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Gorgon Dredge Offset Monitoring Evaluation and Reporting Project

Final Report

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Summary

In August 2009, the “Gorgon Gas Development Revised and Expanded Proposal: Barrow Island Nature Reserve” was approved by the Western Australian Government subject to a number of conditions and a commitment by the Gorgon Joint Venture participants to fund a series of “Additional Gorgon Joint Venture Undertakings”. One of these initiatives was the Dredging Audit and Surveillance Program, hereafter referred to as the Gorgon Monitoring Evaluating and Reporting (MER) program. Under this program, funding was provided to the Department of Biodiversity, Conservation and Attractions (DBCA) (then Department of Environment and Conservation) to allow auditing and surveillance of marine activities during dredging and marine construction, and ongoing auditing of the response and recovery of the marine environment.

To satisfy this requirement, field surveys were undertaken between November 2009 and May 2012 to monitor the changes to the marine environment (particularly coral and fish communities) relative to natural and anthropogenic pressures. Data was collected at monitoring sites established predominantly within the marine conservation reserves along a gradient of increasing distance north and south from the dredging and marine construction activities. The Gorgon MER was conducted in parallel with the industry compliance monitoring programs to enable compatibility between the studies. However, Gorgon MER used a range of methods and sampling procedures that are consistent with those deployed for monitoring coral reef and fish communities in other marine reserves. This allowed temporal and spatial assessment of marine communities both within the Montebello and Barrow islands marine conservation reserves (MBIMCR) and among other reserves. Moreover, sampling at high and medium impact dredging sites was not possible as this was the focus of compliance monitoring and the data presented here assesses the potential for dredge effects within areas of predicted low impact.

For both benthic and fish communities, digital imagery was collected periodically to provide a permanent record of the communities throughout this program that will complement ongoing monitoring of the region. These surveys provided baseline data on the spatial and temporal changes to coral and fish communities at each of the permanent monitoring sites across the marine communities associated with the Montebello and Barrow islands (MBI). Data on coral predators (crown of thorn starfish), seawater temperature anomalies, cyclones, natural sedimentation, and disease provide a background on environmental conditions, whilst measures of sediment plumes and sedimentation give an indication of how dredging effected the environment over the time that fish and coral communities were surveyed. The results presented in this report and associated publications provide useful information for the management and protection of the MBIMCR into the future in addition to understanding the potential impacts of identified anthropogenic pressures at other locations.

The results showed that a distinct plume first appeared 15 days after dredging started and moved predominantly southward. The plume overlapped with the original dredge plume distribution modelling (278.87 km^2), however a further 454.89 km^2 of plume was

outside the modelled area whilst northern areas of the modeled distribution were rarely covered with dredge-related plume. The area of consistent plume exposure, as described by the hotspot analysis, was limited to the port zone <10km from the dredge location, however, the plume was recorded on images up to 35km to the south of the plume along the eastern reefs of the Barrow Shoals. There was also a notable ‘tail’ of plume originating from the spoil ground. Hotspot analysis completed on the cumulative plume exposure showed that Gorgon MER coral reef monitoring sites were covered by the plume from 0 to 296 of the 411 days observed during the dredging activities.

Sediment deposition in traps was measured at eight Gorgon MER monitoring sites (four each north and south of the dredging) along a gradient of increasing distance from the dredging activity. During dredging, a higher proportion of fine grained, silty sediments were detected at sites closer to the dredging operation, and higher proportion of sandy sediments at sites further from the dredge. Sites north of the dredging recorded a higher proportion of coarser sediments than those to the south, as northern sites are more exposed to high energy wave environments. A strong negative correlation ($r = >-0.9$) was detected between the total weight of smaller grain sizes with distance from the dredging activity in a southerly direction.

In the austral summer of 2011, a high water-temperature anomaly was recorded from loggers deployed at seven monitoring sites. Seawater temperature at the MBI increased ~2^{o}C above the long-term average summer temperature, although the maximum degree heating weeks (4-8 Degree Heat Weeks) was below that considered critical for extensive coral mortality. Seawater temperature was highest at sites in the southern Barrow Shoals and decreased to the north along the eastern fringes of the Barrow and Lowendal shoals.

There were significant spatial differences in benthic communities, coral composition, and cover among Gorgon MER sites. Temporally, the benthic measures fluctuated, however, these changes were not statistically significant for most sites, and there was no definitive effect of dredging or thermal stress on benthic communities at our monitoring sites. Of the environmental factors considered in this study we found depth to be the most important determinate of coral community composition.

Similarly, there was no definitive evidence that dredging or coral bleaching affected fish assemblages at monitoring sites. Fish genera typically targeted by fishers had declined at some MBI sites since initial surveys in 2009, particularly those of the genera *Choerodon*. Sites with the highest fishery target densities in 2009 and 2010 were quite low at the end of the study in 2012, whilst fisheries non-target genera and feeding guilds remained relatively stable over the study period. Spatial variation in the benthos, particularly rubble and coral from the genera *Acropora*, best explained spatial variation in fish assemblages suggesting site habitat characteristics had more of an effect on fish than exposure to dredge plume or heat stress during the study.

In addition to the above findings, collaborations with external research partners have led to 15 research publications (plus two in review at time of publication) investigating: large-scale patterns in coral distribution and abundance due to ecological drivers (Speed *et al.* 2013, Zinke *et al.* 2018), including impacts of coral bleaching (Moore *et al.* 2012, Babcock *et al*. in review, Gilmour *et al* 2019); an understanding of macroalgal fish habitats (Evans *et al.* 2012) and crown-of-thorn starfish populations (Haywood *et al*. accepted); and methods for monitoring sea surface temperature (Baldock *et al.* 2014), benthic assemblages (Turner *et al.* 2015, Bennett *et al.* 2016), coral condition (Josephitis *et al.* 2012; Pollock *et al.* 2014, Page *et al.* 2016), and dredge plumes (Evans *et al.* 2012). Review articles documenting the impacts of dredging on fish (Wenger *et al.* 2017, 2018) and invertebrates, seagrasses and macroalgae (Fraser *et al.* 2017) have also been published.

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    9a. Pollock FJ, Lamb JB, Field SN, Heron SF, Schaffelke B, Shedrawi G, Bourne DG, Willis BL (2016) Correction: Sediment and Turbidity Associated with Offshore Dredging Increase Coral Disease Prevalence on Nearby Reefs. PLoS ONE 11:e0165541.

11. Speed CW, Babcock RC, Bancroft KP, Beckley LE, Bellchambers LM, Depczynski M, Field SN, Friedman KJ, Gilmour JP, Hobbs J-PA, Kobryn HT, Moore JAY, Nutt CD, Shedrawi G, Thomson DP, Wilson SK (2013) Dynamic Stability of Coral Reefs on the West Australian Coast. PLoS ONE 8:e69863.

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1 Background

The Montebello and Barrow islands (MBI) are a complex archipelago of >300 islands and islets situated approximately 1,600km north of Perth, Western Australia (WA), in the Pilbara Offshore marine bioregion (IMCRA, 2006). The sub-tidal reef environments adjacent to the MBI include the extensive shoals to the south of Barrow Island, within 25km of the Australian mainland, and the shallow fringing reefs which surround the Montebello Islands (Fig. 1.1). The MBI have expansive intertidal reef flats and shallow pavements which progress to deeper sands offshore. Nearshore limestone or calcarenite pavements are variably covered by sand, gravel and coral (DEC 2007). These pavements allow extensive growth of macroalgal fields, which is the dominant habitat forming benthic cover along with coral reefs. The oceanographic and geomorphological conditions surrounding these remote islands create a diverse range of marine habitats that support a variety of both widely distributed and endemic marine species (DEC 2007). The marine flora and fauna of the MBI are predominantly tropical (DEC 2007) and oceanographic modelling suggests that the MBI are likely to be well connected to coral reefs to the north and adjacent mainland (Feng *et al.* 2016).

The shoals and reefs of the MBI have historically been protected from anthropogenic pressures due to their geographic isolation. Evidence of indigenous occupation until about 5000BC suggests use of marine and coastal resources followed by a period of forced abandonment due to rising sea levels (Veth 1993). Early European explorers briefly visited the islands in the 17th and 18th centuries before a long period of pearl shell harvesting and cultivation from the mid-19th century capitalising on the sheltered clean waters and strong tidal currents (Stanbury 1994). In the 1950s the Montebello Islands saw the detonation of three nuclear devices on Alpha and Trimouille Islands and aboard a vessel anchored off Trimouille Island (Child and Hotchkis 2013). In recent times there has been an increase in commercial and recreational tourism which, like the pearl culture has remained primarily associated with the sheltered waters of the Montebello Islands compared with wave exposed reefs surrounding Barrow Island. Oil and gas facilities have been on Barrow Island since the 1960s and Varanus Island, the largest island of the Lowendal group to the north east of Barrow Island, since 1987. Although largely island based, these facilities have coastal and marine developments which may have impacted the marine environment and the associated wildlife.

Several natural pressures have also been identified as important influences on tropical marine communities (Simpson *et al.* 2015) including thermal stress (Moore 2012, Lafratta *et al.* 2016, Ridgway *et al.* 2016), physical damage from cyclones (Harmelin-Vivien 1994), coral predation (Kayal *et al.* 2012) and coral disease (Haapkylä *et al.* 2013). Combinations of these pressures have the potential to interact and have synergistic effects (Haapkylä *et al.* 2013), though in some cases simultaneous occurrence of disturbances may diminish overall pressure and impact on marine communities (Carrigan and Puotinen 2011). Moreover, predicted increases in the regularity and severity of pressures associated with climate change (Hoegh-Guldberg 1999, Webster *et al.* 2005, Van Hooidonk *et al.* 2016) increases the potential for change in marine communities of the MBI.

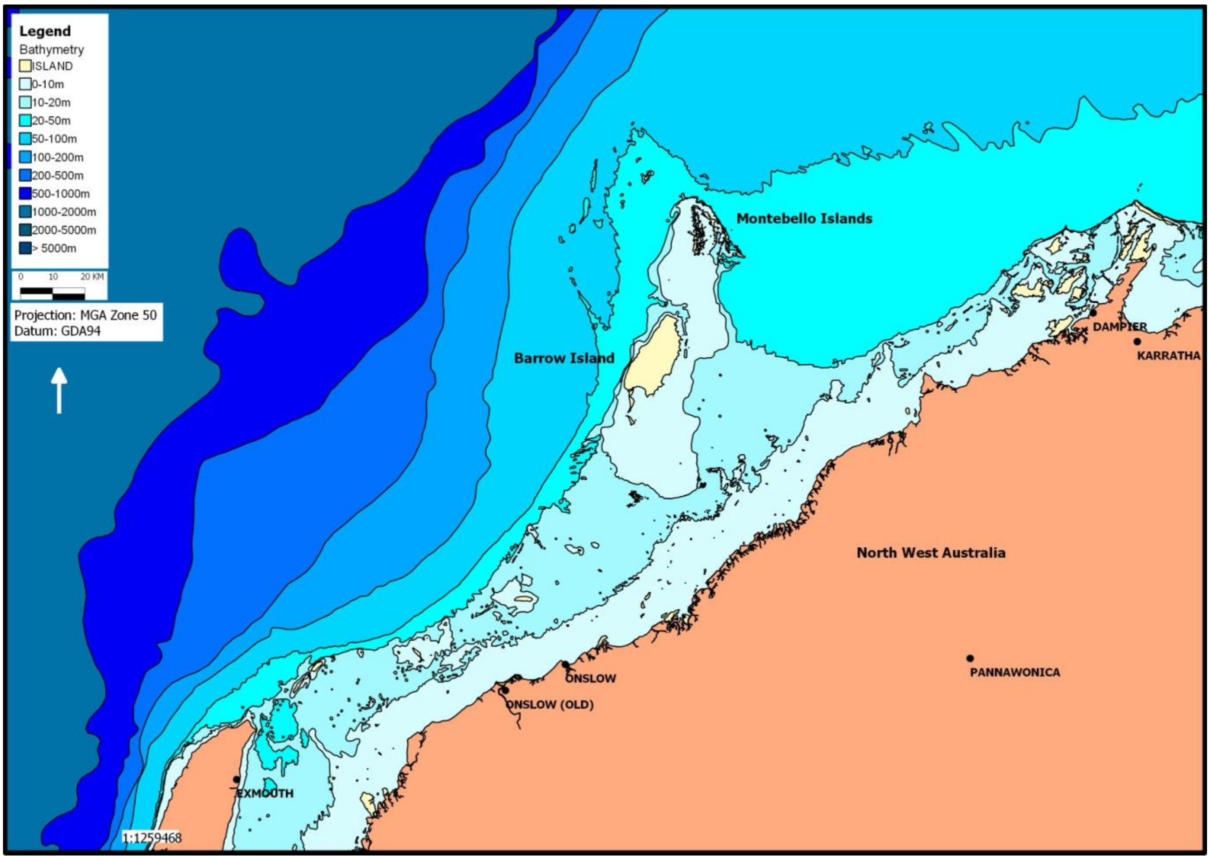

Figure 1.1 Map of Montebello and Barrow islands showing the location of the islands relative to the mainland and the bathymetry of the region.

Recreational and commercial use of the MBI are managed to reduce any related impacts to the islands and the surrounding marine communities. The potential anthropogenic pressures identified for the area include sedimentation, extractive activities (oil and gas, and fishing), pollution events, introduction of marine pests and direct physical impact (Simpson *et al.* 2015). Major industrial and port developments have occurred at the MBI (Lowendal and Barrow islands) since the 1960s in addition to industrial and residential developments at adjacent coastal centres like Exmouth, Dampier, Karratha and Onslow over the past 20 years. These developments have increased the local population and raised awareness of the MBI, increasing commercial tourism and recreational visitation to the islands. These combined factors have markedly increased the potential for human activities to cause impact on the marine environment of the MBI over recent decades. This increase in pressure and potential impact on natural marine communities of the MBI highlights the need for an improved understanding of these communities and provide scientific advice to support their long-term management.

The Montebello and Barrow islands marine conservation reserves (MBIMCR), comprising the Montebello Islands Marine Park, Barrow Island Marine Park and the

Barrow Island Marine Management Area, were created in 2004 to provide a structure for the conservation of the marine communities of the MBI. The management objectives and strategies for these reserves are expressed in the management plan (DEC 2007)

In August 2009, the Western Australian Government approved the Gorgon Gas Development Revised and Expanded Proposal: Barrow Island Nature Reserve. Based on Barrow Island, the Gorgon Project is one of the world's largest natural gas projects and the largest single natural gas project in Australia. The plant includes three 5M tonne/annum LNG trains, with domestic gas piped to the mainland and a 4km long loading jetty for international shipping. The Gorgon Project is subject to a number of Ministerial conditions and a commitment by the Gorgon Joint Venture participants to fund a series of “Additional Gorgon Joint Venture Undertakings”. One of these initiatives was the Dredging Audit and Surveillance Program. This Program enabled the Department of Biodiversity, Conservation and Attractions (DBCA) to undertake auditing and surveillance of marine communities during dredging and marine construction activities, including the response and recovery of the marine environment following the completion of the dredging campaign.

The dredging and related spoil disposal campaign ran from May 2010 to November 2011 and involved the removal and dumping of over 7.5M tonnes of marine sediment. Of the 7.5M tonnes of marine sediment removed during dredging, 1.1M tonnes were removed from the Materials Offloading Facility (MOF) access site while 6.9M tonnes were removed from the Liquid Natural Gas (LNG) shipping terminal turning basin and channel (Fig. 1.2). Of the material removed, 1.6M tonnes were used in the MOF island construction while the remaining 6.4M tonnes were dumped within the allocated spoil disposal ground (Fig. 1.2). Sediment plumes resulting from the dredging campaign were monitored from three origins: sites of material removal, the MOF island construction and spoil disposal in deeper water.

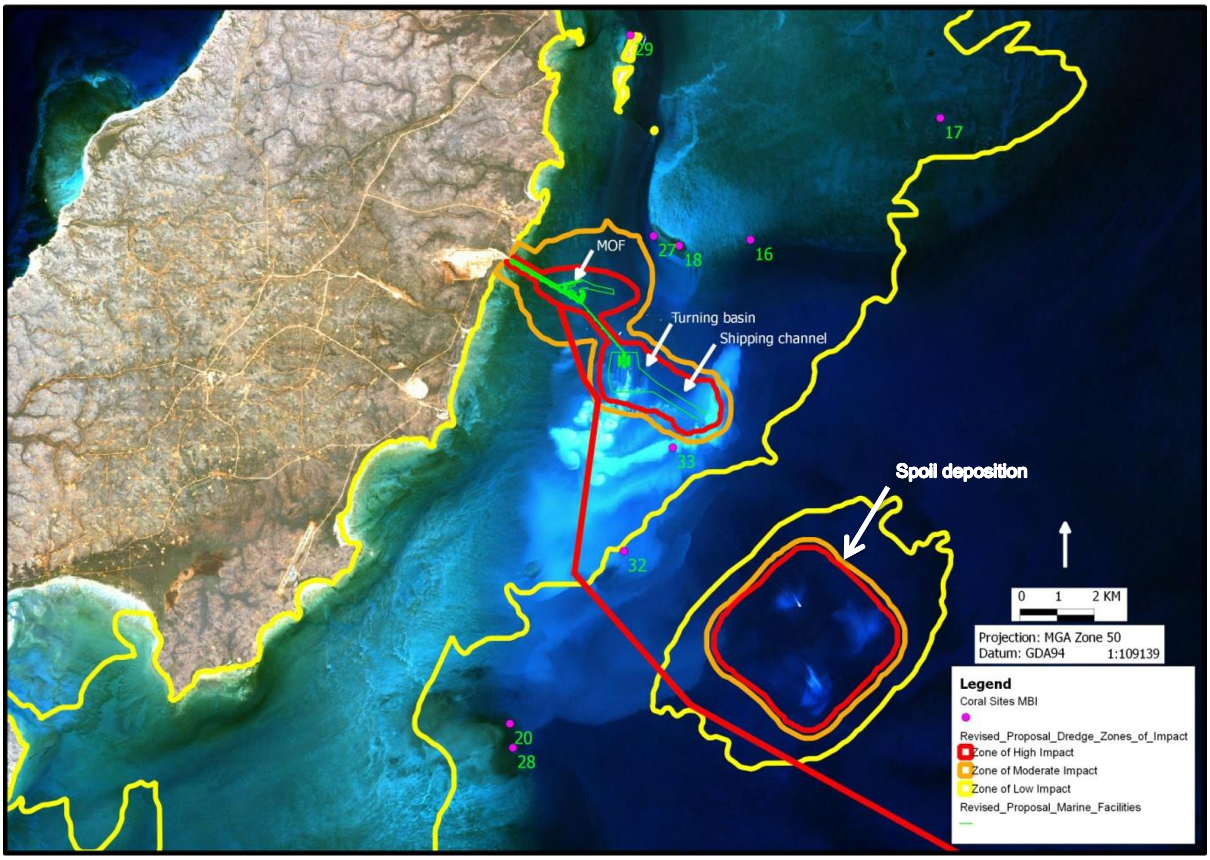

Figure 1.2 Map indicating areas of dredge material removal, including the Marine Offloading Facility (MOF) and liquid natural gas vessel turning basin areas; and spoil deposition offshore from the development. Gorgon MER sites are shown.

The Gorgon Monitoring, Evaluation and Reporting Program (Gorgon MER) was established for several reasons:

- To monitor the responses of key biophysical assets to anthropogenic pressures in the area managed by the Department of Biodiversity, Conservation & Attractions (DEC 2007). In particular, the program was developed to investigate the potential ecological impacts of the dredging and dumping activities on selected marine communities within the MBIMCR (i.e. the 'far-field') whilst also considering the suite of natural, anthropogenic and climate change pressures that have the potential to affect these marine communities.
- To inform future environmental impact assessments (EIA) by improving predictions of the spatial scale and nature of the likely impacts of dredging and dumping activities on sensitive marine communities.
- To increase the knowledge of marine communities in the MBIMCR.
- To complement the work of the environmental compliance and monitoring programs completed by Chevron Australia.

In December 2006, 26 long-term monitoring sites were established at locations of high coral cover within five of the seven identified environmental and geomorphic sectors in the MBIMCR as a component of the DBCA Marine Monitoring Program (Bancroft

2009). The sites were primarily established to provide baseline data to assess spatial and temporal change in benthic coral assemblages. In November 2009, 18 Gorgon MER monitoring sites, including 13 of the 26 long-term monitoring sites and an additional five new sites, were established. These sites were predominantly within the marine conservation reserves and were located along a gradient of increasing distance (both north and south) from the dredging activities (Table 1.1, Fig. 1.3).

*Table 1.1. Details of each of the 18 Gorgon MER monitoring sites at the Montebello and Barrow islands including: 'Distance' from the dredging, measured from the turning basin, the approximate centre of the dredging activity, 'Site rugosity', a relative measure 1 – 5 of least to greatest rugosity following Polunin and Roberts (1993) and 'Depth', the average depth of the site - shallow (<2m), moderate (2-5m) and deep sites (5-10m). * transects not permanently marked with star pickets.*

Site	Location	Distance (km)	Site Rugosity (1-5)	Depth
5*	North Bunsen Channel	50	3	S
8	Stephensons Channel	39	2	M
11	W Ah Chong Is.	35	3	D
12	E Black Rock Is.	30	5	D
14*	N Varanus Is.	25	4	D
15*	S Varanus Is.	16	4	D
16	SE Lowendal	4	4	D
17	E Lowendal	10	4	D
18	S Lowendal	4	1	S
19	Wonnich Reef Flat	35	1	S
20	E Dugong Reef	10	4	M
23	S Batman Reef	16.25	3	M
26	Central E Barrow Is. Shoals	30	2	M
27	W Lowendal Shelf	4	3	M
28	SE Dugong Reef	10	3	M
29	N Double Island	9	3	M
31	Central E Barrow Is. Shoals	26.25	3	M
32*	LNG3	6.5	4	M

Coral and finfish communities are identified as key performance indicators in the MBIMCR management plan (DEC 2007) and were the primary focus of the Gorgon MER project. 'Coral community' in the context of this work is defined as the hermatypic coral component of benthic sessile communities. Finfish communities, in the context of this project, are defined as targeted and non-targeted reef fish assemblages associated with high relief shallow benthic habitats that are being monitored at permanent coral community Gorgon MER sites at the MBI. Finfish communities did not include pelagic or cryptic species.

Also included in this report is an appendix that includes the baseline monitoring of macroinvertebrates inhabiting the shallow marine environments of the MBI.

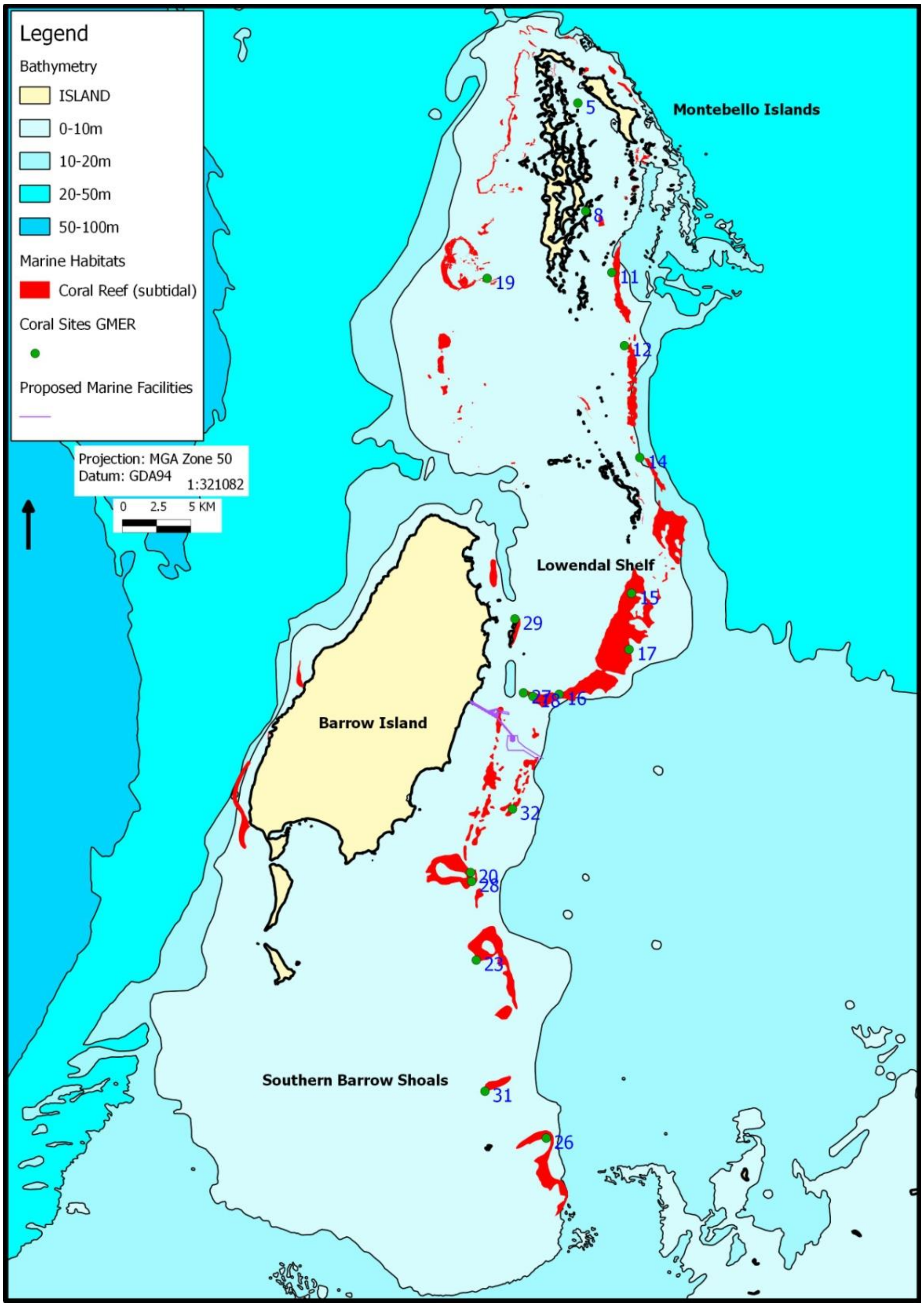

Figure 1.3 Map of the Montebello and Barrow islands, showing the 18 monitoring sites for the Gorgon MER.

2 Spatial and temporal variability of remotely sensed dredge plume extent in relation to sediment deposition, wave and cyclone data.

2.1 Introduction

Australia's coastal environments are under increased pressure from the development of large resource extraction projects and associated port infrastructure. Many of these developments are in tropical or semi-tropical regions and are adjacent to biodiverse coral reefs of high conservation value (Hoeksema 2007). Developments in these areas often involve the excavation, removal and dumping of considerable amounts of marine benthos, which can result in the suspension of large amounts of sediment in the water column. In the north-west of WA approvals have been granted for five large port projects that will collectively require dredging of 163M tonnes of marine sediment (Hanley 2011).

The suspension of sediments due to activity like dredging pose potential threats to surrounding ecosystems (reviewed by Erftemeijer *et al.* 2012), such as physical smothering of benthic communities (Dikou and van Woesik 2006; Jones *et al.* 2016; Smith and Rule 2001), and decreased light availability which limits growth and metabolism of sea grass (Erftemeijer and Robin Lewis III 2006, Strydom *et al.* 2017), sponges (Pineda *et al*. 2017), algae (Lyngby and Mortensen 1996) and coral (Babcock and Davies 1991; Bak 1978; Gilmour 1999; Vargas-Ángel *et al.* 2007), as well as inhibiting reproductive output (Ricardo *et al,* 2015, 2016) and settlement (Ricardo *et al.* 2017) processes in coral. Fine particles from dredge material may also reduce feeding and survival of larval fish important for recreational and commercial fisheries (Partridge and Michael 2010; Wenger *et al*. 2017). The severity of any impact will however depend on the extent and type of sediment released into the marine environment (Erftemeijer *et al*. 2012; Wenger *et al.* 2015), which requires an understanding of the inherent sediment loading dynamics. A quantitative assessment of natural and anthropogenic derived sediment is therefore required if investigating the influence of dredging on marine flora and fauna.

The Gorgon dredging and the related spoil disposal campaign ran from May 2010 to November 2011 and involved the removal and dumping of over 7.5M tonnes of marine sediment. Of the 7.5M tonne total, 1.1M tonnes were removed from the materials offloading facility (MOF) access site while 6.9M tonnes were removed from the Liquid Natural Gas (LNG) shipping terminal turning basin site and channel (See Chapter 1, Fig. 1.2). Of the material removed, 1.6M tonnes were used in the MOF island construction while the remaining 6.4M tonnes were dumped at the designated spoil disposal ground (See Chapter 1, Fig. 1.2). Sediment plumes resulting from the dredging campaign were most prominent within 3km of capital dredging (Fisher *et al.* 2015) with smaller plumes emanating from the MOF island construction site leaching from the bund walls and from the site of spoil disposal in deeper water.

This chapter presents the data collected on the spatial and temporal extent of suspended sediment plumes associated with the Gorgon project's dredging campaign as compared with the background conditions prior to and following the completion of dredging. Several additional natural and anthropogenic pressures were also monitored to understand the influence of these pressures on the extent of the suspended sediment plume and any detected changes to coral and fish communities surveyed in later chapters.

2.2 Methods

2.2.1 Location of study sites

Eighteen sites were selected for monitoring fish and coral communities for the Gorgon MER project (Fig. 2.7). Site locations were chosen to monitor the far field impacts of anthropogenic pressures (particularly dredging) while quantifying a range of other physical factors (i.e. natural turbidity, wave height *etc*) that may influence changes in the marine communities of the MBIMCR. The variables measured included extent of dredge plume coverage, sediment deposition in sediment traps, modelled wave activity, cyclone activity and thermal water events (see Chapter 3).

2.2.2 Physical and oceanographic data

Wave conditions

Differences in wave conditions at sites were determined using data obtained from the Australian Bureau of Meteorology (BoM) AUSWAVE-A product (Durrant and Greenslade 2011) and validated by correlation with *in situ* data from a three-month deployment of an Acoustic Wave and Current (AWAC) meter at site 32, adjacent to the dredging activity. AUSWAVE data are modelled on operational runs of surface wind data from the Australian Community Climate and Earth-System Simulator and output four-times daily. Gridded time-series wave data were extracted by coordinates from a subset of eight of the 18 Gorgon MER sites (26,23,20,30,16,17,14,11) and the resulting data was displayed as daily averages and maxima. The subset of sites was selected to represent the spatial distribution of Gorgon MER sites both north and south of the dredging to indicate the spatial variability of wave energy and its potential to influence sediment plume and deposition among sites.

Cyclones

Annual records of cyclone tracks extracted from BoM were also presented, as site-level wave data is likely to incorporate storm-driven wave activity. Thus, cyclone data is used to aid the interpretation of both the modelled wave and recorded sediment deposition data but is not considered in statistical models.

2.2.3 Suspended sediment monitoring

Suspended sediment was monitored to characterise the spatial extent of the sediment plume from dredging and spoil dumping. Remotely-sensed imagery was used to describe the extent of sediment plumes and sediment traps were employed to measure the rates of sediment deposition at selected Gorgon MER sites.

Mapping Plume Extent

The presence of background turbidity levels and the appearance of a plume from the dredging and spoil disposal activity was monitored using remotely sensed methods. Monitoring was carried out for 12 months prior to dredging to capture background natural turbidity events, for the 18 month duration of the dredging program (19 May 2010 to 7 November 2012) and for an additional four months after dredging finished. It should be noted that no plume was observed until 1 June 2010. The sediment plume boundaries were mapped by digitising the spatial extent of the dredge plume visible on daily MODIS true colour mosaic (250m pixel) images (Evans *et al.* 2012). This provided a cost-effective and relatively accurate method to map the spatial extent of plume presence and provided an understanding of how frequently the plume covered each Gorgon MER monitoring site.

Of the images downloaded, 267 days of a possible 365 were suitable prior to the dredging, 411 days of a possible 538 were suitable during the dredging and 65 days of a possible 114 were suitable post-dredging where the plume was observed and digitised from MODIS Aqua or Terra satellites. Data were not available on the remaining days due to cloud cover and/or incomplete satellite image coverage. In addition to the MODIS imagery, imagery of higher resolution (10m pixel) was also captured to clearly identify the coastline and subsurface structure. Two captures of ALOS AVNIR-2 before (18 November 2006 and 23 November 2008) and one during (29 August 2010) the dredging project were captured and when available, cloud free Landsat 5 TM and 7 ETM+ images were acquired and visually enhanced specifically for the water around Barrow Island to provide the observer with a greater ability to discern the location of subsurface structures amongst from the dredge plume.

Once acquired, MODIS imagery were re-projected into GDA94 MGA zone 50 and displayed in QGIS, a spatial viewer with limited functionality maintained by DBCA. The dredging plume boundary was interpreted by manually digitising (drawing) a vector (digital polygon) around the plume at a scale of 1:450000. This was the optimal scale for digitising the area with 250m by 250m pixels resolution of MODIS. A new polygon was created for each visible plume, and then attributed with the date (Julian day) of the MODIS image being interpreted. A strict file structure and naming convention was employed to aid the quick generation of “clean” datasets that were easy to display when required for quality assurance. Some MODIS images had no plumes and others had more than one plume derived from multiple dredge and spoil disposal locations.

Interpretation

To ensure consistent interpretation of plume boundaries, a single observer followed a set of guidelines (Table 2.1) and was responsible for interpreting all images used in the analyses.

Table 2.1 Plume digitising guidelines

1. Satellite imagery captured on plume-free days under similar tidal and meteorological conditions was utilised as a standard against which to identify dredge and spoil disposal generated plumes. Multiple resolution images (MODIS, Landsat and ALOS) of the same location without a plume were used to identify shorelines and subsurface natural substrate features.

2. Plume areas were only included when the observer had complete confidence that a plume was present (i.e. not reef or bottom features).

3. Based on plume location they were identified as being from one of three potential sources – MOF (both extractive and land reclamation activities), LNG turning circle (extractive activities) and the spoil ground (deposition)

4. Plumes were digitised separately for each location unless there was no discriminating between them in which case one digitised vector was drawn.

Hotspot analysis

A hotspot analysis was run on the cumulative daily digitised plume boundaries to provide a dataset describing the number of days the plume was present at any position within the Barrow Island Marine Management Area and surrounds (Modarres and Patil 2007). This involved appending the datasets in ArcGIS (ESRI 1999) and determining the frequency of plume presence (days) in IDRISI (Eastman 2009) for the entire 525 day dredging period. Data was extracted from the dredge plume frequency dataset to determine the total number of days of plume presence at each site. This method was also used to determine the amount of time the plume overlapped the MBIMCR both north and south of the Barrow Island port boundary.

2.2.4 *In situ* sediment sampling

Sediment deposition

Sediment traps were deployed at eight of the Gorgon MER sites with increasing distance north and south of the dredging area (Fig. 2.1), although not all traps were consistently deployed (Table 2.2). Prior to dredging operations, sediment traps were deployed and retrieved at four monitoring sites (Sites 16,17,14,11) to the north of proposed dredge activity where the plume was predicted to most likely extend.

Sediment traps were deployed at an additional three dredge monitoring sites (Sites 26,23,20), to the south of the dredging operations following the commencement of dredging operations. A fourth site was established to the south (Site 32) in May 2011 to extend the sampling closer to the source of dredge material once it was established that the dominant plume movement was to the south of the dredge operations. Sediment traps were deployed and retrieved three to six times at each site during the dredging operation and twice after dredging had finished.

At each site, three replicate stainless-steel frames were permanently fixed to the substrate using weights and star pickets. Four PVC tubes (300mm long x 50mm diameter) were securely fastened vertically to each frame for the collection of sediment and collectively referred to as the ‘sediment trap’ (Fig. 2.2). Sediment traps were deployed in habitat adjacent and of similar depth and complexity to the areas surveyed for both fish and benthic community composition. Traps were deployed for no longer than 12 weeks to minimise the chance of overfilling and trapped material being re-suspended (Storlazzi *et al.* 2010). To avoid fouling and settlement of flora and fauna, a coarse copper wire mesh was threaded 40 mm from the top of each tube to minimise confounding factors (i.e. filtering organisms, fish removing sediment) that would otherwise influence particle deposition in the tubes.

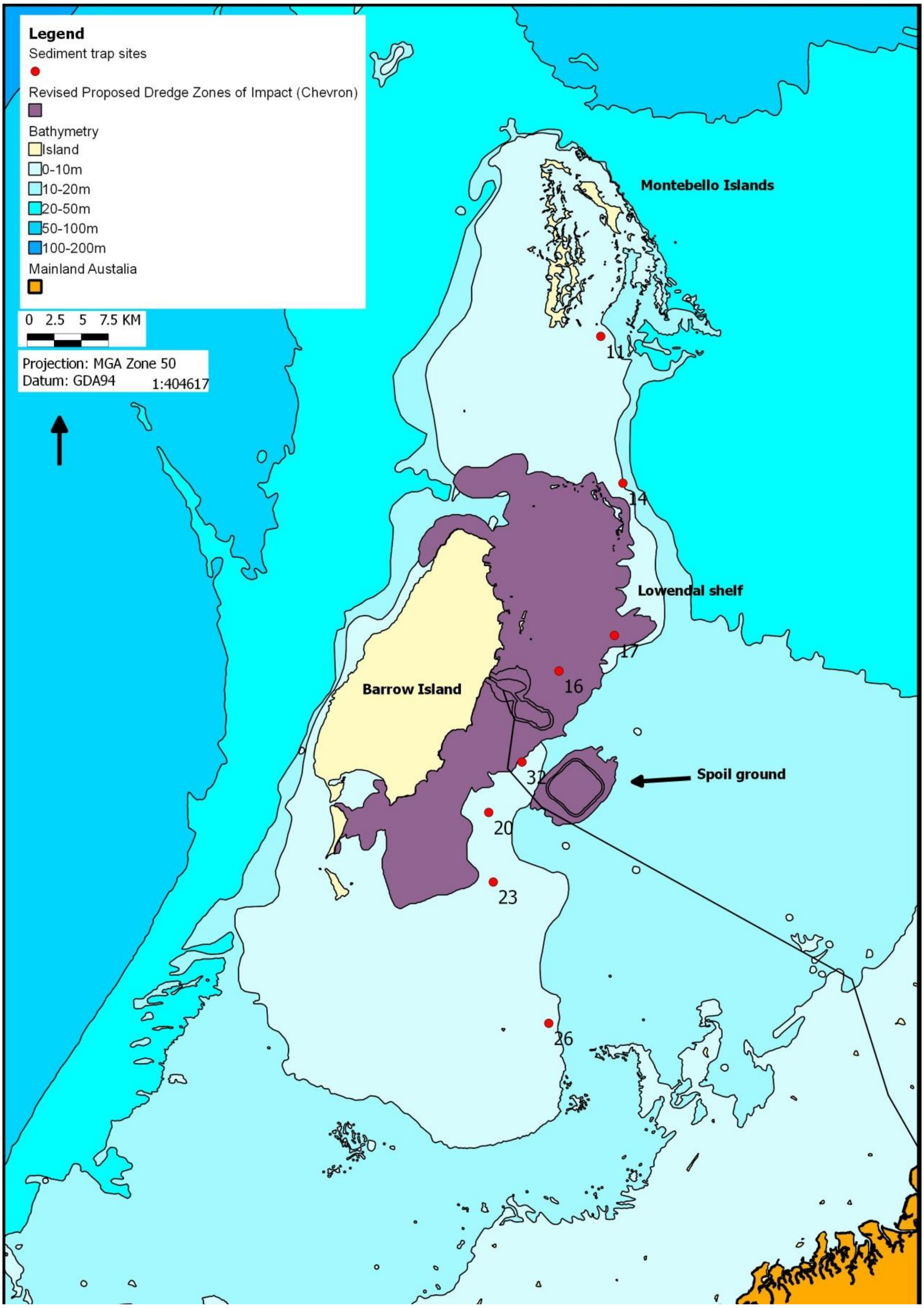

Figure 2.1 Site location of sediment traps relative to the proposed modelling of plume dispersal from dredge operations at Barrow Island.

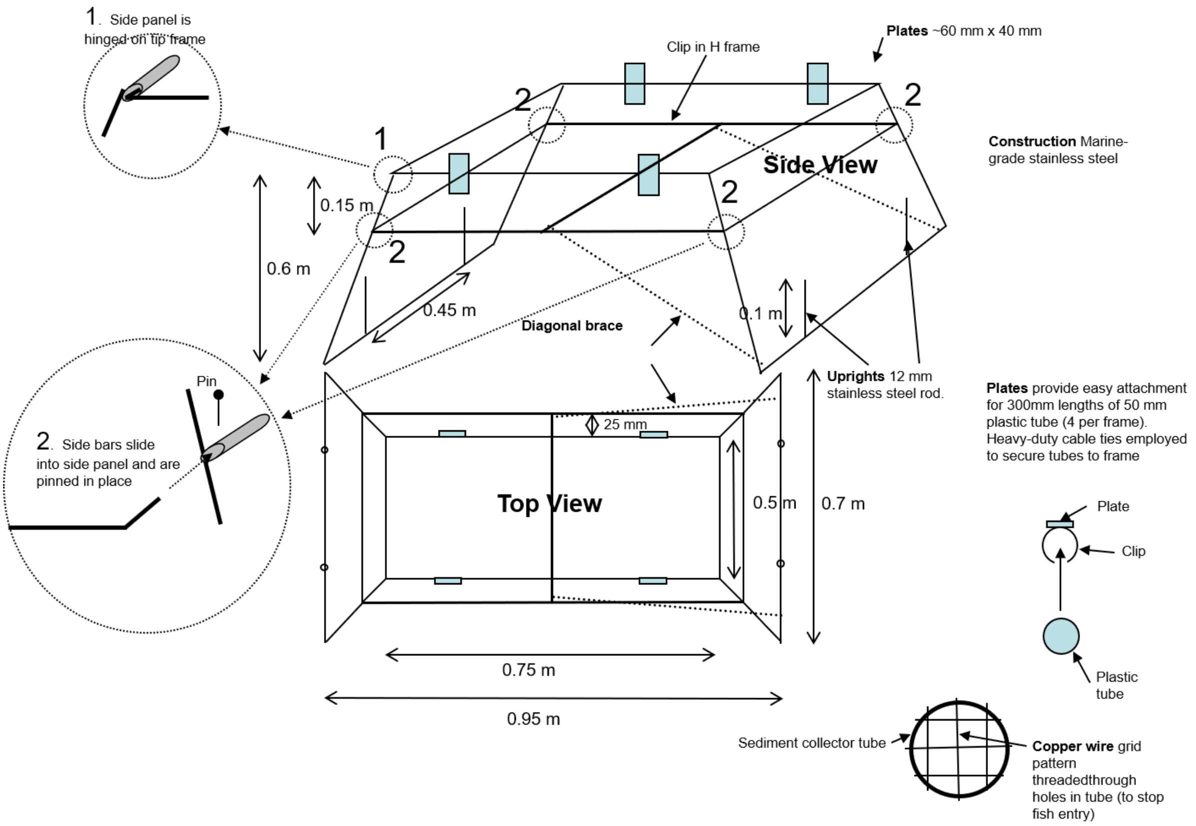

Figure 2.2 Sediment trap design

Sediment analysis

Following retrieval, sediment was quantitatively transferred from the trap to labelled containers, frozen at -20ºC to avoid degeneration of the organic carbon component and returned to the lab for analyses. Sediment from obviously fouled tubes or with faunal inhabitants was discarded to limit biases in differential particle deposition. To quantify total dry weight deposited per trap of each deployment, sediment was thawed to room temperature and transferred to a drying oven at 50°C until a constant dry weight was achieved. Particle size and carbon content of sediment was determined from a subset of dried samples collected before, during and after dredging (Table 2.2).

Particle size distribution – Granulometric analysis

A subset of sediment trap samples from two time points during the dredge (Aug-11 and Sept-11) and after the dredge (Feb-12 and May-12) were processed for precise particle size distribution using a combination of laser diffraction for sub-1000µm particles (Malvern Instruments Mastersizer MS2000) and wet sieving for the remaining larger fraction (see table 2.2). This provided a more precise indication of the percentages of different grain sizes, compared to the qualitative sediment sorting method.

To assess how dredging affects the particle size distribution of sediments, distance from the dredge in a southerly direction (sites 32,20,23,26) was correlated with total and different sediment size classes during and after the dredge. Sediment plume coverage (sites 16,32,20,23,26) as determined by Evans *et al.* (2012) was also correlated with total and different sediment size classes at the two collection times during the dredge operations, to further investigate how extent of dredge exposure influenced sediment distribution. Correlations were conducted in the ggplot2 package in R (Wickham 2009).

Total organic carbon (TOC): Total inorganic carbon (TIC)

Total carbon content of sediment samples was measured using a Labfit CS2000 Carbon and Sulfur Analyser. Each dried sample was weighed and placed into an oxygen rich atmosphere in a combustion furnace regulated at 1400^{o}C. During combustion, carbon in the sample formed carbon dioxide (CO_2). After drying and filtration of the gas flow the CO_2 content of the gas was quantified using an infrared measurement cell.

Total organic carbon (TOC) content of sediment samples was quantified by oxidising organic matter with dichromate ion, $Cr_2O_7^{2-}$ in the presence of sulphuric acid, using heat of dilution to accelerate the reaction according to the formula.

$2Cr_2O_7^{2-}$+ 16H++ 3C → $4Cr^{3+}$+ $8H_2O$ + $3CO_2$

The resultant Cr^{3+} ions are measured spectrophotometrically at 600nm. Total inorganic carbon (TIC) was calculated by subtracting TOC estimates from total carbon estimates. A two-way ANOVA was conducted, within base package of R (R Core Team 2018), to determine if TIC had decreased at each of the sites after the dredging was complete. To minimise analytical costs carbon estimates were only done post-dredging on those sites that were influenced by the dredge based on the digitising analyses, therefore data and subsequent analyses were only completed for those samples that were influenced by the dredge plume, that is Site 16,32,20,23,26.

Table 2.2 Breakdown of analyses completed across sediment monitoring sites and sampling periods for the days of deployment (averaged across sites). Sample analysis included, TD – Total deposition, PSD – Particle size distribution, TICTOC – Total inorganic carbon: total organic carbon. Shaded areas represent sediment sampling during the dredge operation.

Deployed	#Days	Sampling Sites							
		Sites south of dredging				Sites north of dredging			
		26	23	20	32	16	17	14	11
April-May 2010	22	-	-	-	-	TD, PSD	TD, PSD	TD, PSD	TD, PSD
Nov-Jan 2011	65	TD	TD	TD	-	-	-	TD	TD
Jan-Feb 2011	24	TD	TD	TD	-	-	TD	TD	TD
Feb-May 2011	104	TD	TD	TD	-	TD	TD	TD	TD
May-Aug 2011	73	TD, PSD	TD, PSD	TD, PSD	TD, PSD	TD, PSD	TD	TD	TD
Aug-Sep 2011	42	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC
Sept-Nov 2011	43	TD	TD	TD	TD	TD	TD	TD	TD
Nov-Dec 2011	32	TD	TD	TD	TD	TD	TD	TD	TD
Dec-Feb 2012	62	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC	TD, PSD, TICTOC	TD	TD	TD
Feb-May 2012	86	TD, PSD	TD, PSD	TD, PSD	TD, PSD	TD, PSD	TD	TD	TD

2.2.5 Other pressures

Several other pressures exist for marine communities of the MBIMCR. Oil and chemical pollution were monitored through the reporting of pollution events over 20L to the Western Australian Government Department of Transport. Introduced species were monitored over the duration of the study both through the Gorgon MER and by Chevron Australia monitoring as indicated in the Quarantine Management Strategy (Chevron Australia 2009). No incidences of either pressure occurred at the Gorgon MER sites.

2.3 Results

2.3.1 Physical and oceanographic data

Wave conditions

Wave height data modelled by AUSWAVE-A matched the *in situ* AWAC logger wave height data at site 32, although modelled data underestimated some of the *in situ* peaks (Fig. 2.3). High R^2 (0.73) and regression slope (0.85) indicate a good predictive relationship and thus provides some confidence in the modelled AUSWAVE data at other sites.

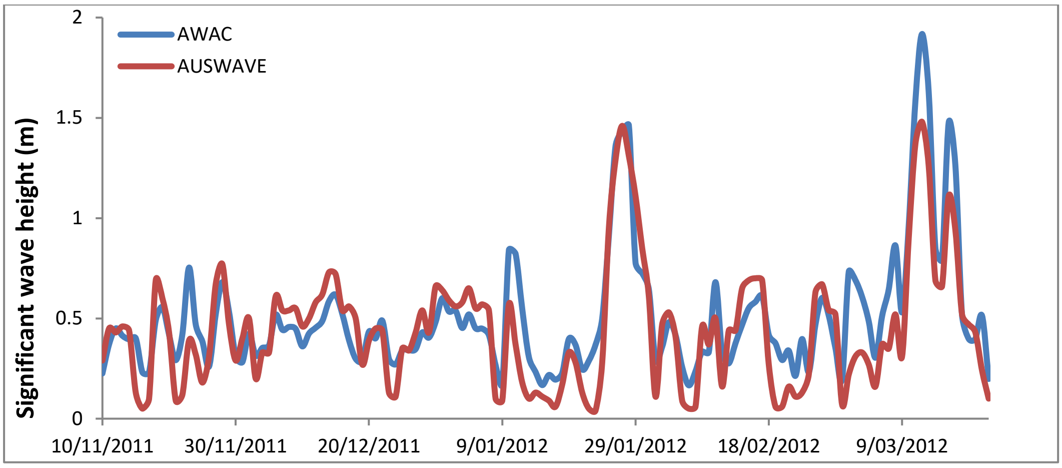

Figure 2.3 Plotted AUSWAVE-A modelled wave height and actual measured wave height Using (AWAC) at site 32 near the dredging operations.

Data modelled from AUSWAVE showed that three sites (17,14,11) on the east side of the Lowendal Shelf and to the north of the dredging operation experienced higher wave heights than the sites further south. Higher median, outlier and extreme wave heights (0.55-0.7m) were recorded compared with sites further south (sites 16,32,20,23,26) (Fig 2.4). Of interest, the extreme outlier events (wave heights greater than 3IQRs from the median) at the sites to the north of the dredging operations were over 1m higher and up to four times more frequent than those to the south of the Lowendal Shelf. Site 16 was an exception as it is located in a sheltered location and experienced low median and extreme wave heights compared to other sites.

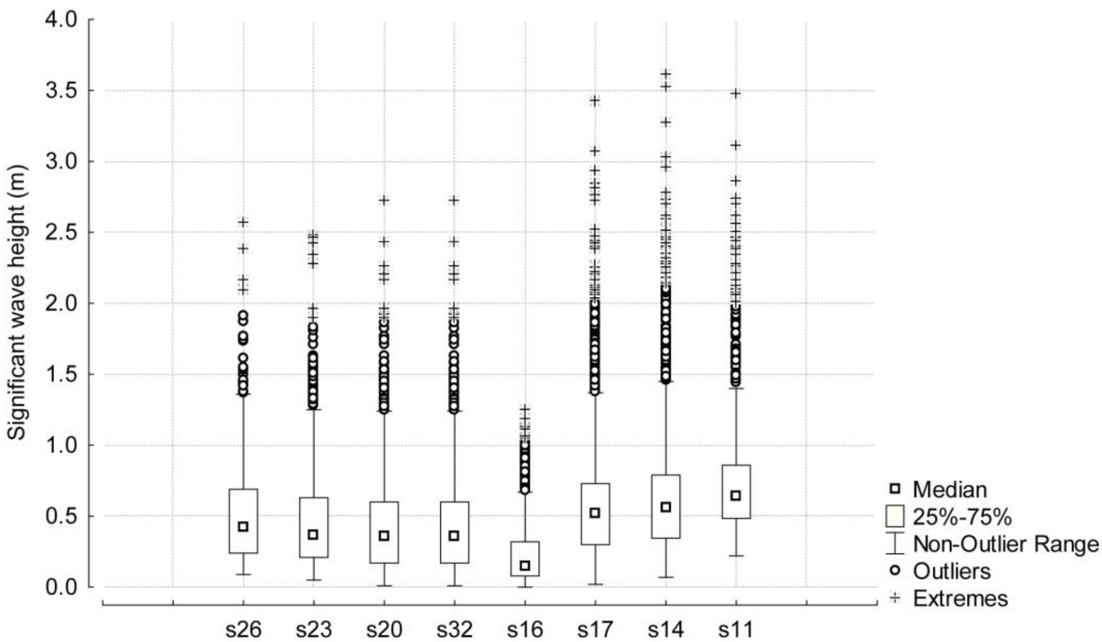

Figure 2.4 Significant wave height regime for the period of AUSWAVE-A operation (1/11/2010 – 05/05/2012). Non-outliers are less than or equal to 1.5x Inter-quartile range (IQR) of the median value, outliers are between 1.5 and 3x IQR's and the extremes are greater than 3x IQR from the median value.

Cyclones

There was large inter-annual variation in the amount of cyclonic and stormy conditions between the years of the Gorgon MER project (Fig 2.5). The 2009-10 season was the least active during the study period with just two named storms forming well to the north-east of the MBI. In contrast, the 2010-11 season was extremely active, with four named storms passing adjacent to the north-west WA coast. Two of these storms passed within 50km of GMER study sites. Additionally, several unnamed tropical storms and depressions formed in the same region during the season. Three cyclones also formed off the north-west coast during 2011-12, however, only one of these passed within 200km offshore from Barrow Island and the survey sites.

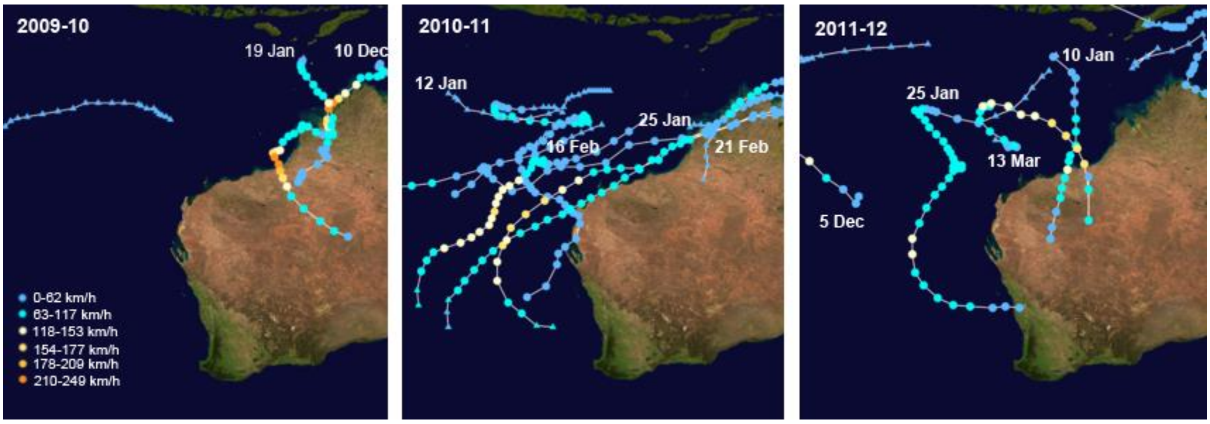

Figure 2.5 Cyclone and storm tracks over the duration of the GMER project (wind strengths indicated by legend on figure). Dates signify the day cyclones were officially named. Australian cyclone season annual summaries created by Potapych - Created using Wikipedia:WikiProject tropical cyclones/tracks. The background image is from NASA [1]. The tracking data is from the Joint Typhoon Warning Center's best track database. Licensed under Public Domain via Commons- https://commons.wikimedia.org/wiki/File:2006_2007_Australian_cyclone_season_summary.jpg#/media/File:2006 2007_Australian_cyclone_season_summary.jpg.

2.3.2 Suspended sediment monitoring

Mapping plume extent

Sediment plumes resulting from the dredging campaign were most prominent from the sites of material removal at both the MOF access site and the LNG shipping terminal turning circle site. Smaller plumes emanated from the MOF island construction site, leaching from the bund walls and from the site of spoil disposal further offshore in deeper (~15m) water. The movement of the sediment plumes was predominantly in a southward direction with minimal days of northward movement for the duration of the project. The cumulative plume exposure over the Barrow Island Marine Management Area was 395 km^2 (Fig. 2.6), albeit at low frequency (Fig. 2.7). A plume was visible, primarily in the port area, for up to three months after dredging finished. Of the 65 days that could be analysed post-dredging, 147km^2 of cumulative plume area was digitised, of which 19km^2 encroached into the Barrow Island Marine Management Area (Fig. 2.6). There was, however, no plume evident from the spoil ground. This method is considered a conservative estimate of plume extent due to the limitations of plume detection.

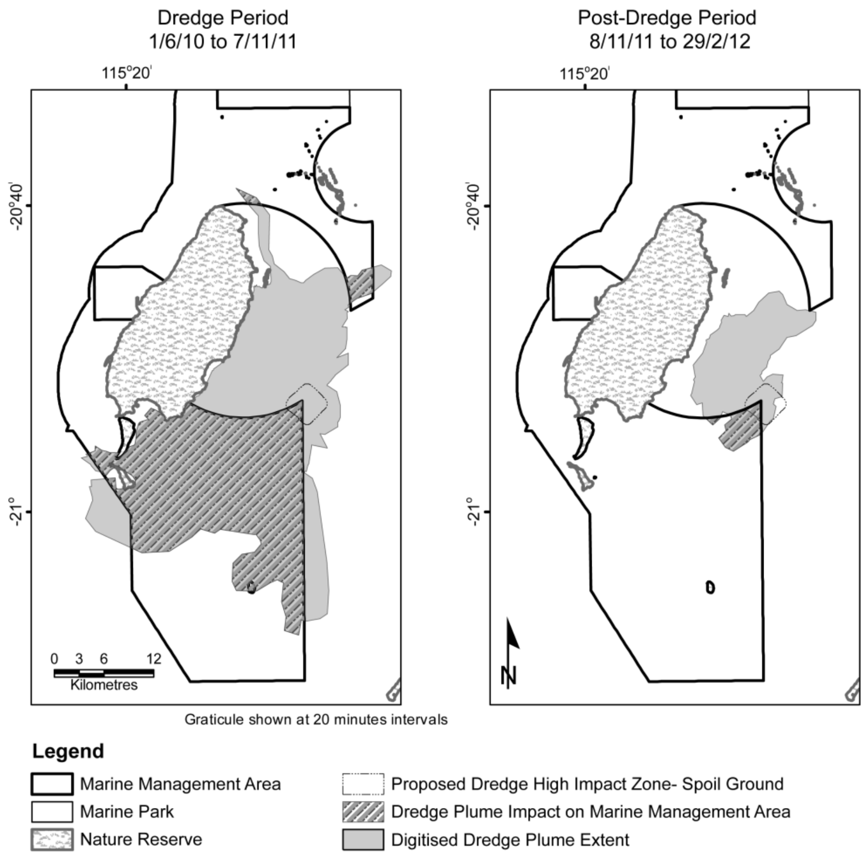

Figure 2.6 Extent of dredge plume incurring into the Barrow Island Marine Management Area boundary during (left) and after (right) dredging.

Hot-spot analysis

The area of consistent plume exposure, as described by the hotspot analysis, was limited to the port zone <10km from the dredge location. However, the plume was also recorded on images up to 35km south of the dredge location along the east side of the Barrow Shoals (Fig. 2.6). There was also at times a noticeable plume originating from the spoil ground. Hotspot analysis completed on the cumulative plume exposure showed that the area of greatest plume exposure, >373 days of the 411 recorded, extended <8km south and <1km north of the dredging location, limiting the greatest plume exposure to the area within port limits (Fig. 2.7). The Gorgon MER sites were exposed to the plume from 0 to 296 of the observed days with the limits of the plume extending >35km to the south and 10km to the north of the dredge location (Table 2.3, Fig. 2.7).

Table 2.3 Number of days the Gorgon MER sites (north to south) were covered by plume as determined by digitising remote sensed MODIS images. Red line indicates where the dredge operation sits relative to the sites.

Site Number	Plume cover (days)
5	0
8	0
11	0
19	0
12	0
14	0
15	0
29	0
17	3
16	43
27	32
18	40
32	296
20	78
28	69
23	9
31	1
26	3

2.3.3 *In situ* sediment sampling

Sediment deposition

Average daily deposition of sediment was highly variable both spatially and temporally with the highest deposition recorded in February 2011 at five of the six sites at which data was collected and lowest in November and December 2011 (Fig. 2.8). Sites to the north of dredging operations (sites 17,14,11) were exposed to the greatest wave energy, and consistently recorded higher average daily deposition compared to the other five sites which were situated in comparatively sheltered positions (Fig. 2.8). Northern sites however experienced little to no exposure to the dredge plume (Fig. 2.7). Higher quantities of sediment were recorded at site 11 in September 2011, which was primarily due to large amounts of coarse sand and gravel in these samples (Fig. 2.8).

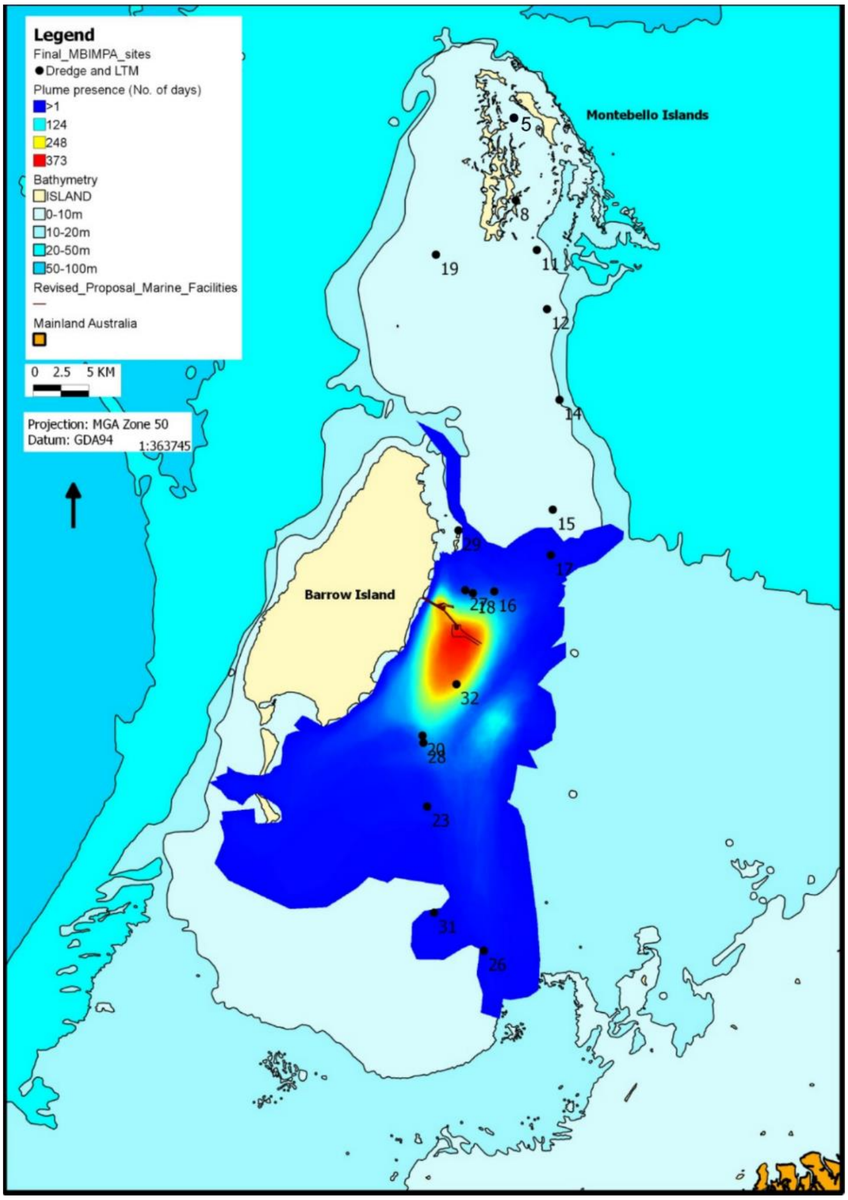

Figure 2.7 Hotspot analysis of daily digitised plume area during dredging campaign 1/6/2010 – 7/11/2011 (n = 411 cloud-free images). Site numbers are shown.

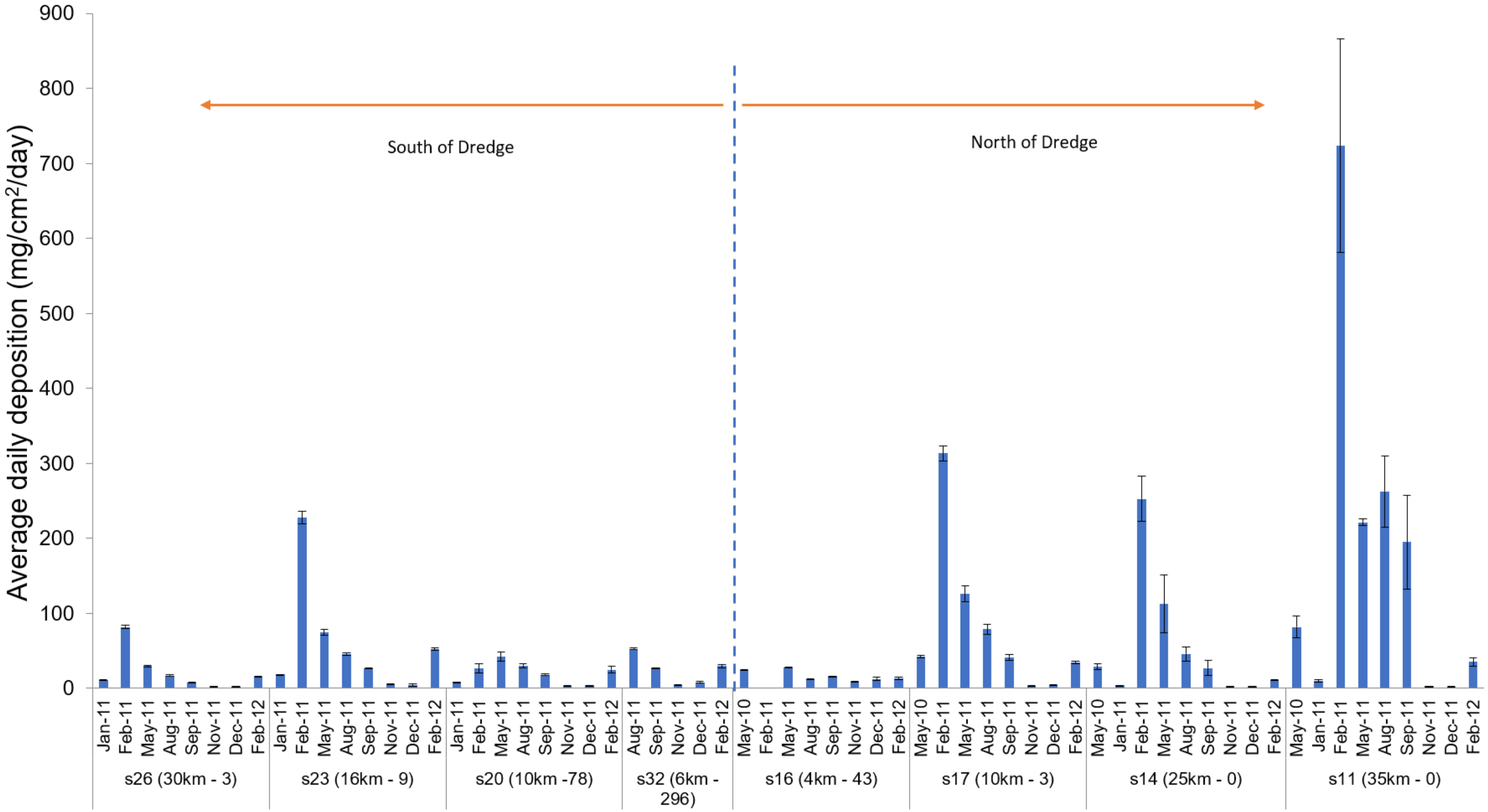

Figure 2.8 Average daily dry weight deposition at sediment trap sites, (direction and distance from dredge operations – number of days of plume exposure). The dredge location is indicated by the blue line. Dredging and related spoil disposal campaign ran from May 2010 to November 2011. Average and standard errors calculated from three samples at each site.

Correlations between sediment size categories and exposure to dredge plume suggest that more fine sediments were present at sites closer to the dredge operations. There was a significant positive relationship between the percentage days of plume coverage and the trapping of finer sediments including the 4µm (clays) and the 4-62µm (silts) fractions (Table 2.4a). There was also a positive relationship between total sediment deposits and the plume coverage days, but the relationship was not significant (Table 2.4a). However, dredge distance was significantly correlated with distance, southerly sites closer to the dredge operation had greater total weight deposited in the traps than those further away (Table 2.4b). More finer sediments (clay and silts) were also captured in the traps closer to the dredge in the southward flow of the plume, while the heavier coarser sediments showed no relationship (Fig. 2.9, Table 2.4b). This relationship continued post-dredging for the smallest size class of sediment (clay) but became weaker for the silts (4-62µm) (Fig. 2.9, Table 2.4c), and further still for the total weight of sediments post-dredging (Table 2.4c).

Table 2.4 Results of the Pearson's correlation of six sediment size classes and total in situ deposition with a) percentage days of plume coverage from Hotspot analysis; b) dredge distance from sampling sites during dredging; and c) dredge distance from sampling sites post-dredging. Only sites to the south of dredge operation are used in this analysis.

Sediment grain size	name	a) Plume coverage n=8		b) Dredge distance (sites to south) n=4		c) Post-dredge distance (sites to south) n=4	
		r	*p*	r	*p*	r	*p*
<4um	**Clay**	**0.88**	**0.001**	-0.94	0.065	**-0.97**	**0.03**
4-62um	**Silt**	**0.83**	**0.003**	-0.92	0.082	-0.87	0.13
62-250um	**Fine sand**	0.27	0.44	-0.35	0.65	-0.16	0.84
250-500um	**Mod sand**	0.38	0.28	-0.45	0.55	-0.12	0.88
500-2000um	**Coarse sand**	0.33	0.36	-0.54	0.46	-0.6	0.4
>2000um	**Gravel**	0.01	0.98	-0.36	0.64	-0.32	0.68
Total Deposit		0.53	0.12	**-0.98**	**0.02**	-0.51	0.49

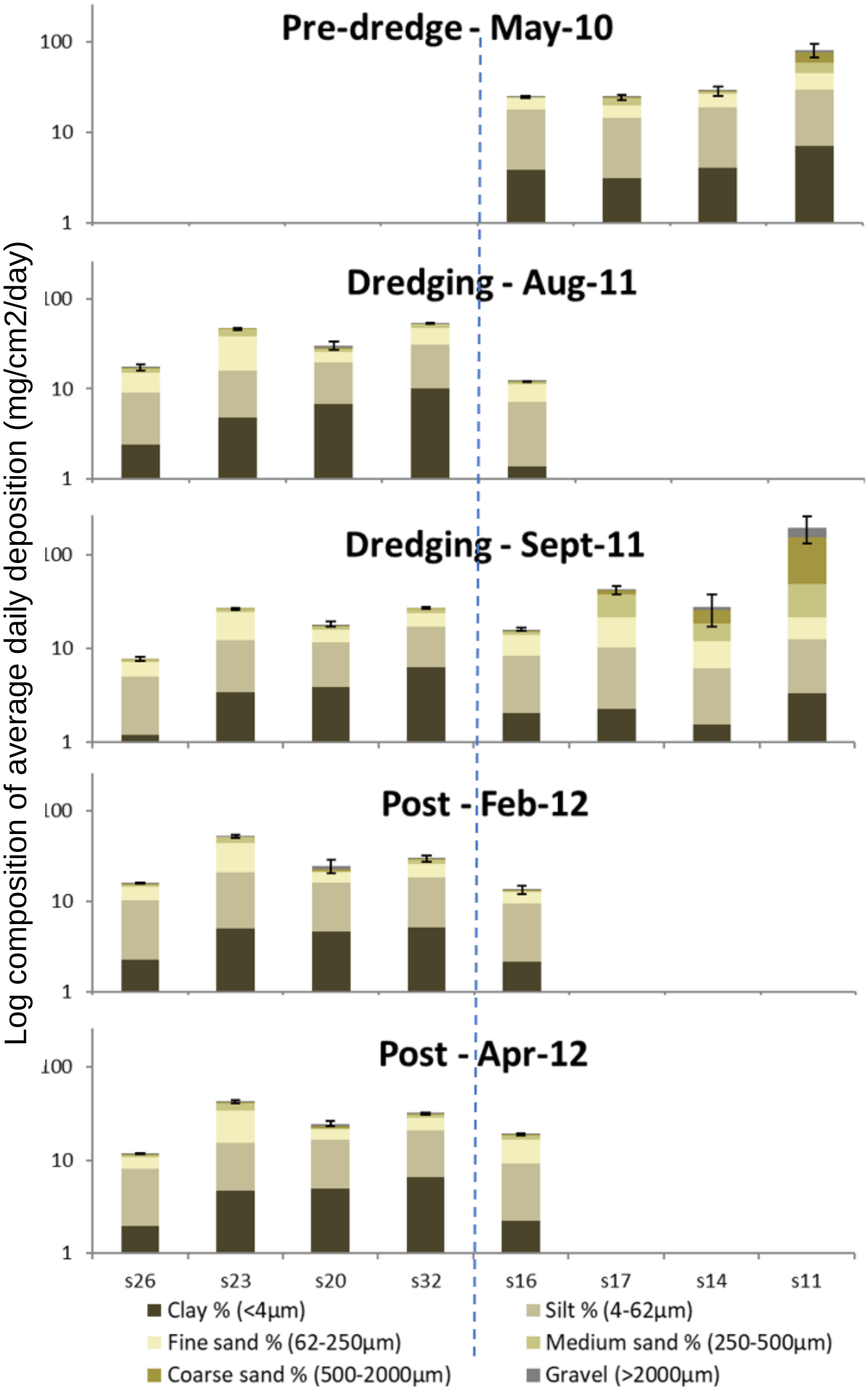

Figure 2.9 Categorised sediment composition (log scale) by total daily deposited dry weight before, during, and after dredging. Error bars represent 1SE of total weight calculated from three replicate sediment traps at each site. Dashed line indicates the dredge location relative to the sites.

Total inorganic (TIC) and organic (TOC) carbon

The relative amounts of inorganic to organic carbon was relatively consistent between samples collected during and after the dredging at sites at sites 32, 20 and 23, where the TOC was less than 10% (Fig. 2.10a). However, both site 26 and 16 had increased levels of TOC post-dredging, site 16 accumulating ~12% and site 26 ~6% more organic matter after dredging was finished. Sites to the north of the dredging (sites 17,14,11) all had high TIC:TOC ratios during dredging, and organic carbon represented only 2-5% of all carbon (Fig. 2.10a). However, the plume rarely covered these sites and no samples were taken post-dredging. A significant decline in TIC after the dredging finished was evident at all but one of the sites (Fig. 2.10b).

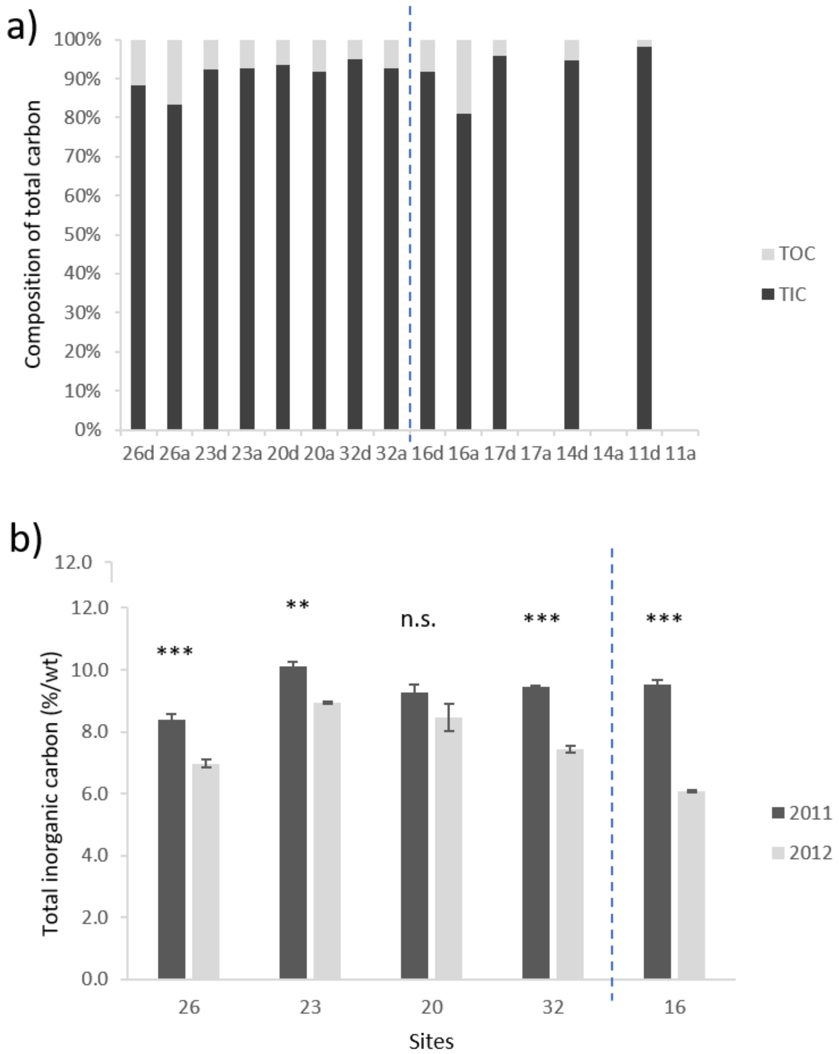

*Figure 2.10 A) Relative total organic carbon (TOC) and inorganic carbon (TIC) by percentage of total sediment sample; d: during dredge (2011); a: after dredge (2012), and B) total inorganic carbon as a percentage of the weight of sediment during (Sept 11) and after (Feb 12) dredging. Blue lines indicate the location of the dredging activity with sites to the north on the right and to the south on the left. Level of significant difference between each site from 2011 to 2012 is represented, ***: P<0.0001; **: P<0.001; *: p<0.05; ns: not significant.*

2.4 Discussion

Sediment deposition at the sites was primarily driven by the amount of wave energy. However, multiple lines of evidence suggest that the plume originating from the dredging operations at Barrow Island was detected at the Gorgon MER coral monitoring sites. Remote sensing and the associated HotSpot analyses provided useful large-scale indications of dredge plume presence and extent, whilst *in situ* sediment traps revealed considerable within-site heterogeneity in sediment deposition characteristics. There were indications of higher proportions of fine (<4 µm) clays and silts (4-64µm) in traps nearer dredging activity, and deposition rates of these fine particles correlated positively with measures of dredge plume presence from HotSpot analysis. Further, carbon fraction analyses of sediment collected during and after dredging indicate that inorganic content of sediment was higher at sites covered by the plume during the dredging than post-dredge. The conclusions drawn from the sediment trap analyses cannot be over-interpreted as there is no baseline data collected prior to the dredging for sites that were affected by the dredge plume. The following chapters investigate if the presence of these dredge constituents had any effect on the coral and fish communities at those sites.

2.4.1 Remote sensing and HotSpot analyses

The digitisation technique (Evans *et al.* 2012) provided spatial and temporal information on the distribution of dredge sediments which enabled quantification of dredging impact during the campaign at a site level (Chevron Australia 2009). Evans *et al.* (2012) found that this method detected dredge plume presence over shallow benthic features more often than the widely used total suspended solids analyses employed elsewhere (Devlin *et al.* 2012; Lambrechts *et al.* 2010). Given the relative ease with which high resolution satellite imagery can now be acquired, this method provides much needed quality control information on dredge plume extent for dredging projects, particularly in the shallow coastal waters of WA where both benthic biota such as corals and seagrass may be affected. The plume was recorded over a total of 734km^2, of which 278.87km^2 overlapped with modelled dredge plume extent (Chevron Australia 2009) with a further 454.89km^2 extending outside the modelled area. Further, coupled with *in situ* monitoring techniques, HotSpot analyses provide quantitative data on the impacts of dredging activities across large spatial scales at relatively low expense. For more details see Evans *et al.* 2012.

2.4.2 Storminess, waves and total sediment deposition

An important process influencing the total deposition of sediment was wind and wave energy, the strength of which was coincident with the extent and intensity of storminess in the region. Seasonal wave energy maxima typically occur in summer with the generation of considerable wind and wave activity associated with intense tropical storms and cyclones (Moore *et al.* 2012), which are a climatic feature of north-west Australia (Goebbert and Leslie 2010). Wind strength and direction are prominent drivers of wave activity in the region (Pearce and Feng 2013), but wind monitoring is limited in the region, and could not provide a site-specific metric across the spatial scale of GMER sites. Direct measurements of wave activity have provided an indication of localised impacts of wind and waves. Data from AUSWAVE-A swell and wave model (source: http://www.bom.gov.au/nwp/doc/auswave/data.shtml), provided site-specific data on swell and waves for the duration of the project, which are useful for understanding natural patterns in the distribution of corals (Dollar and Tribble 1993; Done 1992) and fish (Fulton *et al.* 2005).

Wave energy obviously plays an important role in sediment deposition rates and resuspension of local materials on coral reefs (Crossman *et al.* 2001; Larcombe *et al.* 1995; Storlazzi *et al.* 2004). The plume from dredging in this study did not occur over the three northernmost sites, yet deposition rates were high at these sites suggesting that all sediments accumulated in traps at these sites were a consequence of high wave energy. Indeed, the highest deposition of sediment was recorded at Site 11 (Ah Chong Island), the site with some of the highest wave energy. The wave regimes at sites adjacent to Barrow Island and south to the Barrow Shoals were low compared to northern sites, but relatively similar to each other, which is reflected in the similar deposition amounts recorded for these sites. The dredge plume extended over several of these sites and it is likely that deposition of dredge-derived materials occurred within the sediment traps at these sites. However, the relationship between total sediment deposited and days exposed to the plume is weak and the total sediment load in traps is not likely to be a useful indicator of dredging impact given the strong relationship between site-level wave energy and the amount of sediment accumulated in traps. Some relationship was observed in the smaller particle sizes.

Particle size distribution profiles

There was great variation in particle size distributions between sampling periods at the same site, likely due to temporal variation in wave activity and differential transport and resuspension of different particle fractions. Within the sampling period, wave energy could be broadly considered to be consistent amongst those sites adjacent to Barrow Island, and among the higher energy sites to the north. Thus, comparisons of particle size distributions within, but not between, these areas are warranted. Whilst sedimentation rate may be measured in multiple ways and will necessarily have regional-scale implications for coral assemblages (reviewed by Erftemeijer *et al.* 2012), emerging evidence suggests that sediment grain sizes will exert varying impacts on coral communities (Flores *et al.* 2012; Weber *et al.* 2006), seagrass (see review in Erftemeijer and Robin Lewis III 2006) and fish (Partridge and Michael 2010; Wenger *et al.* 2013). The positive correlation between small (<4µm) clays and percentage days of plume presence, as well as an increasing trend in the smaller (<62µm) fraction of sediments in traps nearer to the dredge, suggests smaller size fractions are characteristic of sediments deposited by the plume. These small particles are more likely to remain in suspension for longer (Storlazzi *et al.* 2010) and thus be advected away from the point of dredging impact towards adjacent areas. Fine sediments have been known to have adverse effects by directly accumulating on coral tissues, and sub-lethal responses were indicated by damage to photosystem II in the symbionts and reductions in lipid content and growth (Flores *et al.* 2012).

A major limitation to the usefulness of this study was the absence of background sedimentation rates and sediment size-class distributions prior to dredging. Prior dredge sampling effort focused on the areas north of Barrow Island, which models suggested would be impacted by significant plume coverage but in fact experienced limited plume presence during the dredging campaign. Based on the digitising method, we know the sites to the north were largely not influenced by the plume. However, these sites do not make good control sites as prevailing wave energy is higher on northern sites and they are ecologically and biologically different (see coral and fish chapters) to those sites impacted by the plume.

TIC:TOC analyses

As the dredging machinery extrudes the limestone or calcarenite sediment at Barrow Island (DEC, 2007), the amount of inorganic suspended sediment settling on the surrounding reefs should increase during the dredge operations, and subsequently decrease once the dredging is complete. The higher levels of inorganics observed during rather than after the dredge operations support this theory. However, consequences for coral impacts must be made cautiously as there was no TIC:TOC assessment of sediments at these sites before dredging and therefore no baseline understanding of inherent sediment composition.

Recent work suggests that total organic carbon may be an important determinant of sediment-driven coral mortality as organic matter proves a medium for microbes that can cause coral mortality (Weber *et al.* 2012). Conversely, many corals ingest and assimilate suspended particulate matter (Anthony, 1999, 2000; Mills *et al.* 2004) and an increased proportion of organic matter in sediments may represent a more viable food source. Carbon fraction analyses of sediment collected during and after dredging indicate that organic content of sediment increased at sites covered by the plume once dredging was completed.

2.4.3 Conclusions and recommendations for future studies

Several important points arise from the results of this study in the context of guiding further monitoring studies of the impacts of sedimentation associated with large-scale developments in coral reef areas along the WA coast. Firstly, a thorough understanding of the physical processes driving the dispersal of sediment plumes and sedimentation is required to disentangle contributions of dredging from natural sediment regimes. This may be achieved with thorough baseline information on the physical environment (e.g. wind, wave and temperature), sediment characteristics and condition of biota before, during and after impact. Secondly, sampling designs for assessing dredge effects on the environment should consider the dimensions and location of potential impacts and sample appropriately but should also consider the possibility that plumes may influence areas outside of the modelled or expected area of impact. Further, a rapid and cost-effective method for monitoring dredge plume dispersal developed by Evans *et al.* (2012) as part of this study should be integrated into the iterative experimental design of sediment monitoring studies.

3 The influence of dredging and environment on the variability of coral communities at the Montebello and Barrow islands

3.1 Introduction

Coral communities are a prominent and ecologically diverse component of the Montebello and Barrow Islands marine conservation reserves (MBIMCR). The most diverse communities exist on the fringing reefs in the relatively clear and high energy waters to the west and south-west of the Montebello Islands, as well as the patch reefs in the more turbid and lower energy waters along the eastern fringes of the Montebello Islands, Lowendal Shelf and the Barrow Shoals (DEC 2007; Fig. 3.1). Coral communities of the MBI are characterised by a high diversity of hard and soft corals with 150 species from 54 genera recorded by Marsh (2000), while rapid visual assessment surveys conducted more recently identified 196 scleractinian species from 48 genera (Chevron 2008) and Richards and Rosser (2012) recorded 204 scleractinian species from 54 genera.

Oceanographic modelling and genetic studies indicate that the MBI are biologically linked to the tropical coral communities of the Pilbara mainland to the east and to the south (Condie *et al.* 2005; Condie *et al.* 2006, D'Adamo 2009, Feng *et al.* 2016, Thomas *et al.* 2017, Di Battista *et al.* 2017), although Evans *et al.* (2019) recently showed some evidence of gene flow restriction between the Northern Montebello Islands to parts of the western Pilbara coastline. The southerly flowing Holloway current also supplies propagules from the north-east (D'Adamo 2009) and counter north-easterly currents in the spring and summer (Cresswell *et al.* 1993) may supply recruits from the south (Underwood *et al.* 2013). However, the MBI are also within the cyclone belt where elevated frequency of high energy winds and waves can alter typical connectivity patterns (Radford *et al.* 2014) making it difficult to ascertain the ecologically important sources of recruits and patterns of connectivity.

Despite their geographic isolation, waters around the MBI have been exposed to various anthropogenic pressures. Pearl harvesting and cultivation has occurred on the MBI since the mid-19th century, capitalising on protected waters and strong tidal currents (Stanbury 1994). In the 1950s three nuclear tests were conducted at the MBI, with devices detonated on Alpha and Trimouille Islands and aboard HMS *Plym* anchored off Trimouille (Child and Hotchkis 2013). In recent times there has also been an increase in commercial and recreational tourism which, which like the pearl culture, has primarily occurred around the Montebello Islands. The discovery of oil on Barrow Island in the 1960s led to the establishment of extraction and processing infrastructure on Barrow Island, followed by the 1987 establishment of similar facilities on Varanus Island, the largest of the Lowendal Islands to the north-east of Barrow Island. The development of oil and gas facilities on Barrow and Varanus islands has resulted in major land and marine-based infrastructure associated with processing and transport, including port developments and shipping activity related to these developments.

The development of industrial infrastructure can cause increased pressure on ecological communities in the marine environment. This is particularly true of dredging projects where increased turbidity and sedimentation can cause light reduction and smothering of benthic communities. This may be detrimental for primary producers such as corals and can cause extensive mortality in areas consistently exposed to a dredge plume (Erftemeijer *et al.* 2012, Jones *et al.* 2016). The influence could be expected to decline as exposure diminishes at sites further away from the dredge. However, the magnitude of dredge plumes will be governed by factors like the extent of the dredging, the geology of the seabed and prevailing winds and the currents. As such, the potential influence from suspended sediments, particularly the finer sediments which remain in suspension, may extend far from the original dredging site.

The dredging and related spoil disposal campaign for the Gorgon gas project ran from May 2010 to November 2011 and involved the removal of over 7.5M tonnes of marine sediment from the materials offloading facility (MOF) access channel and the Liquid Natural Gas (LNG) shipping terminal access channel and turning circle (Chapter 1, Fig. 1.2). Of the sediment removed, 1.6M tonnes was utilised for the construction of the MOF, while the remaining 6.4M tonnes were dumped within the allocated spoil disposal ground in 10-15m of water (Chapter 1, Fig. 1.2). The major area of influence from the dredge plume extended south from the area of dredging and land reclamation as a combined plume with a smaller more isolated plume tail extending south from the spoil ground (Chapter 2; Evans *et al.* 2012). The area consistently influenced by the dredge plume over the 18-month duration of the dredging program was restricted to within 10km south of the dredge site with most of the dredge monitoring sites lying outside the area of influence (Evans *et al.* 2012).

This study presents an analysis of data collected in the MBI region from 2009 to 2012 as part of the Gorgon Monitoring Evaluation and Reporting Project (Gorgon MER). The study was designed primarily to investigate the influence of dredging and marine construction activities on the coral communities outside the zones of high and moderate impact, where modelling data predicted minor to no influence of the dredge plume (see Chapter 2, Fig. 2.1). We investigated whether the dredging and marine construction activities associated with the Gorgon gas project had any measurable effect on the coral communities near the MBI by comparing changes in coral communities to plume exposure. A range of other natural and anthropogenic pressures were also measured to assess their relationship with spatial and temporal differences among coral communities. In particular, the 2011 marine heat wave (Benthuysen *et al.* 2014, Pearce and Feng 2013) occurred during the study, so we explicitly examine the impact of the increased thermal temperature on corals around the MBI to understand the spatial variation in bleaching and survivorship of corals.

3.2 Methods

3.2.1 Study sites

Coral communities of the MBI were initially surveyed at 26 sites in 2006, to characterise the coral communities at the establishment of the Montebello and Barrow

Islands marine conservation reserves (DEC 2007). The sites were selected to document the extensive and variable coral-dominated communities characterising the MBI (Bancroft 2009). Site location was recorded using GPS waypoints at the start and end of three 50m transects surveyed.

In November 2009, 13 of the 26 sites surveyed in 2006, along with three new sites, were surveyed to provide baseline information on the coral communities prior to the start of the Gorgon project dredging program (Fig. 3.1). These sites, known as 'dredge monitoring sites', were selected for the Gorgon MER monitoring program to represent reefs across the marine conservation reserves and enable monitoring of impacts associated with the Gorgon project's dredging operations on coral communities within the marine reserves. An additional site (31) was established at the start of dredging operations in May 2010 to improve spatial distribution of the monitoring sites within the reserve, and site 32 was added in November 2011, once plume dispersal characteristics were identified, to better understand the effects of regular dredge plume exposure on corals (Table 3.1).

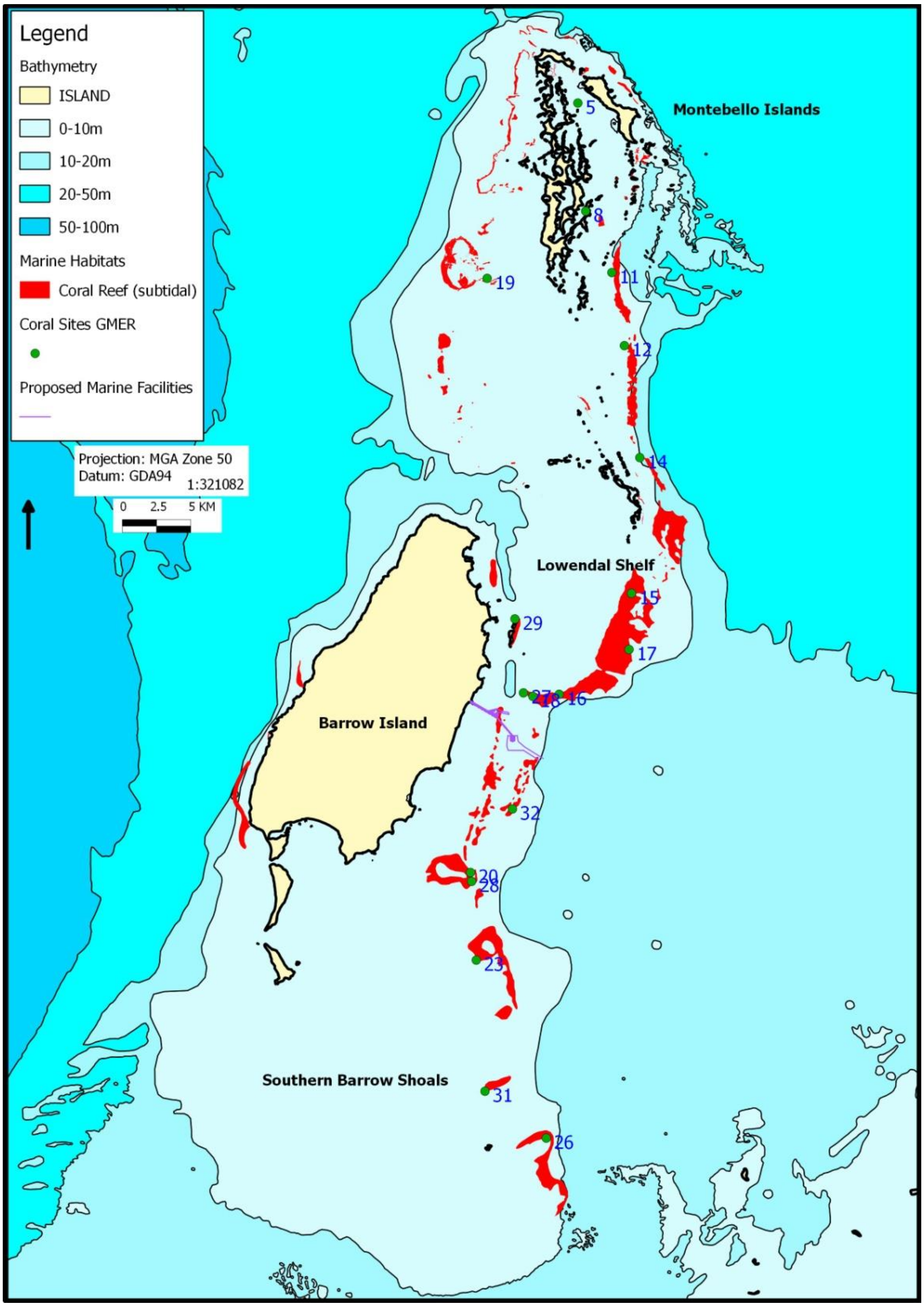

Figure 3.1 Dredge monitoring sites used for assessing the effects of dredging on coral communities. See Figure 1.2 for arrangement of sites relative to zones of predicted impact and Figure 1.3 for arrangement of sites relative to zoning in the Montebello and Barrow islands marine conservation reserves.

These 18 'dredge monitoring sites' were located at increasing distances, both north and south (See Table 3.1, Fig. 3.1) from the site of dredging and spoil disposal to provide a gradient of influence from the dredge activity based on the predicted movement of the dredge plume (Chevron 2005).

At 14 of the ‘dredge monitoring sites’, four 50m transects were permanently marked with star pickets at the start and finish of each transect. These reference points corresponded with the GPS waypoints recorded during initial surveys in 2006 (Table 3.1). Steel rods (~40cm long) were also driven into the substrate at points approximately 15m, 25m and 35m along the transect and plastic cable ties were attached to the substrate at 2-3m intervals between each rod. At the remaining four ‘dredge monitoring sites’ (5,14,15,32), the four transects were only marked using GPS waypoints at the start and finish of each transect.

Diverse weather and technical issues allowed only twelve of the ‘dredge monitoring’ sites to be consistently surveyed twice before, once during and once after dredging, while the remaining six dredge monitoring sites provided additional site-specific spatial information on the benthic communities in the MBI (Table 3.1). These sites will be referred to as the temporal dredge monitoring sites. Digital still images were taken every 1m along the transects using a Canon G12 PowerShot High Sensitivity 10mp camera held perpendicular to the substrate at a fixed height of 1m, providing an image area of 0.85m^2 and ~42.5m^2 per transect.

*Table 3.1 Sampling regime for collection of benthic cover data within n replicate 50m transects. # – Coral bleaching survey; DS – Digital still images. Sites shaded are the 12 temporal dredge monitoring sites that were consistently surveyed over the life of the study. *indicates transects marked with star pickets and steel rods.*

	Survey Periods (n – replicate transects)					
	2009		**2010**	**2011**		**2012**
Site	***n***	**Nov**	**Apr**	**Feb**	**May**	**May**
8*	4	DS	DS	DS#	DS	DS
11*	4	DS	DS		DS	DS
12*	4	DS	DS		DS	DS
16*	4	DS	DS	DS#	DS	DS
17*	4	DS	DS	DS#	DS	DS
18*	4	DS	DS	DS#	DS	DS
19*	4	DS	DS	DS#	DS	DS
20*	4	DS	DS	DS#	DS	DS
23*	4	DS	DS	DS#	DS	DS
26*	4	DS	DS	DS#	DS	DS
27*	4	DS	DS	DS#	DS	DS
29*	4	DS	DS	DS#	DS	DS
5	3	DS				DS
14	3	DS				
15	3	DS			DS	DS
28*	4		DS		DS	DS
31*	4		DS	DS#	DS	DS
32	3					DS

3.2.2 Image analysis

Digital imagery of benthic assemblages was imported into the software package ECOPASS and analysed utilising the ‘point count’ method to quantify benthic cover. In ECOPASS, six random points were overlayed onto each image and each point classified to the highest taxonomic level possible using predetermined hierarchies representing all possible benthic classifications. At a coarse level, points were identified as ‘coral’, ‘macroalgae’, ‘invertebrates’, ‘turf algae or crustose coralline algae (CCA) on stable substrate’, ‘turf algae or CCA on rubble (unstable substrate)’ or ‘sand’. These categories are consistent with those used by the DBCA Marine Monitoring Program and other published classification schemes (Hill and Wilkinson 2004). Hard corals were identified to genus level to investigate the spatial and temporal variability in the composition of coral communities among sites and the response of different genera to environmental and human pressures. Signs of stress on corals were monitored over the duration of the study using the same point count method to record the presence of bleaching, physical damage; sedimentation and disease (see Introduction: Pollock *et al.* 2014, 2016).

3.2.3 Statistical analysis

Spatial and temporal variation in the coarse benthic categories, coral cover and composition were tested for using permutational analysis of variance (PERMANOVA) with 999 permutations, where replicate transects were random factors, nested in random sites and year was a fixed factor. Coarse benthic categories were analysed across all 18 sites to provide the largest spatial coverage of the MBI, however some sites were not sampled every year so temporal analysis of coral cover and composition used only the 12 temporal dredge monitoring sites that were visited in each year of the four-year monitoring program. Analyses were performed on a resemblance matrix created using a Euclidean similarity measure. All data were log(x+1) transformed to reduce the influence of the zero counts and large abundances (Anderson *et al.* 2006). Where significant differences occurred, pairwise comparisons were analysed to explore significant factors or interactions from the PERMANOVA. Results were also plotted using non-metric multidimensional scaling (nMDS) ordination, performed in Primer V6 and PERMANOVA+ with benthic vectors overlayed.

BEST subset modelling was used to assess which environmental variables and pressures best explained variance in the multivariate genera level coral composition. Models were conducted in DISTLM within PERMANOVA+ using a BEST model approach and Akaike Information Criterion modified for small sample size (AICc) (Anderson *et al.* 1994; Burnham and Anderson 2004). Three variables were considered in the BEST model analyses, which included (1) sea surface temperature anomaly (SSTA) (2) the number of days a dredge plume was recorded over each of the sites in 2010-2011, and (3) site depth. SSTA were generated from satellite data collected at 1km resolution on a monthly basis from the Erddap website (Simons 2017). Sediment plume coverage was generated through hotspot analysis of the cumulative daily digitised sediment plumes derived from MODIS imagery as described in Evans *et al.* 2012 (Chapter 2). Best models were selected based on the lowest AICc values with the fewest descriptor variables, that were within 2AICc units of the lowest AICc value (Burnham and Anderson 2004). The best models are presented in table format and in a distance-based redundancy analysis (dbRDA).

3.2.4 2010/2011 bleaching event

The February 2011 survey was conducted approximately four weeks after the onset of an extensive warm water anomaly that affected much of the WA coast (Moore *et al.* 2012, Pearce and Feng 2013). At this time, quantitative surveys were completed at 11 of the 18 dredge monitoring sites. The extent of bleaching at each site was assessed using a bleaching index adapted from McClanahan *et al.* (2004) in the western Indian Ocean (WIO), whereby the bleaching extent of coral colonies >5cm diameter was recorded within each photo quadrat. If several parts of a single colony were bleached in a quadrat, a single bleaching value for that colony was recorded, even if the colony extended outside the frame of the quadrat and included taxa that were not easily identified as an individual colony (i.e. *Acropora* thickets). For each coral colony, the genus was recorded, and the extent of bleaching scored from 1 to 6 where: 1 = Normal colouration, 2 = Pale, 3 = 0-20% of the coral colony bleached, 4 = 20-50% bleached, 5 = 50-80% bleached, 6 = 80-100% bleached (McClanahan *et al.* 2004). Colouration was compared with images taken from the same sites during surveys without any evidence of bleaching. At least 300 colonies were recorded at each site and a bleaching index (BI) was calculated for each genus based on the six categories of bleaching extent.

BI = $0c_1 + 1c_2 + 2c_3 + 3c_4 + 4c_5 + 5c_6$ /6 where c_1 to c_6 are the aforementioned six categories (% occurrence). The sum of the categories was divided by six to normalize the index to a 0–100 scale.

A single measure of bleaching susceptibility was calculated for each site by combining the relative abundance of each genera with the taxa-specific bleaching response. Susceptibility for each GMER site was also calculated from the bleaching response of coral genera in the WIO and the formulae presented in McClanahan *et al.* (2004). Bleaching indices for the MBI based on the local genera response (BI) were then correlated with those based on the WIO bleaching responses (site susceptibility) to see if the regional results are transferable globally. Differences in bleaching index amongst genera were normally distributed but did not meet homogeneity of variance, so were analysed with a one-way Analysis of Variance with a robust covariance matrix to determine robust standard errors in the *Car V3.0* package (Fox and Weisberg 2011) in R, where BI values for each genus at each site were used as replicates. Tukey's post-hoc test was calculated using the '*glht*' function in the *multcomp V1.4-8 package* (Hothorn *et al.* 2008) fitted with a heteroskedastic-consistent estimator *vcovHC* from the package *sandwich V2.4* (Zeileis 2004, 2006) in R. was calculated post-hoc to determine where differences between genera existed.

As the heatwave occurred during the dredging program at Barrow Island, a best subsets approach (Fisher *et al.* 2018) was used to assess the relative importance of thermal stress and the dredge plume on coral bleaching. Generalised additive models were used to account for non-linear relationships between the extent of coral bleaching of all corals, and three prominent genera that represent one susceptible and three relatively stress tolerant genera, with all possible combinations of the following three descriptor variables: 1) weekly sea surface temperature anomaly, generated from satellite data collected at 4 km resolution on a weekly basis, as described in Heron *et* *al.* (2010), 2) sediment plume coverage to February 2011 (date of bleach survey), generated through hotspot analysis of the cumulative daily digitised sediment plumes derived from MODIS imagery (Evans *et al.* 2012) and 3) site depths, which were considered as a potential environmental variable as it is often correlated with light availability (Michael *et al.* 2012) and is often an important predictor of coral bleaching extent during heat stress events (Moore *et al.* 2012).

3.3 Results

3.3.1 Benthic Composition

Coarse measures of benthic community composition across all 18 sites varied both spatially and temporally, as indicated by the interaction between year and site (**Error! Reference source not found.**). There were clear differences among sites, and temporal variation within sites was generally small, indicating stability in benthic communities over the survey period (Fig. 3.2). However, three sites (sites 18,26,27) showed some differentiation in the non-biotic features of sand and rubble between 2009 from the other years (Fig. 3.2). For example, there was more rubble in site 27 and more sand at site 26 in 2009, and site 18 had greater amounts of rubble in 2009 and 2010 than the latter two years, yet coral cover and composition remained quite stable particularly at site 18 (Figs. 3.3 & 3.4). Sites associated with high coral cover included sites 18, 19, 20, 26 and 28, while the sites with mobile substrates such as rubble and sand were typically dominated by large *Porites* bommies with sand interspersed between them (sites 8,16,17,14,31,32).

Table 3.2 Statistical summary of PERMANOVA assessing temporal and spatial variation in coarse measures of benthic cover at 18 sites visited inconsistently from 2009 to 2012.

	Source	Df	SS	MS	Pseudo-F	P(perm)	Unique perms
Benthic cover	Year	3	26.675	8.892	5.054	0.000	9931
	Site	17	438.45	5.791	19.484	0.000	9895
	Year x Site	43	76.15	1.771	4.516	0.000	9810

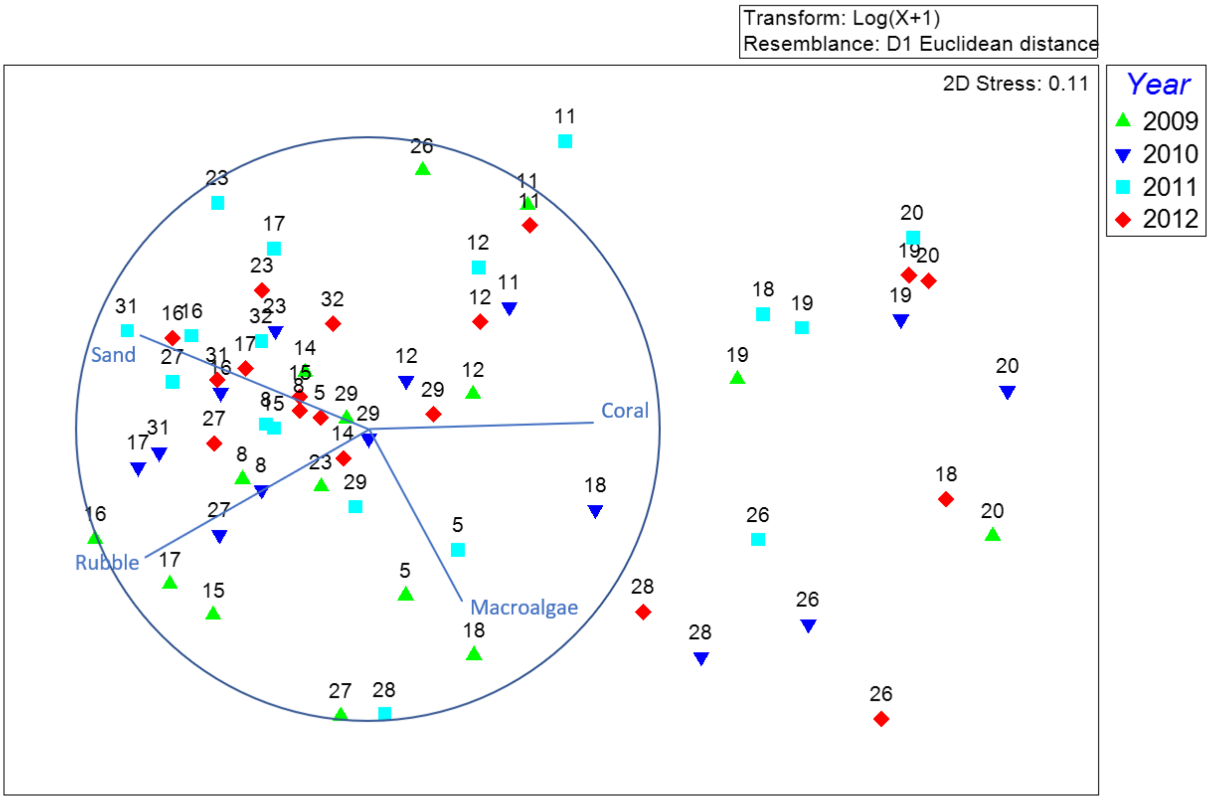

Figure 3.2 Non-metric multi-dimensional scaling plot of the spatial and temporal variation in coarse measures of benthic communities in the Montebello and Barrow islands at 18 sites. Only vectors with Pearson's correlation >0.3 are shown.

3.3.2 Coral Cover and coral composition

Hard coral cover varied among the 12 temporal dredge monitoring sites across years, although the extent of temporal variation differed among sites (significant time and site interaction Table 3.3). At most sites coral cover remained stable over the four survey periods, except at sites 19, 23, 27 and 29 (Table 3.3, Fig. 3.3). At site 19 coral cover declined through the study from 61% to 51% coral cover, so that 2012 was significantly less than 2009 ($p = 0.037$). At site 23, coral cover decreased significantly ($p= 0.032$) from 29% in 2009 to 20% in 2010 and further down to 13% in 2011 with a significant ($p= 0.03$) return to 18% in 2012 (Fig. 3.3). Coral cover at site 27 showed a similar trajectory though declines were less pronounced and were only significant between 2010 and 2011 ($p = 0.03$) (Fig. 3.3). A similar pattern of decline occurred at site 29 from 52% in 2009 to 44% in 2011 and then slight increase up to 50% so that 2012 was significantly greater than 2011 ($p = 0.037$). Notably, similar trends occurred at site 26, though no significant differences were detected. However, over-dispersion, high variance about mean values in each year and p values >0.01 for all post-hoc analyses suggest these temporal trends should be interpreted conservatively (Fig. 3.3).

Table 3.3 Results of the PERMANOVA on the spatial and temporal variation of the univariate coral cover and multivariate composition based on data from the twelve dredge monitoring sites in 2009, 2010, 2011 and 2012.

Dependent variable	Source	Df	SS	MS	Pseudo-F	P(perm)	Unique perms
Hard coral cover	Year	3	0.218	0.073	1.538	0.227	9955
	Site	**11**	**27.711**	**2.519**	**12.018**	**0.0001**	**9927**
	Year x Site	**33**	**1.563**	**0.047**	**4.058**	**0.0001**	**9898**
Coral genera	**Year**	**3**	**7.221**	**2.407**	**1.501**	**0.008**	**9837**
	Site	**11**	**1324.900**	**120.450**	**23.718**	**0.000**	**9862**
	Year x Site	**33**	**52.942**	**1.604**	**1.182**	**0.003**	**9585**

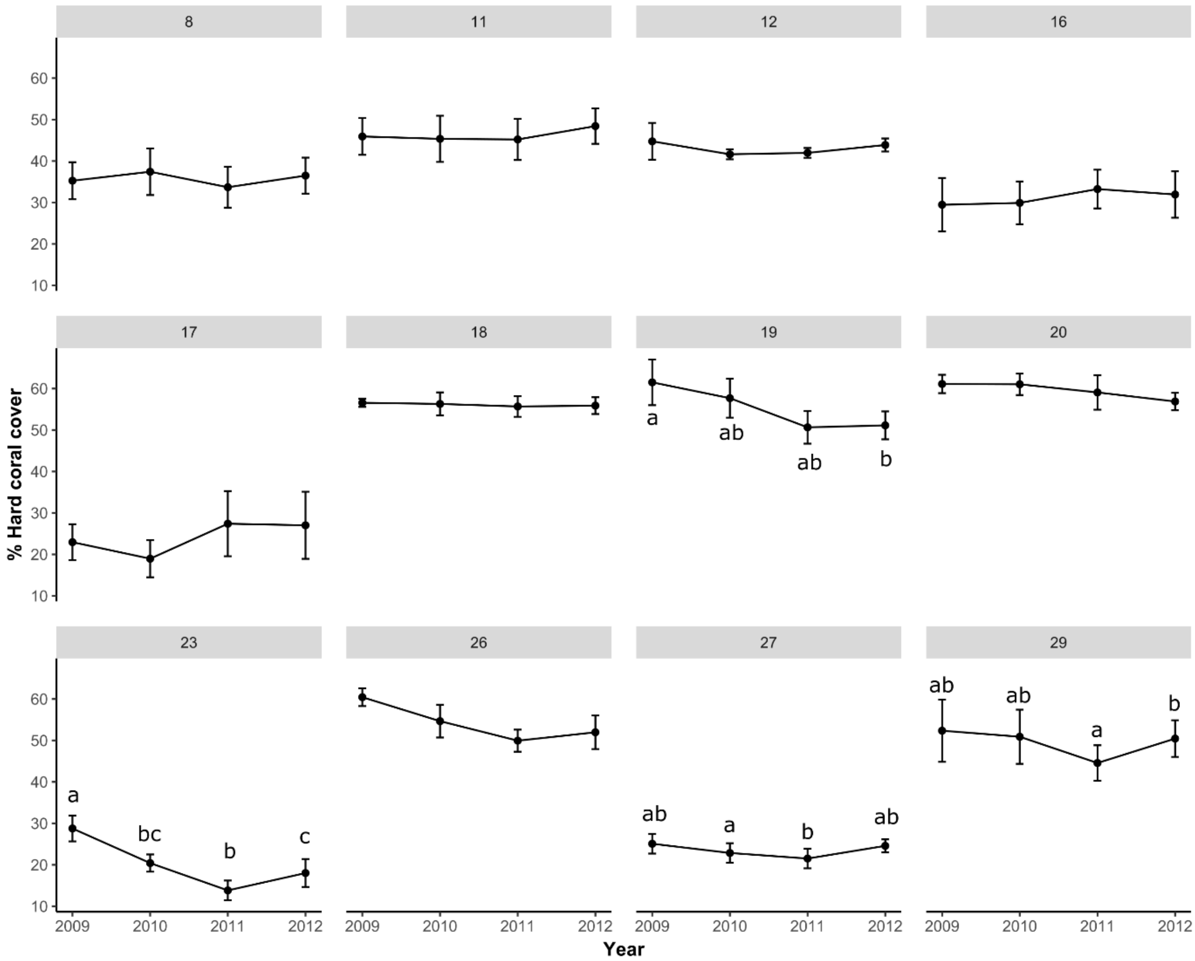

Figure 3.3 Temporal trends in average coral cover (+/- 1 S.E.) at the 12 temporal dredge monitoring sites. Letters show significant groupings based on pairwise comparisons in PERMANOVA. Coral cover at sites without letters did not vary significantly among survey years.

Coral composition within sites remained constant through time at most sites, though there were major differences in coral communities among sites (Fig. 3.4, Table 3.3). A significant interaction term (Table 3.3) was driven by temporal differences at site 27 between 2009 and 2011 (t = 1.909, p = 0.039). The changes are not overall community changes, rather they relate to higher than normal occurrence of rare genera in 2011, e.g. *Pectinia*, *Echinophyllia*, *Leptoseris*. Coral composition at sites 18 and 19 were dominated by *Acropora*, while several sites (11,12,14,16,17), and to a lesser extent site 20, were dominated by large *Porites* bommies, (Fig. 3.4). The rest of the sites (20, 26, 27, 29) were composed of a mixed assemblage.

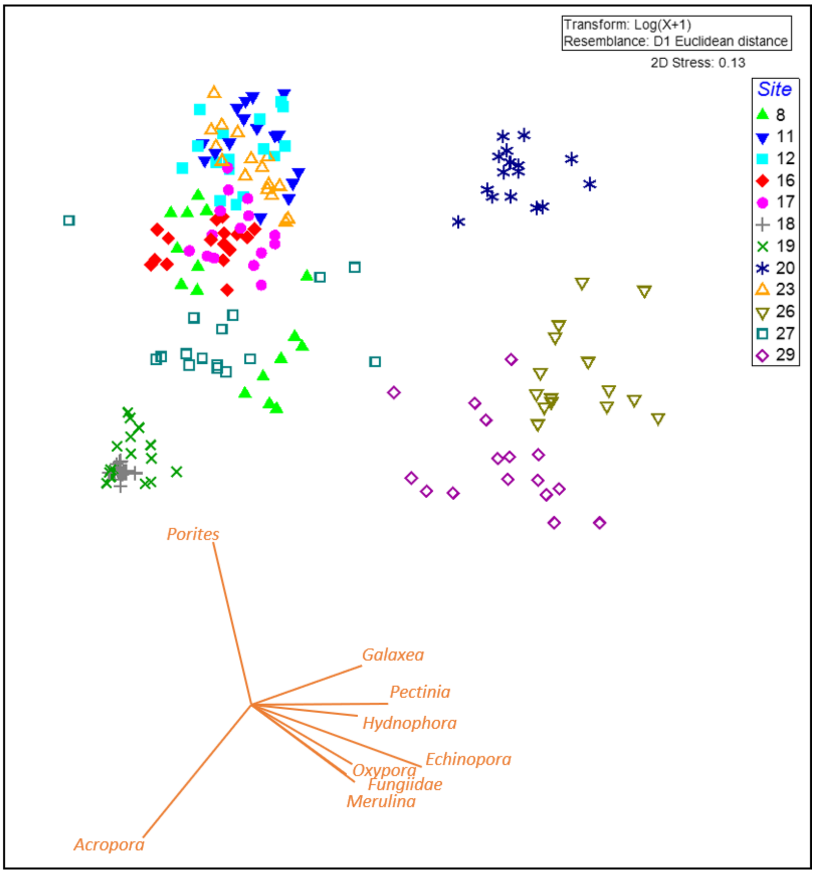

Figure 3.4 Non-metric multidimensional scaling plot of the spatial and temporal variation in coral composition based on coral genera at the 12 sites from 2009 to 2012. Vectors show Pearson's correlation > 0.5.

Variation in coral composition at the MBI is best explained by a model that includes depth and dredge days, although the most parsimonious model is the single variable model that only includes depth (Table 3.4). *Porites* was more prevalent at the deeper sites, whilst *Acropora* and *Echinopora* were characteristic of coral communities at the shallow sites (Fig. 3.5). The number of dredge days correlated strongly with the y-axis, although this explains only 1.4% of the variation.

Table 3.4 Multivariate results within two AICc scores of the Best fit DISTLM model for the coral genera of the Montebello and Barrow islands based on the 12 temporal dredge monitoring sites from 2009-2012.

Predictor variables	No.Vars	RSS	AICc	ΔAICc	R^2
Depth, Dredge days	2	1395.5	384.98	0	0.191
Depth	1	1420	386.23	1.25	0.177
Depth, Dredge days, SSTA	3	1389.7	386.28	1.3	0.194

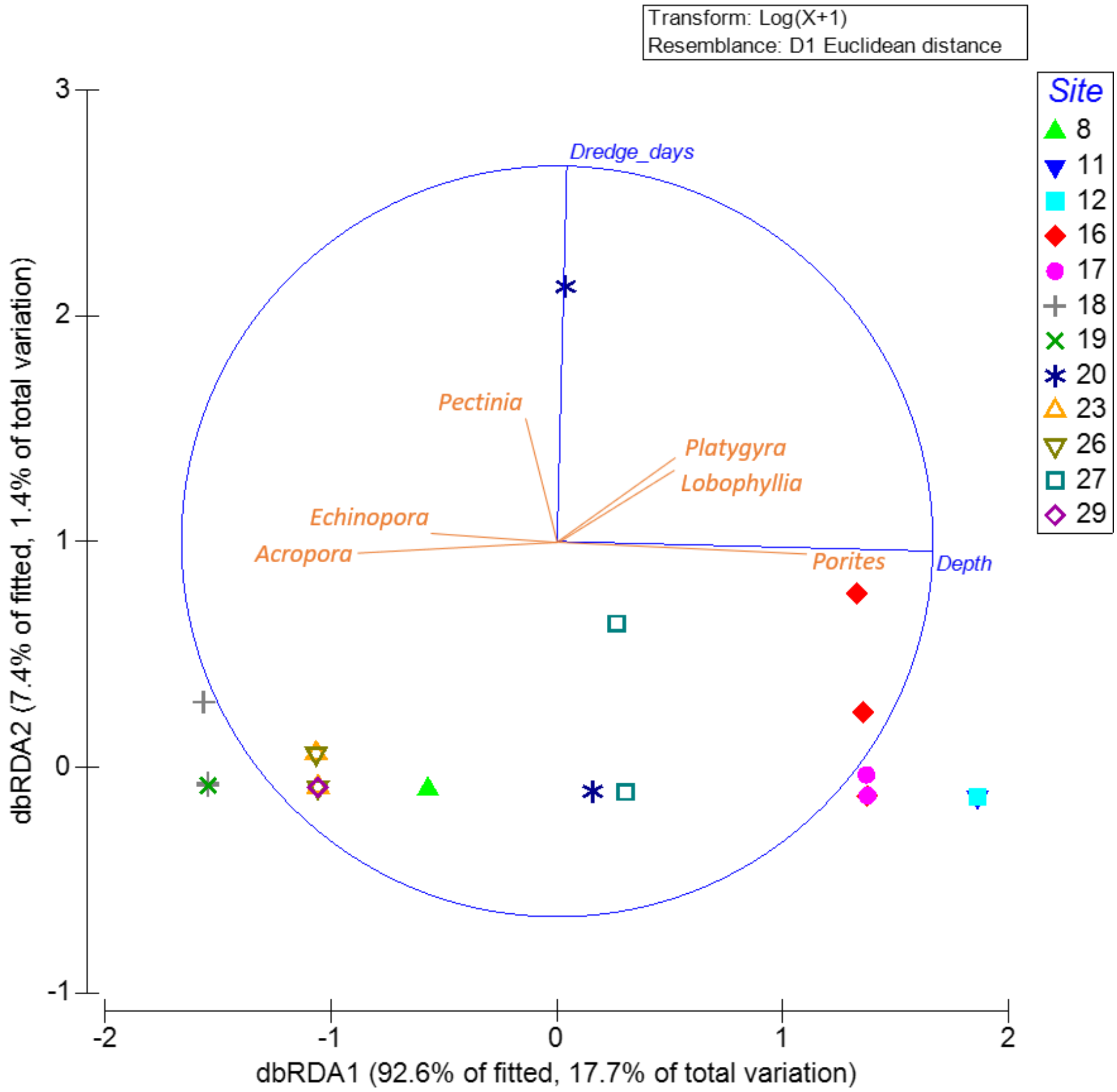

Figure 3.5 Distance based redundancy analysis (dbRDA) of the coral composition (genera) for the twelve temporal dredge monitoring sites from 2009-2012. Blue vectors show the two best descriptor variables derived from DISTLM (Table 3.5). Orange vectors show corals with Pearson's correlation r values > 0.3.

3.3.3 2010/2011 Bleaching Event

A total of 3,373 corals representing 40 genera were surveyed in February 2011 to assess bleaching across eleven sites. The proportion of corals bleached was lowest at sites 17 (25%) and 16 (36.5%) and highest at sites 31 (96%) and 23 (92%) (Table 3.5). Accordingly, bleaching indices calculated from field data at MBI were low at sites 17 (7.11) and 16 (10.87), but highest at sites 8 (50.32) and 31 (48.40). A high index at site 8 was due to a more advanced level of bleaching for each of the colonies surveyed at that site (Table 3.5). Site level indices of bleaching were also high at the three southernmost sites of the Barrow Shoals (26, 31 and 23) and sites 29 and 8, which were fringing reefs in relatively shallow water. The lowest bleaching indices were recorded in the northernmost site of the southern Barrow Shoals (site 20) and the southernmost sites of the Lowendal Shoals (sites 27,16,17) which were all deeper than 5.5m (Table 3.5).

Table 3.5 Number and percentage of bleached coral colonies at 11 sites in the Montebello and Barrow islands during the temperature anomaly in February 2011. Bleaching was recorded from still images taken of each transect. Site susceptibility for each site calculated from bleaching response of coral genera in the western Indian Ocean (WIO) and formulae presented in (McClanahan et al. *2004). Bleaching index calculated form the same formulae but using response values and cover of genera at each site in February 2011.*

Site Number	**26**	**31**	**23**	**20**	**27**	**29**	**16**	**17**	**18**	**19**	**8**
No. of colonies											
Normal	29	13	24	133	86	37	192	230	87	41	36
Pale	75	86	47	110	78	53	56	41	70	152	72
0-20% bleached	53	54	50	25	35	97	36	26	82	63	23
20-50% bleach	37	34	61	14	17	55	6	6	48	17	18
50-80% bleached	27	15	75	5	14	20	9	0	15	18	39
80-100% bleached	80	110	49	17	0	51	3	4	10	10	121
Dead	2	0	0	0	0	0	0	0	0	0	0
Total number of colonies	303	312	306	304	304	313	302	307	312	301	309
% of colonies											
Normal	9.57	4.17	7.84	43.75	28.29	11.82	63.58	74.92	27.88	13.62	11.65
Pale	24.75	27.56	15.36	36.18	25.66	16.93	18.54	13.36	22.44	50.50	23.30
0-20% bleached	17.49	17.31	16.34	8.22	11.51	30.99	11.92	8.47	26.28	20.93	7.44
20-50% bleach	12.21	10.90	19.93	4.61	5.59	17.57	1.99	1.95	15.38	5.65	5.83
50-80% bleached	8.91	4.81	24.51	1.64	4.61	6.39	2.98	0.00	4.81	5.98	12.62
80-100% bleached	26.40	35.26	16.01	5.59	0.00	16.29	0.99	1.30	3.21	3.32	39.16
Dead	0.66	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Depth	3	3	3	5.5	5.8	3	8	8	2	2	4
Number of genera	22	24	22	26	24	17	26	24	7	16	17
Percent of colonies bleached	90.43	95.83	92.16	56.25	47.37	88.18	36.42	25.08	72.12	86.38	88.35
Bleaching index	44.66	48.40	47.66	16.83	13.98	39.78	10.87	7.11	26.07	24.97	50.32
Site susceptibility (WIO)	12.67	10.34	6.98	10.31	11.26	12.54	12.83	10.85	18.85	17.25	11.00

Spatial variation in the site bleaching susceptibility based on the WIO taxa specific bleaching response (McClanahan *et al.* 2004) was inconsistent with the bleaching index based on local coral responses recorded at the MBI (Table 3.5, Fig. 3.6). The highest predicted susceptibilities were at sites 18 (18.85) and 19 (17.25), the *Acropora* dominated sites, while the lowest was recorded at site 23 (6.98), a *Porites* dominated site (Table 3.5, Fig. 3.6). The rest of the sites registered relatively similar site susceptibilities between 10 and 13. The bleaching index based on local coral responses showed a great deal more variation between the sites and there was no significant relationship between the two methods, which suggests region specific coral based bleaching indices are required (Fig. 3.6).

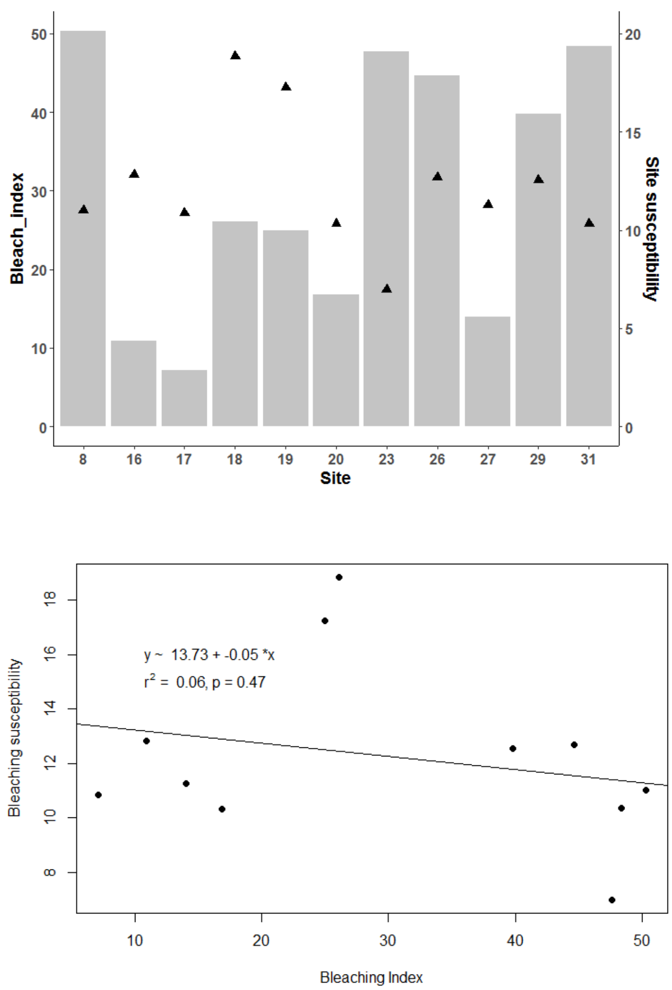

Figure 3.6 In-situ bleaching index (Bars) and the modelled site susceptibility (triangles), and the correlation between them at eleven sites in the Montebello Islands in the 2010/2011 bleaching event. Site susceptibility for each site calculated from bleaching response of coral genera in the Western Indian Ocean and formulae presented in McClanahan et al (2004). Bleaching index calculated form the same formulae but using response values and cover of genera at each site in February 2011.

Bleaching indices varied significantly among coral genera (ANOVA: $F_{(19,160)} = 3.03$; $p<0.0001$), post-hoc testing indicating three overlapping groups of similar value. (Fig. 3.7). Four of the genera (*Galaxea*, *Seriatopora, Pocillopora* and *Monitpora*) recorded a bleaching index >40, while another four genera (*Fungia*, *Lobophyllia*, *Pectinia* and *Oxypora)* had bleaching indices <20. *Acropora*, identified as one of the most susceptible genera in other studies (McClanahan *et al.* 2004, Hoey *et al.* 2016), received a lower than moderate bleaching index of approximately 25 (Fig. 3.7).

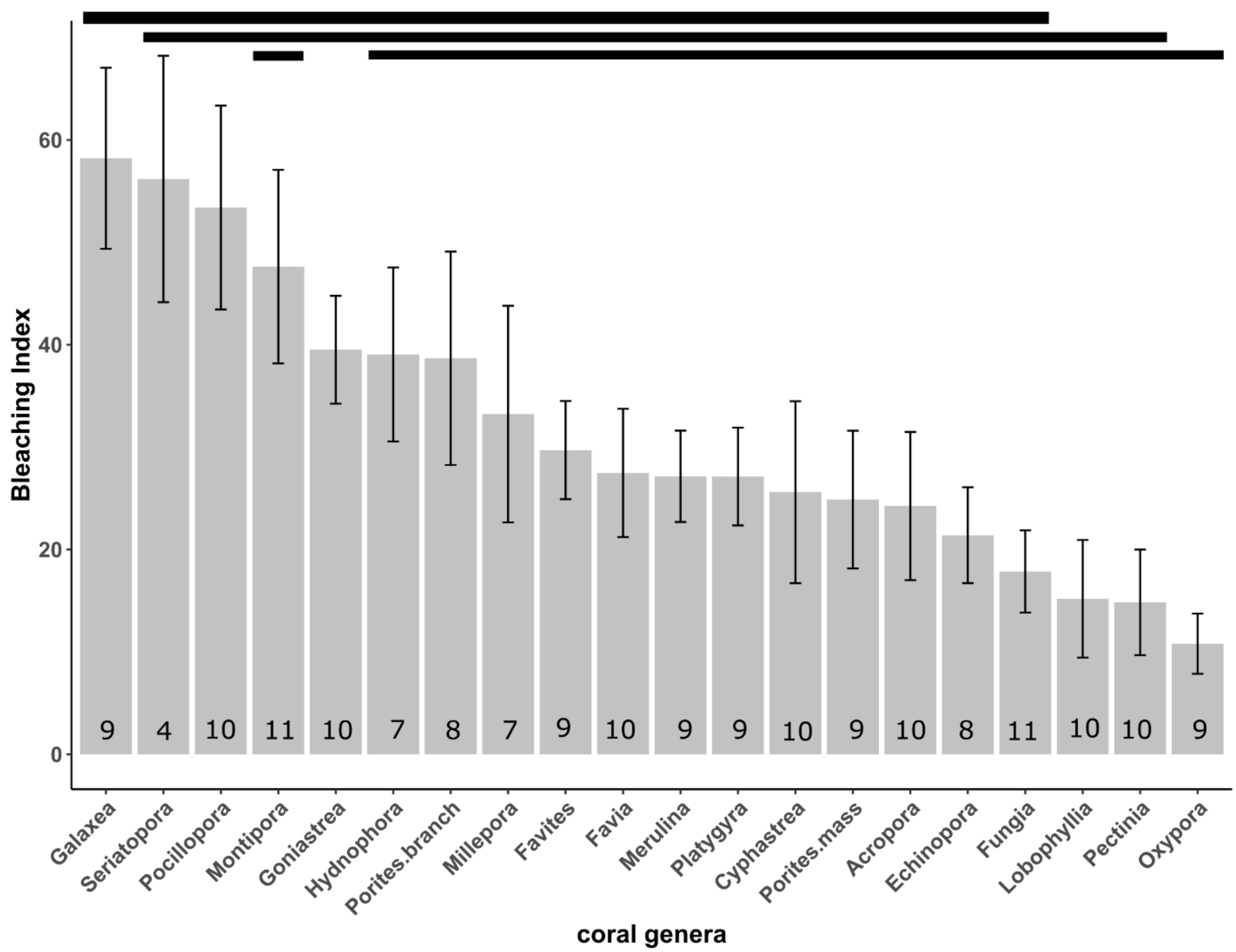

Figure 3.7 Average (+/- S. E.) bleaching indices for genera recorded across the possible 11 sites sampled during the temperature anomaly in February 2011 with results of post-hoc testing indicating four overlapping groups of coral genera. Numbers at base of bars are the number of sites where the coral was observed (either bleached or unbleached). Horizontal bars represent Tukey's post-hoc groupings.

Spatial variation in the bleaching index for most of the response variables in this study, more so for ‘all coral’ and *Favites*, was best explained by depth differences among sites, although for *Acropora* and *Porites* the null model was within 2AICc scores. *Favia* was the exception, best explained by weekly SSTA. The most parsimonious models explained high amounts of the variance in the data ranging from 78 – 85% (Table 3.7). The best models for ‘all coral’ bleaching index and *Favites* bleaching index found that bleaching was highest at shallower sites and declined with depth, while *Favia* bleaching index increased with rising sea surface temperature anomaly (Fig. 3.8, 3.9).

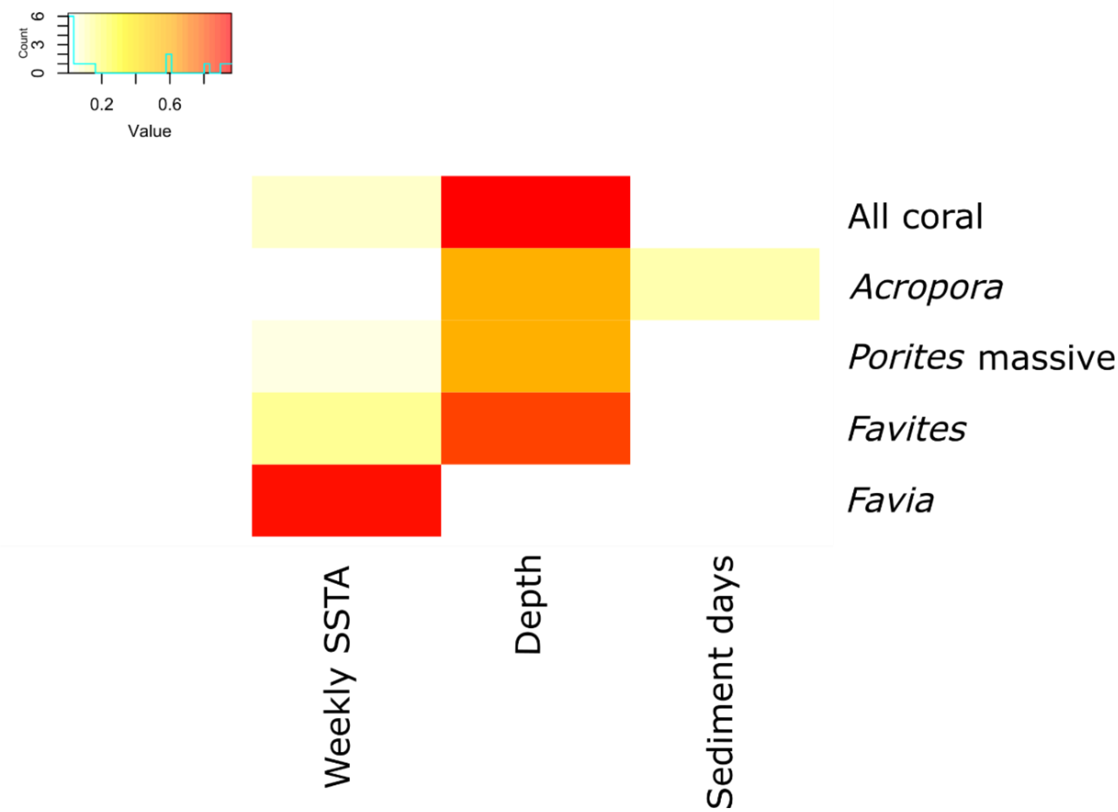

Figure 3.8 Weighted variable importance plots of the descriptor variables for bleaching of total coral and the most abundant genera.

Table 3.6 Results of the best subset models using a general additive model showing the drivers of all coral bleaching index, as well as three abundant coral genera representing one susceptible coral and three relatively heat tolerant genera.

Bleaching Index Response variable	AICc	Delta AICc	R^2	Descriptor variables
All Coral	**89.42**	**0**	**0.78**	**Depth**
Acropora	44.71	**0**	0.74	Depth
	46.33	**1.62**	**0**	**Null**
***Porites* Massive**	25.97	0	0.84	Depth
	27.26	**1.29**	**0**	**Null**
Favites	**70.53**	**0**	**0.85**	**Depth**
Favia	**14.83**	**0**	**0.84**	**SSTA**

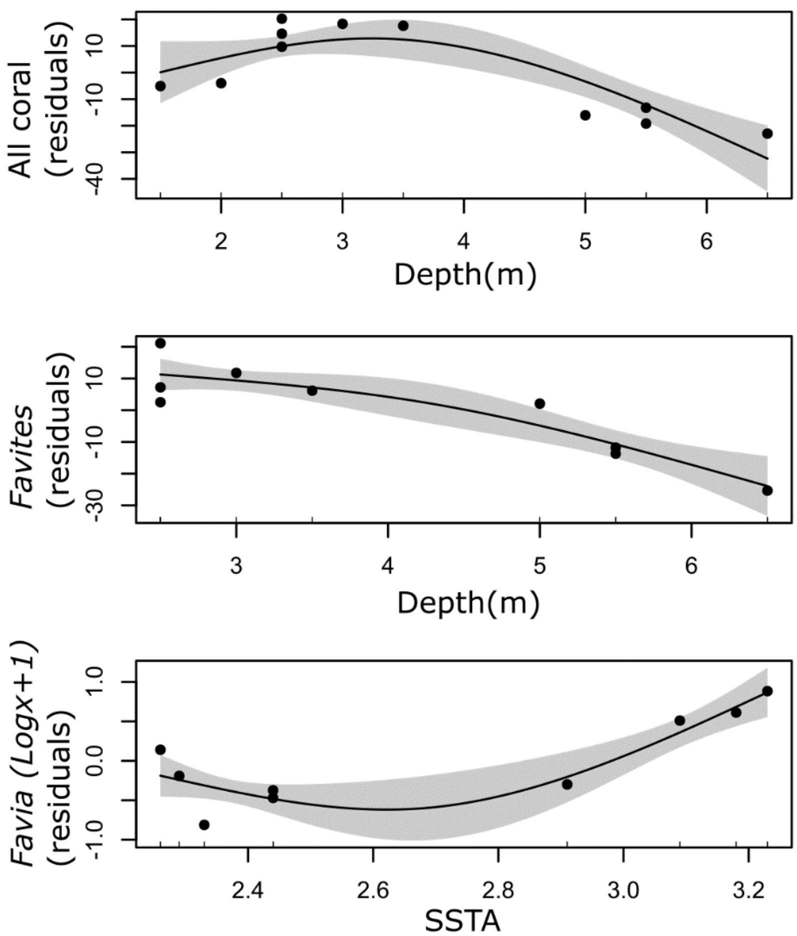

Figure 3.9 Biplots of the best descriptor variables against bleaching index of all coral genera, and the bleaching index for Favites *and* Favia*.*

3.4 Discussion

The MBI are characterised by a diverse range of habitats reflecting a complex geomorphology and variability in environmental conditions. Accordingly, coral cover and composition is spatially variable across the wave exposed bombora and more protected shallow water reefs. Surveys presented in this report further our understanding of the spatial and temporal variation of coral communities in the region and provide a strong baseline for future monitoring of these communities. Of the environmental factors considered in this study we found depth to be the most important determinate of coral community composition, while the number of dredge plume coverage days may have had a minor influence at sites close to the dredge operation. Reef depth also moderated the effect of abnormally high sea surface temperatures on corals in 2011, whereby deeper water reefs experienced lower rates of bleaching.

Coral cover remained relatively stable over the study period, and we conclude that although the dredge plume and warm water event had some influence on corals, the effect of these impacts did not unduly alter coral cover and composition at the 12 monitoring sites. However, the scope of this study does not provide knowledge of the sub-lethal effects of these disturbances and how they impacted the long-term resilience of the ecosystem.

3.4.1 Spatial variation in the MBI

Among the reefs surrounding the Montebello Islands, there is a gradient in coral assemblages from the shallow *Acropora* dominated reef flats of the western Montebello reef flats to the large *Porites* dominated deeper sites on the eastern fringe of the islands. Between these areas, sheltered fringing reefs and reefs within channels experience high tidal movement. The Lowendal Shelf (Fig. 3.1) to the south of the Montebello Islands also has shallow *Acropora*-dominated reef flats in the southwest trailing off to deeper *Porites* bommies in the east. Further south, the reefs of the Barrow Shoals are generally shallower and less complex in structure, potentially due to protection from ocean swells and limited fetch from the wave generating easterly winds (See chapter 2). These sites are a mixture of communities often dominated by either *Acropora* or *Echinopora*. The habitat is characterized by discontinuous reef bombora communities rising from deeper substrates in depths of 5-10m. The physical differences in each location of the MBI and associated environmental conditions reflect the variable taxonomic characterisation of the coral communities between reef types.

Comprehensive and adequate representation of diversity within marine reserves infers that a range of communities be included within the park boundaries. The MBIMCR are divided into different management zones that allow different activities (DEC 2007). Montebello Islands Marine Park includes two large sanctuary (no-take) zones that afford protection from extractive activities to diverse types of coral reef and associated invertebrate and fish communities. However, on the southern Lowendal Shelf and Barrow Shoals in the Barrow Island marine management area, which are closest to the pressure associated with development activities in the MBI, there are currently no management zones that confer a similarly high level of protection (DEC 2007). This is notable as the reefs of the eastern margin of the Lowendal Shelf and Barrow Shoals contain unique coral communities that are not represented elsewhere in the no-take areas of the region. This omission potentially undermines the representativeness of the protection of reef communities with the marine reserves.

3.4.2 Temporal variability of coral communities

Coral mortality in the immediate vicinity of dredging can be caused by the deterioration of water quality associated with suspended and settling sediment (Fisher *et al.* 2017). Furthermore, sub-lethal effects of sedimentation on coral, such as increased energy investment in sediment shedding (Duckworth *et al.* 2017) through mucous sheet production (Bessell-Browne *et al.* 2017), decreased growth rates (Crabbe and Smith 2005), reduced sperm and egg viability (Ricardo *et al.* 2015, 2016a, 2016b), and reduced recruitment (Babcock and Smith, 2002), have been demonstrated in laboratory experiments. Despite a dredging operation, several cyclones and a heat wave over the duration of this study, coral cover and community composition remained stable across most of the twelve dredge monitoring sites. Significant declines in coral cover were detected in 2011 at a couple of sites, however these had partially recovered by 2012. The area consistently influenced by the dredge plume over the 18-month duration of the dredging program was restricted to within 10km south of the dredge site (Evans *et al.* 2012), though water quality was occasionally affected up to 20km away (Fisher *et al.* 2015). As the monitoring sites in this study were either outside, or on the periphery of the major area of dredge influence, this study had limited capacity to investigate the impacts of dredge plumes on coral communities. Nonetheless, low exposure to dredge plume at our monitoring sites (max. 54 days at time of bleaching), may have affected coral composition at the four sites in our temporal study nearest the dredge operations as shown in the DISTLM and dbRDA. However, the finding is weak due to minimum change of coral cover observed and the amount of dredge plume exposure at our sites, so this result should be interpreted with caution.

Furthermore, the period of the dredging operations and associated plume presence coincided with several natural pressure events which complicate any apportioning of changes to coral cover or community complexity to dredging. In January and February 2011, an anomalous warm water event (Pearce *et al.* 2011) caused extensive coral bleaching along the WA coastline, including parts of the MBI and as far south as Perth (Moore *et al.* 2012; Depczynski *et al.* 2013). During the same summer, three cyclones passed within close proximity to the MBI (Moore *et al.* 2012), causing increased wave activity across the normally sheltered sites on the east coast of the MBI and causing resuspension of natural and/or dredge operation sediments at six of the survey sites (Evans *et al.* 2012). Cyclones generally have a negative effect on coral health (Harmelin-Vivien 1994), but as these cyclones did not pass directly over the MBI, they may have increased water mixing (Carrigan and Poutinen, 2014), and reduced stress from solar radiation (Hughes *et al.* 2017), which alleviated the impacts of thermal stress. Cyclone-associated cloud cover and the wind and wave driven resuspension of sediments over large areas at this time also inhibited effective satellite image-derived estimates of dredge plume coverage over the monitoring sites (Evans *et al*. 2012), further complicating our ability to apportion drivers of change in coral communities.

3.4.3 The effect of the 2010/2011 bleaching event on the corals of MBI

Coral bleaching in the MBI decreased in severity at sites with greater depth and increased with greater SSTA in this region. Dredge plume days did not feature strongly in the models. The importance of depth has been confirmed over much broader spatial scales (Moore *et al.* 2012), likely due to depth moderating coral exposure to UV light which contributes to bleaching (Zepp *et al.* 2008, Fabricius *et al.* 2007). Further, coral composition also varies with depth in the MBI, and taxa resilient to heat stress such as *Porites* are commonly found at deeper sites (Zinke *et al.* 2018).

Reports from the recent 2016/17 global bleaching events (Hughes *et al.* 2017, 2018, Le Nohaïc *et al.* 2017; Couch *et al.* 2017, DeCarlo *et al.* 2017, Monroe *et al.* 2018, Lough *et al.* 2018) support predictions of an increase in the regularity and severity of marine heatwaves (Hoegh-Guldberg 1999). The ecological consequences of coral bleaching can be dramatic. The most severe warm water events cause wide spread coral mortality, with flow on effects to reef associated fish and invertebrates (Graham *et al.* 2006, Pratchett *et al.* 2008, Williamson *et al.* 2014, Pratchett *et al.* 2018, Stuart-Smith *et al.* 2018) as well as ecosystem services (Pratchett *et al.* 2014). Therefore, it is important for managers to understand the likelihood of both bleaching and mortality of coral communities at a range of spatial scales. In this study, 4.5 degree heating weeks (DHW) reached was the maximum, relatively low compared to more recent bleaching events, and so the mortality associated with bleaching was limited. However, more prolonged warm water thermal anomalies in 2013 (7.4 DHW) resulted in up to 69% coral mortality in the MBI (Ridgway *et al.* 2016), emphasizing the increasing threat of climate change to corals. Understanding variability in coral response to thermal events allows managers to recognise areas where coral communities or specific taxa are more susceptible to bleaching, potentially requiring greater management control over other potential stressors. For example, increased dissolved nitrogen has been associated with higher coral bleaching (Wooldridge 2009) and controlling nutrients levels around reefs could reduce the impact of warm water anomalies on corals (Wooldridge *et al.* 2017). In comparison, other areas may be characterised by a suite of physical characteristics that moderate elevated seawater temperature events limiting the extent of bleaching and providing potential refugia for corals in the event of warm water events, such as upwelling, rapid flow channels, turbidity and tolerance due to regular exposure to air and heat (reviewed in West and Salm 2003).

Assessments of reef resilience and vulnerability to climate change will be partially based on the coral community and their susceptibility to heat stress. Susceptibility of coral genera to bleaching recorded in this study are somewhat different from those reported at the Great Barrier Reef and Kenya (McClanahan *et al.* 2004). Consequently, measures of reef vulnerability to heat stress calculated from MBI data differ to those calculated from information collated from other locations. Spatial variation in susceptibility of coral taxa to bleaching may relate to colony size (Pratchett *et al.* 2013), the zooxanthellae corals harbour (Sampayo *et al.* 2008), the extent and duration of heat stress, levels of UV radiation (Gleason and Wellington 1993; Hoegh-Guldberg 1999), and oceanographic (Wolanski *et al.* 2017) or atmospheric conditions (Hughes *et al.* 2017). Furthermore, bleaching susceptibility can vary temporally within locations, due to prior exposure to heating (Brown *et al.* 2002; Ainsworth *et al.* 2016). Thus, a comprehensive understanding of local environmental, disturbance history and community composition is required when identifying resilient and vulnerable reefs for spatial planning.

3.4.4 Summary

This study characterised coral reef communities of the MBI across a range of habitats and environmental conditions. There was a strong locational divergence among coral communities that was associated with depth, though environmental factors such as wave exposure and turbidity may also be important in this region (see Chapter 2). Coral cover remained generally stable at most sites for the duration of the study, although there is a suggestion that dredging has influenced the composition of corals at some sites. We postulate that this equivocal finding is because effected sites are on the periphery of the dredge plume, in an environment of inherently high turbidity. Corals on these reefs are therefore adapted to turbid conditions and low levels of exposure to the dredge plume were insufficient to have a large impact. The warm water event in the summer of 2011 further confounded interpretation of dredge impacts on corals. Mortality following this heat event was low, but coral health may have been compromised and elevated water temperature in subsequent years have likely contributed to further coral decline in the region (Ridgway *et al.* 2016; Lafratta *et al.* 2016). To better understand the long-term consequences of climate change, dredging, and ongoing local stress, such as maintenance dredging, long term monitoring of corals and environmental conditions in the region is required. The data presented here provides a strong foundation for this type of monitoring program.

4 Description of Demersal Fish Assemblages of the Montebello and Barrow Islands.

4.1 Introduction

Approximately 456 marine fish species from 75 families inhabit the Montebello and Barrow Islands (MBI) area (Allen 2000). A few are endemic but most of these species have a wide distribution through the Indo-west Pacific region (Allen 2000). To date, two published taxonomic descriptions have suggested the Montebello Islands are similar to the Dampier Archipelago in terms of species abundance and diversity (Allen 2000; Hutchins 2001). However, there has been no examination of how the distribution and abundance of fish communities relate to environmental variables at the MBI.

The fish community of the MBI has been partially protected from extensive and sustained fishing by the remote nature of the archipelago. However, amplified awareness of the marine communities through increased oil and gas activities on and around the islands, an ongoing commercial and charter fishery and the expansion of populations in coastal Pilbara towns, is likely to be causing increasing fishing pressure. Moreover, tropical fish assemblages are inherently related to habitat structure (Pratchett *et al.* 2008; Wilson *et al.* 2014) and pressures such as thermal stress, severe storms, coral predators (e.g. *Acanthaster planci* and *Drupella* spp.), disease and introduction of invasive species pose a threat to prominent fish habitats around the MBI. Importantly, the status of finfish communities is identified as a key performance indicator for management of the Montebello and Barrow Islands marine conservation reserves (MBIMCR) (DEC 2007). It is therefore imperative that the condition of fish is monitored relative to natural, anthropogenic and climate related disturbances as well as different management zones in the reserves.

This chapter presents an analysis of fish data collected in the MBI region from 2009 to 2012, investigating the temporal and spatial distribution of fish communities relative to habitat. The impact of the 18 months of dredging activity associated with the Gorgon gas project on fish is also considered. Survey sites were however located outside the Barrow Island port area, where most dredging pressure were likely to have been experienced (Fisher *et al.* 2015). Nonetheless, surveys investigated whether the effects of dredging could be detected at sites of low plume coverage and provide a basis for long-term fish monitoring in the MBI marine reserves. We also examined the effect of the anomalous water warming and subsequent coral bleaching events in Jan/Feb 2011 (Pearce *et al.* 2011, Moore *et al.* 2012, Pearce and Feng 2013) on fish feeding guilds, and determined if there are detectable impacts of fishing on target species. Quantification of these pressures provides an important baseline for the future, should pressure associated with extractive fishing activities or water temperature anomalies increase.

4.2 Methods

4.2.1 Study location and survey timing

In 2006, finfishes were qualitatively surveyed at 26 sites using underwater visual census (UVC) methods (Bancroft 2009). Of these sites, a subset of 15 'dredge monitoring sites' (Fig 4.1) were chosen to be incorporated into the Gorgon monitoring, evaluation and reporting (GMER) fish sampling program, which was established to investigate the impacts of dredging activities on the marine communities of the MBIMCR. Sites were selected to represent a gradient of pressure from the dredging activities by locating them at increasing distance from the dredging site. Although three sites were in no-take sanctuary zones (hereafter 'protected'), the rest were open to some level of fishing (hereafter 'fished') (Table 4.1, Fig. 4.1). These sites were surveyed annually from 2009 to 2012, although only a subset of ten 'primary sites' were surveyed on all four sampling periods due to bad weather and/or poor visibility (Table 4.1). Only eight of these sites had corresponding data collected when coral bleaching occurred due to anomalously high water temperature. Fish data was collected in the MBIMCR before (October 2009, April 2010), during (May 2011) and after (May 2012) the dredging program (Table 4.1). This sampling design provided a basis for assessing the impact of dredging on fish, however it is possible that seasonal variation between spring 2009 and autumn 2010, 2011 & 2012 surveys influenced results (Bijoux *et al.* 2013).

Table 4.1 Sampling regime for the collection of digital video footage for the fish community. 09 – November 2009; 10 – April 2010 surveys were before dredging; 11 – May 2011 was during dredging and 12 – May 2012 was after bleaching. (Y – quantitative surveys completed). Sites surveyed all four years from 2009 to 2012 are included in the temporal study (a). Sites used in target genera analyses (t). Sites with coral bleaching data used in the DistlM (d).

Site	Location	Protective Status	Analyses	Survey Periods			
				09	10	11	12
8	Stephenson's Channel	Fished	a,t,d	Y	Y	Y	Y
11	W Ah Chong Is	Protected	a,t	Y	Y	Y	Y
12	E Black Rock	Protected	a,t	Y	Y	Y	Y
16	SE Lowendal	Fished	a,t,d	Y	Y	Y	Y
17	E Lowendal	Fished	a,t,d	Y	Y	Y	Y
18	S Lowendal	Fished		Y	Y		Y
19	Wonnich Reef	Protected	a,t,d	Y	Y	Y	Y
20	E Dugong Reef	Fished	a,t,d	Y	Y	Y	Y
21	Central Dugong Reef	Fished		Y	Y		
23	S Batman Reef	Fished	a,t,d	Y	Y	Y	Y
26	Central E Barrow Is. Shoals	Fished	a,t,d	Y	Y	Y	Y
27	W Lowendal Shelf	Fished		Y	Y		Y
28	SE Dugong Reef	Fished		Y	Y		
29	N Double Isl	Fished	a,t,d	Y	Y	Y	Y
31	Central E Barrow Is. Shoals	Fished			Y	Y	Y

4.2.2 Survey procedure

Finfishes were surveyed at each location on scuba using stereo-Diver Operated Video (stereo-DOV). Stereo-DOV surveys were conducted by trained operators using a standardised approach adopted by the Department of Biodiversity, Conservation and Attraction (DBCA) marine monitoring program. Six replicate 50 x 5m belt transects were conducted at each site, each transect taking 5-8 minutes to complete. The stereo-DOV units consisted of two high definition Canon HG21 digital video cameras mounted in stereo configuration on a neutrally buoyant metal bar and were swum along the transect line at a height of approximately 0.5-0.7m above the substrate. The cameras were held in underwater housings and angled inwards at approximately 8°, resulting in the maximum field of view (Harvey and Shortis 1998). The stereo-DOV cameras always remained facing forward along the transect line, inclined slightly toward the substrate, avoiding swinging the cameras to include fishes that may not have been detected in the forward-facing field-of-view (Holmes *et al.* 2013). This technique has been adopted to decrease bias associated with different operators using slightly different search patterns with the stereo-DOV cameras and is a standardised approach used by DBCA's state-wide marine monitoring group.

Video analyses

Stereo-DOV videos were analysed in the laboratory following the completion of field activities by a trained video analyst with experience in processing videos from a variety of locations. The analyst was provided full access to reference material, as well as the option of referring unknown identifications to taxonomic experts. Videos were analysed using the software Event Measure v3.13 (SeaGIS 2016) with the 'Stereo Length Measure' component installed. All individuals observed within the 5m transect belt on the videos were identified to the lowest possible taxonomic level.

Benthic data

Benthic data was collected along four 50m transects at each site. Digital still images were taken every metre along the transects using a Canon G12 PowerShot High Sensitivity 10mp camera held perpendicular to the substrate at a fixed height of 1m, providing an image area of 0.85m^2 and a total area of ~42.5m^2 per transect. Six random points were overlayed onto each image using ECOPASS software and each point classified to the highest taxonomic level possible using predetermined hierarchies representing all possible benthic classifications, the presence of bleaching, physical damage and disease. For details of image analysis see Chapter 3.

Sediment data

Measures of suspended sediment stress used in this chapter are based on the number of days the dredge plume was present at each site. This measure was determined by digitisation of remote sensing MODIS products reported in Chapter 2 of this report and published by Evans *et al.* (2012).

Measure of fishing effort in the MBI

There is no site-specific measure of commercial or recreational fishing pressure available for the MBI area, however information on charter fishing provided by WA's Department of Primary Industries and Regional Development (DPIRD) and fly-over data provided by the Australian Customs and Border Protection Service (Customs) has been used as a proxy for recreational visitation and potential fishing effort across the islands. In the charter fishery, catch per unit effort is based on the catch per number of fisher days. To protect national security, Customs fly-over data is presented as the number of vessel sightings per annual effort as a proportion of the greatest effort (number of flyovers) in any year provided. That is, in 2008 the plane flew over the MBI the greatest number of times relative to the six years from 2007 – 2012. The data presented in each of the other years is presented relative to that amount of effort, for example in 2010 the plane flew over the MBI 58% of the times it flew over in 2008 and is scaled to reflect this lower effort.

Statistical methods

Spatial and temporal variation in adult fish feeding guilds were examined with PERMANOVA where year (2009 to 2012) was a fixed factor and transects nested in sites were both random factors. Pairwise comparisons between years, combined with Canonical Analysis of Principal coordinates with feeding guilds as vectors, were used to interpret significant PERMANOVA results. These analyses were performed in Primer V6 and PERMANOVA+ based on a resemblance matrix created using a modified Gower distance-based measure with a Log_2 transformation to reduce the influence of the zero counts and large abundances (Anderson *et al.* 2006). Fish species were grouped into feeding guilds based on diet information from FishBase (Froese and Pauly 2013) and included, corallivores, large croppers, mobile invertivores, piscivores, scrapers/excavators, small invertivores, small omnivores and zooplanktivores. Three feeding guilds, detritivores, small croppers, omnivores, were omitted from this study due to low numbers and poor distribution.

Species were also grouped into 'fishery target' (including primary and secondary) or 'non-target' genera based on the DPIRD "State of the Fisheries report" (Fletcher and Santoro 2012). Temporal changes to the most abundant fishery target genera, and two non-target genera were examined using PERMANOVA+. The model consisted of four factors, year (2009 to 2012), management status (protected or fished), site nested within management status, and transect nested within site. The PERMANOVA model used a modified Gower distance-based measure with a Log_2 transformation to reduce the influence of species with large abundances (Anderson *et al.* 2006). Target and non-target fish are presented as fished sites (blue) and protected sites (green) in line graphs to show variation between the protective levels over time. All main PERMANOVA and the pair-wise tests on significant factors and their interactions were conducted with 9999 permutations.

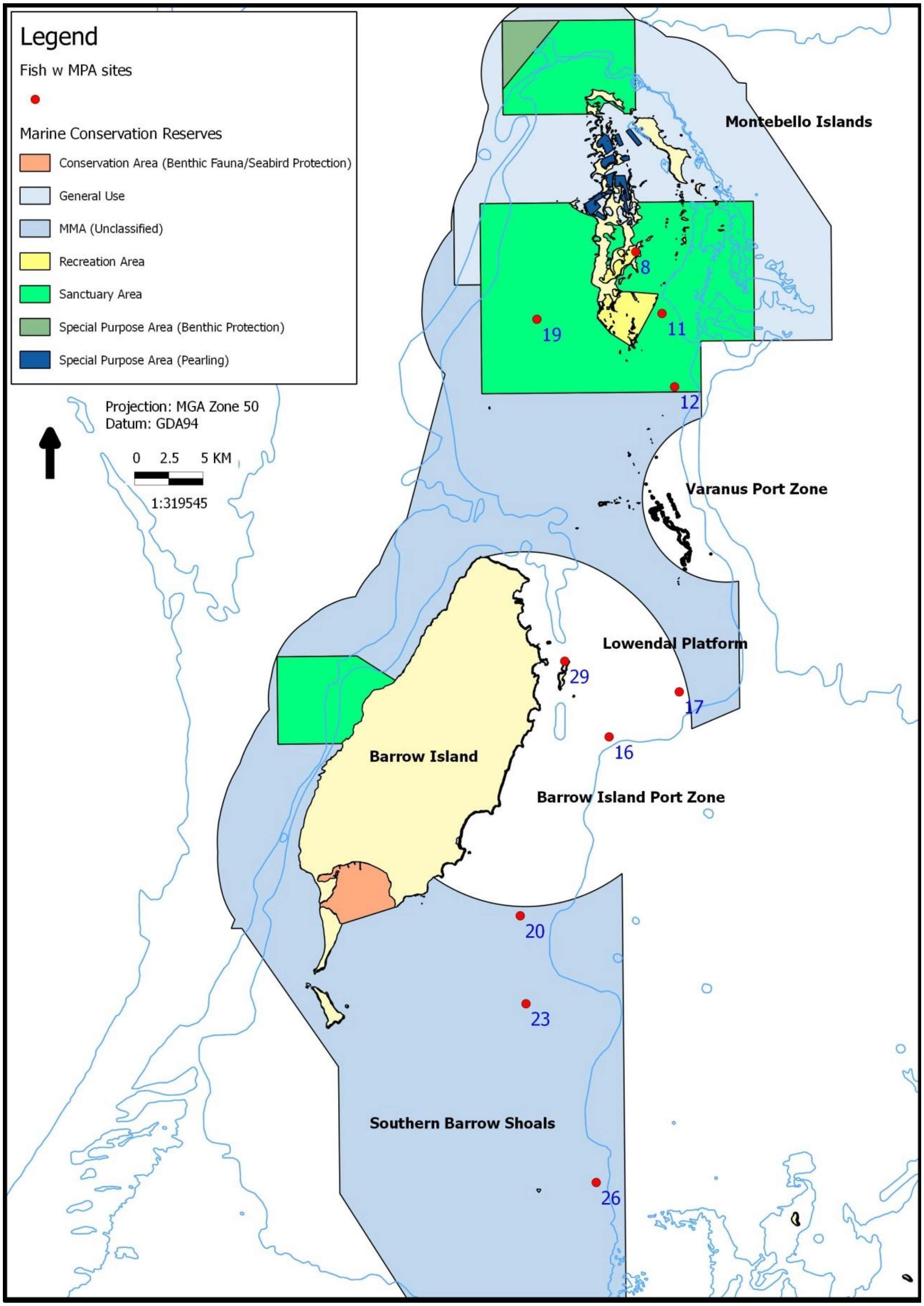

Figure 4.1. Map of sites and the zoning plan for the Montebello and Barrow Islands Marine Conservation Reserves. Note site 8 is not in a sanctuary zone.

BEST subset modelling was used to assess which environmental variables and pressures best explained variance in abundance of fish from different feeding guilds. Model selection was based on Akaike Information Criteria (AIC), which identified the most parsimonious combination of variables from a group of competing models, whilst taking model complexity into account (Bozdogan 1987). Analyses were conducted in DISTLM within PERMANOVA+ using a BEST model approach and AICc selection criteria to account for the limited number of samples through time (Anderson *et al.* 1994, Burnham and Anderson 2004). Nine variables were considered in the BEST model analyses, which included (1) percent coral bleaching of the entire coral assemblage following the heatwave of 2011, as measured in Chapter 3; (2) the number of days the dredge plume was sighted over each of the sites in 2010-2011 (Evans *et al.* 2012); average coral cover at each site of (3) *Acropora* spp., (4) *Echinopora* spp., (5) *Porites* spp., which were the three most prominent genera across all sites; (6) percent cover of rubble, which may be an important benthic habitat for feeding guilds such as mobile invertivores and small invertivores; (7) turf reef; and (8) year – the temporal element 2009–2012. Best models were those within 2AICc of the model with the lowest AICc value with the fewest descriptor variables (Burnham and Anderson 2004). Results were interpreted with distance-based redundancy analysis (dbRDA) conducted in PERMANOVA+.

4.3 Results

4.3.1 Whole community

There was significant temporal variation in the fish community of the MBI ($F_{3, 237}$: 3.745, P = 0.0001). The year 2010 was significantly different from the other three years, most likely driven by high abundances of mobile invertivores in that year. Annual fluctuation in the abundances of the small omnivores, corallivores and zooplanktivores most likely caused the gradual difference between 2009, 2011 and 2012 (Fig 4.2a). Lower abundances of all three feeding guilds were observed in 2009 and 2011 (Fig. 4.2a). There was also significant spatial variation in the fish community of the MBI ($F_{7, 237}$: 17.682, P = 0.0001) (Fig. 4.2b). Pairwise PERMANOVA found all sites to be significantly different from each other. In general, sites 16 and 17 had higher abundances of piscivores, mobile invertivores, zooplanktivores, small invertivores and omnivores, while site 19 had a high abundance of corallivores and detritivores and, to a lesser extent, small croppers (Fig. 4.2b).

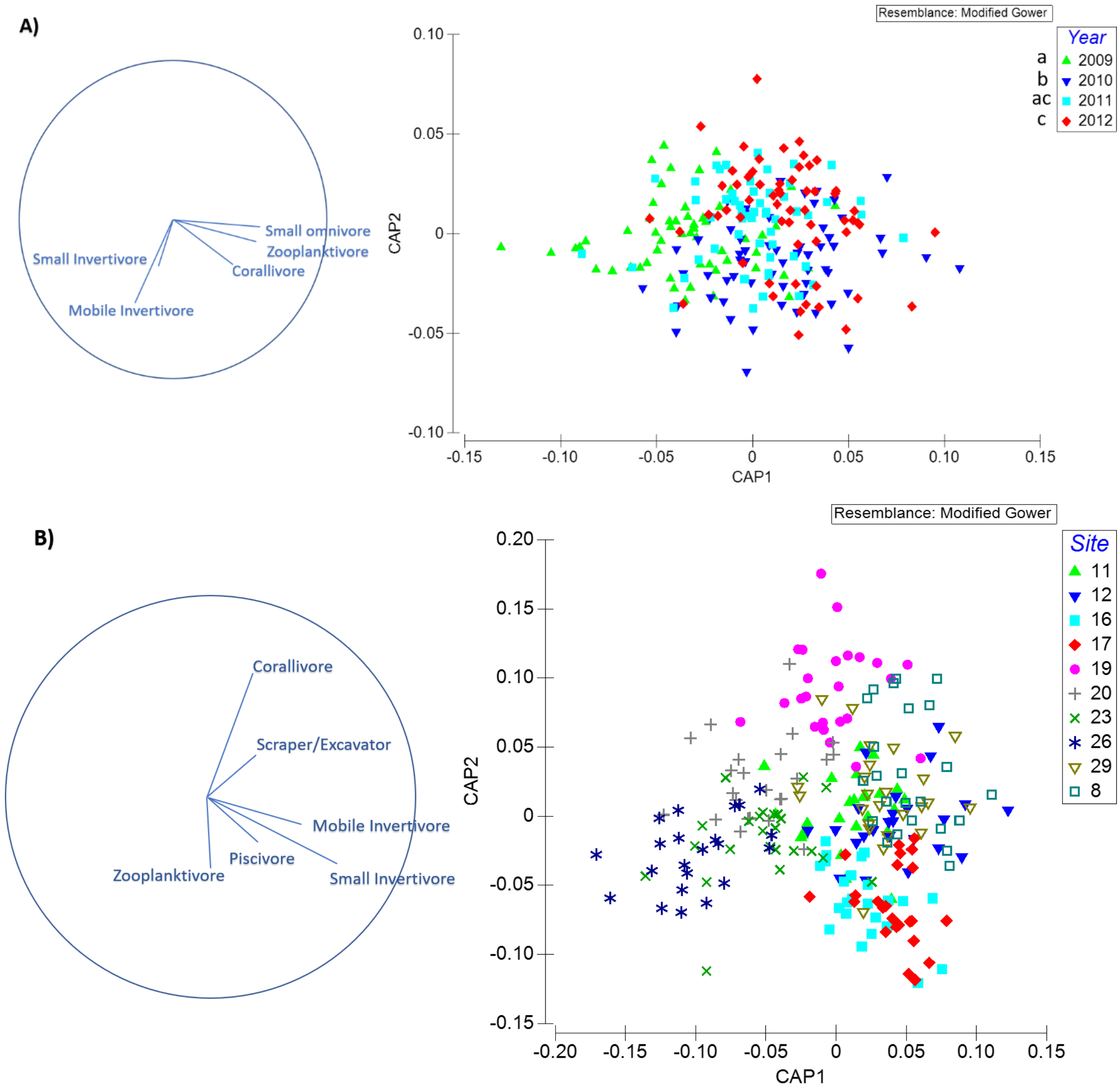

Figure 4.2. CAP analysis of the fish community highlighting differences between (a) years where points represent transects in sites in years 2009–2012, and (b) sites where points represent transects in each site, in the Montebello and Barrow islands based on feeding guilds. Vectors show the fish feeding guilds with a Pearson's correlation >0.3. Letters next to the years in 4.2(a) indicate pairwise PERMANOVA results.

4.3.2 Feeding guild distributions

Abundance of all feeding guilds varied spatially (Table 4.2), and while there was temporal variation in abundance across all feeding guilds, only four guilds varied significantly through time (Table 4.2). Six of the eight feeding guilds; corallivore, large cropper, piscivore, small invertivore, small omnivore and zooplanktivore, increased in abundance from 2009 to 2010, before there was a down-turn from 2010 to 2011. Abundance of some of these feeding guilds increased again in 2012, but this was only significant for the zooplanktivores. Abundances remained low in 2012 for the corallivores, mobile invertivores, piscivores and the small invertivores (Fig. 4.3). Abundance patterns of small omnivores and large croppers fluctuated through time similar to zooplanktivores, however they did not vary significantly. This is likely the result of over-dispersion in the small omnivore data as it did not meet the assumptions of Permdisp across years. Numbers of mobile invertivores declined significantly from 2009 to 2011 and remained at lower levels of abundance in 2012. Scraper/excavator abundance followed a similar pattern, but annual variation was high and temporal changes were not significant.

Table 4.2 Feeding guild PERMANOVAs for the Montebello and Barrow islands.

	Factor	**Site**	**Year**
Feeding Guild	**df**	**7,189**	**3,189**
Corallivore	Pseudo F	17.857	12.892
	p	**0.0001**	**0.0001**
Large Croppers	Pseudo F	6.787	0.371
	p	**0.0001**	0.774
Mobile Invertivores	Pseudo F	26.885	5.511
	p	**0.0001**	**0.005**
Piscivores	Pseudo F	10.917	1.577
	p	**0.0001**	0.212
Scraper excavators	Pseudo F	8.811	2.582
	p	**0.0001**	0.076
Small invertivore	Pseudo F	55.137	3.057
	p	**0.0001**	**0.043**
Small omnivore	Pseudo F	15.336	2.394
	p	**0.0001**	0.082
Zooplanktivore	Pseudo F	10.576	7.544
	p	**0.0001**	**0.0005**

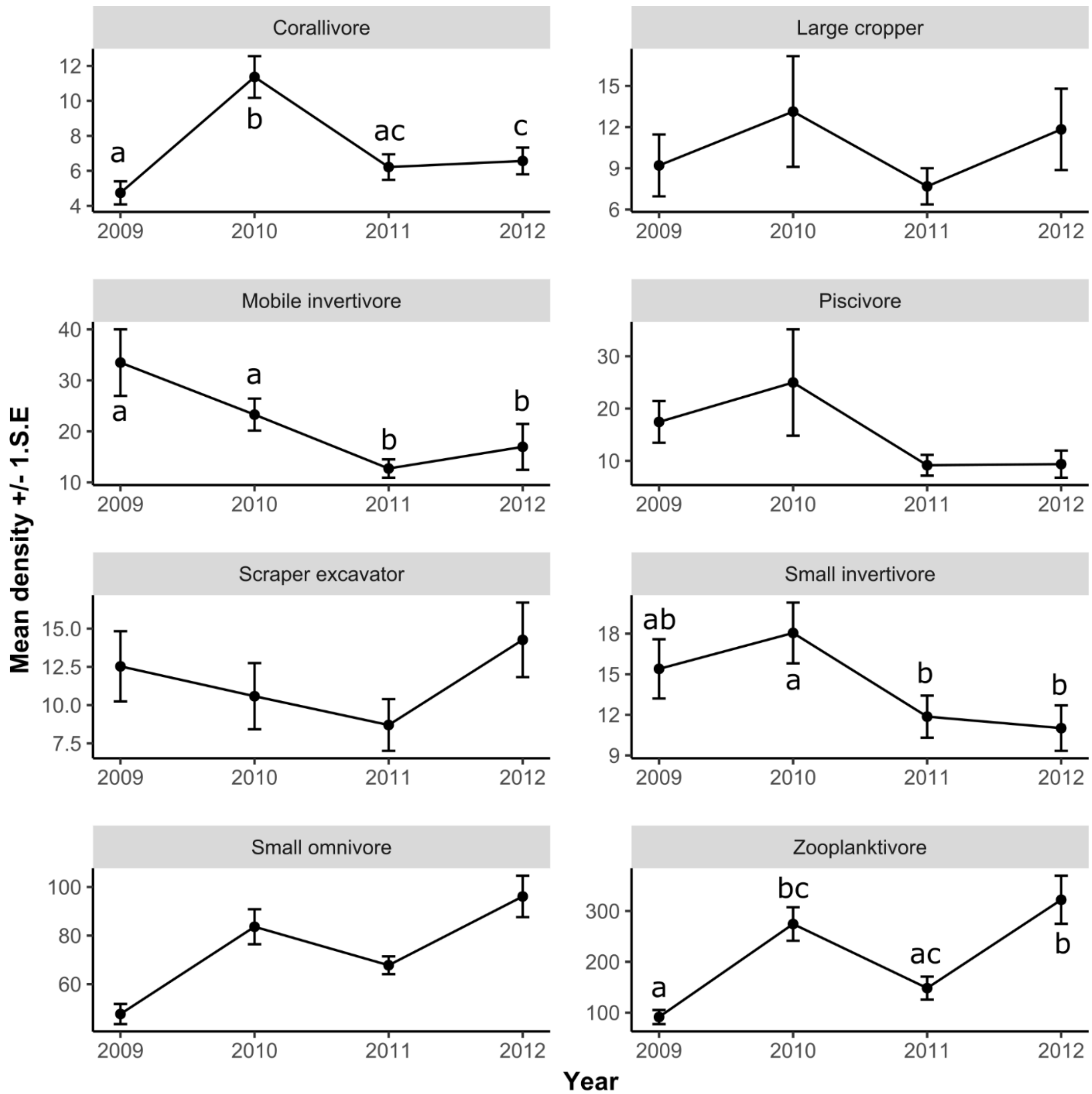

Figure 4.3 Feeding guild densities at the Montebello and Barrow islands from 2009 to 2012. Letters above error bars denote statistical similarities between years based on Pairwise PERMANOVA. Averages and standard errors calculated from 10 sites each year.

4.3.3 The effect of dredging and bleaching on fish in the MBI

Single habitat or pressure variable models did not clearly explain variation of fish communities in the MBI, however combinations of two to seven variables explained between 36% and 61% of the variation. Twenty-nine models were within two AICc values of the best model (Table 4.3), and while the explained variation is quite strong, the high number of competing models and their relatively low AICc weight (Table 4.3) suggests a number of factors influence variation in community composition. The most parsimonious model, with only two predictors, *Acropora* spp. and rubble, described 36% of the variation (Table 4.3). Sites with high rubble cover are associated with more mobile invertivores, piscivores, and small invertivores (Fig. 4.4). Corallivores were positively correlated with the *Acropora* spp. distribution, along with detritivores and small croppers, although this relationship is primarily due to high abundance of these fish and cover of *Acropora* at a single site (site 19, Fig. 4.4). Dredging was not included in the best models of abundance for any of the feeding guilds (Table 4.3). In contrast, while bleaching was not in the most parsimonious model, it was present in 55% of models within 2 AICc values of the best model and is considered a part of the model with the lowest AICc value (Table 4.3).

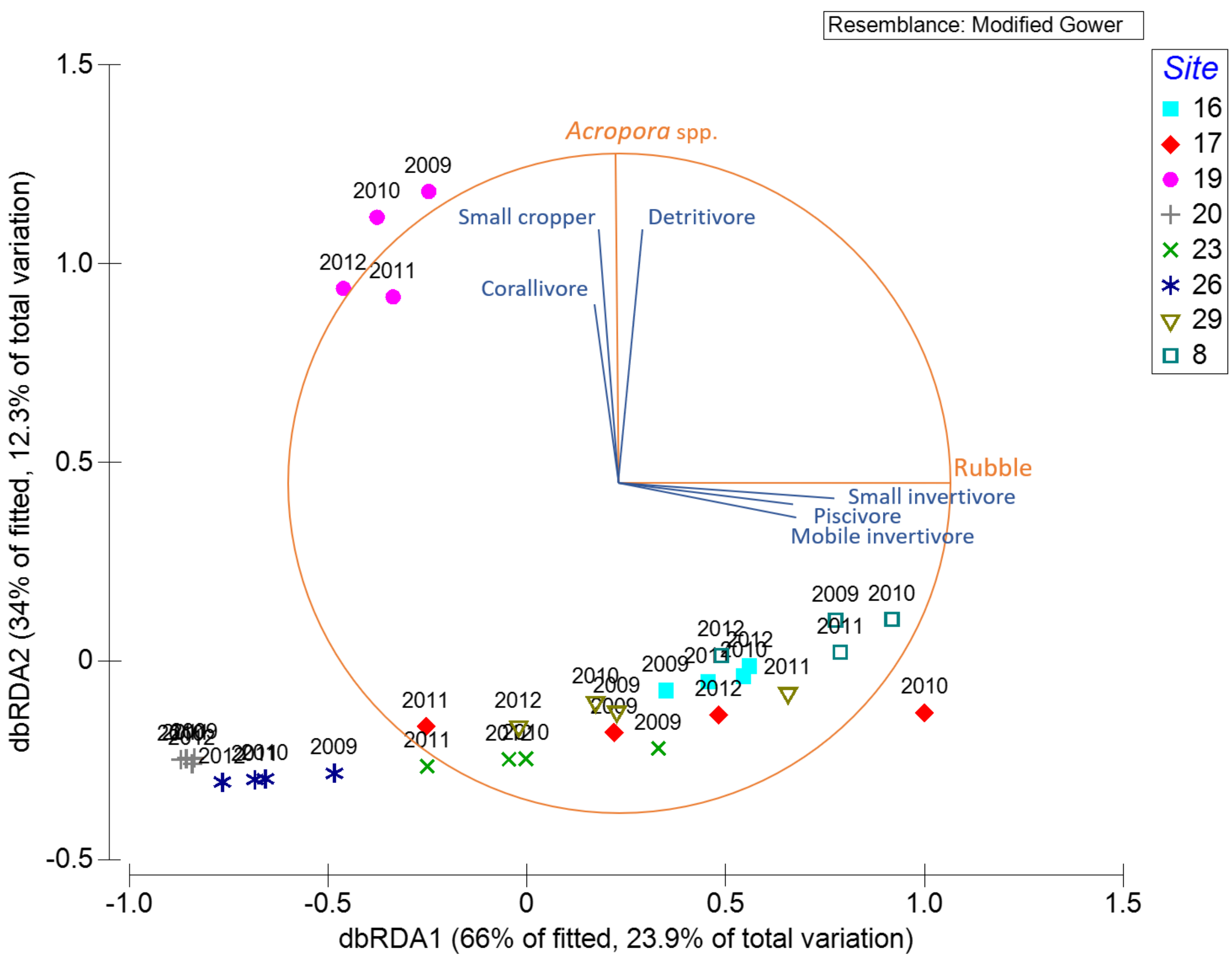

Figure 4.4 dbRDA based on the best predictors (orange vectors) from the DistlM of the fish assemblage, grouped to feeding guilds (Blue vectors, Pearson's r > 0.5) at the MBI.

Table 4.3 DistlM results showing the best predictors within 2AIC values for the distribution of fish (grouped to feeding guild) at the Montebello and Barrow islands. Most parsimonious model is highlighted in bold.

Predictor variables	No. vars	Δ AICc	AICc	AICc Weight	R^2
Bleaching, *Echinopora*, *Porites*, Rubble, Sand	5	0.6310	0.0000	0.0201	0.5372
Bleaching, *Acropora*, *Echinopora*, Rubble, Sand	5	1.0373	0.4063	0.0167	0.5313
Bleaching, *Acropora*, Rubble, Sand	4	1.1734	0.5424	0.0159	0.4822
Echinopora, Porites, Rubble, Sand	4	1.2968	0.6658	0.0152	0.4802
Bleaching, *Echinopora, Porites*, Rubble, Sand, Year	6	1.3149	0.6839	0.0153	0.5737
Acropora, Echinopora, Rubble, Sand	4	1.4125	0.7815	0.0148	0.4783
Bleaching, *Echinopora, Porites*, Rubble, Sand, Turf reef	6	1.4147	0.7837	0.0150	0.5723
Bleaching, *Acropora, Echinopora*, Rubble, Sand, Year	6	1.5406	0.9096	0.0143	0.5706
Acropora, Rubble, Sand	3	1.6182	0.9872	0.0139	0.4265
Acropora, Echinopora, Rubble, Sand, Turf reef	5	1.6730	1.0420	0.0137	0.5219
Bleaching, *Acropora, Echinopora,* Rubble, Sand, Turf Reef	6	1.6870	1.0560	0.0138	0.5687
Echinopora, Porites, Rubble, Sand, Turf Reef	5	1.7101	1.0791	0.0139	0.5214
Bleaching, *Acropora*, Rubble	3	1.8184	1.1874	0.0133	0.4229
Bleaching, *Acropora, Echinopora, Porites,* Rubble, Sand, Turf Reef	7	1.8280	1.1970	0.0134	0.6128
Bleaching, *Acropora, Echinopora, Porites,* Rubble, Sand	6	1.9598	1.3288	0.0127	0.5650
Acropora, Echinopora, Rubble, Sand, Year	5	2.0556	1.4246	0.0123	0.5162
Acropora, Echinopora, Porites, Rubble, Sand, Turf reef	6	2.0755	1.4445	0.0123	0.5634
Bleaching, *Acropora,* Rubble, Sand, Year	5	2.0765	1.4455	0.0125	0.5159
Acropora, Echinopora, Porites, Sand, Turf reef	5	2.1044	1.4734	0.0125	0.5154
Echinopora, Porites, Rubble, Sand, Year	5	2.1542	1.5232	0.0123	0.5147
Bleaching, *Echinopora, Porites,* Rubble,	4	2.1636	1.5326	0.0124	0.4659
Bleaching, *Acropora, Echinopora,* Rubble	4	2.3559	1.7249	0.0114	0.4627
***Acropora*, Rubble**	**2**	**2.3715**	**1.7405**	**0.0114**	**0.3626**
Bleaching, *Acropora,* Rubble, Sand, Turf Reef	5	2.3857	1.7547	0.0115	0.5111
Bleaching, *Acropora, Echinopora, Porites*, Rubble, Sand, Year	7	2.4516	1.8206	0.0113	0.6052
Bleaching, *Echinopora,* Rubble, Sand	4	2.4578	1.8268	0.0113	0.4610
Acropora, Rubble, Sand, Year	4	2.5996	1.9686	0.0107	0.4586
Acropora, Echinopora, Rubble, Year	4	2.6146	1.9836	0.0107	0.4584
Echinopora, Porites, Rubble	3	2.6231	1.9921	0.0108	0.4082

4.3.4 Target genera

Variation in the abundance of the combined target fish species was high in 2009 and 2010, and although the following years had approximately 60% lower abundances and much reduced variation, temporal variation was insignificant (Table 4.4). Similarly, there was no significant difference in the abundance of targeted fish between the fished and protected areas, and there was no interaction between status and year, primarily due to high variation in the 2010 data (Table 4.4, Fig. 4.5). Pairwise tests for the year x status did however suggest significant differences in targeted fish abundance between 2010 and the two later years at fished sites (Fig. 4.5).

The study sites in fished areas had consistently more *Choerodon* than the protected sites through time, but there was a significant decline in abundance at fished sites from 2010 to 2011 (Fig. 4.4). Abundance of all fisheries target and non-target genera varied spatially (Table 4.4), although only *Choerodon* and *Epinephelus* varied temporally and their abundance declined after 2010 (Fig. 4.5). Abundance of *Lethrinus* also declined from 2009 to low levels in the last three years of the study, though high abundance in 2009 was due to large numbers observed at two sites and there is no significant temporal trend.

Table 4.4 PERMANOVA results assessing temporal (year) and management zones (status: protected or fished) influence on the abundance of fisheries target (t) and non-target (nt) genera at the Montebello and Barrow islands.

Genera		**Year x Status**	**Site (Status)**	**Status**	**Year**
df		3,237	8,237	1,237	3,237
All Targets	Pseudo F	0.732	6.98	2.143	1.782
	p	0.697	**0.0001**	0.091	0.092
***Choerodon* spp. (t)**	Pseudo F	0.830	5.417	20.164	4.373
	p	0.486	**0.0001**	**0.003**	**0.014**
***Epinephelus* spp.(t)**	Pseudo F	0.940	19.982	1.360	2.988
	p	0.426	**0.0001**	0.272	**0.046**
***Lethrinus* spp.(t)**	Pseudo F	0.292	3.950	1.928	1.784
	p	0.829	**0.0001**	0.226	0.172
***Lutjanus* spp. (t)**	Pseudo F	0.816	5.025	0.283	0.285
	p	0.500	**0.0001**	0.639	0.826
***Plectropomus* spp. (t)**	Pseudo F	2.333	8.924	0.274	2.416
	p	0.102	**0.0001**	0.611	0.092
***Scarus* spp. (nt)**	Pseudo F	0.738	4.719	0.161	0.128
	p	0.540	**0.0001**	0.691	0.943
***Scolopsis* spp. (nt)**	Pseudo F	0.545	56.496	0.147	0.963
	p	0.656	**0.0001**	0.710	0.432

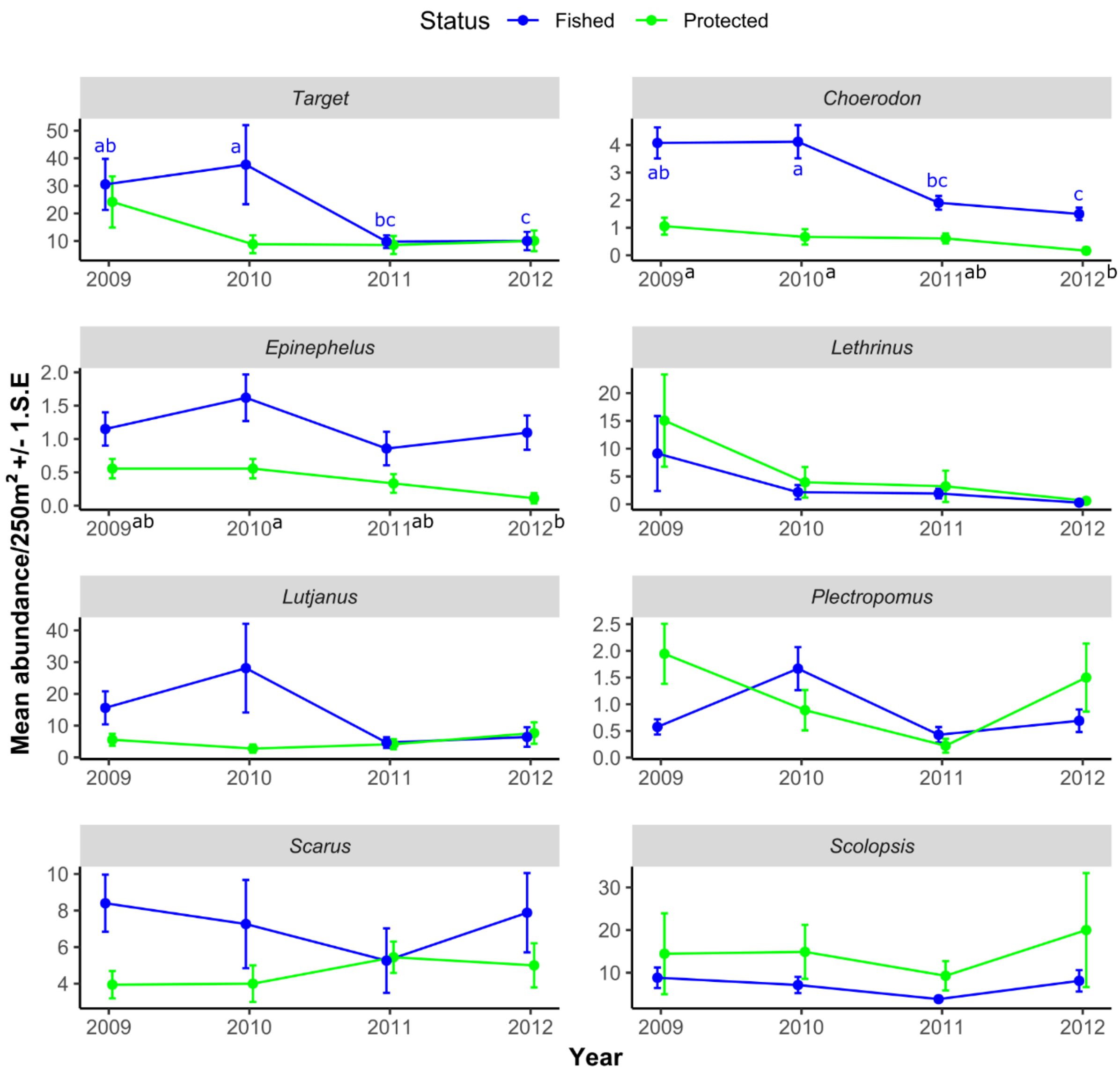

Figure 4.5 Mean abundance of fishery target and non-target genera in fished (blue) and protected (green) sites at the Montebello and Barrow islands from 2009 to 2012. Letters above fished sites (blue line) show statistical similarities and differences using Pairwise tests in PERMANOVA. In genera where a significant difference between years was found, superscript letters above years show statistical similarities using pairwise tests in PERMANOVA.

4.3.5 Measure of fishing effort in the MBI

The number of fisher days and operators within the charter boat industry has fluctuated from year to year at both the Montebello and Barrow islands (Fig. 4.6a, c). Similarly, the catch from charter boats has fluctuated, however the catch per unit effort has generally declined at Barrow Island since 2006 and at the Montebello Islands from 2007 (Fig.4.6b, d).

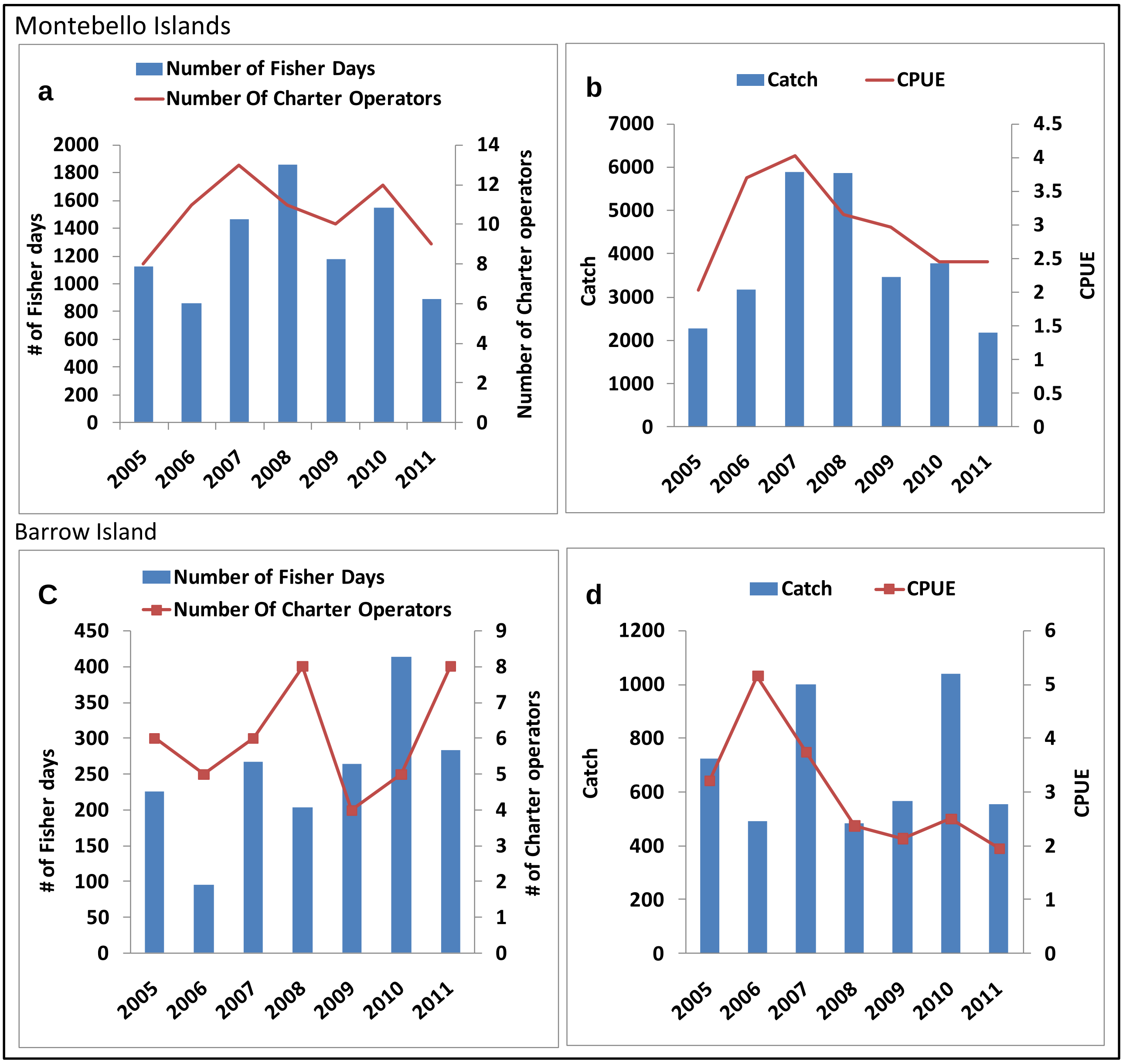

Figure 4.6 Catch and effort data of the charter fishery in the Montebello (a, b) and Barrow islands (c, d) region. Data provided by DPIRD, Western Australia.

Customs flight data suggested an increased visitation and the potential for increased fishing pressure within the MBI. The presence of recreational vessels increased from 2007 to 2011, particularly at the Montebello Islands (Fig. 4.7). However, in 2012 the boat visitation was zero, suggesting a rapid decline in recreational visitation to the area. Effort during 2012 is also relatively low (14% of the maximum flights), although this level of effort is comparable to 2011 when the most boats were spotted relative to 2008.

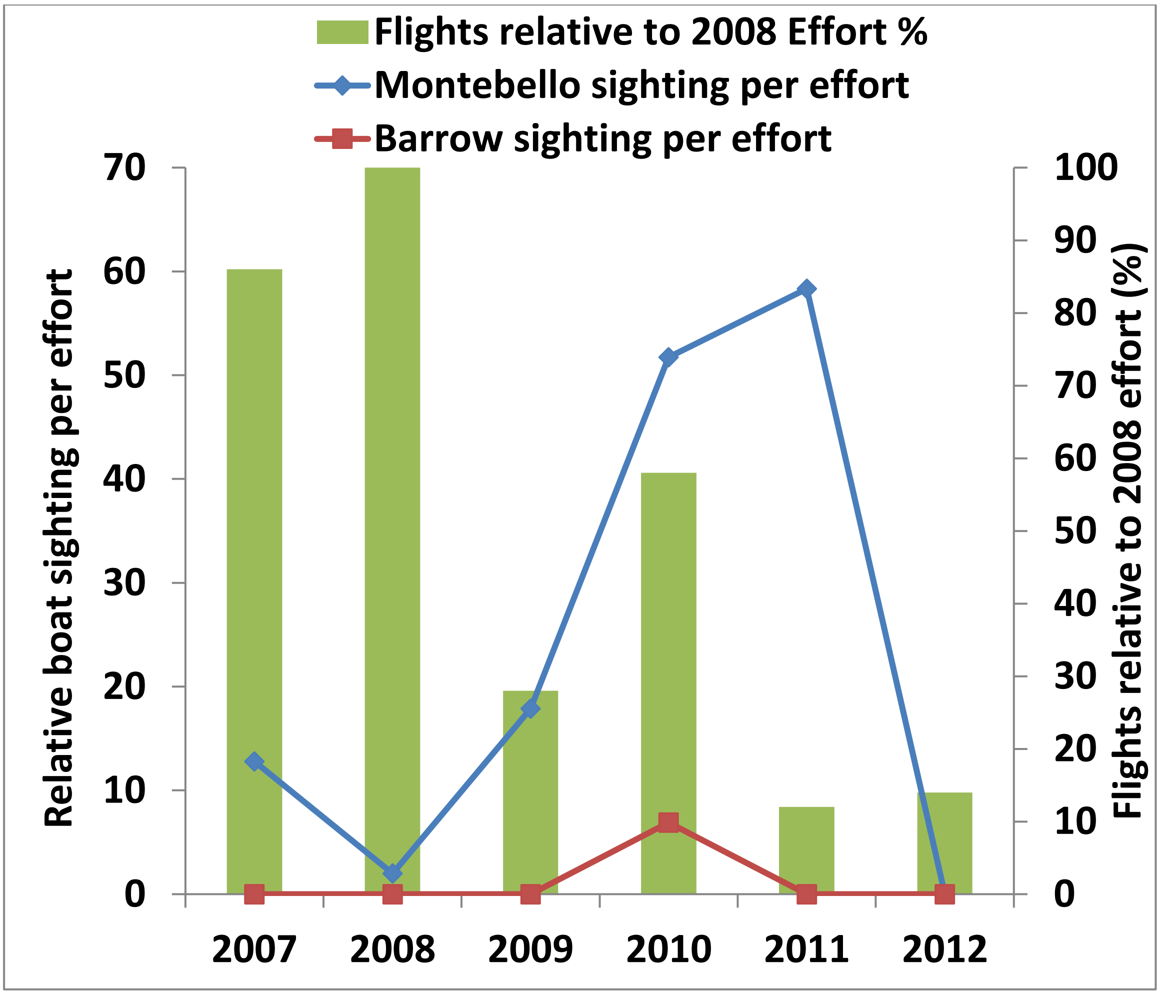

Figure 4.7 The relative number of recreational boats sighted at the Montebello and Barrow islands marine conservation reserves proportional to 2008 which had the greatest amount of effort (annual flyovers). Percentages presented in figure represent the number of flights relative to 2008.

4.4 Discussion

4.4.1 Temporal fish communities 2009 – 2012

Over the four years of this study, abundance of most non-target feeding guilds remained relatively stable; however, there was a downward trend for several feeding guilds and target genera from 2010 to 2011, which coincides with the commencement of dredging in late 2010 (Chevron Australia 2009) and bleaching in early 2011 (Pearce and Feng 2013). Most of the non-target groups increased in abundance again in April/May 2012 after dredging had finished, while most of the fisheries target genera remained low for the remainder of the survey period. Some fish may be directly affected by increased sediments caused by dredging (Wenger *et al.* 2015), or increased temperatures (Munday *et al.* 2008), whilst changes to habitat due to dredging or heat stress may affect fish indirectly (Wilson *et al.* 2006). During the same summer three cyclones passed close to the MBI causing resuspension of natural and/or dredge operation sediments at six of the survey sites (Evans *et al.* 2012) and potentially causing coral mortality. Combined, these events may have caused some biological and behavioural responses from the fish community, although no conclusive evidence exists to suggest that either dredging, cyclones or coral bleaching had strong effects on fish. Similarly, for the target genera, declines in abundance may be a direct effect of increased fishing, but the data we present does not conclusively support this theory. The lack of zoning effect may be a result of one or a combination of factors including: limited fishing pressure early in the study period, lack of suitable habitat in the no-take areas, ineffective compliance in the no-take areas, or there has not been enough time for protected areas to accrue more individuals since they were created in 2005 (Evans and Russ 2004).

The monitoring sites in this study were outside the main zone of dredge influence and can only examine the impact of dredging where dredge plume exposure was sporadic (Chapter 2). Under these conditions we did not find definitive evidence that the MBI reef fish community was affected by dredging. Clearly there is a need to improve understanding of how dredging and associated sediment exposure influences ecological and behavioural response of fish (Wenger *et al.* 2017).

Coral bleaching was also found to have a negligible effect on the abundance of fish. Loss of coral, due to processes like bleaching, has its greatest impact on obligate corallivores (Sano 2004, Wilson *et al.* 2006, Pratchett *et al.* 2008, Emslie *et al.* 2011, Graham *et al.* 2011). The lack of any major decline in corallivores at MBI following coral bleaching therefore suggests any negative relationships with other fish may be coincidental and unrelated to bleaching. Furthermore, loss of coral attributable to this bleaching event was small in the MBI (Moore *et al.* 2012) and consequently any effects on fish habitat were also transient.

4.4.2 The importance of location

The MBI sites covered a range of different coral habitats ranging from near-shore low visibility shoaling reefs south of Barrow Island to clear water lagoonal reef flats inside barrier reefs near the continental shelf margin (DEC 2007). Within this range of coral reef habitats there are back reef bommie fields; sheltered fringing reefs and reefs within channels with high tidal movement. The obvious differences in the fish communities are most likely driven by spatial variation in these benthic and oceanographic variables (Friedlander *et al.* 2003, Travers *et al.* 2006). This finding has implications for management of the multi-use zoning plan. If no-take areas are designed to protect a range of representative habitats, then this has been largely successful in the Montebello Islands where no-take areas cover a range of habitats from east to west. However, the reefs on the Lowendal platform and shoals to the south of Barrow Island are unique (see Chapter 3) and have been zoned for general use or are within port waters. Importantly, the reefs on the Lowendal platform contain the greatest densities of fishery target species, yet fish on these reefs are afforded no zone protection from fishing. Furthermore, the distances from the Lowendal platform to the other parts of the MBI are within known fish dispersal kernels (Harrison *et al.* 2012) and protecting some of these reefs could provide a fishery refuge and source of recruits to the MBI as fishing pressure increases.

4.4.3 Fishery targeted genera

Densities of fishery target species have not previously been reported from the coral reefs of the MBI. The sites in this study were selected for monitoring the impacts of dredging on coral communities, and high variability in abundance estimates indicate there is insufficient replication to investigate any effects of the MBI no-take zones on target species. Nonetheless, results suggest that abundance of target species has declined over the monitoring period, especially among *Choerodon* spp. where densities have declined significantly over time. Statistical differences in temporal declines were difficult to detect, as initial surveys were associated with high abundances and high variability as many of these species tend to aggregate into large schools (De Mitcheson *et al.* 2008). It is, however, noteworthy that these large schools were not observed in later years of the study. Moreover, non-target genera and non-target feeding guilds did not decline in abundance, supporting the suggestion that fishing has affected some taxa in the shallow reefs of the MBI. Increased recreational activity and reduced CPUE from the charter fishery support the notion that decline in target fish abundance is due to fishing. However, these metrics of fishing pressure are coarse and without robust measures of recreational and commercial fishing effort, the effects of fishing remain equivocal.

Linking declines in target species with increased fishing pressure are also confounded by subtle differences in the timing of surveys. Lethrinids are the most targeted fishes in the region (Fletcher and Santoro 2012) and the large number of *Lethrinus* spp. observed at two sites in November 2009 may be fish spawning aggregations. Lethrinid spawning season in the north-west of Australia is during the spring-summer period (Sampey *et al.* 2004, Marriott *et al.* 2010) and abundances during early survey years may have been inflated by spawning aggregations. Similarly, *Epinephelus bilobatus* were seen aggregating (n=33) at a single site in November 2009. These serranids were not seen in such high numbers at other sites in 2009 or in subsequent surveys from 2010 to 2012, suggesting fish were forming a spawning aggregation in November 2009. Continued monitoring of these sites is required to see if these are consistent fish aggregation sites or random events (Claydon 2004).

Although our initial findings are equivocal, our surveys provide a strong basis for monitoring the impacts of fishing on MBI fish communities. Four of the sites (11,12,16,17) have habitat that supports densities of target species similar to other regions in WA. For example, *Plectropomus* abundances at these four sites are similar to those in no-take areas at the Houtman Abrolhos Islands (Nardi *et al.* 2004), and initial estimates of *Lethrinus* abundance (2009-2011) at the same sites were an order of magnitude greater than Ningaloo Reef (Westera *et al.* 2003). Sampling designed to monitor target species at sites with suitable habitat both inside and outside no-take areas is recommended for the MBI.

4.4.4 Summary

There was a strong site divergence among adult fish populations that was most likely associated with geographical differences in habitats, oceanographic and environmental processes. Although there was temporal variation in the abundance of some feeding guilds, evidence that these trends were related to dredging or coral bleaching at our sites is equivocal. These results may however have changed following more extensive bleaching of corals in the region in 2013 and 2014 (Lafratta *et al.* 2016; Ridgway *et al.* 2016). Densities of some fisheries target species have also declined, and relatively stable densities of non-target genera suggests there may be some impact of fishing on targeted genera in shallow (<15m) habitats of the MBI. Importantly, sites and habitats where there are, or have been, high densities of target species are not within current sanctuary zones.

To understand the effects of fishing pressure on the abundance and distribution of target species, further monitoring inside and outside of enforced no-take sanctuary zones is required. Considering the strong locational differences in fish assemblages relate to spatial variation in the benthos, such a sampling program should include matching habitats within and between zones. Further investigations should also consider compliance levels, and a more detailed, location specific, assessment of fishing pressure, and changes to habitat which are the two primary drivers of variance in fish communities.

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Appendices

Appendix 1 Marine macro-invertebrate communities of the Montebello and Barrow islands area

Introduction

Invertebrate fauna of the Montebello and Barrow islands (MBI) is quite diverse, with the most species rich mollusc assemblage (633 species), and among the most diverse echinoderm assemblages (170 species), on the WA coast (Berry & Wells 2000). This is attributable to the wide range of habitats surrounding the mosaic of islands and reefs that make up the MBI. Western and northern shores of Barrow Island are typical of the Pilbara offshore marine bioregion, while the south and eastern shores are more similar to the Pilbara nearshore bioregion. Commercial collection of invertebrates is limited to holothurians and there is some recreational take of *Panulirus* lobsters. Coral predators have been recorded in relatively low densities on some reefs within the MBI. They include *Acanthaster planci* (Crown-of-thorn starfish), *Drupella cornus* and *D. rugosa* (very low densities). This appendix is a summary of collected data.

Methods

Macro-invertebrates (MI) were surveyed using underwater visual census (UVC) at 23 coral reef sites and seven macro-algae sites throughout the MBI (Appendix Table 1). All surveys were carried out between the hours of 0800 and 1700 during November 2011. Taxa were included in this study because they are either relevant to recreation and/or commercial fisheries or because they have significant ecological roles. The UVC method involved laying 3 x 50m transects and counting all MI observed within 1m of the left side of the transect tape (total area 50m^2), except for *Acanthaster* and *Panulirus* (counted 1m either side of the tape (total area 100m^2) and *Drupella* spp (counted on a 0.5m x 20m transect, total area 10m^2).

Appendix Table 1. GPS points for the macroinvertebrate surveys conducted at coral and macroalgal habitats in the Montebello and Barrow islands.

Habitat	Site	Lat	Long	Habitat	Site	Lat	Long
Coral	1	-20.4007	115.5001	Coral	20	-20.9019	115.4617
Coral	1b	-20.4013	115.5	Coral	21	-20.8999	115.4508
Coral	2	-20.3875	115.5642	Coral	22	-20.895	115.4404
Coral	3	-20.3483	115.5138	Coral	23	-20.9598	115.4652
Coral	4	-20.3686	115.5257	Coral	24	-20.951	115.4629
Coral	5	-20.3945	115.542	Coral	25	-20.9946	115.4799
Coral	6	-20.4053	115.5819	Coral	26	-21.0778	115.5132
Coral	7	-20.4527	115.4784	Coral	27	-20.7837	115.5001
Coral	8	-20.4659	115.5468	Coral	28	-20.9078	115.4625
Coral	9	-20.4513	115.6051	Coral	29	-20.7348	115.4946
Coral	10	-20.5085	115.5433	Coral	30	-20.8007	115.504
Coral	11	-20.5066	115.5647	Coral	31	-21.0465	115.4704
Coral	12	-20.555	115.5731	Coral	32	-20.8603	115.4916
Coral	13	-20.556	115.5725	macroalgae	M2	-20.5188	115.5259
Coral	14	-20.629	115.5832	macroalgae	M1	-20.5129	115.5121
Coral	15	-20.7186	115.5766	macroalgae	M11	-20.702	115.5612
Coral	16	-20.7848	115.5251	macroalgae	M12	-20.7458	115.5745
Coral	17	-20.7556	115.5745	macroalgae	MA1	-20.92	115.436
Coral	18	-20.7861	115.5067	macroalgae	MA2	-20.9613	115.4516
Coral	19	-20.5097	115.4773	macroalgae	MA3	-21.0544	115.4609

Appendix Table 2. List of the species within each Genus recorded at coral and macroalgal sites in the Montebello and Barrow islands.

Group	**Common Name**	**Species (Average; SD)**
Coral predator	Crown of thorn starfish (COTS)	*Acanthaster plancii* (0.15/100m^2; 0.56)
	Horn drupe	*Drupella cornus* (3.41/10 m^2; 10.38)
	Rugose drupe	*Drupella rugosa* (0.72/10 m^2; 2.38)
Holothurians	Surf redfish	*Actinopyga echinites* (0.04/50m^2; 0.21)
	Leopard sea cucumber	*Bohadschia argus* (0.01/50 m^2; 0.12)
	Graeffe's sea cucumber	*Bohadschia graeffei* (0.03/50 m^2; 0.24)
	Black lollyfish	*Holothuria atra* (0.29/50 m^2; 0.94)
	Pinkfish	*Holothuria edulis* (0.33/50 m^2; 0.70)
	Tiger tail sea cucumber	*Holothuria hilla* (0.01/50 m^2; 0.12)
	Greenfish	*Stichopus chloronotus* (0.39/50 m^2; 1.00)
	Curryfish	*Stichopus hermanni* (0.04/50 m^2; 0.28)
	Curryfish	*Stichopus variegatus* (0.03/50 m^2; 0.17)
	Sea cucumber	*Stichopus* sp (0.65/50 m^2; 2.61)
Panulirus	Ornate spiny lobster	*Panulirus ornatus* (0.03/100m^2; 0.24)
	Painted rock lobster	*Panulirus versicolor* (0.33/100m^2; 0.78)
Clams & other bivalves	Crocus giant clam	*Tridacna crocea* (0.04/50 m^2; 0.27)
	Smooth giant clam	*Tridacna derasa* (0.01/50 m^2; 0.12)
	Giant clam	*Tridacna gigas* (0.01/50 m^2; 0.12)
	Crocus giant clam	*Tridacna maxima* (0.38/50 m^2; 0.82)
	Fluted giant clam	*Tridacna squamosa* (0.17/50 m^2; 0.42)
	Black lip oyster	*Pinctada margaritifer* (0.48/50 m^2; 1.20)
Seastars and urchins	Cushion seastar	*Choriaster granulatus* (0.01/50 m^2; 0.12)
	Cushion seastar	Culcita novaeguineae (0.01/50 m^2; 0.12)
	Orange seastar	*Echinaster luzonicus* (0.06/50 m^2; 0.38)
	Starfish	*Nardoa* spp (0.03/50 m^2; 0.17)
	Sea Urchins	Diadema spp (3.87/50 m^2; 6.24)
	Blue black urchins	*Echinothrix diadema* (0.04/50 m^2; 0.27)
	Banded urchins	*Echinothrix calamaris* (0.01/50 m^2; 0.12)
	Seastar	*Fromia* spp (0.13/50 m^2; 0.38)
	Slate pencil urchin	*Heterocentrotus mammillatus* (0.17/50 m^2; 0.75)
	Red slate pencil urchin	*Heterocentrotus trigonarius* (0.49/50 m^2; 1.36)
Other gastropods	Star snails	*Astralium* sp1 (1.43/50 m^2; 5.11)
	Pen shell	*Atrina* spp/*Pinna* spp (0.99/50 m^2; 2.02)
	Pyramid top shell	*Tectus pyramis* (1.30/50 m^2; 2.73)
	Commercial top snail	Trochus niloticus (0.04/50 m^2; 0.27)
	Sivermouth turban snail	*Turbo argyrostomus* (0.09/50 m^2; 0.45)

Results

The densities and abundances of observed MI at the genus level (Appendix Fig 1 & 2) and species level (Appendix Table 2) in the MBI vary widely. The following is a comparison of the densities of the most abundant commercially and ecologically important genus/species with other studies within WA and the broader Indian ocean and globally.

Holothurians

Holothuria have clumped distributions and show great variation in their abundance throughout the MBI (Appendix Fig 1). Holuthurians of four genera were observed in the MBI; *Actinopyga, Bohadschia, Holothuria* and *Stichopus*. The two most abundant genera of Holothurian observed were the *Holothuria* spp. (mean 0.55+/-1.05 per 50m^2) and *Stichopus* spp. (mean 0.68 +/- 1.76 per 50m^2). *Holothuria edulis* and *H. atra* were among the most abundant species, with densities similar to those observed using the manta tow method at Ningaloo (100/ha which id equivalent to 0.5/50m^2) (Shiell & Knott 2010). These results are however an order of magnitude lower than those recorded at inshore reefs of the GBR in 1997: 10/100 m^2 (Uthicke 1997); and similar to the lower density reefs on the inshore GBR in 2001, 90.01 – 0.69/ m^2 (Uthicke 2001). *Stichopus* sp. and *Stichopus chloronotus* were also abundant on the MBI reefs. *Stichopus* sp. appeared most similar to the undescribed *Stichopus* sp. shown in field guides (Allen & Steene 2002, Colin & Arneson 1995, Gosliner *et al.* 1996).

Panulirus

Panulirus are recreationally targeted in the region and the densities recorded in this study may be indicative of the remoteness of the MBI. Only two species of *Panulirus* were observed in the MBI; *Panulirus ornatus* and the more abundant *P. versicolor*. *Panulirus* in the MBI had densities similar to other tropical populations in northern Australia recorded in the early 1990s (Pitcher *et al.* 1992), when Torres Strait populations of *Panulirus* were estimated at 2.5 to 90/ha (equiv. 0.25 - 0.9/100 m^2). *Panulirus* densities in the MBI were slightly higher than a recent study at Ningaloo Reef where only one *Panulirus* was detected every 1km of searching, which equates to 0.1/100m^2 (Depczynski *et al.* 2009). However, densities of *Panulirus* recorded on a reef in the southern GBR had densities over an order of magnitude lower than both WA populations (Frisch 2008).

Clams

This is a natural population as clams are not currently harvested at the MBI. Overall clam densities were quite low (0.48 +/- 0.78/50 m^2). *Tridacna maxima* was the most abundant of the giant clams in the MBI although densities were low compared to intertidal habitats at Ningaloo Reef (2 – 413.5/50 m^2) (Black *et al.* 2011). Densities were similar those of *T. maxima* (0.61/50 m^2) to a study in the Lakshadweep Archipelago in the Northern Indian Ocean (Apte *et al.* 2010).

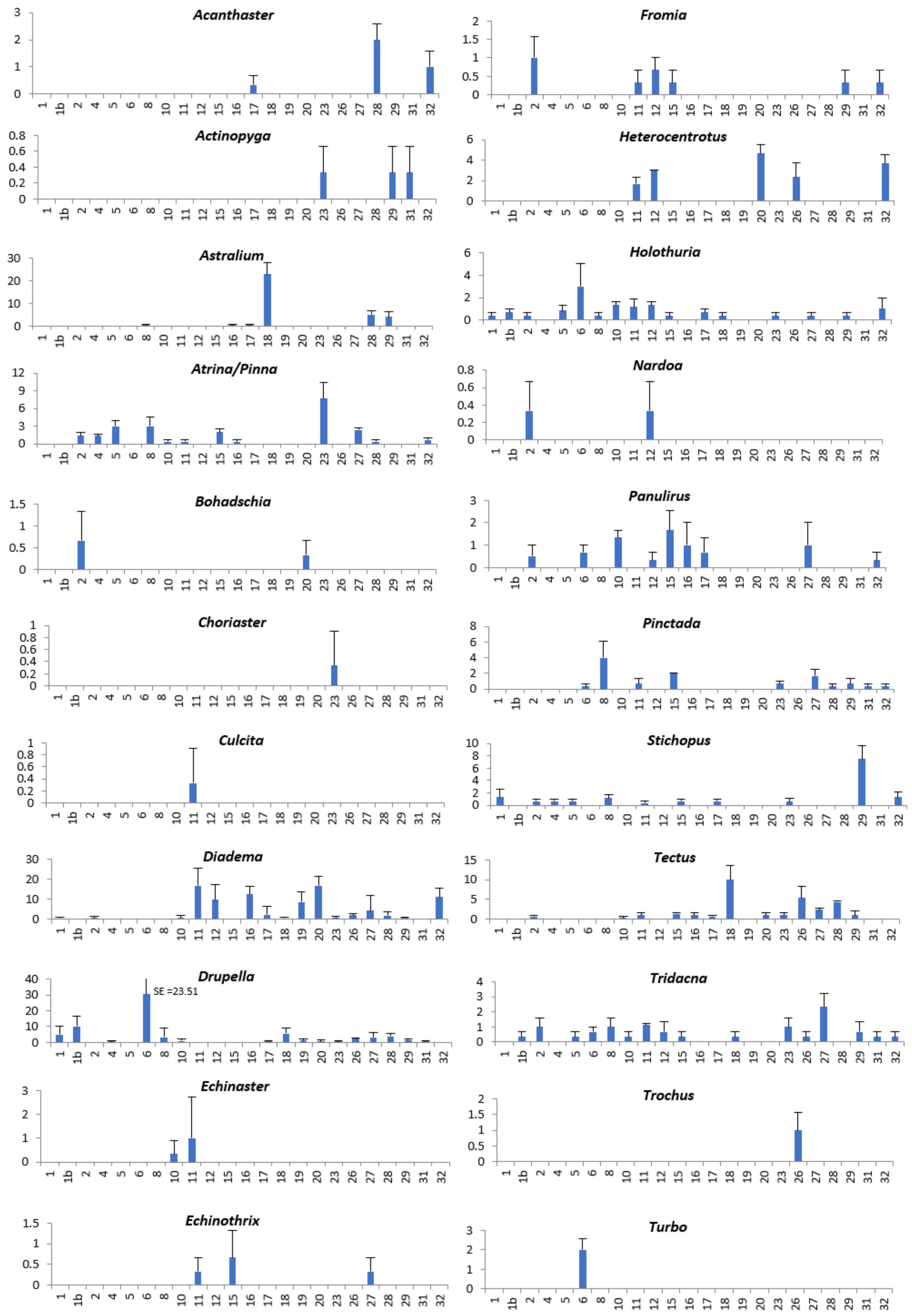
Coral Predators

Crown-of-thorn Starfish (COTS)

COTS densities across sites within the MBI were patchy and ranged from 0 – 2/100 m^2. Higher densities of COTS were observed at site 28 and 32, whilst the rest of the reefs had low and patchy distributions (Appendix Fig 1). All COTS observed were large adults, and no recruits were observed. Densities at most MBI sites were low compared to those found at Lizard Island on the GBR (0.8 – 1.2/ 200 m^2 equiv. 0.4 – 0.6/100 m^2; Pratchett 2005) and historic surveys in the Dampier Archipelago (52/ha, equiv. 0.52/100 m^2; Simpson and Grey 1989).

Drupella spp.

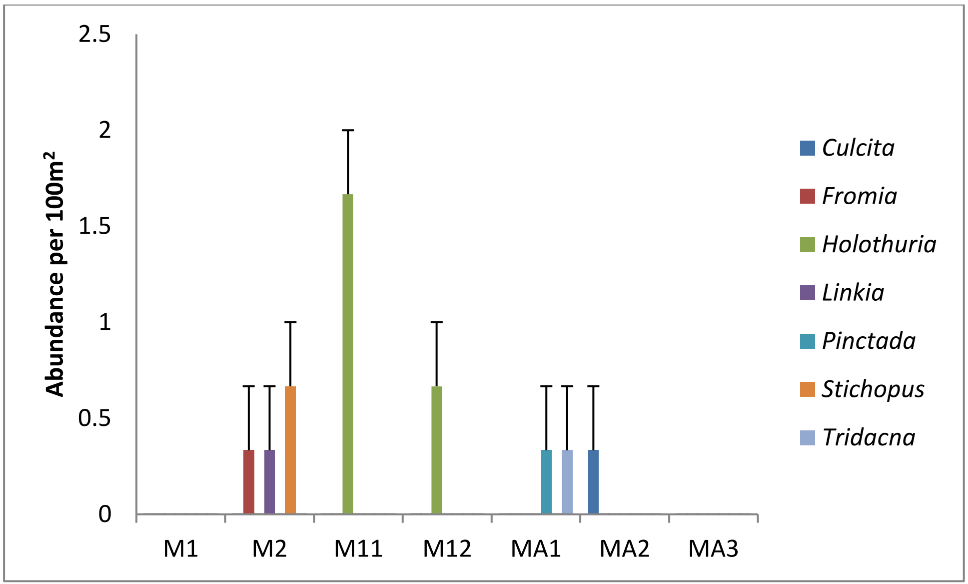
Drupella cornus was the most common species of corallivorous gastropod, although a small number of *D. rugosa* were recorded. Mean density of *Drupella* spp was 3.02 +/- 9.65/10 m^2 which was similar to lower levels of *Drupella* recorded at Ningaloo Reef in 2008 (3.6 – 27.2/10 m^2) (Armstrong unpubl. report) and numerous other studies dating back to 1989 (Armstrong 2007, Ayling & Ayling 1987, Holborn *et al.* 1994, Turner 1994).



Appendix Figure 1. Distribution and mean abundance (+1S.E.) of observed macro-invertebrates at coral reef sites in the Montebello and Barrow islands. X-axis shows the site numbers. All y-axis are units per 50m² except Drupella *(10m²),* Acanthaster *and* Panulirus *(100m²).*

Macroalgal sites

Very few MI were observed in the macroalgal sites (Appendix Fig. 2). The most abundant sea cucumbers, sea stars, and a few bivalves such as *Pinctada* and *Tridacna* (Appendix Fig. 2).



Appendix Figure 2. Distribution and abundance (+1S.E.) of observed macro-invertebrates in macro-algae sites in the Montebello and Barrow islands. X-axis shows the site numbers.

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Appendix 2 List of fish Feeding guilds with their component species and whether they are targeted by fishers.

Feeding guild	Genus species	Targeted
Corallivore	Chaetodon aureofasciatus	no
	Chaetodon bennetti	no
	Chaetodon lunulatus	no
	Chaetodon plebeius	no
	Chaetodon speculum	no
	Chaetodon trifascialis	no
	Oxymonacanthus longirostris	no
Detritrivore	Ctenochaetus striatus	no
	Dischistodus perspicillatus	no
	Dischistodus prosopotaenia	no
	Ecsenius bicolor	no
Large cropper	Acanthurus grammoptilus	no
	Acanthurus nigricans	no
	Kyphosus vaigiensis	no
	Naso brevirostris	no
	Naso unicornis	no
	Siganus doliatus	no
	Siganus lineatus	no
	Siganus punctatus	no
	Siganus trispilos	no
	Siganus virgatus	no
	Zebrasoma scopas	no
	Zebrasoma veliferum	no
Mobile invertebrates	Cheilinus chlorourus	no
	Cheilinus trilobatus	no
	Choerodon cauteroma	yes
	Choerodon cephalotes	yes
	Choerodon cyanodus	yes
	Choerodon rubescens	yes
	Choerodon schoenleinii	yes
	Coris aygula	no
	Coris caudimacula	no
	Diagramma labiosum	no
	Epibulus insidiator	no
	Glaucosoma magnificum	yes
	Hemigymnus fasciatus	no
	Hemigymnus melapterus	no
	Lethrinus atkinsoni	yes
	Lethrinus laticaudis	yes
	Lethrinus nebulosus	yes
	Ostracion cubicus	no

	Ostracion meleagris	no
	Paraplotosus butleri	no
	Parupeneus barberinoides	no
	Parupeneus indicus	no
	Parupeneus spilurus	no
	Pentapodus emeryii	no
	Pentapodus vitta	no
	Plectorhinchus chaetodonoides	no
	Plectorhinchus flavomaculatus	no
	Plectorhinchus gibbosus	no
	Plectorhinchus multivittatus	no
	Plectorhinchus polytaenia	no
	Plectorhinchus vittatus	no
	Sargocentron diadema	no
	Sargocentron rubrum	no
	Scolopsis bilineata	no
	Scolopsis monogramma	no
	Symphorichthys spilurus	no
	Symphorus nematophorus	no
Omnivore	Platax pinnatus	no
	Pomacanthus semicirculatus	no
	Pomacanthus sexstriatus	no
Piscivore	Aethaloperca rogaa	no
	Aulostomus chinensis	no
	Caranx ignobilis	no
	Cephalopholis argus	yes
	Cephalopholis formosa	yes
	Cephalopholis miniata	yes
	Cromileptes altivelis	yes
	Diploprion bifasciatum	no
	Epinephelus bilobatus	yes
	Epinephelus coioides	yes
	Epinephelus fasciatus	yes
	Epinephelus maculatus	yes
	Epinephelus malabaricus	yes
	Epinephelus polyphekadion	yes
	Epinephelus rivulatus	yes
	Hologymnosus annulatus	no
	Labracinus lineatus	no
	Lutjanus argentimaculatus	yes
	Lutjanus carponotatus	yes
	Lutjanus fulviflamma	yes
	Lutjanus lemniscatus	yes
	Lutjanus quinquelineatus	yes
	Lutjanus russellii	yes
	Lutjanus sebae	yes
	Lutjanus vitta	yes

	Parupeneus cyclostomus	no
	Plectropomus leopardus	yes
	Plectropomus maculatus	yes
	Psammoperca waigiensis	yes
	Satyrichthys moluccense	no
	Triaenodon obesus	no
Scraper/excavator	Chlorurus bleekeri	no
	Chlorurus microrhinos	no
	Chlorurus sordidus	no
	Hipposcarus longiceps	no
	Scarus chameleon	no
	Scarus dimidiatus	no
	Scarus frenatus	no
	Scarus ghobban	no
	Scarus globiceps	no
	Scarus oviceps	no
	Scarus prasiognathos	no
	Scarus rivulatus	no
	Scarus schlegeli	no
	Scarus spinus	no
Sesssile invertebrates	Chaetodontoplus duboulayi	no
Small cropper	Plectroglyphidodon lacrymatus	no
	Stegastes fasciolatus	no
	Stegastes lividus	no
	Stegastes nigricans	no
	Stegastes obreptus	no
Small invertivore	Anampses lennardi	no
	Anampses meleagrides	no
	Chaetodon adiergastos	no
	Chaetodon auriga	no
	Chaetodon ephippium	no
	Chaetodon lineolatus	no
	Chaetodon lunula	no
	Chaetodon ulietensis	no
	Chelmon marginalis	no
	Halichoeres nebulosus	no
	Heniochus acuminatus	no
	Heniochus chrysostomus	no
	Labroides dimidiatus	no
	Macropharyngodon negrosensis	no
	Pomacentrus coelestis	no
	Pseudochromis fuscus	no
	Pseudochromis marshallensis	no
	Pseudodax moluccanus	no
	Thalassoma hardwicke	no
	Thalassoma lunare	no
	Thalassoma lutescens	no

	Zanclus cornutus	no
Small omnivore	Aspidontus taeniatus	no
	Centropyge tibicen	no
	Meiacanthus grammistes	no
	Neoglyphidodon melas	no
	Neoglyphidodon nigroris	no
	Plectroglyphidodon dickii	no
	Pomacentrus bankanensis	no
	Pomacentrus limosus	no
	Pomacentrus milleri	no
	Pomacentrus moluccensis	no
	Pomacentrus nigromanus	no
	Pomacentrus vaiuli	no
Zooplanktivore	Abudefduf bengalensis	no
	Abudefduf septemfasciatus	no
	Abudefduf sexfasciatus	no
	Abudefduf sordidus	no
	Abudefduf spp	no
	Abudefduf vaigiensis	no
	Amblyglyphidodon curacao	no
	Amphiprion rubrocinctus	no
	Apogon wassinki	no
	Caesio caerulaurea	no
	Caesio cuning	no
	Caesio teres	no
	Cheilodipterus quinquelineatus	no
	Chromis atripectoralis	no
	Chromis cinerascens	no
	Chromis margaritifer	no
	Dascyllus reticulatus	no
	Dascyllus trimaculatus	no
	Myripristis kuntee	no
	Neopomacentrus azysron	no
	Neopomacentrus cyanomos	no
	Neopomacentrus filamentosus	no
	Pempheris schwenkii	no
	Pterocaesio digramma	no

Glossary

AICc	Akaike Information Criterion modified for small sample size
ALOS	Advanced Land Observing Satellite
ANOVA	Analysis of variance
AWAC	Acoustic Wave and Current device
BI	Bleaching Index
BoM	Bureau of Meteorology
COTS	Crown-of-thorn Starfish
DBCA	Department of Biodiversity, Conservation and Attractions
dbRDA	Distance-based redundancy analysis
DHW	Degree heating weeks
DISTLM	Distanced based linear model
DPIRD	WA's Department of Primary Industries and Regional Development
EIA	Environmental impact assessments
LNG	Liquid Natural Gas
MBI	Montebello and Barrow islands
MBIMCR	Montebello and Barrow islands marine conservation reserves
MER	Monitoring, Evaluating and Reporting
MI	Macro-invertebrates
MODIS	Moderate Resolution Imaging Spectroradiometer
MOF	Marine offloading facility
PERMANOVA	Permutational analysis of variance
PVC	Polyvinyl Chloride
SSTA	Sea surface temperature anomaly
stereo-DOV	Stereo-Diver Operated Video
TIC	Total inorganic carbon
TOC	Total organic carbon
UVC	Underwater visual census
WA	Western Australia
WIO	West Indian Ocean