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Monitoring dugongs within the Reef 2050 Integrated Monitoring and Reporting Program:

Final Report of the Dugong Team in the Megafauna Expert Group

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The Great Barrier Reef Marine Park Authority acknowledges the continuing sea country management and custodianship of the Great Barrier Reef by Aboriginal and Torres Strait Islander Traditional Owners whose rich cultures, heritage values, enduring connections and shared efforts protect the Reef for future generations.

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Executive Summary

- The dugong (*Dugong dugon*) is a coastal marine mammal of high conservation value as the only herbivorous mammal that is strictly marine. The dugong also has high cultural values to Indigenous Australians and is a cultural keystone species.

- The dugong is protected under the Commonwealth Environmental Protection and Biodiversity Conservation Act as a matter of national environmental significance because of its listing as a migratory and marine species.

- The Great Barrier Reef World Heritage Area (the World Heritage Area) supports one of the world's largest population of the dugong (*Dugong dugon*), and the importance of the Great Barrier Reef (the Reef) for dugongs was one of the reasons for the region's World Heritage listing.

- The size of the dugong population in the World Heritage Area is believed to be much less than at the time of European settlement, especially along the urban coast, from Port Douglas south.

- In the World Heritage Area, dugongs are impacted by:
  - indirect pressures that affect their growth, fecundity, movements and mortality by causing changes in the status of the seagrass communities on which they depend for food, and
  - direct pressures that cause mortality.

- The greatest risks to dugongs are from anthropogenic activities that kill adult animals.

- Dugongs have been monitored along the Queensland coast since the 1980s using a series of standardised aerial surveys. The surveys have been loosely coordinated across jurisdictions and largely conducted at the same time of year at approximately five-year intervals. The Queensland coast from the New South Wales border to Cooktown has generally been surveyed in one year, and from of Cooktown through central and western Torres Strait in an adjacent year, while the Gulf of Carpentaria has been surveyed on an opportunity basis. The areas adjacent to the Reef have been included to account for temporary migrations of dugongs across jurisdictional boundaries.

- These surveys have provided long-term information on the distribution and abundance of dugongs, which has informed management. The *Reef 2050 Long-Term Sustainability Plan* (Reef 2050 Plan) commits to continue to survey the dugong population every five years.

- Retrospective Bayesian analyses of the aerial survey data suggest that there was:
  - 0.791 probability (marginally less than the conventional 0.8 power target) that dugong abundance declined in the Reef north of Cooktown between 2005

and 2013. The estimated trend was minus 3.14 per cent per year (SE: 3.844; 95%CI -11.057 - +4.552)
  - 0.689 probability that dugong abundance declined in the Reef south of Cooktown between 2005 and 2016. The estimated trend was minus 1.460 per cent, per annum (SE: 3.345; 95%CI -7.531 – + 5.912)
  - an aggregate decline across the entire Reef of minus 2.301 per cent, per annum (SE: 3.699; 95%CI -9.737 – +5.379) between 2005 and 2016.

- Prospective Baysian power analyses suggest that a minus three per cent per annum decline could be detected with 0.8 probability at intermediate time-horizons (eight years and greater) but shorter time-scales and more frequent surveys are unlikely to provide the requisite power to detect trends, especially if the decline is shallow (for example, minus one per cent per annum).

- Thus, it is very important that dugong monitoring be conducted over long time scales and that the data obtained using new technologies are compatible with the historical data.

- For reasons of human safety and technological superiority, the manned surveys should be replaced by surveys using a large fixed wing unmanned aerial vehicle such as the *SeaEagle* as soon as possible. The work required to compare the *ScanEagle's* capacity to detect dugongs in the Reef with that of the surveys conducted using manned aircraft is detailed in Appendix 4 of this report. Such comparisons are essential to enable a transition to the new technology without losing the value of the historical data sets.

- We recommend that the five-year survey frequency be maintained for the dugong aerial surveys (conducted by manned aircraft or unmanned aerial vehicles) because of:
  - the limited improvement in power achieved by increasing the survey frequency;
  - the statutory five-year reporting period required by the *Great Barrier Reef Marine Park Act 1975* for the Great Barrier Reef Outlook Report; and
  - the requirements of the Reef 2050 Plan.

- The aerial surveys can provide the following indicators of trends in the status of the dugong in the Reef:
  - population size (standardised relative abundance);
  - relative density using the various techniques to correct for standardised relative abundance;
  - percentage calves, an index of fecundity and neonatal survivorship over the previous two years;
  - Area of occupancy; and
  - Percentage overlap of high and medium density dugong habitats as revealed by spatial risk assessment using aerial survey data and spatial data on sources of mortality such as Traditional hunting, gill netting and vessel strike.

- The aerial surveys can also be used to monitor the distribution, abundance and threat exposure of in-water large juvenile and adult turtles in the survey areas.

- The aerial surveys should use the optimised survey design outlined in this report and be coordinated with surveys of the dugong's seagrass habitats.

- The results of the surveys should be used as one of several lines of evidence used to monitor trends in dugong abundance in the Reef. The other lines of evidence should include:

    - Data on dugong mortalities from the Queensland government StrandNet Program, which monitors the distribution and numbers of dugong mortalities and causes of death per month per year, mainly from urban coast. This program also provides evidence of the status of other marine megafauna.
    - Vessel surveys conducted by Indigenous rangers in their sea country and (if Indigenous communities agree) catch monitoring to provide important local scale information on the status of dugongs, dolphins and large marine turtles. Such surveys will not be suitable to monitor dugong at scale of the Reef.
    - Changes in the density of dugong feeding trails, which are an indicator of excavating feeding by dugongs at shallow inshore sites. This information should be used to monitor the dugong's response to local threats (for example, capital dredging) at appropriate sites in concert with reactive seagrass monitoring at those sites.

1.0 Introduction

The *Reef 2050 Long-Term Sustainability Plan (Reef 2050 Plan)* (Commonwealth of Australia 2015) establishes a Reef 2050 Integrated Monitoring and Reporting Program (RIMReP) to measure and report progress, and guide adaptive management towards achieving the outcomes, objectives and targets of the Reef 2050 Plan. The Reef 2050 Plan (Commonwealth of Australia, 2015) will be reviewed on a five-year cycle with an initial mid-term review completed in 2018. RIMReP has established a number of expert groups, including the megafauna working group, which consists of several specialised teams, including one for the dugong (*Dugong dugon*).

1.1 Objectives of the dugong report

The objectives of this report are to determine for the dugong:

- An assessment of the current status of the relevant elements of the Great Barrier Reef (the Reef), including an evaluation of primary drivers, pressures and responses using the Driving Forces, Pressures, States, Impacts, Responses (DPSIR) Framework;
- Identification of priority indicators for monitoring the key values associated with these elements;
- Summary of potential sources of data;
- Evaluation of adequacy of existing monitoring activities within each theme to achieve the objectives and requirements of RIMReP;
- Recommendations for the design of an integrated monitoring program as a component of RIMReP, specifically considering:
    - The information requirements for each key element of the Reef to ensure that appropriate data and information are being collected to meet the fundamental objectives of RIMReP;
    - The spatial and temporal sampling design to ensure that greatest value can be extracted from the data collected;
    - The logistics of the design to ensure that it can be implemented efficiently;
    - Likely funding required to implement the recommended monitoring design (RIMReP Expert Group Project Megafauna EOI 0504117).

1.2 The modified DPSIR Framework

DPSIR (Drivers, Pressures, State, Impact, Response model of intervention) is a cause-and-effect framework for describing the interactions between society and environment. During the workshop held on 15th to 16th February 2017, a slightly modified version of the conceptual diagram adopted from Strawman was developed (Figure 1). The diagram depicts the causal links between nodes (Drivers, Activities, Pressures and States) more explicitly.

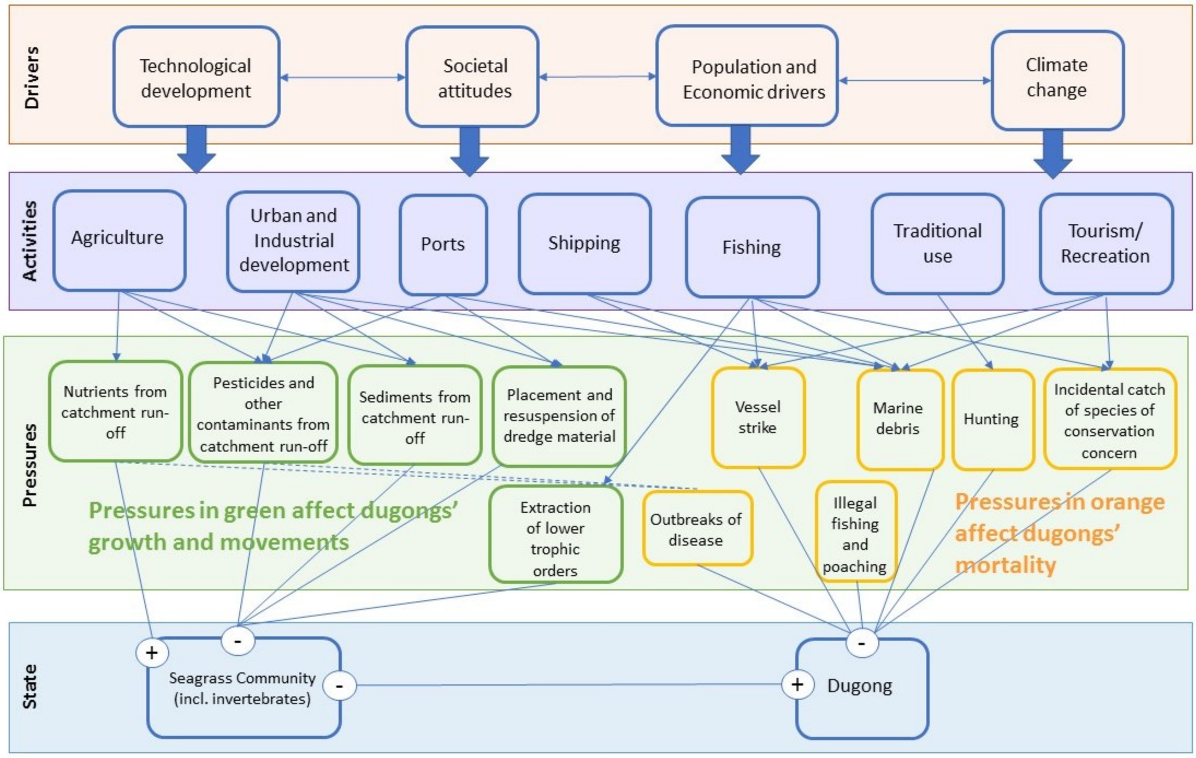

Figure 1. Schematic diagram of the relationships among drivers, activities, pressures and the state of dugong populations in the Reef. The diagram was developed during the February 2017 workshop process.

1.3 Drivers

Drivers are overarching causes that can drive change in the environment (State of the Environment 2011, Great Barrier Reef Marine Park Authority 2014 and 2017). Six drivers of change have been identified for the Reef system, all of which can operate across a range of scales (both in time and space), and are interlinked (Great Barrier Reef Marine Park Authority 2017). The drivers for the dugong DPSIR conceptual model (Figure 1) have been re-arranged to the following four:

1. Technological development
2. Societal attitudes
3. Population and economic drivers
4. Climate change

1.4 Activities

Activities, referred as Pressures in the DPSIR Framework, are the change mechanisms that result from Drivers (Great Barrier Reef Marine Park Authority 2017). For dugongs there are seven activities:

1. Agriculture
2. Urban and industrial development
3. Ports
4. Shipping

5. Fishing
6. Traditional use
7. Tourism/Recreation

One activity is often linked with multiple pressures and one Pressure may be linked to multiple activities (Figure 1).

1.5 Pressures

Dugongs are affected by two broad categories of pressures:

A) Indirect pressures that affect their growth, fecundity, movements and mortality by causing changes in the status of the seagrass communities on which dugongs depend for food;
B) Direct pressures that cause dugong mortality. Studies of dugong demography (Marsh et al. 2011 and below) indicated that the greatest risks to dugongs are from anthropogenic activities that kill adult animals.

Indirect pressures affecting dugong growth, fecundity, movements and mortality by causing changes in the status of the seagrass communities are listed below in no particular order:

1. Nutrients from catchment run-off
2. Pesticides and other contaminants from catchment run-off
3. Sediments from catchment run-off
4. Placement and resuspension of dredge materials
5. Direct loss of seagrass through reclamation, trawling or dredging

Direct pressures that cause dugong mortality in no particular order are:

1. Vessel strike
2. Traditional Hunting
3. Incidental catch
4. Marine debris
5. Illegal fishing, hunting and poaching
6. Outbreaks of disease

1.6 State

As a result of pressures, the ‘state of the environment’ is affected, and the ‘state’ is the combination of the physical, chemical and biological conditions. For dugongs, the relevant states include:

1. Status of the seagrass community
2. Status of the dugong populations

The seagrass community is positively impacted by one of the indirect pressures (nutrients from catchment run-off); and negatively impacted by the remainder (pesticides and other contaminants from catchment run-off, sediments from catchment run-off, placement and resuspension of dredge material, and direct loss of seagrass communities through reclamation, trawling or dredging).

The dugong is negatively impacted by all of the six direct pressures listed above.

2.0 Current understanding of dugong biology

The dugong is a species of high conservation, cultural and biological value, occurring over a large range in the tropical and subtropical Indo-West Pacific Ocean from East Africa to the Solomon Islands and Vanuatu (Marsh et al. 2011). Marsh *et al.* (2011) made a crude estimate of the dugong's extent of occurrence (EOO) as 860,000 square kilometres based on potential habitat (which they defined as waters less than 10 metres deep[1] in its known range). They estimated that the dugong's range spans approximately 128,000 kilometres of coastline across at least 37, and up to 44, countries and territories (IUCN 2017).

The dugong is the only extant species in the family Dugongidae and one of only four extant species in the order Sirenia. The dugong's closest relative, Steller's sea cow (*Hydrodamalis gigas*: Dugongidae), was hunted to extinction in the 18th century, some 27 years after being rediscovered by ship-wrecked sealers (Stejneger 1887). Steller's sea cow had once been distributed in coastal waters from California to Japan but by the time of its rediscovery was restricted to a relict population on the coastal fringes of two islands near the Kamchatka Peninsula of Siberia (Marsh et al. 2011). This extinction has highlighted the vulnerability of sirenian species to human-induced mortality, including wide-ranging species such as the dugong.

Queensland waters including the Great Barrier Reef World Heritage Area (the World Heritage Area) are believed to support one of the world's largest population of the dugong (*Dugong dugon*), a coastal marine mammal that feeds primarily on seagrasses and are seagrass community specialists (Marsh et al. 2011). The dugong is listed as vulnerable to extinction by the International Union for Conservation of Nature (IUCN 2017) and under the Queensland Nature Conservation Act 1992. The dugong is not listed as threatened under the Australian Government's Environment Protection and Biodiversity Act 1999 (EPBC Act). Nonetheless, Australia has an international obligation to conserve the dugongs in its waters because of the dugong's listing in Appendix 11 of the Convention of Migratory Species. The importance of the Reef for dugongs was one of the reasons for its World Heritage listing (Great Barrier Reef Marine Park Authority 1981). Thus, the dugong is protected under the EPBC Act as a matter of national environmental significance because of its listing as a migratory and marine species. The dugong also has high cultural values to Indigenous Australians and is regarded as a cultural keystone species (Butler et al. 2012).

Dugongs occur along the entire Queensland coast from the New South Wales border to Torres Strait and the Gulf of Carpentaria (Marsh et al. 2011). Dugongs are protected by the extensive network of ecosystem-scale Marine Protected Areas (MPAs) including the Moreton Bay Marine Park, the Great Sandy Marine Park, the Great Barrier Reef Marine Park (the Marine Park) and the Great Barrier Reef Coast Marine Park (Marsh et al. 2011). A large area of Torres Strait is designated as The Torres Strait Protected Zone where various regulations are implemented by the Protected Zone Joint Authority to conserve dugongs and marine turtles (for example, Dugong Sanctuary where dugong hunting is banned) (Marsh et al. 2011).

[1] This estimate is likely conservative as dugongs habitats can extend to greater than 30 metres in some areas

The relative abundance and distribution of dugongs have been recorded along the Queensland coast since the 1980s using a series of standardised aerial surveys. The surveys have been loosely coordinated across jurisdictions and largely conducted at the same time of year at approximately five-year intervals. The Queensland coast from the Queensland- New South Wales border to Cooktown has generally been surveyed in one year, and from of Cooktown to Torres Strait in a second survey in an adjacent year while the Gulf of Carpentaria has been surveyed on an opportunity basis. These surveys have provided long-term information on the distribution and abundance of dugongs and the *Reef 2050 Plan* (Commonwealth of Australia 2015) commits to continue to survey the dugong population every five years.

2.1 Genetic structure

Mitochondrial DNA analyses (Blair *et al.* 2014, 2015) show that Australian dugong populations are not pammitic and are genetically distinct from those in other parts of the world. There are two main mitochondrial lineages in Australia. One lineage is found almost exclusively along the east coast of Queensland, the other throughout the entire Australian range of the species. Long periods of isolation during Pleistocene glacial cycles are likely drivers of this separation. Australian dugongs are genetically diverse especially in the Torres Strait and North Queensland regions. Rather unexpectedly, strong population structure has been detected using both mitochondrial sequence data and microsatellites, despite the known ability of individual dugongs to travel long distances.

Seddon et al. (2014) used microsatellite markers to demonstrate low, but significant, population differentiation in southern Queensland. Recent unpublished research by Alexandra McGowan and David Blair et al. using the same genetic markers indicate a significant genetic break along the east cost of Queensland around the Whitsundays, suggesting that there are at least two dugong stocks in the Reef region, north and south of that break.

2.2 Demography

Dugongs are long-lived and slow breeding (Marsh et al. 2011). Carcass analysis indicates that females typically bear one young at a time at intervals of 2.6-6.8 years, after a pre-reproductive period ranging from 6->17 years. The mean generation time is estimated to be 22-25 years. Extreme weather events (for example, cyclones and flooding) have been negatively associated with the following impacts on dugongs at sub-regional (hundreds of kilometres) scales: mass stranding, increased movements presumably in search of food, loss of weight and fat, delayed reproduction and mortality (Heinshohn et al. 1974, Marsh 1989, Preen and Marsh 1995, Marsh and Kwan 2008, Meaguer and Limpus 2014, Fuentes et al. 2016). The relationships between the proportion of dependent calves and the climatic drivers vary spatially and temporally (Fuentes et al. 2016).

2.3 Diet

Dugongs exploit a relatively wide diet including seagrasses, macro-invertebrates and algae within intertidal and subtidal tropical and subtropical seagrass communities (Marsh et al. 2011 and 2018a). The importance of seagrass genera to dugongs differs among locations and may change at the same location during times of seagrass loss. Dugongs feed by excavating or cropping, depending on seagrass morphology and the nature of the sediment. An individual dugong consumes approximately seven per cent of their body weight of seagrass per day (Goto et al. 2004) and can disturb a considerable area of seagrass in a single day, especially in areas with low biomass (Preen 1995). Marked temporal fluctuations in dugong mortality and fecundity (Marsh and Kwan 2008, Meager and Limpus 2014, Fuentes et al. 2015) track major changes in the seagrass communities on which dugongs depend for food.

2.4 Movements

Satellite tracking studies have shown that dugongs undertake long-distance movements of up to 560 kilometres within the Reef region (Sheppard *et al.* 2006). Nonetheless, dugongs do not appear to undertake regular migrations and movements are individualistic (Marsh and Rathbun 1990, Sheppard et al. 2006, Marsh et al. 2011, Gredzens et al. 2014; Cleguer et al. 2015 a and b; Zeh et el. 2015, 2016). Some dugongs are highly mobile and move between regions, especially between Hervey Bay and Moreton Bay in south-east Queensland (Zeh et al. 2016) and Hervey Bay and Shoalwater Bay in the Reef (Sheppard et al. 2006) as well as within the Reef (Marsh and Rathbun 1990, Sheppard et al. 2006, Marsh *et al.* 2011, Gredzens et al. 2014; Cleguer et al. 2015 a and b). Movements have not yet been confirmed across the Whitsundays' genetic break region, although they have been inferred from aerial survey results (Sobtzick et al. 2017). Some dugongs have been relatively sedentary during their tracking period (generally a few months — for example, see Sheppard et al. 2006) but up to 16 months (Marsh and Rathbun 1990). Cope et al. (2015) used pedigree analysis based on individual genetic markers to infer the movements of dugongs between locations in southeast Queensland including Moreton Bay and Hervey Bay. They found that approximately 30 per cent of assigned parents had at least one offspring found in a different locality, implying recent movement of the parent or offspring. This analysis suggested markedly more movement between localities than detected through repeated direct sampling of individuals (Seddon et al. 2014) or through telemetry.

Movements have also been inferred from aerial survey results, especially in the southern Reef where the differences between surveys have been too large to be explainable by dugong demography or survey conditions — for example, see Sobtzick et al. (2012, 2017).

3.0 Current understanding of dugong status in the Great Barrier Reef

3.1 Aerial surveys

Since the 1980s, large scale aerial surveys have provided temporal and spatial information on dugong distribution and abundance in the World Heritage Area along with many other parts of their distribution in Australia. These surveys have been a cost-effective means of assessing the status of the dugong population by providing standardised relative indices of distribution and abundance at vast spatial scales (more than tens of thousands of square kilometres). The surveys have also recorded the percentage of attendant calves, an index of fecundity and neonatal survivorship (Fuentes et al. 2016).

Dugong management on the east coast of Queensland is largely conducted on a jurisdictional basis with separate (and very different) management arrangements for Torres Strait (north of 10.68ºS), the Reef Region (10.68ºS to 24.5ºS) and South East Queensland (24.5 ºS to the Queensland-New South Wales border) (see Marsh et al. 2011 for details). Dugongs make individualistic movements across these jurisdictions (see section on Movements above). In addition, the whole east coast of Queensland has proved too big to survey in a single field season using manned aircraft because of the limited availability of appropriate aircraft, trained observers and funding. For these reasons, aerial survey monitoring of dugong distribution and abundance has been coordinated across jurisdictions at the same time of year over two years at approximately five year intervals. The remote Reef Region north of 16.5ºS has generally been surveyed in the same year as Torres Strait; the entire east Queensland coast south from 16.5ºS in another year. For logistical reasons, the coastline north of north of Shelburne Bay (latitude 11º 54' S) to Newcastle Bay (latitude 11º S) has only been surveyed using a zig-zag shoreline surveys, rather than transects perpendicular to the coast used in other areas. The offshore areas of the World Heritage Area have never been surveyed for dugongs. We consider that neither of these inadequately surveyed areas is likely to be significant dugong habitat.

The survey design has remained largely constant since the beginning of the time series[2] and the field methodology has continued to follow the strip transect aerial survey technique detailed in Marsh and Sinclair (1989) with only minor modifications to take advantages of improved technology. In contrast, the methods for correcting for the detection biases inherent in aerial surveys have improved over time enabled by field experiments external to the surveys (Pollock et al. 2006; Hagihara et al. 2014, 2018; Sobtzick et al. 2015) and the collection of additional data on environmental conditions for surveys conducted from the early 2000s. Three methods have been used to calculate standardised estimates of relative dugong abundance: (1) Marsh and Sinclair (1989) which has been used in dugong aerial surveys since 1986; (2) Pollock et al. (2006) and Hagihara et al. (2014 and 2018). All methods attempt to correct for availability bias (animals not available to observers because of water turbidity), and perception bias (animals visible in the survey transect but missed by observers (terminology established by Marsh and Sinclair, 1989). The most substantive changes have been in the method to correct for availability bias. The Marsh and Sinclair (1989) method averages across conditions within surveys and corrects for environmental differences in availability bias at the scale of an entire survey. The Pollock et al. (2006) method corrects for availability bias at much finer spatial scales by correcting for the spatial heterogeneity in sighting conditions for each group of dugongs. The Hagihara et al. method (2014, 2018) additionally corrects for differences in dugong diving behaviour at different depths. We consider the Hagihara et al. method (2014, 2018) to be superior to the other two methods but it can only be applied on data collected from the early 2000s because the necessary extra data on environmental conditions were not collected during the earlier surveys as explained above.

3.1.1 Northern Great Barrier Reef

The dugong habitats from Cape Bedford through to north of Shelburne Bay (latitude 11º 54' S) were last surveyed in 2013[3] in conjunction with a survey of the western and central

[2] There has been some adaptive rationalisation of transect length and survey intensity. These changes have been minor in the Reef.

[3] The northern Great Barrier Reef will be resurveyed in 2018

regions of Torres St (Sobtzick et al. 2014). The 2013 survey confirmed that the Cape York coast of the World Heritage Area supports globally significant populations of dugongs and includes very large areas of very high and high relative dugong density (Sobtzick et al. 2014). The standardised relative population estimate for the northern Reef was recalculated for this report using the Hagihara *et al.* (2014, 2018) methodology which uses depth-corrected estimates for availability bias. The resultant 2013 estimate (6,558 ± s.e.1,114) was lower than the corresponding 2006 estimate (8449 ± s.e1,803) using the same methodology.

Sobtzick et al. (2014) used data corrected using the Marsh and Sinclair methodology and frequentist statistics to test for significant trends in dugong abundance in the northern Reef since the time series of surveys began in 1987. The differences between surveys were not significant. However, Sobtzick et al. (2014) emphasised that these results should be interpreted with caution because of the difficulty in detecting significant trends in marine mammal populations using this approach unless the trends are large (Taylor et al 2007).

In view of this difficulty, Rankin (Appendix 1) conducted a trend analysis of the data from the 2006 and 2013 surveys relevant to the optimised survey design (see Appendix 2) using Bayesian models for this report. These models incorporated multiple sources of uncertainty, such as imperfect detections (perception bias *sensu* Pollock et al. 2006), availability-bias (*sensu* Hagihara et al. 2014, 2018), parameter uncertainty and model-selection uncertainty, as estimated in previous studies (Marsh and Sinclair 1989, Pollock et al. 2006, Hagihara et al. 2014, 2018). While the analyses detailed in Appendix 1 are Bayesian in computation and interpretation, they are somewhat analogous to conventional 'power analyses' from the Neyman-Pearson school of frequentist statistics. The Bayesian analyses suggest that there was a 0.791 probability (marginally less than the conventional 0.8 power target) that dugong abundance declined in the northern Reef between 2005 and 2013. The estimated trend was minus 3.14 per cent, per year (SE: 3.844; 95%CI -11.057 - +4.552).

3.1.2 Southern Great Barrier Reef

Estimating the trend for the southern Reef is more complex than for the northern Reef. The results for the 2011 survey (Sobtzick et al. 2012) are aberrant, a result largely attributable to dugongs temporarily migrating out of the southern Reef survey area as a result of the poor condition of the seagrass communities on which they depend at the time of the survey. We consider that a more meaningful comparison is between the result for the 2005 and 2016 surveys (adjusted for the optimised survey design) using the Bayesian analysis described in Appendix 1. This analysis suggests that there was 0.689 probability that dugong abundance declined in the southern Reef between 2005 and 2016. The estimated trend was minus 1.460 per cent per annum (SE: 3.345; 95% CI -7.531 – + 5.912).

3.1.3 Whole Great Barrier Reef

The Bayesian analysis indicated that the aggregate trend was minus 2.301 per cent per annum (SE: 3.699; 95%CI -9.737 – +5.379). This result is approximate only because the northern and the southern Reef were not surveyed in the same years (Appendix 1).

3.2 Calf Counts

Environmental and climatic drivers influence key demographic parameters of the dugong. Extreme weather events (for example, cyclones and flooding) have been associated with the following impacts on dugongs: mass stranding, increased movements presumably in search of food, loss of weight and fat, delayed reproduction and mortality (see Marsh et al. 2011 and Meager and Limpus 2014 for details). As explained by Fuentes et al. (2016), the proportion of dependant calves sighted during an aerial survey (calf production) is a reflection of:

(1) births (which are expected to reflect the effect on female fecundity of environmental conditions over the previous several years); and

(2) neonatal survivorship (which can be affected by the more immediate effect of an extreme weather event on the mortality of both mothers and calves as a result of mass stranding associated with a storm surge as well as starvation due to loss of seagrass beds).

Fuentes et al. (2016) investigated how the proportions of dugong calves recorded during the James Cook University (JCU) time series of dugong aerial surveys were associated with various sub-regional and ocean-basin climatic covariates at a range of spatially distinct sub-regions along the east coast of Queensland including the southern Reef and the northern Reef. The relationships between the proportion of dependent calves and the climatic drivers varied spatially and temporally, with climatic drivers influencing calf counts at sub-regional scales. In the northern Reef, the proportion of calves declined in association with the increase in indices of the El Niño phenomenon: both the Southern Oscillation Index (lagged to four years) and Niño 3.4 (lagged to one year). In the southern Reef, the proportion of calves declined with: 1) increasing rainfall above the long-term average (lagged to two and three years); and 2) increases in Niño 3.4 (lagged two years).

In the northern Reef, the proportion of dugong calves declined between 2000 and 2013 (Figure 2). In contrast in Torres Strait, which is north of the main cyclone belt, the percentage of calves increased to 17.9 per cent in 2013. This result suggests that the decline in the dugong population indicated by the Bayesian analysis above is linked to habitat loss. However, the data on the status of seagrass in the northern Reef are not adequate to further evaluate this inference.

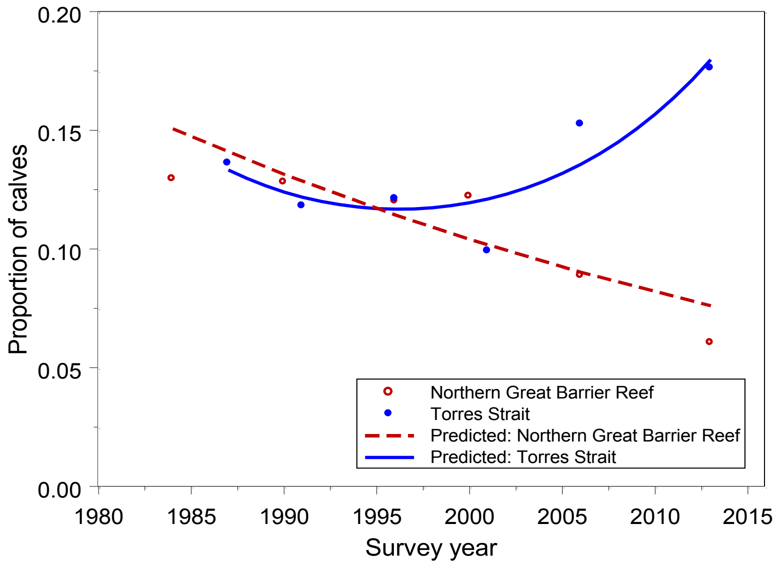

Figure 2. Proportions of calves (y value) plotted against survey year in northern Great Barrier Reef (red open circle) and Torres Strait (blue closed circle). Each line represents the proportion of calves for the northern Reef (dotted line) and Torres Strait (solid line) predicted by logistic regression. (Reproduced from Sobtzick et al. 2017). The patters are clearly very different in the two regions.

In the southern Reef, the percentage of calves increased from zero in 2011 to more than 10 per cent of the population in 2016, a value similar to that in 2005 and within the normal range (Figure 3; Sobtzick et al 2017). The results of this survey add to the evidence that the dugongs in the survey region were in much better condition in 2016 than at the time of the 2011 survey. This change was coincident with improvements in the condition of intertidal seagrass percentage cover which increased in the southern Reef (except in the Wet Tropics; McKenzie et al. 2016) and a reduction in the number of dugong carcasses reported to the Queensland marine wildlife stranding program StrandNet (Meager 2016).

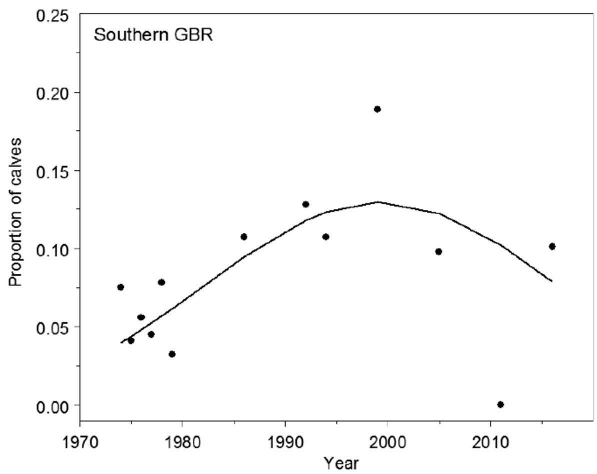

Figure 3. Proportion of calves plotted against survey year for the southern Great Barrier Reef. The line represents the proportion of calves predicted by logistic regression (reproduced from Sobtzick et al. 2017). Note the big increase between 2011 and 2016.

3.3 Spatial distribution of dugongs with respect to protected areas and threats

The information from aerial surveys and/or satellite and acoustic tracking has been used to design and evaluate the effectiveness of management areas in the Reef, including the Dugong Protection Areas (Marsh 2000) and the rezoning of the Reef in 2003 (Fernandes et al. 2003, Dobbs et al. 2008 and McCook et al. 2010).

Grech and her co-workers have developed spatial models using coastal seagrass mapping and dugong distribution and abundance data at the scale of the coastal waters of the entire Reef region. They then overlaid information on the spatial distributions of threats to dugongs and their habitats such as gill-netting, hunting, vessel strike and low-quality terrestrial runoff in the GIS (Grech and Marsh 2007, 2008; Grech et al. 2008). Grech et al. (2008) evaluated the relative risks of various threats to the dugongs and their seagrass habitats. They used expert opinion, spatial information on the distribution of threats, and the spatial models of seagrass and dugongs to identify areas where human impacts posed low, medium and high relative risks to dugongs and their habitats. This technique allowed Grech and her group to explore methodically the ways in which the systematic removal of various threats would likely affect dugong status. The approach identified sites where gill–netting was still occurring in areas of high dugong density, despite the 2003 rezoning of the Marine Park (McCook et al. 2010). Information concerning these sites was provided to a review of the inshore gill net fishery. Spatial risk assessment has also been used to identify areas in the community management areas of Indigenous peoples where hunting does not occur at present because the areas are difficult and expensive to access in small boats. Marsh et al. (2015) has already applied this technique in Torres Strait to identify the high density dugong areas that are rarely and never hunted. Several workshops have been held with local Indigenous peoples, who have Native Title rights to hunt to discuss the possibility of declaring these regions 'no hunting areas' to pre-empt hunting expanding with improved technology.

3.4 StrandNet

StrandNet is an important component of the monitoring of marine wildlife including dugongs on the Queensland Coast, particularly the east coast between Port Douglas and the Queensland-New South Wales border (see Marsh et al. 2018b). The primary focus of the StrandNet database is to record information on where sick, injured, dying and dead animals have been found in Queensland and assess causes of injury and death. StrandNet is the main source of knowledge about human-related mortality factors affecting dugongs in Queensland. Meager and Limpus' (2014) analysis of the StrandNet data has resulted in important insights into the status of the dugong in the southern Reef, especially with regards to: (1) the environmental drivers underpinning stranding events which showed that peak mortality followed sustained periods of elevated freshwater discharge (9 months) and low air temperature (3 months). At a regional scale, these results translated into a strong relationship between annual mortality and an index of El Niño-Southern Oscillation (Meager and Limpus 2014); (2) long-term trends in strandings and bycatch in the Queensland Shark Control Program (Marsh et al. 2005) and health and disease in marine wildlife (for example, bacteria in dugongs, Nielsen et al. 2013). The generic values of StrandNet as a component of RIMReP are considered in detail by Marsh et al. (2018b).

4.0 Evaluation of the adequacy of current dugong monitoring in the Great Barrier Reef

4.1 Aerial surveys

Since the mid-1980s, aerial surveys for dugongs have monitored their: 1) relative abundance, 2) spatial distribution, 3) density and 4) neonatal mortality/fecundity (percentage calves) in six regions: Moreton Bay and Hervey Bay (adjacent to the southern Reef), southern and northern Reef, Torres Strait and Gulf of Carpentaria (adjacent to the Reef in the north), (Table 1 and Figure 4). As explained above, there have been some changes the design of dugong aerial surveys over the past 35 years, largely driven by adaptive monitoring and advances in technology as well as changes in the logistical and financial constraints.

Table 1. Years for which the archival dugong sightings were examined to optimise the survey design for the Great Barrier Reef surveys of dugongs as well as large juvenile and adult turtles which are recorded on the same surveys.

Region	Number of historical surveys	Year
Moreton Bay	8	Nov 1999 Dec 2000 Apr 2001 Nov 2001 Nov 2005 Nov 2011 Jul 2013 Nov 2016
Hervey Bay	11	Jul 1988 Nov 1992 Dec 1993 Nov 1994 Nov 1999 Apr 2001 Nov 2001 Nov 2005 Nov 2006 Nov 2011 Nov 2016
Southern Reef	8	Sep 1986 Sep 1987 Nov 1992 Nov 1994 Oct 1999 Nov 2005 Nov 2011 Nov 2016
Northern Reef	8	Nov 1984 Apr 1985 Nov 1985 Nov 1990 Nov 1995 Nov 2000 Nov 2006 Nov 2013
Torres Strait	9	Nov 1987 Nov 1991 Dec 1994 Nov 1996 Nov 2001 Nov 2005 Nov 2006 Mar 2011 Nov 2013

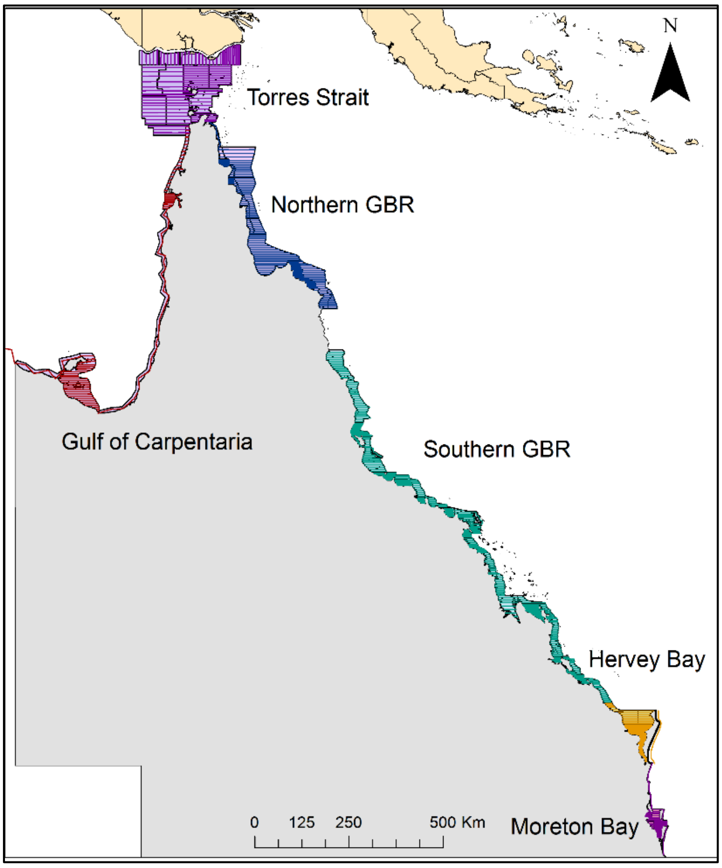

Figure 4. Map of dugong aerial survey regions in Queensland: Moreton Bay (pink), Hervey Bay (orange), southern Great Barrier Reef (green), northern Great Barrier Reef (blue), Torres Strait (purple) and Gulf of Carpentaria (red). The rectangle delineates the Marine Park boundary.

4.1.1 Improving the design of the aerial surveys for dugongs along the eastern coast of Queensland

Optimising the placement and length of transects

In accordance with the principles of adaptive monitoring, all available dugong sighting data (Table 2) were plotted in ArcGIS. Survey intensities in individual blocks were adjusted based on the distribution of the sightings. Survey intensities were reduced in the areas where there were few or no historical dugong sightings and increased in areas with numerous historical dugong sightings. The offshore ends of individual transects were truncated if no dugongs had ever been observed in the offshore area. Aerial survey block sizes were then adjusted to reflect the new survey design (Appendix 2).

Two new blocks were added to the survey design in areas that have not been surveyed to date in a systematic manner for logistical reasons: block C13 in the southern Reef region (between Cairns and Cooktown, Figure 5) and block N15 in the northern Reef (northernmost section of the east coast of the Peninsula, Figure 6). In Torres Strait, Block 3 was split into new blocks 3A and 3B (with increased survey intensity in 3A and increased block size for 3B, Figure 7). No adjustments were made to the survey designs for the Moreton Bay and Hervey Bay regions, which our results suggest already have optimum survey designs.

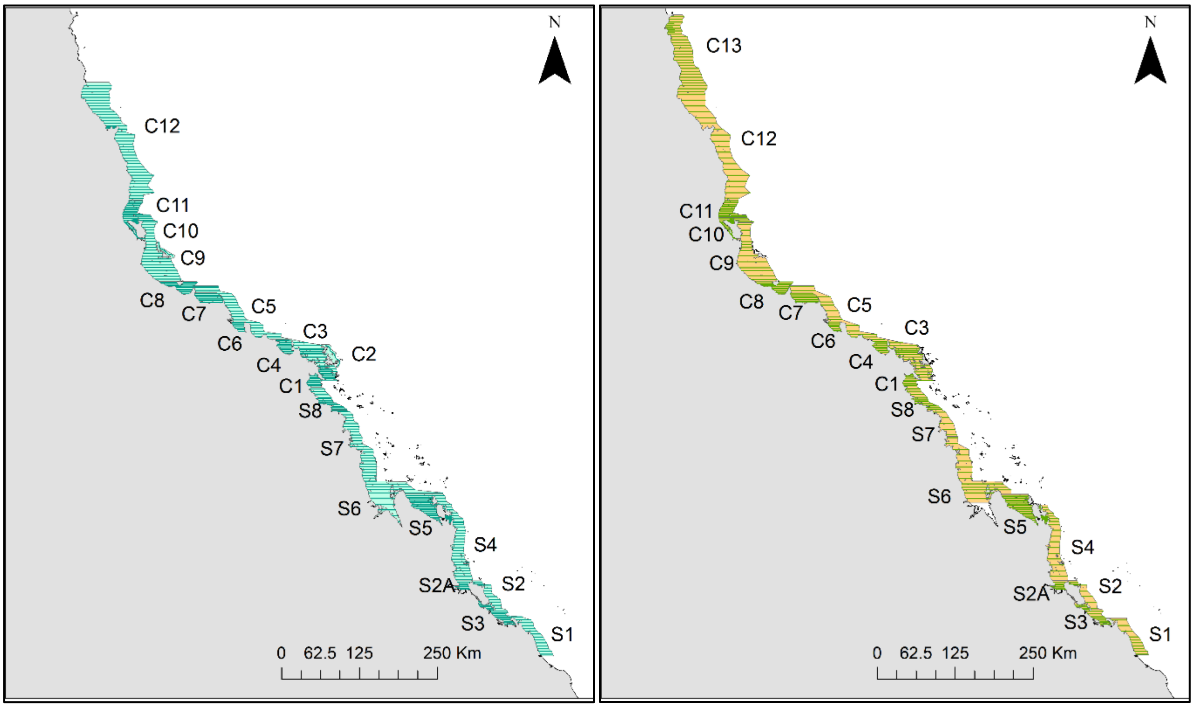

Figure 5. Map of the original (left) and optimised (right) dugong aerial survey designs for the southern Great Barrier Reef. Note the addition of Block C13 in the optimised design.

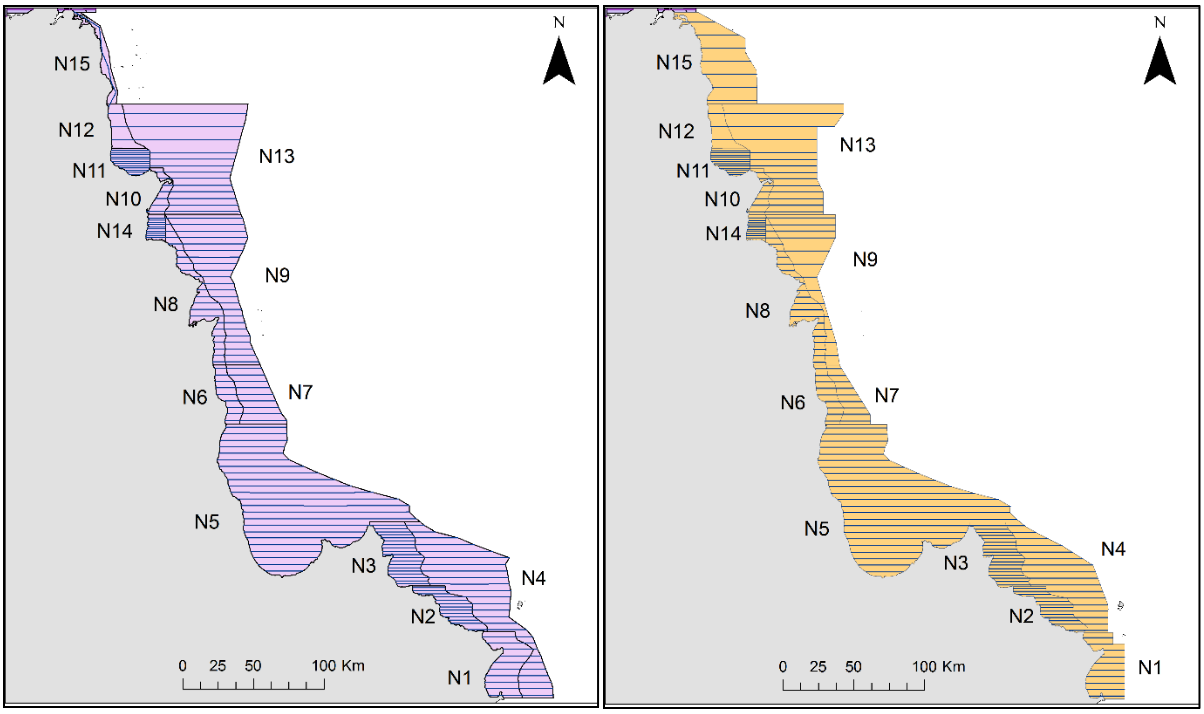

Figure 6. Map of the original (left) and optimised (right) dugong aerial survey designs for the northern Great Barrier Reef. Note the addition of the transects perpendicular to the coast rather than zig-zag transections in Block N15 in the optimised design**.**

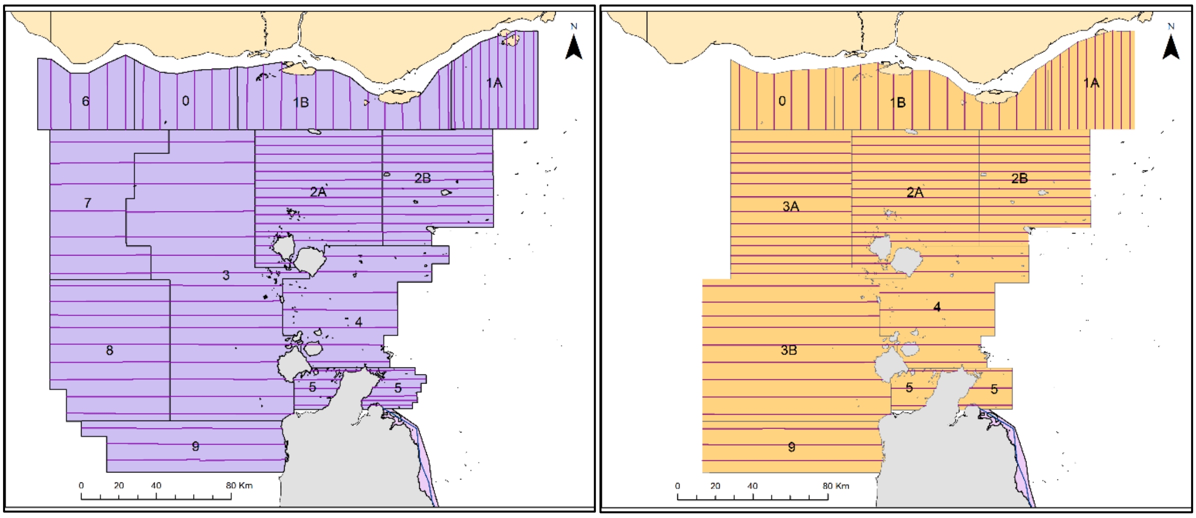

Figure 7. Map of the original (left) and optimised (right) dugong aerial survey design for Torres Strait adjacent to the northern Great Barrier Reef.

<u>Comparisons between the optimised survey designs and the original designs</u>

To compare the optimised survey designs with the original designs, archival dugong and turtle sightings data collected since 2005[4] from the following regions were re-analysed: the southern Reef region, the northern Reef region and Torres Strait (Table 2). Moreton Bay and Hervey Bay were not included since the optimised survey design did not differ from the established design. Survey years were selected due to the availability of information on the sighting conditions (for example, water turbidity) for individual dugong or turtle sightings that are required to estimate population abundance (Hagihara *et al.* 2014, 2018). The information on turtle sightings from the 2006 northern Reef survey were available only in a raw, unadjusted format and were not included in the current analyses.

Table 2. Years for which the archival survey data were reanalysed in the southern Great Barrier Reef, the northern Great Barrier Reef and Torres Strait. Both dugongs and large juvenile and adult marine turtles are recorded during the surveys.

Region	Dugongs	Turtles
Southern Reef	Nov 2005 Nov 2016	Nov 2005 Nov 2016
Northern Reef	Nov 2006 Nov 2013	Nov 2013
Torres Strait	Nov 2006 Mar 2011 Nov 2013	Nov 2006 Mar 2011 Nov 2013

Following Pollock *et al.* (2006) and Hagihara et al. (2014, 2018), the relative abundance of dugongs and large (>1m CCL) in-water turtles was estimated for each region using the

[4] The data required to estimate the Hagihara et al. (2014 and 2018) availability correction factors were not available prior to 2000 as explained above.

optimised survey design and the best available detection probability models for each region (Table 3). Hagihara et al. (2018) found that the diving behaviour of dugongs in the southern Reef, Torres Strait and New Caledonia differed significantly and hence this variation needs to be accounted for in estimating abundance. Thus, for dugongs, all availability bias estimates have been depth-corrected, whereas for turtles the estimates from Hagihara *et al.* (2016) have been depth-corrected while those from Fuentes *et al.* (2015) have not been depth-corrected. Calculating depth-corrected estimates for turtle abundance in the southern and northern Reef following Hagihara et al. (2016) was beyond the scope of this project.

Table 3. Studies from which availability bias estimates were extracted for this report.

	Availability bias estimates	
Region	**Dugongs**	**Turtles**
Southern Reef	Sobtzick *et al.* (2015)	Fuentes *et al*. (2015)
Northern Reef	Sobtzick *et al.* (2015)	Fuentes *et al*. (2015)
Torres Strait	Hagihara *et al*. (2016)	Hagihara *et al.* (2016)

For each region, the dugong relative population estimates using the optimised survey design were within three to six per cent of the estimates obtained using the original survey design and had very similar coefficients of variation (Table 4). Estimates for individual blocks and survey intensities are provided in Appendix 2. Similarly, the turtle population estimations were very similar (three to seven per cent variation between designs) between the original and the optimised survey design with comparable coefficients of variation (Table 5). Estimates for individual blocks and survey intensities are provided in Appendix 3.

Table 4. Percentage change in the population estimates and coefficient of variation (CV) of dugong population sizes using the original and optimised survey designs for the southern and northern Great Barrier Reef and Torres Strait.

Region	Survey Year	Comparison of the original dugong abundance estimates with corresponding estimates from the optimised design[1]	
		Relative population size	Coefficient of variation
Southern Reef	2005	96	102
	2016	104	99
Northern Reef	2006	96	106
	2013	94	103
Torres Strait	2006	97	91
	2011	106	95
	2013	95	87

[1] % change = abundance estimates from the optimised design divided by the abundance estimates from the original design x 100.

Table 5. Percentage change in the population estimates and coefficient of variation (CV) of turtle population sizes using the original and optimised survey designs for the southern and northern Great Barrier Reef and Torres Strait.

Region	Survey Year	Comparison of the original large turtle abundance estimates with corresponding estimates from the optimised design[1]	
		Relative population size	Coefficient of variation
Southern Reef	2005	96	116
	2016	93	112
Northern Reef	2006	-	-
	2013	93	106
Torres Strait	2006	106	95
	2011	103	93
	2013	104	101

[1] % change = abundance estimates from the optimised design divided by the abundance estimates from the original design x 100.

Financial implications of the optimised survey design

Compared to the original survey design, the optimised survey design resulted in a reduction of required flight time, and therefore survey days for all three regions (Table 6).

In both, Torres Strait and the northern Reef, the optimised design requires approximately seven hours less flying time than the original design (equates to one survey day and two days of stand-by time). This reduction equates to a saving of approximately $17,000 in each region.

For the southern Reef, a reduction of 14.5 hours of flight time is estimated to save two days of flying and four days of stand-by time. Based on the costs of the 2016 aerial survey, this equates to a total saving of $35,000.

Table 6. Amounts saved using the optimised survey design compared to the 2016 southern Great Barrier Reef survey plan. Prices based on costs of the 2016 survey, all including GST and assumes that the break between the southern Reef and northern Reef is at Cooktown.

Saving	Torres Strait	Northern Reef	Southern Reef
Aircraft flight hours	7 hrs	7 hrs	14.5 hrs
Aircraft stand-by days	2 days	2 day	4 days
Salaries, accommodation and food	3 days (1 flight day and 2 stand-by days)	3 days (1 flight day and 2 stand-by days)	6 days (2 flights days and 4 stand-by days)
Total savings (approx.)	**$17,000**	**$17,000**	**$35,000**

Prospective power analysis

The prospective power analysis (Appendix 1) used the same approach used in the retrospective analysis to estimate annual trends, but instead of using the observed data, used future hypothetical datasets under different scenarios, using the Negative Binomial distribution fitted during the previous analyses. The purpose of the prospective analysis was to assess the ability of the large scale aerial surveys to detect future declines under a variety of sampling regimes, given 'true' declines of minus one per cent and minus three per cent, per year. The prospective analysis suggested that a minus three per cent decline could be detected with 0.8 probability at intermediate time-horizons (eight years and greater) but shorter time-scales and more frequent surveys are unlikely to provide the requisite power to detect trends. We recommend that the five-year survey frequency be maintained for the dugong aerial surveys, and that these surveys be taken as one of several lines of evidence used to determine important trends in dugong abundance in the Reef. We make this recommendation because of the limited improvement in power achieved by increasing the survey frequency, the statutory five-year reporting period required by the *Great Barrier Reef Marine Park Act 1975* for the Great Barrier Reef Outlook Report, and the requirements of the Reef 2050 Plan (Commonwealth 2018).

Assessment of the resources required to implement the optimised aerial survey design every five years as recommended on the basis of the prospective power analysis

A summary of the required flight hours, total survey duration and plane transit times for the northern and southern Reef regions is provided in Table 7. Additional costs for conducting each survey include:

- salary costs (for pilot, team leader, 4 observers per aircraft, ground support team (typically one driver), project manager);
- food and accommodation costs for field crew plus pilot;
- transport costs (with large capacity troop mover vehicle rental, plus fuel;
- domestic air fares for observers and freight costs;
- additional aircraft fuel costs, especially in remote areas (if not covered by operator);
- safety gear hire (life raft and life jackets); and
- engineering costs for the installation of the survey equipment on the aircraft.

Prior to each survey, a workshop must be conducted to ensure new observers and pilots are familiar with the methodology and confident in correctly identifying marine megafauna. Training course costs vary (depending on the number and the competency of the observers), but training should last at least three days; longer if helicopter underwater escape training is incorporated. The costs of such training have not been included in Table 8.

Table 7. Flight hours, survey duration and aircraft transit times required for each survey of the northern Great Barrier Reef and southern Great Barrier Reef using the optimised design and a manned aircraft.

Item	Reef Region	
	Northern Reef	Southern Reef
Assuming that the junction between the northern and southern Reef is at Cooktown		
Aircraft flight hours	50 hrs	54.5 hrs
Total survey duration	27 days (9 survey days and 18 stand-by days)	30 days (10 survey days and 20 stand-by days)
Aircraft transit time*	17 hrs	13 hrs
Assuming that the junction between the northern and southern Reef is at the Whitsundays, the location of the genetic break in the dugong population		
Aircraft flight hours	80 hrs	24.5 hrs
Total survey duration	12 days (4 survey days and 8 stand-by days)	45 days (15 survey days and 30 stand-by days)
Aircraft transit time*	15 hrs	9 hrs

Item	Adjacent to Reef Region	
	Torres Strait	South-East Qld
Aircraft flight hours	49	27
Total survey duration	25.5 days (8.5 survey days and 17 standby days)	22 days (8.5 survey days and 13.5 standby days)
Aircraft transit time*	20 hrs	6 hrs

*Aircraft transit time will vary depending on the home location of the aircraft, and the number of aircraft used. Presented figures assume using one aircraft and the operator used in 2016.

For reference, the 2018 survey from Cooktown north using the optimised survey design is budgeted to cost $302,000 including staff costs for preparation, training, field work, analysis and report writing. The cost of a survey of the Reef from Cooktown to the southern border of the Reef would be similar.

5.0 New technologies for aerial monitoring dugongs on the Great Barrier Reef

As detailed in Appendix 4, the *ScanEagle* unmanned aerial vehicle has sufficient endurance and range to cover the scale of the areas recommended for monitoring dugongs within the Marine Park using the optimised design with a significant reduction in the human safety risks associated with manned aircraft and likely superior detection of large groups of dugongs.

The coverage can be achieved by using 'hub and spoke' operations whereby repeaters are able to extend the range of the *ScanEagle* by handing off to a nearby communications link. Most (but not all) transects in the optimised design are less than 100 kilometres long. The challenge will be to optimise the placement of the communications links, especially along the remote costs of Cape York and the Shoalwater Bay region where land access is limited. One potential logistical limitation, in using the *ScanEagle*, is that this system flies at half the ground speed of a manned plane, and therefore a survey could take twice as long. However, this unmanned aerial vehicle has enough endurance to fly continually for a whole survey day, as opposed to a manned aircraft where the maximum flight time is three hours before refuelling is necessary.

The two-camera imaging system tested in the most recent trial survey allows for the same survey design and sampling rate as manned dugong surveys (Hodgson et al. in prep). Customised dugong detection and mapping software has been developed (with ongoing improvements) meaning that it is realistic to survey large areas and process images in a cost-effective and expedient timeframe, although some manual review of images is currently still necessary.

Results to date suggest that sighting rates in unmanned aerial vehicle images are not affected by sea state and therefore unmanned aerial vehicle surveys could potentially be flown in a wider range of wind conditions than the manned surveys. This result needs to be tested further as it makes the untested assumption that dugong diving behaviour is unaffected by sea state. The manned aerial surveys overcome this assumption by limiting the sea states in which the surveys are conducted. Both advantages of flight endurance and the ability to survey in a wider range of conditions than manned surveys may counteract the effect of slow flight speed of unmanned aerial vehicles.

5.1 Data compatibility

Trial surveys conducted so far suggest that unmanned aerial vehicles provide comparable dugong sighting rates to manned aerial surveys and that there is no difference in the way sighting rates are affected by environmental conditions when comparing the two platforms. From this result, we can assume that availability of dugongs is comparable between the two platforms, and that it is feasible and appropriate to apply the availability corrections we currently apply to manned surveys, to unmanned aerial vehicle surveys if surveying under the same limited wind conditions (that is, Beaufort sea states of ≤ 3).

However, there are three outstanding matters to resolve before unmanned aerial vehicle surveys can replace manned dugong surveys in the Reef. These matters are presented below in order of likely importance to future Reef dugong surveys.

1. Detection in highly turbid waters and high sea states

   The trial surveys conducted to date have been in Shark Bay, Western Australia, where the water is relatively clear – there is very little of the turbid water characteristic of most dugong habitats within the Reef region. To ensure that future dugong unmanned aerial vehicle surveys are directly comparable with historic manned surveys, similar trial surveys need to be conducted in more turbid dugong habitat than was encountered in Shark Bay. Ideally, such surveys would also incorporate further testing of dugong sighting rates in high sea states. Although previous trial surveys have suggested sea state does not affect sighting rates, there has been relatively little data collected in the higher wind conditions (for example, only three of eight flights were conducted in Beaufort sea state 4 or 5 during the most recent trial surveys (Hodgson et al. in prep)). The *ScanEagle* is capable of flying in these high wind speeds.

2. Detection of large groups

   The trials indicate that unmanned aerial vehicles are better at detecting large dugong groups than human observers in transect mode. This result is currently being investigated and the outcome of these analyses will determine whether further work is needed to resolve this issue. However, large groups of dugongs (more than 10 individuals) are relatively rare in the Reef and have traditionally been stratified out of analyses to correct dugong observations for detection biases in manned surveys and added on at the end as uncorrected counts. Considering the rarity of large groups in Reef survey areas, and that the detection of these groups from manned aircraft is possibly only biased downwards in particular environments (i.e. in shallow murky waters), the increase in detection of large groups by unmanned aerial vehicles will likely not affect Reef dugong population estimates very much. Nonetheless, our understanding of dugong habitat use may be affected by this bias, a defect that may be significant in spatial risk assessments.

3. Availability corrections

   Experiments need to be conducted to ensure that the corrections for availability bias that have been developed for observers in manned aircraft for dugong surveys in the Reef are applicable to unmanned aerial vehicle surveys to maximise the integrity of the historical time series, which is based on standardised indices of relative abundance. The need to do this work would depend on the results from 1) above. The simplest method of conducting this check would be to repeat the dugong model experiments (Hagihara et al. 2016) using a small, multi-rotor unmanned aerial vehicle with a camera similar to that in the *ScanEagle*. The unmanned aerial vehicle could be operated from a boat, similar to methods currently being conducted at Murdoch University by Chris Cleguer. If absolute indices of dugong abundance were required, more elaborate experiments using unmanned aerial vehicles to conduct focal follows of dugongs fitted with satellite transmitters should be conducted as discussed in Appendix 4.

Appendix 4 discusses the work required to conduct the high priority experiments required to compare the *ScanEagle's* capacity to detect dugongs in the turbid waters of the Reef with that of the current surveys conducted using manned aircraft. Such comparisons are essential to enable a transition to the new technology without losing the value of the historical data sets.

6.0 Recommended priority indicators to monitor dugongs and large in-water turtles in seagrass habitats to 25 metres in the Great Barrier Reef

We recommend that dugongs be monitored in the Reef region using several techniques and priority indicators as lines of evidence as summarised in Table 8 below. A several lines of evidence approach process has been used successfully in Torres Strait (Marsh et al. 2015).

Table 8. Indictors of the status of the Great Barrier Reef dugong and large juvenile and adult in-water turtle populations. The indicators that are relevant to both dugongs and turtles are marked with an asterisk.

Method of tracking indicator	Indicators	Frequency	Platform	Coordinated with
Spatial scale: dugong habitats in entire Great Barrier Reef				
Large scale aerial surveys using optimised survey design If manned aircraft are used the surveys should be done over two years using the Hagihara et al. 2018 methodology: Year 1: south of the genetic break in dugong stocks at the Whitsundays + south East Qld if funding available from Qld **Year 2: north** of the genetic break in dugong stocks at the Whitsundays + Torres Strait if funding available from TSRA to account for cross-jurisdictional dugong movements When SeaEagle unmanned aerial vehicle is used it may be possible to survey the entire inshore Reef in a single year	Trends in: • dugong population size (standardised relative abundance)* • dugong relative density using the various techniques to correct for standardised relative abundance* • dugong percentage calves (index of fecundity and neonatal survivorship over the previous 2 years) (Fuentes et al. 2016) • dugong Area of Occupancy (Marsh et al. 2015)* • % overlap of high and medium density dugong habitats with major threats as revealed by spatial risk assessment using aerial survey data and spatial data on sources of potential dugong mortality such as Traditional hunting, gill netting and vessel strike (see Grech and Marsh 2008, Grech et al. 2008 and Marsh et al. 2015).*	5 yearly in November	To 2021; Manned Partenavia aircraft From 2021: SeaEagle unmanned aerial vehicle after transitional work detailed in Appendix 4 is complete	Seagrass habitat assessment monitoring in survey region (Udy et al. 2018)
StrandNet data (See Marsh et al. 2018b)	• Trends in distribution and numbers of dugong mortalities and causes of death per month per year; mainly from urban coast	Ongoing; potential early warning of		

		increased threats		
Spatial scale: local along Reef coast				
Boat surveys conducted by Indigenous rangers (See Bayliss and Fisher 2018)	• Trends in distribution and relative abundance of dugongs and large juvenile and adult in-water turtles in seacountry	Ongoing; potential early warning of increased threats	Ranger vessels	
Catch surveys conducted by Indigenous rangers (See West and Marsh 2018)	• Trends in catch per unit effort of Traditional Hunting by community (this is a potential indicator subject to agreement of Traditional Owners) • Extent of hunting grounds as percentage of areas of high or medium dugong or large marine turtle density determined from aerial surveys (see Marsh et al. 2015).	Ongoing; potential early warning of increased threats e.g. dugongs in poor condition		
Surveys of dugong feeding trails (see Rasheed et al. 2017)	• Changes in density of dugong feeding trails which are an indicator of dugong excavating feeding activity at shallow inshore sites		Human observers, helicopters or small drones	Monitoring of dugong response to local threats e.g. capital dredging

7.0 References

Bayliss, P., and Fischer, M. 2018. Indigenous participation in monitoring megafauna in the coastal waters of the Great Barrier Reef: Review and proposal for integrated monitoring in RIMReP. Report to the Great Barrier Reef Marine Park Authority.

Blair, D., McMahon, A., McDonald, B., Tikel, D., Waycott, M., and Marsh. H. 2014. Pleistocene sea level fluctuations and the phylogeography of the dugong in Australian waters. Marine Mammal Science, 30: 104-121.

Blair, D. 2015. Genetic evidence for distinctiveness and connectivity among Australian dugongs. Australian Marine Mammal Centre Final Report. 25 pp.

Butler, J.R.A., Tawake, A., Skewes, T., Tawake, L. and McGrath, V. (2012) Integrating traditional ecological knowledge and fisheries management in the Torres Strait, Australia: the catalytic role of turtles and dugong as cultural keystone species. Ecology and Society,17: 1-19.

Cleguer, C., Limpus, C.G., Gredzens, C., Hamann, M., Marsh, H. (2015a). Annual report on dugong tracking and habitat use in Gladstone in 2014. Report produced for the Ecosystem Research and Monitoring Program Advisory Panel as part of Gladstone Ports Corporation's Ecosystem Research and Monitoring Program.

Cleguer, C., Limpus, C.G., Hamann, M., Marsh, H. (2015b). Annual report on dugong tracking and habitat use in Gladstone in 2015. Report produced for the Ecosystem Research and Monitoring Program Advisory Panel as part of Gladstone Ports Corporation's Ecosystem Research and Monitoring Program.

Commonwealth of Australia 2015. The Reef 2050 Long-Term Sustainability Plan. Commonwealth of Australia 2015. 111pp.

Cope R.C., Pollett P.K., Lanyon J.M., and Seddon, J.M. 2015. Indirect detection of genetic dispersal (movement and breeding events) through pedigree analysis of dugong populations in southern Queensland, Australia. Biological Conservation 181, 91-101.

Dobbs, K., Fernandes, L., Slegers, S., Jago, B., Thompson, L., Hall, L., Day, J., Cameron, D., Tanzer, J., Macdonald, F., Marsh, H., and Coles, R. 2008. Incorporating dugong habitats into the marine protected area design for the Great Barrier Reef Marine Park, Queensland, Australia. Ocean and Coastal Management 51: 368-375.

Fernandes, L., Day, J., Lewis, A., Slegers, S., Kerrigan, B., Breen, D., Cameron, D., Jago, B., Hall, J., Lowe, D., Innes, J., Tanzer, J., Chadwick, V., Thompson, L., Gorman, K., Simmons, M., Barnett, B., Sampson, K., De'ath, G., Mapstone, B., Marsh, H., Possingham, H., Ball, I., Ward, T., Dobbs, K., Zumend, J., Slater, D., and Stapleton, K. 2005. Establishing representative no-take areas in the Great Barrier Reef: large-scale implementation of Theory on Marine Protected Areas. Conservation Biology 19: 1733-1744.

Flint, M., Patterson-Kane, J.C., Limpus, C.J., and Mills P.C. 2010. Health surveillance of stranded green turtles in southern Queensland, Australia (2006-2009): An epidemiological analysis of causes of disease and mortality. EcoHealth 7: 135-145.

Fuentes, M.M.P.B., Bell, I., Hagihara, R., Hamann, M., Hazel, J., Huth, A., Seminoff, J.A., Sobtzick, S., and Marsh, H. 2015. Improving estimates of in-water marine turtle abundance by adjusting aerial survey counts for perception and availability biases. Experimental Marine Biology and Ecology 471:77–83.

Fuentes, M.M.P.B, Beatty, B., Delean, S., Grayson, J., Lavender, S., Logan, M., Marsh, H. (2016) Spatial and temporal variation in the effects of climatic variables on dugong calf production. PLoS One, 11960 1-14. PONE-D-15-52097R1.

Goto, M, Ito, C., Yahaya, M.S. et al (2004). Effects of age, body size and season on food consumption and digestion of captive dugongs (*Dugong dugon*). Marine and Freshwater Behaviour and Physiology 37:89–97.

Great Barrier Reef Marine Park Authority. 1981. Nomination of the Great Barrier Reef by the Commonwealth of Australia for inclusion on the World Heritage List. UNESCO, 37 pp.

Great Barrier Reef Marine Park Authority. 2017. Drivers of change, pressures and impacts on the Great Barrier Reef http://www.gbrmpa.gov.au/managing-the-reef/reef-2050/Reef-2050-policies/drivers-of-change,-pressures-and-impacts-on-the-great-barrier-reef.

Great Barrier Reef Marine Park Authority 2014, Great Barrier Reef Region Strategic Assessment: Strategic assessment report, GBRMPA, Townsville.

Grech, A., and Marsh, H. 2007. Prioritising areas for dugong conservation in a marine protected area using a spatially explicit population model. Applied GIS 3: 1-14.

Grech, A., and Marsh, H. 2008. Rapid assessment of risks to a mobile marine mammal in an ecosystem-scale marine protected area. Conservation Biology 22: 711-720.

Grech, A., Marsh, H. and Coles, R. 2008. Using spatial risk assessment to evaluate and address the problem of marine mammal bycatch. Aquatic Conservation 18*:* 1127-1139.

Gredzens, C., Marsh, H., Fuentes, M. M., Limpus, C. J., Shimada, T. and Hamann, M. 2014. Satellite tracking of sympatric marine megafauna can inform the biological basis for species co-management. Plos One*,* 9**,** e98944.

Hagihara, R., Jones, R., Grech, A., Lanyon, J., Sheppard, J., and Marsh, H. 2014. Improving population estimates by quantifying diving and surfacing patterns: A dugong example. Marine Mammal Science 30:348–366.

Hagihara, R., Cleguer, C., Preston, S., Sobtzick, S., Hamann, M., Shimada, T., and Marsh, H. 2016. Improving the estimates of abundance of dugongs and large immature and adult-sized green turtles in Western and Central Torres Strait. Report to the National Environmental Science Programme. Reef and Rainforest Research Centre Limited, Cairns, 65 pp.

Hagihara, R., Jones, R.E., Sobtzick, S., Cleguer, C., Garrigue, C., and Marsh, H. 2018. Compensating for geographic variation in detection probability with water depth improves abundance estimates of coastal marine megafauna. PLOSOne 13(1): e0191476.

Hansard Australian Senate. 2017. Senate debates. Wednesday, 13 September 2017. Matters of public importance: management pf protected species. https://www.openaustralia.org.au/senate/?id=2017-09-13.193.2.

Heinsohn, G.E., Spain, A.V. 1974). Effects of a tropical cyclone on littoral and sub littoral biotic communities and on a population of dugongs (*Dugong dugon* [Müller]). Biological Conservation 6:143–152.

Hodgson, a. 2018. Potential use of unmanned aerial vehicles for megafauna monitoring in the GBR: transitioning to the new technology. Report to GBRMPA as a component of the Megafauna RIMReP.

IUCN Red List of Threatened Species. (2017). Version 2017-3. <www.iucnredlist.org>. Downloaded on 14 March 2018.

Marsh, H. 1989. Mass stranding of dugongs by a tropical cyclone. Marine Mammal Science 5: 75-84.

Marsh, H. 2000. Evaluating management initiatives aimed at reducing the mortality of dugongs in gill & mesh nets in the Great Barrier Reef World Heritage Area. Marine Mammal Science 16: 684-94.

Marsh, H., and Kwan, D. 2008. Temporal variability in the life history and reproductive biology of female dugongs in Torres Strait: the likely role of sea grass dieback. Continental Shelf Research 28: 2152-2159.

Marsh, H. and Rathbun, G B. 1990. Development and application of conventional & satellite radio-tracking techniques for studying dugong movements and habitat usage. Australian Wildlife Research 17: 83-100.

Marsh, H., and Sinclair, D.F. 1989. Correcting for visibility bias in strip transect aerial surveys of aquatic fauna. Journal of Wildlife Management 53:1017–1024.

Marsh, H, De'Ath, G, Gribble, N, and Lane, B. 2005. Historical marine population estimates: Triggers or targets for conservation? The dugong case study. Ecological Applications 15: 481-492

Marsh, H., O'Shea, T.J., and Reynolds, J.E. III. 2011. The ecology and conservation of Sirenia: dugongs and manatees. Cambridge University Press. 521pp.

Marsh, H., Grayson, J., Grech, A., Hagihara, R., and Sobtzick, S. 2015. Re-evaluation of the sustainability of a marine mammal harvest by indigenous people using several lines of evidence. Biological Conservation 192:324-330.

Marsh H., Grech A., and McMahon K. 2018a. Dugongs: Seagrass Community Specialists. In: Larkum A., Kendrick G., Ralph P. (eds) Seagrasses of Australia. Springer, Cham. https://doi.org/10.1007/978-3-319-71354-0_19

Marsh, H., Limpus, G.J., Meager, J., Moisel, A., Read, M., Salmon, S., Sobtzick. S. 2018b The role of strandnet in monitoring megafauna in the coastal waters of the Great Barrier Reef: Review and suggestions for upgrading as a key element of RIMReP. Report to GBRMPA as a component of the Megafauna RIMReP.

McCook, L.J., Ayling, T., Cappo, M., Choat, J.H., Evans, R.D., Freitas, D.M., Heupel, M., Hughes, T.P., Jones, G.P., Mapstone, B., Marsh, H., Mills, M., Molloy, F., Pitcher, C.R., Pressey, R.L., Russ, G.R., Sutton, S., Sweatman, H., Tobin, R., Wachenfeld, D.R., and Williamson, D.H. 2010. Adaptive management of the Great Barrier Reef: a globally significant demonstration of the benefits of networks of marine reserves. Proceedings of the National Academy of Science USA. PNAS 107: 18278- 18285.

McKenzie, L.J., Collier, C.J, Langlois, L.A, Yoshida, R.L, Smith, N., Waycott, M. 2016, Marine Monitoring Program. Annual Report for inshore seagrass monitoring: 2014 to 2015. Report for the Great Barrier Reef Marine Park Authority. TropWATER, James Cook University, Cairns. 236pp.

Meager, J.J. 2016. Marine wildlife stranding and mortality database annual report 2013-2015. Dugong. Conservation Technical and Data Report 2016. 2:1-22.

Meager, J.J., and Limpus C. 2014. Mortality of inshore marine mammals in eastern Australia is predicted by freshwater discharge and air temperature. PLOS One: e94849.

Meager, J.J., and Sumpton, W.D. 2016. Bycatch and strandings programs as ecological indicators for data-limited cetaceans. Ecological Indicators 60:987-995.

Nielsen, K.A., Owen, H., Mills, P.C., Flint, M., and Gibson, J.S. 2013. Bacteria isolated from dugongs (*Dugong dugon*) submitted for post-mortem examination in Queensland, Australia, 2000–2011. Journal of Zoo and Wildlife Medicine, 44(1):35-41.

Peltier, H., Dabin, W., Daniel, P., Van Canneyt, O., Dorémus, G., Huon, M., and Ridoux, V. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. Ecological Indicators 18, 278–290.

Perrin, W.F., Würsig, B.G., and Thewissen, J.G.M. 2002. Encyclopedia of Marine Mammals. Academic Press, San Diego.

Pollock, K., Marsh, H., Lawler, I., and Alldredge, M. 2006. Modelling availability and perception processes for strip and line transects: an application to dugong aerial surveys. Journal of Wildlife Management 70:255–262.

Preen, A. 1995. Impacts of dugong foraging on seagrass habitats: observational and experimental evidence for cultivation grazing. Marine Ecology Progress Series 124:201–213.

Preen, A., and Marsh, H. 1995. Response of dugongs to large-scale loss of seagrass from Hervey Bay, Queensland, Australia. Wildlife Research *22*: 507-19.

Rasheed, M.A., O'Grady, D., Scott, E., York, P.H., and Carter, A.B.2017). Dugong feeding ecology and habitat use on intertidal banks of Port Curtis and Rodds Bay – Final Report. Report produced for the Ecosystem Research and Monitoring Program Advisory Panel as part of Gladstone Ports Corporation's Ecosystem Research and Monitoring Program. Centre for Tropical Water and Aquatic Ecosystem Research (TropWATER) Publication 16/14, James Cook University, Cairns, 68 pp.

RIMReP Expert Group Project Megafauna (2017) EOI 0504117 to Australian Institute of Marine Science.

Seddon, J.M., Ovenden, J.R., Sneath H. L., Broderick, D., Dudgeon, C.L. & Lanyon, J.M. 2014. Fine scale population structure of dugongs (*Dugong dugon*) implies low gene flow along the southern Queensland coastline. Conservation Genetics, *15* 6: 1381-1392. doi:10.1007/s10592-014-0624-x.

Sheppard, J., Preen, A.R., Marsh, H., Lawler, I.R., Whiting, S., & Jones, R.E. 2006. Movement heterogeneity of dugongs, *Dugong dugon* Müller over large spatial scales Journal of Experimental Marine Biology & Ecology 334: 64–83.

Sobtzick, S., Hagihara, R., Grech, A. and Marsh, H. 2012. Aerial survey of the urban coast of Queensland to evaluate the response of the dugong population to the widespread effects of the extreme weather events of the summer of 2010-11. Final report to the Australian Marine Mammal Centre and the National Environmental Research Program.

Sobtzick, S., Hagihara, R., Penrose, H., Grech, A., Cleguer, C., and Marsh, H. 2014. An assessment of the distribution and abundance of dugongs in the Northern Great Barrier Reef and Torres Strait. A Report for the Department of the Environment, National Environmental Research Program (NERP). August 2014.

Sobtzick, S., Hagihara, R., Grech, A., Jones, R., and Marsh, H. 2015. Improving the time series of estimates of dugong abundance and distribution by incorporating revised availability bias corrections. Final Report to the Australian Marine Mammal Centre on Project 13/31. Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) Publication, James Cook University, Townsville, 105 pp.

Sobtzick, S., Cleguer, C., Hagihara, R., and Marsh, H. 2017. Distribution and abundance of dugong and large marine turtles in Moreton Bay, Hervey Bay and the southern Great Barrier Reef. A report to the Great Barrier Reef Marine Park Authority. Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) Publication 17/21, James Cook University, Townsville.

State of the Environment Committee. 2011. Australia State of the environment 2011. An independent report presented to the Australian Government Minister for Sustainability, Environment, Water, Population and Communities by the State of the Environment 2011 Committee. 932pp.

Stejneger, L., 1887. How the great northern sea-cow (rytina) became exterminated. American Naturalist 21:1047-1054.

Taylor, B.L., Matinex, M., Gerrodette, T., Barlow, J., 2007. Lessons from monitoring trends in abundance of marine mammals. Marine Mammal Science. 23, 157–175.

Udy, J., Waycott, M., Carter, A., Collier, C., Kilminster, .K., Rasheed,UM., McKenzie, L., McMahon, K., Maxwell, P., Lawrence, E., Honchin, C. 2018. Monitoring seagrass within the Reef 2050 Integrated Monitoring and Reporting Program, Final report of the seagrass expert group. GBRMPA.

West. L., and Marsh, H. 2018. Monitoring the Indigenous dugong and turtle harvest in the Great Barrier Reef region: a preliminary proposal for discussion with Traditional Owners. Report to GBRMPA as a component of the Megafauna RIMReP.

Zeh, D. R., Heupel, M. R., Limpus, C. J., Hamann, M., Fuentes, M. M. P. B., Babcock, R. C., Pillans, R. D., Townsend, K. A. and Marsh, H. 2015. Is acoustic tracking appropriate for air-breathing marine animals? Dugongs as a case study. Journal of Experimental Marine Biology and Ecology*,* 464**,** 1-10.

Zeh, D. R., Heupel, M. R., Limpus, C. J., Hamann, M., and Marsh, H. 2016 . Quick Fix GPS technology highlights risk to marine animals moving between protected areas. Endangered Species Research. 30: 37–44, doi: 10.3354/esr00725.

8.0 Appendix 1 — Trend analysis and probabilities of detecting declining dugong populations

Robert W Rankin, Ph.D. [5]

8.1 Executive Summary

This report presents a trend analysis of the population abundance of dugongs (*Dugong dugon*) in the Great Barrier Reef (the Reef), Australia. The analyses consisted of data from standardised aerial-transects obtained between 2005 and 2016 inclusive using the optimised survey design discussed in the main report.

The analyses attempted to answer two questions: i) Did the large-scale aerial surveys detect a decline in mean dugong counts between 2006 and 2013 in the northern Reef and between 2005 and 2016 in the southern Reef[6,7] ii) How powerful are the data and analysis framework to detect future declines under various scenarios? The analyses consisted of Bayesian hierarchical models which incorporated multiple sources of excess variation and uncertainty, such as imperfect detection, availability-bias, overdispersion, parameter uncertainty, and model-selection uncertainty (the latter were estimated in previous studies (Marsh and Sinclair 1989, Pollock et al. 2006, Hagihara et al. 2014, 2018)). While the analyses were Bayesian in computation and interpretation, they are analogous to conventional 'power analyses' from the Neyman-Pearson school of frequentist statistics.

The results suggested that there was a 0.791 probability that a decline in dugong abundance took place in the northern Reef between 2006 and 2013, whereas there was a 0.689 probability of a decline in the southern Reef between 2005 and 2016. The prospective analysis suggested that a minus three per cent decline could be detected with 0.8 probability at intermediate time-horizons (eight years and greater) but shorter time-scales and more frequent surveys are unlikely to provide the requisite power to detect trends. Given the limited improvement in power achieved by increasing the survey frequency, the statutory five-year reporting period required by the *Great Barrier Reef Marine Park Act 1975* (section 54) for the Great Barrier Reef Outlook Report, and the requirements of the *Reef 2050 Long-Term Sustainability Plan* (Commonwealth 2018), the recommendation is that the five-year survey frequency be maintained for the dugong aerial surveys, taken as one of several lines of evidence used to determine important trends in dugong abundance in the Reef.

8.2 Background

This study focused on the populations of dugongs in the northern and southern Reef. As for most marine mammals, dugong subsurface activity makes it difficult to detect individual animals and monitor changes in abundance. Such observational biases may vary with environmental conditions (such as turbidity and depth) and dugong movement and diving

5 Department of Biology, Georgetown University, Washington D.C.
6 Assuming that the junction between these two regions is at Cooktown.
7 The survey data collected in 2011 for the southern Great Barrier Reef were not included in this analysis because of the aberrant results attributable to temporary immigration out of the survey due to the habitat loss caused by the floods and cyclones during the summer of 2011.

behaviour, as well as changes in the personnel conducting the surveys. Variations in environmental factors and dugong behaviour, especially temporary immigration, as well as observer turnover, can alter the observational bias and confound the trends in population abundances.

Fortunately, several studies have calculated dugong bias correction factors to help improve the accuracy of trend estimates. The populations of dugongs in the northern and southern Reef have been surveyed repeatedly by aerial transects since 1985, according to the protocols of Marsh and Sinclair (1989), and updated by others (Pollock et al. 2006, Hagihara et al. 2014, 2018) as detailed in main report.

The updated survey method facilitates the estimation of several bias correction factors. First, there is the availability probability, whereby animals are only available for detection at the surface with probability (α), due to their subsurface diving behaviour (Pollock et al. 2006). This parameter is conditional on the depth and turbidity of the water (Hagihara et al. 2014, 2018). Secondly, observers have imperfect detection, with detection probability *p*, conditional on the availability of the animal. Previous studies have made significant improvements in estimating these parameters and quantifying their uncertainty. With estimates of *p* and α, Pollock et al. motivated the estimation of 'adjusted counts of dugongs' N^{adj} with a Horvitz-Thompson-like estimator. The adjusted counts were used for estimating trends in population abundance and the probability that there has been decline.

8.3 Methods

8.3.1 Adjusted Counts

For both retrospective and prospective analyses, the Negative-Binomial distribution was used to model the change in adjusted counts $N^{\mathrm{adj}}_{l,t,k,j}$.

$$N^{\mathrm{adj}}_{l,t,k,j} = \frac{N^{\mathrm{obs}}_{l,t,k,j}}{\alpha_{l,k} \cdot p_{l,t,\tau}}$$

$$N^{\mathrm{adj}}_{l,t,k} \sim \mathrm{NB}(\exp\{\mu_{l,t,k}\}, \theta)$$

$$\mu_{l,t,k} = \beta_{0,l} + \beta_l t + \log(\lambda_k)$$

where $N^{\mathrm{obs}}_{l,t,k,j}$ is the *j*[th] observation of (unadjusted) counts of dugongs at location $\ell \in [\mathrm{NGBR}, \mathrm{SGBR}]$, transect *k* and year *t;* $a_{l,k}$ is the availability probability at location *l* and transect *k;* $p_{l,k,\tau}$ is the overall detection probability at location *l,* transect *k*, and team τ;

μ and θ are the log-mean and overdispersion parameters, respectively, of the Negative Binomial distribution; $\beta_{0,l}$ is the intercept and β_l is trend parameter (long-run per-annum change in mean dugong abundance), at location *l*; and $\log(\lambda_k)$ is the length of transect *k* (an 'offset' to account for different transect lengths).

Therefore, the objective of the analyses was to estimate the posterior distribution of the trend variables β_l for the northern, southern and the entire Reef, using samples from $N^{\mathrm{adj}}_{l,t,k}$. This objective was achieved through a Monte Carlo Markov Chains sampler, which sampled from the posterior distribution of $(\beta_l, N^{\mathrm{adj}})$.

Target

After obtaining Monte Carlo Markov Chains samples from the posterior distribution of β_l, it is possible to calculate the probability that the trend was less than zero $\pi(\beta_l < 0|\mathcal{D})$, i.e. the probability that the population at *l* was in decline (conditional on all the input data $\mathcal{D}$).

The posterior probability $\pi(\beta_l < 0|\mathcal{D})$ requires a prior probability on β_l. I used the following truncated normal distribution:

$$\pi(\beta_l) \propto \mathcal{N}(\beta; 0.03, 0.075^2)\mathbb{I}[-0.162 < \beta < 0.067]$$

This distribution is presented in Figure 1 below. The prior places approximately 55 per cent of its density in the region between minus three per cent and three per cent, per annum change in abundance, with an upper cut-off at seven per cent per annum, and a lower cut-off at minus 15 per cent, per annum. The prior was sufficiently vague around zero, and had biologically reasonable upper-bounds on the intrinsic growth rate.

The adjusted dugong counts $N^{\mathrm{adj}}_{l,t,k}$ were stochastic and conditional on random variables α and *p*. The hierarchical nature of the analyses was designed to accommodate the extra observation error due to uncertainty in α and p by assuming that these parameters had probability distributions. These distributions were approximated by using the maximum likelihood estimation point-estimates and 95 per cent confidence intervals from previous frequentist analyses. The uncertainties in α and *p* were propagated through to the adjusted N^{adj} and the final trend estimates, through Monte Carlo Markov Chains integration over the distributions of α and *p*. More details on α and *p* are provided below.

8.3.2 Availability Probability

The availability probabilities α and their standard errors se(α) were estimated based on the turbidity and depth values at each region (northern and southern Reef) per year using the Hagihara method (Hagihara et al. 2014, 2018). Depth and turbidity measurements were discretised into 13 to 15 different categories per year and region. Each positive sighting of a dugong was assigned an availability category and given an α value. For each category, the maximum likelihood estimates $(\hat{\alpha}, \hat{se}(\alpha))$ were transformed into the shape parameters (*a*, *b*) of a Beta distribution, to facilitate sampling of α^*:

$$\alpha^*_v \sim \mathrm{Beta(a,b)}$$

where:

$$\mathrm{a} = \frac{(1-\hat{\alpha})\hat{\alpha}^2}{\hat{se}(\alpha)^2} - \hat{\alpha} \quad \text{and} \quad \mathrm{b} = \mathrm{a}(\frac{1}{\hat{\alpha}} - 1).$$

For example, consider a dugong sighted in a turbidity class 2 and depth between five and 25 metres in the southern Reef in 2005: previous analyses estimated $\hat{\alpha}$=0.652 and $\hat{se}(\alpha)$=0.188 for this sightability class. The equivalent Beta distribution would be Beta(3.53, 1.89).

8.3.3 Detection Probabilities

For each year and region (northern and southern Reef), there were several teams each of four observers in several aircraft. The observers sat in four positions (middle starboard seat, port middle seat, starboard back seat and port back seat), and visually searched for dugongs in the respective quadrants of their visual field. Previous studies used capture-recapture models to estimate the detection probability (correction for perception bias) per observer and per team. Due to the multi-model capture-recapture estimation process, the p estimates should also include model selection uncertainty (as approximated by AIC-weights; Link and Barker 2006), in addition to the parameter uncertainty, as expressed by the parameters' standard errors (standard errors were provided on the logit-scale). This study combined these multiple levels of uncertainty through Monte Carlo Markov Chains sampling.

For example, Team 1 in 2005 in the southern Reef had four possible models for *p;* the models had the following probabilities (as AIC-weights): i) 0.377 for the model with heterogeneous *p* between observers on the starboard side vs. the port side; ii) 0.334 for the model with homogeneous *p* among all observers; iii) 0.170 for the model with heterogeneous *p* for all observers; and iv) 0.119 for the model with heterogeneous *p* between the middle vs. the back seats. Considering only the best model in this example (model *i* above), the port-side detection probability was $p_{\text{port}} = 1 - (1 - p_{\text{mid-port}})(1 - p_{\text{back-port}})$, and a similar calculation defined the starboard side. The other models had similar calculations, as motivated by Pollock et al. (2006).

The entire hierarchical distribution for *p* was sampled in the following three steps. First, the capture-recapture model for *p* was sampled (per team and per year) based on model probabilities $m_{\tau_l} \in M_{\tau_l}$, using AIC-weights as approximations. Secondly, samples of *p* were taken from a logit-Normal distribution (per observer/quadrant):

$$\text{logit}(p^*)|m_{\tau_l} \sim \mathcal{N}\big(\text{logit}(\hat{p}), \hat{se}(p_{\text{logit}})|m_{\tau_l}\big)$$

conditional on the model m^* and previous maximum likelihood estimate parameter estimates. Finally, the observers' detection probabilities were aggregated into overall detection probabilities per side of the aircraft. These detection probabilities were assigned to observations of individual dugong groups and used to adjust the observed counts.

8.3.4 Monte Carlo Markov Chain (MCMC) Sampling

For both the retrospective and prospective analyses, the core elements of the MCMC sampler were as follows.

Inputs

- $N^{\text{obs}}_{l,t,k,j}$ is the *j*th observation of counts of dugongs at transect *k* and year *t* at location *l*. Each $N^{\text{obs}}_{l,t,k,j}$ observation has attribute data consisting of:
    - $s_{l,t,j}$ the side of the aircraft on which the observation took place,
    - $v_{l,t,j}$ the sightability class of the waters in which the observation took place,
    - $\tau_{l,t,j}$ the team of observers who made the *j*th observation.
- $\hat{\alpha}_{v_l}$ and $\hat{se}(\alpha)_{v_l}$ are the maximum likelihood estimate mean and standard error of the availability probability for each sightability class per location *l,*
- $\hat{w}_{\text{AIC}}(m_{\tau_{l,t}})$ is the AIC model weight for the *m*th model of detection probability for team $\tau_{l,t}$ at location *l* and year *t,*
- $\hat{p}^{\text{MP}}_{m,\tau_{l,t}}, \hat{p}^{\text{MS}}_{m,\tau_{l,t}}, \hat{p}^{\text{BP}}_{m,\tau_{l,t}}, \hat{p}^{\text{BS}}_{m,\tau_{l,t}}$ are the maximum likelihood estimate detection probabilities for each observer in four quadrants (*MP*: middle port, *MS*: middle starboard, *BP*: back port, *BS*: back starboard) for model *m* for team $\tau_{l,t}$ at location *l* and year *t; and* $\hat{se}^{\text{MP}}_{m,\tau_{l,t}}, \hat{se}^{\text{MS}}_{m,\tau_{l,t}}, \hat{se}^{\text{BP}}_{m,\tau_{l,t}}, \hat{se}^{\text{BS}}_{m,\tau_{l,t}}$ are their respective maximum likelihood estimate standard errors.

Probability distribution approximations

- <u>*Detection Probability Models*</u> Approximate model probabilities for each team per location and year: $\pi(m_{\tau_{l,t}}) \approx \hat{w}_{\text{AIC}}(m_{\tau_{l,t}})$. Therefore, sampling a random model *m** consists of sampling from a multinomial distribution:

$$m^* \sim \text{Mult}\big(\pi(m_{1,\tau_{l,t}}), \pi(m_{2,\tau_{l,t}}), \ldots, \pi(m_{4,\tau_{l,t}})\big)$$

- <u>*Detection Probabilities*</u> For each model per team per location and per year, define a logit-Normal probability distribution for each quadrants' detection probability (*MP,MS,BP,BS*)

$$\pi(\text{logit}(p)^{\text{MP}}_{\tau_{l,t}}|m^*) \approx \mathcal{N}\big(\text{logit}(\hat{p})^{\text{MP}}_{\tau_{l,t}}, \hat{se}(p)^{\text{MP}}_{\tau_{l,t}}|m^*\big)$$

Therefore, sampling a detection probability consists of a random sample from a Gaussian distribution

$$\text{logit}(p)^* \sim \pi\big(\text{logit}(p)^q_{m^*_{\tau_l,t}}\big) \text{ for } q \in [\text{MP,MS,BP,BS}]$$

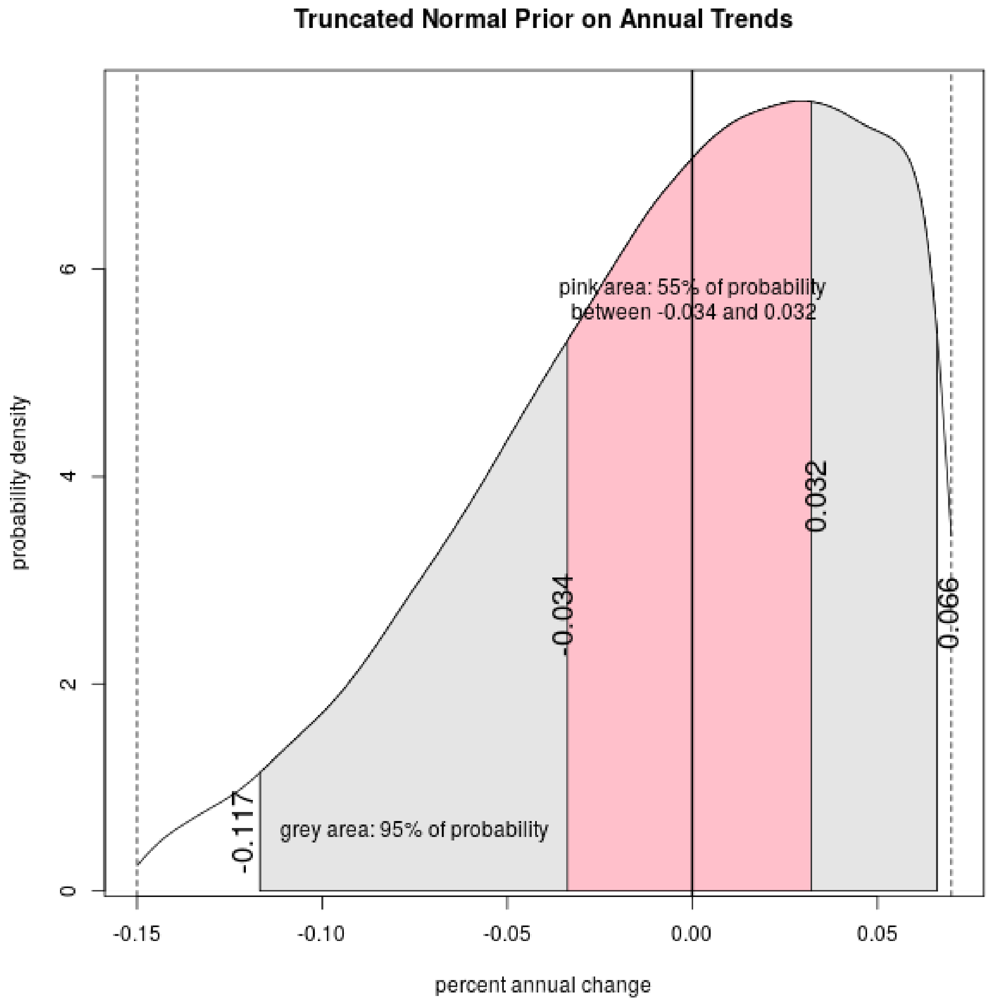

Figure 1. Truncated Normal prior for the change in mean abundance, used for estimating the trends in the northern and southern Great Barrier Reef.

- *Availability Probabilities.* For each location, define a Beta probability distribution for each sightability class v_l: $\pi(\alpha)_{v_l} \approx \mathrm{Beta}(\hat{\mathrm{a}}_l + 1, \hat{\mathrm{b}}_l + 1)$

  where *a* and *b* are Beta parameters calculated from maximum likelihood estimates $\hat{\alpha}_{v_l}$ and $\hat{se}(\alpha)_{v_l}$, plus a Beta(1,1) smoothing prior to ensure that $\pi(\alpha)_{v_l}$ was unimodal.

Sampler

Given the above inputs, the sampler proceeded as follows:

- Set priors on the Negative Binomial parameters:
  - $\pi(\beta_{0,l}) = \mathcal{N}(\beta; 0, 10^2)$ $\ell \in \{\mathrm{NGBR,SGBR}\}$
  - $\pi(\beta_l) \propto \mathcal{N}(\beta; 0.03, 0.075^2)\mathbb{I}[-0.162 < \beta < 0.067]$ for $\ell \in \{\mathrm{NGBR,SGBR}\}$
  - $\pi(\theta) = \mathcal{U}(\theta; 0.4, 30)$
- Initialise Negative Binomial regression coefficients β^* and overdispersion parameter θ^* from their prior distributions.

For each Monte Carlo Markov Chains iteration:

- sample model $m^*_{\tau_{l,t}}$ for each team $\tau_{l,t}$ at location *l* and year *t,*

- sample detection probabilities $p^*_{\tau_{l,t},q}|m^*_{\tau_{l,t}}$ for each quadrant *q* per team $\tau_{l,t}$ at location *l* and year *t,* conditional on model *m**,
- transform quadrant-based detection probabilities into side-based detection probabilities,
    - $p_P = 1 - (1 - p^*_{FP})(1 - p^*_{BP})$
    - $p_S = 1 - (1 - p^*_{FS})(1 - p^*_{BS})$
- sample availability probabilities $\alpha^*_{v_l}$ for each sightability category v_l per location *l*
- assign $(p^*_{\tau_l,t,s}, \alpha^*_{v_j})$ to each *j*th observation N^{obs}_j,
- calculate adjusted counts $N^{\text{adj}}_j = \dfrac{N^{\text{obs}}_j}{p^*_{\tau_l,t,s} \cdot \alpha^*_{v_j}}$ for each *j*th observation,
- sample the conditional posterior distributions of Negative Binomial parameters $\beta_{0,l}, \beta_l$ and θ conditional on the vector of observations, using Slice Sampling (Neal 2003)

$$\pi(\beta_{0,l}|\beta_l, \theta, N^{\text{adj}}) \propto \prod_{t=1}^{T} \prod_{j_t}^{J_t} \text{NB}\big(N^{\text{adj}}_{j_t}; e^{\beta_{0,l}+\beta_l \cdot t + \log(\lambda_k)}, \theta\big)\pi(\beta_{0,l}) \text{ for } l \in [\text{NGBR,SGBR}]$$

$$\pi(\beta_l|\beta_{0,l}, \theta, N^{\text{adj}}) \propto \prod_{t=1}^{T} \prod_{j_t}^{J_t} \text{NB}\big(N^{\text{adj}}_{j_t}; e^{\beta_{0,l}+\beta_l \cdot t + \log(\lambda_k)}, \theta\big)\pi(\beta_l) \text{ for } l \in [\text{NGBR,SGBR}]$$

$$\pi(\theta|\boldsymbol{\beta}, N^{\text{adj}}) \propto \prod_{l \in [\text{NGBR,SGBR}]} \prod_{t=1}^{T} \prod_{j_t}^{J_t} \text{NB}\big(N^{\text{adj}}_{j_t}; e^{\beta_{0,l}+\beta_l \cdot t + \log(\lambda_k)}, \theta\big)\pi(\theta)$$

After n_{mcmc} =80000 iterations, calculate the probability of a decline for each location *l*

$$\pi(\beta_l < 0|\mathcal{D}) \approx \frac{1}{n_{\text{mcmc}}} \sum_{i=1}^{n_{\text{mcmc}}} \mathbb{I}[\beta_l^{(i)} < 0]$$

The above Monte Carlo Markov Chains steps conclude the sampling of the trend parameters θ_l.

8.3.5 Prospective Analysis

The prospective analysis used the same Monte Carlo Markov Chains sampling method outlined in the previous section to estimate annual trends, but instead of using the observed data, the analysis used future hypothetical datasets generated under different scenarios. The datasets were generated using Negative Binomial distributions which matched the empirical characteristics of the fitted Negative Binomial distributions from the retrospective analyses.

The goal of the prospective analyses was to assess the program's ability to detect future declines, as measured by the Bayesian posterior probability of a decline $\pi(\beta < 0|\mathcal{D}^{\text{sim}})$, under a variety of scenarios, given ‘true’ declines of *i*) minus three per cent, per annum and *ii*) minus one per cent, per annum.

The prospective scenarios included variations in the following survey parameters:

- the ‘true’ trend (minus one per cent, per annum and minus three per cent, per annum),
- number of years of surveys (eight,16, and 24 years total),
- inter-year sampling frequency (surveys every two, four, five, and eight years),
- reduction in detection error (same, halved, and set to zero).

The first scenario parameter addressed the question: how many years of data would be necessary to detect a decline in dugong abundance with 80 per cent probability? The second scenario parameter addressed the question: how often should the surveys be conducted in order to increase the probability of detecting a decline? The third scenario parameter addressed the question: can the probability of detecting a decline be increased by reducing observers' probability of missing a dugong (for example, due to better technology, personnel training)?

Generating Simulations

In order to assess future hypothetical scenarios in a realistic manner, Monte Carlo simulations were used, in order to flexibly incorporate key empirical and statistical features of the dugong count data, such as the overdispersed count distribution (Negative Binomial θ), as well as observation errors (detection probabilities *p* and availability probabilities α).

Assumptions. In order to make future projections about key processes, such as *p* and α, additional assumptions were required about the future conditions. In particular, *p* was conditional on the team of observers who conducted surveys, while α was conditional on the water depth and turbidity (as classified by the sightability class *v*). These parameters depend on future unknowns. Although we cannot know such future environmental conditions, for the purpose of the prospective analyses, it is reasonable to assume that future conditions will likely arise from a similar distribution as past conditions. Therefore, bootstrap sampling (i.e. sampling-with-replacement) was used to approximate from the empirical distribution of sightability classes from past transects $\mathrm{EDF}_k(v)$[8]. For each bootstrap draw of a transect k^*, the availability probability α_k was set according to the maximum likelihood estimate of the originating transect's sightability class v_k. Likewise, bootstrap sampling was used to approximate the distribution of teams and their observer error $\tau^* \sim \mathrm{EDF}(\boldsymbol{\tau}_{l,t})$. For each bootstrapped sample of team τ^*, the team’s detection probabilities *p* were set according to the maximum likelihood estimate of the originating team's $\hat{p}$. These were reasonable assumptions for the prospective analyses, barring any drastic changes in the future environmental conditions and/or observation error.

The other Negative Binomial parameters were set according to the posterior means of the retrospective Bayesian analysis, including overdispersion ($\bar{\theta}$ = 0.91) and the intercepts for

8 Measures of depth and visibility were recorded along the aircrafts’ flight paths; these data formed the background distribution of sightability categories which were distinct from the “presence-only” sightability categories recorded at the locations of dugongs.

the average density $\exp\{\beta_{0,l}\}$ per location (northern Reef=-0.766, southern Reef=0.522), which were set according to the estimated posterior mean densities in year 2016, from the retrospective analysis.

The Bootstrap/simulation procedure was as follows:

- set simulation parameters: $\beta_l \in [\log(0.99), \log(0.97)]$ (trend parameter), T^{sim} (number of years of sampling), $\vec{\mathbf{t}}$ (sequence of years with surveys), and ϵ (detection error reduction).
- for each location $\ell \in [\text{SGBR, NGBR}]$, for each year $t \in \vec{t}$, draw a bootstrap sample of transects $k^* \sim \text{EDF}_k$, such that the sum of the transects' length per year equalled 4,870 kilometres and 3,962 kilometres, in the northern and southern Reef respectively. Get the sightability classes and availability probabilities (v^*, α_{v^*}) from the originating transects and assign to the bootstrapped sample of transects.
- for each location $\ell \in [\text{SGBR, NGBR}]$, and for each year $t \in \vec{t}$, bootstrap sample the team of observers, $\tau^* \sim \text{EDF}_\tau$, such that there were two teams per year/location. Get each team's capture-recapture models $M_{\tau_{l,t}}$ for detection probabilities.
- for each τ^* sample of a team, sample from the detection capture-recapture models $m^* \sim \text{Mult}(\mathbf{M}_{\tau_{l,t}})$, and use the maximum likelihood estimate of $\hat{p}$ for each sampled model m^* for the calculation of per-side detection probabilities (p^*_P, p^*_S)
- calculate the mean abundance per transect per year and location:

  $\text{logit}(\mu_{k^*,t}) = \beta_{0,l} + \beta_l \cdot t + \log(\lambda_{k^*})$
- sample counts *N* from a Negative Binomial distribution for each transect *k:*

  $N^*_k \sim \text{NB}(\mu_{k^*,t}, \bar{\theta})$
- randomly assign N^* dugongs to the port and starboard sides of the aircraft with 50 per cent probability: $N^{\text{P}}_k = \text{Bin}(N^*_k, 0.5);\; N^{\text{S}}_k = N^*_k - N^{\text{P}}_k$
- sample counts of observed dugongs for observation *j* per side *s*, including misses due to the imperfect availability probability and detection probability:

  $N^{\text{obs}}_j \sim \text{Bin}(N^s_k, \alpha_{v_k} \cdot p_s)$

With the simulated counts of dugongs N^{obs}_j per transect, per year and per location, the final step was to use the MCMC sampler from the retrospective analysis to estimate the trend $\pi(\beta_l|\mathcal{D}^{\text{sim}})$ as well as calculate the probability of a decline $\pi(\beta_l < 0|\mathcal{D}^{\text{sim}})$. The probability of a decline was averaged over 100 simulations per scenario for reporting.

Results 8.3.6 Retrospective Analysis

The estimated trend in the norther Reef was minus 3.14 per cent, per annum (SE: 3.844; 95%CI -11.057 – 4.552), while the estimate trend in the southern Reef was minus 1.460 per cent, per annum (SE: 3.345; 95%CI -7.531 – 5.912). The composite trend for the entire Reef was minus 2.301 per cent, per annum (SE: 3.699; 95%CI -9.737 – 5.379). The posterior densities are shown in Figure 2. Notice the significant difference between the prior densities (*red lines*) versus the posterior densities, which demonstrates Bayesian learning from the data.

The probability that a decline took place in the northern Reef between 2006 and 2013 was 0.791, just less than the 0.8 power target. In the southern Reef, the probability that a decline took place between 2005 and 2016 was 0.689. The probability that there was an overall decline across the entire Reef between 2005 and 2016 was 0.740.

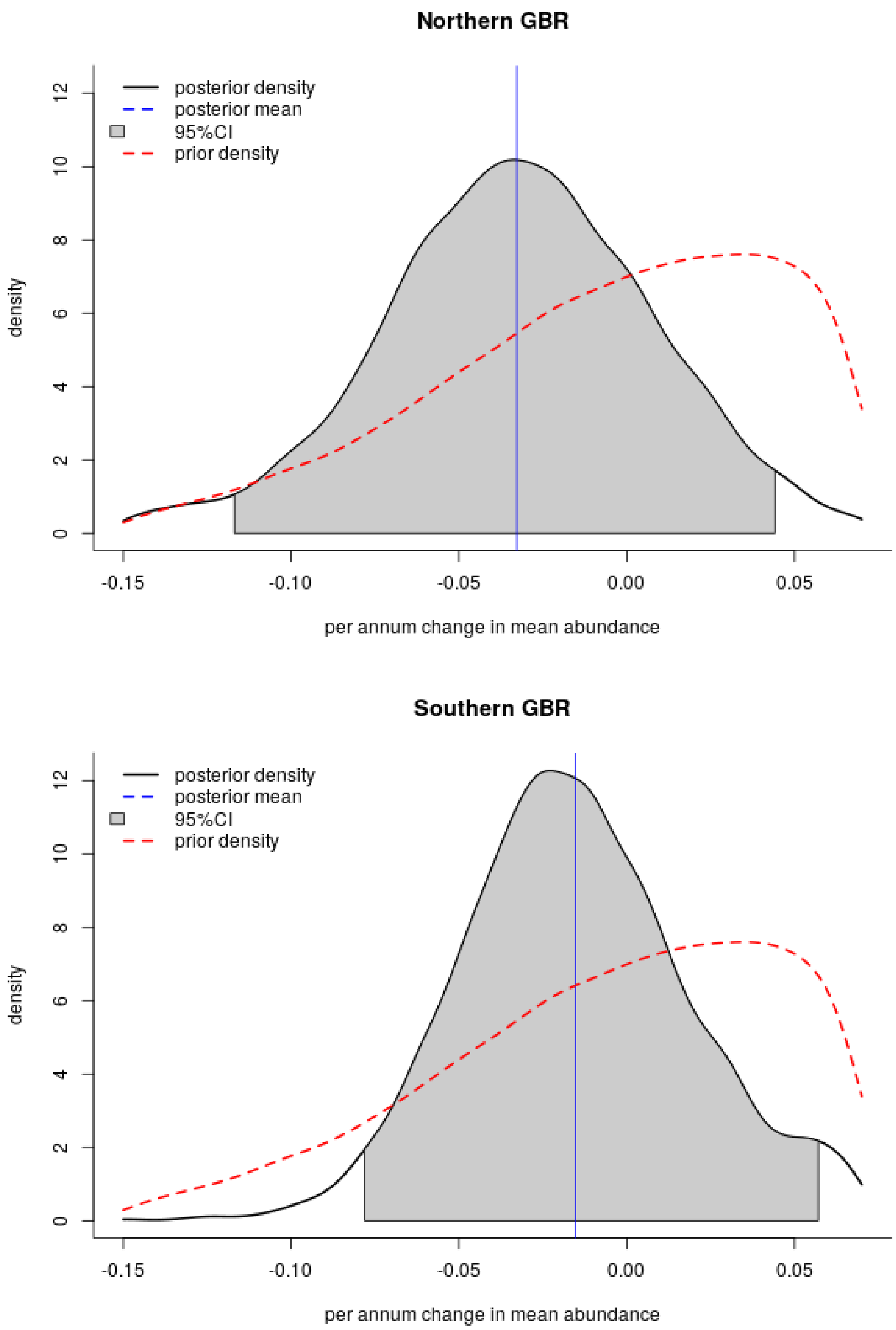

Figure 2. Posterior distribution of trend estimates in dugong abundance for the Great Barrier Reef based on the retrospective analysis of data collected between 2005 and 2016.

8.3.7 Prospective Analysis

In the following results, the focal measure is the Bayesian probability of detecting a decline, averaged over 100 simulations (hereafter, simply referred to as 'power'). The target power was an 80 per cent probability of detecting a decline, which is the Bayesian analogue of the 0.8 power target in conventional frequentist power analyses. Table 1 shows the probability of detecting a future decline under various scenarios, such as different rates of population decline (minus one per cent, per annum decline versus minus three per cent, per annum decline; *major columns*), different sampling frequencies (every two, four, versus eight years; *minor columns*), different time horizons (eight, 16 versus 24 years; *rows*), and under different reductions in the detection error (same, half error, and zero per cent error; *tables A, B and C*). The current survey regime is to try to survey every five years. The five-year results are not shown in Table 1 along with the rest of the results[9], but are discussed below.

8.4 Evaluation of Current Survey Regime

With surveys every five years, the results suggest that it would take at least 20 years to achieve a power of 0.8, given a true decline of minus one per cent, per annum (norther Reef power was 0.8 (SE 0.092), southern Reef was 0.806 (SE 0.084), and their combined power was 0.802 (SE 0.057)). More frequent surveys at two or four years did little to improve these numbers (see Table 1). The time needed to achieve the target power could be reduced to approximately 16 years of surveys if the surveys were conducted every two to four years.

With a steeper population decline of minus three per cent per annum, the target power of 0.8 could be reached after approximately eight years of surveys (an estimated power of 0.788 (SE 0.1) for the northern Reef, 0.828 (SE 0.086) for the southern Reef, and 0.809 (SE 0.068) for both). However, at a duration of five years, the power estimates were much lower at of 0.697 (SE 0.11), 0.735 (SE 0.105), and 0.712 (SE 0.08), respectively.

Table 1. Results of prospective analysis: the probability of detecting a decline under various scenarios (rates of population decline, survey frequencies, different time horizons, and reductions in the detection error.

A) No reduction in detection error

		Decline = -1% per annum			**Decline = -3% per annum**		
Location	**Duration**	**Freq=2**	**4**	**8**	**2**	**4**	**8**
Northern Reef	8	0.634 (0.149)	0.636 (0.138)	0.62 (0.141)	**0.848 (0.083)**	**0.8 (0.106)**	0.79 (0.1)
	16	0.796 (0.095)	0.753 (0.111)	0.729 (0.109)	**0.979 (0.022)**	**0.961 (0.033)**	**0.945 (0.034)**

9 The five year intervals did not divide evenly into the survey durations (8, 16, 24), making the interpretation of the results from the 5 year scenarios slightly more nuanced than the other intervals (which are multiples of 2).

	24	**0.905 (0.064)**	**0.848 (0.073)**	**0.823 (0.092)**	**0.999 (0.001)**	**0.995 (0.004)**	**0.988 (0.008)**
Southern Reef	8	0.679 (0.115)	0.648 (0.11)	0.635 (0.12)	**0.875 (0.061)**	**0.85 (0.066)**	**0.828 (0.086)**
	16	**0.802 (0.091)**	0.77 (0.104)	0.742 (0.106)	**0.987 (0.011)**	**0.974 (0.015)**	**0.96 (0.023)**
	24	**0.921 (0.051)**	**0.871 (0.073)**	**0.84 (0.086)**	**0.999 (0.001)**	**0.996 (0.002)**	**0.992 (0.004)**
Both	8	0.655 (0.101)	0.64 (0.089)	0.624 (0.1)	**0.862 (0.048)**	**0.825 (0.058)**	**0.81 (0.068)**
	16	0.799 (0.069)	0.762 (0.076)	0.736 (0.075)	**0.984 (0.012)**	**0.968 (0.018)**	**0.953 (0.02)**
	24	**0.913 (0.043)**	**0.86 (0.048)**	**0.832 (0.062)**	**0.999 (0.001)**	**0.996 (0.002)**	**0.99 (0.005)**

B) Halving detection error

		Decline = -1% per annum			**Decline = -3% per annum**		
Location	**Duration**	**Freq=2**	**4**	**8**	**2**	**4**	**8**
Northern Reef	8	0.641 (0.129)	0.627 (0.133)	0.664 (0.108)	**0.839 (0.083)**	**0.821 (0.094)**	**0.803 (0.101)**
	16	0.781 (0.092)	0.731 (0.12)	0.738 (0.111)	**0.982 (0.012)**	**0.965 (0.023)**	**0.949 (0.033)**
	24	**0.916 (0.051)**	**0.861 (0.072)**	**0.831 (0.086)**	**0.999 (0.001)**	**0.996 (0.003)**	**0.988 (0.008)**
Southern Reef	8	0.673 (0.129)	0.625 (0.126)	0.644 (0.131)	**0.873 (0.075)**	**0.832 (0.077)**	**0.838 (0.077)**
	16	**0.807 (0.078)**	0.772 (0.096)	0.757 (0.105)	**0.988 (0.009)**	**0.974 (0.018)**	**0.96 (0.023)**
	24	**0.929 (0.041)**	**0.868 (0.075)**	**0.85 (0.066)**	**0.999 (0.001)**	**0.997 (0.002)**	**0.994 (0.004)**

	8	0.657 (0.093)	0.623 (0.097)	0.652 (0.09)	**0.857 (0.057)**	**0.829 (0.058)**	**0.821 (0.061)**
Both	16	0.794 (0.061)	0.753 (0.079)	0.748 (0.083)	**0.985 (0.007)**	**0.97 (0.015)**	**0.955 (0.02)**
	24	**0.923 (0.036)**	**0.865 (0.051)**	**0.842 (0.053)**	**0.999 (0.001)**	**0.997 (0.002)**	**0.991 (0.004)**

C) Zero detection error

		Decline = -1% per annum			Decline = -3% per annum		
Location	**Duration**	**Freq=2**	**4**	**8**	**2**	**4**	**8**
Northern Reef	8	0.641 (0.137)	0.631 (0.13)	0.613 (0.137)	**0.846 (0.08)**	**0.817 (0.088)**	**0.824 (0.098)**
	16	0.789 (0.097)	0.745 (0.109)	0.731 (0.111)	**0.984 (0.011)**	**0.966 (0.025)**	**0.951 (0.027)**
	24	**0.892 (0.063)**	**0.853 (0.08)**	**0.82 (0.092)**	**0.999 (0.001)**	**0.996 (0.003)**	**0.99 (0.008)**
Southern Reef	8	0.665 (0.129)	0.662 (0.115)	0.634 (0.123)	**0.865 (0.076)**	**0.836 (0.079)**	**0.822 (0.087)**
	16	0.826 (0.081)	0.797 (0.09)	0.771 (0.096)	**0.99 (0.006)**	**0.975 (0.017)**	**0.967 (0.018)**
	24	**0.93 (0.036)**	**0.877 (0.064)**	**0.847 (0.084)**	**0.999 (0.001)**	**0.997 (0.002)**	**0.992 (0.005)**
Both	8	0.651 (0.095)	0.647 (0.091)	0.621 (0.09)	**0.856 (0.056)**	**0.828 (0.06)**	**0.823 (0.067)**
	16	**0.808 (0.063)**	0.77 (0.066)	0.751 (0.071)	**0.987 (0.006)**	**0.971 (0.015)**	**0.96 (0.018)**
	24	**0.911 (0.037)**	**0.866 (0.049)**	**0.834 (0.059)**	**0.999 (0)**	**0.997 (0.002)**	**0.991 (0.005)**

As shown in the Table 1, the most influential factors affecting power were: i) the ‘true’ decline (minus one per cent versus minus three per cent, per annum), and ii) the total duration of years monitoring the trend. Other factors, such as the frequency of sampling, or reducing the error of detection probability, were much less influential. Both locations had similar levels of power, although the southern Reef had consistently higher power of about 0.02 to 0.05 probability units compared with the northern Reef.

8.4.1 Detecting trends

More subtle declines were much more difficult to detect than steeper declines. A true population decline of minus one per cent per annum was very difficult to detect with a power of 0.8, except at long time horizons (over 16 years of sampling). For example, a relatively intense frequency of surveying every two years for 16 years resulted in a power of 0.796 (SE 0.095) in the northern Reef, and a power of 0.802 (SE 0.091) in the southern Reef assuming the current detection error. More relaxed sampling frequencies, such as every four years, reduced the power by about 0.04 to 0.06 probability units. The least data-intensive scenario of only two surveys eight years apart resulted in the weakest power of 0.624 (SE 0.1) for both northern Reef and southern Reef combined. However, even this low power resulted in a 'preponderance of evidence' which favoured the declaration of a population decline (i.e. a 1.6:1 odds-ratio in favour of a decline).

In contrast, with steeper population declines of minus three per cent per annum, most scenarios achieved power close to or exceeding 0.8. The least data-intensive scenario with two surveys in eight years resulted in a power of 0.81 (SE 0.068) for the entire Reef. Surveying every four years for eight years resulted in a power of 0.825 (SE 0.06) and sampling every two years for eight years yielded 0.862 (SE 0.048). Survey durations of 16 years or greater resulted in powers much greater than 0.9.

8.4.2 Survey frequency and error reduction

Overall, the survey attributes that were most amenable to managerial discretion (for example, sampling frequency, reducing detection error) were much less effective at increasing power, compared to the influence of intrinsic factors such as the total length of time or the magnitude of the trend. For instance, increasing the frequency of sampling resulted in only minor increases in power. Reducing the observers' detection error had almost no effect on power (this result was likely due to the fact that the existing capture-recapture techniques were highly effective and further improvements to detection probability would only have limited effects). The results suggest that there is intrinsically a lot of statistical uncertainty for the dugong populations which are widely and sparsely distributed.

8.4.3 Duration of monitoring

In contrast, waiting more years to capture a trend was highly influential on power. Consider that with only two surveys over five years of monitoring a trend (at minus three per cent, per annum), the resulting power was 0.697 (SE 0.11), whereas two surveys eight years apart yielded a power of 0.81 (SE 0.068) (for the entire Reef). Longer durations all had very high power.

However, it is important to note that these results only pertain to log-linear declines, and do not include the possibility of catastrophic declines that would be missed if the time between surveys was very long. This is a key reason for using the lines of evidence approach outlined in the main report.

8.5 Conclusions

The Bayesian probability of a population decline in dugongs across the entire Reef was 0.740 for the period 2005 to 2016. For the northern and southern Reef regions, separately, the probabilities were 0.791 and 0.689, or an odds-ratio of 4:1 and 2.2:1 in favour of population declines, respectively. These results were based on Bayesian hierarchical models which incorporated excess uncertainty due to detection error, availability bias, model selection uncertainty, and count overdispersion.

Looking forward, a Bayesian prospective 'power' analysis was used to estimate the ability of the program to detect trends according to a number of different scenarios. I estimated that the current five-year survey regime had a reasonable ability to detect log-linear declines of minus three per cent, per annum (or more extreme) with a 0.8 probability at mid- to long-term time horizons (such as eight years or greater). The ability to detect trends within shorter periods of time, such as five years, was much less reliable (<0.7 power), and may not improve significantly with more frequent surveying or other tweaks to the survey protocol. Shallower trends will be much more difficult to detect, and may only achieve reasonable power at long time horizons (such as 16 to 20 years).

The high amount of heterogeneity in the system (due to factors such as availability bias, overdispersion) make it difficult to detect shallow trends, such as minus one per cent per annum. This situation also means that additional types of survey protocol enhancements (such as increasing sampling frequency or decreasing detection) are estimated to have little impact on improving power. Certainly, there are many other tweaks to the survey protocol which were not explored in this study but could possible improve the power, such as repeat sampling of transects, or increasing the level of sampling stratification[10] by population density. However, such modifications may not be backwards compatible with the existing time-series of counts. As the results demonstrate, having longer time-series is the most important quality to detect a trend.

Lastly, it is noteworthy that the Negative Binomial overdispersion parameter was very small at less than one (a smaller value means more overdispersion), suggesting that the counts were highly unevenly distributed and heavily zero-inflated. Future studies could explore more sophisticated modelling techniques to address this excess heterogeneity, such as incorporating spatial variation in occupancy, zero-inflation, or clustering (Warton and Shepherd 2010). Finally, at low availability probabilities α in areas of extensive deep water seagrass such as Torres Strait, the Horvitz-Thompson-like estimator may be unreliable and could be improved with an N-mixture model (Royle 2004).

8.6 References

Commonwealth 2018. Reef 2050 Long-Term Sustainability Plan—July 2018. Commonwealth of Australia 2018.

[10] The optimised survey design is already highly stratified by dugong density.

Hagihara, R., R. E. Jones, A. Grech, J. M. Lanyon, J. K. Sheppard, and H. Marsh. 2014. Improving population estimates by quantifying diving and surfacing patterns: A dugong example. Marine Mammal Science 30:348–366.

Hagihara, R., R. E. Jones, S. Sobtzick, C. Cleguer, C. Garrigue, and H. Marsh. 2018. Compensating for geographic variation in detection probability with water depth improves abundance estimates of coastal marine megafauna. PLOS ONE 13:e0191476.

Link, W. A., and R. J. Barker. 2006. Model weights and the foundations of multimodel inference. Ecology 87:2626–2635.

Marsh, H., and D. F. Sinclair. 1989. Correcting for visibility bias in strip transect aerial surveys of aquatic fauna. The Journal of Wildlife Management 53:1017–1024.

Neal, R. M. 2003. Slice sampling. The Annals of Statistics 31:705–767.

Pollock, K. H., H. D. Marsh, I. R. Lawler, M. W. Alldredge, and Lubow. 2006. Estimating animal abundance in heterogeneous environments: an application to aerial surveys for dugongs. Journal of Wildlife Management 70:255–262.

Royle, J. A. 2004. N-mixture models for estimating population size from spatially replicated counts. Biometrics 60:108–115.

Warton, D. I., and L. C. Shepherd. 2010. Poisson point process models solve the “pseudo-absence problem” for presence-only data in ecology. The Annals of Applied Statistics 4:1383–1402.

9.0 Appendix 2 — Comparison of dugong population estimates and survey intensities for individual blocks using the original and optimised aerial survey designs.

(a) Southern Great Barrier Reef region

Block	2005		2016		Sampling intensity (%)	
	original survey design	optimised survey design	original survey design	optimised survey design	2016 survey	optimised survey
S1	zzt	zzt	tfs	Tfs	8.82	6.86
S2	tfs	tfs	tfs	Tfs	8.73	4.9
S2A	tfs	tfs	tfs	Tfs	14.17	unchanged
S3	134 (82)	72 (49)	tfs	Tfs	15.9	13.87
S4	zzt	zzt	tfs	Tfs	8.95	5.2
S5	611* (174)	611* (174)	583** (222)	583** (222)	17.3	unchanged
S6	dd	dd	tfs	Tfs	7.26	5.8
S7	zzt	zzt	tfs	Tfs	8.74	4.48
S8	tfs	tfs	122 (88)	122 (88)	13.3	unchanged
C1	tfs	tfs	tfs	Tfs	16.41	unchanged
C2	ns	ns	nds	Nds	5.36	deleted
C3	tfs	tfs	tfs	Tfs	12.5	unchanged
C4	74 (41)	74 (41)	265 (160)	265 (160)	16.62	unchanged
C5	ns	ns	tfs	Tfs	8.54	6.03
C6	173 (82)	173 (82)	tfs	Tfs	16.68	unchanged
C7	tfs	tfs	tfs	Tfs	16.9	unchanged
C8	193 (101)	193 (101)	1171 (423)	1171 (423)	18.18	unchanged
C9	zzt	zzt	361 (252)	487 (300)	8.27	5.44
C10	266 (165)	266 (165)	320 (187)	320 (187)	18.31	unchanged
C11	107 (85)	107 (85)	ns	Ns	ns	unchanged

C12	zzt	zzt	ns	Ns	ns	4.79
C13	ns	ns	ns	Ns	ns	
Total	**1558 (300)**	**1496 (293)**	**2822 (600)**	**2948 (622)**		
CV	**0.193**	**0.196**	**0.213**	**0.211**		

Notes:

1. Data from the 2011 survey were not included because the distribution of dugongs was aberrant due to seagrass loss associated with extreme weather events
2. Previously reported abundance estimates are from Sobtzick et al. (2017).
3. Shaded cells indicate differences in population estimates between original and optimised survey design.
4. *herds of 12, 50 and 70 dugongs;
5. **a herd of 8 dugongs
6. zzt-zig zag transect
7. tfs-too few sightings
8. nds-no dugong sightings
9. ns-not surveyed
10. Only transects from the optimised survey design were retained in the data for 2016.
11. For 2005, data from the zig-zag transects were retained but data from transects not incorporated in the optimised design were removed. Thus the CV for 2005 may not be directly comparable with CVs reported for other years.

(b) Northern Great Barrier Reef region

Block	2006		2013		Sampling intensity (%)	
	original survey design	optimised survey design	original survey design	optimised survey design	2013 survey	optimised survey
N1	tfs	tfs	tfs	tfs	8.8	9.8
N2	1293* (466)	1293* (466)	820*** (278)	820*** (278)	16.9	unchanged
N3	498 (249)	498 (249)	1077 (612)	1077 (612)	17.6	unchanged
N4	1629** (693)	1619** (802)	973 (367)	597 (200)	9	10.06
N5	3061 (1333)	3061 (1333)	1990 (675)	1990 (675)	9	unchanged
N6	tfs	tfs	504 (306)	504 (306)	8.8	unchanged
N7	tfs	tfs	tfs	tfs	9.1	10.14
N8	1407 (725)	1407 (725)	979 (394)	979 (394)	8.7	unchanged
N9	tfs	tfs	tfs	tfs	8.8	6.76

N10	tfs	tfs	tfs	tfs	9.2	unchanged
N11	293 (116)	293 (116)	108 (71)	108 (71)	25.3	unchanged
N12	tfs	tfs	tfs	tfs	3.7	unchanged
N13	492 (211)	189 (105)	nds	nds	6.3	6.97
N14	139 (106)	89 (57)	107 (75)	58 (40)	23.1	25.87
N15	tfs	tfs	nds	nds	9.7	4.55
Total	**8812 (1769)**	**8449 (1803)**	**6558 (1141)**	**6133 (1097)**		
CV	**0.201**	**0.213**	**0.174**	**0.179**		

Notes:

1. Previously reported abundance estimates are from Sobtzick *et al.* (2014). Shaded cells indicate differences in population estimates between the original and optimised survey designs.
2. The original survey design had zig-zag transects in block N15 while the optimised design has transects parallel to the coat which have never been flown. Thus, dugong abundance was not estimated for this block.
3. *herds of 20, 20, 15, 27, 10 dugongs
4. ** herd of 10
5. ***herd of 49 dugongs
6. tfs-too few sightings
7. nds-no dugong sightings

(c) Torres Strait

<table>
<tr><th rowspan="2">Block</th><th colspan="2">2006</th><th colspan="2">2011</th><th colspan="2">2013</th><th colspan="2">Sampling intensity (%)</th></tr>
<tr><th>original survey design</th><th>optimised survey design</th><th>original survey design</th><th>optimised survey design</th><th>original survey design</th><th>optimised survey design</th><th>2013 survey</th><th>optimised survey</th></tr>
<tr><td>TS0</td><td>tfs</td><td>tfs</td><td>3870 (3712)</td><td>3870 (3712)</td><td>2962 (2874)</td><td>2962 (2874)</td><td>4.4</td><td>unchanged</td></tr>
<tr><td>TS1A</td><td>5323 (3478)</td><td>5323 (3478)</td><td>2008 (1191)</td><td>2008 (1191)</td><td>tfs</td><td>tfs</td><td>8.9</td><td>unchanged</td></tr>
<tr><td>TS1B</td><td>7405 (3182)</td><td>7405 (3182)</td><td>9876 (4989)</td><td>9876 (4989)</td><td>10840 (4419)</td><td>10840 (4419)</td><td>4.9</td><td>unchanged</td></tr>
<tr><td>TS2A</td><td>26824 (5050)</td><td>26824 (5050)</td><td>36228 (10026)</td><td>36228 (10026)</td><td>35380 (9412)</td><td>35380 (9412)</td><td>8.2</td><td>unchanged</td></tr>
<tr><td>TS2B</td><td>5166 (2238)</td><td>5166 (2238)</td><td>6609 (3128)</td><td>6609 (3128)</td><td>4516 (1981)</td><td>4516 (1981)</td><td>8.6</td><td>unchanged</td></tr>
<tr><td>TS3A</td><td rowspan="2">24496 (8495)</td><td rowspan="2">21637 (6030)</td><td rowspan="2">16843 (7365)</td><td rowspan="2">23844 (4141)</td><td rowspan="2">38417 (16185)</td><td rowspan="2">33069 (11757)</td><td>part of TS3</td><td>6.6</td></tr>
<tr><td>TS3B</td><td>part of TS3</td><td>4.8</td></tr>
<tr><td>TS4</td><td>15175* (8091)</td><td>15795* (7778)</td><td rowspan="2">1839 (792)</td><td rowspan="2">1789 (949)</td><td>10104 (4859)</td><td>10390 (4726)</td><td>4.4</td><td>4.3</td></tr>
<tr><td>TS5</td><td>nds</td><td>nds</td><td>tfs</td><td>tfs</td><td>11.2</td><td>5.9</td></tr>
<tr><td>TS6</td><td>ns</td><td>ns</td><td>nds</td><td>nds</td><td>nds</td><td>nds</td><td>3.9</td><td>deleted</td></tr>
<tr><td>TS7</td><td>ns</td><td>ns</td><td>nds</td><td>nds</td><td>nds</td><td>nds</td><td>3.6</td><td>deleted</td></tr>
<tr><td>TS8</td><td>ns</td><td>ns</td><td>2636 (1795)</td><td>na**</td><td>tfs</td><td>tfs</td><td>4.9</td><td>deleted</td></tr>
<tr><td>TS9</td><td>ns</td><td>ns</td><td>3463 (2719)</td><td>3734 (3922)</td><td>tfs</td><td>tfs</td><td>4.5</td><td>4.4</td></tr>
<tr><td>Total</td><td>84389 (13797)</td><td>82150 (12231)</td><td>83372 (14693)</td><td>87958 (14754)</td><td>102519 (20146)</td><td>97157 (16759)</td><td></td><td></td></tr>
</table>

CV	**0.163**	**0.149**	**0.176**	**0.168**	**0.197**	**0.172**

Notes

1. Previously reported abundance estimates are from Hagihara *et al.* (2016).
2. Shaded cells indicate differences in population estimates between original and optimised survey design.
3. *a herd of 15 dugongs
4. **eastern side of transect in Block TS8 was incorporated into Block TS3B in the optimised design. Thus dugongs sighted in Block TS8 were incorporated into the abundance estimation for Block 3B.
5. tfs-too few sightings; nds-no dugong sightings; ns-not surveyed; na-not applicable
6. In 2006, Blocks TS 6-8 were not surveyed. Thus, the block size used to estimate abundance was the original block size and no dugong sightings were imputed for Blocks TS 3A and 3B.
7. The survey intensity of Block TS3A was almost doubled in the optimised design and included some of the former blocks TS7 and TS8. To take this increase into account, the relevant dugong sightings were duplicated to accommodate the change in intensity in blocks 3A and 3B.

10.0 Appendix 3 — Comparison of turtle population estimates and survey intensities for individual blocks using the original and optimised survey designs

(a) Southern Great Barrier Reef region

<table>
<tr><th rowspan="2">Block</th><th colspan="2">2005</th><th colspan="2">2016</th><th colspan="2">Sampling intensity (%)</th></tr>
<tr><th>original survey design</th><th>optimised survey design</th><th>original survey design</th><th>optimised survey design</th><th>2016 survey</th><th>optimised survey</th></tr>
<tr><td>S1</td><td>1300 (1020)</td><td>750 (731)</td><td>tfs</td><td>tfs</td><td>8.82</td><td>6.86</td></tr>
<tr><td>S2</td><td>5258 (3559)</td><td>7204 (5458)</td><td rowspan="2">2125 (2010)</td><td>782 (1077)</td><td>8.73</td><td>4.9</td></tr>
<tr><td>S2A</td><td>tfs</td><td>tfs</td><td>nds</td><td>14.17</td><td>unchanged</td></tr>
<tr><td>S3</td><td>1859 (1092)</td><td>1562 (1218)</td><td>1775 (1165)</td><td>1326 (1089)</td><td>15.9</td><td>13.87</td></tr>
<tr><td>S4</td><td>7967 (3788)</td><td>2936 (2635)</td><td>11769 (5250)</td><td>5801 (4687)</td><td>8.95</td><td>5.2</td></tr>
<tr><td>S5</td><td>24437 (5000)</td><td>24437 (5153)</td><td>44063 (9277)</td><td>44063 (9277)</td><td>17.3</td><td>unchanged</td></tr>
<tr><td>S6</td><td>12265 (5077)</td><td>11274 (7307)</td><td>25640 (14148)</td><td>25566 (15649)</td><td>7.26</td><td>5.8</td></tr>
<tr><td>S7</td><td>2712 (1893)</td><td>2862 (2860)</td><td>6327 (3550)</td><td>2792 (1704)</td><td>8.74</td><td>4.48</td></tr>
<tr><td>S8</td><td>6283 (3297)</td><td>6283 (3276)</td><td>10531 (4679)</td><td>10531 (4679)</td><td>13.3</td><td>unchanged</td></tr>
<tr><td>C1</td><td>2123 (2083)</td><td>2123 (2083)</td><td>1821 (1397)</td><td>1821 (1397)</td><td>16.41</td><td>unchanged</td></tr>
<tr><td>C2</td><td>tfs</td><td>tfs</td><td>tfs</td><td>tfs</td><td>5.36</td><td>deleted</td></tr>
<tr><td>C3</td><td>15127 (5349)</td><td>15127 (5349)</td><td>9162 (2500)</td><td>9162 (2500)</td><td>12.5</td><td>unchanged</td></tr>
<tr><td>C4</td><td>619 (513)</td><td>619 (513)</td><td>3206 (1132)</td><td>3206 (1132)</td><td>16.62</td><td>unchanged</td></tr>
<tr><td>C5</td><td>3263 (1898)</td><td>4188 (2526)</td><td>1719 (1637)</td><td>1932 (2183)</td><td>8.54</td><td>6.03</td></tr>
<tr><td>C6</td><td>5251 (3169)</td><td>5251 (3169)</td><td>1265 (1399)</td><td>1265 (1399)</td><td>16.68</td><td>unchanged</td></tr>
<tr><td>C7</td><td>998 (933)</td><td>998 (933)</td><td>tfs</td><td>tfs</td><td>16.9</td><td>unchanged</td></tr>
<tr><td>C8</td><td>1546 (1284)</td><td>1546 (1284)</td><td>5706 (2551)</td><td>5706 (2551)</td><td>18.18</td><td>unchanged</td></tr>
<tr><td>C9</td><td>4158 (2520)</td><td>3359 (2818)</td><td>5859 (3273)</td><td>6987 (4341)</td><td>8.27</td><td>5.44</td></tr>
<tr><td>C10</td><td>3464 (2117)</td><td>3464 (2117)</td><td>4503 (2660)</td><td>4503 (2660)</td><td>18.31</td><td>unchanged</td></tr>
<tr><td>C11</td><td>2361 (1252)</td><td>2361 (1252)</td><td>ns</td><td>ns</td><td>ns</td><td>unchanged</td></tr>
</table>

C12	6388 (2842)	7227 (4325)	ns	ns	ns	4.79
C13	ns	ns	ns	ns	ns	
Total	**107379 (13349)**	**103571 (14915)**	**135471 (19802)**	**125443 (20590)**		
CV	**0.124**	**0.144**	**0.146**	**0.164**		

Notes:

1. 2016 abundance estimates for the original survey design are from Sobtzick *et al.* (2017).
2. Shaded cells indicate differences in population estimates between the original and optimised survey design.
3. Turtle abundance had not been previously estimated for the 2005 survey data and has been estimated for this report
4. tfs-too few sightings for a population estimate
5. ns-not surveyed

(b) Northern Great Barrier Reef region

Block	2013		Sampling intensity (%)	
	original survey design	optimised survey design	2013 survey	optimised survey
N1	1631 (1018)	1709 (1109)	8.8	9.8
N2	3192 (1237)	3192 (1237)	16.9	unchanged
N3	8804 (3068)	8804 (3068)	17.6	unchanged
N4	14545 (3116)	11718 (2964)	9	10.06
N5	37998 (10820)	37998 (10820)	9	unchanged
N6	5056 (2346)	5056 (2346)	8.8	unchanged
N7	1622 (889)	1030 (678)	9.1	10.14
N8	13512 (4194)	13512 (4194)	8.7	unchanged
N9	14464 (4169)	11327 (4148)	8.8	6.76
N10	4617 (1827)	4617 (1827)	9.2	unchanged
N11	3180 (1222)	3180 (1222)	25.3	unchanged
N12	2312 (1989)	2312 (1989)	3.7	unchanged
N13	19575 (5390)	17344 (5084)	6.3	6.97
N14	1039 (567)	1029 (565)	23.1	25.87
N15	1089 (923)*	na*	9.7	4.55
Total	**132636 (14800)**	**122828 (14620)**		
CV	**0.112**	**0.119**		

Notes:

1. All abundance estimates were generated specifically for the comparisons in this table from archived data.
2. Shaded cells indicate differences in population estimates between the original and optimised survey designs.
3. *the northern transects flown in Block N15 in 2013 were zig-zag transects only. Seven transects parallel to the coast were added to Block N15 in the optimised survey design. Thus, an abundance estimate is not yet available (na) for this block for the optimised design.
4. 2006 data not presented for reasons outlined in text

(c) Torres Strait

Block	2006		2011		2013		Sampling intensity (%)	
	original survey design	optimised survey design	original survey design	optimised survey design	original survey design	optimised survey design	2013 survey	optimised survey
TS0	9441 (4961)	9441 (4961)	7286 (6018)	7286 (6018)	4700 (1770)	4700 (1770)	4.4	unchanged
TS1A	11661 (4233)	11661 (4233)	5997 (1848)	5997 (1848)	10575 (2664)	10575 (2664)	8.9	unchanged
TS1B	18289 (5891)	18289 (5891)	13479 (6692)	13479 (6692)	22460 (10094)	22460 (10094)	4.9	unchanged
TS2A	101487 (17799)	101487 (17799)	93032 (25211)	93032 (25211)	172110 (37360)	172110 (37360)	8.2	unchanged
TS2B	19639 (3828)	19639 (3828)	9623 (2436)	9623 (2436)	21848 (4721)	21848 (4721)	8.6	unchanged
TS3A	60764 (10729)	63712 (8091)	58901 (19349)	85489 (20488)	114007 (49101)	146206 (53896)	part of TS3	6.6
TS3B							part of TS3	4.8
TS4	35245 (13424)	37677 (15463)	29758 (19947)	25894 (17718)	53380 (25089)	54809 (24652)	4.4	4.3
TS5	3305 (1983)	4741 (3218)	1857 (892)	2931 (1614)	7306 (4085)	6829 (4679)	11.2	5.9
TS6	ns	Ns	tfs	tfs	tfs	tfs	3.9	deleted
TS7	ns	Ns	3358 (2111)	na*	7330 (4076)	na*	3.6	deleted
TS8	ns	Ns	15056 (9632)	na*	9075 (5281)	na*	4.9	deleted
TS9	ns	Ns	7241 (2882)	8923 (3640)	10217 (5497)	10534 (4984)	4.5	4.4
Total	**250390 (26606)**	**266647 (26900)**	**245588 (40057)**	**252654 (38412)**	**433008 (68278)**	**450071 (71340)**		

CV	**0.106**	**0.101**	**0.163**	**0.152**	**0.158**	**0.159**

Notes:

1. Abundance estimates for the original design are from Hagihara *et al.* (2016).
2. Shaded cells indicate differences in population estimates between the original and optimised survey designs
3. tfs-too few sightings for a population estimate
4. ns-not surveyed
5. **eastern side of transect in Block TS8 was incorporated into Block TS3B. Thus some turtles sighted in Block TS8 were used in the abundance estimation for Block TS3B.
6. In 2006, blocks TS 6-8 were not surveyed. Thus the block size was the original block size and no turtle sightings were imputed for blocks TS3A and TS3B.
7. The survey intensity of Block TS3A was almost doubled in the optimised design and included some of the former blocks TS7 and TS8. To take this increase into account, the relevant turtle sightings were duplicated to accommodate the change in intensity in blocks 3A and 3B.

11.0 Appendix 4 — Potential use of unmanned aerial vehicles for megafauna monitoring in the Great Barrier Reef: Transitioning to the new technology

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11.1 Applicability of unmanned aerial vehicles to marine megafauna monitoring

It is widely recognised that unmanned aerial vehicles (or drones) have the potential to greatly enhance our wildlife research and monitoring capacity. This potential is highly valued for marine mammals. Aerial platforms have long been used to observe or monitor many species of marine mammals because compared to boat-based observation techniques, an aerial perspective offers greater visual penetration through the water column to animals below the surface and the opportunity to observe animals over larger spatial scales. Unmanned aerial vehicles have many advantages over manned aircraft including human safety, superior detection and vastly improved capacity to archive georeferenced images.

Although there have been a number of reviews forecasting the potential applications of unmanned aerial vehicles (Watts et al. 2010, Linchant et al. 2015, Christie et al. 2016, Gonzalez et al. 2016, Fiori et al. 2017, Colefax et al. 2018), there have been very few empirical studies demonstrating these applications, particularly of species that are strictly marine. The applications that have been tested include surveys for monitoring distribution and abundance of dugongs (Hodgson et al. 2013) and whales (Hodgson et al. 2017), determining densities of sharks and rays (Kiszka et al. 2016) and sea turtles (Sykora-Bodie et al. 2017) in nearshore waters, understanding the certainty of detections of humpback whales, killer whales and harbour porpoises (Aniceto et al. 2018), photo-identification mark-recapture studies of bowhead whales (Koski et al. 2015), photogrammetry of various whale species to determine body condition (Christiansen et al. 2016, Durban et al. 2016), behavioural observations of humpback whales (Hodgson et al. 2017) and sea turtles (Bevan et al. 2016). Other studies have shown the use of unmanned aerial vehicles for pinnipeds that are hauled out such as aerial surveys of seals on ice (Moreland et al. 2010) or on land (Johnston et al. 2017) and supplementing aerial surveys of sea lions (Sweeney et al. 2015).

Most of the applications listed above have involved using relatively small unmanned aerial vehicles, flown within line-of-sight and for relatively short durations. Some recent reviews of the potential to use unmanned aerial vehicles for large-scale surveys suggest they are currently only applicable to small-scale surveys as the unmanned aerial vehicles capable of the endurance and range necessary for large-scale surveys are prohibitively expensive, and it is difficult to obtain permits to operate unmanned aerial vehicle systems beyond line-of-sight and at altitudes necessary for such surveys (Christie et al. 2016, Fiori et al. 2017, Colefax et al. 2018). However, Hodgson et al. (2017, in prep) have successfully demonstrated that fixed-wing unmanned aerial vehicles could be used for humpback whale and dugong surveys in Australia. All trial surveys reported by Hodgson et al. were conducted using the *ScanEagle* unmanned aerial vehicle, which was 'wet leased', meaning that a third party company owned and operated the *ScanEagle*. This unmanned aerial vehicle has a range in excess of 100 kilometres from the base station, can fly at altitudes in excess of 5,000 metres, and has an endurance of up to 24 hours. Hodgson et al. conducted trial surveys beyond line-of-sight and the results showed that unmanned aerial vehicle surveys provide comparable, if not superior, data to manned surveys.

Despite these demonstrations, implementing large-scale unmanned aerial vehicle surveys to replace existing long-term monitoring programs in the Reef region, and to implement additional monitoring, including conducting multi-species surveys, requires consideration of the following:

- Integrating historical work — how can we adapt this new methodology whilst ensuring previous surveys are comparable?
    - Can we obtain comparable coverage, whilst capturing the required ground resolution, in a cost-effective time frame?
    - How does detection probability compare between manned and unmanned surveys?
- Are multi-species surveys realistic?
- How do we choose appropriate unmanned aerial vehicles and imaging systems
    - Can alternative camera systems (thermal / hyperspectral) increase detection probability, and how do we integrate this improved detection into long-term datasets?
- Can the image processing be automated to obtain:
    - Sighting data (of multiple species?)
    - Environmental conditions
    - Sampled area (accounting for unmanned aerial vehicle rotations)
    - Location of sightings (accounting for unmanned aerial vehicle rotations)
- What regulations need to be considered?
    - Who will fly the unmanned aerial vehicles?
    - Can we realistically obtain permission to fly beyond line-of-sight and above 400 feet?
    - Do unmanned aerial vehicles cause disturbance to wildlife?

The following review outlines our current understanding about the above considerations. The feasibility of using unmanned aerial vehicles to monitor dugongs, humpback whales, turtles and dolphins, is discussed with the view of replicating the methods currently used for traditional manned surveys, as this would allow us to transition to using unmanned aerial vehicles and maintain consistency and comparability with long-term datasets. However, the applications of unmanned aerial vehicles offer new opportunities to collect and analyse data that are not possible using manned surveys. Therefore, the ultimate aim should not be to simply replicate manned surveys, but continue to improve the data we obtain from marine megafauna monitoring by investigating new ways to exploit unmanned aerial vehicle technology as a vital component of the RIMReP.

11.2 Integrating historical methods and data

Converting from traditional manned surveys to unmanned surveys and the subsequent change to collecting images/video rather than human observations provides the opportunity to collect new types of information. Nonetheless, this change also requires a re-think of the traditional methodology used in aerial surveys (Linchant et al. 2015, Hodgson et al. 2017). The first step is to determine the probability of detecting animals using the unmanned aerial vehicle. Quantifying detection probability will allow for comparisons with traditional manned surveys, which is the key consideration in the transition from previous long-term monitoring programs for marine megafauna in the Reef, to this new technique. The work to date by Hodgson et al. to determine the efficacy of using unmanned aerial vehicles for large-scale marine megafauna surveys, has focused on assessing what factors affect detection rates of dugongs and humpback whales in unmanned aerial vehicle images (Hodgson et al. 2013, Hodgson et al. 2017) and comparing detection rates from manned surveys to unmanned aerial vehicle surveys (Hodgson et al. 2017, Hodgson et al. in prep). Aniceto et al. (2018) also investigated the effects of environmental conditions and unmanned aerial vehicle variables on the certainty of sighting humpback whales, killer whales and harbor porpoises. The following review outlines the components of detection probability for both manned and unmanned surveys, and then for each species of interest, summarises our knowledge to date about how the data from unmanned aerial vehicle surveys compare to manned surveys, and the logistical feasibility of transitioning between manned and unmanned surveys for each species of interest.

11.2.1 Detection probability

In manned aerial surveys, it is commonly understood that the probability of detecting an animal is affected by two factors: (1) availability probability – the proportion of time the animal is actually visible from the air, and (2) perception probability – the probability of an observer actually seeing the animal if it is available (Marsh and Sinclair 1989) (Figure 1). Availability has two components: the animal's diving behaviour, and the environmental conditions such as water turbidity (Pollock et al. 2006). Diving behaviour can be affected by numerous factors such as water depth (Hagihara et al. 2018), group composition (for example, whether a calf is present) (Hodgson et al. 2017), and behaviour (Dorsey et al. 1989). Various methods have been used and experiments conducted to estimate the availability of the species of interest, particularly for dugongs (Pollock et al. 2006, Hagihara et al. 2014, Hagihara et al. 2018). In order to convert to using unmanned aerial vehicles, it is important to understand whether environmental conditions affect the availability of animals differently in the images compared to real-time visual observations from a manned aircraft.

Unmanned aerial vehicles offer an alternative method for assessing availability as this technology allows us to follow and observe the behaviour of marine fauna and directly observe the proportion of time individual animals are available to be seen from the air. This idea was demonstrated by Hodgson et al. (2017) using the *ScanEagle* to follow humpback whales, and a team from Murdoch University is currently using unmanned aerial vehicles to assess the availability of Australian humpback dolphins and bottlenose dolphins. This technique may not be applicable to all species but does have a number of advantages of previously used methods (Hodgson et al. 2017).

When we measure perception probability in manned surveys, we generally compare the detections from two or more observers, and assume that availability is equal for both observers (i.e. that their position in the aircraft does not affect their ability to see the animal). Therefore, perception probability is only dependent on whether the observer happened to see the animal, and is measured empirically from a survey using a mark-recapture approach for observers sitting one behind the other in the aircraft (Pollock et al. 2006). For sightings in unmanned aerial vehicle images, perception probability depends on how the images are processed. If they are manually reviewed, perception probability can be measured by comparing the sightings from two or more reviewers who have processed the same images (Hodgson et al. in prep). If images are reviewed automatically by computer algorithms, then perception probability is the recall rate (proportion of dugongs the algorithm detects), and this can be affected by the image complexity and the number of example images of animals that are accessible to train the algorithm (Maire et al. 2015). The status of computer detection algorithms for the species of interest and the issues surrounding image processing are discussed in 11.10 Image processing).

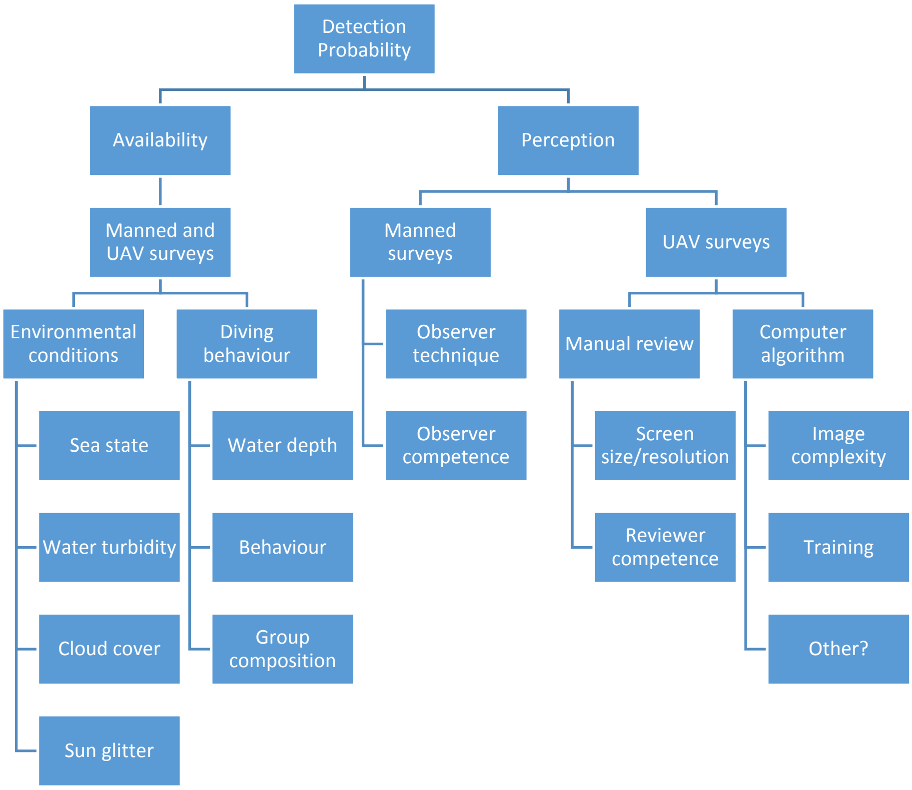

Figure 1. Factors that could potentially affect detection probability in marine mammal aerial surveys.

11.3 Dugongs

Dugong surveys are potentially the easiest to replicate using unmanned aerial vehicles because they have traditionally been conducted using strip transects (Marsh and Saalfeld 1989, Marsh and Sinclair 1989), rather than the line transects used for many other species. Strip transects lend themselves well to the unmanned aerial vehicle design. Hodgson et al. (2013) conducted a series of small-scale trial surveys over an area known to have a high density of dugongs. They used a nadir (downward) facing Single Lens Reflex camera to continually capture high resolution, overlapping images along a set of parallel transect lines at a variety of altitudes, and within a variety of weather conditions. The images were then manually reviewed post-flight to identify dugongs and other animals of interest. These initial trial surveys provided the following key results:

- Dugongs (including calves) could be distinguished within images at a ground resolution of three centimetre per pixel
- Sea state did not affect dugong sighting rates during these trial surveys
- Sun glitter did not affect dugong sighting rates during these trail surveys
- Water turbidity affected the dugong sighting rates, although the observed effect was counterintuitive during these trial surveys (i.e. when comparing shallow clear water to deep unclear water the sighting rate was higher in the latter)
- Overlap between successive images along the transect line was important for countering the effects of sun glitter and for providing multiple opportunities to confirm dugong sightings.

These trial surveys had some limitations, including:

1. The Civil Aviation Safety Authority permit received at this time (these trials were conducted in 2010) meant the unmanned aerial vehicle could only be flown within line-of-sight, therefore limiting the size of the survey area and the distance from shore.
2. The field of view, and therefore effective strip width, from one nadir facing camera was much narrower than the strip width used during traditional manned surveys, meaning that more transects would need to be flown to achieve the same sample rate as manned surveys.

These two limitations were overcome in subsequent trial surveys, where Hodgson et al. (in prep) directly compared dugong sighting rates from manned survey observers to sighting rates in images collected using the *ScanEagle*. Civil Aviation Safety Authority permitted Insitu Pacific to fly beyond line-of-sight and, with the ground control station located at the local airport, the trial surveys covered over 570 square kilometres. Two Single-Lens Relfex cameras with higher resolution than those used in the initial trial surveys, were mounted on the *ScanEagle* at slightly oblique angles, so that combined, they were able to provide a strip width that matched the manned surveys, which is approximately 400 metres (i.e. the combined width of the strips observed on each side of the manned aircraft).

In the direct manned versus unmanned trial, the two platforms were flown over the same transects at the same time. Both platforms covered the same transect strip width, and those survey strips were partially (but not entirely) overlapping. This procedure provided a comparison of dugong sighting rates (i.e. the aim was not to match dugong sightings) in the same areas and under the same environmental conditions (including sea state, turbidity, and cloud cover). Overall, the dugong sighting rate was higher from the unmanned aerial vehicle images than for the observers conducting the manned survey. Analyses conducted to date suggest that the sighting rates from both platforms were affected by the environmental conditions in the same way, and again there was no effect of sea state (however the data for Beaufort sea states greater than four were limited to three out of the eight survey flights conducted). The main difference between the sighting rates from the two platforms was in estimating the sizes of groups of dugongs. There was a higher number of large dugong groups (i.e. group size greater than 10) detected in the unmanned aerial vehicle images than by the manned observers. This effect is still being investigated, but potentially group size is more accurately assessed within the images (where there is sufficient time to see and count all the animals in a group) than by observers in aircraft who have a few seconds to determine how many dugongs are present. This effect seems particularly prevalent in shallow murky water where it can be difficult to see dugongs on the sea floor. Further analysis is being conducted to better understand these results and determine the ultimate effect on population estimates.

11.3.1 Can unmanned aerial vehicles currently be used to survey dugongs?

Logistics

The *ScanEagle* unmanned aerial vehicle used in the trial surveys has sufficient endurance and range to cover the scale of the areas recommended for monitoring dugongs within the Marine Park in the this report. The coverage would be achieved by using ‘hub and spoke’ operations whereby repeaters are able to extend the range of the *ScanEagle* by handing off to a nearby communications link. Most (but not all) transects flown in manned dugong surveys in the Reef are less than 100 kilometres long. The challenge will be to optimise the placement of the communications links, especially along the remote coasts of Cape York and the Shoalwater Bay region where land access is limited. One potential logistical limitation in using the *ScanEagle* is that this system flies at half the ground speed of a manned plane, and therefore a survey could take twice as long. However, this unmanned aerial vehicle has enough endurance to fly continually for a whole survey day, as opposed to a manned aircraft where the maximum flight time is three hours before refuelling is necessary.

The two-camera imaging system tested in the most recent trial survey allows for the same survey design and sampling rate as manned dugong surveys (Hodgson et al. in prep). Customised dugong detection and mapping software has been developed (with ongoing improvements) meaning that it is realistic to survey large areas and process images in a cost-effective and expedient timeframe, although some manual review of images is still necessary (see 11.10 Image processing).

Results to date suggest that sighting rates in unmanned aerial vehicle images are not affected by sea state (see Data compatibility below) and therefore unmanned aerial vehicle surveys could potentially be flown in a wider range of wind conditions than manned surveys. This result needs to be tested further as it makes the untested assumption that dugong diving behaviour is unaffected by sea state. The manned aerial surveys overcome this assumption by limiting the sea states in which the surveys are conducted. The unmanned aerial vehicle advantage of flight endurance coupled with the ability to survey in a wider range of conditions than manned surveys may counteract the effect of slow flight speed of unmanned aerial vehicles.

Data compatibility

Trial surveys conducted so far suggest that unmanned aerial vehicles provide comparable dugong sighting rates to manned aerial surveys and there was no difference in the way sighting rates are affected by environmental conditions when comparing the two platforms. From this result, we can assume that availability of dugongs is comparable between the two platforms, and that it is appropriate to apply the availability corrections currently used for manned surveys, to unmanned aerial vehicles surveys if surveying under the same limited wind conditions (that is, Beaufort sea states of less than or equal to 3). However, there are three outstanding questions to resolve before unmanned aerial vehicle surveys can replace manned dugong surveys and expand the range of wind conditions in which these surveys can be conducted. These are presented below in order of likely importance to future Reef dugong surveys.

1. Detection in highly turbid waters and high sea states

    The trial surveys conducted to date have been in Shark Bay, Western Australia, where the water is relatively clear — there is very little of the turbid water characteristic of most dugong habitats within the Reef region. To ensure that future dugong unmanned aerial vehicles surveys are directly comparable with historical manned surveys, similar trial surveys need to be conducted in more turbid dugong habitat than was encountered in Shark Bay. Ideally, such surveys would also incorporate further testing of dugong sighting rates in high sea states. Although previous trial surveys have suggested sea state does not affect sighting rates, there has been relatively little data collected in the higher wind conditions (for example, only three of eight flights were conducted in Beaufort sea state four or five during the most recent trial surveys (Hodgson et al. in prep)). Aniceto et al. (2018) suggested that high sea states did affect the certainty of sightings of humpback and killer whales during their trial surveys. So collecting more data at Beaufort sea states four and five would confirm the effect of sea state on dugong sighting rates. The *ScanEagle* is capable of flying in these high wind speeds.

2. Detection of large groups

The trials indicate that unmanned aerial vehicles are better at detecting large dugong groups than human observers in transect mode. This result is currently being investigated and the outcome of these analyses will determine whether further work is needed to resolve this issue. However, large groups of dugongs (more than 10 individuals) are relatively rare in the Reef and have been stratified out of analyses to correct dugong observations for detection biases in manned surveys and added on at the end as uncorrected counts. Considering the rarity of large groups in Reef survey areas, and that the detection of these groups from manned aircraft is possibly only biased downwards in particular environments (i.e. in shallow murky waters), the increase in detection of large groups by unmanned aerial vehicles will likely not affect Reef dugong population estimates very much. Nonetheless, our understanding of dugong habitat use may be affected by this bias, a defect that may be significant in spatial risk assessments.

3. Availability corrections

   Experiments need to be conducted to ensure that the corrections for availability bias that have been develop for observers in manned aircraft for dugong surveys in the Reef are applicable to unmanned aerial vehicle surveys to maximise the integrity of the historical time series, which is based on standardised indices of relative abundance. The need to do this work would depend on the results from 1) above. The simplest method of conducting this check would be to repeat the dugong model experiments (Hagihara et al. 2016) using a small multi-rotor unmanned aerial vehicle with a camera similar to that in the *ScanEagle*. The unmanned aerial vehicle could be operated from a boat, similar to methods currently being conducted at Murdoch University by Chris Cleguer. If absolute indices of dugong abundance were required, more elaborate experiments using unmanned aerial vehicles to conduct focal follows of dugongs fitted with satellite transmitters should be conducted as discussed above.

11.4 Humpback whales

To date there are only two published evaluations of using unmanned aerial vehicles to survey humpback whales. Hodgson et al. (2017) conducted a series of trial large-scale surveys, with permission obtained to fly beyond line-of-sight at and an altitude of 2,400 feet. The whales were observed off Stradbroke Island, Queensland, where their migration route passes close to the coastline. The survey area was monitored by land-based observers while the unmanned aerial vehicle surveys were conducted and it was assumed that the land-based observations provided a census of all whales passing, against which the unmanned aerial vehicle survey data could be compared. This comparison provided an understanding of the detection rates from the unmanned aerial vehicle. In addition, the availability component of detection of the whales was assessed by directly observing whale pods using the *ScanEagle* in loitering mode at an altitude high enough (2,400 feet) to give a wide field of view when the target pod had dived out of view. The key results from this assessment were:

- The *ScanEagle* surveys provided detection rates of whales that were within the range reported for previous manned surveys.
- Detection rates were not affected by environmental conditions.
- Focal follows of whale pods using the *ScanEagle* provided detailed availability estimates that represented the spatial and temporal characteristics of the survey area and target population,

from the same perspective as the survey platform. This benign technique also provides additional data on animal behaviour. High whale densities made it difficult to maintain follows of one group, and the ability to plot the tracks of the whales at the ground control station in real time was important for addressing this problem.

- Availability estimates for the pods observed varied immensely, and were much higher for pods with calves that for those without. It is therefore important that the focal follows used to estimate overall availability are representative of the population being surveyed with no bias in sampling.
- The focal follows of whale pods conducted to assess availability showed that group sizes recorded during the unmanned aerial vehicle surveys were underestimates. The data from the follows allowed for group size estimates to be corrected.
- The slower speed of the unmanned aerial vehicles means that movement bias corrections (i.e. correcting for whales missed because of their systematic movement through the survey area) are important for these surveys (although it was also demonstrated that movement bias should also be considered for manned surveys).

Aniceto et al. (2018) conducted some small scale (within line-of-sight, under 400 feet) trial unmanned aerial vehicle surveys of three species of cetaceans — humpback whales, killer whales and harbour porpoises — in two fjords in northern Norway. They found that ‘certainty of detection’ (which was used as a proxy for detectability) was negatively affected by increasing sea state and positively affected by increasing luminosity within the images.

11.4.1 Can unmanned aerial vehicles currently be used to survey humpback whales?

Logistics

According the previous aerial surveys for humpback whales within the Marine Park (Smith and Hedley 2013, Peel et al. 2015) the *ScanEagle* unmanned aerial vehicle used in the trial surveys (Hodgson et al. 2017) has sufficient endurance and range to cover the scale of the areas that might be considered for future monitoring. The coverage would be achieved by using ‘hub and spoke’ operations whereby repeaters are able to extend the range of the *ScanEagle* by handing off to a nearby communications link. This approach should be relatively easier for humpback whales than for dugongs because most sightings are from Cairns south and there is road access along most of the coast.

To date all assessments of unmanned aerial vehicles for conducting aerial surveys have assumed a downwards (nadir) facing camera, and therefore would employ a strip transect sampling design rather than distance sampling. The latter is the commonly used sampling method for aerial surveys of humpback whales and provides a much wider effective strip width than 490 metres achieved by Hodgson et al. (2017). Therefore to achieve the same sample rate as a traditional manned survey, the *ScanEagle* would need to fly more transects. The *ScanEagle* also flies at approximately half the speed of the manned aircraft, further increasing the time needed to cover a comparable sample area. Hodgson et al. (in prep) used a two-camera payload and the cameras had higher resolution than that used for the humpback whale trial surveys (Hodgson et al. 2017). If using this two-camera set-up, it would be possible to achieve the resolution shown to provide certainty of humpback whale detections (11.5 centimetres per pixel) by flying at approximately 5,000 feet. The combination of this altitude and the two-camera set-up would provide a strip width of 1,400 metres with 80 metres of overlap between the two cameras. The disadvantage of flying at such high altitude is the potential to be flying above cloud. Therefore, there is a trade-off among all of these factors when planning an unmanned aerial vehicle survey and the appropriate flight protocol would depend on the research question.

At this point in time, image processing to detect whales in the images would need to be conducted manually because the automated detector has only been developed for dugongs (see 11.10 Image processing). The customised mapping software can be applied to all unmanned aerial vehicle surveys.

Data compatibility

Overall, detection probability of humpback whales from the *ScanEagle* is comparable to that reported for manned surveys (Hodgson et al. 2017). One of the main limitations of humpback whale population estimates from manned aerial surveys has been the difficultly in estimating availability. Methods used to date have resulted in extremely varied availability estimates, even between two studies of the same population, at the same location and time of year (Hodgson et al. 2017). Using unmanned aerial vehicles to conduct aerial surveys and then to conduct focal observations of whales pods to understand availability of the target whale population in situ of the survey would provide a more accurate estimate of availability compared to previous methods, and as a result, would lead to more accurate population estimates.

Although Hodgson et al. (2017) found no effects of environmental correlates on the detection rates of humpback whales, there may not have been a large enough sample size from the various combinations of conditions to adequately quantify their effects on pod detections in the images, especially for high sea states. Although this potential limitation should not prevent unmanned aerial vehicles from being used for humpback whale surveys, we should continue to investigate the effects of environmental conditions of detection rates.

11.5 Turtles

Large juvenile and adult turtles are recorded during manned surveys for dugongs in the Reef as outlined in Sobtzick et al. (2017). In the only published example of using unmanned aerial vehicles to survey turtles, Sykora-Bodie et al. (2017) conducted a relatively small-scale survey of olive ridley turtles at Ostional, Costa Rica. The turtles were aggregating near their nesting beach and the survey design was a single transect perpendicular to the beach where nesting was most concentrated. This three kilometre transect was flown six times, using the eBee (senseFly SA) – a lightweight foam, modular, fixed-wing airframe powered by a single electric motor. The ground resolution achieved by the eBee's near infrared camera, with a flight altitude of 90 metres, was 2.5 centimetres per pixel. This resolution allowed for turtles to be detected with certainty 63 per cent of the time, with the remainder being classified as 'probable' turtles. The researchers could not identify the turtle species with the ground resolution and clarity of images acquired from this particular unmanned aerial vehicle.

Hodgson has unpublished data from *ScanEagle* trial surveys described in Hodgson et al. (2013) where turtles were detected in images collected at three altitudes — 500, 750 and 1,000 feet. The ground resolution in these images was 1.7, 2.5 and 3.4 centimetres per pixel respectively. A total of 103 turtles were detected 'with certainty' by one reviewer and confirmed by a turtle expert. The latter reviewer then classified the turtle detections into species — green turtle, loggerhead turtle and unknown. A much higher proportion of turtle detections could be classified to species 'with certainty' in the higher resolution images than in the lower resolution images. These data have not yet been analysed to determine if any environmental or flight characterises affected whether turtles could be identified to species. Images from subsequent *ScanEagle* trial surveys (Hodgson et al. in prep) could be used to augment this dataset. Most of these were captured at a ground resolution greater than three centimetres per pixel, however, a subset of flights were conducted at a lower altitude where the ground resolution was 2.3 centimetres per pixel. These images could be reviewed to determine whether the particular imaging system and resolution achieved during this survey was sufficient to identify a large proportion of turtles to species.

11.5.1 Can unmanned aerial vehicles currently be used to survey turtles?

Logistics

The logistical feasibility of using unmanned aerial vehicles to survey turtles would depend on the research question, which in turn would determine the size of the animals of interest, and the survey area. Rees et al. (2018) outline various applications of unmanned aerial vehicles to turtle surveys, including beach and coastline surveys, along with in-water surveys. There are many advantages to using unmanned aerial vehicles for surveying nesting beaches, for example, they could provide data in areas that are difficult or dangerous to access on foot, and they limit disturbance to nesting turtles. These surveys could be conducted using small multi-rotor unmanned aerial vehicles, if there were multiple access points along the beaches. Otherwise, larger unmanned aerial vehicles with permission to fly beyond line-of-sight would be required to survey large expanses of coastline. Daytime surveys of turtle nesting activity (tracks and nests) can be conducted using standard RGB cameras, and nocturnal surveys of nesting turtles can be conducted using unmanned aerial vehicles equipped with low-light optical or thermal cameras (Rees et al. 2018) (see 11.9 Alternative imaging systems for more about camera systems).

The logistical challenges of conducting in-water surveys for large immature and adult turtles are similar to those for dugongs. At this time, image processing to detect turtles in the images would need to be conducted manually because the automated detector has only been developed for dugongs (see 11.10 Image processing). The customised mapping software can be applied to all unmanned aerial vehicle surveys.

Data compatibility

Surveying nesting beaches using unmanned aerial vehicles could provide equivalent data to on-foot surveys if the aim was to obtain turtle numbers, track and nest counts. unmanned aerial vehicle surveys could provide more complete beach coverage than on-foot surveys, and provide a permanent archive of actual and potential nesting habitats (Rees et al. 2018).

If in-water surveys are to provide population estimates for turtles, we need an understanding of availability. The dive behaviour of sea turtles has been assessed to produce availability estimates for loggerhead (Thomson et al. 2012) and green turtles (Thomson et al. 2012, Fuentes et al. 2015). Fuentes et al. (2015) and Hagihara et al. (2016) also incorporated the effects of environmental conditions into their estimates. To understand whether these estimates are applicable to unmanned aerial vehicle surveys, it would be necessary to conduct a direct comparison between sighting rates from a drone versus human observers. These data could be extracted from the image set collected during unmanned aerial vehicle trial surveys for dugongs (Hodgson et al. in prep) and a direct comparison of turtle sighting rates from the unmanned aerial vehicle images versus observers on the manned aircraft could be conducted as has been done for dugongs.

The diving behaviour of both loggerhead and green turtles is affected by season (water temperature) and depth (Thomson et al. 2012, Thomson et al. 2013), and Thomson et al. (2012) suggests that "site-specific knowledge of dive–surfacing patterns can be important to mitigate the effects of availability bias during population surveys". It would be possible to gather *in situ* diving behaviour data by conducting focal behavioural observations on individual turtles using drones (similar to Hodgson et al. 2017), but only where turtle densities are relatively low so that individuals could be reliably tracked even when diving out of sight.

In previous manned (Fuentes et al. 2015) and unmanned (Sykora-Bodie et al. 2017) surveys of turtles, they could not be identified to species. To overcome this limitation (Fuentes et al. 2015) suggest:

> *"…conducting separate helicopter surveys in circling mode in the same season as the surveys to identify the species and sex of a large sample of turtles to enable the sex ratio and species composition of sightings in subsequent aerial surveys conducted in passing mode to be estimated…"*

This experiment was subsequently conducted in Torres Strait by turtle expert Mark Hamann (Hagihara et al. 2016). This experiment could also be conducted with a small multi-rotor unmanned aerial vehicle operated from a boat, similar to methods currently being conducted at Murdoch University by Chris Cleguer for dugongs. Alternatively, or in addition, further investigation into Hodgson's unpublished data (as suggested above) could provide a better understanding of the ground resolution needed to identify turtles to species in unmanned aerial vehicle images. Identified to species empirically during a survey would provide the most reliable population estimates and distribution data for all turtle species.

11.6 Dolphins

The only published account of dolphin surveys using unmanned aerial vehicles is Aniceto et al. (2018) who assessed the certainty of detections of harbour porpoises according to environmental variables and image resolution. Small-scale (within line-of-sight) surveys were conducted in an area where harbour porpoises were assumed to be the only species present and the 'certainty' of sightings concerned whether the image reviewers were sure the sightings were actually dolphins.

The ground resolution for these surveys was approximately 2.9 centimetres per pixel (estimated according to the image ground width of view and resolution provided). Of 57 dolphin sightings, only five were considered certain. None of the variables measured — sea state, luminance and pixel size (which changed slightly as a result of pitch, roll and altitude of the unmanned aerial vehicle) — affected the rate of certainty of detections. However, the authors acknowledge that this null result may have been symptomatic of the low rate of certain detections. The authors did not comment on the low rate of detection certainty and what the possible causes might have been.

Hodgson has unpublished data from *ScanEagle* trial surveys (Hodgson et al. 2013) where dolphins were detected in images collected at three altitudes — 500, 750 and 1,000 feet. The ground resolution in these images was 1.7, 2.5 and 3.4 centimetres per pixel respectively. A total of 16 images captured while 'on transect' contained 'certain' dolphin sightings, totalling 26 individual dolphins. If including dolphin sightings captured while 'off-transect', these numbers increase to 28 images and 42 individuals. These images have not yet been systematically reviewed by a dolphin expert, however, Hodgson was not able to classify any sightings to species with certainty, and dolphin experts who have viewed a subset of these images agreed. Most sightings at the study site (Shark Bay, Western Australia) were likely one of two species – bottlenose or humpback dolphins, which look quite similar from the air[11]. It would be worthwhile having a dolphin specialist review these images to better understand the limitations of dolphin species identification on unmanned aerial vehicle images. This dataset could be augmented with images from subsequent *ScanEagle* trial surveys at the same location (Hodgson et al. in prep), which mostly have a ground resolution greater than three centimetres per pixel, although a subset of flights were conducted at a lower altitude where the ground resolution was 1.7 and 2.5 centimetres per pixel. The latter trial surveys produced dolphin sightings in 153 images, totalling 398 individual dolphins in images with the lower resolution. The higher resolution images have not yet been manually reviewed. Dolphin sightings reported here from Hodgson's unpublished unmanned aerial vehicle data include all possible dolphins (including those reported as 'uncertain' by the image reviewers) and double counts of dolphins seen in two images.

11.6.1 Can unmanned aerial vehicles currently be used to survey dolphins?

Logistics

The logistics of conducting large-scale aerial surveys for dolphins within the Reef have some similarities to those for dugongs. However, at this time, image processing to detect dolphins in the images would need to be conducted manually because the automated detector has only been developed for dugongs (see 11.10 Image processing). The customised mapping software can be applied to all unmanned aerial vehicle surveys.

Large-scale aerial surveys for the dolphin species of interest (humpback and snub-finned dolphins) are limited mainly by the low density and restricted habitat use of these dolphins, which mainly occur close to shore in localised habitats close to river mouths and within the 14 metres depth contour (Parra et al. 2006). Large-scale manned surveys for dugongs are typically conducted with a sampling rate in such habitats that is too low to provide sufficient dolphin sightings for reliable or useful abundance estimates and distribution or to monitor changes in dolphin populations (see Coastal Dolphins RIMReP report by Brookes et al.). Cleguer and Hodgson at Murdoch University are investigating methods for using unmanned aerial vehicles to conduct small intense, local-scale surveys for dugongs, which can be conducted from a boat using relatively cheap and easily operated

[11] Snubfin dolphins are likely to be more distinctive as a result of their rounded head

unmanned aerial vehicles. Various survey design methods for conducting these intense surveys and estimating abundance are being investigated. Similar local-scale, boat-based survey designs for dolphin hot spots could be investigated.

There is also potential for small, multi-rotor unmanned aerial vehicles to augment boat-based line-transect surveys. This application is being investigated by NOAA's Southwest Fisheries Science Centre (Marine Mammal Commission 2016). The unmanned aerial vehicles, if operated at relatively low altitude, could be used to assist in identifying dolphin sightings to species and to estimate group size. This assistance could speed up boat-based surveys as the boat would not need to go off transect to collect this information. It could also reduce movement bias as a result of dolphins avoiding or being attracted to boats approaching them.

Data compatibility

To understand whether sighting rates of dolphins from unmanned aerial vehicles is comparable to aerial or vessel surveys, it would be necessary to conduct direct comparisons. The aerial comparison could be achieved using the image set collected during unmanned aerial vehicle trial surveys for dugongs (Hodgson et al. in prep), as dolphin sightings were recorded by the human observers, and during the manual review of the unmanned aerial vehicle images.

It is worth noting that during the Hodgson et al. (in prep) trial surveys, where the target species was dugongs, the human observers in the manned aircraft were also asked to identify the dolphin sightings to species level where possible. They classed their species ID as 'certain', 'probable' or 'guess'. During this trial survey, 45 per cent of dolphin sightings include a 'certain' species ID and 33 per cent were 'probable'. The remaining were 'guess' or unknown species.

In order for unmanned aerial vehicle surveys to replicate these large-scale manned surveys, the ability to identify dolphins to species needs to be further investigated, and a minimum ground resolution assessed. In addition, methods for identifying species from aerial images need to be established.

Hodgson is currently a co-investigator on a research project aimed at establishing estimates of humpback and bottlenose dolphin availability in the Pilbara (and the Perth region for the latter species) in Western Australia, using small multi-rotor unmanned aerial vehicles (Phantoms 4 Pros). Data are currently being analysed for the former species, while field work is about to begin for the latter species. This research has been successful in collecting similar data as obtained by Hodgson et al. (2017) using a much cheaper and more accessible unmanned aerial vehicle. The method could be used to collect in situ dive behaviour observations to assess the availability of any target dolphin species for the purpose of obtaining population estimates from aerial surveys.

11.7 Multispecies surveys

One advantage of conducting aerial surveys with unmanned aerial vehicles, is that sighting data can be collected for multiple species, without the biases introduced when human observers are asked to target particular species, and the likelihood of missed sightings when human observers are asked to call all animals sighted. During the trial unmanned aerial vehicle surveys reported in Hodgson et al. (2013), unpublished sighting data was also recorded for dolphins, turtles, sharks, rays and sea snakes. (All of these species are also recorded during manned surveys for dugongs in the Reef, although much of that data has never been processed, H. Marsh pers comm.). Seabirds could also be clearly seen sitting at the surface of the water in the unmanned aerial vehicle images, although these sightings have not yet been recorded from the images. In the trial unmanned aerial vehicle surveys reported in (Hodgson et al. in prep), sighting data were recorded for dolphins and whales, and is currently being recorded for rays and turtles.

Successful multi-species surveys using unmanned aerial vehicles would require a list of target taxa, and knowledge of the ground resolution required to identify all sightings of these taxa to species. The current knowledge of the resolutions needed to identify 11.3 Dugongs, 11.4Humpback whales, 11.5 Turtles and 11.6 Dolphins are discussed in the relevant sections above. Preliminary investigations have suggested that experts can identify mobula rays to species in the (Hodgson et al. in prep) images, where ground resolution was 3.2 or 3.5 centimetres per pixel (depending on altitude), and these data are currently being reviewed. Kiszka et al. (2016) were able to identify elasmobranchs to species with a ground resolution of approximately 1.8 centimetres per pixel (estimated according to the image ground width of view and resolution provided) in clear shallow water. As many elasmobranch species, particularly rays, spend large amounts of time on the sea floor, turbidity, combined with ground resolution, would dictate the usefulness of multispecies surveys for these taxa. The potential for unmanned aerial vehicles to monitor coral beaching should also be investigated.

The main challenge of multispecies surveys using unmanned aerial vehicles would be to achieve the appropriate ground resolution for all taxa of interest (i.e. including smaller species that require high resolution for species identification), whilst achieving coverage (i.e. effective strip width) that is cost- and time-effective over a large spatial scale.

11.8 Selecting unmanned aerial vehicles

As with any other research tool or vessel, it is essential to select an appropriate unmanned aerial vehicle for the intended research question. There are a large number of unmanned aerial vehicles available, and there are many variables to consider when selecting an appropriate system. The vast majority of unmanned aerial vehicles on the market are from start-up companies and their systems have not been field tested to the extent needed for them to be reliable for marine research. Hodgson has worked with a number of different unmanned aerial vehicle companies and systems, and with the exception of the Phantoms, all unmanned aerial vehicle systems used have been crashed at least once. Therefore it is critical to ensure that any unmanned aerial vehicle system selected has been rigorously field tested, that redundancies are factored into costings, and that if a start-up system is chosen, the company is willing to take responsibility for, and replace, any lost systems. Safety is also a concern when using untested start-up systems, and it would likely be difficult to get Civil Aviation Safety Authority approvals to use such systems in populated areas.

The imaging system (or mounted ‘payload’) is one of the most important considerations when choosing an unmanned aerial vehicle system. Some unmanned aerial vehicles carry a fixed payload that cannot be interchanged, while for others the operator can select from a range of different payloads, and finally some allow the operator to attach their preferred payload. The appropriate payload depends on the desired ground resolution and spectral requirements (for example, normal RGB images, multi-spectral, infrared or thermal images). The imaging system can determine the unmanned aerial vehicle needed to carry the payload.

Other key considerations are the spatial coverage needed and characteristics of the location of operation, which will dictate the range endurance needed and the appropriate launch and retrieval methods. There are many other considerations and all factors are inter-related. Cleguer and

Hodgson at Murdoch University are currently preparing a journal article outlining the considerations and best procedures for selecting unmanned aerial vehicles for wildlife research.

11.9 Alternative imaging systems

All of the research presented here has tested unmanned aerial vehicles that carry normal RGB imaging systems, that is, cameras that capture images in the normal visible colour spectrum. Other types of imagery include thermal, infrared, multispectral and hyperspectral imaging. The applicability of thermal or infrared imaging depends on the context and research question. One main limiting factor is that these two forms of imaging cannot be used to 'view' animals below the water surface, thus negating one of the main advantages of aerial surveys. However, using thermal or infrared cameras on drones could be useful for marine megafauna that haul out, as demonstrated for New Zealand fur seals (Gooday et al. 2018) and grey seals (Seymour et al. 2017), or for detecting nesting turtles at night (Rees et al. 2018).

Multispectral imaging provides the opportunity to capture particular bandwidths of the electromagnetic spectrum, which means that for aerial images over the ocean, it is possible to target the bands that would provide the greatest penetration through the water column and eliminate obstruction from glare, sun glitter, and other reflectants. Schoomaker et al. (2011) showed evidence that multispectral images could provide clearer detections of whales in images, and even reveal whales that were not visible using normal RGB images.

Using imaging systems on unmanned aerial vehicles that provide data on animals that are not visible in normal RGB images, and using these data to estimate animal densities or population sizes, will require recalibration of availability estimates. These non-RGB imaging systems would need to be used to collect focal follow observations of the target species, as per Hodgson et al. (2017) so that new availability estimates could be calculated.

Hyperspectral imagery may provide even more detailed observation data. Both multi- and hyper-spectral imaging systems need to be tested and the potential advantages of these systems investigated. Increasing the detection probability for any species would provide more reliable population estimates and better information about distribution and habitat use.

11.10 Image processing: sighting and spatial data

A key advantage to using unmanned aerial vehicles and imaging systems for marine megafauna monitoring is the potential to standardise data collection. This advantage could apply to the sighting data, spatial data, and environmental characteristics and conditions, if all of these data can be recorded from the images automatically using computer algorithms. Also, by integrating the imagery and the telemetry data (flight characteristics such GPS tracks, altitude and the rotations of the unmanned aerial vehicle in space) which is recorded to relatively high resolution (i.e. to nearest second or millisecond) by most unmanned aerial vehicles, this technology can provide more accurate and higher resolution spatial data than manned surveys.

11.10.1 Automating the collection of sighting data

Weinstein (2018) provides a review of the use of computer vision in animal ecology and, in particular, for identifying animals in images. He recognises the difficulty of this task as the natural world is complex and heterogeneous — changes in illumination and backgrounds, as well as animal appearance and shape, make animal detection difficult. In most cases, the human eye is still better than computer vision systems, however, for large image datasets, it is not time- or cost-efficient to continue to manually record animal sightings.

There are a number of automated detections systems that are being developed for terrestrial animals (Weinstein 2018), as well as images of fish underwater (e.g. Hernández-Serna and Jiménez-Segura 2014). Seymour et al. (2017) present an automated detection algorithm for thermal imagery of hauled out seals. Maire et al. (2015) developed a machine learning system to detect dugongs in unmanned aerial vehicle images using training images provided from the trial surveys described in Hodgson et al. (in prep). The aim of the system is to produce a set of potential dugong detections that are then verified by the researcher. The recall of this system (i.e. the proportion of known dugongs detected in a set of test images) was 80 per cent. The precision of the system (proportion of detections that were true dugongs as opposed to false detections) was 27 per cent. Frederic Maire (Queensland University of Technology), Hodgson and Chris Cleguer from Murdoch University, are continuing to test and improve this system with more labelled dugong images. Immanent testing will include investigating whether particular environmental conditions or characteristics affect the system's recall rate.

The dugong detection system (Maire et al. 2015) could be adapted to detect other marine megafauna. This would require training images sets from manually reviewed images with labelled animal sightings. Such image sets already exist for whales (including sets from Murdoch University as well as Australian Antarctic Division and NOAA), and dolphins (Murdoch University, although not labelled to species), and are currently being produced for turtles and rays (Murdoch University in collaboration with Project Manta, University of Queensland). The ability to automate the whole detection process, including identifying species, so that no human input is required, would depend on the resolution of the images. Species classification has been proven for other taxa (Hernández-Serna and Jiménez-Segura 2014), but this capability currently does not exist for marine megafauna.

11.10.2 Spatial data processing

The challenge with collecting aerial images over water is in georeferencing the images – there are generally no land marks and therefore the standard 'image stitching' software cannot be used for these survey images. Hodgson et al. (2017, in prep), used a customised version of VADAR (www.brahss.org.au/content/vadar.html) to map the outline of all images using GPS data and unmanned aerial vehicles rotation data (pitch, tilt and roll) written to each image at the time of capture. VADAR was also used to plot the GPS location for each individual animal within the images. The data from VADAR were imported into ArcGIS and the total area surveyed could be calculated to a high degree of accuracy. The accuracy is an improvement on manned aerial surveys in which the altitude is not recorded to the same precision and resolution, and rotations of the aircraft are not recorded at all. Therefore, estimates of the spatial coverage from a manned aircraft are much coarser than from an unmanned aerial vehicle. As population estimates include a correction for the sample rate of the survey (i.e. the proportion of the survey area actually covered by the survey), the accuracy population estimates are dependent on establishing an accurate area of coverage. Lisein et al. (2013) also reviews methods to produce survey area coverage from unmanned aerial vehicle images and found that the method used in VADAR is the most accurate and easiest to implement.

As this customised version of VADAR is no longer being supported, Murdoch University is now collaborating with Martin Wieser from TU Wien in Vienna who has developed *OceanMapper*. This new mapping software works very similarly to VADAR, but is integrated with the output from the automated dugong detector, so that every individual animal detected can be easily mapped by importing the csv file produced by the detection system. Similar to VADAR, *OceanMapper* can map the outline of the images, but can also display the actual image on the map.

Murdoch University is seeking funding to investigate the potential to classify images according to environmental characteristics and conditions. This information, including turbidity (traditionally incorporating an assessment of bottom visibility), sea state, glare and cloud cover, is needed to correct sightings for availability (e.g. Pollock et al. 2006). The ability to record these data automatically from the images would result in more reliable, standardised information about the environmental conditions in which the survey was conducted, and ultimately produce more reliable estimates of population size, distribution and habitat use.

11.11 Considerations of regulations

11.11.1 Aviation safety

As previously discussed (11.1 Applicability of unmanned aerial vehicles to marine megafauna monitoring), flying unmanned aerial vehicles in Australia is restricted to within line-of-sight (i.e. the unmanned aerial vehicle can only be flown to a distance where it can still be seen with the naked eye), and a flight altitude of 400 feet. The operator must obtain permission from the Civil Aviation Safety Authority to fly beyond these limits (see https://www.casa.gov.au/aircraft/standard-page/commercial-unmanned-flight-gaining-your-remotely-piloted-aircraft-pilot). Hodgson et al. (2017, in prep) were able to overcome this restriction by working with Insitu Pacific, who were granted the Civil Aviation Safety Authority permit and conducted all the flights. Chris Cleguer (Murdoch University) recently obtained permission to fly beyond these limits in remote areas of the Pilbara in Western Australia, by working with an unmanned aerial vehicle company who undertook the Civil Aviation Safety Authority approvals process. He was able to operate the unmanned aerial vehicle himself under their remotely piloted aircraft operator’s certificate. Any researchers wanting to fly an unmanned aerial vehicle that weighs over two kilograms (including the payload) needs to obtain a remote pilot licence.

11.11.2 Ethics: animal disturbance

The normal ethics approvals required to observe animals for research are applicable to all unmanned aerial vehicle operations. As the use of unmanned aerial vehicles to observe wildlife is relatively new, there are few empirical studies on the potential disturbance effects of this technology, and a number of reviews note that there is still need for policy and legal frameworks to address potential negative impacts on wildlife, and species specific research on disturbance from unmanned aerial vehicles (Smith et al. 2016, Mulero-Pázmány et al. 2017, Wallace et al. 2017). Hodgson and Koh (2016) briefly outline a suggested best practice for observing wildlife with unmanned aerial vehicles, which advocates the precautionary principle and that published studies should note any responses to the unmanned aerial vehicle methodology used.

In one of the few direct behavioural observations of the responses of marine megafauna to unmanned aerial vehicles, Bevan et al. (2018) found that there was no response of sea turtles to a unmanned aerial vehicle (DJI Phantom 4 Pro) flown as low as 10 metres, whilst in-water, or while nesting. However, they did observe responses from crocodiles to the unmanned aerial vehicles being flown below 50 metres and from crested terns to the unmanned aerial vehicle flown below 60 metres. Seals hauled out on ice showed a marked reduction in disturbance responses to a *ScanEagle* flying over compared to a helicopter, with the latter aircraft being the traditional platform for surveys in this instance (Moreland et al. 2015). There was no disturbance noted by land-based observers to the small unmanned aerial vehicles used to survey humpback whales, killer whales and harbour porpoises (Aniceto et al. 2018). A review of documented reactions of wildlife to unmanned aerial vehicles suggested that responses depend on the engine type, where fuel engines which are noisier, elicited the greatest responses, and flight characteristics, where targeted approaches elicited greater responses than passing mode flights. Birds were the most sensitive animal type, while fully aquatic animals were the least affected.

Researching the potential received noise levels from unmanned aerial vehicles, Christiansen et al. (2016) demonstrated that the noise from two types of multi-rotor unmanned aerial vehicles could only be heard above ambient noise when close to the water surface and when the unmanned aerial vehicle was 10 metres or lower. Erbe et al. (2017) states that:

> "*Compared to other platforms of marine observation, underwater levels from the drones in our study were tens of dB lower than those of small motorcraft […], and well below levels considered in environmental regulations of underwater noise […]. Drones therefore may provide a preferable platform in situations where bioacoustic impacts are of concern, or where behavioural responses of marine fauna to the observation equipment would affect data quality and quantity*."

However, Erbe et al. acknowledge that unmanned aerial vehicles can be heard about ambient noise in calm conditions and that the sound from unmanned aerial vehicles is within the hearing range of many species, so some work is required to determine whether disturbance responses occur in these situations.

11.12 Suggested works

I have identified several desktop studies and experiments outlined that would enable the transition to using unmanned aerial vehicles for marine megafauna monitoring within the Marine Park. The following outline explains the resources required to implement key pieces of work suggested.

11.12.1 Desktop analysis

The images collected during unmanned aerial vehicle trial surveys described in Hodgson et al. (2013) and Hodgson et al. (in prep) could be used to provide a better understanding of the ground resolution and imaging systems needed to detect turtles and dolphins in images and identify these taxa to species level. The following table outlines the work required to do this. Note that assessments of whether turtles or dolphins can be identified to species level within the unmanned aerial vehicle images will require experts to assess the images.

Desktop analysis	**Costing unit**	**Number of units**
Turtles - ability to identify species in unmanned aerial vehicle images and direct comparison of manned and unmanned sightings in various environmental conditions		
Review turtle sightings from unmanned aerial vehicle flights described in Hodgson et al. (in prep) and determine whether they can be assigned to species *	Days	14
Analyse data to determine what variables affect ability to identify species (e.g. if turtles are at surface or on bottom, animal size etc.)	Days	5
Direct comparison of turtle sightings in unmanned aerial vehicle images versus from observers - mapping, data processing and analysis	Days	10
Produce journal article	Days	30
Dolphins - ability to identify species in unmanned aerial vehicle images and direct comparison of manned and unmanned sightings in various environmental conditions		
Review dolphin sightings from unmanned aerial vehicle flights described in Hodgson et al. (2013, in prep) and determine whether they can be assigned to species **	Days	8
Analyse data to determine what variables affect ability to identify species (e.g. if turtles are at surface or on bottom, animal size etc.)	Days	5
Direct comparison of dolphin sightings in unmanned aerial vehicle images versus from observers - mapping, data processing and analysis	Days	10
Produce journal article	Days	30

* There were 650 sightings from the manned survey, so we can assume same number matching unmanned aerial vehicle flights. Expect 50 sightings from the images captured at the higher resolution. Assume expert can ID 50 turtles per day including recording sighting characteristics.

** There are 420 individual dolphins already recorded from 170 unmanned aerial vehicle images. Expect each image contains a sighting of one dolphin group and that species ID needed for group. Expect 50 sightings from the images captured at higher resolution. Assume expert can ID 30 dolphin groups per day including recording sighting and group characteristics.

11.12.2 Experiments

The limitation of the trial surveys conducted in Shark Bay by Hodgson et al. (2013, in prep) was that the waters were relatively clear — there was very little deep (i.e. sea floor not visible) turbid water, which is characteristic of some dugong habitat elsewhere. To ensure that future unmanned aerial vehicle dugong surveys are directly comparable to historic manned surveys, similar trial surveys should be conducted in more turbid dugong habitat than that encountered in Shark Bay. Hervey Bay is an accessible site where relatively high densities of dugongs occur in turbid waters similar to much of the Reef, and therefore offers an opportunity to complete the manned and unmanned survey comparison and allow future dugong surveys in the Reef to be conducted with unmanned aerial vehicles. The objective then would be to compare the results from observers on board a manned aircraft, and sightings in images captured by the *ScanEagle* unmanned aerial vehicle, in Hervey Bay, Queensland. The methods would follow those described in Hodgson et al. (in prep). Estimates of resources required for this experiment are provided in the table below and are based on two blocks from the survey plan normally used for Hervey Bay by the JCU group and includes 66 transects of 1618 kilometres total length. This complete survey plan would be covered twice by both aircraft.

Experiment	**Costing unit**	**Number of units**
Dugongs - direct comparison of manned and unmanned sightings in high water turbidity		
Unmanned aerial vehicle flight time (including 3 hours contingency)	Hours	28
Transport of system	Set cost TBA	
Time to review images (260 hrs) *	Days	35
Aircraft charter	Hours	25
Flights to Hervey Bay for *ScanEagle* operators and manned survey team	People	9
Accommodation and meals (10 pax including manned aircraft pilot - have multiplied number of days in field by 10) **	Days	150
Salary for manned survey team (5 pax - have multiplied number of days in field by 5)	Days	75
Two rental cars	Days	15
Planning, mapping and data analysis	Days	25
Field time for Project Leader	Days	15
Produce journal article	Days	15

* Assumes same rate of image capture as previous trial surveys, and time for manned review is 36 seconds per image

** Number of days in field is estimated as number of manned aircraft hours, divided by 5 (assuming total of 5 hours on transect per day) multiplied by 3 (assuming 2 grounded days per 1 flying day).

Depending on the results of this comparison, consideration should be given to exploring further the effects of sea state on unmanned aerial vehicle sightings in the context of the underlying assumptions about dugong diving behaviour and repeating the availability bias experiment using models (Pollock et al. 2006).

11.13 References

Aniceto, A. S., M. Biuw, U. Lindstrøm, S. A. Solbø, F. Broms and J. Carroll (2018). "Monitoring marine mammals using unmanned aerial vehicles: quantifying detection certainty." Ecosphere **9**(3): e02122.

Bevan, E., E. Navarro, M. Rosas, B. M. Z. Najera, L. Sarti, F. Illescas, J. Montano, L. J. Pena and P. M. Burchfield (2016). "Using unmanned aerial vehicle (UAV) technology for locating, identifying, and monitoring courtship and mating behaviour in the green turtle (*Chelonia mydas*)." Herpetological Review **47**(1): 27-32.

Bevan, E., S. Whiting, T. Tucker, M. Guinea, A. Raith and R. Douglas (2018). "Measuring behavioral responses of sea turtles, saltwater crocodiles, and crested terns to drone disturbance to define ethical operating thresholds." PLOS ONE **13**(3): e0194460.

Christiansen, F., A. M. Dujon, K. R. Sprogis, J. P. Y. Arnould and L. Bejder (2016). "Noninvasive unmanned aerial vehicle provides estimates of the energetic cost of reproduction in humpback whales." Ecosphere **7**(10): e01468-n/a.

Christiansen, F., L. Rojano-Doñate, P. T. Madsen and L. Bejder (2016). "Noise Levels of Multi-Rotor unmanned aerial vehicles with Implications for Potential Underwater Impacts on Marine Mammals." Frontiers in Marine Science **3**(277).

Christie, K. S., S. L. Gilbert, C. L. Brown, M. Hatfield and L. Hanson (2016). "Unmanned aircraft systems in wildlife research: current and future applications of a transformative technology." Frontiers in Ecology and the Environment **14**(5): 241-251.

Colefax, A. P., P. A. Butcher, B. P. Kelaher and B. Handling editor: Howard (2018). "The potential for unmanned aerial vehicles (UAVs) to conduct marine fauna surveys in place of manned aircraft." ICES Journal of Marine Science **75**(1): 1-8.

Dorsey, E. M., W. J. Richardson and B. Würsig (1989). "Factors affecting surfacing, respiration, and dive behaviour of bowhead whales, Balaena mysticetus, summering in the Beaufort Sea." Canadian Journal of Zoology **67**(7): 1801-1815.

Durban, J. W., M. J. Moore, G. Chiang, L. S. Hickmott, A. Bocconcelli, G. Howes, P. A. Bahamonde, W. L. Perryman and D. J. LeRoi (2016). "Photogrammetry of blue whales with an unmanned hexacopter." Marine Mammal Science **32**(4): 1510-1515.

Erbe, C., M. Parsons, A. Duncan, S. K. Osterrieder and K. Allen (2017). "Aerial and underwater sound of unmanned aerial vehicles (UAV)." Journal of Unmanned Vehicle Systems **5**(3): 92-101.

Fiori, L., A. Doshi, E. Martinez, M. B. Orams and B. Bollard-Breen (2017). "The Use of Unmanned Aerial Systems in Marine Mammal Research." Remote Sensing **9**(6): 543.

Fuentes, M. M. P. B., I. Bell, R. Hagihara, M. Hamann, J. Hazel, A. Huth, J. A. Seminoff, S. Sobtzick and H. Marsh (2015). "Improving in-water estimates of marine turtle abundance by adjusting aerial survey counts for perception and availability biases." Journal of Experimental Marine Biology and Ecology **471**: 77-83.

Gonzalez, L., G. Montes, E. Puig, S. Johnson, K. Mengersen and K. Gaston (2016). "unmanned aerial vehicles (UAVs) and Artificial Intelligence Revolutionizing Wildlife Monitoring and Conservation." Sensors **16**(1): 97.

Gooday, O. J., N. Key, S. Goldstien and P. Zawar-Reza (2018). "An assessment of thermal-image acquisition with an unmanned aerial vehicle (UAV) for direct counts of coastal marine mammals ashore." Journal of Unmanned Vehicle Systems.

Hagihara, R., R. E. Jones, A. Grech, J. M. Lanyon, J. K. Sheppard and H. Marsh (2014). "Improving population estimates by quantifying diving and surfacing patterns: A dugong example." Marine Mammal Science **30**(1): 348-366.

Hagihara, R., R. E. Jones, S. Sobtzick, C. Cleguer, C. Garrigue and H. Marsh (2018). "Compensating for geographic variation in detection probability with water depth improves abundance estimates of coastal marine megafauna." PLOS ONE **13**(1): e0191476.

Hagihara, R., R. E. Jones, S. Sobtzick, C. Cleguer, M. Hamann, T. Shimada and H. Marsh (2016). Improving the estimates of abundance of dugongs and large immature and adult-sized green turtles in Western and Central Torres Strait. Report to the National Environmental Science Programme. Cairns, Australia, Reef and Rainforest Research Centre Limited**:** 65.

Hernández-Serna, A. and L. F. Jiménez-Segura (2014). "Automatic identification of species with neural networks." PeerJ **2**: e563.

Hodgson, A., D. Peel and N. Kelly (2017). "Unmanned aerial vehicles for surveying marine fauna: assessing detection probability." Ecological Applications **27**(4): 1253-1267.

Hodgson, A. J., N. Kelly and D. Peel (2013). "unmanned aerial vehicles (UAVs) for surveying marine fauna: a dugong case study." PLoS ONE **8**(11): e79556.

Hodgson, A. J., N. Kelly and D. Peel (in prep). "Unmanned aerial vehicles produce superior data over human observers during large-scale aerial surveys of marine wildlife." TBA.

Hodgson, J. C. and L. P. Koh (2016). "Best practice for minimising unmanned aerial vehicle disturbance to wildlife in biological field research." Current Biology **26**(10): R404-R405.

Johnston, D. W., J. Dale, K. T. Murray, E. Josephson, E. Newton and S. Wood (2017). "Comparing occupied and unoccupied aircraft surveys of wildlife populations: assessing the gray seal (Halichoerus grypus) breeding colony on Muskeget Island, USA." Journal of Unmanned Vehicle Systems **5**(4): 178-191.

Kiszka, J. J., J. Mourier, K. Gastrich and M. R. Heithaus (2016). "Using unmanned aerial vehicles (UAVs) to investigate shark and ray densities in a shallow coral lagoon." Marine Ecology Progress Series **560**: 237-242.

Koski, W. R., G. Gamage, A. R. Davis, T. Mathews, B. LeBlanc and S. H. Ferguson (2015). "Evaluation of UAS for photographic re-identification of bowhead whales, Balaena mysticetus." Journal of Unmanned Vehicle Systems **3**(1): 22-29.

Linchant, J., J. Lisein, J. Semeki, P. Lejeune and C. Vermeulen (2015). "Are unmanned aircraft systems (UASs) the future of wildlife monitoring? A review of accomplishments and challenges." Mammal Review **45**(4): 239-252.

Lisein, J., J. Linchant, P. Lejeune, P. Bouché and C. Vermeulen (2013). "Aerial Surveys Using an Unmanned Aerial System (UAS): Comparison of Different Methods for Estimating the Surface Area of Sampling Strips." Tropical Conservation Science **6**(4): 506-520.

Maire, F., L. Mejias and A. Hodgson (2015). Automating Marine Mammal Detection in Aerial Images Captured During Wildlife Surveys: A Deep Learning Approach. AI 2015: Advances in Artificial Intelligence. B. Pfahringer and J. Renz, Springer International Publishing. **9457:** 379-385.

Marine Mammal Commission (2016). Development and Use of UASs by the National Marine Fisheries Service for Surveying Marine Mammals. Bethesda, MD, Marine Mammal Commission.

Marsh, H. and W. K. Saalfeld (1989). "Distribution and abundance of dugongs in the northern Great Barrier Reef Marine Park." Australian Wildlife Research **16**: 429-440.

Marsh, H. and D. F. Sinclair (1989). "Correcting for visibility bias in strip transect aerial surveys of aquatic fauna." Journal of Wildlife Management **53**: 1017-1024.

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

Moreland, E. E., M. Cameron, P. L. Boveng and R. P. Angliss (2010). Surveys of Seals in the Bering Sea Pack Ice using Unmanned Aircraft Systems, 18-22 Jan 2010 Alaska Marine Science Conference, Anchorage, Alaska.

Moreland, E. E., M. F. Cameron, R. P. Angliss and P. L. Boveng (2015). "Evaluation of a ship-based unoccupied aircraft system (UAS) for surveys of spotted and ribbon seals in the Bering Sea pack ice." Journal of Unmanned Vehicle Systems **3**(3): 114-122.

Mulero-Pázmány, M., S. Jenni-Eiermann, N. Strebel, T. Sattler, J. J. Negro and Z. Tablado (2017). "Unmanned aircraft systems as a new source of disturbance for wildlife: A systematic review." PLOS ONE **12**(6): e0178448.

Parra, G. J., R. Schick and P. J. Corkeron (2006). "Spatial distribution and environmental correlates of Australian snubfin and Indo-Pacific humpback dolphins." Ecography **29**(3): 396-406.

Peel, D., N. Kelly, J. N. Smith, S. Childerhouse, T. J. Moore and J. Redfern (2015). Quantitative assessment of the relative risk of ship strike to humpback whales in the Great Barrier Reef, Australia. Final report to the Australian Marine Mammal Centre Grants Programme (Project 13/46)**:** 89.

Pollock, K., H. Marsh, I. R. Lawler and M. W. Aldredge (2006). "Estimating animal abundance in heterogeneous environments: an application to aerial surveys for dugongs." Journal of Wildlife Management **70**: 255-262.

Rees, A. F., L. Avens, K. Ballorain, E. Bevan, A. C. Broderick, R. R. Carthy, M. J. A. Christianen, G. Duclos, M. R. Heithaus, D. W. Johnston, J. C. Mangel, F. Paladino, K. Pendoley, R. D. Reina, N. J. Robinson, R. Ryan, S. T. Sykora-Bodie, D. Tilley, M. R. Varela, E. R. Whitman, P. A. Whittock, T. Wibbels and B. J. Godley (2018). "The potential of unmanned aerial systems for sea turtle research and conservation: a review and future directions." Endangered Species Research **35**: 81-100.

Schoomaker, J. S., Y. Podobna and C. Boucher (2011). "Electro-optical approach for airborne marine mammal surveys and density estimations." US Navy Journal of Underwater Acoustics **61**(4): 968-985.

Seymour, A. C., J. Dale, M. Hammill, P. N. Halpin and D. W. Johnston (2017). "Automated detection and enumeration of marine wildlife using unmanned aircraft systems (UAS) and thermal imagery." Scientific Reports **7**: 45127.

Smith, C. E., S. T. Sykora-Bodie, B. Bloodworth, S. M. Pack, T. R. Spradlin and N. R. LeBoeuf (2016). "Assessment of known impacts of unmanned aerial systems (UAS) on marine mammals: data gaps and recommendations for researchers in the United States." Journal of Unmanned Vehicle Systems **4**(1): 31-44.

Smith, J. N. and S. Hedley (2013). Identification of humpback whale breeding areas in the Great Barrier Reef Marine Park: validation of a spatial habitat model. Final report to the Australian Marine Mammal Centre Grants Programme (Project 11/15)**:** 25.

Sobtzick, S., C. Cleguer, R. Hagihara and H. Marsh (2017). Distribution and abundance of dugong and large marine turtles in Moreton Bay, Hervey Bay and the southern Great Barrier Reef. A report to the Great Barrier Reef Marine Park Authority. Townsville, Australia, Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) Publication 17/21, James Cook University.

Sweeney, K. L., V. T. Helker, W. L. Perryman, D. J. LeRoi, L. W. Fritz, T. S. Gelatt and R. P. Angliss (2015). "Flying beneath the clouds at the edge of the world: using a hexacopter to supplement abundance surveys of Steller sea lions (Eumetopias jubatus) in Alaska." Journal of Unmanned Vehicle Systems **4**(1): 70-81.

Sykora-Bodie, S. T., V. Bezy, D. W. Johnston, E. Newton and K. J. Lohmann (2017). "Quantifying Nearshore Sea Turtle Densities: Applications of Unmanned Aerial Systems for Population Assessments." Scientific Reports (Nature Publisher Group) **7**: 1-7.

Thomson, J. A., A. B. Cooper, D. A. Burkholder, M. R. Heithaus and L. M. Dill (2012). "Heterogeneous patterns of availability for detection during visual surveys: spatiotemporal variation in sea turtle dive–surfacing behaviour on a feeding ground." Methods in Ecology and Evolution **3**(2): 378-387.

Thomson, J. A., A. B. Cooper, D. A. Burkholder, M. R. Heithaus and L. M. Dill (2013). "Correcting for heterogeneous availability bias in surveys of long-diving marine turtles." Biological Conservation **165**(0): 154-161.

Wallace, P., R. Martin and I. White (2017). "Keeping pace with technology: drones, disturbance and policy deficiency." Journal of Environmental Planning and Management: 1-18.

Watts, A. C., J. H. Perry, S. E. Smith, M. A. Burgess, B. E. Wilkinson, Z. Szantoi, P. G. Ifju and H. F. Percival (2010). "Small Unmanned Aircraft Systems for Low-Altitude Aerial Surveys." Journal of Wildlife Management **74**(7): 1614-1619.

Weinstein, B. G. (2018). "A computer vision for animal ecology." Journal of Animal Ecology **87**(3): 533-545.