Australia's National
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Ashmore Reef Marine Park Environmental Assessment

Terrestrial Section (Part 2 of 4)

Final report to Parks Australia

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

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).

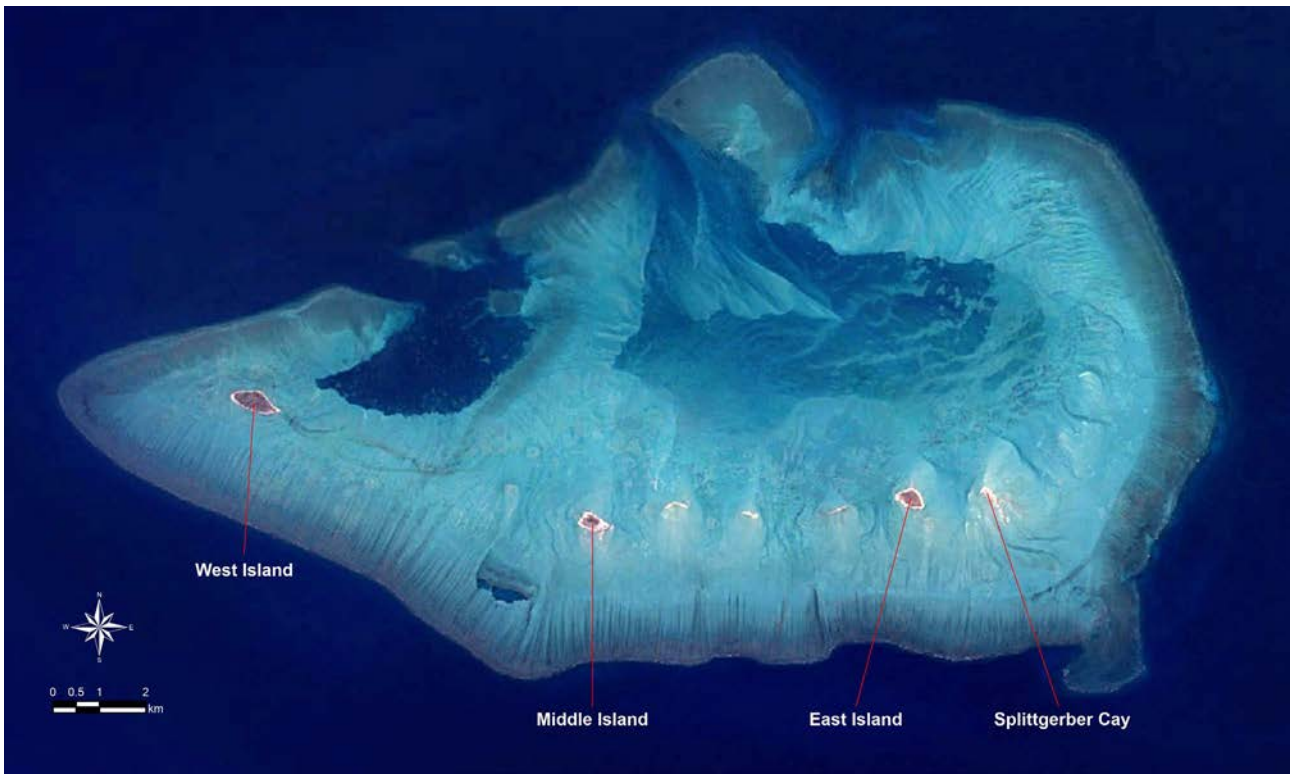


Figure 4. The location of the four islands – West Island, Middle Island, East Island and Splittgerber Cay - within the Ashmore Reef Marine Park that were surveyed for vegetation in May 2019.

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 m² 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.

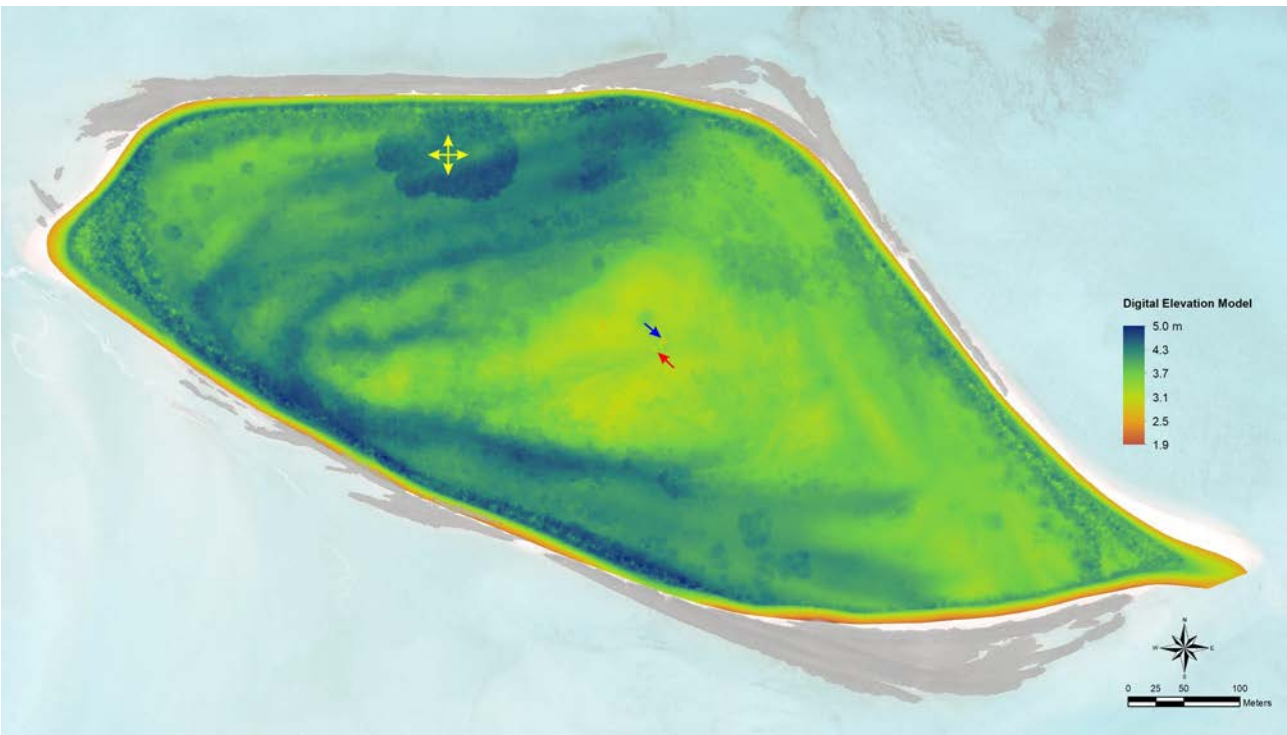


Figure 9. Digital elevation model (DEM) for West Island at Ashmore Reef in May 2019. Locations of island well (blue arrow), excavation (red arrow), the *Spinifex longifolius* patch (yellow arrows) and the surrounding rock shelf structure (light grey) are depicted.

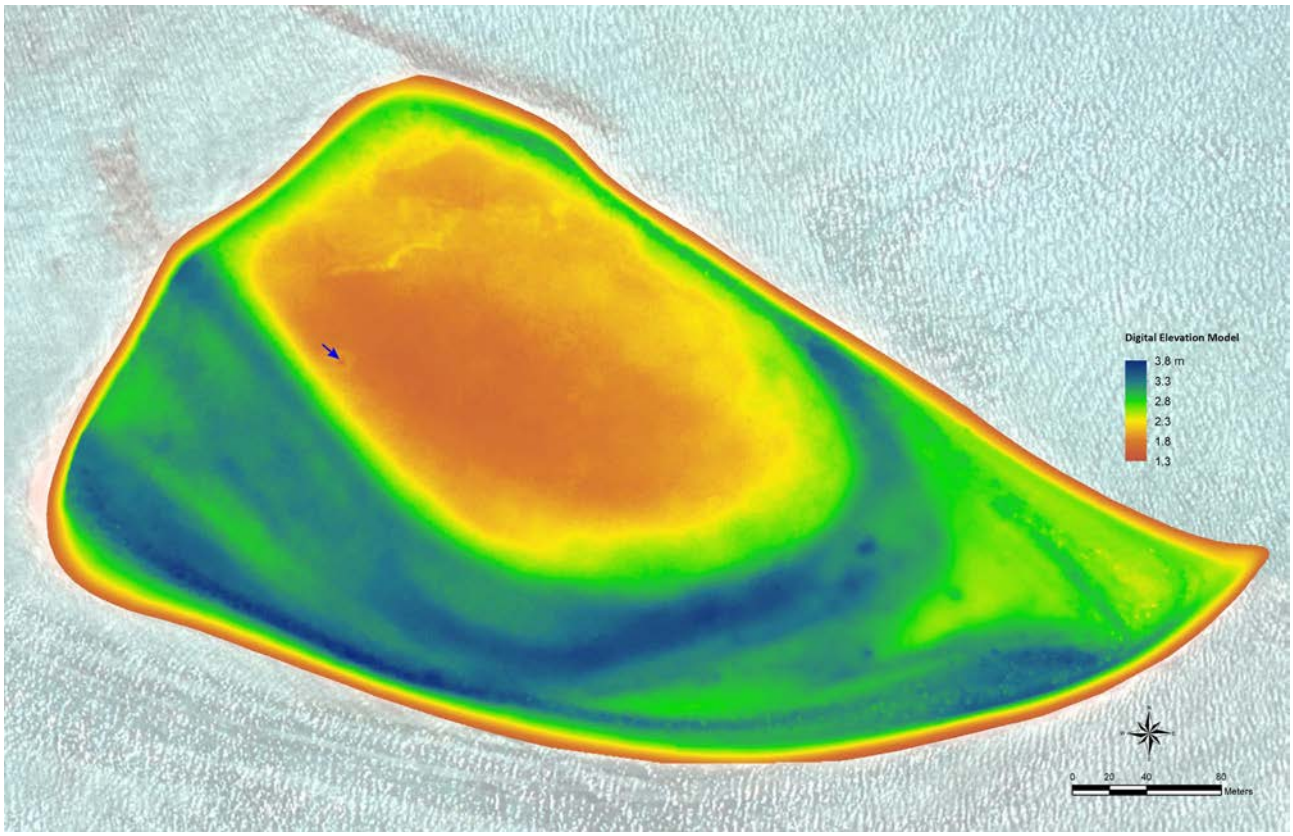


Figure 10. Digital elevation model (DEM) for Middle Island at Ashmore Reef in May 2019. Location of island well (blue arrow) is depicted.

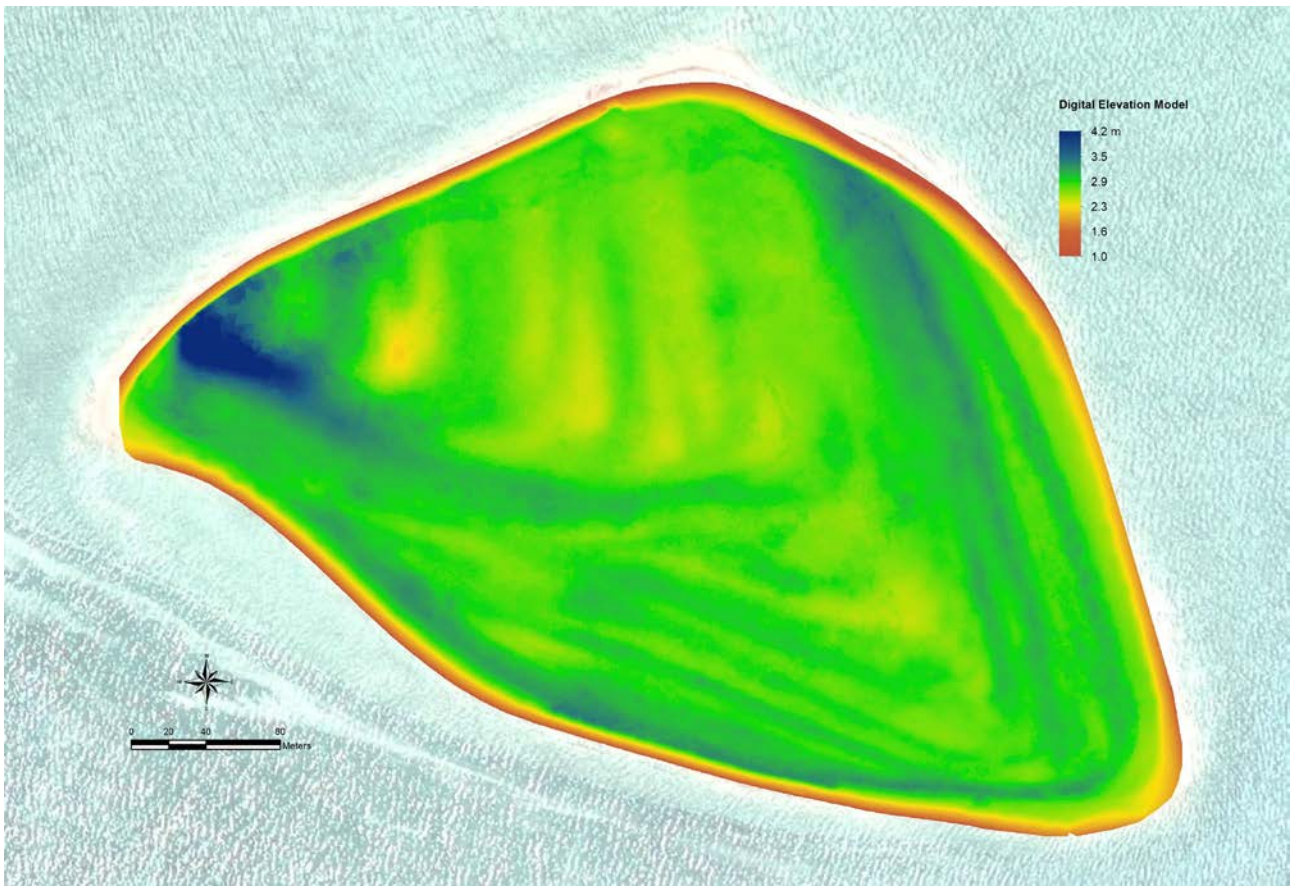


Figure 11. Digital elevation model (DEM) for East Island at Ashmore Reef in May 2019.

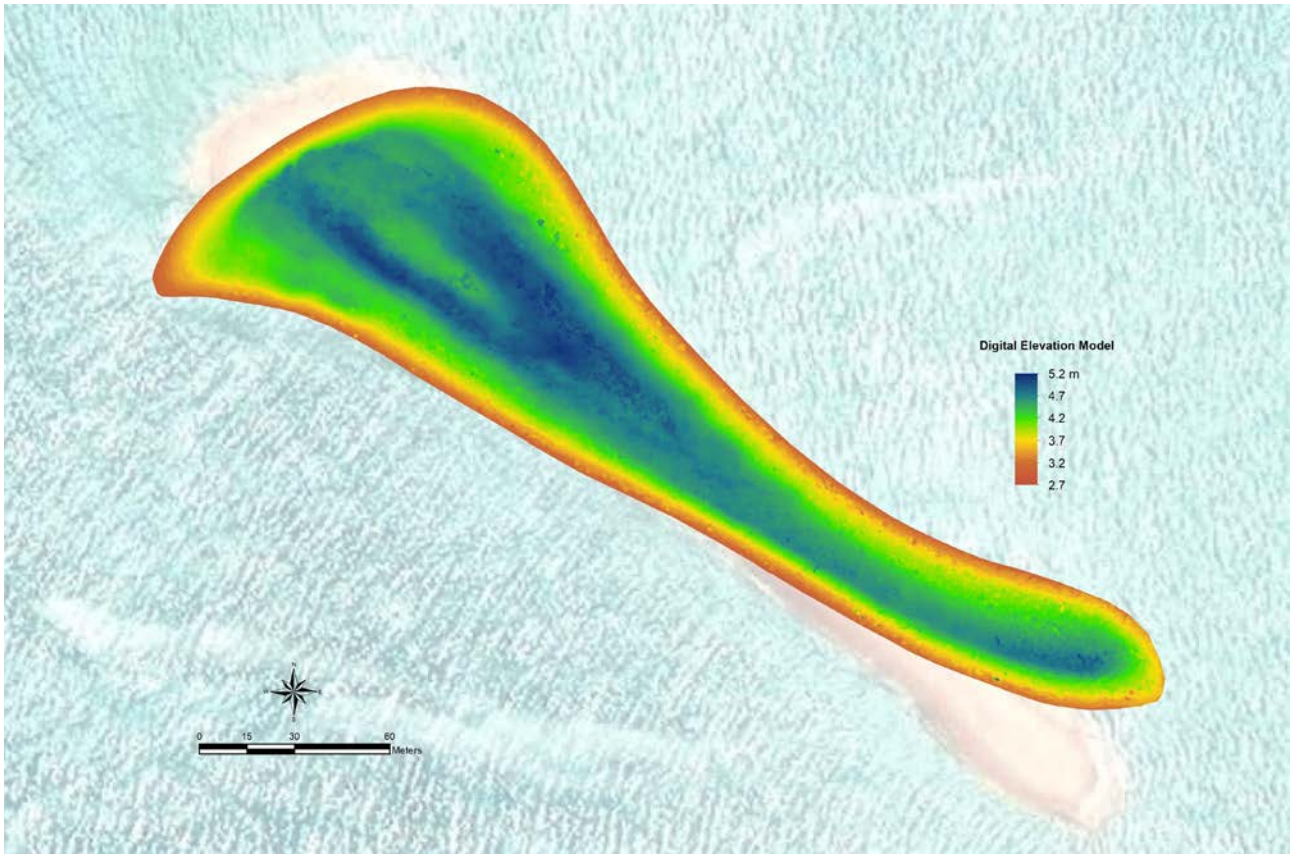


Figure 12. Digital elevation model (DEM) for Splittgerber Cay at Ashmore Reef in May 2019.

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).

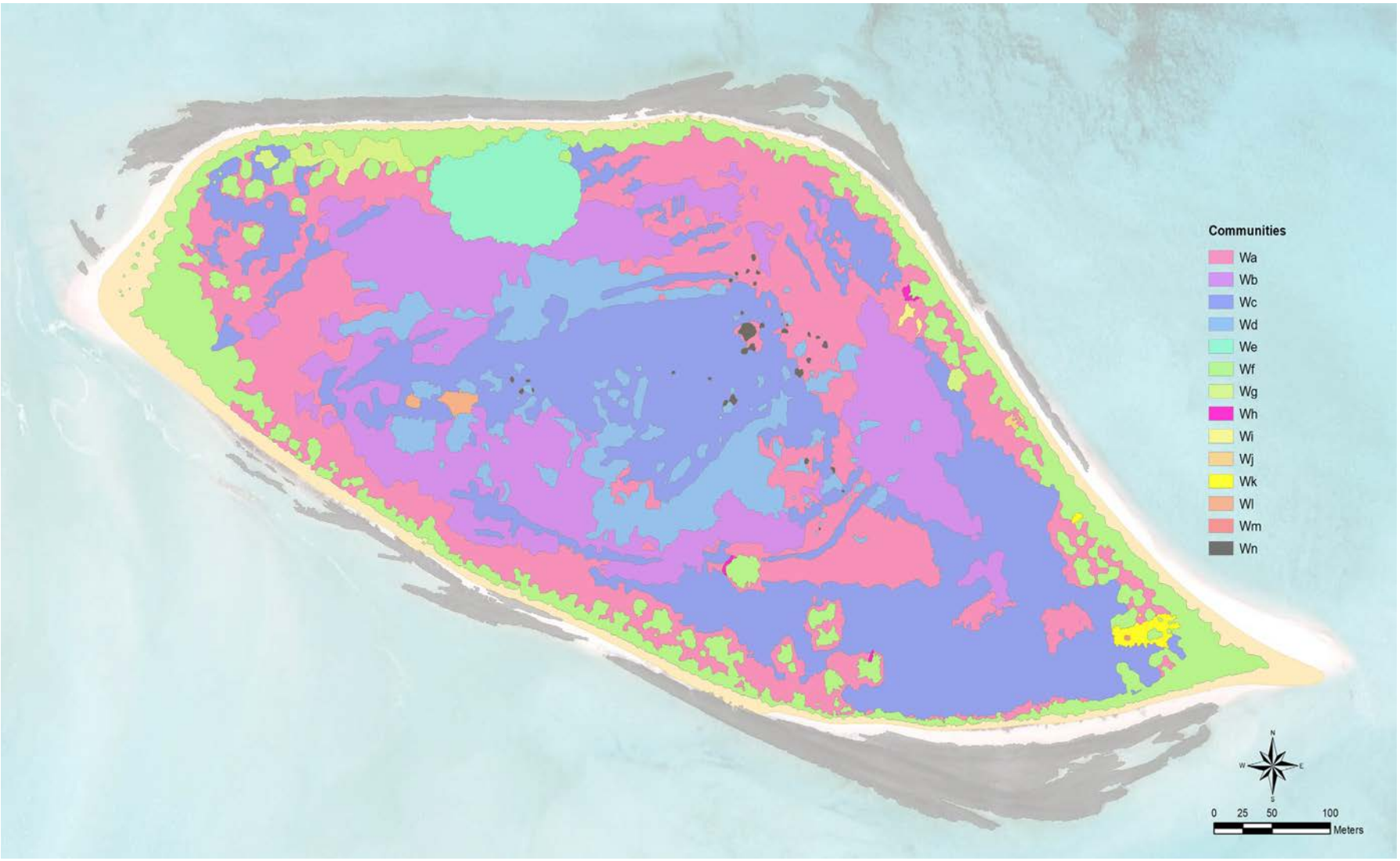


Figure 14. Vegetation community map of West Island at Ashmore Reef in May 2019. Species composition and abundance for each of the 14 communities is detailed in Table 8. Associated island shoreline (cream-yellow) and surrounding rock shelf structure shelf structure (light grey) is also depicted.

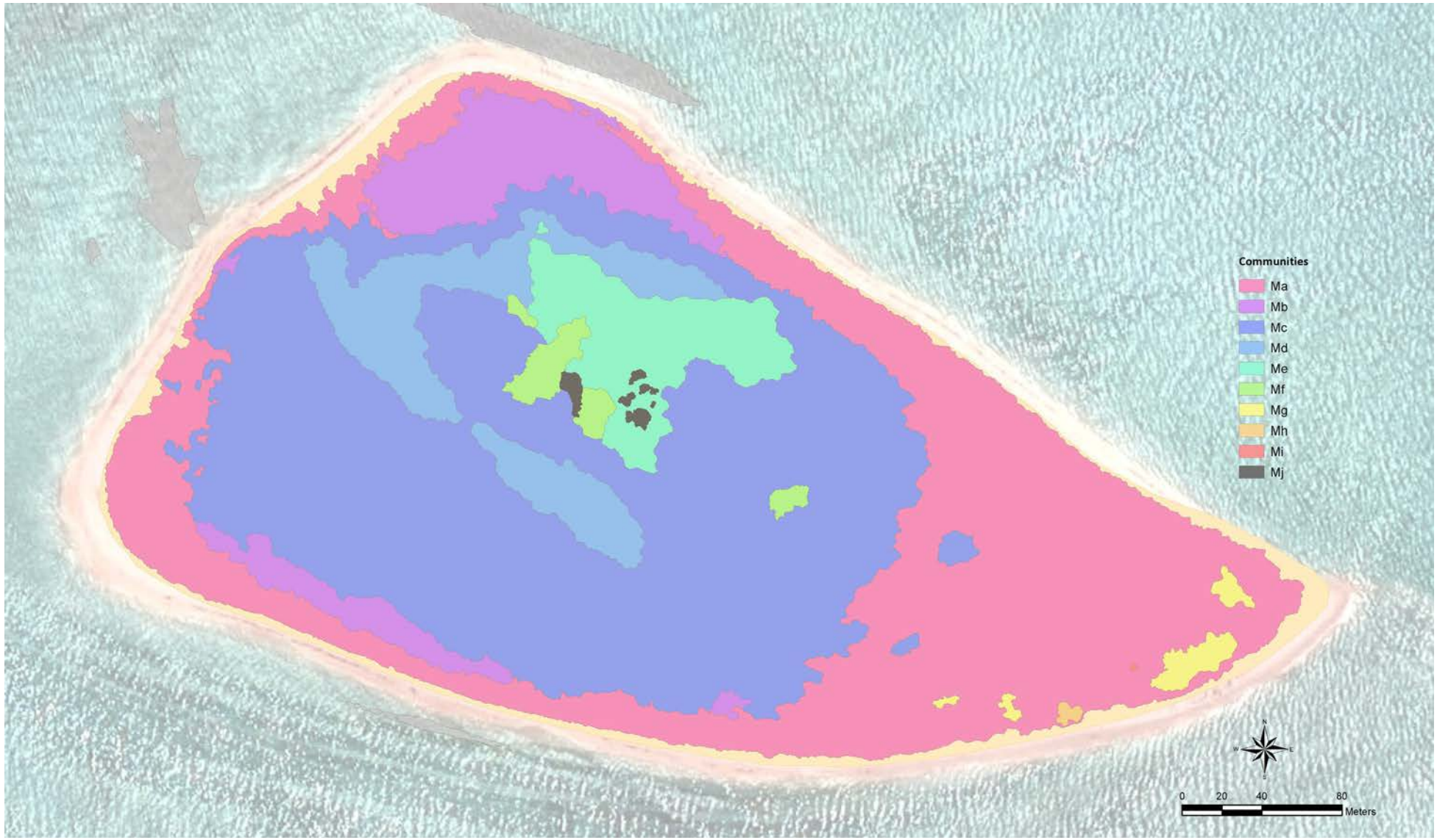


Figure 15. Vegetation community map of Middle Island at Ashmore Reef in May 2019. Species composition and abundance for each of the 10 communities is detailed in Table Table 8. Associated island shoreline (cream-yellow) and surrounding rock shelf structure (light grey) is also depicted.

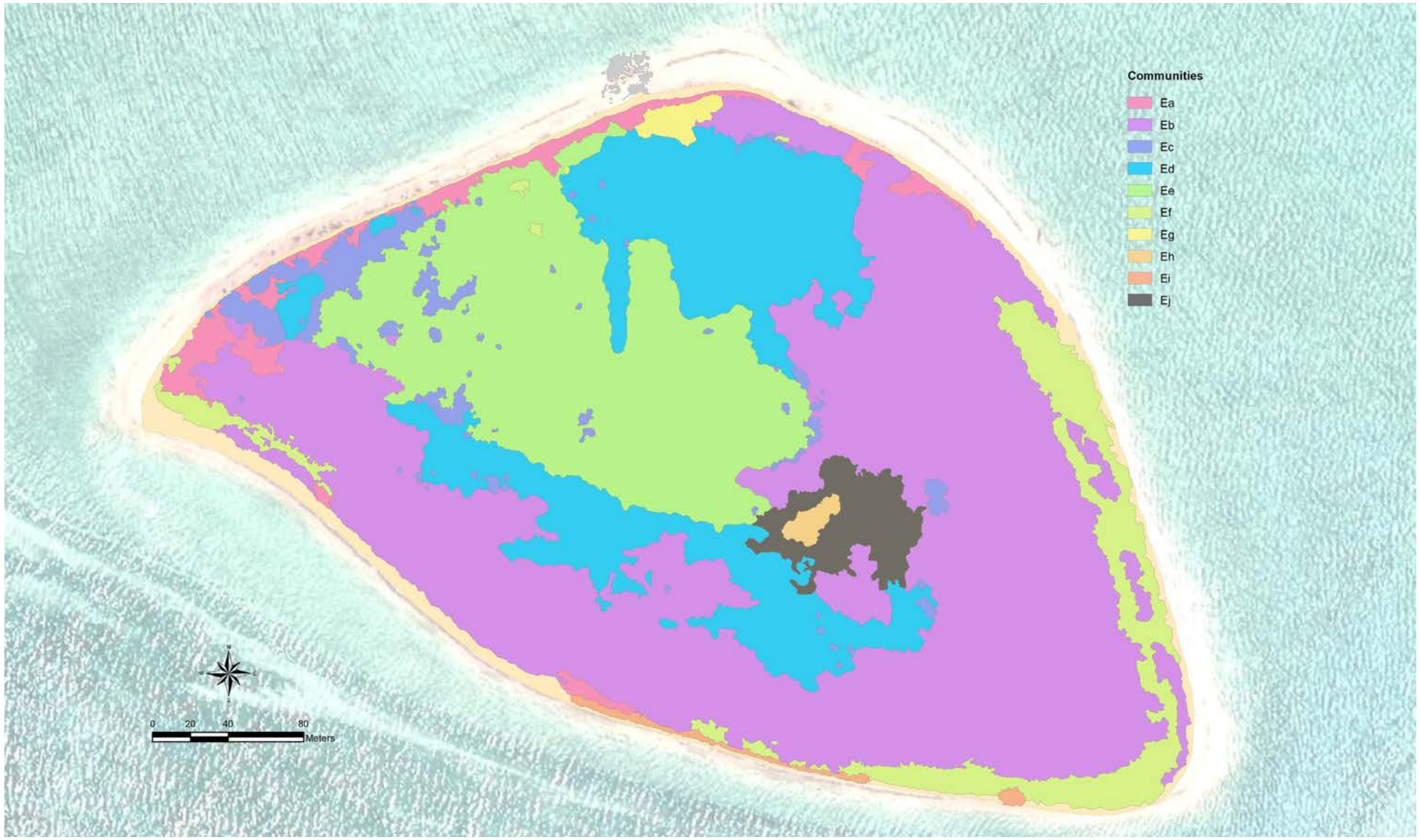


Figure 16. Vegetation community map of East Island at Ashmore Reef in May 2019. Species composition and abundance for each of the 10 communities is detailed in Table 9. Associated island shoreline (cream-yellow) and a fragmented concrete helipad (light grey) is also depicted.

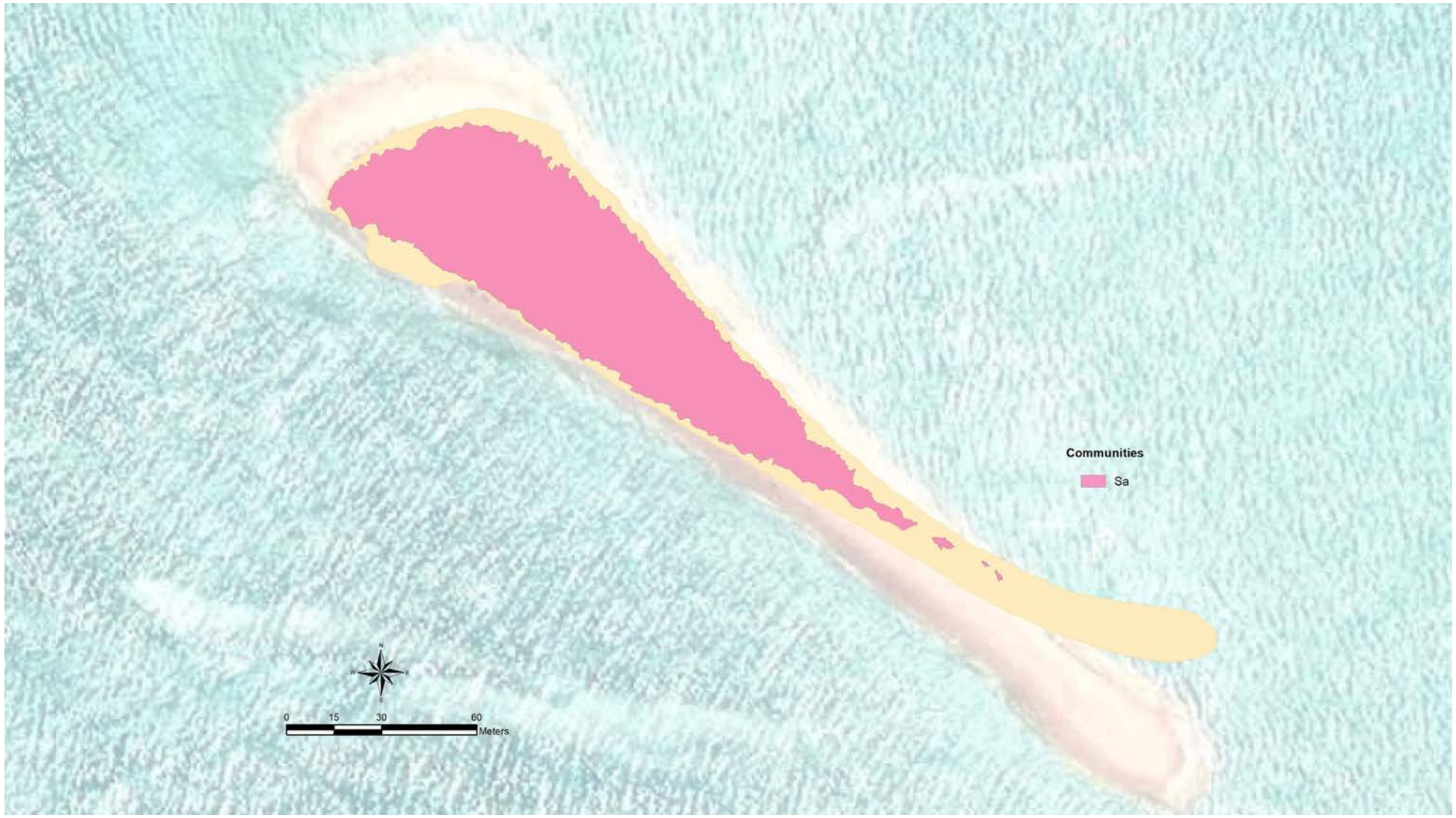


Figure 17. Vegetation community map of Splittgerber Cay at Ashmore Reef in May 2019. Species composition and abundance for the single plant community is detailed in Table 10. Associated island shoreline (cream-yellow) is also depicted.

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).

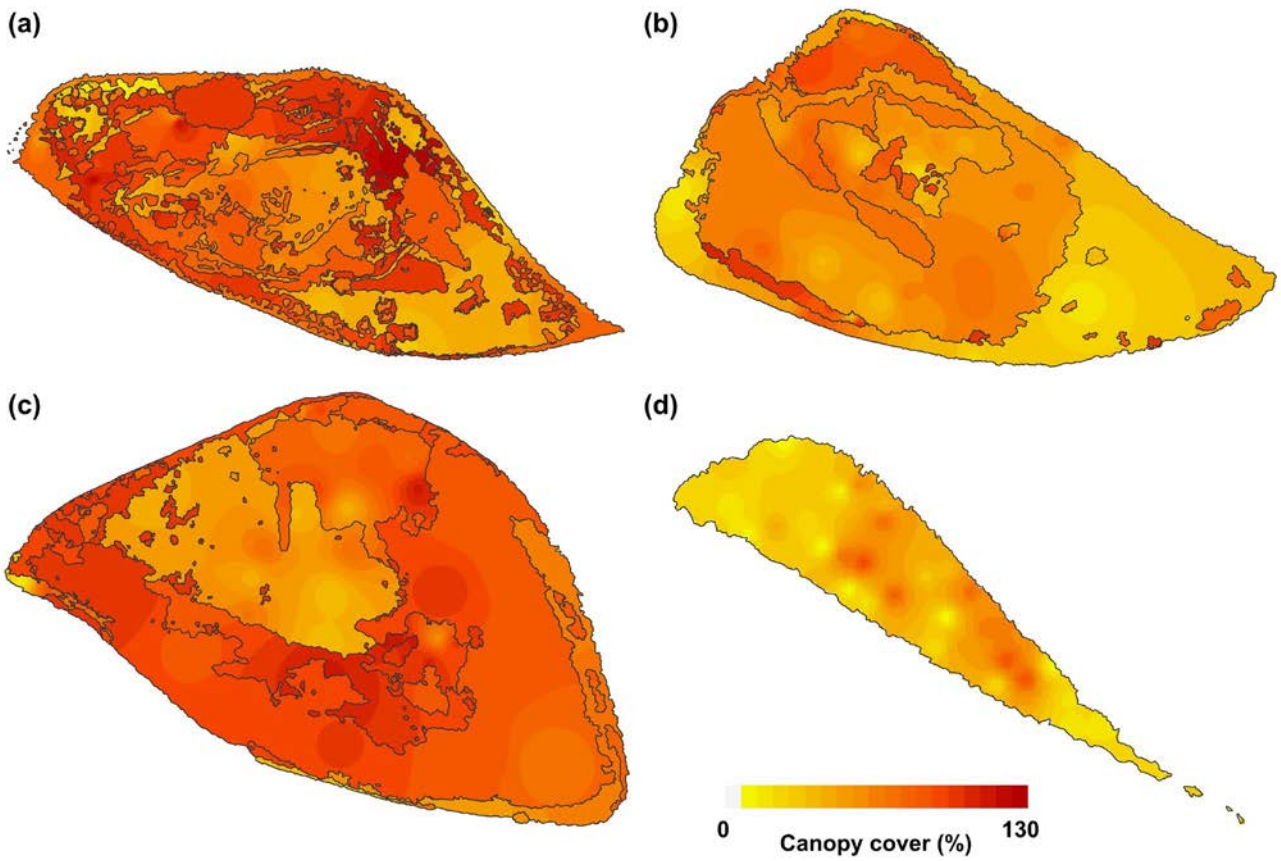


Figure 18. Canopy cover abundance scores for all plant species surveyed the four islands at Ashmore Reef in May 2019 – West Island (a), Middle Island (b), East Island (c) and Splittgerber Cay (d).

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).

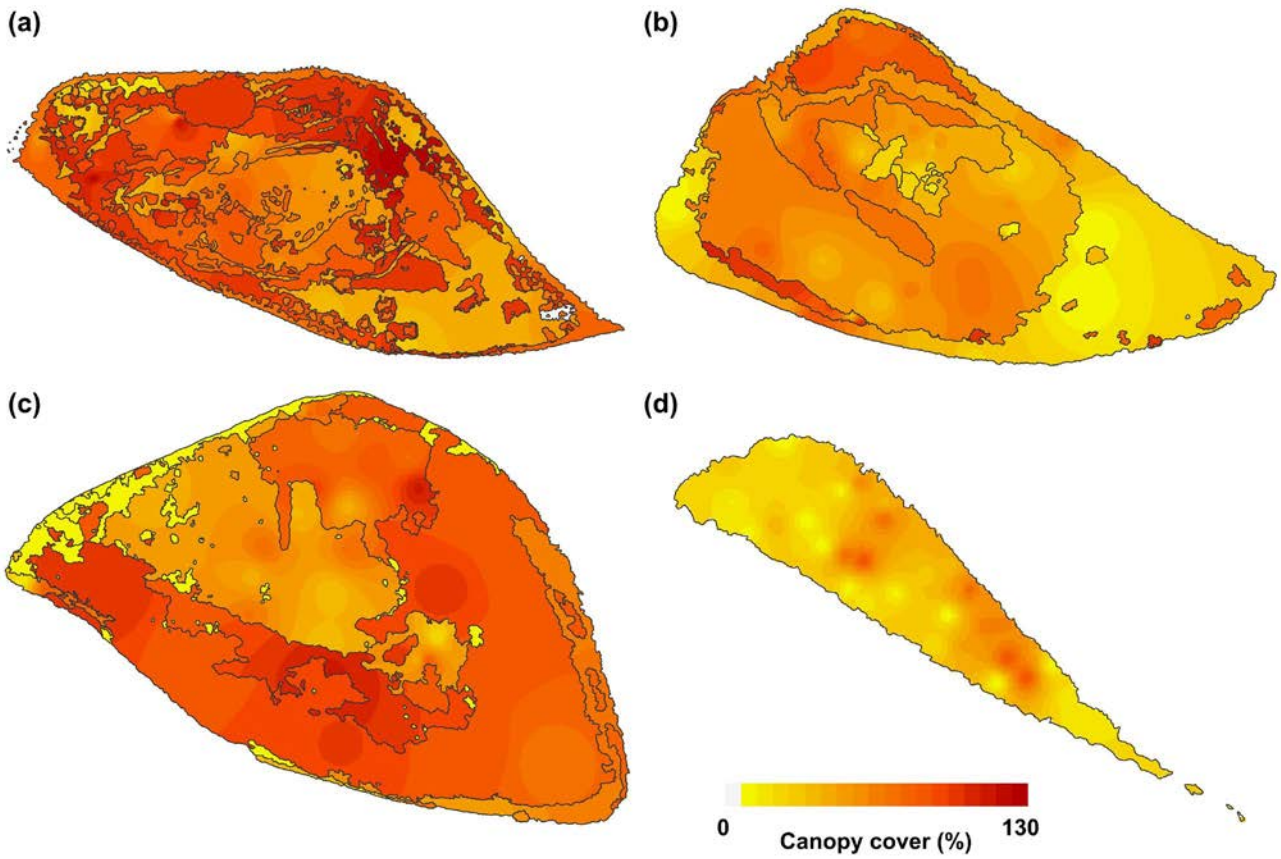


Figure 19. Canopy cover abundance scores for all native plant species surveyed the four islands at Ashmore Reef in May 2019 – West Island (a), Middle Island (b), East Island (c) and Splittgerber Cay (d).

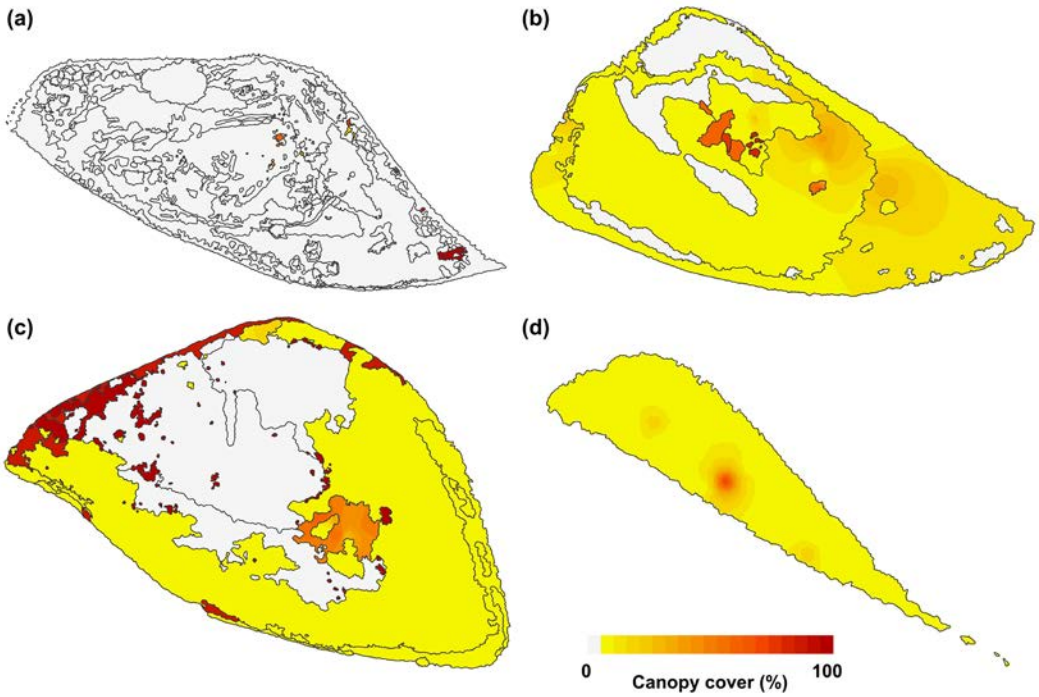


Figure 20. Canopy cover abundance scores for all non-native plant species surveyed the four islands at Ashmore Reef in May 2019 – West Island (a), Middle Island (b), East Island (c) and Splittgerber Cay (d).

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	239161m², 45.1%	56790m², 23.7%	62/101Q, 14C
Boerhavia spp.	202133m², 38.1%	21834m², 10.8%	17/56Q, 7C
Bulbostylis barbata	1985m², 0.4%	68m², 3.4%	5/5Q, 1C
Cordia subcordata	37074m², 7.0%	37m², 0.1%	0/0Q, 1C
Cuscuta victoriana	4140m², 0.8%	652m², 15.7%	11/11Q, 3C
Digitaria mariannensis	262781m², 49.6%	68893m², 26.2%	50/89Q, 11C
Eragrostis cumingii	242986m², 45.8%	76029m², 31.3%	44/120Q, 14C
Guettarda speciosa	37074m², 7.0%	981m², 2.6%	1/17Q, 1C
Heliotropium foertherianum	37085m², 7.0%	5942m², 16.0%	5/17Q, 2C
Ipomoea pes-caprae	37074m², 7.0%	327m², 0.9%	1/17Q, 1C
Ipomoea violacea	258915m², 48.8%	25111m², 9.7%	20/61Q, 8C
Lepturus repens	198463m², 37.4%	31529m², 15.9%	76/100Q, 15C
Pandanus sp.	37074m², 7.0%	37m², 0.1%	0/0Q, 1C
Portulaca tuberosa	229435m², 43.3%	10261m², 4.5%	10/40Q, 7C
Sesbania cannabina	363447m², 68.6%	23609m², 6.5%	32/83Q, 14C
Sesuvium portulacastrum	37074m², 7.0%	7044m², 19.0%	6/17Q, 1C
Sida pusilla	298367m², 56.3%	51065m², 17.1%	59/116Q, 14C
Spinifex littoreus	115m², 0.02%	106m², 92.5%	2/2Q, 1C
Spinifex longifolius	8722m², 1.6%	8544m², 98.0%	4/4Q, 2C
Sporobolus virginicus	1010m², 0.2%	595m², 58.9%	6/6Q, 2C
Suriana maritima	37074m², 7.0%	37m², 0.1%	0/0Q, 1C
*Cenchrus brownii**	529m², 0.1%	255m², 48.2%	11/11Q, 1C
*Cenchrus ciliaris**	891m², 0.2%	822m², 92.3%	3/3Q, 1C

Coverage and abundance (423 community fragments)			
Species	**A_C m², A_I%**	**A_O m², D%**	**Q_P/Q_T, C**
*Cenchrus echinatus**	165m², 0.03%	115m², 70.0%	3/3Q, 1C
*Cenchrus pedicellatus**	16m², 0.003%	15m², 95.0%	1/1Q, 1C
*Cleome gynandra**	61735m², 11.6%	1931m², 3.1%	9/32Q, 4C
*Portulaca oleracea**	295835m², 55.8%	19579m², 6.6%	40/102Q, 13C
*Tribulus cistoides**	119043m², 22.5%	13754m², 11.6%	16/16Q, 7C
*Xenostegia tridentata**	152m², 0.03%	25m², 16.7%	1/3Q, 1C
All species	530078m², 100%	367160m², 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 72092m²	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 110544m²	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 53048m²	7/19Q 4C
										Ashmore Reef (total)				445800 235683m²	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*).

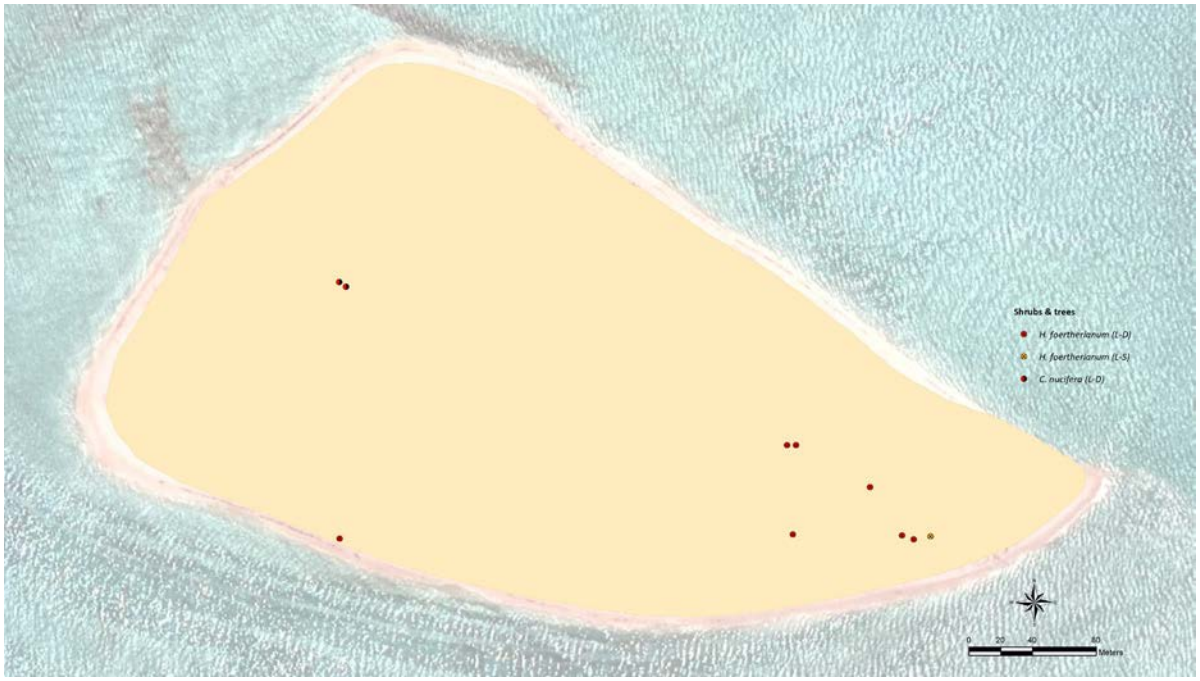
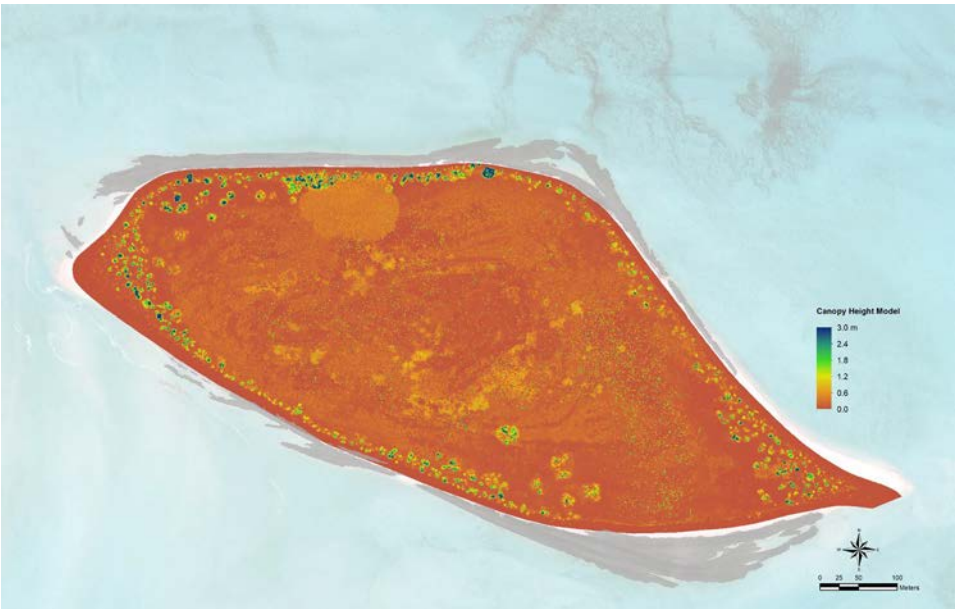
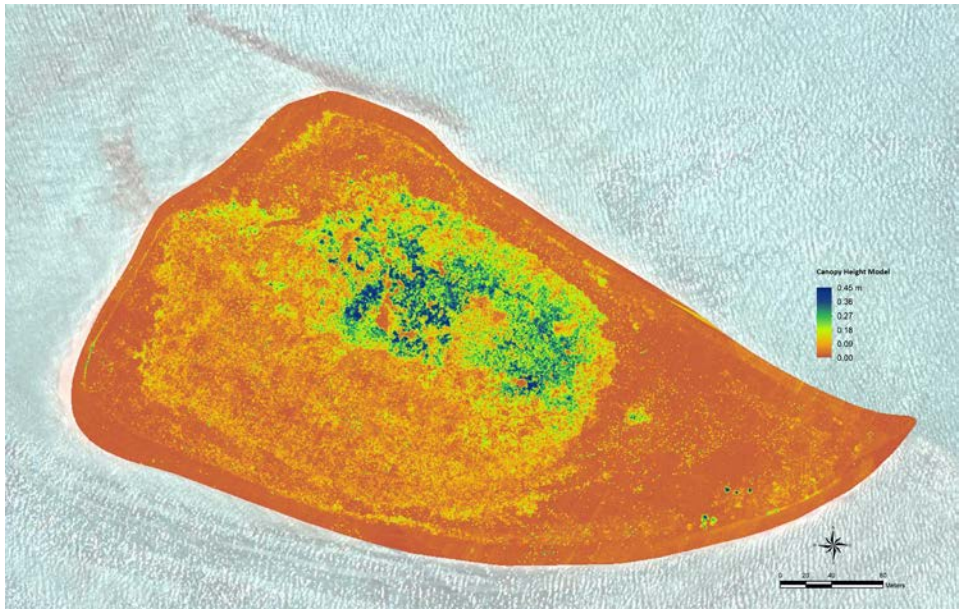


Figure 22. Shrubs and trees on Middle Island at Ashmore Reef in May 2019. The two species, *Heliotropium foertherianum* and *Cocos nucifera* (non-native) were stratified according to canopy diameter into small (S; <2 m dia, lighter shading) and large (L; >2 m dia, darker shading). All individuals were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





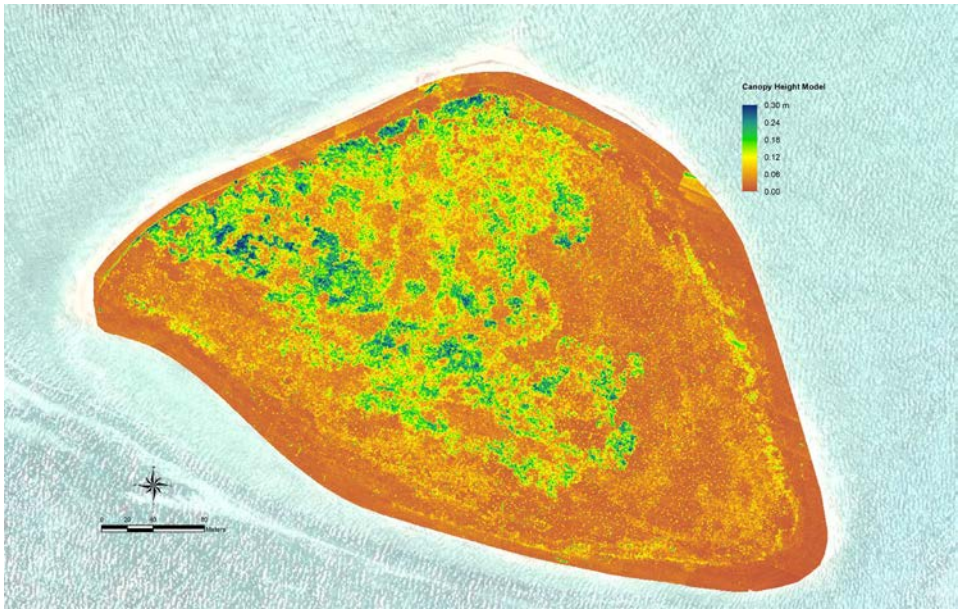


Figure 24. Canopy height models (CHMs) for the three islands at Ashmore Reef that had live shrubs or trees present in May 2019 – West Island (a), Middle Island (b) and East Island (c). Note that the height scale colour gradient (m) differs between plates.

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).

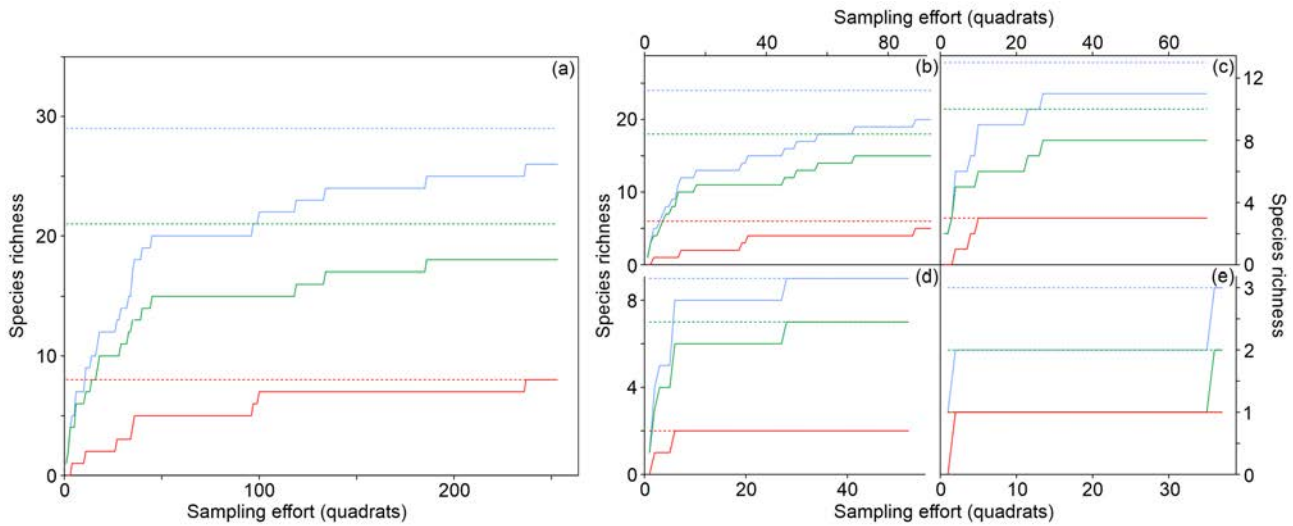


Figure 25. Species accumulation curves comparing species detected during the community survey (dotted lines) with species detected during the quadrat-based abundance survey (solid lines) for all four islands at Ashmore Reef combined (a), as well as broken down for West Island (b), Middle Island (c), 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).

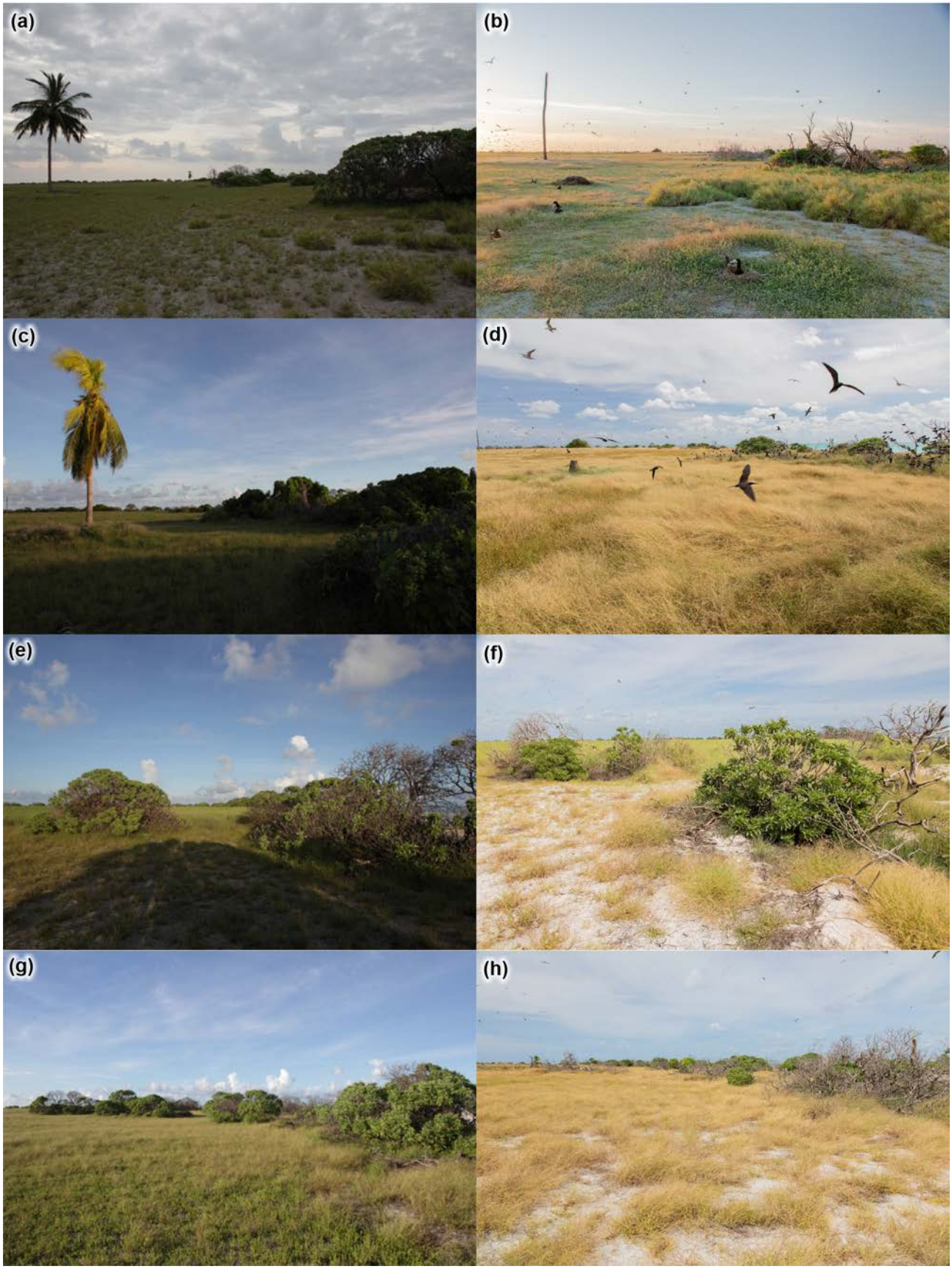
Table 14. Native plant presence over time at Ashmore reef based on field surveys of terrestrial plant abundance and structure in 1977 (Kenneally 1993), 1996-1997 (Pike & Leach 1997), 2002 (Cowie 2004), 2010 (Clarke 2010), 2015 (Westaway 2015) and 2019 (this survey). Splittgerber Cay, a relatively recently 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)

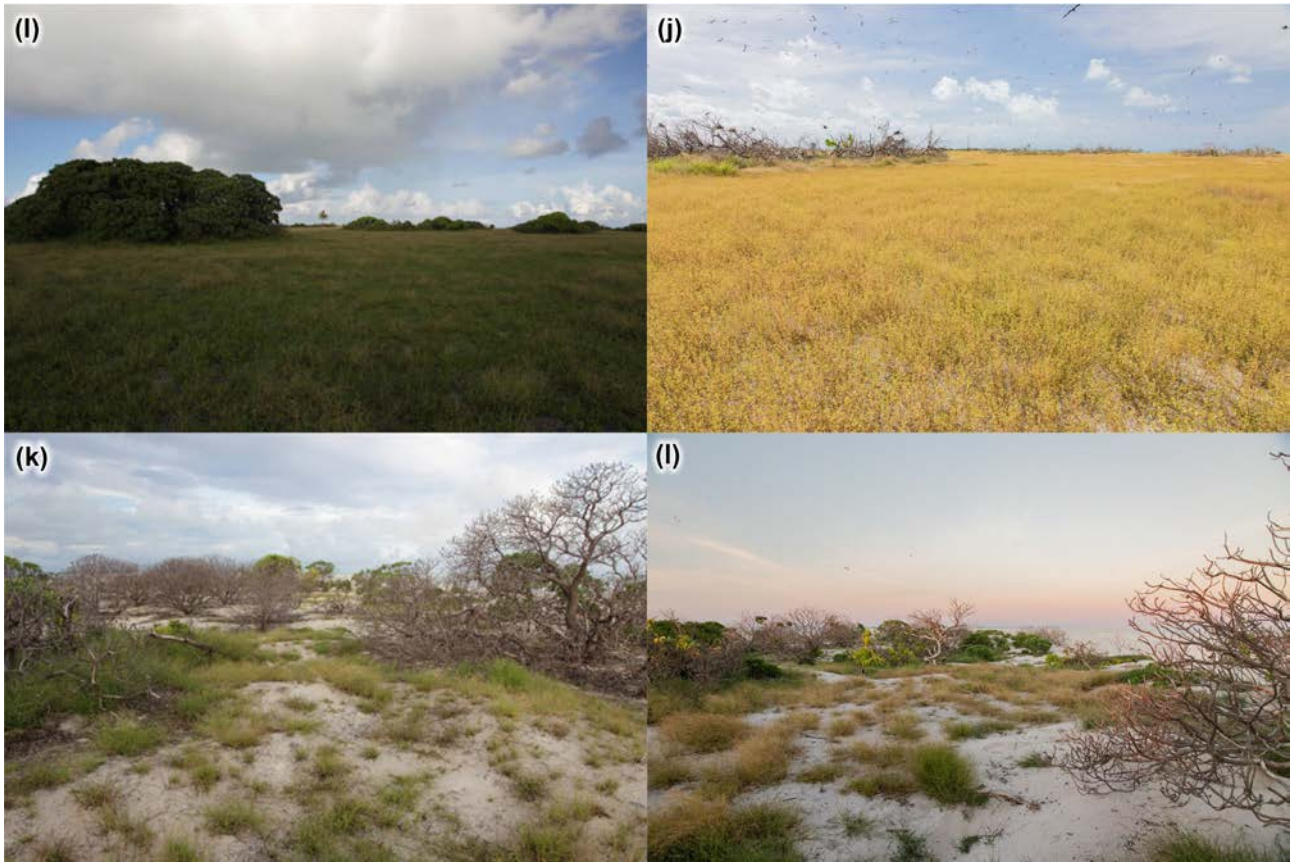


Figure 26. Permanent photo points at West Island, Ashmore Reef, showing change over time between April 2010 (left column) and May 2019 (right column). 2010 photos were from Clarke (2010) and the 2019 resurvey used provided location data where possible (SI Table 1). Photo points 1 (a, b), 2 (c, d), 3, (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).

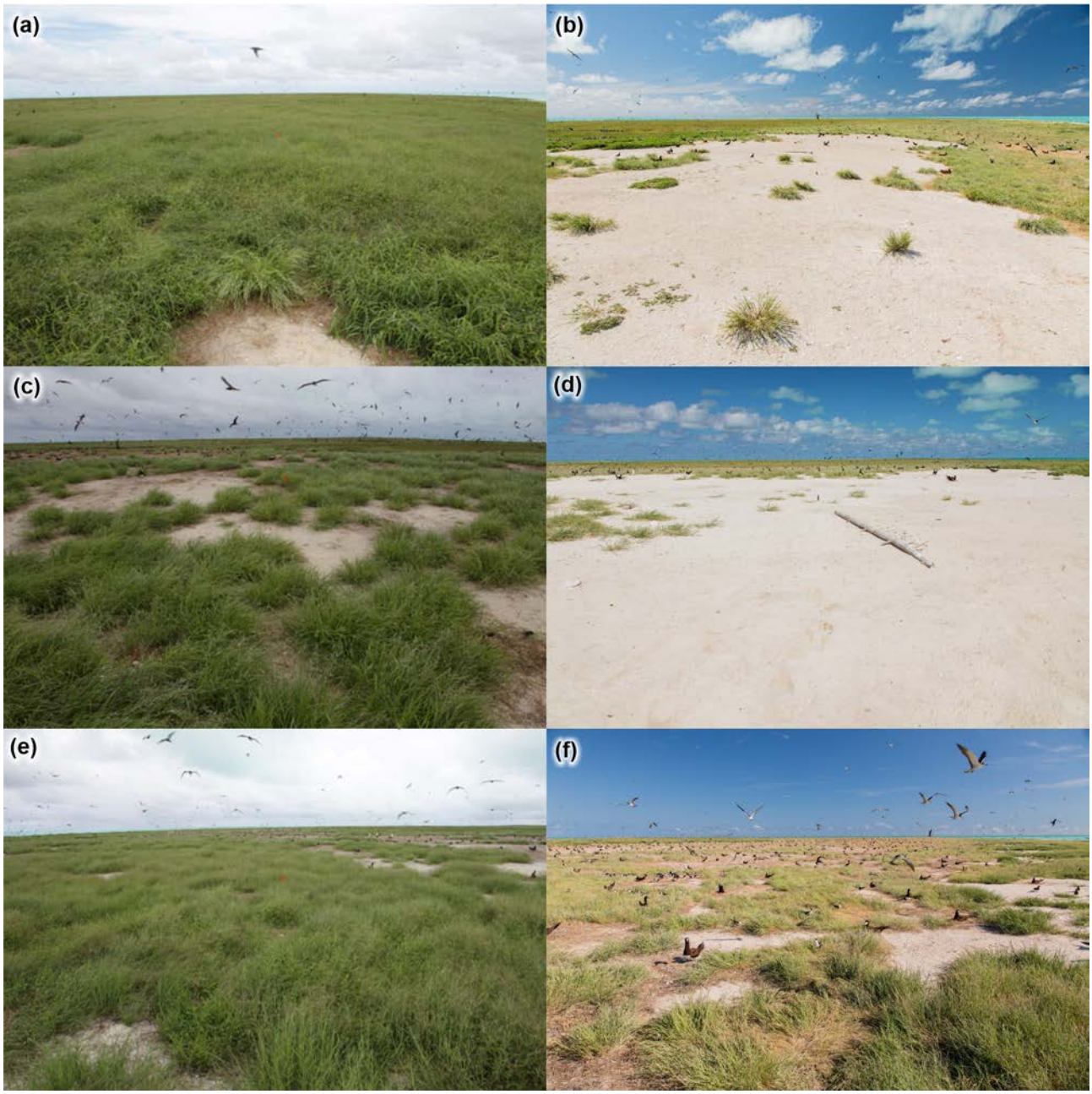


Figure 28. Permanent photo points at East Island, Ashmore Reef, showing change over time between April 2010 (left column) and May 2019 (right column). 2010 photos were from Clarke (2010) and the 2019 resurvey used provided location data where possible (SI Table 1). Photo points 1 (a, b), 2 (c, d) and 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.

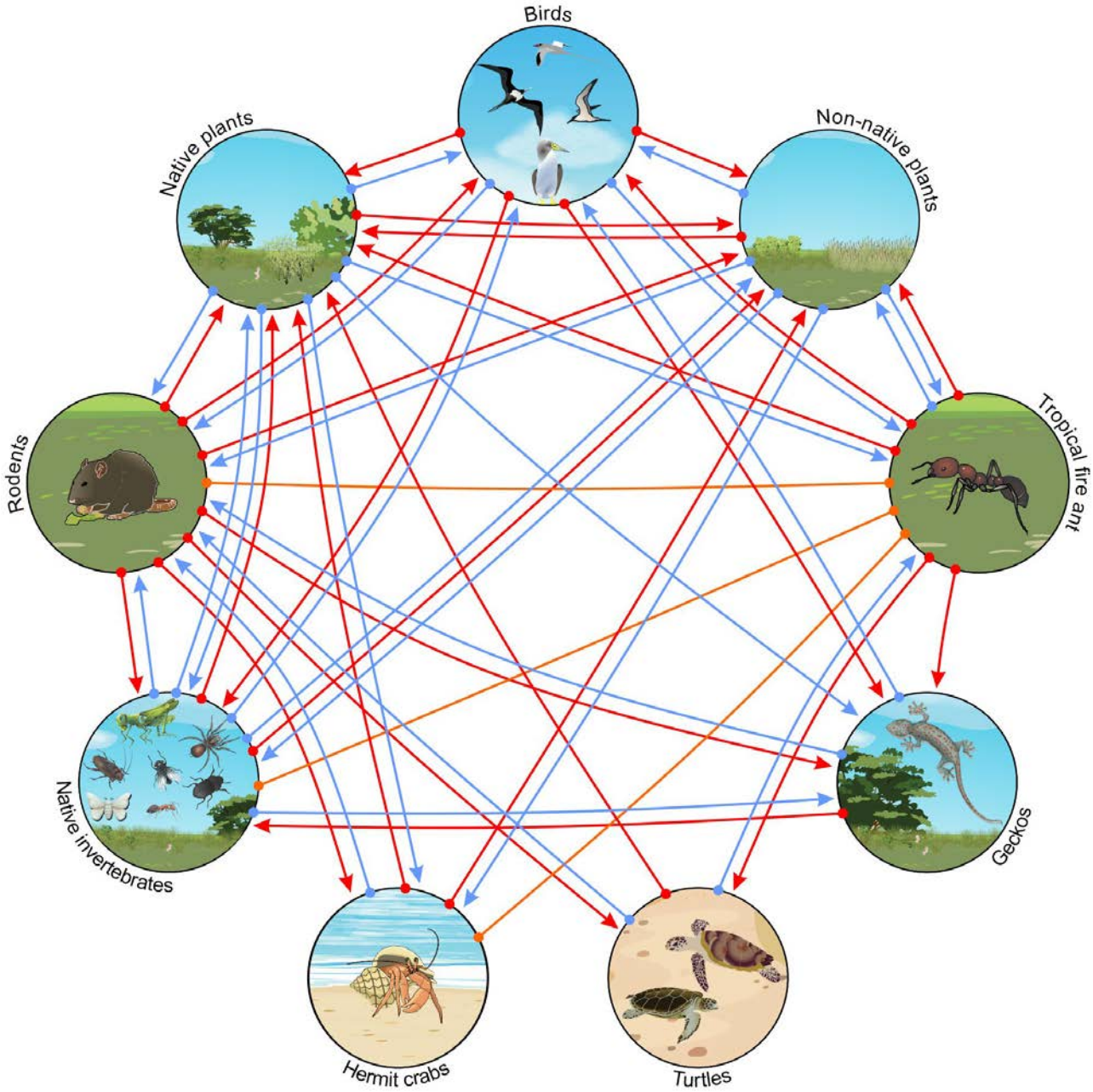


Figure 30. Theoretical species interaction network for the islands of Ashmore Reef. Direct interactions are shown, with positive (blue) and negative (red) outcomes shown from the perspective of the receiver (arrowhead). Orange interactions are likely to exist, but the nature of the interaction is not 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.

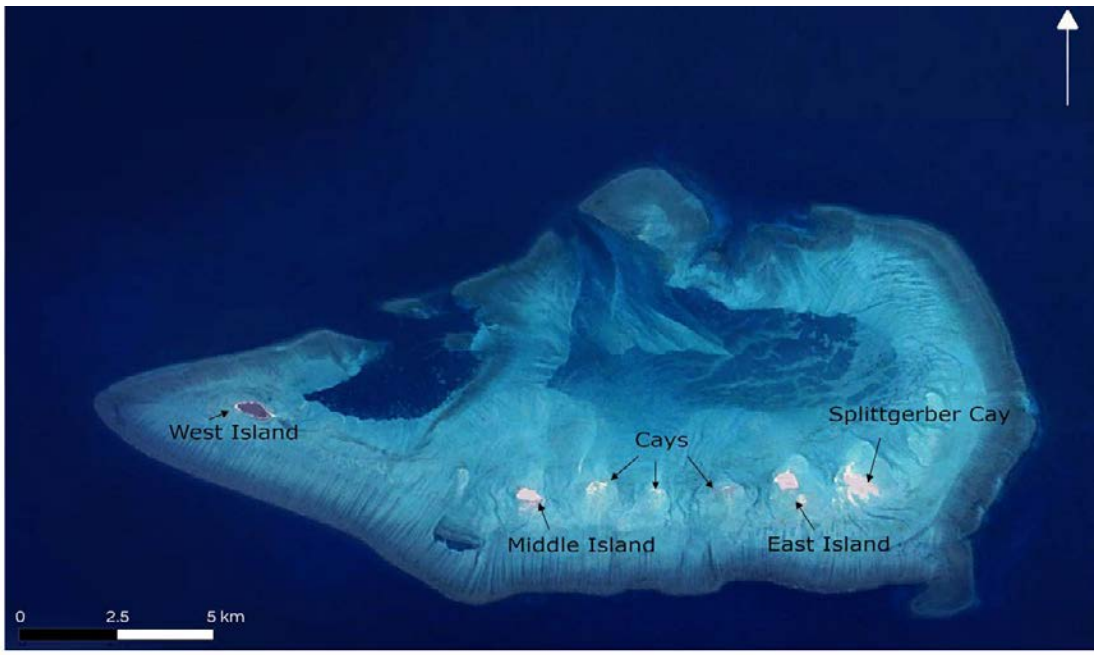


Figure 31. The location of West, Middle and East islands and the four cays within the Ashmore Reef Marine Park that were surveyed for seabirds, egrets, herons and shorebirds in May 2019

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.

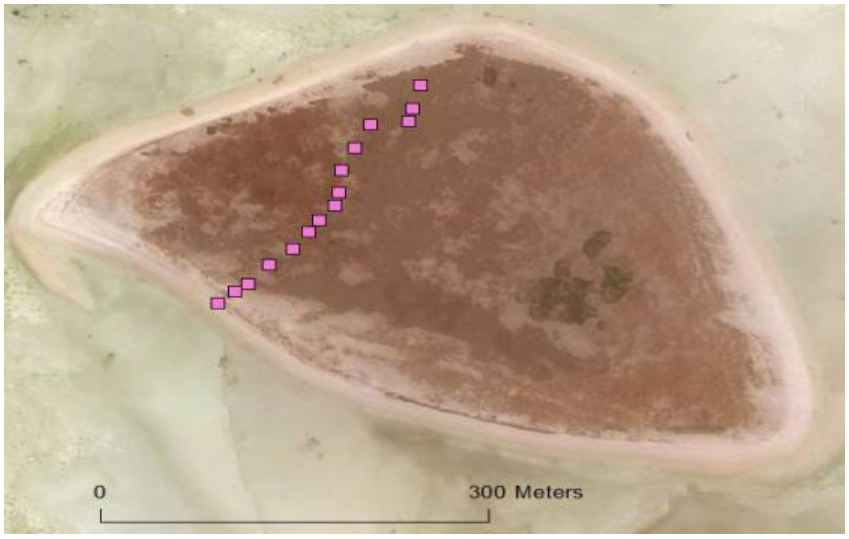


Figure 34. Location of the 15 plots (3 x 5 m) on East Island, Ashmore Reef during 2019 surveys

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).

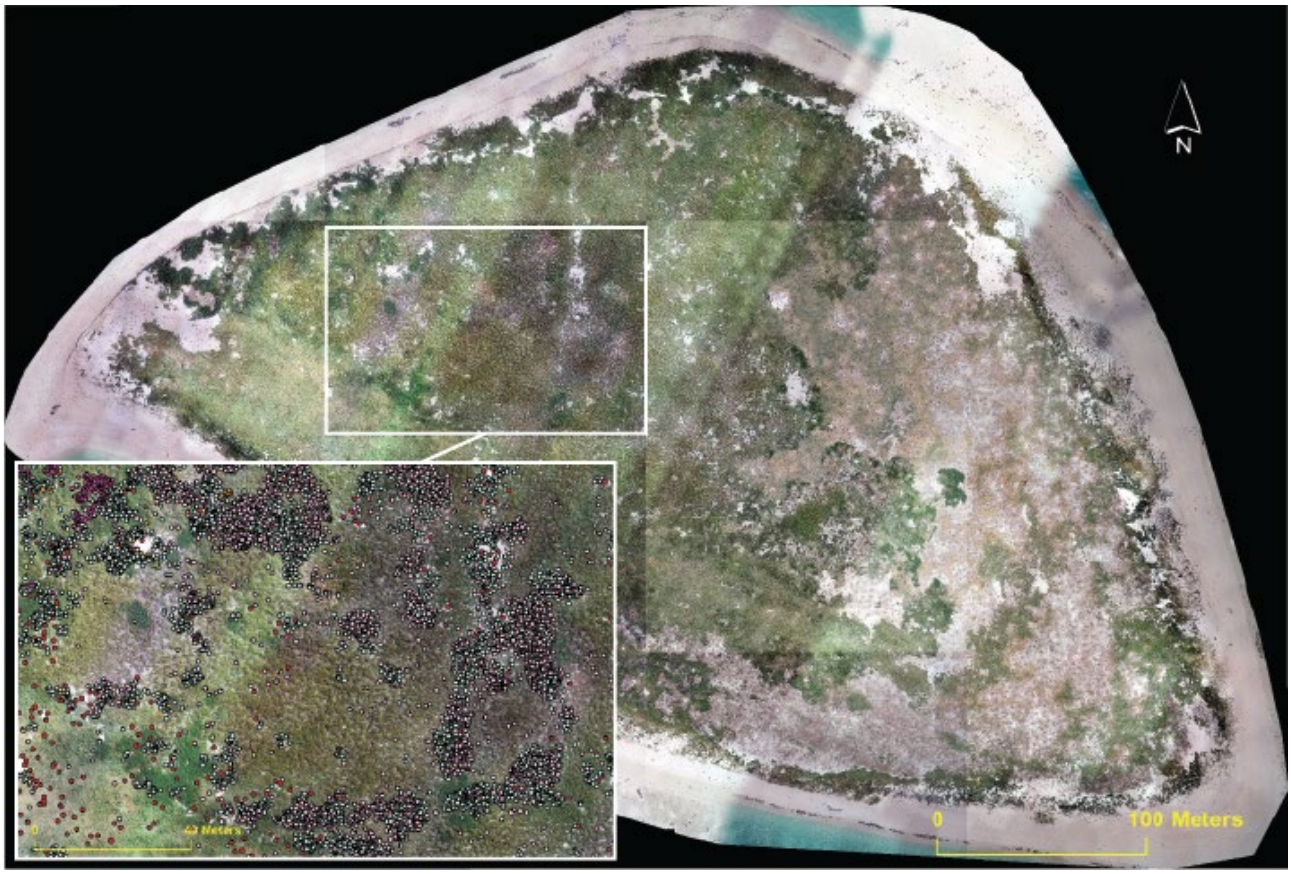


Figure 35. Map of East Island, Ashmore Reef, with inset of a section where Brown Noddies (aqua: n=4,952), Sooty Terns (light pink: n=4,151), Brown Boobies (red: n=228), Lesser Frigatebirds (dark pink: n=178) and Bridled Terns (orange: n=21) were identified in May

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.

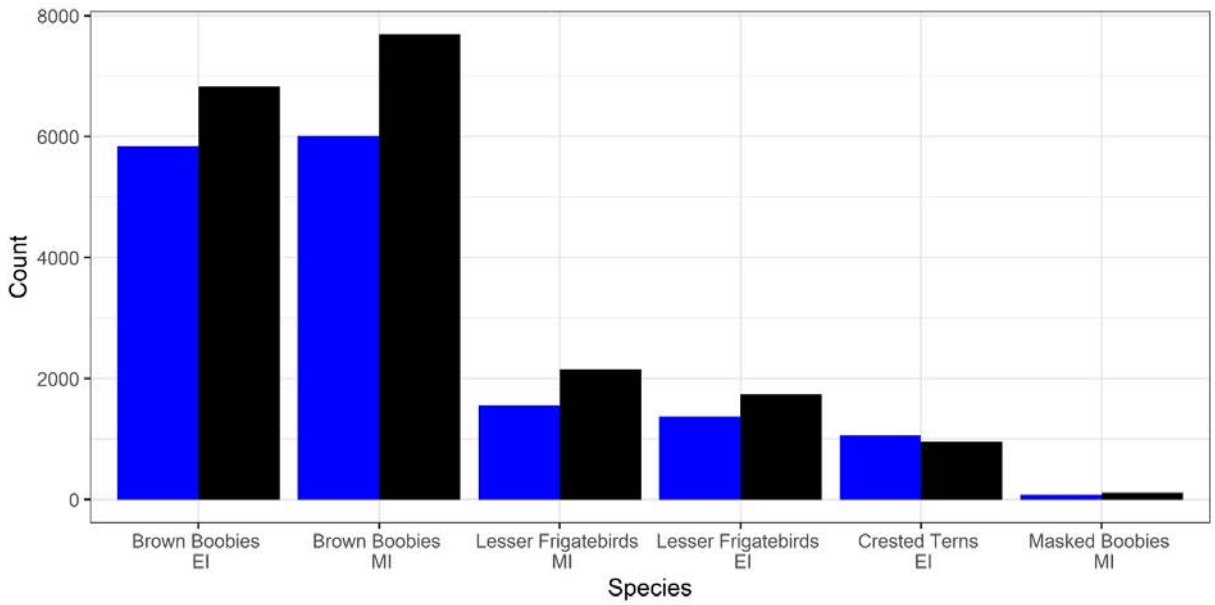


Figure 42. Comparison between ground (blue) and RPS (black) counts of seabirds on Middle and East islands, Ashmore Reef Marine Park, May 2019.

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)

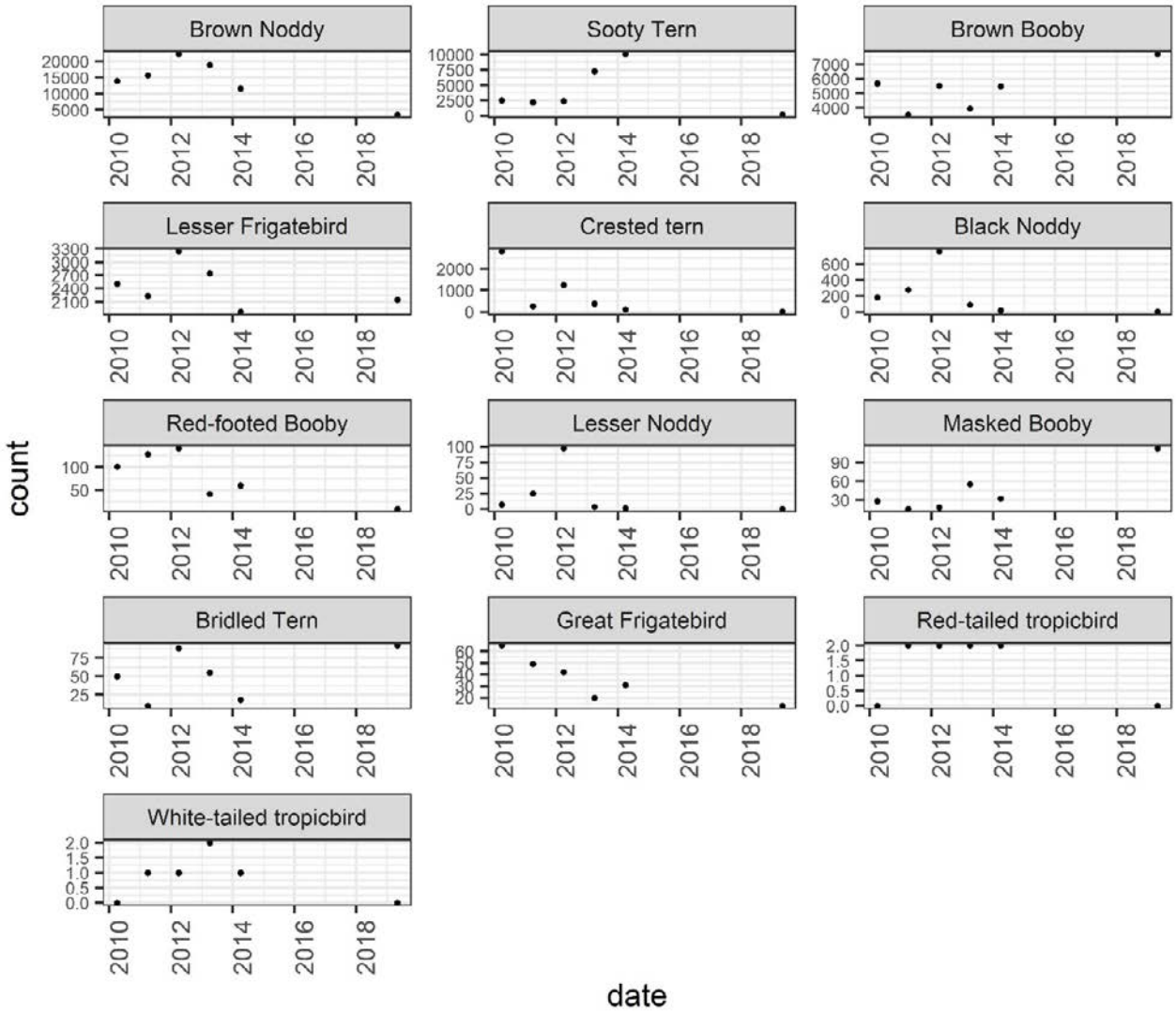


Figure 43. Counts of adults of each species observed on Middle Island, Ashmore Reef Marine Park, in April 2010-2014 (sourced from Clarke and Herrod 2016) and May 2019.

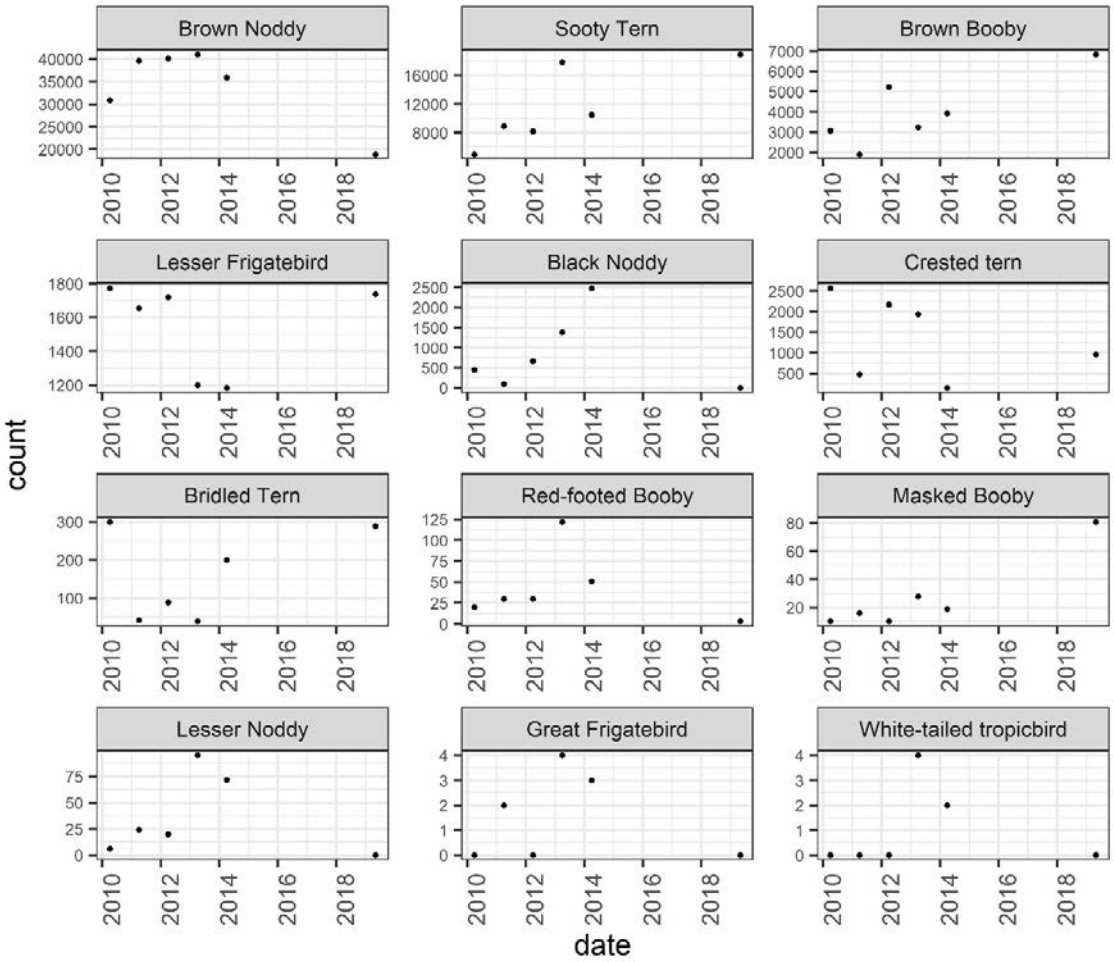


Figure 44. Counts of adults of each species observed on East Island, Ashmore Reef Marine Park, in April 2010-2014 (sourced from Clarke and Herrod 2016) and May 2019.

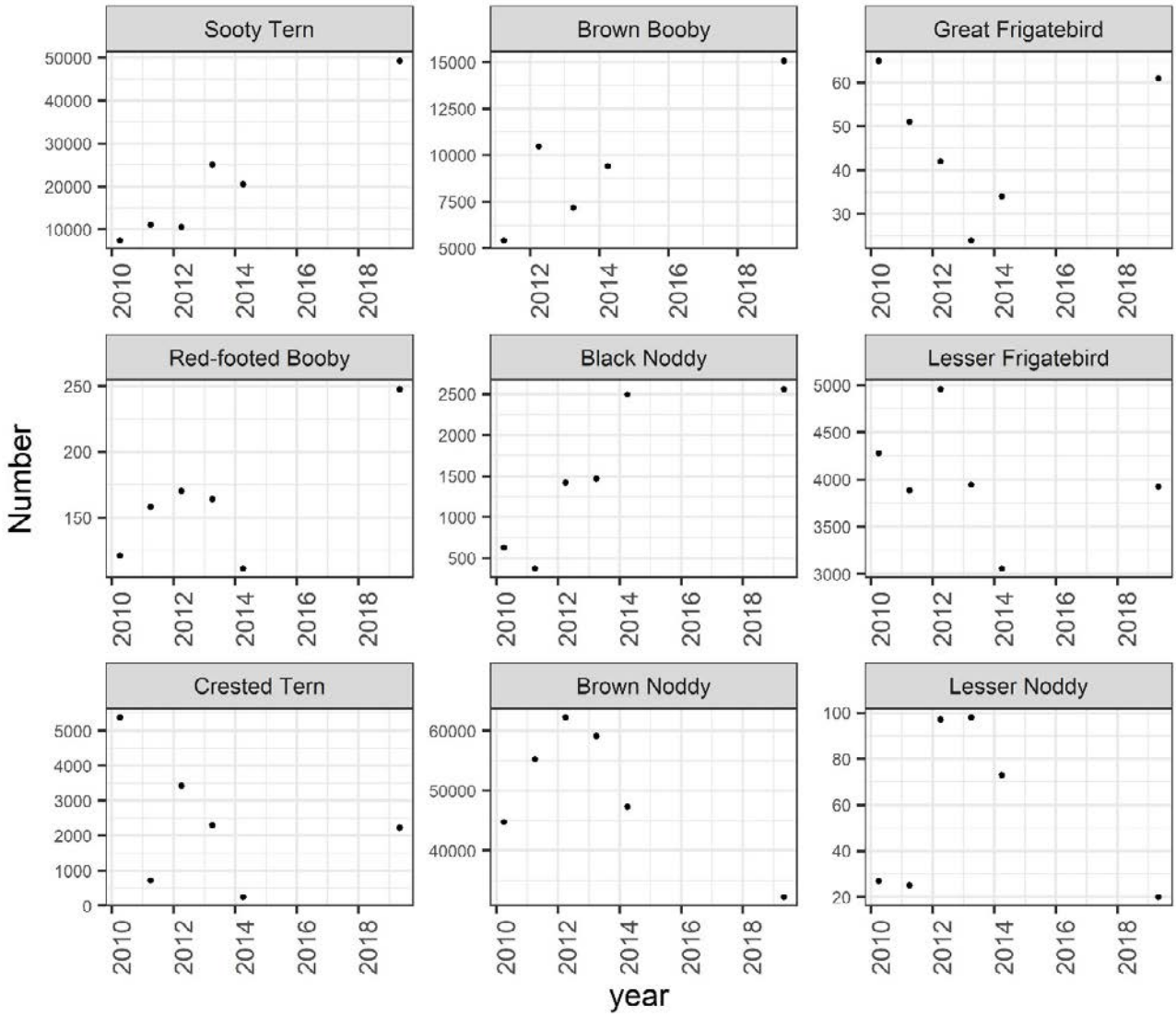


Figure 45. Apparent change in the numbers of adults of 9 seabird species observed across all islands and cays within Ashmore Reef Marine Park from 2010-2019. Data from 2010-2014 from Clarke and Herrod (2016). Note that these numbers are not population estimates, given that many of birds were breeding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

- *Complete a review of the conservation status of all seabirds in Australia*
- *Investigate the impacts of climate variability and change on seabirds and their habitats,*
- *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.

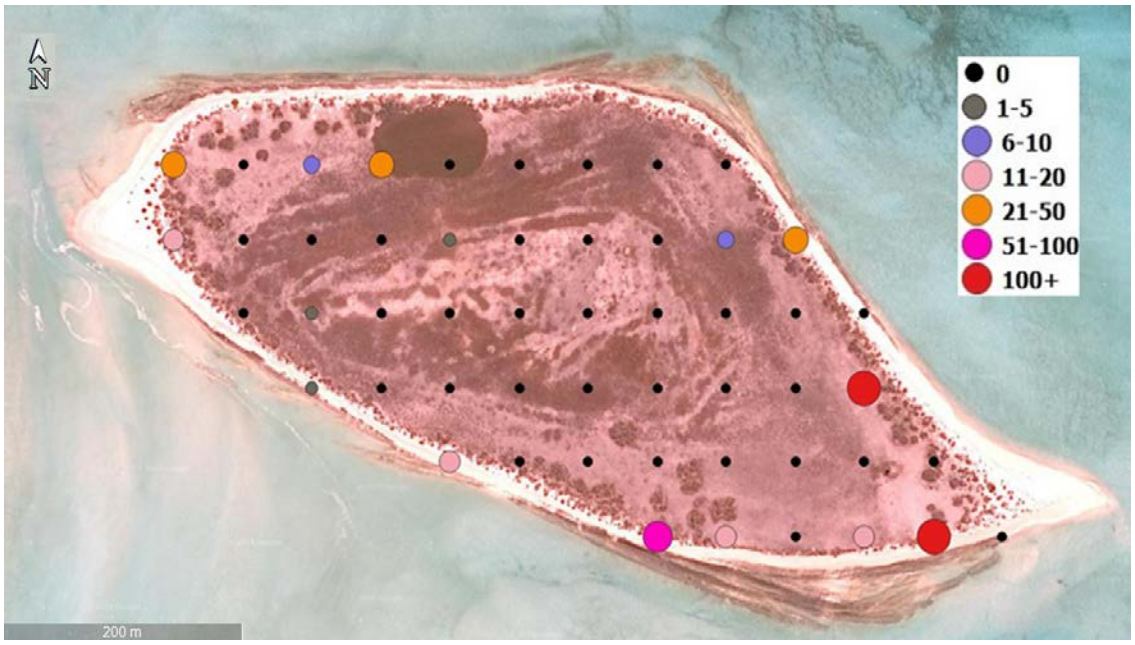


Figure 46. *Solenopsis geminata* abundance on West Island. Data are the average scored abundance of ants at four lures per location, displayed as actual ant abundance ranges

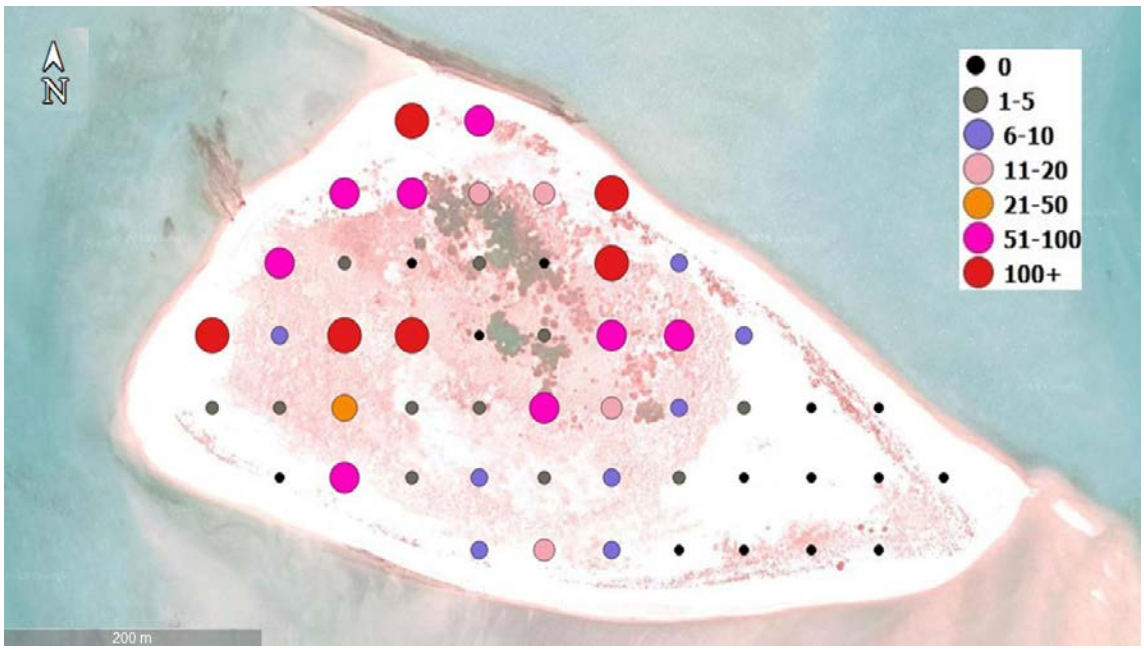


Figure 47. *Solenopsis geminata* abundance on Middle Island. Data are the average scored abundance of ants at four lures per location, displayed as actual ant abundance ranges.

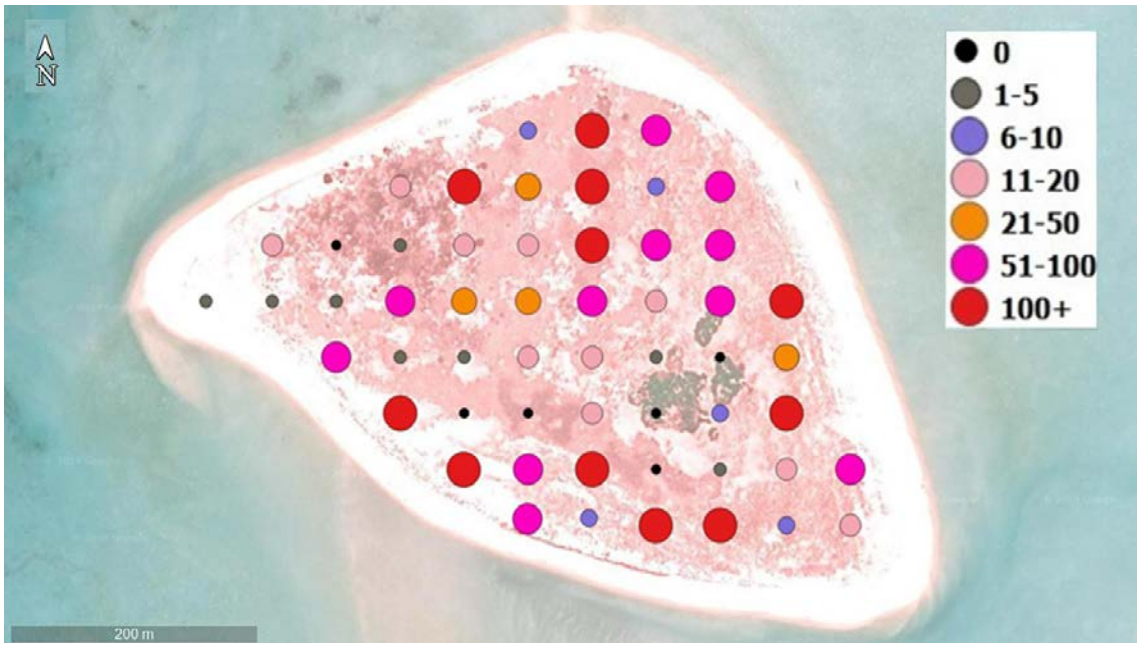


Figure 48. *Solenopsis geminata* abundance on East Island. Data are the average scored abundance of ants at four lures per location, displayed as actual ant abundance ranges

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.

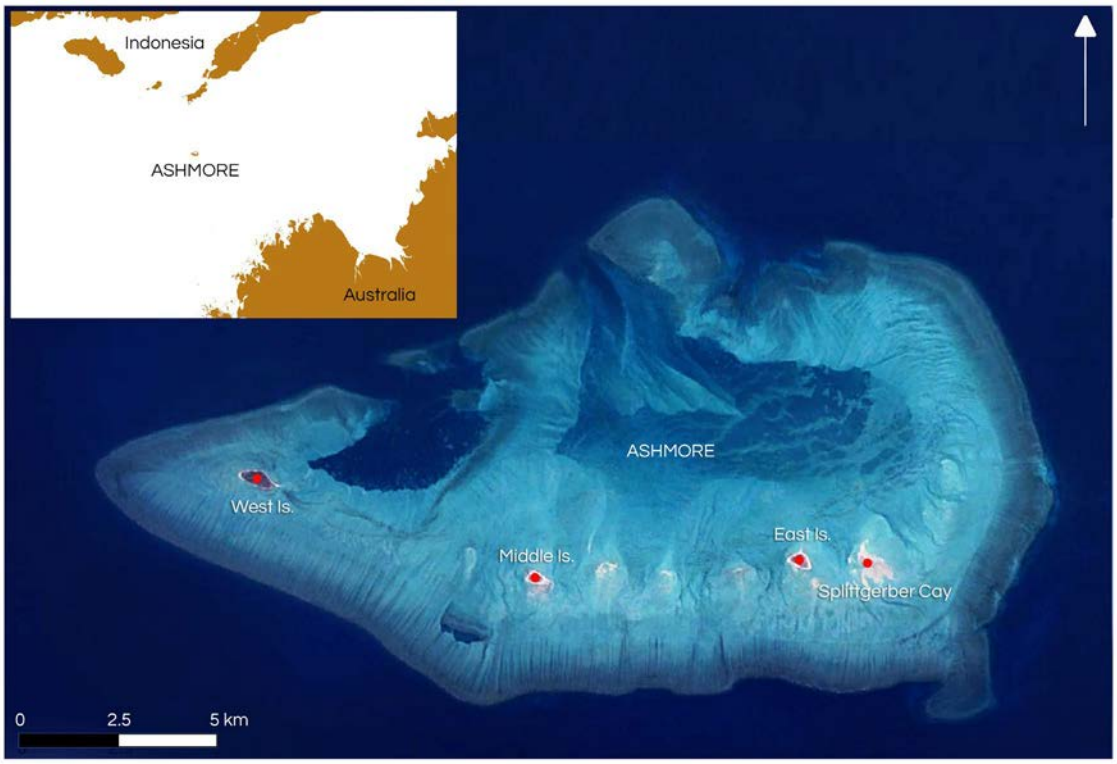


Figure 52. Location of Ashmore (insert) and islands surveyed for the *Hemidactylus frenatus* (red dots) at Ashmore Reef Marine Park in May 2019.

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.

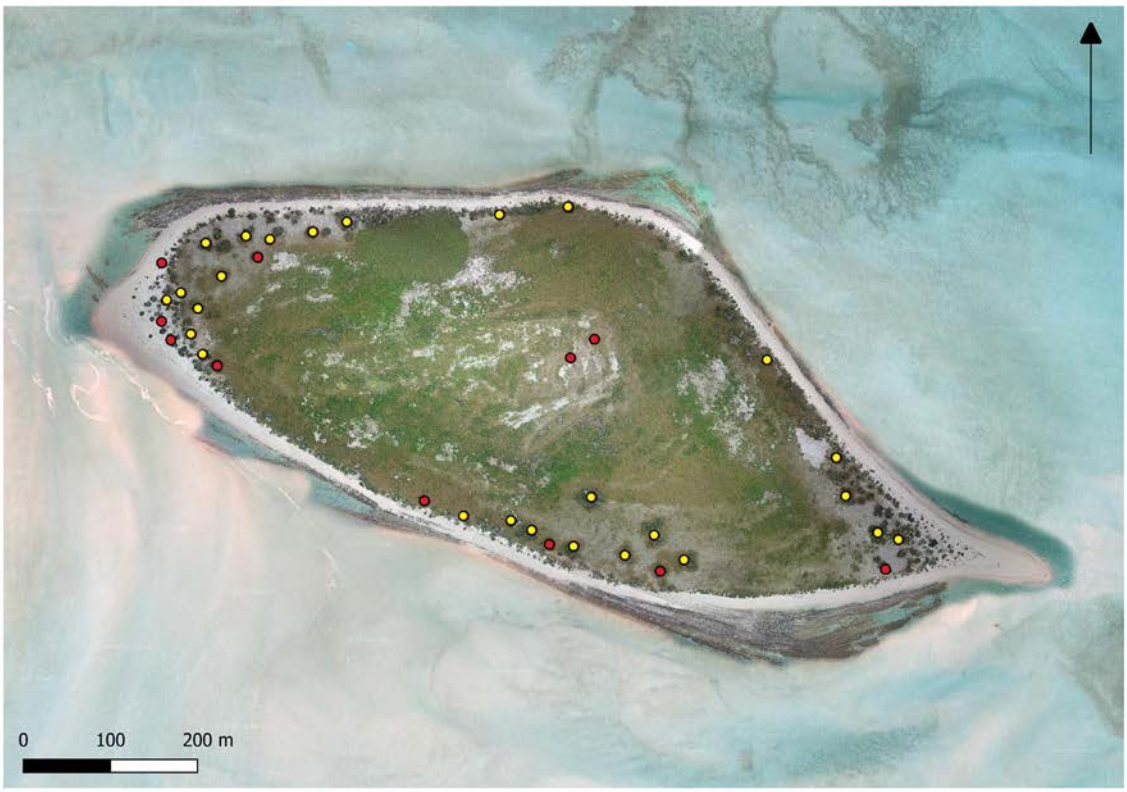


Figure 53. Showing *H. foertherianum* bushes sampled at West Island on 4 and 6 May (shrubs with geckos are shown in yellow)

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.

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