Australia's National
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Ashmore Reef Marine Park Environmental Assessment

Final report to Parks Australia

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The data used in the production of this report can be located at: CSIRO Data Access Portal

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Important disclaimer

Cover image: Survey work on the outer reef at Ashmore Reef Marine Park. Credit: Lauren Hardiman

Contents

Figures

Tables

Supplementary Information Figures

Supplementary Information Tables

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Glossary

AGL	Above Ground Level.
AI/ML	Artificial intelligence and machine learning.
AREAL	Relating to an area, which can be an expanse of space or region.
ARMP	Ashmore Reef Marine Park.
CAD	Critical Approach Distance.
Casual non-native	Non-native plants that may flourish and even reproduce occasionally outside cultivation in an area, but that eventually die out because they do not form self-replacing populations, relying on repeated introductions for their persistence.
CHM	Canopy height model.
DEM	Digital elevation model.
DEWHA	Department of the Environment, Water, Heritage and the Arts.
dGPS	Differential global positioning system, which uses differential correction techniques, either in real time or when post processing data, to provide increased geospatial quality (< 0.5m accuracy) for location data.
eDNA	Environmental DNA - enables the detection of species from environmental samples through analysis of the DNA released by organisms into their environment.
GPS	Global positioning system
GRTS	Generalized random tessellation stratified
Invasive	A population that is rapidly expanding its range. Both native and non-native populations can be invasive. There is no implication of harm or threat associated with an invasive population, whether that be to environmental, agricultural or other social values. Invasive plants are a subset of naturalised plants that produce reproductive offspring, often in very large numbers, at considerable distances from the parent plants, and thus have the potential to spread over a large area.
LiDAR	Light Detection and Ranging
Manta Tow	The manta tow technique is used to provide a general description of large areas of reef and to gauge broad changes in abundance and distribution of organisms on coral reefs.
Native	A population occurring within an appropriate distance from the species region of origin, allowing for natural dispersal potential in a given time frame (Webber and Scott 2012).
Naturalised	Non-native plants that sustain self-replacing populations without direct intervention by people (or in spite of human intervention) by recruitment from seed or other propagules (e.g. tillers, tubers, bulbs, fragments).
Non-native	A population occurring outside its natural past or present range and dispersal potential in the timeframe under consideration, its presence being due to human actions, either deliberate or accidental (Richardson, Pyšek & Carlton 2011)et al. 2011, Webber and Scott 2012). Also known as alien or exotic.
POWO	Plants of the World Online website: http://www.plantsoftheworldonline.org/
Ramsar	The Ramsar Convention of International Importance especially as Waterfowl Habitat is an international treaty for the conservation and sustainable use of wetlands.
RGB	Red, green and blue.
RNA Interference	A biological process in which RNA molecules inhibit gene expression or translation, by neutralizing targeted mRNA molecules.
RPAS	Remotely Piloted Aircraft Systems. Also known as drone or UAV.
RUV	Remote Underwater Video.

Transect-MeasureTM (SEAGIS)	*TransectMeasure* is a program used for the analysis of percentage cover, and measurement of lengths and areas from digital imagery. It is widely used for the analysis of transects or biological sampling from images.
Transformers	A subset of invasive species that have impact to the extent that they change the character, condition, form or nature of ecosystems over a substantial area relative to the extent of that ecosystem (Richardson et al. 2000).
UAV	Unmanned Aerial Vehicle. Also known as RPAS or drone.
UVC	Underwater Visual Census.
Weed	A plant not wanted where it is found. An entirely subjective determination based on value systems within a human context. Weeds usually have detectable economic or environmental effects and are often both non-native and invasive, but this is not always the case (Richardson *et al.* 2000).

EXECUTIVE SUMMARY

1.1 Purpose of this report

Parks Australia commissioned CSIRO to undertake a survey of selected marine and terrestrial environmental values at Ashmore Reef Marine Park. The field work was undertaken on two trips, 01 – 07 May and 14- 22 June 2019. This report presents the findings of that work.

For the terrestrial component, the required elements were to quantify the diversity and abundance of native and non-native plant species, determine the population status and diversity of seabirds and shorebirds on the islands, and to determine the abundance of non-native tropical fire ants. The remit included any evidence of interactions between the tropical fire ants and native fauna, especially birds and turtles. In addition to these required components, non-native geckos and hermit crabs were surveyed for abundance, the latter being native fauna which would likely be most affected by future attempts to eradicate tropical fire ants. Any work on rodents was not included in the project scope.

For the marine component, the required elements were to survey coral reef communities in particular abundance of key invertebrates which have historically been the subject of illegal fishing and over-exploitation (holothurians, trochus and tridacnid clams), the status (cover and health) of key habitat forming benthic assemblages (corals, macroalgae and seagrass) and evidence of sea snakes. In addition, surveys of fish and shark diversity and abundance were undertaken, and monitoring sites were established for seagrass biomass and productivity measurements. Incidental observations of turtle abundance were also made, and water samples collected from a subset of sites for potential future analysis of eDNA.

1.2 Condition statement and key findings

This section provides a set of brief statements about the state of the terrestrial and marine environments at Ashmore Reef Marine Park. These statements summarise key findings and where possible any trends in status, diversity or abundance. More detailed summaries that cover each of the types of biota studied are given later in Section 1.3. Recommendations about gaps or priorities that need addressing, suggested changes to survey methods and the frequency of resurvey are made in Section 1.2.1.

- Island structure – The four islands (East, Middle and West islands and Splittgerber Cay) at Ashmore Reef occupy a total of 56.3 ha with between 2.0 and 0.9m of vertical relief above the shoreline. Physical scars from previous guano mining and other anthropogenic land use still mark West, Middle and East Islands and appear to be a stabilising factor for vegetation communities.

- Terrestrial plants (native) – A total of 20 native plant species, including one new record (a native *Pandanus*) for the islands, were observed across the four islands, as well as up to five *Boerhavia* that were not identified to species level. Together with non-native species, these formed 35 distinct vegetation communities. Tree and large shrubs are dying and their health has decreased markedly. The 2019 survey represents the first detailed quantification of plant abundance for the Ashmore islands, providing a robust baseline for establishing future management success.

- Terrestrial plants (non-native) – A total of eight non-native plant species were recorded from the four islands, including four *Cenchrus* species (*C. brownii*, burr grass; *C. ciliaris*, buffel grass; *C. echinatus*, innocent weed; *C. pedicellatus*, annual mission grass), and an increasing abundance of *Tribulus cistoides* (beach caltrop) and *Cleome gynandra* (spiderwisp). All *Cocos nucifera* (coconut palm) individuals have now died at Ashmore Reef. Past efforts involving infrequent weeding to remove the *Cenchrus* species have been ineffective. This survey has established that there is 1,200 m^2 of *Cenchrus* plants to control.

- Seabirds – Seventeen seabird and two egret and one heron species were recorded on the islands and cays. Twelve seabird species had begun breeding in May 2019. Numbers increased for most species and five species expanded their breeding territories including to other islands than had been previously observed. Overall, the Ashmore Reef Marine Park continues to support significant numbers of seabirds that are generally increasing.

- Shorebirds - Of the 30 species of migratory shorebird known to occur at Ashmore, 17 species were observed overwintering in May 2019.

- Tropical fire ants (non-native) – Tropical fire ants (*Solenopsis geminata*) are present on East, Middle and West Island with distribution and abundance variable. The abundance on Middle Island which had undergone chemical treatments to kill the ant in 2011-12, demonstrated that there is no long-term suppression of the ant in the absence of eradication. No evidence of interactions with bird or turtle nesting was observed and it is recommended that further eradication attempts await the development of RNA interference (RNAi) approaches.

- Asian house gecko – The Asian house gecko (*Hemidactylus frenatus*; non-native) is abundant on West Island. Two nights of surveys found 89 individuals. The potential ecological impact of geckos at Ashmore requires further investigation.
- Coral cover – Coral cover was highly variable across habitats surveyed but was very high on some sections of the southern edge (mean cover range 14 to 70%). Assemblages appeared normal and no coral bleaching or disease, nor signs of stress or major recent disturbance was observed. The nature of previous surveys makes it difficult to be definitive about trends, but 2019 levels of coral cover are believed to be higher than they were in 2005 but are also likely to have been affected by bleaching in 2010 and 2016/17. The 2019 survey represents a sound quantitative baseline for future monitoring.
- Seagrass – Seagrass beds are very widespread on sections of the partially sand covered reef but cover is generally low (10-30% across 7 sites examined). *Thalassia hemprichii* was the dominant species with wide distribution, especially on the western section of the reef flat, although mostly sparse. The surveys indicate seagrass beds are heavily grazed and are a critical habitat despite their sparse nature.

- Holothurians (sea cucumber) – Eighteen species of sea cucumber were recorded. With the exception of three asexually reproducing species, these were in low densities (many <2 per hectare) and have not recovered from heavy exploitation in the mid-1980s. At least one species, *Holothuria lessoni* is locally extinct, and other species that were fished heavily in the past are thought to be so sparsely distributed so as to have reproductively ineffective populations. Deep water habitats (>10 m) in the lagoon and around the reef remain inadequately surveyed.
- Trochus – Trochus abundance was lower than 2013 records, but similar to surveys in 2005 and 2006. The overall trend suggests numbers have increased since 1998. Patchy distribution and high mobility of trochus make temporal comparisons difficult, but Trochus are biologically well-adapted to recover from overfishing.
- Giant clams - Prior to the 2019 survey, the giant clam, *Tridacna gigas* had not been recorded at Ashmore Reef since 2006, although it, and another smaller species, *Hippopus hippopus*, were observed at low densities indicating slow recovery from overfishing in the mid-1980s.
- Fish – A total of 365 species of fish species were observed with the highest densities (>4,900 per hectare) counted on the reef slope and in the western lagoon. Sharks were rare (four species, <1.8 per ha) with a maximum density of 60 sharks per hectare on the reef slope. These densities are consistent with previous studies and represent a very extensive quantitative baseline for future surveys.
- Sea snakes – Of the 17 species previously known to occur at Ashmore, only one olive sea snake *Aipysurus laevis* was observed, despite an extensive search in this study. The cause(s) of the very significant decline in sea snakes at Ashmore Reef between 2002 and 2013 remain unknown and frequent resurvey (i.e. every two years) is warranted to detect if and when recovery occurs.
- Turtles - A quantitative assessment of turtles was not undertaken as part of this study, but the green turtle (*Chelonia mydas*) was present in large numbers. Hawksbill turtles (*Eretmochelys imbricata*) were also present.
- Dugong – A quantitative assessment of dugong was not undertaken, but observations were to be recorded during on water surveys. No dugongs were observed in 15 days of small vessel operations during this study.
- Other – Although not quantitatively surveyed, a large amount of plastic pollution was observed during in water surveys, suggesting that as a known potential threat to marine biota and birds that this should be further investigated.

1.2.1 Summary of key recommendations

The table below provides recommendations for highest priority management actions based on project findings.

Taxa or environmental value	Frequency	Last delivered	Parameters & outputs	Recommendations (including changes to/comments on methods)	Report section
Island topography	5 years	2019	High resolution shoreline, DEM and CHM layers detailing island shape, size and relief, and vegetation canopy height.	Use same methods as in this study.	3.6
Terrestrial vegetation	Immediately & regularly maintained	Never	Unified GIS data asset for all past terrestrial data	Assemble all past data into a unified GIS data asset	3.6
	Immediately, once off with annual updates	Never	Terrestrial ecosystem management plan	Analyse past data and establish ecosystem interaction networks to inform prioritisation, sequence and timing of control and restoration activities	
	Intra- and inter-annually	2019 (in part)	Vegetation survey (presence, abundance), species interaction network validation	Same survey methods as in this study. Species interaction networks established by field sensor array and targeted trapping.	
	Intra- and inter-annually	Never	Coordinated non-native species control program	Implement recommendations as per this report, but with timing and sequence informed by the above management plan	
	Intra- and inter-annually	1999	Coordinated native species restoration program	Implement recommendations as per this report, but with timing and sequence informed by the above management plan	
Seabirds and shorebirds	Intra- and inter-annually	2019	Diversity, abundance and levels of breeding success	Obtain aerial images of each island at 3-4 different time periods each year (May, August/September and November to determine timing of breeding and population estimates. Remote camera systems, AI/ML count pipelines and citizen science should be evaluated Prior to weed management, it will be necessary to clearly identify the potential impacts on breeding seabirds. A range of biological studies for all seabirds and a taxonomic study for Masked Boobies are also recommended	4.7
Tropical fire ants	Immediately prior to any fire ant control actions or eradication attempt	2019	Abundance and distribution	Use same methods as in this study. Evaluate interactions with seabirds and turtles at the same time as surveying of those animals. Consider trials of new technologies e.g. RNAi technology that specifically targets *S. geminata* on Ashmore Reef.	5.6
Hermit crabs	Prior to any fire ant management	2019	Abundance and distribution	Use same methods as in this study	7.6
Geckos	Prior to any weed/ vegetation management	2019	Abundance and distribution of geckos and ecological impact on invertebrate's via stomach content analysis	There is a need to understand the ecological impacts of geckos (e.g. predation pressure on invertebrates).	6.6

Taxa or environmental value	Frequency	Last delivered	Parameters & outputs	Recommendations (including changes to/comments on methods)	Report section
	(given their occurrence and distribution is based on vegetation)			Analysis of stomach contents would support this.	
Coral, other primary producers, sessile reef biota	5 years or in advance of anticipated coral bleaching event if previous survey >2 years	2019	Cover, extent of disease, bleaching or predation	Use same methods as in this study.	9.6
Fish	5 years	2019	Diversity, abundance and distribution	Our surveys excluded cryptic families such as gobies, blennies and moray eels, nocturnal species such as cardinal fishes and pelagic species such as mackerel and tuna, which cannot be sampled accurately by UVC. Consider the use of water sampling for eDNA analysis to complement future biodiversity assessments. We suggest these be conducted every five years.	10.6
Sharks	5 years	2019*	Diversity, abundance and distribution	A targeted survey to monitor sparsely distributed populations sharks at Ashmore Reef is recommended. *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.	10.6
Trochus	5 years	2019	Abundance and distribution	Maintain use of Hosack and Lawrence (2013a,b) survey method but increase density of southern and northern edge sites to account for potentially high levels of mobility and aggregation.	11.6, 12.6
Clams	5 years	2019	Abundance and distribution	Maintain use of Hosack and Lawrence (2013a,b) method	11.6, 12.6
Holothurians	5 years	2019	Diversity, abundance and distribution	Maintain use of Hosack and Lawrence (2013a,b) method A high priority is to extend these surveys to deep water lagoon and back reef habitats as these may represent a refuge for some high value species. The addition of reference sites at other reefs is also recommended.	11.6, 12.6
Other echinoderms and large molluscs	5 years	2019	Diversity, abundance and distribution	Maintain use of Hosack and Lawrence (2013a,b) method	11.6, 12.6
Sea snakes	2 years	2019	Diversity, abundance and distribution	Continue methods used in this study, we suggest more frequent surveys, than for other taxa, given disappearance of sea snakes since the early 2000s until the very recent occasional observations. The addition of reference sites at other reefs is also recommended.	14.6.1
Dugongs	5 years	1996	Abundance and distribution, size structure, behavioural	The last survey was more than 20 years ago. Standardised and systematic survey	n/a

Taxa or environmental value	Frequency	Last delivered	Parameters & outputs	Recommendations (including changes to/comments on methods)	Report section
			patterns relating to feeding grounds	using small vessels and RPAS (as proposed by CSIRO in this study)	
Turtles	5 years	2017	Abundance and distribution, nesting activity and hatching success	Continue use of methods used by Guinea and Mason (2017) with the addition of comparative RPAS surveys	14.6.2
Seagrass	3–5 years	2019	Cover, canopy height, productivity, grazing rates	Full extent of seagrass beds not mapped since 1998, and needs doing Grazing rate experiments not carried out in 2019 and need doing – suggest doing these (grazer exclusion experiments, remote video deployments) at same time as dugong surveys	13.6
Marine debris	5 years	Never*	Abundance of marine debris including macro- and micro-plastics on islands and cays, in marine sediments and the water column	Follow methods developed for Australian beach litter surveys by CSIRO (Hardesty *et al.* 2016) with adaptions to include marine sediments and water column.	14.6.3

*Except to survey frequency of plastic use in bird nests on West Island (Lavers et al. 2013)

1.3 Summary of findings and recommendations

1.3.1 Terrestrial vegetation

This component of the Ashmore Reef survey has delivered the most comprehensive quantitative assessment of vegetation structure, composition and abundance for all four Ashmore Reef islands (East, Middle, West islands and Splittgerber Cay). We used a combination of traditional quantitative ground surveys and aerial Remotely Piloted Aircraft Systems (RPAS) data acquisition to establish vascular plant diversity, abundance and community composition. We then compared this data to past surveys to establish trends over time. We established that the four islands at Ashmore Reef occupy a total of 56.3 ha with between 2.0 and 0.9m of vertical relief above the shoreline. Physical scars from previous guano mining and other anthropogenic land use still mark the islands and appear to be a stabilising factor in the spatial patterns of vegetation communities. A total of 28 vascular plant taxa were recorded as present and with living individuals, as well as up to five *Boerhavia* that were not identified to species level. West Island was the most speciose of the islands, with 23 species identified, compared to Middle Island (13 taxa), East Island (9 taxa) and Splittgerber Cay (3 taxa). This total included 21 native plant species (excluding the *Boerhavia* spp.) and one new record (a native *Pandanus*) for the islands. These taxa formed 35 distinct vegetation communities across the four islands. Trees and large shrubs are dying and their health has decreased markedly since the 1996-97 survey, with 72% of individuals now present either sick or dead and both *Cordia subcordata* (sea trumpet) and *Suriana maritima* (bay cedar) with very few individuals left. It appears that this decline is due to a combination of what appears to be nesting pressure from seabirds combined with shrub recruitment failure, the latter possibly due to seed predation by or competition from non-native species. A total of eight non-native plant species were recorded from the four islands, including four *Cenchrus* species (*C. brownii*, burr grass; *C. ciliaris*, buffel grass; *C. echinatus*, innocent weed; *C. pedicellatus*, annual mission grass) restricted to West Island, many large patches of *Tribulus cistoides* (beach caltrop) on Middle and East Island, and an expanding patch of *Cleome gynandra* (spiderwisp) on Middle Island. All remaining *Cocos* *nucifera* (coconut palm) individuals at Ashmore Reef have now died. Past efforts to control the four *Cenchrus* grasses, consisting of infrequent weeding, have been ineffective. This survey has established that was there is 1,200 m^2 of *Cenchrus* plants to control on West Island.

Recommendations

The 2019 survey represents the first detailed assessment of plant community assembly, spatial patterns and abundance for the Ashmore islands, providing a robust baseline for establishing future management success. It is clear that the stability of the shrub layer is under threat while non-native herbs and grasses could threaten bird nesting habitat and exclude native plants. The interactions between plant, vertebrate and invertebrate taxa need to be taken into account when devising the timing and sequence of management plans to mitigate these threats. For native plants, the taxonomy of *Boerhavia* spp. on the islands needs revision and urgent attention given to turning around the decline in health and mortality of shrub species. It is likely that an active restoration program will be required for all four remaining shrub species, with genetic supplementation considered for all but *Heliotropium foertherianum* (octopus bush). Of the eight non-native species on the islands, there is merit in considering eradication of the four *Cenchrus* species, *Xenostegia tridentata* (African morning vine) and *Cleome gynandra* (spiderwisp) and biological control of *Tribulus cistoides* (beach caltrop). When such restoration and control programs take place, however, they need to be (1) implemented consistently over a sufficiently long duration to ensure successful outcomes, (2) driven by an understanding of multiple direct and indirect interactions between plants, seabirds, ants, crabs and rodents to avoid unintended cascades, and (3) informed by leveraging the unpublished data that remains untapped in regard to understanding past change for the islands of Ashmore Reef.

1.3.2 Birds

Internationally significant numbers of seabirds and shorebirds use the islands and cays within the Ashmore Reef Marine Park. In addition, the Ashmore Reef Ramsar site is located within the boundaries of the Marine Park, and it is an ‘Important Bird Area’ for Lesser Frigatebirds (*Fregata ariel*) and Brown Boobies (*Sula leucogaster*).

Twelve seabird species had begun breeding in May 2019, and a further two species (Bridled Terns, *O. anaethetus* and Roseate Terns, S*terna dougalli*) were observed roosting, displaying courtship behaviour or attending nest sites. The breeding stage of Sooty Terns, *Onychoprion fuscatus* and Brown Noddies, *Anous stolidus* was more advanced on West Island. Sooty Terns were the most numerous birds within the Ashmore Reef Marine Park, with 77,309 counted across East, Middle and West Islands. There has been a shift in the distribution of seabirds across these three islands. For example, Black (*Anous minutus*) and Lesser Noddies (*A. tenuirostris*), previously found breeding only on Middle and East Island, were observed breeding only on West Island. A further five species have expanded their breeding territories from only East and Middle Islands from 1990-2014 to now include West Island in 2019. The numbers of adults observed within the Marine Park have apparently increased for the majority of seabird species.

Recommendations

We propose a series of recommendations which will improve monitoring of the seabirds and shorebirds, including aerial images of each island at 3-4 different time periods each year to estimate seabird and shorebird populations intra- and inter-annually; determining which habitats are important for foraging seabirds; and measuring levels of breeding success. As a priority, before the development of non-native vegetation management strategies within the Marine Park, it will be necessary to clearly identify the potential impacts of non-native plant removal on breeding seabirds. It may be necessary, for example, to have a staggered eradication program, or to plant other nesting habitat species (or install artificial structures) prior to the removal of substantial areas of weeds. As the diversity of seabirds breeding on West Island has increased since 2014, it is recommended that management arrangements around use and island access be reconsidered to ensure impacts on breeding seabirds is minimised.

1.3.3 Tropical Fire Ants

This study provides the latest assessments of tropical fire ant status on Ashmore Reef, with the work conducted comparable with the previous surveys conducted by Hodgson and Clarke (2014). Tropical fire ant (*Solenopsis geminata)* distribution and abundance was assessed, opportunistic observations of nesting seabirds, eggs, and chicks were made for any signs of interference by *S. geminata*. Specifically, we looked for dead chicks in nests, blindness, sting marks on bare skin (feet of adults, bodies of chicks), holes in the webbing of adult feet, and ants crawling over the bodies of birds or eggs. In addition, turtle nesting areas were examined for any signs of interference by *S. geminata,* including dead turtle hatchlings and dead hatchlings clustered at a nest site.

East, Middle and West Island displayed different patterns of *S. geminata* distribution and abundance. West Island had the lowest ant attendance at lures and stations, with ants predominantly occurring just on the island's circumference. Middle Island displayed a gradient from no ants on the relatively bare eastern end, to high abundance on its north-west end. The ant was most prolific on East island, being present at 90% of stations throughout the whole island. Its abundance on the three main islands largely reflected the abundance levels found by Hodgson and Clarke (2014). Unfortunately, spatial information is not provided in Hodgson and Clarke (2014) in any form (visual or data), so no comparison can be made of spatial dynamics. Notably, *S. geminata* was quite abundant on Middle Island which had undergone chemical treatments to kill the ant in 2011-12, demonstrating that there is no long-term suppression of the ant in the absence of eradication.

No instances were found of interference of *S. geminata* with any birds or turtles at Ashmore Reef, which is very surprising given the presence and high abundance of ants over most of the surface of the islands, often with nests and trails within the direct vicinity of nesting birds. However, the lack of interference is consistent with other recent seabird and turtle surveys that also did not find impacts, especially to seabird populations (Clarke & Herrod 2016; Guinea & Mason 2017), indicating that the imperative to eradicate the ant is not as urgent as first envisaged. Explanations of the lack of interaction at our time of observation is purely speculative, but potentially there was a difference in total food resources available to the ants at the time of survey (May) compared to the times when negative interactions had been reported previously (March, September, November). Our survey was conducted at the end of the wet season when there is significant grass seed resources available, and when the grasses had recently dropped their seeds, and when *S. geminata* populations would be expected to be undergoing a seasonal decline. It is also possible that there are strong annual differences driven by the annual climatic conditions that would influence food availability, ant populations and potential impacts.

Ultimately it is clear that impacts by *S. geminata* are dynamic and not always readily observed. But given that the impacts of this ant are well documented globally, and they have been observed at Ashmore previously, the mere continued existence of this ant on the islands should be of great conservation concern and plans to eradicate the ant should continue to be advanced.

Recommendations

Results of this study support the need to manage or eradicate the *S.geminata*, but the imperative to do so immediately is not as urgent as first thought given that there is no visible evidence of impacts on native wildlife, both new and historic. Why *S. geminata* impacts have such inter-annual variability remains speculative, but it is clear that Ashmore Reef's wildlife and breeding success are not under imminent threat. This hiatus in management imperative is opportunistic as it provides time for science globally to provide new baiting technology needed to conduct *S. geminata* management/eradication work on Ashmore Reef with minimal non-target impacts relative to current technology. The most promising of the horizon technology is RNA interference (RNAi), whereby highly targeted "toxic genetics" replace the use of general insecticides as the active constituents in baits. The "toxic genetics" interfere with some specific coding of the target species' genes which ultimately results in the death of the target species, but without affecting non-target species. We recommend that Parks Australia either wait for, or help support, the global initiative to develop/trial the RNAi technology proposed to treat and eradicate *S. geminata* from Ashmore Reef Marine Park.

Should there be a desire to attempt to understand the dynamics of *S. geminata* impacts, a study would need to be conducted that regularly measures numerous variables simultaneously for at least two years, namely ant populations, distribution, impacts (e.g. chick mortality), grass seed supply, and other potential food supply (all protein available to ants from nesting/roosting birds). These data would need to be coupled with stable isotope analysis of the ants to determine if their nutrient uptake is related to the availability of surrounding resources and seasonal environmental conditions. Note though that this knowledge would realistically have no direct impact on the effectiveness of any potential eradication plan.

1.3.4 Hermit crabs

The hermit crab populations at Ashmore Reef are the most susceptible non-target fauna for any proposed tropical fire ant (*S. geminata)* eradication program because they will readily consume the bait and as invertebrates they are also highly susceptible to the active constituents used against ants. Pilot-scale assessments of hermit crab abundance used to date have failed to provide useful data. Here we use a simple method to quantify hermit crab abundance that will enable meaningful comparisons with subsequent surveys for both general population assessments, and pre- and post- any eradication attempt of *S. geminata*. Hermit crabs were present on East, Middle and West islands, being most abundant on West Island (4.9 ± 4.2 mean ± SD) crabs per transect), far more so than on East and Middle Islands (0.6 ± 1.1) and (0.4 ± 0.7) crabs per transect, respectively.

Recommendations

This work established a simple, efficient and effective method of determining hermit crab abundances that should be used in future assessments of the Ashmore Reef hermit crab populations, and especially before and after any attempt to manage or eradicate tropical fire ant from Ashmore Reef Marine Park.

1.3.5 Introduced geckos

Introduced to numerous countries and oceanic islands around the world, the Asian house gecko (*Hemidactylus frenatus*) shows the largest non-native distribution of any gekkonid in the world. Reports suggest that it has been naturalised at Ashmore Reef since the 1990s. Current survey confirms that it is restricted to the West Island and abundant on the octopus bushes (*Heliotropium foertherianum*) above the shoreline. Two nights of sampling recorded 89 individuals (23 juveniles and 66 adults) and 5 eggs on 26 of the 35 *H. foertherianum* shrubs examined at West Island. The potential pathway to impact by *H. frenatus* on the terrestrial invertebrate fauna by direct predation requires further investigation.

Recommendations

There is a need to understand the ecological impacts (e.g. predation pressure on invertebrates) of the geckos in the Ashmore Reef Marine Park. Asian house geckos have been introduced to multiple regions around the world but known instances of the species causing ecological impacts are limited (Lever 2003). Competitive exclusion of sympatric geckos (e.g. Dame & Petren 2006) and transmission of parasites to native species (e.g. Barton 2007) are the studied impacts of this species, but they are not relevant to Ashmore as it is the only reptile species on the islands. Predation pressure on invertebrates is the likely ecological impact of Asian house geckos at Ashmore, but data to evaluate the level of impact does not exist. Analysis of stomach contents of the geckos and evaluation of the relationship between insect and gecko abundance would be needed. This work would be a pre-requisite to any consideration of control.

1.3.6 Corals

Ashmore Reef contains some of the highest coral diversity off the Western Australian coast. Photo quadrat and manta tow surveys were conducted at 216 sites across five habitat types (reef edge, shallow intertidal reef flat, shallow intertidal lagoon sand flat, shallow subtidal lagoon, deep lagoon/bommie habitat) Figure 57. 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% ± 2 s.e, soft coral 8.2% ± 2.5) and sponge cover (1% ± 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% ± 0.6) and approximately 4% of total benthic assemblages (3.9% ± 0.6). No coral bleaching or disease was observed.

Historical comparisons of coral cover presented here need to be regarded with caution because of the small number of areas surveyed prior to 2019. In areas that could be compared, mean coral cover was 20.3% ± 3.4 s.e in 2019. Although 9.1% lower than 2009 surveys (down from 29.4% ± 1.8), coral cover was 10.1% greater than that observed during 2005 surveys (up from 10.2% ± 1.5). 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.

Recommendations

Maintenance of diversity and monitoring cover and health of hard corals at Ashmore Reef are important objectives. The 2019 survey provides a spatially extensive, rigorous, quantitative baseline for monitoring coral cover. We recommend that the monitoring be carried out every five years alongside the regular invertebrate monitoring. It is important that additional surveys should be made in the event of any large perturbation such as a cyclone or a warming event. Forecasts of potential bleaching events 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.

1.3.7 Fish and sharks

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 22,121 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 per hectare, with highest densities observed within the reef slope (4,977 per/ha) and west lagoon (4,902 per/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 per ha) with a maximum density of 60 sharks per hectare on the reef slope and 20 sharks per hectare 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

Recommendations

A targeted survey to monitor sparsely distributed shark populations at Ashmore Reef is recommended as a high priority. 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. Underwater Visual Census (UVC) and Remote Underwater Video, may therefore provide more precise estimates of reef-shark densities at Ashmore Reef.

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 UVC. Previously, over 750 fish species have been reported at Ashmore Reef and Cartier Islands (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 water sampling for eDNA analysis to complement future biodiversity assessments. We suggest these be conducted every five years.

1.3.8 Holothurians, trochus and clams

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 also subjected to incidences of illegal fishing. Where available, data on non-exploited species is included to provide context.

Eighteen species of holothurians were found at Ashmore Reef in 2019. At least one holothurian species, *Holothuria lessoni* is locally extinct, and the abundance of at least 12 other holothurians; *Actinopyga lecanora, A. mauritiana, A. miliaris, Bohadschia argus, B. marmorata, H. whitmaei, H. fuscopunctata, H. fuscorubra, Thelenota ananas*, *T. anax* (all <0.02 /100 m^2)*, H. fuscogilva* (0.05 /100 m^2) and *Stichopus herrmanni* (0.2 /100 m^2) are at very low levels and have not recovered from overfishing in the mid to late 1980s. It is likely that for many of these, 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. When population levels are low it is difficult to detect statistically significant changes between successive surveys. Population size of *T. ananas* declined significantly between 2013 and 2019 and that of two other historically fished species; *H. whitmaei* and *S. herrmanni* are regarded as likely to have declined.

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.

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 and 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 densities of two species of clams; *Tridacna gigas* (<0.05/ 100 m^2) and *Hippopus hippopus* (0.37/ 100 m^2) are also low. 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. 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.

Recommendations

The importance of the deep-water habitat for holothurians and the possibility that some of the species with very low abundance, in particular, *Holothuria fuscopunctata, Holothuria fuscogilva, Stichopus herrmanni, Thelenota ananas* and *Thelenota 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 (Figure 57) 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.

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, impacted by fishing or to a lower level such as Rowley Shoals, Mermaid, Scott Reef, and other reefs in the MOU74 box 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.

With the very low abundance of certain holothurians and clam species, and apparent local extinction or low reproductive rates of others, feasibility or desirability for restorative measures might be considered. Restorative measures however have not been explored here. This would require significant logistical effort and/or radical intervention such as density manipulations within the reef during spawning time, translocation of adults or reseeding of larvae and/or juveniles. A feasibility study would need to be undertaken.

1.3.9 Seagrass

Seagrass was greatest on the reef flats (2.8% ± 0.7) at Ashmore Reef, and sparsely distributed across other habitat types (Figure 57). Although their cover is not high, 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 was collected), although cover is not universally high (10-30% cover at only 7 sites, and is higher on the western part of the reef flat than the eastern or southern sections of the reef. They are vitally important, contributing to productivity, providing food for green turtles and dugong, and stabilise sediments on reef flats creating habitat and nursery areas for marine organisms.

Species diversity, 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 (mean canopy height of 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^2 (range 88.9 to 600 shoots/m^2) across sites. Overall mean total biomass was 116.9 g DW/m^2 across all sites. Mean above ground biomass for *H. ovalis* was 0.015 g DW/m^2 (± 0.003) and below ground biomass was 0.023 g DW/m^2 (± 0.007) with an above and below ground ratio of 1:3. *Thalassia hemprichii* mean above (54.2 g DW/m^2 ± 1.24) and below (224.5 g DW/m^2 ± 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 (± 1.1) and 4.5 mm (± 0.2), respectively. Productivity and growth measurements displayed 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^2 with grazing rate calculated to be 0.4 g DW/m^2/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.

Recommendations

Our study has shown that *Thalassia hemprihii* is 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 the most dependent on seagrass beds for food and to refine the estimates of grazing rates by actual measurement. At present these are only estimated based on canopy height 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.

1.3.10 Sea snakes

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 5 years show that after a complete absence of sea snakes, at least one species, the olive sea snake (*Aipysurus laevis*), appears to be still present at the reef, however, numbers are 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.

Recommendations

Despite a significant search effort, only one individual sea snake was recorded during the surveys, further confirming that the population size at Ashmore, if a resident population even exists, is still extremely low. Determining the reasons for the significant decline in species diversity and abundance remains an important priority. That records of this species have been observed from the reef more than once in recent years may indicate the persistence of a very small population, or that recolonization of the reef following past local decline could be underway. We propose the design and implementation of a standardised surveying and monitoring protocol for sea snakes at Ashmore and few 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.

1.3.11 Turtles and dugongs

Turtle numbers have been consistently high at the reef since surveys started in early 1994. The last survey of turtles at Ashmore Reef was undertaken in 2017. Our 2019 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.

Dugongs at Ashmore Reef represent the most isolated and probably the least known population of Australian dugongs. We did not observe any on either of the 2019 survey trips. The last survey was more than 20 years ago, and the most recent reported sightings were in 2005.

Recommendations

Our surveys did not constitute a quantitative assessment of turtles. 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. Although we did not survey plastics and other marine debris quantitatively, the large amount of floating plastic pollution, principally Indonesian labelled food packaging, in the water at Ashmore Reef was at levels such that would create concern if present anywhere on the coastline of the Australian mainland. 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.

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. The reef is too large and remote to make regular aerial surveys practical, this work should be undertaken using a small vessel and RPAS survey methods, which could easily be done as part of any terrestrial RPAS programs.

PART I INTRODUCTION

Photo: Tommaso Jucker

2 INTRODUCTION

Part of the Australian Territory of Ashmore and Cartier Islands, 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 and Figure 2). The reef is 25 km long, covers 217 km^2 and comprises four low-lying islands (East, Middle and West islands and Splittgerber Cay), several smaller sand cays, two large lagoons and extensive reef and sand flat habitats. Ashmore Reef is the largest of the three emergent reefs present in the north-eastern Indian Ocean and a biodiversity hotspot for both terrestrial and marine species (Bellio *et al.* 2007; DEWHA 2009). Large, isolated offshore coral reefs present a challenge for effective conservation and management. They are difficult and expensive to survey frequently enough to match patterns of diversity and abundance responding to natural and human disturbance events, especially with the pace of anthropogenic climate change. They are also costly and logistically difficult places to undertake management actions. Despite such challenges, these coral reefs represent some of the highest value assets on the planet for conserving native biological diversity, and therefore prioritising their management is a global imperative.

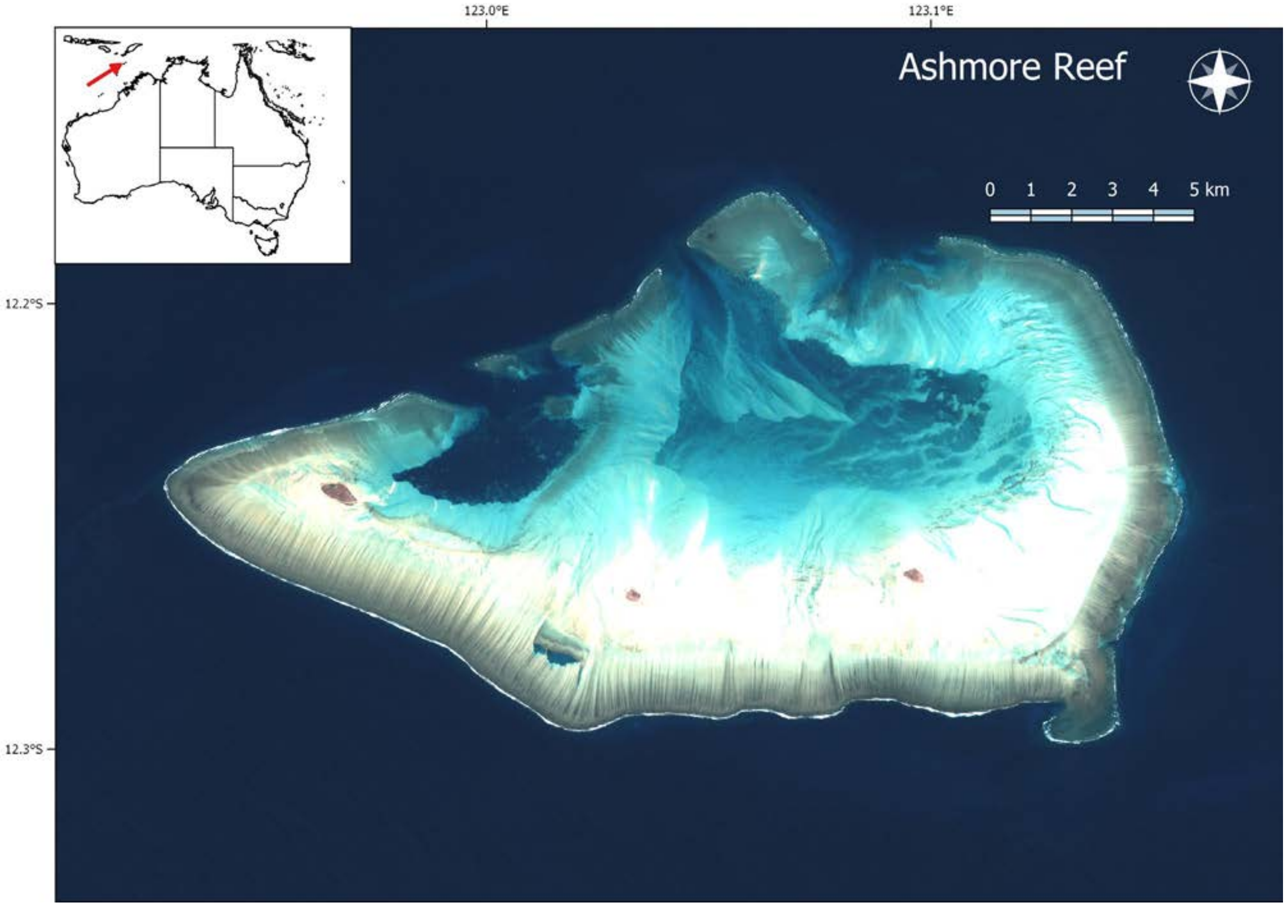


Figure 1. Location of Ashmore Reef and detail showing the islands, lagoons, sand flats reef edge, and reef flats (https://copernicus.nci.org.au/).

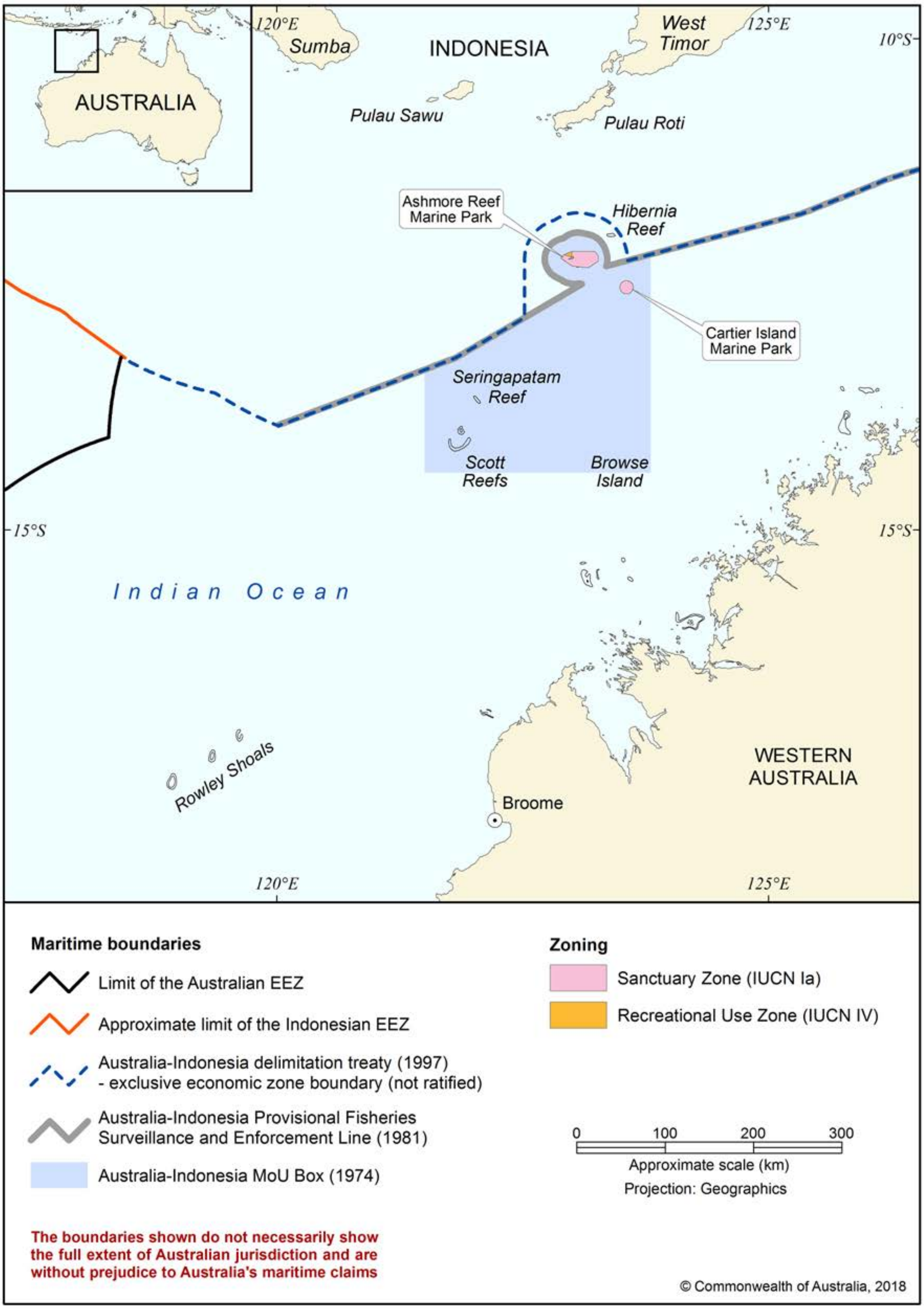


Figure 2. Showing Ashmore Marine Park with zoning locations and maritime boundaries. Taken from https://parksaustralia.gov.au/marine/files/maps/ashmore-cartier-location.png

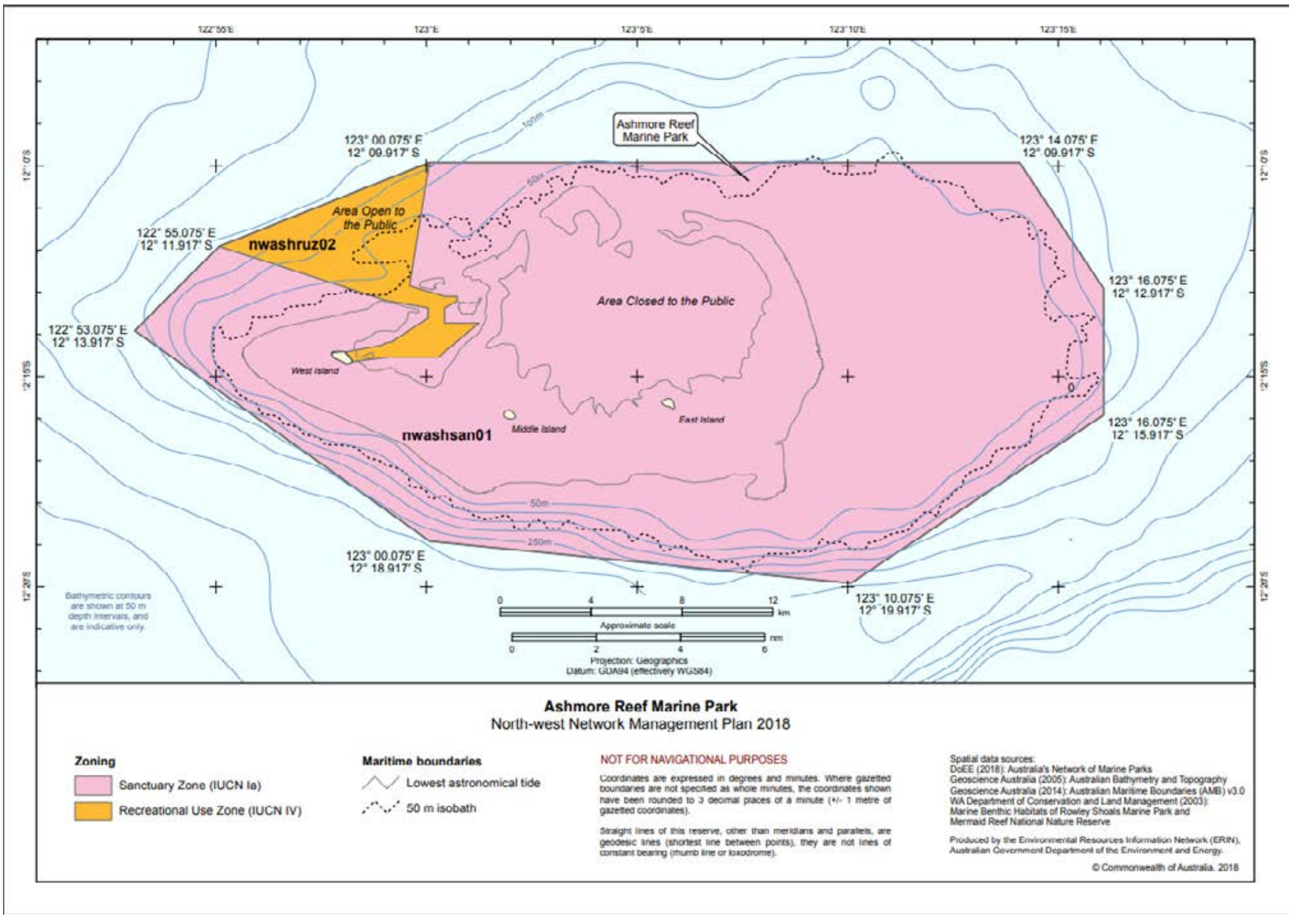


Figure 3. Showing Ashmore Reef Marine Park zones. Taken from https://parksaustralia.gov.au/marine/pub/maps/fnl-mp-2018-nw-map-ashmore-reef-mp.pdf

Ashmore Reef Marine Park (Figure 2 and Figure 3) was established by Australia in 1983, and is primarily a Sanctuary Zone, and therefore afforded Australia's highest level of marine protection. It also includes a small Recreational Use Zone, allowing access to the main lagoon and a narrow strip of land on West Island. The Marine Park is also subject to a Memorandum of Understanding (MOU74) between Australia and Indonesia, first established in 1974, providing limited access to traditional Indonesian fishers within the Recreational Use Zone. This level of access was restricted and the area that could be accessed was greatly reduced in 1989 (Commonwealth of Australia 2002) after concerns were raised about the level of exploitation of resources especially holothurians, trochus and clams (Russell & Vail 1988). Additionally, the Marine Park was designated as a Ramsar site in 2002 (Hale & Butcher 2013), meeting seven of the eight criteria used to identify Wetlands of International Importance. Since 2002, there have been refinements and revisions of the Ramsar criteria, and the Ashmore Reef Ramsar site meets six of the now nine criteria (Hale & Butcher 2013). Under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), the ecological character of the Ramsar site is a protected matter. The ecological character is defined as the critical ecological components, processes and services of the site. For Ashmore reef Marine Park these include near natural wetland types, biodiversity, physical habitats, listed migratory birds, threatened wetland species, marine invertebrates, fish, sea snakes, turtles, seabirds and shorebirds and dugong. Monitoring of the status and changes to these elements is important to inform decisions and management in order to maintain the ecological character of the site (Hale & Butcher 2013).

Following establishment, efforts were made by the Australian Government to ensure a compliance presence in the Marine Park. However, despite formal conservation measures, pressure on the reef's biota from illegal fishing of shark and harvesting of trochus, holothurians and giant clams continued. Surveys in 1998 raised concerns about harvest levels of holothurians in particular (Skewes *et al.* 1999a), and a survey in 2006 recorded further declines (Ceccarelli *et al.* 2007; Ceccarelli *et al.* 2011a).

In addition to its marine life, Ashmore Reef is also an important location for seabirds and shorebirds, as a stopover site for migratory shorebirds on the East Asian Australasian Flyway (Clarke *et al.* 2011; Clarke & Herrod 2016). West Island, the largest of the four islands, was subject to guano mining, which saw all the guano removed by the late 19th century (Pike & Leach 1997). Seabirds and their eggs were also taken in large numbers by visitors to the reef, and terrestrial vegetation was damaged by harvesting for firewood and the construction of wooden racks for the drying of holothurians (Serventy 1952a; Serventy 1952b; Milton 2005; Clarke *et al.* 2011). Ashmore Reef has also experienced the introduction of numerous non-native terrestrial plants and animals that further threaten native species and ecosystem stability (Pike & Leach 1997; Russell, Neil & Hilliard 2004; Hodgson & Clarke 2014).

Since 2008, there has been a permanent Australian Government presence at Ashmore 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. This presence has also reduced human visitation to, and activity on, the Ashmore Reef islands, with only a small area of the West Island now accessible to the public. This reduction in human impacts is likely to have had flow on effects to activities such as disturbance to flora and fauna from the drying of holothurians, the introduction of non-native species, and the opportunistic harvesting of terrestrial species.

Parks Australia manage Ashmore Reef Marine Park and recognise that best practice management requires a systematic and integrated approach to monitoring. To deliver against these goals, and to allow for a more rigorous approach to refining monitoring methods and future management priorities, this project was commissioned to assess the status and trend of key species and habitats in the park. By comparing with past surveys, the project will provide insight into how environmental values in the park are changing over time, and how management can be refined to improve conservation outcomes.

To achieve these aims, we conducted extensive surveys of the marine and terrestrial environmental values of the Ashmore Reef system. Each realm was assessed on separate research voyages, conducted in May (terrestrial) and June (marine) 2019.

For the terrestrial component, the required elements were to quantify the diversity and abundance of native and non-native plant species, determine the population status and diversity of seabirds and shorebirds on the islands, and to determine the abundance of non-native tropical fire ants. The remit included any evidence of interactions between the tropical fire ants and native fauna, especially birds and turtles. In addition to these required components, non-native geckos and hermit crabs were surveyed for abundance, the latter being native fauna which would likely be most affected by future attempts to eradicate tropical fire ants.

For the marine component, the required elements were to survey coral reef communities in particular abundance of key invertebrates which have historically been the subject of illegal fishing and over-exploitation (holothurians, trochus and tridacnid clams), the status (cover and health) of key habitat forming benthic assemblages (corals, macroalgae and seagrass) and evidence of sea snakes. In addition, surveys of fish and shark diversity and abundance were undertaken, and monitoring sites were established for seagrass biomass and productivity measurements. Incidental observations of turtle abundance were also made, and water samples collected from a subset of sites for potential future analysis of eDNA.

PART II TERRESTRIAL SURVEYS

Photo: Booby at Ashmore Reef Credit: Christopher Doropoulos

3 ASHMORE REEF: TERRESTRIAL VEGETATION

Bruce L. Webber, Paul B. Yeoh, Tommaso Jucker and Noboru Ota

3.1 Abstract

- Effective conservation management of our most valuable biological assets requires accurate and up-to-date knowledge on conservation targets, threats to those targets, and the feasibility and resource requirements of management to mitigate threats and build resilience. Ashmore Reef Marine Park is recognised as a wetland of international importance under the Ramsar Convention, is listed on the Commonwealth Heritage List, and is assigned the highest category of protection as a Sanctuary Zone (IUCN category Ia). Yet the Ashmore islands continue to be threatened by non-native invasive species and are losing vital nesting substrate for seabirds due to the decline in shrubs and trees.
- To deliver robust and enduring management programs to improve conservation outcomes for the islands of Ashmore Reef, a detailed understanding of the ecosystems as they are now is needed. Such insight can then be related to past drivers of change and their outcomes, as well as to inform management priorities for the future. To address this need we created the first quantitative assessment of vegetation structure, composition and abundance for all four Ashmore Reef islands (East, Middle and West islands and Splittgerber Cay). A combination of traditional quantitative ground surveys and aerial Remotely Piloted Aircraft Systems (RPAS) data acquisition was used to establish vascular plant diversity, abundance and community composition. We then compared this data to past surveys to establish trends in these metrics over time.
- The four islands at Ashmore Reef occupy a total of 56.3 ha with between 2.0 and 0.9m of vertical relief above the shoreline. Physical scars from previous guano mining and other anthropogenic land use still mark the islands and appear to be a stabilising factor in the spatial patterns of vegetation communities. A total of 21 native plant species (excluding *Boerhavia* spp.), including one new record for the islands (a native *Pandanus*), were observed across the four islands forming 35 distinct vegetation communities. West Island (18 spp.) was the most taxonomically diverse. Trees and large shrubs are dying, and their health has decreased markedly since the 1996-97 survey, with 72% of individuals now present either sick or dead and both *Cordia subcordata* (sea trumpet) and *Suriana maritima* (bay cedar) with very few individuals left. It appears that this decline is due to a combination of what appears to be nesting pressure from seabirds combined with shrub recruitment failure, the latter possibly due to seed predation by or competition from non-native species. A total of eight non-native plant species were recorded from the four islands, including four *Cenchrus* species (*C. brownii*, burr grass; *C. ciliaris*, buffel grass; *C. echinatus*, innocent weed; *C. pedicellatus*, annual mission grass) restricted to West Island, many large patches of *Tribulus cistoides* (beach caltrop) on Middle and East Island, and an expanding patch of *Cleome gynandra* (spiderwisp) on Middle Island. All remaining *Cocos nucifera* (coconut palm) individuals at Ashmore Reef have now died. Past efforts to control the *Cenchrus* species have been ineffective. This survey has established that there was 1,200 m^2 of *Cenchrus* plants to control.

- The 2019 survey represents the first detailed assessment of plant community assembly, spatial patterns and abundance for the Ashmore islands, providing a robust baseline for establishing future management success. It is clear that the stability of the shrub layer is under threat while non-native herbs and grasses could threaten bird nesting habitat and exclude native plants. Some preliminary qualitative associations between seabird nesting locations and dominant plant species are presented in Chapter 4. The interactions between plant, vertebrate and invertebrate taxa need to be taken into account when devising the timing and sequence of management plans to mitigate these threats. For native plants, the taxonomy of *Boerhavia* spp. on the islands needs revision and urgent attention given to turning around the decline in health and mortality of shrub species. It is likely that an active restoration program will be required for all four remaining shrub species, with genetic supplementation considered for all but *Heliotropium foertherianum* (octopus bush). Of the eight non-native species on the islands, there is merit in considering eradication of the four *Cenchrus* species (*C. brownii*, burr grass; *C. ciliaris*, buffel grass; *C. echinatus*, innocent weed; *C. pedicellatus*, annual mission grass), *Xenostegia tridentata* (African morning vine) and *Cleome gynandra* (spiderwisp), and biological control of *Tribulus cistoides* (beach caltrop). When such restoration and control actions take place, however, needs to be driven by an understanding of multiple direct and indirect interactions between plants, seabirds, ants, crabs and rodents, and by leveraging the unpublished data that remains untapped in regard to understanding past change for the islands of Ashmore Reef.

3.2 Introduction

Ashmore Reef Marine Park comprises four coral cay islands - East, Middle and West islands and Splittgerber Cay (Figure 4). The terrestrial vegetation of the these consists largely of widespread species found on tropical coastlines from south-east Asia to northern Australia (Cowie 2004). East, Middle and West islands, the largest, have contrasting vegetation communities, dominated by grasses and low shrubs, while a sandbank to the east that has existed as an island since 2010 (Splittgerber Cay) is currently colonised by three grass species (Pike & Leach 1997; Clarke 2010). Depending on the specific survey, between 29 (Kenneally 1993) and 38 (Cowie 2004) species of terrestrial vascular plants have been described from East, Middle and West Islands across vegetation surveys since 1977, including two taxa not known from the Australian mainland and more than 10 non-native species (i.e. introduced to Ashmore Reef Kenneally 1993; Pike & Leach 1997; Cowie 2004; Westaway 2015).

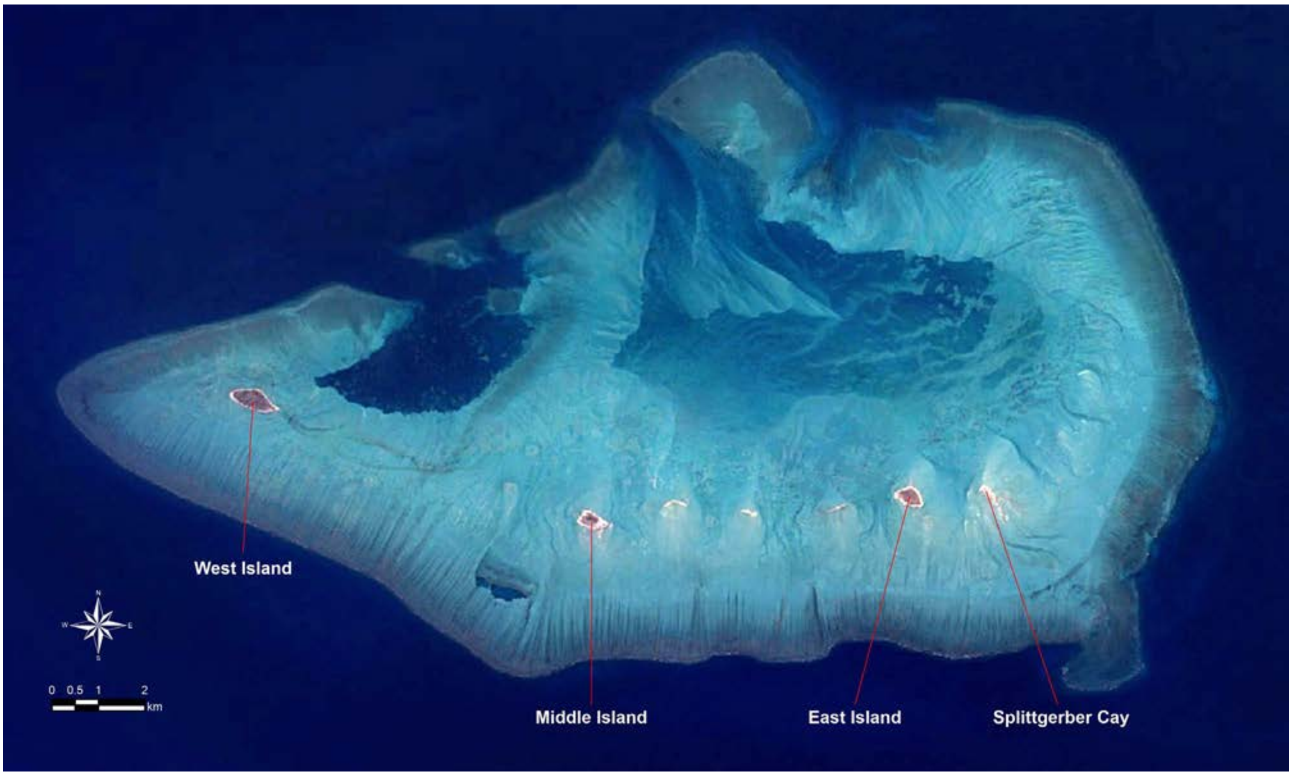


Figure 4. The location of the four islands – West Island, Middle Island, East Island and Splittgerber Cay - within the Ashmore Reef Marine Park that were surveyed for vegetation in May 2019.

This vegetation community undoubtedly plays a significant part in the stability of ecosystem interactions that spans terrestrial and marine communities, including providing structure and shelter for nesting seabirds (Clarke 2010; Clarke *et al.* 2011). The community is known to be dynamic, affected by seasonal changes due to an abundance of annual species, disturbance from animal nesting (birds, turtles) and extreme weather events (Pike & Leach 1997; Hale & Butcher 2013). This makes inter-annual variability in vegetation dynamics an important element to consider when refining management.

There is also a long history of human disturbance on the islands, including phosphate mining, visitation by Indonesian fishers and mining companies, and the introduction of non-native plants and animals. These anthropogenic impacts have shaped the abundance and diversity of native plant species, resulted in the introduction of plant species from Australia and South East Asia, and have led to physical modification of parts of the island (Pike & Leach 1997; Russell, Neil & Hilliard 2004). Some of these introduced plants, including *Cenchrus ciliaris* (buffel grass) and *Cenchrus pedicellatus* (syn. *Pennisetum pedicellatum;* annual mission grass) are known to be transformer weeds elsewhere in Australia and earlier reports from the islands have called for their control to be a priority (Cowie 2004; Hale & Butcher 2013). Furthermore, introduced flora can enhance the ability of these islands to act as important stepping-stones that alter connectivity between South East Asia and Australia, with significant biosecurity implications. For example, flying foxes are known to be vectors of a number of important diseases, and the *Cocos nucifera* (coconut palms) that have been planted by human visitors to the islands have been recorded as roosting sites for vagrant flying foxes (Pike & Leach 1997).

Taken together, this context underscores the need for an up-to-date quantitative understanding of terrestrial vegetation dynamics on the Ashmore islands to underpin effective management actions to mitigate threats and maximise resilience to future global environmental change impacts. Such insight is not only important for the island's vegetation communities, but also for the animals that interact with these communities (e.g. nesting birds, turtles; McDonald 2005; Hale & Butcher 2013). Despite this need, however, the vegetation surveys up to this point have been largely qualitative in nature. Furthermore, a single study on *Heliotropium* (syn. *Argusia argentia*; octopus bush; McDonald 2005) is the only known research to investigate terrestrial vegetation ecology on the Ashmore islands.

This lack of robust information upon which to base management actions is in part due to restricted and costly access to the reef, and that previously completed terrestrial surveys have primarily focused on the bird communities of the island rather than the vegetation. The comprehensive terrestrial vegetation survey of Pike and Leach (1997) produced stylised low-detail hand-sketched maps of broad vegetation communities across East, Middle and West islands. Despite recognising and noting changes in the vegetation since then, Cowie (2004) only described vegetation changes based on opportunistic and ad-hoc surveys at a time when much of the annual component of the vegetation was senesced. Since then, Clarke and colleagues (Clarke 2010) have made general observations and installed a series of qualitative photo points. While these data are useful for getting an overall picture of the health of larger components of the vegetation communities, they are less useful for quantifying how the grass- and herb-dominated communities are changing through time and space. We are aware that additional surveys of the terrestrial vegetation have been performed in recent decades. Some of this work has been recently discovered by the authors in unpublished reports (e.g. Westaway 2015) and in the personal data collections of scientists who have previously worked at Ashmore Reef. While it was not possible to use these recently discovered reports in the design of this study, we have, where possible, included them in the interpretation and contextualisation of the current work. Taken together, a more quantitative approach is required for generating the insight to underpin more robust management recommendations regarding conserving native species at Ashmore. In particular, mitigating the two threats of introduced species and climate change is critical to building resilience in these terrestrial communities to ensure the unique values of this reef ecosystem can be conserved.

3.3 Objectives

The objectives of this study were to combine systematic ground surveys with data layers generated by aerial surveys, and to create the first high-resolution quantitative picture of the vegetation structure, abundance and community composition across Ashmore's islands. This scope includes providing insight into both native and non-native species, surveying for any new arrivals, noting any absences of taxa observed from past surveys, the first estimate of abundance for all plant species present, and a vegetation canopy height model (CHM). Where possible, we relate these findings to past vegetation analyses to assess the magnitude of community change over time. A high-resolution digital elevation model (DEM) for all four islands was also produced, which will allow for island geomorphology, and shoreline / topography change to be tracked with greater precision into the future.

3.4 Methods

3.4.1 Survey logistics

Surveys were conducted across West, Middle and East Islands and Splittgerber Cay (Figure 4). Quarantine and biosecurity protocols were followed closely, including for inter-island movements (i.e. to avoid the spread of non-native species). Due to the monitoring and survey equipment being used, access to the island required careful timing with tides to ensure we could disembark our tenders on the beach without having to negotiate deep water.

An integrated approach was applied to producing the vegetation and topography survey of the island communities. Optical (RGB visible spectrum) and LiDAR (colourised point clouds) surveys obtained from flight missions by Remotely Piloted Aircraft Systems (RPAS or drones) were combined with survey-based ground truthing by quadrats (spanning stratified vegetation communities anchored via georeferenced ground control points) and individual plant mapping for trees and bushes. The stipulated survey period of April/May 2019 coincided with when the detection of the broadest cross-section of the vegetation community is most likely (Cowie 2004; Clarke *et al.* 2011), as well as allowing relationships between vegetation and dominant bird nesting times to be documented (Clarke *et al.* 2011).

After a careful assessment, it was determined to not waste the very limited time at Ashmore Reef undertaking active management of the non-native plants (RFO Schedule 1, A.5 (c) (i)). It was deemed that undertaking *ad-hoc* management was not likely to produce an effective management outcome. Without further understanding of the ecosystem, control efforts may well have adverse off-target impacts. This is because any weed management decision should be made with sufficient baseline knowledge, including identification, distribution, and abundance of the extant population, as well as information on soil seedbank dynamics (Wilson *et al.* 2014). Moreover, there is no understanding as to how these plant species interact with or impact on other components of the island community (either positively or negatively).

3.4.2 RPAS missions

Missions to generate LiDAR data were conducted using a RIEGL miniVUX-1UAV scanner mounted on an RPAS (DJI Matrice 600 pro) flying at an altitude of 70 m above ground level (AGL) at approximately 8 m.s^{-1} in swaths approximately 48 m wide. DJI Ground Station Pro software was used for all mission planning. The RPAS platform also includes a 24 Mp digital camera (Sony A6000 with an f2.8 16mm lens) that is co-registered with the scanner, allowing the 3D LiDAR point-clouds to be colorised and the images to be georeferenced. This platform is able to acquire extremely high-resolution LiDAR point clouds. Every pass the RPAS does at 70m AGL produces 25 to 28 pulses.m^{-2} within the swath (with data added to neighbouring swathes as the LiDAR range is up to 250m from the RPAS). Each pulse is capable of having up to 5 returns depending upon the complexity and density of the vegetation layers present, resulting in >100 points.m^{-2} on a single pass. For each echo signal, high-resolution 16-bit intensity information is provided near infrared. The laser beam footprint is quoted as 160 mm x 50 mm @100m, the LiDAR sensor accurate to 15mm, the IMU accuracy is 0.025 deg Roll/Pitch and 0.08° heading (sample rate 200hz) resulting in a theoretical positional accuracy of <0.05 m horizontally and <0.1 m vertically. This resolution allows the 3D structure of ground topography as well as individual shrubs to be captured in detail. Base station corrections relied upon Rinex files collected with a Trimble GeoX7 differential GPS. Missions to generate optical (visible RGB) data were conducted with a DJI Phantom 4 Pro RPAS (20 Mp camera with an f/2.8-f/11 24mm lens) at both 30 and 120 m AGL with images overlapped by >75% front overlap and >70% side overlap as required for producing orthomosaic maps.

All RPAS operations were conducted under CSIRO Animal Ethics Permit 2019-06 with respect to managing risk relating to potential wildlife interactions. While conducting RPAS missions, all interactions with the resident bird populations were carefully monitored. All missions were launched and landed at a minimum distance of 50 m away from bird colonies (taking off from exposed sandbars if needed). Across all missions with all RPAS at all altitudes, including all landing and take-off procedures, no birds came anywhere near the RPAS, nor showed signs of distress. All birds gave the RPAS a very wide berth in flight. In fact, despite a continually high density of airborne birds circling above the islands and arriving or leaving for foraging trips, on no occasion did we need to manually manoeuvre the RPAS to reduce the risk of what may have been an impending collision. Furthermore, there was no observed change in the number of birds in the air, relative to those on the ground (on beaches, nests or perches) comparing before, during and after RPAS missions. These observations fit with previous pilot studies using a smaller RPAS at Ashmore Reef, which had no impact on the resident bird populations (Hodgson & Koh 2016). Overall, birds were less bothered by RPAS than by people, making RPAS surveys of bird colonies a far less intrusive way to estimate bird numbers relative to manual counts on the ground.

3.4.3 Island topography and vegetation canopy

High resolution (25cm grids) 3D digital elevation model (DEM) maps were derived for all four islands using the LiDAR point clouds for West, Middle and East Islands, and photogrammetry for Splittgerber Cay (where no shrubs were present). Canopy height model (CHM) maps at the same resolution were also derived for the three islands with shrubs and trees – West, Middle and East Islands. The LiDAR data was processed using a combination of Applanix POSPac UAV (Version 8.3) and RIEGL RiProcess (Version 1.8.5) software and the Rinex files from the Trimble GeoX7 differential GPS. Custom-written code was then used to quality control the point cloud data, particularly in relation to erroneous returns from airborne seabirds, to generate DEM and CHM layers for each island. This data was projected using WGS1984 UTM Zone 51S for spatial manipulation and analysis in ArcMap (Esri, V10.8.1). Island shoreline boundaries were defined by altitude based on the high tide mark, as well as by the lowest point at which vegetation was present growing on the beach (Liu *et al.* 2014).

3.4.4 Island imagery

Visible RGB imagery layers were generated at three Ground Sampling Distance (GSD) resolutions: 0.82 cm x 0.89 cm per pixel, 1.70 cm x 1.70 cm per pixel and 3.29 cm x 3.54 cm per pixel for the 30 m, 70 m and 120 m AGL flights, respectively. To map East Island we conducted 4 flights at 30 m (1560 images calibrated & enabled for orthomosaic generation), 1 flight at 70 m (417 photos) and 1 flight at 120 m (306 photos). At Splittgerber Cay, we conducted 1 flight at 30 m (265 photos) and 1 flight at 120 m (90 photos) but did not have enough island access time to fly the lidar at 70 m. An absence of complex vegetation on Splittgerber Cay meant that structure-from-motion (SFM) photogrammetry using the RGB images could still produce high quality point cloud data. Middle Island had 4 flights at 30 m (1639 photos), 2 flights at 70 m (907 photos) and 1 flight at 120 m (294 photos). For West Island we conducted 12 flights at 30 m (4364 photos), 3 flights at 70 m (981 photos) and 2 flights at 120 m (634 photos). We then resurveyed half of West Island with an additional 7 flights at 30 m (2397 additional photos) 3 days after the initial 12 flights were done, to capture any changes in seabird nesting behaviour.

The optical image files collected at 70m AGL with the Sony A6000 camera on the M600 Pro RPA system are not geotagged when initially taken. After calculating PPK GNSS solutions and angles of each lidar return (with Applanix POSPac UAV), the Riegl RiProcess software was used to calculate and geotag the A6000 images with differentially corrected GPS coordinates. The optical image files collected with the Phantom 4 pro RPA system at 30 and 120m AGL are automatically geotagged by the RPA system (hovering accuracy ±0.5 m vertically and ±1.5 m horizontally).

Pix4Dmapper (Version 4.5.2) was used to generate orthomosaic maps from the geotagged images collected at 30, 70 and 120m AGL with the following processing options: Keypoint Image scale=Full, Calibration method=standard, Point Cloud Image scale=multiscale, Half image size as default, Minimum Number of matches=3, Medium resolution 3D textured Mash settings. The Absolute Geolocation Variance RMS (the difference between the initial and computed image positions as calculated by Pix4D) was 1.59 ± 0.20 m (1SE; x-axis), 1.98 ± 0.25 m (y-axis) and 2.13 ± 0.74 m (z-axis) for the 5 orthomosaic maps generated at 30m AGL (note 2 sets of data were collected at different time periods on West Island) and 3.80 ± 0.51 m (x-axis), 2.41 ± 0.15 m (y-axis) and 1.09 ± 0.25 m (z-axis) for the 4 orthomosaic maps generated at 120m AGL. For the 3 orthomosaic maps generated at 70m using the M600 RPA system (there was no data collected on Splittgerber Cay), the values were 1.65 ± 0.91 m (x-axis), 2.33 ± 1.45 m (y-axis) and 1.33 ± 1.04m (z-axis).

Due to time restraints (we could only gain access to the islands at high tide due to shallow sand banks surrounding each island, especially the Splittgerber Cay), it was not possible to set Ground Control Points prior to conducting flight missions with the Phantom 4 Pro. To maximise the accuracy of the orthomosaic maps from this RPAS, we therefore used the orthomosaic maps created from the M600 RPA system (constructed from the more accurate differential GPS PPK GNSS based data sets), as the reference base map for the orthomosaic maps generated from the Phantom Pro4 images at 30 and 120m on West, Middle and East islands. These were subsequently aligned with the M600 RPAS base maps within ArcGIS using the georeferencing tool with manually assigned control points based on clearly discernible landscape features that would not shift location. This data was projected using WGS1984 UTM Zone 51S for spatial manipulation and analysis in ArcMap (Esri, V10.8.1).

3.4.5 Ground surveys

Species identification

Across the prior reports on the terrestrial vegetation of Ashmore Reef, considerable variation exists in relation to (a) the consistency of species determinations, (b) the application of taxonomic nomenclature, and (c) the determination of native or non-native status. To allow for repeatable and consistent analysis within this survey and to be able to compare current results to past work, species nomenclature and native status was reassessed for all species where there was inconsistency either in past reporting from Ashmore Reef or within relevant literature from other locations or sources. All plant species determinations were made by Bruce Webber and Paul Yeoh. Photos of all plant taxa on the islands, including reproductive material where available, were taken to help future identification confirmation in the lab and to illustrate management guides. Collections were made of plants of uncertain identity or unique context to enable verification of field identifications by taxonomic experts and to support future work on the island flora. Voucher specimens of representative plants are being prepared for submission to the Western Australian Herbarium (PERTH). Standard time signals on cameras (Canon EOS7D MkII, Canon EOS5D MkIV, Olympus OM-D EM1) and tracklogs from handheld GPS devices (Garmin 66ST and/or Garmin 64S) together with the software program geosetter (https://geosetter.de/en/main-en/) were used to corroborate and quality control the spatial and temporal alignment of multiple vegetation data sources (photos, quadrat data, dGPS readings, GPS readings, herbarium samples etc.) used for the vegetation surveys.

Taxonomic nomenclature was corrected where possible, and synonymy was tracked to aid in the interpretation of past reports. Species naming conventions were primarily informed by the Australian Plant Census (APC; https://biodiversity.org.au/nsl/services/search/taxonomy), the Australian Plant Name Index (APNI; https://biodiversity.org.au/nsl/services/search/names) and the Atlas of Living Australia (ALA; https://bie.ala.org.au). Common names were chosen to best match previous reports where possible, with names use being the same as Pike and Leach (1997) if available, then Brown, H and Raphael (2008), APC, Australian Plant Common Name database (APCN: http://www.anbg.gov.au/common.names), plantnet.org and the Global Biodiversity Information Facility (GBIF; https://www.gbif.org), in that order of priority. Any deviations were explicitly noted and justified. We detailed the most consistently applied common names for species in the summary and introduction sections of the report as well as summarising these names in the results (Table 4). However, due to ongoing variation and inconsistency in the application of common names to plant taxa found at Ashmore Reef, as well as the situation that many species do not have common names, we used scientific names in the methods, results and discussion sections of this report.

Native and non-native status determinations was based on the definitions of Webber and Scott (2012) and was informed by plant lists, literature and expert opinion where available. Native plants were considered those that could realistically have arrived from a native population elsewhere without human intervention. For example, they could have been transported to the islands via birds, water currents or wind. In contrast, non-native plants either lack dispersal mechanisms that could conceivably allow them to get to the island unless transported by humans (intentionally or accidently) or have arrived on the island by natural means but originating from a non-native population. New arrivals at Ashmore can therefore be native or non-native. All plants with uncertain native status were assessed in this manner and evidence collated to support the decisions.

We actively searched for an additional ten non-native weed species that have never been described before from Ashmore Reef, but which were previously identified by biosecurity experts as targets with a high risk of establishing and having negative impacts (Cowie 2004; Brown, H & Raphael 2008; Table 1).

Table 1. Plant species not known from Ashmore Reef but previously identified by biosecurity experts a high risk of establishing there and having negative impacts.

Scientific name	Family	Common name	Growth habit
Boerhavia erecta L.	Nyctaginaceae	erect tar vine	Robust perennial herb to 80 cm high
Chromolaena odorata (L.) R.M.King & H.Rob.	Asteraceae	Siam weed	Herbaceous to woody perennial to 2 m high
Cleome rutidosperma DC.	Capparaceae	fringed spiderflower	Annual herb to 1 m high
Croton bonplandianus Baill.	Euphorbiaceae		Woody herb or shrub to 1.5 m high
Croton hirtus L'Hér.	Euphorbiaceae	croton	Erect annual herb to 1.2 m high, offensive smell
Indigofera zollingeriana Miq.	Fabaceae	Zollinger's indigo	Small tree 2-3 m high
Mikania micrantha Kunth	Asteraceae	mikania vine	Fast growing creeping or twining plant
Mucuna pruriens (L.) DC.	Fabaceae	cow itch	Semi-woody twining vine
Paederia foetida L.	Rubiaceae	skunk vine	Slender vine
Striga asiatica (L.) Kuntze	Orobanchaceae	witch weed	Obligate parasitic herb to 30 cm high

Community characterisation

RPAS-derived maps of aerial photography (visible RGB collected with the Phantom 4 Pro RPAS at 120 m AGL the day before the anticipated ground surveys, then processed and printed at A3 size overnight), were prepared for all four islands to use during ground surveys. Vegetation communities were identified by the same person (Bruce Webber) across all four islands according to clear thresholds of spatial change in plant species composition and abundance. These community boundaries were identified in the field by hand annotation on the A3 aerial photographs while traversing the islands on foot. GPS track logs were used to document overall survey coverage and to ensure no areas were missed. Due to high seabird nesting densities on some islands, certain areas were avoided to ensure the welfare of nesting animals and binoculars were used to examine some areas from a distance. Viewing these areas from multiple angles and post hoc inspection of high-resolution aerial photography ensured no plants remained unidentified. Documentation of previous locations where non-native species had been observed, where available, was used to guide targeted searching effort over and above the regular survey work. Lastly, unstructured surveys were conducted via a detailed inspection of vegetation communities in public access areas and sites of historic disturbance or settlement (e.g. the public access corridor on West Island, old campsites and weather station sites).

Throughout the ground survey, all species sighted were noted and estimates of overall abundance observed within each of the communities was estimated using a cover-class system scale. Adapting the systems devised by Braun-Blanquet (1932) and Daubenmire (1959), we modified the scale to have lower classes split into finer units so as to account for the many species within community studies that normally fall into these categories (Table 5; Elzinga, Salzer & Willoughby 1998). In some cases, plant species were sighted/noted whist surveying a community but, by chance, the same species was not detected within the quadrats randomly sampled within that community. Species accumulation curves using the quadrat data were generated for each island separately and for the Marine Park as a whole by randomising the order of quadrats (Colwell & Coddington 1994).

Table 2. Vegetation category classes for describing species abundance over a whole plant community for the survey at Ashmore Reef.

Class	Description of abundance	Range (% cover)	Range midpoint
1	Rare (a single large or a few small individuals)	< 1	0.05
2	Occasional (a few large or numerous small individuals)	1-5	3
3	Scattered (or locally common)	5-15	10
4	Common (or locally dense)	15-25	20
5	Co-Dominant (2 or 3 major species)	25-55	40
6	Dominant (single major species)	40-80	60
7	Monoculture	>80	90

Species abundance

Targeted quantification surveys of plant presence and abundance consisted of two field survey methods:

1. To generate quantitative species cover data, all plant communities were surveyed by randomly placed quadrats (1 × 1 m) within identified communities in a stratified random sampling approach. This survey approach is best suited to capturing presence and abundance for the ground layer vegetation (i.e. grasses and herbs), rather than the larger shrubs and trees. Although community boundaries were being identified at the same time as the quadrat surveys (to minimise seabird disturbance), we aimed to ensure at least three quadrats were located in each plant community, with greater numbers of quadrats for communities covering a large area or with multiple community locations across an island. We used virtual non-linear pathways through the plant communities that traversed representative areas of vegetation but that also avoided disturbing nesting seabirds. Distances between quadrats was based on an *a priori* random stride number to ensure random quadrat locations. The total number of quadrats placed was determined by island access, timing and the overall length of the voyage.

   In each quadrat, all plant species present were identified and individually assigned visual percent cover scores based on their vertical projections (live plant material only; noting that as species can overlap each other the total percent cover of all species can be >100%). A score was also assigned to bare ground (i.e. the area not occupied by live plants). Charts with known percent cover (McNaught *et al.* 2006) and the knowledge that 1% cover is an area 10 cm × 10 cm within a 1 m^2 plot were used to assist with standardising the visual assessments. The same person (Paul Yeoh) assessed all cover scores for all quadrats to reduce variation due to assessor bias, a known major source of error (Elzinga, Salzer & Willoughby 1998). Due to vertical layering, cover scores can add up to more than 100%.

2. All shrubs and trees were surveyed via the identification of individual plants in a ground survey. Individual identities of all larger shrubs and trees were recorded by hand-annotating A3 hardcopy aerial maps, as well as recording locations with a handheld GPS device for rare

species. For individuals that had died, branch and/or stem morphology was used, where possible, to attribute likely identity. For *Sesbania cannabina*, a small shrub (to c. 2 m high) occurring at considerable density across the three larger islands, it was not feasible to map each individual. Instead, plant counts were made within each cover data quadrat containing *S. cannabina*. These counts were then averaged and extrapolated over known community areas to give estimated plant numbers at an island level.

Vegetation change over time

Three potential sources existed to understand change over time. First, earlier map products may allow for a spatial assessment of range change. Orthomosaic imagery for each island was compared to the original stylised sketch maps of Pike and Leach (1997) to assess the likelihood of community change over time. These were the only previous vegetation maps identified and available for Ashmore Reef at the time of writing this report, noting those available in Hale and Butcher (2013) are just more inaccurate stylised versions of the Pike and Leach (1997) maps.

Second, survey descriptions and presence/absence data from past vegetation surveys may allow for some understanding of change in presence and/or abundance between islands. We therefore reviewed descriptive vegetation surveys from Kenneally (1993), Cowie (2004), Clarke (2010) and Westaway (2015) spanning 1977 to 2015 to provide further semi-quantitative detail on some components of the vegetation community. Where possible, we generated population metrics for all species found in past surveys, including (1) qualitative and semi-quantitative presence, absence, abundance metrics: 0: no plants found, 1: single individual, 2: a few individuals or localised patches, 3: common &/or widespread, and (2) population stand age metrics: A: adult, J: juvenile, P: mixed age population, D: dead. These population metrics were also generated for our own 2019 survey to allow for comparisons over time. To pick up further ad hoc surveys during the same period we accessed 414 herbarium specimen records from the Atlas of Living Australia (downloaded 16 Jul 2020; https://doi.org/10.26197/5f0f2124f3e7c) as well as direct database downloads from the Australian National Herbarium, Canberra (CANB) and the Northern Territory Herbarium, Darwin (DNA).

Last, permanent photo points were installed on the islands in 2010 (Clarke 2010). This monitoring approach can provide a general indication of topographic change and the health of shrubs and trees, as well as large scale shifts in the herb layer. Efforts were made to find the reference markers and re-take the photos. Images were taken with a Canon EOS5D MkIV with a 24-105mm f/4L IS II USM Lens set at 24mm and f/13. Georeferencing details and photo directions were checked and corrected where possible (SI Table 1). No resources were available during the visit to maintain or replace the reference markers, some of which were missing and many of which were rusted and fragile.

3.4.6 Digitisation and collation of data

Island topography and shorelines

All available satellite imagery was downloaded via sas.planet (build 2018-12-21; http://sasgis.org/) from Bing Maps (www.bing.com/maps) and Google Earth (www.google.com/earth) and date stamped where possible. A selection of this imagery was used to identify island shoreline erosion and accretion over time as well as to provide background imagery for overlaying the mapping products.

Community delimitation

Vegetation community boundaries from the hand-annotated maps were digitised as polygons onto an overlay of the 30m AGL visible RGB survey imagery in ArcMap (Esri, V10.8.1) at a viewing scale of 1:50. Based on visual recognition of dominant species in the high-resolution aerial imagery, minor manual adjustments to these polygon boundaries was undertaken. Adjustments were made only after close examination of maps at 1:20 to check for the identity of individual plants and by comparison with the CHM layer to identify changes in vegetation heights. For certain vegetation (e.g. the distinctive *Tribulus cistoides* and *Spinifex littoreus*), additional small patches that were not recorded during the field survey were able to be added in based on this approach.

Species abundance

All vegetation quadrat locations were geolocated for subsequent mapping with a differential GPS (Trimble GeoX7) used as a rover, and corrected via hourly reference position data from base stations at Broome, Fitzroy, Kununurra, Karratha and Christmas Island (using Trimble, GPS Pathfinder Office Version 5.80 software). From the 15,758 quadrat positions corrected by post-processing against these five base stations (simultaneously), 87.26% were estimated to be accurate to <50 cm, 12.69% between 0.5 and 1.0 m and 0.04% between 1 and 2 m. The exact location of quadrats based on the differential GPS georeferenced positions (precision < 0.5 m) were further refined based on the photo of the quadrat and distinctive vegetation patterns from the aerial imagery.

Island-wide plant cover scores stratified by community were generated by interpolating and extrapolating the quadrat data in ArcMap (Esri, V10.8.1). Inverse distance weighting (IDW) was used with a slow decay rate (search radius 1,200 m, cell size 1.0 m, power 2.0) to avoid areas with low quadrat density producing implausibly low cover scores. These community-stratified layers were then aggregated across communities within islands to generate island-wide abundance maps for each taxon, excluding shrubs and trees.

For shrubs and trees, locations were digitised as points onto an overlay of the 30m AGL visible RGB survey imagery in ArcMap (Esri, V10.8.1) at a viewing scale of 1:30. The CHM layer was used to co-inform decisions in some situations. All individuals were stratified into ‘large’ or ‘small’ cohorts to aid in understanding spatial patterns of recruitment dynamics. For live shrubs this threshold was based on a 2m canopy diameter, a size at which most individuals of the dominant shrub, *Heliotropium foertherianum*, were producing flowers (i.e. maturity). For dead individuals, this size threshold was based on either the diameter of the remaining dead canopy branches where intact, or an assessment of the likely canopy size based on the diameter of the remaining primary trunk.

Vegetation change over time

For the mapping comparison component of the vegetation change analyses, we used ArcMap (Esri, V10.8.1) to align the maps of Pike and Leach (1997) to the orthomosaic images taken during this survey, based on manually assigned control points utilising overall island shape, beach rock and anthropogenic hardware/features common to both layers.

3.5 Results

3.5.1 Island topography

Aerial imagery of the islands revealed considerable vegetation cover across all four islands. The spatial resolution of this imagery not only allowed for confirmation of the location of individual plant species, but also for the counting of individual birds at the species level (Figures 5 to 8; and see Chapter 4). Discarded and fragmenting infrastructure, including old poles and concrete pads are present and visible on East, Middle and West islands. In particular, coastal erosion on the north side of East Island has resulted in the old concrete helipad fragmenting below the high tide mark (Figure 6).

Based on the digital elevation model (DEM) surfaces and the high tide mark, the four islands at Ashmore Reef occupy a total area of 56.3 ha (Table 3). Compared to satellite imagery from earlier years, it appears as if the shape of the islands continues to change. While not possible to quantify due to a lack of points to cross-reference images, it appears that all four islands are changing shape. Relative to satellite images available from between 5 and 10 years ago (Google Earth), West Island is gaining area due to accretion on the eastern tips, Middle Island is gaining area due to accretion on the eastern tip, East Island is losing area due to erosion along the northern edge, and Splittgerber Cay is gaining area due to accretion at the northern edge as well as shifting north-east at its south eastern end (Figures 5 to 8).

Table 3. Island area size for the four islands – West Island, Middle Island, East Island and Splittgerber Cay - at Ashmore Reef. Maximum island width was taken perpendicular to the longest axis (i.e. length) of the island.

Island	Max length (m; orientation)	Max width (m)	Total area (ha)	Max elevation (m)
West Island	1,095 (ESE-WNW)	427	29.0	5.0
Middle Island	617 (E-W)	314	12.7	3.8
East Island	587 (ESE-WNW)	340	13.7	4.2
Splittgerber Cay	315 (SE-NW)	61	0.9	5.2

The digital elevation models revealed that all four islands have very limited vertical relief and a general pattern of higher dunes around the exterior of the island with lower relief toward the centre of the islands (Figures 9 to 12). The largest of the islands, West Island, has a maximum relief (i.e. high tide shoreline to highest dune) of 2.0 m asl, with Middle Island (1.1 m asl), East Island (1.8 m asl) and Splittgerber Cay (0.9 m asl) slightly lower in relief.

Physical scars from the guano mining in the latter part of the 19th century, as well as the old well and another hole dug in the centre of the island, are clearly visible in the DEM map of West Island (Figure 9). The well site on Middle Island is also visible as a point of lower relief (Figure 10), while the old well site on East Island was not detected (Figure 11). The challenges with the LiDAR penetrating the extremely dense vegetation in the patch of *Spinifex longifolius* were also evident in the DEM, with an artefact of slightly higher relief evident in this area (Figure 9).

Figure 5. Orthomosaic image of West Island at Ashmore Reef in May 2019 (a). RPAS missions flown at 30 m altitude were of sufficient resolution to identify individual birds to species (b) and spot introduced plant populations for subsequent ground truthing (c). Insets depict (b) nesting crested terns (*Thalasseus bergii*) and (c) burr grass (*Cenchrus brownii*). Background imagery dates from October 2016, which explains the slight misalignment of the island coastline due to erosion and accretion over time.

Figure 6. Orthomosaic image of Middle Island at Ashmore Reef in May 2019 from RPAS missions performed at 30m altitude. Background imagery dates from up to 10 years previous, which explains the slight misalignment of the island coastline due to erosion and accretion over time.

Figure 7. Orthomosaic image of East Island at Ashmore Reef in May 2019 from RPAS missions performed at 30m altitude. Background imagery dates from up to 10 years previous, which explains the slight misalignment of the island coastline due to erosion and accretion over time.

Figure 8. Orthomosaic image of Splittgerber Cay at Ashmore Reef in May 2019 from RPAS missions performed at 30m altitude. Background imagery dates from up to 10 years previous, which explains the misalignment of the island coastline due to erosion and accretion over time.

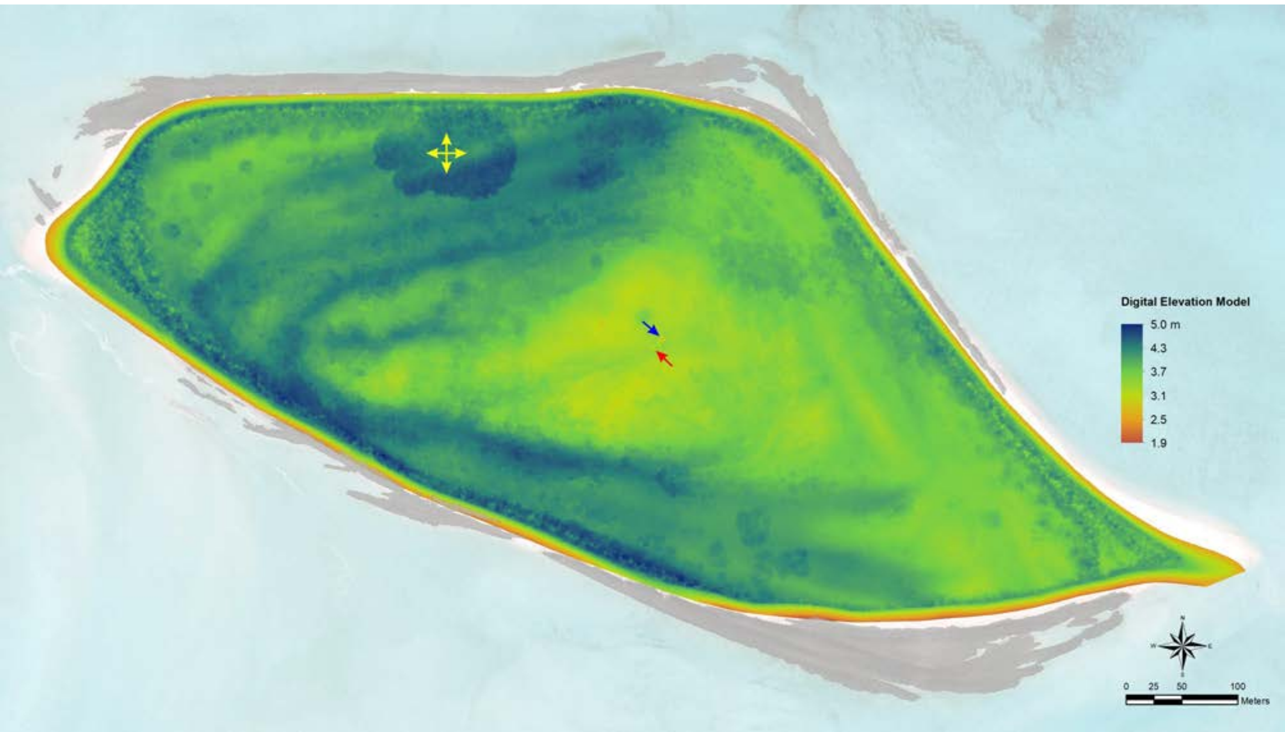


Figure 9. Digital elevation model (DEM) for West Island at Ashmore Reef in May 2019. Locations of island well (blue arrow), excavation (red arrow), the *Spinifex longifolius* patch (yellow arrows) and the surrounding rock shelf structure (light grey) are depicted.

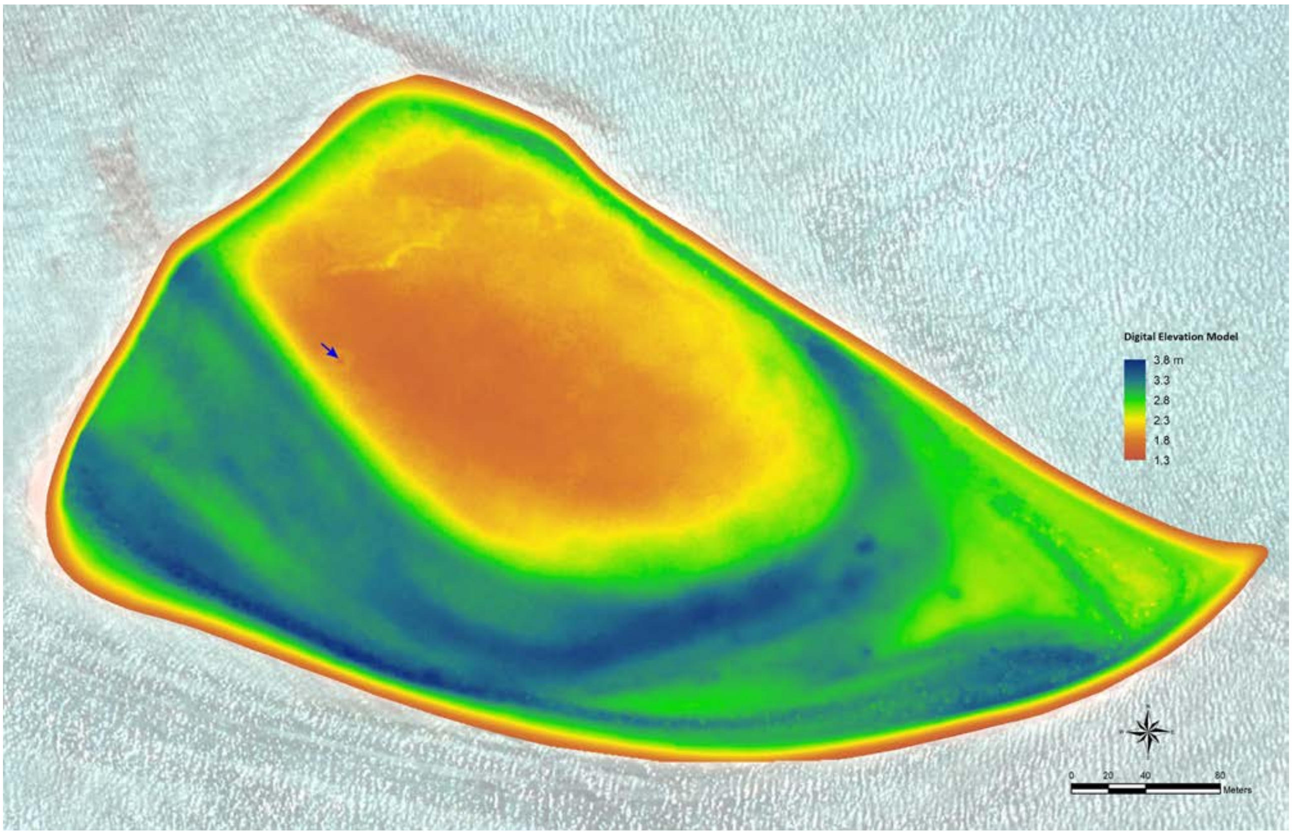


Figure 10. Digital elevation model (DEM) for Middle Island at Ashmore Reef in May 2019. Location of island well (blue arrow) is depicted.

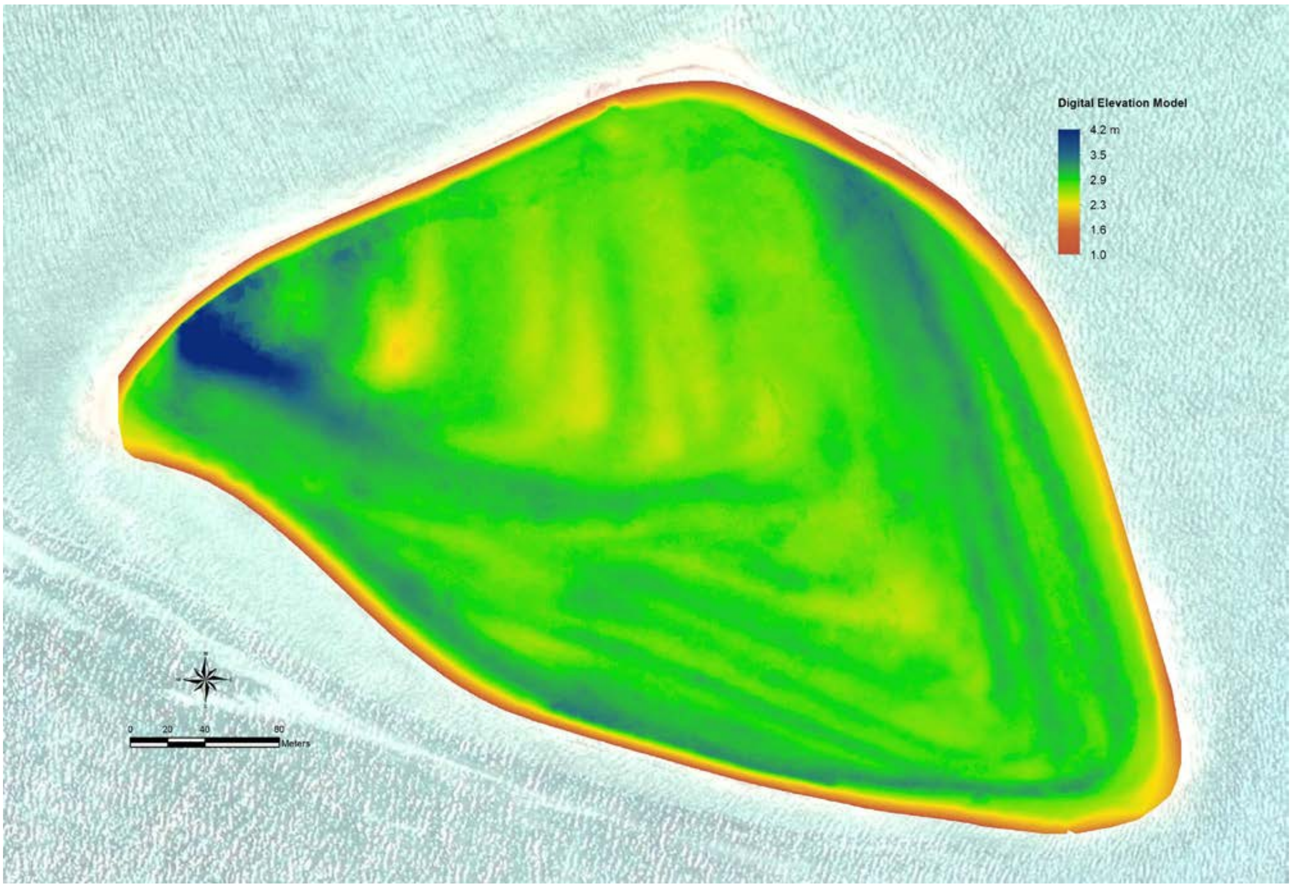


Figure 11. Digital elevation model (DEM) for East Island at Ashmore Reef in May 2019.

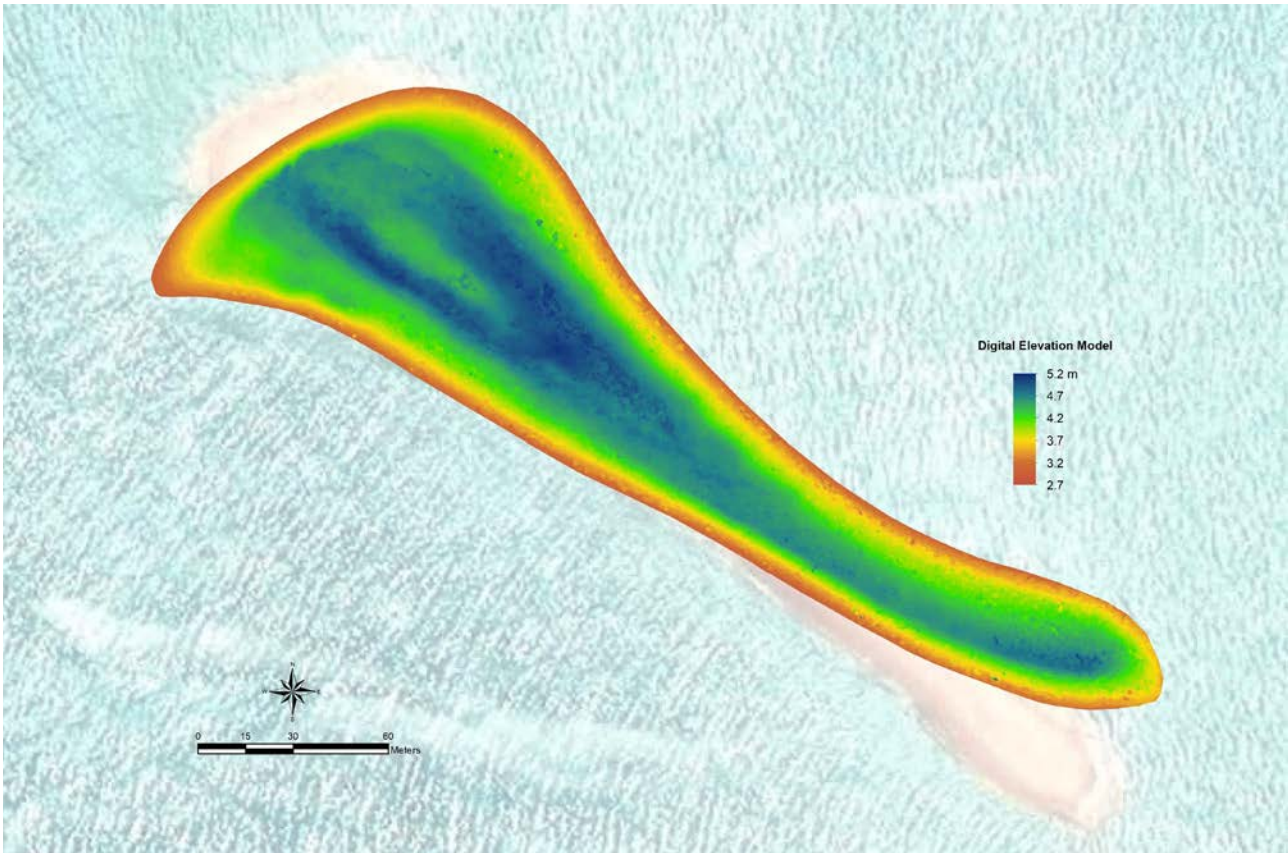


Figure 12. Digital elevation model (DEM) for Splittgerber Cay at Ashmore Reef in May 2019.

3.5.2 Species identification

Across the prior vegetation surveys undertaken at Ashmore Reef, considerable variation exists in relation to the consistency of species determinations (i.e. the assignment of taxonomic identity), the application of taxonomic nomenclature, and the determination of native or non-native status.

Species determinations

A total of 57 taxonomic determinations applied during major surveys or from herbarium specimens collected from Ashmore Reef were refined to address uncertainty, likely errors and inconsistency in earlier determinations (Table 4). Of these determinations, we considered 39 to be valid with reference to terrestrial plant species collected either currently or in the past across the four Ashmore islands. We considered a further six taxa, including *P. pilosa* and five *Boerhavia* species as being of 'uncertain' status, meaning that there is not enough evidence to be confident in these determinations as they apply to survey results or specimens from Ashmore Reef (Table 4). A further 12 names have been applied in the past that are currently invalid in regard to nomenclatural synonymy or simply misidentification (Table 4). Given these findings, we chose to consider all *Boerhavia* species together for the remainder of these surveys, recognising that delimitation at a species level may well be unrealistic and misleading.

Native status

Given the connection between non-native status and the likelihood a species will be considered for active control at Ashmore Reef, all species of dubious origin, or for which there are conflicting reports on native status, were revised and updated as appropriate, taking into account the native/non-native definition framework of Webber and Scott (2012). That is, species were considered native if the population at Ashmore Reef occurs within an appropriate distance from the species region of origin, allowing for natural dispersal potential in a given time frame (Webber & Scott 2012). Seven species were reviewed in this regard:

- *Amorphophallus paeoniifolius*: POWO (2019) states the species is native to south east Asia including Indonesia and the Northern Territory in Australia, while Hay *et al.* (1995) state that the native status in Australia and surrounding islands is uncertain. There is considerable evidence for thousands of years of human movement of the species by Austronesians (Hay & Wise 1991; Matthews 1995; Hetterscheid & Ittenbach 1996; McClatchey 2012). Recent work on genetic diversity and population structuring by Santosa *et al.* (2017) was unable to narrow down a possible native range for *A. paeoniifolius* between centres of diversity in India, Thailand and Indonesia. However, a larger scale study using nuclear and plastid sequences found very strong support for a group that included *A. paeoniifolius* and five other species centred around Thailand, Cambodia and Vietnam, suggesting a mainland south east Asian native range for the plant (Claudel *et al.* 2017).

  The majority of *Amporphophallus* species tend to be very narrow range endemics, with the notable exception of *A. paeoniifolius*. The species has fruits that are a 10-13 mm diameter red berry with 2-3 seeds and are known to be bird dispersed (Hetterscheid & Ittenbach 1996). With frugivorous birds previously observed at Ashmore Reef (albeit rarely, based on recent checklists from Avibase; http://avibase.bsc-eoc.org) that are known for their large gape width (e.g. Ptilinopus regina Swainson, 1825; Westcott *et al.* 2008) it is estimated that the 110 km flight from Roti could be completed in as little as 2-4 hours. While gut retention time (GRT) is not well known for fruit doves, the largest of the frugivorous birds known from Ashmore Reef, other Columbidae have maximum GRTs of up to 2 hours (Tassin & Barré 2010; Rehm *et al.* 2019).

Given that the species has only been observed once, that it was a single individual and that it was found close to a camp on West Island, it suggests that humans brought the species to Ashmore Reef. The plant produces tubers that are widely consumed as a starch crop, which explains the history of widespread biocultural dispersal. The theoretical possibility that the plant came from seed dispersed by birds is bordering on implausible given GRTs and flight times from Roti. Finally, it is somewhat likely that the species may not even be native to Roti, given the long history of biocultural dispersal throughout the region. **Working determination: non-native**.

- *Bulbostylis barbata*: POWO (2019) provides a very large native distribution across the old world, including Ashmore Reef. The genus has about 100 species with a pantropical to warm temperate distribution and concentrations of diversity in tropical Africa and South America (Goetghebeur 1998). Among previous surveys, Cowie (2004) and Westaway (2015) classify the plant as non-native to Ashmore Reef. There is no evidence, despite the broad distribution, to suggest that the species is not native to Indonesia (and it is also native to northern Australia).

  Seeds are 0.5-0.75 mm long with the potential for wind dispersal over long distances as well as being caught in bird feathers or feet. The plant was thought to only have arrived on West Island in c. 1994 and naturalised in close proximity to the old Dept of Territories Caretakers camp near the eastern tip of West Island. Such an association suggests introduction by humans. The current survey, however, has found this distribution to be far more widespread. Given the nearby native status and the low but feasible likelihood of spread via seabirds, we view the presence of *B. barbata* at Ashmore as plausible via natural means from a nearby native population. **Working determination: native**.

- *Cleome gynandra*: POWO (2019) lists the genus as native to Africa and south east Asia, and supposedly Western Australia and the Northern Territory in Australia (but non-native in Queensland). Iltis (1960) considers it native to the old world. Pike and Leach (1997) consider it a native of Africa and possibly Asia and that is was likely planted by Indonesian fishers on Middle Island (it is used for flavouring food, as a perfume and as a medicine). The plant census database at the Western Australian Herbarium (PERTH), which provides the nomenclature for the website FloraBase (Western Australian Herbarium 1998–), lists *C. gynandra* as an excluded name (one that has been referred to in the literature but is not actually present in WA; Parker & Biggs 2014). Kenneally (1993) listed it as native, but Pike and Leach (1997), Cowie (2004) and Westaway (2015) all consider it non-native to Ashmore Reef.

  The fruit are a 150mm long slender capsule (non-fleshy) with 1-1.5mm seeds and are likely wind dispersed. Long distance dispersal either via birds or ocean currents remains an implausible outcome between Roti and Ashmore. **Working determination: non-native**.

- *Portulaca oleracea*: This taxon is regarded as one of the world's most widespread plants, achieving this distribution with considerable assistance from human dispersal (Ridley 1930; Chapman, Stewart & Yarnell 1973; Holm *et al.* 1977). POWO (2019) claim that the original native range is northern and central Africa and the Middle East and consider it introduced in the whole of the Asia-Pacific region and Kenneally (1993) also considers it naturalised. In contrast, Pike and Leach Pike and Leach (1997), Cowie (2004) and Westaway (2015) all consider the taxon native to Ashmore. Bean (2007) considers it non-native to Australia. There is ongoing conjecture about the most likely historical native range, and the most recent insight places this ancestral range in South America (Ocampo & Columbus 2012).

Recent molecular evidence suggests that *P. oleracea* is paraphyletic (Ocampo & Columbus 2012), consisting of up to 15 possible taxa and largely supporting earlier work of that the species complex was variable and that taxa found in Australia and New Zealand differed from those in other parts of the world Danin, Baker and Baker (1979).

Pike and Leach (1997) and others point out the many cultural uses of the plant (Chapman, Stewart & Yarnell 1973; Mitich 1997). The small seeds (c. 0.5mm diam) are within a dehiscent capsule, making both bird and water dispersal possible (the latter sometimes lodged in pumice stone), and have been shown to be viable after weeks in sea water (Ridley 1930; Danin, Baker & Baker 1979). Such traits are also compatible with the species being accidently or intentionally established by human travellers to the island.

Taken together, the likely historical native range, human associations and use, and ongoing taxonomic uncertainty make this taxon as it is currently defined an implausible component of the native community at Ashmore. Inclusion of material in the next revision of the *P. oleracea* species complex would be prudent. **Working determination: non-native**.

- *Portulaca pilosa*: POWO (2019) claim that the species is native to north and south America and with a widespread non-native distribution in Africa and Australia. The only survey to assign this determination to samples from Ashmore Reef was the 1977 survey (Kenneally, 1993). The specimen (CANB 290372.1; collected from behind sand ridges on West Island) is noted as having yellow flowers and tuberous roots. It was identified as *P. pilosa* by R. Geesink (Feb 1979) and named as such in the West Island vegetation description, yet in the species list for the same publication only *P. oleracea* and *P. tuberosa* are named (Kenneally, 1993). Another collection sampled as *P. pilosa* (CANB 391564.1; D.B Carter; 22/02/1984) from Middle Island was subsequently redetermined by I.R. Telford (1993) to be *P. tuberosa*.

  Recent molecular phylogenies have determined that the ancestral origin for *P. pilosa* is South America and that the taxon is likely to be paraphyletic (Ocampo & Columbus 2012). The work also concluded that an earlier assumption that *P. tuberosa* was a synonym of *P. pilosa* was invalid Ocampo and Columbus (2012). We suspect the early records of *P. pilosa* from Ashmore Reef were a misidentification of *P. tuberosa* or *P. oleracea* or perhaps *S. portulacastrum*. However, if the early material was to be confirmed as belonging to the *P. pilosa* species complex, it should be considered non-native at Ashmore Reef. **Working determination: non-native.**

- *Tribulus cistoides:* POWO (2019) and Sheahan (2007) consider *T. cistoides* as introduced to the Americas but native to the east coast of Africa and to certain countries and islands throughout South East Asia and Australia, including Indonesia, the Philippines and Papua New Guinea. Porter (Porter 1971; 1972) considers the native range to be tropical and sub-tropical southern Africa. Kenneally (1993) considered it to be native to Ashmore Reef, while Pike and Leach (1997), Cowie (2004) and Westaway (2015) all considered it non-native during their surveys. *Tribulus cistoides* is known to hybridise with *T. terrestris* in Australia and the Galapagos Islands, and the latter species is also considered non-native in Australia (Morrison & Scott 1993; Barker 1998).

The seeds are encased in a fruit known as a caltrop or trample burr, which can embed themselves with sharp spines into animals for dispersal (including one observation in the toes of an albatross; Barker 1998; Sheahan 2007). However, human dispersal is considered the primary reason for the very wide distribution for *T. cistoides*, relative to other members of the genus (Porter 1972). While a taxonomic revision and molecular phylogeny would help to understand the historical biogeography of *T. cistoides* in south east Asia, the weight of evidence suggests that the presence of *T. cistoides* at Ashmore Reef is implausible via natural means. **Working determination: non-native.**

- *Xenostegia tridentata*: This species was recorded for the first time in the 2015 survey as a small patch on West Island (Westaway 2015). POWO (2019) considers the species to be native across much of its current range in Africa, south east Asia and Australia, and non-native in northern America. Westaway (2015) considered it native to Ashmore Reef. The other five species in the genus are endemic to central and east Africa (POWO 2019).

  The fruit of *X. tridentata* is a dry capsule of 5-7 mm in diameter containing up to 4 seeds each 3-4 mm in diameter (Simões & Staples 2017), suggesting the likelihood of bird dispersal is low. In contrast, the ethnobotanical literature suggests that *X. tridentata* has been intensely traded between Africa, India and south east Asia for at least 2,000 to 3,000 years due to its medicinal properties (Austin 2014). Moreover, the putative sub-species that delimit the African material (*X. t. tridentata*) from the Asian material (*X. t. hastata*). A detailed review of the historical biocultural dispersal and medicinal values of the taxon is presented in Austin Austin (2014). Taken together, the likelihood that the native range for the taxon is east Africa (i.e. the entire south east Asian distribution is non-native) and the low feasibility of long distance dispersal of viable seed via natural means suggest the presence of *X. tridentata* at Ashmore Reef is implausible via natural means. Further phylogenetic work on the historical biogeography of documented biocultural dispersal may help to provide clarity on the status of the species more broadly in south east Asia. **Working determination: non-native.**

Table 4. Historical variation in the consistency of species determinations, the application of taxonomic nomenclature, and notes on the determination of native or non-native status for terrestrial plants at Ashmore Reef. Name status (valid, invalid, uncertain) and origin (native, non-native) refers to interpretations applied specifically to the vegetation of Ashmore Island over past surveys and in the findings of this report. Key surveys as detailed in Table 14 and Table 15 and herbarium records as detailed in Section 3.4.5.

Family	Species	Common name	Name status	Form	Origin	Key surveys	Herb record	Notes
Aizoaceae	*Sesuvium portulacastrum* (L.) L.	sea purslane	valid	herb	native	yes	yes	Earliest known herbarium sample (PERTH 6906192; West Island, 14/9/2004) states "foreshore margin on white sand above high tide mark". Reported as common in Westaway (2015) but not in any earlier key surveys. Foliage is succulent and similar in size to that of *P. pilosa*. Flower is pink. See also section above on native status.
Amaranthaceae	*Amaranthus crispus* (Lesp. & Thevenau) A.Braun ex J.M.Coult & S.Watson		invalid	herb	n/a	no	yes	Applied to multiple herbarium specimens lodged at CANB (e.g. CANB 255978.1). Palmer (2009) states name misapplied to *Amaranthus interruptus* R.Br
Amaranthaceae	*Amaranthus interruptus* R.Br.	native amaranth	valid	herb	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Amaranthaceae	*Amaranthus undulatus* R.Br.		invalid	herb	n/a	no	yes	A single herbarium specimen (PERTH 4567404; collected by Brockway in 1995; ID by Butcher in 2010) notes it was on both East and West Island. Not found before or after in key reports
Araceae	*Amorphophallus paeoniifolius* (Dennst.) Nicolson	stinking arum	valid	herb	non-native	yes	no	Pike and Leach (1997) think an individual was planted on West Island by fisherman but died or was harvested prior to 1997. Not recorded on Ashmore in any subsequent key surveys.
Arecaceae	*Cocos nucifera* L.	coconut palm	valid	palm	non-native	yes	no	Kenneally (1993) states they were planted by Indonesian fishers but had not naturalised. Pike and Leach (1997) says Middle Island planted in 1970's West Island prior to 1980's but also 1987.
Asteraceae	*Melanthera biflora* (L.) Wild	beach sunflower	invalid	herb	native	yes	yes	Homotypic synonym applied in Pike & Leach (1997), Cowie (2004) and Westaway (2015). Accepted name now *Wollastonia biflora* (L.) DC.

Family	Species	Common name	Name status	Form	Origin	Key surveys	Herb record	Notes
Asteraceae	*Wollastonia biflora* (L.) DC.	beach sunflower	valid	herb	native	yes	yes	The synonym *Melanthera biflora* was used by Pike & Leach (1997), Cowie (2004) and Westaway (2015). Pike & Leach (1997) state it was present in 1980's but not recently due to turtle nesting. Herbarium records of 2 reproductive individuals however collected on West Island 23/05/90 (DNA D0048702 & MEL 0222411A). Westaway (2015) did not find it and states it is unlikely shrubs would be accidently overlooked.
Boraginaceae	*Argusia argentea* (L.f.) Heine	octopus bush	invalid	bush	native	yes	yes	Homotypic synonym applied in all previous major surveys and on many herbarium specimens. Accepted name is now *Heliotropium foertherianum* Diane & Hilger (Heliotropiaceae) as per Craven (2005) and Diane *et al.* (2016), but also sometimes reported as *Heliotropium arboreum* (Blanco) Mabb..
Boraginaceae	*Cordia subcordata* Lam.	sea trumpet	valid	bush	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Cleomaceae	*Cleome gynandra* L.	spiderwisp	valid	herb	non-native	yes	yes	Considered native by Keneally (1993). Pike and Leach (1979) suggest it was planted by Indonesian fishermen on Middle Island.
Convolvulaceae	*Cuscuta australis* R.Br.		invalid	creeper	n/a	yes	yes	Incorrect name misapplied to *Cuscuta victoriana* by Pike & Leach (1997) and Cowie (2004).
Convolvulaceae	*Cuscuta victoriana* Yunck.		valid	creeper	native	yes	yes	Previously (and incorrectly) referred to as *Cuscuta australis* by Pike & Leach (1997) and Cowie (2004). Herbarium record (CANB 553603.1) cites redetermination of ID by Costea, M. in 2005 based on a SEM study of flowers/seeds. Westaway (2015) subsequently refers to *C. victoriana* as parasitising *Tribulus* at Ashmore.
Convolvulaceae	*Ipomoea pes-caprae* subsp. *brasiliensis* (L.) Ooststr.	goat's foot convulvulus	valid	creeper	native	yes	yes	Status on Ashmore not specifically stated by Pike & Leach (1997). They note its distribution includes Indonesia and Australia, and that it is an aggressive coloniser of new areas, but also that it is often used by Asian herbalists, Rotinese villagers and Australian Aboriginals. Cowie (2004) considered it native.

Family	Species	Common name	Name status	Form	Origin	Key surveys	Herb record	Notes
Convolvulaceae	*Ipomoea macrantha* Roem. & Schult.	beach moonflower	invalid	creeper	n/a	yes	yes	Incorrect name misapplied to *Ipomoea violacea* L by Kenneally (1993), Pike & Leach (1997) and Cowie (2004). Pike & Leach (1997) considered *I. violacea* a synonym.
Convolvulaceae	*Ipomoea violacea* L.	beach moonflower	valid	creeper	native	yes	yes	Westaway (2015) states all Ashmore specimens previously called *I. marcantha* were misidentified and they are actually *I. violacea*.
Convolvulaceae	*Xenostegia tridentata* (L.) D.F.Austin & Staples	African morning vine	valid	herb	non-native	yes	yes	Westaway (2015) first reported this species on Ashmore (collected 13/3/2015), considering it native. Previously known as *Merremia tridentata* (L.) Hallier f. but this name has never been applied to Ashmore material. See section above on native status for further context.
Cyperaceae	*Bulbostylis barbata* (Rottb.) C.B.Clarke		valid	grass	native	yes	yes	Status on Ashmore not specifically stated in Pike & Leach (1997) but populations first found 1994 near old camps on West Is. Cowie (2004) thinks it was probably introduced and Westaway (2015) classifies it as non-native. See section above on native status for further context.
Euphorbiaceae	*Euphorbia hirta* L.	asthma plant	valid	herb	non-native	yes	yes	No changes to nomenclature or status from previous key surveys.
Fabaceae	*Caesalpinia bonduc* (L.) Roxb.	nicker nut	valid	bush	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Fabaceae	*Sesbania cannabina* (Retz.) Poir.	sesbania pea	valid	bush	native	yes	yes	Listed as non-native by Kenneally (1993). Status on Ashmore not specifically stated by Pike & Leach (1997) but they noted it is native to Australia. They suspect it is transported by birds as it occurs in the island interior. All subsequent key surveys (including this study), consider it native.
Goodeniaceae	*Scaevola sericea* Vahl	Cardwell cabbage tree	invalid	bush	native	yes	yes	Herbarium specimens (CANB 290384.1 by KF Kenneally, 1977 and CANB 373562.1 by PF Berry, 1986) initially determined as *S. sericea* (a heterotypic synonym) but this name not been used in any key survey report.
Goodeniaceae	*Scaevola taccada* (Gaertn.) Roxb.	Cardwell cabbage tree	valid	bush	native	yes	yes	No changes to nomenclature or status from previous key surveys.

Family	Species	Common name	Name status	Form	Origin	Key surveys	Herb record	Notes
Heliotropiaceae	*Heliotropium foertherianum* Diane & Hilger	octopus bush	valid	bush	native	yes	yes	First applied by L. Craven in 2005 as a determination to a 2002 Cowie herbarium specimen from West Island (CANB 553575.1). The synonym *Argusia argentea* (L.f.) Heine has been used in many past surveys. This study follows Craven (2005) and Diane et al. (2016), noting that some sources use the synonym *Heliotropium arboreum* (Blanco) Mabb..
Lauraceae	*Cassytha filiformis* L.	dodder laurel	invalid	creeper	n/a	yes	no	Pike & Leach (1997) are the only surveyors to list *C. filiformis*. Their drawings and descriptions match that of *C. filiformis*. However, their description of the plant's habitat and location fits that of the morphologically similar *Cuscuta victoriana* (also commonly called 'dodder') that has been reported by Cowie (2004), Westaway (2015) and this study (2019). Pike and Leach (1997) state their plant is a "parasitic plant that creates bare areas that are subsequently used for nesting sites", occurring on East Island initially but then spreading to Middle Island prior to 1992. There are no herbarium specimens lodged for *C. filiformis* but numerous for *C. victoriana*. We propose that Pike and Leach (1997) misidentified the dodder species.
Malvaceae	*Sida pusilla* Cav.		valid	herb	native	yes	yes	Status on Ashmore not specifically stated in Pike & Leach (1997), however they state it is widespread on islands throughout the northern Indian and western Pacific oceans and POWO (2020) shows it to be native throughout this region. All other key reports also classify it as native.
Nyctaginaceae	*Boerhavia albiflora* Fosberg		uncertain	herb	native	yes	no	Listed in Kenneally (1993) but no known herbarium specimens lodged from that survey. Pike and Leach (1997) looked for this species as it was listed in Du Puy Du Puy and Telford (1993) as being on Ashmore, however they concluded it was not present. The description of *B. glabrata* in Chen and Wu (2007)) fits the taxon referred to as *B. albiflora* in this study, in that it also had white flowers in clusters and larger leaves mixed with smaller ones.

Family	Species	Common name	Name status	Form	Origin	Key surveys	Herb record	Notes
Nyctaginaceae	*Boerhavia burbidgeana* Hewson		valid	herb	native	yes	yes	This is likely one of the most common species of *Boerhavia* on West Is. It is generally prostate with thin branching stems, terminal pink flowers and long narrow leaves. 18 herbarium specimens have been determined as *B. burbidgeana* between 1983 and 2015. Possible confusion may exist separating this species from others because some young plants have broad basal leaves (broad leaves are a characteristic of *B. gardneri*) and some specimens have white flowers (collected by Mitchell in 2003 and Westaway in 2015).
Nyctaginaceae	*Boerhavia diffusa* L.		uncertain	herb	native	yes	yes	Meikle and Hewson (1984) state "This species does not occur in Australia. The name has been widely misapplied.". Numerous ALA records however state specimens considered to be other species, were re-determined to *B. diffusa*, including CANB 9400801 (Kenneally 1977 West Is. sampled as *B. glabrata*, re-determined by Lally (26/05/05) to be *B. diffusa*), CANB 8704795.1 (Hinchley 1987 East Is. Sampled as *B. glabrata* redetermined by Lally (2005) to be *B. diffusa*) and CANB 555211.1; CANB 555234.1 Cowie 2002 West and East Is. sampled as *B. repens*, redetermined by Lally (2005) to be *B. diffusa*).
Nyctaginaceae	*Boerhavia dominii* Meikle & Hewson		uncertain	herb	native	no	yes	Not mentioned in any key survey. Herbarium sample CBG 9400801.2 Hicks 1983 and CANB 391572.1 Carter 1984 sampled as *B. dominii* but subsequently redetermined to be *B diffusa* and *B repens*. A third specimen (PERTH 3180085 PF Berry 1986) sampled as *B. dominii* is also likely to be a misidentification.
Nyctaginaceae	*Boerhavia gardneri* Hewson		uncertain	herb	native	yes	yes	Similar in form and habit to B. *burbidgeana* but with ovate/elliptic leaves. (Meikle & Hewson 2020) separated it from *B. burbidgeana* based on the flowers being in glomerules or umbels and not large and diffuse as in *B. gardneri* (vs inflorescence a large diffuse cyme; flowers usually solitary in *B. burbidgeana).* Westaway (2015) lists it as common on West Island, but there were no samples lodged. The only herbarium specimens of *B. gardneri* are

Family	Species	Common name	Name status	Form	Origin	Key surveys	Herb record	Notes
								by Brockway in 1995 West Is. (PERTH 4567323; PERTH 4567358; PERTH 4567412 determined by AA Mitchell in 1995). Specimens collected the same time and island and lodged at CANB (CANB 541641.1; CANB 541645.1) were determined to be *B. diffusa* by Lally in 2005.
Nyctaginaceae	*Boerhavia glabrata* Blume		uncertain	herb	native	yes	yes	This species, if present is not common, noting herbarium specimens collected between 1977 and 2003 from West, Middle and East Islands have only been lodged with PERTH and DNA.
Nyctaginaceae	*Boerhavia repens* L.		valid	herb	native	yes	yes	This species is more robust than the other *Boerhavia* species at Ashmore and with a prostrate habit, making it easier to identify. There are numerous herbarium specimens with *B. repens* determinations listed in ALA.
Pandanaceae	*Pandanus* Parkinson sp.		valid	palm	native	yes	no	First described from this survey, see Section 3.5.3
Poaceae	*Cenchrus brownii* Roem. & Schult.	burr grass	valid	grass	non-native	yes	yes	Specimens have been collected from East and West islands, some of which have been re-determined after being sampled as *C. echinatus* (CBG 8600868.1 A. Grant, Middle Is. 1986 & 8405469 J Hicks, East Is. 1984).
Poaceae	*Cenchrus ciliaris* L.	buffel grass	valid	grass	non-native	yes	yes	Specimens of *C. ciliaris* have been collected from East and West island (e.g. CANB 255604.1 KF Kenneally, West island 1977, determination by Lazarides 1978).
Poaceae	*Cenchrus echinatus* L.	innocent weed	valid	grass	non-native	yes	yes	Note that some herbarium samples (CBG 8600868 Middle Is. 1986 & 8405469 1984 East Is.) sampled as *C. echinatus* were subsequently redetermined (by IR Telford, 1989) to be *C. brownii*.
Poaceae	*Cenchrus pedicellatus* (Trin.) Morrone	annual mission grass	valid	grass	non-native	yes	yes	The synonym *Pennisetum pedicellatum* was used by Cowie (2004).
Poaceae	*Pennisetum pedicellatum* Trin.	annual mission grass	invalid	grass	non-native	yes	yes	Homotypic synonym applied in Cowie (2004). Accepted name now *Cenchrus pedicellatus*.

Family	Species	Common name	Name status	Form	Origin	Key surveys	Herb record	Notes
Poaceae	*Digitaria mariannensis* Merr.	finger grass	valid	grass	native	yes	yes	Status on Ashmore not specifically stated by Pike & Leach (1997) who considered it native to Indonesia but not established on Australian mainland. All subsequent reports list it as native.
Poaceae	*Eragrostis amabilis* (L.) Wight & Arn. ex Nees	delicate love grass	valid	grass	non-native	yes	yes	The heterotypic synonym *Eragrostis tenella* was used by Pike & Leach (1997), Cowie (2004) and Westaway (2015).
Poaceae	*Eragrostis tenella* (L.) P.Beauv. ex Roem. and Schult.	delicate love grass	invalid	grass	non-native	yes	yes	Heterotypic synonym applied in Pike & Leach (1997), Cowie (2004) and Westaway (2015). Accepted name now *Eragrostis amabilis.*
Poaceae	*Eragrostis cilianensis* (All.) Vignolo ex Janch.	stinkgrass	invalid	grass	n/a	yes	no	Herbarium sample (CANB 598097 AA Mitchell 2003) initially sampled as *E. cilianensis* but subsequently redetermined to be *E. cumingii*
Poaceae	*Eragrostis cumingii* Steud.	Cuming's love grass	valid	grass	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Poaceae	*Eragrostis elongata* (Willd.) J.Jacq.		invalid	grass	n/a	yes	no	Pike & Leach (1997) state that the sample collected in 1977 was mis-identified and redetermined to be *E. cumingii*
Poaceae	*Lepturus repens* (G.Forst.) R.Br.	stalky grass	valid	grass	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Poaceae	*Spinifex littoreus* (Burm.f.) Merr.		valid	grass	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Poaceae	*Spinifex longifolius* R.Br.		valid	grass	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Poaceae	*Sporobolus virginicus* (L.) Kunth	sand couch	valid	grass	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Poaceae	*Zea mays* L.	maize (corn)	valid	grass	non-native	yes	yes	No changes to nomenclature or status from previous key surveys.
Portulacaceae	*Portulaca oleracea* L.	purslane	valid	herb	non-native	yes	yes	Considered native by Cowie (2004) and Westaway (2015). See section above on native status for further context.

Family	Species	Common name	Name status	Form	Origin	Key surveys	Herb record	Notes
Portulacaceae	*Portulaca pilosa* L.	hairy pigweed (yellow flower)	uncertain	herb	non-native	yes	yes	Herbarium specimen (CANB 290372.1) collected by Kenneally in 1977 behind sand ridges on West Island had yellow flowers and tuberous roots. The accepted form of *P. pilosa* has pink flowers. There are no records of *P. pilosa* from Ashmore after 1984. We suspect these early records were a misidentification of *P. tuberosa* or *P. oleracea* or perhaps *S. portulacastrum*. See section above on native status for further context.
Portulacaceae	*Portulaca tuberosa* Roxb.		valid	herb	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Rhizophoraceae	*Rhizophora stylosa* Griff.	spider mangrove	valid	tree	native	yes	no	No changes to nomenclature or status from previous key surveys.
Rubiaceae	*Guettarda speciosa* L.	fish plate shrub	valid	bush	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Surianaceae	*Suriana maritima* L.	bay cedar	valid	bush	native	yes	yes	No changes to nomenclature or status from previous key surveys.
Zygophyllaceae	*Tribulus cistoides* L.	beach caltrop	valid	herb	non-native	yes	yes	Considered non-native by Pike & Leach (1997) and Cowie (2004) but native by Kenneally (1993). See section above on native status for further context.

3.5.3 Plant diversity

Over all four islands 28 vascular plant taxa, as well as what was likely to be an additional five *Boerhavia* species, were recorded as present and with living individuals during the survey (Table 5). West Island was the most speciose of the islands, with 23 taxa present, compared to Middle Island (13 taxa), East Island (9 taxa) and Splittgerber Cay (3 taxa). Only three taxa – the grasses *E. cumingii* and *L. repens* and the non-native herb *P. oleracea* – were found on all four islands, while 16 species were only found on one of the four islands (all on West Island with the exception of the non-native *C. gynandra* found only on Middle Island; Table 6).

Table 5. Terrestrial plant diversity observed at Ashmore Reef across the four islands – West Island, Middle Island, East Island and Splittgerber Cay. *: Due to ongoing taxonomic uncertainty, the multiple *Boerhavia* species observed during the survey were excluded from this table.

Island	West Island	Middle Island	East Island	Splittgerber Cay	All islands
Native species	17*	10	7	2	20*
Non-native species	6	3	2	1	8
TOTAL	**23***	**13**	**9**	**3**	**28***

Table 6. Vascular plants recorded during terrestrial surveys at Ashmore Reef during the 2019 surveys across the four islands – West Island, Middle Island, East Island and Splittgerber Cay. Due to ongoing taxonomic uncertainty, the multiple *Boerhavia* species observed were treated as a single entity. Island population metrics: A: adult, J: juvenile, P: mixed age population, D: dead; 0: no plants found, 1: single individual, 2: a few individuals or localised patches, 3: common &/or widespread. *: Non-native species

Species	Island presence			
	West	Middle	East	Splittgerber
Amaranthus interruptus	0	P3	P3	0
Boerhavia spp.	P3	0	0	0
Bulbostylis barbata	P3	0	0	0
*Cenchrus brownii**	P3	0	0	0
*Cenchrus ciliaris**	P2	0	0	0
*Cenchrus echinatus**	P2	0	0	0
*Cenchrus pedicellatus**	P2	0	0	0
*Cleome gynandra**	0	P3	0	0
Cordia subcordata	A1	0	0	0
Cuscuta victoriana	0	P2	P3	0
Digitaria mariannensis	P3	P3	0	0
Eragrostis cumingii	P2	P3	P3	A1
Guettarda speciosa	A2	0	0	0
Heliotropium foertherianum	P3	A1	0	0

Species	Island presence			
	West	**Middle**	**East**	**Splittgerber**
Ipomoea pes-caprae	P3	0	0	0
Ipomoea violacea	P3	0	0	0
Lepturus repens	P3	P3	P3	P3
Pandanus sp	J1	0	0	0
*Portulaca oleracea**	P2	P3	P3	P3
Portulaca tuberosa	P3	P2	0	0
Sesbania cannabina	P3	P3	P3	0
Sesuvium portulacastrum	P3	0	0	0
Sida pusilla	P3	P3	P3	0
Spinifex littoreus	P2	0	0	0
Spinifex longifolius	P3	P2	0	0
Sporobolus virginicus	0	0	P3	0
Suriana maritima	A2+J1	0	0	0
*Tribulus cistoides**	0	P3	P3	0
*Xenostegia tridentata**	P2	0	0	0

Native plants

A total of 20 native plant taxa were recorded in the survey, in addition to multiple native *Boerhavia* species that could not be identified to species level with confidence (Table 5). Based on our own determinations, we considered *B. albiflora*, *B. burbidgeana*, *B. diffusa*, *B. gardneri* and *B. repens* to be present on West Island, with *B. burbidgeana* and *B. gardneri* the most widespread. West Island had the greatest native plant diversity, with 17 species. Splittgerber Cay, only colonised by vegetation since 2009, had just two native species present, including *E. cumingii* represented by a single individual. A record from West Island of a single *Pandanus* seedling establishing near the shrub line above the high tide mark was the only terrestrial plant species observed that had not previously been recorded from Ashmore Reef (Figure 13). It was not possible to identify this individual to species level.

Non-native plants

A total of 8 non-native plant taxa were recorded in the survey as naturalised populations (Table 5). On West Island non-native species included four *Cenchrus* grasses – *C. brownii*, *C. ciliaris*, *C. echinatus* and *C. pedicellatus* – as well as *X. tridentata* and *P. oleracea* (the latter found on all four islands). *Cleome gynandra* was restricted to Middle Island, while *T. cistoides* was found across both Middle and East islands. *Bulbostylis barbata*, considered by some earlier surveys to be non-native (but treated in this survey as native), was recorded at both the west and east end of West Island. Lastly, a single standing and multiple fallen trunks from dead *C. nucifera* were recorded on West and Middle islands. None of the 10 weed species identified as targets with a high risk of establishing and having negative impacts (Table 1) were observed in this survey.

Figure 13 . The only new terrestrial plant species to be found at Ashmore Reef was a single record of a Pandanus seedling establishing close to the shrub line on the southern facing beach of West Island. Photo: Bruce Webber

3.5.4 Community characterisation

We identified 14, 10, 10 and 1 distinct terrestrial vegetation communities across West Island, Middle Island, East Island and Splittgerber Cay, respectively (Figure 14 to Figure 17). For the 35 communities identified at Ashmore Reef, no single community was found across multiple islands, with each differing in community composition and/or taxon abundance (Tables 7 to 10).

Community complexity varied considerably, with up to 11 species (Community Wf) found at contrasting overall abundance. Four communities were single species monocultures, including two dominated by spinifex (*S. longifolius* and *S. littoreus*; Tables 7 to 10). The size of each community fragment on the islands varied considerably. Some communities were highly fragmented, such as that dominated by *C. brownii* on West Island (Wn; 30 discrete fragments) and *T. cistoides* on East Island (Ec; 63 discrete fragments; Figure 14 to Figure 17).

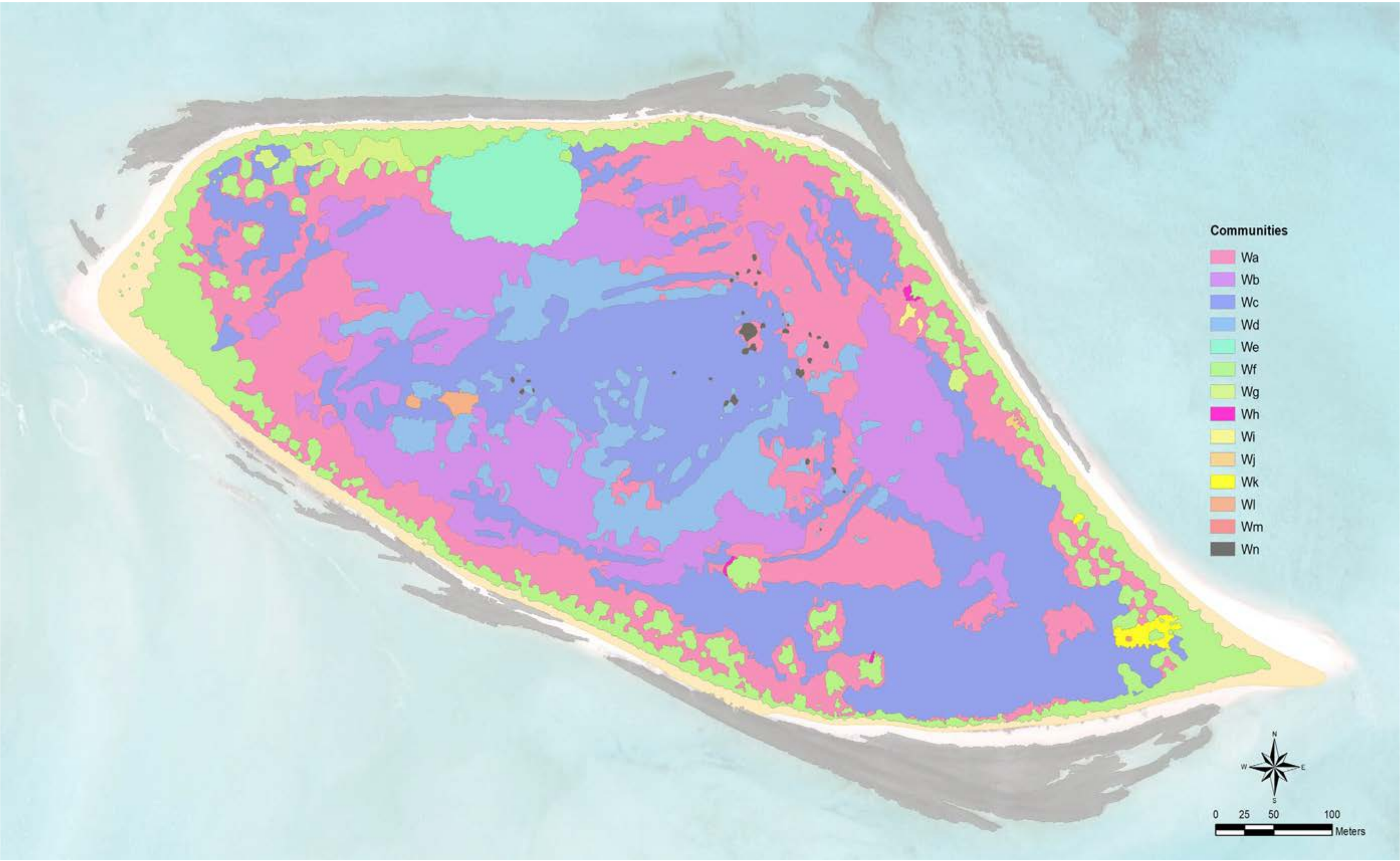


Figure 14. Vegetation community map of West Island at Ashmore Reef in May 2019. Species composition and abundance for each of the 14 communities is detailed in Table 8. Associated island shoreline (cream-yellow) and surrounding rock shelf structure shelf structure (light grey) is also depicted.

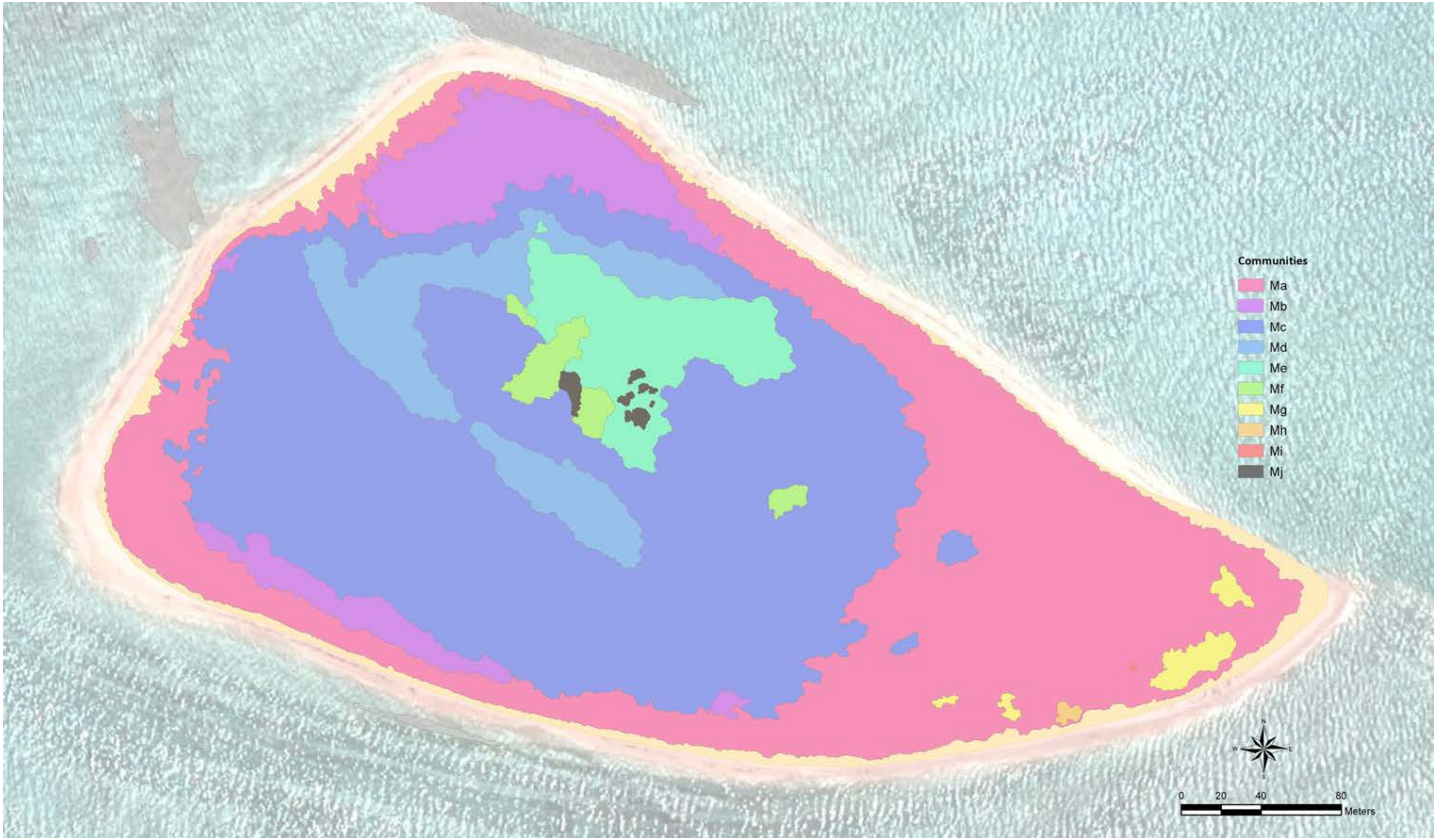


Figure 15. Vegetation community map of Middle Island at Ashmore Reef in May 2019. Species composition and abundance for each of the 10 communities is detailed in Table Table 8. Associated island shoreline (cream-yellow) and surrounding rock shelf structure (light grey) is also depicted.

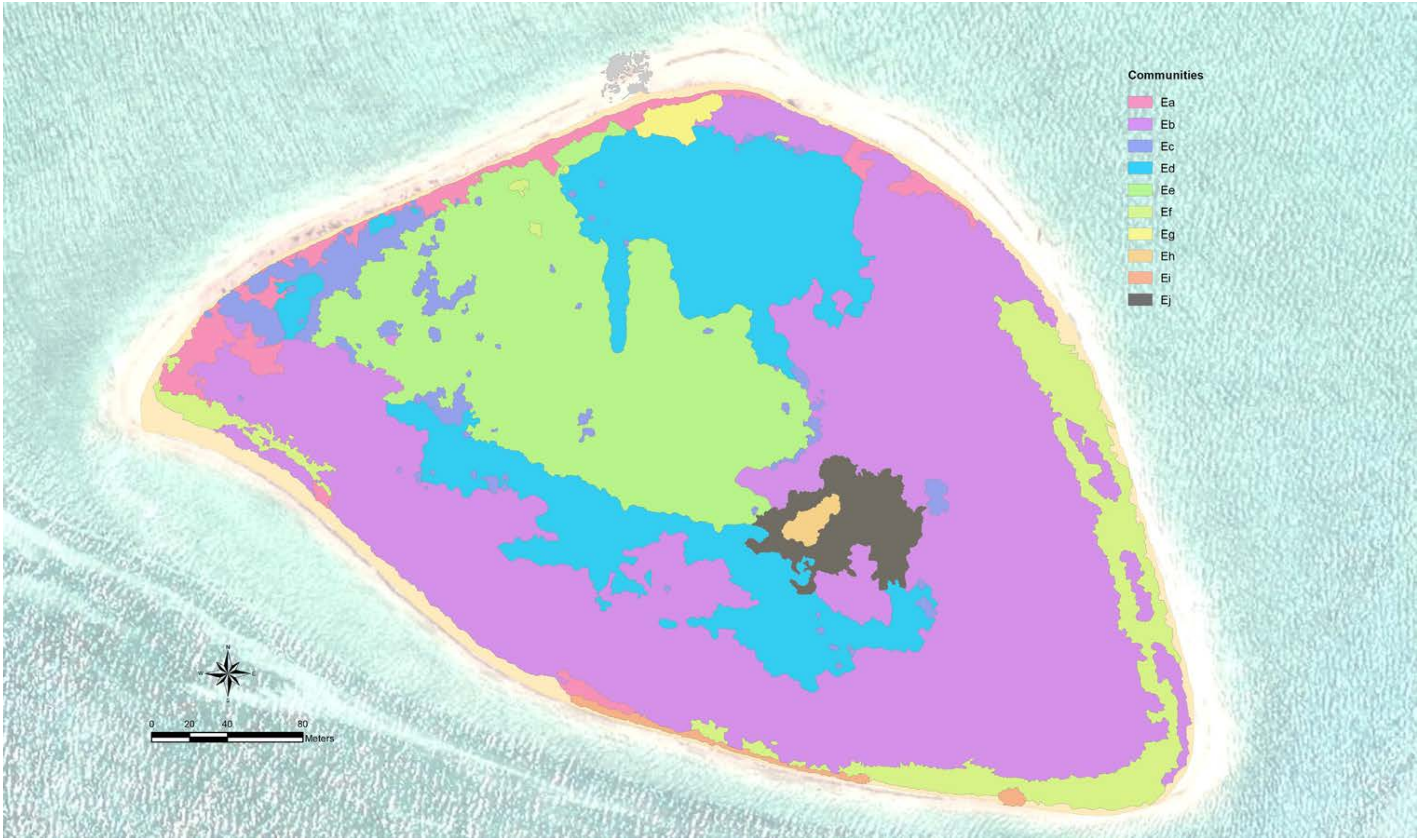


Figure 16. Vegetation community map of East Island at Ashmore Reef in May 2019. Species composition and abundance for each of the 10 communities is detailed in Table 9. Associated island shoreline (cream-yellow) and a fragmented concrete helipad (light grey) is also depicted.

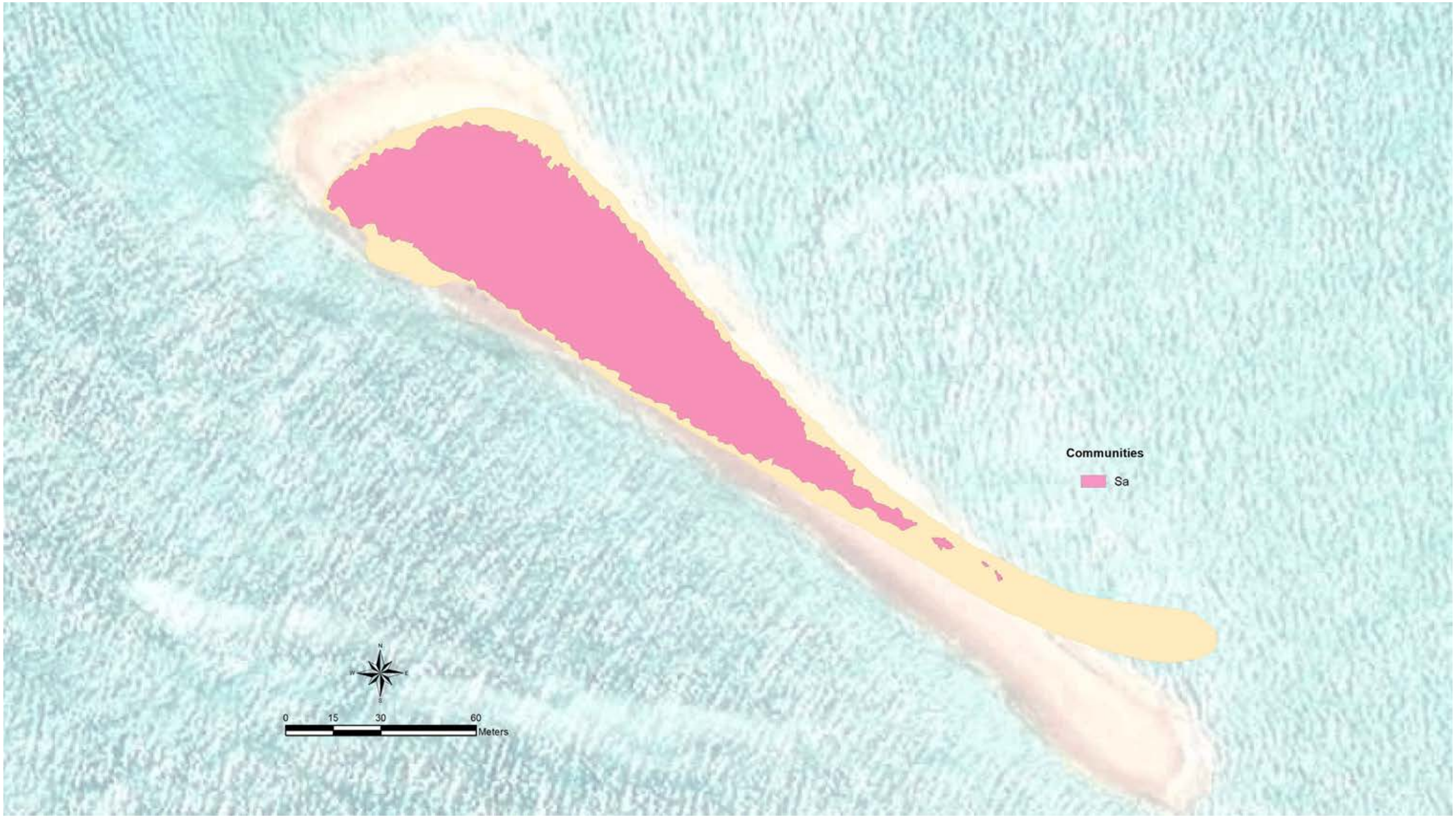


Figure 17. Vegetation community map of Splittgerber Cay at Ashmore Reef in May 2019. Species composition and abundance for the single plant community is detailed in Table 10. Associated island shoreline (cream-yellow) is also depicted.

Table 7. Community composition for the 14 vegetation communities identified and surveyed on West Island, Ashmore Reef, with total area occupied and number of community fragments detailed. Abundance categories follow a modified Daubenmire cover scale (Daubenmire 1959).

Code	Community composition
Wa	DOMINANT: *Digitaria mariannensis* (finger grass); COMMON OR LOCALLY DENSE: *Boerhavia* spp. (native tar vine), *Lepturus repens* (stalky grass) near shrubs on edges, *Portulaca oleracea* (purslane), *Portulaca tuberosa* (‘small leaf’ portulaca), *Sesbania cannabina* (sesbania pea) to 1.2m high, locally dense, mainly towards the interior, *Sida pusilla*; SCA TTERED OR LOCALLY COMMON: *Ipomoea violacea* (beach moonflower) usually overtopping other species.
Wb	DOMINANT: *Digitaria mariannensis* (finger grass) to 60cm high; COMMON OR LOCALLY DENSE: *Boerhavia* spp. (native tar vine), *Ipomoea violacea* (beach moonflower) overtopping, *Portulaca oleracea* (purslane), *Sida pusilla*; SCATTERED OR LOCALLY COMMON: *Sesbania cannabina* (sesbania pea); OCCASIONAL: *Eragrostis cumingii* (Cuming's love grass).
Wc	DOMINANT: *Sida pusilla*; SCATTERED: *Boerhavia* spp. (native tar vine) sometimes locally common, *Digitaria mariannensis* (finger grass), *Ipomoea violacea* (beach moonflower); RARE: *Portulaca tuberosa* (‘small leaf’ portulaca).
Wd	DOMINANT: *Sesbania cannabina* (sesbania pea) to 1.5m high; COMMON: *Ipomoea violacea* (beach moonflower), *Sida pusilla*; SCATTERED: *Digitaria mariannensis* (finger grass) underneath.
We	MONOCULTURE: *Spinifex longifolius* (soft leaf spinifex).
Wf	CO-DOMINANT: *Heliotropium foertherianum* (octopus bush) as a 2-3m tall shrub, *Lepturus repens* (stalky grass) in the ground layer underneath shrubs; SCATTERED & LOCALLY COMMON at times: *Digitaria mariannensis* (finger grass) scattered between shrubs, *Guettarda speciosa* (fish plate shrub) to 2-3m high, *Sesuvium portulacastrum* (shoreline purslane) in large clumps on foredune; SCATTERED: *Portulaca tuberosa* (‘small leaf’ portulaca); RARE: *Cordia subcordata* (sea trumpet) one individual; *Ipomoea pes-caprae* (goat's foot convolvulus) one individual, *Ipomoea violacea* (beach moonflower), *Pandanus* spp. single juvenile individual found near shoreline), *Suriana maritima* (3 individuals).
Wg	CO-DOMINANT: *Bulbostylis barbata* (dainty sedge), *Digitaria mariannensis* (finger grass); COMMON: *Portulaca tuberosa* (‘small leaf’ portulaca), *Sida pusilla*; SCATTERED: *Lepturus repens* (stalky grass) near shrubs on edges; OCCASIONAL: *Boerhavia* spp. (native tar vine) towards the interior.
Wh	DOMINANT: *Cenchrus echinatus* (innocent weed); COMMON: *Digitaria mariannensis* (finger grass), *Ipomoea violacea* (beach moonflower) overtopping, *Lepturus repens* (stalky grass); SCATTERED: *Sida pusilla*; OCCASIONAL: *Boerhavia* spp. (native tar vine).
Wi	CO-DOMINANT: *Digitaria mariannensis* (finger grass), *Lepturus repens* (stalky grass); SCATTERED OR LOCALLY COMMON: *Boerhavia* spp. (native tar vine), *Ipomoea violacea* (beach moonflower), *Sida pusilla*, *Xenostegia tridentata* (African morning vine).
Wj	MONOCULTURE: *Spinifex littoreus* (‘spiky’ spinifex).
Wk	MONOCULTURE: *Cenchrus ciliaris* (buffel grass); RARE: *Digitaria mariannensis* (finger grass) on the periphery.
Wl	MONOCULTURE: *Eragrostis cumingii* (love grass); OCCASIONAL: *Sida pusilla*.
Wm	MONOCULTURE: *Cenchrus pedicellatus* (annual mission grass).
Wn	DOMINANT: *Cenchrus brownii* (burr grass); SCATTERED or LOCALLY COMMON: *Digitaria mariannensis* (finger grass); SCATTERED: *Sida pusilla*; OCCASSIONAL: *Boerhavia* spp. (native tar vine); RARE: *Ipomoea violacea* (beach moonflower).

Table 8. Community composition for the 10 vegetation communities identified and surveyed on Middle Island, Ashmore Reef, with total area occupied and number of community fragments detailed. Abundance categories follow a modified Daubenmire cover scale (Daubenmire 1959).

Code	Community composition
Ma	DOMINANT: *Lepturus repens* (stalky grass); SCATTERED & LOCALLY COMMON: *Portulaca oleracea* (purslane), *Portulaca tuberosa* ('small leaf' portulaca); OCCASIONAL: *Amaranthus interruptus* (native amaranth), *Eragrostis cumingii* (Cuming's love grass), *Sesbania cannabina* (sesbania pea).
Mb	DOMINANT: *Eragrostis cumingii* (Cuming's love grass); OCCASSIONAL: *Amaranthus interruptus* (native amaranth), *Lepturus repens* (stalky grass), *Sida pusilla*.
Mc	DOMINANT: *Amaranthus interruptus* (native amaranth); SCATTERED: *Cleome gynandra* (spiderwisp), *Portulaca oleracea* (purslane), *Sida pusilla*; OCCASIONAL: *Sesbania cannabina* (sesbania pea).
Md	DOMINANT: *Amaranthus interruptus* (native amaranth); COMMON: *Sesbania cannabina* (sesbania pea; seedlings 10 (-30) cm tall); SCATTERED: *Eragrostis cumingii* (Cuming's love grass; healthy).
Me	CO-DOMINANT: *Amaranthus interruptus* (native amaranth) seedlings 10 to 50cm tall, *Sesbania cannabina* (sesbania pea) seedlings 10 to 50cm tall; SCATTERED & LOCALLY COMMON: *Cleome gynandra* (spiderwisp); OCCASIONAL: *Sida pusilla*.
Mf	DOMINANT: *Cleome gynandra* (spiderwisp) to 80cm tall; SCATTERED: *Amaranthus interruptus* (native amaranth), *Sesbania cannabina* (sesbania pea), *Sida pusilla*.
Mg	DOMINANT: *Digitaria mariannensis* (finger grass); COMMON: *Portulaca oleracea* (purslane); SCATTERED: *Lepturus repens* (stalky grass), *Portulaca tuberosa* ('small leaf' portulaca).
Mh	DOMINANT: *Spinifex longifolius* ('soft leaf' spinifex); SCATTERED: *Lepturus repens* (stalky grass), *Portulaca oleracea* (purslane), *Portulaca tuberosa* ('small leaf' portulaca).
Mi	DOMINANT: *Heliotropium foertherianum* (octopus bush) a single live individual.
Mj	DOMINANT: *Tribulus cistoides* (beach caltrop); LOCALLY DENSE: *Cuscuta victoriana*; SCATTERED: *Amaranthus interruptus* (native amaranth), *Cleome gynandra* (spiderwisp), *Sesbania cannabina* (sesbania pea).

Table 9. Community composition for the 10 vegetation communities identified and surveyed on East Island, Ashmore Reef, with total area occupied and number of community fragments detailed. Abundance categories follow a modified Daubenmire cover scale (Daubenmire 1959).

Code	Community composition
Ea	SCATTERED: *Portulaca oleracea* (purslane) over open sand; RARE: *Sesbania cannabina* (sesbania pea).
Eb	DOMINANT (close to a monoculture in most places): *Eragrostis cumingii* (Cuming's love grass); SCATTERED: *Amaranthus interruptus* (native amaranth), *Portulaca oleracea* (purslane), *Sesbania cannabina* (sesbania pea), *Tribulus cistoides* (beach caltrop) in patches).
Ec	DOMINANT (close to a monoculture): *Tribulus cistoides* (beach caltrop) to 40cm tall; SCATTERED: *Amaranthus interruptus* (native amaranth), *Eragrostis cumingii* (Cuming's love grass), *Lepturus repens* (stalky grass).
Ed	CO-DOMINANT: *Amaranthus interruptus* (native amaranth), *Eragrostis cumingii* (Cuming's love grass); SCATTERED: *Lepturus repens* (stalky grass), *Sesbania cannabina* (sesbania pea) seedlings only), *Tribulus cistoides* (beach caltrop).
Ee	DOMINANT: *Amaranthus interruptus* (native amaranth); SCATTERED: *Eragrostis cumingii* (Cuming's love grass), *Sesbania cannabina* (sesbania pea) seedlings only, *Tribulus cistoides* (beach caltrop).
Ef	DOMINANT: *Lepturus repens* (stalky grass); SCATTERED: *Eragrostis cumingii* (Cuming's love grass), *Portulaca oleracea* (purslane).
Eg	DOMINANT: *Amaranthus interruptus* (native amaranth) to 50cm tall; COMMON: *Sesbania cannabina* (sesbania pea); SCATTERED: *Eragrostis cumingii* (Cuming's love grass), *Lepturus repens* (stalky grass), *Portulaca oleracea* (purslane).
Eh	DOMINANT: *Sporobolus virginicus* (sand couch); LOCALLY DENSE: *Tribulus cistoides* (beach caltrop); SCATTERED: *Amaranthus interruptus* (native amaranth), *Cuscuta victoriana* mostly unhealthy, growing over *T. cistoides*.
Ei	DOMINANT: *Sporobolus virginicus* (sand couch); SCATTERED: *Portulaca oleracea* (purslane), *Eragrostis cumingii* (Cuming's love grass), *Lepturus repens* (stalky grass).
Ej	DOMINANT: *Tribulus cistoides* (beach caltrop); SCATTERED: *Cuscuta victoriana* mostly unhealthy, growing over *T. cistoides*; LOCALLY DENSE: *Portulaca oleracea* (purslane) very healthy, *Sida pusilla*; SCATTERED: *Amaranthus interruptus* (native amaranth), *Eragrostis cumingii* (Cuming's love grass).

Table 10. Community composition for the single vegetation communities identified and surveyed on Splittgerber Cay, Ashmore Reef, with total area occupied and number of community fragments detailed. Abundance categories follow a modified Daubenmire cover scale (Daubenmire 1959).

Code	Community composition
Sa	DOMINANT: *Lepturus repens* (stalky grass); COMMON: *Portulaca oleracea* (purslane); RARE: *Eragrostis cumingii* (Cuming's love grass) one individual.

3.5.5 Species abundance

Quadrat surveys

Considerable variation in canopy cover was documented for individual plant species both native and non-native and within and between communities across the four islands (SI Figure 1 and SI Figure 2). When pooled across species, East Island had the highest overall canopy cover, with Splittgerber Cay the lowest cover (Figure 18). The grass-dominated communities had the highest canopy cover values, while the community dominated by *B. barbata* on West Island (Wg) had the lowest canopy cover values (15.9%; SI Table. 2 to SI Table. 4).

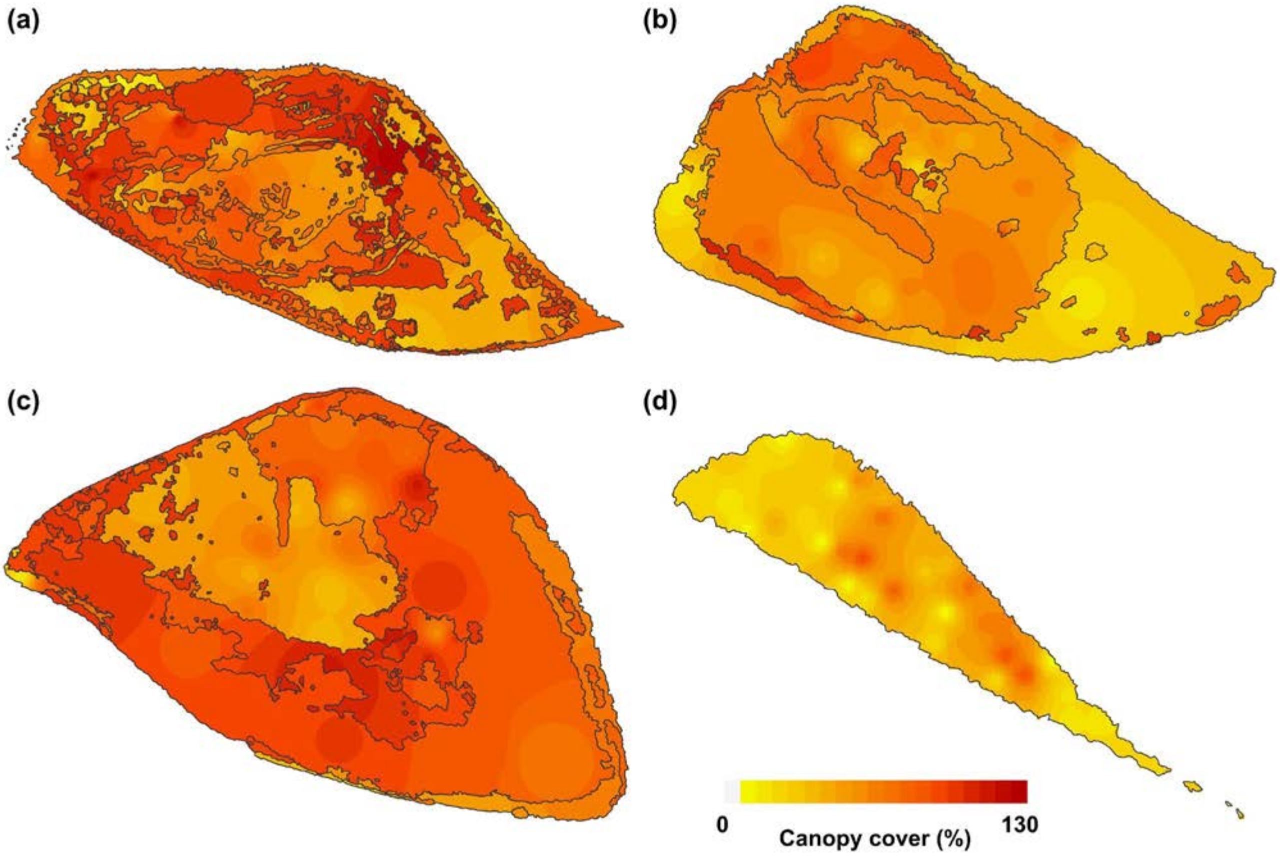


Figure 18. Canopy cover abundance scores for all plant species surveyed the four islands at Ashmore Reef in May 2019 – West Island (a), Middle Island (b), East Island (c) and Splittgerber Cay (d).

When grouped into native and non-native species within the various communities, native components of the communities were spread widely over the four islands at moderate to high abundance (Figure 19. Low overall native species density corresponded either with high non-native species density (e.g. Community Wk), or a high proportion of bare ground (e.g. Community Ma). In contrast, the distribution of non-native species was either large areas of low abundance cover, a pattern explained by a single species, *P. oleracea* (e.g. Community Eb), or small areas of very high abundance (often near-monocultures; e.g. Community Ec; Figure 20).

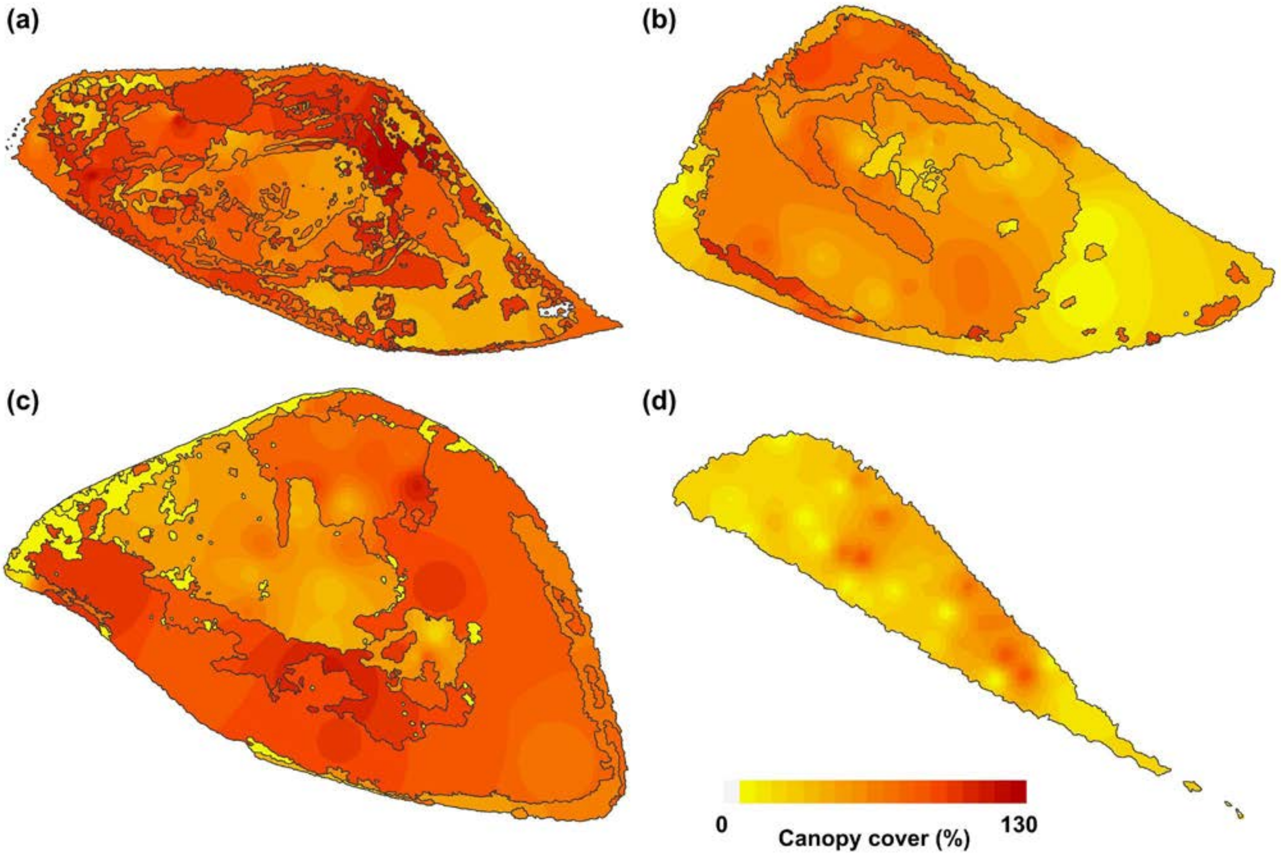


Figure 19. Canopy cover abundance scores for all native plant species surveyed the four islands at Ashmore Reef in May 2019 – West Island (a), Middle Island (b), East Island (c) and Splittgerber Cay (d).

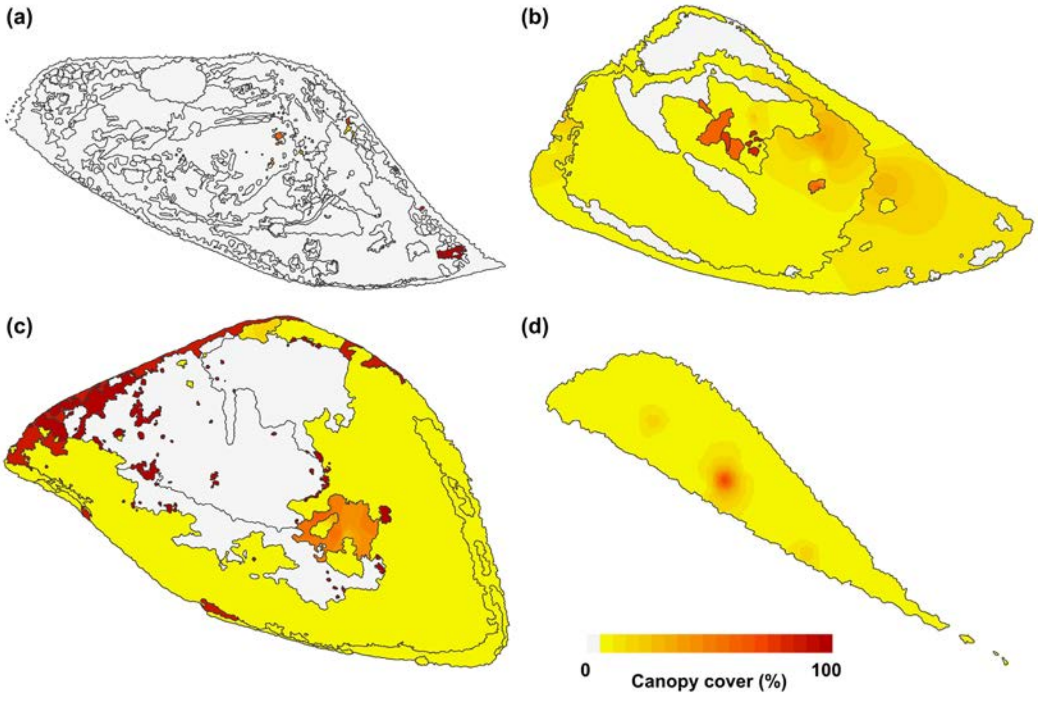


Figure 20. Canopy cover abundance scores for all non-native plant species surveyed the four islands at Ashmore Reef in May 2019 – West Island (a), Middle Island (b), East Island (c) and Splittgerber Cay (d).

Sesbania cannabina had the largest distribution across all four islands at 363,447 m^2, but occurred at an average canopy cover of only 6.5% (Table 11). For plants other than trees and shrubs, *S. pusilla* (298,367 m^2), *P. oleracea* (295,835 m^2), *D. mariannensis* (262,781 m^2) and *I. violacea* (258,915 m^2) also had large distributions, but again occurred at low canopy cover values (6.6 to 26.2%. In contrast, the two spinifex species had smaller ranges (8,722 and 115 m^2 for *S. longifolius* and *S. littoreus*, respectively) but had much higher canopy cover (98.0 and 92.5%, respectively) where found. The four non-native *Cenchrus* species have a range of between 16 and 529 m^2, yet had a canopy cover of between 48.2% and 95%. Therefore, the overall area required to be controlled (i.e. area occupied) is comparatively small, at between 15 m^2 for *C. pedicellatus* and 822 m^2 for *C. ciliaris*. Moreover, these *Cenchrus* distributions are found all on West Island. At the other end of the non-native management scale, the non-native *C. gynandra* has a range of 61,735 m^2 but because it occurs at a relatively low canopy cover (3.1%), the area within that range over which control would need to be deployed is 1,931 m^2 (Table 11).

Table 11. Coverage and abundance of plant communities at Ashmore Reef across the four islands combined. Legend: A_C: Total area for communities containing the species (m^2); A_I: community area as % of all communities; A_O: total area occupied by the species on that island (m^2); D: average canopy cover (%); Q_P: Quadrats with the species present; Q_T: Total quadrats surveyed in communities with the species; C: all communities with that species. *= Non-native species.

	Coverage and abundance (423 community fragments)		
Species	**A_C m^2, A_I%**	**A_O m^2, D%**	**Q_P/Q_T, C**
Amaranthus interruptus	239161m^2, 45.1%	56790m^2, 23.7%	62/101Q, 14C
Boerhavia spp.	202133m^2, 38.1%	21834m^2, 10.8%	17/56Q, 7C
Bulbostylis barbata	1985m^2, 0.4%	68m^2, 3.4%	5/5Q, 1C
Cordia subcordata	37074m^2, 7.0%	37m^2, 0.1%	0/0Q, 1C
Cuscuta victoriana	4140m^2, 0.8%	652m^2, 15.7%	11/11Q, 3C
Digitaria mariannensis	262781m^2, 49.6%	68893m^2, 26.2%	50/89Q, 11C
Eragrostis cumingii	242986m^2, 45.8%	76029m^2, 31.3%	44/120Q, 14C
Guettarda speciosa	37074m^2, 7.0%	981m^2, 2.6%	1/17Q, 1C
Heliotropium foertherianum	37085m^2, 7.0%	5942m^2, 16.0%	5/17Q, 2C
Ipomoea pes-caprae	37074m^2, 7.0%	327m^2, 0.9%	1/17Q, 1C
Ipomoea violacea	258915m^2, 48.8%	25111m^2, 9.7%	20/61Q, 8C
Lepturus repens	198463m^2, 37.4%	31529m^2, 15.9%	76/100Q, 15C
Pandanus sp.	37074m^2, 7.0%	37m^2, 0.1%	0/0Q, 1C
Portulaca tuberosa	229435m^2, 43.3%	10261m^2, 4.5%	10/40Q, 7C
Sesbania cannabina	363447m^2, 68.6%	23609m^2, 6.5%	32/83Q, 14C
Sesuvium portulacastrum	37074m^2, 7.0%	7044m^2, 19.0%	6/17Q, 1C
Sida pusilla	298367m^2, 56.3%	51065m^2, 17.1%	59/116Q, 14C
Spinifex littoreus	115m^2, 0.02%	106m^2, 92.5%	2/2Q, 1C
Spinifex longifolius	8722m^2, 1.6%	8544m^2, 98.0%	4/4Q, 2C
Sporobolus virginicus	1010m^2, 0.2%	595m^2, 58.9%	6/6Q, 2C
Suriana maritima	37074m^2, 7.0%	37m^2, 0.1%	0/0Q, 1C
*Cenchrus brownii**	529m^2, 0.1%	255m^2, 48.2%	11/11Q, 1C
*Cenchrus ciliaris**	891m^2, 0.2%	822m^2, 92.3%	3/3Q, 1C

Coverage and abundance (423 community fragments)			
Species	**A_C m^2, A_I%**	**A_O m^2, D%**	**Q_P/Q_T, C**
*Cenchrus echinatus**	165m^2, 0.03%	115m^2, 70.0%	3/3Q, 1C
*Cenchrus pedicellatus**	16m^2, 0.003%	15m^2, 95.0%	1/1Q, 1C
*Cleome gynandra**	61735m^2, 11.6%	1931m^2, 3.1%	9/32Q, 4C
*Portulaca oleracea**	295835m^2, 55.8%	19579m^2, 6.6%	40/102Q, 13C
*Tribulus cistoides**	119043m^2, 22.5%	13754m^2, 11.6%	16/16Q, 7C
*Xenostegia tridentata**	152m^2, 0.03%	25m^2, 16.7%	1/3Q, 1C
All species	530078m^2, 100%	367160m^2, 69.3%	255/255Q, 35C

Shrubs and trees

The most common shrub species at Ashmore Reef is *S. cannabina*, with an estimated 445,800 plants across West, Middle and East islands (Table 12). Of the large shrubs and trees, *H. foertherianum* is the most common with 619 live individuals. All but one of these plants are found on West Island, a single sick individual is the last remaining shrub on Middle Island, while East Island has no live shrubs or trees (Table 13). Of these live individuals, 72% are large plants (canopy diameter > 2 m), and of these large plants, almost 50% are in poor health, with a significant proportion of the canopy containing dead and dying branches. Together, dead and sick individuals comprise 72% of the bushes and trees still standing at Ashmore Reef. All *C. nucifera* individuals are dead, there is only a single small *C. subcordata* left, and only there are only 2 and 18 large *G. speciosa* and *S. maritima* remaining, respectively (all on West Island). There was evidence of recent recruitment for *H. foertherianum*, *G. speciosa* and *S. maritima*, but with only a single surviving individual for the latter (Table 13).

Almost all large shrubs are found lining the periphery of the islands, with only a few scattered individuals now largely dead closer toward the interior of West, Middle and East islands (Figure 21 to Figure 23). Based on a visual inspection of the data, the only spatial patterning for sickness or mortality for *H. foertherianum* is that those mature trees further inland are mostly dead or dying, with considerable canopy loss and fragmentation. Recruitment of new plants is more common on the western and eastern ends of the island, yet this is in an area where the sand is commonly and significantly disturbed by nesting turtles. All but two of the *G. speciosa* plants occur at the western end of West Island. One of the biggest shrubs on the island, both in terms of height (c. 3 m) and canopy width (c. 16m), is one of the outlier *G. speciosa* individuals on the northern point of West Island (Figure 24a). *Suriana maritima* was represented by just three individuals close to the eastern tip of West Island – two larger shrubs and a smaller one (c. 0.5 m tall).

Table 12. Density and abundance metrics for *Sesbania cannabina* shrubs across vegetation communities on West, Middle and East Islands at Ashmore Reef in May 2019. Vegetation communities are defined in Table 7 to Table 9. Legend: P: Number of plants; A$_O$: total area occupied on that island (m^2); Q$_P$: Quadrats with the species present; Q$_T$: Total quadrats surveyed in communities with the species; C: all communities with that species.

							West Island								
Wa	**Wb**	**Wc**	**Wd**	**We**	**Wf**	**Wg**	**Wh**	**Wi**	**Wj**	**Wk**	**Wl**	**Wm**	**Wn**	**P, A$_O$**	**Q$_P$/Q$_T$, C**
na	18900 1/8Q	na	260300 9/9Q	na	na	na	na	na	na	na	na	na	na	279200 72092m^2	10/17Q 2C
							Middle Island								
Ma	**Mb**	**Mc**	**Md**	**Me**	**Mf**	**Mg**	**Mh**	**Mi**	**Mj**					**P, A$_O$**	**Q$_P$/Q$_T$, C**
2200 1/18Q	na	5900 2/18Q	59000 2/5Q	80500 7/7Q	800 2/4Q	na	na	na	500 2/3Q					149000 110544m^2	16/55Q 6C
							East Island								
Ea	**Eb**	**Ec**	**Ed**	**Ee**	**Ef**	**Eg**	**Eh**	**Ei**	**Ej**					**P, A$_O$**	**Q$_P$/Q$_T$, C**
3700 1/1Q	na	na	6500 2/7Q	3200 1/8Q	na	4200 3/3Q	na	na	na					17600 53048m^2	7/19Q 4C
										Ashmore Reef (total)				445800 235683m^2	33/91Q 12C

Table 13. Shrub and tree counts in May 2019 for Ashmore Reef across West Island, Middle Island and East Island. Plants were first grouped into large (> 2 m canopy diameter) or small (<2 m canopy diameter), and then into healthy (H), sick (S) or dead (D) individuals. No trees or shrubs were found on Splittgerber Cay. *Sesbania cannabina* shrubs were not counted individually (Table 12). *Non-native species

Species	**West Island**						**Middle Island**						**East Island**					
	Large			**Small**			**Large**			**Small**			**Large**			**Small**		
	H	**S**	**D**	**H**	**S**	**D**	**H**	**S**	**D**	**H**	**S**	**D**	**H**	**S**	**D**	**H**	**S**	**D**
Cordia subcordata	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Guettarda speciosa	18	0	0	10	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Heliotropium foertherianum	232	211	384	164	11	64	0	1	7	0	0	0	0	0	1	0	0	0
Suriana maritima	2	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
*Cocos nucifera**	0	0	2	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
TOTAL	**252**	**221**	**386**	**166**	**11**	**64**	**0**	**1**	**9**	**0**	**0**	**0**	**0**	**0**	**1**	**0**	**0**	**0**

Figure 21. Shrubs and trees on West Island at Ashmore Reef in May 2019. The five species, *Heliotropium foertherianum*, *Cocos nucifera* (non-native), *Cordia subcordata*, *Guettarda speciosa* and *Suriana maritima* were stratified according to canopy diameter into small (S; <2 m dia, lighter shading) and large (L; >2 m dia, darker shading), and then into healthy (H, green), sick (S, orange), and dead (D, red) individuals. Legend: (*Canopy diameter-Health*).

Figure 22. Shrubs and trees on Middle Island at Ashmore Reef in May 2019. The two species, *Heliotropium foertherianum* and *Cocos nucifera* (non-native) were stratified according to canopy diameter into small (S; <2 m dia, lighter shading) and large (L; >2 m dia, darker shading). All individuals were dead (D, red). Legend: (Canopy diameter-Health)

Figure 23. Shrubs and trees on East Island at Ashmore Reef in May 2019. The single dead (D) shrub*, Heliotropium foertherianum*, likely had a canopy diameter >2 m dia. Legend: (Canopy diameter-Health)

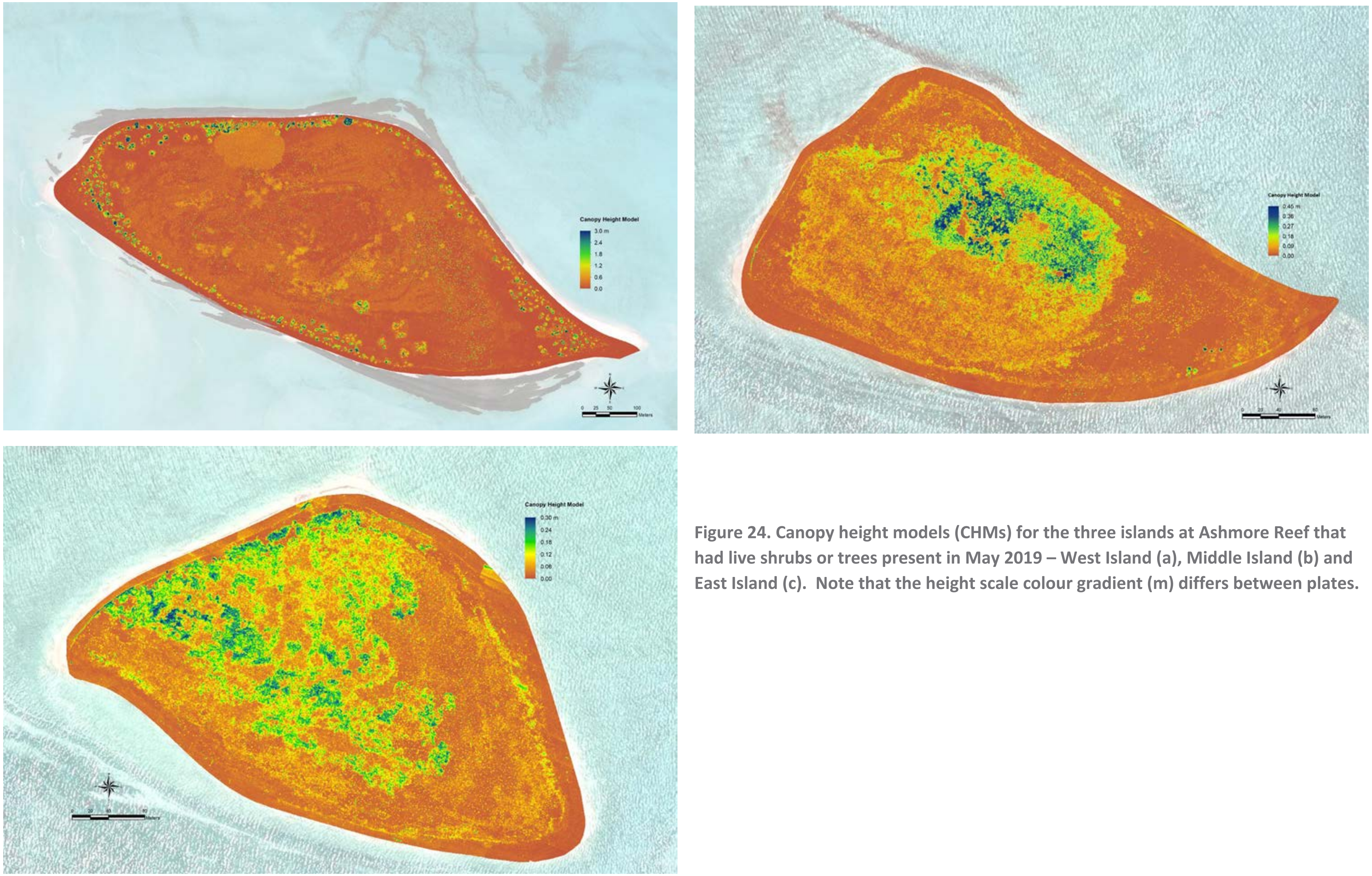


Figure 24. Canopy height models (CHMs) for the three islands at Ashmore Reef that had live shrubs or trees present in May 2019 – West Island (a), Middle Island (b) and East Island (c). Note that the height scale colour gradient (m) differs between plates.

Species accumulation curves

Comparing data from the community survey work and the quadrats, species-area curves revealed that across the four islands combined, the 255 quadrats successfully detected 90% of plant taxa, and 100% of the non-native taxa on the islands (Figure 25a). This pattern was consistent across all four islands (Figure 25b-e), with only one non-native species not detected in the quadrat surveys on West Island (Figure 25b).

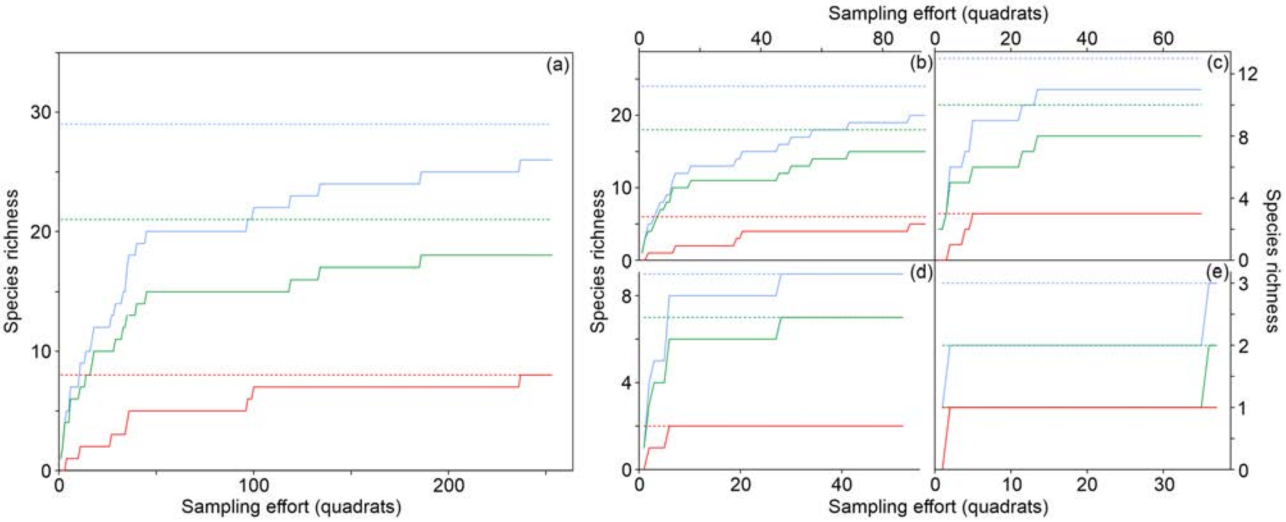


Figure 25. Species accumulation curves comparing species detected during the community survey (dotted lines) with species detected during the quadrat-based abundance survey (solid lines) for all four islands at Ashmore Reef combined (a), as well as broken down for West Island (b), Middle Island (c), East Island (d) and Splittgerber Cay (e). Curves for all species (blue), native species (green) and non-native species (red) are depicted.

3.5.6 Vegetation change over time

Mapping analysis

After digitising the hand-drawn maps of Pike and Leach (1997) and comparing them to the available data from this survey, it was immediately apparent that aligning the two map sources was not going to be possible in a way that would provide robust quantitative insight into change over time. Strong variation in island shape could either be explained by true shifts in the island shoreline from erosion and accretion of sand, or simply from overly stylised hand sketches, with no reliable way to differentiate between either plausible explanation. While fine scale variation in vegetation community locations was not possible, broad similarities were consistent between both time series:

- The fringing *H. foertherianum* bushes and the two locations of *G. speciosa* bushes on West Island;
- The central *C. cannabina* thickets, and the large central fragments of low vegetation dominated by *S. pusilla* and *Boerhavia* spp. on West Island;

  There were, however, also some strong contrasts between the time series:

- While the patch of *S. longifolius* on West Island is in the same location, it is clearly now much larger in size;
- The distribution of the non-native *C. gynandra* has expanded significantly on Middle Island;

- The range of *D. mariannensis* on Middle Island has contracted significantly to a small remnant on the eastern end;
- The location of *C. ciliaris* and *B. barbata* on West Island has changed considerably, with population locations observed in 1997 now not supporting these species, and large new populations found in other locations. In particular, *B. barbata* was only previously described from "a small patch near the Territories Camp" (Pike and Leach, 1997), in a mapped location that now aligns most closely with the largest patch of *C. ciliaris*. In the current survey, however, two larger patches of *B. barbata* were found, both a long way from old infrastructure and one at the western end of the island.

Survey results

By comparing the current survey results to five historical island wide surveys undertaken between 1977 and 2015 and to herbarium specimens collected from the islands during ad hoc surveys over the same period, we were able to reveal species change at the island level for native (Table 14) and non-native (Table 15) species. Of the 41 plant taxa (excluding *Boerhavia* spp.) recorded to have been present on the islands from past surveys, 13 were recorded as absent during the 2019 survey. Of these plants, 11 have previously been recorded as not present on the islands in 2015. Since the most recent 2015 survey, it appears that the non-native *C. nucifera* and *E. hirta* have become extinct at Ashmore Reef (Table 15). The last remaining live *C. nucifera* on West Island has died. Furthermore, the trunks of the two dead *C. nucifera* on Middle Island have now fallen, as has the other *C. nucifera* on West Island. *Euphorbia hirta* was not located in the location from where it was previously documented in 2015 (around the well site on West Island), despite targeted searching. The native *W. biflora* (syn. *M. biflora*), last recorded via a herbarium sample in 2010 from West Island, and the non-native *E. amabilis*, last recorded during the 1997 survey (Pike and Leach, 1997), were carefully searched for but not observed.

There was minimal inter-island variation in native species presence relative to earlier surveys (Table 14). The absence of *Boerhavia* spp. from Middle and East Islands, reported since 2010 (despite abundant populations in earlier surveys) continues. *Digitaria mariannensis* and *I. violacea* were not recorded on East Island in this survey, despite being present in reasonable numbers in 2015 and other earlier surveys. Amongst non-native species, the *Cenchrus* grasses have now not been observed on Middle and East islands since 1997 (Table 15). As part of the 2015 survey, the single *C. pedicellatus* plant on West Island was removed (Westaway, 2015), however the plant was rediscovered as part of this survey in a different location (closer to where it was observed in 2002; Figure 14). It appears that *E. hirta* and *T. cistoides* are now absent from West Island, despite both being present in 2015. While *T. cistoides* is still widespread on both Middle and East islands, *E. hirta* is now absent from the Ashmore islands, at least as live plants, based on this result.

Photo points

All photo points installed by Clarke (2010) were re-surveyed and new comparison images taken (Figure 26 to Figure 29). There was noticeable variation in grass and herb cover between years, however this could easily be due to seasonal differences alone and should not be interpreted as a meaningful change. In contrast, gross differences in shrub and tree health were evident between 2010 and 2019 and these changes reflect more significant and enduring vegetation change. Both *C. nucifera* on West Island had died during the observation period (Figure 26a-d), while all *H. foertherianum* individuals captured on West Island showed significant decline in health, with fragmented canopies and extensive branch loss (Figure 26).

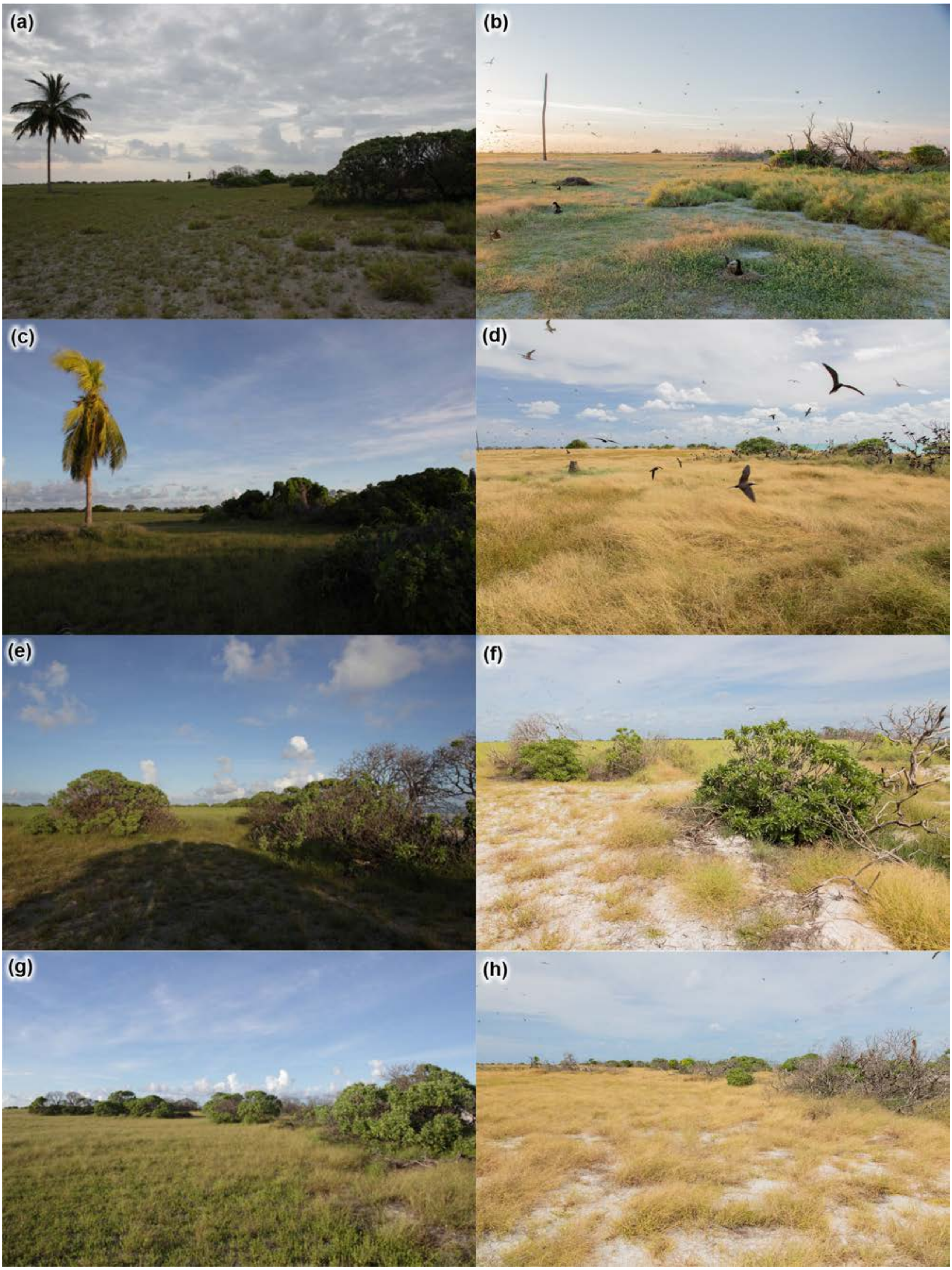
Table 14. Native plant presence over time at Ashmore reef based on field surveys of terrestrial plant abundance and structure in 1977 (Kenneally 1993), 1996-1997 (Pike & Leach 1997), 2002 (Cowie 2004), 2010 (Clarke 2010), 2015 (Westaway 2015) and 2019 (this survey). Splittgerber Cay, a relatively recently formed cay, was first vegetated in 2009. Due to ongoing taxonomic uncertainty, the multiple *Boerhavia* species observed over time have been treated as a single entity. Population structure: A: adult, J: juvenile, U: unknown age, P: mixed age population, D: all dead, NR: not recorded, (NL): no details provided as to which island it was found on. Population abundance (if available): 0: no plants found, 1: single individual, 2: a few individuals or scattered/localised patches (occasionally with number of plants provided: *n=x*), 3: common &/or widespread.

Species	West Island						Middle Island						East Island						Splittgerber Cay		
	1977	1997	2002	2010	2015	2019	1977	1997	2002	2010	2015	2019	1977	1997	2002	2010	2015	2019	2010	2015	2019
Amaranthus interruptus	NR	P2	0	NR	NR	0	P3	P3	P3	P3	P3	P3	P3	P3	P3	P3	P3	P3	0	NR	0
Amaranthus undulatus	NR	0	0	0	0	0	NR	0	0	0	0	0	NR	0	0	0	0	0	0	0	0
Boerhavia spp.	P3	P3	P3	NR	P3	P3	P3	P3	0	NR	NR	0	P3	P3	P3	NR	NR	0	NR	NR	0
Bulbostylis barbata	NR	P2	P2	NR	P3	P3	NR	0	NR	NR	NR	0	NR	0	NR	NR	NR	0	0	NR	0
Caesalpinia bonduc	NR	U2	NR	NR	NR	0	NR	0	NR	0	NR	0	NR	0	NR	0	NR	0	0	NR	0
Cassytha filiformis	P2 (NL)	0	NR	NR	NR	0	P2 (NL)	P2	NR	NR	NR	0	P2 (NL)	P2	NR	NR	NR	0	0	NR	0
Cordia subcordata	P3	A2 (*n=4*)	A2 (*n=4*)	NR	A2	A1	P2	0	NR	0	D1	0	P2	A1	D1	D1	0	0	0	0	0
Cuscuta victoriana	NR	0	NR	NR	NR	0	NR	0	NR	NR	P3	P2	NR	0	P2	NR	NR	P3	0	NR	0
Digitaria mariannensis	P3	P3	P3	NR	P3	P3	P2	P3	P3	NR	P3	P3	P3	P3	P3	NR	P3	0	A1	NR	0
Eragrostis cumingii	P2	P3	NR	NR	NR	P2	NR	P3	P3	NR	P3	P3	NR	P3	P3	P3	P3	P3	P3	NR	A1
Guettarda speciosa	NR	A2	A2	P3	A2	A2	P3	D1	NR	0	NR	0	NR	0	NR	0	NR	0	0	NR	0
Heliotropium foertherianum	P3	P3	P3	P3	P3	P3	P3	P2	P3	A2(*n=2*)	A2 (*n=3*); D2 (*n=3*)	A1; D2 (*n=7*)	P2	D1	A2 (*n=2*)	D2 (*n=2*)	0	0	NR	0	0
Ipomoea pes-caprae	P3	P3	P3	NR	P3	P3	NR	NR	NR	NR	NR	0	NR	D2	NR	NR	NR	0	0	NR	0
Ipomoea violacea	P3	P3	P3	NR	P3	P3	NR	P2	P2	NR	NR	0	NR	P2	P2	NR	P2	0	0	NR	0
Lepturus repens	P3	P3	P3	NR	P3	P3	P2	P3	P3	NR	P3	P3	P3	P3	P3	NR	P3	P3	P3	P3	P3
Pandanus sp.	NR	NR	NR	NR	NR	J1	NR	NR	NR	NR	NR	0	NR	NR	NR	NR	NR	0	NR	NR	0
Portulaca tuberosa	P3 (NL)	P2	P2	NR	P2	P3	P3 (NL)	P2	P2	NR	P2	P2	P3 (NL)	P2	NR	NR	NR	0	0	P2	0
Rhizophora stylosa	NR	J2	NR	NR	NR	0	NR	0	NR	NR	NR	0	NR	0	NR	NR	NR	0	0	NR	0
Scaevola taccada	NR	A2	A1	NR	0	0	NR	A2	A2 (*n=3*)	D2	0	0	A1	0	NR	0	0	0	0	NR	0
Sesbania cannabina	NR	P3	P3	NR	P3	P3	P3	P3	P3	P3	P3	P3	P3	P3	P3	P3	P3	P3	0	NR	0
Sesuvium portulacastrum	NR	NR	NR	NR	P3	P3	NR	NR	NR	NR	0	0	NR	NR	NR	NR	0	0	NR	0	0

Species	West Island						Middle Island						East Island						Splittgerber Cay		
	1977	**1997**	**2002**	**2010**	**2015**	**2019**	**1977**	**1997**	**2002**	**2010**	**2015**	**2019**	**1977**	**1997**	**2002**	**2010**	**2015**	**2019**	**2010**	**2015**	**2019**
Sida pusilla	P3	P3	P3	NR	P3	P3	P3	P3	P2	NR	P3	P3	NR	P3	P2	NR	P3	P3	0	NR	0
Spinifex littoreus	P2	P3	P2	NR	P2	P2	P3	0	NR	NR	NR	0	NR	P3	D2	NR	NR	0	0	NR	0
Spinifex longifolius	P2	P3	P3	P2	P3	P3	NR	NR	NR	NR	P2	P2	NR	NR	NR	NR	NR	0	0	NR	0
Sporobolus virginicus	NR	0	NR	NR	NR	0	P3	P2	0	NR	NR	0	NR	P3	P3	NR	P2	P3	0	NR	0
Suriana maritima	NR	P2	NR	NR	A1+J1	J1+A2 (*n=2*)	P3	P2	P2	NR	0	0	NR	P2	A1	NR	NR	0	0	NR	0
Wollastonia biflora	NR	A+D2	NR	NR	0	0	NR	0	NR	NR	NR	0	NR	0	NR	NR	NR	0	0	NR	0

Table 15. Non-native plant presence over time at Ashmore reef based on field surveys of terrestrial plant abundance and structure in 1977 (Kenneally 1993), 1996-1997 (Pike & Leach 1997), 2002 (Cowie 2004), 2010 (Clarke 2010), 2015 (Westaway 2015), and 2019 (this survey). Splittgerber Cay, a relatively recently formed cay, was first vegetated in 2009. Population structure: A: adult, J: juvenile, U: unknown age, P: mixed age population, D: all dead, NR: not recorded, (NL): no details provided as to which island it was found on. Population abundance (if available): 0: no plants found, 1: single individual, 2: a few individuals or scattered/localised patches (occasionally with number of plants provided: *n=x*), 3: common &/or widespread.

Species	West Island						Middle Island						East Island						Splittgerber Cay		
	1977	1997	2002	2010	2015	2019	1977	1997	2002	2010	2015	2019	1977	1997	2002	2010	2015	2019	2010	2015	2019
Amaranthus crispus	NR	0	0	0	0	0	NR	0	0	0	0	0	NR	0	0	0	0	0	0	0	0
Amorphophallus paeoniifolius	NR	A1	NR	NR	0	0	NR	0	NR	NR	NR	0	NR	0	NR	NR	NR	0	0	NR	0
Cenchrus brownii	NR	P3	P3	NR	P3	P3	P2	P2	NR	NR	NR	0	P3	P2	NR	NR	NR	0	0	NR	0
Cenchrus ciliaris	NR	P2	P3	NR	P3	P2	P2	NR	NR	NR	NR	0	P2	D2	NR	NR	NR	0	0	NR	0
Cenchrus echinatus	NR	P3	NR	NR	P2	P2	P2	NR	NR	NR	NR	0	NR	NR	NR	NR	NR	0	0	NR	0
Cenchrus pedicellatus	NR	P2	P2	NR	A1	P2	NR	0	NR	NR	NR	0	NR	0	NR	NR	NR	0	0	NR	0
Cleome gynandra	NR	0	NR	NR	NR	0	P2	P3	P3	NR	P3	P3	NR	0	NR	NR	NR	0	0	NR	0
Cocos nucifera	A2 (NL)	A2 (*n=2*)	A2 (*n=2*)	A2 (*n=2*)	A2	D2	A2 (NL)	A2 (*n=3*)	A2 (*n=3*)	D2 (*n=2*); A1	0	D2 (n=2)	A2 (NL)	0	NR	0	0	0	0	0	0
Eragrostis amabilis	NR	P3	D2	NR	0	0	NR	0	NR	NR	0	0	NR	0	NR	NR	0	0	0	0	0
Euphorbia hirta	NR	P3	P3	NR	P2	0	NR	0	NR	NR	NR	0	NR	0	NR	NR	NR	0	0	NR	0
Portulaca oleracea	P3 (NL)	P3	NR	NR	P3	P2	P3 (NL)	P3	P2	NR	P3	P3	P3 (NL)	P3	NR	NR	P3	P3	0	P2	P3
Portulaca pilosa	P2$^{\#}$ (NL)	NR	NR	NR	NR	0	P2$^{\#}$ (NL)	NR	NR	NR	NR	0	P2$^{\#}$ (NL)	NR	NR	NR	NR	0	0	NR	0
Tribulus cistoides	NR	P2	P2	NR	P2	0	P2	P2	P2	NR	P3	P3	P3	P2	P3	NR	P3	P3	0	NR	0
Xenostegia tridentata	NR	NR	NR	NR	P2	P2	NR	NR	NR	NR	0	0	NR	NR	NR	NR	0	0	NR	0	0
Zea mays	A2 (NL)	0	NR	NR	0	0	A2 (NL)	D3	NR	NR	NR	0	A2 (NL)	0	NR	NR	NR	0	0	NR	0


(a)
(b)
(c)
(d)
(e)
(f)
(g)
(h)

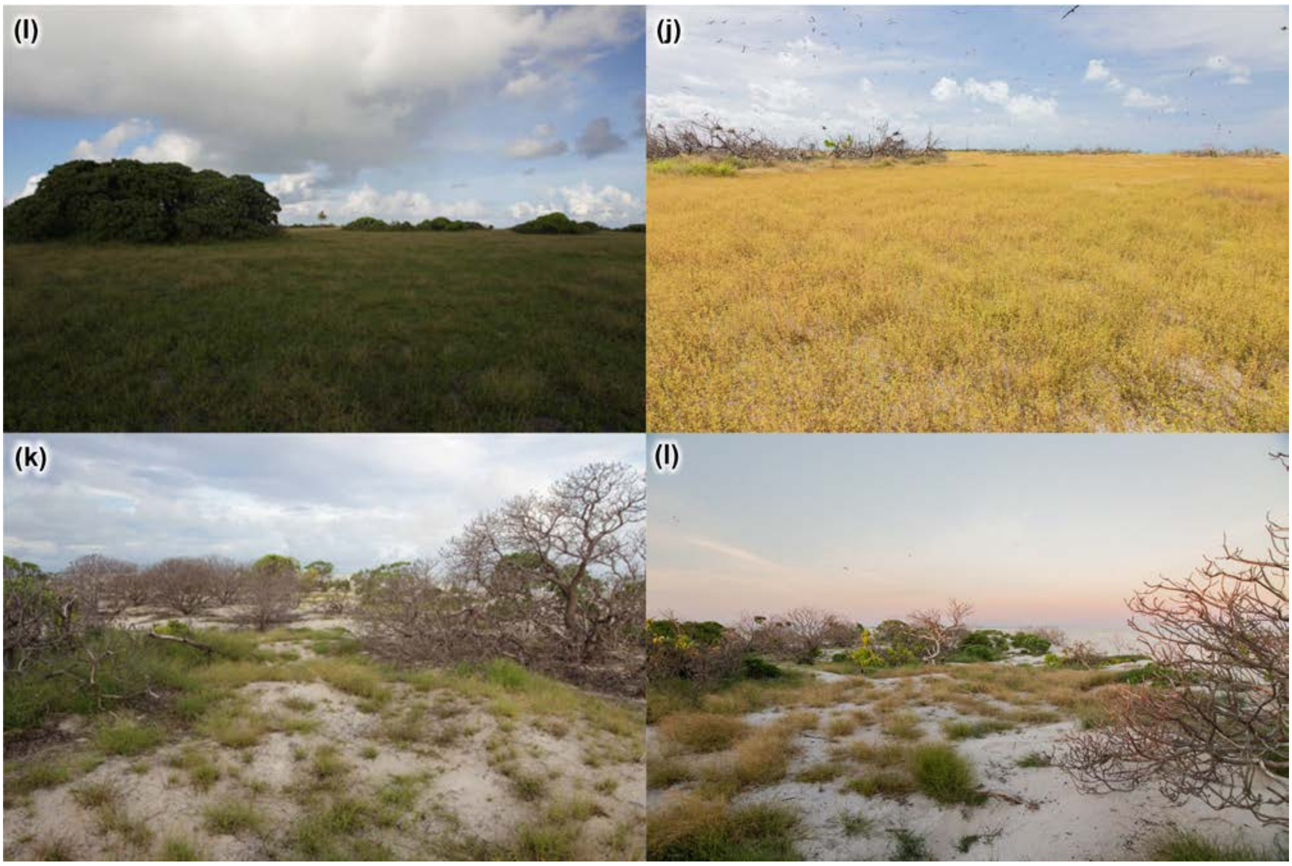


Figure 26. Permanent photo points at West Island, Ashmore Reef, showing change over time between April 2010 (left column) and May 2019 (right column). 2010 photos were from Clarke (2010) and the 2019 resurvey used provided location data where possible (SI Table 1). Photo points 1 (a, b), 2 (c, d), 3, (e, f), 4 (g, h), 5 (I, j) and 6 (k, l) as installed by Clarke (2010) cover a range of vegetation communities across the island (SI Table 1).

Figure 27. Permanent photo points at Middle Island, Ashmore Reef, showing change over time between April 2010 (left column) and May 2019 (right column). 2010 photos were from Clarke (2010) and the 2019 resurvey used provided location data where possible (SI Table 1). Photo points 1 (a, b), 2 (c, d), 3, (e, f) and 4 (g, h) as installed by Clarke (2010) cover a range of vegetation communities across the island (SI Table 1).

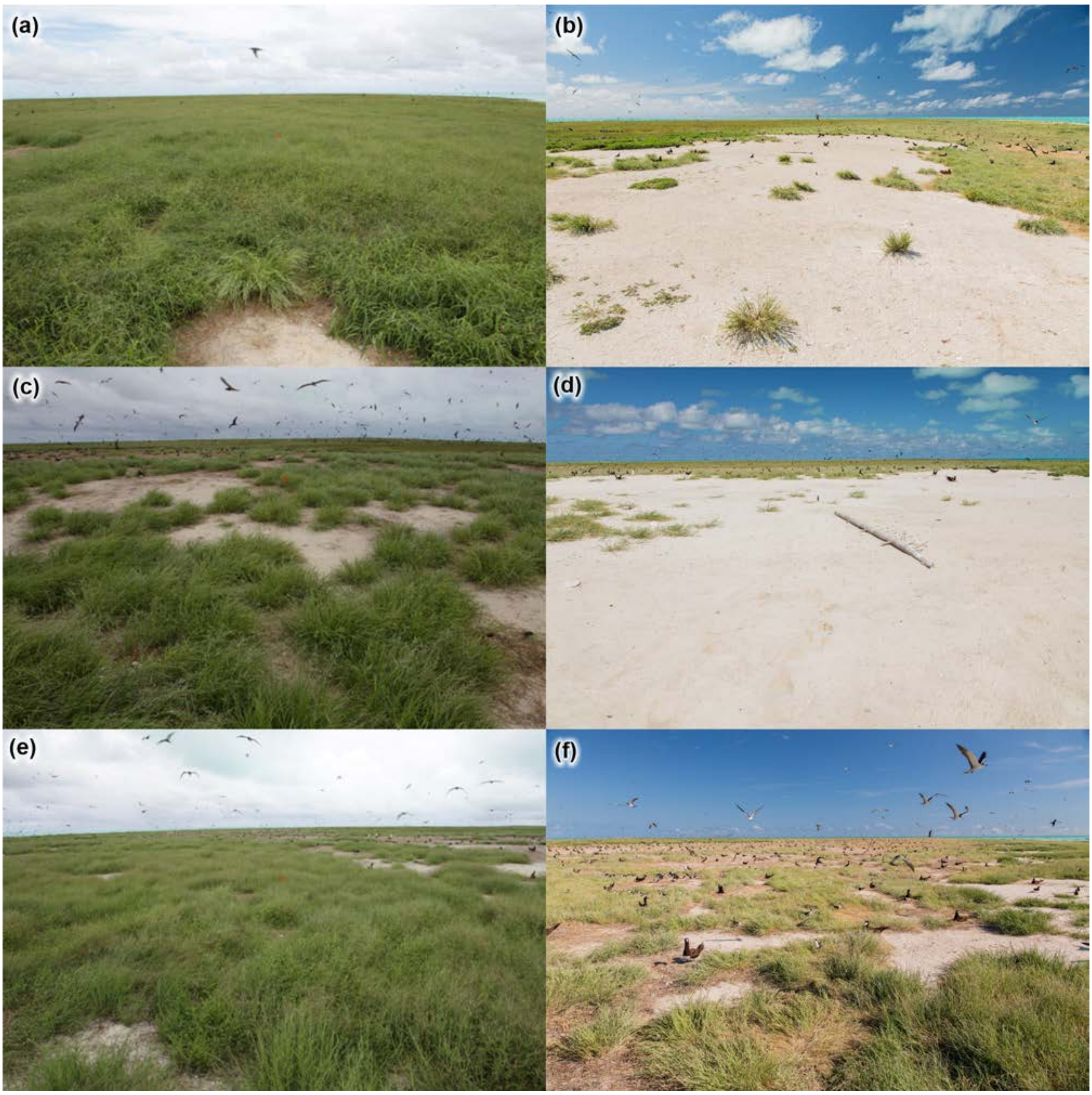


Figure 28. Permanent photo points at East Island, Ashmore Reef, showing change over time between April 2010 (left column) and May 2019 (right column). 2010 photos were from Clarke (2010) and the 2019 resurvey used provided location data where possible (SI Table 1). Photo points 1 (a, b), 2 (c, d) and 3, (e, f) as installed by Clarke (2010) cover a range of vegetation communities across the island (SI Table 1).

Figure 29. Permanent photo points at Splittgerber Cay, Ashmore Reef, showing change over time between April 2010 (left column) and May 2019 (right column). 2010 photos were from Clarke (2010) and the 2019 resurvey used provided location data where possible (SI Table 1). Photo points 1 (a, b) and 2 (c, d) as installed by Clarke (2010) cover a range of vegetation communities across the island (SI Table 1).

3.6 Discussion

This survey builds on a series of vegetation surveys carried out on the Ashmore islands over nearly four decades between 1977 and 2015 (Kenneally 1993; Pike & Leach 1997; Cowie 2004; Clarke 2010; Westaway 2015) to provide the most detailed account of the terrestrial vegetation at Ashmore Reef to date. In doing so, the work has been able to reveal greater detail on the abundance and assemblages of plants across the islands. By comparing this 2019 survey to past surveys, a clearer picture is starting to emerge as to how vegetation metrics have changed over time. The survey has also revealed clear priorities for future management at Ashmore Reef if we are to improve conservation outcomes for this globally important ecosystem.

3.6.1 Island topography

The four islands were revealed to occupy a total area of 56.3 ha, comprising West Island (29.0 ha), Middle Island (12.7 ha), East Island (13.7 ha) and Splittgerber Cay (0.9 ha). Early measurements of overall island area at Ashmore Reef were as high as 93 ha (Langdon 1966). In contrast, the areas calculated during this 2019 survey are comparable to the areas estimated in 2010 (Clarke 2010), and slightly smaller than the areas noted in 1993 (Berry 1993). This similarity is a reassuring outcome for past work given the lower quality of previous survey methods used. Despite the similar overall area, aerial imagery suggests that notable accretion and erosion has changed island shapes over time. For example, a concrete helipad on the northern shoreline of East Island, which was well within the vegetation zone in 1997 (Pike and Leach, 1997), is now crumbling into the sea well below the high tide mark. Shifting shorelines are likely to be more frequent in response to increased extreme weather events, and may well influence the success of vegetation restoration programs, either directly via foredune erosion or indirectly via altering habitat suitability for turtle nesting (Tanaka *et al.* 2007; Feagin *et al.* 2015; Naylor *et al.* 2017). Understanding the stability of Splittgerber Cay, a permanently vegetated island since 2010, and its ongoing value as seabird and/or shorebird habitat will also be important to track over time (Clarke & Herrod 2016). If we are to document this shift in island shoreline over time, taking a georeferenced imagery-based approach to resurvey is the most robust (and arguably most efficient) way to obtain future data. Improved accuracy for both the terrestrial and RPS surveys would be achieved by including the three standard survey marks on West Island (gola.es.landgate.wa.gov.au) into future survey and georeferencing pipelines.

3.6.2 Species identification

With multiple vegetation surveys now available spanning 42 years at Ashmore Reef, it is not surprising that taxonomic nomenclatural changes have impacted on the documented vegetation species lists over time. Of the 57 determinations applied to plant taxa, 39 remained valid with a further six deemed ‘uncertain’ in regard to the relevancy of their application to Ashmore Reef. The primary challenge with taxonomic delimitation and species level clarity at Ashmore Reef is the tar vines (*Boerhavia* spp.). Over time, seven different *Boerhavia* species have been described from the islands. While this situation may be valid, many of these descriptions have been associated with herbarium specimens that have multiple re-determinations and which sometimes conflict between herbaria. Meikle and Hewson (1984) state that *Boerhavia* is a notoriously difficult genus in Australia and that several species have not been determined that may represent undescribed species. Du Puy and Telford (1993) take this further and conclude “a difficult genus requiring a monographic revision,” and clarify that *Boerhavia diffusa* and *Boerhavia repens* should be considered a species complex. Interestingly, the latest revision of *Boerhavia* for the Flora of Australia (Meikle & Hewson 2020) does not even include *B. repens*, which we consider to be one of the more common species currently at Ashmore Reef.

Part of the challenge with accurately capturing the diversity of *Boerhavia* at Ashmore Reef is that the vegetative characters are incredibly variable, with strong morphological change in response to plant age and/or light environments (Chen & Wu 2007). Such variation occurs even within individuals at a single branch level (Webber and Yeoh, unpublished results). Some of the more reliable discriminatory characters require reproductive material and a hand lens or dissecting scope to observe, which makes identification in the field challenging. Furthermore, there are ecological challenges and risks to consider regarding confidently capturing *Boerhavia* diversity at Ashmore Reef. First, the sticky fruits of *Boerhavia* make them a well-documented ‘passenger’ on seabirds, providing an obvious natural dispersal pathway to Ashmore Reef from nearby land masses (Carlquist 1967). Second, it appears that *Boerhavia* populations have entirely disappeared from Middle and East islands since 2002, after once being common there, indicating that the taxa could have varying presence/absence at Ashmore over time. Without good quality herbarium specimens, this temporal variation can make confirming true presence challenging. Last, *Boerhavia* are a biosecurity risk at Ashmore, with *Boerhavia. erecta* viewed as a weed species of concern (Westaway 2015). Noting that *B. erecta* is somewhat easier to differentiate morphologically from other co-occurring taxa, there would be merit in clarifying *Boerhavia* diversity at Ashmore Reef with a robust revision of material collected in the past, along with a systematic collection of new material, and working this in with any new revisions of the genus underway.

An important component of prioritising any ecosystem restoration program is to know which species are non-native, as this information provides a ‘red flag’ to further investigations on ecosystem threats (Webber & Scott 2012; Scott *et al.* 2014). Given that many plant taxa at Ashmore Reef are species with broad global distributions, and that for some species, large parts of their current range have been enabled by anthropogenic dispersal, revisiting assumptions about native status were important. By comparing our work to previous survey reports for Ashmore Reef (Kenneally 1993; Pike & Leach 1997; Cowie 2004; Clarke 2010; Westaway 2015), we found that there was a general agreement in which species are non-native. This includes the four documented *Cenchrus* species, *Cocos nucifera*, *Cleome gynandra* (but note Kenneally, 1993), *Euphoribia hirta* and *Eragrostis amabilis*. For a select group of taxa, however, viewpoints and the rationale for status determinations differed. While this might relate to the way non-native status is ascribed (e.g. Richardson *et al.* 2000; Webber & Scott 2012), previous reports have not been clear in how these determinations were made, despite the connection to informing management choices. In revising seven species here, we reached contrasting conclusions to some previous determinations for four taxa – *Bulbostylis barbata*, *Portulaca oleracea, Tribulus cistoides*, and *Xenostegia tridentata* – noting that there was existing disagreement on status across prior Ashmore Reef vegetation surveys for all but *B. barbata*. These contrasting decisions have been made by considering all available evidence, yet are not likely to change the management recommendation as to whether to control them or not, a decision that should be driven more by negative impacts than native status alone.

3.6.3 Plant diversity

For the most part this survey confirmed what all previous vegetation surveys have found, in that the plant communities are relatively normal in their assemblages for isolated coral reef islands of this region (Kenneally 1993; Pike & Leach 1997). Plants native to the Ashmore islands are dominated by grasses and other herbaceous plants and are widespread across the region due to their effective dispersal ability (Kenneally 1993). The diversity of plants observed in 2019 included 28 taxa, with up to five additional *Boerhavia* taxa that were not identified to species level. This total includes eight species regarded as non-native, and one new native species, an unidentified *Pandanus*, recorded growing on the Ashmore islands for the first time (syncarps have been recorded as flotsam previously; Pike and Leach, 1997). West Island remains the most botanically diverse island, as has been found across all previous surveys (Kenneally 1993; Pike & Leach 1997; Cowie 2004; Clarke 2010; Westaway 2015). As noted previously, both *Digitaria mariannensis* and *Spinifex littoreus* at Ashmore Reef are the only known populations in the Australian territories and therefore represent biogeographical anomalies in the Australian flora. None of the biosecurity target weed species were found during this survey, which is a positive outcome for risk of spread to the Australian mainland, as well as for the risk to local ecosystem disruption (reviewed in Westaway 2015).

3.6.4 Community characterisation

The combination of aerial imagery and ground truthing survey methods allowed for the first detailed mapping of the plant communities at Ashmore Reef. Based on species presence and relative abundance, we mapped the distribution of 35 distinct vegetation communities across the four islands. Earlier surveys have provided generalised sketch maps (Pike & Leach 1997) broad descriptions of general dominant vegetation components (Kenneally 1993; Cowie 2004; Clarke 2010) and occasionally the mapped distribution of certain species (Westaway, 2015; for a discussion of change over time see Section 3.6.5). However, no previous community characterisation has been done at the level required to implement targeted non-native species control, nor to understand how communities may change between years.

That no single community was found on more than one island, even though there was strong overlap in species, suggests different drivers of competition and abundance exist between the islands. Such drivers include past anthropogenic disturbance, such as the guano mining on West Island, petroleum exploration camps, Indonesian fishers activity and wartime activities (Serventy 1952b; Langdon 1966; Pike & Leach 1997; Clark 2000; Dwyer 2000), as well as seabird nesting pressures (Cowie 2004), and threats from non-native species (Pike and Leach, 1997; Cowie, 2004, Westaway, 2015). We discount dispersal limitation between islands as an explanation for any of the plant species being absent, given the short distances between the four islands and the dominance of long-distance propagule dispersal syndromes amongst the island’s plant taxa (Van der Pijl 1982).

3.6.5 Species abundance and change over time

By standardising semi-quantitative descriptions from previous Ashmore vegetation surveys and through a multi-pronged approach to calculating plant abundance during this survey, we have been able to provide unprecedented insight into the spatial patterns of abundance for all species within and between the Ashmore islands. The findings confirm that the communities are grass dominated, although different combinations of grasses dominate in different areas within and between islands. Such insight will be critical for underpinning species management programs, both the control of non-native species as well as the restoration of native vegetation. Amongst the native grasses and herbs, there is almost no previous information from Ashmore Reef to compare our quantitative abundance findings against. However, some reports have specific details on certain species, and historical satellite imagery provides *prima facie* insight on community stability. It is clear that the *S. longifolius* patch continues to expand in size, while the two known patches of *Cenchrus ciliaris* appear similar in size and position to the locations mapped by Westaway (2015). By mapping all plant species and knowing their density, this work has been able to calculate the area occupied by each species, which is incredibly important data for understanding the logistics of non-native species control. The four *Cenchrus* species occupy just over 1,200m^2, all confined to West Island. This ranges from just 15m^2 in one patch for *C. pedicellatus* to 822m^2 across three patches for *C. ciliaris* and 255m^2 across 30 patches for *C. brownii*. While there may well be a seedbank that has a somewhat different distribution, these sizes are all within a feasible size for targeted control (Farrell & Gornish 2019).

In contrast to the grasses and herbs, for the shrub and tree layer our abundance data clearly captures a long-term decline that has been taking place at Ashmore Reef for more than 25 years. Pike and Leach (1997) were the first to mention a general decline in the health of *Heliotropium foertherianum* in particular, including the first island absence of the species (East Island in 1993). The authors also drew attention to the relentless herbivory pressure on *Cordia subcordata* from native invertebrates (predominantly *Coccinella transversalis* syn. *Coccinella repanda*, Coccinellidae), to the point where it was drastically limiting recruitment. The five mature plants observed in 1996-1997 are now reduced to a single sapling on West Island, and there is now a high likelihood that without intervention *C. subcordata* may become extirpated on the Ashmore Islands in coming years. Similarly, *Suriana maritima* is also represented by only two mature plants growing adjacent to each other on the eastern tip of West Island with a small seedling nearby. This population is therefore at risk in the near term due to removal by beach erosion associated with extreme weather events, as well as in the long term from inbreeding depression.

Of greater concern, however, is the population health of *H. foertherianum*. Of the 827 mature shrubs on West Island, 72% are either dead or sick, with considerable dieback affecting those shrubs that remain. Photo point images capture this dieback clearly, with a trend towards greater dieback toward the interior of the island, but equally no part of the West Island remains unaffected. Given the significant role that *H. foertherianum* plays in providing ecosystem services, including foredune stability, habitat structural heterogeneity, this decline is concerning. Regarding the latter, these shrubs provide critical habitat for invertebrates (which in turn support insectivorous birds), as well as for both arboreal- and ground-nesting birds, including the red-tailed tropicbird (*Phaethon rubricauda*; Section 4.5.3 of this report and McDonald (2005); Batianoff *et al.* (2010); Clarke (2010). Efforts to address the shrub layer decline have been attempted in the past. Pike and Leach (1997) refer to a shrub ‘replanting program’ that commenced in 1994, and McDonald (2005) undertook a seedling establishment trial for *H. foertherianum* on West Island in 1999 that saw 21 out of 60 seedling survive the first seven months after planting (their fate after that time remains unknown). As noted by McDonald (2005), there is considerable potential for larger populations of shrubs at Ashmore Reef without impacting on seabirds that require open areas for ground-nesting.

The most plausible primary explanation for this widespread dieback is the notable increase in the seabird populations that utilise the Ashmore islands as nesting sites. All previous reports refer to the increasing nesting burden on a decreasing shrub population, leading to mechanical damage from nesting, as well as guano deposits (possibly leading to nutrient toxicity) as reasons for shrub decline and death (Pike & Leach 1997; Cowie 2004; Clarke 2010; Westaway 2015). What is not commonly focused on is the increase in nesting bird numbers over time that coincides with the shrub decline, and what appears to be a far heavier impact on bird abundance before the mid-1980's by visiting Indonesian fishers than previously acknowledged. Large culls of birds "making heavy inroads on the populations of nesting seabirds" and leaving "heaps of slaughtered bird remains," as well as widespread egg harvesting in the 1950's to the 1980's (Serventy 1952a; Serventy 1952b; Milton 2005; Clarke *et al.* 2011; Clarke & Herrod 2016), could well have significantly ameliorated the nesting burden on the shrubs. Even so, this same visitation led to frequent harvesting of the shrubs for firewood (K.F. Kenneally, unpublished data), so other drivers may well have been influencing the balance. Yet additional threats, such as rodents impacting on seedbanks (and therefore shrub recruitment; Kenneally 1993; McDonald 2005) and other non-native species altering shrub regeneration likelihoods, strongly suggest that multiple factors may be combining to drive shrub decline under increasing seabird numbers. Therefore, understanding why shrubs are in decline would need to consider all of these potential factors together, including their direct and indirect interactions.

Elsewhere, *H. foertherianum* decline has also been observed on Coral Sea islands (Batianoff *et al.* 2010), where prolonged drought was considered a contributing factor. Greater mortality and declines in health towards the West Island interior would fit with lower water availability and water stress being a contributing factor. Another possibly overlooked factor is the residual impact of the 2004 tsunami (Drushka *et al.* 2008), that is likely to have sent a wave of considerable size over the Ashmore islands. Analysing pre- and post- tsunami aerial imagery may well provide insight into any significant damage done. Taken together, while increasing seabird numbers in recent years at Ashmore Reef may be celebrated as a conservation success, the impact these numbers are having on the stability of the island's shrubs should be looked at closely, given the importance of a shrub layer to wider island ecosystem values. That said, we note that multiple reports since 1997 have repeatedly called for shrub restoration as a priority management action at Ashmore (Pike & Leach 1997; Cowie 2004; McDonald 2005; Westaway 2015), yet no meaningful action to address this issue has yet been implemented.

3.7 Management implications and recommendations

The findings of this report have clear implications for guiding management priorities at Ashmore Reef. The recommendations that follow have both near- and long-term deliverables, broken down into theme-based components:

3.7.1 Survey logistics

The 2019 survey represents the first detailed assessment of plant community assembly, spatial patterns and abundance for the Ashmore islands, providing a robust baseline for establishing future management success. Any future plans to actively manage the vegetation on the Ashmore Islands will require adequate monitoring to be able to measure the success of interventions. To make the most of this insight, we recommend that future surveys are done reasonably regularly and adopt a similar quantitative approach as that undertaken in this work. The efficiency of RPAS imagery combined with strategic ground truthing not only improved data quality and time effectiveness, but it also arguably decreased any negative impact of the necessary ground surveys on nesting seabirds (Vas *et al.* 2015; Borrelle & Fletcher 2017; Brisson-Curadeau *et al.* 2017; Pace, Sherley & Elliott 2017).

3.7.2 Taxonomy and native status

The taxonomy of *Boerhavia* spp. on the islands needs revision based on existing collections as well as a thorough set of new specimens from Ashmore Reef, ideally incorporating these updates into any generic revisions underway. As Cowie (2004) points out, there is always a chance that additional plant species known from the region, but not yet naturalised at Ashmore Reef, may arrive via natural dispersal means in the future. How these plants are managed needs careful consideration, as well as improved knowledge on their broader historical biogeography. Multiple factors should influence how to prioritise their management. However, the highest priority for consideration should be the potential impacts that these new arrivals may have on the broader biodiversity values of the Ashmore ecosystems, and whether or not the new arrival may threaten the resilience of these values into the future. Where there is evidence that the species may be present in the region but not native, then we recommend a cautious approach and removal of the species. *Xenostegia tridentata* is a case in point. A recent arrival documented in 2015 (Westaway, (Westaway 2015), the population was previously viewed as native at Ashmore, as some reports consider it native to south east Asia (Simões, Silva & Silveira 2011). However, significant uncertainty about the native range exists, with good evidence it is an ancient human introduction to south east Asia (Austin 2014).

3.7.3 Native species restoration

There is an urgent need to address the ongoing multi-decade decline in health and rising mortality of the shrub layer at Ashmore Reef. For *H. foertherianum*, a focus on restoration via established seedlings sourced from seed on the island appears to be the most feasible (McDonald 2005) and time efficient method. For other shrub species, genetic supplementation and more carefully considered establishment methods may be required for a successful outcome. To mitigate the pressure on these establishing shrubs from seabird nesting, exclosures to protect the shrubs from seabirds and artificial nesting platforms for the birds should be considered, at least until there is a sustainable shrub community established with clear evidence of successful recruitment. Such activity would need to be undertaken in combination with other management strategies that mitigate the other threats to shrub recruitment. The expanding *S. longifolius* patch should be monitored in an ongoing way as part of regular vegetation surveys. If this patch starts to expand so as to impact on ground-nesting birds, then steps to limit its size may be required.

3.7.4 Non-native species control

Of the eight non-native species on the islands, there is merit in considering eradication of the four *Cenchrus* species (*C. brownii, C. ciliaris, C. echinatus* and *C. pedicellatus*), *X. tridentata* and *C. gynandra* as a matter of priority. Furthermore, it is worth pursuing a biological control solution for significantly reducing the abundance of *T. cistoides*, although the timing of such a control program should first considering broader interactions with nesting seabirds (see Section 4.5.3 of this report). Much is known about the threat and control options of *Cenchrus* grasses from their introduction and invasion in other regions (Marshall, Lewis & Ostendorf 2012; Young & Schlesinger 2015; Farrell & Gornish 2019). These perennial weeds are likely to outcompete native vegetation if not controlled. For example, if the plant is able to form dense monocultures, the impact on the recruitment of shrubs on the island can have cascading effects on species that utilise shrub structure for nesting and shelter, as well as disrupting ground-nesting birds that require open ground. As such, the impacts on other community species of not pursuing eradication against *Cenchrus*, including birds and invertebrates, will be variable and dependent on species traits. A commitment to eradicating the four *Cenchrus* species from Ashmore appears feasible from an area under management perspective, but will require a regular commitment for around 10 years to ensure the seedbank is fully depleted. Resourcing this activity efficiently could be improved by a better understanding of growth phenology at Ashmore to more accurately time control trips, as well as considering herbicides that offer longer windows of impact. While the earlier populations of *Cenchrus* on other islands have not been observed for over 10 years (suggesting probable seed bank depletion), a cautious approach to resurveying regularly should be taken, given the risk of dispersal and re-invasion from West Island.

The other high priority species for eradication is *X. tridentata*. Given the recent arrival at West Island and a very small area occupied, manual control may well be sufficient to permanently remove the species. As a creeping and vegetation smothering vine that is common to disturbed areas, and with closely related species considered an invasion threat elsewhere (Meyer 2000), the most prudent course of action would be to remove this vine before any possible impacts are detected. Further ecological understanding of impacts, seed longevity and dispersal ability would be useful for refining control programs. *Cleome gynandra* should be carefully considered for eradication, given the relatively small area occupied on Middle Island. Additional information on response to available treatments (e.g. herbicide) and insight on seed longevity would help to define the most appropriate control program. Lastly, there would be real benefit in investigating the feasibility of a biological control program against *T. cistoides* at Ashmore Reef. This is because seed longevity for the species is upwards of 30 years, making eradication an unrealistic goal. The stem-and-crown-mining weevil *Microlarinus lypriformis* (Coleoptera: Curculionidae) has been a highly successful control agent for *T. cistoides* in other countries, is already known from Papua New Guinea (Maddox 1976; Bennett 1989), and may well reduce any threats from *T. cistoides* to insignificant levels.

There are some non-native plant species (and recently arrived native species) at Ashmore Reef for which we recommend no control. Given the relatively recent arrival of *B. barbata* to the island, while we consider the Ashmore populations native, they should be monitored for invasion and any negative impacts on the overall community. If there are threats observed, which we view as a low likelihood, then control options should be reconsidered. The only other non-native species found during the 2019 survey, *Portulaca oleracea*, is well known as one of the world's most widespread plants, achieving this distribution from a likely native range in South America with considerable assistance from human dispersal (Ridley 1930; Chapman, Stewart & Yarnell 1973; Holm *et al.* 1977; Ocampo & Columbus 2012). However as it is currently defined the taxon is paraphyletic (Ocampo & Columbus 2012), consisting of up to 15 possible taxa, with the taxa found in Australia and New Zealand likely differing from those in other parts of the world (Danin, Baker & Baker

1979; Gorske, Rhodes & Hopen 1979). An effective control program is unrealistic without significant investment, given the widespread but low abundance distribution across three of the four islands and the relatively high likelihood of re-invasion from other land-masses (Ridley 1930). Moreover, there is no clear evidence that *P. oleracea* is causing any negative impacts to other flora or fauna at Ashmore Reef. As such, we recommend that it be monitored as part of regular vegetation surveys and that material from Ashmore Reef is included in any future phylogenetic revision of the *P. oleracea* species complex to help clarify its taxonomy and likely origins.

3.7.5 Biosecurity risks

While no high-risk new weeds were found on the Ashmore islands during the 2019 survey, they remain a risk for as long as they remain present in Indonesia and Australia. As long as non-native plants remain within natural dispersal distance from Ashmore Reef (e.g. Roti), and as long as human visitation (even in a controlled manner) from both countries is allowed on the islands, then active biosecurity protocols and surveillance of the islands should be maintained to a high standard. Ongoing surveys should maintain vigilance for searching for high risk plants and animals, and this risk list should be updated on a regular basis. While *Eragrostis amabilis* and *Euphorbia hirta* were not detected during the 2019 survey, we recommend that they are actively searched for near their last known location as part of regular surveys for the weed species identified as a biosecurity risk by Westaway (2015). The introduction (and re-introduction) of non-native animals, such as rodents and invertebrates, represents additional biosecurity risks to consider and manage.

This broader regional biosecurity need emphasises the value of Australian engagement such as NAQS and Border Force with managing and monitoring non-native species in the wider south east Asian region beyond Ashmore Reef. Similarly, the biosecurity risk of moving species between the islands and mainland Australia should be proactively managed in an ongoing way. The absence of *C. nucifera* at Ashmore, as documented in this survey, is the first time the islands have been free of this species since they were planted in the 1970's (before that time rats in plague proportions prevented establishment; (Pike & Leach 1997). Pike and Leach (2007) noted that the palms were used as roosts for vagrant fruit bats, providing a notable steppingstone for connectivity between Australia and Indonesian populations. Given the risk of disease spread via these animals (Breed *et al.* 2010; Roche *et al.* 2015), maintaining an ongoing absence of *C. nucifera* on the Ashmore islands would be prudent.

3.7.6 Addressing knowledge gaps

During the course of this work it became apparent that there is a large volume of unpublished yet highly informative ecological and sociological data for the Ashmore islands. From an initial assessment of potential sources, these data are widely dispersed in a range of disparate locations and formats. Much of the data was collected before digital records were available and before database management of biological data became common. Such data, if carefully aggregated, quality controlled and analysed, will transform our ability to understand past change trajectories for the islands. We view compilation of this data into a single GIS data asset as a high priority for conservation management at Ashmore Reef, harnessing the considerable resources that have been invested in generating the data over multiple decades. When combined with current understanding, this insight will help to ensure that future management plans deliver against their goals and avoid inadvertent or unintended consequences.

3.7.7 Structuring management

If it is taken that the goal of management at Ashmore Reef is to ensure that the internationally recognised biodiversity values should be maximised now and into the future, then it is clear that a whole of ecosystem approach to management is required. Such an approach is best achieved by incorporating the cascading indirect effects known to occur in complex assemblages, rather than continuing to focus solely on managing one or a couple of species at a time. For many of the conservation challenges at Ashmore Reef, the needs for delivery are less about what to do, and more about when and in what order to do it. This is because of the importance of species interaction networks, that is, the multiple interactions between species in a community, including pollination, herbivory, competition and predation. More than the presence or abundance of individual species per se, these interactions are largely responsible for shaping the resilience and stability of ecosystems (Menge 1995; Tylianakis *et al.* 2008; Sotomayor & Lortie 2015). The findings of the 2019 survey, as well as the changes seen over past surveys, indicate that the stability of the terrestrial communities at Ashmore is under threat. As such, the interactions between plant, vertebrate and invertebrate taxa need to be prioritised when devising the timing and sequence of future management plans.

An ecosystem network approach to management would necessarily need to include the direct interactions that we are aware of at Ashmore Reef (Figure 30). However, it would also need to include the indirect interactions between species, which can account for almost 40% of the change to community structure from ecosystem perturbations (Menge 1995). Furthermore, factoring in terrestrial-marine interactions would help to account for resource flow between the two realms, which has a significant influence on terrestrial ecosystem stability and resilience.

Some of the interactions documented during the broader 2019 Ashmore Reef survey include those between terrestrial vegetation, seabirds and shorebirds, *Solenopsis geminata*, *Hemidactylus frenatus*, turtles and hermit crabs (Figure 30; and see Section 4.5.3 of this report). However, rodents are also known from the islands - rats were first recorded in 1949 (and were most likely introduced during the phosphate mining era in the 1800’s) and since that time very variable numbers of rats and mice have been recorded, from scarce up to plague proportions (Kenneally 1993; Pike & Leach 1997; Hale & Butcher 2013). Rodents can also represent a significant barrier to plant recruitment through their consumption of seeds. For native plants this is a concern, but the interaction between mice and introduced plant seeds may be buffering the potential impact of these non-native plants on the local ecosystem. Moreover, many of the native herb layer species on these islands are short lived annuals, whereas some of the more invasive non-native plants, such as the *Cenchrus* grasses and beach caltrop, are perennials, changing food availability not only for introduced seed predators such as mice, but also for native seed feeders such as hermit crabs. Seabird nesting associations were noted during this survey with both native and non-native plant species and warrant further investigation. Once these networks and the magnitude of their interactions are understood, then the choice of what management option to deploy can be made. Without careful consideration of these interactions and inter-dependencies among species (both native and non-native), unintended consequences could derail management goals.

3.7.8 Conclusions

This detailed survey of the terrestrial environments at Ashmore Reef has provided the most detailed insight into the island ecosystems yet. It has revealed that these islands, which are recognised as globally important conservation assets, are under threat from non-native invasive species and are at risk of losing some of the more significant natural qualities for which they are valued. We have outlined management needs that should be implemented alongside ongoing monitoring to ensure that we can mitigate these threatening processes and build resilience among native species at a community level. We note, however, that many of these management recommendations have been called for repeatedly in the past (Pike & Leach 1997; Cowie 2004; Hale & Butcher 2013; Westaway 2015). While *ad hoc* but ultimately ineffective management programs have occasionally been deployed on the Ashmore islands, no sustained actions to address the threats to terrestrial biodiversity at Ashmore Reef have been implemented. If we are to achieve enduring conservation success for these islands, there is a need to prioritise deploying a carefully planned and executed monitoring and management program of sustained duration to adequately mitigate the challenges identified.

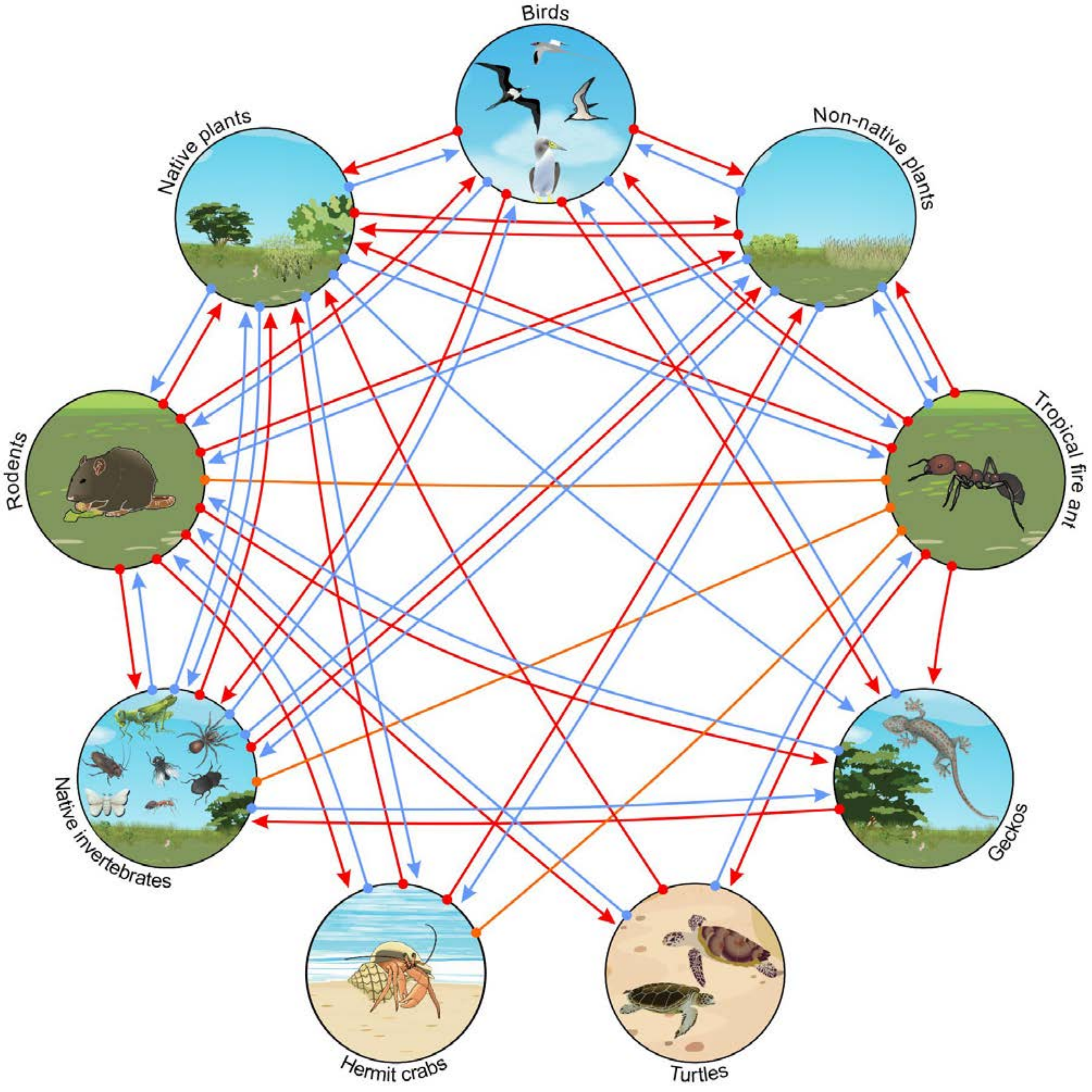


Figure 30. Theoretical species interaction network for the islands of Ashmore Reef. Direct interactions are shown, with positive (blue) and negative (red) outcomes shown from the perspective of the receiver (arrowhead). Orange interactions are likely to exist, but the nature of the interaction is not known. Indirect interactions, as well as marine interactions that influence the terrestrial network are not shown. Icons are adapted from symbols courtesy of the NESP Northern Australia Hub, IAN@UMCES, SciDraw and GraphicsRF.

4 ASHMORE REEF: SEABIRDS AND SHOREBIRDS

Belinda Cannell (University of Western Australia) and Chris Surman (Halfmoon Biosciences)

4.1 Abstract

Internationally significant numbers of seabirds and shorebirds use the islands and cays within the Ashmore Reef Marine Park. In addition, the Ashmore Reef Ramsar site is located within the boundaries of the Marine Park, and it is an ‘Important Bird Area’ for Lesser Frigatebirds (*Fregata ariel*) and Brown Boobies (*Sula leucogaster*). Ground counts, aerial imagery and transects were conducted to determine the population status and diversity of seabirds and shorebirds within the Ashmore Reef Marine Park to inform future monitoring and management. Associations between birds and vegetation were also identified.

Twelve seabird species had begun breeding in May 2019, and a further two species (Bridled Terns, (*O. anaethetus* and Roseate Terns, S*terna dougalli*) were observed roosting, displaying courtship behaviour or attending nest sites. The breeding stage of Sooty Terns, *Onychoprion fuscatus* and Brown Noddies, *Anous stolidus* was more advanced on West Island. Sooty Terns were the most numerous birds within the Ashmore Reef Marine Park, with 77,309 counted across East, Middle and West islands. There has been a shift in the distribution of seabirds across these three islands main islands. For example, Black (*A. minutus*) and Lesser Noddies (*A. tenuirostris*), previously found breeding only on Middle and East Island, were observed breeding only on West Island. A further five species have expanded their breeding territories from only East and Middle Islands from 1990-2014 to now include West Island in 2019. The numbers of adults observed within the Marine Park have apparently increased for the majority of seabird species. Accuracy of counts from aerial RPAS images was higher than that of ground counts. We propose a series of recommendations important for effective management of the seabirds and shorebirds, including aerial images of each island at 3-4 different time periods capturing peak breeding and roosting of the seabirds and shorebirds. E.g. May (egg laying most seabirds), August/September (when many seabird chicks will be visible), November (breeding by Crested and Sooty Terns, egrets and herons: shorebirds also present) to estimate seabird and shorebird populations intra and inter annually. This would enable interrogation of factors that may or may not be correlated with change (e.g. sea surface temperature, cyclone activity, strength of the Indian Ocean Dipole and Leeuwin Current); a necessary precursor for developing management priorities.

4.2 Introduction

Internationally significant numbers of seabirds and shorebirds use the islands and cays within Ashmore Reef Marine Park (Higgins and Davies, 1986, Milton 2005). A total of 15 seabird species and four heron species breed within the marine park (Clarke *et al.* 2011; Clarke & Herrod 2016). It is also an important wintering site for migratory shorebirds, with more than 30 species observed there (Clarke & Herrod 2016). Ashmore Reef Marine Park is assigned as a ‘strict nature reserve’ under the IUCN categories, and consists of two zones: a Sanctuary Zone (550 km^2) and a Recreational Use Zone (34 km^2) (Director of National Parks 2018).

The Ashmore Reef Ramsar site is located within the boundaries of the Marine Park, and the seabirds and shorebirds meet three of the criteria necessary for this designation. The three criteria are:

- Criterion 4: Species and Ecological Communities- *A wetland should be considered internationally important if it supports plant and/or animal species at a critical stage in their life cycles, or provides refuge during adverse conditions*,
- Criterion 5: Waterbirds- *A wetland should be considered internationally important if it regularly supports 20 000 or more waterbirds*, and
- Criterion 6: Waterbirds- *A wetland should be considered internationally important if it regularly supports one per cent of the individuals in a population of one species or subspecies of waterbird* (Hale and Butcher 2013).

Moreover, in 2008 Ashmore Reef was designated as an 'Important Bird Area' (IBA) by Birdlife Australia '(BirdLife 2010). Three species have triggered criterion A4 of the Global IBA criteria, Congregations- *The site is known or thought to hold congregations of ≥1% of the global population of one or more species on a regular or predictable basis*. The species that meet this criterion are Lesser Frigatebirds (*Fregata ariel*), Brown Boobies (*Sula leucogaster*) and Grey tailed Tattlers (*Tringa brevioes*).

Completing surveys of the avifauna has been necessarily pivotal for determining the significance of Ashmore Reef Marine Park. They have been undertaken since 1949, and from 1979–1998, at least two surveys were undertaken on each island in each of the 12 months (Milton 2005). All birds observed were identified and total counts of each species were obtained. However, the reliability of the data prior to 1998 is variable (Milton 2005) and due to the significance of the area to seabirds and shorebirds, Milton (2005) suggested that a monitoring programme be designed to ensure the efficacy of the Nature Reserve (as it was previously called) to maintaining population levels. The suggested annual monitoring programme included two surveys coinciding with peak activity of seabirds and shorebirds, in April-June (seabird breeding) and September –December (shorebird migration) (Milton (2005) through Clarke and Herrod (2016) note peak seabird breeding activity in March-May). However, surveys conducted from 1998–2009 were undertaken in predominantly October/November, and occasionally in January/February. From 2010–2014, following an uncontrolled release of gas, condensate and crude oil from the Montara H1-ST1 Development Well 157 km ESE of Ashmore Reef in 2009, surveys have been undertaken in both April and November (Clarke *et al.* 2011; Clarke & Herrod 2016). No significant declines as a result of the oil spill were detected in any of the seabird or shorebird species in the Marine Park (Clarke & Herrod 2016).

To date, the counts have used standard counting techniques, with experienced seabird counters using spotting scopes, binoculars and the naked eye (Hodgson *et al.* 2016). However, Remotely Piloted Aircraft Systems (RPAs, or drones) have been used in recent years to survey a number of seabird species globally (e.g. Ratcliffe *et al.* 2015; Borrelle & Fletcher 2017; Brisson-Curadeau *et al.* 2017), including Crested Terns (*Sterna bergii*) and Lesser Frigatebirds at Ashmore Reef (Hodgson & Koh 2016). Indeed, accuracy of counts from RPAS footage was found to be as good, if not better, to standard counting techniques (Hodgson & Koh 2016; Brisson-Curadeau *et al.* 2017). Surprisingly, even camouflaged chicks that were invisible to observers on the ground were observed in the RPAS footage (Brisson-Curadeau *et al.* 2017). Responses to RPAS can vary species to species, (Vas *et al.* 2015; Bevan *et al.* 2018; Holldorf 2018), but it is possible to undertake remote sensing that is minimally invasive (Borrelle & Fletcher 2017).

There are a number of pressures that may impact the avifauna in the Marine Park. These include climate change, human presence, non-native invasive species and marine pollution (Director of National Parks 2018). Climate change impacts can include changes in sea surface temperature, sea currents, the intensity and frequency of storms and marine heatwaves. These can have direct impacts on the avifauna, or indirect impacts such as affecting presence and abundance of prey resources (Chambers *et al.* 2011). Tourism and scientific research are allowable activities in the Marine Park and recreational use allowed (Director of National Parks 2018). Human presence associated with these activities has potential to impact avifauna through disturbance and should be managed accordingly (Director of National Parks 2018). Non-native tropical fire ants (*Soleopsis geminata*) were first observed in the Marine Park in 1992 (Bellio *et al.* 2007), and many of the ground nesting seabirds could potentially be impacted by the ants. Whilst there is no direct evidence of adverse impact on the seabirds, it is suggested that the ants could potentially impact the young chicks of Red-tailed and White-tailed Tropicbirds (*Phaethon rubricaudus and P. lepturus* respectively), and both hatching and young chicks of Bridled Terns (*Onychoprion anaethetus*), Crested Terns (*Thalasseus bergii*), Lesser Crested Terns (*T. bangalensis*), Roseate Terns (S*terna dougalli*), Sooty Terns (*O. fuscatus*), Brown Noddies (*Anous. stolidus*) and Black Noddies (*A. minutus*) (Bellio *et al.* 2007). Marine pollution including plastic and other marine debris is known to be ingested by birds and can have both sublethal and lethal outcomes (Roman *et al.* 2016). Noise and light pollution have potential to disturb or disorientate birds. For example, lights on ships have been associated with bird strikes (Black 2005) and simulated aircraft noise of various noise levels increased disturbance behaviours in a seabird colony such as heightened alert levels, startle/avoidance behaviour (where the bird may move off its nest momentarily) and escape, (where the bird flies away) (Brown 1990).

In turn, high densities of nesting seabirds can also have both negative and/or positive impacts on vegetation. These include physical damage, reduced or increased species richness, alteration in community composition and excessive nutrient deposition which also affects the soil pH (Gillham 1960; Ellis 2005). The nature of the impacts on the vegetation occurring on the Ashmore Reef islands remain largely unknown.

4.3 Objectives

To determine the population status and diversity of seabirds and shorebirds within the Ashmore Reef Marine Park to inform future monitoring and management and identify any interactions between tropical fire ants and birds (covered in Chapter 5), as well as associations between birds and vegetation.

4.4 Methods

4.4.1 Survey logistics

Following the methodology of Clarke *et al.* (2011), island-wide ground counts of seabirds, egrets, herons and shorebirds were undertaken during daylight hours from 1/5/2019–6/5/2019 (inclusive) on East, Middle and West islands, Splittgerber Cay (to the east of East Island) and the three tidal sand cays between East Island and Middle Island (Figure 31). Due to tides and tender draft depths, all but West Island were accessible for approximately four hours: two hours on either side of each high tide in a 24-hour period. During the fieldtrip, the time of high tide varied from 0922h–1135h and 2129h–2340h. East and Middle Island were both visited on two occasions. One visit to each island was for approximately four hours, during the morning high tide. The other visit extended through to the evening high tide, though no counts were possible after dusk. West Island was visited on three occasions, and the cays were each visited once.

4.4.2 Aerial surveys

In addition to ground counts, optical (visible RGB) surveys were conducted with a DJI Phantom 4 Pro RPA system at 30 m above ground level with images overlapped by >70% to allow for accurate stitching. Missions were conducted as for the vegetation surveys (see chapter 3) with no observed impact on birds. The missions on each island were conducted prior to other researchers landing on the island. Image processing followed that of the vegetation surveys to produce a single high-resolution georeferenced image for East, Middle and West islands and Splittgerber Cay. The chosen software pipeline treats objects that are not consistent between still images (e.g. moving birds) as anomalies and removes them from the final stitched image. Thus, the final stitched image used for bird counts may represent a slightly more conservative count estimate relative to reality. However, there are no known published data from seabird surveys using RPAS that indicate the percentage of flying birds that would have been removed via image stitching. Furthermore, it is possible that a percentage of flying birds would have been settled, and thus not discounted, by the time the next overlapping survey was conducted. For this reason, birds in flight were not counted.

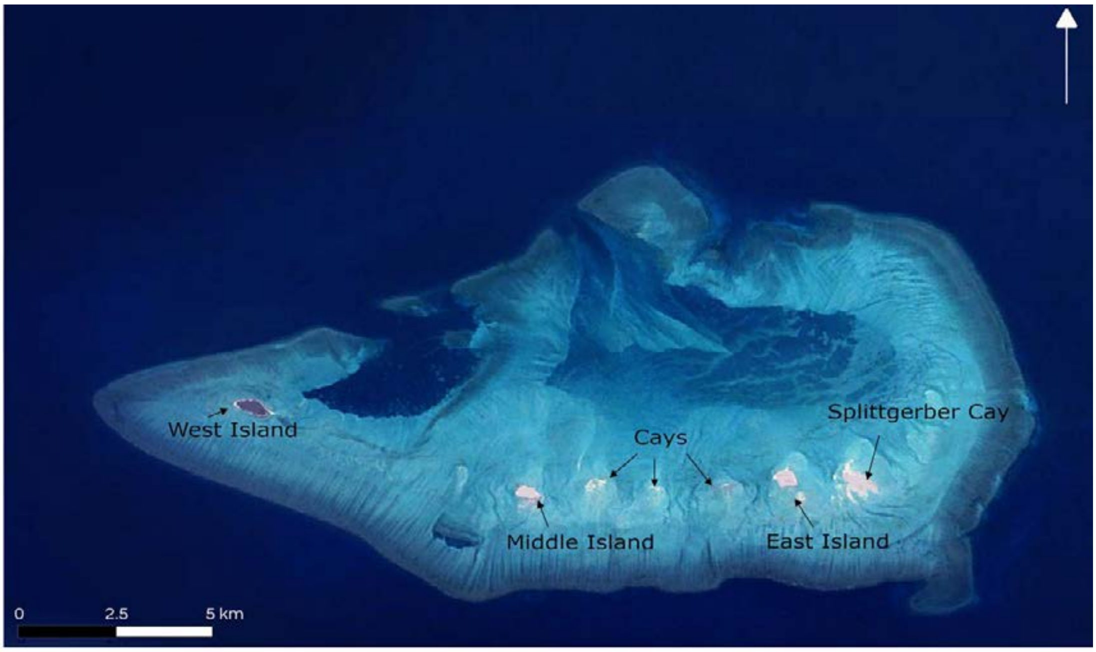


Figure 31. The location of West, Middle and East islands and the four cays within the Ashmore Reef Marine Park that were surveyed for seabirds, egrets, herons and shorebirds in May 2019

4.4.3 Ground surveys: seabirds, egrets and herons

For breeding seabirds, a single count was completed of active nests present on West, Middle and East Islands, and Splittgerber Cay. The exception to this were the counts of Brown Booby, Black

Noddy and Crested Tern nests on West Island. It was not possible to view all the breeding adults on this island without unduly disturbing the large numbers of nesting or roosting Sooty Terns and Brown Noddies. Therefore, estimates of these species were taken from the RPA aerial images alone. Additionally, accurate ground counts of frigatebirds were hampered by their propensity to often nest within dense, low-lying vegetation. Single counts were also conducted of adult non-breeding birds present in small numbers such as Roseate and Bridled Terns. The counts were undertaken by an avian biologist experienced in both the identification of the avian fauna within Ashmore Reef Marine Park and in counting large aggregations of birds. Spotting scopes and binoculars were used for counts, as well as digital SLR photography. GPS coordinates were obtained for all tropicbird nests. Counts were not completed for sooty terns and brown noddies, due to the 1) limited field time, 2) large numbers of individuals and 3) the high Critical Approach Distance (CAD) for both species (i.e. disturbance occurred when researchers were >20 m from groups of birds, with all the birds flying away). The high CAD was due to their pre-breeding/nest building stage, as there is less need for them to remain on site. Thus ground-truthing of the areas used by these two species was completed, and counts were obtained from the RPA aerial images.

Dominant vegetation type associated with breeding seabirds was identified in one of two plot sizes (3 x 5 m or 5 x 5 m) on each island. On West Island, nest sites were counted in eight plots (5 x 5 m), running NE to SW (Figure 32). The first near the centre of the Island was chosen based on the presence of a large number of sooty terns prior to our approach. On Middle Island, nest sites were counted in 49 plots (3 x 5 m) running NE to SW and also in an additional randomised 5 plots (3 x 5 m) (Figure 33). On East Island, nest sites were counted in 15 plots (3 x 5 m) running approximately NE to SW (Figure 34).

Figure 32. Location of 5 x 5 m plots on West Island, Ashmore Reef during 2019 surveys. The location of the first plot near the centre of the island was chosen based on the presence of a large number of sooty terns.

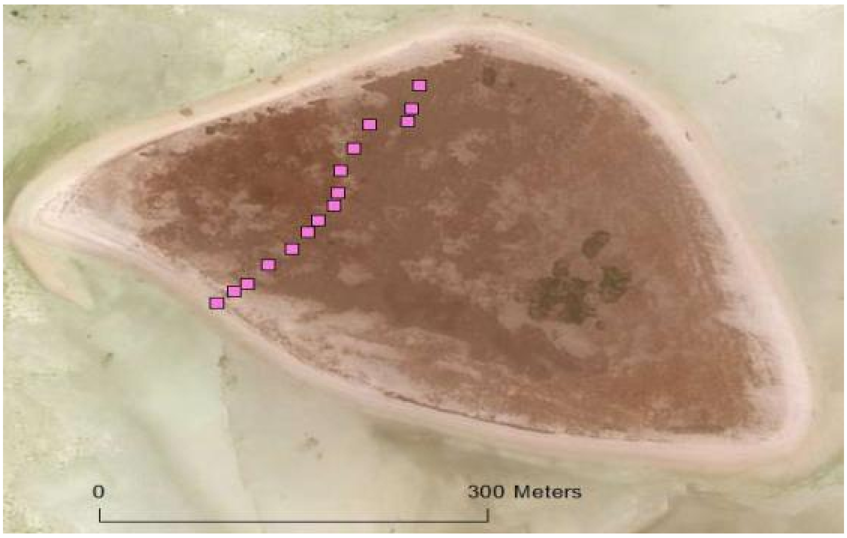


Figure 34. Location of the 15 plots (3 x 5 m) on East Island, Ashmore Reef during 2019 surveys

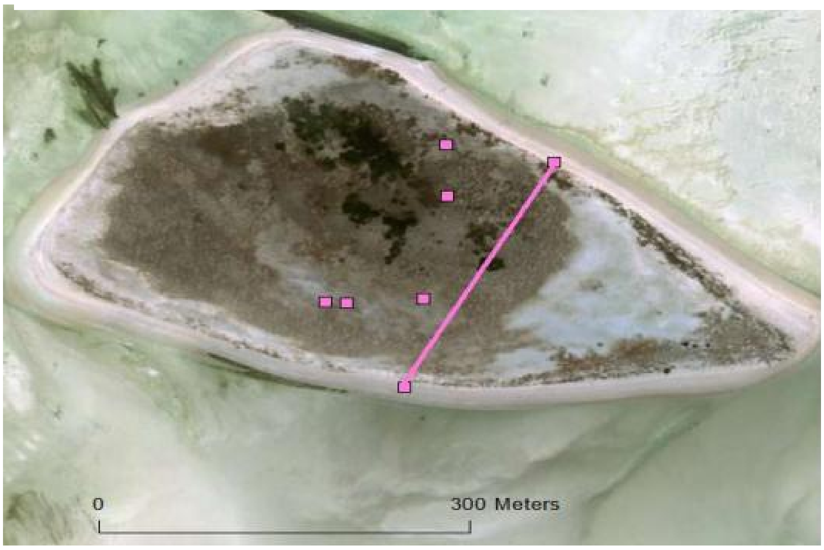


Figure 33. Location of transect, consisting of 49 plots and 5 randomised plots (3 x 5 m) on Middle Island, Ashmore Reef during 2019 surveys.

4.4.4 Ground surveys: migratory shorebirds

Migratory shorebirds were counted on East, Middle and West Islands, Splittgerber Cay and the three tidal sand cays between East Island and Middle Island during high tide. Survey points were selected based on cover and access and all birds were counted before moving to the next survey point until all migratory shorebirds had been counted along each beach. Counts were conducted using spotting scopes and binoculars as well as photography. Digital SLR photographs were taken of each roosting bird flock and were later used to confirm both numbers and species present at each site.

4.4.5 Counts from aerial images

Processed aerial imagery was imported into ArcGIS 10.7, and was used to count the number of breeding and non-breeding seabirds, as well as egrets and herons. Each island was zoomed in to a magnification that allowed identification of individual birds. A mark was placed on each bird, with a different colour and number being assigned to each species. The count of each of the species was obtained by adding all values within the layer's properties (Figure 35).

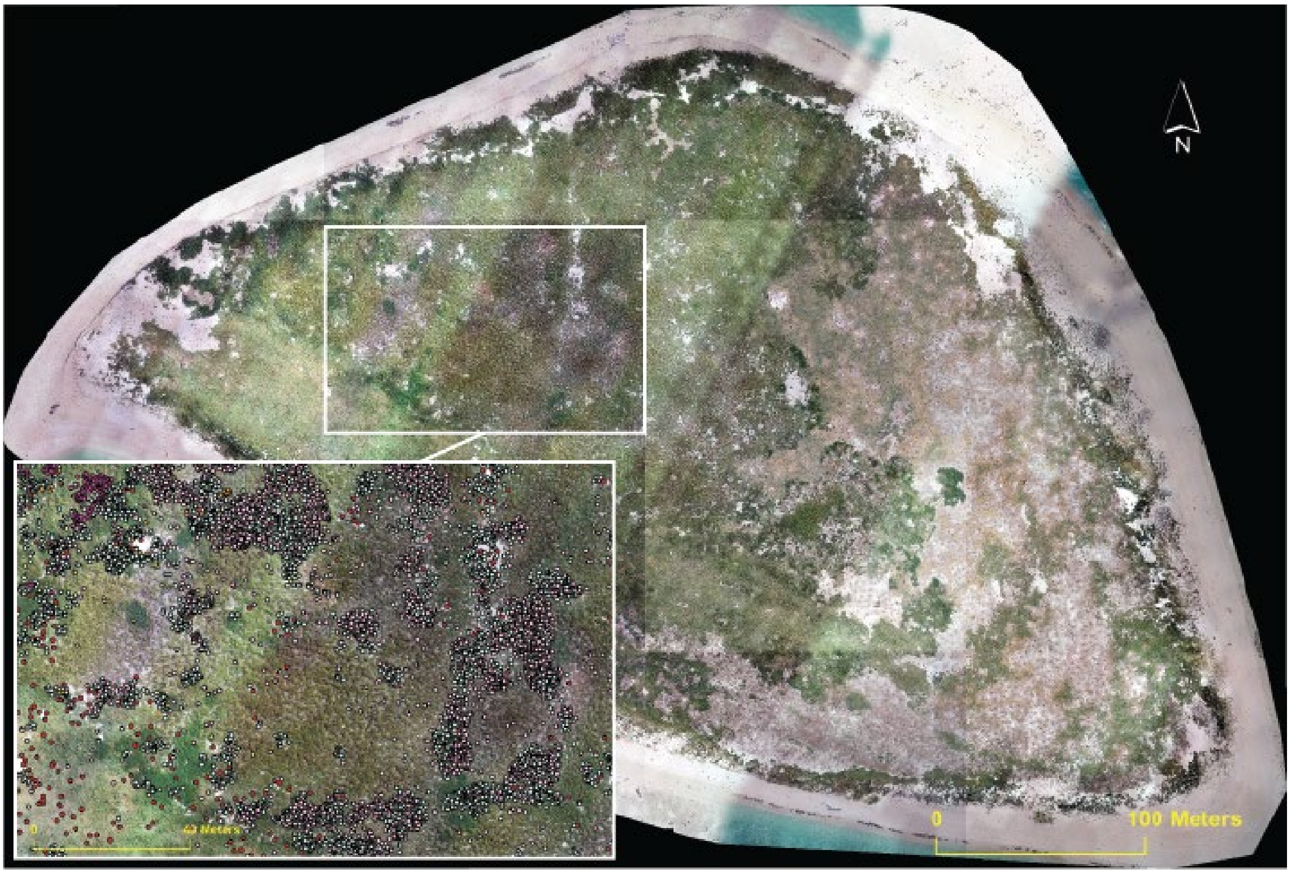


Figure 35. Map of East Island, Ashmore Reef, with inset of a section where Brown Noddies (aqua: n=4,952), Sooty Terns (light pink: n=4,151), Brown Boobies (red: n=228), Lesser Frigatebirds (dark pink: n=178) and Bridled Terns (orange: n=21) were identified in May

The counts of all the seabirds, herons and egrets on each island took several days per island. Due to time restrictions, it was only possible to conduct a single count of all the seabirds, herons and egrets from the stitched RPAS images that were compiled for each island. Therefore, we were unable to assess the degree of variability between counts on the same island.

4.5 Results

4.5.1 Seabird, Egret and Heron diversity and ecology

Seventeen seabird species two egret species and one heron species were recorded on the islands and cays. Twelve species had begun breeding in May 2019 (Table 16): Brown Boobies, Red-footed Boobies (*S. sula*), Masked Boobies (*S. dactylatra*), Crested Terns, Black Noddies, Lesser Noddies (*A. tenuirostris*), Great Frigatebirds (*Fregata minor*), Lesser Frigatebirds, Red-tailed Tropicbirds and White-tailed Tropicbirds (Table 16). Brown Noddies and Sooty Terns were present on each island and were predominantly engaged in a range of behaviours including roosting, courtship behaviour and attending nest sites. Furthermore, laying had just commenced at the time of surveying. As such, there were very few eggs noted. However, there was evidence that Sooty Terns had experienced a failed breeding attempt prior to our visit, with addled eggs found on West, Middle and East islands. Despite this, the current breeding stage of both the Sooty Terns and Brown Noddies was more advanced on West Island compared to both Middle an East islands. As such, on West Island it was possible to differentiate between birds attending nest sites and those prebreeding birds (ie. roosting birds or those displaying courtship behaviour). Bridled Terns and Roseate Terns were displaying courtship behaviour (Table 17). Small numbers of Common (*Sterna hirundo*), Little (*S. albifrons*) and Gull-billed (*Gelochelidion nilotica*) Terns were also observed on the foreshores and the cays (Table 17). Given the number of different stages for the seabirds, the counts have been summarised in either of two categories 1; those that were attending nest sites, i.e. likely to represent a breeding pair (Table 16), and 2; total counts of those birds roosting/pre-breeding (Table 17).

Both species of tropicbirds were found only on West Island. Forty-two breeding Red-tailed Tropicbirds and one breeding White-tailed Tropicbird were observed (Table 16, Figure 36). Additionally, Black and Lesser Noddies were only found on West Island (Table 16). Wedge-tailed Shearwaters (*Ardenna pacifica*) were not observed, but this was expected given that they are wet season (Nov-Apr) breeders. Examination of the ten burrows located during this survey indicate that this site had not been utilised for at least one breeding season. The burrows were amongst overgrown vegetation, the entrances partly collapsed and there were no indicators of recent activity (e.g. recent digging, footprints, guano streaks or chick down at the mouth of any burrows) nor of any burrow maintenance that would be expected if this colony was still extant.

Eastern Reef Egrets (*Egretta sacra*), Little Egrets (*E. garzetta*) and Nankeen Night Herons (*Nycticorax caledonicus*) were observed on West, Middle and East islands (Table 18). Although some of the Eastern Reef Egrets were observed within vegetated sections of Middle Island and East Island, they did not appear to be breeding.

Figure 36. The location of breeding Red-tailed (red triangle) and White-tailed (green triangle) Tropicbirds on West Island, Ashmore Reef Marine Park in May 2019.

4.5.2 Population counts for seabirds, Egrets and Herons

Counting both breeding pairs and single non-nesting birds, Sooty Terns were the most numerous birds within the Ashmore Reef Marine Park, with 77,309 counted across West, Middle and East islands (Table 16 and Table 17). Brown Noddies were the second most abundant, with 40,383 counted. Other significant numbers included Brown Boobies- 30,168, Lesser Frigatebirds- 7,848, Black Noddies-5,126 and Crested Terns- 4,458 (Table 16 and Table 17). Eastern Reef Egrets were the most abundant of the egrets and herons (Table 18), with 393 counted on West, Middle and East islands and Splittgerber Cay.

Table 16. The counts of breeding pairs on East, Middle and West islands and Splittgerber Cay within the Ashmore Reef Marine Park in May 2019. Total counts of adults would therefore be twice the number tabulated. No seabirds were breeding on the other three cays.

Species	East Island	Middle Island	West Island	Splittgerber Cay
Great Frigatebird		13	38	
Lesser Frigatebird	1739	2150	35	
Brown Booby	6835	7693	457	99
Red-footed Booby	3	10	235	
Masked Booby	81	112		
Crested Tern	957		1214	58
Sooty Tern			28031	
Brown Noddy			8127	
Black Noddy			2563	
Lesser Noddy			20	
Red-tailed Tropicbird			42	
White-tailed Tropicbird			1	

Table 17. The counts of adult seabirds on East, Middle and West islands and the four cays within the Ashmore Reef Marine Park in May 2019. Birds were either roosting or pre-breeding.

Species	East Island	Middle Island	West Island	All Cays
Sooty Tern	18897	175	2205	
Brown Noddy	18955	3513	1661	
Bridled Tern	289	92	13	6
Roseate Tern	14	2	57	5
Little Tern				10
Gull-Billed Tern				1
Common Tern	1			

Table 18. The counts of egrets and herons on East, Middle and West islands and the four cays within the Ashmore Reef Marine Park in May 2019

Species	East Island	Middle Island	West Island	All Cays
Eastern Reef Egret	105	88	59	141
Little Egret	11	2	13	9
Nankeen Night Heron	3	2	2	

4.5.3 Vegetation associations amongst seabirds at Ashmore Reef

Ground nesting species

Most species of seabirds that breed at Ashmore Reef nest on bare sandy ground in open areas or amongst the adjacent low herb fields common to the basin of each island.

Crested Terns and Roseate Terns

The two dense colonies of Crested Terns were situated on bare open patches of sand found either along the vegetation/beach interface just above the high water mark (e.g. East Island) or bare sandy patches between areas of *Sesbania cannabina* and *Sida pusilla* inland on West Island (Figure 37). Nests densities were as high as 9/m^2 and followed the shape of bare ground. Nests densities were as high as 9 nest/m^2 and covered all sandy areas to the edges of the fringing vegetation. A single colony of Roseate Terns (14) was observed defending and preparing nest sites amongst the bases of taller tufted grass *Lepturus repens* in the low foredunes of the south-eastern corner of East Island (Figure 38).

Sooty Terns and Brown Noddies

Sooty Terns on West Island were found nesting amongst the extensive herb fields of *S. pusilla* and *S. cannabina* thickets and extending from bare areas, also utilised by Brown Noddies, to areas under the cover of vegetation. Sooty Terns nested often in association with Brown Noddies on the more open sand patches fringed with clumped grasses on each island.

Brown Noddies nested in exposed open areas within the extensive herb fields on clumps of *Digitaria mariannensis* and *Eragrostris cumingii*. Elsewhere, in more coastal regions of East and Middle Islands, a few Brown Noddies were beginning to construct nests atop the dominant tufted grass *L. repens*. At the time of our survey, Brown Noddies were not observed nesting in coastal areas of West Island.

Boobies

Brown Booby nests were found across all low vegetation types in the herb fields, as well as across the beach head and open sand areas devoid of vegetation. In many instances, nest sites and guano fans associated with Brown Booby nests were clearly visible in aerial photography, with a clear central portion around each nest and a fringe of richer vegetation. On West Island, Brown Boobies preferred low growth areas dominated by the grass *D. mariannensis* and the herbs

Boerhavia sp. and *Portulacca tuberosa*. On East and Middle Islands, they nested in loose colonies across the herbaceous *Amaranthus interruptus* and *E. cumingii* as well as the extensive bare sand patches found on the eastern points of each island.

Masked Boobies nested in association with areas of denser Brown Booby colonies on Middle and East Islands. All nest sites were located on exposed sandy areas.

Frigatebirds

Lesser Frigate birds on East and Middle Islands were nesting atop patches of vegetation slightly elevated from the surrounding bare ground or vegetated areas dominated by the grass *D. mariannensis*. They preferred areas of dense *Tribulus cistoides* (beach caltrop) and *A. interruptus* with occasional *Cuscuta victoriana* (Figure 39).

Tropicbirds

Red-tailed Tropicbirds were only found nesting on the ground under the dense canopies of *H. foertherianum* (Figure 41). The single White-tailed tropicbird located was hidden amongst tall *Digitaria mariannensis* at the edge of an *Heliotropium foertherianum* (octopus bush) on West Island.

Tree nesting species

Tree nesting species have adapted to a change in the availability of nesting trees since the last survey in 1997. On Middle Island, the last of the remaining coconut palms have gone, and only one live *H. foertherianum* remains, along with the skeletonised branches of a few recently dead individuals. A few Great Frigatebirds and Red-footed Boobies used these for nesting. However, this change in availability of nesting trees has meant that West Island is the only area with extensive tree or large shrub nesting habitat remaining for Red-footed Boobies, Greater Frigatebirds and Black and Lesser Noddies to utilise.

Frigatebirds

Great Frigate birds nested in living or remnant (dead) shrub structures, principally *H. foertherianum* on West and Middle islands.

Boobies

Red footed Boobies nearly always built their nests in the *H. foertherianum* on West and Middle islands. The single *Cordia subcordata* (sea trumpet) remaining on West Island (c. 1m high) was too small to support any nesting. However, there were three nests on East Island and four nests on Middle Island that were constructed atop low *A. interruptus* bushes. Numbers of nesting Red-footed Boobies has continued to decline at both Middle and East Islands likely due to less suitable nesting habitat.

Noddies

Black and Lesser Noddies at Ashmore Reef built their elaborate nests on the lower branches of the octopus bush and on taller thickets of *Sesbania cannabina* bushes that grow in patches across the central herb fields of West Island (Figure 40). Nesting material included seaweed fronds (*Cyclostoma* sp.), the stems of *Portulacca* sp. and *Amaranthus interruptus* and lined with the leaves of *Ipomoea* sp. cemented together with guano.

Figure 37. a) Crested Terns and b) close up of Crested Terns, nesting on bare open patches between areas of *Sesbania cannabina* and *Sida pusilla* on West Island. Photos: C. Surman

Figure 38. Pre-breeding Roseate terns on East Island, Ashmore Reef Marine Park, May 2019. Photo C. Surman

Figure 39. Adult and juvenile Lesser Frigatebirds nesting on East Island, Ashmore Reef Marine Park, May 2019. Photo C. Surman.

Figure 40. Red-tailed tropicbird nesting under *Argusia argentea* on West Island, Ashmore Reef Marine Park, May 2019. Photo C. Surman

Figure 41. Lesser Noddy nests on West island, Ashmore Marine Reef Marine Park on a) *Heliotropium foertherianum* and b) in *Sesbania cannabina* thickets in May 2019. Photo C.Surman

4.5.4 Comparison of counts between methodologies

The RPAS imagery provided high resolution imagery and high accuracy for counting all species. Ground counts were on average 17±14% lower than those counts of the same species obtained from the RPAS imagery (range 28% lower to 10% higher) (Figure 42). The drawbacks with the RPAS imagery were the difficulty in distinguishing between male Great and Lesser Frigatebirds, and between Sooty and Bridled Terns. For the former, we ground-truthed the location of the Great Frigatebirds during the ground counts. We used the typical behaviour of each of the terns to differentiate between them on the RPAS imagery. Bridled Terns are unlikely to be located in close proximity to one another in large groups. Thus, terns that were observed in groups <6 were considered to be Bridled Terns.

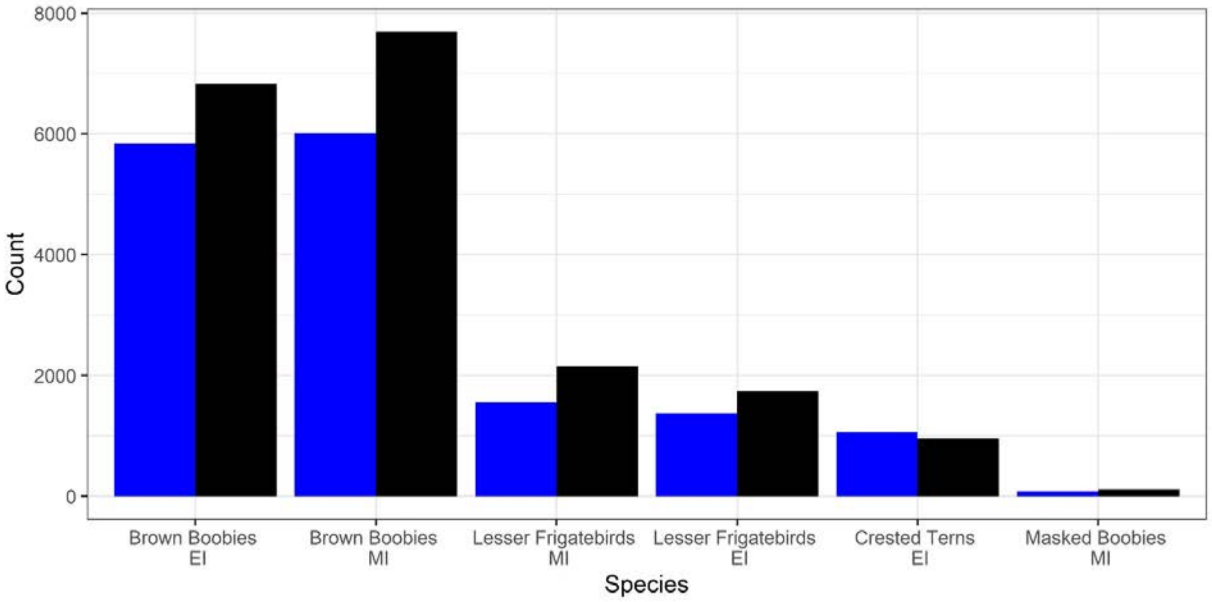


Figure 42. Comparison between ground (blue) and RPS (black) counts of seabirds on Middle and East islands, Ashmore Reef Marine Park, May 2019.

4.5.5 Status of seabird populations on Ashmore Reef

Expansion in breeding distributions

There have been some notable changes in the seabirds breeding on the three largest islands when comparing counts from 2010-2014 (Clarke & Herrod 2016) with this study (Table 19). Five species have expanded their breeding territories from previously only East and Middle Islands between 1990-2014 to now also breeding on West Island in 2019. The species which now also breed on West Island are Red-footed Boobies, Brown Boobies, Lesser Frigatebirds, Sooty Terns and Brown Noddies. Furthermore, Roseate and Bridled Terns were displaying pre-breeding behaviour on West Island in 2019, and presumably would breed there too (Table 19), yet both species had only previously been observed breeding on East and Middle Islands.

Shifts in breeding distributions

Four species appeared to have shifted breeding islands. Black and Lesser Noddies had been regular breeders on East and Middle Islands prior to 2015, albeit numbers were dwindling on Middle Island. Figure 43 and Figure 44). Prior to 2015, these species were not found on West Island. This situation was reversed in 2019, with these two noddies only observed on West Island (Table 19). Great Frigatebirds were observed breeding on Middle Island and occasionally on East Island but not on West Island from 1990-2014. In this study they were not observed on East Island but were breeding on West Island (Table 19). Crested Terns had previously been observed breeding regularly on East and Middle Islands but infrequently on West Island. In this study, they were not observed breeding on Middle Island but were breeding on West Island (Table 19). In addition, Crested Terns were breeding on Splittgerber Cay in 2019, whereas they had not previously been recorded there (Table 19).

It is possible that the expansion in breeding locations within Ashmore Reef Marine Park, was just a redistribution of birds. Certainly, fewer Sooty Terns were observed on Middle Island during this study compared to the maximum count observed in April 2014 (175 compared to 10,028, Figure 43). However, more Sooty Terns were observed on East Island in 2019 (Figure 44). When comparing Middle Island prior to 2015 and the present study, the current abundance of adults was lower for Brown Noddies, Lesser Frigatebirds, Crested Terns, Black Noddies, Red-footed Boobies, Lesser Noddies and Great Frigatebirds (Figure 43). Notably though, the abundance of all these species had been declining since 2012 or before. Similarly, for East Island, reduced numbers were noted for Brown Noddies, Black Noddies, Crested Terns, Red-footed Boobies, Lesser Noddies and Great Frigatebirds (Figure 44). Thus, there has been a significant redistribution of many of the seabird species within the Marine Park.

Changes in abundance of seabirds

The only seabirds previously reported to regularly breed on West Island were the Tropicbirds and Wedge-tailed Shearwaters (Clarke & Herrod 2016). Therefore, it is possible to determine the apparent change in the abundance of adults counted for those nine seabird species that were previously not observed breeding on West Island, and for species never observed to breed on West Island (note this is not a population estimate given that only total numbers of adults, not numbers of breeding pairs, can be sourced from Clarke and Herrod, 2016). It appears that the numbers of adults observed of four species have increased from 2010: Sooty Tern, Brown Booby, Red-footed Booby and Black Noddy (Figure 45). Great Frigatebirds are showing a recovery, i.e. from elevated numbers observed in 2010 followed by a decline in subsequent years (Figure 45), and the current apparent abundance is marginally less than in 2010. The numbers of Lesser Frigatebirds and Crested Terns appear to be recovering from a low in 2014 (Figure 44), but are still in lower numbers compared to 2010. The number of Masked Boobies, only ever observed breeding on Middle and East Islands, has almost quadrupled since 2014, to 193 adults observed on the islands (Table 16).

Only two species appear to be in decline. The number of Brown and Lesser Noddies appear to have peaked in 2012 or 2013, respectively (Figure 44).

Species	East Island 1990-Nov 2014	East Island May 2019	Middle Island 1990-Nov 2014	Middle Island May 2019	West Island 1990-Nov 2014	West Island May 2019	Splittgerber Cay 1990-Nov 2014	Splittgerber Cay May 2019
Wedge-tailed Shearwater	-	-	-	-	Regular	-	-	-
Red-tailed Tropicbird	Infrequent	-	Infrequent	-	Regular	Breeding	-	-
White-tailed Tropicbird	Infrequent	-	Infrequent	-	Regular	Breeding	-	-
Masked Booby	Regular	Breeding	Regular	Breeding	-	-	-	-
Red-footed Booby	Regular	Breeding	Regular	Breeding	-	Breeding	-	-
Brown Booby	Regular	Breeding	Regular	Breeding	-	Breeding	Occasional	Breeding
Great Frigatebird	Occasional	-	Regular	Breeding	-	Breeding	-	-
Lesser Frigatebird	Regular	Breeding	Regular	Breeding	-	Breeding	-	-
Crested Tern	Regular	Breeding	Regular	-	Infrequent	Breeding	-	Breeding
Roseate Tern	Regular	Prebreeding	Regular	Roosting	-	Prebreeding	Occasional	Roosting
Bridled Tern	Regular	Prebreeding	Regular	Prebreeding	-	Prebreeding	-	Roosting
Sooty Tern	Regular	Prebreeding	Regular	Prebreeding	-	Prebreeding/Breeding	-	-
Brown Noddy	Regular	Prebreeding	Regular	Prebreeding	-	Prebreeding/Breeding	-	-
Black Noddy	Regular	-	Regular	-	-	Breeding	-	-
Lesser Noddy	Regular	-	Regular	-	-	Breeding	-	-
Little Egret	Infrequent	Roosting	Infrequent?	Roosting	Occasional	Roosting	-	Roosting
Intermediate Egret	Infrequent	-		-	-	-	-	-
Eastern Reef Egret	Regular	Roosting	Regular	Roosting	-	Roosting	Occasional	Roosting
Nankeen Night-Heron	-	Roosting	Infrequent	Roosting	-	Roosting	-	-

Table 19. Comparison of breeding distribution of seabirds and herons on East, Middle and West islands and Splittgerber Cay, Ashmore Reef, from 1990-November 2014 (sourced from Clarke and Herrod 2016) and May 2019. As there were multiple surveys prior to 2019, it was possible to determine if a seabird was a regular, occasional, or infrequent breeder. However, the 2019 data, the breeding, prebreeding or roosting presence is noted. Dashes indicate 1) breeding activity was not observed (for data 1990-Nov 2014), or 2) no adults were observed (May 2019)

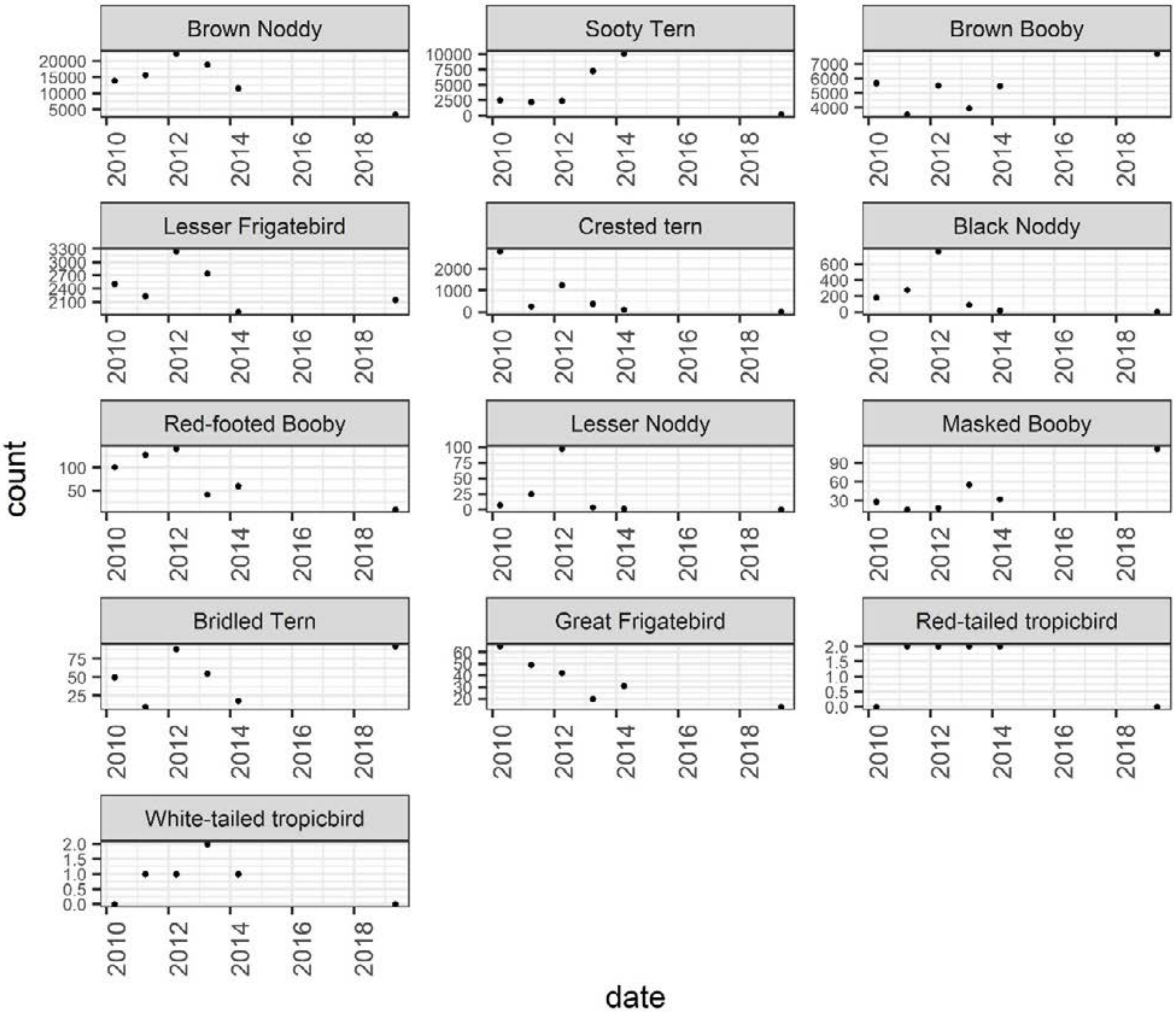


Figure 43. Counts of adults of each species observed on Middle Island, Ashmore Reef Marine Park, in April 2010-2014 (sourced from Clarke and Herrod 2016) and May 2019.

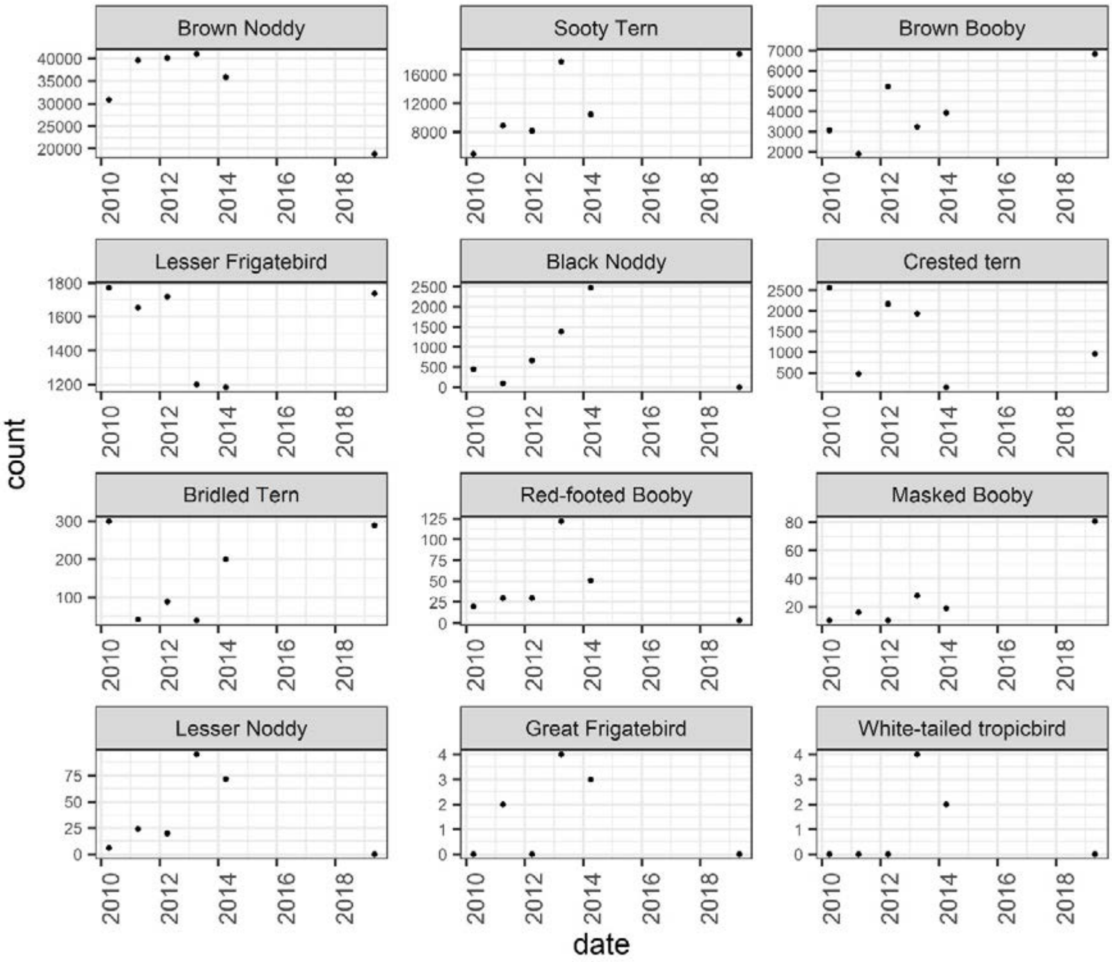


Figure 44. Counts of adults of each species observed on East Island, Ashmore Reef Marine Park, in April 2010-2014 (sourced from Clarke and Herrod 2016) and May 2019.

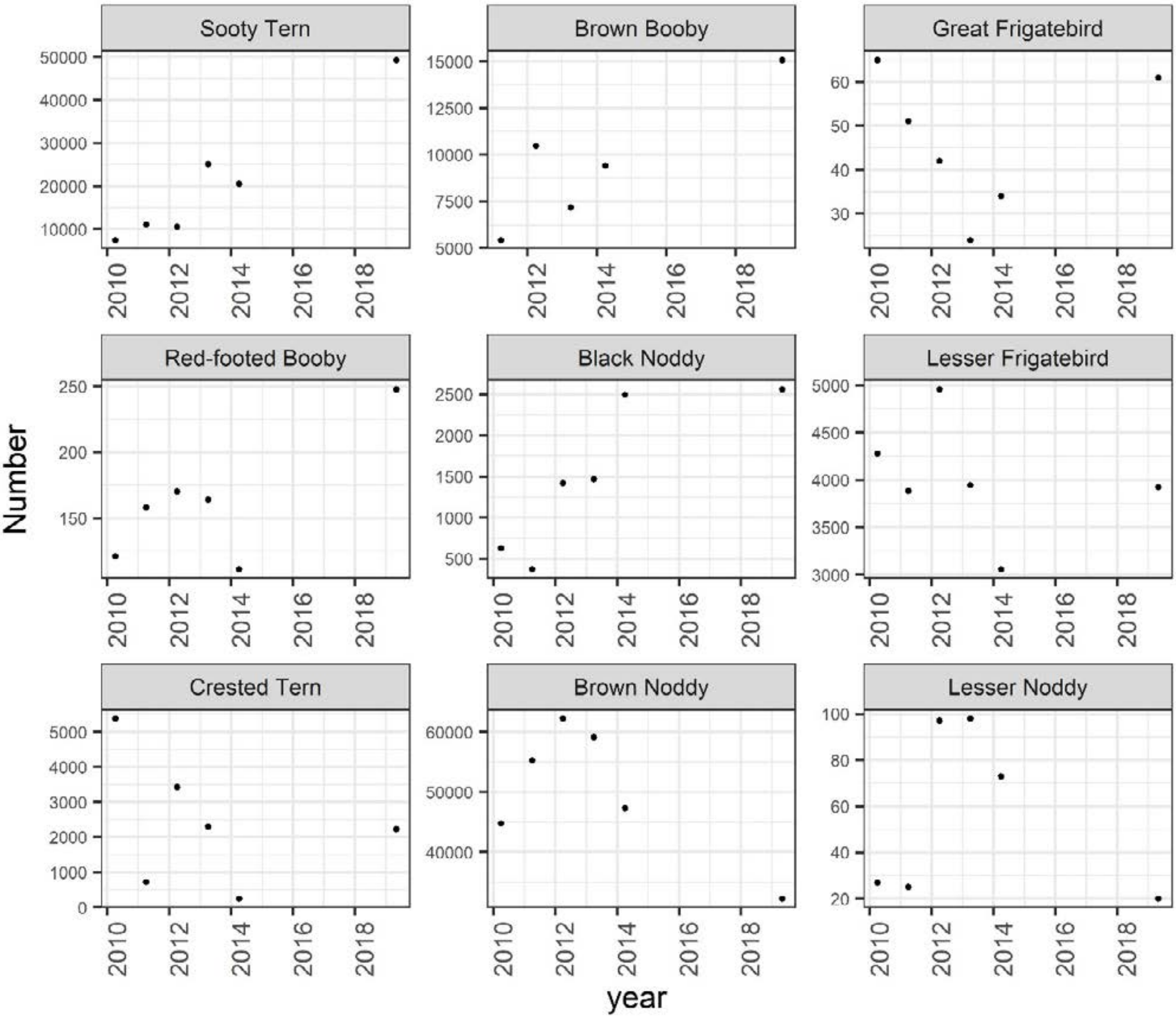


Figure 45. Apparent change in the numbers of adults of 9 seabird species observed across all islands and cays within Ashmore Reef Marine Park from 2010-2019. Data from 2010-2014 from Clarke and Herrod (2016). Note that these numbers are not population estimates, given that many of birds were breeding (at least in 2019), and thus estimates will be much higher than the numbers indicated.

4.5.6 Shorebirds

Whilst 30 species of shorebirds have been observed at Ashmore Reef Marine Park from 1979-2014, numbers and species present were generally greatest from October-January. During this study, 17 species of shorebirds were observed, predominantly on Ashmore's four cays and Middle Island. Only one species was found on East Island (Ruddy Turnstones) and Curlew Sandpipers and Whimbrels were the only shorebirds found on West Island (Table 20).

Table 20. The count of shorebirds observed on East, Middle and West islands and the four cays within the Ashmore Reef Marine Park in May 2019.

Species	Splittgerber Cay	Cay 1	Cay 2	Cay 3	East Island	Middle Island	West Island
Large Sand Plover	99	1	2	12		18	
Mongolian Sand Plover	15	1	5	3		2	
Pacific Golden Plover	6		2	3		1	
Grey Plover	42	24	73	12			
Bar Tailed Godwit	52	37	38	1		10	
Common Greenshank	12	4	1	6		2	
Grey-tailed Tattler	46	17	38	9		2	
Whimbrel	26	7		8			3
Eastern Curlew						1	
Curlew Sandpiper	11			1			1
Terek Sandpiper	4		1				
Sanderling	20	14	5	42		42	
Red-necked Stint	210		2			148	
Ruddy Turnstone	10				30	59	
Great Knot	95	22	44	14			
Red Knot	4			1			
Black-winged Stilt	0	0	0	2		10	

4.6 Discussion

The scope of this study was to determine the population status and diversity of seabirds and shorebirds within Ashmore Reef Marine Park to inform future monitoring and management; to identify any interactions between tropical fire ants and seabirds; and associations between birds and vegetation. Our approach to achieve these aims was to use ground-counts, counts from aerial imagery, and transects in May 2019. The assessment of interaction between seabird and tropical fire ants was undertaken by the research team studying the population of tropical fire ants, and is not covered in this section (See chapter 5). Furthermore, very few shorebirds were present during this study, which is not surprising given that peak numbers are present from October-December (Milton 2005; Clarke & Herrod 2016). It was therefore not possible to investigate the population status of shorebirds even though low numbers of shorebirds from 17 different species were present within the Marine Park during this study. This study has identified that Ashmore Reef Marine Park continues to have internationally significant numbers of seabirds and, from a seabird perspective, meets Criteria 5 (> 20,000 waterbirds) and 6 (more than 1% of the population) for Ramsar Sites. Interestingly, Criterion 6 has only been acknowledged for Sooty Terns. However, the Marine Park also continues to meet Criterion A4 for Important Bird Areas for Lesser Frigatebirds and Brown Boobies (it was not possible in this study to determine if Grey-tailed Tattlers still met this criteria). Thus, these two species should be added to the justification of Ashmore Reef Marine Park as a Ramsar Site. In 2019, breeding appeared to be delayed for several species, whilst others had either expanded their breeding distribution to cover East, Middle and West islands, or had altered the islands on which they were breeding. For the nine species of seabirds previously not found to be breeding on West Island and the one species only found breeding on East and Middle Island, five species have increased, three are recovering after lows in 2013 or 2014, and two are in decline.

Addled Sooty Tern eggs were observed during the transects on the islands. Furthermore, Sooty Terns, Brown Noddies and Bridled Terns were either engaged in pre-breeding behaviour or had just begun to lay eggs (Sooty Terns and Brown Noddies only). This is in contrast to findings from Clarke and Herrod (2016), which suggest that all three species lay eggs in April. In April 2019, a category 1 cyclone (Cyclone Wallace) passed within 64 km of the Marine Park, with wind gusts of up to 85 km/hr (https://web.archive.org/web/20190406133332/http://www.bom.gov.au/cgi-bin/wrap_fwo.pl?IDW27600.txt). It was later upgraded to a Category 3, but by that stage was more than 200 km from the island. A few days later it was followed by Tropical Low 22U that followed a similar path and with sustained wind gusts of 63 km/hr observed at Varanus Island (https://en.wikipedia.org/wiki/2018%E2%80%9319_Australian_region_cyclone_season). The only other cyclone that has occurred in autumn since 2010 was in April 2011, but at its closest was 200 km from the Marine Park. It is thus possible that these two weather systems were responsible for Sooty Terns abandoning their eggs, and for the delayed breeding of the three species.

The seabird species nesting within the Ashmore Reef Marine Park have different nesting habitat requirements. Both Black and Lesser Noddies had shifted from nesting on Middle and East islands prior to 2015 to solely nesting on West Island in 2019. This is because they need to elevate their nest sites, but the larger shrubs that had been on Middle and East islands in recent years, *H. foertherianum*, had died off. Interestingly, at the time of our survey, there was no evidence of Black Noddies amongst any *Sesbania* thickets on these islands even though they were nesting within these thickets on West Island. Lesser Frigatebirds on East and Middle islands were predominantly nesting amongst the non-native *Tribulus cistoides*, thus any weed management program on the islands must consider the potential impacts on this species (see Section 3.7.7 for more details on ecosystem interactions). The number of observed Red-footed Boobies has more than doubled since 2014, with them now principally nesting on West Island. It is likely that they have shifted to West Island due to a reduction in suitable nesting habitat on East and Middle islands, but it is unclear why they did not previously nest in the scattered *H. foertherianum* and *C. subcordata* shrubs on these islands.

In this study, the ground counts of several species were typically around 30% less than those obtained from counting birds on aerial images. Counting seabirds at colonies has been shown to be problematic, even for experienced counters (Hodgson *et al.* 2016; Hurford 2017). Ground counts are difficult especially with horizontal viewpoints where birds in front can obscure those further away, and birds in small undulations or within vegetation are difficult to see. Experienced ornithologists asked to estimate the number of birds in an image within a 30 second period typically underestimated the abundance by an average of 30% (Hurford 2017). Similarly, ground counts of Lesser Frigatebirds by experienced ornithologists on East and Middle islands in April 2014 were found to be underestimated by some 15-40% when compared to counts from RPAS images (Hodgson *et al.* 2016). However, birds that nest under vegetation can potentially be difficult to see from aerial imagery. Despite this, the counts are still likely to be more accurate for such species given that they are generally more visible on RPAS images compared to ground counts and flushing of birds is avoided. Such flushing reduces the ability to count the birds: it is much easier to count birds sitting on a nest or standing within the colony than birds flying away. In addition, birds can become entangled in the vegetation as they attempt to escape when approached, thus such potential negative impacts are avoided by obtaining aerial imagery. In summary, aerial imagery improves count accuracy and allows access to areas that would otherwise be difficult to access whilst reducing disturbance to the seabirds present.

The breeding seabird diversity on West Island was much greater than previously observed by Clarke and Herrod (2016), with at least eight additional species breeding on the island, two likely to breed, and one prior infrequent species also breeding there. In addition, there is an apparent increase in the numbers of observed adults in many of the seabirds across the Marine Park, as well as breeding being established by Brown Boobies and Crested Terns on Splittgerber Cay. The notable exceptions are Brown and Lesser Noddies. It is possible that the Lesser Noddies were underestimated from the aerial counts given their propensity to nest under vegetation. Given that many of the Brown Noddies counted were roosting, i.e. not engaged in breeding, it is possible that many of them were at sea, and thus reducing the observations of adults within the Marine Park. Overall, the Ashmore Reef Marine Park is supporting significant numbers of seabirds which are generally increasing.

Finally, there is some conjecture on the subspecies of Masked Booby breeding within Ashmore Reef Marine Park. It has previously been thought to be a member of *S. dactylatra bedouti* (Bellio *et al.* 2007; Clarke 2010). However, more recently is has been listed as *S.d. personata* (Gill & Donsker 2019). *S.d.bedouti* were thought to breed on four islands only, and as such was considered to be Vulnerable (Bellio *et al.* 2007and references within). Determining the subspecies breeding within Ashmore Reef Marine Park is thus important as this will have potential ramification on its status.

4.7 Management implications and recommendations

We propose a series of recommendations which will improve monitoring of the seabirds and shorebirds at Ashmore Reef Marine Park.

4.7.1 Methodology

Ideally, seabirds should be monitored annually with aerial imagery being undertaken 3–4 times a year capturing peak breeding events of seabirds. to determine

- Intra- and inter- species differences in timing of breeding, and,
- Intra- and inter-annual differences in population estimates.

The surveys should be undertaken in May (egg laying for most seabirds), August/September (when many seabird chicks will be visible, and others will be laying eggs) and November (potentially capturing breeding by Crested and Sooty Terns, Egrets and herons). Note that burrowing seabirds and the tropicbirds (that nest under cover) will not be captured from aerial images. However, a November aerial survey would also provide information on shorebird species presence and abundance.

Camera traps should also be set on each island to determine timing of breeding and breeding success for a subset of several species.

Data obtained from both types of remote photography would ensure that short, medium and long-term changes in breeding parameters and population estimates could be correlated with impacts such as climate variability or degraded breeding habitats. The imagery could be placed on citizen science-based websites, such as Zooniverse.org, where volunteers assist in obtaining data. These two methodologies meet several actions in the Draft Wildlife Conservation Plan for Seabirds (Commonwealth of Australia 2019), including:

- o *Complete a review of the conservation status of all seabirds in Australia*
- o *Investigate the impacts of climate variability and change on seabirds and their habitats,*
- o *Promote the conservation of seabirds and their habitats through strategic programs and educational products*

Temperature gauges should be deployed on at least one of the islands to measure daily terrestrial temperatures. Such data are not currently available, but terrestrial temperatures should be included in models to determine the significance of various environmental variables in seasonal and annual changes in breeding parameters and population estimates.

Whilst citizen science can be used effectively to assist in the counting of birds from aerial imagery, this is a very time-consuming process. Therefore, it would be prudent to research automated techniques for counting birds from aerial images. This would improve monitoring efficiency, and hence understanding changes in population estimates in a timely fashion.

4.7.2 Biology

The foraging habitats of many of the seabird species should be identified. This will provide information on important areas for the seabirds and is a necessary first step in determining likely impacts associated with climate change and marine pollution. Knowledge of areas used by the different species, both for travelling and foraging, can help identify the threatening processes these species are exposed to. Determining foraging habitats assists some of the actions from the Draft Wildlife Conservation Plan for Seabirds (Commonwealth of Australia 2019):

- *Identify important habitats for all seabirds during critical life stages*
- *Enhance contingency plans to prevent and/or respond to environmental emergencies that have an impact on seabirds and their habitats*
- *Obtain baseline data and continue to monitor pollutant concentrations in seabirds and their habitats*
- *Investigate the impacts of climate variability and change on seabirds and their habitats*

Prior to the development of weed management strategies within the Marine Park, it will be necessary to identify the potential impacts of weed removal on breeding seabirds. *Tribulus cistoides* was important nesting habitat for the Lesser Frigatebirds in 2019. It is not known if they nested in the *T. cistoides* in previous years, or indeed if this non-native species has grown over previous nesting sites. No other non-native plant species on the islands were preferentially used for nesting habitat by any other seabirds. It may be necessary, for example, to have a staggered eradication program, or to actively restore other nesting habitat species prior to the removal of substantial areas of non-native plant species (see Section 3.7.7 for factoring ecosystem interactions into management plans).

4.7.3 Taxonomy

The subspecies of Masked Boobies breeding within the Ashmore Reef Marine Park should be determined as this will have potential ramification on its status, that is, whether they are listed as Vulnerable or not. This is important as they have a high fidelity to their breeding colony, with little gene flow between colonies.

4.7.4 Managing use

Five species have expanded their breeding distribution in the Marine Park to include West Island (Brown Boobies, Lesser Frigatebirds, Red-Footed Boobies, Brown Noddies, and Sooty Terns) and four species have shifted to West Island. Of these four species, Black Noddies and Lesser Noddies were only breeding on West Island, Greater Frigatebirds were breeding on Middle and West islands, and Crested Terns were breeding on East and West islands. Six of these species are Listed Migratory species under the Japan-Australia Migratory Bird Agreement and the China Australia Migratory Bird Agreement and all are listed marine species under the EPBC Act. Furthermore, Ashmore Reef Marine Park is an IBA for Brown Boobies and Lesser Frigatebirds. The current Recreational Use Zone (IUCN IV) of West Island allows island access by recreational users to a small area of the island. Therefore, it is recommended that management arrangements around use and island access be considered to ensure impacts on breeding seabirds is minimised.

5 ASHMORE REEF: TROPICAL FIRE ANT (*Solenopsis geminata*)

Ben Hoffmann and Magen Pettit

5.1 Abstract

This study provides the latest assessments of tropical fire ant (*Solenopsis geminata)* status on Ashmore Reef, with the work conducted comparable with the previous surveys conducted by Hodgson and Clarke (2014). Tropical fire ant distribution and abundance was assessed.

Opportunistic observations of nesting seabird eggs and chicks were made for any signs of interference by *S. geminata*. Specifically, we looked for dead chicks in nests, blindness, sting marks on bare skin (feet of adults, bodies of chicks), holes in the webbing of adult feet, and ants crawling over the bodies of birds or eggs. In addition, turtle nesting areas were examined for any signs of interference by *S.geminata*, including dead turtle hatchlings and dead hatchlings clustered at a nest site.

East, Middle and West islands displayed different patterns of *S. geminata* distribution and abundance. West Island had the lowest ant attendance at lures and stations, with ants predominantly occurring just on the island's circumference. Middle Island displayed a gradient from no ants on the relatively bare eastern end, to high abundance on its north-west end. The ant was most prolific on East island, being present at 90% of stations throughout the whole island. Its abundance on the three largest islands largely reflected the abundance levels found by Hodgson and Clarke (2014). Notably, *S. geminata* was quite abundant on Middle Island which had undergone chemical treatments to kill the ant in 2013, demonstrating that there is no long-term suppression of the ant in the absence of eradication. No instances were found of interference of *S. geminata* with any birds or turtles at Ashmore Reef. Explanations of the lack of interaction at our time of observation is purely speculative, but potentially there was a difference in total food resources available to the ants at the time of survey (May) compared to the times when negative interactions had been reported previously (March, September, November). Our survey was conducted at the end of the wet season when the grasses had recently dropped their seeds, and when *S. geminata* populations would be expected to be undergoing a seasonal decline. It is also possible that there are strong annual differences driven by the annual climatic conditions that would influence food availability, ant populations and potential impacts. Ultimately it is clear that impacts by *S. geminata* are dynamic and not always readily observed.

5.2 Introduction

The tropical fire ant *Solenopsis geminata* has been known to be present on Ashmore Reef since 1992, by which time it was already present on both Middle and West islands, and reportedly widespread also on East island soon after (Brown 1999; Curran 2003). It remains unclear as to how the ant arrived on Ashmore Reef, but unpublished genetics indicate that the population is most closely related to populations in Australia than in other locations such as Indonesia. Regardless the ant was no doubt accidentally brought to Ashmore Reef by human mediation rather than self-dispersal. This species, native to Central America and southern North America (Wetterer 2011) is well documented to have adverse environmental impacts (Lowe *et al.* 2000), including killing hatchlings of giant tortoises and birds, and causing physical damage to soft flesh of many vertebrates (Tschinkel 2006).

The first quantitative survey of the ant on Ashmore Reef was conducted in September 2004 (Bellio *et al.* 2007) clearly showing that the ant was present throughout all three islands at the time (Splittgerber Cay did not exist until 2010), also finding that mortality of Common Noddy chicks was positively related to tropical fire ant abundance. Chance observations in 2008 and 2012 found dead sea turtle hatchlings associated with two failed nest emergence events, and mortality was attributed to tropical fire ants (Hodgson & Clarke 2014). Additional seabird research from 2011 to 2013 also found substantial physical damage to foot webbing on seabirds consistent with tropical fire ant damage (Hodgson & Clarke 2014). Clearly there is substantial indirect evidence that tropical fire ant poses a threat to the regionally and internationally significant biodiversity values of Ashmore Reef Marine Park.

A pilot eradication trial was conducted in 2011 after another full quantification of ant abundance over the islands (Hodgson & Clarke 2014), with encouraging results. But since then, additional seabird and turtle surveys have found the imperative to eradicate the ant to not be as urgent as first envisaged (Clarke & Herrod 2016; Guinea & Mason 2017). Eradicating tropical fire ants from Ashmore Reef remains desirable, but for now efforts will continue to improve understanding of tropical fire ant population dynamics and impacts on Ashmore's biota. The primary objective of this study was to provide the latest assessments of tropical fire ant status on Ashmore Reef, with the work to be conducted in such a way as to be comparable with the previous surveys conducted by (Hodgson & Clarke 2014).

5.3 Methods

Tropical fire ant distribution and abundance was assessed using the sampling grid and protocols described in Hodgson and Clarke (2014). Each island had 52 sampling locations (stations) generated on a grid 50 x 50 m on East and Middle islands and 75 x 75 m on West Island (SI Table 5). Ant abundance was quantified at each station using four lures, each consisting of a teaspoon sized amount of cat food placed on an 8 x 8 cm piece of paper placed on the ground at cardinal points (i.e. north, east, south, and west). Due to the great presence of nesting seabirds, the distance of the lures from the centroid of each station was reduced from 5 m (Hodgson & Clarke 2014) to 1 m. To avoid sampling interference individual lures were never placed within the reach of nesting seabirds. The number of ants attending each lure was counted/estimated after 50 minutes. Sampling was undertaken between 0600–0900 h and 1600–1800 h when temperatures were relatively cool and ant activity was not hindered. Additionally, we created four new stations on East Island so that the stations covered the entirety of the island.

Ant abundance was scored according to the following scale: 0=no ants, 1=1 ant, 2=2–5 ants, 3=6–10 ants, 4=11–20 ants, 5=21–50 ants, 6=51–100 ants, 7=>100 ants. The scores of the four lures at each station were summed and averaged to give a single value for each station. Actual abundance counts are provided as supplementary information to aid comparison with future work (SI Table. 6).

Opportunistic observations of nesting seabird eggs, and chicks were made for any signs of interference by *S. geminata*. Specifically, we looked for dead chicks in nests, blindness, sting marks on bare skin (feet of adults, bodies of chicks), holes in the webbing of adult feet, and ants crawling over the bodies of birds or eggs. In addition, turtle nesting areas were examined for any signs of interference by tropical fire ant, including dead turtle hatchlings and dead hatchlings clustered at a nest site.

5.4 Results

West, Middle and East islands displayed different patterns of *S. geminata* distribution and abundance. West Island had the lowest ant attendance at lures and stations, with ants predominantly occurring on the island's circumference (Figure 46). When *S. geminata* was present, it was typically highly abundant on only one or two lures per station. Middle Island displayed a gradient from no ants on the relatively bare eastern end, to high abundance on its north-west end (Figure 47) with the ant present on approximately half of the lures (51%) and at three quarters (73%) of stations. The ant was most prolific on East island, being present throughout the whole island (Figure 48), attending 64% of lures and 90% of stations, and having the greatest abundance at lures across the three islands surveyed (Table 21).

Table 21. Tropical fire ant (*Solenopsis geminata*) abundance across the 2019 Ashmore Reef island surveys, compared to prior surveys at the islands.

Island	**Attendance (%)**		**Mean scaled abundance**	
Our survey (May 2019)	Lure	Station	All	Presence only
East	64.0	89.5	3.87	5.91
Middle	51.0	73.1	2.67	4.99
West	18.8	28.8	1.06	5.37
Hodgson & Clarke (2014)				
East (April 7–8 2011)	71.6	94.2	3.56	4.97
East (May 2–5 2012)	71.2	94.2	3.81	5.35
Middle (April 5–9 2011)	59.6	84.6	3.07	5.15
West (June 21–25 2012)	17.3	46.2	0.63	3.67
West April 2–6 2013)	47.4	72.9	2.2	4.64

Solenopsis geminata abundance at Ashmore Reef largely reflected the abundance levels found by (Hodgson & Clarke 2014) for the three islands surveyed. The ant covers the entirety of West, Middle and East islands, but with varying levels of abundance among and within islands. Unfortunately, spatial information is not provided in Hodgson and Clarke (2014) in any form (visual or data), so no comparison can be made of spatial dynamics. Notably, *S. geminata* was quite abundant on Middle Island which had undergone chemical treatments to kill the ant in 2013, demonstrating that there is no long-term suppression of the ant in the absence of eradication.

No instances were found of interactions or interference of *S. geminata* with any Ashmore Reef biota. Indeed, no ants were observed walking over the body of adult or chick birds despite the ants and the birds very often being in very close proximity (Figure 49 and Figure 50). The surveys found no damage to biota, which was very surprising given the presence and high abundance of ants over most of the surface of the islands, often with nests and trails within the direct vicinity of nesting birds. This was specially so on East Island where both ant and bird densities were very high. Attempting to explain the lack of interaction during our snapshot observation timeframe would be purely speculative. It is possible that there are strong differences driven by seasonal and annual climatic conditions that would influence food availability, ant populations and potential impacts. A prior study that also recently found no observable impacts of fire ants on turtles also speculated that this may have been due to the “bumper” sea-bird breeding season providing so much resources to the ants that it reduced the need for ants to invade sea turtle nests (Guinea & Mason 2017). Ultimately, there is little to no understanding of these interactions and outcomes for all non-native ant species, but it is clear that impacts by *S. geminata* are dynamic and not always readily observed.

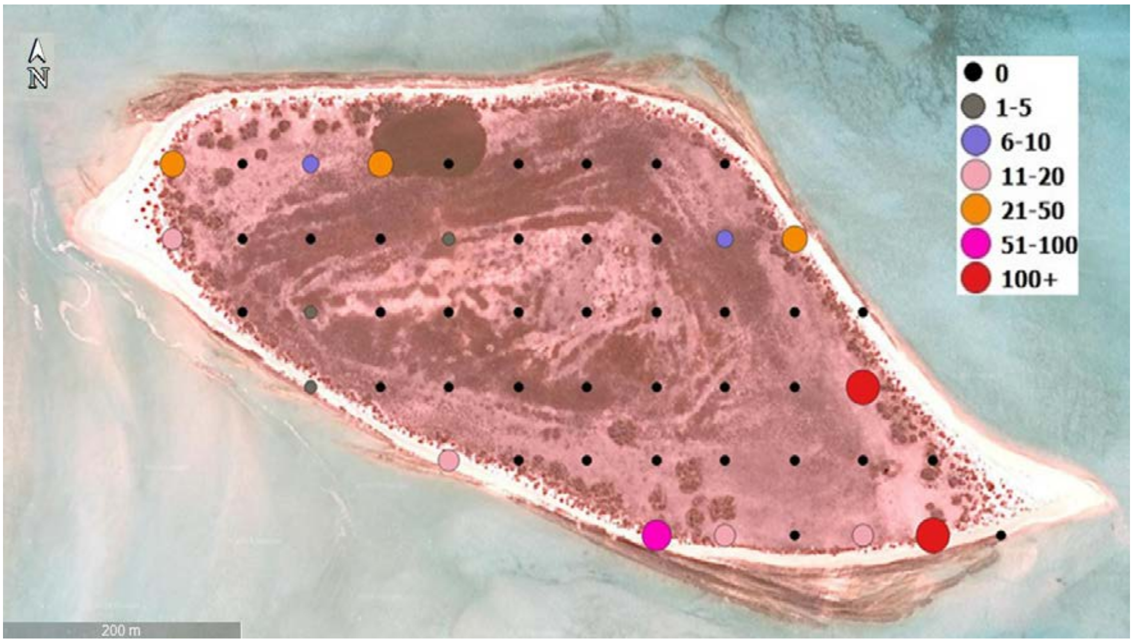


Figure 46. ***Solenopsis geminata*** **abundance on West Island. Data are the average scored abundance of ants at four lures per location, displayed as actual ant abundance ranges**

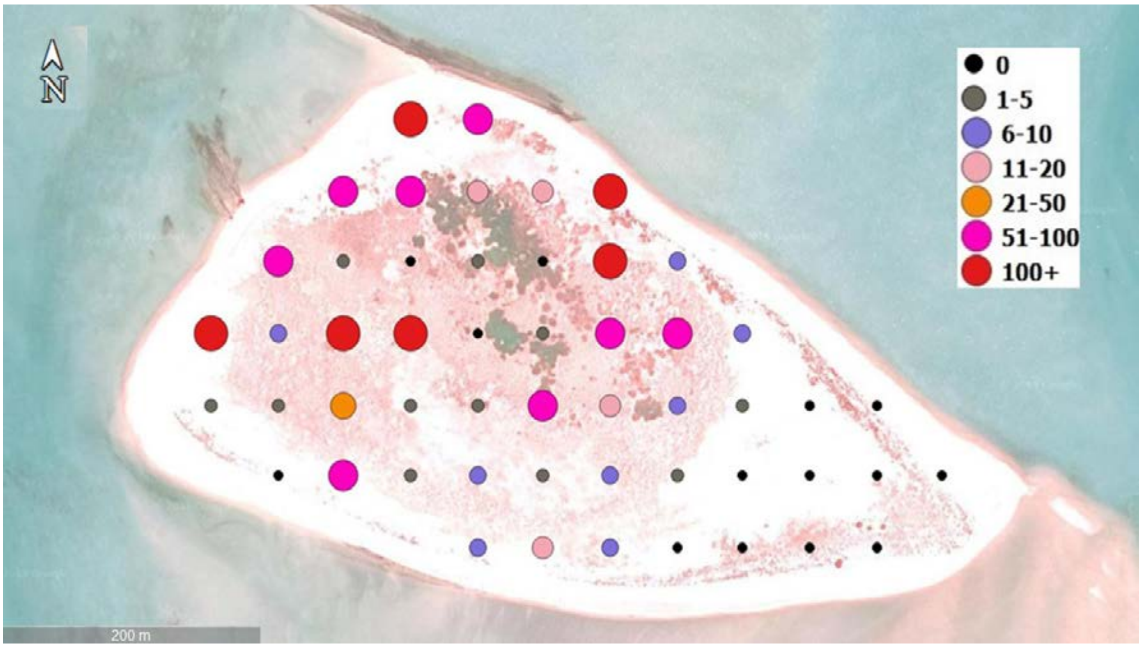


Figure 47. ***Solenopsis geminata*** **abundance on Middle Island. Data are the average scored abundance of ants at four lures per location, displayed as actual ant abundance ranges.**

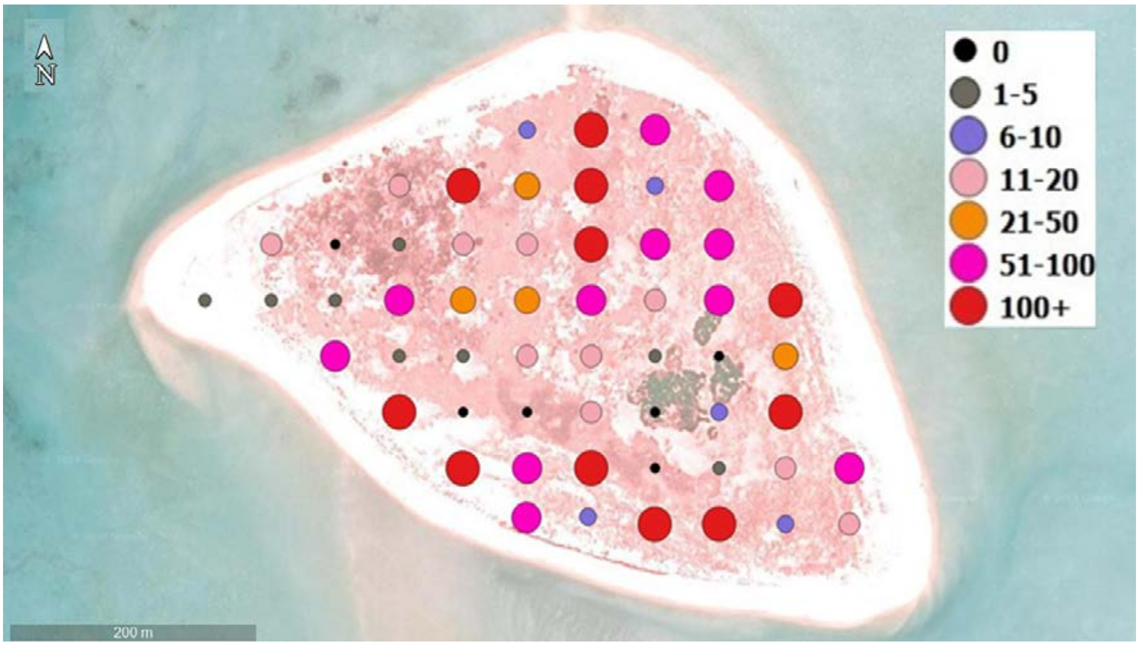


Figure 48. ***Solenopsis geminata*** **abundance on East Island. Data are the average scored abundance of ants at four lures per location, displayed as actual ant abundance ranges**

Figure 49. A brown booby (*Sula leucogaster*) nesting within one metre of a long nest/walkway of *Solenopsis geminata*.

Figure 50. A Brown Booby chick beside a dead Brown Noddy that was covered with *S. geminata.*

5.5 Discussion

5.5.1 *Solenopsis geminata* abundance

Solenopsis geminata abundance on West, Middle and East islands largely reflected the abundance levels found by (Hodgson & Clarke 2014) for these three islands, being also not too indifferent from that found by (Bellio *et al.* 2007). The ant covers the entirety of all three islands, but with varying levels of abundance among and within islands. Unfortunately, spatial information is not provided in (Hodgson & Clarke 2014), so no comparison can be made of spatial dynamics. Notably, *S. geminata* was quite abundant on Middle island which had undergone chemical treatments to kill the ant in 2013, demonstrating that there is no long-term suppression of the ant in the absence of eradication.

5.5.2 Impacts of *S. geminata*

(Hodgson & Clarke 2014) monitored the nesting success of three bird species on two islands, one island having been treated for *S. geminata* and the other untreated. They found that nesting success was greater for two Booby species on the island where treatments for *S. geminata* had been conducted, but no difference was found for nesting success of the Lesser Frigatebird. The report also detailed damage to the webbing of some red-tailed tropicbirds and sub-adult brown boobies that was attributed to *S. geminata.* The report also described two instances of direct *S. geminata* attack on turtle hatchlings that had been observed by another researcher. Nesting success could not be quantified in the “snapshot” survey we undertook, but we were expecting to observe some interactions between birds and the ants, as well as damage to some birds as a result of long-term interactions with ants. That no interactions or damage were found was surprising, especially on East Island where both ant and bird densities were very high. However, the lack of interference is consistent with other recent seabird and turtle surveys that also did not find impacts, especially to seabird populations (Clarke & Herrod 2016; Guinea & Mason 2017).

Attempting to explain the lack of interaction at our time of observation is purely speculative. Notably, the ants were readily attracted to the cat food lures, so they were not averse to consuming protein. Potentially there was a difference in total food resources available to the ants at the time of survey (May) compared to the times when negative interactions had been reported previously (March, September, November). Our survey was conducted at the end of the wet season when the grasses had recently dropped their seeds, and when *S. geminata* populations would be expected to be undergoing a seasonal decline. Given that *S. geminata* is predominantly a seed-harvesting species, it is possible that the ant had full granaries and was satiated with nutritional requirements. In comparison, the other times were either prior to seed drop with very high population levels, or at the beginning of the time of seasonal population increase coupled with low food storage levels. It is also possible that there are strong annual differences driven by the annual climatic conditions that would influence food availability, ant populations and potential impacts. Ultimately, there is little to no understanding of these interactions and outcomes for all non-native ant species, but it is clear that impacts by S. geminata are dynamic and not always readily observed.

5.5.3 Potential and need for control of *S. geminata*

Given that *S. geminata* is a non-native invasive species of global significance and with impacts that are well documented, and such impacts have been observed at Ashmore previously on birds and turtles, the mere continued existence of this ant on the islands should be of great conservation concern, and therefore the need to control or eradicate this species from Ashmore Reef remains. Plans to eradicate the ant should continue to be advanced.

5.6 Management implications and recommendations

This work combined with that of all others has shown that *S. geminata* has varying abundances seasonally and throughout the islands of Ashmore Reef, and that the populations on West, Middle and East islands are persistent and widespread leaving very little land area without its influence. Likewise, its impacts on native biota also vary greatly, both among species, and throughout time. Results found here support the notion that although the need to manage or eradicate the ant remains, the imperative to do so is not as urgent as first thought. This will give the opportunity for science globally to provide new baiting technology needed to conduct such management/eradication work on Ashmore Reef with minimal non-target impacts relative to current technology.

The most promising of the horizon technology is RNA interference, whereby highly targeted “toxic genetics” replace the use of general insecticides as the active constituents in baits. The “toxic genetics” interfere with some specific coding of the target species’ genes which ultimately results in the death of the target species, but without affecting non-target species. We recommend that Parks Australia either wait for, or help support, the global initiative to develop the RNA interference technology proposed to treat and eradicate *S. geminata* from Ashmore Reef.

Should there be a desire to attempt to understand the dynamics of *S. geminata* impacts, a study would need to be conducted that regularly measures numerous variables simultaneously for at least two years, namely ant populations, distribution, impacts (e.g. chick mortality), grass seed supply, and other potential food supply (all protein available to ants from nesting/roosting birds). These data would need to be coupled with stable isotope analysis of the ants to determine if their nutrient uptake is related to the availability of surrounding resources and seasonal environmental conditions. Note though that this knowledge would realistically have no effect on any potential eradication plan.

6 ASHMORE REEF: INTRODUCED GECKOS

Ruchira Somaweera, Paul B. Yeoh, Tommaso Jucker and Bruce L. Webber

6.1 Abstract

Introduced to numerous countries and oceanic islands around the world, the Asian house gecko (*Hemidactylus frenatus*) shows the largest non-native distribution of any gekkonid in the world. Reports suggest that it has been naturalised at Ashmore Reef since the 1990s. Current survey confirms that it is restricted to the West Island and abundant on the octopus bushes (*Heliotropium foertherianum*) above the shoreline. Two nights of sampling recorded 89 individuals (23 juveniles and 66 adults) and 5 eggs on 26 of the 35 *H. foertherianum* shrubs examined at West Island. The potential pathway to impact by *H. frenatus* on the terrestrial invertebrate fauna by direct predation requires further investigation.

6.2 Introduction

The Asian house gecko (*Hemidactylus frenatus*) (Figure 51) has undergone multiple independent introductions to the Australian mainland via ship cargo from South-East Asia since the 1930s, and has now spread extensively across northern and eastern Australia (Hoskin 2011). However, the known populations are mostly centred on urban areas and isolated settlements. Limited information exists on the establishment and status of this species on the offshore islands surrounding Australia. It is known to have established at Cocos (Keeling) Islands since 1930s (Cogger, Sadlier & Cameron 1983), and Christmas Island by 1940 (Smith *et al.* 2012), but no data are available for most smaller islands.

The only confirmed records of a terrestrial reptile species on the Ashmore Islands is of Asian house geckos (Horner 2005). Storr, Smith and Johnstone (1990) first reported the occurrence of this species at Ashmore Reef, but no further information was provided. During a two-week entomological survey at Ashmore Reef in May 1995, Brown (1999) reported this species to be rare, with only a single individual caught in a malaise trap. He did not find any specimens among the ground litter. However, during a 16-h daytime survey in March 2001, Horner (2005), found this species to be abundant in all habitat types on West Island, but absent from the two other islands. Based on this evaluation, it was considered to be 'well established' on West Island (Hale & Butcher 2013).

This survey aimed to gain an updated assessment of the population condition and distribution of *H. frenatus* at Ashmore Reef and to confirm the identity of any other geckos found.

Figure 51. Showing the Asian house gecko (*Hemidactylus frenatus*) Credit: Ruchira Somaweera

6.3 Methods

Field surveys took place in May 2019. To confirm the presence of geckos we conducted targeted visual and auditory encounter surveys during day time at selected locations at East, Middle and West Islands and the sand cay next to East island (Figure 52 – red dots). Although it is a crepuscular species, individuals still call and could be active during day time (Marcellini 1974; Somaweera, pers. obs.). Possible daytime retreats such as under bark on live and dead trees, leaflitter under trees, under and among driftwood on the beaches, and any larger rocky boulders on land were searched for geckos by 1-2 persons during day time (0900–1100 h: 1 h at East Island on 3 May, 1 h at Middle Island on 3 May and 3.5 h cumulatively at West Island on 4 and 6 May) and by three people for 1.5 h at night time at West Island on 6 May. Once caught, we recorded the snout-vent length (SVL) and sex of the individuals (based on presence of hemipenal bulges and preanal pores in males) in and obtained tail tips from five individuals for future genetic analyses. We did not conduct trapping as the non-native tropical fire ants present on the island could have attacked the geckos caught in traps.

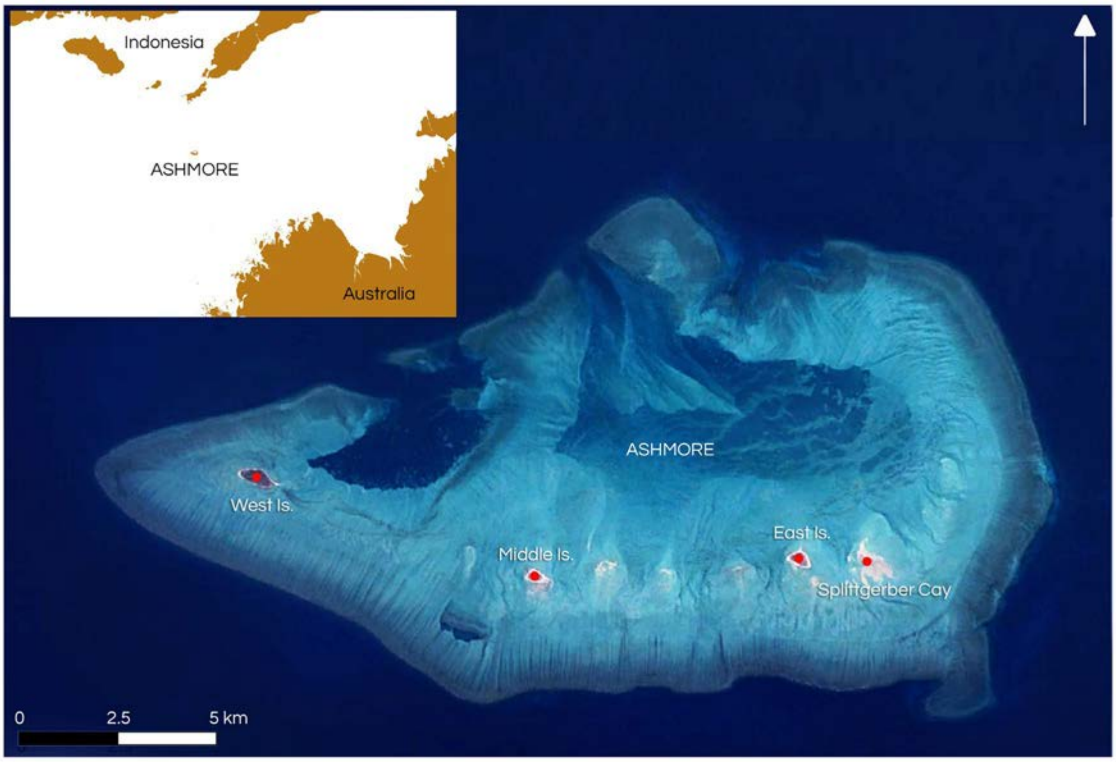


Figure 52. Location of Ashmore (insert) and islands surveyed for the *Hemidactylus frenatus* (red dots) at Ashmore Reef Marine Park in May 2019.

6.4 Results

No geckos or suitable habitats to support gecko populations were recorded at Middle or East Islands or the sand cays. The grass-dominant vegetation communities in those two islands do not provide any suitable cover and places to retreat for the geckos. At West Island, geckos were only observed on *H. foertherianum* shrubs along the outer edge of the island, hiding under bark during daytime but active on the branches and stem during night-time. No geckos were observed among leaf litter, under logs or among driftwood and rocky boulders.

Geckos were observed at 26 of the 35 *H. foertherianum* shrubs sampled at West Island on 4 and 6 May (Figure 53 - trees with geckos are shown in yellow). The 89 specimens observed comprised 23 juveniles (<40 mm SVL) and the rest adults. The 14 adults hand captured consisted of five males and nine females, none of which were gravid. The SVL of the hand captured animals ranged from 46 to 67 mm.

Five eggs were observed under bark in *H. foertherianum* shrubs, as a cluster of 4 in one plant and a single egg in another.

Figure 53. Showing *H. foertherianum* bushes sampled at West Island on 4 and 6 May (shrubs with geckos are shown in yellow)

6.5 Discussion

The exact time and the introduction of Asian house geckos to Ashmore is not known, but based on (Storr, Smith & Johnstone 1990), they were present on the island by the 1990s. Since at least the late 19th century, Indonesian fishers have been known to regularly and frequently visit the Ashmore Islands (Clark 2000). These visits by Indonesian vessels have continued after the World War II. During this period, the islands and the sand cays have been well occupied by the fishers as they cooked and dried harvested fish, clams and seabirds (Serventy 1952a). Therefore, it is possible that stowaway geckos may have been accidentally introduced to Ashmore via these fishers. More recently, merchant ships, commercial fishing vessels from Australia and Indonesia, cruising yachts and charter vessels, as well as government patrol vessels have also accessed the reef and islands at different time periods, with the likelihood of introducing additional individuals to the population.

Although generally considered a species restricted to anthropogenic habitats across much of its native range, *H. frenatus* also inhabits trees and rocks in natural habitats close to humans (Newbery & Jones 2007; Mckay, Griffiths & Crase 2009). At Ashmore Reef, Horner (2005) noted that *H. frenatus* uses any suitable shelter sites at West Island including ground litter, palm fronds, standing and fallen timber and coral rubble. During our visits, we only observed geckos on dead and live *Heliotropium foertherianum* trees, but not among the leaf litter or rubble. The coconut trees (*Cocos nucifera*) mentioned by Horner (2005) are all dead and no fronds were observed. Horner (2005) further mentioned that multiple specimens were usually present when ground litter was moved. We did not observe any specimens among leaf litter during our surveys. A possible explanation is that the non-native tropical fire ants (*Solenopsis geminata*) that established on the island, may prevent geckos occupying ground litter. However, this needs further study and verification.

No population estimates were possible with the data in-hand given the opportunistic nature of our study, but the species is common on suitable shelter sites on *Heliotropium foertherianum* trees along the shoreline of the island. The presence of eggs and several hatchlings and juveniles suggest population recruitment is taking place. The absence of geckos on the Middle and East Islands is not surprising as the vegetation communities in these islands do not have any suitable shelter sites and they are also inhabited by large colonies of seabirds (Clarke *et al.* 2011).

Asian house geckos have been introduced to multiple regions in the Americas, Australia, Oceania and Africa outside their native range (Uetz, Freed & Hošek 2019), but known instances of them causing ecological impacts are far and few (Lever 2003). In fact, Vanderduys and Kutt (2013) argues that *H. frenatus* is a benign threat and is unlikely to become an environmental pest within Australia. However, among the known impacts, the competitive exclusion of sympatric geckos is the most well recognised threat by *H. frenatus* (Rödder, Solé & Böhme 2008; Hoskin 2011). But, given no other gecko or reptile species inhabit Ashmore system, competitive exclusion is not a relevant threat in this system. Accordingly transmission of parasites from geckos to native species (Barton 2007) is also not a possible threat.

A potential pathway of impact of *H. frenatus* at Ashmore Reef is the direct predation of its invertebrate fauna. Like most geckos, *H. frenatus* it is a generalist predator with high efficiency of foraging (Frankenberg & Werner 1981; Lei & Booth 2014). Arguably, the geckos impose an additional predation pressure on the terrestrial invertebrate communities (Russell, Neil & Hilliard

2004) that are already living under harsh environment conditions with a substantial number of avian predators. Terrestrial invertebrates recorded at the Ashmore Islands include 149 species of insects, seven species of spiders, a single species each of centipede, pseudoscorpion and millipede (Pike 1992; Brown 1999). Across Pacific Islands, the invasion of natural habitats by Asian house geckos have been attributed to variation in insect abundance (Petren & Case 1998). While no evaluation of insect abundance across vegetation types at the islands is available, opportunistic field observations during this study suggest that insects are abundant within the vegetation communities inhabited by geckos. Moreover, at Ashmore, it is possible that the distribution of geckos is governed largely by availability of shelter sites rather than the abundance of insects.

6.6 Management implications and recommendations

There is a need to understand the ecological impacts (e.g. predation pressure on invertebrates) of the geckos at Ashmore Reef Marine Park. Asian house geckos have been introduced to multiple regions around the world but known instances of the species causing ecological impacts are limited (Lever 2003). Competitive exclusion of sympatric geckos (e.g. Dame & Petren 2006) and transmission of parasites to native species (e.g. Barton 2007) are the studied impacts of this species, but they are not relevant to Ashmore as it is the only reptile species on the islands. Predation pressure on invertebrates is the likely ecological impact of Asian house geckos at Ashmore, but data to evaluate the level of impact does not exist. Analysis of stomach contents of the geckos and evaluation of the relationship between insect and gecko abundance would be needed. This work would be a pre-requisite to any consideration of control. Further details are discussed in the following publication resulting from the current survey:

Somaweera, R., Yeoh, P. B., Jucker, T., Clarke, R. H., & Webber, B. L. (2020). Historical context, current status and management priorities for introduced Asian house geckos at Ashmore Reef, north-western Australia. *BioInvasions Records*, 9(2), 408-420. DOI: 10.3391/bir.2020.9.2.27

7 ASHMORE REEF: HERMIT CRABS

Ben Hoffmann and Magen Pettit

7.1 Abstract

The hermit crab populations at Ashmore Reef are the most susceptible non-target fauna for any proposed tropical fire ant eradication program because they will readily consume the bait and as invertebrates they are also highly susceptible to the active constituents used against ants. Pilot-scale assessments of hermit crab abundance used to date have failed to provide useful data. Here we use a simple method to quantify hermit crab abundance that will enable meaningful comparisons with subsequent surveys for both general population assessments, and pre- and post- any eradication attempt of tropical fire ant. Hermit crab abundance was quantified by counting the number of crabs found within one minute within 5 x 1 m belt transects placed haphazardly along the high tide mark. Thirty-three transects were used on West Island, 32 transects on Middle Island, and 30 transects on East Island. Hermit crabs were present on West, Middle and East islands, being most abundant on West Island (4.9 ± 4.2 mean ± SD) crabs per transect), far more so than on East and Middle Islands (0.6 ± 1.1) and (0.4 ± 0.7) crabs per transect, respectively. The quick and simple survey technique used was able to provide meaningful and useful abundance data, demonstrating its viability for use in the future to monitor hermit crab populations.

7.2 Introduction

The hermit crab populations at Ashmore Reef are the most susceptible non-target fauna for any proposed tropical fire ant eradication program because they will readily consume the bait and as invertebrates they are also highly susceptible to the active constituents used against ants. Like all other terrestrial invertebrates, the baseline abundance levels of hermit crabs are poorly known, but must be adequately quantified to allow any ant management to be conducted (Hodgson & Clarke 2014). In June 2012, an attempt was made to count hermit crabs within 2 x 2 m quadrats, but the assessment failed to provide useful data due to the low number of crabs detected (Hodgson & Clarke 2014). In October 2012, a mark-recapture approach was used on West Island that was more effective and used to calculate a crab population of 24,460 (Hodgson & Clarke 2014). However, this work found hermit crab abundance was variable making population estimates uncertain. The objective of this work was to implement an effective method to determine hermit crab densities on the islands of Ashmore Reef that would enable meaningful comparisons with subsequent surveys pre- and post- any eradication attempt of tropical fire ant.

7.3 Methods

Hermit crab abundance was quantified by counting the number of crabs found along 5 x 1 m belt transects placed along the high tide mark (Figure 54) within a one-minute time period. Counts were conducted during the late afternoon (after 1630 h) or early morning (before 0730 h) to keep environmental conditions consistent, and with the assumption that there could possibly be reduced hermit crab activity during the warmer parts of the day. Transects were established haphazardly and positioned at least 1 m apart. Transect establishment merely involved dragging the tape along the high-tide mark until the desired position (farther along the high-tide mark from the prior transect) was reached. No regard needed to be, or was given, to hermit crab activity. Following the placement of the tape, two people stood approximately one meter back from the tape on the higher ground, equidistant along the tape, and observed for hermit crabs on the other side of the tape extending out 1m for one minute. At the end of the minute, the observation area was also quickly checked for any hermit crabs that may have been missed. Thirty-three transects were used on West Island, 32 transects on Middle Island, and 30 transects on East Island.

Figure 54. A 5 x 1 m belt transect placed along the high tide mark used to count hermit crabs.

7.4 Results

Hermit crabs were present on West, Middle and East islands, being more abundant on West Island (4.9 ± 4.2 mean ± SD) crabs per transect ($5m^2$), than on East and Middle islands (0.6 ± 1.1) and (0.4 ± 0.7) crabs per transect, respectively.

7.5 Discussion

Hermit crabs were present on West, Middle and East islands and with abundance varying greatly among the islands, so no single island can be used as a proxy for hermit crab abundance on the other islands. Importantly the quick and simple survey technique used was able to provide meaningful and useful abundance data, demonstrating its viability for use to monitor hermit crab populations should any treatments be conducted for *S. geminata* in the future. Although crabs were noticed to be present over the entirety of islands, often in higher abundance than where the transects were placed (Figure 55 and Figure 56) using the high tide mark for surveys allows for consistency and non-bias among islands for the survey habitat. Notably, surveys did not need to be conducted at night, as the crabs were visibly foraging at all hours, especially mornings and late afternoons.

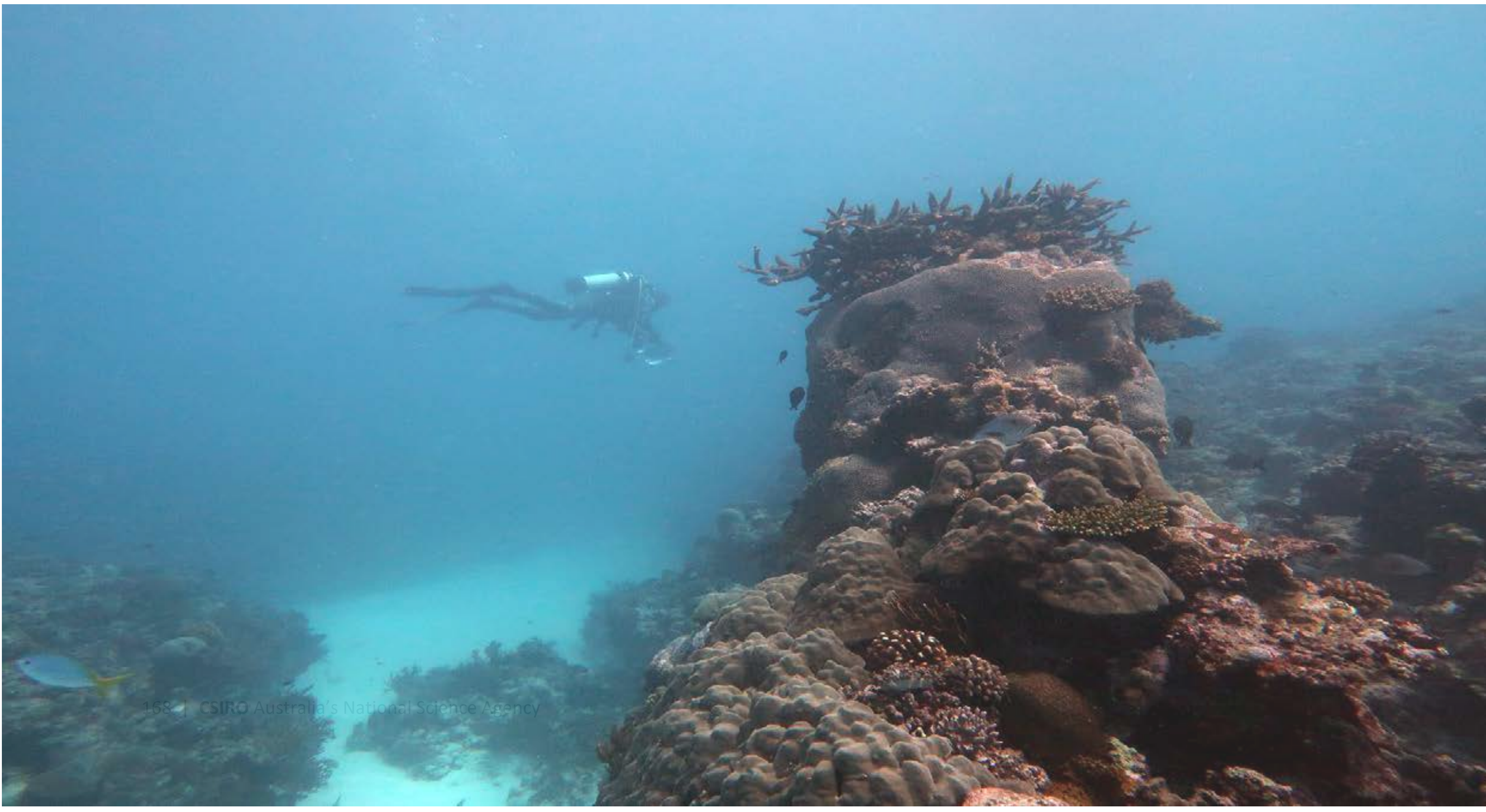
Figure 55. Showing a large congregation of hermit crabs.

Figure 56. Hermit crabs foraging on West Island beach in a relatively small area between the high tide mark and the current tide height.

7.6 Management implications and recommendations

This work established a simple, efficient and effective method of determining hermit crab abundances that can be used in future assessments of the Ashmore Reef hermit crab populations, and especially before and after any attempt to manage or eradicate tropical fire ant from Ashmore Reef Marine Park.

PART III MARINE SURVEYS



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

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

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 maybe 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^{o} 15'S, 123^{o} 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, Cappo & 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).

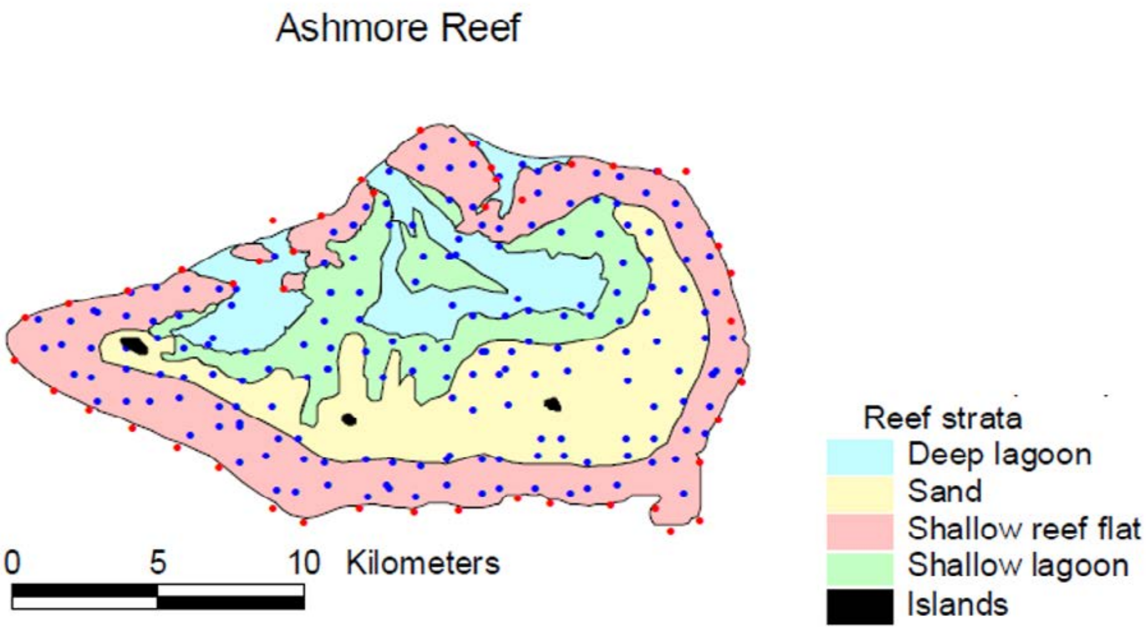


Figure 57. Distribution of habitats at Ashmore Reef adapted from Skewes et al. (1999b) and Hosack and Lawrence (2013). In addition to the habitats shown, a fifth marine habitat (reef edge: red dots) was added to the 2013 (Ceccarelli et al. 2013) and 2019 surveys.

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.

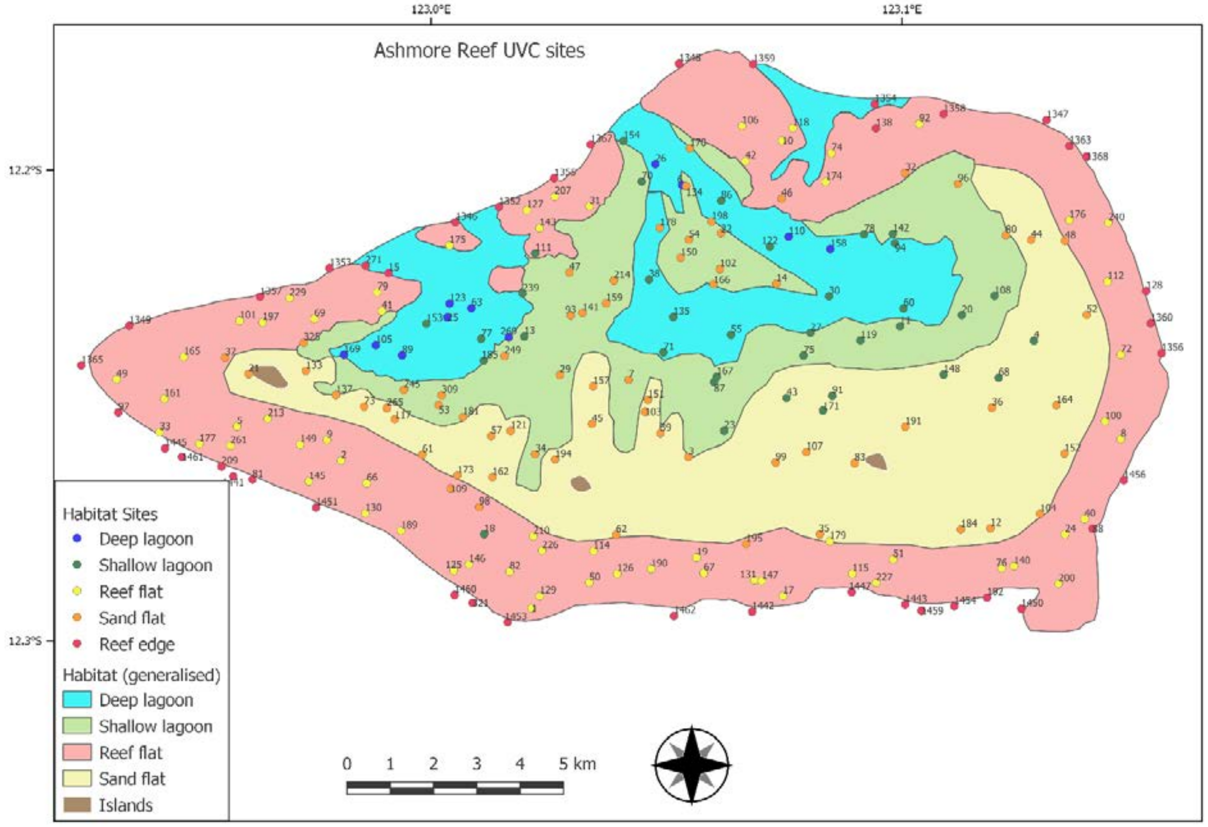


Figure 58. Location of 216 UVC sites surveyed during 2019 Ashmore Reef surveys.

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). 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.

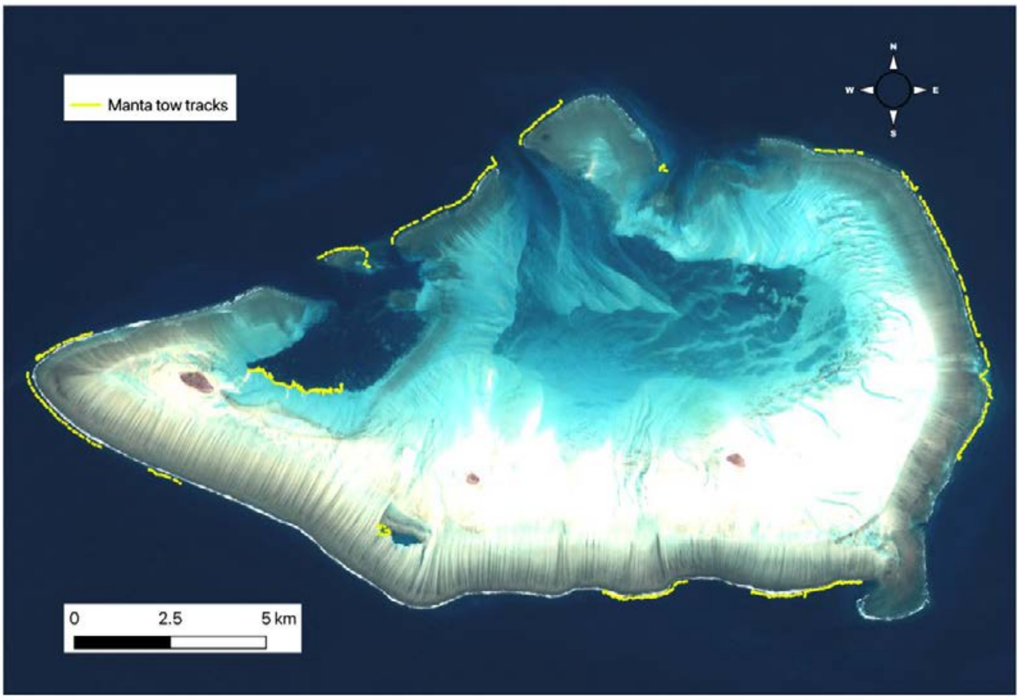


Figure 59. Manta tow tracks (yellow) undertaken during Ashmore Reef during 2019 surveys.

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

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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% ± 2 s.e., soft coral 8.2% ± 2.5) and sponge cover (1% ± 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% ± 0.6) and approximately 4% of total benthic assemblages (3.9% ± 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 overlayed on each image (3 x 2 grid) and taxa directly underneath identified using Transect-MeasureTM (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-MeasureTM (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% ± 2 s.e.), soft coral (8.2% ± 2.5) and sponge cover (1% ± 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% ± 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% ± 2.1 and 64.3% ± 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% ± 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% ± 3.6 s.e.), reef edge (48.3% ± 2.2) and deep lagoon (41.2% ± 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% ± 0.6). Macroalgal cover was greatest on the reef edge (2.7% ± 0.8) and within the shallow lagoon (2.6% ± 0.7) with 20-25% cover observed at some sites Figure 62).

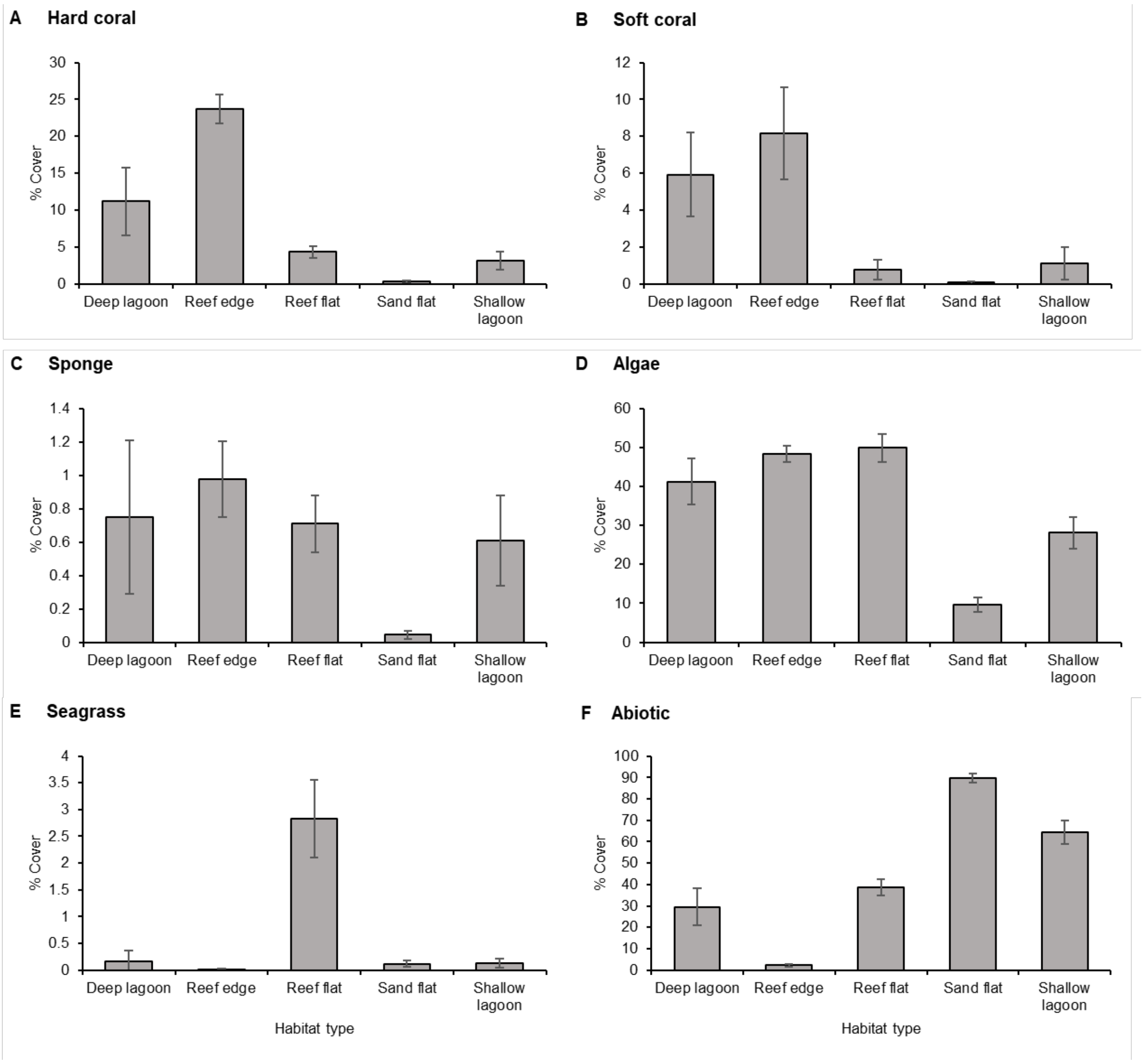
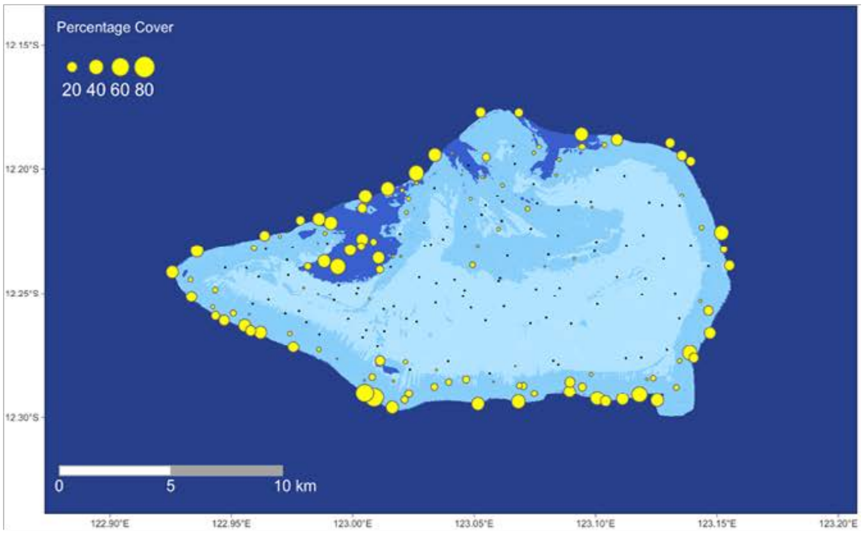
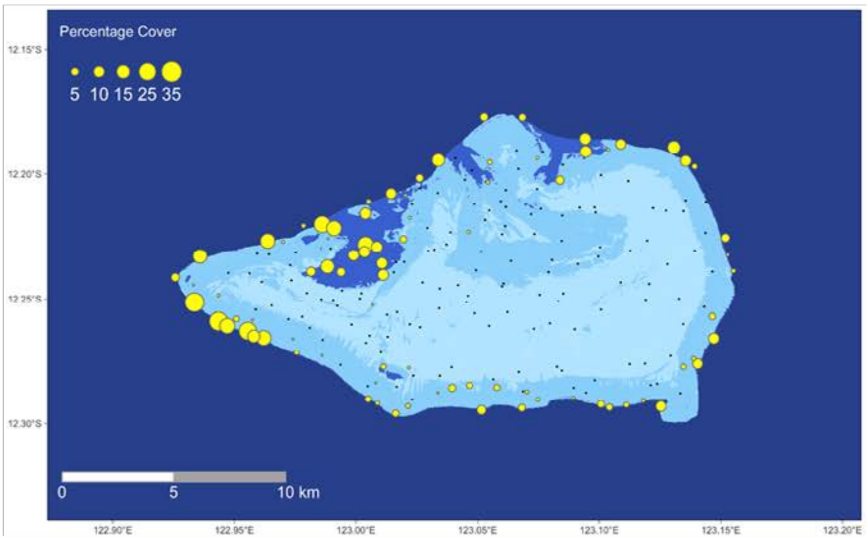


Figure 60. Percent cover of major benthic assemblages across the five habitat types at Ashmore Reef, 2019. Error bars indicate ± 1 s.e.

A Hard coral



B Soft coral



C Sponge

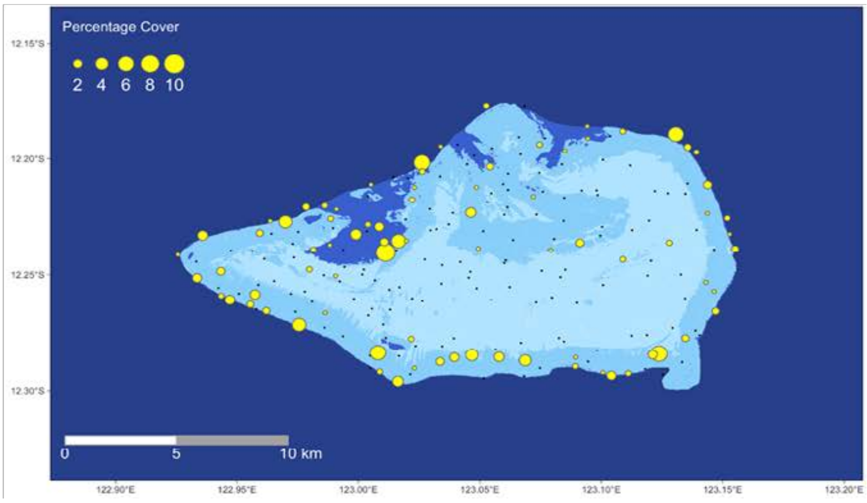
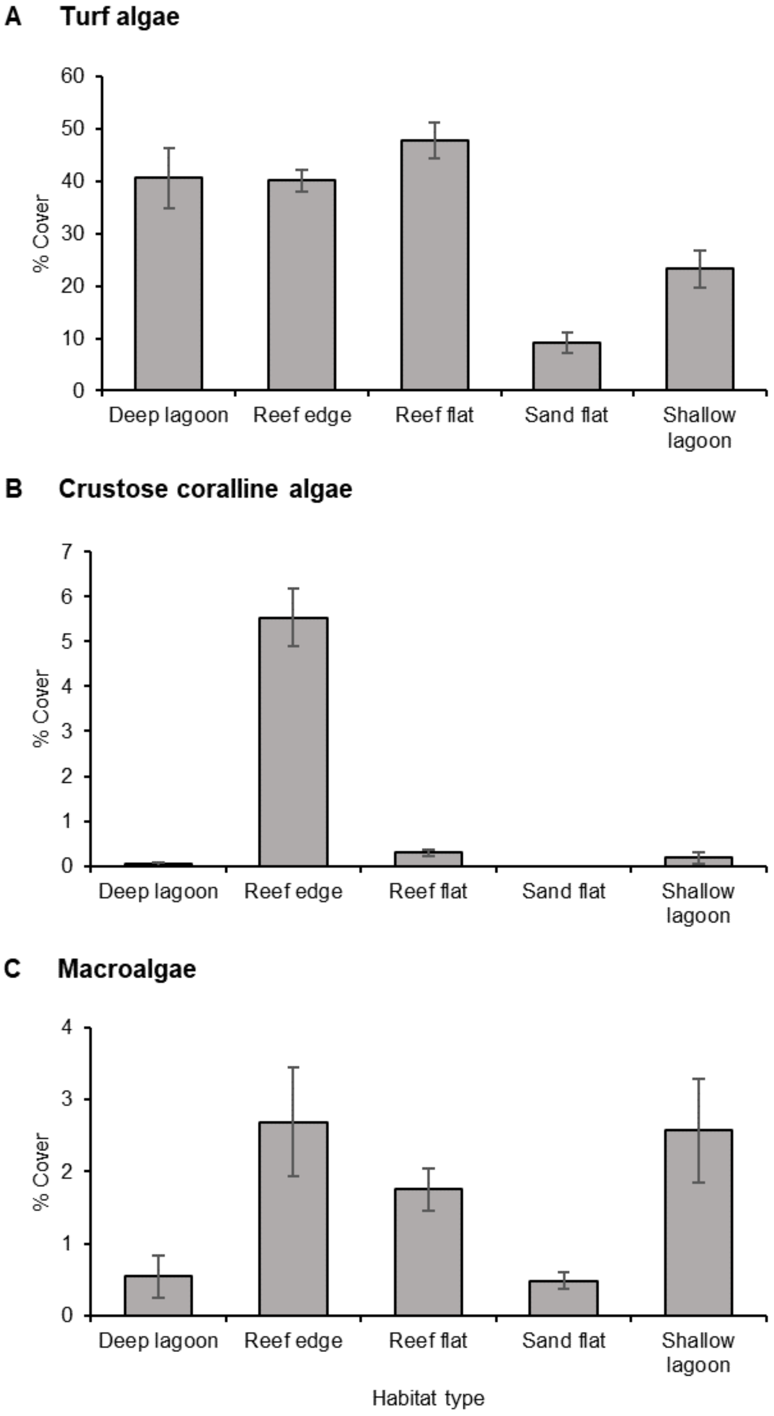


Figure 61. Percent cover of A) hard coral, B) soft coral, and C) sponges across the 216 sites surveyed at Ashmore Reef in 2019


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 ± 1 s.e.

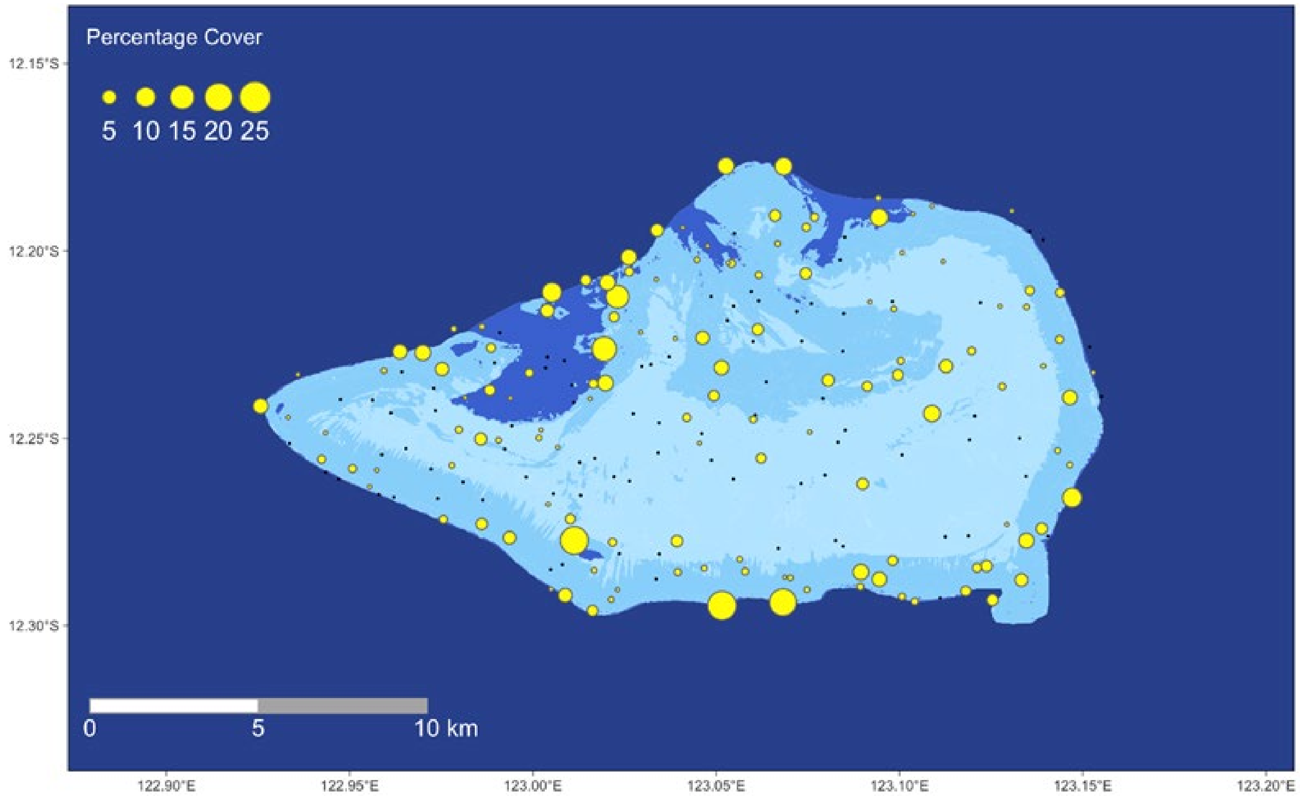


Figure 63. Percent cover of macroalgae across the 216 sites surveyed at Ashmore Reef in 2019.

Acropora, *Pocillopora, Porites*, and Faviidae corals dominated the scleractinian coral community. *Acropora* spp. accounted for half of all live hard coral cover (50.1% ± 0.6 s.e.) and approximately 4% (3.9% ± 0.6) of total benthic assemblages. *Porites* spp. comprised almost one quarter of live coral cover (22.1% ± 0.2) and 1.7% (± 0.2) of all benthic assemblages. Both *Pocillopora* and Faviidae spp. each accounted for approximately 10% (10.5% ± 0.1 and 11% ± 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% ± 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% ± 0.23 and 6.9% ± 0.4). Hard coral cover in reef flats was dominated by *Porites* (7.5% ± 0.4).

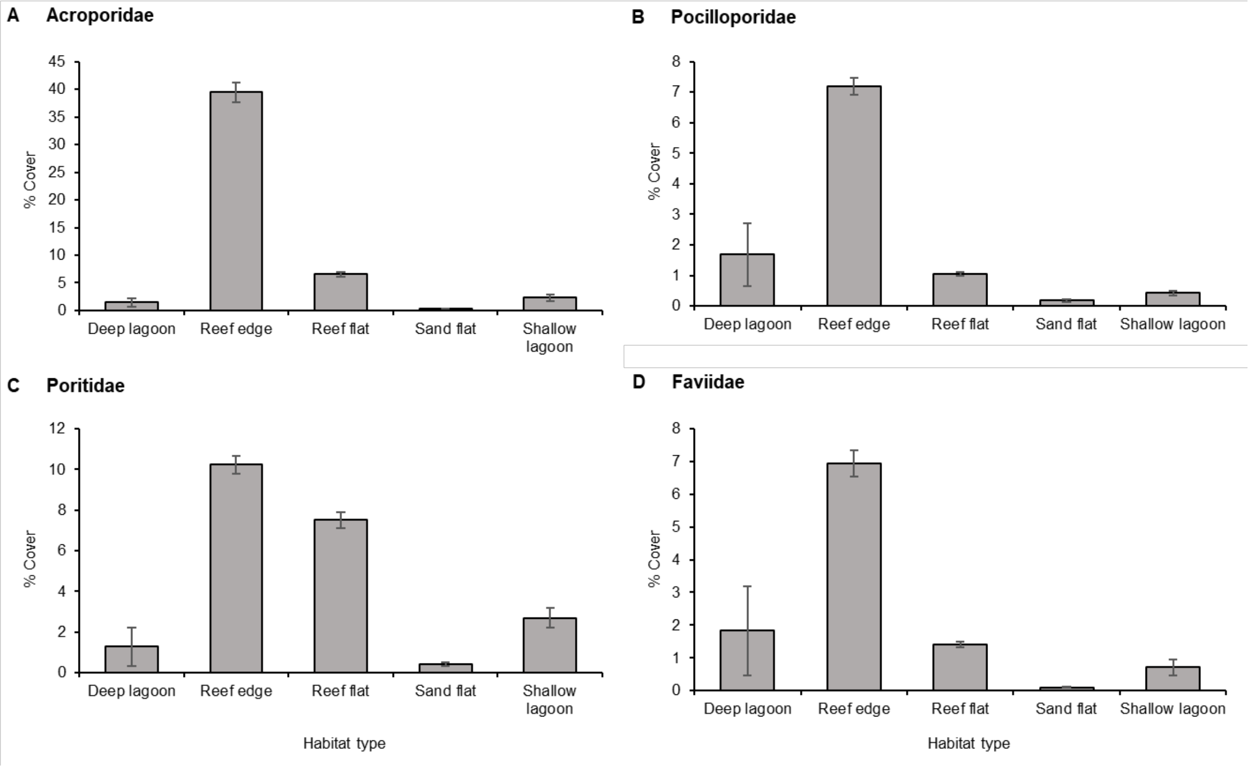


Figure 64. Percent cover of main hard coral genera contributing to overall hard coral cover across the five habitat types at Ashmore Reef: A) Acroporidae, B) Pocilloporidae, C) Poritidae, and D) Faviidae. Error bars indicate ± 1 s.e.

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.

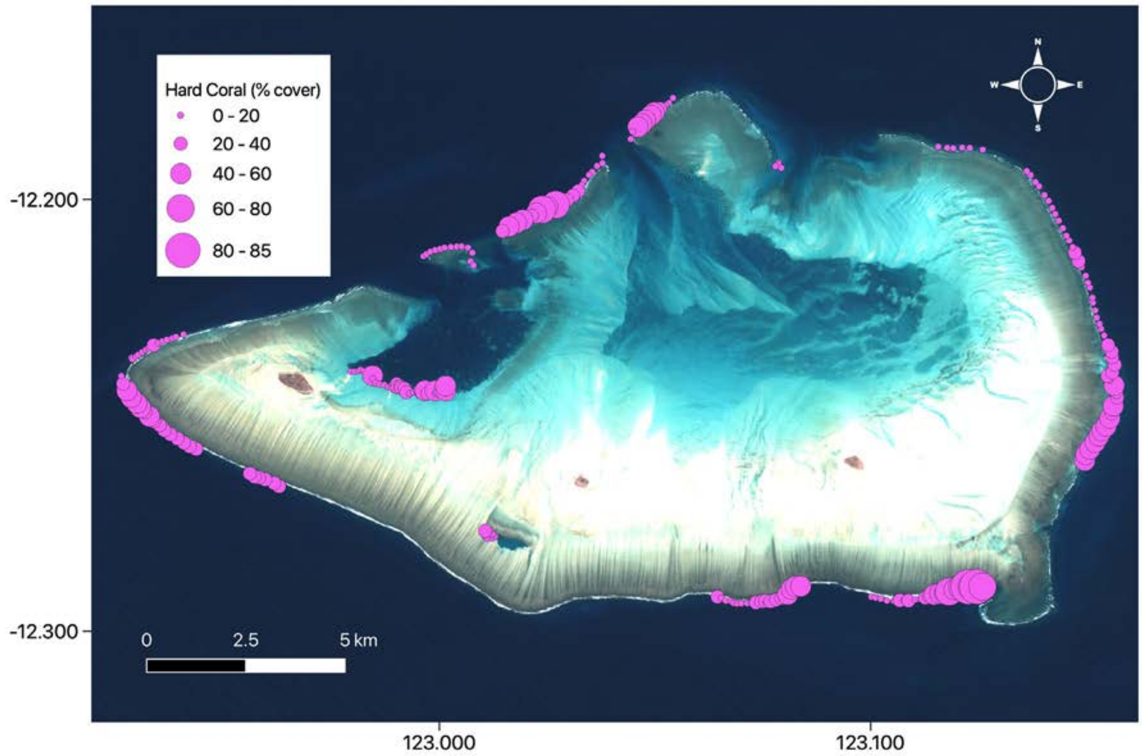


Figure 65. Estimates of percent cover of hard corals from manta tow transects at Ashmore Reef

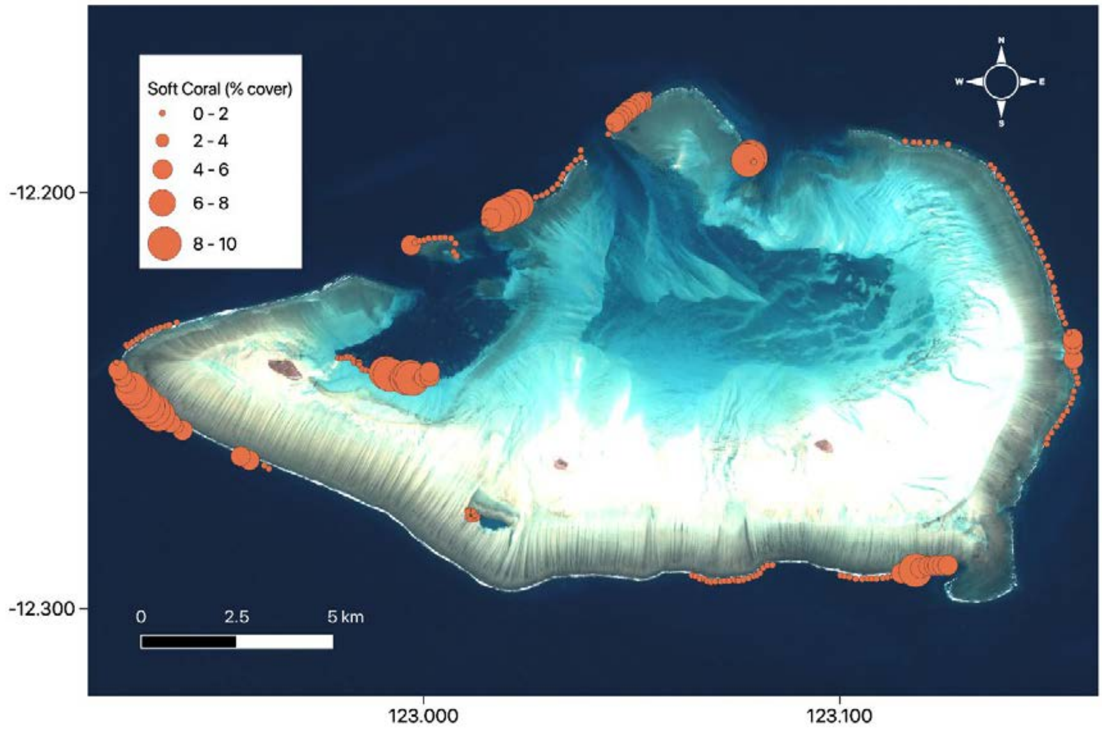


Figure 66. Estimates of percent cover of soft corals from manta tow transects at Ashmore Reef

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).

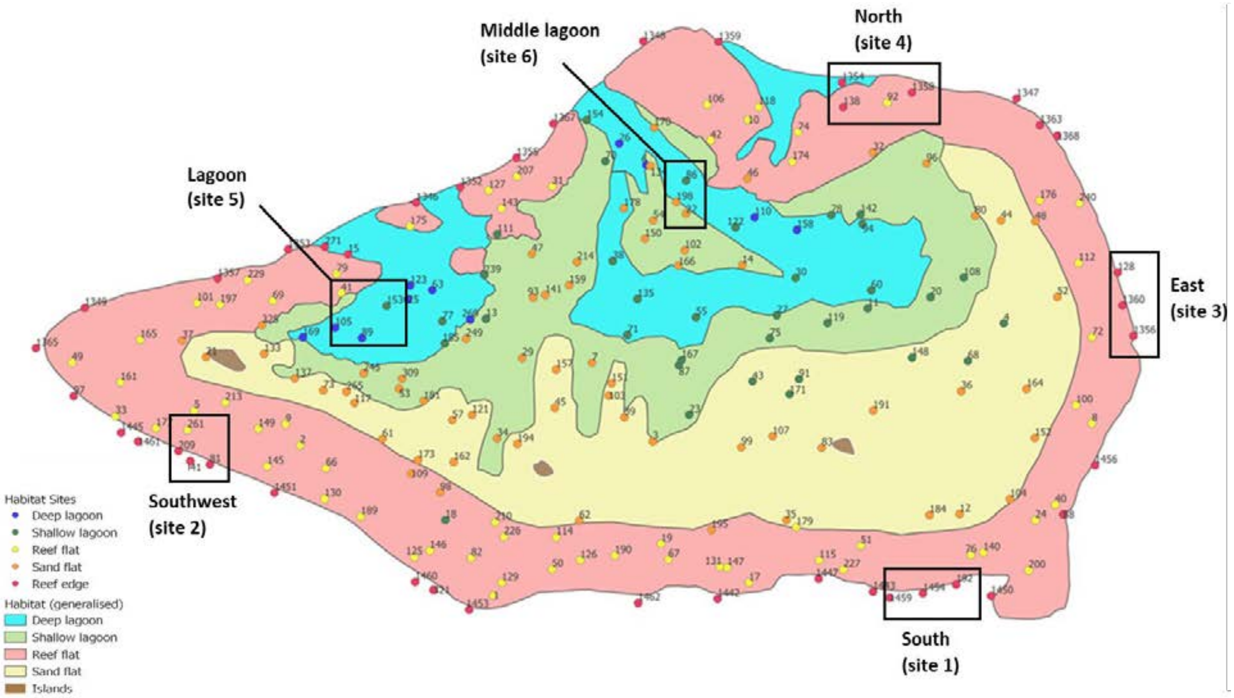


Figure 67. Map showing site groupings in 2019 for comparison with previous studies at Ashmore Reef in 2005 (Kospartov et al. 2005) and 2009 (Ceccarelli et al. 2011).

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			
	261	Reef flat	2			
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			
	1354	Reef edge	6			
Lagoon (5)	41	Reef flat	3	D7, D8	5	Lagoonal
	105	Deep lagoon	8			
	89	Deep lagoon	6.5			
	153	Shallow lagoon	3			
Middle lagoon (6)	86	Shallow lagoon	4	D11, D12	6	Lagoonal
	198	Sand flat	7			
	22	Sand flat	4			

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

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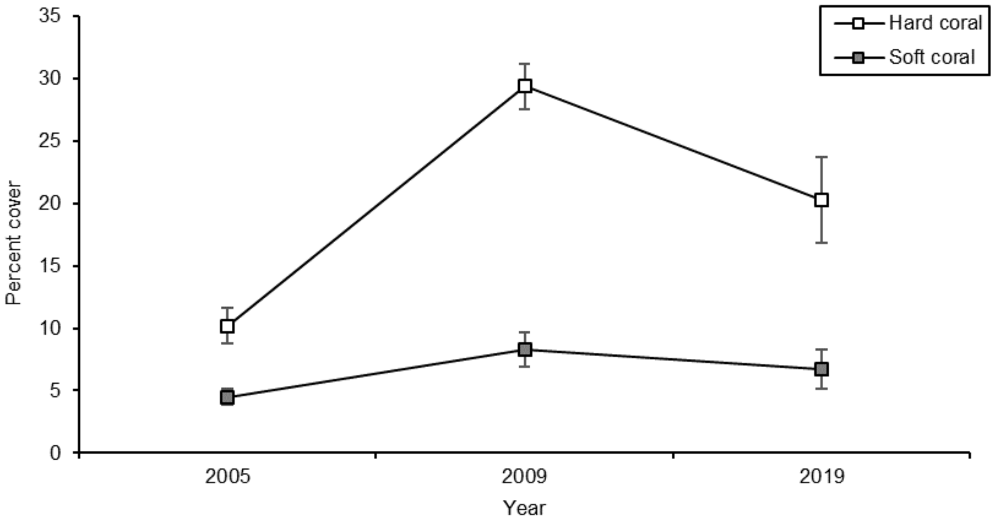


Figure 68. Change in mean percent cover of live hard and soft corals from 2005, 2009 and 2019 benthic surveys. Data from 2005 from Kospartov et al. (2006), data from 2009 from Ceccarelli et al. (2011). Error bars indicate ± 1 s.e.

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% (± 6 s.e.) at the shallow Middle lagoon site to 41% (± 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.

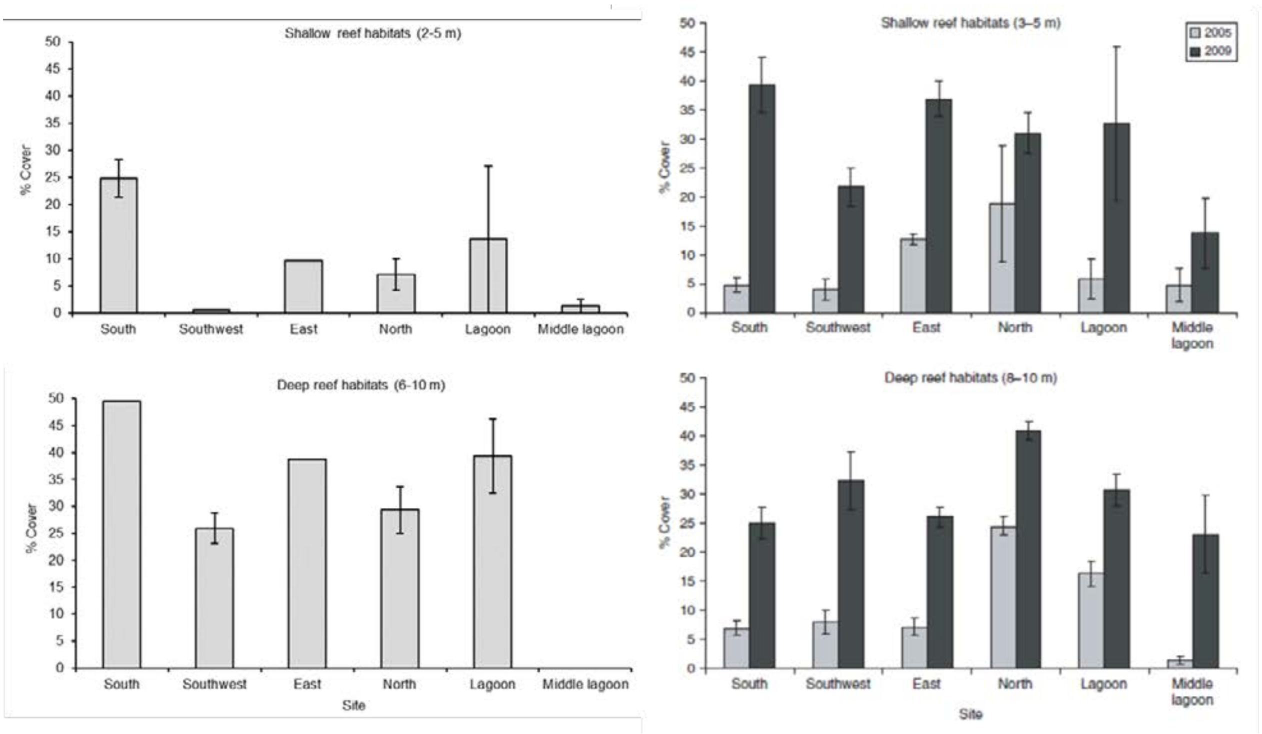


Figure 69. Temporal comparison of mean percent cover of live hard coral during 2019 benthic surveys (left panels) and 2005 (Kospartov et al. 2006) and 2009 (Ceccarelli et al. 2011) benthic surveys (right panels) for shallow (top panels) and deep (bottom panels) reef habitats. Error bars indicate ± 1 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 3rd 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 km2), 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^2), large mobile fishes along 1 x 50 m x 5 m transect (250 m^2) and smaller, site-attached fish surveyed along 1 x 50 m x 2 m belt transect (100 m^2). 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^2) 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.

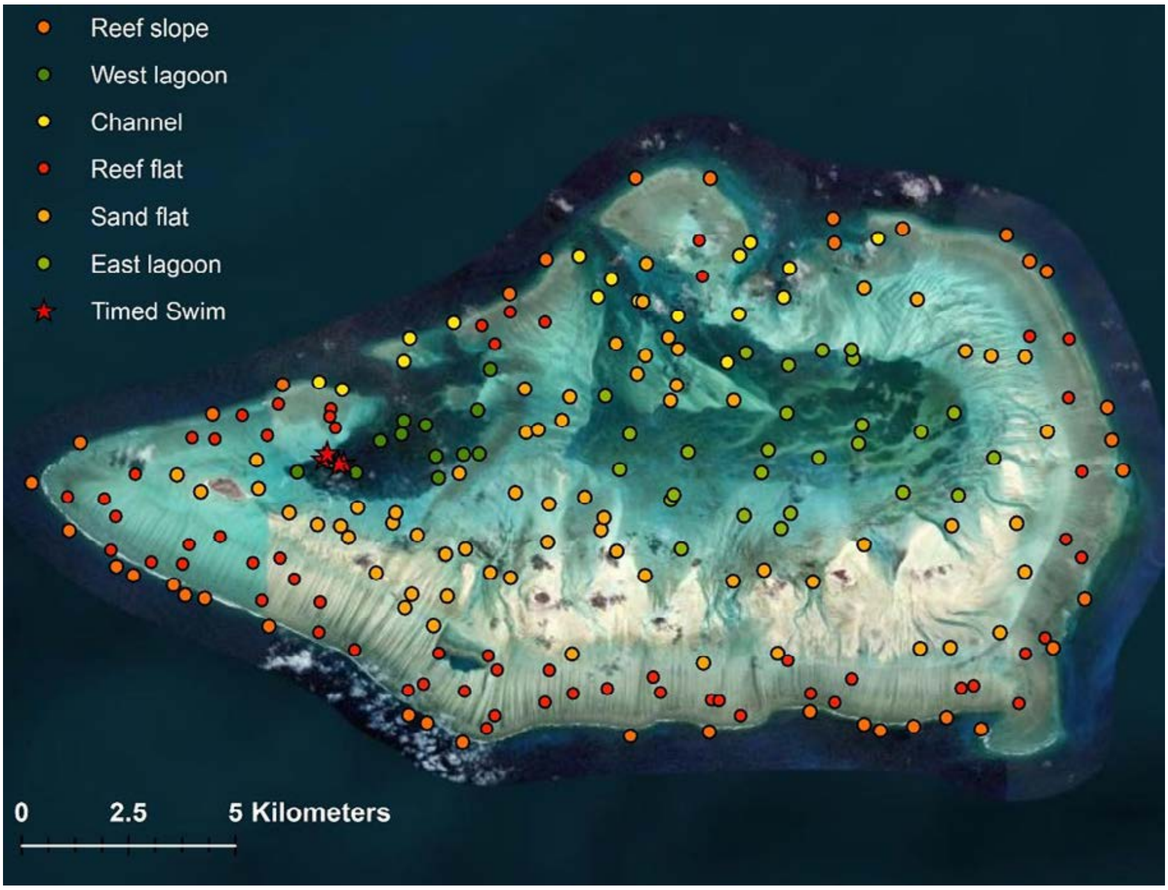


Figure 70. Map showing the location and reef zone classification of the 216 fish and shark survey sites at Ashmore Reef in September 2019. The six reef zones are shown in different colours and location of the four locations where times swims were completed are noted with a red star.

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). The estimated fish lengths were converted to biomass using derived length-weight conversions (Weight = a*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^2 for the 50 m x 10 m transects, 250 m^2 for 50 m x 5 m transects, 100 m^2 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

Classification	Diet
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^2). 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 (fuslliers), 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.

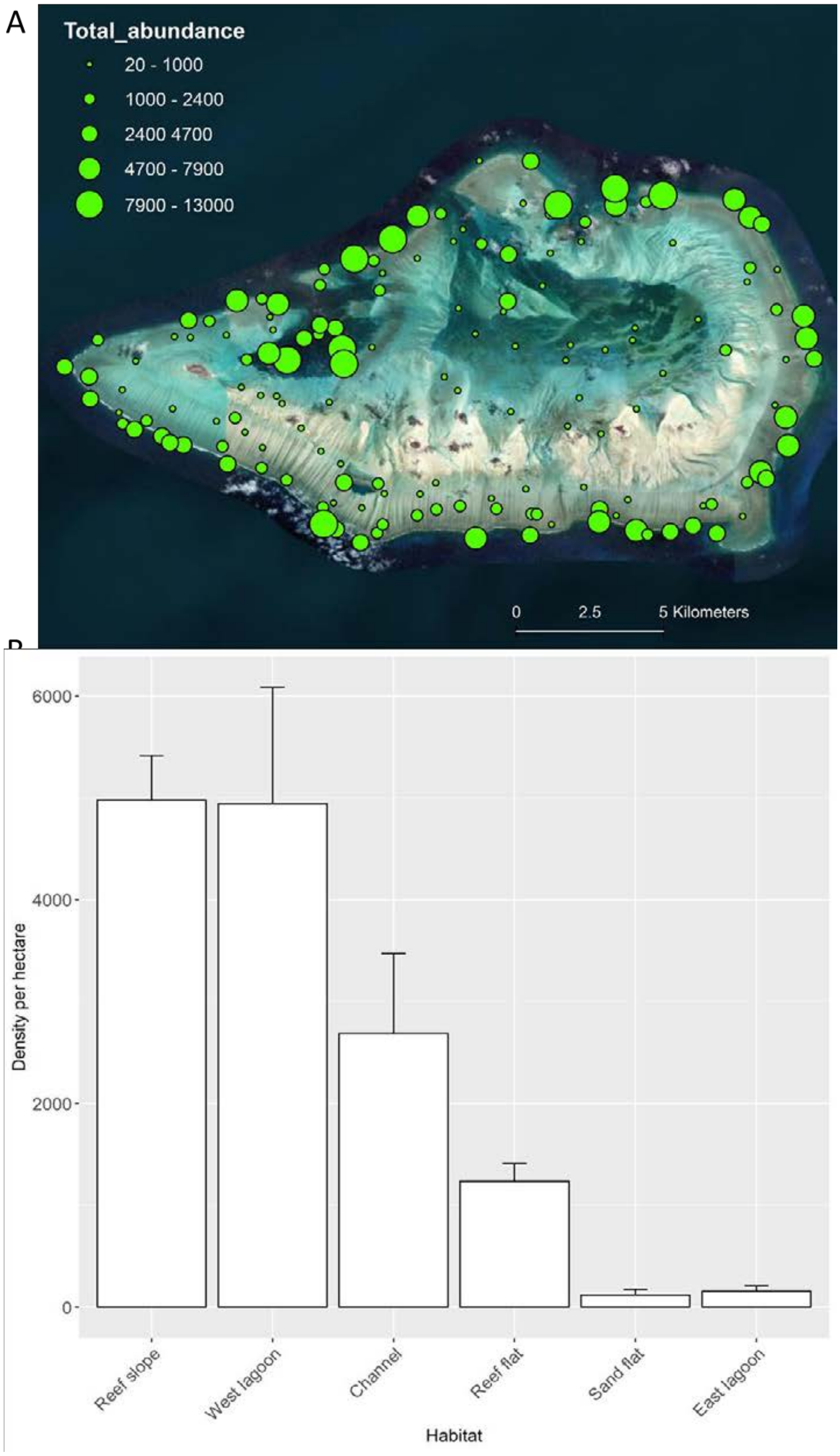


Figure 71. a) Map showing the total density of coral reef fishes observed at each of the 216 survey sites in 2019. Circle size represents the number of fish recorded per site as individuals/ha. b) Plot of mean density (+- se) of fish observed in the six reef habitats in 2019.

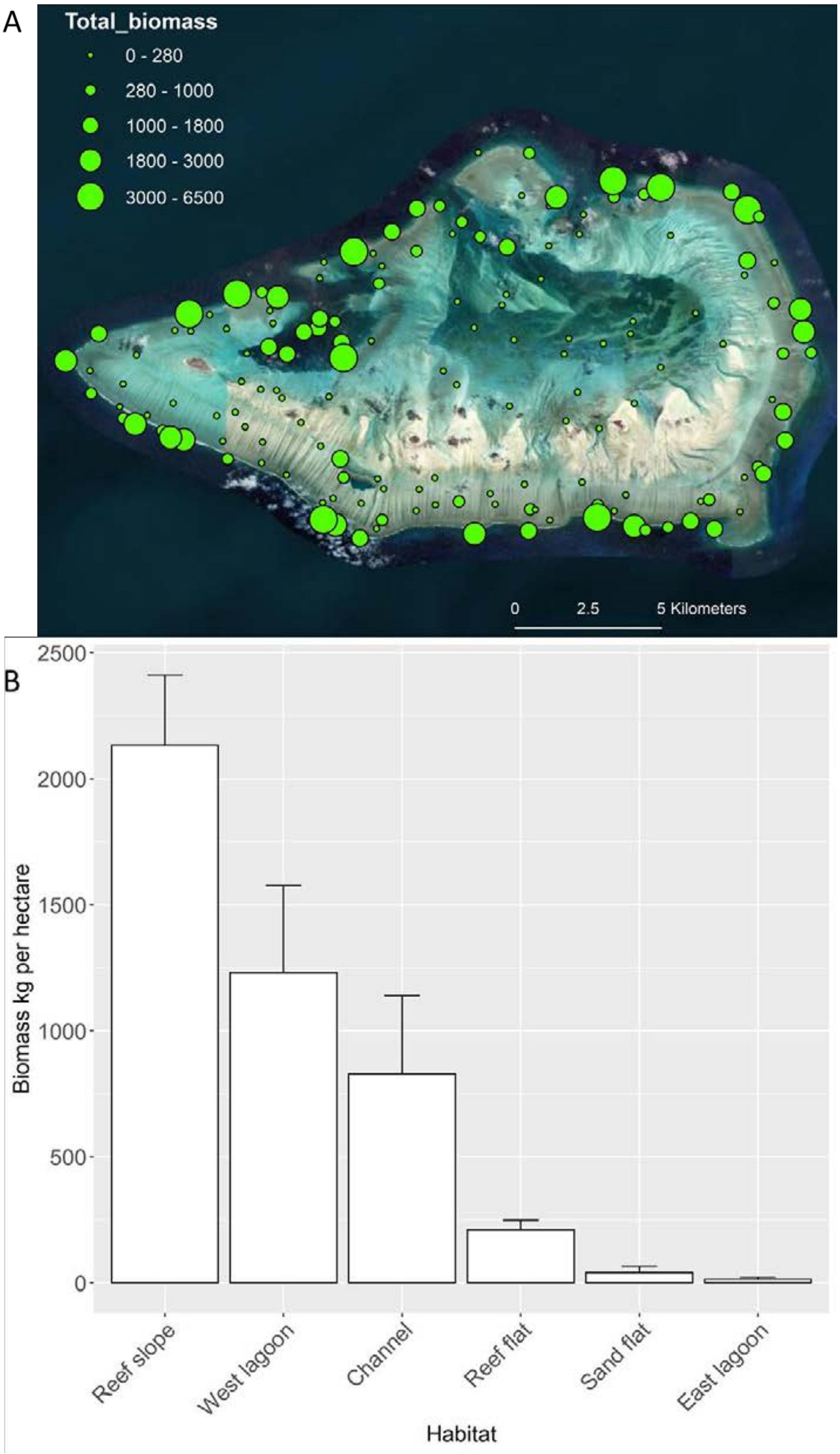


Figure 72. a) Map of biomass of coral reef fishes at Ashmore Reef at 216 survey sites in 2019. Circle size represents the number of fish recorded per site as individuals/ha. b) Mean biomass of fish (+- se) in six reef habitats.

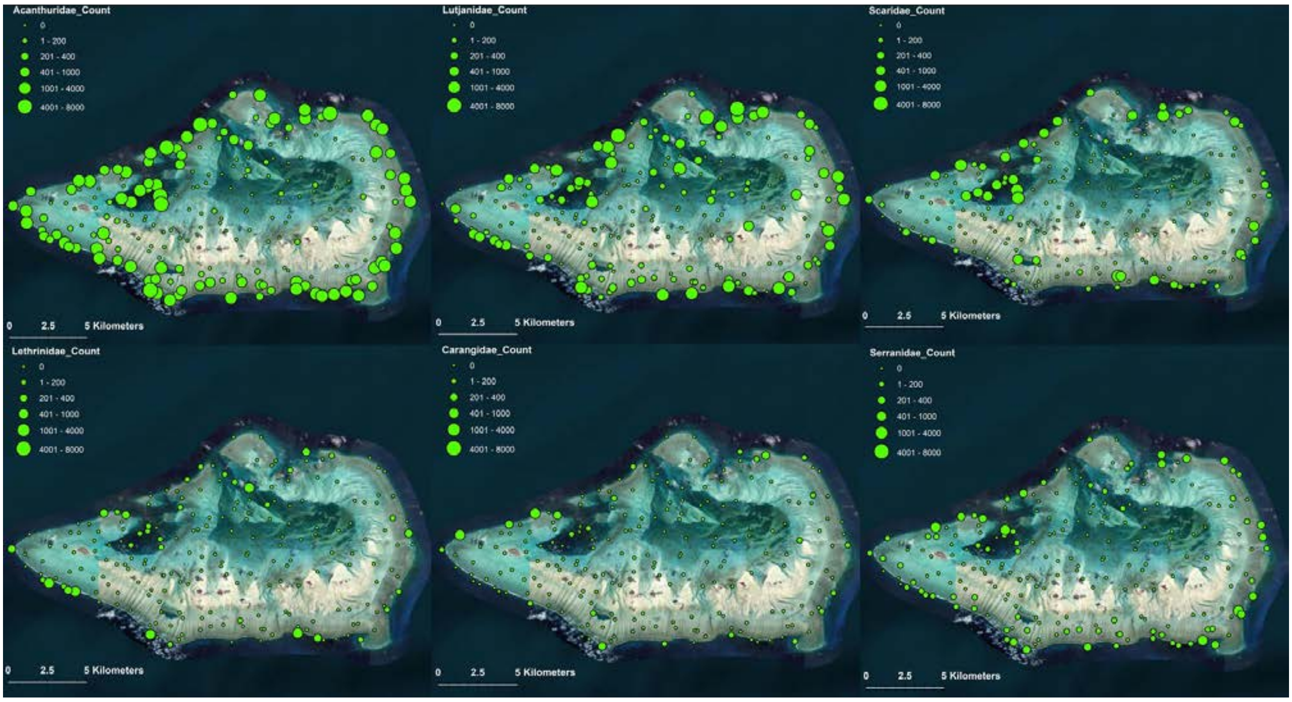

Density per hectare

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.

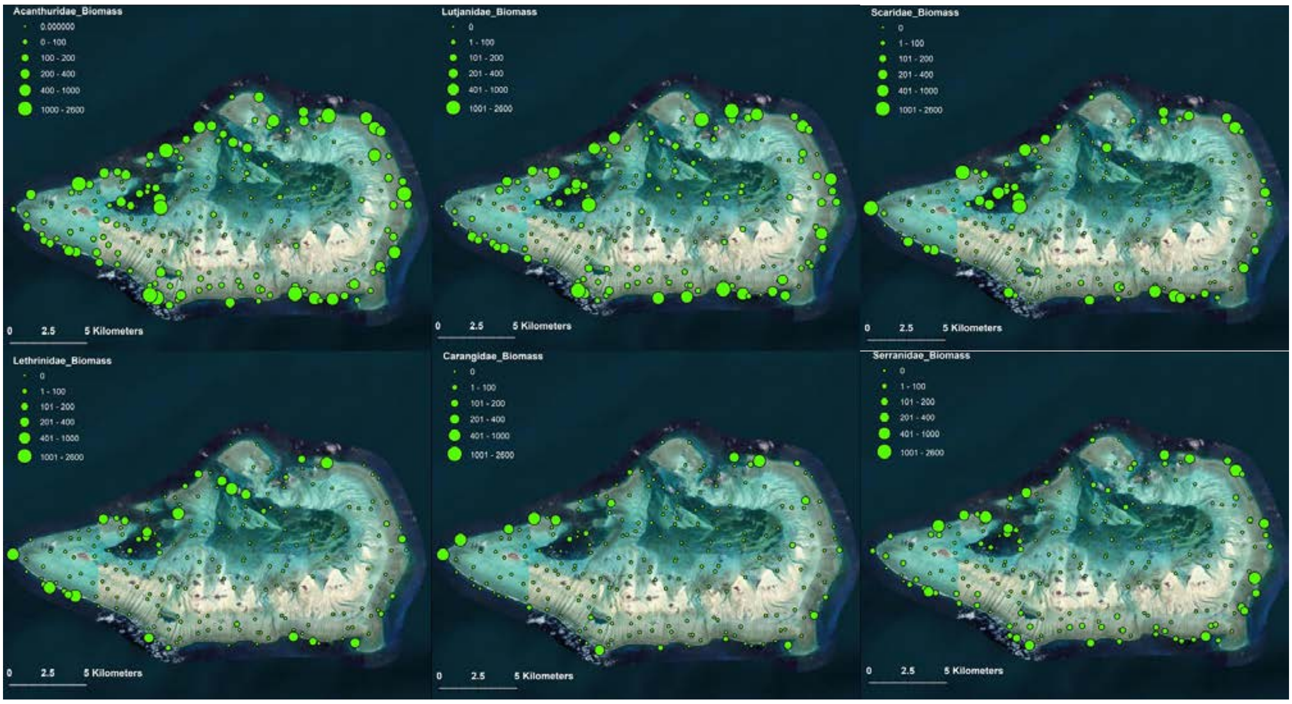

Biomass kg/ha

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.

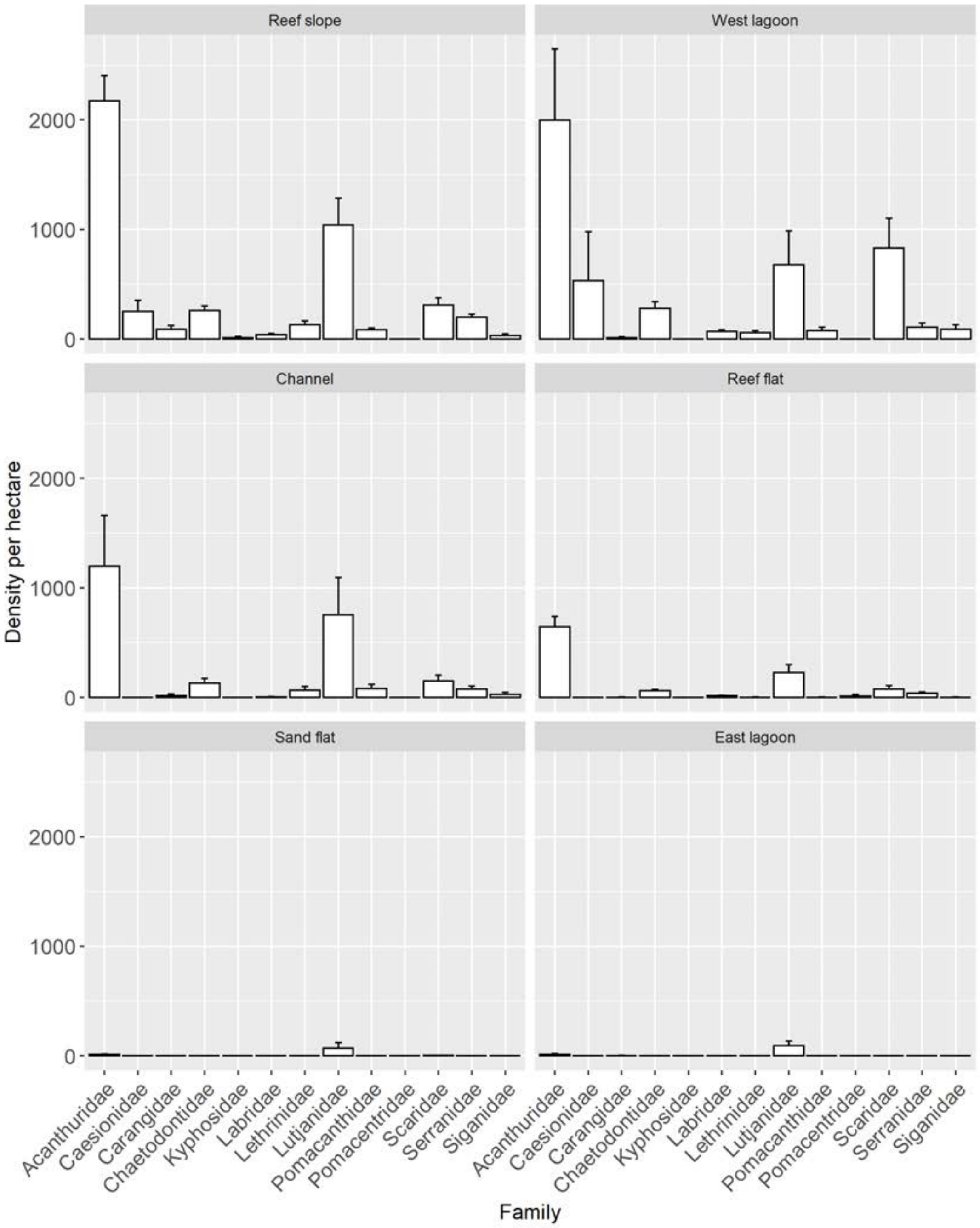


Figure 75. Mean density of the 13 most abundant families of fish within the six surveyed reef zones at Ashmore Reef.

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* (paddlefish), *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.

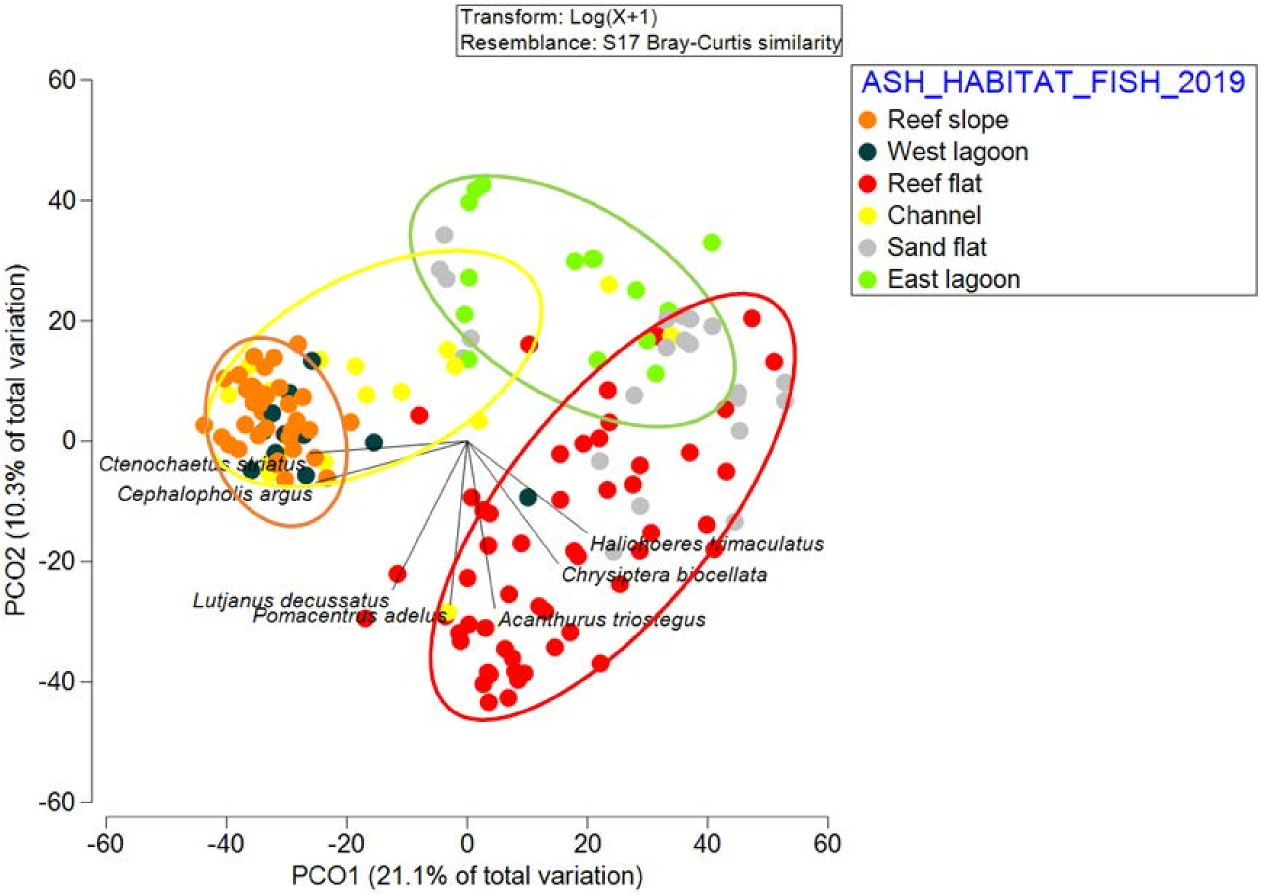


Figure 76. PCO showing the fish species composition of each reef-zone combination. Ellipses show the 95% confidence intervals associated with groups of dots representing each reef zone. The vectors represent fish species most responsible for differences between reef zones. The first two axes show percent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

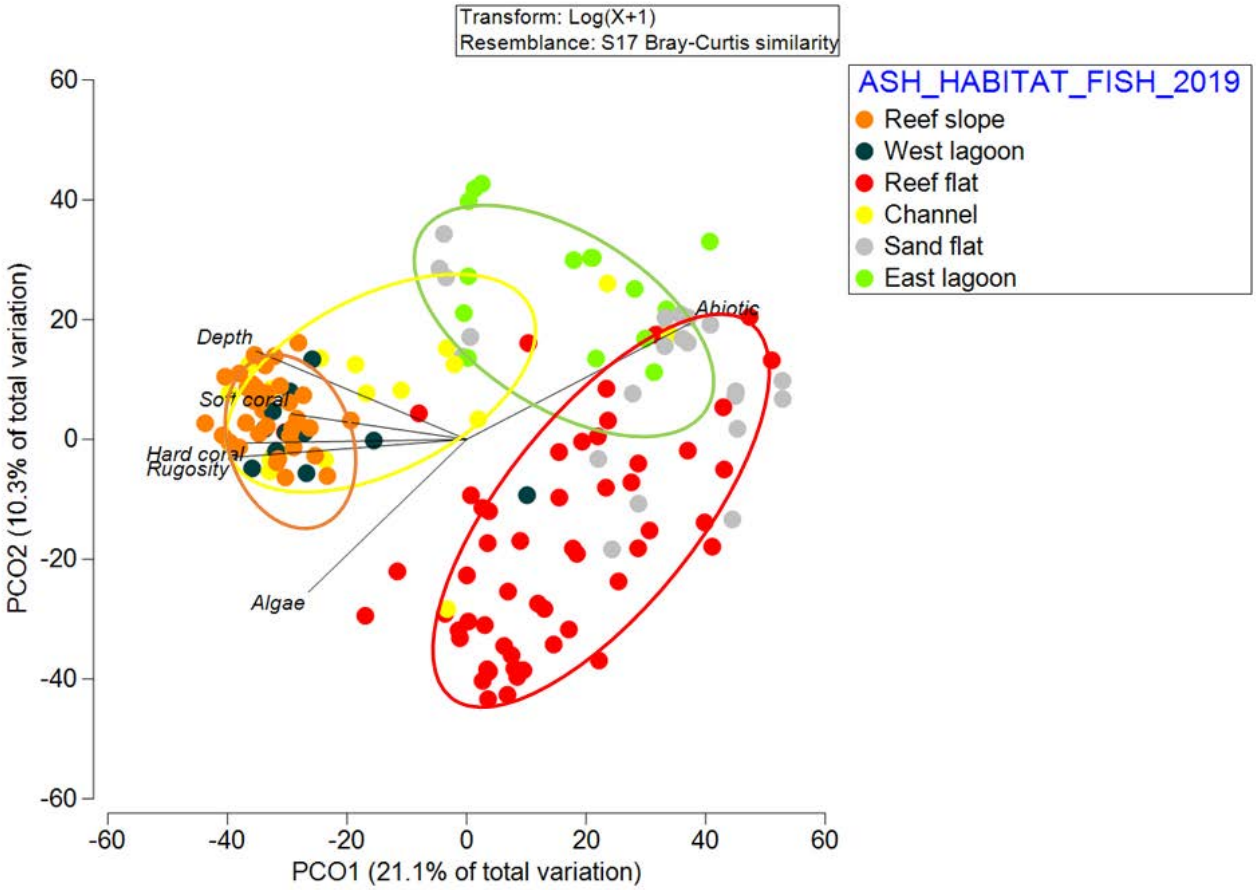


Figure 77. PCO plot showing the fish species composition of each reef-zone combination. Ellipses show the 95% confidence intervals associated with groups of dots representing each habitat combination. The vectors represent environmental variables most responsible for differences between reef zones. The 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).

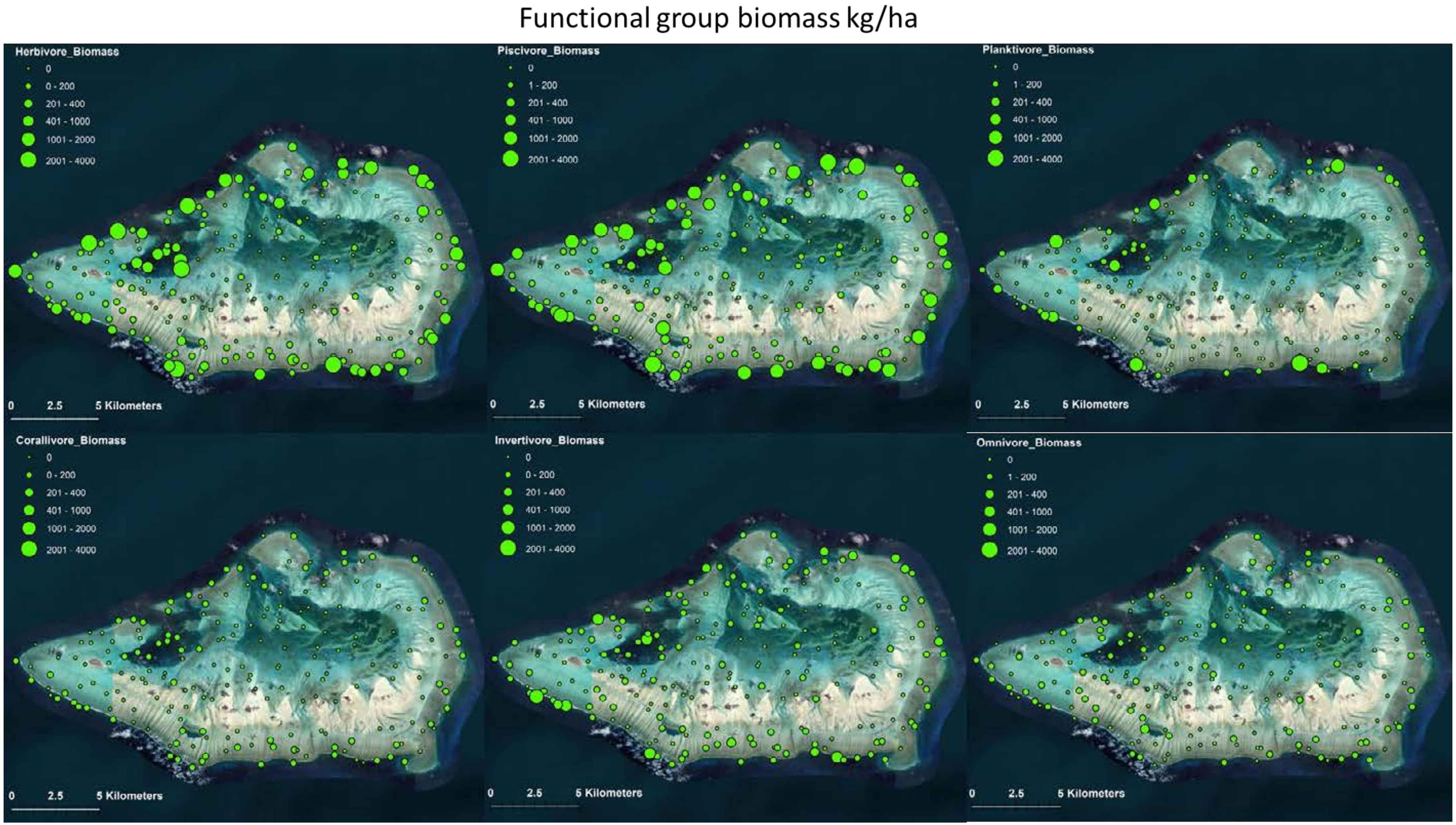


Figure 78. Bubble plots showing the biomass of functional groups of reef fishes observed within the six most abundance families at each of the 216 survey sites in 2019. Circle size represents the biomass of fish recorded per site as kg/ha.

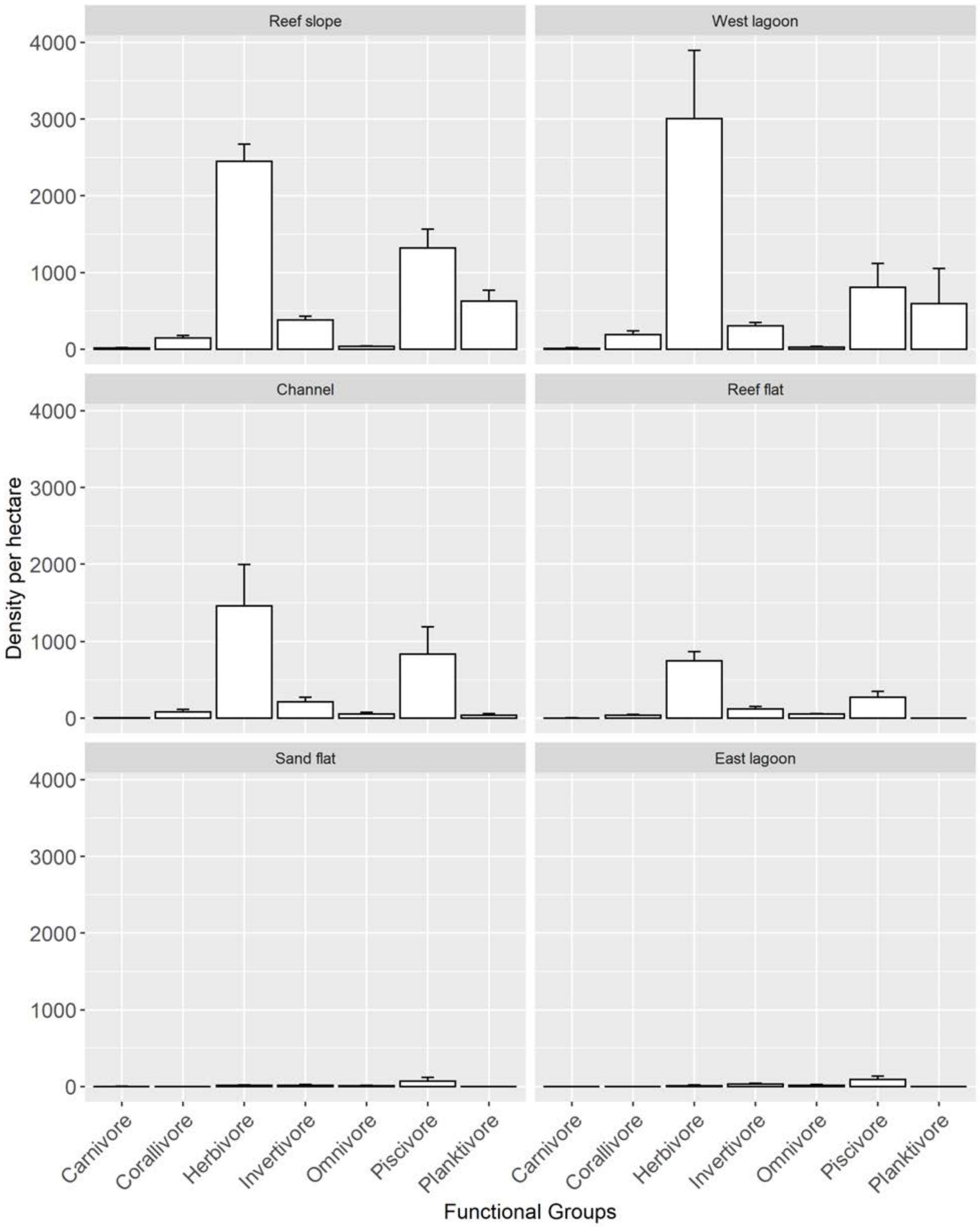


Figure 79. Mean density of the seven functional groups of fish within the six surveyed reef zones at Ashmore Reef.

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).

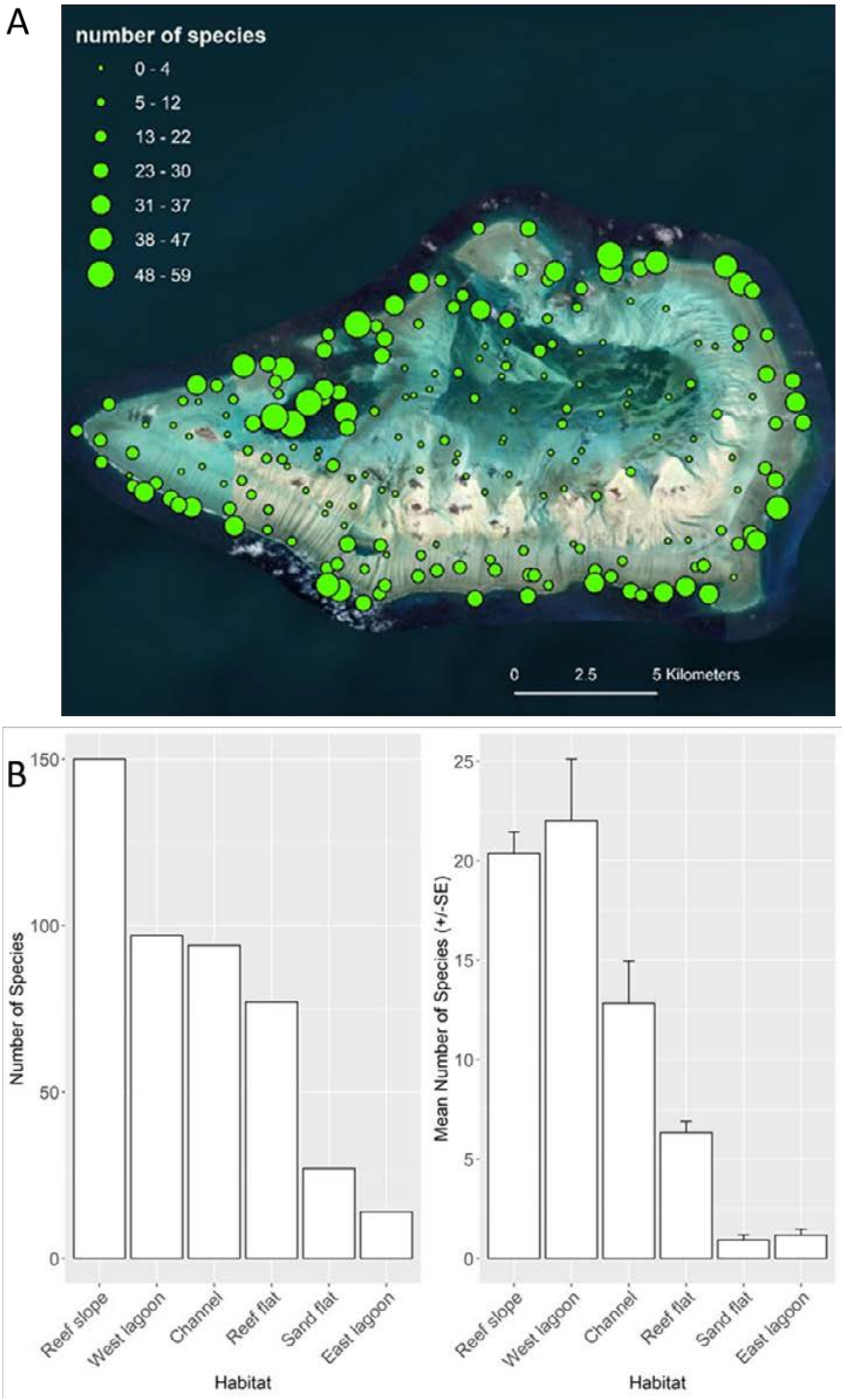


Figure 80a. Map showing the total number of fish species observed at each of the 216 survey sites in 2019. Circle size represents the number of fish species recorded per site. Figure 48b. Total number of species observed in the six reef habitats in 2019. Figure 74c Mean number of species (+-se) observed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

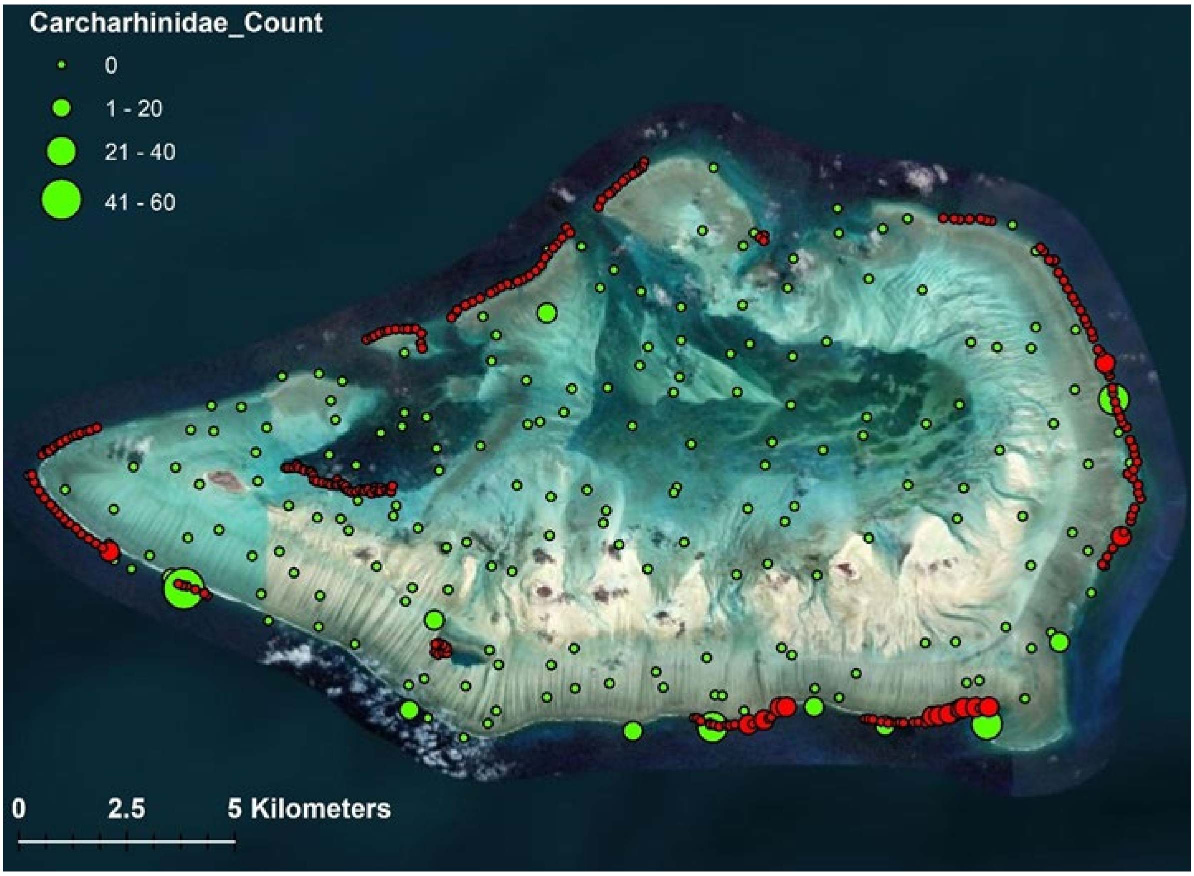


Figure 81. Map showing the density of sharks recorded during UVC surveys (green dots) and manta tow surveys (red dots) at Ashmore Reef in 2019

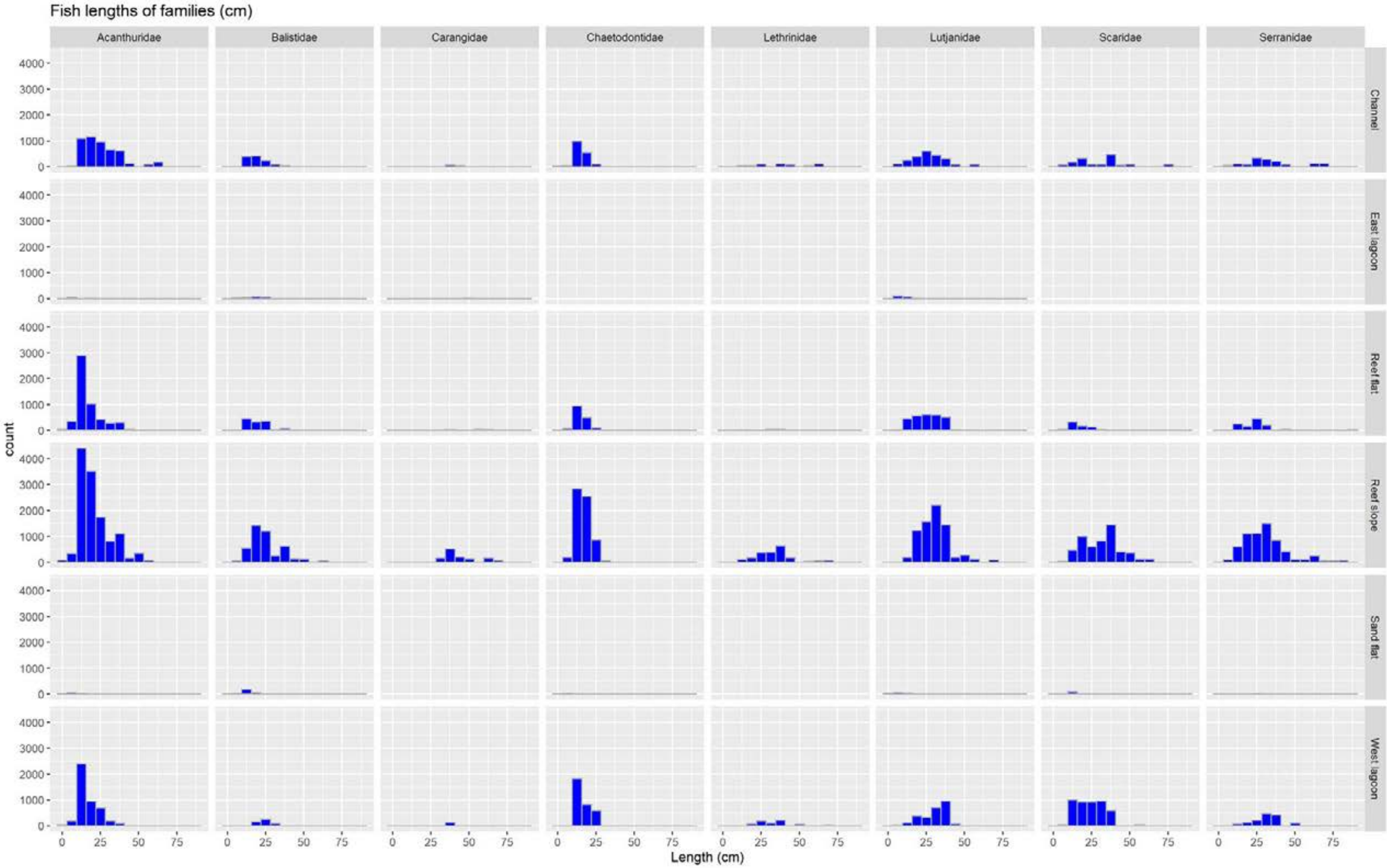


Figure 82. Size-frequency distributions of common fish families within the six reef zones at Ashmore Reef in 2019.

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, were 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).

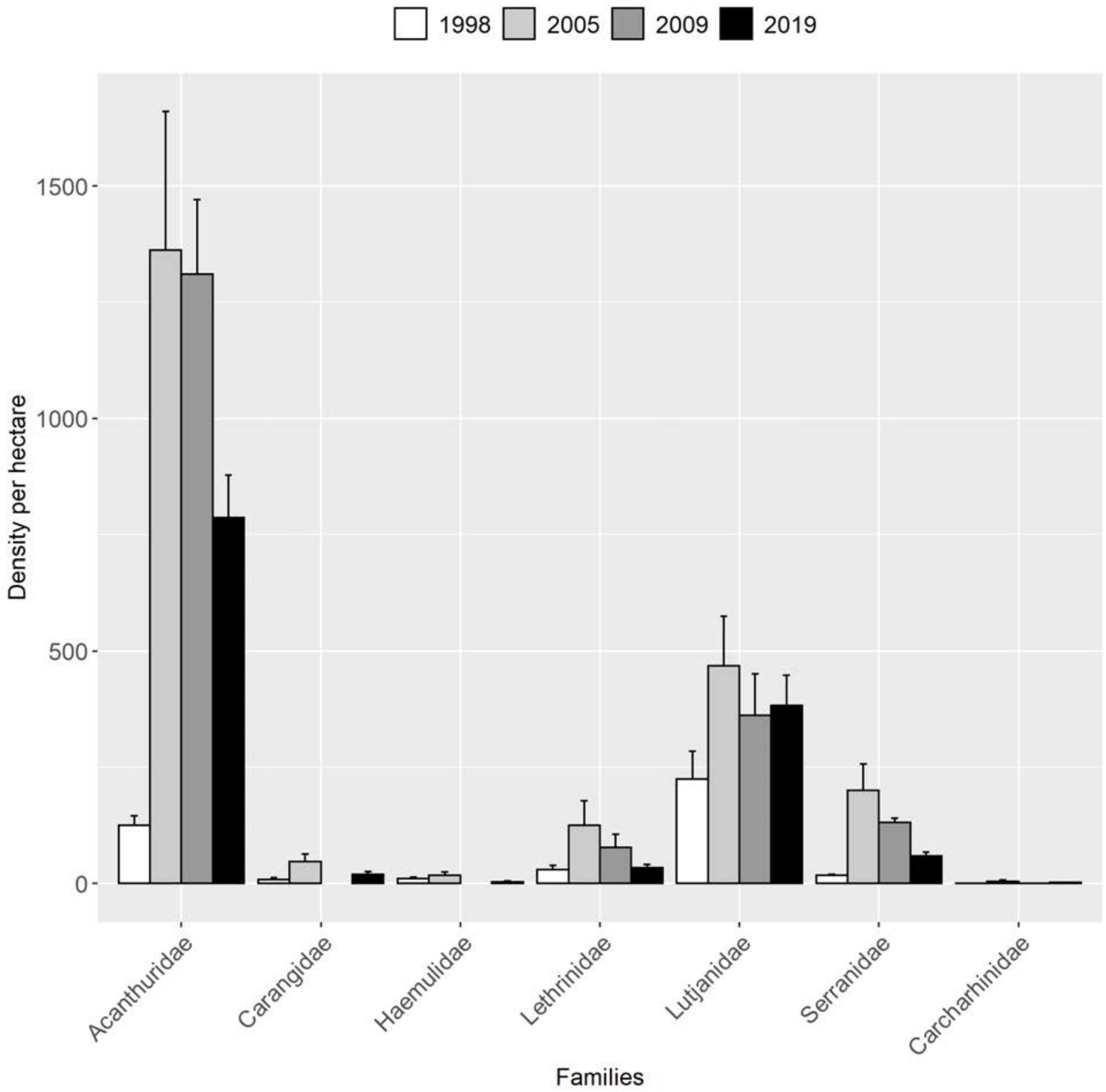


Figure 83. Mean density per ha of fish observed in 1998, 2005, 2009 and 2019 surveys in the 6 major fish families and carcharhinid sharks. SE presented for years where raw data available. Length data for some families were not available for 2009 surveys.

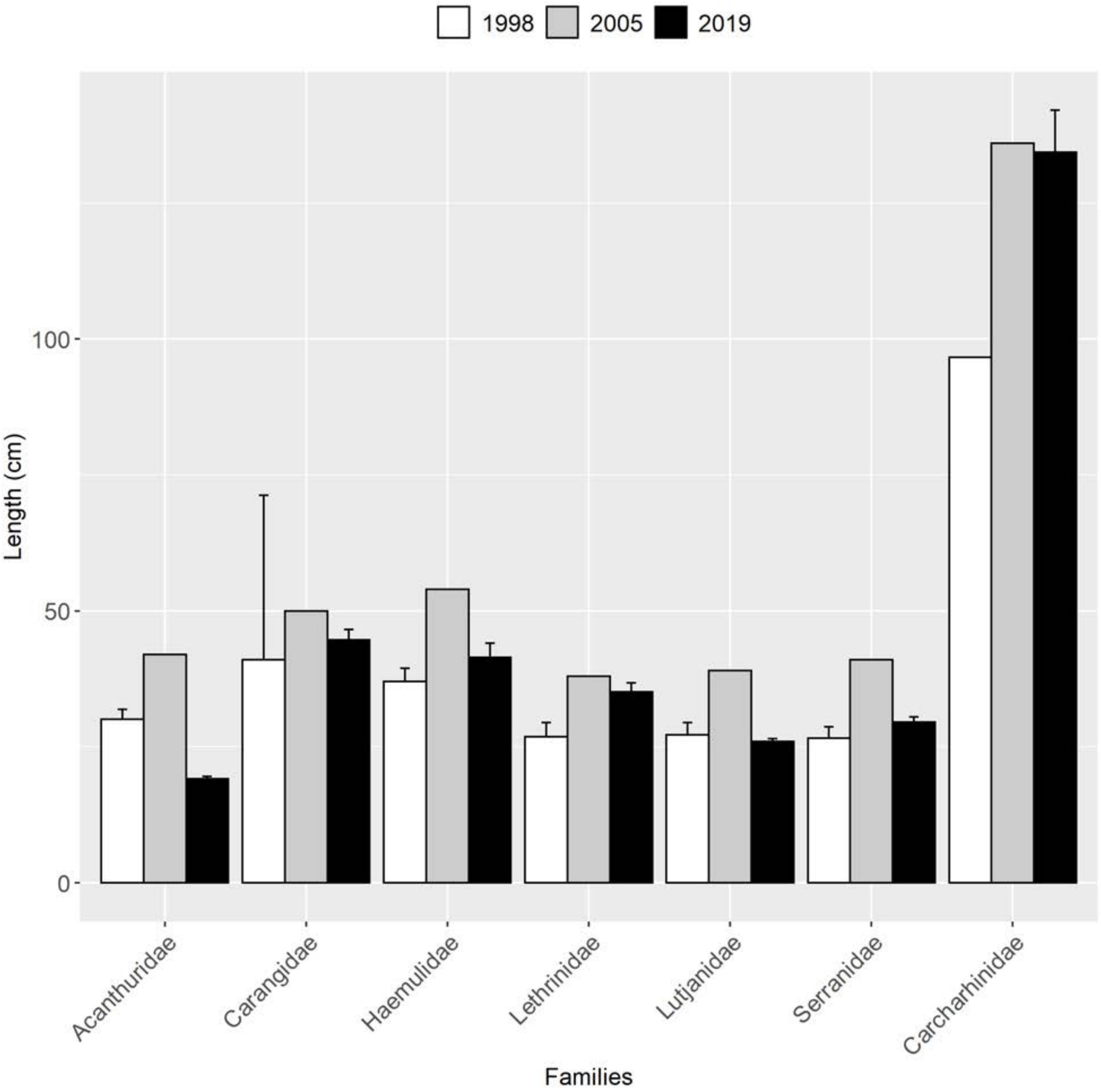


Figure 84. Mean lengths of fish observed in 1998, 2005 and 2019 surveys in the 6 major fish families and carcharhinid sharks. Standard errors are presented for years were raw data were available. Length data for families in 2009 were not available.

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 corelated 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, Cappo & 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. fuscorubra, Thelenota ananas*, *T. anax* (all <0.02 /100 m^2)*, H. fuscogilva* (0.05 /100 m^2), *Stichopus herrmanni* (0.2 /100 m^2), *Tridacna gigas* (<0.05 / 100m^2) and *Hippopus hippopus* (0.37 / 100 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.

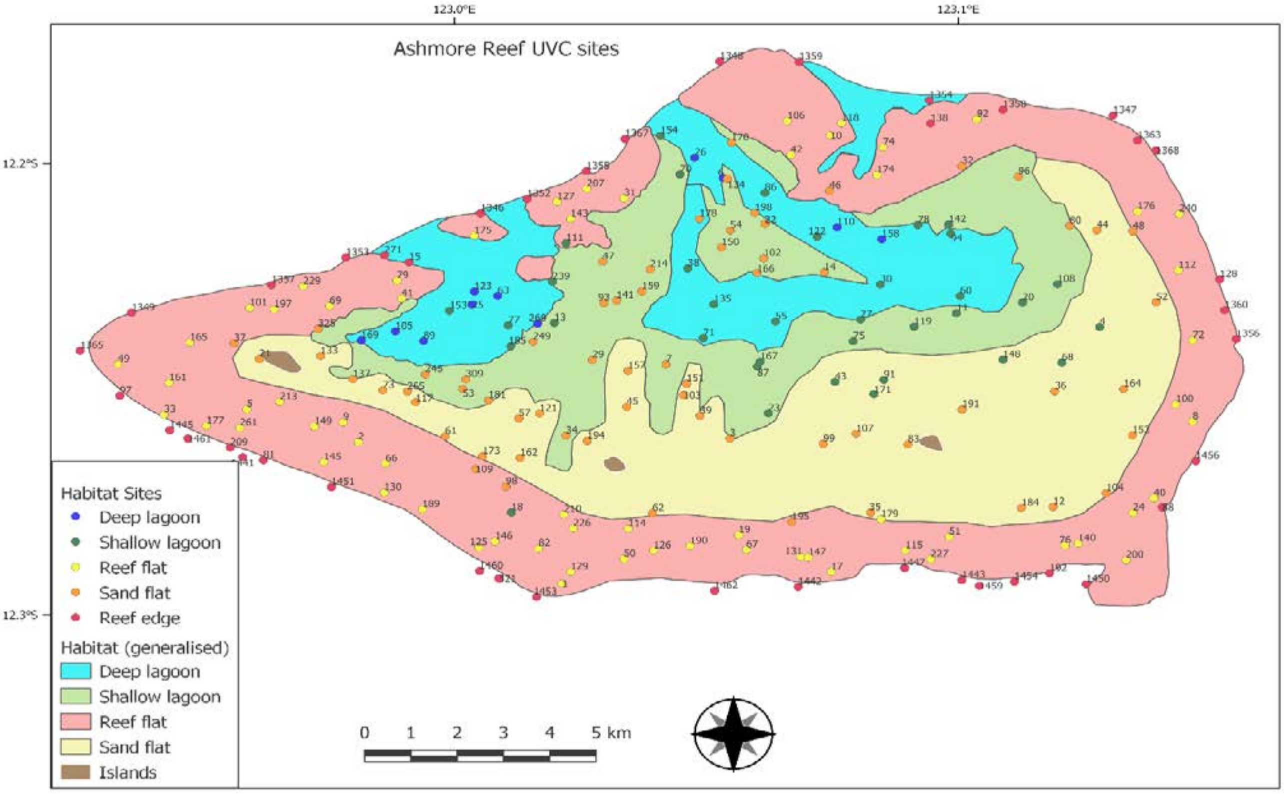


Figure 85. Map of the location of sites surveyed at Ashmore Reef in June 2019 together with habitat stratification.

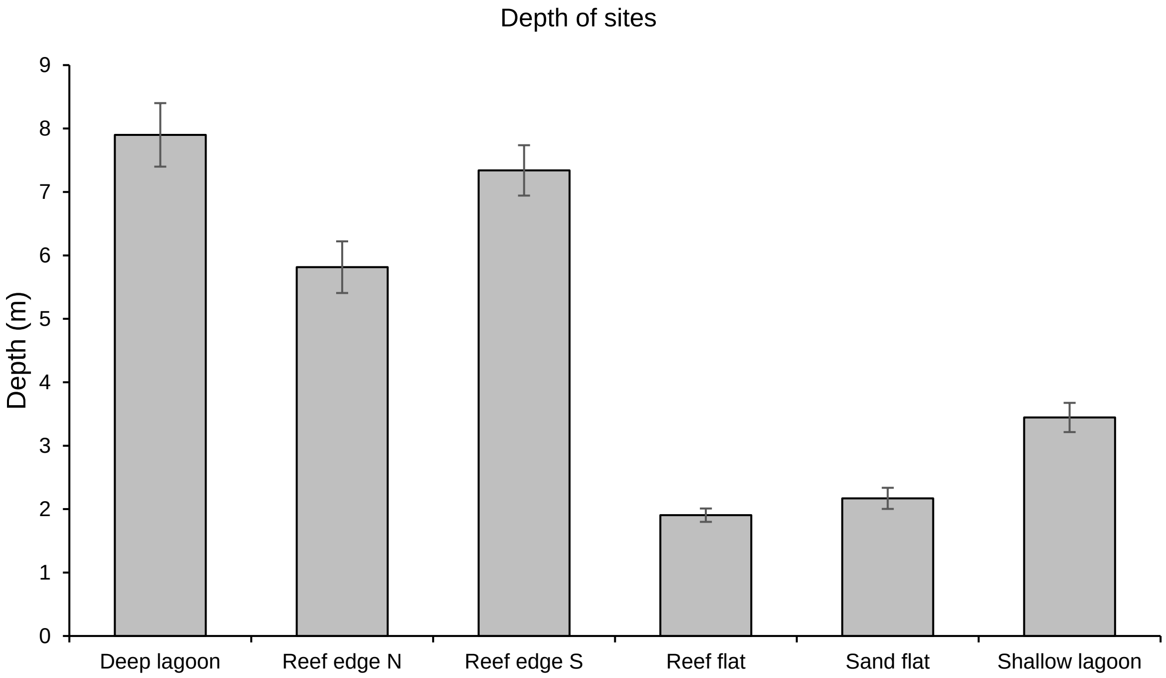


Figure 86. Mean depth of sites in each of the habitat types surveyed, at the time of survey. In general, sites on the reef flat, sand flat and shallow lagoon were surveyed between the mid- to high-tide range, as navigation is not possible at low tides. Reef edge and deep lagoon sites could be surveyed on most tides. Error bars are ±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^2 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^{-1}. 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 I25 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^2 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^2 invertebrate transects. The smaller *Linckia multifora* was only counted on the 100 m^2 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^2 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).

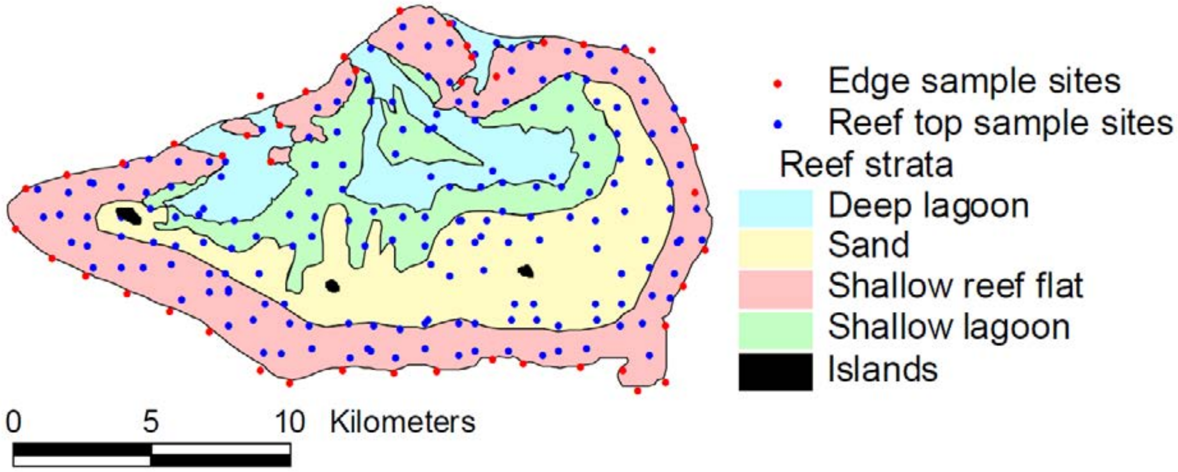


Figure 87. Map of habitat stratification of Ashmore Reef and location of sampling sites taken from figure 2 of Skewes et al. (1999a).

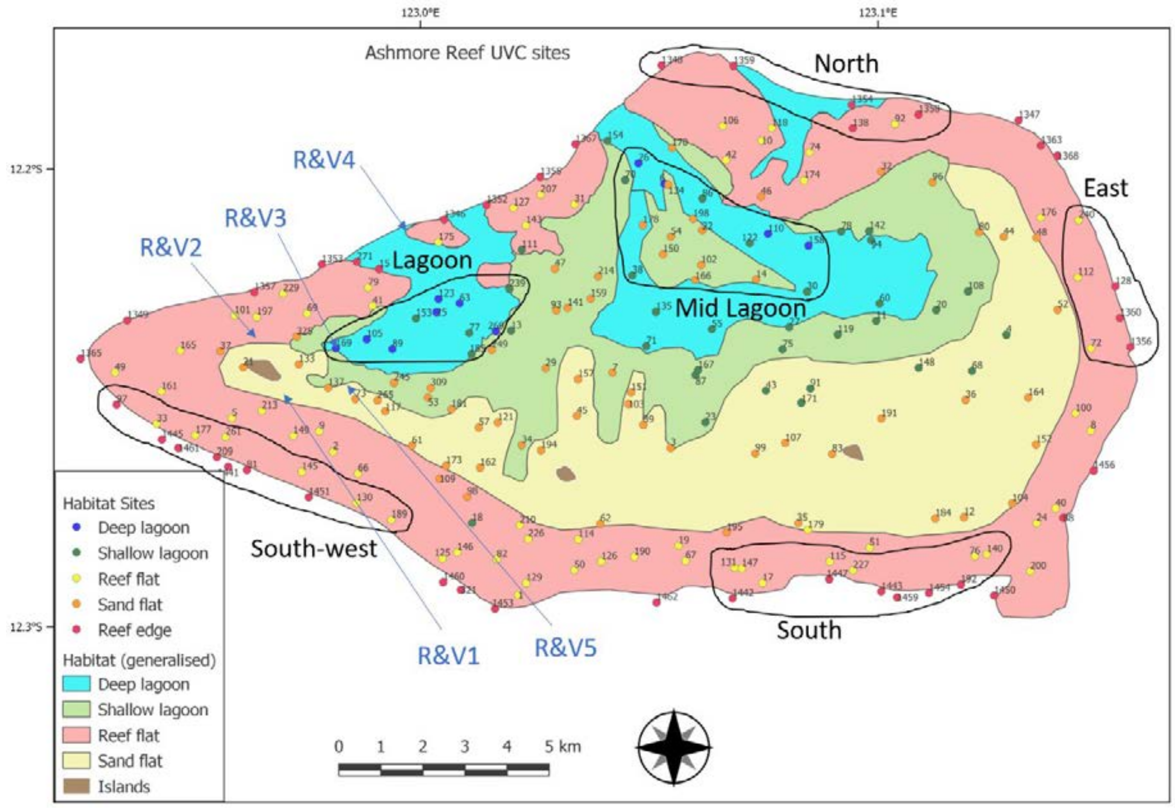


Figure 88. Map of the matchup of the 2019 survey sites (numbered) and the six location-based 2006 and 2009 survey sites used by Ceccarelli et al. (2007) and Richards et al. (2009) labelled South, North, etc. The black areas circled indicate which of the 2019 sites were used to compare with the 2006 and 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). *Acinopyga 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^2 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^2 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.

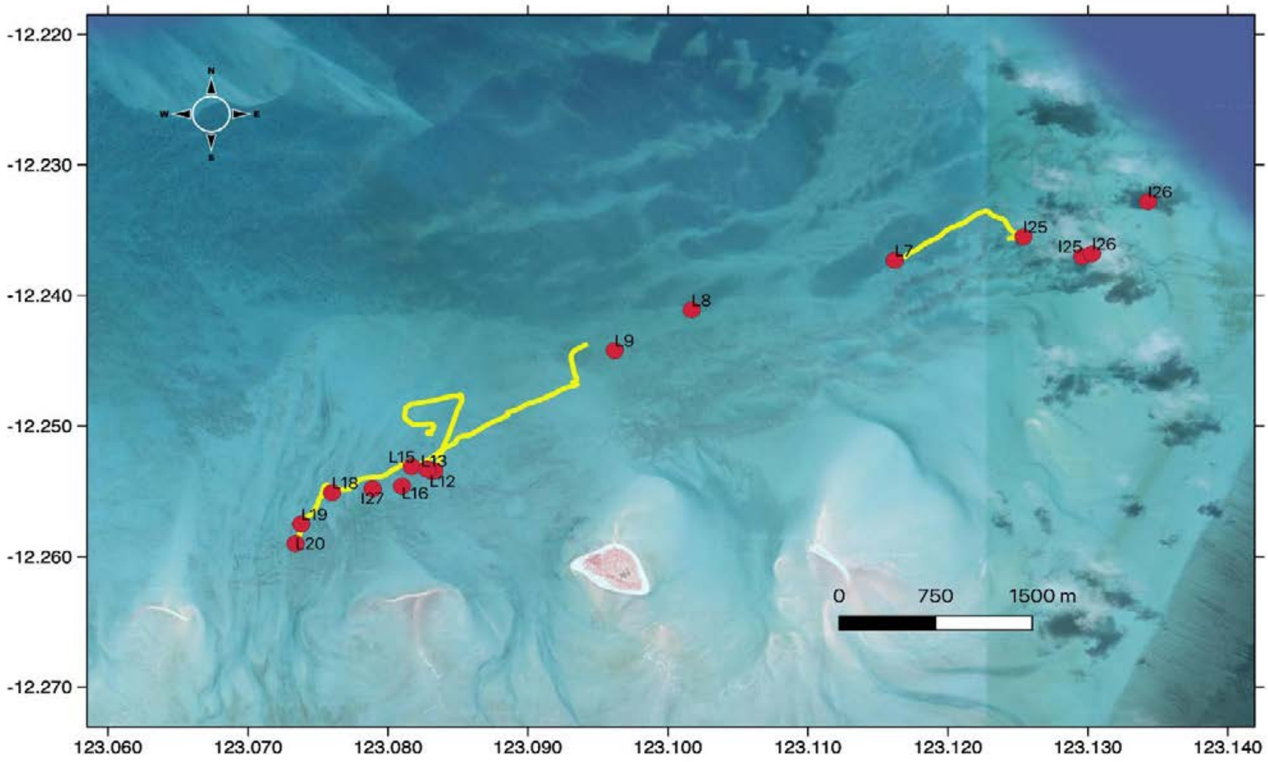


Figure 89. Manta tow tracks (yellow lines) and sites of high-density *H. leucospilota* aggregation reported in 2005 and 2006 by Ceccarelli et al. (2007) (red dots) near East Island.

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

	Estimated density per hectare		
Site	**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).

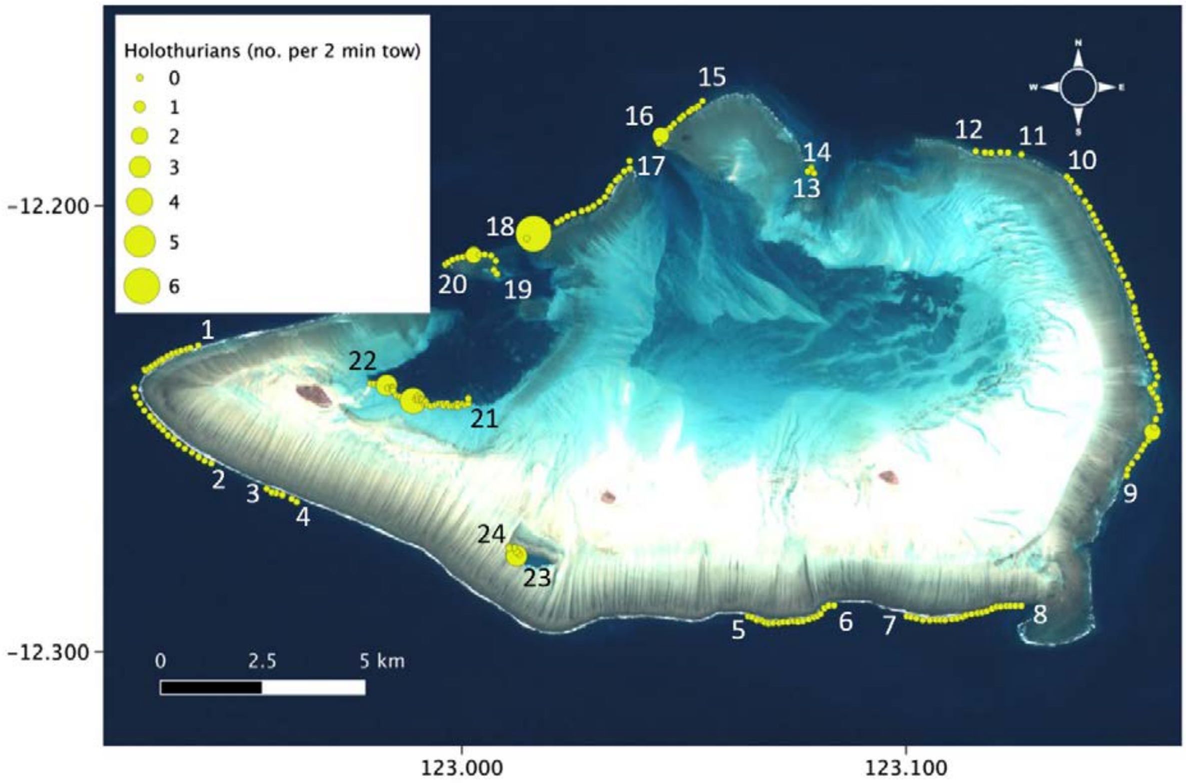


Figure 90. Map of Ashmore Reef showing the number of holothurians recorded throughout the two-minute manta tows. Each two-minute tow is shown as a yellow dot and the numbers refer to waypoints given in the supplementary information Table 13.

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^2) and along the sheltered northern reef edge (1.05/100 m^2), while the deep lagoon, shallow lagoon and sand flats had the lowest densities (ca. 0.3/100 m^2)(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).

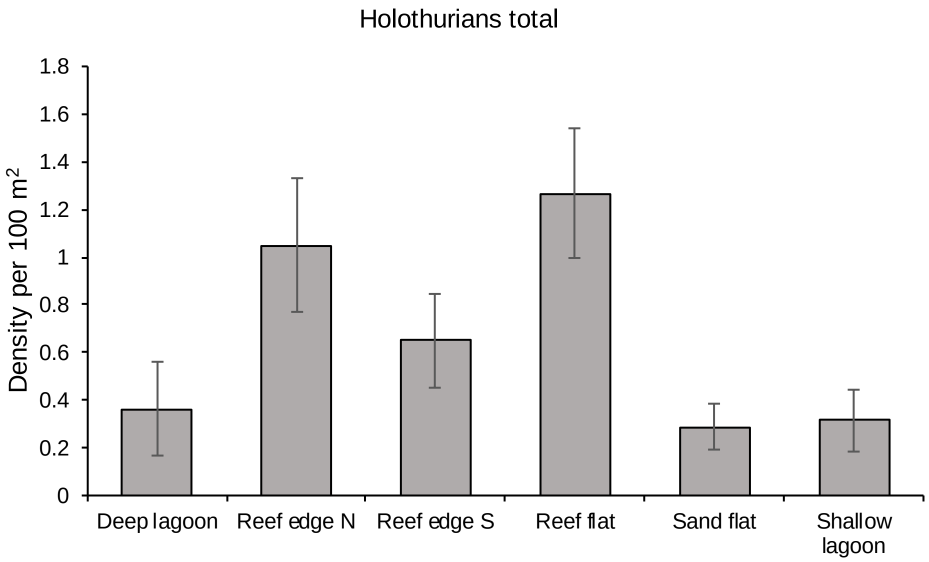


Figure 91. Mean density of holothurians (all species combined) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

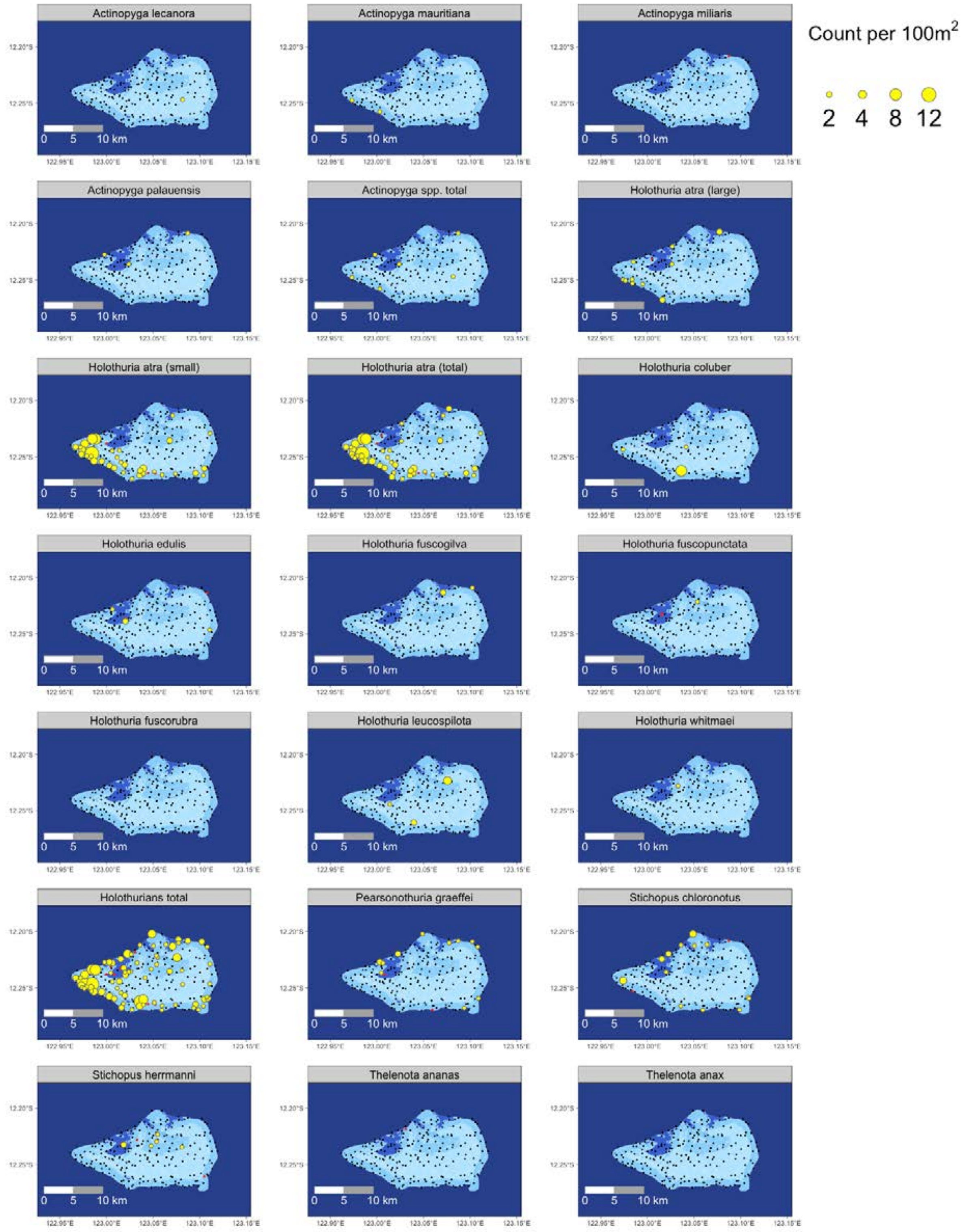


Figure 92. Distribution and density/100 m^2 of all holothurian species across each of the 224 transects surveyed in June 2019. The red dots indicate presence of that species at that site but that the specimen(s) were not within the area of the 100 m^2 quantitative survey transect.

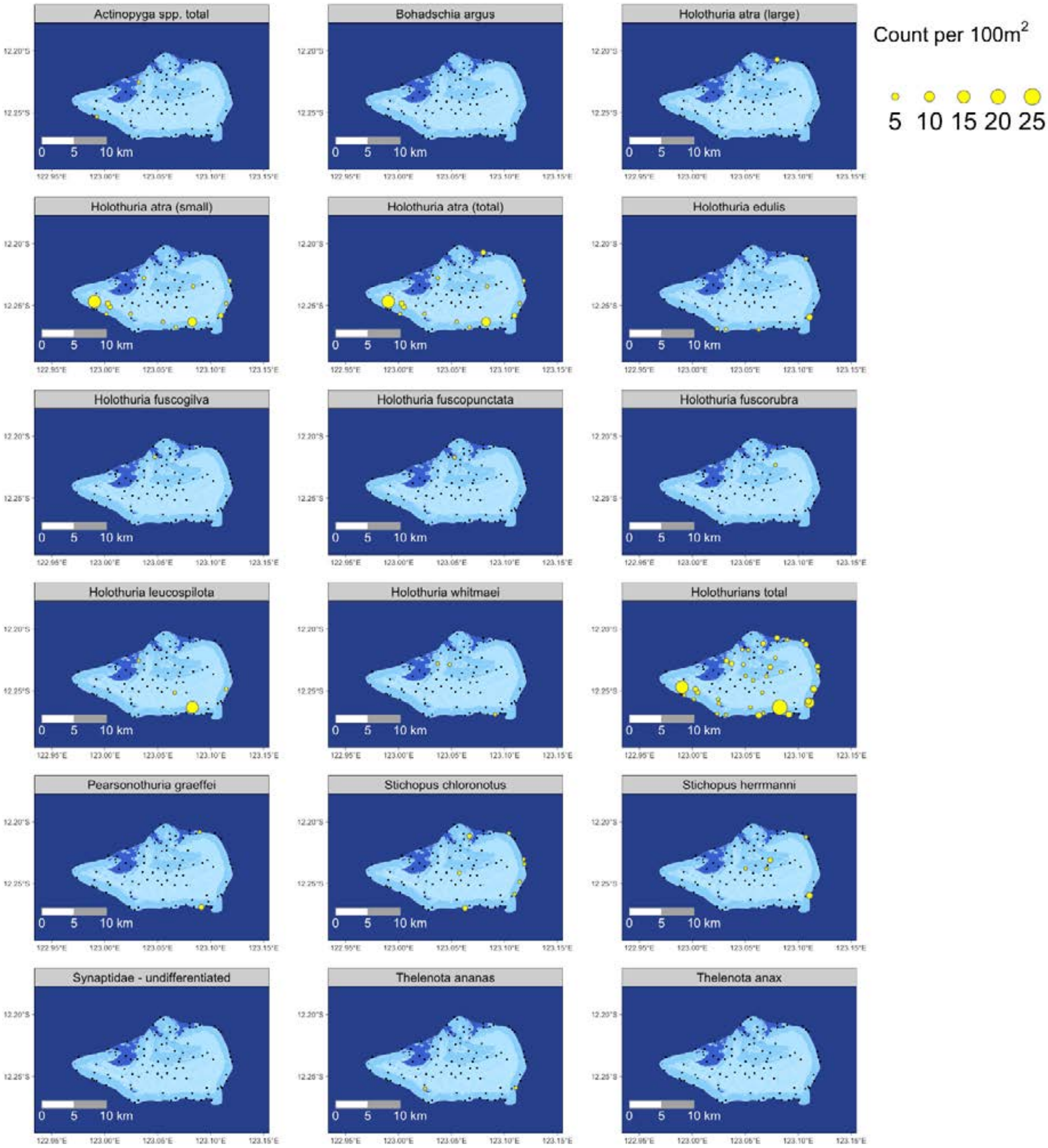


Figure 93. Distribution and density/100 m² of all holothurian species across each of the 95 transects surveyed in March 2013 Data from Ceccarelli et al. (2013) with densities converted to per 100 m² from their 80 m² transects.

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^2) while the asexual form was most abundant on the reef flat (0.96/100 m^2) (Figure 94).

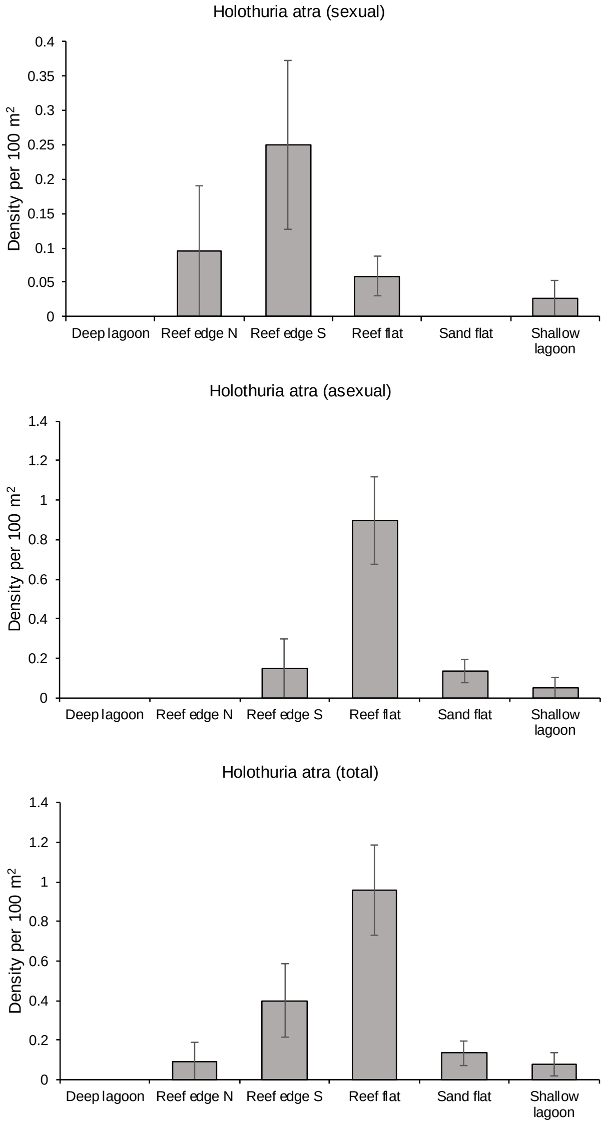


Figure 94. Density of *Holothuria atra* in each of the habitats surveyed at Ashmore Reef in June 2019. Top panel shows the large sexually reproducing form, middle panel shows the small asexually reproducing form and the bottom panel shows the combined total. Error bars are ±1 s.e.

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).

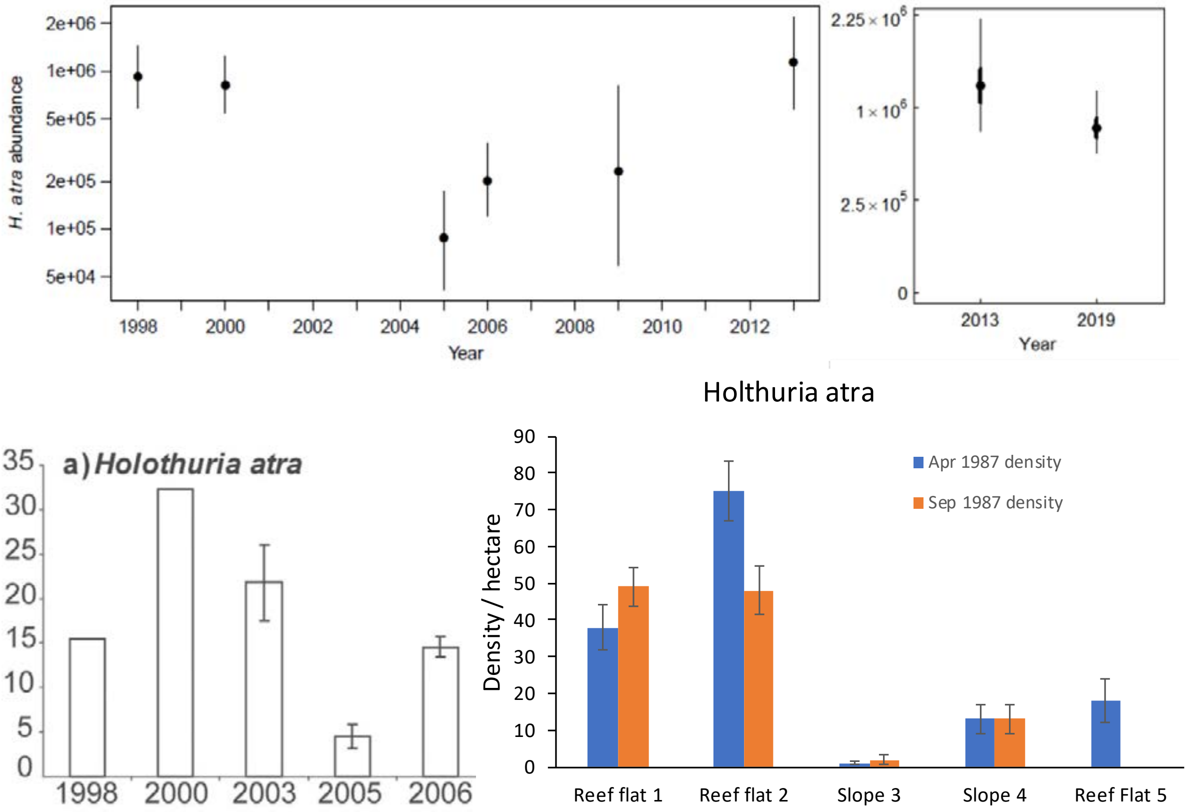


Figure 96. Long term estimates of abundance of *Holothuria atra*. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both panels.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

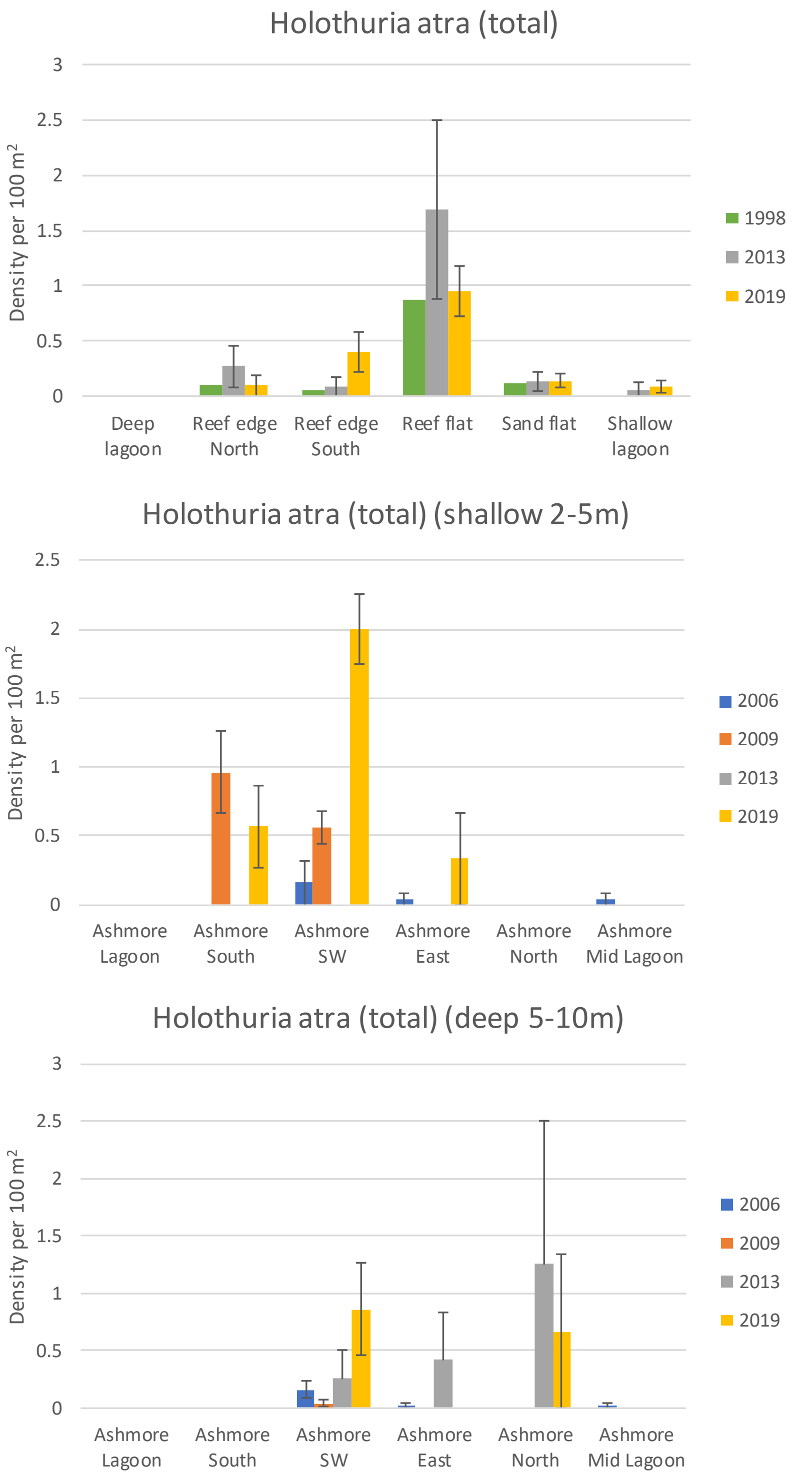


Figure 97. Comparisons of historical and recent surveys of *Holothuria atra* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are ±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^2 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^2 (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^2) and the sand flat (0.02/100 m^2) (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^2 on the reef flat and 0.05/100 m^2 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^2 on the reef flat to 0.05/100 m^2 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^2) 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.

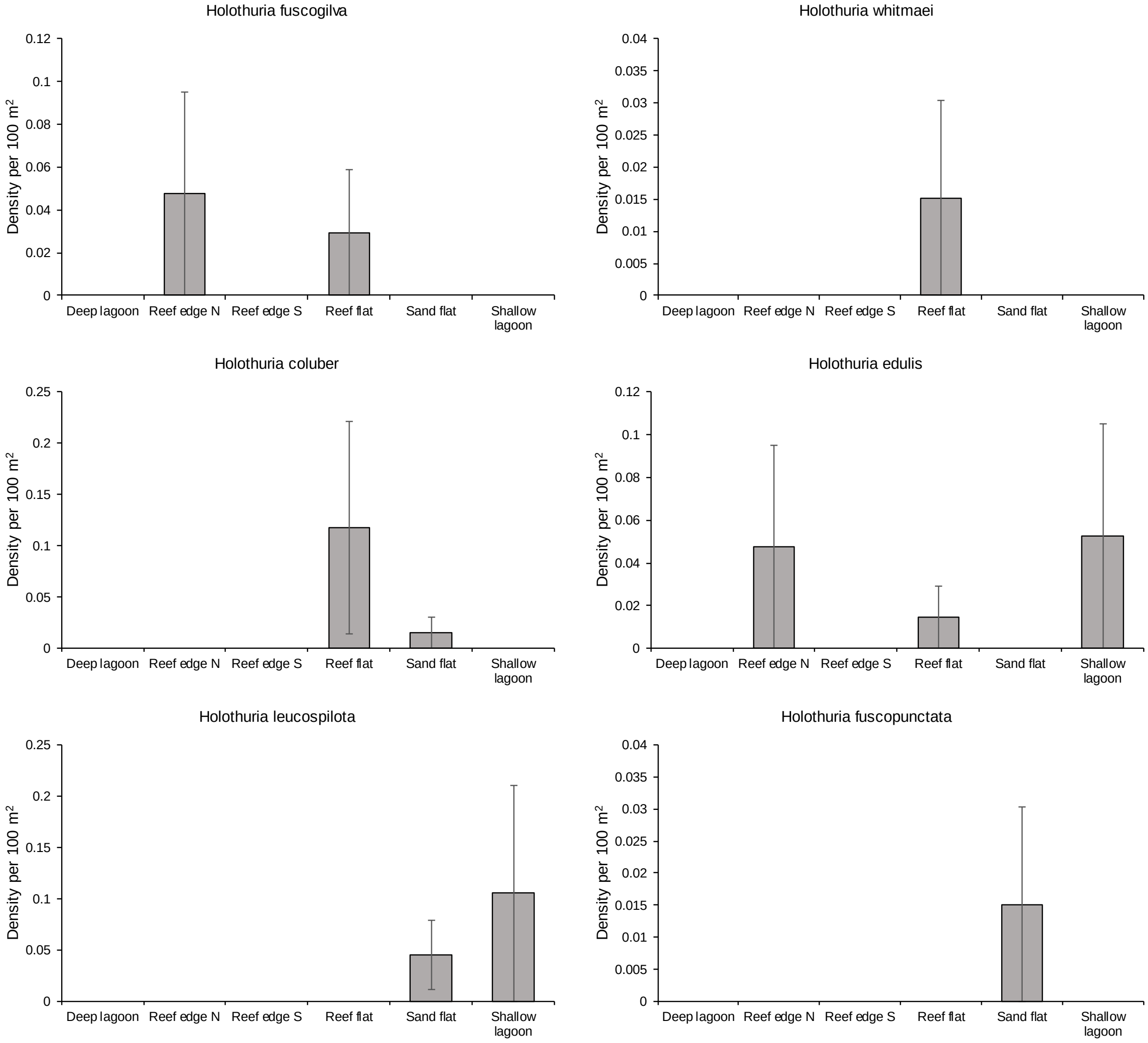


Figure 98. Mean density of holothurians (*Holothuria spp*.) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

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 inneffective 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.

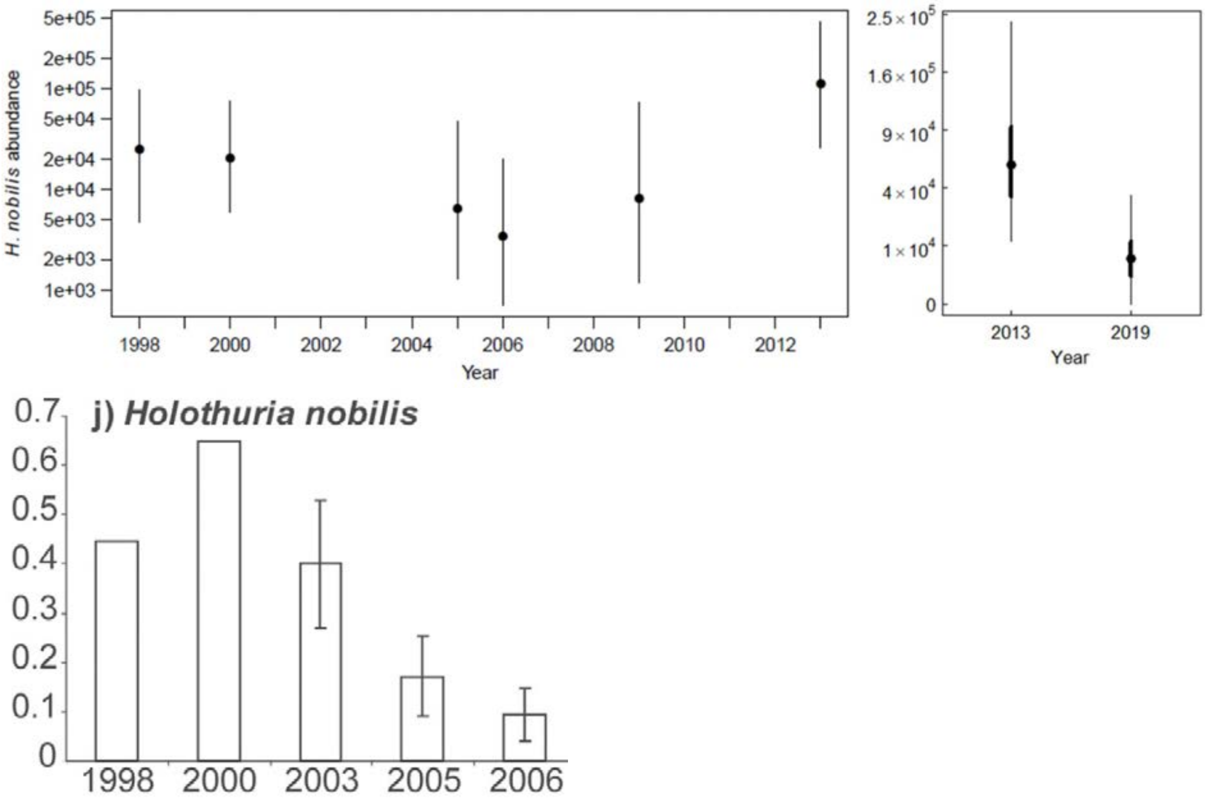


Figure 99. Long term estimates of abundance of *Holothuria whitmaei* (formerly *"nobilis"*). Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are 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 (±1 s.e.) densities for the whole of Ashmore Reef expressed per hectare. See these cited studies for methods.

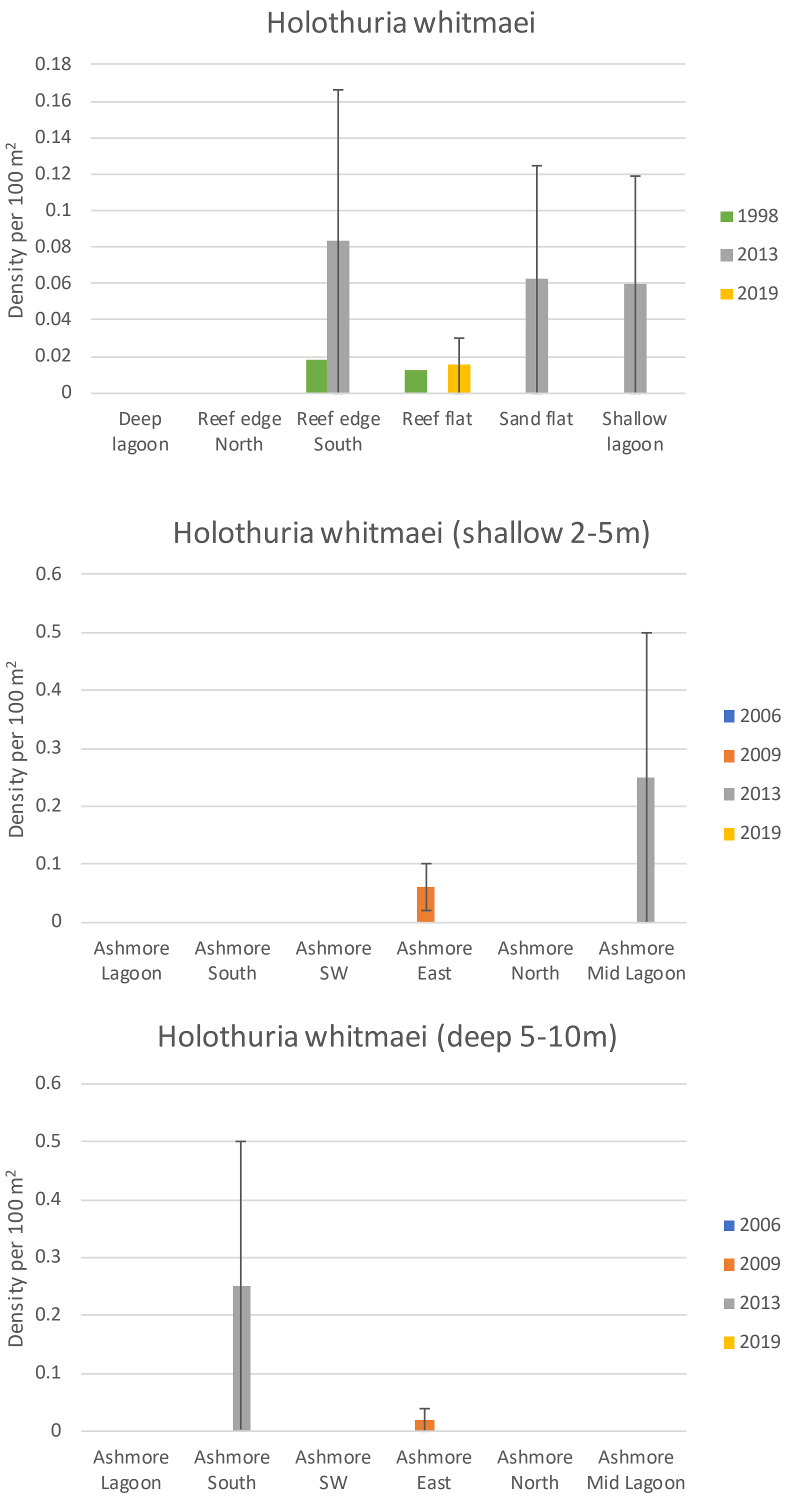


Figure 100. Comparisons of historical and recent surveys of *Holothuria whitmaei* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for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

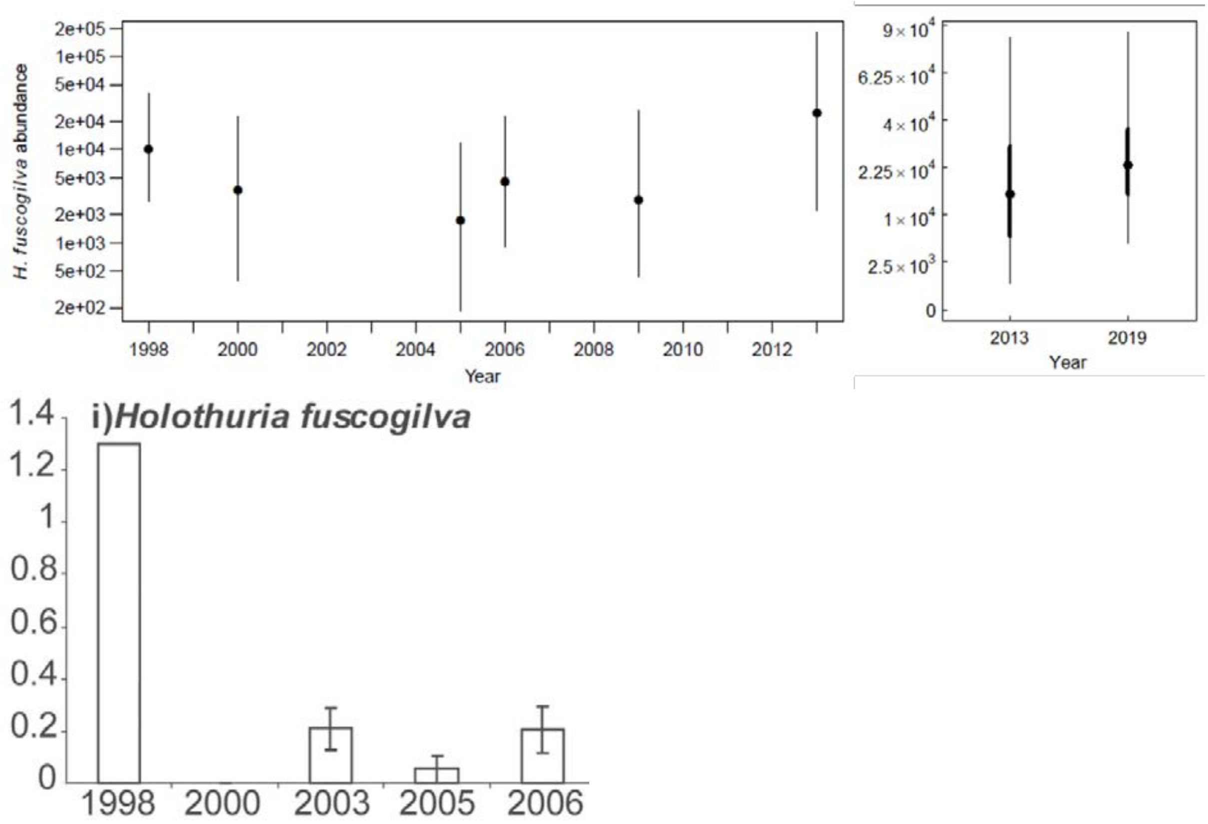


Figure 101. Long term estimates of abundance of *Holothuria fuscogilva*. Upper left panel is from Hosack and Lawrence (2013), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are not the same on both 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 (±1 s.e.) densities for the whole of Ashmore Reef expressed per hectare. See these cited studies for methods.

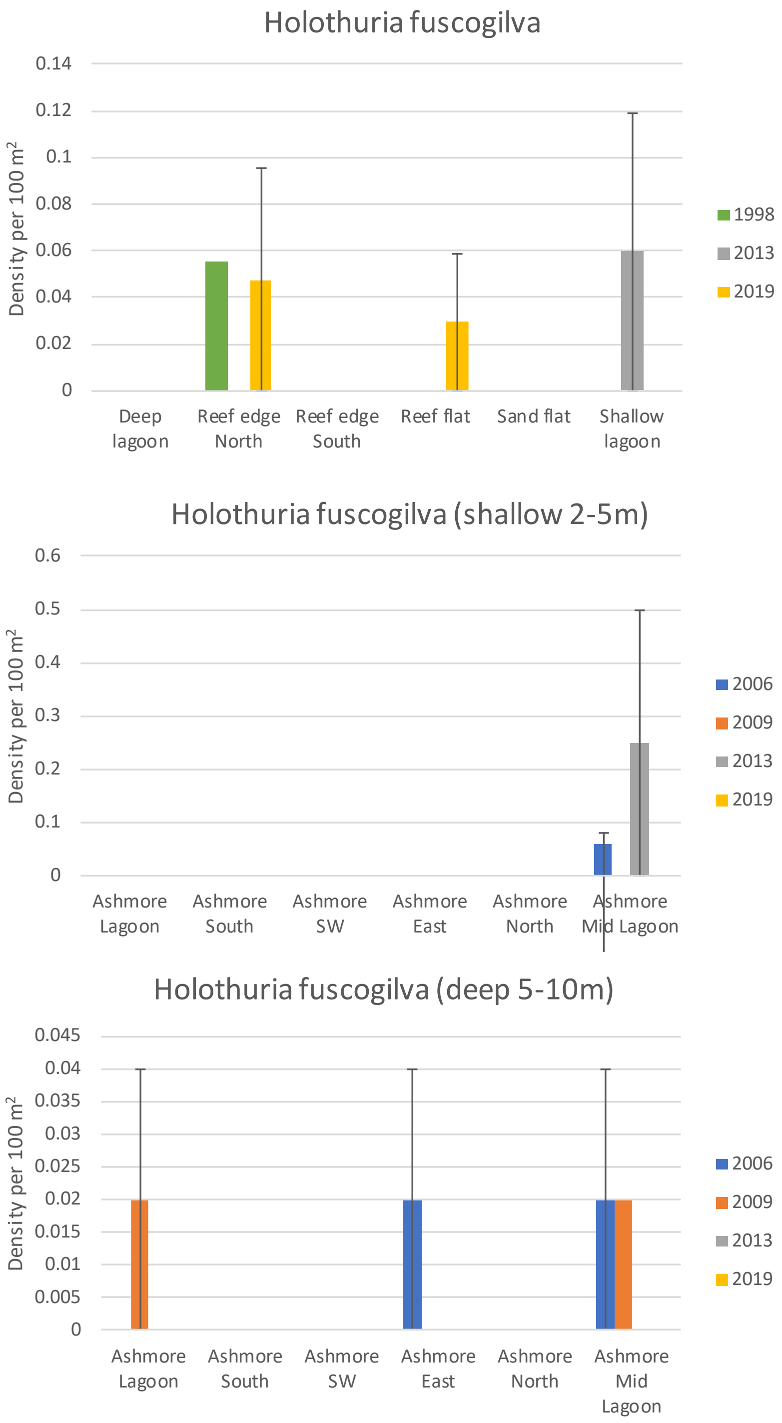


Figure 102. Comparisons of historical and recent surveys of *Holothuria fuscogilva* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for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

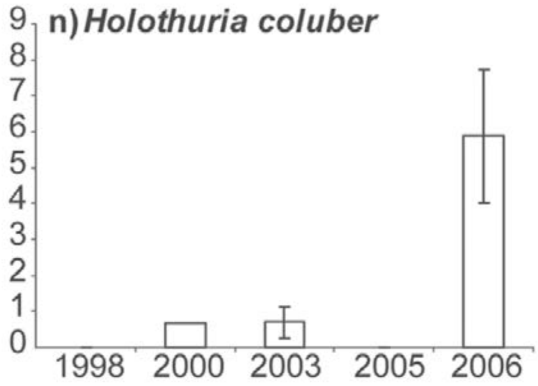


Figure 103. Long term estimates of abundance of *Holothuria coluber* at Ashmore Reef. Figure is from Ceccarelli et al. (2011a) with average or mean (±1 s.e.) densities for the whole of Ashmore Reef expressed per hectare. See cited study for methods.

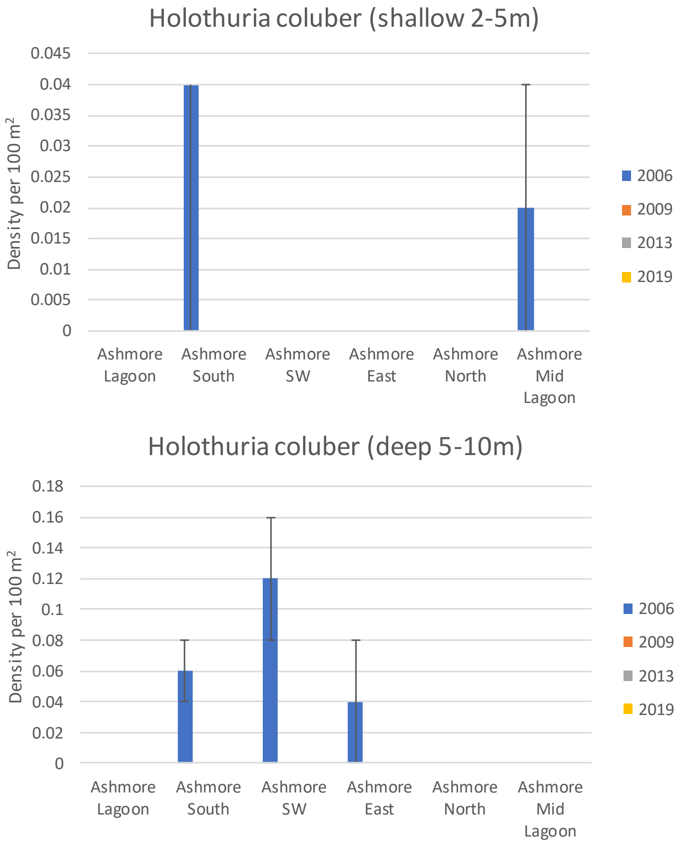


Figure 104. Comparisons of historical and recent surveys of *Holothuria coluber* at Ashmore Reef. Abundances are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are ±1 s.e.

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).

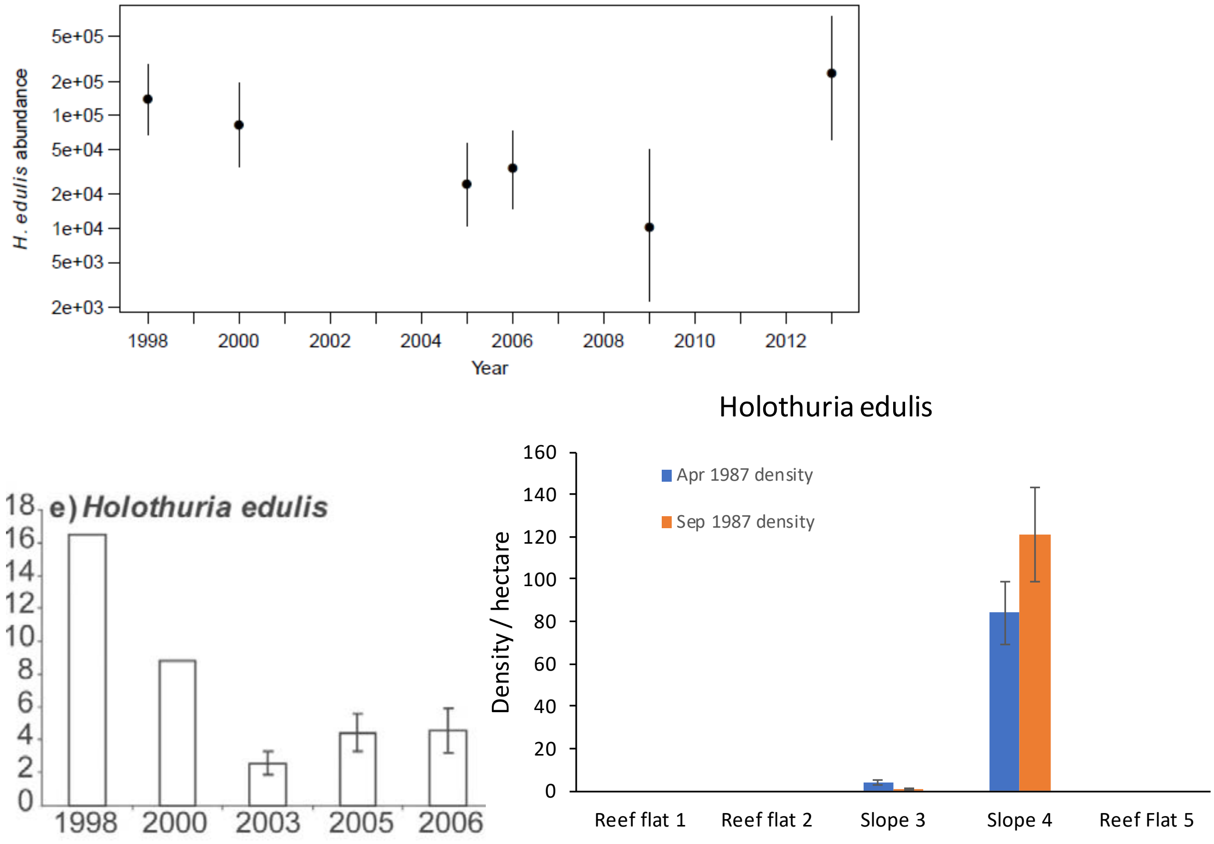


Figure 105. Long term estimates of abundance of *Holothuria edulis* at Ashmore Reef. Upper panel is from Hosack and Lawrence (2013a). These represent the total estimated population on Ashmore Reef. Values are medians and longest error bars are 95% confidence limits. Lower left panel is from Ceccarelli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

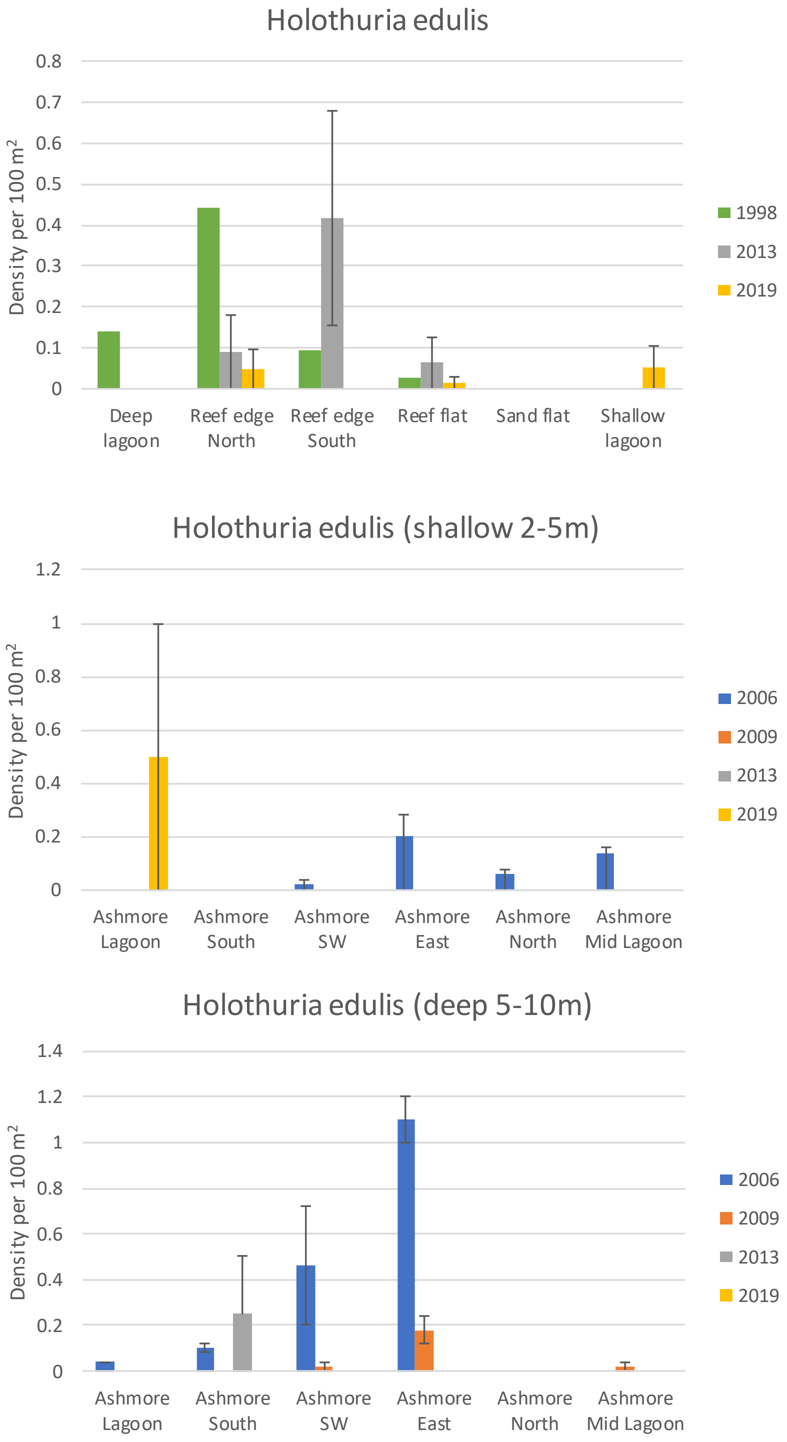


Figure 106. Comparisons of historical and recent surveys of *Holothuria edulis* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are ±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^2 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).

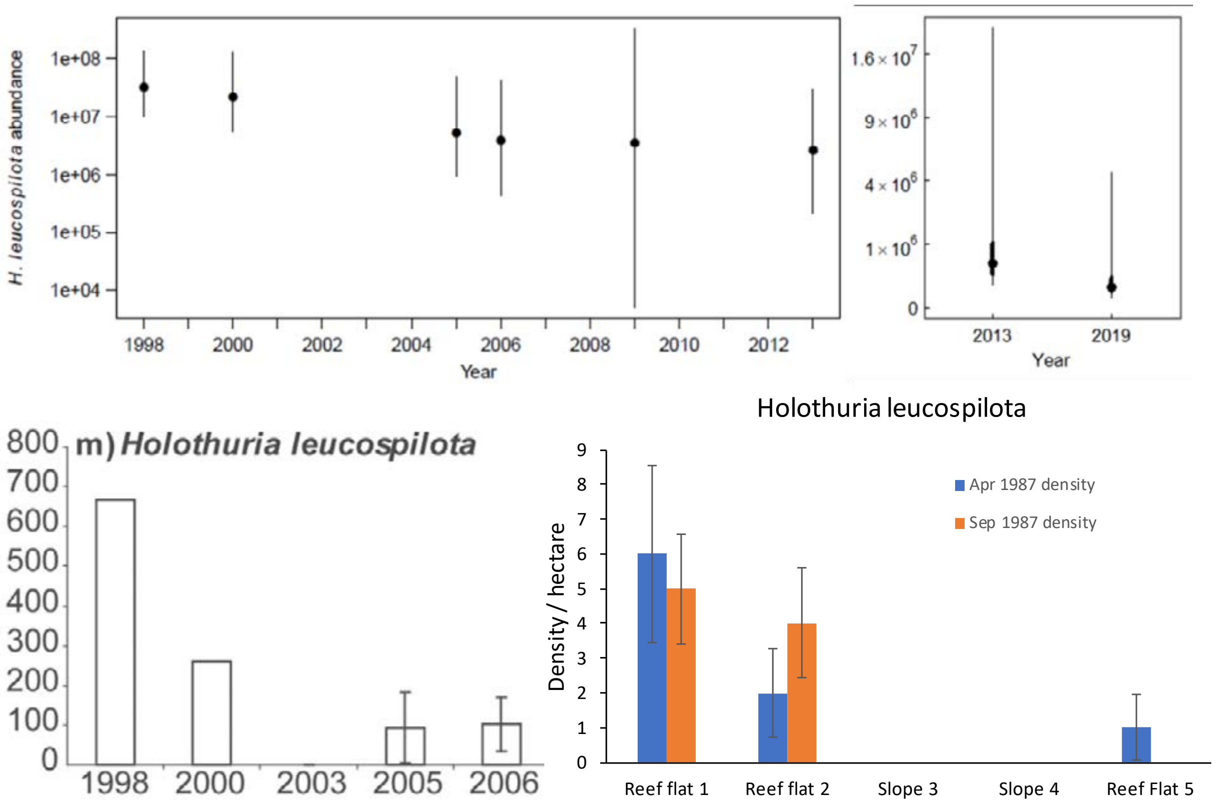


Figure 107. Long term estimates of abundance of *Holothuria leucospilota* at Ashmore Reef. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are 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 (±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 (±1 s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.

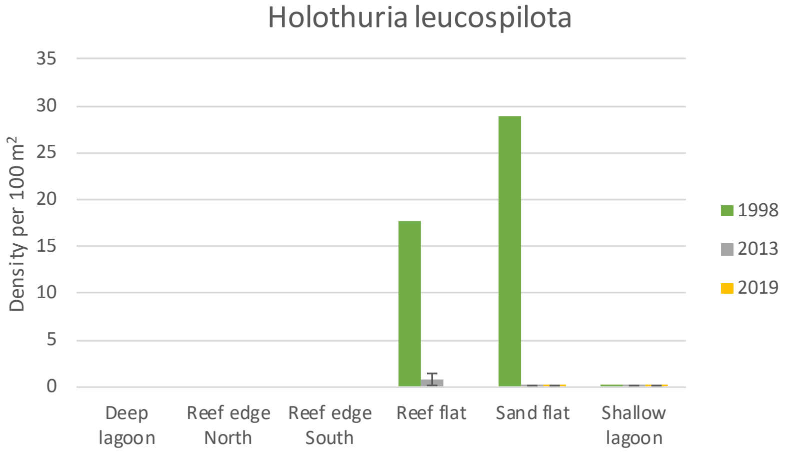


Figure 108. Comparisons of historical and recent surveys of *Holothuria leucospilota* at Ashmore Reef. Abundance is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Error bars are ±1 s.e.

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.

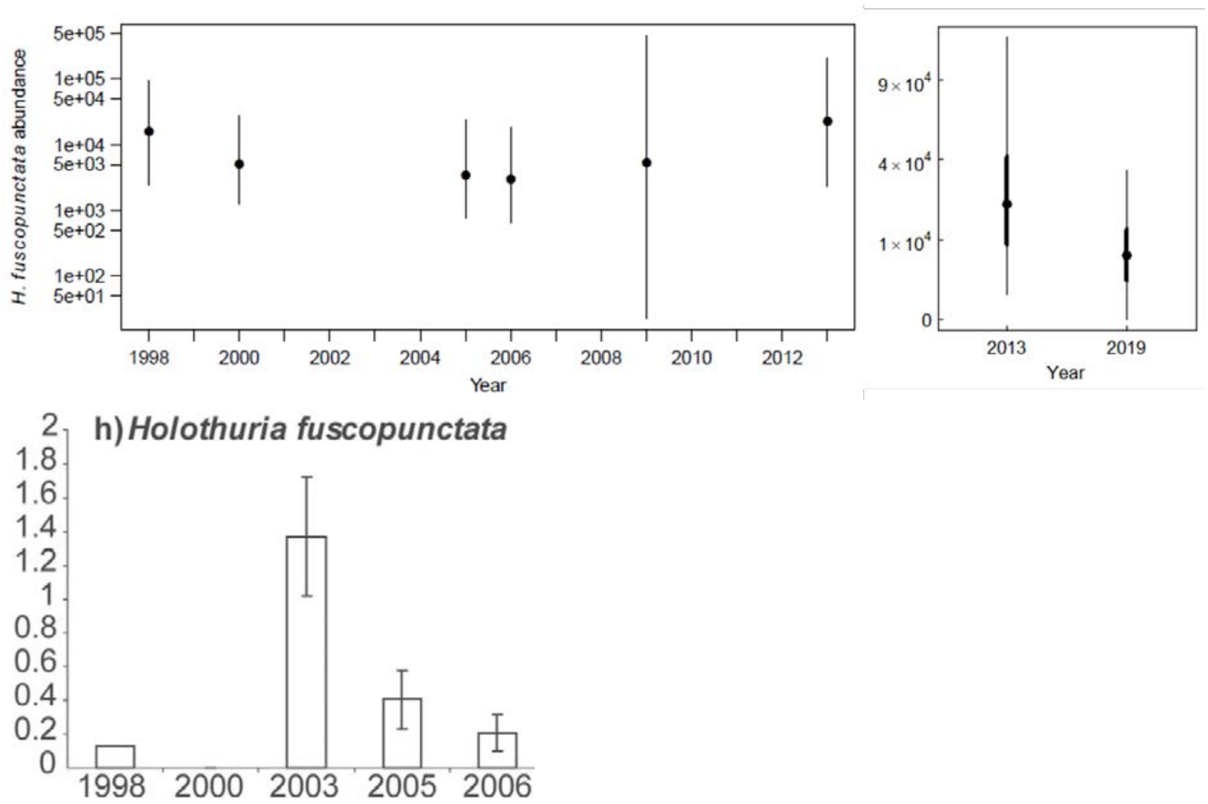


Figure 109. Long term estimates of abundance of *Holothuria fuscopunctata* at Ashmore Reef. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are 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 (±1 s.e.) densities for the whole of Ashmore Reef expressed per hectare.

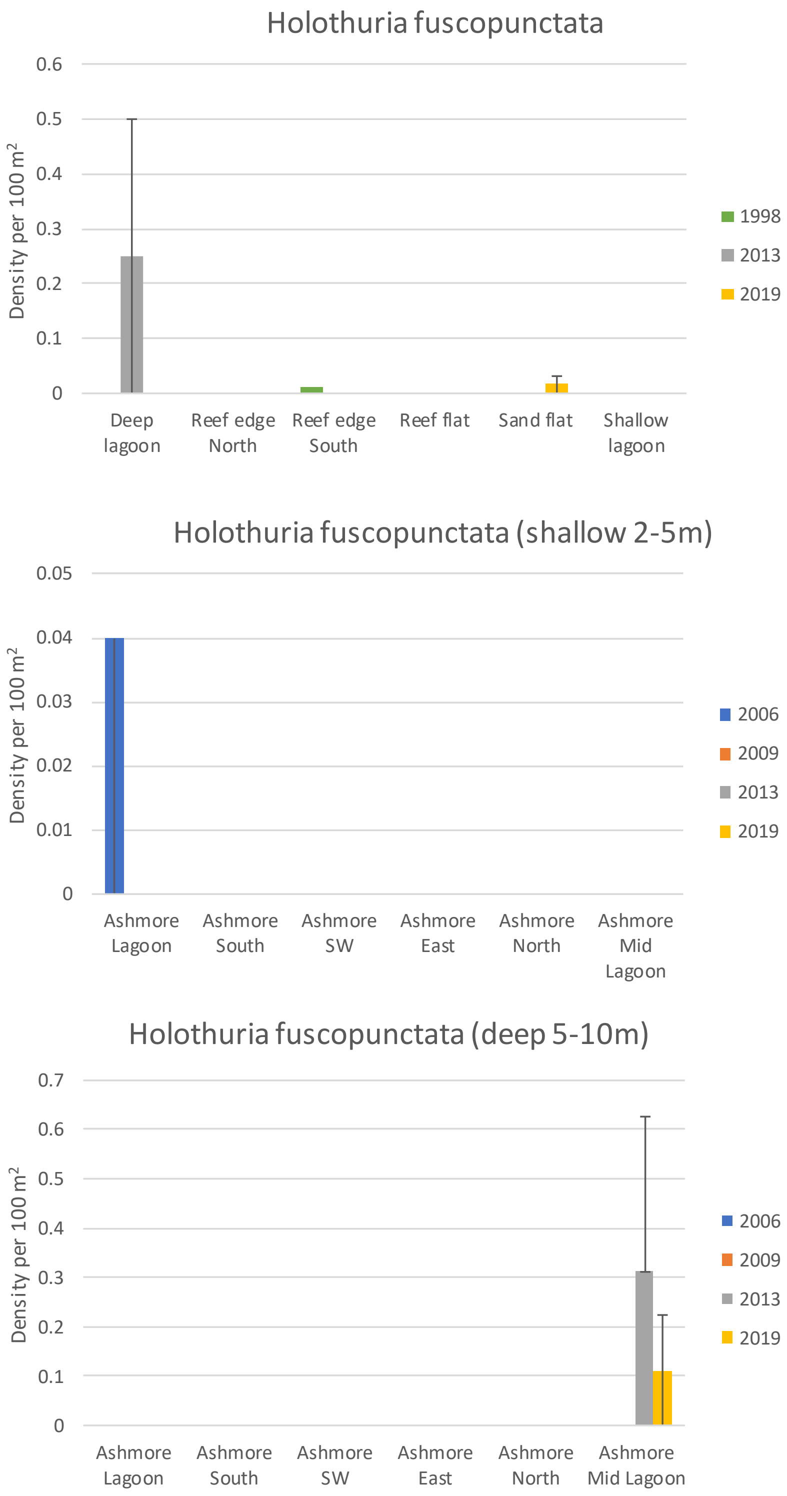


Figure 110. Comparisons of historical and recent surveys of *Holothuria fuscopunctata* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88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

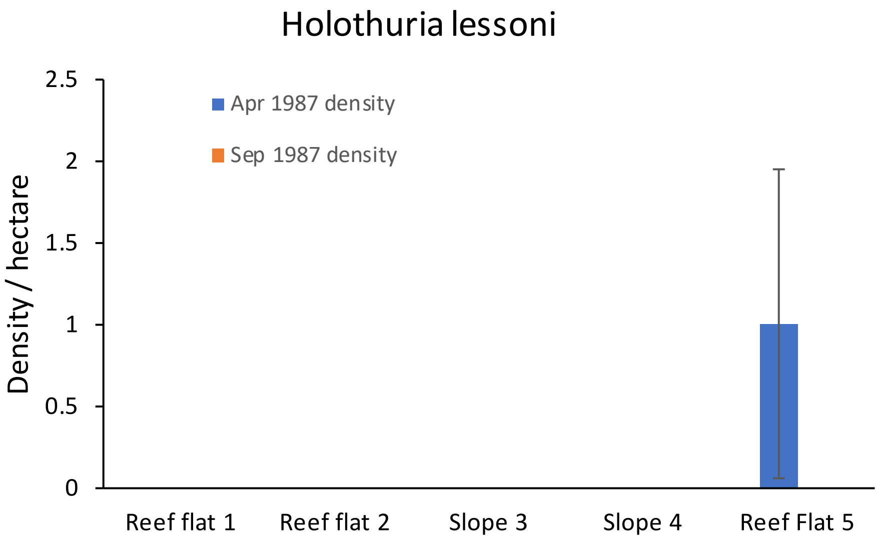


Figure 111. Mean density (±1 s.e.) of *Holothuria lessoni* at five sites surveyed at Ashmore Reef in 1987 by Russel and Vail (1988) See cited study for methods. Values for September 1987 are zeros.

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. *Actinopgya 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 m^2 in all habitats (Figure 113).

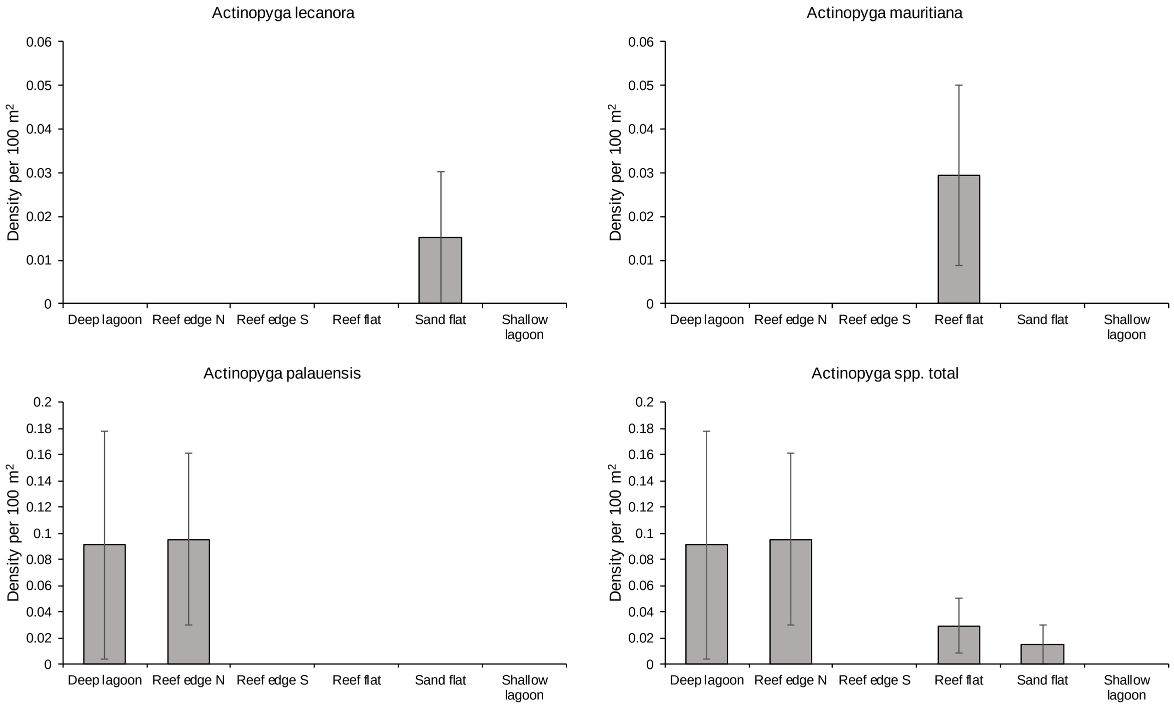


Figure 113. Mean density of holothurians (*Actinopyga* spp.) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

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.

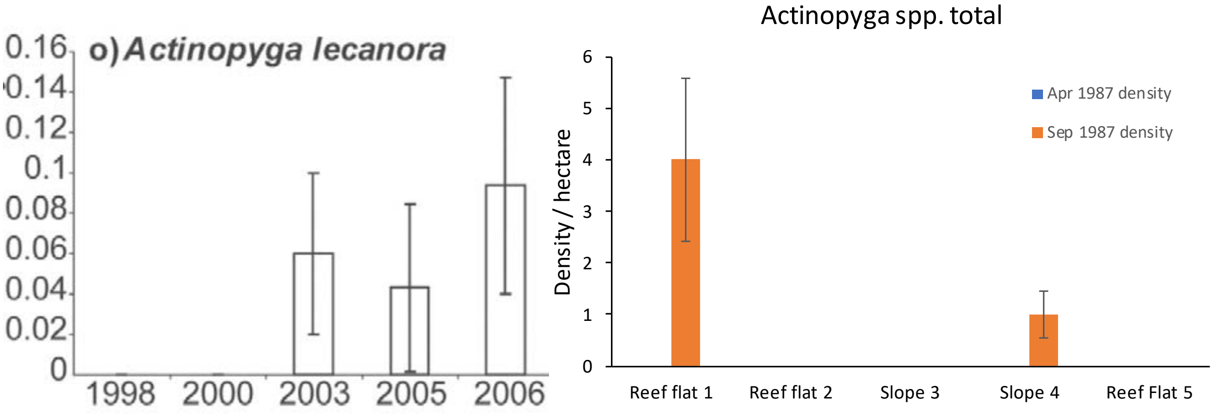


Figure 114. Long term estimates of abundance of *Actinopyga* spp at Ashmore Reef. Left figure is from Ceccarelli et al. (2011a) with mean densities for the whole of Ashmore Reef expressed per hectare (±1 s.e.). Right panel are mean densities (±1 s.e). from each of the small number of sites surveyed by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

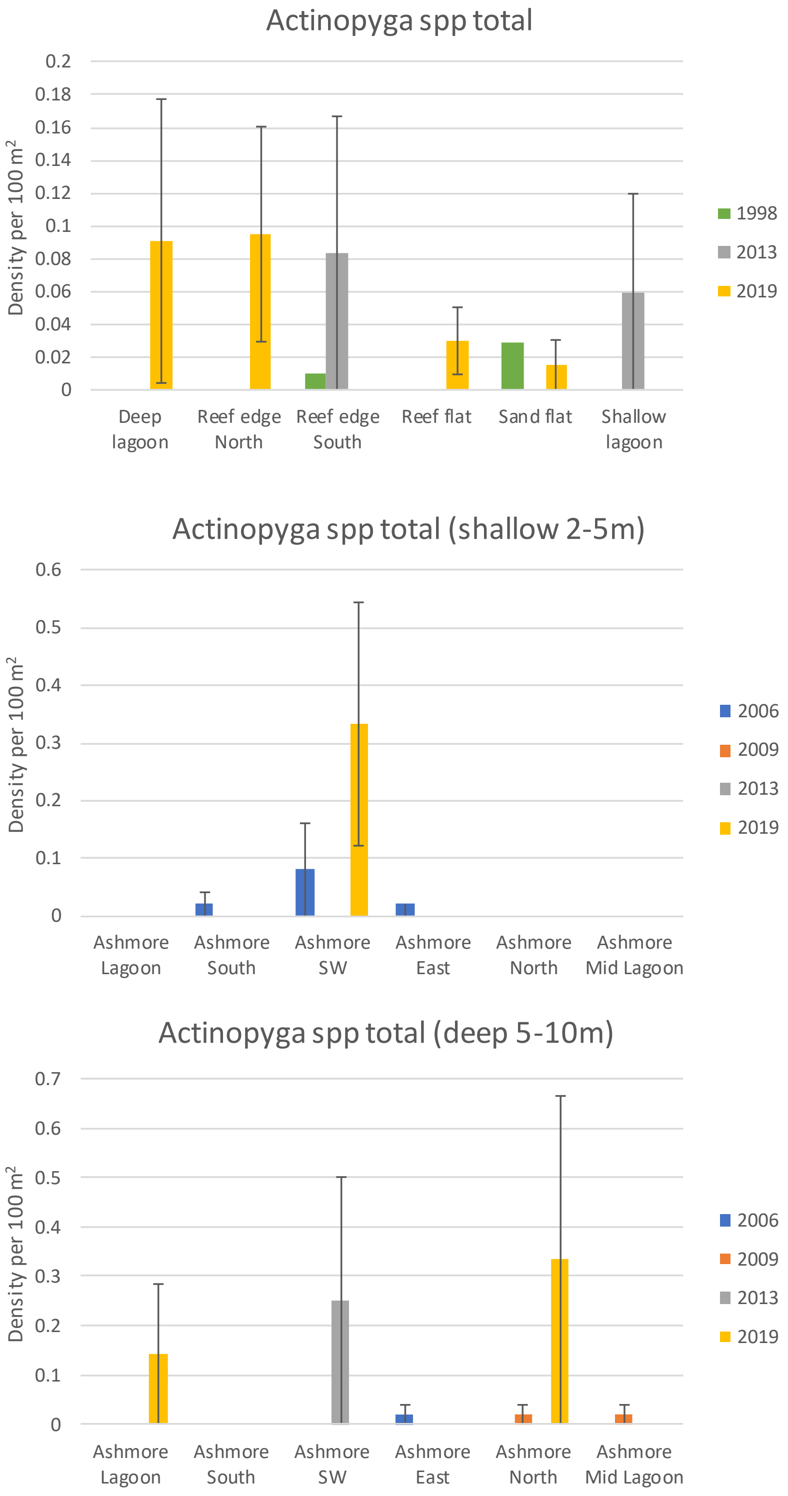


Figure 115. Comparisons of historical and recent surveys of *Actinopyga* spp. at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are ±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.

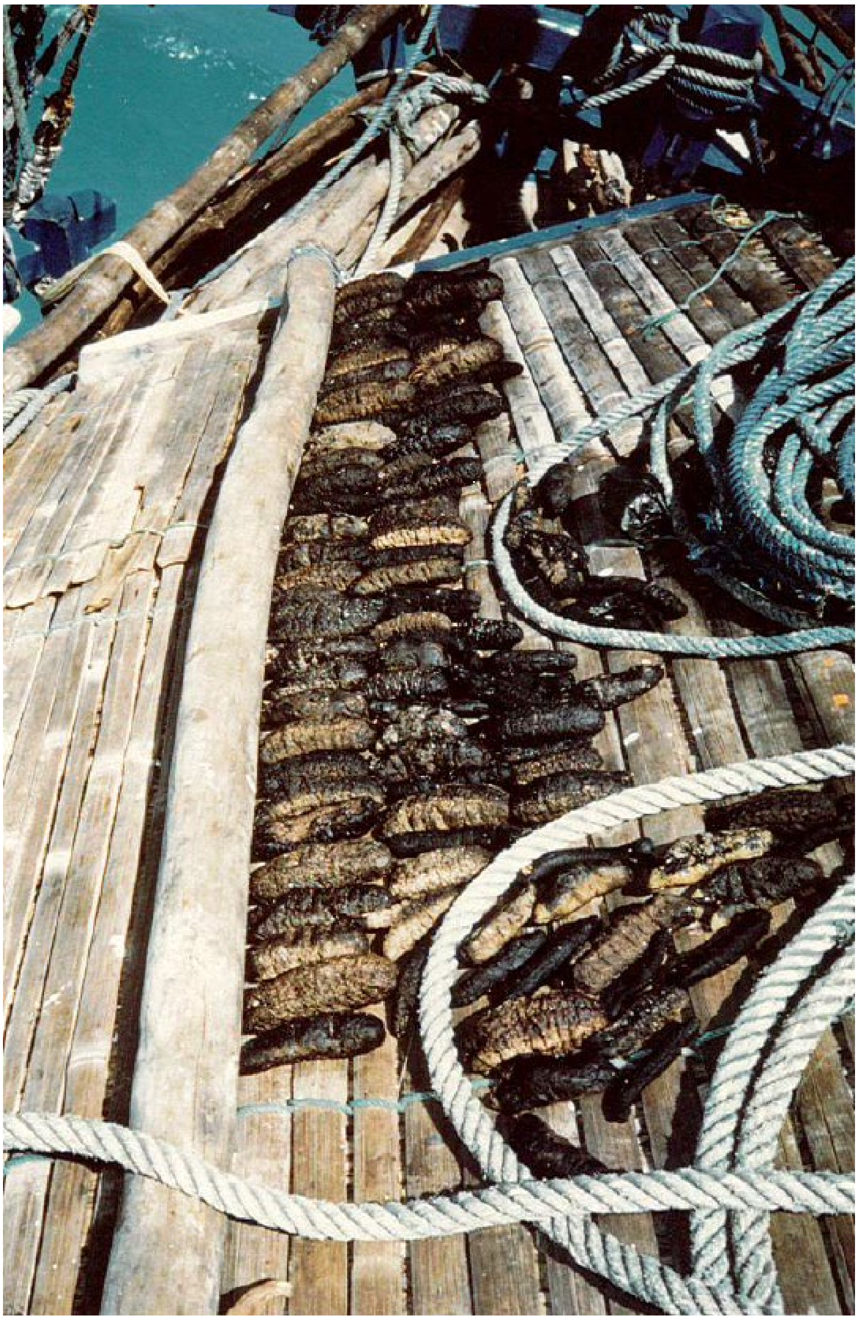


Figure 116. Plate 11 reproduced from Russell and Vail (1988) showing holothurians (mostly *Actinopyga* spp.) being dried on board an Indonesian perahu at Ashmore Reef in 1986/87. Photograph: Lyle Vail.

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^2, 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^2, Figure 117).

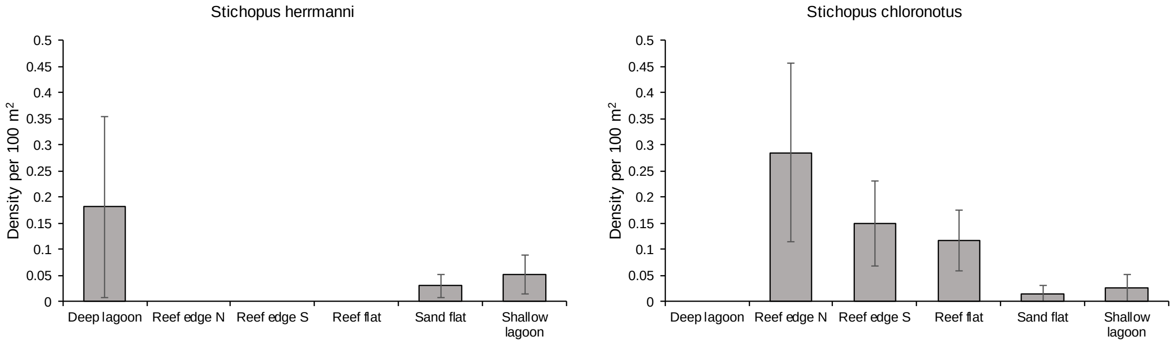


Figure 117. Mean density of holothurians (*Stichopus spp*.) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

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.

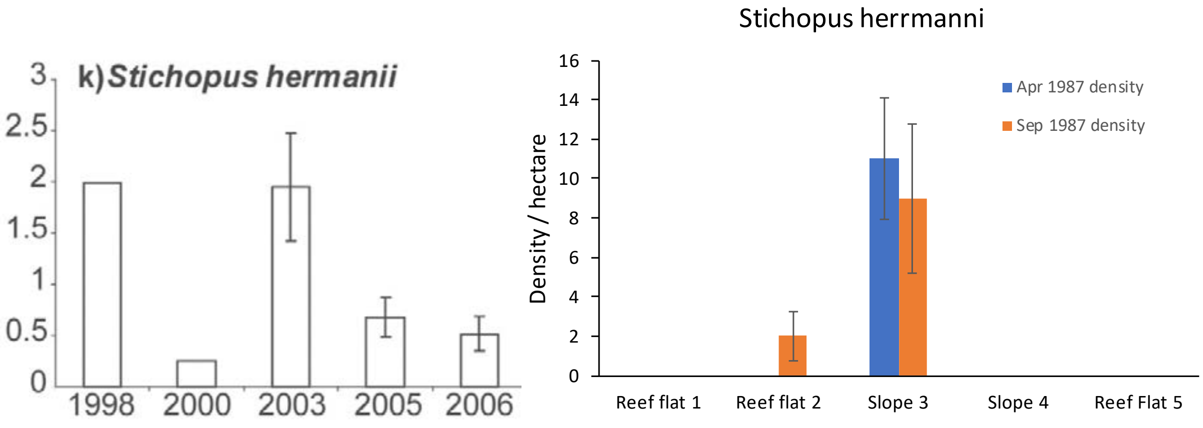


Figure 118. Long term estimates of abundance of *Stichopus herrmanni* at Ashmore Reef. Left figure is from Ceccarelli et al. (2011a) with mean densities for the whole of Ashmore Reef expressed per hectare (±1 s.e.). Right panel are mean densities (±1 s.e.) from each of the small number of sites surveyed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

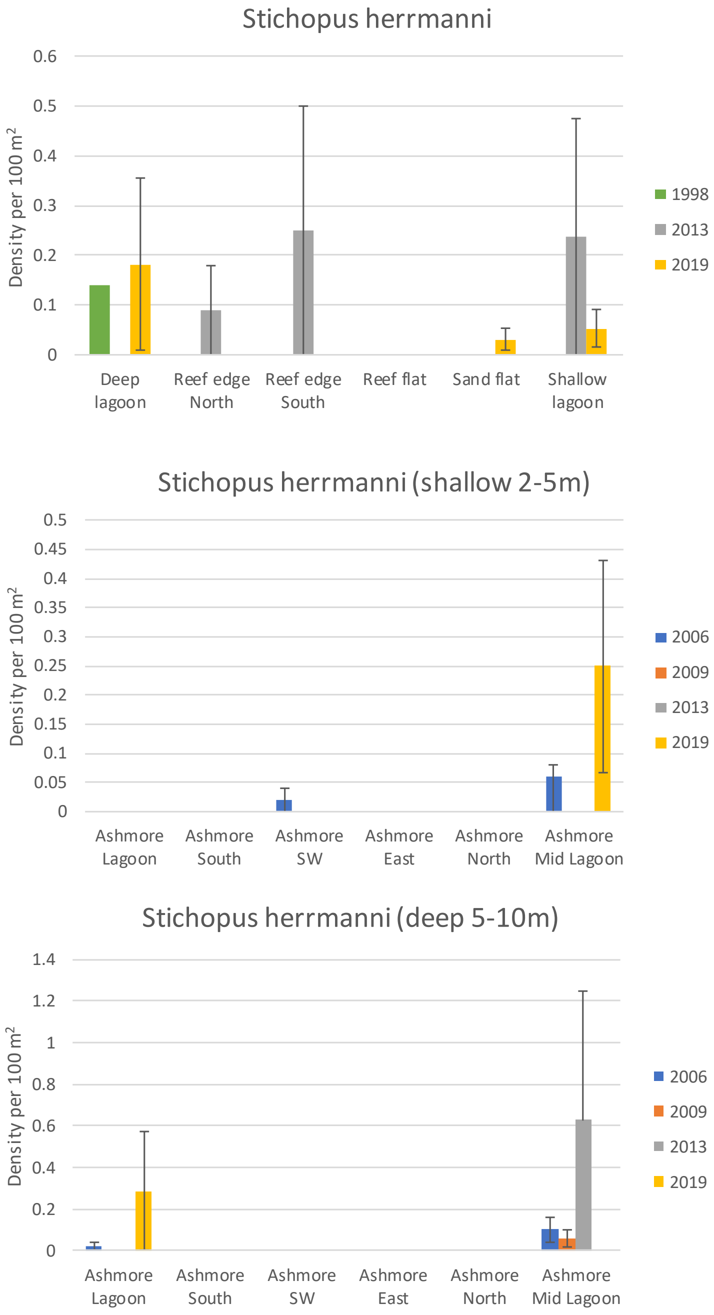


Figure 119. Comparisons of historical and recent surveys of *Stichopus herrmanni* (Herrmann's seas cucumber) at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are ±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. chloronatus* 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.

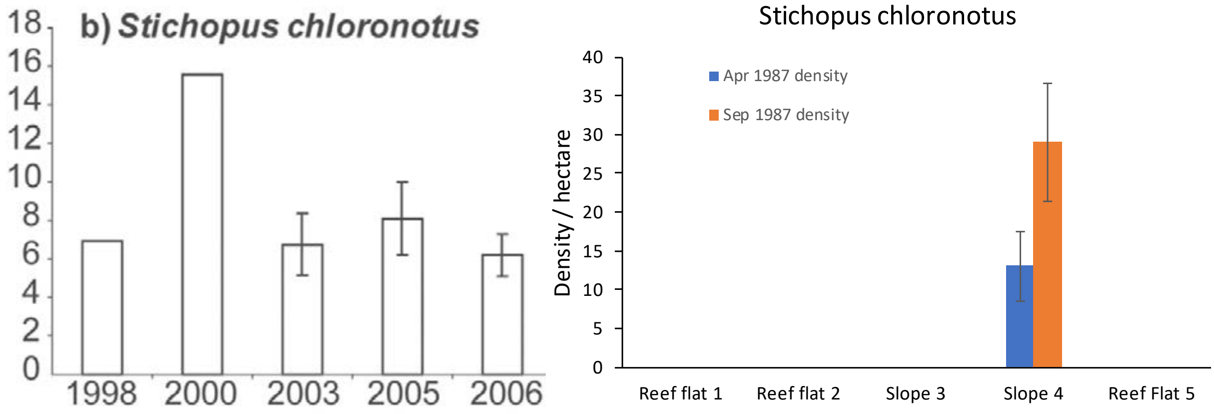


Figure 120. Long term estimates of abundance of *Stichopus chloronotus* at Ashmore Reef Left figure is from Ceccarelli et al. (2011a) with mean densities for the whole of Ashmore Reef expressed per hectare (±1 s.e.). Right panel are mean densities (±1 s.e.) from each of the small number of sites surveyed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

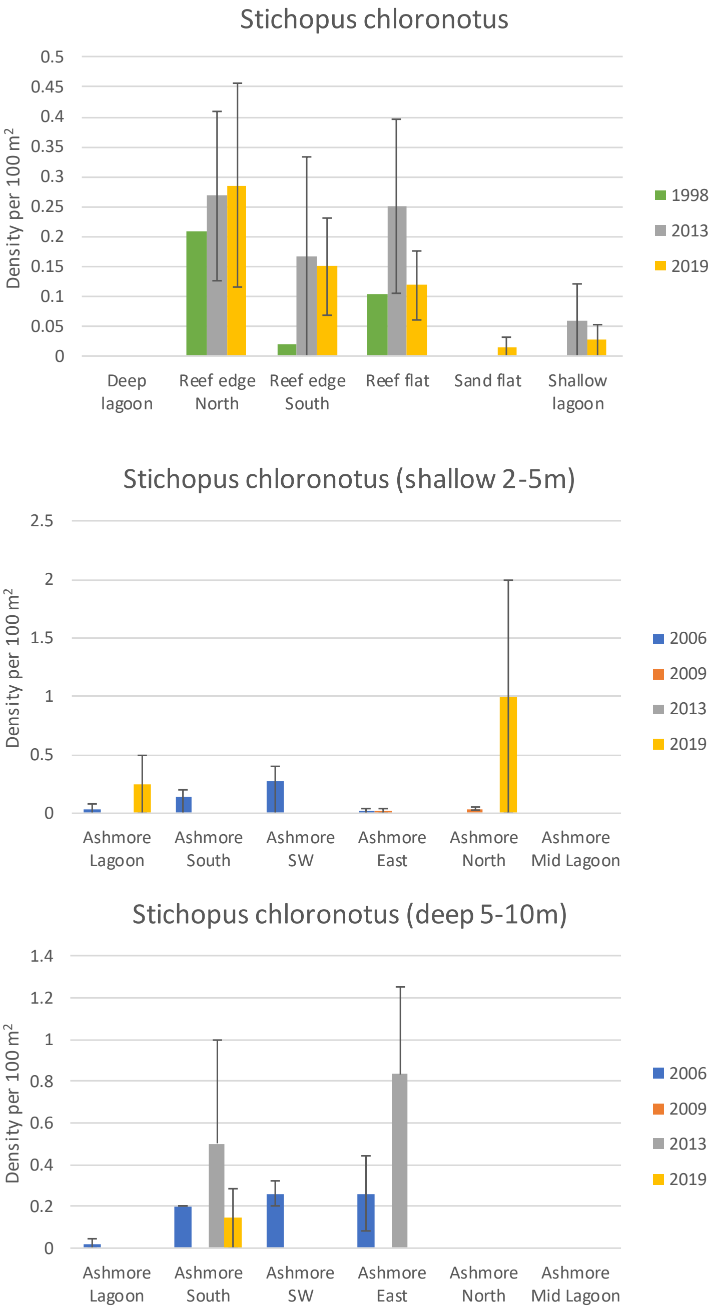


Figure 121. Comparisons of historical and recent surveys of *Stichopus chloronotus* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are ±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^2, Figure 122).

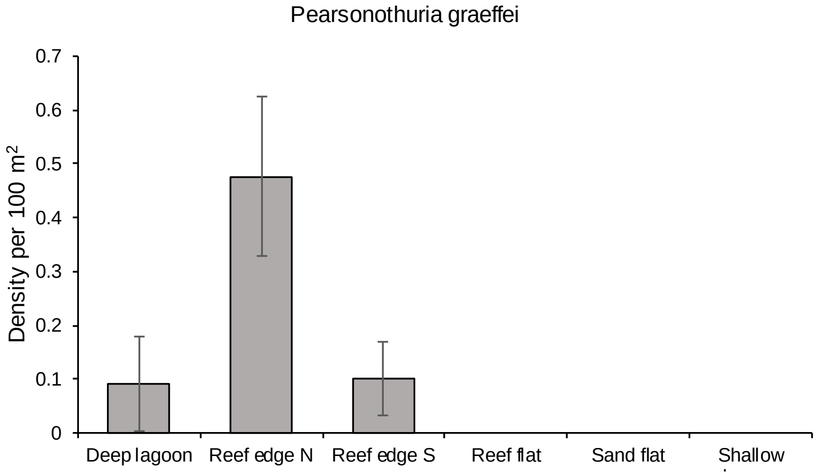


Figure 122. Mean density of the holothurian *Pearsonothuria graeffei* in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

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^2 = 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.

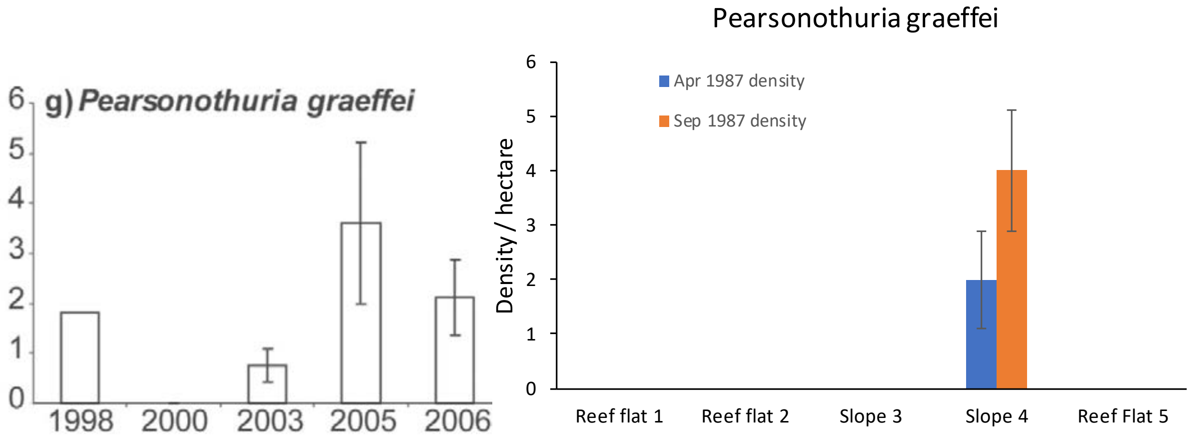


Figure 123. Long term estimates of abundance of *Pearsonothuria graeffei* at Ashmore Reef. Left figure is from Ceccarelli et al. (2011a) with mean densities for the whole of Ashmore Reef expressed per hectare (±1 s.e.). Right panel are mean densities (±1 s.e.) from each of the small number of sites surveyed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

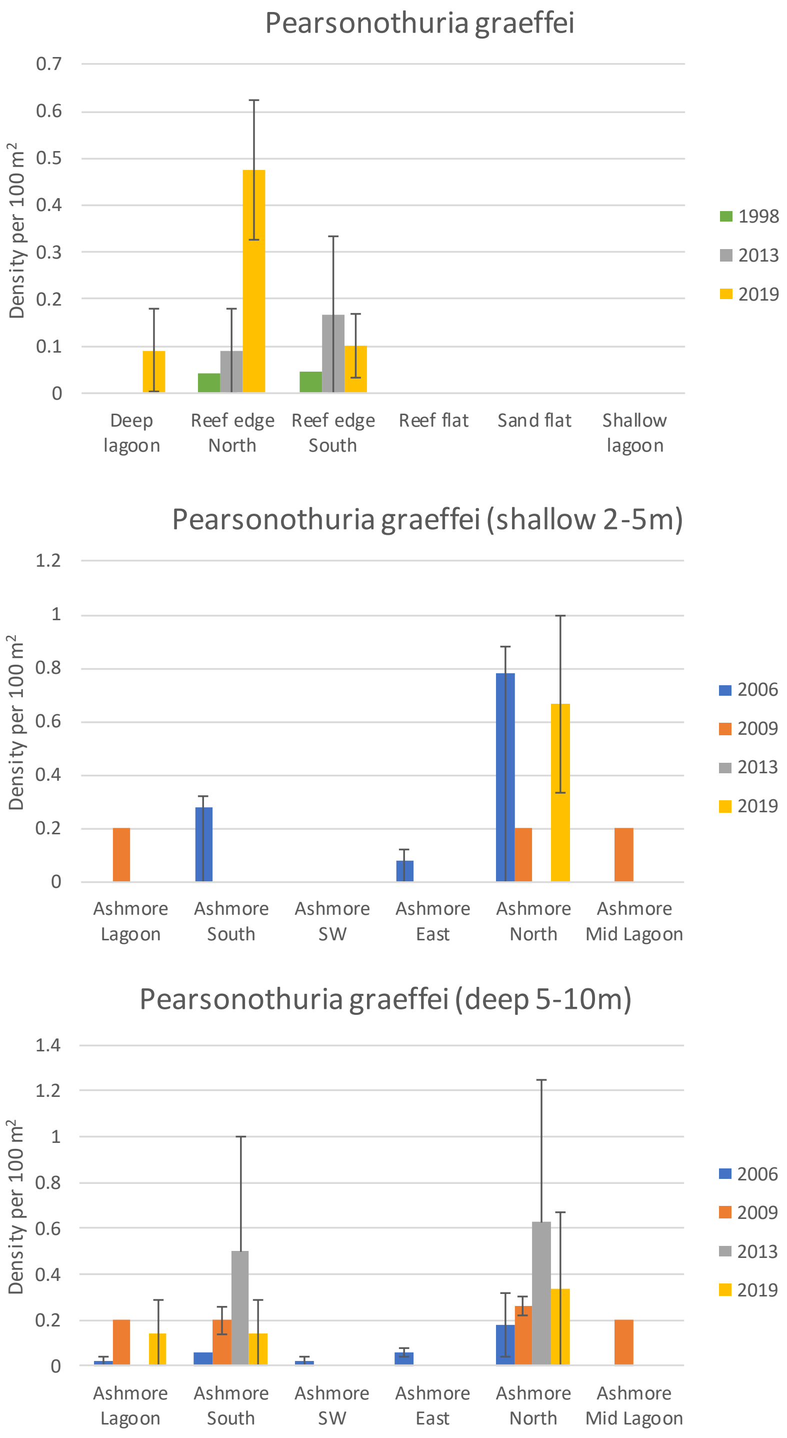


Figure 124. Comparisons of historical and recent surveys of *Pearsonothuria graeffei* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample sites) and compare the 2006, 2009, 2013 and 2019 surveys. Error bars are ±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.

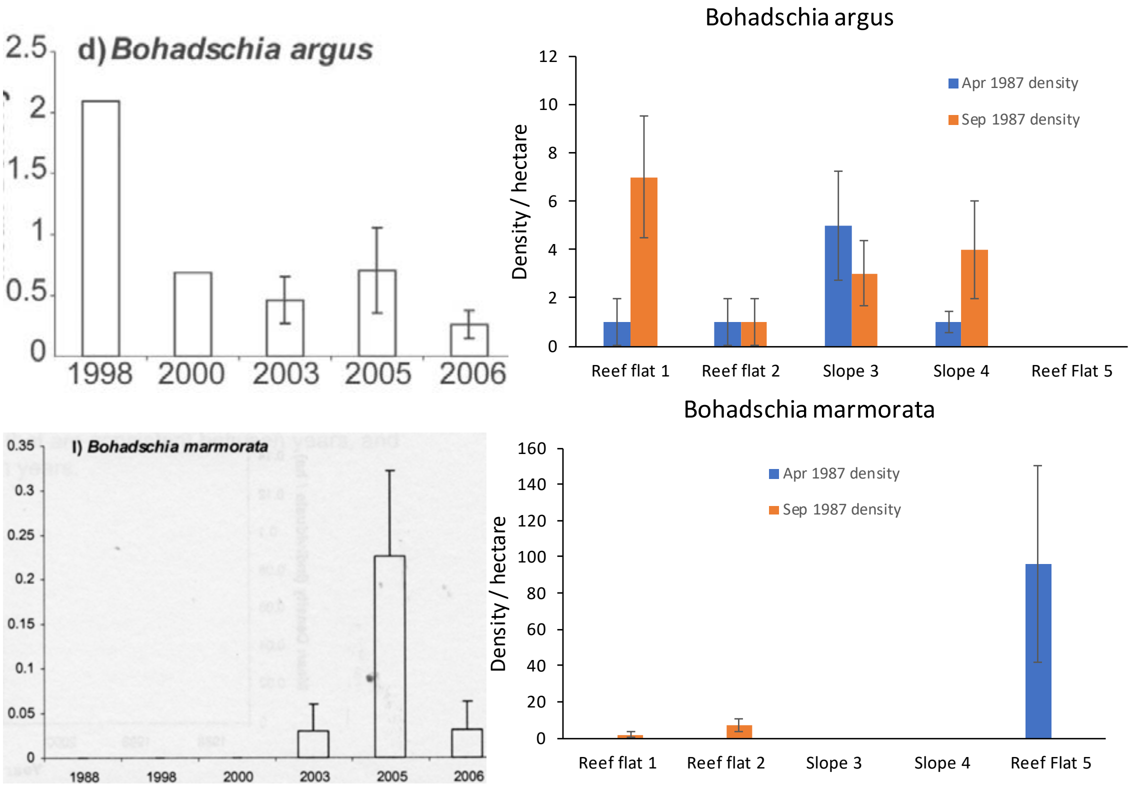


Figure 125. Long term estimates of abundance of *Bohadschia argus* and *Bohadschia marmorata* at Ashmore Reef. Upper left panel is from Ceccarelli et al. (2011a), lower left panel is from Ceccarelli et al. (2007) each with mean densities for the whole of Ashmore Reef expressed per hectare (±1 s.e.).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

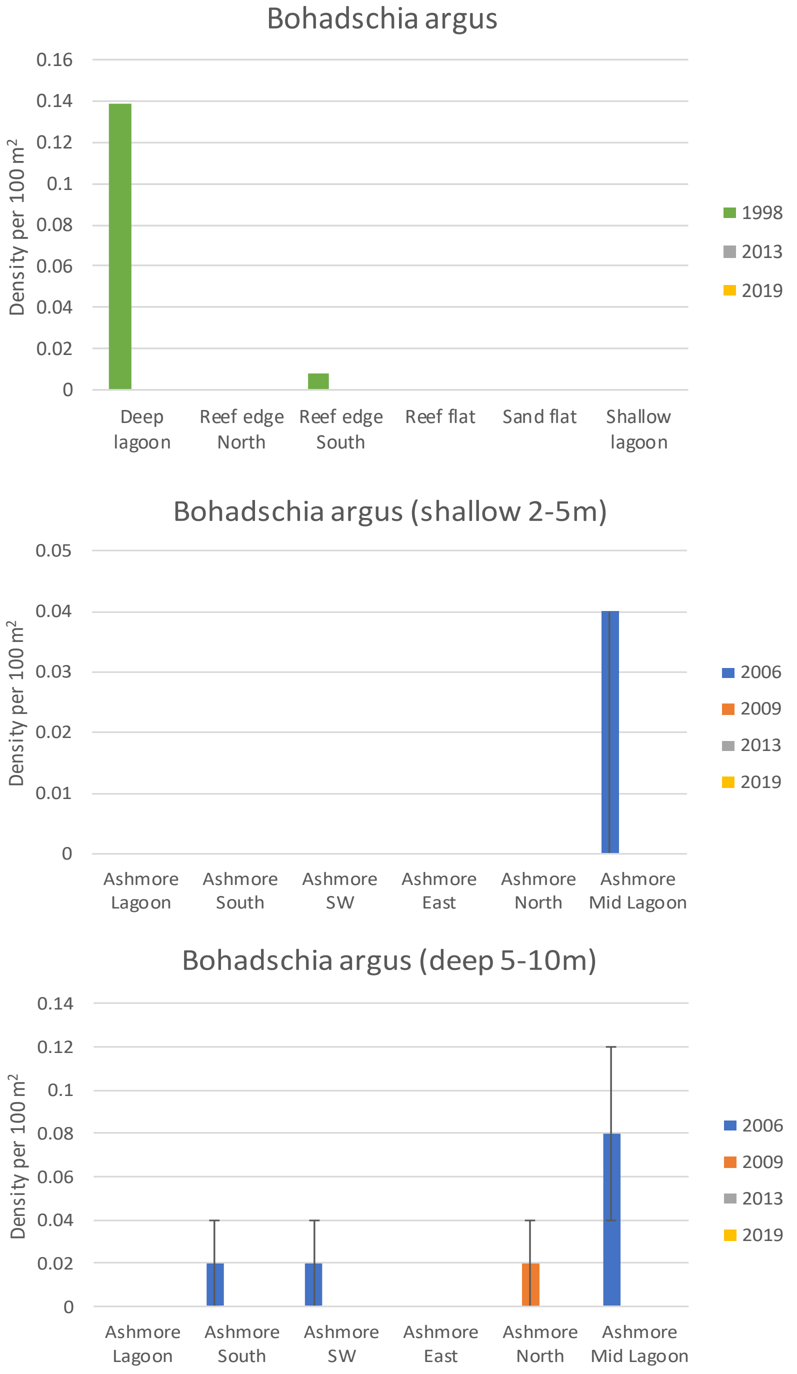


Figure 126. Comparisons of historical and recent surveys of *Bohadschia argus* at Ashmore Reef. Top panel is stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth (see Figure 88 for sample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

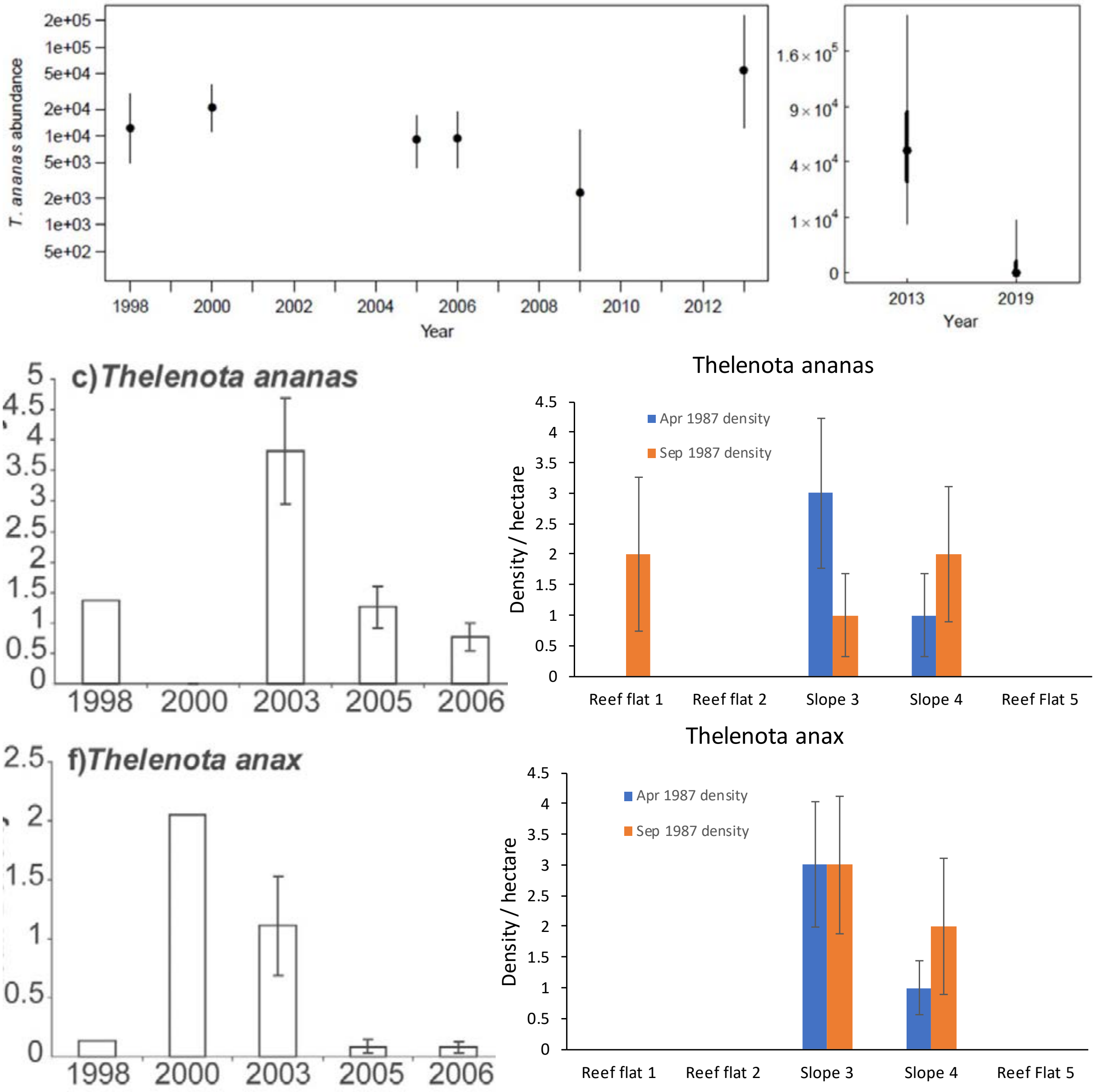


Figure 127. Long term estimates of abundance of *Thelenota ananas* and *Thelenota anax* at Ashmore Reef. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis 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 (±1 s.e.) expressed per hectare for the whole of Ashmore Reef. Middle and lower right panels are mean densities (±1 s.e.) from each of the small number of sites surveyed by Russell and Vail (1988). See cited studies for methods.

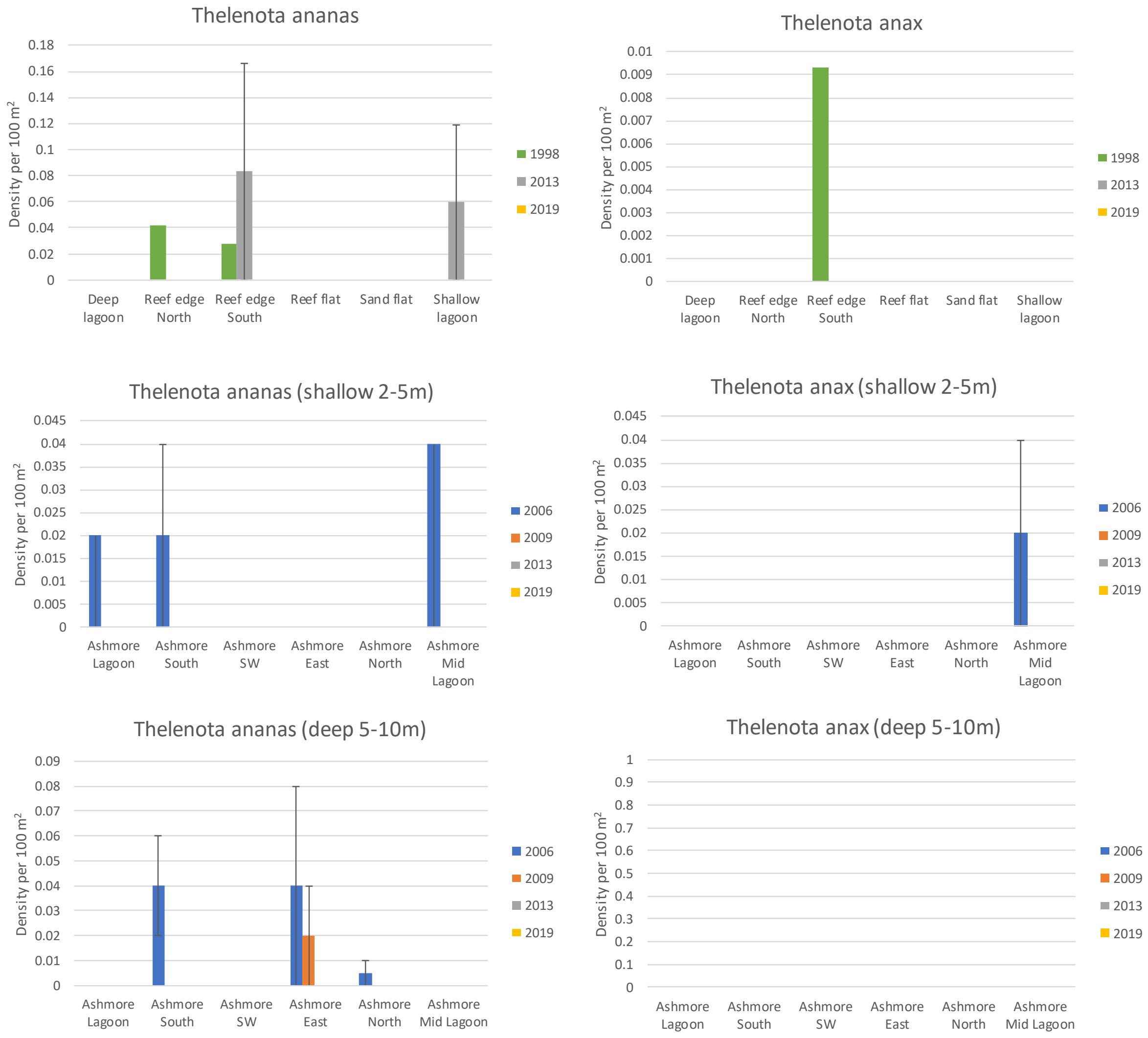


Figure 128. Comparisons of historical and recent surveys of *Thelenota ananas* and *Thelenota anax* at Ashmore Reef. Top panels are stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Bottom two panels are stratified by location on reef and depth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

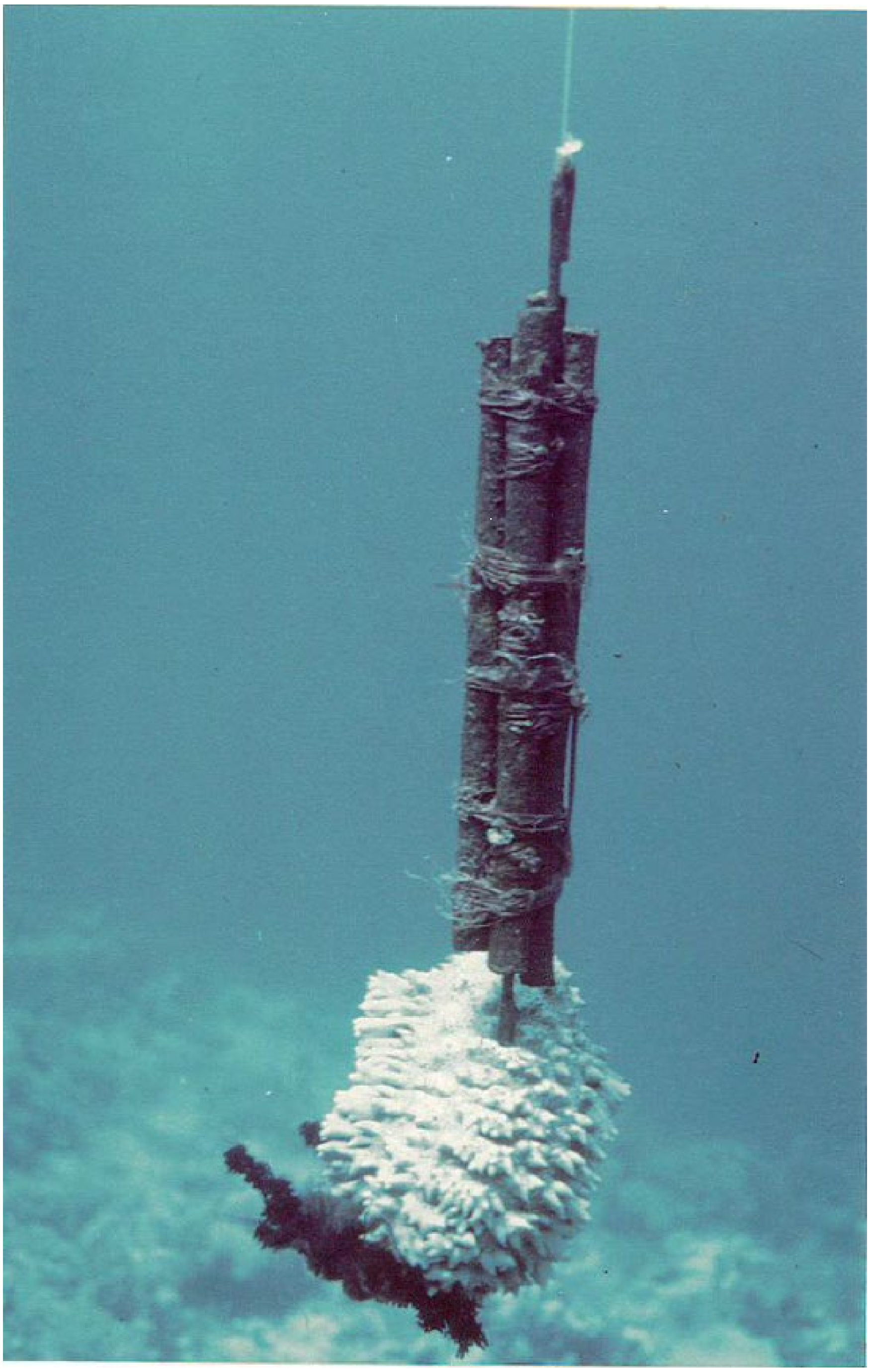


Figure 129. Weighted spear called a ladong used by Indonesian fishers at Ashmore Reef to take deep-water holothurians, in this case *Thelenota ananas*. Reproduced from plate 14 in Russell and Vail (1987). Photograph Lyle Vail.

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^2 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^2) (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^2 to 1.6 – 2.1/100 m^2 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.

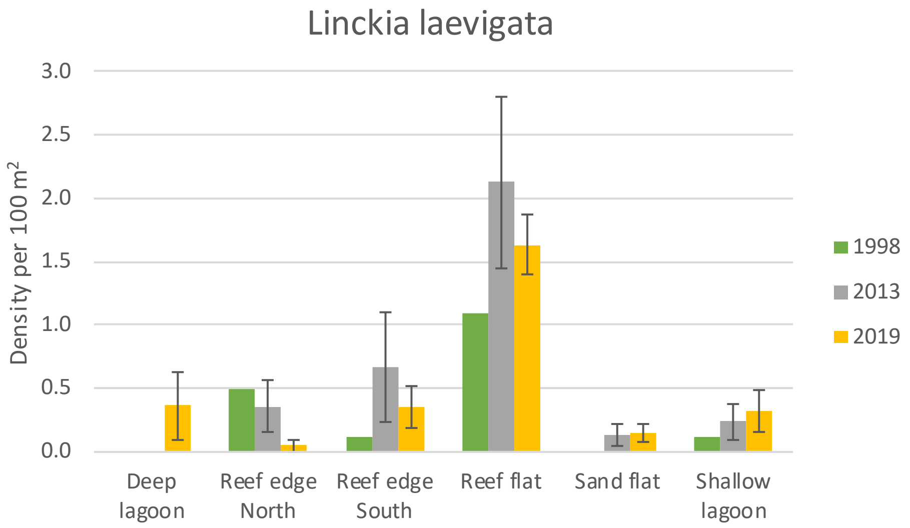


Figure 130. Comparisons of historical and recent surveys of *Linckia laevigata* at Ashmore Reef. Abundances are stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Error bars are ±1 s.e.

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^2 and the latter on both the northern and southern reef edges at densities of 0.25 – 0.3/100 m^2 (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^2 (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).

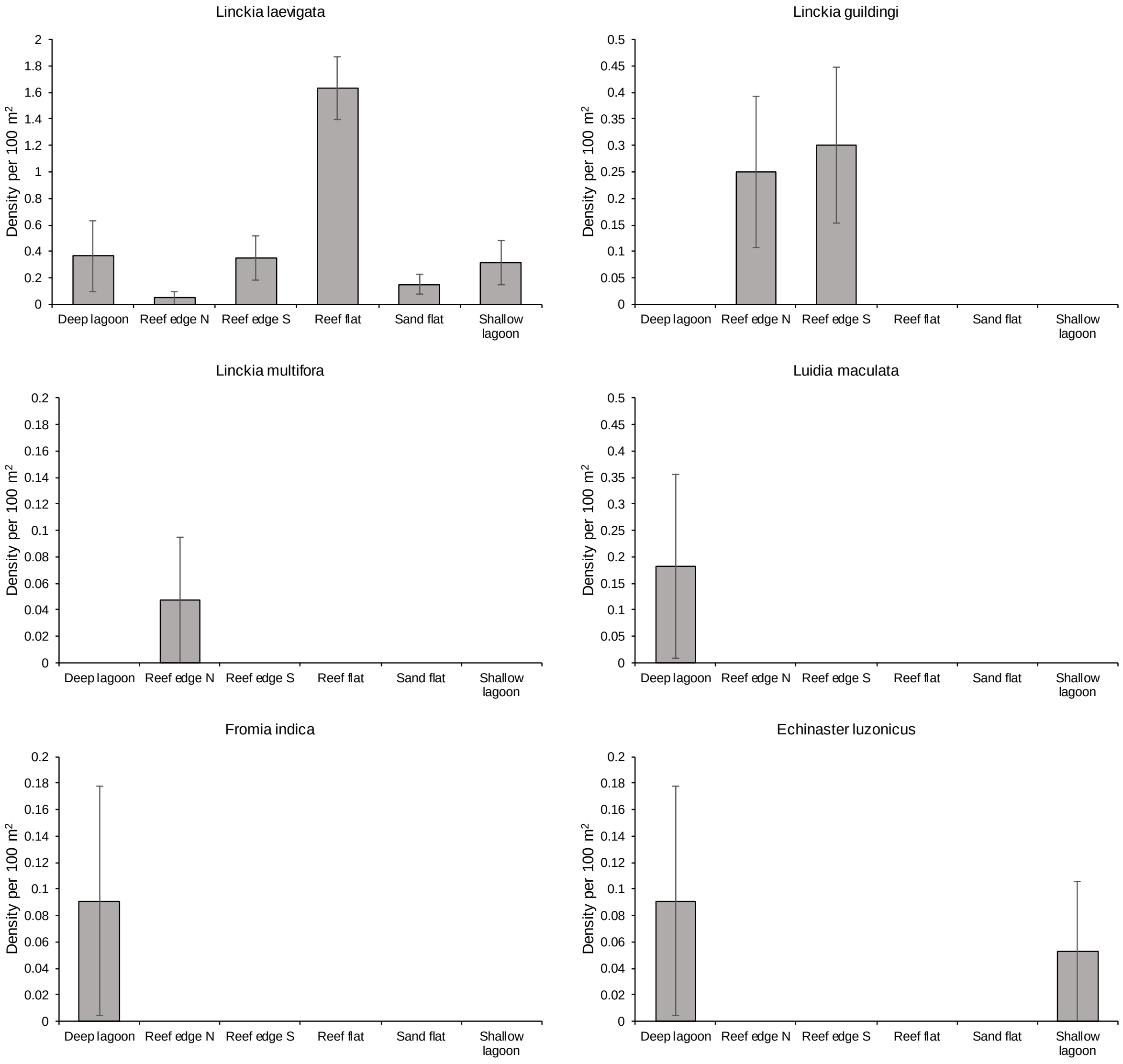


Figure 131. Mean density of starfish in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

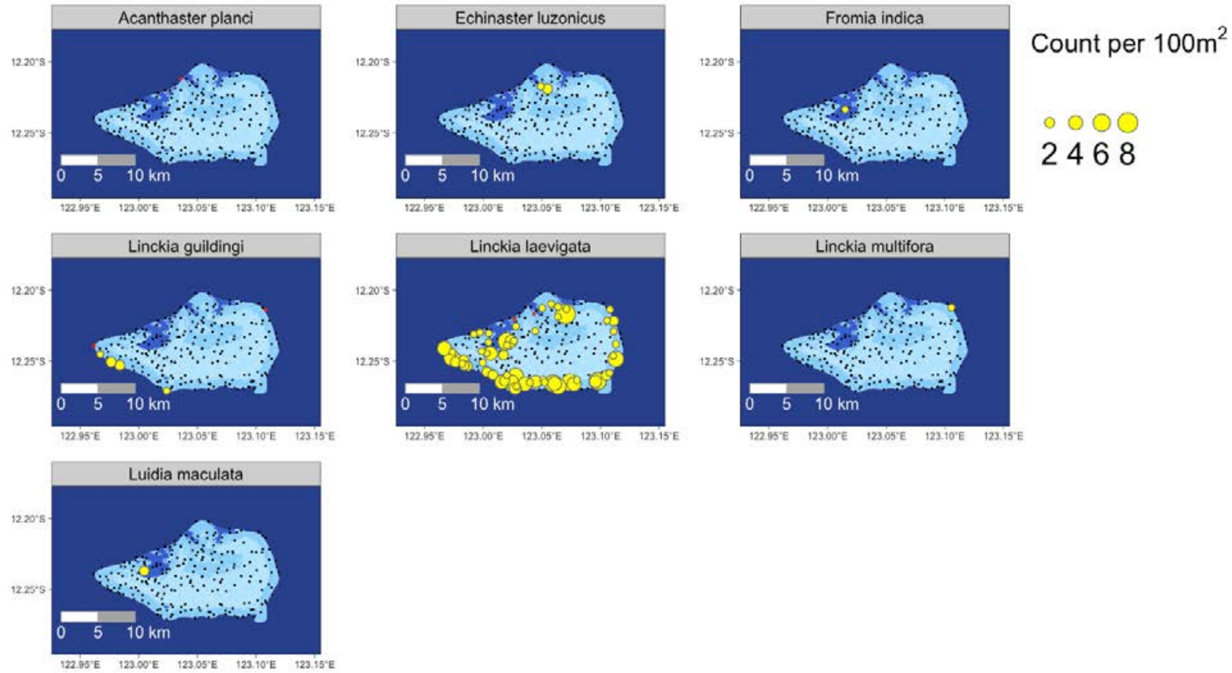


Figure 132. Distribution and density/100 m^2 of all starfish species across each of the 224 transects surveyed at Ashmore Reef in June 2019. The red dots indicate presence of that species at that site but that the specimen(s) were not within the area of the 100 m^2 quantitative survey transect.

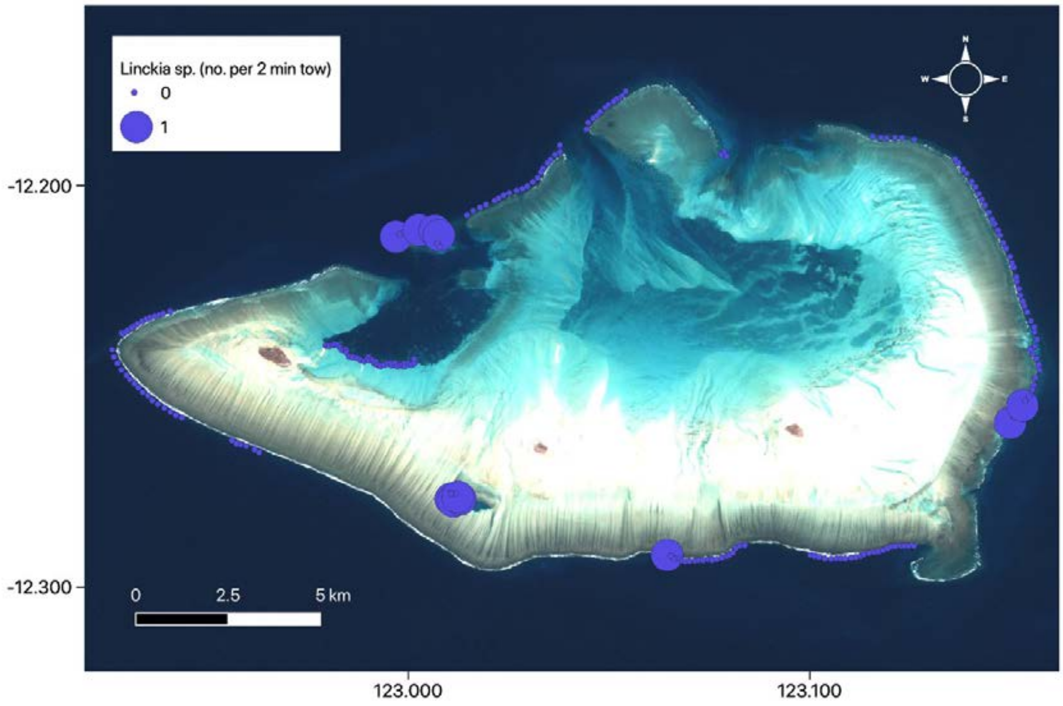


Figure 133.The combined abundance of *Linckia laevigata* and *Linckia guildingi* per each two-minute manta tow at Ashmore Reef.

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^2 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.

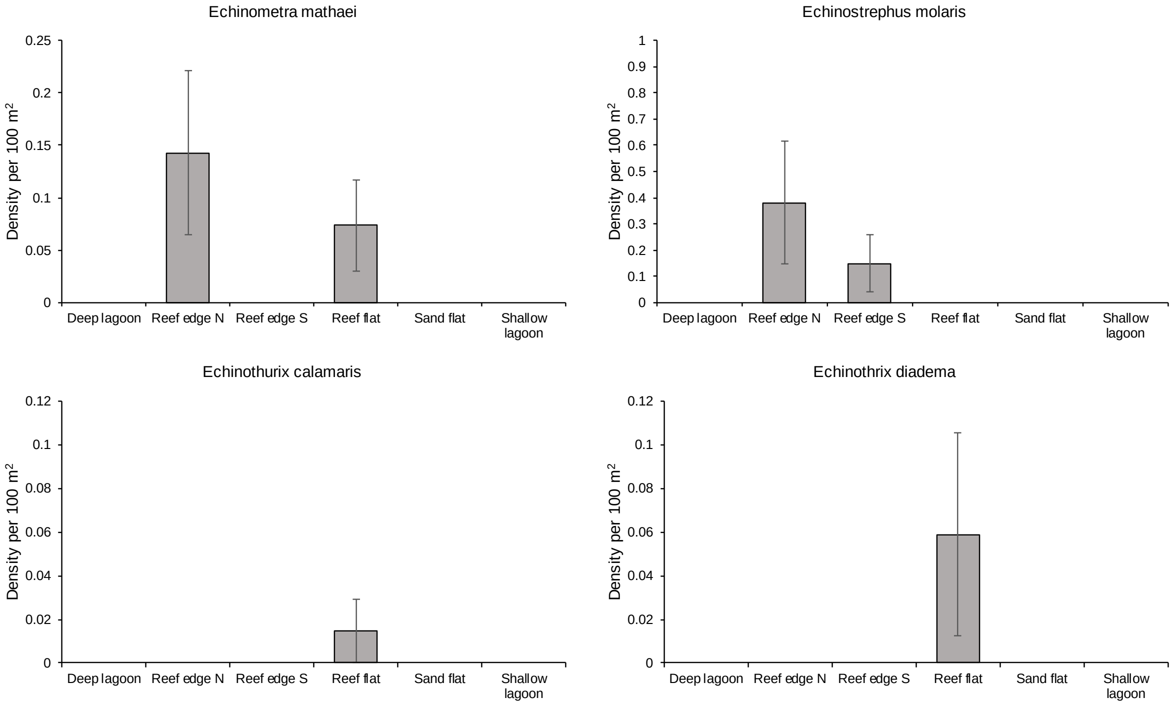


Figure 134. Mean density of sea urchins in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

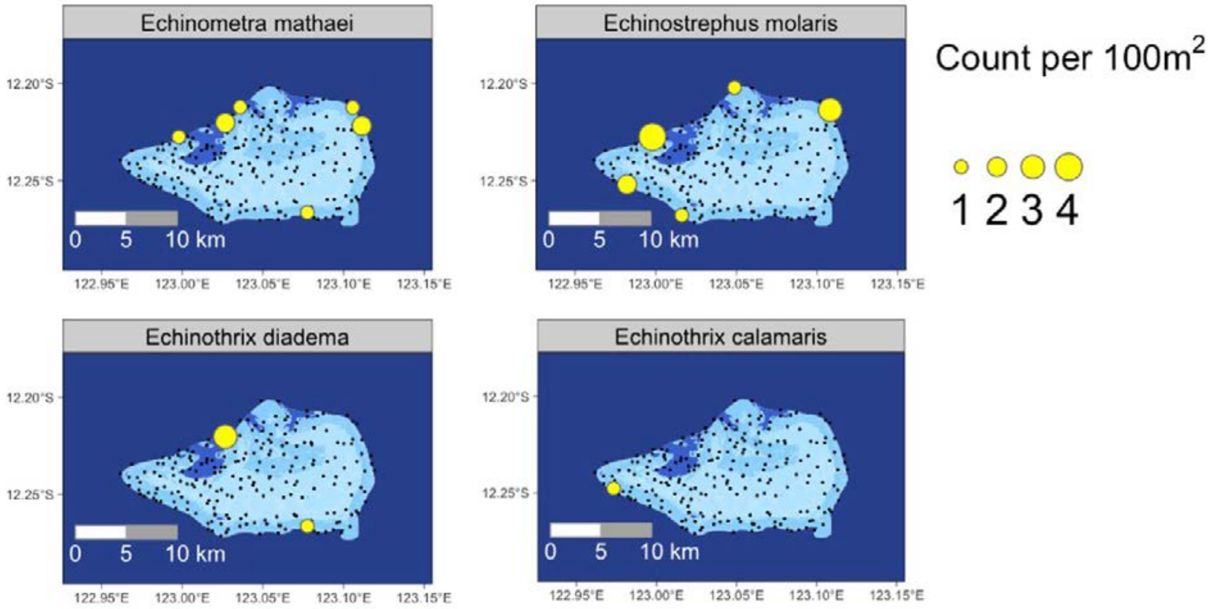


Figure 135. Distribution and density/100 m^2 of all sea urchin species across each of the 224 transects surveyed at Ashmore Reef in June 2019.

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^2 (Figure 136).

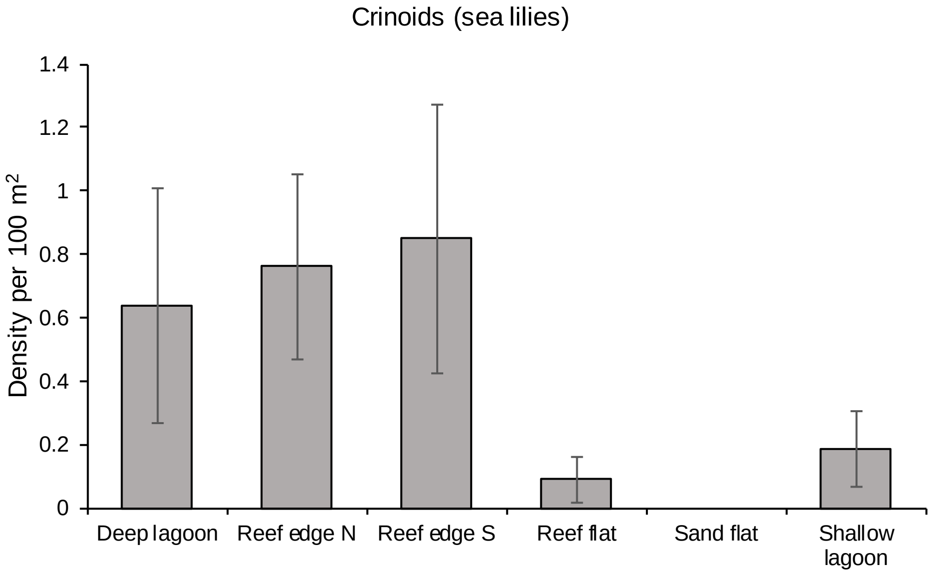


Figure 136. Mean density of crinoids in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

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^2 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^2) 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).

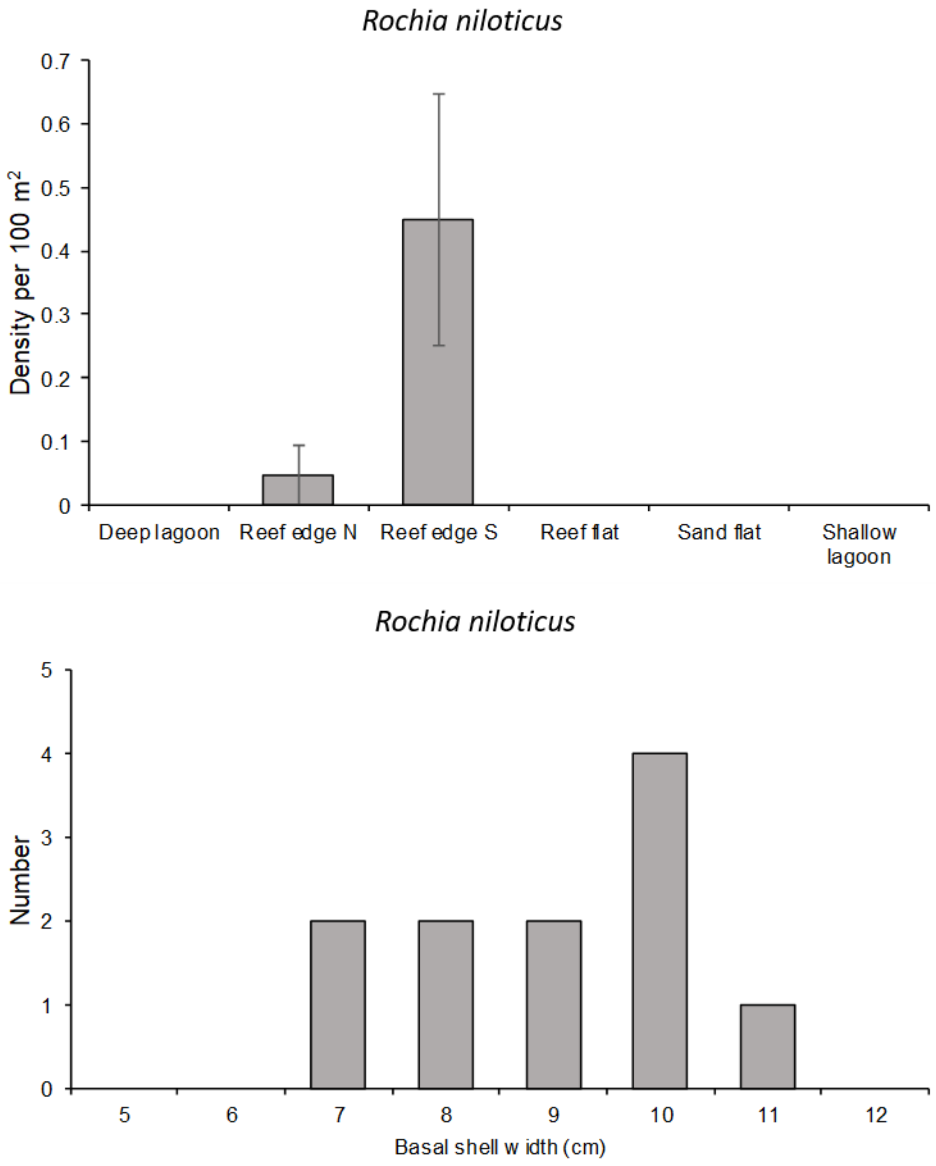


Figure 137. Upper panel is mean density of trochus in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e. Lower panel are shell measurements.

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^2 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.

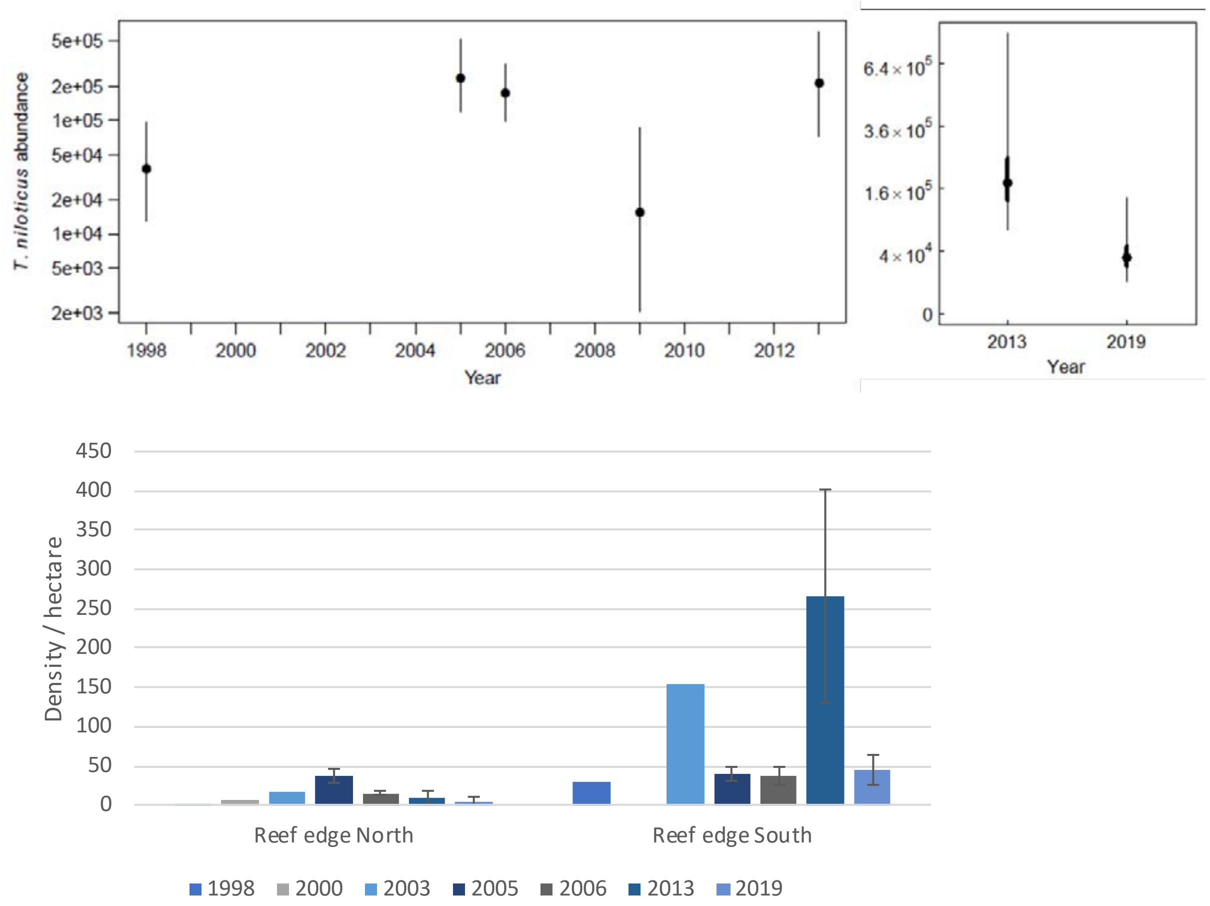


Figure 138. Long term estimates of abundance of trochus *Rochia niloticus* at Ashmore Reef. Upper left panel is from Hosack and Lawrence (2013a), upper right panel is from Hosack et al. in chapter 12 of this study. These represent the total estimated population on Ashmore Reef. Note Y axis scales are 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 (±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^2, 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.

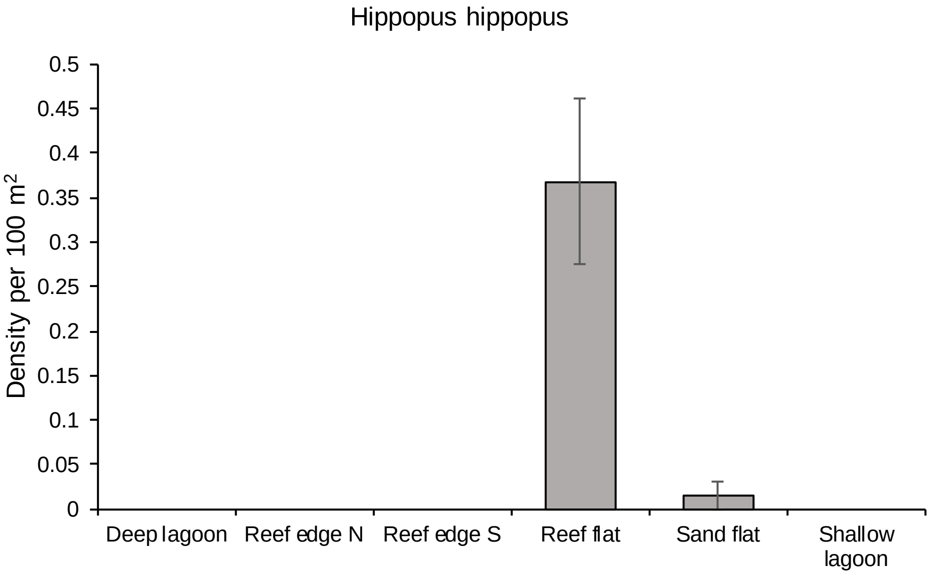


Figure 139. Mean density of the clam *Hippopus hippopus* in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

Tridacna spp. (Giant clams)

The elongate clam *Tridacna maxima* and the fluted gain 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.

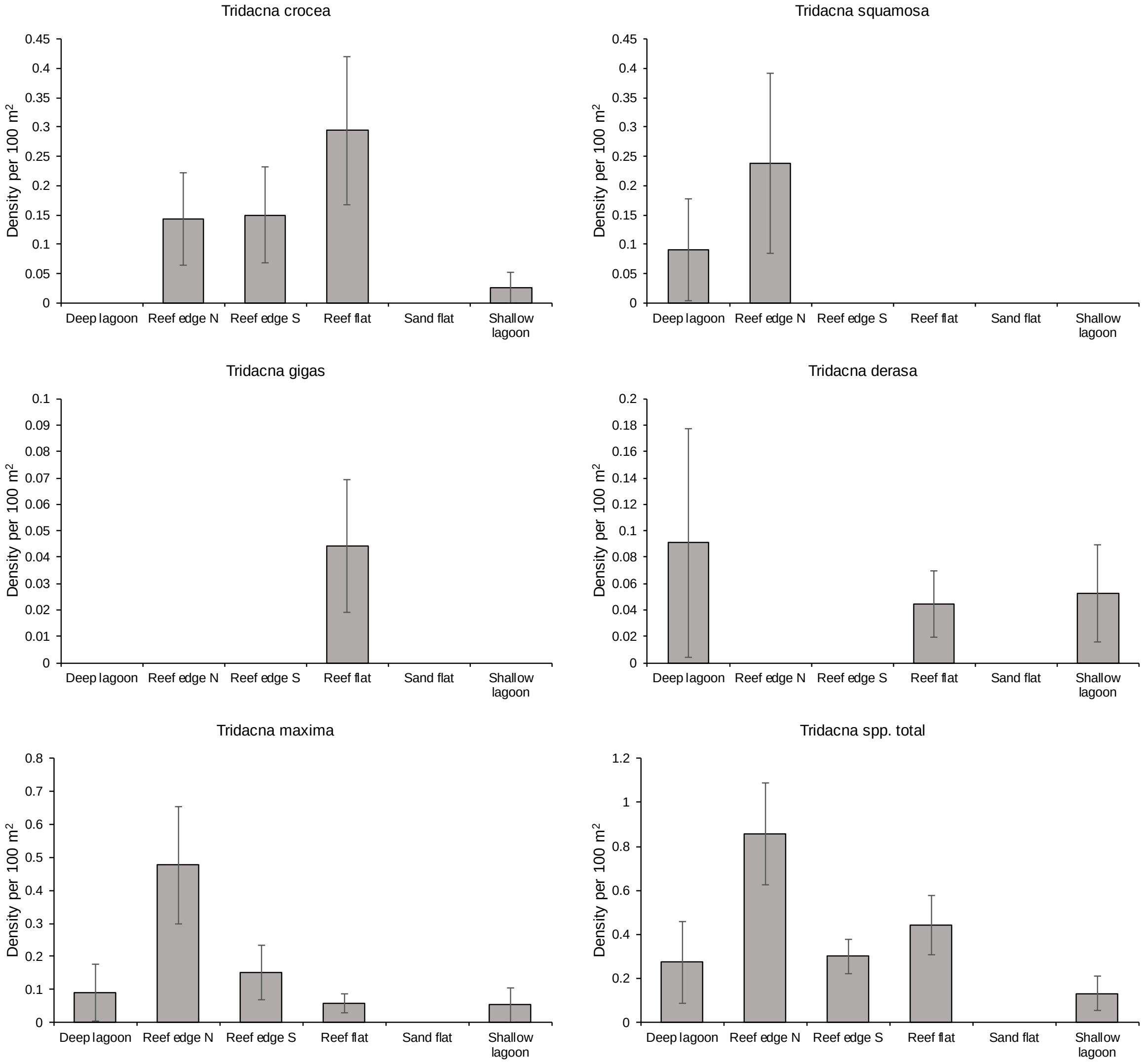


Figure 140. Mean density of tridacnid clam species and combined all species in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

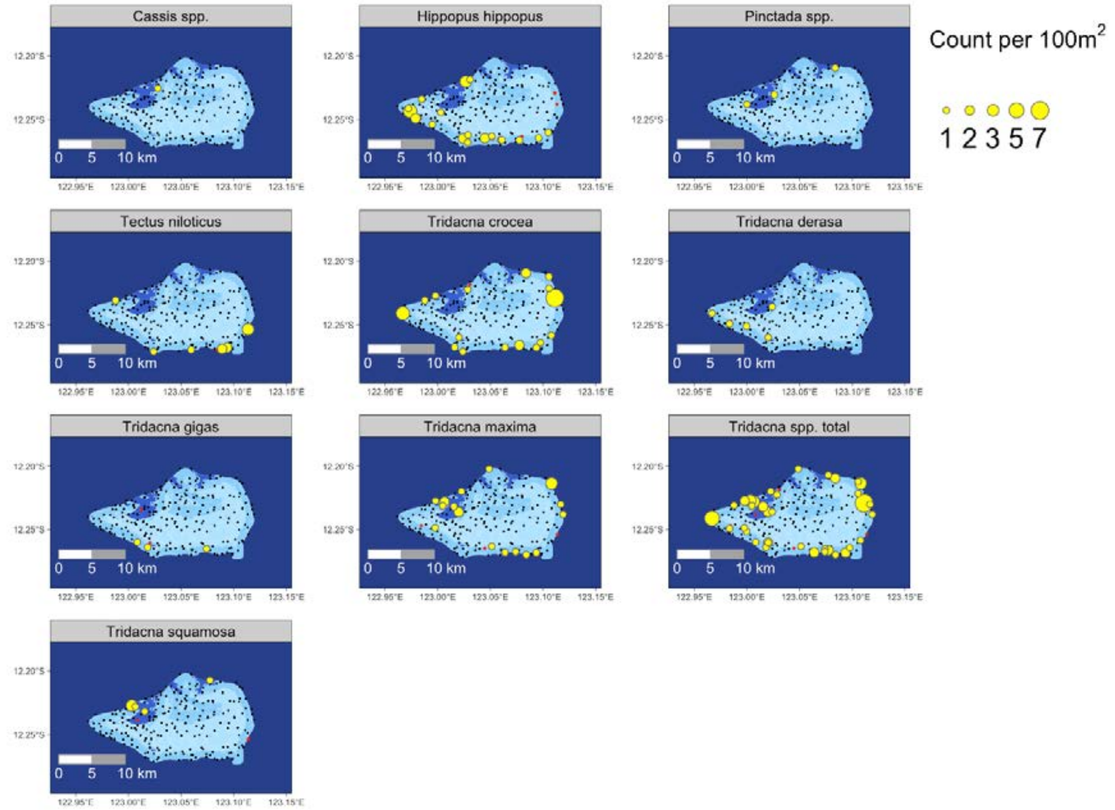


Figure 141. Distribution and density/100 m² of all mollusc species across each of the 224 transects surveyed at Ashmore Reef in June 2019. The red dots indicate presence of that species at that site but that the specimen(s) were not within the area of the 100 m² quantitative survey transect.

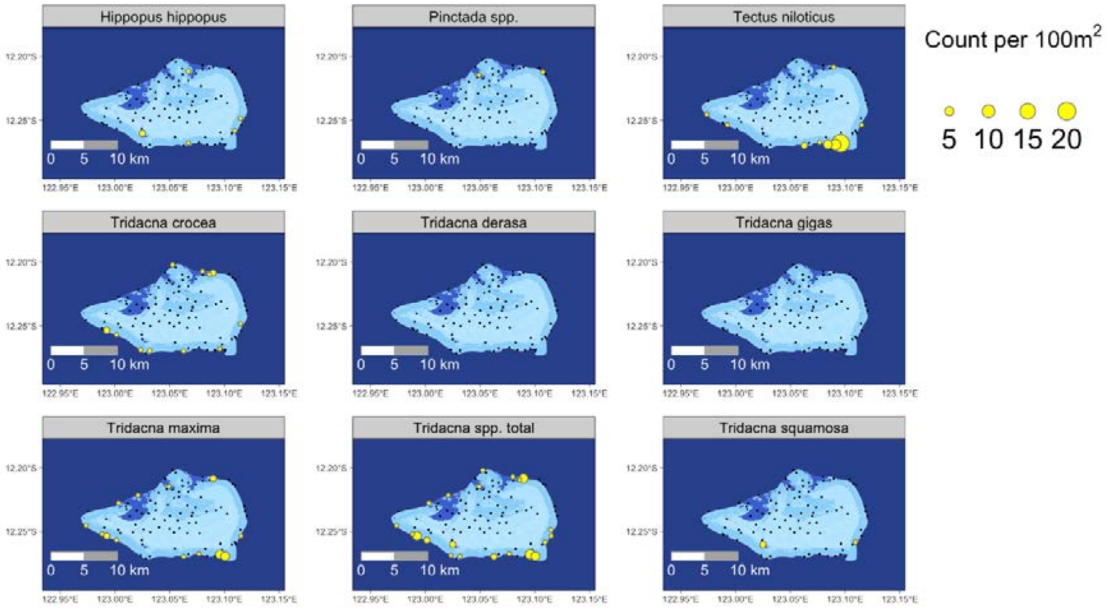


Figure 142. Distribution and density/100 m² of all mollusc species across each of the 95 transects surveyed at Ashmore Reef in March 2013 Data from Ceccarelli et al. (2013) with densities converted to per 100 m² from their 80 m² transects.

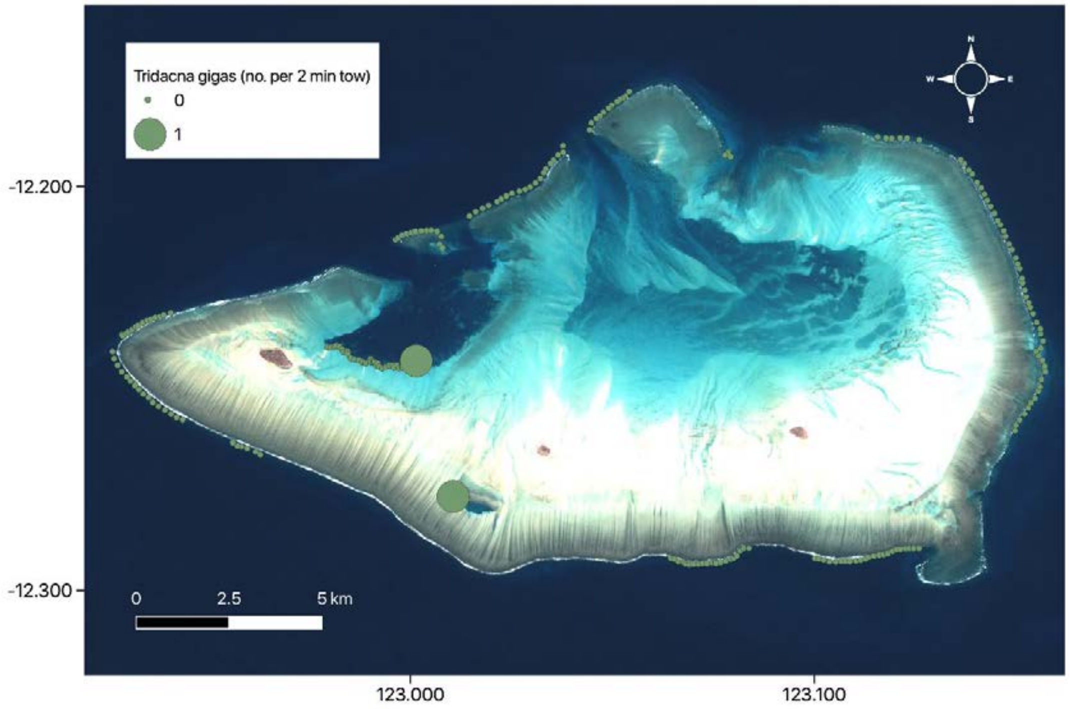


Figure 143. Map of Ashmore Reef showing the location of *Tridacna gigas* counted during the two-minute manta tows

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 perahus 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 of 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^2 are similar to those recorded by Skewes *et al.* (1999a) but the 2013 and 2019 densities of *H. hippopus* are much lower (Figure 145).

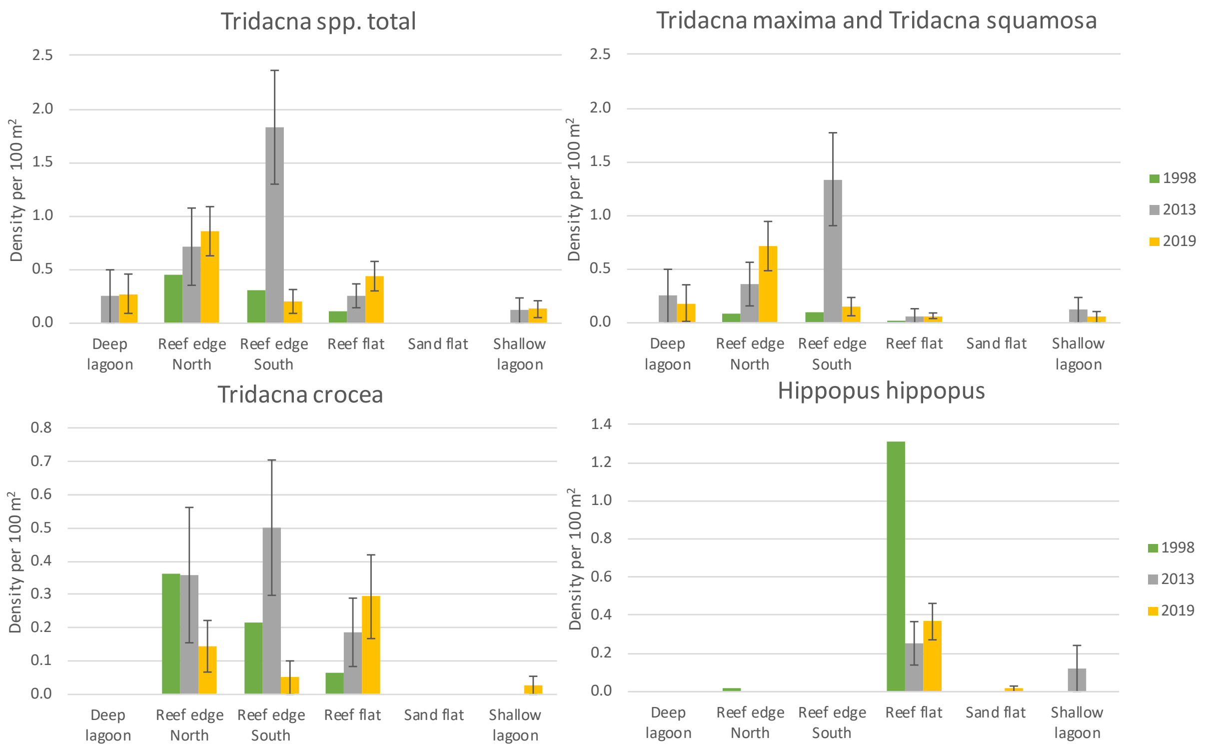


Figure 145. Comparisons of historical and recent surveys of clams at Ashmore Reef stratified by habitat type comparing the 1998, 2013 and 2019 surveys (see Figure 87 for 1998 sample sites). Error bars are ±1 s.e.

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^2) nor helmet shells (*Cassis cornuta*) (0.026/100 m^2) 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^2 in the deep lagoon and 0.09/100 m^2 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.

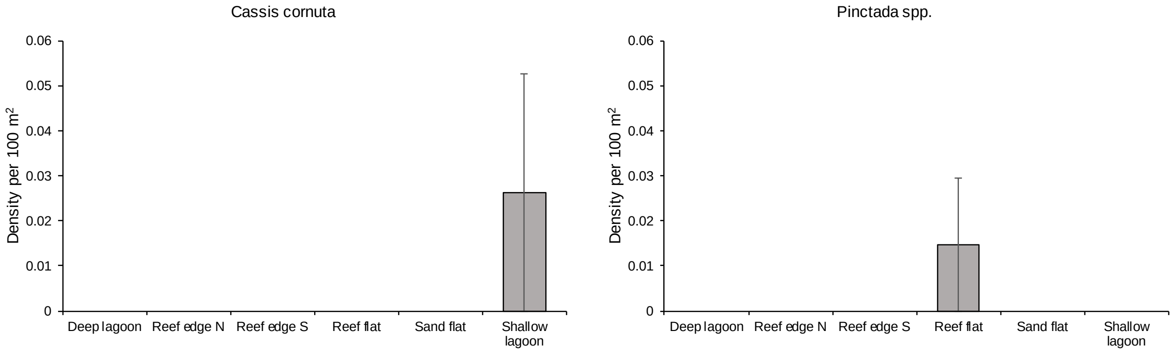


Figure 146. Mean density of helmet shells (*Cassis*) and pearl oysters (*Pinctada*) in each of the habitats surveyed at Ashmore Reef in June 2019. Error bars are ±1 s.e.

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 fuscorubra*. **** Russell and Vail (1988) illustrate both *Holothuria scabra* and *H. fuscorubra* 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
Actinopyga echinites	Hedgehog sea cucumber	Y	-						-	
Actinopyga lecanora	Reef sea cucumber	Y	-		Y	Y	Y		-	Y
Actinopyga mauritiana	Mauritian sea cucumber	Y	-		Y	Y	Y		-	Y
Actinopyga miliaris	Military sea cucumber		-			Y	Y	Y	-	Y
Actinopyga obesa	Plump sea cucumber	Y	-						-	
Actinopyga palauensis	Palauan sea cucumber		-						-	Y
***Actinopyga* scored as a group**	-	-	Y	-	-	-	-	-	Y	-
Bohadschia argus	Eyed sea cucumber or leopardfish	Y	Y	Y	Y	Y	Y	Y		Y
Bohadschia marmorata (aka vitiensis)	Marbled sea cucumber	Y			Y	Y	Y			
Holothuria atra	Black sea cucumber	Y	Y	Y	Y	Y	Y	Y	Y	Y
Holothuria coluber	Snake sea cucumber			y	Y		Y			Y
Holothuria edulis	Unsavoury sea cucumber	Y	Y	Y	Y	Y	Y	Y		Y
Holothuria fuscogilva	White teatfish	Y	Y	Y	Y	Y	Y	Y	Y	Y
***Holothuria fuscopunctata* (aka *axiologa*)**	Black-banded sea cucumber	Y	Y		Y	Y	Y		Y	Y

Holothuria fuscorubra	Orange spotted sea cucumber	****					Y	***	Y	
Holothuria impatiens	Restless sea cucumber	Y								
Holothuria lessoni **(aka** ***timana/aculeata)***	Golden sandfish or Lesson's sea cucumber	Y								
Holothuria leucospilota	Stained sea cucumber	Y	Y	Y	Y	Y	Y		Y	Y
Holothuria scabra	Sandfish	****								
Holothuria whitmaei **(aka** ***nobilis)***	Black teatfish	Y	Y	Y	Y	Y	Y	Y	Y	Y
Holthuria rigida	Rigid sea cucumber									Y
Pearsonothuria graeffei	Graeffe's sea cucumber	Y	Y	Y	Y	Y	Y	Y	Y	Y
Stichopus chloronotus	Green sea cucumber	Y	Y	y	Y	Y	Y	Y	Y	Y
Stichopus herrmanni **(aka** ***variegatus*****)**	Herrmann's sea cucumber	Y	Y	Y	Y	Y	Y	Y	Y	Y
Thelenota ananas	Prickly redfish	Y	Y	Y	Y	Y	Y	Y	Y	Y
Thelenota anax	Royal sea cucumber	Y	Y	Y	Y	Y	Y			Y
Total		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, Culcita 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^2) and shallow lagoon (41.1 km^2) which excludes islands and sand flats (61.2 km^2) and the deep lagoon (31.8 km^2). 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).

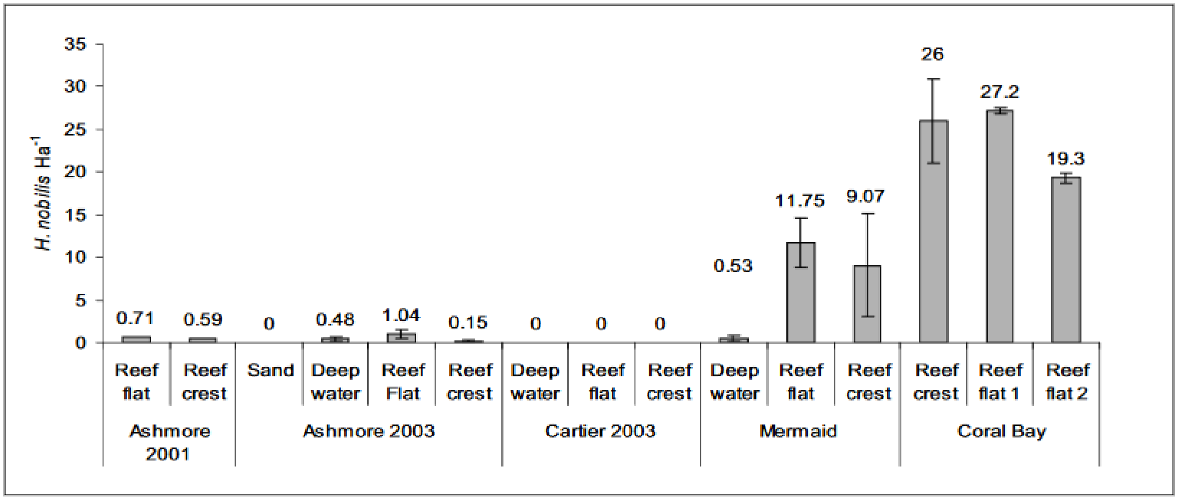


Figure 147. Comparison of densities of *Holothuria whitmaei* at the heavily fished Ashmore and Cartier Reefs with that of the less fished Mermaid Reef and the unfished Coral Bay (Ningaloo Reef) in 2004. The graph shown is figure 2 from Shiell (2005). Data from Ashmore and Cartier Reefs is taken from 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 from 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. fuscorubra, 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^2 in the habitat in which they were most abundant in the 2019 survey with the exception of *H. fuscogilva* (0.05 per 100 m^2) and *S. herrmanni* (0.2 per 100 m^2).

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^2 on southern edge) than those recorded in 2013 (ca. 2.5/ 100 m^2), but similar to surveys in 2005 and 2006 (ca. 0.5/ 100 m^2). 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 low (<0.05/ 100 m^2) 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^2) 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 reseeding 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^2 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,

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

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 β 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 β_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 $w(s^\star)$ 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 ρ and the spatial variance σ^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 ρ 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^2, 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 β 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 ± 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^2) 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).

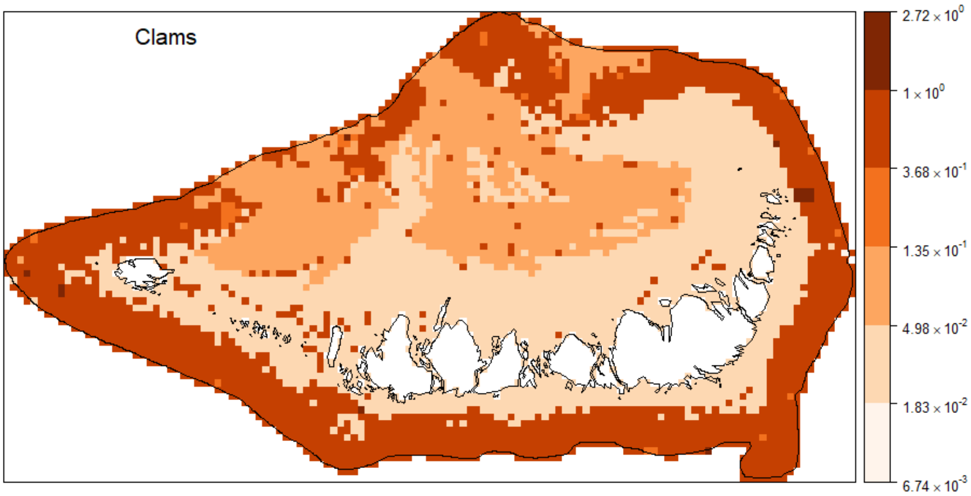


Figure 148. Posterior mean of areal density of all clam species (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

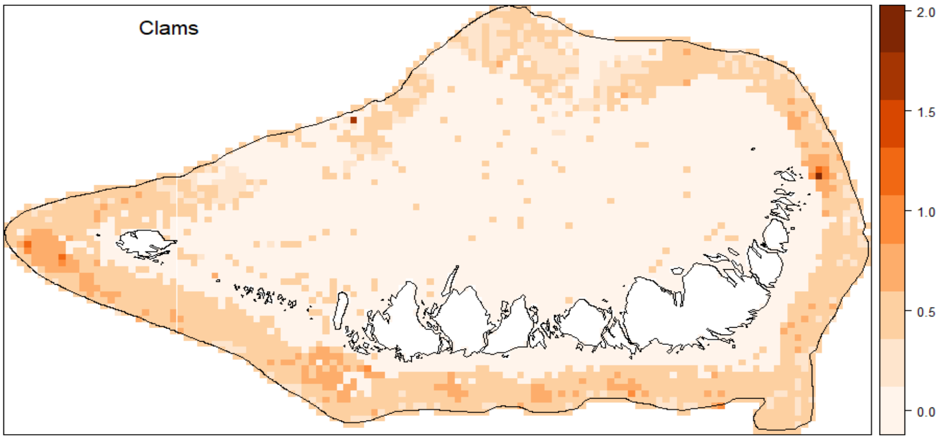


Figure 149. Posterior inter-quartile distance of areal density of all clam species (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

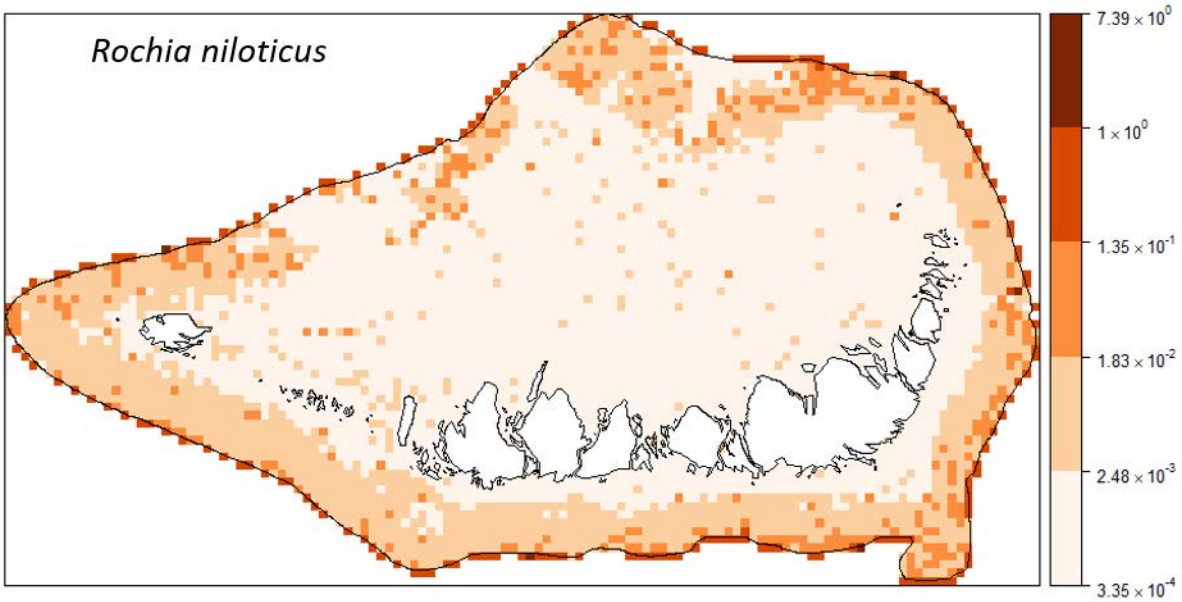


Figure 150. Posterior mean of areal density of trochus (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

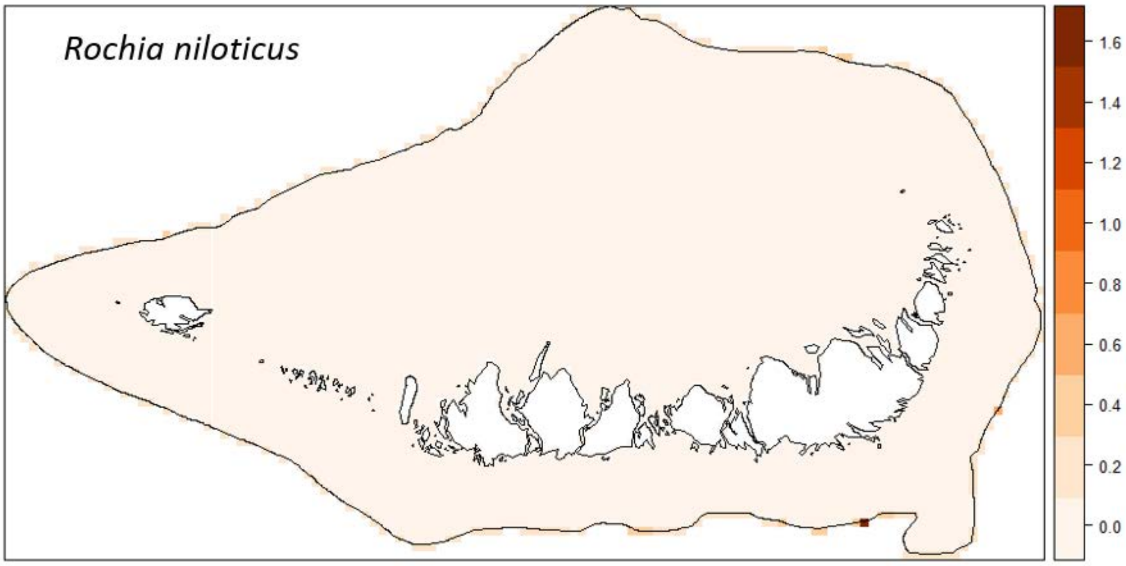


Figure 151.Posterior inter-quartile distance of areal density of trochus (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

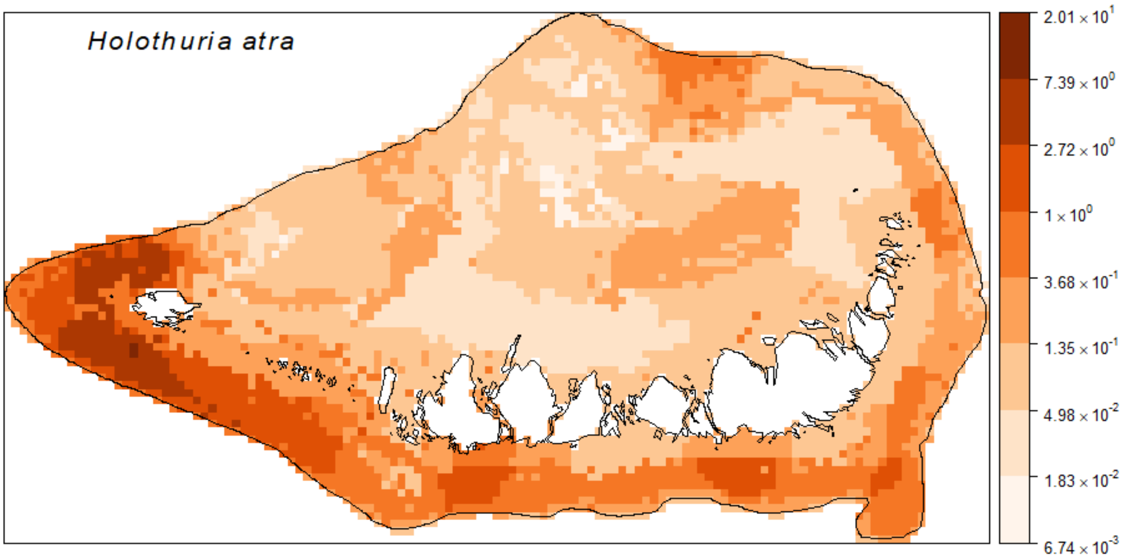


Figure 152. Posterior mean of areal density of *Holothuria atra* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

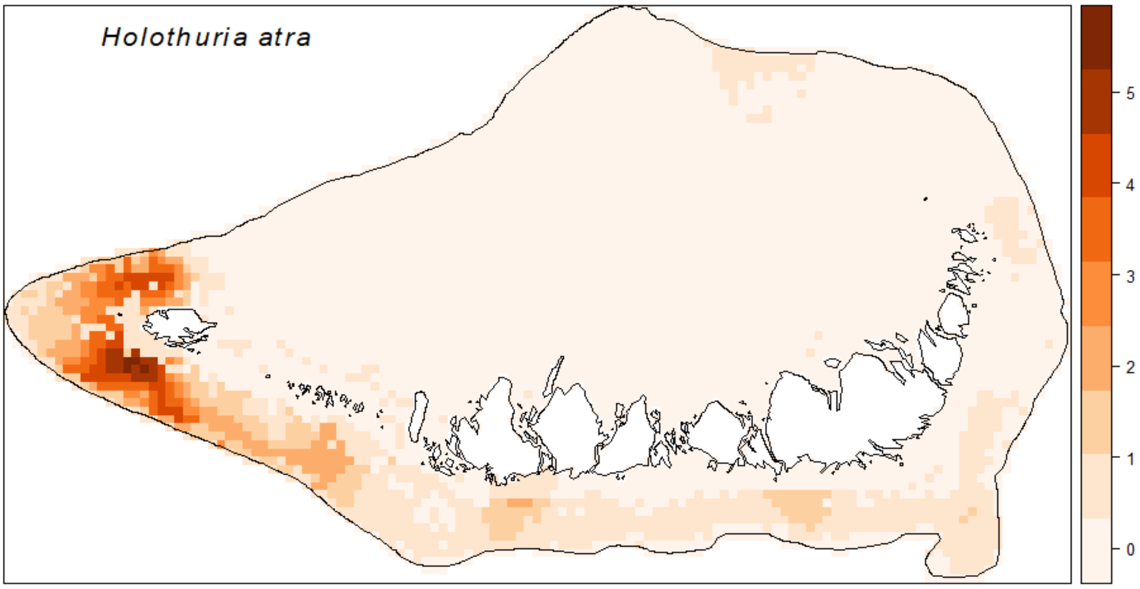


Figure 153. Posterior inter-quartile distance of areal density of *Holothuria atra* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

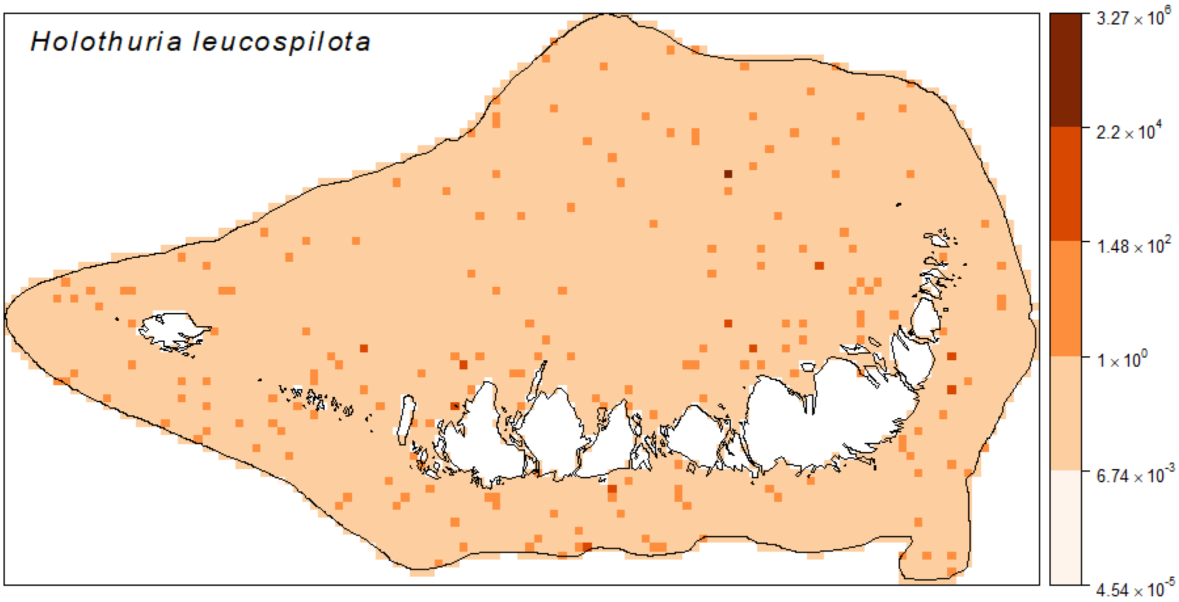


Figure 154. Posterior mean of areal density of *Holothuria leucospilota* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

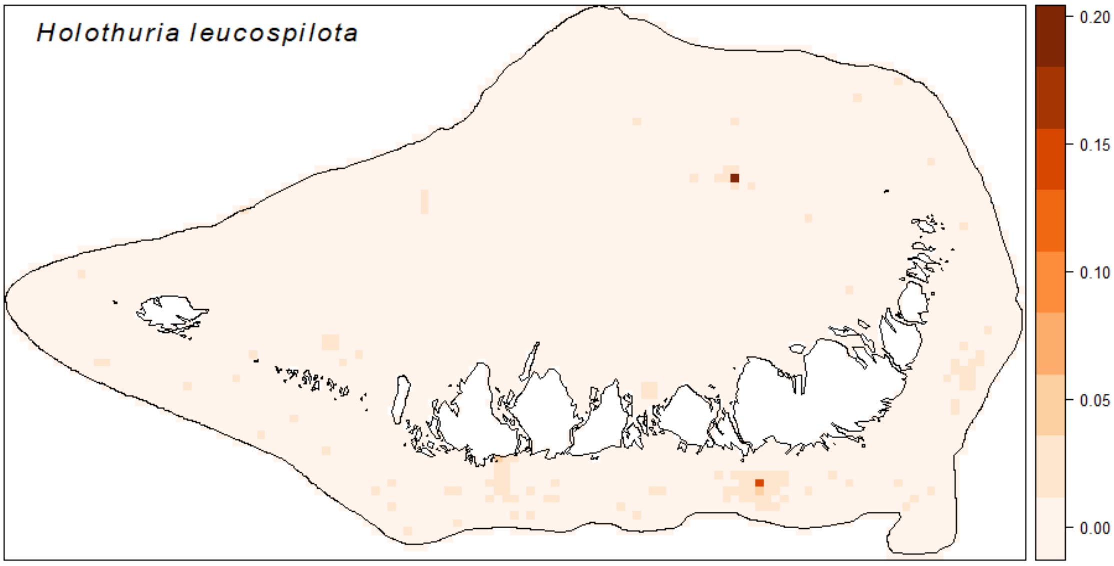


Figure 155. Posterior inter-quartile distance of areal density of *Holothuria leucospilota* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

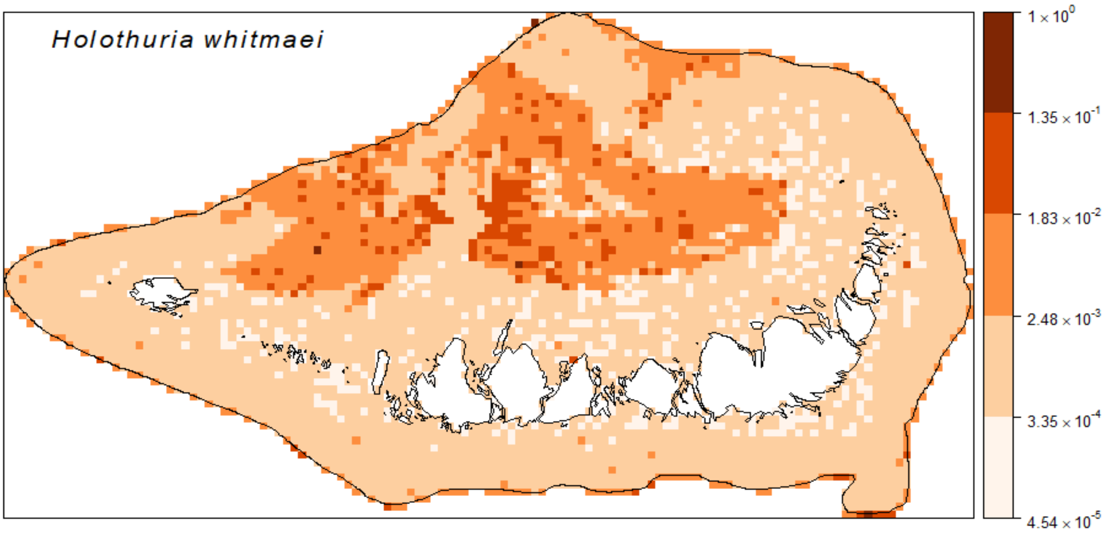


Figure 156. Posterior mean of areal density of *Holothuria whitmaei* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

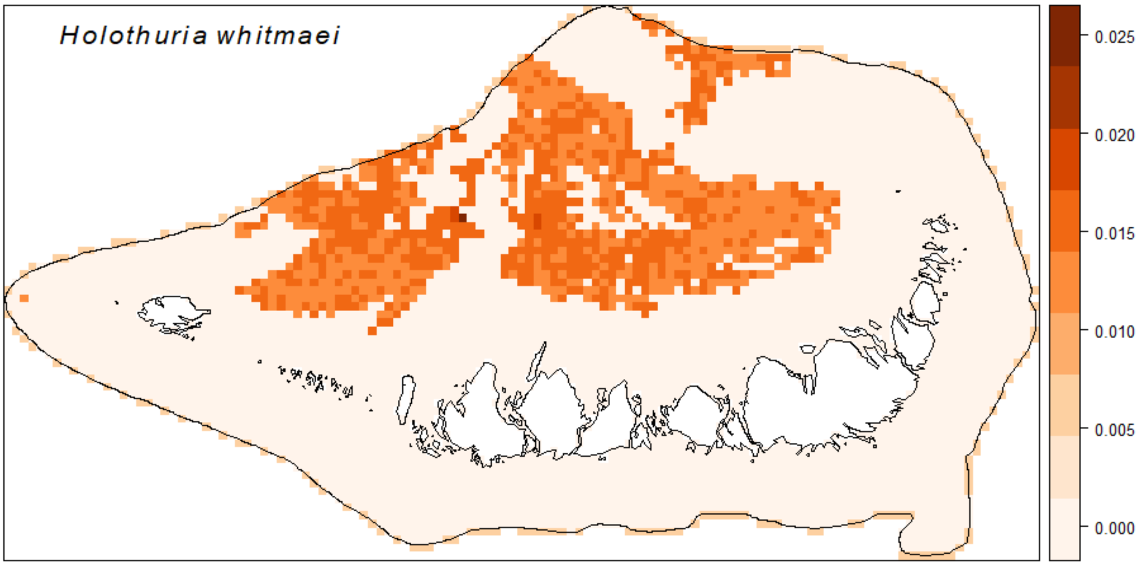


Figure 157. Posterior inter-quartile distance of areal density of *Holothuria whitmaei* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

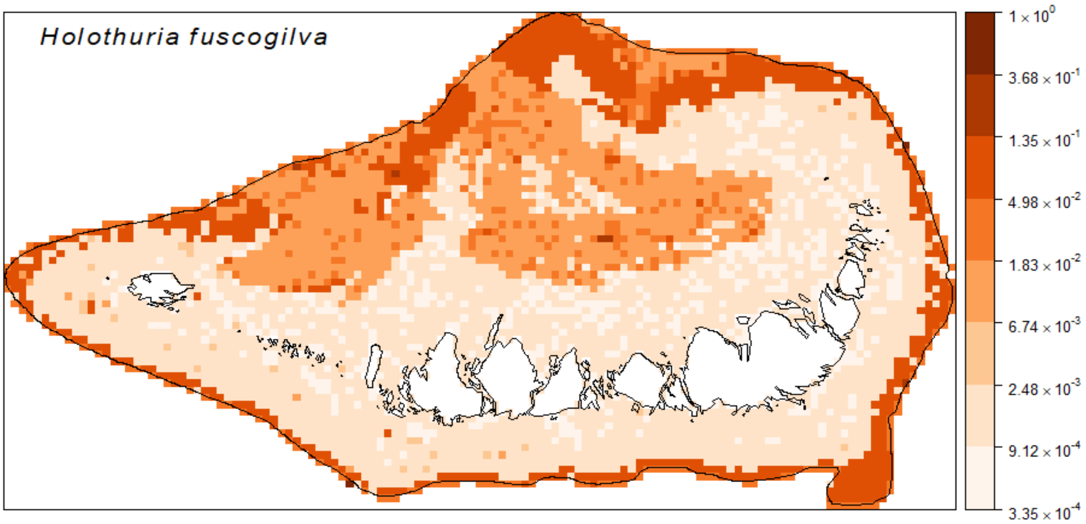


Figure 158. Posterior mean of areal density of *Holothuria fuscogilva* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

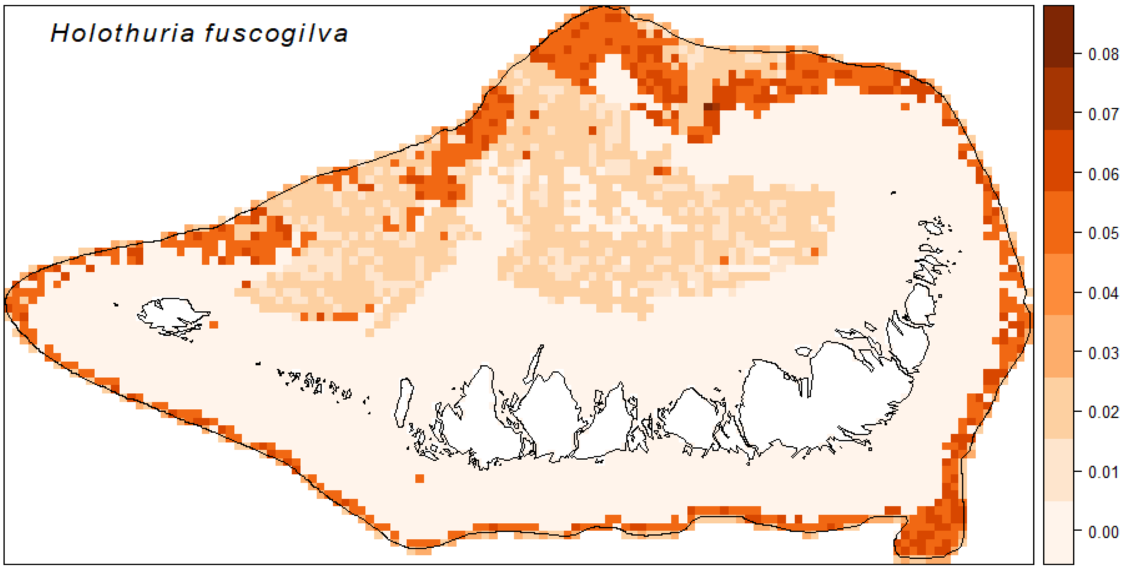


Figure 159. Posterior inter-quartile distance of areal density of *Holothuria fuscogilva* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

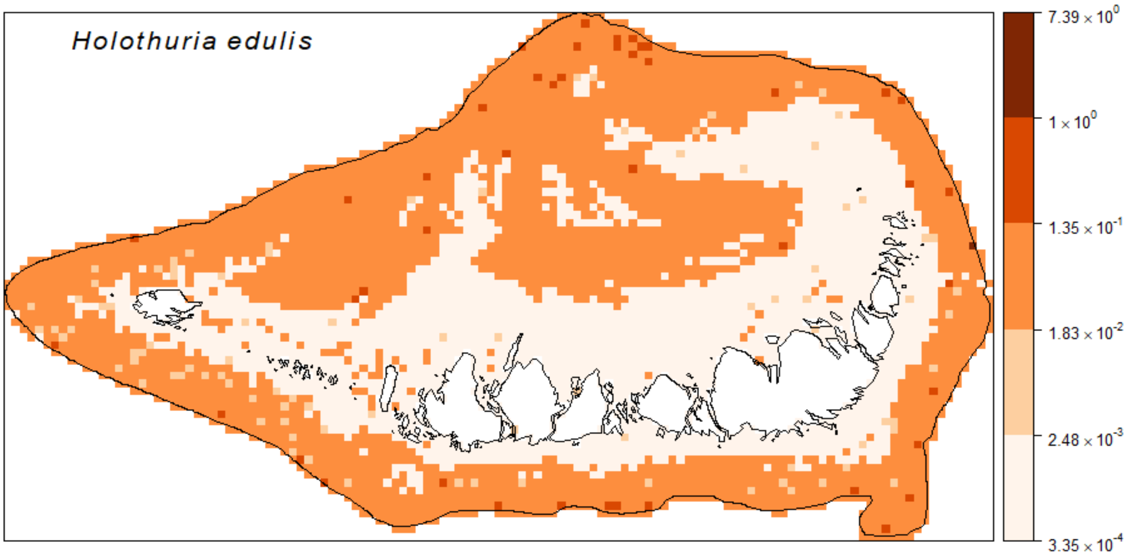


Figure 160. Posterior mean of areal density of *Holothuria edulis* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

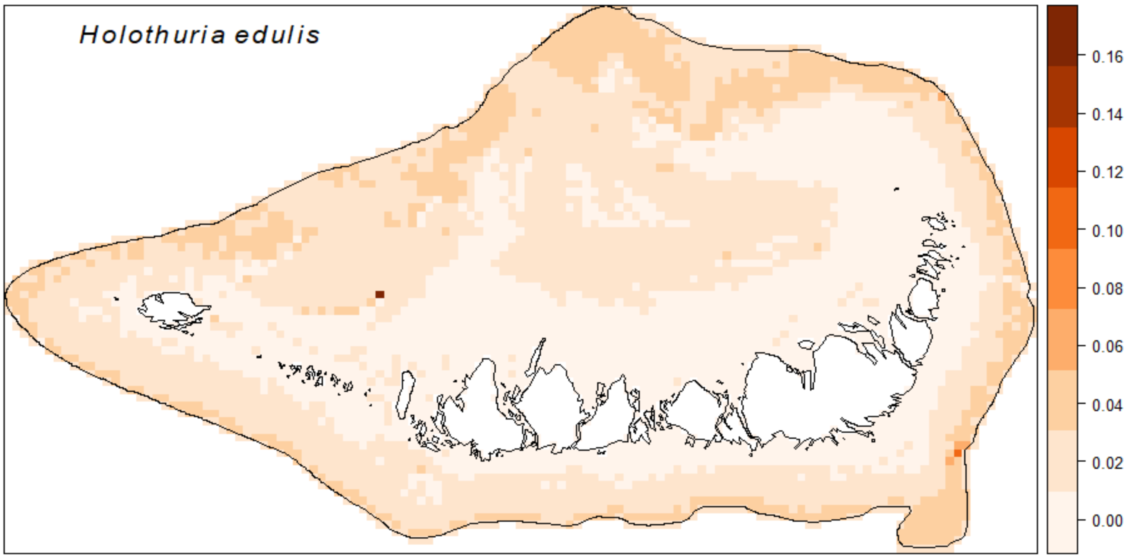


Figure 161. Posterior inter-quartile distance of areal density of *Holothuria edulis* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

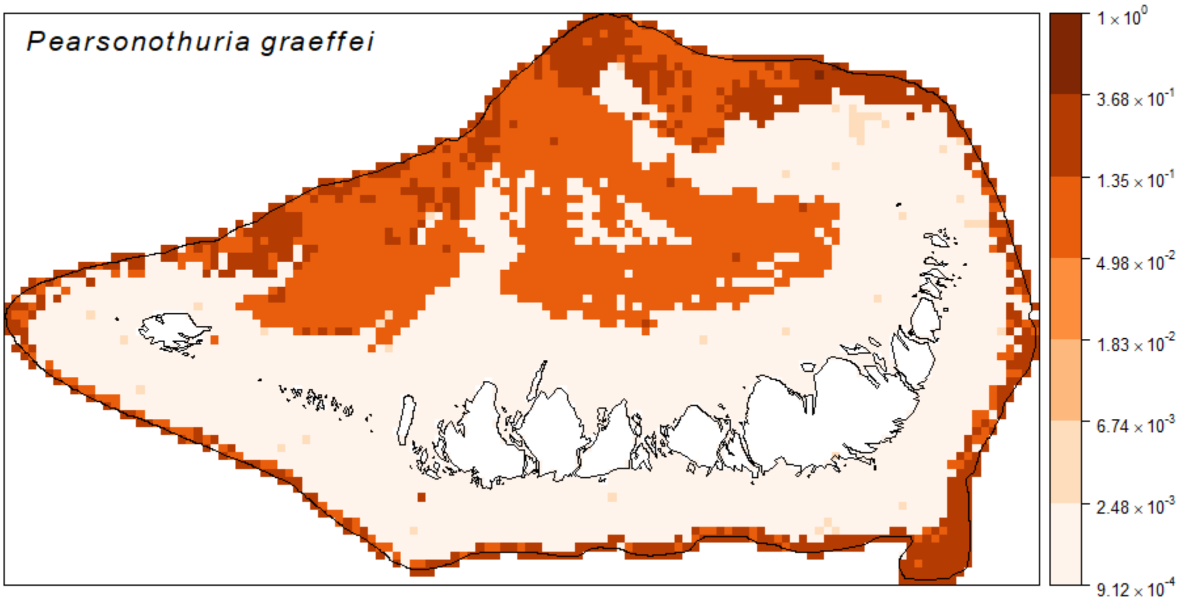


Figure 162. Posterior mean of areal density of *Pearsonothuria graeffei* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

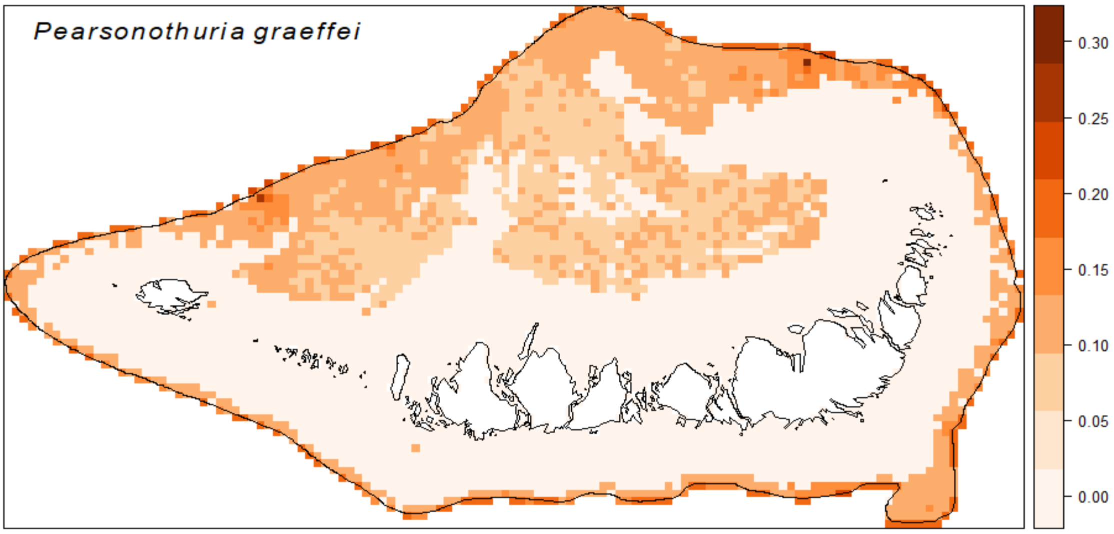


Figure 163. Posterior inter-quartile distance of areal density of *Pearsonothuria graeffei* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

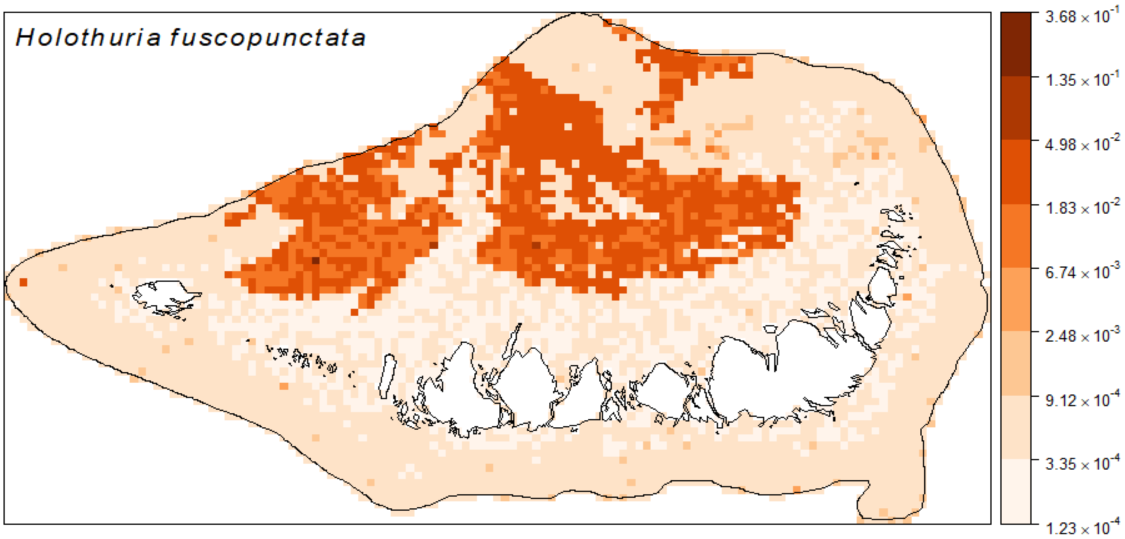


Figure 164. Posterior mean of areal density of *Holothuria fuscopunctata* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

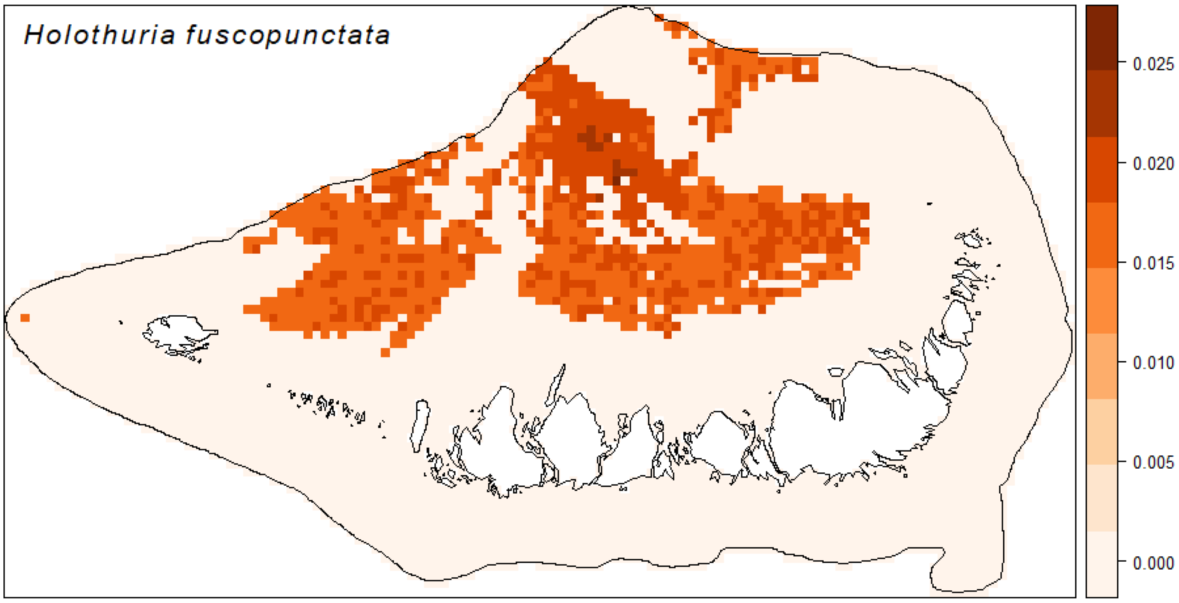


Figure 165. Posterior inter-quartile distance of areal density of *Holothuria fuscopunctata* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

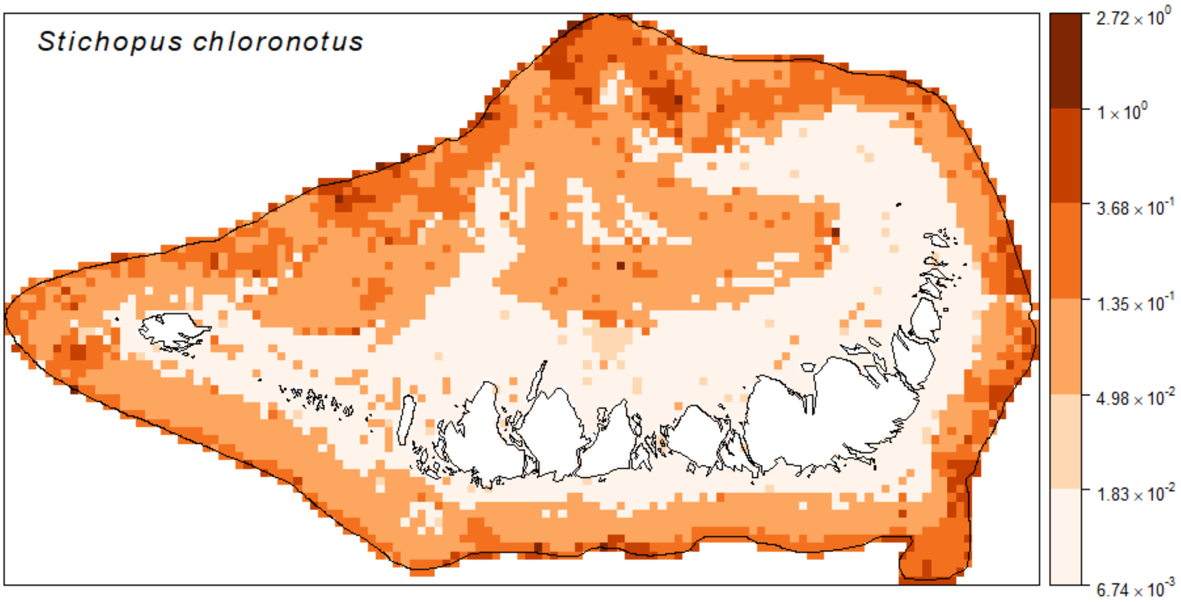


Figure 166. Posterior mean of areal density of *Stichopus chloronotos* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

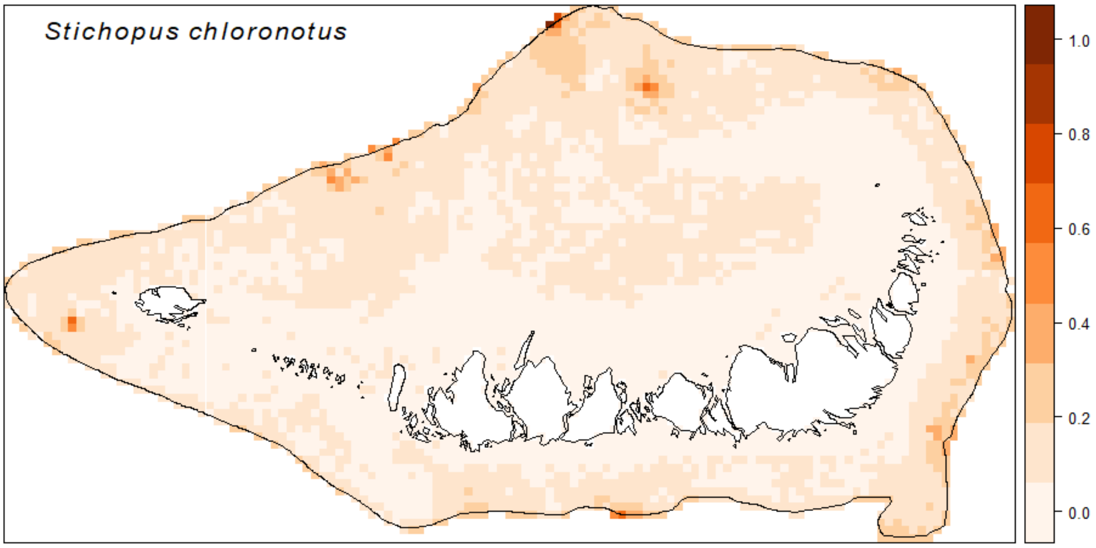


Figure 167. Posterior inter-quartile distance of areal density of *Stichopus chloronotos* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

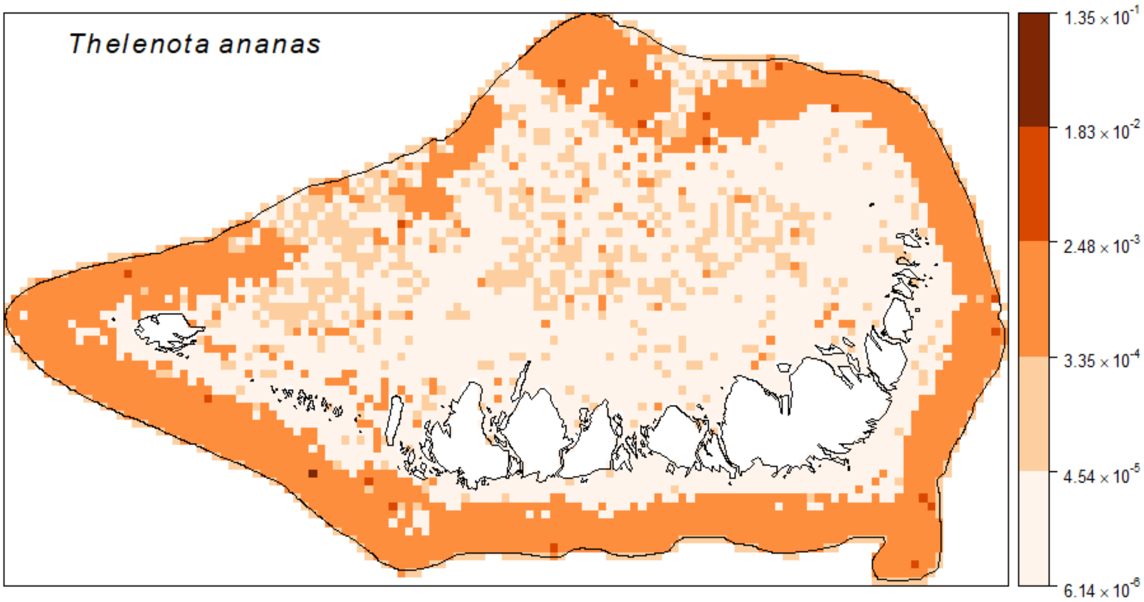


Figure 168. Posterior mean of areal density of *Thelenota ananas* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

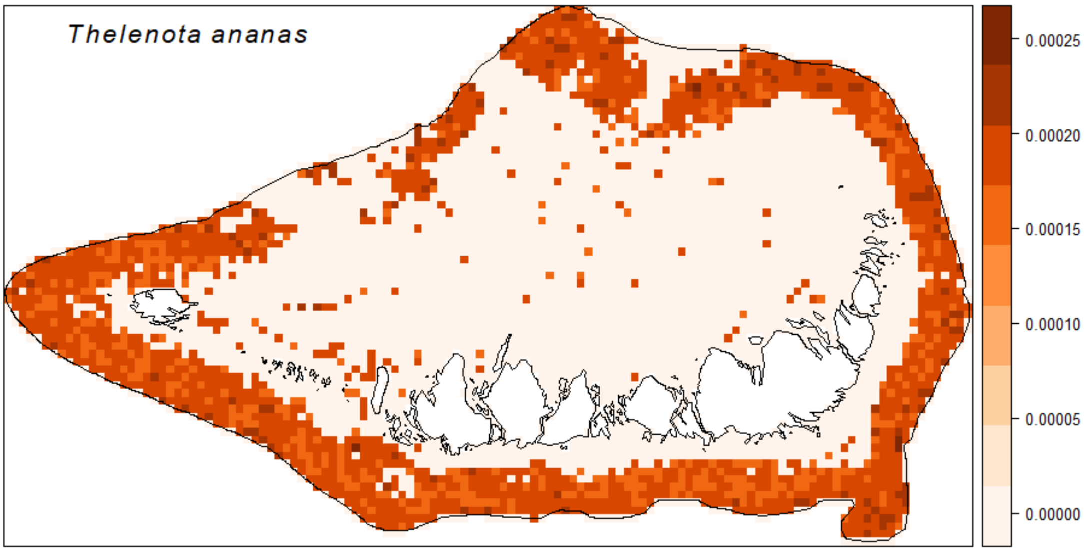


Figure 169. Posterior inter-quartile distance of areal density of *Thelenota ananas* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

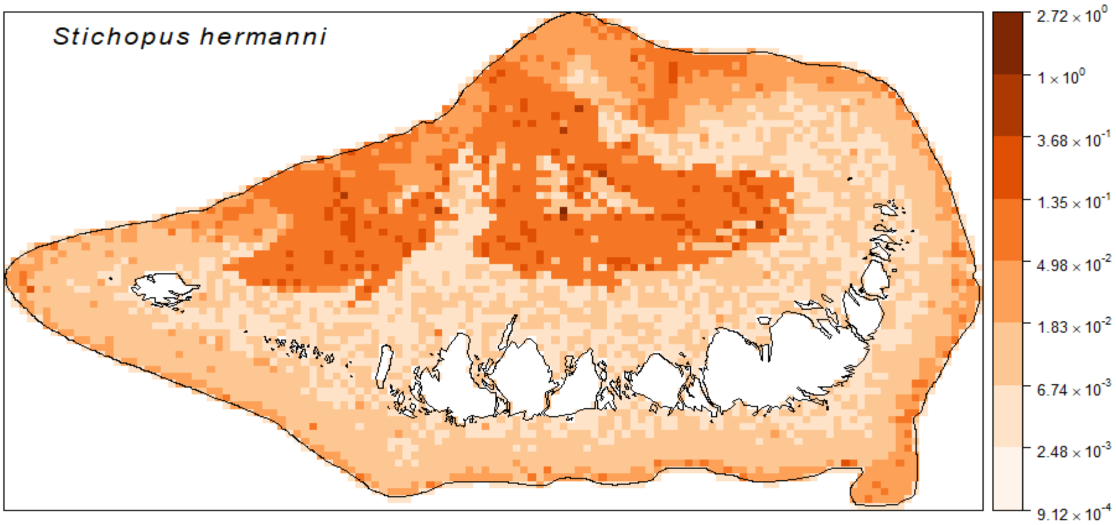


Figure 170. Posterior mean of areal density of *Stichopus herrmanni* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

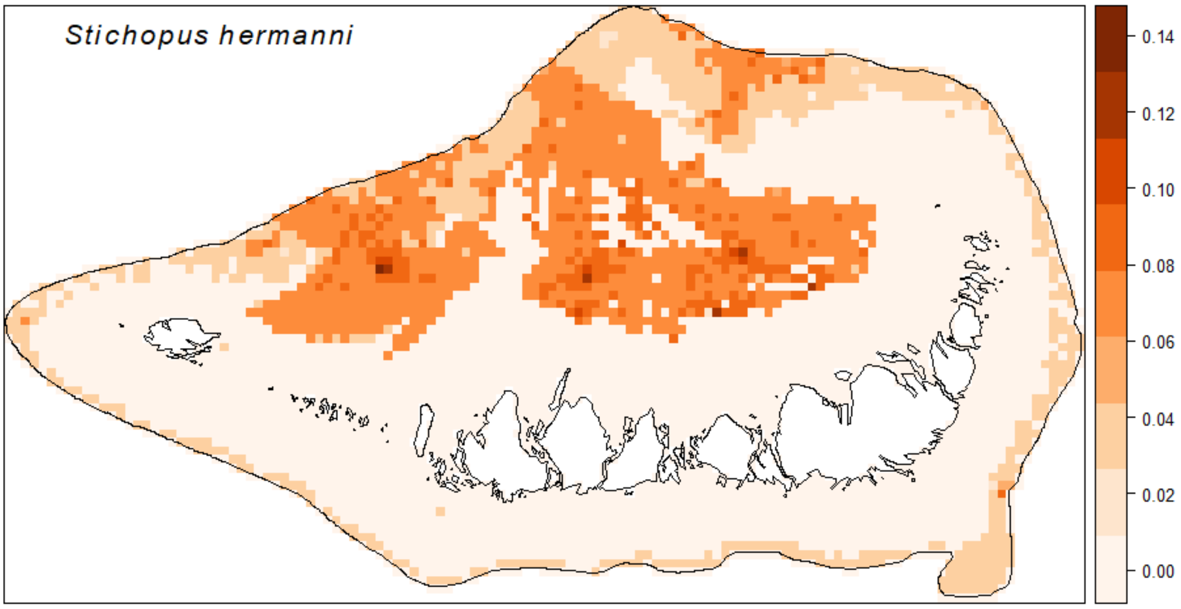


Figure 171. Posterior inter-quartile distance of areal density of *Stichopus herrmanni* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

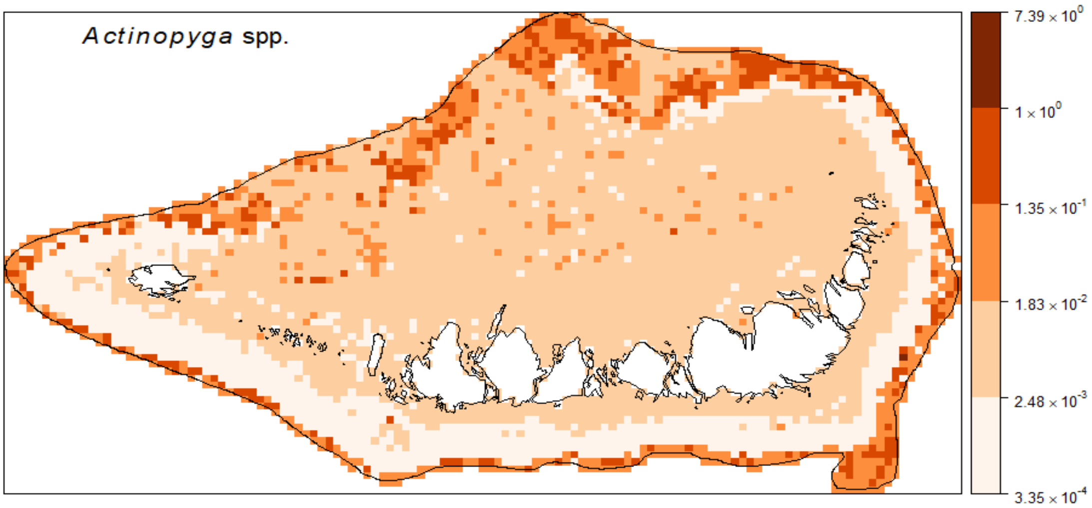


Figure 172. Posterior mean of areal density of all *Actinopyga* species (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

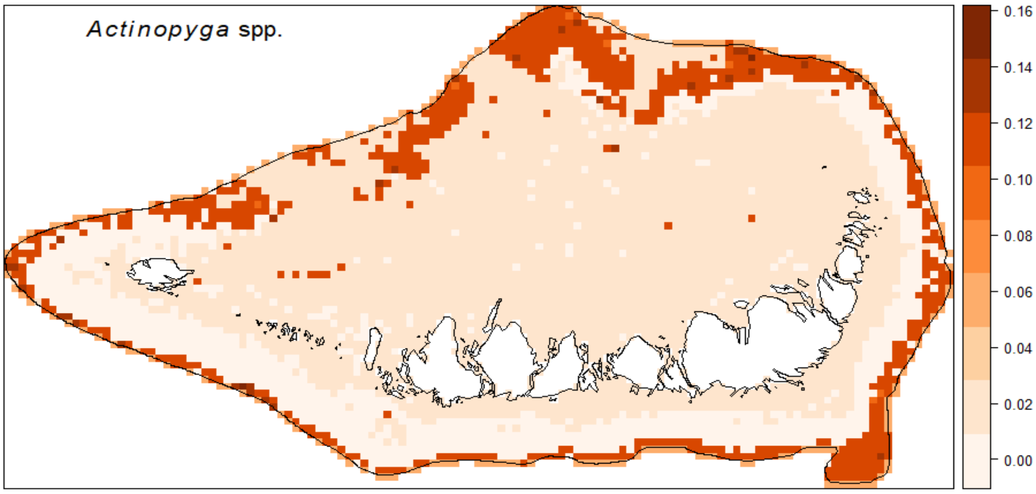


Figure 173. Posterior inter-quartile distance of areal density of all *Actinopyga* species (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

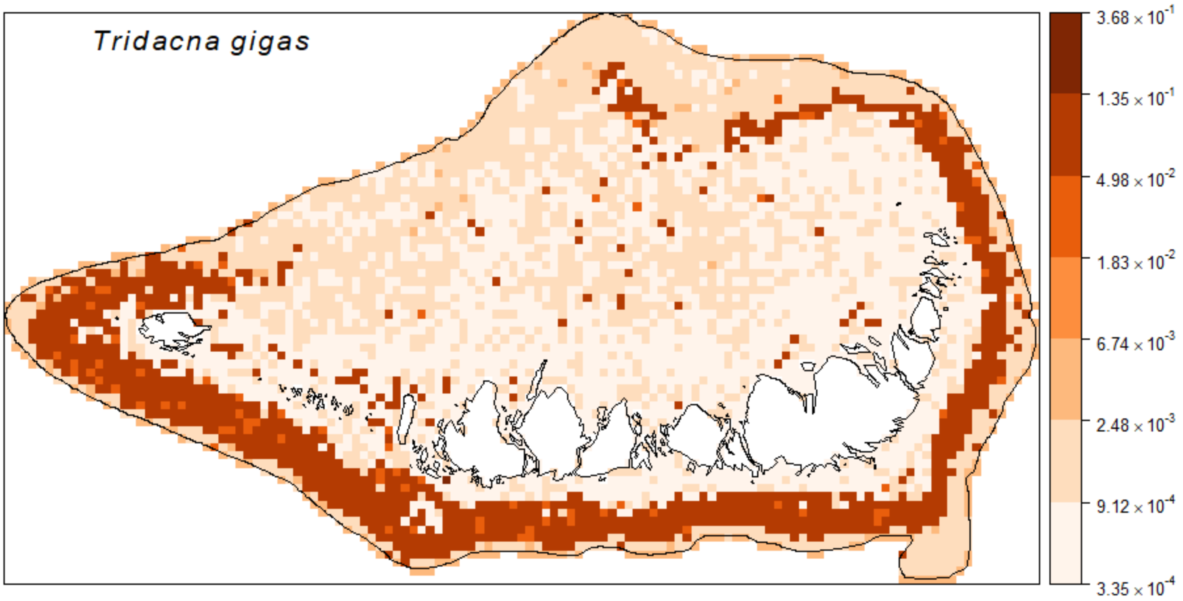


Figure 174. Posterior mean of areal density of *Tridacna gigas* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

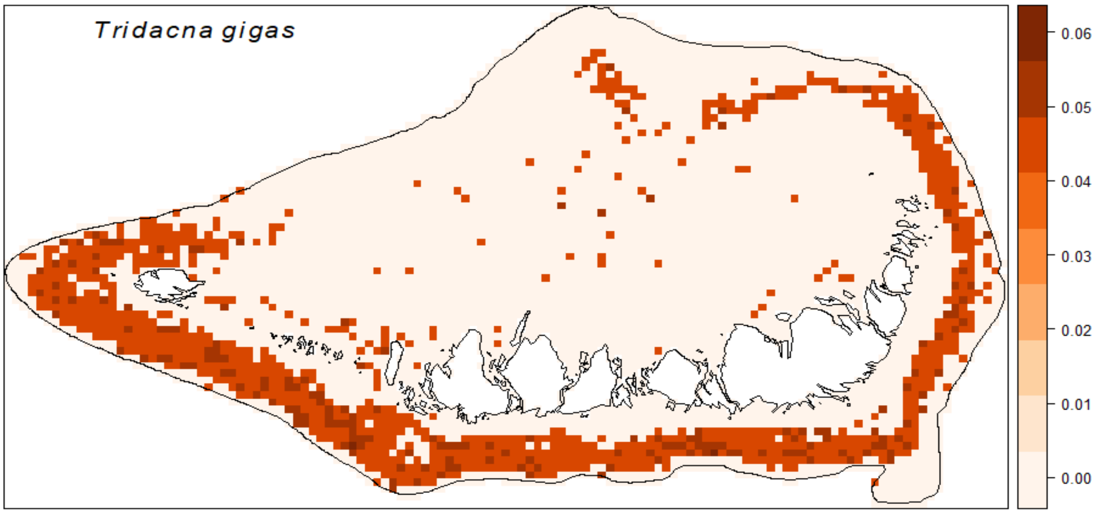


Figure 175. Posterior inter-quartile distance of areal density of *Tridacna gigas* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

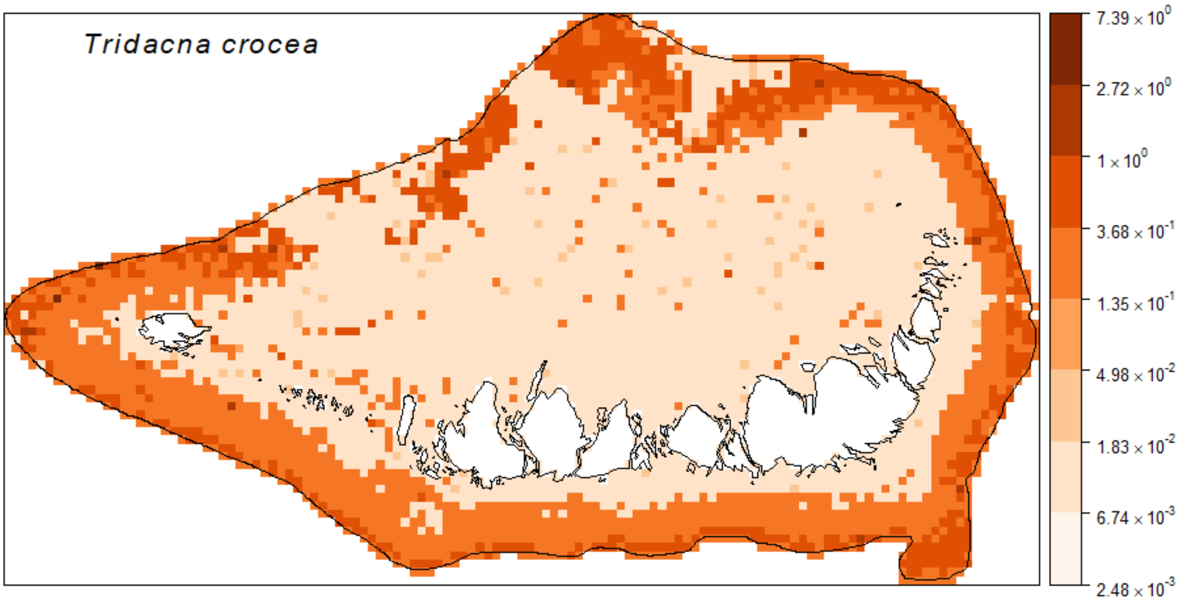


Figure 176. Posterior mean of areal density of *Tridacna crocea* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

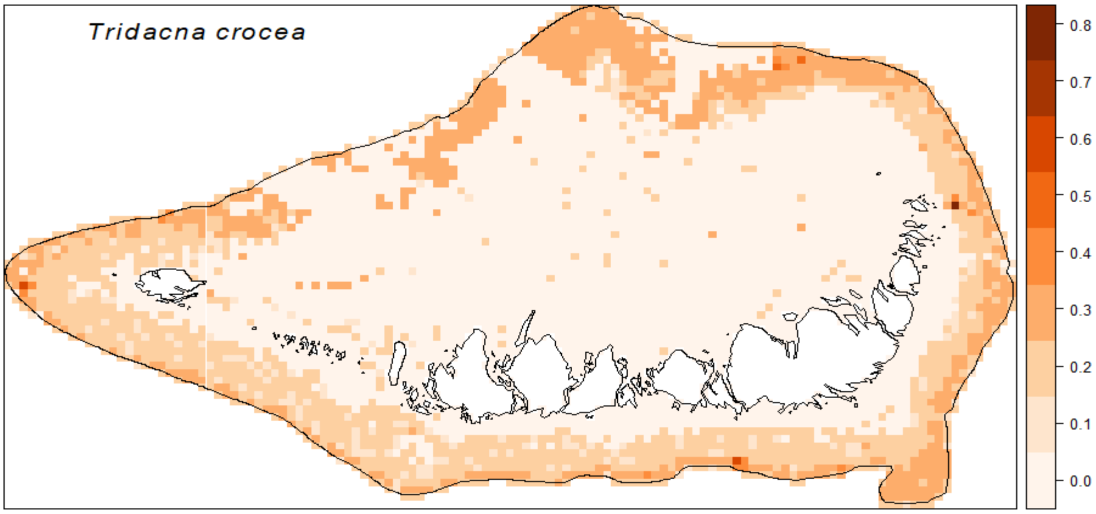


Figure 177. Posterior inter-quartile distance of areal density of *Tridacna crocea* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

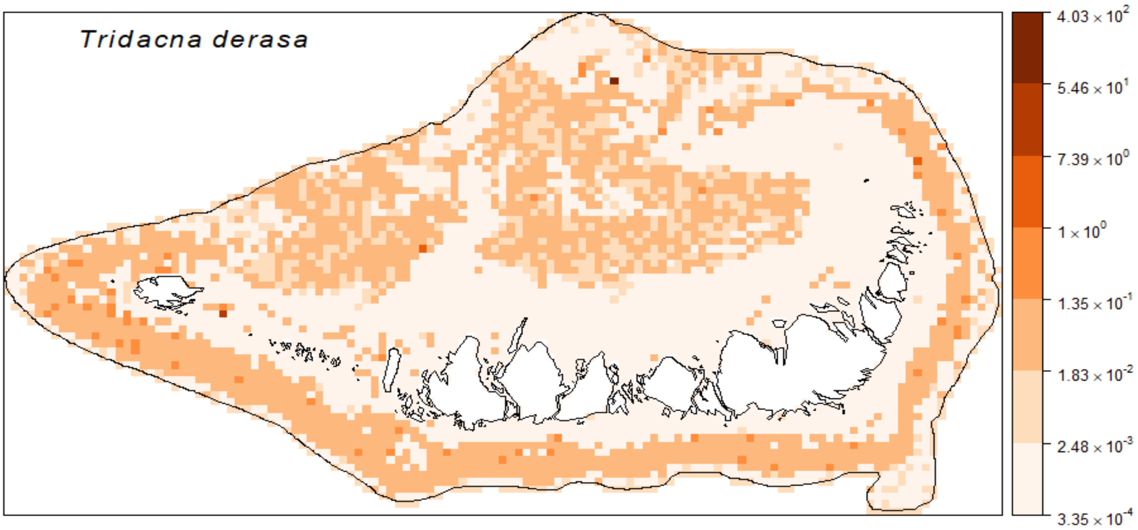


Figure 178. Posterior mean of areal density of *Tridacna derasa* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

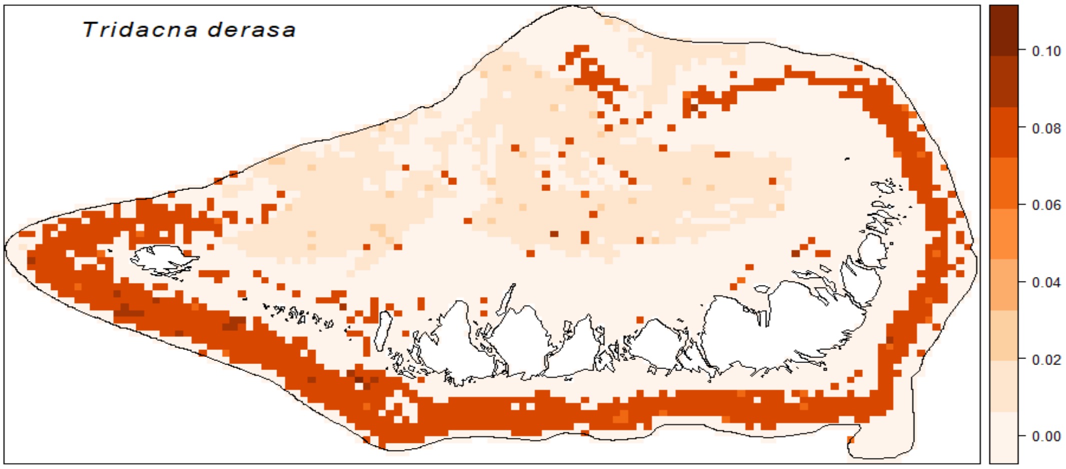


Figure 179. Posterior inter-quartile distance of areal density of *Tridacna derasa* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

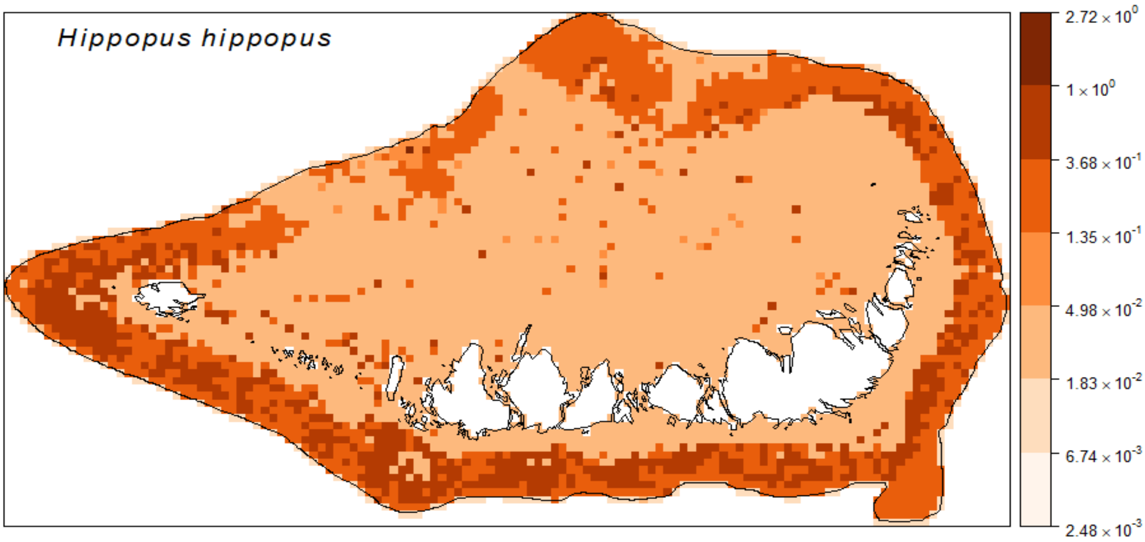


Figure 180. Posterior mean of areal density of *Hippopus hippopus* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

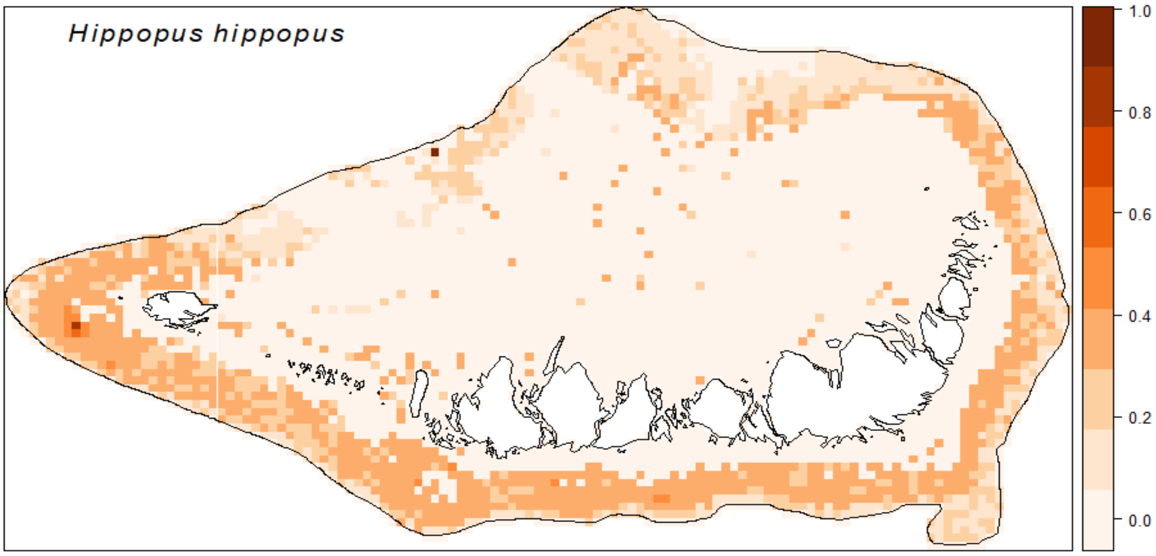


Figure 181. Posterior inter-quartile distance of areal density of *Hippopus hippopus* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

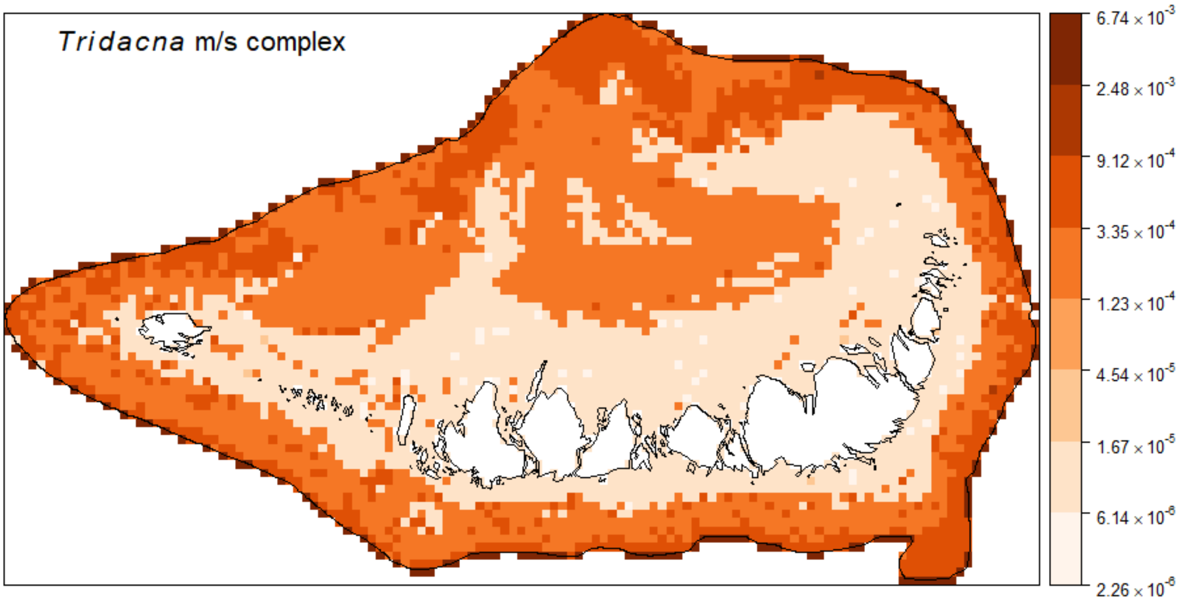


Figure 182. Posterior mean of areal density of *Tridacna maxima* and *Tridacna squamosa* combined (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

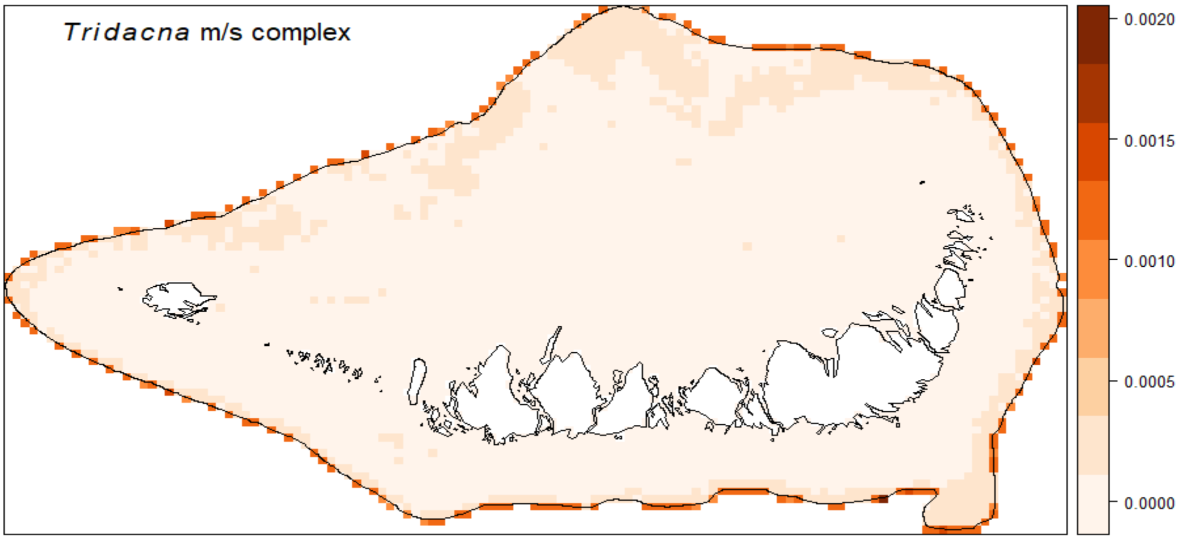


Figure 183. Posterior inter-quartile distance of areal density of *Tridacna maxima* and *Tridacna squamosa* combined (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

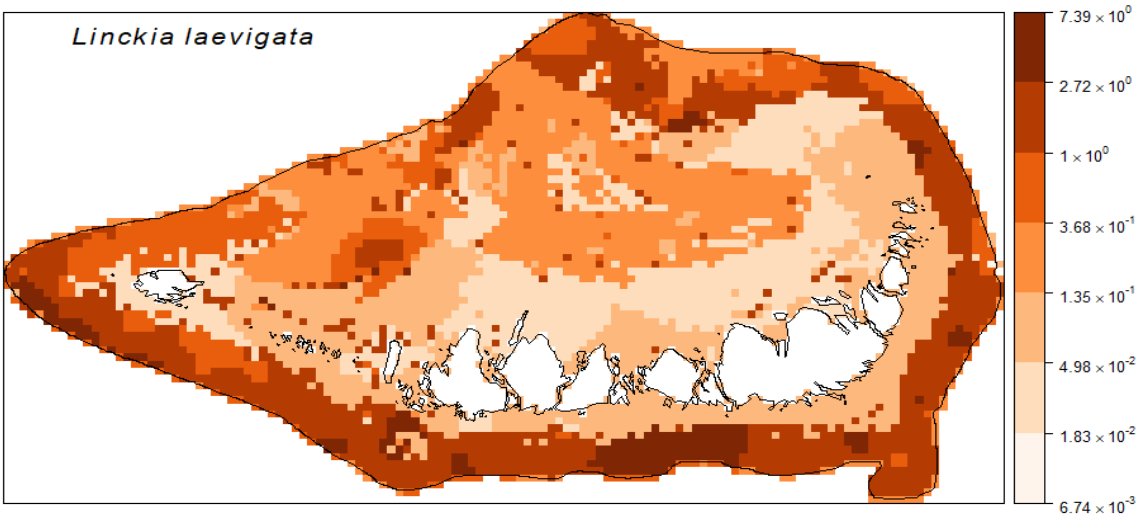


Figure 184. Posterior mean of areal density of *Linckia laevigata* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares). Note logarithmic legend axis.

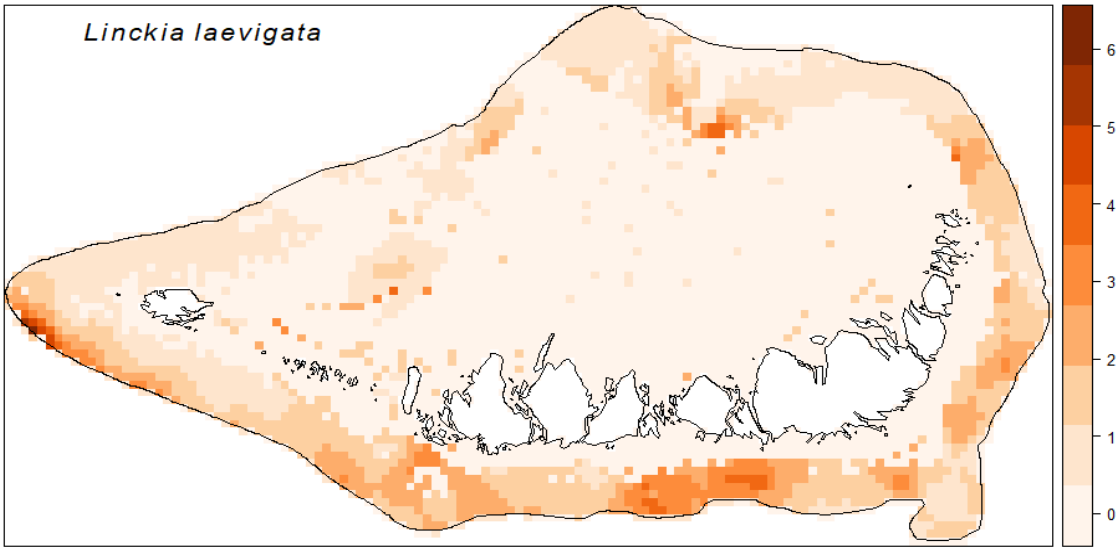


Figure 185. Posterior inter-quartile distance of areal density of *Linckia laevigata* (number per 100 square metres) in 2019. Grid cell size is 200 m x 200 m (4 hectares).

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.

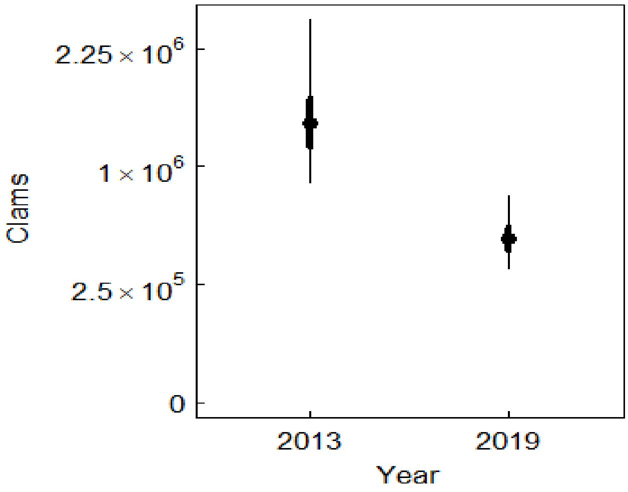


Figure 186. Estimated abundance of all clam species in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

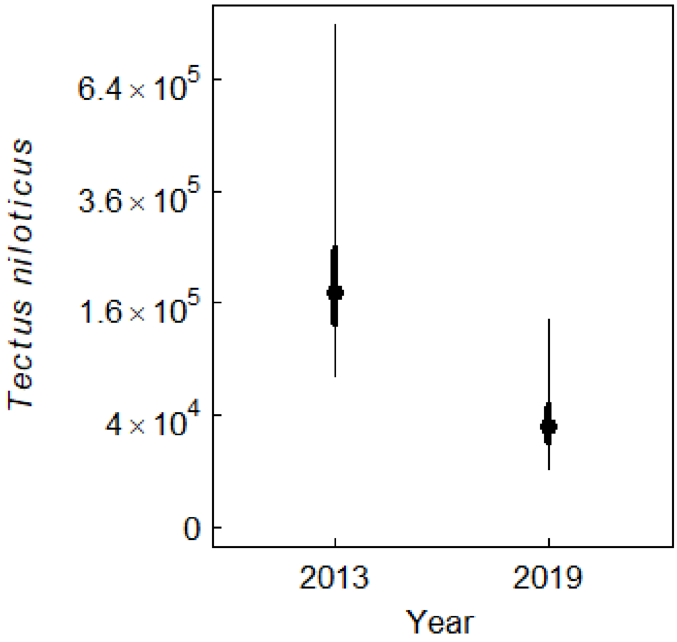


Figure 187. Estimated abundance of trochus in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

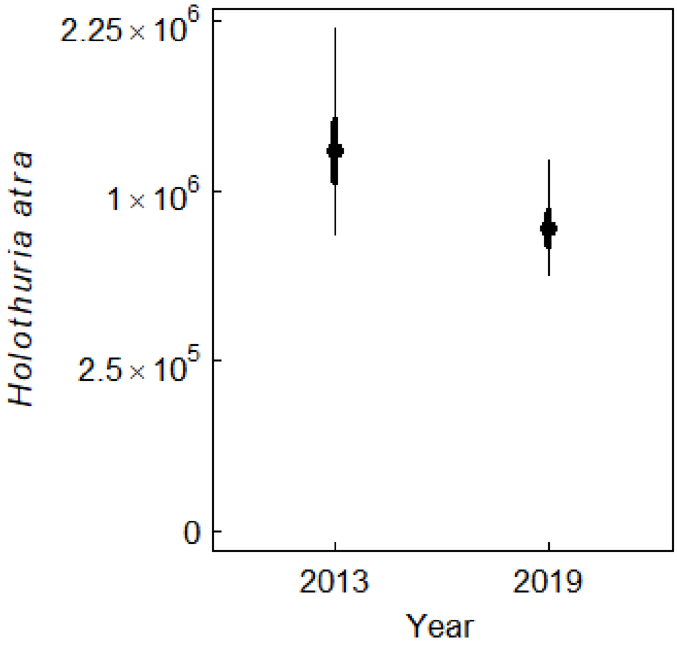


Figure 188. Estimated abundance of *Holothuria atra* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

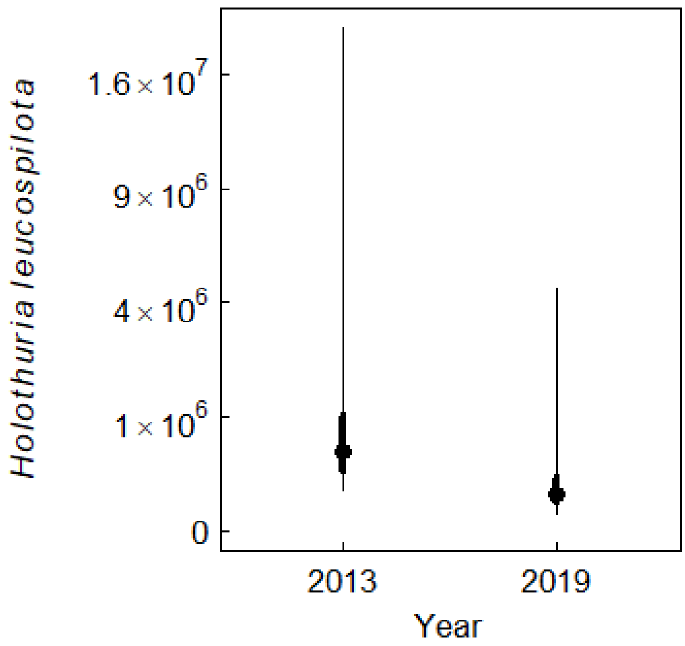


Figure 189. Estimated abundance of *Holothuria leucospilota* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

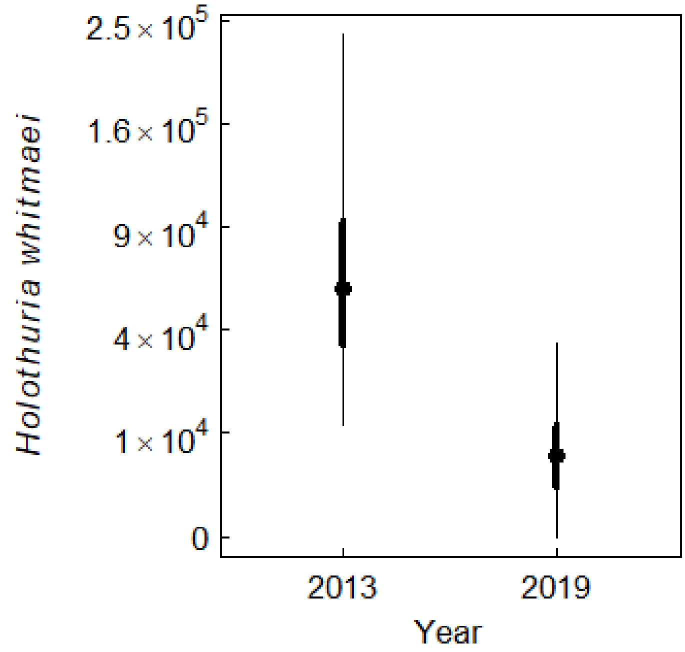


Figure 190. Estimated abundance of *Holothuria whitmaei* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

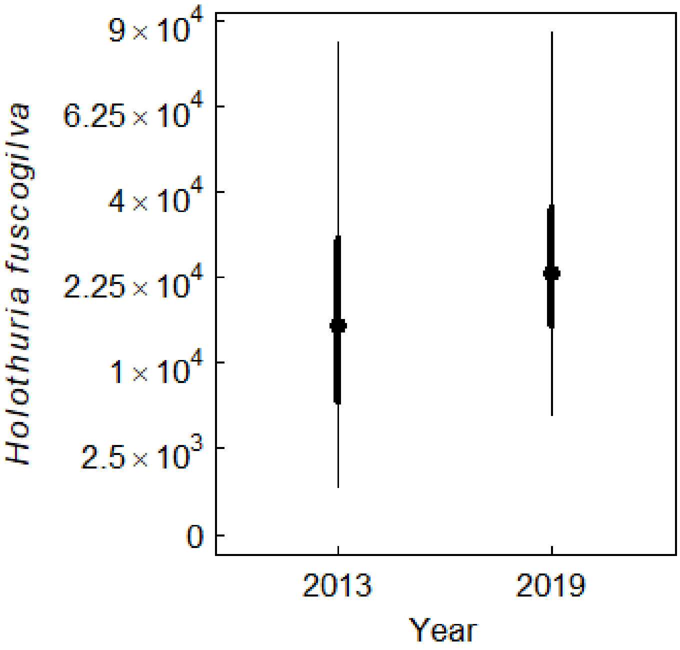


Figure 191. Estimated abundance of *Holothuria fuscogilva* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

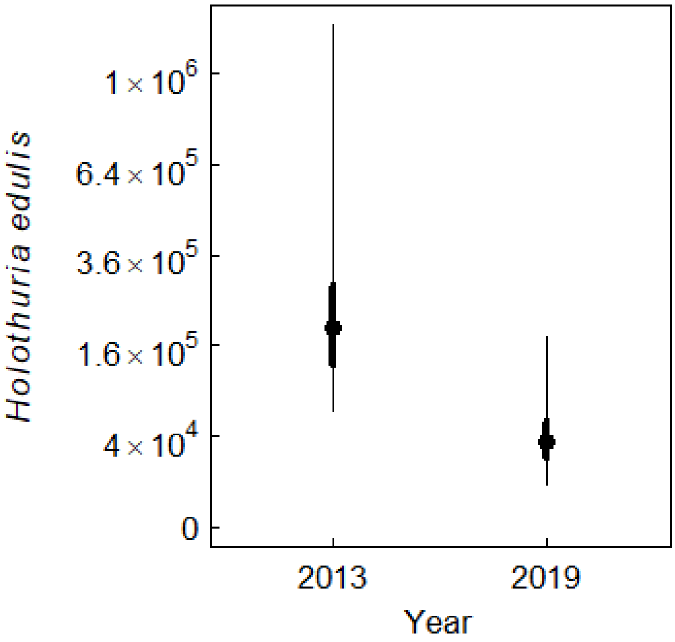


Figure 192. Estimated abundance of *Holothuria edulis* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

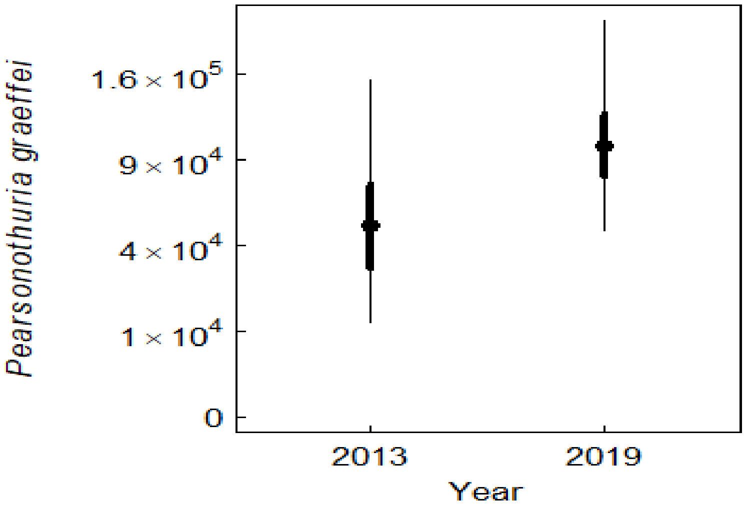


Figure 193. Estimated abundance of *Pearsonothuria graeffei* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

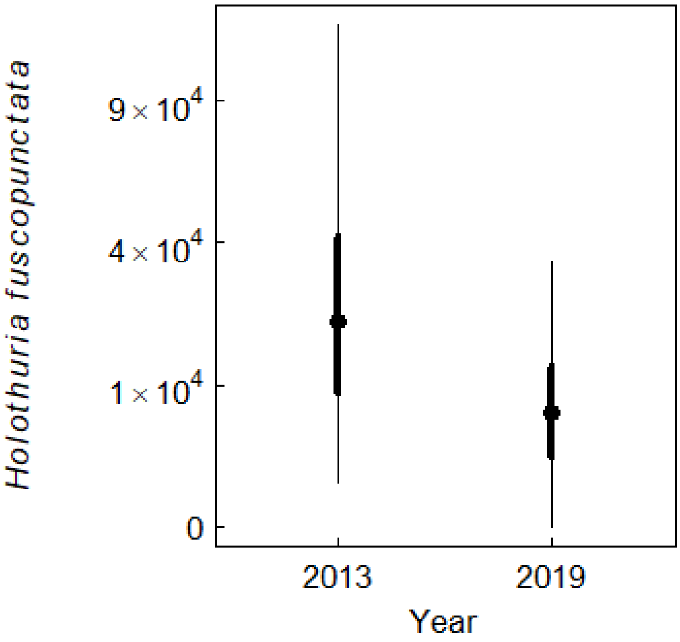


Figure 194. Estimated abundance of *Holothuria fuscopunctata* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

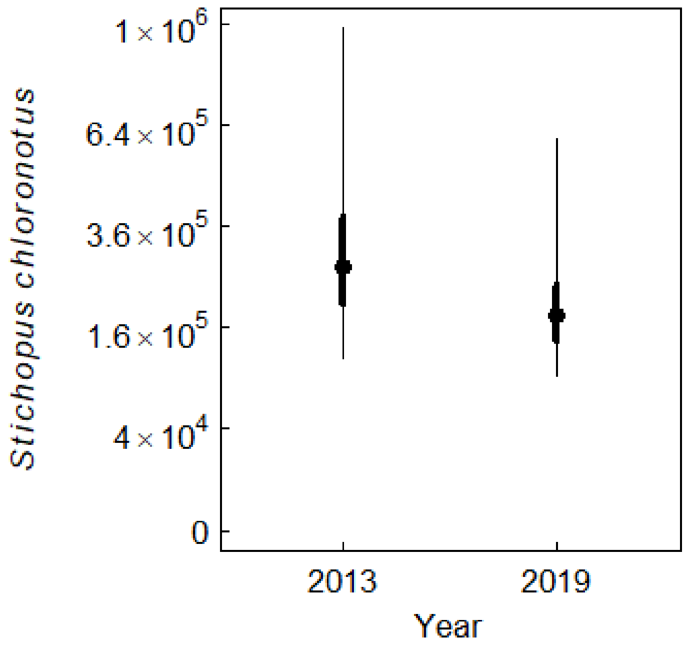


Figure 195. Estimated abundance of *Stichopus chloronotus* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

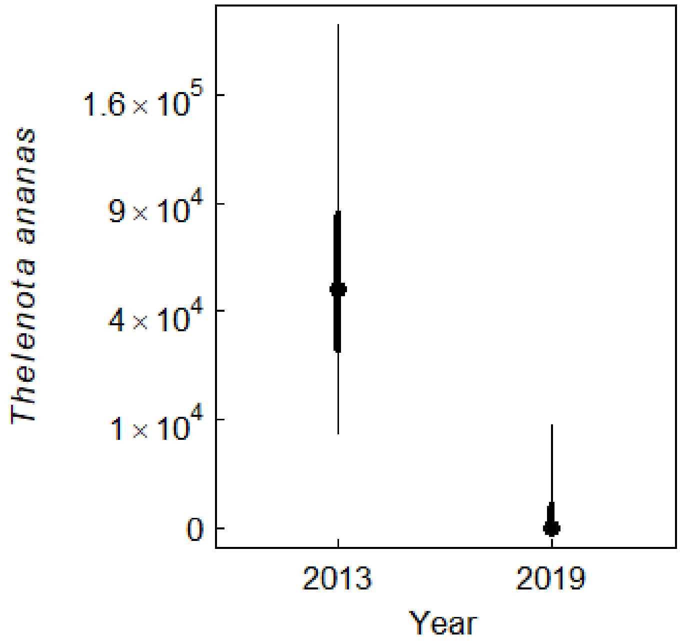


Figure 196. Estimated abundance of *Thelenota ananas* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

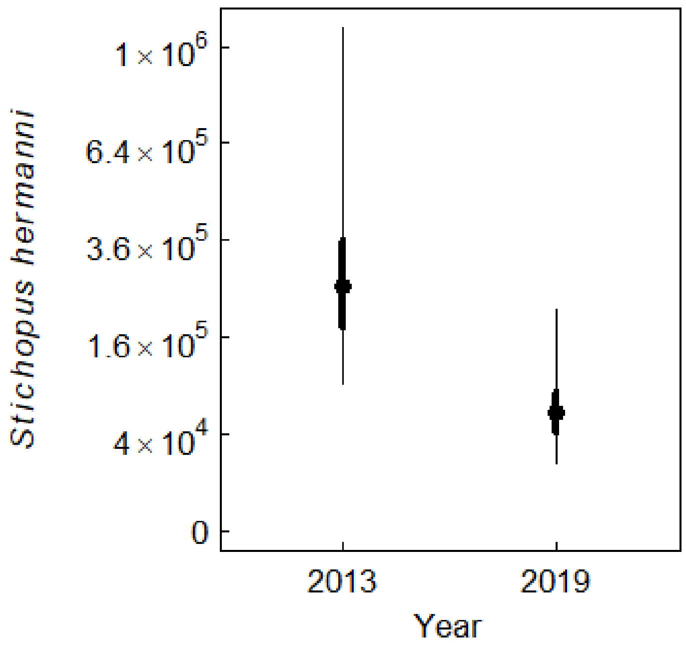


Figure 197. Estimated abundance of *Stichopus herrmanni* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

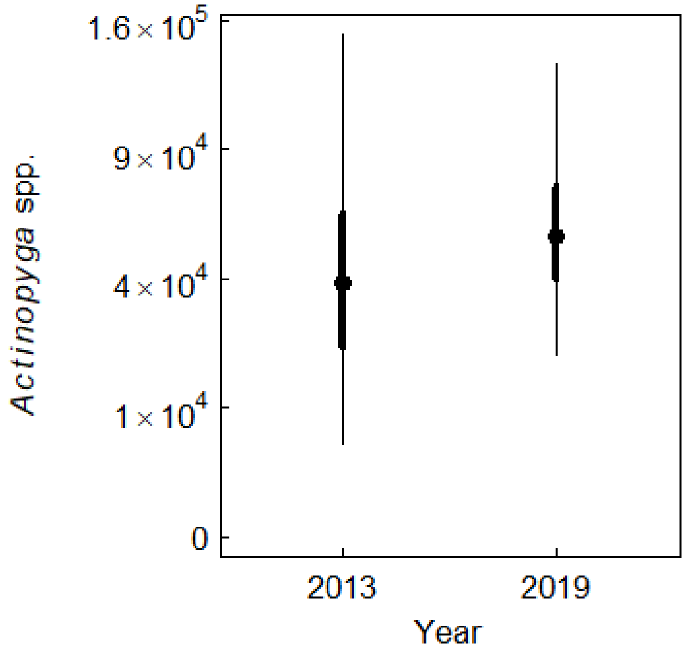


Figure 198. Estimated abundance of *Actinopyga* spp. in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

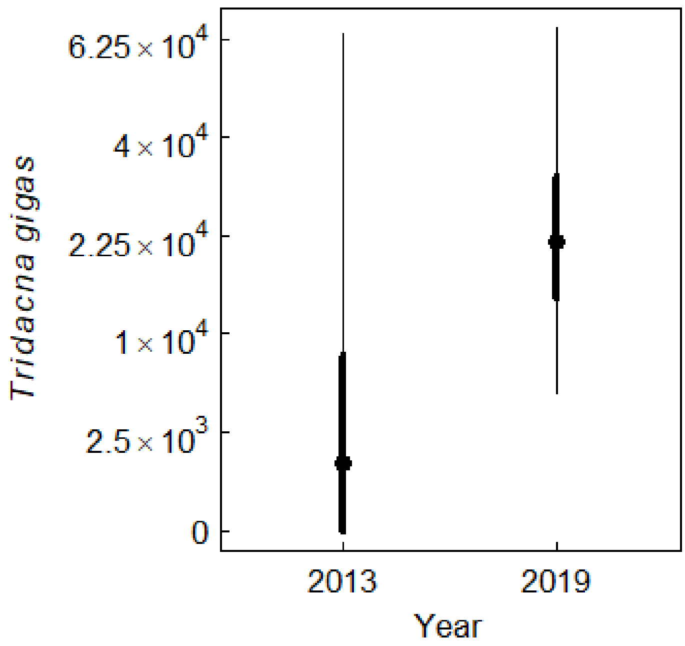


Figure 199. Estimated abundance of *Tridacna gigas* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

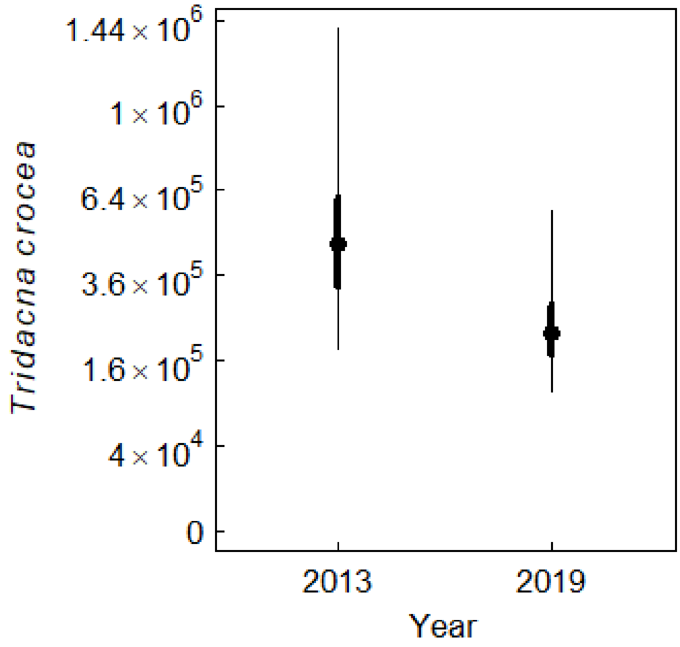


Figure 200. Estimated abundance of *Tridacna crocea* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

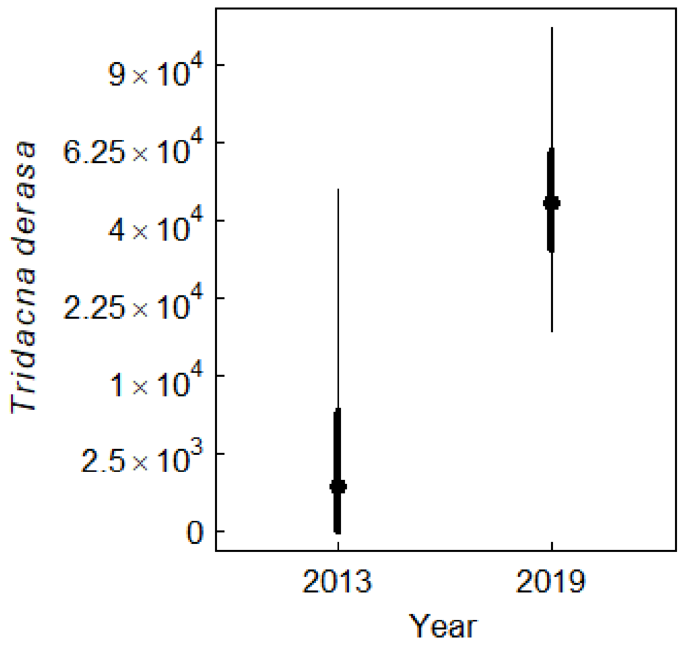


Figure 201. Estimated abundance of *Tridacna derasa* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

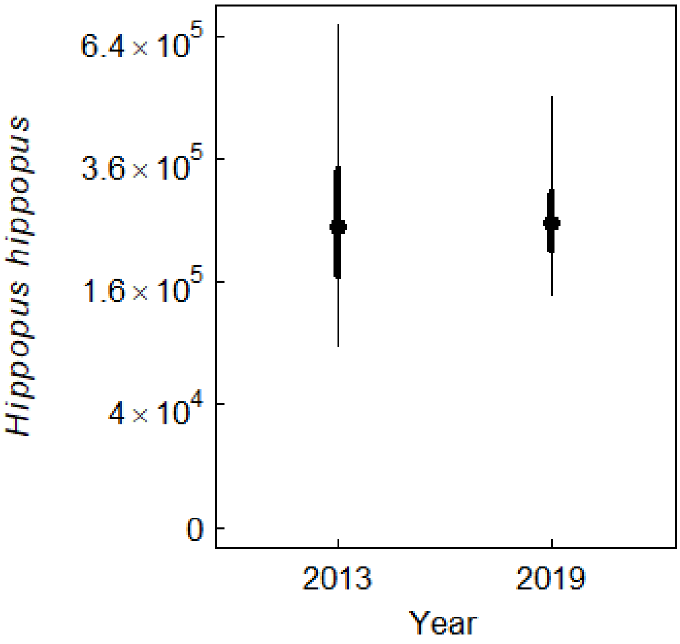


Figure 202. Estimated abundance of *Hippopus hippopus* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

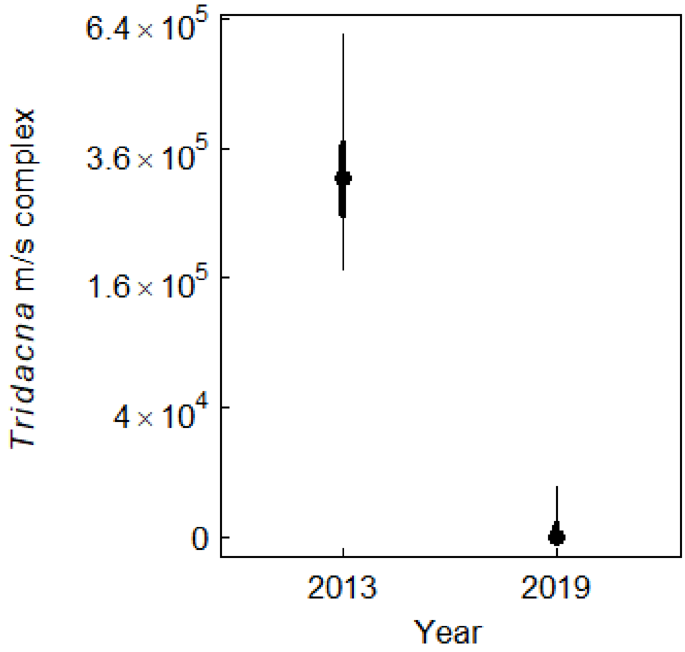


Figure 203. Estimated abundance of *Tridacna maxima* and *Tridacna squamosa* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

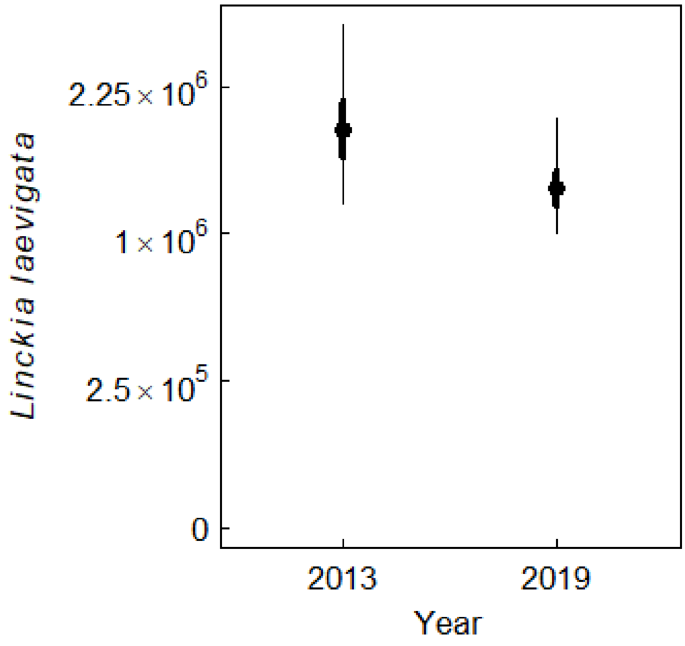


Figure 204. Estimated abundance of *Linckia laevigata* in 2013 and 2019. Estimates are posterior median (points), 0.50 central credible intervals (thick line segments) and 0.95 central credible intervals (thin line segments). Note square root scaled y-axis.

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^2 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^2 (range 88.9 to 600.0 shoots/m^2) across sites. Overall mean total biomass was 116.9 g DW/m^2 across all sites. Mean above ground biomass for *H. ovalis* was 0.015 g DW (± 0.003) and below ground biomass was 0.023 g DW (± 0.007) with an above and below ground ratio of 1:3. *Thalassia hemprichii* mean above (54.2 g DW/m^2 ± 1.24) and below (224.5 g DW/m^2 ± 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 (± 1.1) and 4.5 mm (± 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^2 with grazing rate calculated to be 0.4 g DW/m^2/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.

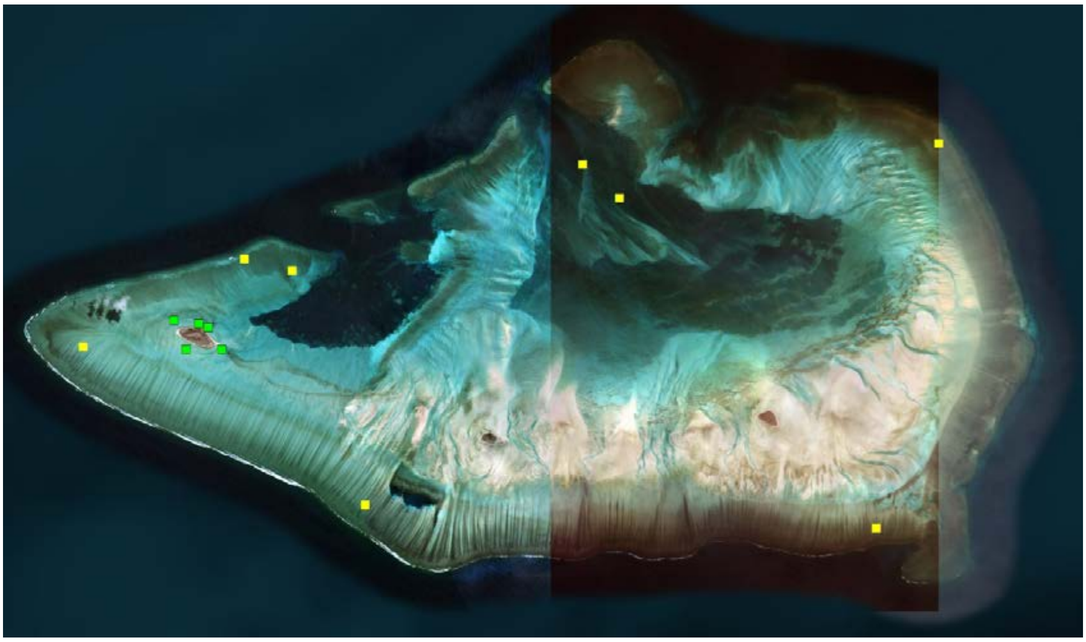


Figure 205. Pre-voyage planned seagrass monitoring sites (yellow) and growth rate sites (green dots around West Island) at Ashmore Reef based on historical studies. These were subsequently replanned following arrival and reconnaissance at Ashmore Reef in 2019.

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^2. Biomass per m^2 was then calculated by multiplying the biomass per shoot by the average shoot density for each 0.1 x 0.1 m quadrat.

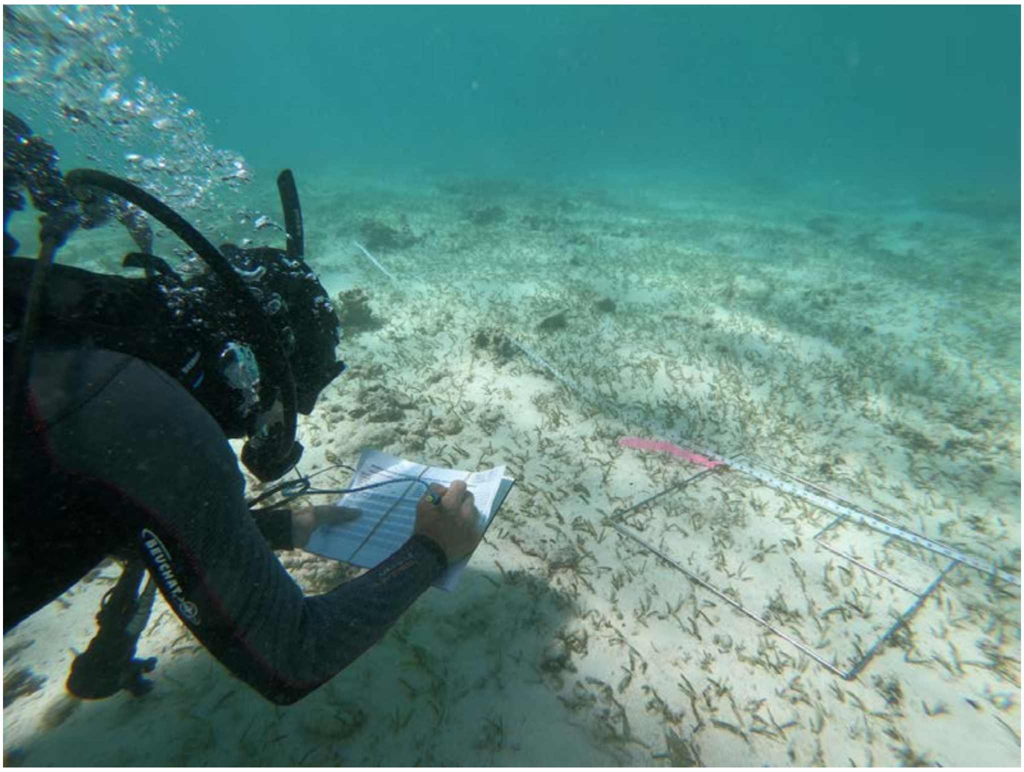


Figure 207. Diver conducting seagrass shoot density and biomass monitoring.

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

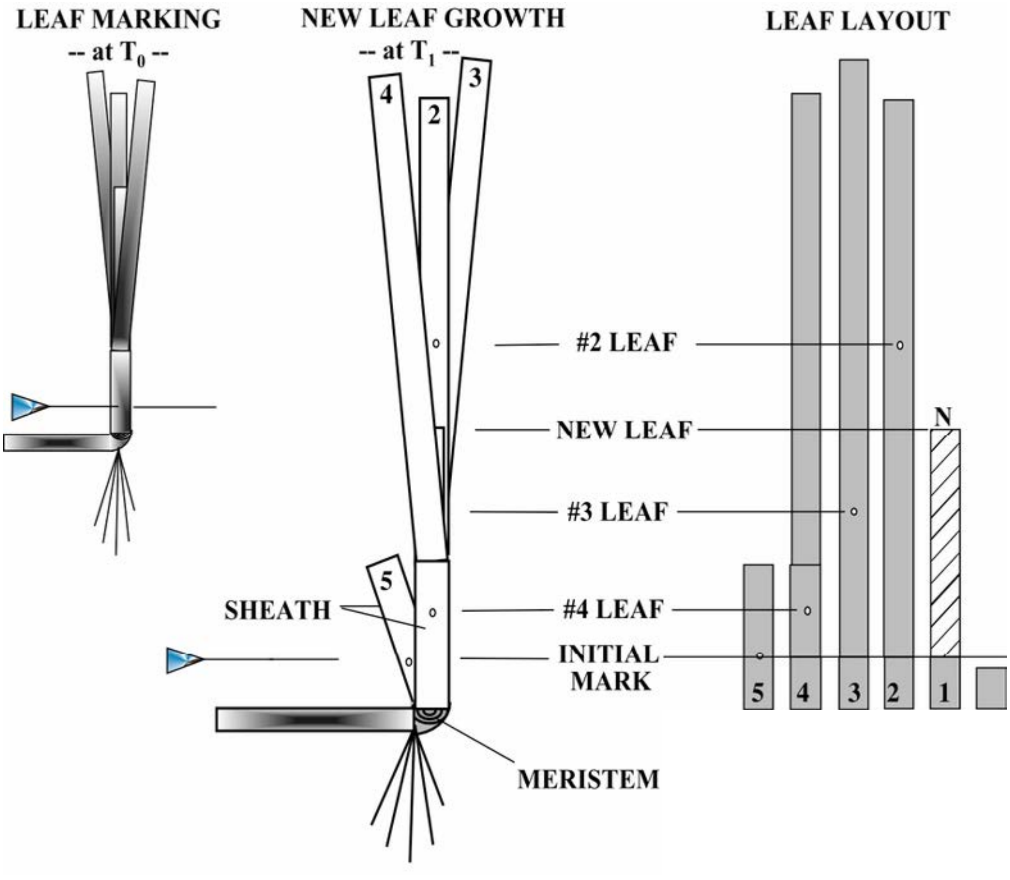


Figure 209. Schematic diagram of leaf growth measurement methodology showing initial mark and new growth (Short and Duarte 2001).

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%).

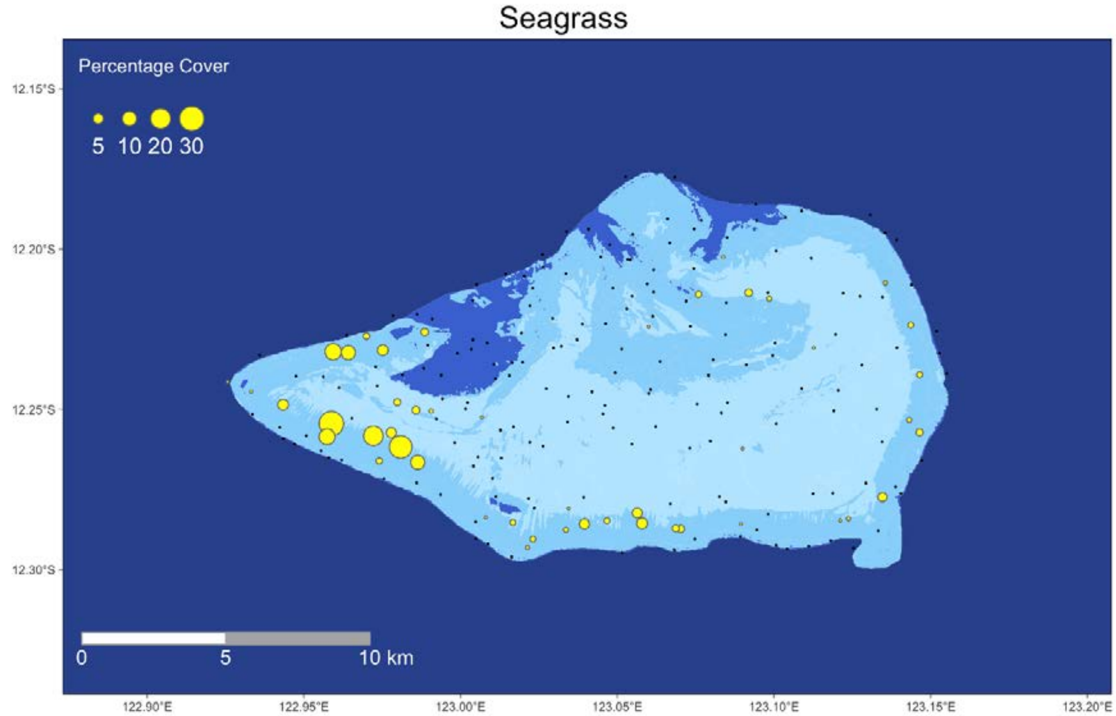


Figure 210. Percent cover of seagrass at Ashmore Reef measured along photo transects at 216 sites. See chapter 8 for details.

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.

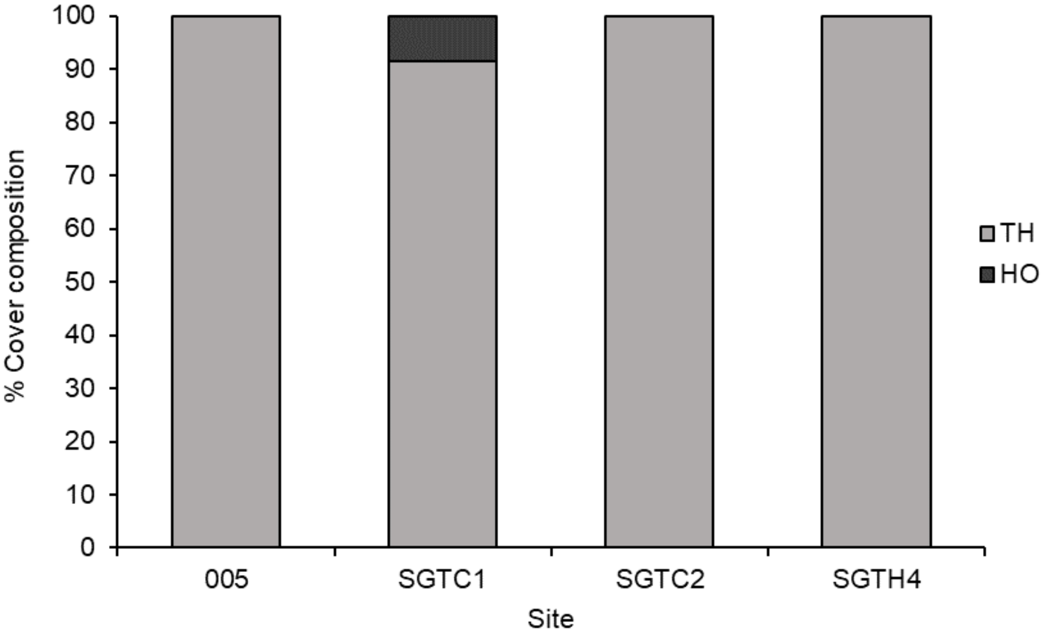


Figure 211. Percent cover composition of the two main seagrass species *T. hemprichii* and *H. ovalis* at each site.

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 ± 1.8 s.e.), while site 005 measured the shortest (40.2 mm ± 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 (± 2.3, range 20 to 50 mm).

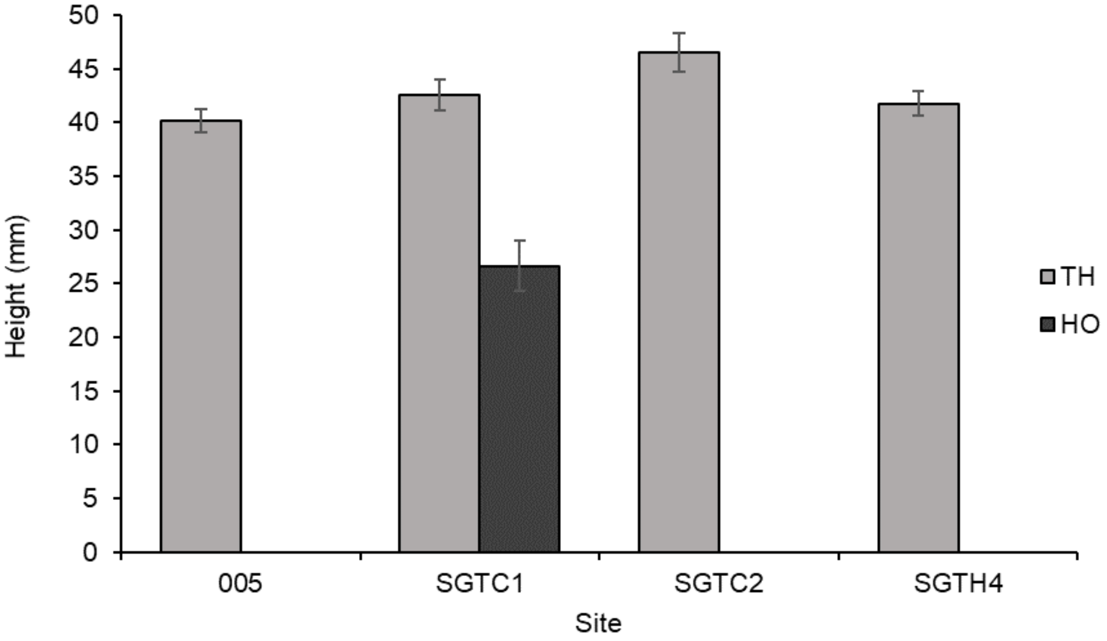


Figure 212. Mean canopy height (mm) of the two main seagrass species *T. hemprichii* and *H. ovalis* at each site. Error bars indicate ± 1 s.e.

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

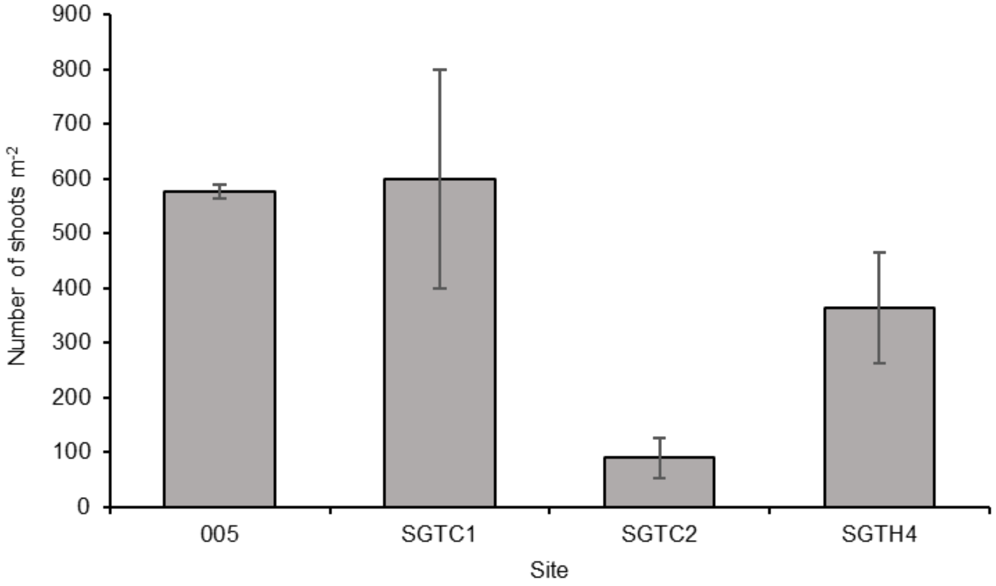


Figure 213. Mean shoot density per m^2 for *T. hemprichii* at each site. Error bars indicate ± 1 s.e.

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^2 across all sites. Site 005 showed the greatest mean total biomass with transect 3 the highest of all transects (290.4 g DW/m^2 ± 54.2 s.e.). The lowest mean total biomass was recorded for site SGTC2 transect 3 (5.1 g DW/m^2 ± 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^2) 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^{-1} (± 0.003 s.e) while below ground biomass (dry root/rhizome weight) was 0.023 g DW shoot^{-1} (± 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^2) was determined at four sites for *T. hemprichii* (Figure 214B). Mean above (54.2 g DW/m^2 ± 1.24 s.e.) and below (224.5 g DW/m^2 ± 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^2 ± 5.1) and below ground biomass (7.8 g DW/m^2 ± 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.

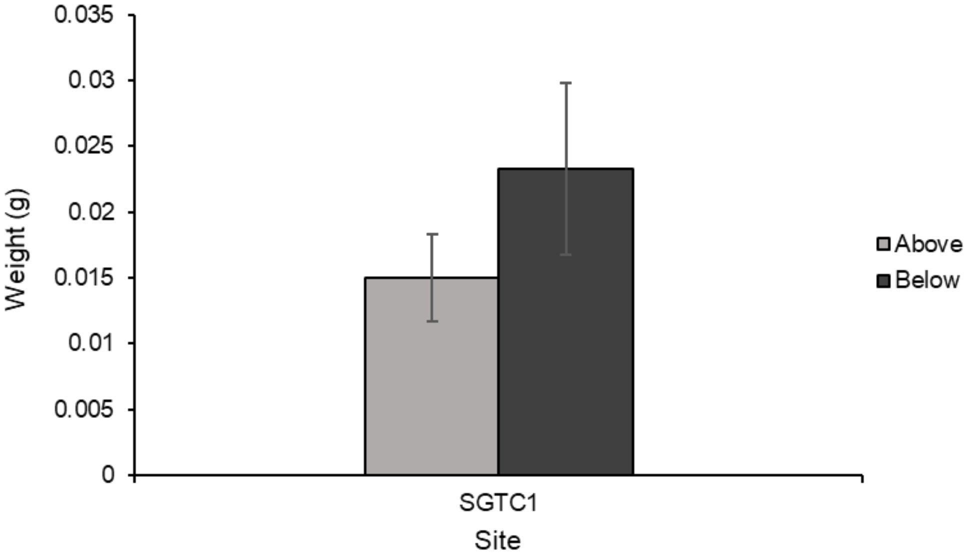


Figure 214. Above and below ground biomass (leaf/sheath and root/rhizome weight (g DW)) for *H. ovalis*. Error bars indicate ± 1 s.e. Dry to wet weight conversion = 7.08 (leaf) and 5.86 (root/rhizome) for *H. ovalis*.

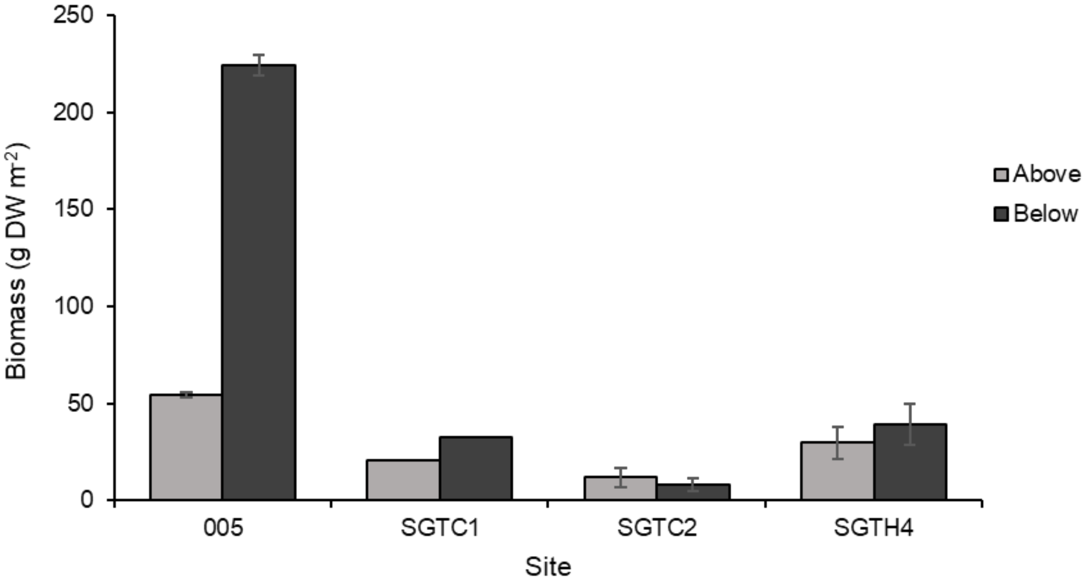


Figure 215. Above and below ground biomass (leaf/sheath and root/rhizome weight (g DW/m^2)) of *T. hemprichii*. Error bars indicate ± 1 s.e. Dry to wet weight conversion = 9.56 (leaf) and 4.63 (root/rhizome).

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 (± 0.1 s.e.) while at site SGTC2 mean growth was 2.2 mm/day (± 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 (± 0.3 s.e.) while at SGTC2 shoot growth was slightly lower at 3.2 mm/day (± 0.6 s.e.).

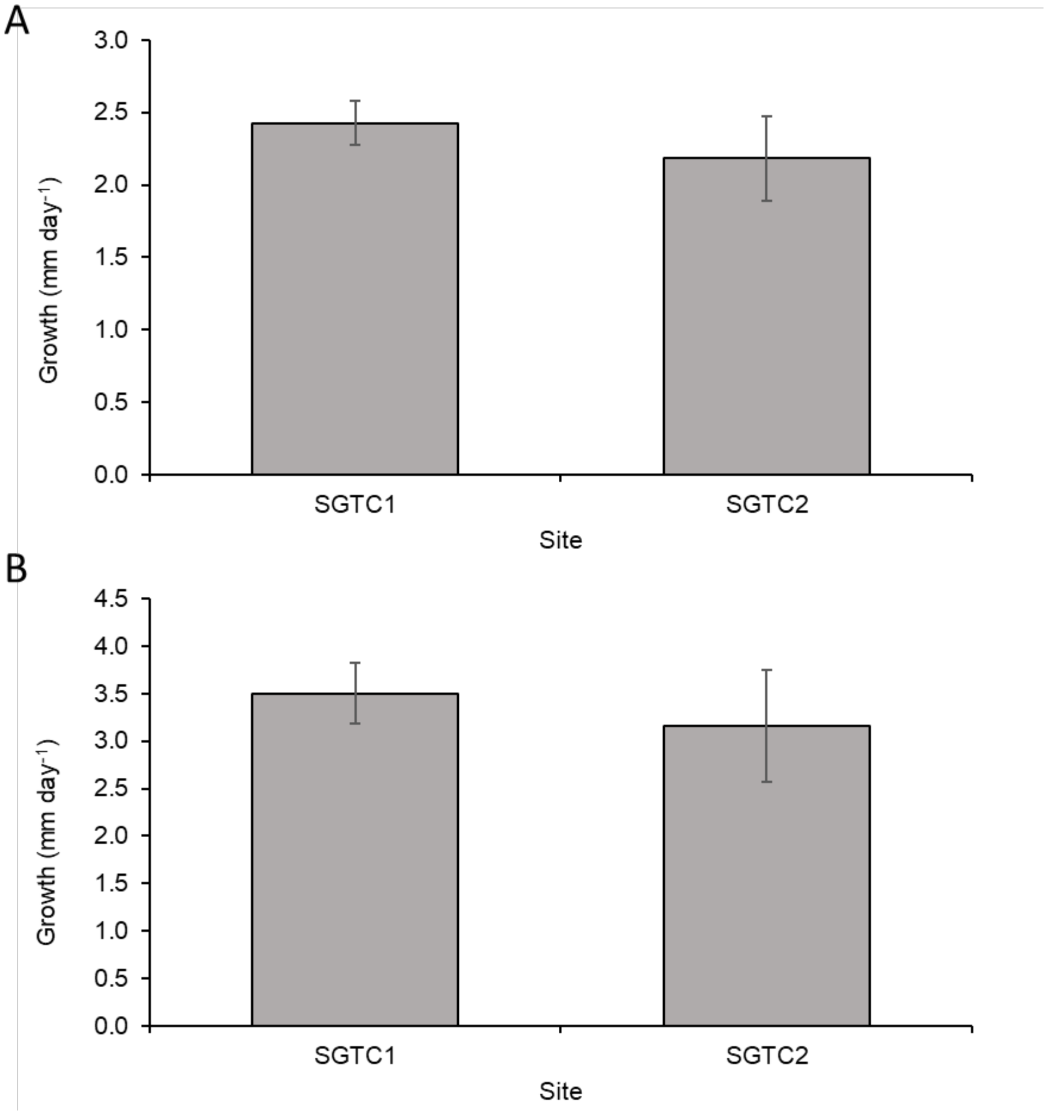


Figure 216. Mean growth (mm/day) per site for *T. hemprichii*: B) mean individual leaf growth, B) mean shoot growth (each comprising multiple leaves). Note: shoot growth does not include new leaves or leaves broken/grazed below punched hole. Error bars indicate ± 1 s.e.

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^2. With an average turnover rate of 19.4 days, it was possible to determine that approximately 0.4 g DW/m^2 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^2 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 sparely 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^2 across sites. Although overall mean density of 407 shoots/m^2 was lower than the 605 shoots/m^2 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^2 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^2 (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^2 – 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 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^2 (range 5.1 to 290.4 g DW/m^2) was obtained for *T. hemprichii* across all sites. A mean above ground biomass of 32.2 g DW/m^2 was recorded while below ground biomass was 85.6 g DW/m^2, 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 Scripts 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^2 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 *et al.* (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 *et al.* (2013)
2006	0.9		Guinea (2007)
2006	2.6	7	Lukoschek *et al.* (2013)
2007	0.2	1	Guinea (2007)
2008	0.2		Guinea (2007)
2009		2	Lukoschek *et al.* (2013)
2010		3	Lukoschek *et al.* (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.

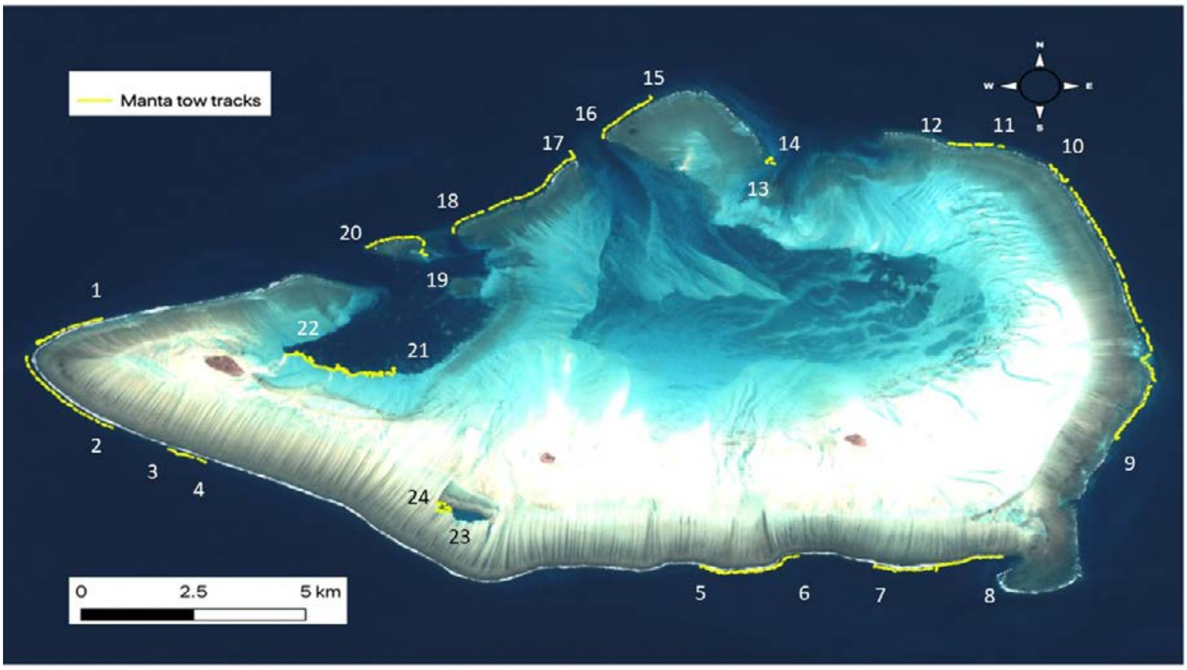


Figure 217. Manta tow transects undertaken for sea snake surveys at Ashmore Reef in 2019.

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.

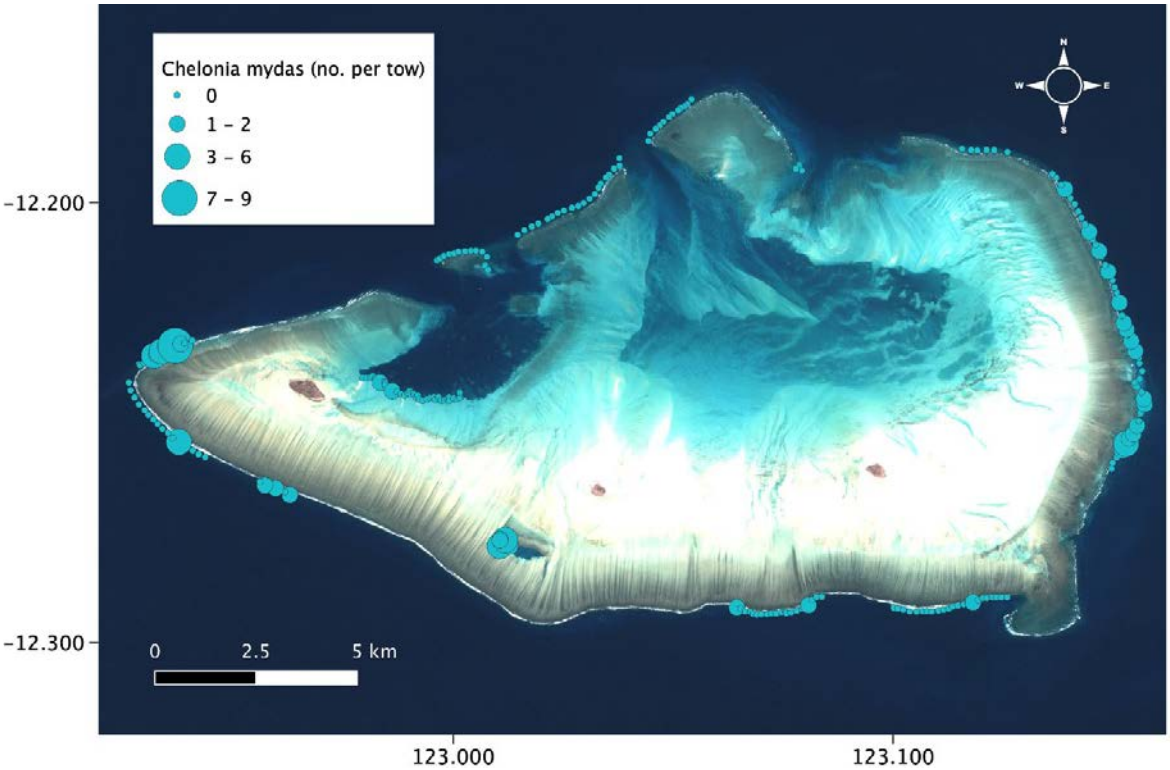


Figure 219. Map of Ashmore Reef showing the number of green turtles (*Chelonia mydas*) per 2-minute manta tow.

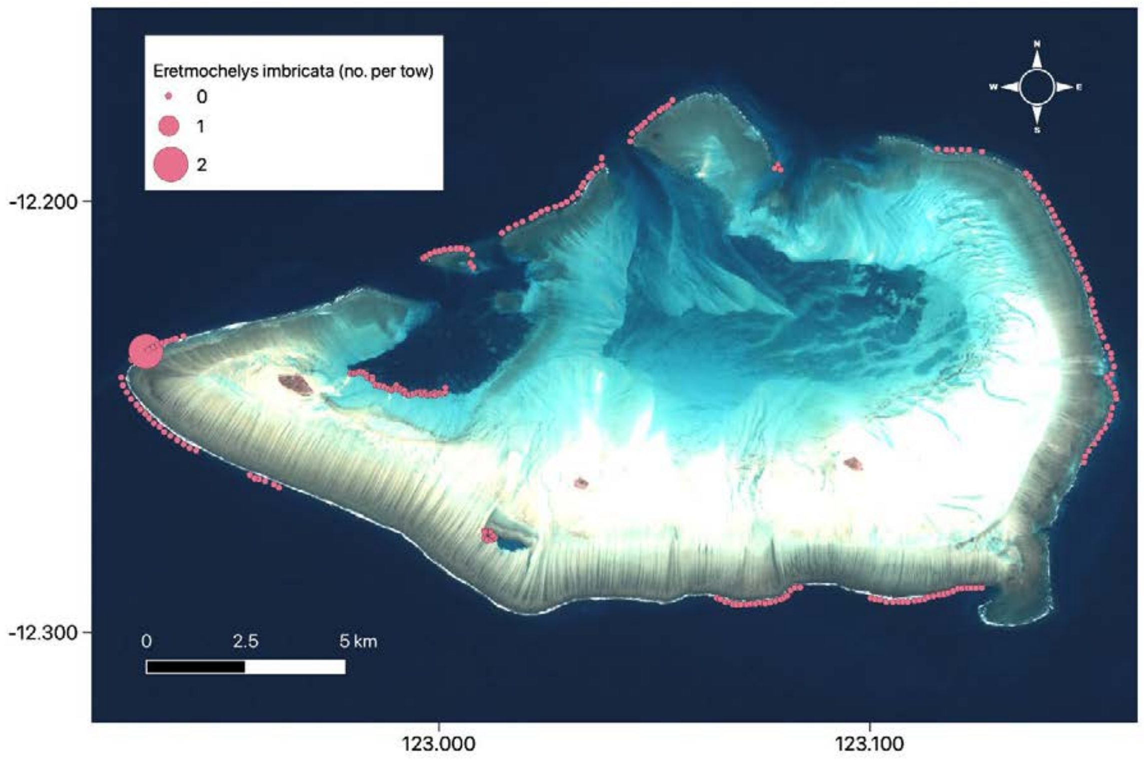


Figure 220. Map of Ashmore Reef showing the number of hawksbill turtles (*Eretmochelys imbricata*) per 2-minute manta tow

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).

15 REFERENCES

Agawin, N.S.R., Duarte, C.M. & Fortes, M.D. (1996) Nutrient limitation of Philippine seagrasses (Cape Bolinao, NW Philippines): in situ experimental evidence. *Marine Ecology Progress Series,* **138,** 233-243.

Alevizon, W., Richardson, R., Pitts, P. & Serviss, G. (1985) Coral zonation and patterns of community structure in Bahamian reef fishes. *Bulletin of Marine Science* **36,** 304-318.

Allen, G.R. (1993) Fishes of Ashmore reef and Cartier Island. . *Marine faunal surveys of Ashmore and Cartier Island, north-western Australia. In: Berry, P.F. (ed.) Records of the Western Australian Museum Supplement* **44,** 1-11.

Allen, G.R. (2008) Conservation hotspots of biodiversity and endemism for Indo-Pacific coral reef fishes. *Aquatic Conservation: Marine and Freshwater Ecosystems,* **18,** 541-556.

Anderson, S.C., Flemming, J.M., Watson, R. & Lotze, H.K. (2011) Serial exploitation of global sea cucumber fisheries. *Fish and Fisheries,* **12,** 317-339.

Austin, D. (2014) Sendera-clandi (*Xenostegia tridentata* (L.) D.F. Austin & Staples, Convolvulaceae: A medicinal creeper. *Ethnobotany Research and Applications,* **12,** 433-454.

Bach, S.S., Borum, J., Fortes, M.D. & Duarte, C.M. (1998) Species composition and plant performance of mixed seagrass beds along a siltation gradient at Cape Bolinao, The Philippines. *Marine Ecology Progress Series,* **174,** 247-256.

Banerjee, S., Gelfand, A.E., Finley, A.O. & Sang, H. (2008) Gaussian predictive process models for large spatial data sets. *Journal of the Royal Statistical Society: Series B (Statistical Methodology),* **70,** 825-848.

Barker, R.M. (1998) A trial key and notes on Tribulus (*Zygophyllaceae*) in Australia, including one new species and validation of *Tribulus suberosus*. *Nuytsia,* **12,** 9-35.

Barley, S.C., Meekan, M.G. & Meeuwig, J.J. (2017) Species diversity, abundance, biomass, size and trophic structure of fish on coral reefs in relation to shark abundance. *Marine Ecology Progress Series,* **565,** 163-179.

Barnes, D.J. & Devereux, M.J. (1984) Productivity and calcification on a coral reef: A survey using pH and oxygen electrode techniques. *Journal of Experimental Marine Biology and Ecology,* **79,** 213-231.

Barnett, A., Braccini, M., Awruch, C. & Ebert, D. (2012) An overview on the role of Hexanchiformes in marine ecosystems: Biology, ecology and conservation status of a primitive order of modern sharks. *Journal of Fish Biology,* **80,** 966-990.

Barton, D.P. (2007) Pentastomid parasites of the introduced asian house gecko, *Hemidactylus frenatus* (Gekkonidae), in Australia. *Comparative parasitology,* **74,** 254-260.

Batianoff, G.N., Naylor, G.C., Olds, J.A., A, F.N. & Neldner, V.J. (2010) Climate and Vegetation Changes at Coringa-Herald National Nature Reserve, Coral Sea Islands, Australia. *Pacific Science,* **64**.

Bean, A.R. (2007) A new system for determining which plant species are indigenous in Australia. *Australian Systematic Botany,* **20,** 1-43.

Bell, J.D., Purcell, S.W. & Nash, W.J. (2008) Restoring small-scale fisheries for tropical sea cucumbers. *Ocean & Coastal Management,* **51,** 589-593.

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

Bellio, M.G., Bayliss, P., Williams, A.J., van Dam, R., Fox, G.J. & Moulen, J.H. (2007) A preliminary ecological risk assessment of the impact of tropical fire ants (*Solenopsis geminata*) on colonies of seabirds at Ashmore Reef. *Report to Department of Environment and Water Resources Supervising Scientist*. Darwin NT.

Bennett, F. (1989) *Microlarinus lypriformis* (Coleoptera: Curculionidae) in Curacao, Venezuela, and Puerto Rico: new distribution records. *Coleopterists Bulletin,* **43,** 390-391.

Berry, O., Richards, Z., Moore, G., Hernawan, U., Travers, M. & Gruber, B. (2019) Oceanic and coastal populations of a harvested macroinvertebrate Rochia nilotica in north-western Australia are isolated and may be locally adapted. *Marine and Freshwater Research,* **71,** 782-793.

Berry, P.F. (1993) Part 1: Historical background description of the physical environments of Ashmore Reef and Cartier Island and notes on exploited species Marine Faunal Surveys of Ashmore Reef and Cartier Island. *Records of the Western Australian Museum,* **Supplement No.44,** 1-11.

Bevan, E., Whiting, S., Tucker, T., Guinea, M., Raith, A. & Douglas, R. (2018) Measuring behavioral responses of sea turtles, saltwater crocodiles, and crested terns to drone disturbance to define ethical operating thresholds. *PLoS One,* **13,** 3.

BirdLife, I. (2010) Marine Important Bird Areas: priority sites for the conservation of biodiversity. *Cambridge, UK: BirdLife International*.

Black, A. (2005) Light induced seabird mortality on vessels operating in the Southern Ocean: incidents and mitigation measures. *Antarctic Science,* **17,** 67-68.

Borrelle, S.B. & Fletcher, A.T. (2017) Will drones reduce investigator disturbance to surface-nesting seabirds? *Marine Ornithology,* **45,** 89-94.

Bouillon, S. & Connolly, R.M. (2009) Carbon exchange among tropical coastal ecosystems. *Ecological Connectivity among Tropical Coastal Ecosystems*, pp. 45-70. Springer Netherlands.

Bour, W., Gohin, F. & Bouchet, P. (1982) Croissance et mortalité naturelle des trocas (*Trochus niloticus L.*) de Nouvelle Calédonie (Mollusca, Gastropoda). *Haliotis,* **12,** 71–90.

Bradbury, R.H., Reichelt, R.E., Meyer, D.L. & Birtles, R.A. (1987) Patterns in the distribution of the crinoid community at Davies Reef on the central Great Barrier Reef. *Coral Reefs,* **5,** 189-196.

Braun-Blanquet, J. (1932) *Plant sociology. The Study of Plant Communities*. New York, NY, US.

Breed, A.C., Field, H.E., Smith, C.S., Edmonston, J. & Meers, J. (2010) Bats Without Borders: Long-Distance Movements and Implications for Disease Risk Management. *EcoHealth,* **7,** 204-212.

Breen, P. (2011) Review of seven 1998–2009 surveys of trochus and holothurians at Ashmore Reef and Cartier Island. *For Temperate Marine Conservation Branch, Department of Sustainability, Environment, Water, Population and Communities*. Technical report. Breen Consulting. .

Brisson-Curadeau, E., Bird, D.E., Burke, C., Fifield, D.A., Pace, P., Sherley, R.B. & Elliott, K.H. (2017) Seabird species vary in behavioural response to drone census. *Scientific reports,* **7,** 17884.

Brouns, J.J.W.M. (1985) A comparison of the annual production and biomass in three monospecific stands of the seagrass *Thalassia hemprichii* (Ehrenb.) aschers. *Aquatic Botany,* **23,** 149-175.

Brown, A.L. (1990) Measuring the effect of aircraft noise on sea birds. *Environment International,* **16,** 587-592.

Brown, B.E. & Suharsono (1990) Damage and recovery of coral reefs affected by El Niño related seawater warming in the Thousand Islands, Indonesia. *Coral Reefs,* **8,** 163-170.

Brown, G.R. (1999) *A preliminary report on the insects of Ashmore Reef Nature Reserve*. Museums & Art Galleries of the Northern Territory.

Brown, K. & Skewes, T.D. (2005) A preliminary assessment of the ecology of seagrasses at Ashmore Reef. *The Beagle***,** 143–152.

Brown, K.M. (2001) Ashmore Reef survey field report. *A study funded by Environment Australia and Adelaide University*. University of Adelaide, Adelaide.

Brown, L., H, J. & Raphael, B. (2008) Northern Australia quarantine strategy: weeds target list. ABARES.

Bruno, J.F. & Selig, E.R. (2007) Regional decline of coral cover in the Indo-Pacific: timing, extent, and subregional comparisons. *PLoS One,* **2,** e711.

Burt, J., Bartholomew, A. & Usseglio, P. (2008) Recovery of corals a decade after a bleaching event in Dubai, United Arab Emirates. *Marine biology,* **154,** 27-36.

Carlquist, S. (1967) The biota of long-distance dispersal. V. Plant dispersal to Pacific Islands. *Bulletin of the torrey Botanical Club,* **94,** 129-162.

Ceccarelli, D., Kospartov, M., Beger, M., Richards, Z. & Birrell, C.L. (2007) Ashmore Reef 2006 survey of the distribution and abundance of Trochus, holothurians and tridacnid clams, to assess the impact of recent illegal fishing on target invertebrate stocks. Technical report, prepared by C&R Consulting for the Department of Environment and Water Resources, Canberra.

Ceccarelli, D., Williamson, D., Ayling, T., Ayling, A. & Pratchett, M. (2013) Ashmore Reef National Nature Reserve marine survey 2013: methods field test. *JCU Report to Department of Sustainability, Environment, Water, Population & Communities***,** 39.

Ceccarelli, D.M., Beger, M., Kospartov, M.C., Richards, Z.T. & Birrell, C.L. (2011a) Population trends of remote invertebrate resources in a marine reserve: trochus and holothurians at Ashmore Reef. *Pacific Conservation Biology,* **17,** 132-140.

Ceccarelli, D.M., Richards, Z.T., Pratchett, M.S. & Cvitanovic, C. (2011b) Rapid increase in coral cover on an isolated coral reef, the Ashmore Reef National Nature Reserve, north-western Australia. *Marine and Freshwater Research,* **62,** 1214-1220.

Chambers, L.E., Devney, C.A., Congdon, B.C., Dunlop, N., Woehler, E.J. & Dann, P. (2011) Observed and predicted effects of climate on Australian seabirds *Emu-Austral Ornithology,* **111,** 235-251.

Chapman, J., Stewart, R.B. & Yarnell, R.A. (1973) Archaeological evidence for Pre-Columbian introduction of *Portulaca oleracea* and *Mollugo verticillata* into Eastern North America. *Economic Botany,* **28,** 411-412.

Charpy-Roubaud, C. & Sournia, A. (1990) The comparative estimation of phytoplanktonic, microphytobenthic and microphytobenthic primary production in the oceans. *Marine Microbial Food Webs,* **4,** 31-57.

Chen, S.H. & Wu, M.J. (2007) A taxonomical study of the genus *Boerhavia* (Nyctaginaceae) in Taiwan. *Taiwania,* **52,** 332-342.

Christianen, M.J.A., van Belzen, J., Herman, P.M.J., van Katwijk, M.M., Lamers, L.P.M., van Leent, P.J.M. & Bouma, T.J. (2013) Low-canopy seagrass beds still provide important coastal protection services. *PLoS One,* **8,** e62413.

Clark, P. (2000) Ashmore Reef: archaeological evidence of past visitation. *Bulletin of the Australian Institute for Maritime Archaeology,* **24,** 1-8.

Clarke, R., Carter, M., Swann, G. & Thomson, J. (2011) The status of breeding seabirds and herons at Ashmore Reef, off the Kimberley Coast, Australia. *Journal of the Royal Society of Western Australia,* **94,** 365-376.

Clarke, R.H. (2010) The status of seabirds and shorebirds at Ashmore Reef and Cartier and Browse Islands: Monitoring program for the Montara Well release - pre-impact assessment and first post-impact field survey. *Prepared on behalf of PTTEP Australasia and the Department of the Environment, Water, Heritage and the Arts, Australia (now the Department of Sustainability, Environment, Water, Population and Communities)***,** 95.

Clarke, R.H. & Herrod, A. (2016) The status of seabirds and shorebirds at Ashmore Reef, Cartier Island & Browse Island. Final impact assessment for the Montara Oil Spill. *Prepared on behalf of PTTEP Australasia and the Department of the Environment. Monash University, Melbourne, Australia***,** 97 pp.

Claudel, C., Buerki, S., Chatrou, L.W., Antonelli, A., Alvarez, N. & Hetterscheid, W. (2017) Large-scale phylogenetic analysis of *Amorphophallus* (Araceae) derived from nuclear and plastid sequences reveals new subgeneric delineation. *Botanical Journal of the Linnean Society,* **184,** 32-45.

Cogger, H.G. (2000) *Reptiles & amphibians of Australia. 6th edition*. Reed New Holland; Sydney, Auckland.

Cogger, H.G., Sadlier, R. & Cameron, E.E. (1983) *The terrestrial reptiles of Australia's island territories*. Australian National Parks and Wildlife Service.

Colwell, R.K. & Coddington, J.A. (1994) Estimating terrestrial biodiversity through extrapolation. *Philosopical Transactions of the royal Society London Biological Sciences,* **345,** 101-118.

Commonwealth of Australia (2002) Ashmore Reef National Nature Reserve and Cartier Island Marine Reserve (Commonwealth Waters) Management Plans, Environment Australia, Canberra.**,** 82.

Commonwealth of Australia (2019) Draft Wildlife Conservation Plan for Seabirds.

Conand, C. (1996) Asexual reproduction by fission in Holothuria atra: Variability of some parameters in populations from the tropical Indo-Pacific. *Oceanologica Acta,* **19,** 209-216.

Cowie, I. (2004) Plant invasions and the Ashmore Islands – species and risks. *In Russell, B.C., Neil, K. and Hilliard, R. (eds.) Ashmore Reef national nature reserve and Cartier Island Marine Reserve*, pp. 82. Marine and terrestrial introduces species prevention and management strategy, Department of Environment and Heritage, Australia.

Craven, L.A. (2005) Malesian and Australian *Tournefortia* transferred to *Heliotropium* and notes on delimitation of Boraginaceae. *Blumea,* **50,** 375-381.

Curran, J. (2003) Survey of Ashmore Reef, Adele Island and other offshore islands of the Kimberley. Australian Quarantine Inspection Service, North Australia Quarantine Strategy, Broome.

Dame, E.A. & Petren, K. (2006) Behavioural mechanisms of invasion and displacement in Pacific island geckos (Hemidactylus). *Animal Behaviour,* **71,** 1165-1173.

Danin, A., Baker, I. & Baker, H.G. (1979) Cytogeography and taxonomy of the *Portulaca oleracea* L. polyploid complex. *Israel Journal of Botany,* **27,** 177-211.

Darling, E.S., Graham, N.A.J., Januchowski-Hartley, F.A., Nash, K.L., Pratchett, M.S. & Wilson, S.K. (2017) Relationships between structural complexity, coral traits, and reef fish assemblages. *Coral Reefs,* **36,** 561-575.

Daubenmire, R. (1959) Canopy coverage method of vegetation analysis. *Northwest Science,* **33,** 43-64.

Depczynski, M., Fulton, C., Marnane, M. & Bellwood, D. (2007) Life history patterns shape energy allocation among fishes on coral reefs. *Oecologia,* **153,** 111-120.

Dethmers, K., E. M., Broderick, D., Moritz, C., Fitzsimmons, N.N., Limpus, C.J., Lavery, S., Whiting, S., Guinea, M., Prince, R.i.T. & Kennett, R. (2006) The genetic structure of Australasian green turtles (*Chelonia mydas*): exploring the geographical scale of genetic exchange. *Molecular ecology,* **15,** 3931-3946.

DeVantier, L., De'ath, A., Done, T. & Turak, E. (1998) Ecological assessment of a complex natural system: a case study from the Great Barrier Reef. *Ecological Applications* **8,** 480-496.

DEWHA (2009) Ashmore Reef National Reserve and Cartier Island Marine Reserve: Information for Visitors. Department of the Environment, Water, Heritage and the Arts.

Diane, N., Hilger, H.H., Förther, H., Weigend, M. & Luebert, F. (2016) Heliotropiaceae. *Flowering Plants. Eudicots: Aquifoliales, Boraginales, Bruniales, Dipsacales, Escalloniales, Garryales, Paracryphiales, Solanales (except Convolvulaceae), Icacinaceae, Metteniusaceae, Vahliaceae* (eds J.W. Kadereit & V. Bittrich), pp. 203-211. Springer International Publishing, Cham.

Director of National Parks (2018) North-west Marine Parks Network Management Plan 2018. Director of National Parks, Canberra.

Done, T., Gilmour, J. & Fisher, R. (2015) Distance decay among coral assemblages during a cycle of disturbance and recovery. *Coral Reefs,* **34,** 727-738.

Doropoulos, C., Ward, S., Roff, G., González-Rivero, M. & Mumby, P.J. (2015) Linking demographic processes of juvenile corals to benthic recovery trajectories in two common reef habitats. *PLoS One,* **10,** e0128535.

Drumm, D.J. & Loneragan, N.R. (2005) Reproductive biology of Holothuria leucospilota in the Cook Islands and the implications of traditional fishing of gonads on the population. *New Zealand Journal of Marine and Freshwater Research,* **39,** 141-156.

Drushka, K., Sprintall, J., Gille, S.T. & Pranowo, W.S. (2008) Observations of the 2004 and 2006 Indian Ocean tsunamis from a pressure gauge array in Indonesia. *Journal of Geophysical Research: Oceans,* **113,** C07038.

Du Puy, D.J. & Telford, I.R.H. (1993) NYCTAGINACEAE. *Flora of Australia,* **50,** 99-107.

Duarte, C.M. & Chiscano, C.L. (1999) Seagrass biomass and production: A reassessment. *Aquatic Botany,* **65,** 159-174.

Dwyer, D. (2000) Disaster and Recovery: A Rotenese Discovery Tale for Ashmore Reef (*Pulau Pasir*), North Australia. *Bulletin of the Australian Institute for Maritime Archaeology,* **24,** 115 -118.

Edgar, G., Ceccarelli, D., Stuart-Smith, R. & Cooper, A. (2017) Reef Life Survey Assessment of Coral Reef Biodiversity in the North-West Commonwealth Marine Reserves Network. Reef Life Survey Foundation Incorporated. 60.

Edgar, G. & Stuart-Smith, R. (2018) Reef Life Survey (RLS): Global reef fish dataset. Institute for Marine and Antarctic Studies (IMAS). https://reeflifesurvey.imas.utas.edu.au/portal.

Elfes, C.T., Livingstone, S.R., Lane, A., Lukoschek, V., Sanders, K.L., Courtney, A.J., Gatus, J.L., Guinea, M., Lobo, A.S. & Milton, D. (2013) Fascinating and forgotten: the conservation status of the world's sea snakes. *Herpetological Conservation and Biology,* **8,** 37-52.

Ellis, J.C. (2005) Marine birds on land: a review of plant biomass, species richness, and community composition in seabird colonies. *Plant Ecology,* **181,** 227-241.

Elzinga, C.L., Salzer, D.W. & Willoughby, J.W. (1998) *Measuring & monitoring plant populations*. U.S. Dept. of the Interior, Bureau of Land Management.

English, S., Wilkinson, C. & Baker, V. (1997) *Survey Manual for Tropical Marine Resources* (2nd Edition) edn. Australian Institute of Marine Science, Townsville.

Erftemeijer, P.L.A., Djunarlin & Moka, W. (1993b) Stomach content analysis of a dugong (*Dugong dugon*) from South Sulawesi, Indonesia. *Marine and Freshwater Research,* **44,** 229-233.

Erftemeijer, P.L.A. & Herman, P.M.J. (1994) Seasonal changes in environmental variables, biomass, production and nutrient contents in two contrasting tropical intertidal seagrass beds in South Sulawesi, Indonesia. *Oecologia,* **99,** 45-59.

Eriksson, H. & Byrne, M. (2015) The sea cucumber fishery in Australia's Great Barrier Reef Marine Park follows global patterns of serial exploitation. *Fish and Fisheries,* **16,** 329-341.

Farrell, H.L. & Gornish, E.S. (2019) *Pennisetum ciliare:* a review of treatment efficacy, competitive traits, and restoration opportunities. *Invasive Plant Science and Management,* **12,** 203-213.

Feagin, R.A., Figlus, J., Zinnert, J.C., Sigren, J., Martínez, M.L., Silva, R., Smith, W.K., Cox, D., Young, D.R. & Carter, G. (2015) Going with the flow or against the grain? The promise of vegetation for protecting beaches, dunes, and barrier islands from erosion. *Frontiers in Ecology and the Environment,* **13,** 203-210.

Field, I.C., Meekan, M.G., Buckworth, R.C. & Bradshaw, C.J.A. (2009) Protein mining the world's oceans: Australasia as an example of illegal expansion-and-displacement fishing. *Fish and Fisheries,* **10,** 323-328.

Finley, A.O., Banerjee, S. & Gelfand, A.E. (2015) spBayes for large univariate and multivariate point-referenced spatio-temporal data models. *Journal of Statistical Software,* **63**.

Fonseca, M.S. & Cahalan, J.A. (1992) A preliminary evaluation of wave attenuation by four species of seagrass. *Estuarine, Coastal and Shelf Science,* **35,** 565-576.

Fonseca, M.S., Fisher, J.S., Zieman, J.C. & Thayer, G.W. (1982) Influence of the seagrass, *Zostera marina* L., on current flow. *Estuarine, Coastal and Shelf Science,* **15,** 351-364.

Foster, S.D., Hosack, G.R., Lawrence, E., Przeslawski, R., Hedge, P., Caley, M.J., Barrett, N.S., Williams, A., Li, J., Lynch, T., Dambacher, J.M., Sweatman, H.P.A. & Hayes, K.R. (2017) Spatially balanced designs that incorporate legacy sites. *Methods in Ecology and Evolution,* **8,** 1433-1442.

Frankenberg, E. & Werner, Y.L. (1981) Adaptability of the daily activity pattern to changes in longitude, in a colonizing lizard, *Hemidactylus frenatus*. *Journal of Herpetology,* **15,** 373-376.

Friedman, K., Eriksson, H., Tardy, E. & Pakoa, K. (2011) Management of sea cucumber stocks: Patterns of vulnerability and recovery of sea cucumber stocks impacted by fishing. *Fish and Fisheries,* **12,** 75-93.

Frisch, A., Ireland, M., Rizzari, J., Lönnstedt, O., Magnenat, K., Mirbach, C. & Hobbs, J.-P. (2016) Reassessing the trophic role of reef sharks as apex predators on coral reefs. *Coral Reefs,* **35,** 459-472.

Fulton, C.J. & Bellwood, D.R. (2005) Wave-induced water motion and the functional implications for coral reef fish assemblages. *Limnology and Oceanography,* **50,** 255-264.

Gill, F. & Donsker, D. (2019) IOC World Bird List (v9.1) https://www.worldbirdnames.org. (Date accessed August 2019).

Gillham, M.E. (1960) Destruction of indigenous heath vegetaion in Victorian sea-bird colonies. *Australian Journal of Botany,* **8,** 277-317.

Gilmour, J., Speed, C.W. & Babcock, R. (2016) Coral reproduction in Western Australia. *PeerJ,* **4,** e2010.

Gilmour, J.P., Cook, K.L., Ryan, N.M., Puotinen, M.L., Green, R.H., Shedrawi, G., Hobbs, J.-P.A., Thomson, D.P., Babcock, R.C., Buckee, J., Foster, T., Richards, Z.T., Wilson, S.K., Barnes, P.B., Coutts, T.B., Radford, B.T., Piggott, C.H., Depczynski, M., Evans, S.N., Schoepf, V., Evans, R.D., Halford, A.R., Nutt, C.D., Bancroft, K.P., Heyward, A.J. & Oades, D. (2019) The state of Western Australia's coral reefs. *Coral Reefs,* **38,** 651-667.

Gilmour, J.P., Smith, L.D., Heyward, A.J., Baird, A.H. & Pratchett, M.S. (2013) Recovery of an isolated coral reef system following severe disturbance *Science,* **340,** 69-71.

Glenn, K.C. (2004) Water properties of Ashmore Reef, North-west shelf, Australia. *In: Records of the Museums and Art Galleries of the Northern Territory, Supplement,* **1,** 9-12.

Goetghebeur, P. (1998) Cyperaceae. *The families and genera of vascular plants. Volume IV - Flowering Plants - Monocotyledons: Alismatanae and Commelinanae (except Gramineae)* (ed. K. Kubitzki), pp. 141-190. Springer, Berlin, Heidelberg.

Goreau, T., McClanahan, T., Hayes, R. & Strong, A. (2000) Conservation of Coral Reefs after the 1998 Global Bleaching Event. *Conservation Biology,* **14,** 5-15.

Gorske, S.F., Rhodes, A.M. & Hopen, H.J. (1979) A numerical taxonomic study of *Portulaca oleracea*. *Weed Science,* **286,** 96-102.

Gouezo, M., Golbuu, Y., Fabricius, K., Olsudong, D., Mereb, G., Nestor, V., Wolanski, E., Harrison, P. & Doropoulos, C. (2019) Drivers of recovery and reassembly of coral reef communities. *Proceedings of the Royal Society B: Biological Sciences,* **286,** 20182908.

Graham, N., Spalding, M. & Sheppard, C. (2010) Reef shark declines in remote atolls highlight the need for multi-faceted conservation action. *Aquatic Conservation: Marine and Freshwater Ecosystems,* **20,** 543-548.

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

Graham, N.A.J., Wilson, S.K., Jennings, S., Polunin, N.V.C., Bijoux, J.P. & Robinson, J. (2006) Dynamic fragility of oceanic coral reef ecosystems. *Proceedings of the National Academy of Sciences of the United States of America,* **103,** 8425-8429.

Green, A. & Bellwood, D. (2009) Monitoring functional groups of herbivorous reef fishes as indicators of coral reef resilience – A practical guide for coral reef managers in the Asia Pacific region. IUCN working group on Climate Change and Coral Reefs. . pp. 70. The International Union for the Conservation of Nature and Natural Resources,, Gland, Switzerland.

Greer, K., Harper, S., Zeller, D. & Pauly, D. (2012) Cocos (Keeling) Islands and Christmas Islands: brief history of fishing and coastal catches (1950–2010). *In Harper S, Zylich K, Boonzaier L, Le Manach F, Pauly D & Zeller D (eds.) Fisheries Catch Reconstructions: Islands, Part III*, pp. 1-13. Fisheries Centre Research Reports.

Greer, K., Harper, S., Zeller, D. & Pauly, D. (2014) Evidence for overfishing on pristine coral reefs: reconstructing coastal catches in the Australian Indian Ocean Territories. *Journal of the Indian Ocean Region,* **10,** 67-80.

Griffith, J. (1997) The corals collected during September/ October 1997 at Ashmore Reef, Timor Sea. *Museum Report to Parks Australia*, pp. 57. Parks Australia, Canberra. Unpublished report.

Guinea, M. (2013) Surveys of the Sea Snakes and Sea Turtles on Reefs of the Sahul Shelf. *Monitoring Program for the Montara Well Release Timor Sea*, pp. 91. School of Environment, Faculty of Engineering, Health, Science and the Environment, Charles Darwin University

Guinea, M. & Mason, E.M. (2017) Surveys of sea snakes and marine turtles and assessment of nesting success of marine turtles at Ashmore Reef Commonwealth Marine Reserve (CMR) with comments on Tropical Fire Ants. pp. 91. Charles Darwin University or AusTurtle Inc, Unpublished report prepared for Parks Australia, May 2017.

Guinea, M. & Whiting, S. (2005) Insights into the distribution and abundance of sea snakes at Ashmore Reef. *Beagle: Records of the Museums and Art Galleries of the Northern Territory,* **Supplement 1,** 199-206.

Guinea, M.L. (1993) Reptilia, Aves and Mammalia. Survey of marine biological and heritage resources of Cartier and Hibernia Reefs, Timor Sea. (eds B.C. Russell & J.R. Hanley), pp. 74-83. Darwin, Northern Territory Museum of Arts and Sciences.

Guinea, M.L. (2006) Final report survey 2005: sea snakes of Ashmore reef, Hibernia reef and Cartier Island. Charles Darwin University, Darwin, Australia.

Guinea, M.L. (2007) Sea snakes of Ashmore Reef, Hibernia Reef and Cartier Island with comments on Scott Reef. *DEWHA Final Report Survey 2007*, pp. 1-20.

Guinea, M.L. (2020) Comments on “Evidence for rapid recovery of shark populations within a coral reef marine protected area”. Speed et al. 2018 220: 308–319. . *Biological Conservation,* **243,** 1-2.

Hale, J. & Butcher, R. (2013) Ashmore Reef Commonwealth Marine Reserve Ramsar Site Ecological Character Description. A report to the Department of the Environment, Canberra. 83.

Harborne, A.R., Mumby, P.J., Micheli, F., Perry, C.T., Dahlgren, C.P., Holmes, K.E. & Brumbaugh, D.R. (2006) The functional value of caribbean coral reef, seagrass and mangrove habitats to ecosystem processes. *Advances in Marine Biology,* **50,** 57-189.

Hardesty, B.D., Lawson, T.J., Van Der Velde , T., LANSDELL, M. & WILCOX, C. (2016) Estimating quantities and sources of marine debris at a continental scale. *Frontiers in Ecology and the Environment,* **15,** 18-25.

Harlin, M.M. (1980) Seagrass epiphytes. In: Phillips, R.C. (eds) Handbook of seagrass biology: an ecosystem perspective. pp. 117-151. Garland STPM press, New York.

Hay, A., Boyce, P.C., Hetterscheid, W.L.A., Jacobsen, N., Murata, J. & Bogner, J. (1995) Checklist of the Araceae of Malesia, Australia, and the tropical western Pacific region. *Blumea. Supplement,* **8,** 1-161.

Hay, A. & Wise, R. (1991) The genus *Alocasia* (Araceae) in Australasia. *Blumea: Biodiversity, Evolution and Biogeography of Plants,* **35,** 499-545.

Hayes, K.R., Hosack, G.R., Lawrence, E., Hedge, P., Barrett, N.S., Przeslawski, R., Caley, M.J. & Foster, S.D. (2019) Designing monitoring programs for marine protected areas within an evidence based decision making paradigm. *Frontiers in Marine Science,* **6,** 1-16.

Heck, K.L. (1976) Community structure and the effects of pollution in sea-grass meadows and adjacent habitats. *Marine biology,* **35,** 345-357.

Heithaus, M.R., Frid, A., Vaudo, J.J., Worm, W. & Wirsing, A.J. (2010) Unravelling the ecological importance of elasobranches. In: Carrier, J., Musick, J.A., Heithaus, M.R (eds.), Sharks and Thier Relatives II-Biodiversity,. *Adaptive Physiology, and Conservation*, pp. 608-633. CRC Press, Boco Raton.

Hernawan, U.E., van Dijk, K.J., Kendrick, G.A., Feng, M., Biffin, E., Lavery, P.S. & McMahon, K. (2017) Historical processes and contemporary ocean currents drive genetic structure in the seagrass *Thalassia hemprichii* in the Indo-Australian Archipelago. *Molecular ecology,* **26,** 1008-1021.

Heslinga, G.A. & Hillmann, A. (1981) Hatchery culture of the commercial top snail Trochus niloticus in Palau, Caroline Islands. *Aquaculture,* **22,** 35-43.

Hetterscheid, W.L.A. & Ittenbach, S. (1996) Everything you always wanted to know about *Amorphophallus*, but were afraid to stick your nose into!!!!! . *Aroideana,* **19,** 7-131.

Heupel, M.R., Knip, D.M., Simpfendorfer, C.A. & Dulvy, N.K. (2014) Sizing up the ecological role of sharks as predators. *Marine Ecology Progress Series,* **495,** 291-298.

Heyward, A., Jones, R., Travers, M., Burns, K., Suosaari, G., Colquhoun, J., Case, M., Radford, B., Meekan, M., Markey, K., Schenk, T., O'Leary, R., Brooks, K., Tinkler, P., Cooper, T. & Emslie, M. (2012) Montara: 2011 Shallow Reef Surveys at Ashmore, Cartier and Seringapatam Reefs. pp. 163. Australian Institute of Marine Science, Townsville, Qld.

Heyward, A., Radford, B., Burns, K., Colquhoun, J. & Moore, C. (2010) Montara Surveys: final report on benthic surveys at Ashmore, Cartier and Seringapatam Reefs. *Report prepared by the Australian Institute of Marine Science (AIMS) for PTTEP Australasia (Ashmore Cartier) Pty. Ltd. in accordance with Contract No. 000/2009/10-23***,** 71.

Hixon, M.A. & Beets, J.P. (1993) Predation, prey Refuges, and the structure of coral-reef fish assemblages. *Ecological Monographs,* **63,** 77-101.

Hodgson, J., Baylis, S., Mott, R., Herrod, A. & Clarke, R. (2016) Precision wildlife monitoring using unmanned aerial vehicles. *Scientific reports,* **6,** 22574.

Hodgson, J. & Clarke, R. (2014) A review of the tropical fire ant *Solenopsis geminata* pilot control program at Ashmore Reef Commonwealth Marine Reserve. *A report prepared for the Director of National Parks*. Monash University, School of Biological Sciences, Melbourne.

Hodgson, J.C. & Koh, L.P. (2016) Best practice for minimising unmanned aerial vehicle disturbance to wildlife in biological field research. *Current Biology,* **26,** R404-R405.

Hoey, A. & Bellwood, D. (2009) Limited Functional Redundancy in a High Diversity System: Single Species Dominates Key Ecological Process on Coral Reefs. *Ecosystems,* **12,** 1316-1328.

Hoey, A.S. & Pratchett, M.S. (2017) Review of research and monitoring relevant to natural values in Australia’s Commonwealth Marine Reserves final report. *Produced for the Department of the Environment and Energy*, pp. 174. James Cook University.

Holbrook, S.J., Schmitt, R.J., Messmer, V., Brooks, A.J., Srinivasan, M., Munday, P.L. & Jones, G.P. (2015) Reef fishes in biodiversity hotspots are at greatest risk from loss of coral species. *PLoS One,* **10,** e0124054.

Holldorf, E. (2018) Avifauna ethological response to unmanned aircraft systems. University of San Francisco.

Holm, L.G., Plucknett, D.L., Pancho, J.V. & Herberger, J.P. (1977) The world's worst weeds: distribution and biology. *University Press of Hawaii***,** 609p.

Horner, P. (2005) Survey for terrestrial reptiles of Ashmore Reef National Nature Reserve. *The Beagle, the Museums and Art Galleries of the Northern Territory,* **Supplement 1,** 131-132.

Hosack, G.R. & Lawrence, E. (2013a) "Survey design for holothurians and *Tectus* at Ashmore Reef. Final Report". Prepared for the Australian Government Department of the Environment., pp. 74. CSIRO Wealth from Oceans Flagship, Hobart, Australia.

Hosack, G.R. & Lawrence, E. (2013b) “Survey Design for Holothurians and Tectus at Ashmore Reef. Interim Report.” Prepared for Department of Sustainability, Environment, Water, Population; Communities. CSIRO Wealth from Oceans Flagship, Hobart, Australia.

Hoskin, C.J. (2011) The invasion and potential impact of the asian house gecko (*Hemidactylus frenatus*) in Australia. *Austral Ecology,* **36,** 240-251.

Hovey, R.K., Statton, J., Fraser, M.W., Ruiz-Montoya, L., Zavala-Perez, A., Rees, M., Stoddart, J. & Kendrick, G.A. (2015) Strategy for assessing impacts in ephemeral tropical seagrasses. *Marine Pollution Bulletin,* **101,** 594-599.

Huisman, J.M. (2015) *Algae of Australia: marine benthic algae of north-western Australia. 1. Green and brown algae*. CSIRO Publishing Melbourne, ABRS, Canberra.

Huisman, J.M. (2018) *Algae of Australia. Marine benthic algae of north-western Australia. 2. Red algae*. CSIRO Publishing Melbourne, ABRS, Canberra.

Huisman, J.M. & Sampey, A. (2014) Kimberly marine biota. Historical data: marine plants. *Records of the Western Australian Museum,* **Supplement 84,** 45-67.

Hurford, C. (2017) Improving the accuracy of bird counts using manual and automated counts in imageJ: an open-source image processing program. *In: Díaz-Delgado R., Lucas R., Hurford C. (eds) The roles of remote sensing in nature conservation*, pp. 249-276. Springer, Cham.

Iltis, H.H. (1960) Studies in the Capparidaceae—VII old world Cleomes adventive in the new world. *Brittonia,* **12,** 279-294.

Jones, G.P., McCormick, M.I., Srinivasan, M. & Eagle, J.V. (2004) Coral decline threatens fish biodiversity in marine reserves. *Proceedings of the National Academy of Sciences of the United States of America,* **101,** 8251-8253.

Kemp, W.M. (1999) Chapter 1: Seagrass ecology and management: an introduction. In: Bortone, S.A. (eds) Seagrasses: Monitoring, ecology, physiology and management. *CRC Marine Science Series*, pp. pp. 336. CRC Press, Florida, USA.

Kendrick, G.A., Vanderklift, M., Bearham, D., McLaughlin, J., Greenwood, J., Säwström, C., Laverock, B., Chovrelat, L., Zavala-Perez, A., De Wever, L., Trapon, M., Grol, M., Guilbault, E., Oades, D., McCarthy, P., George, K., Sampi, T., George, D., Sampi, C., Edgar, Z. & Dougal, K. (2017) Benthic primary productivity: production and herbivory of seagrasses, macroalgae and microalgae. . *Report of 2.2.4 prepared for the Kimberley Marine Research Program,* , pp. 61. Western Australian Marine Science Institution, Perth, Western Australia.

Kenneally, K.F. (1993) Ashmore Reef and Cartier Island. *In A.S. George, A.E. Orchard, & H.J. Hewson (Eds.), Flora of Australia Volume 50, Oceanic Islands* pp. 43-47. Australian Government Publishing Service., Canberra, Australia.

Kinch, J., Purcell, S., Uthicke, S. & Friedman, K. (2008) Population status, fisheries and trade of sea cucumbers in the Western Central Pacific. Sea cucumbers. A global review of fisheries and trade. *FAO Fisheries and Aquaculture Technical Paper*, pp. 7-55.

Kirkman, H. & Kirkman, J. (2000) Long-term seagrass meadow monitoring near Perth, Western Australia. *Aquatic Botany,* **67,** 319-332.

Komyakova, V., Jones, G.P. & Munday, P.L. (2018) Strong effects of coral species on the diversity and structure of reef fish communities: a multi-scale analysis. *PLoS One,* **13**.

Kospartov, M., Beger, M., Ceccarelli, D. & Richards, Z. (2006) An assessment of the distribution and abundance of sea cucumbers, trochus, giant clams, coral, fish and invasive species at Ashmore Reef National Nature Reserve and Cartier Island Marine Reserve: 2005. UniQuest Pty Ltd. pp. 1-298. Department of the Environment and Heritage, Canberra, ACT.

Kramer, M., Bellwood, O. & Bellwood, D. (2013) The trophic importance of algal turfs for coral reef fishes: the crustacean link. *Coral Reefs* **32,** 575-583.

Kulbicki, M., Guillemot, N. & Amand, M. (2005) A general approach to length-weight relationships for New Caledonian lagoon fishes. *Cybium: international journal of ichthyology,* **29,** 235-252.

Langdon, R. (1966) Our far flung empire. 100 miles from Kupang. *New Guinea and Australia, the Pacific and South-East Asia*, pp. 54-58.

Lei, J. & Booth, D.T. (2014) Temperature, field activity and post-feeding metabolic response in the Asian house gecko, *Hemidactylus frenatus*. *Journal of thermal biology,* **45,** 175-180.

Lever, C. (2003) *Naturalized reptiles and amphibians of the world*. Oxford University Press on Demand.

Levitan, D.R. (1991) Influence of body size and population density on fertilization success and reproductive output in a free-spawning invertebrate. *Biological Bulletin,* **181,** 261-268.

Liu, X., Xia, J., Wright, G. & Arnold, L. (2014) A state of the art review on High Water Mark (HWM) determination. *Ocean & Coastal Management,* **102,** 178-190.

Loneragan, N.R., Ahmad Adnan, N., Connolly, R.M. & Manson, F.J. (2005) Prawn landings and their relationship with the extent of mangroves and shallow waters in western peninsular Malaysia. *Estuarine, Coastal and Shelf Science,* **63,** 187-200.

Long, B.G., Andrews, G. & Suharsono, Y. (2004) Sampling accuracy of reef resource inventory technique. *Coral Reefs,* **23,** 378-385.

Lowe, S., Browne, M., Boudjelas, S. & De Poorter, M. (2000) 100 of the World's Worst Invasive Alien Species. A selection from the Global Invasive Species Database *IUCN/SSC Invasive Species Specialist Group (ISSG)*, pp. 11. Auckland, New Zealand.

Loya, Y., Sakai, K., Yamazato, K., Nakano, Y., Sambali, H. & van Woesik, R. (2001) Coral bleaching: the winners and the losers. *Ecology letters,* **4,** 122-131.

Lukoschek, V., Beger, M., Ceccarelli, D., Richards, Z. & Pratchett, M. (2013) Enigmatic declines of Australia's sea snakes from a biodiversity hotspot. *Biological Conservation,* **166,** 191-202.

Maddox, D.M. (1976) History of weevils on puncturevine in and near the United States. *Weed Science,* **24,** 414-419.

Marba, N. & Duarte, C. (1995) Coupling of seagrass (*Cymodocea nodosa*) patch dynamics to subaqueous dune migration. *Journal of Ecology,* **83,** 381.

Marcellini, D.L. (1974) Acoustic behavior of the gekkonid lizard, *Hemidactylus frenatus*. *Herpetologica***,** 44-52.

Marsh, H. & Sinclair, D.F. (1989) An experimental evaluation of dugong and sea turtle aerial survey techniques. *Wildlife Research,* **16,** 639-650.

Marsh, L.M., Vail, L.L., A.K., H. & Rowe, F.W.E. (1993) Echinoderms of Ashmore Reef and Cartier Island. *Records of the Western Australian Museum,* **Supplement No.44,** 53-65.

Marshall, V.M., Lewis, M.M. & Ostendorf, B. (2012) Buffel grass (*Cenchrus ciliaris*) as an invader and threat to biodiversity in arid environments: a review. *Journal of Arid Environments,* **78,** 1-12.

Matthews, P.J. (1995) Aroids and the Austronesians. *Tropics,* **4,** 105-126.

McClatchey, W.C. (2012) Wild food plants of remote Oceania. *Acta Societatis Botanicorum Poloniae,* **81,** 371-380.

McDonald, R.J. (2005) Reproductive ecology and re-establishment of *Argusia argentea* on Ashmore Reef. *The Beagle., Records of the Museums and Art Galleries of the Northern Territory* pp. 153-162.

Mckay, J., Griffiths, A. & Crase, B. (2009) Distribution and Habitat Use by '*Hemidactylus frenatus*' Dumeril and Bibron (Gekkonidae) in the Northern Territory. *The Beagle: Records of the Museums and Art Galleries of the Northern Territory,,* **25,** 107-112.

McKenzie, L.J., Campbell, S.J & Roder, C.A. (2003) Seagrass-Watch: manual for mapping & monitoring seagrass resources by community (citizen) volunteers. pp. 100. QFS, NFC, Cairns.

McKenzie, L.J., Campbell, S.J., Vidler, K.E. & Mellors, J.E. (2007) Seagrass-Watch: manual for mapping and monitoring seagrass resources. pp. 114. Seagrass-Watch HQ Cairns.

McKinnon, J.F., Wesley, D., Raupp, J.T. & Moffat, I. (2013) Geophysical investigations at the Anuru Bay trepang site: A new approach to locating Macassan archaeological sites in Northern Australia. . *Journal of the Australasian Institute for Maritime Archaeology,* **37,** 107-113.

McKnight, C.C. (1976) *The voyage to Marege: Macassan trepangers on northern Australia*. Melbourne University Press, Melbourne.

McNaught, I., Thackway, R., Brown, L. & Parsons, M. (2006) A field manual for surveying and mapping nationally significant weeds. *Bureau of Rural Sciences*, pp. 52. Canberra.

Meekan, M.G., Cappo, M. & Speed, C.W. (2020) Response to Comments on “Evidence for rapid recovery of shark populations within a coral reef marine protected area”. Speed et al., 2018 220:308–319. *Biological Conservation,* **244,** 308-319.

Meikle, R.D. & Hewson, H.J. (1984) Nyctaginaceae *Flora of Australia* **4,** 5-18.

Meikle, R.D. & Hewson, H.J. (2020) *Boerhavia*. Australian Biological Resources Study, Department of Agriculture, Water and the Environment, Canberra.

Menge, B.A. (1995) Indirect effects in marine rocky intertidal interaction webs: patterns and importance. *Ecological Monographs,* **65,** 21-74.

Meyer, J.Y. (2000) A preliminary review of the invasive plants in the Pacific Islands SPREP Member Countries. *In Sherley G., ed. Invasive species in the Pacific: a technical review and regional strategy*, pp. 85–114. South Pacific Regional Environmental Programme (SPREP), Apia, Samoa.

Miller, I., Jonker, M. & Coleman, G. (2009) *Crown-of-thorns starfish and coral surveys using the manta tow and SCUBA search techniques,* 3 edn. Australian Institute of Marine Science, Townsville, Australia.

Milton, D. (2005) Birds of Ashmore Reef National Nature Reserve: an assessment of its importance for seabirds and waders. *The Beagle: Records of the Museums and Art Galleries of the Northern Territory***,** 133–141.

Minton, S.A. & Heatwole, H. (1975) Sea snakes from reefs of the Sahul Shelf. *The Biology of Sea Snakes. University of Maryland Press, Baltimore, Maryland***,** 141-144.

Mitich, L.W. (1997) Common purslane (*Portulaca oleracea*). *Weed Technology,* **11,** 394-397.

Morrison, S.M. & Scott, J.K. (1993) Assessment of the origins of *Tribulus terrestris* in Australia. *In Proceedings I of the 10th Australian Weeds Conference and 14th Asian Pacific Weed*

Science Society Conference, pp. 388-391. Queensland Weed Society, Brisbane, Australia, 6-10

Munro, J.L. (1993) *Giant clams. In Nearshore marine resources of the South Pacific*. Forum Fisheries Agency, Honiara, Guadalcanal, Solomon Islands.

Nagelkerken, I. (2009) Evaluation of nursery function of mangroves and seagrass beds for tropical decapods and reef fishes: patterns and underlying mechanisms. In: Nagelkerken I. (eds) Ecological Connectivity among Tropical Coastal Ecosystems. . *Ecological Connectivity among Tropical Coastal Ecosystems*, pp. 357-399. Springer Netherlands.

Nagelkerken, I., Roberts, C.M., van der Velde, G., Dorenbosch, M., van Riel, M.C., Cocheret del Morinière, E. & Nienhuis, P.H. (2002) How important are mangroves and seagrass beds for coral-reef fish? The nursery hypothesis tested on an island scale. *Marine Ecology Progress Series,* **244,** 299-305.

Naylor, L.A., Spencer, T., Lane, S.N., Darby, S.E., Magilligan, F.J., Macklin, M.G. & Möller, I. (2017) Stormy geomorphology: geomorphic contributions in an age of climate extremes. *Earth Surface Processes and Landforms,* **42,** 166-190.

Newbery, B. & Jones, D.N. (2007) Presence of Asian house gecko Hemidactylus frenatus across an urban gradient in Brisbane: influence of habitat and potential for impact on native gecko species. *Pest or Guest: The Zoology of Overabundance,.(Eds D. Lunney, P. Eby, P. Hutchings and S. Burgin.) pp***,** 59-65.

Ocampo, G. & Columbus, J.T. (2012) Molecular phylogenetics, historical biogeography, and chromosome number evolution of Portulaca (Portulacaceae). *Molecular Phylogenetic Evolution,* **63,** 97-112.

Onuf, C.P. (1994) Seagrasses, Dredging and Light in Laguna Madre, Texas, U.S.A. *Estuarine, Coastal and Shelf Science,* **39,** 75-91.

Osborne, K., Dolman, A.M., Burgess, S.C. & Johns, K.A. (2011) Disturbance and the dynamics of coral cover on the Great Barrier Reef (1995–2009). *PLoS One,* **6,** e17516.

Pace, R., Sherley, B. & Elliott, K.H. (2017) Seabird species vary in behavioural response to drone census. *Scientific Reports* **7,** 17884.

Palmer, J. (2009) A conspectus of the genus *Amaranthus* (Amaranthaceae) in Australia. *Nuytsia,* **19,** 107-128.

Parker, C.M. & Biggs, L.J. (2014) Updates to WA's vascular plant census for 2013. *Nuytsia* **24** 45-63.

Pedersen, O., Colmer, T.D., Borum, J., Zavala-Perez, A. & Kendrick, G.A. (2016) Heat stress of two tropical seagrass species during low tides - impact on underwater net photosynthesis, dark respiration and diel in situ internal aeration. *New Phytologist,* **210,** 1207-1218.

Pennington, J.T. (1985) The ecology of fertilization of echinoid eggs: the consequences of sperm dilution, adult aggregation, and synchronous spawning. *Biological Bulletin,* **169,** 417-430.

Pereira, P., Moraes, R., Santos, M., Lippi, D., Feitosa, J.L. & Pedrosa, M. (2014) The influence of multiple factors upon reef fish abundance and species richness in a tropical coral complex. *Ichthyological Research,* **21,** 375-384.

Petren, K. & Case, T.J. (1998) Habitat structure determines competition intensity and invasion success in gecko lizards. *Proceedings of the National Academy of Sciences,* **95,** 11739-11744.

Pike, D. (1992) Collecting insects in the Ashmore Reef Nature Reserve., pp. 46-55. ANPWS, Ashmore Patrol Report No.1.

Pike, G. & Leach, G. (1997) *Handbook of the vascular plants of Ashmore and Cartier Islands*. Parks & Wildlife Commission of the NT, Australia.

Porter, D.M. (1971) Notes on the floral glands in *Tribulus* (Zygophyllaceae). *Annals of the Missouri Botanical Garden,* **58,** 1-5.

Porter, D.M. (1972) The genera of Zygophyllacea in the southeastern United States *Journal of the Arnold Arboretum,* **53,** 531-552.

POWO (2019) Plants of the World Online. *Facilitated by the Royal Botanic Gardens, Kew*. Published on the Internet. Available: http://www.plantsoftheworldonline.org/ [Accessed Nov 2020].

Pratchett, M., Bay, L., Gehrke, P., Koehn, J., Osborne, K., Pressey, R., Sweatman, H. & Wachenfeld, D. (2011) Contribution of climate change to habitat degradation and loss in Australia's aquatic ecosystems. *Marine and Freshwater Research,* **62,** 1062-1081.

Purcell, S.W., Mercier, A., Conand, C., Hamel, J.F., Toral-Granda, M.V., Lovatelli, A. & Uthicke, S. (2013) Sea cucumber fisheries: global analysis of stocks, management measures and drivers of overfishing. *Fish and Fisheries,* **14,** 34-59.

Rasmussen, A.R., Murphy, J.C., Ompi, M., Gibbons, J.W. & Uetz, P. (2011) Marine Reptiles. *PLoS One,* **6,** e27373.

Ratcliffe, N., Guihen, D., Robst, J., Crofts, S., Stanworth, A. & Enderlein, P. (2015) A protocol for the aerial survey of penguin colonies using AUVs. *Journal of Unmanned Vehicle Systems,* **3,** 95-101.

Rees, M., Colquhoun, J., Smith, L. & Heyward, A. (2003) Surveys Of Trochus, Holothuria, Giant Clams and the coral communities at Ashmore Reef, Cartier Reef and Mermaid Reef, Northwestern Australia. AIMS Report for the Department of the Environment and Heritage. 64.

Rehm, E., Fricke, E., Bender, J., Savidge, J. & Rogers, H. (2019) Animal movement drives variation in seed dispersal distance in a plant-animal network. *Proceedings of the Royal Society B: Biological Sciences,* **286**.

Richards, Z., Beger, M., Hobbs, J.-P., Bowling, T., Chong-Seng, K. & Pratchett, M. (2009) Ashmore Reef National Nature Reserve and Cartier Island Marine Reserve, final report to Department of the Environment, Water Heritage & the Arts. pp. 84.

Richards, Z.T., Bryce, M. & Bryce, C. (2013) New records of atypical coral reef habitat in the Kimberley, Australia. *J Mar Biol.***,** 8.

Richardson, D.M., Pyšek, P. & Carlton, J.T. (2011) *A compendium of essential concepts and terminology in invasion ecology. Fifty years of invasion ecology: the legacy of Charles Elton (ed. D.M. Richardson),*. Blackwell Publishing Ltd, Chichester.

Richardson, D.M., Pyšek, P., Rejmánek, M., Barbour, M.G., Panetta, F.D. & West, C.J. (2000) Naturalization and invasion of alien plants: concepts and definitions. *Diversity and Distributions,* **6,** 93-107.

Rideout, R.S. (1978) Asexual reproduction as a means of population maintenance in the coral reef asteroid *Linckia multifora* on Guam. *Marine biology,* **47,** 287-295.

Ridley, H.N. (1930) *The dispersal of plants throughout the world*. L. Reeve & Company Ltd, Kent, UK.

Robbins, W. (2006) Abundance, demography and population structure of the grey reef shark (*Carcharhinus amblyrynchos*) and the whitetip reef shark (*Triaenodon obesus*) (Fam. Carcharhinidae). PhD thesis, James Cook University.

Roberts, G.O. & Rosenthal, J.S. (2009) Examples of adaptive MCMC. *Journal of Computational and Graphical Statistics,* **18,** 349-367.

Roche, S.E., Costard, S., Meers, J., Field, H.E. & Breed, A.C. (2015) Assessing the risk of Nipah virus establishment in Australian flying-foxes. *Epidemiology Infection,* **143,** 2213-2226.

Rödder, D., Solé, M. & Böhme, W. (2008) Predicting the potential distributions of two alien invasive Housegeckos (Gekkonidae: *Hemidactylus frenatus, Hemidactylus mabouia*). *North-Western Journal of Zoology,* **4,** 236-246.

Roff, G., Bejarano, S., Bozec, Y.-M., Nugues, M., Steneck, R. & Mumby, P. (2014) Porites and the Phoenix effect: Unprecedented recovery after a mass coral bleaching event at Rangiroa Atoll, French Polynesia. *Marine biology,* **161,** 1385-1393.

Roff, G., Doropoulos, C., Rogers, A., Bozec, Y.-M., Krueck, N.C., Aurellado, E., Priest, M., Birrell, C. & Mumby, P.J. (2016) The ecological role of sharks on coral reefs. *Trends in ecology & evolution,* **31,** 395-407.

Roman, L., Schuyler, Q.A., Hardesty, B.D. & Townsend, K.A. (2016) Anthropogenic debris ingestion by avifauna in eastern Australia. *PLoS One,* **11,** e0158343.

Ruppert, J.L.W., Travers, M.J., Smith, L.L., Fortin, M.-J. & Meekan, M.G. (2013) Caught in the middle: combined impacts of shark removal and coral loss on the fish communities of coral reefs. *PLoS One,* **8,** e74648-e74648.

Russell, B., Larson, H., Hutchins, J. & Allen, G.R. (2005) Reef Fishes of the Sahul Shelf. *In* Understanding the Cultural and Natural Heritage Values and Management Challenges of the Ashmore Region. *The Beagle: Records of the Museums and Art Galleries of the Northern Territory***,** 248.

Russell, B. & Vail, L.L. (1988) Report on traditional Indonesian fishing activities at Ashmore Reef nature reserve. *The Beagle, Records of the Museums and Art Galleries of the Northern Territory,* **Supplement 1,** 139-142.

Russell, B.C., Neil, K. & Hilliard, R. (2004) Ashmore Reef National Nature Reserve and Cartier Island Marine Reserve: Marine and Terrestrial Introduced Species Prevention and Management Strategy. *Report for Department of Environment and Heritage***,** 136 pp.

Santosa, E., Lian, C.L., Sugiyama, N., Misra, R.S., Boonkorkaew, P. & Thanomchit, K. (2017) Population structure of elephant foot yams (*Amorphophallus paeoniifolius* (Dennst.) Nicolson) in Asia. *PLoS One,* **12,** e0180000.

Schwerdtner-Máñez, K. & Ferse, S.C.A. (2010) The history of Makassan trepang fishing and trade. *PLoS One,* **5,** e11346.

Scott, J.K., Webber, B.L., Murphy, H.T., Ota, N., Kriticos, D.J. & Loechel, B. (2014) Weeds and climate change: supporting weed management adaptation. *AdaptNRM, www.AdaptNRM.org.*

Serventy, D. (1952a) The bird islands of the Sahul Shelf: with remarks on the nesting seasons of Western Australian sea-birds. *Emu-Austral Ornithology,* **52,** 33-59.

Serventy, D. (1952b) Indonesian fishing activity in Australian seas. *The Australian Geographer***,** 13-16.

Sheahan, M.C. (2007) Zygophyllaceae. *Flowering Plants · Eudicots: Berberidopsidales, Buxales, Crossosomatales, Fabales p.p., Geraniales, Gunnerales, Myrtales p.p., Proteales, Saxifragales, Vitales, Zygophyllales, Clusiaceae Alliance, Passifloraceae Alliance, Dilleniaceae, Huaceae, Picramniaceae, Sabiaceae* (ed. K. Kubitzki), pp. 488-500. Springer Berlin Heidelberg, Berlin, Heidelberg.

Sheppard, C.R.C., Spalding, M., Bradshaw, C. & Wilson, S. (2002) Erosion vs recovery of Coral Reefs after 1998 El Niño: Chagos Reefs, Indian Ocean. *AMBIO: A Journal of the Human Environment,* **31,** 40-48, 49.

Shiell, G. (2005) Density of *Holothuria nobilis* and distribution patterns of common holothurians on coral reefs of northwestern Australia. pp. 231-238. FAO Fisheries and Aquaculture Technical Paper

Short, F. & Duarte, C.M. (2001) Methods for the measurement of seagrass growth and production. *Global seagrass research methods* (eds F.T. Short & R.G. Coles), pp. 155-182. Elsevier Science, Amsterdam, The Netherlands.

Simões, A.R., Silva, H. & Silveira, P. (2011) The Convolvulaceae of Timor with special reference to East Timor. *Blumea,* **56,** 49-72.

Simões, A.R. & Staples, G. (2017) Dissolution of Convolvulaceae tribe Merremieae and a new classification of the constituent genera. *Botanical Journal of the Linnean Society,* **183,** 561-586.

Skewes, T., Dennis, D., Jacobs, D., Gordon, S., Taranto, T., Haywood, M., Pitcher, C., Smith, G., Milton, D. & Poiner, I. (1999a) Survey and stock size estimates of the shallow reef (0-15 m deep) and shoal area (15-50 m deep) marine resources and habitat mapping within the Timor Sea. *Stock Estimates and Stock Status*. CSIRO Division of Marine Research, Canberra:.

Skewes, T., Dennis, D., Jacobs, D.R., Taranto, T.J., Gordon, S.R., Haywood, M., Milton, D., Smith, G.P. & Pitcher, C. (2005) Abundance, stock size and fishery status of the sedentary benthic resources of the reefs and shoals of the MOU74 Box. *The Beagle: Records of the Museums and Art Galleries of the Northern Territory***,** 163-175.

Skewes, T., Smith, L., Dennis, D., Rawlinson, N., Donovan, A. & Ellis, N. (2004) Conversion Ratios for Commercial beche-de-mer Species in Torres Strait. Torres Strait Research Program, Final Report., Brisbane, Australia.

Skewes, T.D., Gordon, S.R., McLeod, I.R., Taranto, T.J., Dennis, D.M., Jacobs, D.R., Pitcher, C.R., Haywood, M., Smith, G.P., Poiner, I.R., Milton, D., Griffin, D. & Hunter, C. (1999b) Survey and Stock Size Estimates of the Shallow Reef (0-15 M Deep) and Shoal Area (15-50 M Deep) Sedentary Marine Resources and Habitat Mapping Within the Timor Sea. MOU74 Box. *Habitat Mapping and Coral Dieback*. CSIRO, Cleveland.

Smith, L., Rees, M., Heyward, A. & Colquhoun, J. (2001) *Survey 2000*. Beche-de-mer and Trochus populations at Ashmore Reef. *Report to Environment Australia*. Australian Institute of Marine Science, Townsville.

Smith, L.D., Gilmour, J. & Heyward, A.J. (2008) Resilience of coral communities on an isolated system of reefs following catastrophic mass-bleaching. *Coral Reefs,* **27,** 197-205.

Smith, M. (1926) *Monograph of the Sea-snakes (Hydrophiidae)*. Wheldon & Wesley, London, British Museum, Department of Zoology.

Smith, M.J., Cogger, H., Tiernan, B., Maple, D., Boland, C., Napier, F., Detto, T. & Smith, P. (2012) An oceanic island reptile community under threat: the decline of reptiles on Christmas Island, Indian Ocean. *Herpetological Conservation and Biology,* **7,** 206-218.

Smith, S.V. (1972) Production of calcium carbonate on the mainland shelf of southern California. *Limnology and Oceanography,* **17,** 28-41.

Smith, S.V. (1981) Marine macrophytes as a global carbon sink. *Science,* **211,** 838-840.

Somaweera, R., Udyawer, V., Guinea, M.L., Ceccarelli, D., Clarke, R.H., Glover, M., Hourston, M., Keesing, J., Rasmussen, A.R., Sanders, K., Shine, R. & Webber, B.L. (2021) Pinpointing drivers of extirpation in sea snakes: a synthesis of evidence from Ashmore Reef. . *Proceedings of the Royal Society B: Biological Sciences,* **8,** 1-19.

Sotomayor, D.A. & Lortie, C.J. (2015) Indirect interactions in terrestrial plant communities: emerging patterns and research gaps. *Ecosphere,* **6,** 103.

Speed, C.W., Cappo, M. & Meekan, M.G. (2018) Evidence for rapid recovery of shark populations within a coral reef marine protected area. *Biological Conservation,* **220,** 308-319.

Stacey, N. (2007) *Boats to burn: Bajo fishing activity in the Australian fishing zone* ANU E Press.

Stevens, D.L. & Olsen, A.R. (2004) Spatially Balanced Sampling of Natural Resources. *Journal of the American Statistical Association,* **99,** 262-278.

Storr, G., Smith, L. & Johnstone, R. (1990) Lizards of Western Australia III Geckos and Legless Lizards. *WA Museum, Perth, WA, Australia*.

Sweatman, H., Delean, S. & Syms, C. (2011) Assessing loss of coral cover on Australia’s Great Barrier Reef over two decades, with implications for longer-term trends. *Coral Reefs,* **30,** 521-531.

Tanaka, N., Sasaki, Y., Mowjood, M.I.M., Jinadasa, K.B.S.N. & Homchuen, S. (2007) Coastal vegetation structures and their functions in tsunami protection: experience of the recent Indian Ocean tsunami. *Landscape and Ecological Engineering,* **3,** 33-45.

Tassin, J., Boissenin, M & Barré, N. (2010) Can *Ptilinopus greyii* (Columbidae) disperse seeds in New Caledonia's dry forests? *Pacific Science,* **64,** 527-532.

Tschinkel, W. (2006) *The fire ants*. The Belknap Press of Harvard University Press.

Tylianakis, J.M., Didham, R.K., Bascompte, J. & Wardle, D.A. (2008) Global change and species interactions in terrestrial ecosystems. *Ecology letters,* **11,** 1351-1363.

Udyawer, V. & Heupel, M. (2017) Spatial and temporal patterns in sea snake populations on the North West Shelf. *Report to the National Environmental Science Programme, Marine Biodiversity*

Uetz, P., Freed, P. & Hošek, J. (2019) The Reptile Database, http://www.reptile-database.org.

Unsworth, R.K.F., Collier, C.J., Henderson, G.M. & McKenzie, L.J. (2012) Tropical seagrass meadows modify seawater carbon chemistry: Implications for coral reefs impacted by ocean acidification. *Environmental Research Letters,* **7,** 024026.

Urquhart, N.S. & Kincaid, T.M. (1999) Designs for detecting trends from repeated surveys of ecological resources. *Journal of Agricultural, Biological, and Environmental Statistics*, 404–414.

Uthicke, S. (1997) Seasonality of asexual reproduction in *Holothuria* (Halodeima) *atra, H*. (H.) *edulis* and *Stichopus chloronotus* (Holothuroidea: Aspidochirotida) on the Great Barrier Reef. *Marine biology,* **129,** 435-441.

Uthicke, S., Schaffelke, B. & Byrne, M. (2009) A boom–bust phylum? Ecological and evolutionary consequences of density variations in echinoderms. *Ecological Monographs,* **79,** 3-24.

Uthicke, S., Welch, D. & Benzie, J.A.H. (2004) Slow growth and lack of recovery in overfished holothurians on the Great Barrier Reef: evidence from DNA fingerprints and repeated large-scale surveys. . *Conservation Biology,* **18,** 1395-1404.

Van der Pijl, L. (1982) Principles of dispersal in higher plants. *Springer-Verlag, Berlin*, 171.

Van Oppen, M. & Gates, R. (2006) Conservation genetics and the resilience of reef building corals. *Molecular ecology,* **15,** 3863-3883.

Vanderduys, E. & Kutt, A. (2013) Is the Asian house gecko, *Hemidactylus frenatus*, really a threat to Australia’s biodiversity? *Australian Journal of Zoology,* **60,** 361-367.

Vas, E., Lescroël, A., D., O, Boguszewski, G. & Grémillet, D. (2015) Approaching birds with drones: first experiments and ethical guidelines. *Biology letters,* **11,** 20140754.

Veron, J.E.N. (1986) Reef-building corals. In: Faunal Surveys of the Rowley Shoals, Scott Reef and Seringapatam Reef, North-Western Australia (ed. Berry, P.F.). *Records of the Western Australian Museum,* **25,** 27-35.

Veron, J.E.N. (1993) Hermatypic corals of Ashmore Reef and Cartier Island. . *Records of the Western Australian Museum,* **Supplement 44,** 13–20.

Vonk, J.A., Christianen, M.J.A. & Stapel, J. (2008) Redefining the trophic importance of seagrasses for fauna in tropical Indo-Pacific meadows. *Estuarine, Coastal and Shelf Science,* **79,** 653-660.

Vroom, P.S., Page, K.N., Kenyon, J. & Brainard, R.E. (2006) Algae-dominated reefs. *American Scientist* **94,** 430–437.

Ward, L.G., Michael Kemp, W. & Boynton, W.R. (1984) The influence of waves and seagrass communities on suspended particulates in an estuarine embayment. *Marine Geology,* **59,** 85-103.

Webber, B.L. & Scott, J.K. (2012) Rapid global change: implications for defining natives and aliens. *Global Ecology and Biogeography,* **21,** 305-311.

Wells, F.E. (1993) Molluscs of Ashmore Reef and Cartier Island. *Records of the Western Australian Museum,* **Supplement 24,** 25-45.

Westaway, J. (2015) Ashmore Plant Health Survey. Northern Australia Quarantine Strategy, Commonwealth Department of Agriculture.

Westcott, D.A., Dennis, A.J., Bradford, M.G., Harrington, G.N. & McKeown, A. (2008) Seed dispersal processes in Australia's tropical rainforests. *Living in a Dynamic Tropical Forest Landscape* (eds N.E. Stork & S.M. Turton), pp. 210-223. Blackwell Publishing, Malden, USA.

Wetterer, J. (2011) Worldwide spread of the tropical fire ant, *Solenopsis geminata* (*Hymenoptera: Formicidae*). *Myrmecological News,* **14,** 21-35.

Whiting, S. (2005) Dugongs of Ashmore Reef and the Sahul Banks: a review of current knowledge and a distribution of sightings. *The Beagle: Records of the Museums and Art Galleries of the Northern Territory,* **Supplement 1,** 207-210.

Whiting, S. & Guinea, M. (2005a) The sea turtles of Ashmore Reef: history, current status and future direction research and management. *Biomarine International & Charles Darwin University*.

Whiting, S.D. (1999) Use of the remote Sahul banks by dugongs including breeding females. . *Marine Mammal Science,* **15,** 609-615.

Whiting, S.D. & Guinea, M.L. (2005b) Sea turtles of Sahul Banks -work completed and required. *The Beagle: Records of the Museums and Art Galleries of the Northern Territory,* **Supplement 1,** 248.

Willan, R.C. (2005) The molluscan fauna from the emergent reefs of the northernmost Sahul Shelf, Timor Sea–Ashmore, Cartier and Hibernia Reefs; biodiversity and zoogeography. *The Beagle, Records of the Museums and Art Galleries of the Northern Territory,* **1**.

Wilson, J.R., Dormontt, E.E., Prentis, P.J., Lowe, A.J. & Richardson, D.M. (2009) Something in the way you move: dispersal pathways affect invasion success. *Trends in ecology & evolution,* **24,** 136-144.

Wilson, J.R.U., Caplat, P., Dickie, I.A., Hui, C., Maxwell, B.D., Nuñez, M.A., Pauchard, A., Rejmánek, M., Richardson, D.M., Robertson, M.P., Spear, D., Webber, B.L., van Wilgen, B.W. & Zenni, R.D. (2014) A standardized set of metrics to assess and monitor tree invasions. *Biological Invasions,* **16,** 535-551.

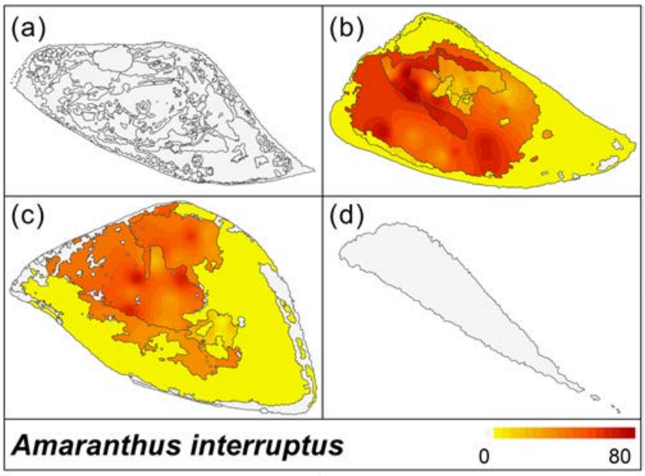
Yokoi, K., Saski, M., Nakagawa, M. & Sato, N.Y. (2011) A nationwide long-term monitoring survey of seagrass and algal beds in the 'Monitoring Sites 1000' Project. *Aquabiology,* **33.,** 283-290.

Young, L. & Schlesinger, C. (2015) Habitat use and behaviour of birds in areas invaded by buffel grass (*Cenchrus ciliaris* L.) and in restored habitat. *Wildlife Research,* **41,** 379-394.

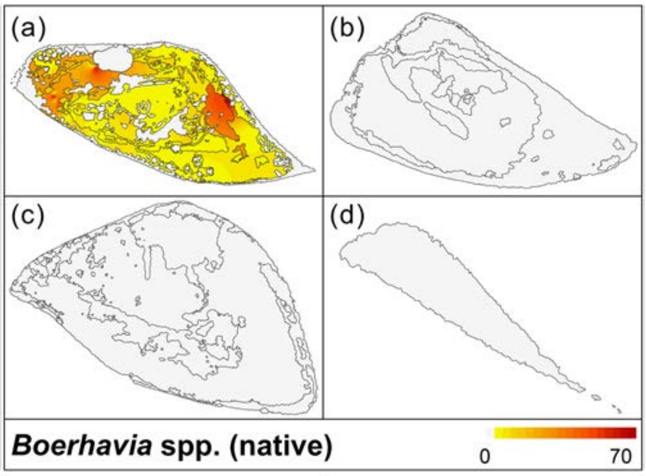
16 Supplementary Information

SI Table 1. Permanent photo point corrections to the location and site description details provided in Clarke (2010) for West Island, Middle Island, East Island and Splittgerber Cay - at Ashmore Reef. If the location marker was found, the condition was noted.

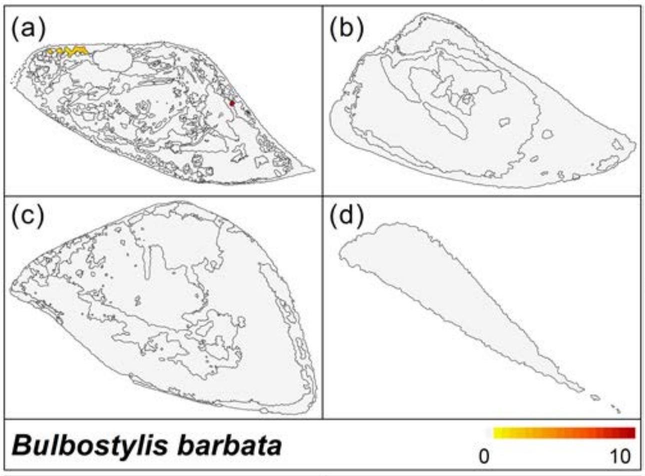
Island – Photo point	Georeference		Bearing		Marker found & condition
	2010	2019	2010	2019	
West Island – P1	51L 496765, 8646464	51L 496769, 8646484	0	317	No | n/a
West Island – P2	51L 496592, 8646715	51L 0496593, 8646718	0	340	No | n/a
West Island – P3	51L 496294, 8646856	51L 0496293, 8646856	270	244	Yes | Good
West Island – P4	51L 495961, 8646796	51L 495960, 8646798	225	180	Yes | Good
West Island – P5	51L 496398, 8646515	51L 496400, 8646516	113	93	Yes | Good
West Island – P6	51L 496802, 8646442	51L 496804, 8646434	90	90	No | n/a
Middle Island – P1	51L 503891, 8643995	51L 503891, 8643996	338	285	Yes | Poor
Middle Island – P2	51L 503476, 8643995	51L 503475, 8643990	45	45	Yes | Poor
Middle Island – P3	51L 503592, 8644247	51L 503592, 8644246	225	215	Yes | Poor
Middle Island – P4	51L 503592, 8644247	51L 503592, 8644246	180	128	Yes | Poor
East Island – P1	51L 510193, 8644641	51L 510193, 8644640	135	103	Yes | V Poor
East Island – P2	51L 510583, 8644722	51L 510582, 8644722	248	243	Yes | Poor
East Island – P3	51L 510676, 8644491	51L 510674, 8644490	315	315	Yes | V Poor
Splittgerber Cay – P1	51L 512037, 8644751	51L 0512042, 8644750	135	135	No | n/a
Splittgerber Cay – P2	51L 512093, 8644704	51L 0512091, 8644704	315	315	No | n/a



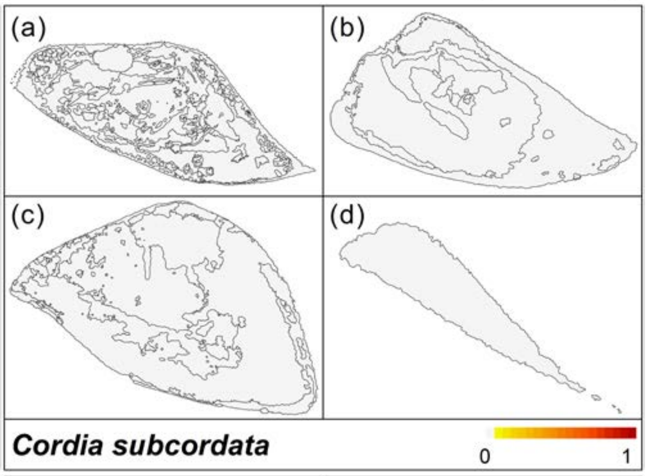
Amaranthus interruptus



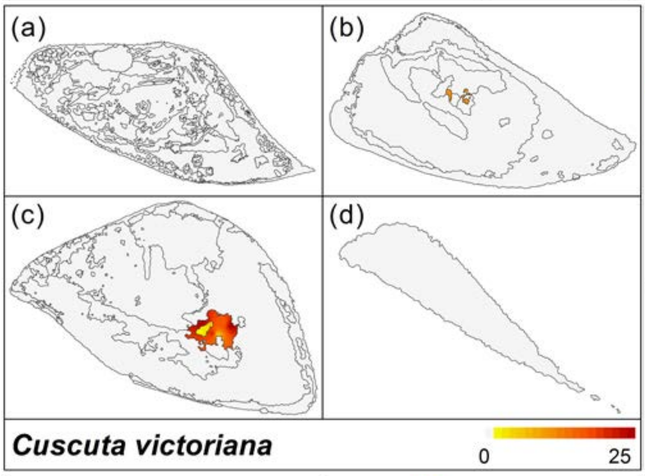
***Boerhavia* spp. (native)**



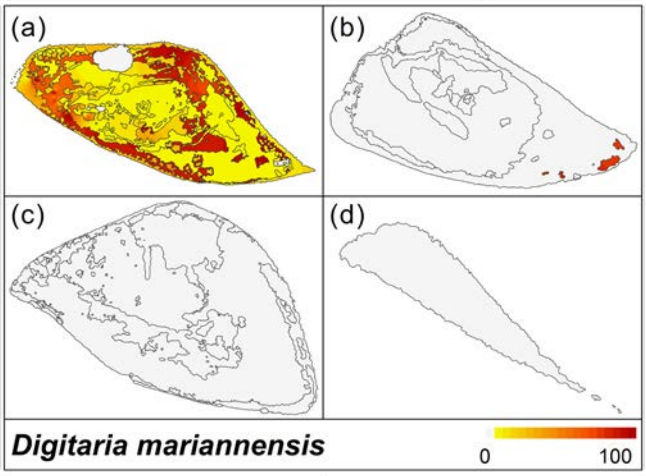
Bulbostylis barbata



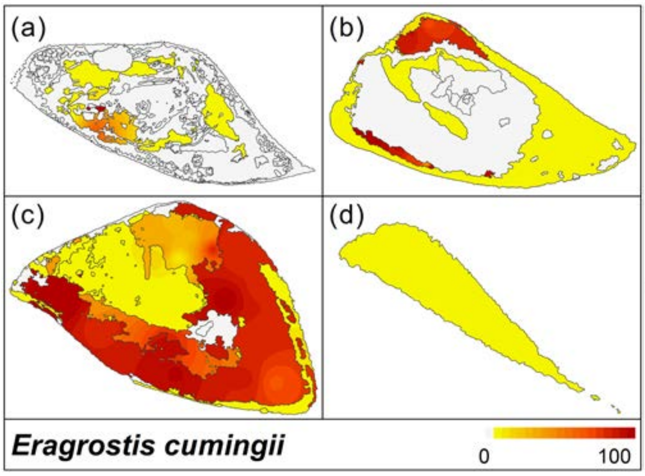
Cordia subcordata



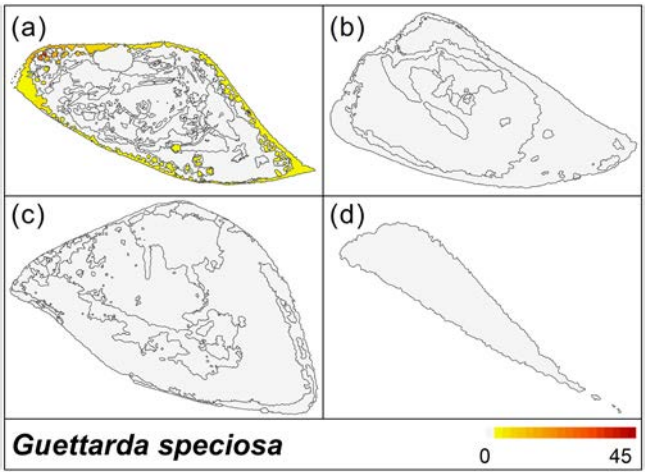
Cuscuta victoriana



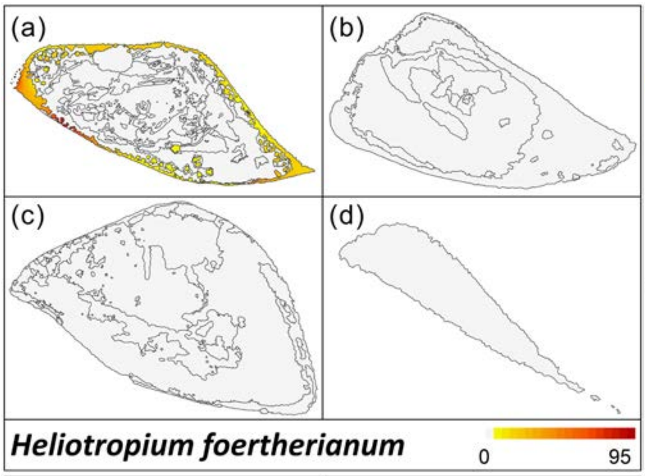
Digitaria mariannensis



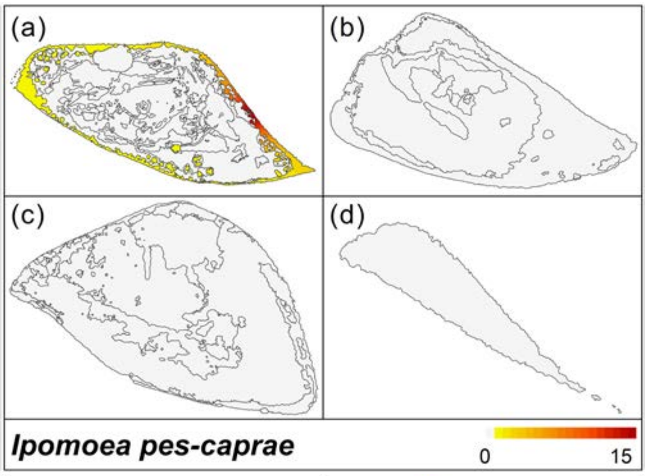
Eragrostis cumingii



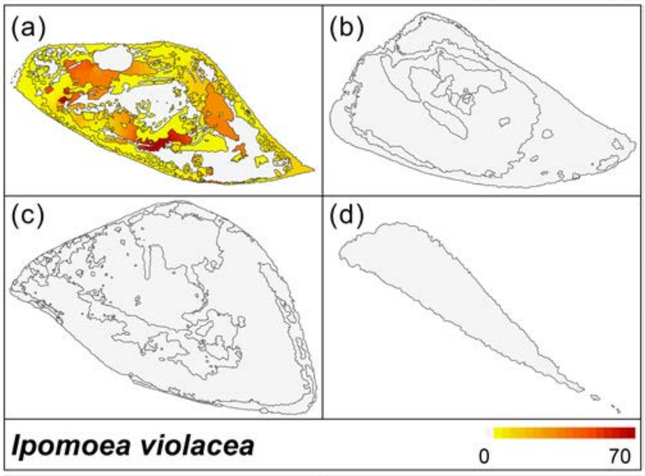
Guettarda speciosa



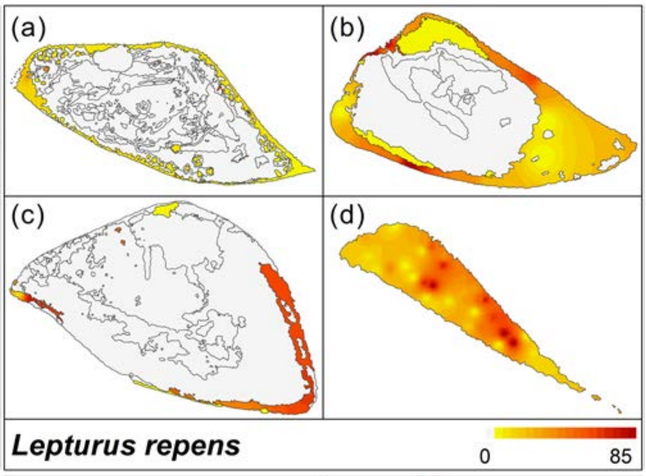
Heliotropium foertherianum



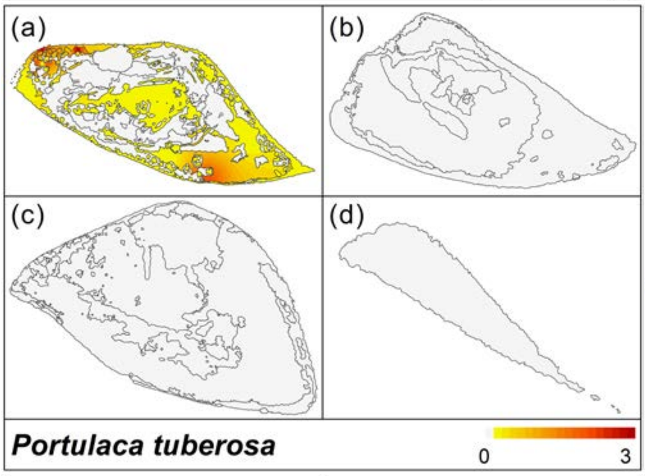
Ipomoea pes-caprae



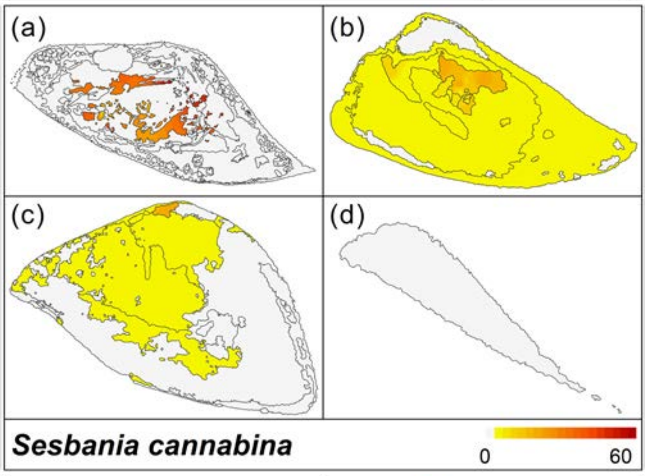
Ipomoea violacea



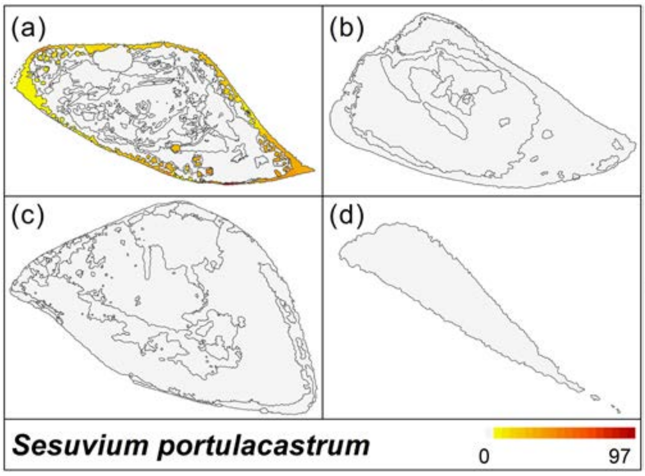
Lepturus repens



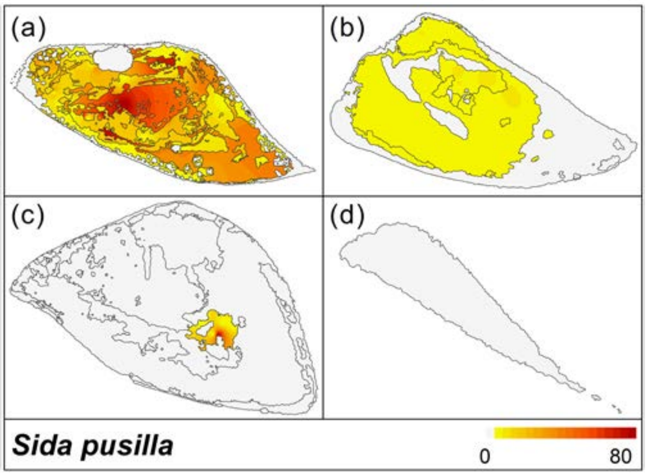
Portulaca tuberosa



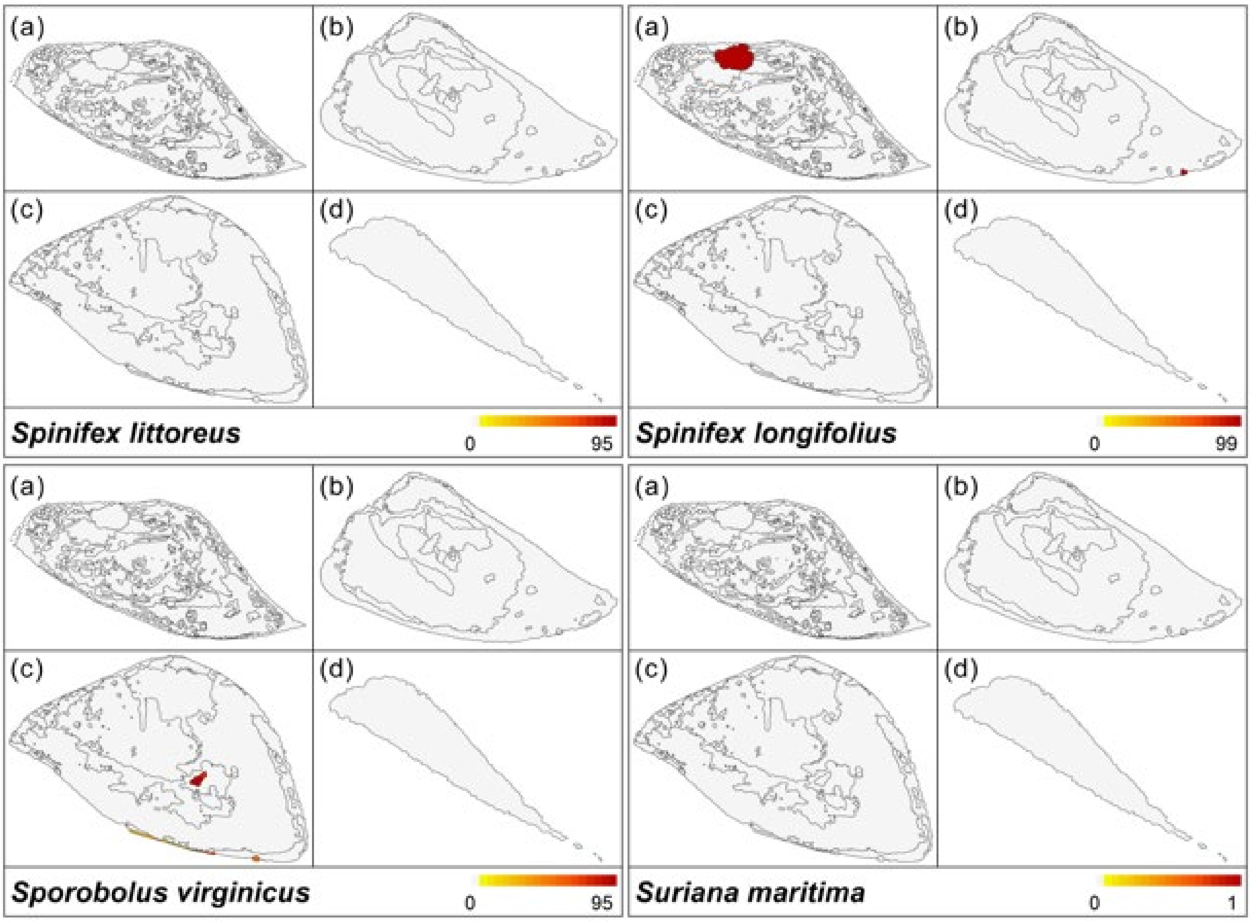
Sesbania cannabina



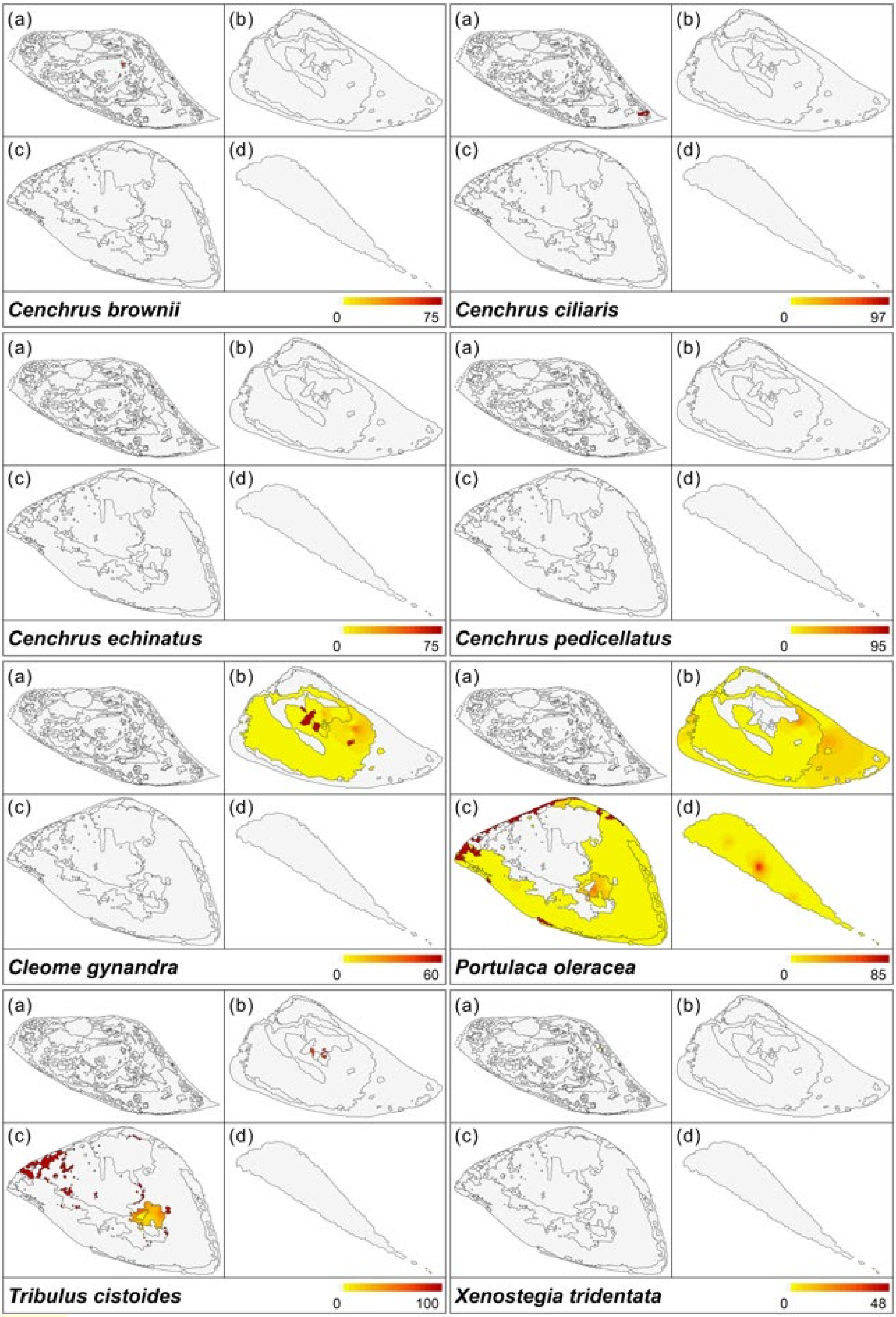
Sesuvium portulacastrum



Sida pusilla



SI Figure 1. Abundance calculations for all native terrestrial plant species stratified by plant community on West Island (a), Middle Island (b), East Island (c) and Splittgerber Cay (d) for each species, respectively. Colour scale represents % cover for each species based on quadrat surveys across communities.



SI Figure 2. Abundance calculations for all non-native terrestrial plant species stratified by plant community on West Island (a), Middle Island (b), East Island (c) and Splittgerber Cay (d) for each species, respectively. Colour scale represents % cover for each species based on quadrat surveys across communities

SI Table 2. Plant species abundance according to vegetation communities on West Island at Ashmore Reef in May 2019. Each header row contains the total area of the community (m^2) and the number of fragments (F) for that community. Within each community, area covered (m^2), average canopy cover (%) and the proportion of quadrats occupied (Q) is shown for each species (cover area estimated from community abundance category if no quadrat data was available). For island-wide totals, A_C: Total area for communities containing the species (m^2); A_I: community area as % of all communities; A_O: total area occupied by the species on that island (m^2); D: average canopy cover (%); Q_P: Quadrats with the species present; Q_T: Total quadrats surveyed in communities with the species; C: all communities with that species. *= Non-native species.

Species	Community coverage and abundance (272 fragments)														Island-wide coverage and abundance		
	Wa: 65635m^2 44 F	**Wb: 50398m^2 14 F**	**Wc: 83269m^2 27 F**	**Wd: 21694m^2 77 F**	**We: 8616m^2 1 F**	**Wf: 37074m^2 60 F**	**Wg: 1985m^2 3 F**	**Wh: 165m^2 3 F**	**Wi :152m^2 2 F**	**Wj: 115m^2 5 F**	**Wk: 891m^2 3 F**	**Wl: 532m^2 2 F**	**Wm: 16m^2 1 F**	**Wn: 529m^2 30 F**	**A_C m^2 A_I%**	**A_O m^2 D%**	**Q_P/Q_T C**
Boerhavia spp.	7982m^2 12.2% 7/13Q	8505m^2 16.9% 4/8Q	5320m^2 6.4% 3/18Q	na	na	na	6m^2 0.3% 0/5Q	1m^2 0.7% 1/3Q	18m^2 11.7% 1/3Q	na	na	na	na	2m^2 0.5% 1/11Q	202133m^2 74.6%	21834m^2 10.8%	17/56Q 7C
Bulbostylis barbata	na	na	na	na	na	na	68m^2 3.4% 5/5Q	na	na	na	na	na	na	na	1985m^2 0.7%	68m^2 3.4%	5/5Q 1C
Cordia subcordata	na	na	na	na	na	37m^2 0.1% 0/17Q	na	na	na	na	na	na	na	na	37074m^2 13.7%	37m^2 0.1%	0/0Q 1C
Digitaria mariannensis	51195m^2 78.0% 13/13Q	5985m^2 11.9% 3/8Q	509m^2 0.6% 3/18Q	3013m^2 13.9% 3/9Q	na	7066m^2 19.1% 10/17Q	111m^2 5.6% 4/5Q	55m^2 33.3% 3/3Q	1m^2 0.3% 1/3Q	na	4m^2 0.5% 0/3Q	na	na	188m^2 35.5% 8/11Q	261791m^2 96.6%	68126m^2 26.0%	48/87Q 10C
Eragrostis cumingii	na	3780m^2 .5% 1/8Q	na	na	na	na	na	na	na	na	na	505m^2 95.0% 1/1Q	na	na	50929m^2 18.8%	4285m^2 8.4%	2/9Q 2C
Guettarda speciosa	na	na	na	na	na	981m^2 2.6% 1/17Q	na	na	na	na	na	na	na	na	37074m^2 13.7%	981m^2 2.6%	1/17Q 1C
Heliotropium foertherianum	na	na	na	na	na	5932m^2 16.0% 5/17Q	na	na	na	na	na	na	na	na	37074m^2 13.7%	5932m^2 16.0%	5/17Q 1C
Ipomoea pes-caprae	na	na	na	na	na	327m^2 0.9% 1/17Q	na	na	na	na	na	na	na	na	37074m^2 13.7%	327m^2 0.9%	1/17Q 1C
Ipomoea violacea	2196m^2 3.3% 3/13Q	15938m^2 31.6% 7/8Q	3997m^2 4.8% 0/18Q	410m^2 1.9% 3/9Q	na	2530m^2 6.8% 4/17Q	na	29m^2 17.4% 0/3Q	10m^2 6.7% 2/3Q	na	na	na	na	1m^2 0.2% 1/11Q	258915m^2 95.5%	25111m^2 9.7%	20/61Q 8C
Lepturus repens	6760m^2 10.3% 0/13Q	na	na	na	na	2510m^2 6.8% 10/17Q	24m^2 1.2% 0/5Q	29m^2 17.4% 0/3Q	85m^2 56.0% 3/3Q	na	na	na	na	na	105011m^2 38.7%	9408m^2 9.0%	13/20Q 5C
Pandanus sp	na	na	na	na	na	37m^2 0.1% 0/17Q	na	na	na	na	na	na	na	na	37074m^2 13.7%	37m^2 0.1%	0/0Q 1C
Portulaca tuberosa	6760m^2 10.3% 0/13Q	na	324m^2 0.4% 4/18Q	na	na	76m^2 0.2% 2/17Q	26m^2 1.3% 4/5Q	na	na	na	na	na	na	na	187963m^2 69.3%	7186m^2 3.8%	10/40Q 4C
Sesbania cannabina	6760m^2 10.3% 0/13Q	2570m^2 5.1% 0/8Q	na	7255m^2 33.4% 9/9Q	na	na	na	na	na	na	na	na	na	na	137727m^2 50.8%	16586m^2 12.0%	9/9Q 3C
Sesuvium portulacastrum	na	na	na	na	na	7044m^2 19.0% 6/17Q	na	na	na	na	na	na	na	na	37074m^2 13.7%	7044m^2 19.0%	6/17Q 1C
Sida pusilla	4297m^2 6.5% 5/13Q	9072m^2 18.0% 7/8Q	31411m^2 37.7% 18/18Q	4941m^2 22.8% 5/9Q	na	na	111m^2 5.6% 4/5Q	16m^2 10.0% 2/3Q	1m^2 0.7% 1/3Q	na	na	43m^2 8.0% 1/1Q	na	55m^2 10.4% 7/11Q	224359m^2 82.8%	49946m^2 22.3%	50/71Q 9C
Spinifex littoreus	na	na	na	na	na	na	na	na	na	106m^2 92.5% 2/2Q	na	na	na	na	115m^2 0.04%	106m^2 92.5%	2/2Q 1C
Spinifex longifolius	na	na	na	na	8444m^2 98.0% 2/2Q	na	na	na	na	na	na	na	na	na	8616m^2 3.2%	8444m^2 98.0%	2/2Q 1C
Suriana maritima	na	na	na	na	na	37m^2 0.1% 0/17Q	na	na	na	na	na	na	na	na	37074m^2 13.7%	37m^2 0.1%	0/0Q 1C
*Cenchrus brownii**	na	na	na	na	na	na	na	na	na	na	na	na	na	255m^2 48.2% 11/11Q	529m^2 0.2%	255m^2 48.2%	11/11Q 1C
*Cenchrus ciliaris**	na	na	na	na	na	na	na	na	na	na	822m^2 92.3% 3/3Q	na	na	na	891m^2 0.3%	822m^2 92.3%	3/3Q 1C
*Cenchrus echinatus**	na	na	na	na	na	na	na	115m^2 70.0% 3/3Q	na	na	na	na	na	na	165m^2 0.1%	115m^2 70.0%	3/3Q 1C
*Cenchrus pedicellatus**	na	na	na	na	na	na	na	na	na	na	na	na	15m^2 95.0% 1/1Q	na	16m^2 0.01%	15m^2 95.0%	1/1Q 1C
*Portulaca oleracea**	6760m^2 10.3% 0/13Q	5191m^2 10.3% 0/8Q	na	na	na	na	na	na	na	na	na	na	na	na	116033m^2 42.8%	11951m^2 10.3%	0/0Q 2C
*Xenostegia tridentata**	na	na	na	na	na	na	na	na	25m^2 16.7% 1/3Q	na	na	na	na	na	152m^2 0.1%	25m^2 16.7%	1/3Q 1C
All species	65670m^2 100% 13/13Q	43279m^2 85.9% 8/8Q	37564m^2 45.1% 18/18Q	15620m^2 72.0% 9/9Q	8444m^2 98.0% 2/2Q	26466m^2 71.4% 17/17Q	316m^2 15.9% 5/5Q	188m^2 114% 3/3Q	140m^2 92.0% 3/3Q	106m^2 92.5% 2/2Q	822m^2 92.3% 3/3Q	548m^2 103% 1/1Q	15m^2 95.0% 1/1Q	501m^2 94.6% 11/11Q	271071m^2 100%	199679m^2 73.7%	96/96Q 14C

SI Table 3. Plant species abundance according to vegetation communities on Middle Island at Ashmore Reef in May 2019. Each header row contains the total area of the community (m^2) and the number of fragments (F) for that community. Within each community, area covered (m^2), average canopy cover (%) and the proportion of quadrats occupied (Q) is shown for each species (cover area estimated from community abundance category if no quadrat data was available). For island-wide totals, A_C: Total area for communities containing the species (m^2); A_I: community area as % of all communities; A_O: total area occupied by the species on that island (m^2); D: average canopy cover (%); Q_P: Quadrats with the species present; Q_T: Total quadrats surveyed in communities with the species; C: all communities with that species. *= Non-native species.

	Community coverage and abundance (36 fragments)										Island-wide coverage and abundance		
Species	**Ma: 40377m^2 7 F**	**Mb: 9285m^2 4 F**	**Mc: 53448m^2 5 F**	**Md: 8431m^2 2 F**	**Me: 6406m^2 2 F**	**Mf: 1514m^2 4 F**	**Mg: 990m^2 4 F**	**Mh: 105m^2 1 F**	**Mi: 11m^2 1 F**	**Mj: 368m^2 6 F**	**A_C m^2 A_I%**	**A_O m^2 D%**	**Q_P/Q_T C**
Amaranthus interruptus	112m^2 0.3% 1/18Q	312m^2 3.4% 5/11Q	27585m^2 51.6% 18/18Q	5059m^2 60.0% 5/5Q	1556m^2 24.3% 7/7Q	151m^2 10.0% 2/4Q	na	na	na	6m^2 1.7% 1/3Q	119828m^2 99.1%	34782m^2 29.0%	39/66Q 7C
Cuscuta victoriana	na	na	na	na	na	na	na	na	na	37m^2 10.0% 3/3Q	368m^2 0.3%	37m^2 10.0%	3/3Q 1C
Digitaria mariannensis	na	na	na	na	na	na	767m^2 77.5% 2/2Q	na	na	na	990m^2 0.8%	767m^2 77.5%	2/2Q 1C
Eragrostis cumingii	449m^2 1.1% 2/18Q	7647m^2 82.4% 11/11Q	na	34m^2 0.4% 1/5Q	na	na	na	na	na	na	58093m^2 48.0%	8129m^2 14.0%	14/34Q 3C
Heliotropium foertherianum	na	na	na	na	na	na	na	na	10m^2 94.5% 1/0Q	na	11m^2 0.01%	10m^2 94.5%	0/0Q 1C
Lepturus repens	13908m^2 34.4% 15/18Q	8m^2 0.1% 1/11Q	na	na	na	na	76m^2 7.7% 0/2Q	11m^2 10.5% 0/2Q	na	na	50756m^2 42.0%	14003m^2 27.6%	16/29Q 4C
Portulaca tuberosa	2988m^2 7.4% 0/18Q	na	na	na	na	na	76m^2 7.7% 0/2Q	11m^2 10.5% 0/2Q	na	na	41472m^2 34.3%	3075m^2 7.4%	0/0Q 3C
Sesbania cannabina	22m^2 0.1% 1/18Q	na	89m^2 0.2% 2/18Q	337m^2 4.0% 2/5Q	897m^2 14.0% 7/7Q	57m^2 3.8% 2/4Q	na	na	na	7m^2 2.0% 2/3Q	110544m^2 91.4%	1410m^2 1.3%	16/55Q 6C
Sida pusilla	na	42m^2 0.5% 1/11Q	386m^2 0.7% 3/18Q	na	183m^2 2.9% 3/7Q	4m^2 0.3% 1/4Q	na	na	na	na	70652m^2 58.4%	615m^2 0.9%	8/40Q 4C
Spinifex longifolius	na	na	na	na	na	na	na	99m^2 94.5% 2/2Q	na	na	105m^2 0.1%	99m^2 94.5%	2/2Q 1C
*Cleome gynandra**	na	na	920m^2 1.7% 2/18Q	na	229m^2 3.6% 2/7Q	776m^2 51.3% 4/4Q	na	na	na	6m^2 1.7% 1/3Q	61735m^2 51.0%	1931m^2 3.1%	9/32Q 4C
*Portulaca oleracea**	1974m^2 4.9% 9/18Q	na	1039m^2 1.9% 1/18Q	na	na	na	153m^2 15.5% 0/2Q	11m^2 10.5% 0/2Q	na	na	94919m^2 78.5%	3178m^2 3.3%	10/36Q 4C
*Tribulus cistoides**	na	na	na	na	na	na	na	na	na	276m^2 75.0% 3/3Q	368m^2 0.3%	276m^2 75.0%	3/3Q 1C
All species	16465m^2 40.8% 18/18Q	8010m^2 86.3% 11/11Q	30020m^2 56.2% 18/18Q	5430m^2 64.4% 5/5Q	2864m^2 44.7% 7/7Q	988m^2 65.3% 4/4Q	767m^2 77.5% 2/2Q	99m^2 94.5% 2/2Q	10m^2 94.5% 1/0Q	332m^2 90.3% 3/3Q	120934m^2 100%	64986m^2 53.7%	70/70Q 10C

SI Table 4. Plant species abundance according to vegetation communities on East Island and Splittgerber Cay at Ashmore Reef in May 2019. Each header row contains the total area of the community (m^2) and the number of fragments (F) for that community. Within each community, area covered (m^2), average canopy cover (%) and the proportion of quadrats occupied (Q) is shown for each species (cover area estimated from community abundance category if no quadrat data was available). For island-wide totals, A_C: Total area for communities containing the species (m^2); A_I: community area as % of all communities; A_O: total area occupied by the species on that island (m^2); D: average canopy cover (%); Q_P: Quadrats with the species present; Q_T: Total quadrats surveyed in communities with the species; C: all communities with that species. *= Non-native species.

	East Island coverage and abundance (114 fragments)													Splittgerber Cay coverage and abundance (1 fragment)			
	Communities										Island-wide			Communities	Island-wide		
Species	Ea 3693m^2 11F	Eb 62130m^2 12F	Ec 4076m^2 63F	Ed 22914m^2 8F	Ee 25783m^2 1F	Ef 8304m^2 14F	Eg 657m^2 1F	Eh 416m^2 1F	Ei 594m^2 2F	Ej 3356m^2 1F	**A_C m^2** **A_I%**	**A_O m^2** **D%**	**Q_P/Q_T** **C**	Sa 6150m^2 1F	**A_C m^2** **A_I%**	**A_O m^2** **D%**	**Q_P/Q_T** **C**
Amaranthus interruptus	na	14m^2 0.02% 1/9Q	448m^2 11.0% 0/5Q	9166m^2 40.0% 7/7Q	11602m^2 45.0% 8/8Q	na	307m^2 46.7% 3/3Q	1m^2 0.3% 1/3Q	na	470m^2 14.0% 3/5Q	119333m^2 90.5%	22008m^2 18.4%	23/35Q 7C	na	na	na	na
Cuscuta victoriana	na	na	na	na	na	na	na	11m^2 2.7% 3/3Q	na	604m^2 18.0% 5/5Q	3772m^2 2.9%	615m^2 16.3%	8/8Q 2C	na	na	na	na
Eragrostis cumingii	na	52727m^2 84.9% 9/9Q	24m^2 0.6% 2/5Q	9493m^2 41.4% 7/7Q	773m^2 3.0% 4/8Q	311m^2 3.8% 4/8Q	45m^2 6.9% 0/3Q	na	0.2m^2 0.03% 1/3Q	238m^2 7.1% 0/5Q	127814m^2 96.9%	63614m^2 49.8%	27/40Q 8C	1m^2 0.01% 1/37Q	6150m^2 100.0%	1m^2 0.0%	1/37Q 1C
Lepturus repens	na	na	448m^2 11.0% 0/5Q	1696m^2 7.4% 0/7Q	na	4121m^2 49.6% 8/8Q	11m^2 1.7% 1/3Q	na	2m^2 0.3% 1/3Q	na	36546m^2 27.7%	6278m^2 17.2%	10/14Q 5C	1840m^2 29.9% 37/37Q	6150m^2 100.0%	1840m^2 29.9%	37/37Q 1C
Sesbania cannabina	74m^2 2.0% 1/1Q	5281m^2 8.5% 0/9Q	na	131m^2 0.6% 2/7Q	6m^2 0.03% 1/8Q	na	121m^2 18.3% 3/3Q	na	na	na	115177m^2 87.3%	5613m^2 4.9%	7/19Q 5C	na	na	na	na
Sida pusilla	na	na	na	na	na	na	na	na	na	503m^2 15.0% 1/5Q	3356m^2 2.5%	503m^2 15.0%	1/5Q 1C	na	na	na	na
Sporobolus virginicus	na	na	na	na	na	na	na	367m^2 88.3% 3/3Q	228m^2 38.3% 3/3Q	na	1010m^2 0.8%	595m^2 58.9%	6/6Q 2C	na	na	na	na
*Portulaca oleracea**	3139m^2 85.0% 1/1Q	483m^2 0.8% 2/9Q	na	na	na	42m^2 0.5% 2/8Q	66m^2 10.0% 3/3Q	na	6m^2 1.0% 1/3Q	503m^2 15.0% 3/5Q	78734m^2 59.7%	4239m^2 5.4%	12/29Q 6C	212m^2 3.4% 18/37Q	6150m^2 100.0%	212m^2 3.4%	18/37Q 1C
*Tribulus cistoides**	na	5281m^2 8.5% 0/9Q	4044m^2 99.2% 5/5Q	1696m^2 7.4% 0/7Q	1367m^2 5.3% 0/8Q	na	na	51m^2 12.3% 3/3Q	na	1040m^2 31.0% 5/5Q	118675m^2 90.0%	13478m^2 11.4%	13/13Q 6C	na	na	na	na
All species	3213m^2 87.0% 1/1Q	53224m^2 85.7% 9/9Q	4068m^2 99.8% 5/5Q	18790m^2 82.0% 7/7Q	12382m^2 48.0% 8/8Q	4474m^2 53.9% 8/8Q	504m^2 76.7% 3/3Q	431m^2 103.7% 3/3Q	236m^2 39.7% 3/3Q	3121m^2 93.0% 5/5Q	131923m^2 100%	100443m^2 76.1%	52/52Q 10C	2053m^2 33.4% 37/37Q	6150m^2 100%	2053m^2 33.4%	37/37Q 1C

SI Table 5. Showing GPS coordinates of the ant stations sampled on West, Middle and East islands at Ashmore Reef in May 2019.

Waypoint	Latitude °)	Longitude °)	Waypoint	Latitude °)	Longitude °)
E1	-12.259633	123.0958	M1	-12.263999	123.0325
E2	-12.259633	123.09625	M2	-12.263999	123.03295
E3	-12.259633	123.0967	M3	-12.264450	123.03205
E4	-12.259999	123.0949	M4	-12.264450	123.0325
E5	-12.259999	123.09535	M5	-12.264450	123.03295
E6	-12.259999	123.0958	M6	-12.264450	123.033383
E7	-12.259999	123.09625	M7	-12.264450	123.033833
E8	-12.259999	123.0967	M8	-12.264883	123.031616
E9	-12.259999	123.09715	M9	-12.264883	123.03205
E10	-12.260383	123.094	M10	-12.264883	123.0325
E11	-12.260383	123.09445	M11	-12.264883	123.03295
E12	-12.260383	123.0949	M12	-12.264883	123.033383
E13	-12.260383	123.09535	M13	-12.264883	123.033833
E14	-12.260383	123.0958	M14	-12.264883	123.034283
E15	-12.260383	123.09625	M15	-12.265333	123.031166
E16	-12.260383	123.0967	M16	-12.265333	123.031616
E17	-12.260383	123.09715	M17	-12.265333	123.03205
E19	-12.260749	123.093533	M18	-12.265333	123.0325
E20	-12.260749	123.094	M19	-12.265333	123.03295
E21	-12.260749	123.09445	M20	-12.265333	123.033383
E22	-12.260749	123.0949	M21	-12.265333	123.033833
E23	-12.260749	123.09535	M22	-12.265333	123.034283
E24	-12.260749	123.0958	M23	-12.265333	123.034716
E25	-12.260749	123.09625	M24	-12.265783	123.031166
E26	-12.260749	123.0967	M25	-12.265783	123.031616
E27	-12.260749	123.09715	M26	-12.265783	123.03205
E28	-12.260749	123.097616	M27	-12.265783	123.0325
E29	-12.261116	123.09445	M28	-12.265783	123.03295
E30	-12.261116	123.0949	M29	-12.265783	123.033383
E31	-12.261116	123.09535	M30	-12.265783	123.033833
E32	-12.261116	123.0958	M31	-12.265783	123.034283
E33	-12.261116	123.09625	M32	-12.265783	123.034716
E34	-12.261116	123.0967	M33	-12.265783	123.035166

E35	-12.261116	123.09715	M34	-12.265783	123.035616
E36	-12.261116	123.097616	M35	-12.266216	123.031616
E37	-12.261483	123.0949	M36	-12.266216	123.03205
E38	-12.261483	123.09535	M37	-12.266216	123.0325
E39	-12.261483	123.0958	M38	-12.266216	123.03295
E40	-12.261483	123.09625	M39	-12.266216	123.033383
E41	-12.261483	123.0967	M40	-12.266216	123.033833
E42	-12.261483	123.09715	M41	-12.266216	123.034283
E43	-12.261483	123.097616	M42	-12.266216	123.034716
E44	-12.261483	123.098066	M43	-12.266216	123.035166
E45	-12.261850	123.0958	M44	-12.266216	123.035616
E46	-12.261850	123.09625	M45	-12.266216	123.03605
E47	-12.261850	123.0967	M46	-12.26666668	123.03295
E48	-12.261850	123.09715	M47	-12.26666668	123.033383
E49	-12.261850	123.0976167	M48	-12.26666668	123.033833
E50	-12.261850	123.098066	M49	-12.266666	123.034283
E51	-12.262216	123.09715	M50	-12.266666	123.034716
E52	-12.262216	123.0976167	M51	-12.266666	123.035166
E53	-12.262216	123.098066	M52	-12.266666	123.035616
E54	-12.261850	123.095346			
E55	-12.262216	123.096697			
E56	-12.262216	123.096228			
E57	-12.262216	123.095795			

W1	-12.241000	122.962833	W27	-12.242350	122.968233
W2	-12.241000	122.963516	W28	-12.242350	122.968916
W3	-12.241000	122.964183	W29	-12.242350	122.969583
W4	-12.241000	122.964866	W30	-12.243033	122.964183
W5	-12.241000	122.965533	W31	-12.243033	122.964866
W6	-12.241000	122.966216	W32	-12.243033	122.965533
W7	-12.241000	122.966883	W33	-12.243033	122.966216
W8	-12.241000	122.967566	W34	-12.243033	122.966883
W9	-12.241000	122.968233	W35	-12.243033	122.967566
W10	-12.241683	122.962833	W36	-12.243033	122.968233
W11	-12.241683	122.963516	W37	-12.243033	122.968916

W12	-12.241683	122.964183	W38	-12.243033	122.969583
W13	-12.241683	122.964866	W39	-12.243700	122.965533
W14	-12.241683	122.965533	W40	-12.243700	122.966216
W15	-12.241683	122.966216	W41	-12.243700	122.966883
W16	-12.241683	122.966883	W42	-12.243700	122.967566
W17	-12.241683	122.967566	W43	-12.243700	122.968233
W18	-12.241683	122.968233	W44	-12.243700	122.968916
W19	-12.241683	122.968916	W45	-12.243700	122.969583
W20	-12.242350	122.963516	W46	-12.243700	122.970266
W21	-12.242350	122.964183	W47	-12.244383	122.967566
W22	-12.242350	122.964866	W48	-12.244383	122.968233
W23	-12.242350	122.965533	W49	-12.244383	122.968916
W24	-12.242350	122.966216	W50	-12.244383	122.969583
W25	-12.242350	122.966883	W51	-12.244383	122.970266
W26	-12.242350	122.967566	W52	-12.244383	122.970933

SI Table 6. Table showing ant abundance recorded at each lure

Waypoint	Lure 1	Lure 2	Lure 3	Lure 4	Waypoint	Lure 1	Lure 2	Lure 3	Lure 4
W1	0	50	>100	30	E1	2	0	50	50
W2	0	0	0	0	E2	>100	>100	>100	>100
W3	>100	6	0	0	E3	>100	>100	50	50
W4	12	50	>100	16	E4	0	>100	0	>100
W5	0	0	0	0	E5	>100	60	>100	>100
W6	0	0	0	0	E6	60	0	>100	49
W7	0	0	0	0	E7	>100	>100	>100	>100
W8	0	0	0	0	E8	0	16	>100	0
W9	0	0	0	0	E9	>100	>100	>100	0
W10	40	15	8	5	E10	>100	0	20	10
W11	0	0	0	0	E11	0	0	0	0
W12	0	0	0	0	E12	0	40	0	0
W13	0	0	0	0	E13	9	0	40	60
W14	0	0	11	0	E14	20	40	40	0
W15	0	0	0	0	E15	>100	>100	>100	>100
W16	0	0	0	0	E16	50	40	>100	>100
W17	0	0	0	0	E17	>100	>100	>100	2

Waypoint	Lure 1	Lure 2	Lure 3	Lure 4	Waypoint	Lure 1	Lure 2	Lure 3	Lure 4
W18	10	80	0	0	E18	0	3	10	20
W19	70	>100	0	>100	E19	23	0	0	0
W20	0	0	0	0	E20	>100	0	0	0
W21	1	0	0	0	E21	0	0	0	40
W22	0	0	0	0	E22	0	>100	>100	>100
W23	0	0	0	0	E23	60	>100	>100	0
W24	0	0	0	0	E24	40	0	>100	>100
W25	0	0	0	0	E25	>100	0	>100	>100
W26	0	0	0	0	E26	>100	0	>100	0
W27	0	0	0	0	E27	>100	>100	>100	0
W28	0	0	0	0	E28	>100	>100	>100	>100
W29	0	0	0	0	E29	0	>100	>100	>100
W30	23	0	0	0	E30	>100	0	0	0
W31	0	0	0	0	E31	>100	0	0	0
W32	0	0	0	0	E32	7	>100	20	0
W33	0	0	0	0	E33	50	0	0	200
W34	0	0	0	0	E34	0	1	0	0
W35	0	0	0	0	E35	0	0	0	0
W36	0	0	0	0	E36	50	200	0	50
W37	0	0	0	0	E37	>100	>100	>100	>100
W38	>100	>100	>100	>100	E38	0	0	0	0
W39	>100	>100	0	0	E39	0	0	0	0
W40	0	0	0	0	E40	1	0	>100	>100
W41	0	0	0	0	E41	0	0	0	0
W42	0	0	0	0	E42	0	60	0	20
W43	0	0	0	0	E43	>100	>100	>100	>100
W44	0	0	0	0	E44	60	>100	20	10
W45	0	0	0	0	E45	>100	20	20	>100
W46	0	0	0	0	E46	>100	>100	>100	>100
W47	50	60	70	>100	E47	0	0	0	0
W48	0	0	>100	>100	E48	0	0	2	6
W49	0	0	0	0	E49	>100	0	>100	1
W50	0	>100	90	0	E50	0	>100	>100	>100
W51	60	>100	>100	>100	E51	>100	70	80	>100
W52	0	0	0	0	E52	0	>100	16	0
					E53	>100	0	300	0

Waypoint	Lure 1	Lure 2	Lure 3	Lure 4	Waypoint	Lure 1	Lure 2	Lure 3	Lure 4
					E54	>100	>100	>100	>100
					E55	>100	70	>100	40
					E56	>100	12	0	0
					E57	>100	0	>100	>100
M1	>100	>100	>100	50	M27	0	40	4	0
M2	>100	7	50	>100	M28	0	0	1	1
M3	45	80	50	>100	M29	60	60	>100	30
M4	>100	>100	0	>100	M30	2	70	>100	0
M5	3	>100	0	>100	M31	0	50	>100	0
M6	>100	0	>100	0	M32	0	0	0	2
M7	>100	>100	60	60	M33	0	0	0	0
M8	3	>100	50	>100	M34	0	0	0	0
M9	10	0	0	0	M35	0	0	0	0
M10	0	0	0	0	M36	60	40	60	50
M11	4	0	0	0	M37	6	0	20	0
M12	0	0	0	0	M38	0	50	80	0
M13	>100	>100	>100	30	M39	0	30	0	0
M14	5	30	0	10	M40	15	0	3	10
M15	>100	>100	>100	>100	M41	0	45	5	0
M16	14	0	0	80	M42	0	0	0	0
M17	>100	>100	>100	>100	M43	0	0	0	0
M18	>100	>100	>100	>100	M44	0	0	0	0
M19	0	0	0	0	M45	0	0	0	0
M20	>100	0	0	0	M46	15	10	8	3
M21	>100	6	>100	>100	M47	80	2	1	50
M22	>100	0	>100	>100	M48	1	30	80	0
M23	25	0	0	25	M49	0	0	0	0
M24	0	0	0	>100	M50	0	0	0	0
M25	0	1	3	0	M51	0	0	0	0
M26	50	>100	>100	0	M52	0	0	0	0

SI Table 7. Table showing GPS coordinates of seagrass monitoring and growth rate sites established at Ashmore Reef during 2019 surveys

Site code	Latitude (	Longitude (
SGHO1	-12.21105	123.0624833
SGHO2	-12.2034333	123.0540833
SGTC1	-12.22483	122.977567
SGTC2	-12.22748	122.988367
SGTH1	-12.1987333	123.1348167
SGTH2	-12.2858333	123.1207
SGTH3	-12.2805667	123.0048667
SGTH4	-12.24478	122.94095

SI Table 8. Table showing locations of all sites at Ashmore Reef where invertebrate, fish and seagrass surveys were conducted

SITE_NAME	LAT (	LON (	X_COORD	Y_COORD	HABITAT_2019	INVERT	FISH	SEAGRASS	INVERT_2013
Site-0001	-12.2931	123.0214	502322	8641041	Shallow	X	X		X
Site-0002	-12.2617	122.9809	497924	8644508	Shallow	X	X		X
Site-0003	-12.2609	123.0547	505945	8644596	Sand	X	X		X
Site-0004	-12.2362	123.128	513926	8647325	Shallow lagoon	X	X		X
Site-0005	-12.2544	122.9588	495518	8645314	Shallow	X	X	X	X
Site-0006	-12.2032	123.0533	505799	8650979	Shallow lagoon	X	X		X
Site-0007	-12.2445	123.042	504563	8646411	Shallow lagoon	X	X		X
Site-0008	-12.2571	123.1465	515930	8645009	Shallow	X	X		X
Site-0009	-12.2573	122.9779	497596	8644999	Shallow	X	X		X
Site-0010	-12.1937	123.0746	508112	8652025	Shallow	X	X		X
Site-0011	-12.2331	123.0996	510832	8647665	Shallow lagoon	X	X		X
Site-0012	-12.2761	123.1188	512919	8642918	Sand	X	X		X
Site-0013	-12.2353	123.0198	502153	8647428	Shallow lagoon	X	X		X
Site-0014	-12.2241	123.0733	507977	8648666	Shallow lagoon	X	X		X
Site-0015	-12.2218	122.991	499017	8648917	Deep lagoon	X	X		X
Site-0017	-12.2904	123.0748	508129	8641332	Shallow	X	X		X
Site-0018	-12.2773	123.0113	501227	8642784	Shallow lagoon	X	X		X
Site-0019	-12.2823	123.0564	506134	8642232	Shallow	X	X		X
Site-0020	-12.2308	123.1127	512259	8647926	Shallow lagoon	X	X		X
Site-0021	-12.2433	122.9613	495787	8646545	Sand	X	X		X
Site-0022	-12.2133	123.0616	506699	8649856	Shallow lagoon	X	X		X
Site-0023	-12.2553	123.0623	506771	8645211	Shallow lagoon	X	X		X
Site-0024	-12.2773	123.1346	514642	8642775	Shallow	X	X		X
Site-0025	-12.2312	123.0035	500377	8647878	Deep lagoon	X	X		X
Site-0026	-12.1987	123.0476	505181	8651477	Deep lagoon	X	X		X
Site-0027	-12.2345	123.0806	508764	8647509	Shallow lagoon	X	X		X
Site-0029	-12.2435	123.0274	502975	8646522	Shallow lagoon	X	X		X
Site-0030	-12.2268	123.0845	509192	8648370	Shallow lagoon	X	X		X

SITE_NAME	LAT (	LON (	X_COORD	Y_COORD	HABITAT_2019	INVERT	FISH	SEAGRASS	INVERT_2013
Site-0031	-12.2076	123.0337	503661	8650486	Shallow	X	X		X
Site-0032	-12.2005	123.1007	510954	8651270	Shallow	X	X		
Site-0033	-12.2556	122.9423	493730	8645179	Shallow	X	X		
Site-0034	-12.2603	123.0221	502405	8644665	Shallow lagoon	X	X		X
Site-0035	-12.2773	123.0826	508982	8642780	Sand	X	X		
Site-0036	-12.2504	123.1191	512957	8645752	Sand	X	X		
Site-0037	-12.2398	122.9563	495242	8646930	Shallow	X	X		
Site-0038	-12.2232	123.0463	505034	8648763	Deep lagoon	X	X		X
Site-0039	-12.2559	123.0487	505298	8645152	Sand	X	X		X
Site-0040	-12.2741	123.1388	515094	8643132	Shallow	X	X		X
Site-0041	-12.2299	122.9895	498861	8648026	Shallow	X	X		X
Site-0042	-12.1981	123.0668	507261	8651544	Shallow	X			X
Site-0043	-12.2484	123.0755	508211	8645982	Shallow lagoon	X	X		X
Site-0044	-12.2148	123.1274	513863	8649694	Sand	X	X		
Site-0045	-12.2539	123.0342	503718	8645370	Sand	X	X		X
Site-0046	-12.206	123.0744	508095	8650663	Shallow	X	X		X
Site-0047	-12.2217	123.0294	503196	8648928	Shallow lagoon	X	X		X
Site-0048	-12.215	123.1346	514643	8649670	Sand	X	X		
Site-0049	-12.2444	122.9332	492736	8646417	Shallow	X	X		
Site-0050	-12.2876	123.0336	503656	8641646	Shallow	X	X		
Site-0051	-12.2827	123.0981	510673	8642187	Shallow	X	X		X
Site-0052	-12.2307	123.1392	515145	8647931	Sand	X	X		
Site-0053	-12.2499	123.0016	500177	8645813	Shallow lagoon	X	X		X
Site-0054	-12.2147	123.0547	505954	8649699	Shallow lagoon	X	X		X
Site-0055	-12.235	123.0637	506928	8647464	Shallow lagoon	X	X		X
Site-0057	-12.2565	123.0128	501387	8645088	Sand	X	X		X
Site-0058	-12.1807	123.0572	506218	8653466	Shallow				X
Site-0060	-12.2293	123.1003	510913	8648084	Shallow lagoon	X	X		
Site-0061	-12.2604	122.9981	499798	8644652	Sand	X	X		X
Site-0062	-12.2774	123.0393	504273	8642767	Sand	X	X		
Site-0063	-12.2293	123.0086	500931	8648092	Deep lagoon	X	X		
Site-0066	-12.2665	122.9863	498513	8643980	Shallow	X	X		
Site-0067	-12.2855	123.0579	506298	8641871	Shallow	X	X		
Site-0068	-12.2441	123.1205	513109	8646453	Shallow lagoon	X	X		

SITE_NAME	LAT (	LON (	X_COORD	Y_COORD	HABITAT_2019	INVERT	FISH	SEAGRASS	INVERT_2013
Site-0069	-12.2315	122.9752	497304	8647843	Shallow	X	X		
Site-0070	-12.2024	123.0447	504866	8651068	Shallow lagoon	X	X		X
Site-0071	-12.2387	123.0493	505367	8647056	Shallow lagoon	X			X
Site-0072	-12.2391	123.1465	515934	8646999	Shallow	X	X		
Site-0073	-12.2502	122.9858	498453	8645778	Sand	X	X		
Site-0074	-12.1963	123.085	509251	8651733	Shallow	X	X		X
Site-0075	-12.2394	123.0791	508605	8646978	Shallow lagoon	X	X		X
Site-0076	-12.2846	123.1212	513177	8641973	Shallow	X	X		X
Site-0077	-12.2358	123.0107	501158	8647372	Deep lagoon	X	X		
Site-0078	-12.2136	123.0919	510001	8649825	Shallow lagoon	X	X		X
Site-0079	-12.2259	122.9886	498757	8648469	Shallow	X	X		X
Site-0080	-12.2138	123.1221	513278	8649804	Shallow lagoon	X	X		
Site-0081	-12.2657	122.9621	495875	8644065	Edge S	X	X		X
Site-0082	-12.2852	123.0167	501815	8641904	Shallow	X	X		
Site-0083	-12.2622	123.09	509784	8644451	Sand	X	X		X
Site-0086	-12.2064	123.0616	506702	8650618	Deep lagoon	X	X		X
Site-0087	-12.245	123.0601	506537	8646354	Shallow lagoon	X	X		X
Site-0088	-12.2761	123.1405	515275	8642907	Edge S	X	X		X
Site-0089	-12.2393	122.9939	499332	8646985	Deep lagoon	X	X		X
Site-0090	-12.1881	123.0424	504617	8652641	Deep lagoon			X	
Site-0091	-12.2479	123.0852	509268	8646034	Shallow lagoon	X	X		X
Site-0092	-12.1901	123.1037	511279	8652420	Shallow	X	X		X
Site-0093	-12.2308	123.0297	503227	8647921	Shallow lagoon	X	X		X
Site-0094	-12.2155	123.0985	510713	8649615	Shallow lagoon	X			
Site-0096	-12.2028	123.1119	512173	8651013	Shallow lagoon	X	X		
Site-0097	-12.2514	122.9336	492777	8645643	Edge S	X	X		X
Site-0098	-12.2715	123.0102	501110	8643419	Shallow	X	X		X
Site-0099	-12.2621	123.0732	507957	8644464	Sand	X	X		X
Site-0100	-12.2533	123.1432	515569	8645431	Shallow	X	X		
Site-0101	-12.232	122.9594	495580	8647795	Shallow	X	X		
Site-0102	-12.221	123.0613	506672	8649010	Shallow lagoon	X	X		
Site-0103	-12.2513	123.0454	504942	8645658	Sand	X	X		

SITE_NAME	LAT (	LON (	X_COORD	Y_COORD	HABITAT_2019	INVERT	FISH	SEAGRASS	INVERT_2013
Site-0104	-12.273	123.1293	514064	8643257	Sand	X	X		
Site-0105	-12.2372	122.9883	498722	8647223	Deep lagoon	X	X		
Site-0106	-12.1906	123.0661	507186	8652374	Shallow	X	X		X
Site-0107	-12.2598	123.0797	508668	8644712	Sand	X	X		
Site-0108	-12.2267	123.1197	513016	8648375	Shallow lagoon	X	X		X
Site-0109	-12.2676	123.0042	500452	8643853	Shallow	X	X		
Site-0110	-12.2141	123.0759	508260	8649771	Deep lagoon	X	X		
Site-0111	-12.2176	123.0221	502407	8649381	Shallow	X	X		X
Site-0112	-12.2236	123.1437	515628	8648714	Shallow	X	X		
Site-0114	-12.2809	123.0345	503750	8642387	Shallow	X	X		
Site-0115	-12.2857	123.0895	509729	8641851	Shallow	X	X		
Site-0117	-12.2529	122.9923	499162	8645485	Sand	X	X		
Site-0118	-12.191	123.0768	508355	8652323	Shallow	X	X		
Site-0119	-12.2361	123.0912	509919	8647332	Shallow lagoon	X	X		
Site-0121	-12.2554	123.0169	501836	8645208	Sand	X	X		
Site-0122	-12.2162	123.072	507827	8649539	Deep lagoon	X	X		
Site-0123	-12.2283	123.0039	500428	8648197	Deep lagoon	X	X		
Site-0125	-12.285	123.0048	500523	8641928	Shallow	X	X		
Site-0126	-12.2857	123.0395	504297	8641851	Shallow	X	X		
Site-0127	-12.2084	123.0203	502213	8650397	Shallow	X	X		
Site-0128	-12.2256	123.1519	516518	8648491	Edge N	X	X		X
Site-0129	-12.2904	123.0231	502511	8641332	Shallow	X	X		
Site-0130	-12.2729	122.986	498482	8643271	Shallow	X	X		
Site-0131	-12.2871	123.0687	507467	8641701	Shallow	X	X		
Site-0133	-12.2426	122.9734	497107	8646619	Sand	X	X		
Site-0134	-12.2034	123.0542	505892	8650959	Shallow lagoon	X			
Site-0135	-12.2312	123.0515	505596	8647884	Shallow lagoon	X	X		
Site-0137	-12.2477	122.9798	497805	8646052	Sand	X	X		
Site-0138	-12.1911	123.0945	510276	8652316	Shallow	X	X		X
Site-0140	-12.2841	123.1238	513459	8642027	Shallow	X	X		
Site-0141	-12.2303	123.0322	503497	8647979	Shallow lagoon	X	X		
Site-0142	-12.2135	123.0981	510668	8649838	Shallow lagoon	X			
Site-0143	-12.2122	123.023	502507	8649977	Shallow	X	X		
Site-0145	-12.2661	122.9741	497179	8644021	Shallow	X	X		
Site-0146	-12.2837	123.008	500875	8642074	Shallow	X	X		

SITE_NAME	LAT (	LON (	X_COORD	Y_COORD	HABITAT_2019	INVERT	FISH	SEAGRASS	INVERT_2013
Site-0147	-12.2872	123.0702	507638	8641684	Shallow	X	X		
Site-0148	-12.2434	123.1089	511840	8646528	Shallow lagoon	X	X		
Site-0149	-12.2582	122.9722	496976	8644890	Shallow	X	X		
Site-0150	-12.2186	123.053	505765	8649276	Shallow lagoon	X	X		
Site-0151	-12.2488	123.046	505008	8645937	Sand	X	X		
Site-0152	-12.2602	123.1345	514630	8644671	Sand	X	X		
Site-0153	-12.2326	122.999	499891	8647728	Deep lagoon	X	X		
Site-0154	-12.1938	123.0408	504443	8652017	Deep lagoon	X	X		
Site-0157	-12.2459	123.0344	503746	8646257	Sand	X	X		
Site-0158	-12.2167	123.0848	509223	8649480	Deep lagoon	X	X		
Site-0159	-12.2283	123.0372	504044	8648203	Shallow lagoon	X	X		
Site-0161	-12.2485	122.9434	493842	8645964	Shallow	X	X		
Site-0162	-12.2652	123.013	501418	8644119	Sand	X	X		
Site-0164	-12.2499	123.1328	514444	8645805	Sand	X	X		
Site-0165	-12.2396	122.9475	494290	8646947	Shallow	X	X		
Site-0166	-12.2242	123.06	506528	8648653	Shallow lagoon	X	X		
Site-0167	-12.2439	123.0607	506598	8646477	Shallow lagoon	X	X		
Site-0169	-12.2392	122.9815	497988	8647000	Deep lagoon	X	X		
Site-0170	-12.1954	123.055	505979	8651843	Deep lagoon	X			
Site-0171	-12.251	123.0832	509053	8645687	Shallow lagoon	X	X		
Site-0173	-12.2648	123.0056	500607	8644164	Sand	X	X		
Site-0174	-12.2025	123.0838	509119	8651057	Shallow	X	X		
Site-0175	-12.216	123.0039	500426	8649567	Shallow	X	X		
Site-0176	-12.2106	123.1356	514745	8650157	Shallow	X	X		
Site-0177	-12.2581	122.9508	494649	8644904	Shallow	X	X		
Site-0178	-12.2122	123.0486	505286	8649980	Deep lagoon	X			
Site-0179	-12.2788	123.0847	509206	8642616	Sand	X	X		
Site-0181	-12.2524	123.0068	500735	8645533	Shallow lagoon	X	X		
Site-0182	-12.1841	123.0708	507700	8653092	Shallow	X			
Site-0184	-12.2763	123.1125	512235	8642892	Sand	X	X		
Site-0185	-12.2404	123.0112	501216	8646860	Deep lagoon	X	X		
Site-0189	-12.2766	122.9936	499307	8642863	Shallow	X	X		
Site-0190	-12.2847	123.0467	505082	8641963	Shallow	X	X		
Site-0191	-12.2545	123.1007	510955	8645305	Sand	X	X		

SITE_NAME	LAT (	LON (	X_COORD	Y_COORD	HABITAT_2019	INVERT	FISH	SEAGRASS	INVERT_2013
Site-0192	-12.2908	123.1181	512841	8641290	Edge S	X	X		X
Site-0194	-12.2614	123.0264	502866	8644538	Sand	X	X		
Site-0195	-12.2794	123.0669	507278	8642552	Shallow	X	X		
Site-0197	-12.2323	122.9642	496107	8647762	Shallow	X	X		
Site-0198	-12.2109	123.0596	506481	8650127	Shallow lagoon	X			
Site-0200	-12.2878	123.1333	514492	8641615	Shallow	X	X		
Site-0207	-12.2056	123.0263	502857	8650716	Shallow	X			
Site-0209	-12.2629	122.9555	495160	8644373	Edge S	X	X		X
Site-0210	-12.2777	123.0217	502361	8642733	Shallow	X	X		
Site-0213	-12.2528	122.9653	496227	8645495	Shallow	X	X		
Site-0214	-12.2234	123.0388	504224	8648742	Shallow lagoon	X	X		
Site-0226	-12.2808	123.0236	502563	8642397	Shallow	X	X		
Site-0227	-12.2876	123.0945	510280	8641638	Shallow	X	X		
Site-0229	-12.2272	122.97	496734	8648326	Shallow	X	X		
Site-0239	-12.2262	123.0194	502115	8648438	Shallow lagoon	X			
Site-0240	-12.2111	123.1438	515646	8650095	Shallow	X	X		
Site-0245	-12.2467	122.9942	499371	8646170	Shallow lagoon	X	X		
Site-0249	-12.2395	123.0156	501697	8646967	Shallow lagoon	X			
Site-0261	-12.2585	122.9574	495368	8644860	Shallow	X			
Site-0265	-12.2505	122.9906	498980	8645744	Sand	X	X		
Site-0269	-12.2354	123.0165	501795	8647411	Shallow lagoon	X			
Site-0271	-12.2203	122.9861	498488	8649091	Shallow lagoon	X	X		
Site-0309	-12.2478	123.0022	500243	8646046	Shallow lagoon	X	X		
Site-0321	-12.2919	123.0088	500957	8641168	Edge S	X	X		X
Site-0325	-12.2366	122.973	497058	8647281	Shallow lagoon	X	X		
Site-1346	-12.211	123.0052	500564	8650111	Edge N	X	X		X
Site-1347	-12.1893	123.1307	514218	8652507	Edge N	X	X		X
Site-1348	-12.1773	123.0527	505730	8653837	Edge N	X	X		X
Site-1349	-12.233	122.9359	493027	8647679	Edge N	X	X		X
Site-1352	-12.2078	123.0144	501568	8650465	Edge N	X	X		
Site-1353	-12.2208	122.9784	497654	8649030	Edge N	X	X		X
Site-1354	-12.1859	123.0942	510250	8652884	Edge N	X	X		X
Site-1355	-12.2017	123.0262	502845	8651146	Edge N	X	X		

SITE_NAME	LAT (	LON (	X_COORD	Y_COORD	HABITAT_2019	INVERT	FISH	SEAGRASS	INVERT_2013
Site-1356	-12.2389	123.1551	516874	8647029	Edge N	X	X		X
Site-1357	-12.2269	122.9637	496050	8648357	Edge N	X	X		
Site-1358	-12.1881	123.1088	511840	8652649	Shallow	X	X		X
Site-1359	-12.1775	123.0684	507441	8653823	Edge N	X	X		
Site-1360	-12.2325	123.1528	516624	8647733	Edge N	X	X		X
Site-1363	-12.1948	123.1356	514747	8651897	Shallow	X	X		X
Site-1365	-12.2414	122.9257	491919	8646748	Edge N	X	X		
Site-1367	-12.1945	123.0338	503682	8651937	Edge N	X	X		
Site-1368	-12.1971	123.1392	515140	8651651	Edge N	X	X		X
Site-1441	-12.265	122.958	495430	8644142	Edge S	X	X		X
Site-1442	-12.2938	123.0682	507416	8640959	Edge S	X	X		X
Site-1443	-12.2923	123.1007	510950	8641125	Edge S	X	X		X
Site-1445	-12.2591	122.9435	493854	8644797	Edge S	X	X		
Site-1447	-12.2896	123.0893	509714	8641421	Edge S	X	X		X
Site-1450	-12.2932	123.1254	513632	8641025	Edge S	X	X		X
Site-1451	-12.2716	122.9756	497343	8643408	Edge S	X	X		X
Site-1453	-12.296	123.0163	501770	8640712	Edge S	X	X		X
Site-1454	-12.2926	123.1111	512084	8641085	Edge S	X	X		X
Site-1456	-12.2659	123.1471	516000	8644041	Edge S	X	X		X
Site-1459	-12.2936	123.1042	511326	8640983	Edge S	X	X		
Site-1460	-12.2902	123.005	500541	8641351	Edge S	X	X		
Site-1461	-12.2609	122.9471	494244	8644595	Edge S	X	X		
Site-1462	-12.2947	123.0516	505610	8640862	Edge S	X	X		
SGTC1	-12.2248	122.9776	497560	8648585	Shallow	X		X	
SGTC2	-12.2275	122.9884	498735	8648292	Shallow	X		X	
SGTH4	-12.2448	122.941	493578	8646378	Shallow	X		X	
Grotto	-12.2771	123.011	501196	8642807	Shallow lagoon	X			

SI Table 9. Table showing the location and habitat classification of the 216 survey sites in 2019

Site number	Latitude °)	Longitude °)	Stratum	Benthic habitat	Fish habitat
1	-12.29305	123.02135	Flat	Reef flat	Reef flat
2	-12.2616977	122.9809109	Flat	Reef flat	Reef flat
3	-12.2608971	123.054665	Flat	Sand flat	Sand flat
4	-12.2361938	123.1280393	Flat	Shallow lagoon	East lagoon
5	-12.2544064	122.9587885	Flat	Reef flat	Reef flat
6	-12.2031744	123.053311	Flat	Deep lagoon	Channel
7	-12.2444859	123.0419548	Flat	Sand flat	Sand flat
8	-12.2571286	123.1464761	Flat	Reef flat	Reef flat
9	-12.2572573	122.9778952	Flat	Reef flat	Reef flat
10	-12.1937103	123.074572	Flat	Reef flat	Channel
11	-12.2331309	123.0995911	Flat	Shallow lagoon	East lagoon
12	-12.2760513	123.1187985	Flat	Sand flat	Sand flat
13	-12.2352914	123.0197952	Flat	Shallow lagoon	West lagoon
14	-12.2240868	123.0733393	Flat	Sand flat	Sand flat
15	-12.2218266	122.9909625	Flat	Reef edge	Channel
17	-12.2904094	123.0747555	Flat	Reef flat	Reef flat
18	-12.2772886	123.0112831	Flat	Shallow lagoon	Reef flat
19	-12.2822749	123.0564074	Flat	Reef flat	Reef flat
20	-12.2307655	123.1127102	Flat	Shallow lagoon	East lagoon
21	-12.2432746	122.9612635	Flat	Sand flat	Sand flat
22	-12.2133283	123.0615871	Flat	Sand flat	Sand flat
23	-12.2553339	123.0622589	Flat	Shallow lagoon	East lagoon
24	-12.2773372	123.1346433	Flat	Reef flat	Reef flat
25	-12.2312227	123.0034662	Flat	Deep lagoon	West lagoon
26	-12.1986719	123.0476288	Flat	Deep lagoon	Channel
27	-12.2345478	123.0805781	Flat	Shallow lagoon	East lagoon
29	-12.243484	123.0273537	Flat	Sand flat	Sand flat
30	-12.2267604	123.0845107	Flat	Shallow lagoon	East lagoon
31	-12.2076359	123.0336566	Flat	Reef flat	Reef flat
32	-12.2005296	123.1007005	Flat	Sand flat	Sand flat
33	-12.2556243	122.9423477	Flat	Reef flat	Reef flat

Site number	Latitude °)	Longitude °)	Stratum	Benthic habitat	Fish habitat
34	-12.2602777	123.0221142	Flat	Sand flat	Sand flat
35	-12.2773126	123.0825957	Flat	Sand flat	Sand flat
36	-12.2504227	123.1191365	Flat	Sand flat	Sand flat
37	-12.2397922	122.9562531	Flat	Sand flat	Sand flat
38	-12.2232155	123.0462817	Flat	Shallow lagoon	East lagoon
39	-12.2558702	123.0487148	Flat	Sand flat	Sand flat
40	-12.2741067	123.138798	Flat	Reef flat	Reef flat
41	-12.2298841	122.989528	Flat	Reef flat	Reef flat
42	-12.1980621	123.06675	Flat	Reef flat	Reef flat
43	-12.2483583	123.0754976	Flat	Shallow lagoon	East lagoon
44	-12.2147707	123.1274498	Flat	Sand flat	Sand flat
45	-12.253901	123.0341866	Flat	Sand flat	Sand flat
46	-12.2060272	123.0744192	Flat	Sand flat	Channel
47	-12.2217257	123.0293833	Flat	Sand flat	Sand flat
48	-12.2149843	123.1346208	Flat	Sand flat	Sand flat
49	-12.2444267	122.9332108	Flat	Reef flat	Reef flat
50	-12.287578	123.0336208	Flat	Reef flat	Reef flat
51	-12.2826701	123.0981476	Flat	Reef flat	Reef flat
52	-12.2307081	123.1392442	Flat	Sand flat	Sand flat
53	-12.249897	123.0016275	Flat	Sand flat	Sand flat
54	-12.2147495	123.0547383	Flat	Sand flat	Sand flat
55	-12.2349592	123.0636976	Flat	Shallow lagoon	East lagoon
57	-12.256453	123.0127534	Flat	Sand flat	Sand flat
60	-12.2293415	123.1003344	Flat	Shallow lagoon	East lagoon
61	-12.2603961	122.9981426	Flat	Sand flat	Sand flat
62	-12.2774398	123.0392932	Flat	Sand flat	Sand flat
63	-12.2292873	123.0085596	Flat	Deep lagoon	West lagoon
66	-12.2664728	122.9863266	Flat	Reef flat	Reef flat
67	-12.2855392	123.0579163	Flat	Reef flat	Reef flat
68	-12.2440829	123.1205312	Flat	Shallow lagoon	East lagoon
69	-12.2315381	122.9752127	Flat	Reef flat	Reef flat
70	-12.2023711	123.0447336	Flat	Shallow lagoon	Channel

Site number	Latitude °)	Longitude °)	Stratum	Benthic habitat	Fish habitat
71	-12.2386518	123.0493461	Flat	Shallow lagoon	East lagoon
72	-12.2391326	123.146503	Flat	Reef flat	Reef flat
73	-12.2502131	122.9857757	Flat	Sand flat	Sand flat
74	-12.1963479	123.0850434	Flat	Reef flat	Channel
75	-12.2393502	123.0791176	Flat	Shallow lagoon	East lagoon
76	-12.2845961	123.1211749	Flat	Reef flat	Reef flat
77	-12.2358	123.01065	Flat	Shallow lagoon	West lagoon
78	-12.2136001	123.0919441	Flat	Shallow lagoon	East lagoon
79	-12.2258779	122.988572	Flat	Reef flat	Reef flat
80	-12.2137783	123.1220711	Flat	Sand flat	Sand flat
81	-12.2657019	122.9620694	Flat	Reef edge	Reef slope
82	-12.2852464	123.0166907	Flat	Reef flat	Reef flat
83	-12.2621991	123.0899655	Flat	Sand flat	Sand flat
86	-12.2064373	123.0616131	Flat	Shallow lagoon	Channel
87	-12.244998	123.0601049	Flat	Shallow lagoon	East lagoon
88	-12.2761405	123.1404635	Flat	Reef edge	Reef slope
89	-12.2392982	122.9938582	Flat	Deep lagoon	West lagoon
91	-12.2478852	123.0852162	Flat	Shallow lagoon	East lagoon
92	-12.1901288	123.1036842	Flat	Reef flat	Channel
93	-12.2308322	123.0296693	Flat	Sand flat	Sand flat
94	-12.215497	123.0984905	Flat	Shallow lagoon	East lagoon
96	-12.2028494	123.1119078	Flat	Sand flat	Sand flat
97	-12.2514263	122.933586	Flat	Reef edge	Reef slope
98	-12.2715462	123.010207	Flat	Sand flat	Sand flat
99	-12.2620865	123.0731659	Flat	Sand flat	Sand flat
100	-12.2533141	123.1431547	Flat	Reef flat	Reef flat
101	-12.2319703	122.959362	Flat	Reef flat	Reef flat
102	-12.2209789	123.0613406	Flat	Sand flat	Sand flat
103	-12.2512949	123.0454407	Flat	Sand flat	Sand flat
104	-12.2729809	123.129326	Flat	Sand flat	Sand flat
105	-12.23715	122.98825	Flat	Deep lagoon	West lagoon
106	-12.1905563	123.0660587	Flat	Reef flat	Reef flat

Site number	Latitude °)	Longitude °)	Stratum	Benthic habitat	Fish habitat
107	-12.259842	123.079703	Flat	Sand flat	Sand flat
108	-12.2267022	123.1196683	Flat	Shallow lagoon	East lagoon
109	-12.2676216	123.0041563	Flat	Sand flat	Sand flat
110	-12.2140934	123.0759383	Flat	Deep lagoon	East lagoon
111	-12.2176298	123.0221291	Flat	Shallow lagoon	West lagoon
112	-12.223625	123.1436811	Flat	Reef flat	Reef flat
114	-12.2808768	123.0344843	Flat	Reef flat	Reef flat
115	-12.2857116	123.0894677	Flat	Reef flat	Reef flat
117	-12.252863	122.9922947	Flat	Sand flat	Sand flat
118	-12.1910148	123.0768051	Flat	Reef flat	Channel
119	-12.2361452	123.0911979	Flat	Shallow lagoon	East lagoon
121	-12.2553676	123.0168819	Flat	Sand flat	Sand flat
122	-12.2161925	123.0719581	Flat	Shallow lagoon	Channel
123	-12.2283379	123.003935	Flat	Deep lagoon	West lagoon
125	-12.28503	123.00481	Flat	Reef flat	Reef flat
126	-12.2857233	123.0395151	Flat	Reef flat	Reef flat
127	-12.208442	123.0203448	Flat	Reef flat	Reef flat
128	-12.225633	123.1518667	Flat	Reef edge	Reef slope
129	-12.2904186	123.0230915	Flat	Reef flat	Reef flat
130	-12.27288	122.98604	Flat	Reef flat	Reef flat
131	-12.287074	123.0686668	Flat	Reef flat	Reef flat
133	-12.2426068	122.9734003	Flat	Sand flat	Sand flat
134	-12.2033551	123.0541659	Flat	Sand flat	Sand flat
135	-12.2311636	123.0514502	Flat	Shallow lagoon	East lagoon
137	-12.2477349	122.9798177	Flat	Sand flat	Sand flat
138	-12.1910726	123.0944643	Flat	Reef edge	Reef slope
140	-12.2841066	123.1237679	Flat	Reef flat	Reef flat
141	-12.2303074	123.0321516	Flat	Sand flat	Sand flat
142	-12.2134805	123.0980761	Flat	Shallow lagoon	East lagoon
143	-12.21224	123.023048	Flat	Reef flat	Reef flat
145	-12.2661012	122.97406	Flat	Reef flat	Reef flat
146	-12.2837094	123.0080464	Flat	Reef flat	Reef flat

Site number	Latitude °)	Longitude °)	Stratum	Benthic habitat	Fish habitat
147	-12.2872274	123.0702393	Flat	Reef flat	Reef flat
148	-12.2434095	123.108863	Flat	Shallow lagoon	East lagoon
149	-12.2582424	122.9721942	Flat	Reef flat	Reef flat
150	-12.2185751	123.0530014	Flat	Sand flat	Sand flat
151	-12.2487717	123.0460471	Flat	Sand flat	Sand flat
152	-12.2601913	123.1345242	Flat	Sand flat	Sand flat
153	-12.2325833	122.999	Flat	Shallow lagoon	West lagoon
154	-12.1937897	123.0408436	Flat	Shallow lagoon	Channel
157	-12.2458796	123.034443	Flat	Sand flat	Sand flat
158	-12.2167224	123.0847925	Flat	Deep lagoon	East lagoon
159	-12.2282811	123.0371805	Flat	Sand flat	Sand flat
161	-12.2485256	122.9433791	Flat	Reef flat	Reef flat
162	-12.2652159	123.0130389	Flat	Sand flat	Sand flat
164	-12.2499372	123.1328088	Flat	Sand flat	Sand flat
165	-12.2396369	122.9475	Flat	Reef flat	Reef flat
166	-12.2242076	123.0600175	Flat	Sand flat	Sand flat
167	-12.2438856	123.0606655	Flat	Shallow lagoon	East lagoon
169	-12.239162	122.9815009	Flat	Deep lagoon	West lagoon
170	-12.1953607	123.0549641	Flat	Sand flat	Sand flat
171	-12.2510238	123.0832403	Flat	Shallow lagoon	East lagoon
173	-12.2648092	123.0055815	Flat	Sand flat	Sand flat
174	-12.2024615	123.0838319	Flat	Reef flat	Channel
175	-12.21595	123.00392	Flat	Reef flat	Channel
176	-12.2105798	123.1355563	Flat	Reef flat	Reef flat
177	-12.2581128	122.9507974	Flat	Reef flat	Reef flat
178	-12.2122095	123.0485965	Flat	Sand flat	Sand flat
179	-12.278795	123.084656	Flat	Reef flat	Reef flat
181	-12.252429	123.0067582	Flat	Sand flat	Sand flat
184	-12.2762891	123.1125088	Flat	Sand flat	Sand flat
185	-12.2404285	123.0111804	Flat	Shallow lagoon	West lagoon
189	-12.2765744	122.9936274	Flat	Reef flat	Reef flat
190	-12.2847093	123.0467338	Flat	Reef flat	Reef flat

Site number	**Latitude °)**	**Longitude °)**	**Stratum**	**Benthic habitat**	**Fish habitat**
191	-12.2544724	123.1007301	Flat	Sand flat	Sand flat
192	-12.2907739	123.1180878	Flat	Reef edge	Reef slope
194	-12.2614258	123.0263533	Flat	Sand flat	Sand flat
195	-12.2793787	123.0669268	Flat	Sand flat	Sand flat
197	-12.2322694	122.9642072	Flat	Reef flat	Reef flat
198	-12.210878	123.0595824	Flat	Sand flat	Sand flat
200	-12.2878279	123.1332692	Flat	Reef flat	Reef flat
207	-12.2055567	123.026265	Flat	Reef flat	Reef flat
209	-12.2629156	122.9554953	Flat	Reef edge	Reef slope
210	-12.2777492	123.0217111	Flat	Reef flat	Reef flat
213	-12.2527705	122.9653079	Flat	Reef flat	Reef flat
214	-12.2234066	123.0388347	Flat	Sand flat	Sand flat
226	-12.2807876	123.0235689	Flat	Reef flat	Reef flat
227	-12.2876333	123.0945333	Flat	Reef flat	Reef flat
229	-12.2271697	122.9699725	Flat	Reef flat	Reef flat
239	-12.2261578	123.0194451	Flat	Shallow lagoon	West lagoon
240	-12.2111363	123.1438399	Flat	Reef flat	Reef flat
245	-12.2466685	122.9942166	Flat	Sand flat	Sand flat
249	-12.2394606	123.0156029	Flat	Sand flat	Sand flat
261	-12.2585118	122.9574086	Flat	Reef flat	Reef flat
265	-12.2505208	122.9906213	Flat	Sand flat	Sand flat
269	-12.2354454	123.0165037	Flat	Deep lagoon	West lagoon
271	-12.22025	122.9861	Flat	Reef edge	Channel
309	-12.2477899	123.0022343	Flat	Sand flat	Sand flat
321	-12.2919025	123.0088008	Flat	Reef edge	Reef slope
325	-12.2366202	122.9729504	Flat	Sand flat	Sand flat
1346	-12.2110291	123.0051851	Edge N	Reef edge	Channel
1347	-12.18933	123.1307	Edge N	Reef edge	Reef slope
1348	-12.17733	123.052667	Edge N	Reef edge	Reef slope
1349	-12.2330148	122.9358891	Edge N	Reef edge	Reef slope
1352	-12.2078274	123.0144151	Edge N	Reef edge	Channel
1353	-12.2208	122.9784333	Edge N	Reef edge	Reef slope

Site number	Latitude °)	Longitude °)	Stratum	Benthic habitat	Fish habitat
1354	-12.1859361	123.0942235	Edge N	Reef edge	Reef slope
1355	-12.2016667	123.02615	Edge N	Reef edge	Reef slope
1356	-12.2388566	123.1551455	Edge N	Reef edge	Reef slope
1357	-12.2268886	122.9636839	Edge N	Reef edge	Reef slope
1358	-12.1880559	123.1088404	Edge N	Reef edge	Reef slope
1359	-12.17745	123.0684	Edge N	Reef edge	Reef slope
1360	-12.2324915	123.1528432	Edge N	Reef edge	Reef slope
1363	-12.1948445	123.1355667	Edge N	Reef edge	Reef slope
1365	-12.2414315	122.9256997	Edge N	Reef edge	Reef slope
1367	-12.1945141	123.033848	Edge N	Reef edge	Reef slope
1368	-12.1970667	123.1391833	Edge N	Reef edge	Reef slope
1441	-12.265005	122.9579776	Edge S	Reef edge	Reef slope
1442	-12.2937842	123.0681995	Edge S	Reef edge	Reef slope
1443	-12.292273	123.1006985	Edge S	Reef edge	Reef slope
1445	-12.2590791	122.9434872	Edge S	Reef edge	Reef slope
1447	-12.2896002	123.0893311	Edge S	Reef edge	Reef slope
1450	-12.2931667	123.1253667	Edge S	Reef edge	Reef slope
1451	-12.2716448	122.9755676	Edge S	Reef edge	Reef slope
1453	-12.2960259	123.0162775	Edge S	Reef edge	Reef slope
1454	-12.2926307	123.1111271	Edge S	Reef edge	Reef slope
1456	-12.2658821	123.1471246	Edge S	Reef edge	Reef slope
1459	-12.2935558	123.1041568	Edge S	Reef edge	Reef slope
1460	-12.2902477	123.0049751	Edge S	Reef edge	Reef slope
1461	-12.2609065	122.9470729	Edge S	Reef edge	Reef slope
1462	-12.294665	123.0515912	Edge S	Reef edge	Reef slope

SI Table 10. Showing category details of fish communities

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Acanthuridae	*Acanthurus olivaceus*	Herbivore	Adults occur singly or in schools and feed on surface film of detritus, diatoms, and fine filamentous algae covering sand and bare rock (Myers, R.F. 1991).
Acanthuridae	*Acanthurus mata*	Planktivore	Adults often seen in schools, feeding mid-water on plankton (Kuiter, R.H. and T. Tonozuka, 2001c).
Acanthuridae	*Naso tuberosus*	Herbivore	Diet consisted a high proportion of the green alga, Caulerpa.
Acanthuridae	*Naso lituratus*	Roving_herbivore	Feed mainly on leafy brown algae (Sargassum and Dictyota)
Acanthuridae	*Naso caesius*	Planktivore	Feeds in midwater on zooplankton (Allen, G.R. and M.V. Erdmann 2012).
Acanthuridae	*Naso vlamingii*	Planktivore/herbivore	Feeds mainly on benthic algae and weeds, also on debris and plankton invertebrates (Myers, R.F., 1991)
Acanthuridae	*Naso brachycentron*	Herbivore	Feeds on benthic algae (Sommer et al. 1996)
Acanthuridae	*Acanthurus xanthopterus*	Herbivore/Invertivore	Feeds on diatoms, hard corals, detritus film of sand, filamentous algae, hydroids, phytoplankton, pieces of fish as well as non-annelids, molluscs and other benthic crustaceans and planktonic invertebrates. (Myers, R.F., 1991, Kuiter, R.H. and T. Tonozuka, 2001)
Acanthuridae	*Acanthurus nigricans*	Herbivore	feeds on filamentous algae
Acanthuridae	*Naso annulatus*	Planktivore/Herbivore	Feeds on large zooplankton during the day and shelter within the reef during the night; also benthic algae (Sommer, C. et al. 1996; Kuiter, R.H. and T. Tonozuka 2001c)
Acanthuridae	*Zebrasoma veliferum*	Roving_herbivore	Feeds on leafy macroalgae
Acanthuridae	*Ctenochaetus striatus*	Roving_herbivore/Invertivore	Feeds on surface film of blue-green algae and diatoms (making this species a key link in the ciguatera food chain) as well as on various small invertebrates.

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Acanthuridae	*Acanthurus thompsoni*	Planktivore/Invertivore	Feeds on zooplankton, particularly large gelatinous forms, and on fish eggs and crustaceans (Randall, J.E. 1956)
Acanthuridae	*Zebrasoma scopas*	Roving_herbivore	Graze on algae
Acanthuridae	*Acanthurus lineatus*	Herbivore/Invertivore	Herbivorous but also feeds on crustaceans (Cornic, A. 1987)
Acanthuridae	*Acanthurus nigricauda*	Herbivore	http://www.iucnredlist.org/details/178017/0 - Its diet is dominated by organic detritus and calcareous sediments with very small portions of identifiable algae (Choat et al. 2002b).
Acanthuridae	*Naso brevirostris*	Planktivore/herbivore	Juveniles and subadults feed on benthic algae; adults feed on zooplankton.
Acanthuridae	*Naso fageni*	Roving_herbivore	Juveniles strict herbivores found on reef - adults forage for zooplankton in pelagic environment (http://reefkeeping.com/issues/2005-07/hcs3/index.htm)
Acanthuridae	*Naso unicornis*	Roving_herbivore	Mainly diurnal feed on coarse leafy brown algae like Sargassum.
Acanthuridae	*Naso hexacanthus*	Planktivore/Herbivore	Mainly diurnal, it feeds on zooplankton such as crab larvae, arrow worms, pelagic tunicates, and occasionally filamentous red algae
Acanthuridae	*Acanthurus triostegus*	Roving_herbivore	Often feeding near freshwater run-offs where certain algae grow on rocks that are grazed (Kuiter, R.H. and T. Tonozuka 2001c). Occasionally form schools; feed on filamentous algae in large aggregations. Feed on a variety of algae (Randall 1961, Abitia-Cardenas et al. 2011)
Acanthuridae	*Acanthurus blochii*	Roving_herbivore	They feed primarily on the algal film covering compacted sand ingesting the usual component of sand which probably aids in the trituration of the algal food in the thick-walled stomach also feed on diatoms and detritus (Randall, J.E. 1985).
Balistidae	*Melichthys niger*	Planktivore/Herbivore	Diet consists primarily of calcareous algae and zooplankton (Myers, R.F. 1991); also feed on phytoplankton (Fischer, W. et al. 1990).
Balistidae	*Sufflamen chrysopterum*	Small_invertivore	Feed on a wide variety of invertebrates
Balistidae	*Rhinecanthus rectangulus*	Invertivore/Herbivore/Piscivore	Feed on algae, detritus, molluscs, crustaceans, worms, echinoderms, fishes, sponges, foraminiferans, and eggs.

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Balistidae	*Rhinecanthus aculeatus*	Invertivore/Herbivore/Piscivore	Feed on algae, detritus, molluscs, crustaceans, worms, sea urchins, fishes, corals, tunicates, forams, and eggs (Randall, J.E. 1985)
Balistidae	*Balistoides viridescens*	Invertivore	Feed on sea urchins, coral, crabs and other crustaceans, molluscs and tube worms.
Balistidae	*Balistapus undulatus*	Herbivore/Invertivore/Piscivore	Feeds on a variety of benthic organisms such as algae, echinoderms, fishes, molluscs, tunicates, sponges, and hydrozoans.
Caesionidae	*Pterocaesio digramma*	Planktivore	Feed on zooplankton in midwater aggregations (Carpenter, K. E., 1988)
Caesionidae	*Caesio teres*	Planktivore	Feeds on zooplankton in large midwater groups (Carpenter, K.E., 1987)
Caesionidae	*Pterocaesio tile*	Planktivore	Feeds on zooplankton in midwater aggregations
Caesionidae	*Caesio cuning*	Planktivore	unspecified zooplankton - fishbase food items
Carangidae	*Scomberoides lysan*	Piscivore/invertivore	Adults feed on small fishes and crustaceans (Fischer, W. et al. 1990) while juveniles feed on scales and epidermal tissues torn from other schooling fishes (Lieske, E. and R. Myers 1994; Allen, G.R. and M.V. Erdmann 2012).
Carangidae	*Caranx melampygus*	Invertivore/Piscivore	Feeds mainly on molluscs, other fishes (Smith-Vaniz, W.F., 1995) and crustaceans (Lieske, E. and R. Myers, 1994)
Carangidae	*Caranx ignobilis*	Large_invertivore	They feed on crustaceans (like crabs and spiny lobsters) and fishes at night (Honebrinnk, R. 1990).
Carcharhinidae	*Triaenodon obesus*	Piscivore/invertivore	Feeds on benthic animals such as fishes, octopi, spiny lobsters and crabs (Compagno 1984).
Carcharhinidae	*Carcharhinus amblyrhynchos*	Piscivore/invertivore	Feeds on reef fishes, squids, octopi, crabs, lobsters and shrimps (Compagno, L.J.V. 1984, Compagno, L.J.V. et al. 1989).
Carcharhinidae	*Galeocerdo cuvier*	Piscivore/invertivore	Nocturnal feeder on other sharks, rays, bony fishes, marine mammals, tortoises, seabirds, sea snakes, squids, gastropods, crustaceans, detritus also including toxic or armoured fish species such as *Lactoria cornuta* or *Diodon hystrix*, porpoises, whales, sea turtles,

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
			cephalopods, domestic animals and humans (Myers, R.F., 1999.). It also feeds on carrion and garbage, including cans, pieces of metal and burlap bags
Chaetodontidae	*Chaetodon lunula*	Invertivore	Adult feeds mainly on nudibranchs, tubeworm tentacles, and other benthic invertebrates, also feeds on algae and coral polyps.
Chaetodontidae	*Chaetodon adiergastos*	Corallivore	Coral- aquarium website
Chaetodontidae	*Chaetodon auriga*	Corallivore	Coral -aquarium websites
Chaetodontidae	*Chaetodon kleinii*	Corallivore	facultative corallivore, feeding on hard and soft corals, as well as algae, hydroids and zooplankton (http://www.iucnredlist.org/details/165628/0)
Chaetodontidae	*Chaetodon ornatissimus*	Corallivore	Feed exclusively on coral tissue (Myers, R.F. 1991).
Chaetodontidae	*Chaetodon lineolatus*	Herbivore/Invertivore	Feed mainly on coral polyps and anemones, but also on small invertebrates and algae (Myers, R.F. 1991).
Chaetodontidae	*Chaetodon pelewensis*	Invertivore/Corallivore	Feed mainly on coral polyps and small benthic invertebrates, also some benthic algae and weeds (Randall, J.E., G.R. Allen and R.C. Steene, 1990)
Chaetodontidae	*Forcipiger flavissimus*	Invertivore	Feed on a wide variety of animal prey including hydroids, fish eggs, small crustaceans but prefers tube feet of echinoderms, pedicilaria of sea urchins, and polychaete tentacles (Myers, R.F., 1991)
Chaetodontidae	*Chaetodon speculum*	Corallivore/Invertivore	Feed on coral polyps and invertebrates (Lieske, E. and R. Myers, 1994.)
Chaetodontidae	*Chaetodon punctatofasciatus*	Herbivore/Invertivore	Feed on filamentous algae, corals, and benthic invertebrates (Randall, J.E., G.R. Allen and R.C. Steene, 1990)
Chaetodontidae	*Chaetodon ephippium*	Herbivore/Invertivore	Feed on filamentous algae, small invertebrates, coral polyps, and fish eggs.

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Chaetodontidae	*Chaetodon melannotus*	Corallivore	Feed on octocorallian and scleractinian coral polyps
Chaetodontidae	*Chaetodon ulietensis*	Herbivore/Invertivore	Feed on plant and benthic invertebrates (Myers, R.F., 1991.)
Chaetodontidae	*Chaetodon unimaculatus*	Invertivore/Herbivore	Feed on soft and hard corals, also on polychaetes, small crustaceans, and filamentous algae (Myers, R.F., 1991)
Chaetodontidae	*Chaetodon ocellicaudus*	Corallivore	Feed on soft coral polyps (e.g. Litophyton viridis and species of the genera Sarcophyton, Nephthia, and Clavularia) (Lieske, E. and R. Myers, 1994)
Chaetodontidae	*Chaetodon trifascialis*	Corallivore	Feed on the polyps and mucus of tabular and staghorn Acropora corals (Myers, R.F., 1991.)
Chaetodontidae	*Chaetodon lunulatus*	Corallivore	Feed solely on live corals (Allen, G.R. et al. 1998).
Chaetodontidae	*Heniochus acuminatus*	Invertivore	Feeds on invertebrates, worms and benthic crustaceans such as shrimp and prawns (Allen, G.R., 2004) (Masuda, H. and G.R. Allen, 1993)
Chaetodontidae	*Chaetodon citrinellus*	Invertivore	It eats coral polyps, algae, and polychaete worms (http://australianmuseum.net.au/citron-butterflyfish-chaetodon-citrinellus)
Chaetodontidae	*Chaetodon vagabundus*	Herbivore/Invertivore	Omnivorous, feed on algae, coral polyps, crustaceans and worms (Cornic, A., 1987)
Chaetodontidae	*Chaetodon baronessa*	Corallivore	The species feeds on the polyps of Acropora corals (http://australianmuseum.net.au/triangular-butterflyfish-chaetodon-baronessa)
Dasyatidae	*Pateobatis fai*	Invertivore/Piscivore	Carnivore - feeds mostly on decapod crustaceans, and on cephalopods and bony fishes (http://fishesofaustralia.net.au/home/species/3528#moreinfo)
Dasyatidae	*Pastinachus ater*	Invertivore/Piscivore	Feeds on small bony fishes and a range of benthic invertebrates including crustaceans, molluscs, polychaete worms and sipunculids. (http://fishesofaustralia.net.au/home/species/2029#moreinfo)
Dasyatidae	*Taeniura lymma*	Small_invertivore	Migrates in groups into shallow sandy areas during the rising tide to feed on molluscs worms shrimps and crabs

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Ephippidae	*Platax orbicularis*	Piscivore/Invertivore/Herbivore	Feed on algae, invertebrates and small fishes (Bacchet, P., T. Zysman and Y. Lefevre, 2006)
Haemulidae	*Plectorhinchus lineatus*	Invertivore/Piscivore	Feeds on benthic invertebrates, crustaceans and fish (Thollot, P., 1996; Lieske, E. and R. Myers, 1994; Masuda, H. and G.R. Allen, 1993)
Kyphosidae	*Kyphosus vaigiensis*	Invertivore	Young (up to 5 cm) found among floating seaweed; feeding on small crustaceans. Adults are carnivorous during summer and autumn, feeding on benthic crustaceans, worms and some fish, however during winter they feed on *Endarachne binghamiae* Masuda, H. et al., 1984)
Labridae	*Thalassoma quinquevittatum*	Invertivore	Feed mainly on benthic crustaceans (crabs, shrimps), small fishes, gastropod molluscs, and sea urchins. (Randall, J.E., G.R. Allen and R.C. Steene, 1990)
Labridae	*Cheilinus chlorourus*	Invertivore	Feed mainly on benthic invertebrates such as molluscs and crustaceans (Nakamura, Y., M. Horinouchi, T. Nakai and M. Sano, 2003; Masuda, H. and G.R. Allen, 1993)
Labridae	*Cheilinus fasciatus*	Invertivore	Feed mainly on benthic, hard-shelled invertebrates, including molluscs, crustaceans and sea urchins (Gomon, M.F. and J.E. Randall, 1984)
Labridae	*Cheilinus fasciatus*	Invertivore	Feed mainly on benthic, hard-shelled invertebrates, including molluscs, crustaceans and sea urchins (Gomon, M.F. and J.E. Randall, 1984)
Labridae	*Bodianus axillaris*	Invertivore	Feed mainly on benthic, hard-shelled, invertebrates such as molluscs and crustaceans (Westneat, M.W., 2001). Also feed on sea stars and urchins (Baensch, H.A. and H. Debelius, 1997)
Labridae	*Cheilinus trilobatus*	Large_invertivore	Feed mainly on shelled benthic invertebrates such as molluscs and crustaceans, but occasionally take fishes (Myers, R.F. 1991).
Labridae	*Hemigymnus fasciatus*	Large_invertivore	Feed mainly on small crustaceans, molluscs and echinoderms (Westneat, M.W. 2001).
Labridae	*Halichoeres marginatus*	Invertivore	Feed on a wide variety of small invertebrates as well as fish eggs. (Randall, J.E., G.R. Allen and R.C. Steene, 1990)
Labridae	*Novaculichthys taeniourus*	Invertivore	Feed on molluscs, sea urchins, brittle stars, polychaetes, and crabs (Fischer, W. et al., 1990)

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Labridae	*Epibulus insidiator*	Invertivore/Piscivore	Feed on small coral-dwelling crustaceans and fishes (Lieske, E. and R. Myers, 1994)
Labridae	*Anampses melanurus*	Invertivore	Feed on small crustaceans, molluscs and polychaetes (Lieske, E. and R. Myers, 1994)
Labridae	*Halichoeres melanurus*	Invertivore	Feed on small invertebrates such as polychaetes, copepods, isopods and forams (Myers, R.F., 1991)
Labridae	*Thalassoma amblycephalum*	Planktivore	Feeds mainly on crustacean zooplankton (Myers, R.F., 1991)
Labridae	*Stethojulis bandanensis*	Planktivore/Invertivore	Feeds mainly on demersal planktonic crustaceans and small benthic invertebrates (Randall, J.E., 2000)
Labridae	*Oxycheilinus unifasciatus*	Piscivore/Invertivore	Feeds mainly on fishes, shrimps and other crustaceans (Westneat, M.W., 2001.)
Labridae	*Labrichthys unilineatus*	Corallivore	Feeds mainly on hard corals such as Acropora and coral polyps (Randall, J.E. and V.C. Springer, 1973; Masuda, H. and G.R. Allen, 1993)
Labridae	*Coris aygula*	Large_invertivore	Feeds mainly on hard-shelled invertebrates including crustaceans, molluscs and sea urchins (Westneat, M.W. 2001).
Labridae	*Halichoeres hortulanus*	Invertivore	Feeds mainly on hard-shelled prey, including molluscs, crustaceans and sea urchins (Westneat, M.W., 2001)
Labridae	*Stethojulis strigiventer*	Invertivore	Feeds mainly on invertebrates and benthic crustaceans such as amphipods and copepods. (Nakamura, Y., M. Horinouchi, T. Nakai and M. Sano, 2003; Masuda, H. and G.R. Allen, 1993)
Labridae	*Coris gaimard*	Invertivore	Feeds mainly on molluscs, crabs, and hermit crabs, and occasionally on ascidians and forams (Myers, R.F., 1991)
Labridae	*Gomphosus varius*	Invertivore	Feeds mainly on small benthic crustaceans, sometimes on small fishes, brittle stars, and molluscs (Randall, J.E., G.R. Allen and R.C. Steene, 1990)

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Labridae	*Thalassoma lunare*	Invertivore	Feeds mainly on small benthic invertebrates and fish eggs (Westneat, M.W., 2001)
Labridae	*Pseudocheilinus hexataenia*	Invertivore	Feeds mainly on small crustaceans (Myers, R.F., 1991)
Labridae	*Hemigymnus melapterus*	Large_invertivore	Feeds mainly on small invertebrates, especially crustaceans, polychaete worms, molluscs and brittle stars (Westneat, M.W. 2001).
Labridae	*Thalassoma hardwicke*	Invertivore/Planktivore/Piscivore	Feeds on benthic and planktonic crustaceans, small fishes, and foraminiferans (Masuda, H. and G.R. Allen, 1993)
Labridae	*Labroides dimidiatus*	Piscivore	Feeds on crustacean ectoparasites and mucus of other fishes (Westneat, M.W., 2001; Kuiter, R.H. and T. Tonozuka, 2001b)
Labridae	*Halichoeres trimaculatus*	Invertivore	Feeds on sand and rubble dwelling invertebrates (crustaceans, molluscs, forams, polychaetes, fish eggs, and small fishes). (Hiatt, R.W. and D.W. Strasburg, 1960
Labridae	*Leptojulis cyanopleura*	Planktivore	Feeds on zooplankton at least a meter above the substrate. (Randall, J.E., 1996)
Labridae	*Labroides bicolor*	Piscivore/Invertivore	Feeds primarily on small fishes and benthic crustaceans (Masuda, H. and G.R. Allen, 1993)
Labridae	*Coris pictoides*	Invertivore	No info
Labridae	*Halichoeres chrysus*	Invertivore	No info
Labridae	*Halichoeres nebulosus*	Invertivore	No info
Labridae	*Anampses caeruleopunctatus*	Large_invertivore	Young feed primarily on small crustaceans and polychaetes adults switch to larger crustaceans and molluscs as well as polychaetes (Myers, R.F. 1991).
Lethrinidae	*Monotaxis grandoculis*	Invertivore	Feed mainly on gastropods, ophiuroids, and echinoids. Pagurids and brachyuran crabs, polychaetes, tunicates, and holothurians are consumed in lesser quantities. (Hiatt, R.W. and D.W. Strasburg, 1960

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Lethrinidae	*Lethrinus obsoletus*	Invertivore	Feed on molluscs, crustaceans, and echinoderms (Sommer, C., W. Schneider and J.-M. Poutiers, 1996)
Lethrinidae	*Gnathodentex aureolineatus*	Invertivore	Feeds at night on benthic invertebrates like crabs and gastropods, occasionally on small fish (Hiatt, R.W. and D.W. Strasburg, 1960) and worms (Masuda, H. and G.R. Allen, 1993)
Lethrinidae	*Lethrinus atkinsoni*	Large_invertivore	Feeds mainly on crustaceans molluscs and fish
Lethrinidae	*Lethrinus rubrioperculatus*	Invertivore/Piscivore	Feeds mainly on crustaceans, fish, echinoderms and molluscs (Sommer, C., W. Schneider and J.-M. Poutiers, 1996)
Lethrinidae	*Lethrinus olivaceus*	Piscivore/invertivore	Feeds mainly on fish, crustaceans, and cephalopods.
Lutjanidae	*Lutjanus fulvus*	Piscivore/invertivore	Adults feed at night on fishes, shrimps, crabs, holothurians and cephalopods (Sommer, C., W. Schneider and J.-M. Poutiers, 1996)
Lutjanidae	*Aprion virescens*	Piscivore/invertivore	Feed mainly on fishes but also on shrimps crabs cephalopods and planktonic organisms (Sommer, C. et al. 1996; Kuiter, R.H. and T. Tonozuka, 2001a)
Lutjanidae	*Lutjanus bohar*	Piscivore/invertivore	Feed mainly on fishes, but also take shrimps, crabs, amphipods, stomatopods, gastropods and urochordates.
Lutjanidae	*Lutjanus fulviflamma*	Piscivore/invertivore	Feed mainly on fishes, shrimps, crabs and other crustaceans (Sommer, C. et al. 1996).
Lutjanidae	*Lutjanus gibbus*	Piscivore/invertivore	Feed on fishes, and a variety of invertebrates including shrimps, crabs, lobsters, stomatopods, cephalopods, echinoderms and ophiuroids (Allen, G.R. 1985).
Lutjanidae	*Lutjanus rivulatus*	Piscivore/Invertivore	Feed on fishes, cephalopods and benthic crustaceans (Sommer, C., W. Schneider and J.-M. Poutiers, 1996)
Lutjanidae	*Lutjanus kasmira*	Piscivore/Invertivore/Herbivore	Feed on fishes, shrimps, crabs, stomatopods, cephalopods, and planktonic crustaceans. Also take a variety of algae (Bagnis, R. et al. 1982).
Lutjanidae	*Lutjanus decussatus*	Piscivore/invertivore	Feed on some fishes and benthic crustaceans (Masuda, H. and G.R. Allen, 1993;)

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Lutjanidae	*Lutjanus monostigma*	Piscivore/invertivore	They feed mainly on fishes and benthic crustaceans, primarily crabs at night (Myers, R.F. 1999).
Lutjanidae	*Macolor niger*	Piscivore/Invertivore	They feed on fishes and crustaceans (Sommer, C., W. Schneider and J.-M. Poutiers, 1996)
Lutjanidae	*Macolor macularis*	Planktivore	They feed primarily on large zooplankton at night as well as some fish and benthic crustaceans (Allen, G.R., 1985; Masuda, H. and G.R. Allen, 1993)
Monacanthidae	*Aluterus scriptus*	Herbivore/Invertivore	Feed on algae, seagrass, hydrozoans, gorgonians, colonial anemones, and tunicates. (Gorelova, T.A. and B.I. Fedoryako, 1986)
Mullidae	*Parupeneus barberinoides*	Large_invertivore	Carnivorous, feeds mainly on worms, crustaceans, brittle stars, and small molluscs and heart urchins.
Mullidae	*Parupeneus indicus*	Large_invertivore	Feed on benthic invertebrates; the diet including small crabs, amphipods, shrimps, small octopuses, polychaete worms, and small fishes (Randall, J.E. 2004).
Mullidae	*Mulloidichthys flavolineatus*	Invertivore	Feed on crustaceans, molluscs, worms, heart urchins and foraminiferans. (Hiatt, R.W. and D.W. Strasburg, 1960)
Mullidae	*Parupeneus cyclostomus*	Large_invertivore	Feed primarily on small fishes, crustaceans, peanut worms, shrimps, crabs, octopi, and small gastropods during the day (Randall, J.E. 1985).
Mullidae	*Parupeneus pleurostigma*	Small_invertivore	Feeds on benthic animals like crabs, polychaetes and other worms during the day (Allen, G.R. and R.C. Steene 1988), also shrimps, heart urchins, gastropods, pelecypods, foraminiferans, brittle stars, and fishes (Myers, R.F. 1999).
Mullidae	*Parupeneus multifasciatus*	Invertivore	Feeds primarily on small crabs and shrimps during the day (Myers, R.F., 1991) also demersal fish eggs, molluscs, and foraminiferans (Myers, R.F., 1999)
Mullidae	*Parupeneus barberinus*	Large_invertivore	Forages on sand-dwelling invertebrates like polychaete worms and crustaceans. Feeding is diurnal; from 19 adult specimens for food-habit study, the prey in order by volume in the stomachs: crabs (portunid, anomuran, xanthid, and raninid), worms (mainly polychaetes but also sipunculids and unidentified), small bivalve molluscs, brachiopods, shrimps, small gastropods, isopods, amphipods, foraminifera, and a small unidentified eel (Randall, J.E. 2004).

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Nemipteridae	*Scolopsis bilineata*	Piscivore/invertivore	Feeds on small fishes and benthic invertebrates.
Nemipteridae	*Pentapodus emeryii*	Piscivore/Invertivore	Feeds on small fishes, crustaceans, ophiuroids and sipunculid worms. (Russell, B.C., 1990)
Pomacanthidae	*Pomacanthus sexstriatus*	Omnivore	Benthic plants, sponges & tunicates
Pomacanthidae	*Pomacanthus imperator*	Invertivore	Feed on sponges and other encrusting organisms; also on tunicates. (Anderson, C. and A. Hafiz, 1987)
Pomacanthidae	*Centropyge tibicen*	Herbivore/Corallivore	Feeds mainly on algae, coral polyps and tunicates (Masuda, H. and G.R. Allen, 1993)
Pomacanthidae	*Apolemichthys trimaculatus*	Invertivore/Herbivore	Feeds mainly on sponges, tunicates and algae (Myers, R.F., 1991)
Pomacentridae	*Amphiprion clarkii*	Herbivore	benthic algae/weeds (http://www.fishbase.org/TrophicEco/FoodItemsList.php?vstockcode=5713&genus=Amphiprion&species=clarkii)
Pomacentridae	*Pomacentrus moluccensis*	Herbivore/Planktivore	Feed mainly on algae and planktonic crustaceans (Allen, G.R., 1991)
Pomacentridae	*Pomacentrus tripunctatus*	Herbivore	Feed mainly on benthic algae (Allen, G.R., 1991.)
Pomacentridae	*Plectroglyphidodon lacrymatus*	Herbivore/Invertivore	Feed mainly on benthic algae and associated small invertebrates and fish eggs (Allen, G.R., 1991)
Pomacentridae	*Hemiglyphidodon plagiometopon*	Herbivore	Feed on algae (Myers, R.F., 1991)
Pomacentridae	*Dischistodus perspicillatus*	Herbivore	Feed on algae and debris (Allen, G.R., 1991)

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Pomacentridae	*Chromis cinerascens*	Herbivore/Planktivore	Feed on algae and zooplankton (Masuda, H. and G.R. Allen, 1993)
Pomacentridae	*Pomacentrus alexanderae*	Herbivore/Invertivore	Feed on algae, barnacle nauplii, copepods, fish eggs, and small gastropods. (Allen, G.R., 1991)
Pomacentridae	*Stegastes nigricans*	Herbivore/Invertivore	Feed on algae, gastropods, sponges, and copepods. (Allen, G.R. and A.R. Emery, 1985)
Pomacentridae	*Stegastes fasciolatus*	Herbivore	Feed on benthic algae and debris
Pomacentridae	*Plectroglyphidodon johnstonianus*	Herbivore/Corallivore	Feed on benthic algae and probably coral polyps. (Randall, J.E., 1985; Gochfeld, D.J., 1991; Masuda, H. and G.R. Allen, 1993)
Pomacentridae	*Pomacentrus vaiuli*	Herbivore/Invertivore	Feed on filamentous algae and small invertebrates (Myers, R.F., 1991)
Pomacentridae	*Plectroglyphidodon dickii*	Herbivore/Invertivore	Feed on filamentous algae, small benthic invertebrates, and occasionally small fishes (Myers, R.F., 1991)
Pomacentridae	*Dischistodus prosopotaenia*	Invertivore/Herbivore	Feed on harpacticoid and calanoid copepods, sponges, algae and debris (Nakamura, Y., M. Horinouchi, T. Nakai and M. Sano, 2003; Masuda, H. and G.R. Allen, 1993)
Pomacentridae	*Chrysiptera hemicyanea*	Planktivore/Herbivore	Feed on plankton and algae (Allen, G.R., 1991; Masuda, H. and G.R. Allen, 1993)
Pomacentridae	*Neopomacentrus azysron*	Planktivore	Feed on unspecified zooplankton (Masuda, H. and G.R. Allen, 1993)
Pomacentridae	*Chromis xanthura*	Planktivore	Feed on zooplankton (Allen, G.R., 1991)
Pomacentridae	*Chromis weberi*	Planktivore	Feed on zooplankton (Masuda, H. and G.R. Allen, 1993)
Pomacentridae	*Abudefduf sexfasciatus*	Planktivore/Herbivore	Feed on zooplankton and algae and aggregates high in the water column (Lieske, E. and R. Myers, 1994)

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Pomacentridae	*Amblyglyphidodon curacao*	Herbivore/Planktivore	Feed on zooplankton and filamentous algae (Hobson, E.S. and J.R. Chess, 1978)
Pomacentridae	*Pomacentrus coelestis*	Planktivore/Herbivore	Feed on zooplankton including larvae, copepods, polychaetes and to a lesser extent on benthic algae. (Hobson, E.S. and J.R. Chess, 1978)
Pomacentridae	*Chromis lepidolepis*	Planktivore	Feed on zooplankton such as copepods and larvae (Allen, G.R., 1991; Hobson, E.S. and J.R. Chess, 1978)
Pomacentridae	*Chromis margaritifer*	Planktivore/Herbivore	Feed on zooplankton such as copepods and larvae and on algae. (Hobson, E.S. and J.R. Chess, 1978)
Pomacentridae	*Neoglyphidodon melas*	Planktivore/Invertivore/Herbivore	Feed on zooplankton, benthic invertebrates and algae (Masuda, H. and G.R. Allen, 1993; Sano, M., M. Shimizu and Y. Nose, 1984)
Pomacentridae	*Pomacentrus chrysurus*	Herbivore/Invertivore	Feed primarily on algae (Allen, G.R., 1991) and copepods (Nakamura, Y., M. Horinouchi, T. Nakai and M. Sano, 2003)
Pomacentridae	*Pomacentrus amboinensis*	Herbivore/Planktivore	Feed primarily on algae, but also takes zooplankton. (Masuda, H. and G.R. Allen, 1993)
Pomacentridae	*Acanthochromis polyacanthus*	Herbivore/Planktivore	Feeds on algae and zooplankton (Masuda, H. and G.R. Allen, 1993)
Pomacentridae	*Dascyllus reticulatus*	Invertivore/Herbivore	Feeds on copepods, mysids, larvae and algae (Hobson, E.S. and J.R. Chess, 1978)
Pomacentridae	*Neopomacentrus cyanomos*	Planktivore	No info
Pomacentridae	*Pomacentrus adelus*	Herbivore	No info
Pomacentridae	*Chromis viridis*	Planktivore	Phytoplankton feeders and males feed on their own unhatched eggs (Hobson, E.S. and J.R. Chess, 1978; Allen, G.R., 1991

FAMILY	**SCIENTIFIC_NAME**	**FUNCTIONAL_GROUP**	**FISHBASE_DIET**
Pomacentridae	*Neoglyphidodon nigroris*	Planktivore/Invertivore/Herbivore	zoobenthos , zooplankton and unspecified algae - http://www.fishbase.org/TrophicEco/FoodItemsList.php?vstockcode=6001&genus=Neoglyphidodon&species=nigroris
Pomacentridae	*Dascyllus trimaculatus*	Planktivore/Herbivore	zooplankton and filamentous algae (http://www.fishbase.org/TrophicEco/FoodItemsList.php?vstockcode=5347&genus=Dascyllus&species=trimaculatus)
Pomacentridae	*Dascyllus aruanus*	Herbivore/Planktivore	zooplankton and plants benthic algae/weeds (http://www.fishbase.org/TrophicEco/FoodItemsList.php?vstockcode=5345&genus=Dascyllus&species=aruanus)
Pomacentridae	*Abudefduf vaigiensis*	Planktivore/Invertivore/Herbivore	Zooplanlkton, Zoobenthos, finfish(?), benthic algae (http://www.fishbase.org/TrophicEco/FoodItemsList.php?vstockcode=6950&genus=Abudefduf&species=vaigiensis)
Scaridae	*Scarus chameleon*	Roving_herbivore	benthic algae/weeds - http://www.fishbase.org/TrophicEco/FoodItemsList.php?vstockcode=5833&genus=Scarus&species=chameleon
Scaridae	*Scarus ghobban*	Herbivore	Feed by scraping algae from rocks and corals (Humann, P. and N. Deloach 1993).
Scaridae	*Chlorurus bleekeri*	Herbivore	Feed mainly on algae (Bellwood, D.R. and J.H. Choat, 1990)
Scaridae	*Scarus dimidiatus*	Herbivore	Feeds mainly on algae (Bellwood, D.R. and J.H. Choat, 1990)
Scaridae	*Scarus flavipectoralis*	Herbivore	Feeds on algae (Bellwood, D.R. and J.H. Choat, 1990)
Scaridae	*Hipposcarus longiceps*	Roving_herbivore	Feeds on benthic algae (Bacchet, P. et al. 2006).
Scaridae	*Scarus schlegeli*	Roving_herbivore	Feeds on benthic algae (Bacchet, P. et al. 2006).

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Scaridae	*Cetoscarus ocellatus*	Herbivore	Feeds on benthic algae (Bacchet, P., T. Zysman and Y. Lefevre, 2006)
Scaridae	*Scarus forsteni*	Herbivore	Feeds on benthic algae (Bacchet, P., T. Zysman and Y. Lefevre, 2006)
Scaridae	*Scarus globiceps*	Herbivore	Feeds on benthic algae (Bacchet, P., T. Zysman and Y. Lefevre, 2006)
Scaridae	*Chlorurus microrhinos*	Roving_herbivore	Feeds on benthic algae (Bacchet, P., T. Zysman and Y. Lefevre, 2006).
Scaridae	*Scarus rubroviolaceus*	Roving_herbivore	Feeds on benthic algae (Sommer et al. 1996).
Scaridae	*Scarus niger*	Herbivore	Feeds on benthic algae (Sommer, C., W. Schneider and J.-M. Poutiers, 1996)
Scaridae	*Scarus psittacus*	Roving_herbivore	Graze on benthic algae (Bruce, R.W. and J.E. Randall 1984).
Scaridae	*Scarus frenatus*	Herbivore	Grazes on benthic algae (Sommer et al. 1996).
Serranidae	*Plectropomus leopardus*	Piscivore	Adults feed mainly on fish (Kailola, P.J. et al. 1993). Juveniles feed on small fish and invertebrates such as crustaceans and squid (Kingsford, M. 1992).
Serranidae	*Epinephelus fasciatus*	Piscivore/invertivore	At Madagascar it feeds night and day on brachyuran crabs, fishes, shrimps, and galatheid crabs (Harmelin-Vivien, M.L. and C. Bouchon 1976). In Kenyan waters it feeds on crabs, stomatopods, fishes, ophiuroids, and octopus (Morgans, J.F.C. 1982). In the Red Sea, mostly fishes and some crustaceans (mainly crabs) are consumed (Randall, J.E. and A. Ben-Tuvia 1983).
Serranidae	*Epinephelus merra*	Piscivore/invertivore	Feed on crustaceans and fishes. Increase of piscivory with age is observed in this species.
Serranidae	*Cephalopholis miniata*	Piscivore/invertivore	Feed on fishes (80%, mainly Pseudanthias squamipinnis) and crustaceans.
Serranidae	*Epinephelus coioides*	Piscivore/invertivore	Feed on small fishes, shrimps, and crabs.

FAMILY	**SCIENTIFIC_NAME**	**FUNCTIONAL_GROUP**	**FISHBASE_DIET**
Serranidae	*Cephalopholis argus*	Piscivore/invertivore	Feeds mainly on fishes (75-95%) and to a lesser extent on crustaceans and polychaetes. (Randall, J.E. and V.E. Brock, 1960; Masuda, H. and G.R. Allen, 1993)
Serranidae	*Epinephelus hexagonatus*	Piscivore/Invertivore	Feeds mainly on fishes and crustaceans (stomatopods and brachyuran crabs) (Heemstra, P.C. and J.E. Randall, 1993)
Serranidae	*Variola louti*	Piscivore/invertivore	Feeds mainly on fishes, and on crabs, shrimps and stomatopods.
Serranidae	*Epinephelus maculatus*	Piscivore/invertivore	Feeds mainly on sand-dwelling fishes and crustaceans, sometimes on octopuses.
Serranidae	*Aethaloperca rogaa*	Piscivore/Invertivore	Feeds mainly on small fishes (including Pempheris sp.), also on stomatopods (Pseudosquilla sp.) (Morgans, J.F.C., 1982)and crustaceans (Myers, R.F., 1999)
Serranidae	*Epinephelus macrospilos*	Piscivore/Invertivore	Feeds on crustaceans (mainly crabs), fishes, octopi, and squid. (Hiatt, R.W. and D.W. Strasburg, 1960; Heemstra, P.C. and J.E. Randall, 1993)
Serranidae	*Cephalopholis cyanostigma*	Piscivore/invertivore	Feeds on crustaceans and fishes (Myers, R.F. 1999).
Serranidae	*Epinephelus fuscoguttatus*	Piscivore/invertivore	Feeds on fishes, crabs, and cephalopods.
Serranidae	*Variola albimarginata*	Piscivore	Feeds on fishes. (Heemstra, P.C. and J.E. Randall, 1993)
Siganidae	*Siganus vulpinus*	Herbivore	Feed on algae growing on the dead bases of Acropora branches. (Baensch, H.A. and H. Debelius, 1997)
Siganidae	*Siganus doliatus*	Roving_herbivore	feed on benthic seaweeds (Woodland, D. 1997; Kuiter, R. H. and T. Tonozuka, 2001c).
Siganidae	*Siganus argenteus*	Herbivore	Feeds on algae (Sommer, C., W. Schneider and J.-M. Poutiers, 1996)
Siganidae	*Siganus punctatus*	Herbivore	Feeds on benthic algae (Woodland, D., 1997)

FAMILY	SCIENTIFIC_NAME	FUNCTIONAL_GROUP	FISHBASE_DIET
Siganidae	*Siganus punctatissimus*	Herbivore	Feeds on benthic seaweeds (Woodland, D., 1997)
Tetraodontidae	*Arothron hispidus*	Herbivore/Invertivore	Feed on fleshy, calcareous, or coralline algae, detritus, molluscs, tunicates, sponges, corals, zoanthid anemones, crabs, tube worms and echinoderms (Myers, R.F., 1991)
Zanclidae	*Zanclus cornutus*	Invertivore	Feed on small encrusting animals (Anderson, C. and A. Hafiz, 1987)

SI Table 11. Abundance of individual fish species observed in UVC survey in the six reef zones at Ashmore Reef in 2019. Species ranked in order from highest to lowest overall abundance. B- additional species sighted on West Lagoon Bommies from cameras and/or timed swims

Species	Channel	East lagoon	Reef flat	Reef slope	Sand flat	West lagoon
Pomacentrus coelestis	426	20	189	726	19	65
Ctenochaetus striatus	197	4	226	706	5	288
Lutjanus gibbus	256	33	138	662	74	133
Halichoeres trimaculatus	57	38	784	6	267	9
Pomacentrus pavo	457	243	47	0	60	217
Pomacentrus lepidogenys	140	0	9	702	0	165
Pomacentrus adelus	39	0	657	53	37	100
Chrysiptera biocellata	15	9	607	0	163	9
Acanthurus nigrofuscus	90	0	87	382	0	56
Chromis weberi	252	0	20	186	0	49
Acanthurus triostegus	8	0	449	30	4	0
Chromis viridis	24	0	39	119	0	297
Acanthurus blochii	86	4	106	152	5	91
Pomacentrus alexanderae	2	20	9	8	0	318
Chlorurus sordidus	42	0	42	92	0	157
Lutjanus decussatus	21	0	174	90	0	37
Dascyllus aruanus	61	25	154	15	2	50
Chromis ternatensis	10	0	0	163	0	114
Pomacentrus bankanensis	42	2	68	159	0	10
Chrysiptera brownriggii	23	0	228	0	19	0
Chromis margaritifer	14	0	0	239	0	3
Pomacentrus amboinensis	88	1	0	110	0	36
Thalassoma lunare	56	3	36	118	1	11
Zebrasoma scopas	0	0	2	129	0	76
Chrysiptera rex	39	0	0	141	0	12
Chrysiptera hemicyanea	0	0	6	108	0	62
Halichoeres nebulosus	45	10	56	30	18	17
Dascyllus reticulatus	89	0	28	18	0	39
Amblyglyphidodon batunai	4	0	4	7	0	149
Pomacentrus chrysurus	13	0	99	29	0	15

Species	Channel	East lagoon	Reef flat	Reef slope	Sand flat	West lagoon
Amblyglyphidodon curacao	11	0	4	2	0	131
Neoglyphidodon oxyodon	0	0	135	0	0	0
Acanthurus lineatus	8	0	2	117	0	7
Cephalopholis argus	10	0	35	76	0	11
Halichoeres melanurus	15	0	45	54	3	13
Melichthys niger	2	0	0	125	0	0
Pomacentrus philippinus	36	0	3	85	0	1
Lutjanus bohar	13	0	3	101	0	6
Hipposcarus longiceps	5	0	0	93	0	24
Chromis lepidolepis	94	0	0	0	0	26
Pterocaesio trilineata	0	0	0	0	0	116
Caesio cuning	0	0	0	87	0	20
Naso unicornis	15	0	6	85	1	0
Monotaxis grandoculis	21	0	0	77	0	7
Abudefduf vaigiensis	0	0	21	82	0	0
Lutjanus fulvus	0	22	1	75	0	1
Pterocaesio tile	0	0	0	98	0	0
Rhinecanthus aculeatus	9	11	59	0	18	0
Dascyllus trimaculatus	65	9	11	6	0	5
Acanthochromis polyacanthus	0	0	6	40	0	48
Pomacentrus vaiuli	22	0	0	52	0	19
Plectroglyphidodon lacrymatus	17	0	0	63	0	12
Acanthurus nigricans	24	0	1	65	0	0
Dischistodus pseudochrysopoecilus	1	0	74	0	6	0
Caranx melampygus	7	0	3	67	0	3
Acanthurus olivaceus	34	0	5	37	0	0
Dischistodus perspicillatus	7	29	32	0	0	5
Scarus psittacus	0	0	35	34	1	2
Chaetodon auriga	10	0	30	27	0	4
Chaetodon lunulatus	4	0	11	35	0	20
Sufflamen chrysopterum	27	3	2	30	3	4

Species	**Channel**	**East lagoon**	**Reef flat**	**Reef slope**	**Sand flat**	**West lagoon**
Thalassoma quinquevittatum	27	0	1	38	0	0
Naso lituratus	8	0	1	49	0	7
Chromis cinerascens	0	0	0	0	0	64
Scarus niger	8	0	30	18	0	7
Centropyge vrolikii	21	0	2	27	0	11
Gomphosus varius	9	0	2	48	0	2
Cirrhilabrus exquisitus	0	0	0	58	0	0
Labroides dimidiatus	13	5	5	26	0	8
Naso brevirostris	2	0	0	50	0	0
Plectroglyphidodon dickii	12	0	0	36	0	2
Scolopsis lineata	0	2	44	0	2	0
Lutjanus fulviflamma	13	1	1	6	25	0
Chaetodon ephippium	3	0	14	20	0	9
Dischistodus melanotus	0	0	42	0	1	3
Stethojulis bandanensis	2	0	26	7	6	4
Scolopsis trilineata	3	0	31	4	3	2
Chaetodon citrinellus	4	0	8	28	0	2
Pomacentrus moluccensis	6	3	4	7	0	22
Halichoeres hortulanus	3	0	7	30	2	0
Parupeneus barberinus	2	3	18	9	4	4
Acanthurus mata	0	0	0	39	0	1
Balistapus undulatus	6	0	1	27	0	6
Chrysiptera glauca	0	0	34	0	5	0
Thalassoma hardwicke	4	0	9	24	0	2
Coris batuensis	16	0	3	14	3	2
Halichoeres marginatus	5	0	22	8	2	1
Thalassoma amblycephalum	17	0	2	13	0	6
Naso brachycentron	8	0	0	29	0	0
Zanclus cornutus	3	0	3	22	0	8
Neoglyphidodon melas	0	0	14	17	0	4
Chaetodon vagabundus	6	0	7	19	0	1
Caesio lunaris	0	0	0	30	0	2

Species	**Channel**	**East lagoon**	**Reef flat**	**Reef slope**	**Sand flat**	**West lagoon**
Chaetodon kleinii	2	0	2	23	1	4
Halichoeres margaritaceus	1	0	20	4	5	2
Naso caesius	0	0	0	32	0	0
Macolor macularis	6	0	0	16	0	8
Naso tuberosus	2	0	0	28	0	0
Gnathodentex aureolineatus	0	0	0	28	0	0
Hemigymnus melapterus	0	0	11	9	0	8
Variola louti	4	0	0	21	0	2
Thalassoma jansenii	2	0	6	17	0	1
Acanthurus xanthopterus	4	0	11	11	0	0
Pseudanthias tuka	0	0	0	25	0	0
Pomacanthus imperator	1	0	0	22	0	2
Naso hexacanthus	0	0	1	18	0	6
Siganus argenteus	4	0	0	20	0	0
Lethrinus olivaceus	6	0	0	12	0	5
Parupeneus pleurostigma	1	3	8	5	4	2
Ctenochaetus binotatus	0	0	0	20	0	3
Balistoides viridescens	1	0	4	16	0	2
Parupeneus multifasciatus	4	0	5	5	0	8
Chaetodon melannotus	0	0	3	5	0	14
Pseudocheilinus hexataenia	2	0	0	14	0	6
Scarus globiceps	0	0	1	5	1	15
Aethaloperca rogaa	4	0	0	17	0	0
Cephalopholis miniata	2	0	1	15	0	3
Mulloidichthys flavolineatus	2	2	9	0	5	3
Acanthurus nigricauda	6	0	2	12	0	1
Melichthys vidua	2	0	0	19	0	0
Centropyge bicolor	7	0	0	13	0	0
Caesio teres	0	0	0	10	0	8
Forcipiger flavissimus	2	0	0	16	0	0
Neoglyphidodon nigroris	5	0	0	7	0	6
Pomacentrus tripunctatus	2	1	1	0	0	14

Species	**Channel**	**East lagoon**	**Reef flat**	**Reef slope**	**Sand flat**	**West lagoon**
Acanthurus pyroferus	1	0	0	12	0	5
Naso annulatus	0	0	0	18	0	0
Chaetodon ulietensis	1	0	0	6	0	9
Plectorhinchus lineatus	0	0	0	13	0	2
Parupeneus indicus	3	0	6	5	0	1
Stegastes fasciolatus	1	0	0	12	0	2
Cirrhilabrus randalli	0	0	0	0	0	15
Chlorurus microrhinos	2	0	0	11	0	2
Carcharhinus amblyrhynchos	0	0	1	13	0	0
Plectropomus laevis	2	0	1	10	0	1
Scomberoides lysan	0	0	0	14	0	0
Scolopsis bilineata	2	0	6	2	0	4
Chaetodon ornatissimus	2	0	0	10	0	2
Chaetodon rafflesii	3	0	5	4	0	2
Cheilinus fasciatus	0	0	1	7	0	6
Epinephelus merra	0	0	9	0	0	4
Siganus puellus	0	0	1	0	0	12
Pygoplites diacanthus	0	0	0	10	1	1
Abudefduf sexfasciatus	11	0	0	0	0	1
Chromis amboinensis	10	0	0	0	0	2
Anampses melanurus	0	0	0	12	0	0
Scarus rubroviolaceus	1	0	0	11	0	0
Ctenochaetus cyanocheilus	8	0	0	1	0	3
Zebrasoma veliferum	4	0	0	3	0	5
Sufflamen bursa	0	0	1	9	2	0
Parupeneus cyclostomus	5	0	0	4	0	2
Plectroglyphidodon johnstonianus	0	0	0	11	0	0
Hemigymnus fasciatus	1	0	1	8	0	1
Siganus doliatus	4	0	2	2	0	3
Siganus vulpinus	2	0	0	2	0	7
Balistoides conspicillum	0	0	0	11	0	0
Cephalopholis urodeta	1	0	0	9	0	0

Species	**Channel**	**East lagoon**	**Reef flat**	**Reef slope**	**Sand flat**	**West lagoon**
Epinephelus hexagonatus	1	0	8	0	1	0
Pterocaesio digramma	0	0	0	10	0	0
Macolor niger	4	0	0	6	0	0
Parupeneus barberinoides	0	5	0	0	0	5
Kyphosus vaigiensis	0	0	0	10	0	0
Chaetodon baronessa	4	0	2	3	0	1
Chaetodon trifascialis	0	0	1	9	0	0
Bodianus axillaris	0	0	0	8	0	2
Cheilinus chlorourus	0	0	7	2	0	1
Stethojulis strigiventer	1	0	9	0	0	0
Cetoscarus ocellatus	4	0	0	6	0	0
Acanthurus nigros	0	0	0	10	0	0
Chaetodon meyeri	6	0	0	2	0	1
Neopomacentrus azysron	9	0	0	0	0	0
Scarus flavipectoralis	0	0	0	0	1	8
Scarus frenatus	1	0	0	7	1	0
Parupeneus trifasciatus	3	0	1	4	0	0
Heniochus varius	0	0	0	4	0	4
Halichoeres chrysus	8	0	0	0	0	0
Rhinecanthus rectangulus	0	0	8	0	0	0
Epinephelus maculatus	4	0	0	0	0	3
Scolopsis margaritifera	1	0	1	0	0	5
Pomacanthus sexstriatus	2	0	0	4	0	1
Leptojulis cyanopleura	2	0	0	5	0	0
Chlorurus bleekeri	0	0	0	0	0	7
Naso fageni	0	0	0	7	0	0
Lutjanus kasmira	6	0	0	0	0	0
Chaetodon adiergastos	0	0	0	6	0	0
Chaetodon lunula	0	0	1	3	0	2
Chaetodon semeion	0	0	0	6	0	0
Centropyge tibicen	3	0	1	1	0	1
Macropharyngodon meleagris	1	0	1	4	0	0

Species	Channel	East lagoon	Reef flat	Reef slope	Sand flat	West lagoon
Novaculichthys taeniourus	0	0	5	0	1	0
Scarus dimidiatus	0	0	0	3	0	3
Paracanthurus hepatus	3	0	0	3	0	0
Pseudobalistes flavimarginatus	1	1	1	0	3	0
Plectropomus leopardus	3	0	0	2	0	0
Variola albimarginata	0	0	0	5	0	0
Lutjanus rivulatus	1	0	0	4	0	0
Chaetodon lineolatus	0	0	0	4	0	1
Heniochus acuminatus	3	0	0	2	0	0
Chaetodontoplus mesoleucus	0	0	0	0	0	5
Stegastes nigricans	0	0	0	0	0	5
Plectorhinchus lessonii	0	0	0	4	0	0
Monotaxis heterodon	0	0	0	1	0	3
Platax orbicularis	0	0	3	1	0	0
Chaetodon pelewensis	2	0	0	2	0	0
Amphiprion ocellaris	0	0	4	0	0	0
Dischistodus prosopotaenia	0	0	1	0	0	3
Anampses caeruleopunctatus	0	0	0	4	0	0
Labroides bicolor	3	0	0	0	0	1
Labropsis australis	0	0	0	1	0	3
Acanthurus thompsoni	0	0	0	4	0	0
Siganus punctatissimus	2	0	0	2	0	0
Rhinecanthus verrucosus	0	0	2	2	0	0
Triaenodon obesus	0	0	0	3	0	0
Pastinachus ater	0	0	0	0	3	0
Pateobatis fai	1	0	0	2	0	0
Cephalopholis leopardus	0	0	0	0	0	3
Caranx ignobilis	0	1	0	2	0	0
Aprion virescens	0	0	0	3	0	0
Lutjanus monostigma	0	0	2	0	0	1
Lethrinus obsoletus	1	0	0	1	0	1
Chaetodon ocellicaudus	2	0	1	0	0	0

Species	Channel	East lagoon	Reef flat	Reef slope	Sand flat	West lagoon
Chaetodon speculum	1	0	0	2	0	0
Amphiprion clarkii	0	0	0	3	0	0
Neopomacentrus cyanomos	0	0	0	3	0	0
Coris gaimard	1	0	0	2	0	0
Epibulus insidiator	1	0	0	0	0	2
Scarus chameleon	0	0	0	3	0	0
Scarus schlegeli	0	0	0	1	0	2
Acanthurus leucocheilus	0	0	0	3	0	0
Naso vlamingii	2	0	0	1	0	0
Pterois antennata	0	0	0	0	2	0
Epinephelus fasciatus	0	0	0	1	0	1
Pentapodus emeryii	0	0	0	0	0	2
Lethrinus harak	0	0	2	0	0	0
Chaetodon punctatofasciatus	0	0	0	2	0	0
Coradion chrysozonus	0	0	0	2	0	0
Heniochus chrysostomus	0	0	0	2	0	0
Amphiprion sandaracinos	0	0	0	2	0	0
Hemiglyphidodon plagiometopon	0	0	2	0	0	0
Bodianus mesothorax	0	0	0	2	0	0
Coris aygula	0	0	0	2	0	0
Oxycheilinus unifasciatus	0	0	0	2	0	0
Siganus corallinus	0	0	0	0	0	2
Siganus punctatus	0	0	0	2	0	0
Galeocerdo cuvier	0	0	0	0	1	0
Taeniura lymma	0	0	0	0	0	1
Gymnothorax javanicus	0	0	0	1	0	0
Cephalopholis cyanostigma	1	0	0	0	0	0
Epinephelus coioides	0	0	0	1	0	0
Epinephelus fuscoguttatus	0	0	0	1	0	0
Epinephelus macrospilos	0	0	0	0	0	1
Lethrinus atkinsoni	0	0	0	1	0	0
Lethrinus erythropterus	0	0	1	0	0	0

Species	**Channel**	**East lagoon**	**Reef flat**	**Reef slope**	**Sand flat**	**West lagoon**
Lethrinus rubrioperculatus	0	0	0	1	0	0
Lethrinus xanthochilus	0	0	0	1	0	0
Chaetodon unimaculatus	0	0	0	0	0	1
Apolemichthys trimaculatus	0	0	0	1	0	0
Amblyglyphidodon leucogaster	0	0	0	0	0	1
Chromis xanthura	0	0	0	1	0	0
Chrysiptera talboti	0	0	0	0	0	1
Cheilinus trilobatus	0	0	0	0	0	1
Cirrhilabrus temminckii	0	0	0	0	0	1
Coris pictoides	0	0	0	0	1	0
Hologymnosus doliatus	0	0	0	1	0	0
Labrichthys unilineatus	0	0	0	1	0	0
Macropharyngodon negrosensis	0	0	0	1	0	0
Macropharyngodon ornatus	0	0	0	1	0	0
Scarus forsteni	0	0	0	1	0	0
Scarus ghobban	0	0	0	0	0	1
Aluterus scriptus	0	0	0	0	1	0
Cantherhines dumerilii	0	0	0	1	0	0
Arothron hispidus	0	0	0	0	1	0
Nebrius ferrugineus	0	0	0	0	0	B
Stegostoma tigrinum	0	0	0	0	0	B
Carcharhinus melanopterus	0	0	0	0	0	B
Saurida gracilis	0	0	0	0	0	B
Synodus variegatus	0	0	0	0	0	B
Myripristis adusta	0	0	0	0	0	B
Myripristis berndti	0	0	0	0	0	B
Myripristis kuntee	0	0	0	0	0	B
Neoniphon sammara	0	0	0	0	0	B
Sargocentron caudimaculatum	0	0	0	0	0	B
Aulostomus chinensis	0	0	0	0	0	B

Species	**Channel**	**East lagoon**	**Reef flat**	**Reef slope**	**Sand flat**	**West lagoon**
Fistularia commersonii	0	0	0	0	0	B
Anyperodon leucogrammicus	0	0	0	0	0	B
Epinephelus multinotatus	0	0	0	0	0	B
Epinephelus polyphekadion	0	0	0	0	0	B
Epinephelus spilotoceps	0	0	0	0	0	B
Pseudanthias huchtii	0	0	0	0	0	B
Pseudochromis fuscus	0	0	0	0	0	B
Priacanthus hamrur	0	0	0	0	0	B
Cheilodipterus artus	0	0	0	0	0	B
Cheilodipterus macrodon	0	0	0	0	0	B
***Cheilodipterus* spp.**	0	0	0	0	0	B
Ostorhinchus compressus	0	0	0	0	0	B
Ostorhinchus cyanosoma	0	0	0	0	0	B
Ostorhinchus sealei	0	0	0	0	0	B
Ostorhinchus wassinki	0	0	0	0	0	B
Pristiapogon exostigma	0	0	0	0	0	B
Taeniamia zosterophora	0	0	0	0	0	B
Zoramia leptacantha	0	0	0	0	0	B
Malacanthus latovittatus	0	0	0	0	0	B
Echeneis naucrates	0	0	0	0	0	B
Caranx lugubris	0	0	0	0	0	B
Symphorichthys spilurus	0	0	0	0	0	B
Scolopsis monogramma	0	0	0	0	0	B
***Diagramma* spp.**	0	0	0	0	0	B
Plectorhinchus chaetodonoides	0	0	0	0	0	B
Lethrinus erythracanthus	0	0	0	0	0	B
Lethrinus nebulosus	0	0	0	0	0	B
Mulloidichthys vanicolensis	0	0	0	0	0	B
Platax teira	0	0	0	0	0	B
Chaetodon bennetti	0	0	0	0	0	B
Chaetodon decussatus	0	0	0	0	0	B
Centropyge eibli	0	0	0	0	0	B

Species	Channel	East lagoon	Reef flat	Reef slope	Sand flat	West lagoon
Centropyge flavissima	0	0	0	0	0	B
Chromis atripectoralis	0	0	0	0	0	B
Chromis atripes	0	0	0	0	0	B
Chromis xanthochira	0	0	0	0	0	B
Dischistodus chrysopoecilus	0	0	0	0	0	B
Stegastes albifasciatus	0	0	0	0	0	B
Cirrhitichthys falco	0	0	0	0	0	B
Paracirrhites forsteri	0	0	0	0	0	B
Sphyraena barracuda	0	0	0	0	0	B
Cheilinus undulatus	0	0	0	0	0	B
Halichoeres claudia	0	0	0	0	0	B
Halichoeres prosopeion	0	0	0	0	0	B
Labroides pectoralis	0	0	0	0	0	B
Oxycheilinus digrammus	0	0	0	0	0	B
Oxycheilinus orientalis	0	0	0	0	0	B
Pseudocheilinus evanidus	0	0	0	0	0	B
Bolbometopon muricatum	0	0	0	0	0	B
Scarus oviceps	0	0	0	0	0	B
Parapercis clathrata	0	0	0	0	0	B
Parapercis millepunctata	0	0	0	0	0	B
Parapercis pacifica	0	0	0	0	0	B
Atrosalarias fuscus	0	0	0	0	0	B
Meiacanthus grammistes	0	0	0	0	0	B
Plagiotremus tapeinosoma	0	0	0	0	0	B
Salarias fasciatus	0	0	0	0	0	B
Amblyeleotris steinitzi	0	0	0	0	0	B
Amblyeleotris wheeleri	0	0	0	0	0	B
Amblygobius nocturnus	0	0	0	0	0	B
Amblygobius phalaena	0	0	0	0	0	B
Amblygobius rainfordi	0	0	0	0	0	B
Ctenogobiops mitodes	0	0	0	0	0	B
Ctenogobiops pomastictus	0	0	0	0	0	B

Species	Channel	East lagoon	Reef flat	Reef slope	Sand flat	West lagoon
Gnatholepis cauerensis	0	0	0	0	0	B
Signigobius biocellatus	0	0	0	0	0	B
Valenciennea sexguttata	0	0	0	0	0	B
Valenciennea strigata	0	0	0	0	0	B
Ptereleotris evides	0	0	0	0	0	B
Ptereleotris heteroptera	0	0	0	0	0	B
Ptereleotris zebra	0	0	0	0	0	B
Acanthurus dussumieri	0	0	0	0	0	B
Naso tonganus	0	0	0	0	0	B
Gymnosarda unicolor	0	0	0	0	0	B
Pseudobalistes fuscus	0	0	0	0	0	B
Oxymonacanthus longirostris	0	0	0	0	0	B
Ostracion cubicus	0	0	0	0	0	B
Arothron stellatus	0	0	0	0	0	B
Canthigaster amboinensis	0	0	0	0	0	B
Canthigaster bennetti	0	0	0	0	0	B
Canthigaster valentini	0	0	0	0	0	B
Grand Total	3478	513	5302	8412	794	3552

SI Table 12. A Permutational MANOVA (PERMANOVA) was conducted to determine whether differences in fish communities between reef zones was statistically significant. There was a significant difference between reef zones, with pair-wise tests indicating fish densities were significantly different between all zones

Main-test

Source	df	SS	Pseudo-F	P(perm)	Unique perms
Reef Zones	5	1.502E+05	10.999	0.001	998
Residual	155	4.2393E+05			
TOTAL	160	5.7434+05			

Pair-wise test

Source	t	P(perm)	Unique perms
Reef flat, Sand flat	3.0229	0.001	999
Reef flat, East lagoon	3.1302	0.001	998
Reef flat, Channel	2.9292	0.001	999
Reef flat, West lagoon	3.0145	0.001	998
Reef flat, Reef slope	5.0324	0.001	999
Sand flat, East lagoon	1.8542	0.003	997
Sand flat, Channel	2.9214	0.001	999
Sand flat, West lagoon	3.2013	0.001	999
Sand flat, Reef slope	4.7016	0.001	996
East lagoon, Channel	2.4038	0.001	999
East lagoon, West lagoon	2.746	0.001	999
East lagoon, Reef slope	3.898	0.001	998
Channel, West lagoon	1.6939	0.001	998
Channel, Reef slope	2.102	0.001	997
West lagoon, Reef slope	2.0475	0.001	998

Table 13. Showing Manta Tow conditions, locations and time

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
14/06/2019	SS20-SS19	1	1	0	Calm	Blue tender		-12.213197	122.996077	15:49	-12.212813	122.997008	15:52
14/06/2019	SS20-SS19	2	1	0	Calm	Blue tender		-12.212733	122.997084	15:52	-12.212731	122.996832	15:54
14/06/2019	SS20-SS19	3	1	0	Calm	Blue tender		-12.212367	122.997693	15:55	-12.212083	122.997916	15:57
14/06/2019	SS20-SS19	4	1	0	Calm	Blue tender		-12.211857	122.998735	15:58	-12.21175	122.999215	16:00
14/06/2019	SS20-SS19	5	1	0	Calm	Blue tender		-12.211439	122.999903	16:01	-12.211342	123.000465	16:03
14/06/2019	SS20-SS19	6	1	0	Calm	Blue tender		-12.211323	123.000595	16:03	-12.21103	123.001466	16:05
14/06/2019	SS20-SS19	7	1	0	Calm	Blue tender		-12.211018	123.002126	16:06	-12.211042	123.003061	16:08
14/06/2019	SS20-SS19	8	1	0	Calm	Blue tender		-12.211051	123.003763	16:09	-12.210984	123.004516	16:11
14/06/2019	SS20-SS19	9	1	0	Calm	Blue tender		-12.21099	123.004615	16:11	-12.210895	123.00544	16:13
14/06/2019	SS20-SS19	10	1	0	Calm	Blue tender		-12.210953	123.006255	16:14	-12.211314	123.006867	16:16
14/06/2019	SS20-SS19	11	1	0	Calm	Blue tender		-12.211923	123.007509	16:17	-12.212814	123.00756	16:19
14/06/2019	SS20-SS19	12	1	0	Calm	Blue tender		-12.213954	123.00735	16:20	-12.21465	123.006997	16:22
14/06/2019	SS20-SS19	13	1	0	Calm	Blue tender		-12.21501	123.007622	16:23	-12.215316	123.008185	16:25
15/06/2019	SS15-SS16	1	1	0	Calm	Blue tender		-12.176186	123.053932	14:50	-12.176767	123.054092	14:52
15/06/2019	SS15-SS16	2	1	0	Calm	Blue tender		-12.177315	123.053555	14:53	-12.17789	123.052847	14:55
15/06/2019	SS15-SS16	3	1	0	Calm	Blue tender		-12.177971	123.052741	14:55	-12.178345	123.052343	14:56
15/06/2019	SS15-SS16	4	1	0	Calm	Blue tender		-12.178349	123.05219	14:56	-12.178594	123.05154	14:58
15/06/2019	SS15-SS16	5	1	0	Calm	Blue tender		-12.178723	123.051527	14:58	-12.179149	123.050807	15:00
15/06/2019	SS15-SS16	6	1	0	Calm	Blue tender		-12.179587	123.050125	15:01	-12.18013	123.049529	15:03
15/06/2019	SS15-SS16	7	1	0	Calm	Blue tender		-12.180238	123.049473	15:03	-12.180789	123.048753	15:05
15/06/2019	SS15-SS16	8	1	0	Calm	Blue tender		-12.181381	123.048154	15:06	-12.181919	123.047553	15:08
15/06/2019	SS15-SS16	9	1	0	Calm	Blue tender		-12.182402	123.047164	15:09	-12.182732	123.046639	15:11
15/06/2019	SS15-SS16	10	1	0	Calm	Blue tender		-12.182858	123.046583	15:11	-12.183438	123.045526	15:13
15/06/2019	SS15-SS16	11	1	0	Calm	Blue tender		-12.184067	123.045059	15:14	-12.18465	123.04457	15:16
15/06/2019	SS15-SS16	12	1	0	Calm	Blue tender		-12.185555	123.044407	15:17	-12.186414	123.044417	15:19

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
15/06/2019	SS17-SS18	1	1	0	Calm	Blue tender		-12.18989	123.037712	15:30	-12.190803	123.038264	15:33
15/06/2019	SS17-SS18	2	1	0	Calm	Blue tender		-12.191028	123.038459	15:33	-12.191918	123.036935	15:36
15/06/2019	SS17-SS18	3	1	0	Calm	Blue tender		-12.191968	123.036803	15:36	-12.192661	123.036216	15:38
15/06/2019	SS17-SS18	4	1	0	Calm	Blue tender		-12.193354	123.035672	15:39	-12.19396	123.035064	15:41
15/06/2019	SS17-SS18	5	1	0	Calm	Blue tender		-12.194071	123.034958	15:41	-12.194689	123.034441	15:43
15/06/2019	SS17-SS18	6	1	0	Calm	Blue tender		-12.195495	123.033797	15:44	-12.196032	123.033313	15:46
15/06/2019	SS17-SS18	7	1	0	Calm	Blue tender		-12.196143	123.033226	15:46	-12.19694	123.032798	15:48
15/06/2019	SS17-SS18	8	1	0	Calm	Blue tender		-12.197773	123.032416	15:49	-12.198323	123.031923	15:51
15/06/2019	SS17-SS18	9	1	0	Calm	Blue tender		-12.198857	123.031298	15:52	-12.19939	123.030622	15:54
15/06/2019	SS17-SS18	10	1	0	Calm	Blue tender		-12.199929	123.029936	15:55	-12.200264	123.029122	15:57
15/06/2019	SS17-SS18	11	1	0	Calm	Blue tender		-12.200686	123.028571	15:58	-12.200833	123.028163	15:59
15/06/2019	SS17-SS18	12	1	0	Calm	Blue tender		-12.201104	123.027496	16:00	-12.201244	123.026436	16:02
15/06/2019	SS17-SS18	13	1	0	Calm	Blue tender		-12.201616	123.025392	16:11	-12.201942	123.024706	16:13
15/06/2019	SS17-SS18	14	1	0	Calm	Blue tender		-12.202229	123.023982	16:14	-12.202563	123.023278	16:16
15/06/2019	SS17-SS18	15	1	0	Calm	Blue tender		-12.203005	123.022701	16:17	-12.203359	123.022043	16:19
15/06/2019	SS17-SS18	16	1	0	Calm	Blue tender		-12.203466	123.021892	16:19	-12.203927	123.021161	16:21
15/06/2019	SS17-SS18	17	1	0	Calm	Blue tender		-12.204594	123.019687	16:23	-12.204952	123.018857	16:25
15/06/2019	SS17-SS18	18	1	0	Calm	Blue tender		-12.205326	123.018046	16:26	-12.205591	123.017472	16:28
15/06/2019	SS17-SS18	19	1	0	Calm	Blue tender		-12.206001	123.016635	16:29	-12.206506	123.015585	16:31
15/06/2019	SS17-SS18	20	1	0	Calm	Blue tender		-12.207056	123.014928	16:32	-12.20771	123.014193	16:34
15/06/2019	SS17-SS18	21	1	0	Calm	Blue tender		-12.208616	123.01372	16:35	-12.209678	123.013689	16:37
16/06/2019	SS02-SS01	1	1	6	Small ripples	Blue tender	6.1	-12.257888	122.943758	14:24	-12.257615	122.943189	14:26
16/06/2019	SS02-SS01	2	1	6	Small ripples	Blue tender	8	-12.257309	122.942445	14:27	-12.257016	122.941788	14:29
16/06/2019	SS02-SS01	3	1	6	Small ripples	Blue tender	4.6	-12.25653	122.941024	14:30	-12.25599	122.940491	14:32
16/06/2019	SS02-SS01	4	1	6	Small ripples	Blue tender	6.4	-12.255556	122.939587	14:33	-12.255089	122.938841	14:35
16/06/2019	SS02-SS01	5	1	6	Small ripples	Blue tender	9	-12.254615	122.938124	14:36	-12.254169	122.937252	14:38
16/06/2019	SS02-SS01	6	1	6	Small ripples	Blue tender	5	-12.25361	122.936553	14:39	-12.253217	122.936073	14:41
16/06/2019	SS02-SS01	7	1	6	Small ripples	Blue tender	6	-12.252509	122.935541	14:42	-12.252075	122.934669	14:44

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
16/06/2019	SS02-SS01	8	1	6	Small ripples	Blue tender	8.2	-12.251486	122.934068	14:45	-12.251085	122.933404	14:47
16/06/2019	SS02-SS01	9	1	6	Small ripples	Blue tender	6	-12.250505	122.932908	14:48	-12.249875	122.932435	14:50
16/06/2019	SS02-SS01	10	1	6	Small ripples	Blue tender	8	-12.249458	122.93194	14:51	-12.248867	122.931533	14:53
16/06/2019	SS02-SS01	11	1	6	Small ripples	Blue tender	4.6	-12.248426	122.930917	14:54	-12.24771	122.930279	14:56
16/06/2019	SS02-SS01	12	1	6	Small ripples	Blue tender	5.6	-12.247593	122.930185	14:56	-12.246777	122.929404	14:58
16/06/2019	SS02-SS01	13	1	6	Small ripples	Blue tender	7.3	-12.246204	122.92891	14:59	-12.24547	122.928255	15:01
16/06/2019	SS02-SS01	14	1	6	Small ripples	Blue tender	6.1	-12.244674	122.92779	15:02	-12.243806	122.927216	15:04
16/06/2019	SS02-SS01	15	1	6	Small ripples	Blue tender	7.7	-12.242917	122.92678	15:05	-12.242175	122.926516	15:07
16/06/2019	SS02-SS01	16	1	6	Small ripples	Blue tender	10.6	-12.241318	122.926266	15:08	-12.240365	122.926286	15:10
16/06/2019	SS02-SS01	17	1	3	Small ripples	Blue tender	7.6	-12.236196	122.929219	15:20	-12.237103	122.928494	15:23
16/06/2019	SS02-SS01	18	1	3	Small ripples	Blue tender	5.2	-12.237105	122.928503	15:23	-12.236881	122.928999	15:26
16/06/2019	SS02-SS01	19	1	3	Small ripples	Blue tender	4.3	-12.236501	122.929589	15:27	-12.236364	122.929811	15:29
16/06/2019	SS02-SS01	20	1	3	Small ripples	Blue tender	6.1	-12.2363	122.929907	15:29	-12.235823	122.930275	15:31
16/06/2019	SS02-SS01	21	1	3	Small ripples	Blue tender	8.4	-12.235474	122.930853	15:32	-12.23535	122.931231	15:34
16/06/2019	SS02-SS01	22	1	3	Small ripples	Blue tender	4.9	-12.234872	122.931783	15:35	-12.234641	122.93223	15:37
16/06/2019	SS02-SS01	23	1	3	Small ripples	Blue tender	7.6	-12.234598	122.932319	15:37	-12.234362	122.932752	15:39
16/06/2019	SS02-SS01	24	1	3	Small ripples	Blue tender	6.2	-12.233944	122.933487	15:40	-12.233684	122.933963	15:42
16/06/2019	SS02-SS01	25	1	3	Small ripples	Blue tender	6.2	-12.233193	122.934782	15:44	-12.233239	122.934775	15:45
16/06/2019	SS02-SS01	26	1	3	Small ripples	Blue tender	5.8	-12.232982	122.935473	15:46	-12.232747	122.935859	15:48
16/06/2019	SS02-SS01	27	1	3	Small ripples	Blue tender	4	-12.232627	122.936443	15:49	-12.232471	122.937014	15:51
16/06/2019	SS02-SS01	28	1	3	Small ripples	Blue tender	3.9	-12.232257	122.937735	15:52	-12.232104	122.938283	15:54
16/06/2019	SS02-SS01	29	1	3	Small ripples	Blue tender	4.5	-12.232078	122.938405	15:54	-12.231666	122.93917	15:56
16/06/2019	SS02-SS01	30	1	3	Small ripples	Blue tender	6.9	-12.231503	122.939786	15:57	-12.230959	122.941608	16:01
16/06/2019	SS04-SS03	1	1	6	Small ripples	Blue tender		-12.266473	122.962846	13:53	-12.266339	122.962818	13:55
16/06/2019	SS04-SS03	2	1	6	Small ripples	Blue tender	6	-12.265969	122.962172	13:56	-12.265542	122.961365	13:58
16/06/2019	SS04-SS03	3	1	6	Small ripples	Blue tender	7.8	-12.265137	122.96037	13:59	-12.264513	122.958731	14:02
16/06/2019	SS04-SS03	4	1	6	Small ripples	Blue tender	7.3	-12.264469	122.958582	14:02	-12.264374	122.957812	14:04
16/06/2019	SS04-SS03	5	1	6	Small ripples	Blue tender	4.3	-12.264357	122.957686	14:04	-12.264042	122.956841	14:06

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
16/06/2019	SS04-SS03	6	1	6	Small ripples	Blue tender	6	-12.264004	122.956819	14:06	-12.263206	122.95536	14:09
16/06/2019	SS24	1	1	6	Small ripples	Blue tender		-12.278281	123.010289	11:56	-12.278038	123.010717	11:58
16/06/2019	SS24	2	1	6	Small ripples	Blue tender		-12.278696	123.011014	11:59	-12.278264	123.011388	12:01
16/06/2019	SS24	3	1	6	Small ripples	Blue tender		-12.278422	123.01206	12:02	-12.278488	123.012633	12:04
16/06/2019	SS24	4	1	6	Small ripples	Blue tender		-12.278421	123.012764	12:04	-12.277675	123.012837	12:06
16/06/2019	SS24	5	1	6	Small ripples	Blue tender		-12.277322	123.012704	12:07	-12.277522	123.012056	12:09
16/06/2019	SS24	6	1	6	Small ripples	Blue tender		-12.27668	123.011939	12:12	-12.276544	123.01166	12:14
16/06/2019	SS24	7	1	6	Small ripples	Blue tender		-12.276362	123.011025	12:15	-12.276322	123.010301	12:17
16/06/2019	SS24	8	1	6	Small ripples	Blue tender		-12.276541	123.010397	12:18	-12.276952	123.010867	12:20
17/06/2019	SS09-SS10	1	1	0	calm	Blue tender	3.6	-12.260263	123.149937	11:19	-12.260121	123.149434	11:21
17/06/2019	SS09-SS10	2	1	0	calm	Blue tender	5	-12.259979	123.149382	11:21	-12.258407	123.150337	11:24
17/06/2019	SS09-SS10	3	1	0	calm	Blue tender	3.1	-12.258361	123.150395	11:24	-12.257293	123.151145	11:26
17/06/2019	SS09-SS10	4	1	0	calm	Blue tender	4.5	-12.256594	123.151606	11:27	-12.25579	123.152318	11:29
17/06/2019	SS09-SS10	5	1	0	calm	Blue tender	4.5	-12.25515	123.152657	11:30	-12.254201	123.15328	11:32
17/06/2019	SS09-SS10	6	1	0	calm	Blue tender	5.8	-12.254116	123.153376	11:32	-12.253215	123.153824	11:34
17/06/2019	SS09-SS10	7	1	0	calm	Blue tender	6.8	-12.253152	123.153898	11:34	-12.251747	123.15513	11:37
17/06/2019	SS09-SS10	8	1	0	calm	Blue tender	6.2	-12.250931	123.155347	11:38	-12.25048	123.155487	11:39
17/06/2019	SS09-SS10	9	1	0	calm	Blue tender	3.9	-12.249777	123.155553	11:40	-12.249128	123.155997	11:42
17/06/2019	SS09-SS10	10	1	0	calm	Blue tender	5.1	-12.248299	123.156081	11:43	-12.24757	123.156587	11:45
17/06/2019	SS09-SS10	11	1	0	calm	Blue tender	6.7	-12.246261	123.157425	11:55	-12.245176	123.156917	11:57
17/06/2019	SS09-SS10	12	1	0	calm	Blue tender	6.5	-12.245029	123.156898	11:57	-12.244274	123.156813	11:59
17/06/2019	SS09-SS10	13	1	0	calm	Blue tender	3.9	-12.243448	123.156852	12:00	-12.242611	123.156098	12:02
17/06/2019	SS09-SS10	14	1	0	calm	Blue tender	4.6	-12.241973	123.1557	12:03	-12.241754	123.155317	12:05
17/06/2019	SS09-SS10	15	1	0	calm	Blue tender	3.6	-12.241103	123.154709	12:06	-12.240663	123.155385	12:08
17/06/2019	SS09-SS10	16	1	0	calm	Blue tender	8	-12.240234	123.15592	12:09	-12.239874	123.156309	12:11
17/06/2019	SS09-SS10	17	1	0	calm	Blue tender	9	-12.238664	123.156641	12:13	-12.237699	123.156437	12:15
17/06/2019	SS09-SS10	18	1	0	calm	Blue tender	5.9	-12.237017	123.156076	12:16	-12.235815	123.155935	12:18
17/06/2019	SS09-SS10	19	1	0	calm	Blue tender	6.6	-12.235654	123.155936	12:18	-12.234816	123.155764	12:20

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
17/06/2019	SS09-SS10	20	1	0	calm	Blue tender	8	-12.23409	123.155224	12:21	-12.233293	123.154924	12:23
17/06/2019	SS09-SS10	21	1	0	calm	Blue tender	8.3	-12.231467	123.15446	12:33	-12.231184	123.153743	12:35
17/06/2019	SS09-SS10	22	1	0	calm	Blue tender	7.9	-12.230458	123.153548	12:36	-12.229452	123.153221	12:38
17/06/2019	SS09-SS10	23	1	0	calm	Blue tender	7	-12.229326	123.153141	12:38	-12.228456	123.153022	12:40
17/06/2019	SS09-SS10	24	1	0	calm	Blue tender	7.6	-12.227787	123.152541	12:41	-12.226968	123.152362	12:43
17/06/2019	SS09-SS10	25	1	0	calm	Blue tender	7	-12.226161	123.152085	12:44	-12.225168	123.151853	12:46
17/06/2019	SS09-SS10	26	1	0	calm	Blue tender	8.4	-12.224322	123.151588	12:47	-12.223397	123.151492	12:49
17/06/2019	SS09-SS10	27	1	0	calm	Blue tender	10	-12.223259	123.151524	12:49	-12.222384	123.151237	12:51
17/06/2019	SS09-SS10	28	1	0	calm	Blue tender	5.6	-12.220756	123.150927	12:53	-12.219869	123.150548	12:55
17/06/2019	SS09-SS10	29	1	0	calm	Blue tender	5.5	-12.219777	123.150506	12:55	-12.218882	123.15024	12:57
17/06/2019	SS09-SS10	30	1	0	calm	Blue tender	4.7	-12.218083	123.150041	12:58	-12.217364	123.149675	13:00
17/06/2019	SS09-SS10	31	1	0	calm	Blue tender	6	-12.215844	123.14905	13:10	-12.215347	123.148539	13:12
17/06/2019	SS09-SS10	32	1	0	calm	Blue tender	7	-12.215253	123.148511	13:12	-12.213664	123.147861	13:15
17/06/2019	SS09-SS10	33	1	0	calm	Blue tender	9	-12.212808	123.147445	13:16	-12.211701	123.147166	13:18
17/06/2019	SS09-SS10	34	1	0	calm	Blue tender	7.6	-12.211556	123.147136	13:18	-12.2107	123.14642	13:20
17/06/2019	SS09-SS10	35	1	0	calm	Blue tender	3.9	-12.209939	123.146199	13:21	-12.209005	123.145867	13:23
17/06/2019	SS09-SS10	36	1	0	calm	Blue tender	4.3	-12.208924	123.145767	13:23	-12.2079	123.145245	13:25
17/06/2019	SS09-SS10	37	1	0	calm	Blue tender	5.3	-12.20712	123.144839	13:26	-12.2062	123.144347	13:28
17/06/2019	SS09-SS10	38	1	0	calm	Blue tender	6.4	-12.205415	123.143829	13:29	-12.204451	123.143305	13:31
17/06/2019	SS09-SS10	39	1	0	calm	Blue tender	5.6	-12.204356	123.143251	13:31	-12.202733	123.14221	13:34
17/06/2019	SS09-SS10	40	1	0	calm	Blue tender	4.4	-12.20262	123.142202	13:34	-12.201464	123.141908	13:36
17/06/2019	SS09-SS10	41	1	0	calm	Blue tender	3.9	-12.200876	123.141394	13:37	-12.199893	123.140937	13:39
17/06/2019	SS09-SS10	42	1	0	calm	Blue tender	7.3	-12.1992	123.140273	13:40	-12.198822	123.140254	13:41
17/06/2019	SS09-SS10	43	1	0	calm	Blue tender	6.1	-12.196677	123.139457	13:47	-12.196776	123.138424	13:50
17/06/2019	SS09-SS10	44	1	0	calm	Blue tender	4.1	-12.196623	123.138378	13:50	-12.195447	123.137906	13:52
17/06/2019	SS09-SS10	45	1	0	calm	Blue tender	3.4	-12.194836	123.137181	13:53	-12.194071	123.136662	13:55
17/06/2019	SS09-SS10	46	1	0	calm	Blue tender	3.7	-12.194062	123.136481	13:55	-12.193026	123.135978	13:57
17/06/2019	SS11-SS12	1	1	0	calm	Blue tender	3.6	-12.18856	123.126299	14:39	-12.188432	123.125406	14:41

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
17/06/2019	SS11-SS12	2	1	0	calm	Blue tender	2.3	-12.188426	123.125265	14:41	-12.188037	123.12411	14:43
17/06/2019	SS11-SS12	3	1	0	calm	Blue tender	2.7	-12.188037	123.12402	14:43	-12.188157	123.121892	14:46
17/06/2019	SS11-SS12	4	1	0	calm	Blue tender	2.7	-12.188126	123.12177	14:46	-12.187997	123.120742	14:48
17/06/2019	SS11-SS12	5	1	0	calm	Blue tender	3	-12.188211	123.119708	14:49	-12.188078	123.118618	14:51
17/06/2019	SS11-SS12	6	1	0	calm	Blue tender	2.3	-12.188061	123.118463	14:51	-12.188056	123.117119	14:53
17/06/2019	SS11-SS12	7	1	0	calm	Blue tender	3.5	-12.187934	123.11631	14:54	-12.187708	123.11516	14:56
17/06/2019	SS14-SS13	1	1	0	calm	Blue tender	3.8	-12.19162	123.078477	15:22	-12.191488	123.078884	15:24
17/06/2019	SS14-SS13	2	1	0	calm	Blue tender	2.8	-12.191583	123.078899	15:24	-12.191932	123.078019	15:27
17/06/2019	SS14-SS13	3	1	0	calm	Blue tender	4.1	-12.191981	123.077981	15:27	-12.192603	123.0779	15:29
17/06/2019	SS14-SS13	4	1	0	calm	Blue tender	4.5	-12.19265	123.078849	15:30	-12.192844	123.079466	15:32
18/06/2019	SS05-SS06	1	2	5	calm	Blue tender	6	-12.291925	123.064365	10:17	-12.291959	123.064937	10:20
18/06/2019	SS05-SS06	2	2	5	calm	Blue tender	4.9	-12.291979	123.065017	10:20	-12.292499	123.065576	10:22
18/06/2019	SS05-SS06	3	2	5	calm	Blue tender	3.5	-12.292547	123.065709	10:22	-12.293038	123.066963	10:25
18/06/2019	SS05-SS06	4	2	5	calm	Blue tender	6.5	-12.292985	123.067618	10:26	-12.293196	123.067595	10:28
18/06/2019	SS05-SS06	5	2	5	calm	Blue tender	10.9	-12.293237	123.067678	10:28	-12.293308	123.06841	10:31
18/06/2019	SS05-SS06	6	2	5	calm	Blue tender	9.2	-12.293355	123.068501	10:31	-12.293707	123.069483	10:34
18/06/2019	SS05-SS06	7	2	5	calm	Blue tender	7.6	-12.293666	123.069546	10:34	-12.293343	123.07038	10:36
18/06/2019	SS05-SS06	8	2	5	calm	Blue tender	5.2	-12.293406	123.071023	10:37	-12.29349	123.071668	10:39
18/06/2019	SS05-SS06	9	2	5	calm	Blue tender	6.7	-12.293466	123.071745	10:39	-12.293301	123.072695	10:41
18/06/2019	SS05-SS06	10	2	5	calm	Blue tender	5.6	-12.293295	123.072761	10:41	-12.293367	123.073895	10:44
18/06/2019	SS05-SS06	11	2	5	calm	Blue tender	5.2	-12.293192	123.074591	10:45	-12.293139	123.074921	10:47
18/06/2019	SS05-SS06	12	2	5	calm	Blue tender	7.3	-12.293152	123.074951	10:47	-12.293172	123.076026	10:50
18/06/2019	SS05-SS06	13	2	5	calm	Blue tender	6.4	-12.293248	123.075634	10:56	-12.293297	123.075828	10:58
18/06/2019	SS05-SS06	14	2	5	calm	Blue tender	6.6	-12.293161	123.076478	10:59	-12.293001	123.077139	11:01
18/06/2019	SS05-SS06	15	2	5	calm	Blue tender	5.4	-12.2928	123.077839	11:02	-12.29261	123.078487	11:04
18/06/2019	SS05-SS06	16	2	5	calm	Blue tender	7.5	-12.292281	123.078909	11:05	-12.292043	123.079734	11:07
18/06/2019	SS05-SS06	17	2	5	calm	Blue tender	7.8	-12.292041	123.079803	11:07	-12.291685	123.080296	11:09
18/06/2019	SS05-SS06	18	2	5	calm	Blue tender	7.4	-12.291673	123.080358	11:09	-12.291184	123.080979	11:11

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
18/06/2019	SS05-SS06	19	2	5	calm	Blue tender	5.7	-12.290417	123.081332	11:12	-12.290116	123.081756	11:14
18/06/2019	SS05-SS06	20	2	5	calm	Blue tender	8	-12.289736	123.082236	11:15	-12.289664	123.082832	11:17
18/06/2019	SS05-SS06	21	2	5	calm	Blue tender	8	-12.289645	123.083528	11:18	-12.289503	123.084285	11:20
18/06/2019	SS05-SS06	22	2	5	calm	Blue tender	6.7	-12.292129	123.099907	11:37	-12.292106	123.100554	11:39
18/06/2019	SS05-SS06	23	2	5	calm	Blue tender	6.6	-12.292079	123.100665	11:39	-12.292292	123.101433	11:41
18/06/2019	SS05-SS06	24	2	5	calm	Blue tender	6.9	-12.292268	123.101517	11:41	-12.292622	123.103208	11:44
18/06/2019	SS05-SS06	25	2	5	calm	Blue tender	6.2	-12.292656	123.103312	11:44	-12.293165	123.104169	11:46
18/06/2019	SS05-SS06	26	2	5	calm	Blue tender	8	-12.292945	123.104843	11:47	-12.292885	123.105806	11:49
18/06/2019	SS05-SS06	27	2	5	calm	Blue tender	6.2	-12.292913	123.106504	11:50	-12.292818	123.107086	11:52
18/06/2019	SS05-SS06	28	2	5	calm	Blue tender	6.4	-12.29284	123.107177	11:52	-12.292895	123.107933	11:54
18/06/2019	SS05-SS06	29	2	5	calm	Blue tender	5.5	-12.292928	123.10799	11:54	-12.2929	123.109424	11:57
18/06/2019	SS05-SS06	30	2	5	calm	Blue tender	5.3	-12.292781	123.110072	11:58	-12.292589	123.110415	11:59
18/06/2019	SS05-SS06	31	2	5	calm	Blue tender	5.9	-12.292692	123.111046	12:00	-12.292478	123.111756	12:02
18/06/2019	SS07-SS08	1	2	5	calm	Blue tender	5	-12.293028	123.1126	12:13	-12.291736	123.11267	12:15
18/06/2019	SS07-SS08	2	2	5	calm	Blue tender	6	-12.291776	123.112714	12:15	-12.29155	123.113948	12:18
18/06/2019	SS07-SS08	3	2	5	calm	Blue tender	10	-12.291566	123.114027	12:18	-12.291383	123.115317	12:21
18/06/2019	SS07-SS08	4	2	5	calm	Blue tender	6	-12.291394	123.115401	12:21	-12.291223	123.116524	12:24
18/06/2019	SS07-SS08	5	2	5	calm	Blue tender	6	-12.291233	123.116613	12:24	-12.291126	123.117341	12:26
18/06/2019	SS07-SS08	6	2	5	calm	Blue tender	6	-12.29109	123.117367	12:26	-12.290666	123.118612	12:29
18/06/2019	SS07-SS08	7	2	5	calm	Blue tender	7	-12.290691	123.11867	12:29	-12.29002	123.119818	12:32
18/06/2019	SS07-SS08	8	2	5	calm	Blue tender	7	-12.290058	123.119925	12:32	-12.289677	123.120524	12:34
18/06/2019	SS07-SS08	9	2	5	calm	Blue tender	6	-12.289746	123.120624	12:34	-12.290027	123.121687	12:37
18/06/2019	SS07-SS08	10	2	5	calm	Blue tender	5	-12.28969	123.122244	12:38	-12.289699	123.122926	12:40
18/06/2019	SS07-SS08	11	2	5	calm	Blue tender	6	-12.289688	123.122983	12:40	-12.289796	123.123739	12:42
18/06/2019	SS07-SS08	12	2	5	calm	Blue tender	7	-12.289807	123.123803	12:42	-12.289871	123.125345	12:46
18/06/2019	SS07-SS08	13	2	5	calm	Blue tender	8	-12.289635	123.125745	12:47	-12.289657	123.126109	12:48
19/06/2019	SS22-SS21	1	3	5	slight	Blue tender	6.8	-12.239847	122.979181	14:23	-12.239872	122.97979	14:25
19/06/2019	SS22-SS21	2	3	5	slight	Blue tender	8.5	-12.239852	122.97984	14:25	-12.239787	122.98065	14:28

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
19/06/2019	SS22-SS21	3	3	5	slight	Blue tender	3.8	-12.239793	122.980715	14:28	-12.239445	122.981436	14:32
19/06/2019	SS22-SS21	4	3	5	slight	Blue tender	5.6	-12.239359	122.981431	14:32	-12.239798	122.981808	14:34
19/06/2019	SS22-SS21	5	3	5	slight	Blue tender	8	-12.239866	122.981848	14:34	-12.239731	122.982744	14:37
19/06/2019	SS22-SS21	6	3	5	slight	Blue tender	2.4	-12.240285	122.982933	14:38	-12.240424	122.983027	14:40
19/06/2019	SS22-SS21	7	3	5	slight	Blue tender	6.4	-12.240433	122.983102	14:40	-12.241248	122.983677	14:43
19/06/2019	SS22-SS21	8	3	5	slight	Blue tender	3.6	-12.240575	122.984238	14:45	-12.240759	122.984498	14:47
19/06/2019	SS22-SS21	9	3	5	slight	Blue tender	1.5	-12.240812	122.984513	14:47	-12.241079	122.985193	14:49
19/06/2019	SS22-SS21	10	3	5	slight	Blue tender	6.6	-12.241461	122.984865	14:50	-12.241782	122.98456	14:52
19/06/2019	SS22-SS21	11	3	5	slight	Blue tender	6.2	-12.24238	122.985051	14:53	-12.242672	122.985552	14:55
19/06/2019	SS22-SS21	12	3	5	slight	Blue tender	6.2	-12.242689	122.985628	14:55	-12.242823	122.986202	14:57
19/06/2019	SS22-SS21	13	3	5	slight	Blue tender	2	-12.242926	122.98685	14:58	-12.242942	122.987466	15:00
19/06/2019	SS22-SS21	14	3	5	slight	Blue tender	7.5	-12.243184	122.98794	15:01	-12.243472	122.988528	15:03
19/06/2019	SS22-SS21	15	3	5	slight	Blue tender	0.8	-12.24347	122.988609	15:03	-12.243858	122.989245	15:05
19/06/2019	SS22-SS21	16	3	5	slight	Blue tender	2.9	-12.243418	122.989519	15:14	-12.24299	122.989946	15:16
19/06/2019	SS22-SS21	17	3	5	slight	Blue tender	2.1	-12.242439	122.989948	15:17	-12.243298	122.990085	15:19
19/06/2019	SS22-SS21	18	3	5	slight	Blue tender	3.7	-12.243427	122.990042	15:19	-12.243411	122.990398	15:21
19/06/2019	SS22-SS21	19	3	5	slight	Blue tender	5.4	-12.2432	122.990829	15:22	-12.243315	122.991087	15:24
19/06/2019	SS22-SS21	20	3	5	slight	Blue tender	1.3	-12.243267	122.991125	15:24	-12.244001	122.991062	15:26
19/06/2019	SS22-SS21	21	3	5	slight	Blue tender	3.7	-12.244034	122.991442	15:27	-12.244303	122.991697	15:29
19/06/2019	SS22-SS21	22	3	5	slight	Blue tender	4.1	-12.243731	122.991779	15:30	-12.244171	122.991864	15:32
19/06/2019	SS22-SS21	23	3	5	slight	Blue tender	2.6	-12.244598	122.992252	15:33	-12.244619	122.992525	15:35
19/06/2019	SS22-SS21	24	3	5	slight	Blue tender	5.6	-12.244891	122.992878	15:36	-12.244998	122.993336	15:38
19/06/2019	SS22-SS21	25	3	5	slight	Blue tender	2.4	-12.244799	122.993835	15:39	-12.244532	122.994379	15:41
19/06/2019	SS22-SS21	26	3	5	slight	Blue tender	3.6	-12.24439	122.994701	15:42	-12.244457	122.994973	15:44
19/06/2019	SS22-SS21	27	3	5	slight	Blue tender	3.6	-12.244522	122.995443	15:45	-12.244504	122.995713	15:47
19/06/2019	SS22-SS21	28	3	5	slight	Blue tender	3.4	-12.244647	122.995628	15:47	-12.244941	122.996165	15:49
19/06/2019	SS22-SS21	29	3	5	slight	Blue tender	3.1	-12.244335	122.996416	15:50	-12.244565	122.996934	15:52
19/06/2019	SS22-SS21	30	3	5	slight	Blue tender	4.1	-12.245169	122.997073	15:53	-12.244359	122.997433	15:55

Date	Transect	Tow no.	Wind (kn)	Cloud cover (okta)	Sea condition	Vessel	Depth (m)	Start Lat °)	Start Long °)	Start Time	End Lat °)	End Long °)	End Time
19/06/2019	SS22-SS21	31	3	5	slight	Blue tender	4	-12.243936	122.997794	16:05	-12.24377	122.997926	16:07
19/06/2019	SS22-SS21	32	3	5	slight	Blue tender	5.5	-12.244524	122.997936	16:08	-12.244691	122.998289	16:10
20/06/2019	SS22-SS21	33	3	5	slight	Blue tender	1.4	-12.244894	122.998539	15:40	-12.244074	122.998811	15:42
20/06/2019	SS22-SS21	34	3	5	slight	Blue tender	1.1	-12.244058	122.998862	15:42	-12.244685	122.999024	15:44
20/06/2019	SS22-SS21	35	3	5	slight	Blue tender	8	-12.244763	122.998983	15:44	-12.245062	122.99903	15:46
20/06/2019	SS22-SS21	36	3	5	slight	Blue tender	1.8	-12.24438	123.000079	15:48	-12.244245	123.000482	15:50
20/06/2019	SS22-SS21	37	3	5	slight	Blue tender	2.9	-12.244538	123.000781	15:51	-12.244839	123.001066	15:53
20/06/2019	SS22-SS21	38	3	5	slight	Blue tender	1.6	-12.244828	123.001132	15:53	-12.243732	123.001452	15:55
20/06/2019	SS22-SS21	39	3	5	slight	Blue tender	1.1	-12.243645	123.001404	15:55	-12.243133	123.001626	15:57

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