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Ecosystem Research and Monitoring Program Advisory Panel as part of Gladstone Ports Corporation’s Ecosystem Research and Monitoring Program

Subject
Annual Report: Migratory Shorebird Monitoring – Understanding Ecological Impact (CA12000284)

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1. EXECUTIVE SUMMARY

This project is focused around integrating migratory shorebird species into understanding the ecological impact of developments. Migratory species range widely, and in theory, the impacted population could be far greater than the spatial footprint of a development itself, because animals could be dependent on the site for only a short period while in transit to non-breeding areas elsewhere, or use the site as part of a much larger foraging circuit. As well as being an interesting scientific problem, this is a key research challenge in minimising the harm of developments.

The scope of works divided the present project into two parts:

- Part A: **Estimate Carrying Capacity**; and
- Part B: **Determine the Size of the Impacted Population**.

We have completed approximately 50% of the work outlined in the scope of works. We had a weather-related setback in deploying the benthic sampling, but in response have accelerated many of the modelling and desktop analyses, such that pending the targeted collection of final field data in summer 2015/16, we will be able to move rapidly into the final analysis phase of the project during winter 2016. Despite the weather setbacks of 2015, we are on track to deliver the final report by the deadline of January 2017 (see timeline in section 4). We have planned all of the activities for the coming year, incorporating a risk assessment. The team is confident of delivering the remainder of the project within the planned timeframe.

In Part A, we have mapped the extent of tidal flats in the ERMP Survey Area, and analysed pilot benthic sampling data showing that most variation in the abundance of the benthos occurs at a scale of ~2–4 m. This unexpected result, coupled with severe weather that interrupted the sampling program has meant we have deferred the full benthic sampling program until November 2015. The design is finalised using a cost-benefit analysis, and will comprise 800 benthic samples across eight major tidal flats from the Fitzroy Delta to Rodds Peninsula.

In view of the delays to the benthic sampling work, we have accelerated the analytical side of the project, by establishing a series of calculations to go from benthic sampling data to estimates of carrying capacity. We provide a worked example that suggests fewer birds occur at Cattle Point (Fitzroy Delta) than could potentially be supported by the available prey base. We have taken 105 videos of focal foraging birds to permit study of diet and further refine the carrying capacity estimation method.
In Part B, we have modelled the movement of birds through the ERMP Survey Area, by developing a method to infer total population size calibrated against historical data available from sites across eastern Australia. Coupled with five targeted counts to discover northward migration phenology in the study area, we have the tools in place to provide a complete estimate of the size of the migratory shorebird population using the study area once we have completed the southward migration counts for all shorebirds in the ERMP Survey Area in late 2015. This work, together with the tidal flat mapping work, has been submitted to an ISI listed journal.

We have begun monitoring the movements of shorebirds, by capturing and banding 45 birds, and have thus far obtained 51 re-sightings of 14 locally and 33 externally marked birds. This revealed some movements of up to 10 km by individual birds, much longer than previously expected. We will conduct a radio tracking study in summer 2015/16 to characterise bird movements in detail. This will allow us to delineate priority areas for shorebirds in the study area and estimate the number of birds potentially affected by future loss or degradation of habitat patches anywhere in the study area.
2. BACKGROUND AND CONTEXT

A remarkable feature of the avifauna of the tidal flats of Australia is that it is not dominated by Australian-bred birds, but by shorebirds (also known as 'waders') from arctic and near-arctic breeding grounds some 6,000–13,000 km away. Shorebirds forage on tidal flats to build up the enormous fuel reserves required for their migrations, almost doubling in mass before setting out on non-stop flights that can exceed 8,000 km and take over a week of flying time.

Coastal shorebirds have attracted particular conservation interest for many reasons. Their elegant appearance and extraordinary migrations may be responsible for inspiring many birdwatchers to focus on this group and to establish active, international groups committed to shorebird research and conservation. Coastal migratory shorebirds also have attributes making them a matter of conservation concern. As their main tidal flat habitat is linear and therefore quite restricted, coastal shorebird populations are not particularly large. Moreover, they are dependent on an international network of breeding regions, non-breeding grounds and stopover sites, and are vulnerable to global population decline should any of these essential habitats deteriorate (e.g. Iwamura et al. 2013; Studds et al. in prep). Migratory shorebirds are listed as matters of National Significance under the Environment Protection and Biodiversity Conservation (EPBC) Act 1999, and are protected under a number of international treaties. In June 2015, two species, the eastern curlew and the curlew sandpiper (see Table 9 for all scientific names), were listed as Critically Endangered in Australia under the EPBC Act, because both have declined by more than 80% in three generations (Studds et al. in prep). Both species occur in the ERMP Survey Area.

Many migratory shorebird species are in severe decline across Australia (Creed & Bailey 1998, Wilson 2001, Minton et al. 2002, Reid & Park 2003, Olsen & Weston 2004, Gosbell & Clemens 2006, Rohweder 2007, Close 2008, Wainwright & Christie 2008, Rogers et al. 2009, Herrod 2010, Wilson et al. 2011, Cooper et al. 2012, Dawes 2012, Milton & Harding 2012, Minton et al. 2012, Szabo et al. 2012), and habitat loss, especially on Asian staging grounds, is considered their most serious threat (e.g. Murray et al. 2014). Habitat loss to shorebirds can take varied forms. Much habitat loss in this flyway has been very direct, with 'reclamation' projects converting enormous areas of tidal flat to agricultural or industrial land that shorebirds are unable to exploit. In addition, declines in the amount of infauna accessible to shorebirds can arise from a variety of causes, including weed invasion, pollution and over-harvesting; there have been a number of studies demonstrating that such declines can cause local or even global declines in population numbers (e.g. Baker et al. 2004; van Gils et al. 2006).

Shorebird numbers at non-breeding sites may also be limited by the availability of high tide roosts – typically very open and undisturbed areas at the water’s edge where shorebirds loaf.
in flocks when the tide is too high for foraging to occur. Roosts can be lost to shorebirds through construction, weed invasion or frequent disturbance. Loss of roosts can force shorebirds to abandon productive foraging areas if there are no suitable roost sites within 'commuting range' (Rogers et al. 2006a).

Coastal developments frequently result in loss, or at least modification of shorebird habitat, with resultant obligations on government or developers to manage shorebird habitats so that shorebird populations do not decline. The concept of 'carrying capacity' has become a baseline for management or offsets in such cases: i.e. the concept that in any particular shorebird site, population size will be limited by the available food supply. In theory, if the size of an area, the prey density and the rate at which the shorebirds are able to catch prey successfully within it, are known, one can calculate how many shorebirds the area can support. This might give clues as to whether a particular development could cause population decline. However, great care is needed in assessing impacts in the light of carrying capacity estimates. For example, birds might often emigrate before carrying capacity is reached (Goss-Custard et al. 2002), and thus predictions based on traditional carrying capacity estimates could under-estimate the effect of a development on shorebirds. Moreover, the number of migratory shorebirds present in an area depends not only on local conditions at the site, but also other sites along the migratory route or that displaced birds could occupy. Arguably, "scientists and managers should not ask whether a proposed change would cause the carrying capacity of the site to be reduced below that required by the number of birds that depend on the site at present. Rather, they should ask whether the change would decrease the survival rate and/or the proportion of birds that depart in good condition on spring migration" (Goss-Custard et al. 2002).

In practice, measurement of carrying capacity of shorebird habitat is difficult. A traditional approach is to analyse population trends, the underlying concept being that if habitats are 'full', counts will be similar from year to year, while if there are substantial annual fluctuations, then they cannot be at capacity (at least in years of low numbers). This may indeed be a scenario that applies to many Australian sites, given mounting evidence that Australian shorebird populations are declining as a result of habitat loss overseas (Wilson et al. 2011; Studds et al. in prep; Clemens et al. in review). However, to be confident in conclusions drawn from this approach, it is necessary to be able to demonstrate that there has been no concurrent decline in habitat quality. This would be challenging in the ERMP Survey Area, given the scantiness of previous data on benthos and shorebird abundance.

Estimates of carrying capacity require knowledge of the number of birds that occur on a site. Shorebirds lend themselves well to direct counts, as they congregate in relatively small roosts
at high tide. However, the number of shorebirds present at a site at any one time can be considerably lower than the numbers that use the site year-round; Queensland Wader Study Group (QWSG) datasets suggest that many shorebirds only use the eastern Queensland coast as a stopover area, migrating to non-breeding regions further south. We have used a modelling method to estimate passage dates and total number of migrants transiting an area (Thompson 1993). In addition to birds captured locally as part of this study, marked birds in the ERMP Survey Area include individuals that have been colour-banded or flagged elsewhere in the flyway; regular systematic scans for such birds will be made at a few accessible roosts to determine their arrival and departure dates.

Radio-telemetry may be a helpful tool in assessing migration dates of the shorebirds in this study. However, it will be used mainly to develop an improved understanding of local movements on non-breeding birds, identifying the scale of movements undertaken by individuals when moving between foraging and roosting sites (Rogers et al. 2006b), and hence the limits of the area in which shorebirds might be affected by Port development. Radio-telemetry of a few selected species will be supplemented by observational work on others, as shorebird species vary in their fidelity to particular non-breeding sites. Mark-recapture studies in Europe (Rehfisch et al. 1996, 2003) have suggested that some species have very high site fidelity and others do not; for example, red knots often move between relatively distant roosts, perhaps because they feed on small bivalves which are patchily distributed.
3. PROGRESS AGAINST PROJECT AIMS

3.1 Summary of aims

The project is divided into Part A and Part B, with four aims within each Part respectively. We list the aims here, along with a brief summary of progress with each one (Table 1).

Overall, approximately 50% of the project is complete. 82 days were spent on field trips, collecting and processing 305 benthic samples, recording 105 videos of foraging birds, conducting six shorebird counts, and catching and banding 45 birds. This represents the most comprehensive field study of shorebirds that has ever been undertaken in the region.

The project is primarily focused in the ERMP Survey Area (Figure 1), although given the mobile nature of migratory species, and the need to understand their movements and habitat linkages in a wider context, we provide several analyses in this report whose spatial scope reaches beyond the ERMP Survey Area.

![Figure 1](image_url) Extent of the ERMP Survey Area (yellow boundary).
Table 1 Progress against project aims, with an estimate of overall progress, a summary of activity to date, and a list of activities still to be completed for each aim.

<table>
<thead>
<tr>
<th>PART A: ESTIMATE CARRYING CAPACITY</th>
<th>% complete</th>
<th>Activities completed to date</th>
<th>Activities required to complete</th>
<th>Due date</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aim A2: Measure benthic prey availability</td>
<td>50%</td>
<td>Collected 305 benthic samples. Analysed spatial variation, and used cost/benefit analysis to design sampling program.</td>
<td>Collect 800 samples during a full sampling in Nov 2015; process the samples Dec 2015 - Feb 2016.</td>
<td>Apr 2016</td>
</tr>
<tr>
<td>Aim A3: Estimate how many birds the area can support</td>
<td>60%</td>
<td>105 videos of foraging birds taken to quantify diet. Method developed, worked example done. Awaiting benthic sampling to complete.</td>
<td>Apply method to produce estimates once data from Aim A2 available (4 weeks' work)</td>
<td>Aug 2016</td>
</tr>
<tr>
<td>Aim A4: Identify priority areas for management</td>
<td>20%</td>
<td>Base mapping complete.</td>
<td>Desktop analysis once data from Aim A2 available (two weeks’ work).</td>
<td>Jul 2016</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>PART B: DETERMINE THE SIZE OF THE IMPACTED POPULATION</th>
<th>% complete</th>
<th>Activities completed to date</th>
<th>Activities required to complete</th>
<th>Due date</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aim B1: Determine how many birds currently use the study area</td>
<td>90%</td>
<td>Review of past data complete. Method developed to use observed counts to estimate total abundance; paper submitted.</td>
<td>Desktop analysis to apply the method to ERMP Survey Area migration count data once Aim B3 complete (two weeks’ work).</td>
<td>May 2016</td>
</tr>
<tr>
<td>Aim B2: Discover how birds move around the study area</td>
<td>15%</td>
<td>45 birds leg-flagged, 51 resightings collected; radio tracking study designed.</td>
<td>Conduct radio tracking study (four months’ work Nov 2015 – Feb 2016).</td>
<td>Feb 2016</td>
</tr>
<tr>
<td>Aim B3: Describe the patterns of flow of birds into the study area</td>
<td>50%</td>
<td>Northward migration counts conducted.</td>
<td>Conduct southward migration counts (three months’ work). Write up results.</td>
<td>Jul 2016</td>
</tr>
<tr>
<td>Aim B4: Identify size of management units</td>
<td>30%</td>
<td>Base mapping complete, dozens of resightings collected.</td>
<td>Desktop analysis based on results of Aims B1-B3; 2 months’ work.</td>
<td>Aug 2016</td>
</tr>
</tbody>
</table>
3.2 Map tidal flat distribution and exposure (Aim A1)

3.2.1. Summary
We have completed mapping of the distribution of tidal flats in the region, as described below. The final step is to overlay bathymetry data to enable us to estimate the time period for which each area of tidal flat is exposed and thus available for shorebirds to feed. This tidal flat mapping work has been submitted to an ISI-listed journal (Dhanjal-Adams et al. in revision).

3.2.2. Background to tidal flat mapping
While many habitats are used by migratory shorebirds, including those above the high tide mark in mangroves, saltmarshes, salt pans and freshwater wetlands, the majority of migratory shorebirds in the ERMP Survey Area rely on intertidal habitats for foraging (GHD 2011a, 2011b; Sandpiper Ecological Surveys 2012a; Wildlife Unlimited 2013, 2014). Because long-distance migrations are so energetically demanding (Blem 1990), shorebirds must feed rapidly and store fat reserves before, during and after migration to ensure survival and reproduction (Drent & Piersma 1990). However, intertidal habitats occur in a relatively narrow strip along the coastline and therefore, in comparison with many other habitat types, have a very restricted distribution. For migratory shorebirds, the likelihood a site will sustain large numbers of birds is strongly correlated with the area of intertidal habitat, with the exposed area being a key factor influencing the availability of benthic prey organisms (Evans & Dugan 1984; Galbraith et al. 2002). Decreases in the extent and distribution of intertidal habitats could reduce the carrying capacity of a site and therefore the number of birds in an area, increasing the risk of local extinctions (Sutherland & Anderson 1993; Sheehy et al. 2011; Iwamura et al. 2013).

As well as providing critically important foraging habitat for shorebirds, tidal flats also provide coastal ecosystem services such as storm protection (Healy et al. 2002). However, tidal flats are declining in area in many regions (e.g. Murray et al. 2014), leading to suspected declines in biodiversity (Szabo et al. 2012, Wilson et al. 2011), loss of ecosystem services (Millennium Ecosystem Assessment 2005, An et al. 2007) and negative impacts to human livelihoods (Millennium Ecosystem Assessment 2005, MacKinnon et al. 2012). As such, an analysis of their current status is urgently required. Yet, mapping of tidal flats across large areas has rarely been attempted and present knowledge of their overall distribution remains remarkably limited. This is primarily due to a deficiency of tidal flat mapping techniques suitable for use over large areas. Here we use a method developed by Murray et al. (2012) based on differencing Landsat scenes taken at high and low water to obtain maps of the extent of tidal flats in the ERMP Survey Area, as part of a broader effort to map tidal flats around the whole coastline of Australia. This piece of work has been submitted for publication (Dhanjal-Adams et al. in revision).
3.2.3. Methods

The method we used to map the extent and distribution of intertidal habitats across Australia was based on a continental-scale mapping project conducted across Asia by Murray et al. (2012, 2014). We first obtained the complete metadata of the freely available Landsat Archive from USGS Earth Explorer (http://earthexplorer.usgs.gov), including in our search all images acquired between 1999 and 2014. We chose this time period as a compromise between (i) achieving sufficient coverage (cloud cover, non-alignment with desired tides and other factors severely constrain the number of suitable images), and (ii) minimising the effects of change through natural episodic events and anthropogenic drivers. We identified all Landsat images that intersected the study area. Using the Tide Model Driver (TMD) MATLAB toolbox for tide modelling, we estimated the tidal elevation at the time of image acquisition with the Indian Ocean, Tasmania and Northern Australia tide models available from the Oregon State University suite of tide models (Egbert & Erofeeva 2002; Padman & Erofeeva 2005). Images acquired within the upper and lower 10% of the tidal range were downloaded and visually reviewed before being selected for the final remote sensing analysis, following Murray et al. (2012). For images not available via Earth Explorer, we obtained the ortho-corrected Landsat Archive images from Geoscience Australia and the Department of Environmental Resource Management (Filmer et al. 2010). Image pre-processing, sorting and pairing followed the procedure in Murray et al. (2012).

The final national image set consisted of 99 pairs of Landsat scenes over 79 path-row footprints of 185-km × 170-km each, with 170 Enhanced Thematic Mapper Plus (ETM+), 28 Landsat Thematic Mapper (TM) satellite images. The mean difference in acquisition time between high and low tide members of the image pairs was 1.49 ± 1.18 years. The Normalised Differenced Water Index (NDWI; McFeeters 1996) and where possible the Modified Normalised Differenced Water Index (MNDWI; Xu 2006) was calculated for each pixel to maximise the likelihood of separating inundated and non-inundated areas, irrespective of the substratum or benthos (McFeeters 1996; Xu 2006). Each image was then classified into a binary land/water image by manually assigning a threshold that most effectively identified the waterline in each image. Images were discarded if a suitable threshold could not be found that consistently identified the waterline throughout the image. After converting each image to a binary format, the classified high and low tide images in each pair were differenced, resulting in a measurement of intertidal habitats as the difference between the two input images (Murray et al. 2012). For further detail on the NDWI differencing method refer to Murray et al. (2012).

The intertidal areas identified from all Landsat images were merged to create an estimate of the distribution of intertidal habitats across Australia at a 30 m resolution. Post-processing
was necessary to remove incorrectly classified pixels (Murray et al. 2012, 2014). We completed an accuracy assessment on the final intertidal habitats map to measure classification error, by comparing a mapped data set with a reference set, using a confusion matrix (Congalton & Green 2008; Roelfsema & Phinn 2013). Using stratified random sampling, we generated 204 sample locations within 10 km of the coastline and within the intertidal class as per the methods of Congalton & Green (2008). Each point was assessed by an independent reviewer and labelled as belonging to one of the two classes to create a reference data set based on a combination of ground-truth information, including low tide Landsat imagery, Google Earth imagery and ESRI World imagery. For each point, the mapped data were extracted from the intertidal habitats map created in this study. Then, using the mapped data and the reference data set, we populated a confusion matrix and quantified the map category user’s and producer’s accuracy and the map overall accuracy (Congalton & Green 2008). User’s accuracy or reliability, represents the probability a pixel classified on the map actually represents intertidal habitats. Producer’s accuracy represents a measure of omission error i.e. the probability a reference pixel is correctly classified (Congalton & Green 2008).

3.2.4. Results

Individual map category user’s accuracy for intertidal class was 100%, and 91.2% for the open ocean class (Table 2), relatively high in comparison with many other remote sensing studies (Congalton & Green 2008; Foody 2009). Errors were likely due to factors including cloud cover, water turbidity confusing land and water, algal blooms, whitewash from waves and vegetation cover (particularly mangroves) which affected the classification output, as well as the limitations inherent in delimiting tidal flat and open water features (Liu et al. 2012; McFeeters 1996; Ryu et al. 2002; Xu 2006). False positive classification errors occurred both landward and seaward in many images. In part, these were due to seasonal changes in water presence, such as flooding and ephemeral wetlands inland appearing in one image but not the other, but most false positive errors occurred when open ocean was classified as intertidal. These errors were corrected in the post-processing phase of the analysis. False negative classification errors, on the other hand, occurred consistently on the landward side, where small strips of intertidal habitat were not correctly classified. These areas of intertidal habitat, probably consisting of salt marsh, un-vegetated tidal flat and shoreline, were caused by the methodological limitation requiring the selection of images within 10% of the high tide, rather than the highest possible tide. Although we used state of the art tide models, errors are likely to remain in the tide predictions due to model resolution and tidal variation across the extent of each Landsat image, as well as variability in timing of Landsat imagery (although note that because we combined multiple images this problem is minimised). Also, the failure of the scan line corrector on the Landsat satellite resulted in much of the later imagery missing up to 22%
of data, but again, by merging extents of tidal flat derived from 15 years of imagery, the effect of this data loss is greatly reduced (see Markham et al. 2004). For further discussion of errors associated with this remote sensing method, refer to Murray et al. (2012).

**Table 2** Confusion matrix for mapping of intertidal habitats. The matrix compares 204 randomly stratified points from the reference and mapped data, in the columns and rows respectively. The proportion of correctly allocated cases indicates the overall classification accuracy.

<table>
<thead>
<tr>
<th>Reference</th>
<th>Intertidal Flat</th>
<th>Other</th>
<th>Sum</th>
<th>User Accuracy</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Mapped</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Intertidal Flat</td>
<td>102</td>
<td>0</td>
<td>102</td>
<td>100.0%</td>
</tr>
<tr>
<td>Other</td>
<td>9</td>
<td>93</td>
<td>102</td>
<td>91.2%</td>
</tr>
<tr>
<td>Sum</td>
<td>111</td>
<td>93</td>
<td>204</td>
<td>Overall accuracy 95.6%</td>
</tr>
<tr>
<td><strong>Producer's accuracy</strong></td>
<td>91.9%</td>
<td>100.0%</td>
<td>Overall accuracy</td>
<td>95.6%</td>
</tr>
</tbody>
</table>

Our map of intertidal habitats achieved 91% coverage of the Australian coastline with an overall accuracy of 95.6% at a 30 m resolution (Table 2). We identified a minimum total of 9856 km$^2$ of intertidal habitat across Australia, with intertidal habitats being largely concentrated into estuaries, embayed coastlines and areas protected by coral reefs (Figure 2). The states with the largest areas of intertidal habitats mapped were Queensland, Western Australia, the Northern territory and South Australia (Figure 2; Table 3). Central and southern Queensland, along with the coast of the Gulf of Carpentaria, support nationally significant intertidal habitats, indicating a large area of suitable habitat for shorebirds south of Cairns, and in the SE Gulf. This sets the context for our analysis of shorebird migration later in this report (see section 3.6).
Figure 2 Distribution of intertidal flats around the Australian coastline. The importance of the Queensland east coast for this habitat is clearly apparent.

Within the Port Alma-Port Curtis study area, extensive tidal flats are apparent, with important concentrations in the Fitzroy Delta area, particularly in the area of Cattle Point, Balaclava Island, Connor Creek, Deception Creek and the Narrows (Figure 3). Substantial tidal flats also occur along the north end of Curtis Island, with a particularly important concentration at Yellow Patch on the NE tip of the Island (Figure 3). Moving south, there are large tidal flats off the SE tip of Curtis Island at Pelican Banks, and also around Facing Island and Boyne Island.
Table 3 Distribution of intertidal habitats in Australia. Note that for the purpose of this analysis we consider Jervis Bay Territory to be part of New South Wales.

<table>
<thead>
<tr>
<th>State</th>
<th>Mapped coastline in km (Percentage of state or territory coastline)</th>
<th>Total Intertidal Habitat in km²</th>
<th>Area of intertidal habitat per km of coastline mapped (km²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>ACT</td>
<td>0 (0)</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>NSW</td>
<td>3793 (100)</td>
<td>95.6</td>
<td>0.03</td>
</tr>
<tr>
<td>NT</td>
<td>10384 (96.68)</td>
<td>2235.1</td>
<td>0.22</td>
</tr>
<tr>
<td>QLD</td>
<td>11235 (97.54)</td>
<td>2682.1</td>
<td>0.24</td>
</tr>
<tr>
<td>SA</td>
<td>4709 (99.99)</td>
<td>925.8</td>
<td>0.20</td>
</tr>
<tr>
<td>TAS</td>
<td>4235 (87.10)</td>
<td>91.8</td>
<td>0.02</td>
</tr>
<tr>
<td>VIC</td>
<td>2404 (99.99)</td>
<td>231.7</td>
<td>0.10</td>
</tr>
<tr>
<td>WA</td>
<td>15611 (80.15)</td>
<td>3593.4</td>
<td>0.23</td>
</tr>
<tr>
<td>AUSTRALIA</td>
<td>52372 (91.08)</td>
<td>9855.6</td>
<td>0.19</td>
</tr>
</tbody>
</table>

3.2.5. Discussion of results to date

Our analysis resulted in a comprehensive map of intertidal habitats in Australia. We identified a minimum total area of 9,856 km², 208 with an accuracy of 95.6%, and 39% of the total extent of intertidal habitats was covered by some form of protected area (Dhanjal-Adams in revision). The ERMP Survey Area supports 275 km² of tidal flats, in keeping with the large numbers of migratory shorebirds known to over-summer in the area. For example, the most recent comprehensive survey of the area found 13,752 migratory shorebirds of 21 species (Wildlife Unlimited 2015).

3.2.6. Next steps to finalise Aim A1

We are currently sourcing bathymetry data, which once overlain onto the distribution of tidal flats, will allow us to estimate the time for which each area of tidal flats is exposed, and therefore available to foraging shorebirds, on each tide. As well as being of inherent interest, for example if there is any spatial patterning to the result, this will be a key input into the carrying capacity estimate (see section 3.4). We expect Aim A1 to be fully complete by the end of August 2015.
3.3 Measure benthic prey availability (Aim A2)

3.3.1. Summary
We have deployed a program of benthic sampling, in which we identified the invertebrates living within the top layer of the sediment (accessible by probing shorebirds) and estimated the densities of prey potentially available to foragers. As explained below, we elected to conduct a detailed study to determine the precise spatial scale at which variation in benthic prey numbers needs to be sampled and this, coupled with severe weather that impacted fieldwork, meant we were unable to complete the full program of sampling in Year 1 as originally planned. This has been the subject of a contract variation, which has provided us with resourcing to complete sampling across one quarter of the study area in summer 2015/16. The design of this benthic sampling program is finalised and 800 samples will be taken in November 2015, with sorting of the samples occurring during summer 2015/16 (see Table 1).

3.3.2. Background to benthic sampling
To estimate how many shorebirds the ERMP Survey Area could support, we must first understand how much prey is available to foraging shorebirds that could potentially use the area. Resolving the abundance and distribution of shorebird prey (benthic invertebrates) is a critical component in the estimation of shorebird carrying capacity. Estimating prey abundance
typically uses a grid-based approach whereby a large grid with 0.5 km or 0.25 km intersects is imposed over the area of interest – usually a single tidal flat (e.g. Gill et al. 2001). In such an approach, samples of benthic prey are taken at each intersection of the grid and the prey densities at un-sampled locations within the grid are spatially interpolated. The advantage of such a coarse grid-based approach is that broad spatial coverage can be achieved, but it makes several key assumptions about the spatial scale of variation in prey abundance. Our initial visits to the sites suggested to us that variation in prey abundance was occurring at much finer scales than would be captured by the typically-used coarse grid. Indeed, benthic communities in soft sediments can be patchily distributed at a range of spatial scales from less than a metre to several kilometres (Morrisey et al. 1992). Given that the only previous benthic sampling in the study area was primarily subtidal (see Currie & Small 2005, 2006), we needed to measure the pattern spatial variation in shorebird-relevant intertidal benthos de novo.

The pilot study incorporated four different spatial scales ranging from 2 m between replicate cores to 50 km between tidal flats. The aims of the study were (i) to identify the spatial scales at which variation in shorebird prey communities is significant for the study area, (ii) to estimate the contribution of different spatial scales to the total variation in shorebird prey abundance between regions in the study area, and (iii) to perform a cost-benefit analysis to determine the optimal allocation of sampling effort in the second field season for estimating prey abundance and distribution.

3.3.3. Methods

Two tidal flats in different regions of the study area were selected for the pilot study, in the Fitzroy Delta and in Port Curtis. These flats were two of the largest within their subregions, providing broad spatial coverage of the study area and represent independent estuarine settings. The roosts near to the tidal flats that were selected have been found to support consistently greater numbers of shorebirds than more distant roosts (over 200 on average) each year during the summer and are thought to be important foraging habitat for migratory shorebirds in the ERMP Survey Area. Shorebirds in these tidal flats tend to forage mostly on the intertidal zone and seldom on the supratidal zone (claypans and dense mangroves), therefore, the latter was excluded from the present analysis.

A large grid with 0.5 km intersects was imposed over each tidal flat. Two grid squares were selected at random in each tidal flat. Each grid square was then further sub-divided so that it consisted of 25 equidistant sampling stations (Figure 4). The distance between adjacent stations was 125 m. Where sample stations intersected with a channel or another obstacle that prevented sampling, the samples were taken from the nearest possible location at which it was feasible to sample.
Figure 4 Pilot benthic sampling locations at (a) Cattle Point, (b) Pelican Banks. At each location, two 500 m x 500 m grids were sampled, with stations spaced 125 m apart in a grid, and three replicate cores 2 m apart taken at each station. Tidal flats mapped in Section 3.2 are shown in yellow.

Sampling was carried out during daylight hours at low tide on the 2nd, 4th and 11th of December 2014 at the Fitzroy Delta site and on the 3rd, 5th, 7th, and 12th of December 2014 at the Port Curtis site. Three replicate core samples spaced 2 m apart were collected at each station. The coring device consisted of a PVC tube 20 cm deep and 18 cm in diameter. Following collection, the entire core samples were immediately fixed in a 7% buffered formalin solution to prevent deterioration of soft-bodied organisms during the sieving process. After approximately one week, samples were transferred to 70% ethanol solution to await sieving and sorting in the laboratory.

The samples were sieved through a 0.5 mm mesh sieve. In shorebird carrying capacity studies, core samples are usually passed through a 1 mm mesh sieve, but because we are calculating carrying capacity for multiple shorebird species ranging in size and feeding habits we elected for the smaller size. For example, some smaller shorebird species such as red-necked stints are known to feed on prey smaller than 1 mm (Dann 1999). It was therefore important to use the smaller mesh size to ensure that the prey community for smaller shorebirds species was not under-estimated. Samples were then sorted to a coarse taxonomic level appropriate to the taxon in question (typically class or order level). Finer level identifications were not performed as:

I. this would have been extremely time consuming; and
II. there is little value in sorting to finer taxonomic resolution as prey selection by shorebirds is unlikely to be influenced by subtle morphological characters that can only be detected with a microscope.

The final component of the pilot study was to perform a cost-benefit analysis. Here, the “cost” of sampling at a particular spatial level is taken to be the total time required to complete sampling at that level while the “benefit” of sampling at each level is considered to be the contribution that level makes to the total variance across all the spatial scales (Underwood 1981). At each spatial scale, all the costs (time) required to complete tasks was recorded. In the final cost-benefit analysis, an average of these costs (times) was then used for each spatial scale based on all sampling done on the two tidal flats.

By chance, the two grids selected at random on Cattle Point were relatively close together with some sampling stations co-occurring. For the cost-benefit analyses, we were interested in estimating how to allocate sampling effort optimally across all the appropriate spatial scales, so it was important that we included analysis of data from grids that were separated at least by 400-500 m. We achieved this by sub-sampling the two grids, to create three smaller grids, each 10 stations in area (5 stations across x 2 stations deep; each station separated by ~125 m).

The data from these three smaller grids were than analysed with a nested (hierarchical) analysis of variance that incorporated the three critical spatial scales (1–2 m, among replicate cores; 100–125 m, among replicate stations; 400–500 m, among replicate grids). The relevant variance components were extracted from the results of an Analysis of Variance (ANOVA; see Underwood, 1981 for appropriate Mean Square estimates and the calculation of the variance components).

It is important to note here that once processing of the samples from the second tidal flat (at Port Curtis) is completed, these analyses can be expanded to incorporate the larger spatial scales (across hundreds of kilometres) and determine how variability in the abundance of the prey is partitioned across all the relevant spatial scales in the study area. This will be the first time such a detailed and carefully designed benthic sampling program has been completed in the ERMP Survey Area, and we expect to produce a publication based on this work.

Cost-benefit analysis then makes use of the combined information of costs (time) and benefits (variance) to minimise the allocation of effort across the different spatial scales providing the optimal sampling design. Here, we defined our total “budget” to be determined by the total number of samples that could be collected for the forthcoming summer sampling program (in this case, 800 samples). We elected to use this measure as the defining cost in the cost
benefit analysis because of the critical financial constraints on the project and the need to sample across a very broad spatial area. The total Cost (C) and Variance (V) are then minimised to provide the most cost effective and accurate sampling program.

3.3.4. Results

In total, 3,790 individuals from 13 different major marine invertebrate orders were counted in the 147 samples collected and sorted from Cattle Point at the mouth of the Fitzroy River. Overall average density of invertebrates was 1,013 per m$^2$ and, of these, bivalves were the most abundant prey group comprising 1,746 individuals in total with an average density of 12 bivalves per m$^2$ (Table 4). Small crustaceans including copepods and amphipods were the next most abundant groups comprising 1,461 and 384 individuals respectively. Polychaetes were the next most abundant group with 150 individuals counted in total.

Table 4 Mean richness, abundance and density of major shorebird prey groups at Cattle Point, Fitzroy Delta.

<table>
<thead>
<tr>
<th></th>
<th>Total number of individuals</th>
<th>Bivalves</th>
<th>Copepods</th>
<th>Amphipods</th>
<th>Polychaetes</th>
<th>Number of Taxa</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance (Individuals per core sample)</td>
<td>25.78 ± 4.08</td>
<td>11.88 ± 2.42</td>
<td>9.94 ± 2.41</td>
<td>2.61 ± 0.56</td>
<td>1.05 ± 0.12</td>
<td>2.24 ± 0.12</td>
</tr>
<tr>
<td>Density (Individuals per m$^2$)</td>
<td>1013.18 ± 160.17</td>
<td>466.76 ± 94.76</td>
<td>390.57 ± 94.76</td>
<td>102.65 ± 22.05</td>
<td>41.44 ± 4.66</td>
<td></td>
</tr>
</tbody>
</table>

We used the abundance of all prey items, sampled across the tidal flat at Cattle Point, as the variable for our determination of the optimal sampling strategy. We opted for using the total prey items, rather than the abundance of a specific taxon (e.g. bivalves), in these analyses because we needed to design a strategy that would be broadly applicable to a wide range of shorebirds which individually forage on many different prey items. Further analyses for the final report will, however, investigate possible optimal sampling designs for specific shorebirds based on their individual diets in order to guide any future studies done in the Port Alma - Port Curtis area.
Table 5 ANOVA results for the total abundance of benthic prey items at Cattle Point, Fitzroy Delta across three spatial scales ranging from metres (replicate cores) to separate grids (>500m).

<table>
<thead>
<tr>
<th>Source of Variation</th>
<th>DF</th>
<th>Effect (F/R)</th>
<th>Sums of Squares</th>
<th>Mean Square</th>
<th>Denom. df</th>
<th>Denom. MS</th>
<th>F-ratio</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grids</td>
<td>2</td>
<td>F</td>
<td>16953.09</td>
<td>8476.54</td>
<td>27</td>
<td>3586.67</td>
<td>2.36</td>
<td>0.113</td>
</tr>
<tr>
<td>Stations (Grids)</td>
<td>27</td>
<td>R</td>
<td>96840.07</td>
<td>3586.67</td>
<td>60</td>
<td>1323.71</td>
<td>2.71</td>
<td>0.001</td>
</tr>
<tr>
<td>Error</td>
<td>60</td>
<td></td>
<td>79422.67</td>
<td>1323.71</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Despite the relatively large spatial separation of the grids that were laid out across Cattle Point (400–500m), the total abundance of all prey items did not differ significantly among the grids (Table 5; grids, P=0.113). The abundance of the prey available to the shorebirds was, however, extremely patchy at the spatial scale of 100’s of metres, i.e. among stations across the grid (Table 5; stations (grids), P<0.001).

Table 6 Estimates of the variance components for each of the spatial scales sampled in the pilot study at Cattle Point, Fitzroy Delta, using the results of the ANOVA above (see Table 5)

<table>
<thead>
<tr>
<th>Source of Variation</th>
<th>DF</th>
<th>Mean Square</th>
<th>MS Estimate</th>
<th>Estimated Variance</th>
<th>% contribution</th>
</tr>
</thead>
<tbody>
<tr>
<td>Replicate Grids across the Site</td>
<td>2</td>
<td>8476.54</td>
<td>$\sigma_1^2 + n_3\sigma_2^2$</td>
<td>$\sigma_1^2 = 163.00$</td>
<td>7.3</td>
</tr>
<tr>
<td>Replicate Stations in each Grid</td>
<td>27</td>
<td>3586.67</td>
<td>$\sigma_3^2 + n_3\sigma_2^2$</td>
<td>$\sigma_2^2 = 754.32$</td>
<td>33.7</td>
</tr>
<tr>
<td>Replicate Cores at each Station</td>
<td>60</td>
<td>1323.71</td>
<td>$\sigma_3^2$</td>
<td>$\sigma_3^2 = 1323.71$</td>
<td>59.1</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>100.00</td>
<td></td>
</tr>
</tbody>
</table>

Approximately 59% of the variation in the abundance of prey items at Cattle Point was associated with the smallest spatial scale, among the replicate cores collected at each station (Table 6). A further ~34% of the variation in the prey abundance was among the replicate stations (~125m apart) within each of the grids. Therefore nearly 94% of the variation in the abundance of the prey resource for the shorebirds occurs at the smallest spatial scales on even large tidal flats. This indicates that the benthic prey resources are very patchy at a fine scale, suggesting the shorebirds will have to move frequently in order to access food as rapidly as possible.
Table 7 Calculations of the costs (time) and benefits (variances) associated with the pilot sampling program at Cattle Point, Fitzroy Delta, for each of the relevant spatial scales.

<table>
<thead>
<tr>
<th>Costs of Sampling</th>
<th>Cost/Time (mins)</th>
<th>% Cost</th>
<th>Variance</th>
<th>% Variance</th>
</tr>
</thead>
<tbody>
<tr>
<td>$C_1$ = time per grid</td>
<td>54</td>
<td>29.5</td>
<td>163.00</td>
<td>7.3</td>
</tr>
<tr>
<td>$C_2$ = time per station</td>
<td>15</td>
<td>8.2</td>
<td>754.32</td>
<td>33.7</td>
</tr>
<tr>
<td>$C_3$ = time per core</td>
<td>114</td>
<td>62.3</td>
<td>1323.71</td>
<td>59.1</td>
</tr>
</tbody>
</table>

The real costs of the sampling program arise for the spatial scales of the replicate cores, with 63% of effort/cost associated with work at this smallest spatial scale (Table 7).

Each core must be fixed in order to preserve and protect the soft-bodied animals for subsequent counting and identification. These cores are then carefully sieved and processed to extract the small animals from among the sediment particles. It is this processing of the samples that results in benthic sampling programs being relatively expensive, to obtain the critical information needed to assess the abundance and variability in the food resource available to the shorebirds.

Complexity in designing the future sampling program arises because of the relative contribution to the costs (time) associated at the larger spatial scale of grids spread across each tidal flat (~30%) compared with the contribution to the benefits (variance) in the total abundance of prey at that spatial scale (~7%). The large number of samples collected within each grid (number of stations x number of replicate cores) results in considerable time being spent packing these samples for transfer to the laboratory.

We will be modifying our procedures to minimise the time spent on these tasks to optimise further the sampling program for the forthcoming summer period. Optimisation of the sampling strategy therefore must compete with minimising costs ($C$ – time) that are relatively large at the greatest spatial scale compared with benefits ($V$ – variance for the mean number of prey items per core) that are relatively unimportant. The subsequent sampling strategy will focus on:

- Reducing costs, especially associated with processing of samples from each grid in preparation for transfer to the laboratory. The number of grids allocated to a tidal flat is determined in order to ensure we obtain accurate estimates of the abundance of the food resource across each of the tidal flats.
- Ensuring sufficient numbers of stations are allocated within each grid on a tidal flat. Shorebirds are capable of and do undertake small-scale movements across individual...
tidal flats in search of food, so our future sampling strategy must incorporate adequate replication at this meso-scale (~125 m).

- The greatest cost and contribution to total variance occurs at the smallest spatial scale – among replicate cores – indicating that we must continue to allocate appropriate effort at this spatial scale, on each of the tidal flats in each of the four study regions.

### 3.3.5. Discussion of results to date

The results show that the pattern of spatial variation in benthos relevant to shorebirds is primarily occurring at fine spatial scales within the sampled areas. This has informed the design of the sampling regime for summer 2015/16 which we outline previously in Section 3.3.1. While we expected variation to be relatively fine scale, we were surprised by the important amount of variation at the station level, and without this carefully constructed pilot study there would have been considerable risk of under-sampling at these fine and meso scales, weakening inference about carrying capacity.

### 3.3.6. Next steps to finalise Aim A2

We will conduct the full-scale sampling program in November 2015, which will sample eight tidal flats across four regions in the study area (Table 8). Sampling across each flat will be allocated according to the strategy in section 3.3.4.

**Table 8** Sampling locations for full benthic sampling program.

<table>
<thead>
<tr>
<th>Region</th>
<th>Site</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fitzroy Delta</td>
<td>Cattle Point</td>
</tr>
<tr>
<td></td>
<td>Deception Point</td>
</tr>
<tr>
<td>North Curtis Island</td>
<td>Warner Point</td>
</tr>
<tr>
<td></td>
<td>Yellow Patch</td>
</tr>
<tr>
<td>Gladstone Harbour</td>
<td>Pelican Banks</td>
</tr>
<tr>
<td></td>
<td>Facing Island</td>
</tr>
<tr>
<td>Rodds Peninsula</td>
<td>Mundoolin Rocks</td>
</tr>
<tr>
<td></td>
<td>Rodds Bay</td>
</tr>
</tbody>
</table>
3.4 Estimate how many birds the area can support (Aim A3)

3.4.1. Summary
We have extensively reviewed the literature on estimating carrying capacity, and present here the results of this review, together with a worked example of turning benthic sampling data (from the pilot study described above) into estimates of the number of foraging shorebirds a tidal flat can support. For example, initial results from Cattle Point, in the Fitzroy Estuary, suggest that the site is capable of supporting up to five times the number of bar-tailed godwits than currently occur there. However, the calculations include a number of estimated parameters that will be tested with further analysis of field data.

3.4.2. Background to estimating carrying capacity
Carrying capacity is a concept that has been employed in a wide range of disciplines as wide ranging as population ecology, wildlife management, anthropology and mechanical engineering. Coined as early as 1845 within the field of mechanical engineering to determine the amount of duties that could be imposed on a ship (Sayre 2008), the term was subsequently introduced in the field of ecology and applied to grazing livestock (Hadwen & Palmer 1922) and herbivorous wildlife (Leopold & Brooks 1933). It was not until Odum (1953) defined it in the context of population limitation as the upper asymptote of a logistic population growth trajectory (also known as K, or equilibrium level) that its popularity increased dramatically within the field of ecology (Dhondt 1988; Odum 1953).

Over the years the concept of carrying capacity has led to considerable confusion and uncertainty in definitions and measurement methods, possibly caused by its use across a range of disciplines, and has thus also received considerable criticism (Sutherland 1996; Sutherland & Parker 1985). Part of this criticism relates to difficulties in substantiating the natural limit to population growth (Price 1999) and whether this limit is a dynamic quantity that may vary in time (e.g. amount of food in a pasture) or a fixed constraint (Sayre 2008). Moreover, in an ecological context, carrying capacity may not only depend on habitat features, but also on the behaviour of the animals living there (Newton 1998).

For instance, the carrying capacity of an area could vary in relation to the extent of territoriality and gregarious behaviours and (at least partially) independently of resource availability. A complication in the case of highly mobile animals, such as migrants, may arise from the fact that carrying capacity at one place could be influenced by events at other locations (Goss-Custard 1993).
Furthermore, the definition of the term 'carrying capacity' has been ambiguous. One of the more complex definitions is “the user-specified quality biomass of a particular species, under the influence of social or behavioural constraints, for which a particular area, having user-specified objectives, will supply all energetic and physiological requirements over a long (but specified) period” (Giles 1978). Conversely, it has also been simply defined as “the maximum population of a given organism that a particular environment can sustain” (Allaby 2014). In the study of migratory organisms it has been defined as the “maximum numbers of migratory animals that can be supported in a particular locality at a particular time of year” (Goss-Custard et al. 2002).

Given the diverse usages and ambiguity of the carrying capacity concept, it is not surprising that some researchers suggested it be abandoned (Dhondt 1988). Nonetheless, it has substantial utility, notably in the field of conservation biology, potentially allowing the quality evaluation of different habitats and sites in terms of their suitability to harbour a certain number of individuals of target species.

Carrying capacity estimates could inform prioritisation processes, shed light on important manageable factors that may affect population size and assist in predicting the effects of environmental change on key species. The applied values of the carrying capacity concept may be particularly relevant to the conservation of migratory shorebirds along the East Asian-Australasian Flyway given their alarming population declines (Moores et al. 2008; Wilson et al. 2011), which have been linked to the rapid disappearance of their coastal habitats (Murray et al. 2014; Studds et al. in prep). There is thus an urgent need for rapid assessment protocols that can quickly assess habitat change at impacted sites along the flyway. Therefore, we retain and recommend the carrying capacity concept as a highly valuable paradigm, under the condition that a clear definition is provided and results are interpreted with care. In this study we follow the approach of Goss-Custard et al. (2002), and define carrying capacity as the "maximum number of migratory shorebirds that can be supported in the ERMP Survey Area during the non-breeding season".

For the measurement of carrying capacity both demographic (Dasmann 1964) and energetic approaches (de Leeuw 1997) have been considered. The former is a numerical approach focusing on the demographic analyses of the number of organisms using an area but the outcome from such approach is often inconclusive because concurrent measurement of local habitat quality and potential influences beyond the study area are often unfeasible or neglected. The alternative, energetic carrying capacity is a more deterministic, functional approach that examines the biological mechanisms limiting the utilisation of food resources (de Leeuw 1997). Models that have been employed to estimate energetic carrying capacity
include, in order of increasing complexity, Daily Ration Models, Spatial Depletion Models and Spatially-explicit Individual-based Models (Stillman & Goss-Custard 2010).

Daily Ration Models estimate the total number of bird-days an area can support based on the total amount of food available and the requirement of an average individual (Alonso et al. 1994; Stillman & Goss-Custard 2010). In some cases critical prey density, below which food intake rate rapidly diminishes or ceases, is also taken into consideration (Gill et al. 2001). Despite its successful application in a number of cases (e.g. Alonso et al. 1994; Gill et al. 2001), it has been criticised for its simplicity. Adding complexity, the Spatial Depletion Models consider spatial variation in food abundance and track how identical foragers utilise the different food patches using game theoretical approaches (Stillman & Goss-Custard 2010). Spatial Depletion Models consider fixed critical prey densities and, like Daily Ration Models, assume all individuals in the population to be identical. However, it has been shown that assuming a fixed critical prey density may not be appropriate in spatially heterogeneous environments with varying costs in different patches (van Gils et al. 2004), while individual organisms are unlikely to be identical in competitive ability and foraging efficiency (Ens & Goss-Custard 1984). As a result, a population does not instantly disappear if prey abundance declines below a fixed critical prey density; instead, some individuals will have the foraging or competitive skills to remain in preferred patches while others do not, resulting in patchy and gradual declines in abundance rather than an instant (and easily interpreted) exodus. Therefore, more sophisticated approaches have been developed based on individual-based models, which assume individuals are usually different and behave in ways that maximise their fitness (Railsback & Grimm 2011). These Spatially-explicit Individual-based Models tend to be far more complex, tracking large numbers of individuals with different entities rather than identical individuals. Often these models are data hungry, requiring a lot of data for model parameterisation and calibration (Stillman & Goss-Custard 2010).

In the current project, given the limited human resources and the lack of previous foraging studies on shorebirds in the area, we are adopting a Daily Ration Model to estimate the carrying capacity for shorebirds of the Port Curtis / Port Alma area. The fundamental data required for models of this kind are measures of prey abundance and availability (i.e. considering tidal exposure of the potential foraging areas and burrowing depths of potential prey species), shorebird abundance, daily food requirements of the shorebirds and assessment of the food intake rate in relation to prey density (to estimate critical prey density; Sutherland & Anderson 1993). To this end we will map tidal exposure across the region using a remote sensing method recently developed at UQ (Dhanjal-Adams et al. in revision). We will use the QWSG database of leg-flagged birds and remotely-sensed data to map all tidal flats known to be used by or potentially useful to shorebirds in the Port Curtis / Port Alma region.
For the sampling of intertidal benthic invertebrate communities (i.e. potential shorebird prey), we will use a random and stratified (i.e. considering depth distribution of benthos) design. In conjunction with published data on shorebird diets and foraging behaviour yielding critical prey density estimates), we will subsequently estimate and map migratory shorebird carrying capacity across the study area.

3.4.3. Worked example based on pilot data from 2014-2015
In this report, we include a worked example to show how shorebird carrying capacity is assessed, in this case for bar-tailed godwit, using the spatially limited benthic data set collected at Cattle Point. We conducted focal bird observation in the field and combined it with information from published literature to quantify the potential diet for different shorebird species (Higgins & Davies 1996). A total of 105 video clips from 12 different shorebird species was recorded (average duration 3:56 minutes) and a range of different prey types have been observed being taken by different shorebird species (Figure 5).
Food availability

In this worked example, food availability was derived from our benthic pilot sampling in the Fitzroy Estuary region. Details on the benthic sampling method can be found in the benthic sampling section of this report. Given 99% of all benthic organisms collected belong to five groups (i.e. Polychaeta, Brachyura, Bivalvia, Amphipoda and Copepoda), our remaining analysis will only focus on these. We assumed no net production and all benthos depletion took place by shorebirds only.
To convert count data of benthic organisms into prey energy content, we took the following steps:

1. We randomly selected and measured the size (in mm) of at least 20 individuals from each of the 5 benthic groups, except Brachyura for which only 7 individuals were available. All measured individuals were harvestable and ingestible food items for bar-tailed godwits (Choi pers. obs.).

2. We used published benthic size-biomass (ash-free dry mass, hereafter AFDM) relationships of the focal or closely related taxa (Rogers 2006a; Zwarts & Wanink 1993) to work out the AFDM of all measured individuals. We then took the average AFDM among individuals in each of the five benthic groups.

3. For each of the five benthic groups we calculated the total AFDM for each core sample based on the total prey counts within each benthic group.

4. We converted AFDM to gross energy content using 22 kJ / g AFDM for Bivalvia and 21.48 kJ / g AFDM for all other taxa. Next we converted these values to digestible energy content through multiplication with the assimilation efficiency for which 0.8 was assumed (Castro et al. 2008; Kersten & Piersma 1987; Zwarts & Wanink 1993).

5. We next obtained the total digestible energy content in each core sample by summing the digestible energy represented in the five benthic groups.

6. We converted the unit of total digestible energy from core sample surface area into per kilometre square.

7. Finally, we averaged the digestible energy per kilometre square among the 147 samples collected (17,070,463.64 kJ per km$^2$) and multiplied this by the total area of intertidal flat (4.79 km$^2$) to yield the total digestible energy available to bar-tailed godwits at Cattle Point, which amounted to 81,767,520.84 kJ.
Daily energy requirement in bar-tailed godwits

For this exercise we assumed the bar-tailed godwits in the ERMP Survey Area spend similar amount of time in this non-breeding area as their counterparts in New Zealand (i.e. 176 days; Conklin & Battley 2011a). In the final report we will estimate duration spent in the study area more directly, through analysis of count data. We moreover considered that fuel-deposition in preparation of migration, requiring elevated intake rates, takes place during the final one-third of their stay. We also assumed the birds largely operate under thermally neutral conditions (Kersten et al. 1998) and that their Daily Digestible Energy Requirements (DDER) are twice their Basal Metabolic Rate (BMR), increasing to 4–5 times BMR during the fuel deposition period (Piersma 2002). Body mass of bar-tailed godwits was obtained from averaging the weight data for birds of different ages and sexes and in different months as caught in southeast Australia (Higgins & Davies 1996). BMR was estimated using the allometric relationship between BMR and body mass as measured in non-breeding migratory shorebirds in Africa (Kersten et al. 1998). From these calculations, it followed that the average DDER of bar-tailed godwits was 450 kJ per day.

Carrying capacity

Given that the total digestible energy available to bar-tailed godwits at Cattle Point was 81,767,520.84 kJ, divided by the average DDER of 450 kJ/day, this gave 181,706 bird-days for Cattle Point during the non-breeding season (i.e. 1,032 individuals for the entire non-breeding season), which was substantially higher than the average count of bar-tailed godwits (205 ± 28, n = 4) in the nearest main high tide roost (S23.49278°, E150.86599°, surveyed on 8-Oct-14, 28-Nov-14, 21-Jan-15 and 18-Mar-15). However, it should be noted that the same food source considered in this worked example is also shared among other shorebird species and not solely to bar-tailed godwits. In fact the amount of food available to shorebirds based on our limited benthic samples was substantially lower than the other tidal flats around the world (Figure 6).
3.4.4. Discussion of results to date

The main purpose of this worked example is to demonstrate the calculation procedures involved in estimating carrying capacity. The result as such must be considered tentative only. The resulting estimate in the next season will be improved substantially by conducting a large-scale benthic sampling survey that is more representative of the food availability in the entire region than the pilot study conducted to date. The survey will also be conducted earlier in the non-breeding season to avoid potential depletion effects. The estimate of total amount of digestible energy will be improved by taking the critical prey densities into account and exclude non-profitable patches because shorebirds generally prefer more profitable patches and do not feed on everything on the intertidal flat (Goss-Custard et al. 2006). Moreover, in future we will also use our own benthic size-biomass relationships based on our own samples collected in the field rather than the ones now used from the literature.

Our pilot benthic sampling data might not be representative of the entire foraging range of the bar-tailed godwits, and the relative abundance of bar-tailed godwits in the shorebird community was lower in the Fitzroy Estuary than in other sub-regions of the study area (Figure 7). The same distribution pattern was also apparent in two other large-sized shorebird species (eastern curlews and whimbrels), which have much higher energetic requirements than small-sized shorebird species. In other words, there were hints that Cattle Point and Fitzroy Estuary as a whole may not be as attractive a region for large-sized shorebird species as other regions in the area.

**Figure 6** The biomass (ash free dry mass in grams per m²) of intertidal benthic organisms at Cattle point (in red) in comparison with those around the world (in black; Piersma et al. 1993).
In addition to our carrying capacity estimates, we will try to qualify the suitability of the various regions within the ERMP Survey Area for foraging shorebirds. If we assume that the consumption by shorebirds has no effect on the standing stock of benthos (i.e. that benthos production is sufficient to replace the prey lost to shorebird predation, at least over the term of a full year), we can generate maps showing digestible energy intake rates on different tidal flats (based on prey densities in different tidal flats and functional responses of shorebirds, Goss-Custard et al. 2006), and variation in tidal flat availability according to tide conditions (based on bathymetry map). With our calculation of DDER, we can then evaluate the minimum number of hours birds will have to forage to satisfy their DDER and how this varies over the non-breeding season.

3.4.5. Next steps in to finalise aim A3

Now that we have established a series of calculations that can turn our benthic sampling data into estimates of carrying capacity, two aspects remain. First, we will continue to refine and improve our method, in particular checking assumptions against published data, our own existing observations, and strategically targeted future observations. Second, we will apply this method to produce carrying capacity estimates for about one quarter of the study area (see Table 8) for a broad range of shorebird species.
3.5 Identify priority areas for management (Aim A4)

3.5.1 Summary
We have assembled the base mapping that will be used to delineate tidal flats (See section 3.2), and we have roost site data available from previous surveys (GHD 2011a, 2011b; Sandpiper Ecological Surveys 2012a; Wildlife Unlimited 2013, 2014). We are now awaiting benthic sampling and carrying capacity estimates that will be available in August 2016. This will enable us to identify priority tidal flats that have the potential to support large numbers of birds.
3.6 Gather data on local movements of shorebirds in the area (Aim B1)

3.6.1. Summary
Shorebirds are highly mobile, and so count data cannot be used alone to estimate how many birds are using an area, as they do not account for flux of birds over time as they move through the site. Moreover, the situation analysis completed by Driscoll (2013), together with our review of past data indicated that the study area has been too infrequently surveyed in the past to make a clear judgement about movements of shorebirds through the region, and hence existing data are too weak to estimate how many birds are using the area on migration and during over-summering. Bi-weekly or monthly counts during migration periods are needed to achieve this definitively for a site. To circumvent this, we have developed a novel approach to estimating the numbers of birds using the area by modelling the arrival and departure of migratory shorebirds along areas of eastern Australia for which there are sufficient data, and have applied these models to count data from the ERMP Survey Area. These models are performing well, and we have already produced quantitative estimates of the numbers of birds using the area during northward migration. The final step is to complete this picture with an analysis of the southward migration during spring 2015, and then this aim will be complete.

3.6.2. Previous data on bird movements from the ERMP Survey Area
No banding, flagging or telemetry activity has previously occurred in the ERMP Survey Area. Resighting activity has been noted in previous surveys (Wildlife Unlimited 2014) but only a few resights have been recorded due to the time constraints in those surveys, and the low numbers of marked birds at large in the field and available for resightings. During our first field season, 51 resight records were made. These involved birds originally banded in Alaska, Yalu Jiang and Chongming Dongtan in China, Hokkaido in Japan, Sakhalin Island in Russia, Northwest Australia, Victoria and other parts of Queensland. We also managed to capture and band 45 individuals and put engraved flags on 31 of them. Ten of these individuals (bar-tailed godwits, great knots and pied oystercatchers) were seen again in the field, mostly near at the capture site but two great knots were seen to have moved 10 km away from the capture site. Nonetheless, there have been no records of movement across the five subregions so far. These findings provide important information to help us design our radio tracking study, especially in terms of where to set up the automatic radio receivers.

3.6.3. Background to understanding bird movements
Across this project we are taking four complementary approaches to estimate the size of the potentially impacted migratory shorebird population in the ERMP Survey Area. The first is to assess the importance of the region as stopover site for migratory shorebirds by reviewing existing count data (including the literature review already completed by Driscoll 2013). The
second is to collate resightings data for all individually flagged birds that spend the non-breeding season in the region (See Aim B2). The third is to monitor shorebirds passing through the region during at least one southward (to be completed) migration and at least one northward migration (completed) using detailed counts (see Aim B3). The fourth is to use an automatic radio-telemetry array to track the individual movements of foraging birds (great knot and bar-tailed godwit) across the region during the mid-summer period when the non-breeding population is stable (see Aim B4). We have thus far reviewed the existing count data, monitored the shorebirds passing through during northward migration and collated resighting data in the region.

In each of the last five Austral summers (October-February), the ERMP Survey Area supported at least 11,000 migratory shorebirds (GHD 2011a, 2011b; Sandpiper Ecological Surveys 2012a; Wildlife Unlimited 2013, 2014), dominated by the bar-tailed godwits, red-necked stints, whimbrels, eastern curlews, lesser sand plovers, Terek sandpipers, grey-tailed tattlers, great knots, greater sand plovers and grey plovers. These seven species, together with the broad-billed sandpiper counts in March 2015, exceed 1% of the total flyway population estimates (Bamford et al. 2008), indicating the international importance of the ERMP Survey Area in the conservation of these species. Based on the counts conducted in January or February between 2011 and 2014, the north Curtis Island and Fitzroy Estuary supported the largest number of shorebirds, followed by Colosseum Inlet, Port Curtis and Rodds Peninsula (Figure 7). Migratory shorebird species richness was 20, 20, 19, 19 and 15, respectively among these regions. The species composition was similar in most regions, dominated by bar-tailed godwits and whimbrels. However, the composition in the Fitzroy Estuary was substantially different to other regions, favoured by small-sized shorebird species such as red-necked stints (61%), broad-billed sandpipers (71%), curlew sandpipers (68%) and sharp-tailed sandpipers (47%) (Figure 7, Table 9).
Table 9  The distribution of each migratory shorebird species in the ERMP Survey Area, represented as the percentage of occurrence per region based on the averaged counts from surveys conducted in January or February between 2011 and 2014 (GHD 2011a, 2011b; Sandpiper Ecological Surveys 2012a; Wildlife Unlimited 2013, 2014).

<table>
<thead>
<tr>
<th>Species</th>
<th>Fitzroy Estuary</th>
<th>North Coast</th>
<th>Port Curtis</th>
<th>Mundoolin / Colosseum</th>
<th>Rodds Peninsula</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bar-tailed godwit <em>Limosa lapponica</em></td>
<td>15</td>
<td>32</td>
<td>21</td>
<td>21</td>
<td>11</td>
</tr>
<tr>
<td>Black-tailed godwit <em>Limosa limosa</em></td>
<td>67</td>
<td>33</td>
<td>0</td>
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<td>0</td>
</tr>
<tr>
<td>Broad-billed sandpiper <em>Limicola falcinellus</em></td>
<td>71</td>
<td>26</td>
<td>0</td>
<td>3</td>
<td>0</td>
</tr>
<tr>
<td>Common greenshank <em>Tringa nebularia</em></td>
<td>11</td>
<td>19</td>
<td>23</td>
<td>47</td>
<td>0</td>
</tr>
<tr>
<td>Common sandpiper <em>Actitis hypoleucos</em></td>
<td>33</td>
<td>17</td>
<td>33</td>
<td>0</td>
<td>17</td>
</tr>
<tr>
<td>Curlew sandpiper <em>Calidris ferruginea</em></td>
<td>68</td>
<td>29</td>
<td>0</td>
<td>3</td>
<td>0</td>
</tr>
<tr>
<td>Eastern curlew <em>Numenius madagascariensis</em></td>
<td>8</td>
<td>29</td>
<td>28</td>
<td>24</td>
<td>11</td>
</tr>
<tr>
<td>Great knot <em>Calidris tenuirostris</em></td>
<td>4</td>
<td>59</td>
<td>8</td>
<td>22</td>
<td>8</td>
</tr>
<tr>
<td>Greater sand plover <em>Charadrius leschenaultii</em></td>
<td>47</td>
<td>44</td>
<td>1</td>
<td>6</td>
<td>3</td>
</tr>
<tr>
<td>Grey plover <em>Pluvialis squatarola</em></td>
<td>31</td>
<td>33</td>
<td>2</td>
<td>29</td>
<td>5</td>
</tr>
<tr>
<td>Grey-tailed tattler <em>Tringa brevipes</em></td>
<td>13</td>
<td>26</td>
<td>30</td>
<td>21</td>
<td>11</td>
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<tr>
<td>Latham’s snipe <em>Gallinago hardwickii</em></td>
<td>0</td>
<td>0</td>
<td>100</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Lesser sand plover <em>Charadrius mongolus</em></td>
<td>24</td>
<td>52</td>
<td>10</td>
<td>9</td>
<td>5</td>
</tr>
<tr>
<td>Marsh sandpiper <em>Tringa stagnatilis</em></td>
<td>14</td>
<td>0</td>
<td>29</td>
<td>57</td>
<td>0</td>
</tr>
<tr>
<td>Pacific golden plover <em>Pluvialis fulva</em></td>
<td>43</td>
<td>23</td>
<td>13</td>
<td>21</td>
<td>0</td>
</tr>
<tr>
<td>Red knot <em>Calidris rufa</em></td>
<td>8</td>
<td>65</td>
<td>11</td>
<td>13</td>
<td>3</td>
</tr>
<tr>
<td>Red-necked stint <em>Calidris ruficollis</em></td>
<td>61</td>
<td>26</td>
<td>2</td>
<td>8</td>
<td>2</td>
</tr>
<tr>
<td>Ruddy turnstone <em>Arenaria interpres</em></td>
<td>1</td>
<td>17</td>
<td>8</td>
<td>53</td>
<td>21</td>
</tr>
<tr>
<td>Sanderling <em>Calidris alba</em></td>
<td>0</td>
<td>94</td>
<td>0</td>
<td>6</td>
<td>0</td>
</tr>
<tr>
<td>Sharp-tailed sandpiper <em>Calidris acuminata</em></td>
<td>47</td>
<td>4</td>
<td>35</td>
<td>1</td>
<td>13</td>
</tr>
<tr>
<td>Terek sandpiper <em>Xenus cinereus</em></td>
<td>26</td>
<td>11</td>
<td>22</td>
<td>20</td>
<td>21</td>
</tr>
<tr>
<td>Whimbrel <em>Numenius phaeopus</em></td>
<td>5</td>
<td>53</td>
<td>21</td>
<td>16</td>
<td>4</td>
</tr>
</tbody>
</table>
Migratory Shorebird Monitoring: Understanding Ecological Impact

Figure 7 Shorebird abundance and species composition in five different regions of the ERMP Survey Area, based on averaged counts from surveys conducted in January or February between 2011 and 2014 (GHD 2011a, 2011b; Sandpiper Ecological Surveys 2012a; Wildlife Unlimited 2013, 2014).

The major difficulties in estimating the number of migratory shorebirds in ERMP Survey Area using peak counts and relative distributions are: 1) overlooking the number of transiting birds due to the incomplete temporal coverage during the migration period; 2) underestimating the total number of birds using the area if some individuals in the population stop briefly leave the region before all the non-breeding population arrive. This first problem could be illustrated by our records of at least 140 Red Knots in mid-October 2014 and 382 broad-billed sandpipers in late March 2015. Both records exceeded the highest count recorded from previous surveys (86 and 32, respectively), even though we only counted 19 out of the possible 151 high tide roosts (Wildlife Unlimited 2015).

Recent attempts to overcome the second problem have taken detection probability, sampled proportion of study area, length of stay or residence probability into account when estimating the number of birds transiting (Cohen et al. 2010; Farmer & Durbian 2006).
However, a reliable estimate of the length of stay or residence probability often requires radio tracking, unbiased capture-recapture, or resightings of individual birds (Cohen et al. 2010; Farmer & Durbian 2006; Frederiksen et al. 2001; Matechou et al. 2013), and none of this information is available for the shorebirds in the ERMP Survey Area since very little has been done in the region before. Moreover, some of these modelling estimates tend to be less reliable with small sample sizes (Frederiksen et al. 2001) and the ways to estimate stopover duration remain debatable (Efford 2005; Pradel et al. 2005).

We now describe a method we have developed to use count data to estimate total numbers of birds using the study area. The method relies on understanding the migration of shorebirds to and from the ERMP Survey Area, and is calibrated with data on migration phenology from across the East Asian – Australasian Flyway. It is only by understanding how the ERMP Survey Area fits into the migration cycle of the birds that we are able to estimate the total numbers of birds using the ERMP Survey Area, which is critical for understanding ecological impacts.

3.6.4. Methods
We began by estimating passage dates and total number of migrants transiting an area (Thompson 1993). This is a simple modelling approach that generates estimates by assuming normally distributed arrival and departure times. Although this is a relatively coarse approach to determine passage times, it has so far generated estimates that correspond well to results from direct tracking of individual birds (Choi et al. 2015; Rogers et al. 2010; Thompson 1993). The passage date parameters generated can then be used to: 1) provide key parameters to model the total number of birds using the ERMP Survey Area; 2) foresee the important period for shorebird survey so transiting species will not be missed; 3) provide passage dates in species that could be used for radio-tracking study in the second field season, minimising the risk of tagging transiting individuals that do not stay in the study area.

We started with investigating the migration phenology of shorebirds during southward migration, using count data from 15 key stopping and non-breeding sites along the East Asian-Australasian Flyway where shorebird counts have been carried out at least once a month. Two of these sites were on the east coast of Asia, and numerous resightings of Australian-flagged shorebirds at these sites (Anon 2011; Minton et al. 2011b; Riegen et al. 2014) indicate strong migratory connectivity with Australia, where the remainder of our study sites were located. The count data used in our phenology investigation were collected from a series of 15 regularly monitored shorebird areas from the northern Yellow Sea to southern Australia (Figure 8, Table 10).
Detailed descriptions of shorebird usage and roosts are available for most of the sites:

- Yalu Jiang coastal wetland in northern China (Choi et al. 2015),
- Mai Po Inner Deep Bay Ramsar Site in Hong Kong (Anon 2011),
- northern beaches of Roebuck Bay in north-western Australia (Rogers et al. 2006c; Rogers et al. 2011; Rogers et al. 2006d),
- Lee Point to Buffalo Creek in Darwin NT (Lilleyman et al. in review),
- Cairns, Mackay, Bundaberg, Great Sandy Strait, northern Moreton Bay (Pumicestone Passage), central Moreton Bay, southern Moreton Bay, Tweed River (Milton & Driscoll 2006), Hunter Estuary (Spencer 2010),
- Botany Bay and the Western Treatment Plant (Rogers et al. 2013).

All the sites are monitored by experienced local observers, and shapefiles describing the exact areas monitored in Australia are held on the Shorebirds 2020 database at Birdlife Australia.
Figure 8 The fifteen areas used to parameterise a model of how migratory shorebirds flow through the ERMP Survey Area: YLJ, Yalu Jiang coastal wetland; HK, Mai Po Inner Deep Bay; NWA, northwest Australia; DA, Darwin; CA, Cairns; MA, Mackay; BU, Bundaberg; GS, Great Sandy Strait; MBPP, northern Moreton Bay (Pumicestone Passage); MBCE, central Moreton Bay; MBSO, southern Moreton Bay; TR, Tweed River; HE, Hunter Estuary; BB Botany Bay; WTP, Western Treatment Plant (Base map © Esri, DeLorme, NAVTEQ 2015).

Following Clemens et al. (2014), each of the routinely counted locations within a ‘shorebird site’ is termed a ‘count area’ – usually a specific shorebird roost with clearly defined natural boundaries. A ‘shorebird site’ comprises contiguous and non-contiguous ‘count areas’ with frequent interchange of birds. Most of the shorebird sites considered in our study comprised more than one count area and the number of each species counted in these count areas were summed to yield the total count for each species in each shorebird site per survey. Error through double-counting or overlooking birds was reduced by surveying count areas at a consistent survey time relative to high tide, and by minimising the elapsed time between counts in adjacent count areas. In general each survey was completed in a single day or on consecutive days. The longest survey was five days, but that rarely occurred. If more than one day was used to complete the survey in a shorebird site, then the mean date was used for analysis. In cases when more than one count was made in the same count area within the single survey period, either an average or the maximum was taken for every species counted, depending on the likelihood of double-counting. We also standardised the count effort in each shorebird site by selecting count areas that were consistently counted throughout the multi-year study period.

As we were specifically interested in southward migration, we truncated the datasets to the months between July and December with two exceptions. In New South Wales and Victoria we worked on data collected from July-January, as inspection of scatterplots indicated that shorebird numbers often peaked in January. From Yalu Jiang we included counts in June as this is the northernmost site, with returning birds known to be present in this month.

Data were analysed using Thompson’s modelling approach (Thompson 1993) to understand migration phenology. This approach generates passage date and transiting population size estimates using repeated counts by assuming normally distributed arrival and departure times. We first examined the count data with scatter plots to determine the most appropriate model to use, depending mostly on: 1) whether the species was stopover or spending the entire non-breeding season at the shorebird site; 2) if there were individuals that did not migrate and were already present at the shorebird site. This situation often arose in species with delayed maturity (Rogers et al. 2006b), in which immatures do not migrate north until they are 2-4 years old or undergo partial migration and spend the breeding season at a stopover site.
All models were calibrated using the non-linear modelling procedure in SYSTAT 12 (Systat Software Inc. 2007) with a least-squares loss function and more importance was given to higher counts. Before analysis, calendar days were transformed into the number of days since 1-June. Starting values of the parameters to be estimated are needed for the calibrations and these values were estimated based on the count data.

A range of starting values was then used to check how robust our estimates were and only robust results were presented. The quality of the estimate was also evaluated based on the R-square, asymptotic standard error and the test statistic of the parameter estimates. A significance level of 80% was used in the test statistic of the parameter estimates due to the small samples and uncertainty in the precision of count data. Occasionally, the models did not converge or the parameter estimates were not significantly different from zero, implying a poor fit of the model to the data. To overcome such problem, we:

I. tried different models,
II. aggregated multi-year data before analysis,
III. excluded years with a low observed count because these years were likely to yield nonsignificant estimates, or
IV. hard-wired count estimates before generating the passage date estimates.

If the model still did not converge or the parameter estimates remained non-significantly different from zero, then the results were not presented. Finally, the accuracy of estimated passage dates was evaluated by comparison with known passage dates based on satellite tracking and geo-locator studies on the same populations using z-test (Zar 1999) and a significance level (α) of 0.05.

The models that were used in this study included the following: in cases where birds arrived and left the shorebird site within the study period, we calibrated the following model:

\[
\text{Count}_{j,\text{day}} = \sum_{j=1}^{\infty} \left( a_j \cdot n_j \right) \cdot \left( ZC \left( \text{day}, m1, s1 \right) - ZCF \left( \text{day}, m2, s2 \right) \right)
\]

Equation 1

where \( j \) is the year index: 1 for year 1, 2 for year 2, 3 for year 3 etc.,
\( \text{Count}_{j,\text{day}} \) is the observed number of birds present on the indicated day in year \( j \),
a\(_j\) is a dummy variable set to 1 for observations in year \( j \), and 0 otherwise,
\( n_j \) is the estimated size of the transiting population in year \( j \),
m\(_1\), s\(_1\) are the estimated mean and standard deviation of arrival dates,
m\(_2\), s\(_2\) are the estimated mean and standard deviation of departure dates,
ZCF(day, m, s) is the cumulative normal distribution for a mean of m and a standard deviation of s.

In cases where birds arrived and remained in the shorebird site over the study period, the following model applies.

$$\text{Count}_{j, \text{day}} = \sum_{j=1}^{\infty} (a_j n_j) \times (ZCF(\text{day}, m1, s1))$$

Equation 2

In single years or aggregated cases where some birds did not participate in the previous northward migration, and birds arrived and remained in the shorebird site over the study period, the following model applies:

$$\text{Count}_{\text{day}} = R + n \times (ZCF(\text{day}, m1, s1))$$

Equation 3

where R is the number of birds which did not participate in the previous northward migration.

The mean arrival date estimates generated from the modelling approach described above were used to examine if mean arrival date increase with latitude within Australia, which indicates that birds may “hop” (Piersma 1987) from the north to south of Australia. Weight data used in these regressions were means in the non-breeding season (using data specifically from January when these were available) from Marchant & Higgins (1993) and Higgins & Davies (1996).

3.6.5. Results
Historical shorebird survey data from 15 shorebird sites were analysed (Table 10). The comparison of passage dates among these areas in different latitudes indicated overlap between departure dates from the northern Yellow Sea and arrival dates in Australia for eastern curlew, *menzibieri* bar-tailed godwit, whimbrel and great knot (Figure 9), implying direct flights or rapid movement between continents. On the other hand, there was no overlap between the departure dates of broad-billed sandpiper from the northern Yellow Sea and its arrival in Australia, indicating an indirect flight between continents. Several other smaller species for which there were no departure data from the Yellow Sea arrived in Australia at a similar time to broad-billed sandpiper (Figure 9).
Table 10 Survey areas, monitoring periods and frequencies

<table>
<thead>
<tr>
<th>Site</th>
<th>Country</th>
<th>GPS</th>
<th>Years</th>
<th>Survey interval</th>
<th># surveys / season</th>
<th>Data source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Yalu Jiang coastal wetland</td>
<td>China</td>
<td>39.816° N, 123.948° E</td>
<td>2012</td>
<td>Fortnightly during migration period (June-October)</td>
<td>12</td>
<td>Q. -Q. Bai</td>
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<tr>
<td>Mai Po Inner Deep Bay Ramsar Site</td>
<td>China</td>
<td>22.489° N, 114.029° E</td>
<td>2007-2012</td>
<td>Weekly during migration period (July-October)</td>
<td>18</td>
<td>Hong Kong Bird Watching Society</td>
</tr>
<tr>
<td>Northwest Australia, Western Australia</td>
<td>Australia</td>
<td>17.981° S, 122.337° E</td>
<td>2007</td>
<td>Weekly during migration period (August-September)</td>
<td>11</td>
<td>K.G. and D.I. Rogers</td>
</tr>
<tr>
<td>Mackay, Queensland</td>
<td>Australia</td>
<td>20.998° S, 149.057° E</td>
<td>2007-2012</td>
<td>Mainly monthly</td>
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<td>QWSG</td>
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<td>Bundaberg, Queensland</td>
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Overlapping departure dates from Southeast Asia (Hong Kong) and arrival dates in Australia were found in all populations that may travel between the two locations, namely the whimbrel, greater sand plover, marsh sandpiper, curlew sandpiper and red-necked stint, indicating likely direct flights between the areas.
Meanwhile, overlap of arrival dates in Southeast Asia and Australia was found in *menzbieri* bar-tailed godwit, black-tailed godwit and great knot, indicating birds may arrive in these regions at similar times (Figure 9).

After arriving in Australia, there was a gradual delay in arrival date in areas further south for red knot, curlew sandpiper, sharp-tailed sandpiper and red-necked stint, suggesting that these species might make stopovers before arriving in the more southerly areas. This was confirmed by regression analysis of mean arrival date against latitude within Australia, which showed a significant positive relationship in small-sized shorebirds (pooling species weighing ≤ 130 g; $p = 0.002$, adjusted $R^2 = 0.19$, 95% C.I. = 0.18 to 0.67). In contrast, eastern curlew, bar-tailed godwit, whimbrel, black-tailed godwit and great knot arrived in different regions of Australia on similar dates, suggesting that they mostly fly directly to their non-breeding ground instead of making a series of short flights within the continent (Figure 9). There was no significant relationship between mean arrival date and latitude in large-sized shorebirds (pooling species weighing > 130 g; $p = 0.13$, adjusted $R^2 = 0.05$, 95% C.I. = -0.09 to 0.6).

**Figure 9** Estimated passage dates of adult shorebirds during southward migration for 19 shorebird species at 15 shorebird sites from China to Australia.
With the exception of Northwest Australia (NWA), locations are arranged in N-S order (see caption to Figure 8 for details of site abbreviations). Values for each site are the modelled mean and standard deviation of date of passage. Species are arranged in order of decreasing mass from the top left to bottom right (based on Marchant and Higgins (1993) and Higgins and Davies (1996)). The significant terms refer to whether the parameters estimated from the model were significantly different from zero. Non-significant results imply high uncertainties in the parameter estimates. Bar-tailed godwit subspecies *menzbieri* occurred in YLJ, HK and NWA while those along the east coast of Australia were mainly *baueri* (Battley et al. 2012; Wilson et al. 2007). Detailed results of the parameter estimates can be found in Table 11 for selected species relevant to the ERMP Survey Area.

Table 11 Comparison between southward migration passage dates estimated and those reported in tracking studies (presented as mean ± standard deviation). Locations within Australia were grouped into Northern Australia (NWA, DA and CA), Central East Coast (MA, BU, GS, MBPP, MBCE, MBSO and TR) and Southeast Australia (HE, BB and WTP). Tracking results were based on published satellite tracking and geolocator studies (Battley et al. 2012; Conklin & Battley 2011b; Driscoll & Mutsuyuki 2002; Gosbell et al. 2012; Minton et al. 2011a; Minton et al. 2013). Asterisk denotes significant difference between the passage date estimated and that recorded from tracking studies using a z-test. Non-significant estimates were excluded from analysis.

<table>
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<tr>
<th>Species</th>
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<th>Event</th>
<th>Thompson model estimate</th>
<th>Tracking observations</th>
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<tr>
<td>Bar-tailed godwit ssp. menzbieri</td>
<td>Yalu Jiang coastal wetland</td>
<td>Arrival</td>
<td>24-Jul ± 16.2</td>
<td>20-Jul ± 4.8 (n=8, Yellow Sea)</td>
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<td>Yalu Jiang coastal wetland</td>
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<td>2-Sep ± 0.5</td>
<td>30-Aug ± 5.0 (n=8, Yellow Sea)</td>
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<td>Arrival</td>
<td>9-Sep ± 16.1</td>
<td>10-Sep ± 12.8 (n=7, Northwest Australia)</td>
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<td>Ssp. baueri</td>
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<td>Arrival</td>
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<td>23-Sep ± 14.3 (n=24, New Zealand)</td>
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<td>16-Jun ± 4.7 (n=5, Yellow Sea)</td>
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<td>Departure</td>
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<td>Greater sandplover</td>
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<td>21-Aug ± 76.9 (NWA, DA)</td>
<td>15-Aug ± 12.1 (n=4 Northwest Australia)</td>
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<td>Grey-tailed curlew</td>
<td>Central East Coast</td>
<td>Arrival</td>
<td>1-Sep ± 21.4</td>
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Modelled passage dates were generally very similar to dates estimated from remotely-tracked birds, with no significant differences found in bar-tailed godwit, eastern curlew, grey-tailed tattler, greater sand plover and ruddy turnstone ($p > 0.05$ in 16 out of 17 cases; Table 12). The only case with a significant difference was the estimated departure date of red knot from Darwin (northern Australia), which was six weeks later than departures of tracked red knots from Papua New Guinea and the Gulf of Carpentaria (Table 12).
### Table 12

Estimates of passage dates around the flyway for seven shorebird species that use the ERMP Survey Area in internationally important numbers (excluding broad-billed sandpiper, greater sand plover and grey plover). Dates are represented as the number of days since 1st June. Single asterisk denotes significant different to zero at 0.2 level, double asterisks denotes significant to zero at 0.05 level in all parameter estimates, cases without asterisk denote non-significant different to zero. Model types refer to the models described in the method section, with R denoting non-migrating individuals, ‘agg’ denoting aggregated datasets and ‘multy’ denoting multi-year datasets. Locations are coded as follows: YLJ, Yalu Jiang coastal wetland; HK, Mai Po Inner Deep Bay; NWA, northwest Australia; DA, Darwin; CA, Cairns; MA, Mackay; BU, Bundaberg; GS, Great Sandy Strait; MBPP, northern Moreton Bay (Pumicestone Passage); MBCE, central Moreton Bay; MBSO, southern Moreton Bay; TR, Tweed River; HE, Hunter Estuary; BB Botany Bay; WTP, Western Treatment Plant.

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<th>Mean departure</th>
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### 3.6.6. Discussion of results to date

The passage dates estimated in this study correspond closely to tracking records, giving us confidence in the results of this first population-scale, cross-species comparison of movement patterns of shorebirds along the EAST ASIAN-AUSTRALASIAN FLYWAY. Our results indicate that southward migration strategies differ between species and body size might play an important role in the migration strategy used by shorebirds.

*Do shorebirds migrate directly from the Yellow Sea to the ERMP Survey Area on southward migration?*
A non-stop flight from the northern Yellow Sea to Australia would be expected to take some 4–6 days, given an average ground speed of \( \sim 50 \text{ km h}^{-1} \) and by comparison with observed duration of direct flights from the Yellow Sea to Australia by Eastern curlew and bar-tailed godwit ssp. *menzbieri* (Battley *et al.* 2012; Minton *et al.* 2013). This brief time lapse is consistent with the difference between departure times from the northern Yellow Sea, and arrival times in at least northern Australia, for a number of the larger species investigated in this study: eastern curlew, bar-tailed godwit, whimbrel, great knot and possibly grey plover and terek sandpiper, indicating that these species may fly directly from the north Yellow Sea to Southeast Asia, or even Australia, during southward migration.

It was not possible to estimate passage times for smaller species from Yalu Jiang in the northern Yellow Sea, because of their small numbers and potential complications from late-arriving juveniles. However it was noteworthy that for one small species, broad-billed sandpiper, arrivals in North-western Australia were almost two months later than departures from the northern Yellow Sea, strongly suggesting that there must be other stopover areas between the northern Yellow Sea and Australia. A known refuelling site for smaller shorebird species during southward migration is Chongming Dongtan in the southern Yellow Sea, where significant increases in body condition with time were found in most of the calidrid sandpipers (Choi *et al.* 2009).

*Do shorebirds stopover in northern Australia before moving on to the ERMP Survey Area?*

Our analysis approach can describe the average phenology of the population, but may not be suited to reveal a small proportion of birds that may have different phenologies. Given that northern Australian shorebird populations are much larger than those in southern Australia, we could have overlooked some stopover areas in northern Australia. Nevertheless, it was noteworthy that in several medium- to large-sized shorebird species, such as eastern curlew, whimbrel, bar-tailed godwit and great knot, arrival times were very similar in the north and south of Australia, suggesting direct flights to the non-breeding destination whether it was in north or south of Australia. In contrast, there was an apparent tendency for small-sized shorebird species to arrive earlier in northern Australia than in southern Australia, suggesting some stopover may occur in northern Australia.

*Are movements from northern to southern Australia made in a series of short ‘hops’ along the coast?*

Small-sized species such as red knot, curlew sandpiper, sharp-tailed sandpiper and red-necked stint may make several ‘short hops’ from northern Australia to southeast Australia, with arrival time significantly delayed in areas further southeast. The interpretation of gradual delay in arrival date estimates as ‘short hops’ was further supported by tracking study of Red Knots
that some individuals migrated south by stopping at more than one site in Papua New Guinea or northeast Australia (10–40 days at each stopover, P. Battley unpubl. data) before flying to New Zealand, which led to the gradual delay in arrival dates further south along the coast (Figure 9).

Our study indicated that the same shorebird site can serve different purposes for different species. Some of the shorebird sites in the north of Australia, such as Darwin NT, may be used as a stopover area for some shorebird populations while others treat it as the non-breeding area where they spend most of the austral summer. It seems that small-sized shorebird species make a series of stops on their way south and therefore need a series of wetlands to complete their migration journey. This is consistent with other areas along the East Asian-Australasian Flyway, such as Chongming Dongtan, which was dominated by small-sized refuelling calidrid sandpipers during southward migration (Choi et al. 2009).

Differentiating the different roles that different shorebird sites play for different shorebird species will be important when prioritising site-based management efforts.

Model validation
For several species, it was possible to assess the accuracy of our modelling estimates through comparison with independent studies in which satellite transmitters or geolocators provided empirical observations of timing of passage. In most cases the correspondence of count-based estimates and tracking studies was remarkably close; there were no significant differences in estimates of arrival time, and often the average arrival date determined by the two approaches were within 1-4 days of each other. 'Thompson models' thus appear to show great promise, especially as they can be applied to any species that can be counted regularly. An important assumption of Thompson models is that arrival dates and departure dates are normally distributed. While this seems plausible, the assumption has not been fully tested in shorebirds. It would be worthwhile for researchers reporting on tracking studies involving large numbers of individuals to assess whether their observations of arrival and departure date are normally distributed.

Our modelling results gave reliable estimates of passage days, with non-significant differences between our estimates and that recorded from tracking studies in 16 out of 17 cases. A high degree of agreement was found in cases such as the passage dates at Yalu Jiang and arrival date in northern Australia in bar-tailed godwit ssp. menzbieri, arrival date of eastern curlew in northern Australia, greater sand plover arriving in northwest Australia, and red knot arriving in northern and southeast Australia. The only discrepancy between our modelling estimate and tracking result was found in red knots departing from Darwin, which was perhaps complicated by the late-arriving juveniles. A substantial difference was also found in the arrival date of
eastern curlews in Yalu Jiang, in the northern Yellow Sea (p = 0.06). The five tagged individuals arrived and left 3–4 weeks earlier than the modelled estimates (Table 11). This difference might have arisen because all the tagged individuals were probable failed breeders (Driscoll & Mutsuyuki 2002; Gosbell et al. 2012). Our late passage date estimates for eastern curlew were unlikely to be affected by the late arrival of juveniles because juveniles comprised less than 1% of the stopover population, based on age-scans, each involved more than 1,000 individuals, from 4 different surveys (Q.-Q. Bai, unpubl. data). Given that eastern curlew is now listed as Critically Endangered in Australia, there is an urgent need to understand the cause of decline and monitor reproductive activities by documenting age proportions in different sites (Rogers et al. 2003).

In general, the modelled estimate for arrival dates matched tracking records more closely than departure dates did (Table 11). Departure dates were likely to be confounded by the late arrival of young birds, which were not taken into account in tracking studies. Moreover, the potential late arrival of juveniles may cause the departure of early-arriving adults undetectable in our modelling approach. This could partly explain the significantly earlier tracked departure date than modelled departure date for red knot in Darwin. On the other hand, the relatively poorer performance (more non-significant estimates) of our approach in grey plover (D. Rogers unpubl. data) and sharp-tailed sandpiper could be a result of differential migration in these species. Many juvenile sharp-tailed sandpipers are known to use a more easterly migration route than adults (Handel & Gill 2010; Lindstrom et al. 2011), potentially leading to a relatively high proportion of juveniles in the northeast Australia.

Annual variation in breeding success and perhaps availability of inland habitats for inland-favouring species also cause variation in the pattern of the number of birds through non-breeding season, yielding poor modelling estimates in some areas. The impact of late-arriving juveniles was best illustrated by great knots in the Yalu Jiang coastal wetland. The modelled results gave poor estimates of the number of birds and passage dates if both age-classes were combined for analysis. However, the modelled results improved substantially when the two age classes were analysed separately (Figure 10). Importantly, these models indicated that the total number of great knots transiting was almost twice as many as the highest count observed at any one time, as there were two peaks composed of different age-classes (Bai et al. 2012). The impact of different age-classes on modelled estimates was not as large in species such as bar-tailed godwit, eastern curlew and grey plover at Yalu Jiang coastal wetland because their populations were dominated by adults. In short, our modelling approach yielded reliable passage date estimates that were consistent with those recorded from tracking studies and it is important to differentiate the different age-classes wherever possible. In
addition to improving the modelling estimates for adults, documenting the migratory timing of juveniles would be of considerable interest.

Figure 10 The number of great knots stopped over at Yalu Jiang during southward migration in 2012. Black bars represent count data from surveys and the solid line the modelled estimate for adults while grey bars and dotted line for juvenile birds.

3.6.7. Next steps to finalise Aim B1

The amount of time that migratory shorebirds spend in non-breeding grounds such as Australia is generally known (Higgins & Davies 1996), but much more precise information at the species level on both arrivals and departures at all shorebird sites is needed to improve our understanding on the life cycle of individual species. This study has shown that reasonable estimates of arrival and departure dates at stopover sites, and arrival dates in non-breeding grounds, can be made using the results from regular counts conducted by the public.

The precise passage time estimates provided in this study could be used for population estimates in the studied or nearby areas (i.e. ERMP Survey Area) for the species that were modelled. They could for example be used to assess the proportion of the local population expected to be present at a site on a particular date, potentially allowing the peak number of birds in an area to be estimated from smaller number of counts in remote areas where regular counts are not possible. This would be of particular value in stopover sites where shorebird numbers peak briefly, and where the number of birds present at any one time may be considerably lower than the total number of birds that stopover locally on migration. Thompson models can be used to estimate the total number of staging birds in such situations.
There was evidence that red knots, grey-tailed tattler and sharp-tailed sandpiper may leave the ERMP Survey Area during non-breeding season. The results hinted that the end of September would be an important period to survey the transiting red knot population. The results also indicated that most of the large-sized species would arrive the ERMP Survey Area by October, while smaller species such as red-necked stints, may take longer and keep arriving until the end of October. The latter would mean care must be taken to minimise the chance of attaching radio-transmitters on transiting individuals.

We made a series of counts focused on the northward migration in the ERMP Survey Area in 2014/2015 (Figure 14), and will fit similar models using method described above to these data. Models described above will be fitted to the new data from the southward migration collected between September and November 2015. The northward migration data illustrate the variation in migration strategy through the area, with for example grey plovers departing relatively early, and red knot stopping at the ERMP Survey Area having spent the summer further south.
3.7 Discover how birds move around the study area (Aim B2)

3.7.1. Summary
We have finalised the design of the radio tracking study that will be carried out during summer 2015/16, and outline this below. Despite mounting a large field campaign to capture and mark birds during summer 2014/15, it has proven challenging reliably to catch birds using cannon-netting. However, we have banded 45 birds so far, and achieved 51 resightings of individually marked birds in the study area.

3.7.2. Background to bird movement study
Shorebird movements occur on a variety of spatial scales. On the basis of our previous studies, supplemented by our recent field experiences in the study area, we think shorebird movements are likely to be divisible into three categories, each with a different function. These categories have not been formally described previously, but we suspect they may be broadly applicable to coastal shorebirds. We refer to them as:

- “commuting” (see section 3.7.3 below). The minimal local movements made by shorebirds when they have established foraging areas.
- “exploratory” (see section 3.7.4 below). Non-migratory movements to locate locally rich foraging areas in dynamic coastal habitats.
- “migratory” (see section 3.6). Stopovers to rest and refuel in the course of ongoing migration to breeding or non-breeding grounds.

The varied scale of these movements, ranging from <10 km (commuting movements) to thousands of km (migration) poses challenges for shorebird biologists. The ideal tool for their study would be GPS tags that log spatial position regularly and transmit the data to biologists via satellite. While it is likely that such tools will be developed in the relatively near future, prototypes at present remain too heavy to be carried by most shorebird species, and too costly for this study.

As this approach is not yet feasible on shorebirds, we intend to assess the movement scales of the shorebirds of the ERMP Survey Area using a combination of automatic radio-telemetry and behavioural observations to inform on local movements, supplemented by analysis of shorebird count data and resightings of individually marked birds to inform on movements of larger scale.
3.7.3. **Measuring commuting movements in 2015/16**

'Commuting flights' include movements made by shorebirds between high tide roosts and low tide foraging areas on a daily basis. They also include regular movements made over two-week periods, as movement routines often vary over a tide cycle. During neap periods tidal flat exposure is restricted when the tide is low; during spring periods tidal flat exposure is much greater when the tide is low, but this may be offset by more restricted availability of roost sites when the tide is high (Rogers *et al.* 2006a,b). Moreover, movements can differ between day and night; several previous studies have reported shorebirds selecting different roost sites at night (Rogers 2006b), apparent reasons including nocturnal exploitation of dry roosts that have an inhospitably warm microclimate by day, and the reduction of risk of depredation at night when approaching predators are more difficult to detect. There have also been studies demonstrating differences in foraging site by night and by day (Sitters *et al.* 2001), including cases where foraging site choice of shorebirds was influenced by the presence of artificial lighting (Dwyer *et al.* 2013), a striking feature in some of the ERMP Survey Area.

In tidal systems where there are two low and two high tides per day, shorebirds make four commuting flights every 24 hours. The energetic costs of commuting can therefore be considerable in regions where foraging sites are far from suitable roosts; regular commuting flights of up to 30 km (i.e. 120 km per day) have been observed in some studies (e.g. Sanzenbacher & Haig 2002; Rogers 2006b). In the ERMP Survey Area, where roosts are relatively numerous and widespread, opportunistic behavioural observations made during the first season of fieldwork suggest commuting flights range from about 1–10 km.

The main tool we intend to use to document 'commuting' is radio-telemetry; we will attach VHF radio-transmitters to a sample of birds, and document their local movements with an array of automatic radio-receivers (ARTS; Figure 11, 12). These radio-transmitters have a detection range of ~0.5–3 km in open habitats (depending largely on the elevation of the receiving antenna). Data from the ARTS will be supplemented by handheld radio-telemetry and behavioural observations to build a picture of local movement routines according to tide, weather and time of day. They will also be helpful in assessing the amount of time spent foraging in different areas (thus supporting the carrying capacity component of this study).
Figure 11 Radio-tagged red-necked stint about to be released; the transmitter is concealed by plumage, but the antenna can be seen projecting beyond the tail.
Our original intention was to deploy approximately 50 'pip' transmitters in the study, each transmitting a signal on a different frequency. However, with our increased experience of the region we now believe that coded transmitters would be better suited to our study. Coded transmitters send a unique digital signal (greatly reducing the number of pulses that need to be logged to confirm identification of a signal).

**Figure 12** Example of an automatic radio-tracking station.
Coded transmitters can be set with a slower pulse interval than 'pip' transmitters, and therefore have a considerably longer battery life. Use of coded transmitters will allow us to track birds throughout the non-breeding season, rather than confining the radio-telemetry to an intensive 6–8 week study. The longer battery life will increase our chances of detecting relatively infrequent 'exploratory' movements (see section 3.7.4).

There are some disadvantages to coded transmitters. With their slower pulse intervals, they are not as well suited to handheld radio-telemetry as pip transmitters. They are also more costly. As a result we can only commit to deploying 30 coded transmitters in this study. Our target is to deploy 15 coded transmitters on bar-tailed godwit and great knot. These species are intensely gregarious, roosting in large mixed-species flocks at high tide, so it is likely that the radio-tagged birds will alert to us any key neap or night-tide roosts that may have escaped former detection. They do however forage in somewhat different areas, and we suspect this will drive some important differences in commuting routines. Both species can be difficult to catch and we have developed a list of fall back species that we can track instead, should we be unable to catch the required samples of knots and godwits.

We intend to deploy an array of nine automatic receivers in October 2015, shortly before a catching expedition in November 2015. Most of the receivers will initially be deployed at roosts and key foraging areas in the area around Pelican Banks, South Curtis Island and Facing Island, where we intend to make our cannon-net catches. Coded transmitters will be superglued to the trimmed rump feathers of the targeted species, and the birds will then be released. Their signals will be detected when they occur within range of an automatic receiver. We will probably attach more than one antenna to most receivers: an omnidirectional antenna that detects birds at relatively short range, and a larger, 6-element Yagi antenna that detects birds at longer range, but only in a limited bearing. In combination with signal strength (recorded automatically by the data-loggers) experiments of detection range of each receiver (made with test transmitters carried by researchers) this will give us a reasonably precise indication of the location of birds when signals are detected (see Figure 13 for examples from another study), and of their general movement routines.
Figure 13 Field plot of data logged by an array of automatic radio-receivers for great knot in Roebuck Bay, North-western Australia. We compiled and checked plots like this regularly for an ongoing understanding of local movements. In this plot, data is shown on the Y-axis, time of day on the X axis, and different symbols are used to denote detections by receiving stations in different locations. The wavy diagonal lines show the mean tide level – i.e. the demarcation between high and low tides. This particular individual usually foraged in the same location at low tide, whether it was day or night, with signals fading at the lowest point of the tide when if followed the sea-edge beyond the reach of the automatic receivers. The preferred night-time roost at high tide (red crosses) differed from its preferred daytime roost (green triangles). Signals stopped abruptly on 29 March when the bird departed on northward migration.

3.7.4. Measuring exploratory movements in 2015/16

Foraging patches in coastal regions are often impermanent, not least because shorebird predation can cause local depletion of prey abundance. In addition some prey resources are naturally temporary: for example immature bivalves favoured by red knots can occur in high local densities after a spat fall, but once the bivalves grow to full size they become too large for knots to swallow. Shorebirds appear to be remarkably skilled at locating new sites where prey is abundant, suggesting that they must explore to sample different areas and find patches with high prey abundance.
Relatively little is known about these exploratory flights, in part because they are likely to be 'medium-length' movements, too lengthy for ready detection in brief radio-tracking studies, but not long enough for confident identification in satellite-telemetry movements in which the geographical precision of 'fixes' is often rather coarse. There is some evidence that the range covered in shorebird explorations varies among species (Rehfisch et al. 1996, 2003) and even between individuals (Bijleveld et al. 2014).

In the ERMP Survey Area we plan to increase our probability of detecting exploratory flights by deploying coded transmitters, giving us a long field season in which we can detect rather infrequent exploratory flights. After resolving the general routine of commuting flights in the South Curtis Island region, we will redeploy some of the transmitters further afield, at roosts around Rodds Peninsula, and possibly also at selected sites in the Fitzroy Estuary. We will explore supplementing this work with chartered flights to search for 'missing' tags from the air; detection range is considerably increased by aerial survey, so this approach has a reasonable chance of success. We will also continue searches for individually-marked birds, with the intention of finding how often birds make movements beyond the local scale of commuting flights.
3.8  Describe the patterns of flow of birds into the study area (Aim B3)

3.8.1. Summary
As outlined elsewhere in this report (section 3.6), we have carried out analyses of count data from a series of sites along the eastern Australia coast to develop a regional understanding of probable shorebird passage in the ERMP Survey Area. We counted birds in the ERMP Survey Area during the northward migration in early 2015 (Figure 14), and will conduct southward migration counts in late 2015. We will formally analyse these data to determine overall numbers of birds using the study area (section 3.6), but will also describe the pattern of migration flow in relation to impacts, as per the scope of works.

![Figure 14](image)

**Figure 14** Counts of migratory shorebirds in the ERMP Survey Area in 2014/2015. The counts are focused on the northward migration period between March and May. The February count from Wildlife Unlimited (2015) will fill the gap for the non-breeding period.
3.9 Identify size of management units (Aim B4)

3.9.1. Summary
We have completed the base mapping on which these spatially explicit calculations will be based. Combined with resightings data, radio tracking data, and estimates of total numbers of birds using the sites, we will conduct desktop analysis to identify the appropriate size of management units for migratory shorebirds in the ERMP Survey Area.
4. TIMELINE

In Figure 15, we provide a high level timeline of the major tasks to be completed to finalise this project. We have scheduled fieldwork into the first part of summer 2015/16, which will minimise the probability of weather disruptions, which tend to occur more commonly after the new year.

![Timeline of major tasks remaining.](image-url)

**Figure 15** Timeline of major tasks remaining.
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