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Desktop analysis to inform the design for megafauna monitoring within the Reef 2050 Integrated Monitoring and Reporting Program:

Final Report of the Seabirds 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 current seabird monitoring strategy for the Great Barrier Reef Marine Park (Marine Park) is the *Coastal Bird Monitoring and Information Strategy - Seabirds 2015-2050* (CBMIS-2015). This strategy is built around monitoring breeding populations of indicator species that represent different feeding guilds at identified essential breeding sites. Patterns of visitation aim to maximise the likelihood of surveys coinciding with the breeding of 20 species while minimising operational effort. Of necessity, the overall strategy is a compromise between the number of sites, visitation rates and logistic constraints. The Reef 2050 Integrated Monitoring and Reporting Program (RIMReP) review process undertaken here assesses whether the CBMIS-2015 strategy, designed within these constraints, is adequate to meet the needs of the *Reef 2050 Long-Term Sustainability Plan* (Reef 2050 Plan).

Prior to implementation of the CBMIS-2015, seabird monitoring in the Marine Park had declined since the early 1980s with major spatial and temporal gaps occurring in monitoring activities. Since implementation there has been a marked improvement in the number and consistency of survey outcomes, implying that the CBMIS-2015 is improving the ability of monitoring to detect trends in numbers of breeding birds at essential sites. The overall CBMIS-2015 sampling design appears to be robust against unforeseen logistical problems within any given season and thus is likely to be maintainable into the future.

CBMIS-2015 aims to detect adverse changes in population size in order to inform management and reporting. However, no required level of significant change is identified in this or any previous seabird management strategy. This is problematic because it is not possible to gauge the effectiveness of a monitoring program if the level of impact it is trying to detect is unknown. We have addressed this issue by estimating critical rates of change for seabird populations of the Great Barrier Reef (the Reef) based on internationally accepted International Union for Conservation of Nature criteria. These criteria suggest that the CBMIS-2015 needs to be able to detect approximately a 1.5 to two per cent change per annum within 10 years. This equates to an overall population decline of approximately 15 to 20 per cent over this same period.

Simulation-based power analyses suggest that the current CBMIS-2015 strategy requires sampling periods of between 20 and 25 years to detect an approximately 1.5 per cent per annum decline (warranting *Endangered* listing) and about 15 years to detect an approximate 3.2 per cent per annum decline (warranting *Critically Endangered* listing), there being some variation between species examined. This means that the CBMIS-2015 provides adequate power to detect population declines at essential sites within the period of the Reef 2050 Plan, particularly if existing historical data can be used to complement newly generated data. Use of historical data from the more intensively monitored Michaelmas Cay population for comparative analysis also improves the utility of CBMIS-2015 data for identifying more general trends across the Reef within Reef 2050 Plan timeframes.

However, the CBMIS-2015 strategy does not provide sufficient power (greater than or equal to 80 per cent) to detect trends resulting in a *Critically Endangered* listing within 10 years, as specified by the IUCN criteria. Consequently, it cannot trigger management actions within this timeframe. This lack of power stems from two sources. The first is an inaccurate estimation of peak breeding activity, which can significantly influence the utility of the data obtained. We provide specific recommendations for improving the current strategy in this regard. The second is natural variation in the breeding index being measured due to differential between-season survival, recruitment and breeding deferment. Currently, the ability of the CBMIS-2015 to further understand and quantify natural levels of variation and so improve predictive power is limited by the monitoring of only a single index of population change. Use of a single index also significantly limits the ability of the current strategy to identify and quantify the relative importance of different threatening processes impacting breeding populations. This limitation is particularly important in the context of the Reef 2050 Plan, which aspires to use monitoring data to understand environmental influences so that management can become more proactive.

To overcome these limitations we provide detailed recommendations for the monitoring of additional indices in conjunction with total breeding population. These indices provide high-resolution information on short-term changes in reproductive success and identify the potential ecological and threatening processes that drive these changes. We outline the utility and potential information gain from the indices including each additional index either individually or in combination.

Overall, we recommend an approach that uses a small number of additional indices (particularly *fledging success*) and mark-recapture across a subset of essential sites spanning the latitudinal range of the Reef, in combination with intense monitoring of a comprehensive set of indices within specially developed sub-populations of key indicator species where it is possible to use artificial nest sites and autonomous data collection to minimise logistic constraints and costs.

Seabird subgroup: Desktop analysis phase

Within the larger Megafauna Expert Group the Seabird subgroup has been tasked with '…*evaluating the adequacy of existing 'seabirds' monitoring activities and indices to achieve the objectives and requirements of RIMReP'.* This evaluation was undertaken by directly addressing the specific objectives and outcomes provided for the 'Desktop Analysis Phase' of the Reef 2050 Integrated Monitoring and Reporting Program (RIMReP) process as applicable to seabirds. These five objectives are detailed below.

Seabird objectives (terms of reference)

1. Provide a synopsis of all current monitoring and modelling activities relevant to the theme of the Expert Group (seabirds), identifying potential sources of data describing proposed indicators
2. Describe the current status of the relevant communities and define the desired environmental or social state and develop potential thresholds for each proposed indicator (i.e. for each proposed indicator, is there a credible number that defines a healthy state/condition that we should be aiming to achieve/maintain and that we can track progress toward? How might this number vary between locations/regions?)
3. Evaluate the adequacy of current monitoring and modelling of proposed indicators to achieve the objectives of RIMReP. The evaluation should consider:
   a. The accuracy of monitoring and modelling;
   b. The power to detect change in proposed indicators at magnitudes and spatial and temporal scales that are relevant for managers, stakeholders and for assessing the effectiveness of the Reef 2050 Plan.
   c. The adequacy of sampling methods, and
   d. The adequacy of the spatial and temporal resolution of current monitoring and modelling.
4. Identify gaps in (or issues with) current monitoring and modelling of proposed indicators. Gaps might be spatial (i.e. where an indicator is not measured), temporal (i.e. when indicators are not measured with sufficient frequency to maintain adequate knowledge of condition) or, in some cases, indicators might not be measured at all.
5. Evaluate new monitoring technologies for their potential to increase efficiency or statistical power and their compatibility with long-term datasets.

The following report documents, in turn, the individual desktop analyses, results and conclusions associated with fulfilling each of these objectives.

1.0 Current seabird monitoring and modelling activities

1.1 What is the current seabird monitoring strategy?

The current seabird monitoring strategy used throughout the Great Barrier Reef Marine Park (Marine Park) is outlined in detail in the *Coastal Bird Monitoring and Information Strategy - Seabirds 2015-2050* (CBMIS-2015) (Hemson et al. 2015, Appendix A). This is a recently developed strategy document based on a comprehensive evaluation of historic and current seabird monitoring practices over an extended period of time and throughout the Marine Park. The CBMIS-2015 has been operational since mid-2015 and it is anticipated that this strategy will remain in place until formal review in 2020. The overall aim of the CBMIS-2015 is 'to establish how populations of seabirds in Queensland change through time and to alert us to undesirable trends so that we might understand, reverse or mitigate them' (Hemson et al. 2015).

Hemson et al. (2015) provide the following executive summary of the CBMIS-2015:

'The Strategy is built around four indicator species representative of coastal, inshore, offshore and pelagic feeding guilds. Initial site selection prioritised these species and subsequent sites were added to improve coverage of species less well represented in the initial selections.

The sites and timing of visits laid out in the strategy aim to maximise the likelihood of obtaining useful data on 20 species of seabird while minimising operational effort.

The Strategy is divided into a list of *essential* sites and visits to be made each year, and a list of *significant* sites that will contribute valuable data if resources are available to include them.

The Strategy defines a maximum period of five years between visits for any *significant* site to ensure that major changes are not overlooked and highlights the need to integrate this condition with other requirements for visitation.'

The importance of timing and consistency (of sampling) are explained in detail (within the CBMIS-2015 document, Appendix A), as are matters of governance with respect to altering the strategy prior to the formal review in 2020.

The CBMIS-2015 'is a revision of the seabird component of the previous Coastal Bird Monitoring and Information Strategy (CBMIS) (McDougall 2011) and is considered to have been created using the best available data, expert opinion, commissioned reports and operational expertise. The strategy encompasses the east coast of Queensland and so is the strategy used throughout the Marine Park region.

Through necessity the CBMIS-2015 was developed using historic data that have been influenced by previous inconsistent monitoring methodologies and visitation schedules, combined with a decision support process that modified potentially ideal survey designs in response to a number of operational and/or logistic constraints (Hemson et al. 2015). Consequently, the overall strategy 'is a compromise between data quality and operational feasibility' (Hemson et al. 2015).

In essence this means that task of the RIMReP review process undertaken here is to assess whether the CBMIS-2015 strategy designed within these constraints is adequate to meet the needs of the Reef 2050 Plan for seabird populations of the Reef.

1.2 What data sources were used to develop the CBMIS-2015?

The data that formed the basis of the decision support tool used to develop the CBMIS-2015 were extracted from the Queensland Government's WildNet database. This is the official repository for all seabird population data obtained by State and Federal government staff throughout the Reef region. Any issues with these data that may have affected the design of the CBMIS-2015 are fully documented in Hemson et al. (2015), as are the rationale for including these datasets and the steps taken to ensure maximum data quality. Other seabird data sources beyond surveys undertaken by Queensland Parks and Wildlife Service andGreat Barrier Reef Marine Park Authority (the Authority) staff are included in the WildNet database, but it is likely that additional data still exists outside of WildNet that are held by researchers. Currently, the exact details of these datasets are unknown and likely unobtainable in many cases.

1.3 To what degree are the historic data compatible with data obtained in the CBMIS-2015?

Field methods and/or the breeding categories that are recorded during current seabird surveys have not changed significantly from those used historically. However, overtime there have been several iterations of field methods particularly at Raine Island. For example, at this location there have been gridded counts, estimates of nesting from numbers of flying birds, and counts from the historic lighthouse tower. Currently, total counts are undertaken for all the larger species (boobies, frigates), transects counts for cryptic nesters such as common noddies, plus detailed habitat searches for red tailed-tropic birds. Elsewhere, (for example, Swains Reefs cays or other sand cays including Michaelmas Cay) total counts have likely been the consistent methodology.

In general, the current survey methodology produces one or more estimates of the total number of birds breeding at a site and by taking the largest of these observations estimates total breeding population. Most previous survey methods have produced an equivalent measure and so are broadly compatible with the current system of monitoring. However, in some earlier years population counts were often discretised into broad categories (for example, zero, 10, 100, 1000), which limited the resolution of the counts and their utility for assessing population change. This practice does not occur under the CBMIS-2015. However, there has been, or is, no measure of the level of accuracy (that is, error) in the single per annum estimates for either the previous or present methodologies. Therefore, it is not possible to determine if current or previous methodologies are more or less accurate. Problems associated with this lack of error estimate are discussed elsewhere (Section 4.1.1).

1.4 What are the currently monitored species/foraging guilds and rational?

Indicator species in the CBMIS-2015 were selected through an expert and stakeholder group evaluation of their values as indicators of a particular foraging guild of seabirds, their predictability (site fidelity and phenology) and their geographic range. This process identified four species as broadly representative of coastal, inshore, offshore and pelagic feeding guilds.

Indicator species identified were:

- Little Terns (*Sternula abifrons*), coastal forager;
- Crested tern (*Thalasseus bergii*), inshore forager;
- Brown booby (*Sula leucogaster*), offshore forager; and
- Wedge-tailed shearwater (*Ardenna pacifica*), pelagic forager.

The strategy was created to ensure the best possible coverage of the indicator species and then sites and visits were added to improve coverage for all other species. It is important to note most significant seabird breeding sites support breeding populations of indicator and non-indicator species although breeding seasons may not completely coincide. Whether the focus on monitoring four key indicator species is sufficient to be able to extrapolate to all species that breed on the Reef depends firstly on whether all species in a guild respond similarly to environmental influences and threats, and secondly, whether the CBMIS-2015 strategy will obtain sufficient information on non-indicator species so as to be able to compare and extrapolate the findings obtained for indicator species.

The answer to the first question depends on the overlap in life-history characteristics among focal and non-focal taxa. This could be overlap in breeding or non-breeding food resources including prey types/size classes and foraging locations, species-specific nesting habitat and/or direct susceptibility to other non-starvation associated causes of mortality.

Species groupings within the CBMIS-2015 are organised around foraging guilds with different predicted foraging ranges. Therefore, beyond species-specific information the different key taxa primarily provide information on foraging resource availability during breeding at different distances from breeding colonies. The utility of this process assumes that the availability of prey of different sizes/types at similar distances from the colony likely varies in unison. Whether this is true for seabirds of the Reef, and exactly what the overlap is in this and other types of life-history characteristics and susceptibilities is largely unknown (see Section 4.1.2).

The answer to the second question depends on obtaining sufficient data on non-indicator species during a monitoring regime that is aimed at surveying indicator species during peak breeding. Whether this is likely to occur based on the CBMIS-2015 monitoring protocols is discussed Section 1.5.2 below and Section 4.1.4.

1.5 What is the current spatial and temporal pattern of monitoring and rationale?

The CBMIS-2015 divides monitoring activities into *essential* and *significant* sites with surveys at *significant* sites occurring once every five years. This means that it is only possible to assume minimum data acquisition at *essential* sites for this RIMReP review process.

1.5.1 Monitoring at essential sites

Monitoring protocols at *essential* sites are outlined in detail in Hemson et al. (2015). Briefly the CBMIS-2015 follows the spatial and temporal sampling pattern provided in Fig. 1.1. For each location/colony it is considered a high priority to undertake surveys within months shown in green, and a medium priority in a month shown in yellow. The overall strategy comprises a minimum of two surveys per year at the majority of sites and up to four per year in the Capricorn-Bunker Island group, with the two principal surveys at each site per year being conducted in non-consecutive green bands.

Both the spatial and temporal pattern of sampling shown in Fig. 1.1 were developed using data on relative population sizes at different locations and breeding phenology extracted from the WildNet database. This procedure provided an objective estimate of the importance of each site to each species. Monitoring site selection was then made on the basis of this index of relative importance in combination with expert opinion, after which site selection was also vetted for feasibility. Spatially, the final sampling program covers the entire Reef, from Lady Elliott Island in the south to Raine Island in the north, with no significant gaps occurring for known breeding colonies. There is a possibility that a number of smaller unknown colonies in the Mackay region have not been included in the sampling regime, but this is unlikely due to the general geomorphology of the region combined with a lack of sand cays (Hemson et al. 2015). However, there are significant seabird breeding colonies outside of the Reef region, particularly in the Torres Strait, that are not included in this monitoring program.

It is particularly important to note that within the general program outlined in Figure 1.1, monthly counts of breeding seabirds are undertaken at Michaelmas Cay. The seabird data from Michaelmas Cay are the longest-term, highest resolution data on breeding participation for any site on the Reef. The long-term monitoring at this site combined with the implementation of monthly counts means that Michaelmas Cay is currently the only breeding colony on the Reef where breeding peaks for ground nesting tern species can be clearly identified and where the likelihood of being able to estimate robust, long-term trends in breeding seabird populations is highest; along with the correlation of these trends with patterns of environmental variation (Section 2).

Much of the current CBMIS-2015 strategy has been designed based on the findings of a previous power analysis that used data from Michaelmas Cay and it will continue to be used as an important dataset against which potential environmental impacts and population declines at other locations can be assessed. The maintenance of this sampling regime at Michaelmas Cay is therefore a critical component of the ongoing CBMIS-2015 strategy and needs to be documented as such (Section 4.1.3).

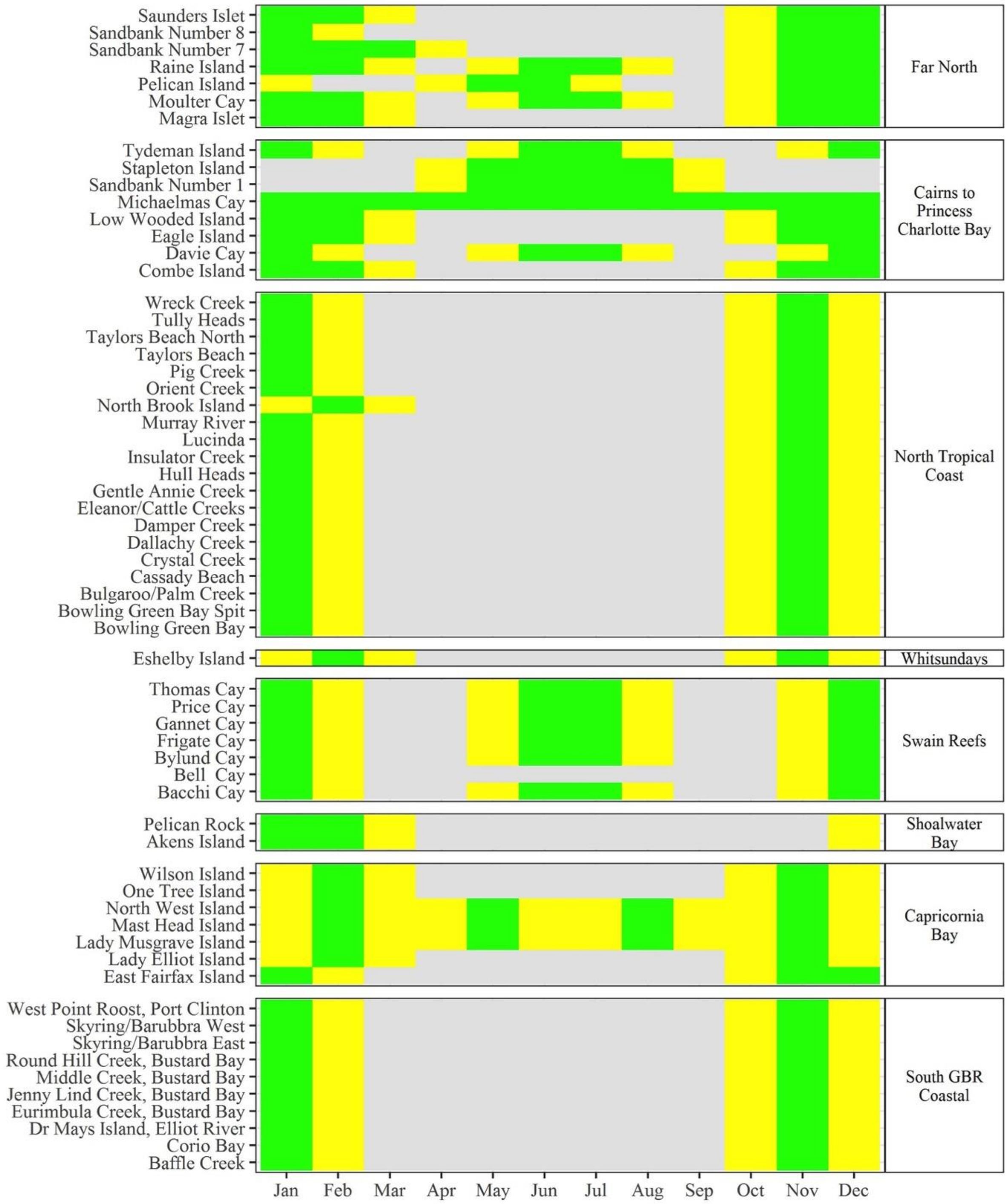

Figure 1.1. *Essential* site monitoring scheme adapted from Table 2 of Hemson et al. 2015. Essential sites (left axis) are grouped by region (right axis). High priority surveys months are shown in green, medium priority months are shown in yellow, and low priority months are shown in grey. In general, CBMIS-2015 recommends that at least one survey be conducted in each contiguous set of high priority (green) months at each site. Prioritisation of survey months was based on the focal species present at each site and best available information on the timing of their breeding.

Figure 1.2 summarises the number of surveys that have been conducted per year across all *essential* sites since the 1980's using a similar colour scheme to Hemson et al. (2015) (patterns of visitation for each essential site since 2012 when the previous monitoring scheme was implemented is provided in Appendix B). It is obvious from this figure that until implementation of the CBMIS-2015, the intensity of monitoring of seabird populations on the Reef had been steadily declining since the mid to late 1980s, with the mid 2000s seeing all-time lows in monitoring activities across the reef. This hiatus and lack of associated high quality data has the potential to significantly impact trend analyses attempted using this data set, particularly given the relative lack of activity at *essential* sites. In contrast, since the full implementation of the CBMIS-2015 there have been near all-time highs in the total number of surveys undertaken. In addition, approximately twice as many surveys have occurred during high priority months (Table 2 CBMIS-2015, Appendix A) than have occurred in previous years. Since 2012, most of the 57 *essential* sites in the Reef have been visited at least once with most visits occurring within high priority time periods. This has led to at least one survey, per site, per year, per high priority time period (Appendix B).

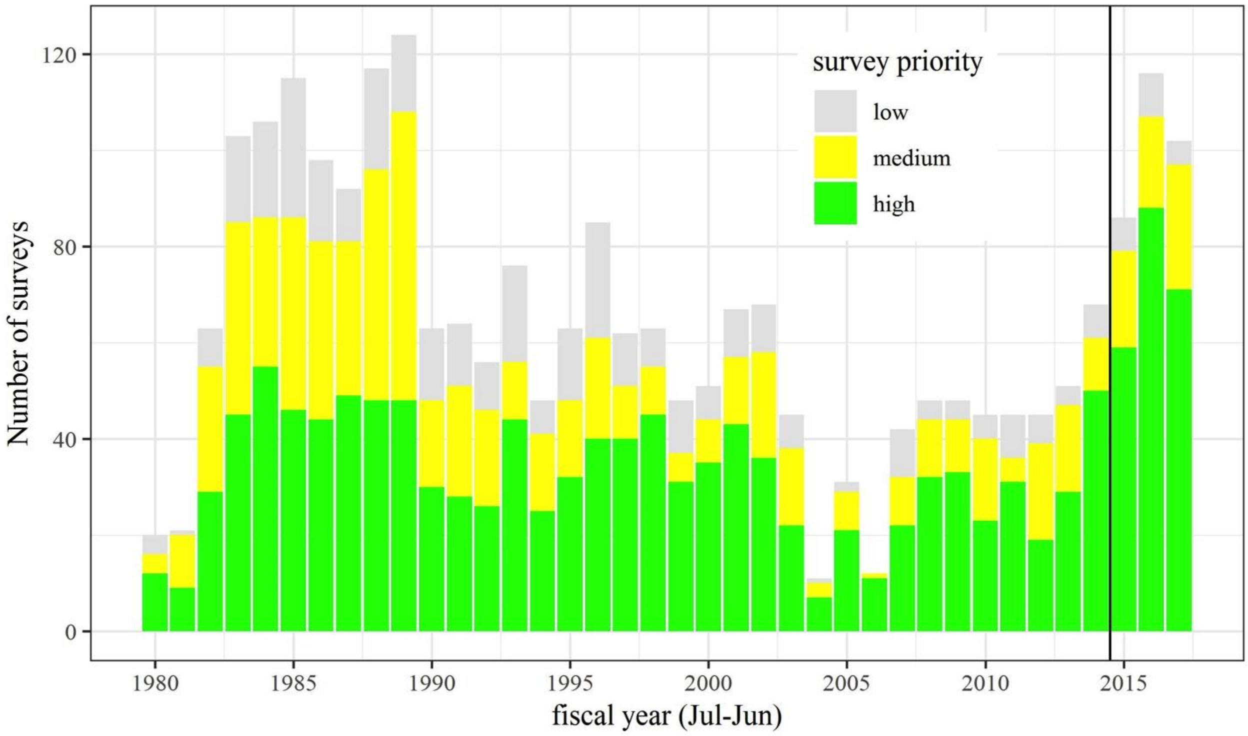

Figure 1.2. Summary of seabird surveys conducted at *essential* sites as of March 2018. The stacked bar chart highlights that, since implementation of the CBMIS in the 2015/16 fiscal year, (i) the proportion of surveys conducted in high priority (green) and medium priority (yellow) months is higher than at any point in history, and (ii) the total number of surveys is similar to all-time highs that occurred in the 1980s. Only surveys conducted since July 1980 are shown. A detailed breakdown of essential site visits since 2012 is available in Appendix B.

In general, these data suggest a significant improvement in the number and consistency of survey outcomes due to the implementation of the CBMIS-2015 implying that this monitoring strategy is having a real impact on the timing of visits, and potentially improving the ability to determine breeding peaks.

However, while visits have primarily occurred within high priority periods, there are occasional gaps. Any problems in meeting the specified survey regime occur largely due to logistic issues, such as vessel breakdown and scheduling conflicts, thus implying that there is always likely to be some level of attrition due to unforeseeable circumstances. A buffer against these types of logistic constraints has been built into the CBMIS-2015 via a time window of availability over which counts at any particular *essential* site can occur. Over the 2016 to 2018 period when the CBMIS-2015 had been in full operation, actual surveys undertaken had been between 90 and 95 per cent of those planned, with no apparent systematic bias towards missing counts for individual taxa or at specific sampling locations. Therefore, this buffer appears to have been effective to date in reducing the impact of unforeseen logistic issues. Consequently, the overall sampling design appears to be maintainable into the future.

1.5.2 Monitoring of non-indicator species

In the CBMIS-2015, the monitoring of non-indicator species is to occur at essential sites *"when possible"* (Hemson et al. 2015). In 'practice', this equates to all species at an *essential* site being counted during any visit to that site regardless of whether they are indicator species or not. At no time are non-indicator species not counted when a breeding site is visited due to time constraints or other logistic concerns (Hemson *pers comm*). Assuming this continues to be the case (see Section 4.1.4), then species status (indicator verses non-indicator) only affects *essential* site selection and visitation rates. This means that surveying of non-indicator species may or may not occur during breeding peaks for those species and so data on non-indicator species may or may not be sufficient to produce robust statistical analysis.

However, as most of the *essential* sites are important for many species and the timing of breeding is broadly similar for many species, it is likely that much of the data on non-indicator species may be of very similar quantity to that on key species data. This allows robust comparisons between indicator and non-indicator species at the same locations. However, it must also be noted that the CBMIS-2015 strategy does not guarantee that these comparisons are possible, particularly for species that are consistently out of breeding synchrony with the indicator species present (see Section 4.1.4).

Also, since visits are not designed specifically to occur during breeding peaks for non-indicator species, it becomes particularly important to know the stage of breeding associated with counts for any non-indicator species and whether any lack of data on a particular species at a particular site is a real absence or a missing survey. In the CBMIS-2015, monitoring protocols specified for non-indicator species are the same as for indicator species. They include the documentation of different life-history stages (such as nesting adults, eggs, chicks) with the quality of data acquisition being essentially the same for both indicator and non-indicator groups. Therefore, the main concerns regarding types of data obtain for non-indicator species is whether breeding synchrony occurs with indicator species and whether adequate documentation of breeding absences occurs in such a way as to be easily identifiable in the resulting database (see Section 4.1.4).

1.5.3 Monitoring at Significant sites

According to CBMIS-2015 protocols, *Significant* sites are to be visited at least once every five years for a single total-observed population count on all species present. The timing of visits to *Significant* sites are scheduled around predicted peak breeding in the same manner as for *Essential* sites thus providing useful comparative data between the two. However, at this level of visitation it is likely that no trend analyses are possible using the data obtained from these sites.

However, the quantitative use of count data obtained from *Significant* sites once every five years is not given as the intent of the strategy outlined in the CBMIS-2015, with data acquisition on breeding numbers at these locations only being an opportunistic consequence of visitation. The principal rationale given for visits to *Significant* sites is to check for emerging threats such as the presence of feral/exotic plants and animals etc. and to supplement the broader monitoring program with all available relevant data that can be gathered during a visit.

The CBMIS-2015 considers it important not lose site of the value of maintaining a careful watch on colonies for the presence of these emerging threats. Early identification of rat, cat, exotic insect/pathogen or weed infestations that have the potential to dramatically change ecosystem structure and function has considerable value for conserving breeding populations of seabirds even if the associated seabird population data are not quantitatively robust. Visits to these breeding locations once every five years, as part of the CBMIS-2015, is considered adequate only because additional visits are undertaken as part of the ongoing *Island Watch* program. This program is specifically designed to monitor for pests and other threats while assessing the general health of island habitats. Therefore, a visitation rate of every five years specified by CBMIS-2015 should only be considered adequate in conjunction with these other ongoing monitoring activities.

1.6 What are the currently monitored indices?

Currently the CBMIS-2015 estimates the maximum number of breeding pairs per species per year at each monitoring site. It does this to provide an index of the "total breeding population", which reflects the number of individuals that survive or recruit into the population *and* which participate in breeding in any one year. Potential issues with the use of this single index are discussed in Section 4.2.

Monitoring and counting of active nests occurs using two methodologies:

1. Visual counts
    a. Standard full population counts during breeding by trained observers with stage of breeding (adults vs. adolescent) and numbers in different breeding sub-categories (nests, eggs, chicks, young) recorded.
    b. Drone counts by trained pilots — *Still in developmental phase (see Section 5.1.1)*
    c. Static preprogramed cameras mounted at height adjacent to colonies — *Still in developmental phase (see in Section 5.1.2)*
2. Acoustic estimation
   Remote sensed full breeding cycle population size index based on species-specific call rates — *Still in developmental phase (see Section 5.1.3)*

2.0 Current status of seabirds of the Great Barrier Reef and potential thresholds

2.1 What is the current status of the relevant communities based on the currently monitored indices?

A formal analysis of the current status of the seabird populations in the Reef is beyond the scope of this report, but is currently underway, having been commissioned by the Queensland Department of Environment and Science. The objective of these analyses is to test for and quantify recent and long-term temporal trends in abundance of breeding pairs of a suite of seabird species for which adequate numbers of breeding records exist. Since 1980, over 6000 records of breeding pairs of 20 seabird species have been recorded in the region, including approximately 500 breeding records across 20 species since implementation of the current monitoring scheme in 2015. Of the approximate 200 sites where breeding pairs have been recorded, nine sites have accounted for approximately 50 per cent of breeding records and 57 sites have accounted for 90 per cent of breeding records. Michaelmas Cay alone has accounted for approximately 18 per cent of all breeding records. The species with the most breeding records are brown booby (n = 900), crested tern (n = 700), bridled tern (n = 670), common noddy (n = 665), black-naped tern (n = 600), silver gull (n = 565), sooty tern (n = 480), and masked booby (n = 480), with the remaining twelve species having fewer than 400 breeding records (Figure 2.1). Among the four focal species, trend analyses are likely to yield the most robust results for brown boobies and crested terns due to the high number of breeding records for these species. Data for little terns are currently too sparse for population assessment, with fewer than 40 breeding records in the region between 1980 and November 2017. Wedge-tailed shearwaters are the most abundant breeding seabird in the region, but the fact that they nest in burrows means that alternative methods for counting and analysing abundance of breeding pairs are needed (Hemson et al. 2015 and discussed in Section 5.1.3).

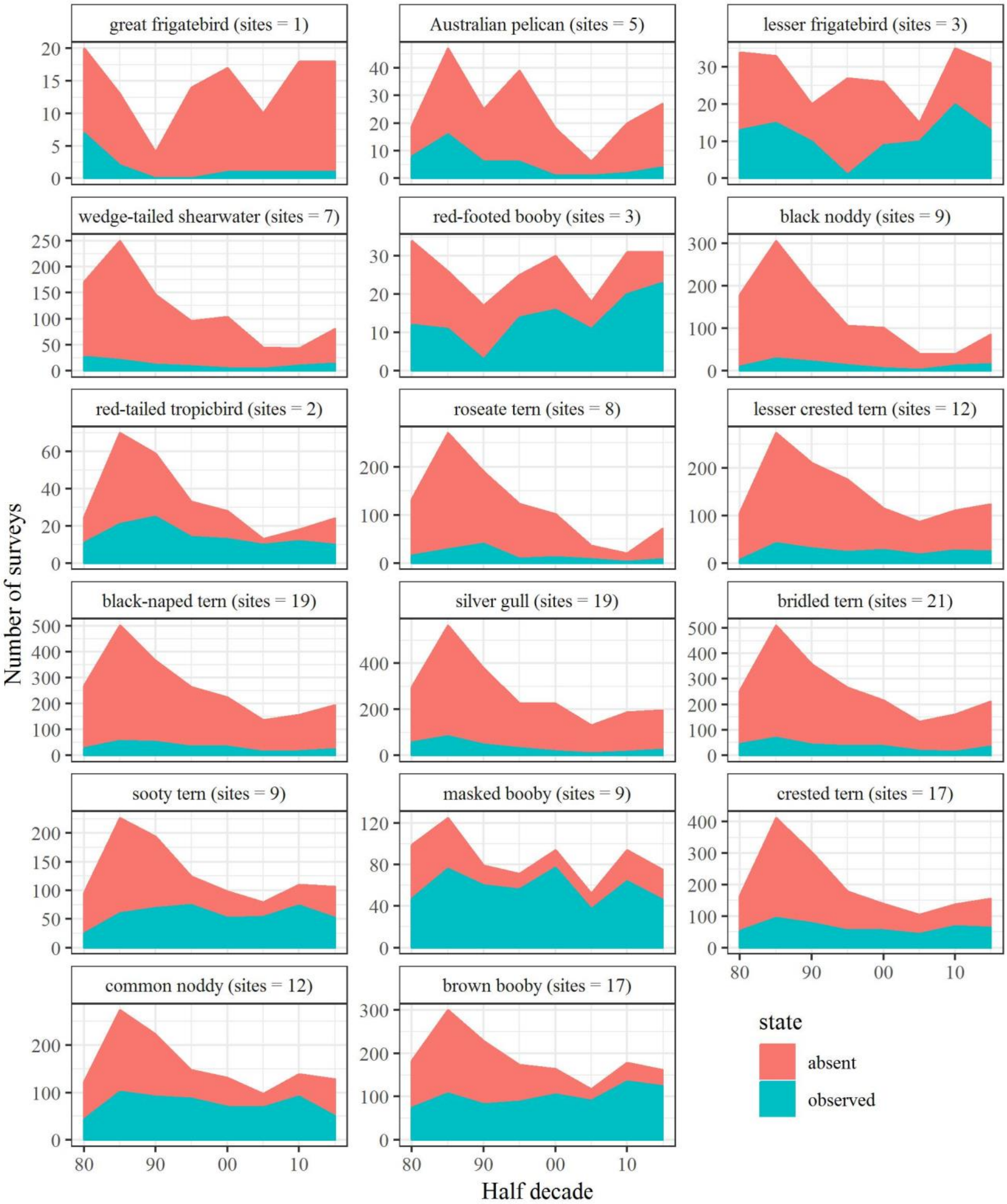

Figure 2.1. Summary of surveys reporting breeding pairs of seabirds at five-year intervals between 1980 and March 2018. For each species, blue areas show the number of surveys that reported breeding pairs and pink areas show the number of surveys where no breeding pairs were reported. The combined height of the blue and pink areas equals the total number of surveys conducted at known breeding sites for each species. The number of essential sites where breeding

has been recorded for each species are shown in parentheses. Only data from essential sites and for species with at least 10 records of breeding pairs are shown.

2.2 What magnitude of change/criteria needs to be detected to identify problems/trigger management actions?

Accurately assessing seabird population trends for management purposes requires identifying the level of decline/change considered ecologically significant and the statistical power required for detecting a trend of this magnitude. No required level of trend detection is specified in the CBMIS-2015, or any previous management plan or strategy. In the case of the CBMIS-2015, this is because the original intent was to ensure that the best possible data were available to inform management and reporting within current operating constraints. However, this is a problem when trying to assess the ability of the strategy to meet the needs of the Reef 2050 Plan. The assessment criteria required stem from the desired management outcomes and/or the need to raise flags for management intervention at appropriate times. Therefore, to aid the RIMReP process we have attempted to develop appropriate criteria against which to undertake this assessment.

The Reef 2050 Plan focuses on the Reef maintaining the outstanding universal values for which the World Heritage Convention listed it as a World Heritage Area in 1981. Maintaining World Heritage status requires an assessment process undertaken against a set of internationally developed criteria. Consequently, internationally developed and accepted criteria on what constitutes significant, or undesirable, ecological change in avian populations are also likely to be the most appropriate criteria for detecting change in seabird populations of the Reef.

A number of international and regional criteria associated with detecting significant negative trends in other avian systems have been developed. Most of these are in some way linked to, or developed from, the International Union for Conservation of Nature criteria for communicating the risk of extinction and the listing of species/populations threatened with extinction (Table 2.1). These include but are not limited to:

- International Union for Conservation of Nature, Red List of Threatened Species criteria. Birdlife International use these criteria.
- Nature Conservation Act, is the legislation used in Queensland. This Act lists species in Queensland using International Union for Conservation of Nature criteria.
- Environmental Protection and Biodiversity Conservation Act (EPBC Act). This is the Federal Government legislation. This Act also lists Australian species using International Union for Conservation of Nature criteria.
- Back on Track species prioritisation framework is a Queensland species prioritisation process. This process is not statutory but assesses whether recovery is likely and other factors to prioritise recovery actions.

Table 2.1. Summary of International Union for Conservation of Nature population reduction criteria for the evaluation of the risk of extinction (IUCN 2012).

Category	Criteria
Critically Endangered (CR)	Population size reduction of ≥ 90% over the last 10 years or three generations, whichever is the longer, where the causes of the reduction are clearly reversible AND understood AND ceased
	OR
	Population size reduction of ≥ 80% over the last 10 years or three generations, whichever is the longer, where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible
Endangered (EN)	Population size reduction of ≥ 70% over the last 10 years or three generations, whichever is the longer, where the causes of the reduction are clearly reversible AND understood AND ceased
	OR
	Population size reduction of ≥ 50% over the last 10 years or three generations, whichever is the longer, where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible
Vulnerable (VU)	Population size reduction of ≥ 50% over the last 10 years or three generations, whichever is the longer, where the causes of the reduction are: clearly reversible AND understood AND ceased
	OR
	Population size reduction of ≥ 30% over the last 10 years or three generations, whichever is the longer, where the reduction or its causes may not have ceased OR may not be understood OR may not be reversible

The Australian Government Department of Environment and Energy, as well as the Queensland Department of Environment and Science, currently use International Union for Conservation of Nature criteria for assessing total species status under the Nature Conservation Act, the EPBC Act, and *The Action Plan for Australian Birds 2000*. The adoption of these criteria by multiple Australian government agencies implies that they are also the most appropriate internationally recognised criteria for assessing the risk of extinction of seabird species at critical breeding locations on the Reef. Note that several species/population characteristics are used by the International Union for Conservation of Nature including geographic range but for the purposes of this review we have focused on changes in population size as this is the population index closest to the value estimated by the CBMIS-2015.

We have applied the International Union for Conservation of Nature criteria specifically to seabird populations breeding on the Reef. The current International Union for Conservation of Nature status of each species, and the per cent population decline required to move a particular species from one International Union for Conservation of Nature category to another over a specific time period, are given in Table 2.2. To generate Table 2.2, we calculated the average annual per cent declines required using generation times provided by the Birdlife International Portal (http://datazone.birdlife.org/species/spcpop, last accessed 23 May 2018) based on available life-history data. Calculations of annual average per cent declines assumed exponential population growth (see *Section 3* for details).

Table 2.2. Current threat categories of Great Barrier Reef seabird species and minimum per cent decline thresholds over 10 years to qualify for *Vulnerable* (VU), *Endangered* (EN), and *Critically Endangered* (CR) status based on the International Union for Conservation of Nature 2012 population reduction criteria outlined in Table 2.1. Methods for calculating per cent declines are described in Section 3.1.3. Annual average per cent declines for focal species are presented in Section 3.

Species	Generation length	Threat category by conservation body				Per cent decline over 10 years to meet IUCN threat category criteria		
		NCA	BOT	EPBC	IUCN	VU	EN	CR
Australian pelican	16	LC	L	LC	LC	7.2	13.5	28.5
beach stone curlew	10.5	V	H	LC	NT	10.5	19.5	39.5
black noddy	10.8	LC	L	LC	LC	10.2	18.9	38.6
black naped tern	11	LC	L	LC	LC	10.2	18.9	38.6
bridled tern	11.3	LC	L	LC	LC	10	18.4	37.7

brown booby	17.3	LC	L	LC	LC	6.6	12.5	26.6
Caspian tern	12.2	LC	L	LC	LC	9.2	17.1	35.3
common noddy	12.9	LC	L	LC	LC	8.7	16.3	33.8
crested tern	10.5	LC	L	LC	LC	10.5	19.5	39.5
fairy tern	11	LC	L	V	V	10.2	18.9	38.6
greater frigatebird	15.2	LC	L	LC	LC	7.5	14	29.5
herald petrel	15.6	E	CE	L	LC	7.3	13.7	29
lesser crested tern	11	LC	L	LC	LC	10.2	18.9	38.6
lesser frigatebird	15.5	LC	L	LC	LC	7.3	13.7	29
little tern	10.9	E	H	LC	LC	10.2	18.9	38.6
masked booby	16.3	LC	L	LC	LC	7	13.2	28
red footed booby	13	LC	L	LC	LC	8.7	16.3	33.8
red tailed tropic bird	13	LC	L	LC	LC	8.7	16.3	33.8
roseate tern	10.2	LC	L	LC	LC	10.9	20	40.5
silver gull	11.5	LC	L	LC	LC	9.7	18	36.9
sooty tern	10.9	LC	L	LC	LC	10.2	18.9	38.6
wedge-tailed shearwater	16.5	LC	L	LC	LC	6.9	12.9	27.5

These results suggest that, in general, the CBMIS-2015 needs to be able to detect declines of between approximately five and 10 per cent over a 10-year period if it is to identify species or populations that are *Vulnerable,* and approximately 15 to 20 per cent over 10 years for *Endangered*. Detecting a *Critically Endangered* species requires identifying declines of approximately 25 to 30 per cent over 10 years for seven of the 11 species listed, including most large bodied species such as boobies, frigatebirds and shearwaters and declines of between 30 and 35 per cent over 10 years for the remaining taxa.

The results in Table 2.2 imply that for offshore and pelagic foraging indicator species, i.e. wedged-tailed shearwaters, and brown boobies, changes in IUCN status occur at lower percentage declines than for other species due to their longer generation times. This further justifies their use as indicator/focal species. For inshore and coastal indicator species such as crested terns, or those with smaller body size, changes in IUCN status occur at higher percentage declines than for non-indicator species with similar life histories. This implies that if status change is occurring in these indicator species then equivalent or greater changes are also likely to be occurring in non-indicator species with similar ecology and life histories.

Using our metrics of success, table 2.2 suggests that in general the CBMIS-2015 needs to be capable of detecting a 15 to 20 per cent change over 10 years, which equates to an approximate 1.5 to two per cent change per annum over this same period. This level of change (approximately 1.5 to two per cent, per annum) would identify declines in most species before they become *Endangered*. The smaller the changes the strategy can detect the more useful it is for triggering management aimed at stabilising trends in populations before they undergo significant or irreversible decline.

2.3 Statistical power required to detect a trend of a specified magnitude

The stability of a population at any point in time is assessed against accumulated baseline data on “normal” intrinsic year-to-year variation in total breeding population. Importantly, these baseline data are assumed not to reflect variation associated with major ecological perturbations, other possible external influences, such as anthropogenic disturbance, and/or variation introduced due to the sampling strategy of the monitoring program itself. These data also need to be collected over a sufficient period and at appropriate sampling intervals so as to provide a robust ability to forward project future trends.

As outlined by Fuller and Dhanjal-Adams (2012), in practice the choice of statistical power thresholds depends on the purpose of the monitoring program and is often influenced by the needs of a range of stakeholders and funders. In general, standard scientific analyses attempt to reject a null hypothesis of no change. This requires very strong statistical evidence that an observed effect is real. However, in the case of diagnosing threats it is usually more prudent to take a precautionary approach, such that a population might be considered to be in decline unless it is certain that it is stable or increasing. This is precisely the approach recommended by IUCN (2012) for assessing species for admission to the Red List of Threatened Species. The IUCN (2012) also recommend that the attitude to risk (precautionary versus evidentiary) should be explicitly documented. The downside of more liberal thresholds is the cost of management actions that are inevitably triggered more frequently at lower statistical thresholds. The costs of this potentially unnecessary management can be explicitly estimated to help decide on appropriate thresholds, noting that a threshold of being 95 per cent confident is almost always too high for most environmental management decisions (Field et al. 2004).

For these reasons we recommend that seabird monitoring on the Reef take an appropriate precautionary approach favouring a lower threshold for declaring a statistically significant trend, such as greater than or equal to approximately 80 per cent statistical likelihood of decline, over the more standard 95 per cent significance threshold. This has the effect of favouring action over statistical certainty.

3.0 Adequacy of current monitoring and modelling

We evaluated statistical power to detect trends in abundance of the four focal seabird species (crested tern, brown booby, wedge-tailed shearwater, and little tern) at essential sites within the Marine Park as outlined in Hemson et al. 2015. As previously described, the current monitoring scheme relies on counts of breeding pairs for assessing the status and trends of populations and thus, requires that the surveys align with timing of breeding. For some species and sites (for example, Michaelmas Cay), timing of breeding is well known, but the breeding phenology of many species, at many of the essential sites, is less well understood. Thus, when evaluating the effectiveness of the monitoring scheme to detect trends in seabird abundance, we needed to account for uncertainty in breeding phenology and variability in survey effort across seasons. To do so, we adopted a four-step procedure that involved simulating many replicate datasets of inter-annual variability in breeding phenology and survey effort, as well as time-series of *true* numbers of breeding pairs of different lengths (in years) and magnitudes of decline. These three simulated datasets were then combined to produce time-series of *observed* numbers of breeding pairs, which were then analysed alongside time-series of *true* numbers of breeding pairs to determine power to detect temporal trends and the extent to which uncertainty in breeding phenology and variability in survey effort reduced power to detect trends. Detailed explanations of the four analysis steps are provided below. All data operations, analysis, and simulations were conducted in R 3.5.0 (R Core Team 2018).

3.1 General methods for power analysis and simulations

3.1.1 Breeding phenology

We extracted data on timing and duration of breeding of each of the four focal species from published studies, local expert knowledge and, where possible, historical Reef count data, to parameterise simulations of inter-annual variability in breeding phenology. This information included timing of egg-laying, length of the incubation period, and length of the nestling period (egg hatch to fledging), and nest survival to fledging. We used published studies whose focus was to measure different aspects of the breeding biology, namely timing of breeding, duration of the breeding cycle, and reproductive success, as the primary data source of for parameterising breeding phenology simulations for two main reasons. First, historical monitoring data are generalised (i.e. the focus was on counting breeding pairs, not detailed monitoring of breeding events), as well sparse for a majority of sites and species.

Second, surveys have historically been biased in space and time, including rather few surveys outside optimal breeding times for many of the key species, rendering a full analysis of timing of breeding peaks problematic. Data extracted from published studies are summarised in Appendix C.

For each species we simulated time-series of high and low success breeding seasons with equal probability (Figure 3.1.1). Under both scenarios, breeding pairs accumulated at a site following a cumulative exponential distribution function for several weeks until the peak number of breeding pairs was reached at the end of egg laying and beginning of incubation (blue line in Figure 3.1.1). After peak breeding, the proportion of breeding pairs underwent sigmoidal decline until the end of the breeding season when all young were assumed to have fledged or died. In scenarios of high breeding succession, proportion of breeding pairs declined following a sigmoidal growth curve with alpha = 0.005 and beta = 8 (Fig. 3.1.1a), whereas in scenarios of lower breeding success, proportion of breeding pairs declined following a sigmoidal growth curve with alpha = 0.005 and beta = 1.5 (Figure 3.1.1b). Proportions present at the end of the breeding season were based on high and low estimates of fledging success from published studies. Although the shape of the breeding curve could alternate between high and low success among years, we assumed that each species' breeding season length was constant (i.e. there are no stochastic events, such as cyclones or large predation events, that drastically alter the duration of the breeding season).

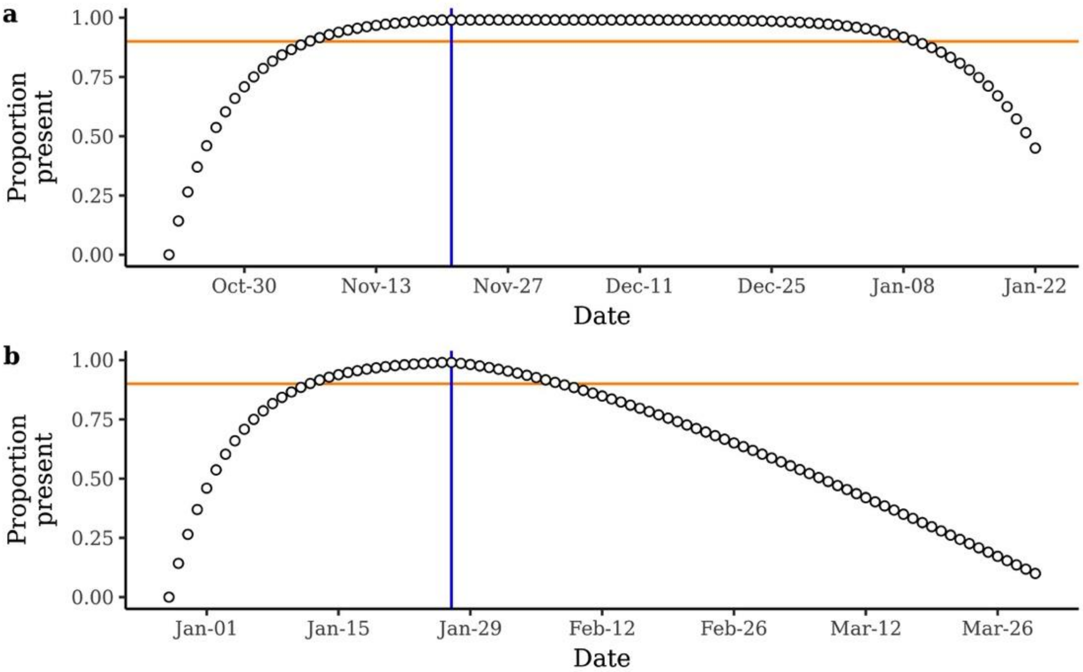

Figure 3.1.1. Examples of the proportion of the breeding pairs of crested terns present (and thus observable) over time for seasons of high (a) and low (b) breeding success. Peak abundance of breeding pairs (blue line) is expected near the end of egg-laying and beginning of incubation. In simulations, seasons of high and low breeding success had an equal probability of occurrence in each year. The orange line indicates 90 per cent of the population.

3.1.2 Survey scheme

Time-series of site visits were simulated based on the most common monitoring scheme for essential sites for each of the four focal species (outlined in Hemson et al. 2015 and summarised in Section 1.5). For crested terns and brown boobies, which overlap across much of their distributions, we simulated two site visits, with the first site visit occurring in November and the second site visit occurring two months later. For wedge-tailed shearwaters, we simulated a single survey in December of each year. For little terns, we simulated two site visits, with the first site visit in November and the second occurring one month later. In all simulations, the first survey could occur on any day within the first month with equal probability, whereas the second site visit occurred exactly 60 days (crested terns and brown boobies) or 30 days (little terns) after the first.

3.1.3 Breeding abundance

Annual numbers of breeding pairs were simulated for all combinations of six time-series lengths (five to 30 years, in five year intervals), three levels of decline [based on the IUCN 2012 criteria for *Critically Endangered*, *Endangered*, and *Vulnerable*], and a range of species-specific initial population sizes representative of individual island populations. Population declines were simulated following an exponential model of population growth:

$$N_t = N_0 * e^{rt} \tag{1}$$

In equation 1, N_t is the population size at time t, N_0 is the initial population size, r is the exponential rate of increase, and e is the mathematical constant equal to 2.78128. From the equation 1, the average annual rate of increase r was derived using the following equation:

$$r = [\log_e(N_t / N_0)] / t \tag{2}$$

Declines were simulated over three generation lengths at levels of 30%, 50%, and 80% in concert with IUCN 2012 criteria for *Vulnerable*, *Endangered*, and *Critically Endangered* threat statuses, respectively (Table 2.1). We introduced inter-annual variation in numbers of breeding pairs following a negative binomial distribution:

$$\begin{gathered} N_t \sim NegativeBinomial(\lambda_t, k) \\ \log_e(\lambda_t) = N_0 + rt \end{gathered} \tag{3}$$

In equation 3, k is the dispersion parameter, whose size relative to the expected population size (λ_t) determines the level of inter-annual variability in breeding pairs. For crested terns and brown boobies, we simulated inter-annual variability based on the observed range and variability of breeding pairs at selected well-monitored essential sites. For little terns and wedge-tailed shearwaters, data on the range and inter-annual variability of population sizes are sparse, so we simulated and analysed time-series of breeding pairs ranging from low to high levels of inter-annual variability.

3.1.4 Power analysis

We evaluated power to detect trends from a time-series of both *true* and *observed* numbers of breeding pairs. By comparing results from analyses of true and observed numbers of breeding pairs, we could evaluate differences in power attributable to the observation process; specifically, the extent to which mismatches between peak breeding and survey timing reduce power to detect trends. For trend estimation, we fitted generalised linear models with a single fixed effect for year and a negative binomial error structure. Time-series of true numbers of breeding pairs were those generated in Section *3.1.3.* Time series of observed numbers of breeding pairs, were generated by multiplying true numbers of breeding pairs by the proportions observed (as simulated in *3.1.2*). For simulations where two or more surveys were conducted in a single breeding season, we analysed the higher of the two counts.

A consequence of using breeding pairs as a measure of population size is that inter-annual variation in observed counts can reflect changes in population size due to inter-annual variation in multiple factors, including survival and recruitment, breeding participation, and timing of surveys relative to the peak breeding season. Assuming population closure, the only true zero that arises is due to local extinction, whereas zeroes due to low breeding participation and mistimed surveys represent non-detections. To account for non-detections and improve model fit, counts of zero breeding pairs were treated as missing values rather than true zeroes.

In all subsequent results and figures, power is defined as the proportion of models where the slope estimate for the year effect on numbers of breeding pairs was both negative AND statistically significant ($\alpha = 0.2$).

3.2 Species-specific methods and results

3.2.1 Crested terns

Breeding phenology

For simulations of breeding phenology of crested terns, we specified the total length of the breeding season as the sum of the pre-laying (30 days), incubation (25 days), and nestling to fledging (39 days) periods (Appendix C). Survival to fledging was specified as 0.45 for years of typical breeding success and 0.1 for years of low reproductive success. High and low breeding success were assigned equal probability of occurrence. We assumed that breeding participation was 100 per cent in each year and peak breeding was constrained to between the 15 November and the 15 February (Appendix C).

Surveys

Following the current monitoring scheme, we simulated two surveys per year with the first survey occurring in November and the second survey occurring two months afterwards. Simulations revealed that, at sites where peak numbers of breeding pairs occur between January and March, such as Michaelmas Cay (Fig. C1a), visits in November are likely to miss the breeding peak (Fig. C2). However, when two surveys are conducted per season, at least one of the two is likely to capture a high proportion of the population.

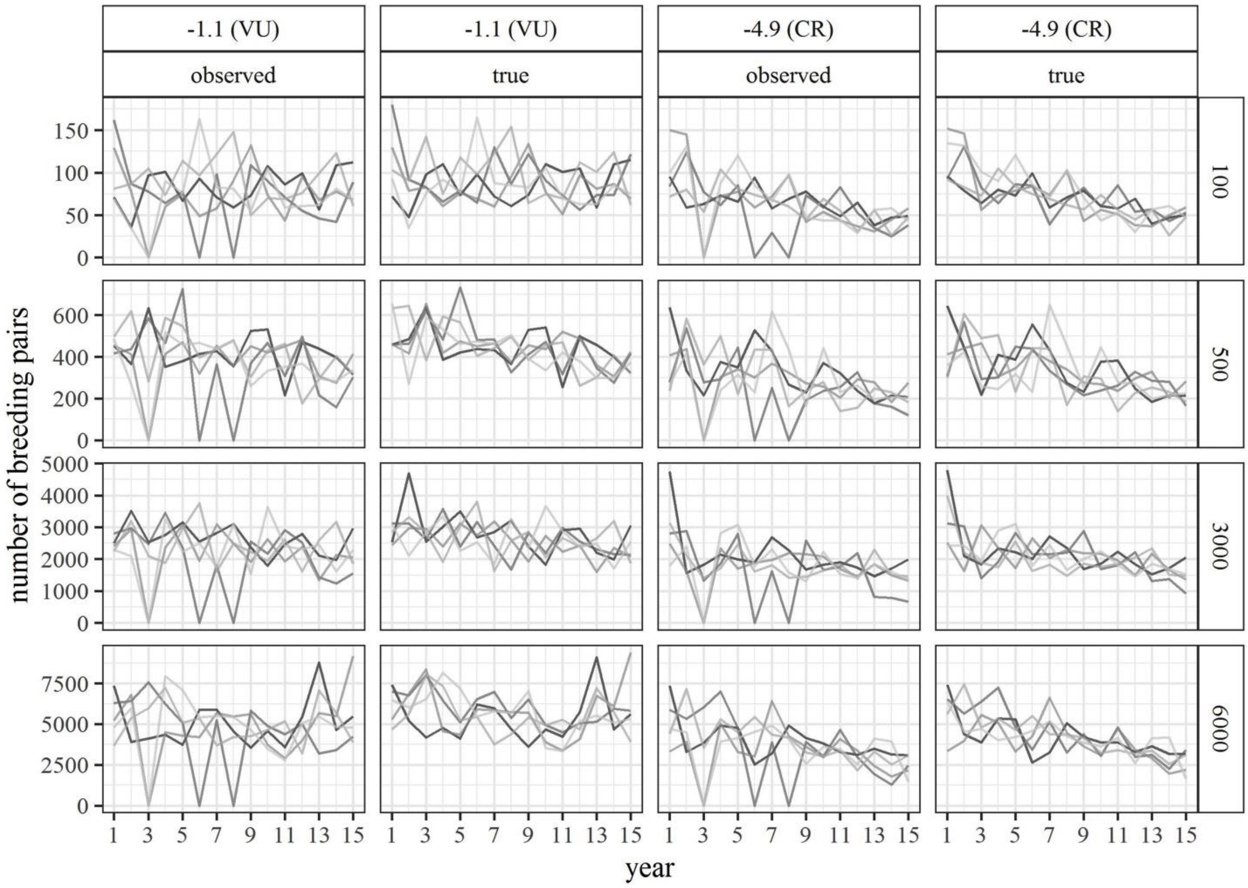

Figure C1. **(a) Monthly and (b) annual maximum counts of breeding pairs of crested terns at Michaelmas Cay where monthly monitoring has taken place since 1983.** Only years where surveys were conducted in at ≥ 9/12 months are shown. (a) Monthly data show that peak numbers of breeding pairs occur in February at Michaelmas. The regression line and 95 per cent confidence interval was estimated from a local polynomial regression implemented using the 'loess' function in R with a smoothing parameter (or span) equal to 0.7. (b) Annual maximum counts of breeding pairs at Michaelmas ranged from 835 to 6165 and averaged 3050.

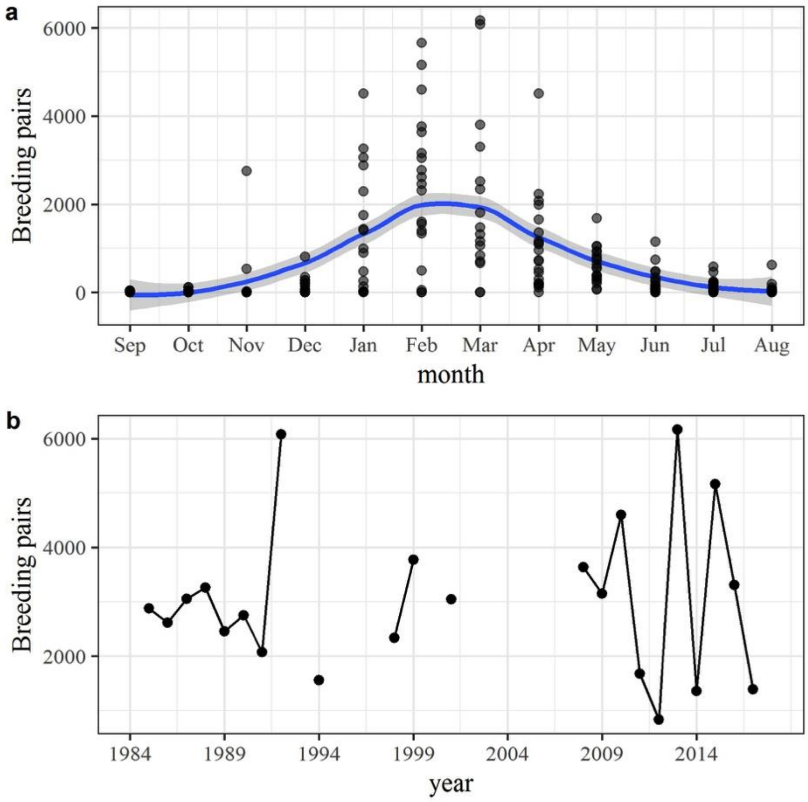

Figure C2. Simulated distribution of the proportion of breeding pairs observed during each of two surveys as well as the distribution when the *best* of the two surveys was taken.

Abundance

We simulated time-series of abundance of breeding pairs with initial population sizes ranging from 100 to 6000 breeding pairs; these upper and lower bounds approximated the minimum and upper 95 per cent of maximum annual breeding pairs observed at essential sites since 1983. Based on three generation lengths equal to 32 years (Table 2.2), we simulated average annual per cent declines for crested terns of 1.1 per cent for *Vulnerable*, 2.1 per cent for *Endangered*, and 4.9 per cent for *Critically Endangered* (Figure C3). Inter-annual variability was introduced using a negative binomial distribution with a dispersion parameter of 20. Coefficients of variation for true and observed numbers of breeding pairs over 30 years with an initial population size of 3,000 and average per cent declines of 1.1 per cent averaged 0.24 and 0.39, respectively (Figure C3). By comparison, the coefficient of variation for annual maximum counts of breeding pairs of crested terns at Michaelmas Cay (excluding years where surveys were conducted in fewer than nine months; Figure C1b) was 0.47, suggesting that simulated levels of inter-annual variability were similar to, if not slightly lower than, actual inter-annual variability.

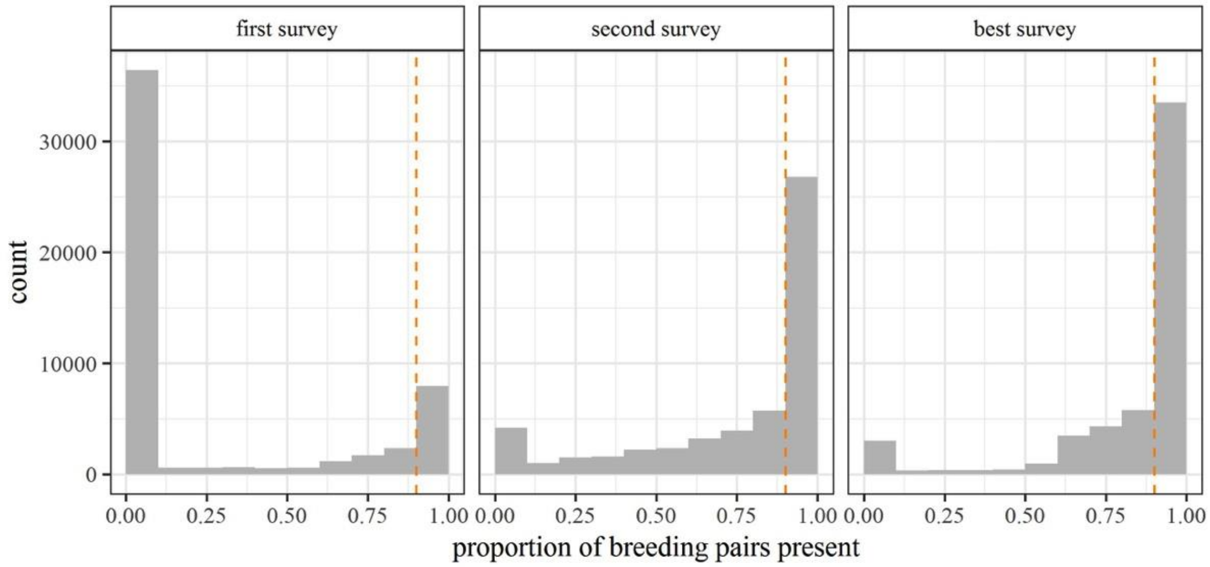

Figure C3. Five simulated 15-year time series of breeding pairs of crested terns undergoing average annual declines that meet the IUCN criteria of *Vulnerable* (-1.1 per cent per year) and *Critically Endangered* (-4.9 per cent per year). For each initial population size and decline level, time-series of both true and observed numbers of breeding pairs are shown. Time-series were simulated for initial population sizes ranging from 100 to 6000 breeding pairs.

Power

Analysis of simulated datasets show increasing power to detect site-level trends with increasing time-series length and magnitude of decline (Figure C4a). Power losses attributable to uncertainty in breeding phenology and the observation process ranged from zero to 20 per cent and were highest for long time-series and low levels of decline and short time-series and moderate to levels of declines (Figure C4b). Power losses due to the survey process were generally negligible when time-series were long (over 20 years) and declines were steep (greater than or equal to 4.9 per cent, per year).

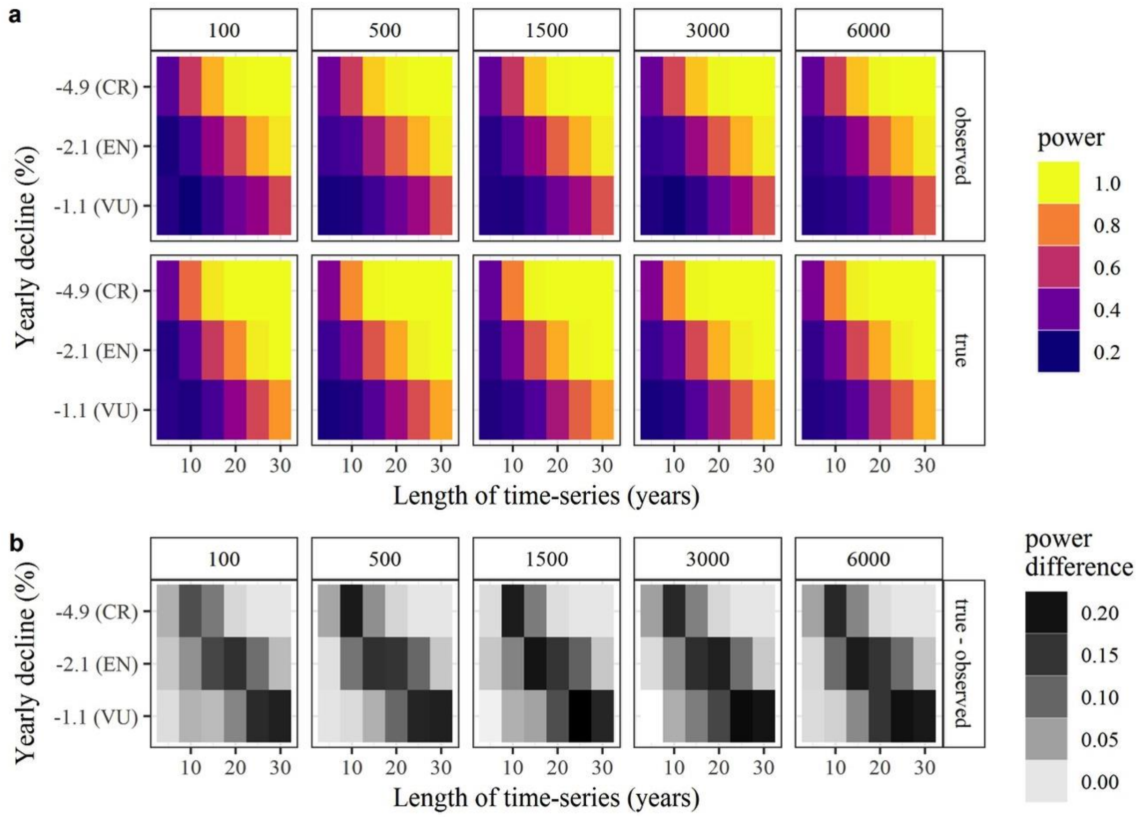

Figure C4. (a) Power to detect trends in numbers of breeding pairs of crested terns in relation to time-series length, initial population size, and extent of decline. (b) Difference in power to detect trends from time-series of true and observed numbers of breeding pairs.

3.2.2 Brown booby

Breeding phenology

For simulations of breeding phenology of brown boobies, we specified the total length of the breeding season as the sum of the pre-laying (30 days), incubation (43 days), and nestling to fledging (107 days) periods (Appendix C). Survival to fledging was specified as 0.58 for years of typical breeding success and 0.1 for years of low reproductive success. High and low breeding success were assigned equal probability of occurrence. Due to the tendency for this species to breed throughout the year, we specified breeding participation of 75 per cent for the summer breeding period when most surveys occur. Peak breeding was constrained to between the 1 September and the 31 December (Figure B1).

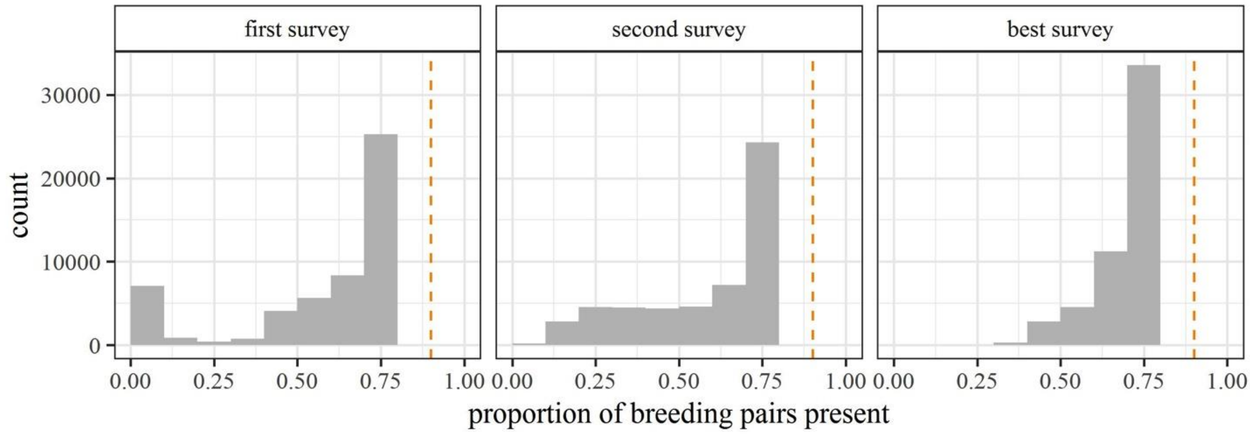

Figure B1. Data showing counts of breeding pairs of brown boobies in relation to calendar month at Michaelmas Cay (where monthly monitoring has occurred since 1983), as well as three sites in the Great Barrier Reef (Raine Island, East Fairfax Island, and Sandbank Number 8) where brown boobies are abundant.

Surveys

Following the current monitoring scheme, we simulated two surveys per year with the first survey occurring in November and the second survey occurring two months afterwards. Because we specified a breeding participation of 75 per cent for the summer breeding period, the maximum proportion of the population observable on any given survey was 75 per cent (Figure B2). Simulations revealed that, provided the timing of breeding is reasonably well known, the majority of the breeding population is likely to be observed on both the first and second surveys due to the long duration of the breeding season (approximately 170 days).

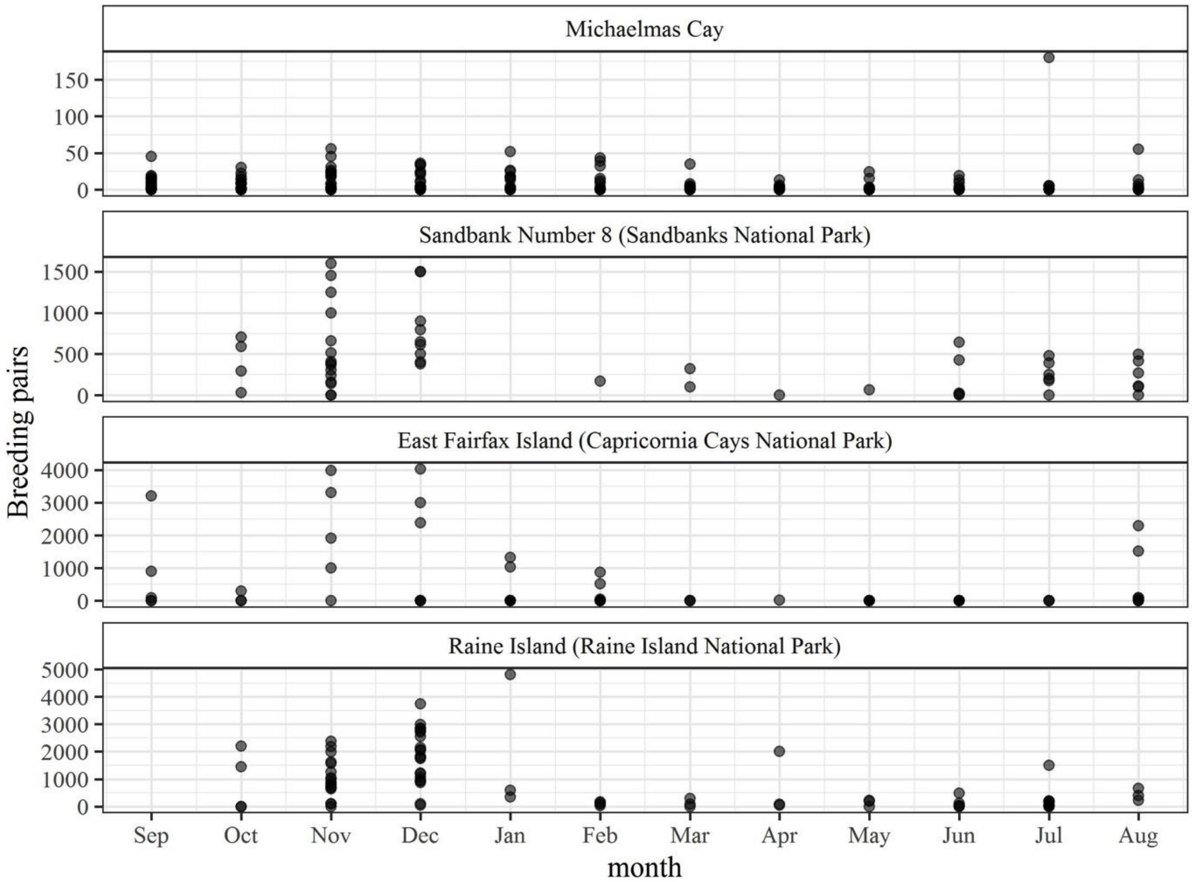

Figure B2. Simulated distribution of the proportion of breeding pairs of brown boobies observed during each of two surveys as well as the distribution when the best of the two surveys was taken.

Abundance

We simulated time-series of abundance of breeding pairs with initial population sizes ranging from 50 to 4500 breeding pairs; these upper and lower bounds approximated the range of breeding populations at four observed at four essential sites on the Reef since 2012 (Figure B3). Based on three generation lengths equal to 32 years (Table 2.2), we simulated average annual per cent declines for crested terns of 0.7 per cent for *Vulnerable*, 1.3 per cent for *Endangered*, and 3.1 per cent for *Critically Endangered* (Figure B4). Inter-annual variability was introduced using a negative binomial distribution with a dispersion parameter of 20. Coefficients of variation for true and observed numbers of breeding pairs over 10 years with average per cent declines of 0.7 per cent averaged 0.23 and 0.26, respectively. By comparison, the coefficient of variation for annual maximum counts of breeding pairs of brown boobies since 2012 at the four sites in Figure B3 averaged 0.36 (range = 0.31-0.47), suggesting that simulated levels of inter-annual variability approximated, if not slightly underestimated, actual inter-annual variability.

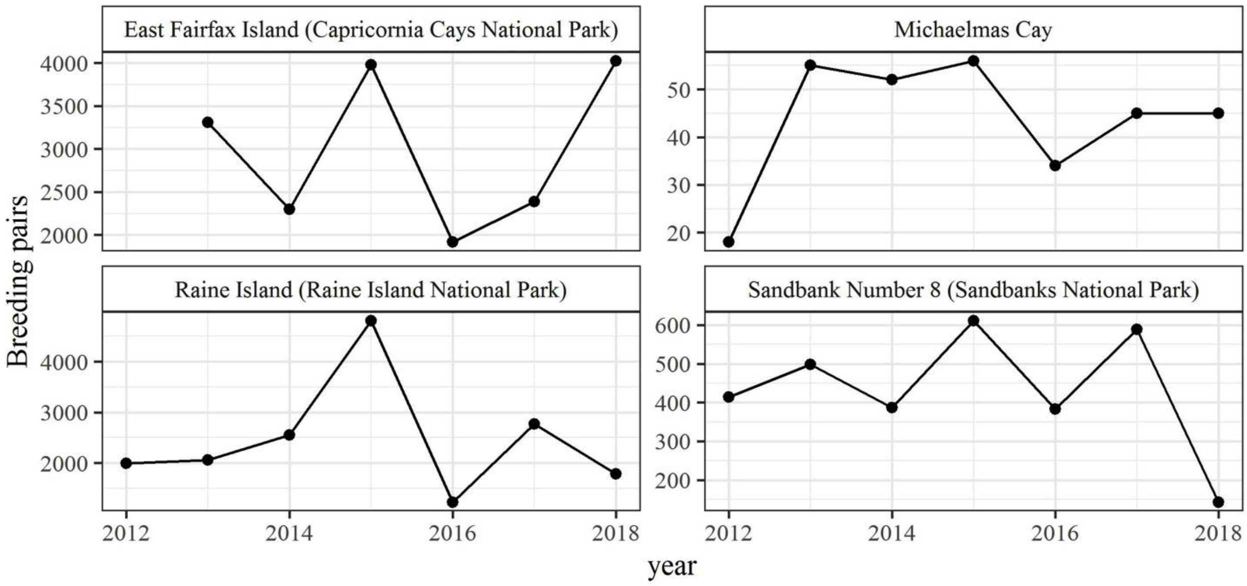

Figure B3. Annual maximum counts of breeding pairs of brown boobies at Michaelmas Cay and three other essential sites where the species is most abundant since 2012.

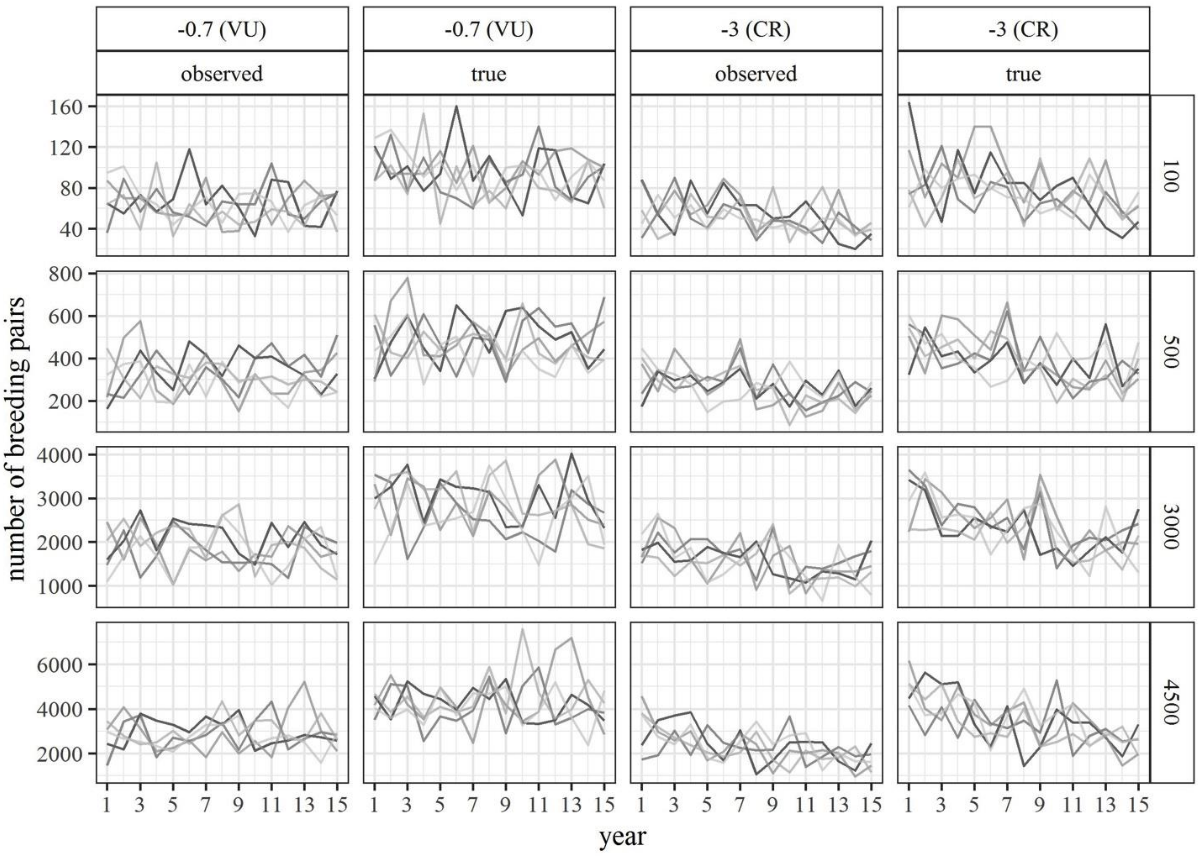

Figure B4. Five simulated 15-year time-series of breeding pairs of brown boobies undergoing average annual declines that meet the IUCN criteria of *Vulnerable* (-0.7 per cent per year) and *Critically Endangered* (-3.0 per cent per year) over three generations. For each initial population size and decline level, time-series of both true and observed numbers of breeding pairs are shown. Inter-annual variability was introduced using a negative binomial distribution with a dispersion parameter of 20. Time-series were simulated for initial population sizes ranging from 50 to 4500 breeding pairs.

Power

As with crested terns, analysis of simulated datasets show increasing power to detect site-level trends with increasing time-series length and magnitude of decline (Figure B5a). Power losses attributable to uncertainty in breeding phenology and the observation process ranged from zero to nine per cent and were highest for long time-series and low levels of decline and short time-series and moderate levels of decline (Figure C4b). The lower decline threshold required for brown boobies to reach *Vulnerable*, *Endangered*, or *Critically Endangered* status resulted in a lower average power to detect trends compared to crested terns for equivalent time-series lengths.

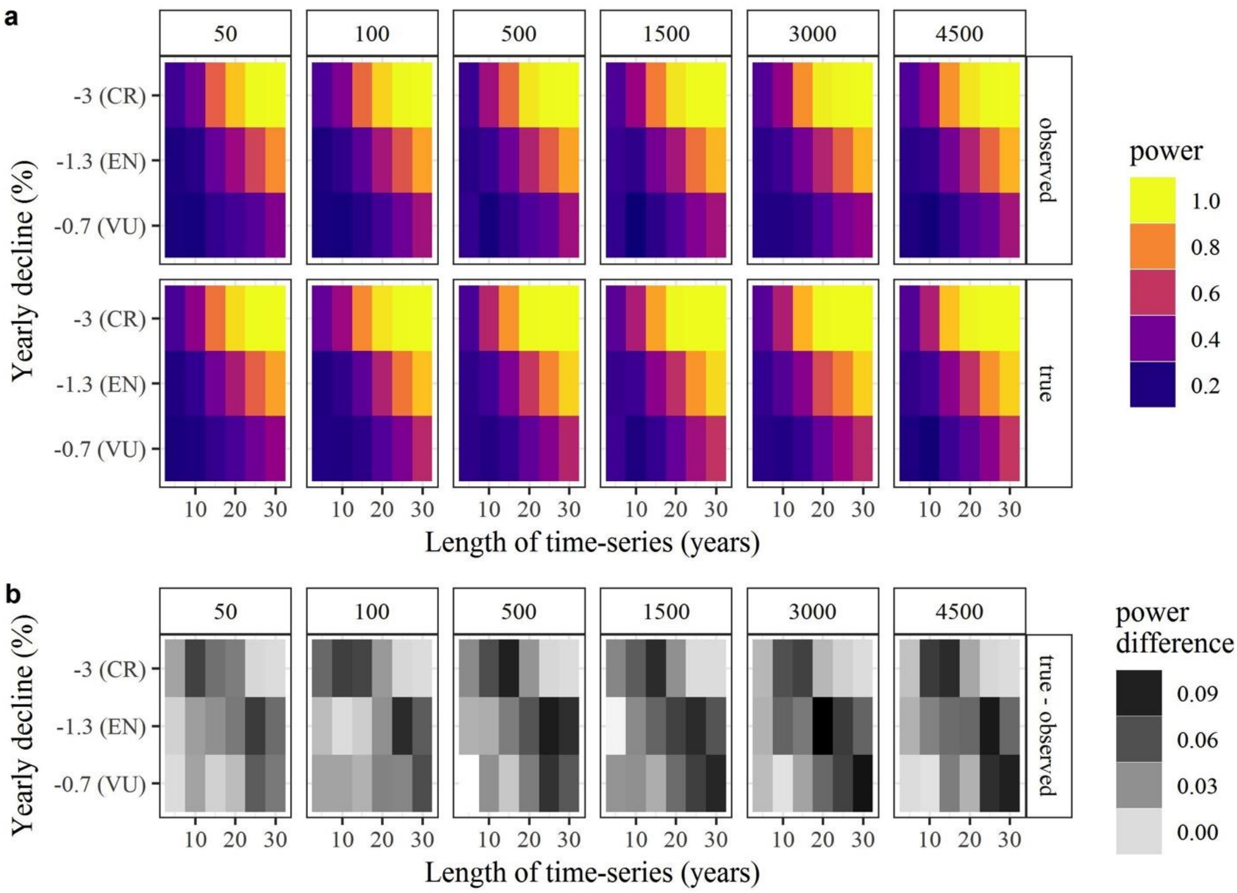

Figure B5. Power to detect trends in numbers of breeding pairs of brown boobies in relation to time-series length, initial population size, and extent of decline. (a) Power to detect trends from

time-series of true and observed numbers of breeding pairs. (b) Difference in power to detect trends in time-series of true and observed numbers of breeding pairs.

3.2.3 Little tern

Breeding phenology

For simulations of breeding phenology of little terns, we specified the total length of the breeding season as the sum of the pre-laying (assumed to be 14 days), incubation (22 days), and nestling to fledging (22 days) periods. Lengths of the incubation and nestling periods represent the approximate midpoints reported by Gochfeld et al. 2018. Due to a lack of data on nest success for little terns, we used the same values for survival to fledging for years of typical breeding success (0.45) and low reproductive success (0.1) as for crested terns. High and low breeding success were assigned equal probability of occurrence. We assumed that breeding participation was 100 per cent in each year and peak breeding was constrained between the 1 November and the 31 December based on the observed distribution of breeding pairs by month (Figure L1). For many sites, particularly those at the northern end of the region where breeding may occur throughout the year, a two-month breeding window likely under-represents variance in breeding phenology.

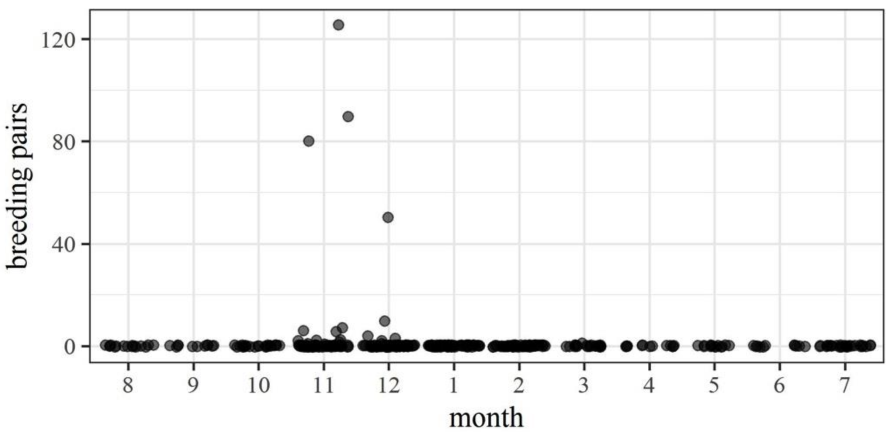

Figure L1. Counts of maximum numbers of breeding pairs of little terns by month at essential sites on the Great Barrier Reef since 1980. One record of 1000 breeding pairs is not shown because such a high count seems unlikely for this species.

Surveys

Following the current monitoring scheme, we simulated two surveys per year with the first survey occurring in November and the second survey occurring one month afterwards. The results show that due to the short breeding season and high variability in timing of peak breeding that two surveys are necessary to ensure that the majority of breeding pairs are detected on at least one of two surveys (Fig. L2). That 90 per cent or greater of breeding pairs were detected on at least one of two surveys may be optimistic for northern reef sites and elsewhere where little terns may breed throughout the year, as opposed to only November/December as specified here.

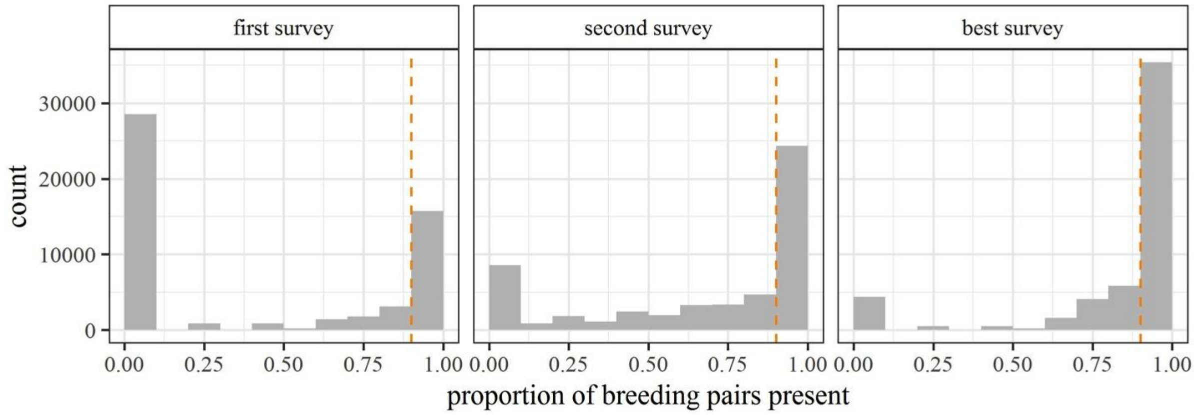

Figure L2. Simulated distribution of the proportion of breeding pairs observed during each of two simulated surveys as well as the distribution when the *best* of the two surveys was taken.

Abundance

We simulated time-series of abundance of breeding pairs with initial population sizes ranging from 10 to 125 breeding pairs; these upper and lower bounds approximated the range of numbers of breeding pairs at essential sites on the Reef since 1980 (Figure L2). Based on three generation lengths equal to 33 years (Table 2.2), average annual per cent declines for little terns were 1.1 per cent for *Vulnerable*, 2.1 per cent for *Endangered*, and 4.8 per cent for *Critically Endangered*. As with brown boobies and crested terns, inter-annual variability was introduced using a negative binomial distribution with a dispersion parameter of 20 (Figure L3). However, due to tendencies for low site fidelity in this species and insufficient historical count data for comparing simulated and actual levels of variation for this species, we also simulated and analysed time-series of true numbers of breeding pairs with three additional levels of inter-annual variability (dispersion parameters = 2, 5, 10; Figure L4).

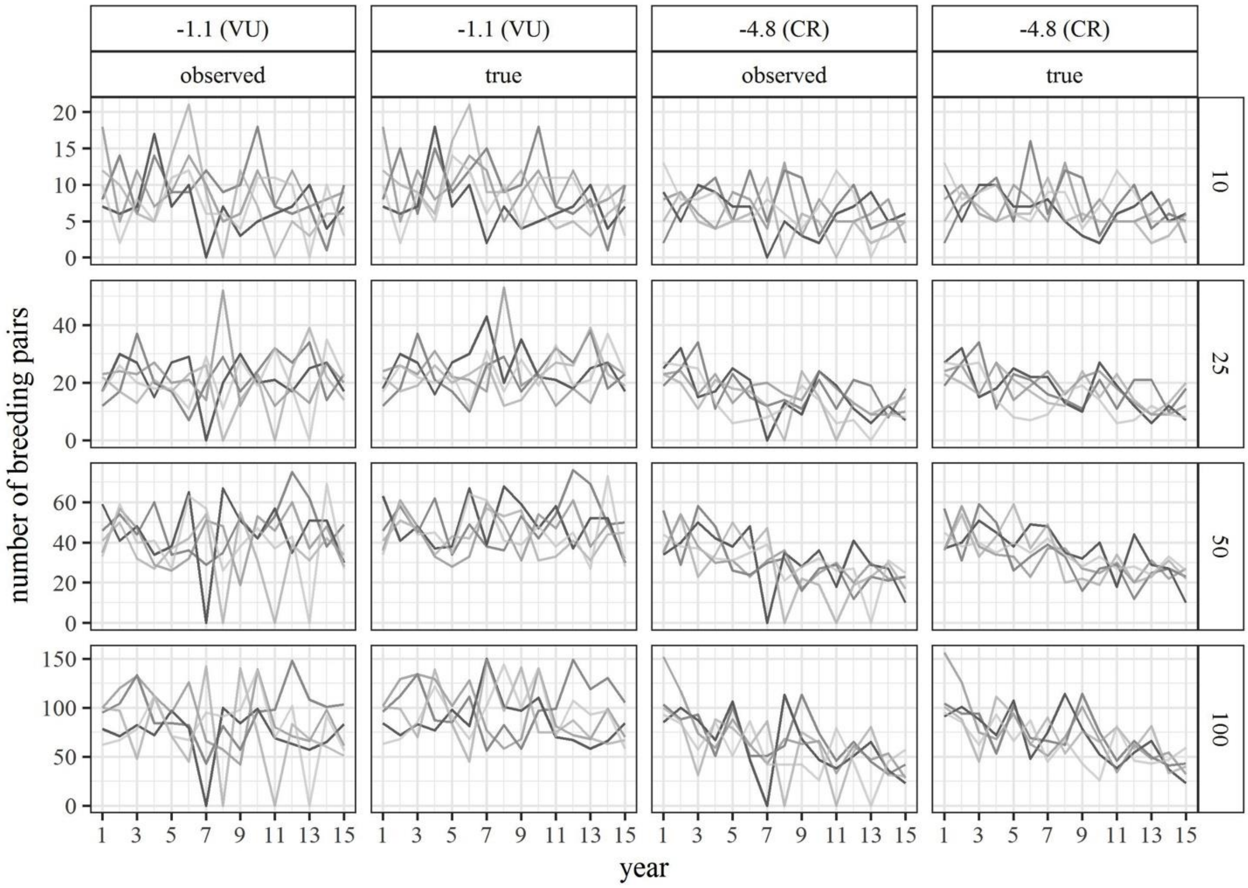

Figure L3. Five simulated 15-year time-series of breeding pairs of little terns undergoing average annual declines consistent with the IUCN 2012 population reduction criteria for *Vulnerable* (-1.1 per cent per year) and *Critically Endangered* (-4.8 per cent per year) threat statuses over three generations. For each initial population size and decline level, time-series of both true and observed numbers of breeding pairs are shown. Inter-annual variability was simulated using a negative binomial distribution with a dispersion parameter of 20.

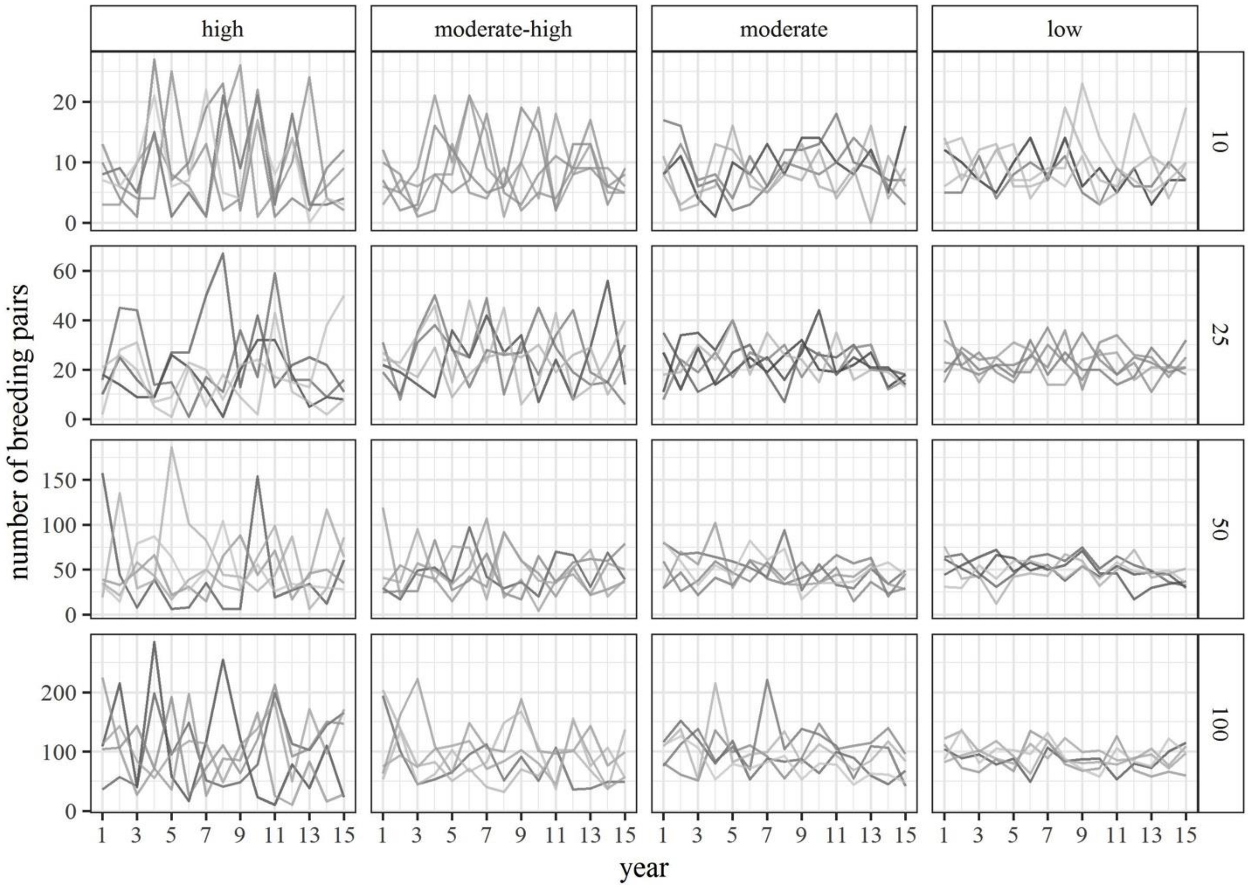

Figure L4. Five simulated 15-year time-series of breeding pairs of little terns undergoing average annual declines that meet IUCN 2012 population reduction criteria for *Vulnerable* (-1.1 per cent per year) threat status over three generations. For each initial population size and decline level, time-series of true numbers of breeding pairs are shown under different levels of inter-annual variability (high [k = 2], moderate-high [k = 5], moderate [k = 10], and low [k = 20]).

Power

Figure L5a shows power to detect trends from time-series of observed and true numbers of breeding pairs when inter-annual variability in population size is low (Figure L2). At this level of inter-annual variability, power losses attributable to uncertainty in breeding phenology and the observation process ranged from zero to 10 per cent and were highest for long time-series and low levels of decline and short time-series and moderate to levels of declines

(Figure L5b). When inter-annual variability in population size was high, we found reductions in power in excess of 40 per cent relative to simulations when variability was low (Figure L6).

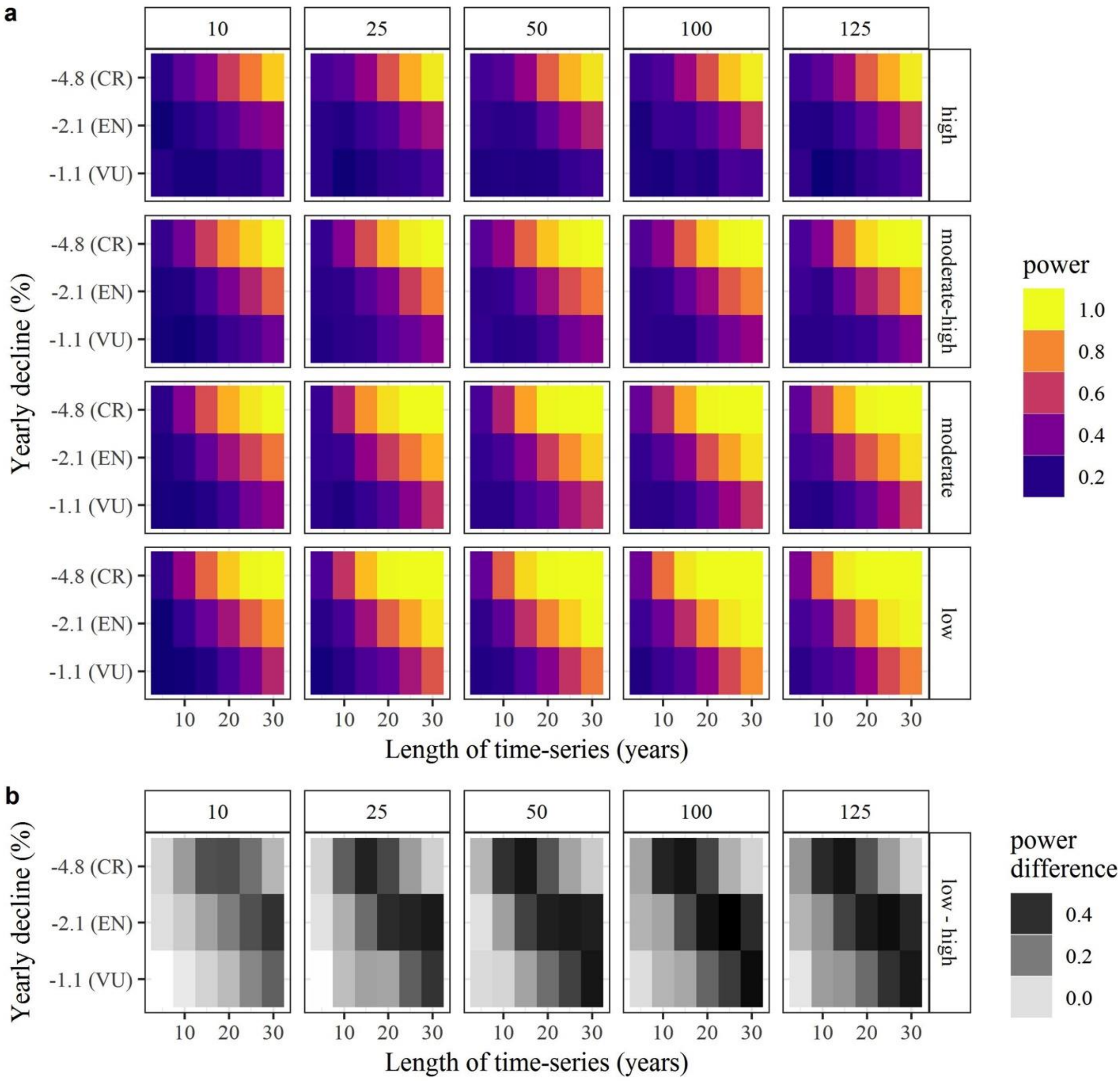

Figure L5. Power to detect trends in numbers of breeding pairs of little terns in relation to time-series length, initial population size, and extent of decline. (a) Power to detect trends from time-series of true and observed numbers of breeding pairs with low inter-annual variability (dispersion = 20). (b) Difference in power to detect trends from time-series of true and observed numbers of breeding pairs.

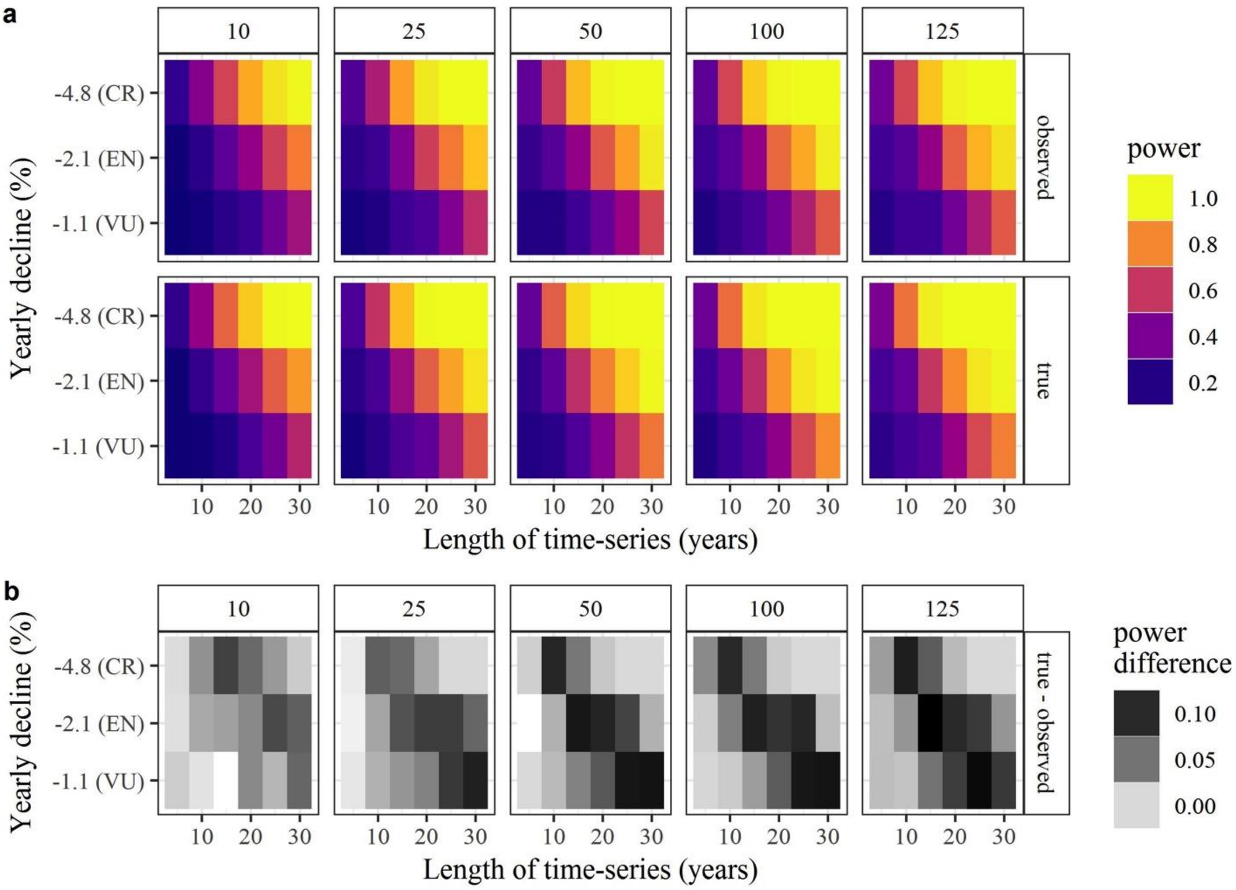

Figure L6. (**a) Power to detect trends in numbers of breeding pairs of little terns in relation to time-series length, initial population size, magnitude of declines, and different levels of inter-annual variability in population size. Variability in Fig L5 is equivalent to low variability here. (b) Difference in power to detect trends in time-series of true numbers of breeding pairs between low and high variability.**

3.2.4 Wedge-tailed shearwater

Breeding phenology

For simulations of breeding phenology of little terns, we specified the total length of the breeding season as the sum of the pre-laying (60 days), incubation (53 days), and nestling to fledging (98 days) periods (Appendix C). Survival to fledging was specified as 0.52 for years of typical breeding success and 0.1 for years of low reproductive success. High and low breeding success were assigned equal probability of occurrence. We assumed that breeding participation was 100 per cent in each year and peak breeding was constrained to within 14 days on either side of the 15 January.

Surveys

Following the current monitoring scheme, we simulated a single survey between the 15 December and the 31 January. Higher certainty about the timing of breeding for this species and well-timed surveys resulted in 90 per cent or more of breeding pairs being detected by a single survey on nearly every occasion (Figure W1).

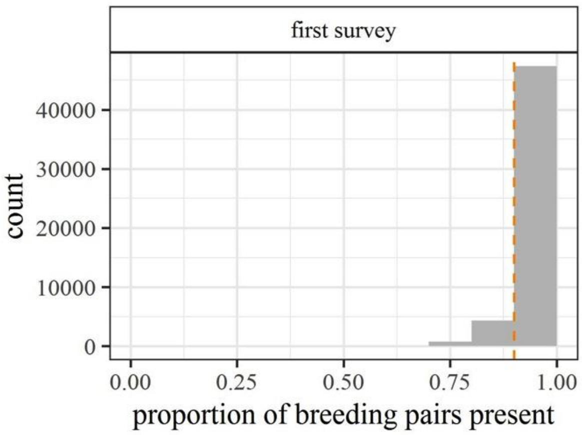

Figure W1. Simulated distribution of the proportion of breeding pairs of wedge-tailed shearwaters observed on a single survey representative of a scenario where timing of breeding is well understood.

Abundance

We simulated breeding pairs with initial population sizes that approximated the lower bounds of the population estimates for Heron Island (n = 10281), Lady Musgrave Island (n = 6196), Mast Head Island (n = 52,282), and North West Island (*n* = 162,808) from the Capricornia Cays Acoustic Experiment report from 2017 (Roberts and McKown 2018). Because inter-annual variability of wedge-tailed shearwaters are not available from the data, we could not compare actual and observed levels of population variability so instead simulated and analysed time-series with high (dispersion parameter k = 20), moderate (k = 40), and low (k = 80) dispersion. Furthermore, because there was nearly no observation error introduced by mistimed surveys or uncertainty in breeding phenology (Figure W1), we restricted our analyses to time-series of true population sizes only.

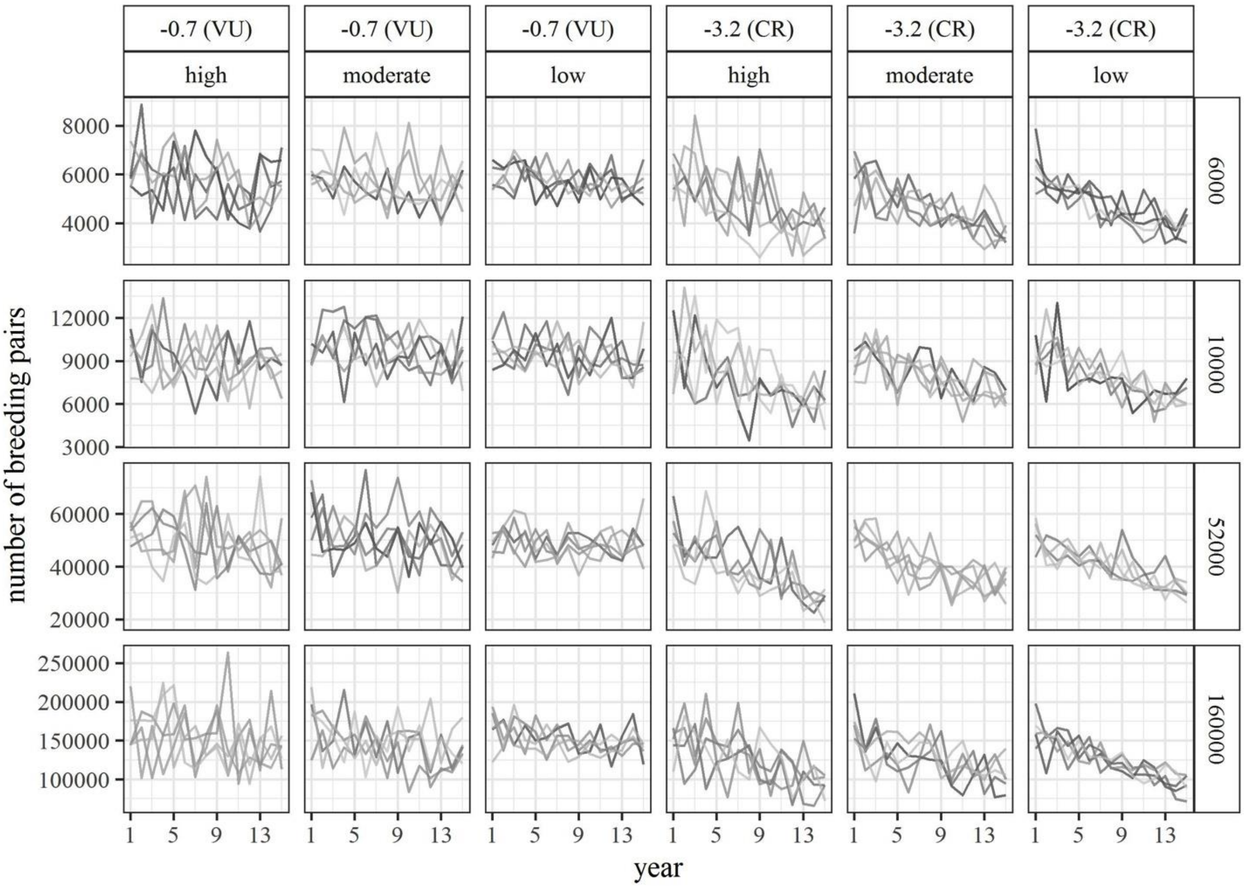

Figure W3. Five simulated 15-year time-series of breeding pairs of wedge-tailed shearwaters undergoing average annual declines that meet the IUCN criteria of *Vulnerable* (-0.7 per cent per year) and *Critically Endangered* (-3.2 per cent per year) over three generations. For each initial population size and decline level, time-series of true numbers of breeding pairs are shown under different levels of inter-annual variability (high, moderate, and low).

Power

Analysis of simulated datasets revealed overall high power to detect trends from time-series of true numbers of breeding pairs of wedge-tailed shearwaters. When inter-annual variability in population size was increased from low to high, power was reduced by up to 30 per cent (Figure L6). Further study of actual levels of variability will be required to validate simulated levels of inter-annual variability.

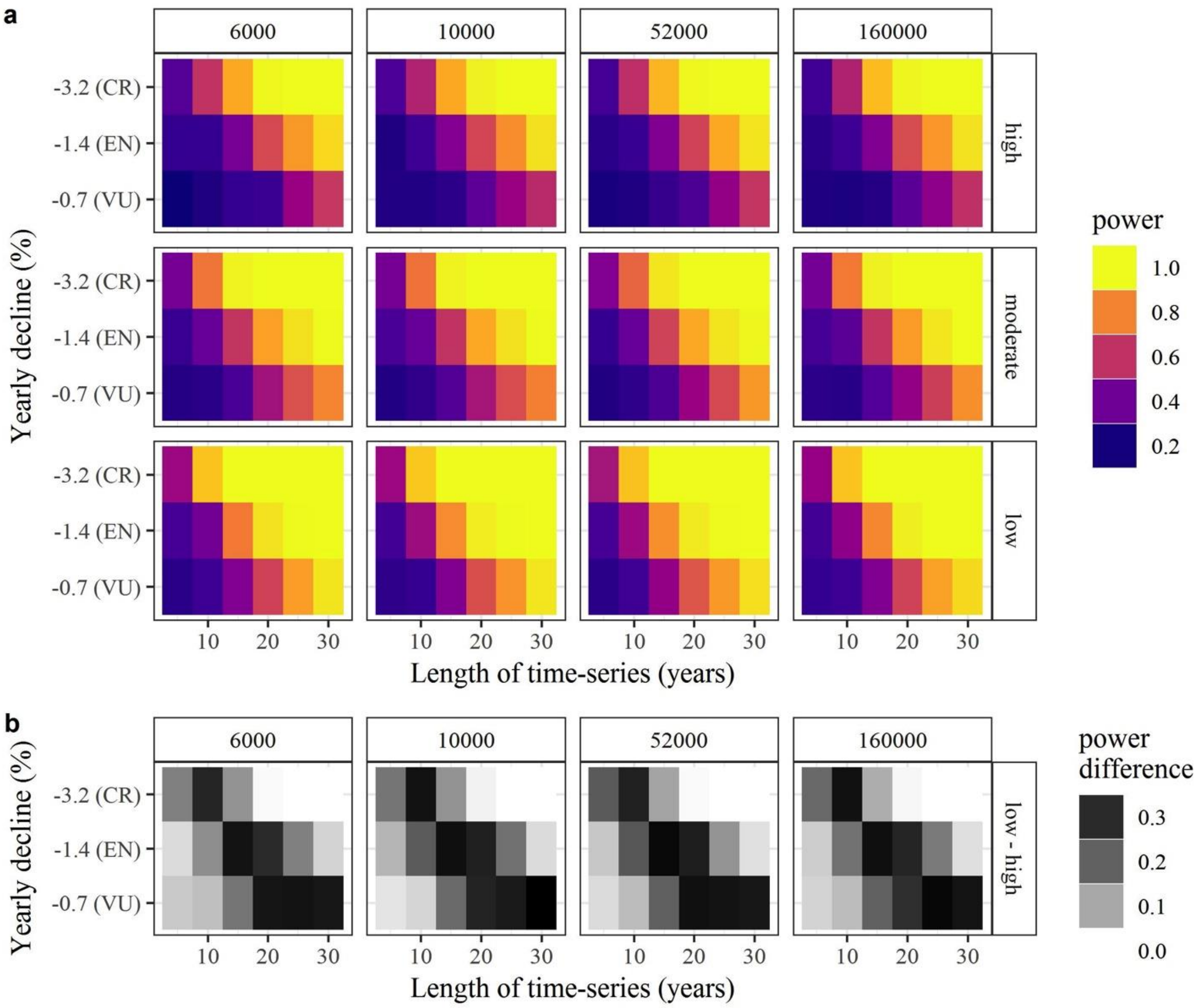

Figure W4. (**a) Power to detect trends in numbers of breeding pairs of wedge-tailed shearwaters in relation to time-series length, initial population size, magnitude of decline, and level of inter-annual population variability. (b) Difference in power to detect trends in time-series of true numbers of breeding pairs between low and high variability.**

4.0 Gaps in current monitoring and modelling of proposed indicators

4.1 What are the potential issues/problems with the current strategy?

Summarise any identified potential problems or issues with current strategy relative to the logistic constraints within which it was originally designed and/or based on the modelling results.

4.1.1 Issues from Section 1.3: Are the historic data compatibility with data obtained in the CBMIS-2015?

Generally historic and future data to be obtained with the CBMIS-2015 are compatible, but the lack of error measurement for each single-figure total breeding population estimates per annum is a problem. Because of this issue it is not possible to determine if current or previous methodologies are more or less accurate.

Obtaining this measure would help to remove noise in total population estimates per season due to observer error/bias and so significantly improve estimates of between-season variation in breeding participation. Improvements in estimating between-season variance in breeding pairs due to natural processes (for example, survival, reproduction, and breeding participation) vs. the observation process are essential for reducing the period of time over which the CBMIS-2015 protocol will be able to identify robust trends in breeding numbers.

A final consideration regarding existing historical datasets is that their utility for detection of current and future changes in populations will depend on whether large perturbations to the system have occurred during the sampling period. If large environmental shifts have occurred then historical population levels may not represent an appropriate baseline by which to assess current population status and trends.

4.1.2 Issues from Section 1.4: What are the currently monitored species/foraging guilds and rational?

As indicated previously, whether a focus on four indicator species combined with additional data on non-indicator species provide sufficient information across the range of different seabirds species breeding throughout the Reef depends on the level of overlap in life-history characteristics among indicator and non-indicator taxa. The current strategy assumes some level of overlap in breeding or non-breeding food resources including prey types/size classes and foraging locations, species-specific nesting habitat and/or direct susceptibility to other non-starvation associated causes of mortality. Whether such overlap actually occurs in these and other types of life-history characteristics or susceptibilities specifically for seabirds of the Reef is largely unknown.

4.1.3 Issues from Section 1.5.1: Monitoring at Essential sites

Michaelmas Cay is an important dataset against which potential environmental impacts and population declines at other less intensively monitored locations can be assessed. As such we believe it is an important component of the ongoing CBMIS-2015 strategy. However, the rationale for maintaining the intensity of sampling at Michaelmas Cay and the way that data from this site interface with the overall CBMIS-2015 strategy are not explicitly outlined in the CBMIS-2015. In the long term this has the potential to lead to a decoupling of these two interlinked sampling programs and the downscaling of sampling at Michaelmas Cay, which would significantly undermine the ability of the CBMIS-2015 to assess population trends throughout the Reef region.

4.1.4 Issues from Section 1.5.2: Monitoring of Non-indicator species

Independent analyses of breeding abundance of non-indicator species and comparison to data for indicator species are considered important to the overall CBMIS-2015 (*Hemson pers comm*), particularly given the current lack of understanding about life history and response overlap among taxa (Section 4.1.2).

Therefore, there are three potential issues associated with the monitoring of non-indicator species that need to be considered. These are;

1) That the requirement to obtain data on all non-indicator species during an essential site visit is not currently a prescribed component of the sampling strategy within the CBMIS-2015 program document.

2) That the importance of, and a process for, accurately recording zero sightings of a species at known breeding locations for both indicator and non-indicator species in such a way that this information is easily retrievable from the resulting database is not currently a documented component of the CBMIS-2015 strategy.

3) That the level to which survey timing built around breeding peaks for indicator species effectively captures breeding peaks of other non-indicator species has not been quantitatively assessed so that significant mismatches can be identified.

4.2 What are the weaknesses of the current index and what threatening processes are detectable using these indices?

4.2.1 A "Conceptual Model" for seabirds of the Great Barrier Reef

As part of this review process we have developed a comprehensive conceptual model linking different seabird life history components to critical resource requirements, measurable indices that are influenced by each of these critical resources, and to the threats that impact these same resources. This model is provided in detail in Appendix D.

This model can be used to identify weaknesses associated with using the currently monitored index (numbers of breeding pairs) and also provide information on the practical utility of monitoring other additional indices.

4.2.2 What are the weaknesses of the current index?

The CBMIS-2015 attempts to estimate the maximum numbers of active nests each season as a measure of *total breeding population*. Counts focused on estimating numbers of active nests will consistently underestimate absolute population size but as long as the non-breeding proportion of each population remains relatively constant among years it is possible to use *total breeding population* to undertake the robust relative comparisons required for accurate trend analyses. However, for many seabirds, the numbers of adults that attempt to breed in any given season can be directly linked to the availability of pre-breeding resources.

This implies that much of the inter-annual variation observed using the current index may relate only to changes in the ratio of non-breeding to breeding adults in the populations. An inability to account for or estimate variation due to this phenomenon using the current index, means that considerably longer data series are required to observe statistically significant trends. The use of additional indices is needed to overcome this challenge.

As indicated previously, a further potential shortcoming of the current monitoring strategy and/or existing data is the lack of any systematic attempt to measure within-season error associated with the estimates obtained. Because of this it is difficult to estimate the level of inter-annual variation attributable to observer error(s) and uncertainty in counts of breeding pairs. Levels of uncertainty could be recorded as part of the existing monitoring protocol with minimal additional effort and cost, thereby allowing better separation of natural population variability from variability due the observation process, leading to improved trend detectability.

In general, it is the magnitude of inherent year-to-year variation that makes estimating population trends difficult and generates the need for long monitoring programs. Therefore, the power of any monitoring program can be substantially increased and its length considerably reduced if the level of year-to-year variation not associated with real population changes can be kept to a minimum.

4.2.3 What threatening processes are detectable using the current monitoring?

The CBMIS-2015 monitors a single seabird population index in isolation. This means that data collected by the current monitoring strategy on its own will be useful for establishing that a decline of a particular magnitude has occurred over a particular sampling period. However, identifying links between observed population decline(s) and known or anticipated threatening processes requires either:

1) Correlating long-term trends in seabird population size, reproduction, or survival with equivalent long-term data sets on variation in specific environmental or anthropogenic factors. Correlative analyses of this type are also the only way to search for previously unidentified or unanticipated effects.
2) Focused intensive studies using experimental designs developed specifically to detect the influences/impacts of specific environmental or anthropogenic drivers of change. This is really only applicable where the possible impacts of specific factors have already been highlighted by long-term correlation analysis or through comparisons with known impacts in other systems.

Therefore, the utility of measures of *total breeding population* or *breeding participation* as obtained by the CBMIS-2015 is that they can be correlated with environmental data sets obtained over similar time periods to develop testable hypotheses on the potential mechanisms driving any observed declines. Consequently, this type of index is useful for exploratory analysis on the relative importance of different environmental phenomenon. Importantly, such analyses can only be undertaken if data on the environmental or anthropogenic phenomena are also being collected concurrent to the seabird monitoring program. The CBMIS-2015 does not specifically include a framework or procedures for the collection of accompanying environmental or anthropogenic data and so by itself does not provide for these types of correlative analyses.

In addition, measures of *"breeding participation"* in any one season, as obtained in the CBMIS-2015, are strongly influenced by a) survivorship during the equivalent non-breeding season, b) whether surviving adults reach the body condition required to breed and c) the availability of appropriate breeding habitat. Consequently, this index likely responds most directly to threatening processes that are impacting food availability in non-breeding areas, or to changes in nesting habitat availability. Its association with long-term reproductive success and recruitment at breeding colonies is less direct, particularly since for most seabirds there is a five to 10 year lag between poor breeding success in any one year and the associated recruitment back into the breeding population of that same failed cohort.

4.3 Issues from Section 3: Simulations and power analysis

Results of the modelling process in Section 3 imply that for each of the four indicator species the current CBMIS-2015 monitoring program requires sampling periods of over 30 years to have sufficient power to detect declines of approximately 0.7 per cent per annum and so identify a change in threat status from *Near Threatened* to *Vulnerable*. This is the maximum period of time included in the modelling process. To detect declines from *Near Threatened* to *Endangered* at approximately 1.5 per cent per annum the time periods are shortened but are still relatively long at 20 to 25 years for crested terns, little terns and wedge-tailed shearwaters (at moderate levels of inter-annual variation) and 25 to 30 years for brown boobies. For all species except crested terns approximately 15 years of CBMIS-2015 data are required to detect declines from *Near Threatened* to *Critically Endangered* at approximately 3.2 per cent per annum. For crested terns, this time period is shortened to approximately 10 to 15 years. To summarise across all species the current monitoring program appears to require a 15 to 25 year timeframe to detect approximately 2 per cent decrease per annum, or, in other words, requires a minimum of 15 years to detect a population decline of approximately 30 per cent.

Therefore, based on our modelling, the current strategy will have limited power inside 15 to 25 year time frames to detect average annual changes of less than approximately 3 per cent. However, importantly, the current strategy is likely to succeed in providing alerts of any precipitous or abrupt changes in excess of approximately three to five per cent per annum over shorter periods of time. Being able to detect changes of approximately two per cent per annum after approximately 15 to 25 years also produces outcomes within the timeframe given for the Reef 2050 Plan. Remembering that it may be possible to improve the predictive power of the CBMIS-2015 strategy within this 15 to 25 year period (i.e. within the Reef 2050 Plan time frame) using existing historical data to augment incoming data. However, successful use of these historic data is subject to several caveats (Section 4.2.3); the extent to which these caveats interfere with use of historical data may well be revealed by ongoing analysis of population trends (see Section 2). It must also be remembered that being able to undertake comparative analyses using the much more intensively monitored Michaelmas Cay population as a reference substantially increases the utility of the data currently being obtained.

Using the IUCN level of change as the benchmark the current monitoring strategy needs to be capable of detecting a 15 to 20 per cent change over 10 years, which equates to a 1.5 to two per cent change per annum over this same period 10-year period. The current strategy does not meet this detection level. To do so requires a minimum 0.5 to one per cent improvement in trend detection per annum over a five to 10 year shorter time period. Within IUCN time frames the current strategy is only capable of detecting shifts to *Critically Endangered*. The inability of the currently monitored index (*breeding participation*) to produce statistically informative results on population trends within a 15 to 25 year time frame means that the current strategy is unable to assess trends in seabird abundance within the time period IUCN considers appropriate for the implementation of management aimed at mitigating species becoming *Endangered*.

There are multiple reasons why the CBMIS strategy is unable to detect population trends at the required levels. One is to do with the very high levels of inter-annual variation in breeding participation. The measure of inter-annual variation using any trend analyses has two components. The first is inter-annual variation due to observer effects. These include survey timing relative to breeding peaks (as modelled here), but also observer bias/errors that were not estimated during the modelling process. Consequently, the real power of the current survey methodology maybe be lower than the model outcomes. Some of the inter-annual variation due to survey methodology can be removed by improved survey processes (see recommendations). The improvement in predictive power gained by removing this component of variation can be seen as the ‘power difference’ between the ‘observed’ and ‘true’ values generated for each of the indicator species (Figures C4, B4, and L5). Therefore, these analyses suggest that improvements to the timing of survey methodology alone are capable of decreasing the time required to identify shifts from one IUCN category to another by five years (approximately). This still does not allow the current strategy to accurately assess trends in seabird abundance within the time periods considered appropriate by the IUCN. The additional improvement that could be gained from removing individual observer error and bias is unknown but is potentially important.

The second component of inter-annual variation affecting the predictive power of the current monitoring strategy is the actual biological year-to-year variation on breeding participation within a breeding colony. This combined with the general lack of within and between year breeding synchrony in tropical species likely adds considerable uncertainty to measures of between-season breeding participations for Reef breeding taxa and so limits the predictive power of any monitoring strategy. The effect of inter-annual variation on the predictive power of the monitoring program can clearly be seen in Figures L6 and W4. In fact, at even the lowest levels of ‘true’ inter-annual variation modelled, there appears to be no simple methodological change to the CBMIS-2015 strategy that would allow robust detection of trends that meet the IUCN criteria of 0.5 to one per cent per annum decline over a 10-year period.

Importantly, this outcome is partly a consequence of what we consider to be the ‘true’ levels of inter-annual variation we have included in the model. We believe that inter-annual variation has been estimated realistically, if not conservatively, in the modelling process based on comparisons of coefficients of variation for simulated time-series of breeding pairs (true and observed) and actual counts of breeding pairs extracted from selected *essential* sites of known importance for two of the four focal species, crested terns and brown boobies. For both species, simulated time-series of true and observed counts were, on average, less variable than actual counts. This suggests that actual power to detect trends may be slightly lower than reported here. Similar comparisons of simulated and actual population variability for wedge-tailed shearwaters and little terns were not possible due to insufficient data, but based on results for the other two species we expect that the range of simulated levels of inter-annual variability likely encompassed actual population variability for little terns and wedge-tailed shearwaters (also see Section 3.2 for species-specific rationale). But by accurately quantifying inter-annual variation it becomes possible to further test these assumptions and fine-tune the model and sampling strategy to better overcome this limitation.

5. Evaluation of new monitoring technologies

5.1 What new monitoring strategies are possible for the current indices?

A range of alternative technology is currently undergoing field trials at seabird breeding colonies throughout the Marine Park. These trials aim to determine whether it is possible use drones, cameras and/or acoustic recorders as alternatives to standard survey methodology. Trials were begun in 2012 for automated cameras, audio recorders and acoustic pattern analysis and more recently for drones.

An up-to-date evaluation of these experiments and recommendations regarding further field trials are provided in Hemson et al. (2017) (Appendix E). The summaries from this document for each automated option are also provided here for completeness and ease of access. As of May 2018, these trials are on-going and so it is not possible for us to undertake further review or make additional recommendations at this time as part of the RIMReP process.

A general summary to date would suggest that with additional work it may be possible to accurately estimate chick numbers at fledging and get a good approximation of numbers of chicks hatching and fledging for species like boobies and shearwaters, and possibly also frigatebirds as well as common and black noddies. But acoustics may never be useful for ground nesting terns. However, in addition to data being obtained on individual indices, it is also worth noting that autonomous-sentinel systems of any sort provide phenological data that enable more reliable estimations of breeding peaks.

5.1.1 Drones

Drones have the potential to reduce bias and error but at present still require staff to be present in the field and a staff member to analyse imagery manually. Counts from drone imagery likely generate much more accurate estimates of the numbers of large birds at a site than on-ground counts if visibility is reasonable. However, it is the number of breeding birds that we are most interested in and for many species it is difficult to identify breeding birds from non-breeding birds using drone imagery because of brooding adults covering chicks. Identification is dependent upon the altitude, magnification and resolution of the drone and its camera. Future work with automated image analysis may overcome this.

Drone-in-a-box type systems, a drone inside a box or hangar that charges the drone and deploys it onto pre-programmed routes when weather conditions are suitable, already exist. These and automated counting algorithms will improve and become cheaper in the near future. In addition, there is evidence showing that counts using both drone and ground observers may be more accurate and precise than counts using either method alone. This combination of current and future potential make further assessment of drones worthwhile.

5.1.2 Static cameras

The trial of the camera has been less compelling. The complexity of installing and operating the device, reliance on staff to count birds from footage, and questionable reliability of the technology detract from the underlying promise of the concept. The difficulties associated with having a single fixed perspective, and in discriminating between species and between breeding and non-breeding birds over distance, adds a level of variability to the data that is difficult to overcome. While pattern recognition software is likely to help in the future, until that eventuates it seems unlikely that these types of cameras will be useful other than for monitoring colonial beach nesters and/or monitoring priority species that do not lend themselves to acoustic survey and that are in locations that are extremely challenging to access regularly.

5.1.3 Acoustic

Findings to date suggest that acoustic sensors are simple to use and robust, and can produce results that scale reliably with the number of seabirds in an area. As such, they show great promise in producing robust indices of abundance or, with more work, actual estimates of abundance. The inconsistent application of monitoring methods during visual surveys may introduce biases to data; therefore acoustic data may be less error prone than first person observations. An additional advantage is that prolonged deployments allow season-wide monitoring rather than single days. This provides the opportunity to monitor both breeding participation (the number of breeding pairs and the size of the population) and success (the numbers of chicks hatched and raised until fledging). As changes in reproductive success only influence the size of the breeding population several years later when birds from the effected cohort first return to breed, monitoring of breeding success has potential as an important early warning indicator of future population decline. This lag otherwise limits our capacity to understand and react to change or manage threats in a timely manner.

However, analyses of these data are currently outsourced to contractors in the United States of America and this comes at a cost. Time and effort also need to be invested to establish that acoustic measures scale reliably with each species. This requires counting nesting birds near recorders several times to correlate these counts with data from acoustic recordings taken from equivalent periods. In some cases these experiments may reveal weak correlations between counts and acoustic recording of breeding birds and hence involve risk that resources may be ‘wasted’. These experiments have been undertaken for several species and information gained has been used to develop guidelines about the types of species and colonies that lend themselves to acoustic monitoring. These include two of the key seabird species identified in the CBMIS-2015 strategy; brown boobies and wedge-tailed shearwaters. This approach would likely be useful for any species that breeds in colonies spread out over quite large areas and/or that breed in a predictable location every year, including brown, masked and red-footed boobies, wedge-tailed shearwaters, black and common noddies, sooty and bridled terns. Lesser frigatebirds, which despite nesting in discrete colonies, may nest in predictable colonies in the same areas each year so may too be suitable.

5.2 What other indices could be monitored and what threatening processes could these indices detect?

Based on the "Conceptual Model" for seabirds of the Reef developed in Section 4.2.1 (Appendix D), potential additional monitoring activities can be broken down into four modules having different levels of monitoring intensity and associated effort. Each of these modules contains a unique set of indices that are sensitive to different potential threatening processes at different time scales. The indices within each module are related in such a way that information on each is obtainable via similar sampling methodologies and/or visitation rates. These modules are not mutually exclusive and can be combined for higher resolution and finer-scale additional information. The potential utility of each module to the overall seabird monitoring strategy is outlined for each. Modules are placed in order of the perceived effort/resources required to obtain the data, not necessarily according to their potential utility.

5.2.1 Module 1. Indices of between-season reproductive success

Additional indices that can be monitored:

- Nesting participation — number of established nests pre-laying
- Laying success — number of eggs laid
- Hatching success — number of chicks post-hatch
- Fledging success — number of chicks at fledging
- Breeding phenology — shifts in the timing of the events above

Data on these indices is obtained for each species of interest before and after the beginning of incubation, post-hatching and at chick fledging. Data on each index is obtainable during a single colony visit. Therefore, acquisition of data on all indices requires visual surveys/counts at four to five specific times during the breeding season for each indicator species. For some indicator species, it is likely one visit will be necessary to establish exactly when egg laying will commence so that the other visits can be timed appropriately. Data on each of these indices apart from *laying success* is also potentially obtainable using acoustics, with or without drones, after appropriate calibration of these methodologies.

What do these indices monitor and what improvements would they provide in the overall strategy?

These indices respond directly to change in the following critical breeding resources or reproductive components, all of which need to remain within acceptable limits of change for successful reproduction to occur:

- Nesting habitat availability
- Non-breeding and pre-breeding adult food supply

- Egg mortality
- Chick mortality (starvation and non-starvation combined)

A range of threatening processes is known to impact each of these specific reproductive components or breeding resources. These can be identified using the seabird conceptual model (Appendix D). For example, changes in chick mortality (starvation and non-starvation combined) as measured by differences between hatching and fledging success may occur as a response to either changes in food availability to chicks (starvation), or increases in predation, pathogens, pollution or natural/anthropogenic disturbance (non-starvation). A single measure of chick mortality across a breeding season cannot distinguish among these possibilities, but it does identify a specific life-history phase that is being impacted and a subset of possible threatening process for further examination using higher resolution indices or experimental designs. This is also true of the other life-history phases monitored using these indices.

Breeding participation as currently measured by the CBMIS-2015 is similar to a combined measure of *nesting participation*/*laying success* in the indices above, with similar associated limitations. However, acquiring data on these two components independently, as well as on the additional indices of this module allows significantly better estimates of between-season variation in all components of reproductive success. Importantly, this decreases the length of time needed to detect significant trends in *breeding participation.* In addition, because these indices provide information on the year-to-year variation in specific life-history components they can be used to monitor for significantly more rapid fluctuations, or downward trends in breeding success than are observable using *breeding participation* alone. Similarly, because they better identify the life-cycle stage being impacted, when combined with data on background environmental variability they can be used to isolate potential threatening processes impacting each life-history stage at seasonal scales.

Within this set of indices, measures of *fledging success* relative to *breeding participation* are particularly useful, as they offer the greatest potential window into short-term changes in year-to-year reproductive success due to the combined influence of egg mortality, near-colony decreases in food availability and non-food related mortality of chicks.

5.2.3 Module 2. Mark recapture of adults and fledglings at nest sites

Additional indices that can be monitored:

- Adult overwinter survivorship
- Adult inter-annual breeding participation rate
- Adult pre-breeding condition
- Fledgling pre-breeding survivorship and recruitment
- Individual adult breeding phenology

When combined with monitoring of egg/chick breeding indices from Module 1 additional information is also obtained on;

- Adult age-specific reproductive success
- Relationships between adult survivorship, breeding participation and reproductive success

Mark-recapture analysis is an alternative standard method of obtaining population estimates. Mark-recapture methods are considerably more powerful than the total nest counts currently used by the CBMIS-2015 because they enable estimates of the non-breeding population in each season and over time, and can also provide information on a range of other demographic and life-history parameters such as age and sex-specific mortality, recruitment and patterns of movement. However, mark-recapture techniques are also very labour intensive, time consuming and hence expensive. Their accuracy is also highly dependent on a range of specific analytical assumptions being valid and on data quantity, accuracy and robustness being assured. Unless careful consideration is given to these requirements, mark-recapture estimates are often highly inaccurate because the base models are seldom more than a vague approximation of reality. So data quality control becomes of particular importance with the use of these techniques.

What do these indices monitor and what improvements would they provide in the overall strategy?

When done effectively, mark-recapture methodologies allow significant improvements in accurate population estimation and trend analysis, enabling trends to be established over much shorter sampling periods.

Crucially, these indices respond to changes in food supply and survivorship in at-sea wintering areas. This allows the relative importance of impacts in breeding verses non-breeding components of the system to be identified and quantitatively isolated, thus providing a window into potential threatening processes affecting reproductive success away from breeding grounds on the Reef. Some information on these processes can also be gleaned from *breeding participation* if *nesting participation* and *laying success* can be separated. However, it is only via mark-recapture that season-to-season changes in *breeding participation* due to poor adult condition and deferment of breeding can be separated from adult mortality. When applied to fledglings, mark-recapture methodology is also the only way to attempt to identify the impact of post-fledging mortality on patterns of recruitment into the breeding population.

These types of data are clearly important for management as they allow managers to determine at what spatial and temporal scales management must be applied to counter specific tends in overall population numbers and what the potential effectiveness of specific regional management options may be. This allows management to be more targeted and cost effective.

5.2.3 Module 3. Chick weights and measures — measurements obtained on chicks at nest sites during breeding.

Additional indices that can be monitored:

- Fledging weight
- Provisioning rates
- Meal sizes
- Chick growth
- Chick condition
- Starvation rate
- Non-starvation mortality

Apart from *fledging weight*, which is obtained once per season, this is an interrelated set of indices that are all generated from measurements taken of chicks at regular intervals throughout the breeding season. This is a very powerful set of indices but one that requires either repeated observation of nests and handling of chicks over short time intervals throughout the season, or an automated system of data acquisition and retrieval. For this reason, these indices are generally only obtainable for a specific subsample of nests at easily accessed colonies. Consequently, the collection of this information is more amenable to species that remain at a single nest site during the entire breeding season and reuse nesting sites from year to year such as shearwaters, and tropicbirds. It is also easier to obtain this type of data for species that will use artificial nest sites, which can be prepared prior to breeding so that the data can be obtained remotely. While it is not as easy to obtain data on these indices for ground nesting species where chicks become mobile early and form crèches, it is still possible, particularly for smaller subsections of a specific population.

What do these indices monitor and what improvements would they provide in the overall strategy?

These indices are particularly useful for monitoring and understanding changes in within and between breeding season food availability to chicks. Therefore, they give early warning of threatening processes directly influencing food availability at breeding sites both within and between seasons, such as effects of increasing sea-surface temperatures or other changing oceanographic conditions (Appendix D).

In addition, these indices provide a direct measure of the relative importance of changes in local food availability versus other factors that influence chick survivorship such as predation, pathogens etc., thereby allowing quantification of the effects of threatening processes unrelated to food availability. This is particularly true when these indices are monitored concurrently with those in Module 1 (above) across a larger sample of nests.

Therefore, Module 3 indices are useful for identifying consistently poor reproductive output and potential reproductive collapse along with the associated causes of these problems prior to recruitment being affected. They also narrowly define the period within each breeding attempt when threatening processes impact and allow the magnitude of these impacts to be accurately quantified. Without such indices, ongoing reproductive failures and poor recruitment are observable only as decreases in total breeding populations over extended periods of time such as 15 to 20 years or longer. Such trends may not even be detectable because of high levels of between-season variation in breeding participation due a number of factors other than changing recruitment (see Section 3). Similarly, without these indices the potential threatening processes driving any identified population level declines remain unknown.

These indices also provide information specifically on the relative importance of components of the reproductive system that can be influenced by local management; that is, impacts on food supply in near-colony foraging grounds and on within-island causes of chick mortality. This, provides the opportunity for management to be timely and focused on the specific problem influencing reproductive success.

5.2.4 Module 4. Adult breeding weights, measures and behaviours — measurements obtained on adults at nest sites

Additional indices that can be monitored:

- Adult weight change
- Desertion rate
- Patterns of adult nest attendance
- Adult on-island non-starvation mortality

These are indices generated from measurements of adults taken at regular intervals throughout the breeding season. Therefore, similar to Module 3, acquisition of these data requires either repeated observation of nests and/or handling of adults at short time intervals throughout the season, or an automated system of data acquisition and retrieval. For this reason these indices are generally only obtainable for a specific subsample of nests/adults.

Regular handling of adults in this way also has the potential to disrupt provisioning and/or cause desertions and so needs to be pre-trialled and undertaken with care. If possible, obtaining the data to generate these indices with automated systems is preferable.

What do these indices monitor and what improvements would they provide in the overall strategy?

These indices monitor food availability to adults across the duration of each breeding season and its influence on body condition, nest attendance and rates of chick provisioning. In addition, because they provide information on rates of desertion relative to body condition, these indices can also be used to estimate levels of within-season mortality of adults.

Because seabirds are relatively long lived and breed over an extended number of years, when breeding most adults will preferentially maintain their own body condition over that of their chicks during food shortages. This means that when the same pool of foraging resources is used by adults and chicks then food shortages are more likely to show up firstly, as a lack of food being provided to the chicks, followed by chick starvation and subsequent adult desertion. These types of impacts are most easily observed in the indices of Module 3.

However, information on adult condition becomes increasingly important when the food resources used by adults and chicks are not the same, such as is known for wedge-tailed shearwaters on the Reef. Adults of this species use a discrete set of foraging locations at great distances from the breeding colony for self-provisioning and a different set of near-colony resources for chick provisioning. At present it is not known if other seabird species breeding on the Reef also partition resources in this way. But based on evidence from elsewhere, it is possible that both the males and females of many breeding species on the Reef self-provision using different foraging locations, prey types and/or prey size classes.

Chick food availability determines fledging success in any one season and may impact later recruitment for the same cohort. However, unless chick food supplies remain depressed for long periods, the loss of fledglings in poor seasons can be compensated for by increased fledging rates in better years. This is true as long as adult survival is not impacted. But without sufficient adult-specific foraging resources reproductive output in any one season completely collapses and higher adult mortality during migration and over winter becomes more likely. Consequently, the maintenance of adult-specific foraging resources becomes critical to the long-term stability of any breeding seabird population. The only way to monitor if food availability to adults is changing is to monitor adult-specific food acquisition rates and adult condition via the indices in this module.

Therefore these indices become uniquely important for disentangling the impact of changing food regimes on adults and chicks and so for looking at the stability of any adult-specific resources critical to the continuing long-term stability of a breeding colony. They also identify whether problems with these resources are occurring at adult-only foraging sites that are outside currently managed areas.

6. Recommendations for monitoring seabirds on the Great Barrier Reef

6.1 Recommendations for the current CBMIS-2015 strategy

The CBMIS-2015 strategy was always intended to be adaptive in its ability to utilise incoming information to better develop and refine survey methodology. Based on the issues identified previously (Section 4), a number of recommendations can be made to facilitate this adaptive process and further improve the potential ability of the current CBMIS-2015 to detect long-term trends.

6.1.1 Error measurements for breeding participation estimates (from Sections 1.3 and 4.1.1)

The lack of error measurement for each single-figure total breeding population estimates per annum is a problem. To overcome this issue, measures of observer error/bias need to be incorporated into the standard CBMIS-2015 sampling program that, as far as possible, remove any subjectivity from the estimation process. The preferred option is via the use of at least two independent observers undertaking multiple counts (minimum two per observer) for each taxa, at each site. Alternatively, but less preferably, this error could be estimated via multiple independent counts by a single observer usually on different days. This is less preferable because, unless the independence of counts is stringently maintained during the multi-count process, a single observer's bias is often just reinforced, giving a more precise but equally less accurate result. After either of these procedures is used the resulting data needs to be stored so that each of the individual counts, their details and the associated observers are retrievable for further quality control and analyses. Significantly less preferable is that these data could be obtained via a general error estimate associated with each count. This process is potentially no more useful than the single figure count because of the introduction of significant subjectivity into the estimation process that has no real quantitative basis apart from the observer's original estimate.

6.1.2 Are four indicator species enough? (from Section 1.4 and 4.1.2)

How effectively the focused surveying of four indicator species combined with non-indicator species monitoring identifies potential impacts across the broader range of seabird species breeding on the Reef is largely unknown. We are unaware of any formal review process having been undertaken to assess the life-history overlap among species and suspect that the data probably do not exist specifically for breeding populations of the Reef. However, such a review should, at least in part, be possible using the broader seabird literature and looking at general life-history and ecological overlap in the same or similar species when they breed together elsewhere. We recommend that such a review be undertaken to identify the degree to which the current CBMIS-2015 may provide information across a broader spectrum of taxa. Such a review should also aim to highlight species for which too little data are available to make an assessment so that it is clear that the applicability of the current CBMIS-2015 to these species is unknown.

Clear rationale explaining the extent to which indicator species are to be used as surrogates for other taxa could then be included in the CBMIS-2015 program documents. Thus identifying both the broad-scale utility of the current program and knowledge gaps and research priorities for further improving the monitoring effectiveness. Such a review would also be useful for deciding on the utility of using single-species tracking data for identifying and designing Important Bird Areas (IBP) and Marine Protected Areas (MPA) for seabirds within the Reef region.

6.1.3 Sampling at Michaelmas Cay (from Sections 1.5.1 and 4.1.3)

The maintenance of the monthly sampling regime at Michaelmas Cay is an important on-going component of the CBMIS-2015 strategy. As such, we recommend that the rationale for maintaining sampling at its current intensity at Michaelmas Cay and the way that data from this site interfaces with the overall CBMIS-2050 strategy be explicitly outlined and emphasised in the CBMIS-2015 program documents. This is so as to avoid any possibility that the two long-term and inter-related sampling programs become decoupled, resulting in the sampling at Michaelmas Cay being down-scaled without the impact of this on the CBMIS-2015 as a whole being considered.

6.1.3 Monitoring of non-indicator species (from Sections 1.5.2 and 4.1.4)

Data on non-indicator species are considered important to the overall monitoring strategy (see Section 1.5.2 and 4.1.4). Consequently, it is important that the requirement to obtain data on all taxa at a site be a fully identified, prescribed and maintained component of the current CBMIS-2015 strategy. Therefore, we make recommendations on the three potential issues associated with the monitoring of non-indicator species.

1) That the requirement to obtain count data of equivalent quality on all non-indicator species during an *essential* site visit be a clearly outlined component of the strategy within the CBMIS-2015 document.

2) That zero sightings of any species at known breeding locations are recorded for both indicator and non-indicator species in such a way that this information is easily retrievable from the resulting database as well as distinguishable from missed surveys.

3) That the level to which survey timing built around breeding peaks for indicator species effectively captures data on other non-indicator species needs to be quantitatively assessed so that significant mismatches can be identified. It is possible, at least in part, to use the existing data from the WildNet database to check for obvious mismatches. A program also needs to be established for continuing to screen incoming CBMIS-2015 data for mismatches that could trigger the need for additional monitoring for non-indicator species of concern.

6.1.4 What level of change needs to be detected? (from Section 2.2)

No required level of trend detection was specified in the CBMIS-2015, or any previous management plan or strategy. Consequently, we have developed a set of detection criteria for the Reef's breeding seabirds that identify the level of decline/change considered ecologically significant and against which the effectiveness of the current or future monitoring programs can be assessed. These levels are based on IUCN international standards making them appropriate for use within the Reef 2050 Plan. We recommend these criteria be adopted and documented as a component of any ongoing monitoring programs.

6.1.5 Issues with the power of the current CBMIS-2015 program (Sections 3 and 4.3)

6.1.5.1 Survey methodology and observer effects

The inability of the currently monitored index of *breeding population size* to produce statistically informative results on population trends for indicator species against criteria of acceptable change within a 15 to 25 year time frame is a significant limitation to its use for effective management of seabird populations of the Reef. This lack of power is due to both natural and observer introduced inter-annual variation in the count data obtained.

Therefore, the primary recommended adjustments or alterations to the current methodology should aim to remove, as far as possible, any inter-annual variation in *breeding population size* due to observer affects, so as to shorten the potential time frame within which robust predictions can be made. This requires sampling the breeding peaks for indicator species as accurately and consistently as possible. The current CBMIS-2015 monitoring process already attempts to do this in the most cost effective way. Increasing the number of surveys undertaken at specific sites would further improve estimates of both breeding peaks and the levels of inter-annual variation (as can be seen at Michaelmas Cay), with remote continuous monitoring being the highest resolution version of this option for improvement. Our modelling suggests a total reduction of approximately five years in the time needed to accurately predict trends could be obtained from further fine tuning of survey methodology in this way. The relative merits of attempting to achieve these gains in power via either increasing survey frequency, or by using remote monitoring ultimately depend on the relative cost and site-specific applicability of each. A full, site-specific comparative review of these alternatives is beyond the scope of this report given the current on-going trialling of remote options.

Additional recommendations (also see Section 6.1.1) associated with these issues are:

1) Continued analysis of incoming data to ensure that the timing of surveys is optimal for measuring breeding peaks in both indicator and non-indicator species. This is also to identify specific sites/species where current survey intensity is potentially inadequate and increased rates of visitation may provide significant and cost effective improvements in trend detection.
2) Continued development and testing of remote methodology for its potential to provide cost effective, higher resolution, non-subjective information on breeding participation and other possible indices of reproductive success.

6.1.5.2 Natural inter-annual variation

If these adjustments are successful in removing observer influences on levels of inter-annual variation then the models presented here still suggest that a minimum time frame in the region of 15 years is required to achieve high power to detect trends for some indicator species (crested terns, wedge-tailed shearwaters) at the levels of annual decline considered significant under IUCN criteria and approximately 20 years for others (brown boobies).

Consequently, a further recommendation is that, as far as possible, the actual level of inter-annual variation should be quantified. This would enable further testing of predictions of the modelling process undertaken here and also more accurate quantification of the period required for statistically robust trend analyses. This needs to be done individually for each indicator species via both an analysis of past data (currently underway, see Section 2), and as an on-going analysis of incoming data from the CBMIS-2015 combined with information from the monitoring of additional indices as recommended (see Section 6.2).

Combined with this is the recommended analysis of, and continued checking on, level of mismatch between breeding of indicator and non-indicator species so as to allow effective comparative analysis and utilisation of non-indicator species data.

6.2 Recommendations for the use of additional indices

6.2.1 Fledging success and mark-recapture indices

As noted previously a single index of *breeding participation* may be of use within a 15 to 25 year timeframe for highlighting abrupt change over shorter periods against a background of smaller incremental change. However, the lack of any explanatory context that can be used to inform decision makers about the drivers of such change is a real problem with the current strategy. Even if a major decline has been identified, the use of a single index leaves managers none the wiser about what has happened, or whether they are able to do anything about it. Under the current monitoring program, managers would have to begin to look for drivers of change only after the date of adverse trend discovery.

To further improve estimates of natural inter-annual variation and also avoid costly, potentially catastrophic delays in the implementation of appropriate management options, we recommend the monitoring of additional key variables/indices. These indices provide information on what is likely to be driving change and so potentially enable declines to be managed proactively, before population stability is compromised. A range of additional indices has been identified previously for this purpose (Section 5.2).

The most useful subset of additional indices are those focused on components of the ecological system that the Authority can potentially influence and improve via direct management actions. In addition to these are also indices that alert the Authority to the influence of emerging threats both within and outside management jurisdictions. Useful indices will also provide information sufficient to trigger the need for additional higher resolution monitoring or information needs when potential impacts are identified.

The minimum level of additional monitoring that would be potentially useful is the monitoring of one to two other indices apart from breeding population size. We would recommend, in the first instance, the monitoring of additional indices from Module 1, particularly *fledging* *success*, because *fledging success* in combination with measures of *breeding participation* likely offers the greatest potential early warning of local reproductive failures likely to impact future population stability. *Fledging success* as measured against breeding/nesting participation identifies significant shorter-term changes in year-to-year reproductive success due to the combined influence of egg mortality, near-colony decreases in food availability and non-food related mortality of chicks. It can therefore be used to highlight decreases in reproductive output that are likely to impact on later recruitment and so follow through to overall breeding numbers. Monitoring of *fledging success* alone provides limited additional information and is the minimum that you would want to add to the existing CBMIS-2015 program.

Importantly, as with quantifying *breeding participation,* obtaining data on the additional indices from Module 1, such as *fledging success,* also requires accurate estimates of when peak breeding occurs. This is so that forward predictions can be made for the timing of additional visits that maximise the quality of the resulting data. The utility of additional indices such as *fledging success* can only be realised if observer introduced inter-annual variation in the count data obtained for these indices is also reduced to a minimum (see Section 6.1.5).

Additional indices such as *fledging success* would of course need to be monitored for all indicator species at appropriate breeding colonies for the reasons that these species and colonies were identified as important in the development of the original CBMIS-2015. Principal breeding colonies would need to include, but not be limited to, Raine Island, Michaelmas Cay, and the Capricorn-Bunker group of islands.

The more indices that are monitored, the more targeted management response can be. Therefore, measuring *fledging success* in combination with additional indices is preferred. Measurements of *fledging success* can be logistically combined with mark-recapture (Module 2) from a single round of capture and banding undertaken at the same time that fledging success is measured. Mark-recapture at this time can be undertaken on both adults that have successfully bred and chicks that are fledging. Combining these two processes into a single sampling period significantly improves both logistic feasibility and the breadth of life history and threatening processes that are being monitored. Additional marking of birds at the beginning of the breeding season when *breeding participation* surveys are undertaken further increases the range of information obtainable on over-winter patterns of adult survivorship relative to inter-annual breeding success without the need for additional visits to colonies. Therefore, an efficient overall strategy combining mark-recapture with surveys of *breeding participation* and *fledging success* would look at accessing each colony twice per season; at both the beginning and end of breeding.

6.2.2 Intensively monitored model sub-populations

The most informative strategy would of course be to monitor all of the additional indices described. However, it is clearly not possible to do such intensive monitoring at all locations or on all species for financial, technical and/or logistic reasons. However, this strategy can be recommended for a subset of species at frequently visited breeding locations where it is feasible to establish a continually banded and intensively monitored sub-population. Monitoring of this type at selected essential sites can not only provide a greater understanding of demographic causes of population change at the intensively monitored sites, but also provide insight into drivers of population change for other species and locations where fewer indices are monitored.

As explained previously, intensive monitoring is best done on species that use a single nest site during the entire breeding season, that reuse nest sites from year to year, and will use artificial nest sites which can be prepared prior to breeding. Such species definitely include wedged-tailed shearwaters and black noddies, likely red-tailed tropicbirds, and potentially boobies and frigatebirds. While it is not as easy, it is also possible to obtain data on these indices for smaller ground nesting species such as sooty terns where chicks become mobile early and crèche.

Using current technology, it is most logistically feasible to establish a monitored sub-population of this type for wedge-tailed shearwaters and/or black noddies in the Capricorn-Bunker Island group. It would be particularly useful to establish such an intensively monitored population for wedge-tailed shearwaters because wedge-tailed shearwaters have been quantitatively shown to be the most useful model species for a large range of other seabird taxa. They are known to be sensitive to a number of threatening processes (such as the El Niño Southern Oscillation and Sea Surface Temperatures) that have also been shown to significantly impact other smaller ground nesting species. Wedge-tailed shearwaters also forage at a range of different distances from breeding colonies and access foraging environments and prey types/sizes used by many other species, particularly smaller-bodied terns that are particularly difficult to obtain similar information from. The extensive data already obtained for wedge-tailed shearwaters on foraging site use and drivers of foraging resource availability, means that general models of the influence of these phenomena on population change can be more easily and quickly developed and applied for management purposes.

Currently, acoustic monitoring trials and protocols are most advanced for wedge-tailed shearwaters. This means that only for this species would it be possible to quantitatively compare results from all three monitoring methodologies (observer counts, acoustics and intensive sub-population monitoring) to inform development of future monitoring processes. The use and acceptance of artificial nest sites that allow this type of intensive monitoring has already been trialled and used successfully on wedge-tailed shearwaters, as has GPS tracking with automated systems of data recovery. This means such a monitoring program could be relatively quickly established using current techniques and any impacts affecting reproductive success be directly associated with specific foraging sites and resources. The logistics for this species at Heron Island also lend themselves most easily to continued introduction of remote data acquisition options so that the monitoring process can become fully automated. This ease of logistics also make this the best species and site for developing automated monitoring systems that could be used for more intensive monitoring of sub-populations of other species at remote locations, for example, red-tailed tropicbirds or other petrel species at Raine Island.

6.3 Recommendations for monitoring of environmental indices

If the intended goal of future monitoring programs is to collect data that will facilitate timely adaptive management intervention, then monitoring of seabird breeding indices alone will not suffice. Analyses linking identified changes in reproductive parameters/indices directly to driving processes can only be undertaken if data on environmental or anthropogenic phenomena are being collected concurrent to the seabird-monitoring program. The CBMIS-2015 does not specifically include a framework or procedures for the collection of accompanying environmental or anthropogenic data and so by itself does not provide for these types of correlative analyses.

Therefore, it is a recommendation of this RIMReP report that current seabird monitoring programs need to be thoroughly integrated with both ongoing dedicated monitoring of background environmental variation and focused research studies into key associated ecological processes. Specifically, general environmental monitoring programs aimed at large-scale environmental process thought to be important for other biological components of the Reef, such as coral cover, also need to specifically consider the spatial and temporal scale of data requirements necessary to examine potential drivers of seabird food availability and breeding success, as outlined in the seabird conceptual model (Appendix D). Key amongst these are changing patterns of oceanography at regional scales and potential shifts in the distribution and abundance of both forage-fish and key sub-surface predators that forage in association with seabirds.

References

Field, S.A., Tyre, A.J., Jonzén, N., Rhodes, J.R. & Possingham, H.P. 2004. Minimizing the cost of environmental management decisions by optimizing statistical thresholds. Ecology Letters, 7, 669–675

Fuller, R.A. & Dhanjal-Adams, K.L. 2012. Monitoring seabirds in the Great Barrier Reef: A power analysis. Report to Queensland Department of Environment and Resource Management.

Gochfeld, M., Burger, J. & Garcia, E.F.J. 2018. Little Tern (*Sternula albifrons*). In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A. & de Juana, E. (eds.). *Handbook of the Birds of the World Alive*. Lynx Edicions, Barcelona. (retrieved from https://www.hbw.com/node/54031).

Hemson, G., McDougall, A., Dutoit, J. 2015. Coastal Bird Monitoring and Information Strategy: Seabirds 2015-2020. Brisbane: Department of National Parks, Sport and Racing, Queensland Government. (Appendix A)

Hemson, G., McDougall, A. & Melzer, R. 2018. Autonomous Monitoring of Seabird Breeding Sites, Ecological Assessment Unit, QPWS

McDougall, A. 2011. *Coastal Bird Monitoring and Information Strategy*, State of Queensland, Department of Environment and Resource Management, Rockhampton.

Reef 2050 Long-Term Sustainability Plan, Commonwealth of Australia 2015

Roberts, P. & Mckown, M. 2018. Acoustic monitoring of wedge-tailed shearwater and black noddy populations in the Capricornia Cays. Prepared for G. Hemson and A. McDougall from Queensland Parks and Wildlife Service on 15-February-2018.

IUCN. (2012). IUCN Red List Categories and Criteria: Version 3.1. Second edition. Gland, Switzerland and Cambridge, UK: IUCN. iv + 32pp. Available at: http://www.iucnredlist.org/technical-documents/categories-and-criteria

WildNet database Environment and Science, Queensland Government, Qld wildlife data API, licensed under Creative Commons Attribution 4.0 sourced on 30 May 2018 (https://data.qld.gov.au/dataset/qld-wildlife-data-api)

Appendix A — Coastal Bird Monitoring and Information Strategy - Seabirds 2015-2050

Summary

This Coastal Bird Monitoring and Information Strategy – Seabirds 2015-2020 (Strategy) revises the seabird component of the current Coastal Bird Monitoring and Information Strategy (CBMIS) (McDougall 2011) and was created using the best available data, expert opinion, commissioned reports and operational expertise. The strategy encompasses the east coast of Queensland and excludes the Gulf of Carpentaria – not because the Gulf is any less significant for seabirds but because it is beyond current operational capacity.

- The Strategy is built around four indicator species representative of coastal, inshore, offshore and pelagic feeding guilds. Initial site selection prioritised these species and subsequent sites were added to improve coverage of species less well represented in the initial selections.
- The sites and timing of visits laid out in the strategy will maximise the likelihood of obtaining useful data on 20 species of seabird while minimising operational effort.
- The Strategy is divided into a list of *Essential* sites and visits to be made each year and a list of *Significant* sites that will contribute valuable data if resources are available to include them.
- The Strategy defines a maximum period of five years between visits for any significant seabird site to ensure that major changes are not overlooked and highlights the need to integrate this condition in with other requirements for visitation.
- The importance of timing and consistency are explained in detail, as are matters of governance with respect to altering the strategy prior to the formal review in 2020.

Introduction

Seabirds are conspicuous natural values of the coast and islands of Queensland. They are of interest to conservation and tourism stakeholders and, as high order predators, their demography reflects the processes and condition of the ecosystems within which they feed and nest (Catry et al. 2011, Dunlop et al. 2002). They can have profound influences on island ecosystems, bringing nutrients from the sea to the land (Ellis, Fariña & Witman 2006, Towns et al. 2009). These nutrients may be vital to the fertility and biodiversity of some of our highest value island national parks in the form of guano and the by-products of breeding such as unconsumed regurgitate, dead chicks and eggs.

Several species travel across international boundaries and are the foci of international treaties (see appendix A.1) such as the Convention on the Conservation of Migratory Species of Wild Animals (Bonn Convention) and bilateral conservation agreements such as the *Agreement Between the Government of Australia and the Government of Japan for the Protection of Migratory Birds and Birds in Danger of Extinction and their Environment*, mercifully known as the Japan-Australia Migratory Bird Agreement (JAMBA). These agreements require Australia to ensure that cited species are given sufficient protection to prevent long term population decline and to gather information that allows us to advise bilateral partners as to changes in their status. In addition, Queensland's and Australia's conservation legislation affords different levels of protection to species based upon their conservation status which is largely derived from assessments of demography and distribution. The value of seabird populations is captured in commitments to manage and monitor them in all levels of Great Barrier Reef management from the Statement of Outstanding Universal Value through the Intergovernmental Agreement and into the Field Management Program's annual business plans. The commitment to "Monitor and report on key seabird populations to establish trends" in the Long Term Sustainability Plan is quite specific.

In Queensland and particularly the Great Barrier Reef World Heritage Area (World Heritage Area) there are concerns about seabird populations that stem from publications describing declines of seabird populations at Raine Island, Michaelmas Cay and the Swain Reefs' cays (Batianoff & Cornelius 2005, Devney et al. 2009, Heatwole et al. 1996). While our interpretation of this material suggests that reported declines may be over stated or significantly influenced by methodological bias they re-emerge in most reporting on seabirds in Queensland and the Reef and underpin most of the concern for seabirds in these areas

(Congdon et al. 2007, Great Barrier Reef Marine Park Authority 2009). While the reported declines appear plausible when viewed in the global context of seabird declines (Croxall et al. 2012), subsequent efforts to confirm these trends and to detect similar patterns across Queensland have been undermined by the inadequacy of the data (Driscoll 2013). While two monitoring strategies have resulted in significant improvements in determining how, when and where we gather coastal bird data, these improvements did not include quantitative considerations (McDougall 2011, Turner 2002). The Driscoll report highlights the variability in how and when data has been gathered as a major impediment to its usefulness for establishing patterns in seabird demography. In response to these and related concerns the seabird component of the 2012 Coastal Bird Monitoring and Information Strategy is replaced by this Strategy with a specific view to improving our understanding of how seabird populations are changing through time and across the region (Driscoll 2013). The shorebird/wader component of the CBMIS (McDougall 2011) is still current but will likely be reviewed in the future.

The Strategy sets out the minimum combination of sites and visits to give the Queensland Government and the Authority the ability to evaluate the status of seabird populations and their demographic trends. The monitoring described is surveillance or foundational in that it provides us with data that reflects the status of species across years rather than details about their ecology (Legg & Nagy 2006, Melzer 2013). However, this does not preclude using the data for more detailed analysis and correlation with ecological drivers in the future. This Strategy does not describe performance or management effectiveness monitoring for impacts of, or recoveries after, a management intervention and does not replace the need for this targeted project specific monitoring. The objective of the monitoring described in this strategy is to establish how populations of seabirds in Queensland change through time and to alert us to undesirable trends so that we might understand, reverse or mitigate them.

The Strategy is a compromise between data quality and operational feasibility. Sites have been selected for both seabird values and for operational feasibility. Many sites that host significant breeding populations of seabirds do not feature in this Strategy because of this compromise. It is important to clarify that while a current monitoring site may not have been included in the two lists in the Strategy, monitoring at the site should continue if there is the capacity and rationale for doing so.

Methods for Selecting Sites and Visitation Strategies

To select the sites and estimate optimal visitation frequencies and timings we extracted data on seabird breeding from the Queensland Government's WildNet database to form the basis of a decision support tool. We excluded sites in the Coral Sea and the Gulf of Carpentaria from the data used in the tool as this best reflected the Queensland Parks and Wildlife Service's (QPWS) operational limitations and the likelihood of vessels available to the management agencies being able to transport staff to these locations to undertake surveys.[1] However, we acknowledge that some seabird populations from the east coast of Queensland are highly likely to be mixing with populations in these two areas and even further afield.

From these data we calculated crude estimates[2] of the average breeding populations for each species in Queensland and at each site and then the approximate proportion of each species breeding at each location for which there were records. This calculation provided an objective estimate of the importance of each site to each species. An example is provided in Figure 1 in which a value of one indicates that 100 per cent of the State's population of this species breeds at a location.

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[1] We have retained the data from the Gulf of Carpentaria and the Coral Sea and have requested that DEHP consider whether they can assess the current significance of Rocky and Manowar Islands in the southern Gulf.

[2] We consider the estimates crude as no effort was made to separate surveys undertaken when seabird breeding was most likely from those when seabird breeding was less likely. This was partly a time consideration but partly an issue of practicality. Seabird breeding is not uniform in Queensland with some species breeding very seasonally in some areas and throughout the year in others. For many species and sites we could not reliably determine which surveys were in a likely breeding season.

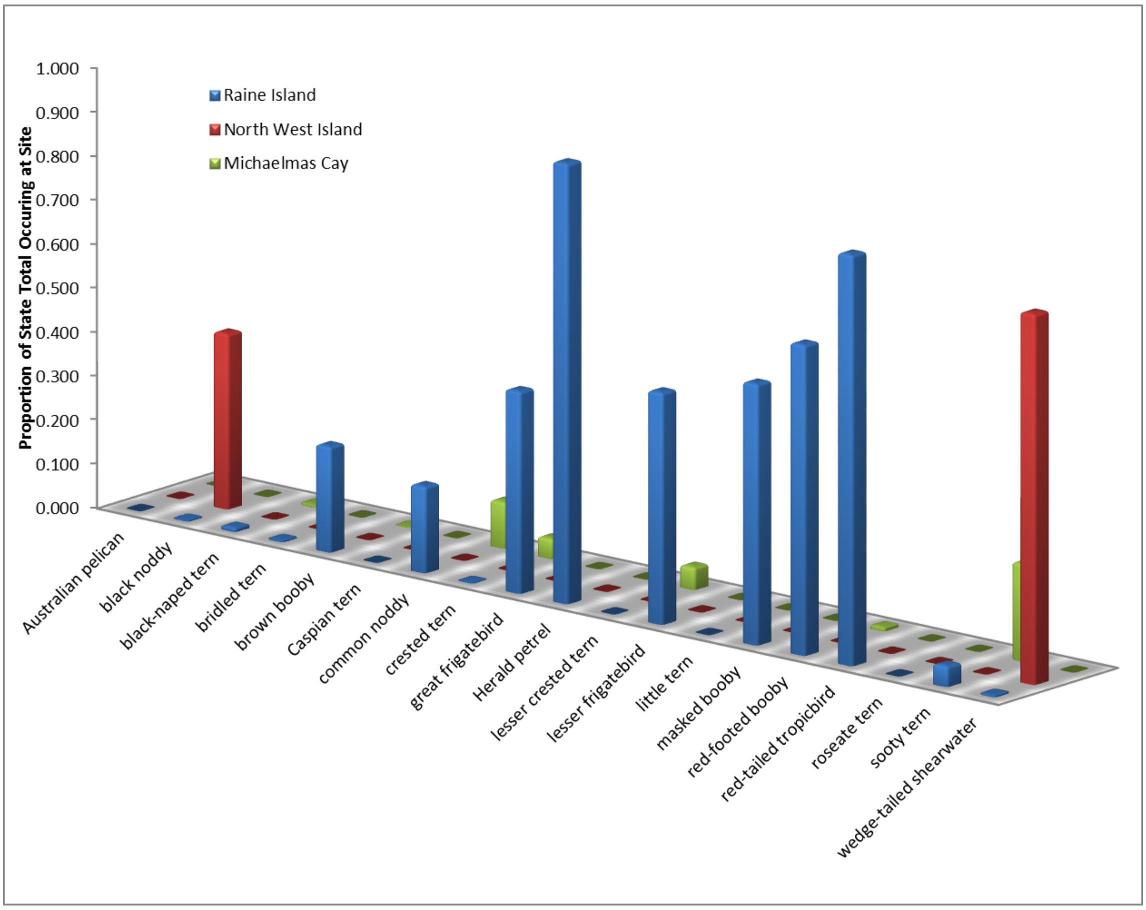


Figure 1: The approximate proportion of Queensland's total of each species breeding at three locations: Raine Island, Michaelmas Cay and North West Island.

Note that while Raine has a much higher proportion of several of the rarer pelagic species Michaelmas has more inshore species that are rare at Raine and North West hosts two species that are rare at Raine.

By querying the data we were able to depict the timing of breeding at a site Figures 2 and 3 and by examining both the importance of breeding events and the timing of breeding we were able to form the basis of a method for selecting sites and visitation strategies.

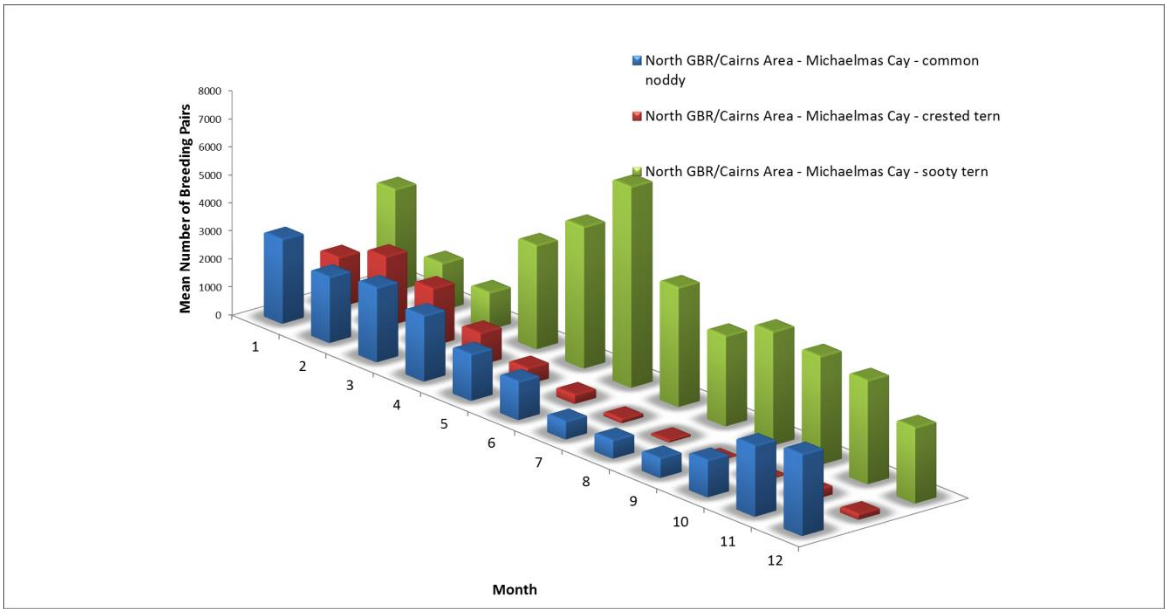


Figure 2: The mean number of breeding pairs of three species recorded at Michaelmas Cay during each month of the year.

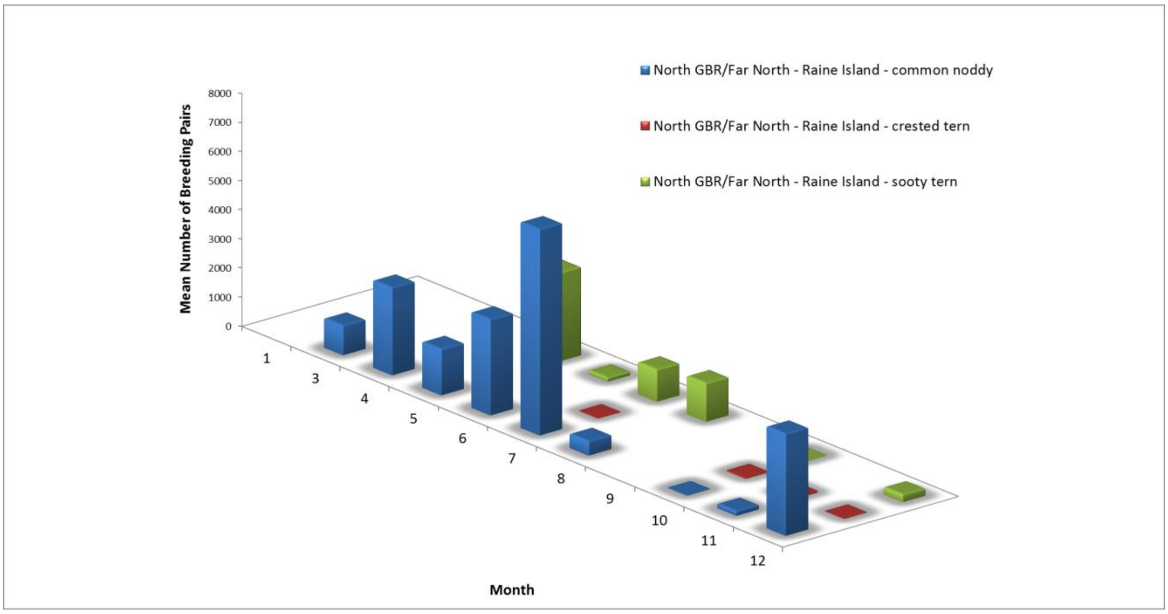


Figure 3: The mean number of breeding pairs of three species recorded at Raine Island during each month of the year.

There are issues with the data which must be clarified for transparency.

1. Many older records provide no numerical estimates for the numbers of breeding birds, recording only presence. These records were not included in the calculations of breeding populations.
2. There are 47 sites with a single count of breeding seabirds and 40 sites with only two records. In some of these cases one of the records is unusually large for the species. These can significantly bias the averages calculated for these sites and it is impossible to know whether these rare observations are truly indicative of the significance of the site or whether they are errors. Most of these outliers were removed from the data after consultation with the original observer or experienced experts.
3. Most sites have not been visited frequently enough to get a good understanding about when breeding occurs. In these cases we assimilated expert knowledge and data from nearby sites into further discussions about the best times to visit a site. Even so, in some cases we may be proven wrong in time.
4. The mean breeding figures are not true mean peak breeding effort figures for each species as the timing of many visits and their records may not have coincided with peaks of breeding. It is not possible with the available data and resources to determine when the species-specific breeding seasons are for each site or to validate whether each data point is likely to fall within a breeding season. This is due partly to a limitation of human resources and partially to an inadequate understanding of whether seasons exists for some species at all and in other cases whether seasons

can be generalised over larger geographic areas from sites where we are better informed.

Notwithstanding these limitations the use of data to inform decision making was viewed as preferable to relying solely on expert opinion. The decision support tool allowed us to apply a consistent level of objectivity to our evaluation of the majority of sites under consideration while expert opinion was often restricted to fewer sites in a more restricted geographic range. This is not to say that the data was better than expert judgement but that it provided a common baseline in more areas than expertise could. Thorough assessments of the data were undertaken by experts to remove or flag suspect data from the system. It is also important to note that the data was used to inform decisions and was not used to determine selections automatically.

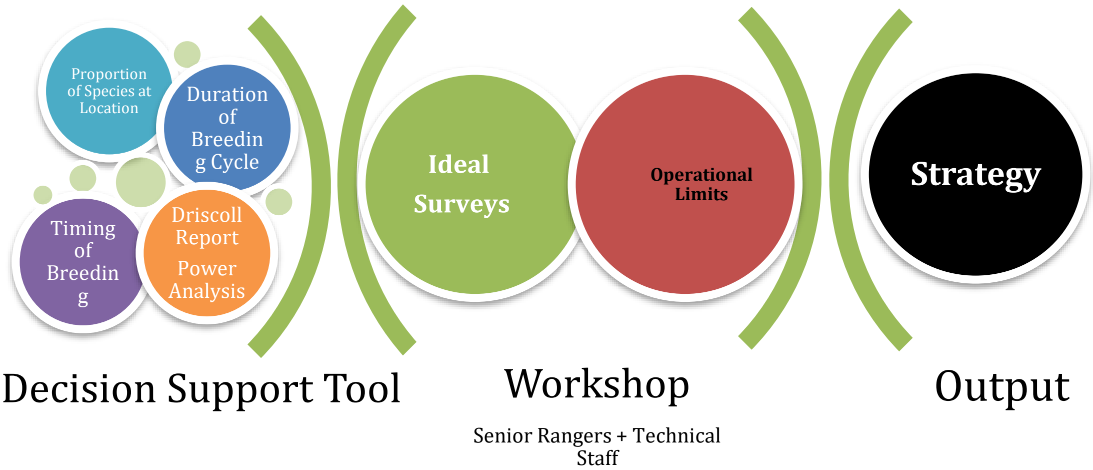


Figure 4: A schematic of the process used to develop the Strategy.

The decision support tool was used to guide discussions at a workshop held in Townsville in September 2014 (see Figure 4). The workshop in Townsville in September 2014 was attended by members of the Ecological Assessment Unit in Operational Support, Senior Rangers and Technical Support staff from the Great Barrier Reef Region, a manager from the Threatened Species Unit and ecologists and managers from the Authority. The group was tasked to create a strategy – using their knowledge of the sites, the decision support tool and their knowledge of operational logistics and vessel tasking – that would allow government agencies to understand and report on the status of seabirds in Queensland.

After the workshop the draft strategy was vetted again by Senior Rangers for feasibility and double checked by the Ecological Assessment Unit staff before being finalised.

How many visits do we need at each site per year?

In trying to answer this question we re-evaluated advice received in Fuller and Dhanjal-Adams' 2012 report that suggested single annual visits were unlikely to be sufficient and that two visits during the breeding season were the best trade-off between effort and data quality (Fuller & Dhanjal-Adams 2012). In re-evaluating this advice we considered that the analysis used data from four smaller species of seabird breeding on Michaelmas Cay. Generally these birds have shorter and less predictable breeding cycles than larger species and repeated site visits simply minimise the risk of arriving before or after a breeding event and recording a false negative. A more thorough consideration of the issue and the supporting logic was extended to all the species covered in this strategy and led us to revise these recommendations for species with different breeding cycles.

Any species' with a breeding cycle (time from nesting to fledging) of three months or less will require two site visits within a six month season (winter or summer) in order to have an acceptable chance of detecting peak breeding events each year. Less frequent visits would result in a high rate of false negatives in the data. In this context a false negative will be when a visit misses the breeding of a species at a location and records a zero when in fact the species has bred before or after the visit.

As the length of a species' breeding cycle increases beyond three months, the chance of missing a seasonal breeding event with a well-timed site visit drops and the value of two seasonal visits is reduced. As such species with cycles longer than three months can be monitored once in a breeding season. It is important to note that no species have four month breeding cycles and the next shortest breeding cycle after three months is five (Figure 5). Most species with short breeding cycles are small inshore foragers (http://www.gbrmpa.gov.au/__data/assets/pdf_file/0003/21729/gbrmpa-VA-InshoreCoastalSeabirds-11-7-12.pdf) and most with longer breeding cycles are larger offshore or pelagic birds (http://www.gbrmpa.gov.au/__data/assets/pdf_file/0013/21730/gbrmpa-VA-OffshorePelagicSeabirds-11-7-12.pdf) (see Table 1).

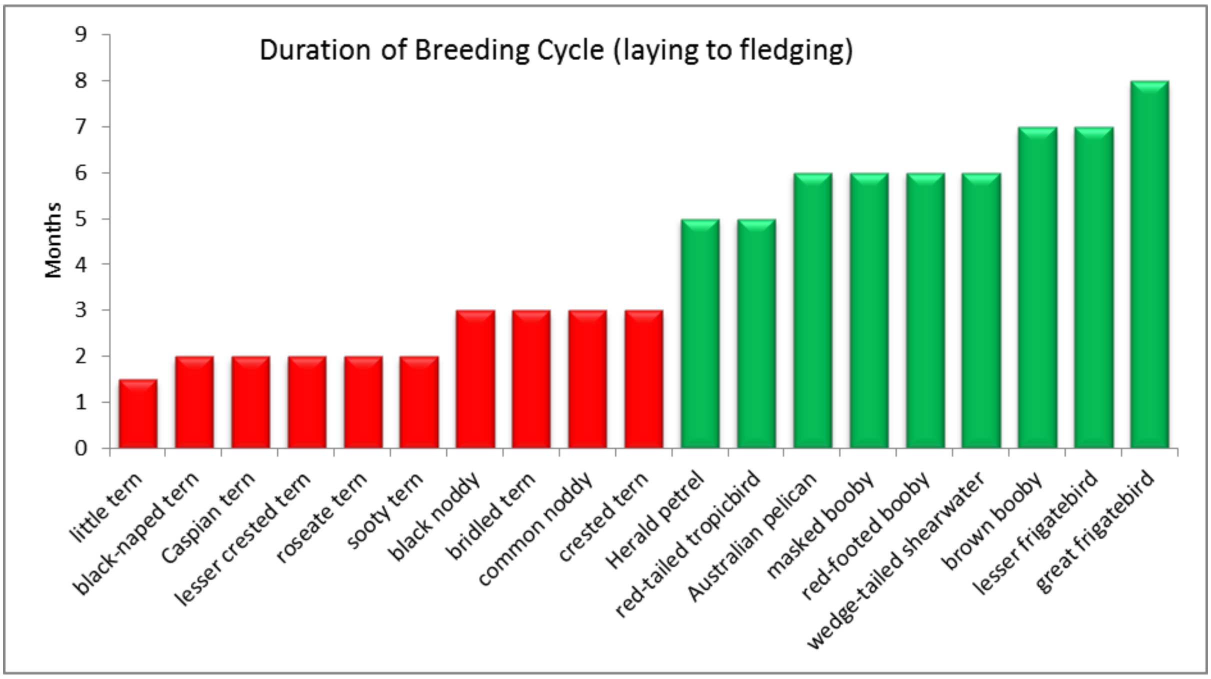
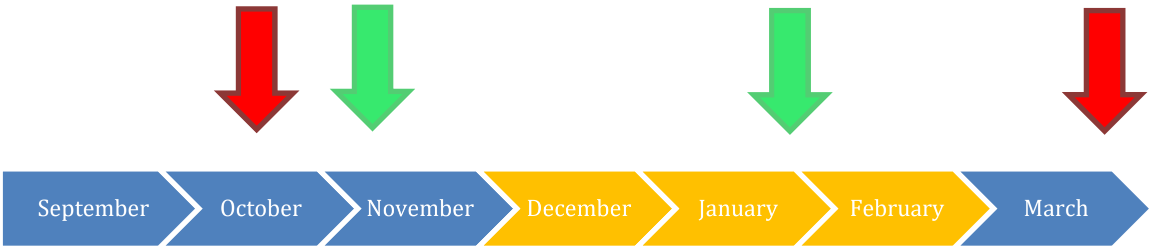


Figure 5: The duration of seabird breeding cycles in months. Red columns indicate species that will require two visits per season.

Despite single seasonal visits being sufficient, for the purpose of monitoring birds with longer breeding cycles (those longer than five months), the timing of visits is still critical. Visits must be targeted at the middle of their known or predicted breeding cycle to ensure that breeding is not missed. If a management unit can do more site visits than the minimum required it would be valuable; both in terms of gathering more robust data and improving our understanding of the timing of breeding events so monitoring effort can be focussed more effectively in the future. As Figure 6a illustrates, having two site visits in a season to detect a species with a three month cycle is counterproductive if the visits are five months apart as an entire breeding event can occur between visits. It is also important not to time site visits too close together. While we have assigned six month seasons to bird breeding (summer and winter), many rookeries may not be conveniently restricted to one season or another. Spacing your visits such that you can catch breeding early or late in the other season is a good tactic to minimise false negatives. In the second example (Figure 6b) the November visit would also detect a breeding event if it was initiated as early as September although a December visit would not.

Visits within any given season in a year visits should be timed so that there are never more months between surveys or before or after surveys within a season than there are months in the breeding cycle of the species of interest. Where possible these spaces should be equal unless more detail is known about the breeding phenology of the species' of interest. This requirement is particularly important when considering rescheduling bird surveys due to competing work priorities or unscheduled interruptions to field management activities.

6a



6b

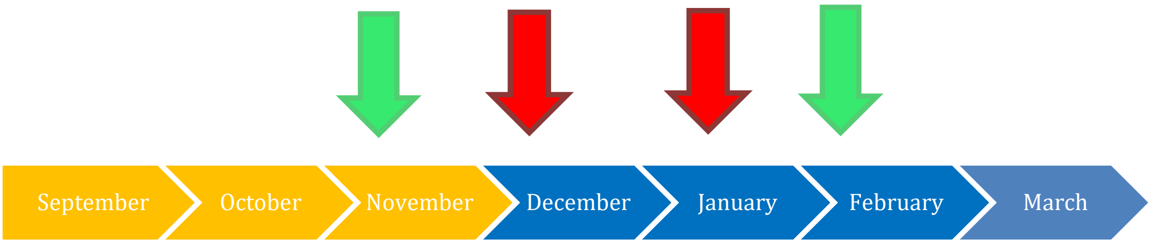


Figure 6a and 6b: How to schedule site visits. Yellow months are when the hypothetical bird bred. Red arrows indicate poorly timed visits and green arrows visits that are far more likely to capture a breeding event.

Although a handful of species have highly predictable breeding events our knowledge of the timing of seabird breeding is often fairly poor, especially for smaller and/or inshore species. Variability and unpredictability complicate any attempts to monitor populations and many seabird species, particularly terns, change both the location and timing of breeding, nesting where and when conditions are right (Palestis 2014). Generally speaking, the smaller the colony the less predictable the breeding behaviour of its residents. At the other extreme, hundreds of thousands of wedge-tailed shearwaters arrive to breed in the Capricornia Cays in mid-October every year. They excavate their nests and mate in November and then in early December leave to forage at sea before returning 7-14 days later to lay eggs (Figure

7). Wedge-tailed shearwaters also exhibit very high site fidelity, returning to the same area of the same island, if not the same burrow, each year to breed. The species is highly predictable in time and space and in these respects at least the wedge-tailed shearwater is a relatively easy species to monitor as we know exactly when we should visit breeding locations to attempt counts.

The broad approach to measuring anything that is variable or unpredictable is to increase the frequency of site visits and the number of sites sampled depending on the nature of the variation expected. With this in mind it is important to recognise that it is currently not feasible to allocate sufficient resources to gather robust data for all species. Our approach to managing these limitations was to select indicator species with broadly similar ecological niches that might indicate trends within a feeding guild of seabirds. These species and their most important breeding sites provide a framework around which to build the Strategy.

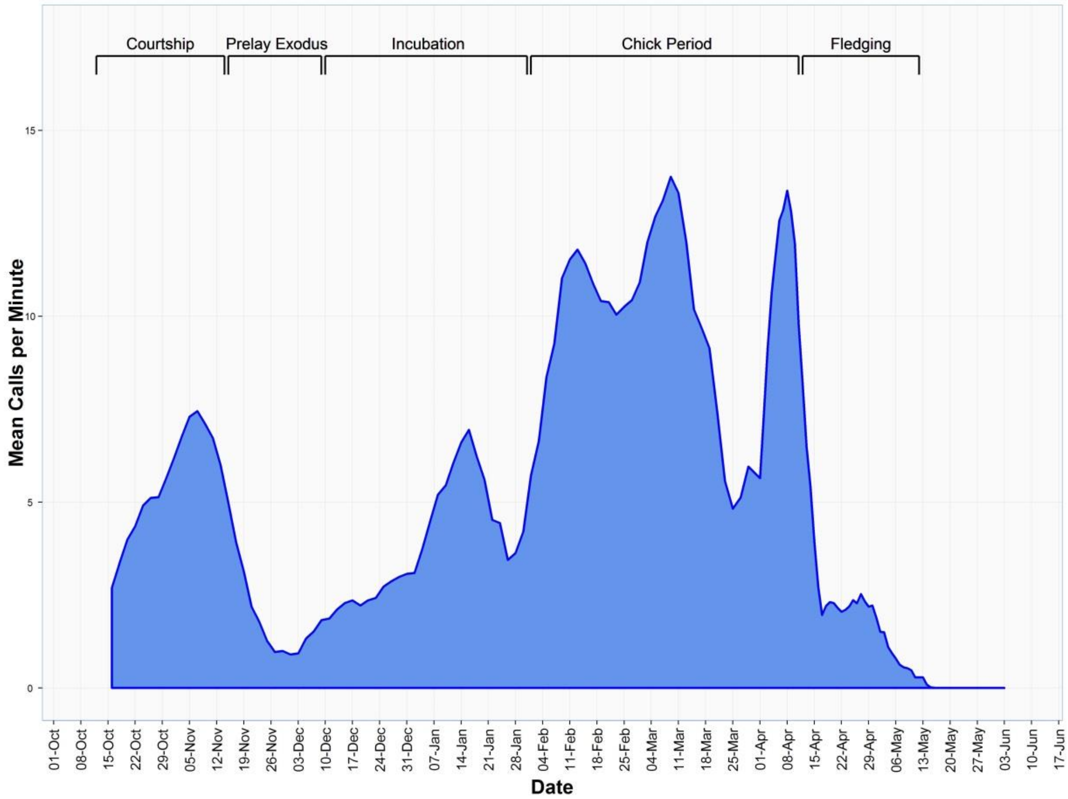


Figure 7: The Breeding Patterns of wedge-tailed shearwaters at North West Island as indicated by Vocal Activity.

Indicator Species

Indicator species were selected through an expert and stakeholder group evaluation of their values as indicators of a particular foraging guild of seabird, their predictability (site fidelity and phenology) and their geographic spread. This process identified four species as broadly representative of coastal, inshore, offshore and pelagic feeding guilds:

Little Terns (*Sternula abifrons*), coastal forager

The little tern was identified as a species of high conservation interest and has only very recently been down-listed from Endangered to Least Concern in Queensland based largely on changes to management elsewhere in Australia. Little terns are widespread in inshore coastal waters and occur in tropical through to warm temperate latitudes from the central Pacific to the west coast of Africa and over much of Europe. It is rated Least Concern by the IUCN with a global population of between 190,000 and 410,000 but is not common anywhere.

While it was recognised that the species has an unusually predictable phenology for a small tern, with most breeding records occurring in November and December, it also exhibits signs of having very low site fidelity, potentially moving between breeding sites in different years. Monitoring the numbers of any species that moves breeding sites is challenging as it could require the visitation of very large numbers of potential sites to ensure that birds are not moving. This is not feasible for a species that is as widespread as the little tern. To ensure that we have useful data on the species it was agreed that we should focus on three broader areas in which little terns breed and gather adequate data from all sites within these areas rather than spreading our effort across sites evenly spread along the coast. We selected areas based on existing projects, operational convenience and the abundance of little terns.

Crested tern (*Thalasseus bergii*), inshore forager

The crested tern is an inshore species that is common, easy to identify, conspicuous and has a longer breeding cycle than most other inshore species. It occurs throughout Queensland’s inshore waters and breeds across a similar geographic range. Globally the species occurs throughout the coastal Indian Ocean and Western Pacific although its population is very poorly understood with an estimated global population of 150,000 - 1,100,000. Its large range and population size and the lack of data to indicate noteworthy downward trends have meant the species is listed as Least Concern on the IUCN’s Red List.

The species appears to breed mainly in summer with peaks in breeding from November till April. However there are records of breeding in every month of year. The crested tern has a three month breeding cycle from laying eggs till fledging of young and breeding colonies typically host hundreds of birds. It is not known how much site fidelity the species shows but the occurrence of regularly occurring large colonies suggests more fidelity than birds that occur in very small ephemeral colonies such as roseate terns.

Brown booby (*Sula leucogaster*), offshore forager

The brown booby is an offshore species that is common, easily identified and occurs in several large breeding colonies from the Capricornia Cays to Raine Island and into the Gulf of Carpentaria. The species is known to exhibit high levels of site fidelity, and has a long breeding cycle and breeding is generally predictably timed (O'Neill et al. 1996). Beyond Queensland the species occurs across northern Australia and across much of the tropical Pacific, Atlantic and Indian Oceans. The global population has been estimated as approximately 200,000 and is thought to be in decline due to nest predation by invasive species and mortality, potentially as fisheries by-catch. Nevertheless the rate of decline is not thought sufficient to list the species as Near Threatened and the IUCN has assessed the species as Least Concern.

In Queensland the species' breeding generally peaks in summer often commencing between August and October and with a breeding cycle that lasts for seven months. Year-round breeding has however been recorded and winter breeding is common in the Swain Reefs (O'Neill et al. 1996, Heatwole et al. 1996). This is notably unusual as breeding on nearby East Fairfax Island in the Capricornia Cays is more typical of elsewhere in the state.

Wedge-tailed shearwater (*Ardenna pacifica*), pelagic forager

The wedge-tailed shearwater is a common and abundant pelagic species which breeds predictably in time and space. In Queensland it breeds from the Sunshine Coast to Raine Island, with the main breeding colonies in the Capricornia Cays. The species occurs throughout the tropical Pacific and Indian Oceans with major Australian breeding populations in Western Australia and Queensland and others in New South Wales. The global population has been estimated at around 5,000,000 and is believed to be declining through predation, exploitation, fisheries by-catch and over exploitation of tuna fisheries. The decline is not thought to be occurring at a rate sufficient for the species to be listed as Near Threatened so is assessed as Least Concern by the IUCN.

In the Capricornia Cays, wedge-tailed shearwater breeding is highly predictable and consistent – starting with courtship in mid-October, egg laying in December and through to fledging in May every year (Figure 6). While less information is available for the rest of the State a similar pattern is anticipated.

While we have structured the strategy around three indicator species, the Authority and the Queensland Government require data on all seabirds encountered, particularly when they are breeding. Some details on the distribution and conservation status of these species is provided at the end of this document and tabulated below (Table 1).

Table 1: Some key attributes of Queensland's seabird species

Species	NCA Status	Back on Track	EPBC Status	IUCN	JAMBA	CAMBA	ROKAMBA	Convention on the Conservation of Migratory species of	Pelagic	Offshore	Inshore	Foraging range from	Breeding Cycle	Global Population	Global Population Trend
Herald petrel	Endangered	Low	Critically Endangered	Least Concern					Yes			?	5	1,500,000	decreasing
red-tailed tropicbird	Vulnerable	Low		Least Concern					Yes	Yes		673	5	32,000	stable
wedge-tailed shearwater	Least Concern	Low		Least Concern	Yes				Yes	Yes		?	6	5,200,000	decreasing
lesser frigatebird	Least Concern	Low		Least Concern	Yes	Yes	Ye s		Yes	Yes		?	7	>10,000[3]	decreasing
great frigatebird	Least Concern	Low		Least Concern	Yes	Yes			Yes	Yes		385	8	>10,000[3]	decreasing

Species	NCA Status	Back on Track	EPBC Status	IUCN	JAMBA	CAMBA	ROKAMBA	Convention on the Conservation of Migratory species of	Pelagic	Offshore	Inshore	Foraging range from	Breeding Cycle months	Global Population	Global Population Trend
masked booby	Least Concern	Low		Least Concern	Yes		Yes		Yes	Yes		196	6	>10,000[3]	decreasing
red-footed booby	Least Concern	Low		Least Concern	Yes	Yes			Yes	Yes		115	6	1,000,000	decreasing
brown booby	Least Concern	Low		Least Concern	Yes	Yes	Yes		Yes	Yes		90	7	200,000	decreasing
sooty tern	Least Concern	Low		Least Concern					Yes	Yes		?	2	21,500,000	unknown
bridled tern	Least Concern	Low		Least Concern	Yes	Yes			Yes	Yes		15	3	750,000	unknown
common noddy	Least Concern	Low		Least Concern	Yes	Yes			Yes	Yes		83	3	640,000	stable

black-naped tern	Least Concern	Low		Least Concern	Yes	Yes				Yes		2	2	>10,000[3]	unknown
lesser crested tern	Least Concern	Low		Least Concern		Yes				Yes		?	2	>10,000[3]	stable
roseate tern	Least Concern	Low		Least Concern	Yes	Yes				Yes		?	2	76,000	unknown
black noddy	Least Concern	Low		Least Concern						Yes	Yes	60	3	>10,000[3]	stable
silver gull	Least Concern	Low		Least Concern						Yes	Yes	?		>10,000[3]	increasing
crested tern	Least Concern	Low		Least Concern						Yes	Yes	15	3	625,000	stable

[3] In cases where the IUCN and Birdlife International has no estimate for a species but knows that it is abundant it uses >10,000 to signify that the species is abundant. This does not mean that there are less black-naped terns (>10,000) than little terns (300,000), for example, we just don't know how many black napes there are.

Caspian tern	Least Concern	Low		Least Concern	Yes	Yes				Yes	Yes	60	2	330,000	increasing
fairy tern[4]	Least Concern	Low	Vulnerable	Vulnerable						Yes	Yes		1.5	5,000	decreasing
Australian pelican	Least Concern	Low		Least Concern							Yes	?	6	190,000	stable
little tern	Least Concern	High		Least Concern	Yes	Yes	Yes	Yes			Yes	4	1.5	300,000	decreasing

[4] Recent observations of fairy terns of the subspecies *exsul* in the Swain Reef's suggest there may be a breeding population in the area but it is not yet confirmed

Site Selection and Visitation Strategy

There have been several valuable syntheses of the important seabird sites in Queensland. Brian King's 1993 and Lavery and Grime's 1971 papers are excellent starting points (King 1993, Lavery & Grimes 1971) as well as the Australian Bird Study Association's series on important seabird islands [5]. These sources and previous monitoring strategies provide a valuable baseline to help validate the outputs of the decision support tool.

The participants of the 2014 Townsville workshop initially used the tool to select sites ensuring an adequate representation for indicator species. Where and when possible, sites were selected to ensure a complete latitudinal (north to south) coverage for each indicator species along the coast. Additional sites were then added to the list based on their value to species that were poorly represented in the initial list. These choices were made based upon the site's significance to an indicator species, the value to other species and the logistical challenges in implementing the desired strategy for that site and as a whole. The list of essential sites and visits that should be completed each year is in Table 2 and the maps of the corresponding sites in Figures 8, 9, 10 and 11.

Some sites identified in previous strategies and publications have not been included in the essential list because of operational constraints. It is critically important for the user of this strategy to understand that important seabird breeding sites that are not in the essential list that can be visited, should be visited. When visited standard seabird counts should be undertaken and assessments of threat levels or other changes made as frequently. These data must be gathered in the same way as for essential sites and incorporated in WildNet. It will be used when analyses are conducted. We have listed the most important of the sites that have not made the essential list as significant sites in Table 3.

The selection of which significant sites to visit in a year should be determined with other operational requirements as part of the business planning cycle. When these sites are visited seabird surveys, using the same methods used for the essential list, should be conducted as well as assessments of threats such as weeds and pests. In the Great Barrier Reef Region these issues should be examined during the Pod natural resource management planning workshops, and in the equivalents South East, and Sunshine and Fraser Coast Regions. Table 3 provides a list of important desirable seabird sites and some guidance as

[5] http://www.absa.asn.au/Seabird%20Islands/List%20of%20Islands/LIST%205a.htm

to their likely significance and the times of year when a visit is most likely to detect seabird species of interest. While we need to keep a level of surveillance on lower priority sites we should be mindful of effort creep – the process by which we add more sites to the essential list and end up being unable to service our consistency requirements.

Little Tern Areas

Because of the potential low site fidelity of little terns we determined that monitoring all potential breeding locations in three larger areas would be a useful approach to establishing their population condition. By sampling in this way we hope to minimise noise in the data caused by birds changing breeding locations due to local condition changes (e.g. spit erosion after storms).

Moreton Bay/Gold Coast

> The Department of Environment and Heritage Protection with assistance from QPWS have taken over the monitoring of a significant little tern breeding area on South Stradbroke Island. DEHP and QPWS staff will continue to monitor these sites until 2019 at least.

Central Queensland

> There are ten sites between Shoalwater Bay and Bundaberg that will likely host breeding colonies of little terns in the summer months. Shared responsibility for visiting these sites has been agreed between staff from the Sunshine and Fraser Coast Region, Great Barrier Reef Region and tour operators based in the Town of 1770.

Tropical Coast

> Twelve sites between the Bowling Green Bay Spit and Lucinda with a high likelihood of hosting little tern breeding colonies have been selected as primary sites. They will be visited by staff from the Great Barrier Reef Region.

Other Tern Areas

Several of the sites on the essential lists are parts of three aggregations of sites that should be monitored collectively if possible. These aggregations are important for tern species (crested, lesser crested, roseate, black-naped) that may change breeding locations between seasons. Using the same logic employed to monitor little terns it is likely that monitoring all sites in these three aggregations will result in more useful data on these species.

Table 2: The list of essential sites and visit timings to be undertaken each year.

The numbers indicate the number of surveys for each green band which in turn represent months that are preferred for surveys. The month in which the number appears is indicative of the best month. Yellow months are less preferred for that survey and to be used only if required (see section “How many visits do we need at each site per year?”).

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
Far North													
Moulter Cay (Raine Island National Park)[6]	1					1					1		**brown booby**, **crested tern**, black noddy, masked booby, sooty tern
Raine Island (Raine Island National Park)[6]	1					1					1		**brown booby, wedge-tailed shearwater**[7]**, crested tern**, red-footed booby, red-tailed tropic bird, Herald petrel, masked booby, common noddy, lesser frigatebird, great frigatebird, black-naped tern, sooty tern, bridled tern, black noddy.

[6] Important Bird Area (IBA) Dutson et al (2009).

[7] No reliable method currently available for monitoring wedge-tailed shearwaters on Raine Island.

Saunders Islet (Saunders Island National Park)	1										1		**crested tern**, black-naped tern , bridled tern, common noddy, lesser crested tern, roseate tern[8], sooty tern
Magra Islet (Saunders Island National Park)	1										1		**crested tern**, lesser crested tern, lesser frigatebird, roseate tern[8]
Sandbank Number 8 (Sandbanks National Park)											1		**brown booby**, black noddy, bridled tern, common noddy, lesser crested tern,
Sandbank Number 7 (Sandbanks National Park)		1									1		**brown booby, crested tern**, black-naped tern, common noddy, lesser crested tern, masked booby, sooty tern
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.

[8] These sites will be targeted for autonomous monitoring technologies because of the logistic difficulty of visiting Magra and Saunders twice in the summer and the significance of these sites to crested and roseate terns.

Pelican Island (Claremont Isles National Park)[9]					1						1		**Australian pelican**, black noddy, black-naped tern, bridled tern, Caspian tern, crested tern, lesser crested tern, little tern, roseate tern
[Cairns to Princess Charlotte Bay]()													
Davie Cay						1						1	**brown booby**, crested tern, black-naped tern, common noddy, **sooty tern**.
Tydeman Island						1						1	**brown booby**, crested tern, black-naped tern, bridled tern, lesser crested tern, **sooty tern**.
Sandbank Number 1 (Sandbanks National Park)						1							**brown booby**, lesser crested tern, masked booby.
Stapleton Island (Howick Group National Park)[4]						1							**brown booby, crested tern**, Australian pelican, black-naped tern, bridled tern, common noddy, lesser crested tern, lesser frigatebird, **great frigatebird,** red-footed booby, sooty tern.

[9] It is anticipated that the Lama Lama Rangers will take over this responsibility from QPWS after receiving training. Andrew Simmonds at the Great Barrier Reef Marine Park Authority has taken the lead on this.

Combe Island (Howick Group National Park)	1										1		**crested tern**, **wedge-tailed shearwater,** Australian pelican, black noddy, bridled tern, common noddy, lesser crested tern, roseate tern,.
Eagle Island (Lizard Island National Park)	1										1		**crested tern**, black-naped tern, bridled tern, lesser crested tern , roseate tern, sooty tern.
Low Wooded Island	1										1		**crested tern**, black-naped tern, bridled tern, roseate tern.
Michaelmas Cay[4]	1	1	1	1	1	1	1	1	1	1	1	1	**crested tern, brown booby**, black-naped tern, common noddy, lesser crested tern, sooty tern

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	**Seabird species recorded breeding at each site with those prioritised for the site in bold.**
North Tropical Coast													
North Brook Island (Brook Islands National Park)[4]		1									1		**crested tern**, black-naped tern, birdled tern, lesser crested tern, roseate tern
Hull Heads	1										1		**little tern**
Tully Heads	1										1		**little tern**
Murray River	1										1		**little tern**
Dallachy Creek	1										1		**little tern**
Wreck Creek	1										1		**little tern**

Pig Creek	**1**										**1**		**little tern**
Damper Creek	**1**										**1**		**little tern**
Lucinda	**1**										**1**		**little tern**
Gentle Annie Creek	**1**										**1**		**little tern**
Taylors Beach North	**1**										**1**		**little tern**
Taylors Beach	**1**										**1**		**little tern**
Cassady Beach	**1**										**1**		**little tern**

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
Bulgaroo/Palm Creek	1										1		little tern
Orient Creek	1										1		little tern
Eleanor/Cattle Creeks	1										1		little tern
Insulator Creek	1										1		little tern
Crystal Creek	1										1		little tern
Bowling Green Bay[10]	1										1		little tern
Bowling Green Bay Spit	1										1		little tern

[10] Bowling Green Bay includes four sites Bowling Green Bay A, B and C and Sheepwash Creek (see Figure 9)

Whitsundays													
Eshelby Island		1								1			**crested tern**, bridled tern, lesser crested tern

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
Swain Reefs[6]													
Bell Cay (Swain Reefs National Park)												1	**brown booby, crested tern,** bridled tern, common noddy, lesser crested tern, **lesser frigate**, masked booby, sooty tern.
Thomas Cay (Swain Reefs National Park)						1						1	**brown booby, crested tern**, black-naped tern, bridled tern, common noddy, lesser crested tern, **masked booby**
Bacchi Cay (Swain Reefs National Park)						1						1	**brown booby, crested tern**, common noddy, lesser crested tern, **masked booby**
Frigate Cay (Swain Reefs National Park)						1						1	**brown booby, crested tern**, black-naped tern, bridled tern, common noddy, lesser crested tern, **masked booby**, roseate tern
Bylund Cay (Swain Reefs National Park)						1						1	**brown booby**, black-naped tern, common noddy, **masked booby**, roseate tern

Price Cay (Swain Reefs National Park)						1						1	**brown booby, crested tern**, black-naped tern, bridled tern, common noddy, lesser crested tern, **masked booby**, roseate tern, sooty tern
Gannet Cay (Swain Reefs National Park)						1						1	**brown booby, crested tern**, black-naped tern, bridled tern, common noddy, lesser crested tern, **masked booby**
Shoalwater Bay													
Akens Island	1												**Australian pelican**
Pelican Rock	1												crested tern, **Australian pelican, Caspian tern**

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
Capricornia Cays[6]													
North West Island (Capricornia Cays National Park)	[11]	1			1			1			1		wedge-tailed shearwater, black noddy
Wilson Island (Capricornia Cays National Park)		1									1		wedge-tailed shearwater, **black-naped tern**, bridled tern, **roseate tern**
One Tree Island (Capricornia Cays National Park)		1									1		**crested tern**, black noddy, **black-naped tern**, bridled tern, lesser crested tern, **roseate tern**
Mast Head Island (Capricornia Cays National Park)	[11]	1			1			1			1		**crested tern**, **black-naped tern**, black noddy bridled tern, **roseate tern**

[11] Acoustic monitoring is currently being evaluated as an alternative to the physical surveys of wedge-tailed shearwaters and black noddies. If proved useful this system will replace most regular summer surveys of wedge-tailed shearwaters in 2016 or 2017.

East Fairfax Island (Capricornia Cays National Park)												1	**brown booby**, crested tern, black-naped tern, lesser crested tern, roseate tern
Lady Musgrave Island (Capricornia Cays National Park)	[11]	1			1			1			1		black noddy, wedge-tailed shearwater, **black-naped tern**, bridled tern, **roseate tern**
Lady Elliot Island (Capricornia Cays National Park)		1									1		**crested tern,** wedge-tailed shearwater**, black-naped tern**, bridled tern, **common noddy**, **red-tailed tropicbird, roseate tern**, **sooty tern**

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
Southern Reef Coastal													
West Point Roost, Port Clinton	1										1		**little tern**
Corio Bay (Great Barrier Reef Coast Marine Park)	1										1		**little tern**
Jenny Lind Creek, Bustard Bay (Eurimbula Regional Park)[12]	1										1		**little tern**
Middle Creek, Bustard Bay (Eurimbula National Park)[12]	1										1		**little tern**

[12] Will be monitored by Larc Tours in the Town of 1770

Round Hill Creek, Bustard Bay (Eurimbula National Park)[12]	**1**										**1**		**little tern**
Eurimbula Creek, Bustard Bay (Eurimbula National Park)[12]	**1**										**1**		**little tern**
Baffle Creek	**1**										**1**		**little tern**
Skyring/Barubbra East	**1**										**1**		**little tern**
Skyring/Barubbra West	**1**										**1**		**little tern**
Dr May’s Island, Elliot River	**1**										**1**		**little tern**

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
Moreton Bay and Stradbroke Islands													
Bribie Island (North tip) (Bribie Island National Park)	1										1		**little tern**
North Moreton (Moreton Island National Park)	1										1		**little tern**
Mirapool Beach (Moreton Island National Park)	1										1		**little tern**
North Stradbroke (Swan Bay)	1										1		**little tern**
South Stradbroke (Northern tip)	1										1		**little tern**

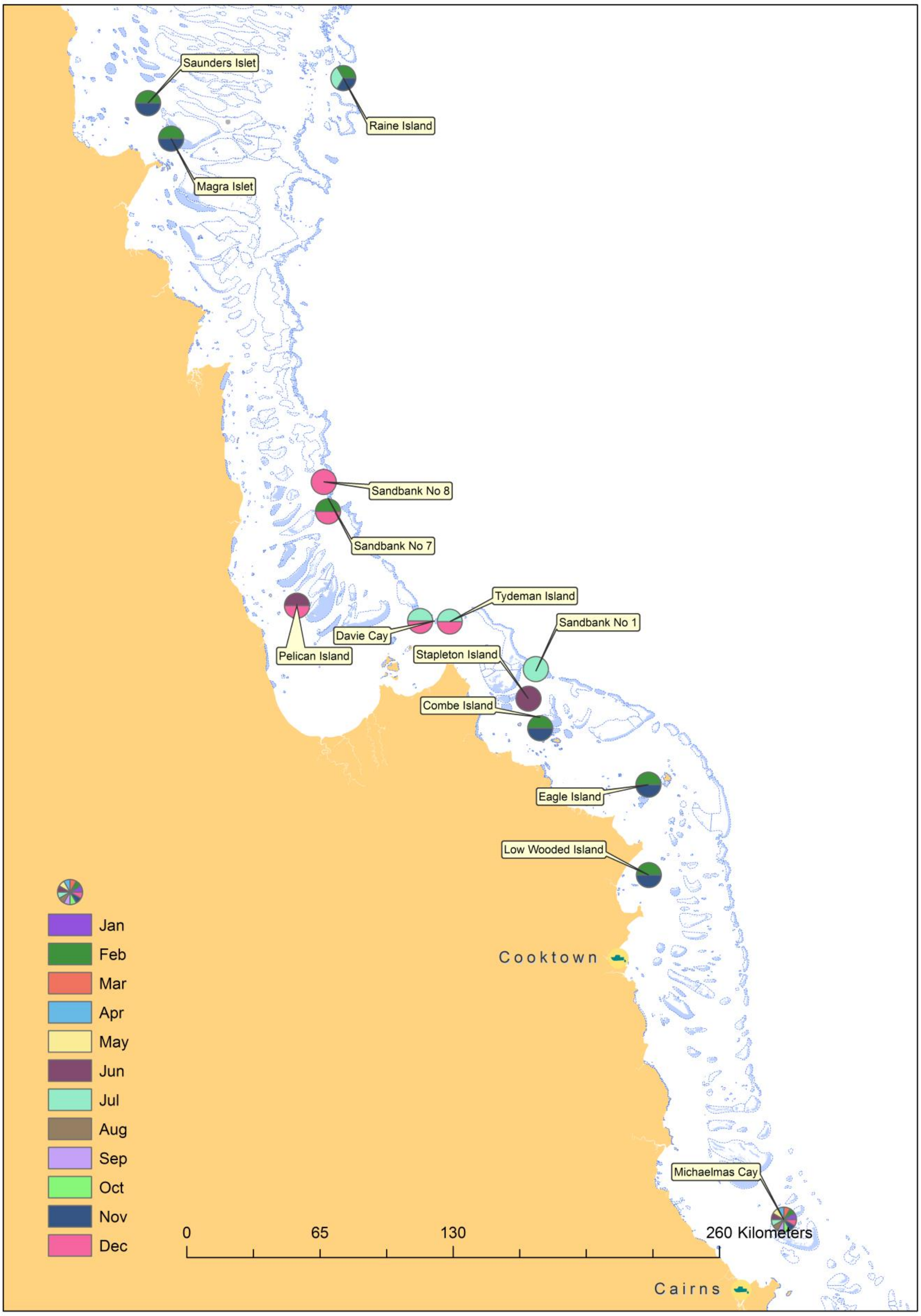


Figure 8: A map of the essential sites from Cardwell to Moulter Cay and the months in which they should be surveyed.

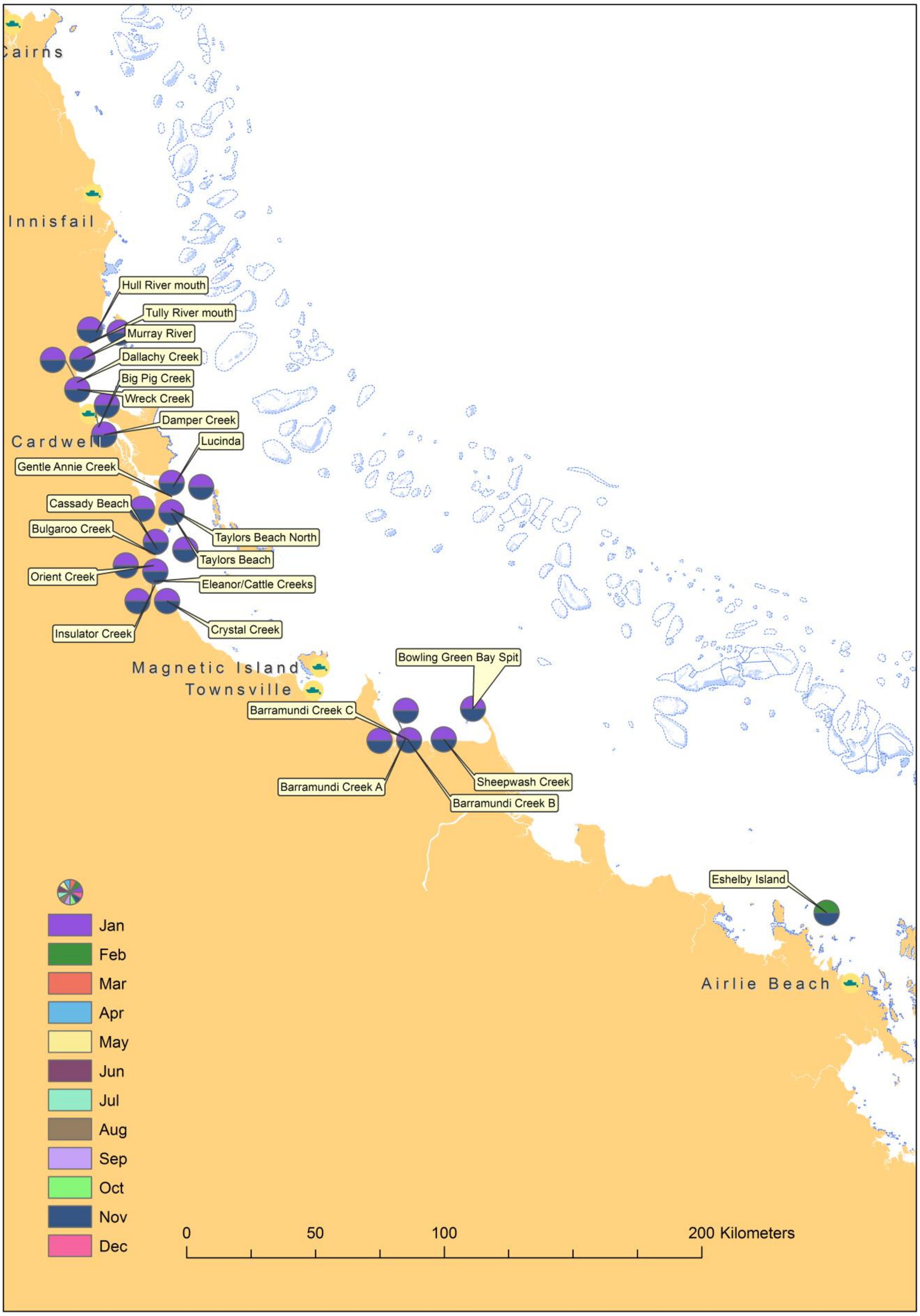


Figure 9: A map of essential sites from Eshelby to North Brook and the months they should be surveyed in.

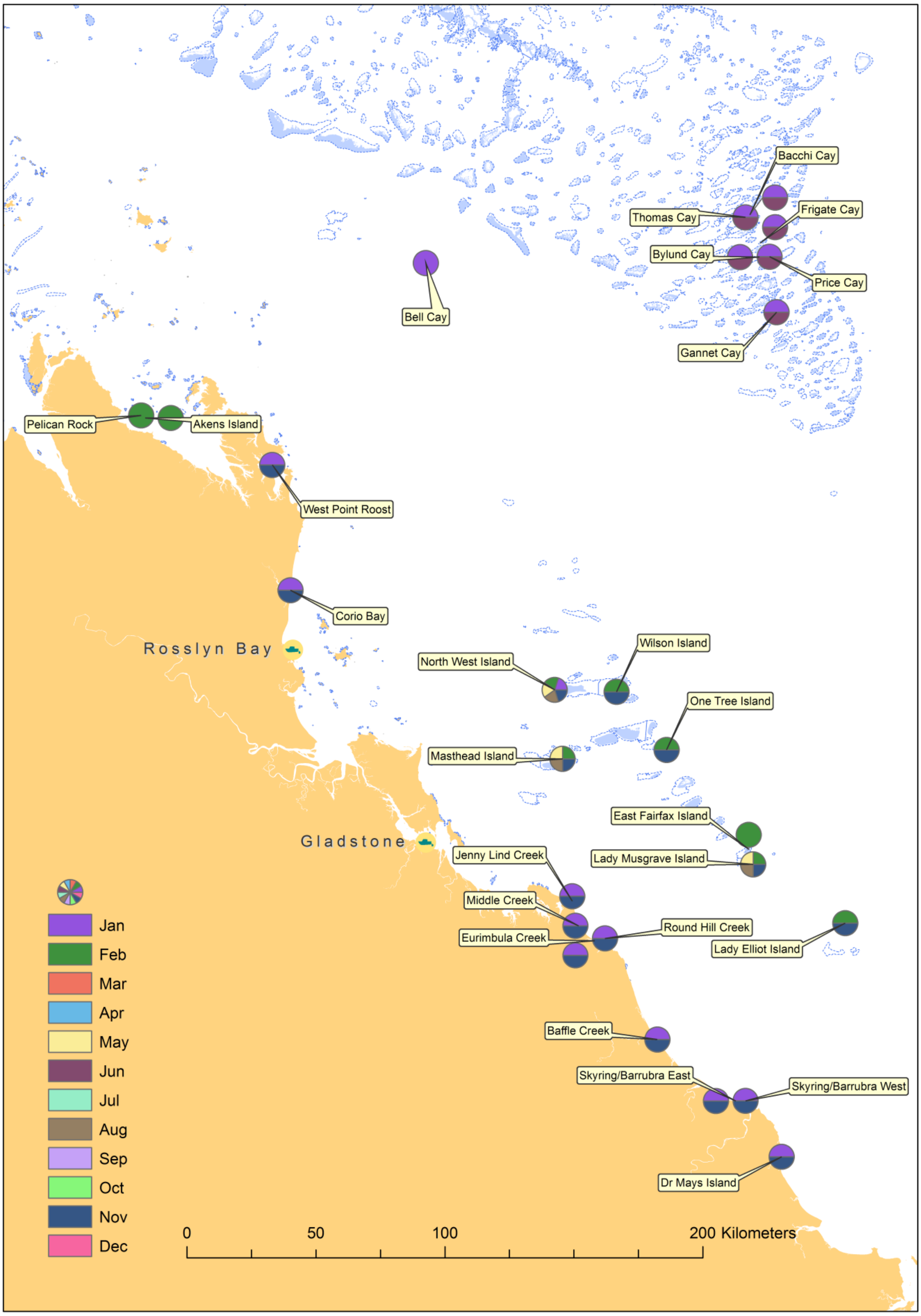


Figure 10: A map of essential sites from Dr Mays Island to Bacchi Cay and the months they should be surveyed in.

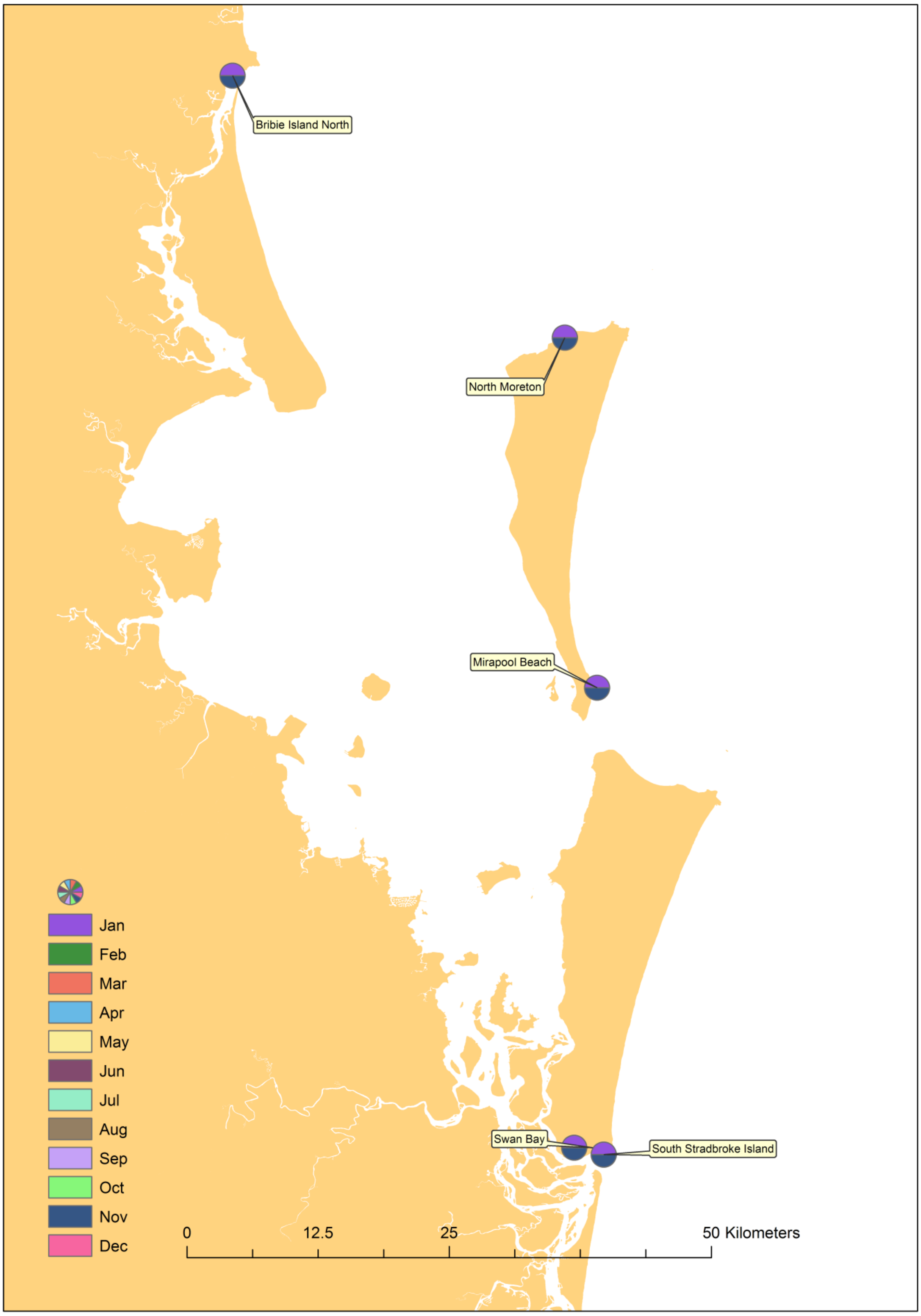


Figure 11: A map of essential sites from South Stradbroke Island and Bribie Island and the months they should be surveyed in.

Table 3: Significant seabird breeding sites and recommended visits.

Italicised sites are not previously included in the essential sites above (e.g. West Fairfax Island), sites in standard font (e.g. East Fairfax Island) are already essential sites but have a new schedule which is preferred to that in the essential list. The numbers indicate the number of surveys for each green band which in turn represent months that are preferred for surveys. The month in which the number appears is indicative of the best month. Yellow months are less preferred for that survey and to be used only if required (see section "How many visits do we need at each site per year?").

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
Far North													
Womer Cay[13]		1									1		**brown booby, crested tern,** Australian pelican, black noddy, black-naped tern, bridled tern, common noddy, lesser crested tern, lesser frigatebird, sooty tern,

[13] Important Bird Area (IBA) Dutson et al (2009)

Wallace Islet (Denham Group National Park)[14]		1				1					1		crested tern, black noddy, black-naped tern, bridled tern, common noddy, lesser crested tern, **roseate tern**, sooty tern
Mitirinchi (Quoin) Island[15]						1						1	crested tern, **brown booby,** black-naped tern, black noddy, bridled tern, common noddy, **greater frigatebird**, **lesser frigatebird,** sooty tern.
Lowrie Islet[16]		1									1		black-naped tern, bridled tern, **lesser crested tern**, little tern

[14] Three of the largest 10 counts of roseate terns 16 records of seabird breeding; no counts since 1996. Records of breeding in June and December.

[15] More than 1/3rd of the east coast's lesser frigatebird, and 1/10th great frigatebird breeding records. Breeding "peaks" in June/July and November/December likely reflecting operational convenience rather than a seasonal pattern. Eight other species use the island.

[16] Six visits since 1990 all in summer may host a large colony of lesser crested terns (2,900 pairs in 1999)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
Cairns to Princess Charlotte Bay													
Ingram Island (Howick Group National Park)[17]		1									1		**crested tern**, black-naped tern, bridled tern, lesser crested tern, little tern, **roseate tern**
Nymph Island[18]		1			1			1			1		**crested tern, black-naped tern**, bridled tern,
Turtle Group (Number 3, 5 and 6) (Turtle Group National Park)[18]		1			1			1			1		**crested tern,** black-naped tern, bridled tern, **lesser crested tern, little tern, roseate tern**

[17] Five visits since 1994; all breeding records in summer.

[18] AGGREGATION: Nymph and the Turtle Group are a potentially valuable cluster of islands for monitoring the smaller tern species. Breeding appears to be mainly in summer but there are records in winter records; access each site four times a year until we are able to ascertain the timing of breeding with more accuracy.

Two Islands (Three Islands Group National Park)[19]		1									1		**black-naped tern**, bridled tern, lesser crested tern, **little tern, roseate tern**, sooty tern
Sudbury Cay[20]		1									1		**crested tern,** black noddy, black-naped tern, common noddy, lesser crested tern, sooty tern

[19] Five visits 1989-1999; all December

[20] Ten visits from 1983-1989 that recorded significant crested and lesser crested tern breeding. Most visits were in summer, the exception being one in April during which only common noddies were recorded. Some evidence it may be occasionally inundated.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	**Seabird species recorded breeding at each site with those prioritised for the site in bold.**
North Tropical Coast													
Mound (Purtaboi) Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Dunk Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Mung Um Gnackum[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Kumboola Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Woln Garin Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern

[21] AGGREGATION: The Islands between Mound (Purtaboi) and Hudson (Coolah) support good breeding populations of terns and may make another good aggregation to monitor as a unit.

Richards (Bedarra) Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Pee Rahmn Ah (Battleship) Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Wheeler (Toolgbar) Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Smith (Kurrumbah) Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Coombe Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Seabird species recorded breeding at each site with those prioritised for the site in bold.**
Bowden (Budg-Joo) Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Hudson (Coolah) Island[21]		1									1		crested tern, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern

Swain Reefs[6]													
Bell Cay (Swain Reefs National Park)		1				1					1		**brown booby, crested tern,** bridled tern, common noddy, lesser crested tern, **lesser frigate**, masked booby, sooty tern.
Thomas Cay (Swain Reefs National Park)		1				1					1		**brown booby, crested tern**, black-naped tern, bridled tern, common noddy, lesser crested tern, **masked booby**
Bacchi Cay (Swain Reefs National Park)		1				1					1		**brown booby, crested tern**, common noddy, lesser crested tern, **masked booby**
Frigate Cay (Swain Reefs National Park)		1				1					1		**brown booby, crested tern**, black-naped tern, bridled tern, common noddy, lesser crested tern, **masked booby**, roseate tern
Bylund Cay (Swain Reefs National Park)		1				1					1		**brown booby**, black-naped tern, common noddy, **masked booby**, roseate tern
Price Cay (Swain Reefs National Park)		1				1					1		**brown booby, crested tern**, black-naped tern, bridled tern, common noddy, lesser crested tern, **masked booby**, roseate tern, sooty tern

Gannet Cay (Swain Reefs National Park)		1				1					1		**brown booby, crested tern**, black-naped tern, bridled tern, common noddy, lesser crested tern, **masked booby**
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	**Seabird species recorded breeding at each site with those prioritised for the site in bold.**
Capricornia Cays[22]													
North Reef Island (Capricornia Cays National Park) [22]		1									1		**crested tern, black-naped tern**, bridled tern, **lesser crested tern**, little tern, **roseate tern**
Tryon Island (Capricornia Cays National Park)[22]		1									1		**crested tern, black-naped tern**, bridled tern, **lesser crested tern**, little tern, **roseate tern**
Broomfield Cay (Capricornia Cays National Park)[22]		1									1		**crested tern, black-naped tern**, bridled tern, **lesser crested tern**, little tern, **roseate tern**

[22] AGGREGATION: The Capricornia Cays support large numbers of nesting terns and there may well be significantly more movement between islands in the archipelago than to sites further afield. As such if it is possible to count all cays in the area twice during a summer this would be useful.

Wreck Island (Capricornia Cays National Park)[22]		1									1		**crested tern, black-naped tern**, bridled tern, **lesser crested tern**, little tern, **roseate tern**
Heron Island (Capricornia Cays National Park)[22]		1									1		**crested tern, black-naped tern**, bridled tern, **lesser crested tern**, little tern, **roseate tern**
Erskine Island (Capricornia Cays National Park)[22]		1									1		**crested tern, black-naped tern**, bridled tern, **lesser crested tern**, little tern, **roseate tern**
East Hoskyn Island (Capricornia Cays National Park)[22]		1									1		**brown booby, crested tern**, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Seabird species recorded breeding at each site with those prioritised for the site in bold.
West Hoskyn Island (Capricornia Cays National Park)[22]		1									1		**brown booby, crested tern**, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
East Fairfax Island (Capricornia Cays National Park)[22]		1									1		**brown booby, crested tern**, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
West Fairfax Island (Capricornia Cays National Park)[22]		1									1		**brown booby, crested tern**, black-naped tern, bridled tern, lesser crested tern, little tern, roseate tern
Keppel Bay													
Creek Rock					1			1					**Caspian tern**[23]

[23] Three records of breeding Caspian terns in July 1995 and 1996.

Sunshine Coast													
Mudjimba Island[24]												1	wedge-tailed shearwater

[24] A survey will have to be designed before this is attempted. Another potential candidate for acoustic monitoring.

Threats

While this strategy is principally about where and when we should count seabirds we must also be observant for other issues at seabird sites. Seabirds are highly susceptible to invasive species and lack many of the adaptations of terrestrial bird species to predators or the flexibility to select new sites in response to weed infestations or fire modified habitats. Rats, cats and rabbits rate as the worst invasive threats to seabird populations, but anything that may modify the habitat, compete for nesting species or destroy nests, eggs, chicks or adults is potentially important. Field staff must also be mindful that native species can also be invasive when they appear in an area that they do not normally occur in. A spotted quoll or Tasmanian devil might pose as significant a threat to a colony of nesting seabirds on an island as a cat or rat might. Nesting birds may also be susceptible to disturbance by people, vehicles or domestic animals.

Staff must be observant for potential threats while conducting seabird surveys. It is generally significantly easier to prevent a small outbreak from spreading through early action than it is to control an outbreak once it is well established. When at a site allow time to look around for evidence of invasive vertebrates. As most are nocturnal these may be signs of feeding, droppings or tracks. Make time to take a look at the flora, ask yourself are these species all native to this area? Are there signs of disturbance such as vehicle, foot or dog tracks or is there evidence of nest predation?

Threats need not be obvious or immediate. Is the site at risk of being immersed; are nests or cadavers filled with plastic debris that might have choked birds? Are those ants benign natives or are they red imported fire ants? There are many potential threats to seabird breeding colonies and many are hard to detect. Staff should not be encouraged to get the seabird survey done as rapidly as possible, but rather to make as detailed a site inspection as possible within operational constraints.

If you observe or suspect that a threat is increasing or imminent then make a note on the data sheet and contact the regional seabird thematic coordinator at your earliest convenience to ensure that the threat is given further consideration.

Quality Control and Capacity

While adhering to this Strategy will take us a significant way towards gathering useful data it is only one aspect of the task. We also need to maintain the quality of data being entered into WildNet to ensure that subsequent analyses are not biased by poor counts, misidentified species or incorrect site allocation.

Quality control of the information gathered will occur at several levels:

1. Staff training will be rolled out to all staff who gather coastal bird data. It is a requirement that someone with this training should be present within any team that records coastal bird data.
2. Standard operating procedures and guides will be created to provide staff with guidance as to how an individual site or a type of site (e.g. non-vegetated cay) should be approached. These guides can be taken into the field.
3. Ongoing mentoring and validation of the currency of skills will continue after training. This roll out will include an expectation that staff will either retrain or be signed off as retaining competency by the program officer (Andrew McDougall).
4. Data entered or to be entered into the corporate databases (Wetland Information Capture System and Wild Net) will continue to be validated by regional coastal bird coordinator and the program officer.

Accommodating Change and Uncertainty

In response to shortcomings in the rigour previously applied to seabird monitoring, this strategy prescribes an essential list of sites that must be visited consistently. Keeping the timing of visits consistent and using the same methods during each site visit will help ensure that the data gathered will provide a realistic indication of what is happening to seabird populations in the Great Barrier Reef and eastern Queensland. Visiting a small number of sites regularly and frequently has limitations – one being that a narrower focus may result in changes being missed elsewhere.

In the context of seabirds we should expect change. Coastlines and islands are dynamic places and significant changes can occur over a short period of time. Cays and spits are frequently eroded or accreted by cyclones and king tides and may appear in areas for which we have no previous records. In areas that do not favour rapid sand accretion, cays may not reform or may take years to do so. Several significant breeding cays have already disappeared. Following inundation Upolo, Maclennan and Beaver Cays no longer host significant breeding rookeries and Michaelmas Cay appears to be eroding away.

Management actions may also change the significance of a seabird breeding site. Recent interventions to eradicate rats at Boydong Island and to manage Guinea grass at Three Isles may lead to significantly more use of these sites by seabirds in the future. Sand replenishment at Raine Island may alter the breeding habitat of common noddies and red-tailed tropicbirds.

These sorts of fluctuations may result in significant changes to the seabird breeding landscape that would go unnoticed if we do not make an effort to spread our surveillance beyond the essential sites. To meet these ends we propose that other valuable seabird breeding sites, as identified in Table 3, are visited as frequently as possible and at a minimum once every five years during the most likely breeding season. Other sites for consideration can be found in King (1993) and Lavery and Grimes (1971). However these sources and our secondary site list will not include sites which we have never visited or newly formed cays or spits.

One area of interest in this regard is an expanse of approximately 70,000 square kilometres of offshore reef area between the Swain Reefs and Hinchinbrook Island from which we have no records of seabird breeding despite the occurrence of several cays (Figure 12). A rapid approach to assessing whether potential features are significant breeding sites would be to observe potential sites from an aircraft during high tides within seabird breeding seasons. Further effort might also be directed to identifying other potential sites from remote sensed data (Bob Beaman *pers comm*).

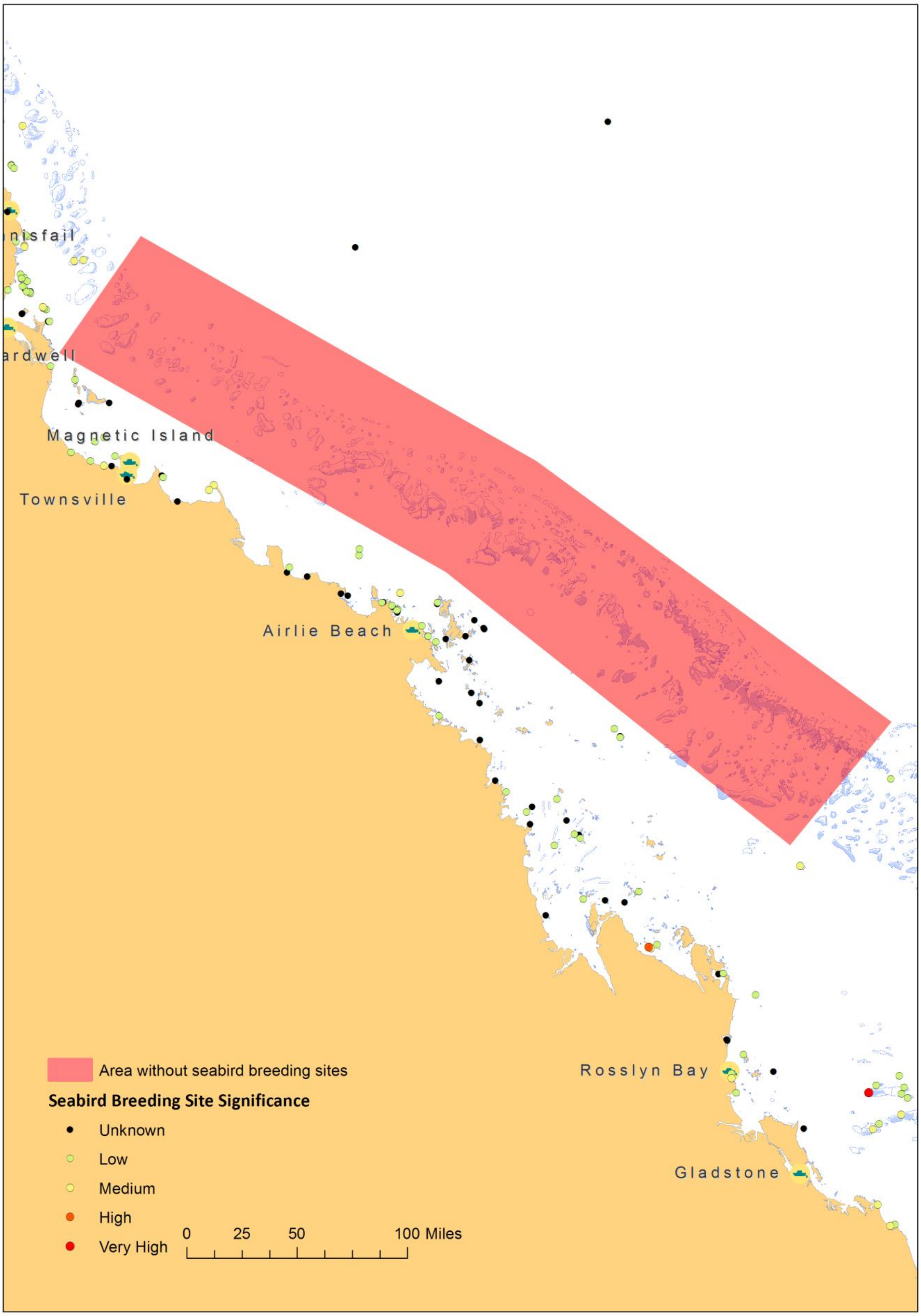


Figure 12: The area between the Swain Reefs and Hinchinbrook Island

Governance and managing changes to the strategy

This Strategy promotes the value of consistency above all else. However changes in the physical, ecological and political environment may result in a need to alter the sites we visit and/or the timing of these visits. If change is required it must be managed to preserve objectives of the Strategy. Any modification must be approved by **a minimum group of staff responsible for the management of the program**. This quorum will compromise of the program manager, Technical Services (currently Graham Hemson) and Principal Rangers or their equivalent from the regions concerned, and the Manager of the Field Management Program. Proposed changes should be directed to the program manager for consideration by this group. These will be considered in April each year unless an urgent case for change is apparent and communicated by the proponent. This timing will allow changes to be factored into planning for the following financial year and allow information from the summer breeding season to be incorporated into any suggestions.

Using the data

The value of seabird data will grow with time and the accumulation of baseline and trend information. Until our baseline grows there are only a handful of sites (e.g. Michaelmas Cay and the Capricornia Cays) for which we have sufficient data to make any confident comparisons. Notwithstanding these initial limitations, the Ecological Assessment Unit and regional staff dedicated to managing the implementation of seabird monitoring will provide annual summaries of the data gathered at the end of each financial year. Notable observations and any emerging patterns will be highlighted in these summaries.

In 2019 a more thorough review of the Strategy and the data gathered will be conducted to coincide with the next Great Barrier Reef Outlook Report. This review will establish whether the data being gathered is of the required quantity and quality and identify problems and propose solutions. The review will provide guidance as to the sensitivity and precision of species specific condition and trend data and will likely require the contracting of a statistician. It will also examine any data from lower priority sites to assess whether changes to the essential site list should be considered. The results of the review will determine any significant changes to the strategy before its renewal.

Subsequent minor analyses of the data will occur annually with trend information for sites of interest being updated and reporting on adherence to the Strategy. Major analyses of the data from all seabird monitoring described in this strategy will occur every five years after 2019.

Other Species

Bridled Terns (*Onychoprion anaethetus*)

Bridled terns are widespread seabirds generally found within 15 kilometres of land and in close association with continental land masses throughout the tropics. While little is known about their numbers they occur widely in tropical and temperate waters in all major ocean systems and are assessed as Least Concern by the IUCN. They frequently nest in vegetation that precludes observers from counting aggregations without disturbing them. This has led to the development of some innovative attempts to estimate the numbers of bridled terns at large colonies including drive-by photography of flocks of birds spooked into flight by the presence of a vessel. These methods are perhaps only useful in detecting extremely large changes in the size of breeding populations such as near complete breeding failures as we don't know what proportion of the birds take-off and whether this is consistent between surveys and locations. If we want to monitor bridled terns meaningfully then better methods need to be identified.

Sooty tern (*Onychoprion fuscatus*)

The sooty tern is the most globally widespread and abundant seabird that breeds in Queensland. The global population is estimated at more than 21 million and it occurs in tropical and sub-tropical oceans across the globe. While there is no certainty about how the population is tracking it is assessed as Least Concern by the IUCN by virtue of the size of it range and population. It is thought to be highly pelagic when not breeding and nests in more open habitat than the very similar bridled tern. Nevertheless in areas of low shrubs and long grass the species may remain a challenging prospect to count reliably because it occurs in such large numbers. It is unusual in Queensland as while most species breed more often in summer there are larger numbers of sooty terns breeding in winter months. However they can breed year round.

Black noddy (*Anous minutus*) and common noddy (*Anous stolidus*)

The black and common noddies are small offshore seabirds that feed by surface dipping for bait fish. While very similar in appearance and feeding strategies, they use very different habitats for breeding. Black noddies nest in trees or shrubs on vegetated cays and common noddies generally nest on the ground on unvegetated or sparsely vegetated cays. Both species form quite large colonies with black noddies forming very large breeding colonies from September until May in the Capricornia Cays. Common noddies are more widespread and are more common in the north of the state. In the lower latitudes such as Michaelmas Cay and Swain Reefs, common noddies appear to breed predominately, but not exclusively, in summer while at Raine Island they breed mainly in winter. Black noddies are slightly more predictable summer breeders. Both species are widespread in tropical and sub-tropical waters in all major ocean systems and are listed as Least Concern by the IUCN.

Breeding black noddies may be difficult to count, especially in larger colonies such as in the Capricornia Cays where total counts are not feasible. In these cases doing complete counts in small areas or plots, measuring the total area in which they occur and multiplying the mean density from the plots by the total area will give a reasonable estimate. The more plots you do the better the resulting estimate. Plots should be placed randomly within noddy habitat or use a regular pattern (e.g. every 50 metres). Your technical support team or the state-wide ecological assessment unit will be able to provide guidance on this if you are unclear.

Roseate (*Sterna dougallii*), black-naped (*Sterna sumatrana*) and lesser crested (*Thalasseus bengalensis*) terns

These largely inshore species breed in more than 200 small colonies along the coast without any especially significant foci of breeding activity. Colonies are typically 100 pairs or less although confirmed records of breeding events of more than 5000 pairs exist from the Queensland coast. Globally all three species are widespread but relatively rare with global populations in tens or hundreds of thousands rather than millions. Nevertheless all are rated as Least Concern by the IUCN without any specific evidence of rapid decline or imminent extinction. With so many small colonies in Queensland it is not possible to monitor a few sites and observe the majority, or even a significant proportion, of these populations as is the case with species such as brown boobies and wedge-tailed shearwaters. In addition these species have short unpredictable breeding cycles and are significantly more difficult to monitor as effectively as species with higher site fidelity, more predictable timings and longer breeding cycles.

Without the capacity to monitor large numbers of sites frequently, the Strategy must rely on the untested assumptions that the movements of birds between sites is not extensive and will not bias the data and that the relatively small proportion of the population observed will be representative of the condition of the species across Queensland. These species will be prioritised as candidates for autonomous monitoring technologies to extend our capacity to monitor more sites.

If additional resources are available we suggest that a similar strategy to that proposed to monitor little terns be applied. By undertaking counts at all sites within an aggregation of sites we might limit the influence of movements on the assumption that these are more likely to be local than long distance. The table of significant sites includes three proposed aggregations of breeding sites that might afford a more reliable assessment of these species than that currently included in the essential list.

Masked booby (*Sula dactylatra*) and red-footed booby (*Sula sula*)

These species are similar in size and behaviour to the brown booby. Both species are more pelagic than the closely related brown booby but all plunge-dive spectacularly to capture prey and have very similar global ranges. Both are listed as Least Concern by the IUCN but are suspected of being in decline. The masked and red-footed boobies are markedly less common and widespread than the brown booby but have similar breeding ecology. Masked boobies breed mainly on cays in the Swain Reefs, Raine Island and Moulter Cay. Red-footed boobies breed mainly at Raine Island, Moulter Cay and Stapleton Island in the north of the state.

Red-tailed tropicbird (*Phaethon rubricauda*)

The red-tailed tropicbird is rare in Australia although widespread across the tropical Indian and Pacific Oceans. Its rarity in continental Australia may pertain more to its pelagic lifestyle and preference for breeding sites than any local decline although the most recent global population estimate is from 1992. In Queensland the species breeds only at Raine and Lady Elliot Islands, both notable for their proximity to deep pelagic waters. The red-tailed tropicbird is listed as Vulnerable in Queensland due to its small local breeding population but is globally considered Least Concern without evidence of declines.

Counting this species requires a unique technique as it often nests in rock crevices and hollows and is not immediately obvious to the observer.

Herald petrel (*Pterodroma heraldica*)

The Herald petrel is another widespread pelagic that is locally rare. In Australia the Herald petrel only breeds on Raine Island in far north Queensland. Because of this restricted range the species has secured a conservation status of Critically Endangered under national environmental protection law despite being regionally abundant and listed by the IUCN as Least Concern. Elevated interest in the species is common in agencies concerned with threatened species due in part to its unpredictable and poorly understood breeding ecology and perhaps the remote allure of its breeding site. At present there is no reliable method for estimating the size of the breeding population on Raine Island as the species' nests are very hard to find and it only returns to them on dusk or after dark. They have a five month breeding cycle with most records between July and October.

Silver gull (*Chroicocephalus novaehollandiae*)

The silver gull is a very common species that occurs nearly ubiquitously along the coast and on islands but also around inland waters and ephemeral wetlands. It is largely restricted to Australia and New Zealand but is extremely adaptable and can be locally abundant. It breeds in colonies of varying sizes and may form seasonal aggregations around food sources such as turtle nesting beaches and other seabird nesting colonies. It is listed as Least Concern by the IUCN and the population is thought to be increasing, a rarity in seabirds, and likely due to its adaptability and ability to coexist with human settlements and activities and in some cases profit from them.

Lesser (*Fregata ariel*) and great frigatebirds (*Fregata minor*)

Frigatebirds are pelagic, offshore birds that are occasionally sighted inshore. While they are accomplished klepto-parasites, stealing food from other birds at colonies, most of their food is fish and squid captured in the water. Both species are widespread across the tropical Indian and Pacific Oceans but breed only on a handful of sites in low numbers in Queensland with larger colonies in the Gulf of Carpentaria and the Coral Sea. Both species and are listed as Least Concern by the IUCN. However they are thought to be declining through nest predation and unsustainable exploitation. They have the longest breeding cycles of seabirds found breeding in Queensland: seven months for the lesser frigatebird and eight for the great frigatebird. There are breeding records for every month of the year.

Australian pelican (*Pelicanus conspicillatus*)

The pelican is a common inshore and wetland species which breeds mainly in ephemeral inland water systems. The species occurs only in Australian and Papua New Guinea but is widespread in both countries. Pelicans have six month breeding cycles and often breed at the same location as Caspian terns. Records from Shoalwater Bay in the southern Reef are from January and March while those from the northern Reef are from every month except January, August and October although this likely reflects the infrequent visits to pelican breeding sites.

Caspian tern (*Hydroprogne caspia*)

The Caspian tern is an extremely widespread inshore and wetland tern that occurs in many small scattered colonies across the globe. While it is common nowhere, it is extremely widespread. In Queensland it is frequently located at sites used by Australian pelicans including ephemeral inland wetlands and a handful of marine sites.

References

Batianoff, GN & Cornelius, NJ 2005, ' Birds of Raine Island: Population trends, breeding behaviour and nesting habitats.' *Proceedings of the Royal Society of Queensland,* vol. 112, pp. 1-1-29.

Catry, P, Dias, MP, Phillips, RA & Granadeiro, JP 2011, 'Different means to the same end: Long-distance migrant seabirds from two colonies differ in behaviour, despite common wintering grounds.' *PLoS ONE,* vol. 6, no. 10.

Congdon, BC, Erwin, CA, Peck, DR, Baker, GB, Double, MC & O'Neill, P 2007, "Chapter 14 Vulnerability of seabirds on the Great Barrier Reef to climate change" in *Climate Change and the Great Barrier Reef: A Vulnerability Assessment.* Eds. J.E. Johnson & P.A. Marshall, Great Barrier Reef Marine Park Authority, Townsville, pp. 427-463.

Croxall, JP, Butchart, SHM, Lascelles, B, Stattersfield, AJ, Sullivan, B, Symes, A & Taylor, P 2012, *Seabird conservation status, threats and priority actions: A global assessment*.

Devney, CA, Short, M & Congdon, BC 2009, 'Cyclonic and anthropogenic influences on tern populations.' *Wildlife Research,* vol. 36, no. 5, pp. 368-378.

Driscoll, PV 2013, *Phase 1 Analysis of Coastal Bird Atlas data*, Great Barrier Reef Marine Park Authority, Townsville.

Dunlop, JN, Long, P, Stejskal, I & Surman, C 2002, 'Inter-annual variations in breeding participation at four Western Australian colonies of the Wedge-tailed shearwater Puffinus pacificus', *Marine Ornithology,* vol. 30, no. 1, pp. 13-18.

Ellis, JC, Fariña, JM & Witman, JD 2006, 'Nutrient transfer from sea to land: The case of gulls and cormorants in the Gulf of Maine', *Journal of Animal Ecology,* vol. 75, no. 2, pp. 565-574.

Fuller, RA & Dhanjal-Adams, KL 2012, *A Power Analysis: Monitoring Seabirds in the Great Barrier Reef*, Uniquest, Brisbane.

Great Barrier Reef Marine Park Authority 2009, *Great Barrier Reef Outlook Report 2009*, Great Barrier Reef Marine Park Authority, Townsville.

Heatwole, H, O'Neill, P, Jones, M & Preker, M 1996, *Long-term population trends of seabirds on the Swain Reefs, Great Barrier Reef.*, CRC Reef Research Centre, Townsville, Australia.

King, BR 1993, 'The Status of Queensland Seabirds', *Corella,* vol. 17, no. 3, pp. 65-65-92.

Lavery, HJ & Grimes, RJ 1971, 'Sea-birds Of The Great Barrier Reef', *Queensland Agricultural Journal,* vol. 97, pp. 106-106-113.

Legg, CJ & Nagy, L 2006, 'Why most conservation monitoring is, but need not be, a waste of time', *Journal of environmental management,* vol. 78, no. 2, pp. 194-199.

McDougall, A 2011, *Coastal Bird Monitoring and Information Strategy*, State of Queensland, Department of Environment and Resource Management, Rockhampton.

Melzer, RI 2013, *Building knowledge in QPWS: Assessment and Monitoring - some basics.* , Queensland Government, Rockhampton.

O'Neill, P, Heatwole, H, Preker, M & Jones, M 1996, *Populations, movements and site fidelity of brown and masked boobies on the Swain Reefs, Great Barrier Reef, as shown by banding recoveries*, CRC Reef Research Centre, Townsville.

Palestis, BG 2014, 'The role of behavior in tern conservation', *Current Zoology,* vol. 60, no. 4, pp. 500-514.

Towns, DR, Wardle, DA, Mulder, CPH, Yeates, GW, Fitzgerald, BM, Richard Parrish, G, Bellingham, PJ & Bonner, KI 2009, 'Predation of seabirds by invasive rats: Multiple indirect consequences for invertebrate communities', *Oikos,* vol. 118, no. 3, pp. 420-430.

Turner, M 2002, *Coastal Bird Monitoring Strategy for the Great Barrier Reef World Heritage Area*, Great Barrier Marine Park Authority, Townsville.

Appendix A1.1 Legislative and international obligations

The Australian and Queensland governments are required to protect the values, particularly birds and threatened species, within marine parks they have jurisdiction over. These species are outlined under a variety of legislation and international treaties including:

Legislation –

1. Nature Conservation Act 1992
2. Nature Conservation (Wildlife) Regulation 2006
3. Marine Parks Act 2004 (includes link to zoning plans for Great Barrier Reef Coast, Great Sandy and Moreton Bay).
4. Great Barrier Reef Marine Park Act 1975
5. Environment Protection and Biodiversity Conservation Act 1999

International Conventions and Agreements –

1. The World Heritage Convention
2. Convention on Biological Diversity
3. JAMBA (Japan Australia Migratory Bird Agreement)
4. CAMBA (China Australia Migratory Bird Agreement)
5. ROKAMBA (Republic of Korea Australia Migratory Bird Agreement)
6. The Ramsar Convention on Wetlands
7. Convention on the Conservation of Migratory Species of Wild Animals (Bonn)

Links for printed out strategies (as above)

Legislation –

1. http://www.legislation.qld.gov.au/LEGISLTN/CURRENT/N/NatureConA92.pdf
2. http://www.legislation.qld.gov.au/LEGISLTN/CURRENT/N/NatureConWiR06.pdf
3. http://www.legislation.qld.gov.au/Acts_SLs/Acts_SL_M.htm
4. http://www.comlaw.gov.au/comlaw/management.nsf/lookupindexpagesbyid/IP200401513?OpenDocument
5. http://www.comlaw.gov.au/comlaw/management.nsf/lookupindexpagesbyid/IP200401830?OpenDocument

International Conventions and Agreements –

1. http://www.environment.gov.au/heritage/about/world/convention.html
2. http://www.cbd.int/
3. http://www.environment.gov.au/biodiversity/migratory/waterbirds/bilateral.html#jambacamba
4. http://www.environment.gov.au/biodiversity/migratory/waterbirds/bilateral.html#jambacamba
5. http://www.environment.gov.au/biodiversity/migratory/waterbirds/bilateral.html#jambacamba

6. http://www.ramsar.org/cda/en/ramsar-home/main/ramsar/1_4000_0__
7. http://www.cms.int/

Appendix B — Summary of essential site visits since 2012

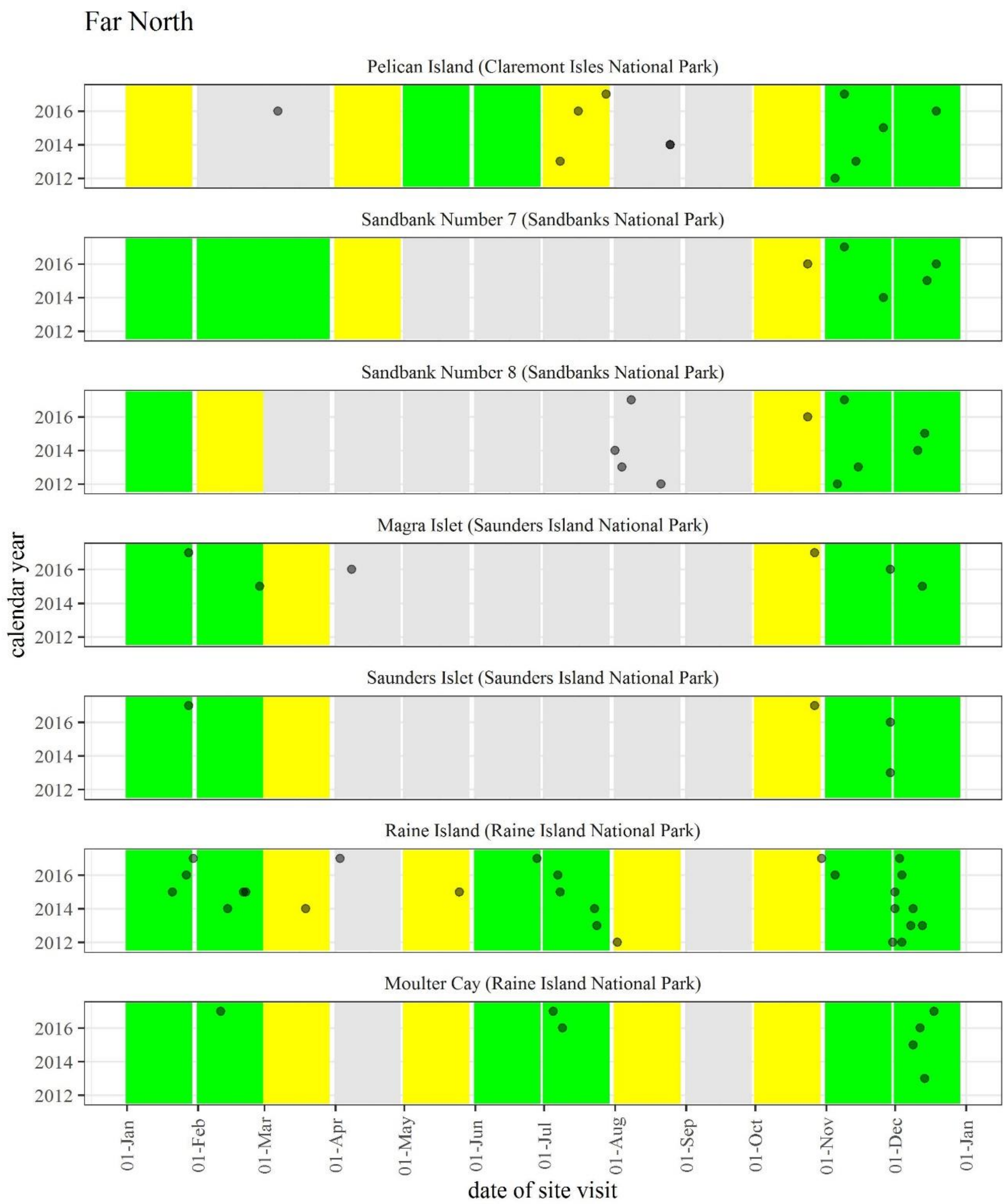


Figure 1. Summary of site-specific survey implementation from 2012-2017 for essential sites of the Far North region. Each point indicates a site visit and background colours indicate high (green), medium (yellow), and low (grey) priority survey times following Hemson et al. 2015.

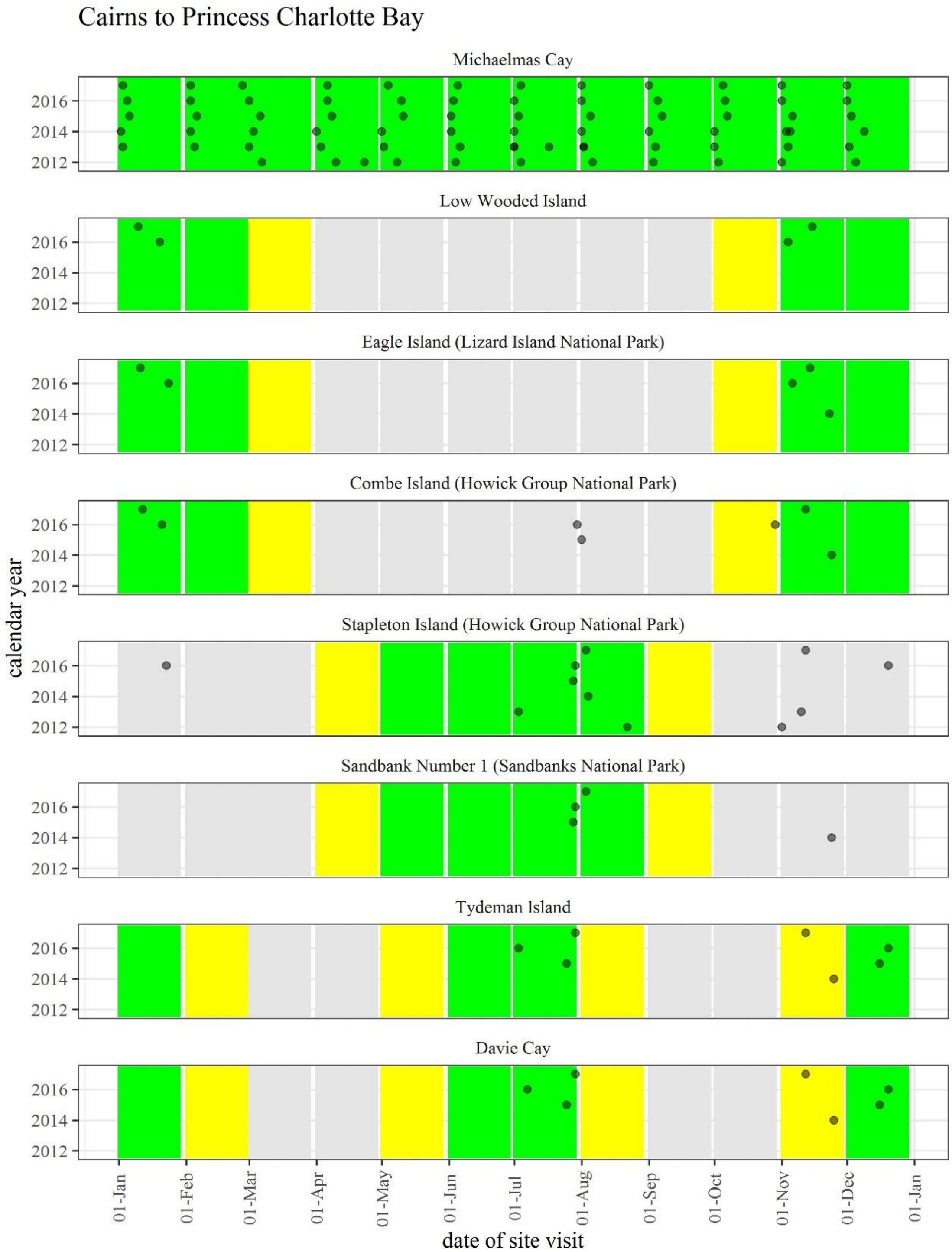


Figure 2. Summary of site-specific survey implementation from 2012-2017 for essential sites of the Cairns to the Princess Charlotte Bay region. Each point indicates a site visit and background colours show high (green), medium (yellow), and low (grey) priority survey months following Hemson et al. 2015.

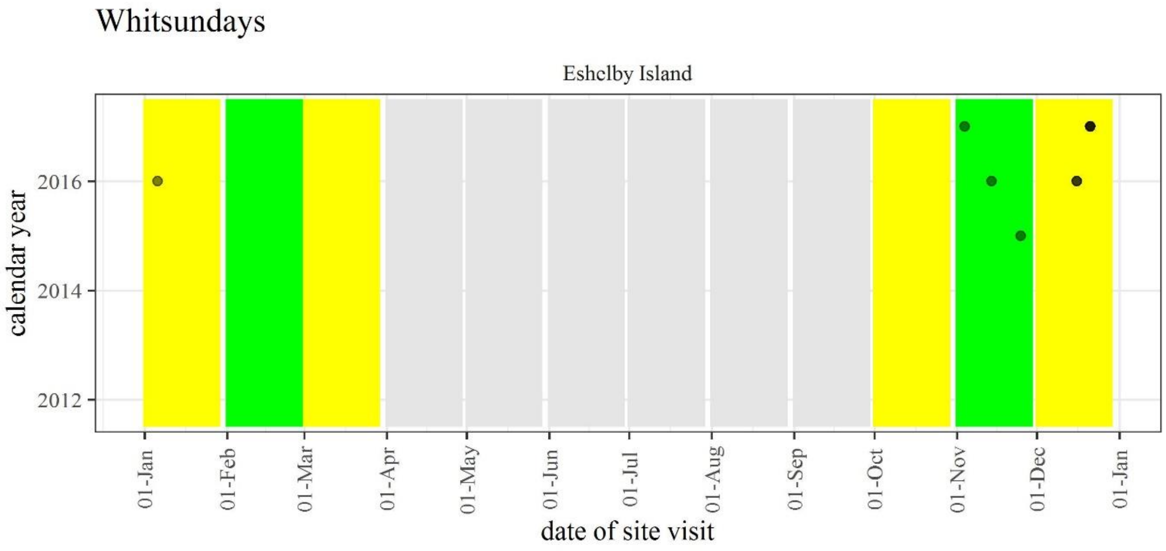


Figure 3. Summary of site-specific survey implementation from 2012-2017 for essential sites of the Whitsundays region. Each point indicated a site visit and background colours show high (green), medium (yellow), and low (grey) priority survey months following Hemson et al. 2015.

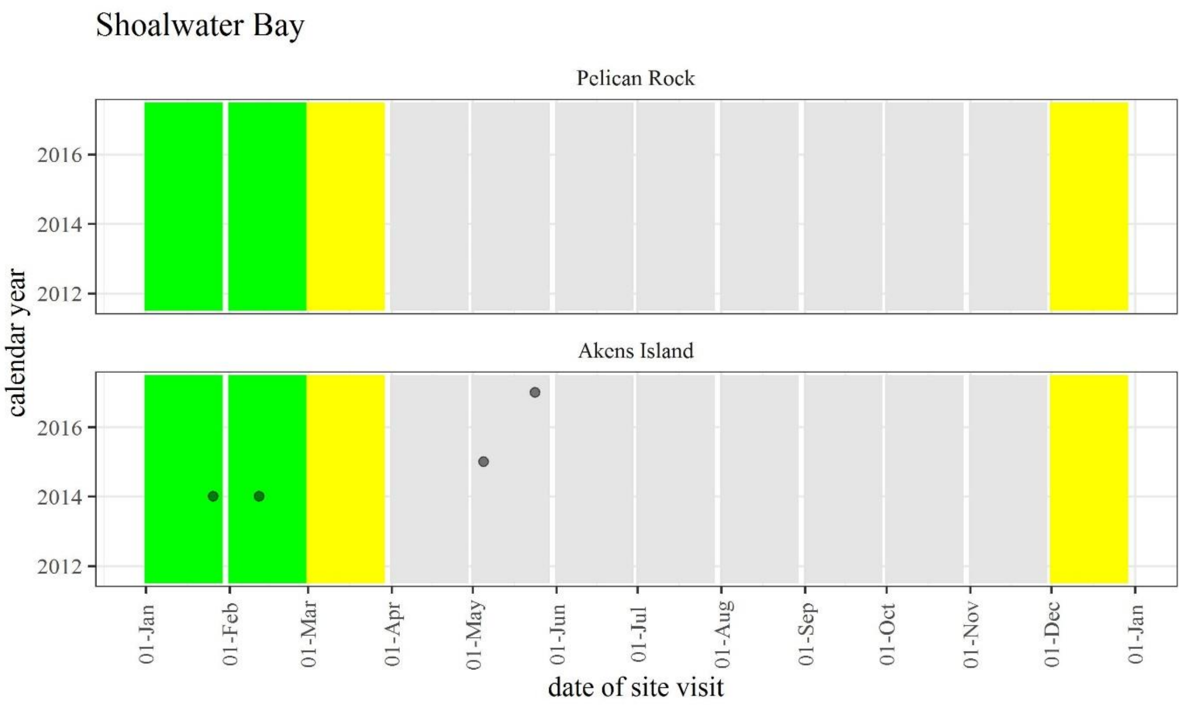


Figure 4. Summary of site-specific survey implementation from 2012-2017 for essential sites of the Shoalwater Bay region. Each point indicates a site visit and background colours show high (green), medium (yellow), and low (grey) priority survey months following Hemson et al. 2015.

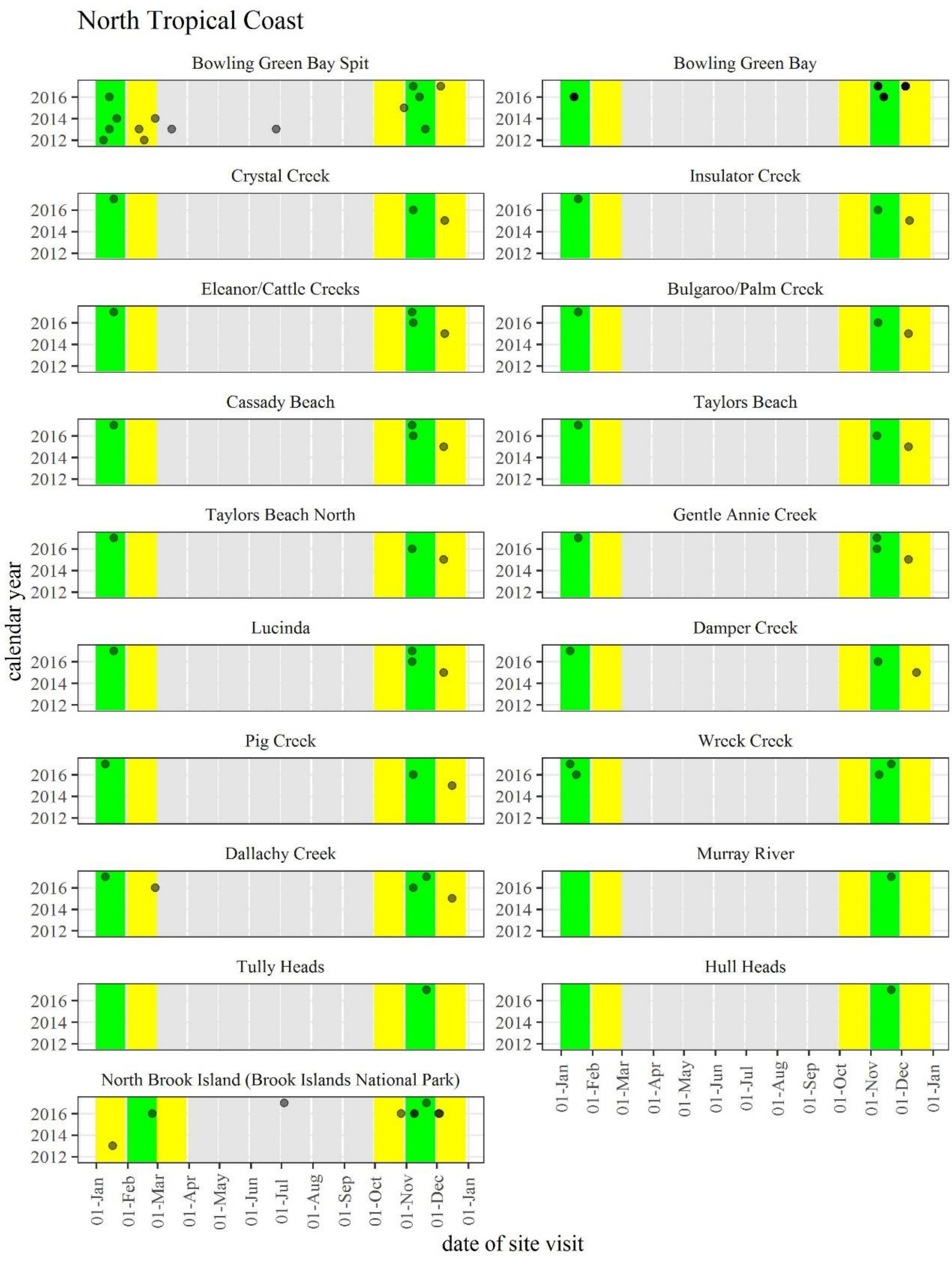


Figure 5. Summary of site-specific survey implementation from 2012-2017 for essential sites of the North Tropical Coast region. Each point indicates a site visit and background colours show high (green), medium (yellow), and low (grey) priority survey months following Hemson et al. 2015.

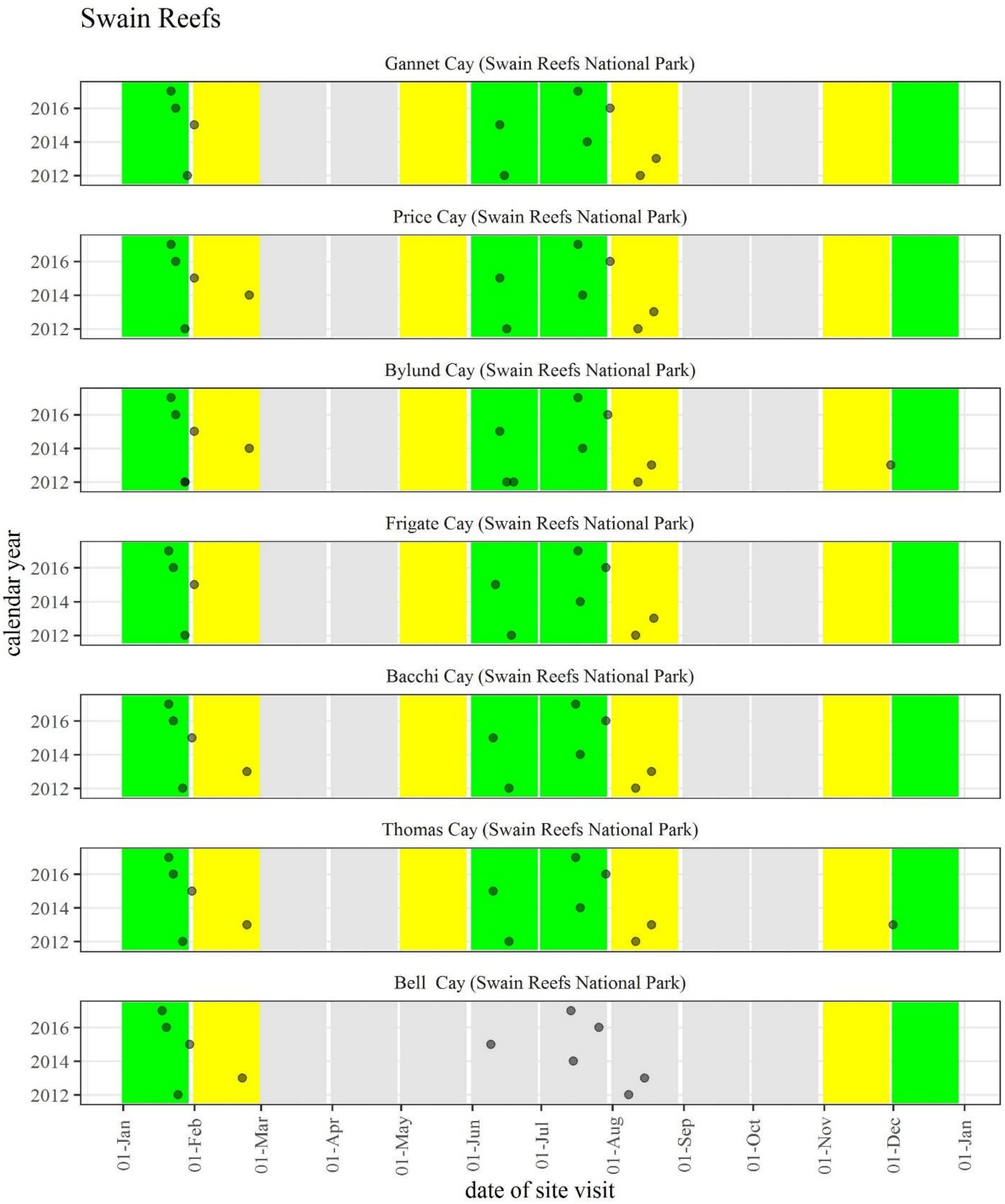


Figure 6. Summary of site-specific survey implementation from 2012-2017 for essential sites of the Swain Reefs region. Each point indicates a site visit and background colours show high (green), medium (yellow), and low (grey) priority survey months following Hemson et al. 2015.

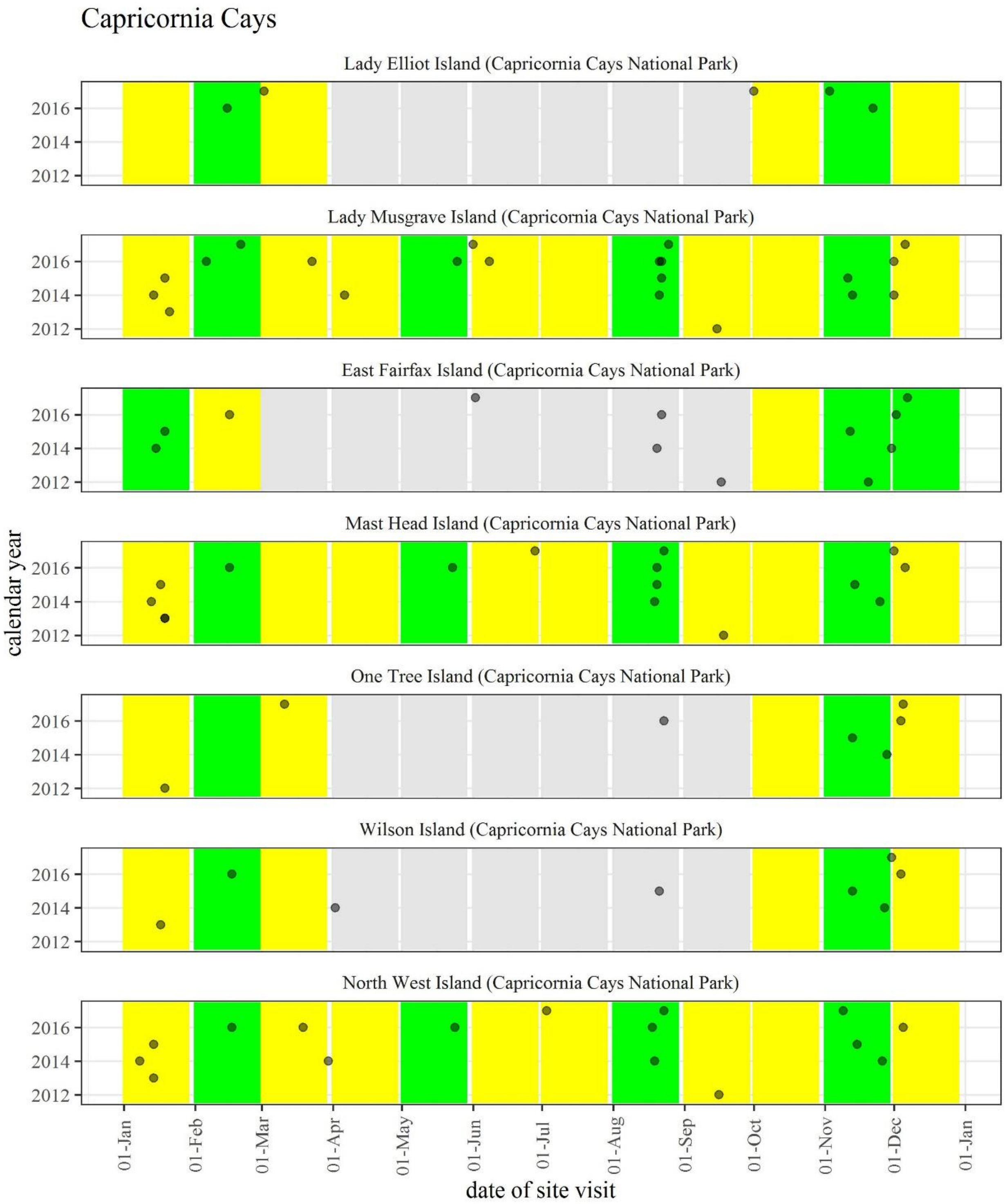


Figure 7. Summary of site-specific survey implementation from 2012-2017 for essential sites of the Capricornia Cays region. Each point indicates a site visit and background colours show high (green), medium (yellow), and low (grey) priority survey months following Hemson et al. 2015.

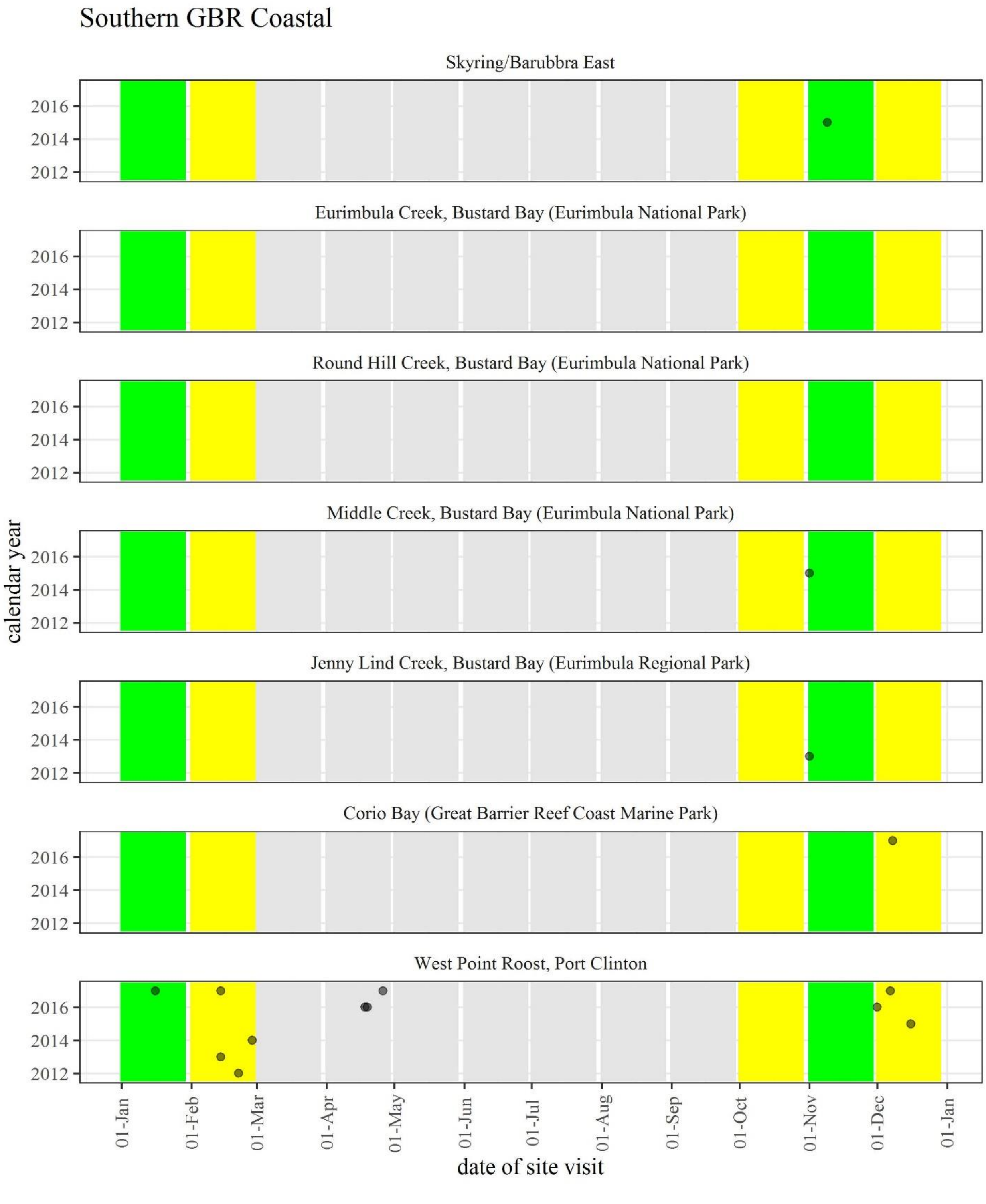


Figure 8. Summary of site-specific survey implementation from 2012-2017 for essential sites of the Southern Reef Coastal region. Each point indicates a site visit and background colours show high (green), medium (yellow), and low (grey) priority survey months following Hemson et al. 2015.

Appendix C — Species-specific breeding phenology summaries

Parameter	Wedge-tailed Shearwater

Length of breeding season	Pre-laying period (arrival to laying) protracted at Heron Island (and potentially other southern Reef colonies) relative to more southern colonies but laying date similar across Southern Reef & NSW colonies
	
	pre-laying period including pre-laying of exodus ~ two weeks[1]:
	~ 90 days (Muttonbird Island)
	~ 60 days (Heron Island)[2]
	
	Incubation duration 53 days (Muttonbird Island; 1971)[1]
	Incubation duration 48-55 days (global range)[2]
	
	hatching-fledging duration 98 days (Muttonbird Island; 1971)[1]
	
	Total duration:
	min 60+48+98=206 days;
	max 90+55+98=243 days;
	likely Heron island: 60+53+98=211 days

Time of peak breeding	Asynchronous arrival at E. Australian Colonies (see Fig 1): Heron Island 29 Oct ± 2.5 d (2012)[3] 1-14 Oct (multiple studies)[2] Mudjimba Island & North Stradbroke Island 19-28 Aug. (multiple studies)[2] Muttonbird Island 2-12 Aug. (multiple studies)[2] Raine Island and Rocky Islet Shearwaters recorded on Raine Island in all months except May. Potential breeding starts in June, mating in July, laying in Oct. with fledging in Feb.[2] Departure more synchronised: Heron Island/ Capricorn Group Adult departure 21 May ± 1.1 (2012)[3] Fledgling departure 20-25 May (1964-79)[1] Muttonbird Island/ NSW Adult departure 4 May Fledgling departure 11 May (1970)[1]

Breeding success	43% success (55.8% egg survival; 78.8% chick survival; Muttonbird Island; 1979-80)[8] 61% success egg to fledging (Heron Island; 1993)[7] 5-55% success egg to fledging (Western Australia; 1994-2001)[4] Chick mortality rate 3.5% (2001), 50-100% (2002), 10% (2003) at Heron Island (Feb-Mar)[5,6]
Clutch size and egg replacement	One egg per season, no replacement[1]
Breeding desertion rate	Variable, complete breeding failure (100% desertion) is known from Heron Island [6]
Annual breeding participation	19.3-83.6% breeding participation between 1994-2001(measured as number of active burrows during pre-laying that then had an egg laid in them) (Western Australia)[4]

Parameter	**Brown Booby**	**Masked Booby**
Length of breeding season	Pre-laying period duration uncertain. Up to 30 days needed for egg formation in Red-rooted Booby[10] Incubation duration 42.8 days (Christmas Is [Pac])[9] Average hatching-fledging duration 96 days (Christmas Island [Pac]), even weak chicks capable of flight after 119 days.[9] Total duration: min 30+43+96=169 days max 30+43+119=192 days	Pre-laying period duration uncertain. Up to 30 days needed for egg formation in Red-rooted Booby[10] Incubation duration 43.6 days (43-49) at Kure Atoll[1] and 42-46 days at Ascension Island[16] Average hatching -fledging duration 120 days (Ascension Island)[16], and 113-120 (Galapagos)[1] Total duration: min 30+42+113=185 days max 30+49+120=199 days
Time of peak breeding	Breeds year-round with distinct nesting peak in summer. Peak laying Sept.-Nov. at Raine Island[14] and Far Northern Reef colonies[12], and Swains Reefs (but with more variability)[13]	Breeds year-round with distinct nesting peak in summer. Peak laying Sept.-Nov. and peak chicks/fledglings Nov.-May (Pandora Cay and Raine Island)[1]

	Peak laying Aug.-Oct. (Chesterfield Is)[11]	Peak laying June-Oct. and chicks seen until end of Apr. (Chesterfield Is)[11]
Breeding success	58% success egg to fledging: 68% egg survival; 81% chick survival (Christmas Island [Pac])[1] 10% success egg to fledging (Ascension Island; c. 1960)[1]	Egg-fledging success varied from 51.1% to 90.3% over six seasons at Kure Atoll[1]
Clutch size and egg replacement	one-three eggs laid, two most common (multiple studies)[9] Only one chick raised, but rare observations of two from Raine Island. Replacement clutches laid in < 50% of nests (Kure Atoll), after 20-34 days (Christmas Island [Pac])[9]	one-three eggs laid, two most common (multiple studies)[1] Only one chick raised, but rare observations of two from Raine Island. Replacement clutches laid in 43% of nests (Kure Atoll), after 17-59 days (Christmas Island [Pac])
Breeding desertion rate	Variable, complete breeding failure (100% desertion) seen at Christmas Island [Pac] due to El Nino[15] and suspected at Swains Reefs[13]	Variable, complete breeding failure (100% desertion) seen at Christmas Island [Pac] due to El Nino[13] and suspected at Swains Reefs[12]
Annual breeding participation	unknown	unknown

Parameter	**Crested Tern**	**Sooty Tern**	Little Tern
Length of breeding season	Gather at pre-nesting area 'club' (100's of metres to several kilometres from nesting grounds) during pre-laying period, one-two months before nesting.[1] Incubation duration 21-24 days (One Tree Island; 1973-76)[18], 28±1 days (One Tree Island; 1979)[17], and 29 days (South Australia; 1967-70)[1]	Gather at breeding colony one-two months prior to nesting, rarely landing during the day.[211] Incubation duration 28-30 days globally.[1]However, reported as under 26 days at Michealmas Cay[19] Average hatching-fledging 28 days[21], up to 56 days globally during poor conditions	

	Average hatching-fledging 35-43 days, can be longer due to bad weather (One Tree Island)[17] Total duration: min 30+21+35=86 days max 60+29+43=112 days	Total duration: min 30+26+28=84 days max 60+30+43=133 days	
Time of peak breeding	Majority of Reef colonies are summer breeding, however some appear to breed in winter in the more northerly Reef. Sept.-Nov. (mid Nov. peak) arrival at pre-breeding gathering grounds. Laying Nov.-Dec. Adults and fledglings depart by end of Feb. (One Tree Island)[17] Nov. arrival at pre-breeding gathering grounds. Laying Dec.-Jan. Adults and fledglings depart by end of Mar. (Eagle Island, Lizard Island)[20] Breeding peaks in summer at Raine Island and far northern Reef colonies (Nov.-Apr.), however a few colonies breed in winter (May-Oct.)[12] Breeding Dec.-July (peak Jan./Feb.) at Michealmas Cay (1984-90)[19] Breeding season Jan.-July(Chesterfield Island)[11]	Complicated phenology across the Reef: "In the Sooty Tern, therefore, we appear to have a situation of year-round sub-annual breeding with an 8.5 month cycle at the southern end of its range, with annual winter breeding further north, and sub-annual year-round breeding further north again"[22] Sub-annual breeding at Michealmas Cay with birds breeding on a 8.5 month cycle. Some form of breeding found in almost every month of the year.[19] Winter breeding reported at Raine Island (Apr.-Aug.), Stapleton Island, Tydeman Cay, Davie Cay, Sandbank No 8, Moulter Cay and MacLennan Cay.[22]	
Breeding success	Success egg to fledging 41.7% (31.1-63.9%) (Eagle Island)[20] Success egg to fledging 47.1% (0.5-59%). Egg hatching success 53.8% (0.5-69%) (One Tree Island)[17]	Success egg to fledging 11.3-47.5%(over seven seasons Michaelmas Cay)[19]	

Clutch size and egg replacement	Only a single egg laid, almost always[1] Up to three replacement clutches laid at 10, 11, and <20 days after loss[1]	Normally one egg, occasionally two[1] Up to two replacement clutches laid, usually within two weeks of loss[1]	
Breeding desertion rate	22% desertion rate of eggs over four seasons (One Tree Island)[17]	Variable, complete desertion following some cyclones[19]	
Annual breeding participation	Unknown	unknown	

References

Marchant, S. & Higgins, P. in *Handbook of Australian, New Zealand and Antarctic birds. Vol. 1: Ratites to Ducks, Part A-Ratites to Petrels, Part B-Australian Pelican to Ducks* 593–602 (Oxford, University Press, 1990).

Dyer, P. K. & Carter, J. L. Synchronous Breeding: Wedge-tailed Shearwaters Puffinus pacificus in Eastern Australia. *Emu* **97,** 305–309 (1997).

McDuie, F. & Congdon, B. Trans-equatorial migration and non-breeding habitat of tropical shearwaters: implications for modelling pelagic Important Bird Areas. *Mar. Ecol. Prog. Ser.* **550,** 219–234 (2016).

Dunlop, J. N., Long, P., Stejskal, I. & Surman, C. Inter-annual variations in breeding participation at four Western Australian colonies of the Wedge-tailed shearwater Puffinus pacificus. *Mar. Ornithol.* **30,** 13–18 (2002).

Smithers, A., Peck, D. R. A., Krockenberger, A. K. A. & Congdon, B. C. A. Elevated sea-surface temperature, reduced provisioning and reproductive failure of wedge-tailed shearwaters (Puffinus pacificus) in the southern Great Barrier Reef, Australia. *Mar. Freshw. Res.* **54,** 973–977 (2003).

Peck, D. R., Smithers, B. V., Krockenberger, A. K. & Congdon, B. C. Sea surface temperature constrains wedge-tailed shearwater foraging success within breeding seasons. *Mar. Ecol. Prog. Ser.* **281,** 259–266 (2004).

Carter, J. L., Hill, G. J. E. & Dyer, P. K. Breeding cycle of wedge-tailed shearwaters Puffinus pacificus at Heron Island, Great Barrier Reef. *Emu* **96,** 195–198 (1996).

Floyd, R. B. & Swanson, N. M. Wedge-tailed shearwaters on muttonbird island: An estimate of the breeding success and the breeding population. *Emu* **82,** 244–250 (1983).

Nelson, B. *The Sulidae: gannets and boobies*. *1978* (Oxford University Press, 1978).

Grau, C. R. in *Seabird Energetics* (eds. Whittow, C. C. & Rahn, H.) 33–58 (Plenum, 1984).

Borsa, P., Pandolfi, M., Andréfouët, S. & Bretagnolle, V. Breeding Avifauna of the Chesterfield Islands, Coral Sea: Current Population Sizes, Trends, and Threats. *Pacific Sci.* **64,** 297–314 (2010).

Blaber, S. J. M., Milton, D. a., Farmer, M. J. & Smith, G. C. Seabird Breeding Populations on the Far Northern Great Barrier Reef, Australia: Trends and Influences. *Emu* **98,** 44–57 (1998).

Heatwole, H., O'Neill, P., Jones, M. & Preker, M. *Long-term population trends of seabirds on the Swain Reefs, Great Barrier Reef*. (1996).

Batianoff, G. N. & Cornelius, N. J. Birds of Raine Island: population trends, breeding behaviour and nesting habitats. *Proc. R. Soc. Queensland.* **112,** 1–29. (2005).

Schreiber, R. W. & Schreiber, E. A. Central Pacific Seabirds and the El Niño Southern Oscillation : 1982 to 1983 Perspectives. *Science (80-.).* **225,** 713–716 (1984).

Doward, D. F. Comparative Biology of the White Booby and the Brown Booby Sula Spp. At Ascension. *Ibis (London. 1859).* **103,** 174–220 (1962).

Langham, N. P. & Hulsman, K. The breeding biology of the crested tern sterna bergii. *Emu* **86,** 23–32 (1986).

Hulsman, C. Feeding and breeding biology of six sympatric species of tern (Laridae) at One Tree Island, Great Barrier Reef. (University of Queensland, 1977).

King, B. R., Hicks, J. T. & Cornelius, J. Population changes, breeding cycles and breeding success over six years in a seabird colony at Michaelmas Cay, Queensland. *Emu* **92,** 1–10 (1992).

Smith, G. C. Feeding and breeding of crested terns at a tropical locality-comparison with sympatric black-naped terns. *Emu* **93,** 65–70 (1993).

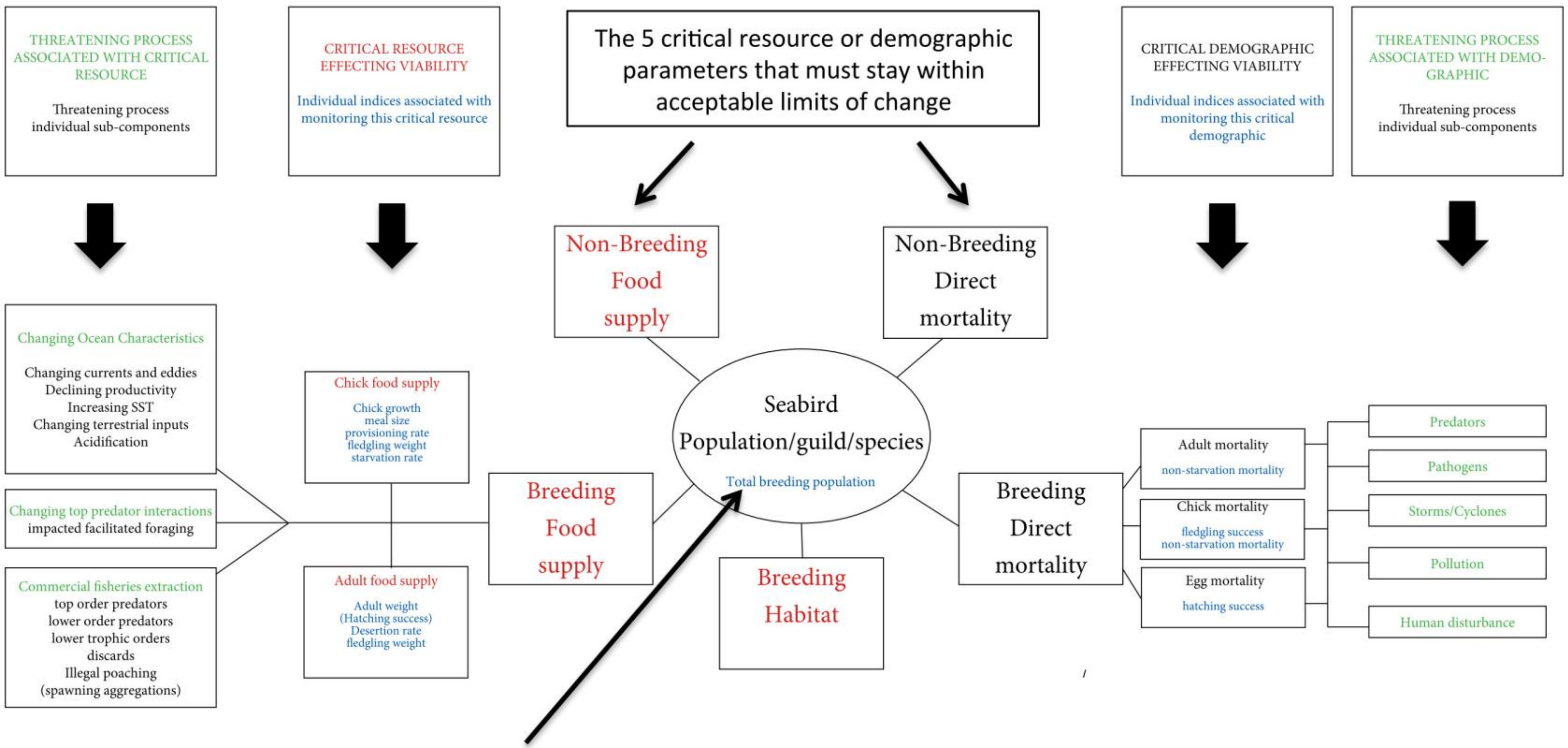
Devey, Carol Ann (2012) Climate variation and population dynamics in tropical seabirds. PhD thesis, James Cook University. (2012).

King, B. The status of Queensland seabirds. *Corella* **17,** 65–92 (1993).

Appendix D — Seabird conceptual model

Seabird conceptual model:

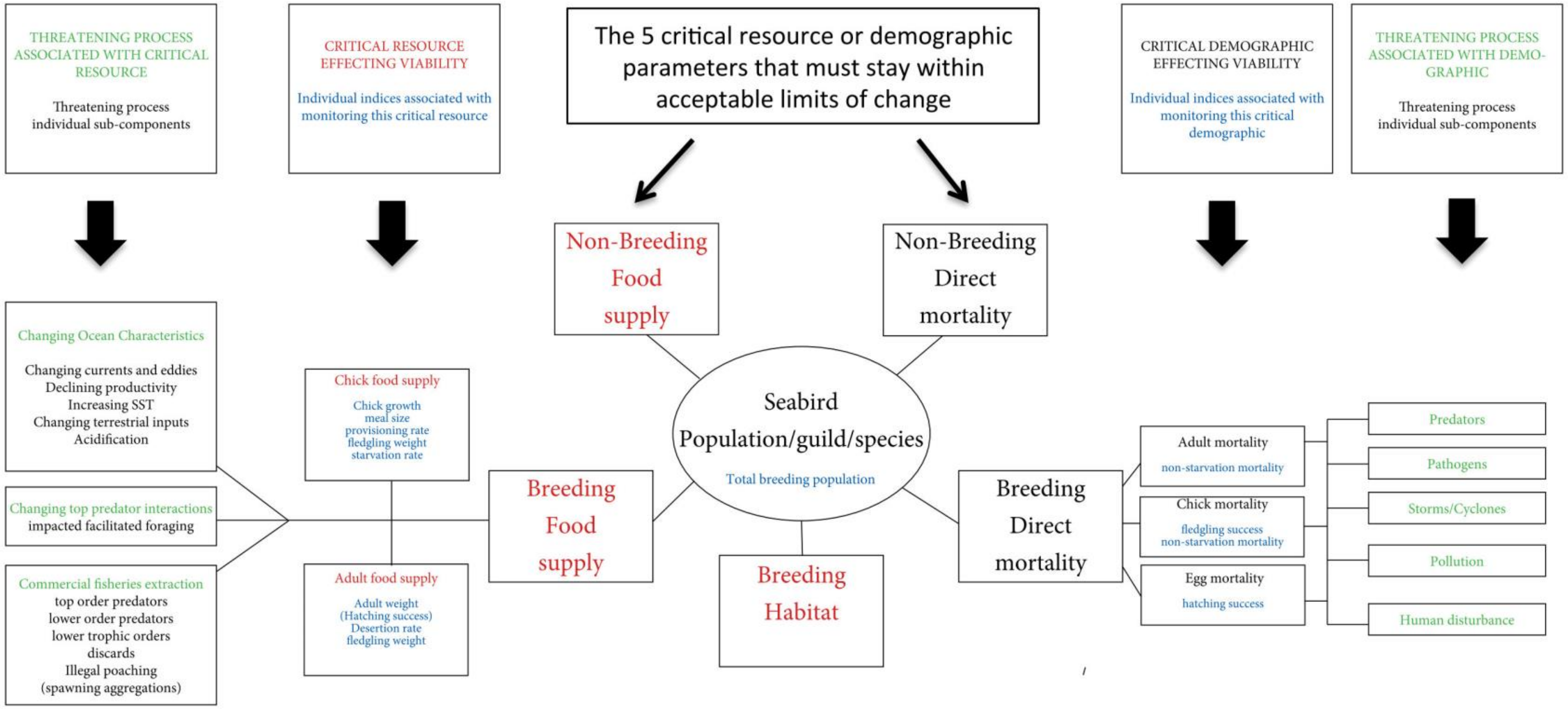
Expanded example for two of the five critical resource/demographic sub-components



Current monitoring of *"total breeding population"* can only detect that some unknown threat is impacting. It does not distinguish between any of these threatening processes

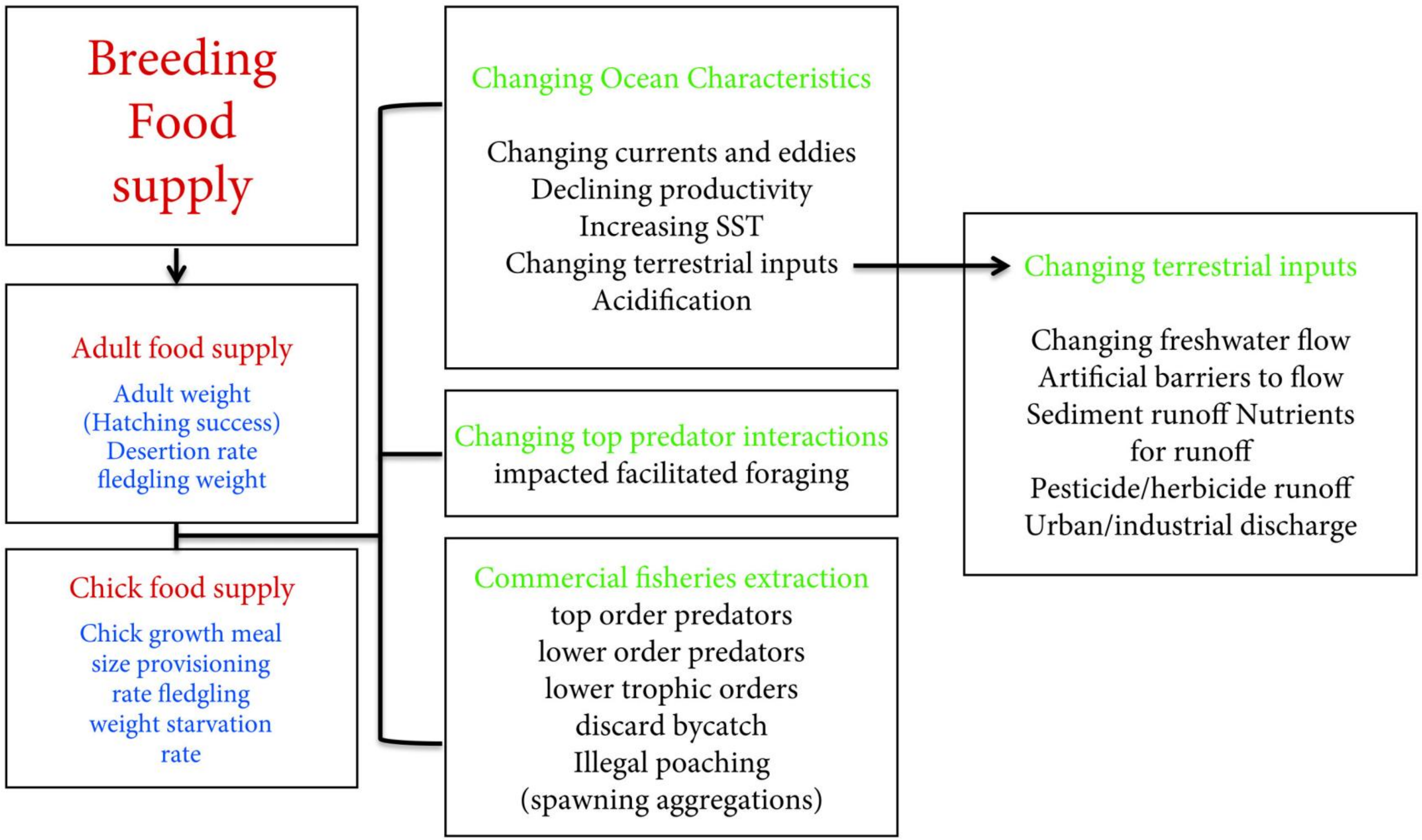
Also, it most likely indicates threatening processes that have impacted *non-breeding grounds food supply*, or the availability of *breeding habitat*.

What other indices could be monitored and what pressures could they detect?

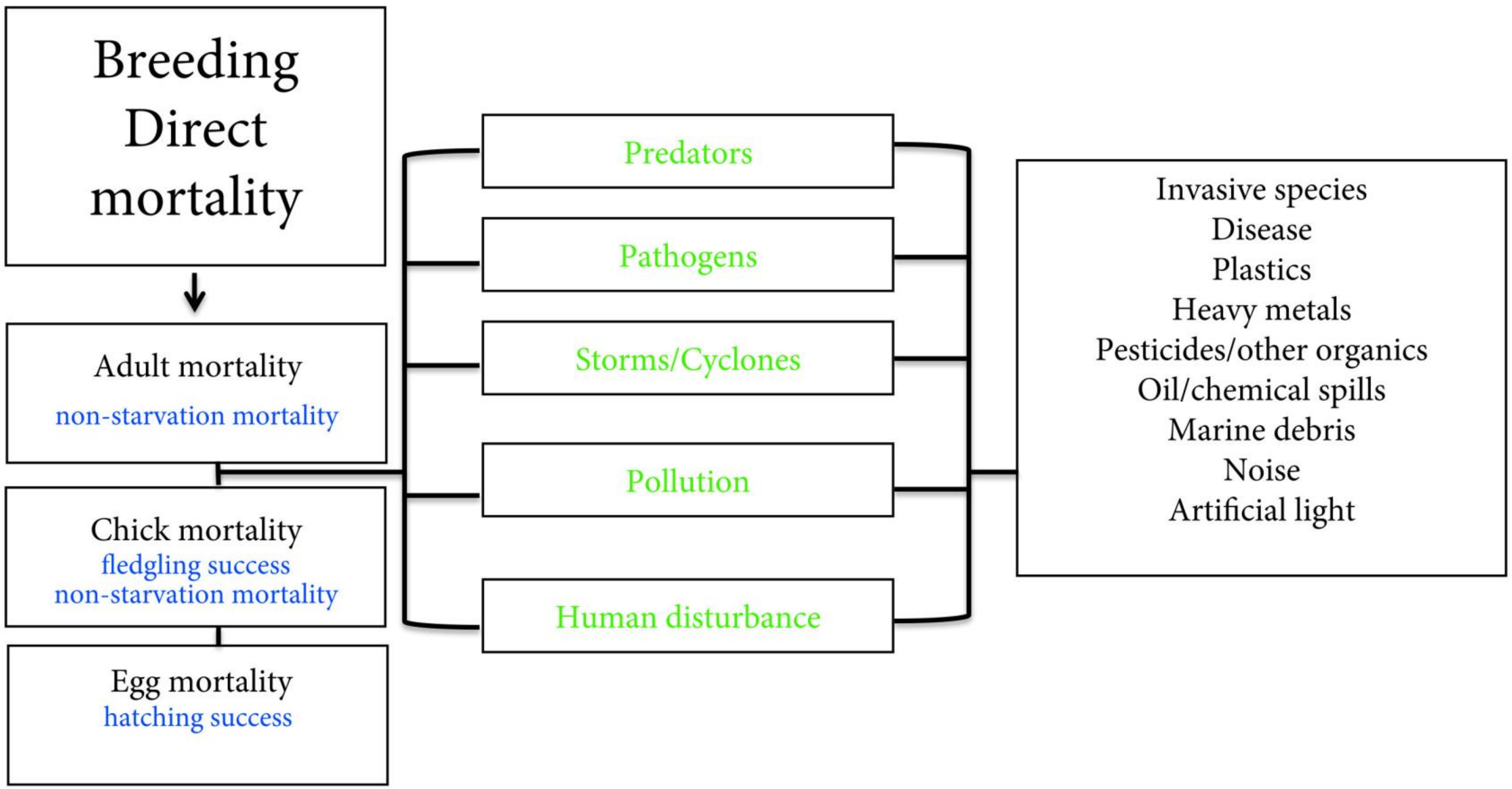


Examples of other indices that could be monitored (blue) along with the general (green) and specific (black) threatening processes they provide information on respectively, for two of the five critical resource/demographic sub-components.

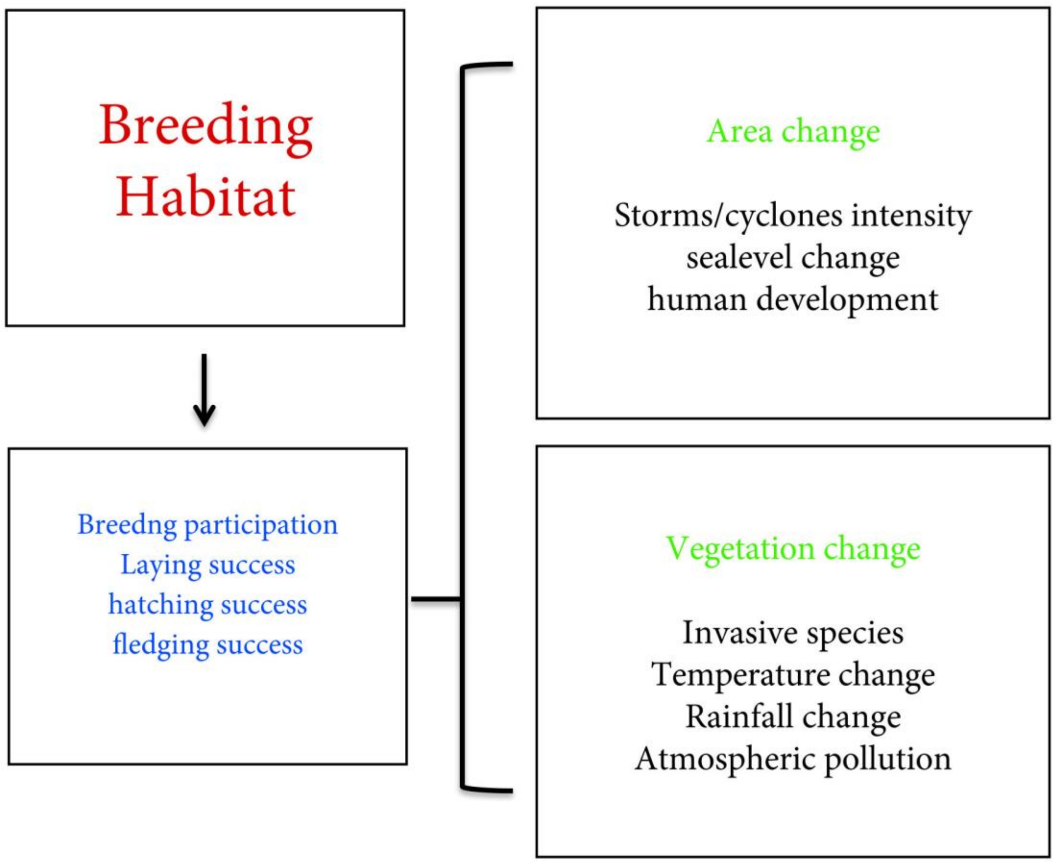
Expanded list of seabird indices (blue) and associated general (green) and specific (black) threatening processes that impact *Breeding Food Supply*



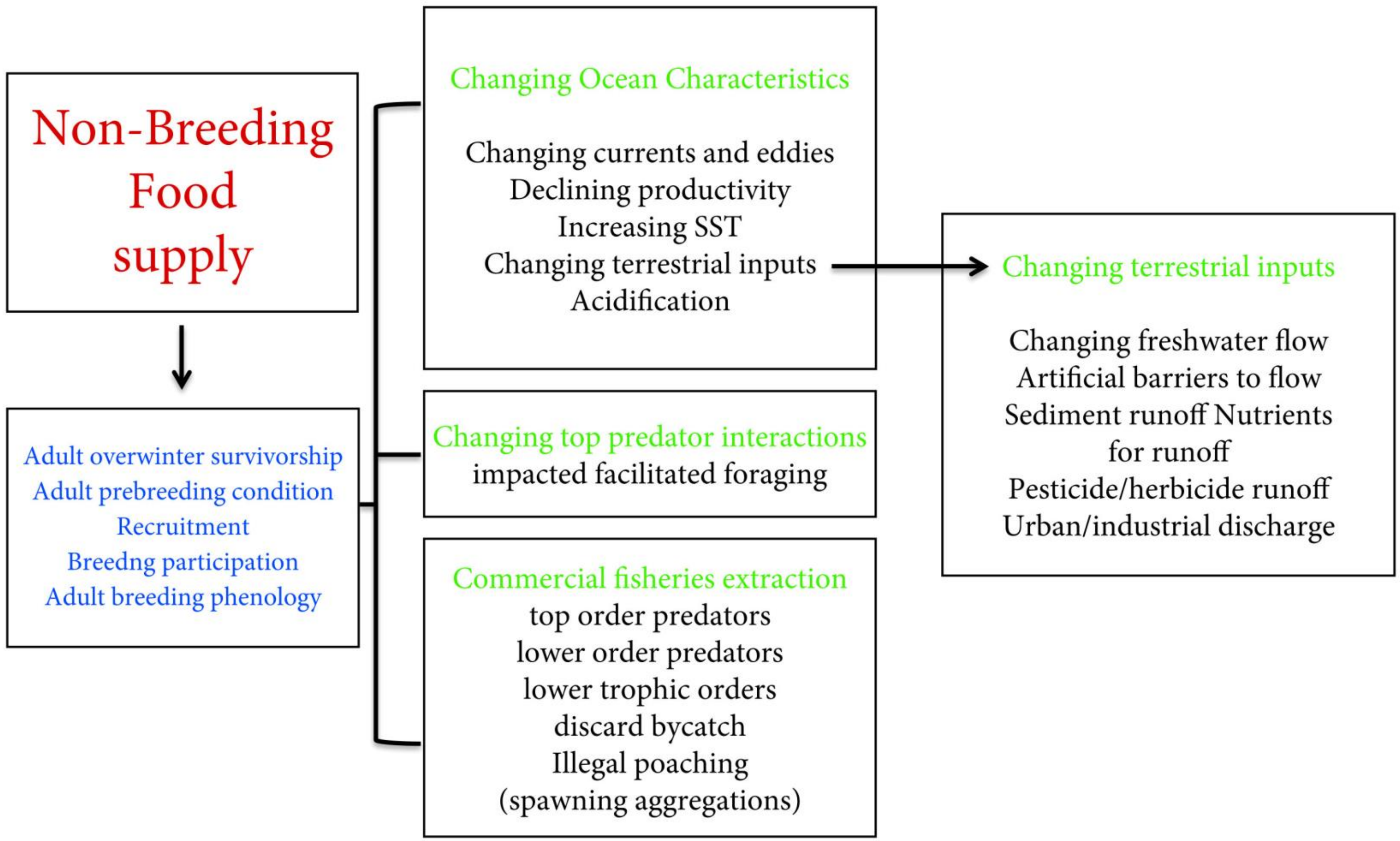
Expanded list of seabird indices (blue) and associated general (green) and specific (black) threatening processes that impact *Breeding Direct Mortality*



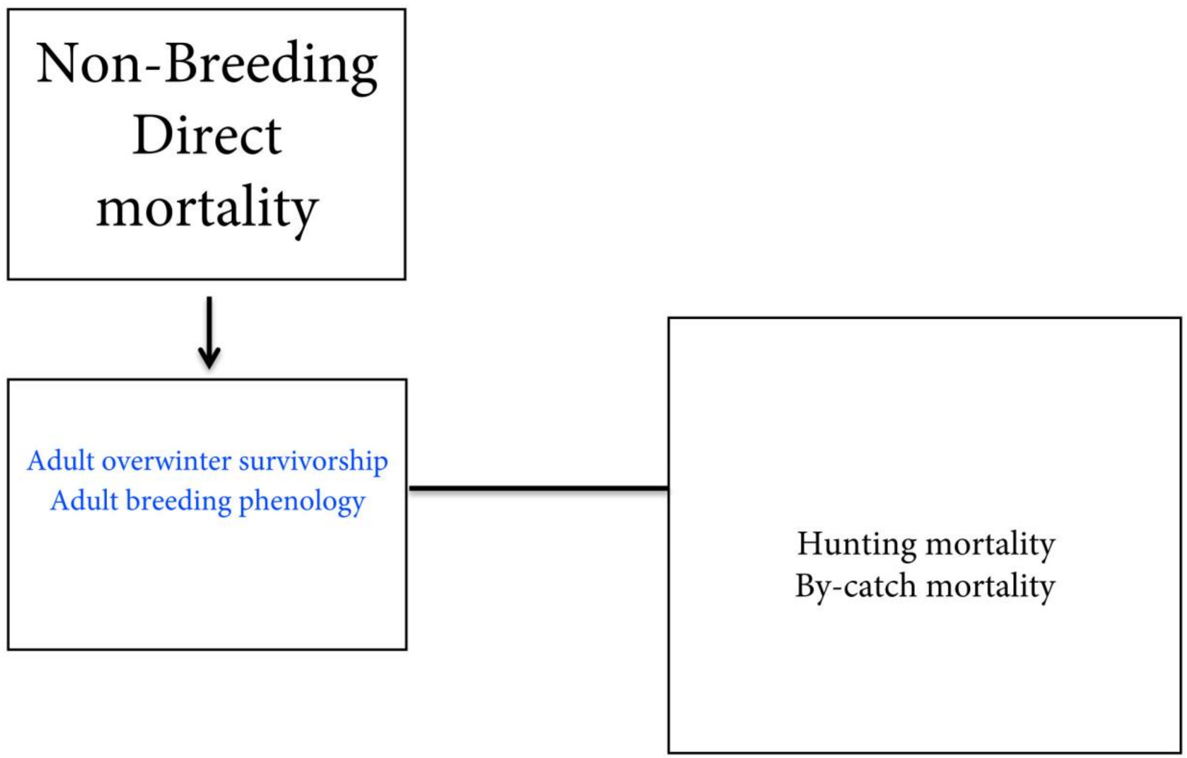
Expanded list of seabird indices (blue) and associated general (green) and specific (black) threatening processes that impact *Breeding Habitat*



Expanded list of seabird indices (blue) and associated general (green) and specific (black) threatening processes that impact *Non-breeding food supply*



Expanded list of seabird indices (blue) and associated general (green) and specific (black) threatening processes that impact Non-*Breeding Direct Mortality*



Appendix E — Drone and acoustic sampling report

Summary

This report summarises experiments conducted to test whether we can use cameras, acoustic recorders and drones as alternatives to site visits to survey seabird-breeding colonies. It recommends a direction for future work and seeks endorsement of the direction described.

The experiments are not complete, but we wanted to update people with an interest in the project. We now have enough information to provide informed recommendations and directions for future work. While each method has unique advantages and disadvantages, we conclude that we should use acoustic sensors wherever the conditions are suitable and it is economically sensible to do so. We also recommend that we further investigate the use of drones, as this technology is improving very rapidly. The potential of autonomous drones housed in solar powered containers located at remote sites is of particular interest in this regard. Establishing the drone specifications needed to monitor seabirds can occur at the same sites as the testing of acoustic sensors. This will result in faster acoustic tests and advancing our evaluation of drones at the same time.

Seabird breeding aggregations are increasingly limited to islands as nest predators such as foxes, dogs, rats and cats have spread and increased in abundance on mainland Australia. The majority of islands in Queensland are within the Great Barrier Reef World Heritage Area and many of these are managed by and most accessible to the Queensland Parks and Wildlife Service (QPWS). QPWS leads or contributes to most seabird related monitoring and management actions in the Reef Long Term Sustainability Plan 2050 (http://hdl.handle.net/11017/2934).

The QPWS Ecological Assessment Unit supported by the Great Barrier Reef and Marine Parks Region has been testing automated cameras, audio recorders and acoustic pattern analysis since 2012 and has recently started looking at drones as tools for monitoring seabird-breeding colonies. We refer to these technologies as autonomous tools. The knowledge that we had not previously been surveying seabirds regularly or frequently enough to establish trends has in part driven this project.

Here we outline the work that is occurring and has occurred, the most significant results from this work, and provide an initial evaluation of the different approaches. We assess for which species and in which circumstances they are best suited, how the development of the technologies may change this and how we believe QPWS should proceed to get the best bang for our buck.

Acoustics

Our findings to date suggest that acoustic sensors are simple to use and robust, and can produce results that scale reliably with the number of seabirds in an area. As such, they show great promise in producing robust indices of abundance [25]or, with more work, estimates of abundance. Inconsistent applications of monitoring methods may introduce biases to data; therefore acoustic data may be less error prone than first person observations. An additional advantage is that prolonged deployments allow season wide monitoring rather than single days. This means we have an opportunity to monitor both reproductive participation (the number of breeding pairs and the size of the population) and a measure of breeding success (the numbers of chicks hatched and raised until fledging). As changes in reproductive success only influence the size of the breeding population several years later, when birds from the effected cohort first return to breed, this is potentially important. This lag otherwise limits our capacity to understand and react to change or manage threats in a timely manner.

However, we outsource the analyses of these data to contractors in the USA and this comes at a cost. We also have to invest time and effort to establish that acoustic measures scale reliably with each species. This requires us to count nesting birds near recorders several times to correlate these counts with data from acoustic recordings taken from equivalent periods. In some cases, these experiments may not produce the results we want and hence involve risk that resources may be ‘wasted’. We have undertaken these experiments for several species and have used what we have learned to develop guidelines about the types of species and colonies that lend themselves to acoustic monitoring. These include two of the key seabird species identified in the seabird strategy; brown boobies and wedge-tailed shearwaters. It would likely be useful for any species that breeds in colonies spread out over quite large areas and/or that breeds in a predictable location every year. Because of this, acoustic monitoring should be limited, at least in the short to medium term, to those species

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[25] An index is a value that we know increases and decreases with simultaneous changes in population density but from which we cannot estimate actual numbers, i.e. we can say that population has increased, decreased or doubled or halved but cannot say that it consists of an estimated number of birds.

that disperse themselves relatively evenly across nesting sites. These include brown, masked and red-footed boobies, wedge-tailed shearwaters, black and common noddies, sooty and bridled terns. Lesser frigatebirds, which despite nesting in discrete colonies may nest in predictable colonies in the same areas each year may also be suitable.

Camera

The trial of the camera has been less compelling. The complexity of installing and operating the device, the reliance on staff to count birds from footage and the questionable reliability, detract from the underlying promise of the concept. The difficulties associated with having a single fixed perspective, and in discriminating between species and between breeding and non-breeding birds over distance, adds a level of variability in the data that is difficult to overcome. While pattern recognition software is likely in the future, until that eventuates, it seems unlikely that these types of cameras will be useful other than for monitoring priority species that do not lend themselves to acoustic survey and that are in locations that are extremely challenging to access regularly.

Drones

Drones have potential to reduce bias and error but at present still require staff to be present in the field and a staff member to analyse imagery manually. Counts from drone imagery likely generate much more accurate estimates of the numbers of large birds at a site than on-ground counts if visibility is reasonable. However, it is the number of breeding birds that we are most interested in and for many species it is difficult to identify breeding birds from non-breeding birds using drone imagery. Identification is dependent upon the altitude, magnification and resolution of the drone and its camera. Future work with automated image analysis may overcome this.

Drone-in-a-box type systems, a drone inside a box or hangar that charges the drone and deploys it onto pre-programmed routes when weather conditions are OK, already exist. These and automated counting algorithms will improve and become cheaper in the near future. In addition, we have evidence that counts using both drone and ground observers may be more accurate and precise than counts using either method alone. This combination of current and future potential make assessing drones worthwhile. Using the combination of ground and drone counts at acoustic sensor trials sites will hasten the trials of acoustic sensors and allow further assessment of drones.

Recommendation

QPWS should use acoustic sensors and analysis wherever the method is likely to be cost effective. We should use first person observations combined with drone imagery to speed-up the validation of acoustic trials, and to improve the precision and accuracy of surveys at sites where acoustics may not be suitable. This approach will also allow QPWS to establish: the optical specifications required to monitor seabirds; how weather constrains drone flight; and whether “drone in a box” options could be useful in the future. We should stop trials of static cameras until we understand the scope of acoustic sensors and drones but keep an eye on their potential for sites for which other methods are unsuitable.

Figure 1: Crested terns (A.McDougall)

Background

The issues

In 2013, a consultant completed a report that attempted to detect trends and patterns in population size from almost 30 years of seabird data gathered by QPWS and the Field Management Program within the Great Barrier Reef (Driscoll 2013). There was disappointment in the finding that less than half of the data was useful for this purpose and a close reading of the report suggested that trend data was only reliable from Michaelmas Cay from where we had monthly surveys. The reasons for these results were:

- The data had been gathered to document where and when seabirds bred, i.e. an atlas, rather than to understand trends.
- The timing of site visits was irregular – the counts often being an ‘add on’ to other scheduled work and often didn’t adequately consider the breeding biology.
- Methods for counting seabirds varied along the coast, and at individual sites through time, introducing an unknown and in some cases unknowable level of bias into the data.

To rectify these issues the Ecological Assessment Unit undertook a significant revision of the seabird parts of the 2012 Coastal Bird Monitoring and Information Strategy and associated field methods. We also designed and implemented a state-wide training package for field staff. However, mindful that monitoring uses resources that could otherwise be used on other priorities, we sought to establish whether new technologies offered cost effective alternatives to first person site visits to monitor important seabird colonies and populations.

Our initial intent was to explore technologies that enabled QPWS to gather data from remote, logistically challenging or costly sites, without regular site visits by staff. However, we expanded this remit to include options that could improve the precision and accuracy of estimates of the numbers of breeding seabirds whether staff are present or absent. Precision is a key aspect of the data we gather and is a measure of our confidence in the data. Imprecise data means large confidence intervals. It is easier to detect change in a series of precise data points than an equivalent series with lower precision. Improving precision improves our capacity to detect changes and to do so more quickly. Ideally, our data would be accurate and precise, but accuracy is only essential if knowing the actual size of the population is critical (see Figure 2). It is often more important, in terms of species management, to know with certainty (precision) that a population is increasing or decreasing than to have an accurate estimate of population size.

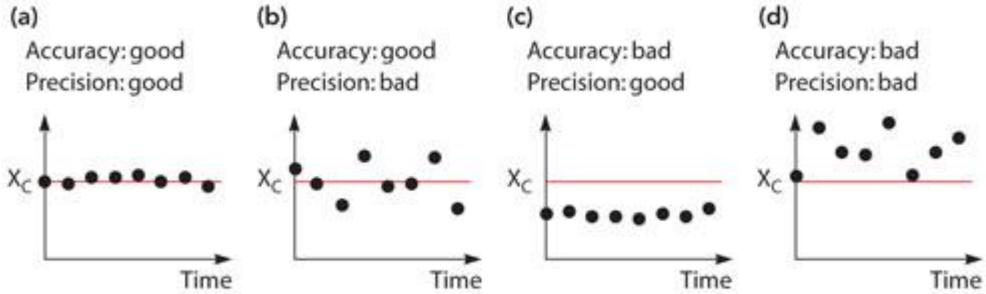

Figure 2: The difference between precision and accuracy, the red line is the trend of interest and the black dots are data points.

Literature review

A literature review by UQ Masters student and EHP Conservation Officer, Rebecca Richardson, highlighted the possibility of using acoustic monitoring and confirmed that cameras, acoustics and drones all had unproven potential (Appendix 1). She noted that trail cameras were being used extensively to monitor terrestrial mammals and that some people had started using drones for monitoring wildlife. While resources required for owning and operating longer-range drones were prohibitive, those associated with smaller drones were not. Rebecca noted that acoustic surveys were also being trialled for a few species with distinctive calls and that the University of California had started detecting seabirds from acoustic data.

In reviewing Rebecca's findings, we noted that drones required a staff member to attend the site and therefore did not meet our expectations of an autonomous system. Cameras and sound recorders offered the greatest potential to gather data on remote seabird colonies without staff presence. We subsequently found out about a camera system for monitoring Adelie penguins in the Antarctic (Southwell & Emmerson 2015). Our interest in drones resumed as our concerns over the role of observer bias in our results increased, because they offer a tool for improving precision (Hodgson *et. al*. 2016).

Figure 3: Sooty terns. A. Mcdougall.

Methods

Camera

In 2012, we built and tested a Pan Tilt Zoom (PTZ) camera on North Keppel Island. We designed this proof of concept to evaluate whether a camera could record imagery of sufficient quality, and from a wide enough area, to identify and count seabirds on a typical sand cay. We also tested and developed the wireless links and solar charging systems to maintain the camera in the field.

In 2013, this unit was redeployed to Michaelmas Cay near Cairns. This trial sought to establish whether we could count nesting seabirds reliably, over what range species could be differentiated, whether loafing and nesting seabirds could be separated, whether automated PTZ patrol routine and data storage would work, and to assess reliability in a more exposed location.

The camera deployed was a Vivotek SD836e (http://www.vivotek.com/sd8363e/#views:view=jplist-grid-view) which recorded 1080p HD video and 2 megapixel still images. It could rotate horizontally through 360° and vertically through just over 180°. This enabled a 360º view of everything between slightly above the horizon and vertically underneath the camera; slightly more than a hemisphere. It had a 20X optical zoom creating a 3° to 55° field of view. Imagery was stored on a solid-state thumb drive attached to a network attached storage (NAS) device. The entire system connected back to the QPWS intranet via a 3G mobile phone data connection.

Figure 4: Installing the fixed camera at Michaelmas Cay. A. McDougall.

The camera malfunctioned after unanticipated king tide flooding in February 2013. A replacement system, with improved water proofing, was deployed in May 2014. This in turn failed in March 2015 after more king tide flooding (see Figure 17).

We programmed the camera to perform a PTZ routine designed to record imagery from the entire island several times a day. Each routine consisted of several 360° horizontal rotations each at a lower tilt and wider angle of zoom than the preceding rotation. Rotations when the camera pointed closest to the horizontal used higher levels of zoom to cover the more distant parts of the cay in high resolution. As the camera tilted down the zoom was opened up to wider angles to take in equivalent areas (Figure 2). Each rotation consisted of several adjacent frames in which the camera would pause for a few seconds to record multiple still images or a short video before panning to the next. The frames overlapped slightly to ensure total coverage.

Three observers (Andrew McDougall, Gemma Haley and David Stewart), provided with identical training, counted the number of birds of each species they estimated to be breeding or loafing in each field of view from a subset of this imagery.

Figure 5: An example of how images at different zooms are used to cover the entire island. Small images are those viewed at the highest zoom.

Figure 6: A view of sooty terns closer than 10 metres to the camera pole.

Figure 7: A view of sooty terns, common noddies and brown boobies further than 40 metres from the pole.

Acoustics

We are conducting acoustic trials at North West Island, Heron Island, East Fairfax Island, One Tree Island, Raine Island, Michaelmas Cay and Sisters Island (Table 1). Across these islands, we target wedge-tailed shearwaters, black noddies, common noddies, crested terns, bridled terns, red-footed, masked and brown boobies and Herald petrels. We deployed acoustic recording units with schedules that made most recordings around sunset and sunrise (1 minute every 10-30 minutes) when nesting seabirds are departing from or arriving at nests and are most vocal, with fewer recordings (1 minute per hour) spread across the rest of the day and night. We then counted the number of birds nesting and loafing within 5 and/or 10 and/or 20 metres (depending on the trial) of each recorder as frequently as possible. These counts provided a measure of the true density of nesting birds that we could then compare with estimates *of density obtained from acoustic signals. Trials began using* Wildlife Acoustics' Songmeter SM2+'s and later extended to Frontier Lab's Bioacoustic Recorders *(BARs) which are smaller and lighter and use rechargeable batteries.*

Data from both devices were recorded onto Class 10 UHS 1 (30-80 MB/s) SD (HC or XC) cards (32-128Gb)[26]*.* We retrieved these data at the end of the deployment or when batteries were changed in the devices. We transferred the data to a 14Tb Network Attached Storage device (NAS) in Rockhampton. Data were sent via the internet and File Transfer Protocol (FTP) or on hard drives in the post to Conservation Metrics Incorporated (CMI) in California who did the call analysis.

In situations where birdcalls do not frequently overlap and where ambient noise is relatively low, automated acoustic analysis of field recordings was carried out with custom detection and classification software developed by CMI. The approach uses a machine learning technique known as Deep Neural Networks to automate the detection of sounds on field recordings that have properties matching those measured from signals produced by target species. Deep Neural Networks (DNNs) are a powerful classification tool used to perform speech recognition, image recognition, and computer vision tasks.

CMI's approach splits field recordings into 2second clips and extracts measurements of 10 spectro-temporal features typically found in animal sounds. CMI then train a DNN classification model for each species of interest using training and cross-validation datasets containing examples of "positive" sounds (vocalizations from target species) and a representative example

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[26] *for more info on SD cards;*
https://en.wikipedia.org/wiki/Secure_Digital#Ultra_High_Speed_.28UHS.29_bus

of “negative” sound clips (i.e. sound clips from the soundscape at all survey sites that do not contain the species of interest). The DNNs learn which combination of spectro-temporal features best differentiates target sounds from other sounds in the environment, and each model can then classify sounds on new acoustic data from survey sites. Each classification model returns a probability that a given 2second window of field recordings contains the target species vocalization. These data are then added to calculate the numbers of calls per minute in each sound file.

Initially all events flagged by the automated classification model were manually reviewed to confirm correct identification and remove misidentified sounds. The proportion of events manually reviewed can be reduced as confidence in the algorithm improves with repeated use.

As bridled terns frequently created saturated soundscapes in which calls overlapped and were indistinguishable from one another, CMI also analysed spectral energy in the bandwidth between 2200 HZ and 2400 Hz where bridled tern calls have the most energy.

Table 1: Acoustic trial sites

Site	*Year*	*Units*	*Species*
North West, Mast Head, Wreck and West Hoskyns	*2012/13*	*7*	*Wedge-tailed shearwater, black noddy*
North West	*2013/14*	*7*	*Wedge-tailed shearwater, black noddy*
North West	*2014/15*	*9*	*Wedge-tailed shearwater, black noddy*

North West	*2015/16*	*30*	*Wedge-tailed shearwater, black noddy*
North West, Mast Head, Heron, Lady Musgrave	*2016/17*	*45*	*Wedge-tailed shearwater, black noddy*
Sisters Island	*2014/15*	*4*	*Bridled tern*
One Tree Island	*2016/17*	*3*	*Bridled tern*
Michaelmas Cay	*2015/16/17*	*3*	*Crested tern, common noddy, sooty tern*
Bushy Island	*2016/17*	*3*	*Black noddy*
East Fairfax	*2015/16*	*7*	*Brown booby*
East Fairfax	*2016/2017*	*9*	*Brown booby*
Raine Island	*2017/18*	*25*	*Commom noddy, brown booby, masked booby, red-footed booby, Herald petrel*

Drones

In November 2016, we engaged ecological consultant Ian Denley to fly his drone, a DJI Phantom 4 (http://www.dji.com/phantom-4), over the acoustic test sites at East Fairfax Island and One Tree Island (Table 1). Each acoustic sensor had small "cairns" of yellow spray painted rocks describing f5 metres and 10 metres radii around each acoustic sensor (see figures 8, 24 and 25).

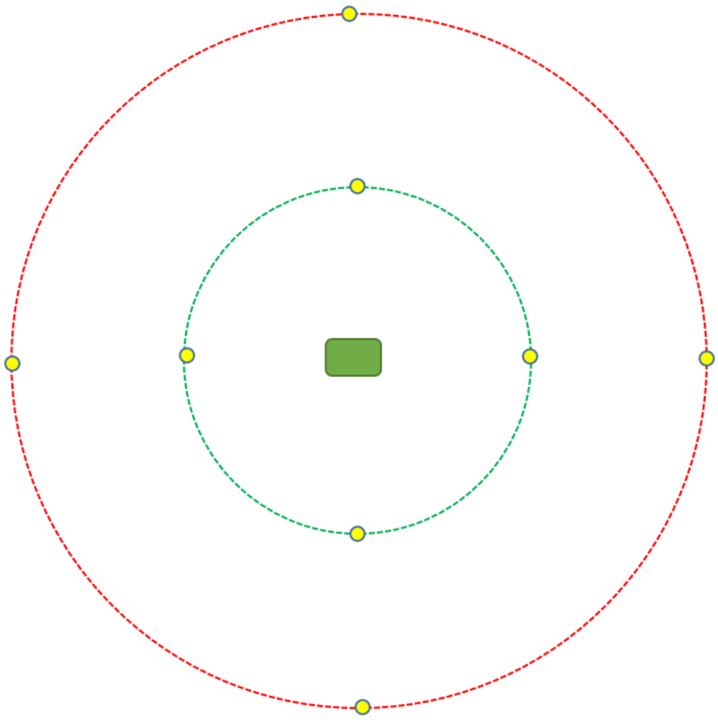

Figure 8: A representation of the layout of an acoustic trial with green oblong at the centre being the recorder and the two circles different radii from the recorder and the yellow dots being markers.

The drone recorded stills (12.4 million effective pixels with a field of view 94° or 20 millimetre as measured in 35 millimetre camera format equivalent) and 4K video (4096×2160 pixels, same field of view) from 50 metres and 30 metres above ground level (AGL). Transits across the island occurred at 50 metres and the drone descended from 50 metres to 30 above each site. An observer (Graham Hemson) obtained ground counts at each site from a three-step stepladder to improve his angle of viewing. Brown boobies and bridled terns were divided into nesting (birds on eggs, nests and chicks) and loafing (birds loafing or otherwise present but not associated with a nest).

In April 2017, a DJI Inspire 1 with a Zenmuse X3 camera (effectively identical to the camera on a Phantom 4) was used to obtain imagery from above two colonies of lesser frigatebirds at Raine Island. All imagery was recorded from 30m above ground and ground counts were done by Graham Hemson. All adult lesser frigatebirds on the ground within a breeding colony were assumed to be nesting as frigatebirds only aggregate on the ground to nest.

Counts from imagery were done in two ways:

1. Stills were opened in Inkscape, a photo editing program that allows markers to be pasted onto a photograph and which can count these markers. Markers were pasted onto each nesting booby and then the number of markers totalled automatically by Inkscape.
2. Video was analysed using a 4K monitor to ensure the benefits of the full resolution could be realised. Video was observed and counts made. Video was paused rewound and cued as required.

All counts; ground, video and still, were undertaken by the same observer, Graham Hemson.

Results

PTZ Camera; Michaelmas Cay

Costs

The camera, solar panel, mounts, battery, charge regulator, NAS, Ethernet switch and router cost in the order of $10,000. Adding satellite connectivity to enable remote access at sites beyond mobile phone range would add approximately $5000 to the build cost. The review and analysis of the recorded imagery is potentially as time consuming as a site visit. It involves selecting images with enough clarity, reviewing each image and tagging each breeding bird or target of interest and then calculating totals. However this action can be scheduled at a time convenient to staff, whereas gathering data in the field is constrained by work programs, the availability of vessels and weather.

Counts

Counts of all adult seabirds (breeding and loafing) by three different observers from 11 different images showed that counts were generally similar and graphs show that all observers tracked the same underlying patterns of abundance (Figures 9-11). However, when we asked the same observers to determine the number of breeding birds from the same images the results were markedly inconsistent (Figures 12-14). In one example one observer failed to identify any breeding common noddies in a series of images, another 30 and the final observer 258 (Figure 13). Similar results were recorded for sooty terns and brown boobies although the numbers of boobies were an order of magnitude lower than terns and noddies. Coefficients of variation (a measure of variability proportional to the total number of observations) were not consistently higher in images at higher or lower zooms indicating that variability in counts did not simply increase with distance.

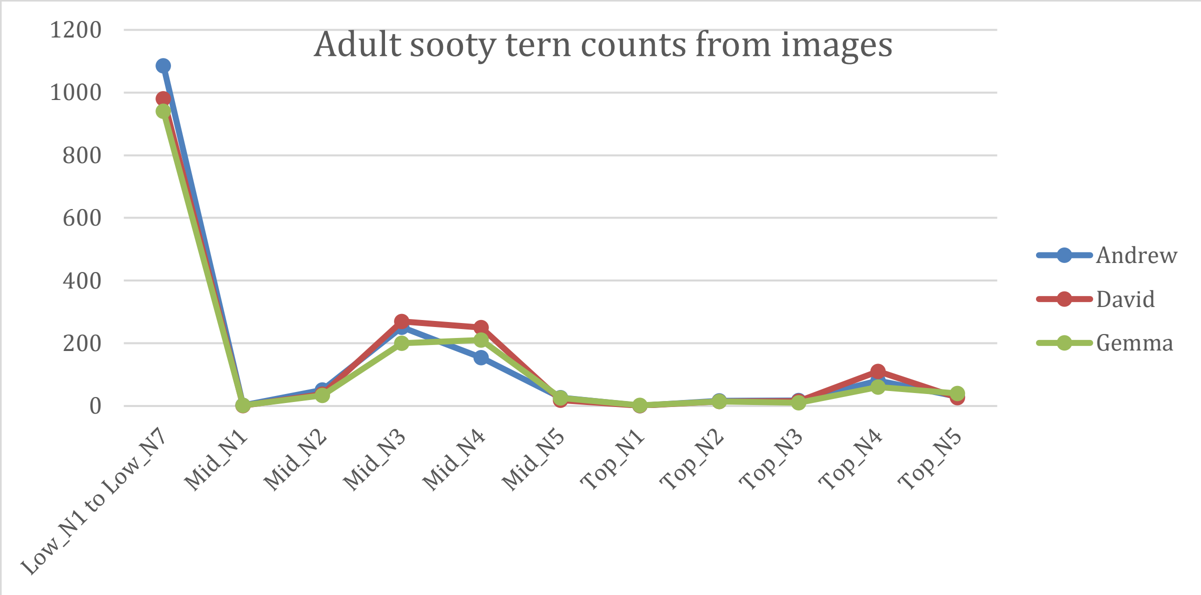

Figure 9: Numbers of adult sooty terns estimated from 11 images by three observers

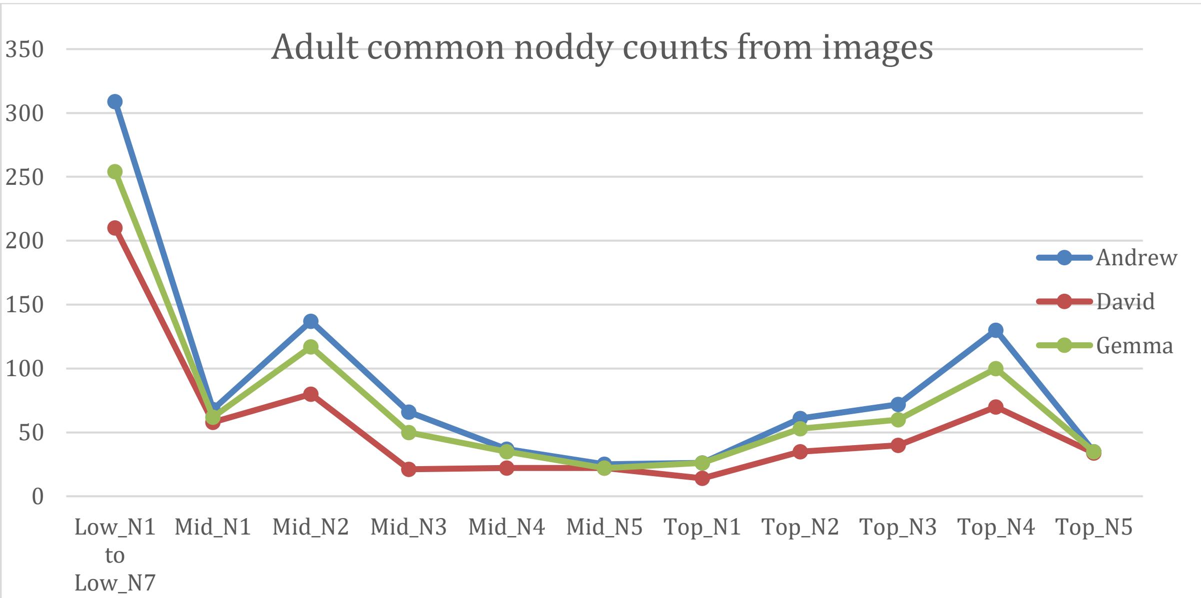

Figure 10: Numbers of adult common noddies estimated from 11 images by three observers.

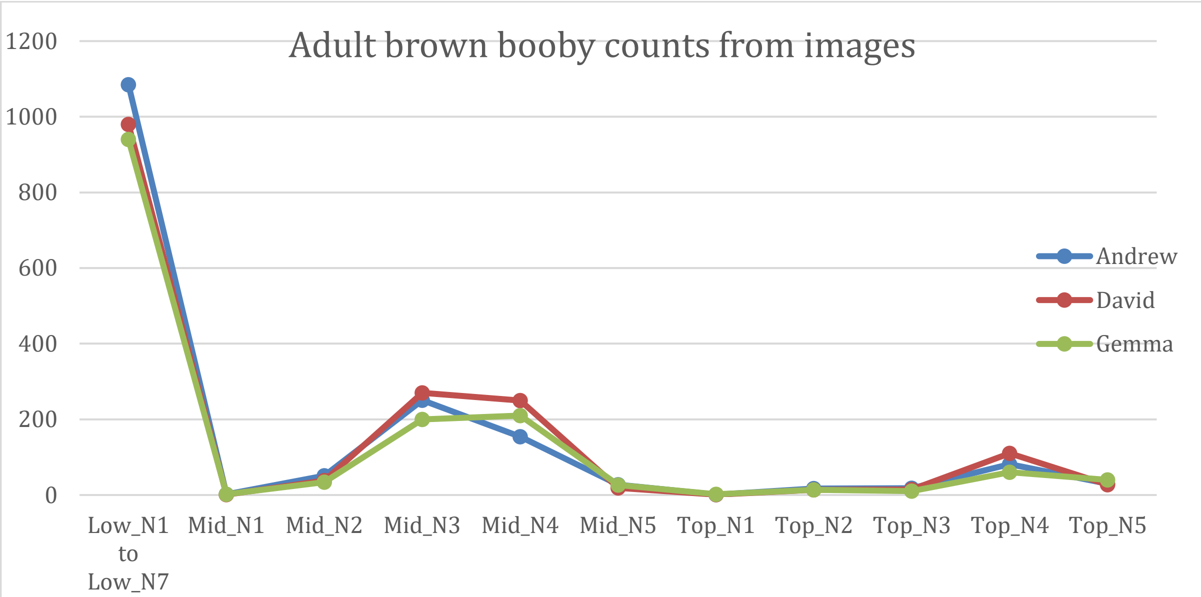

Figure 11: Numbers of adult brown boobies estimated from 11 images by three observers.

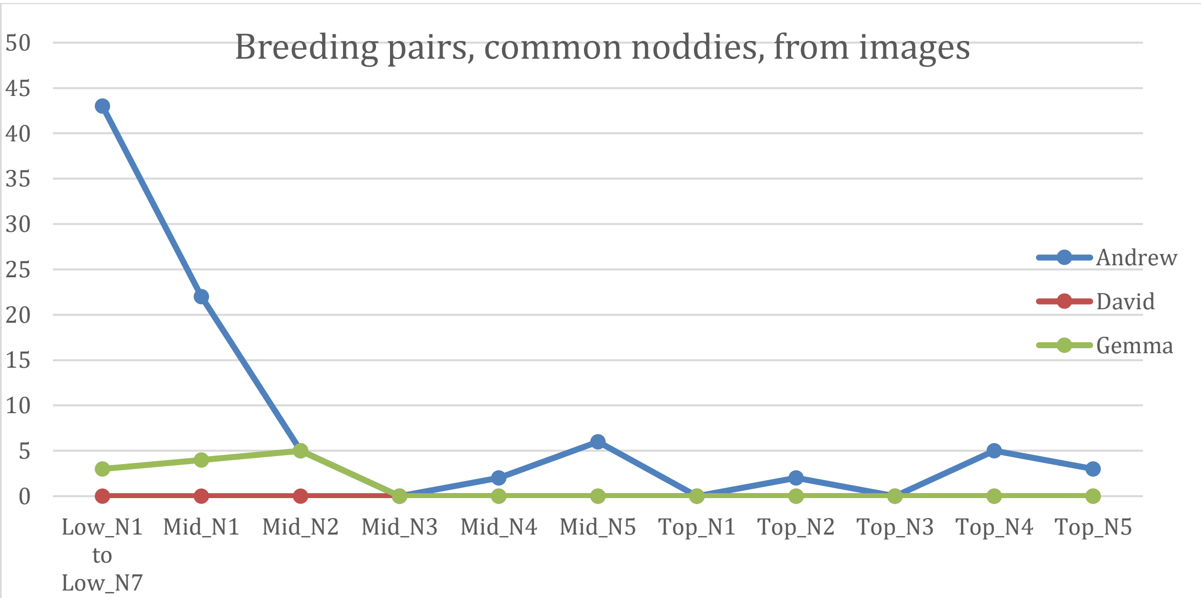

Figure 12: Numbers of breeding pairs of common noddies as estimated from 11 images by three observers.

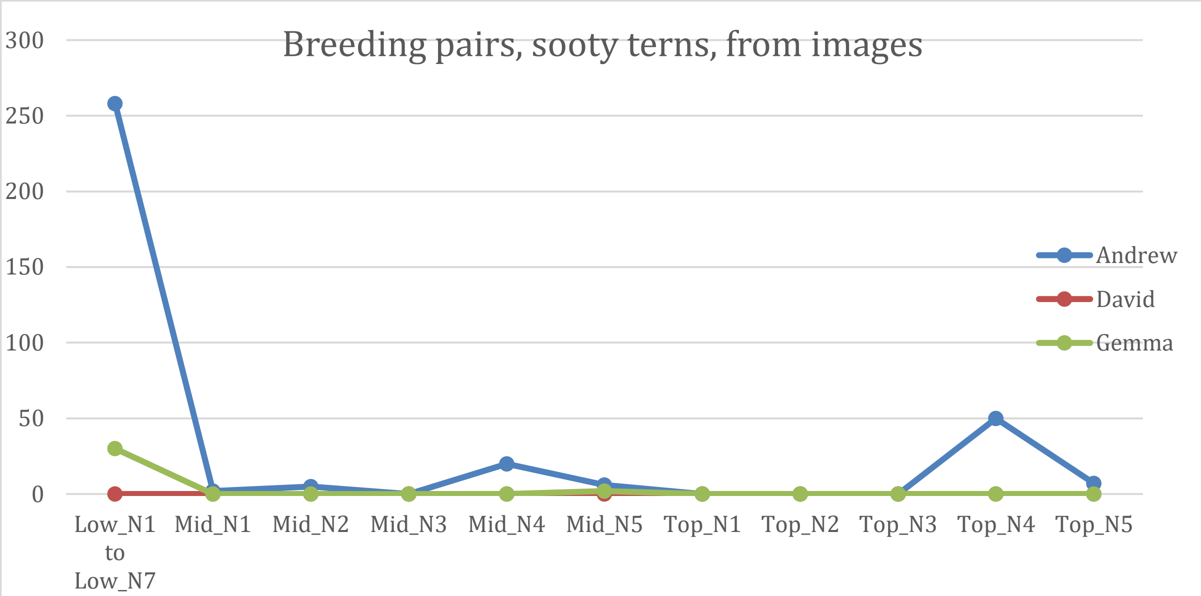

Figure 13: Numbers of breeding pairs of sooty terns as estimated from 11 images by three observers.

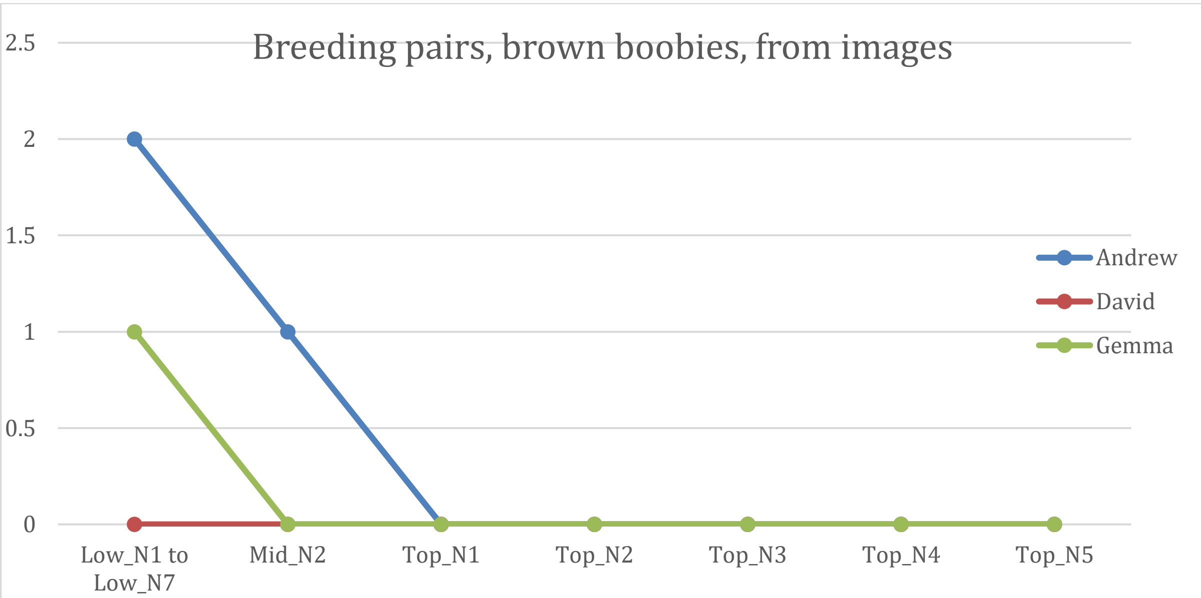

Figure 14: Numbers of breeding pairs of brown boobies as estimated from 11 images by three observers.

The data from the camera could be used to investigate seasonal patterns in breeding and to detect other events at the site. We observed tourists moving outside the area allowed for visitors, the landing of an emergency helicopter (Figure 15), and the flooding of the island during king tides (Figure 16). The flooding and associated reduction in vegetation has prompted concern over the security and sustainability of Michaelmas as a seabird nesting location and discussion about whether intervention is feasible or appropriate.

Figure 15: A medical rescue helicopter landing at Michaelmas Cayas recorded by the Michaelmas camera.

Figure 16: Flooding during a king tide at Michaelmas Cay as recorded by the Michaelmas camera.

Acoustics

Costs

We currently use Frontier Labs, Bioacoustic Recorders (BARs) which cost approximately $1000 per unit. The number of BARs required for a deployment is determined in part by the area to be surveyed and in part by our needs in terms of data precision and accuracy. At North West Island, we deploy 30 units, at Raine Island there are 25 units, while at Mast Head, Heron and Lady Musgrave Island there are five apiece. The batteries supplied are rechargeable and all unit failures (four from 60) have been warranty replacements. In the longer term, microphones and batteries will need replacing every three or so years as their performance deteriorates. Microphones cost $80, batteries cost $15 each and each unit uses four.

Data is currently analysed by an external contractor in the United States, Conservation Metrics Incorporated (CMI) who are world leaders in the analysis of patterns in acoustic and visual data. Their prices depend on the number of species analysed and the volume of data to be analysed. The volume is determined by the period of interest, the number of recordings per day and number of sensors. Costs are more during the initial set-up due to the development of species specific recognisers and the need to identify any site or species specific problems and iron these out before going operational.

The cost of analysing one month of data, from one sensor for one species, is $100. During the initial experimentation, we need to analyse more months for each species to identify the correct time window before refining the analysis down to concentrate on the appropriate time. In the initial phase we need to analyse every month of the breeding cycle to calculate the best period in which to focus future monitoring. The cost will therefore depend on the length of the cycle. For frigatebirds, it may be eight times the cost of the final deployment and for common noddies it may be three times. In terms of the numbers of sensors needed, this will be dependent on the size of the site and variation observed. The largest deployment we have is 30 sensors on North West for two species (+/- $6000 per annum to analyse one key month), 25 at Raine for five species and five on Heron, Mast Head and Lady Musgrave for two species.

Counts

Capricornia Cays; wedge-tailed shearwaters and black noddies

Trials to establish the relationship between an acoustic measure of abundance and actual abundance for wedge-tailed shearwaters in the Capricornia Cays were successful. Correlations between the wedge-tailed shearwater calls per minute scaled linearly, strongly and significantly with nest density within 10 metres (Figure 6). The relationship was consistent between years.

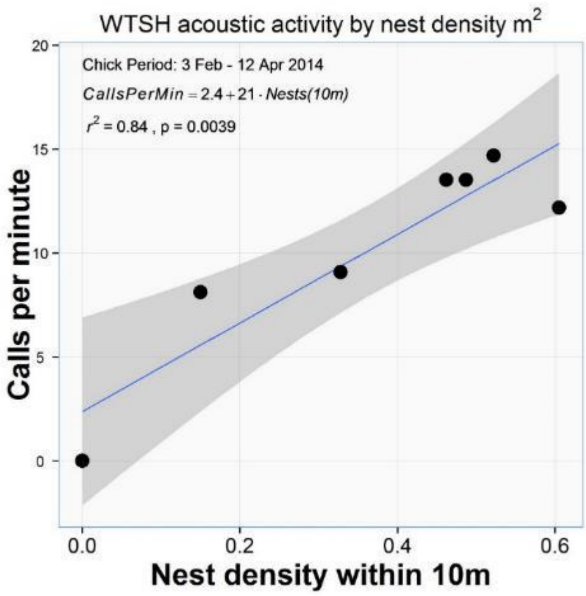

Figure 17: A correlation between wedge-tailed shearwater nest density and wedge-tailed shearwater calls per minute as identified by the deep neural network.

The correlation between black noddy calls and density was less strongly correlated, but was still linear, significant and positive.

CMI also developed recognisers for both wedge-tailed shearwater and black noddy chicks. In 2016 they identified a significant correlation between wedge-tailed shearwater chick call rates and burrow densities within 10 metres and 20 metres of the sensor at each site (10 metres: r-squared = 0.4, p = 0.00024; 20 metres: r-squared = 0.44, p = 0.000091) (Figure 17). There were also significant correlations between black noddy chick call rates and nest densities within 10 metres and 20 metres of each sensor (10 metres: r-squared = 0.25, p = 0.0059; 20 metres: r-squared = 0.29, p = 0.0028).

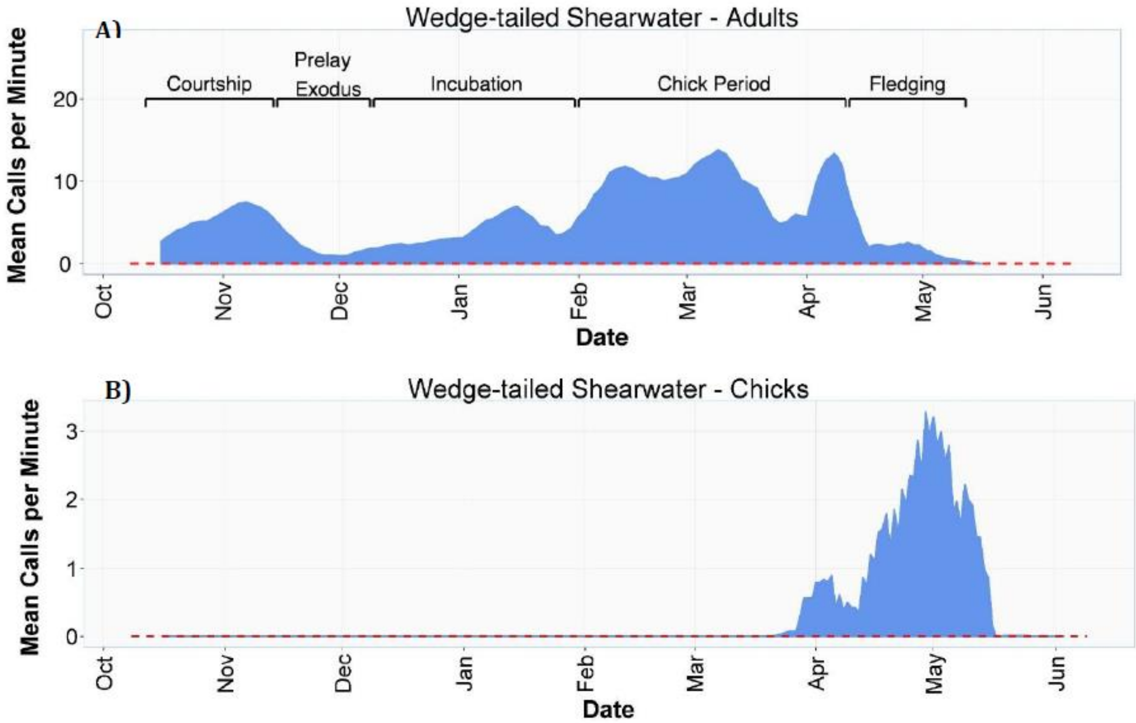

Figure 18: The frequency of calling of adult and chicks and the postulated phases of their reproductive cycle.

In addition to data on density of nesting pairs and chicks we were able to collect information on other aspects of the ecology of both species. We are able to identify exactly when wedge-tailed shearwaters arrived, when they left after courting and returned to lay eggs (prelay exodus). We could identify when chicks began to hatch, when the adults left, and when the last chicks fledged and left (figure 18). We could also investigate changes in behaviour across each day and how daily variations in calling associated with events like lunar phase and storms.

Capricornia Cays; brown boobies

Trials at East Fairfax Island on brown boobies also found significant strong positive linear correlations between nest density and call frequency. For boobies it is possible to separate calls of males and females and both correlated with nest density (figure 19 and 20). However, counts made around the sensors by drone highlighted that the ground counts are underestimates (Figure 23) and we need more drone counts at the site to establish what the actual numbers around each sensor are. It is highly likely that the significant relationship will remain but that the slope of the line may be different. We have not yet attempted to identify chick calls.

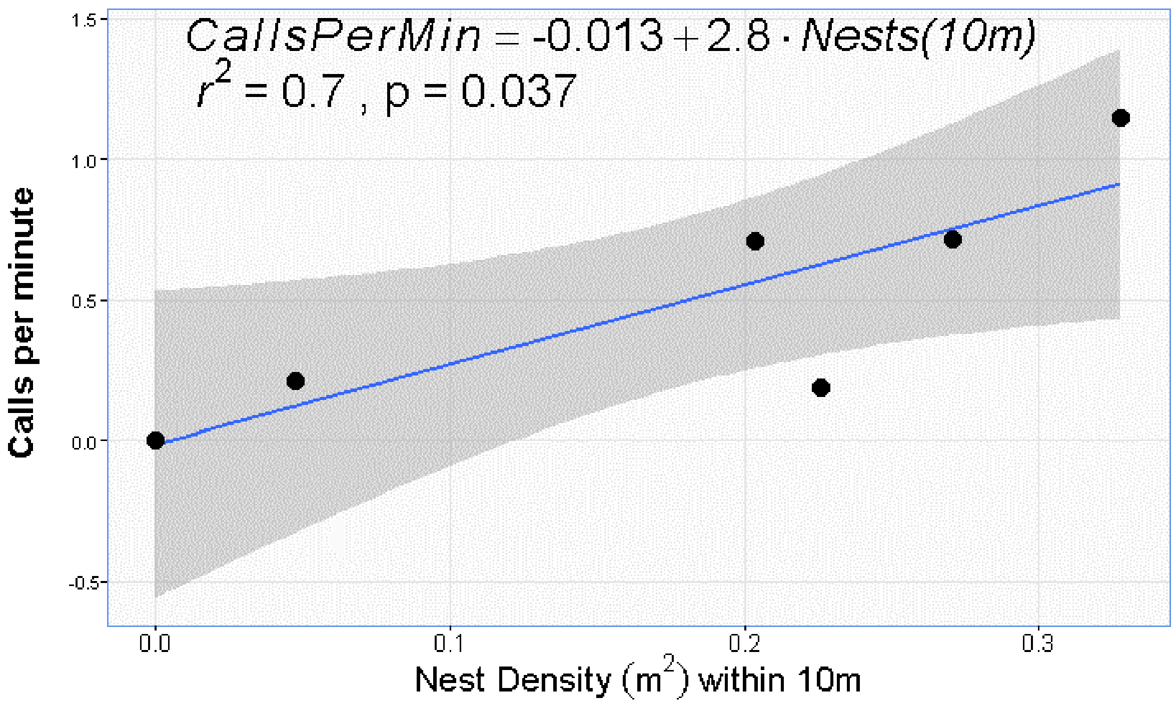

Figure 19: The density of brown booby nests within 10m of the recorder and the frequency of female calls.

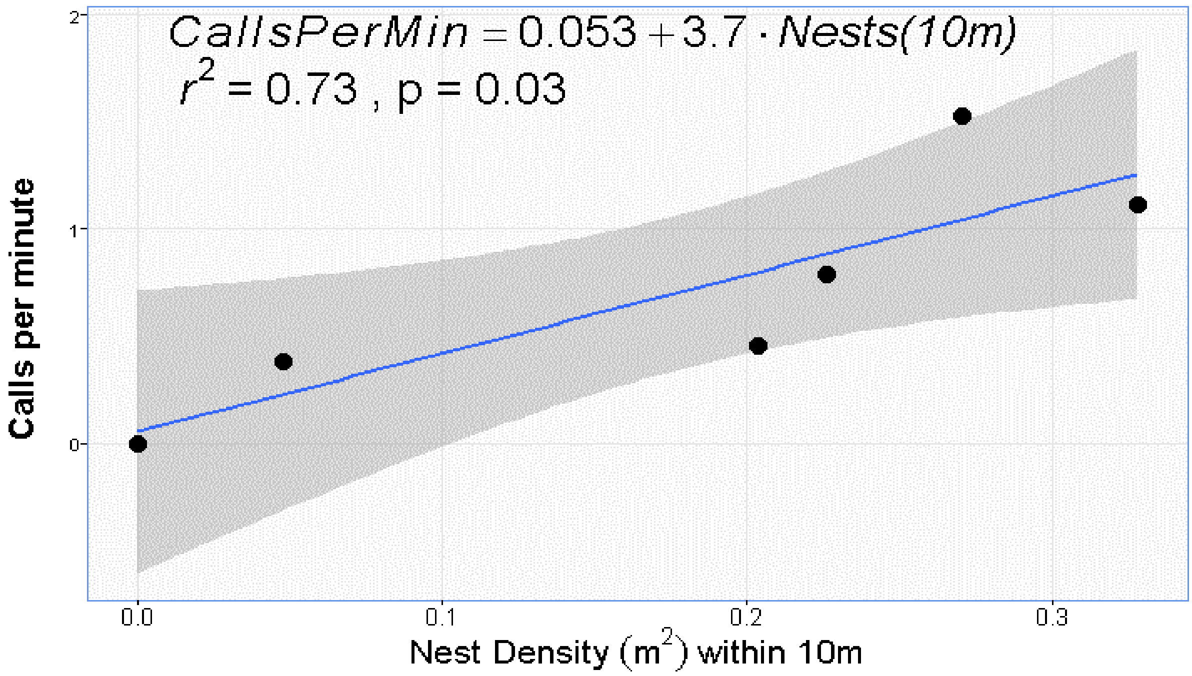

Figure 20: The density of brown booby nests within 10m of the recorder and the frequency of male calls.

Sisters Island; bridled tern

Ground counts of bridled terns were very difficult as they nest in thickly vegetated areas and hide their nests in cover. We were only able to establish five trial sites and get a single nest count during the deployment. Despite these limitations our counts were significantly positively correlated with call frequency (r^2=0.67, p=0.046). Spectral energy and nest density were also positively correlated but the relationship was not significant (r^2=0.78, p=0.088). Spectral energy and call frequency were extremely strongly correlated (r^2=0.94, p=0.001). This last correlation infers both spectral energy and call rates are tracking the same underlying variation; nest density.

One Tree Island; bridled tern

It proved impossible to count the numbers of nesting bridled terns around the acoustic sensors at One Tree Island because of the thick vegetation and we were unable to validate the relationship between calls and birds at this site.

Raine Island and Bushy Island

We have not downloaded the recorders or collated validation counts so far.

Figure 21: Lesser frigatebirds. G. Hemson.

Drones

Costs

Drones, including control equipment and software, can be purchased for as little as $3000 (DJI Phantom 4 Pro) but a robust all weather professional drone with high-resolution cameras better suited to the task (e.g. a DJI Matrice) is likely to be approximately $20,000. The processing of drone imagery is currently manual and would take-up a similar amount of time as site visit or counts made from a fixed camera. However, as for the fixed camera, this analysis can be scheduled at the analyst’s convenience. It is currently theoretically possible to count seabirds using pattern recognition/deep neural network type software. Pattern recognition using deep neural networks and other machine learning algorithms is a rapidly developing field and it is therefore unlikely that we would have to count birds in drone images manually for longer than five years.

Counts

Counts from drones and from the ground were only possible at a single site at One Tree Island as the vegetation prevented counts from either method at the remaining two sites. At the only site where a ground count could be made it proved impossible to discriminate between bridled terns and black noddies, or to establish whether either species were nesting, from the drone imagery.

The work at East Fairfax was more successful. Counts from the drone imagery at East Fairfax consistently identified more nesting birds than ground counts (Figure 18 and 19). However, they consistently underestimated the numbers of loafing birds (birds that were present on the ground but were not breeding). The certainty associated with identifying nesting and loafing birds was generally less than from a ground count. We rarely saw chicks, nests or eggs in drone imagery and instead were reliant on visual cues such as dead vegetation, nesting material and radial patterns of guano, to infer nesting was occurring (Figure 22).

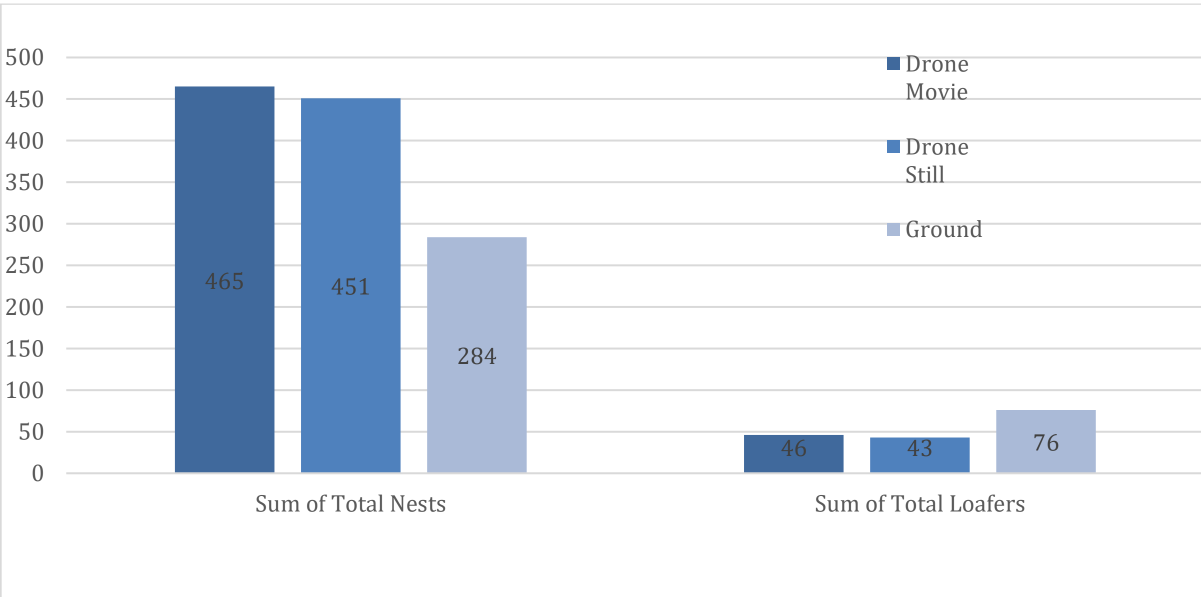

Figure 22: The numbers of brown booby nests and loafers as estimated from seven sites on East Fairfax Island using movies and stills captured from a Phantom 4 drone and from first person observation from the ground.

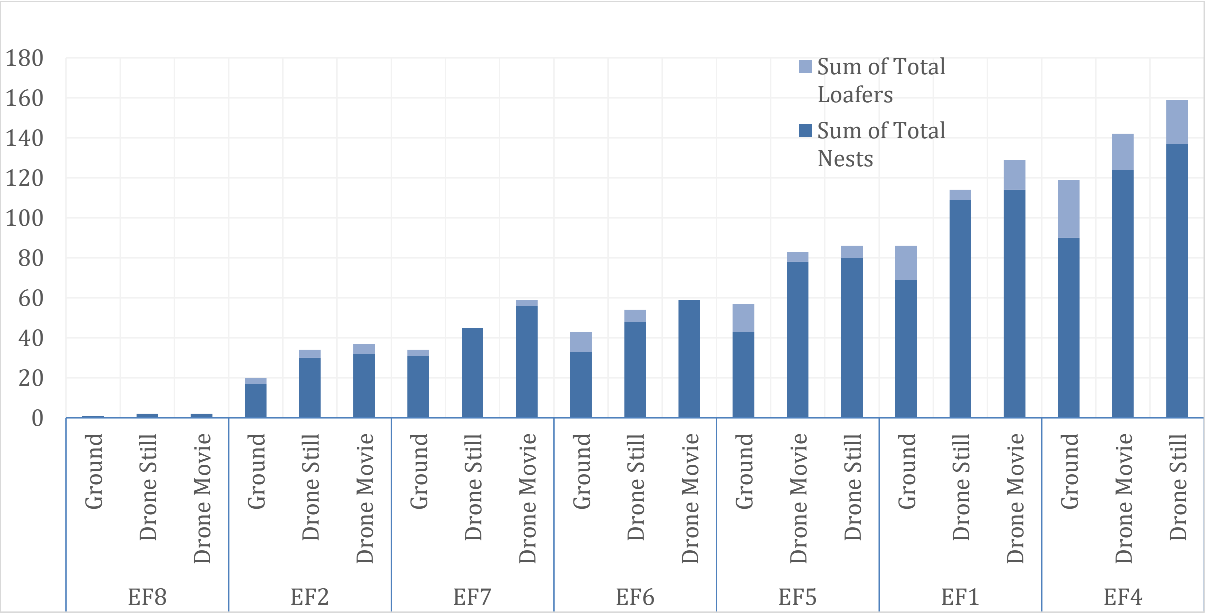

Figure 23: A site by site comparison of the numbers of brown booby nests and loafers as estimated from seven sites on East Fairfax Island using movies and stills captured from a Phantom 4 drone and from first person observation from the ground.

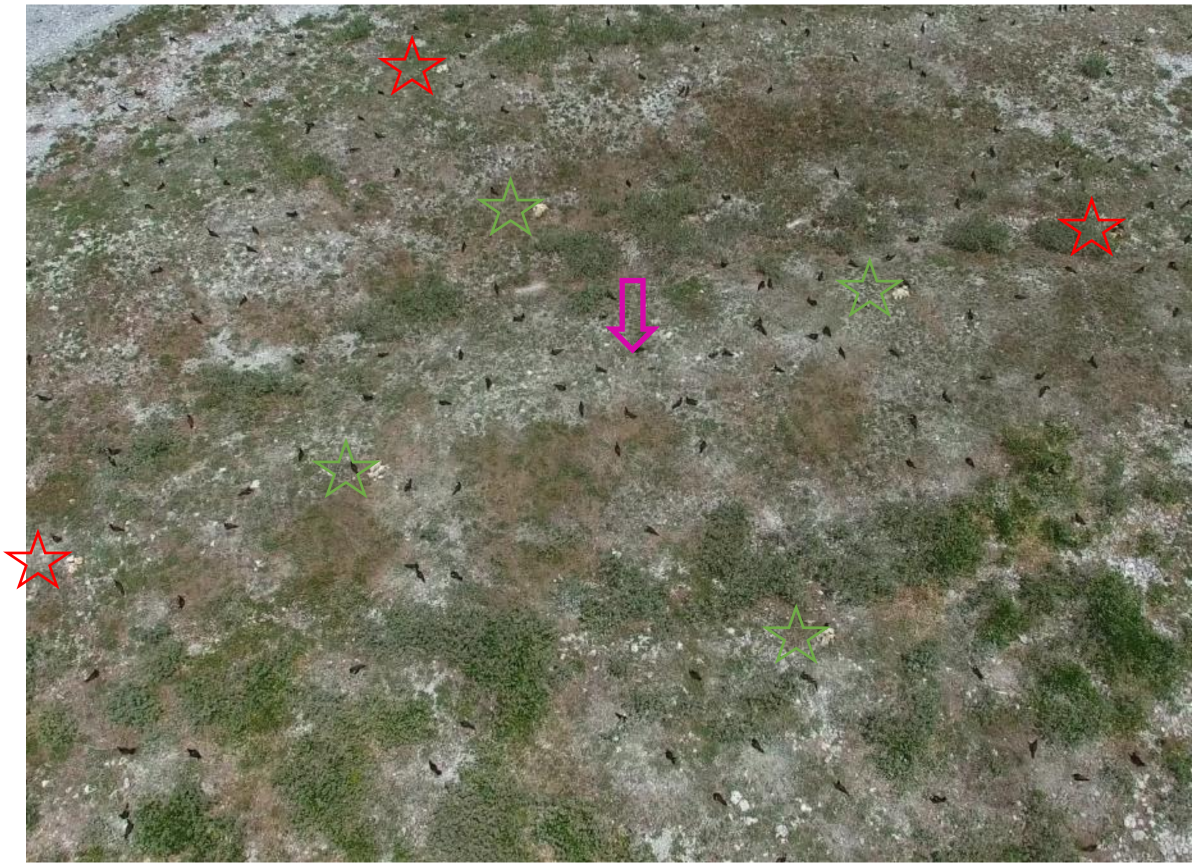

Figure 24: An oblique view of an acoustic trial site at East Fairfax Island highlighting the recorder (indicated by the purple arrow) and 5 metres (green stars) and 10 metres (red stars) radii markers used to count nesting brow boobies.

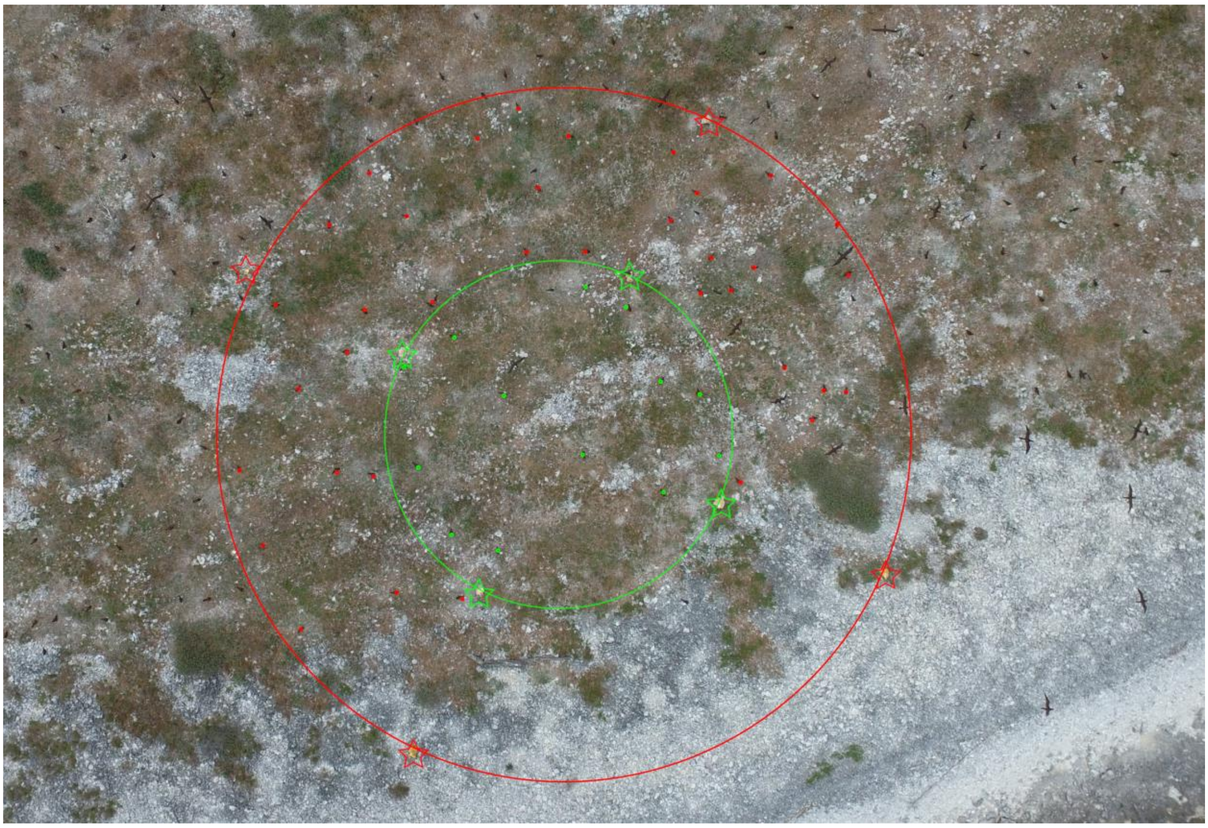

Figure 25: A vertical view of an acoustic trial site at East Fairfax Island highlighting 5 metres (green stars) and 10 metres (red stars) radii markers used to count nesting brow boobies (red dots).

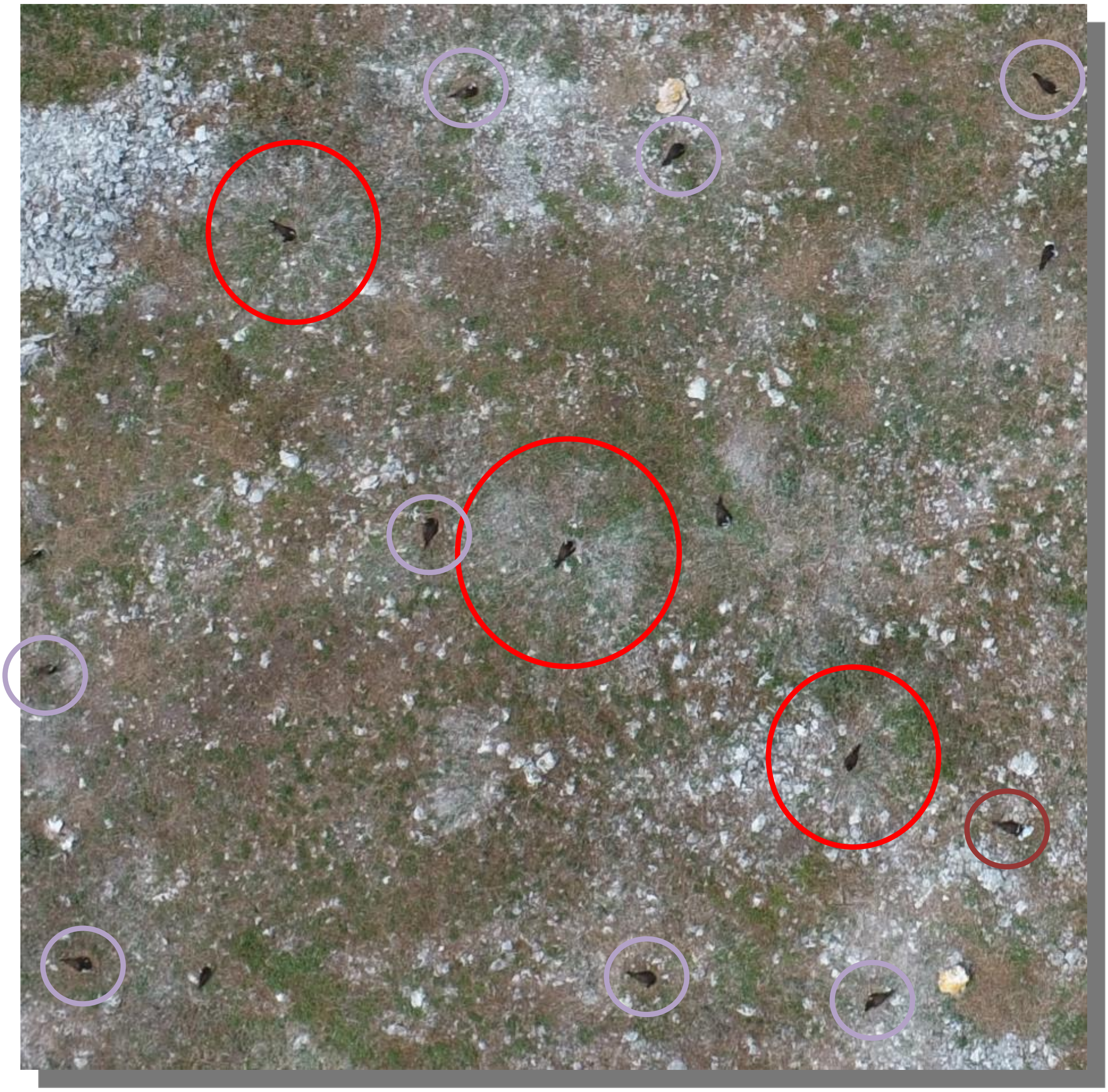

Figure 26: A cropped section of a vertical image. Note the faint radial 'spokes' or sprays of guano around some birds (red circles). These would seem to indicate a long term association of birds with a location and are a potential indicator of a nest. The birds in yellow circles appear to have a small brown area around them that suggests nesting material. The bird in the orange circle looks like it is on rocks and is probably loafing. The remaining four birds are not associated with clear cues that suggest they are loafing or nesting.

When compared with counts from drone-generated imagery, ground counts of two lesser frigatebird colonies at Raine both underestimated the number of birds in the two colonies.

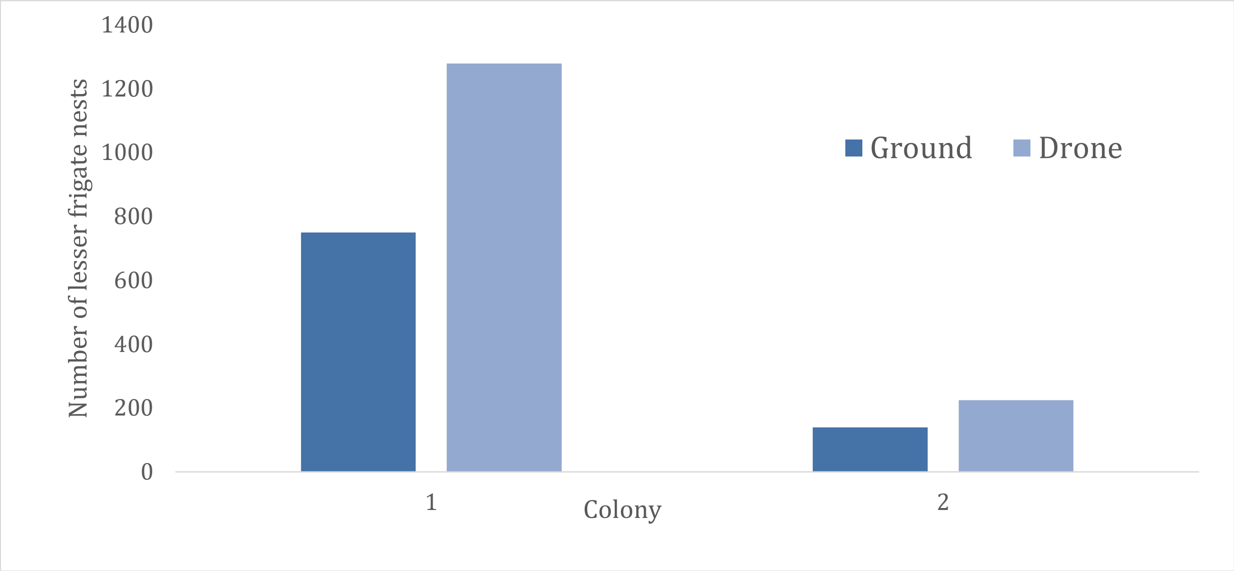

Figure 27: A graph comparing the numbers of lesser frigatebird nests in two colonies as estimated from ground counts and drone imagery.

Figure 28: An image of a lesser frigatebird colony (Colony 1 from Figure 27) photographed from a DJI Inspire 1 using a Zenmuse X3 (see Figure 34) camera at 30m AGL. We tagged each nesting bird with a red dot in Inkscape.

Discussion

Autonomous devices have the capacity to generate accurate and precise data and in many circumstances may be cost effective. Acoustics, cameras and drones have unique and varying strengths and weaknesses that lend them to, or make them unsuitable for, surveys of different species and habitat. Reductions in costs for autonomous devices and automated analysis are almost certain in the near future.

Cameras

Cameras can potentially gather data year round however the reliance on people to review and analyse imagery, and the relative complexity and expense of the systems, are impediments to operationalising their use.

Precision

The results for the camera were similar to, but worse than, those from the drone derived imagery. While all three observers estimated very similar numbers of birds from the imagery there was considerable variation amongst their estimates of the numbers of nesting birds. It seems likely that this variability was due to how each observer interpreted the instructions and the imagery, and that more practice and improved methods and instructions, would reduce it. An observer's field experience and observational skills are likely to influence their interpretation of the imagery.

Distance, height occlusion and perspective

In reviewing the imagery from the camera we noted that while it was relatively easy to count birds it did get harder with distance. Chromatic aberration and other optical limits of compact zoom lenses make high zoom images 'softer' and 'fuzzier' making identification difficult for very similar species. As distance increases, more and more birds and more of each bird are/is potentially occluded by birds or other objects in the foreground as the angle of vision becomes more acute. This becomes particularly problematic when determining whether birds are nesting. Firstly, birds often space themselves evenly when nesting thereby creating a regular pattern when viewed from above. This spacing might be the distance one bird can reach with its bill to peck a neighbour. Secondly, at acute angles the chance of viewing a chick or eggs sticking out from underneath a bird is greatly reduced by the occlusion noted above. However, the same is also true for vertical or near vertical imagery where no lateral view is available. As such looking down on nesting seabirds affords a useful view when assessing patterns but reduces the capacity to view under birds for eggs and chicks and extremely lateral views limit the ability to see patterns and eggs and chicks.

Ground based observers do better than the camera at detecting nesting because they can continually shift their perspective to minimise occlusion.

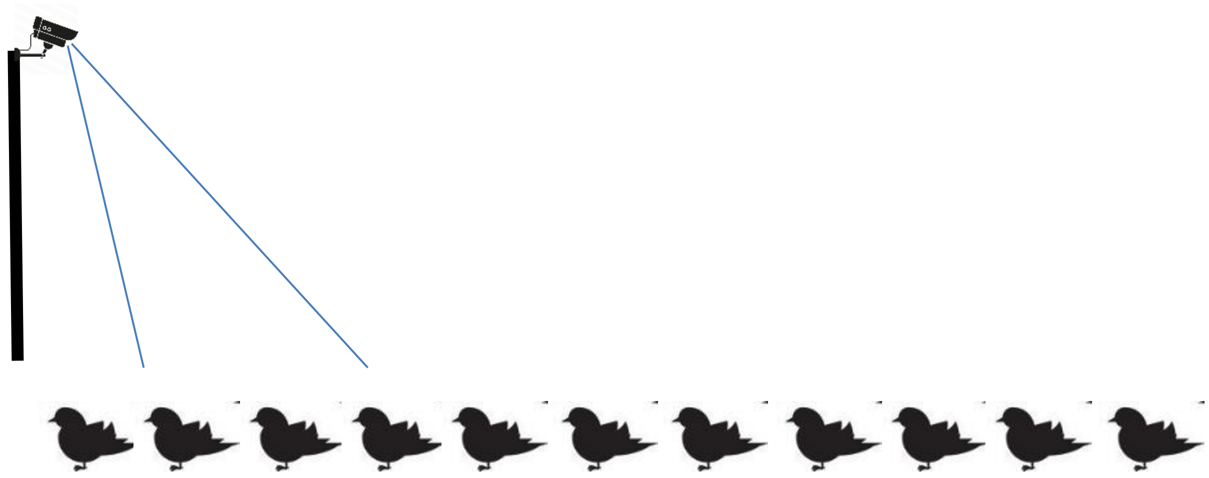

Figure 29: A schematic of how images of nesting birds near to the camera are largely free of occlusion interference by other birds but may cover eggs and chicks under the birds.

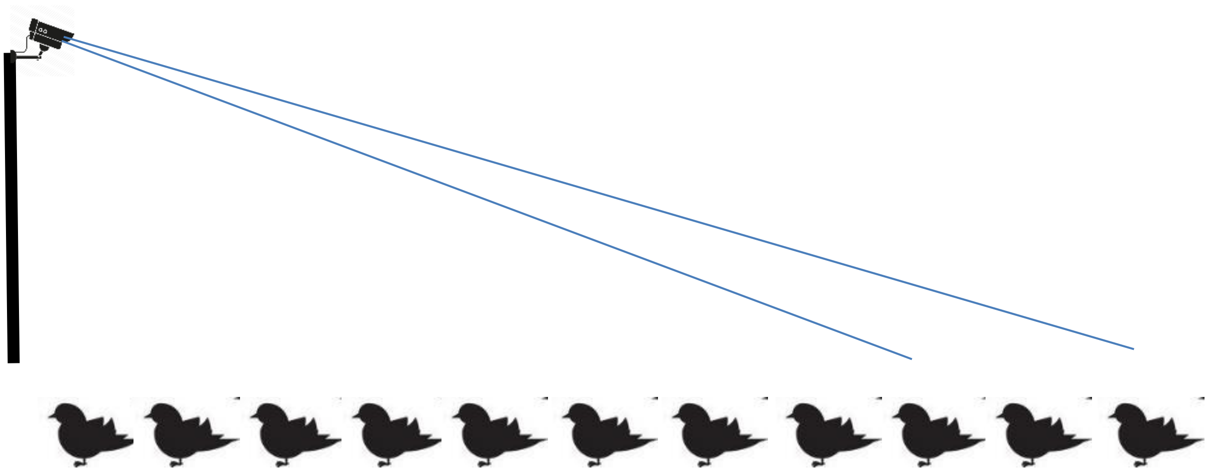

Figure 30: A schematic of how images of nesting birds far from the camera partially occlude each other and evidence of breeding from the camera.

Logistics

The system consists of a camera, custom mount and pole to ensure adequate height for coverage, stays to prevent wind wobble, a solar panel or wind turbine to generate power, a battery and charging system, and a network attached storage device. If communication or live updates are required the unit can be connected to mobile or satellite data coverage and requires an antenna and router. Many of these components are sensitive to temperature and moisture. Reliable housing for low-lying cays will need to be designed. There is currently no off-the-shelf integrated system designed for remote marine conditions. The Australian Institute of Marine Science has the capacity to design better systems than those deployed so far.

We are not sure whether fouling by bird droppings is a significant issue for remote deployments as people may have cleaned the solar panel and camera dome from time-to-time. It seems likely that frequent rain may keep important surfaces clean enough to function but clarity is of particular importance when it comes to identifying species and assessing breeding status. Misty, fogged or glass streaked with bird droppings will certainly limit the acuity.

None of the impediments are beyond our capacity to resolve but there is a considerable body of work to refine a camera system to the point where it is effective operationally.

Figure 31: A typical sight at Michaelmas Cay, sooty terns, brown boobies and common noddies. G. Hemson.

Acoustics

Acoustic sensors enable year round surveillance at a site and automated analysis free from observer bias. However, they are dependent upon the nests being relatively predictably or widely distributed and validation of the relationship between calls and nest density may take two or more seasons and some additional field work.

Precision

Acoustic trials have been successful for all species tested but not for all sites. The exceptions being: Michaelmas Cay, where the intensity of seabird noise and the density of birds was so high that it effectively saturated recordings making call recognition impossible; and for bridled terns on One Tree Island where the density of the vegetation made it impossible to do the on-ground count necessary to validate the acoustic data. That we have been able to validate bridled tern data from Sister's Island is encouraging and we may be able to extrapolate to sites like One Tree. At all the other sites there was a positive correlation between the numbers of calls identified per minute and the number of nests in proximity to the microphone. At some sites the correlation was not statistically significant. This was probably due to a combination of small sample size and errors in the ground counts used to validate the method. The errors in the ground counts may have arisen because dense cover prevented accurate counts (One Tree) or an inability to see the entire circular area around a microphone and vegetation such as at East Fairfax (see results from drone trials conducted around the acoustic sensors). In the case of East Fairfax where ground counts underestimated numbers, this may not adversely influence precision, but it has compromised our accuracy. While we are quite confident that the increase in call frequency is proportional to an increase in nest density we do not have the actual numbers counted correct so we cannot reliably predict numbers. From our longest running and most successful trials we have established significant linear positive relationships between calling and nest density for wedge-tailed shearwater chicks and adults and have modelled the statistical power for our array of 45 sensors to detect change.

Distance and dispersion

Sound attenuates in proportion to the square of the distance from the source. This is not a problem in a relatively homogenously dispersed colony of birds, such as brown boobies on East Fairfax and wedge-tailed shearwaters on North West, because we are aiming to sample variation in the density of nests from a wide area. We use this variation to calculate an average density from which to extrapolate a population. As long as there are sufficient microphones spread across a representative sample of the range of density, as calculated from a power analysis, we should produce robust estimates of abundance. In the case of seabirds that nest in small discrete colonies, the attenuation of sound and the lack of dispersion creates uncertainty as to whether a lack of detected calls is due to the absence of birds or just that they have nested away from the microphones. A different acoustic measure is obtained 10 metres from a colony of 100 crested terns compared to 20 metres or 50 metres from the same colony. This creates a sampling problem for species such as many of the terns that nest in tightly packed discrete colonies that may move between seasons.

Because of this issue, acoustic monitoring should be limited, at least in the short to medium term, to those species that disperse themselves relatively evenly across nesting sites. These include brown, masked and red-footed boobies, wedge-tailed shearwaters, black and common noddies, sooty and bridled terns. Lesser frigatebirds, which despite nesting in discrete colonies may nest in predictable colonies in the same areas each year may also be suitable.

It may be possible to overcome the aforementioned problems associated with species that nest in discrete colonies, at some locations, by putting out enough sensors such that there will always be one within range of a colony. This strategy would need to be modelled, tested, and the cost of implementation evaluated against other monitoring options.

The analysis

The use of automated pattern recognition and deep neural network machine learning algorithms is both a strength and weakness for acoustics. The only things staff have to do currently is download cards, copy data and post hard drives because the analysis is outsourced. The analysis costs money. In the long-term, it is probable that the hardware and software to undertake these analyses will become more widely available and easier to use. We are working on ways to reduce the volume of data that has to be analysed and this will have impacts on the costs of ongoing analyses.

Analyses for new species take longer than for species for which we have identified peak daily and annual calling times and for which we have established accurate call recognition algorithms. For the latter species we no longer need to analyse an entire seasons worth of recordings from across the day but can instead focus on a particular window of time to obtain the data we need.

Entire recordings can be stored for future analysis if new methods or questions arise in the future.

The vital correlation

One important aspect of the acoustic method is that it requires ground truthing. We need to calculate the relationship between the true density of nesting birds and the acoustic index. The speed at which this can be achieved is proportional to the number of sensors deployed (the more the better), the number of well-spaced counts (i.e. not many counts on one day) and the accuracy of these counts. Too few counts or sensors, or inaccurate counts, slows down the process. The most accurate and precise counts can be obtained by combining drone imagery with ground counts. The former ensures no birds are overlooked and the latter provides proof of nesting and can supply a minimum number of loafers and/or a proportion of nesters to loafers.

Figure 32: An acoustic trial site at East Fairfax Island. G. Hemson.

Drones

The precision of counts from drones was found to be dependent on vegetation cover, the size of the target species, the similarity of other species cohabiting at a site, and the ability to detect evidence of breeding. In the One Tree Island trial the target species, bridled tern, was relatively small, occurred with a similar sized and coloured species (black noddy), and nested in relatively thick low vegetation. Because of these factors it was impossible to count bridled terns or identify evidence of breeding. In contrast, the work at East Fairfax Island with relatively large brown boobies, and at Raine Island with lesser frigatebirds in single species colonies, was much more successful. Counts from the ground made at the same time were noticeably short of the actual numbers recorded in the drone imagery. However, discriminating loafing birds from nesting birds on the drone imagery was difficult or impossible from direct evidence (nests, chicks, eggs). Instead, we either assumed nesting based on our understanding of the biology of the species (lesser frigatebirds only aggregate on land to breed) and ground observations of nesting, and/or from visible cues that are likely to be indicators of nesting. These cues included evidence of protracted presence (rings of ejected guano and dead vegetation) or nesting material and regular spacing between birds.

Breeding or loafing and associated errors and bias

Overall drones enable us to get better counts of the total number of large birds nesting in the open (e.g. boobies, frigatebirds) than ground counts. These results are consistent with other published work on the subject (Hodgson et. al., 2016). However, we cannot reliably discriminate between nesting and loafing birds and this may introduce an unknown amount of error into our estimates. The magnitude of this error is likely to fluctuate between species, the time in the breeding season and the time of day, as the proportion of nesting/loafing birds changes. In the case of lesser frigatebirds, it may be reasonable to assume that the error is insignificant as adults birds are rarely on the ground in large numbers unless breeding. Furthermore, the spacing and size of the colonies are potential indicators of breeding activity that need little validation. In the case of boobies, the magnitude of the error is likely to be higher and less predictable as large numbers of birds often rest among nesting birds.

We can speculate that birds that space themselves in a specific manner to nest, or select specific habitats or time of year to do so, may be amenable to accurate estimation of breeding numbers from drone imagery. However, for other species we need to identify correlates of breeding that can be used to identify breeding birds from drone imagery.

Resolution and size

The drone trials we have undertaken so far involved consumer quality 4K wide-angle fixed focal length cameras from 30-50 metres above ground. It is evident from reviewing this imagery that our capacity to identify species or evidence of nesting is limited by the effective resolution of imagery and the number of pixels that a bird occupies on the sensor. Put simply, if all other factors are equal, the more pixels a bird occupies the easier it is to identify as there is no definition beyond the pixel. The number of pixels a target occupies is known as the effective resolution. Effective resolution can be increased by increasing the focal length (zoom) of a lens, increasing the number of pixels per unit area in the sensor, and flying lower (Figure 33, 34 and 35).

We have adopted guidelines limiting flight over seabird colonies to no less than 60m to minimise disturbance. It is therefore unlikely that the effective resolution from current consumer drones and wide-angle lenses will be sufficient for many situations and species. We could only reliably detect the largest species in single species colonies from a Phantom 4. However, it is highly probable that the effective resolutions obtained from consumer drones will improve rapidly (e.g. Phantom 4 Pro).

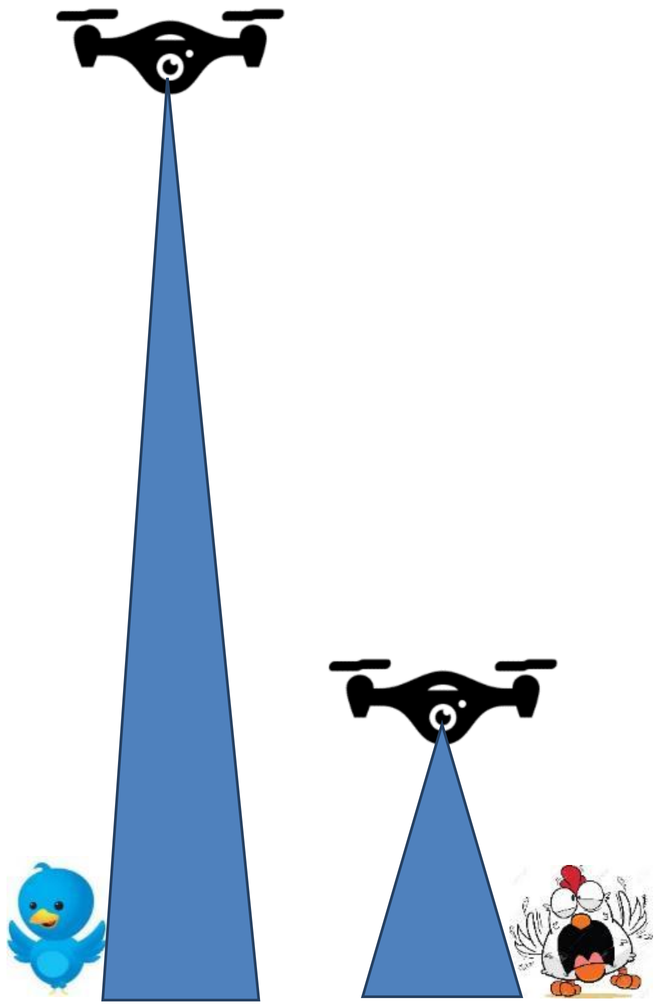

Figure 33: A schematic showing the relationship between zoom, altitude and disturbance. If all other factors are equal, a drone with more zoom can get the same resolution imagery and coverage from a higher altitude than a drone with wide-angle lens.

The camera used in initial trials (Phantom 4) had 12.4 million pixels on its sensor and a 4 millimetre lens to get wide-angle imagery on a small sensor. A 4 millimetre lens is extremely wide-angle, making it prone to curvilinear distortion in the images (fisheye type effects). The equivalent consumer drone today, the Phantom 4 Pro, has 20 million pixels on a larger sensor and a 9 millimetre lens and the latest X7 camera for the Inspire or Matrice professional drones has 24 million pixels and interchangeable lens options from 50-16 millimetres. The resulting improvements in effective resolution are calculable (Figure 34) and improvements in image clarity on top of this are very likely. The best path to understanding what effective resolution we require will be to use a drone like the Matrice 200 or Inspire 2 that allows the use of high-resolution sensors and interchangeable lenses. By using these at varying altitudes, we can quantify the capacity to identify and count different species in different settings.

The Phantom 4 used at East Fairfax in 2016 would have had an effective resolution of 3.5 cm per pixel at 60m above ground, while the new Phantom 4 Pro would have 2.2 cm/pixel and an X7 camera on an Inspire 2 or Matrice 200 would have between 0.5 cm/pixel and 1.6 cm/pixel.

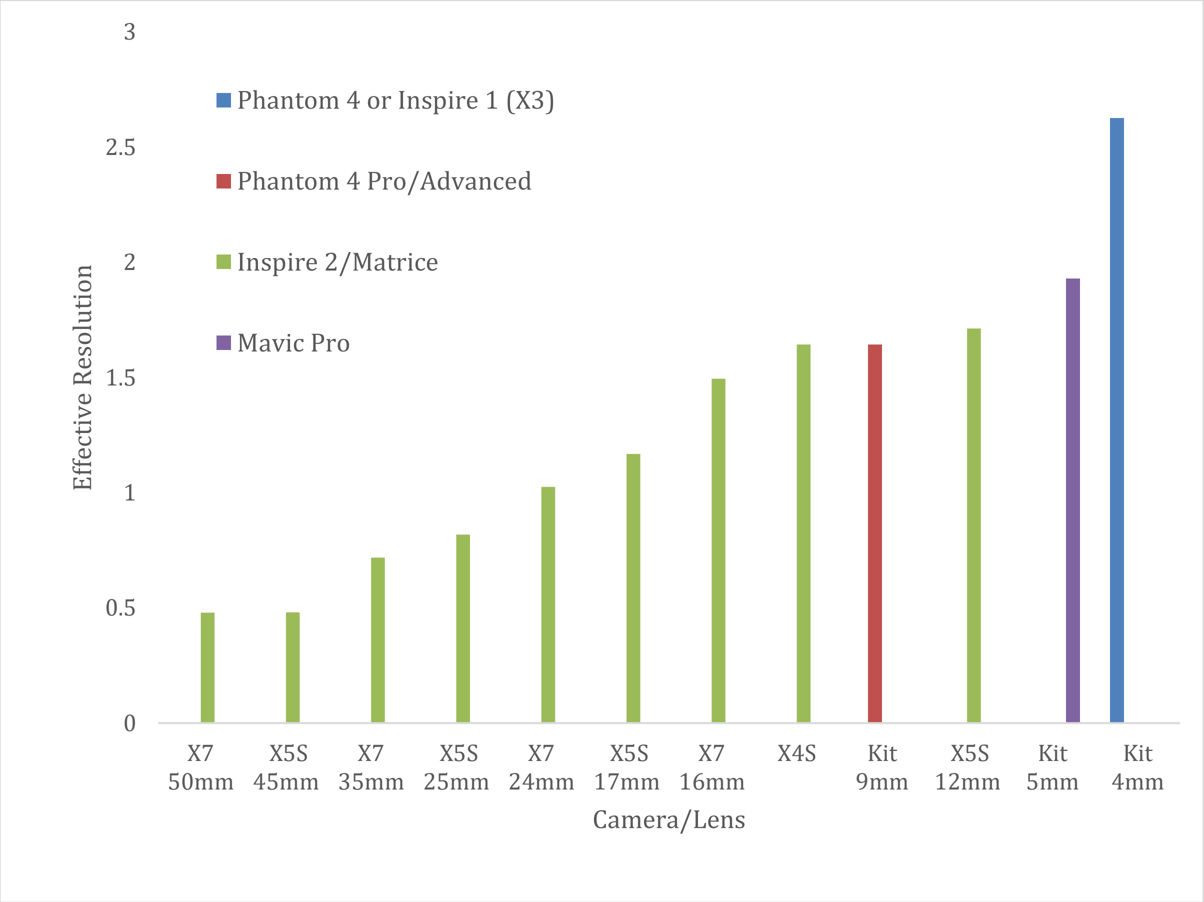

Figure 34: The effective resolution in cm per pixel at the sensor at 60 metres above ground with different drone and lens combinations.

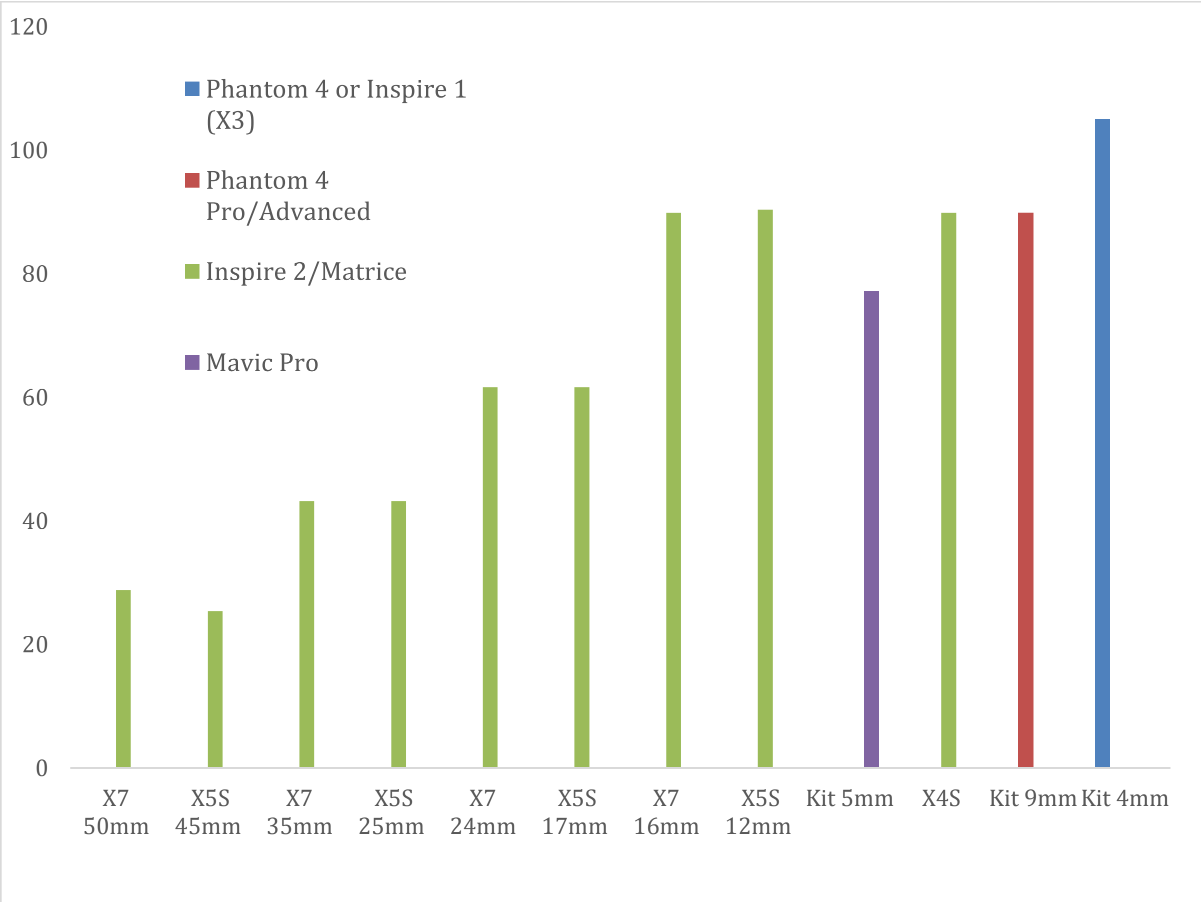

Figure 35: The swathe width in metres at 60 metres above ground with different drone and lens combinations.

Logistics

While fully autonomous drones that can operate over hundreds of kilometres (e.g. Boeing's Scan Eagle, https://insitu.com/information-delivery/unmanned-systems/scaneagle) can be hired, the costs are prohibitive – probably more than a conventionally piloted fixed wing or helicopter. The drones currently available to us are less than 10 kilograms in weight and must be flown within line of sight of the pilot. Because of the line of sight requirement drone deployments do not offer any significant saving in vessel or staff time to conduct surveys at remote locations. Furthermore, drones and particularly cheaper consumer drones, are more limited by weather than ground based observers. Drones generally can't be flown in winds above 15 or 20 knots and/or in rain. This is a significant liability because it restricts surveys to a narrower set of conditions than for ground observations.

A drone flight would likely take a similar, or slightly longer, amount of time as a ground count at a small open site with relatively few nesting birds. It is likely to be quicker than a ground count at larger sites such as Raine or East Fairfax Island and at sites, such as Michaelmas, where there are very large numbers of birds.

Analysis

We do not have an automated analytical software package or tool to analyse drone counts. Instead, a staff member reviews and analyses the imagery. This limitation means that even when flight times are quicker than ground counts there is a time commitment in the office. The combined field and office time is likely to be comparable to, or greater than for ground counts.

Future

It is likely that machine learning and neural and deep neural network software will provide avenues to both automate counts and enhance our capacity to detect nesting. Queensland University of Technology believe that automating the counting of seabirds would be achievable and will continue to get easier as the resolution in imagery improves. The development of species specific, or even habitat specific, detection algorithms in deep neural networks into tools that FMP staff can use will probably require financial investment.

It may be possible to overcome weather and logistic issues by adopting “drone in a box” type solutions. These are solar powered enclosures/boxes containing a drone, processing power, charging and communication tools, and weather sensors. This emerging technology can analyse the weather and, when conditions permit flight, deploy a multi-copter drone to perform a predefined flight over an area of interest. The drone then returns to its protective enclosure, and downloads the data which can then be transmitted back to a server by the enclosure. The data may be raw image data or may be counts of seabirds as calculated by the computer and software inside the enclosure. We are communicating with H3 dynamics (https://www.h3dynamics.com/products/drone-box/) who have developed functioning prototypes. Deploying these units would currently cost approximately $100,000 but it is likely that costs will come down in the next two to five years.

At present light drones would be most useful for improving the speed at which we can ground truth relationship between acoustic indices and nest density and establishing the effective resolution required for all future use of imagery. We should establish how wind and rain limits there deployment and what units permit operations in the most challenging conditions.

We now know that drones imagery allows more accurate adult bird counts but a reduced ability to discriminate between breeding and loafing birds than a ground based observer. However the weaknesses of both methods can be overcome by combining the techniques to generate a single “corrected” count that is both more accurate and more precise than either could offer alone. We will generate much more precise data on the numbers of nesting seabirds around sensors and for any other drone test sites. More precision means that we can establish the nature of the relationship between acoustic indices and actual numbers in less time and with fewer repetitions.

The drone that currently offers a robust weather proof platform and the capacity to carry a camera capable of testing the full range or effective resolutions we are interested in is the DJI Matrice 200 (http://www.dji.com/matrice-200-series).

Figure 36: Rangers, a contractor and a Phantom 4 at East Fairfax in 2016. G. Hemson.

Summary

Each survey method/technology has its own distinct benefits and disadvantages. All require consideration of the need to store significant amounts of digital media.

Cameras

Advantages

- Can capture data over long periods of time[27], although note disadvantage about unit failure
- Imagery easy to understand and interpret
- Can detect other events of interest such as trespass, flooding and weed incursion.
- A single unit can record useful imagery from approximately 50m radius[28]

Disadvantages

- Analyses are currently manual
- Data interpretation by different observers may be inconsistent, reducing precision an accuracy
- Can be difficult to reliably identify which birds are breeding and which are not
- Vegetation, debris, obstacles or other birds can occlude birds and nests
- Expensive per unit (+/- $10-20k)
- Trial deployments stopped working within a year

Acoustics

Advantages:

- Unbiased data
- Easy to deploy and maintain.
- Can capture data over long periods of time
- Largely unaffected by vegetation
- Analyses outsourced

Disadvantages

- Analyses outsourced and costly
- Must be placed within or a fixed or known distance from a colony. Potentially making it unsuitable for any species that nests in tight colonies at locations and timings that cannot be predicted in such a way that ensures sensors are deployed within them

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[27] If lens and charging input (e.g. solar panel) do not require cleaning.

[28] Dependent on topography and vegetation

Drones

Advantages

- Data analysis may be automated in the near future which would generate unbiased data
- Elevated perspective avoids topography and grasses occluding most birds
- Easy to generate accurate counts in colonies of easily visible species
- Drone in a box options may offer the potential to capture data over long periods of time

Disadvantages

- Staff must currently be present (although note “drone in a box” potential)
- Weather may restrict operations
- Currently difficult to discriminate breeding from loafing
- Difficult or impossible to detect birds under thick overhanging vegetation or structures

Future direction

When the site is expensive to access and the species is suitable for acoustic monitoring then that is our recommended option. The technology for recorders and analysis is already available and the probability that recorders and analysis will improve and reduce in costs in the future is high. To speed up the experimental testing of the correlations between acoustic indices and actual abundance for each species we recommend using ground counts augmented with counts from drone data. This approach provides optimal precision and accuracy enabling us to progress to an autonomous acoustic deployment more rapidly and confidently.

If the species or site is not suitable for acoustic monitoring and is relatively convenient to access then we recommend counts using drones and ground observers. This strategy should ensure that we are able to compare the ground counts with all historic data and provide a correction based upon the additional information provided from the drone. We anticipate that “drone in a box” options will become feasible in the next five years and this work will pave the way for their use in all sites for which acoustics are not suitable. As the cost of drones comes down, and their capacity to perform autonomously at range improves, we may be able to use them in place of ground counts.

If the site is not convenient AND is not suitable for acoustic surveys then we should evaluate “drone in a box” or fixed camera solutions against regular ground counts augmented with drone imagery.

To progress these recommendations we propose the following:

1. Expand and conclude the acoustic trials.
    a. Complete trials at East Fairfax and Raine.
    b. Explore the most resource[29] efficient options for automated analysis.
    c. Prioritise additional trials for locations which are costly to access and species for which we currently have no useful acoustic data.
2. Evaluation of drones at acoustic trial sites and some other sites selected because they are difficult to count from the ground and are not suitable for acoustic deployments. We will specifically try to work out whether drone imagery can be;
    a. generated at the resolution required to identify all seabirds,
    b. captured under the prevailing conditions, and
    c. analysed automatically.
3. Desktop exploration of the cost/value comparison of fixed cameras and “drone in a box” options for remote locations and species for which neither acoustics nor staff piloted drones are sufficient.

References

Driscoll, P. V. 2013, ‘Phase 1 Analysis of Coastal Bird Atlas Data’. A Report to GBRMPA

Hodgson, J. C., Baylis, S. M., Mott, R., Herrod. A., & Clarke. R. H 2016, ‘Precision wildlife monitoring using unmanned aerial vehicles’. *Scientific Reports,* Vol 6, 2257

Southwell, C & Emmerson, L 2015, 'Remotely-operating camera network expands Antarctic seabird observations of key breeding parameters for ecosystem monitoring and management', *Journal for Nature Conservation,* vol. 23, pp. 1-8.

[29] We use the term resource rather than cost to indicate that we would need to factor FMP FTE into any calculation.

Appendix E1.1

Literature Review

CONS6014

An analysis on the use of remote monitoring techniques to gather useful data on seabirds

Rebecca Richardson

Masters of Science

University of Queensland

This literature review was prepared as part of the Coastal Bird Program and the Remote Monitoring Project. The purpose of this review is to collate the available information on the use of remote or autonomous systems for gathering information on fauna populations and particularly nesting seabirds. The original material was provided by Rebecca Richardson and has been edited and augmented by Graham Hemson.

Introduction

The breeding biology and ecology of seabirds is relatively well known compared to many other taxonomic groups (Duffy 1992; Schreiber and Burger 2001). However they forage over enormous areas and generally breed in remote and therefore terrestrial predator free environments, making accurate monitoring of seabird population trends logistically challenging. A seabird research program by the Coastal Conservation Action Lab at the University of California, Santa Cruz identified that the monitoring of seabird populations is made difficult by three factors: 1) the cost of deploying and maintaining survey teams on remote islands, 2) the ability of teams to regularly arrange travel to remote sites, and 3) the disturbance that survey teams can cause while working in seabird colonies.

Croxall et al (2012) found that seabirds (particularly pelagic seabirds) are more threatened and that their conservation status is worsening more rapidly than other birds. Key threats include overfishing, bycatch, nest predation by invasive species and the influences of climate change (Croxall, 2012 and Cury et. al., 2011). The Great Barrier Reef is the world's largest reef system and supports approximately 1.5 million breeding seabirds of 22 species (Appendix 1). With more than 900 islands, monitoring seabirds is logistically challenging. A report commissioned by the Authority by Driscoll (2010) indicated negative population trends for several seabird species in the Great Barrier Reef. However this same report was also critical of the ability of the available data set to reliably indicate these trends and became the motivation for much of the subsequent work on coastal bird monitoring strategies and methods.

In 2012, QPWS used funds from the Authority to engage scientists at the University of Queensland to undertake a power analysis of data gathered from Michaelmas Cay, the longest monthly seabird breeding dataset in Queensland. The final report highlighted that the current frequency and regularity of site visits to gather seabird breeding data was insufficient to detect even quite large changes in seabird breeding populations. Instead the analysis indicated that at least two surveys during the breeding season of each bird species were required each year to ensure that the data could reliably discern trends from noise, a result similar to that determined by Johnson and Khron (2001). In addition the report recommended extending surveillance to lower priority sites to accommodate potential overspill from saturated sites or relocations from declining sites. Cognisant of the resource impost inherent in increasing monitoring effort, the report provided further endorsement of the Remote Monitoring Project by recommending that remote monitoring technologies were evaluated.

In order to understand seabird population trends long-term monitoring is important. Most seabirds are characterised by low reproductive rates, relative longevity, delayed maturity (three-nine years), and high adult survival rates, so that timescales for population processes are relatively long. This presents a challenge for conservation management as the lags inherent in looking for the effects of previous years on recruitment make active management difficult. As such other agencies have prioritized the detection of breeding success metrics (≈ population condition) as well as breeding effort (≈ population size) to provide more immediate relevance to environmental conditions (Mitchell and Parsons, 2007). The data from long-term seabird monitoring and correlations with environmental changes are increasingly important if we want to manage proactively and predict when and where seabirds are most likely to be threatened by climatic events such as booms or collapses of prey populations, El Nino events, cyclones and disease (Nisbet 1989, Devney et. al., 2009).

To date, all large scale monitoring of seabird populations has been conducted using human observers on the ground, in boats and in aircraft.

This review will examine camera traps, audio recorders, aerial photography using piloted small aircraft and unmanned aerial vehicles and satellite imagery as alternative methods to conducting human field surveys and observations to monitor seabird breeding colonies in remote areas.

Results

The majority of 'technique' papers reviewed were associated with the use of camera traps as a means of remote monitoring. There were very few papers discussing the use of remote monitoring techniques to monitor colonial seabirds across all of the 'technique' types, particularly for more recent studies.

There exists very little peer reviewed scientific literature on the use of satellite imagery or UAVs to estimate species populations, both of which are newly emerging technologies. Most of the literature associated with the satellite imagery technique was concentrated in the Antarctic, although the studies were relevant to colonial seabirds. All but one of the papers associated with UAV survey trials were conducted in the United States of America. The literature on aerial surveys via piloted small fixed wing aircraft probably had the most examples of the use of remote monitoring techniques using birds. However this was mainly associated with nest predation, and nesting behaviour, with many of the studies mostly waterfowl.

The literature associated with camera traps was primarily focused on the use of cameras traps for monitoring nest predation, animal behaviour and presence / absence of species, particular, rare or secretive species. The main taxonomic groups studied were birds (mostly non colonial species) and mammals (small to large carnivores). A number of technical reports were found for the use of camera traps that were more specific to the topic and more recently undertaken: Cunha et al (2008) 'Development and Methods for Monitoring Seabirds on Castle Rock NWR; Lorentzen et al (2010) 'Estimating chick survival in cliff-nesting seabirds – a hazard made easy with monitoring cameras, SEAPOP; Dickinson et al (2008) 'Autonomous monitoring of cliff-nesting seabirds using computer vision'. Further details on some of the key findings from the literature are summarized below in section 3.1.

Types of Remote Monitoring

Camera Traps

A review of the literature on the use of remote photography in wildlife ecology by Cutler and Swann (1999) found that the most common use for cameras was nest predation, followed by feeding ecology and nesting behaviour. The high proportion of papers associated with nest predation and nest behaviour found through this review continue to support Cutler and Swann's (1999) findings. Few examples exist on the use of camera traps to monitor seabird populations. There also appeared to be a lack of more recent papers on the use of remote cameras to monitor species population trends.

As technology has advanced more recent papers (i.e. Locke et al 2005) have recognised the need for self-sustaining cameras in remote locations that can operate in near real-time. Probably the most relevant and recent study is by Newbery and Southwell (2009) who designed a monitoring technique using a camera system for automated recording of digital images at remote sites in polar environments. The design places emphasis on low maintenance, low environmental impact, autonomous operation, ability to withstand high winds and low temperatures with very low electrical power requirements.

The key advantages of data captured from camera traps are that the information can be reviewed by other researchers. Since each photograph includes the exact time it was taken, camera traps collect detailed data on the activity patterns of many species (van Schaik & Griffiths, 1996; Gómez et al., 2005; Azlan & Sharma, 2006)

There are numerous options for designing and customizing camera traps to meet monitoring objectives. Differences in field conditions and target species will influence the choice of camera traps and set up. There are no set guidelines available on which equipment and methods are best for certain applications; researchers must often learn by an expensive trial and error process (Cutler & Swann 1999). As each of the approaches and subject of the study vary, so to do the problems associated use of the camera traps. Cutler and Swann (1999) suggest that the still camera is not feasible in applications in which the subject makes frequent movements that would trigger cameras or for applications in which detailed information on the behaviour is needed. Bolton et al (2007) found that cameras did not operate effectively at high light levels for example at nests in open situations and exposed to full sunlight. Light intensity was reduced by attaching an infrared filter to the front of the lens at nests in exposed locations.

Kawakami (2002) acknowledged power supply as the biggest obstacle for field applications with camera traps and suggested using a generator high capacity battery. Lock et al (2005) overcame power supply issues by using two deep cycle marine batteries that were continuously recharged via solar panels, making them essentially self-sufficient. However, Locke et al (2005) did identify that cost was a disadvantage of this system, although with advances in technology this has most probably declined over time. Margalida et al (2006) used a wind powered battery charger with an adaptor volt regulator at one of the monitoring sites and found this to be suitable alternative for solar power in their study.

Bolton et al (2007), King et al (2001), and Thompson et al (1999) found that cameras did not influence behaviours of parent nesting birds. However, this is in contrast to Pietz & Granfers (2000) and Cain (1985) who found substantial abandonment of monitored nests in response to camera deployment.

Trathan (2004) found that total counts of penguin species were impractical for large colonies, and reliable methods for subsampling were needed. However, subsampling can cause biases, particularly in complex landscapes where high relief may block the observer's view or where parts of the colony can be assessed only from a poor vantage point. In such situations poor estimates of average nest density potentially compromise the reliability of the count

TEAM network (2008) recommended that if surveys are repeated over years for monitoring species diversity, the same camera trap sites should be used every year, and that camera traps should be run for approximately the same number of days every year to achieve a comparable sampling effort.

Acoustic Monitoring

Of the reviewed literature on acoustic monitoring the majority of studies used this method for assessments of species richness or occurrence patterns, or counts of birds that index abundance (Dawson & Efford 2009). There are limited studies on the use of this technique in remote areas for the long-term monitoring of seabird populations. However, a few studies have suggested that nocturnal burrow nesting seabirds such as shearwaters would make good candidates for acoustic monitoring since many species vocalize at night when they return to their colony (Brooke 2004; Bardeli et al 2010; Buxton & Jones 2012).

The investigations that have compared acoustic recordings and the traditional survey means of point counts suggest that acoustic recordings can perform as well as or better than point counts for species richness in land-bird dominated systems (Haselmayer and Quinn 2000, Hobson et al 2002, Acevedo and Villanueva-Rivera 2006, Celis-Murillo et al. 2009 Celis-Murillo 2012). A recent comparative study by Celis-Murillo (2012) supports this finding and suggests that detection probabilities were higher for acoustic recordings than point counts and that acoustic recordings can perform just as well across a variety of tropical vegetation types including in vegetation ranging from forests to pastures. This is in contrast to findings by Hutto and Stutzman (2009) who suggest that significantly more bird species were detected using point counts than acoustic recordings. Bardeli et al (2010) identified two aspects essential to successfully carrying out acoustic monitoring. These are: 1) there needs to be pattern recognition algorithms for the automatic detection 2) appropriate techniques for the estimation of the number of individuals.

The ability to reliably detect acoustic events depends largely on the structure of the recording environment, the signal-to noise ratio (SNR) and the complexity and variability of the signals to be detected and that the minimum number of recording nights will depend on call type and activity level (Blumestein et al 2011; Buxton et al 2012; Bardeli et al 2010). The study by Bardeli et al

(2010) found that in the best case, calls of the bittern could be recorded over distances of about 1 kilometre with a complexity level somewhere between the simple situation of a laboratory recording and the extremely complex situation of bird choirs at daytime. Whereas Buxton & Jones (2012) found that under ideal conditions they could detect calls up to 50 metres away. Under this study the call recognition software identified over 50 per cent of the calls of target species. Recognition models were most useful at sites with moderate levels of call activity. However, at locations with numerous overlapping calls of different species the information was unusable as the software was unable to discern individual calls. At these sites, the models identified <10 per cent of calls. Bardeli et al (2010) found that wind plays a crucial role in the reliability of pattern recognition algorithms and that recordings were clearer at sites where there was more wind shelter.

Bardeli et al (2010) found one of the main problems with wind is the high number of false positives it produces. Blumstein et al (2011) recommends that systems should be designed to err on the side of false positive, as the cost of false negatives is that the sound of interest was not recorded and cannot be examined.

The devices can be programmed to record vocal activity over specified time periods and they can simultaneously record at multiple sites over entire seasons, facilitating spatial and temporal comparisons of activity. The recording data collected can then be processed in batches using automatic signal detection software to identify composition or quantify calling rates (Goh 2011; Peterson and Dorcas 1994 cited in Buxton 2012). There have been logistical advancements associated with using recording devices. Battery operated recording devices are commercially available and have been used to monitor a diversity of species (Dorcas et al 2010, Thompson et al 2010) and have the capacity to store up to 100GB of data which means they can easily be deployed for months in remote locations to automatically record bird calls.

Buxton & Jones (2012) conducted a study over a period of 2 months using a 32 GB memory card and predicted battery life (high capacity 12,000mAh batteries) of 100 h allowed for 32 nights at three h/night of recording. At this rate with the device was programmed to record in 15 minute on/off cycles from dusk to dawn batteries needing to be changed once every 32 days.

Another advantage of this method is that a permanent digital record is made available for the scrutiny of several experts and if these recordings were archived through time, the same interpreter can be used to evaluate population trends. Thus, it should be possible to control the confounding factors of inter-observer bias, changes in observer, or observer ability over time (Cyr 1981) for both short and long-term studies (Hobson et al 2002).

Further work is needed to determine if call activity can be used to infer the size of seabird populations, to relate call rates to independent measures of seabird numbers (Haselmayer and Quinn 2000) and to determine what proportion of birds are detectable (Dawson and Efford 2009).

Aerial Surveys Piloted Fixed Wing Aircraft

The amount of variability in aerial estimates using observers for individual colonies suggests that aerial surveys are most useful for locating and determining qualitative properties of a colony (e.g., size categories such as small, medium, or large) (Tracey et al 2005). The accuracy of aerial surveys varies with observer, colony size, species composition and canopy cover, especially with slow rapid flights (Rodgers et al 2005). The accuracy of counts reduces rapidly as the number and density of birds to be counted increases (Arbib 1972, Samuel & Pollock, 1981; Prach & Smith 1992). Not all birds that are present in an area are detected, and the numbers recorded are only estimates of actual numbers seen (Prach & Smith 1992).

Bajzak & Piatt (1990) found that visual counts of aerial photographs can vary by up to 100 per cent between different observers, depending on their counting technique and those repeated by a single observer can vary up to 40 per cent. Gibbs et al (1988) found that although aerial observers consistently underestimated the number of nests at colony sites (aerial-visual estimates averaged 87per cent of ground count; aerial photographs averaged 83 per cent of ground counts) the precision of counts was high. This high precision allowed their use in combination with conversion factors to predict colony size.

More precise counts can be obtained if photographs are digitized and bird images are quantified on the basis of photographic density. Manually counting individual birds from highly aerial photography is time-consuming and a major disadvantage to aerial survey work. However, with improved colour photography and the advent of high-resolution digital scanners, automated counting methods based on computerized image-analysis techniques are now feasible (Trathan 2004).

The computer aided counting technique used for aerial photographs by Bajzak et al (1990) suggested that the method could be used to sort counted birds into size and tonal classes and has the potential for use in counting different species and sex or age classes of birds in mixed waterfowl assemblages.

One of the primary problems associated with aerial surveys is noise levels that will likely be disruptive to the wildlife. In addition manned aircraft is totally dependent on weather and humans, which usually overrides the temporal and spatial considerations on population distributions.

Aerial Surveys Unmanned Aerial Vehicles (UAVs)

The literature suggests that the use of UAVs for wildlife purposes does occur internationally, although this is still at an early stage of development. Within Australia, there has been limited application of UAVs in wildlife research. The most recent study undertaken my Hodgson et al (2010) investigated the use of UAVs to monitor marine mammals. No examples were found on the use of UAVs to monitor remote seabird populations in Australia.

Different research applications will call for different UAV systems according to the sensors required (autonomous control, video, still photos, radio telemetry, etc.) and the distance required to reach or cover the area of interest (Jones et al 2006). Each UAV system can only accommodate a limited set of research applications, classified primarily by their range, time aloft, and image resolution requirements (Jones et al 2006). A UAV useful to wildlife research is not one size fits all. As UAV systems increase in size and complexity, they become more expensive and difficult to operate. The larger the range (and time aloft), the larger the UAV needs to be. Appendix 2 summarises the widely accepted classification and characteristics of various UAV systems.

Jones et al (2006) states that UAVs have promise as a scientific monitoring tool, but only when combined with appropriate sensors and with established sampling protocols and statistical analysis. The UAV offers enhanced capabilities in dealing with habitat/population relationships on small scales, since it has spatial/temporal capabilities that traditional aircraft and ground research simply does not (Jones et al 2006).

Remotely sensed imagery acquired from unmanned aerial vehicles (UAVs) allows for image capture close to the ground surface, and thereby provides superior spatial resolution, which may be necessary to detect individual species.

The use of small UAVs to monitor wildlife may address some of the problems that are often associated with piloted aerial surveys, including safety, cost, statistical integrity and logistics (Jones et al 2006, Wong et al 1997). UAVs can be mobilized rapidly and often to meet monitoring requirements, while collection of remotely sensed imagery from airplanes and sensors requires considerable mobilization time and expense (Balbach et al 2007). One of the key advantage of using the UAV rather than conducting manned flights is that the UAV system produces precise records of the flight parameters, allowing the plane to fly a specific programmed path.

Knowing the field of view and angle of the camera,
together with the exact altitude, pitch, roll, heading and GPS track provides the opportunity to

determine the exact area surveyed and therefore the proportion of the survey area sampled. This then allows a more accurate population estimate to be calculated (Hodgson et al 2010).

Hodgson et al (2010) found three key limitations to the use of UAVs for surveys, these are:

(1) the quality and reliability of the imagery to identify animals, particularly small marine mammals (it was recommended that high definition video be used). This was supported by the study conducted by Balbach et al (2007) which found that in most cases, aerial photography and satellite imagery does not provide adequate spatial resolution to determine the presence or number of individual threatened species.

(2) stabilisation of the imaging platforms is needed to produce high quality images and reliably survey the area of interest, but to date most stabilising systems are prohibitively heavy and/or expensive, and (3) permitting requirements include collision avoidance methods such as autonomous sense and avoid systems, however these methods are still in the research and development stage.

Satellite imagery

Although numerous scientific papers discuss the use of satellite imagery for monitoring landscape scale and vegetation changes, the literature available on the use of satellite imagery to determine breeding populations of seabirds is limited. In the past satellite imagery has been used most often for the evaluation of species habitat, predicting species distributions and detecting landscape level change (Kerr et al 2003; Reeves et al 1976; Sidle et al 2002; Conner 2002). This technique of information gathering has been considered as a method to estimate the size of colonies of seabirds (Schwaller et al 1989; Guinet et al. 1995; Woehler & Riddle 1998). A search of the literature only found a small number of papers that discussed this method as a possible approach to monitoring breeding populations of colonial seabirds. All of these papers refer to the study of penguins in the remote arctic environment. The most recent studies by Fretwell et al (2012) and Lynch et al (2012) have presented the use of satellite remote sensing to survey an entire population of penguins and estimate penguin abundance. Earlier studies were undertaken by Barber-Mayer et al (2007) and Guinet et al (1995).

Both Fretwell et al (2012) and Guinet et al (1995) found that penguin species, breeding in large colonies in relatively flat areas with individuals regularly spaced, are suitable for conducting this kind of investigation. It was suggested that this was due to the high level of contrast with the penguins and the surrounding environment (sea ice) making them easier to count in remote sensing imagery. Lynch et al (2012) has found that this is not the case with smaller penguin species that breed on topographically complex terrain composed of mixed substrates.

Fretwell et al (2012) Suggested that image quality and cloud cover may make identification from a nominal resolution ~10 metre difficult. Smaller colonies of less than 200 individuals were also found to be more difficult to identify using imagery at this resolution (Fretwell et al 2012; Lynch et al 2012). The count study by Fretwell (2012) only included adult birds at the breeding site and did not include chicks or non-breeding adults not present at the breeding colony.

In the majority of cases penguins grouped into close clusters and their shadows overlapped, meaning that individuals cannot be differentiated and a different approach is needed. Differentiating between penguins, shadows and guano was a limiting factor causing large errors between actual and estimated counts and almost certainly resulted in an over estimation of the population. This is similar to the findings of Barber-Mayer et al (2007) who used satellite imagery to differentiate between relatively small (<3,000 adult birds) and larger colonies (>5,000 adult birds), but also found that their analysis was hampered by excessive guano and shadows. Lynch et al (2012) the visibility of breeding populations depends sensitively on matching the timing of imagery to the phenology of the target species. Target visibility may peak before or after peak abundance depending on whether the goal is to identify individual animals, which often appear more sharply against snow earlier in spring or early summer, or to identify colonies, which are highlighted later in the breeding season by the accumulation of guano at the site.

While the spot scene appears to be a useful tool in monitoring medium to long-term changes in colony size, the use of this technique in monitoring inter-annual fluctuations in breeding population size needs on site validation, but it appears to be doubtful (Guinet 1995). Fretwell et al (2012) and Lynch et al (2012) both recognise the value of this tool for its capability to discover new colonies.

Discussion

In general, a lack of literature exists on the use of the various remote monitoring ‘techniques’ for gathering data on seabird populations. Of the techniques discussed camera trapping appears to be the most feasible in the present environment. The literature suggests that the use of UAV and satellite remote sensing technology do not appear to be as well advanced in their application and further studies are needed before either of these techniques could be applied with some confidence. The study by Fretwell et al (2012) and Lynch et al (2012) with satellite remote sensing did find that the technique was beneficial in detecting new colonies or confirming previously suspected colonies. Wong et al (1998) predicted that the market for civilian UAV applications will grow significantly. However, the literature suggests that no significant progress has been made in Australia, on the use of UAV’s for the purposes of wildlife management. Although, this may not be the case for other civilian UAV applications. For shorter distances smaller UAVs could be trialled, however the greater the distance to travel the more complex and costly the system becomes

As vocalisations are one of the primary means of communication for birds it makes them very amenable to acoustic monitoring. However up to this point, there has not been conclusive evidence on whether acoustic monitoring could provide a reliable method of obtaining numbers of breeding pairs of seabirds in an area. Therefore before further development of the monitoring system can proceed, several important issues need to be considered.

The UCSC Coastal Conservation Action Lab and computer scientists at Lorax Analytical are currently developing an easy-to-use, low-cost automated acoustic sensors for monitoring changes in seabird populations breeding on islands helping to expand the geographic and temporal scale of seabird research in remote locations (http://ccal.ucsc.edu/wam/index.html). The results of this study may be worth further investigation.

For seabirds, on any one island, there is usually a mix of breeding species and based on data by King (1993) in the Great Barrier Reef this can range from one breeding pair to up to eleven breeding pairs of species on one island. Although software designed to detect and recognize species vocalizations autonomously have been used successfully in other acoustic monitoring studies (Figueroa and Robbins 2008, Trifa et al. 2008), the ability to reliably detect acoustic events depends largely on the complexity and variability of the signals to be detected and this should be taken into consideration, particularly on islands were multiple species breed.

Remote photography can be less consuming, costly and invasive than traditional research methods for many applications. However researchers should be prepared to invest time and money troubleshooting problems with remote camera equipment, be aware of potential effects of equipment on animal behaviour, and recognize the limitations of data collected with remote photography equipment (Cutler & Swan 1999)

A recommendation for researchers proposing monitoring trials in remote areas would be to conduct trials with the units set in an easily accessible location beforehand or simultaneously. This may help to identify and resolve any issues prior to revisiting remote location trail sites, potentially preventing multiple visits to remote locations during trials and reducing loss of important data later.

The size of many seabird colonies, and sometimes the geographical spread of species breeding in small, scattered groups, often makes counting or monitoring all birds or breeding attempts impractical. The usual approach to this problem is to study a sample of the colony or colonies, and to use results from this sample to infer, for example, how the population of a species in the colony as a whole is changing, or how successfully birds are breeding in the colony. Estimates of numbers in sample areas can also be extrapolated to provide an estimate of numbers for the whole colony, provided good information on colony area is available. Therefore, models that explicitly include detection probability must be used when analysing changes in diversity over time and space. Royle & Dorazio (2008) propose a hierarchical multi-species site- occupancy model to analyse temporal changes in community composition. Application of these models to analyse camera trap data is under development (T. O'Brien, personal communication), and they have great potential for data from multiple sites or multiple years

Conclusion

A recent study of seabird conservation status by Croxall et al (2012) found that overall, seabirds are more threatened than other comparable groups of birds and that their status has deteriorated faster over recent decades. Island birds also dominate the list of extinct taxa in Australasia. Of Australia's threatened or extinct bird taxa a total of 77 out of 132 (58 per cent) are island birds (Kirkwood & O'Connor 2010). Unfortunately it is unclear whether the substantial population declines observed for the many species of tropical seabirds breeding on the Great Barrier Reef have continued and recent and systematic data for most important seabird breeding colonies in the region are limited (Kirkwood & O'Connor 2010). The main research actions recommended by Croxall et al (2012) to understand the trends in seabird populations was to undertake more and better coordinated monitoring in order to permit evaluation of population size and trends for as many species as possible, particularly those already in adverse conservation status.

Taking into consideration that many of our seabirds occur on remote islands, that there are challenges associated with their monitoring and that there is a lack of long-term data on population; seabirds lend themselves well to trialling new techniques in data collection and analysis. Trials would benefit seabirds and species occurring in remote areas more broadly. In Australia seabirds choose a range of different breeding sights, but these can generally be grouped into three broad classes of nesting habitat: 1)burrowing, 2)surface nesting, (however, the proportion of seabirds that are surface nesters decreases with increasing latitude) and 3)tree nesting. The fact that one island can have seabirds that display two or more breeding habitats means that an appropriate monitoring strategy needs to apply.

The challenges associated with monitoring in remote areas have been recognised in the scientific literature. However, there appears to be a lack of recent studies on attempting to overcome these issues, particularly associated to gathering data on seabird populations. Rapid advances in technology render many of the components in the existing literature obsolete, particularly in relation to the logistical issues (power supply, image resolution, data storage capacity) associated with applying techniques such as camera traps. This, in addition to a lack of recent scientific literature makes it difficult to build solid conclusions about the logistical aspects of each technique. One approach to ensure we can learn from current research experiences in this field may be to establish a small specialised international network to help facilitate the exchange of information on the various approaches. This is particularly important for seabirds where a number of research programs have recently or are currently being undertaken, but where recent peer-reviewed scientific literature is lacking.

Some of the impediments to monitoring programs are overcome when monitoring becomes a required activity, such as fulfilling legal requirements. However, this may also be achieved by reducing some of the factors that impede the commitment to long term monitoring for species conservation, particularly in remote areas (cost, design, operational and logistical requirements of data collection, data application).

Reference

Acevedo, MA, and Villanueva-Rivera, LJ 2006, Using automated digital recording systems as effective tools for the monitoring of birds and amphibians, *Wildlife Society Bulletin*, 34, 211–214.

Balbach, H, Pitts, H, Meyer, W & Tweddale, S 2007, *Threatened and endangered species surveillance in inaccessible areas: A feasibility study*, Final Report prepared for US Army Corps of Engineers, Washington DC, USA.

Bajzak, D & Piatt, JF 1990, Computer-aided procedure for counting waterfowl on aerial photographs, *Wildlife Society Bulletin*, 18(2), 125-129.

Barber-Meyer, S, Kooyman, GL & Ponganis, PJ 2007, Estimating the relative abundance of emperor penguins at inaccessible colonies using satellite imagery, *Polar Biology*, 30, 1565-1570.

Bardeli, R, Wolff, D, Kurth, F, Koch, M, Tauchert, K-H & Frommolt, K-H 2010, Detecting bird sounds in a complex acoustic environment and application to bioacoustic monitoring, *Pattern Recognition Letters*, 31, 1524-1534.

Beever, E 2006, Monitoring biological diversity: strategies, tools, limitations, and challenges, *Northwestern Naturalist*, 87(1), 66-79.

Blumstein, DT, Mennill, DJ, Clemins, P, Girod, L, Yao, K, Patricelli, G, Deppe, JL, Krakauer, AH, Clark, C, Cortopassi, KA, Hanser, SF, McCowan, B, Ali, AM & Kirschel, ANG 2011, Acoustic monitoring in terrestrial environments using microphone arrays: applications, technological considerations and prospectus, *Journal of Applied Ecology*, 48, 758-767.

Brook, M 2004, *Albatrosses and petrels across the world*, Oxford University Press, USA.

Buxton, RT & Jones, IL 2012, Measuring nocturnal seabird activity and status using acoustic recording devices: applications for island restoration, *Journal of Field Ornithology*, 83(1), 47-60.

Celis-Murillo, A, Deppe, JL & Allen, MF 2009, Using soundscape recordings to estimate bird species abundance, richness and composition, *Journal of Field Ornithology*, 80(1), 64-78.

Celis-Murillo, A, Deppe, JL & Ward, MP 2012, Effectiveness and utility of acoustic recordings for surveying tropical birds, *Journal of Field Ornithology*, 80(2), 166-179.

Conner, LM 2002, A technique to locate isolated populations using satellite imagery, *Wildlife Society Bulletin*, 30(4), 1044-1049.

Croxall, JP, Butchart, SHM, Lascelles, B, Stattersfield, AJ, Sullivan, B, Symes, A & Taylor, P 2012, Seabird conservation status, threats and priority actions: a global assessment, *Bird Conservation International*, 22(1), 1-34.

Cutler, TL, & Swann, DE 1999, Using remote photography in wildlife ecology: a review, *Wildlife Society Bulletin*, 27(3), 571-581.

Cyr, A 1981, 'Limitation and variability in hearing ability in censusing birds', In: *Estimating numbers of terrestrial birds* (Ralph & Scott eds), Cooper Ornithological Society, Studies in Avian Biology No. 6.

Dawson, DK & Efford, MG 2009, Bird population density estimated from acoustic signals, *Journal of Applied Ecology*, 46, 1201-1209.

Dickinson, P, Freeman, R, Patrick, S & Lawson, S 2008, 'Autonomous monitoring of cliff-nesting seabirds using computer vision', in: *International Workshop on Distributed Sensing and Collective Intelligence in Biodiversity Monitoring*, Amsterdam.

Dixon, PM, Olsen, AR, Kahn, BM 1998, Measuring trends in ecological resources, *Ecological Applications*, 8(2), 225-227.

Driscoll, PV 2010, *Phase 1 Analysis of Coastal Bird Atlas data*, Great Barrier Reef Marine Park Authority, Townsville, Australia.

Duffy, DC 1992, Biodiversity and research on seabirds, *Coastal Waterbirds*, 15(1), 155-158.

Field, S, Tyre, AJ, Possingham, HP 2005, Optimizing allocation of monitoring effort under economic and observational constraints, *The Journal of Wildlife Management*, 69(2), 473-482.

Fretwell, PT, LaRue, MA, Morin, P, Kooyman, GL, Wienecke, B, Ratcliffe, N, Fox, AJ, Fleming, AH, Porter, C & Trathan, PN 2012, An emperor penguin population estimate: the first global, synoptic survey of a species from space, *PLoSONE*, 7(4), e33751.

Gibbs, JP, Woodward, S, Hunter, ML & Hutchinson, AE 1988, Comparison of techniques for censusing great blue heron nests, *Journal of Field Ornithology*, 58(2), 130-134.

Gibbs, JP, Droege, S, Eagle, P 1998, Monitoring populations of plants and animals, *Conservation Biology*, 48(11), 935-940.

Gibbs, JP, Snell, HL, Causton, CE 1999, Effective monitoring for adaptive wildlife management: lessons from the Galapagos Islands, *Journal of Wildlife Management*, 63(4), 1055-1065.

Goh, M 2011, ‘Developing an automated acoustic monitoring system to estimate abundance of Cory’s Shearwaters in the Azores’, PhD Thesis, Imperial College, London.

Goldsmith, FB (ed) 1991, *Monitoring for conservation and ecology*, Chapman and Hall, London, United Kingdom.

Green, RE, Balmford, A, Crane, PR, Mace, GM, Reynolds, JD & Turner, RK 2005, A framework for improved monitoring of biodiversity: responses to the World Summit of Sustainable Development, *Conservation Biology*, 19(1), 56-65.

Guinet, C, Jouventin, P & Malacamp, J 1995, Satellite remote sensing in monitoring change of seabirds: use of Spot image in king penguin population increase at Ile aux Cochons, Crozet Archipelago, *Polar Biology*, 15, 511-515.

Haselmayer, J & Quinn, JS 2000, A comparison of point counts and sound recordings as bird survey methods in Amazonian Southeast Peru, *The Condor*, 102(4), 887-893.

Hobson, KA, Rempel, RS, Greenwood, H, Turnball, B & Van Wilgenburg, SL 2002, Acoustic surveys of birds using electronic recordings: new potential from an omnidirectional microphone system, *Wildlife Society Bulletin*, 30, 709-720.

Holling, CS (ed.) 1978, *Adaptive environmental assessment and management*, John Wiley & Sons, New York, New York, USA.

Hodgson, A, Noad, M, Marsh, H, Lanyon, J & Kniest, E 2010, Using unmanned aerial vehicles for surveys of marine mammals in Australia: test of concept, Final Report to the Australian Marine Mammal Centre

Hutto, RL & Stutzman RJ 2009, Humans versus autonomous recording units: a comparison of point-count results, *Journal of Field Ornithology*, 80, 387-398.

Jones, GP, Pearlstine, LG, Percival, HF 2006, ‘An assessment of small unmanned aerial vehicles for wildlife research’, *Wildlife Society Bulletin*, vol. 34, no. 3, pp. 750-758.

Johnson, CM and Krohn, WB. 2001. Monitoring nesting seabirds: Reply to commentary by Ian C.T. Nisbet. *The International Journal of Waterbird Biology*, 24(3), 461 – 466.

Jones, GP, Pearlstine, LG, Percival, HF 2006, An assessment of small unmanned aerial vehicles for wildlife research, *Wildlife Society Bulletin*, 34(3), 750-758.

Kerr, JT & Ostrovsky, M 2003, From space to species: ecological applications for remote sensing, *Trends in Ecology and Evolution*, 18(6), 299-305.

King, BR 1993, The status of Queensland seabirds, *Corella*, 17, 65-92.

Kirkwood, J & O'Connor, J (compilers) 2010, The State of Australia's Birds 2010: islands and birds, *Wingspan*, vol. 20, no. 4, supplement.

Legg, CJ & Nagy, L 2006, Why most conservation monitoring is, but need not be, a waste of time, *Journal of Environmental Management*, 78, 194-199

Lindenmayer, DB & Burgman, M 2005, *Practical Conservation Biology*, CSIRO Publishing, Collingwood, Victoria, Australia.

Lindenmayer, DB & Likens, GE 2010, *Effective ecological monitoring*, CSIRO Publishing, Collingwood, Victoria, Australia.

Lynch, HJ, White, R, Black, AD & Naveen, R 2012, Detection, differentiation, and abundance estimation of penguin species by high-resolution satellite imagery, *Polar Biology*, 35, 963-968.

Locke, SL, Cline MD, Wetzel, DL, Pittman, MT, Brewer, CE, Harveson, LA 2005, From the field: A web-based digital camera for monitoring remote wildlife, *Wildlife Society Bulletin*, 33(2), 761-765.

McDonald-Madden, E, Baxter, PWJ, Fuller, RA, Martin, TG, Game, ET, Montambault, J, Possingham, HP 2010, Monitoring does not always count, *Trends in Ecology and Evolution*, 25(10), 547-550.

Newbery, KB & Southwell C 2009, An automated camera system for remote monitoring in polar environments, *Cold Regions Science and Technology*, 55, 47-51.

Nichols, JD & Williams, BK 2006, Monitoring for conservation, *Trends in Ecology and Evolution*, 21(12), 668-673.

Nisbet, ICT 1989, Long-term ecological studies of seabirds, *Journal of Colonial Waterbird Society*, 12(2), 143 – 230.

Noss, RF 1990, Indicators for monitoring biodiversity: a hierarchical approach, *Conservation Biology*, 4(4), 355-364.

Peterson, CR & Dorcas, ME 1994, ‘Automated data acquisition’. In: *Measuring and monitoring biological diversity: standard methods for amphibians* (Heyer et al eds.) Smithsonian Institution Press, Washington DC.

Pollock, KH, Nichols, JD, Simons, TR, Farnsworth, GL, Bailey, LL, Sauer, JR 2002, Large scale wildlife monitoring studies: statistical methods for design and analysis, *Environmetrics*, 13(2), 105-109.

Prach, RW & Smith, AR 1992, Breeding distribution and numbers of black guillemots in Jones Sound, N.W.T, *Arctic*, 45(3), 111-114.

Reeves, HM, Cooch, FG & Munro, RE 1976, Monitoring arctic habitat and goose production by satellite imagery, *The Journal of Wildlife Management*, 40(3), 532-541.

Ringold, PL, Alegria, J, Czaplewski, RL, Mulder, BS, Tolle, T, Burnett, K 1996, Adaptive monitoring design for ecosystem management, *Ecological Applications*, 6(3), 745 -747.

Rodgers, JA, Kubilis, PS & Nesbitt, SA 2005, ‘Accuracy of aerial surveys of waterbird colonies’, *Waterbirds: The International Journal of Waterbird Biology*, 28(2), 230-237.

Schreiber, EA, & Burger, J 2001, *Biology of Marine Birds*, CRC Press, Boca Raton, Florida.

Schwaller, MR, Olson, CE, Ma, Z, Zhu, Z & Dahmer, P 1989, Remote sensing analysis of Adelie penguin rookeries, *International Journal of Remote Sensing*, 28, 199-206.

Sidle, JG, Johnson, DH, Euliss, BR & Tooze, M 2002, Monitoring black-tailed prairie dog colonies with high-resolution satellite imagery, *Wildlife Society Bulletin*, 30(2), 405-411.

Smyth, AK, & James, CD 2004, Characteristics of Australia’s range- lands and key design issues for monitoring biodiversity, *Austral Ecology*, 29(1), 3-15.

Teder, T, Moora, M, Roosaluste, E, Zobel, K, Pärtel, M, Kõljalg, U, Zobel, M 2007, Monitoring of Biological Diversity: A Common-Ground Approach, *Conservation Biology*, 21(2), 313-317.

Tracey, JP, Fleming, PJS & Melville, GJ 2005, Does variable probability of detection compromise the use of indices in aerial surveys of medium-sized mammals?, *Wildlife Research*, 32, 245-252.

Trathan, PN. 2004. Image analysis of color aerial photography to estimate penguin population size, *Wildlife Society Bulletin*, 32(2), 332-343.

Wong, KC, Bil, C, Gordon, D, Gibbens, PW 1997, *Study of the Unmanned Aerial Vehicle (UAV) Market in Australia*, Final Draft, Aerospace Technology Forum Report.

Yoccoz, NG, Nichols, D, Boulinier, T 2001, Monitoring biological diversity in space and time, *Trends in Ecology and Evolution*, 16(8), 446-453.

Woehler, EJ & Riddle, MJ 1998, Spatial relationships of Adelie penguin colonies: implications for assessing population changes from remote sensing, *Antarctic Science*, 10, 449-454.

Appendix 1.

Table 1. Key breeding seabirds in the Great Barrier Reef. Source: Driscoll (2010).

Common name	**Scientific name**
Red-tailed Tropicbird	*Phaethon rubricauda*
Wedge-tailed Shearwater	*Ardenna pacifica*
Herald Petrel	*Pterodroma heraldica*
Lesser Frigatebird	*Fregata ariel*
Great Frigatebird	*Fregata minor*
Masked Booby	*Sula dactylatra*
Red-footed Booby	*Sula sula*
Brown Booby	*Sula leucogaster*

Common Noddy	*Anous stolidus*
Black Noddy	*Anous minutus*
Bridled Tern	*Onychoprion anaethetus*
Sooty Tern	*Onychoprion fuscata*
Little Tern	*Sternula albifrons*
Caspian Tern	*Hydroprogne caspia*
Roseate Tern	*Sterna dougallii*
Black-naped Tern	*Sterna sumatrana*
Lesser Crested Tern	*Thalasseus bengalensis*

Crested Tern	*Thalasseus bergii*
Silver Gull	*Chroicocephalus novaehollandiae*

Appendix 2.

Table 2: UAV Tier Classification and Characteristics Source: Wong et al (2007).

CATEGORY	DESIGNATION	MAX ALTITUDE	RADIUS	SPEED	ENDURANCE	EXAMPLE
Tier I	Interim-Medium Altitude, Endurance	Up to 15,000 ft	Up to 250 km	60-100 kts	5-24 hrs	Pioneer, Searcher
Tier II	Medium Altitude, Endurance	3,000 ft to 25,000 ft	900 km	70 kts cruise	More than 24 hrs	Predator (Used in Bosnia)
Tier II Plus	High Altitude, Endurance	65,000 ft max	Up to 5,000 km	350 kts cruise	Up to 42 hrs	Global Hawk (expected to fly in early 1998)
Tier III Minus	Low Observable-High Altitude, Endurance	45,000 ft to 65,000 ft	800 km	300 kts cruise	Up to 12 hrs	Darkstar (expected to enter service in 1999)