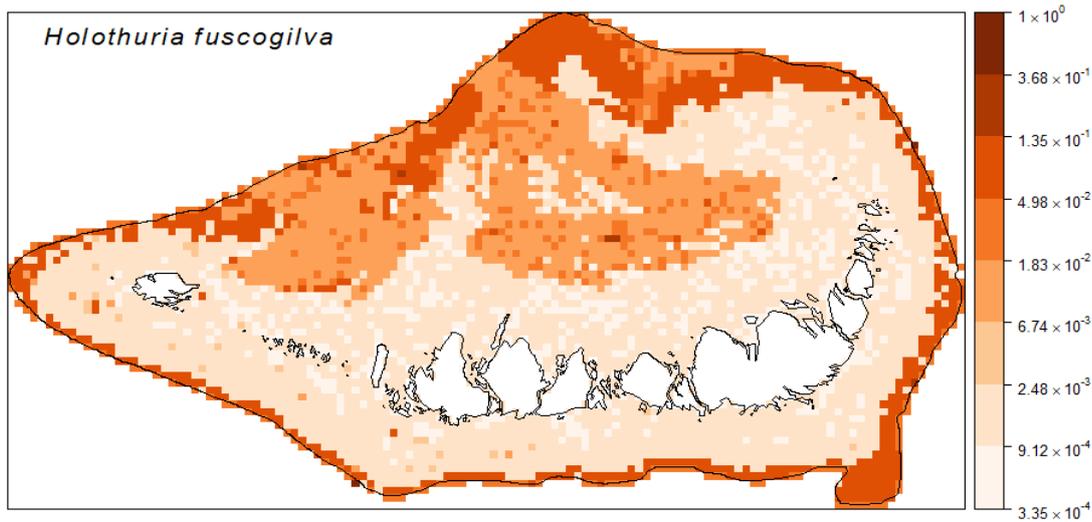


Figure 158. Posterior mean of areal density of *Holothuria fuscogilva* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

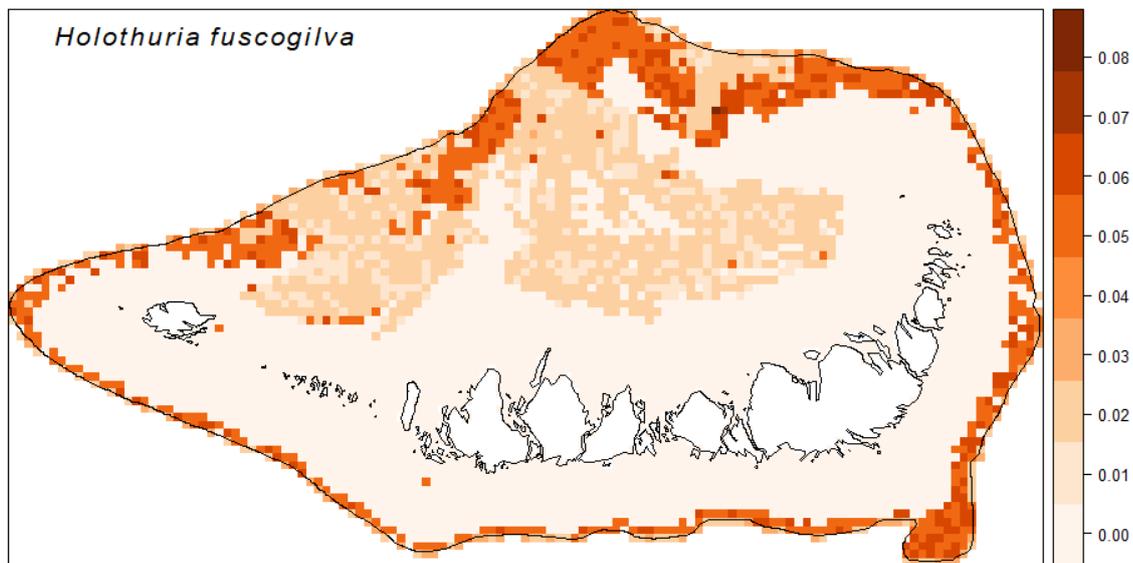


Figure 159. Posterior inter-quartile distance of areal density of *Holothuria fuscogilva* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

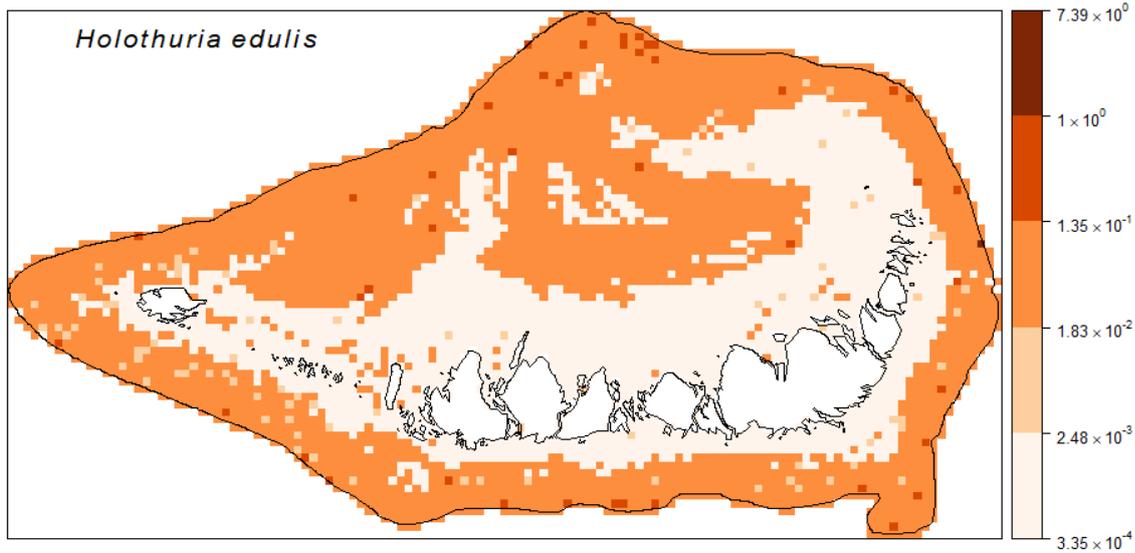


Figure 160. Posterior mean of areal density of *Holothuria edulis* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

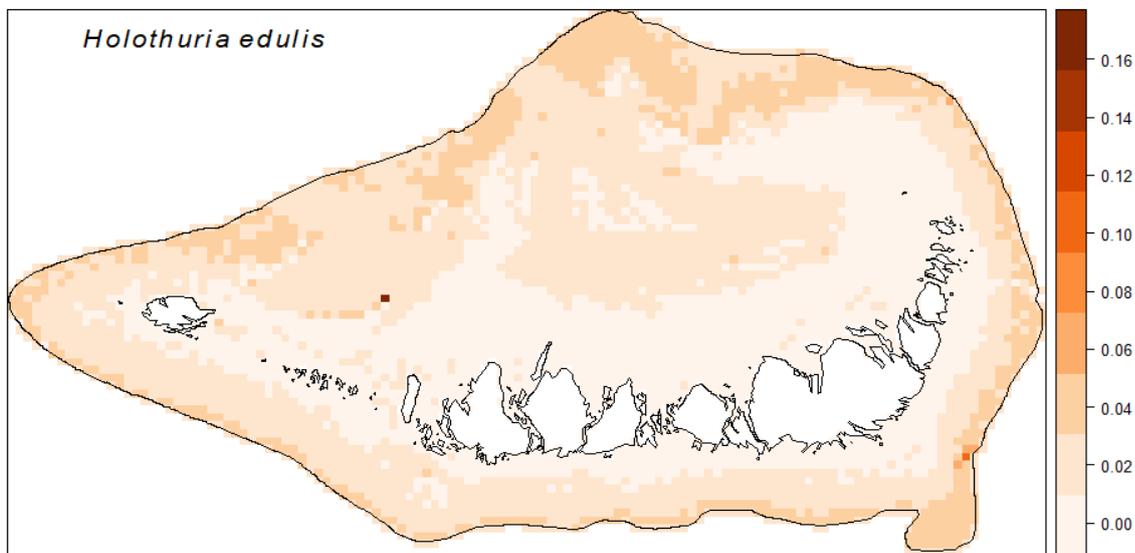


Figure 161. Posterior inter-quartile distance of areal density of *Holothuria edulis* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

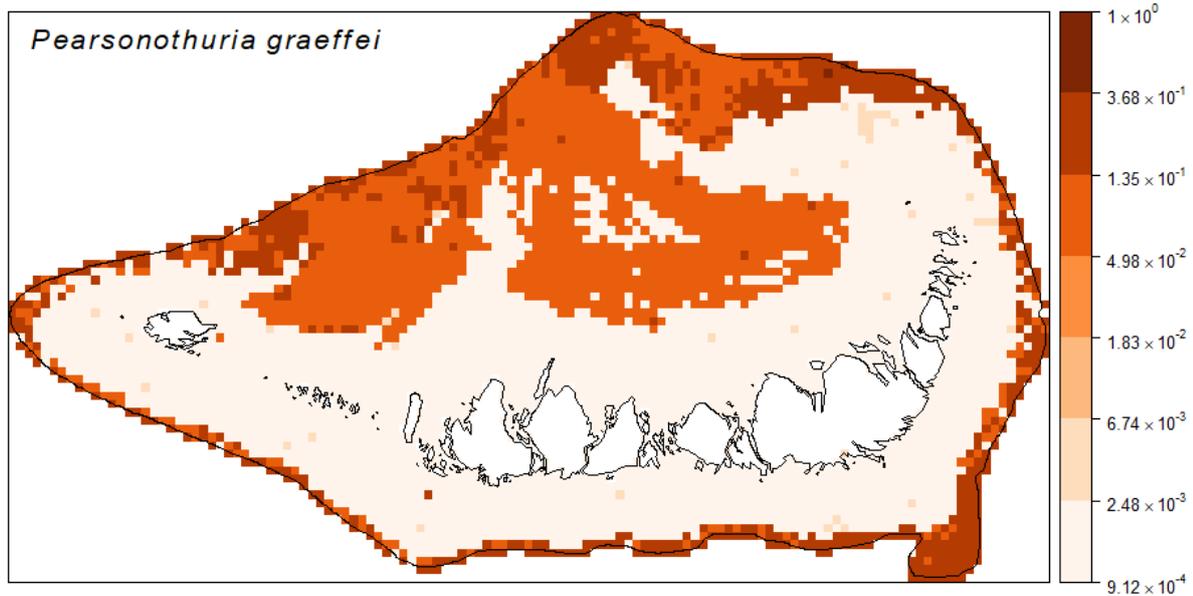


Figure 162. Posterior mean of areal density of *Pearsonothuria graeffei* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

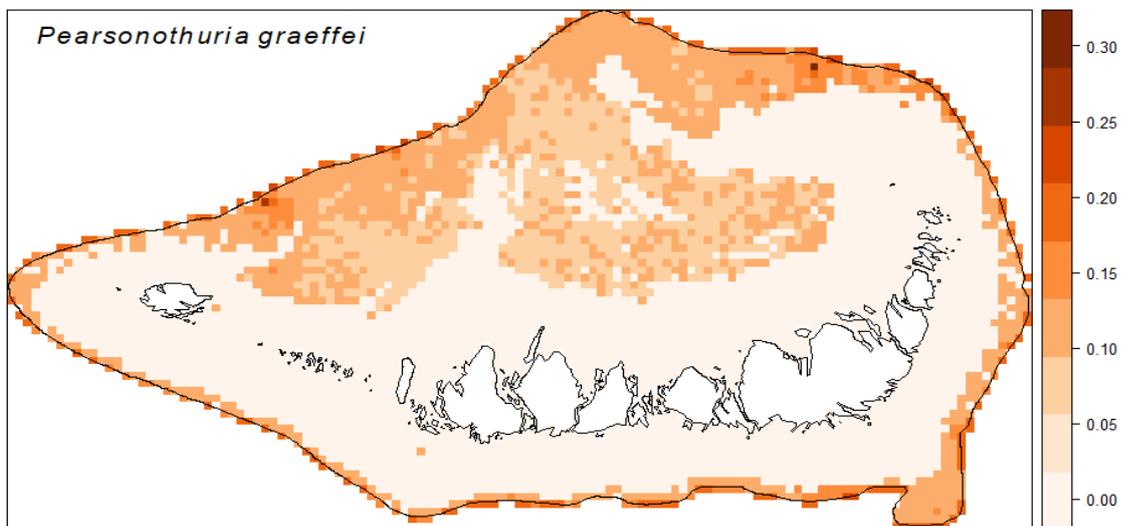


Figure 163. Posterior inter-quartile distance of areal density of *Pearsonothuria graeffei* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

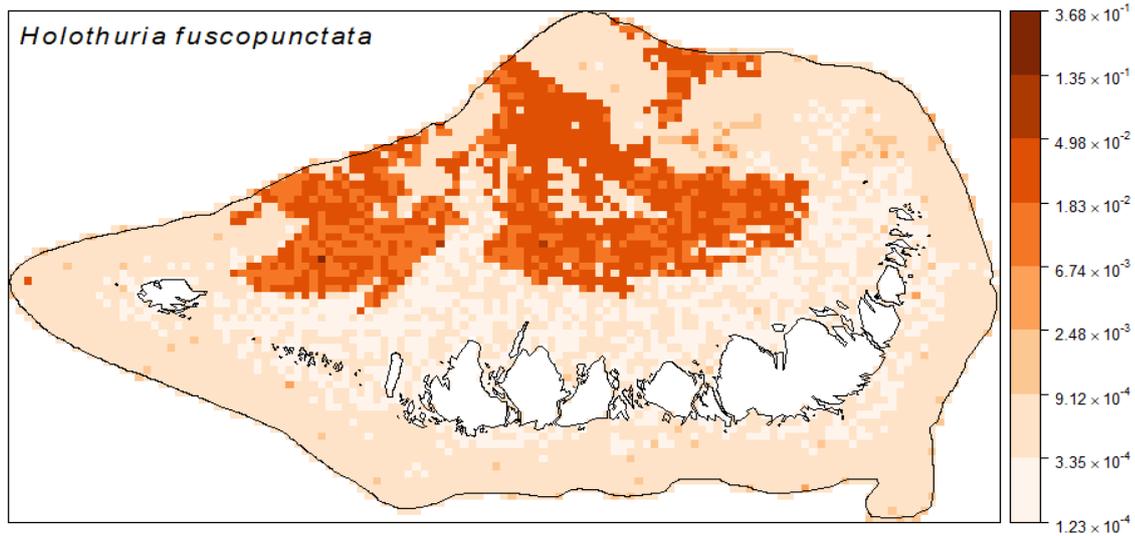


Figure 164. Posterior mean of areal density of *Holothuria fuscopunctata* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

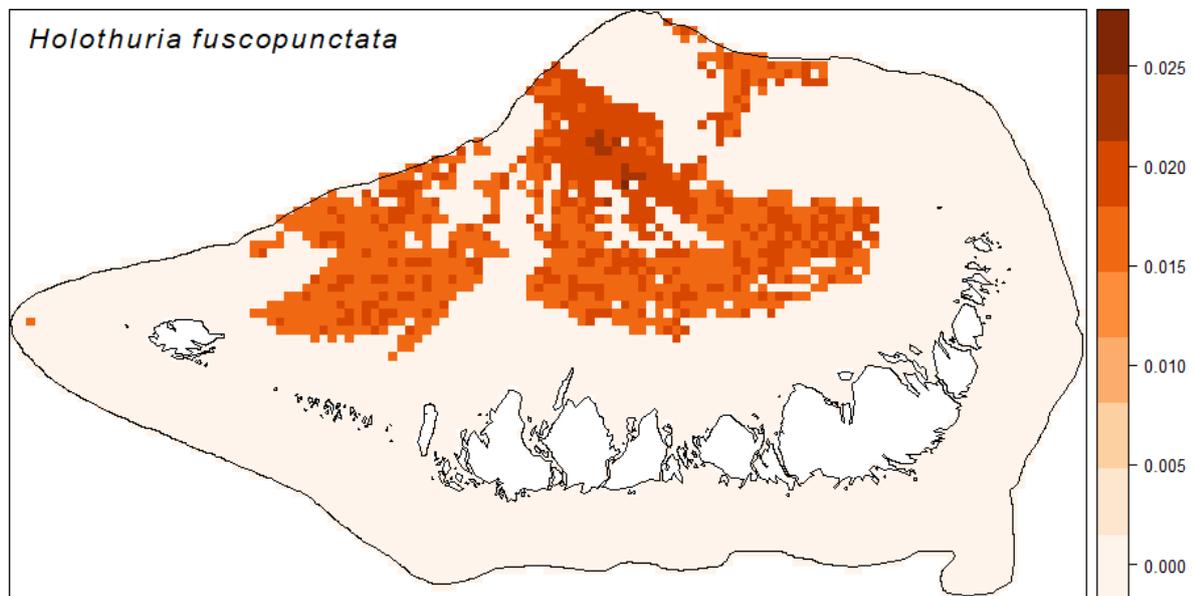


Figure 165. Posterior inter-quartile distance of areal density of *Holothuria fuscopunctata* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

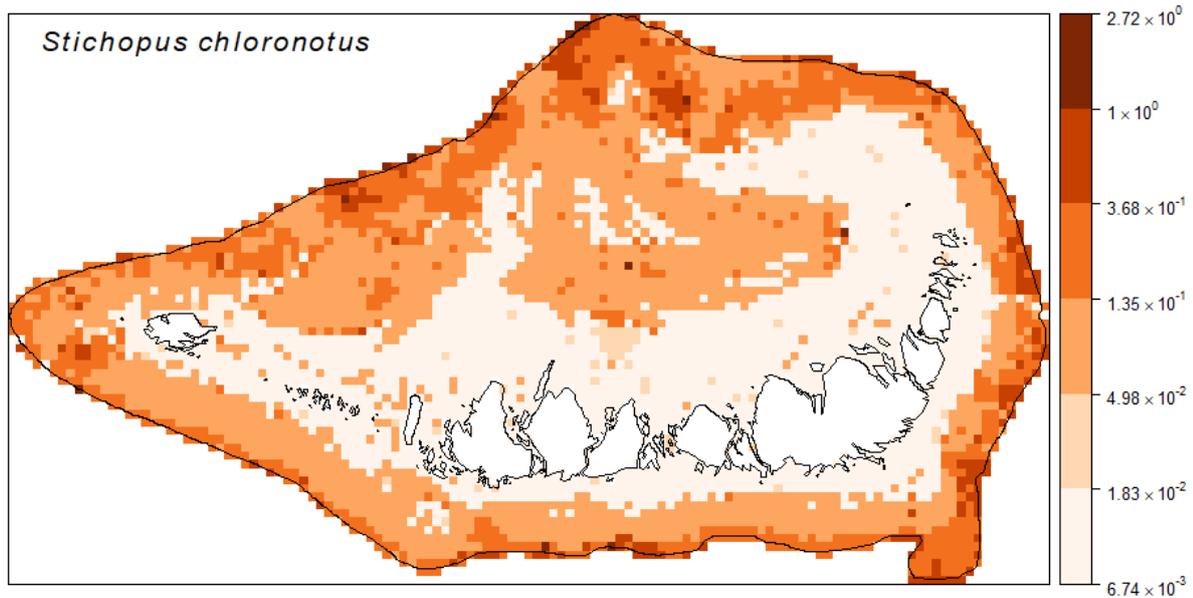


Figure 166. Posterior mean of areal density of *Stichopus chloronotus* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

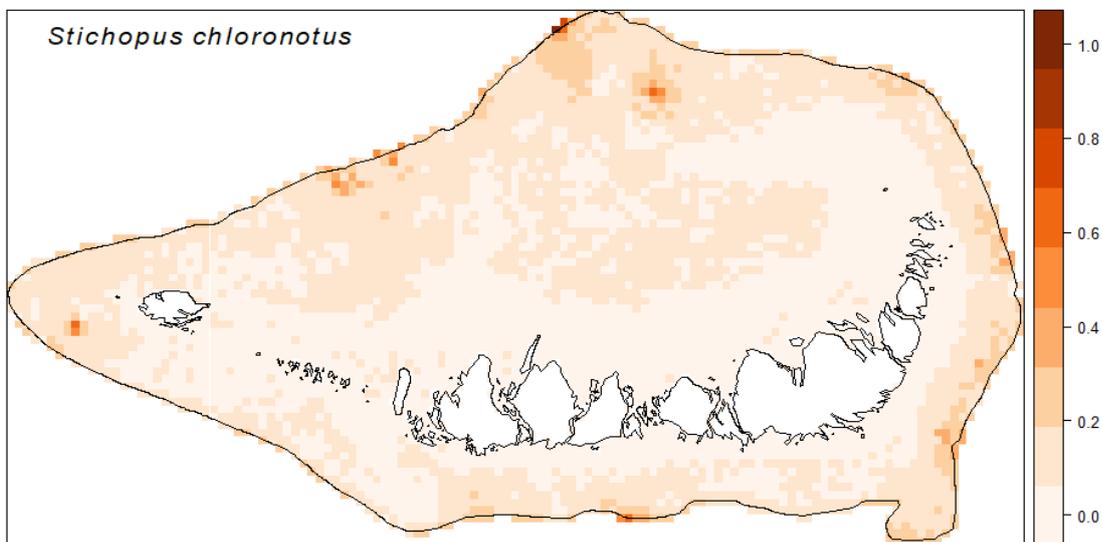


Figure 167. Posterior inter-quartile distance of areal density of *Stichopus chloronotus* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

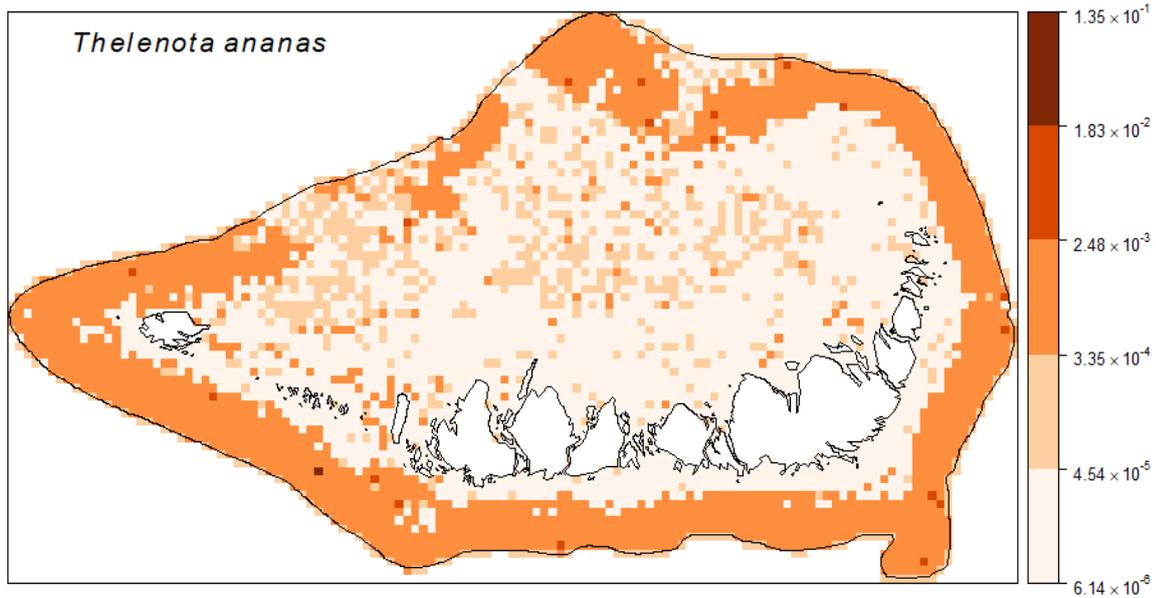


Figure 168. Posterior mean of areal density of *Thelenota ananas* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

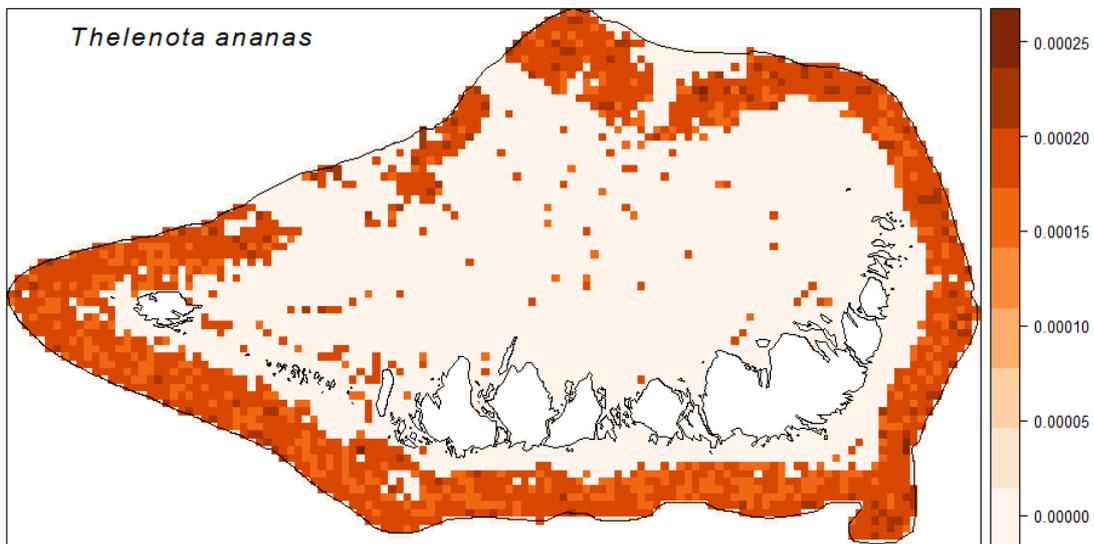


Figure 169. Posterior inter-quartile distance of areal density of *Thelenota ananas* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

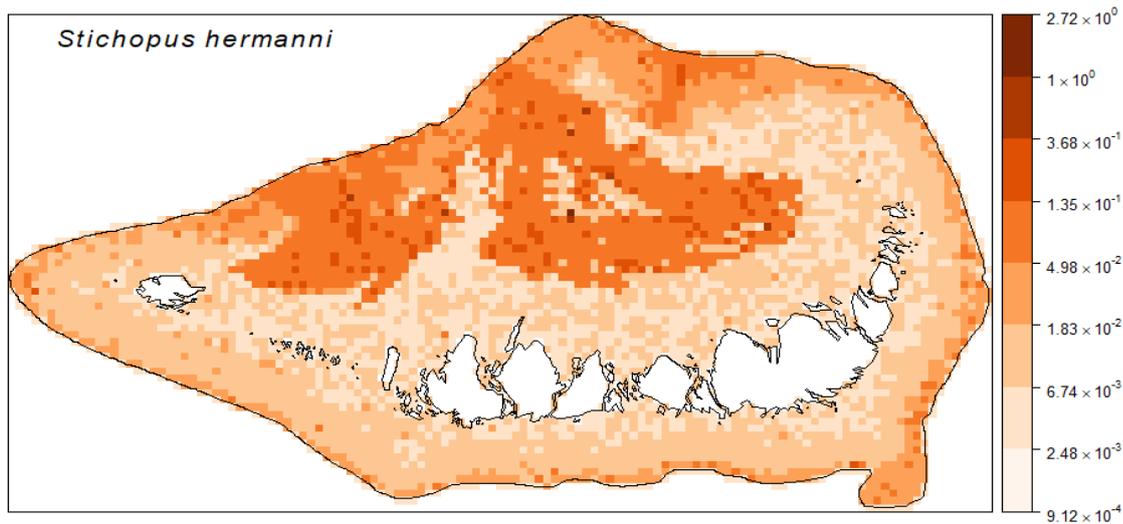


Figure 170. Posterior mean of areal density of *Stichopus hermanni* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

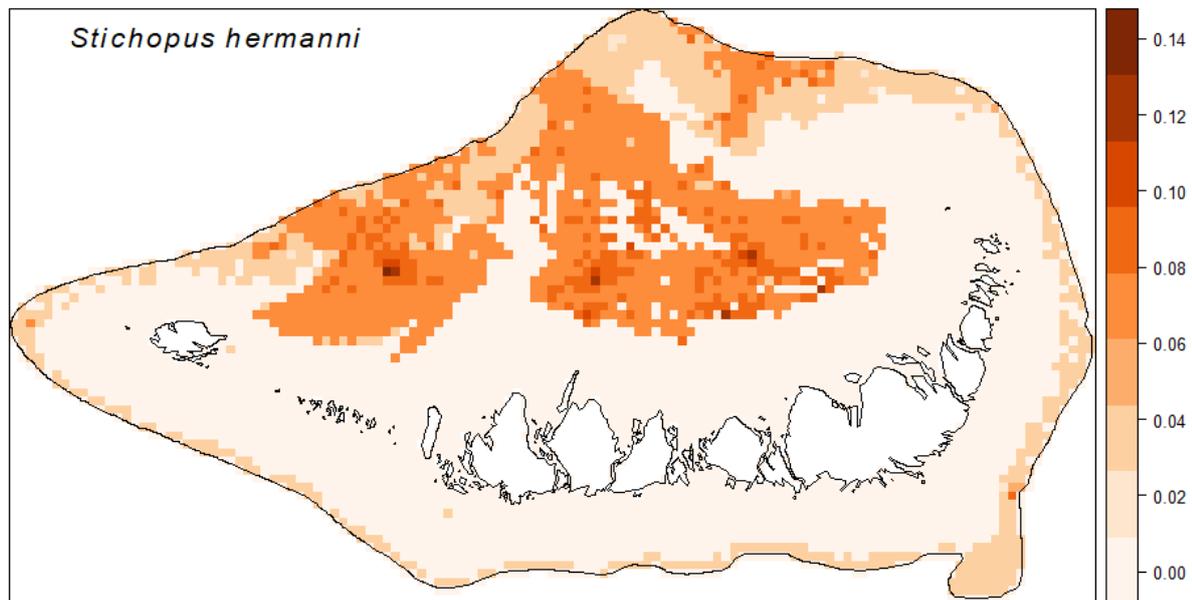


Figure 171. Posterior inter-quartile distance of areal density of *Stichopus hermanni* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

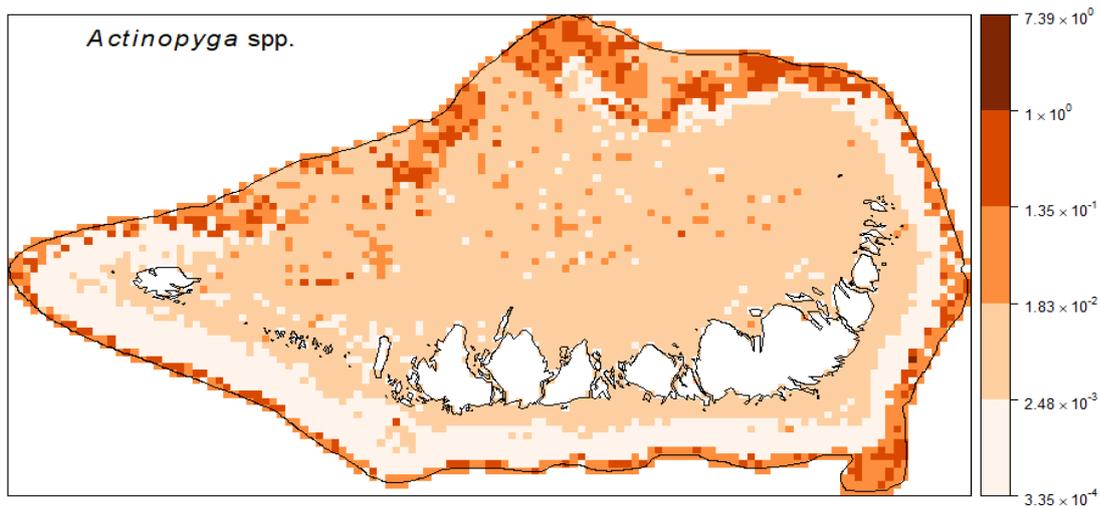


Figure 172. Posterior mean of areal density of all *Actinopyga* species (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

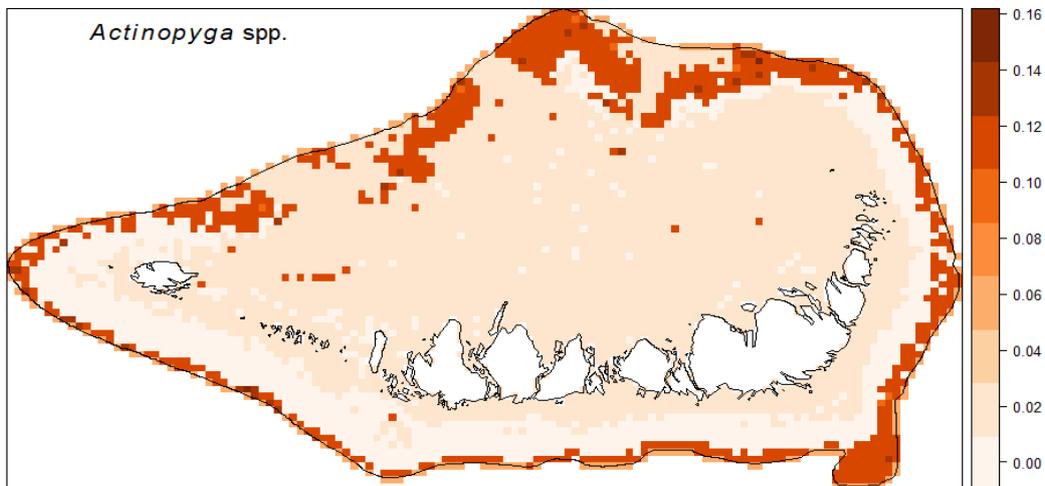


Figure 173. Posterior inter-quartile distance of areal density of all *Actinopyga* species (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

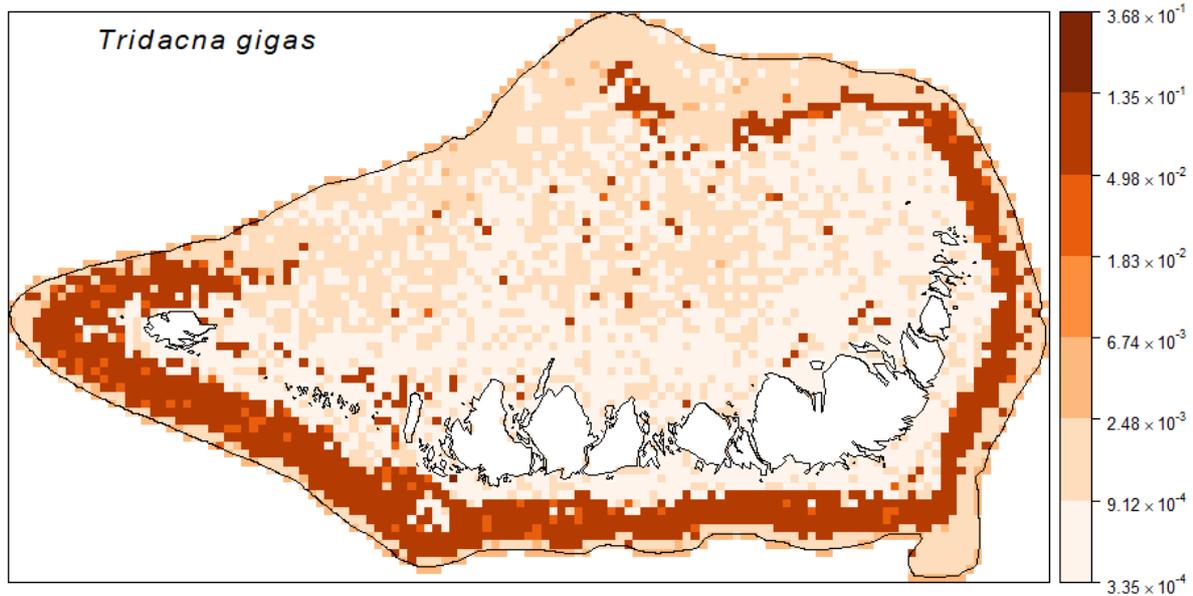


Figure 174. Posterior mean of areal density of *Tridacna gigas* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

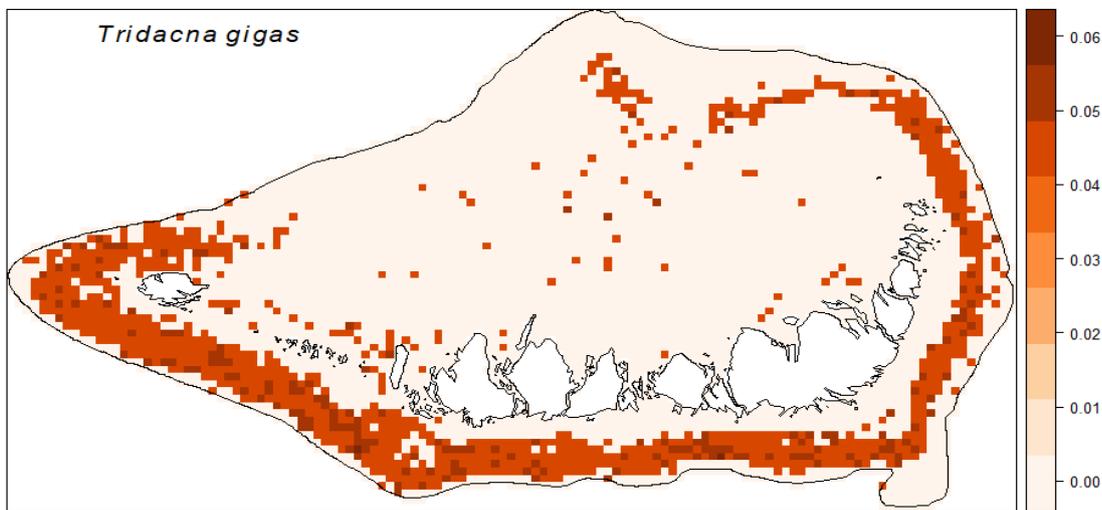


Figure 175. Posterior inter-quartile distance of areal density of *Tridacna gigas* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

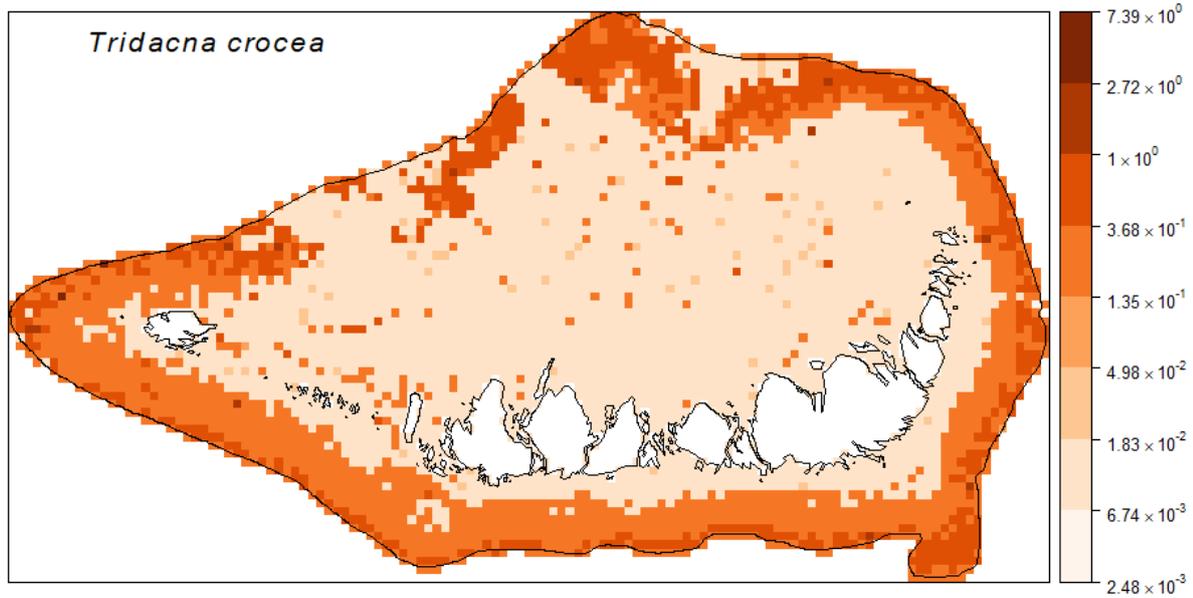


Figure 176. Posterior mean of areal density of *Tridacna crocea* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

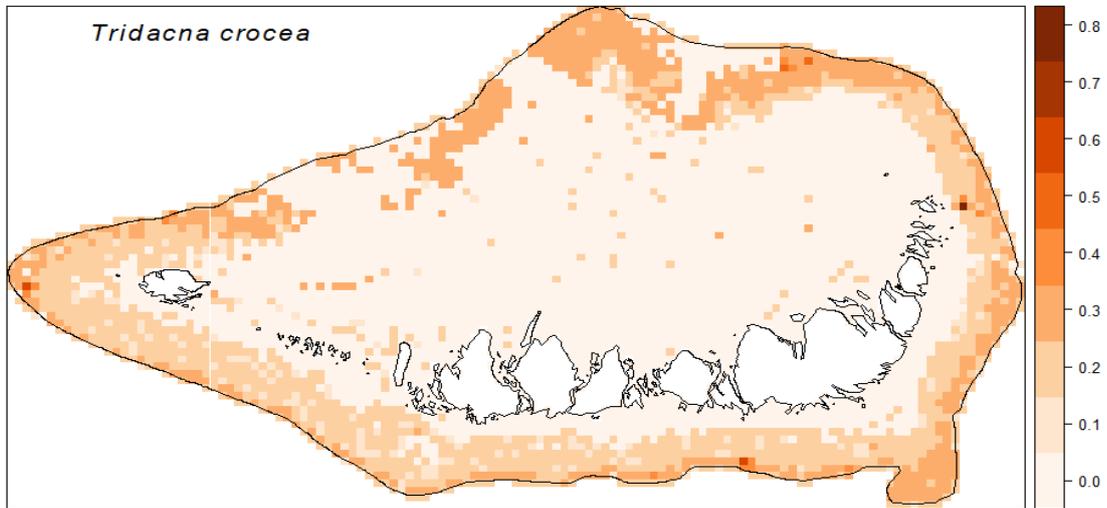


Figure 177. Posterior inter-quartile distance of areal density of *Tridacna crocea* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

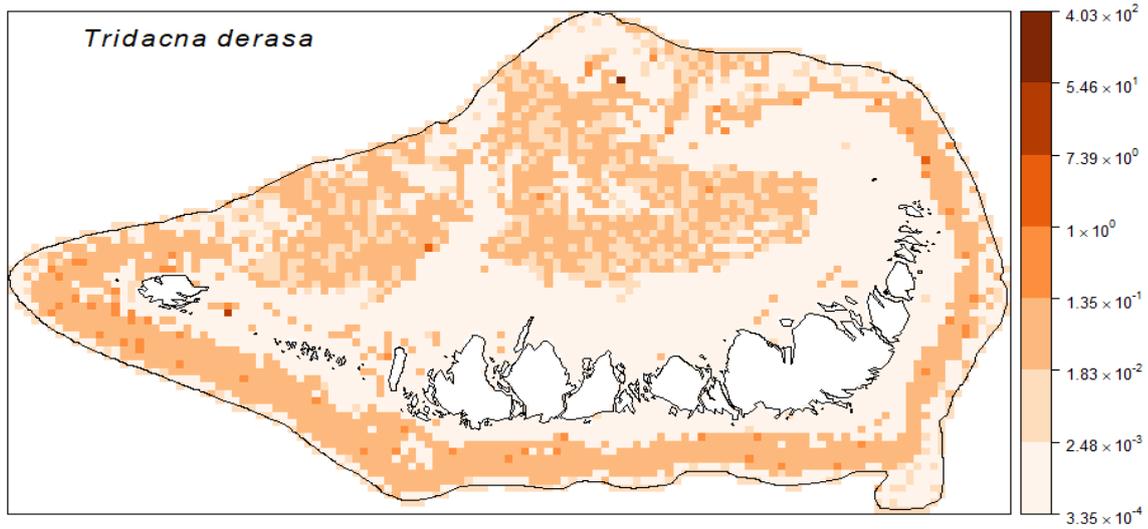


Figure 178. Posterior mean of areal density of *Tridacna derasa* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

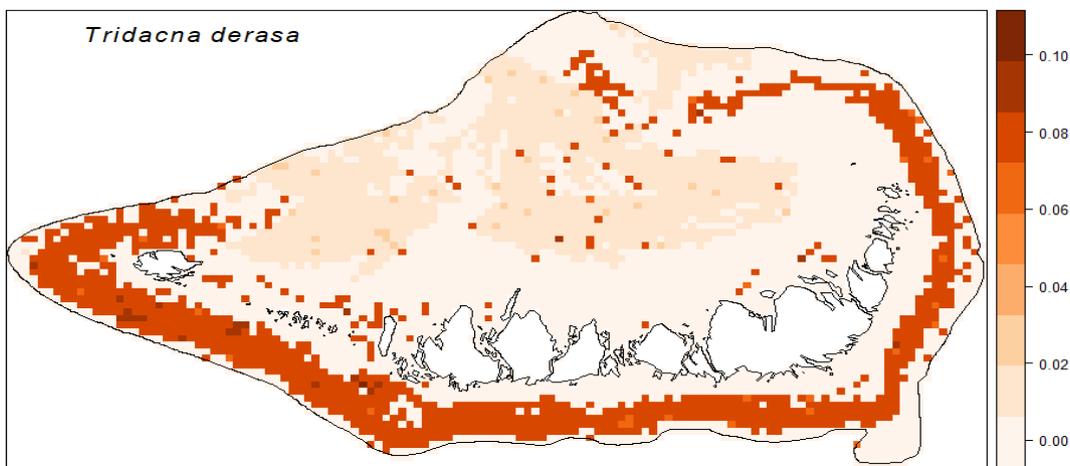


Figure 179. Posterior inter-quartile distance of areal density of *Tridacna derasa* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

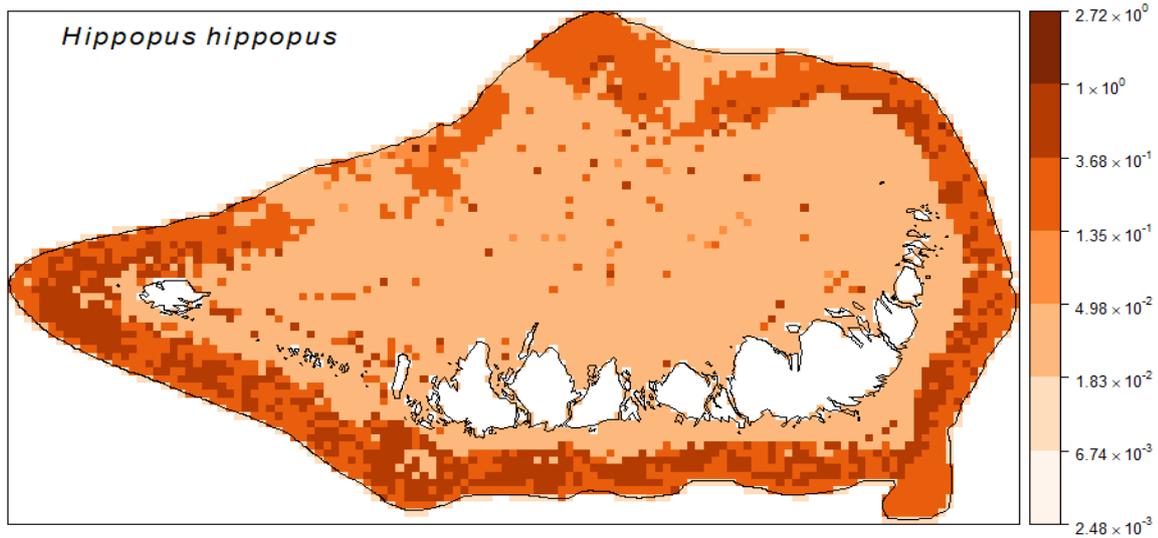


Figure 180. Posterior mean of areal density of *Hippopus hippopus* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

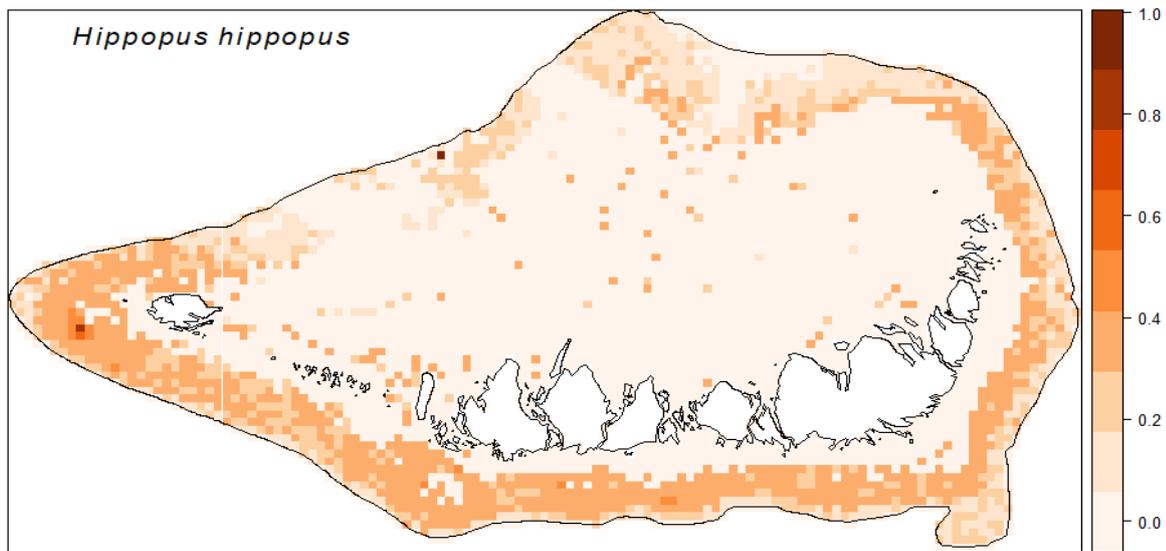


Figure 181. Posterior inter-quartile distance of areal density of *Hippopus hippopus* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

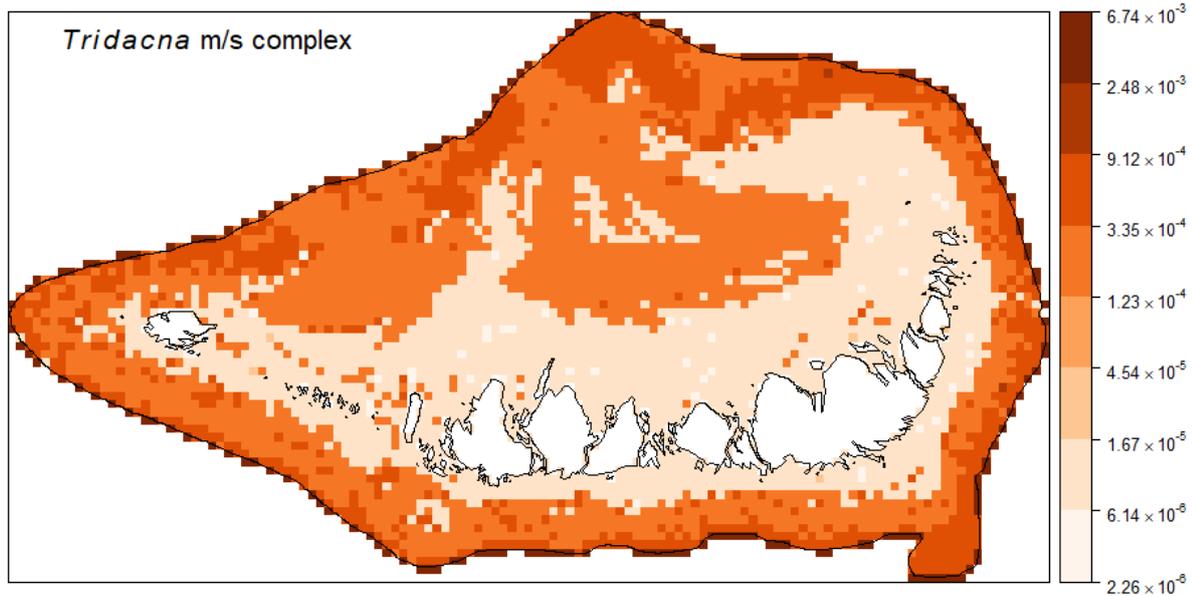


Figure 182. Posterior mean of areal density of *Tridacna maxima* and *Tridacna squamosa* combined (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

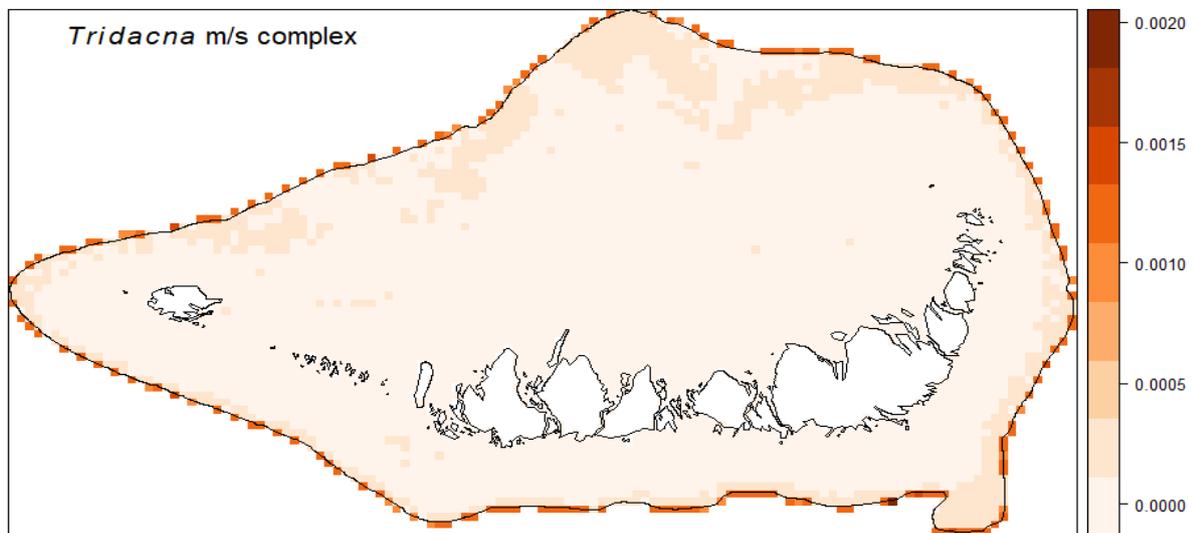


Figure 183. Posterior inter-quartile distance of areal density of *Tridacna maxima* and *Tridacna squamosa* combined (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

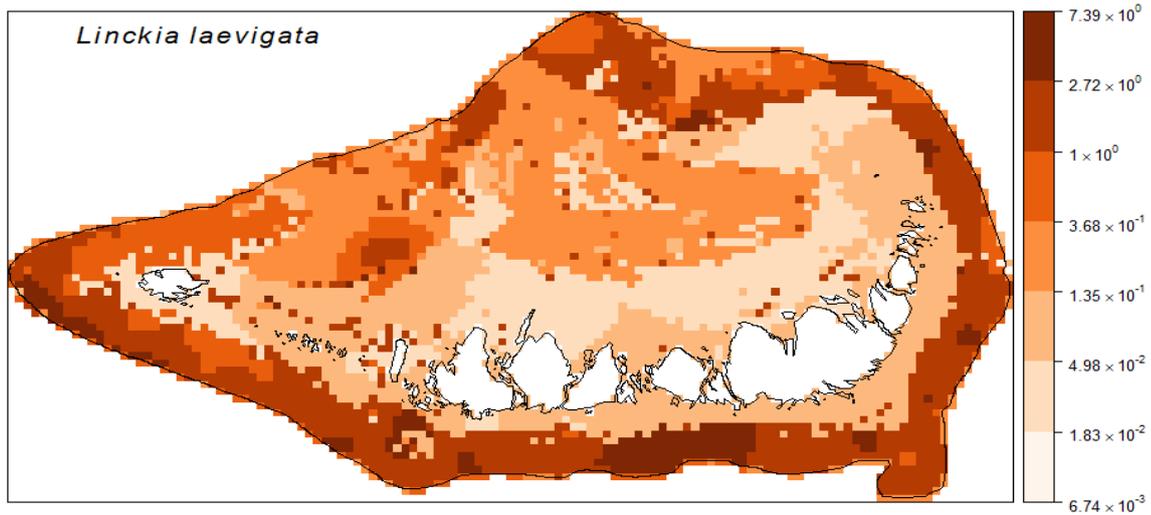


Figure 184. Posterior mean of areal density of *Linckia laevigata* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

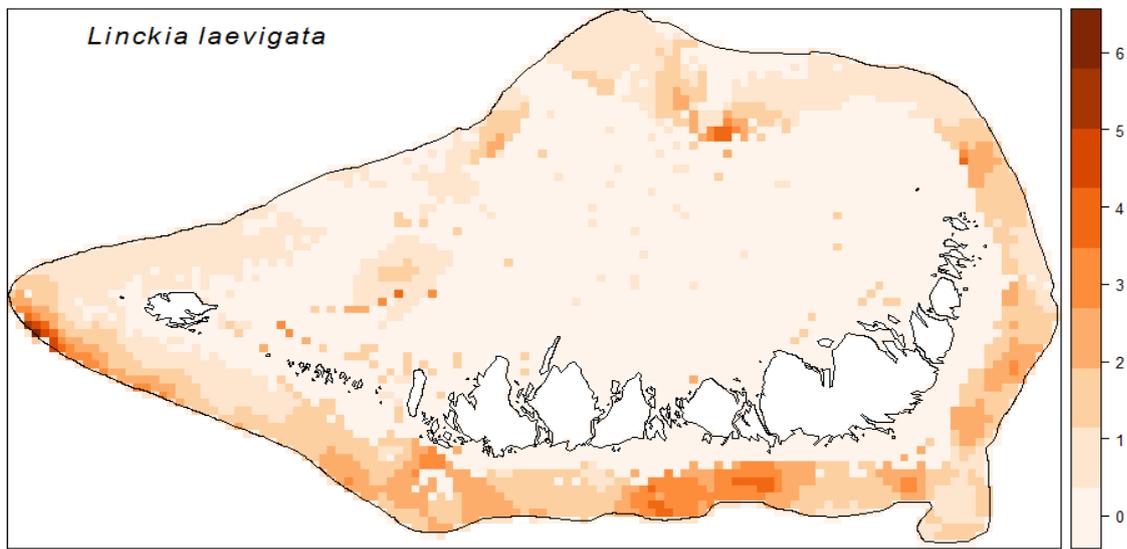


Figure 185. Posterior inter-quartile distance of areal density of *Linckia laevigata* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

## 12.4.2 Temporal Estimates

The estimated abundance in years 2013 and 2019 were obtained from the predictive joint posterior distribution of the observations for each year. The predicted abundances were scaled up by the size of the grid cells and multiplied by the number of grid cells within the Ashmore Reef sample frame to obtain a total abundance in 2013 and 2019.

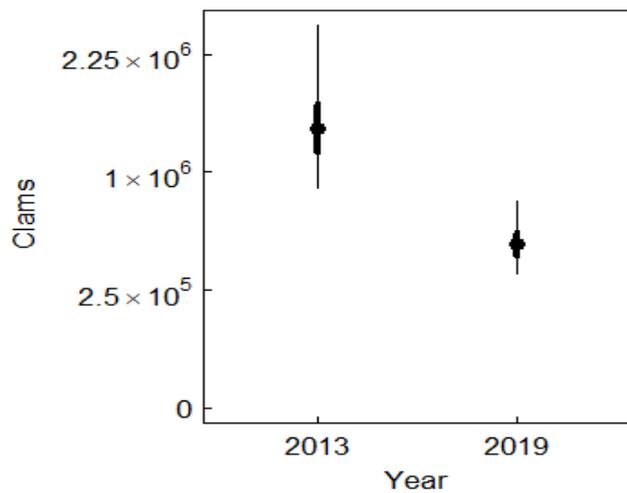


Figure 186. Estimated abundance of all clam species in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

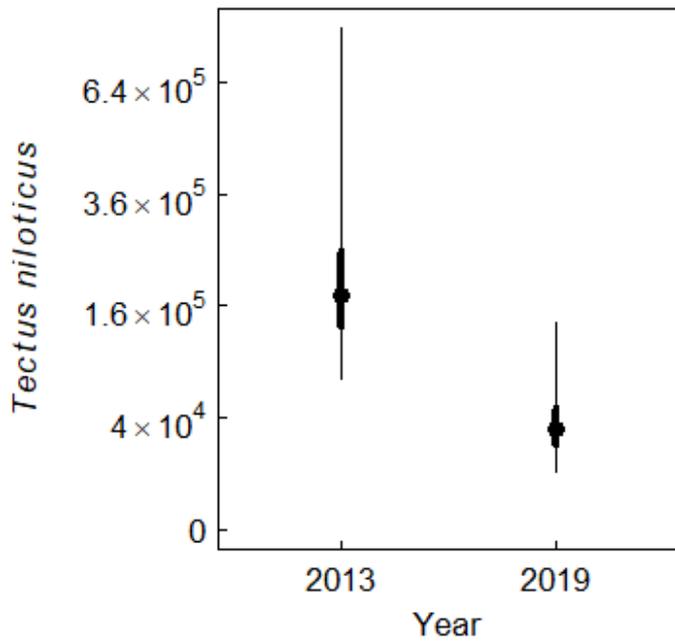


Figure 187. Estimated abundance of trochus in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

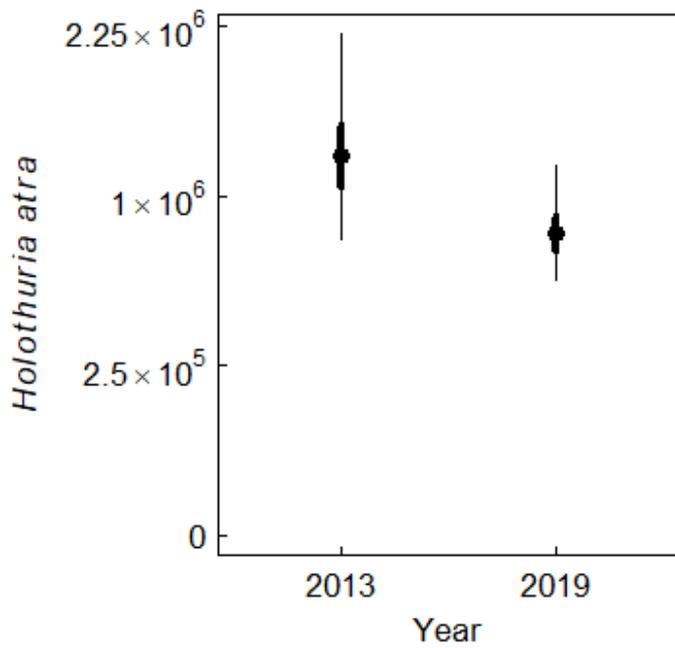


Figure 188. Estimated abundance of *Holothuria atra* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

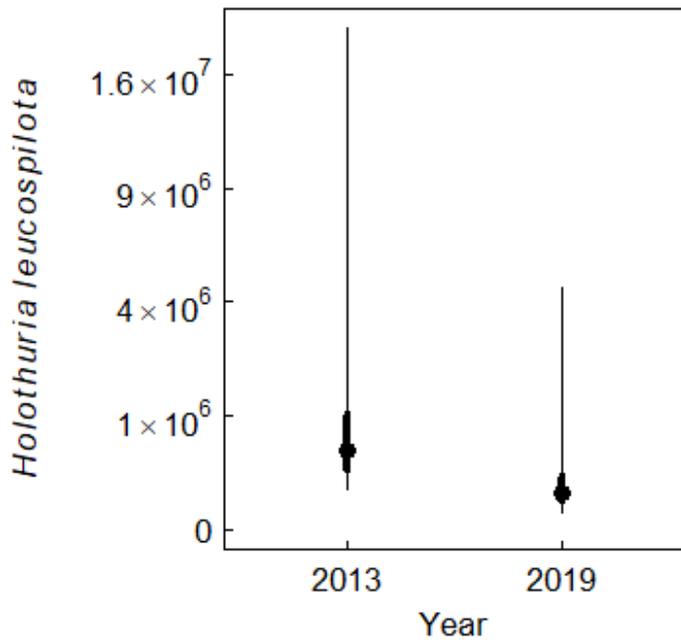


Figure 189. Estimated abundance of *Holothuria leucospilota* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

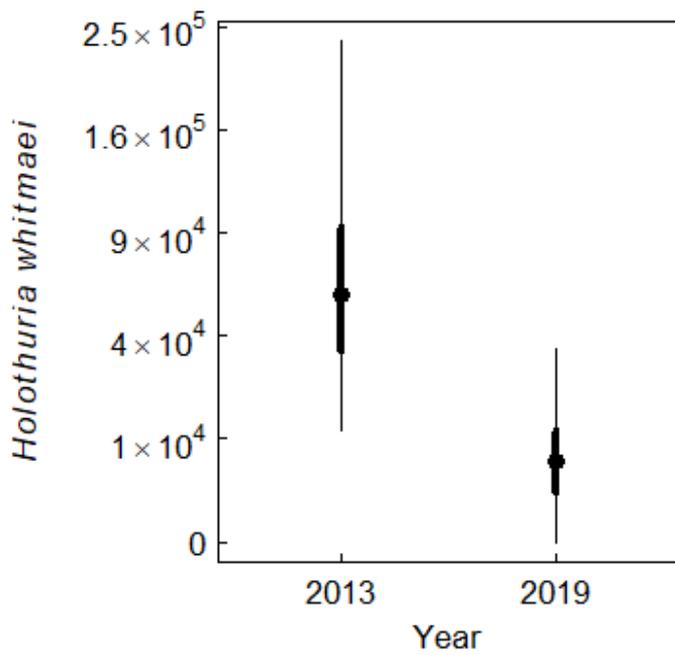


Figure 190. Estimated abundance of *Holothuria whitmaei* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

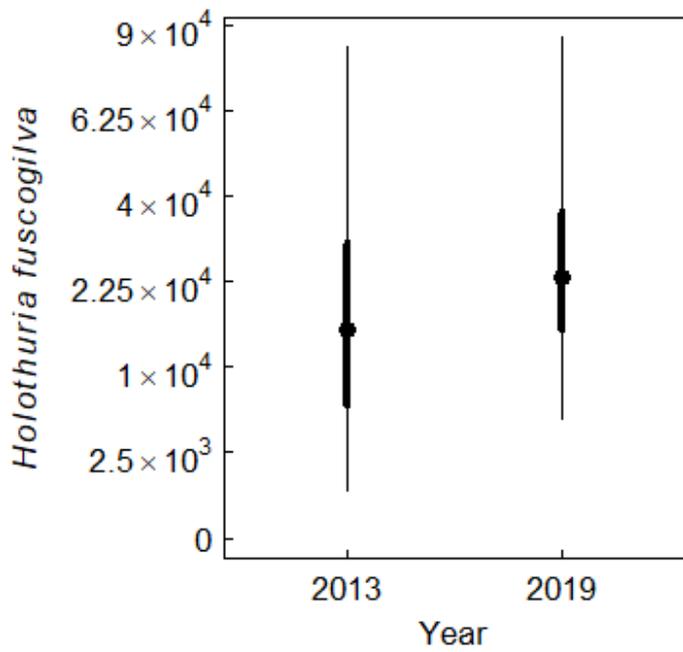


Figure 191. Estimated abundance of *Holothuria fuscogilva* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

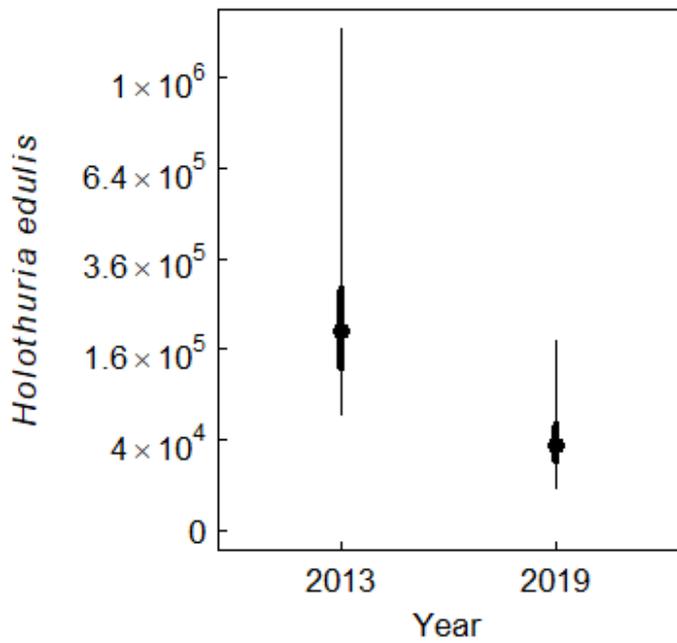


Figure 192. Estimated abundance of *Holothuria edulis* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

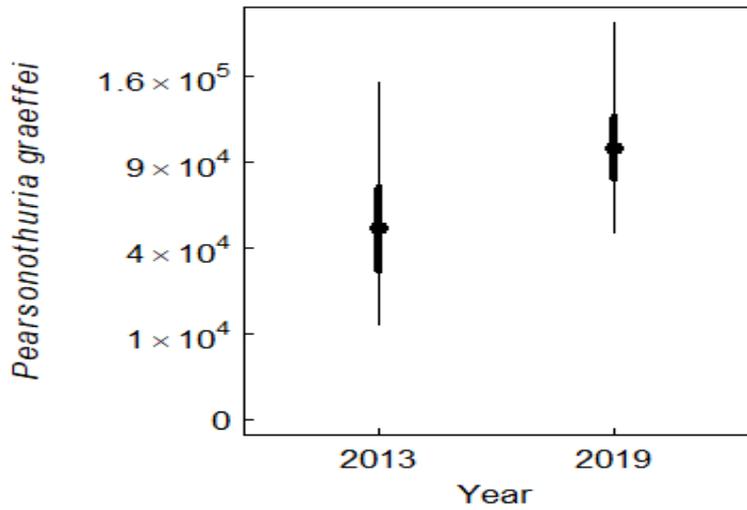


Figure 193. Estimated abundance of *Pearsonothuria graeffei* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

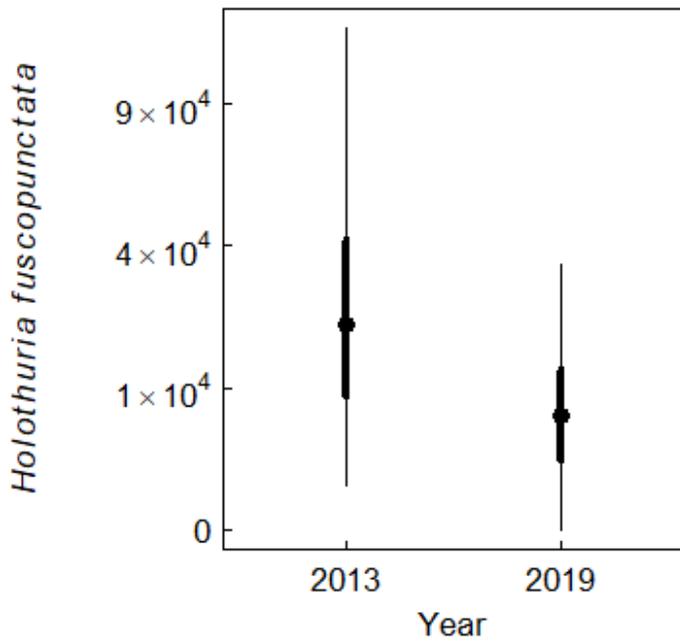


Figure 194. Estimated abundance of *Holothuria fuscopunctata* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

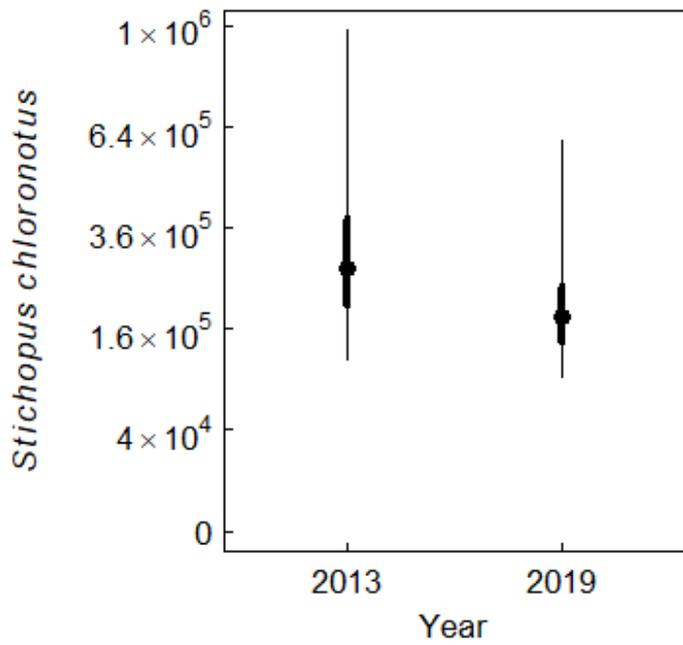


Figure 195. Estimated abundance of *Stichopus chloronotus* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

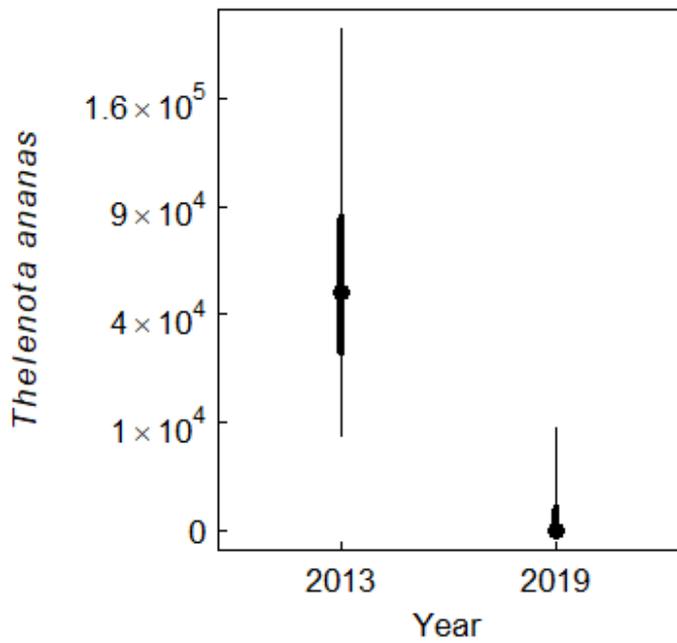


Figure 196. Estimated abundance of *Thelenota ananas* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

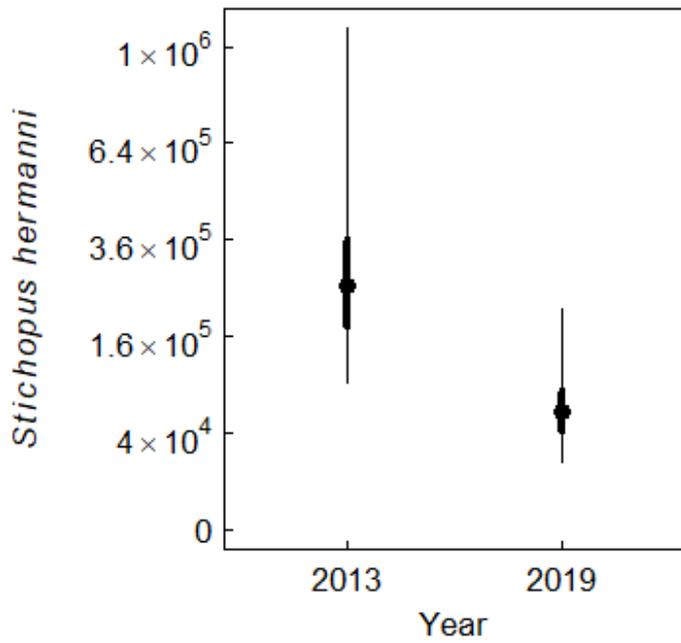


Figure 197. Estimated abundance of *Stichopus hermanni* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

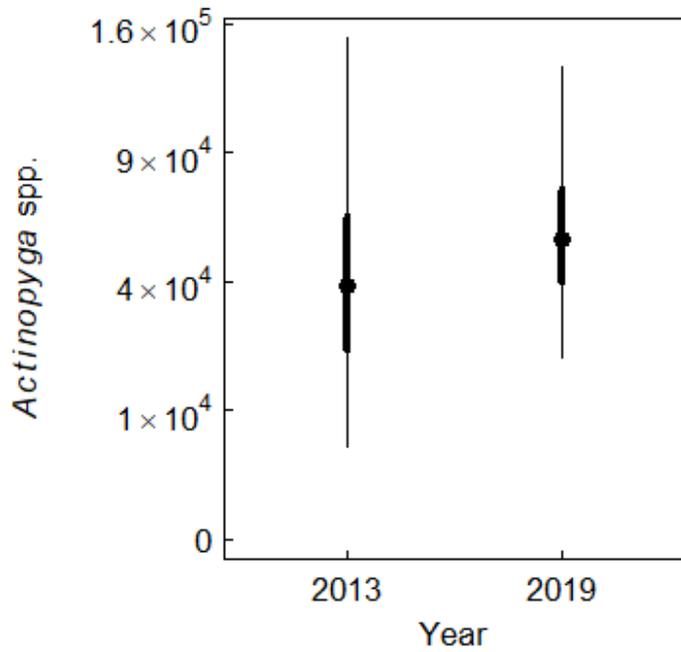


Figure 198. Estimated abundance of *Actinopyga* spp. in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

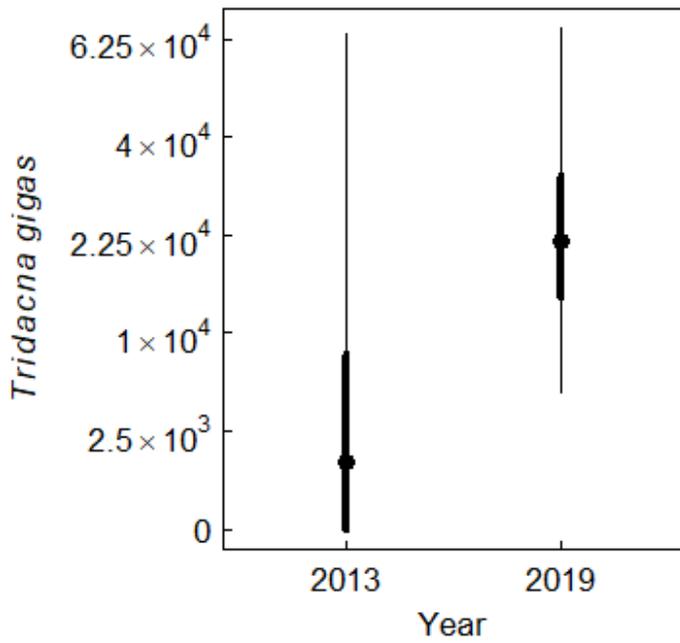


Figure 199. Estimated abundance of *Tridacna gigas* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

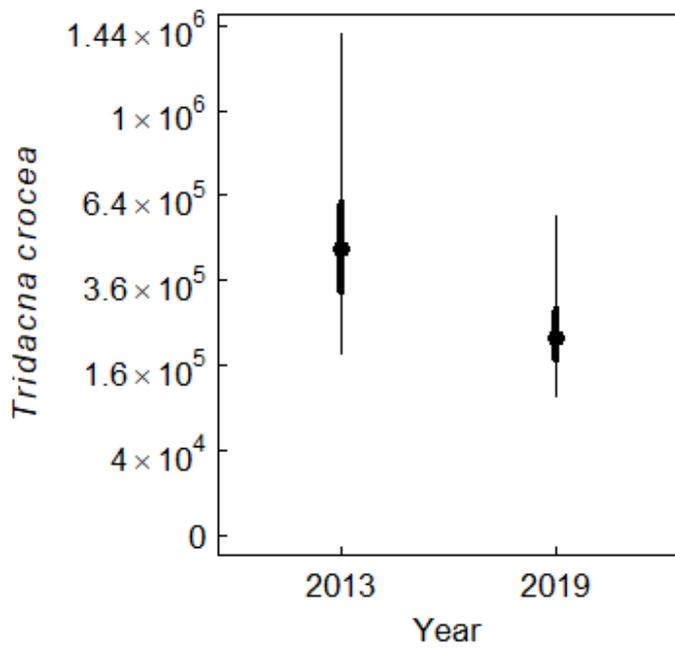


Figure 200. Estimated abundance of *Tridacna crocea* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

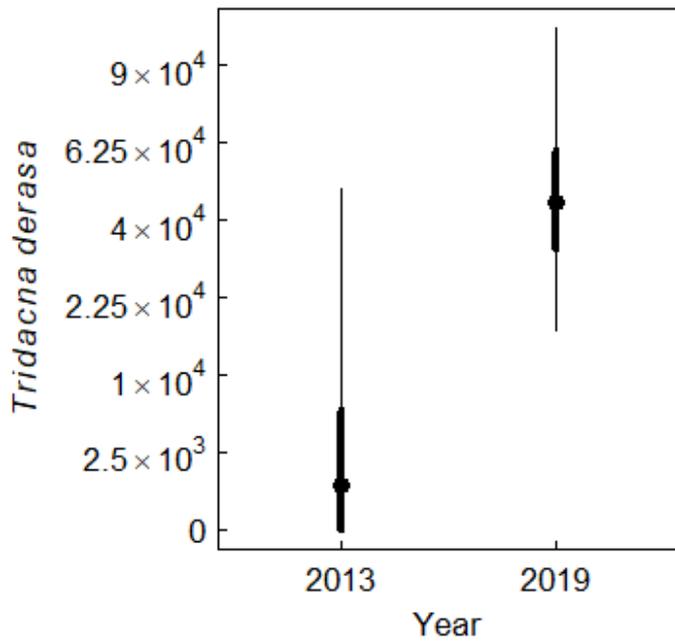


Figure 201. Estimated abundance of *Tridacna derasa* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

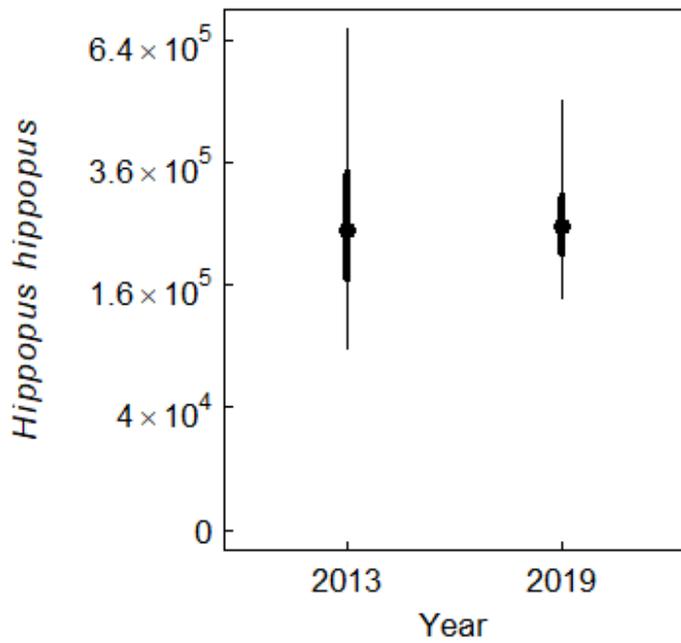


Figure 202. Estimated abundance of *Hippopus hippopus* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

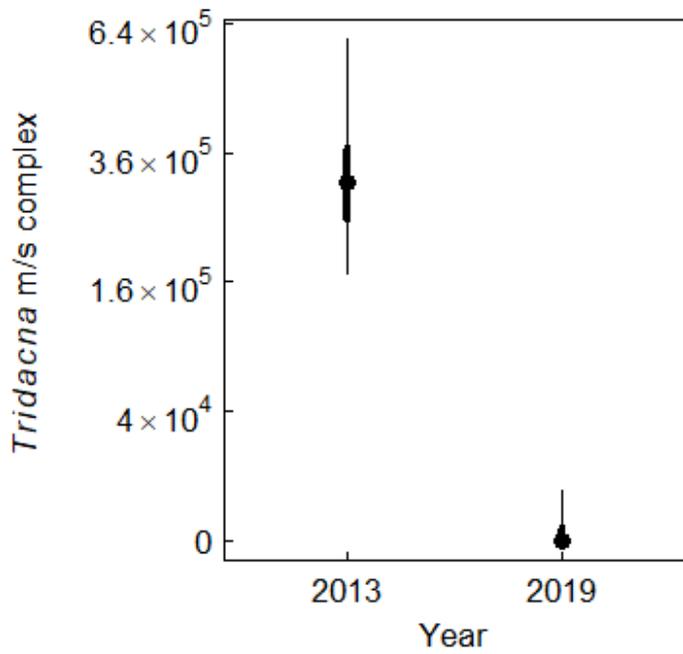


Figure 203. Estimated abundance of *Tridacna maxima* and *Tridacna squamosa* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

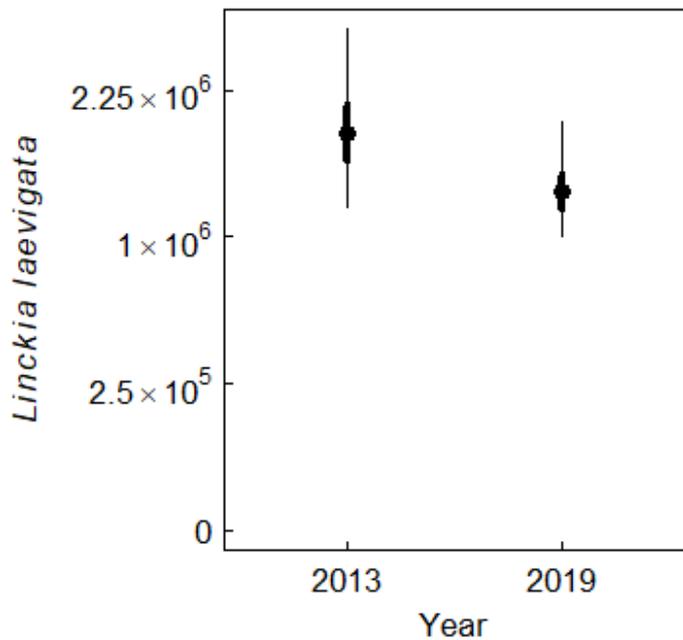


Figure 204. Estimated abundance of *Linckia laevigata* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

## 12.5 Discussion

The spatially balanced design of the 2019 Ashmore Reef swim surveys of benthic invertebrates ensured broad coverage of benthic habitats, which revealed different patterns of habitat and niche differentiation by benthic taxa. The 2019 survey for benthic invertebrates at Ashmore Reef also followed the same study design as the 2013 survey Ceccarelli *et al.* (2013). Both the 2013 and 2019 surveys used the spatially balanced design and field protocols documented in Hosack and Lawrence (2013a). The common field protocols and study design between the two surveys enabled a direct comparison of changes in abundance over the 5+ year interval between surveys. Such temporal comparisons are more difficult when comparing previous surveys at Ashmore Reef, which implemented a wide range of study designs and field protocols that degrade the ability to estimate temporal change in abundance of benthic invertebrates (Hosack & Lawrence 2013b). The 2013 and 2019 surveys provide the opportunity to assess temporal change at the reef scale based on a spatially balanced design implemented using common field protocols.

The implementation of the GRTS design permits use of both design-based (Stevens & Olsen 2004) and model-based analyses (see preceding sections). Design based estimators must account for spatial autocorrelation that arise from shared environmental factors among neighbouring sites (Stevens & Olsen 2004). The model-based approach implemented in this analysis and by Hosack and Lawrence (2013b) flexibly accommodates spatial dependence among sites using Gaussian processes that are estimated from the observed data. Moreover, the model-based approach allows flexible accommodation of site drop out and replacement due to sampling limitations in the field that result in departures from the pre-specified GRTS design. The model-based approach to total abundance estimation considers habitat information and spatial dependence that would not be addressed by the standard GRTS design-based estimator. The model-based approach thereby permits prediction of areal densities and observations to unsampled locations, which would not be possible with the design-based approach.

The model-based analysis of benthic invertebrate abundance revealed strong habitat differentiation and spatial patchiness among the taxa of interest. Both habitat covariates and spatial dependence are important features for benthic invertebrates at Ashmore Reef, as shown by the spatial maps of taxa areal densities in 2019. Big differences in spatial distributions are apparent with clear evidence of different habitat preferences across taxa. Clams, for example, were most abundant along the reef flat. *Rochia niloticus* abundance was also higher on the reef flat compared to sand and lagoon habitats but abundance was highest along the reef edges.

Among holothurians, *Holothuria atra* and *Stichopus choloronotus* were abundant along the reef flat with highest abundances of the former species estimated at the western end of Ashmore Reef. In contrast, *Holothuria leucospilota* was much more widely spread with no clear habitat preferences; the distribution of this species is associated with occasional very localised patchiness. The highest abundances of *Actinopyga* spp. were concentrated along the reef edge. The remaining holothurians had high relative abundances in the lagoon habitat. *Holothuria fuscopunctata* was predominately in lagoon. *Holothuria edulis* also extended up to the reef flat, *Stichopus herrmanni* along the exterior of the reef flat, and *Holothuria whitmaei* along the reef edge. Temporally, increased mean abundances were estimated for the holothurians *Holothuria fuscogilva*, *Pearsonothuria graeffei* and *Actinopyga* spp. Decreased mean abundances from 2013 to 2019 were estimated for several species of holothurians, including *Holothuria atra*, *Holothuria whitmaei*, *Holothuria edulis*, *Holothuria fuscopunctata*, *Thelenota ananas*, *Stichopus choloronotus*

and *Stichopus herrmanni*. Uncertainty arose due to the high heterogeneity of species' distributions. The 95% credible intervals overlapped for these taxa with the exception of *Thelenota ananas*, which appeared to have the most dramatic decline between surveys.

The 2019 survey also performed manta–tow sampling, mostly along the edge habitat of Ashmore Reef but also within the lagoon habitats. This form of sampling is particularly well-suited for species such as *Linckia laevigata* (see chapter 8.3.4). The 2-minute manta tows that observed the presence of *L. laevigata* were roughly associated with high density areas along the reef flat as identified by the swim surveys. Analysis of the latter data suggest that the preference gradient for this species is reef flat, lagoon, and reef edge or sand. Within these habitats, some regions of the reef tend to have high abundance of *L. laevigata*, which suggested spatial patchiness within habitat types.

The shared sampling methodology between surveys (Hosack & Lawrence 2013a,b), combined with the hierarchical spatial modelling framework described above, allowed an efficient estimation of spatial variability for each taxon of interest and also permitted temporal comparisons. Returning to this spatially balanced design between years means that the site locations become reference sites, also known as legacy sites (Foster *et al.* 2017). These reference sites were selected in way that is representative of Ashmore Reef and also efficient because of the spatially balanced design (Hosack & Lawrence 2013a). Following the same design across years increases the chances of detecting temporal changes among benthic invertebrates at Ashmore Reef. In contrast, following a different design each year, or selecting sites on an ad hoc or preferential basis, would have introduced increased uncertainty among any inter-year comparisons of abundance. The increased uncertainty would be a direct result of increased spatial variability, were future surveys to diverge from the current sampling design. It is for this reason that reference sites are deliberately targeted in recurring ecological surveys, so as to increase the ability of trend detection (Urquhart & Kincaid 1999), which has been the approach for the 2013 and 2019 Ashmore Reef surveys.

Temporal variation in abundance of benthic invertebrates at Ashmore Reef can be driven by a range of possible factors. For example, it is possible that taxa have been impacted either directly by heat stress or cyclone impacts, which have been documented to occur between the 2013 and 2019 sampling years (Gilmour *et al.* 2019), or indirectly by the resulting impact on coral. Coral cover at Ashmore Reef declined from 36% to 24% following cyclones and coral bleaching in 2016 and 2017 (Gilmour *et al.* 2019). Another possibility is that Allee effects may have constrained the ability of some taxa, such as *Thelenota ananas*, to recover following a period of suspected high exploitation in the 1980s (see chapter 11 ). Sampling variability is another possible cause of temporal variability. For instance, in the 2013 survey 50% of trochus were discovered on a single 80 m<sup>2</sup> transect (see chapter 11 ). High spatial aggregation increases the uncertainty associated with the abundance estimates thereby making inter-year comparisons difficult. Such sample variability can in principle be reduced by increasing the amount of surveyed area, for example, by increasing the number of surveyed sites. Another option would be to increase the area of the transect survey at each site. However, the results above demonstrate that increasing the number of sites is the better strategy as it helps estimate the degree of spatial dependence that is shown to be important for many of the target taxa. Both options of course lead to increased survey effort and hence directly impact the cost of future survey implementation.

Although the common methods employed in the 2013 and 2019 surveys enables estimation of temporal change in abundances, it is important to note that the estimated total abundance of benthic invertebrate taxa at Ashmore Reef across years is an uncontrolled time series, where the

term “uncontrolled” refers to the absence of an experimental control. Uncontrolled time series lessen the ability to attribute changes in abundance to causal factors (Hayes *et al.* 2019). In the context of Ashmore Reef, it is nevertheless a step forward to be able to compare these recent survey data derived from a common field methods protocol and spatially balanced sample design.

The detection of temporal trends is a reasonable objective for specific taxa at Ashmore Reef, particularly those of the more common taxa documented in the previous section. On the other hand, the testing of defined hypotheses for causal mechanisms that might explain observed changes in abundance will require an additional step toward a broader information base. The ability to attribute temporal change of Ashmore Reef benthic invertebrates to causal factors, which may include but are not limited to cyclones and climatic drivers, is currently limited by lack of comparison to other reefs that have a different suite of exposures to plausible factors. A careful expansion of the survey design to additional reef ecosystems is required to test hypotheses of attribution among a well-defined set of potential causal factors such as cyclones, extreme weather events and heat stress that impact coral reef health.

## 12.6 Management implications and recommendations

As indicated above, the ability to attribute temporal change of Ashmore Reef benthic invertebrates to causal factors, is currently limited by lack of comparison to other reefs. Expansion of the survey design to add reference sites is required to test hypotheses of attribution of change in abundance to a specific cause. These points are discussed in more detail in chapter 11.

## 13 ASHMORE REEF: SEAGRASS MONITORING

Emma Westlake, John Keesing, Lauren Hardiman, Mark Tonks, Margaret Miller, Russ Babcock, Cindy Bessey, Daniella Ceccarelli, Thais Costa Dalseno, Christopher Doropoulos, Michael Haywood, Ylva Olsen and Damian Thomson

### 13.1 Abstract

Seagrasses are vitally important for the structure and function of coral reefs, contributing to productivity, providing food, habitat and nursery areas for marine organisms, and they are widely recognised for their ability to modify tidal currents, sediment composition and sediment stabilisation. Species diversity, seagrass cover, canopy height, shoot density and biomass were measured using transects and quadrats at four sites. Growth rates of *Thalassia hemprichii* were measured at two monitoring sites. Of the five species of seagrass previously recorded at Ashmore Reef, only two species were recorded at the sites surveyed in 2019 – *T. hemprichii* and *Halophila ovalis*, with seagrass cover predominantly composed of *T. hemprichii* (mean 8.6%, range 1 to 35%). *Thalassia hemprichii* was present at all sites with a mean canopy height of 42.8 mm (range 10 to 80 mm) while *H. ovalis* was only recorded at one site (26.7 mm, range 20 to 50 mm). Mean shoot density or mean biomass were not calculated for *H. ovalis* due to its sparsity. *Thalassia hemprichii* mean shoot density was 407 shoots/m<sup>2</sup> (range 88.9 to 600.0 shoots/m<sup>2</sup>) across sites. Overall mean total biomass was 116.9 g DW/m<sup>2</sup> across all sites. Mean above ground biomass for *H. ovalis* was 0.015 g DW ( $\pm$  0.003) and below ground biomass was 0.023 g DW ( $\pm$  0.007) with an above and below ground ratio of 1:3. *Thalassia hemprichii* mean above (54.2 g DW/m<sup>2</sup>  $\pm$  1.24) and below (224.5 g DW/m<sup>2</sup>  $\pm$  5.2) ground biomass was greatest at site 005. Above and below ground biomass ratio was 1:2.5. The average blade length and width of *T. hemprichii* was 37.6 mm ( $\pm$  1.1) and 4.5 mm ( $\pm$  0.2) respectively. Productivity and growth measurements showed mean leaf growth of 2.2–2.4 mm/day and shoot growth of 3.2–3.5 mm/day. Canopy height increased by 5.3–5.7%/ day with a turnover rate of 17.5–21.3 days. Mean biomass of emergent leaves was 8 g DW/m<sup>2</sup> with grazing rate calculated to be 0.4 g DW/m<sup>2</sup>/day or 123 kg DW/ha/month. Biomass and productivity results obtained in 2019 were similar to those of previous surveys conducted at Ashmore Reef, however, were lower than in other areas of the world. Smaller blade dimensions, above ground to below ground biomass ratios and growth rates were observed in seagrass at Ashmore Reef. This physiological variability may be afforded to the environmental conditions to which it's subjected including exposure to high temperatures and low salinity, low nutrients and high current velocity.

## 13.2 Introduction

Macroalgae and seagrasses are important components of tropical reefs. They contribute significantly to the productivity of reef ecosystems, provide a major source of food for marine organisms (Skewes *et al.* 1999b; Brown 2001; Vonk, Christianen & Stapel 2008) and provide habitats for many marine species, including nursery refugia for juvenile fish (Nagelkerken *et al.* 2002; Nagelkerken 2009). Seagrass beds provide vital coastal protection services including wave attenuation and prevention of erosion (Christianen *et al.* 2013) and modify hydrochemistry (Unsworth *et al.* 2012) and biogeochemical cycling, (Harborne *et al.* 2006; Bouillon & Connolly 2009). Furthermore, seagrasses are significant contributors to primary production on a global scale (Smith 1981; Charpy-Roubaud & Sournia 1990; Harborne *et al.* 2006).

Ashmore Reef has the highest seagrass cover of the bioregion, including Scott Reef and Seringapatam Reef (Skewes *et al.* 1999b). The total seagrass area at Ashmore Reef has been estimated at 470 ha (Skewes *et al.* 1999b). Much of the seagrass coverage in this area is considered sparse with only 220 ha of area covered by > 10% seagrass (Brown & Skewes 2005). Seagrass cover is relatively high on the shallow reef flats and sparse and patchy across the sand flats (Skewes *et al.* 1999b). Ashmore Reef has been recognised as a globally significant site for grazing vulnerable green (*Chelonia mydas*) turtles (Brown & Skewes 2005) and a small population (10-60 individuals) of dugongs (*Dugong dugon*) (Whiting & Guinea 2005b).

Five species of seagrass have been recorded at Ashmore Reef; *Thalassia hemprichii*, *Thalassodendron ciliatum*, *Halophila ovalis*, *H. decipiens* and *Halodule pinifolia* (Pike & Leach 1997). The latter two have not been observed in recent surveys of the reef, possibly due to the naturally low cover and shoot density, and highly seasonal and ephemeral nature of these small seagrass species (Hovey *et al.* 2015). Of these, *T. hemprichii* is the dominant species comprising up to 85% of the total cover in the meadows (Skewes *et al.* 1999b). Although located closer to Indonesia than the Australian coastline, *T. hemprichii* at Ashmore Reef may be genetically distinct to that of Indonesia, aligned more closely with that of other parts of the Indian Ocean and Australia (Hernawan *et al.* 2017).

Seagrasses are important indicators or biological markers of relative health of tropical and subtropical estuarine ecosystems (Kemp 1999). Ongoing seagrass monitoring allows the collection of information on the status of ecosystems, the detection of signs of degradation (Yokoi *et al.* 2011) and subsequent recovery following disturbance (Kirkman & Kirkman 2000). Over the last 30 years, a global trend of regional declines in seagrass abundance has occurred, with most of these declines attributed to human-induced disturbances including dredging, siltation and turbidity adjacent to urban centres (Heck 1976; Onuf 1994; Bach *et al.* 1998). While the location of Ashmore Reef away from mainland influences substantially reduces the risk of such human-induced disturbances, it is not immune to changes associated with natural climatic cycles (Marba & Duarte 1995). By establishing monitoring sites within the seagrass meadows at Ashmore Reef using established standardised methodologies (McKenzie, Campbell & Roder 2003), this project has established the opportunity for a long term time series of data on seagrass biomass and productivity to be collected to periodically determine status and trends in seagrass health which will be comparable with work across the broader Indian Ocean region.

## 13.3 Methods

### 13.3.1 Seagrass monitoring

Eight monitoring sites were selected pre-voyage based on prior reports of the distribution of *T. hemprichii*, *T. ciliatum* and *H. ovalis* (Figure 205). Easily accessible sites around West Island were also selected for seagrass productivity measurements based on these previous studies.

However, after arriving at Ashmore Reef in 2019, no evidence of *H. ovalis* or other seagrass was found at sites SGHO1 or SGHO2, or *T. ciliatum* at sites SGTC1 or SGTC2. Nor were there any seagrass beds in the shallow waters at the five sites chosen around West Island (Figure 205). However, *T. hemprichii* was found to be abundant at SGTC1 and SGTC2 and *H. ovalis* was common at SGTC1. Sites SGTH1, SGTH2 and SGTH3 were considered too remote from the main vessel mooring to establish sites that could be regularly monitored. Ultimately, four sites were selected for monitoring *T. hemprichii*. These were SGTC1 and SGTC2 north-east of West Island and, SGTH4 and SGTH5 to the south-west of West Island (Figure 206). The last of these sites, SGTH5 coincided with one of the reef flat fish/invertebrate transect sites (site 005) and is thus referred to as site 005 hereafter.

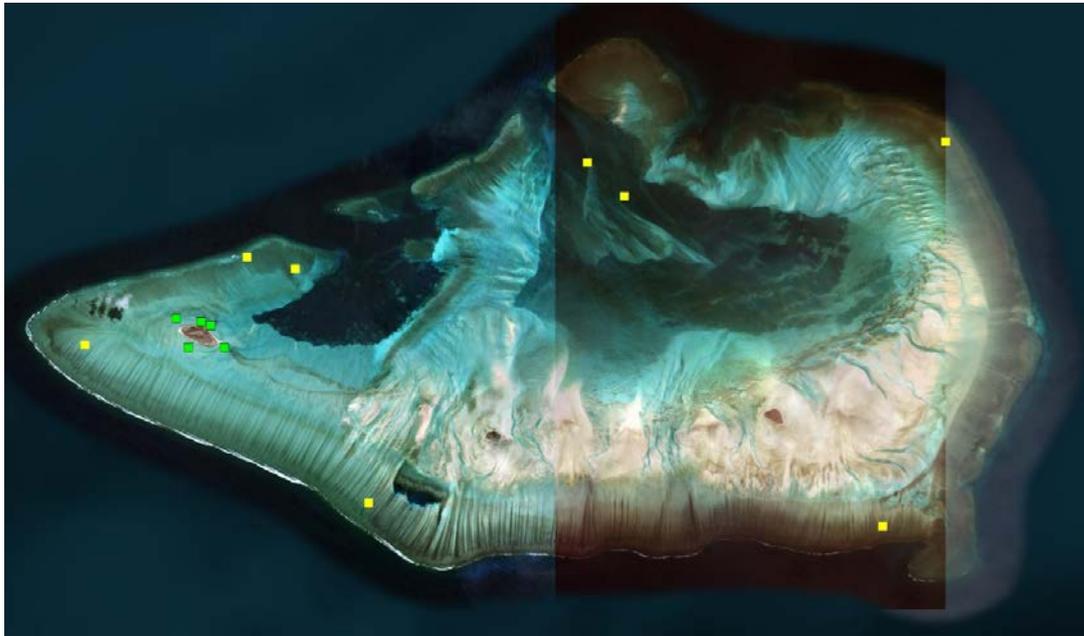


Figure 205. Pre-voyage planned seagrass monitoring sites (yellow) and growth rate sites (green dots around West Island) at Ashmore Reef based on historical studies. These were subsequently replanned following arrival and reconnaissance at Ashmore Reef in 2019.



Figure 206. 2019 seagrass monitoring (yellow and green) and growth rate sites (green), remote sites not surveyed (purple) and sites where *H. ovalis* was previously recorded but absent during 2019 surveys (orange) at Ashmore Reef. Note: site SGTH5 coincided with invertebrate and fish monitoring site 005 and is referred to as 005 hereafter

### 13.3.2 Seagrass shoot density and biomass monitoring

The four seagrass monitoring sites (Figure 206, SI Table 1) were established based on methodologies and standard protocols comparable to previous work conducted across the broader Indian Ocean region. At each site, three 50 m transects were laid out starting at random points perpendicular to West Island shoreline. A 0.5 x 0.5 m quadrat was positioned every 5 m along the transect. Species diversity, seagrass cover, canopy height, epiphyte abundance and macroalgae cover were estimated by divers using standardised Seagrass-Watch protocols (McKenzie *et al.* 2007) and photographed (Figure 207). Counts of shoot density and abundance of flowers and fruits were made within a smaller 0.1 x 0.1 m. At the beginning and end of each transect, several ramets (sections of rhizome with attached shoots and roots) were collected until at least 10 shoots of *T. hemprichii* were obtained. Where *H. ovalis* was present, the same method was repeated. In the laboratory, specimens were separated into below- (roots and rhizomes) and above ground (leaves and sheaths) components, dried at 60°C and each component weighed to obtain dry mass. Leaf area (length and width measurements) and weights were obtained to calculate biomass per mm<sup>2</sup>. Biomass per m<sup>2</sup> was then calculated by multiplying the biomass per shoot by the average shoot density for each 0.1 x 0.1 m quadrat.



Figure 207. Diver conducting seagrass shoot density and biomass monitoring.

### 13.3.3 *Thalassia hemprichii* seagrass productivity

Growth rates of *T. hemprichii* were measured at two monitoring sites in the intertidal meadows north-east of West Island (sites SGTC1 and SGTC2, see Figure 206). Initial marking of seagrass leaf sheaths was conducted on 15 June 2019, with subsequent monitoring on 22 June 2019. At each site, six 0.2 x 0.2 m quadrats were selected and all shoots within each were marked for leaf productivity by SCUBA divers using a leather punch (Figure 208) following a standard technique (Short & Duarte 2001, Figure 179). This required fanning away the sediment by hand and exposing the sheath at the base of the shoot. Once the leaf sheaths were punched, sand was redeposited over the area to cover the rhizomes. The number of shoots punched was recorded and quadrats marked with steel pegs and flagging tape. Seven days later, allowing sufficient time to measure growth in this species (Pedersen *et al.* 2016; Kendrick *et al.* 2017), divers collected all shoots within the quadrats by first fanning away the sediment to expose the rhizomes and carefully collecting all shoots in the marked area. The shoots were then frozen onboard for subsequent analysis.



Figure 208. Image of seagrass leaf growth measurement highlighting new growth following marking.

The distance of the hole-punch marks above leaf sheaths were later measured in the laboratory and converted into productivity per day (Figure 209). Productivity was measured as the distance (in mm) the hole in the leaves had moved from that in the sheath and converted into growth rate/day (in mm). Note that this method does not consider new growth occurring at the terminal end of the ramet where new rhizome and new shoots are being produced. This growth was not measured due to time constraints.

Calculations were as follows:

Leaf growth/day = mean leaf growth (mm)/day

Shoot growth/day = mean total growth (mm) of all leaves/shoot/day

Canopy height increase (%)/day = mean leaf growth (mm)/mean canopy height (mm) x 100

Turnover rate in days = mean canopy height (mm)/mean leaf growth (mm)/day

Mean biomass of emergent leaves per square metre of reef flat seabed:

1. Mean leaves/m<sup>2</sup> = mean shoots/m<sup>2</sup> x mean leaves/shoot
2. Mean emergent leaf area (mm<sup>2</sup>)/m<sup>2</sup> seabed = mean leaves/m<sup>2</sup> x mean canopy height (emergent leaves in mm) x mean leaf width (mm)
3. Mean biomass of emergent leaves/m<sup>2</sup> of reef flat = mean emergent leaf area (mm<sup>2</sup>)/m<sup>2</sup> of seabed x mean DW (g) of 1 mm<sup>2</sup> of leaf

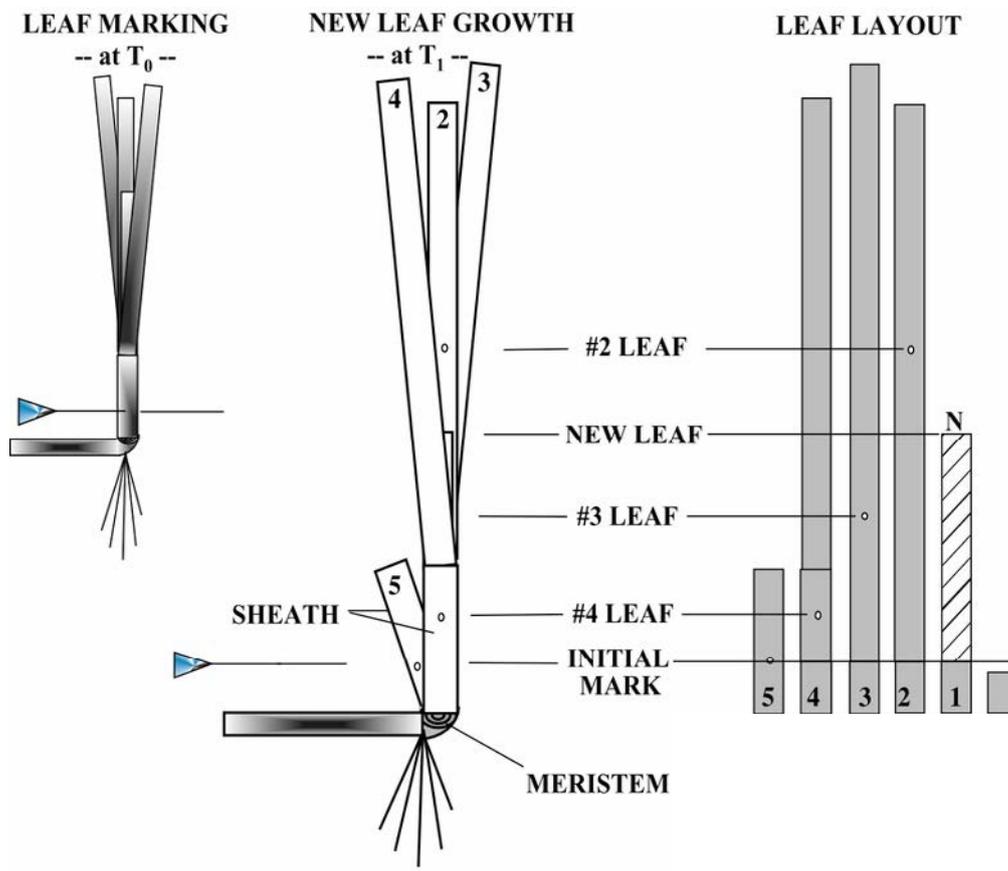


Figure 209. Schematic diagram of leaf growth measurement methodology showing initial mark and new growth (Short and Duarte 2001).

## 13.4 Results

### 13.4.1 Diversity and distribution

Monitoring and productivity sites were selected pre-voyage based on prior observations of *T. hemprichii*, *T. ciliatum* and *H. ovalis* distribution (Figure 205). Following arrival and reconnaissance in 2019, neither *H. ovalis* nor *T. ciliatum* were observed at their respective sites. The original sites were reconsidered (as discussed in the Methods section 13.3) resulting in a total of four sites chosen for seagrass shoot and biomass monitoring and *T. hemprichii* productivity measurements.

Of the five species of seagrass previously recorded at Ashmore Reef, only two species were recorded at the sites surveyed in 2019 – *T. hemprichii* and *H. ovalis*.

Seagrass was observed at multiple sites during benthic surveys (Figure 210). Percent cover was generally low across Ashmore Reef with highest cover recorded on the western reef flat (5-30%). Much lower cover and sparse distribution was observed along the southern and eastern reef flat (< 5%).

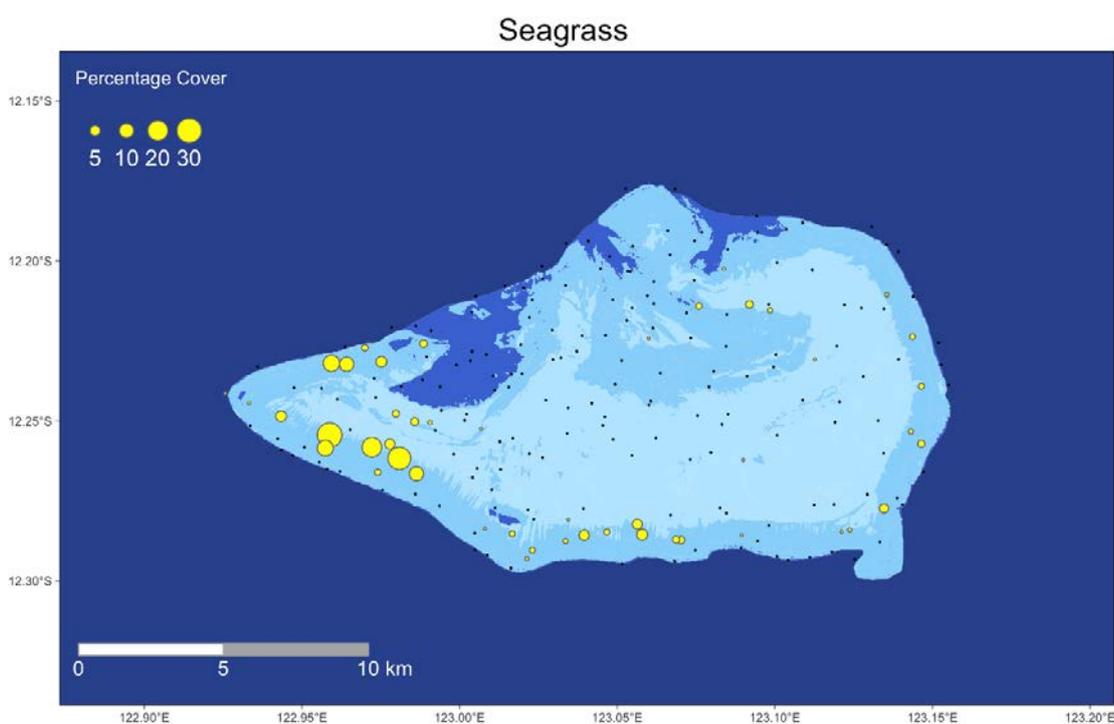


Figure 210. Percent cover of seagrass at Ashmore Reef measured along photo transects at 216 sites. See chapter 8 for details.

### 13.4.2 Seagrass shoot and biomass monitoring

Seagrass cover was predominantly composed of *T. hemprichii* across the four sites (Figure 211). While *T. hemprichii* was present at all sites, *H. ovalis* was only recorded at one site (SGTC1) comprising 8.5% of seagrass cover composition.

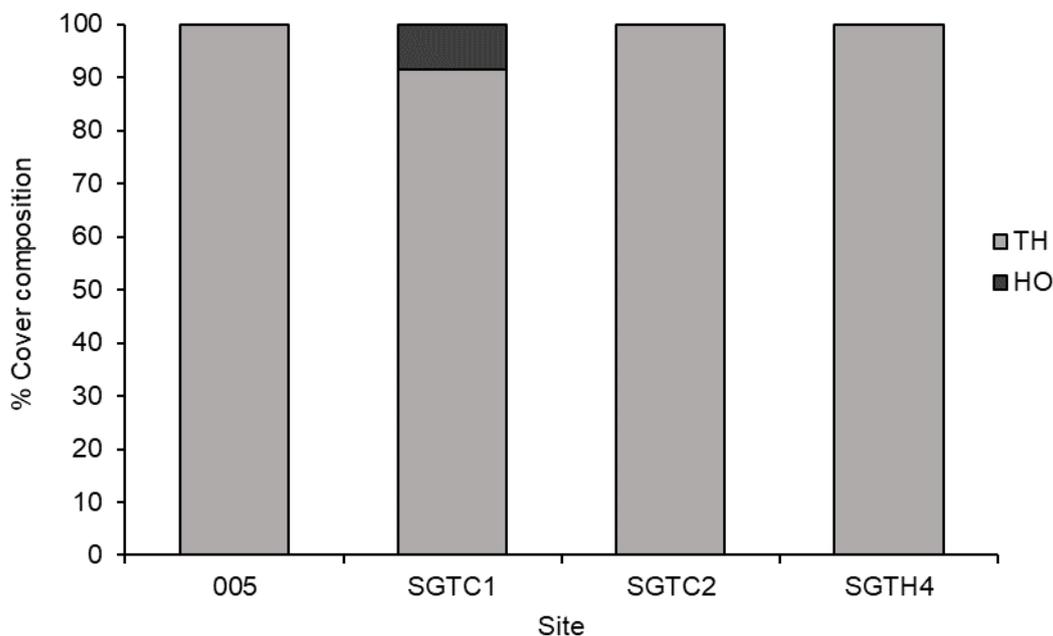


Figure 211. Percent cover composition of the two main seagrass species *T. hemprichii* and *H. ovalis* at each site.

Mean canopy height was greatest for *T. hemprichii* (mean 42.8 mm, range 10 to 80 mm) across all sites (Figure 212). Site SGTC2 measured the tallest mean canopy height for *T. hemprichii* (46.5 mm  $\pm$  1.8 s.e.), while site 005 measured the shortest (40.2 mm  $\pm$  1.1). It should be noted that the canopy height referred to here for *T. hemprichii* is the height above the sediment. At Ashmore Reef, the sediment generally covered the meristem and sheath such that only the leaves were emergent from the sediment. *Halophila ovalis* was only recorded at one site, with mean canopy height measured at 26.7 mm ( $\pm$  2.3, range 20 to 50 mm).

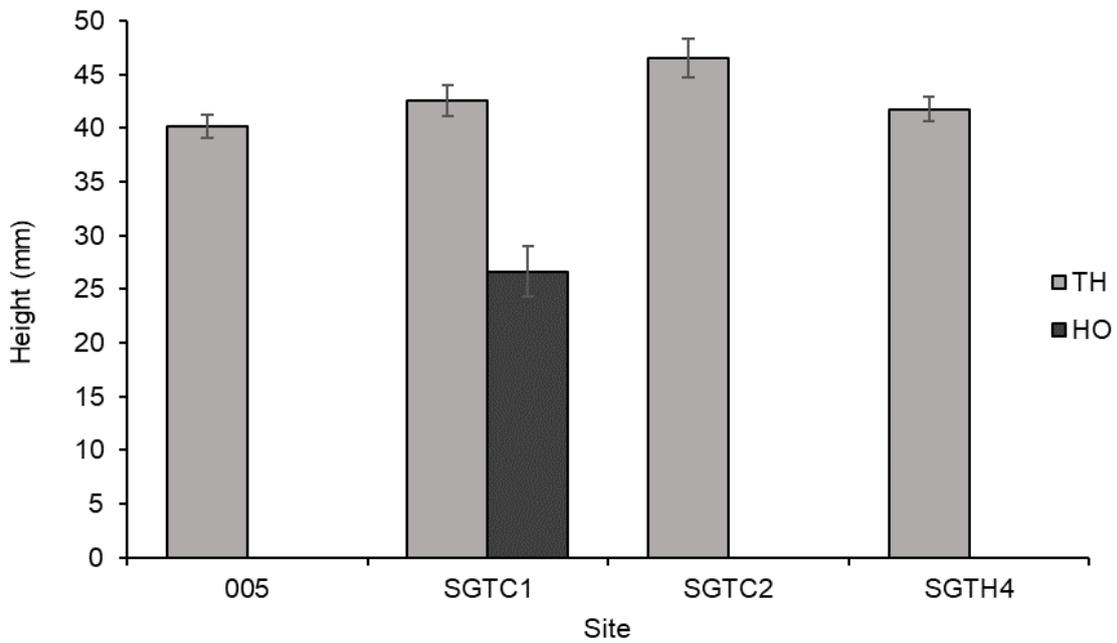


Figure 212. Mean canopy height (mm) of the two main seagrass species *T. hemprichii* and *H. ovalis* at each site. Error bars indicate  $\pm 1$  s.e.

Due to the sparsity of *H. ovalis* and minimal shoots detected during surveys, shoot density/m<sup>2</sup> was only calculated for *T. hemprichii* (Figure 213). Overall mean density was 407 shoots/m<sup>2</sup>. Mean shoot density greatest at site SGTC1 with a mean of 600 shoots/m<sup>2</sup> ( $\pm 200$  s.e.). Site 005 showed the second highest mean density with 575.8 shoots/m<sup>2</sup> ( $\pm 13.2$ ). The lowest mean shoot density was recorded at site SGTC2 with only 88.9 shoots/m<sup>2</sup> ( $\pm 36.8$ ).

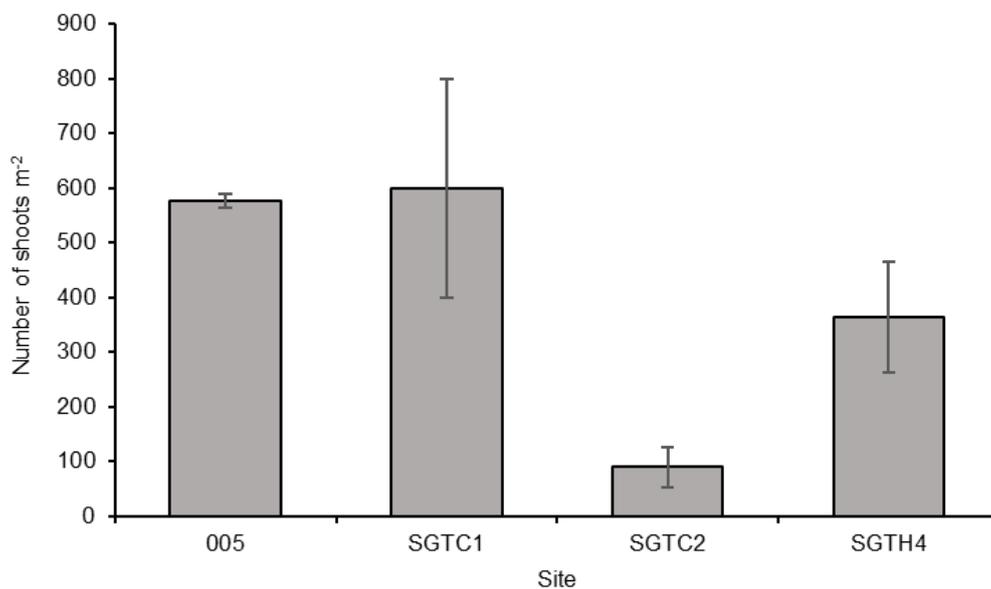


Figure 213. Mean shoot density per m<sup>2</sup> for *T. hemprichii* at each site. Error bars indicate  $\pm 1$  s.e.

Due to the sparse density of *H. ovalis*, mean total biomass was calculated for *T. hemprichii* only (Table 28). Overall mean total biomass was 116.9 g DW/m<sup>2</sup> across all sites. Site 005 showed the greatest mean total biomass with transect 3 the highest of all transects (290.4 g DW/m<sup>2</sup> ± 54.2 s.e.). The lowest mean total biomass was recorded for site SGTC2 transect 3 (5.1 g DW/m<sup>2</sup> ± 3.8). Neither transect 1 nor 2 of site SGTC1 had shoot density recorded during surveys resulting in a lack of biomass data available for these transects.

Table 28. Total mean biomass (g DW/ m<sup>2</sup>) for *T. hemprichii* across transects at each of the four sites. Dry to wet weight conversion = 9.56 (leaf) and 4.63 (root/rhizome).

Site	Transect	Mean biomass	stdev	s.e.
005	1	277.22	209.98	63.31
005	2	268.42	76.12	22.95
005	3	290.43	179.81	54.22
SGTC1	1			
SGTC1	2			
SGTC1	3	53.23	47.09	14.20
SGTC2	1	25.18	38.48	11.60
SGTC2	2	42.31	61.41	18.51
SGTC2	3	5.13	12.56	3.79
SGTH4	1	106.76	41.73	12.58
SGTH4	2	46.49	38.22	11.52
SGTH4	3	53.38	51.40	15.50

Although it was not possible to calculate mean total biomass for *H. ovalis*, it was possible to determine above and below ground biomass based on samples processed in the laboratory (Figure 214A). Mean above ground biomass (dry leaf weight) was 0.015 g DW shoot<sup>-1</sup> (± 0.003 s.e) while below ground biomass (dry root/rhizome weight) was 0.023 g DW shoot<sup>-1</sup> (± 0.007). Above and below ground ratio was 1:3, with below ground biomass being on average 3 times that of above ground biomass.

Mean above and below ground biomass (g DW/m<sup>2</sup>) was determined at four sites for *T. hemprichii* (Figure 214B). Mean above (54.2 g DW/m<sup>2</sup> ± 1.24 s.e.) and below (224.5 g DW/m<sup>2</sup> ± 5.2) ground biomass was greatest at site 005, with below ground biomass approximately 6-times greater than that of sites SGTC1 and SGTH4. Site SGTC2 recorded both the lowest above (11.8 g DW/m<sup>2</sup> ± 5.1) and below ground biomass (7.8 g DW/m<sup>2</sup> ± 3.4). Above and below ground biomass ratio was 1:2.5, with below ground biomass being on average 2.5 times that of above ground biomass. The average blade length and width of *T. hemprichii* was 37.6 mm (± 1.1 s.e.) and 4.5 mm (± 0.2) respectively.

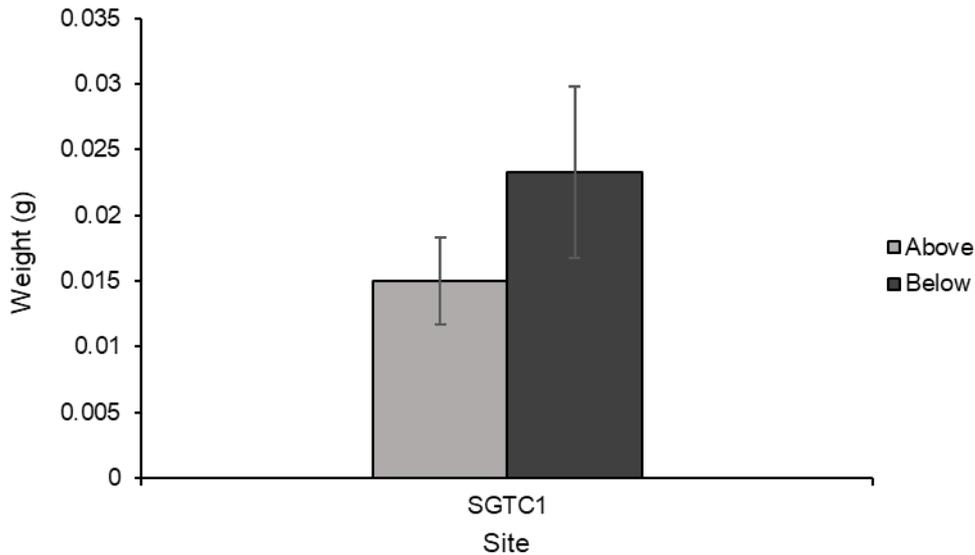


Figure 214. Above and below ground biomass (leaf/sheath and root/rhizome weight (g DW)) for *H. ovalis*. Error bars indicate  $\pm 1$  s.e. Dry to wet weight conversion = 7.08 (leaf) and 5.86 (root/rhizome) for *H. ovalis*.

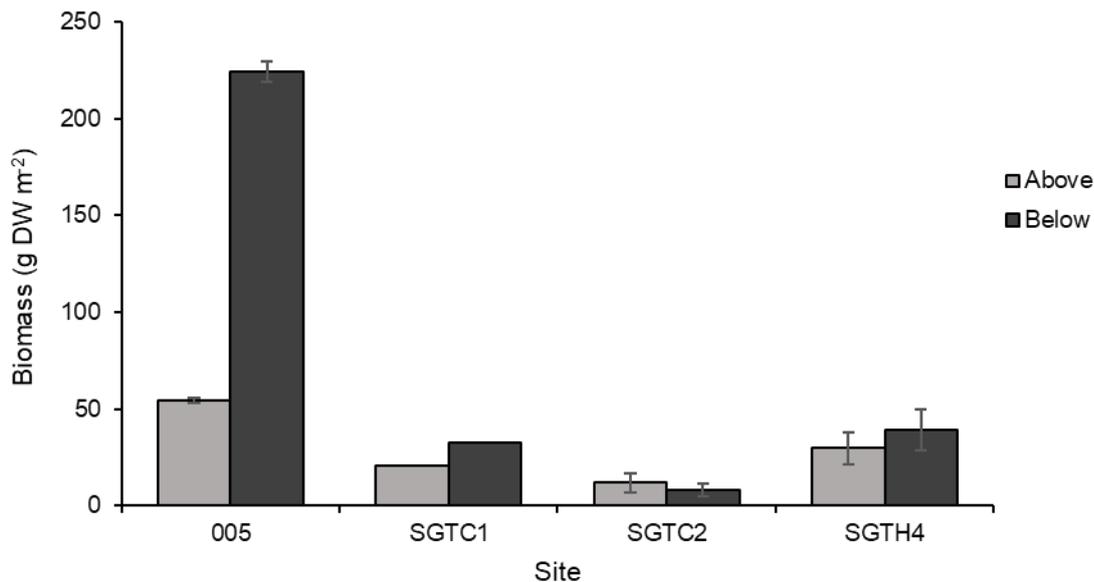


Figure 215. Above and below ground biomass (leaf/sheath and root/rhizome weight (g DW/m<sup>2</sup>)) of *T. hemprichii*. Error bars indicate  $\pm 1$  s.e. Dry to wet weight conversion = 9.56 (leaf) and 4.63 (root/rhizome).

### 13.4.3 *Thalassia hemprichii* productivity

Productivity and growth measurements were taken at two sites in 2019. Both leaf and shoot growth were greater at site SGTC1 than SGTC2. Mean leaf growth at site SGTC1 was measured at 2.4 mm/day ( $\pm 0.1$  s.e.) while at site SGTC2 mean growth was 2.2 mm/day ( $\pm 0.3$  s.e.) (Figure 216A).

Shoot growth (each comprising multiple leaves) was slightly higher than leaf growth at both sites (Figure 216B). Mean shoot growth at site SGTC1 was 3.5 mm/day ( $\pm 0.3$  s.e.) while at SGTC2 shoot growth was slightly lower at 3.2 mm/day ( $\pm 0.6$  s.e.).

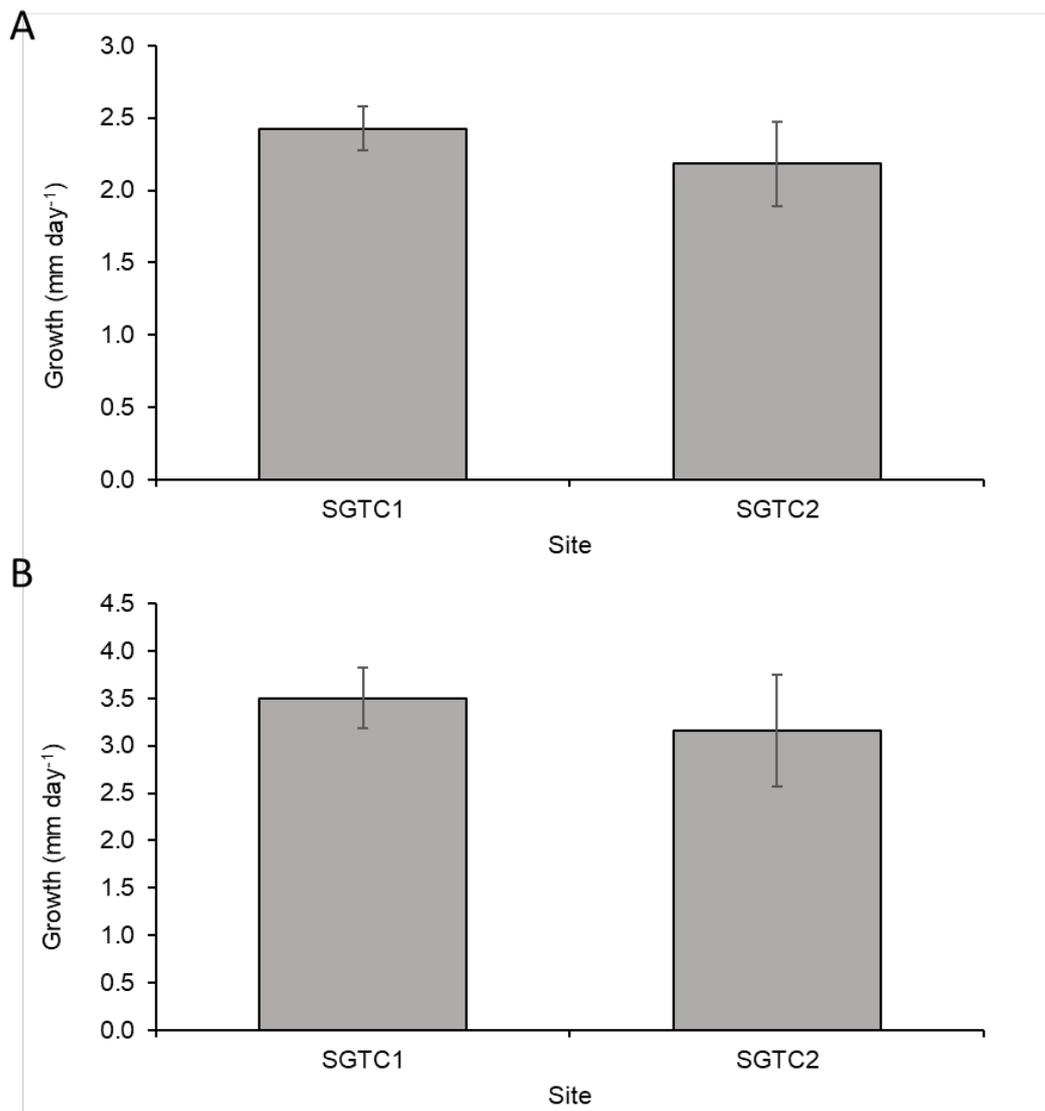


Figure 216. Mean growth (mm/day) per site for *T. hemprichii*: A) mean individual leaf growth, B) mean shoot growth (each comprising multiple leaves). Note: shoot growth does not include new leaves or leaves broken/grazed below punched hole. Error bars indicate  $\pm 1$  s.e.

Canopy height increased by 5.7%/day at site SGTC1 while this increase was slightly lower at site SGTC2 (5.3%). The turnover rate for *T. hemprichii* was calculated to 17.5 days for site SGTC1 and 21.3 days for site SGTC2.

Mean biomass of emergent leaves was calculated to be 8 g DW/m<sup>2</sup>. With an average turnover rate of 19.4 days, it was possible to determine that approximately 0.4 g DW/m<sup>2</sup> is grazed each day or 123 kg DW/ha each month. Using a consumption weight of 150 g DW/day (Brown & Skewes 2005), a turtle would need to graze 18.8 m<sup>2</sup> each day in order to obtain its daily seagrass intake.

## 13.5 Discussion

### 13.5.1 Shoot and biomass monitoring

The seagrass beds of Ashmore Reef are considered extensive by comparison with adjacent reef systems, e.g. Browse, Cartier, Hibernia, Scott and Seringapatam Reefs (Pike & Leach 1997; Skewes *et al.* 1999b; Willan 2005). Consisting predominantly of *Thalassia hemprichii* and to a far lesser extent *Halophila ovalis*, the number of species present at Ashmore Reef is relatively low compared with the Kimberley to the south (Huisman & Sampey 2014; Kendrick *et al.* 2017) and coastal areas of Timor to the north (Pike & Leach 1997). While previous surveys have identified five different species of seagrass at Ashmore Reef - *T. hemprichii*, *Thalassodendron ciliatum*, *H. ovalis*, *H. decipiens* and *Halodule pinifolia* (Pike & Leach 1997), more recent surveys have not detected the latter two (Skewes *et al.* 1999b; Brown & Skewes 2005). Of these, *T. hemprichii* has been identified as the most dominant species at Ashmore Reef, which, in 1999 was found to comprise over 85% of total seagrass cover, although sparsely distributed (Skewes *et al.* 1999b). *Halophila ovalis* was also recorded in 2019 however only comprising a very small portion of seagrass composition and, like previous surveys, is not considered a major component of seagrass beds at Ashmore Reef (Pike & Leach 1997). While *T. hemprichii* is a common seagrass on shallow reef flats (Skewes *et al.* 1999b), *H. ovalis* is only found in small isolated patches amongst *T. hemprichii* and has been discovered on the deeper sandy reef slopes in water up to 20 m deep (Pike & Leach 1997). Seagrass monitoring in 2019 occurred predominantly on the reef flat. Therefore, it is possible *H. ovalis* was present in deeper areas at Ashmore Reef. However, photo quadrats during invertebrate and fish surveys failed to identify *H. ovalis* in these areas.

Seagrass has been identified as the main dietary component of dugong (*Dugong dugon*) and green turtle (*Chelonia mydas*) (Pike & Leach 1997). A small population of dugong (10-60 individuals) feed on the seagrasses at Ashmore Reef (Whiting & Guinea 2005b). Although known to feed on all species of seagrass, dugongs preferentially feed on *Halodule* sp. and *Halophila* sp. due to the high nitrogen and low fibre content (Erftemeijer, Djunarlin & Moka 1993b). Ashmore Reef has also been recognised as a globally significant site for grazing turtles (Brown & Skewes 2005) with an estimated 10,000 green turtles feeding predominantly on *T. hemprichii* (Whiting & Guinea 2005b). In 2019, *T. hemprichii* mean shoot density ranged from 88.9 to 600 shoots/m<sup>2</sup> across sites. Although overall mean density of 407 shoots/m<sup>2</sup> was lower than the 605 shoots/m<sup>2</sup> recorded by Brown and Skewes (2005), shoot densities at two of the four sites surveyed in 2019 were similar to those previously recorded at Ashmore Reef (Brown & Skewes 2005) and within the range of 40–1,200 shoots/m<sup>2</sup> recorded in the Kimberley (Kendrick *et al.* 2017). Using measurements of density and biomass, the dietary requirements of an adolescent turtle weighing 50 kg has previously been calculated to be 150 g DW/day of seagrass, the equivalent of grazing 3.75 m<sup>2</sup> (Brown & Skewes 2005). While based on similar density values, biomass calculations in 2019 showed that 150 g DW/day would be the equivalent of 18.8 m<sup>2</sup> – approximately five times that found in 2005. Although these values vary considerably, canopy height used in 2019 was based on blade height above the substrate and did not factor in shoot material buried below the surface of the sand. When complete above ground biomass values of 34–36 g DW were used for the same calculations, the resulting values were comparable. Interestingly, substantially longer leaves were observed near and between rocks and coral, possibly suggesting that these shoots may not be accessible to grazing turtles.

(Brown 2001) identified three habitat environments provided by seagrasses. The presence of roots and rhizomes provide a stable infaunal habitat while the increased organic matter, nutrient uptake by seagrass roots, presence of nitrogen-fixing and sulphate-reducing bacteria, and oxygen release from roots produce a favourable environment for infaunal organisms (Brown & Skewes 2005). The presence of shorebirds, that feed amongst seagrass beds at low tide, may be indirect evidence of the importance of such infaunal organisms and the role of seagrass as habitat providers (Brown & Skewes 2005). Secondly, the blade surface provides habitat to algal and calcareous epiphytes, both vitally important in marine ecosystems for contributing to total plant biomass and sediment respectively (Smith 1972; Harlin 1980). Third, seagrass has been reported to modify tidal currents and wave energy in shallow water, subsequently providing protection between plants (Fonseca *et al.* 1982; Ward, Michael Kemp & Boynton 1984; Fonseca & Cahalan 1992) for organisms including holothurians, fish, gastropods and bivalve species (Skewes *et al.* 1999b; Brown 2001). Seagrass cover decreases physical stress on the sediment-water interface by redirecting water flow and decreasing current velocity as blades and the shoot canopy bend (Fonseca *et al.* 1982). This bending also protects the plant by reducing the exposed shoot surface area, ultimately reducing drag forces and internal stresses on the blade (Fonseca *et al.* 1982).

Yet, the protection afforded between plants may be restricted at Ashmore Reef due to reduced leaf length. Mean canopy height for *T. hemprichii* was 42.8 mm across all sites while average blade length and width was 37.6 mm and 4.5 mm respectively. Although substantially shorter and narrower than the taxonomic description of 100–300 mm blade length and 5–10 mm width (Pike & Leach 1997), these measurements are consistent with those of Brown and Skewes (2005) who found blade length and width to be 40 mm x 3 mm. They suggested *T. hemprichii* at Ashmore Reef may be considered relatively stunted (Brown and Skewes 2005). This notion was also suggested by Pike and Leach (1997) who reported seagrass beds within the reef as patchy and stunted, possibly due to shifting unconsolidated sand resulting in the progressive burial and establishment of seagrass beds.

An average biomass of 116.9 g DW/m<sup>2</sup> (range 5.1 to 290.4 g DW/m<sup>2</sup>) was obtained for *T. hemprichii* across all sites. A mean above ground biomass of 32.2 g DW/m<sup>2</sup> was recorded while below ground biomass was 85.6 g DW/m<sup>2</sup>, on average 2.5 times that of above ground biomass and similar to that recorded by Duarte and Chiscano (1999). For *H. ovalis*, above ground to below ground biomass ratio was 1:3. These values vary somewhat from previous above to below ground ratios for *T. hemprichii* at Ashmore Reef of 1:5 (Brown 2001). Brown and Skewes (2005) suggested such an above ground to below ground biomass ratio may indicate nutrient deficiency in seagrasses at Ashmore Reef. *Thalassia hemprichii* has been reported to respond to reduced nutrient levels by increasing below ground biomass (Erftemeijer & Herman 1994; Agawin, Duarte & Fortes 1996). However, unless grazing rates of the above ground standing stock are taken into account, it is difficult to compare across studies and nutrient levels in seagrass have not been studied at Ashmore Reef.

### 13.5.2 *Thalassia hemprichii* productivity

*Thalassia hemprichii* shoot growth was 3.2–3.5 mm/day while leaf growth rate was slightly lower at 2.2–2.4 mm/day. This was substantially lower than the leaf growth rates of 5–26 mm/day obtained in the Kimberley (Kendrick *et al.* 2017) and 8.4 mm/day obtained by Brouns (1985) in Papua New Guinea.

Canopy height increased by 5.3–5.7% with a turnover rate of 17.5–21.3 days. Rate of leaf growth has been suggested to be dependent on the number of days following emergence of new leaves. Brouns (1985) observed constant rapid growth during the first 13 days, before a subsequent decrease then cessation of growth within a few days following new leaf emergence on the shoot (after approximately 24 days). However, Kendrick *et al.* (2017) found that *T. hemprichii* was seasonal with the highest productivity occurring between August and February. As such, lower productivity observed in 2019 may be due to the timing of surveys (conducted in June), possibly coinciding with a period of slower growth.

The combination of biomass and productivity measurements for *T. hemprichii* may indicate that the conditions for growth may not be ideal at Ashmore Reef and may suggest this seagrass species may be growing under stress (Brown & Skewes 2005). Although tolerant to high water temperature and occasional exposure to low tides (Skewes *et al.* 1999b), *T. hemprichii*, like other *Thalassia* spp., cannot tolerate prolonged high temperatures or long-term desiccation (Brouns 1985). Aerial exposure of blades during the semi-diurnal tide present at Ashmore Reef (Glenn 2004) may result in seagrass being subjected to high temperatures above 35°C (Brown 2001, 2001). However, there is also evidence that *T. hemprichii* copes well with high levels of environmental stressors. Kendrick *et al.* (2017) found that despite the extreme tidal fluctuations and exposure temperatures which may exceed 40°C in the coastal Kimberley, seagrasses are still able to persist, grow and produce new leaf biomass. This highlights their ability to adapt and thrive across a wide range of environmental conditions.

## 13.6 Management implications and recommendations

Based on our survey results, seagrass beds at Ashmore Reef are extensive. They are mostly restricted to the reef flat habitat (present on 37 of 65 reef flat sites for which photo-transect data were collected), although cover is not universally high (10-30% cover at only 7 sites, and was higher on the western part of the reef flat than the eastern or southern sections of the reef). Nevertheless, our study has shown that *Thalassia hemprichii* was very productive and is grazed intensively. Our survey did not map the full extent of the seagrass beds and this should be a priority for future surveys. In addition, we recommend grazer exclusion experiments and remote video deployments to determine which species are most dependent on seagrass beds for food, and to refine the grazing rate and growth rate measurements made in this study. We also recommend that the four monitoring sites we established at Ashmore Reef be periodically monitored, every 3-5 years, and additionally in the event of any large perturbation such as a cyclone or warming event.

# 14 ASHMORE REEF: SEA SNAKES AND TURTLES

Ruchira Somaweera, John Keesing, Lauren Hardiman and Daniella Ceccarelli

## 14.1 Abstract

With 17 species of sea snakes recorded from Ashmore Reef, it was once considered a global hotspot of sea snakes. However, the diversity and abundance of sea snakes at Ashmore Reef collapsed dramatically since the early 2000s, for reasons still not understood. Surveys during the last five years show that after a complete absence of sea snakes, at least one species, the olive sea snake (*Aipysurus laevis*), appears to be recolonising the reef, however, numbers are still extremely low. Despite an extensive search effort using day and night boat surveys, reef walks, manta tows and 224 quantitative transects made by two divers for fish and invertebrates, only a single specimen of *A. laevis* was reported during the current survey.

In contrast, the marine turtle numbers have been consistently high at the reef since surveys started in early 1994. The current survey did not make a quantitative assessment of turtles at Ashmore Reef but we recorded two species, the hawksbill turtle (*Eretmochelys imbricata*) and the green turtle (*Chelonia mydas*), the latter in large numbers. Manta tows of approximately 26 km of the reef perimeter and lagoon edges recorded 73 green turtles and 6 hawksbill turtles.

## 14.2 Introduction

### 14.2.1 Sea snakes

Some 71 species of sea snakes inhabit tropical and subtropical waters globally (Rasmussen *et al.* 2011), reaching the highest species diversity in the tropical coastal waters of Australia and the Indo-Malay region (Cogger 2000; Elfes *et al.* 2013). Within this region, the Timor Sea, extending from the southern coast of Timor to Australia's north-west coast, was widely considered the sea snake biodiversity hotspot, harbouring 17 species, which is ca. 24% of all sea snake species (Minton & Heatwole 1975; Guinea & Whiting 2005).

Historical records suggest that sea snakes were once abundant at Ashmore Reef (see Table 29). All 17 species of sea snakes hereto reported from the Timor Sea have been recorded at Ashmore Reef, nine as breeding residents and the rest as vagrants (Minton & Heatwole 1975; Cogger 2000). In 1926, Dr Malcolm Smith who had negotiated with Malay navigators to collect reptile species throughout South-East Asia, obtained 100 specimens of sea snakes belonging to five species from Ashmore Reef (Smith 1926). His collectors indicated that many more specimens could have been obtained easily. Later, in 1973, researchers from the RV *Alpha Helix* from Scripps Institute collected more than 350 sea snakes from nine species in less than two weeks at Ashmore Reef noting that 'many more were observed' (Minton & Heatwole 1975). Subsequent surveys in the 1990s further supported the theory that Ashmore Reef supports the greatest diversity of sea snakes in the world (Guinea & Whiting 2005; Guinea 2007), with an estimated standing stock of almost 40,000 sea snakes on the 174 km<sup>2</sup> reef flat (Guinea & Whiting 2005). Mark and recapture studies over three

years indicated between 94 and 192 turtle-headed sea snakes (*Emydocephalus annulatus*) alone frequented a single coral head 30 m in diameter (Guinea & Whiting 2005).

However, surveys conducted from 2005 onwards noticed a substantial decline in sea snakes diversity and numbers (Lukoschek *et al.* 2013). This decline in sea snake numbers to below the level of detection was unprecedented and unexplained. The system has been surveyed using a wide array of methods including boat surveys, SCUBA, manta tows, snorkelling surveys, reef walks and night-time spotlighting. Despite these differences in survey methods, the numbers of sea snakes recorded at Ashmore Reef declined from 46 snakes/day during the first year of survey in 1973 to less than half of that (21 snakes/day) three decades later in 2002 and then were completely absent by 2013 (Table 29). This disappearance was not uniform across all species. Specialist feeders such as horned sea snake (*Hydrophis peronii*) that only feed on longiformes fish on the sand flats and in the lagoon were among the first to disappear, followed by turtle-headed sea snake (*E. annulatus*) that feed on fish eggs. Generalist feeders such as the olive sea snake (*Aipysurus laevis*) that have a wide diet, were the last to disappear (Guinea 2013). However, in 2016, three olive sea snakes were recorded in baited camera traps set west of Ashmore Reef (Conrad Speed, pers. comm.). In 2017, a 10-day survey detected four olive sea snakes at the extreme south-east outer reef and another one in the West Island channel, and later the same The most recent survey by the University of Tasmania in 2018 did not record any sea snakes within Ashmore Reef, while numerous specimens from three species were observed at nearby Scott, Seringapatam, Hibernia and Cartier reefs during the same survey (Graham Edgar, pers. comm.).

**Table 29. Scientific survey history and sea snake abundance at the inner reef at Ashmore Reef Marine Park. Sea snake sighting rates were reported as either per hour or per day or both.**

Year	Sea snake sighting rate (per hour)	Sea snake sighting rate (per day)	Reference
1973		46.0	Minton and Heatwole (1975)
1994	16.3	42.0	Guinea and Whiting (2005)
1996	26.5		Guinea and Whiting (2005)
1998	60		Guinea and Whiting (2005)
1999	8		Guinea (2007)
2000	21		Guinea (2007)
2002		21	Lukoschek <i>et al.</i> (2013)
2003	6.9		Guinea (2007)
2004	0.3		Udyawer and Heupel (2017)
2004	1.5		Guinea (2006)

2005	1.6		Guinea (2006)
2005		4	Lukoschek <i>et al.</i> (2013)
2006	0.9		Guinea (2007)
2006	2.6	7	Lukoschek <i>et al.</i> (2013)
2007	0.2	1	Guinea (2007)
2008	0.2		Guinea (2007)
2009		2	Lukoschek <i>et al.</i> (2013)
2010		3	Lukoschek <i>et al.</i> (2013)
2013	0		Guinea (2013)
2017		0.5	Guinea and Mason (2017)
2018	0		Edgar and Stuart-Smith (2018)

### 14.2.2 Turtles

Ashmore Reef is an internationally important region for foraging and nesting turtles (Whiting & Guinea 2005a; Guinea 2013; Guinea & Mason 2017). Of the six species of marine turtles known from the Indian Ocean, three have been recorded nesting at Ashmore Reef. The most abundant of them, the green turtle (*Chelonia mydas*) that mainly nest on the West Island (sometimes in large numbers: Whiting and Guinea 2005), while the hawksbill turtle nests on West, Middle and East islands. Loggerhead turtles feed on the reef flat and there is also a record of this species nesting on the West Island (Guinea 2013). The reefs at Ashmore were estimated to support over 10,600 green turtles (Guinea & Whiting 2005) and studies by (Dethmers *et al.* 2006) showed that the those nesting at Ashmore Reef form a distinct management unit alone with those nesting on Cartier Island.

Turtle nesting on the islands of Ashmore Reef has been reported since the early 1950s (Serventy 1952a). However, monitoring of the population only started in early 1980s and was conducted on an ad-hoc basis until the early 1990s (Guinea 2013). Standardised assessment of turtles was only initiated in 1994 (Whiting & Guinea 2005a). While the surveys for turtles at Ashmore has varied in methods and timing from year to year, in general, they have consistently reported large numbers of green turtles.

The seagrass and algae habitats on the reef flats are critically important to sustain Ashmore Reef's population of turtles (Brown & Skewes 2005). Any decline in these resource habitats will have a detrimental impact on turtle populations. There has also been evidence of non-native species (tropical fire ants) impacting the populations of turtles through nesting interference (Guinea 2013).

The last survey of turtle populations at Ashmore Reef was in 2017 (Guinea & Mason 2017). Other surveys have included tagging nesting green and foraging sub-adult green turtles from 1987 to

2004 (Whiting & Guinea 2005b). Our study did not involve a formal assessment of turtles, however they were recorded during manta tow surveys made to survey sea snakes.

### 14.3 Objectives

To determine the abundance and diversity of sea snakes around Ashmore Reef and record observations of turtles made at the same time.

### 14.4 Methods

Multiple survey methods outlined below were used to detect, count and identify sea snakes during two voyages undertaken from 1-7 May 2019 and 12–24 June 2019. Counts of turtles were only made on the second voyage.

#### 14.4.1 Boat surveys

Daytime surveys of the reef flat were conducted over 5 days at high tide in May 2019. Two to four observers per tender boat actively looked for sea snakes while travelling at approximately 4 knots over the reef flat. Night-time spotlighting for sea snakes took place for 2 h in the western channel on 3 May, and for 3.5 h on 4 May in the eastern lagoon. A spotter and a skipper travelled at ~4 knots while scanning the surface for snakes using handheld spotlights.

#### 14.4.2 Reef walks

Surveys on foot of the intertidal flat north and south of West Island once exposed at low tide were conducted to locate sea snakes in May 2019. A total of 2.1 person hours was spent at the exposed reef on 3 May.

#### 14.4.3 Manta tow and fish/invertebrate transects

Sea snakes and turtles were surveyed by manta towing large sections of the reef perimeter and lagoon areas (Figure 217; see chapter 8.3.4 for detail of manta tow methods). Any snakes and turtles observed in each two-minute tow were identified and counted. Sea snakes were also surveyed during fish/invertebrate transects (even if observed off transect). Any observed specimens were counted and identified. Incidental observations of sea snakes from the survey vessels were also recorded.

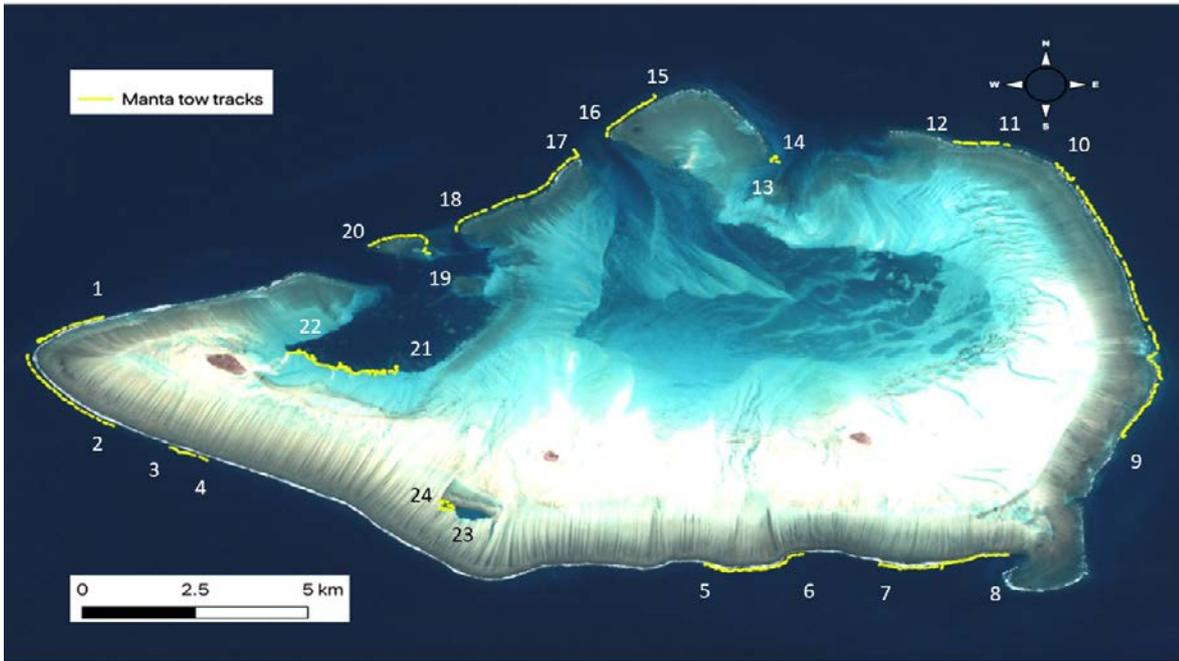


Figure 217. Manta tow transects undertaken for sea snake surveys at Ashmore Reef in 2019.

## 14.5 Results

### 14.5.1 Sea snakes

No sea snakes were reported during the May 2019 survey. No sea snakes were recorded in any of the manta tow surveys during the June 2019 voyage. However, one olive sea snake *A. laevis* was observed from the diving tender on the south-eastern corner perimeter of the reef near the feature known as “the Hook” at Ashmore Reef.

### 14.5.2 Turtles

Although no standard surveys were conducted in May 2019, numerous green turtles and hawksbill turtles were opportunistically observed during sea snake surveys. A large aggregation of close to 100 turtles were observed near West Island around 1000 h on the 3 of May (Figure 218).

During the June 2019 manta tow survey, a total of 73 green turtles (*Chelonia mydas*) (Figure 219) and 6 hawksbill turtles (*Eretmochelys imbricata*) were observed (Figure 220).



Figure 218. Large aggregation of green turtles (*Chelonia mydas*) near West Island on 3 May 2019 Photo: Tommaso Jucker.

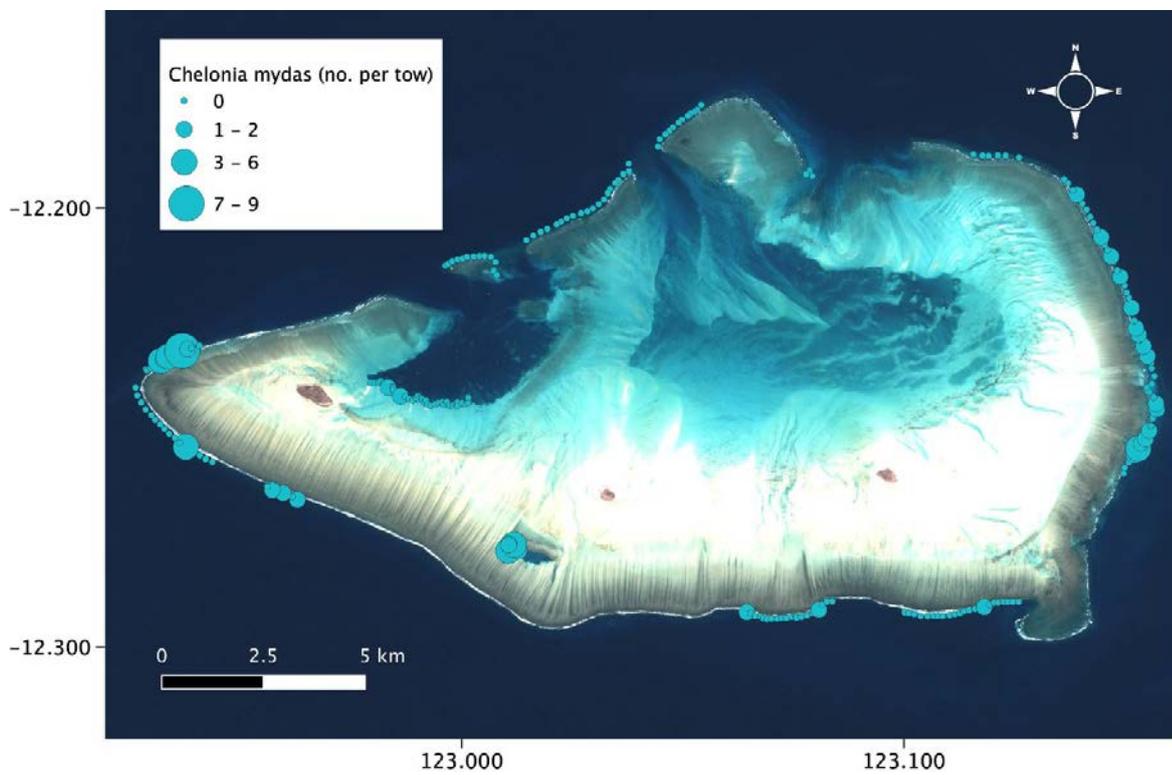


Figure 219. Map of Ashmore Reef showing the number of green turtles (*Chelonia mydas*) per 2-minute manta tow.

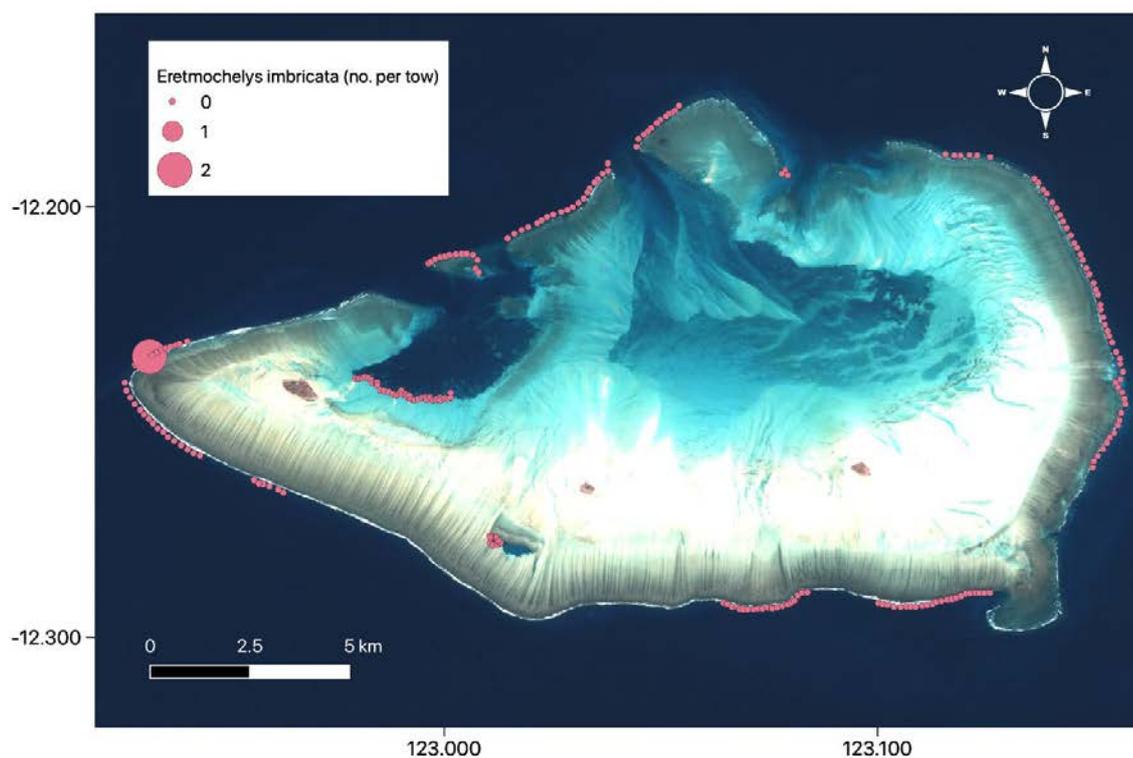


Figure 220. Map of Ashmore Reef showing the number of hawksbill turtles (*Eretmochelys imbricata*) per 2-minute manta tow

## 14.6 Discussion, management implications and recommendations

### 14.6.1 Sea snakes

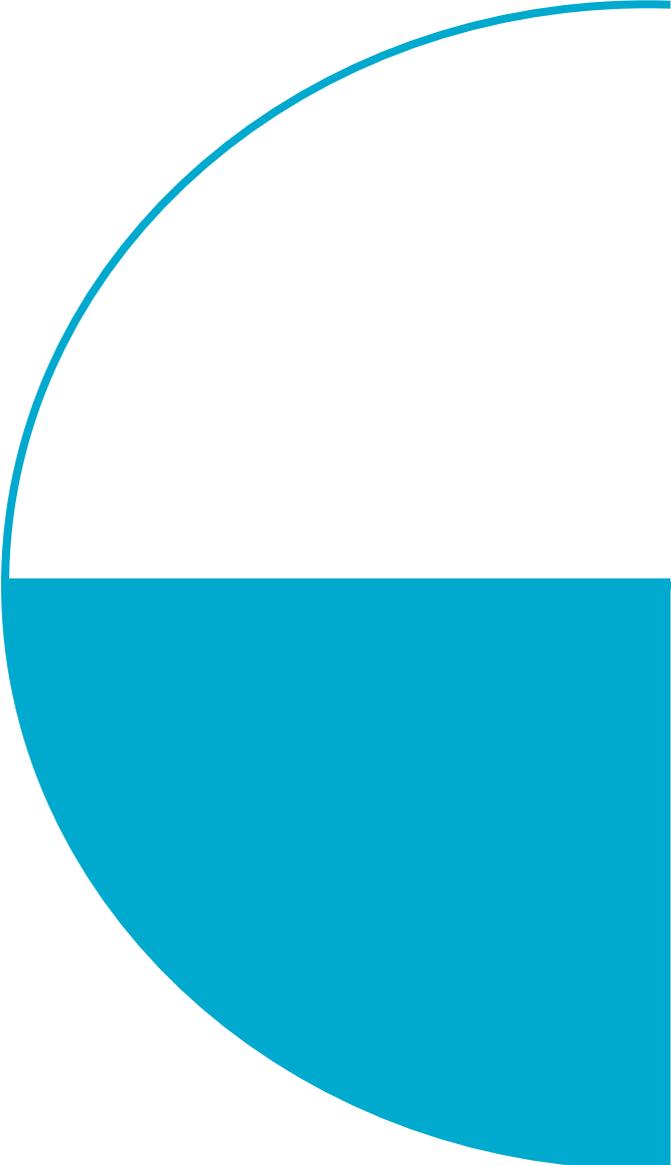
Despite a significant search effort, only one individual sea snake was recorded during the surveys, further confirming that the population size at Ashmore Reef remains extremely low. However, records of this species also further indicate that recolonization of the reef could be underway. No explanation exists as to the declines of sea snakes that took place in early 2000s at Ashmore Reef (Lukoschek *et al.* 2013). However it is possible that a trophic cascade and/or physical impacts from increasing presence of vessels, and/or a pathogen may have contributed to the decline (Somaweera *et al.* 2021). Having a better understanding of the pathways of impact at Ashmore Reef would be important to detect, identify and mitigate future declines in other regions too. To that end, we propose the design and implementation of a standardised surveying and monitoring protocol for sea snakes at Ashmore and selected reference sites, where monitoring can be repeated at least once every two years. This two-year survey period is due to the significant decline of sea snakes Ashmore Reef and therefore require more frequent monitoring to detect any patterns in recovery. There is also a need for establishment and deployment of data loggers to monitor abiotic conditions at the reefs.

### **14.6.2 Turtles**

Our surveys did not constitute a quantitative assessment of turtles and the last such survey was conducted relatively recently in 2017 (Guinea & Mason 2017). However, we recommend some additional work on the interaction between green turtle populations and seagrass, including mapping the extent of seagrass beds and grazer exclusion experiments to quantify the importance of the seagrass beds to turtles at Ashmore Reef (see chapter on seagrass for more detail).

### **14.6.3 Other observations**

Although we did not survey plastics and other marine debris quantitatively, a large amount of plastic pollution was observed floating in the water during in-water surveys. This comprised mostly of soft plastic food packaging labelled as made in Indonesia, and was considered at levels substantial enough to cause concern if present anywhere on the Australian mainland coastline. We did not observe any negative interactions between plastic and marine life but recommend its potential impacts on turtles, birds and other biota should be assessed (Section 1.1).



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