Department of **Biodiversity, Conservation and Attractions**

GOVERNMENT OF WESTERN AUSTRALIA

Ecological monitoring in the Ningaloo marine reserves 2017

Marine Monitoring Program Report 1

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Inset top to bottom: Mangrove Bay. Photo - Jamie Campbell/DBCA; Acropora thicket. Photo - DBCA; Spangled emperor. Photo - Suzanne Long/DBCA.

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Summary

This report presents a synthesis of ecological monitoring within the Ningaloo marine reserves up to the end of 2016. 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 Exmouth 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 four of the park's six ecological/physical value Key Performance Indicators (KPI's) are presented. The monitoring of turtles and coastal biological communities does not currently occur as a part of the program coordinated through MSP. Those ecological/physical values presented here are:

- Water quality
- Finfish communities
- Coral reef communities
- Mangrove communities

The main risk to ecological values within 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), and indirect impacts through the degradation of essential habitats (e.g. changes in fish communities as a result of losses in live coral and reef complexity). The condition of corals at shallow water monitoring sites within the reserves has declined since 2010. Declines have been most severe in the eastern sector, whilst the southern sector has suffered less dramatic but continued losses. The northern sector in comparison has remained relatively stable. The primary cause of the coral loss has been thermal stress caused by increasing sea water temperatures, with anomalously high summer temperatures between 2011 and 2013 causing a majority of the loss.

The abundance of targeted fishes have been stable across the limited monitoring period. However, there is some concern that the abundance of the heavily fished spangled emperor (*Lethrinus nebulosus*) is currently low in relation to historical observations. There have been widespread declines in the abundance of corallivorous fishes at shallow water monitoring sites throughout the park, most likely driven by the loss of coral described above.

The spatial extent of mangroves within Mangrove Bay increased between 2006 and 2014. However, recent observations indicate signs of declining condition, associated with sedimentation from a flooding event in 2014 and declining sea level caused by the regional influence of the Southern Oscillation Index cycle.

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 (Department of Parks and Wildlife, 2014).

WA currently (as of 2017) 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 instances 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 Battershill, 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 that ensures 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 last monitoring survey. The frequency of sampling is also assessed in relation to the natural variability associated with different ecological values.

1.2 Ningaloo Marine Reserves

The Ningaloo marine reserves (NMR), comprising Ningaloo Marine Park and the Muiron Islands Marine Management Area, are located in State waters adjacent to WA's North West Cape, approximately 1,200km north of Perth (Figure 1.1). The reserves do not represent a discrete ecosystem, but instead incorporate shallow coastal reef communities that are connected to varying extents with several marine bioregions recognised in the Pilbara and Gascoyne regions (Department of the Environment and Heritage, 2006). Ningaloo Marine Park (263,343ha) was created in 1987 and in 2004 was re-zoned and extended southwards to include the full extent of Ningaloo Reef. The Muiron Islands Marine Management Area (28,616ha) was gazetted in 2004 and extends northwards from Ningaloo Marine Park (Department of Conservation and Land Management, 2005).

While supporting a relatively small resident population of ~2500, the Ningaloo area is an iconic tourist and recreation destination with more than 200,000 visitors to the region annually (Gascoyne Development Commission, 2012). Areas of particular growth in recent years have been nature-based tourism, such as manta ray, cetacean and whale shark tours, and coastal camping along the west coast of North

West Cape. The annual number of passengers on whale shark viewing tours, for example, has increased from about 4,000 in the mid-1990s to about 20,000 by 2015 (Department of Biodiversity, Conservation and Attractions, unpublished data). Ningaloo is also a focus for recreational fishing, scuba diving and snorkelling and access for these activities has improved over the last two decades with improved roads and boat launching facilities at some locations.

The physical/ecological values identified in the current (2005-2015) Ningaloo marine reserves management plan, including those listed as KPIs, are:

- Geomorphology.
- Sediment quality.
- Water quality (**KPI**).
- Coral reef communities (**KPI**).
- Filter-feeding communities.
- Intertidal reef communities.
- Soft-sediment communities.
- Macroalgal & seagrass communities.
- Mangrove communities (**KPI**).
- Coastal biological communities (**KPI**).
- Seabirds, shorebirds & migratory waders.
- Finfish (**KPI**).
- Invertebrates.
- Sharks and rays.
- Whale sharks.
- Manta rays.
- Whales and dolphins.
- Turtles (**KPI**).
- Dugong.

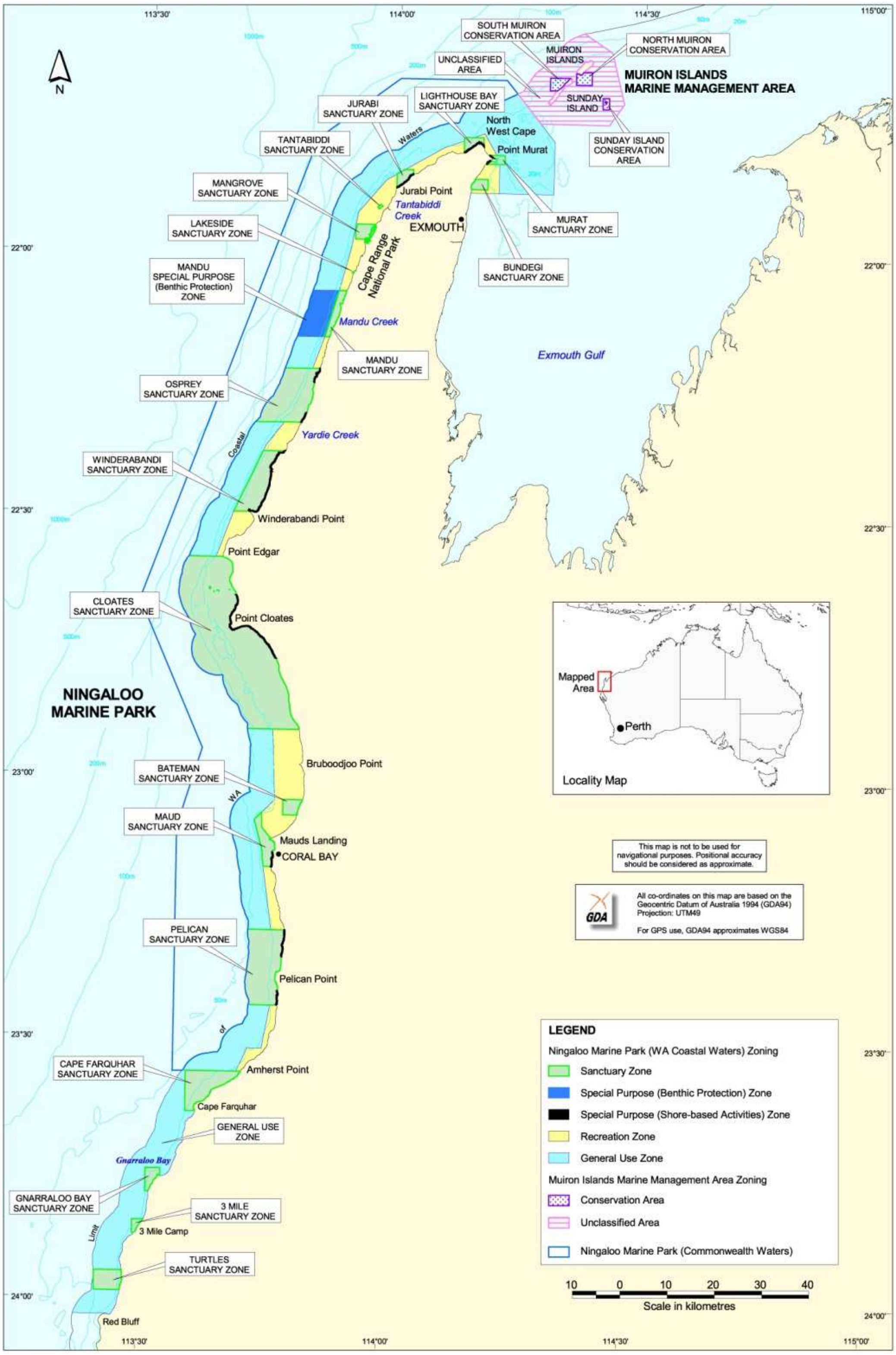

Figure 1.1: Map of the location and management zoning of the Ningaloo marine reserves, comprising Ningaloo Marine Park and Muiron Islands Marine Management Area. The boundary of the Ningaloo Marine Park (Commonwealth waters) is also marked.

1.3 Scope of this report

This report provides a summary of ecological marine monitoring undertaken at the Ningaloo marine reserves up to 2017, under Science Project Plan 2012/008. 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 Ningaloo 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 Ningaloo 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 Ningaloo marine reserves.

The data presented here have been primarily collected by MSP, Exmouth District staff and collaborators, but also includes analyses of data obtained by agreement from the Department of Primary Industries and Regional Development (DPIRD), and data from the Bureau of Meteorology (BoM), CSIRO, AIMS 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, coral communities and mangrove communities. Additional monitoring information relating to the Ningaloo 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

- With the exception of increasing seawater temperature, water quality is considered to be in overall good condition due to the relatively small local population and the lack of terrigenous sources of pollution. Low rainfall, a lack of riverine discharge and the absence of intensive agriculture on North West Cape limit broad impacts on water quality, and existing pressures are likely to be localised and associated with the few coastal population centres.
- There has been a slow increasing trend in seawater temperature in the eastern, northern and southern sectors across the last 32 years. 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.
- Small scale sampling at Coral Bay suggests that pathogen levels along the coastal margin of Bills Bay may continue to be slightly elevated, but are unlikely to pose a risk to human health or influence biological communities within the marine reserves.

2.2 Indicator Summary

	Trend	Confidence
Condition		
Seawater Temperature	Increasing	High
Pathogens (Coral Bay)	Stable	Low
Pressure		
Climate change	Increasing	High
Terrestrial Runoff	Stable	Low

2.3 Condition Indicators

Condition indicators are measures of water quality ‘state’ relative to pressures acting within 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 within the Ningaloo marine reserves and presented in this report are:

- Seawater temperature.
- 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, causing, for example, coral bleaching, 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). 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). Nitrogen (N) and phosphorous (P) are the two relevant 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. Given the lack of significant agriculture that utilises nitrogen and phosphorous enriched fertilisers along the Ningaloo coast, the most likely source of anthropogenic inputs into the marine reserves are from sewage in settled areas. As such, pathogen and nutrient concentrations are only likely to be relevant condition indicators close to coastal settlements like Coral Bay. Whilst pathogens are currently being assessed, a program to appropriately monitor nutrient concentrations is yet to be developed.

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 like upwelling, currents (Morales *et al*., 1996; Sokolov and Rintoul, 2007) or storm activity (Chang *et al*., 1996) and by anthropogenic inputs such as riverine discharge or dredging (Brodie *et al*., 2010). Appropriate methods for monitoring chlorophyll-a at a scale relevant to the marine reserves are still being developed, and are not 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 50km^2 virtual stations (http://coralreefwatch.noaa.gov/satellite/vs/) in conjunction with *in situ* temperature loggers (Table 2.1). Analyses are based on southern, northern and eastern sectors of the Ningaloo marine reserves (Table 2.1, Figure 2.1), which were identified according to a natural separation of the physical oceanographic patterns of the region (Taylor and Pearce, 1999; Woo *et al*., 2006b; Taebi *et al*., 2012).

Table 2.1: Location of NOAA 50km^2 virtual stations and *in situ* temperature loggers used to derive mIST time series for the eastern, northern and southern sectors of the Ningaloo marine reserves

Sector	Virtual Station	Logger Location
Eastern	North West Cape	Bundegi
Northern	North West Cape	Tantabiddi
Southern	Coral Bay	Coral Bay

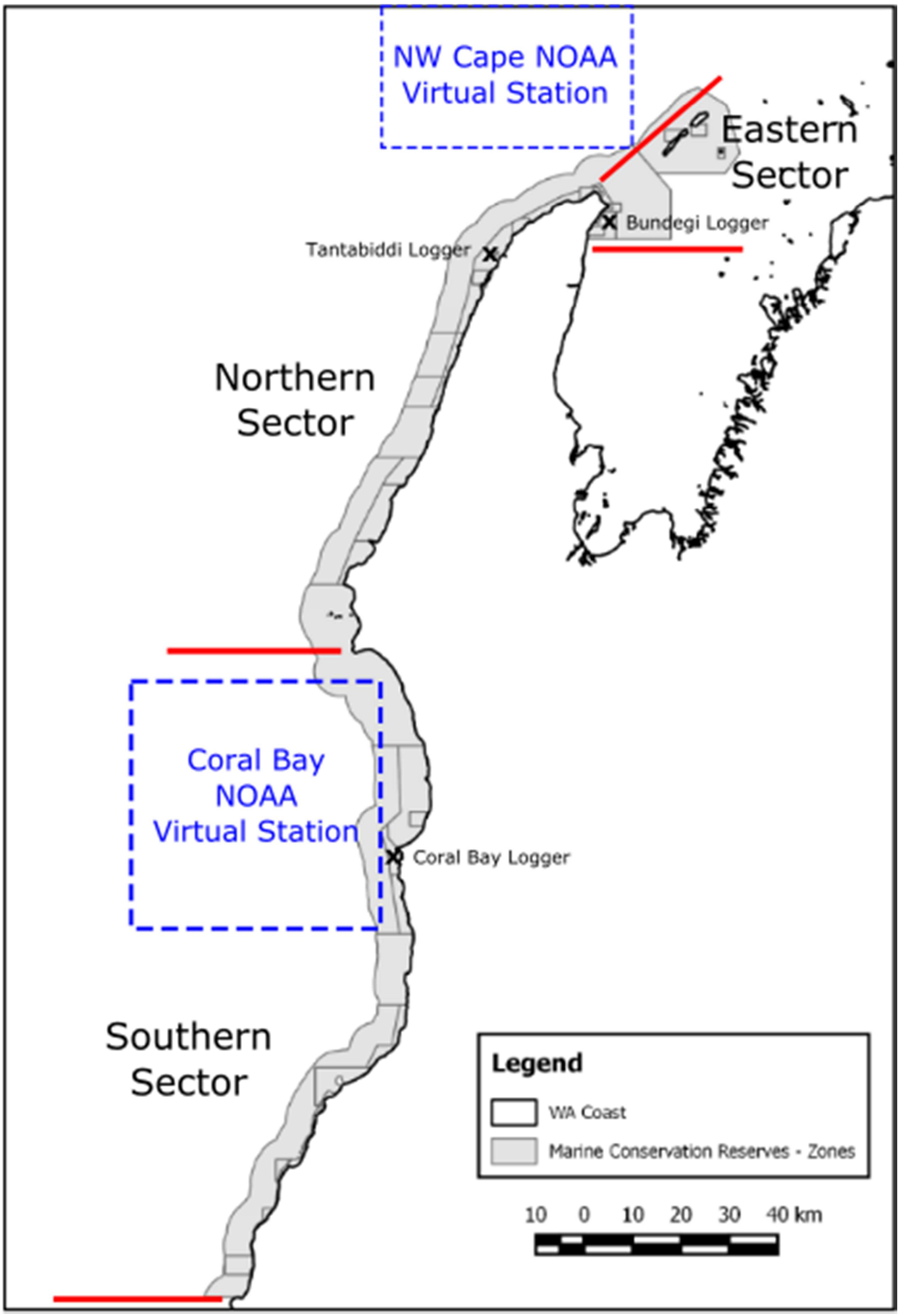

Figure 2.1: The southern, northern and eastern sectors of the Ningaloo marine reserves identified in this report and the positions of *in situ* temperature loggers and NOAA 50km2 sea surface temperature virtual stations.

All sectors show medium-term oscillations (5-10 year cycles) in seawater temperature resulting from regional climatic cycles associated with the Southern Oscillation Index (Figure 2.2). All three sectors also exhibit a slowly increasing significant trend in seawater temperature from 1985 to 2017, amounting to a change of approximately 0.8-1.0°C over this 32 year period (Figure 2.2; seasonally adjusted Mann-Kendall; eastern sector T=0.08, 2-sided p<0.01; northern sector T=0.0841, 2-sided p<0.01; southern sector T=0.0886, 2-sided p<0.01).

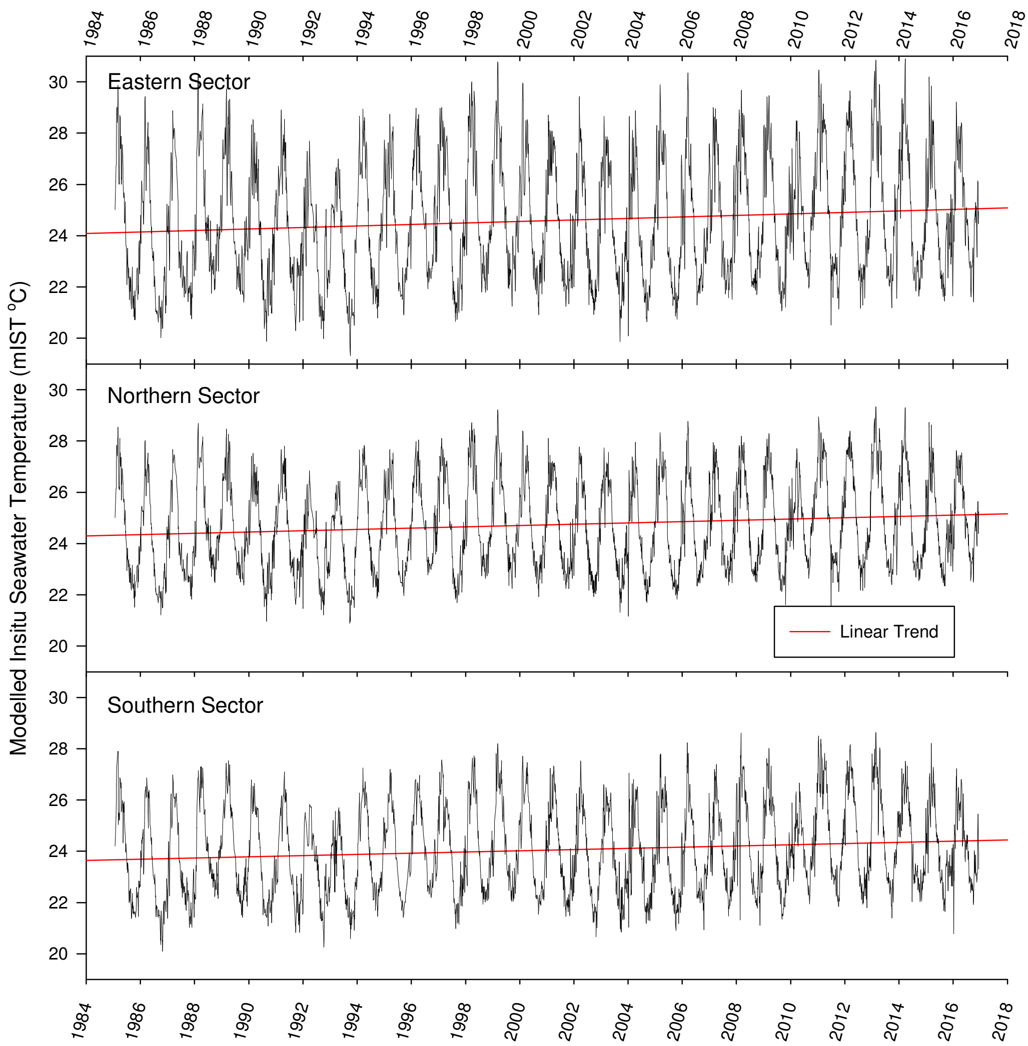

Figure 2.2: Mean seawater temperature for the eastern, northern and southern sectors of the Ningaloo marine reserves, 1985-2017. Mean seawater temperature is calculated using modelled *in situ* seawater temperature (mIST) averaged twice weekly across nocturnal periods (n=104 per year). Red lines indicate significant trends based on seasonally adjusted Mann-Kendall trend analyses.

Source: NOAA (2016).

Instances of anomalously high seawater temperature (identified as exceeding two standard deviations of the long-term mean) have occurred in all three sectors since July 2010 (Figure 2.3). These events occurred from October 2010 to January 2011 and from December 2012 to February 2013, with some variation between sectors. Anomalously high seawater temperatures also occurred briefly in the northern sector during October 2015.

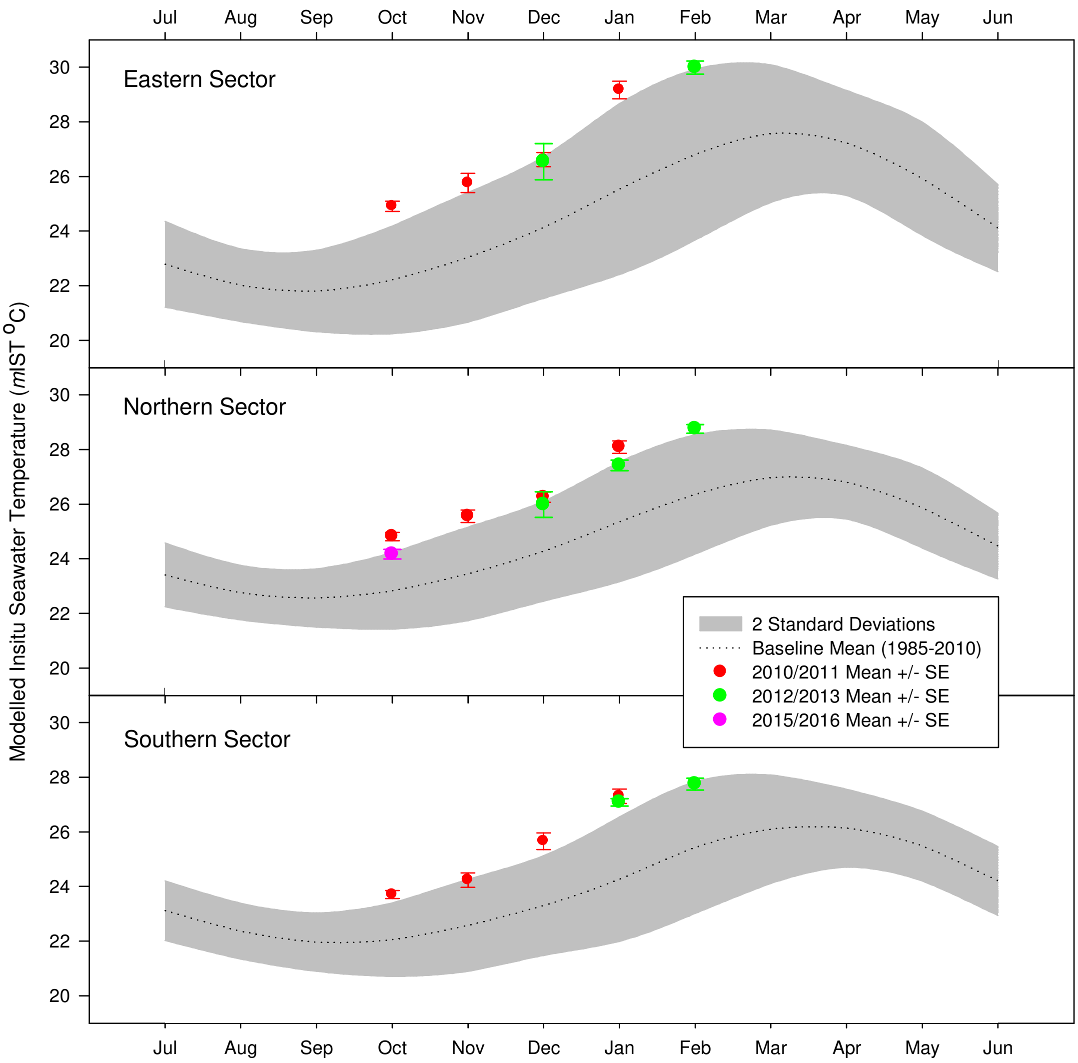

Figure 2.3: Anomalous mean (±1 SE) monthly seawater temperatures for the eastern, northern and southern sectors of the Ningaloo marine reserves, July 2010 – June 2016. Values are calculated using modelled *in situ* seawater temperature (mIST) averaged twice weekly across nocturnal periods (n=8 per month). 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.

Source: NOAA (2016).

2.3.2 Pathogens

Pathogens are measured as the concentration of *Enterococci* spp. and are used as an indicator of contamination by human waste. No assessment has been made of pathogen concentrations across the whole marine reserves as this risk is considered to be low. However, sampling has been conducted at Coral Bay due to the elevated risk associated with old septic systems close to the reserve shoreline (Stoddart, 1990; Simpson and Field, 1995). Water samples were collected at one metre depth from three replicate shoreline sites at Coral Bay and three replicate reference sites south of Moncks Head. In September of 1989 and 1995 median concentrations of

Enterococci spp. at the Coral Bay sites were approximately 8 and 6 cfu.100mL^{-1} respectively (Figure 2.4). The most recent samples collected in September 2016 recorded a median of 7 cfu.100mL^{-1}. In comparison, all reference site samples collected across 1995 and 2016 returned median concentrations of 0 cfu. mL^{-1} and 0.5 cfu.mL^{-1} respectively. Note that no reference site samples were collected in 1989.

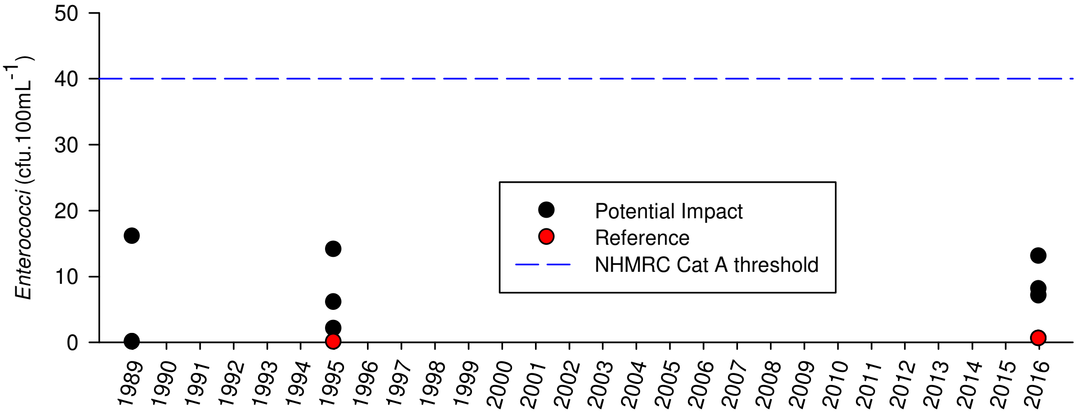

Figure 2.4: *Enterococci* concentrations in nearshore coastal waters at Coral Bay, Ningaloo marine reserves (1989-2016). Coral Bay potential impact site; all occasions n=3 and Moncks Head reference site; 1989 n = 0; 1995 n=2; 2016 n=3.

Source: Department of Biodiversity, Conservation and Attractions (2016); Stoddard (1990); Simpson and Field (1995).

2.4 Pressure Indicators

Pressure indicators are considered to be reliable measures of the primary causes of change in water quality within the marine reserves. Those indicators relevant to the Ningaloo 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 Ningaloo 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 probably before (IPCC, 2014). While greenhouse gas emissions are clearly a 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 0.8-1.0°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 Ningaloo marine reserves. The most appropriate indicator of climate change pressure acting on water temperature in the Ningaloo marine reserves is still being developed.

Sewage input is likely a combination of seepage from septic tanks and terrestrial runoff. No permanent waterways enter the Ningaloo marine reserves and the transport of terrestrial contaminants into coastal waters primarily occurs from below ground seepage or runoff associated with episodic rain events. This pressure is considered minor in the context of the whole marine reserves, but could be significant in settled areas like Coral Bay. Methods for assessing the amount of sewage in vulnerable coastal storage are not currently incorporated into the monitoring program and as such are not presented in this report.

2.4.1 Terrestrial Runoff

Rainfall recorded at Exmouth is presented as a coarse indicator of terrestrial runoff into the Ningaloo marine reserves. While it is not a direct measure of terrestrial runoff and there can be significant variation in rainfall between Exmouth and the broader Ningaloo region, reliable rainfall data is only available for this location and it is currently the best available surrogate for above and below ground runoff/seepage. Data from 1968 to 2016 indicates that rainfall for the last four years was above the long-term mean of ~280mm, with 2016 (290mm) just above average. Mann Kendall trend analysis indicate that there is no discernible long-term increasing or decreasing trend (Figure 2.5). For 2015/2016, monthly rainfall totals (Figure 2.6) were either on or below the baseline mean (1968-2010) for all months except June which was well above the baseline mean. However, this value was still below the upper control limit of two standard deviations suggesting that it was not an anomalous event.

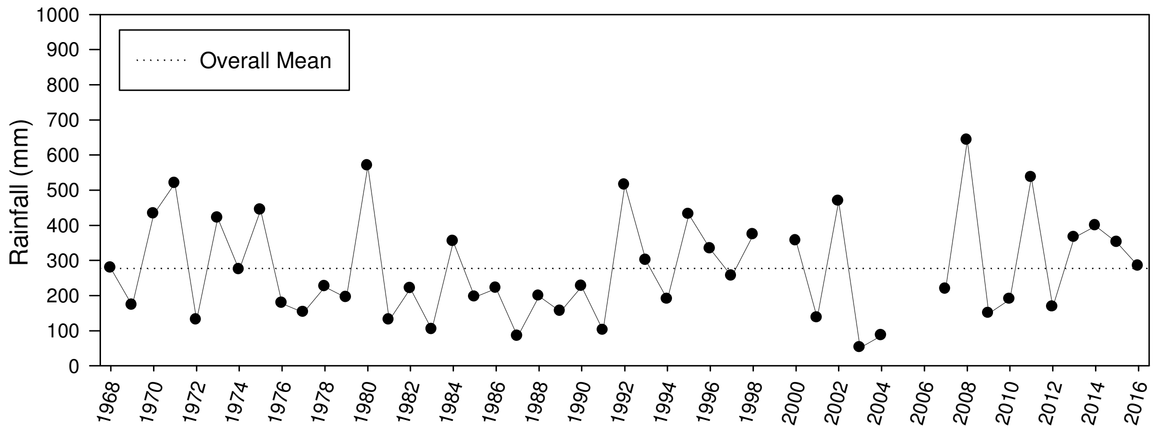

Figure 2.5: Total annual rainfall for Exmouth, 1968-2016. Black dashed line indicates overall mean rainfall for 1986-2010.

Source: Bureau of Meteorology (2016).

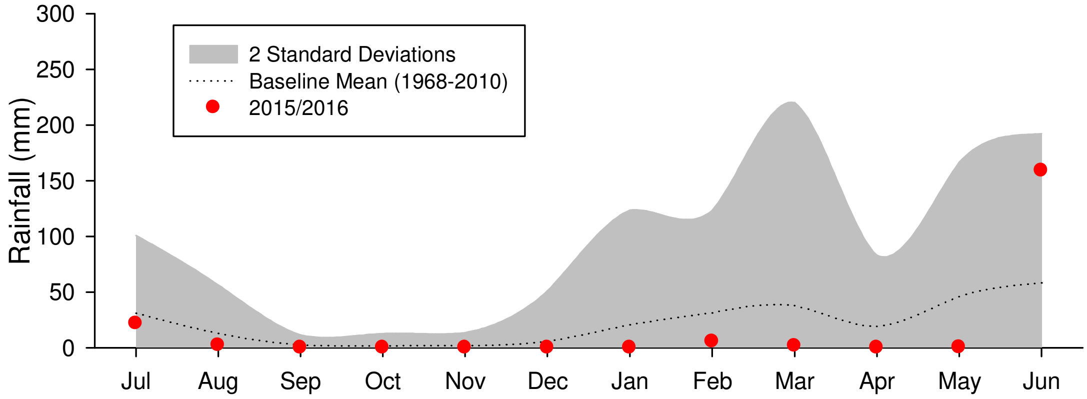

Figure 2.6: Total monthly rainfall for Exmouth, 2015-2016. Dashed line indicates baseline rainfall mean for 1968- 2010. Grey shading indicates two standard deviations from the mean, with values outside of this considered anomalous.

Source: Bureau of Meteorology (2016).

2.5 Synthesis

The low number of coastal settlements, lack of intensive agriculture and river systems and low rainfall (<300mm mean annual rainfall) indicate that overall water quality within the Ningaloo marine reserves (excluding water temperature) is likely to be in a relatively good condition, with pressures limited to localised impacts. While there have been no long term temporal trends in rainfall across the 48 years of sampling, it is notable that five of the past six years have recorded rainfall at or above the long term mean.

The long-term trends in seawater temperature across the three sectors of the reserves are relatively consistent, with evidence of an increasing trend in all sectors (eastern, northern, and southern). This trend of increasing seawater temperature in the Ningaloo marine reserves over the past 30 years is consistent with those recorded elsewhere in Western Australia (Abdo *et al.*, 2012), nationally (CSIRO and Bureau of Monitoring, 2016; Hoegh-Guldberg and Ridgway, 2016) and internationally (Abraham *et al.*, 2013). Of particular note was the abnormally high water temperature that occurred across all three sectors in the summer months of 2010/2011 and 2012/2013. These events were part of regional seawater warming that occurred along much of the Western Australian coastline during these years, resulting from 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 finer-scale, 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.

Coral Bay has a history of nutrient and pathogens entering Bills Bay from septic tanks (Simpson and Field, 1995). While effort has been placed into reducing this pressure by improving infrastructure, small scale sampling conducted in 2016 indicates that slightly elevated pathogen levels may still occur in Bills Bay. While these levels are above those collected from reference locations at Moncks Head, it should be noted that they fall well below standard water quality threshold limits and are unlikely to be a threat to human health (National Health and Medical Research Council, 2010). However, more detailed assessments are warranted to more accurately quantify the extent of this pressure and its potential impact on ecological and social values of the marine reserves.

3 Finfish Communities (KPI)

3.1 Key Points

- There are no clear trends in the abundance of targeted fishes at shallow water (lagoonal) sites across the limited sampling period (2011-2016). While surveys conducted in 1999 and 2000 indicate abundances may have been higher at this time, methodological differences make direct comparisons with more recent data difficult.
- While other trophic groups have remained stable, the abundance of corallivorous fishes has declined at numerous sites, most noticeably at Bundegi, North Muiron and Pelican. This indicates that changes to coral condition are currently the primary driver of change in fish community structure in the reserves.
- The number of temporal sampling points in deep water locations (two) is currently too low to allow for meaningful interpretation of patterns in this fish community.
- A declining trend in the abundance of fish recruits over the past five years is most likely associated with La Niña/El Niño cycles and is not currently of management concern.
- Current assessments of recreational fishing effort are based on estimates of effort and catch from boat-based recreational fishing in the Gascoyne Bioregion, a scale far larger than the marine reserves, and incorporate just two sampling points. As such, they are considered to have low levels of confidence.
- 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/medium levels of confidence, this will improve as the program is refined and spatial and temporal resolution improves.

3.2 Indicator Summary

	Trend	**Confidence**
Condition		
Target abundance – shallow water	Stable	Low
Target abundance – deep water	Increasing	Low
Community composition – shallow water	Stable	Medium
Species richness – shallow water	Stable	Medium
Species richness – deep water	Increasing	Low
Fish recruitment	Decreasing	Low
Pressure		
Recreational fishing	Stable	Low
Charter fishing	Decreasing	Medium
Benthic habitat loss - coral	Increasing	High

3.3 Condition Indicators

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

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

The abundance of target species is indicative of pressure exerted directly by fishing on particular species, generally within higher trophic groups (referred to as 'top-down' effects). 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 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 Ningaloo marine reserves management plan (Department of Conservation and Land Management, 2005). Fish recruitment is indicative of future supply and recovery potential within fish communities and represents a significant source of natural variation in the abundance of species and structure of fish communities. Understanding population variance caused by recruitment is important when seeking to understand any impacts caused by pressures on target species and broader fish communities.

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 derive this information 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 (ie fishing and habitat change), and the management strategies used to conserve biodiversity within the marine reserves (i.e. spatial zoning, catch and size limits). Sites are spread along the length of the marine reserves and evenly between management zones where fishing is permitted (general use and recreational zones) and prohibited (sanctuary zones and marine conservation areas). Monitoring occurs at 42 shallow water lagoonal (2-12m depth) and 10 deep water slope (40-60m depth) sites, with specific locations chosen based on habitat criteria. Shallow surveys use stereo-DOV on six replicate 50 x 5 m belt transects per site and are completed between autumn and spring (April-September). While monitoring has occurred since 2010, spatial coverage prior to 2014 was patchy. As such, whole reserve assessments currently only incorporate a subset of sites where data has been consistently collected across the sampling period. Historical data collected in 2006/2007 at a limited number of sites, but using the same sampling area and method are also incorporated into site specific assessments for Mandu and Osprey. For assessments of community composition, data collected in 1999/2000 at the same sites and using the same sampling area, but an alternate method (Underwater Visual Census), were incorporated (Westera, 2003). The stereo-DOV and Underwater Visual Census methods have previously been found to produce comparable results when examined at a functional level (Holmes *et al.*, 2013).

Deep surveys use stereo-BRUV with 10-18 replicate 60 minute camera drops per site. One sanctuary and one general use site are positioned at the Osprey and Cloates locations, whilst one sanctuary and two general use sites are positioned at the Mandu and Pelican locations. While stereo-DOV estimates the real abundance of fishes within a defined area, stereo-BRUV provides a relative measure of abundance, referred to as Max N.

Fish recruit surveys are conducted using UVC at sites dominated by coral (the primary recruitment habitat for most reef associated species) and macroalgae (the primary recruitment habitat for targeted lethrinid species) within the shallow lagoon (Wilson *et al.*, 2010a). Nine replicate 30 x 1m belt transects are surveyed at each site. The timing of the sampling occurs at the end of the major recruitment period for the region (end of summer) and all fish determined to be less than one year old based on known growth rates and colour patterns for each species, are classified as recruits.

3.3.1 Target Species Abundance – Shallow Water

Target species are identified based on DPIRD phone diary surveys and boat ramp inspections within the reserves. Only those species present in the depth range being sampled and considered to be in the top twenty most targeted species within the whole reserves are included. The main demersal target species in shallow (2-12 m) waters of the Ningaloo marine reserves are spangled emperor (*Lethrinus nebulosus*), rankin cod (*Epinephelus multinotatus*), chinaman rockcod (*Epinephelus rivulatus*), yellow-spotted rockcod (*Epinephelus areolatus*), orange-spotted rockcod (*Epinephelus coioides*), tomato rockcod (*Cephalopholis sonnerati*), coral trout (*Plectropomus* spp.), mangrove jack (*Lutjanus argentimaculatus*) and baldchin grouper (*Choerodon rubescens*) (Ryan et al., 2013, 2015, DPIRD *unpublished data*, *pers comm*). While the baldchin grouper (*C. rubescens*) is not within the twenty most targeted species within the reserves, it has been added to this list as it is known to be targeted at a number of localised areas where it is present. Although all of these species are present within the Ningaloo marine reserves, they are generally only present in moderate numbers in the context of the overall fish community. 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 species targeted by fishing in the Ningaloo marine reserves is the spangled emperor (*L. nebulosus*) (Ryan et al., 2013; Ryan et al., 2015). This species constitutes 86% of all targeted fish recorded in shallow water sampling across years (Figure 3.1).

Data indicates that the mean abundance of targeted fish species at shallow water monitoring sites has generally been low across the limited sampling period (2014-2016), ranging from 0.16 – 1.2 fish per 250m^2 (Figure 3.1). While mean abundances were higher in 2015 (~1-1.1 fish per 250m^2), these values were associated with higher levels of spatial variability. This was a result of aggregations of *L. nebulosus* on a single transect at each of the Osprey recreation zone and Mandu sanctuary zone sites. Mean abundances were similar in both management zones, except in 2016 when higher abundances were recorded in sanctuary zones than in recreational zones (~1.1 ± 0.4 SE and ~0.11 ± 0.08 SE, respectively). When examined at the site level, the incorporation of earlier patchy sampling indicates that mean abundances have remained relatively consistent at the Mandu and Osprey sites since 2006, and at the Maud sites since 2010 (Figure 3.2). Although the data suggests a declining trend in target species abundance at the Bundegi sanctuary site between 2010 and 2013, this is driven by an aggregation of *L. nebulosus* on a single transect in 2010.

This interpretation is based on limited sampling events and patchy spatial coverage prior to 2014. Hence, caution should be used when interpreting the data at this stage and the confidence associated with any assessment is rated as low. This confidence is likely to increase over time as the spatial and temporal resolution of the data increases.

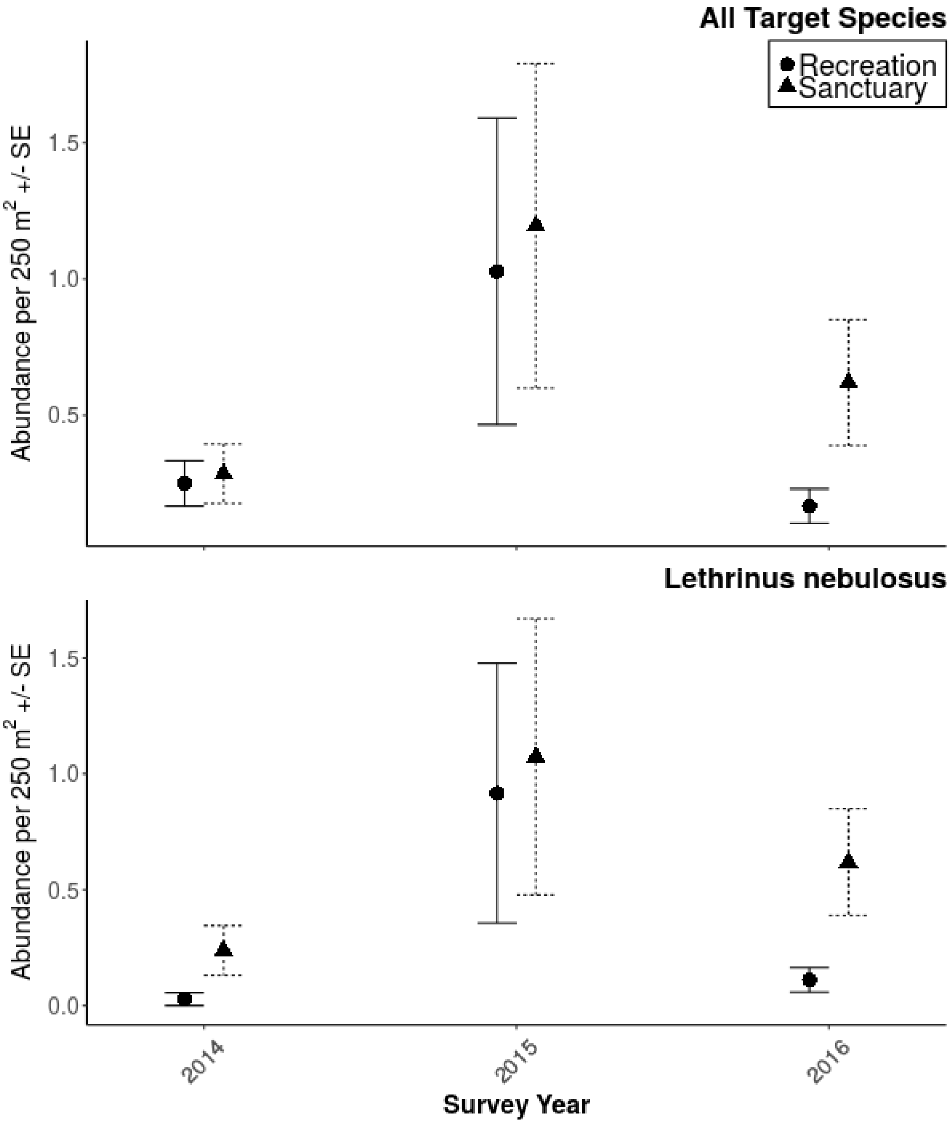

Figure 3.1: Mean abundance (±1 SE) of all targeted fish species and *Lethrinus nebulosus* in the Ningaloo marine reserves, based at shallow water back-reef/leeward sites in recreational (open to fishing) and sanctuary (closed to fishing) zones, 2014-2016. Data is based on sites sampled at Bundegi, Tantabiddi, Mandu, Osprey and Maud locations only. Recreational zone n = 36, sanctuary zone n = 42.

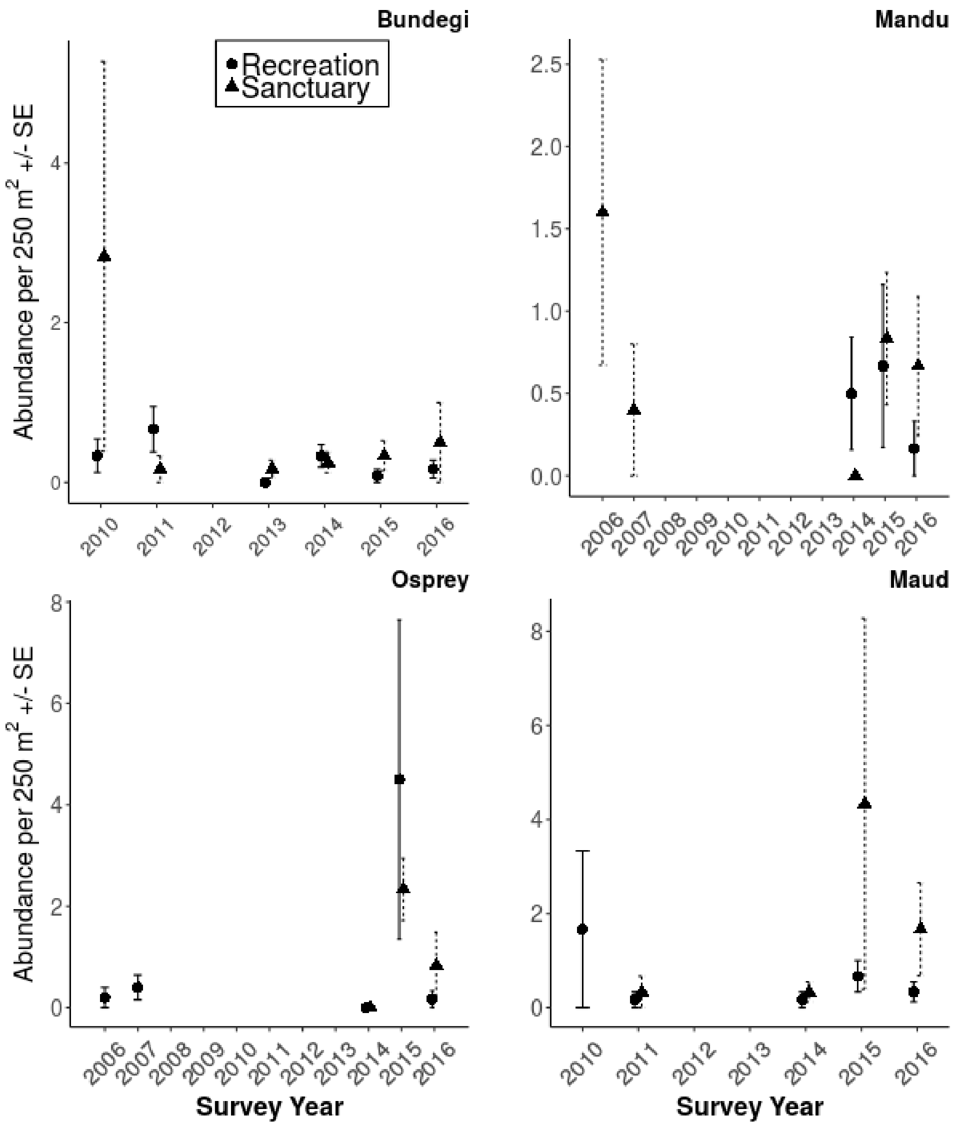

Figure 3.2: Mean abundance (±1 SE) of all targeted fish species at Bundegi, Mandu, Osprey, and Maud locations in the Ningaloo marine reserves, based at shallow water back-reef/leeward sites in recreational (open to fishing) and sanctuary (closed to fishing) zones, 2006-2015. Bundegi (recreational zone n = 12, sanctuary zone n = 12); Mandu, Osprey and Maud (recreational zone n = 6, sanctuary zone n = 6).

Source: Department of Biodiversity, Conservation and Attractions (2010-2015), Ben Fitzpatrick (2006-2007).

3.3.2 Target Species Abundance – Deep Water

Target species are identified based on DPIRD phone diary surveys and boat ramp inspections within the reserves. Only those species present in the depth range being sampled and considered to be in the top twenty most targeted species within the whole reserves are included. The main demersal target species in deep (40-60m) water the Ningaloo marine reserves are spangled emperor (*Lethrinus nebulosus*), red-throat emperor (*Lethrinus miniatus*), robinson's seabream (*Gymnocranius gradoculis*), red emperor (*Lutjanus sebae*), goldband snapper (*Pristipomoides multidens*), rankin cod (*Epinephelus multinotatus*), yellow-spotted rockcod (*Epinephelus areolatus*), orange-spotted rockcod (*Epinephelus coioides*), tomato rockcod (*Cephalopholis sonnerati*), coral trout (*Plectropomus* spp.) and coronation trout (*Variola* spp.) (Ryan et al., 2013; Ryan et al., 2015, DPIRD *unpublished data*, *pers comm*). This list does not incorporate pelagic target species, which are not sampled as well as demersal species using DBCA fish monitoring methods.

The abundance of targeted fish species in deep (40-60m) water increased across all zones and locations from 2013 to 2015 (Figure 3.3). These changes were most apparent in general use and sanctuary zones at Pelican and in sanctuary zones at Osprey (Figure 3.4). Changes were mostly attributed to increases in the numbers of *Lethrinus nebulosus*, *Lethrinus miniatus*, *Epinephelus multinotatus* and *Variola* spp. recorded in 2015.

This interpretation is based on only two sampling events and for this reason the confidence associated with any trends is currently rated as low. This confidence will increase over time as more data are collected.

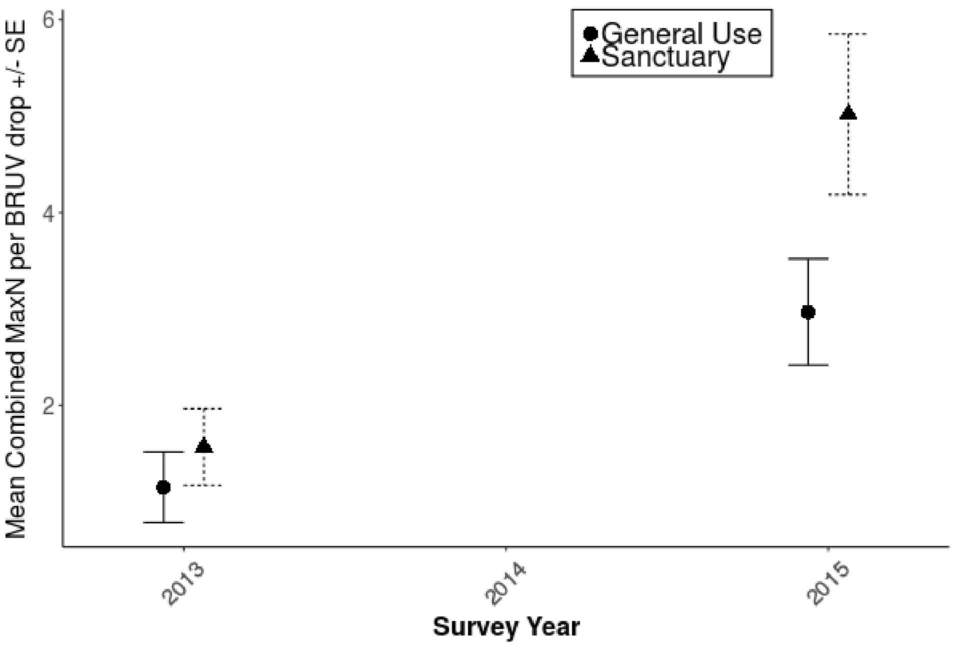

Figure 3.3: Mean relative abundance (±1 SE) of targeted fish species in the Ningaloo marine reserves, based at randomly allocated deeper water (40-60m) sites in general use (open to fishing) and sanctuary (closed to fishing) zones, 2013-2015. General use zone n = 63-93, sanctuary zone n = 57-60.

Source: Department of Biodiversity, Conservation and Attractions, UWA, CSIRO.

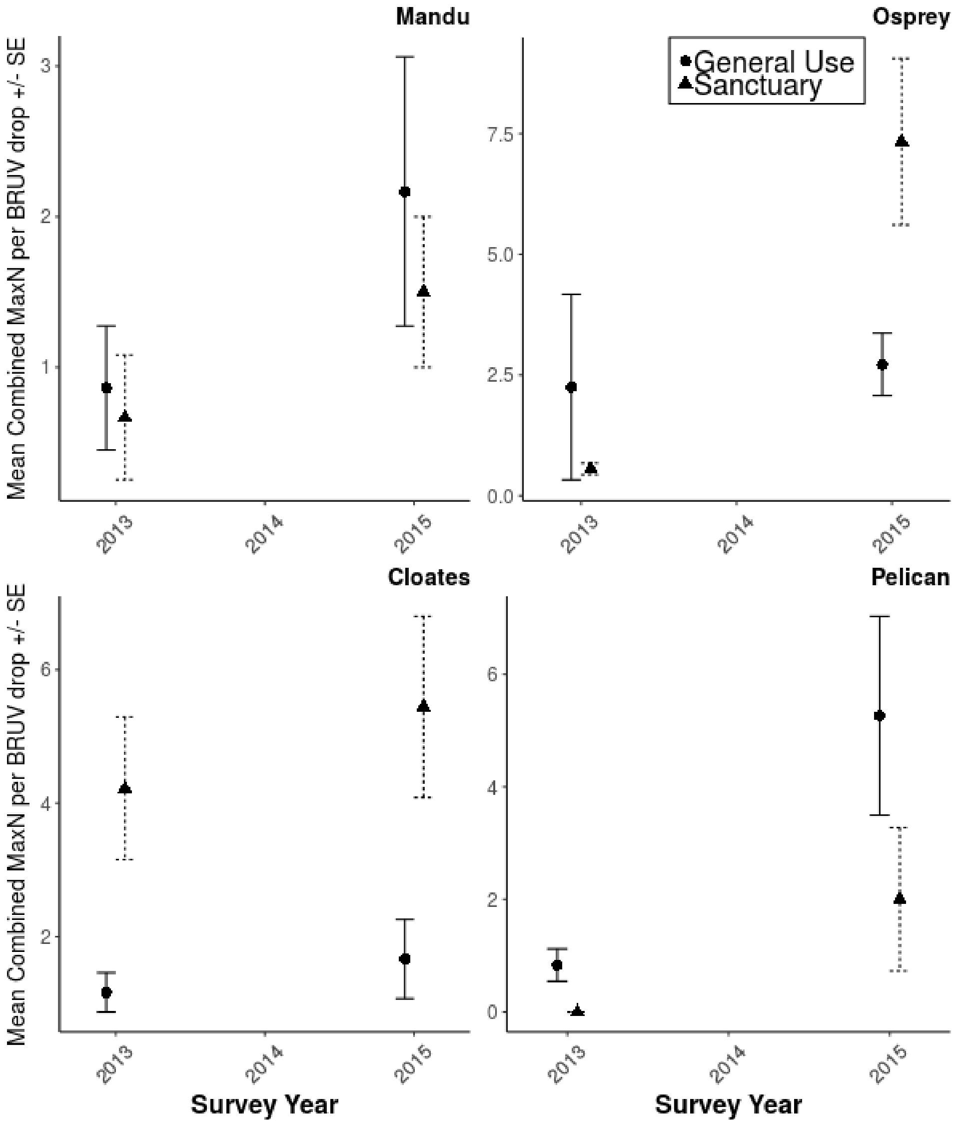

Figure 3.4: Mean relative abundance (±1 SE) of targeted fish species at Mandu, Osprey, Cloates and Pelican locations in the Ningaloo marine reserves, based at randomly allocated deeper water (40-60m) sites in general use (open to fishing) and sanctuary (closed to fishing) zones, 2013-2015. General use zone (Mandu n = 18-29, Osprey n = 16-18, Cloates n = 12-18, Pelican n = 15-30), sanctuary zone (Mandu n = 10-12, Osprey n = 18, Cloates n = 18, Pelican n = 10-12).

Source: Department of Biodiversity, Conservation and Attractions, UWA, CSIRO.

3.3.3 Community Composition – Shallow Water

The abundance of piscivores and large herbivores at shallow sites has remained relatively stable across the survey years both inside and outside of sanctuary zones (Figure 3.5). However, the presence of large variance around many of the mean values makes it difficult to identify clear trends in these data at this time. The abundance of mobile invertivores was much lower in 2014 than all other survey years (mean ~8 fish $250m^{-2}$ as opposed to ~13-22 fish $250m^{-2}$) and this change was primarily because of changes to the abundance of yellow-strip goatfish (*Mulloidichthys flavolineatus*), two-lined monocle bream (*Scolopsis bilineata*), cigar wrasse (*Cheilio inermis*) and clown coris (*Coris aygula*), none of which are fishing target species.

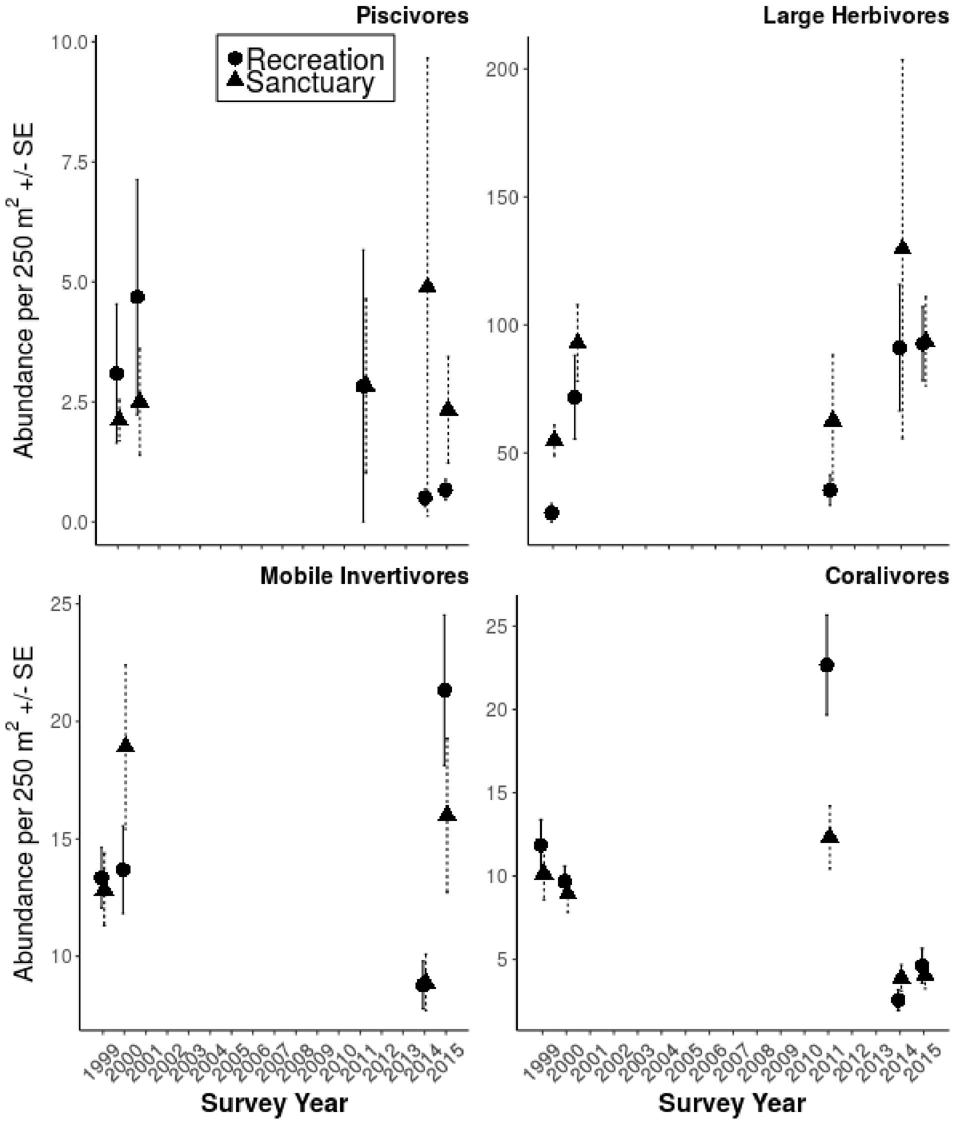

Figure 3.5: Mean abundance (±1 SE) of piscivores, large herbivores, mobile invertivores and corallivores in the Ningaloo marine reserves, based at shallow water back-reef/leeward sites in recreation (open to fishing) and sanctuary (closed to fishing) zones, 1999-2015. Due to limited spatial coverage in 1999-2000 surveys, analysis is based on sites at Mandu, Osprey and Maud locations only. Recreational zone n = 18-23, sanctuary zone n = 24.

Source: Department of Biodiversity, Conservation and Attractions (2011-2015), Mark Westera (1999-2000).

There has been a major decline in the abundance of corallivores both inside and outside of sanctuary zones throughout the park from between 1999 and 2015 (Figure 3.5). This decline occurred in all management zones during or soon after 2011, and was most obvious at Bundegi, North Muiron, Mandu and Pelican (Figure 3.6). The most dramatic change was recorded at Bundegi, where the abundance of corallivores declined from ~12 fish 250m^{-2} in 2010 to less than 1 fish 250m^{-2} from

2011 onwards. While monitoring timeframes are relatively short at Bundegi, North Muiron and Pelican, the Mandu data extends across a 16 years and demonstrates a decline over a longer time period.

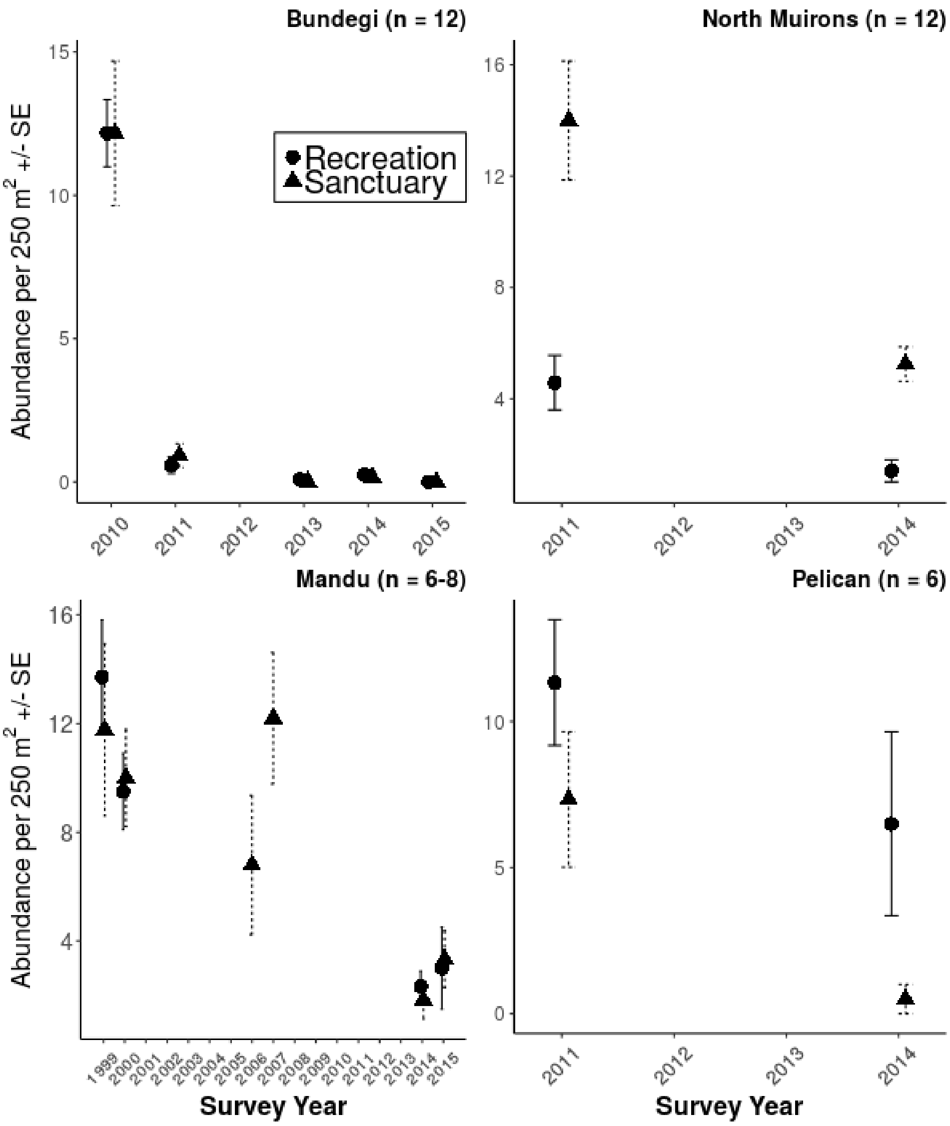

Figure 3.6: Mean abundance (±1 SE) of corallivorous fish at Bundegi, North Muiron, Mandu and Pelican locations in the Ningaloo marine reserves, based at shallow water back-reef/leeward sites in recreational (open to fishing) and sanctuary (closed to fishing) zones, 1999-2015.

Source: Department of Biodiversity, Conservation and Attractions (2010-2015), Ben Fitzpatrick (2006-2007), Mark Westera (1999-2000).

3.3.4 Species Richness – Shallow Water

The species richness of fishes at shallow water monitoring sites was stable between 2010 and 2014, at ~26-29 species per 250m^2 (Figure 3.7). However, richness declined following 2014 in both recreational and sanctuary zones, to ~21-24 species per 250m^2 in 2016. Examination at the site level indicates that this pattern is driven by the Bundegi location, where species richness has declined from ~27-30 species per 250m^2 in 2011 to ~18 species per 250m^2 in 2016. All other locations have remained relatively stable across the sampling period.

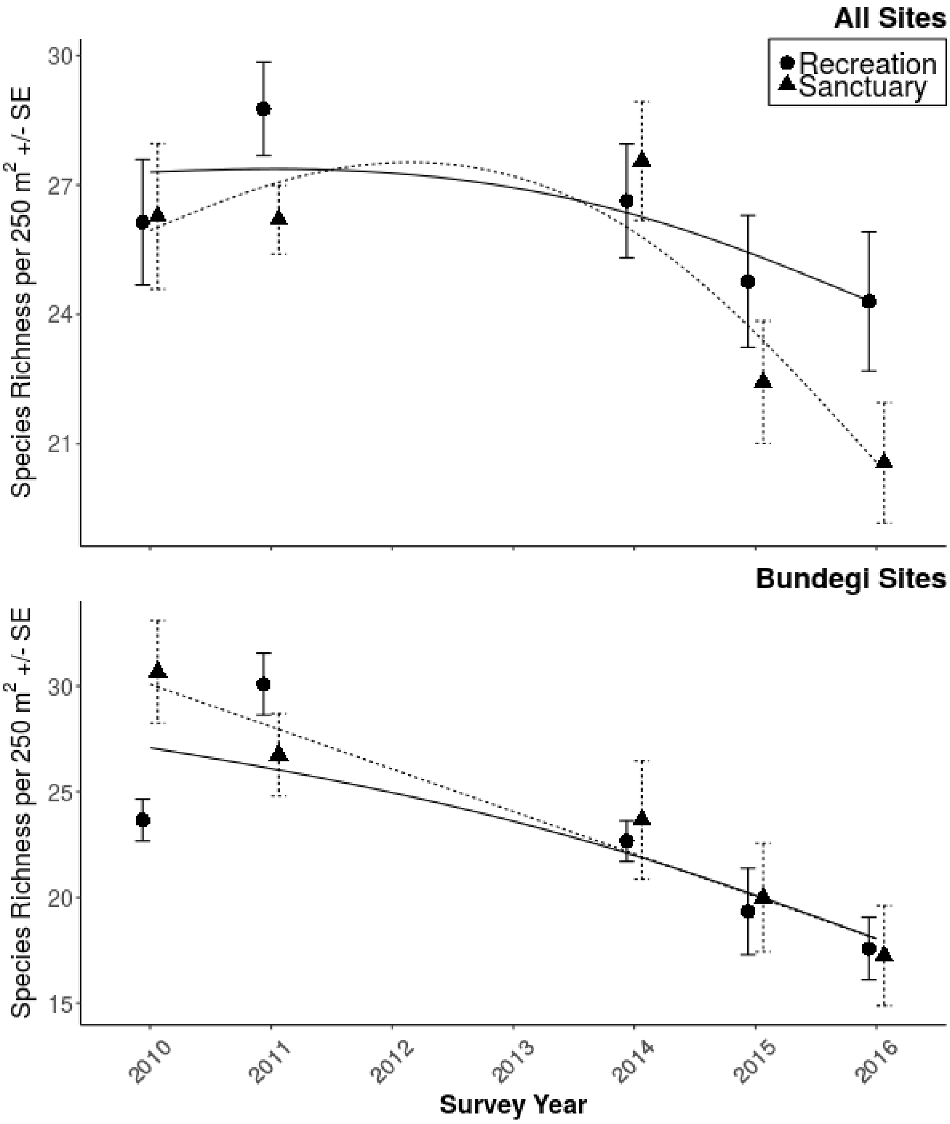

Figure 3.7: Mean species richness (±1 SE) of fishes at all sites and at Bundegi sites only in the Ningaloo marine reserves at shallow water back-reef/leeward locations in recreational (open to fishing) and sanctuary (closed to fishing) zones, 2010-2016. Richness values do not include cryptic species, which are not well sampled using stereo-DOV. Data is based on sites sampled at Bundegi, Tantabiddi, Mandu, Osprey and Maud locations only. All sites (recreational zone *n* = 36, sanctuary zone *n* = 42), Bundegi (recreational zone *n* = 12, sanctuary zone *n* = 12. Full (recreational zone) and dashed (sanctuary zone) lines are fitted Generalized Additive Models indicating data trends.

3.3.5 Species Richness – Deep Water

Species richness of fishes at deep water sites was notably higher in 2015 compared to 2013 (Figure 3.8) and this difference was consistent across management zones.

This interpretation is based on only two sampling events and for this reason the confidence associated with any trends is currently rated as low. This confidence will increase over time as more data are collected.

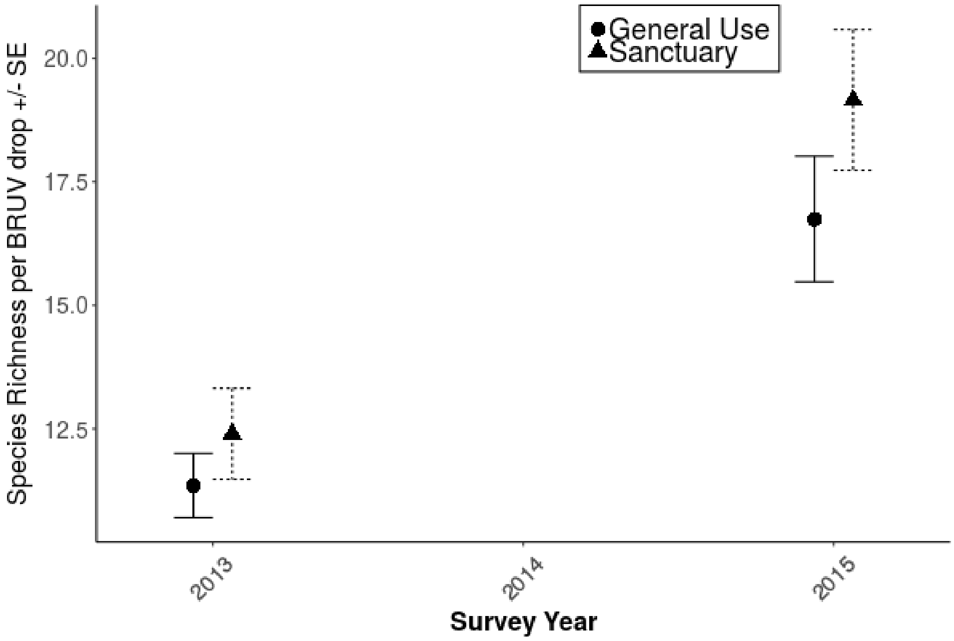

Figure 3.8**:** Mean species richness (±1 SE) of fishes at deep (40-60m) water sites in the Ningaloo marine reserves, based at randomly allocated deep water (40-60m) sites in general use (open to fishing) and sanctuary (closed to fishing) zones, 2013-2015. General use zone n = 63-93, sanctuary zone n = 57-60.

Source: Department of Biodiversity, Conservation and Attractions, UWA, CSIRO.

3.3.6 Fish Recruitment

The abundance of fish recruits increased from a low in 2010 to a peak in 2011, before following a steady decline to another low in 2016 (Figure 3.9). Patterns were consistent across management zones and habitats (coral associates and macroalgal associated lethrinids). The only exception to this trend was a spike in the abundance of lethrinid recruits in recreation zones in 2014, caused by higher abundances at a small number of sites. This assessment is made with a medium level of confidence due to the moderate levels of variance present and continuous sampling across a seven year period.

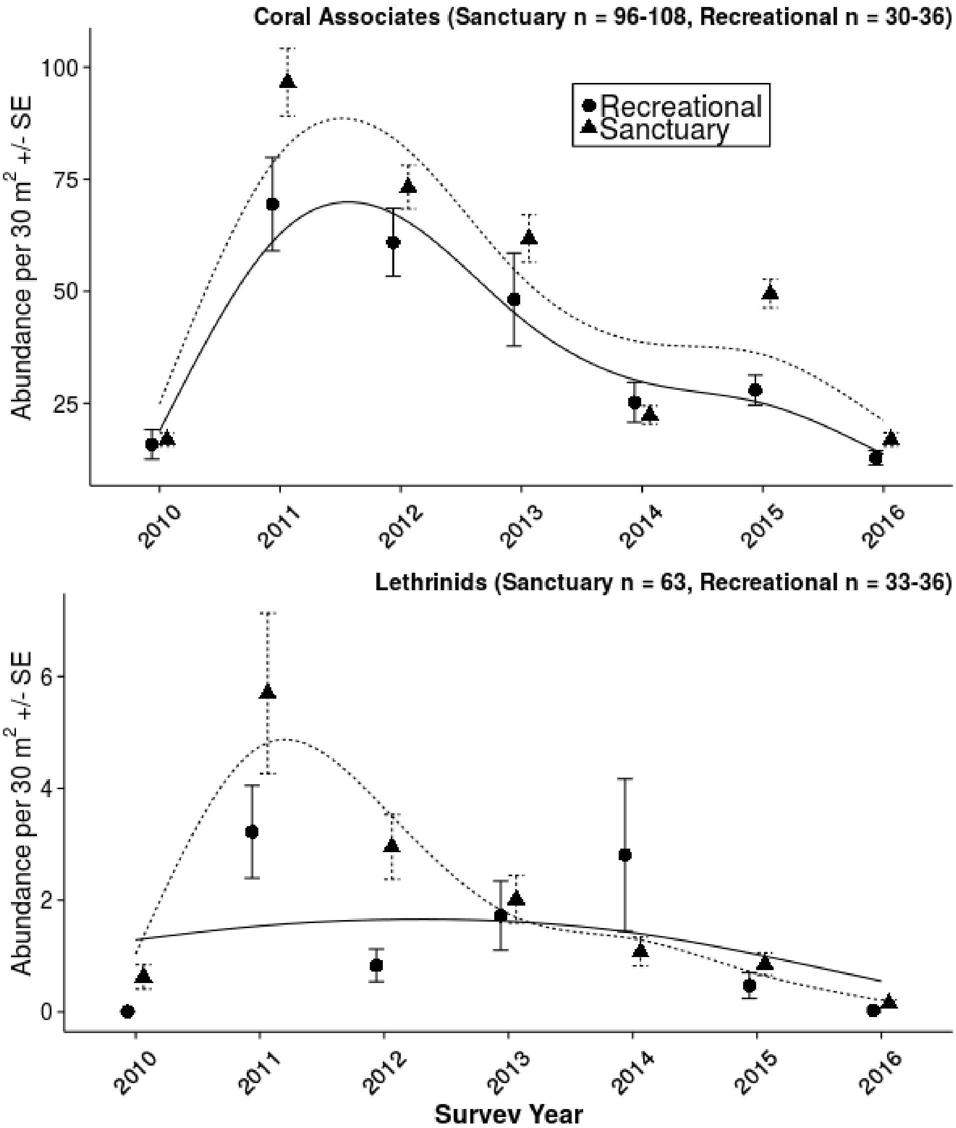

Figure 3.9**:** Mean abundance (±1 SE) of juvenile lethrinids (emperor) in macroalgal habitat and coral associates (pomacentrids, labrids, scarids, chaetodontids and acanthurids) on back-reef coral habitat in recreational (open to fishing) and sanctuary (closed to fishing) zones of the Ningaloo marine reserves, 2010-2016. Full (recreational zone) and dashed (sanctuary zone) lines are fitted Generalized Additive Models indicating data trends.

Source: Department of Biodiversity, Conservation and Attractions, AIMS, ANU.

3.4 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 Ningaloo marine reserves and presented in this report are:

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

Commercial fishing has not been permitted between Point Maud and Tantabiddi from the coast to the edge of the Exclusive Economic Zone for 30 years. Therefore recreational fishing is the primary fishing activity occurring within the Ningaloo marine reserves (Fletcher *et al.*, 2017). While information specific to the marine reserve area is not currently available, state-wide surveys conducted by DPIRD provide a general perspective of trends in fishing activity in the Gascoyne Coast bioregion, and catch composition in the Ningaloo zone (Ryan *et al.*, 2013; 2015) (Figure 3.10). Recreational fishing activity is assessed through phone-diary surveys of fishers sampled from Recreational Boat Fishing Licence (RBFL) holders to provide estimates of ‘boat-based fishing effort’ and ‘catch’ of targeted species (Ryan *et al.*, 2013; 2015). Estimates 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 Beckley, 2012; 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.

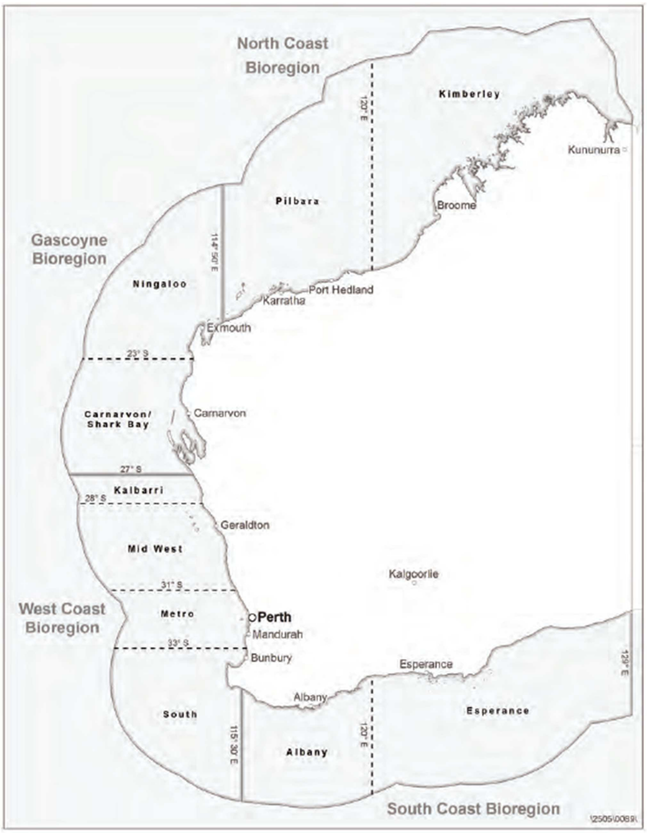

Figure 3.10: Map of Western Australian coastline, highlighting the geographic range of the Gascoyne Coast bioregion (incorporating both the Ningaloo and Shark Bay marine reserves) and Ningaloo zone (incorporating parts of the Ningaloo marine reserves) used to make regional assessments of recreational fishing effort (Ryan *et al.* 2015).

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

Coral reefs and macroalgae are considered the primary benthic habitats important for finfish communities in the reserves. Both are considered key habitats for adult and juvenile fishes. While information on the condition of coral reef communities is included within this report, DBCA does not currently have a standardised monitoring program for macroalgal communities within the reserves, and as such no information is currently available for this habitat.

3.4.1 Recreational Fishing

Boat-based recreational fishing effort within the Gascoyne Coast bioregion remained relatively stable between 2011/12 (61,616 ± 3,895 SE fishing events) and 2013/14 (56,334 ± 3,849 SE fishing events) (Ryan *et al*. 2015). Trends in recreational fishing prior to the survey period are largely unknown. However, oral history indicates that pressure may have markedly increased along the west coast of the Ningaloo marine reserves in the 1970s and 1980s, when infrastructure was upgraded and visitation to the region increased (Fowles, 2007).

Recent surveys indicate the most common species retained by recreational fishing activity in the Ningaloo zone of the Gascoyne Coast bioregion (Figure 3.10) are (in order of total catch retained): spangled emperor (*Lethrinus nebulosus*), chinaman rockcod (*Epinephelus rivulatus*), goldband snapper (*Pristipomoides multidens*), redthroat emperor (*Lethrinus miniatus*), red emperor (*Lutjanus sebae*), spanish mackerel (*Scomberomorous commersoni*) and rankin cod (*Epinephelus multinotatus*) (Ryan *et al.,* 2013; 2015). Of these, *Lethrinus nebulosus* and *Epinephelus rivulatus* make up the highest proportion of the catch, while the catch of *Pristipomoides multidens* within the marine reserves is likely to be significantly lower than reported due to their preferred depth range being largely outside of the reserve area. Grass emperor (*Lethrinus laticaudis*) and spanish mackerel (*Scomberomorous commersoni*) are also reported as having high catch numbers within the Ningaloo zone, but these are likely to come from outside of the marine reserves in the Exmouth Gulf or are not well sampled under current condition assessments, respectively.

The most recent (2006) spatial analysis of fishing activity indicated that most recreational fishing occurred close to coastal access points, with the regions close to Coral Bay, Bundegi and Tantabiddi boat ramps having the highest effort. However, significant pressure is also present around coastal camps at Winderabandi Point, Lefroy Bay, 14 Mile and Gnarraloo Bay (Smallwood *et al*., 2011). The most recent assessments of recreational fishing compliance with management zones in 2006 place boat-based non-compliance at 8-12% and shore-based non-compliance at 2-4% (Smallwood and Beckley, 2012).

3.4.2 Charter Fishing

Charter fishing effort (assessed as the annual number of days fished) in the marine reserves increased from 79 in 2009 to a peak of 482 in 2012, before declining again to 317 in 2016 (Figure 3.11).

The total number of fish retained by charter operators within 5 x 5 nm recording blocks intersecting with the marine reserve increased from 1,446 in 2008 to a peak of 3,065 in 2011, before declining again to 1,224 in 2016 (Figure 3.11). The five most popular species retained by charter operators were spangled emperor (*Lethrinus nebulosus*), redthroat emperor (*Lethrinus miniatus*), rankin cod (*Epinephelus multinotatus*), spotcheek emperor (*Lethrinus rubrioperculatus*) and goldband snapper (*Pristipomoides multidens*). With the exception of the spotcheek emperor, retained catches of all other species peaked in the period of 2010 to 2012 which generally coincided with the peak in annual effort in terms of days fished (Figure 3.11).

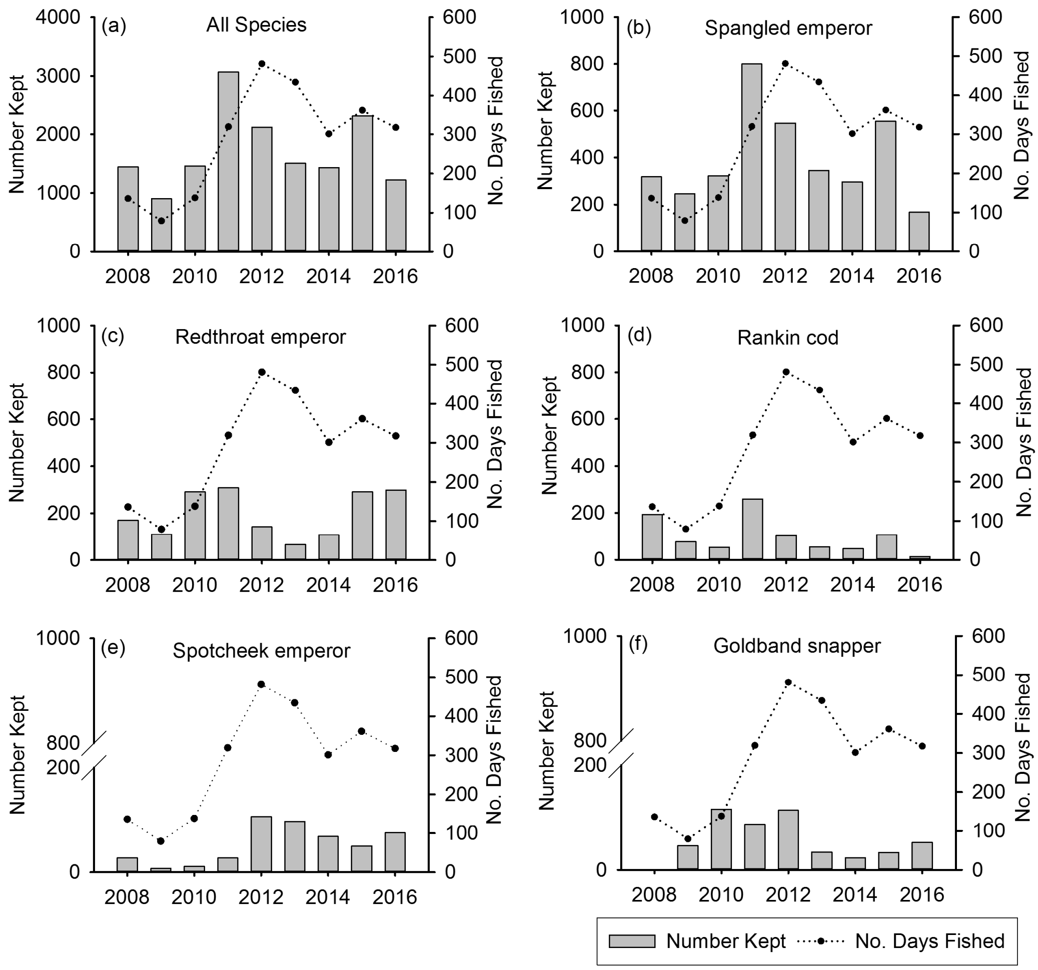

Figure 3.11: Total charter fishing catch and annual number of days fished (a) and for the five most retained species, spangled emperor (b), redthroat emperor (c), rankin cod (d), spotcheek emperor (e) and goldband snapper (f) for the Ningaloo marine reserves, 2008-2016. Information was obtained using charter operator logbook returns for all 5 x 5 nm recording blocks that intersect with marine reserve boundaries.

Source: Department of Primary Industries and Regional Development.

3.4.3 Benthic Habitat Loss – Coral

Hard coral is one of the primary benthic habitats that influence the structure of fish communities in the Ningaloo 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 in the marine reserves, please refer to page 41 of this report.

3.5 Synthesis

There is currently no evidence to suggest that there have been changes in the abundance of targeted fish species at shallow water monitoring sites across the limited monitoring period (four sampling events between 2011 and 2016). The incorporation of sampling at Mandu and Osprey sites from 2006 and 2007 further suggests that the target fish community has likely remained relatively stable across the past decade. While Westera (2003) indicated that target fish abundances may have been higher at the same sites in 1999/2000 (~2.2 - 4.6 fish per 250m^2 in 1999/2000 as opposed to ~0.11 – 2.5 fish per 250m^2 from 2011 – 2016) methodological differences (the 1999/2000 samples utilised Underwater Visual Census (UVC) while all other sampling was conducted using stereo-DOV) make comparisons with more recent sampling events difficult. Holmes *et al.* (2013) found that UVC recorded higher abundances of targeted fish species than stereo-DOV at this location. Although the degree of difference is likely to differ depending on the experience of the person conducting the UVC, it is possible that this factor would account for the higher abundances observed in 1999/2000.

The slightly higher abundances of target species observed across management zones in 2015 and in sanctuary zones in 2016 are associated with relatively high variation within the sites. Given that fishing effort is unlikely to vary markedly over the short-term (between years), these higher abundances may be primarily driven by natural population dynamics associated with recruitment variability. Indeed, recruitment has exhibited high temporal variability and the data suggests that increases in abundance of adult target fishes may follow recorded peaks in recruitment. Furthermore, supply of juvenile fishes in the Ningaloo lagoon is tightly associated with regional climatic processes such as the Southern Oscillation Index (Wilson *et al.*, 2017), supporting the notion that natural dynamics are influencing these trends.

There is a lack of clear differences in the abundance of targeted fish between management zones. Documented evidence of the effectiveness of sanctuary zones subject to appropriate compliance is common in the literature (e.g. Abesamis *et al.*, 2006; Russ *et al.*, 2008), yet there is little evidence presented here to suggest that the abundance of targeted fish is higher in sanctuary zones sampled at Ningaloo. This may be due to the relatively small number of sampling sites currently included into the presented data, with the effects of spatial fishing restrictions often being subtle in nature (Wilson et al., 2012). Alternately, this may be associated with the relatively low fish abundances recorded overall, with the stereo-DOV method perhaps being a sub-optimal method for assessing target fish communities at this location (see Holmes *et al*., 2013; Haberstroh, 2016). Additional factors that may underlie the observed trends are illegal fishing in sanctuary zones (Smallwood *et al*., 2012) and/or the movement of fishes across zone boundaries. *Lethrinus nebulosus*, for example, have home ranges that may exceed the size of most sanctuary zones in the Ningaloo reserves (Pillans *et al*., 2014) meaning that they are at times likely to be exposed to fishing pressure. At this time, however, it is difficult to clearly identify the primary factors that cause the observed patterns.

Widespread declines in the abundances of corallivores are likely associated with localised declines in coral cover, the benthic habitat on which this fish group is dependent. This is most clearly demonstrated at Bundegi, North Muiron and Pelican which sustained significant coral loss in 2010/11 (Moore *et al*., 2012; Depczynski *et al*., 2013) and between 2011 and 2013, respectively (refer to Coral Reef Communities chapter). Declines in corallivore abundance at Mandu are more surprising, given that coral cover at monitoring sites in this region has been relatively stable through time. However, fish and coral are not monitored at the same sites, and this result may be due to losses of coral away from the fixed coral monitoring locations. Newly developed techniques will be applied to future and past stereo-DOV imagery that will allow the extraction of habitat information to support the coral monitoring program and assist with the interpretation of the fish data set (Bennett *et al*., 2016).

Although there have been noted declines in coralivores, the abundances of other trophic groups, such as piscivores and large herbivores, have remained relatively stable (although with considerable variance in the data). This indicates that habitat degradation is currently likely to be the most important pressure influencing the trophic structure of fish communities within the reserves (see Wilson *et al*., 2012).

While fish recruitment has declined since 2011, this is likely linked to natural climate fluctuations and is not currently considered to be a management concern. Fish recruitment is a highly dynamic process and evidence suggests that recruitment in the Ningaloo reserves is strongly influenced by regional climatic cycles such as the Southern Oscillation Index (Wilson *et al*., 2017). The SOI peaked in 2011 and has been on a downward cycle into an *El Nino* period since. Fish recruitment has followed this cycle closely, with weaker south-flowing currents corresponding with relatively low fish recruitment across the summer period.

The abundance of targeted fish in sanctuary zones and overall species richness in deeper waters were markedly higher in 2015 than in 2013. The cause of differences in targeted fish abundance and species richness between 2013 and 2015 is likely due in part to methodological inconsistencies between surveys, with structurally complex habitat being more accurately sampled in 2015. It should also be noted that this result is based on just two sampling events, meaning that it is inappropriate to draw firm conclusions regarding trends in the data. Overall, targeted fish abundance and species richness was higher within sanctuary than general use zones. This pattern was largely driven by data from the Cloates Sanctuary Zone, which is known to support more complex habitat in deeper water relative to other areas.

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. Anecdotal information suggests that fishing activity in the reserves increased greatly during the 1970s and 1980s when infrastructure upgrades, particularly along the west coast of North West Cape, improved access and increased visitation to the region (Fowles, 2007). Current assessments are based on estimates at bioregional scales from state-wide surveys conducted by the DPIRD and, while these are likely to provide indicative information, they are not directly applicable to the marine reserve area. The Ningaloo 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 (2011/2012 to 2013/2014). 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.

With the exception of *Lethrinus nebulosus* and *Epinephelus rivulatus*, all target species identified by Ryan *et al.* (2013; 2015) are exclusively found in deeper waters outside of the lagoon, indicating that boat-based effort is most likely focused in those areas (although it should be noted that *L. nebulosus* and *E. rivulatus* are the two most targeted species, and are both predominantly found inside the lagoon). While this information is extrapolated from a wider bioregional dataset, unpublished surveys and anecdotal observations indicate that it is likely to be indicative of patterns within the marine reserves. 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.

The most recent spatial assessment of recreational fishing activity in the Ningaloo marine reserves was in 2007 (Smallwood *et al*., 2011). This survey found that fishing effort was mostly concentrated close to boat ramps (i.e. Bundegi, Tantabiddi and

Coral Bay), followed by high-use coastal camping areas such as Winderabandi Point, Lefroy Bay, 14 Mile and Gnarraloo Bay. As these patterns of visitor use have not markedly changed since then, fishing effort is likely to still be concentrated in these areas, although effort may have extended further away from Coral Bay and Tantabiddi with the development of improved boat ramps at these locations in 2007 and 2014, respectively. Anecdotal information also suggests that some fishing effort may have moved further offshore into deeper areas of the marine reserves, including into Commonwealth waters. These surveys need to be repeated to provide updated information on the spatial extent of fishing in the reserves.

In comparison to boat-based recreational fishing effort, charter fishing effort in the marine reserves is relatively small (approximately 1,500 target fish retained by charter operators compared to over 14,000 target fish retained by boat-based recreational fishers in 2013/2014). While many charter vessels operate within the park, much of the fishing effort occurs in deeper waters outside the boundary of the state managed reserve. Charter fishing effort peaked in 2012 (482 fishing days) and while it has since declined (317 fishing days in 2016), is still well above the low recorded in 2009 (79 fishing days). There has been an increase in the percentage of fish released/discarded following capture (56.9% in 2016 compared with 26.7% in 2008), either indicating a movement towards catch/release in the charter industry, stricter DPIRD bag limits, or an increase in the proportion of under-sized fishes in the overall catch.

4 Coral Reef Communities (KPI)

4.1 Key Points

- Severe declines in coral cover occurred at shallow water monitoring sites in the eastern sector in 2011 (80-95% coral loss at Bundegi) and 2013 (~50% coral loss at Muiron Islands) due to thermal stress and cyclones. Coral cover at these sites is currently at historically low levels of less than 5%.
- Coral cover has continued to decline at shallow water lagoonal monitoring sites in the southern sector since 2011, most likely due to occurrences of thermal stress from 2011 to 2013.
- Coral loss at shallow water monitoring sites is most apparent in the dominant family Acroporidae, although losses of the less abundant families Pocilloporidae and Poritidae have also occurred.
- Coral recovery in the eastern sector may be inhibited by consistently low levels of recruitment recorded in this region since 2011.
- The condition assessment is considered indicative of shallow water lagoonal and leeward reefs within the marine reserves, with little quantitative data currently existing for coral communities on the reef slope or deeper waters.
- The confidence in the condition and pressure indicators is generally high due to the length of time over which monitoring has been conducted, and the number of survey events across this period. It should however be noted that temporal variability in the data prior to 2010 may be higher due to inconsistent spatial coverage between sampling events.

4.2 Indicator Summary

	Trend	**Confidence**
Condition		
Coral cover	Decreasing	High
Community composition	Changing	High
Recruitment	Decreasing	Medium
Pressure		
Thermal stress	Increasing	High
Coral predators	Stable	High
Cyclones	Stable	Low

4.3 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 Ningaloo marine reserves and presented in this report are:

- Coral cover.
- Community composition.

- Recruitment.

Coral cover, a broad metric that encompasses the abundance of live coral, 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, 1990; McClanahan *et al.*, 2007; Berkelmans, 2009). Changes in community composition can have major implications for the structure of associated reef organisms given the diversity of roles that different taxa and growth forms play as habitat or food (Connell and Kingsford, 1998; Wilson *et al.*, 2010b). Measures of coral recruitment are indicative of future supply and recovery potential within coral communities and represent a source of natural variation in the abundance of corals and structure of coral communities (Hughes *et al.*, 1999). Understanding the relationship between live coral abundance and recruitment is therefore important when seeking to understand the long-term impact of pressures on recovery and changes in coral communities (Hughes and Tanner, 2000; Acosta *et al.*, 2011).

The survey design for assessing coral community condition is based around the primary pressures acting on them (i.e. thermal stress, coral predators and cyclones). As pressures are spatially and temporally variable, the design encompasses sites spread across the entire length of the marine reserves, including sites where historical information is most readily available (prior to the commencement of the standardised coral monitoring program in 2010). The incorporation of historical information is important as it provides valuable information on the medium-term dynamics of the condition indicators, assisting with the interpretation of more recent data. All survey sites are in protected shallow water lagoonal and leeward reef areas that are readily accessible in varied weather conditions, and are most sensitive to the major pressures acting on the marine reserve. While interpretation of the condition indicators should largely be confined to these areas only (and not, for example, on the exposed reef slope), they are considered a good representation of pressure effects on coral communities at Ningaloo.

Coral cover and community composition are measured with standard monitoring methods (Loya, 1978; Brown *et al.*, 2004). The survey uses a nested sampling design, with sites nested within the three sectors of the marine reserves; eastern (6 sites), northern (6 sites) and southern (8 sites) (Figure 2.1). These sectors were identified according to a natural separation of the physical oceanographic patterns of the region, which may differentiate how the benthic community in each sector responds to pressures identified in the marine reserves (Taylor and Pearce, 1999; Woo *et al.*, 2006a; Taebi *et al.*, 2012). While the spatial coverage of coral monitoring was patchy prior to 2010, with a subset of only 7-15 sites included in each survey, all data have been included in these analyses. Importantly, interpretation of the results must consider that survey effort was lower and inconsistent prior to 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 from 2010 onwards. Using this methodology, fifty (1m height and ~ 1m^2) or one hundred (0.5m height and ~ 0.5m^2) downward facing benthic photo-quadrats are taken at regular intervals along each transect, depending on water depth. Benthic video was used to collect imagery along transects and 50 equidistant photo quadrats were taken from the video recordings between 1999 and 2010, whilst line intercept transects were used to record benthic communities between 1991 and 1999. To estimate cover and taxonomic classification of live hard coral or categorise alternate substrata (e.g. sand, dead hard coral, turf algae), six or three points are placed on each 1m^2 and 0.5m^2 photo-quadrat respectively for 2010 onwards, whilst five sampling points per image were used between 1999 and 2010 (Abdo *et al.,* 2004). The line intercept transect method used between 1991 and 1999 records the distance of each taxa or substrata category along the transect line and divides this length by the total length of each transect to determine the relative proportional cover of each category.

Coral recruitment is measured as the number of individuals that settle to standardised terracotta tiles deployed during the coral spawning and subsequent recruitment period at Ningaloo (Babcock *et al.*, 1994; Rosser, 2013). Five replicate settlement tiles were deployed at three sites nested within three (eastern and northern sectors) and four (southern sector) locations in each sector (Figure 2.1).

4.3.1 Coral Cover

Coral cover was relatively stable across all monitoring sites from 1991 to 2010 (Figure 4.1). Since 2010, however, there has been a decline in overall coral cover, from approximately 30-40% cover prior to 2011, to <20% cover in 2015. Analyses indicate that this decline is primarily caused by change in the eastern and southern sectors, where coral cover declined from approximately 50% to <15% and from 45% to 20%, respectively (Figure 4.2).

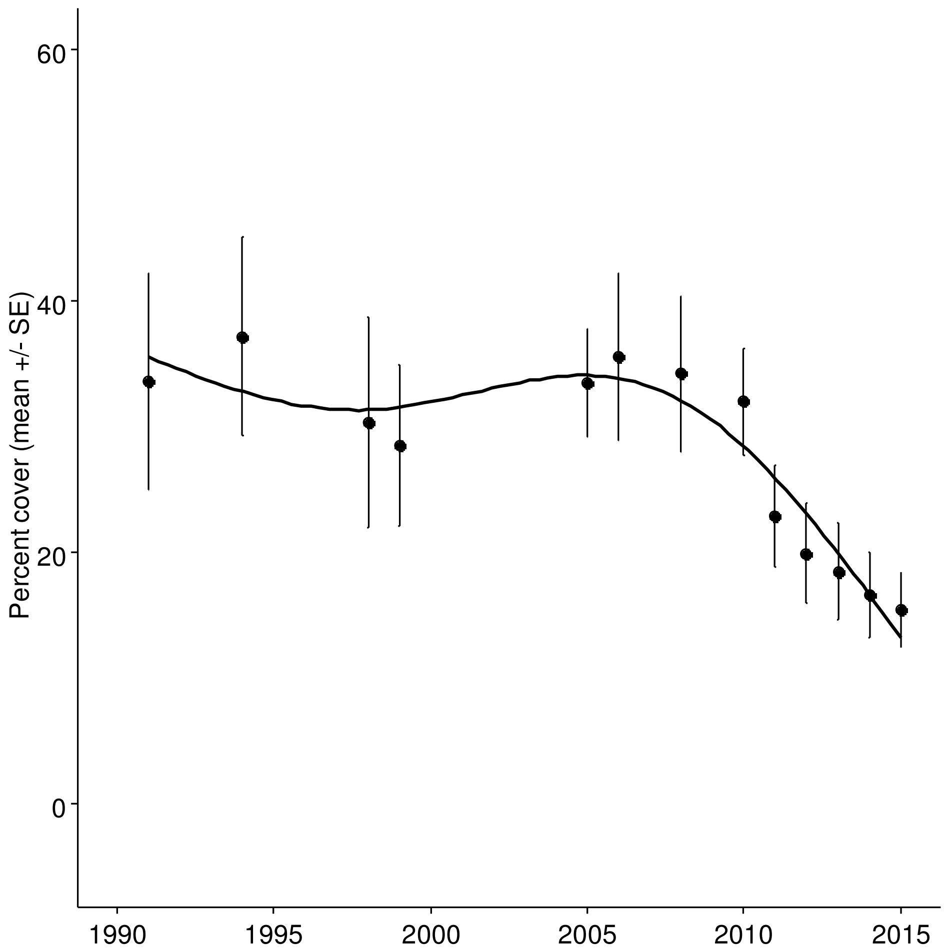

Figure 4.1: Mean coral cover (±1 SE) at all sites located within the Ningaloo marine reserves (n = 9 from 1991 to 1994; n = 11 in 1998, n = 7 in 1999, n = 12 in 2005, n = 10 in 2006, n = 7 in 2008; n = 20 from 2010 to 2015). Black line represents a fitted non-parametric generalised additive model (Adjusted R^2 = 0.86).

Severe (80-95%) declines in coral cover occurred at all four Bundegi sites in the eastern sector in 2011, with the two Muiron islands sites also suffering ~50% declines in coral cover in 2013. Whilst coral cover is currently at historically low levels in the eastern sector, this area has exhibited higher variability in coral cover over the 25 years of surveys compared to the northern and southern sectors. This suggests that corals in the eastern sector may have been resilient to periodic disturbances prior to 2011. Coral cover in the southern sector appears to have undergone a less sudden but sustained decline between 2011 and 2015. While initial coral loses occurred at the most southern reef areas of Gnarraloo Bay, Three Mile and Turtle Bay, declines have also occurred in the last three years at sites further north in the sector, including Point Cloates, Coral Bay and Pelican Point. In contrast to the other two sectors, coral cover at sites in the northern sector has remained relatively stable across the survey period (Figure 4.2).

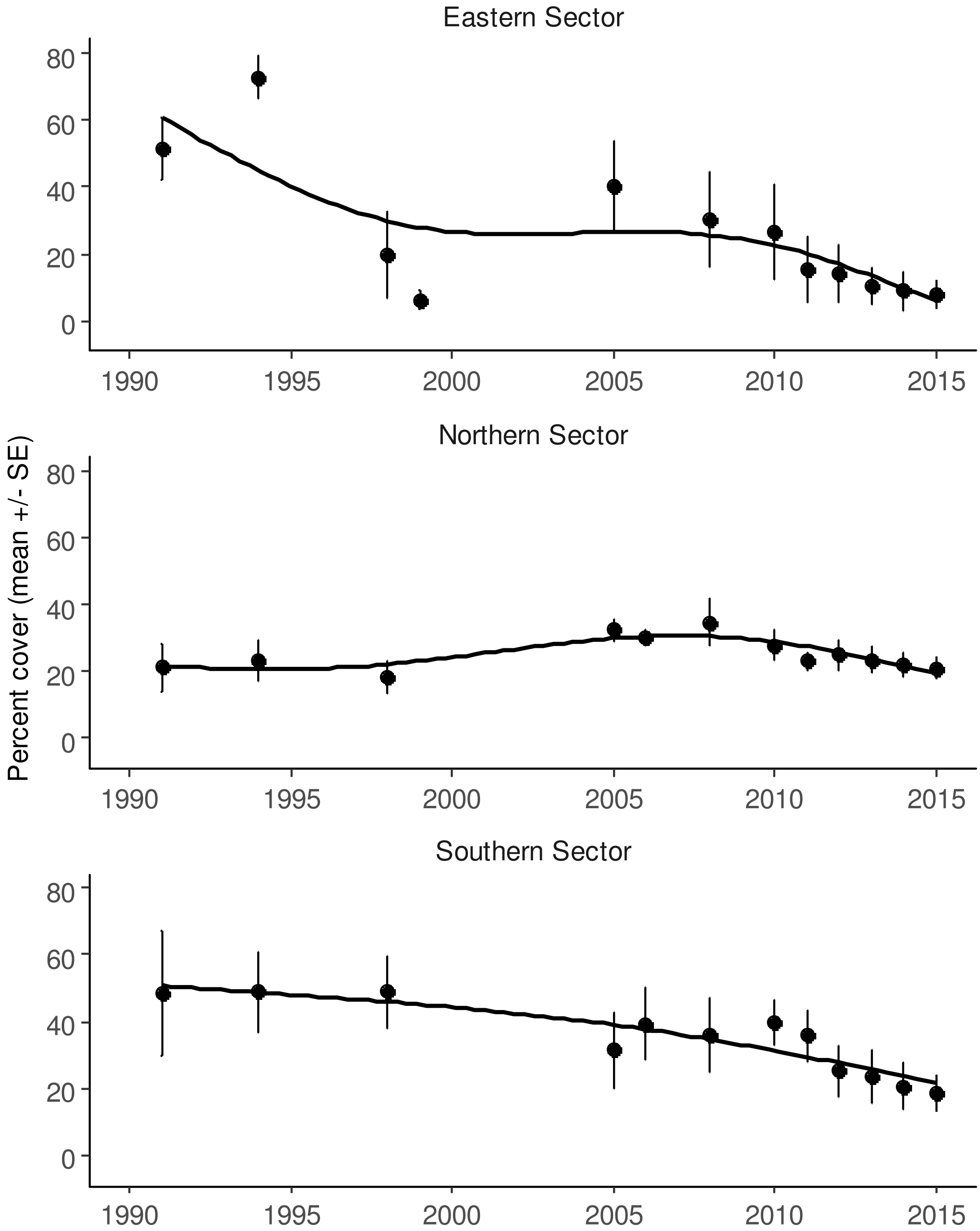

Figure 4.2: Mean coral cover (±1 SE) at sites in the eastern, northern and southern sectors of the Ningaloo marine reserves (n = 9 from 1991 to 1994; n = 11 in 1998, n = 12 in 2005, n = 10 in 2006, n = 7 in 2008; n = 20 from 2010 to 2015). Black lines represent fitted non-parametric generalised additive model (eastern sector Adjusted R^2 = 0.51; northern sector Adjusted R^2 = 0.84; southern sector Adjusted R^2 = 0.82).

4.3.2 Coral Community Composition

Coral community composition is assessed by measuring the percentage cover of the four most common families; Acroporidae, Faviidae, Pocilloporidae and Poritidae. There were notable declines in the cover of acroporids, pocillioporids and poritids at monitoring sites following 2010, with acroporids continuing to decline in all subsequent years (Figure 4.3). Given that acroporids are by far the most common hard corals, and pocilloporids and poritids constitute relatively low proportions of coral cover at monitoring sites (<5% combined in 2015), the loss of acroporids represents the most notable change in community composition.

Examination of acroporid cover within each of the sectors (Figure 4.4) indicates that changes to this taxa closely reflect the changes to overall coral cover (Figure 4.2), with major declines in both the eastern and southern sectors since 2010. This indicates that recent declines in coral cover throughout the reserves are primarily a consequence of changes to acroporids.

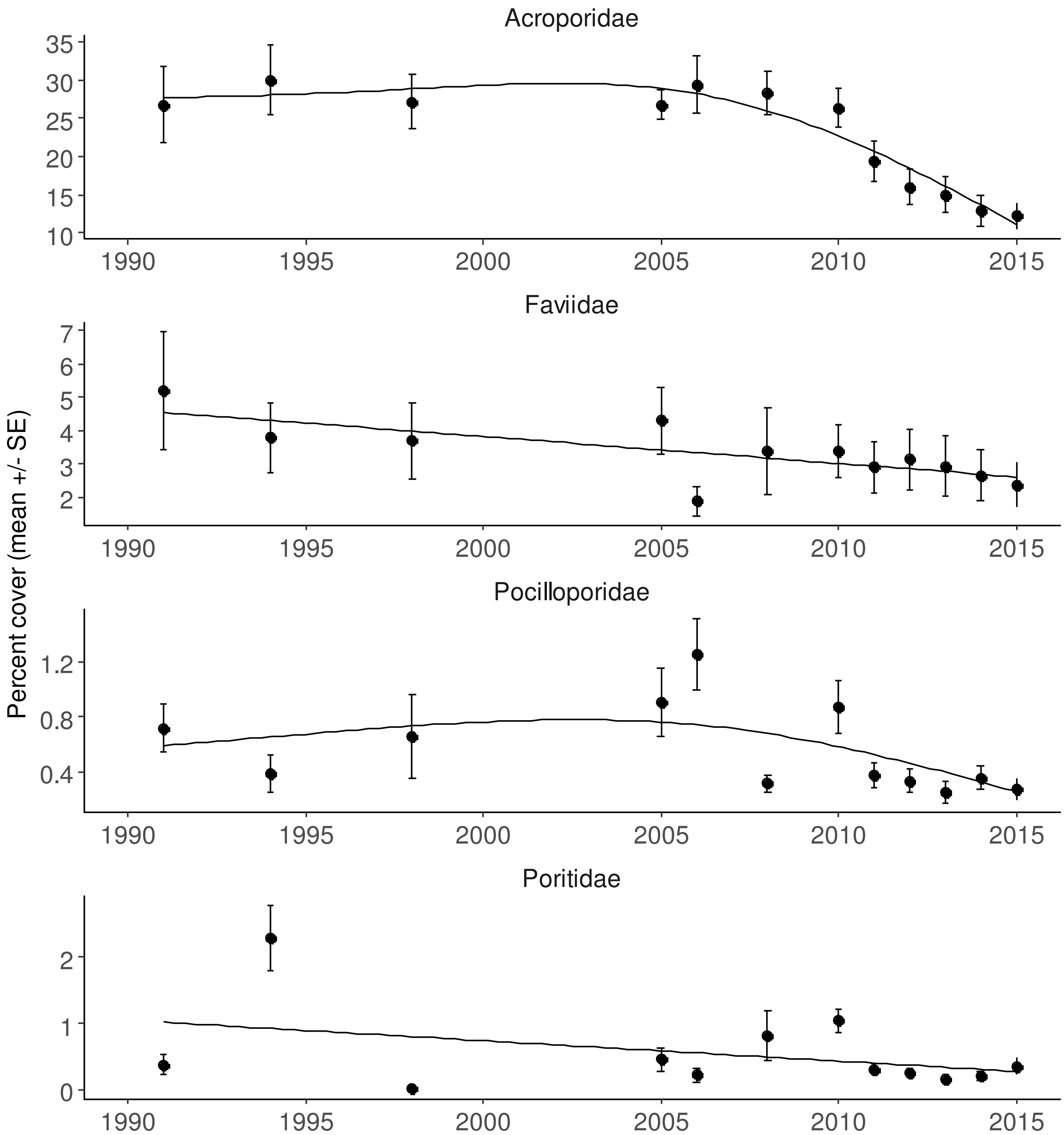

Figure 4.3: Mean percent cover (±1 SE) of the four dominant coral families at all sites located in the Ningaloo marine reserves (n = 9 from 1991 to 1994; n = 14 in 1998, n=12 in 2005, n = 10 in 2006 and 2008; n = 20 from 2010 to 2015). Black lines represent fitted non-parametric generalised additive model (Acroporidae, Adjusted R^2 = 0.91; Faviidae Adjusted R^2 = 0.49, Pocilloporidae Adjusted R^2 = 0.23, and Poritidae Adjusted R^2 = 0.07).

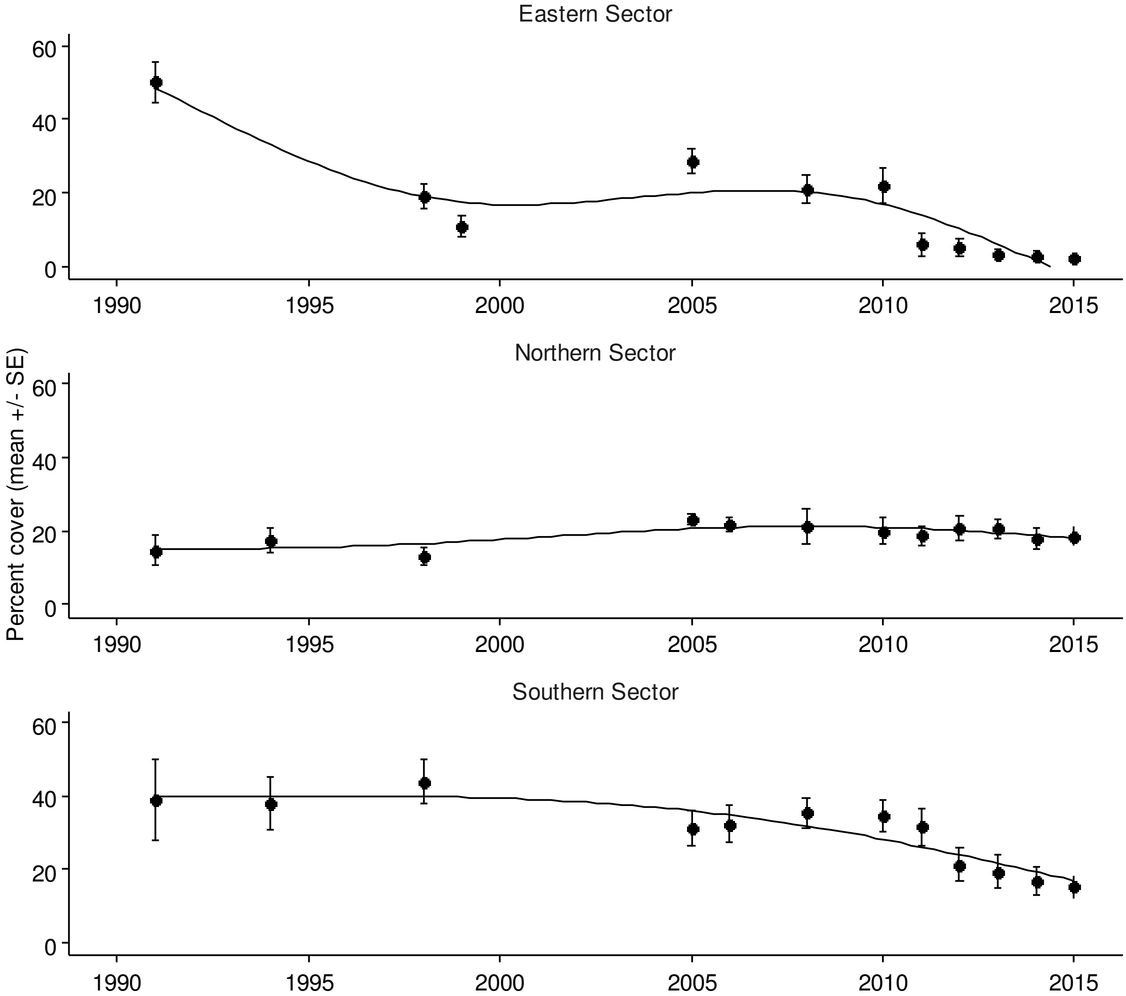

Figure 4.4: Mean percent cover (± 1 SE) of acroporid corals at sites in the eastern, northern and southern sectors of the Ningaloo marine reserves (n = 9 from 1991 to 1994; n= 14 in 1998, n = 4 in 1999, n=12 in 2005, n = 10 in 2006 and 2008; n = 20 from 2010 to 2015). Black lines represent fitted non-parametric generalised additive models (eastern sector Adjusted R^2 = 0.62; northern sector Adjusted R^2 = 0.58; southern sector Adjusted R^2 = 0.81).

4.3.3 Coral Recruitment

Coral recruitment has been variable across all sites over the monitoring period (Figure 4.5). While there is some evidence to suggest a decline in mean recruitment from 2009 to 2016, this apparent trend is largely caused by relatively high levels of recruitment recorded in the first survey year (2009), after which variability then obscures any clear patterns of change from 2010-2016. The decline in recruit numbers across the reserves is predominantly the result of very low recruitment measured within the eastern sector from 2011-2016, which contrasts with extremely high recruitment recorded in 2009 and 2010 (Figure 4.6). Recruitment in the northern and southern sectors however has been more variable, with no evidence to suggest a changing trend over this time (Figure 4.6).

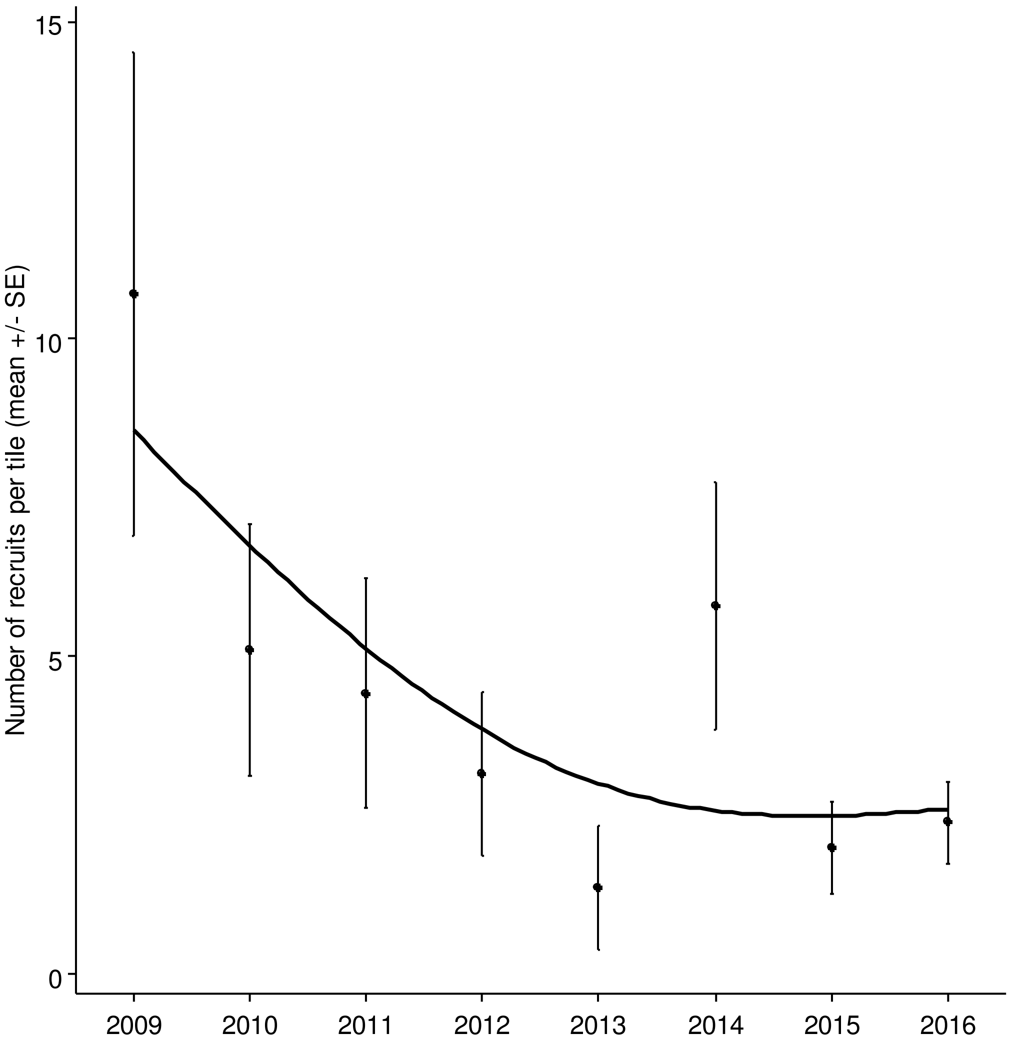

Figure 4.5: Mean coral recruitment (± SE) at the Ningaloo marine reserves (n = 150). Five tiles at three sites nested within three locations for eastern (Bundegi) and northern (Tantabiddi) sectors, and four locations in the southern sector (Coral Bay). Black line represents a fitted non-parametric generalised additive model (Adjusted $R^2 = 0.54$).

Source: Australian Institute of Marine Science (2009, 2010) and Department of Biodiversity, Conservation and Attractions (2011-2016).

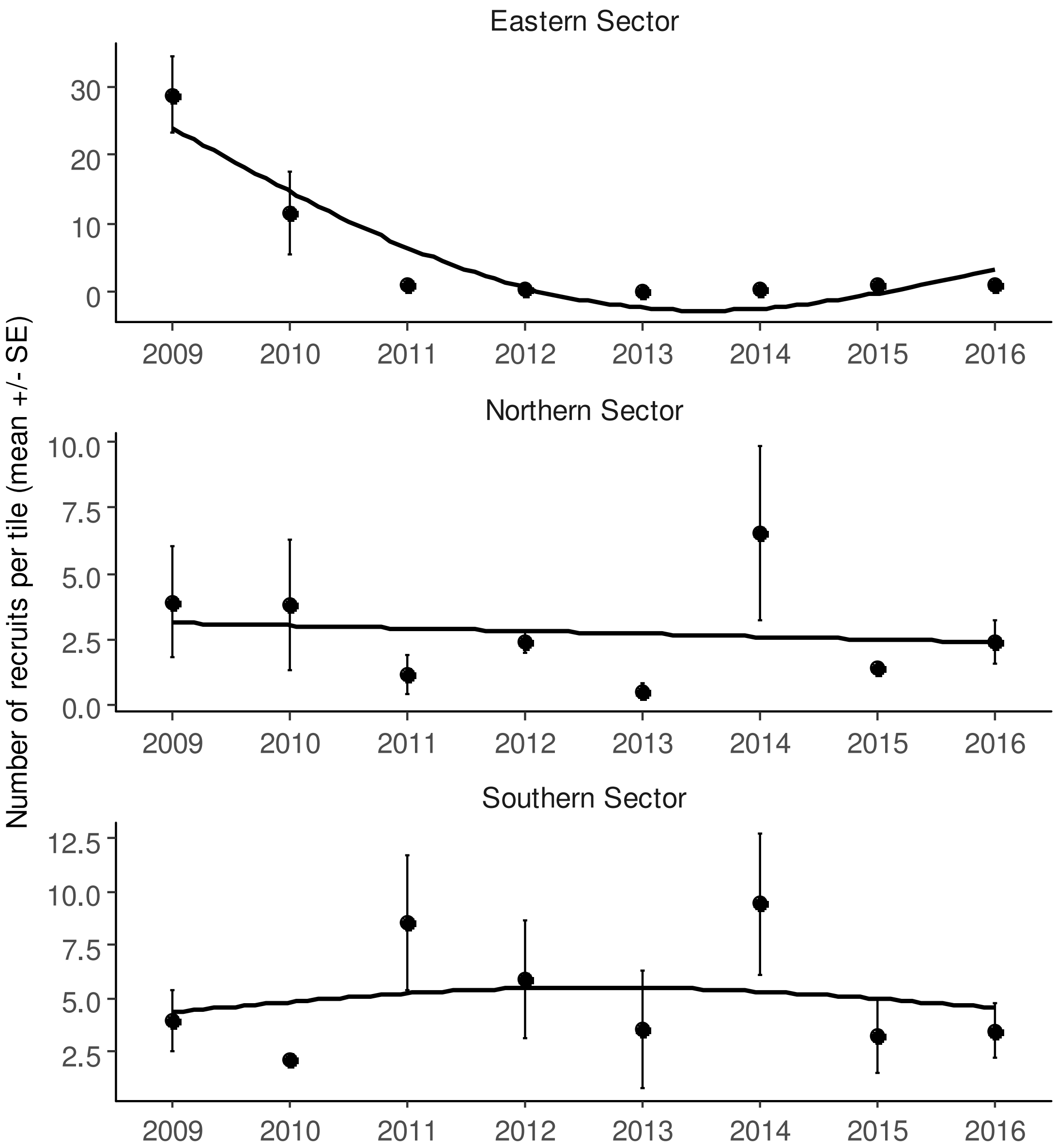

Figure 4.6: Mean coral recruitment (± SE) within eastern (n = 45), northern (n = 45) and southern (n = 60) sectors of the Ningaloo marine reserves. Black lines represent fitted non-parametric generalised additive models (eastern sector Adjusted R^2 = 0.60; northern sector Adjusted R^2 = 0.01; southern sector Adjusted R^2 = 0.01

Source: Australian Institute of Marine Science (2009, 2010) and Department of Biodiversity, Conservation and Attractions (2011-2016).

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 Ningaloo marine reserves and presented in this report are:

- Thermal stress.
- Coral predators.
- Cyclones.

When severe, thermal stress can lead to coral bleaching and mortality and is assessed by measuring the accumulated temperature stress during a 12 week period in which the seawater temperature exceeds the normal local conditions (known as climatology) by 1°C. This is referred to as Degree Heating Weeks (DHW) (Liu *et al.*, 2006; Glynn and D'croz, 1990). The primary coral predator relevant to the marine reserves is the corallivorous gastropod *Drupella cornus*. *D. cornus* has previously caused significant loss of coral in parts of the Ningaloo marine reserves, with the last major impacts occurring in the early 1990s (Turner, 1994). Crown of thorns starfish (*Acanthaster planci*) also feed on coral and occur in the reserves, but are most prevalent around the Muiron Islands and are not currently considered to be a significant threat to coral communities. Cyclones have been identified as having 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 coral 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 cyclones in the Ningaloo marine reserves are currently made using the number of cyclones that pass within a 50km and 200km radius of Coral Bay (Speed *et al.*, 2013).

4.4.1 Thermal Stress

Sea surface temperature (SST) data from National Oceanic and Atmospheric Administration (NOAA) virtual stations at Ningaloo (http://coralreefwatch.noaa.gov/satellite/vs/) 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 for the northern and southern sectors is based on DHW calculated from 50 x 50 km satellite pixels located to the north-west of North West Cape and adjacent to Coral Bay, respectively (Figure 2.1). There is currently no comparable way to assess DHW in the eastern sector as this region is oceanographically distinct from the northern and southern sectors (Taylor and Pearce, 1999; Taebi *et al.*, 2012) and the available data are not considered to be representative of local thermal stress. However, assessing sea surface temperature trends that have been corrected for *in situ* seawater temperature (Baldock *et al.*, 2014), provides a relative indication of the thermal pressures on coral communities within this sector (Figure 2.2).

Both the northern and southern sectors experienced significant thermal stress (assessed as Level 8 DHW) in summers from 2011-2013 (Figure 4.7). This was most apparent in 2011 and 2013 in the northern sector and 2011 to 2013 in the southern sectors, with the number of DHWs being higher in the southern sector in each of these years. Whilst DHW was not assessed for the eastern sector, corrected seawater temperature shows that temperatures were also high during the 2011-2013 period, with temperatures exceeding that of the southern and northern sectors. Thus, corals in this sector are also likely to have been exposed to excessively high cumulative thermal stress across this period.

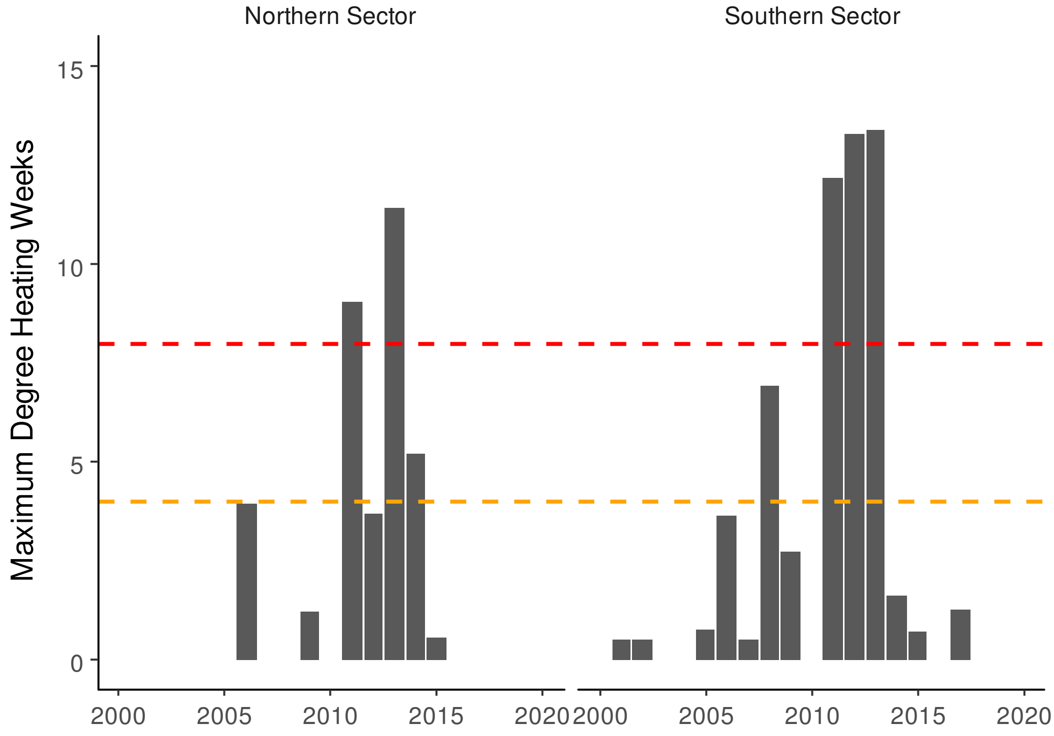

Figure 4.7: Maximum degree heating weeks (DHW) recorded in the northern and southern sectors of the Ningaloo marine reserves. No similar assessment of the eastern sector was possible as the NOAA satellite virtual stations are only located northwest of Point Murat (northern sector) and west of Coral Bay (southern sector). Orange dashed line indicates a four DHW threshold likely to cause bleaching, and red dashed line indicates an eight DHW threshold likely to cause mortality.

Source: NOAA coral reef watch.

4.4.2 Coral predators

The abundance of *D. cornus* is surveyed along a belt 0.5 metres wide on the same transects used to assess coral communities (i.e. three replicate 50m transects at each site). The abundance of *D. cornus* has decreased in all sectors since monitoring began in 1991 (Figure 4.8). The highest mean abundance (±1 SE) of *D. cornus* currently recorded is 5.9 (±0.49) individuals per m^2 at Pelican Point in the southern sector. While this is much higher than all other monitored sites, it is still considered lower than historical outbreak levels of approximately 18 per m^2 recorded at Ningaloo Reef in the early 1980s and early 1990s (Ayling and Ayling, 1987; Turner, 1994).

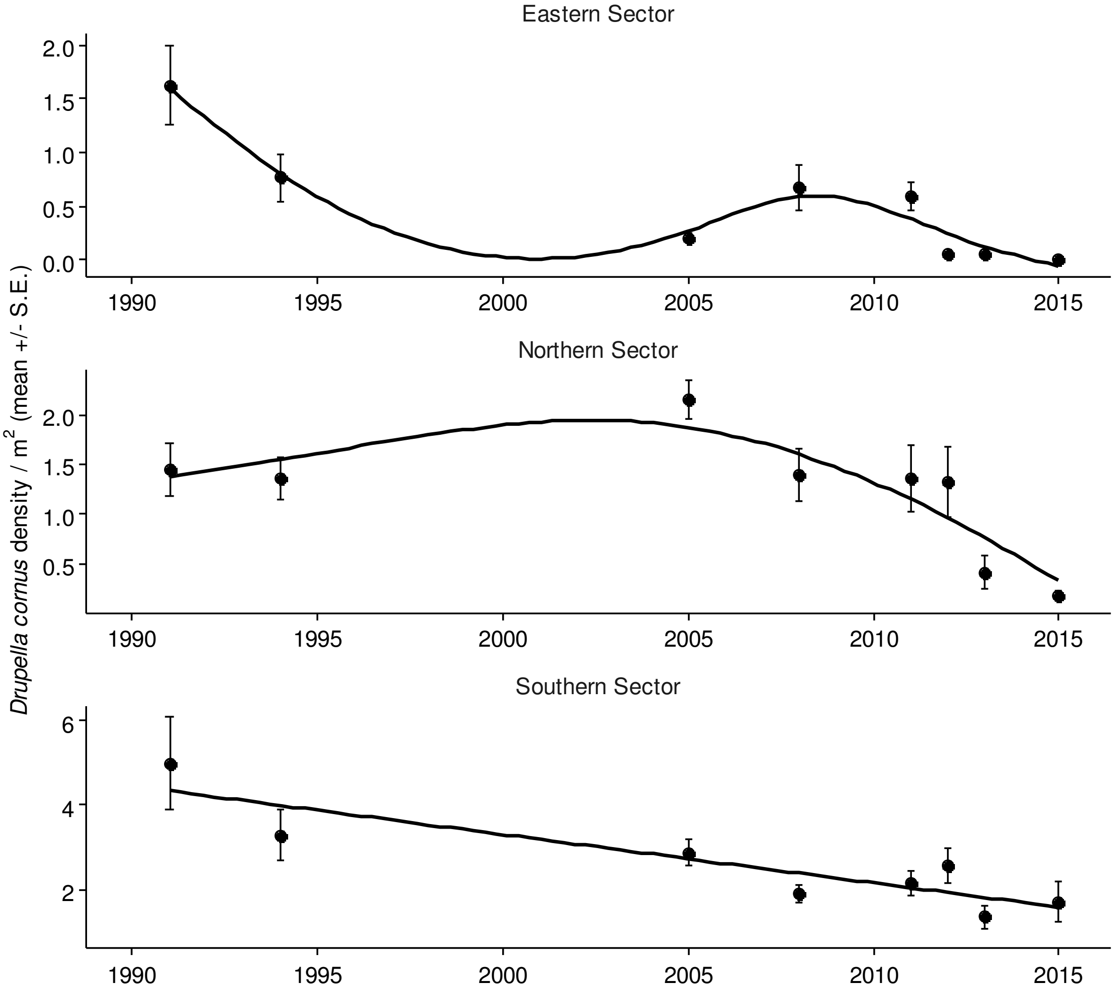

Figure 4.8: Mean *Drupella cornus* density per m^2 (+/-1 SE) in the eastern, northern and southern sectors of the Ningaloo marine reserves (n = 9 from 1991 to 1994; n = 15 in 1999, n = 9 in 2006 and 2008; n = 20 from 2010 to 2015) Black lines represent fitted non-parametric generalised additive models (eastern sector Adjusted R^2 = 0.71; northern sector Adjusted R^2 = 0.42; southern sector Adjusted R^2 = 0.75).

4.4.3 Cyclones

Cyclone frequency and intensity information were acquired from BoM (http://www.bom.gov.au/cyclone/history/tracks/index.shtml). All cyclones classified as category 1 or above that passed within 50km and 200km radii of Ningaloo (using Coral Bay as a centre point) were identified as potential sources of wave damage to coral in the Ningaloo marine reserves (Speed *et al.*, 2013).

Three cyclones passed within a 50km radius of the Ningaloo marine reserves between 1999 and 2006; Vance in 1999, Carlos in 2011, and Olwyn in 2015. Another four cyclones passed within a 50-200km radius; Steve in 2000, Alistair in 2001, and Daryl and Glenda in 2006.

4.5 Synthesis

The condition of coral communities at shallow water monitoring sites in the Ningaloo marine reserves has declined since 2010. This decline is primarily due to the loss of coral in the eastern and southern sectors, with the northern sector remaining relatively stable across the monitored period. The most severe declines occurred in 2011 in the eastern sector, driven by coral loss of 80-95% at the Bundegi monitoring sites. Subsequent changes in this sector were primarily due to coral loss at sites around the Muiron Islands from 2013 onwards. The main loss of coral cover in the southern sector occurred in 2012, with smaller but consistent declines being measured in each sampling event since then. These declines were initially driven by the loss of 60-80% of corals at the southern-most sites (Gnarraloo Bay, Three Mile and Turtle Bay) since 2010. More recently, however, there has been coral loss at other sites in the sector (Pelican Point, Coral Bay and Cloates), such that six of the eight southern sector monitoring sites now exhibit declining trends of coral cover since 2010. It should be noted that the current monitoring program is based on sites situated in protected shallow water lagoonal and leeward reef locations, where coral cover is generally highest and considered most vulnerable to stressors. As such, the condition and pressure indicators presented in this report are considered indicative of these areas only, and not of reef slope, deeper water or lagoonal bommie coral communities in the marine reserves.

Coral loss has been most apparent amongst acroporid corals, which dominate the coral community at most monitoring sites (exceptions include sites at Tantabiddi and one at Coral Bay which mostly comprise faviid corals). Relatively fast-growing and structurally complex, acroporids are the preferred prey of key coral predators such as *D. cornus* and *A. plancii* and are relatively sensitive to other environmental pressures such as heat stress (Pratchett, 2001; McClanahan *et al.*, 2007; Berkelmans, 2009). However, there have also been small losses of pocillioporid and poritid corals in the same period, particularly from 2010 to 2011. Combined, these two taxa presently comprise <5% of coral cover at all monitoring sites, but have declined by ~60% since 2010. This indicates that changes in community composition cannot be solely attributed to the loss of the dominant acroporids.

Increased thermal stress in the summers from 2011-2013 most likely caused declines in coral condition at shallow water monitoring sites throughout the reserves. While there is no DHW data for the eastern sector, anomalously high water temperatures were known to have occurred there in 2011, resulting in extremely high coral bleaching and mortality that reduced coral cover to <5% at the monitoring sites (Moore *et al.,* 2012; Depczynski *et al.,* 2013; Speed *et al.*, 2013). Thermal stress was also consistently high in the southern sector from 2011-2013 (as high as ~14 DHW in a single year), which appears to have resulted in gradual but sustained coral mortality in the region since 2011. Similar levels of thermal stress in the lagoon in this region were calculated by Zhang *et al.* (2013), providing further confirmation of these results. The differential timing and rates of loss between sites in the southern sector are likely a result of localised fluctuations in seawater temperatures and residence times.

Thermal stress was less in the northern sector and there has been no significant change in coral condition at these monitoring sites. However, anecdotal reports of coral loss at other northern sector locations have occurred during the same period, suggesting that bleaching impacts may be patchy and not always measured by monitoring using current methods. This highlights the need to continually adjust seawater temperature and DHW models to provide improved information for lagoonal coral communities within each of the sectors, as well as investigating new methods that enable coral community assessments at larger spatial scales and at reef slope locations.

The effects of cyclones on the condition of coral communities across the marine reserves are less clear. Cyclone Olwyn (category 3) passed almost directly over Coral Bay in 2015 yet did not have a measurable impact on coral communities at monitoring sites in this area. In contrast, cyclone Carlos (category 2) contributed to major coral loss in the eastern sector in 2011 (Moore *et al.,* 2012). There also appears to have been a reduction in coral cover in the eastern sector between 1998 and 1999 in response to cyclone Vance (category 5), although this is uncertain as coral monitoring was sporadic at the time.

Coral recruitment has been relatively stable in the northern and southern sectors, but has been consistently low in the eastern sector since 2011. This has likely contributed to the lack of coral recovery in the eastern sector following major losses in 2011 and 2013. Low recruitment is potentially due to conditions that affect larval settlement (e.g. high siltation, abundance of coral rubble, ongoing temperature stress) and/or the reduced supply of gametes from source areas (Hughes *et al.*, 2000). Given the significant loss of stable reef structure and low coral cover at sites currently monitored in the eastern sector (e.g. Bundegi), recovery at this location is expected to be limited for some time (Hughes *et al.*, 2000). However, if a period now occurs without the pressures that caused this coral loss, recovery in other depleted areas may be more rapid, due to more consistent recruitment, higher cover of existing live coral and the availability of stable recruitment habitat.

5 Mangrove Communities (KPI)

5.1 Key Points

- The spatial extent of mangroves at Mangrove Bay increased between 2006 and 2014, most likely due to decreases in porewater salinity across the same period favouring recruitment and growth.
- The declining porewater salinity is primarily a result of natural oscillations in sea level caused by regional climatic cycles (Southern Oscilation Index).
- A decline in canopy density, identified in 2014 by remote sensing and confirmed by subsequent field measurements, indicates that there has been some loss of condition and increased mortality at the southern end of the mangroves. This was most likely caused by smothering/erosion from storm run-off in April 2014 and a recent decline in sea level as a part of natural oscillations.
- The current assessments of mangrove community condition are made with medium levels of confidence due to the length of time over which monitoring has occurred (eight years) and the number of survey periods (four). The canopy cover indicator is likely associated with moderate levels of variability due to its sensitivity to sub-lethal effects. The confidence level in these assessments will increase as more data is added over time.

5.2 Indicator Summary

	Trend	Confidence
Condition		
Species diversity	Stable	Medium
Spatial extent	Increasing	Medium
Canopy density	Decreasing	Medium
Pressure		
Atmospheric temperature	Stable	High
Groundwater availability	Stable	High
Sea level	Stable	Low
Cyclones	Stable	Low

5.3 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 Ningaloo marine reserves and presented in this report are:

- Species diversity.
- Spatial extent.
- Canopy density.

Spatial 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). Species diversity/composition is an indicator of community structure and the relative contribution of different species. As species have different environmental tolerances (Saenger, 2002), each is likely to respond to pressures and environmental change differently. Techniques for quantitatively assessing species composition are still being refined and as such, species diversity is used as a coarser surrogate in this report.

The survey design for assessing mangrove communities encompasses all significant mangrove stands within the reserve area. While small stands exist at Yardie Creek and along the coastal strip to the south of Bundegi, these are unlikely to contribute greatly to local ecosystem function and are not well assessed using the current remote sensing methodologies. As such, the assessment within the Ningaloo marine reserves is limited to Mangrove Bay in the northern sector (2.1).

5.3.1 Species diversity

The diversity of mangroves at Mangrove Bay was determined using a combination of satellite (ALOS AVINIR-2 and SPOT 6), aerial imagery and ground-based verification. *Avicennia marina* and *Rhizophora stylosa* have been the only two mangrove species recorded at Mangrove Bay since 2006.

5.3.2 Spatial extent

The spatial extent of mangroves at Mangrove Bay was calculated using the spectral response of mangrove foliage from ALOS AVINIR-2 and SPOT 6 satellite imagery (25-30m pixel footprints) verified by field surveys. The spatial extent has increased steadily across the monitoring period from approximately 23ha in 2006 to approximately 29ha in 2014 (Figure 5.1). The greatest rate of gain occurred between 2006 and 2008, but has subsequently slowed. While there has been a net gain in the overall spatial extent during the monitoring period, there has been both loss and gain at different parts of the site (Figure 5.2).

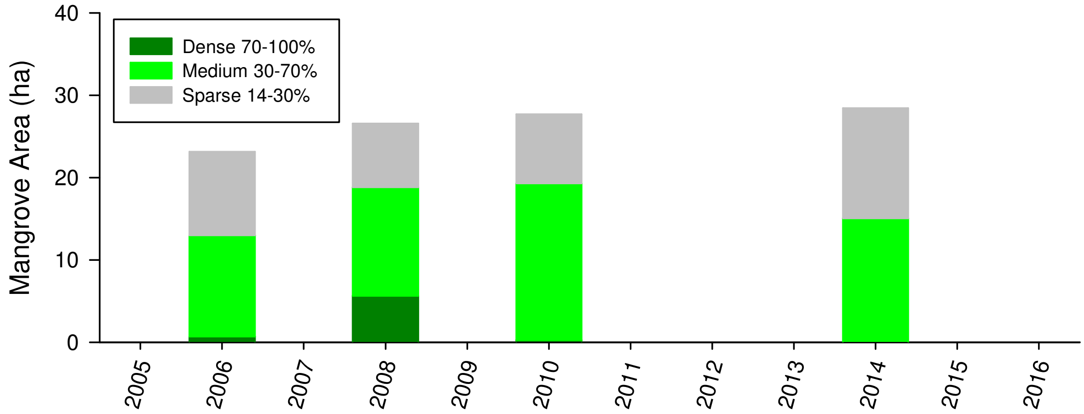

Figure 5.1: The spatial extent of mangroves (measured in hectares) at Mangrove Bay in the Ningaloo marine reserves, determined from remote sensing imagery and incorporating the projected foliage cover (PFC). PFC categories: dense (70-100% foliage cover); medium (30-70% foliage cover); and sparse (14-30% foliage cover). N.B. <14% PFC is considered too sparse to reliably classify at this location.

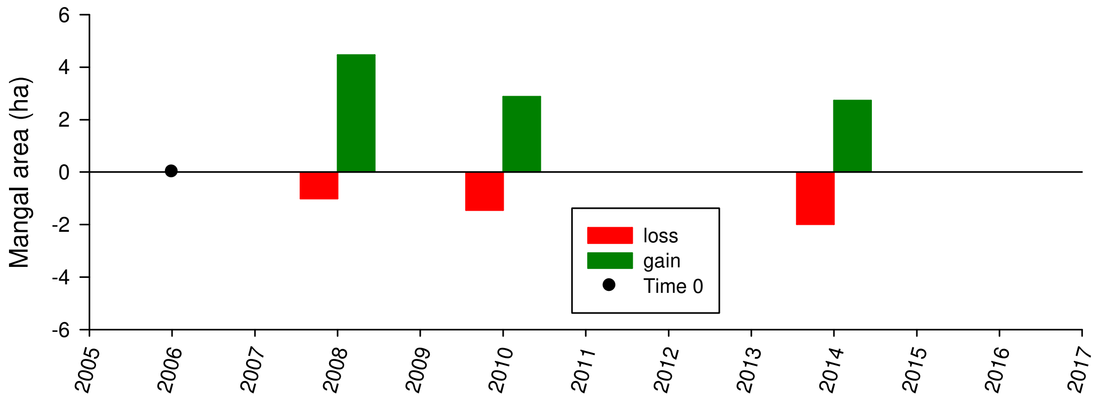

Figure 5.2: Loss and gain in spatial extent of mangroves (measured in hectares) at Mangrove Bay in the Ningaloo marine reserves, determined from remote sensing imagery. Loss and gain for each survey period is measured relative to spatial extent at the previous survey period, with 2006 being considered the baseline.

5.3.3 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 Mangrove Bay. PFC is categorised as dense (70-100% foliage cover), medium (30-70% foliage cover) or sparse (14-30% foliage cover). PFC measurements below 14% were not used in this assessment as this category is considered too sparse at the Mangrove Bay location to be accurately represented within the remote sensor’s 25-30m image pixels.

The proportion of dense PFC increased from 2006 to 2008 (Figure 5.1). Although the overall spatial extent has continued to increase since this period, there has been a decline in the proportion of dense (2010) and medium (2014) PFC, while sparse PFC has increased from ~30% in 2008 to >45% of the overall community in 2014 (Figure 5.1).

5.4 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 Ningaloo marine reserves and presented in this report are:

- Atmospheric temperature.
- Groundwater availability.
- Sea level.
- Cyclones.

Atmospheric and water temperature are considered to be among the most important structuring factors for mangroves on a global scale (Saenger, 2002; Osland *et al.*, 2017). Atmospheric temperature 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 coastal strip of the Ningaloo marine reserves are sea level and groundwater availability (Lovelock *et al.,* 2017). Sea level is assessed through a Bureau of Meteorology logging station located at Exmouth, with observed sea levels measured in metres above Tide Gauge Zero and adjusted to the Australian Height Datum (Bureau of Meteorology, 2017). Local rainfall in the Cape Range region is currently used as a surrogate for groundwater availability, given that this is considered to be the source of effective freshwater recharge at this location (Allen, 1993). Cyclones can have detrimental effects on mangrove communities through wind damage and sediment erosion, causing defoliation, physical damage or uprooting (Gilman *et al.*, 2008; Paling *et al.*, 2008).

5.4.1 Atmospheric Temperature

Air temperature recorded at Exmouth is being used as a representative for temperatures in the Mangrove Bay area. While there are likely to be localised fluctuations between the weather station location and the west coast of North West Cape, this is considered to be the most suitable surrogate of long-term temperature trends.

The long-term data highlights a stable trend in daily maximum air temperature (Figure 5.5). Since July 2011, mean maximum monthly air temperatures have frequently exceeded the long-term mean across all months (Figure 5.6). Above average temperatures were most consistent during the spring (September-November), 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.

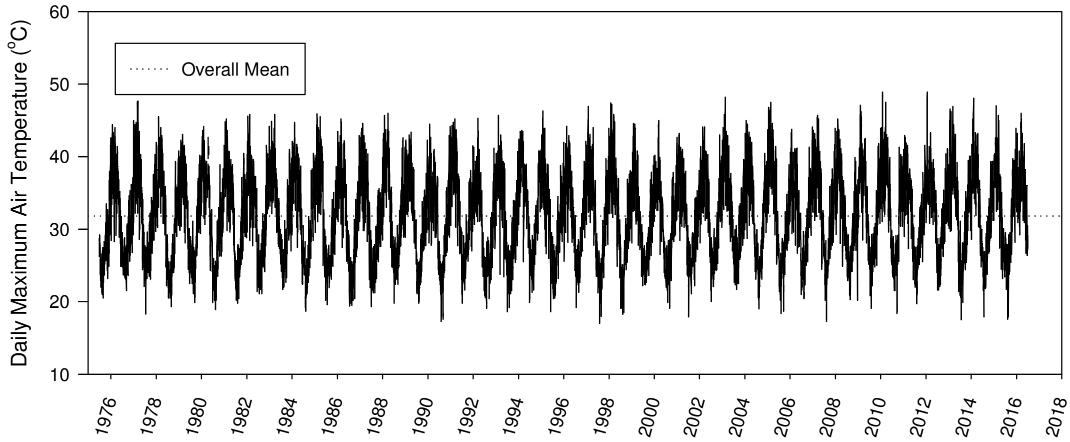

Figure 5.5: Daily maximum air temperature (°C) recorded at Exmouth, 1975-2016. Dotted line indicates overall mean. A Mann-Kendall trend analysis indicates that there is no discernible trend.

Source: Bureau of Meteorology (2016).

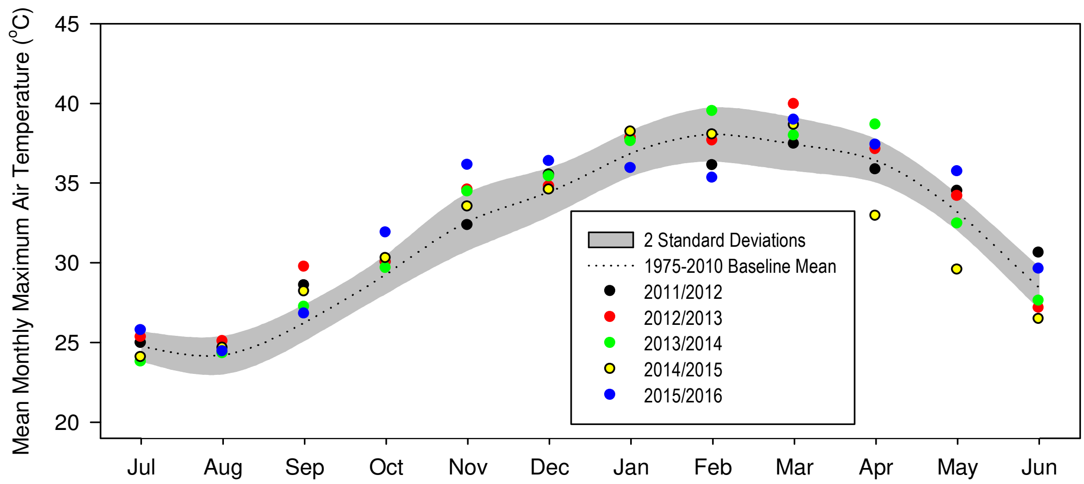

Figure 5.6: Mean monthly maximum air temperature from July 2010 to June 2016 recorded at Exmouth. Dashed line indicates baseline mean for 1975-2010 and grey shading indicates two standard deviations from the long term mean. Mean values beyond two standard deviations are considered 'anomalous.' Monthly means are based on daily maximum air temperature measurements.

Source: Bureau of Meteorology (2016).

5.4.2 Groundwater availability

Rainfall recorded at Exmouth township (Bureau of Meteorology 2016) is being used as a representative measure of groundwater availability in the Mangrove Bay area. While it is not a direct measure of groundwater availability and there can be significant variation in rainfall throughout the Ningaloo region, reliable rainfall data only exists at BoM weather stations and Exmouth is the best available surrogate for above and below ground water availability. Annual rainfall at Exmouth has been variable across the monitoring period, but exhibits no clear increasing or decreasing trend over time (Figure 5.3). However, when examined across the mangrove condition monitoring period (2006-2014), there were two years of extremely high annual rainfall (2008 and 2011) and two years of above average rainfall (2013 and 2014) indicating that there was likely to be a high potential for groundwater recharge across this period.

In April 2014, there was an anomalous rain event above two standard deviations (95th percentile) followed by a wet May (Figure 5.4) which caused extensive flooding and some sedimentation at Mangrove Bay.

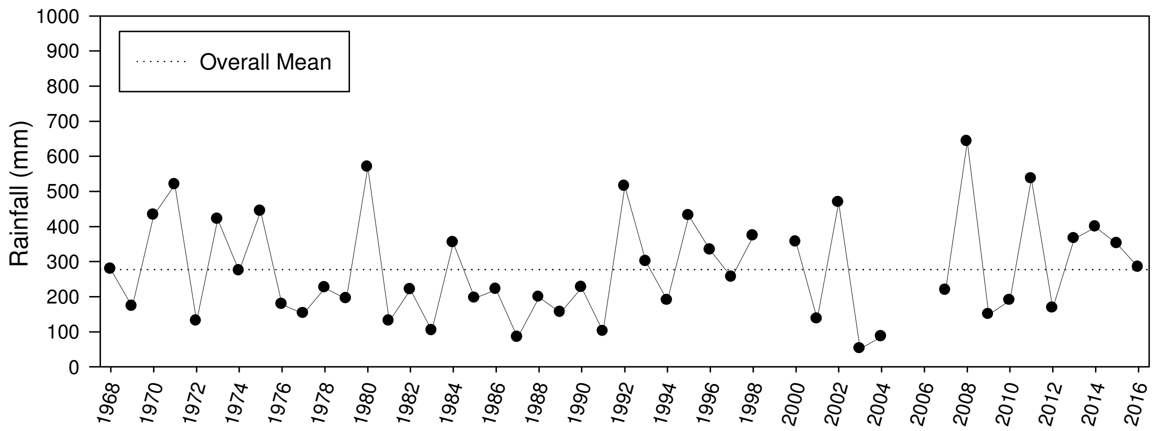

Figure 5.3: Total annual rainfall for Exmouth, 1968-2016. Black dashed line indicates overall mean rainfall for 1986-2010.

Source: Bureau of Meteorology (2016).

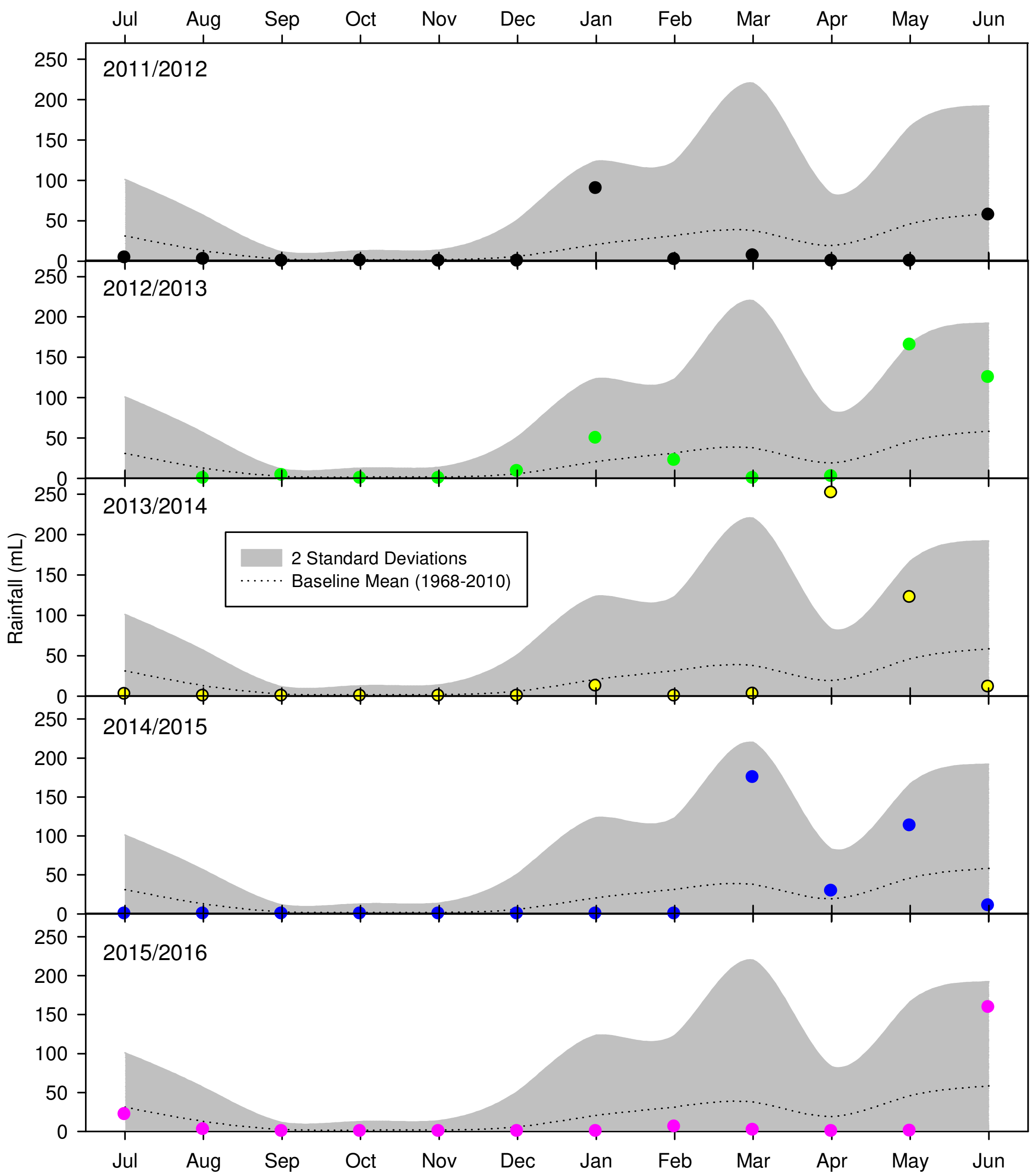

Figure 5.4: Total monthly rainfall for July 2011 – June 2016 recorded at Exmouth. Dotted line indicates baseline rainfall means (1968-2010) and the shaded area indicates two standard deviations from the mean, with values outside this considered anomalous.

Source: Bureau of Meteorology (2016).

5.4.3 Sea Level

Trends in sea level between 1998 and 2015 display medium-term oscillations resulting from regional climatic cycles associated with the Southern Oscillation Index (Figure 5.7). Annual mean sea level has ranged from ~1.41m in 2003, to ~1.61m in 2011. 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 (2006-2014), there was an increasing trend in sea level from 2006 to 2011, followed by a plateau from 2011-2013, and a decline from 2013 to 2014.

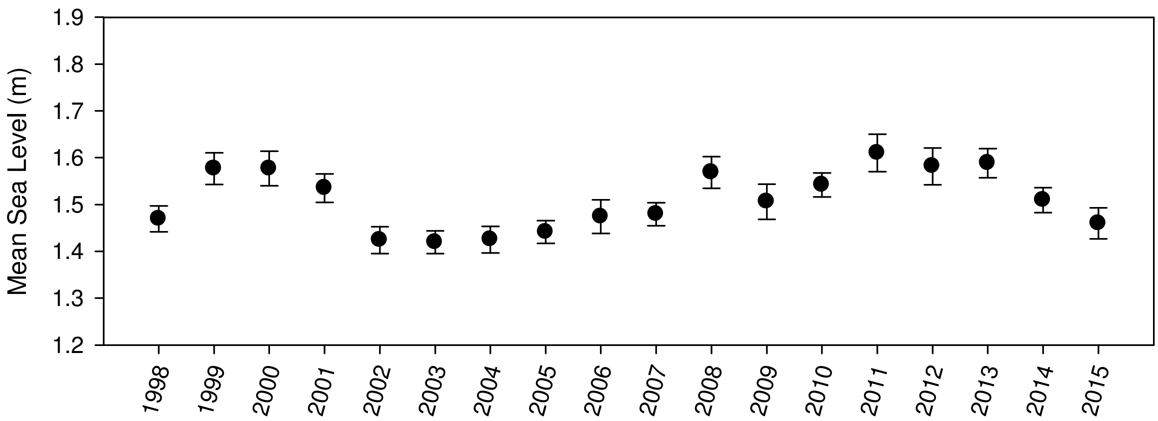

Figure 5.7: Mean annual sea level (± 1 SE) at Exmouth marina, 1998-2015. Mean values are calculated based on monthly sea level recordings within each year (annual n = 12), and measured above Tide Gauge Zero adjusted to the Australian Height Datum.

Source: Bureau of Meteorology, 2017.

5.4.4 Cyclones

Cyclone frequency and intensity information were acquired from BoM (http://www.bom.gov.au/cyclone/history/tracks/index.shtml). All cyclones classified as category 1 or above that passed within a 50km and 200km radius of Ningaloo (using Mangrove Bay as a centre point) were identified as potential sources of wind damage to mangroves in the Ningaloo marine reserves (Paling *et al.*, 2008).

Three cyclones passed within a 50km radius of the Ningaloo marine reserves between 1999 and 2006; Vance in 1999, Carlos in 2011, and Olwyn in 2015. Another four cyclones passed within a 50-200km radius; Steve in 2000, Alistair in 2001, and Daryl and Glenda in 2006.

5.5 Synthesis

The mangroves at Mangrove Bay have steadily increased in extent since 2006. This is most likely due to favourable conditions created by increasing sea level and generally high annual rainfall across the monitoring period (2006-2014). Both of these factors have the potential to lower porewater salinity, and changes in sea level have specifically been shown to have a close inverse relationship with porewater salinity at this location (Lovelock *et al.*, 2017). Lovelock *et al.* (2017) also found a close relationship between porewater salinity and canopy condition (with a delayed response of approximately 2 years), highlighting that sea level is likely to be a major driver of overall mangrove condition at Mangrove Bay. As sea levels oscillate over medium time-frames (10-20 years) due to regional climate processes (Southern Oscillation Index), this is likely to mean that there is a degree of natural fluctuation in mangrove condition over similar time scales. However, this same study noted that canopy condition was far higher in 2013/2014 than during the last period of comparably high sea level in 2001/2002. Given the relatively high levels of annual rainfall recorded immediately prior to 2013/2014, it is possible that this factor further decreased salinities and contributed to positive growth conditions on the landward fringes. While temperature regimes also play a key role in structuring mangrove communities in WA (Osland *et al.*, 2017), this is less likely to have resulted in the observed increases at this location given the variability in this indicator between 2006 and 2014.

While extent continued to increase up to 2014, the monitoring data shows that the peak in canopy density occurred in 2008 (recorded as a high proportion of dense PFC) and has since declined, particularly between 2010 and 2014. The peak in 2008 was likely a result of high rainfall recorded during the same year causing a short-term spike in foliage growth. The decline in the condition of canopy density observed in 2014 (recorded as a large increase in the proportion of sparse PFC) was likely due to a combination of increasing porewater salinity (which began increasing again in 2014 as a result of a decline in sea levels due to natural oscillations; Lovelock *et al.*, 2017) and a single high intensity rainfall event that occurred in April 2014, which is known to have resulted in unusually high levels of terrestrial runoff and sedimentation at the southern end of Mangrove Bay. Field observations indicate that this caused localised erosion and smothered mangrove pneumatophores, resulting in condition declines in the affected area at the time of satellite image capture in August 2014. More recent field observations indicate that this may have subsequently caused the death of some trees, although the extent of this needs to be confirmed via analysis of more recent satellite imagery.

Cyclones can damage mangroves by stripping leaves, breaking branches and uprooting trees (Lee, 2003; Gilman *et al.*, 2008; Paling *et al.* 2008; McIvor *et al.*, 2012). While it is difficult to make definitive conclusions due to the gap in monitoring between 2010 and 2014, it is possible that cyclone Carlos damaged the mangroves in 2011 and contributed to the decline in canopy density recorded in 2014. The potential effects of cyclone Olwyn (2015) are currently unknown, with more recent assessments of condition required.

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