Department of **Biodiversity, Conservation and Attractions**

GOVERNMENT OF WESTERN AUSTRALIA

Biodiversity and Conservation Science

Ecological monitoring in the Shark Bay marine reserves 2019

Marine Monitoring Program Report 2

February 2019

Thomas Holmes, Michael Rule, Kevin Bancroft, George Shedrawi, Kathy Murray, Shaun Wilson and Alan Kendrick

Department of Biodiversity, Conservation and Attractions
Locked Bag 104
Bentley Delivery Centre WA 6983
Phone: (08) 9219 9000
Fax: (08) 9334 0498

dbca.wa.gov.au

February 2019

This report/ was prepared by the Marine Science Program.

Questions regarding the use of this material should be directed to:
Dr Thomas Holmes
Marine Science Program
Department of Biodiversity, Conservation and Attractions
Locked Bag 104
Bentley Delivery Centre WA 6983
Phone: (08) 9219 9769
Email: thomas.holmes@dbca.wa.gov.au

The recommended reference for this publication is:
Department of Biodiversity, Conservation and Attractions, 2019, *Ecological monitoring in the Shark Bay marine reserves 2019*, Department of Biodiversity, Conservation and Attractions, Perth.

This document is available in alternative formats on request.

Cover images
Main: Faure Sill – Kevin Crane/DBCA
Inset top: Dugong – Kevin Crane/DBCA
Inset centre: Fish and Turbinaria coral communities – Eva Boogaard/DBCA
Inset bottom: Monitoring seagrass – Mike Rule/DBCA

‘

Contents

Acknowledgments

We wish to thank Dave Holley and Claire Ross from the Department of Biodiversity, Conservation and Attractions, and Brett Molony, Brent Wise, Mike Travers, Claire Smallwood and Karina Ryan from the Department of Primary Industries and Regional Development for editorial comments provided.

Summary

This report presents a synthesis of ecological monitoring within the Shark Bay marine reserves up to the end of 2017. The work presented here is managed by the Department of Biodiversity, Conservation and Attractions’ (DBCA) Marine Science Program (MSP) as part of Science Project Plan 2012/008 and is implemented in collaboration with DBCA Shark Bay District staff, with additional information provided by the Department of Primary Industries and Regional Development (DPIRD) and the Bureau of Meteorology (BoM). Detailed information on the condition of and pressures acting on five of the park’s seven ecological/physical value Key Performance Indicators (KPIs) are presented. Although mangroves and saltmarshes are listed together in the management plan as a single KPI, only mangroves are currently monitored. While periodic monitoring of the Shark Bay dugong population does occur, the data is currently still being synthesized and will be included in later versions of this report. The monitoring of microbial communities and Monkey Mia dolphins does not currently occur as a part of the program coordinated through MSP. The ecological/physical values presented here are:

- Water quality.
- Seagrass communities.
- Mangroves.
- Coral reef communities.
- Finfish communities.

The main risk to ecological values in the reserves is currently increasing seawater temperature associated with global climate change. This is likely to have direct impacts through thermal stress (e.g. coral bleaching, seagrass loss), and indirect impacts through the degradation of essential habitats (e.g. changes in fish communities as a result of losses of primary benthic habitat). There have been major declines in the areal extent of seagrass communities within the marine reserves, primarily as a result of thermal stress caused by anomalously high seawater temperature during the summer of 2010/11 and the synergistic effects of high river discharge in early 2011. The most severe losses of persistent meadow-forming seagrasses occurred on the Wooramel Bank and in the northern parts of the western gulf, primarily due to declines in the extent of *Amphibolis antarctica*. However, the other dominant species, *Posidonia australis*, appeared less effected. More recent data suggests that there has been some seagrass recovery, although losses have continued in the areas of the Wooramel Bank, Faure Sill and Faure Island.

There was a major decline (loss of approximately 135ha) in the areal extent of mangroves in the Carnarvon coast region between 2010 and 2015. The specific cause of this is unknown but was most likely a combination of high terrigenous discharge from the Gascoyne River in 2011, natural oscillations in sea level, below average local rainfall and the effects of cyclone Olwyn in 2015. There were also declines in the extent of mangroves in the Peron Peninsula and west coast regions following 2013. These were most likely caused by natural oscillations in sea level and the effects of cyclone Olwyn in 2015.

The condition of coral communities in the Shark Bay marine reserves remains stable. However, there have been dramatic declines in coral community condition immediately to the north of marine reserves (but within the World Heritage estate) near Bernier and Dorre islands, as a result of major coral bleaching in early 2011. Differential effects between the reserves and the islands immediately to the north are likely a result of localised oceanography and major differences in the composition of coral communities between the locations.

It is currently not plausible to make conclusions on the condition of finfish communities on coral habitat within the marine reserves, due to the short time period over which data has currently been collected. A program focussing on finfish communities within the dominant seagrass communities is in the process of being developed. It should be noted however that there have been recent declines in commercial and charter fishing effort within the region (2008-2016). While recreational fishing effort is considered stable over more recent times (2011/12 – 2015/16), there was a major decline between 1998/99 and 2010 that resulted from changes in fisheries management in the region.

Water quality immediately adjacent to Monkey Mia is considered stable, but the inclusion of appropriate control sites into the sampling design is required to improve confidence in these assessments.

1 Introduction

The Department of Biodiversity, Conservation and Attractions (DBCA) in Western Australia (WA) works with traditional owners to manage and conserve WA's natural and cultural heritage and the State-wide system of terrestrial and marine parks and reserves plays a key role in attaining this goal. In addition to conserving wildlife and landscapes, conservation reserves enrich the lives of all Western Australians, are important locations for tourism and recreation, and provide areas where Aboriginal people can maintain their cultural values (DPaW 2014).

WA currently (as of 2018) has 20 marine parks and reserves created under the *Conservation and Land Management Act 1984*, vested in the Conservation and Parks Commission and in some cases now jointly managed by DBCA and traditional owners. These marine reserves are located from the south to north of the state, across approximately 20° of latitude and more than 13,500 km of coastline. The reserves occur in tropical, sub-tropical and temperate climatic zones and protect a high diversity of habitats and species, including many endemic to the state (Phillips 2001; Roberts 2002; Tittensor *et al*. 2010). Numerous important ecological values are recognised in WA's marine reserves, including many of exceptional conservation significance. Examples include some of the most diverse and extensive seagrass communities currently known, a large marine stromatolite community in Hamelin Pool, and globally significant populations of large marine fauna such as cetaceans, turtles and dugong. The global importance of the WA marine environment has been recognised by the inclusion of both Shark Bay and Ningaloo Reef on the World Heritage register.

The management of WA's marine parks and reserves is guided by management plans which define the key social and ecological values and the primary aims of management, along with the strategies to achieve these aims over a period of ten years. Marine reserve management plans define *performance measures*, which are indicators of management effectiveness, *management targets* which represent the desired endpoint of management and *key performance indicators* (KPIs) which represent social and ecological values of notably high conservation significance and are used as measures for the overall effectiveness of management. A range of management strategies are provided to guide implementation of the management plan by DBCA and other relevant agencies such as the Department of Primary Industries and Regional Development. DBCA's Marine Science Program (MSP) has the role of conducting or facilitating research and monitoring to assist and inform the management of marine reserves and threatened or specially protected marine fauna. The Department's marine monitoring program is managed by MSP and implemented in collaboration with regional management staff.

1.1 Marine Reserve Ecological Monitoring

Monitoring is the collation and analysis of repeated observations over time to detect stability or change (Kingsford and MacDairmid 2000; Koss *et al*. 2005). Scientifically designed and implemented long-term monitoring provides robust quantitative data on trends in the condition of ecological values, the natural and anthropogenic pressures acting on those values and, where relevant, suitable indicators of management response. Using such a condition-pressure-response (CPR) model, monitoring is a key component of adaptive management by providing knowledge to assess management effectiveness and refine practices (see Evans *et al*. (2017) for an example). A key challenge of ecological monitoring for management purposes is being able to distinguish between the effects of natural variation and changes caused by anthropogenic influences (Magurran *et al*. 2010). Aspects of the natural environment can vary considerably over timeframes that may be diurnal, seasonal or annual in response to, for example, the influences of oceanography (e.g. tidal cycles), ecological processes (e.g. recruitment, spawning behaviour) or even periodic disturbance events like cyclones. Such variability must be considered and accounted for when designing spatial and temporal sampling to monitor key ecological values like fishes and benthic cover in marine reserves (Underwood 2000). Importantly, the capacity for monitoring to inform management with a high degree of confidence increases over time. The collection of extended time-series data provides a far stronger ability to understand trends than time-series comprised of relatively few data points and increases the capacity (expressed in this report as ‘confidence’) to inform management with greater certainty.

The DBCA marine monitoring program is being implemented incrementally under a structured framework to ensure that consistent indicators and monitoring methods are used, where possible, to enable both temporal and spatial comparisons of long-term data. This program focuses on key ecological values (with priority given to KPIs) and employs a variety of monitoring indicators and methods. Sampling also takes place at varying spatial and temporal scales to account for localised characteristics, pressures and management objectives. Annual monitoring priorities are primarily based on trends in the condition of ecological values, the significance of pressures acting on the value and time since the previous monitoring survey. The frequency of sampling is also assessed in relation to the natural variability associated with different ecological values.

1.2 Shark Bay Marine Reserves

The Shark Bay marine reserves (SBMR), comprising Shark Bay Marine Park and Hamelin Pool Marine Nature Reserve, are located on WA’s arid central coast between approximately 24 and 26˚ S and approximately 650 km north of Perth (Figure 1.1). Shark Bay is a large north-west facing embayment that forms Australia’s most westerly point. The bay comprises two gulfs separated by the Peron Peninsula and bounded to the west by ridges of Pleistocene Tamala limestone that form the Edel Land Peninsula and west coast of Dirk Hartog Island. Shark Bay forms an extensive low-energy and mostly shallow marine environment on an otherwise high-energy rocky coast extending southwards along the Zuytdorp Cliffs and northwards towards Quobba and Red Bluff (Playford 1990). The unique physical and biological features of Shark Bay and its outstanding conservation values have led to it being recognised as a discrete marine bioregion (DEH, 2006). In 1991, the whole of Shark Bay was inscribed on the World Heritage List for its outstanding natural values.

Shark Bay Marine Park (748,725ha) and Hamelin Pool Marine Nature Reserve (132,000 ha) were created in 1990 (CALM, 1996) and together represent the WA’s largest marine protected area outside of the Kimberley region. Shark Bay’s distinctive physical features include hypersaline inner gulf environments, most notably in Hamelin Pool, which are created and maintained by the combined influences of geology, high evaporation, limited oceanic circulation and limited freshwater input from rain or waterways. In fact, no permanent surface water exists in the Shark Bay region and the Wooramel and Gascoyne rivers typically only discharge periodically in response to cyclones. The bay supports numerous important biological communities, including extensive seagrass meadows, rare marine stromatolites, and extensive populations of large marine fauna such as dugong. It is generally considered to have limited ecological connectivity with regions to the north and south (Thomas *et al*. 2017; DiBattista *et al*. 2017).

The resident population of the Shark Bay area is centred at the small towns of Carnarvon (population ca. 5,000), Denham (population <1000) and Useless Loop (population <150), which is a company settlement associated with solar salt production (http://www.censusdata.abs.gov.au). The broader hinterland supports only low-density pastoralism and intensive agriculture on the flood-plains of the Gascoyne and Wooramel rivers. The Shark Bay area is a popular domestic and international tourist destination with more than 97,000 visitors to the region annually (Tourism Research Australia 2015). Camping in caravan parks or coastal camps and recreational fishing, including shore-based, private boat-based and charter fishing, are popular visitor activities (Smallwood and Gaughan 2013; Tourism Research Australia 2015). There is also significant nature-based tourism focusing on wild dolphin encounters and fauna viewing from vessels at Monkey Mia. The Shark Bay Beach Seine and Mesh Net Managed Fishery is the sole commercial fishery that operates entirely within the marine reserve boundaries.

The physical/ecological values presented here are identified from the current Shark Bay marine reserves management plan (1996-2006). While KPIs are not identified in this management plan, they are identified here using expert local and scientific knowledge of the primary values of ecological and social importance to the reserves. These are:

- Geomorphology.
- Sediment quality.
- Water quality (**KPI**).
- Microbial communities (**KPI**).
- Seagrass communities (**KPI**).
- Mangroves and saltmarshes (**KPI**).
- Coral reef communities.
- Finfish communities (**KPI**).
- Dugong (**KPI**).
- Monkey Mia dolphins (**KPI**).
- Macroalgal communities.
- Invertebrate communities.
- Cetaceans.
- Marine turtles.
- Sea snakes.
- Seabirds.

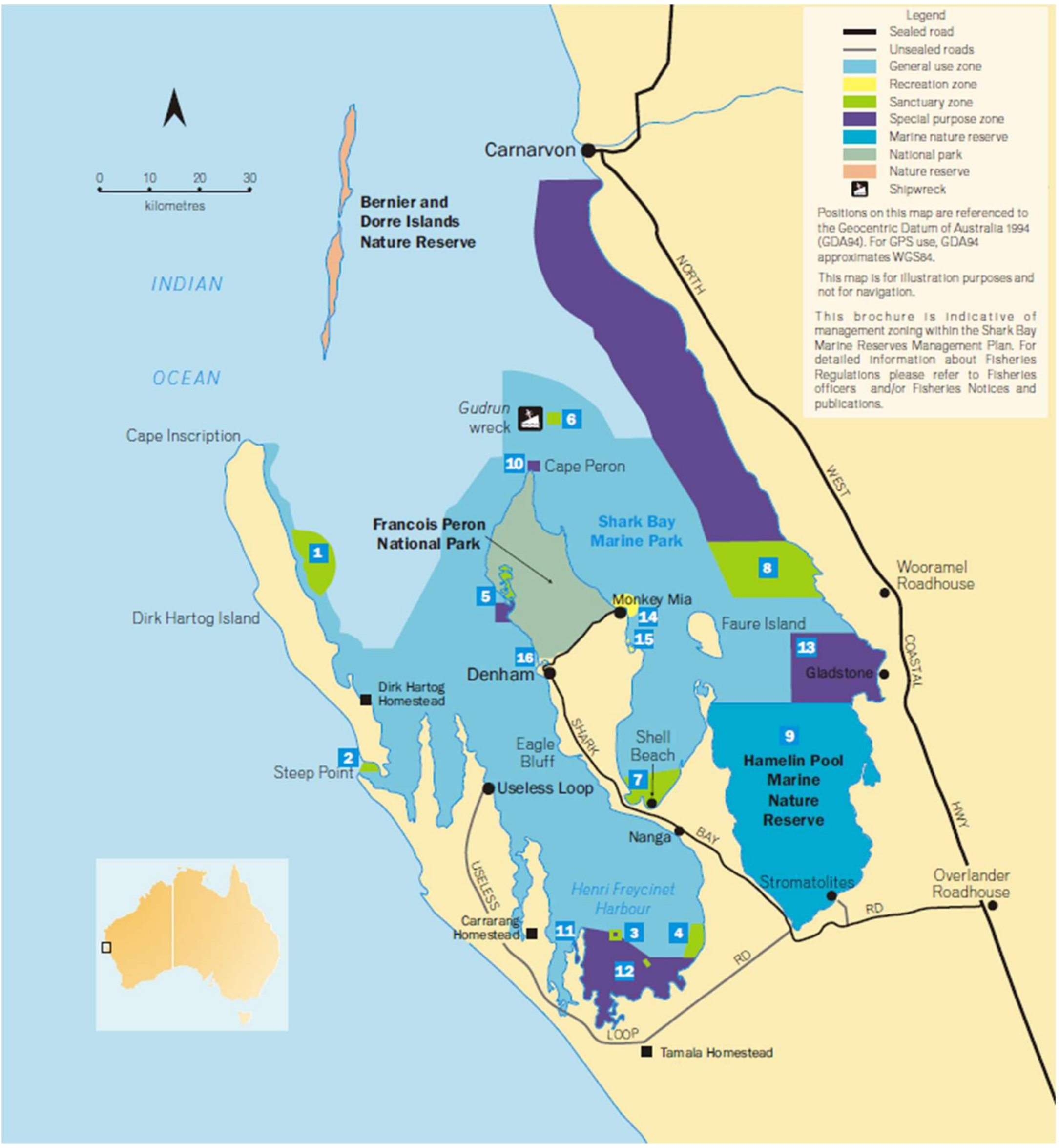

Figure 1.1: Map of the location and management zoning of the Shark Bay marine reserves, comprising Shark Bay Marine Park and Hamelin Pool Marine Nature Reserve.

1.3 Scope of this report

This report provides a summary of ecological marine monitoring undertaken at the Shark Bay marine reserves up to 2017. This work comprises part of DBCA's marine monitoring program that is managed by MSP and develops consistent indicators and sampling methods to monitor key ecological values across WA's marine parks and reserves. While the program conforms to a condition-pressure-response model, the focus so far has been on developing quantitative condition and pressure time-series data, and indicators of management response are not yet reported.

The information in this report provides a benchmark assessment of the condition of key ecological values of the Shark Bay marine reserves and some of the pressures acting on them. The analyses and synthesis information provided here and in subsequent updated reports will inform adaptive management of the Shark Bay marine reserves by providing a knowledge-based understanding around key management objectives. The information in the report can also be used by managers for performance assessment reporting and to inform external committees and agencies in relation to, for example, World Heritage and State of the Environment reporting. The information can also be used to inform stakeholder engagement, to respond to public inquires and to develop educational material relating to the Shark Bay marine reserves.

The data presented here have been primarily collected by MSP, Shark Bay District staff and collaborators. Also included are analyses of data obtained by agreement from the Department of Primary Industries and Regional Development (DPIRD), as well as data from the Bureau of Meteorology (BoM), CSIRO and National Oceanographic and Atmospheric Administration (NOAA) that is in the public domain. Data have only been included if they conform to Departmental requirements of utility and quality as part of a long-term CPR monitoring program. For this reason, data associated with short-term research are typically not included unless they align with appropriate indicators, site selection criteria and methods and can form the basis for ongoing monitoring. To date, information is being presented for water quality, finfish communities, seagrass communities, coral communities and mangrove communities. Additional monitoring information relating to the Shark Bay marine reserves, such as for marine turtles, is managed and reported by other branches of the Department and is not included here.

Importantly, this report comprises a benchmark summary of a developing monitoring program. Trends in condition and pressure indicators are characterised as stable, increasing or decreasing based on the direction of change. The relative effect of that trend on the overall state of the ecological value being assessed (i.e. no effect, positive effect, negative effect) is indicated based on the colour coding system displayed in Table 1.1. For the purposes of this report, a ‘trend’ is defined as either a consistent increase or decrease in a metric over three or more consecutive sampling periods, or a statistically significant modelled trend over longer time frames. Each indicator assessment is associated with a level of confidence which provides the reader with an indication of certainty associated with interpreting the data. This confidence level is based on;

- the number of sampling periods,
- the time period over which they have occurred relative to the level of variability in the data being collected (e.g. more mobile groups such as large fish are typically associated with high levels of variability)
- the spatial scale over which data has been collected relative to distribution.

The volume and breadth of monitoring data will increase over time and this will provide greater confidence in the analysis and interpretation of trends. This developing complexity will be reflected in subsequent reports.

Table 1.1: Colours used in summary indicator tables to indicate the relative effect of temporal trends in condition and pressure indicators on the overall ecological value being assessed.

Effect on Ecological Value	Indicator Cell Colour
Positive Effect	
No Effect	
Negative Effect	

2 Water Quality (KPI)

2.1 Key Points

- There has been a slow increasing trend in seawater temperature in all sectors across the last 32 years, with the greatest increase occurring in the western gulf (east) and eastern gulf. It is highly likely that this trend is associated with climate change. Confidence in this assessment is high due to the extended time over which continuous data is available.
- Targeted sampling at Monkey Mia suggests that nutrient (nitrogen and phosphorous) and pathogen concentrations have remained stable over the monitoring period and are comparable to control levels. However, unusually high concentrations of nitrogen and phosphorous were recorded in the most recent samples from 2016, although the cause of this is currently unknown.

2.2 Indicator Summary

	Trend	**Confidence**
Condition		
Seawater Temperature	Increasing	High
Nitrogen (Monkey Mia)	Stable	Medium
Phosphates (Monkey Mia)	Stable	Medium
Pathogens (Monkey Mia)	Stable	Medium
Pressure		
Climate Change	Increasing	High
Terrestrial Runoff (Monkey Mia)	Stable	Medium

2.3 Condition Indicators

Condition indicators are measures of water quality 'state' relative to pressures acting in the marine reserves. While many indicators can be considered for identifying changes in water quality condition (Scheltinga *et al*. 2004), those most relevant to water quality in the Shark Bay marine reserves and presented in this report are:

- Seawater temperature.
- Nitrogen.
- Phosphates.
- Pathogens.

Water temperature is a major driver of marine community composition (Hoegh-Guldberg and Bruno 2010). Changes in water temperature can cause thermal stress to benthic communities, which is often assessed with local temperature thresholds (Eakin *et al*. 2010). As such, anomalous temperature fluctuations or longer term changes can have significantly impacts, such as coral bleaching, seagrass die-off, changes in fish physiology, or even range shifts if such conditions are sustained (Sorte *et al*. 2010; Neuheimer *et al*.

2011; Hughes *et al*. 2017). Nitrogen (N) and phosphorous (P) are the two primary elements involved in marine nutrient cycling (Scheltinga *et al*. 2004). While both occur naturally and underpin productivity in coastal ecosystems, an overabundance can cause eutrophication. Pathogens, such as faecal streptococci/enterococci and faecal coliforms (*Escherichia coli*), are indicative of the presence of human waste in the water column (Scheltinga *et al*. 2004). The most likely sources of anthropogenic nutrient and pathogen inputs into the marine reserves are from horticulture within the catchments of the Gascoyne and Wooramel rivers and sewage leaching from settlements. The Monkey Mia settlement on the east coast of Peron Peninsula has been identified as a potential source of sewerage input. As such pathogen and nutrient concentrations are assessed at this location (EPA 1989). Sewage storage and treatment at other settlements adjacent to the marine reserves (i.e. Denham, Carnarvon and Useless Loop) are currently considered adequate to prevent discharge into local waters. While nutrient concentrations may be elevated in the eastern gulf area when the Gascoyne and Wooramel rivers discharge, an appropriate program to monitor these indicators at a relevant scale is yet to be implemented.

Chlorophyll-*a* is used as a proxy measure of phytoplankton biomass and is a commonly used indicator of water column eutrophication (Scheltinga *et al.* 2004). Changes in the concentration of chlorophyll-*a* can be driven by natural oceanographic processes, including upwelling, currents (Morales *et al.,* 1996; Sokolov and Rintoul, 2007) or storm activity (Chang *et al*. 1996), as well aanthropogenic inputs such as fertilizer runoff or dredging (Brodie *et al*., 2010). Appropriate methods for monitoring chlorophyll-*a* at a scale relevant to the marine reserves are currently being developed and are not yet reported here.

2.3.1 Seawater Temperature

Current assessments are based on modelled *in situ* seawater temperature (mIST) (Baldock *et al.* 2014) derived from NOAA satellite sea surface temperature estimations from a 50 x 50 km virtual station to the north-west of Dirk Hartog Island (http://coralreefwatch.noaa.gov/satellite/vs/) in conjunction with *in situ* temperature loggers (Table 2.1). Analyses were based on eastern gulf, Hamelin Pool, western gulf (west) and western gulf (east) sectors of the Shark Bay marine reserves (Table 2.1, Figure 2.1), which were identified according to a natural separation of the physical oceanographic patterns of the region (Burling *et al.*, 2003; Nahas *et al.*, 2003; Hetzel *et al.*, 2015). The western gulf (south) is identified as another distinct sector, but temperature loggers are yet to be installed to assess trends in this area.

Table 2.1: Location of NOAA 50 x 50 km Northwest Dirk Hartog virtual station and *in situ* temperature loggers used to derive mIST time series for the eastern gulf, Hamelin Pool, western gulf (east) and western gulf (west) sectors of the Shark Bay marine reserves.

Sector	Logger Location
Eastern gulf	Redcliffe Bay
Hamelin Pool	Hamelin Pool
Western gulf (east)	Denham
Western gulf (west)	Sandy Point

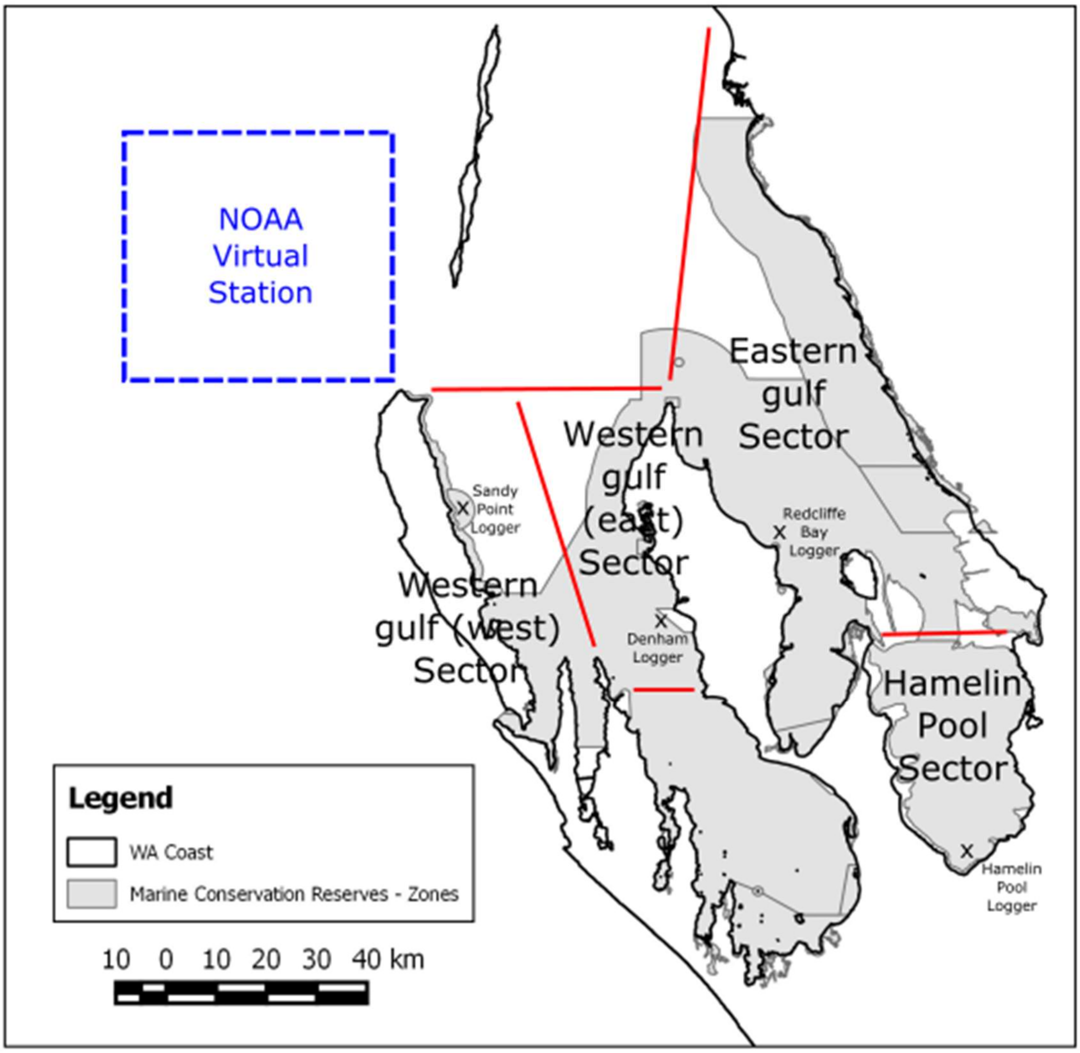

Figure 2.1: The eastern gulf, Hamelin Pool, western gulf (west) and western gulf (east) sectors of the Shark Bay marine reserves identified in this report and the positions of *in situ* temperature loggers and NOAA 50 x 50 km sea surface temperature virtual station.

All sectors show medium-term oscillations in seawater temperature over 5-10 years resulting from regional climatic cycles associated with the Southern Oscillation Index (Figure 2.2). All four sectors also exhibit a slowly increasing trend in seawater temperature from 1985-2017, amounting to a change of between 0.34 and 0.91^{o}C across this period (Figure 2.2; seasonally adjusted Mann-Kendall; eastern gulf Q= -0.163, p<0.001; Hamelin Pool Q= -0.248, p<0.001; western gulf (east) Q= -0.183, p<0.001; western gulf (west) Q= -

0.128, p<0.001). Temperature increase was greatest in the western gulf (east) and eastern gulf sectors.

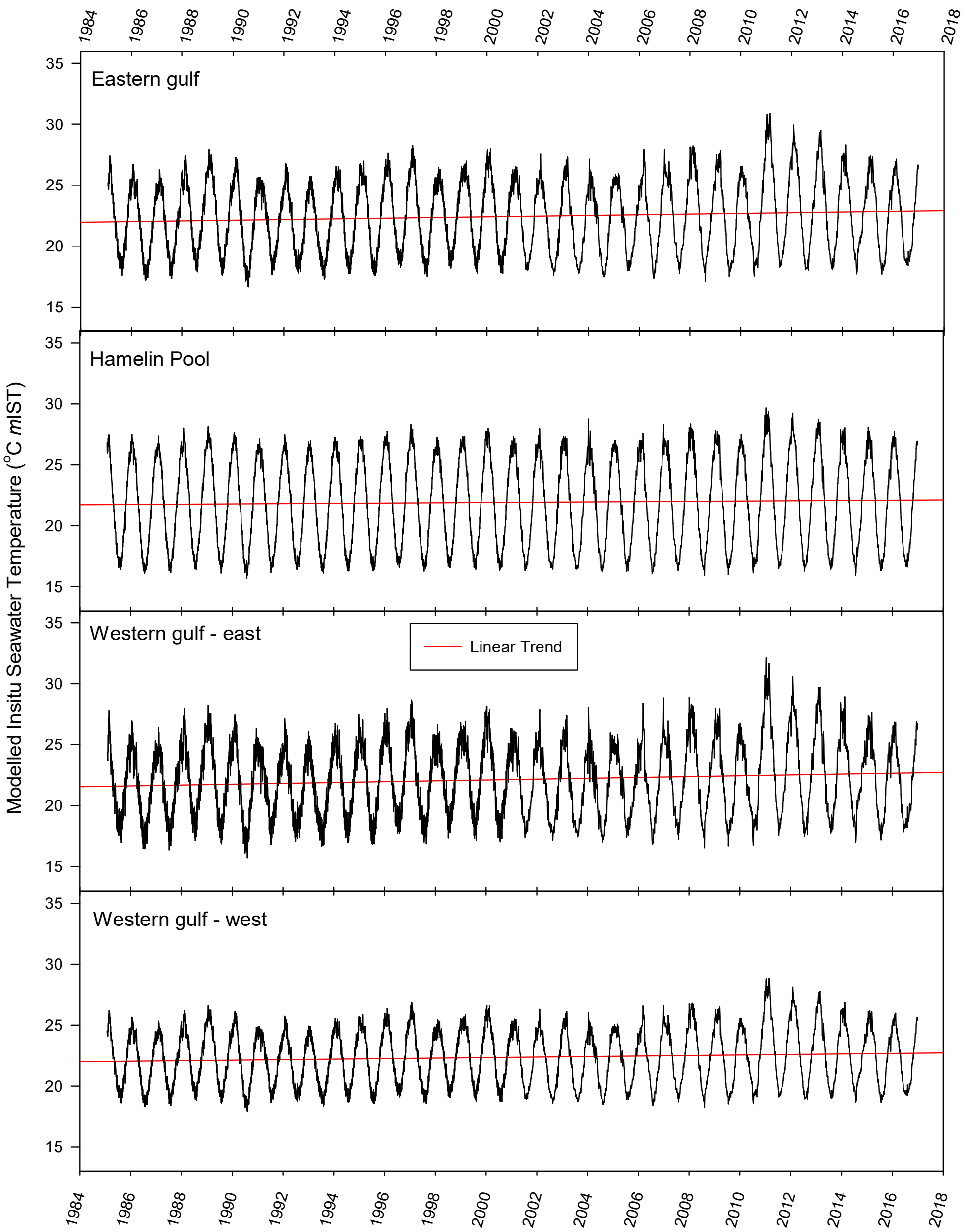

Figure 2.2: Mean seawater temperature for the eastern gulf, Hamelin Pool, western gulf (east) and western gulf (west) of the Shark Bay marine reserves, 1985-2017. Mean seawater temperature is calculated using modelled *in situ* seawater temperature (mIST) averaged twice weekly across nocturnal periods. Red lines indicate significant trends based on seasonally adjusted Mann-Kendal trend analyses.

Source: NOAA (2017)

Instances of anomalously high seawater temperature (identified as exceeding two standard deviations from the long-term mean) occurred in all four sectors since July 2010 (Figure 2.3). These events typically occurred in the summer and autumn months, between December 2010 and April 2011, December 2011 and March 2012, and January 2012 and March 2013. Anomalously high temperatures were most severe in 2010/11.

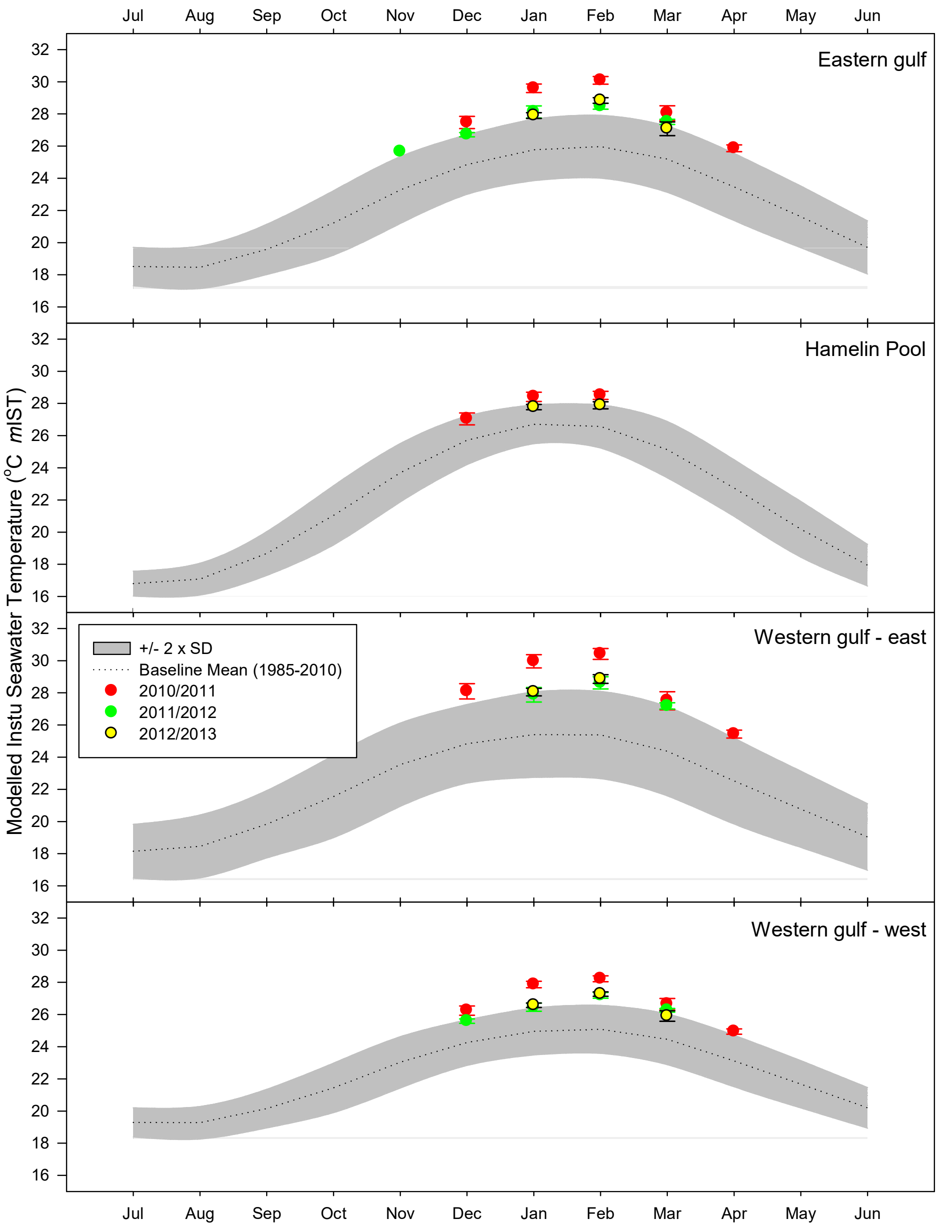

Figure 2.3: Anomalous mean (± SE) seawater temperature for the eastern gulf, Hamelin Pool, western gulf (east) and western gulf (west) sectors of the Shark Bay marine reserves from July 2010 to June 2016. Values were calculated using modelled *in situ* seawater temperature (mIST) averaged twice weekly across nocturnal periods (n=8 per month). Dashed lines indicate baseline means for 1985-2010 and grey shading indicates two standard deviations around these means. Mean values beyond two standard deviations are considered anomalous and are marked in colour.

Source: NOAA (2017).

2.3.2 Nitrogen

Nitrogen is assessed as both total nitrogen (both the soluble and insoluble nitrogen forms) and dissolved inorganic nitrogen (the bio-available nitrogen species most readily absorbed by plants). No assessment has been made of nitrogen concentrations across the whole marine reserves. However, sampling has been conducted near the Monkey Mia resort since 2001 by the private owners as an operational condition, due to the risk associated with sewage leaching from septic systems (EPA 1989) and the resort wastewater treatment facility. Water samples are collected in 1m water depth from five replicate sites along the foreshore adjacent to the Monkey Mia resort. As no control sites are incorporated as a part of compliance monitoring conditions, ANZECC/ARMCANZ guideline levels for nitrogen in inshore waters are being utilised for reference purposes (ANZECC/ARMCANZ 2000). It is, however, acknowledged that these threshold levels are indicative only and may not be appropriate for this location, and there is a need to incorporate control sites into the current sampling design.

The median concentrations of total nitrogen have remained relatively stable at approximately 200-300 μg N.L^{-1} since 2001 (Figure 2.4). The exceptions to this were 1989 and 2004, when concentrations were elevated with a median of approximately 480 μg N.L^{-1} and 410 μg N.L^{-1}, respectively, and 2016 when concentrations were highly elevated with a median of approximately 600μg N.L^{-1}. Although median concentrations have been relatively stable across much of the monitoring period, it should be noted that nine of the ten sampling years returned median total nitrogen concentrations above existing ANZECC/ARMCANZ guideline levels for inshore waters of 230 μg N.L^{-1} (ANZECC/ARMCANZ 2000).

The median concentration of dissolved inorganic nitrogen has generally remained stable at approximately 8-25 μg N.L^{-1} across the monitoring period (Figure 2.4). Exceptions to this were elevated concentrations in 2004 (median ~60 μg N.L^{-1}) and 2002 (median ~100 μg N.L^{-1}), and highly elevated concentrations in 1989 and 2016 (both with a median of ~250 μg N.L^{-1}). It should be noted that all sampling years have recorded median concentrations on or above existing ANZECC/ARMCANZ guideline levels for inshore waters of 10 μg N.L^{-1} (ANZECC/ARMCANZ 2000).

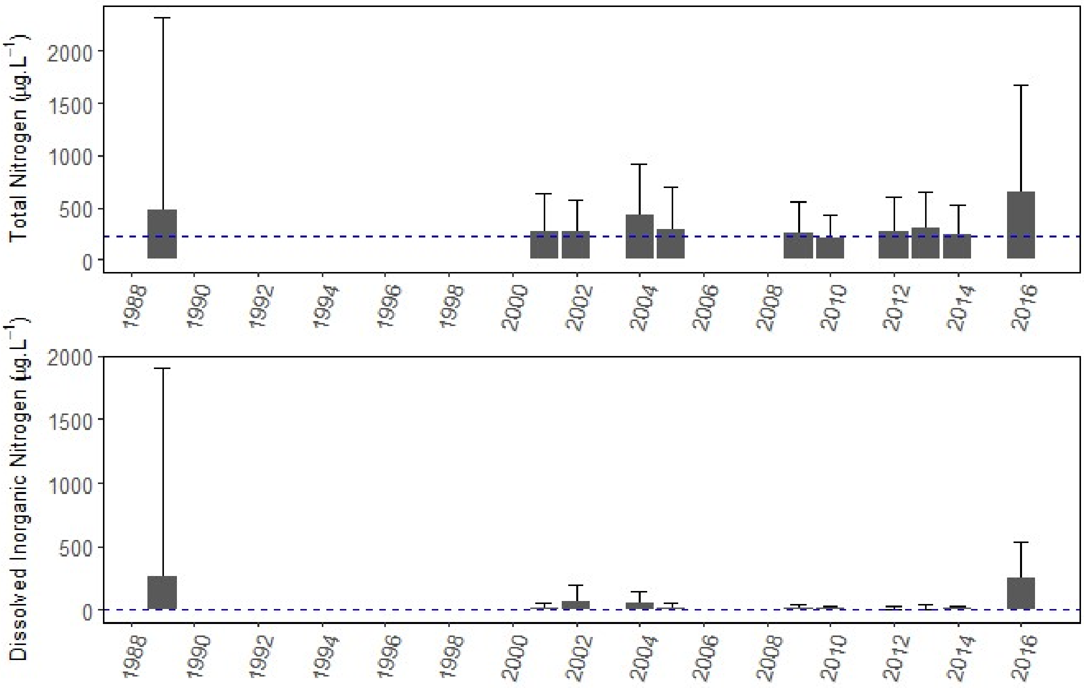

Figure 2.4: Median total nitrogen and dissolved inorganic nitrogen concentrations (± 90th percentiles) for nearshore waters near Monkey Mia in the Shark Bay marine reserves, 1989-2016. 1989 n = 3; 2001, 2004 and 2009 n=10; 2002, 2010-2014 n=5; 2016 n=4. Blue dashed lines indicate threshold water quality limits for inshore waters set by Australia and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand. This indicates generic guidelines for comparison and should not be considered as baseline concentrations for this location.

Source: Environmental Protection Authority; Department of Biodiversity, Conservation and Attractions; RAC Monkey Mia Resort.

2.3.3 Phosphates

Phosphates are assessed as both total phosphorous (both dissolved and particulate) and orthophosphate (the bio-available phosphate species most readily absorbed by plants). No assessment has yet been made of phosphate concentrations across the whole Shark Bay marine reserves. However, sampling has been conducted around the Monkey Mia resort due to the risk associated with sewage input from septic systems (EPA 1989) and the resort wastewater treatment facility. Water samples are collected at a depth of 1m from five replicate sites along the foreshore adjacent to the Monkey Mia resort. Given that no control sites are incorporated into ongoing monitoring, ANZECC/ARMCANZ guideline levels for phosphates in inshore waters are being utilised for reference purposes (ANZECC/ARMCANZ 2000). However, it is acknowledged that these threshold levels may not be appropriate for this location, and there is a need to incorporate control sites into the current sampling design.

Between 1989 and 2005, median total phosphorous concentrations were relatively stable, ranging from approximately 22 to 42 μg P.L^{-1} (Figure 2.5). Median concentrations then declined and were stable at approximately 8-19

μg P.L^{-1} between 2009 and 2014. Concentrations were elevated during the most recent survey in 2016 (median of approximately 55 μg P.L^{-1}), albeit with high levels of variance. The 2016 median concentration was also well above existing ANZECC/ARMCANZ guideline levels for inshore waters of 30 μg P.L^{-1} (ANZECC/ARMCANZ, 2000).

Orthophosphate concentrations have only been assessed in 1989 and between 2010 and 2014 (Figure 2.5). Median concentrations were initially high in 1989 at approximately 6 μg P.L^{-1}, and above existing ANZECC/ARMCANZ guideline levels for inshore waters (5 μg P.L^{-1}) (ANZECC/ARMCANZ, 2000). However, median concentrations recorded between 2010 and 2014 were lower and ranged from approximately 1 to 4 μg P.L^{-1}.

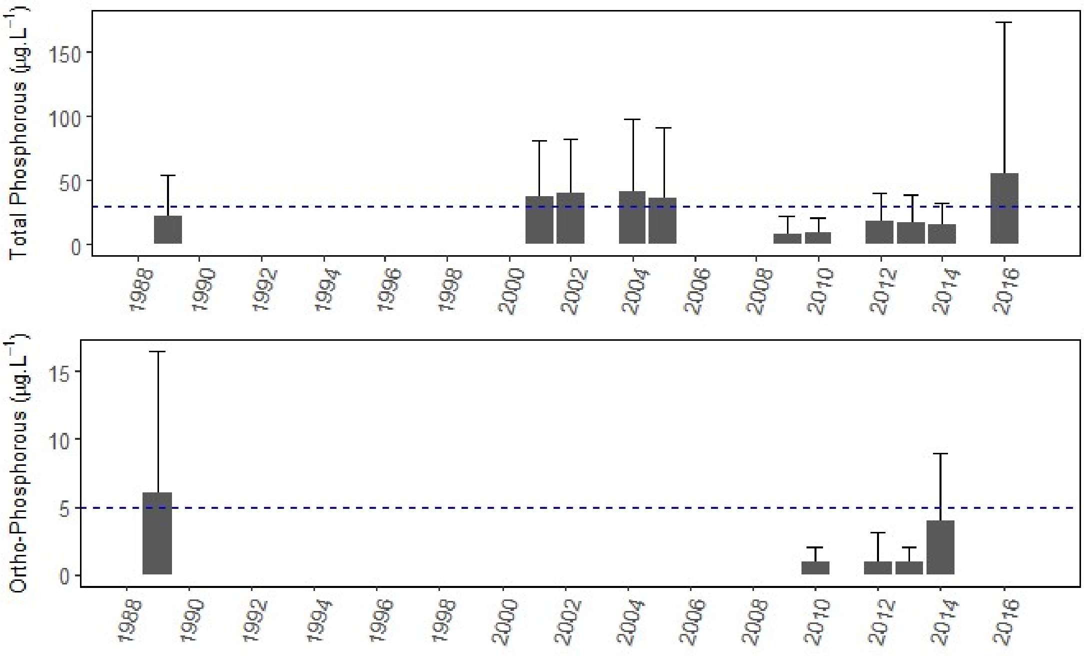

Figure 2.5: Median total phosphorous and orthophosphate concentrations (± 90th percentiles) for nearshore waters around Monkey Mia in the Shark Bay marine reserves, 1989-2016. 1989 n=3; 2010-2014 n=5; 2016 n=4. Blue dashed lines indicate threshold water quality limits for inshore waters set by Australia and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand. This indicates generic guidelines for comparison and should not be considered as baseline concentrations for this location.

Source: Environmental Protection Authority; Department of Biodiversity, Conservation and Attractions; RAC Monkey Mia Resort.

2.3.4 Pathogens

Pathogens are measured as the concentrations of faecal *Enterococci* spp. and are an indicator of contamination by human waste. No assessment has been made of pathogen concentrations across the whole of the marine reserves as this risk is considered to be relatively low. However sampling has been conducted at Monkey Mia due to the elevated risk associated with historical waste treatment in this area (EPA 1989; Stoddart 1990). Water samples are collected at a depth of 1 m from five replicate sites along the foreshore of Monkey Mia. Increased sampling effort occurred in 2010 because unusually high concentrations were measured in initial samples. As no control sites are incorporated into ongoing monitoring, ANZECC/ARMCANZ guideline levels for pathogens in inshore waters are used for reference purposes (ANZECC/ARMCANZ, 2000). It is, however, acknowledged that these threshold levels may not be appropriate for this location, and there is a need to incorporate control sites into the current sampling design.

The annual median *Enterococci* concentration has remained stable at 0-10 cfu 100 mL^{-1} across the monitoring period (Figure 2.6), well below existing ANZECC/ARMCANZ guideline levels for inshore waters (42 cfu 100 mL^{-1}) (ANZECC/ARMCANZ, 2000).

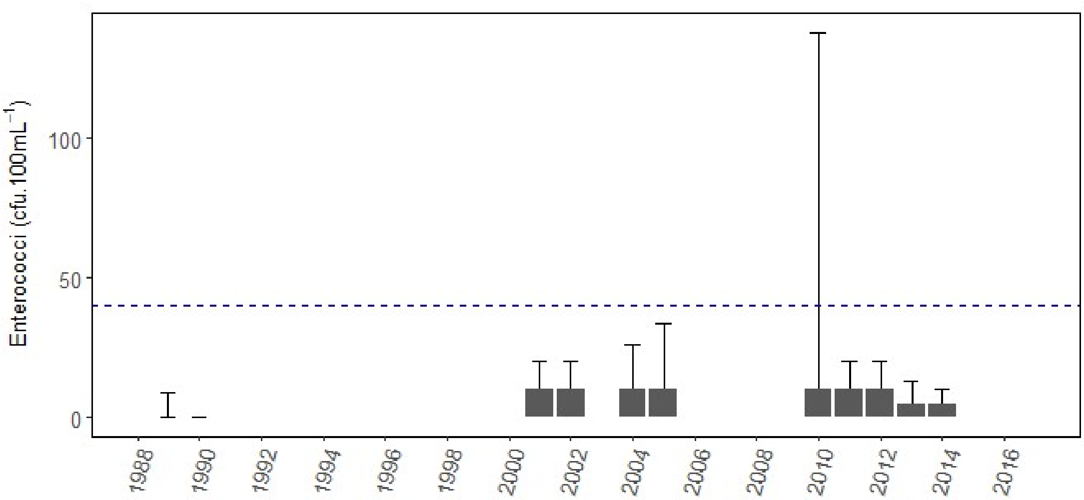

Figure 2.6: Median *Enterococci* concentrations (± 90th percentiles) in nearshore waters near Monkey Mia in the Shark Bay marine reserves, 1989-2016. 2005, 2016 n=4; 1989, 2004, 2012, 2014 n=5; 2001, 2011, 2013 n=10; 2010 n =105). Blue dashed lines indicate threshold water quality limits for inshore waters set by Australia and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand. This indicates generic guidelines for comparison and should not be considered as baseline concentrations for this location.

Source: Department of Biodiversity, Conservation and Attractions; Environmental Protection Agency; RAC Monkey Mia Resort.

2.4 Pressure Indicators

Pressure indicators are reliable measures of the primary causes of change in water quality within the marine reserves. Those indicators relevant to the Shark Bay marine reserves and presented in this report are:

- Terrestrial runoff.

Climate change and sewage input are considered the primary pressures acting on water quality in the Shark Bay marine reserves. It is now highly likely that anthropogenic greenhouse gas emissions are driving climate change that has substantially contributed to increases in the heat content of the upper oceans since at least the 1970s and most likely before (IPCC 2014). While greenhouse gas emissions are the primary driver of climate change and ocean warming at a global scale, the effect is often spatially patchy and identification of warming at smaller spatial scales (i.e. 10's-100's km) at this time must be considered in relation to this variation and natural factors (Rhein *et al*. 2013). However, it is highly likely that climate change is the primary cause of the 0.34-0.91°C increase in mean seawater temperature since 1985 presented here, and climate change represents the most significant pressure acting on water quality condition in the Shark Bay marine reserves. The most appropriate indicator of climate change pressure acting on water temperature in the Shark Bay marine reserves is still being developed.

The Wooramel and Gascoyne rivers discharge into or near the eastern gulf sector of the Shark Bay marine reserves. No other major waterways flow into the reserve area and runoff primarily occurs due to episodic rain events within the catchment which can deliver terrestrial nutrients, sediments and contaminants into adjacent coastal waters. While this is a major pressure on water quality in the reserves, the relevant condition indicators are not currently monitored at locations appropriate to this pressure (i.e. throughout the eastern gulf sector and specifically along the east coast). As such, this pressure is not presented in this report.

Sewage input is likely a combination of septic tank seepage and terrestrial runoff. This pressure is considered minor in the context of the whole marine reserves and is likely isolated to areas of human settlement where vulnerable sewage storage may be present, such as Monkey Mia (EPA 1989) and Nanga Bay Resort. Methods for assessing the risk of seepage from vulnerable sewage storage are not currently incorporated into the monitoring program and as such are not presented in this report.

2.4.1 Terrestrial Runoff

Terrestrial runoff is currently only reported for the Monkey Mia area, where relevant condition indicators are also assessed (i.e. nitrogen, phosphorous, pathogens). Rainfall recorded at Denham is being used as a coarse indicator of terrestrial runoff in this region. While it is not a direct measure of terrestrial runoff, it is the best available surrogate for above and below ground runoff/seepage. Data from 1945 to 2016 highlights the large degree of variation in rainfall across this period. Mann Kendall trend analysis indicates that there is no discernible long term increasing or decreasing trend (Figure 2.7). Since 2000, there have been high levels of total annual rainfall (350-380 mm) recorded in 2000, 2008, 2011 and 2015, compared to the long-term mean.

Rainfall is typically highest in the Denham area between May and August, although there is increased variability between January and April associated with cyclones (Figure 2.8). Since July 2010, there have been anomalous rainfall events (more than 2 standard deviations above the long-term mean) in December 2010, January and February 2011 and March 2015.

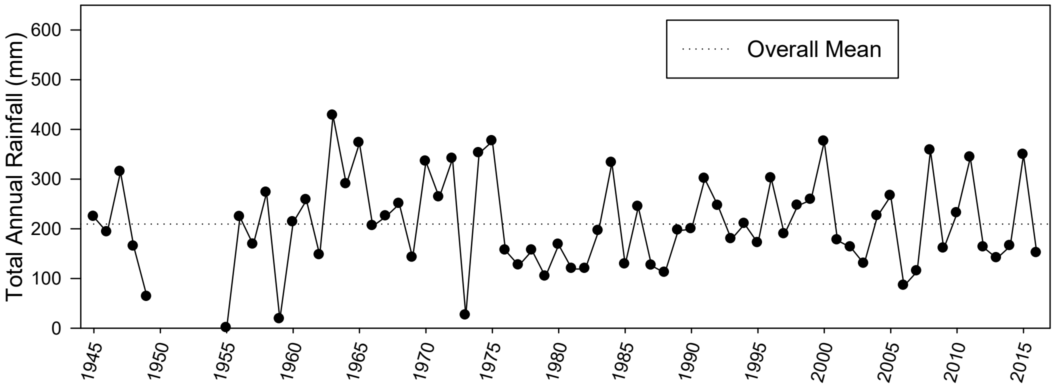

Figure 2.7: Annual rainfall for Denham, 1945-2015. Black dashed line indicates overall mean rainfall for 1945-2010.

Source: Bureau of Meteorology (2017).

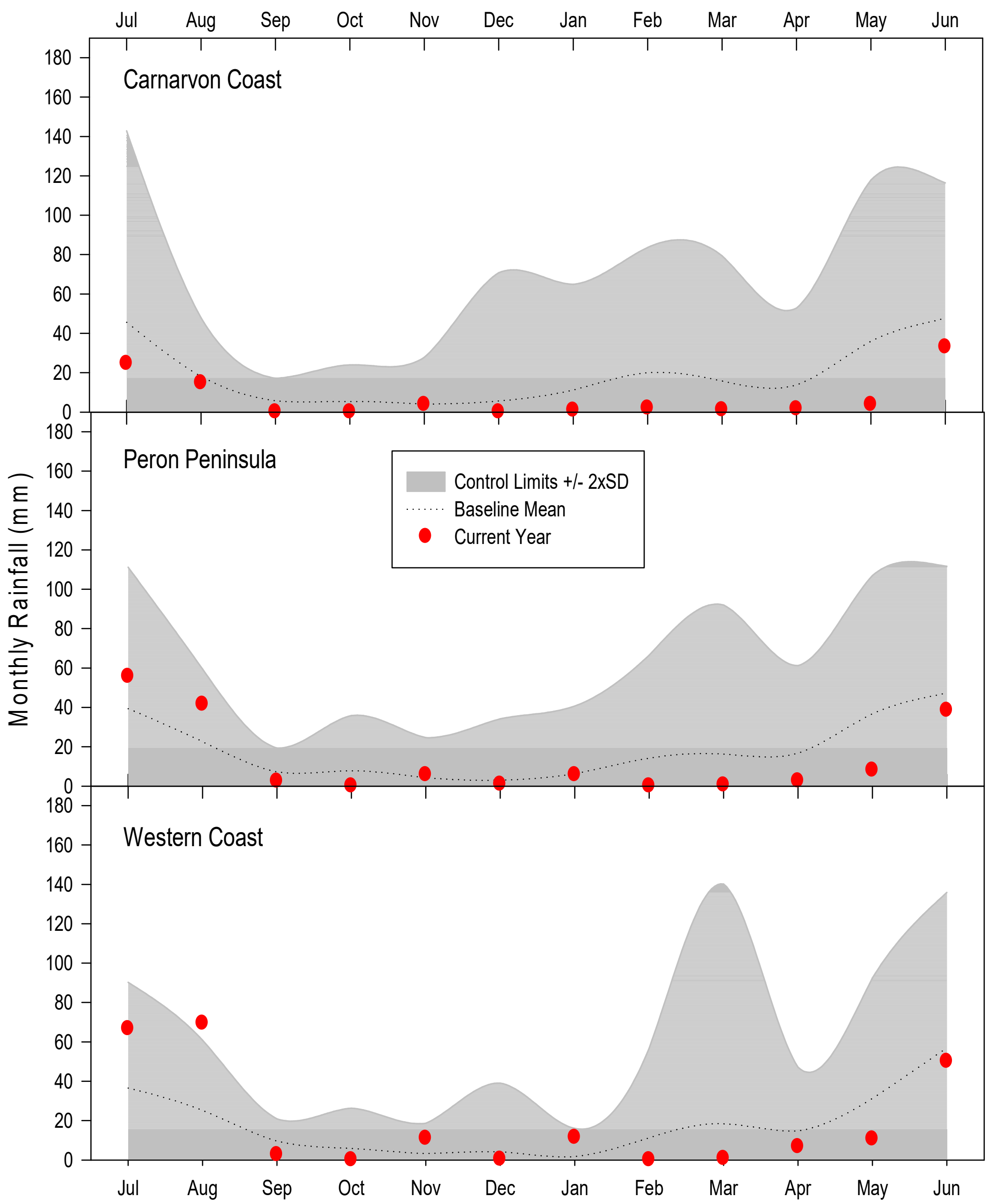

Figure 2.8: Total monthly rainfall for Denham, July 2010 - June 2016. Dashed line indicates baseline rainfall mean (1945-2010). Grey shaded area indicates two standard deviations from the mean, with values outside this considered anomalous.

Source: Bureau of Meteorology (2016).

2.5 Synthesis

The highly variable and sporadic nature of rainfall in the Shark Bay region indicates that water quality condition (excluding water temperature) in the marine reserves is likely to be primarily influenced by runoff and river flow from episodic rain events. While discharge from the Gascoyne and Wooramel rivers likely plays a significant role in this, the current monitoring program focuses on a single point source of potential impact around the Monkey Mia area. Methods for assessing water quality condition and pressure over larger spatial scales, and the incorporation of river discharge, are currently being developed.

The long-term trends in seawater temperature across the four sectors of the marine reserves are relatively consistent, with evidence to suggest an increasing trend across the monitoring period in all sectors (eastern gulf, Hamelin Pool, western gulf (east) and western gulf (west)). Seawater temperature increases were greatest in the eastern gulf and western gulf (east) sectors. This trend of increasing seawater temperature in the Shark Bay marine reserves over the past 30 years is consistent with those recorded elsewhere in Western Australia (Abdo *et al.* 2012), nationally (CSIRO and BOM 2016; Hughes *et al.* 2017) and internationally (Abraham *et al*. 2013). Of particular note was the abnormally high-water temperature that occurred across all four sectors in the summer months of 2010/11, 2011/12 and 2012/13. These events were part of seawater warming that occurred along much of the Western Australian coastline during these years due to strong La Niña conditions (Pearce *et al.* 2011). These trends are most likely caused by climate change driven by anthropogenic greenhouse gas emissions (Levitus 2001; Rhein *et al*. 2013). New remote sensing products from NOAA will enable the application of 5km satellite-derived seawater temperature data to models of SST based on *in situ* temperature loggers currently deployed throughout the reserves. This will enable a more detailed and accurate assessment of seawater temperature trends across the reserves.

The Monkey Mia settlement has been identified as a point source for increased nutrients and pathogens entering nearshore waters as a result of surface runoff and ground water ingress (EPA 1989; Stoddart 1990). While improved wastewater treatments are now in place, recent sampling indicates that nearshore nutrient levels (nitrogen and phosphorous) are still relatively high. However, pathogen concentrations in the same samples were low, indicating that human faecal contamination from land-based sources was unlikely to be the cause. Further, rainfall around the 2016 sampling event was low, decreasing the possibility that faecal matter entered the nearshore waters through surface or ground water runoff. In the absence of this pressure source, it is possible that heightened nutrients resulted from an accumulation of decomposing wrack in nearshore waters or along the shoreline (Dugan *et al*. 2011). This may also explain why nutrient concentrations have frequently exceeded the ANZECC/ARMCANZ thresholds in this area without clear cause, with wrack accumulation and decomposition possibly being a natural process at this location. Consideration needs to be given to including appropriate control locations and wrack surveys into the monitoring program in order to fully understand the dynamics of water quality at this location.

3 Seagrass Communities (KPI)

3.1 Key Points

- A major decline in the extent of persistent seagrass occurred across Shark Bay between 2002 and 2014.
- This loss was primarily due to a decline in the extent of *Amphibolis antarctica*.
- While *Posidonia australis* does not appear to have been affected to the same extent, there has been localised loss recorded in some locations (e.g. Monkey Mia).
- Seagrass loss was most likely caused by a sustained period of anomalously high seawater temperature during the summer of 2010/2011, and the synergistic effects of river discharge in early 2011.
- There is currently insufficient data from the Wooramel Bank area to assess changes in seagrass shoot density and extent from this part of the marine park.
- While the assessments of *in-situ* indicators are generally considered to have low/medium levels of confidence, this will improve as the spatial and temporal resolution of monitoring improves.

3.2 Indicator Summary

	Trend	**Confidence**
Condition		
Areal extent	Declining	High
Community composition	Changing	Low
Posidonia australis shoot density	Stable	Low
Posidonia australis canopy height	Declining	Low
Pressure		
Seawater temperature	Increasing	High
River discharge	Stable	High

3.3 Condition Indicators

Condition indicators are measures of seagrass community ‘health’ relative to pressures acting on them. The indicators relevant to seagrass communities within the Shark Bay marine reserves and presented in this report are:

- Areal extent.
- Community composition.
- *Posidonia australis* shoot density.
- *Posidonia australis* canopy height.

Shark Bay contains some of the most diverse and extensive seagrass meadows in the world (Walker *et al.* 1988). Large meadows are composed predominantly of *Amphibolis antarctica* or *Posidonia australis*, both of which are temperate species occurring near the northern extent of their range in WA. Meadows often contain smaller proportions of colonizing species, such as *Halodule uninervis* and *Halophila ovalis*. Seagrass monitoring indicators were chosen to be consistent between the two major persistent seagrass species in Shark Bay, and to provide measures of the key structural elements of these meadows. Assessments of areal extent were conducted over large spatial areas in order to measure changes in seagrass distribution at the landscape scale. Community composition, measured as changes in percent canopy cover, is assessed over much smaller spatial scales and provides detailed information on the relative cover of the dominant species (*A. antarctica* and *P. australis*) within the overall Shark Bay seagrass community. Shoot density is used as an indicator of overall abundance and canopy height is used to characterise the structural complexity of the meadows (Duarte and Kirkman 2001). While this report only presents density and canopy height assessments of *P. australis*, initial assessments of *A. antarctica* have also been collected, such as stem density, number of clusters per stem, number of leaves per cluster and canopy height. These will be presented in future reports as the spatial and temporal resolution of sampling improves.

Areal extent was estimated by classifying Landsat (30 x 30 m pixel resolution) or Sentinel II (10 x 10 m pixel resolution) satellite imagery into different habitat categories (dense seagrass, >40% cover; sparse seagrass, <40% cover) based on pixel colour and extensive ground-truthing. Appropriate imagery that was not obscured by cloud cover was sourced in 2002, 2014 and 2016, with assessments calculated among these years. Currently, this method cannot easily distinguish between the major seagrass species, resulting in combined values for total seagrass extent of the persistent seagrasses. There are also issues with water depth and clarity which means that it is not always possible to accurately estimate seagrass extent across the whole of Shark Bay.

The survey design for in water assessments of seagrass communities is based around the distribution of the two main seagrass species, such as homogeneous meadows in shallow areas (2-7 m) as well as the primary pressure acting on them (thermal stress). Based on these factors, seagrass monitoring in the marine reserves is undertaken in four sectors: western gulf, Peron, Monkey Mia and eastern gulf. As thermal stress is spatially and temporally variable, the design encompasses sites spread across all four sectors, including sites where historical data exists (i.e. prior to the commencement of the standardised seagrass monitoring program in 2010). The incorporation of historical data, where possible, is important as it provides valuable information on the medium-term dynamics of the condition indicators, assisting with the interpretation of more recent data. Seagrass in deeper (>8 m) water is not currently represented in the survey design, and as such any findings should only be interpreted to represent shallow areas.

Broad-scale assessments of community composition were made at seven 30 x 30 m sites in the western gulf (n=4), Peron (n=1) and Monkey Mia (n=2). While these comprise a mix of *P. australis* and *A. antarctica*-dominated sites, *A. antarctica* occurred at all sites. At each site, 30 random benthic images are obtained using a remotely-operated drop camera. Images are obtained from approximately 1 m height and analysed using a point count method to measure canopy cover. More recently this was supplemented with *in situ* data collected at fourteen *P. australis* sites spread across all four sectors of the marine reserves: western gulf (n=4), Peron (n=3), Monkey Mia (n=2), and eastern gulf (n=2) (Figure 3.1). While information on *A. antarctica* has also been collected at nineteen sites spread across the four sectors, comparable information for *P. australis* is currently limited to a single time point and is hence not presented in this report. Measurements of canopy cover were conducted along three permanent 10 m transects at each site, with ten benthic photographs taken from approximately 1 m height at 1 m intervals and analysed using a point count method to obtain measurements of species-specific canopy cover.

Assessments of *P. australis* density and canopy height were made at the same sites, with data collected from eight replicate 20 x 20 cm quadrats at 1.5 m intervals along each of the three 10 m transects. In each quadrat, shoot density (the number of individual shoots) and canopy height are measured.

The Department's *in situ* seagrass monitoring program in the marine reserves began in 2010 and has been incrementally increased since. Monitoring sites were initially established in *P. australis*-dominated meadows and these have been surveyed in 2011, 2014 and 2016. These sites are located across Shark Bay but access to the Wooramel Bank sites is difficult and only limited time-series data exists for these sites. Several permanent *A. antarctica* sites were established in the eastern gulf in 2014 and additional sites for both species were established in the western gulf during 2016. Because of the limited history of surveying permanent seagrass sites in Shark Bay it is currently difficult to identify trends in condition for most *in situ* indicators and more data are needed before trends in condition can be interpreted with confidence.

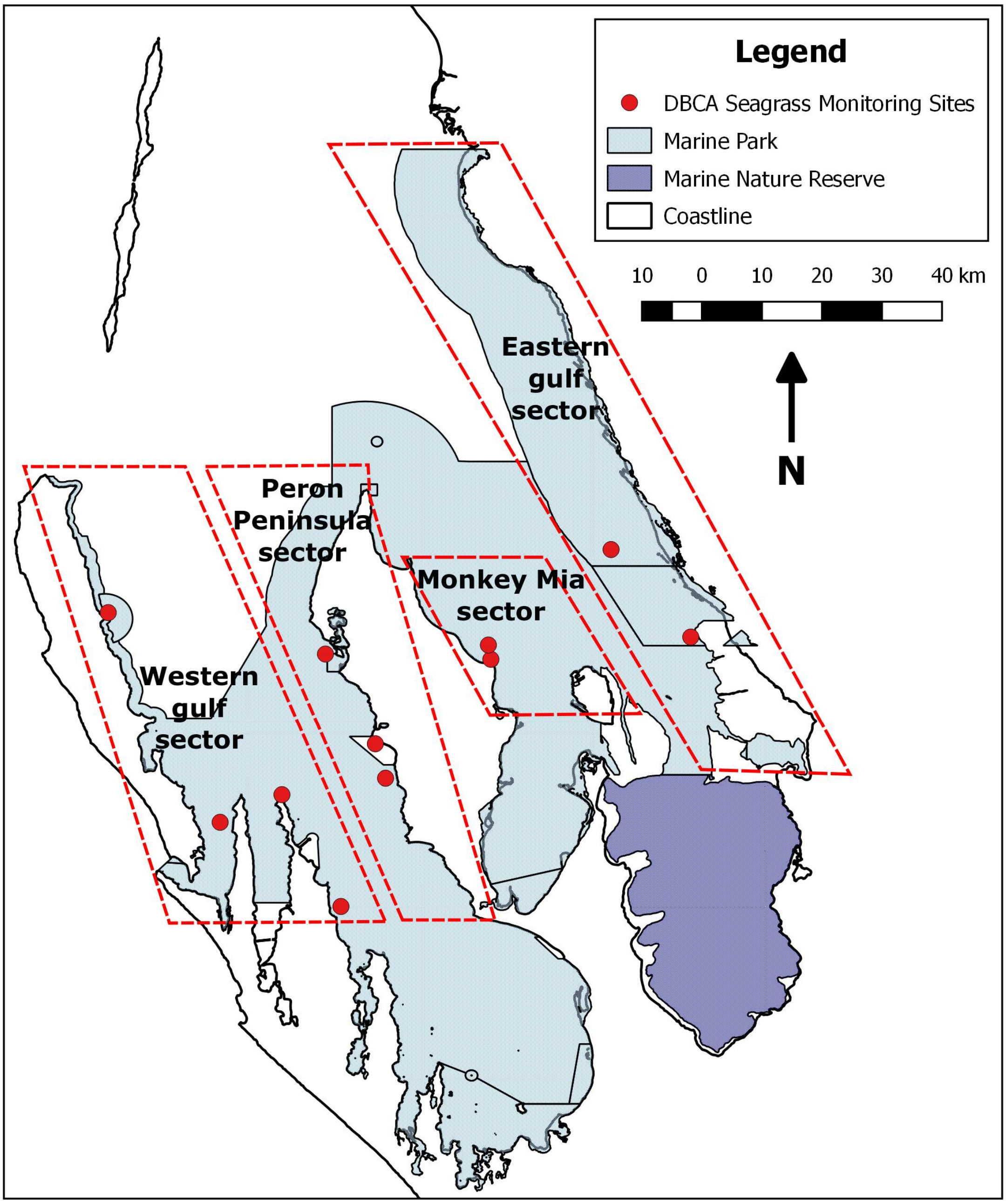

Figure 3.1: Location of DBCA *in situ* seagrass sites, along with the western gulf, Peron Peninsula, Monkey Mia and eastern gulf sectors used for monitoring in the Shark Bay marine reserves.

3.3.1 Areal extent

The Shark Bay marine reserves cover approximately 8900 km^2, of which 78% (approximately 7000 km^2) has been assessed to determine the extent of identifiable seagrass (>5% cover) in this report. The remaining 22% of the area has not been included because the analysis of satellite images was constrained by water depth, clarity or the presence of cloud cover. The areas that were excluded in any year have been removed from the temporal analyses, so that the same area has been assessed in each time period. Research is currently being conducted to correct issues in the areas not reported here, and to increase the temporal scale of the assessment.

Assessing the extent of seagrass using satellite imagery is appropriate as areas of dense seagrass in Shark Bay have little co-occurring reef/algal habitat that could potentially complicate interpretation.

In 2002, approximately 2700 km^2 of identifiable seagrass habitat was identified in the marine reserves. Subsequent mapping in 2014 revealed that the area of seagrass habitat had declined to approximately 2110 km^2; a reduction of approximately 600 km^2 (21.5% loss; Figure 3.2). Most of this loss occurred in the northern parts of the western gulf and Wooramel Bank (Figure 3.3), and additional areas of loss were identified across the northern part of Faure Island, Monkey Mia, the northern part of Hamelin Pool and along the coastal margin of Henri Freycinet Harbour.

From 2014 to 2016, the total area of identifiable seagrass had increased by 250 km^2 to approximately 2360 km^2 (Figure 3.2, Figure 3.4), although losses were still observed around the Wooramel Bank, Faure Sill and Faure Island (Figure 3.4).

In addition to the loss of total seagrass area, there was a major decline in the proportion of dense (>40% cover) seagrass between 2002 and 2016. In 2002, approximately 72% (1925 km^2) of the total area of seagrass comprised dense canopy (Figure 3.2), whereas in 2014 dense seagrass had declined to 58% of the total area. This declined even further to 55% of the total area in 2016, despite slight increases in the overall extent (Figure 3.2).

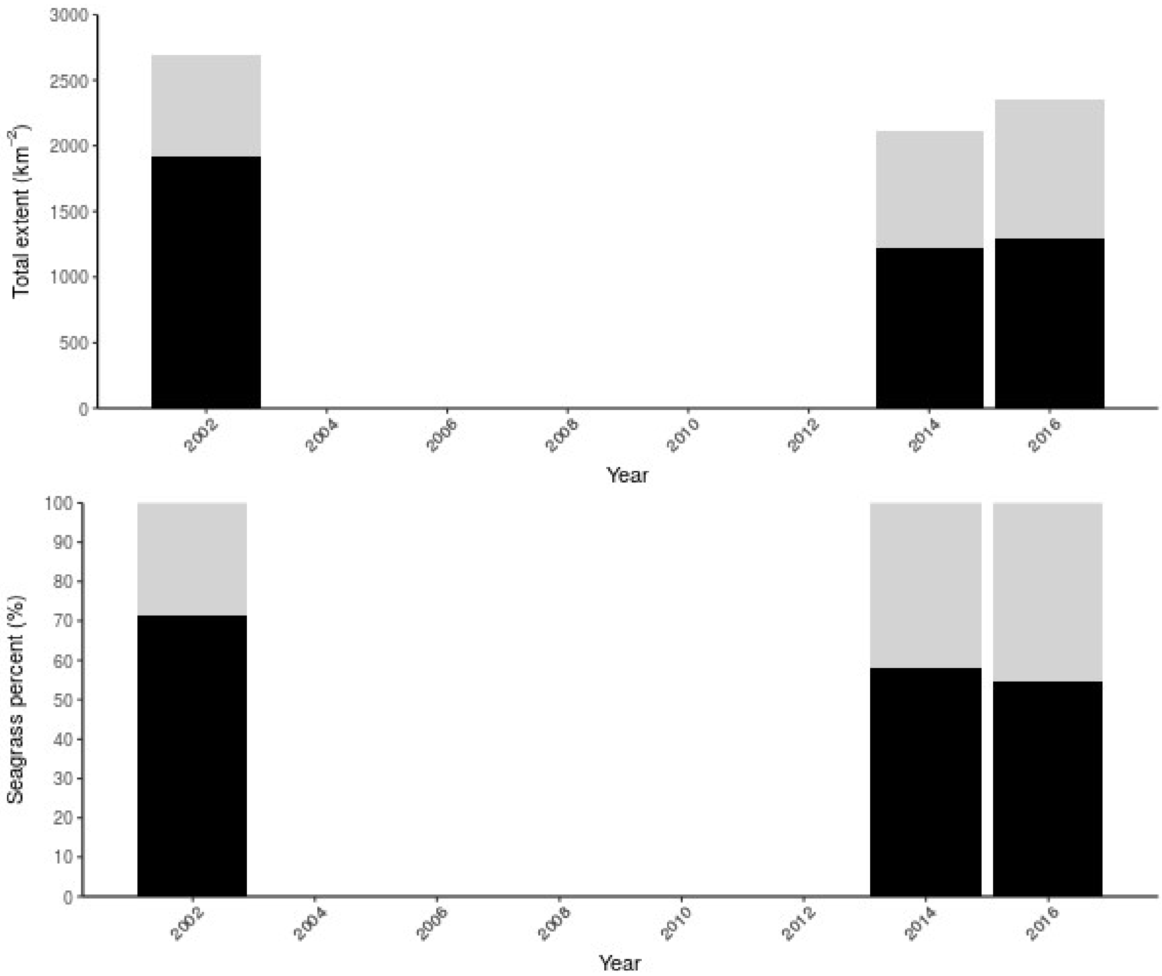

Figure 3.2: The extent of seagrass mapped in the Shark Bay marine reserves, 2002-2016. a) total extent (km^2) and b) the proportion of dense and sparse seagrass in each year of mapping. These values are derived from habitat maps of 78% of SBMP; the remaining 22% has not yet been adequately mapped. Black bars = dense (>40% cover) seagrass; grey bars = sparse (5-39% cover) seagrass.

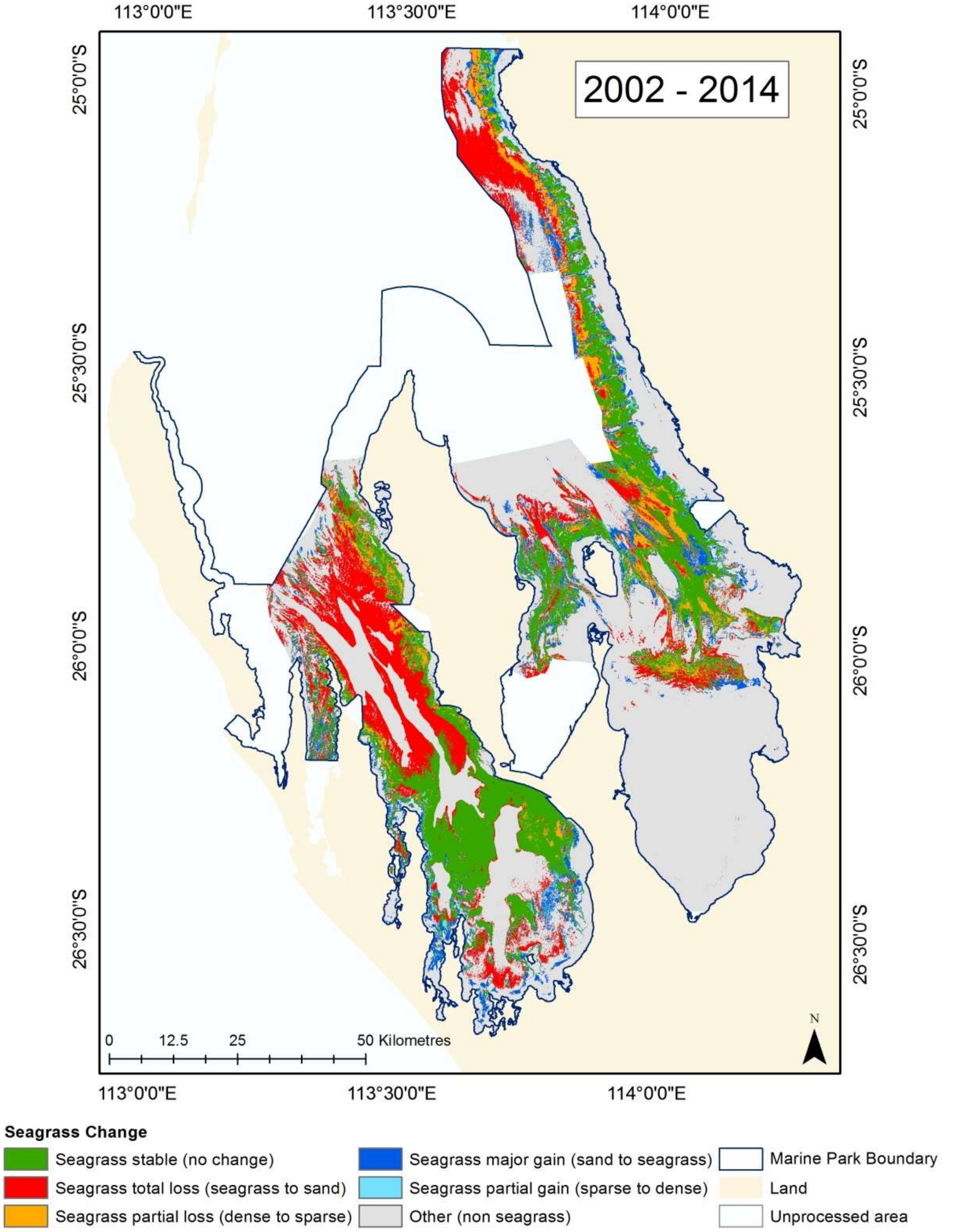

Figure 3.3: The loss and gain of seagrass extent within the Shark Bay marine reserves between 2002 and 2014. These values were derived from habitat maps of 78% of the marine reserves; the remaining 22% (i.e. white areas) has not yet been adequately mapped.

Source imagery: Landsat 7 TM and 8 OLI, USGS (glovis.usgs.gov/).

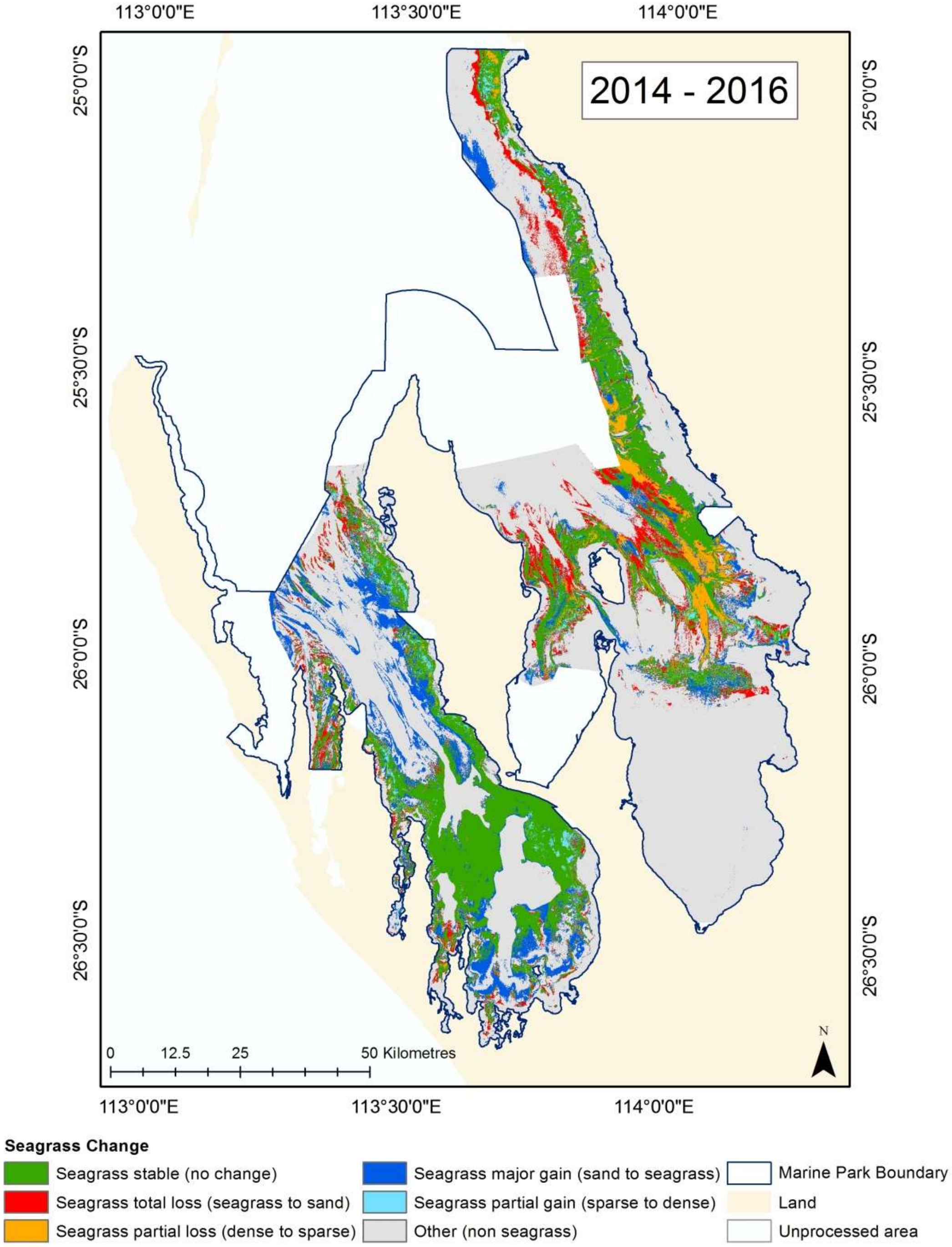

Figure 3.4: The loss and gain of seagrass extent in the Shark Bay marine reserves between 2014 and 2016. These values were derived from habitat maps of 78% of the marine reserves; the remaining 22% (i.e. white areas) has not yet been adequately mapped.

Source imagery: Landsat 7 TM and 8 OLI, USGS (glovis.usgs.gov/).

3.3.2 Community composition

Broad-scale community composition assessments indicate a major decline in the canopy cover of *A. antarctica* over the past 20 years, from 54.9% (±13.5 SE) in 1996 to 1.5% (±0.99 SE) in 2016 (Figure 3.5). Although no measures were available in 2013, there was a slight increase in the cover of *P. australis* over the same time period, from 22.7% to 45.2%, although measures were associated with high levels of variance.

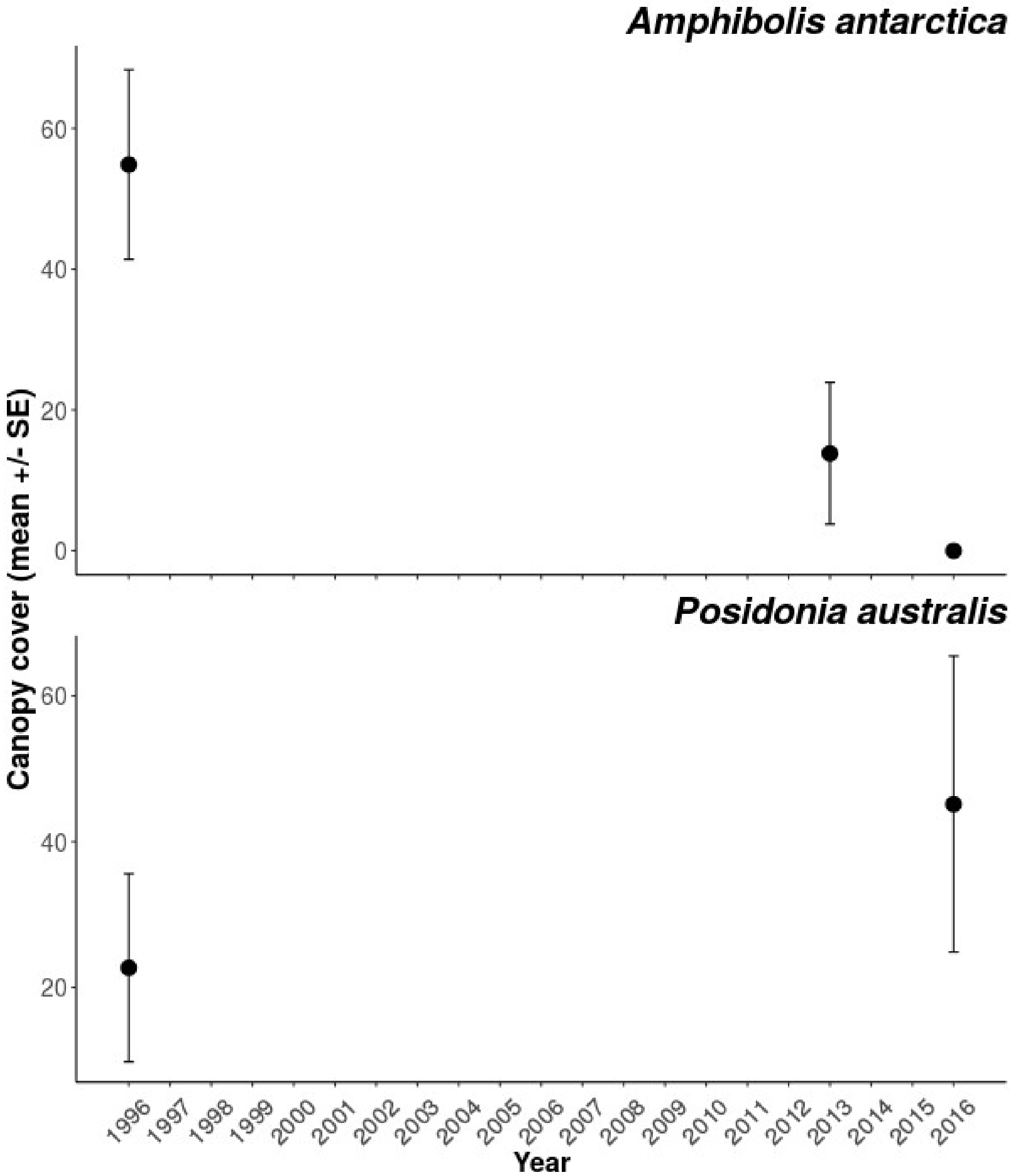

Figure 3.5: Mean (± SE) canopy cover of the dominant seagrass species, *Amphibolis antarctica* and *Posidonia australis* in the Shark Bay marine reserves (1996-2016), obtained using drop camera. Seven 30 x 30 m sites were surveyed across the western gulf (n = 4), Peron (n = 1) and Monkey Mia (n = 2) sectors.

At fixed monitoring sites, *in situ* assessments of *P. australis* indicated that canopy cover was stable in the eastern gulf (26-28%), Peron (34-42%) and Western gulf (19-23%) sectors from 2010/11 to 2014 (Figure 3.6). While no sampling occurred in the eastern gulf in 2016, there was an increase in canopy cover from 2014 to 2016 in the Peron (from 34.5% to 56.1%) and western gulf (from 19.8% to 42.2%) sectors. Assessments of the Monkey Mia sector are currently based on a single site, meaning that no temporal comparisons can be made at this time.

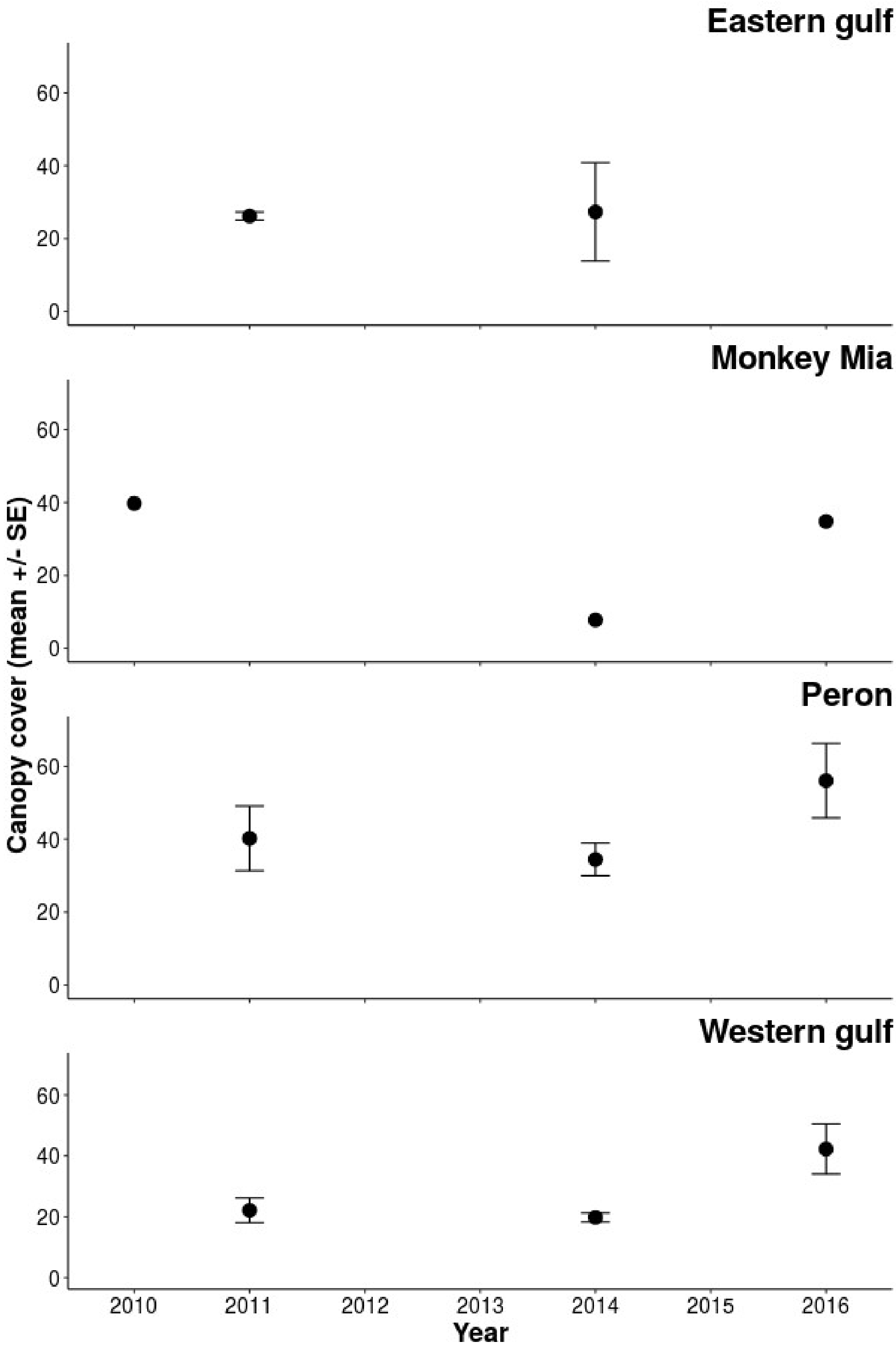

Figure 3.6: Mean (± SE) percent cover of *Posidonia australis* across the eastern gulf, Monkey Mia, Peron, and western gulf sectors of the Shark Bay marine reserves, 2010-2016. Assessments were made from thirty benthic photographs taken at each site approximately 1 m above the substrate. Eastern gulf n=2; Monkey Mia n=1; Peron n=3, western gulf n=3.

3.3.3 *Posidonia australis* shoot density

The overall mean density of *P. australis* was relatively stable at 175-200 shoots m^{-2} across the limited sampling period from 2011-2016 (Figure 3.7). Shoot density was highest in the eastern gulf (277-394 shoots m^{-2}), followed by the western gulf and Peron (168-227 shoots m^{-2} and 163-184 shoots m^{-2} respectively) and Monkey Mia (45-100 shoots m^{-2}) (Figure 3.8). While shoot density was relatively stable in the western gulf and Peron sectors between 2011 and 2016, there was an increase in shoot density in the eastern gulf sector between 2011 (approximately 277 shoots m^{-2}) and 2014 (approximately 394 shoots m^{-2} in 2014). However, there was a decline in shoot density in the Monkey Mia sector from 2010 (approximately 100 shoots m^{-2}) to 2014 and 2016 (approximately 52 shoots m^{-2} and approximately 46 shoots m^{-2} respectively). It should be noted that Monkey Mia was the only sector sampled prior to 2011.

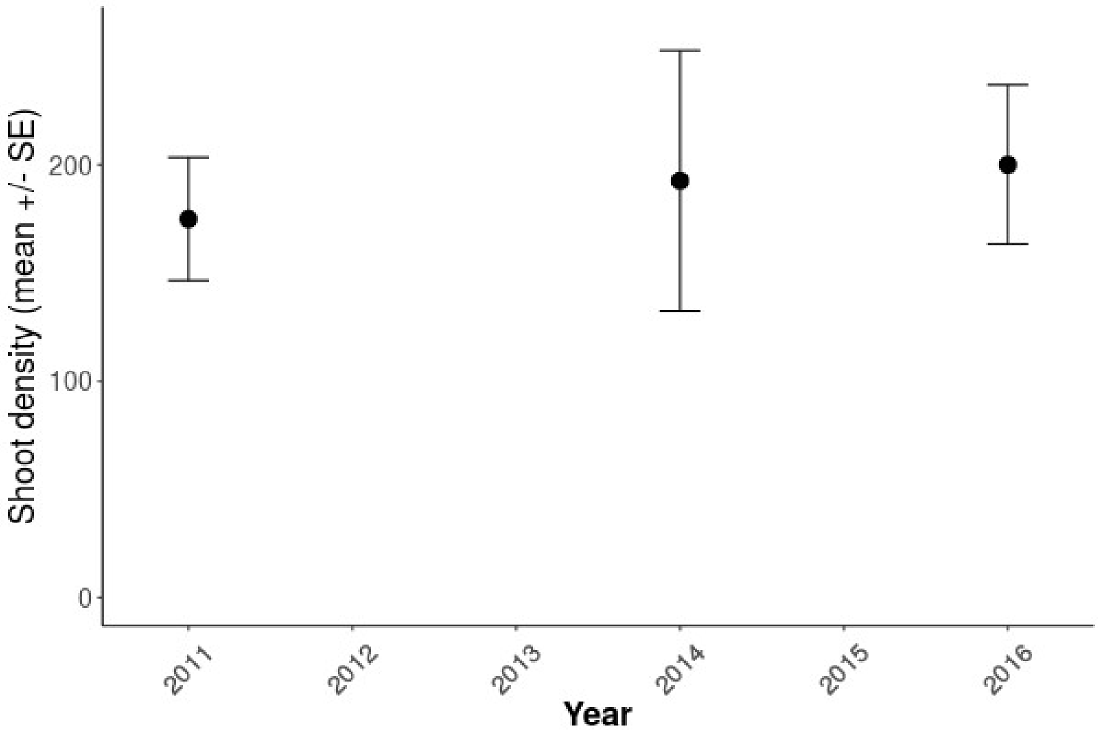

Figure 3.7: Mean (± SE) shoot density m^{-2} of *Posidonia australis* across all permanent sites sampled consistently across years in the Shark Bay marine reserves, 2011-2016. Data was collected from twenty-four 20 x 20 cm quadrats at each of seven sites spread across the Peron (n=3) and western gulf (n=4) sectors of the park.

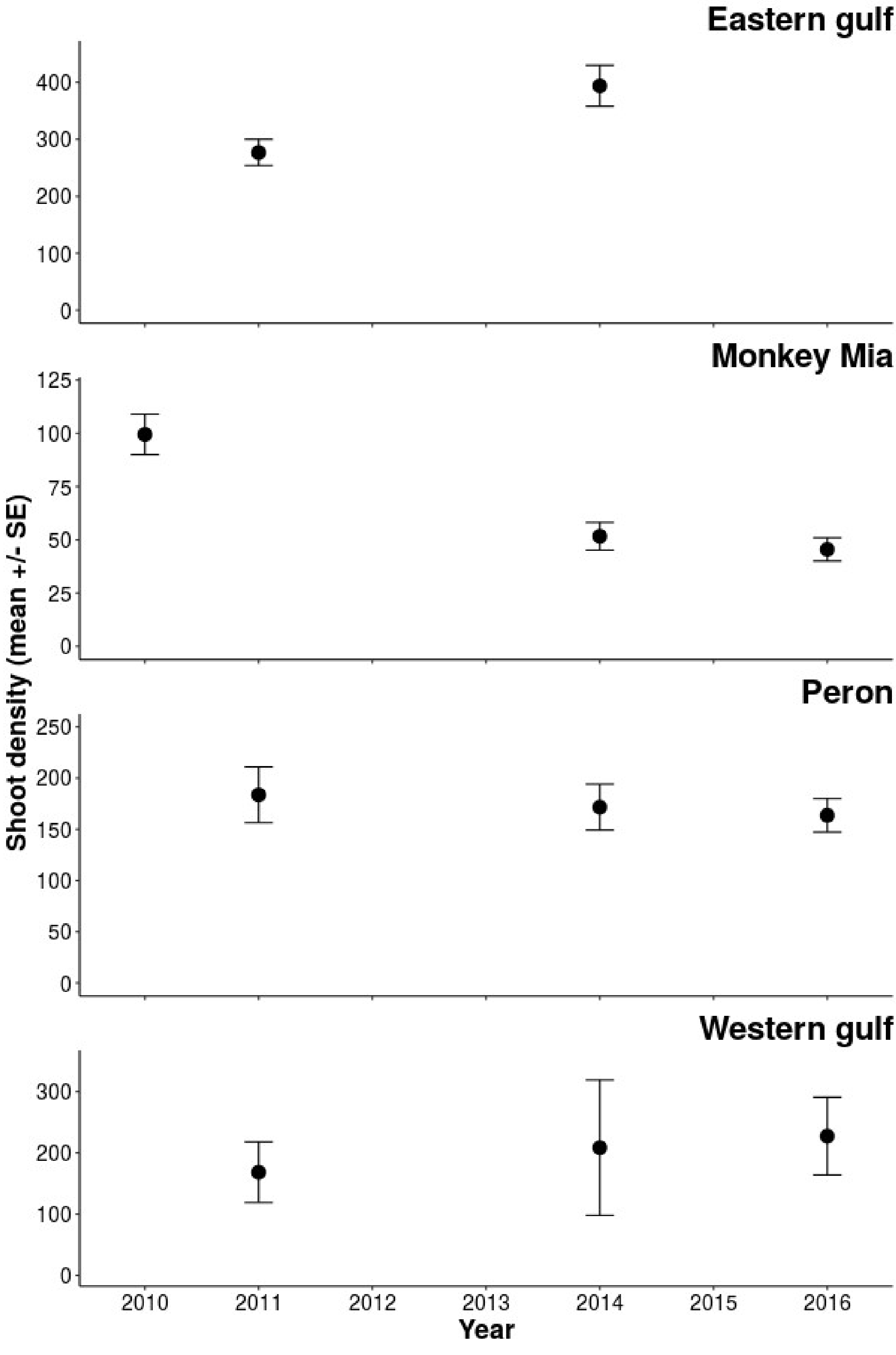

Figure 3.8: Mean (± SE) shoot density per m^{-2} of *Posidonia australis* across the eastern gulf, Monkey Mia, Peron, and western gulf sectors of the Shark Bay marine reserves, 2010-2016. Shoot density was collected from twenty-four 20 x 20 cm quadrats at each site; eastern gulf (n=2); Monkey Mia (n=2); Peron (n=3), and western gulf (n=4).

3.3.4 *Posidonia australis* canopy height

The mean maximum height of *P. australis* canopy across all consistently sampled sites within the marine reserves declined from approximately 362 mm in 2011 to approximately 237 mm in 2014 (Figure 3.9). However, it then increased slightly again to approximately 298 mm in 2016. The decline between 2011 and 2014 was largely a result of the pattern in the Peron sector, where mean maximum canopy height decreased from approximately 443 mm to approximately 233 mm (Figure 3.10). The Monkey Mia sector was not included in the overall reserve assessment as sites in this sector were surveyed in 2010, and not 2011. However, a major decline in mean maximum canopy height up to 2014 was also observed there, from approximately 382 mm in 2010 to approximately 200 mm in 2014. While canopy height increased again slightly in 2016 in the Peron sector (approximately 283 mm), it continued to decrease across the same period in the Monkey Mia sector across the same period (to approximately 155 mm in 2016), albeit with increased variability. Mean maximum canopy height in the eastern and western gulf sectors remained relatively stable across the monitoring period (Figure 3.10).

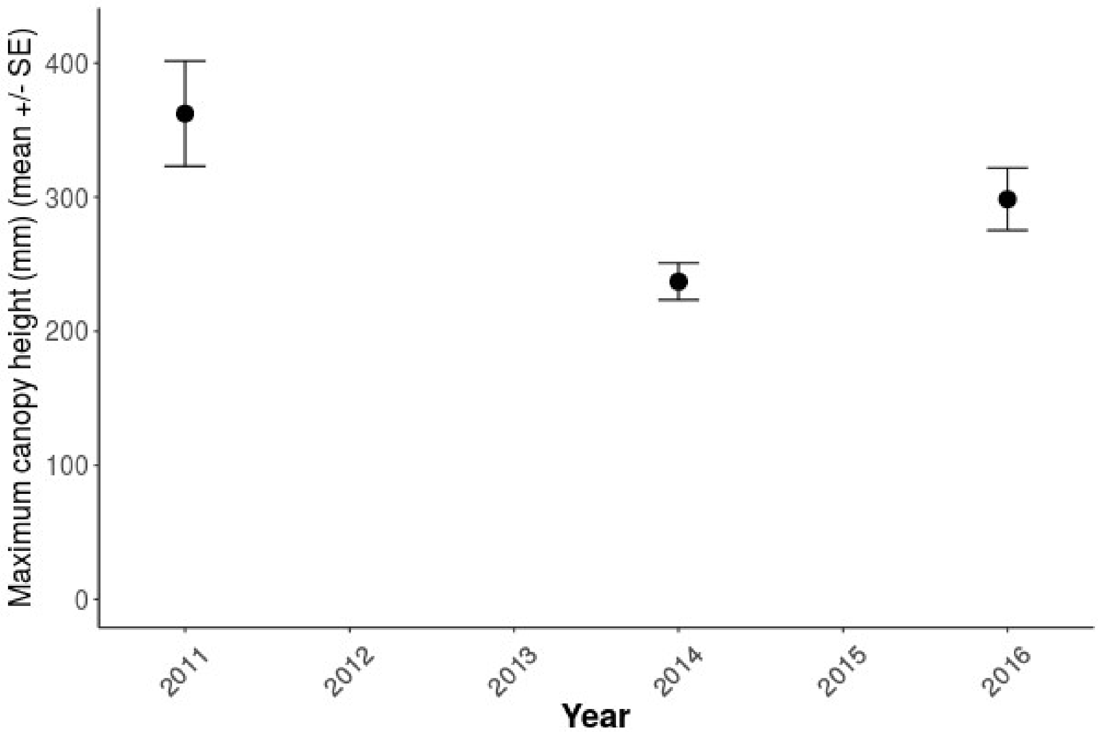

Figure 3.9: Mean (± SE) maximum canopy height m^{-2} of *Posidonia australis* across all permanent sites in the Shark Bay marine reserves, 2011-2016. Canopy height was collected from twenty-four 20 x 20cm quadrats at each site spread across the Peron (n=3) and western gulf (n=4) sectors of the park.

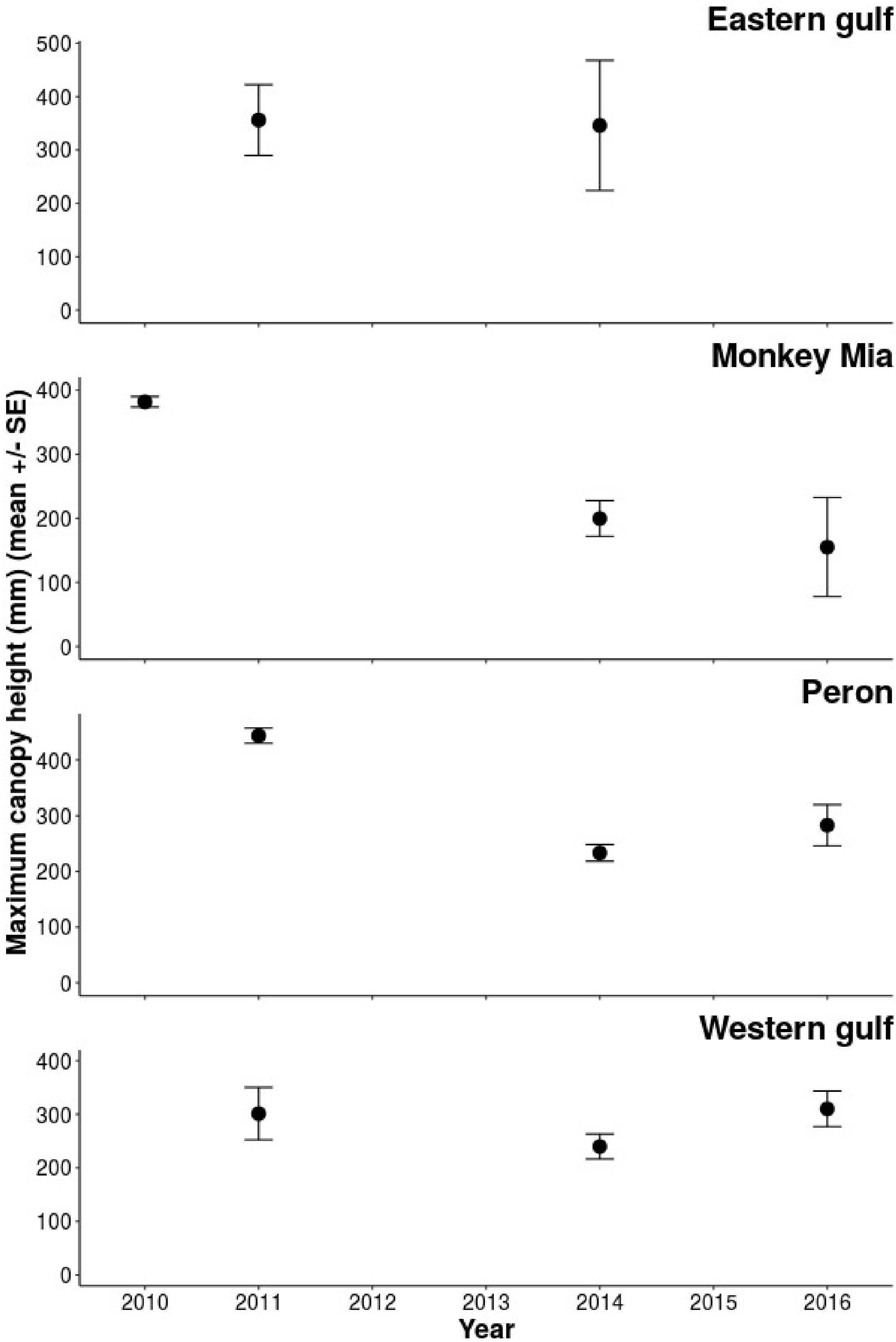

Figure 3.10: Mean (± SE) maximum canopy height m^{-2} of *Posidonia australis* across the eastern gulf, Monkey Mia, Peron, and western gulf sectors of the Shark Bay marine reserves, 2010-2016. Maximum canopy height was collected from twenty-four 20 x 20 cm quadrats at each site; eastern gulf (n=2); Monkey Mia (n=2); Peron (n=3), and western gulf (n=4).

3.4 Pressure Indicators

Pressure Indicators are considered to be reliable measures of the primary causes of change in seagrass communities within the marine reserves. Those indicators relevant to the Shark Bay marine reserves and presented in this report are:

- Seawater temperature.
- River discharge.

Water temperature is one of the key physical drivers of seagrass distribution and condition, with exceedance of thermal optima known to cause declines in seagrass condition (Garrabou *et al.* 2009; Jordà *et al*. 2012). For example, the widespread decline in Shark Bay seagrasses following the 2010/11 marine heatwave was likely due to an exceedance in thermal tolerance, as could be expected from a population occuring near the northern extent of its biogeographic range (McComb *et al*. 1981; Fraser *et al*. 2014). Riverine discharge is also known to impact seagrasses either due to acute changes in salinity (Seddon *et al*. 2000) and/or reductions in light availability (Longstaff *et al*. 1999, Waycott *et al*. 2009). Thus, when turbidity increases due to terrigenous sediments or excess nutrients in freshwater flows leading to algal blooms, seagrasses may be impacted. In Shark Bay, the simultaneous occurrence of increased water temperature and reduced light (from high turbidity associated with riverine discharge of the Wooramel River) correlated with loss of *A. antarctica* after the 2010/2011 marine heat wave (Fraser *et al*. 2014). Historically, freshwater discharge into the Shark Bay marine reserves is generally low, however, episodic flooding from the Gascoyne and Wooramel rivers does discharge into the eastern gulf sector. While direct measures of salinity and turbidity are not currently recorded in this area, annual discharge from these rivers is currently used as a measure of potential impact from these sources.

3.4.1 Seawater temperature

For a full assessment of seawater temperature within the marine reserves, please refer to page 7 of this report.

3.4.2 River Discharge

Annual discharge from the Wooramel and Gascoyne rivers have been highly variable across the monitoring period but exhibit no clear increasing or decreasing trend over time (Figure 3.11). While there is some variability between the two systems, overall patterns are similar as the catchments may be influenced by the same rainfall events. When examined across the seagrass condition monitoring period (1996-2016), exceptionally high discharge from both the Gascoyne and Wooramel rivers occurred following significant rainfall events in January and February of 2011. High levels of river discharge also occurred from the Wooramel River in 2000 and 2010 and the Gascoyne River in 2000, 2006 and 2015.

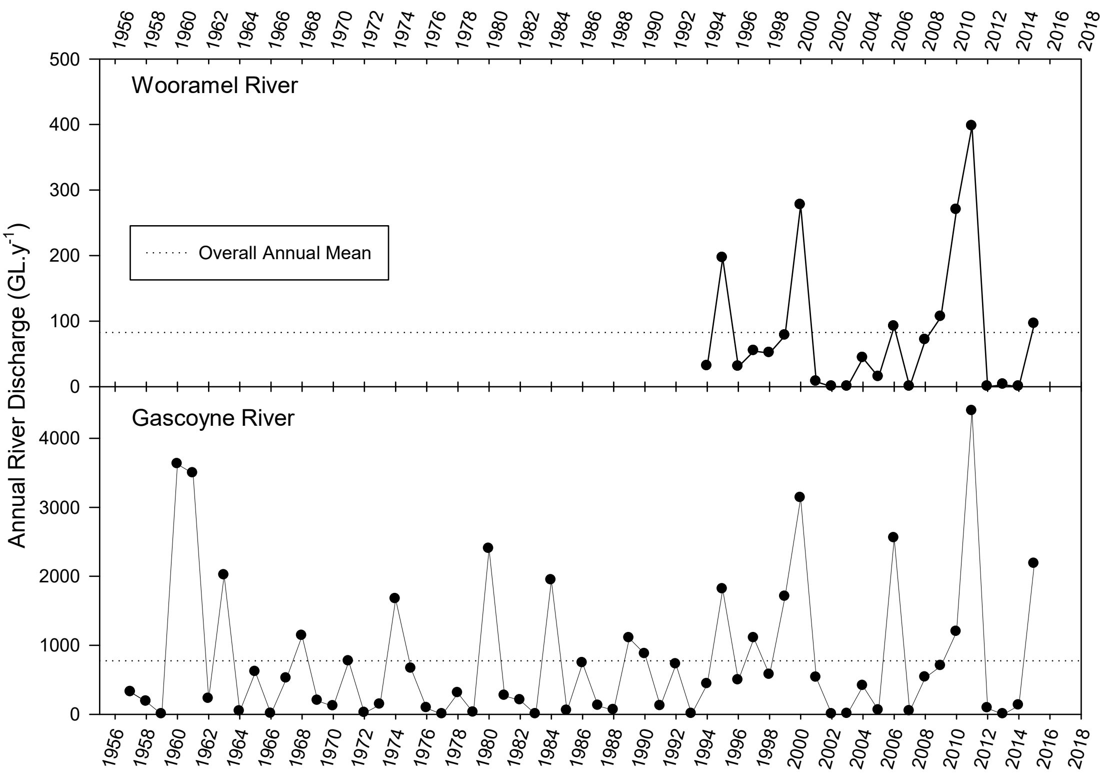

Figure 3.11: Total annual river flow (GL.y^{-1}) for the Wooramel (1994-2015) and Gascoyne (1957-2015) rivers which discharge into the eastern gulf of the Shark Bay marine reserves. Black dotted line indicates overall mean river flow over the whole sampling period.

Source: Department of Water (2017)

3.5 Synthesis

There was a dramatic decline in the area of persistent seagrass in the Shark Bay marine reserves between 2002 and 2014, representing a reduction of approximately 600 km^2 or 21.5% of the total measurable seagrass area. Although widespread throughout the reserves, this loss was most severe around the Wooramel Bank, Faure Sill and north-western gulf. In addition to the loss of seagrass, there was a major change in seagrass cover from dense (>40%) to sparse (<40%) across large areas of the bay. The area of dense seagrass, which comprised approximately 72% of the total area of seagrass in 2002, had declined to only 58% by 2014. While total areal extent increased slightly between 2014 and 2016 (an increase of approximately 250 km^2), the total area of seagrass remained well below the extent recorded in 2002. This increase mostly occurred in the ‘sparse’ seagrass density category and was generally attributed to gains in the western gulf, with losses continuing around the Wooramel Bank and Faure Sill across this period. The increased prevalence of sparse seagrass may represent some recovery of persistent seagrasses. Alternatively, the increase in sparse seagrass cover may represent the colonisation of bare habitats by different species which are known to respond following environmental disturbances (Anderson 1994; Preen *et al*. 1995). Flowers were observed at many of the *A. antarctica* sites surveyed in 2016 which suggests some potential for long-term recovery.

While detailed data are currently limited, it appears that these changes were mostly due to the decline of *Amphibolis antarctica*, likely caused by extremely high seawater temperature during the 2010/11 summer. Successive years of anomalously high seawater temperature in 2011/12 and 2012/13, and prolonged discharge events from the Gascoyne and Wooramel rivers into the eastern gulf in early 2011 were also likely contributing factors. Evidence suggests that the other dominant seagrass, *Posidonia australis*, was less impacted. However, there were noted declines in the condition of *P. australis* in the Monkey Mia sector between 2010 and 2014. However, the Monkey Mia sector was the only location where more detailed *in-situ* information was collected prior to the extreme seawater temperature event in 2010/11. While the limited drop camera data suggests that the canopy cover of *P. australis* communities may have increased in general across the past two decades, *in-situ* data presented here at the reserve-scale and for the Peron, western gulf and eastern gulf sectors individually is post-disturbance and may not accurately reflect the shorter-term effects (i.e. 1-5 years) of the warm water event on *P. australis* communities.

The mapping work presented here confirms the losses of seagrass in the Shark Bay marine reserves reported in other sources. Fraser *et al*. (2014) reported widespread defoliation of *A. antarctica* within 15 km of the mouth of the Wooramel River as a result of the freshwater plume and increased turbidity following the floods in 2010/11. While these authors observed some recovery in leaf biomass two years after these events, the below-ground biomass continued to decline, potentially leading to reduced resilience in *A. antarctica* meadows to future disturbances in Shark Bay. Thomson *et al.* (2015) reported ‘catastrophic’ declines (>90%) in *A. antarctica* cover areas around the shallow waters of the Peron Peninsula following the 2010/11 marine heat wave. The subsequent failure of sexual reproductive effort (e.g. seeds aborted) of *P. australis* meadows were recorded following 2011 (Thomson *et al*. 2015).

While *A. antarctica* has the potential to recover above-ground biomass reasonably rapidly, this requires a source of carbohydrates stored in healthy below-ground rhizomes (Walker *et al*. 2006; Fraser *et al*. 2014). Although some recovery in the total area of seagrass was recorded in 2016, the community composition of this is currently largely unknown. Recent observations have confirmed an increase in the presence of more tropical species, such as *Halodule uninervis,* colonising the sediment where *A. antarctica* was lost (Nowicki *et al*. 2017).

Amphibolis antarctica is a large, temperate species which in some parts of Shark Bay can be greater than 2 m tall (M. Rule, *pers. obs*). This seagrass forms a dense canopy which provides habitat to numerous other species and is a significant part of the Shark Bay ecosystem (Walker *et al*. 1988). While we have found a moderate recovery in the total area of seagrass, this is unlikely to represent a recovery of ecosystem function if *A. antarctica* has been replaced by other species. For example, *A antarctica* exceeds the size, standing stock, and productivity of *H. uninervis* by several orders of magnitude (Walker 1985). In addition, *H. uninervis* has a short, and simple canopy structure and is unlikely to contribute to sediment accumulation (Fonseca and Fisher 1986), or carbon storage (Fourqurean *et al*. 2012) to the same extent as *A. antarctica.* The long-term consequences and flow-on effects of these losses are yet to be realised.

The results presented here are largely interpreted from remote sensing outputs, which provide landscape level information (i.e. meadow extent) and cannot provide information regarding fine-scale changes in community composition or structure. The more detailed *in situ* monitoring data has been collected quite sporadically since 2010, due to the intense time and financial constraints associated with sampling in a remote and spatially large area. As such, the *in-situ* sampling effort to date has focused primarily on *P. australis*, in order to remain consistent with the seagrass monitoring program in other WA marine reserves. Overall, the limited amount of *in situ* data points mean that definitive trends in finer-scale seagrass condition are not yet clear but will improve as the spatial and temporal resolution of monitoring improves for both persistent seagrass species in Shark Bay.

4 Mangroves and Saltmarshes (KPI)

4.1 Key Points

- There was a major decline in the areal extent of mangroves in the Carnarvon coast region (loss of ~135 ha) between 2010 and 2015. While the specific cause is not yet known, it may be a combination of 2011 flood plume effects from the Gascoyne River, increasing porewater salinity resulting from natural oscillations in sea level and below average local rainfall and the effects of cyclone Olwyn in 2015.
- The areal extent of mangroves increased in the Peron Peninsula and west coast regions up to 2013, but then declined dramatically to 2015. The cause is likely a combination of increasing porewater salinity resulting from natural oscillations in sea level and below average local rainfall, and the passing of cyclone Olwyn in 2015.
- The current assessments of mangrove community condition are made with medium levels of confidence due to the limited length of time over which monitoring has occurred (eight years) and the number of survey periods (five). The confidence level in these assessments will increase as more data is added over time.

4.2 Indicator Summary

	Trend	Confidence
Condition		
Areal extent	Declining	Medium
Canopy density	Stable	Medium
Pressure		
Atmospheric temperature	Increasing	High
Groundwater availability	Stable	Medium
Sea level	Stable	Low
River discharge	Stable	High
Cyclones	Stable	Low

4.3 Condition Indicators

Condition indicators are measures of mangrove community ‘health’ relative to pressures acting in the marine reserves. Those indicators relevant to mangrove communities within the Shark Bay marine reserves and presented in this report are:

- Areal extent.
- Canopy density.

Areal extent quantifies the total area covered by mangrove communities and is responsive to longer term (e.g. prolonged drought) or more intense environmental changes or pressures (e.g. anomalous temperatures, physical removal). Canopy density is more sensitive to change and is likely to identify sub-lethal effects or early changes in overall community condition that may lead to changes in spatial extent over longer time-frames (English *et al*. 1997; Lovelock *et al.* 2017). Measures of species diversity/composition are not considered here as mangroves in the Shark Bay marine reserves comprise only a single species species (*Avicennia marina*).

Mangrove monitoring in the Shark Bay marine reserves is based on sites located in Carnarvon coast, Peron Peninsula and west coast regions that have been identified on the basis of different environmental characteristics and pressures that are likely to influence condition (Figure 4.1). Condition indicators are assessed for major mangroves in these regions only and these regions are indicative of mangroves across the entire reserve area.

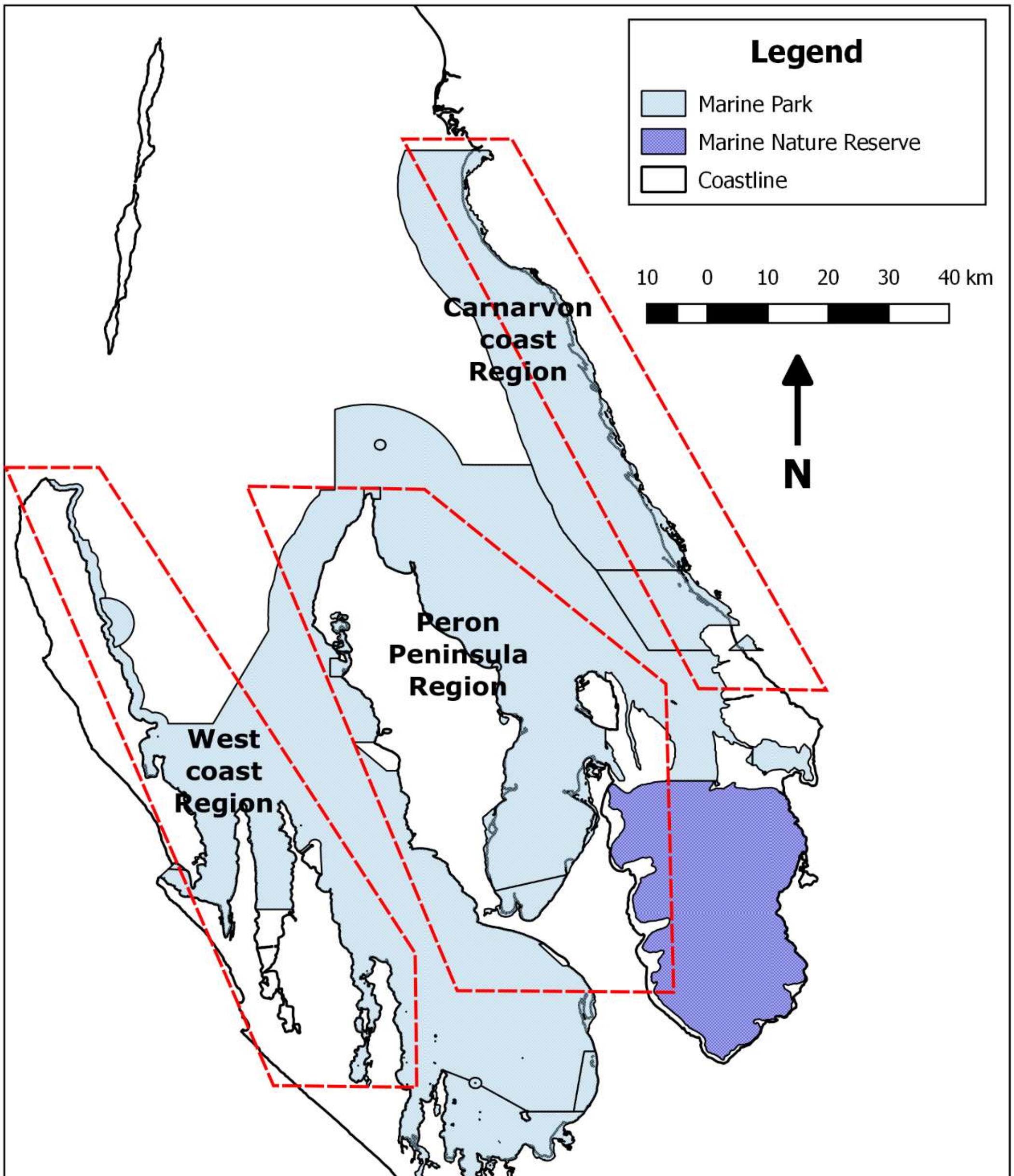

Figure 4.1: Location of the Carnarvon coast, Peron Peninsula and west coast regions used for mangrove monitoring in the Shark Bay marine reserves.

4.3.1 Areal Extent

The areal extent of mangroves was calculated using spectral response of mangrove foliage from ALSO AVINIR-2 and SPOT 6 satellite imagery (10-25 m pixel footprints) and aerial imagery (0.5 m pixel footprints) verified by field surveys. The most significant extent of mangrove in the marine reserves is in the Carnarvon coast region (approximately 718 ha in monitoring sites in 2015), followed by the Peron Peninsula (approximately 96 ha extent in monitoring sites in 2015) and west coast regions (approximately 16 ha extent in monitoring sites in 2015).

The areal extent of mangroves declined in all three regions in the period leading up to 2015 (Figure 4.2). Proportional loss was greatest in the west coast and Peron Peninsula regions, with 34% (approximately 9 ha total area) and 30% (approximately 39 ha total area) of mangrove area being lost between 2013 and 2015, respectively. While proportional losses in the Carnarvon coast region were lower at 16%, this represented the greatest loss of total extent in the marine reserves, with approximately 135 ha being lost between 2010 and 2015. While no monitoring data was available in this region between these years, visual examination of available Google Earth imagery suggests that this decline may have commenced around 2012/2013. Prior to these declines, areal extent was either relatively stable (Carnarvon coast and west coast) or slightly increasing (Peron Peninsula) across all regions, despite relatively high rates of both loss and gain in different parts of the mangrove communities (Figure 4.3).

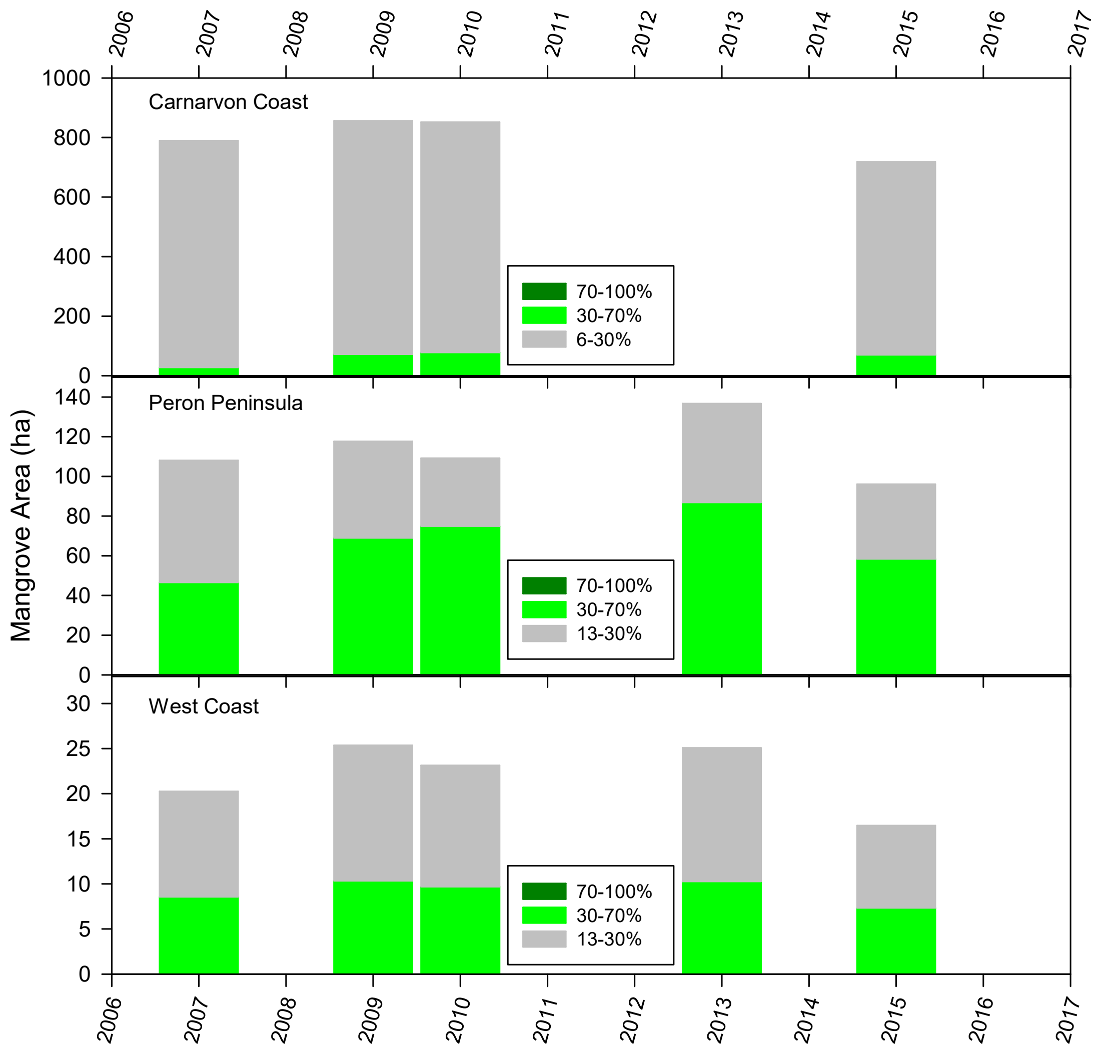

Figure 4.2: The areal extent of mangroves (ha) in the Carnarvon coast, Peron Peninsula and west coast regions of the Shark Bay marine reserves from 2007 to 2015, determined from remote sensing imagery and incorporating the projected foliage cover (PFC). PFC categories: dense (70-100% foliage cover); medium (30-70% foliage cover); and sparse (<30% foliage cover). Note that the lower limit for detected Sparse PFC can differ between locations depending on local variability. At this location, PFC less than 6-13% is considered too variable to be used in this analysis. The Carnarvon coast region was not assessed in 2013.

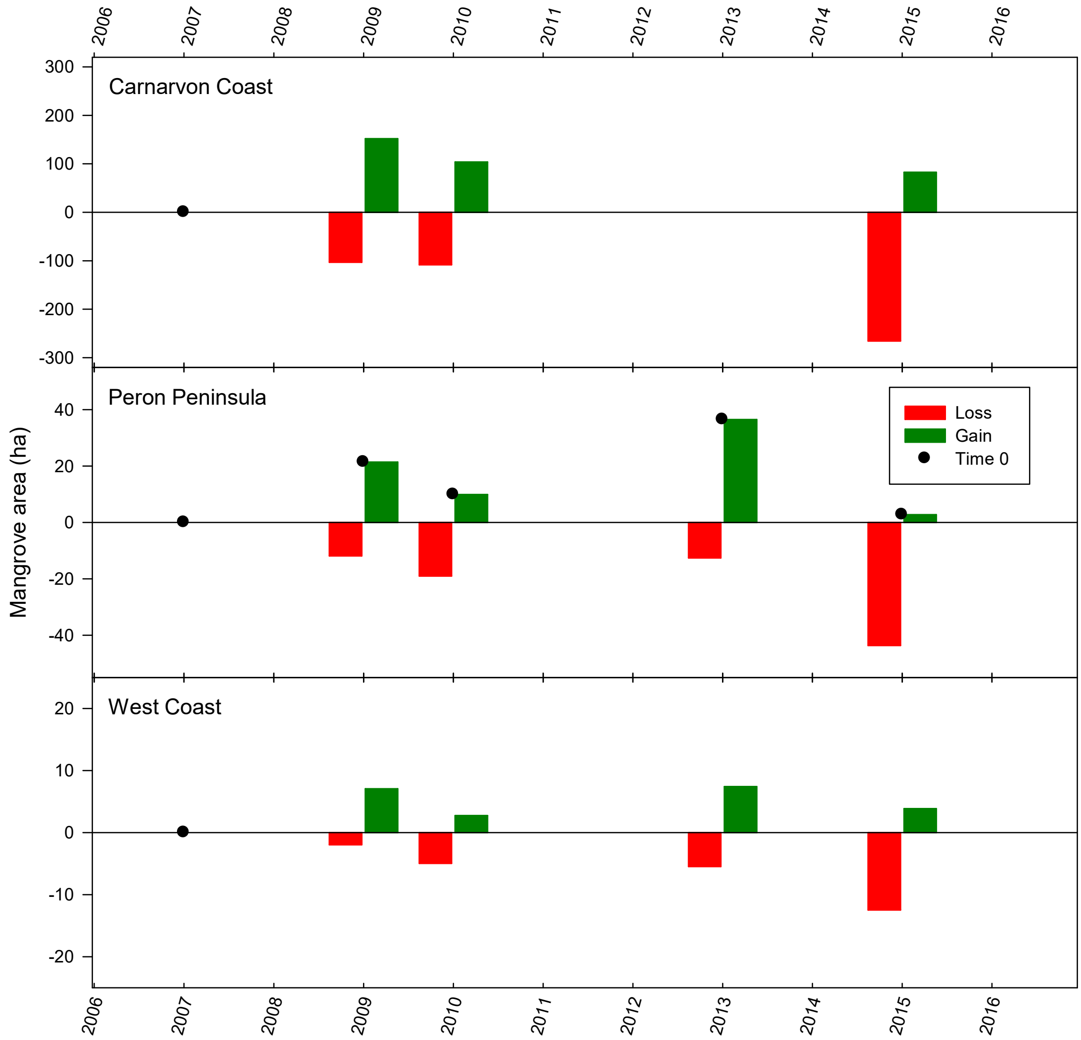

Figure 4.3: Loss and gain in areal extent of mangroves (ha) at the Carnarvon coast, Peron Peninsula and west coast regions of the Shark Bay marine reserves from 2007 to 2015, determined from remote sensing imagery. Loss and gain for each survey period is measured relative to spatial extent at the previous survey period, with 2007 being considered the baseline. Note that the Carnarvon coast region was not assessed in 2013.

4.3.2 Canopy density

Remotely sensed imagery (ALOS AVINIR-2 and SPOT 6) was coupled with field measurements of canopy cover to assess projected foliage cover (PFC) at sites in all sectors. PFC is categorised as dense (70-100% foliage cover), medium (30-70% foliage cover) or sparse (<30% foliage cover). The lower limit for detected sparse PFC can differ between locations depending on local variability. PFCs below 6% in the Carnarvon coast region and 13% at the Peron Peninsula and west coast regions were considered too variable to be used in this analysis.

Mangrove communities across the Shark Bay marine reserve are generally sparse in nature, with no dense PFC detected in any region across the monitoring period (Figure 4.2). Mangroves in the Carnarvon coast region mostly consist of the sparse PFC category (90-96% of total extent), with declines in extent between 2010 and 2015 largely due to losses in this category. The Peron Peninsula and west coast regions have higher proportions of the medium PFC category (contributing 42-67% and 40-47% of the total extents respectively) with changes in extent across the monitoring period more evenly distributed between the sparse and medium PFC categories.

4.4 Pressure Indicators

Pressure Indicators are considered to be reliable measures of the primary causes of change in coral communities within the marine reserves. Those indicators relevant to the Shark Bay marine reserves and presented in this report are:

- atmospheric temperature.
- groundwater availability.
- sea level.
- river discharge.
- cyclones.

Atmospheric temperature is considered to be among the most important structuring factors for mangroves on a global scale (Saenger 2002; Osland *et al.* 2017), and is currently used as the primary temperature indicator in this report. Porewater salinity is also considered a major structuring factor, creating conditions that are more or less favourable for plant recruitment, growth and survival (Smith and Duke 1987; Eslami-Andargoli *et al.* 2009). There is currently no direct measure of porewater salinity incorporated into the Department's monitoring program. However, the primary drivers of porewater salinity within the Shark Bay marine reserves are likely to be sea level (Lovelock *et al*., 2017) and groundwater availability (Smith and Duke 1987; Eslami-Andargoli *et al.* 2009). Sea level is assessed through a Bureau of Meteorology logging station located at Carnarvon (BOM 2017). Local rainfall within each of the regions is currently used as a surrogate for groundwater availability, given that this is likely to be the source of effective freshwater input at this location. Sedimentation above the normal levels can result in suffocation of trees through the burial of pneumatophores and prop roots essential for respiration (Ellison 1999). The primary source of excess sediments into mangrove communities within the reserves is through discharge from the Gascoyne and Wooramel River's or storm related shoreline changes during episodic events. While sediment levels within the mangroves are not directly measured, river discharge is currently being used as a surrogate for potential sedimentation for communities in the Carnarvon coast region (preliminary examination of satellite imagery indicates that this is the only area potentially affected by sediments in river discharge). Cyclones can have detrimental effects on mangrove communities through wind damage and sediment erosion, causing defoliation, physical damage or uprooting (Paling *et al*. 2008; Gilman *et al*. 2008).

4.4.1 Atmospheric temperature

Atmospheric temperature recorded at Carnarvon and Denham are being used as representative in the Carnarvon coast and, Peron Peninsula/west coast regions, respectively. While there are likely to be localised fluctuations between the Denham weather station and the west coast region, this is considered to be the most suitable surrogate of long-term temperature trends.

The long-term data at both Carnarvon (1945-2016) and Denham (1988-2016) highlights an increasing trend in daily maximum air temperature (seasonally adjusted Mann-Kendall; Carnarvon tau = 0.0766, 2-sided $p<0.001$; Denham tau = 0.0533, 2-sided $p<0.001$) (Figure 4.4). This has amounted to an increase of approximately 0.8°C over 71 years at Carnarvon, and approximately 0.2°C over 28 years at Denham.

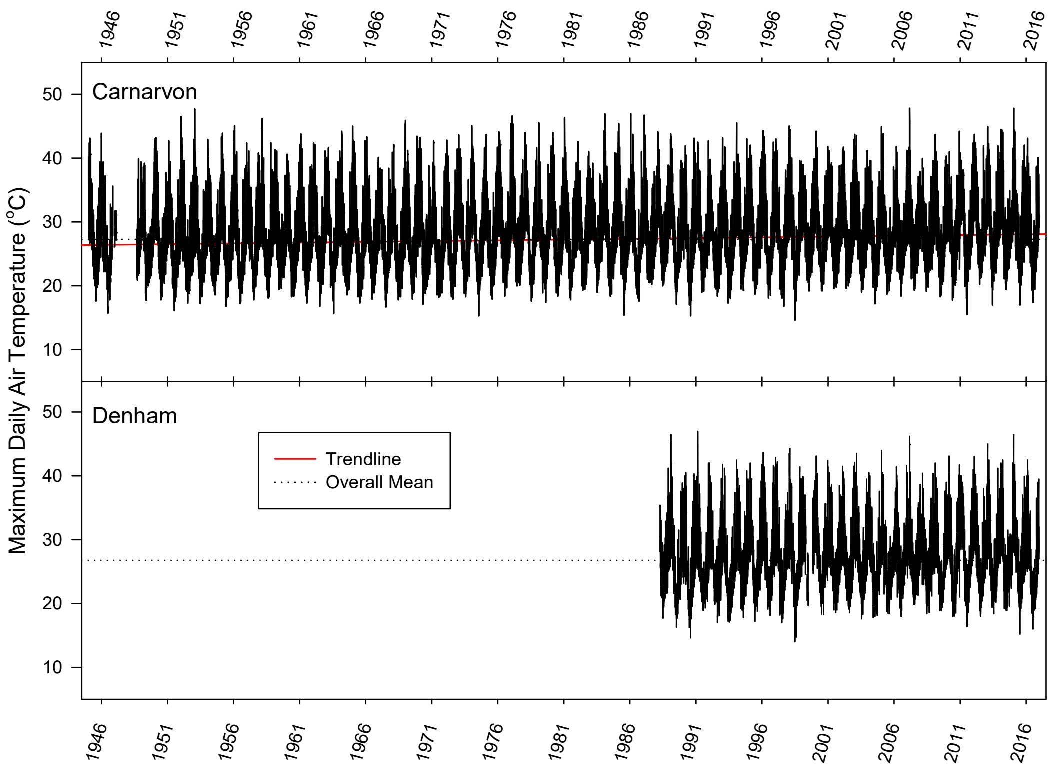

Figure 4.4: Daily maximum atmospheric temperature (°C) recorded at Carnarvon (1945-2016) and Denham (1988-2016), representing the Carnarvon coast and Peron Peninsula/west coast regions of the Shark Bay marine reserves, respectively. Dotted line indicates overall mean. Red lines indicate significant trends based on seasonally adjusted Mann-Kendall trend analyses (Carnarvon tau = 0.0766, 2-sided $p<0.001$; Denham tau = 0.0533, 2-sided $p<0.001$).

Source: Bureau of Meteorology (2017).

Since July 2011, mean maximum monthly atmospheric temperatures have frequently exceeded the long-term mean across all months at both Carnarvon and Denham locations (Figure 4.5). Atmospheric temperatures have been particularly high at the Carnarvon location. Above average temperatures were most consistent during the spring and summer periods (August-February), with all years recording at least one anomalously high mean temperature (more than two standard deviations above the long-term mean) during this period. Temperatures were also anomalously high during the spring (March-June), although this was less consistent between years.

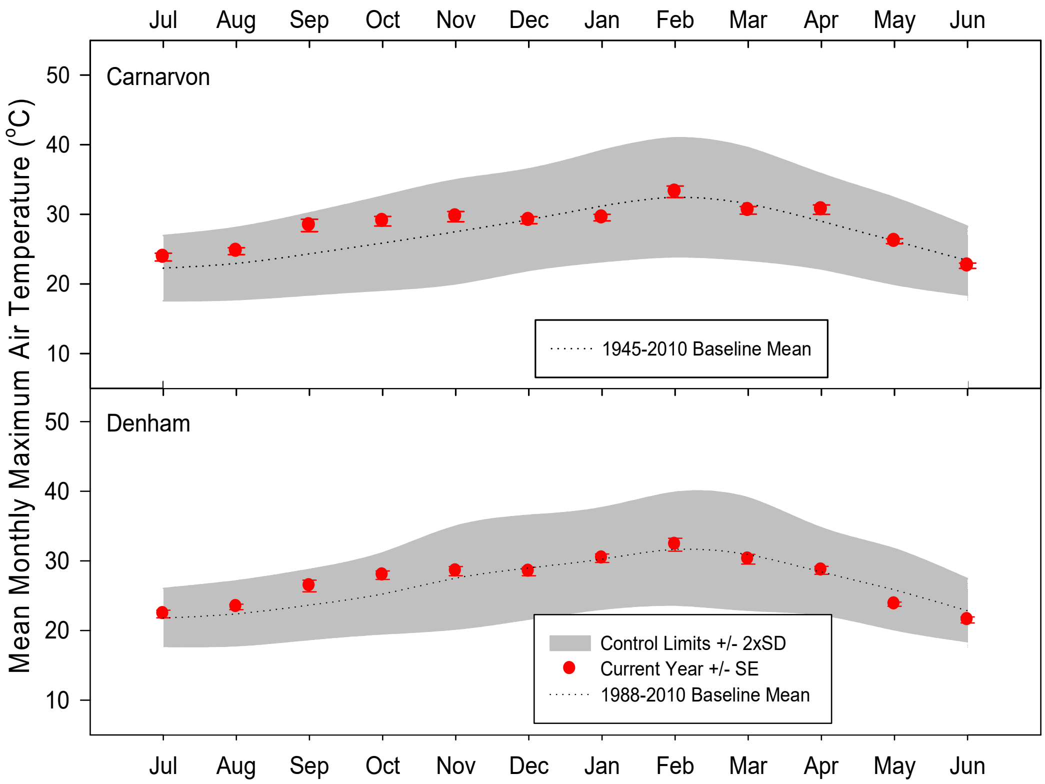

Figure 4.5: Mean monthly maximum atmospheric temperature from July 2011 to June 2016 recorded at Carnarvon and Denham, representing the Carnarvon coast and Peron Peninsula/west coast regions of the Shark Bay marine reserves, respectively. Dashed lines indicate baseline means (Carnarvon 1945-2010 and Denham 1988-2010). Grey shading indicates two standard deviations from the long term mean. Mean values beyond two standard deviations are considered 'anomalous.' Monthly means are based on daily maximum air temperature measurements.

Source: Bureau of Meteorology (2017).

4.4.2 Groundwater availability

Rainfall recorded at Carnarvon, Denham and Useless Loop is being used as representative indicators of surface runoff and ground water availability for the Carnarvon coast, Peron Peninsula and west coast regions, respectively. While it is not a direct measure of ground water availability and there can be significant variation in rainfall throughout the Shark Bay region, reliable rainfall data only exists at BoM weather stations at these locations and are the best available surrogates for above and below ground water availability.

Annual rainfall at all three regions has been variable across the monitoring period but exhibit no clear increasing or decreasing trends over time (Figure 4.6). There is also considerable variation between regions, with patterns in annual rainfall rarely being consistent across the different areas. When examined across the mangrove condition monitoring period (2007-2015), the Carnarvon coast received relatively high levels of rainfall in 2008, 2010 and 2011, followed by five consecutive years of below average rainfall. The Peron Peninsula received relatively high levels of rainfall in 2008, 2011 and 2015, while the west coast received relatively high levels of rainfall in 2008 and 2015. This indicates that the potential for surface runoff and groundwater recharge was likely to differ between regions across this period. However, groundwater availability is likely to have been relatively low in the Carnarvon coast since 2012. Subsequent seasonally adjusted Mann Kendall trend analyses indicated that there were no discernible trends in annual rainfall collected at the three stations.

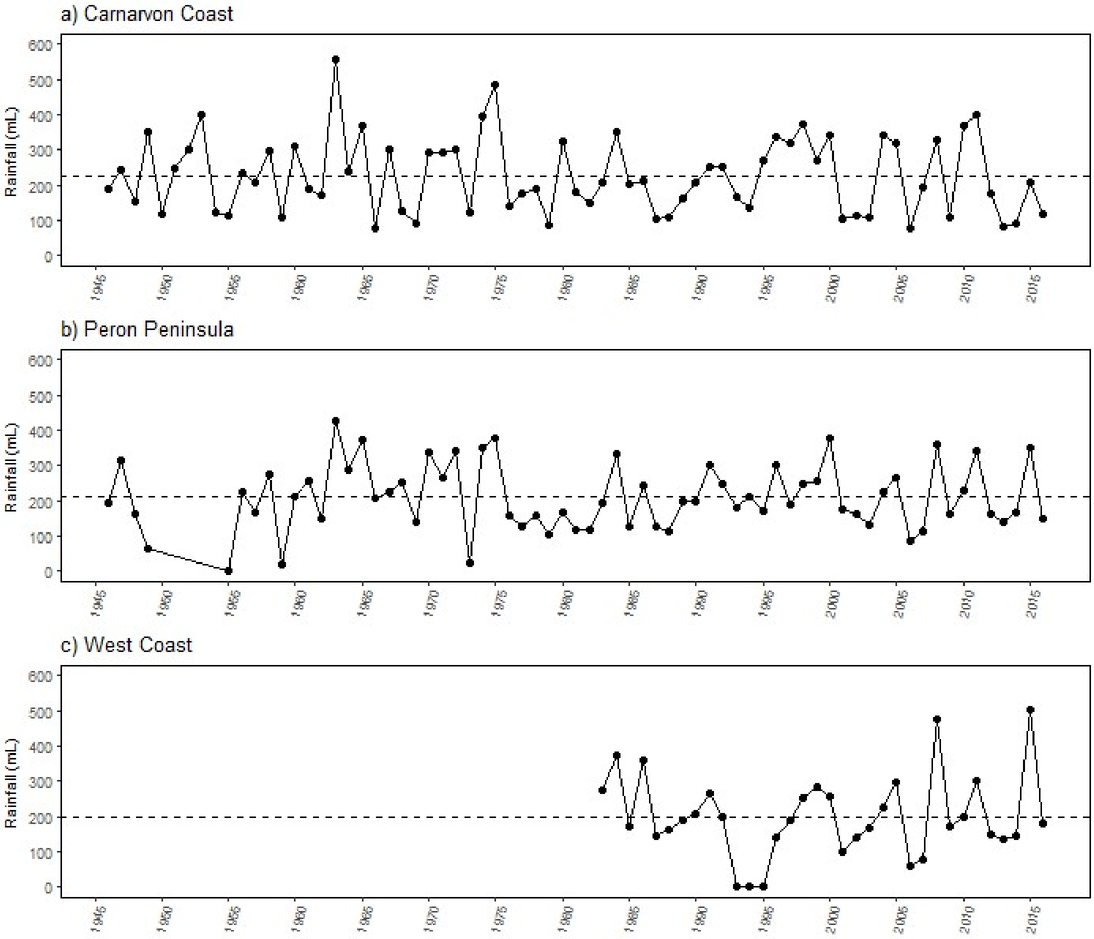

Figure 4.6: Total annual rainfall for Carnarvon (1945-2015), Denham (1945-2015), and Useless Loop (1983-2015), representing the Carnarvon coast, Peron Peninsula and west coast regions of the Shark Bay marine reserves. Black dashed line indicates overall mean rainfall across the monitoring periods.

Source: Bureau of Meteorology (2017).

Rainfall across the three regions is typically highest and most variable in February-March and May-July (Figure 4.7). Outside of these periods rainfall is generally very low. The Carnarvon coast received anomalous rainfall events (above two standard deviations from the long-term monthly mean) in January and February 2011 and March 2015. The Peron Peninsula received anomalous rainfall events in December 2010, January and February 2011 and March 2015. The west coast received anomalous rainfall events in August and December 2010, January 2011, and June and August 2015.

Error! Reference source not found. Figure 4.7: Total monthly rainfall for the Carnarvon coast, Peron Peninsula and west coast regions of the Shark Bay marine reserves, July 2010 to June 2016. Dotted line indicate baseline rainfall means (1945-2010, 1945-2010 and 1982-2010 respectively) and the shaded area indicates two standard deviations from the mean, with values outside this considered anomalous.

Source: Bureau of Meteorology (2017).

4.4.3 Sea level

Trends in sea level between 1985 and 2015 display medium-term oscillations resulting from regional climatic cycles associated with the Southern Oscillation Index (Figure 4.8). Annual mean sea level has ranged from a peak of approximately 1.21 m in 2011 to a low of approximately 0.93 m in 1993. As the time series currently incorporates less than two full cycles of the SOI climatic cycle, it is not considered appropriate to assess long term trends in this indicator. However, when examined across the mangrove condition monitoring period (2007-2015), there was an increasing trend in sea level from 2007 to 2011, followed by a decline from 2011 to 2015.

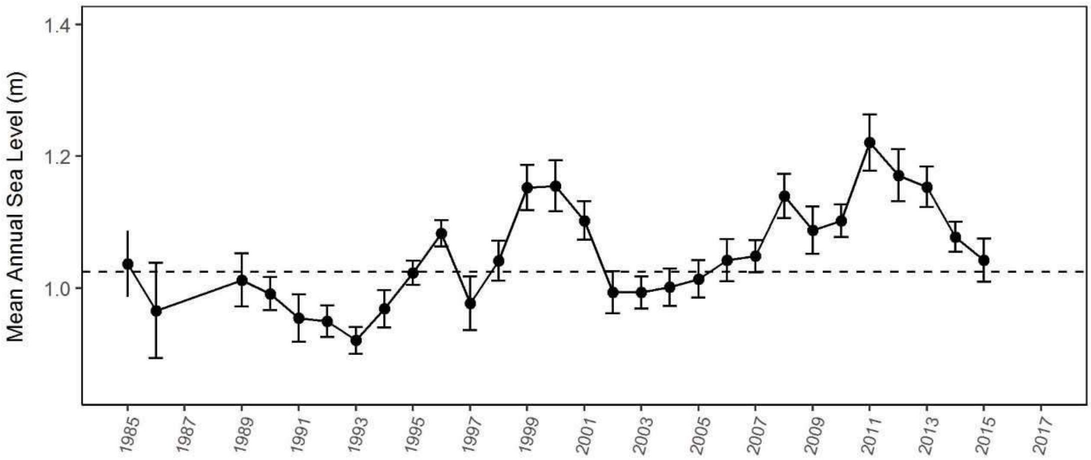

Figure 4.8: Mean annual sea level (± SE) at Carnarvon, 1985-2015. Mean values are calculated based on monthly sea level recordings within each year (annual n = 12) and measured above Tide Gauge Zero adjusted to the Australian Height Datum.

Source: Bureau of Meteorology, 2017.

4.4.4 River Discharge

Annual discharge from both the Wooramel and Gascoyne rivers have been highly variable across the monitoring period but exhibit no clear increasing or decreasing trends over time (Figure 4.9). While there is some variability between the two systems, overall patterns show a degree of similarity indicating that catchments are influenced by the same rainfall events. When examined across the mangrove condition monitoring period (2007-2015), there was exceptionally high discharge from both the Gascoyne and Wooramel rivers in 2011, following significant rainfall events in January and February of that year. There were also high levels of river discharge from the Wooramel in 2010 and the Gascoyne in 2015.

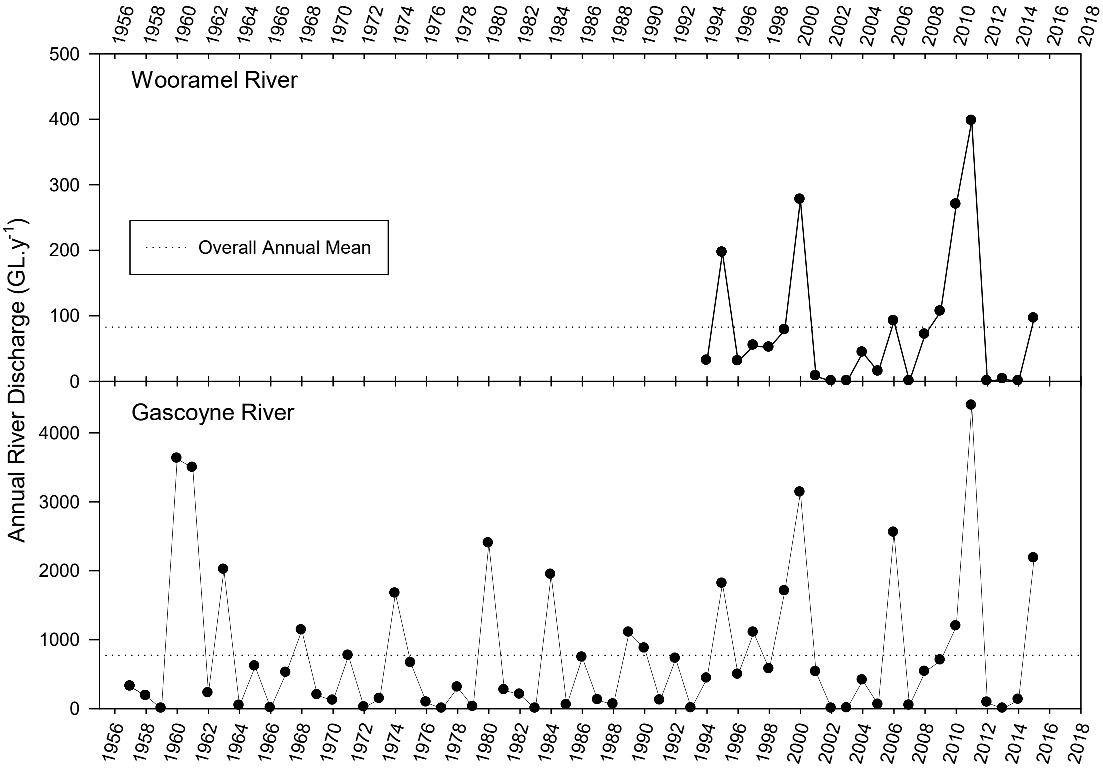

Figure 4.9: Total annual river flow (GL.y^{-1}) for the Wooramel (1994-2015) and Gascoyne (1957-2015) rivers, which discharge into the western gulf of the Shark Bay marine reserves. Black dotted line indicates overall mean river flow over the whole sampling period.

Source: Department of Water (2017)

4.4.5 Cyclones

Cyclone frequency and intensity information were acquired from the Bureau of Meterology (http://www.bom.gov.au/cyclone/history/tracks/index.shtml). All cyclones classified as category 1 or above that passed within 50 km and 200 km radii of Shark Bay (using Denham as a centre point) were identified as potential sources of wind damage to mangroves in the Shark Bay marine reserves

Three cyclones passed within a 50 km radius of the Shark Bay marine reserves between 1997 and 2015. These were: Frank in 1996, Steve in 2000, and Olwyn in 2015. Five cyclones passed within a 50-200 km radius. These were: Rhonda in 1997, Elaine in 1999, Alistair in 2001, Glenda in 2006, and Nicholas in 2008.

4.5 Synthesis

There have been major declines in the spatial extent and canopy cover of mangrove communities in all three regions of the Shark Bay marine reserves. Declines occurred in the Peron Peninsula and west coast regions between 2013 and 2015, and in the Carnarvon coast region between 2010 and 2015. Although proportionally lower than in the other regions, the decline in the

Carnarvon coast region represents the most significant in the marine reserves, with the loss of ~135ha of mangrove (~16% of total mangrove area in the region) (Figure 4.10). While no monitoring occurred between 2010 and 2015 due to the unavailability of appropriate remote sensing imagery, examination of larger scale satellite imagery on Google Earth indicates that declines were already occurring in 2012/13. This suggests that the declines in condition at this location likely began at least 1-2 years prior to those in the Peron Peninsula and west coast regions.

The specific cause of the decline in mangrove extent across the marine reserve is largely unknown. However, given the apparent time disparity in their commencement between the Carnarvon coast and the Peron Peninsula/west coast regions, it is possible that they were influenced by different pressures. While atmospheric temperature was frequently anomalously high across the reserves between 2011 and 2015, it is unlikely that this exceeded the tolerance of *Avicennia marina* or represents the main cause of declining mangrove condition. The decline in spatial extent in the Peron Peninsula and west coast regions following 2013 is consistent with a downward oscillation in sea level in the Shark Bay area, which is likely to have resulted in increased porewater salinity in intertidal soils. Lovelock *et al*. (2017) found a close relationship between mean sea level, porewater salinity and canopy condition (with a delayed response of approximately 2 years) in mangroves at the Ningaloo marine reserves, highlighting that sea level is also likely to be a major driver of overall condition within the Shark Bay marine reserves. As sea levels oscillate over medium time-frames (10-20 years) due to regional climate processes (Southern Oscillation Index), there is likely to be natural variation of mangrove condition over similar time scales. Below average rainfall between 2012 and 2014 and may have further increased porewater salinity and contributed to mangrove decline. In addition, category 3 tropical cyclone Olwyn passed directly over the Shark Bay area in early 2015, exposing the mangroves to potentially damaging winds. It is likely that the decline in the spatial extent of mangroves in the Peron Peninsula and west coast regions resulted from of a combination of these factors, with no clear evidence to suggest that any single pressure was the primary cause and that it may be the combined influence of these pressures.

While the same pressures may also have contributed to the losses in extent in the Carnarvon coast region following 2013, it is unlikely that they were the cause of the initial declines observed prior to this. The most likely cause was an extremely high discharge event that occurred from the Gascoyne River in February 2011, resulting from high rainfall in the catchment. This is known to have caused an extensive sediment plume that persisted for several weeks in the area where declines in mangrove condition were detected. While the most likely explanation is that mangrove pneumatophores were smothered by terrestrial sediments deposited by the plume, it is also possible that pollutants from agricultural areas in the catchment were carried down the river and into the mangrove area. There is, however, currently no evidence identifying the exact cause.

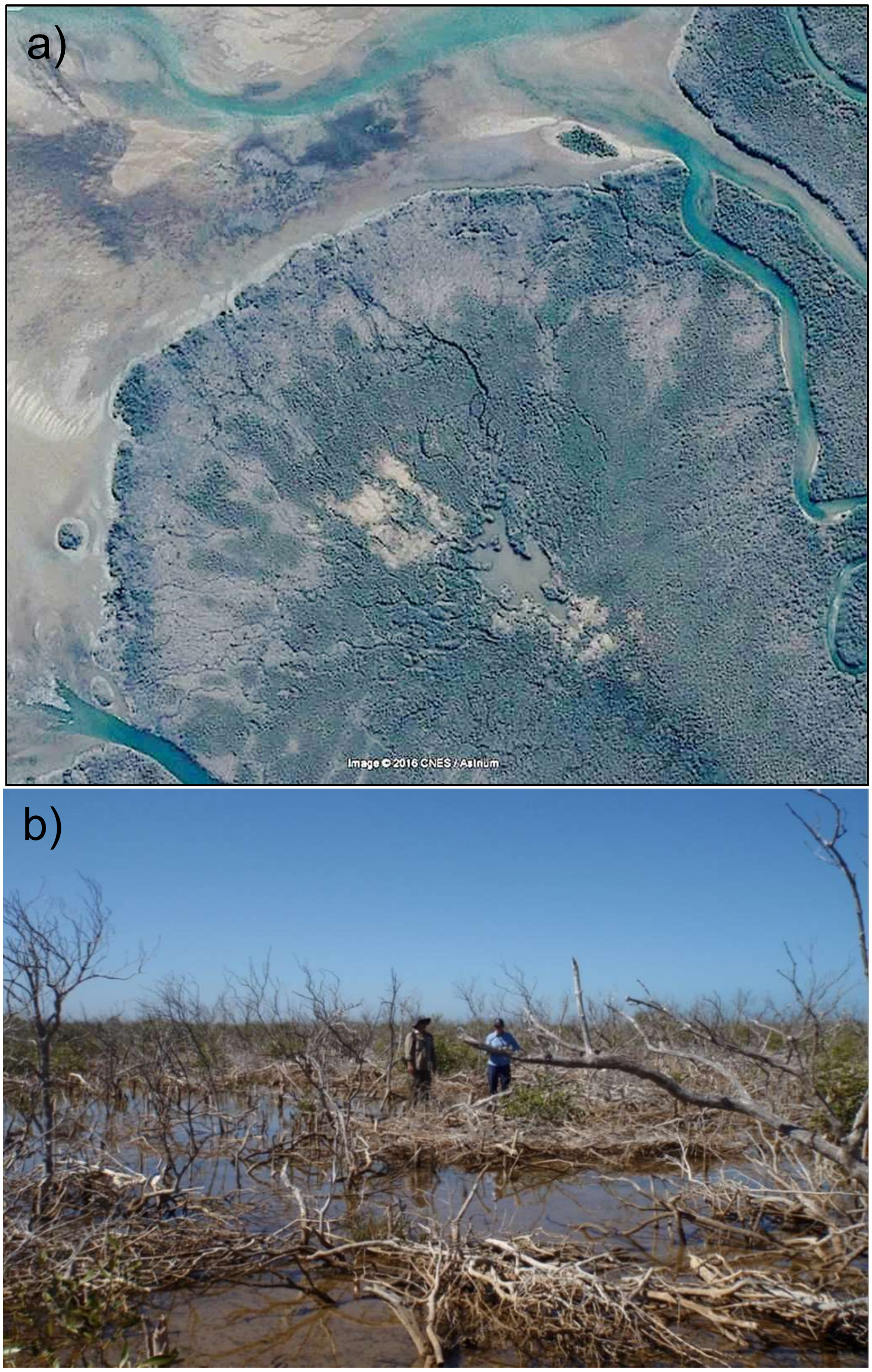

Figure 4.10: Images of mangrove loss in the Carnarvon coast region from (a) 2016 Google Earth imagery and (b) ground based assessments.

Source: Google Earth 2017 & DBCA

5 Coral Reef Communities (KPI)

5.1 Key Points

- Coral cover within the marine reserves has remained stable across the survey period (1996-2015) at approximately 15-25%
- Coral cover to the north of the marine reserves near Bernier and Dorre islands has declined by approximately 90-95% since 2010, as a result of thermal stress during the austral summer of 2010/2011.
- While declines in coral cover near Bernier and Dorre islands have occurred across the entire community, the greatest loss has occurred in the dominant family Acroporidae.
- The different coral condition trends between areas of Shark Bay is likely the result of local environmental/oceanographic conditions and the more robust life-history characteristics of the dominant coral family (Dendrophylliidae) within the western gulf.
- The confidence in the current condition assessment is low, due to the limited number of sampling points and the significant temporal gap between surveys in 1996 and 2010/2011. This assessment confidence will improve as more temporal data points are added. However, it should be noted that declines in coral condition observed near Bernier and Dorre islands following 2010 should be treated with a high degree of certainty.

5.2 Indicator Summary

	Trend	**Confidence**
Condition		
Coral cover	Stable	Low
Community composition	Stable	Low
Pressure		
Thermal stress	Increasing	High
Cyclones	Stable	Low

5.3 Condition Indicators

Condition indicators are measures of coral reef community ‘health’ relative to pressures acting in the marine reserves. Those indicators relevant to coral communities within the Shark Bay marine reserves and presented in this report are:

- Coral cover.
- Community composition.

Coral cover is a broad metric that encompasses the abundance of healthy live coral and is a suitable measure that is influenced by a range of pressures including heat stress, cyclones and pollution (Connell 1978; Rogers *et al.*

1983; Connell and Keough 1985). Community composition is also an important measure of condition as specific taxa are known to have differential responses to environmental variables and pressures (Rogers 1983; Rogers 1990; McClanahan *et al*. 2007; Berkelmans 2009). Changes in community composition can have major implications for the structure of reef-associated organisms given the diversity of roles that different taxa and growth forms play as habitat or food (Connell and Kingsford 1998; Wilson *et al*. 2010). Measures of coral recruitment are indicative of future supply and recovery potential within coral communities and represent a source of natural variation in coral abundance and the structure of coral communities (Hughes *et al.* 1999). Understanding the relationship between live coral abundance and recruitment is therefore important for understanding the long-term impact of pressures on recovery and changes in coral communities (Hughes and Tanner 2000; Acosta *et al*. 2011). While the inclusion of a measure of coral recruitment within the monitoring program is planned, to date only a single data point has been collected (i.e. in 2016) across coral sites in the western gulf and near Bernier and Dorre islands. As such, coral recruitment is not yet reported for the Shark Bay marine reserves.

The survey design for assessing coral community condition is based around the primary pressures acting on them (thermal stress and cyclones). As pressures are spatially and temporally variable, the design encompasses sites spread across coral dominated habitat within the marine reserves and at sites where historical information is most readily available (i.e. prior to the commencement of the standardised coral monitoring program in 2011). The incorporation of historical information is important as it provides valuable information on the medium-term dynamics of the condition indicators and can assist with the interpretation of more recent data. Public education is assumed to have a uniform spatial influence on pressures across the reserves and is hence only incorporated as a temporal component of the design. Significant coral habitat is limited to the western gulf area, with sites predominantly located along the shallow coast around Dirk Hartog Island and on shallow flats within the gulf (Figure 5.1). The design also incorporates sites outside of the marine reserves, but within the World Heritage Area near Bernier and Dorre islands, due to their proximity and relevance to regional coral reef processes. However, there is likely to be a degree of natural separation between the two areas relating to the physical oceanographic characteristics of the region (Burling *et al*. 2003b; Nahas *et al.* 2003; Hetzel *et al*. 2015).

Coral cover and community composition are measured with standard monitoring methods (Loya 1978; Brown *et al.* 2004). Five survey sites are located in the western gulf of the marine reserves and an additional four are located outside of the marine reserves along the east coast of Bernier and Dorre islands. The spatial coverage of sites prior to 2011 was sporadic, with only one of the five sites in the western gulf being surveyed in 1996 and three of the four sites at Bernier and Dorre islands being surveyed in 1996 and 2010. As such, the interpretation of some of the patterns observed is made with low confidence, with the exception of the substantial declines observed in coral cover after 2010.

Coral survey methods have changed slightly since monitoring began due to advances in camera technology. Three replicate 50m permanent transects have been used to collect benthic imagery at each site. Transect start and end points were fixed in 2010 and 2011 using GPS waypoints from the 1996 survey. From 2010 onwards, fifty downward facing benthic photo-quadrats (1 m height and approximately 1 m^2 image) were taken at regular intervals along each transect. Benthic video was used to collect imagery along transects in 1996 and 50 equidistant photo quadrats were taken from the video transects. To estimate cover and taxonomic classification of live hard coral or to categorise alternate substrata (e.g. sand, dead hard coral, turf algae), six points were placed on each 1m^2 photo-quadrat (Abdo *et al*. 2004).

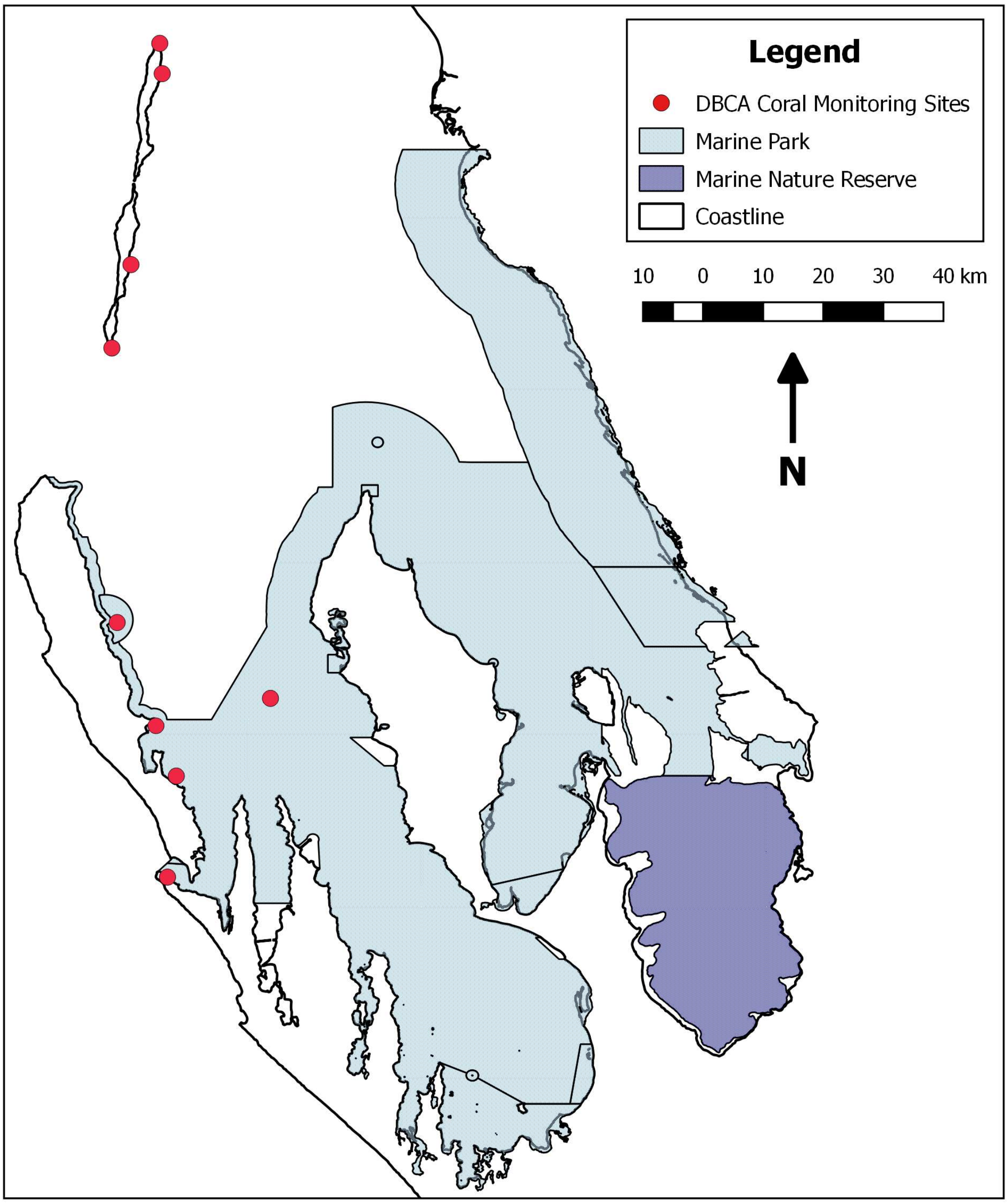

Figure 5.1: Location of DBCA coral monitoring sites in the Shark Bay marine reserves.

5.3.1 Coral cover

Coral cover at sites within the marine reserves has remained relatively stable at approximately 15-25% from 20010 to 2015 (Figure 5.2). A single data point prior to 1996 within the marine reserve indicates that coral cover has remained stable over the temporal range. In contrast, since 2010 there has been a major decline in coral cover near Bernier and Dorre Islands, from approximately 45% in 2010 to <2% in 2013 and 2015 (Figure 5.2). Data prior to 2010 indicates that coral cover was also relatively high in 1996 at approximately 30%.

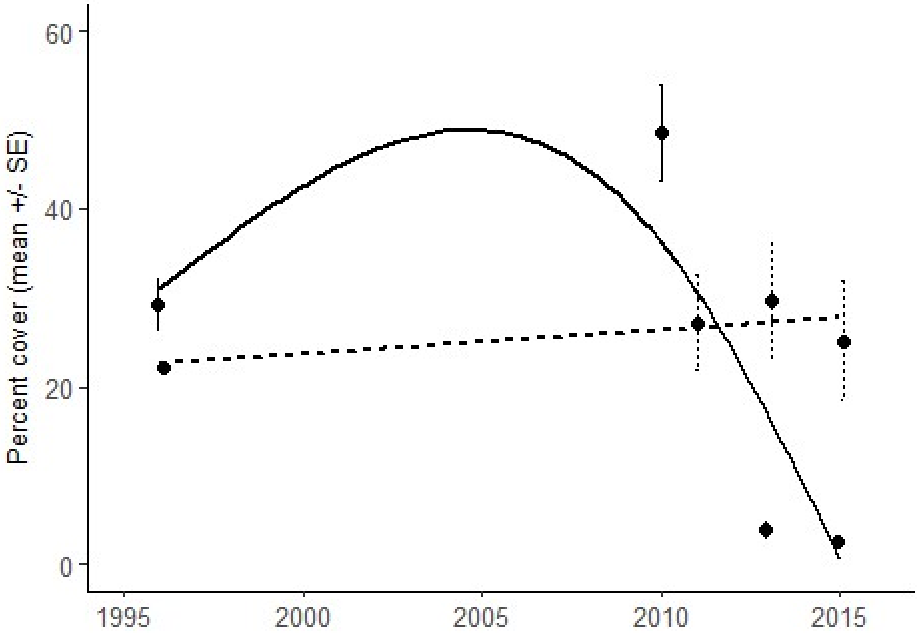

Figure 5.2: Mean coral cover (± SE) at all sites located in the Shark Bay marine reserves (dotted trend-line) and near Bernier and Dorre islands (solid trendline); Bernier and Dorre islands; n = 3 in 1996 and 2010, n = 4 in 2013 and 2015; marine reserves; n = 1 in 1996, n = 5 in 2011 to 2015. Trend-lines represent fitted non-parametric generalised additive models (Bernier and Dorre islands, Adjusted R^2 = 0.75; marine reserves Adjusted R^2 = 0.97).

5.3.2 Coral Community Composition

Coral community composition is assessed by measuring the percentage cover of the most common families at this location, which are Dendrophylliidae and Acroporidae. Mean cover of dendrophylliids in the marine reserves has remained stable (Figure 5.3). While only accounting for a relatively small percentage of overall benthic cover in the reserves, acroporids have generally declined across the same period, from approximately 4% in 1996 to approximately 1% in 2013 and 2015. Near Bernier and Dorre Islands, the mean cover of acroporids increased from approximately 10% in 1996 to approximately 40% in 2010 (Figure 5.3). This was followed by a sharp decline in 2013 and 2015 when mean cover dropped to 2-4%. The cover of dendrophylliids has declined consistently across the survey period at this location, from approximately 5% in 1996 to less than 1% in 2015. Interpretation of this information prior to 2010/2011 should, however, be treated with caution due to the large time gap between surveys in 1996 and 2010/2011 as well as and the limited number of sites surveyed in 1996.

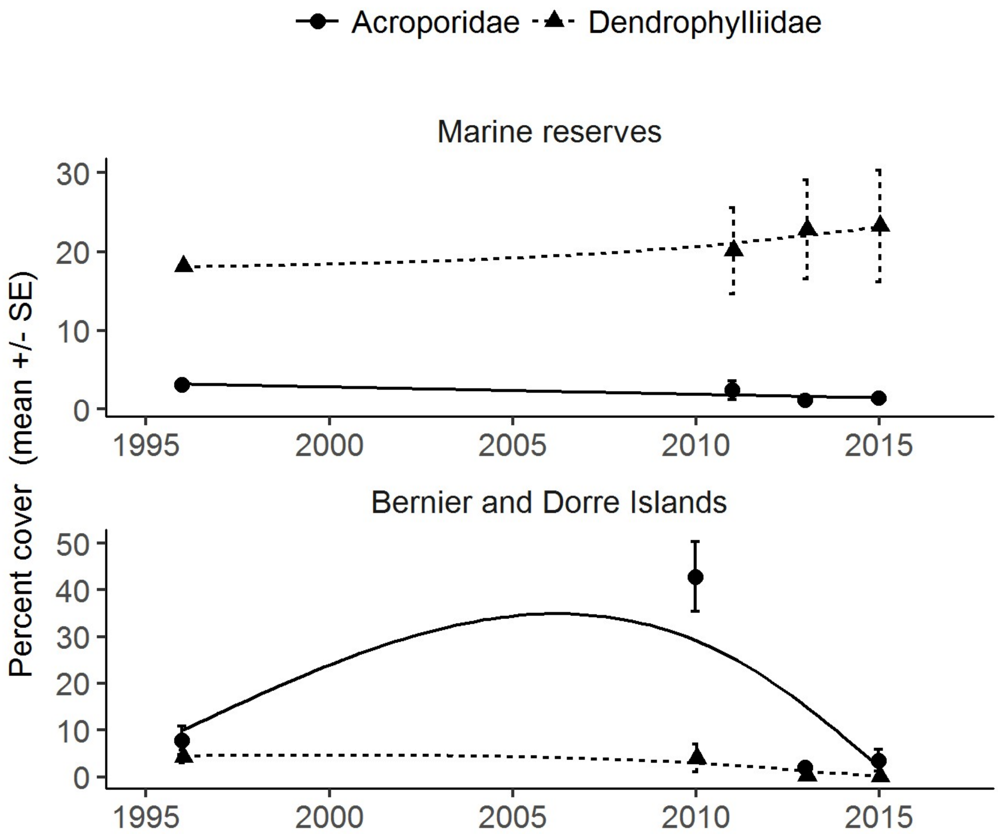

Figure 5.3: Mean percent cover (± SE) of the two dominant coral families at all sites located in the Shark Bay marine reserve and near Bernier and Dorre Islands; (Bernier and Dorre islands; n = 3 in 1996 and 2010, n = 4 in 2013 and 2015; marine reserves; n = 1 in 1996, n = 5 in 2011 to 2015). Lines represent fitted non-parametric generalised additive models for each location and taxa (Marine reserves Acroporidae Adjusted R2 = 0.61, Dendrophylliidae, Adjusted R2 = 0.87; Bernier and Dorre islands, Acroporidae, Adjusted R^2 = 0.32, Dendrophylliidae, Adjusted R^2 = 0.70).

5.4 Pressure Indicators

Those indicators relevant to the Shark Bay marine reserves and presented in this report are:

- Thermal stress.
- Cyclones.

Persistent thermal stress can result in coral bleaching and mortality and is assessed by measuring the accumulated temperature stress across a twelve-week period in which seawater temperature exceeds the local seawater climatology by 1°C. This is referred to as Degree Heating Weeks (DHW) (Glynn and D’croz 1990; Liu *et al*. 2006). Cyclones have a major influence on the structure of coral reef communities (Harmelin-Vivien 1994). While the effects can be highly variable (Wakeford *et al*. 2007) and dependent on local geomorphology and cyclone path, cyclones can cause significant damage by intense wave action (Done 1992; Massel and Done 1993; Tanner 1995; Blakeway 2004; Fabricius *et al.* 2008). While their effects on coral communities in Western Australia are generally poorly understood due to a lack of temporal data, assessments of cyclone pressure within the Shark Bay marine reserves are currently made using the number of cyclones that pass within a 50 km and 200 km radius of Denham (Speed *et al.* 2013).

Water quality issues such as sedimentation and nutrient input resulting from river discharge may also exert pressure on benthic communities in some parts of the marine reserves. However, they are not considered relevant in relation to the location of coral communities in the western gulf. Additionally, while coral predators are considered significant pressures on coral communities globally and in other Western Australian marine reserves (e.g. Ningaloo marine reserves), baseline surveys indicate that predators such as *Drupella cornus* and *Acanthaster planci* are not common in the Shark Bay marine reserves.

5.4.1 Thermal Stress

Sea surface temperature (SST) data from National Oceanic and Atmospheric Administration (NOAA) virtual stations at Shark Bay (http://coralreefwatch.noaa.gov) were used to assess thermal stress on corals as measured by DHW (Liu *et al*. 2006). DHW threshold levels four and eight are defined as those levels likely to cause bleaching and mortality of coral tissue, respectively. The assessment is based on DHW calculated from a 50 x 50 km satellite pixel formerly located off the west coast of Dirk Hartog Island. This measure provides a broad understanding of the potential for thermal stress within the marine reserves, without accounting for localised geomorphology and oceanography which are likely to alter SST at local scales (see Figure 2.2 for variation in SST at finer scales).

The Shark Bay area experienced significant thermal stress (assessed as level eight DHW) in the summer periods from 2011 to 2013, with cumulative stress measures being up to three times greater than thresholds thought to cause coral mortality in this region (Figure 5.4). Thermal stress was most intense in 2011 (24 DHW) followed by 2012 (18 DHW) and 2013 (12 DHW). The bleaching threshold (assessed as above four DHW) was also exceeded in 2008 (7.5 DHW) but did not reach the threshold likely to cause mortality.

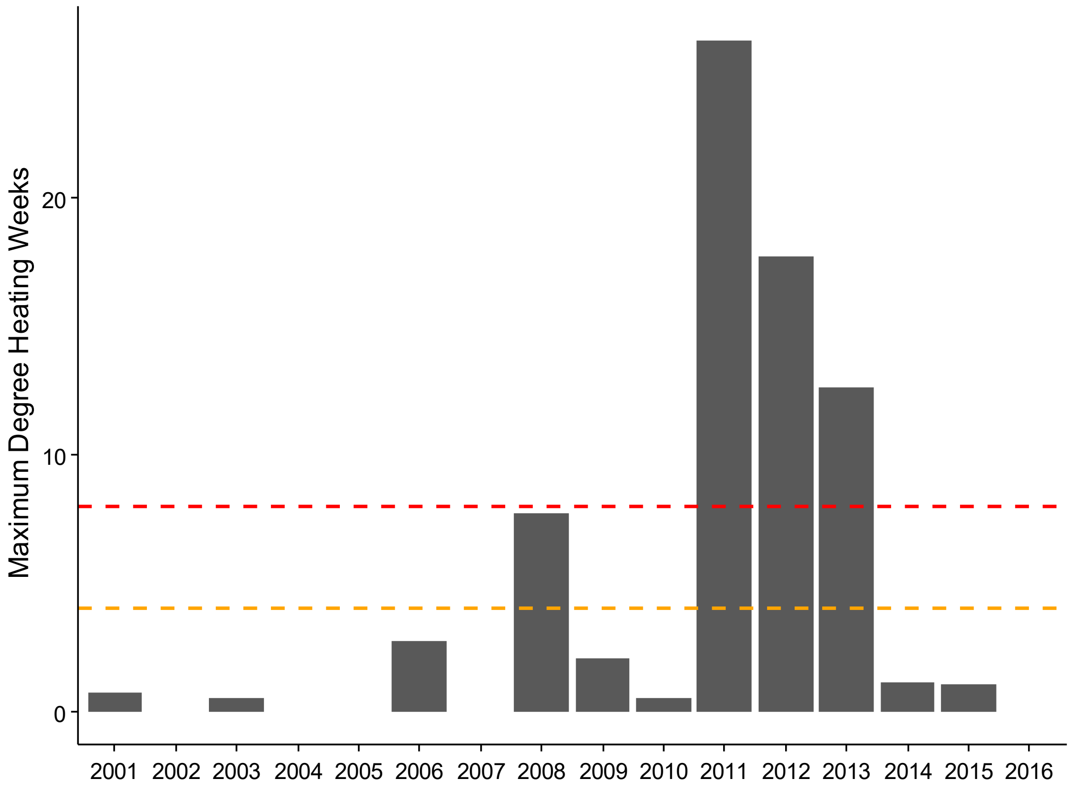

Figure 5.4: Maximum degree heating weeks (DHW) recorded in the Shark Bay marine reserves, based on a 50 km^2 satellite pixel formerly located off the west coast of Dirk Hartog Island. Orange dashed line indicates a four DHW threshold likely to cause severe bleaching, and red dashed line indicates an eight DHW threshold likely to cause mortality.

Source: NOAA coral reef watch

5.4.2 Cyclones

Cyclone frequency and intensity information were acquired from Bureau of Meoerology (http://www.bom.gov.au/cyclone/history/tracks/index.shtml). All cyclones classified as category 1 or above that passed within 50 km and 200 km radii of Shark Bay (using Denham as a centre point) were identified as potential sources of wave damage to coral in the Shark Bay marine reserves (Done 1992; Massel and Done 1993; Tanner 1995; Blakeway 2004; Fabricius *et al*. 2008).

Three cyclones passed within a 50 km radius of the Shark Bay marine reserves between 1997 and 2015. These were: Frank in 1996, Steve in 2000, and Olwyn in 2015. Five cyclones passed within a 50-200 km radius. These were: Rhonda in 1997, Elaine in 1999, Alistair in 2001, Glenda in 2006, and Nicholas in 2008.

5.5 Synthesis

The condition of coral communities in the Shark Bay marine reserves has remained relatively stable since monitoring began in 1996. While coral cover has generally remained stable, there have been differing trends for the two dominant coral families, with an increasing trend in the dominant family Dendrophylliidae and a decrease in the less dominant family Acroporidae. In contrast, coral communities immediately to the north of the reserves near Bernier and Dorre Islands have drastically declined since 2010, from approximately 45% to less than 2% in 2013 and 2015. The measured declines have occurred for both acroporid and dendrophylliid corals, although they are most apparent among acroporids, which were the dominant family up until 2010. While the small number of temporal data points across the 20-year survey period indicates a low level of confidence in the assessment of overall trends in coral condition, there is a high degree of certainty around the declines that occurred near Bernier and Dorre islands following 2010 (i.e. the decline in coral cover was severe with relatively small levels of variance around the measures).

The decline of acroporid corals in the western gulf and overall coral cover near Bernier and Dorre Islands can be attributed to significant thermal stress resulting from anomalously high seawater temperatures in January 2011 (Moore *et al*. 2012). Additional thermal stress in April 2012 and 2013 may have also contributed to the recorded declines. Sites in the marine reserves were less affected, potentially due to local oceanographic processes resulting in lower thermal stress in the western gulf or because of the differences in coral community composition to those at the highly affected sites near Bernier and Dorre islands. Coral communities in the marine reserves are dominated by *Turbinaria* spp. (Family Dendrophylliidae), which is known to be relatively tolerant to stress events. In comparison, the sites near Bernier and Dorre Islands were dominated by acroporids prior to 2011, which are more sensitive to environmental pressures such as thermal stress (McClanahan *et al*. 2007; Berkelmans 2009). However, while declines in acroporid cover were recorded at both locations, there was also a decline in the cover of dendrophylliids near Bernier and Dorre Islands, indicating that sites in this region may have been exposed to greater levels of thermal stress than those in the western gulf. Given that calculations of DHW are made from a satellite pixel in oceanic waters to the west of Dorre Island, it is most likely that the thermal stress reported here is indicative of the Bernier and Dorre islands area only, and less so of the inner gulf regions where oceanographic or environmental processes may have mitigating effects.

The effects of cyclones on the condition of coral communities across the marine reserves have been less clear. While the reserves have been exposed to multiple cyclones over the last 20 years, the temporal coverage of the condition indicators does not currently allow for interpretation with regards to this pressure. However, other reefs in Western Australian marine reserves have exhibited extremely high spatial variability in structural damage resulting from cyclones. For example, cyclones Vance, Carlos, and Bianca contributed to losses of corals in the eastern sector of the Ningaloo marine reserves in 2011 but did not have a measurable impact on coral in the northern and southern sector (Moore *et al*., 2012; DBCA, *unpublished data*). Anecdotal observations from 2016 suggest that coral condition within the Shark Bay reserves may have been affected by Cyclone Olwyn in early 2015, but this information needs to be verified by quantitative survey.

Although coral recruitment is not presented in this report, data for this condition indicator was collected for the first time in 2016. This indicated relatively high recruitment inside the reserves in the western gulf, but minimal recruitment near Bernier and Dorre Islands. However, the stochastic nature of recruitment means that information needs to be collected over an extended time period before conclusions can be made.

6 Finfish Communities (KPI)

6.1 Key Points and Recommendations

- Condition assessments are currently focused on finfish communities associated with coral habitat and are considered to be stable within the marine reserves across the monitoring period (2010-2015).
- While community composition within the reserves has remained stable, there has been a decline in the abundance of coral dependent fishes (corallivores) immediately to the north of the reserves near Bernier and Dorre islands, as a result of declining coral condition at this location.
- Boat-based recreational fishing effort declined between 1998/99 and 2000/01 but remained stable from 2000/01 to 2010 and across the presented monitoring period (2011-2016).
- Commercial and charter fishing effort have both declined across the monitoring period (2008-2015).
- Assessments of finfish community condition are typically associated with high variance due to their mobile nature and often patchy distributions. While the current assessments are considered to have low levels of confidence, this will improve as the program is refined to include non-coral habitats and spatial and temporal resolution improves.

6.2 Indicator Summary

	Trend	Confidence
Condition		
Target abundance	Stable	Low
Community composition	Stable	Low
Species richness	Stable	Low
Pressure		
Charter fishing	Decreasing	Medium
Commercial fishing	Decreasing	Medium
Recreational fishing	Stable	Medium
Benthic habitat loss - coral	Stable	Low

6.3 Condition Indicators

Condition indicators are measures of finfish community ‘health’ relative to pressures acting on them. Those indicators relevant to finfish communities within the Shark Bay marine reserves and presented in this report are:

- Target species abundance.
- Community composition.
- Species richness.

Changes in the abundance of target species can be indicative of pressure exerted directly by fishing on particular species, in conjunction with habitat and environmental variability. This generally affects higher trophic groups and is commonly referred to as a ‘top-down’ effect. The broader measure of fish community composition is indicative of the ecosystem services provided by finfish, with healthy systems containing a diverse trophic structure, measured by relative numbers of piscivores, large herbivores, mobile invertivores and corallivores, that is characteristic of healthy coral reef function (McClanahan and Shafir 1990; Hughes *et al*. 2007). Changes within different trophic levels are likely to indicate different pressures (e.g. fishing activity acting on upper trophic groups, or habitat changes acting on lower and mid trophic groups), so this measure is particular helpful for identifying causes of change (Wilson *et al*. 2008). Although species richness (a measure of community diversity) is not considered to be a strong indicator of community condition (Holbrook *et al*. 1994), it is reported here as it is an important indicator for regional comparisons and is included as a performance measure in the current Shark Bay marine reserves management plan (CALM 1996).

Size structure and biomass are also considered to be key indicators for target fishes as these measures are highly sensitive to fishing effort (Jennings and Dulvy 2005; Nash and Graham 2016). However, these indicators are not as yet reported here as research is still examining how best to extract these data from imagery collected using the Diver Operated stereo-Video (stereo-DOV) and Baited Remote Underwater stereo-Video (stereo-BRUV) used by the Department for fish monitoring *(Holmes et al*. 2013).

The sampling design for assessing fish community condition is based around the primary pressures acting on them (fishing effort and habitat change), and the management strategies used to manage biodiversity conservation within the marine reserves (spatial zoning, catch and size limits). Fish communities are currently only monitored at coral dominated sites, despite being a relatively minor benthic habitat in the overall Shark Bay marine reserves. Appropriate methods to monitor fishes in Shark Bay’s dominant seagrass communities are currently being developed as a part of a Departmental research project and will be integrated into the monitoring program when results become available. Sites are located on coral dominated habitat in the western gulf and spread between management zones where fishing is permitted (general use and recreational zones) and prohibited (sanctuary zones and Hamelin Pool Marine Nature Reserve). Monitoring currently occurs at two shallow water (2-6 m depth) sites, one within the Sandy Point Sanctuary Zone, the other in the general use zone (Figure 6.1). A further three sites (one sanctuary zone, two general use zone) have been surveyed once, but will not be included within the reported dataset until such time as further sampling points are collected. An additional three shallow water sites have been surveyed outside of the marine reserves, along the east coasts of Bernier and Dorre islands, and have been included within this report due to their close proximity and relevance to regional and overall Shark Bay ecosystem and its management. Surveys use stereo-DOV on six replicate 50 x 5 m belt transects per site and are completed during the winter months (April-September).

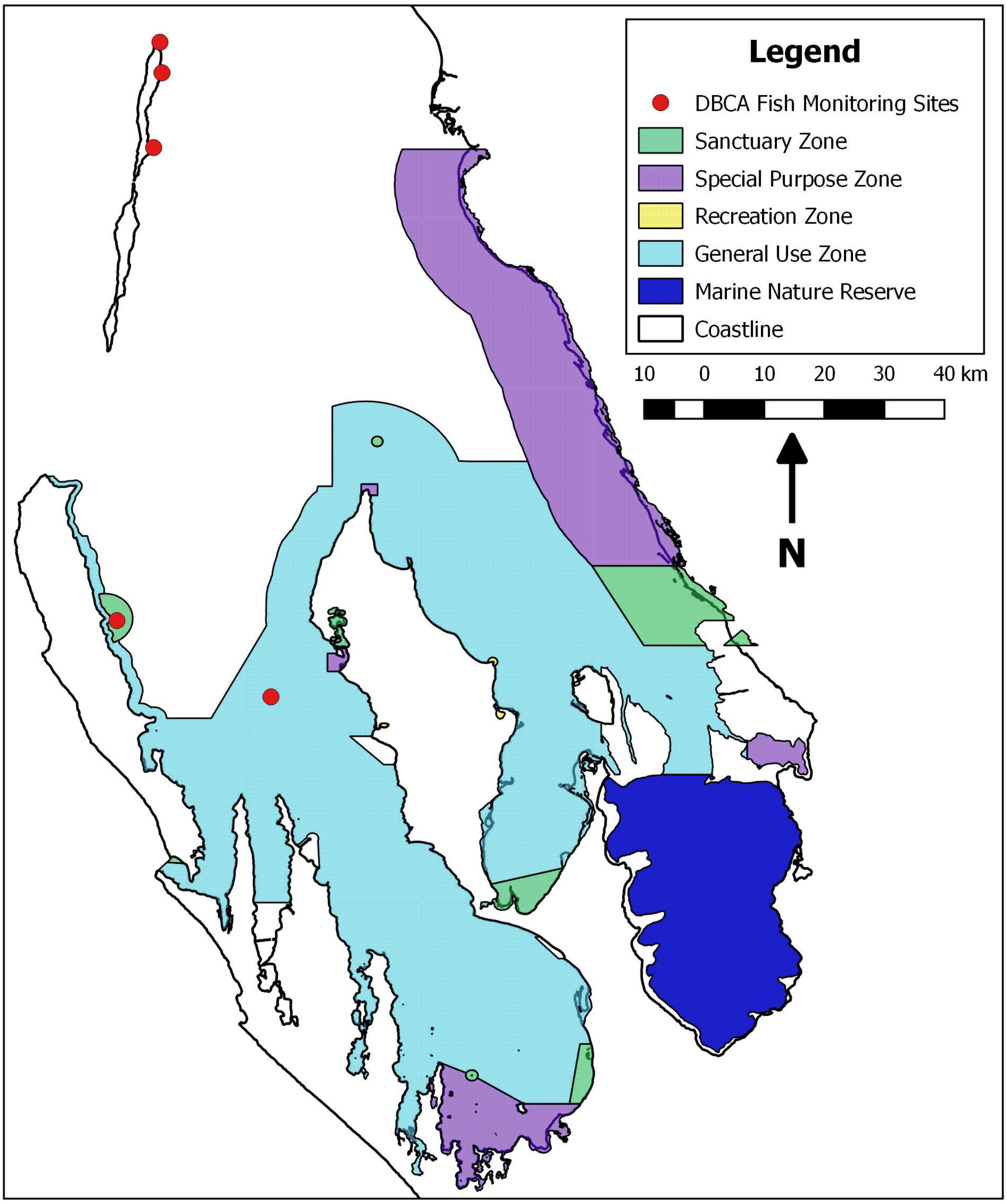

Figure 6.1: Location of DBCA fish monitoring sites on coral dominated habitat in the Shark Bay marine reserves.

6.3.1 Target Species Abundance

Target species are identified based on Department of Primary Industries and Regional Development's (DPIRD) phone diary surveys across the Carnarvon/Shark Bay zone of the Gascoyne Coast bioregion (Ryan *et al*. 2015, 2017). Only those species present in the depth range being sampled are included. The main demersal target species potentially present in shallow (2-6m) waters of the Shark Bay marine reserves are pink snapper (*Chrysophrys auratus*), grass emperor (*Lethrinus laticaudis*), baldchin groper (*Choerodon rubescens*), spangled emperor (*Lethrinus* *nebulosus*), orange-spotted grouper (*Epinephelus coioides*), Malabar grouper (*Epinephelus malabaricus*), rankin cod (*Epinephelus multinotatus*), chinaman rockcod (*Epinephelus rivulatus*), coral trout (*Plectropomus* spp.) and stripey snapper (*Lutjanus carponotatus*). Although all species occur in the Shark Bay marine reserves, they typically represent only a moderate proportion of the overall fish community. It is also acknowledged that the depth range and coral dominated habitat currently sampled is likely to represent sub-optimal habitat for grass emperor (*L. laticaudis*), pink snapper (*C. auratus*) and rankin cod (*E. multinotatus*). As such, abundances are likely to be naturally low within the presented data. This list does not include targeted pelagic species, which are not well sampled using the stereo-DOV method at shallow water sites.

The most common targeted species recorded at the coral dominated sites were *L. carponotatus* (approximately 76% of all targeted fish observed) and *Plectropomus* spp. (approximately 17% of all targeted fish observed). The recorded abundances of *C. rubescens*, *L. laticaudis* and *L. nebulosus* were all minimal (together approximately 6% of all targeted species), while *C. auratus* was not observed at all. This is considered representative of the preferred habitats of each of the species, with both *L. carponotatus* and *Plectropomus* spp. both generally associated with reef structure and/or coral habitat.

Monitoring is currently limited to two surveys across five years, making it inappropriate to make conclusions on temporal trends in target abundance at this time. While considered stable, confidence in this assessment is hence low. The high levels of variance also make interpretation between zones difficult at this point, with no consistent trend apparent across both survey years (Figure 6.2). The higher abundance of targeted species recorded in general use zones in 2015 is likely a result of site placement, with appropriate habitat being more effectively sampled and marked at this location during this sampling period.

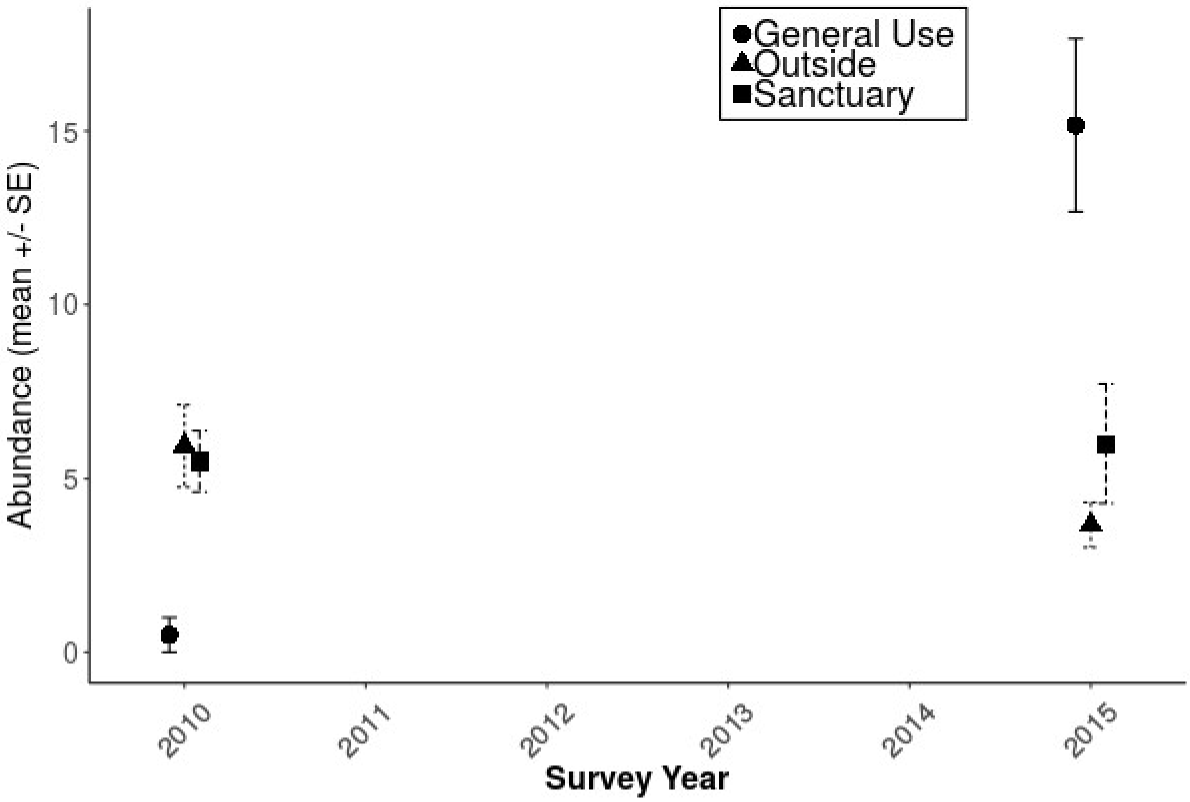

Figure 6.2: Mean abundance 250 m^{-2} (± SE) of targeted fish species in the Shark Bay marine reserves, based at shallow coral dominated sites in general use zones (open to fishing), sanctuary zones (closed to fishing) and outside the reserves near Bernier and Dorre islands 2010-2015. General use zone n = 6, sanctuary zone n = 6, outside n = 18.

6.3.2 Community Composition

Monitoring is currently limited to two surveys across five years, making it inappropriate to make conclusions on temporal trends in community composition at this time. There are however some interesting patterns worthy of note. There were higher abundances of fishes recorded in 2015 across all four trophic groups in the general use zone (Figure 6.3), but this is likely associated with a slight shift in the placement of this site over more appropriate habitat prior to the survey. There was also a noted decline in the abundance of corallivores at sites outside of the marine reserves near Bernier and Dorre islands, from approximately 12.5 fish 250 m^{-2} to approximately 1 fish 250 m^{-2}.

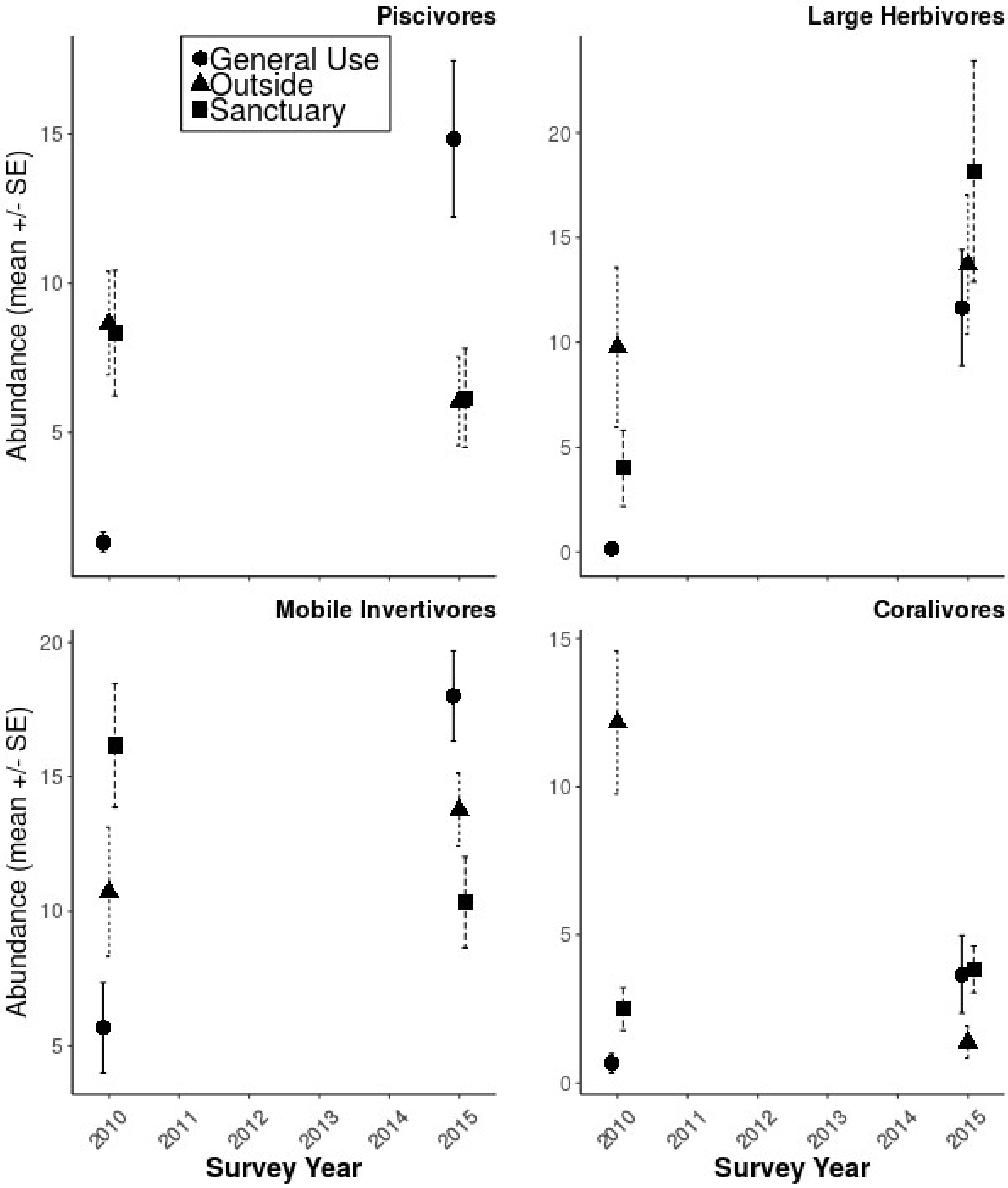

Figure 6.3: Mean abundance 250 m^{-2} (± SE) of piscivores, large herbivores, mobile invertivores and corallivores in the Shark Bay marine reserves, based at shallow coral dominated sites in general use zones (open to fishing), sanctuary zones (closed to fishing) and outside of the reserves near Bernier and Dorre islands, 2010-2015. General use zone n = 6, sanctuary zone n = 6, outside n = 18.

6.3.3 Species Richness

Monitoring is currently limited to two surveys across five years, making it inappropriate to make conclusions on temporal trends in species richness at this time. Species richness was generally higher at sites outside of the marine reserves near Bernier and Dorre islands, followed by the sanctuary zone (Sandy Point Sanctuary Zone) and the general use zone (Figure 6.4). While alternate explanations are possible, the higher species richness recorded within the general use zone in 2015 relative to 2010 is most likely associated with a slight shift in the placement of this site over more appropriate habitat prior to the survey.

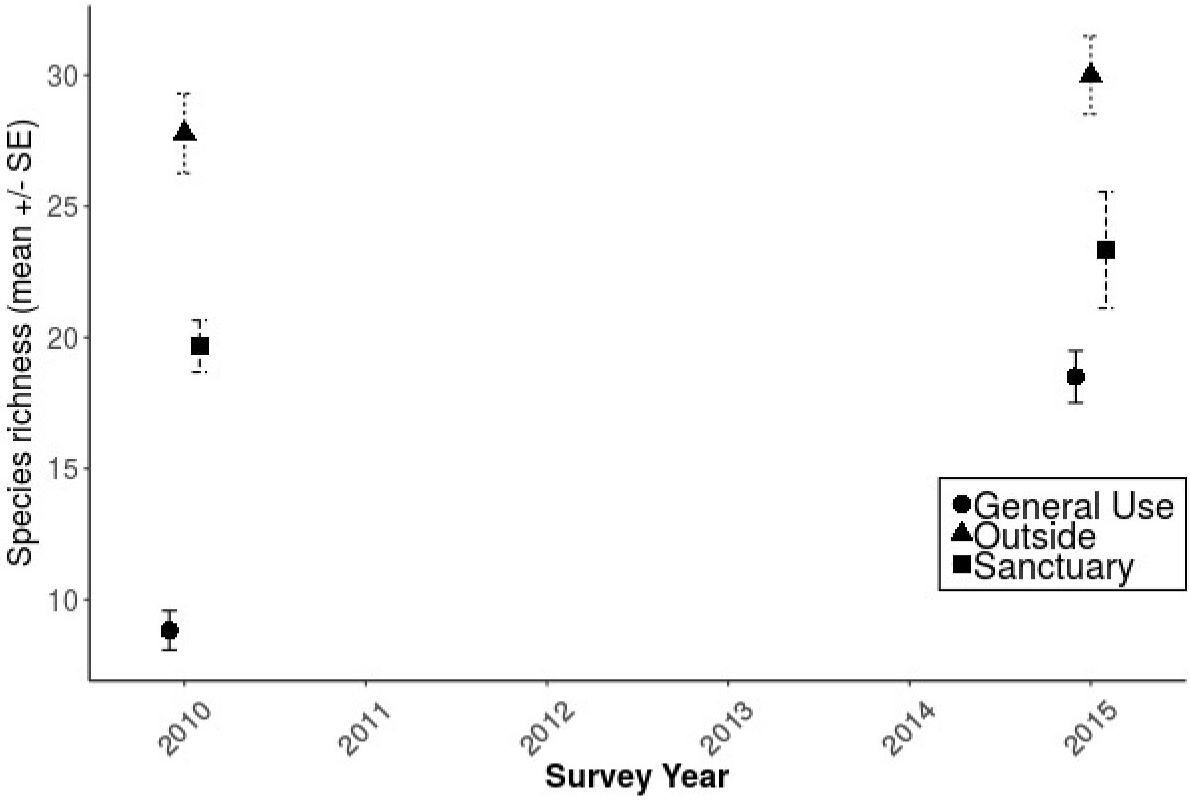

Figure 6.4: Mean species richness 250 m^{-2} (± SE) of fishes in the Shark Bay marine reserves, based at shallow coral dominated sites in general use zones (open to fishing), sanctuary zones (closed to fishing) and outside of the reserve near Bernier and Dorre islands 2010-2015. General use zone n = 6, sanctuary zone n = 6, outside n = 18.

6.4 Pressure Indicators

Pressure Indicators are considered to be reliable measures of the primary causes of change in fish communities within the marine reserves. Those indicators relevant to the Shark Bay marine reserves and presented in this report are:

- Recreational fishing.
- Charter fishing.
- Commercial fishing.
- Benthic habitat loss – coral.

Recreational fishing contributes the greatest proportion to overall fishing effort within the marine reserves (DPIRD, *pers comm*). While information specific to the marine reserve area is not currently available, state-wide DPIRD surveys provide a general perspective of trends in boat-based fishing activities in the Gascoyne Coast bioregion, and catch composition within the Carnarvon/Shark Bay zone (Ryan *et al.*

2013, 2015, 2017) (Figure 6.5). Recreational fishing activity is assessed through phone-diary surveys of fishers sampled from the Recreational Boat Fishing Licence (RBFL) to provide estimates of ‘boat-based recreational fishing effort’ and ‘catch’ of targeted species (Ryan *et al.* 2013, 2015, 2017). Prior to 2011/12, data on boat-based recreational fishing effort in the Shark Bay area was collected between 1998/99 and 2010 via standardised 12 month bus-route surveys boat ramp surveys (Wise *et al.* 2012). However, differences in methods and spatial coverage mean that the two datasets are not directly comparable, making long-term interpretation difficult. Estimates for both datasets do not include shore-based fishing activity, which does not require a licence but may still contribute significantly to overall recreational fishing effort (Smallwood and Gaughan 2013). As recreational fishing information is both broader in spatial scale and indicative of boat-based recreational fishing only it may not be fully representative of trends within the marine reserves.

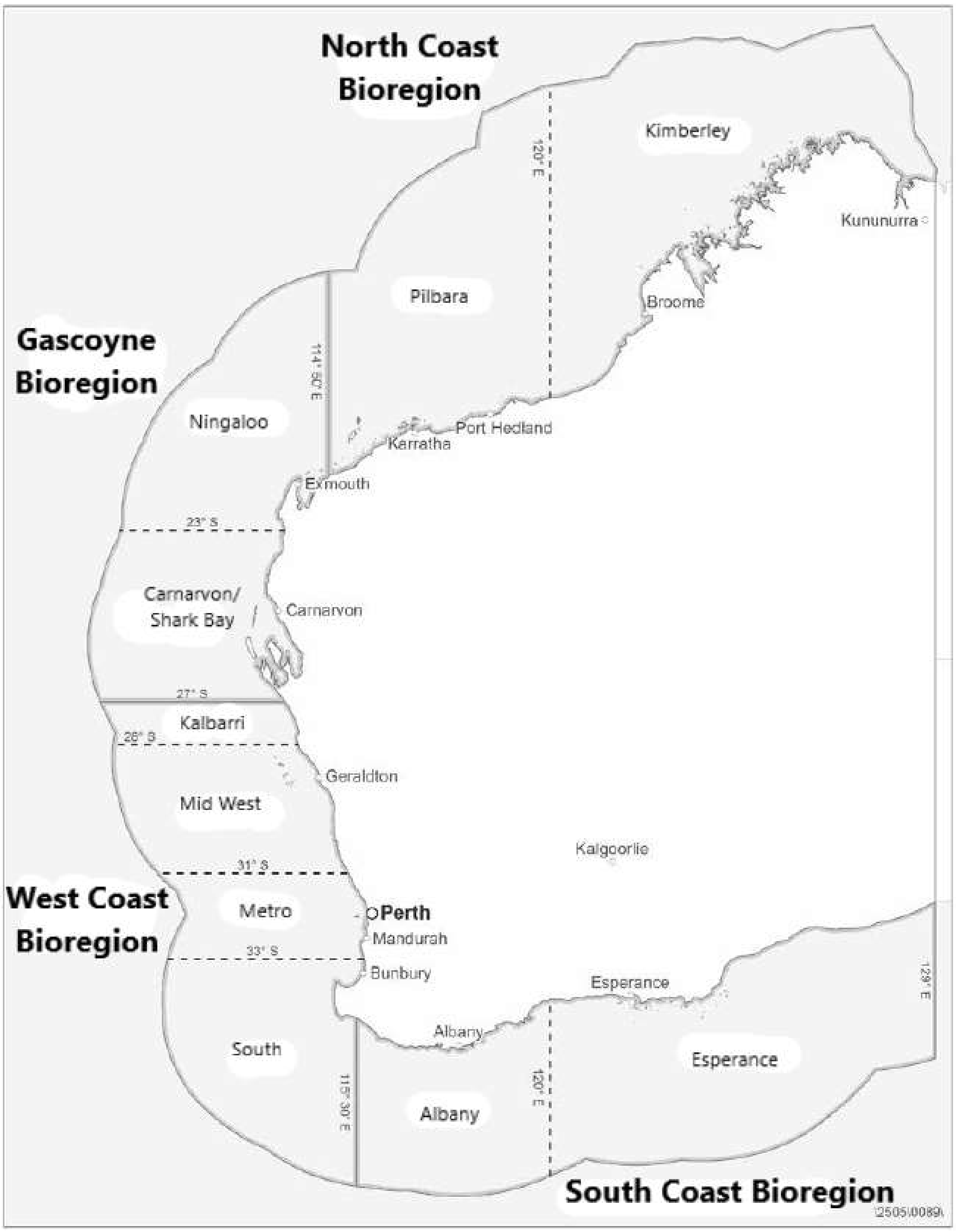

Figure 6.5: Map of the Western Australian coastline, highlighting the geographic range of the Gascoyne Coast bioregion (incorporating both the Ningaloo and Shark Bay marine reserves) and Carnarvon/Shark Bay zone (incorporating the Shark Bay marine reserves) used to make regional assessments of recreational fishing effort (Ryan *et al*., 2015).

Charter fishing activity is assessed through Tour Operator Returns maintained by licensed charter operators, which are recorded in 5 x 5 nm blocks (DoF 2012). This information has been restricted to the Shark Bay Marine Park by using GIS to overlay the charter catch and effort reporting blocks to that of the marine park boundary. Data were restricted to grid blocks that overlapped the marine park and cover an area larger than the park (i.e. the estimates include data from outside of the marine park, or non-marine park fishing effort and are likely to overestimate the effort and catch).

Commercial fishing activity is assessed through daily logbook returns from licensed commercial operators, which are recorded in 10 x 10 nm blocks. This information has been restricted to the Shark Bay Marine Park by using GIS to overlay the commercial catch and effort reporting blocks to that of the marine reserve boundary. Data were restricted to grid blocks that overlapped the marine park and cover an area larger than the marine park (i.e. the estimates include data from outside of the marine park, or non-marine park fishing effort and are likely to overestimate the effort and catch). Data from the Shark Bay Beach Seine and Mesh Net managed Fishery (SBBSF) have been provided. It should be noted that the Gascoyne Demersal Scalefish Managed Fishery does not operate inside either of the gulfs that comprise the marine reserve area and occurs outside of the line between the northern tip of Dirk Hartog Island and the coast (Fletcher *et al.* 2017).

Seagrass, coral reefs and mangroves are considered the primary benthic habitats important for finfish communities in the reserves. While seagrass is the dominant habitat, all are considered key habitats for adult and juvenile fishes. However, as the sampling design for fish communities only incorporates coral dominated sites, only information on the condition of coral reef communities is included within this section. While the Department does have a standardised monitoring program for seagrass communities (Chapter 3) and mangroves (Chapter 4) in the reserves, changes to these habitats will not be considered a relevant pressure on fish communities until the survey design appropriately incorporates seagrass and mangrove sites.

6.4.1 Recreational Fishing

Boat-based recreational fishing effort within the Shark Bay area declined between 1998/99 (approximately 70,000 hours fished) and 2000/01 (approximately 39,000 hours fished), before staying relatively stable through to 2010 (37,000 to 49,000 hours fished) (Wise *et al*. 2012). More recently, within the wider Gascoyne Coast bioregion, boat-based recreational fishing effort declined slightly between 2011/12 (253,930 ± 17,245 SE hours fished), 2013/14 (211,967 ± 15,671 SE hours fished) and 2015/16 (169,312 ± 12,914 SE) (Ryan *et al.* 2017). Despite the slight recent decline, recreational fishing effort is currently considered stable due to the short temporal resolution of the more recent data collection process, and issues directly comparing the two data sets.

The most recent surveys indicate that the most common species caught (both retained and released) by recreational fishing activity in the Carnarvon/Shark Bay zone of the Gascoyne Coast bioregion (Figure 6.4) are (in order of total catch retained): pink snapper (*Chrysophrys auratus*), grass emperor (*Lethrinus laticaudis*), spangled emperor (*Lethrinus nebulosus*), baldchin groper (*Choerodon rubescens*), stripey snapper (*Lutjanus carponotatus*), rankin cod (*Epinephelus multinotatus*), orange-spotted grouper (*Epinephelus coioides*), Malabar grouper (*Epinephelus malabaricus*), and coral trout (*Plectrpomus* spp.) (Ryan *et al.* 2017). Of these *C. auratus* and *L. laticaudis* make up the highest proportion of the catch, together comprising approximately 76% of the total catch amongst the above list. Red emperor (*Lutjanus sebae*), Redthroat emperor (*Lethrinus miniatus*), Spanish mackerel (*Scomberomorous commersoni*) and Cobia (*Rachycentron canadum*) are also reported as having high catch numbers within the Carnarvon/Shark Bay zone, but these are not well sampled under current condition assessments.

The most recent (2007/2008) spatial analysis of fishing activity indicated that most shore-based recreational fishing occurs throughout the western gulf, to the south of Carnarvon and directly adjacent to Monkey Mia. However, significant fishing activity also occurs out of coastal camping/access areas at Bush Bay, Cape Peron, Steep Point and Eagle Bluff (Smallwood and Gaughan 2013). There have been no assessments of recreational fishing compliance relating to marine reserve management zones.

6.4.2 Charter Fishing

There were between 5 and 12 active licence holders operating annually in the Shark Bay Marine Park between 2008 and 2014. Charter fishing effort (assessed as the number of days fished each year) in the marine park decreased from 261 fishing days in 2008 to a low of 129 fishing days in 2011, remaining less than 186 fishing days through to 2016 (Figure 6.6).

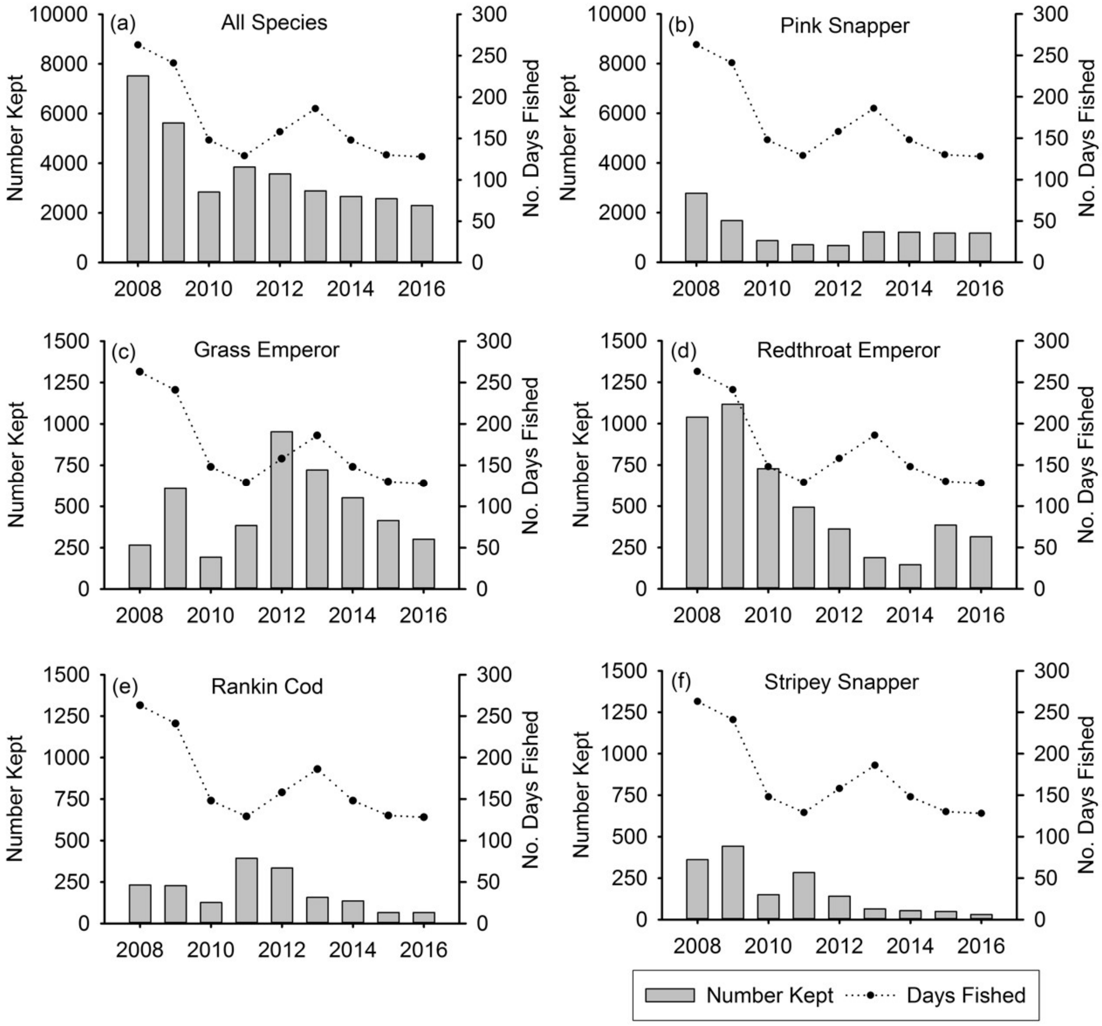

Figure 6.6: Total charter fishing catch and annual number of days fished (a) and for the five most retained species, pink snapper (b), grass emperor (c), redthroat emperor (d), rankin cod (e), and stripey snapper (f) in the Shark Bay Marine Park, 2008-2016. Information was obtained using charter operator logbook returns for all 5 x 5 nm recording blocks that intersect with the marine park boundary.

Source: Department of Primary Industries and Regional Development.

The total number of fish retained by charter operators within the Shark Bay Marine Park decreased from 7,519 in 2008 to a low of 2,297 in 2016 (Figure 6.6). The five most popular species retained by charter operators were pink snapper (*Chrysophrys auratus*), grass emperor (*Lethrinus laticaudis*), redthroat emperor (*Lethrinus miniatus*), rankin cod (*Epinephelus multinotatus*), and stripey snapper (*Lutjanus carponotatus*). The catch of *C. auratus* decreased from ~2,800 in 2008 to ~700 in 2012, and then remained ~1,300 through to 2016 (Figure 6.6). The catch of *L. laticaudis* peaked in 2012 (953) and then declined through to 301 in 2016. Catches of both *E. multinotatus* and *L. carponotatus* were less than 500 for all years in the reporting period. The trend in catch through the reporting period overall and for each species tended to follow that for fishing effort, *i.e*. days fished (Figure 6.6).

The most recent spatial analysis of charter fishing activity in 2007/08 indicated that activity mostly occurred at the northern tip of Dirk Hartog Island, Cape Peron and within South Passage (Marriott *et al.* 2012).

6.4.3 Commercial Fishing

The catch reported by the SBBSF for all species combined in blocks intersecting with the Shark Bay Marine Park, show a catch of approximately 260 tonnes annually between 2008 and 2010, decreasing to approximately 180 tonnes in 2015 (Figure 6.7). The three principal species caught by the SBBSF are sea mullet (*Mugil cephalus*), western sand whiting (*Sillago schomburgkii*) and tailor (*Pomatomus saltatrix*), all of which show similar patterns to the total catch. Recent declines in catches since 2011 are due to reductions in effort throughout the fishery (Fletcher *et al*. 2017).

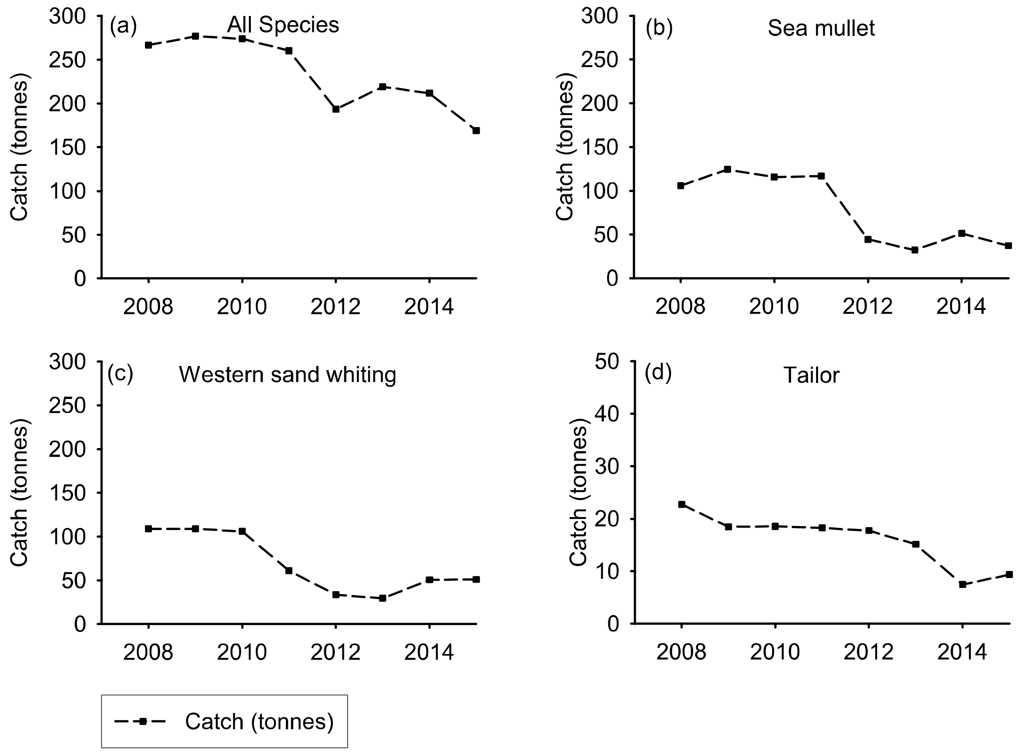

Figure 6.7: Total commercial catch for all species (a), sea mullet (b), western sand whiting (c), and tailor (d) of the Shark Bay Beach Seine and Mesh Net Managed Fishery reporting blocks that overlap with the boundary of the Shark Bay Marine Park. Data reported for this fishery are derived from 60 x 60 nm reporting blocks rather than 10 x 10 nm blocks.

Source: Department of Primary Industries and Regional Development.

6.4.4 Benthic habitat loss – Coral

Hard coral is one of the primary benthic habitats that influence the structure of fish communities within the Shark Bay marine reserves, serving as a food source, predator refuge and recruitment habitat for many species. As such, any changes in the condition of hard coral communities are likely to affect associated fish communities.

For a full assessment of the condition of coral reef communities within the marine reserves, please refer to page 52 of this report.

6.5 Synthesis

Current assessments of fish community condition are based on data collected from 2010 and 2015 only. As such, it is inappropriate to make conclusions on temporal trends at this time and condition is considered stable but with a low assessment confidence. Additionally, the spatial coverage within the park is still limited and is focussed solely on a single habitat (coral reef) that is associated with only a portion of the overall marine reserve fish community. There are however some points of interest from these data that are worthy of note. Species richness appears to be slightly higher within the sanctuary zone (Sandy Point Sanctuary Zone) than at sites in the general use zone. This is likely related to habitat characteristics, with coral cover at the general use zone sites being heavily dominated by a single species from the family Dendrophylliidae, whilst the sanctuary zone site contains a more diverse benthic assemblage. However, the relative abundance of targeted species between the different management zones is currently mixed across the two sampling periods.

The dominance of *L. carponotatus* and *Plectropomus* spp. amongst the observed target species is indicative of the habitat sampled, with this group tending to associate more with high relief and hard coral structure (Light and Jones 1997). This does however highlight the need to extend the monitoring program into seagrass and macroalgal habitats, with a completely different community suite and greater abundance of other target species likely to be recorded in these areas (Heithaus 2004). The development of methods for surveying fish communities in seagrass areas is currently awaiting the completion of a Departmental research project and will be implemented when results are available.

While not inside the reserves, the Bernier and Dorre islands area is considered to be an integral component of the overall Shark Bay ecosystem, and provide an important linkage between the more coral dominated habitats in the western gulf and southern end of the Ningaloo reef (DEC 2008). It is interesting to note the sharp decline in the abundance of corallivorous fishes near Bernier and Dorre islands in 2015. This decline was likely caused by major declines in hard coral cover (the benthic habitat on which this fish group is dependent) in this area following thermal stress in 2011 and 2012 (Moore *et al.* 2012). In comparison, coral cover and associated corallivorous fish remained relatively unchanged within the reserves following this event.

Surveys of finfish communities are typically associated with high levels of variability, particularly in the case of schooling and mobile species or those with strong habitat associations that mean they are not evenly distributed in the area being sampled (Sale and Douglas 1984; Holbrook *et al*. 1994). As such, it often requires a large number of survey events across a significant period of time before temporal patterns of change are clear (see Emslie *et al*. 2008). Variability is further heightened by the complex nature of finfish distributions, making short term comparisons between sites and locations difficult without also considering various biological (e.g. habitat type, complexity; Friedlander and Parrish 1998, Gratwicke and Speight 2005), environmental (e.g. depth, wave exposure, water movement; Letourner 1996; Friedlander *et al*. 2003), anthropogenic (e.g. long-term fishing pressure) and ecological process (e.g. recruitment; Sale *et al*. 1984) characteristics. While the survey design used here does account for many of these factors, this inherent variability will mean that the power to reliably identify long-term trends will be dependent on appropriate sample sizes and increased temporal coverage.
There is currently no quantitative data on recreational fishing effort specifically relating to the marine reserve area. Current assessments of fishing effort are based on estimates at bioregional scales from state-wide surveys and, while these are likely to provide indicative information, they are not directly applicable to the boundaries of the marine reserve area. The Shark Bay marine reserves are located within the Gascoyne Coast fisheries management bioregion, with data suggesting that boat-based recreational fishing effort across this area has been stable across the available survey period (2011/12, 2013/14 and 2015/16). While a broad assumption can be made that the same pattern exists within the marine reserve area, it highlights the need to identify appropriate methods for assessing recreational fishing directly within the state's marine reserves.

Catch data highlights that the main recreationally target species in the Carnarvon/Shark Bay zone of the Gascoyne Coast bioregion are *Chrysophrys auratus* (pink snapper) and *Lethrinus laticaudis* (grass emperor), together contributing approximately 76% of the total catch. While highly mobile, both species are more closely associated with habitats other than coral, further highlighting the need to monitor fishes in a range of habitats. While this information is extrapolated from a wider bioregional dataset, it is broadly assumed that similar patterns may exist within the marine reserve area. However, effort should be placed into developing appropriate methods for quantifying recreational catch of these species directly within the marine reserves to identify spatial and temporal trends.

Both commercial and charter fishing effort have declined from 2008 to 2015 and 2016 respectively, due largely to reductions in fishery effort (Fletcher *et al*. 2017) and charter operators (from 12 to 5) working within the reserves.

References

Abdo D, Coleman G, Ninio R, Osborne K (2004). Surveys of benthic reef communities using underwater video - Long term monitoring of the Great Barrier Reef - Standard Operating Procedure Number 2. Australian Institute of Marine Science, Townsville.

Abdo DA, Bellchambers LM, Evans SN (2012). Turning up the heat: increasing temperature and coral bleaching at the high latitude coral reefs of the Houtman Abrolhos Islands. *PLOS ONE* **7**, e43878.

Abraham JP, Baringer M, Bindoff NL, Boyer T, Cheng LJ, Church JA, Conroy JL, Domingues CM, Fasullo JT, Gilson J, Goni G, Good SA, Gorman JM, Gouretski V, Ishii M, Johnson GC, Kizu S, Lyman JM, MacDonald AM, Minkowycz WJ, Moffitt SE, Palmer MD, Piola AR, Reseghetti F, Schuckmann K, Trenberth KE, Velicogna I, Willis JK (2013). A review of global ocean temperature observations: Implications for ocean heat content estimates and climate change. *Reviews of Geophysics* **51**, 450–483.

Acosta A, Dueñas LF, Pizarro V (2011). Review on hard coral recruitment (Cnidaria: Scleractinia) in Colombia. *Universitas Scientiarum* **16**, 200–218.

Anderson PK (1994). Dugong distribution, the seagrass *Halophila spinulosa*, and thermal environment in winter in deeper waters of eastern Shark Bay, Western Australia. *Wildlife Research* **21**, 381–388.

ANZECC/ARMCANZ (2000). National Water Quality Management Strategy: Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Environment and Conservation Council and the Agriculture and Resource Management Council of Australia and New Zealand, Canberra, Australian Capital Territory.

Baldock J, Bancroft KP, Williams M, Shedrawi G, Field S (2014). Accurately estimating local water temperature from remotely sensed satellite sea surface temperature: A near real-time monitoring tool for marine protected areas. *Ocean & Coastal Management* **96**, 73–81.

Berkelmans R (2009). Bleaching and mortality thresholds: how much is too much? In *Coral Bleaching*. Ecological Studies. Springer, Berlin, Heidelberg. pp. 103–119.

Blakeway DR (2004). Patterns of mortality from natural and anthropogenic influences in Dampier corals: 2004 cyclone and dredging impacts. In: *Corals of the Dampier Harbour: Their survival and reproduction during the dredging programs of 2004*. Mscience, pp 65–76.

Brodie J, Schroeder T, Rohde K, Faithful J, Masters B, Dekker A, Brando V, Maughan M (2010). Dispersal of suspended sediments and nutrients in the Great Barrier Reef lagoon during river-discharge events: conclusions from satellite remote sensing and concurrent flood-plume sampling. *Marine and Freshwater Research* **61**, 651.

Brown E, Cox E, Jokiel P, Rodgers K, Smith W, Tissot B, Coles SL, Hultquist J (2004). Development of benthic sampling methods for the Coral Reef Assessment and Monitoring Program (CRAMP) in Hawai'i. *Pacific Science* **58**, 145–158.

BOM (2017) Tides and sea level: tide gauge metadata and observed monthly sea levels and statistics. Bureau of Meteorology, Australian Government. http://www.bom.gov.au/oceanography/projects/ntc/monthly/.

Burling MC, Pattiaratchi CB, Ivey GN (2003a). The tidal regime of Shark Bay, Western Australia. *Estuarine Coastal and Shelf Science* **57**, 725–735.

Burling MC, Pattiaratchi CB, Ivey GN (2003b). The tidal regime of Shark Bay, Western Australia. *Estuarine, Coastal and Shelf Science* **57**, 725–735.

CALM (1996). Management Plan for the for the Shark Bay Marine Reserves 1996-2006, Management Plan No. 34. Department of Conservation and Land Management, Government of Western Australia. Perth, Western Australia.

CALM (2005). Management Plan for the Ningaloo Marine Park and Muiron Islands Marine Management Area 2005-2015, Management plan No. 52. Department of Conservation and Land Management, Government of Western Australia. Perth, Western Australia.

Chang J, Chung C-C, Gong G-C (1996). Influences of cyclones on chlorophyll a concentration and Synechococcus abundance in a subtropical western Pacific coastal ecosystem*. Marine Ecology Progress Series* **140**, 199–205.

Connell J, Keough M (1985). Disturbance and patch dynamics of subtidal marine animals on hard substrata.

Connell JH (1978). Diversity in tropical rain forests and coral reefs. *Science* **199**, 1302–1310.

Connell SD, Kingsford MJ (1998). Spatial, temporal and habitat-related variation in the abundance of large predatory fish at One Tree Reef, Australia. *Coral Reefs* **1**, 49–57.

CSIRO, BOM (2016). State of the Climate 2016. Commonwealth of Australia.

DEC (2008). Shark Bay World Heritage Property Strategic Plan 2008-2020. Department of Environment and Conservation, Government of Western Australia. Perth, Western Australia.

DoF (2012). A review of the management arrangements and licensing framework for the aquatic tour industry in Western Australia. Department of Fisheries, Government of Western Australia. Perth, Western Australia.

DEH (2006). A guide to The Integrated Marine and Coastal Regionalisation of Australia - version 4.0 June 2006. In *A guide to The Integrated Marine and Coastal Regionalisation of Australia - version 4.0 June 2006*. Department of Environment and Heritage, Commonwealth of Australia.

http://www.environment.gov.au/resource/guide-integrated-marine-and-coastal-regionalisation-australia-version-40-june-2006-imcra.

DiBattista JD, Travers MJ, Moore GI, Evans RD, Newman SJ, Feng M, Moyle SD, Gorton RJ, Saunders T, Berry O (2017). Seascape genomics reveals fine-scale patterns of dispersal for a reef fish along the ecologically divergent coast of Northwestern Australia. *Molecular Ecology* **26**, 6206-6223.

Done TJ (1992). Phase shifts in coral reef communities and their ecological significance. *Hydrobiologia* **247**, 121–132.

DPaW (2014). Department of Parks and Wildlife strategic directions 2014-2017. Department of Parks and Wildlife, Government of Western Australia. Perth, Western Australia.

Duarte CM, Kirkman H (2001). Methods for the assessment of seagrass abundance and depth distribution. In Short FT, Coles RG (eds) *Global seagrass research methods*. Elsevier, pp 141–153.

Dugan JE, Hubbard DM, Page HM, Schimel JP (2011). Marine macrophyte wrack inputs and dissolved nutrients in beach sands. *Estuaries and Coasts* **34,** 839–850.

Eakin CM, Morgan JA, Heron SF, et al (2010). Caribbean corals in crisis: record thermal stress, bleaching, and mortality in 2005. *PLoS ONE* **5,** e13969.

Ellison JC (1999). Impacts of sediment burial on mangroves. *Marine Pollution Bulletin* **37,** 420–426.

Emslie M, Cheal A, Sweatman H, Delean S (2008). Recovery from disturbance of coral and reef fish communities on the Great Barrier Reef, Australia. *Marine Ecology Progress Series* **371**, 177–190.

English S, Wilkinson C, Baker V (1997). Mangrove Ecosystems. In: *Survey Manual for Tropical Marine Resources*. Australian Institute of Marine Science, Townsville, pp 119–196

EPA (1989). Disappearance of dolphins at Monkey Mia: findings of a preliminary investigation. Environmental Protection Agency, Government of Western Australia. Perth, Western Australia.

Eslami-Andargoli L, Dale P, Sipe N, Chaseling J (2009). Mangrove expansion and rainfall patterns in Moreton Bay, Southeast Queensland, Australia. *Estuarine, Coastal and Shelf Science* **85**, 292–298.

Evans K, Bax N, Smith D (2017). Australian state of the environment 2016: marine environment. Department of the Environment, Commonwealth of Australia.

Fabricius KE, De'ath G, Puotinen ML, Done T, Cooper TF, Burgess SC (2008). Disturbance gradients on inshore and offshore coral reefs caused by a severe tropical cyclone. *Limnology and Oceanography* **53**,690–704.

Fletcher W, Mumme M, Webster F (2017). Status reports of the fisheries and aquatic resources of Western Australia 2015/16: the state of the fisheries. Department of Fisheries, Government of Western Australia. Perth, Western Australia.

Fourqurean JW, Duarte CM, Kennedy H, Marba N, Holmer M, Mateo MA, Apostolaki ET, Kendrick GA, Krause-Jensen D, McGlathery KJ, Serrano O (2012). Seagrass ecosystems as a globally significant carbon stock. *Nature Geoscience* **5**, 505–509.

Fraser MW, Kendrick GA, Statton J, Hovey RK, Zavala-Perez A, Walker DI (2014). Extreme climate events lower resilience of foundation seagrass at edge of biogeographical range. *Journal of Ecology* **102**, 1365-2745.

Friedlander AM, Brown EK, Jokiel PL, Smith WR, Rodgers KS (2003). Effects of habitat, wave exposure, and marine protected area status on coral reef fish assemblages in the Hawaiian archipelago. *Coral Reefs* **22**, 291–305.

Friedlander AM, Parrish JD (1998). Habitat characteristics affecting fish assemblages on a Hawaiian coral reef. *Journal of Experimental Marine Biology and Ecology* **224**, 1–30.

Garrabou J, Coma R, Bensoussan N (2009). Mass mortality in northwestern Mediterranean rocky benthic communities: effects of the 2003 heat wave. *Global Change Biology* **15**, 1090–1103.

Gilman EL, Ellison J, Duke NC, Field C (2008). Threats to mangroves from climate change and adaptation options: a review. *Aquatic Botany* **89**, 237–250.

Glynn PW, D’Croz L (1990). Experimental evidence for high temperature stress as the cause of El Nino-coincident coral mortality. *Coral Reefs* **8**,181–191.

Gratwicke B, Speight MR (2005). The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. *Journal of Fish Biology* **66**, 650–667.

Harmelin-Vivien ML (1994). The effects of storms and cyclones on coral reefs: a review. *Journal of Coastal Research* **Special Issue 12**, 211–231.

Heithaus MR (2004) Fish communities of subtropical seagrass meadows and associated habitats in Shark Bay, Western Australia. *Bulletin of Marine Science* **75**, 79–99.

Hetzel Y, Pattiaratchi C, Lowe R, Hofmeister R (2015). Wind and tidal mixing controls on stratification and dense water outflows in a large hypersaline bay. *Journal of Geophysical Research: Oceans* **120,** 6034–6056.

Hoegh-Guldberg O, Bruno JF (2010). The Impact of climate change on the world’s marine ecosystems. *Science* **328,** 1523–1528.

Holbrook SJ, Kingsford MJ, Schmitt RJ, Stephens JS (1994). Spatial and temporal patterns in assemblages of temperate reef fish. *American Zoologist* **34**, 463–475.

Holmes TH, Wilson SK, Travers MJ, Langlois TJ, Evans RD, Moore GI, Douglas RA, Shedrawi G, Harvey ES, Hickey K (2013). A comparison of visual- and stereo-video based fish community assessment methods in tropical and temperate marine waters of Western Australia: Comparison of fish community assessment methods. *Limnology and Oceanography: Methods* **11,** 337–350.

Hughes TP, Baird AH, Dinsdale EA, Moltschaniwskyj NA, Pratchett MS, Tanner JE, Willis BL (1999). Patterns of recruitment and abundance of corals along the Great Barrier Reef. *Nature* **397**, 59–63.

Hughes TP, Kerry JT, Álvarez-Noriega M, et al (2017). Global warming and recurrent mass bleaching of corals. *Nature* **543,** 373–377.

Hughes TP, Rodrigues MJ, Bellwood DR, Ceccarelli D, Hoegh-Guldberg O, McCook L, Moltschaniwskyj N, Pratchett MS, Steneck RS, Willis B (2007). Phase shifts, herbivory, and the resilience of coral reefs to climate change. *Current Biology* **17**, 360–365.

Hughes TP, Tanner JE (2000). Recruitment failure, life histories, and long-term decline of Caribbean corals. *Ecology* **81,** 2250–2263.

IPCC (2014). Climate Change 2014: Sythesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland.

Jennings S, Dulvy N (2005). Reference points and reference directions for size-based indicators of community structure. *ICES Journal of Marine Science* **62**, 397–404.

Jordà G, Marbà N, Duarte CM (2012). Mediterranean seagrass vulnerable to regional climate warming. *Nature Climate Change* **2**, 821–824.

Kingsford MJ, MacDairmid AB (2000). Large gastropods, urchins and crustaceans of subtidal reefs. In Kingsford MJ, Battershill CN (eds) *Studying temperate marine environments: a handbook for ecologists*. Canterbury University Press, Christchurch, pp 115–131.

Koss R, Gilmour P, Wescott G, Bunce A, Miller K (2005). Sea Search: Community-Based Monitoring of Victoria's Marine National Parks and Marine Sanctuaries: Intertidal Rocky Shore Monitoring. 50 pgs.

Letourner Y (1996). Dynamics of fish communities on Reunion fringing reefs, Indian Ocean. I. Patterns of spatial distribution. *Journal Of Experimental Marine Biology And Ecology* **195**, 1–30.

Levitus S (2001). Anthropogenic Warming of Earth's Climate System. *Science* **292**, 267–270.

Light PR, Jones GP (1997). Habitat preference in newly settled coral trout (*Plectropomus leopardus*, Serranidae). *Coral Reefs* **16**, 117–126.

Liu G, Strong AE, Skirving W, Arzayus LF (2006). Overview of NOAA coral reef watch program's near-real time satellite global coral bleaching monitoring activities. In: *Proceedings of the 10th International Coral Reef Symposium*. June.

Lovelock CE, Feller IC, Reef R, Hickey S, Ball MC (2017). Mangrove dieback during fluctuating sea levels. *Scientific Reports* **7**, s41598.

Loya Y (1978). Plotless and transect methods. In: Stoddart D, Johannes R (eds) *Coral reef research methods*. UNESCO, Paris, pp 581–591.

Magurran AE, Baillie SR, Buckland ST, Dick J, Elston DA, Scott EM, Smith RI, Somerfield PJ, Watt AD (2010). Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends in Ecology & Evolution* **25**, 574–582.

Marriott R, Jackson G, Lenanton R, Telfer C, Lai E, Stephenson P, Bruce C, Adams D, Norriss J (2012). Biology and stock status of inshore demersal scalefish indicator species in the Gascoyne Coast Bioregion. Department of Fisheries, Government of Western Australia. Perth, Western Australia. 216 pgs.

Massel SR, Done TJ (1993). Effects of cyclone waves on massive coral assemblages on the Great Barrier Reef: meteorology, hydrodynamics and demography. *Coral Reefs* **12**, 153–166.

McClanahan T, Ateweberhan M, Graham N, Wilson SK, Sebastian CR, Guillaume MMM, Bruggemann JH (2007). Western Indian Ocean coral communities: bleaching responses and susceptibility to extinction. *Marine Ecology Progress Series* **337**, 1–13.

McClanahan TR, Shafir SH (1990). Causes and consequences of sea urchin abundance and diversity in Kenyan coral reef lagoons. *Oecologia* **83**, 362–370.

McComb AJ, Cambridge ML, Kirkman H, Kuo J (1981). The biology of Australian plants. In: Pate JS, McComb AJ (eds) *The biology of Australian plants*. University of Western Australia Press, Nedlands, Western Australia, pp 258–293.

Moore JAY, Bellchambers M, Depczynski MR, Evans RD, Evans SN, Field SN, Friedman KJ, Gilmour JP, Holmes TH, Middlebrook R, Radford BT, Ridgway T, Shedrawi G, Taylor H, Thomson DP, Wilson SK, Alvarez I (2012). Unprecedented mass bleaching and loss of coral across 12° of latitude in Western Australia in 2010-11. *PLoS ONE* **10**, e51807.

Morales CE, Blanco JL, Braun M, Reyes H, Silva N (1996). Chlorophyll-a distribution and associated oceanographic conditions in the upwelling region off northern Chile during the winter and spring 1993. *Deep Sea Research Part I: Oceanographic Research Papers* **43**, 267–289.

Nahas EL, Jackson G, Pattiaratchi CB, Ivey GN (2003). Hydrodynamic modelling of snapper Pagrus auratus egg and larval dispersal in Shark Bay, Western

Australia: reproductive isolation at a fine spatial scale. *Marine Ecology Progress Series* **265**, 213–226.

Nash KL, Graham NAJ (2016). Ecological indicators for coral reef fisheries management. *Fish and Fisheries* **17**, 1029–1054.

Neuheimer AB, Thresher RE, Lyle JM, Semmens JM (2011). Tolerance limit for fish growth exceeded by warming waters. *Nature Climate Change* **1**, 110–113.

Nowicki RJ, Thomas JM, Burkholder DA, Fourqurean JW, Heithaus MR (2017). Predicting seagrass recovery times and their implications following an extreme climate event. *Marine Ecology Progress Series* **567**, 79-93.

Osland MJ, Feher LC, Griffith KT, Cavanaugh KC, Enwright NM, Day RH, Stagg CL, Krauss KW, Howard RJ, Grace JB, Rogers K (2017). Climatic controls on the global distribution, abundance, and species richness of mangrove forests. *Ecological Monographs* **87**, 341–359.

Paling EI, Kobryn HT, Humphreys G (2008). Assessing the extent of mangrove change caused by Cyclone Vance in the eastern Exmouth Gulf, northwestern Australia. *Estuarine, Coastal and Shelf Science* **77**, 603–613.

Pearce A, Lenanton G, Moore J, Feng M, Gaughan D (2011). The “marine heat wave” off Western Australia during the summer of 2010/11. Department of Fisheries, Government of Western Australia. Perth, Western Australia.

Phillips J (2001). Marine macroalgal biodiversity hotspots: why is there high species richness and endemism in southern Australian marine benthic flora? *Biodiversity and Conservation* **10**, 1555–1577.

Playford P (1990). Geology of the Shark Bay area, Western Australia. In Berry P, Bradshaw S, Wilson B (eds) *Research in Shark Bay*. Western Australian Museum. pp 13–32.

Preen AR, Lee Long WJ, Coles RG (1995). Flood and cyclone related loss, and partial recovery, of more than 1000 km2 of seagrass in Hervey Bay, Queensland, Australia. *Aquatic Botany* **52**, 3–17.

Rhein M, Rintoul SM, Aoki S, Campos E, Chambers D, Feely RA, Gulev S, Johnson GC, Josey SA, Kostianoy A, Mauritzen C, Roemmich D, Talley LD, Wang F, Stocker TF, Qin D, Plattner GK, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM (2013). Observations: Ocean. In Stocker TF, Qin D, Plattner GK, et al. (eds) *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp 255–315.

Roberts CM (2002). Marine biodiversity hotspots and conservation priorities for tropical reefs. *Science* **295**, 1280–1284.

Rogers CS (1990). Responses of coral reefs and reef organisms to sedimentation. *Marine Ecology Progress Series* **62**, 185–202.

Rogers CS (1983). Sublethal and lethal effects of sediments applied to common Caribbean Reef corals in the field. *Marine Pollution Bulletin* **14**, 378–382.

Rogers CS, Gilnack M, Fitz HC (1983). Monitoring of coral reefs with linear transects: a study of storm damage. *Journal of Experimental Marine Biology and Ecology* **66**, 285–300.

Ryan K, Hall N, Lai E, Smallwood CB, Taylor SM, Wise BS (2015). State-wide survey of recreational boat-based fishing in Western Australia 2013/14. Department of Fisheries, Government of Western Australia. Perth, Western Australia.

Ryan KL, Hall NG, Lai EK, Smallwood CB, Taylor SM, Wise BS (2017). Statewide survey of boat-based recreational fishing in Western Australia 2015/16. Department of Primary Industries and Regional Development, Government of Western Australia. Perth, Western Australia.

Ryan KL, Wise BS, Hall NG, Pollock KH, Sulin EH, Gaughan DJ (2013). An integrated system to survey boat-based recreational fishing in Western Australia 2011/12. Department of Fisheries, Government of Western Australia. Perth, Western Australia.

Saenger P (2002). Mangrove ecology, silviculture and conservation. Kluwer Academic Publishers, The Netherlands. 363 pgs.

Sale PF, Doherty PJ, Eckert GJ, Douglas WA, Ferrell DJ (1984). Large scale spatial and temporal variation in recruitment to fish populations on coral reefs. *Oecologia* **64**, 191–198.

Sale PF, Douglas WA (1984). Temporal variability in the community structure of fish on coral patch reefs and the relation of community structure to reef structure. *Ecology* **65**, 409–422.

Scheltinga DM, Counihan R, Moss A, Cox M, Bennet J (2004). Users’ guide to estuarine, coastal and marine indicators for regional NRM monitoring. Prepared for the Department of Environment and Heritage, the Monitoring and Evaluation Working Committee and the Intergovernment Coastal Advisory Group by the Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management, Brisbane, Queensland.

Smallwood CB, Gaughan DJ (2013). Aerial surveys of shore-based recreational fishing in Carnarvon and Shark Bay: June to August 2012. Department of Fisheries, Government of Western Australia. Perth, Western Australia.

Smith TJ, Duke NC (1987). Physical determinants of inter-estuary variation in mangrove species richness around the tropical coastline of Australia. *Journal of Biogeography* **14**, 9-19.

Sokolov S, Rintoul SR (2007). On the relationship between fronts of the Antarctic Circumpolar Current and surface chlorophyll concentrations in the Southern Ocean. *Journal of Geophysical Research: Oceans* **112**, 2156–2202.

Sorte CJB, Williams SL, Carlton JT (2010). Marine range shifts and species introductions: comparative spread rates and community impacts: Range shifts and non-native species introductions. *Global Ecology and Biogeography* **19**, 303–316.

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, Alvarez I (2013). Dynamic stability of coral reefs on the West Australian coast. *PLoS ONE* **8**, e69863.

Stoddart JA (1990). Analysis of water quality in Shark Bay and Coral Bay: August - October 1989. *Landnote* **1/90**, 1–11.

Tanner JE (1995). Competition between scleractinian corals and macroalgae: An experimental investigation of coral growth, survival and reproduction. *Journal of Experimental Marine Biology and Ecology* **190**, 151–168.

Thomas L, Kennington WJ, Evans RD, Kendrick GA, Stat M (2017). Restricted gene flow and local adaptation highlight the vulnerability of high-latitude reefs to rapid environmental change. *Global Change Biology* **23**, 2197–2205.

Thomson JA, Burkholder DA, Heithaus MR, Fourqurean JW, Fraser JW, Statton J, Kendrick GA (2015). Extreme temperatures, foundation species, and abrupt ecosystem change: an example from an iconic seagrass ecosystem. *Global Change Biology* **21**, 1463–1474.

Tittensor DP, Mora C, Jetz W, Lotze HK, Ricard D, Berghe EV, Worm B (2010). Global patterns and predictors of marine biodiversity across taxa. *Nature* **466**, 1098–1101.

Tourism Research Australia (2015). Tourism profile for Shark Bay, Western Australia. http://www.tra.gov.au/research/Regional-overview.html.

Underwood AJ (2000). Experimental ecology of rocky intertidal habitats: what are we learning? *Journal of Experimental Marine Biology and Ecology* **250**, 51–76.

Wakeford M, Done TJ, Johnson CR (2007). Decadal trends in a coral community and evidence of changed disturbance regime. *Coral Reefs* **27**, 1–13.

Walker DI (1985). Correlations between salinity and growth of the seagrass *Amphibolis antarctica* in Shark Bay, Western Australia, using a new method for measuring production rate. *Aquatic Botany* **23**, 13–26.

Walker DI, Kendrick GA, McComb AJ (2006). Decline and Recovery of Seagrass Ecosystems-The Dynamics of Change. In Larkum AWD, Orth RJ, Duarte CM (eds) *Seagrasses: Biology, Ecology, and Conservation*. Springer, Dordrecht, The Netherlands, pp 553–565.

Walker DI, Kendrick GA, McComb AJ (1988). The distribution of seagrass species in Shark Bay, Western Australia, with notes on their ecology. *Aquatic Botany* **30**, 305–317.

Wilson SK, Depczynski M, Fisher R, Holmes TH, O'Leary RA, Tinkler P (2010). Habitat associations of juvenile fish at Ningaloo Reef, Western Australia: the importance of coral and algae. *PLoS ONE* **5**, e15185.

Wilson SK, Fisher R, Pratchett MS, Graham NAJ, Dulvy NK, Turner RA, Cakacaka A, Polunin NVC, Rushton SP (2008). Exploitation and habitat degradation as agents of change within coral reef fish communities. *Global Change Biology* **14**, 2796–2809.

Wise BS, Telfer CF, Lai EKM, Hall NG, Jackson G (2012). Long-term monitoring of boat-based recreational fishing in Shark Bay, Western Australia: providing scientific advice for sustainable management in a World Heritage Area. *Marine and Freshwater Research* **63**, 1129.