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Waterbird responses to flooding, stressors and threats

A review of the literature

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Foreword

The Murray-Darling Basin Environmental Water Knowledge and Research Project (MDB EWKR) requires a review of current knowledge during its first phase in order to inform future development of priority questions for research. Waterbirds are likely to be key targets or indicators for which priority questions are raised, and the review will need to identify the current level of understanding regarding outcomes for waterbirds expected from environmental flows. It will also need to identify the state of knowledge regarding how various other stressors affect waterbirds (e.g. pest species, habitat fragmentation or loss, pollution), and how these interact with the effects of flows. While the effects of flows on waterbirds have been reviewed in recent years with important knowledge gaps and research priorities identified, the role of other stressors is much-neglected in Australian literature and practice and a review of their effects and interactions is timely.



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Introduction

The Murray-Darling Basin Environmental Water Knowledge and Research Project (MDB EWKR) requires a review of current knowledge during its first phase in order to inform future development of priority questions for research. Waterbirds are likely to be key targets or indicators for which priority questions are raised.

This review summarises known waterbird responses to flows and flooding in Australia, with an emphasis on the Murray-Darling Basin, and synthesises knowledge gaps highlighted in the literature. It also reviews the international literature regarding how various other stressors and threats affect waterbirds (habitat loss, fragmentation and change, predation, climate change, pollution, disease, human disturbance, competition), and how these interact with the effects of flows.

The review was conducted in two main stages.

Part 1: Construction of a summary table of known and unknown relationships between waterbird responses and flows/flooding in Australia, with specific reference to the Murray-Darling Basin, based on previous reviews

Part 2: Synthesis of the international literature on waterbird responses to other stressors, including habitat loss, predation, climate change, pollution, disease, and human disturbance, the likely mechanisms behind these responses and interactions with responses to flows/flooding

Both stages endeavoured to establish:

a. What is known, with reference to the quality of the evidence and our present ability to develop a quantitative models, conceptual models, hypotheses supported by evidence, or speculative hypotheses
b. Knowledge gaps and suggestions for ways forward identified by previous reviews

This was not a formal 'systematic review' that exhaustively sourced and criticised all literature relative to each topic, however extensive literature searches were undertaken using Web of Science and Google Scholar, and relevant references were also sourced from previous reviews. Specifically, the method was as follows:

1. Create a conceptual model of response variables, drivers and other stressors (updated as needed).
2. Design MS Excel spreadsheet tables based on conceptual model variables. These tables contain the detailed results for Part 1 and should be referred to along with the current report.
3. Extract general statements of known relationships and knowledge gaps from previous reviews (reports and journal papers). Sort based on dominant drivers/explanatory variables.
4. Extract examples of known relationships re. responses to flows and flooding from previous reviews, the most relevant journal papers and reports referred to by reviews, and from journal papers published since 2010 (Google Scholar and Web of Science searches); file under applicable general statements and response variables; update general statements as necessary.
5. Assess availability and quality of empirical data and potential knowledge status with further research for general statements related to flows or flooding.
6. Extract known relationships and knowledge gaps re. responses to stressors and threats from relevant journal papers (full literature search; Google Scholar and Web of Science); sort under applicable general statements and response variables; update general statements as necessary.
7. Assess evidence quality, availability of empirical data, and potential knowledge status with further research for waterbird responses to stressors and threats.
8. Summarise interactions and their potential influence.

For Part I, assessment of the quality of the evidence was tabulated in the accompanying MS Excel spreadsheet using the following categories, some of which were requested by the MDFRC:

Empirical data:

- None
- Limited
- Moderate
- Good
- Extensive

Evidence quality (knowledge status):

- Quantitative - quantitative models
- Conceptual - evidence based conceptual models
- Established influence - good evidence to support relationship
- Likely influence - some evidence to support relationship
- Possible influence - evidence from other systems/theory

Potential knowledge status with further research

- Quantitative - develop quantitative models
- Conceptual - develop/improve conceptual models
- Interaction - develop knowledge of interaction among factors
- Established influence - evaluate relationship and significance
- Influence - identify if factor influences outcomes

Significance of flood frequency, dry spell duration, antecedent flows, flow event /area/extent/size, flow event duration, flow depth, flow connection, flow timing/season

- Highly significant
- Significant
- Significant in context
- Significant sometimes
- Not significant

Flow event type 1

- Natural
- Combination
- Managed
- Experimental
- Any

Flow event type 2

- Cease flow
- In-channel pulse
- Minor flood
- Major flood
- Any

For Part II, the quality of the evidence and availability of empirical data are discussed in the text.

Key messages

While increasing waterbird populations and maintaining waterbird diversity are important targets for environmental flows in Australia, knowledge gaps exist that affect our ability to manage and predict waterbird populations and diversity at appropriate scales. The largest of these gaps in knowledge are:

1. **Demographics**

- Survival and mortality rates, especially of fledglings and juveniles (and therefore recruitment)
- Population age structures and sex ratios
- Population and sub-population boundaries

2. **Movements**

- Immediately following and between breeding events – timing, distances travelled, differences between juveniles and adults, site fidelity, key foraging habitat locations and characteristics, effects of habitat availability, quality and productivity on bird condition and survival
- During breeding events – distances travelled, habitat characteristics, effects of habitat availability, quality and productivity on breeding site choice, site fidelity, event size and success
- Mechanisms, cues or drivers behind bird movements and choices and how these interact

3. **Effects of interactions between flow-related drivers of waterbird responses and other stressors, pressures or threats, especially:**

- Habitat loss, fragmentation and change
- Predation - rates, species, and timing
- Climate change and adverse or extreme weather

These knowledge gaps exist even for common and conspicuous taxa such as colonially-nesting waterbirds that are often thought to be relatively well-understood. They are particularly severe for cryptic and uncommon taxa. Filling these knowledge gaps will assist managers to:

- Identify, maintain, and/or restore key waterbird habitats – especially critical foraging habitats
- Better understand the spatial and temporal scales at which key habitat characteristics are required
- Better target water, vegetation and threat management actions to ensure 'event readiness' at nesting sites between flooding events and maximise waterbird recruitment
- Better predict the effects of water management and threats



Executive summary

This review summarises known waterbird responses to flows and flooding in Australia, with an emphasis on the Murray-Darling Basin, and synthesises knowledge gaps highlighted in the literature. It also reviews the international literature regarding how various other stressors and threats affect waterbirds (habitat loss, fragmentation and change, predation, climate change, pollution, disease, human disturbance, competition), and how these interact with the effects of flows.

While the effects of flows on waterbirds have been reviewed in recent years by various authors with important knowledge gaps and research priorities identified, the role of other stressors and threats is much-neglected in Australian literature.

Waterbird responses to flows and flooding

While the maintenance of waterbird populations and diversity are critical targets for environmental flows in Australia, there is a dearth of data on their demographics and movements, making modelling and prediction of the effects of expensive management actions and climate change difficult. Information regarding local abundance and aspects of breeding, especially nesting, is far more common and readily available than information regarding population parameters such as survival / mortality and movements associated with foraging and dispersal between breeding events. Data are especially scarce for juveniles, whose recruitment is critical for long term population persistence.

Waterbird abundance responses to flows are thought to be relatively well understood for ducks, cormorants, pelicans, and colonially-nesting waterbirds - especially at the local scale. However drivers of abundance of the more cryptic species are poorly understood, spatially or temporally, and our understanding for the more conspicuous groups is based on annual aerial surveys of only a portion of their ranges and limited small-scale local studies. Waterbird movement is an important complicating factor. Kingsford and Norman (2002) rated information on the abundance of species as 'good' in three of the more conspicuous waterbird orders (Anseriformes – ducks, geese and swans; Pelecaniformes – pelicans, cormorants etc.; and Ciconiiformes - bitterns, egrets, herons, ibis, storks) but 'poor' for all the others (e.g. Gruiformes - cranes, rails, crakes and gallinules), based on long-term surveys.

Population sizes and age structures for most species are effectively unknown, especially those that are cryptic and/or use habitats other than major wetlands. Species thought to be in low numbers and/or with restricted range are of most concern for conservation but there is usually poor information to determine whether these populations are increasing or decreasing. Other than gross reductions in total populations we have very little data on waterbird population functioning in general. Even basic biological data such as age at sexual maturity are poorly documented for many Australian waterbird species, although estimates are available based on overseas data and limited local studies (e.g. egrets and straw-necked ibis are estimated to reach reproductive maturity at 18-24 months). The time between successful breeding events and the size of those events are almost certainly critical in determining population age structure. Age structures are likely to change in association with boom and bust periods. They have also most likely changed with the increase in river regulation and subsequent reductions in breeding event frequency and size, perhaps for some species more than others. The impacts of loss of habitat on population sizes and age structures of both resident and migratory waterbird species is poorly understood.

In Australia, we still lack fundamental knowledge on the movements of most waterbird species, including an explicit understanding of the spatial and temporal scales at which individual species interact with their habitats and the triggers for movement - particularly for those using habitats in inland Australia. For the six orders of waterbirds considered by Kingsford and Norman (2002), knowledge of movements was categorised as 'moderate' for Anseriformes and Ciconiiformes, but 'poor' for the other orders. This remains

the case 12 years later. Although sufficent data on movements are lacking, it is believed that individuals of most species are capable of dispersing at the scale of the continent, and so populations of most species are thought not to be confined to the Basin. However we have limited understanding of whether waterbird populations in Australia function effectively as one large population, or whether there are subpopulations that function largely independently of each other. Questions that remain include: What are the wetland connectivity/network and associated movement requirements for juvenile dispersal and survival? How do juvenile birds interact with the landscape? Which sites and habitats are critical for their movements? What proportions or age groups of populations move long distances vs. locally? Do movement distances of young waterbirds differ from those of adults?

Initiation of breeding may not result in breeding success. However examination of threshold triggers for breeding initiation is much more common than quantitative assessment of breeding success of any kind, whether proportion of nests successful, fledging rate, or juvenile survival. While basic information about the reproductive ecology of many waterbird species is known, information is poor for Gruiformes (cranes, rails, crakes and gallinules) and Charadriiformes (sandpipers, stilts and terns), and detailed information is confined to only a few species (e.g. magpie goose, black swan, chestnut teal, Australian wood duck, some cormorants and egrets). Data on waterbird fledging rates and their drivers in Australia are scarce, with some exceptions for particular species or groups in certain locations such as egrets. These data can be difficult to collect in the field for logistical and ethical reasons, however detailed, long-term monitoring is essential if we are to build better predictive models that allow assessment of the potential quantity, timing and distribution of water required to ensure successful recruitment.

We currently know very little about survival rates in Australian waterbirds, or how important survival rates are to the maintenance of waterbird populations, or which wetland habitats play a key role for survival of waterbirds. Research needs to be established to quantify adult and juvenile survival and determine whether it is affected by the availability of suitable habitat between flooding events. Juvenile survival is not readily quantified from observations at local scales. Consequently modelling of population dynamics in response to flows is difficult, and reliant on estimates from overseas or different species. Some key questions include: Is recruitment occurring at a sufficient rate (if at all) to maintain populations, considering the relatively long lives of Australian waterbirds? Are any Australian waterbird species facing extinction debts? What are the main drivers of juvenile survival/mortality, and do these vary spatially or temporally? How important are small to medium sized floods for juvenile survival between the large breeding events triggered by large floods? How do juveniles use the landscape around them to move and feed? At what scales do they move and interact, and how do these affect juvenile survival? Can co-ordinated broad scale surveys that include marking and tracking of juvenile cohorts provide sufficient data to answer these questions?

Waterbird responses to stressors and threats

General characteristics of the literature

Following an extensive international literature search, of over 900 references identified that were directly relevant to waterbird responses to stressors and threats other than changes in flow/flood regime, only 40 (< 5%) were from Australia. Many of the latter were short anecdotal records in small bird club journals regarding one or two species or single events. Threats and stressors for waterbirds other than changes in flows/flooding are mentioned in some Australian reviews, but usually only briefly or as factors complicating results and requiring consideration. No comprehensive or integrative studies were located of major stressors and threats for Australia's waterbirds, or how these interact with each other or flows and flooding – except for migratory shorebirds whose populations are of international concern.

Habitat alteration (loss, fragmentation and change) and predation were by far the most common stressors in the literature reviewed (approximately 25% of references each), and were often discussed as interacting factors. The influence of climate change (and weather) was frequently mentioned but not often directly studied (approx. 14% of references). Pollution, disease, human disturbance and competition were all

regarded as significant threats to waterbirds, however were relatively less frequently encountered in the literature reviewed.

Breeding (especially nesting) was the most common response variable addressed by far (63%). Effects of stressors on foraging or feeding were at least mentioned in approximately 26% of references reviewed, and survival, mortality, or recruitment were mentioned in approximately 20% - however very few studies actually measured these variables. The term recruitment was rarely used in titles or abstracts (3% of references).

In terms of stressors and threats, relevant literature is relatively scant for Gruiformes (cranes, rails, crakes and gallinules; approximately 4% of literature reviewed), for Pelecaniiformes (pelicans and cormorants; 9% and for Ciconiiformes (bitterns, egrets, herons, ibis, storks; 29%). Anseriformes (waterfowl: ducks, geese and swans - especially migratory species) and Charadriiformes (especially migratory shorebirds: sandpipers, stilts and terns) have been far more commonly studied (46% and 44% respectively).

Here, key issues identified in the international literature are highlighted.

Specific drivers and responses

Habitat loss, fragmentation, and change are arguably the most important drivers of changes in waterbird abundance, population size and diversity worldwide. Most empirical studies of relationships between habitat and bird populations have focused on the numbers breeding in a particular locality. For practical reasons, these studies have often ignored the sometimes large numbers of non-breeders and have often not considered in much detail processes outside the breeding season such as foraging, dispersal, juvenile survival and adult survival and the habitats required for these. Modelling suggests that in long-lived species, survival of juveniles and adults is key to maintaining healthy populations (Arthur 2011; Sovada et al. 2001). To understand the behaviour of bird populations, and to predict how they might respond to major habitat loss and change, data and models are needed that cover large geographic areas and incorporate both breeding and non-breeding individuals and all critical life stages and habitats (Goss-Custard et al. 1995).

- Waterbird species that are highly mobile may be affected by habitat loss occurring long distances from their breeding sites at non-breeding, foraging or refuge sites (Dorfman and Kingsford 2001). Processes in one place influence numbers in another, at the same and at different times of year (Goss-Custard et al. 1995a; b; Goss-Custard et al. 2006). As non-breeding (foraging) habitat is lost, the increasing competition for the feeding areas that remain can increase mortality rates and thus lower population size. Conversely, the resulting reduced levels of competition for territories in the breeding season may raise reproductive rates and increase population size (Goss-Custard et al. 2006). The distance between foraging habitats and nesting habitats can also influence nesting success, with increased distances and reductions in the area, availability or quality of nearby foraging habitat resulting in decreased success rates (Van de Pol 2010; Zharikov and Milton 2009).
- Habitat loss can often be more about site condition, quality and characteristics than clearing or fragmentation. Habitats used by waterbirds vary in quality as a function of interactions between water regime, vegetation type, food abundance, predation danger and competition. Changes in habitat characteristics will often favour some species and individuals while disadvantaging others, leading to changes in both abundance and diversity. Habitat loss, fragmentation and change can cause food shortages and subsequent starvation of nestlings, fledglings and juveniles. Invasive plant species have the potential to dramatically alter the availability of waterbird foraging and breeding habitats. Water and land management actions to promote or protect agriculture, industry and residential areas in floodplains and wetlands often conflict with waterbird use of breeding and foraging habitats provided by floodwater. Grazing of waterbird habitats can have negative or positive effects depending on species, location and timing. Since individual species have different habitat preferences (e.g. reed-beds and other wetland herbaceous vegetation at the local scale for rail species), there is an obvious risk of habitat loss and tradeoffs at the local scale.
- Landscape scale changes in habitat configuration can also affect species occupancy and abundance. Habitat type, area, and configuration at landscape scales can influence predation rates on nests and young and consequently nest success and fledging rates. Loss in area or fragmentation of wetlands can concentrate waterbirds and predators in the remaining patches of suitable habitat and reduce the abundance of alternative prey species (Cowardin et al. 1983, Sovada et al. 2001).
- Worldwide, there has been a documented increase in the use of agricultural lands and water impoundments for foraging by waterbirds as their natural foraging habitats are lost. This can result in conflict with farmers,

depending on the foraging species, and result in expansion of some populations and contraction of others. Habitat losses - and in particular loss of foraging habitat or declines in natural food availability - have been associated with diet switches in some species from their natural foods in natural foraging locations to new foods in agricultural foraging locations (e.g. crops). This can have negative effects for waterbirds that no longer receive sufficient variety or nutrition in their diets (Jefferies and Drent 2006).

Predation is a natural component of waterbird population biology. However anthropogenic changes, in particular the introduction of feral predators and habitat alteration, have changed the nature and importance of its impact on waterbird populations (Sovada et al. 2001). Changes in predation may cause prey populations to decline to extinction, or to stabilise at lower levels where they may be more susceptible to extinction from other causes (MacDonald and Bolton 2008). Populations that have declined due to other causes are also more likely to suffer more from predation. Usually more than one factor is involved when predation negatively impacts waterbird populations (Sovada et al. 2001).

- Many studies have shown that predation on waterbirds occurs mainly during nesting, and is dominated by egg predation. Predators also take nestlings or fledglings, or scavenge those that die due to other causes such as starvation. Predation on adult waterbirds is relatively rare, but is probably additive to mortality due to other factors (e.g. hunting, pollution; Sovada et al. 2001 citing King and Derksen 1986, Raveling 1989, Sargeant and Raveling 1992). Predators can reduce the survival of waterbirds and consequently population size either through direct predation, or indirectly, by causing adults to desert their nests or foraging sites, competing for habitat or food, or affecting other predators and prey (Cruz et al. 2013; MacDonald and Bolton 2008; Skorka et al. 2014).
- Adult waterbirds are most commonly preyed upon by avian and mammalian predators. Introduced predators, particularly mammals, may cause local extinction of breeding populations.
- Fluctuations in predation pressure on waterbirds may result from fluctuations in the availability of other prey. The relative importance of waterbirds in predator diets changes with fluctuations in their availability, however some species and individuals can be selective in terms of which species, ages, and sizes they kill, and in which habitats or locations.
- Nest predation is well known to be of great importance to birds, not only in terms of population regulation, but also in nest placement and distribution (McKilligan 1987) and nesting density (Sugden & Beyrsbergen 1986; Dorfman and Read 1996). Most nest predation involves eggs rather than chicks (though the latter is rarely measured). Consequently it can be difficult to ascribe changes in populations definitively to changes in nest predation, as other parameters can have a large influence. Factors that may influence rates of predation on nests include time of season, habitat type, habitat management, nest concealment, nest density, distance to habitat features used by predators (e.g. perches, nests, dens, shelter), and of course, predator abundance. There is conflicting evidence for most of these, and it is likely that the influence of each on rates of predation is dependent on location and species (MacDonald and Bolton 2008; Sutherland et al. 2012).
- Waterbird eggs and chicks may be preyed upon by avian, mammalian, or reptilian predators – and occasionally fish. Raptors (e.g. eagles, falcons) and corvid species (e.g. ravens, crows) are major predators at waterbird nests worldwide and are regularly associated with waterbird nesting colonies. Mammals are also critical and often more cryptic predators of nests.
- A change in predation is often one of the main mechanisms via which land use or habitat change affect waterbird breeding success (Douglas et al. 2014).
- Water depth under or around waterbird nests significantly affects predation rates and consequently nest success. Colony size, nest location, nest dispersion and nest density within the colony can all affect predation rates. Usually, predation rates are higher in smaller colonies and at the periphery of colonies (Angulo-Gastelum et al. 2011).
- Waterbird responses to predation are often species-specific. Many species have behavioural adaptations to predation, and some can respond flexibly to different situations.

Changes in temperatures, the timing, intensity, amount and duration of precipitation, and the frequency and severity of extreme weather events all have the potential to influence waterbird populations positively and negatively, and directly and indirectly (Sutherland et al. 2012). Climate change already appears to be affecting population movements such as migration and dispersal, as well as breeding initiation and success and survival between breeding events. Chambers et al. (2005) reviewed the effects of climate change on birds that have been documented or predicted, with particular reference to Australian species. Potential impacts included changes in geographic range, movement patterns, morphology, physiology, abundance,

phenology and community composition. They found that the evidence suggests that these changes are already happening, both overseas and in Australia, but more research is needed to determine their extent and how to conserve birds in the face of climate change. Relative to the northern hemisphere, little is known about the effect of climate change on southern hemisphere birds, although the impact could be significant. Management options suggested included promoting adaptation and resilience, intensive management of sensitive species, and improved planning for mitigation techniques and monitoring.

- Making predictions about how populations might respond to future climate change depends on an adequate understanding of important ecological processes and their interactions at appropriate spatial and temporal scales (Norris et al. 2004). In many cases, the effects of climate change will be manifest through indirect effects on land-use change, prey availability, the condition of seasonal wetlands, changes in matching of the timing of arrival dates and prey dynamics, predation effects, disease and parasitism. There may be complex interactions between these factors.
- The complexity of the interactions between climate change and other variables driving waterbird populations makes predictions of waterbird responses very difficult. Sutherland et al. (2012) suggested that given the nature of climate changes experienced to date, we need many more analyses of existing data and further data collection in order to assess the magnitude of this driver on waterbirds. To-date, the cost and logistical difficulties involved in long-term studies of the population ecology of waterbirds have meant that detailed information on breeding parameters and their relation to climate and weather patterns is relatively scarce (Ganter and Boyd 2000; Van de Pol et al. 2010). However such information is essential as input into models of the potential impact of various climate change scenarios on bird populations. Long-term data in particular (typically decades for birds and mammals) is required in order to untangle spatial and temporal variability from responses to change.
- Van de Pol et al. (2010) stated that although it is well established that climate change may strongly affect population dynamics, the general mechanisms causing climate induced population change are still poorly understood. They suggested that four major unresolved questions are: (1) Does climate change mainly affect population dynamics through its effects on survival or fecundity, and how does this vary between species and environments? (2) How important is the contribution of climate change to population fluctuations in comparison to other stochastic and deterministic processes? (3) What is the relative importance of changes in the mean and variability of climatic drivers? (4) Does increased interannual climatic variability typically reduce population viability as predicted by classical stochastic population theory, or can it also improve population viability as more recently put forward?

Worldwide, many populations of waterbirds have suffered declines in abundance as a result of toxic effects of pollution or contaminants in their environment. While pesticides (herbicides and insecticides) have received much-deserved attention in the last 30 years, waterbirds may also be affected by many other contaminants, including heavy metals, nutrients, and other chemicals.

- Often, a considerable number of contaminants are present in the environment and consequently it can be difficult to associate toxic effects with any one contaminant. Very similar chemicals can vary widely in their toxicity, and it is also likely that some contaminants interact to increase toxicity. Critical effect concentrations of contaminants are often unknown, making risk assessment difficult (De Luca-Abbott et al. 2001).
- Diet has a significant effect on the concentration of contaminants accumulated. Predatory waterbirds consuming fish, amphibians, reptiles, invertebrates and other prey are situated at the top of the wetland food web. Biomagnification or bioaccumulation of heavy metals may occur in these species, with a proportionate increase in concentration with increasing trophic level. The effects of these accumulations are not always apparent in terms of mortality or other parameters, however in some cases they may allow waterbirds to act as indicators of pollution accumulation in affected ecosystems (Kushlan 1993; De Luca-Abbott et al. 2001).
- The vast majority of effects are expressed in reproductive failure, and subsequent population declines may be masked by pairs laying repeat clutches after failures.
- Direct adult mortality of waterbirds as a result of pollutants is relatively rare, but incidents have been recorded (De Luca-Abbott et al. 2001; Howarth et al. 1981; Kim et al. 2007).
- Besides their direct toxic effects, pesticides and herbicides can reduce food availability for waterbirds, depending on their diet.

Disease and parasites can affect waterbird nest success, fledging rates, juvenile survival, and adult survival. Infectious diseases are an important and frequently dominant mortality factor in waterbird populations.

- Bacteria such as *Clostridium botulinum* and viruses such as avian influenza, West Nile Virus, Newcastle Disease Virus, avian poxvirus, duck plague, avian bornavirus, reoviruses and adenoviruses may contribute to population declines of both wild and domestic waterbirds. The infection rate by *Plasmodium* parasites (avian malaria) is rapidly increasing in many birds and there are high infection rates of Campylobacteria in waders (Sutherland et al. 2012 and references therein). Changes in land use and global climate may result in a stronger concentration of waterbirds on remaining high-quality sites, making them potentially more vulnerable to infections (Krauss et al. 2010 cited by Sutherland et al. 2012).
- Ticks parasitising nestlings can reduce survival and nest success, and potentially also transmit viruses.
- Botulism is fatal, is now present in every continent apart from the Antarctic, and is a leading cause of death for waterbirds. Given that the bacteria responsible for botulism have resistant spores that can survive for years these problems can persist, and may well have population-level impacts. This is most likely to be the case for species with small populations (Sutherland et al. 2012). Affected birds are also more likely to be subjected to predation. Data collected by the Australian Wildlife Health Network (AWHN) of *reported* Australian botulism cases over a six year period documented sixty eight events of suspected or confirmed avian botulism between 2006 and 2012 (WHA 2013). In ten of these events multiple species of birds from several bird orders were affected, however Anseriformes (ducks, geese and swans) were involved at the majority of events. The factors that predispose to botulism outbreaks are complex and incompletely understood. Further study into the environmental conditions associated with these outbreaks in Australia may help to understand why they occur and mitigate their impact (WHA 2013).

Disturbance from human activities can cause changes in behaviour and affect temporal and spatial distribution of waterbirds. Human disturbance can be equivalent to habitat loss or degradation because waterbirds may avoid or underuse areas (Gill & Sutherland 2000 cited by Fernandez and Lank 2008). Waterbirds may waste time and energy responding to human disturbance, and the consequences for individuals, populations, and sites can be difficult to determine. Temporary loss of foraging habitats can occur and the capacity to compensate by foraging for longer periods may vary between individuals and species (Sutherland et al. 2012 and references therein). During the breeding season, human disturbance may influence nest incubation and chick rearing.

- The effect of human disturbance on waterbirds is dependent on the nature and location of the disturbance, and aspects such the type, frequency, intensity and duration of disturbance can all interact.
- Waterbirds may be more sensitive to disturbances during particular periods, such as nesting, hatching, brood care and moulting.
- To some extent, some species can compensate for disturbance by altering their behaviour or habituating to human activities, and this may interact with predation (Madsen 1995; Baudains and Lloyd 2007; Fitzpatrick and Bouchez 1998). The effects of human disturbance may increase predation, or may not, depending on the characteristics of the disturbance and the species present.
- Buffers are often used to separate threatening stimuli, such as humans, from wildlife but with few exceptions buffer widths are based on little empirical information.
- Since rates of disturbance differ between species and may be correlated to foraging strategies, habitat requirements, and type of disturbance, managers of public lands should consider life histories of waterbirds when setting seasonal recreation dates or planning scientific surveys (Schummer and Eddleman 2003). Larger waterbird species often have greater 'flush distances' than smaller species to the approach of a disturbance such as a human or boat, and therefore management buffer distances need to take this into account (Rodgers and Schwikert 2003).
- The role of hunting or harvest in the dynamics of waterbird populations continues to be debated among scientists and managers internationally. Other than the obvious mortality, hunting causes significant disturbance and consequently spatial and temporal displacement of quarry species (Madsen 1998). There is a large international literature on the subject that is beyond the scope of this review, with contrasts between research concerning waterfowl (ducks, geese and swans) and research concerning other waterbird species such as shorebirds. Waterbird hunting in Australia has significantly decreased in popularity, frequency and impact in recent decades.

Both inter- and intra-specific competition for resources can affect waterbird condition, behaviour, abundance, distribution, and survival, and can vary both spatially and temporally. While food and habitat niches are usually sufficiently different to avoid competition between species in a native community under normal conditions, the introduction of invasive species, increases in populations of particular native species

driven by other factors, loss of critical habitat or food sources, or extreme weather conditions such as drought can significantly alter competition intensity and timing. When such changes take place concurrently, it can be difficult to identify causative mechanisms for shifts in waterbird distribution or changes in population parameters.

- Competition may particularly affect the movements, condition and survival of young, female, or subordinate portions of a population (Gyimesi et al. 2010; Amat and Rilla 1994; Stirneman et al. 2012; Minias et al. 2014).
- The mechanisms of bird density and prey density effects on competition can be complex, differing between species (Vahl et al. 2005)
- Depending on climate or weather over time, waterbird communities may exhibit periods of intense inter-specific and/or intra-specific competition, alternating with times of habitat or food abundance where competition may be insignificant or absent.
- Scarcity of preferred nesting sites is probably the key ecological factor underlying competitive behaviours during breeding, driven by natal philopatry, nest-site fidelity, aggressive competition for nest sites and intraspecific parasitism (Semel and Sherman 2001).
- Invasive or pest species may compete with waterbirds for habitat or food.

Where influences on waterbird populations are natural processes (e.g. predation, disease, competition), anthropogenic influences have almost always altered those processes. Consequently such processes are often cited as management problems, even though they are fundamentally natural. However it may be ultimately unproductive for the ecosystem as a whole if species are managed by simply trying to influence these processes, regardless of whether or not they have been altered by human activities.

Mechanisms, rather than just patterns, are important sources of rich information about why certain factors may be stressors or threats and how managers may intervene. For example, some of the reasons why human disturbance at nesting sites is a stressor are because bird responses to such disturbance use time and energy that would otherwise be spent in individual feeding or reproduction. This is important detail because it may guide specific targeted management responses. This type of mechanistic information is critical for more innovative management of biodiversity.

Uncertainty and complexity are core characteristics of floodplain wetland ecosystems. There are many factors that influence waterbird populations through multiple different mechanisms which are often context-dependent. There are interactions between flooding and all of the stressors and threats discussed in this review in terms of impacts on waterbirds. These interactions and their effects are difficult to untangle, especially without integrated studies with long term data.

Part I Waterbird responses to flows and flooding in Australia



1 Conceptual model: Drivers of waterbird species persistence and diversity

Flow-related variables that affect waterbirds include (Figure 1):

- **Flow regime – local to basin scales**
  - **Location, frequency, volume, duration, timing**
  - **Separation of environmental flow component from natural and regulated flows**
- **Flood regime – local to basin scales**
  - **Location, frequency, extent, duration, timing, depth, rate of change in depth, interflood period (dry duration)**
- **Local and catchment rainfall, evaporation and temperature**

These interact with pressures such as:

- **habitat loss, fragmentation and degradation**
- **predation (native, introduced)**
- **competition**
- **disease and parasites**
- **pollution**
- **human disturbance and hunting**
- **climate change**

At different spatial and temporal scales, these interactions affect the following cues/factors:

- **food abundance or availability**
- **habitat availability, distribution and quality (breeding (courtship, nesting), foraging, roosting and refuge)**
- **connectivity between habitats/sites and within the population**
- **condition of critical individual sites (nesting, foraging, roosting and refuge)**
- **catchment condition**

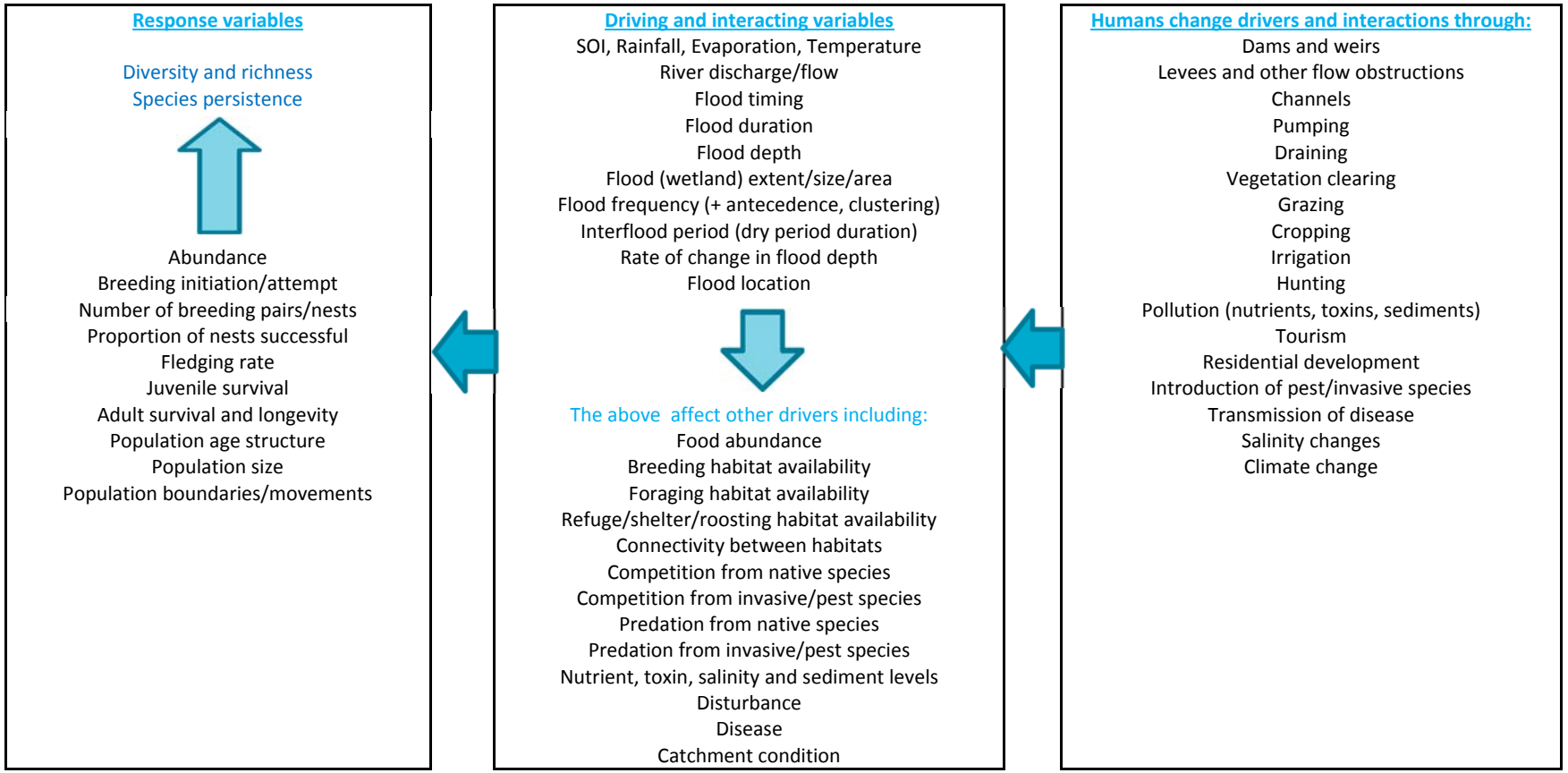
These variables affect waterbird abundance, through their influence on bird condition, breeding initiation (courtship, nesting and egg-laying), and recruitment of juveniles into the adult population. Some of these can be measured by the number of pairs/nests, the number of eggs laid, egg survival, hatchling survival, fledgling survival, juvenile survival, adult survival, individual and population movements, population size and age structure, and hence waterbird diversity.

For individual flow events at local scale, the critical waterbird response variables are:

- Numbers of nests and breeding pairs per species
- Fledging rates (taking into account repeat-nesting – i.e. nest success)
- Juvenile survival

For population maintenance/growth at Basin to continental scales, the frequency of breeding initiation is also important (Figure 1).

See also Section 11 for an additional diagrammatical conceptual model (in three stages) of links and interactions between flows/flooding, stressors and threats, and waterbird response variables.



Finally, the evolutionary adaptations, life histories and associated varying requirements of individual birds, species and guilds all affect waterbird responses - including bird memory, experience, behavioural characteristics and site attachment. These may vary considerably even within species.

Figure 1 Conceptual model of drivers of waterbird species persistence and diversity.

2 Summary of known relationships

2.1 Abundance

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
Waterbird abundance at individual sites or wetland complexes is highly variable and often unpredictable both temporally and spatially. Explanatory models have found wetland area (often with a 1 year lag), river flow, and rainfall to be the best predictors of waterbird abundance (positive relationships), with the strength of the relationship dependent on the primary mode of local wetland filling and the scale of the predictor and driver variables.	Kingsford and Norman 2002; Briggs 1977; Gosper et al. 1983; Woodall 1985; Kingsford and Porter 1994; Halse et al. 1998; Roshier et al. 2001a; Frith 1959; Briggs and Holmes 1988; Norman and Nicholls 1991; Kingsford 1996; Kingsford et al. 1999; Kingsford et al. 2012; Kingsford et al. 2013; Wen et al. 2011
Species/group abundance changes in response to wetland availability/area/flood extent at continental, basin, catchment and local scales	Roshier et al. 2002; Reid et al. 2009; Kingsford et al. 2013; Kingsford et al. 1999; Maher and Braithwaite, 1992; Kingsford and Thomas 2004; Wen et al. 2011; Kingsford et al. 2012
Total wetland area in the Murray-Darling Basin and the Lake Eyre Basin can explain the abundance of waterbirds in some individual wetlands and catchments - in particular, Lake Eyre Basin wetland area strongly influences waterbird abundances in the Murray-Darling Basin. The influence of wetland area in the Lake Eyre Basin on waterbird abundance in the Murray-Darling Basin is particularly strong in the Lower Cooper, Corop wetlands, Fivebough Swamp, Lowbidgee, Cuttaburra/Paroo, Menindee Lakes, and Lower Coorong. This influence can be stronger when time lags are taken into account. Roshier et al. (2002) concluded that Lake Eyre Basin wetland availability exerted stronger effects on waterbird abundance in the Paroo basin than did local habitat resources, at least for some functional groups.	Kingsford et al. 2013; Roshier et al. 2002; Kingsford et al. 2010;
Within the Murray-Darling Basin, two key factors explaining about half of the annual variation in abundance of waterbirds recorded by aerial surveys were current flow (year of the survey) across the MDB and wetland area (assessed during aerial survey) lagged by a year.	Kingsford et al. 2013
Within the River Murray catchment, abundance of waterbirds was significantly related to lagged wetland area and lagged rainfall by a year, and occasionally flow in the River Murray.	Kingsford et al. 2013
Kingsford and Thomas (2004) found a significant relationship between river discharge and waterbird abundance in the Lowbidgee. In contrast, modelling at the local scale by Wen et al. 2011 for the Lowbidgee found river discharge was **not** significant as a predictor of waterbird abundance when compared to a more direct surrogate for flood area/extent (using aerial survey data).	Wen et al. 2011; Kingsford and Thomas 2004
Mean abundance of all functional groups of waterbirds in wetlands of north-western NSW was significantly greater on large (1200+ha) than on small wetlands (<120ha), but a general model of increasing waterbird abundance with increasing wetland area was NOT supported.	Roshier et al. 2002

Known relationships	References
Waterbird abundance at a location can change rapidly by orders of magnitude over short periods of time (3 months or less), ultimately in response to changes in food abundance driven by changes in flood area/extent either locally or elsewhere. For example, numbers of functional groups and of individual species were highly variable and changed markedly in the three months between surveys of wetlands in north-western NSW during 1990, where estimates of abundance of pink-eared duck in the Paroo River catchment changed from 28,268 to 1,154 between March and June (Roshier et al. 2002). Low numbers of waterbirds are often observed at individual sites immediately after flooding, when habitat is extensive and waterbird numbers have not increased by immigration or breeding, while in contrast, there may be large concentrations of waterbirds during dry times, when birds are forced to move and concentrate on the remaining, more permanent wetlands, often in coastal areas. Abundance is often highest at inland floodplain sites during the drying phase.	Kingsford et al. 1999; Maher and Braithwaite 1992; Roshier et al. 2002; Kingsford and Norman 2002 citing White 1987; Morton et al. 1990a, 1990b; Norman and Nicholls 1991; White 1993; Kingsford 1996
Changes in food web structure over time can be reflected by changes in the abundance of particular waterbird functional groups, e.g. invertebrate feeders, herbivores and fish-eating birds	Kingsford et al. 2012; Kingsford and Porter 1994; Kingsford et al. 2004
Species/group abundance changes in response to rainfall at local, catchment, and basin scales.	Kingsford et al. 1999; Kingsford et al. 2012; Kingsford et al. 2013; Wen et al. 2011; Padgham 2011
In a broad-scale climatic analysis of associations between rainfall and waterbird abundance using the eastern Australian aerial survey data, the strongest climatic influence on avian abundance was shown to be the coherence of large-scale precipitation systems with spectral frequencies of around 15 days that propagate south-east into the interior of the continent from the tropical north. Directly aggregated rainfall presents a secondary influence. These models were able to explain over 75% of the inter-annual variance in the aggregate abundance of over 90 species of waterbirds.	Padgham 2011
In analyses conducted for the Lower Murrumbidgee floodplain wetlands, five of seven functional groups were positively related to both annual rainfall and water usage, defined as the total water volume intercepted by the river reach [surrogate for flood area/extent], and the models indicated that rainfall was slightly more important. Rainfall was not a significant predictor for grazing waterfowl (e.g. AWD, ASD, PWD) and small waders.	Wen et al. 2011
Species/group abundance at a site may be associated with temperature (season)	
In the Lower Murrumbidgee, temperature also appeared to be associated with waterbird abundance for some groups: the maximum summer temperature negatively influenced the abundance of dabbling ducks, shoreline foragers and fish eaters, while the minimum winter temperature positively affected the abundance of dabbling ducks and shoreline foragers.	Wen et al. 2011;
Species/group abundance responses are greater in response to shallow extensive inundation of intermittently flooded or ephemeral wetlands than in static, permanently inundated wetlands, particularly in the north-western and western parts of the Murray-Darling Basin.	Reid et al. 2009; Maher and Braithwaite 1992; Kingsford and Norman 2002; Lawler and Briggs 1991; Kingsford and Porter 1993; Kingsford et al. 1999a; Kingsford et al. 2004; Briggs et al. 1985; Maher and Carpenter 1984; Maher 1984; Crome 1986, 1988; Kingsford and Thomas 2004; Scott 1997
The relatively flat gradient floodplains in the north-western and western parts of the Basin are conducive to peak bursts or booms in aquatic production. The increasing temperature gradient to the north interacts with and partly reinforces the low topographic gradient effects, giving rise to a putative south-east to north-west productivity gradient. Maher and Braithwaite (1992) described waterbird densities in the Paroo wetlands as being 3-4 times that of contemporaneous waterbird densities elsewhere in the Basin. An alternative and partly countervailing productivity gradient probably occurs from the north-east to south-west consistent with the direction of flow in most catchments and the distribution of extensive floodplains in the lower reaches. This suggests the need for powerful covariates in basin-wide models of waterbird abundance.	Reid et al. 2009; Maher and Braithwaite 1992

Known relationships	References
Key individual wetlands are consistently important and should be prioritised for management of long-term waterbird abundance.	Kingsford et al. 2013; Kingsford et al. 2012 citing Morton et al. 1990a,b, 1993a,b,c; Garnett 1987; Halse et al. 1998; Jaensch and Vervest 1990a,b; Roshier et al. 2002; Kingsford et al. 2011; Wade and Hickey 2008; Kingsford et al. 1999; Kingsford and Porter 1993; Halse et al. 2005
These wetlands include the following 17 wetlands: Booligal wetlands, Currawinya Lakes, Cuttaburra channels, Gwydir wetlands, Kerang wetlands, Lake Buloke, Lake Cowal, Lake Brewster, Lindsay-Walpolla-Chowilla system, Lowbidgee, Lower Lakes and Coorong, Macquarie Marshes, Menindee Lakes, Narran Lakes, Paroo overflow lakes, Tallywalka system and Yantabulla Swamp. Generally, these same wetlands were highly ranked in terms of breeding indices. All are already identified as important through protected area, Ramsar, Living Murray icon site or HIS designations. Over the 30 years of the 1983-2012 aerial surveys, about 80% of all waterbirds in the Murray Darling Basin could be found on the 20 highest ranked wetlands of a year's survey. Even the 10 highest ranked wetlands had about 60% of all waterbirds.	Kingsford et al. 2013
Waterbird numbers can be heavily concentrated in relatively few wetlands, e.g. during the 2008 National Waterbird Survey, over 50% of recorded waterbirds occurred in only 41 wetlands or 1.1% of the total number of wetlands surveyed. 39% of all recorded waterbirds occurred on the top 20 wetlands, as ranked by waterbird abundance, with over 6% occurring in the highest ranked wetland alone (Eighty Mile Beach) [most of these were migratory shorebirds]	Kingsford et al. 2012
The inland, Lake Eyre and Bulloo-Bancannia drainage divisions in central Australia are particularly important for duck species because of the productivity of their ephemeral wetlands	Kingsford et al. 2012
Species/group abundance is greater following clusters of floods over two to four years	Reid et al. 2009 p.123 and pers. Comm. refs therein; Puckridge et al. 2000; Briggs and Holmes 1988;
Some studies have explored the influence of the Southern Oscillation Index (and sometimes the associated La Nina / El Nino events) and have found relationships with individual species abundances, often with lags of 1-5 years taken into account. Short-term recovery in waterbird abundance can occur following widespread flooding driven by La Nina events and environmental flows.	Kingsford et al. 1999; Kingsford et al. 2013
Extended wet periods may allow waterbirds to produce more than one or two clutches successively, resulting in rapid increases in abundance.	Kingsford and Norman 2002 citing Braithwaite 1981a, Fullagar et al. 1988; Cowan 1973
Different functional groups and species differ significantly in abundance and can heavily influence overall abundance assessments (and community/media impressions of abundance)	
Across Australia, the most abundant functional group of waterbirds found during aerial surveys in 2008 was the herbivores (e.g. Australian shelduck, Eurasian coot, black swan). The most abundant species in Australian wetlands during 2008 were the magpie goose, small waders, plumed whistling-duck, grey teal, large waders, egrets, banded stilt, wandering whistling-duck, pink-eared duck, terns, black swan, and Eurasian coot. These species accounted for over 82% of all waterbirds counted. In contrast, at least 43 least abundant species comprised less than 1% of all waterbirds surveyed.	Kingsford et al. 2012

Known relationships	References
Within the Murray-Darling Basin, during aerial surveys 1983-2012 more than half the total number of waterbirds consisted of ducks (one third of which were grey teal), followed by almost a quarter comprising 'herbivores' (black swan, magpie goose, cape barren goose, eurasian coot, black-tailed native hen), followed by large wading birds, piscivores, and shorebirds. Grey teal comprised approximately one third of all waterbirds counted during aerial surveys in the MDB 1983-2012, while the other 11 species in the top twelve each comprised less than 10% (in order, Eurasian coot (8%), pink-eared duck, straw-necked ibis, pacific black duck, Australian shelduck, hardhead duck, Australian wood duck, black swan, Australian pelican, black-winged stilt, whiskered tern (3%); other species 19%).	Kingsford et al. 2013
Species/group abundance may be partly determined by bird experience/memory/site attachment	
Significant long-term declines in waterbird abundance have occurred in areas subject to river regulation / flow regime change in the Murray-Darling Basin, but not to the same extent in the unregulated Lake Eyre Basin and Paroo wetlands. This applies to all functional groups of waterbirds, with declines of similar magnitude across groups. (Except possibly deep-water foragers / piscivores? Wen et al. 2011)	Kingsford et al. 2013; Kingsford et al. 2012; Rogers and Paton, 2009; Kingsford and Thomas 2004; Brandis et al. 2009; Nebel et al. 2008; Kingsford and Thomas 1995; Kingsford and Johnson 1998; Wen et al. 2011; Paton et al. 2009?
There was a 74% decline in mean abundance of all waterbirds during the first decade (1983-1992) to the last decade (2003-2012) [of the AWSEA] across the Murray-Darling Basin. Similarly, there was a decline of 66% in mean abundance across the River Murray catchments. The level of decline in mean abundance between the two decades varied for the 13 individual wetlands between 28% and 97%.	(p.9 Kingsford et al. 2013).
Within the MDB, long-term declines have been recorded in some of the most common duck species including grey teal, pacific black duck, Australian shelduck, Australian wood duck, and hardhead duck. Some species increased in abundance ('recovered') during the 2010-2012 flood period, including Eurasian coot, pink-eared duck and straw-necked ibis.	Kingsford et al. 2013

2.2 Population size and age structure

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
Most species that require flooding of wetlands for breeding are highly mobile and hence they do not necessarily need to breed in the same location every year to maintain overall population size. Provided sufficient nesting habitat and nearby foraging resources are made available somewhere within a species' range during its lifespan with sufficient frequency and for sufficient duration, populations should remain stable. However with river regulation these requirements may no longer be met for some species. Flooding events of a size known to trigger and support bird breeding events have been reduced significantly in frequency in all major wetlands of the MDB that are subject to regulation.	Leslie 2001; Arthur 2011; Kingsford et al. 2013; Roshier et al. 2001; Scott 1997
Early modelling suggested that breeding would have occurred almost annually in Barmah Forest if none of the storages had been constructed and with no diversions. There were 20 'excellent' breeding episodes in 100 years (modelled). This has now been reduced to breeding four times a decade on average, with 'excellent' breeding only occurring three times per century. Modelling by Leslie (2001) indicated that the frequency of successful breeding episodes has been reduced by 80%.	Leslie 2001;
An age-structured population model for egrets (generic egret) suggested that most egrets must breed every one to two years for populations to persist, and therefore a high frequency of sufficiently large flood events is required within their range.	Arthur 2011
Very large inland floods are probably critical for maintenance of populations of many waterbird species in the long term, because such boom events result in very large breeding events, often with repeated breeding.	Kingsford et al. 1999
Braithwaite (1975) postulated that waterbird populations were controlled in the long term by the availability of wetlands during dry periods. The food resources of drought refuges determine the number of birds that can exploit the next breeding opportunity (Maher 1991).	Scott 1997; Braithwaite 1975; Maher 1991
The number of waterbirds recorded during the National Waterbird Survey 2008 was 4.55 million; an estimate of the true number of waterbirds at the time, extrapolating from randomly surveyed wetlands, was calculated to be 4.65 million. Some species appear to have maintained a steady population size up until 2008 (e.g. magpie geese and banded stilt), while others appear to have declined considerably (e.g. grey teal).	Kingsford et al. 2012
In 2008, magpie goose population size was estimated at 900,000, compared with past estimates of around 1 million (although one regional estimate suggested 1.6 million)	Kingsford et al. 2012
In 2008, banded stilt population size was estimated at 200,000, compared with past estimates of around 206,000 (in 2006)	Kingsford et al. 2012
In 2008, grey teal population size was estimated at 320,000, compared with past estimates of over 1 million	Kingsford et al. 2012
There may be an average of more than 1.5 million waterbirds using the wetlands and rivers of the MDB, and this may have ranged from 280,000 and 8.5 million from 1983-2012. The range of **annual** estimates of waterbirds across the Murray-Darling Basin from aerial surveys 1983-2012 varied from 38,000 to more than 1 million. These estimates do not necessarily include or sufficiently reflect populations of relatively cryptic species.	Kingsford et al. 2013

2.3 Population boundaries / movements

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
Long-distance broad-scale movements occur in response to cues that indicate changes in food and habitat availability. These cues may include rainfall, flooding, drought, season, and bird memory/site attachment. Individuals may travel between locations thousands of kilometres apart.	Frith 1957, 1959, 1962, 1963; Roshier et al. 2006; Kingsford and Norman 2002; Marchant and Higgins 1990; Minton et al. 1995; Kingsford et al. 1999; Alcorn and Alcorn 2000; Roshier et al. 2002; Griffioen and Clarke 2002; Lawler and Briggs 1991; Maher and Braithwaite 1992; Lawler et al. 1993; Kingsford and Porter 1993, 1994, 1999; Halse et al. 1998; Kingsford 1996; McKilligan 1975; Carrick 1962; Kingsford, Curtin and Porter 1999; Kingsford, Wong, Braithwaite and Maher 1999; Reid and Jaensch 2004; Woodall 1985; Briggs and Holmes 1988; Norman and Nicholls 1991; Halse et al. 1992; Briggs 1992; Leach and Hines 1991; Lavery 1970; Norman 1971; McKean and Braithwaite 1976; Llewellyn 1983
There is considerable variability in spatial and temporal movement patterns within and between species	See references above
Species that prefer permanent water environments and have regular breeding seasons tend to be relatively limited and regular in their movements - e.g. the musk duck, blue-billed duck and black swan.	Frith 1959
Species that are flexible in their requirements or prefer ephemeral water environments generally breed opportunistically and can travel long distances in response to rainfall or flooding - e.g. grey teal, pink-eared duck. There can be some bias in movements toward the more well-watered coast during extended drought periods or during spring, but movements switch inland following flooding or rainfall.	Frith 1959; Frith 1962; Frith 1982; Gosper et al. 1983; Braithwaite 1975; Lawler et al. 1993; Scott 1997; Llewellyn 1983; refs above
In grey teal, long-distance movements vary markedly at the individual level in terms of timing and/or direction, and the movements of this so-called nomadic species do not appear to be random wanderings between adjacent wetlands.' 'The observed pattern of movement suggests that Grey Teal interact with available habitat at broad scales and that movement at these scales is, at least in part, based on experience and spatial memory' .	Roshier et al. 2006
Species that are intermediate in habitat preferences (e.g. using both permanent and temporary habitats) tend to be intermediate in degree of nomadism and regularity of breeding season - e.g. the pacific black duck. Moulting requirements can also dictate movements - e.g. in south-eastern Australia, Australian Shelducks move from breeding areas to large semi-permanent or permanent wetlands. Movements of many tropical waterbirds are relatively predictable, spreading out onto inundated floodplains during the wet season and then retreating to remnant wetlands in the dry season.	Frith 1959; Braithwaite 1975; Kingsford and Norman 2002 citing Bayliss and Yeomans 1990; Morton et al. 1990a, 1990b, 1993a, 1993b; McKean and Braithwaite 1976; Norman 1983
Some species, especially small waders and shorebirds, are migratory and follow a regular pattern of movement across Australia and the MDB en route to and from breeding grounds in the northern hemisphere. Their movements may be less predictable when they follow temporary inland wetlands, and a few individuals remain in Australia, adding to the complexity. The tropical, Indian Ocean and Western Plateau drainage divisions in north-western Australia are particularly important as staging areas and over-wintering sites for migratory shorebirds.	Scott 1997; Kingsford and Norman 2002 citing Thomas 1970; Lane 1987; Tulp et al. 1994; Kingsford and Porter 1993; Kingsford et al. 1999a; Lane 1987; Kingsford et al. 2012
Networks or mosaics of wetland habitat across the continent provide essential corridors or	Brandis 2010 citing Haig et

Known relationships	References
stepping stones for movements, and must be preserved to support population persistence and species diversity. Directions of waterbird movement and distances travelled vary depending on wetland habitat availability, natal sites and species.	al. 1998; Dorfman and Kingsford 2001; Roshier et al. 2001; Kingsford and Norman 2002
In Australia's arid zone, there is always wetland habitat available providing a network across the landscape in any 12 month period (Roshier et al. 2001).	Roshier et al. 2001
Following breeding, colonial nesting waterbirds disperse in small groups, pairs, or singly as the wetland dries, often to coastal locations or the tropics (Carrick 1962; Marchant and Higgins 1990), but little is known about this process (Kingsford and Norman 2002).	Kingsford and Norman 2002 citing Carrick 1962; McKilligan 1975; Geering et al. 1998; Marchant and Higgins 1990
Movement patterns may vary with age in ibis, with young ibis moving further than older birds	McKilligan 1975
Cormorants bred in the lower Murray tend to disperse eastwards seeking permanent water; however there is variation in the direction of dispersal	Llewellyn 1983
Waterbird movements in the Murray-Darling Basin are influenced by flooding patterns in the Lake Eyre Basin and other inland areas	Roshier et al. 2002; Kingsford et al. 2010; Kingsford et al. 2013
While there are limited data on the actual movements of waterbirds, for fish-eaters, dabbling ducks, grazing waterfowl, shoreline foragers, large waders and small waders in the Paroo River catchment the best predictor of abundance was change in wetland area of +/- 100,000 ha in the Lake Eyre Basin and/or one of its component catchments, accounting for 40-56% of the observed deviance. When wetland area in the Lake Eyre or Bulloo basins decreased by 100,000 ha or more, waterbird numbers increased in the Paroo. Changes in wetland area in the Paroo itself, or in the adjacent Warrego or Darling catchments did not predict waterbird abundance, except for grazing waterfowl. Only for shoreline foragers and small waders was change in wetland area in the local catchment a significant factor.	Roshier et al. 2002
Australian Pelicans move across the continent in response to habitat availability driven by flooding or the lack of it. Pelicans were rare around Brisbane when Lake Eyre in central Australia was full in 1974-76 but became numerous again in 1978-79 when the lake dried, and similarly in northern Australia. Great cormorants appear to respond in similar ways. Black-tailed native hens rapidly increased in abundance in south-east Australia after the wet years inland in 1973 and 1974	Kingsford and Norman 2002; Woodall 1985; Draffan et al. 1983; Matheson 1978; Briggs 1990
The ranges or population boundaries of some species have contracted, but the causes of contractions are not well understood (e.g. Magpie Goose, Cotton Pygmy-goose, Green Pygmy-goose, Black-necked Stork, Brolga)	Kingsford and Norman 2002

2.4 Breeding initiation

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
For most Australian waterbirds, breeding occurs when their preferred food resources are approaching, or are at, a maximum, and appropriate nesting habitat is available. There is a recognised temporal sequence of breeding initiation for different species/guilds during a flood event. Many species have a lag period before breeding initiation, for courting (establishment of pairing) and fat storage to enhance reproduction.	Reid et al. 2009 citing Leslie 2001; Carrick 1959, 1962; Briggs and Thornton 1999; Kingsford and Norman 2002 citing Briggs 1991; McKilligan 2001; Braithwaite 1977; Miller 1980; Kingsford and Norman citing Carrick 1962; Braithwaite and Frith 1969; McKilligan 1975; Miller 1980; Braithwaite 1982; Llewellyn 1983; Crome 1986; Kingsford 1989b; Maddock and Baxter 1991; Whitehead and Saalfeld 2000
In temperate areas of southern Australia, most waterbirds require suitable flood and seasonal conditions to coincide to be induced to breed. In the tropics, predictable breeding coincides with the summer wet season. However breeding may be initiated at any time in the arid and semi-arid zones following flooding or rainfall. Waterbird breeding attempts in the Murray-Darling Basin are influenced by flooding patterns in the Lake Eyre Basin.	Roshier et al. 2002; Kingsford et al. 2010; Kingsford et al. 2013; Leslie 2001 citing Chesterfield et al. 1984, Briggs 1990, Scott 1997; Kingsford and Norman 2002 citing Lavery 1970b; McKilligan 1975; Frith 1982; Whitehead and Saalfeld 2000; Chatto 2000; Maddock 2000; Braithwaite 1976a; Halse and Jaensch 1989; Kingsford 1989b; Carrick 1962; Braithwaite and Frith 1969; Lawler and Briggs 1991; Maher and Braithwaite 1992; Kingsford and Johnson 1998; Ley 1998; Briggs and Thornton 1999; Maddock and Baxter 1991; McKilligan 2001; Lavery 1970b
Some Australian waterbirds breed after heavy rainfall alone, especially if rainfall is sufficient to fill wetlands	Kingsford and Norman citing Maddock and Baxter 1991; McKilligan 2001; Lavery 1970b; Geering 1992; Kingsford 1989b
Egret colonies re-established on the north coast of NSW when heavy rainfall followed a dry period, and Australian wood ducks have been recorded breeding in autumn and spring following rainfall	Kingsford and Norman 2002 citing Geering 1993 and Kingsford 1989b
Grey teal may engage in reproductive behavioural displays after a single heavy downpour of rain	Kingsford and Norman 2002 citing Braithwaite 1976b
Straw-necked ibis move to breeding areas after rain	Kingsford and Norman 2002 citing McKilligan 1975
Local rainfall inland can create foraging and breeding habitat for waterbirds that can last for up to 2 years, but such events are infrequent. For example, the most important recorded breeding event for banded stilts on Lake Callbonna in 1931 resulted from local rainfall.	Kingsford et al. 1999 citing McGilp and Morgan 1931
Flooding is not required for initiation of breeding in some species (e.g. musk duck, blue-billed duck, AWD, PWD, ASD), but most waterbird species in Australia respond to flow-related triggers	Frith 1959; Reid et al. 2009 citing Reid 2006 and waterbird sections in McCarthy et al. 2006)
A simple rise in water depth in rivers or wetlands can stimulate breeding in some species	Reid et al. 2009 citing MFAT Young et al. 2003 and Leslie in Bennett (2000)
In the grey teal, sexual activity is triggered immediately by an increase in water level, whether rain has fallen or not - eggs may be laid 7-10 days later.	Frith 1959
Flood area/extent is a key driver of breeding initiation for many waterbirds, with positive	Reid et al. 2009; Brandis

Known relationships	References
associations between measurements or surrogate measurements of flood area and probabilities of breeding. For some species, threshold areas of inundation may exist for breeding initiation.	2010;
The pink-eared duck begins its sexual cycle following flooding of low-lying land, and is not triggered by a rise in river levels alone. This is related to the timing of suitable food availability for ducklings.	Frith 1959
At a regional scale, the number of breeding waterbird species is strongly 'explained' by wetland area but less so by the number of wetlands	Kingsford et al. 2012
For some species, threshold durations of inundation exist for breeding initiation. For example, modelling has demonstrated that there are thresholds of flow that must be exceeded, usually for a minimum number of days, for initiation of colonial nesting waterbird breeding in individual wetlands (ibises, egrets, herons, spoonbills). Lag times between threshold exceedence and initiation of breeding vary depending on the site and species.	Arthur et al. 2012; Briggs and Thornton 1999; Scott 1997; Reid et al. 2009;
In the southern Murray-Darling Basin, modelling of long-term datasets has found that exceedence of daily flow thresholds is usually required for 30-50 days to trigger breeding in most colonial nesting waterbird species. For all species, the maximum probability of attempted breeding occurs after a daily flow threshold is exceeded for about 50 days. Ibises respond to a 5-10 day shorter period of threshold exceedence than egrets. Thresholds may be exceeded for 10-15 fewer days for attempted breeding to occur at the Macquarie Marshes than at Barmah-Millewa Forest.	Arthur et al. 2012
The total flow volume July-December also serves as a good predictor of breeding initiation of CNW in Barmah-Millewa Forest and the Macquarie Marshes. However there is greater variation in species responses to volume at the Macquarie Marshes than at Barmah-Millewa, with egrets requiring greater volumes before breeding initiation.	Arthur et al. 2012
In earlier modelling for the Macquarie Marshes, total annual river flow at Oxley of over 200,000 ML was required to induce colonially nesting waterbirds to breed. Breeding was positively related to flow in the three months before breeding.	Kingsford and Johnson 1998; Kingsford and Auld 2005
Ibis nesting at Lake Merreti is associated with a maximum flow threshold in September of ~25,000 ML/day	Arthur et al. 2012
Nankeen Night Heron breeding initiation is closely related to maximum daily flow in October in Barmah-Millewa Forest and the Macquarie Marshes	Arthur et al. 2012
Early modelling for the Barmah Forest found that the Algeboia Plain colony (Great Egret, Intermediate Egret, Nankeen Night Heron) initiated breeding when river discharge maximums exceeded 18330 ML/day during September and/or October (rarely November)	Leslie 2001
At Narran Lakes, modelling found recorded ibis breeding events were associated with thresholds for three flow measures: total flow event volumes >100,012 ML, mean daily flows >1,552 ML, and flow durations >63 days at Wilby Wilby. An annual flow threshold for breeding of 160,183 ML at Wilby Wilby was identified as the trigger for breeding.	Brandis et al. 2010
Australian white ibis and straw-necked ibis breed 1 month and 2 months, respectively, after flooding begins	Carrick 1962
Average lag times between flooding under nest trees and first nests of colonial nesting waterbirds (darter, cormorants, herons, egrets, ibis and spoonbills) in the mid-Murrumbidgee wetlands ranged from 2-4 months following spring floods, and from 4-7 months following autumn floods. The average lag times from flooding under nest trees to first broods of ducks observed were 1-2 months, regardless of season.	Briggs and Thornton 1999
In the Paroo system, waterbird breeding (egg laying) may not commence until 8 weeks following the beginning of flooding, or two weeks after maximum inundation area is reached, and breeding peaks four to six months after flooding. Birds may remain in the system for up to 30 months after which all wetlands are dry.	Maher and Braithwaite 1992
For some species, threshold depths of inundation exist for breeding initiation	Reid et al. 2009
Some species require water to be present underneath or surrounding their nesting sites in order to initiate and complete breeding e.g. straw-necked ibis usually build nests on emergent vegetation in water 2-3 feet deep, but sometimes less, and also occasionally on dry ground or trees surrounded by floodwater.	Briggs and Thornton 1999; Briggs et al. 1997; Leslie 2001; Carrick 1962; Maher 1991; Maher and Braithwaite 1992
Breeding initiation at a site may be prevented by lack of suitable habitat - e.g. aquatic and semi-aquatic plants, live and dead trees, tree hollows, islands - even if water and species are present. Appropriate vegetation structure and composition for feeding and nesting is	Brandis et al. 2011 citing Bren 1988; Brock and others 2006; Capon 2005; Briggs et al. 1997; Kingsford and

Known relationships	References
essential for breeding waterbirds and floodplain vegetation is significantly affected by changes in flood frequency, duration, depth, and interflood periods.	Norman 2002 citing Marchant and Higgins 1990, 1993, 1996;
Many species require water to be present underneath or surrounding particular vegetation types in order to initiate and complete breeding e.g. straw-necked ibis usually build nests on emergent vegetation in water 2-3 feet deep, but sometimes less, but also occasionally on dry ground that is surrounded by floodwater. Lignum is a particularly important plant species for nesting habitat inland.	Briggs and Thornton 1999; Briggs et al. 1997; Leslie 2001; Carrick 1962; Maher 1991; Maher and Braithwaite 1992
Australian wood ducks are obligate tree hollow nesters, breeding on or near wetlands with trees of appropriate age to provide such hollows	Kingsford and Norman 2002 citing Frith 1982; Kingsford 1992
Dominant vegetation form is an important determining variable for probability of breeding of intermediate egret, little egret, straw-necked ibis, pied cormorant and Australian pelican, reflecting specific nesting requirements for these species.	Brandis 2010
Dense wetland vegetation is essential for nesting of many herons, egrets, ibises and bitterns	Kingsford and Norman 2002 citing Frith 1982; Marchant and Higgins 1990
Islands in lakes are preferred breeding habitat for banded stilt, Australian pelican, silver gull, caspian tern and gull-billed tern	Kingsford and Norman 2002 citing Burbidge and Fuller 1982; Waterman and Read 1992; Kingsford and Porter 1993, 1994; Kingsford et al. 1999a
The availability of nutritious and abundant vegetation determines when herbivorous waterbirds such as black swans and Australian wood ducks breed. Changes in food availability may account for seasonal differences in clutch size of black swans.	Kingsford and Norman 2002 citing Braithwaite 1976a, 1982; Kingsford 1989b
Breeding initiation at a site can be partly determined by bird experience/memory/site attachment	
Colonial nesting waterbirds can develop traditional attachments to nest sites that provide reliable nesting and foraging habitats. 'Such nesting site fidelity can equip individuals with prior knowledge regarding resources and risk (Renken and Smith 1995). There is anecdotal evidence that site fidelity exists in colonially breeding waterbirds in Australia. For example, marked intermediate egrets returned to natal sites in the Macquarie Marshes (R. Jones pers. comm.)'. Egrets regularly move to and from the Macquarie marshes and female Chestnut Teal often return to breed each year in the same or nearby artificial nest boxes in Victoria. However site fidelity may increase a species' vulnerability to changes in habitat.	Brandis 2010; Scott 1997; Leslie 2001 citing Marchant and Higgins 1990; Brandis et al. 2009; Arthur 2011; Maddock and Baxter 1991; McKilligan 2005; Kingsford and Norman citing Llewelyn 1983; Maddock and Geering 1993; McKilligan et al. 1993; Geering et al. 1998; Baxter 1994; Kingsford and Norman citing pers comms and unpub data
Long-term decline in the frequency of breeding events, the number of waterbirds breeding and the number of breeding species has been demonstrated across the Murray-Darling Basin (e.g. evidence from 1983 - present aerial surveys)	Kingsford et al. 2012; Kingsford et al. 2013; Leslie 2001;
Brandis et al. (2010) estimated that reductions in total annual flow volume in the Narran Lakes from river regulation would decrease opportunities for ibis breeding on large flow events by 30% and change the frequency of breeding from once every four years to once in 43.5 years. They also found that water management rules for waterbird breeding did not coincide well with the timing of flow events of sufficent size or the majority of waterbird breeding events.	Brandis 2010
The frequency and extent of breeding by colonial nesting waterbirds has been significantly reduced in the Macquarie Marshes because of reductions in flood area of at least 40-50%	Kingsford and Thomas 1995; Kingsford and Johnson 1998
Brandis et al. (2010) found 959 records of colonial waterbird breeding in Australia from 1899-2008. Breeding was recorded at 276 unique wetlands. A breeding event for one of the nine studied colonial nesting species occurred on average once every two years. Only three percent of wetlands recorded >10 breeding events - most (64%) wetlands had only one event recorded. Ibis breeding accounted for 36% of all records.	Brandis 2010
Prolonged dry periods may stop or extensively modify breeding of waterbirds. In dry periods, waterbirds may not breed, or breed later, produce fewer clutches and smaller eggs, have higher nest densities, lower reproductive success, increased rates of nest abandonment, low or no juvenile survival, and increased intraspecific nest parasitism and predation pressure.	Kingsford and Norman 2002
Conversely, prolonged inundation may kill or significantly alter nesting habitat (such as live	Kingsford and Norman 2002

Known relationships	References
trees) to the point that breeding is reduced or no longer occurs at a site	citing Briggs et al. 1994
Mean annual temperature was identified as an influencing variable for probability of breeding of Australian Pelicans and Great Cormorants in modelling conducted by Brandis (2010), however its influence was outweighed significantly by wetland area	Brandis 2010

2.5 Number of breeding pairs / nests

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
The number of breeding pairs/nests increases with increasing flood extent and duration	Reid et al. 2009 citing MFAT Young et al. 2003 and Leslie in Bennett (2000); Reid et al. 2009 p.123 and pers. Comm. refs therein; Puckridge et al. 2000; Briggs and Holmes 1988;
At regional and local scales, the number of breeding waterbirds is strongly 'explained' by wetland area and the number of wetlands, which are in turn correlated	Kingsford et al. 2012; Briggs et al. 1997
There is a positive relationship between flow in the Macquarie River (positively related to flood area/extent in the Macquarie Marshes) and the number of nests of colonial nesting waterbirds in the Macquarie Marshes. The best predictor of number of nests was the river discharge in the 3 months prior to breeding. The strength of the relationship varied with species. The species with the strongest relationships were the most numerous: intermediate egret, rufous night heron, glossy ibis, straw-necked ibis and Australian white ibis. A six-month lag was slightly better for the intermediate egret, while a nine-month lag was best for great egret and cormorants. The strongest relationship was for rufous night heron and Australian white ibis.	Kingsford and Johnson 1998; Kingsford and Auld 2005
The number of breeding pairs/nests increases following extended wet periods and/or clusters of floods over two to four years	
Large breeding events triggered by very large flood events are critical for population maintenance over time. e.g. The 2010 breeding event in the MDB was about six times larger than during any other year from 1983-2012, and resulted in increased waterbird abundance during 2011 and 2012	Kingsford et al. 2013
Two main abiotic factors are at play – La Nina rainfall clusters and lowered hydrological resistance (already wet and unobstructed channels). Briggs and Harper use the term 'memory' to describe the breeding response by colonial nesting waterbirds, whereby their abundance and breeding effort can increase in a second or third successive year of wetland flooding, presumably because they have not moved far away in the periods between consecutive breeding events (and/or recent positive memories). Puckridge et al. (2000) formally documented this phenomenon for fish, in particular, but also other instream biota in the Coongie Lakes, Cooper Creek, SA, over a four year flood cluster from 1988 to 1992. It is important to note that two to three consecutive years of flooding does not imply stable high water levels over this period – it is still important that much of the floodplain dries out between successive flood peaks.'	p.123 Reid et al. 2009 citing Puckridge et al. 2000
The number of breeding pairs/nests is typically greater in intermittently flooded wetlands that experience productivity booms than in permanently inundated wetlands	Kingsford et al. 2004; Briggs et al. 1985; Maher and Carpenter 1984; Maher 1984; Crome 1986, 1988; Kingsford and Thomas 2004;
Ducks breed most prolifically at sites that have reflooded after a dry period	Briggs and Thornton 1999; Maher and Carpenter 1984; Crome 1986; Briggs et al. 1997
Key locations are consistently important and should be prioritised for management of waterbird breeding	Kingsford et al. 2013; Brandis et al. 2009
Most records of colonial nesting waterbird breeding are in the MDB. The top 10 wetland complexes identified by Brandis et al. 2009 for colonial nesting waterbird breeding (based on historical records) were Narran Lakes, Lowbidgee Wetlands, Macquarie Marshes, Kerang Wetlands, The Coorong, Paroo Floodplain, Gwydir Wetlands, Barmah-Millewa Floodplain, Lachland Wetlands, and Menindee Wetlands. These include arid, semi-arid, and temperate zone wetlands. Most colonial nesting waterbird breeding occurs in flooded lakes, floodplains and other low-lying areas.	Brandis et al. 2009
Large recorded breeding events (>10,000 individuals) of colonial nesting waterbirds occurred on only 24 individual wetlands during the period 1899-2008	Brandis 2010
The impact of river regulation on waterbird breeding - at least colonial-nesting waterbirds -	Kingsford et al. 2013;

Known relationships	References
is reasonably well documented. Significant long-term declines in the number of breeding pairs/nests have occurred in areas subject to river regulation in the Murray-Darling Basin, but not in the unregulated Lake Eyre Basin	Brandis et al. 2009; Kingsford and Johnson 1990
In Barmah Forest, river management since the mid 1950's is associated with the loss of breeding colonies of at least eight formerly abundant species, a significant decline in breeding numbers of at least a further 11 species, and abandonment of numerous traditional nest sites. Species dependent on non-emergent macrophytes effectively stopped breeding in Barmah Forest by 1980 (e.g. Whiskered Terns, Black Swans, Eurasian Coots) because of the decline in suitable breeding habitat and food sources. Numbers of breeding tree nesting waterbirds declined by at least one order of magnitude. When initiation of breeding occurs, nests and fledging frequently fail. Leslie (2001) attributed this to reduced flood duration acting to decrease nest security and food availability during fledging. Species that construct their nests in emergent macrophytes have also reduced in breeding numbers by an order of magnitude. There was a 20-year lag between the time that river management began to affect waterbird breeding and waterbird abundance and diversity declining signfiicntly in the forest.	Leslie 2001
In the mid-Murrumbidgee wetlands, breeding in Black Swans, Pacific Black Ducks and Grey Teal was found to be positively related to the maximum area of emergent aquatic plants. Few ducks or other precocial species bred at wetlands where water levels were highly controlled. Nesting in altricial species (mostly colonial-nesting waterbirds) was related to maximum area of river red gum, and to area of river red gum that flooded for four months or longer.	Briggs et al. 1997
In coastal NSW, breeding populations of Intermediate Egrets declined by 98% between 1988 and 1998	Kingsford and Norman 2002 citing Maddock 2000
In the Macquarie Marshes, upstream dam construction (Burrendong and Windamere) has resulted in significant reductions in the number of nests, as well as in the frequency of breeding events	Kingsford and Johnson 1998

2.6 Proportion of nests successful

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
Rapid reductions in flood depth and/or reductions below a threshold depth can cause nest/egg/chick abandonment and reduce the proportion of successful nests. After taking into account the time needed for birds to prepare behaviourally, nutritionally and hormonally for breeding, egg incubation, fledging, and acquisition of foraging skills by juveniles, nest sites and nearby foraging areas need to be flooded for several months, with minimum flood durations dependent on species, pre-flood site conditions, and the season during which flooding began.	Reid et al. 2009 citing Carrick 1962; Jensen 1983; Scott 1997; Kingsford 1998; Leslie 2001 (cited latter 2 refs); Brandis et al. 2011; McCosker 1996; Briggs et al. 1997 citing Miller 1980; Maher 1988, 1990; Marchant and Higgins 1990; Leslie 2001; Marchant and Higgins 1990; Briggs and Thornton 1999
During a 2008 ibis breeding event in Narran Lakes (mostly straw-necked ibis), in one colony 60% of eggs hatched and 94% of chicks fledged, while in another colony 40% of eggs hatched with only 17% of chicks fledging. Statistical analyses found that water depth was a significant variable in determining reproductive success. Rapid falls in water level during the chick stage in the second colony resulted in decreased chick and overall offspring success. (Colony 2 experienced a fall in water depth of more than 30 cm over 40 days, coinciding with chick development. Similarly in the Barmah-Millewa, breeding success declined when there was a fall by 30 cm in water levels at nesting site or within foraging areas (Leslie 2001).)	Brandis et al. 2011
In Narran Lakes during January 2010 following local rainfall, an estimated 20,000 straw-necked ibis began nesting and egg laying but all nests were abandoned following no further rain or river flows.	Brandis 2010
Australian pelicans abandoned their colony on Lake Eyre in 1990 after eggs were laid because the flood was no longer sufficient	Kingsford et al. 1999 citing Kingsford and Porter 1993
Between 1886-2001 in the Macquarie Marshes, there were six nest desertion events directly attributable to rapidly falling water levels, with losses of hundreds or thousands of eggs and chicks.	Kingsford and Auld 2005
Reproductive performance is influenced by changes in water level as little as 0.3m under nest sites or within nearby foraging areas in Barmah Forest	Leslie 2001
A rate of fall of 1440 ML/day/day sustained for 1 month during October, November or December was associated with abandoned breeding at the Barmah Forest Algeboia Plain nest site because of rapid water level recession	Leslie 2001
By reducing the number AND duration of floods, river management has reduced the frequency of 'successful' breeding episodes by more than 80% from the natural precedent in Barmah Forest	Leslie 2001
Temperature (and therefore season and flood timing) can also affect the success of nests	
Breeding success is thought to be higher following a flood in late winter, spring, or early summer, because increasing temperatures favour the production of food sources such as invertebrates and wetland plants.	Scott 1997 citing Briggs 1990, 1994, Maher 1991, Johnson pers.comm
Cold temperatures are thought to affect nest success by causing adult desertion and chick death. This is likely also driven by the effects of cold temperatures on food resources such as emerging invertebrates and plants. Scott (1997) cites a report of breeding abandonment in Narran Lakes in very cold weather in early June, even though there was no change in water level, that resulted in the abandonment of 5000-10000 young birds.	Brandis 2010 citing McCosker 1996; Taft et al. 2000; Scott 1997 citing B. Johnson pers. Comm.

2.7 Fledging rate

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
For successful fledging of most nesting waterbird species to occur, a shifting spatiotemporal mosaic of wetland inundation needs to occur over a lengthy period, e.g. >5 months. After taking into account the fact that breeding is often staggered or staged, and the time needed for birds to prepare behaviourally, nutritionally and hormonally for breeding, egg incubation, fledging, and acquisition of foraging skills by juveniles, nest sites and nearby foraging areas need to be flooded for several months, with minimum flood durations dependent on species, pre-flood site conditions, and the season during which flooding began.	Reid et al. 2009; Brandis et al. 2011; Briggs and Thornton 1999; Leslie 2001; Marchant and Higgins 1990;
Taking egg-laying, incubation of eggs and fledging of young into account, waterbirds need 5-10 months inundation under nest trees to complete breeding. In the mid-Murrumbidgee, most darters, cormorants, herons, egrets, ibis and spoonbills nested at wetlands which retained water permanently in their deeper, open areas. In contrast, most ducks bred at wetlands which dried out fully between floods.	Briggs and Thornton 1999
The great egret requires 7 months of flooding to occur before it begins breeding, and 3 months after that for egg laying, chick rearing and fledging. Consequently the **minimum** duration of flooding required for successful fledging of chicks is 10 months, and maximum reproduction success would probably require 12 months flooding or more. In contrast, some ducks only require 3-4 months of flooding to occur to successfully raise chicks (however a more desirable duration for maximum reproduction success/numbers would be 6-7 months).	Briggs et al. 1994; Briggs and Thornton 1999
The area of nearby foraging habitat is positively correlated to nestling and fledgling survival. By reducing flood size and duration, food availability is reduced for adults, the nestlings they are feeding, fledglings and juveniles (who also need time to learn foraging skills), reducing their survival.	Leslie 2001 citing Butler 1994
During a 2008 ibis breeding event in Narran Lakes (mostly straw-necked ibis), in one colony 60% of eggs hatched and 94% of chicks fledged, while in another colony 40% of eggs hatched with only 17% of chicks fledging. Statistical analyses found that water depth was a significant variable in determining reproductive success. Rapid falls in water level during the chick stage in the second colony resulted in decreased chick and overall offspring success.	Brandis et al. 2011
Fledging success of all three species of egret has been measured in coastal colonies in NSW, with success of Intermediate egret nests in particular being positively related to local rainfall over several months. Baxter (1994) suggested that widespread, long-duration shallow inundation associated with wet periods was particularly important in providing suitable food sources for Intermediate egrets. Nesting success ranged from 51 to 71% over 2 years. In other studies from outside Australia, summarised in Baxter 1994, fledging success ranged from 0.9 to 2.95 young fledged per nest.	Baxter 1994; Arthur 2011 citing Maddock and Baxter 1991;

2.8 Juvenile survival

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
The area of nearby foraging habitat and its availability over time is probably positively correlated with juvenile survival. However quantitative data are lacking in Australia.	Leslie 2001 citing Butler 1994; Marchant and Higgins 1990; Briggs and Thornton 1999
Studies have shown that immature wading birds can be less efficient foragers than adults (Quinney and Smith 1980; Bildstein 1983) and Butler (1994) hypothesised that survival of first-year birds may be dependent on their acquisition of foraging skills, as has been shown for some terrestrial bird species (e.g. Sullivan 1989). If flood waters are only maintained until birds fledge this may mean that suitable foraging conditions to acquire foraging skills are not available in the local area, with potential consequences for juvenile survival.'	Arthur 2011 and citations therein

2.9 Adult survival and longevity

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
Adult survival is critical for population persistence and is dependent on the availability and quality of foraging and refuge habitat, especially during dry periods or droughts. However quantitative data on adult survival are lacking in Australia for most species.	Kingsford et al. 1999; Kingsford and Norman 2002
An age-structured population model for egrets ('generic egret') found that high adult survival rates are critical for population persistence, suggesting that management needs to consider foraging / refuge habitat availability between breeding events.	Arthur 2011
The food resources and other characteristics of drought refuges determine the number of individuals that can exploit the next breeding opportunity. Drought refuges consist of semi-permanent and permanent inland wetlands, coastal wetlands, and other wetlands permanently inundated.	Scott 1997 citing Maher 1991
If water and/or food availability declines at a site, waterbirds may die, move, or choose to change their diet temporarily. There is an energetic conflict between moving to another location or remaining where decreasing resources may become abundant again.	Kingsford and Norman 2002 citing unpublished data and Barnard 1927; Kingsford et al. 1999
For example, about 200 black swans died when aquatic macrophytes declined at Lake Altibouka in in arid Australia, and Australian pelicans sometimes stay and die during bust periods inland when wetlands are drying or dry completely (Barnard 1927). Nearly 1000 dead cormorants were recovered from a concrete tank with 30-50 cm of water after Lake Eyre dried up in 1974. A similar mass mortality of cormorants ocurred on Lake Salisbury some 370 km to the east of Lake Eyre.	Kingsford and Norman 2002 citing unpublished data and Barnard 1927, Kingsford 1989a; Norman 1983; Kingsford et al. 1999 citing Barnard 1927; Blakers et al. 1984; Kingsford et al. 1999 citing Barker et al. 1989, Kingsford unpub data
Australian waterbirds may be longer-lived than their Northern Hemisphere counterparts, which has implications for assessment of their conservation prospects and true status. However longevity data are mostly based on infrequent recovery of banded individuals or captives, and we have limited understanding of average longevity in the wild vs. in captivity.	Reid et al. 2009; Leslie 2001;
Most studies have shown that waterfowl [ducks, geese, and swans] seldom survive more than about three years, though they may live more than 10 years in captivity.	Kingsford and Norman 2002
Straw-necked ibis may live up to 29 years (ABBBS 2009) but the average life span is more likely to be 15-20 years, with adult stage reached after 3-4 years (Marchant and Higgins 1990). Brandis (2010) presented a table of lifespan data gleaned from the ABBS scheme on pages 151-152 of her thesis: Herons and egrets: 7-14 years; Cormorants: 10-26 years; Pelicans: 15-20 years; Ibis and spoonbills: 7-12 years	Brandis 2010 citing ABBBS 2009, Marchant and Higgins 1990; McKilligan 2005

2.10 Diversity and richness

General statements of what is known are presented in blue text, with relevant examples or detail presented in black.

Known relationships	References
The Murray-Darling Basin provides habitat for more than 120 foraging and breeding waterbird species (more than 50 detected by aerial surveys; Kingsford et al. 2013). Approximately 30 species have a large component of their breeding habitat in the wetlands of the MDB. There are geographical gradients in waterbird distribution (and community composition) across Australia and the Murray-Darling Basin, consistent with known biogeographic trends reflecting climate. Generally, some species are largely restricted to the tropics ('northern') with occasional irruptions to the south and inland during drought in the north. Others are more restricted to the south ('southern'), while still others range across the continent as nomads ('continental').	(p.154 Reid et al. 2009, citing Fjeldsa 1985; Kingsford and Norman 2002); Kingsford et al. 2012; Kingsford et al. 2013; Reid et al. 2013; Scott 1997; Frith 1982; Kingsford and Halse 1999; Roshier et al. 2001; Braithwaite 1975; 1976;
Catchment condition affects waterbird community composition - in particular, the abundance of colonial nesting waterbirds (egrets, herons, ibis and spoonbills, darters and pelicans) is positively associated with catchments in moderate to good condition.	Reid et al. 2013; Reid et al. 2009
Different locations support different species/guilds/communities at different times	Kingsford and Norman 2002
Key individual wetlands are consistently important in terms of waterbird diversity and should be prioritised for management of long-term waterbird species diversity/richness. Key individual wetlands are also important for particular waterbird species/groups, e.g. magpie geese or migratory shorebirds. There can be considerable discrimination between wetlands by waterbirds on the basis of water quality and various other habitat parameters.	Kingsford et al. 2013; Kingsford et al. 2012; Kingsford et al. 2012 citing Cale et al. 2004; Kingsford and Porter 1994
Species/group diversity is often greater in intermittently flooded or ephemeral wetlands that experience productivity booms than in static, permanently inundated wetlands	Kingsford et al. 2004; Briggs et al. 1985; Maher and Carpenter 1984; Maher 1984; Crome 1986, 1988; Kingsford and Thomas 2004; Scott 1997; Kingsford and Norman 2002
Species richness is signficantly correlated with abundance, with high species numbers generally occurring on wetlands supporting high numbers of waterbirds	Kingsford et al. 2012
Waterbird community composition changes markedly as a flood event progresses and recedes, mostly because of changes in habitat, food type and abundance, which are in turn driven by flood regime.	Reid and Jaensch 2004; Reid et al. 2009
For example, in Yantabulla Swamp (northern MDB), differences in species assemblages during January 2008 increased as drying accelerated toward the end of the flood pulse, with herbivorous species increasing as expanses of herbaceous plants appeared.	Brandis et al. 2009 citing Kingsford et al. 2008
A diversity of ***breeding*** species requires a range of ***nesting*** habitats	(see Reid et al. 2009 for a list and example species); Leslie 2001; Kingsford and Norman 2002
For example, cormorants, herons and egrets nest in trees along or within rivers and wetlands, Australian pelicans are ground nesters, usually on sandy islands or beaches, and ibis and spoonbills usually nest on emergent macrophytes, including lignum and phragmites. These nesting habitats are dependent on appropriate flood regimes for persistence.	Brandis 2010 (thesis); Leslie 2001; Briggs et al. 1997
Associations of breeding darters, great cormorants, and pacific herons with wetlands containing areas of dead trees (especially river red gum) reflect the preference of these waterbirds for nesting in wetlands with prolonged inundation, even if sufficiently prolonged to kill the trees. Conversely, little black cormorants, little pied cormorants, white-faced herons and yellow-billed spoonbills use wetlands with live trees that are flooded for at least four months.	Briggs et al. 1997
A diversity of ***feeding*** species requires a range of ***foraging*** habitats and foods	Frith 1959; Reid et al. 2009; Kingsford and Norman 2002
The composition of waterbird communities on a wetland often reflects the availability of food. Food items are often consumed according to their availability, with the proportions and types of vegetable matter, invertebrates and vertebrates consumed varying with time and location.	Kingsford and Norman 2002 citing Kingsford and Porter 1994; McDougall and Timms 2001
Waterbird community composition varies depending on wetland type - e.g. large complex floodplain wetlands with a range of habitats vs. freshwater lakes vs. coastal wetlands	Kingsford et al. 2013; Frith 1959

Known relationships	References
e.g. different duck species have different preferences re. rivers, anabranches, effluent streams, storage dams, irrigation channels, billabongs, swamps, temporary floodplain water - usually because of food and nesting habitat requirements	Frith 1959
Large dams such as Burrendong, Coolmunda and Lake Mokoan are associated with piscivorous communities; large floodplain complexes such as Lowbidgee and Macquarie Marshes are associated with large wading birds such as egrets and straw-necked ibis; freshwater wetlands of the inland MDB such as the Paroo overflow lakes, Cuttaburra channels are associated with grey teal, Eurasian coot, and whiskered terns.	Kingsford et al. 2013
Species diversity/richness at a site can be partly determined by bird experience/memory/site attachment	Reid et al. 2009; Leslie 2001
Changes in river and wetland condition, particularly those related to altered flows, are reflected by changes in watebird communities. Wetlands affected by flow regulation have different waterbird community composition than unregulated wetlands	Kingsford et al. 2012; Kingsford and Thomas 1995; 2004; Leslie 2001; Kingsford et al. 2004; Kingsford and Auld 2005; Kingsford and Porter 2009
Significant long-term declines in waterbird species richness have occurred in areas subject to river regulation in the Murray-Darling Basin, but not in the unregulated Lake Eyre Basin	Kingsford et al. 2013
In Barmah Forest, diversityof colonially-nesting waterbirds has declined since the introduction of water resource development. There was a lag of 20 years following construction of Hume Dam, with differences becoming noticable during the 1950's.	Leslie 2001
Significant long-term shifts in waterbird community composition occurred in regulated wetlands of the MDB from 1983-2009, particularly in the Lowbidgee and Macquarie Marshes	Kingsford et al. 2012



3 Knowledge gaps

3.1 Abundance

Knowledge gaps	
Waterbird abundance responses to flows are thought to be relatively well understood for ducks, cormorants, pelicans, and colonially-nesting waterbirds - especially at the local scale. However drivers of abundance of the more cryptic species are poorly understood, spatially or temporally, and our understanding for the more conspicuous groups is based on annual aerial surveys of only a portion of their ranges and limited small-scale local studies. Waterbird movement is an important complicating factor. Kingsford and Norman (2002) rated information on the abundance of species as 'good' in three of the more conspicuous waterbird orders (Anseriformes, Pelecaniformes, Ciconiiformes) but 'poor' for all the others, based on long-term surveys. Questions that remain include:	Kingsford and Norman 2002 citing Braithwaite et al. 1986; Briggs and Holmes 1988; Bayliss and Yeomans 1990; Morton et al. 1990a, 1990b; Norman and Nicholls 1991; Briggs et al. 1993; Kingsford et al. 1994a, 1999b; Halse et al. 1998
Have abundances of the more cryptic species changed over time and/or space? What are the drivers of abundance for these species?	
How do we differentiate spatial shifts in distribution (redistribution of the population) from real population abundance trends through time? How do we model changes in abundance if the scale and frequency of movements is poorly understood? Are basin-scale estimates of abundance/density/reporting rate the best spatial scale of data aggregation at which to analyse trends of whole or substantial proportions of populations?	Reid et al. 2009; Maher and Braithwaite 1992; Roshier et al. 2002
Has there been a greater reduction in waterbird abundance than percent reductions in flows in regulated rivers?	Reid et al. 2009
Are there critiical benchmarks or thresholds in abundance for particular species or guilds?	Kingsford et al. 2013
Does flood clustering (ramping up) result in true increases in waterbird abundance? How much of such increases are a result of movement vs breeding and recruitment?	Reid et al. 2009
What factors govern the presence and abundance of waterbird food sources over space and time, and how do these drive the distribution and abundance of waterbirds?	Kingsford and Norman 2002

3.2 Population size and age structure

Knowledge gaps	
Population sizes for most species are effectively unknown, especially those that are cryptic and/or use habitats other than major wetlands. Species thought to be in low numbers and/or with restricted range are of most concern for conservation but there is usually poor information to determine whether these populations are increasing or decreasing.	Kingsford et al. 2012; Kingsford and Norman 2002
Population age structures and their drivers are largely unknown or poorly documented for most waterbird species in Australia. Other than reductions in total populations we have very little data on waterbird population functioning in general. Even basic biological data such as age at sexual maturity are poorly documented for many Australian waterbird species, although estimates are available based on overseas data and limited local studies (e.g. egrets and straw-necked ibis are estimated to reach reproductive maturity at 18-24 months). The time between successful breeding events and the size of those events are almost certainly critical in determining population age structure. Age structures are likely to change in association with boom and bust periods. They have also most likely changed with the increase in river regulation and subsequent reductions in breeding event frequency and size, perhaps for some species more than others.	Brandis 2010; Arthur 2011; Brandis et al. 2009; McKilligan 1975, 2005; Baxter 1994; Cezilly 1997; Leslie 2001; Scott 1997; Arthur 2011; Kingsford et al. 2013; Roshier et al. 2001;
The impacts of loss of habitat on population sizes and age structures of both resident and migratory waterbird species is poorly understood.	Brandis et al. 2009

3.3 Population boundaries / movements

Knowledge gaps	References
In Australia, we still lack fundamental knowledge on the movements of most waterbird species, including an explicit understanding of the spatial and temporal scales at which individual species interact with their habitats and the triggers for movement - particularly for those using habitats in inland Australia. For the six orders of waterbirds considered by Kingsford and Norman (2002), knowledge of movements was categorised as 'moderate' for Anseriformes and Ciconiiformes, but 'poor' for the other orders. Questions that remain include:	Roshier et al. 2002; Reid et al. 2009; Kingsford et al. 2013; Arthur 2011; Kingsford and Norman 2002
Over what spatial scales do waterbird species populations actually function? Although sufficent data on movements are lacking, it is believed that individuals of most species are capable of dispersing at the scale of the continent, and so populations of most species are thought not to be confined to the Basin. However we have limited understanding of whether waterbird populations in Australia function effectively as one large population, or whether there are subpopulations that function largely independently of each other.	Arthur 2011; McKilligan 2005; Reid et al. 2009
Exactly what cues are being used/detected to trigger long-distance movements? Are atmospheric cues required in addition to environmental flows?	
Are different proportions of a population (e.g. Grey Teal) affected by different movement cues at different spatial or temporal scales? What causes differences in spatial and temporal patterns of movement within and between species?	Kingsford and Norman 2002
What are the wetland connectivity/network and associated movement requirements for juvenile dispersal and survival? How do juvenile birds interact with the landscape? Which sites and habitats are critical for their movements?	Brandis et al. 2009
What proportions or age groups of populations move long distances vs. locally? Do movement distances of young waterbirds differ from those of adults? Apart from recent studies of the Grey Teal (Roshier et al. 2008) there is very little direct data to bear upon the proportions of populations that move great distances vs. locally.	Reid et al. 2009; Kingsford and Norman 2002
Are long-distance movements exceptional, or repeated frequently in a long-lived bird's life?	Reid et al. 2009
Are tropical drainage divisions used as refuges by waterbird species from southern temperate drainage divisions during extended drought?	
How do waterbirds use wetland mosaics both regionally and within a wetland complex? Data on wetland mosaic use could be collected by the marking/tagging of birds, tracking their movements and monitoring their wetland use during and after flood events. This information would provide a greater understanding of the contribution of particular wetlands or mosaic patches to waterbird management. It would also inform management decisions regarding management targets in relation to waterbirds.	Brandis et al. 2009
What is the impact of changes in waterbird movements on plant and invertebrate dispersal?	Brandis et al. 2009

3.4 Breeding initiation

Knowledge gaps	References
Are patterns of breeding initiation at nesting sites changing over time? If so, why?	Brandis 2010
Do undocumented nesting areas exist in Australia?	Brandis 2010
What are the precise effects of seasonal changes in flood timing on waterbird breeding initiation?	Scott 1997
Has there been a greater reduction in breeding effort and success than in waterbird numbers per se? If so, for which species, and why?	Leslie 2001; Reid et al. 2009
Are particular guilds/groups/species being prevented from breeding by lack of suitable nesting habitat? E.g. grebes, swans, coots (species that nest of floating or attached macrophytes in the interior of wetlands). Does the abundance of adults mask a lack of sufficient recruitment to sustain populations?	
Is there a requirement for a natural trigger or cue for breeding initiation? i.e. will an artificial flood produce a breeding attempt if it is not associated with a natural trigger such as rainfall? Does this vary by species/group/guild?	
For which other species do threshold durations of inundation exist for breeding initiation, and what are those thresholds?	
For which species do threshold areas of inundation exist for breeding initiation, and what are those thresholds?	

3.5 Number of breeding pairs / nests

Knowledge gaps	References
Does the length of the preceding dry period affect the number of breeding pairs/nests? Is it a hump-shaped relationship, or something different?	Reid et al. 2009
Does flood clustering (ramping up) truly result in higher numbers of breeding pairs/nests? Is this because they remain in the area and repeat breed, or because they move in?	Reid et al. 2009
Does the number of breeding pairs/nests scale with flood area/extent linearly, or are there thresholds for different species?	Kingsford et al. 2013

3.6 Proportion of nests successful

Knowledge gaps	References
Initiation of breeding may not result in breeding success. However examination of threshold triggers for breeding initiation is much more common that assessment of breeding success of any kind, whether proportion of nests successful, fledging rate, or juvenile survival. While basic information about the reproductive ecology of many waterbird species is known, information is poor for Gruiformes (cranes, rails, crakes and gallinules) and Charadriiformes (sandpipers, stilts and terns), and detailed information is confined to only a few species (e.g. magpie goose, black swan, chestnut teal, Australian wood duck, some cormorants and egrets).	Brandis 2010; Kingsford and Norman 2002 (p. 56) and references therein
What drives the proportion of nests that are successful, other than flood duration, extent and depth? Do drivers vary between different locations? If so, how and why?	
Do rapid reductions in flood depth and/or reductions below a threshold depth directly cause nest/egg/chick abandonment and reduce the proportion of successful nests? It needs to be clarified whether rapid draw-down itself causes colony desertion (as articulated by Leslie 2001) or whether desertion occurs as a result of inundation depth or area declining below some threshold, i.e. distinguishing between an effect caused by the rate of fall (or rate of reduction in flow volumes) or upon reaching a threshold depth or area. Ibis are thought to be more sensitive to depth changes - are there species differences in response?	Reid et al. 2009
Does the length of the preceding dry period affect the proportion of nests that are successful?	
Does flood clustering (ramping up) result in a greater proportion of successful nests?	Reid et al. 2009
How do interactions between waterbird diet and breeding energetics affect the proportion of nests that are successful?	Kingsford and Norman 2002

3.7 Fledging rate

Knowledge gaps	References
Data on waterbird fledging rates and their drivers in Australia are scarce, with some exceptions for particular species or groups in certain locations such as egrets. These data can be difficult to collect in the field for logistical and ethical reasons, however detailed, long-term monitoring is essential if we are to build better predictive models that allow assessment of the potential quantity and timing of water required to ensure successful fledging.	Brandis et al. 2011
Are there thresholds (e.g. duration of wetland inundation or total flow volume) at which the fledging rate plateaus? How do changes in food abundance affect fledging rates, and can these be predicted?	Reid et al. 2009

3.8 Juvenile survival

Knowledge gaps	References
Juvenile survival is not readily quantified from observations at local scales. For example, no estimates of adult, subadult, or juvenile survival are available for egrets in Australia, despite long-term monitoring at a range of nest sites. Consequently modelling of population dynamics in response to flows is difficult, and reliant on estimates from overseas or different species. Some other relevant questions include:	Roshier et al. 2002; Marchant and Higgins 1990, 1993; Briggs 1992; McKilligan 2005; Arthur 2011
What are the main drivers of juvenile survival/mortality, and do these vary spatially or temporally?	
Are there thresholds in depth/extent/area of flooding around the nesting site that affect juvenile survival? Does loss of floodwater immediately after fledging have a detrimental effect on juvenile survival?	Arthur 2011
How important are small to medium sized floods for juvenile survival between the large breeding events triggered by large floods?	Scott 1997
How do juveniles use the landscape around them to move and feed? At what scales do they move and interact, and how do these affect juvenile survival?	
Can co-ordinated broad scale surveys that include marking and tracking of juvenile cohorts provide sufficient data to estimate survival?	

3.9 Adult survival and longevity

Knowledge gaps	References
We currently know very little about survival rates in Australian waterbirds, or how important survival rates are to the maintenance of waterbird populations, or which wetland habitats play a key role for survival of waterbirds. Research needs to be established to quantify adult survival and determine whether it is affected by the availability of suitable habitat between flooding events.	Reid et al. 2009; Arthur 2011; also citing Cezilly 1997; Brandis 2010
No estimates of adult, subadult, or juvenile survival are available for egrets in Australia	Arthur 2011 citing McKilligan 2005
Is recruitment occurring at a sufficient rate (if at all) to maintain populations, considering the relatively long lives of Australian waterbirds? Are any Australian waterbird species facing extinction debts? A time lag of at least 20 years to detect declines was suggested by Leslie (2001).	Reid et al. 2009

3.10 Diversity and richness

Knowledge gaps	References
Which species are 'winners' or 'weedy' species, which have declined significantly in abundance or distribution, and which are likely to decline in the future? (p122 Reid et al. 2009)	
How do patterns of waterbird diversity and abundance at a national scale relate to patterns of diversity and abundance of other aquatic organisms, e.g. frogs, fish, wetland plants?	p.xvii Kingsford et al. 2012
What spatial patterns exist in the distribution of waterbird species at a national scale and how do these relate to wetland area and type, climate, hydrology, landuse and landscape factors, e.g. proximity to other wetlands or urban centres?	p.xvii Kingsford et al. 2012
To what extent can the effects of climate, changes to flow (e.g. river regulation) and other landscape factors, e.g. landuse, be identified in temporal and spatial patterns of waterbird community composition?	p.xvii Kingsford et al. 2012
How vulnerable are wetlands of importance to waterbirds to climate change and water resource development in different regions of Australia in terms of projected exposure?	p.xvii Kingsford et al. 2012
The range of some species has contracted away from the southern MDB to the north, including the magpie goose, brolga, black-necked stork and jacana. It is not clear what the drivers of these range contractions are/were.	Scott 1997 citing Kingsford pers. Comm., Slater et al. 1989
Knowledge of the distributions or ranges of cryptic or rare species is poor	Kingsford and Norman 2002

3.11 Other general knowledge gaps

Knowledge gaps

- There have been very few studies on the impact of other environmental factors on waterbirds besides flow/flooding. These may include climate change, pest species, changes in fish community structures, clearing, grazing, and avian diseases. Greater understanding of these impacts may be acheived through collection of baseline data and then conducting manipulative experiments (Brandis et al. 2009).
- While general waterbird responses to habitat variables are known, quantitative relationships based on empirical data are rare, and where they exist are often based on limited single-site or single-species studies or information from North America, Europe or Africa. Development of quantitative relationships could be done with collection of new field data and through manipulative experiments (Brandis et al. 2009).
- Identification of thresholds of concern requires detailed data for target species - this is lacking for most species of waterbirds, especially those that are not colonial-nesters (Brandis et al. 2009).
- Current gaps in our understanding of waterbirds mean that management policies need to be based on an adaptive management framework, and our ability to predict responses is poor. Further research into waterbirds and the way they interact with the environment will improve this situation (Brandis et al. 2009).
- If we are to manage waterbirds such as egrets for their long-term persistence in the MDB, then we need to understand how the various demographic parameters are likely to influence their long-term persistence (Arthur 2011).
- There is scarce information on moulting in Australian waterbirds, especially regarding timing (Kingsford and Norman 2002).
- There is considerable variation in knowledge status among species, and knowledge is generally poor for cryptic species such as rails and crakes (Kingsford and Norman 2002 citing Braithwaite 1975; Frith 1982; Marchant and Higgins 1990; Maddock 2000).
- What are the mechanisms via which river regulation may affect whole ecosystems (e.g. food webs, feeding and nesting areas) for waterbirds? (p.xvii Kingsford et al. 2012)

Part II Waterbird responses to stressors and threats



4 Habitat loss, fragmentation, and change

Habitat loss, fragmentation, and change are arguably the most important drivers of changes in waterbird abundance, population size and diversity worldwide. In particular, loss of preferred vegetation whether through changes in flood regime, clearing, isolation, land management, fire regime, salinity or weed invasion is a key stressor for waterbird populations and is a major factor associated with decreases in waterbird abundance and diversity (Kingsford and Thomas 1994). Habitat characteristics from local to landscape and continental scales can influence waterbird abundance, including features such as vegetation type, wetland area, flood duration, connectivity (distance to the nearest wetland), and food availability.

Most empirical studies of relationships between habitat and bird populations have focused on the numbers breeding in a particular locality. For practical reasons, these studies have often ignored the sometimes large numbers of non-breeders and have often not considered in much detail processes outside the breeding season such as foraging, dispersal, juvenile survival and adult survival and the habitats required for these. Modelling suggests that in long-lived species, survival of juveniles and adults is key to maintaining healthy populations (Arthur 2011; Sovada et al. 2001). To understand the behaviour of bird populations, and to predict how they might respond to major habitat loss and change, data and models are needed that cover large geographic areas and incorporate both breeding and non-breeding individuals and all critical life stages and habitats (Goss-Custard et al. 1995).

For resident populations, the simplest starting assumption is that populations will decrease in proportion to amounts of habitat lost or degraded. However, to predict the population consequences of habitat loss, we also need to understand the role of density dependence (Fernandez and Lank 2008). Habitat loss is thought to result in increased competition within and between bird species for nesting and foraging space (Goss-Custard et al. 1995). How will populations be affected if more waterbirds attempt to occupy less space during breeding and non-breeding periods? It is likely that body condition and rates of survival will decline due to changes in competition for food and/or intensity of predation at fewer sites. Whether waterbirds starve or emigrate may not be of immediate concern for a particular site, but could have an important effect on that site and on broader populations in the long term. The impact of habitat loss is probably more severe for the periods or areas in which density-dependence is stronger (Fernandez and Lank 2008). To predict the consequences of habitat loss on population size, therefore, the strength of the density-dependence in both breeding and non-breeding areas needs to be known (Fernandez and Lank 2008).

Waterbird species that are highly mobile may be affected by habitat loss occurring long distances from their breeding sites at non-breeding, foraging or refuge sites (Dorfman and Kingsford 2001). Processes in one place influence numbers in another, at the same and at different times of year (Goss-Custard et al. 1995a; b; Goss-Custard et al. 2006).

For example, several cormorant species use seagrass beds in estuaries as critical foraging habitats, and loss of those habitats reduces food availability and consequently bird abundance (Dorfman and Kingsford 2001). For migratory shorebirds, many of the current and future threats relate to changing availability of wintering, stopover and breeding habitats along their migratory pathways (Sutherland et al. 2012). Migratory species differ from other species because individuals depend on multiple locations that may be spread over continents, and individual sites can support substantial proportions of entire populations during the course of annual migrations. The loss of key locations at any point on migratory routes can therefore have far-reaching consequences for whole populations. The capacity of migratory species to alter migratory routes or migration timing in response to environmental changes is not well understood, but the range of species that have recently changed their migration suggest some ability to adapt (Stirnemann et al. 2012; Godet et al. 2011; Moller et al. 2006). Migratory behaviour clearly has the capacity to respond to large-scale environmental changes. However, the success of any shifts in migratory behaviour would also be contingent upon the availability of alternative locations or habitats for the species at risk (Sutherland et al. 2012).

As non-breeding (foraging) habitat is lost, the increasing competition for the feeding areas that remain can increase mortality rates and thus lower population size. Conversely, the resulting reduced levels of competition for territories in the breeding season may raise reproductive rates and increase population size (Goss-Custard et al. 2006).

The effects of habitat loss on ducks, geese, swans and migratory shorebirds have been relatively well researched compared to other waterbird families. Habitat loss, hunting, and exotic introductions are the major causes of globally threatened status in Anatidae (ducks, geese and swans), affecting 73%, 48%, and 33% of threatened species respectively in this group (Green 1996). Although the habitat use patterns of threatened and non-threatened Anatidae are similar, inland lentic wetland and forest inhabitants are most threatened by habitat loss, whereas marine

ecosystem, grassland, tundra, arable land, and scrub dwellers are least threatened. There is an exceptional concentration of 7 threatened, migratory taxa confined to the east-Asian flyway (Green 1996).

The distance between foraging habitats and nesting habitats can influence nesting success, with increased distances and reductions in the area, availability or quality of nearby foraging habitat resulting in decreased success rates (Van de Pol 2010; Zharikov and Milton 2009).

Loss of foraging habitat near breeding sites may influence nest success because of the energetic costs of travel, however this is likely to vary with species, location, foraging habitat preferences or adaptability, and food abundance. A study of the foraging flights of four species of nesting adult wading birds at Lake Okeechobee, Florida, from 1989-1992 during chick-rearing periods Smith (1995) found little evidence that increasing foraging flight distances influenced levels of nesting success and nestling production, except for tricolored herons, for which linear regressions of annual median flight distances versus colony-specific estimates of nest success and productivity revealed significant negative relationships. The latter species preferred to forage in natural habitats, which forced it to travel further. Median flight distances were: Great Egret (*Casmerodius albus*, N = 356) 3.7 km, range 0.1-33.3; Snowy Egret (*Egretta thula*, N = 236) 2.8 km, range 0.1-29.8; Tricolored Heron (E. *tricolor*, N = 82) 2.4 km, 0.1-22.3; and White Ibis (*Eudocimus albus*, N = 286) 2.7 km, range 0.1-33.3 (Smith 1995).

Habitat loss can often be more about site condition, quality and characteristics than clearing or fragmentation. Habitats used by waterbirds vary in quality as a function of interactions between water regime, vegetation type, food abundance, predation danger and competition. Changes in habitat characteristics will often favour some species and individuals while disadvantaging others, leading to changes in both abundance and diversity.

For example, drastic changes have occurred in the Camargue wetlands in France (Tamisier and Grillas 1994). The loss of c. 40,000 ha of natural areas (33,000 ha of wetlands) was related to the extension of agriculture, salt exploitation and industry. On most of the remaining wetlands, management involved division of the marshes into smaller dyked units and large inputs of freshwater, resulting in a decrease in mean water salinity and an increase in duration of flooding. The original unpredictable variation in water level and water salinity, a characteristic of Mediterranean seasonal marshes, has been replaced by predictable permanent freshwater marshes. Consequently the species composition of aquatic habitats has changed from diversified Mediterranean to monospecific continental-type communities. The higher plant biomass favours use by wintering waterfowl which are, in turn, are probably limited by high hunting pressure. These changes result in losses of biological diversity at the intra- and inter-habitat, as well as at the Mediteranean and continental, levels. The authors suggested that new concepts of conservation should be proposed which take into consideration the biodiversity of whole landscapes, including private properties, instead of isolated management plans for reserves only.

Individuals typically vary considerably in how they exploit food resources, and in their susceptibility to predation and interference competition (Durell 2000, Ydenberg et al. 2002 cited by Fernandez and Lank 2008). Habitat quality depends on both benefits and costs, and the best habitat choice for any individual thus involves condition- or state-dependent tradeoffs that balance metabolic requirements, safety priorities, and social status or dominance (Fernandez and Lank 2008). Such individual variations have important implications for the effect the population consequences of habitat loss or change.

Anthropogenic installations or structures in a site or landscape can also affect waterbird breeding and survival.

Detailed postmortem examinations on 167 free-ranging Eurasian Cranes (*Grus grus*) from Germany collected over a 10-year period found that by far the most common causes of mortality were traumatic injuries from collisions with power lines and wire fences (Fanke et al. 2011). Mortality from collisions with power poles and power lines has also been recorded for young cattle egrets in NSW/QLD (McKilligan et al. 1993). In another study the number of breeding duck pairs was thought to be negatively affected by wind turbine presence for wind energy production (Loesch et al. 2013).

Habitat loss, fragmentation and change can cause food shortages and subsequent starvation of nestlings, fledglings and juveniles.

If habitat loss or change occurs, young birds are likely to be more affected more than adults. Young waterbirds can have specific habitat and food preferences, and therefore habitat and associated food loss may have a disproportionate effect on their survival (Nummi and Poysa 1995). Young birds in the nest are dependent on abundant food brought to them by their parents, and loss of foraging habitat during nesting can have significant negative effects. A study of cattle egrets in South Africa found that 66% of eggs failed to become fledged young, and attributed this to chick starvation following food shortage (Siegfried 1972 cited by McKilligan 1987). Starvation of cattle egret nestlings was also an issue for 'younger' chicks in each nest - e.g. the fourth chick - in Queensland Australia (McKilligan 1987).

Invasive plant species have the potential to dramatically alter the availability of waterbird foraging and breeding habitats.

This is most often documented for shorebirds - for example, Ge et al. (2009) found that an introduced species of smooth cordgrass *Spartina altemiflore* had rapidly spread in the Yangtze River estuary in China, invading the bare mudflat and bulrush zones and reducing the availability of high-tide roosting habitat for shorebirds.

Water and land management actions to promote or protect agriculture, industry and residential areas in floodplains and wetlands often conflict with waterbird use of breeding and foraging habitats provided by floodwater.

Nesting waterfowl suffer from chronic deterioration and loss of critical habitats over much of their breeding range because of encroachment by developments (Sovada et al. 2001 citing Petersen and Hogan 1996). The effects can also be very rapid. For example, in Turkey's Gediz delta, urbanization, industrialization, pollution, overgrazing, disturbance, and illegal reed cutting and burning significantly affected the breeding populations of several waterbird species in just 4 years (Onmus and Siki 2013). The mean total number of breeding birds per UTM grid slightly decreased in areas under threat, but significantly increased ($P < 0.05$) in areas without threat. The mean total number of breeding waterbirds significantly decreased ($P < 0.05$) in both the grids under and without threats. Of 30 species, 23 shifted their breeding locations in areas with threatening processes. The total number of breeding pairs of 9 species decreased, 7 species increased, and 6 species were stable (Onmus and Siki 2013).

Twedt (2013) found that proposed levees and pumps to protect cropland along the Mississippi River in Missouri USA would reduce shorebird foraging habitat by 80 %. European declines in shorebird and wader populations have been attributed to alterations of habitat associated with agricultural intensification (Sutherland et al. 2012). Changes in the wetting regime of breeding and wintering habitats are typically one of the first impacts of agricultural intensification. Declines of waders in British uplands have been associated with changes in sheep and game management, afforestation, habitat cover, forest edge exposure, grouse moor management intensity and crow abundance, depending on species (Douglas et al. 2014; Amar et al. 2011). Such changes are predicted to have reduced habitat and food availability and increased predation pressure for a globally unique suite of breeding birds of international conservation importance. Intensively managed grasslands generally have drier soils with reduced prey availability, may be subject to heavy grazing or mowing, and present homogenized habitat at the landscape scale (Sutherland et al. 2012).

Grazing of waterbird habitats can have negative or positive effects depending on species, location and timing

There can be complex relationships between stock grazing intensity, vegetation density and height, habitat availability, nest density, and nesting success of waterbirds (Warren et al. 2008). Consequently grazing is a controversial tool for waterbird managers (West and Messmer 2006). In Europe, the creation and the maintenance of grass cover states suitable for both the objectives of production and wildlife conservation are, in a large extent, determined by the timing and frequency of grazing and mowing (Tichit et al. 2005) and when the impact of grazing is detrimental, it is not so much due to the presence of livestock per se, but rather a consequence of the adopted intensity of land use. Some have reported livestock grazing removes vegetation and thus is detrimental to nesting waterbirds. High-intensity grazing by domestic livestock may threaten waterbird breeding sites through trampling and associated reduction of vegetation cover and diversity (Jansen and Robertson 2005; 2001). In the UK, the abundance of Common Redshank *Tringa totanus* breeding on saltmarsh declined by about 23% between the mid-1980s and mid-1990s, and the decline is thought to have been caused by an increase in grazing pressure (Norris et al. 2004). However, in some locations even low-intensity grazing may reduce productivity and nesting success by causing declines in insect density (Sutherland et al. 2012 citing Székely et al. 1993).

Others argue that the disturbance provided by domestic stock is necessary to maintain vegetation health. Often, this effect and its significance depend on the timing, intensity and duration of the grazing as well as the timing of nesting of the individual waterbird species (Lapointe et al. 2000). It is important that grazing management is considered throughout the year (not only when birds are present on their nesting sites) in order to create the vegetation structure desirable for the waterbird species under question and in accordance with their timing of breeding (Durant et al. 2008). There may be delayed effects of grazing regimes in different seasons (Tichit et al. 2005). Targeted grazing practices may even enhance plant biodiversity, eliciting increases in the number or density of species (Durant et al. 2008). However there is not a single scale of investigation for assessing the effect of grazing on habitat suitability for waterbirds. A variety of habitats at a range of spatial scales (from the site to landscape, basin and continent) is required by waterbirds. Therefore management and grazing practices must be tailored and targeted to sites and species in the context of the appropriate spatial and temporal scales (Durant et al. 2008).

The effects of grazing intensity on waterbirds extend beyond their breeding sites. The condition of adjacent areas is critical for chick rearing and foraging by fledglings or juveniles and for species that need vegetation corridors or stepping stones to move between foraging and breeding habitats. For example, in Europe pastures are favourable for

chick rearing, although meadows are almost never used. If the surrounding fields of an optimal nesting site are mainly made of hay meadows, it is likely that the site will not be used or waterbirds may be forced to make very long trips to find good foraging sites with a greater risk of chick mortality. It is insufficient to give recommendations on grazing management (e.g. in terms of stocking rates or timing) in breeding sites without considering the importance of adjacent or nearby areas for foraging and chick rearing (Durant et al. 2008).

The effects of various grazing regimes can vary depending on location, season, and year, and long-term studies are necessary to avoid confounding of results by other factors. Carroll et al. (2007) demonstrated that rotational grazing can be successfully integrated with provision of summer nesting habitat for dabbling ducks in California USA. Interestingly, although Ignatiuk and Duncan (2001) did not detect greater duck nest success on rotational grazing systems compared to season-long pastures, there was a year x treatment effect interaction wherein nest success differed between years on rotational pastures but not on season-long pastures. The authors suggested that rotational systems could be beneficial if they preserve or improve suitable habitat, attract more ducks from less productive habitats, or increase duckling survival (Ignatiuk and Duncan 2001).

Since individual species have different habitat preferences (e.g. reed-beds and other wetland herbaceous vegetation at the local scale for rail species), there is an obvious risk of habitat loss and tradeoffs at the local scale.

For example, the abundance and diversity of various waterfowl species in semi-arid South Africa varies in relation to season, wetland characteristics and land-use (Raeside et al. 2007). Reduction or elimination of one habitat type to favour another for particular species is usually not a feasible or scientifically sound option in the long term, even for threatened species management (Thompson et al. 2012). For example, Thompson et al. (2012) found that management efforts focusing on removing woody vegetation were unlikely to provide improvements in nest survival rates for grassland-specialist breeding waterfowl in Minnesota USA, except to the extent that such management is necessary to maintain large tracts of grassland.

Landscape scale changes in habitat configuration can also affect species occupancy and abundance.

For example, a negative effect of spatial isolation was found for Water Rails (*Rallus aquaticus*) in Italy (Brambilla et al. 2012). One study on neotropical waterbirds (Guadagnin and Maltchik 2007) found that the most important predictors of presence, richness and abundance of waterbird species at a site were wetland area, rice field matrix permeability, microhabitat richness, wetland connectivity, and wetland isolation. Discriminant analysis showed that fragments richer in species than expected from their wetland areas were found in landscapes with greater connectivity and matrix permeability than the species-poor fragments. The total area of rice fields was not related to the richness, abundance or presence of waterbirds in the wetlands fragments. Wetland area is a critical parameter (see Part I of this report); the relationship between the lost area of the best breeding habitats and the size of population deficits for Mallards and Northern Pintails in the entire Canadian prairie-parkland region was significant for both species ($P < 0.0027$ and $P < 0.0001$, respectively; Bethke and Nudds 1995).

Habitat type, area, and configuration at landscape scales can influence predation rates on nests and young and consequently nest success and fledging rates. Loss in area or fragmentation of wetlands can concentrate waterbirds and predators in the remaining patches of suitable habitat and reduce the abundance of alternative prey species (Cowardin et al. 1983, Sovada et al. 2001).

Appropriate habitat provides both food and shelter resources for nests and young waterbirds and thus plays a critical role in their growth, development, and survival. The effects of habitat patch size on the proportion of nests successful have been studied for ducks in the USA, with some indications that larger habitat patches increase nest success, but results are often inconclusive and highly dependent on interactions between predator species, habitat type, and time (Sovada et al. 2000). Few studies have examined whether and how particular elements of habitat affect fledging rates or juvenile survival. Simpson et al. (2007) investigated relationships of duckling survival rates with distance of overland travel, wetland vegetation composition, water permanency, and surrounding upland vegetation for 116 mallard (Anas *platyrhynchos*) broods in the Great Lakes region from 2001 to 2003. They found that the probability, on hatch day, that a mallard duckling will survive to 55 days was positively related to the proportion of wetland area that was vegetated and negatively related to the proportion of forest cover within 500 m of duckling locations. They found little support for relationships between duckling survival rates and the proportions of grasslands or seasonal wetlands or to distances traveled overland by broods (Simpson et al. 2007). Red foxes (*Vulpes vulpes*) have been shown to select and use different types of plant cover in different landscape types of North Dakota, USA, which in turn influenced duck nesting success (Phillips et al. 2003). While habitat edge-effects on nesting success in fragmented landscapes have been suggested for many bird species, their effects on waterbird nesting success are not well studied and are unclear. In Canadian prairies, large vegetation plots (200 ha) had increasing duck nest success with distance from edge up to 250 m, but then declined. In small plots (50 ha), no significant edge effect was detected (PasitschniakArts and Messier 1996; Pasitschniak-Arts et al. 1998).

Feature Abstract

Bird populations are influenced by many factors at multiple scales. Forcey et al. (2014) evaluated the influences of regional climate and land-use variables on the Northern Harrier (Circus cyaneus), Black Tern (Childonias niger), and Marsh Wren (Cistothorus palustris) in the prairie potholes of the upper Midwest of the United States. These species were chosen because their diverse habitat preference represent the spectrum of habitat conditions present in the Prairie Potholes, ranging from open prairies to dense cattail marshes. Land-use covariates were evaluated at three logarithmic spatial scales (1,000 ha, 10,000 ha, and 100,000 ha) and models constructed a priori using information from published habitat associations and climatic influences. The strongest influences on the abundance of each of the three species were the percentage of wetland area across all three spatial scales and precipitation in the year preceding that when bird surveys were conducted. Even among scales ranging over three orders of magnitude the influence of spatial scale was small, as models with the same variables expressed at different scales were often in the best model subset. Examination of the effects of large-scale environmental variables on wetland birds elucidated relationships overlooked in many smaller-scale studies, such as the influences of climate and habitat variables at landscape scales. Given the spatial variation in the abundance of the focal species within the prairie potholes, the model predictions are especially useful for targeting locations, such as northeastern South Dakota and central North Dakota, where management and conservation efforts would be optimally beneficial. This modeling approach can also be applied to other species and geographic areas to focus landscape conservation efforts and subsequent small-scale studies, especially in constrained economic climates.

Worldwide, there has been a documented increase in the use of agricultural lands and water impoundments for foraging by waterbirds as their natural foraging habitats are lost. This can result in conflict with farmers, depending on the foraging species, and result in expansion of some populations and contraction of others.

Agricultural development has benefitted some waterbird species by converting habitats such as woodlands to grasslands, providing food via crops, and providing habitat, water and food in impoundments. Agricultural areas can provide important foraging habitat for some waterbird species such as ibis and geese, both during and between breeding events. For example, breeding glossy ibis (*Plegadis falcinellus*) will use cultivated grass, wet areas, uncut meadows, and recently mowed hayfields, and are often more common on farms with a high density of cows (> 10 cows per ha) and streams (Trocki and Paton 2006). Nevertheless, species may only profit from agricultural intensification up to some threshold level, and species that cannot persist in intensively managed lands have declined in Europe and in North America (Sutherland et al. 2012).

Water storages and flooded ricefields can provide waterbird habitat, especially in regions with severe wetland drainage and degradation, although they are generally inferior to existing natural wetland habitat (Czech and Parsons 2002; Fasola and Ruiz 1996; Richardson and Taylor 2003; Taft and Elphick 2007; Sizemore and Main 2012). The value of such habitats depends on factors such as distance from breeding sites, feeding and roosting habitat preferences (e.g. water depth and vegetation type or cover), and food availability and diversity. For example, a study of a breeding population of the Cormorant *Phalacrocorax carbo sinensis* in south Bohemia found that out of 749 Cormorants recorded, 92.3% were observed on ponds located in distance less than 10 km from the breeding colony, and ponds with water surface area larger than 20 ha (Musil et al. 1995). The cormorants preferred ponds with water depth of 1-2 m, surrounded by wood and well developed shore line. Interestingly, 87% of cormorants occurred on ponds with Carp *Cyprinus carpio* older than 1 year, and the cormorants preferred fish sized between 100-200 mm (Musil et al. 1995). Thus, changes in the timing, depth, or duration of flooding of water storages or rice fields can affect their use by waterbirds (Sutherland et al. 2012). In Japan, some species reliant on rice field staging sites have declined after the introduction of efficient drainage systems, however the effects are dependent on species habitat and food preferences (Lane and Fujioka 1998; Amano 2009, Amano et al. 2010 cited by Sutherland et al. 2012).

The expansion of particular waterbird species may be related to their ability to use anthropogenic habitats such as ricefields, other crops and water storages. Examples include cattle egrets in south-east Queensland, Australia (McKilligan 1984; 1997) and north American greater snow geese (Gauthier et al. 2005), both of which switched their foraging locations and diets in the 1970s and 1980s - and both of which have significantly increased in population size following the switch. In France, ricefields and other agricultural habitats are commonly used by cattle egrets, an invasive species in southern Europe; whereas little egrets, which are native to the Camargue, tend to select natural freshwater marshes and lagoons (Lombardini et al. 2001). Similarly, in Portugal, the number of cattle egret nests depends on the area of dry pasture and crops within 5-km, while the number of little egrets (*Egretta garzetta*) depends on the presence of freshwater and saltwater habitats (Farinha and Leitao 1996). Compared with other areas

in the Mediterranean, Portuguese heronries are dominated mainly by cattle egrets, possibly related to a comparative lack of freshwater feeding habitats and abundance of dry pasture and crops (Farinha and Leitao 1996).

Habitat losses - and in particular loss of foraging habitat or declines in natural food availability - have been associated with diet switches in some species from their natural foods in natural foraging locations to new foods in agricultural foraging locations (e.g. crops). This can have negative effects for waterbirds that no longer receive sufficient variety or nutrition in their diets (Jefferies and Drent 2006).

The ability of waterbirds to satisfy daily energy requirements in flooded or dry agricultural fields or water impoundments has been little studied, but depends on the individual species. For example, the ability to meet daily energy requirements for Little Blue Herons (*Egretta caerulea*) and Great Egrets (*Ardea alba*) foraging in flooded rice and fallow fields of the Everglades Agricultural Area (EAA) USA was evaluated by Sizemore and Main (2012) during April-June 2008 and 2009. Within flooded fields prey density was measured and foraging sites and random locations were compared. Habitat variables did not differ between foraging sites and random locations. Vegetation cover and prey abundance increased in rice fields over time and were greater in rice than fallow fields. Small prey, dominated by fish, were captured by both species and corresponded to prey sampled. Most wading birds in June were observed in newly flooded fallow fields despite lower prey densities. Little Blue Herons met daily energy requirements for both years; but Great Egrets did not, likely clue to predominantly small prey, increasing vegetative cover in rice fields, and lower prey densities in newly flooded fallow fields. Although Great Egrets did not meet daily caloric requirements, the EAA may still function as an important transitional habitat at a time when foraging resources in the region are limited.

In breeding adult purple herons (*Ardea purpurea*), proportion of foraging time has been shown to be higher in rice fields than on rivers, with the relationship between biomass ingested in relation to foraging time greater on rivers. The success of feeding attempts was similar in both habitats, but most Purple Herons preferred to forage along rivers, and rice fields while a viable alternative are probably not as valuable (Campos and Lekuona 2001).



5 Predation

5.1 Introduction

Predation is a natural component of waterbird population biology. However anthropogenic changes, in particular the introduction of feral predators and habitat alteration, have changed the nature and importance of its impact on waterbird populations (Sovada et al. 2001). Changes in predation may cause prey populations to decline to extinction, or to stabilise at lower levels where they may be more susceptible to extinction from other causes (MacDonald and Bolton 2008). Populations that have declined due to other causes are also more likely to suffer more from predation. Usually more than one factor is involved when predation negatively impacts waterbird populations (Sovada et al. 2001).

Many studies have shown that predation on waterbirds occurs mainly during nesting, and is dominated by egg predation (see next section). Predators also take nestlings or fledglings, or scavenge those that die due to other causes such as starvation. Predation on adult waterbirds is relatively rare, but is probably additive to mortality due to other factors (e.g. hunting, pollution; Sovada et al. 2001 citing King and Derksen 1986, Raveling 1989, Sargeant and Raveling 1992).

However the precise effects of nest failure due to predation on population trends can be difficult to establish because other parameters, such as repeat nesting, juvenile and adult survival and individual movements are also important. Egg and chick survival in nests is much more readily quantified in the field than other population parameters, and this partly explains the large number of studies that have reported nest success relative to those reporting juvenile survival, adult survival or overall population trends (MacDonald and Bolton 2008).

Predators can reduce the survival of waterbirds and consequently population size either through direct predation, or indirectly, by causing adults to desert their nests or foraging sites, competing for habitat or food, or affecting other predators and prey (Cruz et al. 2013; MacDonald and Bolton 2008; Skorka et al. 2014).

For example, evidence from video cameras placed at black-fronted tern (*Sterna albostriata*) nests in New Zealand (Keedwell 2005) suggested predators were a cause of nocturnal and permanent desertion of both eggs and chicks, and that single predators could destroy entire colonies.

Adult waterbirds are most commonly preyed upon by avian and mammalian predators.

In Europe, raptors are key predators of adult waterbirds outside of the breeding season and raptor abundance has increased globally due to protective measures implemented in the 1990s (Kirk & Hyslop 1998, Kjellén & Roos 2000 cited by Sutherland et al. 2012). There have been reports of abbreviated migratory stopovers, inadequate weight gain during the wintering period and predation–starvation risk trade-off, as in the case of Common Redshanks in Scotland, which experience higher mortality risk in cold weather because they are obliged to move from safer, but less profitable, areas to risky foraging areas with more profitable prey (Sutherland et al. 2012; Cresswell & Whitfield 2008). In a study of declining Dusky Canada Geese (*Branta canadensis occidentalis*) in Alaska, radio-monitoring of goslings found that almost all gosling mortality (96%; 81 of 84) was due to predation, with mink (*Mustela vison*) and Bald Eagles (*Haliaeetus leucocephalus*) the most important predators. Bald Eagles are major nest predators and thus appear to play a key role in limiting the breeding productivity of these geese (Fondell et al. 2008).

In Australia, raptors, corvids, and feral predators including foxes, cats and pigs are probably the most abundant and important waterbird predators.

Dingoes are also known to prey on waterbirds (generally coot and swan), with predation rates correlated with waterbird abundance/density (Newsome et al. 1983). There are records of waterbirds preying on each other, usually taking nestlings or precocial young such as ducklings (e.g. silver gulls taking ducklings, Haddon 1987); but occasionally as adults – for example an Australian pelican consuming a grey teal (Crawford 1987). However very little (if any) targeted research has been published quantifying predation of waterbirds in Australia or even identifying dominant predators, let alone separating effects on breeding success, recruitment, demographics or population size. Reports of predation are few and tend to occur in asides within papers or reports on other topics. For example, during a banding and tagging study over six years of young cattle egrets in NSW/QLD, the proximate cause of death or injury was given or suggested for only 14 bird recoveries - these were raptor (4), car (2), dog or fox (3), electrocution (2), colliding with power pole (2) and shot (1). There was no obvious association between cause of death and age of the bird, with juveniles and adults represented in all the above categories of two or more birds. The ages of eight described as 'injured' ranged from 2 to 47 months (McKilligan et al. 1993).

Introduced predators, particularly mammals, may cause local extinction of breeding populations

Introduced Hedgehogs *Erinaceus europaeus* have substantially reduced shorebird abundance on the Western Isles of Scotland (Jackson *et al*. 2004 cited by Sutherland et al. 2012; Calladine et al. 2014), and population declines of several species of waterfowl in North America have been linked with low recruitment attributed to high rates of predation on eggs, ducklings and adults (Cowardin et al. 1985; Sovada et al. 2001).

Invasive predators can have both direct and indirect effects on waterbird communities, including competition for breeding habitat, changes in the spatial distribution of nests and alteration of predation rate by native predators. The overall effects may include population changes in both native prey and native predators as well as alteration of predator-prey interactions (Skorka et al. 2014). For example, a population increase of the invasive Caspian gull at a lake in Poland displaced all the native waterbird species from their preferred breeding islets, forcing them to lower-quality islets close to the shoreline. The latter were frequently visited by magpies, which hunted nests, leading to an up to threefold decrease in nesting success as compared with nests located in the preferred islets in the centre of the invaded reservoir. Thus although direct predation by the invasive Caspian gulls was rarely observed, their arrival resulted in increased predation rates for local waterbird species regardless (Skorka et al. 2014).

Fluctuations in predation pressure on waterbirds may result from fluctuations in the availability of other prey.

Predation of breeding Charadriiformes in braided river systems of New Zealand was significantly greater immediately after rabbit control, suggesting a shift in predator diet immediately after rabbit population declines (Norbury and Heyward 2008). In the Arctic, fluctuations in predation pressure linked to cyclic changes in abundance of lemmings and voles can affect annual reproductive success of entire populations of Arctic-breeding birds, even though they usually occur at a local or regional scale (Ganter and Boyd 2000). Consequently habitat management or control measures put in place for alternative prey, pest herbivores or weeds should also consider potential effects on predators and waterbirds.

The relative importance of waterbirds in predator diets changes with fluctuations in their availability, however some predators are selective in terms of which species, ages, and sizes they kill, and in which habitats.

For example, Geoffroy's cat (*Oncifelis geoffroyi*) appears to change its kill rate with waterbird abundance in Argentina. Distance of waterbird prey before attack and prey size were also significant predictors of waterbird consumption, with the cat preferring large prey close to vegetation cover (Canepuccia et al. 2007). In Mozambique, the diets of feral cats (*Felis catus*) shifted dramatically from insects, rats and mice outside the tern breeding season to primarily terns when terns were breeding (Peck et al. 2008).

Agricultural intensification may lead to increased rates of predation on waterbird nests.

Possible mechanisms include: smaller and more dispersed waterbird populations may be less effective at deterring predators by mobbing; anti-predator vigilance may be compromised by changes in habitat; and nests may be more vulnerable to predation due to reductions in suitable cover. Additionally, numbers of some nest predators may have increased in response to agricultural intensification (MacDonald and Bolton 2008). In North America, predation is most severe where a large proportion of the landscape has been converted to cropland (Sovada et al. 2001). The population declines of waders in Europe are widely considered to have resulted from habitat loss and degradation due to agricultural changes, and recent empirical evidence suggests that levels of predation on wader nests are unsustainably high in many cases, even in some situations where breeding habitat is otherwise favourable. Sutherland et al. (2012) reviewed the published and 'grey' literature on nest predation on waders in Europe and quantified the relative importance of the major predators.

5.2 Predation on nests and young

Nest predation is well known to be of great importance to birds, not only in terms of population regulation, but also in nest placement and distribution (McKilligan 1987) and nesting density (Sugden & Beyrsbergen 1986; Dorfman and Read 1996).

Although the *distribution* of waterbirds and their abundance is often related to abundance of food, fluctuations in population size can be substantially affected by nest predation (Grant 1970; Paine et al. 1990 cited by Dorfman and Read 1996). Numerous studies have drawn attention to high rates of nest predation as responsible, at least in part, for low productivity in waders outside Australia (MacDonald and Bolton 2008 and references therein). In Europe, nest predation rates are so high in some cases that, without immigration, population decline is inevitable, even if chick survival rates were high. Adult survival rates are available for most European wader species, and of the studies that measured number of chicks reared to fledging, few have reported rates high enough to maintain a stable population

based on these or similar survival rates, at least in most years. Many report clutch failure rates of over 50% attributable to predation alone (MacDonald and Bolton 2008 and references therein).

Most nest predation involves eggs rather than chicks (though the latter is rarely measured). Consequently it can be difficult to ascribe changes in populations definitively to changes in nest predation, as other parameters can have a large influence.

Many waterbird species will re-nest, either as multiple brooding or in response to nest failure, which may compensate, to some degree, for low nest survival. Chick survival is an important demographic parameter affecting productivity that is more rarely measured than nest (egg) survival. Despite high levels of nest (egg) predation, chick predation was considered to have had more impact on wader productivity in meadows in the Netherlands, and productivity of Avocets in the Wadden Sea coast of Germany was not related to hatching success, but was positively related to chick survival (Teunissen *et al*. 2005 and Hötker & Segebade 2000 cited by MacDonald and Bolton 2008). Sutherland et al. (2012) suggest that future studies quantify chick survival, as well as examining the predator community, wherever possible. In addition, while predator removal usually results in increased survival of young such as ducklings, this is not always the case, and the effects of predators may be outweighed by the effects of habitat and food availability (Amundson and Arnold 2011). Consequently the latter variables are also essential co-variates for monitoring programs and analyses.

It is possible that in some situations nest predation is compensatory rather than additive. If young that would have hatched from predated clutches are doomed to death from other causes, or if fledged birds cannot find suitable breeding habitat, then reducing nest predation will not be sufficient to increase population sizes. However in some situations, stable populations are not possible at current rates of nest predation, regardless of other factors. In these situations, reduced nest predation may be considered necessary, but not necessarily sufficient, to stabilize/improve population trends (MacDonald and Bolton 2008).

Factors that may influence rates of predation on nests include time of season, habitat type, habitat management, nest concealment, nest density, distance to habitat features used by predators (e.g. perches, nests, dens, shelter), and of course, predator abundance. There is conflicting evidence for most of these, and it is likely that the influence of each on rates of predation is dependent on location and species (MacDonald and Bolton 2008; Sutherland et al. 2012).

Waterbird eggs and chicks may be preyed upon by avian, mammalian, or reptilian predators – and occasionally fish

For example, during surveys of magpie goose nests in the Northern Territory, the failure of 42% of marked nests (prior to inundation losses) was attributed to predators, including whistling kites *Haliastur sphenurus*, Torresian crows *Corvus orru*, water rats *Hydromys chrysogaster*, and water pythons *Lialis fuscus* (Whitehead and Tschirner 1990). Whistling kites were frequently seen harassing incubating magpie geese. There were earlier reports of even higher predation losses of magpie goose eggs (72% of eggs; Frith and Davies 1961). Similarly, long-term studies of radiomarked North American wood duck (*Aix sponsa*) ducklings and females that nested in artificial structures and used floodplain palustrine, riverine, and lacustrine wetlands in Mississippi and Alabama (Davis et al. 2009) found composite estimates of duckling mortality rate for the brood rearing period across years and areas were avian (0.46; n = 155), aquatic predators (0.23; n = 79), snakes (0.06; n = 2 1), mammals (0.05; n = 18), exposure-related (0.02; n = 7), and unknown causes (0.13; n = 44). Often the composition of the nest predator community depends on landscape type, with mammals typically predominating in wooded or forested landscapes and birds in agricultural areas of Europe and the USA (Padysakova et al. 2010).

Fish and ducks often belong to the same local food web, and some studies indicate that there is a general negative effect of fish on breeding ducks. This pattern has so far been addressed mainly within the framework of competition for common invertebrate prey. However predation by large fish has also been suggested as a driver of settlement and abundance patterns in ducks (Dessborn et al. 2011). Dessborn et al. (2011) found that two duck species spent less time on lakes with predatory pike during the brood-rearing season, and observed fewer ducklings of one species on those lakes, however there was no causal link established between pike predation and duck behaviour or duckling survival, and pike abundance did not affect the selection of lakes as nesting habitats by ducks.

Corvid species (e.g. ravens, crows) are major predators at waterbird nests worldwide and are regularly associated with waterbird nesting colonies.

Corvids are known to adaptively prey on waterbird eggs and young in Australia, reducing nest success or even causing nest failure of entire small colonies. For example, a small colony of Little Pied Cormorants *Phalacrocorax melanoleucos* in Sydney failed to raise any young in 1993, and this was attributed to egg predation by Australian Ravens and Little Crows (Dorfman and Read 1996). Australian Ravens have been observed to have an adaptable hunting strategy when preying cooperatively on young Great Egrets (Baxter 1988 cited by Dorfman and Read 1996) and to occupy areas near egret colonies, waiting for the opportunity to prey on nestlings. Their ability to take effective advantage of small windows of opportunity may explain corvids' regular association with cormorant nesting colonies (Dorfman and Read

1996). A small number of corvids can potentially have a large impact on nesting populations. For example, in the summer of 1981-82, a pair of Torresian Crows *Corvus orru* took 54 eggs from a nesting colony of Cattle Egrets *Ardea ibis* in south-east Queensland, accounting for 46% of eggs lost that season (McKilligan 1987). The apostlebird *Struthidea cinerea* has also been observed to peck into cattle egret eggs in Australia (McKilligan 1987).

An investigation of nest predation and associated foraging behaviors by resident Common Ravens (*Corvus corax*) in the San Francisco Bay area, California USA (Kelly et al. 2005) indicated that occupation of heronries, predation of Great Egret (*Ardea alba*) nests, duration of patrol flights, landing rates, and number of interactions with ardeids varied with the productivity of resident ravens. Annual increases in raven predatory behaviors were consistent with increases in foraging experience for a few to several years after ravens became resident at colony sites. However, overall nest predation did not increase at three sites from 1999-2004, and at one of these sites, predation did not differ from levels measured before ravens were resident, suggesting that ravens may have interfered with the nest predatory activities of other species. Ravens at one colony site obtained most or all of their energy needs from the heronry. Predation of Great Egret nestlings was most likely 14-29 days after first hatch, when parental attendance begins to decline. Regional monitoring of heronries indicated highly variable rates of nest predation by Common Ravens and a low overall presence of ravens, even though ravens occurred throughout the region (Kelly et al. 2005). Nest predation by *Corvus corax* has been linked to the decline of the Doublecrested Cormorant *Phalacrocorax auritus* in the central United States of America (Grant 1970; Post 1988 cited by Dorfman and Read 1996). At Tatoosh Island, Washington, USA, a population of Pelagic Cormorants P, pelagicus and Common Murres Uria aalge increased markedly as nest predation by Northwestern Crows *C. caurinus* diminished when Peregrine Falcons *Falco peregrinus* immigrated to the area (Paine et al. 1990 cited by Dorfman and Read 1996).

Mammals are also critical and often more cryptic predators of nests.

Predators of nests may vary greatly between sites, even where habitat and management appear similar (Grant et al. 1999 cited by MacDonald and Bolton 2008). In Europe there is growing evidence from remote monitoring devices, where bias is minimized, that in many situations the majority of predation occurs at night and is attributable to mammalian species. In Europe, the use of temperature loggers initially, and nest cameras more recently, has shown that the widely held belief that birds (particularly corvids) are the major predators of wader nests is frequently not true. The limited quantitative information currently available suggests that the most important nest predator species include Fox, Badger and Stoat (MacDonald and Bolton 2008). A number of mammalian predators also have recently increased in abundance in Europe and these can have an impact on breeding success (Smith *et al*. 2010; Fletcher *et al*. 2012 cited by Sutherland et al. 2012). Black rat predation is an important factor affecting the breeding performance of great egrets in the Hara Biosphere Reserve in the Persian Gulf (Neinavaz et al. 2013).

Changes have occurred among predator communities and populations in many breeding areas, which have adversely affected waterbird populations (Sovada et al. 2001). Often these changes are driven by habitat change. A change in predation is often one of the main mechanisms via which land use or habitat change affect waterbird breeding success (Douglas et al. 2014).

The replacement of larger predators (e.g., gray wolf, coyote) by smaller species, which often are more numerous, has been demonstrated as being detrimental to nesting ducks (Johnson and Sargeant 1977, Sargeant et al. 1984, Sargeant et al. 1993 cited by Sovada et al. 1995). In North America (as in Australia) an increase in numbers of red foxes has been a particular problem, since they are a principal predator of nesting ducks and eggs (Johnson et al. 1989, Sovada et al. 1995). Johnson and Sargeant (1977) demonstrated that predation on nesting mallard hens by red foxes may be responsible for the imbalanced sex ratio common in that species. Sovada et al. (1995) demonstrated that nest success was greater in areas occupied by coyotes than in areas with foxes, and it is possible that this principle would also apply to the replacement of Australian predators such as dingoes and thylacines by red foxes and feral cats. Similarly, changes in the avian predator community triggered by anthropogenic activities may have affected predation rates on Australian waterbirds as they have for North American species.

Water depth under or around waterbird nests significantly affects predation rates and consequently nest success.

For example, significant losses of waterbird eggs and young can occur at sites previously isolated that are made accessible to predators, such as through drying up of water around trees or islands (Sovada et al. 2001). In a study of nest success of Ferruginous Ducks *Aythya nyrocaas* in Hungary (Purger and Meszaros 2006), as water levels surrounding nests decreased, nests became more accessible to Wild Boar *Sits scrofa* and other land mammal predators, which increased the rate of predation. The authors suggested that maintaining water levels would not only decrease the predation rate of nests, but would also maintain feeding areas for ducks. Wild Boar were the main cause of clutch loss in this area, and therefore by management measures, such as a reduction in their abundance or attracting them away from potential nesting sites by providing food elsewhere, the breeding success of the Ferruginous Ducks may be further improved. Predation was also the major cause of nest failure in a study population of Great Bitterns in Poland (Polak 2007). Although no relationship between nest site vegetation type and daily nest

survival rate was found, a logistic regression model indicated significant effects of water depth and vegetation cover on nest predation. There was a tendency towards better success for nests in dense emergent vegetation with higher water depth. The survival of nests at edges and in the interior of reed beds was similar.

Outside of Australia, the importance of flooding to birds that nest in floodplain forests is relatively poorly understood. A study of cavity-nesting Wood Ducks (*Aix sponsa*) in the Mississippi River floodplain (Nielsen and Gates 2007) found that nests were depredated prior to and after, but not during, a four-week flood, and in general nest predation was lower and nest success was higher in the floodplain during floods. Flood conditions did not significantly affect nest success or nest predation rates in the adjacent upland, but nest success in upland forest was higher than in the floodplain during dry periods. The authors concluded that historically, spring flooding of riparian areas may to some extent have mitigated nest predation in floodplain forests (Nielsen and Gates 2007).

Colony size, nest location, nest dispersion and nest density within the colony can all affect predation rates. Usually, predation rates are higher in smaller colonies and at the periphery of colonies (Angulo-Gastelum et al. 2011).

The relative effects of nest density vs nest dispersion (nearest-neighbour distance) on predation rates of nests have been debated for some time, with some variation in results depending on location and species (predators and prey). For example, Ringelman et al. (2012) found no evidence for density-dependent predation on artificial duck nests at the 1-ha scale, whether predation rates were very high or moderate. They also found little evidence for dispersion-dependent predation on artificial nests, however for natural nests, nest survival increased with shorter nearest-neighbor distances, and neighboring nests were more likely to share the same nest fate than non-adjacent nests. They suggested that local nest dispersion, rather than larger-scale measures of nest density per se, may play a more important role in nest predation. Similarly, despite differences in local predator communities, Padysakova et al. (2010) found no evidence of different survival rates of artificial solitary nests and nests placed in high-density patches in either agricultural or forested areas. Ackerman et al. (2004) found no evidence of density-dependent predation on duck nests at any scale of analysis, in contrast to a number of previous studies. They suggested that variation among geographical locations in the degree to which predation is density-dependent may reflect the composition of the predator community and the availability of alternate prey.

5.3 Behavioural adaptations to predation

Many waterbird species nest on islands, islets, hummocks or trees, often surrounded by water, that improve visibility for incubating birds and limit access by mammalian predators, especially foxes

Nest initiation on such sites may be delayed until they are surrounded by water (or ice-free in freezing latitudes), inhibiting mammalian access during nesting (Spaans et al. 1998; Liljesthrom et al. 2014). Inland nest sites are accessible to most predators, but often these nests are dispersed and well concealed, reducing the chance of egg depredation. Nest sites near water and foraging areas probably increase survival of young and adults.

Adults of most nesting waterfowl are very attentive to their nests, especially after egg laying, and many actively defend nests and care for young.

Some species have developed antipredator behaviors, such as injury displays to divert predators from nest sites or feigning death. Species nesting in dense colonies may benefit from deterrent effects of nest defense by colony members and communal brood rearing. The increase in prey abundance associated with colonies probably both attracts and minimises the impact of predator species (Sovada et al. 2001).

Some species of waterbirds nest in association with raptors or other species

Nesting in the vicinity of a potential predator may garner benefits from its nest defence behaviour, a situation referred to as “predator protection” (Richardson and Bolen 1999 cited by Jones et al. 2013). For example, the Great Blue Heron *Ardea herodias fannini* in the Pacific northwest appears to have modified nesting behaviour in response to the strong recent recovery of the Bald Eagle (*Haliaeetus leucocephalus* population. Previously undescribed, herons now often nest in close association with some breeding eagles, even though eagles depredate heron nestlings, are implicated in the recent reproductive decline of herons, and may induce abandonment of heron breeding colonies. We tested the hypothesis that breeding herons gain protection from the territorial behaviour of eagles. Natural observations and simulated incursions showed that nesting eagles actively repel other eagles within at least 250 m around the nest site, thereby establishing a relatively safe place for herons to nest. Surveys showed that 70% of heron nests and 19% of heron colonies were located within 200 m of eagle nests with high reproductive success. These herons had greater reproductive success than those nesting far from eagle nests (Jones et al. 2013). Redbreasted Geese (*Branta ruficollis*) may nest in close association with Snowy Owls (*Bubo scandiacus*) or Peregrine Falcons (*Falco peregrinus*) that aggressively defend the area around their nests from arctic foxes, which are effective predators of ground-nesting

birds. Geese thereby gain safety, but occasionally lose nestlings to these raptors, or may even fall prey themselves (Quinn et al. 2003; Jones et al. 2013).

Nesting waterbirds may change their behaviour depending on their assessment of predation pressure at a particular location.

For example, increased nest defence of upland-nesting ducks has been documented in response to experimentally reduced risk of nest predation (Dassow et al. 2012). Parent birds take greater risks defending nests that have a higher probability of success. In the latter study, because ducks modified nest defence in the breeding season immediately following predator removal, the authors suggested that ducks may be able to assess predator abundance indirectly (e.g. by UV reflection from urine) rather than by seeing or interacting directly with the predators (Dassow et al. 2012; Colcherc et al. 2010). Other studies have also suggested that nesting ducks will assess predator abundance via mammalian urine and choose where to nest accordingly in order to avoid predation (Eichholz et al. 2012). Ducks may be able to detect mammalian urine either by ultraviolet light reflectance or by odor. Eichholz et al. (2012) simulated increased predator abundance on experimental plots by using Red Fox (*Vulpes vulpes*) urine to make artificial scent marks and used water in a similar fashion on control plots. On 16 pairs of plots over 2 years, fewer ducks nested on experimental plots than on control plots (97 vs. 143 nests).

Waterbird responses to predation are often species-specific.

For example, Cruz et al. (2013) found species-specific responses to both river flows and predators by four species of Charadriiformes in New Zealand. They cite previous studies on multiple bird species that have also found species-specific responses by Charadriiformes to habitat alteration (Garvey et al., 2013), water extraction (Nebel et al., 2008), and predation (Cresswell and Whitfield, 2008), and caution against the use of the indicator species concept for birds without prior evaluation of its suitability for a given system.

Feature extract

'Parents of altricial young with biparental care face a dilemma: they can choose to guard their nestlings against predation with their physical presence at the nest and ensure the fledging of at least one chick, or they can choose to leave the young unguarded so they can forage for more food, increasing provisioning rates and the probability of fledging more young. In group-nesting birds, parents may be able to dilute the risk of nest predation while leaving the nest to forage if they leave the nest unguarded when there are many other unguarded nests in the colony. The complexity of the parental decision to leave a nest unguarded is highlighted by variation in the chicks' age, within a nest and within the colony, and by differences in effects on nest success and nest fecundity. Our results suggest that the incremental fitness advantage of reducing predation risk by continuing to guard the nest is eventually outweighed by opportunities to reduce nestlings' risk of starvation. The effect is likely to reduce fledgling production within a colony when less food is available, which reinforces the importance of healthy and productive wetland ecosystems in maintaining viable populations of the Great Egret.' (Colcherc et al. 2010).

5.4 Predator identification

There are multiple methods for identification of nest predators. Information on the identity of nest predators is vital to inform management intended to decrease levels of nest predation. Nest predation is not easily observed but the means of identifying nest predators and of quantifying the contribution of the various predators of wader nests have improved in recent years (MacDonald and Bolton 2008 and references therein). In particular, nest cameras offer a highly accurate and minimally biased method of identifying and quantifying nest predators. Camera studies have often been conducted in combination with nest temperature loggers. In Europe, these methods have indicated that nocturnal mammalian predators make significant and previously unrecognised contributions to waterbird nest predation.

Many studies on nest predation have used artificial nests and eggs – particularly for ducks. However there are problems in applying the results of artificial nest studies to real nests. Predation rates on real and artificial nests can differ in unpredictable and inconsistent directions, primarily because each type can attract different predators for different reasons. This finding has been borne out in several studies of predation of artificial nests (MacDonald and Bolton 2008 and references therein). The use of artificial nests, the effects of human disturbance, the composition of the predator community and the availability of alternate prey may be responsible for conflicting results among studies and geographical locations in the degree to which nest predation is density-dependent (Ackerman et al. 2004; MacDonald and Bolton 2008). Anthony et al. (2006) found that artificial goose nests with cameras had higher rates of abandonment than natural nests, especially during laying. Abandonment rates were reduced by deploying artificial

eggs late in laying and reducing time at nests. Predation rates for goose nests with cameras were slightly lower than for nests without cameras. Wax-filled artificial eggs caused mortality of embryos in natural nests, but were better for identifying predator marks at artificial nests. Use of foam-filled artificial eggs in natural nests was the most cost effective means of monitoring nest predation.

Transmitters (radio, satellite) attached to birds can provide valuable information on mortality rates and causes, including predation; however they must be applied with consideration to the age and size of the individual birds. In a study of declining Dusky Canada Geese (*Branta canadensis occidentalis*) in Alaska, daily survival rate for goslings to 28 d of age was lower (0.011; 95% CI 0.002-0.024) for those with radio transmitters than for those without, but did not differ for older goslings (29-45 d). Although finer resolution in the timing of the transmitter effect within the first 28 d was not possible, the authors found that, by limiting the sample to goslings that survived until after 2-3 d posthatching, support for a transmitter effect was much reduced. Younger, smaller birds are inherently more vulnerable than older birds to transmitter effects. In addition, the process of radio-marking may have delayed the departure of goslings from nests and increased their risk of mortality shortly after hatching. Although radio transmitters may often be the only practical means for determining causes of mortality for young waterbirds, the authors suggested caution in using transmitters because of their potential negative effects, particularly during the first few days after hatching (Fondell et al. 2008).

Analysing DNA from eggshell and carcass remains is a relatively new tool in wildlife research and management that can identify predator species. However it may not indicate their relative importance, because predator species are not equally detectable using DNA, and predator DNA is not always left at the predated nest. A study of predators at black-fronted tern (*Chlidonias albostriatus*) nests on the Wairau braided riverbed in Marlborough, New Zealand (Steffens et al. 2012) found that video footage was essential to identify predators at most nests, because predators often left no eggshell or other remains as evidence, and predator DNA was not always detectable even where predation had occurred or evidence remained.

5.5 Predator management and its effects

Predation control issues are controversial and complex because they raise questions involving humane treatment of animals and ethical concerns about management of one species in favor of another, particularly for native predators. Ultimately, predation management must be accepted by the public, whose support is critical to the maintenance of waterbird populations, and contribute to long-term management goals (Sovada et al. 2001).

If the goal of predator management is to enhance or maintain population size, evaluation should focus on changes in recruitment. Other measurements (number of predators removed, amount of nesting habitat planted, number of wetlands restored) are easy and appealing to document, but may have little relation to actual number of offspring recruited (Sargeant et al. 1995 cited by Sovada et al. 2001).

Intensive management designed to keep predators away from nests can be grouped into methods that physically separate or conceal prey from predators, those that alter foraging behavior of predators or their food availability, and those that affect predator distribution or abundance (Greenwood and Sovada 1996).

In North America, most methods are aimed at protecting the nesting adults and eggs from mammalian predators; few focus on avian predators or enhancing survival of young. Methods appropriate for the local scale include isolating mechanisms (nesting structures, barrier fences, and moats) and mechanisms that alter predator foraging (conditioned taste aversion, supplemental foods). The former are effective in increasing nest success, although they can be resource- and labor-intensive (Lokemoen 1984). The latter have not proven to be effective (Greenwood et al. 1998, also see Greenwood and Sovada 1996). Methods that influence predator distribution, such as managing canid populations to favor coyotes and reduce red fox populations, have potential to positively affect nest success over large areas of landscape (Sovada et al. 1995). Several studies have shown lethal methods (e.g., toxicants, traps, shooting) to be effective in controlling predators and increasing duck nest success (Duebbert and Lokemoen 1980, Greenwood 1986, Sargeant et al. 1995, Garrettson et al. 1996). Use of toxicants to reduce predator numbers could be especially effective over large areas of the landscape, but effects on nontarget species are of great concern (Sovada et al. 2001). In unstable and disturbed environments, where wetland conditions change continually, the challenge is to know when, where, and how to apply existing options to manage nesting habitat and predation. To make informed decisions, managers need to know habitat conditions, predator community composition, and nest success rates at the landscape scale (Sovada et al. 2001).

Predator management can improve waterbird nesting success, often beyond levels believed necessary for population maintenance, but in most cases is likely to be restricted in its benefits to the local area (Pieron et al. 2012; Pieron M. R. & Rohwer 2010) .

If recruitment, survival of breeding females, and/or breeding site fidelity is increased on predator-reduced sites, then local breeding populations may increase in subsequent years. Excluding large mammalian predators from wetlands with predator-exclusion fencing has been shown to improve waterbird nest success and productivity, allowing breeding success to exceed the levels required for population recovery (Malpas et al. 2013). However such fencing can be prohibitively expensive when managing large or remote sites. In areas where nesting habitat is scant and predation rates are high, it makes little sense to attempt to attract more birds through wetland restoration or to increase nesting habitat without undertaking significant long-term predator control. Some landscapes are so severely altered that it would be impossible to restore enough nesting habitat to return nest success to sustainable levels. Under these conditions, it may be more appropriate to isolate areas of nesting habitat from predators or reduce predator populations (Sovada et al. 2001).

In contrast, Pieron et al. (2013) found limited and equivocal evidence that breeding populations (number of pairs) of waterfowl in North Dakota increased over time following predator management, despite 1.4- to 1.9-fold increases in nest success. Possible reasons suggested included the fact that some species do not show extensive natal or breeding philopatry; lack of recruitment into the population; and territorial behaviour. They suggest that managers should not assume that increased production as a product of elevated nest success will be compounded over years.

Assessing the effectiveness of predator control is difficult, but where data are sufficient, using population viability analyses (PVA) which identify the population growth rate (lambda) and extinction risk of threatened species may offer one solution (Whitehead et al. 2010). PVA provide the opportunity to compare the relative effectiveness of various management options and can identify knowledge gaps to prioritize research efforts. For example, population viability analyses of a New Zealand riverine duck (whio, *Hymenolaimus malacorhynchos*) found that populations with no predator control and low productivity will rapidly decline to extinction. Increasing productivity through predator control increased population viability but populations still showed a declining trajectory. A perturbation analysis showed that the growth rate of whio populations was largely driven by adult survival. Therefore, future research should target obtaining more robust estimates of adult survival, particularly how it is affected by predator control. Overall, their analysis indicated that large-scale predator control increases the short-term viability of whio populations but is insufficient for long-term population persistence (Whitehead et al. 2010).

Specialisation by individual predators at particular sites and on particular species, nests and life stages has been recorded (Cruz et al. 2013).

Therefore predation rates may not relate directly to overall predator abundance if particular predator species or individuals cause a disproportionate amount of predation events by specialising on prey types. Consequently there are complexities in evaluating the benefits of predator management on multiple prey species in the same ecosystem. Consequently multiple authors have called for studies in conjunction with control programs that assess multiple factors, including multiple bird species and life-stages (Cruz et al. 2013).

Nest predator responses to different habitats are complex, taxon specific, and context dependent. Conservation efforts for waterbirds may need to be customised according to the nest-predator species primarily responsible for local nest mortality and the nature of the landscape mosaic (Padysakova et al. 2010).

Nest predators may immigrate into vacancies created by predator removal.

Consequently predator removal should be applied intensively over a larger geographic area in order to be a viable management strategy (Meckstroth and Miles 2005).



6 Climate change

Changes in temperatures, the timing, intensity, amount and duration of precipitation, and the frequency and severity of extreme weather events all have the potential to influence waterbird populations positively and negatively, and directly and indirectly (Sutherland et al. 2012; Chambers et al. 2005). Climate change already appears to be affecting population movements such as migration and dispersal, as well as breeding initiation and success and survival between breeding events. Chambers et al. (2005) reviewed the effects of climate change on birds that have been documented or predicted, with particular reference to Australian species. Potential impacts included changes in geographic range, distribution, movement patterns, morphology, physiology, abundance, phenology and community composition. They found that the evidence suggests that these changes are already happening, both overseas and in Australia, but more research is needed to determine their extent and how to conserve birds in the face of climate change. Relative to the northern hemisphere, little is known about the effect of climate change on southern hemisphere birds, although the impact could be significant. Management options suggested included promoting adaptation and resilience, intensive management of sensitive species, and improved planning for mitigation techniques and monitoring.

In Australia's Murray-Darling Basin, warmer temperatures, reduced rainfall, increased evaporation and decreased surface-water flows are expected to occur within the next few decades. Importantly, such changes will reduce the availability of both breeding and foraging habitats for waterbirds by reducing flood frequency, duration, depth and extent, as well as changing vegetation composition and condition. Critically, if breeding opportunities do not occur with sufficient frequency and size, recruitment may not be sufficient to maintain some waterbird populations (Arthur 2011). We have very few data with which to explore the potential impacts of such changes via modelling.

Unfavourable weather events can negatively influence breeding success in a variety of ways.

Depending on the nature and timing of severe weather, waterbirds may opt out of breeding entirely or suffer population losses through decreased clutch size, reduced hatching success, increased juvenile mortality, or even adult mortality (Smith et al. 2010; Duff et al. 2011). Cold temperatures, heavy precipitation and strong winds in particular can depress growth rates in waterbird chicks, thereby reducing chances of survival (Kasprzykowski et al. 2014).

General correlations between Arctic weather, particularly spring temperatures, and overall breeding success of Arctic bird populations have been well established (Ganter and Boyd 2000). Populations of precocial birds seem to be especially dependent upon climate during the breeding season, while populations of altricial birds covary with climatic variables during the nonbreeding season (Sæther et al. 2004 cited by Van de Pol et al. 2010). However, little is known about the relative importance of the various mechanisms that may depress breeding success. Because of the cost and logistical difficulties involved, long-term studies of the population ecology of waterbirds that provide detailed information on breeding parameters and their relation to weather patterns are relatively scarce. Detailed information on the relationship between weather and reproduction is needed as input into scenarios for assessing the potential impact of global climate change on bird populations (Ganter and Boyd 2000).

The effect of climate change on rainfall has a major influence on the nesting numbers and breeding success of waterbird populations - however this may depend on location. Unusually dry weather can cause nest desertion or starvation, and intense precipitation events such as storms can cause nest and egg destruction and chick death (McKilligan 1987). Insufficient or poorly timed rainfall can also result in mortality of both juvenile and adult waterbirds.

Magpie goose goslings die in large numbers if breeding and brood-rearing swamps dry before they are fledged and able to fly to often distant dry season habitat (Frith and Davies 1961). Starvation due to food shortage following periods of low rainfall seems likely to be a major variable in Cattle Egret mortality worldwide. Primary foods for cattle egrets such as earthworms are more abundant in moist soils and are likely to be difficult to obtain during dry periods (Siegfried 1972; McKilligan 2001). The importance of spring rainfall to egret breeding success has been well-established in some Australian locations (McKilligan 1997; 1997; 2001); however in contrast Maddock and Baxter (1991) found that the breeding success of Cattle Egrets did not vary consistently with wet and dry seasons, over six nesting seasons. McKilligan (2001) suggested that the lack of stability in the annual nesting numbers of Cattle Egrets at a long-term monitoring site in Queensland, compared with the situation in France, for example (Hafner et al. 1992), may be largely a reflection of the rainfall-determined 'stop–go' environmental conditions characteristic of much of Australia (McKilligan 1997; 2001; Maddock and Baxter 1991). Autumn rainfall is also thought to be particularly important for cattle egrets because it promotes the growth of populations of the egret's prey at a critical time for the survival of recently independent juvenile egrets (McKilligan 2001). Annual variation in mortality of juvenile and adult Cattle Egrets probably explains most fluctuations in the size of the breeding population, and most egret mortality

occurs outside the breeding season when the birds are dispersed around coastal south-east Australia (McKilligan et al. 1993).

Changes in the timing and intensity of rainfall and subsequent flooding as a result of climate change may result in increased frequency of incidents of nest flooding resulting in reduced breeding success.

Loss of nests during large floods has been repeatedly documented for magpie geese in Australia's Northern Territory (Whitehead and Tschirner 1990). Frith and Davies (1961) reported flooding in colonies on the Adelaide River plain in one of three years of study, and recounted reports of nest losses due to flooding at Oenpelli in two of the same three years. In the first year of a 2-year study at one of Frith and Davies' sites, many nests were drowned following heavy rainfall associated with a tropical cyclone (Whitehead and Tschirner 1990 citing Dexter 1988).

Feature abstract

Understanding the demographic mechanisms through which climate affects population dynamics is critical for predicting climate change impacts on biodiversity. In arid habitats, rainfall is the most important forcing climatic factor. Rainfall in arid zones is typically variable and unpredictable, and we therefore hypothesise that its seasonality and variability may be as important for the population ecology of arid zone animals as its total amount. Here we examine the effect of these aspects of rainfall on reproduction and age specific survival of blue cranes (Anthropoides paradiseus Lichtenstein) in the semi-arid eastern Nama Karoo, South Africa. We then use our results to predict the effect of changes in rainfall at the population level. 3. Using combined capture-mark-resighting and dead-recovery models, we estimated average survival of cranes to be 0.53 in their first year, 0.73 in their second and third year, and 0.96 for older birds. 4. We distinguished between three seasons, based on the blue cranes' breeding phenology: early breeding season, late breeding season and nonbreeding season. Cranes survived better with increasing rainfall during the late but not early breeding season. Based on road counts and success of monitored nests, reproduction was positively associated with rainfall during the early but not late breeding season. 5. A matrix population model predicted that population growth rate would increase with increasing rainfall. A stochastic analysis showed that variation in early breeding season rainfall increased population growth slightly due to the nonlinear relationship between rainfall and reproduction. This effect was opposed by the effect of variation in late breeding season rainfall on survival and overall, variation in rainfall had a negligible effect on population growth. 6. Our results allow predictions to be made for a range of climate-change scenarios. For example, a shift in seasonality with drier springs but wetter summers would likely decrease reproduction but increase survival, with little overall effect on population growth (Altwegg and Anderson 2009)

Extreme cold or hot weather conditions can result in high mortality rates.

Several mass mortality incidents have been recorded in wild water birds in England in association with prolonged freezing weather (Duff et al. 2011). During these incidents, the ground froze, preventing access to food sources and causing starvation. Some flocks moved to feeding on landfill sites, where they are thought to have contracted and spread avian botulism (Duff et al. 2011).

Temperature may be a major driver of community composition

Although effects of climate change have widely been studied at the species level, less is known about community responses. One study has assessed how the composition of wader (Charadrii) assemblages, breeding in high latitude and wintering from Europe to Africa, is affected by climate change over 33 years (Godet et al. 2011). Using the temporal trend in the community temperature index (CTI), which measures the balance between cold and hot dwellers present in species assemblages, a steep increase was found in the CTI, reflecting a profound change in assemblage composition.

Wind strength and direction can directly affect waterbird migration and dispersal

Changes in typical weather patterns including wind strength and direction and storms may also influence the capacity of birds to move between key locations along migratory or dispersal routes (Sutherland et al. 2012; McKilligan et al. 1993).

Climate change is already affecting the timing and success of waterbird migrations worldwide. The effect of climate and weather on timing of migrations is important, partly because birds arriving early at nesting sites get the best choice of nest locations, have a longer period in which to try again if nesting fails through factors such as predation, and in general are more successful at producing young that survive (Bridgman et al. 1998; McKilligan 1987; 1997; 1997). In addition, climate affects the timing of food availability for migrating and dispersing waterbirds in both

foraging and breeding habitats, and disjuncts between this and the movement of individuals can significantly influence survival / mortality and breeding success.

Climate change may cause changes in primary productivity on migratory staging areas. For example, shorebird wintering and staging sites tend to be concentrated on intertidal flats with high levels of near-shore primary productivity and generally exclude sites with low productivity (Butler et al. 2000). With the exception of the Pacific Americas Flyway, most shorebird flyways have wide separations between these localities. Changes in ocean current circulation and/or onshore winds could change the distribution of such sites (Sutherland et al. 2012).

Natal dispersal and subsequent survival of juveniles may also be affected by climate change.

For example, Moller et al. (2006) investigated long-term patterns of natal and breeding dispersal in a coastal seabird, the Arctic tern *Sterna paradisaea*, that experiences specific climatic conditions in the northern temperate and Arctic climate zones during breeding and different climatic conditions in the Antarctic during winter. Long natal and breeding dispersal distances were costly as shown by their effects on delayed breeding. Dispersal distances varied significantly among years, with natal dispersal showing a strong temporal increase during the last 70 years. Annual differences in dispersal distance could be accounted for by climate conditions in the breeding grounds and the winter quarters. Natal dispersal was related to climate conditions in both the year of hatching and the year of breeding, whereas breeding dispersal was only related to climate conditions in the second year of the dispersal event. Only the north Atlantic oscillation (NAO) index for winter showed a consistent temporal trend, suggesting that the temporal trend in natal dispersal distance must be caused by changes in the NAO (or associated phenomena).

Feature extract

The timing of the migration season depends on several things (see Richardson 1978, 1990; Elkins 1988; Alerstam 1990). There may be a response to a seasonal change in food resources, as cooler conditions become more dominant over time. Birds may migrate to find better seasonal shelter, and to minimise the danger from predators. There may be competition for resources with other species or from members of their own species, which require expansion of territory over parts of the year. The distance from the wintering grounds can be important. Birds prepare for migration by building fat stores, to give them energy for long periods of flight. There is a positive correlation between the amount of fast stored and the distance of migration. Several decades of study of the migration timing of different bird species in the Northern Hemisphere suggest that, once fat reserves are obtained by the birds, the decision to migrate depends mainly on meteorological "cues" such as rising pressure, falling temperature, and falling humidity. Favourable synoptic conditions include at least partially clear skies, light and variable wind or a steady following wind of light to moderate strength, and some suggestion of persistence of positive weather over time. Early in the migration period, birds may be more selective regarding departure conditions. However, the later the delay in good weather conditions, the more likely that migration departure may occur in marginal conditions. Migrating birds wish to avoid unstable conditions, such as frontal situations, cloudiness, precipitation, and strong changing winds. Under these situations birds may have no control over destinations, and flocks of birds are scattered in many directions (Bridgeman et al. 1998).

Making predictions about how populations might respond to future climate change depends on an adequate understanding of important ecological processes and their interactions at appropriate spatial and temporal scales (Norris et al. 2004). In many cases, the effects of climate change will be manifest through indirect effects on land-use change, prey availability, the condition of wetlands, changes in matching of the timing of arrival dates and prey dynamics, predation effects, disease and parasitism, and interactions between these factors - for example:

- In south-western Australia, the most noticeable impacts of climate change are likely to be on wetland species, especially those using coastal wetlands that are expected to be lost – including internationally important migratory wader species (Arnold 1988 cited by Chambers et al. 2005).
- North American greater snow geese now winter further north along the US Atlantic coast, leading to reduced hunting mortality. Their migratory routes now include portions of southwestern Quebec where corn production has increased exponentially. Since the mid-1960s, average temperatures have increased by 1-2.4 degrees C throughout the geographic range of geese, which may have contributed to the northward shift in wintering range and an earlier migration in spring (Gauthier et al. 2005).
- A long-term study of heron populations of NW Italy found that different species were sensitive to environmental and climatic changes (in particular rainfall), temporal variation in human disturbance, changes in foraging habitats, and competition between species, to different extents and at different times (Fasola et al. 2010). Specifically, grey herons increased following a decrease in human-induced mortality, as quantified by an index of

hunting pressure, and an increase in winter temperatures. Little egrets increased mainly with the increase of the extent of ricefields, whereas squacco herons increased with increasing rainfall in the African wintering range. Black-crowned night herons were also positively affected by increasing African rainfall, but only during 1972-1988, whereas in later years competition with other herons could have affected the species' decline.

- It is thought that gradual climatic warming observed in the western boreal forest of North America has contributed to and may continue to exacerbate population declines of duck species such as scaup and scoter (Drever et al. 2012). Duck population models incorporated with snow cover duration models derived from climate simulations under the A2 emission scenario have suggested that late-nesting duck species will experience the most severe population declines. Spring snow cover (duration, timing e.g. of 50% cover etc.) is often used as a proxy for phenological timing of wetland ecosystems used by ducks in these environments (Drever et al. 2012).
- With projected climate change, the waterbirds of high-arctic Greenland may face more unstable breeding conditions, and in the long term some of the wader species may be hampered by overgrowing of the high-arctic tundra with more lush low-arctic vegetation (Meltofte et al. 2008).
- One study of climate impacts on habitat for migratory waterbirds in Italy found that impacts followed a geographic trend, changing the distribution of suitable habitats for migrants and highlighting a latitudinal threshold beyond which the connectivity reaches a sudden collapse. There was relatively poor reliability of most sites in dealing with changing habitat conditions and ensuring long-term connectivity, with possible consequences for the persistence of species (Bellesario et al. 2014)
- Given the disproportionate impact that climate change may have on arctic ecosystems, changes in the timing of snowmelt and plant growth and invertebrate activity are likely to markedly influence waterbirds at higher latitudes and altitudes. However, some animals and plants may simply tolerate or adapt to various climatic effects, in which case the effects may be weaker than modelled responses would assume (Sutherland et al. 2012).

Feature abstract

Whooping crane (Grus americana), a rare and critically endangered species, are wetland dependent throughout their life cycle. The whooping crane's small population size, limited distribution, and wetland habitat requirements make them vulnerable to potential climate changes. Climate change predictions suggest overall temperature increases and significant changes in precipitation regimes throughout North America. At the individual level, temperature changes should have neutral to positive effects on thermoregulation and overall energy expenditure throughout the whooping crane's range. In the breeding grounds, earlier snow melt and increasing temperatures should improve food resources. However, increased precipitation and more extreme rainfall events could impact chick survival if rainfall occurs during hatching. Increased precipitation may also alter fire regimes leading to increased woody plant abundance thus reducing nesting habitat quality. During winter, higher temperatures will lead to a northward shifting of the freeze line, which will decrease habitat quality via invasion of black mangrove. Large portions of current winter habitat may be lost if predicted sea level changes occur. Stopover wetland availability during migration may decrease due to drier conditions in the Great Plains. Current and future conservation actions should be planned in light of not only current needs but also considering future expectations (Chavez-Ramirez and Wehtje 2012).

The potential effects of climate change on waterbird breeding initiation are significant, and interact with other factors such as predation.

In 11 years of monitoring of 12 species in Canada (Smith et al. 2010), timing of breeding was related to the date of 50% snow melt, with later snow melt resulting in delayed breeding. Higher predator abundance resulted in earlier nesting than would be predicted by snow cover alone. The authors hypothesised that when predation risk is high, the value of potential re-nesting exceeds the energetic risks of early breeding. In addition, synchrony of breeding was significantly higher in late breeding years suggesting a relatively fixed date for the termination of nest initiation, after which nesting is no longer profitable (Smith et al. 2010).

As another example, in 1992 the aerosol cloud resulting from the 1991 eruption of Mount Pinatubo (Philippines) reached the high northern latitudes and caused significant cooling in most of the Arctic, with widespread negative consequences for Arctic-breeding birds. At the same time, low abundance of small rodents and high abundance of predators presented additional problems for breeding birds in parts of the Palearctic. A review of multiple studies from that year across the Arctic (Ganter and Boyd 2000) reported a higher proportion of nonbreeders and a delayed onset of nest initiation compared to other years. Hatching and fledging success of the low number of late breeders was reduced. In addition, some projects reported lower clutch sizes and increased adult mortality. Detailed data from

field studies are complemented by data on overall reproductive success of waterfowl and wader populations collected from staging and wintering grounds. In total, there was an almost complete reproductive failure for waders and waterfowl throughout the Arctic in 1992, suggesting a short-term effect on global waterbird populations. This is an example of climatic fluctuations influencing reproductive biology of a group of species on a circumpolar scale (Ganter and Boyd 2000).

The timing of hatching in the mallard (*Anas platyrhynchos*) and the common goldeneye (*Bucephala clangula*) is associated with the timing of lake ice break-up (a local climate index) in Finland (Oja and Poysa 2007). Sjoberg et al (2007) conducted an egg-hatching experiment that found that duckling survival on boreal lakes was not affected by a 12-day delay in hatching date. Since they did not find any consistent trends in abundance of aquatic prey, i.e. neither clear peaks nor differences between treatment periods, they hypothesised that moderate climate change has minor effects on resource abundance and hence also on mallard duckling survival in boreal environments.

Feature abstract

Nesting migratory geese are among the dominant herbivores in (sub) arctic environments, which have undergone unprecedented increases in temperatures and plant growing days over the last three decades. Within these regions, the Hudson Bay Lowlands are home to an overabundant breeding population of lesser snow geese that has dramatically damaged the ecosystem, with cascading effects at multiple trophic levels. In some areas the overabundance of geese has led to a drastic reduction in available forage. In addition, warming of this region has widened the gap between goose migration timing and plant green-up, and this mismatch between goose and plant phenologies could in turn affect gosling development. The dual effects of climate change and habitat quality on gosling body condition and juvenile survival are not known, but are critical for predicting population growth and related degradation of (sub) arctic ecosystems. To address these issues, we used information on female goslings marked and measured between 1978 and 2005 (4125 individuals). Goslings that developed within and near the traditional center of the breeding colony experienced the effects of long-term habitat degradation: body condition and juvenile survival declined over time. In newly colonized areas, however, we observed the opposite pattern (increase in body condition and juvenile survival). In addition, warmer than average winters and summers resulted in lower gosling body condition and first-year survival. Too few plant growing days in the spring relative to hatch led to similar results. Our assessment indicates that geese are recovering from habitat degradation by moving to newly colonized locales. However, a warmer climate could negatively affect snow goose populations in the long-run, but it will depend on which seasons warm the fastest. These antagonistic mechanisms will require further study to help predict snow goose population dynamics and manage the trophic cascade they induce (Aubry et al. 2013).

The complexity of the interactions between climate change and other variables driving waterbird populations makes prediction and management of waterbird responses very difficult.

Sutherland et al. (2012) suggested that given the nature of climate changes experienced to date, we need many more analyses of existing data and further data collection in order to assess the magnitude of this driver on waterbirds. To-date, the cost and logistical difficulties involved in long-term studies of the population ecology of waterbirds have meant that detailed information on breeding parameters and their relation to climate and weather patterns is relatively scarce (Ganter and Boyd 2000). However such information is essential as input into models of the potential impact of various climate change scenarios on bird populations. Long-term data in particular (typically decades for birds and mammals) is required in order to untangle spatial and temporal variability from responses to change.

Van de Pol et al. (2010) stated that although it is well established that climate change may strongly affect population dynamics, the general mechanisms causing climate induced population change are still poorly understood. They suggested that four major unresolved questions are: (1) Does climate change mainly affect population dynamics through its effects on survival or fecundity, and how does this vary between species and environments? (2) How important is the contribution of climate change to population fluctuations in comparison to other stochastic and deterministic processes? (3) What is the relative importance of changes in the mean and variability of climatic drivers? (4) Does increased interannual climatic variability typically reduce population viability as predicted by classical stochastic population theory, or can it also improve population viability as more recently put forward? Answering these questions and identifying the mechanisms involved are crucial for identifying species that are most at risk. Unfortunately, the duration and level of detail of field data required to gain these insights is typically unavailable for the species for which these insights are actually most needed (i.e. those of conservation concern). We are still a long way from accurately predicting long-term consequences of climate change for most waterbird populations (Van de Pol et al. 2010).

The establishment and maintenance of conservation areas are among the most common measures proposed to mitigate loss of biodiversity due to climate change. However, recent advances have challenged the ability of such areas to cope with predicted changes. The availability and connectivity of suitable habitats, together with species-specific characteristics, can limit adaptation options. The extent to which existing protected areas are able to ensure the persistence of species still remains unclear (Bellesario et al. 2014). Results of recent predicted future climate scenarios are being used to suggest that waterfowl conservation be shifted away from currently important areas in the western and central portions of the U. S. eastward, to locations where wetland and climate models suggest may become more conducive for providing wetland habitat for breeding ducks in the future. However an economic assessment of the biological risk of such a shift (Loesch et al. 2012) found that maintaining the current focus of habitat protection appears to be the most cost effective approach for waterfowl habitat conservation efforts. Additionally, the authors suggested that continued intensive monitoring activities designed to detect changing waterfowl populations and habitats as they relate to anthropogenic impacts and climatic changes will provide precise results to inform and adapt management and conservation activities (Loesch et al. 2012).



7 Pollution

7.1 Introduction

Worldwide, many populations of waterbirds have suffered declines in abundance as a result of toxic effects of pollution or contaminants in their environment (Powell and Powell 1986; Peterle 1991; Grasman et al. 1998 cited by De Luca-Abbott et al. 2001). While pesticides (herbicides and insecticides) have received much deserved attention in the last 30 years, waterbirds may also be affected by many other contaminants.

For example, nutrient pollution was hypothesised as being a key driver of population declines in 16 waterbird species reviewed by Poysa et al. (2013), particularly in those species that preferred eutrophic lakes as breeding habitat. Increased rates of sediment delivery may change nutrient dynamics, reduce water clarity, reduce food availability and eventually cause wetland loss (Sutherland et al. 2012). Spills of other toxic substances from a range of industries and activities, such as waste chemicals, oil and boat fuel, are well known for their effects on fish and waterbirds.

Often, a considerable number of contaminants are present in the environment and consequently it can be difficult to associate toxic effects with any one chemical. Very similar chemicals can vary widely in their toxicity, and it is also likely that some chemicals interact to increase toxicity. Critical effect concentrations of contaminants are often unknown, making risk assessment difficult (De Luca-Abbott et al. 2001).

Residue levels of toxins associated with reproductive effects vary widely from species to species, and are complicated by interactions between toxins, sometimes depending on geographic location. There is variability in accumulation of contaminants between species, sexes, ages and tissue types (De Luca-Abbott et al. 2001). Consequently, studies of contaminant effects must establish the relationships between toxin levels and each of these variables if they are to yield meaningful results.

Diet has a significant effect on the concentration of contaminants accumulated.

Predatory waterbirds are susceptible to bioaccumulation (or biomagnification) of organochlorines (e.g. DDT, dieldrin, other chlorinated pesticides, PCBs) and some metals (e.g. mercury). Piscivorous waterbirds are more susceptible to bioaccumulation through the food chain than insectivores or herbivores - for example, the aquatic-foraging piscivorous intermediate egret has been found with higher levels of contaminants than the mostly terrestrial and insectivorous cattle egret (although both have varied diets) (Burger and Gochfeld 1996 cited by De Luca-Abbott et al. 2001; Burger et al. 1992).

The vast majority of effects are expressed in reproductive failure, and subsequent population declines may be masked by pairs laying repeat clutches after failures.

Bioaccumulation has been associated with breeding failures in some waterbird species (Faber et al. 1972; Cooke et al. 1976; Price 1977; Sanderson et al. 1994 cited by De Luca-Abbott et al. 2001). Eggs and developing chicks are generally more susceptible to toxins and contaminants than adult birds (Kushlan 1993; De Luca-Abbott et al. 2001). There is wide variation in susceptibility between species. Symptoms of toxicity include eggshell thinning, reduced egg weight, embryonic deformities and embryonic mortality (reduced hatch rate), reduced chick size and increased chick mortality, aberrant adult behaviour such as egg-eating, and reduced fledging rates and nest success.

Direct adult mortality of waterbirds as a result of pollutants is relatively rare, but incidents have been recorded (De Luca-Abbott et al. 2001; Howarth et al. 1981; Kim et al. 2007).

In a study of 167 Eurasian Cranes (*Grus grus*) from Germany, a group of 28 (16.8%) died from organophosphate intoxication (Fanke et al. 2011). These and the associated pollutant levels in tissues are reviewed by De Luca-Abbott et al. (2001) for ardeids.

Besides their direct toxic effects, pesticides and herbicides can reduce food availability for waterbirds, depending on their diet.

Spraying of these toxins often coincides with breeding seasons of waterbirds, increasing the pressure on more susceptible eggs and young and the difficulty in obtaining sufficient food for nestlings, fledglings and juveniles (Rusch et al. 1989). For most invertebrate taxa, biomass will remain low after spraying until ovipositing adults recolonize the pond; this may take weeks or years depending on the taxa (Gibbs et al. 1984 cited by Rusch et al. 1989).

7.2 Pesticides and other toxins

Historically, the major pesticides of concern for waterbirds have been organochlorine (chlorinated hydrocarbon) pesticides, including DDT-type compounds and chlorinated alicyclics. Examples include DDT, dicofol, heptachlor, endosulfan, chlordane, aldrin, dieldrin, endrin, mirex, kepone and pentachlorophenol. Many studies have found associations between levels of DDE (residues of DDT) in the environment and eggshell thinning, with thinning of more than 16-18% resulting in population declines (De Luca-Abbott et al. 2001 and references therein). The concentration of DDE required to produce critical eggshell thinning varies between species and locations. Dieldrin has been implicated in mass deaths of egrets (De Luca-Abbott et al. 2001), and reduced breeding success in cormorants (*Phalacrocorax carbo sinensis*) nesting in the heavily contaminated sedimentation area of the rivers Rhine and Meuse in the Netherlands has been attributed to eggshell thinning and increased embryonic mortality from chlorinated hydrocarbon pollution reducing hatching success (Dirksen et al. 1995).

DDT and similar toxins were widely used in the 20th century as pesticides, and were banned in most developed countries in the 1970s - 1980s. Many studies report that contaminant levels in birds have declined since 1970-80 (De Luca-Abbott et al. 2001 and refs therein). However these chemicals are not banned in all countries used by migratory waterbirds, and levels of other contaminants are increasing in some locations. Persistent DDT and other organochloride residues continue to be found in humans and mammals worldwide (De Luca-Abbott et al. 2001).

Anticholinesterase compounds are found in organophosphorus and carbamate insecticides, and they disrupt nerve function and eventually cause death. The organophosphate insecticide monocrotophos has caused deaths of globally threatened Sarus cranes *Grus antigon* and common cranes *Grus grus* in India, through ingestion of treated seed (Pain et al. 2004). Waterbird with aquatic diets are not as exposed to anticholinesterase as terrestrial species, except where the compounds are used for mosquito control - the latter use can result in high mortality rates (De Luca-Abbott et al. 2001).

Polychlorinated biphenyls (PCBs) are toxins that were once commonly used electrical insulators and heat transfer agents. Their use has generally been phased out due to health concerns. PCBs were replaced by polybrominated diphenyl ethers (PBDEs), which bring similar toxicity and bioaccumulation concerns. The toxicology of PCBs, PCDDs and PCDFs is complex because of the presence of many congeners of widely different toxicity (De Luca-Abbott et al. 2001).

Many species of algae, including dinoflagellates, diatoms and cyanobacteria, can produce neurotoxins, hepatotoxins and dermatotoxins that can poison waterbirds through direct ingestion or bioaccumulation in prey (Sutherland et al. 2012). Harmful algal blooms are increasing in frequency due to climate warming and eutrophication. The impact of algal blooms on waterbird mortality is probably underestimated due to carcass predation, decomposition and water action, although there is evidence that algal blooms may have triggered several mass mortality events (Buehler *et al*. 2010 cited by Sutherland et al. 2012). However, some waterbird species avoid prey or habitats contaminated by algal bloom toxins (Kvitek & Bretz 2005 cited by Sutherland et al. 2012).

7.3 Heavy metals

Predatory waterbirds consuming fish, amphibians, reptiles, invertebrates and other prey are situated at the top of the wetland food web. Biomagnification or bioaccumulation of heavy metals may occur in these species, with a proportionate increase in concentration with increasing trophic level. The effects of these accumulations are not always apparent in terms of mortality or other parameters, however in some cases they may allow waterbirds to act as indicators of pollution accumulation in affected ecosystems (Kushlan 1993; De Luca-Abbott et al. 2001).

Mercury accumulation can be a particular problem in migratory waterbirds (Gerstenberger 2004). While mercury can cause toxicity and ultimately death directly, it is more likely to increase waterbird susceptibility to cold and other adverse conditions. Non-marine species are more susceptible than marine foragers. Selenium toxicity has also been a problem in some areas, associated with embryo death and deformities. Herons may be less susceptible than ducks (De Luca-Abbott et al. 2001). Purple heron eggs and nestling feathers have been used as bioindicators for heavy metal pollution; feathers in particular showing pollutant impacts on a strict local basis (Cotin et al. 2012).

In Australia, some studies on shorebirds (predominantly those using coastal sites) have found no indication of toxic levels of accumulation of metals in bird tissues and organs (Howarth et al. 1981; Kim et al. 2007). Tissues taken from Australian crested terns were analysed for cadmium, lead, copper, chromium, manganese, zinc and iron by Howarth

et al. (1981). The birds were taken from a breeding colony in a non-industrialized region and from one near an industrialized region. The birds from the industrialized region did not show any consistent evidence of significant heavy metal contamination when compared to those of the non-industrialized region. The individual tissues were also compared to assess the distribution of the various metals, and the relative degree to which each tissue accumulates the metals. Some accumulation of chromium, copper, manganese, lead and, to a lesser degree, cadmium and zinc was found in the salt glands. The kidneys had a relatively large cadmium content.

Lead toxicosis following ingestion of spent shot has long been recognised as a significant cause of mortality in waterfowl. Studies in Australia initially indicated a low incidence of ingested shot compared with the United States or Europe, and the issue received little early attention (Whitehead and Tshirner citing Lavery, 1971; Norman, 1976). Subsequent studies demonstrated significant levels of lead intoxication in waterfowl at particular sites, for example at Bool Lagoon in South Australia, and Howard Springs Hunting Reserve in the Northern Territory (Whitehead and Tshirner citing Koh & Harper 1988; Harper & Hindmarsh 1990). Geese appear to be particularly susceptible to lead ingestion and poisoning, deriving from their feeding habits. Even sublethal exposures may have important longer term effects, including reduced body weight, lower reproductive output, impairment of motor function increasing probability of predation, or death through collision of flying birds with obstacles (Whitehead and Tschirner 1991 and references therein).

In a study of swan mortality in Washinton USA, lead poisoning was the primary cause of death, accounting for 29% of the mortalities. Other causes of mortality identified were aspergillosis (17%), illegally shot (11%), and other traumatic factors (12%) (Lagerquist et al. 1994). Lead contamination from the use of leaded petrol and generally high pollution levels has also been blamed for extremely high concentrations of lead and other heavy metals found in feathers of young night herons, little egrets, great egrets and cattle egrets in Hong Kong (Burger and Gochfeld 1993). Adult great egrets had higher concentrations than young individuals.



8 Disease and parasites

8.1 Introduction

Disease and parasites can affect nest success, fledging rates, juvenile survival, and adult survival.

Infectious diseases are an important and frequently dominant mortality factor in waterbird populations.

An early survey of waterfowl deaths in North America indicated 80-90% of birds had died of disease (excluding those shot (Holmes 1982 cited by McKilligan 1987). Bacteria such as *Clostridium botulinum* and viruses such as avian influenza, West Nile Virus, Newcastle Disease Virus, avian poxvirus, duck plague, avian bornavirus, reoviruses and adenoviruses may contribute to population declines of both wild and domestic waterbirds. The infection rate by *Plasmodium* parasites (avian malaria) is rapidly increasing in many birds and there are high infection rates of Campylobacteria in waders (Sutherland et al. 2012 and references therein). Changes in land use and global climate may result in a stronger concentration of waterbirds on remaining high-quality staging sites, making them potentially more vulnerable to infections (Krauss et al. 2010 cited by Sutherland et al. 2012).

Avian Newcastle Disease (ND) is a highly contagious and infectious disease that affects all avian species including poultry, cage and wild birds and can cause high mortalities reaching 100%, however in Australia, deaths have not been reported in wild birds (WHA 2011). Wild migratory waterfowl populations are known to carry it (Takakuwa et al. 1998). Reoviruses have been identified in a limited number of species of European geese and ducks (Palya et al. 2003), causing hepatic and arthritic symptoms; and an adenovirus has been identified as a source of mortality in long-tailed ducks in Alaska and a possible contributing factor in population declines (Hollmen et al. 2003). Detailed postmortem examinations on 167 free-ranging Eurasian Cranes (*Grus grus*) from Germany collected over a 10-year period found that pathologic changes due to infectious diseases were associated with Aspergillus spp. (n=7, 4.2%), endoparasites (n=7, 4.2%), avian poxvirus (n=6, 3.6%), Mycobacterium spp. (n=2, 1.2%), and adenovirus infection (n=1, 0.6%), while s severe Strigea spp. infection (n =1, 0.6%) and a leiomyosarcoma (n=1, 0.6%) were newly recognized diseases (Fanke et al. 2011).

Ticks parasitising nestlings can reduce survival and nest success, and potentially also transmit viruses.

In a 3 year study of the cattle egret at a heronry of 3500 nests in south-east Queensland, 34% of eggs in 711 nests observed failed to survive to become fledglings (11 day old chicks). Fledgling mortality seemed much lower. The main sources of mortality were crow predation on eggs, failure of the incubated egg to develop fully, starvation of nestlings and tick infestation of nestlings. These were of about equal importance overall but their individual magnitudes varied considerably among the three seasons. The overall loss of cattle egret nestlings to tick infestation was calculated as 10-12% depending on the year (McKilligan 1987). An experiment using acaricides found that mean tick load of cattle egret chicks was inversely correlated with chick longevity (McKilligan 1996). However a subsequent study (McKilligan 1997) found that annual differences in sizes of advanced cattle egret broods were mostly influenced by chick starvation, not by the sizes of early broods or the burden of parasites that season. Viruses transmitted by ticks can also increase mortality of waterbirds and chicks, but the interactions between the effects of ticks vs the effects of the viruses are less well understood (Standfast et al. 1986; McKilligan 1987 citing Converse et al. 1975; Duncan et al. 1978; Reid et al. 1978).

Below, the effects of three of the most widely occurring and researched diseases are summarised in more detail: Botulism, Avian Influenza, and West Nile Virus.

8.2 Botulism

Botulism is a paralytic disease that is caused by a toxin produced by the bacterium *Clostridium botulinum*. It occurs worldwide (except Antarctica). *C. botulinum* is a soil bacterium that is particularly common in wetland environments. Spores produced by the bacterium persist in the environment for years, so outbreaks may occur repeatedly in the same location. Most outbreaks of botulism occur in the summer. The majority of birds that develop botulism and are not treated will die. Botulism can occur in any bird species, but is most frequently seen in ducks, geese, swans, ibis, egrets and pelicans (WHA 2013).

The earliest cases of botulism recorded occurred in the United States and Canada in the early 1900s, and the first reported outbreaks in Australia took place around the 1930s, with many more recorded since (WHA 2013). Data

collected by the Australian Wildlife Health Network (AWHN) of reported botulism cases over a six year period documented sixty eight events of suspected or confirmed avian botulism in Australia between 2006 and 2012 (WHA 2013). In ten of these events multiple species of birds from several bird orders were affected, however Anseriformes (ducks, geese and swans) were involved at the majority of events. Most of the reported cases of avian botulism took place during a six month window from November to April, whilst fewer cases were reported in the cooler months of the year.

Botulism outbreaks with one million or more waterbird deaths have been reported in the USA, Canada, and Russia (WHA 2013). Type E Botulism is an important source of mortality for waterbirds on the lower Great Lakes (Shutt et al. 2014). On six islands in eastern Lake Ontario, Canada, July-Nov 2004-2009, over 6600 dead/dying birds were located; five species accounted for >98% of the birds found: double-crested cormorant (*Phalacrocorax auritus*), herring, ring-billed and great black-backed gulls (*Larus argentatus, Larus delawarensis, Larus marinus*, respectively) and Caspian tern (*Hydroprogne caspia*). Cormorants accounted for 65-78% of all birds annually. Mortality was greatest in 2005 (7.5%). Botulism deaths accounted for >100% of the great black-backed gulls breeding there. The species was extirpated from Lake Ontario during the study period. Most carcasses (91%) were documented on four islands, leading Shutt et al. 2014 to emphasise that mortality assessments must document both roosting and breeding sites. In Australia, numbers of birds that die in botulism outbreaks are generally less than 100, but the death of up to 1,500 birds (predominately grey teal [(*Anas gibberifrons gracilis*)]) has been recorded in one outbreak in Victoria (WHA 2013).

Botulism is now present in every continent apart from the Antarctic and is a leading cause of death for waterbirds. Given that the bacteria responsible for botulism have resistant spores that can survive for years these problems can persist, and may well have population-level impacts. This is most likely to be the case for species with small populations (Sutherland et al. 2012). Affected birds are also more likely to be subjected to predation. The factors that predispose to botulism outbreaks are complex and incompletely understood. Further study into the environmental conditions associated with these outbreaks in Australia may help to understand why they occur and mitigate their impact (WHA 2013).

8.3 Avian influenza

While waterfowl and waders form the natural reservoir for all avian influenza (AI) viruses, they usually show no clinical signs of disease (WHA 2009). The Anseriformes (ducks, swans, geese) and Charadriiformes (gulls, terns and shorebirds) make up most of the natural reservoir. Wild ducks of the genus Anas are hosts for a large genetic diversity of influenza A viruses in which co-infections with different virus genotypes are frequent and result in high rates of genetic reassortment (Lebarbenchon et al. 2012). The Asian highly pathogenic AI (HPAI) H5N1 is a virulent AI virus strain with the potential to cause deaths in wild and domestic birds, humans and some other species. This strain is currently not efficiently transmitted from birds to humans or from human-to-human, and has not been detected in Australia. Other virus subtypes have been detected in Australia in wild duck, wader, tern, shearwater, gull and other waterbird species (WHA 2009).

Although waterfowl are usually asymptomatic carriers of the influenza A virus, low-pathogenic avian influenza (LPAI) infection in Bewick's swans (*Cygnus columbianus bewickii*) has been shown to negatively affect migration stopover time, body mass and feeding behaviour (Van Gils et al. 2007). In migratory mallards (*Anas platyrhynchos*), body mass has been found to be significantly lower in infected ducks than in uninfected ducks (mean difference almost 20 g over all groups), and the amount of virus shed by infected juveniles was negatively correlated with body mass (Latorre-Margalef et al. 2009). There was no general effect of infection on staging time, except for juveniles in September, in which birds that shed fewer viruses stayed shorter than birds that shed more viruses. LPAI infection did not affect speed or distance of subsequent migration. Shedding time decreased during the season, suggesting that mallards acquire transient immunity for LPAI infection. Over 6000 birds, including more than 3000 Bar-headed Geese *Anser indicus,* died at Lake Qinghai in northern China during an outbreak in 2005 (Chen *et al*. 2006).

The practices of culling wild birds and/or the destruction of their habitat (eg wetlands, water bodies) are not supported and are recognised by key international AI expert organisations as ineffective measures for the control of avian influenza (WHA 2009).

8.4 West Nile Virus

West Nile Virus (WNV) is a mosquito-borne zoonotic arbovirus belonging to the Japanese encephalitis virus serocomplex. The virus cycle is maintained in birds, and mammals are incidental hosts. The rapid global expansion of

WNV has recently raised concerns regarding its possible spread into new countries (Yeh et al. 2011) and has prompting widespread testing, including experimental studies of the role of particular bird species as early-warning indicators (Bingham et al. 2010). In Australia, a subtype of WNV called Kunjin virus, or KUNV is found in the Northern Territory and northern Western Australia. Kunjin is less virulent than the current USA strain of WNV to which it is closely related.

In Europe, Africa, the Middle East and Asia, mortality in birds associated with WNV infection is rare. In contrast, the virus is highly pathogenic for birds in the Americas. Members of the crow family (*Corvidae*) are particularly susceptible in the USA (but apparently not in Australia; Bingham et al. 2010) but the virus has been detected in dead and dying birds of more than 250 species. Birds can be infected by a variety of routes other than mosquito bites, and different species may have different potential for maintaining the transmission cycle. For example, West Nile virus (WNV) causes significant mortality of American White Pelican chicks at colonies. Shedding of the virus could enable chick-to-chick transmission and help explain why WNV spreads rapidly in colonies. It is possible that passive transfer of maternal antibodies occurs. In a study of near-fledged chicks, 41% tested positive for anti-WNV antibodies, indicating that they survived infection. Among years and colonies, cumulative incidence of WNV in chicks varied from 28% to 81%, whereas the proportion of chicks surviving WNV (i.e., seropositive) was 64-75%. Infection of chicks is pervasive, and significant numbers of chicks survive infection (Sovada et al. 2013). However WNV kills older chicks that are no longer vulnerable to other common mortality factors (e.g., severe weather, predation) and typically would have survived to fledge; thus WNV appears to be an additive mortality factor (Sovada et al. 2008; 2013).

Although herons and egrets in the family Ardeidae frequently have been associated with viruses in the Japanese encephalitis virus serocomplex, communal nesting colonies do not appear to be a focus of early season and rapid amplification of WNV in California (Reisen et al. 2009; 2005). Evidence for repeated WNV infection was found by testing living and dead nestlings collected under trees with mixed species ardeid colonies nesting above in an oak grove near the University of California arboretum in Davis and in a Eucalyptus grove at a rural farmstead. However, mosquito infection rates at both nesting sites were low and positive pools did not occur earlier than at comparison sites within the City of Davis or at the Yolo Bypass wetlands managed for rice production and waterfowl habitat. Black-crowned night herons (*Nycticorax nycticorax*) were the most abundant and frequently infected ardeid species, indicating that WNV may be an important cause of mortality among nestlings of this species (Reisen et al. 2009; 2005).

Temporal variation in exposure and (or) susceptibility to disease-causing agents may result in changing disease risks for offspring of seasonally reproducing organisms (Gurney et al. 2014). Although increases in disease risk and disease-related mortality have been observed during the course of the breeding cycle in some systems, the extent to which this is a generalized ecological pattern remains uncertain. For example, Lesser Scaup (*Aythya affinis*) duckling known-fate survival models have been used to assess whether daily survival rate (DSR) was related to age, hatch date, immunogenic challenge, vector abundance, and risk of WNV infection. Ducklings produced late in the breeding cycle had lower survival probabilities, relative to earlier conspecifics, but the best predictor of DSR was relative risk, suggesting that reduced survival of late-hatched individuals may have been related to increasing exposure to WNV-infected vectors (Gurney et al. 2014).



9 Human disturbance

9.1 Introduction

Disturbance from human activities can cause changes in behaviour and affect temporal and spatial distribution of waterbirds.

Human disturbance can be equivalent to habitat loss or degradation because waterbirds may avoid or underuse areas (Gill & Sutherland 2000 cited by Fernandez and Lank 2008). Waterbirds may waste time and energy responding to human disturbance, and the consequences for individuals, populations, and sites can be difficult to determine. Temporary loss of foraging habitats can occur and the capacity to compensate by foraging for longer periods may vary between individuals and species (Sutherland et al. 2012 and references therein). During the breeding season, human disturbance may influence nest incubation and chick rearing.

The effect of human disturbance on waterbirds is dependent on the nature and location of the disturbance.

Carney and Sydeman (2000) identified three main categories of disturbance via human presence in waterbird colonies:

1. Scientific investigators, people who often need to closely monitor demographic parameters of colonial waterbirds. Their work often presents the most intense kinds of disturbance: entering colonies, handling nest contents, and capturing adults.
2. Ecotourists, people who travel primarily to experience free-ranging wildlife. Ecotourists (including wildlife photographers) can also introduce high levels of disturbance to nesting waterbirds. They often approach wildlife to close distances, return repeatedly to the same places, and visit wildlife areas year-round.
3. Recreators, people who visit natural areas for reasons other than viewing or interacting with wildlife. Recreators can disturb nesting colonial waterbirds as well, but often to a lesser extent. Hikers, joggers, bikers, boaters, etc. are more likely to come into contact with wildlife incidentally and usually do not remain in close proximity to wildlife for extended periods of time. Though they tend to limit their activities to certain times of the year, peak use of natural areas is coincident with the breeding season of colonial waterbirds (e.g. during summer).

The type, frequency, intensity and duration of disturbance can all interact to affect waterbirds

A behaviour-based model of the impact of human disturbance on oystercatchers (*Haematopus ostralegus*) on their intertidal feeding grounds in the Exe estuary in winter predicted that, for the same overall area disturbed, numerous small disturbances would be more damaging than fewer, larger disturbances. When the time and energy costs arising from disturbance were included, disturbance could be more damaging than permanent habitat loss. Preventing disturbance during late winter, when feeding conditions were harder, practically eliminated its predicted population consequences (West et al. 2002).

In nineteen nesting colonies of Great Blue Herons (*Ardea herodias*) surveyed in northeast Ohio and western Pennsylvania USA, fledging success was correlated with the type of barrier forming a buffer zone around each site, and the types of human disturbances experienced. Fledging success was not related to colony size or to width of the buffer zone (Carlson and McLean 1996).

Waterbirds may be more sensitive to disturbances during particular periods, such as nesting, hatching, brood care and moulting

Carney and Sydeman (1999) reviewed 64 papers documenting the effects of human disturbance on colonial nesting waterbirds, and synthesised these into recommendations for site managers based on bird family. They found that nesting colonial waterbirds are particularly vulnerable to human disturbance. When approached by humans, they often flush from nests in an attempt to either intimidate a perceived predator or to flee. Nest contents may be spilled, exposed to predation, or suffer from exposure to the elements (Carney and Sydeman 1999; Burger 1982; Anderson 1988; Randler 2003).

Responses to disturbance vary with species, populations, and timing, with some species more being sensitive early in the breeding season than late, and vice versa. Temporary or permanent abandonment of nests as a result of disturbance leaves eggs and young susceptible to predation or exposure, and they may also be crushed as adults leave the nest. Smith et al. (2012) found that nest predation in Arctic shorebirds was positively related to the proportion of time that each species left the nest unattended. Even a single disturbance early in the nesting season can have

dramatic negative impacts on reproductive success of nesting Brown Pelicans (*Pelecanus occidentalis*); once disturbed, breeding areas usually remain unoccupied for the rest of the breeding season. In this species, clutch size, hatching success, fledging success, and overall reproductive success are lowered as a result of human and aircraft disturbance (Carney and Sydeman citing Schrieber and Risebrough 1972; Anderson and Keith 1980; Boellstorff et al. 1988; Bunnel et al. 1981). Even in areas relatively distant from the site of disturbance, waterbirds can be sensitive to the presence of humans, fleeing at the sound of the alarm calls of other birds (Carney and Sydeman 1999). Human disturbance has also been shown to lower productivity of cormorant colonies. For example, when a person entered a Double-crested Cormorant (*Phalacrocorax auritus*) colony, adults often flushed from their nests, exposing contents to predation or the elements (Ellison and Cleary 1978). Investigator disturbance also discouraged late-nesting birds from settling in affected areas.

Similarly, human disturbance associated with logging operations has affected colony size, nest occupancy rate, and fledging rate of Great Blue Herons (*Ardea herodias*) nests, with nesting activity shifting away from the point of disturbance. Vos et al. (1985) found the response of nesting Great Blue Herons to disturbance was dependent upon the type of intruder (boater, horseback rider, or person on foot). Herons were most sensitive to land-related intrusions; 61% of these disturbances resulted in nest abandonment. When appropriate buffers were set up and monitored; however, human visitation of heronries resulted in no ill-effects (Burger et al. 1995). Heronries surrounded by a buffer zone of only 50 m that were visited daily by tourists suffered no short-term reproductive losses, and birds seemed generally unconcerned with human presence near the colony. However, when a group of tourists at an unwardened site entered a heronry, nest mortality rates of 15-28% per heron species resulted (Burger et al. 1995).

To some extent, some species can compensate for disturbance by altering their behaviour or habituating to human activities, and this may interact with predation (Madsen 1995; Baudains and Lloyd 2007; Fitzpatrick and Bouchez 1998). The effects of human disturbance may increase predation, or may not, depending on the characteristics of the disturbance and the species present.

In some species and colonies, intensive study involving trapping, banding and repeated nest visitations can be conducted without detrimental impacts on reproductive success (Shealer and Haverland 2000; Olsen and Rohwer 1998). This may depend on location and other factors. While Tremblay and Ellison (1979) found that nest checking and marking provoked abandonment of newly-constructed black-crowned night heron nests and intensified predation of nest contents by gulls and ravens, Parsons and Burger (1982), suggested that black-crowned night heron chicks habituated to investigator disturbance and handling. At three weeks of age, all regularly-handled experimental chicks remained in their nests during disturbance. All control chicks, however, left the vicinity of the nest, sometimes fleeing the nesting tree altogether. In contrast, Davis and Parsons (1991) found that Snowy Egret (*Egretta thula*) chicks did not habituate to human intrusion, but no differences in survivorship were found between chicks handled twice daily from hatching and chicks handled only during banding (age 7-10 d). Frequency of visitation also had no effect on the reproductive success of Tricolored Herons (*Egretta tricolor*) during courtship and early egg-laying (Frederick and Collopy 1989 cited by Carney and Sydeman 1999).

The presence of scientific temperature loggers and nest cameras, and the associated disturbance in placing these devices, might be expected to increase rates of predation. However, this was not found to be the case for temperature loggers in Piping Plover *Charadrius melodus* and Long-billed Curlew *Numenius americanus* nests in the USA, for video cameras at wader nests in New Zealand, or for digital nest cameras at Lapwing nests in the United Kingdom (Hartman & Oring 2006, Schneider & McWilliams 2007; Bolton et al. 2007; Sanders & Maloney 2002; cited by MacDonald and Bolton 2008). Increased egg losses in duck nests are associated with the presence of scented attractors such as duck feces and cracked eggs, but the effects of human trails to nests are equivocal (Olsen and Rohwer 1998).

St Clair et al. (2010) measured the flushing distances of incubating Two-banded Plovers in response to a controlled human approach to the nest. They found that flushing distances were increased at sites where mammalian predators were present and decreased where exposure to humans was high. These effects were additive, and interpreted as the effects of generalization and habituation, respectively. Mikola et al. (1994) studied the consequences of boat disturbance and gull predation on brood survival of velvet scoter *Melanitta fusca* in the Archipelago of South-West Finland in 1990 and 1991. Each brood was exposed to disturbance by boats on average 8.5 times a day in 1990 and 3.5 times a day in 1991. Disturbance lengthened the swimming distances of ducklings and reduced the time used for feeding. Broods disturbed more frequently than average were smaller than those disturbed less frequently. At least 60% of ducklings died before the age of three weeks, and the frequency of gull attacks was 3.5 times higher in disturbed than in undisturbed situations (Mikola et al. 1994).

Comparatively little is known about how reactions to disturbance may impact on the dispersal of waterbirds and, ultimately, on their overall population dynamics (Madsen 1995). One study has shown that capture of breeding adult

little egrets (*Egretta garzetta*) at their nest and handling (i.e., disturbance) did not increase the probability that they would move to a new colony in the subsequent breeding season (Henry et al. 2004).

9.2 Implications for disturbance management

Buffers are often used to separate threatening stimuli, such as humans, from wildlife but with few exceptions buffer widths are based on little empirical information.

Glover et al. (2011) measured the distance at which a response (i.e., flight initiation distance [FID]) occurred among 28 of Australia's 36 regularly occurring shorebird species when presented with an approaching human (n = 760 approaches in Victoria, south eastern Australia). Species differed in their FID. with species with higher body masses having longer FIDs ($F(1,2G) = 36.830$. $p < 0.001$; $R(2) = 0.586$). Mean FIDs for species were 18.6-126.1 m (n = 370 approaches by a walker). Depending on the species, FID was significantly influenced by the starting distance of the human approach, flock size, previous exposure to humans and stimulus type (walker, jogger, walker with dog). The FIDs reported suggest that current buffer designations will reduce disturbance to many but not all shorebird species tested (Glover et al. 2011).

Studies of mixed colonies of waterbirds containing herons, egrets and ibis have found that most initiate flight (flush) when humans approach to between 30-50 m, and buffer zones of 100 m have been suggested to minimize disturbance and prevent nest desertion, temporary or permanent (Carney and Sydeman 1999 citing Erwin 1989; Rodgers and Smith 1995). The mode or route of approach may also make a difference, even when the frequency of disturbance is high. A study in British uplands found that while high levels of human disturbance via foot traffic can affect habitat usage by waders, this is generally only in limited circumstances where visitor pressure is very high (> 30 visitors per weekend day). Research suggested that access to such areas can be permitted for larger numbers of visitors without impacting upon wader reproductive performance through the provision of a well-surfaced route (Pearce-Higgins et al. 2007).

Since rates of disturbance differ between species and may be correlated to foraging strategies, habitat requirements, and type of disturbance, managers of public lands should consider life histories of waterbirds when setting seasonal recreation dates or planning scientific surveys (Schummer and Eddleman 2003). Larger waterbird species often have greater 'flush distances' than smaller species to the approach of a disturbance such as a human or boat, and therefore management buffer distances need to take this into account (Rodgers and Schwikert 2003).

In most situations that have been studied, the creation of refuge areas for migratory waterfowl has generally increased the number of waterfowl using a site. However, experimental evidence to show that the increases were caused by reduced disturbance and not by confounding factors, e.g. improvement of habitat quality of the sites or general population increases, is generally lacking (Madsen 1998). To be able to answer questions about the effects of disturbance, a better theoretical framework, based on optimal foraging theory incorporating predation risk, and field experiments are required. Furthermore, we need to study waterbirds throughout their ranges to interpret the overall impacts of disturbance (Madsen 1995).

9.3 Hunting or harvesting – a special case

The role of hunting or harvest in the dynamics of waterbird populations continues to be debated among scientists and managers internationally. Other than the obvious mortality, hunting causes significant disturbance and consequently spatial and temporal displacement of quarry species (Madsen 1998). There is a large international literature on the subject that is beyond the scope of this review, with contrasts between research concerning waterfowl (ducks, geese and swans) and research concerning other waterbird species such as shorebirds. Waterbird hunting in Australia has significantly decreased in popularity, frequency and impact in recent decades. Here, examples of reviews of the impact of hunting on waterbird populations compared to other stressors are summarised.

Hunting by humans has severely threatened several shorebird species to the point of endangerment or even possible extinction (e.g. Eskimo Curlew Numenius borealis, Gillet al. 1998, Graves 2010 cited by Sutherland et al. 2012). Although hunting of shorebirds is banned in some developed countries, hunting and poaching elsewhere may undermine conservation measures. In the European Union (EU), for example, many waders listed under Annex II of

the EU Birds Directive (quarry list) are hunted. Reporting of the 'take' of these birds is clearly important to determine impacts (Sutherland et al. 2012).

Sedinger and Herzog (2012) reviewed the evidence for an impact of hunting on duck populations in North America, and found that understanding of the effects of harvest is limited because effects are typically confounded with those of population density; regulations are typically most liberal when populations are greatest. Consequently, even where hunting appears additive to other mortality, this may be an artifact of ignoring effects of population density. Overall, they found no compelling evidence for strong additive effects of harvest on survival in duck populations that could not be explained by other factors.

Similarly, Poysa et al. (2013) found that in general population trends of 16 species of waterbird were not strongly associated with hunting pressure in Finland or in Europe. Nor were basic life history characteristics (body mass and clutch size) associated with population trends of the species. In contrast, recent population declines were associated with habitat preferences of the species: those breeding mainly in eutrophic lakes had more negative population trends than those breeding in oligotrophic lakes or generalist species. Suggested reasons for the relatively poor status of species preferring eutrophic lakes included over-eutrophication of nutrient-rich lakes resulting in less abundant food resources, and increased nest depredation.

To examine the importance of hunting mortality in the population dynamics of waterfowl in southwestern Australia, Halse et al. (1993) estimated survival and recovery rates of 19,523 Pacific black ducks (*Anas superciliosa*) and 2,487 grey teal (*A. gibberifrons*) banded in 1968-76 and recovered by 1978. Both species showed strong evidence of year-to-year variation in survival rates, which averaged 63 +/- 4(SE)% and 56 +/- 6% for adult and young Pacific black ducks, respectively, and 55 +/- 17% for adult grey teal. First-year recovery rates indicated the average annual hunting mortality over 6 years for Pacific black ducks and grey teal exposed to heavy hunting pressure was 23 +/- 2% and 17 +/- 2%, respectively. Hunting mortality was 9% in 1 year for Pacific black ducks exposed to light hunting. Estimates of mortality rate on opening day of hunting seasons at a site of intense hunting varied between 7 and 20% for Pacific black ducks and 5 and 16% for grey teal. In an intensively hunted population in southwestern Australia, nearly 60% of all mortality of Pacific black ducks and 40% of grey teal mortality resulted from hunting. Hunting probably caused <25% of mortality in most other parts of southwestern Australia, however, and there was no evidence of hunting reducing survival rates (Halse et al. 1993).



10 Competition

Both inter- and intra-specific competition for resources can affect bird condition, behaviour, abundance, distribution, and survival, and can vary both spatially and temporally. While food and habitat niches are usually sufficiently different to avoid competition between species in a native community under normal conditions, the introduction of invasive species, increases in populations of particular native species driven by other factors, loss of critical habitat or food sources, or extreme weather conditions such as drought can significantly alter competition intensity and timing. When such changes take place concurrently, it can be difficult to identify causative mechanisms for shifts in waterbird distribution or changes in population parameters.

Competition for food may be classed as either interference (contest) or exploitative (scramble) competition. Interference competition may reduce the food intake of subordinate individuals, while exploitative competition may simply reduce local food availability (Gyimesi et al. 2010). Increasing numbers of a particular species with specific food and habitat requirements may result in increased local density-dependent interference and exploitative competition. This is particularly likely when decreases in habitat or food availability occur.

Such competition may particularly affect the movements, condition and survival of young, female, or subordinate portions of a population (Gyimesi et al. 2010; Amat and Rilla 1994; Stirneman et al. 2012; Minias et al. 2014). Individual 'despots' are also known to occur and may strongly affect habitat use and food intake of other individuals (Harper 1982). However the mechanisms of bird density and prey density effects on competition can be complex, differing between species (Vahl et al. 2005).

Depending on climate or weather over time, waterbird communities may exhibit periods of intense inter-specific and/or intra-specific competition, alternating with times of habitat or food abundance where competition may be insignificant or absent.

One study of a community of North American dabbling ducks (Anatidae) found that intensity of interspecific competition and its effect on community structure was seasonal, with greater resource limitation and, therefore, resource partitioning during "lean" winter months than during "fat" summer months (DuBowy 1988). In Alaska, it is unclear whether shifts in the distribution of goose species are caused by climate and ecological change (erosion and saltwater intrusion) altering optimal foraging habitats for moulting birds, or by interspecific competition, with one rapidly increasing species excluding another from preferred habitats (Flint et al. 2008). Gurd (2008) suggested that better understanding of the mechanisms regulating assemblages of waterbirds might emerge if patterns of resource use and species co-occurrence were predicted taking into account performance trade-offs and how they affect resource selection in the context of varying resource and competitor densities.

Competition for food between adults and juveniles can significantly affect juvenile condition, behaviour and possibly survival.

In migratory waterbird species, adults typically leave nesting sites before juveniles, and this is probably partly explained by the need for juveniles to accumulate higher body fat stores prior to travelling. However it also appears that juveniles are disadvantaged by the simultaneous presence of adults at stopover sites, with competition from older conspecifics causing juveniles to 'refuel' more slowly and attain lower fat reserves (Minias et al. 2014). Juveniles forced to compete with adults will also leave a site more quickly. Thus it is thought that delayed departure from natal grounds may be adaptive for juvenile waders, allowing them to mismatch the timing of their first migration with the peak of adult passage and, thus, reduce the negative consequences of intraspecific competition during migration (Minias et al. 2014). Departure dates for migrating Whooper Swans (*Cygnus Cygnus*) in Ireland are also thought to be affected by competition on wintering grounds (Stirneman et al. 2012).

An inverse association between egret breeding success and nesting numbers over 14 years at a colony in Queensland, Australia, has been suggested to be a density-dependent process, possibly based on competition for food (McKilligan 2001). This competition may occur between adults, or juveniles, or even between chicks in the nest. Cattle egret chicks hatch asynchronously, and it has been demonstrated experimentally that chick growth and mortality in the nest is significantly affected by competition between siblings for food and direct sibling aggression, with older chicks occasionally killing younger chicks (Fujioka 1985). Intraspecific predation has also been reported among juvenile White Ibis (*Eudocimus albus*) in the Florida Everglades (Herring et al. 2005).

Scarcity of preferred nesting sites is probably the key ecological factor underlying competitive behaviours during breeding, driven by natal philopatry, nest-site fidelity, aggressive competition for nest sites and intraspecific parasitism (Semel and Sherman 2001).

Both inter and intra-specific competition can occur between ducks via parasitism, where similar or the same species lay their eggs in each other's nests (Semel and Sherman 2001; Peron and Koons 2012). For example, in North America, Canvasback duck numbers have been found to positively affect Redhead duck fecundity, whereas Redhead numbers negatively affect Canvasback fecundity, due to parasitism. This interaction was modulated by wetland habitat availability, with Redhead hens exhibiting non-parasitising nesting behavior more frequently during years when more ponds were available (Peron and Koons 2012).

Some authors suggest that in waterbird populations strong competition is more likely to take place in connection with the breeding season (particularly pre-laying) than during non-breeding (e.g. Meltofte 1996). However this is not always the case (Fox and Bergersen 2005), and competition may occur at any time if prompted by conditions.

Invasive or pest species may compete with waterbirds for habitat or food.

Invasive species can compete with waterbirds directly or indirectly, and such competition may interact with habitat loss and predation. For example, a population increase of the invasive Caspian gull at a lake in Poland displaced all the native waterbird species from their preferred breeding islets, forcing them to lower-quality islets close to the shoreline. The latter were frequently visited by magpies, which hunted nests, leading to an up to threefold decrease in nesting success as compared with nests located in the preferred islets in the centre of the invaded reservoir. Thus although direct predation by the invasive Caspian gulls was rarely observed, their arrival resulted in increased predation rates for local waterbird species regardless (Skorka et al. 2014).

Competition may even occur in terms of destruction of nests and eggs. For example, European rabbits *Oryctolagus cuniculus* have been photographed burying eggs and destroying artificial nests of terns (Nolfo-Clements and Clements 2011). Competition for food from invasive species can also be important. Introduced rats and mice have been shown to deplete insect populations in the Falkland Islands and the Antipodes Islands, which could affect waterbirds (Sutherland et al. 2012 citing St Clair et al. 2011; Marris 2000).

Direct competition between introduced or native fish and waterbirds for a food resource may also affect waterbird abundance. For example, tufted duck (*Aythya fuligula*) and introduced roach (*Rutilus rutilus*) in a lough in Northern Ireland are thought to be the only significant consumers of molluscs, and their population fluctuations mirror each other (Winfield and Winfield 1994). Fish density has been shown to affect the condition and mortality rates of mallard (*Anas platyrhynchos*) ducklings, primarily through competition for invertebrate prey (Hill et al. 2008).

In Australia, there is potential for competition between increasing populations of the self-introduced cattle egret and some waterbird species with similar foraging preferences such as the white-faced heron. This could lessen the chances of survival or reproduction of both groups when food is scarce. The central coastal areas of eastern Australia that have attracted the greatest number of cattle egrets are important dry season refuges for birds such as the straw-necked ibis, which nest in inland wetlands (McKilligan 1975). Diet studies have indicated that interspecific competition for food between cattle egret and intermediate egret nestlings is unlikely, since they have only small diet overlap (Baxter and Fairweather 1989). However a detailed comparison of the food, feeding ecology and feeding behaviour of the cattle egret and native species at a time of food shortage is needed to assess the extent to which they compete for food (McKilligan 1984).

Feature abstract

Although the common carp (Cyprinus carpio), an invasive benthic fish from Eurasia, has long been strongly implicated in the disappearance of vegetative cover and reduced waterfowl abundance in North American shallow lakes, the details of this relationship are obscure. This study documented ecological changes in a recently restored shallow lake (Hennepin and Hopper Lakes, IL, USA) at a time that it was experiencing a large increase in its carp population. We estimated the abundance and biomass of carp 7 years after this lake had been restored and then back-calculated carp population size across time while examining changes in the lake's plant and waterfowl communities. We found that the biomass of carp remained below similar to 30 kg/ha for 5 years following restoration, but then increased to similar to 100 kg/ha in the sixth year following a strong recruitment event. Although a carp biomass of <30 kg/ha had no discernible effects on vegetative cover (which exceeded 90%) or waterfowl (which exceeded 150,000 individuals during fall censuses), the increase to 100 kg/ha was associated with a similar to 50% decrease in both vegetative cover and waterfowl. A further increase in carp biomass to over 250 kg/ha during the seventh year coincided with a decrease in the vegetative cover to 17% of the lake's surface and a decline in waterfowl use to similar to 10% of its original value. These data suggest that the common carp is extremely damaging to the ecological integrity of shallow lakes when its density exceeds similar to 100 kg/ha. Since the biomass of carp in Midwestern shallow lakes commonly exceeds this value by 3-4 times, it seems likely that carp are responsible for the large-scale habitat deterioration described in many of these ecosystems (Bajer et al. 2009).

11 Interactions and links between flooding, stressors, threats and waterbird responses

There are interactions between flooding and all of the stressors discussed in this review in terms of impacts on waterbirds. Many of these are discussed in the previous sections. These interactions and their effects are complex and difficult to untangle. Some examples of interactions mentioned in the literature are described briefly here, followed by conceptual model diagrams showing the complexity of the interactions and links.

All population parameters may be affected by interactions between climate, weather, food and habitat availability, predation, disturbance, pollution, disease, and intra- and inter-specific competition (Bridgeman et al. 1998; McKilligan 1987; 1997; 1997; Drent et al. 2007).

Significantly, these interactions are just as important at foraging (non-breeding) sites as they are at breeding sites, and frequently take effect over multiple spatial and temporal scales.

For example, linking spring migratory itineraries of individual Arctic-breeding geese to their eventual breeding success has provided evidence that accumulation of body stores (protein, fat) at stop-over sites is crucial. This is because geese nesting in the Arctic depend at least in part on these stores for synthesis of eggs and supporting incubation (for the female, a phase of starvation). Estimates of the body stores needed for successful reproduction (eggs + incubation) in relation to measured rates of accumulation of these stores make clear that meeting the demands solely by feeding at the breeding grounds is not an option for geese. The time constraint does not allow this, because early laying is a necessity in the Arctic to ensure survival of the progeny. Although the parents can exploit the early spring growth along the flyway, they get ahead of the wave of growth when they arrive on the breeding site and hence the parental timetable can only be met by drawing on body stores. Results from tracking studies in six goose species underline the conclusion that egg formation commences along the flyway before arrival at the nesting colony. In some cases, signatures of stable isotopes in egg components and parental body tissues in relation to the signature in forage plants support the notion of a mixed endogenous/exogenous origin. The close match between migratory timing and the spring flush of plant foods makes geese particularly vulnerable to the impact of climate change. There is an increasing mismatch along the NE Atlantic Flyway, where a warming trend in NW Europe conflicts with stable or even cooling trends in the Arctic target areas (Drent et al. 2007).

Flooding, water levels, landscape configuration and predation can interact to influence waterbird population dynamics, particularly through their effects on nest success (Heath et al. 2006).

Reduced flows may increase predator access to nests, while very high flows may inundate nests and kill eggs or young (Cruz et al. 2013). A study of cavity-nesting Wood Ducks (*Aix sponsa*) in the Mississippi River floodplain (Nielsen and Gates 2007) found that nests were depredated prior to and after, but not during, a four-week flood, and in general nest predation was lower and nest success was higher in the floodplain during floods. Flood conditions did not significantly affect nest success or nest predation rates in the adjacent upland, but nest success in upland forest was higher than in the floodplain during dry periods. The authors concluded that historically, spring flooding of riparian areas may to some extent have mitigated nest predation in floodplain forests (Nielsen and Gates 2007). In a study of nest success of Ferruginous Ducks *Aythya nyrocaas* in Hungary (Purger and Meszaros 2006), as water levels surrounding nests decreased, nests became more accessible to Wild Boar Sits scrofa and other land mammal predators, which increased the rate of predation. The authors suggested that maintaining water levels would not only decrease the predation rate of nests, but would also maintain feeding areas for ducks. Wild Boar were the main cause of clutch loss in this area, and therefore by management measures, such as a reduction in their abundance or attracting them away from potential nesting sites by providing food elsewhere, the breeding success of the Ferruginous Ducks may be further improved (Purger and Meszaros 2006).

Predator abundance, predation rates and habitat type also interact with weather and climate to influence breeding success (Smith et al. 2010; Phillips et al. 2003).

Predator control may mitigate the effects of land use change or climate change by enhancing waterbird breeding success, particularly where climate effects coincide with changing land use (Douglas et al. 2014; Padysakova et al. 2010).

Disturbance by humans and predator abundance also interact to affect parent bird behaviour at nests (St Clair et al. 2010). Disturbance by humans is also known to influence predation rates, making prey species more susceptible to predators (Dorfman and Read 1996).

For example, Mikola et al. (1994) studied the consequences of human disturbance and gull predation on brood survival of velvet scoter *Melanitta fusca* in the Archipelago of South-West Finland in 1990 and 1991. Each brood was exposed to disturbance by boats on average 8.5 times a day in 1990 and 3.5 times a day in 1991. Disturbance lengthened the swimming distances of ducklings and reduced the time used for feeding. Broods disturbed more frequently than average were smaller than those disturbed less frequently. At least 60% of ducklings died before the age of three weeks, and whe frequency of gull attacks was 3.5 times higher in disturbed than in undisturbed situations (Mikola et al. 1994). Predation rates at a breeding site may also vary with changes in abundance of other prey (e.g. rabbits or rodents) and changes in habitat (Norbury and Heyward 2007). Changes in the behaviour of parent birds in response to predation pressure, can in turn affect nest success, depending on food and habitat availability at the time. Birds may even adjust their nesting habitat selection in response to an increase in the risk of egg predation (Rothenbach and Kelly 2012; Colcherc et al. 2010).

Changes in land and water use and global climate may result in a higher concentration of waterbirds on remaining high-quality sites, making them potentially more vulnerable to disease and competition (Krauss et al. 2010 cited by Sutherland et al. 2012). Flood regimes may also interact with disease to influence waterbird populations.

For example, spores produced by the bacterium responsible for Botulism persist in the environment for years, so outbreaks may occur repeatedly in the same location. The frequency and timing of inundation both interact with spore presence to influence the effects of Botulism on birds using a site.

The introduction of invasive species, increases in populations of particular native species driven by other factors, loss of critical habitat or food sources, or extreme weather conditions such as drought can significantly alter inter- and intra-specific competition intensity and timing.

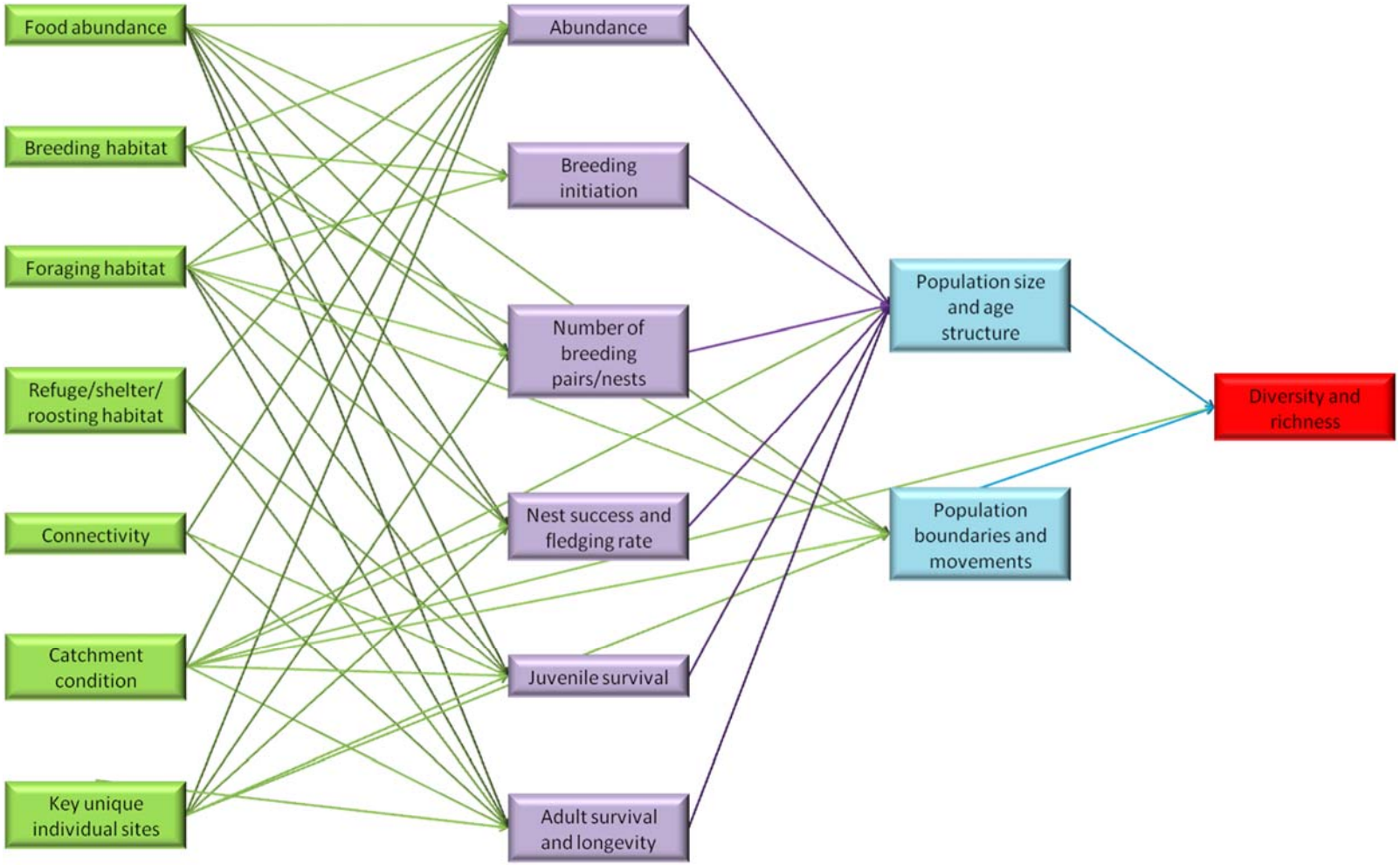


11.1 Conceptual model of interactions and links

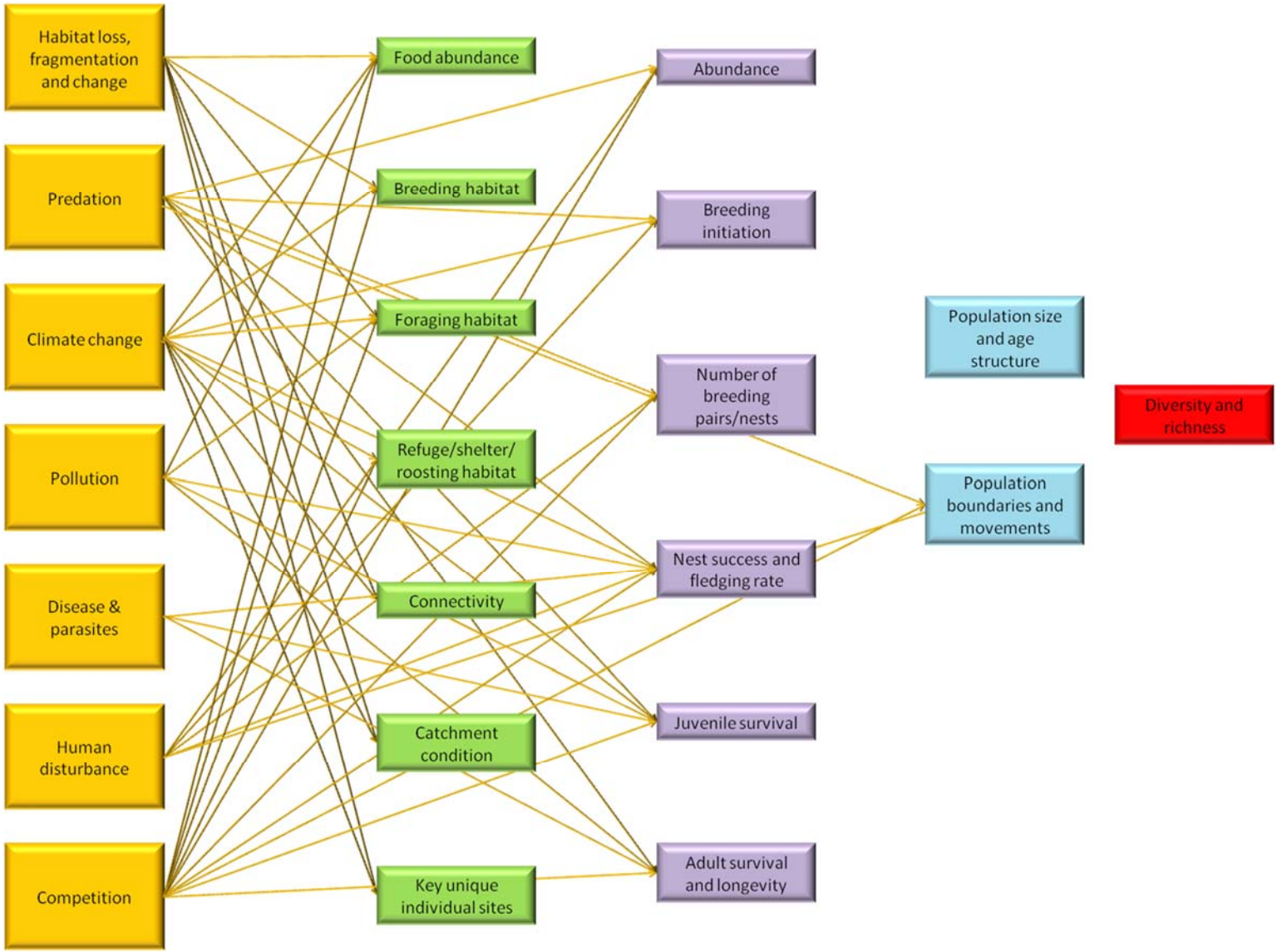
It is impossible to clearly represent a conceptual model in two dimensions that includes detailed relationships and interactions between all the different drivers, stressors, threats and responses discussed in this review. In order to simplify this complexity and communicate the primary variables of concern that are discussed in the literature, this conceptual model is presented in three diagrammatic stages:

1. A diagram simply depicting important food and habitat related drivers highlighted in the literature and their main links with waterbird response variables. For example, relatively few 'key unique individual sites' can be of disproportional importance for waterbird foraging and nesting, affecting waterbird population movements and nest success. During the 2008 Australian National Waterbird Survey, over 50% of recorded waterbirds occurred in only 41 wetlands or 1.1% of the total number of wetlands surveyed. Nearly forty percent of all recorded waterbirds occurred on the top 20 wetlands.
2. A similar diagram adding stressors and threats, depicting how they may act through multiple mechanisms, often through key habitat or food drivers, but also directly. These may effect change either simultaneously or independently. For example, climate change may affect nesting and foraging habitats and therefore nest success indirectly, or it may directly affect nest success through extreme weather events reducing fledging rates.
3. An integrative diagram adding flow/flooding variables, but still including all major variables presented in the previous two diagrams. This diagram is structured differently because it is intended to represent the multi-layered complexity of the many potential interactions that may occur in different directions. For example, climate change may affect flood frequency, which in turn exacerbates habitat change/loss, which may play out via nesting, foraging, or refuge habitats or via loss of connectivity, loss of key unique individual sites, changes in catchment condition, or reduced food abundance. These in turn may affect a range of waterbird response variables, such as fledging rates or juvenile survival, which ultimately affect species persistence and both local and overall species diversity in the long term.

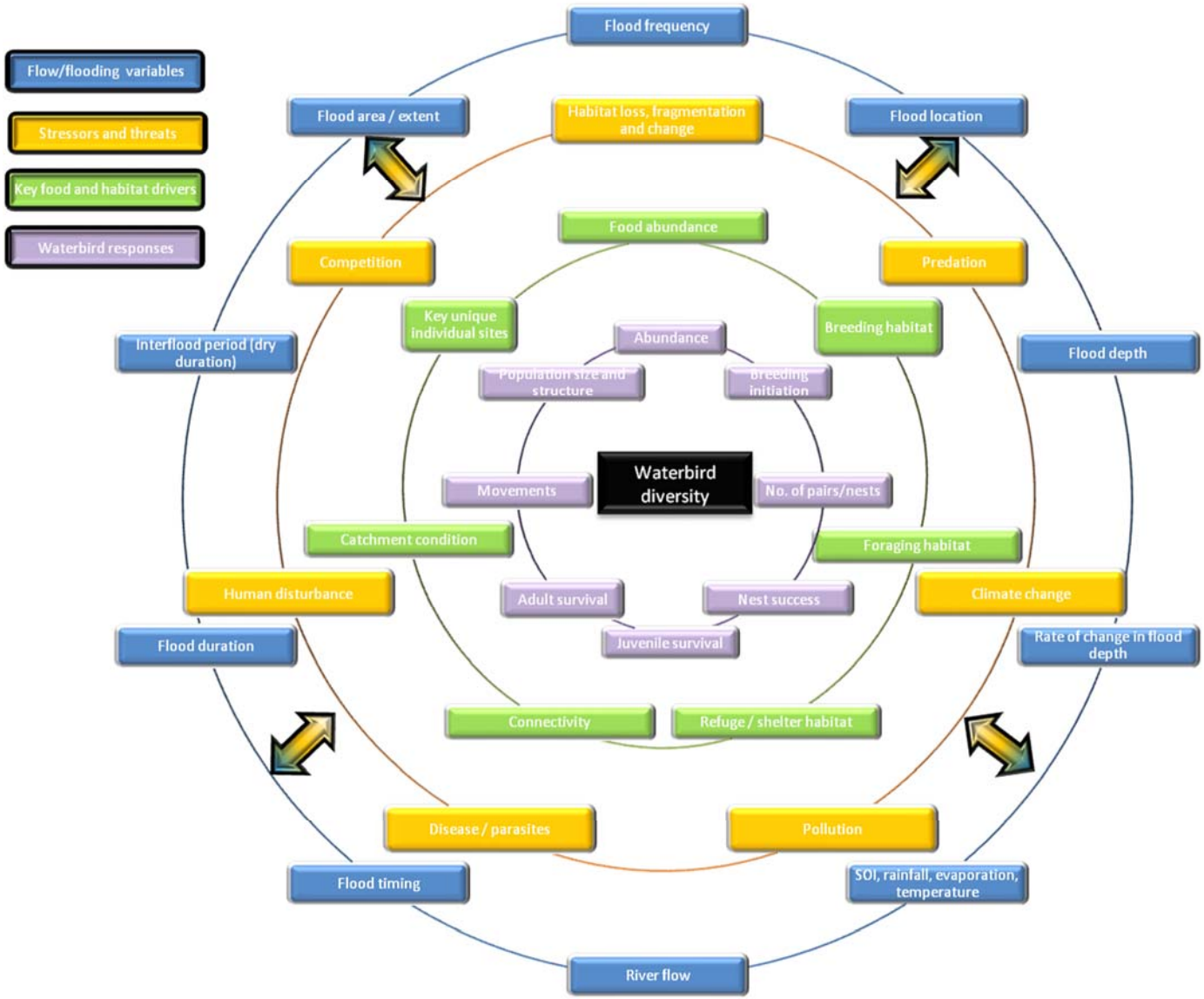
1. At the core are key food and habitat drivers of waterbird demographics, behaviour and life history, which determine population size, age structure, population boundaries and movements, and ultimately diversity and richness. All of the key food and habitat drivers are strongly driven by flows and flooding.



2. Stressors and threats act through multiple mechanisms, often through key habitat or food drivers, but also directly. They may effect change either simultaneously or independently.



3. Flows/flooding, stressors and threats may each act directly on key food and habitat drivers to affect waterbird responses, but also frequently interact.



Appendix A Part I Summary table of waterbird responses to flows and flooding in Australia

Please see attached MS Excel spreadsheet:

'Part A Summary of waterbird responses to flows.xlsx'

Appendix B Part II Web of Science Advanced search string example

An example of the literature search strings used in Web of Science Advanced for Part II of the review is presented below. Multiple searches with various modifications were conducted, including a specific search for Australian literature. Repeat searches were also conducted in Google Scholar.

(TI=vegetat* OR TI=clear* OR TI=fragment* OR TI=connect* OR TI="habitat loss" OR TI=loss* OR TI=degrad* OR TI="land use" OR TI=landuse OR TI=graz* OR TI=crop* OR TI=harvest* OR TI=hunting OR TI=irrig* OR TI=pollut* OR TI=toxic* OR TI=poison* OR TI=insecticide* OR TI=herbicide* OR TI=sedimentation OR TI=anoxic OR TI=blackwater OR TI=eutrophi* OR TI=nutrient* OR TI=touris* OR TI=agricultur* OR TI=intensification OR TI=industr* OR TI=residential OR TI=disturbance OR TI=disease* OR TI=virus* OR TI=illness* OR TI=salini* OR TI="climate change" OR TI=temperature OR TI=competit* OR TI=predat* OR TI=invasi* OR TI=pest* OR TI=feral* OR TI=exotic* OR TI=alien* OR TI=foreign OR TI=threat* OR TI=fox* OR TI=vulpes OR TI=cat* OR TI="Felis catus" OR TI=pig* OR TI="Sus scrofa" OR TI=carp OR TI="Cyprinus carpio") AND (TI=recruit* OR TI=mortality OR TI=declin* OR TI=surviv* OR TI=abundance OR TI=forag* OR TI=breed* OR TI=nest* OR TI=fledg* OR TI=juvenile OR TI=young OR TI=immature OR TI=longevity OR TI=population OR TI=movement* OR TI=dispers* OR TI=migrat* OR TI=divers* OR TI=richness OR TI=communit* OR TI=persistence OR TI=occupancy OR TI=success OR TI="food-web" OR TI="food web" OR TI=ecolog* OR TI=biolog* OR TI="life history" OR TI="life-history") AND (TI=waterbird* OR TI="water bird" OR TI=waterfowl OR TI="water fowl" OR TI=shorebird* OR TI="shore bird" OR TI=podicipedi* OR TI=pelecanid* OR TI=anhingid* OR TI=phalacrocoracid* OR TI=ardeid* OR TI=plataleid* OR TI=anatid* OR TI=gruid* OR TI=rallid* OR TI=charadrii* OR TI=rostratulid* OR TI=scolopacid* OR TI=recurvirostrid* OR TI=larid* OR TI=pelican* OR TI=ciconiiform* OR TI=anseriform* OR TI=gruiform* OR TI=duck* OR TI=cormorant* OR TI=egret* OR TI=spoonbill* OR TI=heron* OR TI=ibis* OR TI=bittern* OR TI=crake* OR TI=grebe* OR TI=geese OR TI=goose OR TI=swan* OR TI=gallinule* OR TI=crane* OR TI=wader* OR TI=tern* OR TI=snipe) AND (SU=Agriculture OR SU=Behavioural Sciences OR SU=Biodiversity OR SU=Conservation OR SU=Biodiversity & Conservation OR SU=Biophysics OR SU=Environmental Sciences & Ecology OR SU=Ecology OR SU=Entomology OR SU=Environmental OR SU=Evolutionary Biology OR SU=Biology OR SU=Fisheries OR SU=Forestry OR SU=Genetics & Heredity OR SU=Genetics OR SU=Marine & Freshwater Biology OR SU=Freshwater OR SU=Water OR SU=Plant Sciences OR SU=Veterinary Sciences OR SU=Zoology OR SU=Physical Geography OR SU=Water Resources) NOT (TI=patient* OR TI=cell* OR TI=enzyme* OR TI=embryo* OR TI=yolk* OR TI=sac* OR TI=clinical OR TI=cultur* OR TI=speciation OR TI=phylogenetic* OR TI=Holocene OR TI=Pleistocene OR TI="deep-sea" OR TI=marine OR TI=reef* OR TI=seabird OR TI="sea bird" OR TI=sea OR TI=ocean* OR TI=stygo* OR TI=diatom* OR TI=shear OR TI=plate* OR TI=uplift OR TI=laser* OR TI="nutrient reserves" OR TI=duckweed* OR TI=Ducke OR TI=Pekin OR TI=meat* OR TI=digestive OR TI=antibacterial OR TI=supplement* OR TI=polyculture OR TI=H5N1 OR TI="avian influenza" OR TI=H6N2 OR TI=HPAI OR TI=blackbird*)

Of over 900 references identified that were directly relevant to waterbird responses to stressors and threats other than changes in flow/flood regime, only 40 (< 5%) were from Australia. Many of the latter were short anecdotal records in small bird club journals regarding one or two species or single events. Threats and stressors for waterbirds other than changes in flows/flooding are mentioned in some Australian reviews, but usually only briefly or as factors complicating results and requiring consideration. No comprehensive or integrative studies were located of major stressors and threats for Australia's waterbirds, or how these interact with each other or flows and flooding – except for migratory shorebirds whose populations are of international concern.

Habitat alteration (loss, fragmentation and change) and predation were by far the most common stressors in the literature reviewed (approximately 25% of references each), and were often discussed as interacting factors. The influence of climate change (and weather) was frequently mentioned but not often directly studied (approx. 14% of references). Pollution, disease, human disturbance and competition were all regarded as significant threats to waterbirds, however were relatively less frequently encountered in the literature reviewed.

Breeding (especially nesting) was the most common response variable addressed by far (63%). Effects of stressors on foraging or feeding were at least mentioned in approximately 26% of references reviewed, and survival, mortality, or recruitment were mentioned in approximately 20% - however very few studies actually measured these variables. The term recruitment was rarely used in titles or abstracts (3% of references).

In terms of stressors and threats, relevant literature is relatively scant for Gruiformes (cranes, rails, crakes and gallinules; approximately 4% of literature reviewed), for Pelecaniiformes (pelicans and cormorants; 9% and for Ciconiiformes (bitterns, egrets, herons, ibis, storks; 29%). Anseriformes (waterfowl: ducks, geese and swans - especially migratory species) and Charadriiformes (especially migratory shorebirds: sandpipers, stilts and terns) have been far more commonly studied (46% and 44% respectively).

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DRAFT MANUSCRIPT IN PREPARATION FOR SUBMISSION TO A SCIENTIFIC JOURNAL FOR PUBLICATION

Satellite-tracking ibis and spoonbill movements and habitat selection: Implications for water and wetland management

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DRAFT

Abstract

Flow regimes, water management and threats such as habitat change and habitat loss affect the availability and quality of foraging sites for waterbirds at multiple scales. These in turn affect bird survival, particularly of young birds - and consequently recruitment and population persistence. While increasing waterbird populations and maintaining waterbird diversity are often important goals for environmental water management, knowledge gaps exist that affect our ability to manage and predict waterbird populations and diversity at appropriate scales. One of the largest of these gaps in knowledge is nomadic waterbird movements and their drivers, including: Timing; distances travelled; differences between juveniles and adults; key foraging habitat locations and characteristics; effects of habitat availability; quality and productivity on site choice, bird condition and survival; site fidelity; and mechanisms, cues or drivers inducing movements and choices and how these interact.

To begin filling these knowledge gaps, we used GPS satellite transmitters and the ARGOS satellite network to track the movements of colonial-breeding wading birds in the Threskiornithidae family – including straw-necked ibis (*Threskiornis spinicollis*), Australian white ibis (*T. molucca*), and royal spoonbills (*Platalea regia*). These species are of particular interest for management and policy decision-makers and are good representatives of colonial-breeding wading birds dependent on environmental water.

Specifically, the research aimed to address the following questions: 1) How far, how often, and when do these species move? 2) What are the characteristics of foraging and roosting habitats selected? 3) What are the characteristics of movement routes? and 4) Are there differences in the above between species, age categories, capture sites or years?

Keywords

Threskiornithidae; environmental water; satellite tracking; foraging habitat; nomadic; partial migration

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Introduction

Flow regimes, water management and threats such as habitat change and habitat loss affect the availability and quality of foraging sites for waterbirds at multiple scales. These in turn affect bird survival, particularly of young birds - and consequently recruitment and population persistence. While increasing waterbird populations and maintaining waterbird diversity are often important goals for environmental water management, knowledge gaps exist that affect our ability to manage and predict waterbird populations and diversity at appropriate scales. One of the largest of these gaps in knowledge is nomadic waterbird movements and their drivers, including: Timing; distances travelled; differences between juveniles and adults; key foraging habitat locations and characteristics; effects of habitat availability; quality and productivity on site choice, bird condition and survival; site fidelity; and mechanisms, cues or drivers inducing movements and choices and how these interact.

These knowledge gaps exist even for common and conspicuous taxa such as colonially-nesting waterbirds – taxa that are often thought to be relatively well-understood. Filling these knowledge gaps would assist water and land managers to:

- Identify, maintain, and/or restore key waterbird habitats – especially critical foraging habitats
- Better understand the spatial and temporal scales at which key habitat characteristics are required
- Better target water, vegetation and threat management actions to ensure 'event readiness' at nesting sites between flooding events and maximise waterbird recruitment
- Better predict the effects of water management and threats

To begin filling these knowledge gaps, we used GPS satellite transmitters and the ARGOS satellite network to track the movements of colonial-breeding wading birds in the Threskiornithidae family – including straw-necked ibis (*Threskiornis spinicollis*), Australian white ibis (*T. molucca*), and royal spoonbills (*Platalea regia*). These species are of particular interest for management and policy decision-makers and are good representatives of colonial-breeding wading birds dependent on environmental water.

Specifically, the research aimed to address the following questions:

1. How far, how often, and when do these species move? (Movement statistics)

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2. What are the characteristics of foraging and roosting habitats selected? (Habitat statistics)
3. What are the characteristics of movement routes? (Mapping)
4. Are there differences in the above between species, age categories, capture sites or years?

Methods

Species

Colonially-breeding wading waterbird species in the Threskiornithidae family were chosen for tracking because: a) They are important species for environmental flows management and policy – increased knowledge regarding these species will assist land and water managers to achieve their stated aims and objectives; b) Locations of major colonies are known, as are some of the breeding thresholds related to flows and inundation; c) however there are major knowledge gaps regarding their movements, demographics, recruitment and behaviour.

Straw-necked ibis (*Threskiornis spinicollis*) were chosen as the primary species receiving the majority of the trackers because they are good representatives of colonially-breeding waterbirds, are known to nest in large numbers in nearly all major wetlands managed with environmental flows, and are a species of particular interest to managers. Straw-necked ibis forage for prey in a range of habitats including wetlands, grazing lands, crops, feedlots, and sports grounds. On agricultural land they eat pest insects such as locusts. When in wetlands they eat frogs, aquatic invertebrates, spiders, fish, molluscs and small reptiles. They are highly mobile and nomadic.

Two other species that frequently nest together with straw-necked ibis were also tracked in smaller numbers, in order to explore differences between species: The royal spoonbill (*Platalea regia*) and the Australian white ibis (*Threskiornis molucca*). Royal spoonbills are also highly mobile and nomadic and are known to nest in most major wetlands managed with environmental flows. Yet they have different preferences, preferring to forage in shallow water consuming fish, crustaceans and aquatic invertebrates. The Australian white ibis is thought to occupy foraging habitat niches between those of the straw-necked ibis and royal spoonbill, foraging on a wide range of food sources. In the last few decades the Australian white ibis has colonised and begun breeding in significant numbers in some of Australia's

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major coastal cities, while the straw-necked ibis and royal spoonbill are more restricted to natural habitats.

Transmitters

Transmitters were solar-powered and recorded hourly diurnal location fixes 0700-1900 hr and a midnight fix, usually with a resolution of <26 m. Ground checks found that resolution of stationary transmitters was frequently better than 15 m. Location fixes were recorded and stored for 36 hours, and then transmitted during an 8-hour window via the ARGOS satellite system. The ARGOS satellite system provides continual full coverage within Australia and worldwide, with no recapture or base station required, thereby avoiding issues with phone network coverage in inland Australia and providing data suited to highly nomadic species. It also allows transmitters to be located and recovered when they eventually fall off the bird or if the bird dies.

Three transmitter sizes were used: 22g, 40g and 52g. Birds were weighed prior to fitting transmitters to ensure that transmitters were less than 5% of bird weight. Transmitter percentages of adult weight ranged from 1.3% to 4.2%; percentages of juvenile weight ranged from 1.0% to 3.2%. Any adult bird lighter than 1,100 grams and any juvenile bird lighter than 1,000 g was not fitted with a transmitter and was immediately released. Initial transmitter weight limits were determined by examination of adult and juvenile bird weight data from HANZAB, the Australian National Wildlife Collection, and previous studies. Weight ranges for straw-necked ibis are generally 1,100g – 1,500g; Australian white ibis 1,400 - 2,500g; and royal spoonbill 1,500-1,800g (Marchant and Higgins 1990). Juveniles ready to leave the nesting area are generally similar weights to adults. For example, juvenile straw-necked ibises captured as part of this study ranged from 1,000-1,806g, average weight 1,501g; while adults ranged from 1,230-1,850g, average weight 1,546g.

The transmitters were fitted using a Teflon tape harness stitched with cotton thread, designed to degrade over a period of time (3-4 years) after which the harness will break and fall off the bird. The Teflon tape is ~10mm wide and ~1mm thick, non-absorbent and widely used in tracking studies in Australia and worldwide as it reduces the likelihood of the harness interfering with plumage. Juvenile harnesses were fitted with an adjustment mechanism in the form of a gather using dissolvable sutures and/or elastic material for the keel-strap, allowing the harness to expand with the growth of the bird.

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Field deployment

Satellite transmitters were deployed on 64 birds of three species over two years (Table): 42 Straw-necked Ibis (*Threskiornis spinicollis*), 15 Royal Spoonbill (*Platalea regia*), and seven Australian White Ibis (*Threskiornis molucca;* Table).

In the first year, 2016-2017, 20 GeoTrak ARGOS GPS tracking transmitters were fitted to straw-necked ibis; 10 adults and 10 juveniles. Five transmitters were fitted to adults in the Macquarie Marshes, northern NSW (Figure). Five transmitters were fitted to adults at Barmah-Millewa Forest, on the NSW/VIC border, together with10 transmitters fitted to juveniles. The decision to deploy five transmitters in the Macquarie Marshes in 2016-2017 was driven by a major breeding event occurring there, which was unlikely to occur again during the life of the project and was an opportunity to gain some information about where ibis from the Macquarie Marshes go versus where ibis from the Barmah-Millewa Forest go in the same year (northern Murray-Darling Basin vs southern Murray-Darling Basin).

Another 30 22g GeoTrak ARGOS transmitters were deployed during the 2017-2018 season. Also, 14 recovered transmitters were re-deployed. In total, 22 straw-necked ibis, 15 royal spoonbills, and seven Australian white ibis were fitted with transmitters in 2017-2018. All of the ibis were captured in Kow Swamp and Kerang Middle Lake in Victoria, while 13 of the royal spoonbills were captured in Barmah-Millewa Forest and two were captured in Kerang Middle Lake. The 2017-2018 season was more restricted than 2016-2017 in numbers of breeding locations and breeding birds. Breeding commenced later in the year at most known sites.

Table 1 Number of transmitters deployed per species, age group and breeding site

	Breeding site				
Species and age group	**Barmah Millewa**	**Kerang**	**Kow Swamp**	**Macquarie Marshes**	**Grand Total**
Australian white ibis			**7**		**7**
Juvenile			7		7
Royal spoonbill	**13**	**2**			**15**
Adult	1				1
Juvenile	12	2			14
Straw-necked ibis	**15**	**7**	**13**	**5**	**40**
Adult	5	1	3	5	14
Juvenile	10	6	10		26
Grand Total	**28**	**9**	**20**	**5**	**62**

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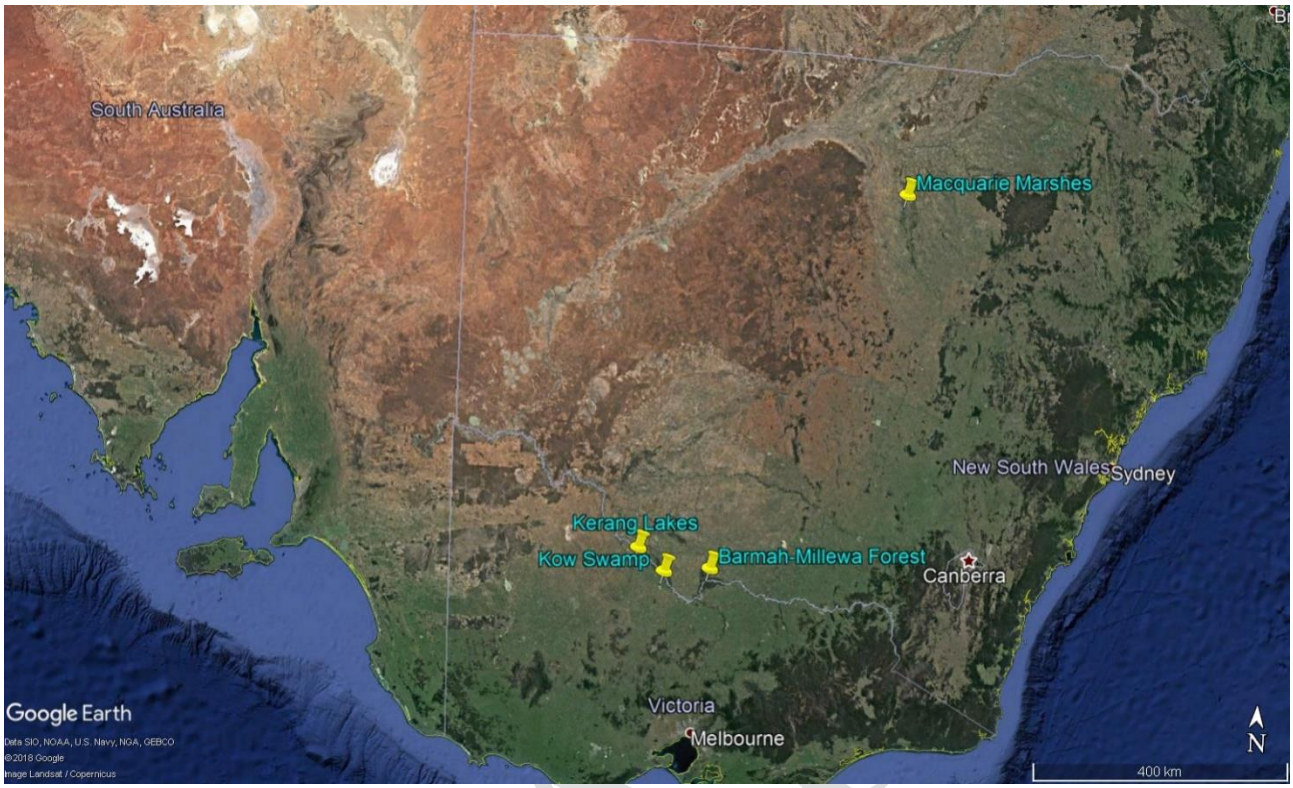


Figure 1 Sites at which birds were captured for satellite-tracking, south-eastern Australia (yellow markers).

Birds were caught either by hand, or with a slip noose trap or with a hand held net or net launcher, and placed into calico bags for processing. Each bird was sexed, weighed, measured for bill and head-bill length, tarsus length, tail length and wing length, sampled for five feathers, and photographed. Where possible, each bird was also leg-banded. Banded birds were fitted with an orange band over a stainless steel numbered band on the right tibia (above the knee) – the orange indicating that it is a bird from our study. Birds were also fitted with two colour bands on the left tibia to identify the individual from a distance.

The tracking harness with the tracker attached was placed over the bird's head and the head placed back into the bag to keep the bird calm. Each wing was placed through the harness loops and the keel strap connecting the two loops that have been placed over the wings positioned on the keel. From the back of the transmitter the loops were tightened to the appropriate fit and the position of the transmitter and keel strap checked. Adjustments were made as required, then the final knot was tied, glued and stitched with cotton thread and the excess material trimmed. Adult and juvenile birds were released at the capture location immediately after processing.

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Data analysis

Mapping and classification

Data were downloaded from ARGOS weekly using custom software. Mapping was automated using Python and ArcGIS including conversion to local time, colour-classification of morning, afternoon and midnight fixes and export of kmz, jpg and pdf files.

Classification of the tracking data was also conducted in ArcGIS. The classification produced three base datasets: Points, Lines, and Midnights (midnight reflecting roosting locations for ibis species). Each dataset contained a range of calculated movement variables and classes such as the activity at each point (roost, forage, travel) and the maximum distance travelled from the roost per day (see Table and supplementary material table).

Each bird was assigned one of three status types: Active; Missing; or Deceased.

- **Active** birds were still alive and mobile with their satellite transmitter sending data at the time specified during analysis (eight individuals at the time of analysis).
- **Missing** birds were birds for which satellite data had not been received for an unusual duration at the time specified or that had lost their transmitters. Loss of satellite data may indicate insufficient solar recharge of the transmitter (e.g. from feather coverage of the solar panels or because the bird is deceased) or damage to the transmitter.
- **Deceased** birds were confirmed dead or presumed dead based on available evidence at the time specified during analysis.

The date of decease was determined from the satellite tracking data, including spatial and temporal variation in GPS fixes and transmitter measures of activity and battery voltage.

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Table 2 Primary movement variables and classes derived from ArcGIS analyses

Base dataset	Variable/factor	Name	Categories / description
Points	Activity At Point	Activity	Roost, forage, travel
Points	Daily Translocation Type	Translocation	Stayed, shifted, moved, travelled
Lines	Hourly Movement Type	Movement	Staying, shifting, moving, travelling
Lines	Hourly Movement Length Type	Length	Minimal, short, medium, long, very long, extremely long, unusually long
Midnights	Total Distance Travelled Daily	DistTravelled	Cumulative distance travelled along line segments during previous day (km)
Midnights	Distance Ranged	DistRanged	Maximum straight line distance between previous midnight and any point during the day (km)
Midnights	Range Type	Range	None, adjacent, near, local, far, distant
Midnights	Roost Type	RoostType	Reside, Explore, Layover
Midnights	Distance From Previous Night Roost (km)	DistPrevNight	Straight line distance between this and previous midnight roost (km)

Movement statistics

Basic movement statistics were calculated for both the raw data and for randomly sampled data. The randomly sampled dataset consisted of a repeat random sample of 40 (non-consecutive) days of data for each bird, 100+ times, to reduce temporal autocorrelation and avoid having particular birds with long tracking records biasing results. Comparisons between the raw and the randomly sampled data indicated no significant influence of temporal autocorrelation or bias.

Comparisons between habitats selected and habitats available

To facilitate comparisons between habitat selected and habitats available, a buffer was created around the lines (movements) where buffer distance was the distance travelled along that line from one point to the next point. Within the buffer area for the entire 24 hours, 14 random points were generated (the same as the maximum number of tracking fixes per day). Using this method, long distance movements were therefore allocated larger buffer areas (and choice of habitats) than were short distance movements. The date, point ID and bird ID were then exported to a separate file. These random points were then used to sample the context layers / data using ArcGIS, with the results exported separately for each individual.

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Results

The average duration of tracking per individual was 174 days, with a maximum of 588 days (Table ; March 2019, eight birds still active). Even with short-duration tracking, detailed data were available due to the hourly frequency of location fixes – particularly for the post-nesting period when adults and juveniles dispersed from the nesting sites to forage. Overall, the tracking produced ~10,800 days of tracking data and ~150,000 location fixes across the Murray-Darling Basin.

Table 3 Summary statistics of the total number of days tracked per bird, by species

	Transmitter count	Mean	SD	Min	Max
Australian White Ibis	7	261	210	11	462
Royal Spoonbill	15	69	61	15	256
Straw-necked Ibis	40	198	174	3	588
All	**62**	**174**	**169**	**3**	**588**

Altitude and speed

Six transmitters were fitted with altitude and speed recording capability; four straw-necked ibis and two royal spoonbills. Three of the straw-necked ibis left their nesting sites (one adult and two juveniles), while one had a malfunctioning transmitter. The two juvenile royal spoonbills remained at or near the nesting site before their transmitters ceased to function. Long-distance flights were associated with higher altitudes and faster speeds. The maximum altitude recorded was 2,524 m, recorded by a juvenile straw-necked ibis. The maximum speed was 62 km/h, recorded by an adult female straw-necked ibis.

Table 4 Altitude and speed summary statistics

	Sp.	Sex	Age	Capture Location	Altitude (m)				Speed (km / hr)			
ID					**Records**	**Max**	**Mean**	**SD**	**Records**	**Max**	**Mean**	**SD**
165104	SNI	F	A	Kow Swamp	2996	1587	50	109	3122	62	1	6
041446	SNI	M	J	Kow Swamp	1643	1020	278	128	1721	53	1	4
041454	SNI	M	J	Kow Swamp	1543	2524	148	137	1604	56	1	5
133517	RBS	U	J	Barmah Millewa	1199	266	96	16	1236	47	1	2
133518	RBS	U	J	Barmah Millewa	332	180	98	14	359	18	0	1

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Departures from natal / nesting sites

Departures from nesting sites tended to occur in clusters. Royal spoonbills departed nest sites months later than straw-necked ibis and white ibis. Tracking showed that spoonbills forage for extended periods in and around the nesting site – particularly juveniles who have just left the nest. In contrast, straw-necked ibis and Australian white ibis typically left the nest site not long after starting to fly. Satellite-tagged adults left within days after tagging unless they were actively raising a clutch.

- In 2016-2017, tracked adult straw-necked ibis departed the Macquarie Marshes between 29/10/2016 and 16/11/2016.
- In 2016-2017, tracked juvenile straw-necked ibis departed Barmah-Millewa forest between 20/1/2017 and 20/2/2017.
- In 2017-2018, tracked juvenile royal spoonbills departed Barmah-Millewa forest between 27/3/2018 and 12/4/2018, with three departing on the 2/4/2018.
- Tracked juvenile straw-necked ibis departed Kerang wetlands between 10/12/2017 and 24/12/2017. The only juvenile royal spoonbill to leave Kerang wetlands left on 26/2/2018.
- Tracked juvenile straw-necked ibis departed Kow Swamp between 3/12/2017 and 23/1/2018. The juvenile Australian white ibis departed between 10/1/2018 and 15/2/2018.

Direction of initial flight

Straw-necked ibis adults and juveniles tracked from Barmah-Millewa, Kow Swamp and Kerang flew mostly north-east or east on leaving their nest sites, while the five Macquarie Marshes straw-necked ibis adults all flew south. Post-breeding, the adult male royal spoonbill flew north directly to Queensland. One juvenile royal spoonbill flew NE, another flew S, and other flew ESE. Juvenile royal spoonbills from Barmah-Millewa did not travel long distances on leaving the nest site, however one of the juvenile spoonbills from Kerang dispersed north overnight to Lake Cargelligo and then continued north past the Macquarie Marshes, where transmissions ceased. Australian white ibis juveniles did not travel far from the nesting site

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and flew in a range of directions. None of the tracked birds flew SW and very few flew west (Figure) – those that did didn't travel far and were simply foraging near the nesting sites.

Most juvenile straw-necked ibis initially flew north-east after leaving the nest site (Figure). Some flew briefly ESE before turning NE. One juvenile royal spoonbill flew NE, another flew S, and other flew ESE. Australian white ibis juveniles did not travel far from the nesting site and flew in a range of directions. None of the tracked birds flew SW and very few flew west – those that did didn't travel far and were simply foraging near the nesting sites.

There were distinct temporal patterns in departure times and dates for longer trips. Straw-necked ibis typically departed on long-distance journeys at approximately 10am and travelled throughout the day, stopping at approximately 4-5pm. In contrast, royal spoonbills typically departed at approximately 6pm, travelling through the night. Weather changes appear to be associated with long-distance movements, including temperature drops, rainfall, and shifts in wind direction, however these have not yet been quantified statistically.

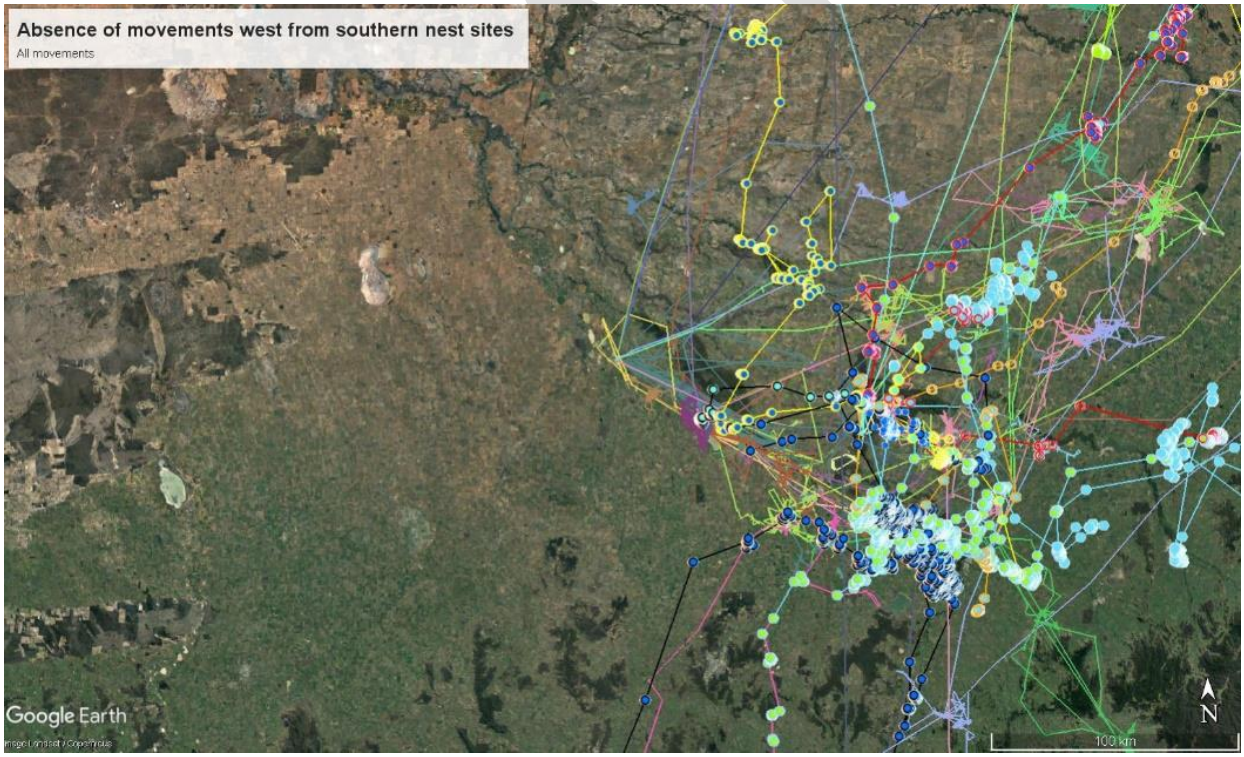


Figure 2 Movements from and around the southern nest sites. Colours indicate individual birds.

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Long-distance movements vs limited movements

Tracking bird movements revealed distinct variation in movement behaviour between and within species.

Within the 40 individual straw-necked ibis tracked, there were at least three movement patterns. Some individuals remained relatively resident (or sedentary) in one area, while others moved nomadically across a region or sub-catchment, and others moved thousands of kilometres up and down the Murray-Darling Basin - with one behaving in the same way an 'obligate' migrant might, moving seasonally at the same times to the same places each year. This mixed movement behaviour within species or populations is known as 'partial migration'. Partial migration is common among birds and is thought to be a strategy that copes at the species level with spatial and temporal variation in resources, environmental conditions and their predictability. Within partially migratory species, the movement behaviour of an individual may remain rigid or fixed over its life, or may change or be flexible.

Mechanisms that have been suggested as driving these differences within partially migratory species include genetics and associated endocrine differences, sex, body size, environmental conditions and social dominance – however the relative roles of these are still poorly understood for most. In partially migratory populations, it is often more common for females, juveniles and birds of smaller body size to migrate or to leave a site early. The tracking of straw-necked ibis in this study is not yet sufficient to quantify this, but there are some possible indications suggesting that a sex bias may exist – for example, all of the tracked straw-necked ibis that flew north to Queensland were female.

In general, birds were most mobile after leaving the nesting site and during autumn and spring. No long-distance movements were observed during winter. All tracked individuals of all species, ages and sexes 'settled down' from autumn to winter, using highly localised over-wintering areas often of only a few square kilometres or less. The locations and site characteristics of these over-wintering sites varied. Some were in the northern Murray-Darling Basin; others were in the south. One adult female straw-necked ibis was tracked seasonally migrating from Victoria to Queensland, back to Victoria, and then returning to exactly the same Queensland over-wintering site, showing site fidelity.

In general, the tracked Australian white ibis were far more 'resident' than the straw-necked ibis and royal spoonbills post-dispersal from the nest site. Most remained in the region of the

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nesting site, roosting and foraging around irrigated dairy farm buildings or irrigated cropping. None of the tracked white ibis moved to the east coast. One juvenile moved to Melbourne, spending approximately at year foraging in the Werribee area at the rubbish tip, golf courses and sports grounds. In early 2019 he then began a circuit of Port Phillip Bay, with short residences at rubbish tips, golf courses and sports grounds along the way.

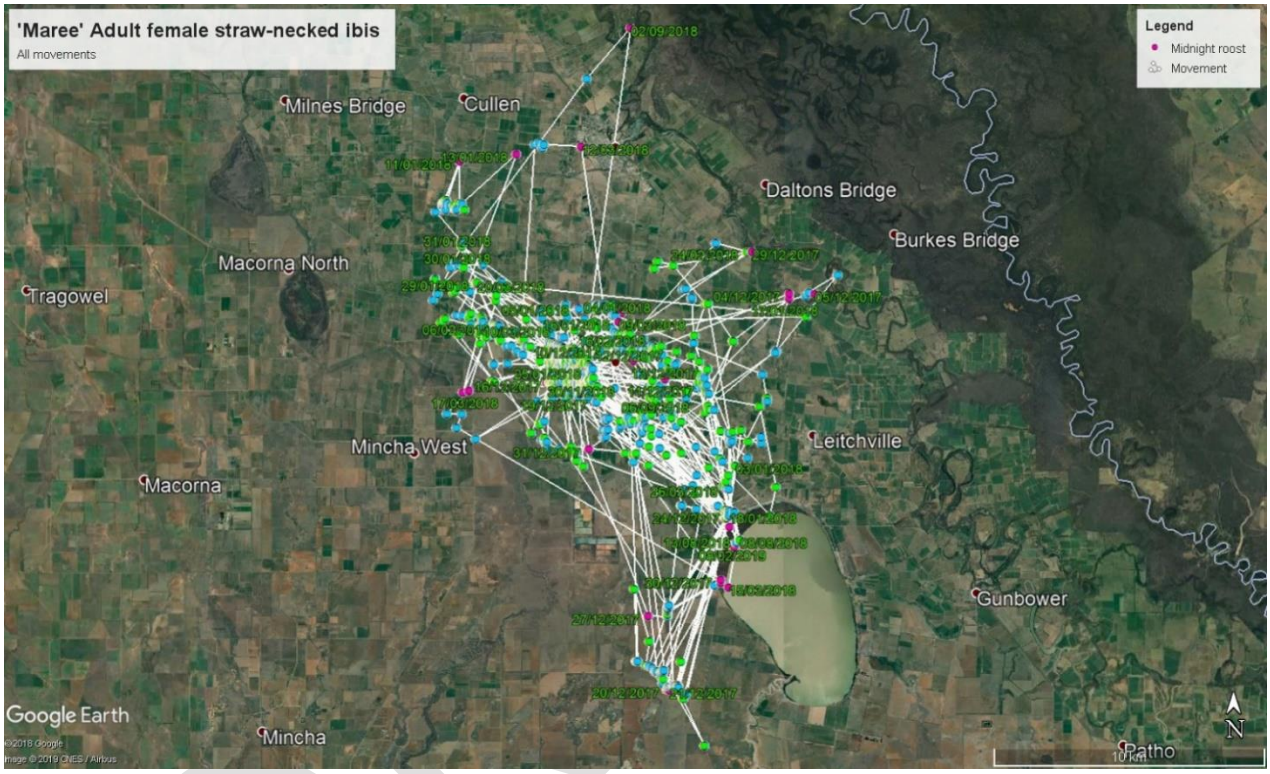
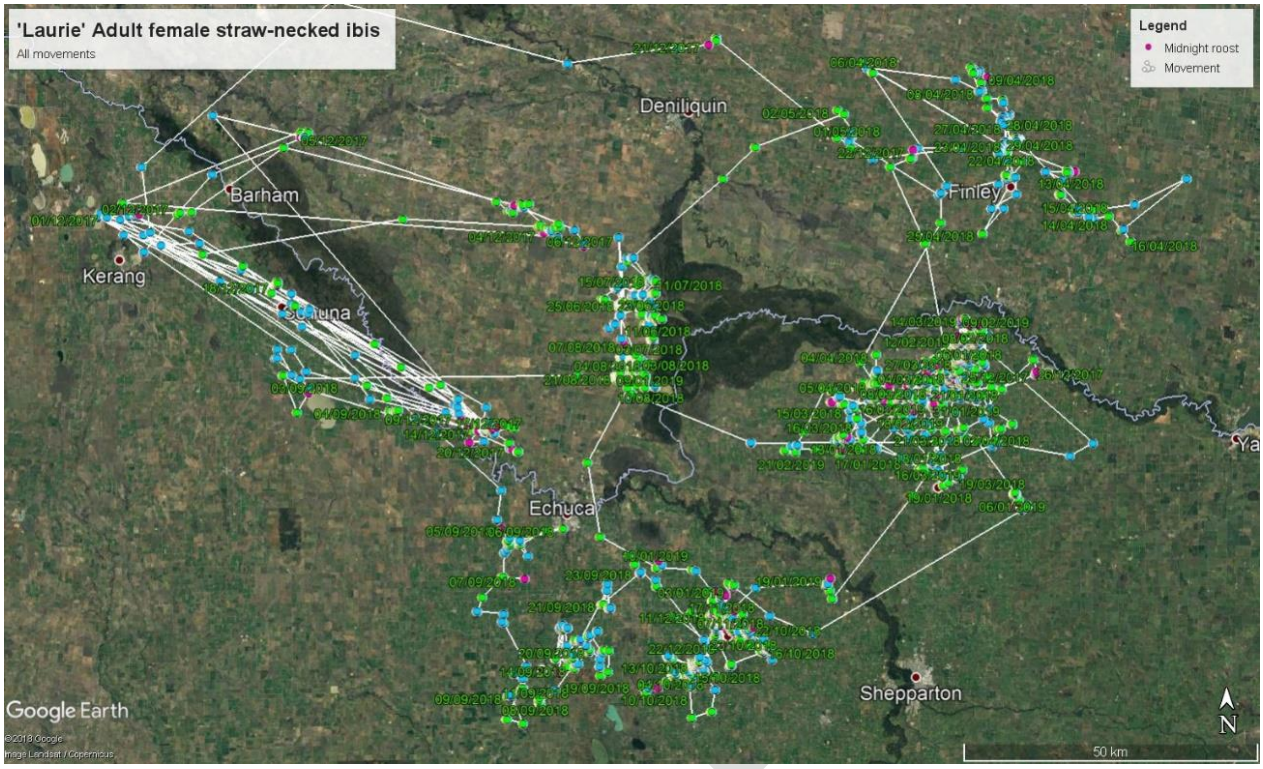


Figure 3 Resident or sedentary movements of an adult female straw-necked ibis in northern Victoria.

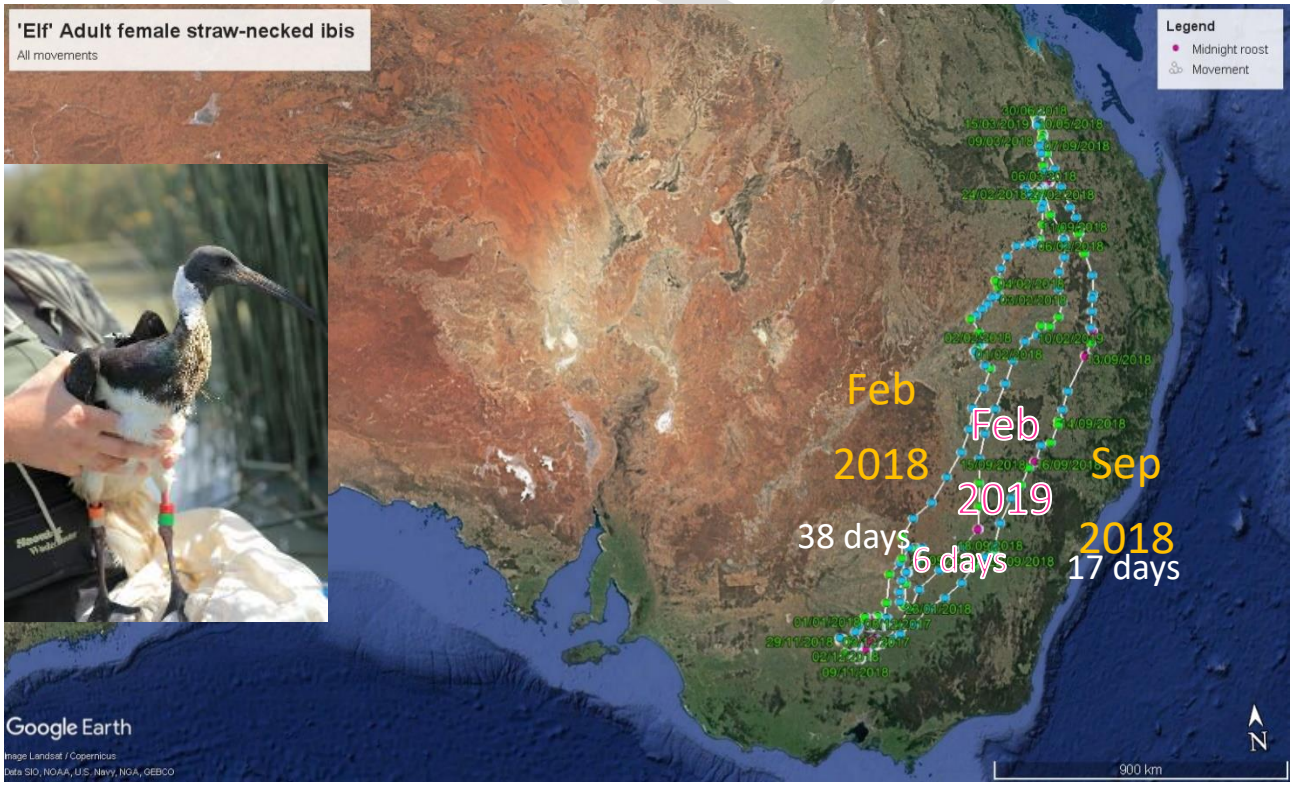
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340 **Figure 4 Nomadic movements of an adult female straw-necked ibis in southern NSW and northern VIC**

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343 **Figure 5 Migratory movements of an adult female straw-necked ibis between Victoria and Queensland.**

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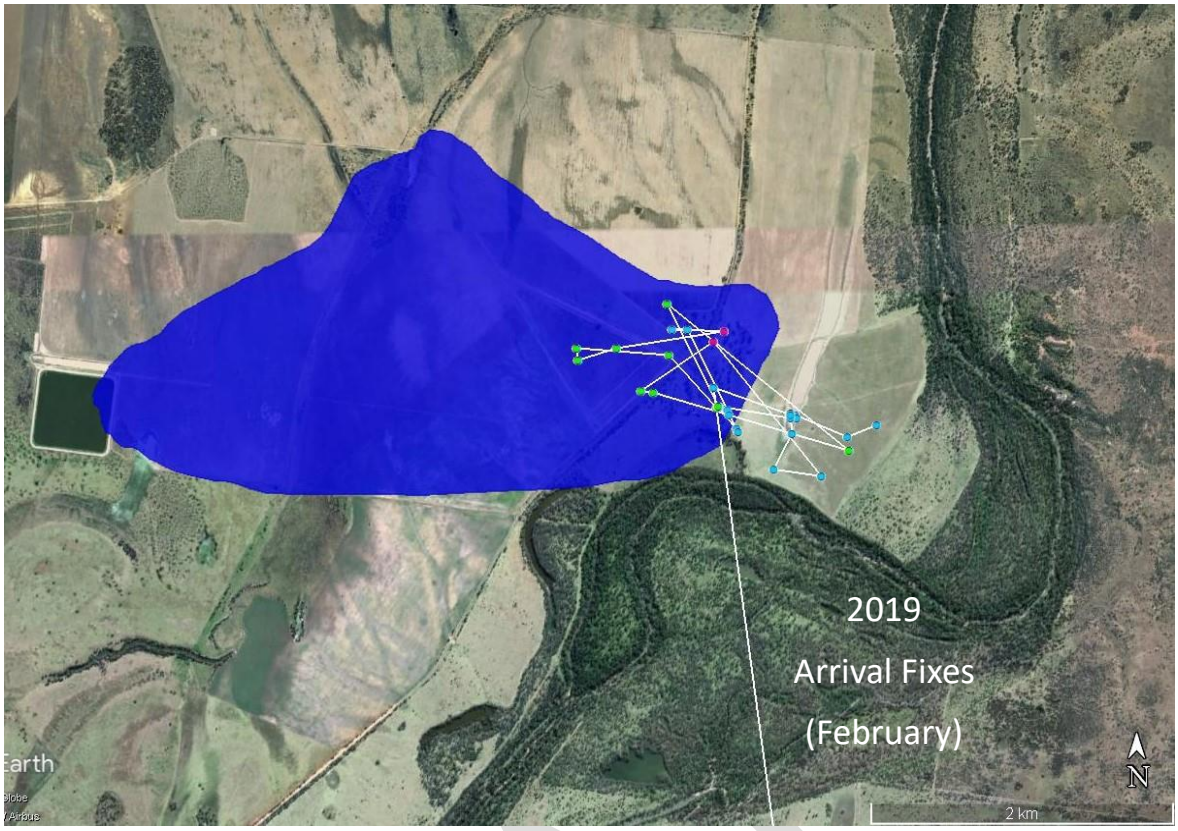


Figure 6 Site fidelity shown by an adult female straw-necked ibis. Blue shading indicates 2018 over-wintering area; white lines and dots indicate movements on arrival at the same site in February 2019.

Common long-distance routes for different individuals and species

A common long-distance NE-SW route was used in both years by straw-necked ibis juveniles and adults and by juvenile and adult royal spoonbills (Figure). This route was used in both directions, sometimes by the same birds, from both northern basin and southern basin sites. The route avoids dry, high elevation, or forested areas and corresponds with climatic / bioregional zones (Figure). It follows zones or zonal boundaries that are temperate to sub-tropical, with no distinct 'dry season', high soil moisture, and low elevation. The route generally follows areas with low to moderate rainfall variability that have a mean annual rainfall of ~600-700mm (ranging between 400-1000mm). Approximately 25% of tracked birds have used this common route to-date; 15% of tracked birds used it flying to Queensland.

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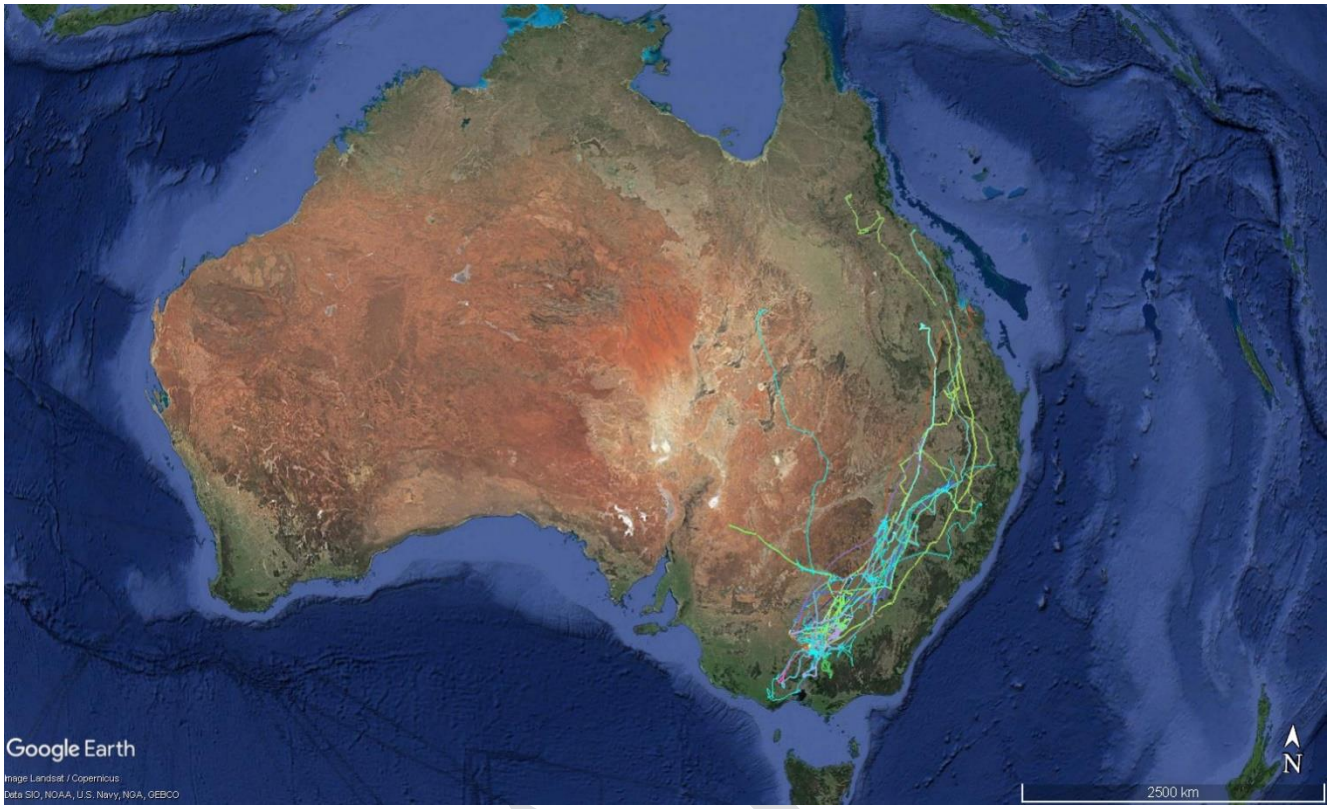


Figure 7 Tracked bird movements across eastern Australia; each coloured line is an individual bird.

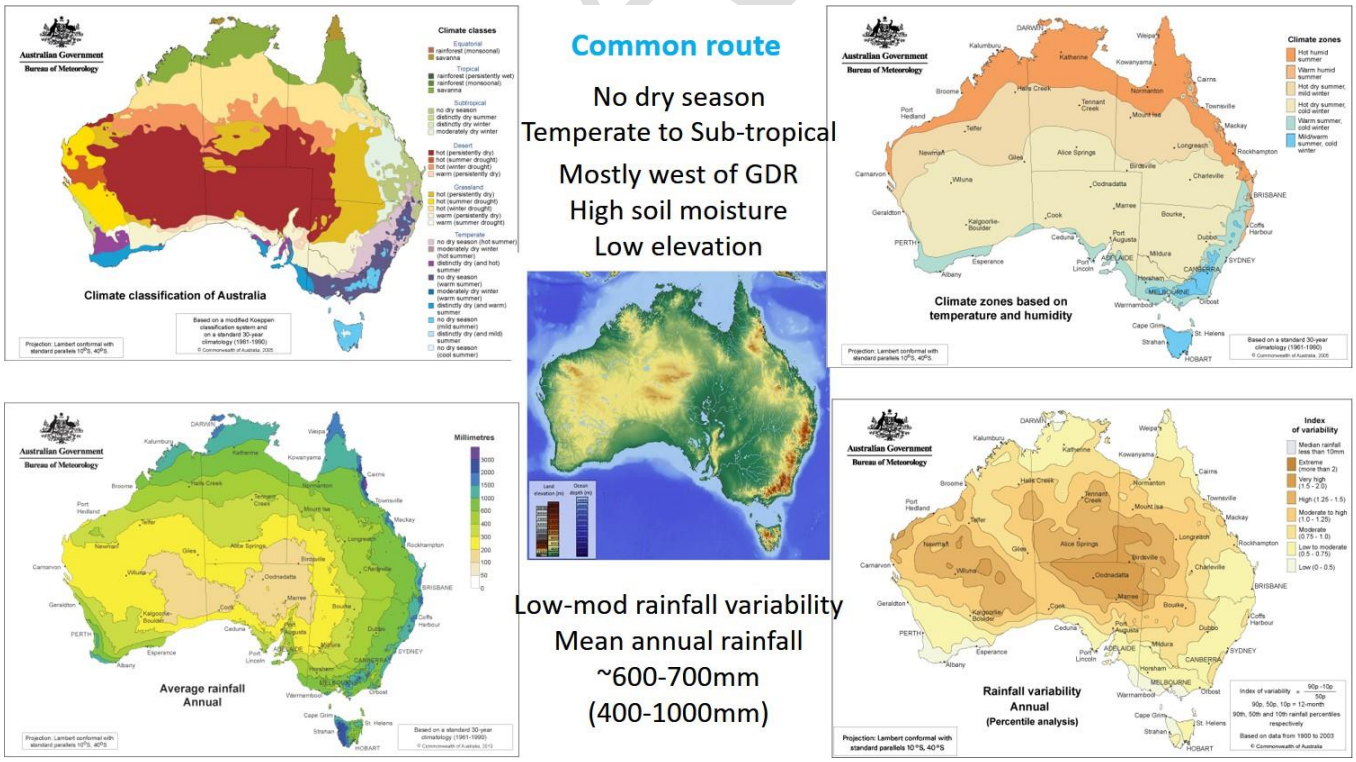


Figure 8 Climatic and bioregional zones of Australia with boundaries that correspond to the common route being used by tracked straw-necked ibis. Images sourced from the Australian Bureau of Meteorology.

Key foraging and stopover areas

Tracking revealed re-use of some sites by different birds at different times. For example, in the wet year of 2016-2017, there was extensive flooding around the mid-Lachlan River near Condobolin, NSW, and several tracked birds from the Macquarie Marshes and from Barmah-Millewa Forest spent time foraging in the area. In contrast, during the dry year of 2017-2018, foraging was more intensive in the dairy farming country of northern Victoria, where pasture irrigation was taking place. However the mapped density of use of an area is biased by capture locations and may not reflect whole of population patterns at this stage. For example, with most birds captured in the mid-Murray area, most location fixes and movements were concentrated around this area and may not reflect habitat use by the whole population across the Basin. Further tracking of individuals captured at different sites across the Basin will clarify key foraging and stopover areas.

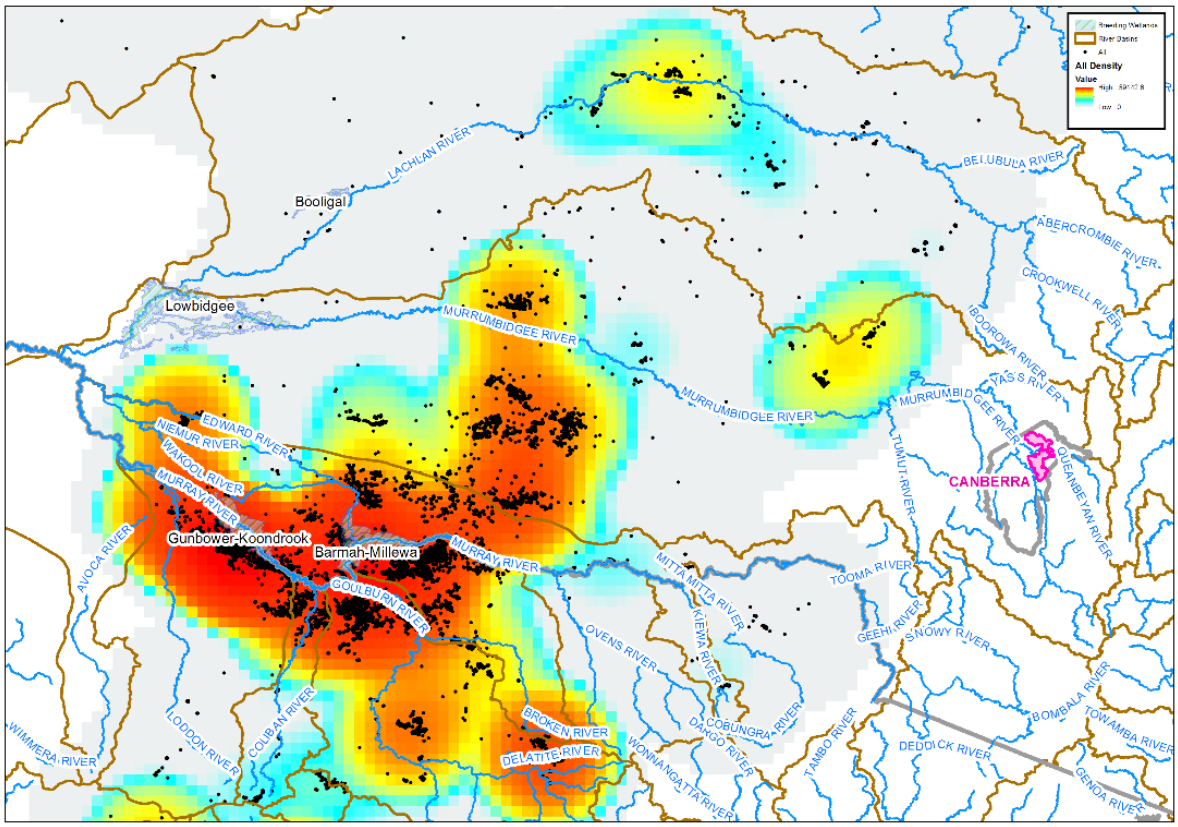
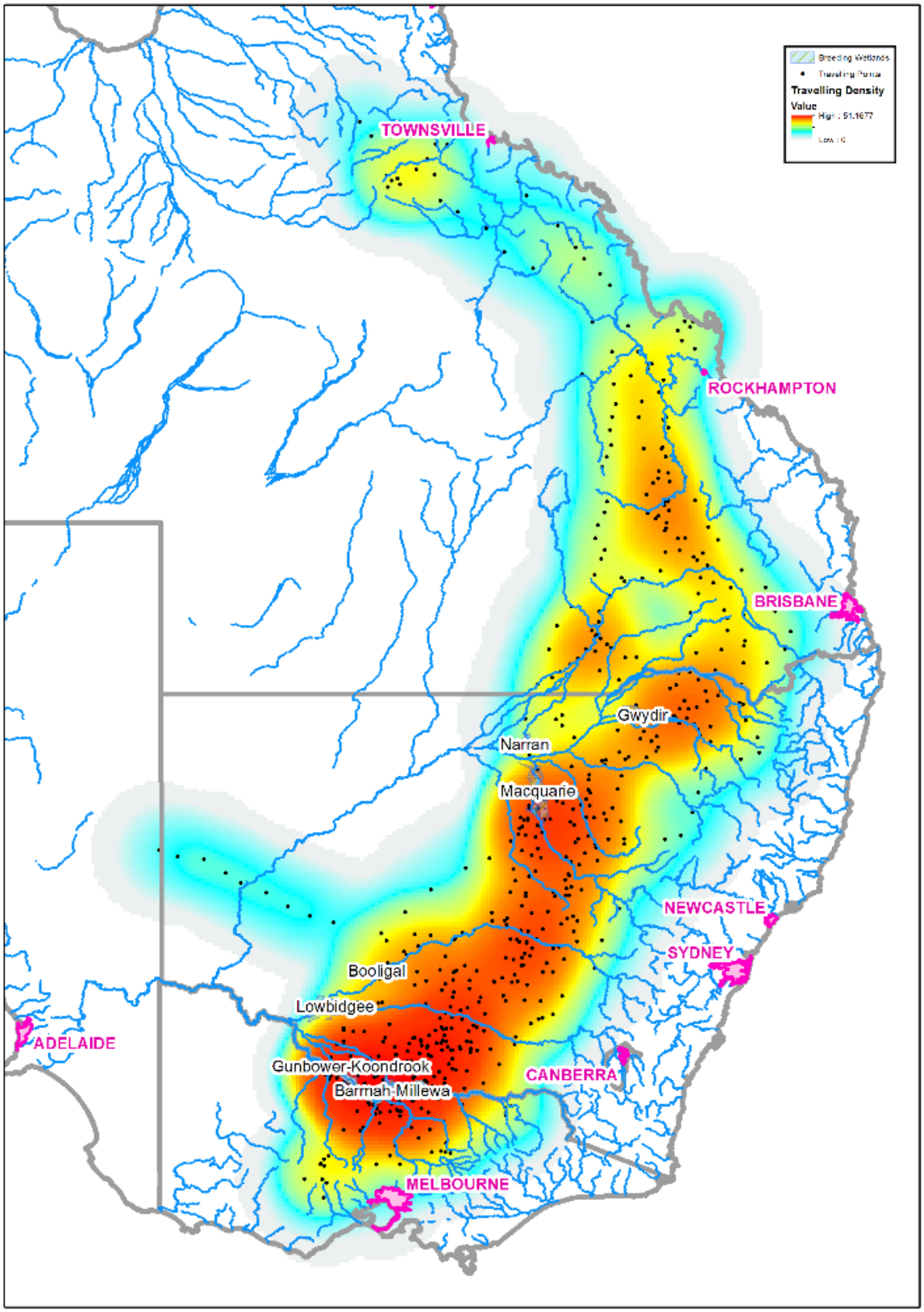


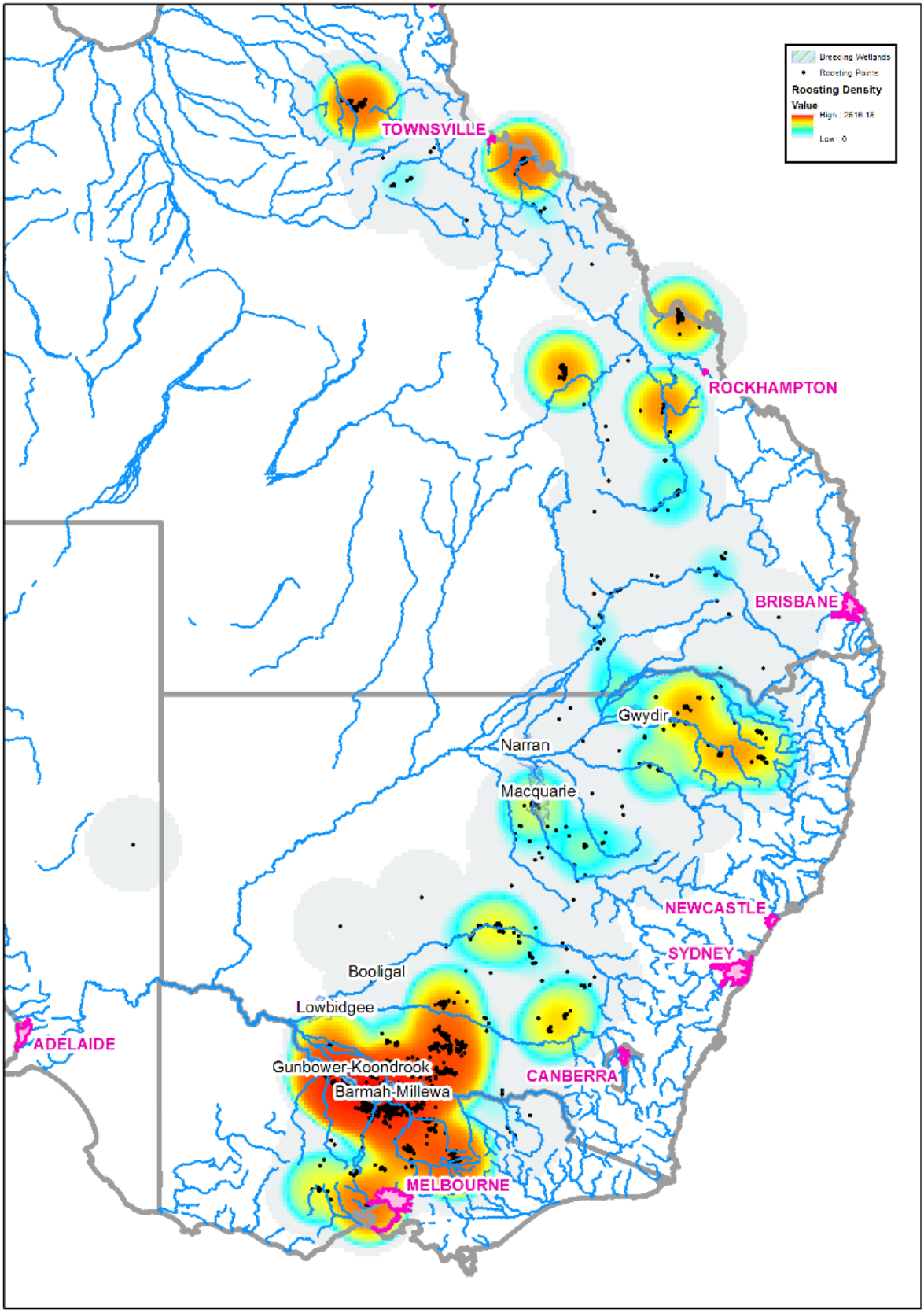
Figure 9 Density of location fixes around the Murray, Murrumbidgee and Lachlan rivers



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387 **Figure 10 Density of location fixes for tracked birds travelling in eastern Australia**

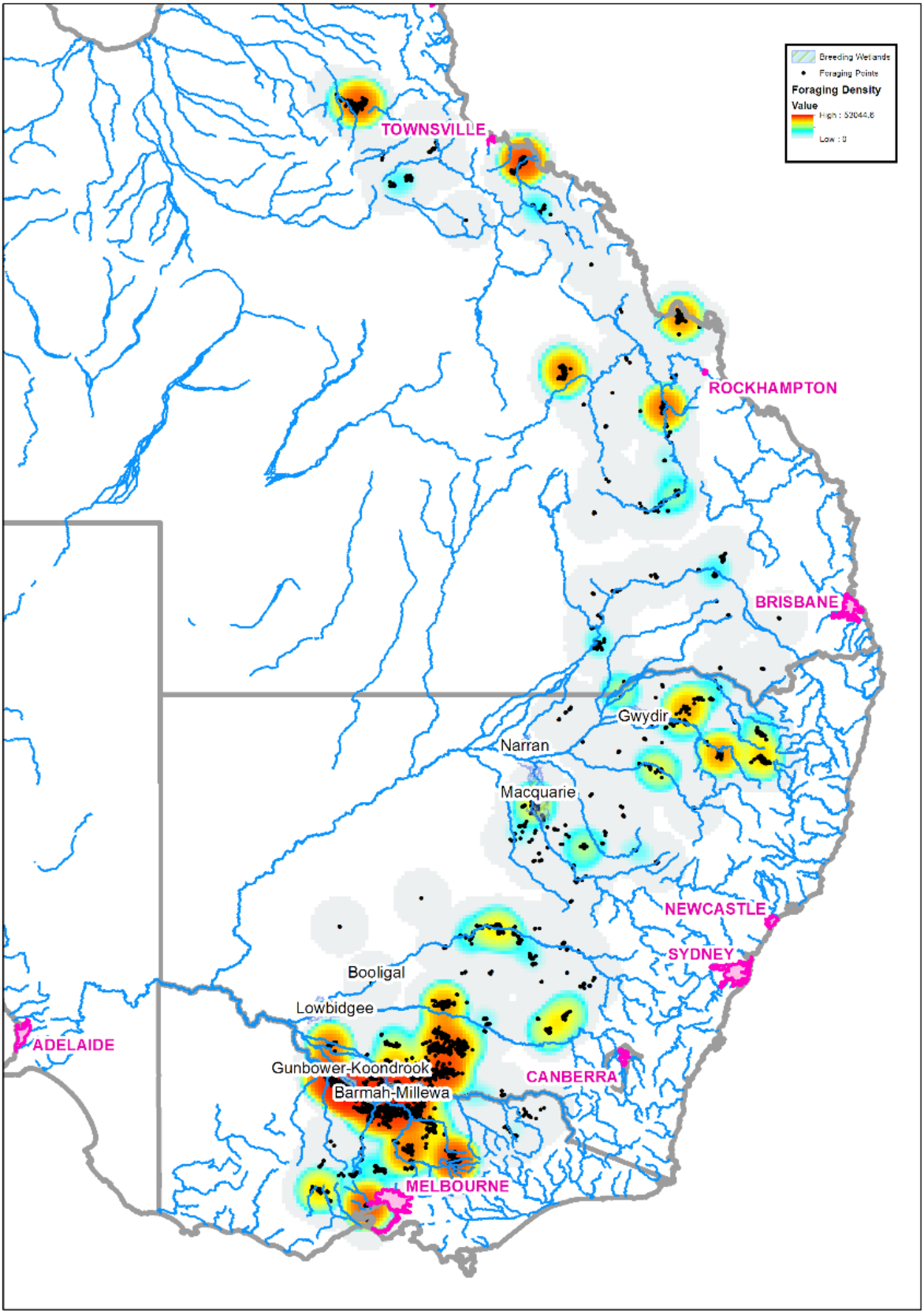
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390 **Figure 11 Density of roosting location fixes for tracked birds roosting in eastern Australia**

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393 **Figure 12 Density of foraging location fixes for tracked birds foraging in eastern Australia**

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Foraging movements

Examination of bird movement tracks and fixes together with satellite imagery shows a distinct 'pairing' between roosting habitats and foraging habitats.

Straw-necked ibis usually actively forage during the day and sleep during the night, coming in to roost at around sunset. The most commonly used foraging habitats have adjacent remnant vegetation, with trees for roosting that are next to or overhang water. Straw-necked ibis use these trees mostly for sleeping at night but also for taking shelter from the weather during the day. When resident in an area, adult straw-necked ibis most commonly foraged within approximately 2 km of their roosting trees (median 2.21 km); while juvenile straw-necks foraged slightly closer to their roosts (median 1.34 km).

Royal spoonbill habits were less predictable than those of straw-necked ibis. They often actively foraged at night and during the morning, and slept during the afternoon. Royal spoonbills prefer tall roosting trees next to or overhanging water; however will also rest in dense wetland vegetation such as rushes or on wetland banks. Tracking data for royal spoonbills is limited to one adult and is biased by juvenile movements being mostly within the nesting site and the fact that night fixes were not received between midnight and 7am or between 7pm and midnight. The most common maximum distance travelled by the adult from wherever he was at midnight each day was 2.66 km; while the median distance travelled by juveniles was 259 m.

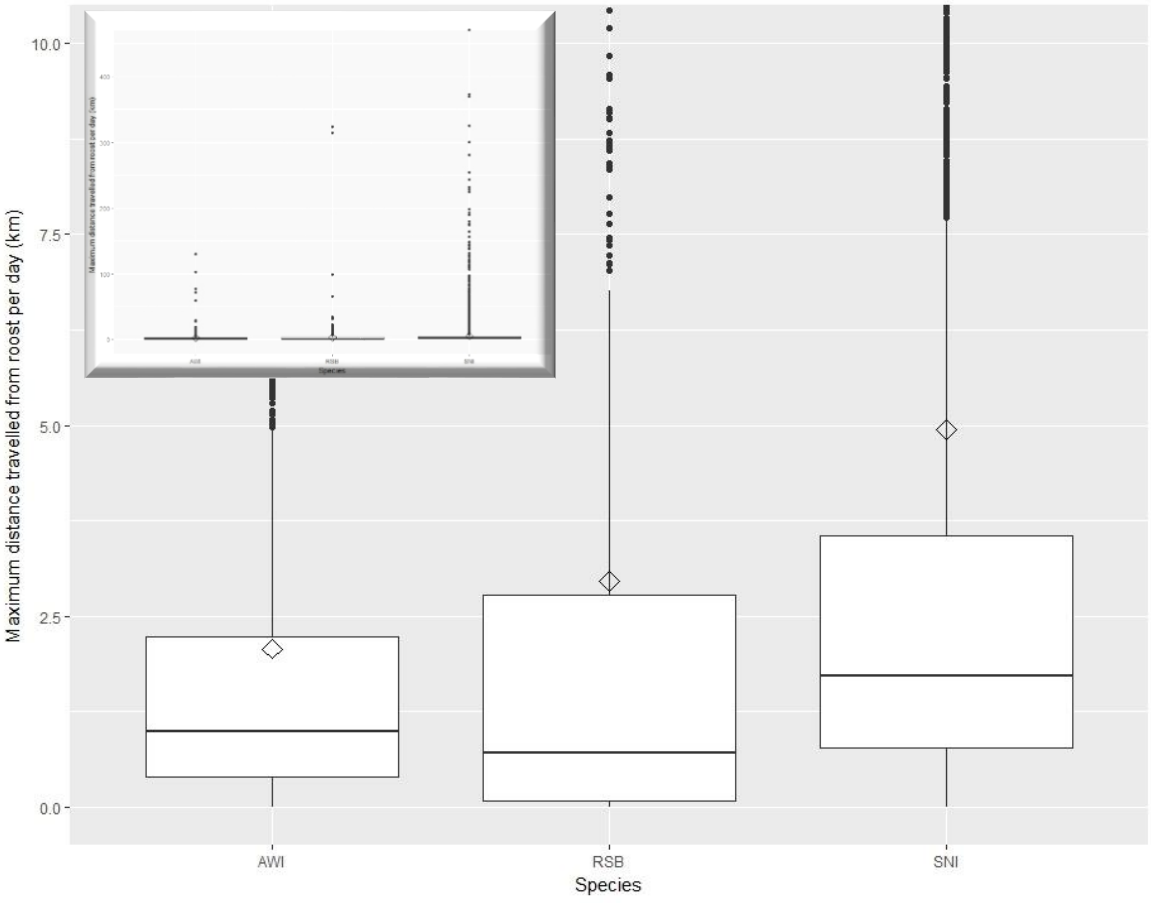
In contrast, Australian white ibis frequently roosted away from water and in or on farm buildings and did not travel as far to forage. All of the tracked white ibis were juveniles, with a median foraging distance from their roosts of 1 km.

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418 **Table 5 Maximum distance ranged per day summary statistics, all raw data, by roost type, species and age**

RoostType	Sp	Age	n	mean	median	sd
Reside	SNI	Juvenile	3685	3.82	1.34	16.6
Reside	SNI	Adult	3217	6.22	2.21	18.6
Reside	RSB	Juvenile	708	2.07	0.259	12.8
Reside	RSB	Adult	241	5.57	2.66	21.7
Reside	AWI	Juvenile	1650	2.07	1	5.45
Explore	SNI	Juvenile	69	12.5	12.1	5.19
Explore	SNI	Adult	116	11.9	10.5	5.78
Explore	RSB	Juvenile	4	9.96	10.1	2.24
Explore	RSB	Adult	9	12.3	13	7.26
Explore	AWI	Juvenile	8	12.6	11.7	5.57
Layover	SNI	Juvenile	53	126	83.2	113
Layover	SNI	Adult	94	121	74.7	118
Layover	RSB	Adult	4	171	198	136
Layover	AWI	Juvenile	4	73.3	66.9	44.3

419
420



421

422 **Figure 13 The maximum distance travelled from the midnight fix per day, by species, with the y-axis**
423 **restricted to 10 km in order to better show the distribution of the majority of the data and the inset graph**
424 **showing the full dataset. Solid horizontal lines within boxes indicate the median; diamonds indicate the**
425 **mean. AWI = Australian white ibis; RSB = Royal spoonbill; SNI = Straw-necked ibis.**

426

Nesting vs not nesting

Five adult individuals either continued nesting or re-nested while being tracked: four straw-necked ibis (three females and one male) and one royal spoonbill (male).

For straw-necked ibis, when nesting, the mean maximum distance travelled from the nest to forage per day was generally longer than the distance travelled from the roost per day when not nesting and resident, but not statistically significant. For the royal spoonbill adult male, the reverse was true. Foraging trips generally became longer over time as the nesting season progressed. Foraging distances during nesting at Macquarie Marshes in the northern Basin were longer than those at Barmah-Millewa Forest or Kow Swamp in the southern basin. The straw-necked ibis tracked during nesting at Kerang displayed surprising and unusual long-distance movements that were very different to those of other tracked birds and have yet to be explained.

The tracking data for juveniles at nesting sites displayed a distinct pattern of roosting at night in one part of the nesting colony, but spending the day in another part of the colony up to 100m away. It is likely that the former is the nest site, while the latter is a crèche site.

Table 6 Individual birds that were tracked during nesting and summary statistics describing the maximum distance travelled from the nest to forage per day.

Species	Sex	Nesting dates	Location	Nesting?	Mean	Median	SD	SE	N
SNI	M	26/10/2016 – 16/11/2016	Macquarie Marshes	Nesting	18.8	8.42	19.6	4.7	17
				Not nesting	4.67	1.51	16.3	0.7	535
SNI	F	15/12/2016 – 30/1/2017	Barmah-Millewa Forest (Boals Deadwoods)	Nesting	5.67	6.03	4.49	0.7	44
				Not nesting	2.0	0.7	3.5	0.7	24
SNI	F	3/12/2017 – 25/12/2017	Kow Swamp	Nesting	7.02	6.79	2.34	0.5	21
				Not nesting	3.69	3	3.04	0.3	91
SNI	F	1/12/2017 – 21/12/2017	Kerang	Nesting	44.3	56.7	29.4	6.8	19
				Not nesting	5.32	3.41	7.6	0.4	406
RSB	M	27/2/2018 – 7/4/2018	Barmah-Millewa Forest (Reed Beds)	Nesting	4.1	2.5	3.87	0.6	36
				Not nesting	5.83	2.66	23.5	1.6	204

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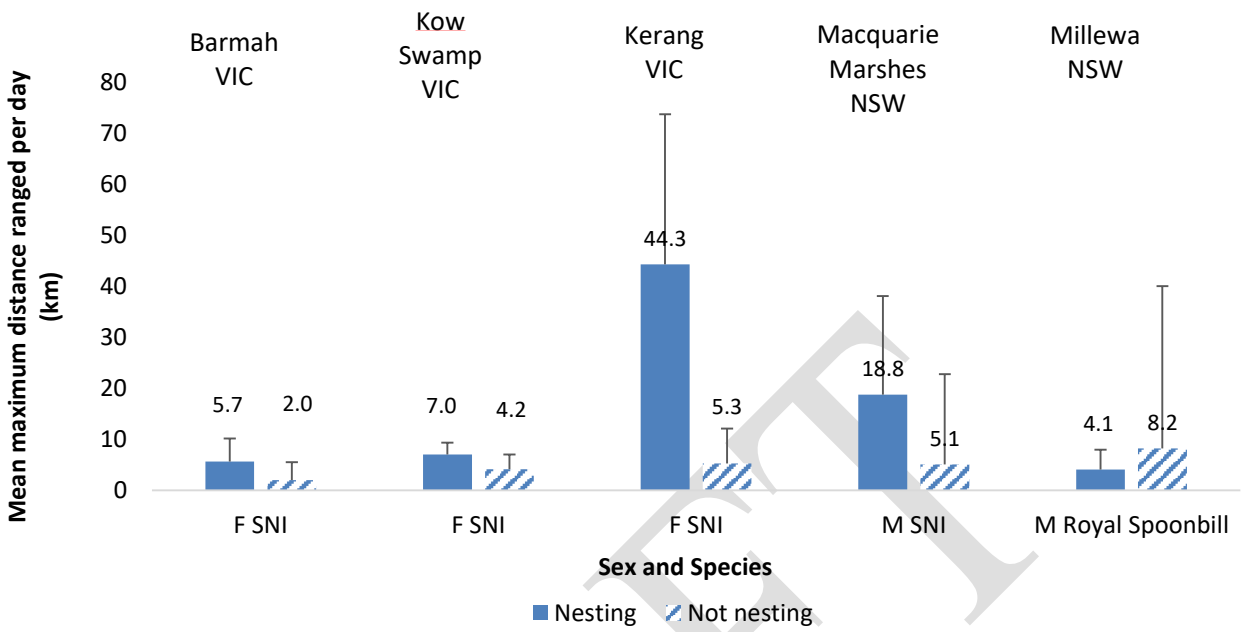


Figure 14 Mean (+/- 1SD) maximum distance ranged from the midnight fix per day for each of the five individuals tracked during nesting and not during nesting. Solid bars = nesting; hatched bars = not nesting but resident.

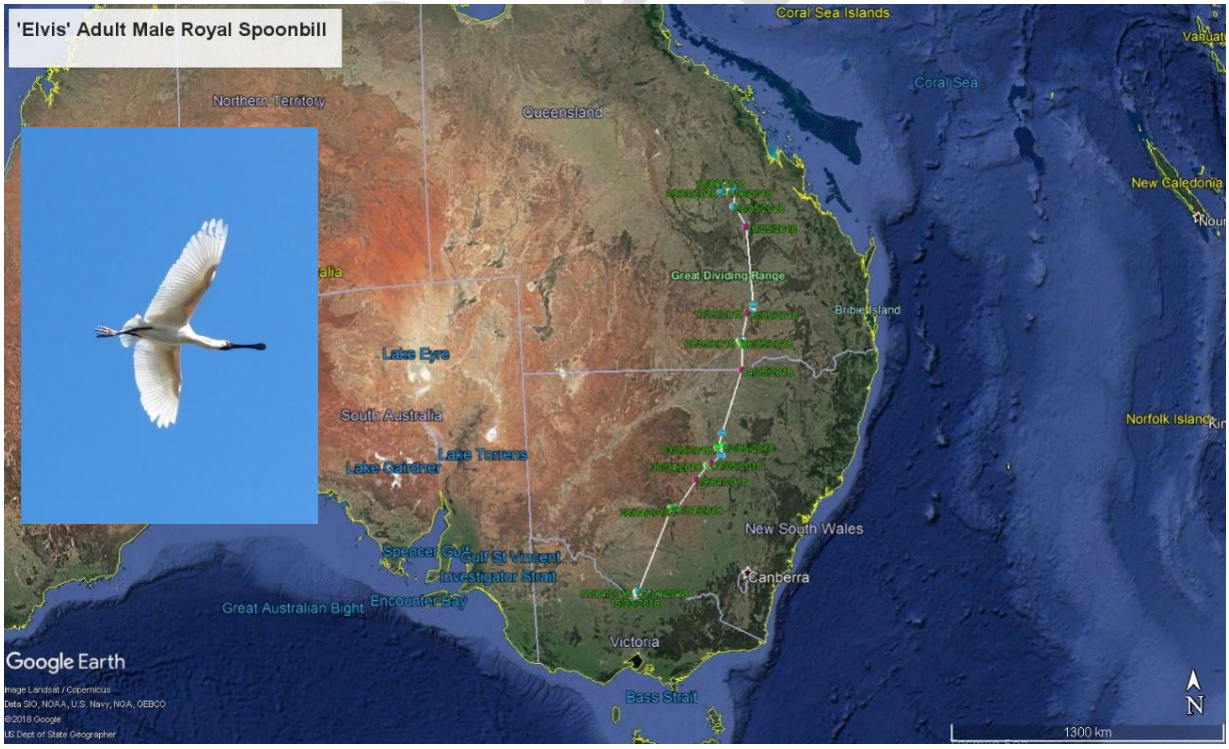
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455 **Figure 15 Foraging movements of an adult male royal spoonbill during nesting in Barmah-Millewa Forest.**

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457

458 **Figure 16 Movement of an adult male royal spoonbill from southern Australia to northern Australia post-**
459 **nesting.**

460

Habitat characteristics

Habitat characteristics of foraging sites used by satellite-tracked ibis and spoonbills generally reflect a combination of habitat availability and dietary preferences.

Most of the available spatial habitat mapping data in Australia is far less accurate in resolution than the tracking data; consequently analysing tracking data using existing mapping information provides a broad indication of habitat characteristics rather than a specific classification. This is particularly relevant for roosting sites, which if they are in a small or narrow stand of trees amongst agricultural land may not be classified by digital spatial maps as 'trees' or 'native vegetation'; instead they are usually classified according to the surrounding agricultural land use. However examination of satellite imagery together with the satellite tracking data allows visual assessments.

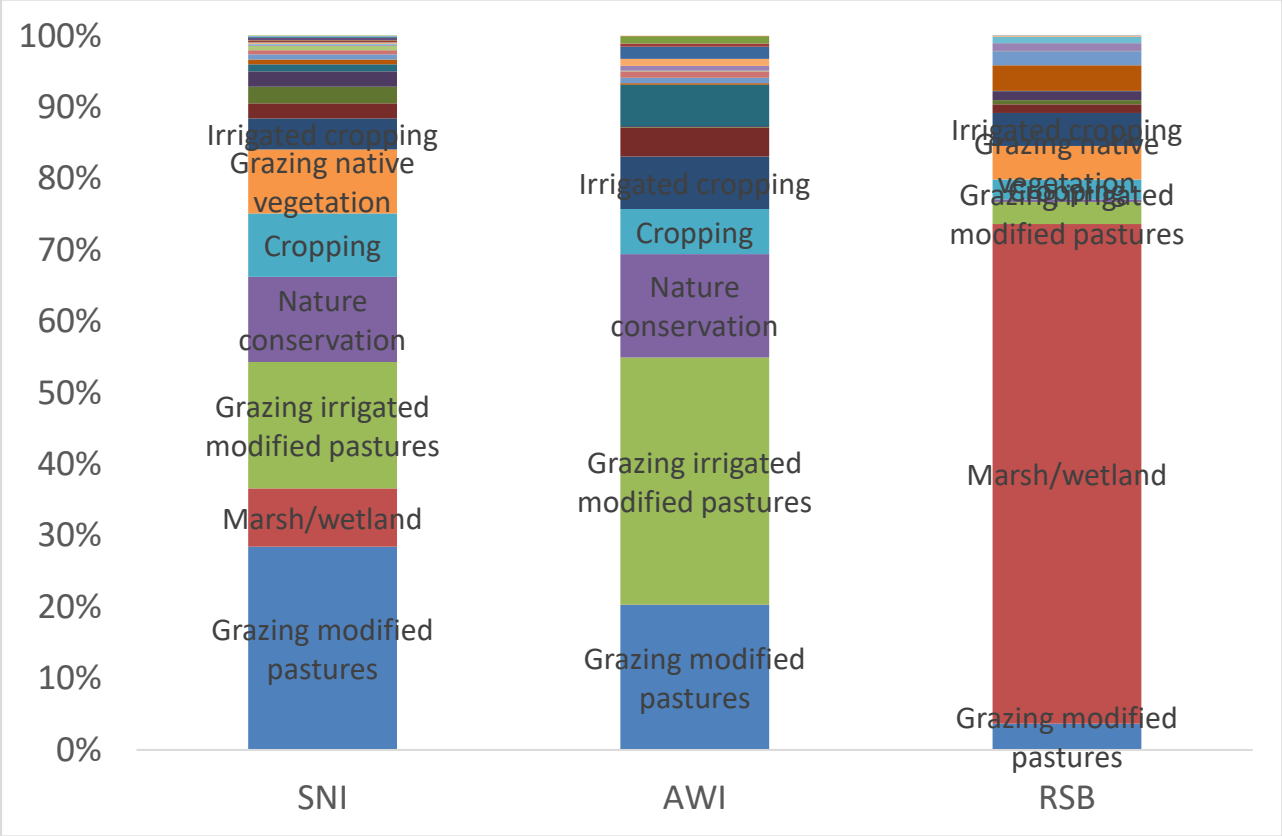


Figure 17 Landuse types (ALUM 2016) used by straw-necked ibis (SNI), Australian white ibis (AWI) and royal spoonbills (RSB) during tracking.

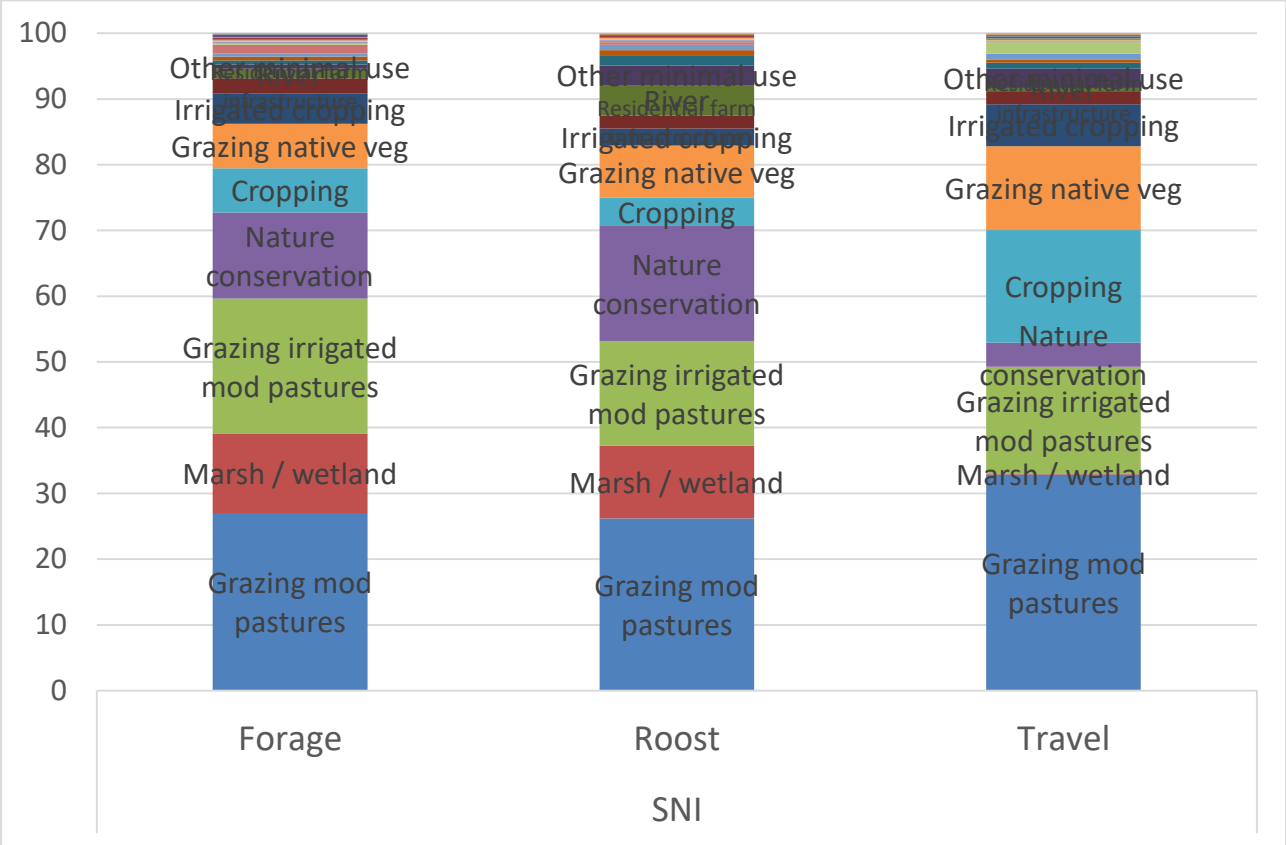


Figure 18 Landuse types (ALUM 2016) used by straw-necked ibis (SNI) when foraging, roosting and travelling.

Nesting vs not nesting

Foraging habitats used when nesting differed substantially from those used when not nesting for all but one of the five birds for which data were available. Landuses when nesting varied by site and by species (note: there was only one bird tracked during nesting from each site). Overall, the tracked royal spoonbill spent significantly more time foraging in marsh/wetland environments when nesting than tracked straw-necked ibis.

An adult male straw-necked ibis nesting in the Macquarie Marshes (Monkeygar Swamp) spent 30% of his time foraging in areas classed as 'marsh/wetland', 25% in 'nature conservation' areas, and 24% in areas classed as 'grazing native vegetation' (Figure 19a). His remaining foraging time when nesting was primarily split between areas classed as 'grazing modified pastures', 'other minimal use' and 'river'. This individual initially foraged relatively close to the nesting site, with increasing distances travelled to forage over time. When not nesting, this individual ranged throughout the mid-Murray-Darling Basin, spending 57% of his foraging time in 'grazing irrigated modified pastures', 22% in 'grazing modified pastures',

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and the remainder in areas classed as ‘residential and farm infrstructure’, ‘irrigated cropping’, ‘transport and communication’ (mostly roadsides), ‘river’, ‘nature conservation’ and ‘manufacturing and industrial’ (Figure 19b).

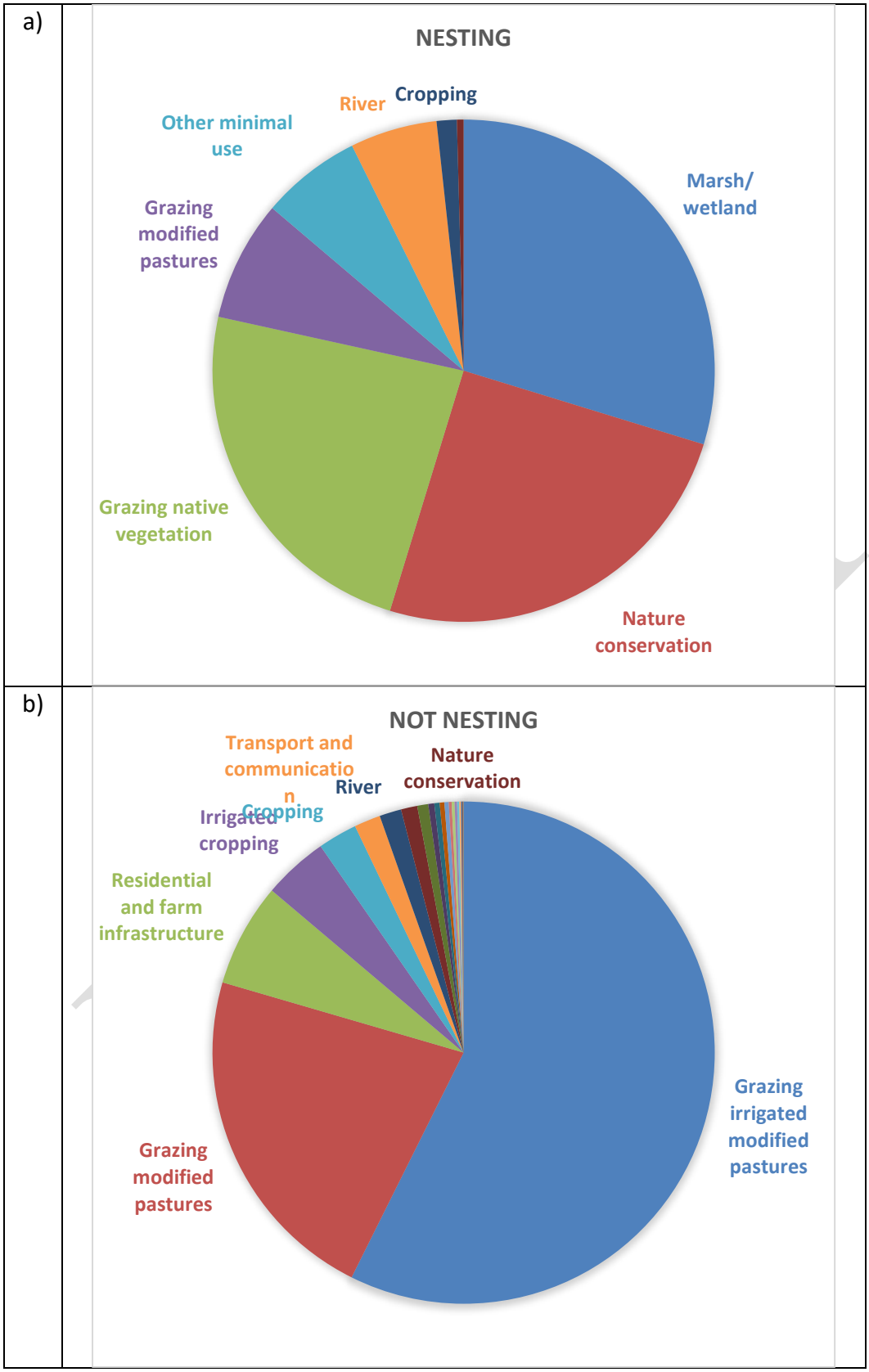
An adult female straw-necked ibis tracked when nesting in Barmah Forest (Boals Deadwoods) spent 67% of her foraging time in nature conservation areas, 20% in cropping and the remainder in grazing modified pastures and transport and communication areas (Figure 20a). When not nesting, she did not move far from Barmah-Millewa Forest and foraged in areas classed as ‘grazing modified pastures’ (48%), nature conservation (27%); production native forests (10%), cropping (8%), grazing irrigated modified pastures (4%) and marsh/wetland (3%) (Figure 20b).

An adult female straw-necked ibis tracked when nesting in Kow Swamp (northern VIC) spent 77% of her foraging time in ‘grazing irrigated modified pastures’ dominated by dairy farms (Figure 21a). She also foraged in irrigated cropping (18%). When not nesting, 94% of foraging locations were located in ‘grazing irrigated modified pastures’ or dairy farms (Figure 21b). This individual was highly resident, relative to other tracked straw-necked ibis. After nesting, she remained within an area of approximately 40 square kilometres and rarely travelled more than 5km from the nesting site.

An adult female straw-necked ibis tracked when nesting in Kerang Wetlands (northern VIC) spent 68% of her foraging time in areas classed as ‘grazing modified pastures’ (Figure 22a) and the remainder in ‘grazing irrigated modified pastures’ (11%), irrigated cropping (9%), nature conservation (5%), cropping (3%), marsh/wetland (2%), residential and farm infrastructure (1%) and transport and communication (mostly roadsides; 1%). When not nesting the proportion of ‘grazing irrigated modified pastures’ (mostly dairy farming) increased to 56% and ‘grazing modified pastures’ decreased to 13% (Figure 22b).

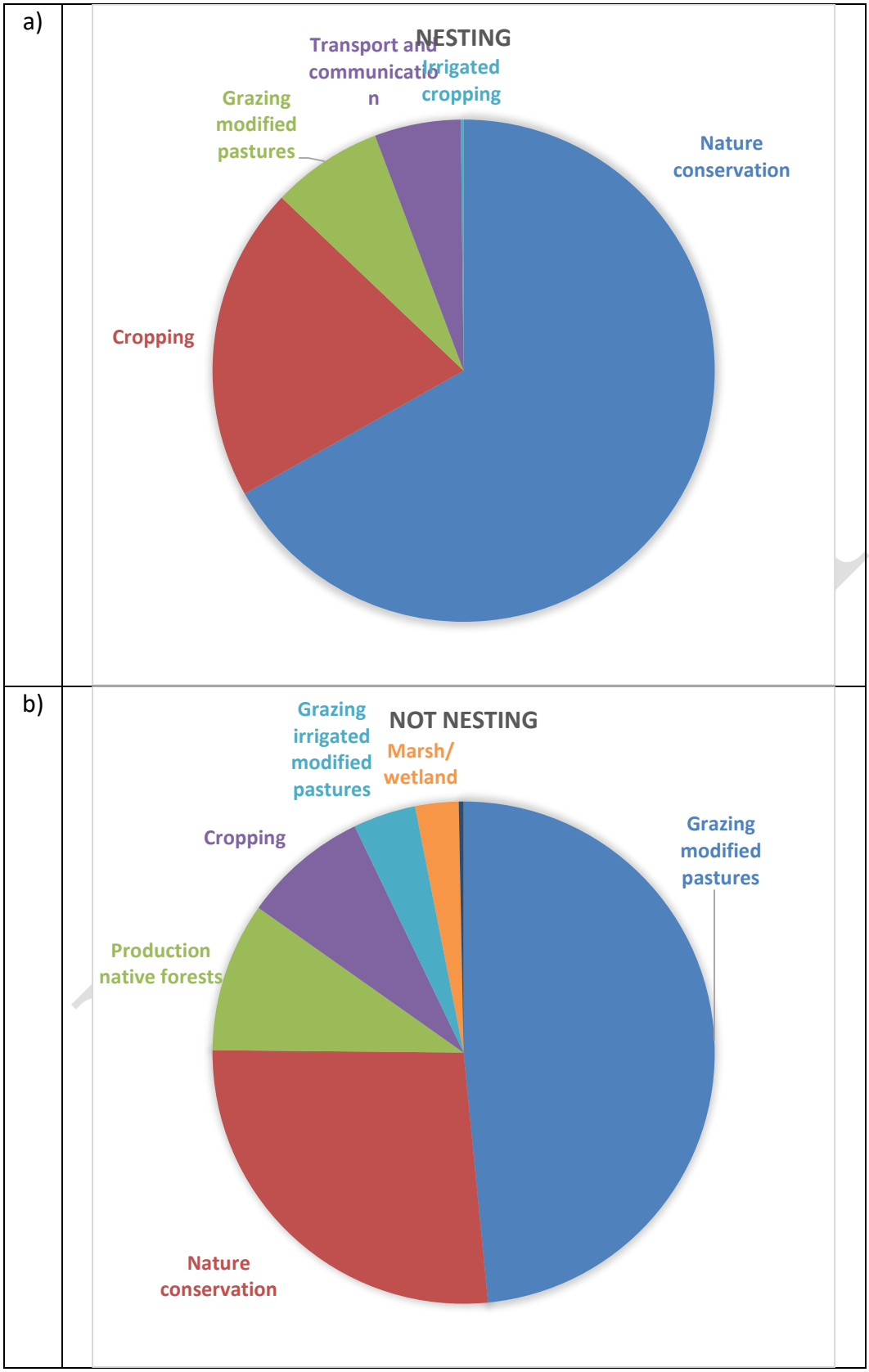
Finally, an adult male royal spoonbill tracked when nesting in Millewa Forest (Reed Beds, NSW) spent 94% of his foraging time in marsh/wetland habitats and the remainder in ‘production native forests’ and ‘other minimal use’ (Figure 23a). When not nesting (in Queensland), he spent 62% of his foraging time in ‘grazing native vegetation’ landuses, 20% in reservoirs/dams, 7% in marsh/wetland and the remainder in other minimal use, irrigated cropping, production native forests, lake, channel/aqueduct and grazing modified pastures (Figure 23b).

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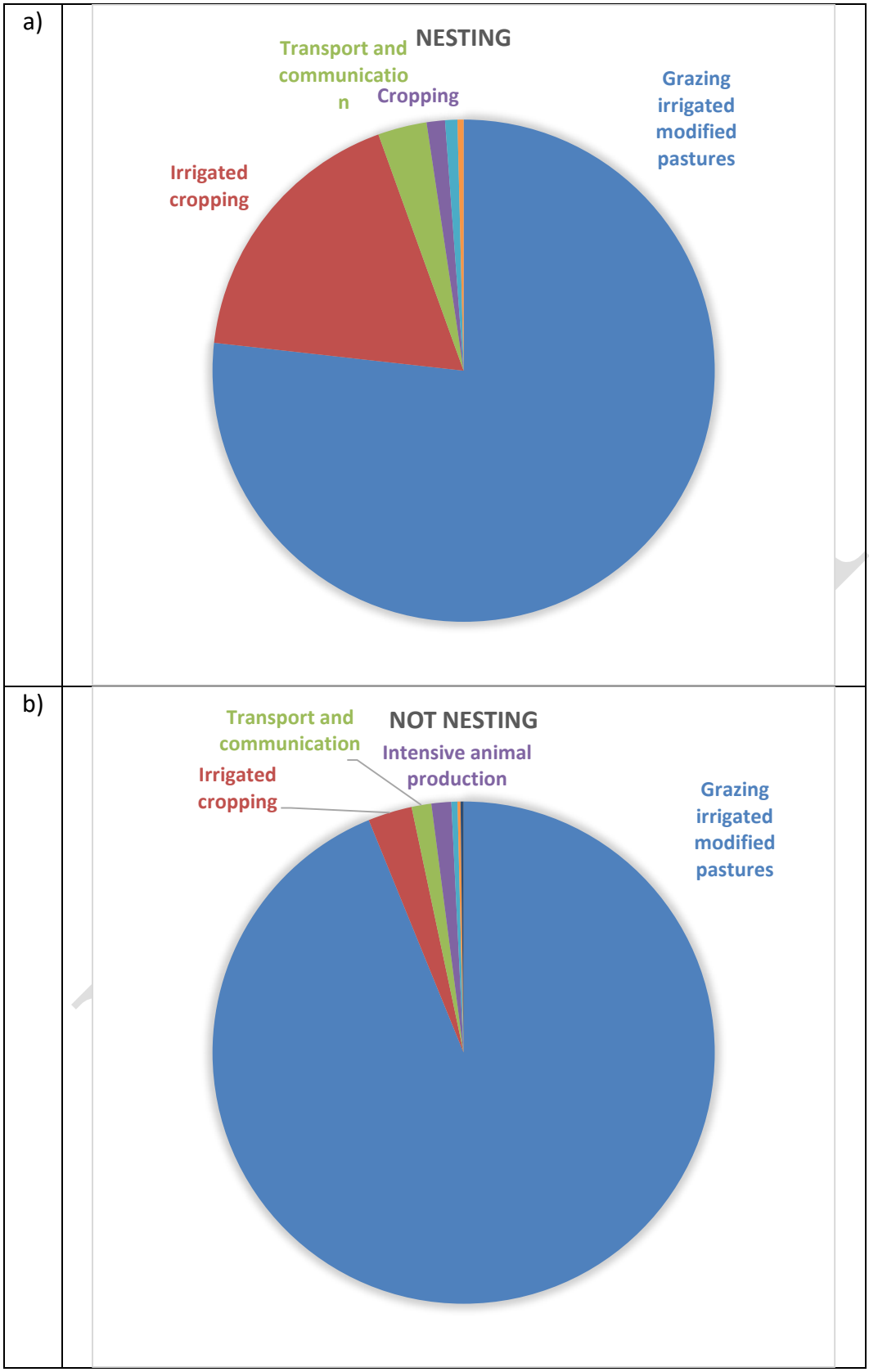
527 **Figure 19 Landuse types (ALUM 2016) used by an adult male straw-necked ibis (SNI) when foraging. a) During**
528 **nesting in the Macquarie Marshes; b) when not nesting (foraging across the Murray-Darling Basin).**

529



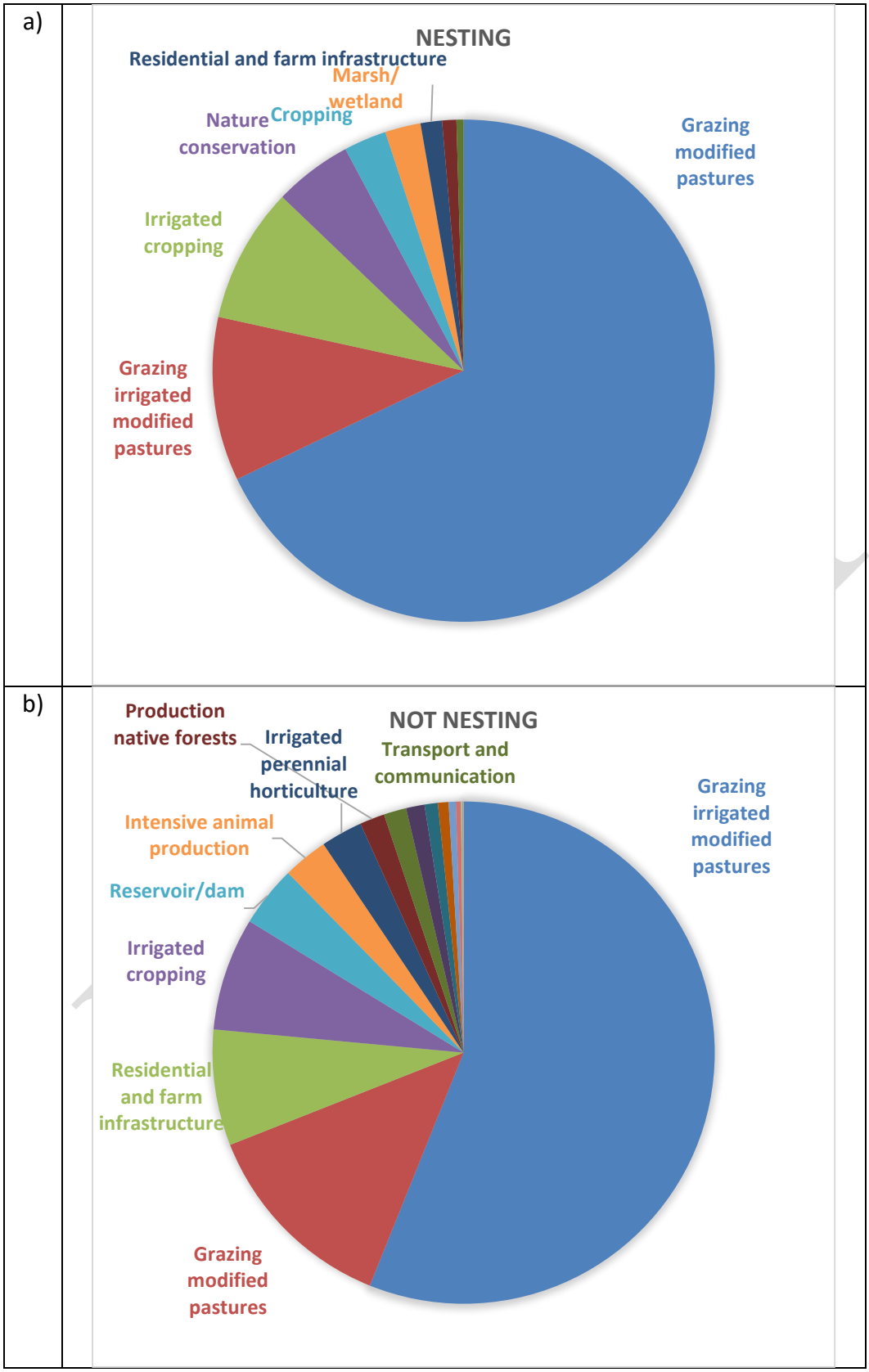
530 **Figure 20 Landuse types (ALUM 2016) used by an adult female straw-necked ibis (SNI) when foraging. a)**
531 **During nesting in Barmah Forest; b) when not nesting.**

532



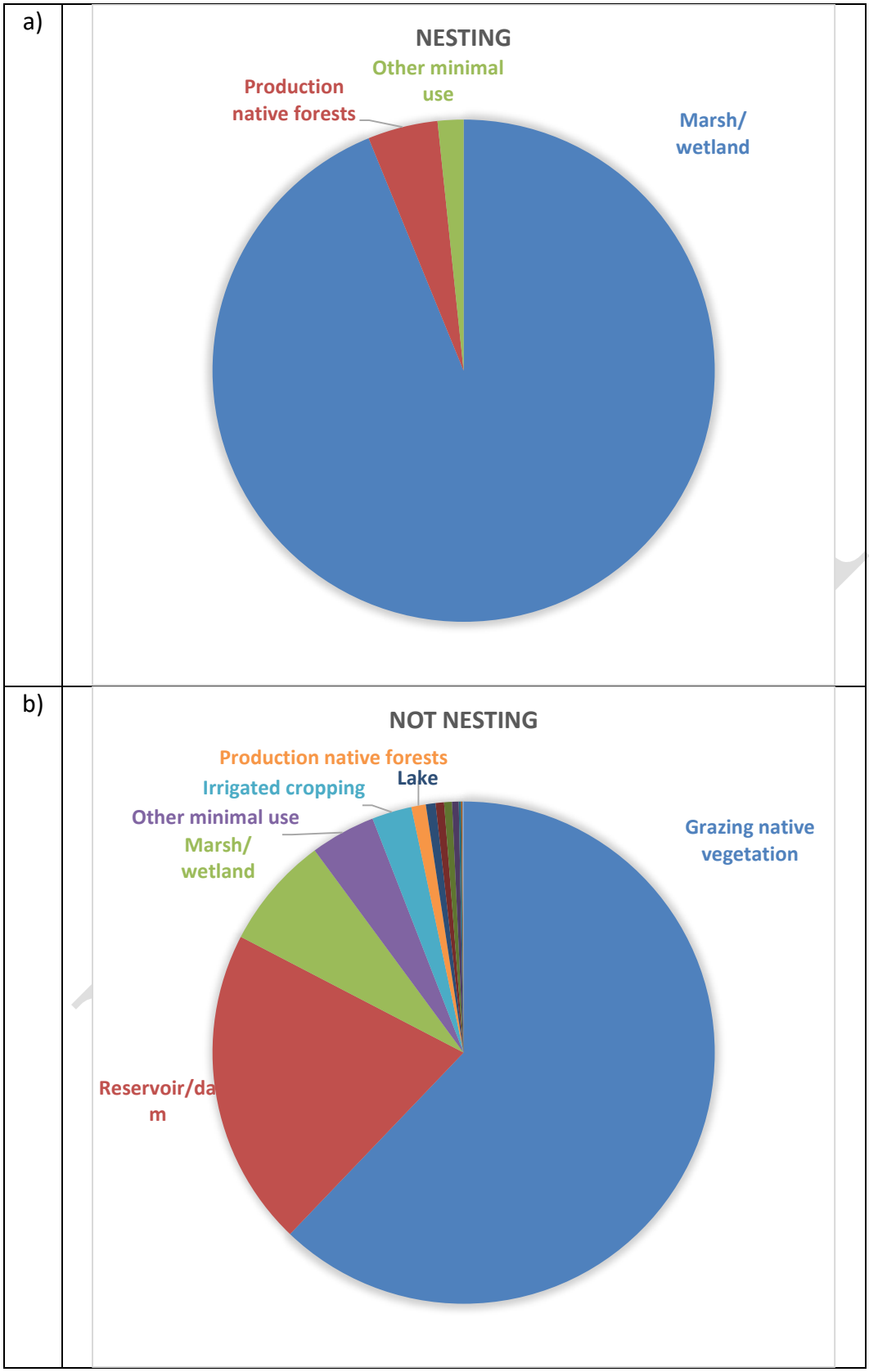
533 **Figure 21 Landuse types (ALUM 2016) used by an adult female straw-necked ibis (SNI) when foraging. a)**
534 **During nesting at Kow Swamp, VIC; b) when not nesting (foraging adjacent to Kow Swamp)**

535



536 **Figure 22 Landuse types (ALUM 2016) used by an adult female straw-necked ibis (SNI) when foraging. a)**
537 **During nesting at Kerang Wetlands, VIC; b) when not nesting**

538



539 **Figure 23 Landuse types (ALUM 2016) used by an adult male royal spoonbill (RSB) when foraging. a) During**
540 **nesting at Millewa Forest, VIC; b) when not nesting**

541

Mortality drivers

Satellite tracking of movements has for the first time allowed preliminary investigations into the relative importance of various drivers of mortality for juveniles and adults. While some birds were tracked for extended periods well over a year (), many others died or went missing. It can be difficult to establish the cause of death, but the timing, location and other factors can be established from transmitter data and other sources and sites visited to gather additional evidence and recover transmitters. An evidence scoring system is in development using data downloaded from the satellite transmitters (e.g. temperature, activity, solar recharge), the condition of the satellite transmitters when recovered, other on-ground evidence and weather data.

The main factors associated with mortality in the satellite-tracked birds were: Heat exhaustion / cold / exposure (weather extremes) (28%); Predation (e.g. raptors, foxes) (26%); Shooting (6%); Disease (e.g. botulism, widespread in 2016–17) (4%); Vehicle impact (4%); and unknown (26%) causes. Mortality drivers that may fall into the 'unknown' class may include any of the above causes, or other diseases, poisoning and toxins, starvation / malnutrition, parasites and entanglement in fencing or powerlines. Often it is not possible to determine what has occurred or birds go missing for no apparent reason. Interactions are likely – e.g. starvation or toxins/disease increasing susceptibility to predation or weather extremes.

Record-breaking heatwaves occurred in eastern Australia in both the 2016-2017 and 2017-2018 years during which birds were tracked. Australian summer and autumn temperatures have been rising steeply over recent decades (Australian Bureau of Meteorology 2019). Preliminary analyses indicate that associations between weather extremes and mortality or missing status for tracked birds are strongest where there is a shift from heat to cold and windy conditions within two days. Such conditions are likely to affect bird thermoregulation and can result in immune system depression and increased susceptibility to diseases, toxins and parasites. Juveniles are more susceptible than adults to these pressures, at least in part because they are naïve, are still learning to feed themselves and find shelter, use a lot of energy on first leaving the nesting area, and require high-quality food.

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A female straw-necked ibis on the day of capture with a new 22g satellite transmitter, fitted with a neoprene pad to prevent feather coverage of solar panels. This individual has been tracked moving within and between Victoria and Queensland over 18 months, covering over 10,000 km. She is still active.	
The same individual as that above, photographed foraging in Victoria in January 2019 after 411 days (13 months) of tracking. Straw-necked ibis hold their back feathers up as a heat management strategy and it was over 40^{o}C when this photo was taken. The raised feathers are not from the transmitter.	
A juvenile Australian white ibis with a 40g satellite tracker, photographed foraging in Victoria in January 2019 after 410 days (13 months) of tracking. He is still active.	

Figure 24 Photographs of tracked individuals taken after 13 months of continuous tracking.

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Discussion

Satellite tracking of waterbird movements has emphasised the need for Basin scale thinking and coordination in planning environmental water allocations and in managing expectations regarding responses. There is clearly physical population connectivity between the northern Basin and the southern Basin – and birds can move very quickly from north to south and vice versa. Planning and response predictions may need to account for varying population movement strategies – migration, nomadism and residency – and also for differences between species.

The existence of a common movement route between the northern and southern Basin suggests that for maximum impact, water and site management for foraging and stopover (refuelling) could be embedded within this route. However individual sites differ in importance among years and species and we don't yet fully understand this variation. Additional tracking of species dependent on surface water to feed will provide further information about key sites and movement routes that can be managed with environmental water.

Foraging habitat availability needs management both during and between breeding events. For example, tracking suggests that ibis and spoonbills target watered foraging sites within 1-3 km of appropriate roosting or nesting habitat. Environmental water could be used to increase the number or area of foraging sites within 10 km of nest sites after breeding, potentially extending watering for foraging into autumn and winter and possibly staggering inundation of foraging sites to maximise food productivity over a period of months to support juvenile survival.

At nesting sites used by royal spoonbills, satellite tracking has revealed the importance of extended duration of water availability for foraging by juveniles. Use of environmental water to extend nest and adjacent site flood duration during and after nesting beyond just the 'fledging' time threshold may facilitate increased juvenile survival. It is important to recognise that species such as spoonbills feeding in surface waters require different foraging-habitat provision and management to species with mixed terrestrial and aquatic diets such as ibis.

Management of over-wintering sites and foraging habitats may also be important for juvenile survival. Tracking indicates that over winter, even small habitat areas may be important, and

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there may be some site fidelity for such sites. Overwintering occurs in both the north and south of MDB, yielding a range of opportunities for supporting juveniles and sub-adults using environmental water to create or sustain foraging habitats and food sources. This may be particularly important in areas where irrigated agriculture and other water sources effectively 'dry up' over winter.

Overall, increased knowledge of the interaction of waterbirds with their environment, their movements and their life histories (Figure 25) is important for both basic understanding of their responses but also for informing policy and management decisions and predictions aimed at increasing waterbird abundance and maintaining waterbird diversity across the Basin.

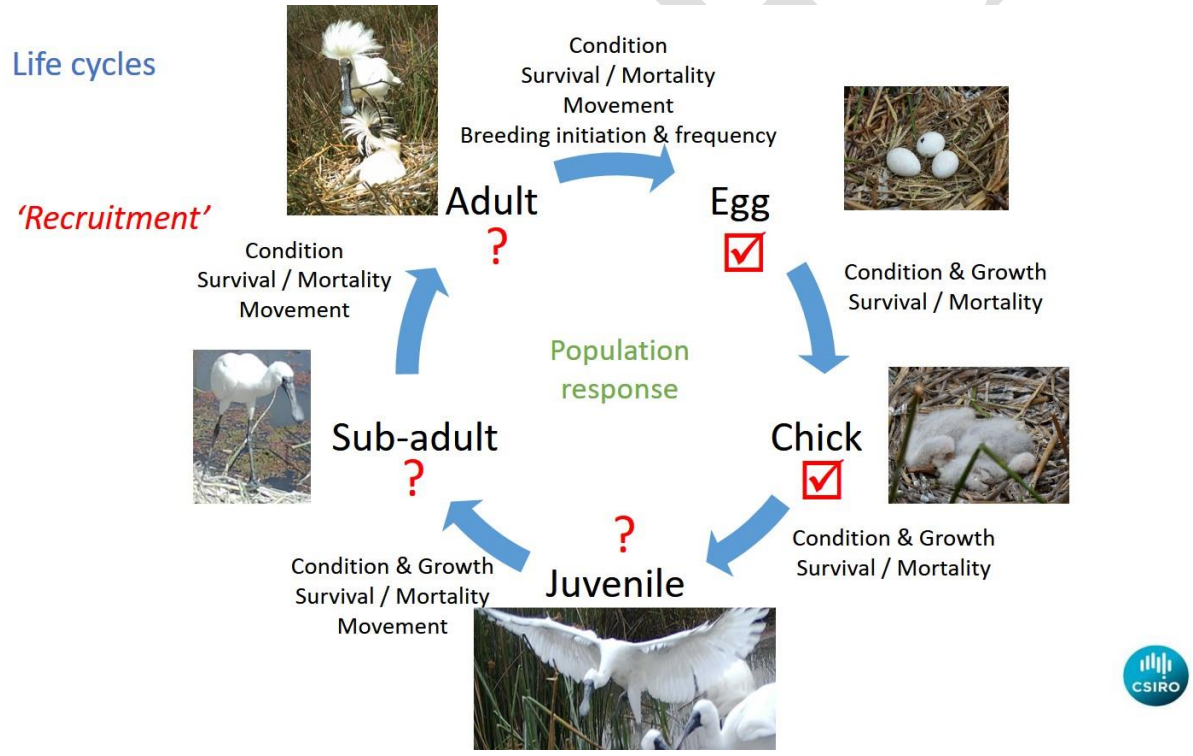


Figure 25 Major life cycle stages for colonial-nesting waterbird species and associated parameters affecting overall population responses and trajectories. The relative lack of information for the juvenile, sub-adult and adult stages (encompassing recruitment of chicks into the adult breeding population) is highlighted in red.

Acknowledgements

This work was supported by the Australian Commonwealth Environmental Water Office through the Murray-Darling Basin Environmental Knowledge and Research project's Waterbird Theme. The Waterbird Theme benefited from co-investment by CSIRO Land and Water and from in-kind support from the Royal Botanic Garden Sydney (RBGS; John Martin), NSW OEH and the Goulburn-Broken Catchment Management Authority (Keith Ward). The research was co-ordinated by Heather McGinness (CSIRO) with a leadership group comprising Richard Kingsford (UNSW), Veronica Doerr (CSIRO) and Ralph Mac Nally (UC) providing research direction and design advice.

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637 **Supplementary information**

638 **Table 7 Weights and measurements of tracked birds**

	Species and age	Mean	Min	Max	StdDev
Weight (g)	**SNI**	**1517**	**1000**	**1850**	**191**
	Adult	1546	1230	1850	199
	Juvenile	1501	1000	1806	189
	RSB	**1573**	**1280**	**1926**	**155**
	Adult	1926	1926	1926	-
	Juvenile	1548	1280	1740	125
	AWI	**1649**	**1380**	**2106**	**268**
	Juvenile	1649	1380	2106	268
	Grand Total	**1548**	**1000**	**2106**	**195**
Bill length (mm)	**SNI**	**112**	**75**	**183**	**29**
	Adult	146	110	183	22
	Juvenile	94	75	123	11
	RSB	**113**	**79**	**225**	**35**
	Adult	225	225	225	-
	Juvenile	105	79	127	17
	AWI	**97**	**88**	**106**	**8**
	Juvenile	97	88	106	8
	Grand Total	**111**	**75**	**225**	**29**
Head + bill length (mm)	**SNI**	**159**	**120**	**228**	**30**
	Adult	194	160	228	23
	Juvenile	140	120	170	11
	RSB	**148**	**125**	**250**	**29**
	Adult	250	250	250	-
	Juvenile	141	125	154	9
	AWI	**144**	**128**	**154**	**9**
	Juvenile	144	128	154	9
	Grand Total	**154**	**120**	**250**	**29**
Tarsus length (mm)	**SNI**	**91**	**78**	**105**	**8**
	Adult	81	78	85	3
	Juvenile	92	78	105	7
	RSB	**131**	**117**	**150**	**8**
	Adult	135	135	135	-
	Juvenile	131	117	150	8
	AWI	**100**	**90**	**106**	**6**
	Juvenile	100	90	106	6
	Grand Total	**104**	**78**	**150**	**19**
Wing length (mm)	**SNI**	**332**	**275**	**390**	**29**
	Adult	368	340	390	22
	Juvenile	326	275	381	26
	RSB	**308**	**250**	**385**	**37**
	Adult	385	385	385	-

	Juvenile	302	250	345	31
	AWI	**309**	**270**	**366**	**31**
	Juvenile	309	270	366	31
	Grand Total	**322**	**250**	**390**	**33**
Tail length (mm)	**SNI**	**130**	**90**	**173**	**15**
	Adult	134	127	140	7
	Juvenile	130	90	173	16
	RSB	**105**	**86**	**137**	**13**
	Adult	137	137	137	-
	Juvenile	102	86	114	9
	AWI	**121**	**101**	**137**	**12**
	Juvenile	121	101	137	12
	Grand Total	**122**	**86**	**173**	**18**

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641 **Table 8 Attributes of base datasets derived from classification of the satellite tracking data**

Point attributes	
OBJECTID	
ID	
Latitude	
Longitude	
Accuracy	
AquistionTime	
DateTimeLocal	
Time	
ClosestHour	
Date	
Interpolated	Yes/No
DateTimeUTC	
BandId	
GPSFixID	
Activity	Classification of purpose of being at each point (If midnight then activity = Roost, Elseif translocation = Travelled and previous and next movements = Travelled or Moved then activity = Travel, Else activity = Forage
Translocation	Type of translocation occurring on that day (derived from midnights for that day: Stayed < 0.025 < Shifted < 7 < Moved < 21 < Travelled)
Midnight (roost) attributes (Midnights are all points with ClosestHour=24)	
OBJECTID	
ID	
Latitude	
Longitude	
Accuracy	
AquistionTime	
DateTimeLocal	
Time	
ClosestHour	
Date	
Interpolated	Yes/No
DateTimeUTC	
BandId	
GPSFixID	
DistPrevNight	Straight line distance between this and previous midnight (km)
DistTravelled	Cumulative distance travelled along line segments during previous day (km)
Translocation	Classification of DistancePrevNight (Stayed < 0.025 < Shifted < 7 < Moved < 21 < Travelled)
DistRanged	Maximum straight line distance between previous midnight and any point during the day
FurthestRangedFixID	GPXFixID of maximum range point
Range	Classification of DistRanged (None = 0 < Adjacent < 1 Near < 4 < Local < 11 < Far < 20 Distant)

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RoostType	Classification based on previous and next day's translocation (If previous = Moved and (next = Moved or next = Travelled) then type = Explore, Elseif previous = Travelled and next = Travelled then type = Layover Else type = Reside
Line attributes:	
OBJECTID	
ID	
BandID	
FromDateLocal	
ToDateLocal	
FromID	
ToID	
ClosestHour	
Date	
DISTANCE	
BEARING	
Length	Classification of distance travelled along line segment (Minimal < 0.025 < Short < 1.25 < Medium < 7 < Long < 24 < Very L:ong < 50 < Extremely Long < 100 < Unusually Long)
Movement	Classification of the type of movement for the line segment (Staying < 0.025 < Shifting < 7 < if(Translocation = Travelled, Travelling, Moving))
Translocation	Type of translocation occurring on that day (derived from midnights for that day)
Shape_Length	

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DRAFT MANUSCRIPT IN PREPARATION FOR SUBMISSION TO A SCIENTIFIC JOURNAL FOR PUBLICATION

Quantifying egg and chick mortality in ibis and spoonbill (Threskiornithidae) nests using wildlife monitoring cameras

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Abstract

Significant management resources are frequently allocated to maintaining water levels at colonial-nesting waterbird breeding sites using 'environmental water'. While the effects of managed watering on breeding initiation, numbers of nests and nest abandonment are frequently monitored and reported, egg and chick survival rates and their drivers during environmental watering events are less commonly quantified. Such basic biological data are particularly scarce in Australia, even for relatively common species. Quantifying egg and chick survival rates allows better understanding of the population outcomes of the resources invested in supporting completion of a breeding event. It provides information on how many chicks are produced by the nesting event that (if they survive the juvenile and sub-adult stages) may consequently become recruits into the adult population. Using monitoring cameras to capture such data also facilitates explanation of perceived failures, by providing additional information describing the effects of other pressures, stressors or threats such as predation, disturbance, starvation, competition and other bird behaviour. This study used automatic wildlife cameras to monitor nests during three managed environmental water events for three colonial nesting waterbird species in the Threskiornithidae family. It had two aims: First, to quantify survival rates and the relative impacts of detected drivers for the species concerned; and second, to explore and test a range of methodologies and options for camera use in colonial-nesting waterbird colonies for various types of data collection.

Keywords: *Platalea regia, Threskiornis molucca, Threskiornis spinicollis*, survival, mortality, nest, colonial

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Introduction

Worldwide, freshwater ecosystems and associated waterbird populations are in decline. Recognition of this decline has led to a range of interventions including the development of international (e.g. JAMBA, Ramsar, EU, Americas), national (e.g. Australian Water Act, US Endangered Species Act 1973), regional and local scale management agreements and initiatives. In particular, interventions via the provision of 'environmental flows' are increasing, recognising the dependence of many waterbird species on flooding for nesting (Brandis *et al.* 2011; Kingsford and Auld 2005) Baker et al., 2015), migration (Krapu et al., 2014, Oneal et al., 2008; King and Allen 1996), and foraging (Xia et al., 2016).

Long-term declines in waterbird diversity and abundance are associated with decreases in breeding event frequency, size and success. This association is thought to be strongest for colonial-nesting waterbirds, including ibis, spoonbills, herons and egrets, and is frequently driven by water resource development or river regulation for human use. For example, in Australia long-term declines in the number of colonial-nesting breeding pairs/nests have occurred in areas subject to river regulation in the Murray-Darling Basin, but not in the unregulated Lake Eyre Basin (Kingsford et al. 2013; Brandis et al. 2009; Kingsford and Johnson 1990. Kingsford and Johnson 1998). Simultaneously, declines in waterbird abundance have occurred in areas subject to river regulation in the Murray-Darling Basin, but not to the same extent in the unregulated Lake Eyre Basin and Paroo wetlands (Kingsford et al. 2013; Kingsford et al. 2012).

Many colonial-nesting waterbird species require water to be present underneath or surrounding their nesting sites in order to initiate and complete breeding (Briggs and Thornton 1999; Briggs et al. 1997; Leslie 2001; Carrick 1962; Maher 1991; Maher and Braithwaite 1992). After taking into account the time needed for birds to prepare behaviourally, nutritionally and hormonally for breeding, egg incubation, fledging, and acquisition of foraging skills by juveniles, nest sites and nearby foraging areas need to be flooded for several months, with minimum flood durations dependent on species, pre-flood site conditions, and the season during which flooding began (Reid et al. 2009; Carrick 1962; Kingsford 1998; Leslie 2001; Brandis et al. 2011; Briggs et al. 1997; Maher 1988, 1990; Briggs and Thornton 1999). In the southern Murray-Darling Basin, modelling of long-term datasets has found that exceedence of daily flow thresholds is usually required for 30-50 days to trigger breeding in most colonial nesting waterbird species. For all species, the maximum probability of attempted breeding occurs after a daily flow threshold is exceeded for about 50 days. Ibis respond to a 5-10 day shorter period of threshold exceedence than egrets (Arthur et al. 2012).

Rapid reductions in flood depth or reductions below a threshold depth can cause nest abandonment and reduce the proportion of successful nests. Consequently, significant management resources are

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frequently allocated to maintaining water levels at breeding sites using 'environmental water'. While the impacts of managed watering on breeding initiation, numbers of nests and nest abandonment are frequently monitored and reported, egg and chick survival rates and their drivers during environmental watering events are less commonly quantified. Such basic biological data are particularly scarce in Australia, even for relatively common species. Quantifying egg and chick survival rates allows better understanding of the population outcomes of the resources invested in supporting completion of a breeding event. It provides information on how many chicks are produced by the nesting event that (if they survive the juvenile and sub-adult stages) may consequently become recruits into the adult population. Using monitoring cameras to capture such data also facilitates explanation of perceived failures, by providing additional information describing the effects of other pressures, stressors or threats such as predation, disturbance, starvation, competition and other bird behaviour.

This study used automatic wildlife cameras to monitor nests during three managed environmental water events for three colonial nesting waterbird species in the Threskiornithidae family. It had two aims: First, to quantify survival rates and the relative impacts of detected drivers for the species concerned; and second, to explore and test a range of methodologies and options for camera use in colonial-nesting waterbird colonies for various types of data collection.

Methods

Context

The Barmah-Millewa Forest is the largest River Red Gum (*Eucalyptus camaldulensis*) forest in Australia and the world. It comprises Barmah Forest in Victoria and the Millewa Forests in New South Wales. It covers approximately 66,000 hectares of floodplain and wetlands surrounding the Murray River. The Barmah-Millewa Forest is one of six icon sites of The Living Murray (TLM); a Federal and State Government initiative aimed at improving environmental health in the Murray River system through the monitoring of bird, vegetation and fish responses to environmental watering. The Barmah-Millewa Forest contains internationally significant wetlands listed under the Ramsar Convention, and was gazetted into two National Parks in 2010 (Barmah National Park, NSW and Millewa National Park, Victoria).

Vegetation communities in Barmah-Millewa Forest comprise a range of significant habitats for flora and fauna, including swamps and marshes, rush beds, lakes and billabongs, open grassland plains with Moira Grass (*Pseudoraphis spinescens*) plains, River Red Gum forests and woodlands, and Black Box (*Eucalyptus largiflorens*) woodlands. During appropriate seasonal conditions and

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flooding the forest and its wetlands provide important foraging and breeding habitat for waterbirds, with 54 recorded breeding species, of which 25 are colonial nesting species. Large waterbird nesting events have been recorded within the Forest since early in the 20th century (Mattingley, 1907, 1908). Although these breeding events still occur, they are much smaller in size compared to historical events (Maher, 1993) and occur less frequently (Leslie, 2001). This has been attributed largely to water abstraction and flow regime change.

Study site & hydrology

Reed Beds Swamp is a naturally occurring wetland located in Millewa Forest, east of Mathoura, NSW, and is within the Ramsar and TLM icon sites. The wetland is bordered by River Red Gum forest and is dominated by extensive beds of Common Reed (*Phragmites australis*) and Giant Rush (*Juncus ingens*) and with some areas of Cumbungi (*Typha sp.*). Reed Beds is divided into distinct north and south regions the Gulpa channel or cutting, which restricts water flow from Gulpa Creek into the northern region of Reed Beds (named Reed Beds North in this study) (Figure 1). The southern region (Reed Beds South) receives unregulated water flows from Gulpa Creek, and is prone to sudden rises and falls in water level (Figure 1). The water flow into Reed Beds North is controlled by regulators, enabling the control of water levels throughout the year. The release of water into Reed Beds North is aimed at triggering and sustaining breeding events for colonial nesting waterbirds when the season is deemed suitable. Breeding of colonial nesting waterbirds in Reed Beds occurs in the spring and summer months (usually September-February) and may occur in both northern and southern regions of Reed Beds. The timing of the delivery of environmental flows into Reed Beds North aims to correspond with the colonial nesting waterbird breeding season.

For the purpose of this study and to better distinguish the spatial location of nests in the wetland, Reed Beds North was further classified into west and east regions (named Reed Beds West and Reed Beds East) (Figure 1), corresponding to access points into the wetland.

Generally, over the periods of monitoring river flows decreased from 1000 to less than 500 megalitres per day. This was reflected by water depths (Figure 2). In each monitoring season, the water depth steeply decreased by up to 46 cms (one foot and a half) in one season (weeks).

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Figure 1 Satellite image from 2018 of Reed Beds Swamp, NSW, showing the three regions where monitoring cameras were deployed. West and East regions make up what is known as Reed Beds North, with the Gulpa channel separating Reed Beds South.

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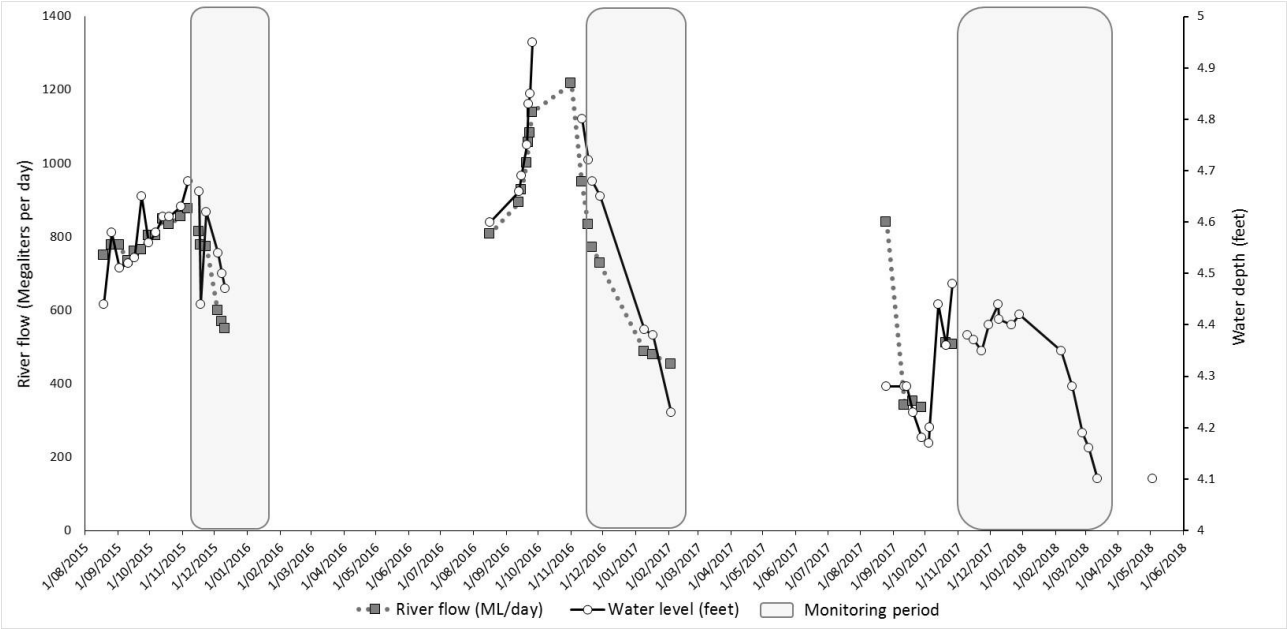


Figure 2. Water depth measured in feet at Reed Beds (the field site) over time and during the camera monitoring periods. The shaded rectangles represent the time periods of camera monitoring.

Monitoring camera deployment

Monitoring cameras were deployed in Reed Beds Swamp in three consecutive waterbird breeding seasons; November 2015- February 2016; November 2016- February 2017 and October 2017- April 2018. The 2015-2016 breeding season deployment was a pilot study aimed at developing and testing methods in the field. The species monitored were Australian White Ibis (*Threskiornis molucca;* AWI), Straw-necked Ibis (*Threskiornis spinicollis;* SNI), and Royal Spoonbill (*Platalea regia;* RSB). Straw-necked Ibis were not monitored in 2017-2018 because they did not breed in Reed Beds. The two main categories of data collected by monitoring cameras were: (1) egg and chick counts over time, yielding information about incubation periods, hatching, egg survival rates and chick survival rates per nest (nest success) and (2) bird behaviour and predation events.

Three different monitoring camera brands were used over the course of the study: (1)'Swift' cameras (https://outdoorcameras.com.au/shop/swift-3c3g-solar-package/), which are solar-powered and transmit live still images via the mobile phone network; (2) 'Reconyx PC900' cameras (http://reconyx.com.au/Professional-pc900-full-black-flash.php), take still images and record to a 32 GB SD card, powered by Eneloop rechargeable batteries, and (3) 'Boblov RD1003' (https://www.boblov.com/m/prod_view.aspx?TypeId=75&Id=173&Fid=t3:75:3), take still images and record to a 32GB SD card, powered by Eneloop rechargeable batteries. Each brand had different features which made it suitable for collecting slightly different data types. Reconyx cameras are extremely robust, can record both time-lapse and motion-sensed imagery at the same

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time, and produce high-quality images. Boblov cameras were an affordable alternative to Reconyx cameras for capturing time-lapse images. Swift cameras enable high quality live imagery to be sent via the 3G telephone network and be accessed in real-time via a website, eliminating the need to change over full SD cards. Swift cameras also enable battery recharge via solar panel, eliminating the need to replace batteries in the field. Swift cameras were chosen for their perceived benefit of less time spent on camera maintenance in the field and therefore less disturbance to nests and more reliable long term monitoring data.

For the 2015-2016 season monitoring cameras were deployed in the field mounted on a bolted set of 1.8m (19mm x 19mm wide) hardwood posts. For the following 2016-2017 and 2017-2018 seasons, 25mm x 25mm wide posts were used for greater stability and to prevent snapping. Prior to field deployment, two identically positioned holes (15cm apart) were drilled in the non-staked ends of both poles to enable attachment via bolts; the attached two poles measuring a total height of over 3m. Cameras were either clamped onto the post during deployment or screwed into the post before deployment. Bird spikes were glued to the top of the cameras and on the top edge of the Swift solar panels (image 1) to prevent birds standing on the cameras. The Swift cameras also required a bracket to mount the battery pack below the camera and a bracket at the top of the post for the solar panel; this was prepared prior to field deployment to minimise deployment time and disturbance to nests (Image 1).

After a nest or group of nests were selected for monitoring, the first post was positioned at least 1.5 metres from the nest edge and hammered into the ground, ensuring the bolt holes were above water to prevent the wood around the holes swelling. The second post with the mounted camera was then attached to the top of the hammered-in pole with two bolts (Image 1). The Swift cameras were mounted under the solar panel to provide extra protection from birds and rain, and the battery cord connected to the camera (Image 1). The angle of the camera was adjusted to ensure the nest/s were in camera view. In the second and third breeding seasons cable ties were also used to attach the camera to the pole to prevent the camera mount loosening and the camera angle view moving off nests.

Monitoring camera locations were GPS marked and mapped for later reference. Other measurements and data taken at the time of camera deployment included: the number of eggs and chicks per nest; estimated chick age; the depth of water below the nest; and the height of the nest above water. In the third breeding season (2017-2018) a Kestrel DROP D2 environmental data logger was used to more accurately record the ambient temperature within the breeding colony and rectify camera temperature data.

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Monitoring cameras were checked every 3-4 weeks throughout the breeding season, allowing full SD cards and depleted batteries to be replaced, and cameras to be maintained as required. Cameras on empty nests were also moved to new active nests; this was considered a new camera deployment. For this reason each breeding season has multiple camera deployment periods. At the completion of each breeding season, usually when most chicks had left nests, all monitoring camera equipment was removed from the field.

Monitoring camera settings

Monitoring camera settings evolved throughout the study in an effort to optimise the quality and type of data extractable from images, and to extend and maintain the period of nest monitoring. The addition of Boblov and Swift cameras in the second and third breeding seasons enabled the camera brands to be designated specific data collection purposes.

2015-2016 breeding season (pilot study)

During 2015-2016, four field trips were conducted to deploy and maintain monitoring cameras, with three deployment periods covered:

1) November to December 2015: 30 Reconyx cameras, comparing two camera settings (Setting 1: 20 cameras; Setting 2: 10 cameras; Table 1)
2) December 2015 to January 2016: 29 Reconyx cameras, comparing two camera settings (Setting 1: 1 camera; Setting 2: 1 camera; Setting 3: 26 cameras; Setting 4: 1 camera)
3) January to February 2016: 15 Reconyx cameras, comparing three camera settings (Setting 5: 5 cameras; Setting 6: 5 cameras; Setting 7: 5 cameras).

Reconyx cameras were first deployed in Reed Beds East and West in November 2015 on 42 Australian white ibis nests and 1 straw-necked ibis nest with eggs. Cameras were also deployed in Reed Beds West on trampled reed beds where royal spoonbill pairs were observed (a sign of pair-bond formation and early nest building), to obtain dates for egg laying and incubation. It was assumed that cameras programmed to both motion-sensing and time-lapse would capture changes in egg and chick numbers over time via time-lapse and sufficiently capture mortality events and parental attendance via motion-sensing (Table 1). Cameras with time-lapse only were set to capture images more regularly than those cameras with both settings, as a measure of comparison for time-lapse settings (Table 1).

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After reviewing imagery from the first deployment it became apparent the motion sensing settings weren't capturing parents departing and arriving at the nest. Motion sensitivity was increased to 'high' and the number of images per trigger was increased to '3' for the second camera deployment (December 2015) (Table 1). Sixty-five nests were monitored during this period with cameras kept on 20 Australian white ibis nests from the November 2015 deployment. Cameras were set up on single royal spoonbill nests with and without eggs in Reed Beds West and on groups of straw-necked ibis or Australian white ibis nests with eggs and chicks in Reed Beds East and West.

In the third deployment (January 2016) camera settings were further compared by setting up three cameras with different settings on each of five single royal spoonbill nests with eggs and chicks. The overall aim was to capture all events in the nest by using camera settings to capture specific data. The comparison of images of the same time point between the three cameras enabled identification of the best settings for capturing changes in egg and chick numbers, mortality events and parental attendance.

Table 1: Reconyx PC900 monitoring camera settings for pilot study test deployments (2015-2016).

Deployments	**November 2015**	**November 2015; December 2015**	**December 2015**		**January 2016**		
Setting #	**1**	**2**	**3**	**4**	**5**	**6**	**7**
Program	Time-lapse & motion sensing	Time-lapse	Time-lapse & motion sensing	Time-lapse & motion sensing	Time-lapse & motion sensing	Time-lapse & motion sensing	Time-lapse
Motion sensitivity	Medium-High	N/A	High	Medium-High	High	High	N/A
Pictures per trigger for motion sensing	1	N/A	3	1	3	3	N/A
Period armed for motion sensing (hours)	24	N/A	24	24	24	24	N/A
Time-lapse interval (minutes)	30	5	30	5	1 hour	8am and 6pm only	1 minute
Pictures per trigger for time-lapse	1	1	1	1	1	1	1
Period armed for time-lapse (hours)	24	24	24	24	7:00 – 19:00	24 hours	7:00 – 19:00

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Picture rate (seconds)	N/A	N/A	N/A	N/A	Rapidfire	Rapidfire	N/A
Quiet period	0	None	0	0	30	0	None
Number of nests in camera view	0 to 15	0 to 15	1 to 10	0 to 15	1	1	1

2016-2017 breeding season

During 2016-2017, five field trips were conducted to deploy and maintain monitoring cameras, with three deployment periods covered:

1) November to December 2016: 25 Boblov cameras (one stolen during deployment), 28 Reconyx cameras (one stolen and one damaged) and 28 Swift cameras (Table 2)
2) December 2016 to January 2017: 22 Boblov cameras (18 remained on nests from Nov/Dec, four were deployed onto new nests), 27 Reconyx cameras (8 remained on nests from Nov/Dec deployment, 19 deployed onto new nests) and 29 Swift cameras (19 remained on nests from Nov/Dec deployment, 10 deployed onto new nests (Table 2)
3) January to February 2016: 17 Boblov cameras (1 remained on nests from Dec/Jan deployment, 16 deployed onto new nests), 22 Reconyx cameras (all deployed onto new nests) and 24 Swift cameras (3 remained on nests from Nov/Dec deployment, 21 deployed onto new nests) (Table 2)

A total of 54 new cameras (25 Boblov cameras; 29 Swift cameras) were deployed with the existing 28 Reconyx cameras at the beginning of the 2016-2017 season.

The settings were different for each camera brand to capture different data (Table 2) and remained fixed for the entire breeding season. Reconyx cameras were prioritised to perform motion-sensing to capture detail of mortality and disturbance events as well as parental attendance on 1-5 nests (Table 2). Boblov cameras were set to time-lapse only to monitor changes in egg and chick numbers in 5-15 nests over the entire breeding season (Table 2). It was assumed that Reconyx cameras would run out of battery or SD cards fill up between deployments due to the frequency of movement on nests. It was also assumed that Boblov camera batteries and SD cards would last between deployments as a fixed maximum number of images were taken each day. Therefore a Boblov camera was mounted on the same pole as a Reconyx camera and focused on the same area of nests in an effort to maintain monitoring between deployments. Swift cameras were installed on single high-priority nests where detailed data tracking a nest from start to finish was desired.

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Cameras were deployed to monitor single royal spoonbill nests with eggs and chicks in Reed Beds West, and groups of nests with eggs and chicks of straw-necked ibis and Australian white ibis in Reed Beds East, West and South.

Table 2: Monitoring camera settings during the 2016-2017 breeding season. N/A= not applicable

Camera brand	**Boblov RD1003**	**Reconyx PC1900**	**Swift**
Camera purpose	Egg and chick counts (hatching rates, nest success)	Behaviour, feeding rates, predation, disturbance	Egg and chick counts, bird behaviour, feeding rates, predation, disturbance
Program	Time-lapse	Motion sensing & Time-lapse	Motion sensing
Motion sensitivity	None	Medium-High	High
Pictures per trigger - Motion sensing	N/A	3	3
Period armed - motion sensing (hours)	N/A	24	24
Time-lapse interval	30 minutes	8am, 6pm	30 minutes
Pictures per trigger - time-lapse	1	1	1
Period armed - time-lapse	Diurnal	Diurnal	Diurnal
Picture rate	N/A	RAPIDFIRE	High
Quiet period (seconds)	N/A	0	0
No of nests in camera view	5 to 15	1 to 5 with good visibility	1 to 5 with good visibility

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2017-2018 breeding season

During the 2017-2018 breeding season, five field trips were conducted to deploy and maintain monitoring cameras, with four deployment periods covered:

1) October to November 2017: 28 Swift cameras (Table 3)
2) December 2017 to January 2018: 25 Swift cameras (Table 3)
3) January to February 2018: 20 Swift cameras (4 remained on nests from Dec/Jan deployment, 16 deployed onto new nests)
4) February to April 2018: 18 Swift cameras (15 remained on nests from Jan/Feb deployment, 3 deployed onto new nests)

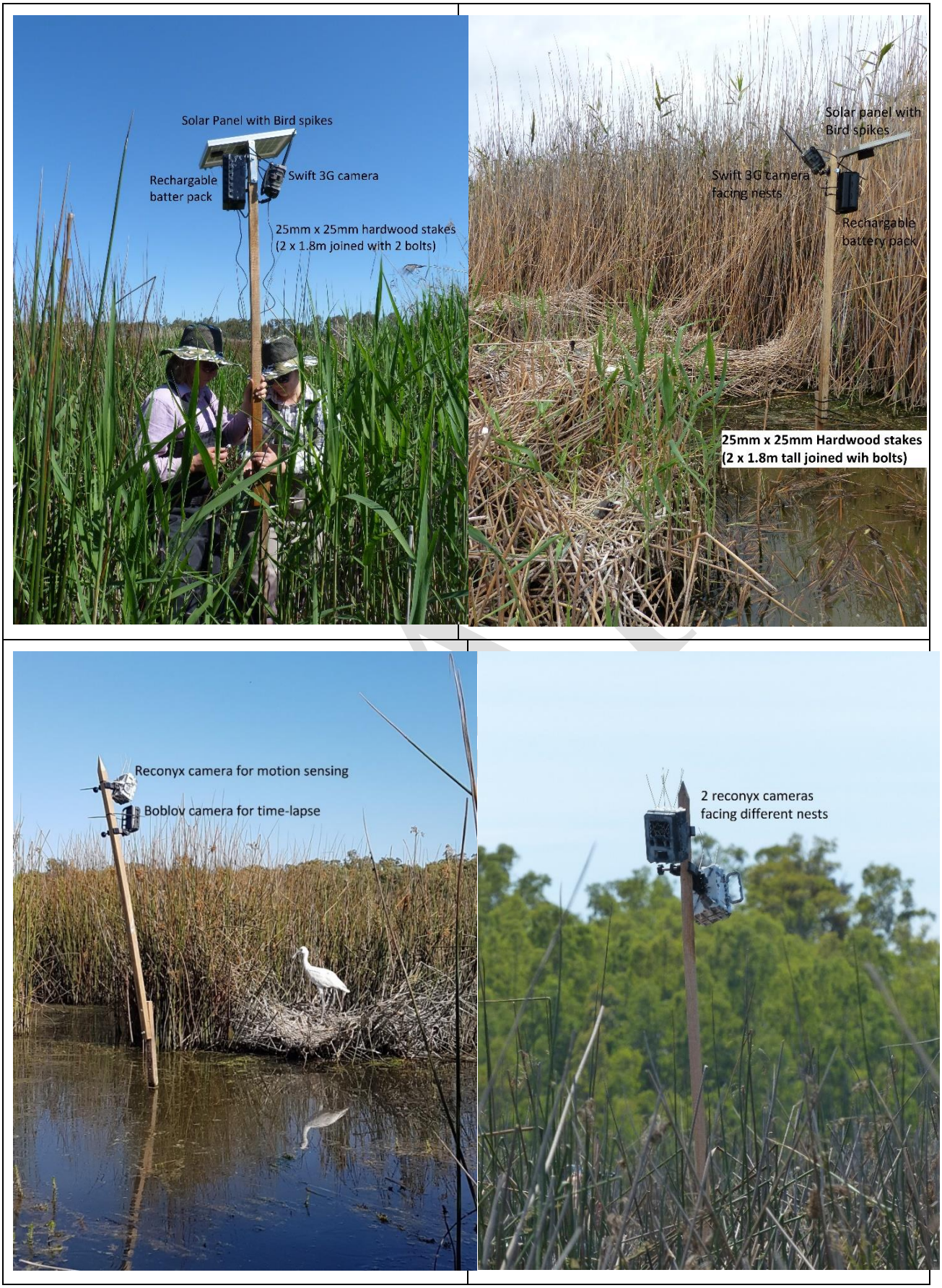
In October 2017 28 Swift cameras were deployed in Reed Beds East and West prior to birds arriving. Cameras were set up in areas of previous ibis and spoonbill nesting to capture any nest area re-use, arrival time and behaviours such as reed trampling, courting or nest-building. Cameras were positioned to focus on reed beds from varying distances (some within or close to vegetation, or facing an open channel) to cover a range of nesting scenarios and positions.

By November 2017 birds were foraging and nesting within view of some cameras, however too far away to determine egg numbers, so all cameras were repositioned on single royal spoonbill nests containing eggs and chicks in Reed Beds East and West, and groups of Australian white ibis nests containing eggs and chicks in Reed Beds East, West and South. Straw-necked ibis did not nest. Swift camera settings were changed from 2016-2017 to only take a single image per trigger for motion sensing with no quiet period between triggers (Table 3). This change resulted from previous technical difficulties with image transmission to the website. The issue was that when the cameras took a sequence of images, the camera only sent the first image to the web site and stored the remaining pictures on the SD card. Once the SD card was full the camera overwrote the old images with new ones. By taking one photo per trigger, all images were received on the website for analysis. Deployed cameras in Reed Beds decreased over the season as chicks left nests. Reconyx and Boblov cameras were not deployed due to an insufficient number of nests to warrant their use.

Table 3: Swift monitoring camera settings during the 2017-2018 breeding season.

Camera purpose	Egg and chick counts, bird behaviour, feeding rates, predation, disturbance
Program	Time-lapse & Motion sensing
Motion sensitivity	Medium
Pictures per trigger - Motion sensing	1
Period armed - motion sensing (hours)	24
Time-lapse interval (minutes)	30
Pictures per trigger - time-lapse	1
Period armed - time-lapse	Diurnal 7am-7pm
Pic rate (secs)	High
Quiet period (seconds)	0
Number of nests in camera view	1 to 5 with good visibility

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321 Figure 3 Various camera deployment combinations in the field.

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Data extraction and analysis

Reconyx and Boblov camera images were transferred from SD cards onto hard drives for processing, and Swift camera images were processed via the website, at the end of each breeding season. All images were processed manually with data extracted visually and entered into Microsoft Excel workbooks at the end of each season (see Supplementary material for details of variables recorded). Nest exposure variables were assessed using ArcGIS, satellite imagery and field data (

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329 Table 4).

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331 Table 4 Nest exposure parameters

Parameter	Description/definition
Nesting group ID	A 'nesting group' is defined as an area of nests within close proximity which can be separated from another area of nests based on distance, differences in nest layout within the area, differences in overall vegetation shape and structure, and differences in water and element exposure.
Nesting group shape	The dominant shape of the nesting group. **Open Fringe** (nesting group on reed bed fringe adjacent to expanse of open water with little or no reed vegetation). **Closed Fringe** (nesting group on reed bed fringe adjacent to water with sparser or shorter reed vegetation. eg. phrag reed bed surrounded by juncus). **Pocket** (nesting group within reed bed situated in open 'pocket' of water. Little or no open water channel joining pocket; reeds surrounding pocket are dense, ie. navigating canoe through requires pushing reeds over. **Interior Channel** (nesting group within reed bed but on fringe adjacent to open water channel. Water channel must be free of tall reeds; ie. wide enough to navigate canoe through without pushing through reeds. **Amphitheatre** (nesting group has trampled down and nested within the interior of an island reed bed. Nests usually of varying heights with shorther nests at front and taller nests at rear. Tall reed wall still surrounds nesting group. Structure usually crescent moon shaped.) **Closed archipelago** (nesting group within a chain of small 'island' clumps of reed/rush [ie. clumps too sparse and discontinuous to be considered a 'bed', but in close proximity of another]. Islands surrounded by a matrix of reeds or rushes). **Open archipelago** (nesting group within a chain of 'island' clumps of reed/rush [ie. clumps too sparse and discontinuous to be considered a 'bed']. Islands surrounded by open water and/or sparse rushes).
Nest aspect	Each nest was visually assessed in ArcGIS with the aim of determining in which cardinal direction the nest was the most exposed (to heat, light, wind). Cardinal directions were only determined to N, NE, E, SE, S, SW, W or NE. If a nest was offset from the vegetation, creating some clearance in all directions, the direction with the most clearance or exposure was chosen. If a nest was offset from the vegetation such that there was equal exposure on all 4 sides, it was given an A (All).
Vegetation wall type	Vegetation wall defined as a continuous and intact reed/rush barrier that has not been trampled or bent. When deciding what is or isn't a vegetation wall, consider whether the vegetation would provide the nest with shelter from the elements.
Vegetation wall protection @ nest scale	How far is the nest from the nearest vegetation wall? Think in terms of number of nests away from the wall.
Open water exposure @ nest scale	Is open water visible surrounding nest? 'Open water' for this purpose is defined as an area where more water surface is visible in the image than reed/rush vegetation.
Min nest height from water (cm)	The minimum height in cm of the nest heights recorded for the nest being assessed.
Max nest height from water (cm)	The maximum height in cm of the nest heights recorded for the nest being assessed.
Average nest height from water (cm)	The average in cm of the nest heights recorded for the nest being assessed.

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Results

Timing of nesting and number of nests

The number of breeding pairs and species present and the timing of breeding varied between years.

During the 2015-16 water season, 60GLs of environmental water were delivered to Millewa Forest from mid-August to late January, with 15,300 ML delivered specifically to support waterbird breeding in Reed Beds (Borrell and Webster 2016). Australian white ibis had begun nesting by October. On 23 October 2015 there were with approximately 120 AWI displaying breeding behaviour (posing on trees and trampling), and approximately 90 nests, some at first egg stage (Keith Ward GBCMA aerial survey; Figure 4). By the week of 9 November 2015 when cameras were deployed, there were approximately 150 AWI nests, most with eggs. Some straw-necked ibis were also observed nesting among the AWI with an additional 150 birds observed trampling in preparation to nest. At this time approximately 20 royal spoonbill nests were recorded, some with eggs, together with evidence of additional trampling and courting, with more than 120 and up to 200 individuals recorded. Over the breeding season, approximately 430 pairs of ibis and 230 pairs of royal spoonbills were observed nesting in Reed Beds as part of approximately 1077 colonial-nesting waterbird pairs in the entire Reed Beds complex (Borrell and Webster (2016).

During the 2016-17 water season, environmental water was delivered to Millewa Forest for 86 days from November to the end of January. The environmental water releases included the delivery of 46 gigalitres (GL) of environmental water delivered into Millewa Forest and 4.488GLs delivered into Reed Beds. River flows were elevated over spring/summer due to the delivery of River Murray Increased Flows (RMIF), Planned Environmental Water (PEW) and 110GL from the Commonwealth. Colonial waterbirds began nesting later than the previous year (Borrell et al. 2017). There were ~1018 colonial-nesting waterbird pairs recorded in Reed Beds by NSW NPWS surveys. Straw-necked ibis nested twice, once at the start of the season mixed in with the white ibis at sites in Reed Beds North, and once in Reed Beds South, with eggs still present in February. Flows were held up in the Gulpa Creek until the end of February to ensure a successful fledging and a slow recession was implemented throughout March for these late eggs.

During the 2017-2018 season, rainfall and flows were much more limited than in previous years. However, Barmah-Millewa forest received flows throughout spring that inundated 25% of the forest (Borrell 2018 citing Kuo and Thomas, 2018). An environmental water release included the delivery of 5.6 gigalitres (GL) of targeted environmental water delivered through Millewa Forest regulators and 6.7 GLs delivered into Reed Beds (Borrell 2018). Due to a late increase in waterbird breeding numbers, an extension of three weeks was delivered via the Gulpa Creek offtake regulator to

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maintain water levels. Over 250 pairs of colonial nesting waterbirds were recorded at Reed Beds, including 102 pairs of royal spoonbills and 148 pairs of Australian white ibis. Straw-necked ibis were flying in groups around the wetlands early in the season, however none were recorded nesting. Australian white ibis began nesting in late November with eggs and chicks being produced throughout December to late February and fledging mostly complete in February. Royal spoonbills began constructing nests from late October to December and eggs were present from mid-November to mid-February, with chicks fledging in March and April.



Figure 4 Aerial photo of part of Reed Beds Swamp (Keith Ward, GBCMA)

Monitored nests and periods

A total of 412 nests were monitored by cameras over the 3 year period: 248 AWI, 90 SNI and 74 RSB (Table 5). Initially, the duration of camera monitoring at each nest was often short and the dataset was complex because of variation in settings being tested, memory and battery limitations, camera swivel attachment problems and other logistical issues. In addition, where nests did not progress or were subject to predation, cameras were relocated. Consequently many cameras did not provide a continuous record of activities at each nest throughout the breeding period. These issues were addressed in subsequent deployments. In particular, certain cameras were dedicated to monitoring nest fates through time-lapse imagery for the entire nesting period. Despite these issues,

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the dataset yielded results of interest describing aspects of clutch sizes, egg and chick survival, predation, nest attendance and bird behaviour. All aspects of survey effort (Table 5) dramatically increased each monitoring season after method testing during the pilot study of 2015-2016. Fewer nests were monitored in the 2017-18 season due to drought conditions but the average duration of monitoring, continued to improve.

At the beginning of each monitoring session some monitored nests already had chicks present. Because of this, the dataset contains both direct and indirect observations of chick survival from the egg stage. For analysis, these were separated into direct records only vs all records (Table 5) in order to: a) give the most accurate statistics of chick survival from our direct observations from egg stage to nest departure; and b) present overall broader estimates. The indirect observations include the chicks observed to hatch plus the chicks already present at the beginning of monitoring.

Table 5. Summary data of the survey effort for each species and each monitoring season.

								Monitoring days		
Year	Species	No. nests	Direct records	Indirect records	No. nests full record	No. of clutches	No. of eggs full record	Mean	Min	Max
2015-2016 Pilot study	Australian White Ibis	98	66	-	4	103	4	11.10	1	57
	Royal Spoonbill	23	11	-	0	23	0	22.87	1	44
	Straw-necked Ibis	24	24	-	0	24	0	12	1	33
2016-2017	Australian White Ibis	96	150	76	16	127	33	28.66	1	88
	Royal Spoonbill	26	73	16	17	38	50	47.89	7	88
	Straw-necked Ibis	66	33	52	3	84	4	22.86	2	87
2017-2018	Australian White Ibis	54	70	9	10	54	17	28.35	3	94
	Royal Spoonbill	21	35	3	12	21	28	47.23	6	97
	Straw-necked Ibis	0	-	-	-	-	-	-	-	-

Clutch sizes

Greater accuracy in estimating the summary statistics of clutch size was gained with the dramatic increase in sample size of direct records and the duration of monitoring per nest for each species in the years following the pilot study. Median clutch sizes were 2 eggs for straw-necked ibis, 2-3 eggs for Australian white ibis, and 3-4 eggs for royal spoonbills (Table 6).

Table 6. Summary statistics of clutch size for each species and each monitoring season (all records).

Year	Species	Mean	SD	Median	Max	Min	Number of clutches
2015-2016 Pilot study	Australian White Ibis	2.12	1.17	2	5	0	103
	Royal Spoonbill	0.87	1.36	0	3	0	23
	Straw-necked Ibis	2	0.72	2	3	1	24
2016-2017	Australian White Ibis	2.17	1.39	3	6	0	127
	Royal Spoonbill	3.03	1.59	4	4	0	38
	Straw-necked Ibis	1.60	1.17	2	3	0	84
2017-2018	Australian White Ibis	2.33	0.61	2	3	1	54
	Royal Spoonbill	2.70	0.75	3	4	1	21
	Straw-necked Ibis	-	-	-	-	-	-

Egg and chick mortality and survival

Hatching and chick survival rates varied between seasons and species (Table 7; Table 8). Generally, if a chick hatched, its likelihood of survival to the end of monitoring significantly increased. Approximately 20-50% of all eggs laid survived, while 75-97% of all hatched chicks survived (Table 7; Table 8). The overall estimate of survival from monitored nests is inherently biased to be greater than the survival rate of direct records because there would have been chicks and/or eggs present that died that we did not observe, hence statistics based on 'direct records' are more accurate.

In the pilot study, the number of nests monitored that had chicks present at the end of monitoring was low (Table 9). This was not due to a lack in survival but rather was because of the timing of camera deployments. Many nests at the end of monitoring in the pilot study were still at the egg stage (Table 10). Of those with chicks present, most were runners aged 10-16 days old (Table 10). In subsequent survey years, the timing of monitoring aligned better with the cycle of breeding for

each of the three species and consequently at the end of monitoring, most nests had chicks at the runner stage or older present (Table 9).

Table 7 Summary statistics of the hatching, mortality and survival rates of direct records where each egg was monitored continuously through hatching to its eventual fate. Where means are presented the standard deviation follows in brackets.

Survey year	2015-2016			2016-2017			2017-2018		
Species	AWI	RSB	SNI	AWI	RSB	SNI	AWI	RSB	SNI
Mean clutch size (±SD)	2.12 (1.17)	0.87 (1.36)	2 (0.72)	2.17 (1.39)	3.03 (1.59)	1.60 (1.17)	2.65 (0.95)	2.76 (0.83)	-
Total no. of eggs observed laid	218	20	48	275	115	134	143	58	-
Mean chicks hatched (±SD)	0.64 (1.17)	0.48 (1.08)	1 (1.02)	1.18 (1.37)	1.92 (1.89)	0.39 (0.78)	1.46 (1.24)	1.81 (1.44)	-
Total # chicks with direct record	66	11	24	150	73	33	70	35	-
Mean egg loss (±SD)	0.72 (1.03)	0.04 (0.21)	0.13 (0.34)	0.50 (0.80)	0.35 (0.59)	1.04 (1.17)	0.53 (0.68)	0.38 (0.65)	-
Total chick loss	7	1	2	16	4	7	16	6	-
Mean # chicks died per clutch (±SD)	0.07 (0.32)	0.04 (0.21)	0.08 (0.28)	0.28 (0.58)	0.20 (0.41)	0.23 (0.52)	0.53 (0.73)	0.46 (0.66)	-
Total survived	59	10	22	134	68	26	54	29	-
Direct survival estimate (% eggs laid)	27%	50%	45.8%	48.7%	59.1%	19.4%	37.8%	50%	-
Direct survival estimate (% chicks hatched)	80.4%	90.9%	91.7%	89.3%	93.2%	78.8%	77.1%	82.9%	-

Table 8 **Overall** summary statistics of the hatching, mortality and survival rates of all records (**direct plus indirect records**). Where means are presented the standard deviation follows in brackets.

Survey year	2015-2016			2016-2017			2017-2018		
Species	AWI	RSB	SNI	AWI	RSB	SNI	AWI	RSB	SNI
Mean clutch size	2.12 (1.17)	0.91 (1.38)	2 (0.72)	2.17 (1.39)	3.03 (1.59)	1.60 (1.17)	2.65 (0.95)	2.76 (0.83)	-
Total chicks observed	66	11	24	226	89	81	79	38	-

Overall mean chick death	0.07 (0.32)	0.04 (0.21)	0.08 (0.28)	0.19 (0.47)	0.16 (0.44)	0.23 (0.52)	0.33 (0.61)	0.33 (0.58)	-
All surviving chicks at end of monitoring	59	10	22	203	86	62	61	31	-
Overall survival estimate	89.4%	90.9%	91.7%	89.8%	96.6%	76.5%	77.2%	81.6%	-

436

437 Table 9 The number of nests with chicks present at the end of monitoring and the maximum age
438 class of those chicks. T=Total number of chicks.

	2015-2016						**2016-2017**						**2017-2018**					
	AWI		RSB		SNI		AWI		RSB		SNI		AWI		RSB		SNI	
	%	T	%	T	%	T	%	T	%	T	%	T	%	T	%	T	%	T
Hatchlings	9	2	0	0	46	6	6	5	0	0	8	2	16	5	7	1	-	-
Squirters	4	1	0	0	0	0	7	6	9	2	16	4	3	1	0	0	-	-
Runners	61	14	75	3	54	7	38	32	9	2	32	8	16	5	7	1	-	-
Flappers	17	4	25	1	0	0	28	24	9	2	16	4	45	14	7	1	-	-
Flyers	9	2	0	0	0	0	21	18	73	15	28	7	20	6	79	11	-	-
Total		**23**		**4**		**13**		**85**		**21**		**25**		**31**		**14**	-	-

439

440 Table 10 The state of the nest at the end of each monitoring season (number of nests)

	2015-2016			**2016-2017**			**2017-2018**		
Nest state	AWI	RSB	SNI	AWI	RSB	SNI	AWI	RSB	SNI
Abandoned	8	1	0	1	0	0	2	1	-
Abandoned at egg stage	34	0	1	23	9	23	2	0	-
Abandoned at chick stage	1	0	0	3	0	3	0	0	-
Eggs	32	10	10	14	2	8	5	1	-
Depredated	0	0	0	1	0	0	13	6	-
Empty pre-breeding	0	15	0	0	0	0	0	0	-
Empty post-breeding	2	0	0	9	14	1	4	10	-
Chicks (all ages)	23	4	13	76	13	38	26	3	-

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Drivers of egg and chick mortality

Drivers of egg and chick mortality captured by monitoring cameras varied between years and species. Some mortalities could not be definitively assigned drivers based on the information available (16% overall). These included instances where the camera field of view changed or stopped working temporarily and when the view was restored the eggs or chicks were no longer present. Overall, other than the unknown causes of mortality, the greatest driver of mortality in the 2015-16 pilot study was predation of eggs (

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Table 11). In the 2016-17 survey year, the leading cause of mortality was abandonment of eggs. In the 2017-18 survey year the leading cause of mortality was again predation of eggs.

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Table 11 Summary of the primary cause of mortality for egg and chick loss each year. Total percentages are of all eggs laid or chicks hatched.

		2015-2016			2016-2017			2017-2018			TOTAL
		AWI	RSB	SNI	AWI	RSB	SNI	AWI	RSB	SNI	**All sp**
Unknown	Eggs	4	1	0	33	9	0	20	5	-	**108**
	Chicks	2	0	0	15	6	0	7	6	-	(16%)
Predation	Eggs	37	0	1	1	12	1	12	7	-	**81**
	Chicks	3	0	0	2	0	0	4	1	-	(12%)
Abandonment	Eggs	7	0	0	26	0	0	6	5	-	**45**
	Chicks	0	0	0	1	0	0	0	0	-	(7%)
Anthropogenic disturbance	Eggs	0	0	0	16	12	1	7	6	-	**43**
	Chicks	0	0	0	1	0	0	0	0	-	(6%)
Rejection	Eggs	15	0	1	0	0	1	0	0	-	**18**
	Chicks	1	0	0	0	0	0	0	0	-	(3%)
Conspecific damage	Eggs	0	0	0	4	4	0	3	0	-	**11**
	Chicks	0	0	0	0	0	0	0	0	-	(2%)
Competition	Eggs	7	0	0	0	0	0	5	0	-	**12**
	Chicks	0	0	0	0	0	0	0	0	-	(2%)
Starvation	Eggs	-	-	-	-	-	-	-	-	-	**9**
	Chicks	0	0	1	5	0	1	2	0	-	(1%)
Other	Eggs	1	0	0	3	0	0	1	0	-	**9**
	Chicks	0	0	0	0	0	0	4	0	-	(1%)
TOTAL	**Eggs**	**71** (33%)	**1** (5%)	**3** (6%)	**83** (30%)	**37** (32%)	**3** (2%)	**56** (39%)	**23** (40%)	-	
	Chicks	**6** (9%)	**0**	**1** (4%)	**24** (11%)	**6** (7%)	**1** (1%)	**18** (23%)	**7** (18%)	-	

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Predation

More eggs than chicks were lost to predation (Table 12). Five native bird species were documented by cameras eating ibis or spoonbill eggs or chicks: swamp harriers (*Circus approximans*); white-belled sea-eagles (*Haliaeetus leucogaster*); whistling kites (*Haliastur sphenurus*); purple swamp hen (*Porphyrio porphyrio*); and Australian raven (*Corvus coronoides*) (Table).

Australian white ibis were documented via images as being responsible for egg loss in adjacent nests. Two mammal species were documented via images and direct observations in nesting areas – the feral fox (*Vulpes vulpes*) and the feral pig (*Sus scrofa*). Other predators seen in or adjacent to nesting areas included wedge-tailed eagles (*Aquila audax*); peregrine falcons (*Falco peregrinus*); brown goshawks (*Accipiter fasciatus*); nankeen kestrels (*Falco cenchroides*); and black kites (*Milvus migrans*). It is likely that feral cats and owls are also present.

Swamp harriers (adults and juveniles) were responsible for the greatest loss over all three years. Some species, such as white-bellied sea-eagles, were observed returning repeatedly to a location and consuming eggs from multiple nests. Purple swamphens were the primary predator of Australian white ibis over all three years.

Predation events correlated with known peak activity times for many birds. Over the three seasons of study, peak predation activity occurred in the early morning (between 6 and 8am) and late afternoon/evening (4 – 8pm; Figure 5).

Since all predators captured by cameras as taking eggs or chicks were avian and water level is irrelevant to their access to nests, no relationship was found between water depth and predation impacts. This is reflected in Figure 6 where it appears that the frequency of avian predation is better correlated with timing in the breeding cycle than with water depth, with high impacts on eggs and low impacts on chicks. However when water levels were low, land-based mammalian predators including pigs, foxes and cats (all feral species) were observed more frequently in images around nests. These species were not directly captured in images depredating any eggs or chicks in Reed Beds Swamp and consequently data were insufficient to quantify any relationship between water depth and the direct impacts of these species. Despite this, their presence in the colony indicates some level of mammalian predation or disturbance pressure; feral pigs were documented by cameras at a separate site across the river eating eggs from Australian white ibis nests during the late 2017-2018 breeding season. Rakali (water rats; *Hydromys chrysogaster*) were also present in the colony and have been known to consume waterbird eggs and chicks.

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Table 12 The number and proportion of chicks and eggs killed by observed predator species.

			2015-2016			2016-2017			2017-2018			
Predator species	**Predator scientific name**		**AWI**	**RSB**	**SNI**	**AWI**	**RSB**	**SNI**	**AWI**	**RSB**	**SNI**	**Total**
Purple Swamp Hen (adult)	*Porphyrio porphyrio*	Eggs	2	0	0	2	0	0	4	0	-	8
		Chicks	0	0	0	2	0	1	0	0	-	3
Swamp Harrier (adult)	*Circus approximans*	Eggs	8	0	1	0	9	0	9	7	-	34
		Chicks	0	0	0	0	0	1	2	1	-	4
Swamp Harrier (juvenile)	*Circus approximans*	Eggs	0	0	0	0	3	0	0	0	-	3
		Chicks	0	0	0	0	0	0	0	0	-	0
Whistling Kite (adult)	*Haliastur sphenurus*	Eggs	0	0	0	1	0	0	0	0	-	1
		Chicks	0	0	0	0	0	0	0	0	-	0
White-bellied Sea Eagle (juvenile)	*Haliaeetus leucogaster*	Eggs	10	0	0	0	0	0	0	0	-	10
		Chicks	0	0	0	0	0	0	0	0	-	0
Australian Raven	*Corvus coronoides*	Eggs	3	0	0	0	0	0	0	0	-	3
		Chicks	0	0	0	0	0	0	0	0	-	0
Australian White Ibis	*Threskiornis moluccus*	Eggs	0	0	0	5	0	0	0	0	-	5
		Chicks	0	0	0	0	0	0	0	0	-	0
Unknown predator	*NA*	Eggs	3	0	0	0	0	0	0	0	-	3
		Chicks	0	0	0	0	0	0	1	0	-	1
Grand Total			**26**	**0**	**1**	**10**	**12**	**2**	**16**	**8**	**-**	

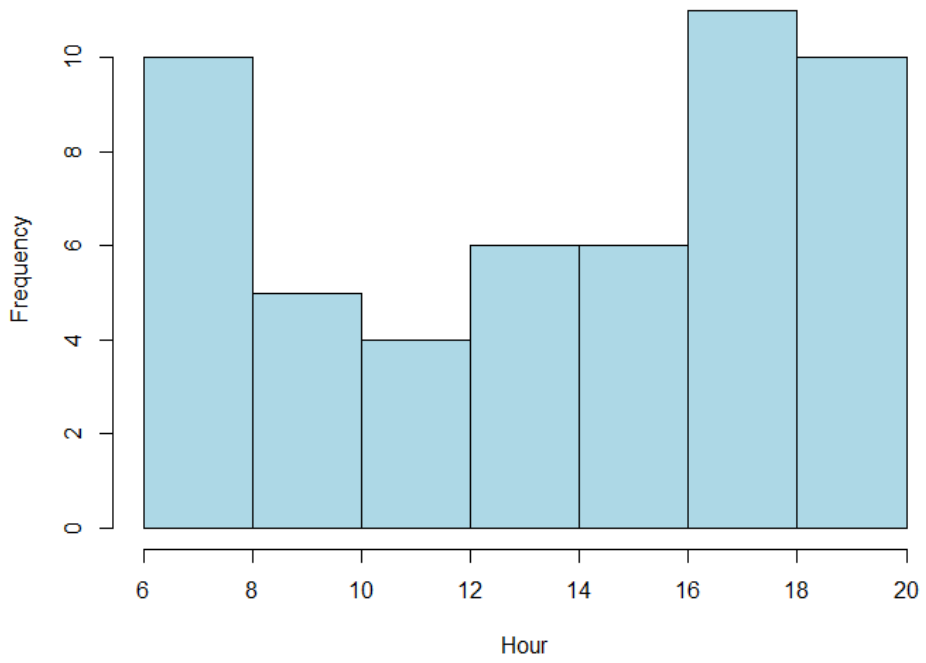


Figure 5. The frequency of predation events per hour (in 24hour format) for all predators and all survey years.

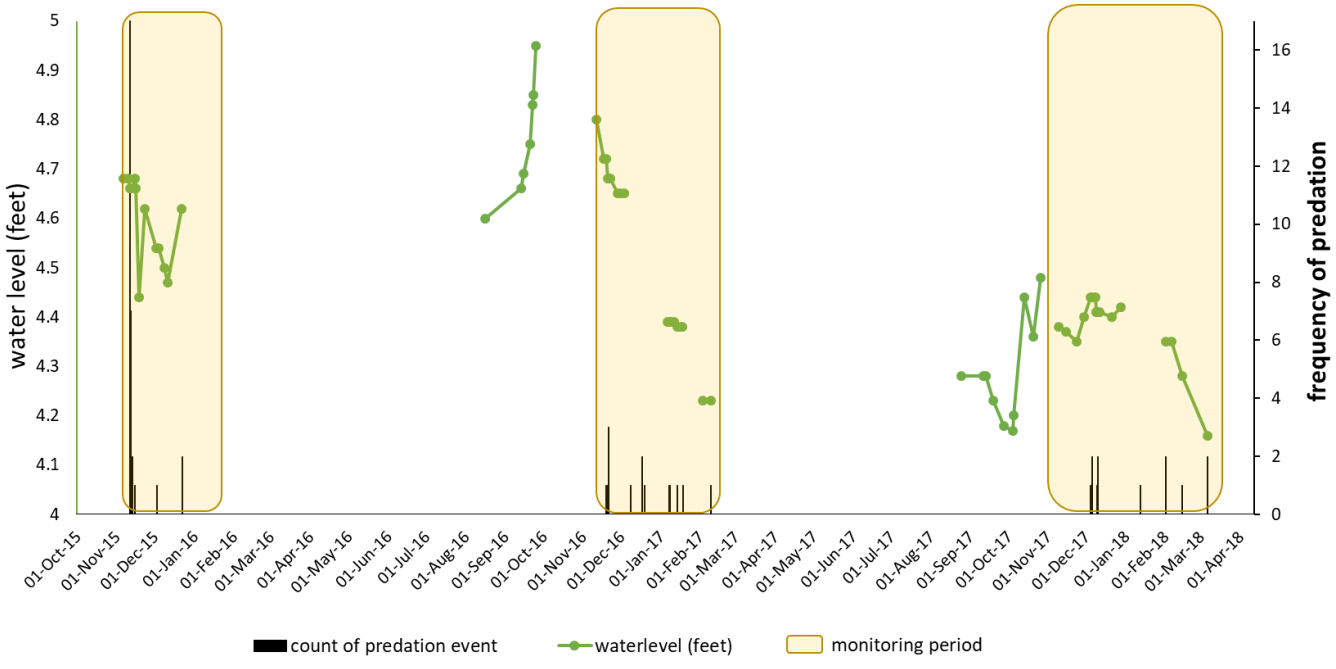


Figure 6. Frequency of predation related to water depth. Water level was measured at Reed Beds, Millewa by the permanent Gulpa Regulator gauge.

Abandonment

Abandonment of eggs occurred far more frequently than of chicks; indeed chick abandonment only occurred once throughout the study. Abandonment was the leading known primary cause of egg mortality for Australian white ibis but was not observed for either of the other two species until 2017-2018.

The cause of abandonment was often difficult to ascertain, but abandonment was sometimes a secondary cause of mortality after an anthropogenic disturbance, or after clear egg rejection or egg loss due to some factor not attributable to predation, conspecifics or human disturbance (e.g. sometimes eggs would 'sink' into the nest). Abandonment could not be quantitatively correlated with water depth at the nests, however there appears to be an association between a decrease in water depth and some nest abandonment events (Figure 7). It is likely that there are interactions between the effects of water depth change, breeding stage, predation, disturbance and other factors. For example, a fall in water level during the egg stage may be more likely to trigger abandonment than a fall in water level during the chick stage. Adding predation or disturbance at the same time may increase the chances of abandonment (Figure 8). Anthropogenic disturbance and subsequent abandonment was one of the leading causes of egg loss during the 2016-17 survey year but was not as important in the other two years and was generally low over the whole study period. Occasionally parents of one nest would fight with other parents and deliberately prevent them from returning to their own eggs/chicks, thus leading secondarily to abandonment.

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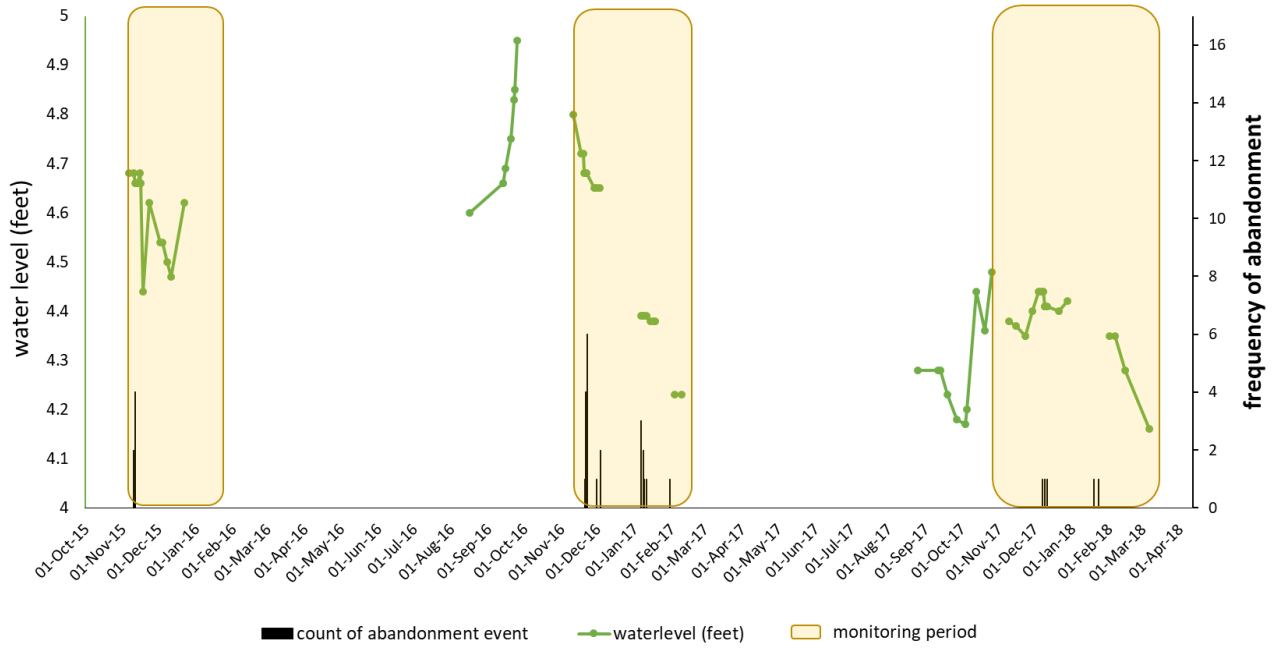


Figure 7 Frequency of abandonment related to water depth. Water level was measured at Reed Beds, Millewa by the permanent Gulpa Regulator gauge.

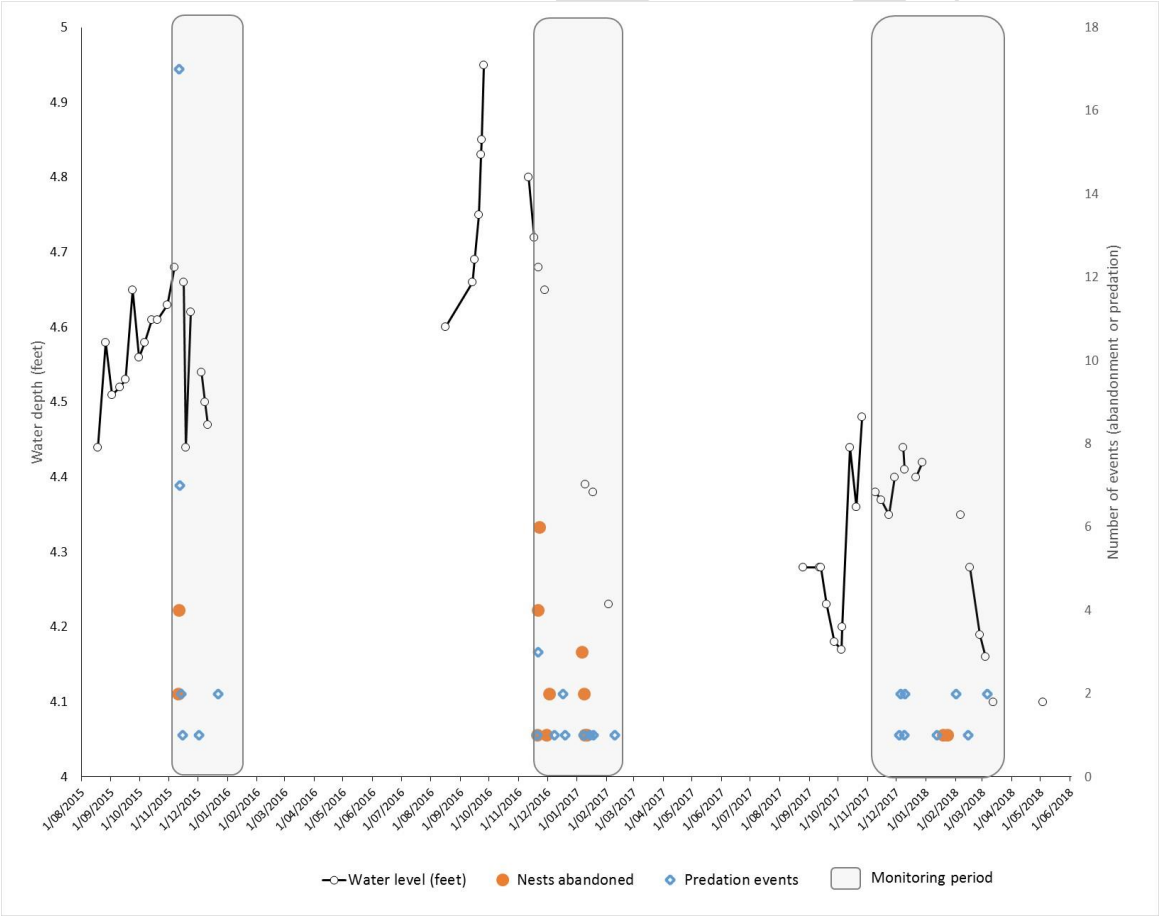


Figure 8 Water depth at the Gulpa Gauge and the number of nest abandonment and predation events at monitored nests in the Reed Beds wetland over time.

Egg damage, rejection and failure to hatch

Damage from conspecifics such as parents or other waterbirds in the vicinity was a relatively rare cause of death (2% of egg loss) but did occur. Events observed that fall into this category include accidental damage to eggs by parents, i.e. squashing eggs, knocking them off the nest into the water and trampling. We also observed some adults deliberately destroying the eggs of other nests or dropping them into the water. Egg rejection occurred frequently among Australian White Ibis in the pilot study but was a rarely recorded event for all species in the following years. Rejection was classified as an event where a parent deliberately destroyed the egg. We observed several instances where parents would kick eggs out of the nest into the water, often only hours after the eggs had been laid. Where it was observed that eggs failed to hatch but others in the nest did, those eggs were recorded as 'failed to hatch' and were potentially infertile or died. This category accounted for only 1% of all chick and egg loss; however this may be an underestimate. Adults may be able to detect when eggs are infertile or dead and therefore what appeared as abandonment or rejection could actually have been a response to infertility.

Starvation

Observed starvation of chicks was rare, with only nine chicks suspected to have starved, seven of which were Australian white ibis and two straw-necked ibis. Royal Spoonbills were not recorded to have any loss from starvation.

Nest aspect

Nests with a north-west or south-east aspect had the highest average egg losses over all species, with the north-north-east aspect also being associated with relatively high egg losses for ibis species (Figure 9). The number of surviving chicks was significantly lower in nests with a western aspect (0.000257; Figure 10). Nests with a south-west aspect also had a relatively low mean number of surviving chicks. The mean number of surviving chicks was highest in nests with a NNE, S or E aspect (Figure 10).

Nesting group shape

There were no statistically significant differences in total egg loss between nesting group shapes. However mean egg loss was generally highest in pockets and open fringes and lowest in closed archipelagos and amphitheatre (Figure 11). Closed archipelago nesting groups had a significantly higher number of surviving chicks ($Pr(>|z|)=0.0227$ for just ibis species, 0.0000282 for all species; Figure 12). Open archipelago groups also had higher chick survival than other shapes, but this was only statistically significant for all species (0.00205). Results for the effect of nesting group shape

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on egg loss and chick survival were biased by royal spoonbill nests, which were more successful than ibis nests and mostly occurred in closed or open archipelagos.

Nest exposure to open water

Nests with the lowest exposure to open water exposure had the lowest mean egg loss, while those with the highest open water exposure had the highest mean egg loss (Figure 13). However there were no statistically significant differences. Results for the effect of nesting group shape on chick survival were again biased by royal spoonbill nests, which were more successful than ibis nests and mostly occurred facing open water. Results were not statistically significant. When all species are analysed together, the highest open water exposure is associated with the highest mean number of surviving chicks (Figure 14). However when only ibis species are analysed, the average number of surviving chicks is lowest with highest water exposure and highest with lowest water exposure.

There was no significant relationship or pattern in response to nest height above water for either total egg loss or number of surviving chicks.

Nest vegetation wall protection

Across all species, nests with the lowest vegetation wall protection had the highest mean egg loss (Figure 15). The number of surviving chicks was significantly higher in nests with the highest vegetation wall protection (0.00183; Figure 16).

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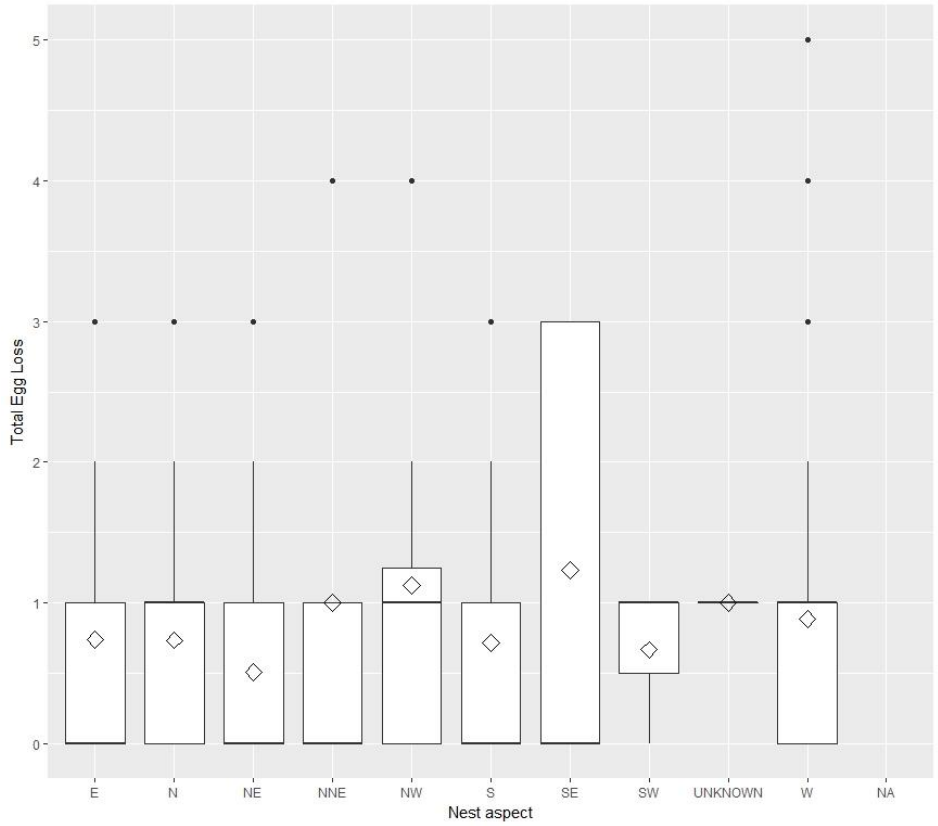


Figure 9

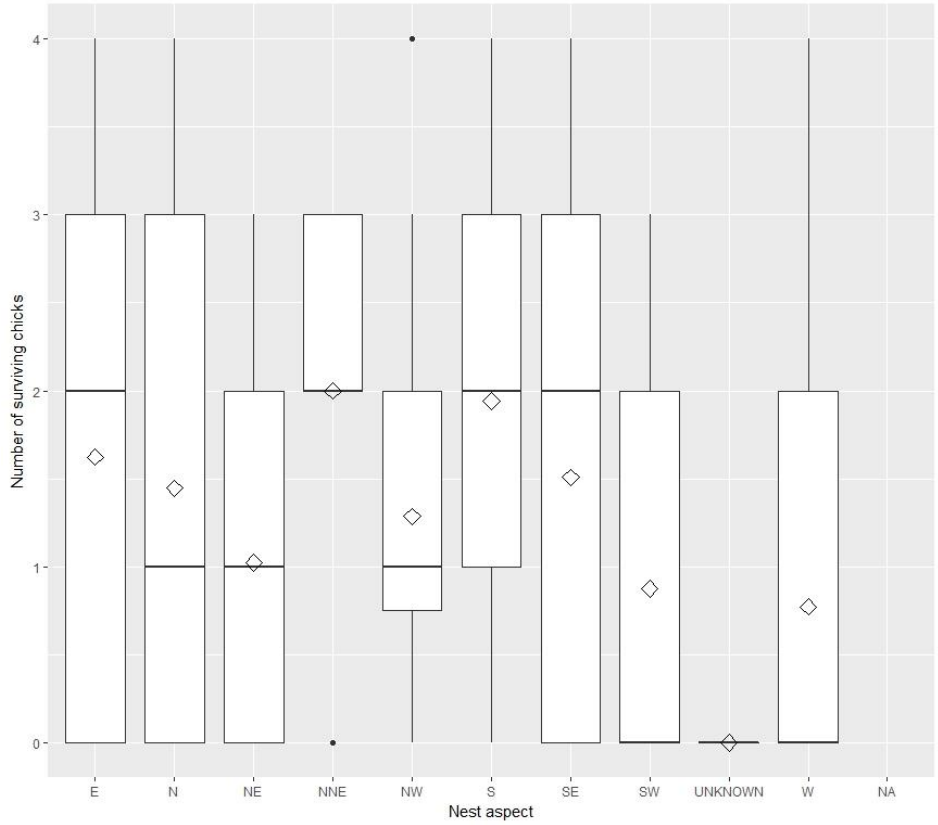


Figure 10

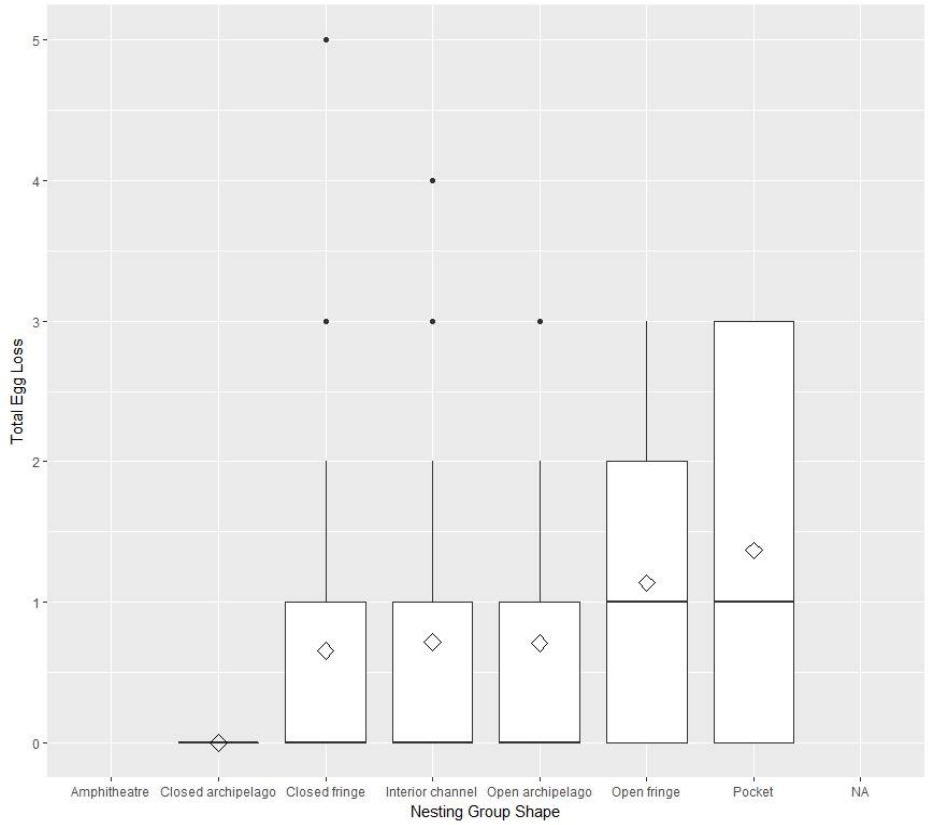


Figure 11

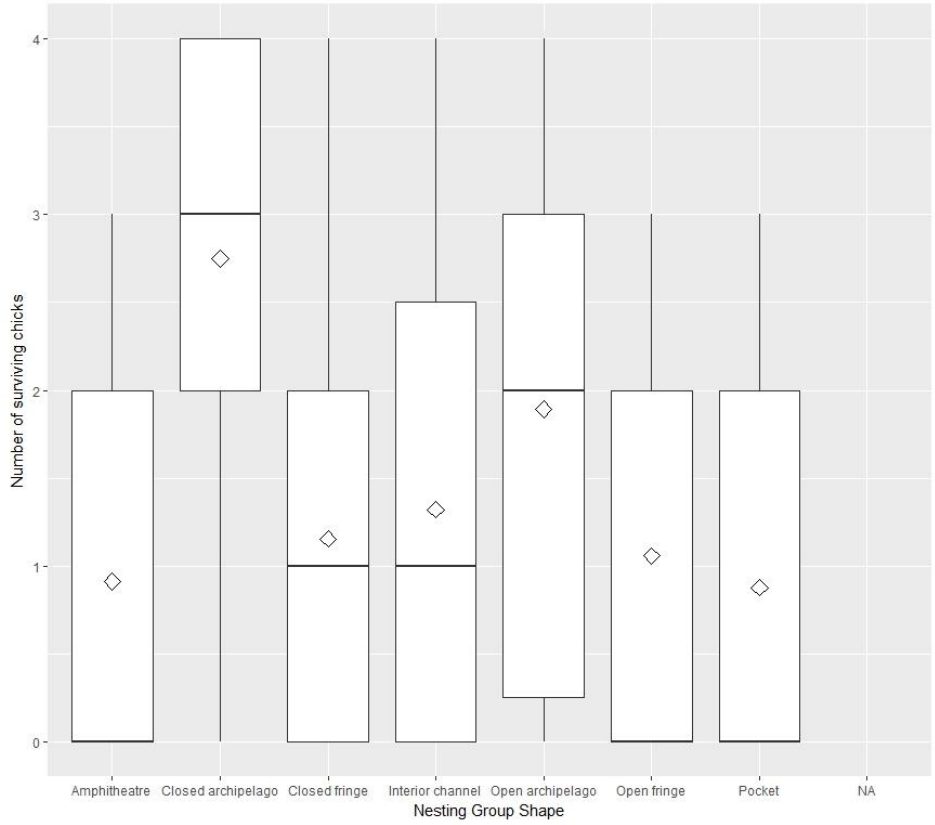


Figure 12

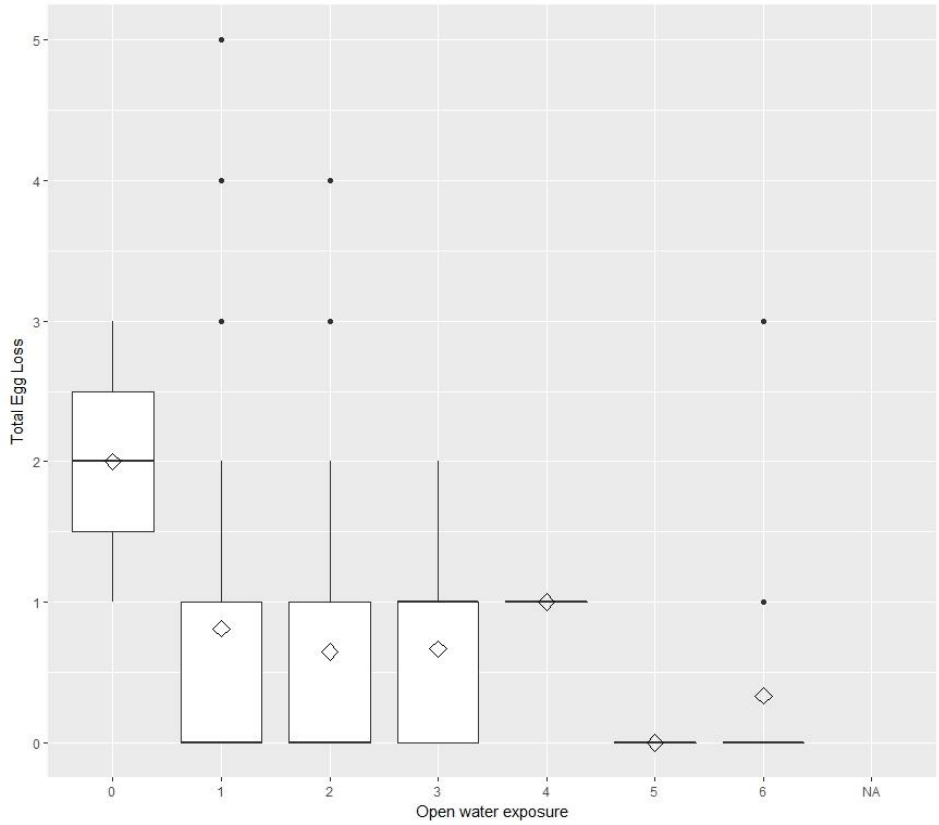


Figure 13

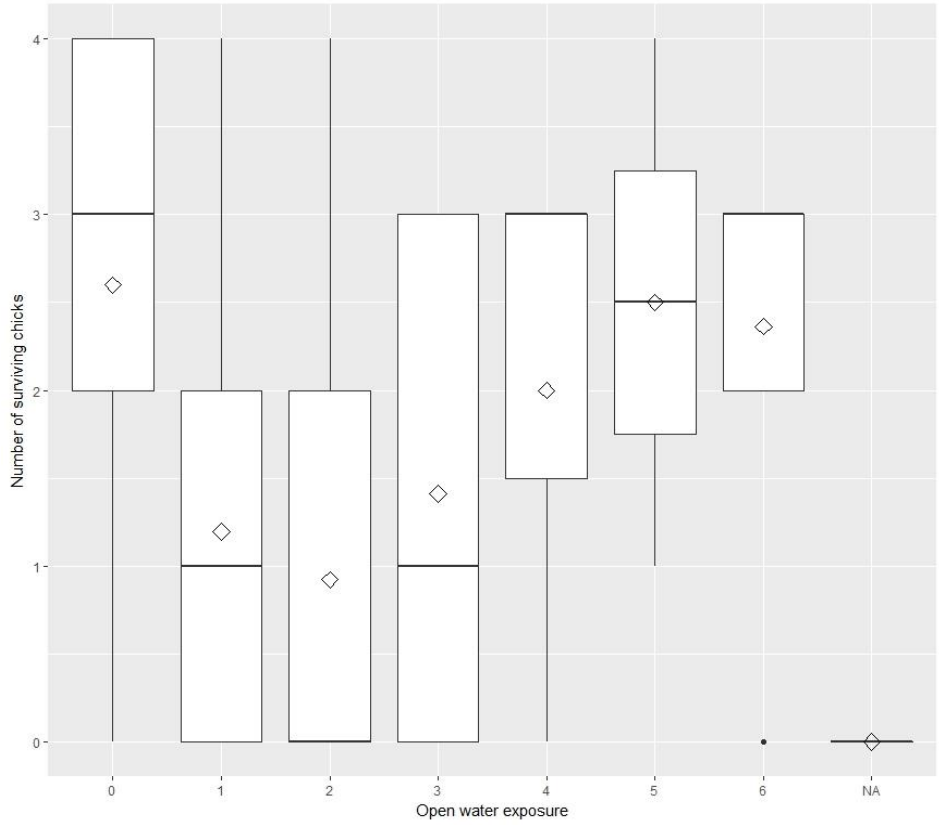


Figure 14

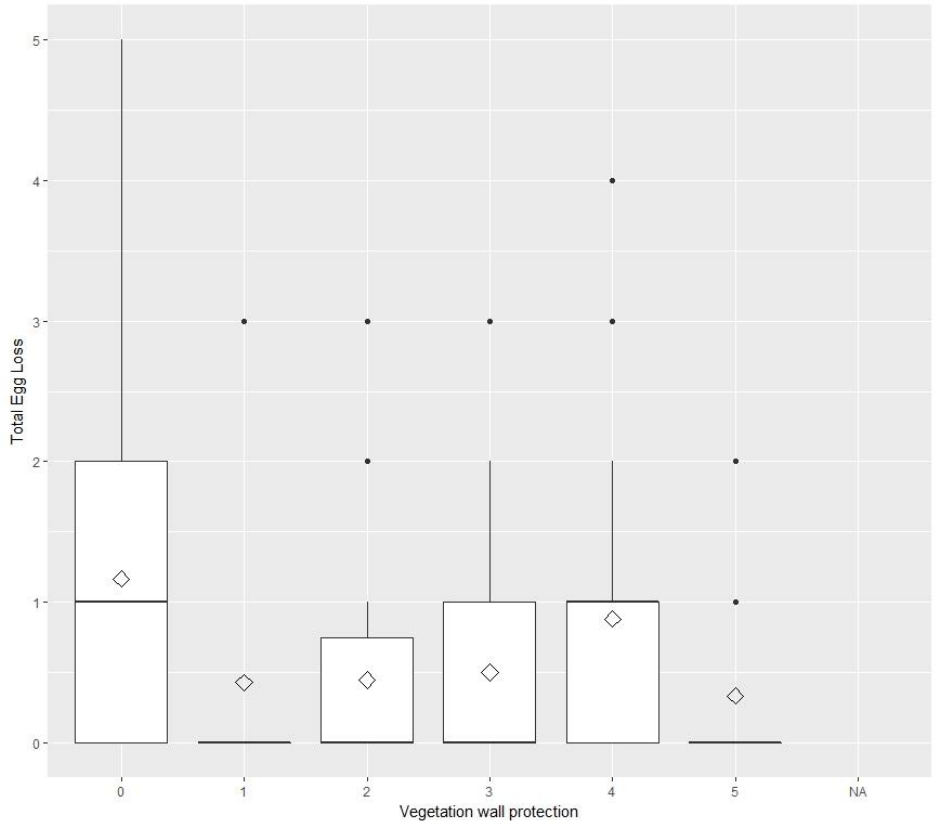


Figure 15

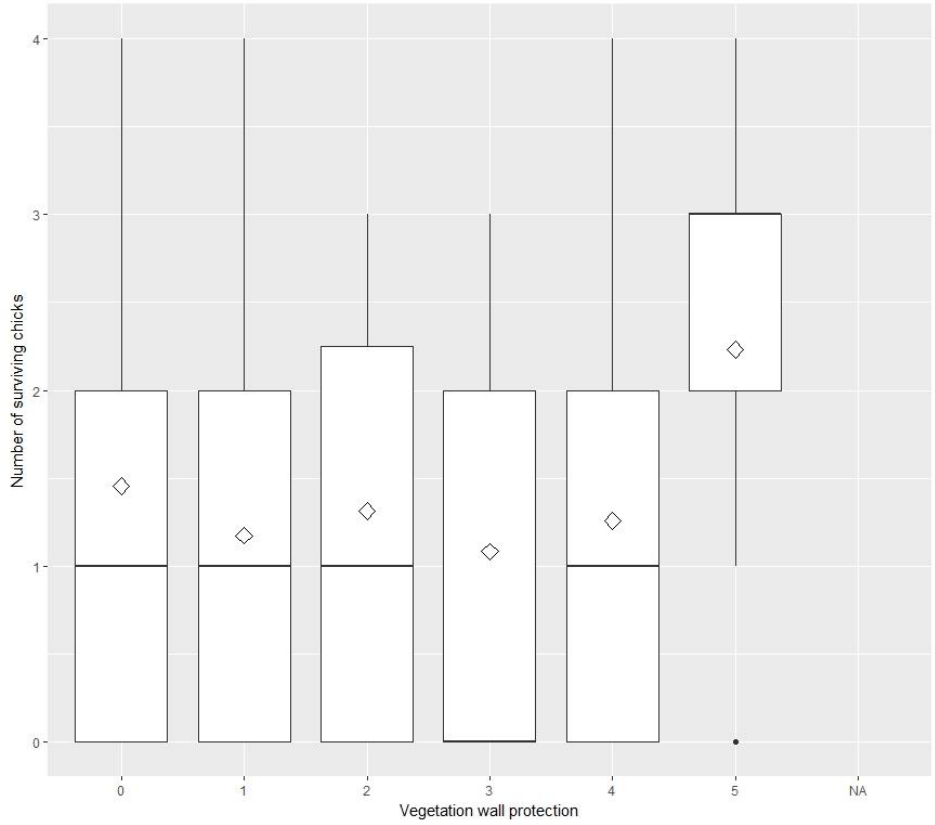


Figure 16

Discussion

Clutch sizes

Over all three breeding seasons the mean clutch size for Australian white ibis in the study wetland was within the range of means in the literature (Table), but lower than the overall mean, median and mode. Minimum and maximum clutch sizes of AWI were similar to literature values, where minimum clutch sizes range from 1-3 eggs with a mode of 1 egg (n= 20 rates) and maximum clutch sizes range from 3-6 eggs with a mode of 5 eggs (n= 25 rates).

The mean clutch size for royal spoonbills during 2015-2016 was the lowest recorded in the literature, while 2016-2017 and 2017-2018 means were within the range of literature means. The reason for such a low clutch size in that year is unclear. The minimum clutch size was lower than recorded minimums in the literature however the maximum was similar. Minimum egg clutch sizes in the literature range from 2-3 eggs, with a mode of 2 eggs (n=8 rates) and maximum clutch sizes range from 3-7 eggs with a mode of 4 eggs (n=10 rates).

The mean and maximum clutch sizes for straw-necked ibis were within the range of means and maximums presented in the literature; however minimum clutch size for 2016-2017 was lower than the range of minimums in the literature. Minimum clutch sizes in the literature range from 1-3 eggs with a mode of 1 egg (n= 12 rates), while maximum clutch sizes range from 3-7 eggs with a mode of 5 eggs (n= 16 rates).

Table 13 Summary statistics for clutch sizes described in the literature

Species	Range of means	Mean	Mode	Median	No. of rates	No. of clutches	No. of sites	References
Australian white ibis	2-3.5	2.71	3	2.69	37	7-1004	20	Magrath 1992; Lowe 1984; Vestjens 1984; Kentish 1999; Smith 2009; Corben et al. 2006; Beilharz 1988; Murray et al. 2006
Royal spoonbill	2.58-4.25	3.26	3	3.06	10	3-40	6	Brandis 2011; Magrath 1992; Vestjens 1977; Brandis 2011; Lowe 1984
Straw-necked ibis	0.65-3	2.15	1.95	2.17	32	12-781	14	Brandis et al. 2011; Brandis et al. 2014; Magrath 1992; Vestjens 1977; Kingsford et al. 2003; Merritt et al. 2016; Kingsford et al. 2008; Brandis 2010; Lowe 1984;

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Egg and chick mortality rates

Quantitative measurements of ibis and spoonbill egg and chick mortality or survival rates are uncommon in Australia. Rates are generally more readily available for species that nest in urban environments, including the Australian white ibis. Urban egg and chick mortality estimates for this species are available from Healesville Sanctuary VIC (Lowe 1984;), Brisbane Waters, Woy Woy, NSW (Smith 2009); and Lake Gillawarna, Bankstown NSW (Smith et al. 2013). Non-urban studies with published rates include those from Green's Swamp, Clump Swamp, Gemmell's Swamp, Heifer Swamp, Coolart Lagoon, and Rhyll Swamp, Phillip Island, VIC (Lowe 1984). In these studies, the percentage of Australian white ibis eggs producing hatchlings ranges from 0 to 68%, with a mean of 49% (9 rates). This corresponds closely to the mean hatching rate during the 2016-2017 season described here (48.7%). Lowe (1984) found that less than 60% of AWI eggs produced hatchlings across eight breeding colonies in southern Victoria. The mean number of chicks hatched per clutch also varies by location, from 0.7 (SD=1.2) (Smith et al. 2013); to 0.97 (SD=0.04) at Brisbane Waters, Woy Woy NSW (Smith 2009); and 1.29 chicks (SD=0.01, n=216 clutches) at Lake Gillawarna, Bankstown NSW. No published rates describing the mean number of chicks hatched per clutch (hatching rate) were found in the literature for royal spoonbills or straw-necked ibis.

Chick survival rates reported in the literature are highly variable within and between studies (Table), corresponding with the high variability recorded during the present study.

The mean percentage of chicks fledging from Australian white ibis eggs in the current study (38%) was similar to a mean value derived from the available literature (41%), whereas the mean percentage of chicks fledging from royal spoonbill eggs (53%) was higher than the mean literature-based rate (44%). The mean percentage of chicks fledging from straw-necked ibis eggs in the current study was considerably lower (33%) than the mean derived from the literature (72%). It is not yet clear why this is the case.

The mean percentages of chicks fledging from hatchlings (chick survival rate) were higher for Australian white ibis and straw-necked ibis chicks in the current study (82% and 85% respectively) than those derived from the literature (68% and 55% respectively). Royal spoonbill data were not available from the literature, but in the current study the royal spoonbill chick survival rate from hatchling stage was also high, at 89% - and higher than either of the two ibis species.

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Table 14 Table Chick survival rates derived from the published literature

Species	**Number of fledged chicks per clutch**		**% of chicks fledging from eggs**		**% of chicks fledging from hatchlings**		**References**
	Mean	Range	Mean	Range	Mean	Range	
Australian white ibis	1.14	0.36 - 2.64	40.58% (10 rates)	25% - 100%	67.83% (4 rates)	60% – 79%	Smith et al. 2013, Corben 2006; Smith 2009; Smith 2013; Marchant et al. 1990; Kentish 1994; Beilharz 1988
Royal spoonbill	1.03	0.33 - 3.18	44% (8 rates)	27% - 79%	No rates available	No rates available	Brandis et al. 2011; Lowe 1984; Magrath 1992; Butterfield 2009; Hoyo et al. 1992
Straw-necked ibis	0.65	0.6 – 0.7	72% (7 rates)	18% - 95%	55% (2 rates)	17% - 94%	Lowe 1984; Merritt et al. 2016; Brandis et al. 2011; Brandis 2010; Brandis et al. 2014

Drivers of egg and chick mortality

Threskiornithidae (ibis and spoonbill) nest success worldwide is influenced by a wide range of factors or drivers (

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Table 15). These drivers are likely to vary by site, species and season. Those most commonly observed and most detectable by monitoring cameras are predation, abandonment, nest exposure (and weather), competition and conspecific damage, and starvation. There are also several possible drivers of egg and chick mortality occurring at breeding sites that are difficult to quantify visually through surveys or camera monitoring. These include toxins, diseases, parasites, and interactions between these and other pressures. Evaluating the roles of these factors in affecting egg and chick survival will require testing of swabs, blood or other samples collected in the field.

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Table 15 Major causes of mortality for international species of the *Platalea*, *Plegadis* and *Threskiornis* genera documented in the published scientific literature.

Genera	Egg mortality drivers	Chick mortality drivers
***Platalea*:** *Platalea ajaja* (roseate spoonbill), *P. alba* (African spoonbill), *P. leucorodia* (Eurasian spoonbill), *P. minor* (black-faced spoonbill)	abandonment (Allen 1942; Lok *et al.* 2017; Spanoghe *et al.* 2012); broken (Whitelaw 1968); competition between nests(Kwon *et al.* 2012); disturbance by predator (Overdijk 2004; Spanoghe *et al.* 2012); exposure to elements (Allen 1942; Lok *et al.* 2017; Triplet *et al.* 2008); disturbance by humans (Collar *et al.* 2001; Hodgson and Paul 2013; Lorenz *et al.* 2002; Society *et al.* 1994); researcher disturbance (Lorenz *et al.* 2002); hunted (Guo-An *et al.* 2005; Hancock *et al.* 2010); falling from nest (Kwon *et al.* 2012); parents breaking pair-bond (Kwon *et al.* 2012); predation (Allen 1942; Chong and Pak 2000; Hancock *et al.* 2010; Society *et al.* 1994; Spanoghe *et al.* 2012; Triplet *et al.* 2008); pesticide poisoning (Lorenz *et al.* 2002); infertile or unviable (Guo-An *et al.* 2005; White *et al.* 1982; Whitelaw 1968; Wylie 1982); changes in water levels at nesting site (Lok 2013; Triplet *et al.* 2008).	abandonment (Chastant and Gawlik 2015; Spanoghe *et al.* 2012); infanticide(Aguilera 1990); competition between nests (Andrianarimisa and Razafimanjato 2012; Kwon *et al.* 2012; Whitelaw 1968); disease (Lorenz *et al.* 2002; Society *et al.* 1994); disturbance by predator (Overdijk 2004; Spanoghe *et al.* 2012); exposure to elements (Allen 1942; Cramp and Simmons 1983; Lok *et al.* 2014; Lok *et al.* 2017; Lorenz *et al.* 2002; Society *et al.* 1994; Triplet *et al.* 2008); human disturbance (Hodgson and Paul 2013; Lorenz *et al.* 2002; Society *et al.* 1994); researcher disturbance (Lorenz *et al.* 2002); hunted (Allen 1942; Hancock *et al.* 2010); falling from nest (Chong and Pak 2000; Kwon *et al.* 2012); trampled by siblings (Whitelaw 1968); entangled (Whitelaw 1968); parents breaking pair-bond (Kwon *et al.* 2012); parasites (Allen 1942; Society *et al.* 1994); predation (Allen 1942; Hancock *et al.* 2010; Society *et al.* 1994; Spalding and Forrester 1991; Spanoghe *et al.* 2012; Triplet *et al.* 2008; White *et al.* 1982); starvation (Cramp and Simmons 1983; Lorenz *et al.* 2002; Society *et al.* 1994; Sweeney 1993); heavy metal poisoning (Lorenz *et al.* 2002); pesticide poisoning (Lorenz *et al.* 2002); changes in water levels at nesting site (Hoyo *et al.* 1992; Kwon *et al.* 2012; Lok 2013; Lorenz 2014; Triplet *et al.* 2008; Vespremeanu 1968).
Threskiornis: *Threskiornis molucca* (sacred ibis), *T. melanocephalus* (black-headed ibis), *T. bernieri* (Madagascar sacred ibis)	abandonment (Parsons 1977; Urban 1974); broken (Devkar *et al.* 2006); exposure to elements (Brown *et al.* 1982; Cramp and Simmons 1983); human disturbance (Benson and Penny 1971; Hancock *et al.* 2010; Penny 1974; Urban 1974); hunted (Andrianarimisa and Razafimanjato 2012; Hancock *et al.* 2010; Hoyo *et al.* 1992; Iqbal and Hasudungan 2012; Perschke 2006); falling from nest (Benson and Penny 1971; Sclater 1878); predation (Ali and Ripley 1978; Brown *et al.* 1982; Dayananda and Hosetti 2008; Hancock *et al.* 2010; Hoyo *et al.* 1992; Manry 1978; Parsons 1977; Theba and Vyas 2013; Urban 1974), unviable (Kopij 1999); changes in water levels at nesting site (Brown *et al.* 1982).	siblicide (Kopij 1999); exposure to elements (Brown *et al.* 1982; Cramp and Simmons 1983; Hancock *et al.* 2010; Kopij 1999; Narwade *et al.* 2014; Urban 1974); human disturbance (Hancock *et al.* 2010; Penny 1974; Urban 1974); hunted (Andrianarimisa and Razafimanjato 2012; Hancock *et al.* 2010; Hoyo *et al.* 1992; Iqbal and Hasudungan 2012); falling from nest (Brown *et al.* 1982; Kopij 1999; Parsons 1977; Urban 1974); entangled (Brown *et al.* 1982); predation (Ali and Ripley 1978; Dayananda and Hosetti 2008; Devkar *et al.* 2006; Hancock *et al.* 2010; Hoyo *et al.* 1992; Manry 1978; Parsons 1977; Smits *et al.* 2010; Urban 1974); starvation (Brown *et al.* 1982; Manry 1978).
Plegadis: *Plegadis chihi* (white-faced ibis), *P. ridgwayi* (Puna ibis)	abandonment (Capen 1977; Capen 1978; Garrett 1996; Hothem *et al.* 2006; Kaneko 1972; King *et al.* 1980; Taft *et al.* 2000; Zaun *et al.* 2003); broken(Capen 1977; Capen 1978; Henny *et al.* 1985; Kaneko 1972; King *et al.* 1980; Steele 1980); exposure to elements (Capen 1977; Hancock *et al.* 2010; Ryder 1967; Ryder 1994); human disturbance (Capen 1977; Ryder 1994); researcher disturbance (Belknap 1957; Capen 1978; Steele 1980); hunted (Coke *et al.* 2004; Hancock *et al.* 2010); predation (Capen 1977; Capen 1978; Dark-Smiley and Keinath 2003; Garrett 1996; Hancock *et al.* 2010; Hothem *et al.* 2006; Ryder 1994; Stahlecker 1989; Steele 1980; Taft *et al.* 2000; Zaun *et al.* 2003); pesticide poisoning (Garrett 1996; Hancock *et al.* 2010; Henny *et al.* 1985; Henny and Herron 1989; King *et al.* 2003); infertile (Garrett 1996); changes in water levels at nesting site (Ryder 1994).	abandonment (Taft *et al.* 2000); competition between nests (Capen 1977; Capen 1978); exposure to elements (Belknap 1957; Capen 1977; Capen 1978; Dark-Smiley and Keinath 2003; Hancock *et al.* 2010; Ryder 1994); researcher disturbance (Belknap 1957; Steele 1980); hunted (Hancock *et al.* 2010); entangled (Belknap 1957); predation (Belknap 1957; Capen 1977; Dark-Smiley and Keinath 2003; Garrett 1996; Hancock *et al.* 2010; Hothem *et al.* 2006; Ryder 1994; Steele 1980; Taft *et al.* 2000; Zaun *et al.* 2003); heavy metal poisoning (Flickinger and Meeker 1972); pesticide poisoning (Capen 1978; Flickinger and Meeker 1972; King *et al.* 1980; Ryder 1994); changes in water levels at nesting site (Earnst *et al.* 1998; Ryder 1994).

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Predation

Predation is a natural component of waterbird population biology. However anthropogenic changes, in particular the introduction of feral predators and habitat alteration, have changed the nature and importance of its impact on waterbird populations (Sovada et al. 2001). A change in predation is often one of the main mechanisms via which land use or habitat change affect waterbird breeding success (Douglas et al. 2014). Changes in predation may cause prey populations to decline to extinction, or to stabilise at lower levels where they may be more susceptible to extinction from other causes (MacDonald and Bolton 2008). Populations that have declined due to other causes are also more likely to suffer more from predation. Usually more than one factor is involved when predation negatively impacts waterbird populations (Sovada et al. 2001).

Nest predation is well known to be of great importance to birds, not only in terms of population regulation, but also in nest placement and distribution (McKilligan 1987) and nesting density (Sugden & Beyrsbergen 1986; Dorfman and Read 1996). Many studies have shown that predation on waterbirds occurs mainly during nesting, and most nest predation involves eggs rather than chicks (though the latter is rarely measured). Predators also take nestlings or fledglings, or scavenge those that die due to other causes such as starvation. Predation on adult waterbirds is relatively rare, but is probably additive to mortality due to other factors (e.g. hunting, pollution; Sovada et al. 2001 citing King and Derksen 1986, Raveling 1989, Sargeant and Raveling 1992).

Waterbird eggs and chicks may be preyed upon by avian, mammalian, or reptilian predators – and occasionally fish. Raptors (e.g. eagles, falcons) and corvid species (e.g. ravens, crows) are major predators at waterbird nests worldwide and are regularly associated with waterbird nesting colonies. Mammals are also critical and often more cryptic predators of nests. Adult waterbirds are most commonly preyed upon by avian and mammalian predators. Introduced predators, particularly mammals, may cause local extinction of breeding populations.

Predators can reduce the survival of waterbirds and consequently population size either through direct predation, or indirectly, by causing adults to desert their nests or foraging sites, competing for habitat or food, or affecting other predators and prey (Cruz et al. 2013; MacDonald and Bolton 2008; Skorka et al. 2014). Fluctuations in predation pressure on waterbirds may result from fluctuations in the availability of other prey. The relative importance of waterbirds in predator diets changes with fluctuations in their availability, however some species and individuals can be selective in terms of which species, ages, and sizes they kill, and in which habitats or locations.

Factors that may influence rates of predation on nests include water depth, colony size, time of season, habitat type, habitat management, nest concealment, nest location, nest density, distance to

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habitat features used by predators (e.g. perches, nests, dens, shelter), and of course, predator abundance. Usually, predation rates are higher in smaller colonies and at the periphery of colonies (Angulo-Gastelum et al. 2011). There is conflicting evidence for most of factors, and it is likely that the influence of each on rates of predation is dependent on location and species (MacDonald and Bolton 2008; Sutherland et al. 2012).

The existing literature contains no quantified mortality rates from predation for Australian white ibis, straw-necked ibis or royal spoonbill eggs or chicks; however predation events and the presence of predator species are occasionally recorded. Most predation records are for Australian white ibis and straw-necked ibis; royal spoonbill predation records are rare. Avian predators are the most commonly reported in the literature, dominated by *Corvus* species preying on eggs and chick (Lowe 1984; Martin 2012, Smith et al. 2008, Smith 2009, Smith et al. 2013; Vestjens 1977; Campbell 1974). Other reported avian predator species include: White-bellied sea eagle (*Haliaeetus leucogaster)* recorded predating Australian white ibis eggs and chicks (Martin 2012); Whistling Kite *(Haliastur sphenerus)* predating Australian white ibis eggs and chicks (Hancock et al. 2010); Pelicans (*Pelecanus conspicillatus*) predating Australian white ibis eggs at Brisbane Waters, Woy Woy (Smith 2008); and Australian white ibis predating royal spoonbill eggs (Bekle 1982).

Feral mammals are also known predators of ibis and spoonbill eggs and chicks. In particular, feral pigs (*Sus scrofa*) have been reported predating Australian white ibis chicks in Macquarie Marshes (Johnson pers. comm in Scott 1997); straw-necked ibis eggs in Eulimbah Swamp, Lower Murrumbidgee (Brandis et al 2014); straw-necked ibis eggs and chicks in the Macquarie Marshes (Kingsford et al. 2003); and straw-necked ibis eggs and chicks in the Narran Lakes (Terrill 2008). Foxes *(Vulpes vulpes)* have been reported predating Australian white ibis chicks at Healesville Sanctuary (Beilharz 1988) and straw-necked ibis eggs and chicks at the Macquarie Marshes (Kingsford et al. 2003) and Narran Lakes (Terrill 2008). Feral cats (*Felis cattus)* have also been recorded predating straw-necked ibis eggs and chicks in Macquarie Marshes (Kingsford et al. 2003).

Documented reptilian predators include lace monitors (*Varanus varius*), reported taking royal spoonbill eggs and chicks at Lake Cowal (Vestjens 1977) and straw-necked ibis eggs and chicks at Macquarie Marshes (Kingsford et al. 2003); and tiger snakes (*Notechis scutatus*) reported taking 21 straw-necked ibis chicks on flooded lignum at Lake Cowal (Vestjens 1976).

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Abandonment

For successful breeding of most colonial-nesting waterbird species to occur, a shifting spatiotemporal mosaic of wetland inundation needs to occur over a lengthy period. This period varies by species, with species such as great egrets requiring more time than species such as straw-necked ibis (Briggs and Thornton 1999; Arthur et al. 2012). Flow thresholds have been identified for some of the major breeding sites in the Murray-Darling Basin below which colonial-nesting waterbirds will not initiate breeding (Arthur et al. 2012). For ibis and spoonbills in Barmah-Millewa, modelling has shown that breeding initiation generally commences after a 15,000 ML/day flow threshold is exceeded for 30-50 days and is related to total flow volumes being over ~2.3×10^6 ML between July and December (Arthur et al. 2012). Arthur et al. (2012) suggested that these thresholds exist in Australian because time is required for food sources to build up and for pair bonds to form once flooding occurs (McKilligan 1975; Kingsford and Norman 2002). Following breeding initiation, eggs are incubated for approximately 21 days and chicks fed at the nesting site for 35-50 days. Overall, inundation of nesting and foraging sites is required for a minimum of 85 – 121 days (3-4 months) to achieve fledging of a single cohort of ibis or spoonbill chicks. This does not take into account the time needed by juveniles to acquire foraging skills and by adults to rebuild body mass following breeding. In addition, the total number of colonial-nesting waterbird nests tends to increase with area of floodplain inundation, suggesting food requirements for breeding and successful recruitment are correlated with area inundated (Kingsford and Johnson 1998; Kingsford and Auld 2005).

Premature reductions in flood depth or extent are well known to be associated with nest abandonment by ibis and spoonbill species. For example, changes in water level as little as 0.3m under nest sites or within nearby foraging areas in Barmah Forest have been associated with abandonment events (Leslie 2001). A rate of fall of 1440 ML/day/day sustained for 1 month during October, November or December was associated with abandoned breeding at the Barmah Forest Algeboia Plain nest site because of rapid water level recession (Leslie 2001). It has been estimated that by reducing the number and duration of floods, river management has reduced the frequency of 'successful' breeding events by more than 80% from the natural rate in Barmah Forest (Leslie 2001). Similarly, from 1886-2001 in the Macquarie Marshes there were six nest desertion events attributed to rapidly falling water levels, with losses of hundreds or thousands of eggs and chicks (Kingsford and Auld 2005). During a 2008 ibis breeding event in Narran Lakes (mostly straw-necked ibis), analyses found that water depth was a significant variable in determining reproductive success. Rapid falls in water level of more than 30cm over 40 days during the chick stage resulted in significantly decreased chick survival (Brandis et al. 2011). During January 2010 following local

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rainfall, an estimated 20,000 straw-necked ibis began nesting and egg laying in Narran Lakes, but all nests were abandoned following no further rain or river flows (Brandis 2010). Leslie (2001) attributed nest abandonment to reduced flood duration acting to decrease nest security and food availability during fledging.

Table 16 Reports of changes in water levels in major breeding sites causing abandonment of straw-necked ibis and Australian white ibis eggs and chicks.

Year	Location	Species	Lifestage	Water level	Scale / impact	References
1954	Macquarie Marshes, NSW	Straw-necked ibis	Chicks	Falling water levels (natural)		Kingsford 2003
1961-1967	Coolart Lagoon, VIC	Australian white ibis	Chicks	Falling water levels (natural)		Davis 1974
1989	Macquarie Marshes, NSW	Australian white ibis	Eggs; Chicks	Falling water levels (managed)	100 nests	Kingsford 2005
1989, 1990-1991	Eulimbah Swamp, NSW	Straw-necked ibis	Eggs; Chicks	Falling water levels (managed)		Magrath 1992
1990-1991	Eulimbah Swamp, NSW	Australian white ibis	Eggs	Falling water levels (unspecified)		Magrath 1992
1990-1991	Booligal Swamp, NSW	Straw-necked ibis	Eggs	Falling water levels (managed)	Fully feathered chicks	Magrath 1992
1993	Macquarie Marshes, NSW	Straw-necked ibis	Eggs; Chicks	Falling water levels (managed	>200 nests	Kingsford 2005
1995-1996	Macquarie Marshes, NSW	Australian white ibis & straw-necked ibis	Eggs; Chicks	Falling water levels (managed)	800-1000 chicks	Kingsford 2005
1996	Macquarie Marshes, NSW	Straw-necked ibis	Eggs; Chicks	Falling water levels (managed)	400 nests (1000 chicks)	Scott 1997
1997	Narran Lakes, NSW	Straw-necked ibis	Chicks	Falling water levels (unspecified)		Merritt 2016
2008	Narran Lakes, NSW	Straw-necked ibis	Chicks	Falling water levels (managed)		Kingsford 2008; Brandis 2011; Terrill 2008
2008	Narran Lakes, NSW	Straw-necked ibis	Eggs	Rising water levels (natural)	40% egg mortality	Brandis 2011
2008	Narran Lakes, NSW	Straw-necked ibis	Eggs	Rising water levels (managed)		Terrill 2008
2010	Narran Lakes, NSW	Straw-necked ibis	Chicks	Falling water levels (unspecified)		Merritt 2016
2010	Narran Lakes, NSW	Straw-necked ibis	Eggs	Falling water levels (unspecified)		Brandis 2010
2015-2016	Narran Lakes, NSW	Straw-necked ibis	Chicks	Falling water levels (natural)		Merritt 2016

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Disturbance

Disturbance from human activities can cause changes in behaviour and affect temporal and spatial distribution of waterbirds. Human disturbance can be equivalent to habitat loss or degradation because waterbirds may avoid or underuse areas (Gill & Sutherland 2000 cited by Fernandez and Lank 2008). Waterbirds may waste time and energy responding to human disturbance, and the consequences for individuals, populations, and sites can be difficult to determine. Temporary loss of foraging habitats can occur and the capacity to compensate by foraging for longer periods may vary between individuals and species (Sutherland et al. 2012 and references therein). During the breeding season, human disturbance may influence nest incubation and chick rearing.

The effect of human disturbance on waterbirds is dependent on the nature and location of the disturbance and its type, frequency, intensity and duration (Carney and Sydeman 1999; West et al. 2002). Waterbirds may be more sensitive to disturbances during particular periods, such as nesting, hatching, brood care and moulting (Carney and Sydeman 1999; Burger 1982; Anderson 1988; Randler 2003). For example, at Lake Cowal NSW, 200 pairs of straw-necked ibis abandoned nests containing eggs after nearby disturbance from duck shooters (Vestjens 1977).

Responses to disturbance vary with species, populations, and timing, with some species more being sensitive early in the breeding season than late, and vice versa. Temporary or permanent abandonment of nests as a result of disturbance leaves eggs and young susceptible to predation or exposure, and they may also be crushed as adults leave the nest.

To some extent, some species can compensate for disturbance by altering their behaviour or habituating to human activities, and this may interact with predation (Madsen 1995; Baudains and Lloyd 2007; Fitzpatrick and Bouchez 1998). The effects of human disturbance may increase predation, or may not, depending on the characteristics of the disturbance and the species present. In some species and colonies, intensive study involving trapping, banding and repeated nest visitations can be conducted without detrimental impacts on reproductive success (Shealer and Haverland 2000; Olsen and Rohwer 1998). This may depend on location and other factors such as adult or chick habituation to humans (Tremblay and Ellison 1979; Parsons and Burger 1982; Davis and Parsons 1991; Frederick and Collopy 1989).

The presence of scientific temperature loggers and nest cameras, and the associated disturbance in placing these devices, might be expected to increase rates of predation. However, this was not found to be the case for temperature loggers in Piping Plover *Charadrius melodus* and Long-billed Curlew *Numenius americanus* nests in the USA, for video cameras at wader nests in New Zealand, or for digital nest cameras at Lapwing nests in the United Kingdom (Hartman & Oring

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2006, Schneider & McWilliams 2007; Bolton et al. 2007; Sanders & Maloney 2002; cited by MacDonald and Bolton 2008). Increased egg losses in duck nests are associated with the presence of scented attractors such as duck feces and cracked eggs, but the effects of human trails to nests are equivocal (Olsen and Rohwer 1998).

Comparatively little is known about how reactions to disturbance may impact on the dispersal of waterbirds and, ultimately, on their overall population dynamics (Madsen 1995). One study has shown that capture of breeding adult little egrets (*Egretta garzetta*) at their nest and handling (i.e., disturbance) did not increase the probability that they would move to a new colony in the subsequent breeding season (Henry et al. 2004).

Nest exposure, temperature and other weather-related variables

There are no published weather-related mortality rates for Australian white ibis, straw-necked ibis or royal spoonbills and few records of mortality events related to nest exposure, temperature and other weather-related variables. Australian white ibis chick mortalities have been reported from unusual tides and storms (Smith et al. 2013). At Lake Cowal NSW, royal spoonbill chicks were predated after exposure to elements (unspecified) led to parents leaving nests unattended (Vestjens 1977). In Blenheim, New Zealand, royal spoonbill adults have abandoned nests due to high tides and storms (Holdaway 1980).

Unfavourable weather events can negatively influence breeding success in a variety of ways. Depending on the nature and timing of severe weather, waterbirds may opt out of breeding entirely or suffer population losses through decreased clutch size, reduced hatching success, increased juvenile mortality, or even adult mortality (Smith et al. 2010; Duff et al. 2011). Cold temperatures, heavy precipitation and strong winds in particular can depress growth rates in waterbird chicks, thereby reducing chances of survival (Kasprzykowski et al. 2014; Ganter and Boyd 2000; Van de Pol et al. 2010). Unusually dry weather can cause nest desertion or starvation, and intense precipitation events such as storms can cause nest and egg destruction and chick death (McKilligan 1987). Extreme cold or hot weather conditions can result in high mortality rates. For example, several mass mortality incidents have been recorded in wild water birds in England in association with prolonged freezing weather (Duff et al. 2011). During these incidents, the ground froze, preventing access to food sources and causing starvation. Some flocks moved to feeding on landfill sites, where they are thought to have contracted and spread avian botulism (Duff et al. 2011).

Insufficient or poorly timed rainfall can also result in mortality of both juvenile and adult waterbirds. Magpie goose goslings die in large numbers if breeding and brood-rearing swamps dry

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before they are fledged and able to fly to often distant dry season habitat (Frith and Davies 1961). Starvation due to food shortage following periods of low rainfall seems likely to be a major variable in Cattle Egret mortality worldwide. Primary foods for cattle egrets such as earthworms are more abundant in moist soils and are likely to be difficult to obtain during dry periods (Siegfried 1972; McKilligan 2001). The importance of spring rainfall to egret breeding success has been well-established in some Australian locations (McKilligan 1997; 1997; 2001); however in contrast Maddock and Baxter (1991) found that the breeding success of Cattle Egrets did not vary consistently with wet and dry seasons, over six nesting seasons. McKilligan (2001) suggested that the lack of stability in the annual nesting numbers of Cattle Egrets at a long-term monitoring site in Queensland, compared with the situation in France, for example (Hafner et al. 1992), may be largely a reflection of the rainfall-determined 'stop–go' environmental conditions characteristic of much of Australia (McKilligan 1997; 2001; Maddock and Baxter 1991). Autumn rainfall is also thought to be particularly important for cattle egrets because it promotes the growth of populations of the egret's prey at a critical time for the survival of recently independent juvenile egrets (McKilligan 2001). Annual variation in mortality of juvenile and adult Cattle Egrets probably explains most fluctuations in the size of the breeding population, and most egret mortality occurs outside the breeding season when the birds are dispersed around coastal south-east Australia (McKilligan et al. 1993).

Colonial-nesting waterbird breeding success is thought to be higher following a flood in late winter, spring, or early summer, because increasing temperatures favour the production of food sources such as invertebrates and wetland plants (Briggs 1990, 1994, Maher 1991). Cold temperatures are thought to affect nest success by causing adult desertion and chick death. This is likely also driven by the effects of cold temperatures on food resources such as emerging invertebrates and plants. Scott (1997) cites a report of breeding abandonment in Narran Lakes in very cold weather in early June, even though there was no change in water level, which resulted in the abandonment of 5000-10000 young birds (Brandis 2010 citing McCosker 1996; Taft et al. 2000; Scott 1997 citing B. Johnson pers. Comm.).

Little is known about the relative importance of the various mechanisms that may depress breeding success because of weather. Because of the cost and logistical difficulties involved, long-term studies of the population ecology of waterbirds that provide detailed information on breeding parameters and their relation to weather patterns are relatively scarce. However detailed information on the relationship between weather and reproduction is needed as input into scenarios for assessing the potential impact of global climate change on bird populations (Ganter and Boyd 2000).

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Competition, conspecific damage, rejection and other behavioural factors

Mortalities due to competition and other behavioural factors are rarely quantified or reported for Australian white ibis, straw-necked ibis and royal spoonbills, however they do occur. At Healesville Sanctuary VIC, Lowe (1984) observed groups of noisy and mobile Australian white ibis congregating at nest sites, which resulted in chicks and eggs being pushed from nests and not retrieved. These congregations can also result in nest abandonment and the trampling of eggs and chicks (Hancock et al. 2010). Lowe (1984) suggested these congregations were associated with courtship and nest-site takeovers. Similarly, an Australian white ibis nest with eggs was reportedly destroyed by an interloping female over competition for a breeding partner (Hancock et al. 2010).

Both inter- and intra-specific competition for resources can affect bird condition, behaviour, abundance, distribution, and survival, and can vary both spatially and temporally. While food and habitat niches are usually sufficiently different to avoid competition between species in a native community under normal conditions, the introduction of invasive species, increases in populations of particular native species driven by other factors, loss of critical habitat or food sources, or extreme weather conditions such as drought can significantly alter competition intensity and timing. Increasing numbers of a particular species with specific food and habitat requirements may result in increased local density-dependent interference and exploitative competition. This is particularly likely when decreases in habitat or food availability occur. Such competition may particularly affect the movements, condition and survival of young, female, or subordinate portions of a population (Gyimesi et al. 2010; Amat and Rilla 1994; Stirneman et al. 2012; Minias et al. 2014). Individual 'despots' are also known to occur and may strongly affect habitat use and food intake of other individuals (Harper 1982). However the mechanisms of bird density and prey density effects on competition can be complex, differing between species (Vahl et al. 2005). An inverse association between egret breeding success and nesting numbers over 14 years at a colony in Queensland, Australia, has been suggested to be a density-dependent process, possibly based on competition for food (McKilligan 2001). This competition may occur between adults, or juveniles, or even between chicks in the nest. Cattle egret chicks hatch asynchronously, and it has been demonstrated experimentally that chick growth and mortality in the nest is significantly affected by competition between siblings for food and direct sibling aggression, with older chicks occasionally killing younger chicks (Fujioka 1985). Intraspecific predation has also been reported among juvenile White Ibis (*Eudocimus albus*) in the Florida Everglades (Herring et al. 2005). Scarcity of preferred nesting sites is probably the key ecological factor underlying competitive behaviours during breeding, driven by natal philopatry, nest-site fidelity, aggressive competition for nest sites and intraspecific parasitism (Semel and Sherman 2001).

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Starvation

Mortality rates attributable to starvation have not been quantified for ibis and spoonbill species in Australia, however starvation is thought to be an important driver of chick mortality. At several Australian white ibis breeding sites in southern Victoria over multiple consecutive years, breeding success appeared to be limited by food shortage and chick death from starvation was apparent (Lowe 1984). Starvation of the smallest chick in broods of three was found to be the most important factor limiting nest success (nests fledging young) in Australian white ibis nests at Healesville Sanctuary (Beilharz 1988). Clutches with four eggs didn't fledge significantly more chicks than three egg clutches (Beilharz 1988), suggesting starvation of younger or smaller chicks in a brood was a primary mortality factor. In Australian white ibis urban colonies at Woy Woy and Bankstown, evidence of chick malnourishment has been observed; most young captured at 28 days old or older had protruding keel bones and many of these died in subsequent weeks (Smith 2009).

Habitat loss, fragmentation and change can cause food shortages and subsequent starvation of nestlings, fledglings and juveniles. Young birds in the nest are dependent on abundant food brought to them by their parents, and loss of foraging habitat during nesting can have significant negative effects. A study of cattle egrets in South Africa found that 66% of eggs failed to become fledged young, and attributed this to chick starvation following food shortage (Siegfried 1972 cited by McKilligan 1987). Starvation of cattle egret nestlings was also an issue for 'younger' chicks in each nest - e.g. the fourth chick - in Queensland Australia (McKilligan 1987). Starvation due to food shortage following periods of low rainfall seems likely to be a major variable in colonial-nesting waterbird mortality worldwide. Primary foods for ibis and egrets are more abundant in moist soils and are likely to be difficult to obtain during dry periods (Siegfried 1972; McKilligan 2001). The importance of spring rainfall to egret breeding success has been well-established in some Australian locations (McKilligan 1997; 1997; 2001); however in contrast Maddock and Baxter (1991) found that the breeding success of Cattle Egrets did not vary consistently with wet and dry seasons, over six nesting seasons. Autumn rainfall is also likely to be particularly important for juvenile colonial-nesting waterbirds because it promotes their prey at a critical time for their survival (McKilligan 2001).

Toxins

Toxin-associated mortality rates for ibis and spoonbill eggs and chicks have not been quantified in Australia. However it is likely that toxins play a role in egg and chick mortality, and mortality

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events are occasionally reported. For example, in the 1995 breeding season at the Macquarie Marshes, NSW, it was estimated that approximately 500 straw-necked ibis chicks were poisoned by pesticide (chlorpyrifos) from prey fed to them; adult deaths were also reported (Johnson pers. comm in Scott 1997). Australian white ibis eggs have been analysed for toxin content, however no studies state whether toxin concentrations caused mortality. For example, Ridoutt et al. (2011) found a large variation in concentration of DDT in eggs collected from eleven Australian white ibis colonies in urban, peri-urban and inland sites, however no difference between the sites was found. PCB, PBDEs and dioxin concentrations in eggs collected from urban colonies in Sydney and Brisbane were 7-9 times higher than inland sites at Mathoura, Lowbidgee and Macquarie Marshes (Ridoutt et al. 2011). Furan concentrations were significantly higher at inland sites (Mathoura, Lowbidgree, Macquarie Marshes) compared to urban sites and four times higher than at peri-urban sites.

Worldwide, many populations of waterbirds have suffered declines in abundance as a result of toxic effects of pollution or contaminants in their environment (Powell and Powell 1986; Peterle 1991; Grasman et al. 1998 cited by De Luca-Abbott et al. 2001). While pesticides (herbicides and insecticides) have received much deserved attention in the last 30 years, waterbirds may also be affected by many other contaminants. For example, nutrient pollution was hypothesised as being a key driver of population declines in 16 waterbird species reviewed by Poysa et al. (2013), particularly in those species that preferred eutrophic lakes as breeding habitat. Spills of other toxic substances from a range of industries and activities, such as waste chemicals, oil and boat fuel, are well known for their effects on fish and waterbirds.

Often, a considerable number of contaminants are present in the environment and consequently it can be difficult to associate toxic effects with any one chemical. Very similar chemicals can vary widely in their toxicity, and it is also likely that some chemicals interact to increase toxicity. Critical effect concentrations of contaminants are often unknown, making risk assessment difficult (De Luca-Abbott et al. 2001). Residue levels of toxins associated with reproductive effects vary widely from species to species, and are complicated by interactions between toxins, sometimes depending on geographic location. There is variability in accumulation of contaminants between species, sexes, ages and tissue types (De Luca-Abbott et al. 2001). Consequently, studies of contaminant effects must establish the relationships between toxin levels and each of these variables if they are to yield meaningful results.

Diet has a significant effect on the concentration of contaminants accumulated. Predatory waterbirds are susceptible to bioaccumulation (or biomagnification) of organochlorines (e.g. DDT, dieldrin, other chlorinated pesticides, PCBs) and some metals (e.g. mercury). Piscivorous waterbirds are more susceptible to bioaccumulation through the food chain than insectivores or

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herbivores - for example, the aquatic-foraging piscivorous intermediate egret has been found with higher levels of contaminants than the mostly terrestrial and insectivorous cattle egret (although both have varied diets) (Burger and Gochfeld 1996 cited by De Luca-Abbott et al. 2001; Burger et al. 1992).

The vast majority of effects are expressed in reproductive failure, and subsequent population declines may be masked by pairs laying repeat clutches after failures. Bioaccumulation has been associated with breeding failures in some waterbird species (Faber et al. 1972; Cooke et al. 1976; Price 1977; Sanderson et al. 1994 cited by De Luca-Abbott et al. 2001). Eggs and developing chicks are generally more susceptible to toxins and contaminants than adult birds (Kushlan 1993; De Luca-Abbott et al. 2001). There is wide variation in susceptibility between species. Symptoms of toxicity include eggshell thinning, reduced egg weight, embryonic deformities and embryonic mortality (reduced hatch rate), reduced chick size and increased chick mortality, aberrant adult behaviour such as egg-eating, and reduced fledging rates and nest success.

Besides their direct toxic effects, pesticides and herbicides can reduce food availability for waterbirds, depending on their diet. Spraying of these toxins often coincides with breeding seasons of waterbirds, increasing the pressure on more susceptible eggs and young and the difficulty in obtaining sufficient food for nestlings, fledglings and juveniles (Rusch et al. 1989). For most invertebrate taxa, biomass will remain low after spraying until ovipositing adults recolonize the pond; this may take weeks or years depending on the taxa (Gibbs et al. 1984 cited by Rusch et al. 1989).

Diseases

Infectious diseases are an important and frequently dominant mortality factor in waterbird populations. An early survey of waterfowl deaths in North America indicated 80-90% of birds had died of disease (excluding those shot (Holmes 1982 cited by McKilligan 1987). Bacteria such as *Clostridium botulinum* and viruses such as avian influenza, West Nile Virus, Newcastle Disease Virus, avian poxvirus, duck plague, avian bornavirus, reoviruses and adenoviruses may contribute to population declines of both wild and domestic waterbirds. Some of the most widely occurring and researched diseases are Botulism, Avian Influenza, Avian Newcastle Disease and West Nile Virus. The infection rate by *Plasmodium* parasites (avian malaria) is also rapidly increasing in many birds and there are high infection rates of Campylobacteria in waders (Sutherland et al. 2012 and references therein). High densities of birds make them more vulnerable to infections (Krauss et al. 2010 cited by Sutherland et al. 2012).

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In Australia, mortality rates of ibis and spoonbills from disease have not been quantified. Botulism events do occur and are sometimes reported, however the effects of these events in terms of numbers per species and effects on different age groups are poorly understood.

Parasites

Ticks and other parasites affecting nestlings can reduce survival and nest success, and potentially also transmit viruses. The impacts of parasites on Australian white ibis, straw-necked ibis and royal spoonbill chicks have not been quantified or reported; however reports of impacts exist for other colonial-nesting species. For example, in a three-year study of the cattle egret at a heronry of 3500 nests in south-east Queensland, 34% of eggs in 711 nests observed failed to survive to become fledglings (11 day old chicks). Fledgling mortality seemed much lower. The main sources of mortality were crow predation on eggs, failure of the incubated egg to develop fully, starvation of nestlings and tick infestation of nestlings. These were of about equal importance overall but their individual magnitudes varied considerably among the three seasons. The overall loss of cattle egret nestlings to tick infestation was calculated as 10-12% depending on the year (McKilligan 1987). An experiment using acaricides found that mean tick load of cattle egret chicks was inversely correlated with chick longevity (McKilligan 1996). However a subsequent study (McKilligan 1997) found that annual differences in sizes of advanced cattle egret broods were mostly influenced by chick starvation, not by the sizes of early broods or the burden of parasites that season. Viruses transmitted by ticks can also increase mortality of waterbirds and chicks, but the interactions between the effects of ticks vs the effects of the viruses are less well understood (Standfast et al. 1986; McKilligan 1987 citing Converse et al. 1975; Duncan et al. 1978; Reid et al. 1978).

Management implications

Water management actions to support survival of eggs and chicks already include provision and maintenance of water within breeding habitats. It is well known that rapid or early declines in water levels can trigger nest abandonment. Abandonment is more likely during the egg stage than the chick stage, however it is still important to maintain water levels late in the breeding season, for two primary reasons: 1) Maintaining food and foraging habitat availability; and 2) Preventing feral predator access. In addition to provision of environmental water for breeding sites, provision of water to support nearby foraging sites and food before, during and after a breeding event is likely to affect the size and success of the event.

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Predation is a natural component of waterbird population biology. However anthropogenic changes, in particular the introduction of feral predators and habitat alteration, have changed the nature and importance of its impact on waterbird populations (Sovada et al. 2001). A change in predation is often one of the main mechanisms via which land use or habitat change affect waterbird breeding success (Douglas et al. 2014). Changes in predation may cause prey populations to decline to extinction, or to stabilise at lower levels where they may be more susceptible to extinction from other causes (MacDonald and Bolton 2008). Populations that have declined due to other causes are also more likely to suffer more from predation. Usually more than one factor is involved when predation negatively impacts waterbird populations (Sovada et al. 2001).

Actions that can be taken to complement environmental flows, reduce pressure on adults, eggs and chicks, and increase egg and chick survival include: Feral predator control (pigs, foxes and cats); Invasive plant species control (native and exotic); Habitat modification and maintenance to provide a range of vegetation types and configurations to choose from; and disturbance control. It is likely that these pressures vary between different locations and will therefore require different levels of effort and investment across the Basin.

There are also several possible drivers of egg and chick mortality occurring at breeding sites that are difficult to quantify visually through surveys or camera monitoring. These include toxins, diseases, parasites, and interactions between these and other pressures. Evaluating the roles of these factors in affecting egg and chick survival will require testing of swabs, blood or other samples collected in the field.

Disturbance management

Buffers are often used to separate threatening stimuli, such as humans, from wildlife but with few exceptions buffer widths are based on little empirical information. Glover et al. (2011) measured the distance at which a response (i.e., flight initiation distance [FID]) occurred among 28 of Australia's 36 regularly occurring shorebird species when presented with an approaching human (n = 760 approaches in Victoria, south eastern Australia). Species differed in their FID. with species with higher body masses having longer FIDs ($F(1,2G) = 36.830$. $p < 0.001$; $R(2) = 0.586$). Mean FIDs for species were 18.6-126.1 m (n = 370 approaches by a walker). Depending on the species, FID was significantly influenced by the starting distance of the human approach, flock size, previous exposure to humans and stimulus type (walker, jogger, walker with dog). The FIDs reported suggest that current buffer designations will reduce disturbance to many but not all shorebird species tested (Glover et al. 2011).

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Studies of mixed colonies of waterbirds containing herons, egrets and ibis have found that most initiate flight (flush) when humans approach to between 30-50 m, and buffer zones of 100 m have been suggested to minimize disturbance and prevent nest desertion, temporary or permanent (Carney and Sydeman 1999 citing Erwin 1989; Rodgers and Smith 1995). The mode or route of approach may also make a difference, even when the frequency of disturbance is high. A study in British uplands found that while high levels of human disturbance via foot traffic can affect habitat usage by waders, this is generally only in limited circumstances where visitor pressure is very high (> 30 visitors per weekend day). Research suggested that access to such areas can be permitted for larger numbers of visitors without impacting upon wader reproductive performance through the provision of a well-surfaced route (Pearce-Higgins et al. 2007).

Since rates of disturbance differ between species and may be correlated to foraging strategies, habitat requirements, and type of disturbance, managers of public lands should consider life histories of waterbirds when setting seasonal recreation dates or planning scientific surveys (Schummer and Eddleman 2003). Larger waterbird species often have greater 'flush distances' than smaller species to the approach of a disturbance such as a human or boat, and therefore management buffer distances need to take this into account (Rodgers and Schwikert 2003). In most situations that have been studied, the creation of refuge areas for migratory waterfowl has generally increased the number of waterfowl using a site. However, experimental evidence to show that the increases were caused by reduced disturbance and not by confounding factors, e.g. improvement of habitat quality of the sites or general population increases, is generally lacking (Madsen 1998). To be able to answer questions about the effects of disturbance, a better theoretical framework, based on optimal foraging theory incorporating predation risk, and field experiments are required. Furthermore, we need to study waterbirds throughout their ranges to interpret the overall impacts of disturbance (Madsen 1995).

Climate change

Changes in temperatures, the timing, intensity, amount and duration of precipitation, and the frequency and severity of extreme weather events all have the potential to influence waterbird populations positively and negatively, and directly and indirectly (Sutherland et al. 2012; Chambers et al. 2005). Climate change already appears to be affecting population movements such as migration and dispersal, as well as breeding initiation and success and survival between breeding events. Chambers et al. (2005) reviewed the effects of climate change on birds that have been documented or predicted, with particular reference to Australian species. Potential impacts included changes in geographic range, distribution, movement patterns, morphology, physiology, abundance, phenology and community composition. They found that the evidence suggests that these changes

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are already happening, both overseas and in Australia, but more research is needed to determine their extent and how to conserve birds in the face of climate change. Relative to the northern hemisphere, little is known about the effect of climate change on southern hemisphere birds, although the impact could be significant. Management options suggested included promoting adaptation and resilience, intensive management of sensitive species, and improved planning for mitigation techniques and monitoring.

In Australia's Murray-Darling Basin, warmer temperatures, reduced rainfall, increased evaporation and decreased surface-water flows are expected to occur within the next few decades. Importantly, such changes will reduce the availability of both breeding and foraging habitats for waterbirds by reducing flood frequency, duration, depth and extent, as well as changing vegetation composition and condition. Critically, if breeding opportunities do not occur with sufficient frequency and size, recruitment may not be sufficient to maintain some waterbird populations (Arthur 2011). We have very few data with which to explore the potential impacts of such changes via modelling.

Conclusion

Data quantifying waterbird fledging rates and their drivers in Australia are scarce, with some exceptions for particular species or groups in certain locations such as egrets. These data can be difficult to collect in the field, however detailed, long-term monitoring is essential if we are to build better predictive models that allow assessment of the potential quantity and timing of water required to ensure successful fledging (Brandis et al. 2011).

With ongoing declines in waterbird populations, it is important that we understand how many birds are being produced by breeding events, together with what is affecting those numbers. Using monitoring cameras to capture such data facilitates explanation of perceived failures, by providing additional information describing the effects of other pressures, stressors or threats such as predation, disturbance, starvation, competition and other bird behaviour.

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Disclosure statement

The authors declare no potential conflicts of interest.

Data availability statement

The data that support the findings of this study are available on reasonable request from the corresponding author, H.M. The data are not publicly available due to intellectual property restrictions.

Ethics statement

This research was conducted in compliance with the CSIRO Animal Ethics Committee Authority Number 2015-17 and the University of NSW Animal Care and Ethics Committee Authority Number 17/122B.

Author contributions

H.M. Conceived the idea, obtained funding, supervised the research, led design and methods development, conducted fieldwork, collected data, analysed data, and co-wrote the paper.

F.R. Assisted with methods development, conducted fieldwork, collected data, analysed data, and co-wrote the paper.

M.P. Assisted with methods development, conducted fieldwork, collected data, analysed data, and co-wrote the paper.

J.H. Analysed data and co-wrote the paper.

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Supplementary material

Data extraction and entry

The Excel workbooks comprised multiple spreadsheets. One spreadsheet outlined instructions for data extraction and entry, and another spreadsheet contained set variables for data entry linked to drop down boxes in the data entry spreadsheets, to ensure consistency between individuals and seasons. The next spreadsheet was for a key to colour codes on the data tabs. The colour codes defined who entered each section of the data in the 'All variables spreadsheet' (explained below) and which data was from the different camera types in the 'Nest Summary data' tab. Two spreadsheets were for data entry; an 'All Variables' spreadsheet for all camera, image and nest event data throughout monitoring, collected both in the field and extracted from images; and a 'Nest summary data' spreadsheet which summarised each nest from the start to end of camera monitoring.

<u>All variables data spreadsheet</u>

Data collected during camera deployment were entered into the 'All variables' spreadsheet. This included information on camera deployment, camera settings and observations of the nest/s at deployment (Table 7). This data was entered across a single row for each camera and its respective SD card. A deployment code was also created at the time of image processing to distinguish which cameras and SD cards belonged to which deployment (Table 7). For example if a camera had an SD card replaced during a deployment, a new row in the data would be created to represent a change in SD card.

When the SD cards were swapped, or cameras retrieved, the date and time of retrieval were recorded. This information was also included in the spreadsheet. Once this data was entered for a nest, lines/ rows were copied and inserted for individual nests and 'events' as the images are reviewed. Every image and every nest for which an event happened and data recorded had its own row.

SD cards were split between staff for the processing of images and data was obtained for the sub-categories of 'SD card information' (Table 7). The first image in which the camera was left stable was used to assign each nest with a unique 'nest ID', and the image was appropriately labelled and saved for later reference (Image 4). The nest ID started with a three-letter code for the area of the wetland where the camera was located then a nest number starting at 001 (i.e RBW001). When multiple nests were monitored on one camera, nests were numbered from left to right then down (Image 4). Some nests were captured by multiple cameras hence the importance of the nest ID image.

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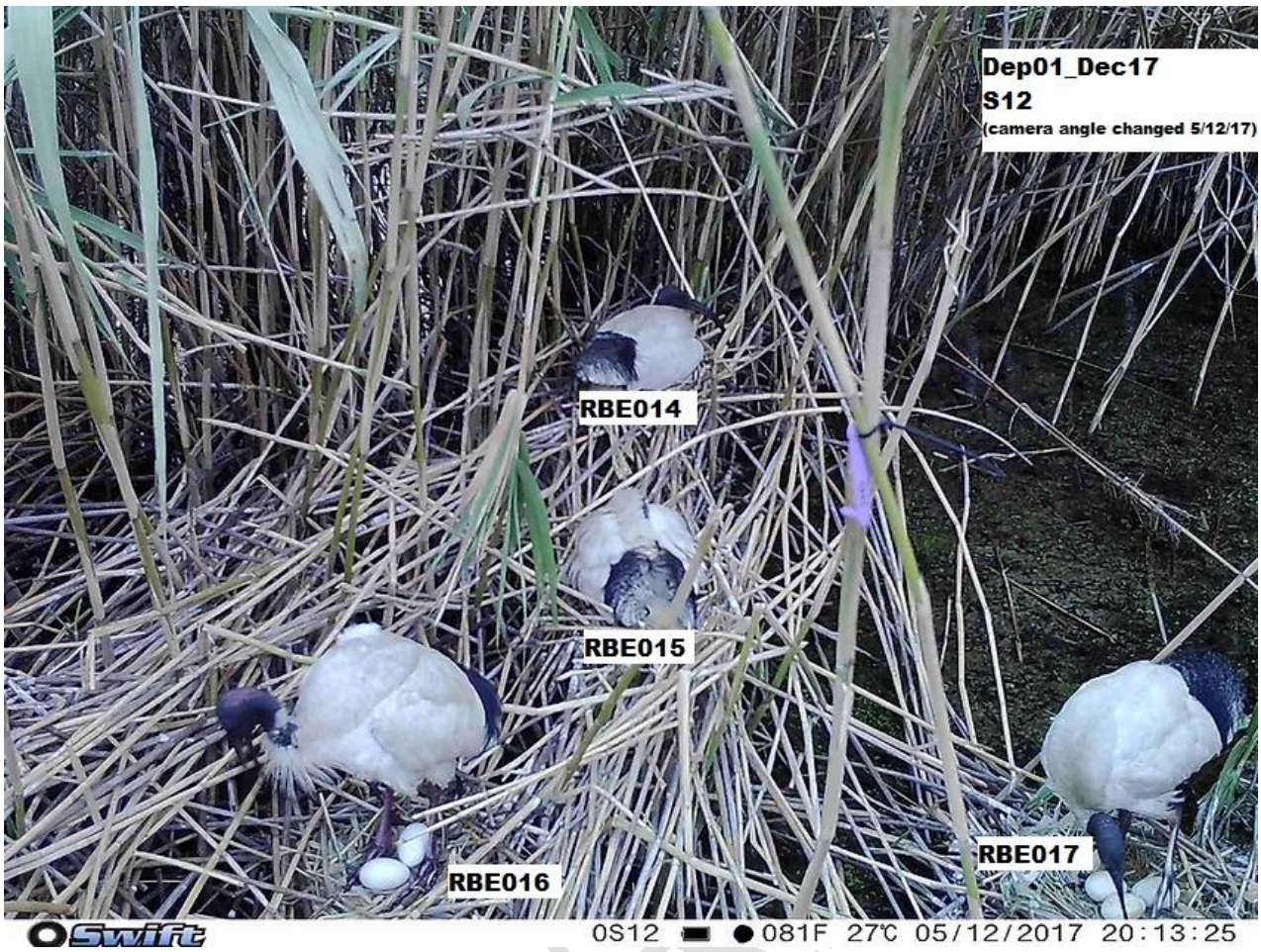
Once a nest ID was assigned to the nest/s the images were scanned through quickly in 'extra-large icon' view in windows explorer until a recordable event was observed. A recordable event included all the sub-categories listed in Table 7 under the categories of: 'nest information', 'predation mortality', 'other mortality', 'disturbance' and 'parental attendance'. Changes in egg and chick numbers in the nest were recorded under the 'nest information' category and egg and chick death recorded under 'predation mortality' or 'other mortality' (Table 7). Each time a mortality event or change in egg and chick numbers occurred for a nest a new row was created in the spreadsheet tab and the data entered under the relevant sub-category (Table 7). For example in Image 4, when an adult stands up you might notice a new egg has been laid; this change in egg numbers would be recorded in a new row. If a parent left the nest in an image a new row in the tab would also be created and the time of return in a subsequent image recorded under the 'disturbance' or 'parental attendance' categories (Table 7). Each event or change in the nest required all sub-categories under 'image information' (Table 7) and a nest ID to be recorded. Other events of interest or observations that added information to data were recorded under the 'comments and observations' category. Data for the nest was entered into the 'all variables' spreadsheet until it could not be determined which chicks belonged to the nest; usually highly mobile chicks of the flapper or flyer stage or chicks that were creching.

Nest summary data spreadsheet

A second spreadsheet, 'Nest summary data', summarised the data for each nest from the 'All variables' spreadsheet. This pooled all the data available to a nest, combining the data extracted from multiple cameras, SD cards and deployments. Each nest ID was assigned a single row and data taken from or calculated from the 'All variables' spreadsheet, and for some sub-categories, like 'max ages of chicks in view' and 'nest state in last useful image' from specific images of the nest.

Image 4

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Dep01_Dec17
S12
(camera angle changed 5/12/17)
RBE014
RBE015
RBE016
RBE017
Swift
0S12 081F 27℃ 05/12/2017 20:13:25

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Table 17: The sub-categories under which data was entered into the 'all variables' spreadsheet, organised by main category of data. The origin of the data (collected in the field or during image processing) and a description of the data type is given for each sub-category, along with an example of the format the data was entered into the spreadsheet.

Main category	Data from field collection or image processing?	Sub-category	Description of sub-category	Example of data entry
Camera deployment	Image	Deployment code	The deployment trip number for the season, deployment month and year	Dep02_Jan_18
	Field	Deployment date	The date of camera deployment	13/01/2018
	Field	Deployment time	The time of camera deployment in 24 hour time	15:42
	Field	Camera ID	Each camera was assigned a unique combination of a number and a letter respective of its brand (S=Swift, R=Reconyx, B=Boblov)	S01
	Field	SD ID	The SD card images were saved upon; each SD card was assigned a unique number and letter ID, for example A1. For Swift camera images accessed via the website 'WEB' was entered.	WEB
	Field	Location	The region within Reed Beds in which the camera was deployed (East, West or South)	Reed Beds South
	Field	GPS_UTM	The GPS point for the camera in UTM format	314358.28 m E; 6031584.48 m S
Camera settings	Field	Motion sensitivity	The camera's sensitivity to motion; this varied depending on what was programmable for the camera brand but generally could be set at 'Low', 'Medium' or 'High'	High
	Field	Pics per trigger	The number of images taken each time the camera is triggered by motion; this generally was 1-3 images	1
	Field	Time-lapse interval	The programmed time interval between time-lapse images	30 minutes
	Field	Period armed	The time period during which the camera was programmed to take pictures	24 hours
	Field	Quiet period	The time period between the camera being triggered by motion	2 minutes
Nest observations	Field	Species in area	The species observed on or around the monitored nest/s at camera deployment	Australian white ibis
	Field	Breeding stage at deployment	The stage of breeding (trampling/nest building, eggs or chicks) in the monitored nest/s at camera deployment	Eggs

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at camera deployment	Field	No. nests in cam view	An estimate of the number of nests in camera view at camera deployment	2
	Field	No. nests in clump	The number of nests situated in the nesting 'clump'(continuous reed bed with nests) of the nest/s being monitored by camera, at the time of deployment	8
	Field	Egg count at deployment	The number of eggs in each nest being monitored by camera (individual nests separated by commas in data entry), at camera deployment	3, 3
	Field	Chick count at deployment	The number of chicks in each nest being monitored by camera (individual nests separated by commas in data entry) at camera deployment	0, 0
	Field	Nest exposure	An estimate of the main aspect of a nest at camera deployment	North-east
	Field	Water depth (cm)	The depth of water where the camera is deployed	40
	Field	Nest height from water (cm)	The height of the nest above water (individual nests separated by commas in data entry) at camera deployment	29, 36
SD card information	Image	Time camera left stable	The time of the first image showing that the camera is stable (and therefore fully deployed) after deployment, in 24 hour time	15:50
	Image	Download/retrieval date	The date the SD card was collected from the camera in the field	16/02/2018
	Image	Download/retrieval time	The time the SD card was collected from the camera in the field	10:07
	Image	Number of images	The number of images on the SD card	22021
	Image	Date of last image	The date of the last image on the SD card	10/02/18
	Image	Number of days of images	The number of days of images on the SD card	29
	Image	Date of last useful image on nest/s	The date of the last image where the camera is focused on the nest/s	6/02/18
	Image	Number of days of useful images on nest/s	The number of days of images on the SD card where the camera is focused on the nest/s	25
Image information	Image	Number of nests in camera view	The number of nests in the image being processed	2
	Image	Image date	The date of the image being processed	13/01/18
	Image	Image time	The time of the image being processed, in 24 hour time	15:50
	Image	Image type	The type of image being processed; either motion-sensing or time-lapse	Motion-sensing
	Image	Temperature (°C)	The temperature on the image being processed	26

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Nest information	Image	Nest ID	A unique code given to each nest to identify it based on its location in Reed Beds (East, West, South)	RBW001 (Reed Beds West 001)
	Image	Species in nest	The species in the nest	Australian white ibis
	Image	Eggs	The number of eggs in the nest for the image being processed (if no eggs leave blank).	2
	Image	Hatchlings	The number of hatchlings in the nest for the image being processed (if no hatchlings leave blank)	2
	Image	Squirters	The number of squirters in the nest for the image being processed (if no squirters leave blank)	2
	Image	Runners	The number of runners in the nest for the image being processed (if no runners leave blank)	1
	Image	Flappers	The number of flappers in the nest for the image being processed (if no flappers leave blank)	1
	Image	Flyers	The number of flyers in the nest for the image being processed (if no flyers leave blank)	1
	Image	Adults	The number of adults or parents on the nest for the image being processed (if no adults leave blank)	1
Predation mortality	Image	Predation?	Record 'Yes' if mortality in the nest occurred from predation for the image being processed. Leave blank if no predation in image.	Yes
	Image	Observed predator species	Record the observed predator species (species name) for the image being processed	*Corvus coronoides*
	Image	Suspected predator species	Record a suspected predator species (species name) if the predator wasn't captured directly predating the nest in the image. For example, if a predator species is hanging around the nest or the nest is predated in a certain way.	*Corvus coronoides*
	Image	No. eggs taken/killed	The number of eggs in the nest taken or killed by the predator, for the image being processed	2
	Image	No. chicks taken/killed	The number of chicks in the nest taken or killed by the predator, for the image being processed	1
Other mortality (not predation)	Image	Chick death (not predation) count	The number of chicks in the nest that died or disappeared (were not recorded as predated), for the image being processed. This is to capture deaths/losses from other possible causes like abandonment, starvation or falling from the nest. Any observed or suspected mortality cause is recorded under the 'general comments' sub-category.	2
	Image	No. eggs rejected/failed to hatch (not predated)	The number of eggs in the nest which didn't hatch, were rejected by the parent or disappeared (i.e. didn't result in a chick and were not recorded as predated), for the image being processed. This is to capture deaths/losses from other possible causes like parents discarding, accidently breaking or abandoning eggs.	3
Disturbance	Image	Birds disturbed/flushed?	A record of the parent/s being flushed from the nest in the image, due to a disturbance event. Record 'Yes'	Yes

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	Image	Cause of disturbance	The cause of disturbance if known	Predator
	Image	Time parent returned to nest	The time (in 24 hour time) the parent returned to the nest after being flushed (the time of the next image the parent is present after being flushed)	16:00
	Image	Date parent returned to nest (if different from leaving date)	Record of the date the parent returned to the nest if it left on a different date.	12/02/18
Parental attendance	Image	Parent leaving (but not flushed)?	A record of the parent/s leaving the nest in the image, but not due to a disturbance event. Record 'Yes'.	Yes
	Image	Reason	The reason for the parent leaving the nest in the image (not a disturbance event). Reasons for leaving could be to feed, drink/forage next to the nest or swap minding duties with other parent.	Drinking
	Image	Parent on neighbouring nest?	A record of parents on other nests in the image. Record 'Yes'	Yes
	Image	Time parent returned to nest	The time (in 24 hour time) the parent returned to the nest after leaving (the time of the next image the parent is present after leaving)	17:00
	Image	Date parent returned to nest (if different from leaving date)	Record of the date the parent returned to the nest if it left on a different date.	12/02/18
Comments and observations	Image	Behavioural observations	Comments only. These are comments about bird behaviour that are not captured in the other categories. Comments could be a general comment about parental attendance during the deployment, i.e. very attentive, most of the time a parent was on the nest. Some other comments could be that a flock of birds, the same species attacked the parent and drove them off the nest (nest crash), parent returning having an injury. Juveniles foraging with parents in the water next to nest. Parent visited the nest for days after all eggs predated.	Parent very attentive
	Image	Camera observations	Comments only. These are comments about the camera functionality. For example, not taking as many shots as programmed, image colour is not correct, camera being triggered excessively by vegetation, camera moved, camera settings changed	Camera not picking up parents leaving
	Image	General comments	Comments only. Any other comments that add information to the data entered. For example, chick fell off nest and did not return; predator seen around nest; egg discarded by parent from nest after not being incubated overnight.	Chick fell from nest

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Table 18: The sub-categories under which data was entered or calculated from the 'all variables' spreadsheet into the 'nest summary data' spreadsheet, organised by main category of data. Some data was also taken directly from images as specified in the table. A description of the data type is given for each sub-category, along with an example of the format the data was entered into the spreadsheet.

Main category	**Sub-category**	**Description of sub-category**	**Example of data entry**
Nest information	Nest ID	The unique code identifying the nest. Taken from the 'all variables' spreadsheet.	RBW001
	Camera IDs	The cameras deployed on the nest over the entire monitoring period; some nests may have been monitored by one or several cameras. Taken from the 'all variables' spreadsheet.	S01, S02
	SD ID	The SD cards containing images of the nest; some nests may have multiple SD cards. For Swift cameras 'WEB' is entered. Taken from the 'all variables' spreadsheet.	WEB
	Species in nest	The species in the nest. Taken from the 'all variables' spreadsheet.	Australian white ibis
	Nest vegetation	The vegetation type the nest is made out of; provide species name and common name in brackets. Recorded from the camera images.	*Juncus ingens* (giant rush)
Egg and chick survival and mortality	Total eggs laid in nest during monitoring	The total number of eggs laid over the entire period the nest was monitored. If re-laying of eggs occurs, use a comma to delineate first clutch from second clutch, eg. 3, 2. If the number of eggs laid is not known then record a '0' and record a presumed number under 'presumed no. eggs'. Calculated from the 'all variables' spreadsheet.	3
	No. eggs failed to hatch, destroyed or rejected	The number of eggs which did not hatch an alive chick. Calculated from the 'all variables' spreadsheet.	1
	Presumed no. eggs	The presumed number of eggs laid based on the number of chicks in the nest. The laying of eggs may not have been captured as monitoring may have started after hatching. If the number of eggs laid is known then record a '0'. Calculated from the 'all variables' spreadsheet.	0
	No. chicks hatched	The number of chicks hatched alive, Calculated from the 'all variables' spreadsheet.	1
	Total chicks died in nest	The total number of chicks who died in the nest. Calculated from the 'all variables' spreadsheet.	1
	Unexplained chick/egg loss count	The number of eggs or chicks who were presumed dead (and unable to specify an egg or chick death) due to gaps in the monitoring period, resulting from dead camera batteries, a malfunctioning camera or the nest no longer in camera view. Calculated from the 'all variables' spreadsheet.	0
	No. chicks that left the nest	The number of chicks who reached the flapper and flyer stage; it usually couldn't be determined which chicks belonged to the nest when they were creching. Taken from the 'all variables' spreadsheet.	1

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	No. eggs in last useful image	The number of eggs in the oldest useful image of the entire monitoring period. An image was useful if the nest was still in view. Taken from the 'all variables' spreadsheet.	0
	Total eggs laid	The total number of eggs the nest produced, combining eggs from multiple clutches. Calculated from the 'all variables' spreadsheet.	5
Chick survival by age stage	No. hatchlings at end of monitoring	The number of live chicks at the hatchling stage from the nest at the end of monitoring. Taken from the 'all variables' spreadsheet.	0
	No. squirters at end of monitoring	The number of live chicks at the squirter stage from the nest at the end of monitoring. Taken from the 'all variables' spreadsheet.	0
	No. runners at end of monitoring	The number of live chicks at the runner stage from the nest at the end of monitoring. Taken from the 'all variables' spreadsheet.	1
	No. flappers at end of monitoring	The number of live chicks at the flapper stage from the nest at the end of monitoring. Taken from the 'all variables' spreadsheet.	2
	No. flyers at end of monitoring	The number of live chicks at the flyer stage from the nest at the end of monitoring. Taken from the 'all variables' spreadsheet.	0
	No. surviving chicks	The number of live chicks (regardless of stage) from the nest at the end of monitoring. Taken from the 'all variables' spreadsheet.	3
	Max ages of chicks in view	The maximum stage of chicks (hatchlings, squirters, runners, flappers, flyers) in the entire camera view (not just the chicks belonging to the nest) at the end of monitoring. Record NA (not applicable) if no chicks are in camera view at the end of monitoring. Taken from the oldest camera image with chicks in view.	Flappers
Nest state and summary	Nest state in last useful image	A short description of the state of the nest in the last useful image; this captures if the nest is still in use or not at the end of monitoring. Nests may depredated, contain eggs or a specific chick stage, be empty post-breeding, etc. Taken from the oldest camera image with nest in view.	Empty post-breeding
	Primary cause of egg/chick mortality	The main cause of mortality for eggs or chicks in the nest. This may include abandonment, starvation, predation, disturbance, unknown, etc. Record NA (not applicable) if no mortality occurred. Taken from the 'all variables' spreadsheet (if mortality cause known and recorded).	Abandonment
	Secondary cause of egg/chick mortality	Mortality of eggs or chicks in the nest arising from the primary mortality cause. For example a parent may have abandoned the nest (primary cause) and then predation occurred as a secondary event. Record NA (not applicable) if no mortality occurred overall or no secondary cause of mortality occurred. Taken from the 'all variables' spreadsheet (if mortality cause known and recorded).	Predation
	Nest state notes	Any additional comments on the state of the nest at the end of monitoring and any events which led to this state. Taken from the 'all variables' spreadsheet.	Egg sunk into nest after parent abandoned nest

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Nest monitoring period	Dates monitored	The dates the nest was monitored. If gaps in the monitoring period occurred then enter as separate periods delineated by a comma. Taken from the 'all variables' spreadsheet.	15/01/18- 20/01/18, 10/03/18- 17/03/18
	Deployments	The camera deployment periods during which the nest was monitored; record deployment codes. Taken from the 'all variables' spreadsheet.	Dep02_Jan_18; Dep03_Feb_18
	Total days monitored, entire season	The total number of days of images for the nest. Calculated from the 'all variables' spreadsheet.	14
Nest exposure		nesting group shape, vegetation, score on protection and open water, number of nests in clump, nest heights from water and neighbours, nest aspect (see table ? for definition of terms) The nest exposure category reflects decisions birds make on nest placement. This data is captured when setting up the cameras, camera images and aerial shots of the wetland.	

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Variables recorded included:

- Image information – Image date, time, type (motion or time-lapse) and temperature.
- Nest – Species in nest, No. eggs, hatchlings, squirters, runners, flappers, flyers, adults.
- Predation – predator species observed or suspected, no. of eggs or chicks taken or killed in the nest.
- Non predatory death/rejection – no. eggs rejected or failed to hatch, no. chicks died, No of eggs chicks in the last image on the camera.
- Disturbance – Birds disturbed/flushed, cause of disturbance, time parent returned, date returned (if different to date left), length of time parent absent
- Parental attendance – parent leaving but not flushed, reason, presence on parent on a neighbouring nest, time parent returned, date returned (if different to date left), length of time parent absent.
- Behavioural observations – as a comment
- Camera observations – as a comment
- General comments
- Nest information – Nest ID, camera ID's used to monitor this nest and SD ID's, Nest tags (used by UNSW), Species, vegetation.
- Nest activity – total eggs laid during monitoring, number of eggs failed to hatch, destroyed or rejected, presumed number of eggs, no. chicks hatched, no. chicks died in the nest, unexplained chick/egg loss, No. chicks fledged, No. eggs in last useful image, total eggs laid.
- Chick survival – No. of hatchlings, squirters, runner, flapper, flyers at the end of monitoring. No. of total surviving chicks at the end of monitoring. Max ages of chicks in camera view.
- Nest – Nest state in last useful image, primary and secondary causes of egg/chick mortality, notes on nest state, Dates monitored by all cameras on this nest, total number of days monitored. Comments on behavioural observations and camera settings.
- Nest exposure (see Supplementary material) – nesting group shape, vegetation, score on protection and open water, number of nests in clump, nest heights from water and neighbours, nest aspect

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- **No. of nests:** simply the number nests monitored. Cameras were deployed in such a manner that multiple nests could be monitored per camera.
- **Direct record:** these are those individuals that we observed on camera from egg stage- to hatching stage and then to its eventual fate i.e. survive/die.
- **Indirect records:** these are for those individuals that were present in the nest as chicks before and at the beginning of monitoring. In these cases we do not know how many eggs there actually were in the nest, what the fate of all eggs in the nest were etc… these records are included in overall statistical estimates but not in direct statistical estimates because their do positively inflate numbers /estimates in some cases.
- **No. nests with a full record:** the no of nests that follow from egg through to chick departure/'flyer' chick stage.
- **No. of clutches:** Some nests had multiple egg laying events therefore some summaries are more accurate at the clutch level than the nest level. This is the number of clutches that were monitored. Most data summaries are at the clutch level and are clearly stated if otherwise.
- **No. eggs full record:** the number of eggs (therefore individuals) that were monitored from egg to chick departure/'flyer' chick stage.
- **Monitoring days; mean, min and max:** the summary statistics for the number of days that each species was monitored.
- **Nest crash:** a flock of birds arrive at a nest and attack parent until the leave the nest. The flock then destroys eggs/chicks and the nest

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Murray-Darling Basin Environmental Water Knowledge and Research Project

Waterbird Theme Research Report

Appendix 4

Monitoring ibis and spoonbill nest success through on-ground tagged nest and water depth monitoring

Authors: Kate Brandis, Maria Bellio, Diane Callaghan, Emily Webster, Roxane Francis, UNSW

Objectives

To collect detailed information on nesting success of colonially breeding waterbirds at Reed Beds, Barmah-Millewa Forest.

Methods

Within the colony site a random subset of nest clumps were marked using numbered cattle ear tags (Figure 1). Each clump consisted of between 2-30 nests. Individual nests were marked using small numbered sheep ear tags. During the first survey marked nests contained eggs or a combination of eggs and very young chicks. Nests without eggs were not marked as it is not possible to determine clutch sizes or reproductive success. Each nest was relocated during each survey and the following variables recorded; number of eggs or chicks, chick development stage, evidence of predation and nest abandonment. Individual nest surveys continued fortnightly until chicks had reached runner stage, where chicks become highly mobile and begin créching behaviour; congregating on nests as large groups of chicks. At this stage it is not possible to attribute chicks to specific nests so counts are done at a nest clump scale. Nest clump monitoring continued until chicks had fledged or were no longer associated with the nest clump site.

Water depth data was recorded at each nest clump each survey using a 1 m folding ruler. Water quality data was collected in 2107/18 at four random locations within the nest monitoring area at each colony site during each survey. Water quality variables recorded included: dissolved oxygen, temperature, pH, conductivity, total dissolved solids.

To minimise disturbance to the colonies all ground surveys were limited to two periods during the day; early morning (6-11am) or late afternoon (3-8 pm) to avoid causing heat stress to nesting birds and their offspring. Access to the colony sites was by canoe. Surveys were conducted in accordance with UNSW Animal Care and Ethics 14/148A and NSW Scientific Licence 101452.

The nests of three colonially breeding waterbird species are monitored in Reed Beds, Barmah-Millewa Forest: Australian white ibis (*Threskiornis molucca*), Straw-necked ibis (*T. spinicollis*), and Royal spoonbill (*Platelea regia*). On-ground nest surveys were conducted over three breeding seasons 2015-2018. (Table 1).

Table 1 Details of on-ground nest surveys at Reed Beds, Barmah-Millewa 2015 – 2018.

Breeding season	No. of surveys	Total clumps marked	Total nests marked	Straw-necked ibis nests	Australian white ibis nests	Straw-necked/Aust. white ibis	Royal spoonbill nests
2015	3 (9/11, 2/12, 16/12)	29	167	22	101	22	20
2016/17	5 (15/11, 3/12, 15/12, 27/12, 6/01/2017)	91	506	40	278	161	27
2017/18	7 (16/11, 30/11, 16/12, 28/12, 11/01, 27/01, 8/02/2018)	47	153	0	113	0	40

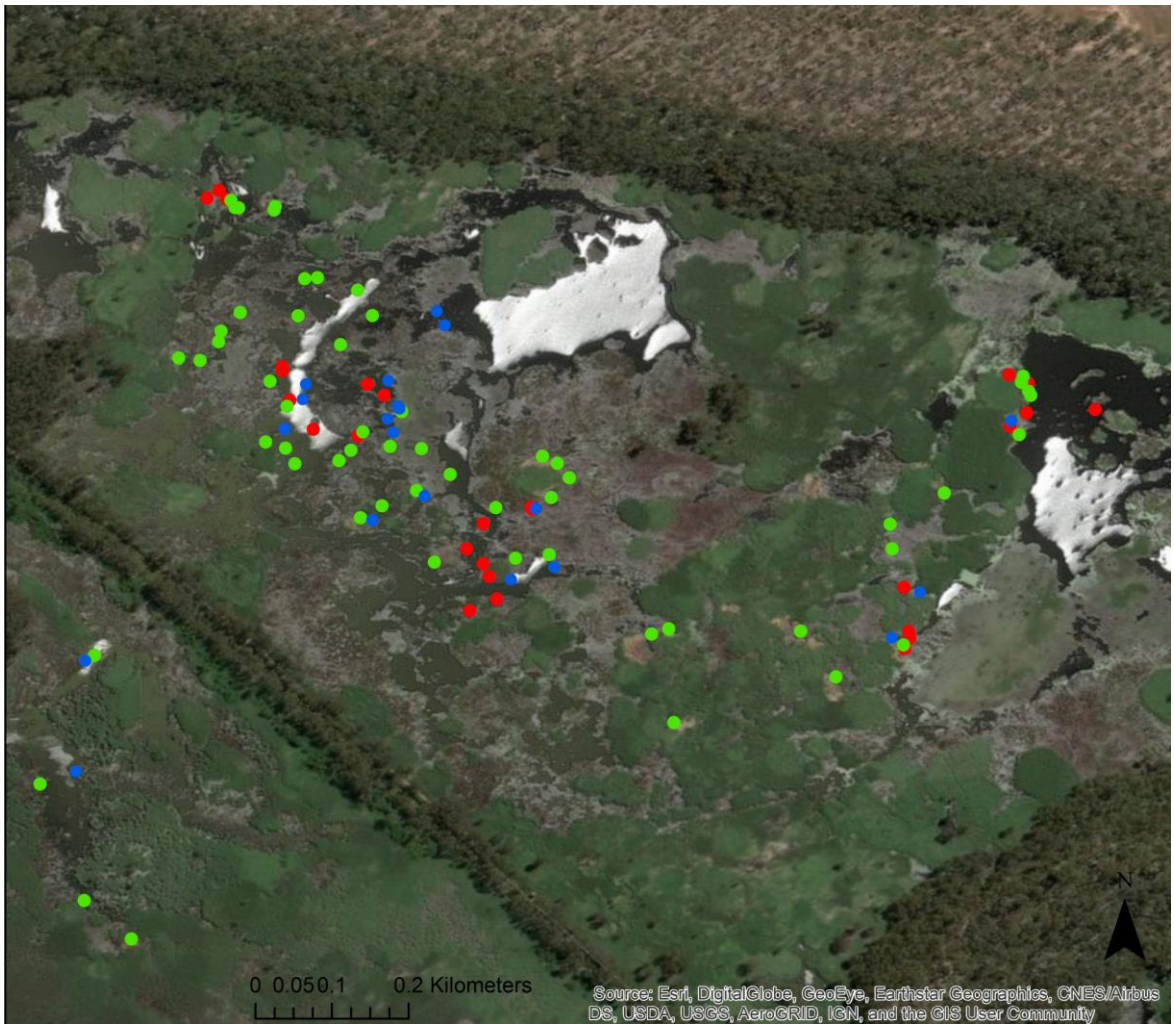


Figure 1 Locations of nest clumps monitored in Reed Beds monitored during 2015 (red dots), 2016/17 (blue dots) and 2017/18 (green dots).

Calculating reproductive success

Reproductive success was calculated for each period between surveys and across all surveys. Nest success is a measure of a particular nest producing offspring. This is a difficult measure to obtain for species such as ibis which crèche. Créching by birds is where young leave their nest to form groups with other young birds. For ibis this typically occurs from ~38 days (12 days post hatching). For this reason, nest success is not a measure that can be used reliably for ibis post the creching stage. Depending upon timing of the first survey, and chick development at this time it may be possible to calculate nest success for the first survey period, but not after that.

Offspring success is a combined measure of eggs and/or chicks that survive compared to total eggs laid. This measure is calculated for the period between each survey and over the total nesting period. Between survey measures, allow for greater understanding of the timing of mortality (e.g. due to predation, starvation, abandonment) with relation to chick development stage and hydrological (e.g. water depth) and environmental (e.g. weather events, temperature) conditions. The numbers of eggs and chicks per nest were measured on the first survey, but due to créching behaviours this is not always possible for second and subsequent surveys. For second and subsequent surveys offspring numbers are calculated for the clump (group of nests) and then totalled for the colony.

Development stages

Egg laying and chick development was staggered across the colony site with many nests being established after the first survey. Royal spoonbill began nesting later than ibis with most eggs being laid in early December. Eggs and chicks were counted per nest from the egg stage (days 1-20) until runner stage (days 30-35). At runner stage chicks are mobile and move between nest sites making it difficult to obtain accurate counts of young.

Surveys are timed to coincide with chick development stages, however this can vary depending upon the timing of the initial survey and any additional nesting that occurs.

Survey	Chick development stage
T1	Egg stage (1-20 days); chick stage (days 21-25)
T2	Chick /squirter stage (days 26-30)
T3	Squirter/runner stage (days 31-35)
T4	Runner/flapper stage (days 36-40)
T5	Fledging (days 41-45) On-going surveys – monitoring cohort 2 nesting
T6	On-going surveys – monitoring cohort 2 nesting

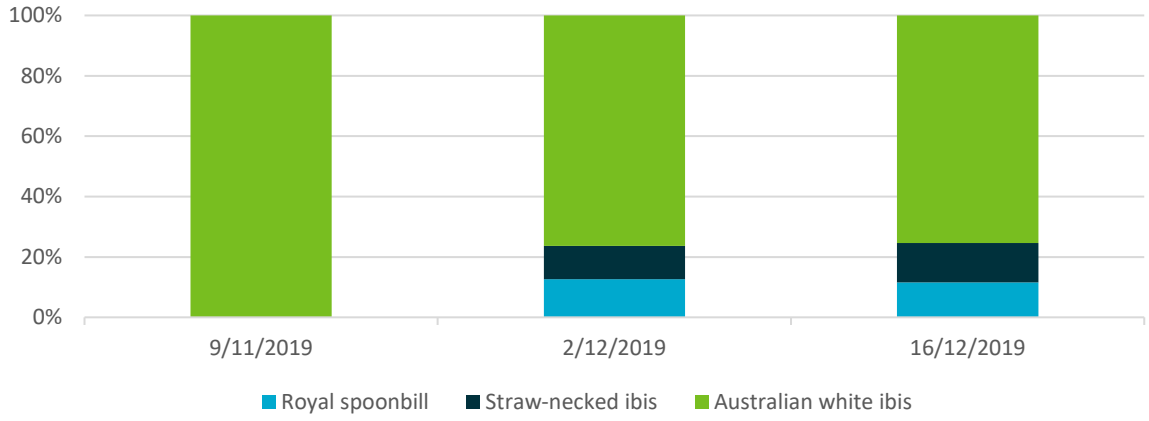
Results

Colonial waterbirds nested in Reed Beds in each of the three years of EWKR 2015 – 2018. Nesting period was generally October – February. Ibis nests were built primarily on *phragmites australis* with royal spoonbill also nesting in Eleocharis spp. along channel edges.

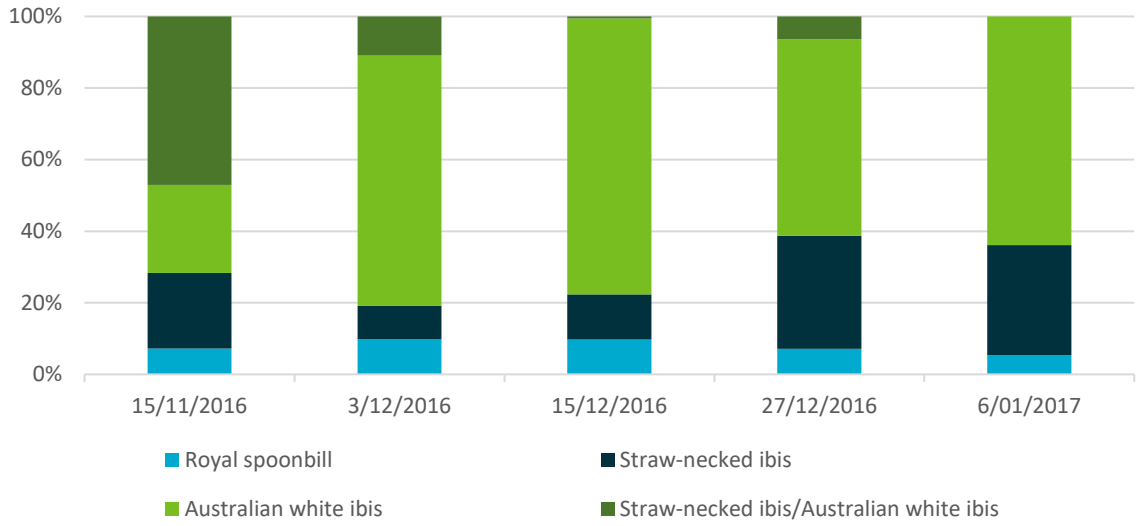
Species composition of the colony

Australian white ibis were consistently the most abundant nesters at Reed Beds over the three breeding seasons 2015 – 2017. Australian white ibis and Royal spoonbill nested in all years surveyed, while Straw-necked ibis nested in 2 out of the 3 years (2015, 2016/17, Figure 2).

a)



b)



c)

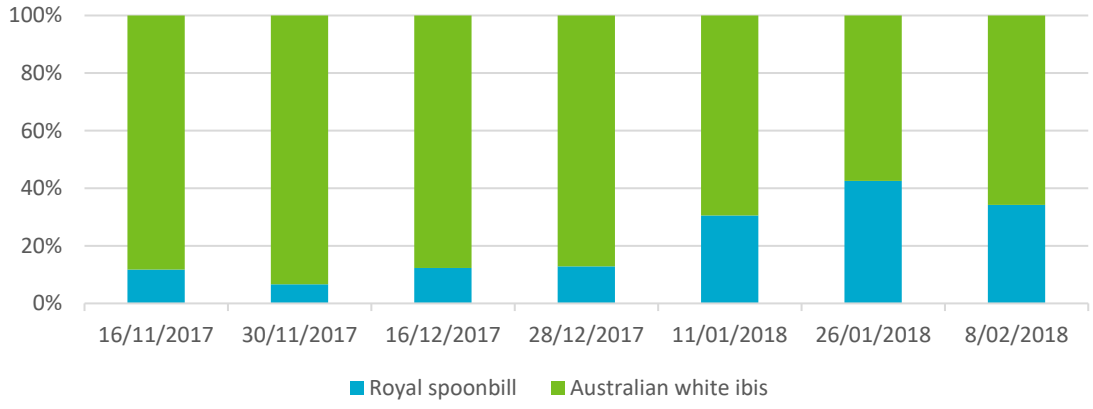


Figure 2 Species composition of Reed Beds colony a) 2015, b) 2016/17, c) 2017/18.

Reproductive success

2015

Surveys undertaken in 2015 were part of a pilot project for method testing and development. Consequently, only three surveys were completed. During the first survey (T1) (Table 1) in early November only nests with eggs of Australian white ibis were present (cohort 1) (Figure 3). During survey 2 (T2), in early December there were additional Australian white ibis nests (cohort 2), and new royal spoonbill and straw-necked ibis nests. Reproductive success varied between stages of development corresponding with nest surveys, with losses highest during egg and chick stages (T1 – T2) (Figure 3). Australian white ibis had the highest reproductive success rate observed during these limited surveys. Australian white ibis that nested later (cohort 2) were more successful than birds that nested early (cohort 1).

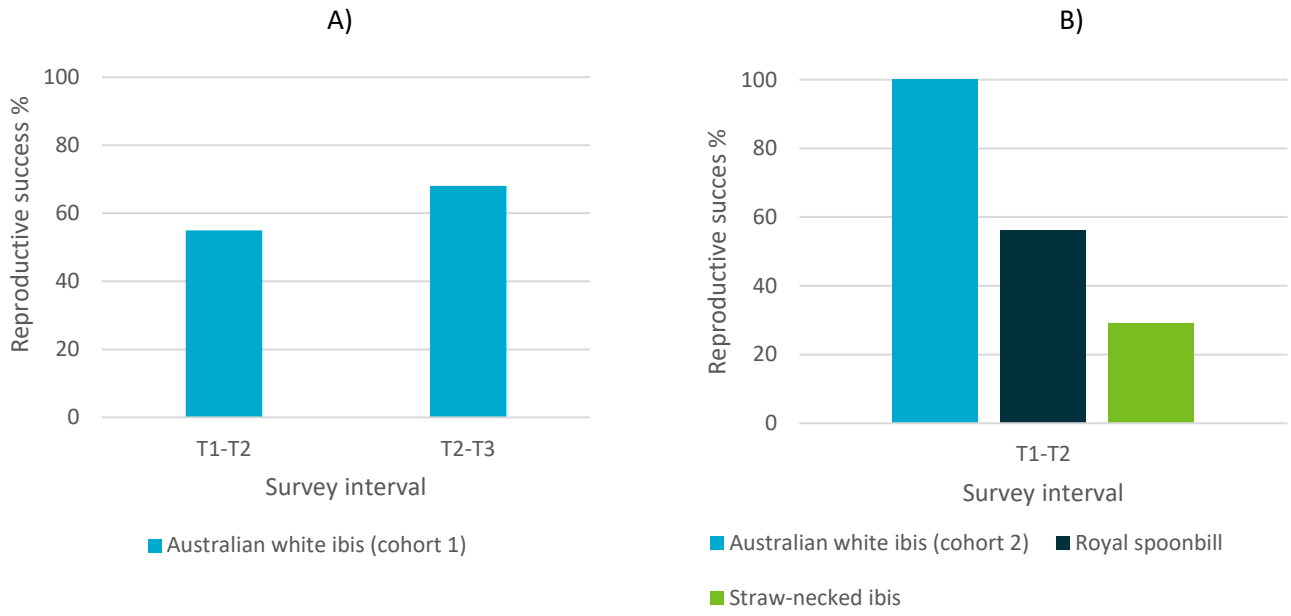


Figure 3 Reproductive success for colonial waterbird breeding in Reed Beds, Barmah-Millewa 2015 A) cohort 1 – the first nester and B) cohort 2 – second nesters.

2016/17

Five surveys were undertaken in 2016/17. During the first survey Australian white ibis and straw-necked ibis were nesting. Royal spoonbill began nesting in the interval between surveys T1 and T2. Reproductive success varied between surveys which corresponded with chick development stages. Australian white ibis had the highest success rates for cohort 1 (Figure 4) while cohort 2 had lower rates for T1-T2 (egg/chick), but higher rates for T2-T3. Reproductive success rates for straw-necked ibis were higher for early nesters (cohort 1) (Figure 4) than later nesters (cohort 2). Royal spoonbill had consistently high reproductive success rates (>60%) for each stage of nesting (Figure 4).

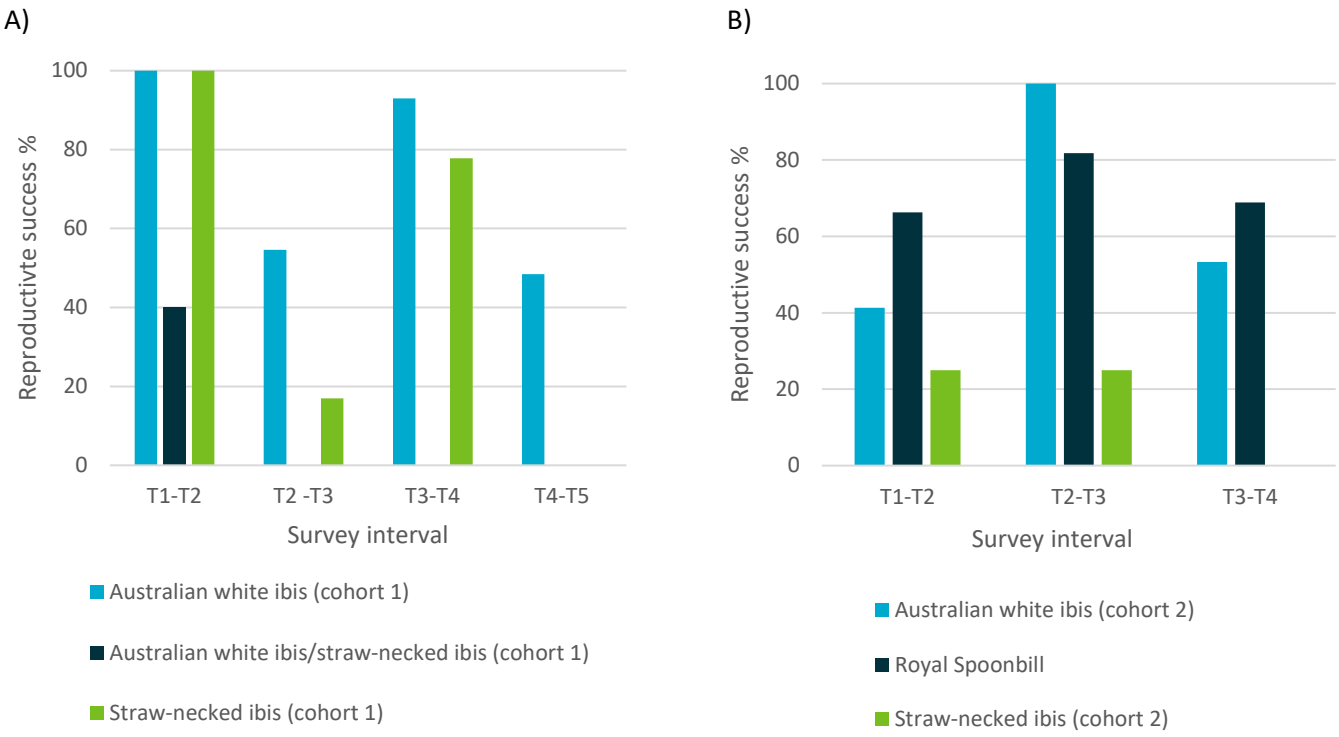


Figure 4 Reproductive success for colonial waterbird breeding in Reed Beds, Barmah-Millewa 2016/17 A) cohort 1 – the first nesters and B) cohort 2 – second nesters.

2017/18

Two colonial species nested at Reed Beds in 2017/18, Australian white ibis and royal spoonbill. 2017/18 was a protracted nesting season with 3 nesting cohorts for Australian white ibis (Figure 5) and 4 nesting cohorts for royal spoonbill (Figure 6). To account for this a total of seven nest surveys were done (Table 1). Australian white ibis early nester (cohort 1) had the highest reproductive success rates (>70%) at each survey interval (Figure 5) while royal spoonbill later nesters (cohort 3) had the highest reproductive success rates (Figure 6).

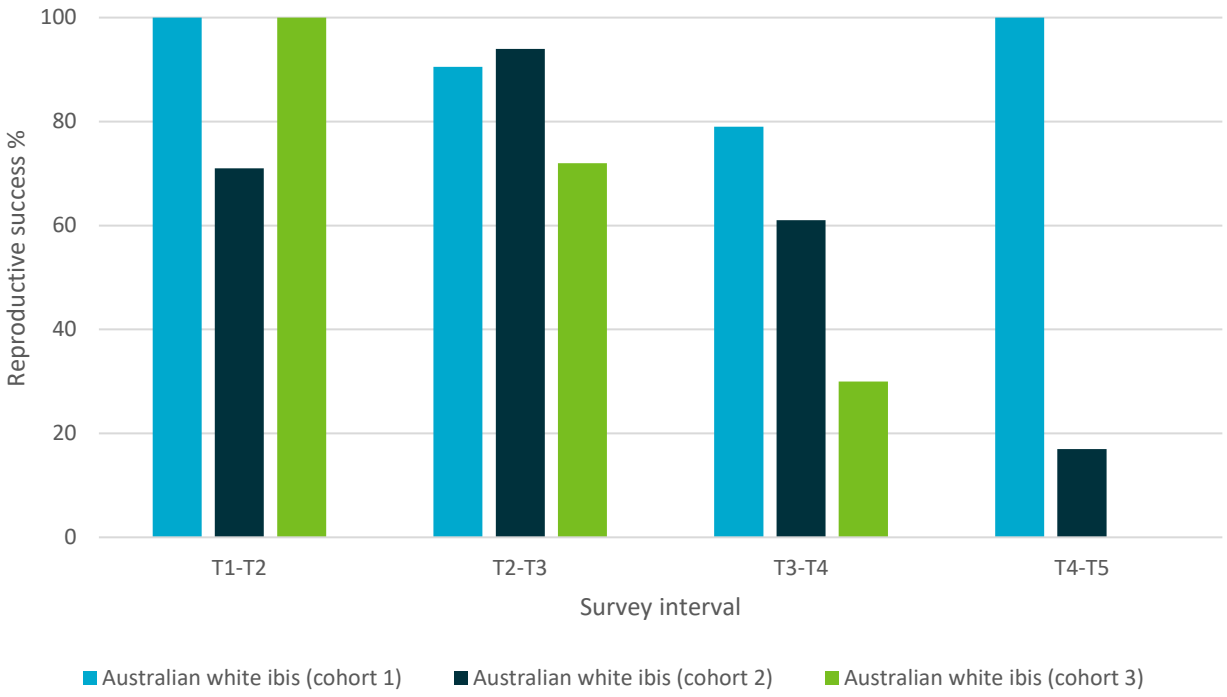


Figure 5. Reproductive success for Austrlaian white ibis in Reed Beds, Barmah-Millewa 2017/18.

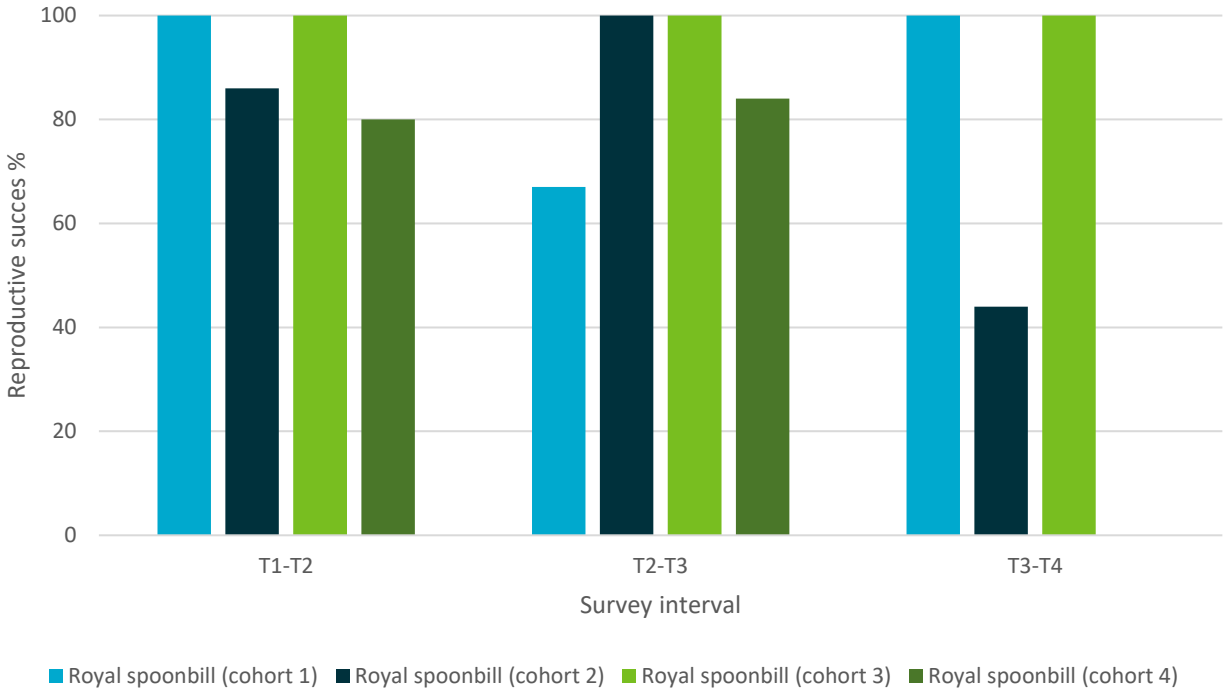


Figure 6 Reproductive success for royal spoonbill in Reed Beds, Barmah-Millewa 2017/18.

Water depth

Water depth at the start of nesting was deepest in 2016/17 with a median depth of 80cm (2015 – 70 cm; 2017/18 – 57cm). Water levels fell throughout the nesting period in 2015 (Figure 7) and 2016/17 (Figure 8), but remained stable during 2017/18, although relatively shallow when compared to other years (Figure 9).

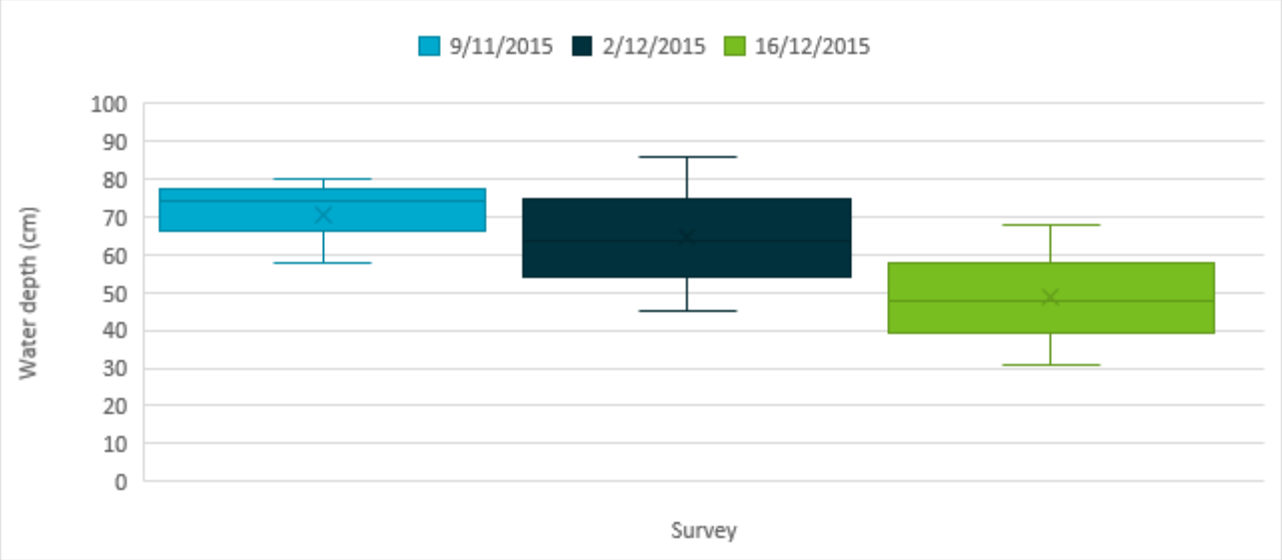


Figure 7 Mean (line), median (cross), minimum and maximum water depth at nest sites in Reed Beds colony site during each 2015.

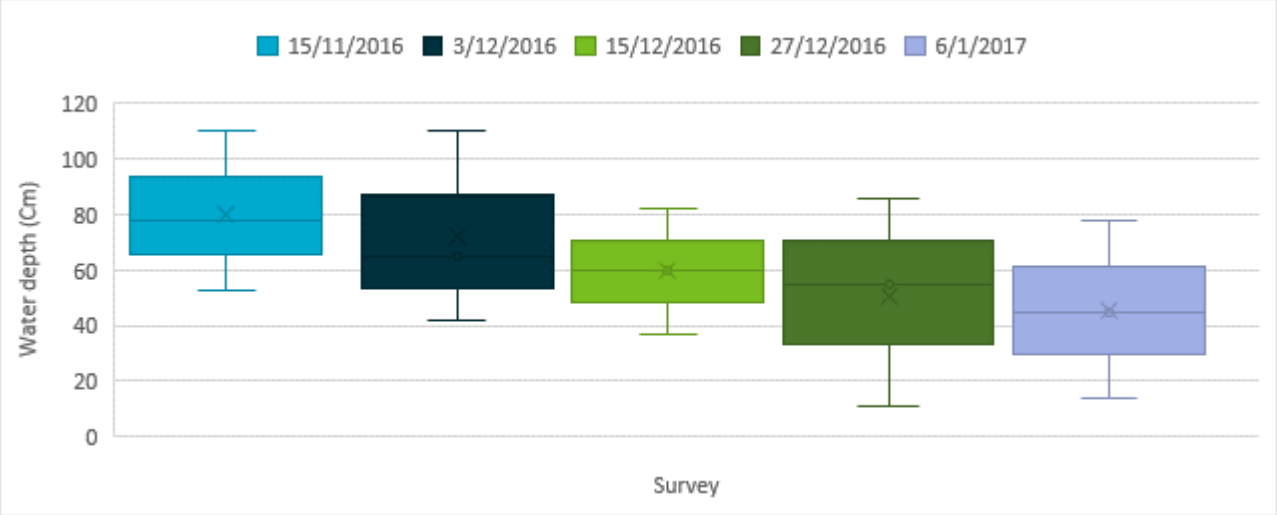


Figure 8 Mean (line), median (cross), minimum and maximum water depth at nest sites in Reed Beds colony site during each 2016/17.

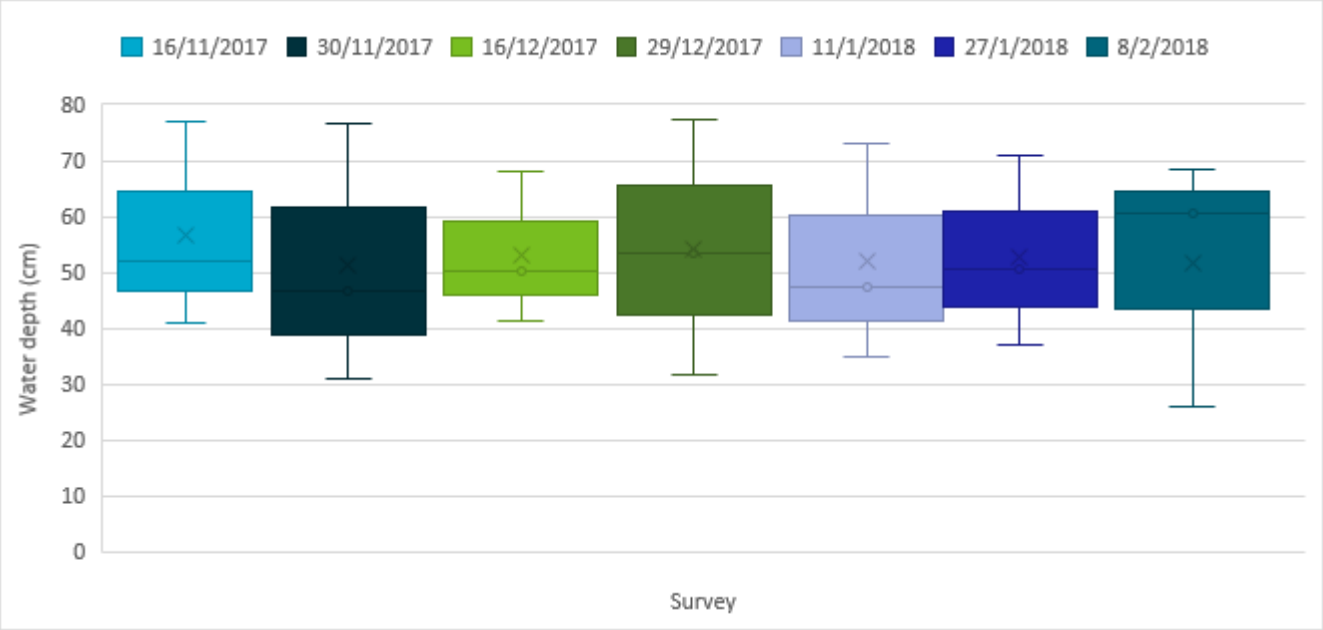


Figure 9 Mean (line), median (cross), minimum and maximum water depth at nest sites in Reed Beds colony site during each 2017/18.

Interaction between water depth and reproductive success

Analysis of water depths and reproductive success in 2015 and 2016/17 found that relationships between reproductive success and water depths were strongest for straw-necked ibis. Offspring success of straw-necked ibis was significantly lower than that of Australian white ibis, with greatest loss occurring at young chick and egg stages. Straw-necked ibis responded later to flow thresholds triggering breeding, nesting later than Australian white ibis (see "Waterbird breeding response to flow" activity report).

Water quality

Random water quality measurements were taken during all surveys. Average values consisting of all sites at each survey time show that values were relatively consistent and within normal range throughout the season (Table 2).

Table 2 Mean water quality values for each survey date 2017/18.

Date	pH	Dissolved Oxygen %	Temp (°C)	Conductivity (µS/m)	Total Dissolved Solids (ppm)
16/11/2017	6.52	34.47	20.58	31.22	25.08
30/11/2017	6.47	31.87	27.48	32.65	24.75
16/12/2017	6.41	21.02	24.36	37.36	26.5
28/12/2017	6.37	12.36	25.07	40.34	27.42
11/01/2018	6.44	21.28	25.21	38.69	24.83
27/01/2018	6.72	40.62	27.35		25.2
8/02/2018	7.05	40.12	27.6		25

Discussion

The colony was largest in 2016/17 with all three species nesting. Reproductive success was similar across all years although highest in 2017/18. In all three years there was staged nesting with different cohorts of either one or more species.

This study has provided a better understanding of timing of breeding and variability between years including colony size and species breeding.

Water management application

This study has demonstrated that Reed Beds is a colony site that is used annually by colonially breeding waterbirds. Australian white ibis and royal spoonbill bred in each of the three years, while straw-necked ibis bred in two years. The role of regular small breeding events is likely to be a key contributor to maintaining populations. While unlikely to be contributing to population growth, they are important in maintaining genetic diversity in the population and providing some new recruits each year.

Further analysis of the interactions between water depth and reproductive success will inform water managers of what water depths in the colony site are optimal for successful reproductive success.

Murray-Darling Basin Environmental Water Knowledge and Research Project

Waterbird Theme Research Report

Appendix 5

Ibis and spoonbill nest and colony mapping using drone imagery

Authors: Kate Brandis, Justin McCann, Mitchell Lyons

Objectives

To acquire high resolution imagery of the colony area from which accurate nest counts and colony boundaries could be obtained.

Methods

Drone image (still photos and video) data were collected during 2016 and 2017 using a DJI Phantom 3 Professional quad-copter, with the stock single sensor red/green/blue (RGB) camera. The drone was launched from the bird hide adjacent to the colony site.

Flights were conducted using parallel flight lines, at ~100 m and speed of 5-10 ms-1 (Lyons et al. 2018; Lyons et al. in review). We aimed to acquire imagery with ~70% forward and lateral overlap to ensure adequate coverage for post-processing.

The drone imagery was processed using the commercial software Pix4DMapper (v4.19, Pix4D SA), which uses a photogrammetry technique called 'structure from motion' to identify points in overlapping images, ultimately generating a 3D point cloud reconstruction of the landscape. The 3D information is then used to generate a digital surface model and an orthorectified image mosaic.

Nests were manually and systematically counted by visual interpretation and each nest marked in a geospatial database. We imposed a 50 x 50 m grid of quadrats on each colony, and digitally annotated every visible nest. We used this gridded method for two reasons: 1) it enabled an observer to sequentially work through the whole colony, while reducing distraction (and computer memory overhead) from surrounding areas; and 2) it reflected real-world practices when users choose only a limited number of training quadrats to manually count nests. During the field work, we also counted nests (in situ) for GPS-tagged clumps at each colony which we used to test the accuracy of the drone-based manual counting.

Results

2016-2017

Drone footage of the Reed Beds Swamp colony was successfully captured on 13-15th December 2016. From this imagery the colony area was estimated at 50 ha and 1,645 nests were counted (Figure 1).

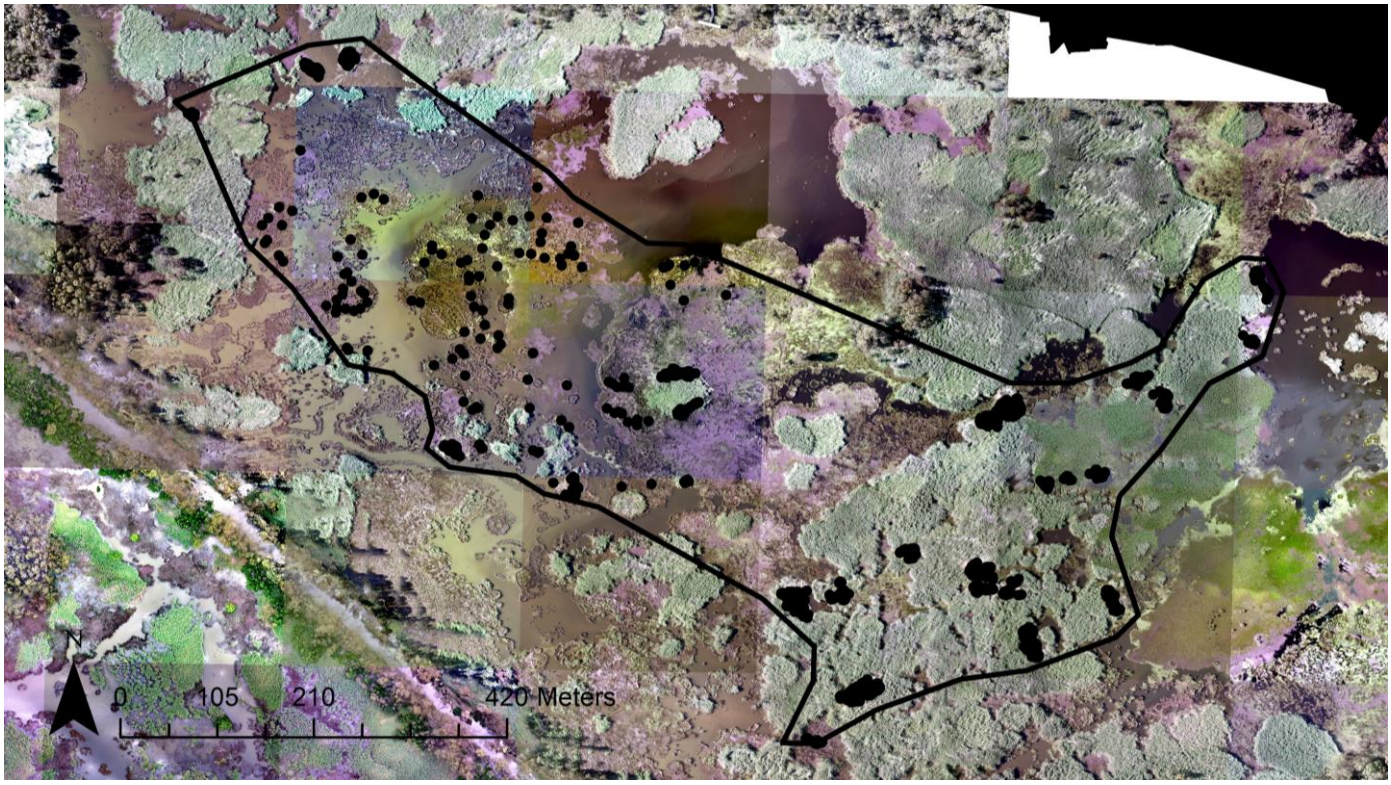


Figure 1 Nest mapping and colony boundary mapping from drone survey 2016. Each dot represents an individual nest (1,645 nests).

2017-2018

Drone footage of the Reed Beds Swamp colony was successfully captured on 16th December 2017. From this imagery the colony area was estimated at 17 ha and 260 nests were counted (102 Royal spoonbill; 148 Australian white ibis; 10 unsure (Figure 2)**.**

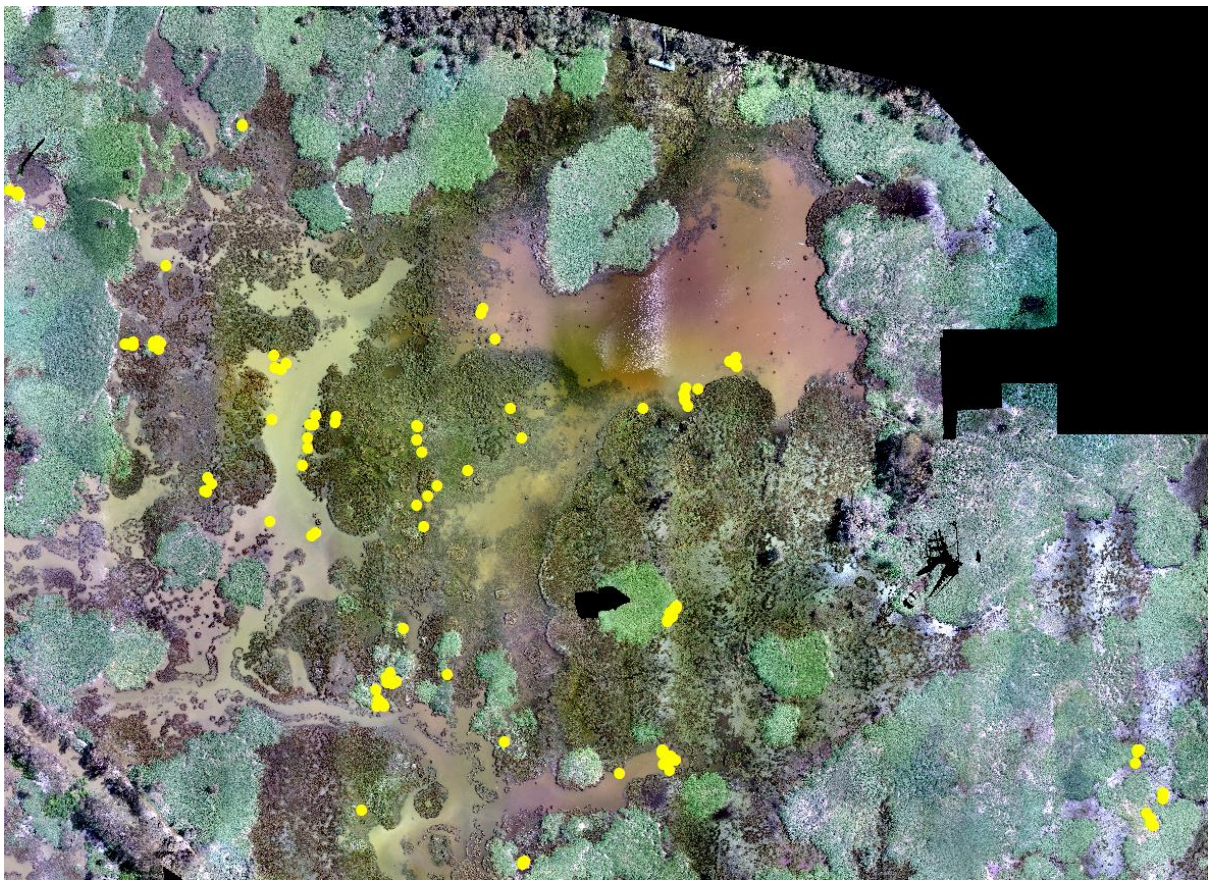


Figure 2 Nest counts from drone survey. Each symbol is an individual nest. 260 nests 2017.

Conclusions and applications

Drone imagery allowed for the accurate counting and locating of nests throughout the Reed Beds colony site. Mapping of nests using this method allows for the accounting of nests that are not accessible by ground counts i.e. those surrounded by vegetation. It also allows for the mapping of vegetation in the colony sites (Dyer et al. 2017).

Water management application

The Reed Beds colony site is a traditional nesting site used each year. The data collected from the drone data analysis provides insights into the habitat layout that the birds are utilising for nesting. This helps inform water managers to ensure that these characteristics can be maintained or enhanced. For example:

- Provision and maintenance of suitable nesting habitat at Reed Beds would be a priority as birds are reusing the site.
- Site knowledge by birds may result in increased reproductive success (Greenwood and Harvey 1982, Haas 1998). Therefore, ongoing provision of suitable nesting habitat at this site may result in more successful breeding events.
- Visual analysis of nest distribution shows that nests are frequently located near areas of open water. This assists water and wetland managers in vegetation management and watering decisions to prevent the infilling of channels and open water areas.

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Lyons, M., Brandis, K., Murray, N., Wilshire, J., McCann, J., Kingsford, R.T., Callaghan, C. (2019 in review *Methods in Ecology and Evolution*) Monitoring large and complex wildlife aggregations with drones.

Lyons, M., Brandis, K., Wilshire, J., Murray, N., McCann, J., Kingsford, R. & Callaghan, C. (in review *Australian Zoologist*) A protocol for using drones to assist monitoring of large breeding bird colonies.

Lyons, M., Brandis, K., Callaghan, C., McCann, J., Mills, C., Ryall, S. & Kingsford, R. (2018) Bird interactions with drones, from individuals to large colonies. *Australian Field Ornithology,* 35, 51-56.

Acknowledgements

Drone surveys were conducted in accordance with UNSW ACEC 16/3B. We wish to thank Diane Callaghan, Roxane Francis, Ben Feit, and Ana Feit for assisting with data collection or processing.

Murray-Darling Basin Environmental Water Knowledge and Research Project

Waterbird Theme Research Report

Appendix 6

Ibis and spoonbill chick energy source assimilation

Manuscript in early stages of preparation for submission to a scientific journal for publication.

Authorship to be confirmed. Current authors include: Paul McInerney, Kate Brandis, Heather McGinness, Ralph Mac Nally.

Background

This component is a Waterbird Theme - Food Web Theme collaboration involving analysis of waterbird scat, regurgitate and feather samples.

- Project designed by Waterbird Theme and Foodweb Theme coordinators and leadership groups
- Samples collected and provided by Waterbird Theme (CSIRO).
- Laboratory work and data extraction from literature completed by the Waterbird Theme (UNSW)
- Collaborative write-up

Three components:

1. Visual ID and wet/dry weights of what chicks are being fed / what birds are eating (regurgitate and scat analyses)
2. Review and calculation of the energy value of the items being consumed (literature review)
3. Analysis of the trophic position of the food items being consumed and the food chain length from basal resources to waterbirds (isotope analyses, including feathers from tracked birds and chicks)

Questions this paper can start to address:
1. What are Australian ibis and spoonbills eating (diet composition)?
2. What is the energy value of different types of ibis and spoonbill food sources?
3. Which food sources appear to be assimilated better than others ~~and vice versa~~?

For each question, there are these sub-levels:

- Are there differences between species?

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- Are there differences between chick, juvenile and adult diets?
- Are there differences between sites?

If we have time, we could also look at: Based on O'Brien and McGinness (2018/19) bulk energy requirements paper, how much of each of these different sources (in energy and biomass) would be needed to raise a chick? How much of each would be needed for an entire breeding event?

Other issues that we can then discuss:

- Are there differences between previous studies and this study?
- Are there differences to what they ate or what was available in the past? If so probable whys and what are the implications?
- For individuals that we have two or three kinds of samples for (feather, regurgitate and scat), are there any patterns or differences to what we would expect?
- How might the availability of high-value and high-assimilation food sources have changed over time (with WRD)?
- What is the food web supporting ibis and spoonbills? Can we describe/conceptually map it?
- How might the food web have changed over time (with WRD)?

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Abstract

The use of environmental water within the basin has often been focused on supporting the completion of waterbird breeding events at key colonial nesting sites. However, managers and policy-makers are becoming increasingly conscious of the need also to manage feeding sites at basin scales. Providing the 'right amounts of water to the right places at the right times' will increase recruitment and, hopefully, elevate the long-term persistence of waterbird populations in changing climates. To understand how environmental water provision ('ewater') can support important dietary components of colonial waterbirds, we must first better understand their feeding requirements. This activity set out to answer three primary questions:

1. What are Australian ibis and spoonbills eating (diet composition)?
2. What is the energy value of different types of ibis and spoonbill food sources?
3. Which food sources appear to be assimilated best?

Based on regurgitate sampling (the fresh contents of a bird's crop), diets of Royal spoonbills and Straw necked ibis are significantly different. Royal spoonbills fed predominately on fish and yabbys (71% and 15% of total diet respectively, **Error! Reference source not found.**). In contrast straw–necked ibis fed primarily on beetles (49%), centipedes (11%) and other indeterminate insects (11%). These results from regurgitates form an interesting contrast with diet results based on scats (Section 2.8), suggesting that sample type is important for diet assessment.

Overall, the calorific value of the primary diet sources for each species were similar (Fish ~ 5761 compared to beetles ~ 5556 calories/gm dry weight), though slightly higher for Royal spoonbills. Stable isotopes ratios of $\delta^{15}N$ and $\delta^{13}C$ from feathers suggest that for both species, regurgitate contents largely correspond to what food items the birds are assimilating. However, for Royal spoonbills isotopic analyses indicated that yabbys and prawns were under represented in regurgitate samples compared to tissue assimilation and for Straw–necked ibis, frogs and crickets were under represented in regurgitate samples compared to tissue assimilation (**Error! Reference source not found.**).

Our results suggest that consideration of specific waterbird species trophic requirements are important when tailoring management of environmental watering to maximise food availability to support waterbird recruitment. For example, if Royal spoonbills are known to breed in a particular location, watering should be targeted to maximise the abundance of aquatic food resources, such as small fish, yabbies and prawns. If however if a location is known to support primarily straw necked ibis colonial breeding, environmental water might be better used to promote diverse terrestrial/riparian foraging habitat that supports a diverse array of vertebrate and invertebrate prey.

Introduction

The global scarcity of freshwater effectively has arisen from the almost unbridled demands by humans for domestic use, agriculture and mining (Rodell *et al.*, 2018). In many parts of the world, this effective scarcity will continue to be exacerbated by climate change (e.g. Carrão *et al.*, 2016), or at least, by the increasingly high amplitudes in precipitation (O'Gorman, 2015). Global warming trends have elevated evapotranspiration, which lessens availability for the biota and humans alike (Yang *et al.*, 2019). From an ecological perspective, the abstraction of freshwaters for human consumption when coupled with climate change has meant that many freshwater ecosystems have become deprived of the amounts of water that they used to experience, both on average and in the exposure to large flooding events (Thompson *et al.*, 2013). Many ecological components of freshwaters, riparian zones and floodplains have been adversely affected by lessened water availability, including fish (Bond *et al.*, 2015), aquatic invertebrates (Paul *et al.*, 2018), riparian vegetation (Thomson *et al.*, 2012; Capon & Pettit, 2018), in-stream ecological processes (e.g. 1° production, Cook *et al.*, 2015) and colonial waterbirds (Kingsford, 2000; Leslie, 2001). Some governments have responded to the decline in ecological condition by re-acquiring rights ('environmental share') of water (e.g. Adamson & Loch, 2018) and using this share to increase average flows or to complement floods in volume or duration (Poff *et al.*, 2017).

Our focus is on colonial waterbirds of Australia's most important river basin and agricultural bread-basket, the Murray-Darling Basin in southeastern Australia (MDB, a basin similar in area to the Republic of South Africa) and, particularly, on the energy requirements and provisioning needed to successfully produce fledglings. The use of environmental water within the basin has often been focused on supporting the completion of waterbird breeding to fledging at key colonial nesting sites (Brandis *et al.*, 2011; Arthur *et al.*, 2012). However, managers and policy-makers are becoming increasingly conscious of the need also to manage feeding sites at sites distributed throughout the MDB because of the birds' high mobilities. Providing the 'right amounts of water to the right places at the right times' is likely to increase recruitment and, hopefully, elevate the long-term persistence of waterbird populations in changing climates. To understand how environmental water provision ('ewater') can support important dietary components of colonial waterbirds, we must first better understand their feeding and dietary requirements.

The family Threskiornithidae comprises 34 species of large wading ibises and spoonbills that are distributed almost worldwide. Straw-necked ibises (*Threskiornis spinicollis*, SNI) and royal spoonbills (*Platalea regia*, RS) are two broadly distributed Australian species that are of particular interest for ewater managers and policy-makers because both species' nesting and foraging habitats are influenced by water availability. SNIs use their markedly curved bills to probe for prey in shallow water, mud, damp soil, crevices, holes, logs, and grasses, consuming a wide range of prey, including terrestrial and aquatic invertebrates and small vertebrates, such as fish and frogs (Table) (Carrick, 1959). SNIs forage and travel during the day and roost at night in trees (McKilligan, 1979). RSs are active hunters that use a range of foraging methods seeking mostly aquatic prey while wading in shallow water. RSs travel and forage in

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Commented [Uo8]: Are they also good models for overseas colonial waterbirds to draw the wider interest?

both day and night. Juveniles are usually much less successful foragers than adults, but any differences between adult, juvenile and chick foraging effectiveness and diets are poorly understood. Past studies have found that the RS diet is diverse and differs by location (Mathews, 1909; Vestjens, 1975; Lowe, 1982)(Table). Fish are the dominant prey in freshwaters, while crustaceans are more commonly taken in estuarine or coastal environments (Hobbs, 1957; Howard & Lowe, 1984; HANZAB., 2006)

The feeding ecology of birds has been examined historically by linking stomach or crop contents with data on foraging behavior. However, stomach or crop contents reflect short-term intake and differing prey digestibility may bias assessments of the components that are assimilated by consumers. Ecological tracers, such as stable isotopes, are increasingly used to overcome the limitations of standard methods because such tracers can reflect nutrient and energy assimilation from the diet at the time of tissue synthesis.

Based on earlier dietary studies, we expected a greater reliance on terrestrial food sources for *T. spinicollis* than for *P. regia*. Due to the differing foraging strategies of the two species, we expected there to be little dietary overlap between the two species and, although we expect *T. spinicollis* to have greater dietary breadth, we anticipated that *P. regia* would have more spatial diet plasticity due to decreased potential prey sources. We evaluate our results in the context of maximizing environmental water for water bird recruitment success.

Methods

Sample collection

Samples were collected from four nesting sites in the Murray-Darling Basin, Australia: Macquarie Marshes (Monkeygar Swamp, New South Wales), Barmah-Millewa Forest (Reed Beds Wetland, New South Wales), Kow Swamp (Victoria) and the Kerang Lakes (Reedy Lake, Victoria).

Data were obtained from three sources. (1) Feathers. Three to five feathers were taken from the breast or belly of birds captured for satellite tracking or allometry measurements as part of related research (*for now, refer to O'Brien and McGinness 2018*). Feathers from 30 individuals per species were analysed, including adults, juveniles and chicks (Table). (2) Regurgitates. Samples were collected opportunistically from 10 individuals per species. SNIs and RSs frequently regurgitate the contents of their crop as a defence mechanism if disturbed. Regurgitates were collected incidentally (not by induction) from the same birds sampled for satellite tracking, allometry, or from nearby nests. No regurgitate samples were available from the Macquarie Marshes. (3) Scats. Ten scat samples per species were collected from the individuals referred to above or opportunistically from nests. Samples were frozen for transport to the laboratory prior to analysis.

Table Number of samples of each type by species and age group

Species and age group	Feather	Regurgitate	Scat	Grand Total
RS	**30**	**10**	**10**	**50**
Adult	1			1
Chick	15	2	9	26
Juvenile	14	8	1	23
SNI	**30**	**10**	**10**	**50**
Adult	10			10
Juvenile	20	10	7	37
Unknown			3	3

Commented [MH(BM9]: **This is important and should be expanded and emphasised. Our paper is original and useful because it does two extra things: 1. It looks at the energy value of the prey being consumed; 2. It looks at what is actually being assimilated. It may also do a third important thing: Calculate how much of each type of prey would be needed to raise a chick based on O'Brien and McGinness 2018/19**
RM: There have been severe critiques of stable-isotopic methods too, so that need bearing in mind.
Some basic information on items consumed and relative numbers of items consumed/identifiable is available in the literature based on stomach, crop or scat contents and some observations. This information is limited and varies greatly in quantity and quality by species, study, and location. I have some draft tables that can be updated for inclusion. For refs see HANZAB, Vestjens et al., Kate's Narran study, Emily Webster's study, Obrien and McGinness 2018 allometry and energy study. In summary, there is little or no existing information on the:
-mass / weight / energy of food items consumed or required by any life stage
-quality of food items consumed
-assimilation of food items consumed
-relative abundance/availability of food items consumed in foraging habitats (let alone how this varies in different places or at different times or with flow or flood characteristics)
-what drives differences in the relative abundance/availability of food items and their consumption
-differences between chick, fledgling, juvenile and adult diets

Commented [MH(BM10]: We are doing more than comparing sites.

Commented [MH(BM11]: Need a short para on differences between the two species in general and in foraging strategies and habitats and diets, including what is already known about diet, before bringing it up below. I have tried to do this above.

Commented [Uo12]: Perhaps use acronyms instead of species names?

Commented [RN13]: Not sure how this arises? Decreased relative to what?

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RM: Supp Info presumably

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Grand Total	60	20	20	100

Table Number of samples of each type by site and species

Site and species	Feather	Regurgitate	Scat	Grand Total
Barmah Millewa	**31**	**12**	**4**	**47**
RS	16	8	3	27
SNI	15	4	1	20
Kerang Lakes	**17**	**3**	**9**	**29**
RS	12	1	6	19
SNI	5	2	3	10
Kow Swamp	**7**	**5**	**4**	**16**
RS	2	1	1	4
SNI	5	4	3	12
Macquarie Marshes	**5**		**3**	**8**
SNI	5		3	8
Grand Total	**60**	**20**	**20**	**100**

Combined table

Site, species and age group	Feather	Regurgitate	Scat	Grand Total
Barmah Millewa	**31**	**12**	**4**	**47**
RS	**16**	**8**	**3**	**27**
Adult	1			1
Chick	3		2	5
Juvenile	12	8	1	21
SNI	**15**	**4**	**1**	**20**
Adult	5			5
Juvenile	10	4	1	15
Kerang Lakes	**17**	**3**	**9**	**29**
RS	**12**	**1**	**6**	**19**
Chick	10	1	6	17
Juvenile	2			2
SNI	**5**	**2**	**3**	**10**
Juvenile	5	2	3	10
Kow Swamp	**7**	**5**	**4**	**16**
RS	**2**	**1**	**1**	**4**
Chick	2	1	1	4
SNI	**5**	**4**	**3**	**12**
Juvenile	5	4	3	12
Macquarie Marshes	**5**		**3**	**8**
SNI	**5**		**3**	**8**
Adult	5			5
Unknown			3	3
Grand Total	**60**	**20**	**20**	**100**

Sample processing

All samples were frozen immediately after collection. Total wet weight of each regurgitate sample was recorded, then samples were thawed and sorted removing all body parts (whole or incomplete) of prey items and any plant material. Wet weight for each food item was recorded. Food items were oven dried (60^0C for 48 hours) and dry weight was recorded. Food items were classified to order when possible. The twelve most abundant food type in the regurgitates were analysed for ^{13}C and ^{15}N stable isotopes. Five replicates of each food item were sampled. If five replicates were not available, all samples of that item were pooled and five sub-samples were analysed. Samples were ground using a ball grinder (instrument details here).

Total wet weight and dry weight were recorded for each scat sample.

Feathers were cleaned using a mixture of 2:1 chloroform:methanol solvent as per the method in Paritte and Kelly (2009) to remove any surface contaminants. The distal tip of the feather was trimmed and weighed for stable isotope analysis. This part represents the first grown part of the feather and a record of diet during growth (Hobson and Wassenaar 2008). Feathers are composed of keratin which is chemically inert once grown, providing a fixed record of diet.

All samples were weighed into tin caps and stable carbon (^{13}C) and nitrogen (^{15}N) isotopes were analysed at the Bioanalytical Mass Spectrometry Facility (BMSF) at the Mark Wainwright Analytical Centre (MWAC) University of New South Wales. A Delta V Advantage Isotope Ratio Mass Spectrometer and Flash 2000 Organic Elemental Analyzer fitted with a Conflo 4 was used.

Statistical analysis

Diet composition

We used 2-factor analyses (fixed factors: species; 2 levels and location; 3 levels) primarily to identify differences in regurgitate composition and dry weight of scats from *T. spinicollis* and *P. regia* at different wetlands. We used permutational analysis of variance (PERMANOVA) in PRIMER v7 (Plymouth Marine Laboratories, 2015) to test for differences in univariate and multivariate response variables among treatments. All regurgitate and scat data were square–root–transformed to meet the assumption of a normal distribution before calculation of Bray–Curtis similarities. We used SIMPER to identify which food components contributed to differences in diet composition between species and locations.

Diet component energy values

Diet component assimilation and trophic positions

Diet mixing models

Stable isotope data were analysed used R package 'Simmr' (Parnell and Inger 2016). Models were developed for each species at each wetland using only those food items relevant to the species; i.e. prey items sources from *P. regia* regurgitate samples were used in *P. regia* models. We ensured dietary items were within the convex hull that was the 95% confidence interval (Smith et al 2013). Consumer samples that were outside this range were removed from analyses.

Commented [Uo22]: Is this by species, by species age class, or generally?

Commented [KB23]: Do we want a table/list of what these were? It becomes obvious in the results. In Supp Info if at all. Might have to be deposited in Dryad depending on journal

Commented [KB24]: KB to get instrument name

Commented [KB25]: Paritte, J. M. and J. F. Kelly (2009). "Effect of cleaning regime on stable-isotope ratios of feathers in Japanese Quail (*Coturnix japonica)*." The Auk **126**: 165-174

Commented [KB26]: Hobson, K. A. and L. I. Wassenaar (2008). Tracking animal migration with stable isotopes. USA, Elsevier Academic Press.

Commented [Uo27]: What about age classes?

Commented [Uo28]: Location strictly is a random factor – these are not the 'universe' of possible sites and we want to generalize to all sites.

Commented [Uo29]: Not sure BC requires normal distributions. Transforms tend to emphasize or de-emphasize abundant taxa.

Commented [KB30]: Parnell, A., and Inger, R. (2016) Stable Isotope Mixing Models in R with simmr. R package https://cran.r-project.org/web/packages/simmr/vignettes/simmr.html

Commented [Uo31]: Age classes pooled?

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Commented [KB33]: Need to redo models removing some samples based on convex hull analysis

Table: Results of testing for likelihood that the consumer diet can be explained by the food items included in the model. Consumer numbers refer to feather sample. Value is the probability that the food items included in the mixing model explain the diet of this consumer. Bold denotes samples that were outside the 95% confidence range for dietary contribution.

Consumer – *P. regia*	Kerang	Barmah-Millewa		
1	**0.00**			
2	0.40			
3	0.35			
4	0.21			
5	**0.04**			
6	0.09			
7	0.49			
8	**0.00**			
9	**0.00**			
10	0.36			
11	0.18			
12	0.62			
13				
14				
15				

Commented [Uo34]: No idea what this table does ;-)

Results

1. What are Australian ibis and spoonbills eating?
2. What is the energy value of different types of ibis and spoonbill food sources?
3. Which food sources appear to be assimilated better than others and vice versa?

For each question, there are these sub-levels:

- Are there differences between species?
- Are there differences between chick, juvenile and adult diets?
- Are there differences between sites?

For individuals that we have two or three kinds of samples for (feather, regurgitate and scat), are there any patterns or differences to what we would expect?

Commented [MH(BM35]: We could lay this out in a few different ways. One is by sample type (regurgitate, scat, feather); another is by topic, e.g. diet composition, then diet energy value, then trophic positions, etc. I like the idea of building the story up from the basics (composition) to the energy, assimilation and foodweb concepts.

RM – yes, this is very clumsily organized at present and makes for a dull presentation of outcomes

Based on O'Brien and McGinness (2018/19) bulk energy requirements paper, how much of each of these different sources (in energy and biomass) would be needed to raise a chick? How much of each would be needed for an entire breeding event?

Diet composition

Regurgitate samples

There were significant differences in diet composition from regurgitate samples of *P. regia* and *T. spinicollis* (*Pseudo–*$F_{(1, 18)}$ = 2.97, $P < 0.001$), although species' diets were not location dependent (*Pseudo–*$F_{(2, 18)}$ = 0.97, $P = 0.53$). *P. regia* regurgitate samples (dry mass, g) were mostly fish (71%) and yabbies (15%, Figure 1a). Fish contributed 23% to dissimilarity of regurgitate contents between consumer species, accounting for 99% of within-species similarity (SIMPER). *T. spinicollis* diet (dry mass, g) was mostly coleopterans (49%), centipedes (11%) and unidentified insects (11%, Figure 1b). Anuran amphibians and coleopterans each contributed 11% to dissimilarity between regurgitate composition of *T. spinicollis* and *P. regia*, while centipedes contributed 17% to within-species similarity of *T. spinicollis* diets. Fish were a major dietary component for consumers at Kerang (average abundance 13.73, SIMPER), contributing 27% to dissimilarity between Kerang and Kow Swamp and 48% between Kerang and Barmah–Millewa. Centipedes contributed 16% to dissimilarity between consumer diets at Barmah–Millewa and Kow Swamp (average abundance 17.14 and 0.07 respectively, SIMPER).

Age group differences

Scat samples

The mean dry weight of *P. regia* scats (0.20 g ±0.03 SE) was significantly less than scats of *T. spinicollis* (1.37 g ±0.49 SE) (*Pseudo–*$F_{(1, 17)}$ = 5.63, $P < 0.05$). We found significant differences (*Pseudo–*$F_{(3, 17)}$ = 4.20, P =0.01) in scat weights from different locations for each species. For *P. regia*, the mean weight of scats collected from Barmah–Millewa was 0.11 g ±0.03 SE compared to 0.25 g ±0.09 SE at Kerang. For *T. spinicollis*, the mean scat weight was similar at Kow Swamp (0.46 g ±0.37 SE) and Kerang (0.65 g ±0.32 SE) and larger at Maquarie Marshes (2.98 g ±0.81 SE).

Diet-component energy values

Regurgitate samples

Mean calorific values differed between species, with *T. spinicollis* regurgitate (18957 calories ± 5258 SE) greater than that of *P. regia* (13398 calories ± 3683 SE) (*Pseudo–*$F_{(1, 16)}$ = 3.89, $P < 0.05$).

Species differences

Age group differences

Site differences

Scat samples

Species differences

Age group differences

Site differences

Diet component assimilation and trophic position

Commented [MH(BM36]: We don't have to keep these headings – they're just for our own navigation for now. We may only need a table combining all types.

Commented [PM37]: We can test site and species diffs, but not age, since they are all from juveniles, except for two chick samples. More or less Same for scats

Commented [PM38]: I can only find data for dry weight, not composition

Feather samples

Species differences

Age group differences

Site differences

The mean values of $\delta^{15}N$ were similar for the two species (10.44 ± 0.24 for *T. spinicollis* and 10.50 ± 0.23 for *P. regia*), indicating similar trophic positions (Fig. 1). Mean values of $\delta^{13}C$ were significantly different ($t = -4.12$, $P < 0.05$), suggesting diet sources differed between the two species. Regurgitate analyses showed that *P. regia* was more reliant on aquatic food sources than *T. spinicollis*, and these data were supported by relatively depleted mean $\delta^{13}C$ values (–26.27 ± 0.40), indicating that nutrient and energy assimilation for *P. regia* was primarily from aquatic prey items. The more enriched mean $\delta^{13}C$ value (–19.63 ± 0.10) for *T. spinicollis* is consistent with nutrient and energy assimilation primarily arising from a broad range of terrestrial resources.

Bayesian mixing models (Simmr) were used to estimate the percentage contribution of potential diet sources and showed differences in the relative contribution of food sources between locations for the two waterbird species. For *P. regia* at Barmah-Millewa, the crayfish (yabby, *Cherax destructor*) contributed most to dietary proportions (mean 0.62 ± 0.22 SD, Fig. 2a), while in the Kerang Wetlands, fish contributed most (mean 0.37 ± 0.16 SD, Fig. 2b). Results from the mixing model were supported by analyses of regurgitate samples (Fig. 3), with yabbies (as g per dry weight) contributing most to *P. regia* stomach contents at Barmah–Millewa (37%) and fish contributing most at Kow Swamp (99%) and Kerang Wetlands (95%, Fig. 3). While prawns were little represented in stomach contents, they were an important dietary source among all locations from the isotopic mixing models.

For *T. spinicollis*, mixing models showed that coleopterans and anuran amphibians contributed strongly to dietary assimilation at Barmah-Millewa (Fig 2c). As for *P. regia*, these data were supported by manual examination of stomach contents, where beetles contributed 85% and frogs 5% to total diet at Barmah–Millewa (Fig. 3a). At Kerang Wetlands and Kow Swamp, mixing models indicated that *T. spinicollis* diet was evenly spread among all dietary items, with gryllid crickets contributing most to assimilated tissue at both locations. These results were consistent with stomach-content analyses, which revealed high proportions of other gut contents not selected for isotopic discrimination, such as unclassifiable insects, vegetation, and seeds.

Dietary overlap

Ibis diet was more broad, but was similar between locations. There was greater diet plasticity in spoonbills between locations.

Discussion

- Are there differences to what they ate or what was available in the past? If so probable whys and what are the implications?
- How might the availability of high-value and high-assimilation food sources have changed over time (with WRD)?
- What is the food web supporting ibis and spoonbills? Can we describe/conceptually map it?
- How might the food web have changed over time (with WRD)?

Commented [MH(BM39]: Need to break up the following material to fit into the new structure

Commented [RN40]: Some anal referees may ask for a formal test here …

Commented [RN41]: Would be better to cite the values first before stating that they are different

Commented [RN42]: It's either significant or not in frequentist analyses – 0.001 does not mean anything more than 0.05.

Commented [MH(BM43]: Careful – regurgitate is crop contents, not gut. What about scats? Or this this combined?

Commented [RN44]: Needs a citation to support why this deduction can be made

Commented [RN45]: What are the uncertainties in this paragraph?

Commented [RN46]: Many reviewers get narky about mixing Bayes and frequentist analyses in the one MS …

Commented [RN47]: I thought we had only 2 sites?

Commented [RN48]: Provide data

Commented [RN49]: This is confusing – are you referring to the ibis result (1st sentence) or spoonbill results?

Commented [RN50]: Which data – is this inference from isotope or gut-contents analyses?

Commented [MH(BM51]: I have existing food/diet tables based on the literature that can be updated and used for this

Commented [MH(BM52]: I have a couple of existing foodweb diagrams we can use / modify for this

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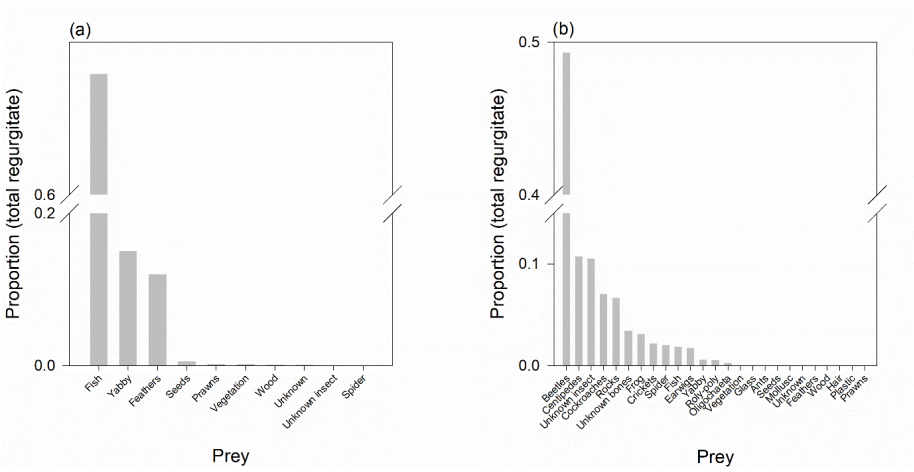


Fig. 1 Proportion of prey taxa (total dry weight (g)) from combined regurgitate from all locations of (a) *P. regia* and (b) *T. spinicollis*.

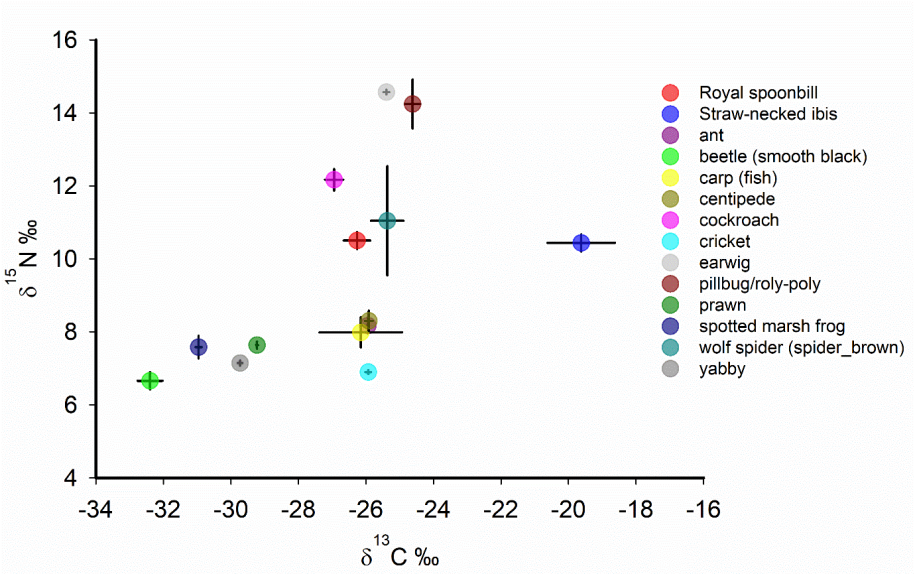


Fig. 2: Mean ±SE stable carbon and nitrogen isotopic signatures of food sources and consumers.

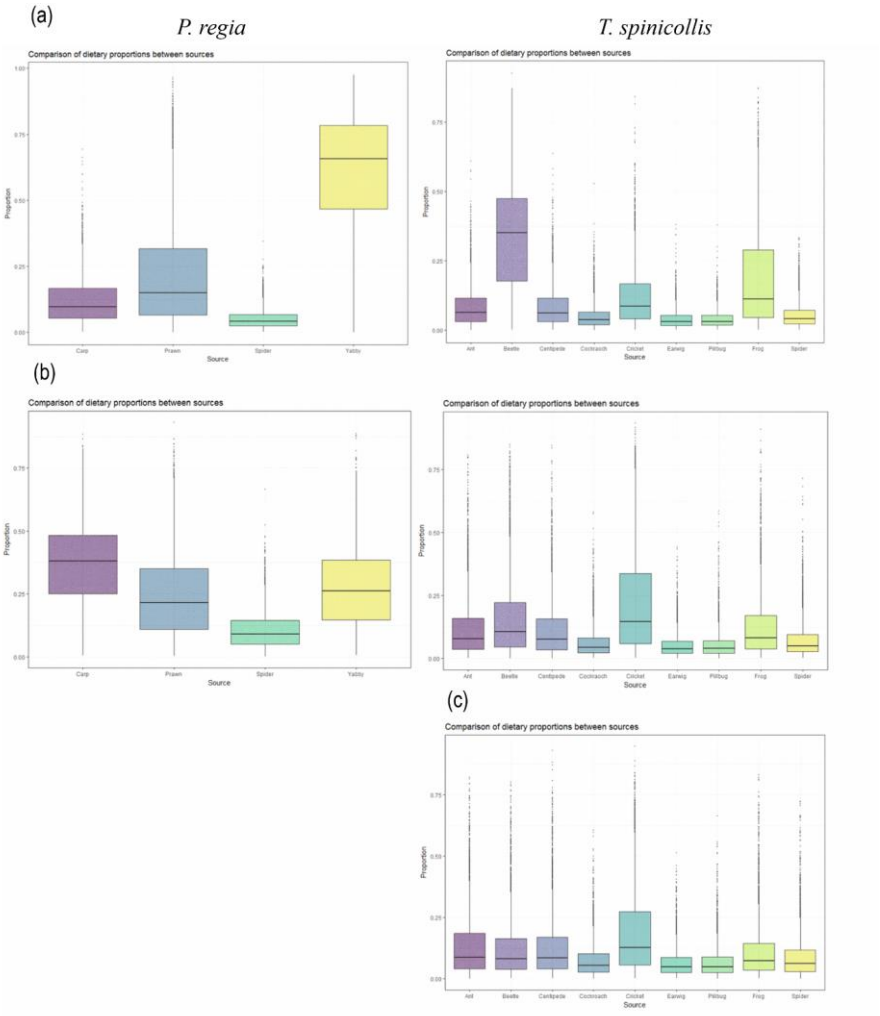
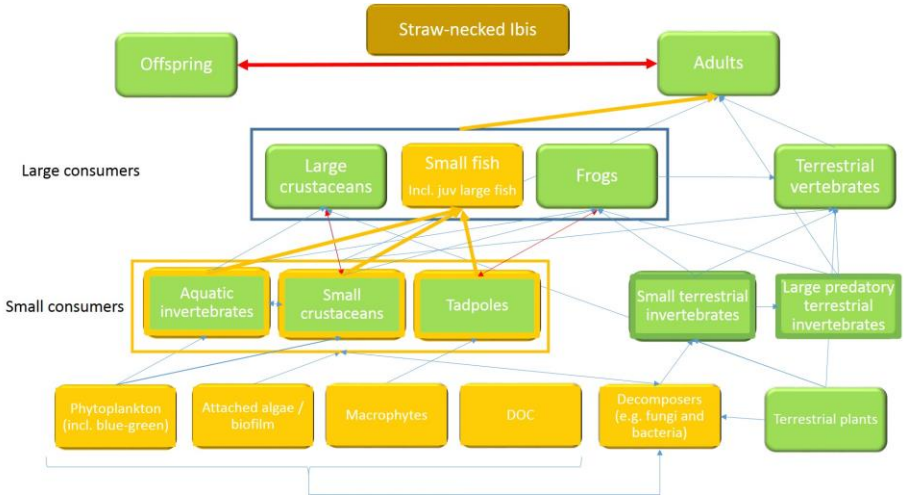


Fig. 2 Diet compositions of *P. regia* and *T. spinicollis* from (a) Barmah Millewa, (b) Kerang Wetlands and (c) Kow Swamp.

This is an old SNI foodweb diagram that we could update


Straw-necked Ibis
Offspring
Adults
Large consumers
Large crustaceans
Small fish
Incl. juv large fish
Frogs
Terrestrial vertebrates
Small consumers
Aquatic invertebrates
Small crustaceans
Tadpoles
Small terrestrial invertebrates
Large predatory terrestrial invertebrates
Phytoplankton (incl. blue-green)
Attached algae / biofilm
Macrophytes
DOC
Decomposers (e.g. fungi and bacteria)
Terrestrial plants

369

370

RESEARCH ARTICLE

Ibis and spoonbill chick growth and energy requirements: Implications for wetland and water management

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Abstract

Colonial-nesting waterbirds such as ibis and spoonbills (Threskiornithidae) can account for a significant proportion of energy flow through wetlands, particularly during large breeding events. However when food availability is reduced, chicks may starve and adults may abandon nests. If the energy required to rear chicks could be calculated, data quantifying prey energy value and availability could be used to develop landscape scale management targets to ensure that food requirements are met to support chicks until they attain independence, thereby maximising recruitment. We calculated ibis and spoonbill chick biometrics and energy requirements through a) an international literature review, extracting and synthesising the best available growth and energy data; b) new measurements of ibis and spoonbill chick biometrics for selected species; and c) analysis of the resulting databases to construct growth curves and predict energy requirements for selected species. Here we present the first models of Royal Spoonbill growth and of Royal Spoonbill and Australian White Ibis chick energy requirements. The total energy estimated to raise a single Royal Spoonbill chick from hatching to independence was 71,290 kJ and for an Australian White Ibis chick was 67,160 kJ. Using prey energy values from the literature, extrapolations indicate that for either species, a nesting event of 1000 nests producing three chicks per nest would require an estimated ten tonnes of freshwater crayfish (*Cherax destructor*) or eight tonnes of invasive juvenile carp (*Cyprinus carpio*) to support chicks from hatching to independence. Effective water and wetland management is critical to optimise both energy availability in foraging sites and breeding success.

Keywords: Threskiornithidae, *Platalea regia*, *Threskiornis molucca*, food, energy, biometrics

Introduction

Colonial-nesting waterbirds such as ibis and spoonbills (Threskiornithidae) can account for a large proportion of energy flow through wetlands and are often used as bio-indicators, usually with a focus on the size, frequency, and success of their breeding events (Kingsford et al. 2013). Colonial-nesting species may congregate in tens to hundreds of thousands (10,000's to 100,000's) to breed, often concentrating foraging efforts within specific limited areas (Jenkins et al. 2009). It is estimated that raising young may double the daily energy requirement of parent birds (Hancock et al. 1992; Kushlan 1977a).

When food availability is reduced during breeding events, colonial-nesting waterbirds may change their behaviour to reduce energy expenditure (Elliott et al. 2014). In cases where the availability of food drops below a certain level, chicks may starve or be abandoned (Brandis 2010; Herring et al. 2011; Kingsford and Auld 2005). For many species, food availability is positively related to the extent, depth and duration of flood inundation of suitable habitat. This is because inundation is required to promote the growth and reproduction of their preferred food sources, for which many colonial-nesting waterbirds have developed specialised foraging techniques and associated morphological adaptations. Prematurely or rapidly falling water levels are therefore an indicator of potential loss of food resources and foraging opportunities (Brandis 2010; Herring et al. 2011; Kingsford and Auld 2005).

Allocation of valuable management resources such as 'environmental water' is frequently targeted at supporting completion of breeding events, usually by supplying water to prevent nest abandonment due to falling water levels (Brandis 2010; Brandis et al. 2011; Kingsford and Auld 2005). In Australia, breeding sites of Straw-necked Ibis (*Threskiornis spinicollis*), Glossy Ibis

(*Plegadis falcinellus*), and Royal Spoonbills (*Platalea regia*) are intensively managed in this way (Brandis et al. 2011; Kingsford and Norman 2002). However, while knowledge exists regarding water management of nesting sites to prevent nest abandonment, there is limited knowledge about food, energy, or foraging habitat requirements to prevent chick starvation and ensure sufficient recruitment to maintain a sustainable population size. Demographic data for Threskiornithidae populations are rare, particularly in Australia, making it difficult to estimate previous or existing recruitment rates or how many chicks must be recruited to ensure breeding adults are replaced in the population. However while ibis and spoonbill species in Australia are not yet listed as species of conservation concern, their populations are thought to be in steep decline due to changes in flood inundation regimes and associated resource loss, and similar species internationally are listed as endangered for similar reasons. While effective recruitment requires survival of eggs, chicks in the nest, juveniles out of the nest, and sub-adults up to adult breeding age (thought to be approximately three years for some species), supporting chick survival at the nesting site is a first step via which managers of wetlands and water can support populations.

Better understanding of chick energy requirements is needed to inform management of food sources and improve chick survival. However direct metabolic measurements are rare worldwide and do not exist for Australian species. In order to estimate chick energy requirements without direct metabolic measurement, data are required describing waterbird biometrics at a range of age stages, in order to apply allometric relationships between metabolism and body size (White 2011). Body mass in particular has a strong allometric relationship with metabolism and accounts for over 90% of interspecific variation in energy expenditure (White 2011). Using allometric models, the daily energy expenditure of a species can be calculated from body mass,

100 then combined with an estimate of digestion efficiency to give a daily energetic requirement
101 (Kendeigh 1970; Nagy 1987). However, biometric data are rare worldwide and especially in
102 Australia, even for species of management interest (Kingsford and Norman 2002). In Australia,
103 some limited data exist describing the growth of Australian White Ibis *Threskiornis molucca*
104 (Marchant and Higgins 1990; Smith et al. 2013) and Glossy Ibis (Miller and Burger 1978;
105 Williams 1975), but no growth data are available for the other three species of Australian
106 Threskiornithidae, the Straw-necked Ibis, Royal Spoonbill, and Yellow-billed Spoonbill
107 (*Platalea flavipes*). These species are of particular management interest in Australia when
108 considering environmental water allocations.

109 If the energy required by Threskiornithid chicks to survive to independence can be calculated,
110 meaningful management targets can be set to ensure that sufficient food can be provided to
111 support a specified number of chicks. The aim of this study was develop chick growth and
112 energy models to inform management, through a) an international literature review, including
113 extraction and synthesis of all available Threskiornithid biometric and energy data; b) field
114 measurement waterbird chick biometrics for two Australian species, Royal Spoonbills and
115 Australian White Ibis; and c) analysis of the resulting databases to construct growth curves and
116 predict chick energy requirements.

Methods

Chick growth and energy models were developed using two sources of data: 1) an international literature review, including extraction and synthesis of all available Threskiornithid biometric and energy data; 2) field measurement of waterbird chick biometrics. Biometric data were used to construct predictive growth models, which were then used to develop models of chick energy requirements for three species: Royal Spoonbill, Australian White Ibis and Glossy Ibis.

Synthesis of growth and energy data

An extensive global literature review was conducted with the aim of extracting all existing data describing biometrics including body mass, growth, energy requirements and energy expenditure of species in the family Threskiornithidae. The approach used transparent and consistent search and filtering methods with specific inclusion and exclusion criteria (CEE 2018). Final searches were conducted in: Web of Science Core Collection; Web of Science All Databases; NRM Knowledge Online; Trove; and Google Scholar (Supplementary Material Table S1).

In order to use all available data, mean values were estimated where none were given, assuming a normal distribution, and maximum and minimum values were used to represent variability. Where sample sizes were not reported, the minimum possible sample size for the reported values was allocated. Hence, the synthesised values presented here represent the mean of published means, the maximum and minimum published values, and the minimum possible sample sizes for each species. Where more accurate estimates of variability and sample size were available, standard deviations are presented instead.

To compare chick body mass growth between Threskiornid species for which data were available, values were divided by the mean adult body mass of the species and graphed as the

proportion of adult body mass. Where adult body mass was provided, this value was used to calculate the proportion of adult body mass. Where no adult body mass was provided the mean adult body mass derived from the literature review was used. The same approach to the selection of adult body mass values was used to derive mass specific energy expenditures where none were calculated in the relevant studies.

Field data collection

Due to the scarcity of chick biometric data in the literature, additional field data collection was conducted to provide empirical data for modelling of growth and energy requirements. Royal Spoonbill and Australian White Ibis chicks were measured during December 2017 and January 2018 at the Reed Beds Wetland, Mathoura, NSW (35.8393°S, 144.9475°E) and Middle Lake, Kerang, Victoria, (35.6639°S, 143.8754°E), Australia. These are sites at which ibis or spoonbill breeding is estimated to occur in approximately seven of every 10 years. Reed Beds Wetland dries out each year, while Middle Lake is permanently inundated. Environmental conditions were within the range of variability usual for these sites. Chick biometrics were collected for 35 Australian White Ibis chicks ranging from 2-33 days old, and 27 Royal Spoonbill chicks ranging from 9-30 days old (up to the age of flight when capture becomes difficult). Body mass, bill length, head-bill length, wing length, and tarsus length were recorded for each chick using Reichenow's (1913) standard techniques. Chicks were allocated an age in days using species specific diagnostic features derived from monitoring camera data (Wenger and McGinness 2018). Rather than perform repeated measurements on the same chicks, measurements were taken of chicks of varying age classes. While single measurements on multiple chicks mean that there is potential lack of independence of observations at site level, this approach was taken to reduce disturbance to individual chicks, maintain independence of data points, and obtain results

representing colony level growth. All animal handling was conducted under UNSW ACEC (Animal Care and Ethics Committee) Authority approval no. 17/122B.

Predictive growth models

Predictive growth models were developed for three species: Royal Spoonbill, Australian White Ibis and Glossy Ibis. If well fitted, species specific growth models may be used to predict growth parameters at ages for which no measured data exist (Ricklefs 1968). Avian growth curves often cease at 25-40 days of age, when values approach 90% of their adult values and recapture of chicks becomes difficult due to their mobility (Kushlan 1977b; Smith et al. 2013). However, the success of a managed breeding event at a particular site may be judged by the number of chicks that survive to the age at which foraging independence from the parent birds is reached. This is approximately 50 days for Threskiornithids, since first flight occurs around 30 days of age (Wenger and McGinness 2018) and foraging independence is estimated to occur ~21 days after the chick's first flight (Baynard 1913; Dementiev and Gladkov 1951; Marchant and Higgins 1990). This is beyond the scope of most existing growth curves. However if the average mass of an adult of the species is known, a predictive growth curve allows the estimation of chick mass up until foraging independence (Ricklefs 1968). This predictive model can then be used in conjunction with allometric equations to estimate daily chick energy requirements from hatching to independence (Kushlan 1977c). Following this logic, we used existing data from the literature together with the field-collected data to create growth curves that were then extrapolated to 50 days (foraging independence) based on adult mass data from the literature and subsequently used to estimate daily chick energy requirements. This process is described in more detail below.

When fitting growth curves to biological data a model must be chosen that accurately describes the growth pattern of the organism, but also allows for meaningful interspecific and intraspecific comparisons (Ricklefs 1967; Tjørve and Tjørve 2010). Avian growth follows a sigmoidal curve, with the most commonly fitted curves being the Gompertz, logistic, and von Bertalanffy models (Lok et al. 2014; Ricklefs 1968; Tjørve and Tjørve 2010).

Gompertz model: $$y_t = A\,exp(-exp(-k(t - T_i))) \quad (1)$$

Logistic model: $$y_t = {A}/{(1 + exp(-k(t - T_i)))} \quad (2)$$

von Bertalanffy model: $$y_t = A(1 - exp(-k(t + A_0)))^3 \quad (3)$$

Where t represents a given age, and y_t represents the value of the growth parameter at that age. These curves mainly differ in the placement of their inflection point (T_i), which describes the age at which the maximum growth rate occurs. The inflection point T_i occurs earliest in the logistic model and, because it reaches an earlier asymptote, is more suited to fast growing parameters compared with the other two models, with T_i occurring latest in the von Bertalanffy curve. Examples of fast growing biometric parameters suited to a logistic growth model include wing length in Red-throated Loon (*Gavia stellata*) chicks (Rizzolo et al. 2015) and tarsus length in American White Ibis (*Eudocimus albus*) chicks (Kushlan 1977b). In addition to T_i, descriptive parameters predicted by these growth models include:

- the growth rate constant (k), which describes the rate that the slope changes with age
- the upper asymptote (A)
- the lower asymptote (A_0), for the von Bertalanffy model only

To improve the reliability of avian growth models, a fixed asymptote may be used (Tjørve and Tjørve 2010). Because the upper asymptote of a growth model (A) effectively represents the eventual adult mass or length of a physical parameter, where numerous and reliable adult measurements are available, the model can be better informed by affixing the asymptote to known adult values (Tjørve and Tjørve 2010). Alternatively, a free asymptote (estimated or fitted) may be used (Tjørve and Tjørve 2010).

Gompertz, logistic, and von Bertalanffy models were all fit to the data for each of the three species (Royal Spoonbill, Australian White Ibis and Glossy Ibis) using the nls function from the stats package in R (R Core Team 2016; Wickham 2017). In the interest of maximising the dataset and increasing sample size, mean body mass values for Australian White Ibis chicks (n=7) from the literature review were included in the analyses (Marchant and Higgins 1990; Smith et al. 2013). Both free and fixed asymptote models were tested for all three species and goodness of fit was assessed using Akaike's Information Criterion (AIC) (Motulsky and Christopoulos 2004). The most representative models for the Royal Spoonbill and Australian White Ibis had the asymptote (A) fixed to the mean adult value for the relevant biometric parameter, obtained by calculating the mean of means for each species from the literature review, while a free asymptote provided a better fit for Glossy Ibis. Model sensitivity to the fixed asymptote value for the Australian White Ibis was analysed by re-running the best fitting body mass growth model with the maximum and minimum study means for adult Australian White Ibis from the literature review (max = 1901g, Marchant and Higgins 1990; min = 1685g, Smith et al. 2013) and comparing total predicted energy requirements from hatching to foraging independence with the base model. Initial parameters for the models were estimated from growth data for the same species, obtained from the literature search for Glossy Ibis (Miller and Burger

1978; Williams 1975) and from both the literature and from field data from this study for Royal Spoonbill and Australian White Ibis (Table 1). The only exception was that for Royal Spoonbill head-bill length, a mean value for Eurasian Spoonbill (*Platalea leucorodia*) was used, as no adult head-bill values were available for Royal Spoonbill.

Chick energy requirement models

The energy needs of growing chicks are most commonly reported in terms of field metabolism, or daily energy requirements, rather than the intake of energy required for survival (Fort et al. 2011; Rizzolo et al. 2015; Santoro et al. 2015). To convert energy expended into energy required, an estimate of the digestive efficiency, meaning the proportion of energy consumed that is assimilated, is required (Ricklefs 1974; Sutherland et al. 2004). Experimentally determined estimates of Threskionithid digestion efficiency are limited to Kushlan's (1977a; 1977c) American White Ibis (*Eudocimus albus*) studies, which both resulted in estimates of 79%. Consequently, Santoro et al.'s (2015) data describing wild Glossy Ibis chick daily energy expenditure derived from doubly-labelled water measurements were extracted and divided by Kushlan's (1977c) mean estimated digestive efficiency for American White Ibis chicks. While gut length and digestive efficiency may differ between these species, we expect the effect of any differences to be relatively minor and the models can be updated when species-specific data become available. A linear model was fit to the $\log_{10}$ of the resultant data using the lm function in R, with goodness of fit assessed using the coefficient of determination (R Core Team 2016). The assumptions of linear regression were tested with the gvlma package in R (Pena and Slate 2014). A log-linear model was used for consistency with accepted allometry techniques (White 2011) and so that a scaling coefficient, the slope of the logarithm of metabolism and body mass, could be produced for comparison with the existing literature (White 2011). Energetic models

were based on body mass because it is the best biometric predictor of daily energy requirements (Santoro et al. 2015).

Prior to estimating the daily energy requirements of Royal Spoonbills and Australian White Ibis, a model was created based on the allometric energy requirements of Glossy Ibis chicks and cross-validated with all available measured Threskiornithid chick energetic requirements from the literature. The body mass growth model calculated for Glossy Ibis was used to predict Glossy Ibis body mass for 0 – 50 days of age. The predicted body masses were then used with the derived allometric equation to model the daily energy requirement of a Glossy Ibis chick from hatching to independence. These results were then cross-validated with all mass specific energy requirements (energy required per gram of body mass) for ibis species with published chick and/or adult values from the literature review (Table 2, Supplementary Material Table S2). The mass specific energy requirement was calculated by dividing the total daily energy requirement by the body mass of the measured bird (Table 2). Further cross-validation was performed by comparing energetic requirements of ibis species extracted from the literature review with modelled energetic requirements for a Glossy Ibis of the same body mass or age, where known. Energetic models were then created for the Australian White Ibis and the Royal Spoonbill and estimations of mean individual energy requirements calculated 0 – 50 days of age.

Results

Synthesis of growth and energy data

Literature searches located 105 studies describing biometrics including body mass, growth, energy requirements and energy expenditure of species in the family Threskiornithidae,

including 53 references extracted from Hancock et al. (1992). Additional studies on non-Threskiornithid species were used to assess and inform methods and models. Adult biometric data were collected from the literature for 30 species and juvenile biometric data were collected for seven species across 66 studies from 18 countries (Supplementary Material Table S2). This represents the most extensive collation of biometric data on Threskiornithid species to-date.

Predictive growth models

Growth patterns in body mass have been described for eight species of ibis and two species of spoonbill globally. Of these, descriptive growth models have only been applied to four species: American White Ibis, Red-naped Ibis (*Pseudibis papillosa*), Eurasian Spoonbill, and Australian White Ibis (Kushlan 1977c; Lok et al. 2014; Smith et al. 2013; Soni et al. 2009). Body mass growth records generally ceased at around 30-40 days of age, when an asymptote was approached (Kushlan 1977b; Kushlan 1977c; Lok et al. 2014). While chicks can reach up to 90% of adult body mass by 23 days old, it may take more than a year for full adult body mass to be reached (Kushlan 1977c). Maximum growth rates in body mass occurred just prior to 10 days of age (T_i = 7-10 days old, Table 3). Variability in body mass increased with bird age, between and within species. Between species, mean hatching body mass ranged from 3-5% of mean adult body mass in wild chicks, but by 30 days of age species differed by 34% of mean adult body mass (Figure 1). Within species, the growth rate of captive chicks followed a similar curve shape but was slower and reached a lower asymptote than wild chicks (Archibald et al. 1980; Kushlan 1977c).

Predictive growth curves were fit to biometric data for Glossy Ibis, Australian White Ibis, and Royal Spoonbill chicks (Figures 2 - 4). The Gompertz and von Bertalanffy models provided better goodness of fit for all biometric parameters compared to the logistic model, except for

Royal Spoonbill wing growth (Table 4). There was little difference in the goodness of fit of the von Bertalanffy and Gompertz models for most parameters, so the model with the lowest AIC was selected in most cases (Supplementary Material Table S3, S4). An exception was made for the Royal Spoonbill body mass model, because the von Bertalanffy model (AIC evidence ratio 1.3) provided an unrealistic y-intercept, predicting chick mass to be only 3 grams at hatching. The Gompertz model provided a chick mass of 64 grams at hatching, which is more in line with published values for Threskiornithid hatchlings at zero days (Australian White Ibis 34-70g, Marchant and Higgins 1990; African Spoonbill (*Platalea alba*) 33-39g, Sweeney 1993). A free asymptote was selected for the Glossy Ibis body mass growth model because it provided a better fit for chick growth from 0-30 days than the fixed asymptote model (AIC evidence ratio 951, Figure 2). Consequently, the predicted asymptotic value of this model fell toward the lower end of adult body mass values from the literature review (predicted = 484.7 ± 21.4g, literature review min-max: 485–768g).

Variability in all biometric measurements increased with age in Australian White Ibis chicks, particularly for body mass and tarsus length. Bill and head-bill length were the least variable parameters for both Australian White Ibis and Royal Spoonbill chicks (Supplementary Material Figures S1 and S2). Biometric parameters differed within species in the rate at which they approached asymptotic mass. Tarsus length reached an asymptote before 30 days of age in both Royal Spoonbills and Australian White Ibis, and 90% of adult body mass was attained by both species before the independence age of 50 days. Bill and head-bill length were the slowest growing parameters for both species, with only 78% and 83% for Australian White Ibis and 73% and 78% for Royal Spoonbill of mean adult size reached by independence, respectively (Supplementary Material Figures S1 and S2). There were also interspecies differences in body

mass growth. Royal Spoonbill chicks approached asymptotic body mass earlier than Australian White Ibis chicks, reaching 90% of mean adult body mass at just 25 days old, compared to 38 days old for white ibis chicks.

Chick energy requirements

Only five studies have examined energy expenditure for the growth and survival of wild or captive Threskiornithids (Table 2). All five focused on ibis species, with two of these exploring chick energy requirements. No estimations of energetic allometry specific to the Threskiornithidae were found. No energetic studies have been performed on Australian spoonbills or ibis of any age group. Experimentally determined estimates of Threskionithid digestion efficiency are limited to Kushlan's (1977a; 1977c) American White Ibis studies, which both resulted in estimates of 79% (Table 5).

Glossy Ibis energy requirements

The Glossy Ibis energetic allometry model produced a scaling coefficient (log-linear slope) of 0.78 ± 0.11 with an intercept of 0.73 ± 0.29 ($P = 0.000$, $R^2 = 0.55$, Figure 5) to fit the equation:

$$log_{10}DER = 0.7315 + 0.7825 * log_{10}mass \quad (4)$$

All assumptions of linear regression were met. This equation was then used in conjunction with predicted daily body mass values from the Glossy Ibis von Bertalanffy growth model to predict daily energy requirements for the first 50 days of age, then compared with the mass specific daily energy requirements (kJ g^{-1} day^{-2}, mDER) from the literature. The energy model took the following mathematical form, where *DER* is the daily energy requirement of the chick in kilojoules and *t* is the age in days:

$$DER = 10\text{^}(0.7315 + 0.7825 * log_{10}(484.7421 * (1 - exp(-0.1406 * (t + 2.2375)))\text{^}3)) \quad (5)$$

The predictive model produced similar mDERs to those observed by Santoro et al. (2015) for Glossy Ibis chicks (Figure 5; Supplementary Material Table S5). However, while Kushlan's (1977c) mDER for the American White Ibis (1.99 ± 0.80) and the predicted values (1.51 ± 0.15) from 5–40 days of age were not significantly different (P = 0.14, F = 2.45), the predicted values did not decline as sharply as Kushlan's (1977c) values over the 35 days of growth compared. For further validation, the allometric prediction was compared to Bairlein et al.'s (2015) doubly-labelled water measurement of adult energy requirements for the Northern Bald Ibis (*Geronticus eremita*) using mean adult body mass from the literature (1393 g) and found to be within the published range of error (predicted = 1,555 kJ day^{-1}, published = 1,823 ± 911 kJ day^{-1}).

Royal Spoonbill and Australian White Ibis energy requirements

The cross-validated Glossy Ibis energetic allometry model was applied to the estimated daily body mass of Royal Spoonbill and Australian White Ibis chicks (Figure 6). The energy models took the following mathematical forms, where *DER* is the daily energy requirement of the chick in kilojoules and *t* is the age in days:

Royal Spoonbill:

$$DER = 10\text{^}(0.7315 + 0.7825 * log_{10}(1{,}685 * exp(-exp(-0.1395 * (t - 8.4962))))) \quad (6)$$

Australian White Ibis:

$$DER = 10\text{^}(0.7315 + 0.7825 * log_{10}(1{,}847 * (1 - \exp(-0.08229 * (t + 3.63412)))\text{^}3)) \quad (7)$$

Royal Spoonbill chick daily energy requirements (DER) began at 139 kJ day^{-1} at hatching and reached a maximum daily increase in energy demand with growth at ~8.5 days old (Figure 6) before peaking at 1,800 kJ day^{-1} at 50 days old. Australian White Ibis DER ranged from 81 kJ day^{-1} at hatching to 1,884 kJ day^{-1} at 50 days old, reaching a maximum daily increase in energy demand slightly later than Royal Spoonbill chicks at ~10 days old (Figure 6).

Based on the predictive models, the total energy required for a single Royal Spoonbill chick from hatching to foraging independence is 71,290 kJ and for that for an Australian White Ibis chick is 67,160 kJ. The total energy requirement from hatching to foraging independence in Australian White Ibis chicks was robust to variations in the model asymptote, based on asymptotes derived from minimum and maximum study means for adult Australian White Ibis body mass from the literature review, providing a range of 65,588 - 67,563 kJ (-2.3% - +0.6% variation from base model).

Discussion

There is a paucity of data available describing the growth and energy requirements of the Threskiornithidae. The most important knowledge gaps identified via this review were: 1) No direct measurement of spoonbill energetics worldwide, and very limited direct measurement of ibis energetics; 2) No estimates of the biomass of food required for wild Threskiornithid chick rearing; 3) No direct measurements or estimates of the energetics of any Australian spoonbill or ibis species; 4) No growth curves for the Australian spoonbills or Straw-necked Ibis. This study has begun to fill some of these knowledge gaps, through the extraction and synthesis of the best available data globally on Threskiornithid energetics, growth, and biometrics, and by creating the

first growth model for Royal Spoonbill chicks and developing the first predictive energetic models of Australian waterbird chick energy requirements. The general agreement between modelled predictions and measured values for other species (Bairlein et al. 2015) suggests that these models and predictions could be extrapolated to other similar species.

Growth

Body mass was the most commonly recorded growth parameter in Threskiornithids, being the most important biometric predictor of many aspects of physiology and life history (Calder 1984; Cornioley et al. 2017). However measurement of body mass often ceases well before independence. The Royal Spoonbill and Australian White Ibis growth models developed here were able to estimate biometric parameters up until independence, at 50 days of age, by using fixed asymptotes based on empirical data. The Australian White Ibis body mass growth model produced similar predictions to Smith et al.'s (2013) model from the 2005-2006 breeding season in Greater Sydney for young chicks, and provided improved predictions after 20 days of age (Supplementary Material Figure S1e). Reasonable model accuracy during the later stages of chick growth is important because older birds become progressively more difficult to catch and age with confidence as they gain mobility (Lok et al. 2014; Smith et al. 2013).

Some common patterns in growth were apparent in the Australian White Ibis and Royal Spoonbill growth models that were supported by patterns identified in the literature. The tarsus grew fastest in all Threskiornithid species with full adult length reached before fledging. The tarsus is important for both terrestrial locomotion and foraging in Threskiornithids (Ntiamoa-Baidu et al. 1998), so a larger tarsus may allow birds to commence foraging and escape predators at an earlier age, hence reducing reliance on the parent bird for food and increasing the chance of survival (Soni et al. 2009). The bill and head-bill length were the slowest growing and least

variable parameters for all Threskiornithids, only reaching 70-80% of adult size by fledging (Supplementary Material Figures S1 and S2) (Kushlan 1977b; Lok et al. 2014; Smith et al. 2013; Soni et al. 2009). This slower bill growth rate could enable chicks to receive food directly from the parent bird for as long as possible via the insertion of the chick's bill into the parent's gullet - a smaller bill may facilitate this process.

Body mass in Royal Spoonbill chicks approached mean adult mass 13 days faster than the Australian White Ibis, before reaching a 9% lower mean adult mass (Figures 3 and 4). The smaller comparative size of Australian White Ibis fledglings may confer metabolic savings during the first 30 days of life, until the chick becomes more mobile and can forage to supplement parental food deliveries (Kushlan 1977b).

Due to the relatively early asymptote and variability in body mass, bill or head-bill measurements may be a better predictor of chick age if hatching date is not known (Lok et al. 2014; Smith et al. 2013). Bill and head-bill measurements grow slower than all other commonly measured biometric parameters; for example, these parameters only reached ~60% of adult size at 40 days of age in wild Red-naped Ibis (*Pseudibis papillosa*), Eurasian Spoonbills, and Australian White Ibis (Lok et al. 2014; Smith et al. 2013; Soni et al. 2009). In black-faced spoonbills (*Platalea minor*) and American White Ibis the bill continued to grow in length until ~70 days old (Jong-Ryol et al. 1999; Kushlan 1977b). In addition, the variability in bill and head-bill size at a given age remains relatively consistent in comparison to body mass, despite sexual dimorphism in bill size (Babbitt and Frederick 2007; Lok et al. 2014; Smith et al. 2013; Soni et al. 2009). Such differences are useful in determining chick age in the field for management purposes, including predicting how much longer water may need to be supplied in the nesting wetland.

Within species, the growth rate of captive chicks followed a similar curve shape to that of wild chicks, but was slower and reached a lower asymptote than wild chicks (Archibald et al. 1980; Kushlan 1977c). This is particularly interesting since captive chicks were offered food regularly until satiated, and hence probably received a greater quantity of food than their wild counterparts (Kushlan 1977c; Sweeney 1993; Xi et al. 2001). This suggests that the quality or diversity of the food offered and/or the biomechanics of wild existence may play an important role in the accumulation of body mass during growth, highlighting the importance of field studies for ecological relevance. Kushlan's (1977c) captive American White Ibis chicks also fledged 10 days later than their wild counterparts, reinforcing the potential importance of body mass energy reserves in fledging. Variability in body mass growth may further be explained by sexual size dimorphism, which is exhibited by all spoonbills and ibis (Figuerola et al. 2006; Fuchs et al. 2017; Kushlan 1977d), hatching order (Jong-Ryol et al. 1999; Kopij et al. 2000), and food availability (Lorenz 2014; Miller and Burger 1978).

Energy

The energy models presented here used the best available empirical energetic measurements of wild Threskiornithid chicks (Table 4) together with empirically collected growth data to predict spoonbill and ibis energetics (this study). The models used doubly-labelled water measurements from the Glossy Ibis (Santoro *et al.* 2015), a species that breeds in the same Australian wetlands as the Royal Spoonbill and Australian White Ibis (Brandis et al. 2011), as the basis of the allometry model rather than generalised non-passerine equations as previous studies have done (Bildstein 1987; Kushlan 1977a; Kushlan 1977c; Santoro et al. 2015). The use of wild chicks meant that no adjustments were needed to account for differences in captivity, further increasing the reliability of the models (Kendeigh 1970; Kushlan 1977c). Ideally, the energy models would

453 have been created with species or genus specific energetic measurements, but the similarity in
454 mDER between the Glossy Ibis metabolism used in the models and measured adult Northern
455 Bald Ibis metabolism increases confidence in the transferability of the allometry model to other
456 Threskiornithid species (Bairlein et al. 2015; Santoro et al. 2015). No other estimates of
457 spoonbill energetics exist in the literature for any spoonbill species globally.

458 The allometric model developed suggests that chicks require more energy per gram of body mass
459 than adult birds to survive, highlighting the need to quantify chick energy requirements to inform
460 management. The scaling coefficient of 0.78 derived from the Glossy Ibis chick allometric
461 model is slightly steeper than the range of values published in White's (2011) literature review of
462 adult bird metabolism (0.635 – 0.744). However, steeper metabolic scaling in growing birds is
463 not surprising. Chicks must amass energy during growth to support them as they recruit into the
464 adult population (Kushlan 1977c). They require energy to maintain their current physiology as
465 well as additional energy for biosynthesis and growth not required by adult birds (Ricklefs
466 1974).

467 Starvation is most threatening to Threskiornithid chicks during the first 10 days after hatching,
468 with the highest mortality from starvation in Eurasian Spoonbills and greatest variability in
469 White-faced Ibis (*Plegadis chihi*) growth rates recorded during this time (Garrett 1996; Lok et al.
470 2014). The energy models predicted the maximum rate of increase in chick energy requirements
471 at the same time as maximum growth, around 8.5 days for the Royal Spoonbill and 10 days for
472 the Australian White Ibis. This matches the published range in time of maximum growth rates
473 for Threskiornithid chicks (T_i = 7-10 days old), and may explain why chicks are more
474 susceptible to starvation at this age, with a sharp increase in energy requirements increasing the
475 difficulty of parental provision of sufficient food (Kushlan 1977c; Nagy 1987). This difficulty is

likely to be exacerbated where insufficient foraging habitat within range or insufficient food sources are available due to insufficient flood area or duration, habitat modification or other factors.

The predicted daily energy requirements of the Australian White Ibis and Royal Spoonbill were nearly identical at 50 days (foraging independence), being 1,884 kJ day^{-1} and 1,800 kJ day^{-1} respectively. These predictions are in close agreement with the mean daily energy requirement of Northern Bald Ibis at 1,823 ± 911 kJ day^{-1} (Bairlein et al. 2015), which at adult size has a similar mean body mass (1,393) to juvenile Australian White Ibis and Royal Spoonbills (1,576 and 1,543 g respectively), affirming the utility of the models.

Although generalised non-passerine energetic allometry models exist, there is substantial variability between and within bird families due to differing life strategies (Fort et al. 2011; Kendeigh 1970; White 2011). Application of a generalised energetic model to a species whose metabolism has not been measured in the development of the model may lead to inaccurate predictions and create a model that is not biologically relevant (Fort et al. 2011). In addition, these generalised models often predict basal metabolic rate or existence metabolism, which must first be converted to free-living daily energy expenditure, and then to daily energy requirement (White 2011). These extra steps often rely on non-species specific relationships and would likely result in compounding errors, giving an unreliable result. Pennycuick (1989) estimated that calculations of flight metabolism based on basal metabolic rates can be in error up to a factor of 10. Hence, the models put forward in this study represent some of the most accurate energetic estimations for the Threskiornithidae to date.

Management implications

The predictions from the energy models developed in this study can be used to assess the needs of spoonbill and ibis breeding events in terms of wetland ecosystem energy flow and be incorporated into ecological models to meet management targets such as fish reproduction and recruitment. This type of modelling will support efforts to meet targets in wetland ecosystem and waterbird population management. For example, the Australian Murray Darling Basin Authority (2014) has identified key environmental water management targets that include increased waterbird breeding and abundance by 2024. Using the Royal Spoonbill and Australian White Ibis energy models, the amount of energy required for the successful fledging of a selected number of birds can be estimated and flows and wetland habitats managed to promote the availability of sufficient food to meet those energy needs.

The main components of Royal Spoonbill and Australian White Ibis diets are small fish, crustaceans, amphibians and invertebrates (Barker and Vestjens 1989). The proportions of different prey types in the diet vary spatially and temporally (Barker and Vestjens 1989). Two of the most abundant and widespread prey items consumed by these species are the freshwater crayfish commonly known as the yabbie (*Cherax destructor*) and the invasive common carp (*Cyprinus carpio*). Mean energy content values reported for yabbies range from 14.60 +/- 0.15 kJ/g (Jones et al. 1996) to 20.6 +/- 0.1 kJ/g (Musgrove and Geddes 1995); values reported for carp range from 22.36 kJ/g to 27 kJ/g (Heinitz et al. 2018). Using these values, it is possible to calculate an approximate biomass of yabbies or carp needed in nearby foraging habitats to support a specific breeding event – acknowledging that ibis and spoonbills feed their chicks a variety of prey species. For example, a small-scale breeding event of approximately 1000 nests with average clutch sizes of 2-4 eggs producing an average of three chicks per nest will require

enough food to support 3000 chicks. Using the energetic model produced in this study, for an Australian white ibis or royal spoonbill breeding event of this size an estimated ten tonnes of yabbies or eight tonnes of juvenile carp would be required to support chick energy requirements from hatching to independence. If the mean energy content of the diet of a bird species is known, another approach can be used to calculate the biomass of food required. For example, Kushlan's (1977a) mean value for American White Ibis in the Florida Everglades USA of 3.4 kJ g^{-1} wet food results in an estimate of 3.7 tonnes of mixed prey required to support fewer than 200 chicks from hatching to independence.

In terms of whole-of-system wetland management, these calculations are conservative, because they do not include the food required by the nesting adults or by juveniles older than 50 days. Additionally, it is likely that adult energy needs are increased during breeding periods relative to non-breeding periods. Further research is required to quantify adult and juvenile energy needs, to refine models and predictions taking into account diet composition and variability, and to quantify food and energy availability in a range of habitats and compare it to waterbird needs.

These models emphasise the large quantity of food needed to support successful waterbird breeding and the consequent importance of effective wetland management to support foraging habitats and waterbird food sources. Management options to promote food availability include water management interventions or actions targeted toward the reproduction, recruitment and survival of preferred prey, such as 'environmental flows' and aquatic refuge conservation. For example, temporally staggered watering of a series of wetlands within foraging range of the nesting site will promote successional changes in vegetation and prey populations and provide a range of foraging and prey options. Other options include vegetation management actions such as provision of appropriate wetting regimes and removal or control of invasive weeds; pest

543 animal management actions such as feral pig and fox control; and minimisation of the impacts of
544 toxins, diseases and poisons such as herbicides and insecticides on both prey and waterbirds.
545 Timing will be important, with a balance to be found between maximising positive impacts on
546 prey populations and minimising adverse impacts on waterbirds and other fauna.

547

Figures

Figure 1 Plot of mean wild chick body mass as a proportion of mean adult mass of species in the Threskiornithidae family. Data were sourced from the literature (*n* = 94, hollow points), with individual values for the Royal Spoonbill (*n* = 27) and Australian White Ibis (*n* = 35) (filled points) sourced from the current study (Garrett 1996; Kopij 1999; Kushlan 1977b; Kushlan 1977c; Marchant and Higgins 1990; Miller and Burger 1978; Smith et al. 2013; Williams 1975).

Figure 2 Body mass growth models for Glossy Ibis chicks based on data extracted from the literature review (*n* = 43). The unbroken line represents the best fitting curve, the broken curve represents a curve modelled with fixed asymptote, and the horizontal broken line represents the average adult body mass derived from the literature review.

Figure 3 Growth models for body mass of Royal Spoonbill chicks (*n* = 27) in the Murray Riverina region during the 2017-2018 breeding season. Unbroken lines represent the best fitting curve and broken lines represent the average adult value for the parameter derived from the literature review.

Figure 4 Growth models for body mass of Australian White Ibis chicks in the Murray Riverina region during the 2017-2018 breeding season (*n* = 35) and from the literature review (*n* = 7).

Unbroken lines represent the best fitting curve and broken lines represent the average adult value for the parameter derived from the literature review.

Figure 5 Graphs of Glossy Ibis chick energetic allometry (left) created using data collected by Santoro et al. (2015) (n = 44) and Glossy Ibis predictive energy model up until independence age (right) based on the resultant allometric equation and growth model predictions. Daily energy requirements were also predicted using the mean mass specific daily energy requirement from Santoro et al. (2015) for comparison (broken line).

Figure 6 Predictive energy models for Royal Spoonbill and Australian White Ibis chicks up until independence age, based on species specific growth model predictions.

Tables

Table 1 Initial parameter values used to fit growth curves to ibis and spoonbill biometric data, derived from the literature review and field measurements. Values for the asymptote (A) equal the mean of the relevant biometric from the literature review.

Species	Parameter	Body mass	Wing	Bill	Head-bill	Tarsus	References
Royal Spoonbill	A	1,685	366.9	186.9	248.5*	127.1	(Lok et al. 2014) and field measurements
	A_0	-1.3	-3.1	-8	-12	-7	
	T_i	9.7	14.7	11.3	7.9	10.6	
	k	0.147	0.095	0.056	0.052	0.157	
Australian White Ibis	A	1,847	356.5	167.1	223	93	(Smith et al. 2013) and field measurements
	A_0	-1.8	-4.2	-6.7	-11.2	-12.3	
	T_i	8	15.5	11.6	7.4	9.2	
	k	0.124	0.077	0.041	0.055	0.116	
Glossy Ibis	A	617	-	-	-	-	(Miller and Burger 1978; Williams 1975)
	A_0	-1.8	-	-	-	-	
	T_i	7	-	-	-	-	
	k	0.178	-	-	-	-	

* Mean value for Eurasian Spoonbill (*Platalea leucorodia*) used as no adult head-bill values available for Royal Spoonbill.

Note: The growth rate constant was derived using the equation $k = (e * g_{max}) / A$, where g_{max} is the maximum growth rate (Lok et al. 2014).

Table 2 Summary of the available data on the daily energy expenditure of spoonbills and ibis (± SD). Energy requirement was calculated by multiplying the daily energy expenditure by Kushlan's (1977a; 1977c) digestion efficiency of 79% for American White Ibis, and the weight specific energy requirement was calculated by dividing the total daily energy requirement by the body mass of the measured bird.

Species	Age class or mass	N	Breeding? (Y/N)	Energy expenditure (kJ $bird^{-1}$ day^{-2})	Energy requirement (kJ $bird^{-1}$ day^{-2})	Weight specific energy requirement (kJ g^{-1} day^{-2})	Method	References
American White Ibis *Eudocimus albus*	Adult	NA ^	Y	680*	972	1.09	Allometry (Wiens and Dyer 1977)	(Bildstein 1987)
	Adult	NA ^	Y	689	864	0.911	Allometry (Kendeigh 1970)	(Kushlan 1977a)
	Chick 0–40 days**	5	N	719 ± 159	909 ± 187	1.99 ± 0.86	Free feeding Kendeigh, 1970)	(Kushlan 1977c)
Glossy Ibis *Plegadis falcinellus*	Chick 300–550g	44	N	481 ± 80	609 ± 101	1.46 ± 0.17	Doubly labelled water (Butler et al. 2004)	(Santoro et al. 2015)
Northern Bald Ibis *Geronticus eremita*	Adult	9	N	1,440 ± 720	1,823 ± 911	1.31 ± 0.65 ***	Doubly labelled water (Visser and Schekkerman 1999)	(Bairlein et al. 2015)

^ Sample sizes not given because data were calculated theoretically by the authors and not directly measured.

* Represents the mean of the author's calculated values for the male (745.6 kJ day^{-1}) and female (614.6 kJ day^{-1}) of the species.

** Chicks reared in captivity

*** Body mass not given; mean adult body mass of 1393 g used to calculate weight specific energy requirement.

Table 3 Growth model parameters for selected species derived from the literature. Asymptote values are given in grams or millimetres.

Species	Model	Biometric parameter (g or mm)	k (growth rate constant)	A (upper asymptote)	Ti (time of inflection; days)	Reference
American White Ibis *Eudocimus albus*	Logistic	Body mass	0.185	7000	-	Kushlan, 1977b
	Gompertz	Bill	0.043	125.5	-	
	Logistic	Tarsus	0.180	94.5	-	
	von Bertalanffy	Tail	0.020	95	-	
Red-naped Ibis *Pseudibis papillosa*	Gompertz	Bill	0.0013	362	-	Soni et al., 2009
		Tarsus	0.1359	84	-	
		Tail	0.1189	165	-	
Eurasian Spoonbill[A] *Platalea leucorodia*	Gompertz	Body mass	0.147 ± 0.005	1555 ± 29	9.7 ± 0.2	Lok et al., 2014
		Wing	0.095 ± 0.004	355 ± 7	14.7 ± 0.3	
		Bill	0.056 ± 0.002	129 ± 3	11.3 ± 0.5	
		Head-bill	0.052 ± 0.002	184 ± 5	7.9 ± 0.6	
	Logistic	Tarsus	0.157 ± 0.005	152 ± 2	10.6 ± 0.3	

			Model parameters[B]				
Species	**Model**	**Biometric parameter**	**a**	**b**	**c**	**d**	**Reference**
Australian White Ibis *Threskiornis molucca*	Quasi-Newton sigmoid[B]	Body mass	761.73 ± 48.75	0.18 ± 0.02	-11.28 ± 0.60	614.81 ± 31.58	Smith et al., 2013
		Wing	163.40 ± 7.14	0.13 ± 0.01	-17.07 ± 0.38	151.33 ± 3.17	
		Bill	53.04 ± 5.37	0.10 ± 0.01	-11.57 ± 1.18	46.49 ± 2.82	
		Head-bill	69.73 ± 6.90	0.10 ± 0.01	-9.04 ± 1.33	71.49 ± 4.43	
		Tarsus	44.61 ± 3.06	0.17 ± 0.01	-9.41 ± 0.71	47.82 ± 2.22	
		Tail	59.76 ± 4.25	0.18 ± 0.02	-23.56 ± 0.60	65.41 ± 2.44	

[A] Parameters for Lok et al. (2014) represent the average value for male and female growth curves combined.

[B] Equation fit by Smith et al. (2013) $y_t = a\frac{\exp(b(t+c))-1}{\exp(b(t+c))+1} + d$ where t is age in days and y_t is the biometric parameter. Model parameters: a = the lateral position of the curve (values where inflection points lie); b and c determine the asymptote; d = the specific growth rate.

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Table 4 Estimated parameters for the best fitting growth models for Royal Spoonbill (n = 27) and Australian White Ibis (n = 35) chicks based on data collected during the breeding season of 2017-2018. Also includes values derived from the literature for Australian White Ibis chicks (body mass n = 7, all other parameters n = 1) (Marchant and Higgins 1990; Smith et al. 2013) and Glossy Ibis chicks (body mass n = 98) (Miller and Burger 1978; Williams 1975) ± standard error. No standard error is given for the upper asymptote (A) because values were fixed to those of the relevant mean adult value from the literature review, except for Glossy Ibis body mass where a free asymptote was used.

Biometric parameter	Best fitting model	Upper asymptote (A)	Growth rate constant (k)	Age at inflection (T_i)	Lower asymptote (A_0)
Royal Spoonbill					
Body mass	Gompertz	1,685	0.140 ± 0.018	8.5 ± 0.1	-
Wing length	Logistic	366.9	0.135 ± 0.014	14.1 ± 0.4	-
Bill length	Bertalanffy	186.9	0.0307 ± 0.0023	-	24.9 ± 3.0
Head-bill length	Bertalanffy	248.5	0.0332 ± 0.0028	-	26.7 ± 3.5
Tarsus length	Gompertz	127.1	0.202 ± 0.023	7.8 ± 0.5	-
Australian White Ibis					
Body mass	Bertalanffy	1,847	0.0823 ± 0.0065	-	3.6 ± 1.4
Wing length	Gompertz	356.5	0.104 ± 0.0059	12.8 ± 0.4	-
Bill length	Bertalanffy	167.1	0.0377 ± 0.0015	-	17.3 ± 1.4
Head-bill length	Bertalanffy	223	0.0394 ± 0.0016	-	21.0 ± 1.5
Tarsus length	Gompertz	93	0.173 ± 0.021	5.8 ± 0.5	-
Glossy Ibis					
Body mass	Bertalanffy	484.7 ± 21.4	0.140 ± 0.013	-	2.2 ± 0.6

Table 5 Summary of the available data on digestion efficiency in spoonbills and ibis (± SD).

Species	Age class	N	Digestion efficiency (%)	Source	References
American White Ibis *Eudocimus albus*	Adult	NA^	70	Literature source (Kale 1965)	(Bildstein 1987)
	Adult	3	79.7	Experimentally determined (Kendeigh 1970)	(Kushlan 1977a)
	Chick 0 – 40 days	5	79 ± 5	Experimentally determined (Kendeigh 1970)	(Kushlan 1977c)

^ Sample size not given because data were calculated theoretically by the author and not directly measured.

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SUPPLEMENTARY MATERIAL

Ibis and spoonbill chick growth and energy requirements: Implications for wetland and water management

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Figure S1 Growth models for (**A**) wing length, (**B**) bill length, (**C**) head-bill length, and (**D**) tarsus length of Australian White Ibis chicks in the Murray Riverina region during the 2017-2018 breeding season (n = 35) and from the literature review (n = 1 per parameter). Unbroken lines represent the best fitting curve and broken lines represent the average adult value for the parameter derived from the literature review. Plot (**E**) compares the growth model for Australian White Ibis developed by Smith et al. (2013) with the model developed in this study.

Figure S2 Growth models for (**A**) wing length, (**B**) bill length, (**C**) head-bill length, and (**D**) tarsus length of Royal Spoonbill chicks (n = 27) in the Murray Riverina region during the 2017-2018 breeding season. Unbroken lines represent the best fitting curve and broken lines represent the average adult value for the parameter derived from the literature review.

Table S1 Final search string implemented in Web of Science Core Collection (modified versions were used for Web of Science All Databases and for Google, Trove and NRM Knowledge Online grey literature searches). The full string used is shown and the number of different drafts of that string that were explored until test papers were adequately captured is given.

Theme	Search String	# drafts
Therskiornithid biometrics and energy requirements	TI=(ibis* OR spoon$bill* OR threskiornithidae OR threskiornis OR Platale* OR Pseudibis OR Geronticus OR Nipponia OR Bostrychia OR Theristicus OR Cercibis OR Mesembrinibis OR Phimosus OR Eudocimus OR Plegadis OR Lophotibis OR Thaumatibis) AND TS=(allometr* OR morphometr* OR (bill* NEAR/10 *length) OR (head$bill* NEAR/10 *length) OR (culmen* NEAR/10 *length) OR (wing* NEAR/10 *length) OR weight OR (tail* NEAR/10 *length) OR grow* OR mass OR *tarsus* OR energy OR *calorie* OR *joule* OR nutrition* OR *energetic* OR metaboli* OR biomass*) NOT TS=(patient* OR cell* OR culture* OR speciation OR phylogenetic OR Holocene OR Pleistocene OR deep-sea OR marine OR soil$ OR shear OR plate OR neural OR neuro* OR brain* OR virus* OR disease* OR politic* OR ion OR erosion OR alloy OR polymer* OR gas OR medicin* OR laser* OR "integrated$biosphere simulator" OR *plankton OR "inter$birth interval*" OR parasit* OR "trace element*" OR mercury OR pathogen* OR organochlor* OR contaminat* OR gamma* OR medicin* OR paleo* OR cancer*) NOT SU=(astronomy astrophysics OR computer science OR engineering OR geology OR plant sciences OR physics OR physical geography OR business economics OR forestry OR chemistry OR meteorology atmospheric sciences OR science technology other topics OR oceanography OR psychology OR mathematics)	23

Supplementary material

Table S2 Summary of the available juvenile and adult biometric measurements of spoonbills and ibis around the world in alphabetical order by scientific name. The sample size (n) is given in brackets next to the mean value - because the exact sample size was often not stated, the listed n represents the minimum possible sample size.

Common name	Scientific name	Mean body mass (g) (*n*)	Body mass range (g)	Mean wing length (mm) (N)	Wing length range (mm)	Mean bill length (mm) (N)	Bill length range (mm)	Mean head-bill length (mm) (N)	Head-bill length range (mm)	Mean tail length (mm) (N)	Tail length range (mm)	Mean tarsus length (mm) (N)	Tarsus length range (mm)
Juvenile													
Eurasian Spoonbill	*Platalea leucorodia*	900 (1)											
Black-faced spoonbill	*Platalea minor*	1,338 (1)	1,338 – 1,338										
Olive ibis	*Bostrychia olivacea*			280 (1)		76 (1)							
American White Ibis	*Eudocimus albus*	785 (31)	695 - 969			116.3 (25)	87 - 145			96.6 (25)	89 - 106	96.5 (24)	86 - 106
Glossy Ibis	*Plegadis falcinellus*	536 (7)	445 - 620										
Australian White Ibis*	*Threskiornis molucca*	1,576 (6)	1,380 – 1,940	297.8 (5)	270 - 320	94.2 (5)	88 - 105	139.8 (5)	128 - 148	119 (5)	101 - 130	99.2 (5)	90 - 105
Straw-necked Ibis*	*Threskiornis spinicollis*	1,491 (28)	1,000 – 1,806	324.3 (30)	271 - 381	94.1 (30)	75 - 123	139.1 (30)	112 - 170	128.3 (30)	90 - 173	91.5 (30)	78 - 105
Adult													
Roseate spoonbill	*Platalea ajaja*	1,370 (7)	1,200 – 1,800	347 (34)	322 - 375	163.2 (33)	145 - 180			97 (31)	85 - 108	108.5 (31)	97 - 124
African spoonbill	*Platalea alba*	1,723 (3)	1,590 – 1,790	383.7 (25)	363 - 415	192.8 (21)	160 - 230			131.6 (21)	105 - 152	144.9 (21)	130 - 166
Yellow-billed spoonbill	*Platalea flavipes*	1,784 (12)	1,500 – 2,000	381.3 (21)	186 - 424	212.6 (53)	179 - 238			159.3 (9)	131 - 175	128.3 (24)	113 - 140
Eurasian Spoonbill	*Platalea leucorodia*	1,838 (21)	1,130 – 2,080	371 (140)	308 - 412	194.6 (129)	145 - 238	248.5 (11)	231 - 266	119.9 (41)	108 - 132	136.3 (123)	102 - 165
Black-faced spoonbill	*Platalea minor*	1,660 (13)	1,470 – 1,900	359.2 (39)	334 - 376	180.6 (60)	145 - 207			106.4 (27)	100 - 122	124.4 (38)	106 - 152

Common name	Scientific name	Mean body mass (g) (*n*)	Body mass range (g)	Mean wing length (mm) (N)	Wing length range (mm)	Mean bill length (mm) (N)	Bill length range (mm)	Mean head-bill length (mm) (N)	Head-bill length range (mm)	Mean tail length (mm) (N)	Tail length range (mm)	Mean tarsus length (mm) (N)	Tarsus length range (mm)
Royal Spoonbill*	*Platalea regia*	1,685 (27)	1,290 – 2,070	366.9 (43)	325 - 390	186.9 (74)	158 - 220			117.5 (6)	104 - 137	127.1 (62)	100 - 143
Wattled ibis	*Bostrychia carunculata*			369 (7)	358 - 380	118.5 (7)	110 - 127			165 (7)	150 - 180	64.5 (7)	63 - 66
Hadada ibis	*Bostrychia hagedash*	1,262 (18)		355 (27)	330 - 392	142.4 (27)	117 - 174			152.2 (23)	137 - 170	68.7 (23)	63 - 73
Olive ibis	*Bostrychia olivacea*			309.1 (18)	233 - 362	89.8 (18)	68 - 116			130.1 (16)	88 - 165	63.4 (17)	50 - 73
Spot-breasted ibis	*Bostrychia rara*			274 (16)	253 - 291	125.2 (16)	115 - 131			112.7 (16)	102 - 120	60.2 (16)	56 - 65
Sharp-tailed ibis	*Cercibis oxycerca*			397 (13)	376 - 415	157.2 (13)	144 - 168			267.4 (13)	250 - 301		
American White Ibis	*Eudocimus albus*	911 (92)	593 – 1,261	273.3 (249)	205 - 310	139.7 (277)	102 - 169			103 (76)	84 - 121	91.9 (180)	79 - 110
Scarlet ibis	*Eudocimus ruber*	764 (224)	676 - 852	270.5 (224)	258 - 283	149.2 (224)	134 - 164					83.9 (224)	78 - 90
Southern bald ibis	*Geronticus calvus*			386 (2)	369 - 403	146 (2)	140 - 152			187.5 (2)	175 - 200	69.5 (2)	66 - 73
Northern Bald Ibis	*Geronticus eremita*	1,393 (73)	880 – 1,571	405.3 (4)	390 - 420	131.9 (12)	115 - 147			208 (4)	196 - 220	70 (4)	68 - 72
Madagascan ibis	*Lophotibis cristata*			355 (1)		134 (1)	134 - 134			184 (1)		64 (1)	
Green ibis	*Mesembrinibis cayennensis*	750 (2)	715 - 785	282.5 (10)	263 - 300	109.7 (10)	103 - 119			139.8 (10)	127 - 155	60.5 (10)	57 - 64
Crested Ibis	*Nipponia nippon*			419 (1)		165 (1)				165 (1)		89 (1)	
Barefaced ibis	*Phimosus infuscatus*			277.4 (53)	251 - 299	116.5 (53)	101 - 136			121.7 (53)	105 - 134		
White-faced ibis	*Plegadis chihi*	567 (90)	385 - 679	250 (100)	220 - 280	140.5 (106)	92 - 143			91.6 (104)	73 - 110	88.5 (106)	63 - 121
Glossy Ibis	*Plegadis falcinellus*	617 (11)	485 - 768	276.6 (70)	248 - 306	120.3 (48)	98 - 144			99.1 (44)	90 - 111	93 (46)	68 - 113
Puna ibis	*Plegadis ridgwayi*			283.9 (18)	266 - 302	98.8 (18)	87 - 127			112.7 (18)	106 - 125	70.3 (18)	59 - 88
Red-naped Ibis	*Pseudibis papillosa*			400.8 (3)	365 - 419	172.5 (3)	138 - 197			204.3 (3)	165 - 229	81.5 (3)	75 - 85

Common name	Scientific name	Mean body mass (g) (*n*)	Body mass range (g)	Mean wing length (mm) (N)	Wing length range (mm)	Mean bill length (mm) (N)	Bill length range (mm)	Mean head-bill length (mm) (N)	Head-bill length range (mm)	Mean tail length (mm) (N)	Tail length range (mm)	Mean tarsus length (mm) (N)	Tarsus length range (mm)
Giant ibis	*Thaumatibis gigantea*			546 (2)		238.5 (2)	236 - 241			273 (2)	267 - 279	110 (2)	108 - 112
Plumbeous ibis	*Theristicus caerulescens*			399.1 (20)	360 - 450	144.2 (20)	126 - 167			177.8 (20)	165 - 192		
Buff-necked ibis	*Theristicus caudatus*			391.9 (59)	351 - 429	137.4 (59)	109 - 172			192.7 (59)	158 - 224		
Sacred ibis	*Threskiornis aethiopicus*	1,515 (3)	1,500 – 1,530	371.7 (202)	357 - 396	159.3 (202)	135 - 183			135.9 (202)	127 - 149	98.5 (202)	88 - 116
Oriental white ibis	*Threskiornis melanocephalus*			368.8 (3)	343 - 381	169.8 (3)	139 - 185					113.8 (3)	99 - 119
Australian White Ibis*	*Threskiornis molucca*	1,847 (216)	1,300 – 2,500	356.5 (164)	167 - 398	167.1 (323)	131 - 210	223 (26)	204 - 242	120.6 (91)	103 - 135	93 (159)	81 - 108
Straw-necked Ibis*	*Threskiornis spinicollis*	1,507 (22)	1,100 – 1,850	370.2 (19)	340 - 415	147 (35)	110 - 185	193.8 (14)	160 - 228	135.6 (18)	126 - 152	82.5 (19)	75 - 102

* Data collected during the current study have been included in these biometric summary statistics.

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Table S3 Akaike Information Criterion (AIC) scores for Gompertz, Logistic, and von Bertalanffy growth models for Australian White Ibis, Royal Spoonbill, and Glossy Ibis with free asymptotes

Species	Parameter	Gompertz model AIC	Logistic model AIC	Bertalanffy model AIC	Preferred model
Royal Spoonbill	Body mass	343.98	344.92	343.67	Bertalanffy*
	Wing length	233.42	233.09	233.69	Logistic
	Bill length	153.05	153.85	163.85**	Bertalanffy
	Head-bill length	171.26	171.72	177.94**	Bertalanffy
	Tarsus length	193.34	193.73	193.26	Gompertz
Australian White Ibis	Body mass	564.64	566.94	564.38	Bertalanffy
	Wing length	328.36	329.48	329.13	Gompertz
	Bill length	237.98	238.81	237.86	Bertalanffy
	Head-bill length	257.68	258.34	257.65	Bertalanffy
	Tarsus length	256.36	257.03	256.72	Gompertz
Glossy Ibis	Body mass	398.99	405.70	398.95	Bertalanffy

* Gompertz model was used for final Royal Spoonbill body mass growth model, despite von Bertalanffy model being AIC preferred, due to von Bertalanffy model producing negative weights for chicks under five days old. This is most likely due to the paucity of field data collected for young chicks.

** von Bertalanffy free asymptote model would not run for Royal Spoonbill bill length and head-bill length due to insufficient data and parameters to achieve an acceptable fit, so results from the equivalent fixed asymptote model are presented instead.

Table S4 Akaike Information Criterion (AIC) scores for preferred Australian White Ibis, Royal Spoonbill, and Glossy Ibis growth models with free and fixed asymptotes

Species	Parameter	Model type	Fixed asymptote model AIC	Free asymptote model AIC	Preferred model
Royal Spoonbill	Body mass	Gompertz	342.48	343.98	Fixed
	Wing length	Logistic	249.64	233.09	Free
	Bill length	Bertalanffy	163.85	NA*	Fixed
	Head-bill length	Bertalanffy	177.94	NA*	Fixed
	Tarsus length	Gompertz	191.52	193.73	Fixed
Australian White Ibis	Body mass	Bertalanffy	562.51	564.38	Fixed
	Wing length	Gompertz	326.45	328.36	Fixed
	Bill length	Bertalanffy	235.86	237.86	Fixed
	Head-bill length	Bertalanffy	255.91	257.65	Fixed
	Tarsus length	Gompertz	265.36	256.36	Free
Glossy Ibis	Body mass	Bertalanffy	412.67	398.95	Free

* von Bertalanffy free asymptote model would not run for Royal Spoonbill bill length and head-bill length due to insufficient data and parameters to achieve an acceptable fit.

Table S5 Comparison of Glossy Ibis chick predicted mass specific daily energy requirements (mDER) (kJ g^{-1} day^{-2}) with measured DERs of Glossy Ibis chicks (Santoro et al. 2015) and American White Ibis chicks (Kushlan 1977). A single value ± SD is given for Santoro et al. (2015) as chick ages were not recorded.

Chick age (days)	Model predicted mDER (kJ g^{-1} day^{-2})	Kushlan's (1977c) measured mDER (kJ g^{-1} day^{-2})	Santoro et al.'s (2015) measured mDER (kJ g^{-1} day^{-2})
5	1.88	3.88	1.46 ± 0.17
10	1.60	2.45	
15	1.49	1.93	
20	1.45	1.75	
25	1.42	1.84	
30	1.41	1.55	
35	1.41	1.43	
40	1.41	1.11	

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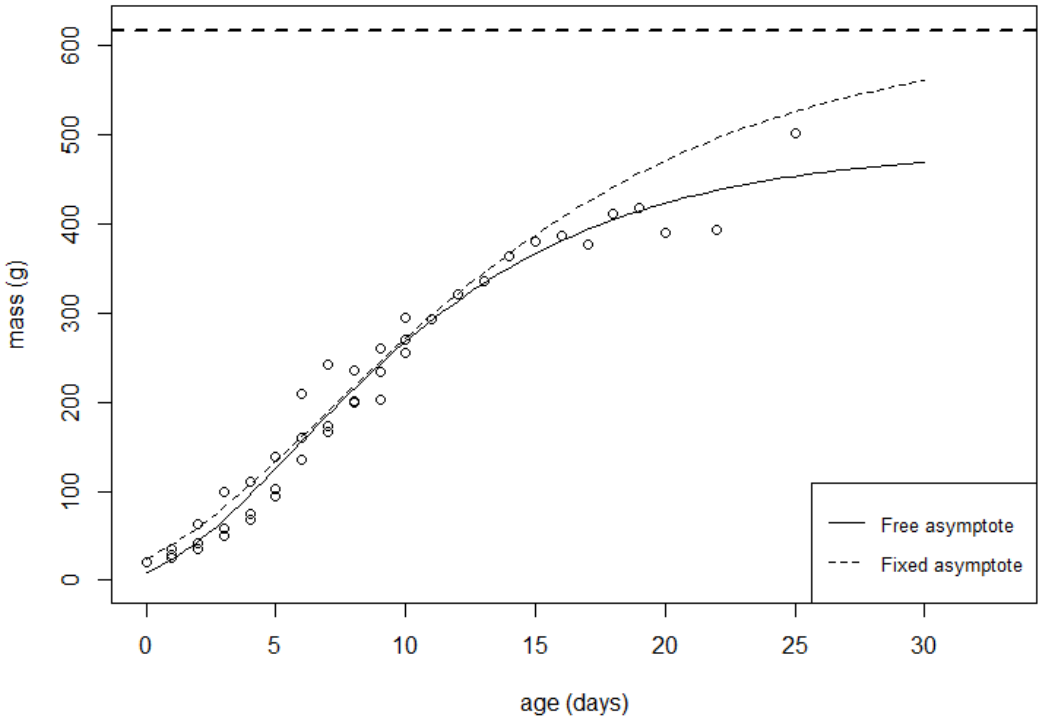


Figure 1

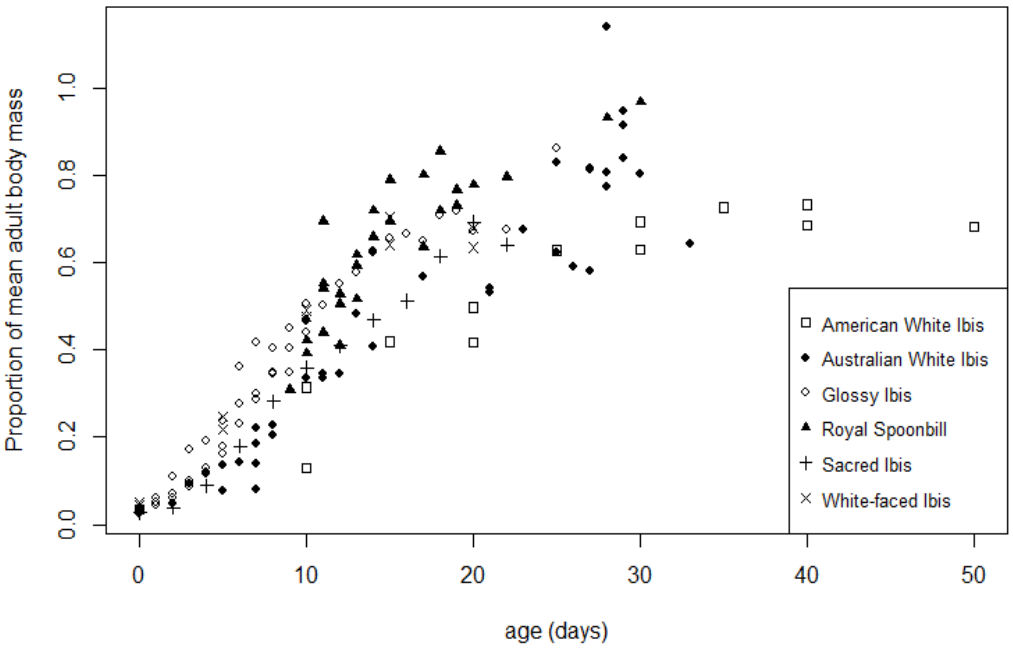


Figure 2

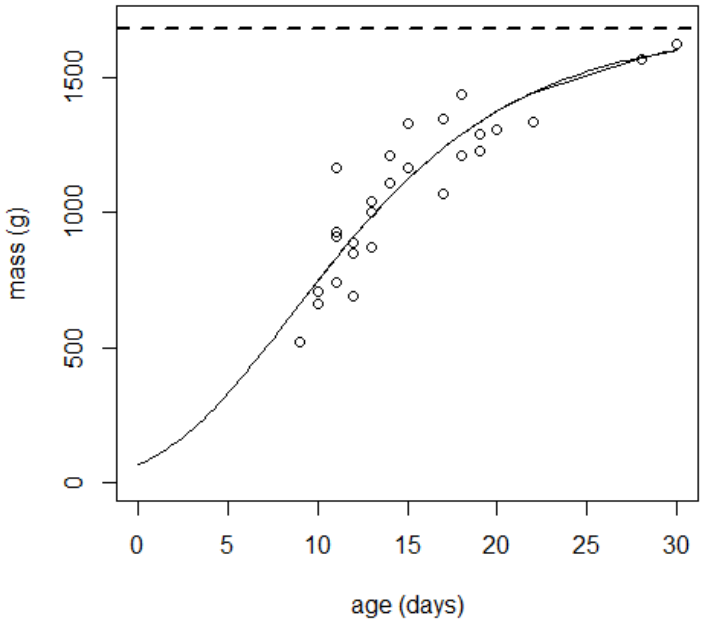


Figure 3

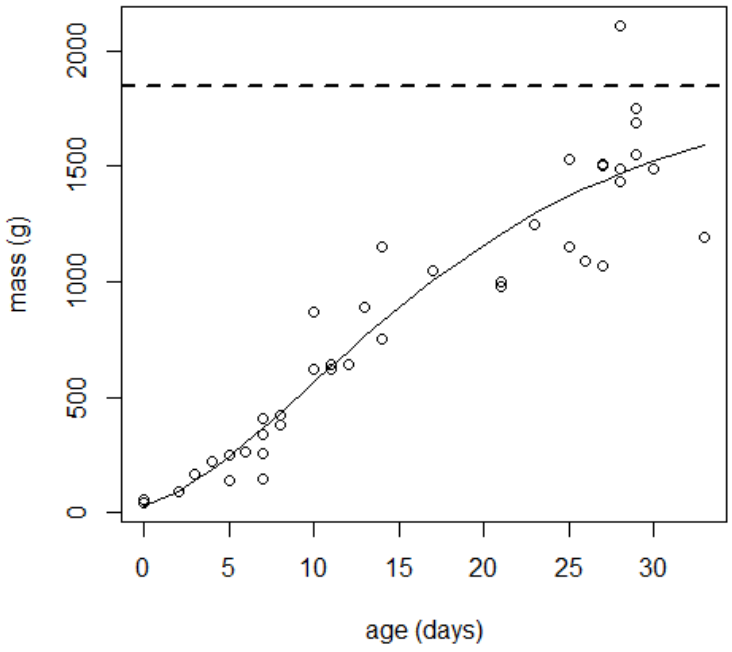


Figure 4

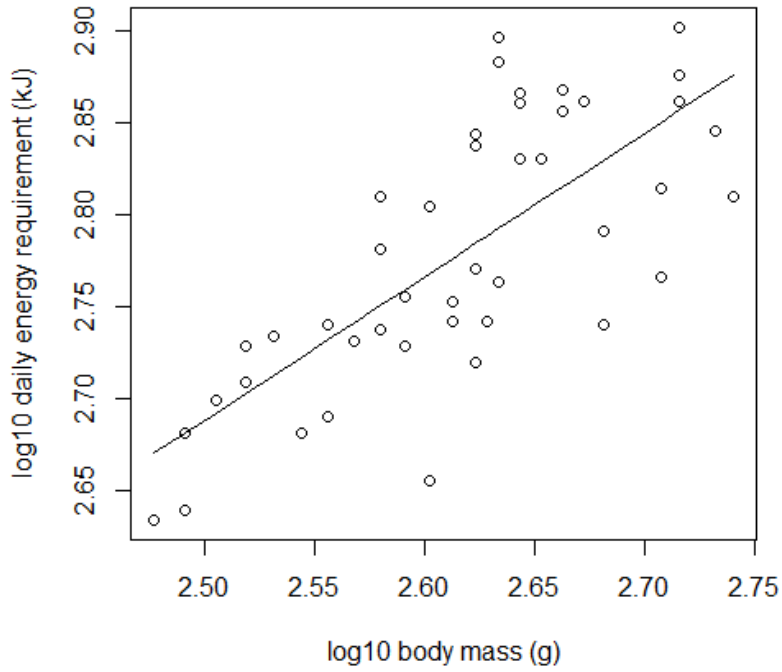


Figure 5a

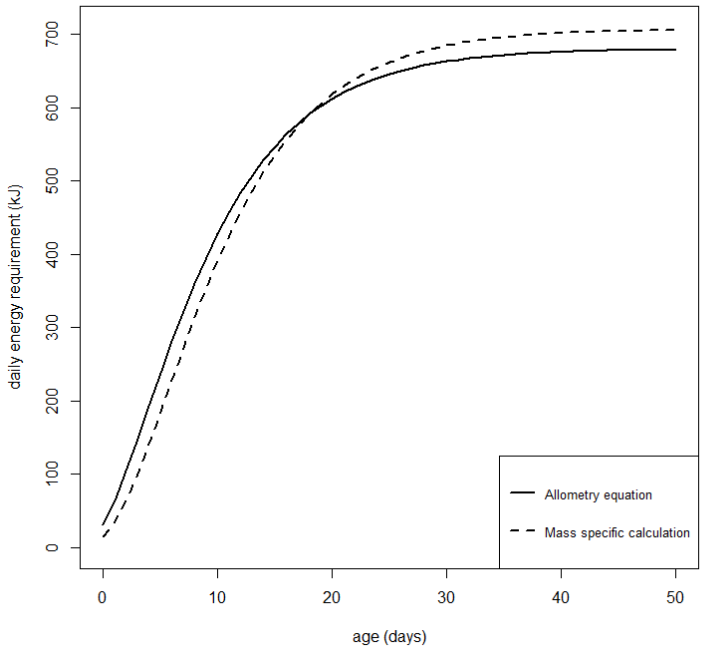
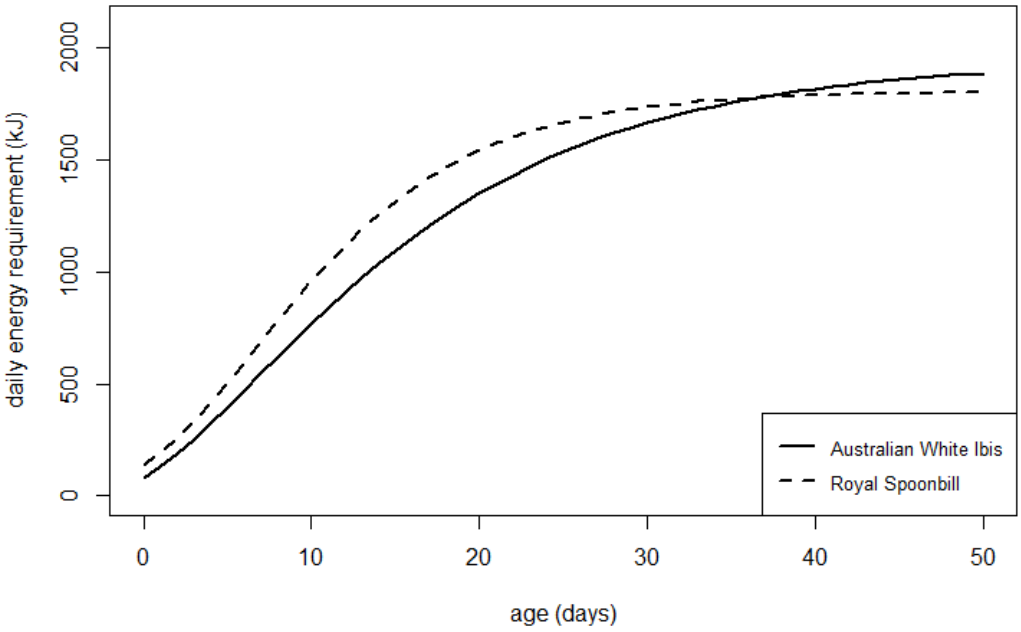
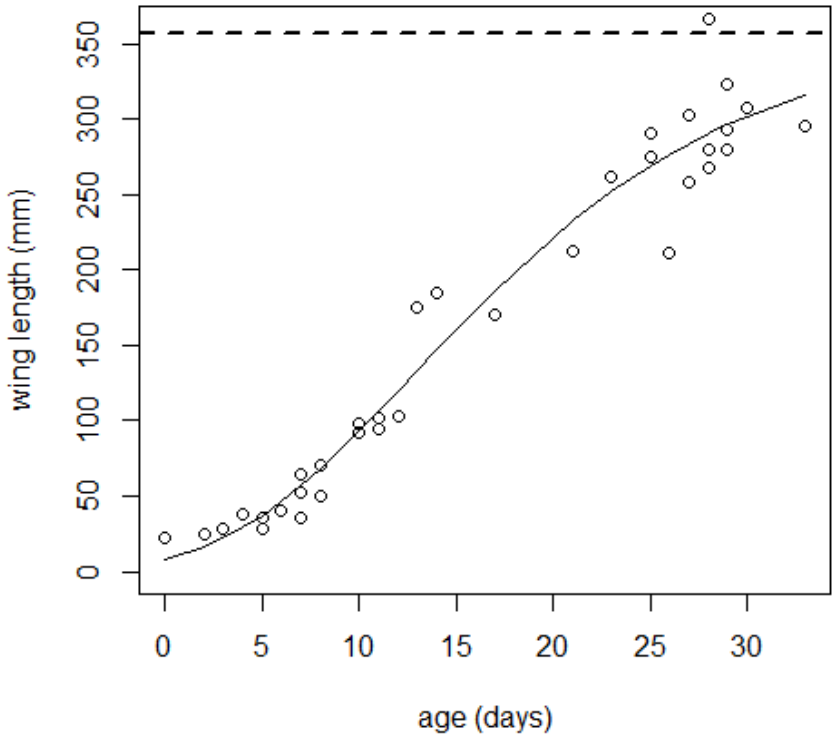


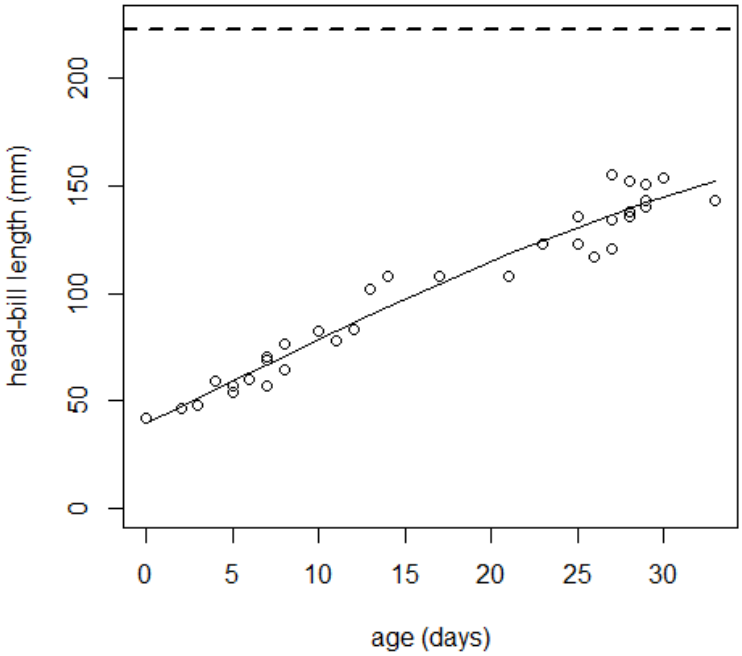
Figure 5b



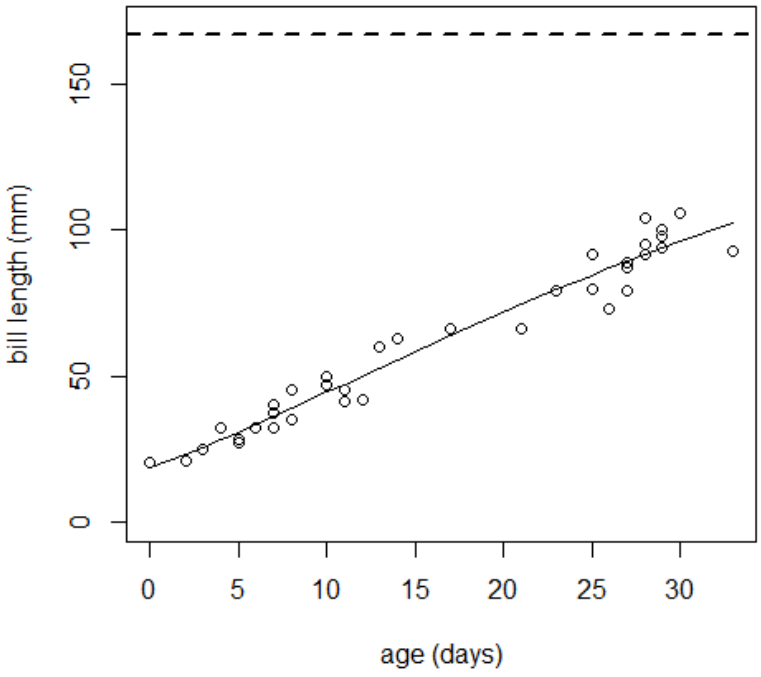
Fgiure 6



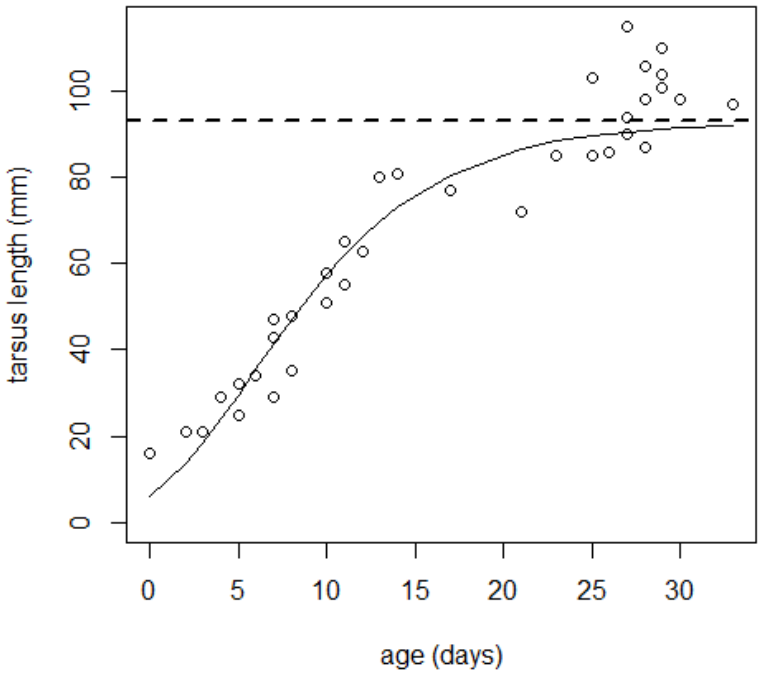
Supplementary material Figure S1a



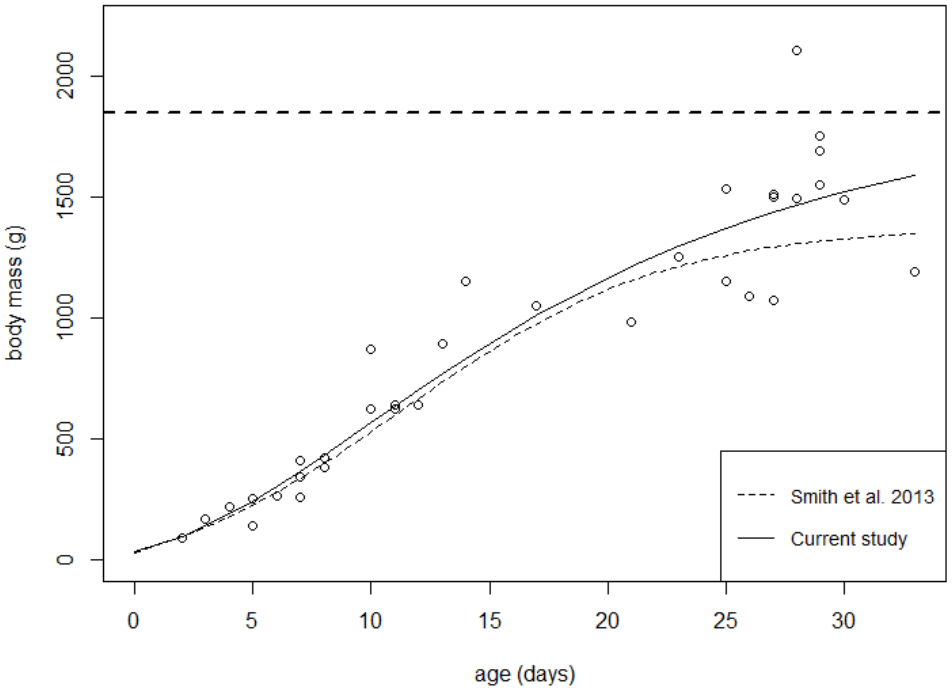
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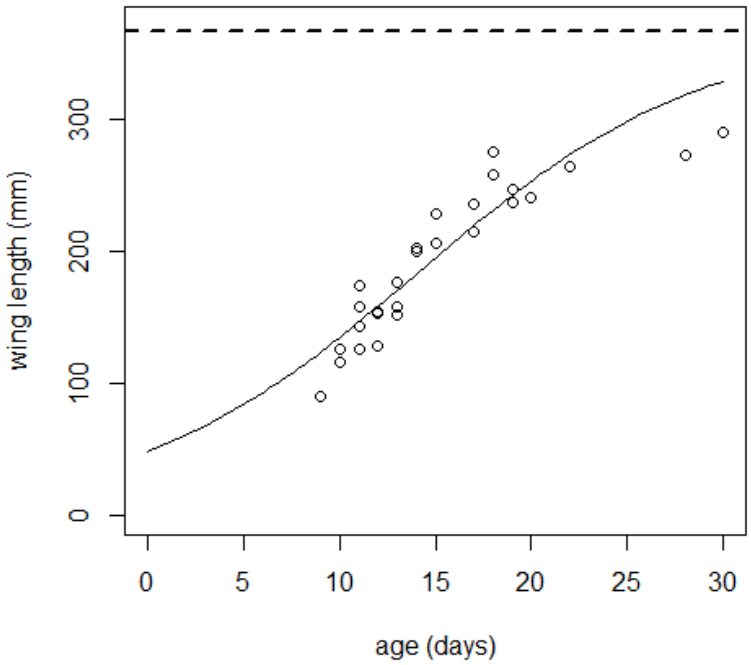
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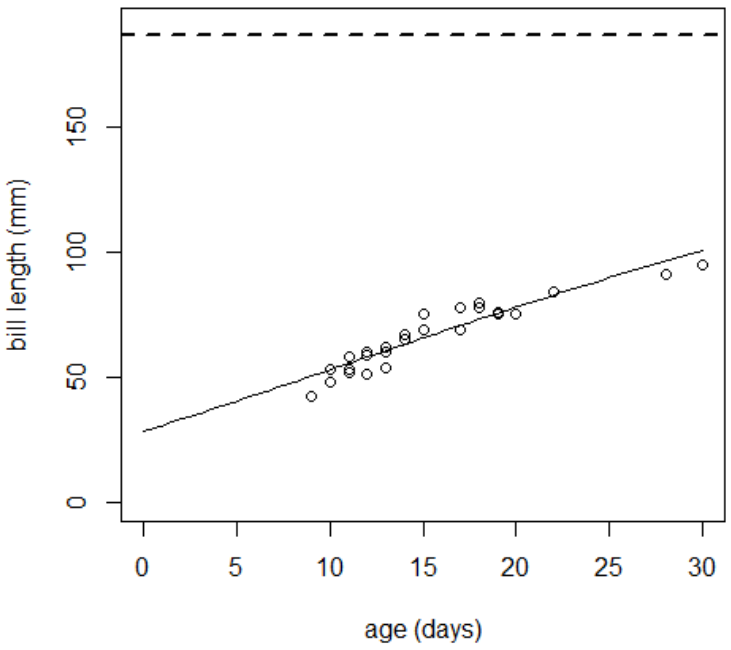
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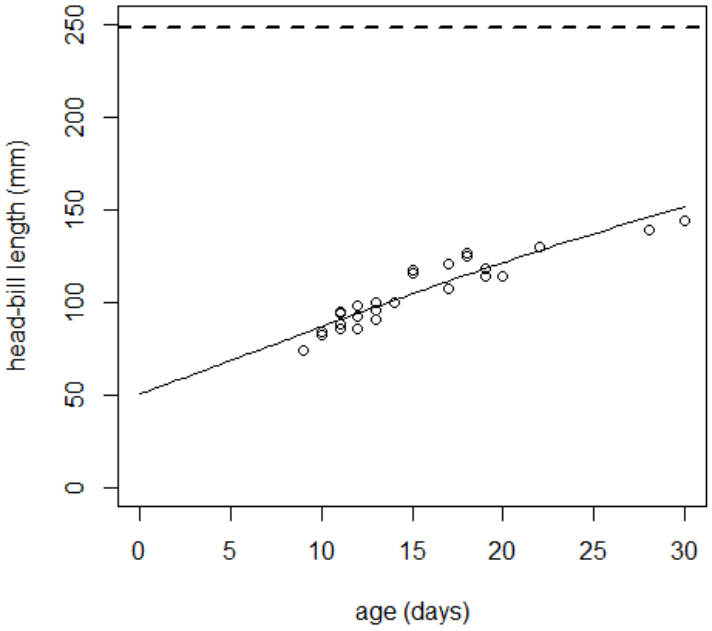
Supplementary material Figure S1e



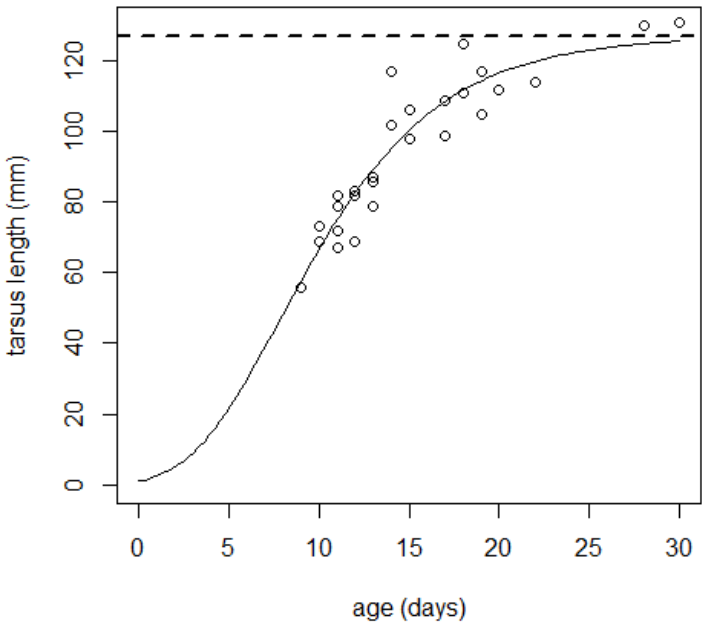
Supplementary material Figure S2a



Supplementary material Figure S2b



Supplementary material Figure S2c



Supplementary material Figure S2d

Waterbird Chick Development

A visual guide to selected Australian species

Australian White Ibis

Straw-necked Ibis

Royal Spoonbill

Lucy Wenger & Heather McGinness







Waterbird Chick Development

A Visual Guide to
Selected Australian Species

Australian White Ibis
Threskiornis molucca

Straw-necked Ibis
Threskiornis spinicollis

Royal Spoonbill
Platalea regia

First Edition

Lucy Wenger & Heather McGinness

Citation

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Cover images: A *Phragmites australis* reed wetland in Barmah-Millewa Forest, south east Australia (Heather McGinness); Australian White Ibis, Straw-necked Ibis and Royal Spoonbill (CSIRO monitoring camera images).





Australian Government
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Many Australian waterbird populations depend on nesting and feeding habitats provided by extended flood events. However flood events of sufficient size and duration for successful breeding are now less frequent than in the past. Flooding may begin, triggering initiation of waterbird breeding, but be cut short, causing nest abandonment and recruitment failure. Consequently the timing and release of managed 'environmental water' is often focused on supporting successful completion of waterbird breeding events – particularly for species which breed and nest in large colonies.

Effective monitoring of waterbird responses to water management is critical to facilitating chick survival and the recruitment of juveniles into waterbird populations. Yet our understanding of the basic biology, demography and behaviour of many common Australian waterbird species is surprisingly still in its infancy.

This booklet aims to assist those monitoring waterbird nest success and behaviour by providing a simple visual guide to accurately aging chicks.

This first edition describes the chicks of the Australian White Ibis (*Threskiornis molucca*), Straw-necked Ibis (*Threskiornis spinicollis*) and Royal Spoonbill (*Platalea regia*). These species frequently nest together in large colonies sometimes comprising tens of thousands of birds.

Age Class Definitions

The following definitions are a general summary of the key identifiers for each age class for the three species

Young hatchling Mostly immobile/asleep, with squinting eyes. Clean fluffy down.

Older hatchling Eyes open. Down less fluffy and thinner on ear covert and side-neck, and still no pin-feathers.

Squirter Squatting. Wing pin-feathers present.

Young runner Try standing (unsteady). Wing-tips emerge from sheaths.

Older runner Walking, creching. Tail feathers emerge, and feathers look 'messy'.

Young flapper Start flapping/stretching wings. Wings feathers not fully overlapping.

Intermediate flapper Flapping regularly, wing feathers almost fully emerged. Underwing coverts developing.

Older flapper Almost able to fly. Other traits species-dependent.

Younger flyer Able to fly, but still returns to nest. Bill upper and lower mandibles almost or completely dark grey.

Older flyer Able to fly, but no longer attends nest.

Age class	Australian White Ibis	Straw-necked Ibis	Royal Spoonbill
Young hatchling	1 - 3	1 - 3	1 - 3
Older hatchling	4 - 6	4 - 7	4 - 6
Squirter	7 - 9	8 - 10	7 - 9
Young runner	10 - 12	11 - 13	10 - 13
Older runner	13 - 16	14 - 16	14 - 16
Young flapper	17 - 21	17 - 20	17 - 20
Intermediate flapper	22 - 25	21 - 24	21 - 25
Older flapper	26 - 29	25 - 27	26 - 30
Young flyer	30 - 35	28 - 34	31 - 45
Older flyer	36+	35+	46+

Ibis and spoonbill chick ages (in days post-hatching) for each age class

Hatchling, squirter, runner, flapper and flyer terminology based on Brandis et al. *(2010, 2011).*

In this guide, stages and indicators of chick development are described based on photographs taken by motion-sensing and time-lapse wildlife cameras positioned at ibis and spoonbill nests in wetlands of the Barmah-Millewa Forest, south-east Australia, in 2016-17.

Cameras captured chick development continuously from hatching (day 1). This allowed more precise ageing than has been possible before, based on distinct physical traits. Each chick's progress was followed until the chick had left the nest or died, or images of it were no longer reliable, or until the camera batteries were flat. A total of seven nests/chicks were followed in detail for each species, with supporting data and images also taken from the total 188 nests monitored by cameras.

For each nest, the day of first appearance of a trait was recorded for the eldest chick. The day of first appearance of each trait is given as the **mean, with the minimum and maximum in brackets** (min, max). Important, easily visible indicative traits are highlighted by **coloured arrows** (➤➤➤).

Basic categories were assigned to age stages according to chick age and indicative traits, using terminology based on Brandis et al. (2010, 2011). Summary statistics for biometrics such as bill length are based on work described in O'Brien and McGinness (2018).

Australian White Ibis

Threskiornis molucca





Day 1



Day 2



Day 3

Behaviour

- Mostly immobile & asleep but starting to watch surroundings

Eyes

- ◊ Squinty eyes *d2 (1-3)*, fully open *d3 (3-4)*

Plumage

- Body covered in white down, head black down (*NB: chick in images has atypical white patch on head*)

Bill

- ◊ Bulbous, then flat *d3 (1-5)*
- ◊ Bright orange-pink
- Bill 20-25 mm
- Head-bill 42-48 mm



Day 4



Day 5



Day 6

Behaviour

◊ Alert, mostly sleeping and sitting

Plumage

➤ Body pale grey down *d6 (6-7)*, no pin-feathers

◊ Down thin on ear-covert, below eye and side-neck *d6 (5-6)*

➤ Bare grey below eye *d5 (4-5)*

Bill

◊ Flat

◊ Duller orange-pink *d6 (5-8)*, with black at base of lower mandible *d5 (4-5)*

➤ Bill 27-32 mm

➤ Head-bill 54-60 mm



Day 7



Day 8



Day 9

Behaviour

◊ Squatting *d7 (6-8)*

Plumage

➤ Wing pin-feathers *d7 (7-7)*, body downy

◊ Grey, mostly bare ear-covert, below eye, side-neck *d7 (7-8)*, with thin down on hindneck *d8 (8-9)*

Bill

◊ More angular *d7 (6-7)*

➤ Bill 32-45 mm

➤ Head-bill 57-76 mm


Day 10


Day 11


Day 12


Day 12

Behaviour

- ◊ Try standing (unsteady) *d10 (8-11)*,
- ◊ Standing on edge of nest *d11 (10-12)*

Plumage

- ◊ Body covered in mix of feathers and down, with untidy 'pink & stringy' wings when closed *d12 (11-13)*
- ➤ Wing feather tips emerge from sheaths *d12 (11-13)*
- ◊ Pin-feathers on shoulder *d10 (9-11)* and tail *d12 (11-12)*

Bill

- ◊ Slightly down-turned *d10 (9-12)*
- ➤ Bill 41-50 mm
- ➤ Head-bill 78-83 mm



Day 13



Day 14



Day 15



Day 15



Day 16



Day 16

Behaviour

- ◊ Strong legs *d13 (13-14)*, explore away from nest *d15 (13-17)*, creching *d15 (13-18)*
- ◊ Flap/stretch wings *d14 (13-16)*

Plumage

- ◊ Untidy white body feathers *d15 (14-16)*, wing feathers non-overlapping
- ◊ 2 black spots on upper back *d15ish* (tertials)
- ➤ Tail feathers emerge
- ◊ White feathers start covering lower throat *d15 (12-18)*

Bill

- ◊ Black on upper mandible reaches nostril *d15 (11-22)*
- ➤ Bill 60-63 mm
- ➤ Head-bill 102-108 mm



Day 17



Day 17



Day 17



Day 18



Day 19



Day 19



Day 20



Day 20



Day 21



Day 21

Behaviour

- ◊ Wings often open and away from nest for extended periods *d20 (18-21)*

Plumage

- ◊ Wing feathers almost completely overlapping but don't look very powerful *d17 (16-18)*
- ◊ Thicker brown/black feathers on hindneck *d17 (16-19)*

Bill

- ◊ More down-turned *d21 (20-22)*
- ➤ Bill tip grey *d18 (17-18)*, then distal half of upper mandible grey-pink *d20 (19-21)*
- ➤ Bill 66 mm
- ➤ Head-bill 108 mm


Day 22


Day 22


Day 22


Day 23


Day 23


Day 25

Behaviour

◊ Often flapping and away from nest

Plumage

◊ Clean, fully emerged white feathers, wings look powerful *d22 (22-22)*

➤ Under-wing coverts developing

Bill

➤ Almost completely dark grey, slightly pinkish *d24 (22-25)*

➤ Bill 79-92 mm

➤ Head-bill 123-136 mm



Day 26



Day 28



Day 29



Day 26-29



Day 26-29



Day 26-29

Behaviour

◊ Flap & jump, almost flying. Often away from nest.

Plumage

◊ Wings look powerful

➤ Under-wing coverts more developed

◊ Small white feathers appear at base of hindneck and side-neck *d26 (25-26)*

➤ Black patch when wings folded (tertials) smaller or same size as head

Bill

◊ Colour unchanged

➤ Bill 73-104 mm

➤ Head-bill 117-155 mm



Day 30



Day 30



Day 35

Behaviour

◊ Flying *d33 (30-35)*, leaving nest completely *d30 (27-35)*

Plumage

◊ Wings look powerful

➤ Black patch when wings folded (tertials) bigger than head *d31 (28-35)*

➤ White feathers speckle all neck, to nape *d30*

◊ Tail & under-wing coverts longer

Bill

◊ Completely grey, but pink inside mouth *d30*

➤ Bill 93-106 mm

➤ Head-bill 143-154 mm



Day 36+



Day 36+



Day 36+



Day 36+



Day 36+



Day 36+



Day 36+

Behaviour

◊ Not attending nest, flying and relatively independent

Plumage

◊ Under-wing coverts well developed
◊ Tail feathers longer
➤ Black tips on second row of wing feathers (primary coverts)
◊ Black patch when wings folded larger
➤ Dappled grey feathers above black patch

Bill

◊ Longer, more decurved
◊ Length not measured

Straw-necked Ibis

Threskiornis spinicollis





Day 1



Day 2



Day 3

Behaviour

◊ Mostly immobile/ asleep but starting to watch surroundings

Eyes

◊ Squinty eyes *d1*, fully open *d2*
◊ Eye-ring grey-pink

Plumage

➤ Body downy black, underparts white

Bill

◊ Flat
◊ Base dark grey and tip bright orange-pink
◊ *Biometrics not measured for Straw-necked Ibis*



Day 4



Day 6



Day 7

Behaviour

◊ Alert, start squatting *d6 (4-9)*

Eyes

➤ Eye-ring grey *d5 (2-6)*

Plumage

➤ Body dark grey down *d5 (4-6)*, head black, no pin-feathers

➤ Down thin on ear-covert, below eye and side-neck *d4 (3-5)*

Bill

◊ Flat, becoming more angular *d7 (6-9)*

◊ Duller orange-pink *d6 (5-8)*, with grey reaching nostrils *d7 (5-9)*



Day 8



Day 9



Day 10

Behaviour

◊ Often squatting, neck long

Eyes

◊ Eye-ring same colour as facial skin *d8 (7-8)*

Plumage

➤ Wing pin-feathers *d9 (7-11)*

◊ Hindneck greyish with thin black down *d10 (9-11)*

◊ Grey, mostly bare ear-covert, below eye, lower throat *d8 (7-9)*

Bill

◊ Greying between nostrils *d10 (5-13)*



Day 11



Day 12



Day 13

Behaviour

◊ Try standing, but unsteady (use wings as support) *d11 (10-14)*

Plumage

➤ Wing feather tips emerge from sheaths *d13 (12-15)*, body mix of down and feathers

➤ Rump/tail black-speckled *d13 (11-13)*

◊ Thin brown feathers on lower throat *d12 (10-13)* then hindneck *d13 (11-14)*

Bill

◊ Grey colour increasing



Day 15



Day 15



Day 15



Day 16



Day 16

Behaviour

◊ Strong legs *d15 (14-15)*, exploring away from nest *d15 (12-16)* & creching *d16 (12-19)*

Plumage

➤ 'Curvy cross' back *d14 (13-15)*, then 'butterfly' back *d15 (14-17)* with untidy body feathers

◊ Thicker brown feathers on neck, parted at front *d14 (13-15)*. Thin white feathers around pink upper throat *d15 (13-19)*

Bill

◊ Slightly down-turned *d14 (12-15)*

➤ Distal half beige *d15 (14-17)* with grey tip *d14 (13-15)*



Day 17



Day 18



Day 18



Day 19



Day 20



Day 20



Day 20

Behaviour

- ◊ Away from nest for extended periods *d17 (14-20)*
- ◊ Flap/stretch wings *d17 (15-18)*

Plumage

- ◊ 'Skull' back *d17 (15-18)*, then 'slit' back *d19 (18-21)*
- ➤ Wing edge half beige *d18 (16-20),* then mostly black (except shoulder patch) *d20 (18-21)*
- ◊ Neck fluffy feathers *d19 (16-21)*
- ◊ Thin black collar *d18 (17-20)*. Depends on sex (M: just sides, F all round)

Bill

- ➤ Grey with beige tinge *d19 (18-20)*
- ➤ Nostril grey *d19 (17-22)*



Day 21



Day 21



Day 23



Day 23



Day 24

Behaviour

◊ Often flapping and exploring away from nest

Plumage

➤ Back fully black (wings open) *d20 (18-22)* but brown slit still present when wings closed

◊ Wing feathers fully overlapping, start to look powerful *d23 (21-25)*

➤ Wing feathers fully black (except shoulder patch and blue sheaths) *d21 (20-23)*

◊ Black collar thicker

Bill

◊ More down-turned *d21 (19-23)*

➤ Dark grey but pink inside mouth *d21 (16-23)*


Day 25-27


Day 25-27


Day 25-27


Day 25-27

Behaviour

◊ Often flapping and away from nest (hard to ID chicks)

Plumage

➤ Wings completely black (shoulder patches & blue sheaths disappear)

➤ White under-wing coverts start developing

◊ Back and wings have glossy rainbow patches

Bill

◊ Shape and colour as previous


Day 29


Day 29


Day 28-34


Day 28-34

Behaviour

- ◊ Flying *d28 (27-29)*
- ◊ Leave nest completely *d27 (22-33)*

Plumage

- ◊ Back glossy, rainbow
- ◊ Wings look powerful
- ➤ Tail longer
- ➤ White under-wing coverts longer
- ◊ White feathers of upper neck extending to jaw/face
- ◊ Feathers on head very thin, brown/black behind head
- ◊ Black collar thick, joined in front (F only). Lower neck brown (F) or pale/white (M). Very variable.

Bill

- ➤ Completely grey, even on the inside



Day 35+



Day 35+



Day 35+



Day 35+



Day 35+



Day 35+



Day 35+



Day 35+



Day 35+

Behaviour

- ◊ No longer attending nest, flying and relatively independent

Plumage

- ◊ Neck plumage becomes more defined, but still very variable between sexes and individuals
- ➤ Under-wing coverts fully developed
- ◊ Tail feathers longer
- ➤ White feathers on neck, extending to hind-neck
- ◊ Head almost bald

Bill

- ◊ Longer, more curved
- ◊ Colour unchanged

Royal Spoonbill
Platalea regia





Day 1



Day 2



Day 3

Behaviour

◊ Mostly immobile/ asleep

Eyes

◊ Squinty eyes *d1 (1-4)*
◊ Eye-ring orange, becoming grey

Plumage

➤ Downy white body

Bill

◊ Bulbous/up-turning, then flat *d3 (2-4)*
◊ Orange
◊ Length not measured



Day 4



Day 5



Day 6

Behaviour

- ◊ Alert, mostly sleeping and sitting

Eyes

- ◊ Fully open *d4 (4-4)*
- ➤ Eye-ring grey *d4 (3-4)*

Plumage

- ➤ Body pale grey down *d6 (5-7)*, no pin-feathers
- ◊ Forehead dull orange *d6 (5-7)*

Bill

- ◊ Flat
- ◊ Colour unchanged
- ◊ Length not measured



Day 7



Day 8



Day 8



Day 9

Behaviour

◊ Squatting *d7 (6-8)*

Plumage

➤ Wing pin-feathers *d7 (7-7),* body downy

➤ Grey, mostly bare ear-covert, below eye, side-neck *d7 (7-8)*, with thin down on hindneck *d8 (8-9)*

◊ Dark blue-grey 'tear-shape' below eye *d8 (7-9)*

Bill

◊ More angular *d7 (6-7)*

◊ Dull orange *d8 (8-9)*

◊ Length not measured



Day 11



Day 11



Day 12



Day 12



Day 12



Day 13

Behaviour

- ◊ Try standing (unsteady) *d10 (8-11)*, stand on edge of nest *d11 (10-12)*

Plumage

- ➤ Wing feather tips emerge from sheaths *d12 (11-13)*, body mix of feathers and down
- ◊ Pin-feathers on shoulder *d10 (9-11)* and tail *d12 (11-12)*
- ◊ 'Pink & stringy' wings when closed *d12 (11-13)*

Bill

- ◊ Slightly down-turned *d10 (9-12)*
- ➤ Becoming grey *d12 (11-12)*
- ➤ Bill 48-60 mm



Day 15



Day 15



Day 16



Day 16

Behaviour

◊ Strong legs *d15 (14-15)*

Plumage

◊ Untidy body feathers *d14 (14-15)*, wing feathers non-overlapping

➤ Tail feathers emerge

◊ Skin around/below eye almost same colour as forehead *d14 (13-15)*

Bill

◊ Continuing to become grey

➤ Bill 54-75 mm



Day 17



Day 17



Day 18



Day 19



Day 20



Day 20

Behaviour

- ◊ Flap/stretch wings *d17 (16-19)*

Plumage

- ◊ Wing feathers almost completely overlapping but don't look very powerful *d17 (17-18)*
- ➤ Under-wing coverts start developing
- ➤ Skin around/below eye same grey as forehead *d18 (17-18)*

Bill

- ◊ Nostril ridges present *d18 (15-20)*
- ◊ Mostly grey with orange tinge *d19 (17-20)*
- ➤ Bill 69-80 mm



Day 21



Day 22



Day 24



Day 24



Day 25

Behaviour

◊ Often flapping, explore away from nest *d21 (20-21)*

Plumage

➤ Clean, fully emerged white feathers, wings look powerful *d21 (20-21)*

◊ Under-wing coverts continue to develop

Bill

➤ Nostrils grey (previously orange) *d21 (20-21)*

➤ Bill 84 mm



Day 26



Day 26



Day 26



Day 28



Day 28

Behaviour

- ◊ Flapping & jumping, almost flying
- ◊ Sleep squatting *d26 (24-27)*, then standing

Plumage

- ◊ As before
- ◊ Under-wing coverts well-developed

Bill

- ➤ Nostril ridges more defined *d28 (27-31)* and faint bill edge groove
- ➤ Upper mandible completely grey *d26 (25-26)*, lower mandible pinkish inside
- ➤ Bill 91 mm



Day 31



Day 31



Day 33



Day 36



Day 39



Day 42



Day 44



Day 47



Day 52

Behaviour

- ◊ Flying *d31 (30-31)*
- ◊ Leave nest completely *d45 (38-47)*

Plumage

- ◊ Wings look powerful, wingtip 'fingers' more pronounced
- ◊ Under-wing coverts longer
- ◊ Tail longer

Bill

- ➤ Edge grooves and nostril ridges well defined
- ➤ Upper and lower mandibles dark grey *d41 (38-44)*
- ◊ Length not measured

Australian white ibis	Young hatchling	Older hatchling	Squirter	Young runner	Older runner	Young flapper	Intermediate flapper	Older flapper	Flyer
Body mass (g)	130 (90 – 170)	219 (140 – 265)	362 (260 – 420)	678 (620 – 870)	1020 (890 – 1150)	1015 (980 – 1050)	1310 (1150 – 1530)	1525 (1070 – 2106)	1338 (1190 – 1486)
Bill length (mm)	23 (21–25)	30 (27 – 32)	38 (32 – 45)	45 (41 – 50)	62 (60 – 63)	66 (66 – 66)	84 (79 – 92)	91 (73 – 104)	100 (93 – 106)
Head-bill length (mm)	47 (46 – 48)	58 (54 – 60)	67 (57 – 76)	81 (78 – 83)	105 (102 – 108)	108 (108 – 108)	127 (123 – 136)	139 (117 – 155)	149 (143 – 154)
Wing length (mm)	27 (25 – 28)	36 (28 – 40)	55 (35 – 71)	98 (92 – 103)	180 (175 – 185)	191 (170 – 212)	276 (262 – 290)	286 (211 – 366)	301 (295 – 307)
Tail length (mm)	0 (0 – 0)	0 (0 – 0)	0 (0 – 0)	5 (0 – 15)	50 (45 – 55)	44 (43 – 45)	91 (82 – 103)	105 (61 – 137)	120 (115 – 125)
Tarsus length (mm)	21 (21 – 21)	30 (25 – 34)	40 (29 – 48)	58 (51 – 65)	81 (80 – 81)	75 (72 – 77)	91 (85 – 103)	99 (86 – 115)	98 (97 – 98)

Values represent mean and range. All data taken from O'Brien & McGinness (2018)

Royal spoonbill	Young hatchling	Older hatchling	Squirter	Young runner	Older runner	Young flapper	Intermediate flapper	Older flapper	Flyer
Body mass (g)	–	–	520*	872 (660 – 1170)	1205 (1110 – 1330)	1271 (1070 – 1440)	1340*	1600 (1570 – 1630)	–
Bill length (mm)	–	–	42*	56 (48 – 62)	69 (65 – 75)	76 (69 – 80)	84*	93 (91 – 95)	–
Head-bill length (mm)	–	–	74*	91 (82 – 100)	111 (100 – 117)	118 (107 – 127)	130*	142 (139 – 144)	–
Wing length (mm)	–	–	90*	147 (116 – 176)	209 (200 – 228)	244 (215 – 275)	264*	281 (272 – 290)	–
Tail length (mm)	–	–	5*	29 (8 – 47)	66 (50 – 73)	78 (54 – 88)	109*	109 (108 – 109)	–
Tarsus length (mm)	–	–	56*	77 (67 – 87)	106 (98 – 117)	111 (99 – 125)	114*	131 (130 – 131)	–

* only one chick measured

Species	**Australian White Ibis** *Threskiornis molucca*	**Straw-necked Ibis** *Threskiornis spinicollis*	**Royal Spoonbill** *Platalea regia*
Body mass (g)	1847 (1300 – 2500)	1507 (1100 – 1850)	1685 (1290 – 2070)
Bill length (mm)	167 (131 – 210)	147 (110 – 185)	187 (158 – 220)
Wing length (mm)	357 (167 – 398)	370 (340 – 415)	367 (325 – 390)
Tail length (mm)	121 (103 – 135)	136 (126 – 152)	118 (104 – 137)
Tarsus length (mm)	93 (81 – 108)	83 (75 – 102)	127 (100 – 143)

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Brandis, K.J., Kingsford, R.T., Ren, S. & Ramp, D. (2011) Crisis Water Management and Ibis Breeding at Narran Lakes in Arid Australia. *Environmental Management, 48*, 489-498.

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Murray-Darling Basin Environmental Water Knowledge and Research Project

Waterbird Theme Research Report

Appendix 9

Honours Project extracts: Similarities and differences in breeding ecology of straw-necked ibis *Threskiornis spinicollis* and Australian white ibis *T. moluccus* in response to environmental flows

Emily Webster, Honours student, UNSW (2017)

Webster, E. (2017) Similarities and differences in breeding ecology of straw-necked ibis Threskiornis spinicollis and Australian white ibis T. moluccus in response to environmental flows. Honours thesis, UNSW.

Ibis diet analysis

Authors: Emily Webster and Kate Brandis

Webster, E. (2017) Similarities and differences in breeding ecology of straw-necked ibis *Threskiornis spinicollis* and Australian white ibis *T. moluccus* in response to environmental flows. Honours thesis, UNSW.

This work was undertaken by Emily Webster as part of an BSc honours (2016-2017) degree at the UNSW supervised by Richard Kingsford, Kate Brandis and John Martin.

Questions/Aims/Objectives

To examine chick diet in relation to 1) differences between straw-necked ibis and Australian white ibis, and 2) chick age.

Methods

To investigate chick diets, 114 scats (51 straw-necked and 63 Australian white ibis), were collected directly from chicks (46%) and from nest platforms (54%) throughout November-December 2016/17. For direct scat collection, one chick from each nest and no more than 7 nests in a cluster were sampled, recording GPS waypoints to avoid repeat sampling. Chicks were captured by hand and placed in a calico bag. Head-bill length and tarsus length were measured to verify their classification into age classes (Smith 2009) (**Error! Reference source not found.**). Chicks were sampled at 23-35 days old, as these readily produced a scat, when handled for less than 10 minutes, while young chicks (hatched - 20 days) did not. Older chicks (>35 days) were too mobile to catch. After release, chicks were monitored for 10 minutes from 20-50m away, recording any stress signals, abnormal behaviour, and when parents returned to the nests. All samples were stored in plastic bags and placed in a hard cooler with icepacks, before freezing in storage. Samples were oven dried at 60°C for 48 hours (Søreide et al. 2006; Logan et al. 2008). A random subsample of 15 chick scats from each species were selected for analysis. Each sample was weighed, and all discernible skeletal parts of prey animals were counted and classified to order where possible. Mean proportional contribution of these taxa (counts) to each sample for each species' and stage combination was

calculated. Counts were standardised for each different prey part type, dividing by sample weight, calculating the total number of skeletal parts per scat. Prey richness was also calculated (the number of discernably different parts), standardised by sample weight, and Shannon diversity (H') (Magurran 1988) of these parts, for each scat. Linear models were used to separately examine relationships of each of these response variables with 5 predictor variables (species, chick stage, date of collection, site and water depth adjacent to nest) and their interactions.

Results

Diversity of prey was significantly higher in Australian white ibis than straw-necked ibis (F1, 11=5.197, p=0.044), and varied significantly among sites (F2, 11=0.835, p=0.033), but not water depth, age or date. Standardised total count of prey varied significantly among dates (F2, 11=5.522, p=0.022), but not species, age, water depth or site (Appendix 2). Standardised richness did not vary significantly among species, ages, depth, date or site (Appendix 2). There was no significant interaction between species and stage for any of these metrics. Sampled chicks all obtained scores of 1 out of 3 or lower when monitored for signs of stress, with rapid breathing accounting for most non-zero scores. Few chicks (<10)

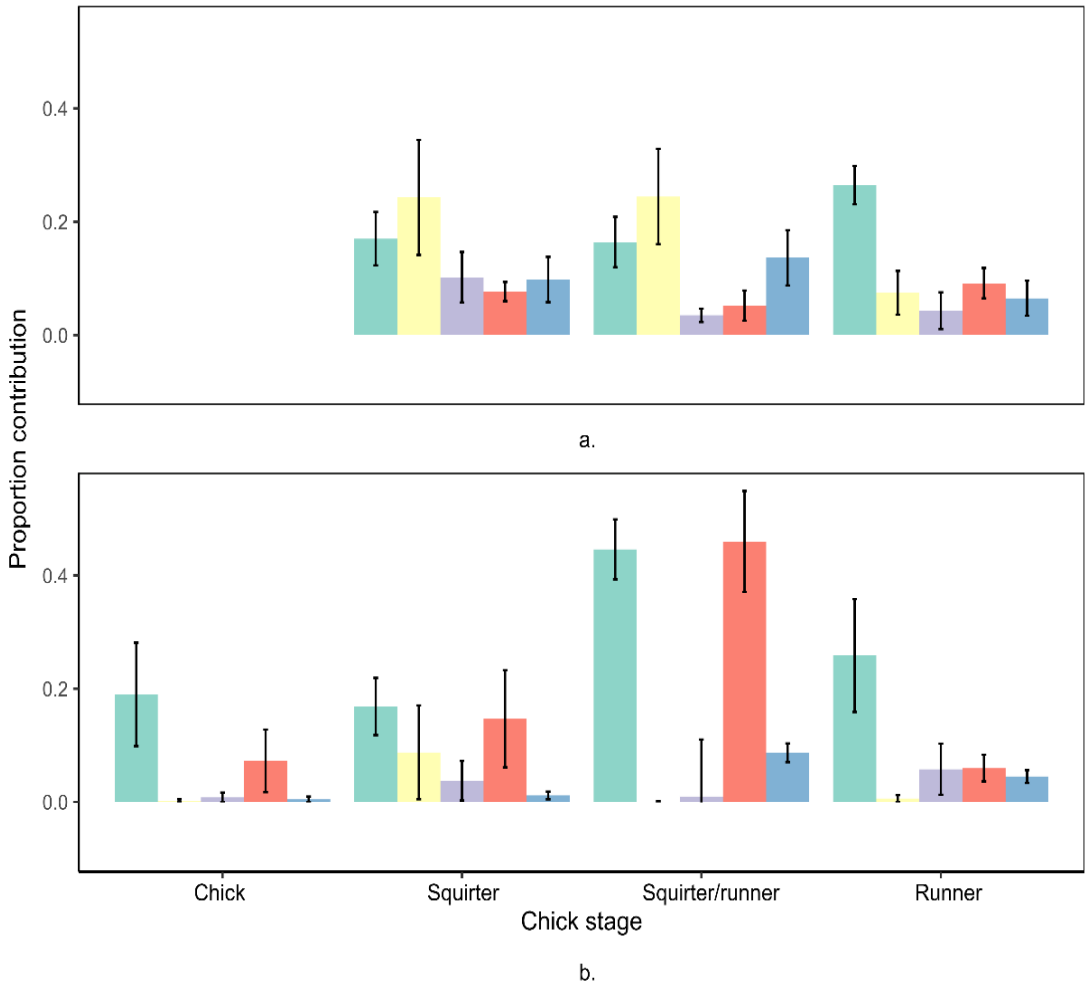


Figure 1 Mean (±SE) proportional contributions of 5 groups of prey taxa (coleoptera=green, decapoda=yellow, fish=purple, orthoptera=red, vertebrates=blue), identified in the scats of a. Australian white ibis (n=15) and b. straw-necked ibis (n=15) at different developmental stages.

Key taxa identified in straw-necked ibis scats were coleoptera and orthoptera (Figure 1). Coleoptera contributed greater than 15% to straw-necked ibis scats at all chick development stages, with greatest contributions to squirters (44.59%) and squirter/runners (25.85%). Orthoptera contributed most to straw-necked ibis squirters (45.99%) and runners (14.69%), and over 5% at all stages. Fish were present at all stages but made greatest contributions at squirter/runner (5.79%) and runner (3.76%) stages. Decapoda were most abundant at runner stage (8.78%) and vertebrates at squirter stage (8.70%). Taxa present in low abundance (<4%) were anura in scats of chicks and squirters, aranea in chicks and squirter/runners, hymenoptera in all stages and odonata in all except chicks. Hemiptera were not recorded for straw-necked ibis. Key taxa identified in Australian white ibis scats were coleoptera and decapoda (**Error! Reference source not found.**). Coleoptera contributed over 16% at all stages and most to squirter/runners (26.48%). Decapoda contributed over 24% to runners and squirters, and 7.47% to squirter/runners. Vertebrates contributed 13.63%, 6.53% and 9.82% to squirters, squirter/runners and runners respectively. Fish contributed to all stages and most to squirters (10.21%), and orthoptera contributed greater than 5% at all stages. Taxa present in low abundance (<4%) were anura in scats of squirter/runners, hemiptera in squirter and squirter/runners, and hymenoptera and odonata in all stages. Araneae were not recorded for straw-necked ibis. Unidentified teeth parts were observed in samples from both species, suggesting birds were also feeding on small mammals. All identified (49 of 166) discernibly different prey parts are listed in Appendix 3.

Discussion

The composition of chick scats suggested a preference of terrestrial invertebrates as prey by straw-necked ibis, and aquatic invertebrates by Australian white ibis. In particular, orthoptera or grasshoppers were abundant in the scats of straw-necked ibis chicks, contrasting an abundance of decapoda for Australian white ibis: the former primarily occupies terrestrial habitats and the latter, aquatic. Coleoptera were abundant in chick scats collected from both species. The relative lack of hemiptera, fish and decapoda in straw-necked ibis scats, and the presence of araneae, not represented in Australian white ibis, further suggests reliance on terrestrial rather than aquatic habitats. There was also a higher diversity of prey items in Australian white ibis scats, compared to those of straw-necked ibis, which could either indicate a higher diversity of prey within aquatic habitats than terrestrial habitats, or that Australian white ibis are more generalists. These analyses are similarly indicative, as scat samples did not adequately reflect dietary intake, given mastication and differential digestion of prey. For example, straw-necked ibis feed on freshwater snails (Marchant & Higgins 1990), however no soft-bodied prey was represented in their scat. In support of the patterns found, voluntarily regurgitated boluses obtained from sampled chicks of both species contained complete bodied prey, confirming the identification of some parts observed in scats.

Water management application

The dietary differences between straw-necked ibis (terrestrial feeders) and Australian white ibis (more aquatic feeders) illustrate niche partitioning between two closely related species to avoid interspecific competition (May and MacArthur 1972). The implications for water management include:

- Recognising that waterbird species differ in their dietary requirements,
- Different diets are sourced from different foraging habitats,
- Ensuring that a mosaic of habitats is available for foraging,
- Not all waterbirds forage in wetland areas, managing these terrestrial foraging sites is often beyond the remit of the water manager.

References

Smith, A.C.M. 2009. Population ecology of the Australian White Ibis, *Threskiornis molucca*, in the urban environment. PhD thesis. University of Technology, Sydney.

Magurran, A. E. 1988. Ecological Diversity and its Measurement. Princeton University Press, Princeton.

Marchant, S., Higgins, P. 1990. Handbook of Australian, New Zealand and Antarctic birds. Vol. 1: Ratites to Ducks, Part A-Ratites to Petrels, Part B-Australian Pelican to Ducks. Oxford University Press, Melbourne.

Supplementary material

Appendix 1. Description of chick stage classifications.

Egg: 1-20 days old

Chick: recently hatched, downy feathered, 21-25 days old

Squirter: large chicks with some feather development, remained in nests, 26-30 days old

Squirter/Runner: large chicks able to move pushing with legs, but not to walk, 29-32 days old

Runner: mixture of developed and down feathers, ability to leave nest on foot, 31-35 days old

Flapper: could not fly, flapped while moving between nests, 36-40 days old

Flyer: young juvenile that can fly, 41-45 days old

Appendix 2: Summary of analysis of variance comparing relationships of species, straw-necked ibis and Australian white ibis, stage, depth (cm), site and date on standardised total counts of prey parts, standardised richness of parts and diversity of parts in scats collected directly from chicks.

Standardised total	df	MS	*F*	*p*
Species	1	194152	1.265	0.285
Stage	3	159938	1.042	0.412
Depth	9	22370	1.454	0.275
Site	1	1135	0.007	0.933
Date	2	847744	5.522	0.022*
Species: stage	2	20041	0.196	0.825
Residuals	11	153510		
Standardised richness				
Species	1	12845	2.493	0.143
Stage	3	3363	0.653	0.598
Depth	9	6302	1.223	0.370
Site	1	467	0.091	0.763
Date	2	11278	2.189	0.158
Species: stage	2	1234	0.239	0.791
Residuals	11	5153		
Diversity				
Species	1	0.943	5.197	0.044*
Stage	3	0.438	2.415	0.122

Depth	9	0.076	0.416	0.901
Site	1	1.072	5.907	0.033*
Date	2	0.152	0.835	0.460
Species: stage	2	0.0049	0.027	0.973
Residuals	11	0.182		

Appendix 3 List prey parts identified to order, showing reference photographs and mean standardised counts for species and chick stages in which samples were found. Fish and vertebrates are included as taxa.

Order	Further classification	Item code
Coleoptera		Carapace B
		Carapace K
		Carapace M
		Carapace P
		Foot A
		Head E
	Curculionoid	Head G
		Joint A
	Dyticid	Leg L
		Leg LB
		Leg M
		Leg O
		Leg P
		Mandible C
		Mandible F
		Organism B
		Organism C
		Part B
		Part CB
		Part Q
		Part S
		Rostrum B
		Sting J
		Wing A
		Wing B
Vertebra		Vertebra A

		Teeth A
		Unidentified bone
Anura		Leg bone
		Pelvic bone
		Skin A
Fish		Fin A
		Long bone
		Otolith
Hemiptera		Gill A
		Leg J
Hymenoptera		Head A
		Spine D
Odonata		Spine C
		Spine F
Orthoptera		Carapace A
		Carapace F
		Leg A
		Leg E
		Leg K
		Mandible A
		Mandible H
		Part A
		Part G
Araneae		Leg Pink
Neuroptera		Spine D
Decapoda		Carapace I
		Carapace L
		Leg Ci

Tracking of ibis movements using GSM transmitters

Authors: Emily Webster, Kate Brandis

Webster, E. (2017) Similarities and differences in breeding ecology of straw-necked ibis *Threskiornis spinicollis* and Australian white ibis *T. moluccus* in response to environmental flows. Honours thesis, UNSW.

This work was undertaken by Emily Webster as part of an BSc honours (2016-2017) degree at the UNSW supervised by Richard Kingsford, Kate Brandis and John Martin.

Objectives

To examine foraging movements during a breeding event and identify key foraging areas for straw-necked and Australian white ibis. This study was designed to be a short-term movement study examining the daily movements of ibis between the colony site and foraging sites. GSM transmitters were appropriate for this study as the area of deployment and surrounding areas were within the range of mobile communications.

Methods

At nests in the Reed Beds Swamp colony, 10 adult birds (6 straw-necked ibis and 4 Australian white ibis) and one fledging juvenile Australian white ibis (sex not determined) were captured in November 2016 (Table 1). Each captured bird was placed in a calico bag, moved from the nest platform, and fitted with a solar-recharging GPS Global System for Mobile Communication (GSM) transmitter (Ecotone, SAKER-H model). Transmitters can work for up to 2 years, allowing for long-term monitoring of movements and habitat use during and after breeding. Transmitters weighed 25g, < 3% of average adult body weight of straw-necked ibis (1100g - 1500g) and Australian white ibis (1400 - 2500g) (Marchant & Higgins 1990). They were fitted according to the backpack and keel strap method (Kenward 2000), with a Teflon tape harness (10 mm wide and 1 mm thick) (Martin et al. 2011). The stitches securing the harness were designed to degrade over the first year of deployment and fall off the bird. Weight, head-bill length and bill length were measured, and birds were individually numbered by attaching Australian Bird and Bat Banding Scheme bands to the right tibia, and individually coded colour bands to the right and left tibia of all birds captured before 24 November (Appendix 4). After processing, birds were immediately released and monitored them for 10 minutes at 20-50 m, recording behavioural and physiological signs of stress. Transmitters were programmed to record hourly GPS locations each day (0400-2100 Australian Eastern Daylight Time or 0300-2000 Australian Eastern Standard Time), effectively capturing roosting locations, as no notable displacement was observed between the last point and the first point for the next day. More frequent data were not collected to ensure transmitter battery voltage did not fall below 3.8 V, risking below-optimal operation. The GSM capability of the transmitter allowed remote monitoring of transmitter operation, in response to battery voltage and temperature (Ecotone Telemetry 2013, http://ecotone-telemetry.com/). Due to variable GSM coverage and transmitter failure, tracking data were incomplete for all birds. To standardise the distribution of data frequency throughout the day, and the study period, a subsample of 4 points per day for each individual was randomly selected. Male straw-necked ibis THRE05 and male Australian white ibis THRE08 and THRE09 were excluded from movement analysis due to lack of data. Transmission ceased completely for 6 of 11 birds during the study period.

A separate two-factor analyses of variance was used to test for differences in weight, number of tracked days, data received and maximum displacement from the capture site between the 2 sexes and species, for all birds excluding Australian white ibis male THRE08 (no data received) and juvenile THRE10 (sex not determined). Data was checked for normality with residual and normal Q-Q plots for each of the response variables. Movement behaviour of the two tracked ibis species was

compared, separating parents provisioning chicks from non-breeding birds, in the T-LoCoH package in R (RStudio Team 2015; Lyons et al. 2013), which created time-local convex hulls based on point data. The hulls were based on a fixed number of nearest neighbour points, the fixed-k method (RStudio Team 2015; Lyons et al. 2013). This was preferred over the adaptive-a method, which overestimated utilised areas for very dense points, particularly breeding birds at nest sites. Time-use maps were created, defined by visitation (number of separate visits with an inter-visitation gap of 6 hours) and duration of visit (mean number of locations per visit). Two nesting birds were identified based on their movement patterns.

To examine associations with habitats, location data was plotted for 10 birds (6 straw-necked ibis and 4 Australian white ibis) and used spatial joins of data points with land use (Geoscience Australia 2016, retrieved from http://services.ga.gov.au/site_1/rest/services/Land_Cover_WM/MapServer/layers) and surface hydrology (Geoscience Australia 2012, retrieved from http://www.ga.gov.au/metadata-gateway/metadata/record/83134) layers to attribute a land use class and wetland type to each point. Frequency of use was determine (number of locations) within different habitat classes. A 1 km circular buffer was drawn around points for each group and defined "available" habitat as the minimum convex polygon (Mohr 1947; McClean et al. 1998) around the vertices of this buffer. The buffer was included to allow for transmitter error and account for movement and habitat use that occurred between point locations. Point data captured only discrete locations and could be recorded in transit, not accurately reflecting habitat use, while a minimum convex polygon without a buffer excluded areas immediately adjacent to points that were likely used. After trialling 20 m and 14 km (Martin et al. 2011) buffers, a 1 km buffer best represented the average scale of moments in one-hour intervals (the smallest possible interval between points for the random sub-selection). We calculated the proportional area of each habitat class within the availability polygon area and conducted a composition analysis, with availability proportions and use counts (as a proportion of the total count for each group) used to calculate Manly's selection ratios (Manly et al. 2007, Eqn. 1). Composition analysis determined whether habitats were significantly selected for, or selected against (de-selection) and was conducted in R, as a design type I study, with the AdehabitatHS package (RStudio Team 2015; Calenge 2007):

$$w_i = \frac{u_j}{a_j} \qquad \text{(Eqn. 1)}$$

where u_j was the proportion use of each habitat class j and a_j was the proportion availability of habitat class j.

Results

Straw-necked ibis (n=6) and Australian white ibis adults (n=4) were tracked respectively for an average of 110 days (SE=20.59) and 56 days (SE=31.16), recording average maximum displacements of 117.6 km (SE=51.10) and 158.33 (SE=51.22, Table 1). Mass, days tracked, number of points received, and maximum displacement of individual birds were not significantly different between species or sexes (Appendix 5).

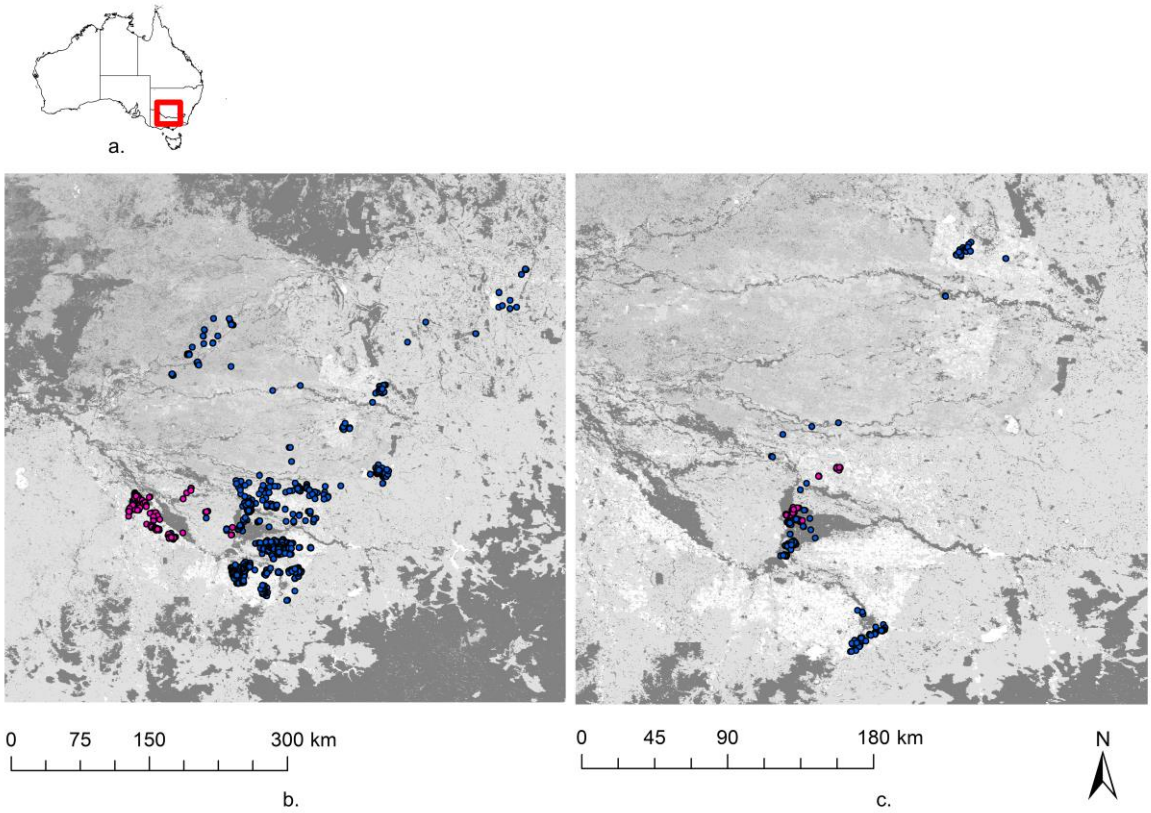


Figure 2 Study area and locations of tracked breeding (filled pink circles) and non-breeding (filled blue circles) b. straw-necked ibis (breeding n=1 and non-breeding n=4) and c. Australian white ibis (breeding n=1, non-breeding n=2).

Table 1 Sex, breeding individuals (B-breeding), tracker identification number, capture date and mass of 11 straw-necked ibis and Australian white ibis, deployed with GPS GSM transmitters, showing the number of days tracked, total number of GPS fixes recorded between deployment and 21 April 2017, and maximum displacement travelled from the capture site

Species	Sex	Identification	Capture date	Mass (g)	Days tracked	GPS fixes	Max displacement (km)
Straw-necked ibis	F	THRE01	19/11/16	1275	153	2191	78
	F	THRE06	29/11/16	1400	130	1951	229
	F	THRE11	22/11/16	1525	85[b]	1042	375
	M (B)	THRE03	24/11/16	1620	147	1873	102
	M	THRE04	29/11/16	1615	124	1545	104
	M	THRE05[a]	20/11/16	1525	19[b]	342	
Australian white ibis	F	THRE02	19/11/16	1560	144	2289	235
	F (B)	THRE07	2/12/16	1568	83[b]	815	42
	M	THRE08[a]	24/11/16	1830	0[b]	0	
	M	THRE09[a]	22/11/16	1615	3[b]	40	
	J	THRE10	8/01/16	1560	20[b]	563	198

[a] Excluded from movement analysis due to insufficient data

[b] Transmitter failed.

Breeding birds

A single bird of each species displayed breeding behaviour, frequently returning to known breeding colonies for 2016/17 (one straw-necked ibis bred at Kerang Lakes, 35°40'00.7"S, 143°52'46.3"E, 98 kilometres west of the capture site on the River Murray, and one Australian white ibis bred at the Barmah-Millewa Forest). They remained relatively close to the capture site compared to other birds (Figure 3), with maximum displacements of 26 km (straw-necked ibis in Kerang wetlands) and 9 km (Australian white ibis in the Reed Beds Swamp) during breeding, and dispersal 50 km and 42 km from the breeding site post-breeding.

THRE03 moved within the Reed Beds Swamp for several hours before travelling 93 km west toward Kerang wetlands over 3 days (Figure 3), where he arrived at a confirmed nesting site for the 2016/17 breeding event. Throughout December, he returned to this point, travelling northeast daily to foraging sites up to 26.5 km away, generally departing for 6-7 daylight hours. He also roosted overnight away from the nesting site about once every 3 days. When away from the colony, he frequented sites on the floodplain of the Murray River near Murrabit, and farmlands west of Myall. By 5 March 2017, he had stopped visiting the colony. Movements of up to 40 km at a time were regularly recorded into April, followed by short localised movements for up to 14 days (Figure 4). THRE03 had a mean visitation rate of 127.02 nsv overall, with a maximum of 233 nsv at the nest site, and a low mean duration of 1.66 mnlv (Figure 3, Appendix 4). Conversely, visitation was low (<50 nsv) and duration of visit high (>2.0 mnlv) post-breeding. Most points (78%) for THRE03 fell outside the bounds of habitat classes defined by the wetland layer (Figure 4), and 16% of points fell in the lake class, for which selection was significant (Wi= 47.745, p<0.001, **Error! Reference source not found.**, Appendix 6) as was selection against land subject to inundation (Wi=0.155, p=0.003, Appendix 6). The most frequent land-use habitat class visited was rainfed pasture (Fig. 8b), with 55% points. Inland waterbodies and irrigated crop were also important, containing 17% and 16% of points respectively. Selection for inland waterbodies (Wi=46.089, p<0.001), irrigated crop (Wi=1.871, p<0.001) and rainfed pasture (Wi=2.429, p<0.001), were significant (**Error! Reference source not found.**, Appendix 6), as was de-selection of rainfed crop (Wi=0.143, p<0.001), trees-open (Wi=0.418, p<0.001), trees-sparse (Wi=0.035, p<0.001), tussock grasses-closed (Wi=0.025, p<0.001) and wetland (Wi=376, p=0.001) classes.

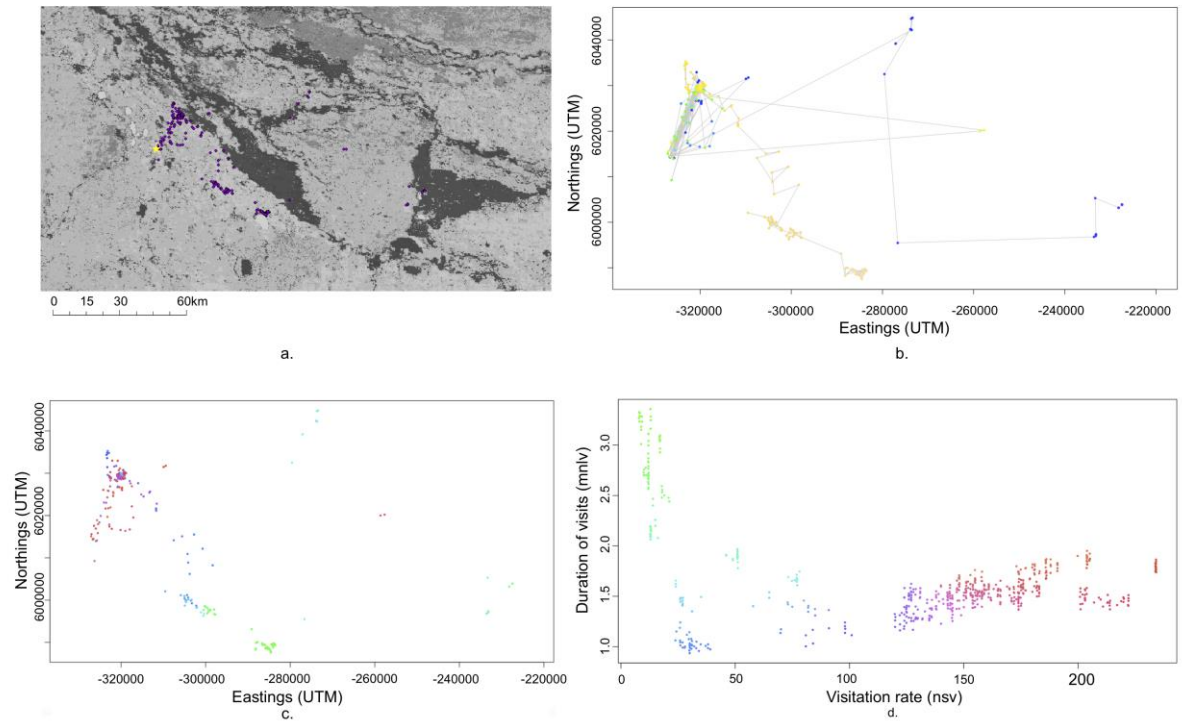


Figure 3 . Locations (filled purple circles) of male straw-necked ibis (THRE03), nesting in Middle Lake, Kerang wetlands (23 December 2016-6 March 2017, yellow star), b. flight paths between spatially distributed points, coloured in a gradient according to date (24 November 2016-21 April 2017: blue to green to yellow to orange), and his c. GPS locations (UTM eastings and northings), showing time-use space where coloured points represented the d. two-dimensional interaction between visitation (nsv) and duration (mnlv) rates, calculated from time-local convex hulls with 628 points (k=25, s=0.05, kmin=0).

The Australian white ibis female, THRE07, bred in the Barmah Millewa Forest (Table 1, Figure 4). She made daily excursions from the nest site to areas up to 9 km north and 7 km east within the Barmah-Millewa Forest, close to main channels, for 22 days, departing for 8-10 daylight hours. Mean visitation rate was 65.19 nsv, with a maximum of 133 at the nesting site, while mean duration was lowest of all birds at 1.47 mnlv (Appendix 5). During January, she continued, moving only 3-4 km away from the nest site, and alternating roosting locations for 2-3 days at the nest site and 2-3 days elsewhere. After 20 January she did not return to the nest site, but remained in the Barmah-Millewa Forest until 13 February, when her chicks had likely fledged (Figure 4). She had high visitation rates when nesting and roosting within the Barmah-Millewa Forest (Figure 4). On 13 February, she flew 33 km north, where she continued to exhibit frequent localised movement until transmission ceased on 24 February. Her most frequented habitat was land subject to inundation (74%), with < 25% of points in undefined regions of the wetland layer (**Error! Reference source not found.**). She had a higher selection ratio for swamp (Wi=4.63) than for land subject to inundation (**Error! Reference source not found.**), with de-selection significant for the latter (Wi=0.87, p=0.024, **Error! Reference source not found.**, Appendix 6). In the land use layer, most points (76%) fell in trees-open (**Error! Reference source not found.**), and no other land-use habitat class contained more than 10% of points. Selection for rainfed crop (Wi=0.301, p<0.001) and trees-open (Wi=2.490, p<0.001) were significant, as was de-selection of rainfed pasture (Wi=0.055, p<0.001) and trees-sparse (Wi=0.264, p<0.001, **Error! Reference source not found.**, Appendix 6).

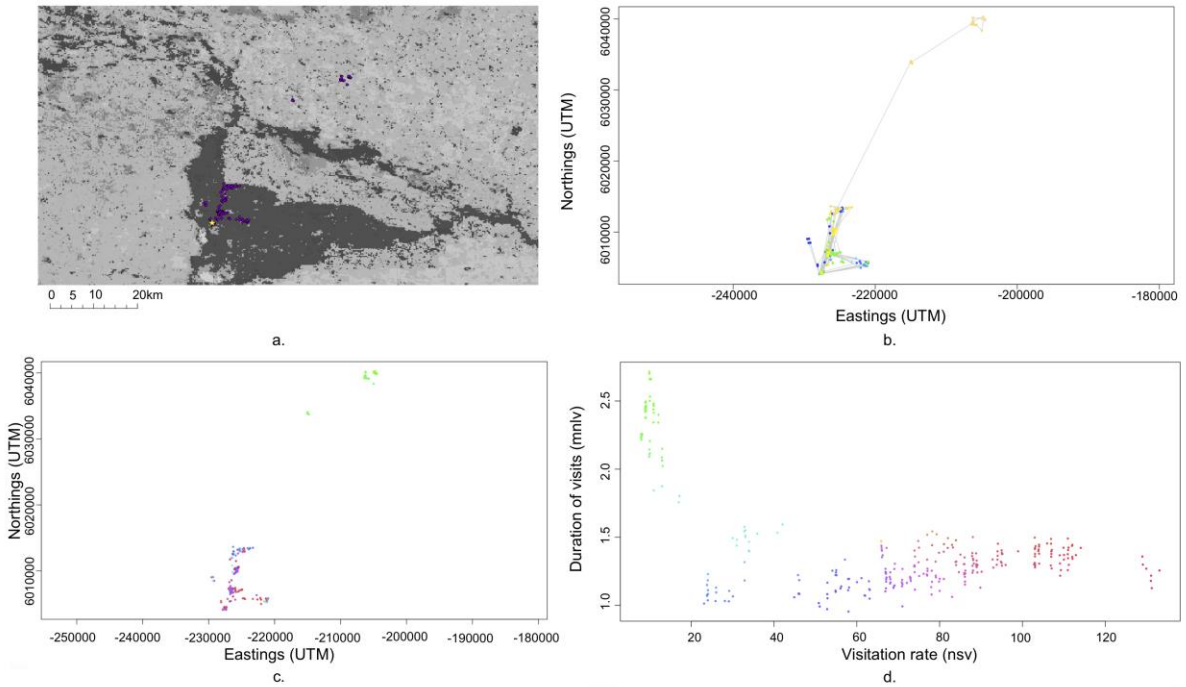


Figure 4 a. Locations (filled purple circles) of female Australian white ibis (THRE07), nesting in Reed Beds Swamp, Barmah Millewa Forest (2 December 2016-13 February 2017, yellow star), b. flight paths between spatially distributed points, coloured in a gradient according to date (2 December 2016-23 February 2017: blue to green to yellow to orange), and her c. GPS locations (UTM eastings and northings), showing time-use space where coloured points represented the d. two-dimensional interaction between visitation (nsv) and duration (mnlv) rates, calculated from time-local convex polygons with 303 points (k=15, s=0.035, kmin=0.7).

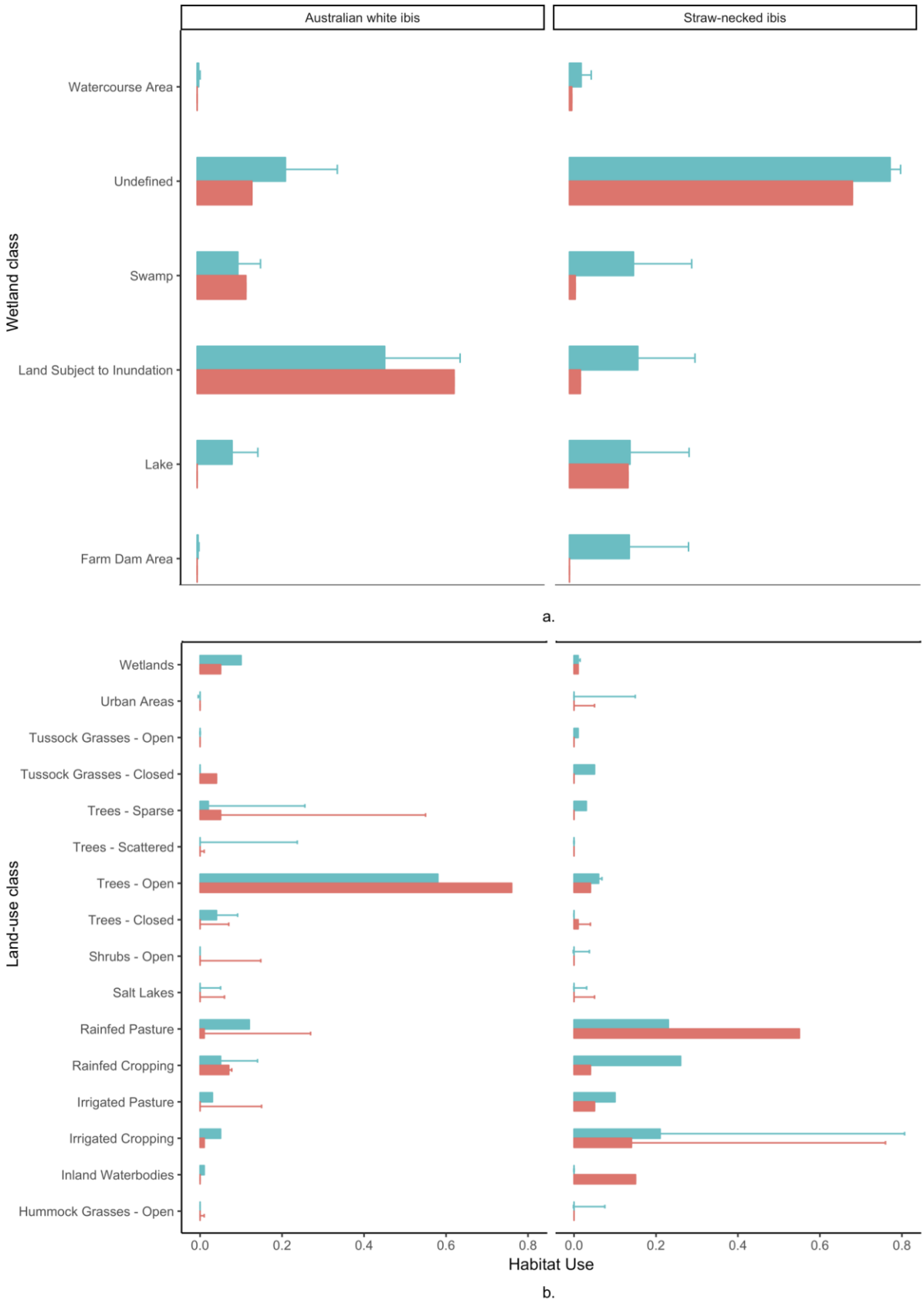


Figure 5 Mean (±SE) habitat use for breeding (one male straw-necked ibis and one female Australian white ibis) and non-breeding individual birds (3 female and 1 male straw-necked ibis, and 1 female and 1 juvenile Australian white ibis), calculated for a. wetland and b. land-use habitat classes.

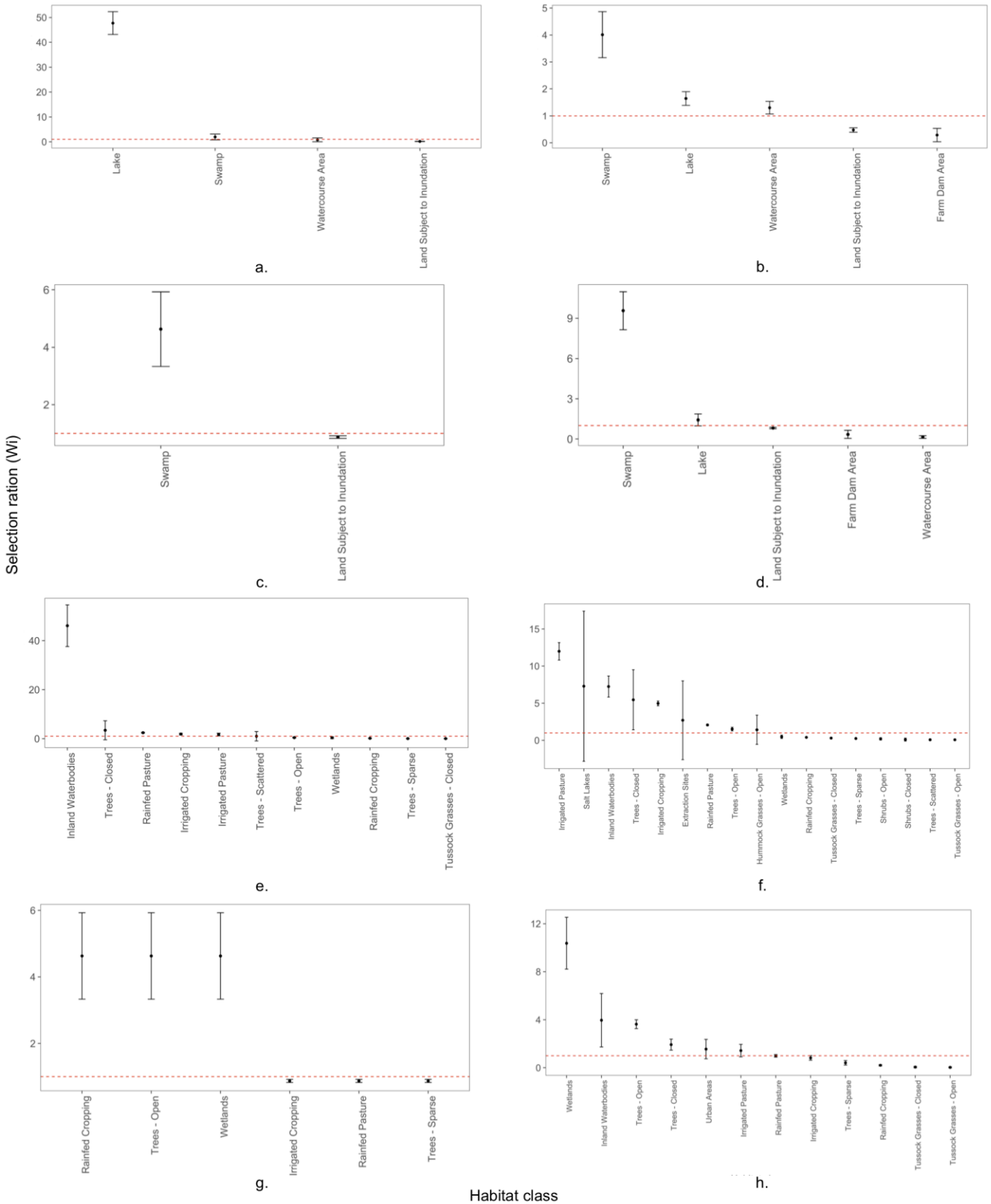


Figure 6 Habitat selection (Manly selection ratio, Wi) (Manly et al. 2007) in wetland habitat classes for a. breeding straw-necked ibis (one male), b. non-breeding straw-necked ibis (3 females and one male), c. breeding Australian white ibis (one female) and d. non-breeding Australian white ibis (one female and one juvenile), and in land-use habitat classes for e. breeding straw-necked ibis, f. non-breeding straw-necked ibis, g. breeding Australian white ibis and h. non-breeding Australian white ibis.

Non-breeding birds

There was a consistent difference between movements of non-breeding straw-necked ibis and Australian white ibis (Figure 7). Straw-necked ibis dispersed further on average from the capture site, and in all directions post-release, while Australian white ibis primarily dispersed north and south. Both species exhibited multidirectional localised movements as well as periodic long flights, however both localised movement and long flights were further for straw-necked ibis than Australian white ibis. Non-breeding straw-necked ibis dispersed an average maximum of 151.7 km (SE=52.55) from the capture site. Throughout the study period, females THRE01, THRE06 and THRE11, and male THRE04, regularly took long flights of 20-70 km, followed by small multidirectional localised movements of up to 5 km, for up to 96 days. When moving locally, they frequently revisited (>50 nsv) locations, while transitory locations were visited infrequently (<30 nsv) and for short durations (<2.0 mnlv). Few straw-necked ibis remained within the Barmah-Millewa Forest for more than 5 days post-release, though THRE06 and THRE11 returned to their nest site for 2 and 9 days respectively post-release. Individual straw-necked ibis also visited Booligal wetlands, the banks of Goulburn and Broken Rivers, rice paddocks, and farmlands.

For non-breeding straw-necked ibis, 82% of points fell outside the wetland layer (Figure 5), and > 15% of points fell within land subject to inundation, farm-dam area, lake and swamp classes. De-selection was significant for land subject to inundation (Wi=0.473, p=0.024). The most frequented habitat for non-breeding birds in the land-use layer was rainfed crop (26%), closely followed by rainfed pasture (23%) and irrigated crop (21%) (Figure 5). Selection was significant for rainfed pasture (Wi=2.072, p<0.001) and trees-open (Wi=1.518, p<0.001), as was de-selection for rainfed crop (Wi=0.405, p<0.001), and trees-sparse (Wi=0.250, p<0.001) classes (Figure 6).

Transmitter failure occurred in 3 of 4 non-breeding Australian white ibis. Individuals dispersed an average maximum of 122.8 km (SE=54.91) from the capture site. Non-breeding Australian white ibis THRE02 did not return to the nest site after release. Throughout the study period she occasionally flew up to 72 km, and then remained within 1 km of a location for 3 to >50 days. She spent most time foraging between the banks of the Goulburn River near Shepparton, and the Broken River, before flying 124 km south to the Dandenong Ranges National Park in April. Mean visitation was high, at 61.47 nsv (Appendix 4), as was mean duration at 2.03 mnlv (Appendix 4). THRE02 reached a maximum displacement of 235 km from the capture site. THRE10, the juvenile Australian white ibis (Figure 7) remained within 1.3 km of the nest site until 12-13 February, in which it departed from the natal wetland and flew north 186 km to Griffith. It then made localised movements of 5-10 km in fields and paddocks until transmission ceased on 21 February. Mean visitation was 29.03 nsv (Appendix 4), and mean duration was 1.99 mnlv (Appendix 4) for THRE10. It achieved a maximum displacement of 198 km from the capture site.

Less than 25% of points fell in undefined regions of the wetland layer for Australian white ibis (Figure 7), and the most frequented habitat was land subject to inundation for non-breeding (51%) birds. Selection was significant for swamp (Wi=9.565, p<0.001); as was de-selection for farm-dam area (Wi=0.343, p<0.001), land subject to inundation (Wi=0.812, p<0.001) and watercourse area (Wi=0.147, p<0.001). The majority of points for non-breeding Australian white ibis (58%) in the land-use habitats fell in trees-open, and greater than 10% of points fell in rainfed pasture (Figure 7). Selection was significant for trees-closed (Wi=1.927, p<0.001), trees-open (Wi=3.633, p<0.001) and wetlands (Wi=10.377, p<0.001), as was de-selection for rainfed crop (Wi=0.204, p<0.001), trees-sparse (Wi=0.410, p<0.001), and tussock grasses-open (Wi=3.633, p<0.01) and closed (Wi=1.927, p<0.001).

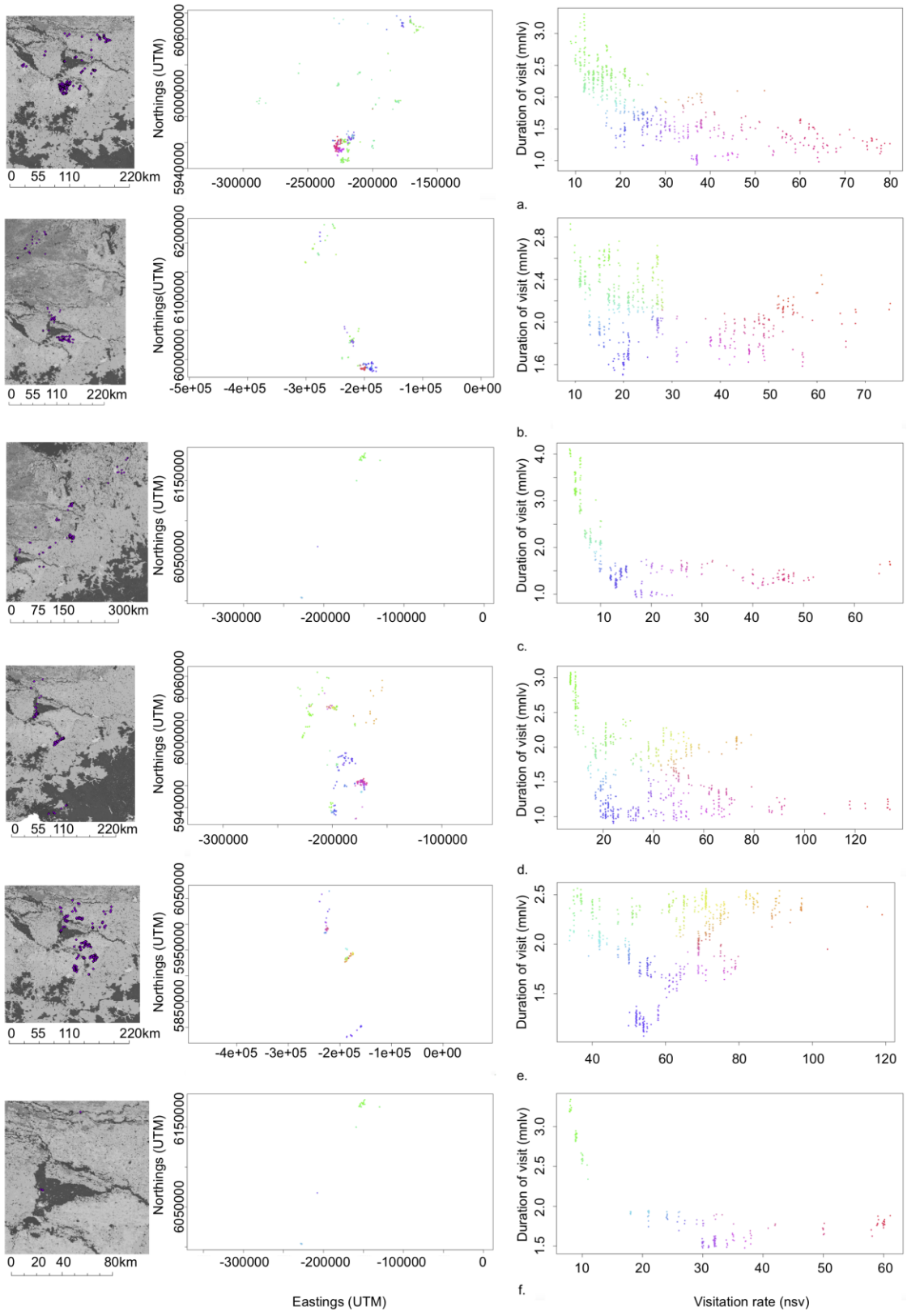


Figure 7 , Study area map for non-breeding birds; straw-necked ibis females a. THRE01, b. THRE06 and c. THRE11, d. straw-necked ibis male THRE04, e. Australian white ibis female THRE02 and f. Australian white ibis juvenile THRE10, showing UTM eastings and northings where coloured points represent the two-dimensional interaction between visitation and duration rates, calculated with time-local convex from a. 616 points (k=25, s=0.01, kmin=0), b. 528 (k=25, s=0.028, kmin=0.2), c. 345 points (k=15, s=0.003, kmin=0.2),

d. 604 points (k=20, s=0.018, kmin=0), e. 644 points (k=55, s=0.02, kmin=0.3), f. 165 points (k=25, s=0.2, kmin=0.11).

Water management application

The movement of ibis to foraging sites can assist in our understanding of what habitats, other than wetlands, are important for ibis in the landscape. For example, the importance of agricultural lands as a foraging site for straw-necked ibis, while the more aquatic habitats were used by Australian white ibis. This knowledge has implications for water and wetland managers. For straw-necked ibis, which forage in agricultural lands, which are frequently beyond the management scope for water and wetland managers, it highlights the life history stage for which water management is most important in supporting straw-necked ibis. Due to the specific water requirements that straw-necked ibis need for successful breeding (Brandis et al. 2011, 2018) it is during this life history stage that water management is most critical. For Australian white ibis, the water requirements for breeding appear to be less stringent than those for straw-necked ibis, however the provision of aquatic habitat for foraging is also important in supporting breeding events.

Acknowledgements

Funding for purchase of the GMS transmitters was provided by NSW National Parks and Wildlife Service. Assistance with field work was provided by EWKR Waterbirds Team. Thank you to J. McCann and C. Callaghan from UNSW for their assistance with statistical analyses, and to K. Ward from the Goulburn Broken Catchment Management Authority for sharing his knowledge of environmental flow provision to the Barmah-Millewa Forest. Thank you also to M. Bellio, members of CSIRO, NSW National Parks and the Australian Bird and Bat Banding Scheme for facilitating fieldwork.

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Appendix 4 ABBBS bands, individual colour bands (DG- dark green, W- white, B- blue, Y- yellow, R-red, O-orange, BL- black) and summary of time-use metrics for individual birds.

Species	Sex	Tracker number	ABBBS band number	Individual colour bands	Mean duration	Range duration	Mean visitation rate	Range visitation rate
Straw-necked ibis	F	THRE01	121-60761	DG-W	1.88	1.05-3.25	28.50	9-80
	F	THRE06			2.12	1.6-2.89	27.96	9-75
	F	THRE11	121-60768	DG-B	1.95	1-4	18.25	4-67
	M	THRE03	121-60779	DG-DG	1.66	1-3.38	127.02	8-233
	M	THRE04			1.67	1-3	38.4	7-134
	M	THRE05[a]	121-60763	DG-Y				
Australian white ibis	F	THRE02	121-60762	DG-R	2.03	1.15-2.51	61.47	34-119
	F	THRE07			1.45	1.04-2.7	65.19	8-133
	M	THRE08[a]	121-60767	DG-O				
	M	THRE09[a]	121-60764	DG-G				
	J	THRE10		BL	1.99	1.48-3.38	29.03	8-61

[a]Excluded from time-use analysis due to lack of data points

Appendix 5 Summary of two-factor analyses of variance testing differences in weight, number of days tracked, number ofdata points received and maximum displacement from the capture site across the 2 species and sexes for all tracked birds excluding juvenile THRE10.

Weight	df	MS	F	p
Species	1	15371	2.077	0.209
Sex	1	45506	6.150	0.559
Species: sex	1	8495	1.148	0.333
Residuals	5	7400		
Days tracked				
Species	1	2178	0.803	0.411
Sex	1	5859	2.159	0.202
Species: sex	1	3296	1.215	0.321
Residuals	5	2713		
Data points received				
Species	1	391908	0.628	0.464
Sex	1	136415	2.188	0.199
Species: sex	1	496643	0.796	0.413
Residuals	5	624123		
Max displacement				
Species	1	2814	0.139	0.728
Sex	1	18551	1.183	0.338
Residuals	5	15684		

2 observations missing (THRE05 and THRE09)

Appendix 6 Summary of *p*-values from composition analysis of habitat selection (Manly selection ratio (Manly et al. 2007)) by straw-necked ibis and Australian white ibis breeding and non-breeding birds, indicating selection or de-selection of habitat classes (RStudio team 2016; Calenge 2011).

Habitat class- Wetland	Straw-necked ibis		Australian white ibis	
	Breeding	Non-breeding	Breeding	Non-breeding
Farm dam area		<0.001[b]		<0.001[b]
Lake	<0.001[a]	<0.001[a]		0.063
Land subject to inundation	<0.001[b]	<0.001[b]	0.024[b]	<0.001[b]
Swamp	0.110	<0.001[a]	0.663	<0.001[a]
Watercourse area	0.599	0.119		<0.001[b]
Habitat class- Land-use				
Extraction sites		0.528		
Hummock grasses		0.676		
Inland waterbodies	<0.001[a]	<0.001[a]		0.009
Irrigated pasture	0.012	<0.001[a]		0.106
Irrigated cropping	<0.001[a]	<0.001[a]	0.480	0.064
Rainfed pasture	<0.001[a]	<0.001[a]	<0.001[a]	0.912
Rainfed cropping	<0.001[b]	<0.001[b]	<0.001[b]	<0.001[b]
Salt lakes		0.222		
Shrubs - closed		<0.001[b]		
Shrubs - open		<0.001[b]		
Trees - closed	0.219	0.030		<0.001[a]
Trees - open	<0.001[b]	<0.001[a]	<0.001[a]	<0.001[a]
Trees - sparse	<0.001[b]	<0.001[b]	<0.001[b]	<0.001[b]
Trees - scattered	0.986	<0.001[b]		
Tussock grasses - closed	<0.001[b]	<0.001[b]		<0.001[b]
Tussock grasses - open		<0.001[b]		<0.001[b]
Urban areas				0.178
Wetlands	0.001[b]	<0.001[b]	0.034	<0.001[a]

[a] Selection[b] De-selection

[c] Significant values defined by Bonferroni correction factor for each group within wetland layer (straw-necked ibis breeding=0.0125, non-breeding=0.01 and Australian white ibis breeding=0.025, non-breeding=0.01), and the land-use layer (straw-necked ibis breeding=0.005, non-breeding=0.003, Australian white ibis breeding=0.008, non-breeding=0.004) analyses.

Webster, E. (2017) Similarities and differences in breeding ecology of straw-necked ibis Threskiornis spinicollis and Australian white ibis T. moluccus in response to environmental flows. Honours thesis, UNSW.

Waterbird breeding responses to flows

Authors: Emily Webster, Kate Brandis

Webster, E. (2017) Similarities and differences in breeding ecology of straw-necked ibis *Threskiornis spinicollis* and Australian white ibis *T. moluccus* in response to environmental flows. Honours thesis, UNSW.

This work was undertaken by Emily Webster as part of an BSc honours (2016-2017) degree at the UNSW supervised by Richard Kingsford, Kate Brandis and John Martin.

Objective

To quantify the relationships between river flow volumes in the Murray River and the breeding responses on colonial waterbirds in Barmah-Millewa.

Introduction

The Barmah-Millewa Forest supports breeding of straw-necked ibis and Australian white ibis populations (Leslie 2001; Hale & Butcher 2011), which is triggered by exceedance of two flow thresholds: (1) the total volume of flow > 2,300,000 megalitres (ML) for 6 months before breeding (July-December), and (2) flow > 15,000 ML day^{-1} for 30-50 days, measured downstream of Yarrawonga Weir (Leslie 2001; Arthur et al. 2012). Despite reasonable understanding of the relationships between these abiotic drivers and total breeding effort (Arthur et al. 2012), little is known about the proximate role of biotic drivers, particularly food requirements for raising chicks and how this might change with flooding and vary among species. Straw-necked ibis and Australian white ibis may nest together or separately in the Barmah-Millewa Forest (Leslie 1995; Leslie 2001), and forage in mixed-species flocks.

Breeding of colonial waterbirds

The breeding colony of straw-necked ibis and Australian white ibis at Reed Beds Swamp in the Barmah-Millewa Forest was monitored in 2015 and 2016/17. I collated historical daily flow data (ML day^{-1}) from Yarrawonga Weir (409025, 1961-2017) on the Murray River, comparing this to breeding events of the two ibis species (1961-2012 historical data, retrieved from http://waterinfo.nsw.gov.au; Leslie 1995; Leslie 2001; Arthur et al. 2012; 2013-2017 data, Kingsford & Porter 2016, retrieved from http://aws.ecosystem.unsw.edu.au; and this study). There were 11 years of missing data. I calculated flow metrics for the two thresholds that trigger breeding (Arthur et al. 2012) and examined their relationship with linear regression. Timing of breeding in 2015 and 2016/17 was compared to daily flows for the two species. Water depth was measured adjacent to nests within Reed Beds Swamp each fortnight (2015) and every 3-4 days (2016/17). Declining flood

depth can cause nest abandonment by colonial waterbirds (Leslie 2001; Kingsford & Auld 2005; Brandis et al. 2011a). I tested the relationship between depths at these sites and flow, allowing for a transmission time of 3 days (K. Ward, pers. comm.), with linear regression. Colony sizes for the 2 species in both breeding events were also estimated from aerial survey data (Kingsford & Porter 2016, retrieved from http://aws.ecosystem.unsw.edu.au).

Nests of straw-necked ibis and Australian white ibis were monitored within Reed Beds Swamp in 2015 and 2016/17. Data were collected every 2 weeks during the breeding events: 11 November (T1), 2 December (T2), 16 December (T3) in 2015/16, and 16 November (T1), 3 December (T2), 15 December (T3), 27 December (T4), 7 January (T5) and 19 January (T6) in 2016/17. Nest clusters of both species were marked during the first trip (25 in 2015/16 and 47 in 2016/17) with new clusters (4 in 2015/16 and 7 in 2016/17) added subsequently. Clusters included 30-300 nests on contiguous platforms of trampled vegetation, surrounded by water or walls of thick reeds and rushes. The number of eggs and chicks present was recorded for each observable nest in a cluster, with chicks classified into 4 age cohorts: “chick”, “squirter”, “runner” or “flapper” (Brandis et al. 2011b). Some data from individual nests were missing across trips because tags were lost, or nests or clusters were destroyed, meaning nest comparisons could not be made between surveys. When chicks were mobile and moved between nests and clusters (runners and flappers), the total number of chicks of each stage for the cluster was recorded. I assumed 100% success when counts of old (runner and flapper) chicks that had probably crèched on a cluster, exceeded previous nest estimates. Reproductive success was measured at the offspring level, as the percentage offspring surviving between consecutive surveys (separated by stage: egg, chick, squirter, runner, flapper); and at cluster (“clump”) level, where clumps survived if they contained live offspring. There were also some large mixed-species clusters in the colony, which could not be separated into species.

Average clutch size was the number of chicks and eggs in each nest of an active cluster when offspring were first detected for each species, divided by the total number of active nests. I used a linear mixed effects model to compare clutch size between species and years, checking for significant interactions and including clump identification numbers within sites as a random factor (**Error! Reference source not found.**). I separately tested for relationships between reproductive success of offspring and clumps in relation to species, year, site, trip and mean water depth, as well as interactions among species, year and water depth. I tested for assumptions of normality and variance with residual and normal Q-Q plots in R (RStudio Team 2016).

Results

Breeding responses to flow

There was a positive linear relationship between the 2 flow thresholds for breeding, total volume July-December >2,300,000 ML and >30 consecutive days > 15,000 ML day^{-1} (Figure 8, number of days exceeding 15,000 ML day^{-1} = 403024 x (total volume July-December) + 1360588, R2=0.80, p<0.001). Breeding occurred in all years, when both thresholds were met (n=25 years) (1961-2017): 24 years when both species bred and in the year when only Australian white ibis bred in 1981 (Figure 8). Neither threshold was met in 2014/15, or 2015/16 but both species bred (Figure 8). When only the daily flow threshold was met, breeding occurred in all years (Figure 8). When only the total volume threshold was met, breeding occurred in all but one year (1968) (Figure 8). For the 11 years where breeding data were missing (1961-2017), this strong relationship predicted likely breeding in 1970 (Figure 8). This meant that breeding frequencies for straw-necked ibis and Australian white ibis in the Barmah-Millewa Forest were respectively 29 and 30 times since 1961 (1961-2017), about 7 times in every 10 years for both species, coinciding with periods of high flow in the River Murray (Figure 8).

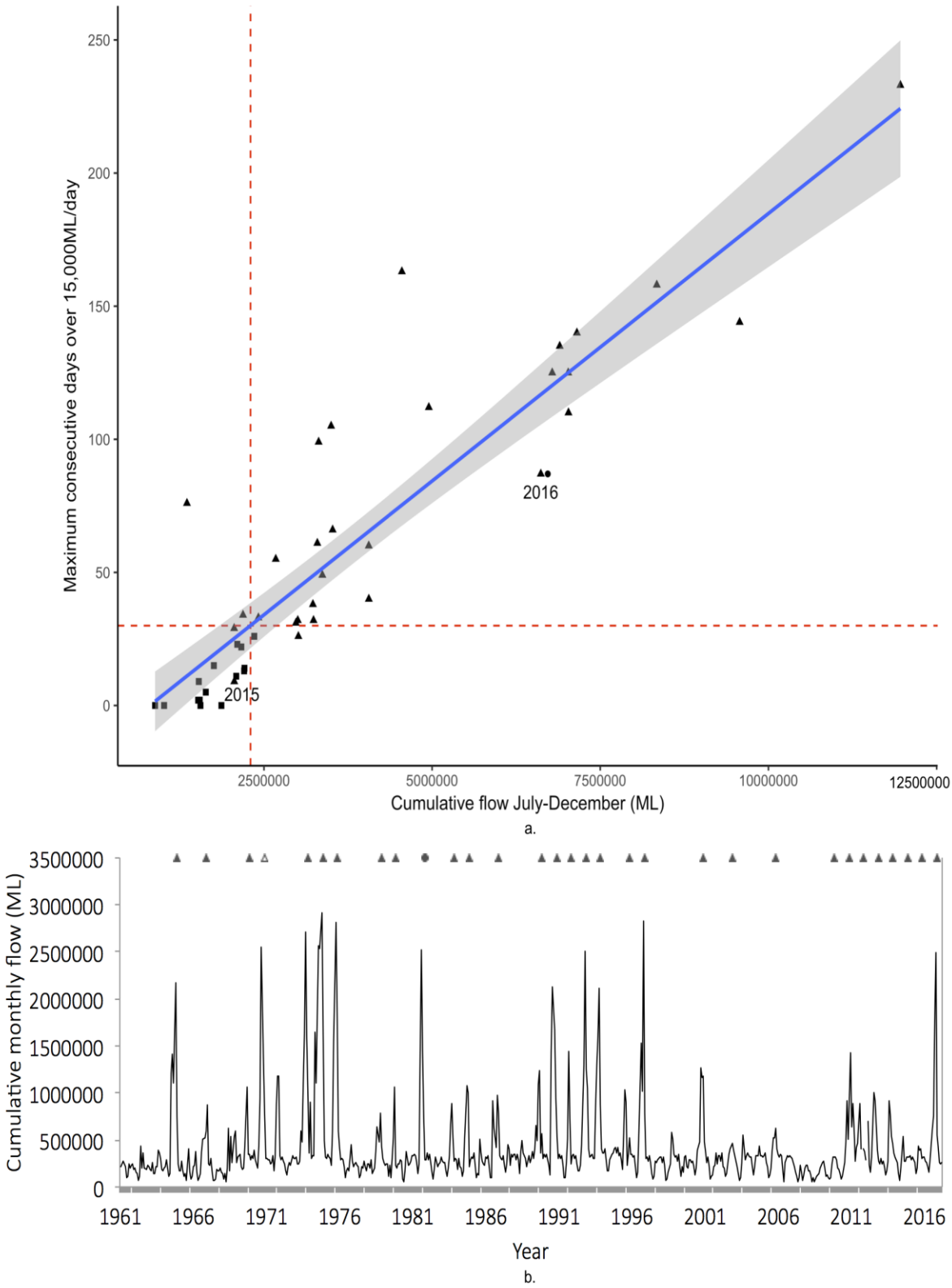


Figure 8 a. Line of best fit (±95% confidence intervals) between the 2 breeding flow thresholds in the Barmah-Millewa Forest, measured at Yarrawonga flow gauge (dashed lines, total volume July-December >2,300,000 ML and >30 consecutive days > 15,000 ML day^{-1}; Arthur et al. 2012) (study years labelled) and b. cumulative monthly flow (ML, 1961-2017), measured at Yarrawonga flow gauge, showing known (filled triangles) and predicted (unfilled triangles) breeding years for straw-necked ibis and Australian white ibis and known breeding year of Australian white ibis only (filled circles).

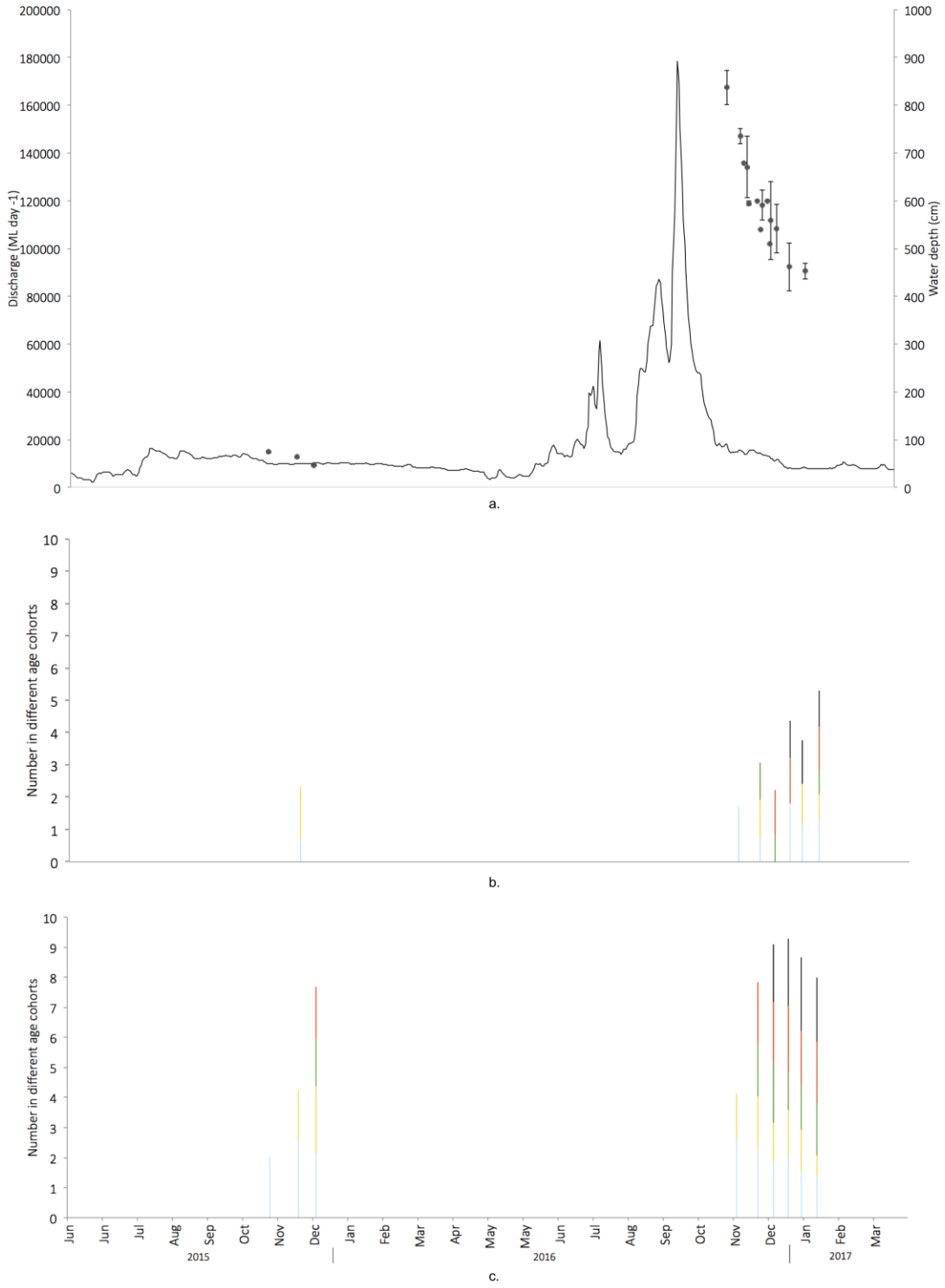


Figure 9 a. Daily flow (ML day^{-1}, June 2015-April 2017), with mean (±SE) water depth (filled circles) averaged across 4 sites (East, West, North, South) at the Reed Beds Swamp colony in the Barmah-Millewa Forest and b. total number (log transformed) in straw-necked ibis chick stages (egg (blue), chick (yellow), 'squirter' (green), 'runner' (red) 'flapper' (black) counted at all sites in 2015 and 2016/17, and the equivalent for c. Australian white ibis.

In 2015 and 2016/17, there were respectively 987 (43.36% (n=428) straw-necked ibis and 56.64% (n=555) Australian white ibis) and 1857 (58.93% (n=1092) straw-necked ibis and 41.07% (n=761) Australian white ibis) breeding birds recorded, and about 500 (40% straw-necked ibis and 60% Australian white ibis) and 1500 (55% straw-necked ibis, 45% Australian white ibis) nests in the Reed Beds Swamp colony. Single species colonies of Australian white ibis occurred, although the two species predominantly nested together.

Straw-necked ibis laid in late November (Figure 9) while Australian white ibis continued to lay eggs into December in 2015 (Figure 9). Both species had laid by mid-November in 2016/17 (Figure 9), and Australian white ibis chicks had begun to hatch, with straw-necked ibis hatching in late November. In 2015, flow volume was low (Figure 9), compared to the following year (2016/17) when both breeding thresholds were exceeded, with several large peaks in flow July-October (Figure 9). Water depth was significantly positively related to flow measured at Yarrawonga (East R^2 =0.57, p<0.01, North R^2 =0.80, p<0.01, South R^2 =0.56, p=0.05, West R^2 =0.54, p<0.01,).

Average clutch size was similar for both species ($F_{1, 29}$=0.117, p=0.790, Table 2, **Error! Reference source not found.**), but slightly larger for both species in 2016/17 compared to 2015, although not significantly so ($F_{1, 29}$=0.016, p=0.900, **Error! Reference source not found.**).

Table 2 Mean (±SD) clutch size and percentage success at offspring and clump level for straw-necked ibis and Australian white ibis nesting in the Reed Beds Swamp of the Barmah-Millewa Forest in 2015 and 2016/17, across surveys (T1-T2, T2-T3, T3-T4, T4-T5, T5-T6) showing number of clumps and nests (used to calculate clutch size/total monitored).

Breeding	Species	Clutch size ±SD		Number of clumps	Number of nests	T1-T2[a]	T2-T3[b]	T3-T4[c]	T4-T5[d]	T5-T6[e]
2015	Straw-necked ibis	2	Offspring	1/1	22/22	N/A	0			
			Clump			100	100			
	Australian white ibis	2.13±0.662	Offspring	8/15	104/152	81.22±0.037	77.74±0.032			
			Clump			62.5±0.180	100±0.000			
2016/17	Straw-necked ibis	2.34±0.299	Offspring	3/8	21/25	43.14±0.246	93.33±0.236	93.33±0.293	47.22±0.477	37.50±0.142
			Clump			100±0	66.67±0.333	100±0	40±0.577	100±0
	Australian white ibis	2.15±0.817	Offspring	22/27	183/201	74.88±0.100	76.15±0.259	94.70±0.057	71.53±0.186	57.42±0.352
			Clump			90.48±0.143	91.30±0.102	100±0	87.50±0.167	80.95±0.236

[a] 11 November-2 December 2015 and 16 November-3 December 2016

[b] 2 December-16 December, 2015 and 3 December-15 December, 2016

[c] 15 December-27 December 2016

[d] 27 December 2016-6 January 2017

[e] 6 January-19 January 2017

There were significant differences in offspring success between the 2 species ($F_{1, 28}$=9.609, p=0.004) and throughout the breeding events (T1-T6, $F_{3, 28}$=5.962, p=0.001). Number of offspring surviving through to the next survey was less for straw-necked ibis than Australian white ibis (Table 2). The model at the offspring level explained a reasonable amount of the reproductive success (R^2=0.647), with a significant interaction between year and species ($F_{1, 28}$=5.009, p=0.033) (Figure 10). Offspring success of straw-necked ibis significantly more affected by year compared with success of Australian white ibis (Figure 8). Offspring success was also low early and late in the event in both years for both species ($F_{4, 28}$=5.962, p=0.001, Table 2). Offspring success was not significantly related to year, site or water depth (**Error! Reference source not found.**).

Success at the clump level also varied significantly across the survey trips (Table 2, $F_{3, 28}$=2.864, p=0.042), with significant interactions between year and species ($F_{1, 28}$=4.472, p=0.044, Figure 10, Appendix 5), and species and depth ($F_{1, 28}$=4.229, p=0.049, Figure 10, **Error! Reference source not found.**). The model at the clump level explained some of the reproductive success (R^2= 0.549). Clump success was generally high but was lowest in 2015 for Australian white ibis (T1-T2) and in 2016/17 for straw-necked ibis (T4-T5), (Table 2). Clump success was significantly more related to species in 2016/17 than 2015, and more related to water depth for straw-necked ibis compared to Australian white ibis.

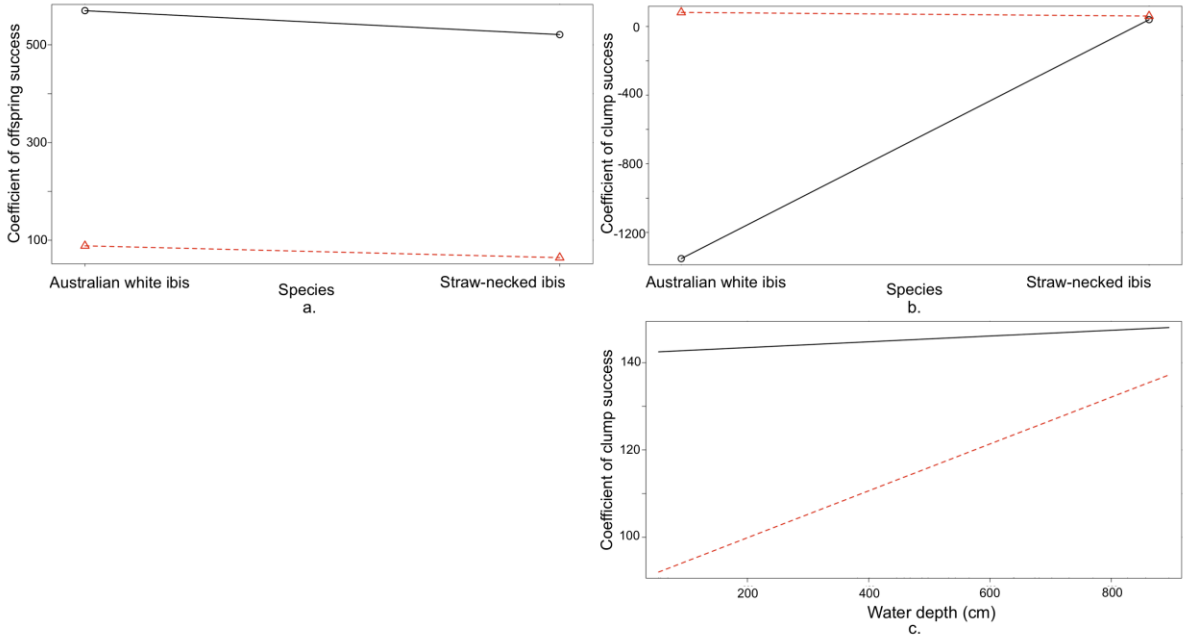


Figure 10 Significant interaction effects for a) reproductive success of offspring (i.e. percentage surviving through to next survey) between year (2015- red, dashed and 2016/17- black, solid) and species (straw-necked ibis and Australian white ibis) and for clump success (percentage remaining active through to next survey) between b) year (2015- red, dashed and 2016/17- black, solid) and species, and c) water depth (cm) and species (straw-necked ibis- red, dashed and Australian white ibis- black, solid).

Discussion

Water depths in Reed Beds were positively correlated with flows at Yarrawonga, relationships between reproductive success and water depths were strongest for straw-necked ibis.

In 45 years, both species bred about every 7 years in 10, triggered when high flow thresholds were met, but also breeding in 2014/15 and 2015 when flows were small. Larger flows in 2016/17 triggered a breeding event 88.15% larger than in the previous year. Offspring success of straw-necked ibis was significantly

lower than that of Australian white ibis, with greatest loss occurring at young chick and egg stages. Straw-necked ibis responded later to flow thresholds triggering breeding, nesting later than Australian white ibis.

Conclusions and applications

This study is the first that compares breeding responses of two closely related ibis that nest in mixed colonies. The results of this study resulted in improved understanding of relationships between river flows and nesting habitat inundation and the role this plays in reproductive success. This study provided insight into the different responses of straw-necked ibis and Australian white ibis and identified that the reproductive success of straw-necked ibis is more closely linked to water depth than it is for Australian white ibis.

Management applications

This study supports the identified flow requirements for colonial waterbird breeding at Reed Beds with a positive linear relationship between the 2 flow thresholds for breeding, total volume July-December >2,300,000 ML and >30 consecutive days > 15,000 ML day^{-1}. Australian white ibis and straw-necked ibis bred on average 7 out of 10 years, coinciding with periods of high flow in the River Murray. This long-term study presents a baseline and target against which long term water management plans can be developed.

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Murray-Darling Basin Environmental Water Knowledge and Research Project

Waterbird Theme Research Report

Appendix 10

Theme Data Inventory

Data	Custodian	Contact	Location
Review and synthesis of known waterbird responses to flows and flooding in Australia, with an emphasis on the Murray-Darling Basin and knowledge gaps	CSIRO	Heather McGinness	McGinness, H. M. (2016) *Waterbird responses to flooding, stressors and threats*. CSIRO, Australia. Literature review report. www.ewkr.com.au
Review and synthesis of international literature regarding how various other stressors and threats affect waterbirds (habitat loss, fragmentation and change, predation, climate change, pollution, disease, human disturbance, competition), and how these interact with the effects of flows.	CSIRO	Heather McGinness	McGinness, H. M. (2016) *Waterbird responses to flooding, stressors and threats*. CSIRO, Australia. Literature review report. www.ewkr.com.au
Detailed movement and habitat-use studies of individual birds over the duration of the EWKR project using solar-powered GPS tracking devices	CSIRO	Heather McGinness	CSIRO databases and publications; https://research.csiro.au/ewkrwaterbirds/
Motion-sensing and time-lapse camera nest monitoring	CSIRO	Heather McGinness	CSIRO databases and publications; https://research.csiro.au/ewkrwaterbirds/
On-ground tagged-nest and water depth monitoring	UNSW	Kate Brandis	UNSW databases and publications
Colony mapping via drone survey	UNSW	Kate Brandis	UNSW databases and publications
Analysis of chick diet/energy sources	UNSW, CSIRO	Straw-necked ibis and royal spoonbill samples: Heather McGinness Australian white ibis samples: Kate Brandis	CSIRO and UNSW databases and publications
Stable isotope analysis of chick energy source assimilation	UNSW, CSIRO	Straw-necked ibis and royal spoonbill samples: Heather McGinness Australian white ibis samples: Kate Brandis	CSIRO and UNSW databases and publications

Measurement of chick biometrics and modelling of chick energy requirements	CSIRO	Heather McGinness	CSIRO databases and publications; O'Brien, L., and McGinness, H. (In review) Ibis and spoonbill chick energy requirements. Wetlands Ecology and Management journal. O'Brien, Lauren and McGinness, Heather M. (2018) *Waterbird chick energy requirements.* CSIRO, Australia. Technical Report.

Murray-Darling Basin Environmental Water Knowledge and Research Project

Waterbird Theme Research Report

Appendix 11

Theme outputs

See also Appendix 4: Theme engagement and communications activities

- MDB EWKR progress reports
- McGinness, H. M. (2016) *Waterbird responses to flooding, stressors and threats*. CSIRO, Australia. Literature review report. www.ewkr.com.au
- McGinness HM, Brandis K (2016) *Waterbird breeding success in the Reed Beds wetlands, Millewa Forest NSW 2015-2016*.
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- *MDB EWKR Waterbird Theme Factsheet: Background and Work Program.* www.ewkr.com.au
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- McGinness, H., Brandis, K., Doerr, V., Kingsford, R., Mac Nally, R., Martin, J. (2017) *Feathering their nests.* RipRap Edition 40, 9-11
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- O'Brien, Lauren and McGinness, Heather M. (2018) *Waterbird chick energy requirements.* CSIRO, Australia. Technical Report.
- *'We are not all bin chickens'* A short video explaining that Australia has three types of ibis: https://open.abc.net.au/explore/254033;
- McGinness (2018) *Australian project using remote nest cameras and GPS satellite tracking to study ibises and spoonbills*. IUCN SSC Stork Ibis and Spoonbill Specialist Group website http://storkibisspoonbill.org/news/australian-project-using-remote-nest-cameras-and-gps-satellite-tracking-to-study-ibises-and-spoonbills/
- ECOS article *'The story of Australian waterbirds - the length and breadth of the Murray-Darling Basin'.* ECOS Issue 243 https://blogs.csiro.au/ecos/ibis/;
- *'Waterbird's stories revealed through satellite tracking'* Spatial Source: https://www.spatialsource.com.au/gpsnav/waterbirds-stories-revealed-through-satellite-tracking.
- Wenger, L., and McGinness, H. (2018) *'Waterbird Chick Development: A Visual Guide to Selected Australian Species'.* CSIRO, Canberra, Australia. *Available for preview and purchase (at cost) here: http://au.blurb.com/bookstore/invited/7714996/2775e42c847608953b28c7020e788bd36b9f0770*

- A 5-minute film/ video describing the MDB EWKR Waterbird Theme research and preliminary results https://open.abc.net.au/explore/276633
- Waterbirds Australia: Science for Management. Facebook Page. Co-designed the content, co-wrote the text and supplied images. https://www.facebook.com/ColonialWaterbirdScience/
- Waterbird breeding and movements: Knowledge for water managers. Website. Co-designed the content, co-wrote the text and supplied images. https://research.csiro.au/ewkrwaterbirds/
- Regular email updates to subscription list of over 100 water managers, policy makers, scientists, and landholders
- 'Water management for waterbirds in Australia: Lessons learned' (in development) Glossy full-colour magazine-style booklet.
- Waterbird species, sex and age-group profiles distributed through social media, email and websites (examples are presented below)
- *Also: General public and interest group presentations, Stakeholder presentations, Scientific presentations, Radio interviews, Newspaper interviews*

Scientific publications in preparation

O'Brien, L., and McGinness, H. (In review) **Ibis and spoonbill chick energy requirements: implications for management**	Submitted and undergoing review for Wetlands Ecology and Management journal.
McGinness, H.M. et al. (TBC) **Satellite tracking ibis and spoonbill movements and habitat choices**	Including critical habitat characteristics (nesting, foraging, roosting, stopover); Foraging trip distances; Pairing of roosting and foraging sites; Critical routes/movement corridors; Land and water management actions/policies required
McGinness, H.M. et al. (TBC) **Quantifying ibis and spoonbill egg and chick survival rates and mortality drivers using motion-sensing and time-lapse cameras**.	With an emphasis on the roles of stressors/threats/pressures other than water (predation, habitat change, weather extremes, disturbance etc.).
Authors (TBC) **Ibis and spoonbill diets: energy value/content and trophic structure.**	Based on the Waterbird Theme – Foodweb Theme collaboration project
McGinness, H.M. et al. (TBC) **Meeting ibis and spoonbill energy requirements during breeding events.**	Combining the results from the energy requirements and energy values projects to model real world scenarios with management implications.
Brandis, K. et al (TBC) **Colonial-nesting waterbird breeding attempts, success and flow relationships in the Murray Darling Basin.**	Bringing together EWKR, LTIM and historical data from all colonies feasible
McGinness, H.M. et al. (TBC) **Limits to colonial-nesting waterbird population growth and implications for management.**	Including a review of Threskiornithidae survival/mortality rates and their drivers in Australia and overseas.
McGinness, H.M. et al. (in prep) **Pressures on waterbird populations: Managing more than just water.**	Based on the EWKR Waterbird Theme literature review report

Waterbirds Australia

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The field guide to chick development produced by Lucy Wenger and Heather McGinness (CSIRO) ***Available for preview and purchase (at cost) here: http://au.blurb.com/bookstore/invited/7714996/2775e42c847608953b28c7020e788bd36b9f0770***

HOW TO DISTINGUISH BETWEEN MALE AND FEMALE STRAW-NECKED IBIS....

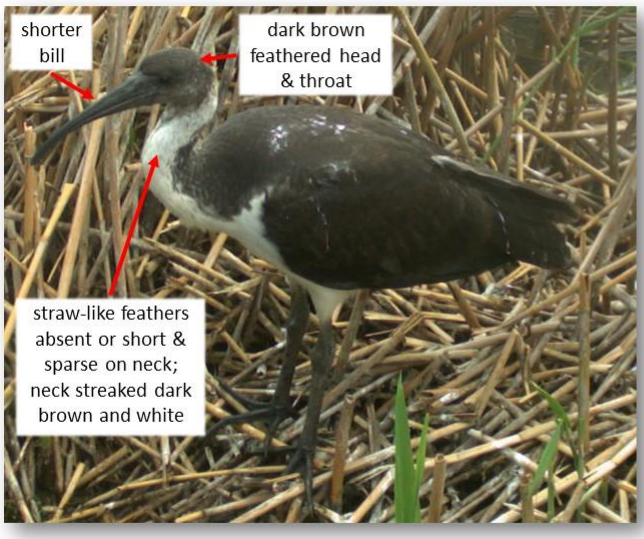


Female
Solid black band across chest

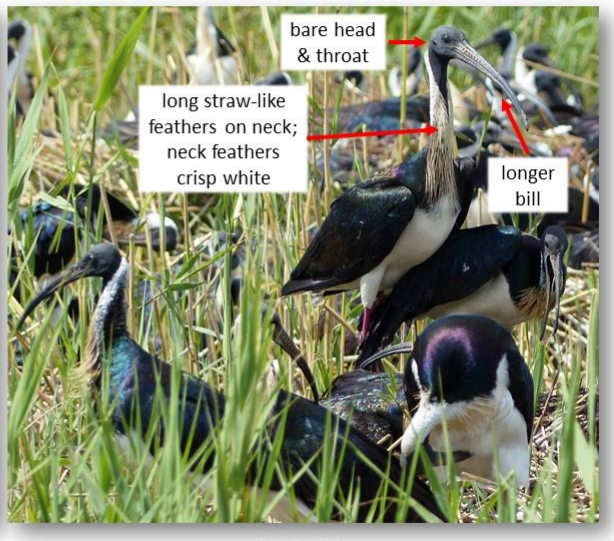
Male
Broken black band across chest



HOW TO DISTINGUISH BETWEEN JUVENILE AND ADULT STRAW-NECKED IBIS....



Juvenile



Adult

Pictorial guides for straw-necked ibis produced by Freya Robinson and Heather McGinness (CSIRO) and distributed through social media, email and websites.

PROFILE #1:

Australian White Ibis





Species name: *Threskiornis molucca*

Lives: Widespread throughout south-east, south-west and northern Australia. Resident in southern New Guinea and New Zealand.

Breeds: Largest breeding colonies in inland south-east Australia and east Australian coast.

Feeds: Often seen wading in wetlands and marine habitats with bill immersed probing for fish, frogs and aquatic invertebrates, or in flocks on grasslands and ovals looking for insects.

Did you know...

Young ibis like to hang out together?

Once chicks are a few weeks old they begin to group together with others from neighbouring nests, forming 'crèches'. It's not uncommon to have crèches with both Australian white ibis and straw-necked ibis chicks.

Species profile produced by Freya Robinson and Heather McGinness (CSIRO) and distributed through social media, email and websites.

PROFILE #2:

Straw-necked Ibis



Species name: *Threskiornis spinicollis*

Lives: Originally endemic to Australia however now occurs in southern New Guinea. Widespread throughout Australia, preferring drier inland to coastal habitats.

Breeds: Largest breeding colonies in inland south-east Australia.



Feeds: Feed more on inland grasslands and pasture than Australian white ibis, favouring locusts and other insects.

Did you know...

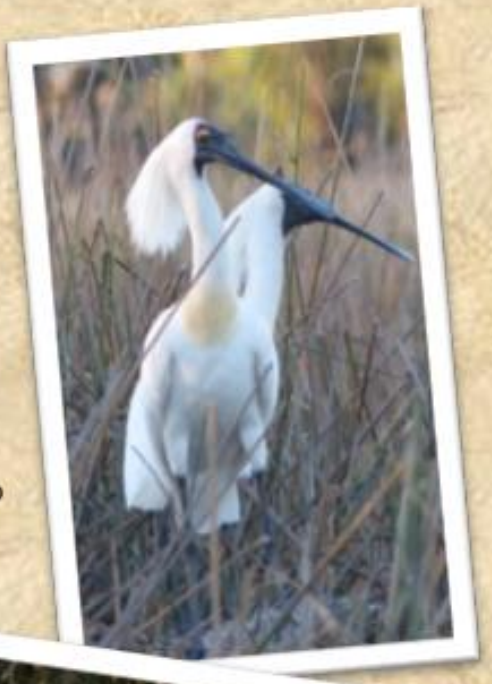
Straw-necked ibis parents are dedicated?

Both parents take it in turns to incubate the eggs and tend to young at the nest, continuing to feed their young until they can fly.

Species profile produced by Freya Robinson and Heather McGinness (CSIRO) and distributed through social media, email and websites.

PROFILE #3:

Royal Spoonbill



Species name: *Platalea regia*

Lives: Originally endemic to Australia with populations now in NZ. Occurs throughout eastern and northern Australia.

Breeds: Largest breeding colonies thought to be in inland and coastal south-east Australia.

Feeds: Feed in open shallow water of inland wetlands, sheltered marine habitats and submerged grasslands by wading and sweeping spatula-shaped bill through water to locate fish and aquatic invertebrates.



Did you know....

Spoonbills like to impress their partner?

Spoonbills use all kinds of display techniques to court potential mates and reaffirm partnerships throughout the breeding season. This includes head shaking and presenting their partner with sticks or nesting material.

Species profile produced by Freya Robinson and Heather McGinness (CSIRO) and distributed through social media, email and websites.

Murray-Darling Basin Environmental Water Knowledge and Research Project

Waterbird Theme Research Report

Appendix 12

Theme Engagement and Communications Activities

- **Websites:** https://research.csiro.au/ewkrwaterbirds/ and http://ewkr.com.au/
- **Facebook and Twitter: Waterbirds Australia**
  https://www.facebook.com/ColonialWaterbirdScience/
- **Regular email updates on satellite tracked birds: Contact Heather to subscribe**

Date	Person(s)	Type	"Location"	Audience	Content / Activity
28/07/2015	Heather McGinness	Workshop	Jurisdictional workshop, Dubbo	Stakeholders and/or Water Managers	Presented Waterbird Theme material and participated in discussions
27/08/2015	Heather McGinness	Meeting	Scientific Advisory Group (SAG), Sydney	Stakeholders and Academic	Presented Waterbird Theme material and participated in discussions
17/09/2015	Heather McGinness	Meeting	Michael Wilson, MDBA and Kate Brandis, UNSW, Teleconference	Stakeholders and/or Water Managers	Discussion re. waterbird satellite tracking research, Participated in the meeting and supplied information
19/10/2015	Heather McGinness	Meeting	Keith Ward, GBCMA and Waterbird Theme staff, Teleconference	Stakeholders and/or Water Managers	Discussion re. waterbird research, Organised the meetings, wrote the agenda, wrote and circulated meeting notes / minutes
10/02/2016	Heather McGinness	Workshop	EWKR/DoE Theme Coordinators, JRG and SAG Workshop, Canberra	Stakeholders and Academic	Presented Waterbird Theme material and participated in discussions
19/05/2016	Heather McGinness	Workshop	Jurisdictional workshop, Dubbo	Stakeholders and/or Water Managers	Presented Waterbird Theme material and participated in discussions
09/06/2016	Heather McGinness	Workshop	Stakeholders / jurisdictional workshop, Canberra	Stakeholders and/or Water Managers	Presented Waterbird Theme material and participated in discussions
01/08/2016	Heather McGinness	News article	Australian River Restoration Centre	General	Colonial-nesting waterbirds: Science for management., Co-wrote the text and supplied images. https://arrc.com.au/
14/06/2016	Heather McGinness	Workshop	Waterbird-Food Webs Theme workshop, Canberra	Stakeholders and Academic	Organised the workshop and catering, wrote the agenda, prepared material, facilitated the workshop, wrote and circulated meeting notes / minutes

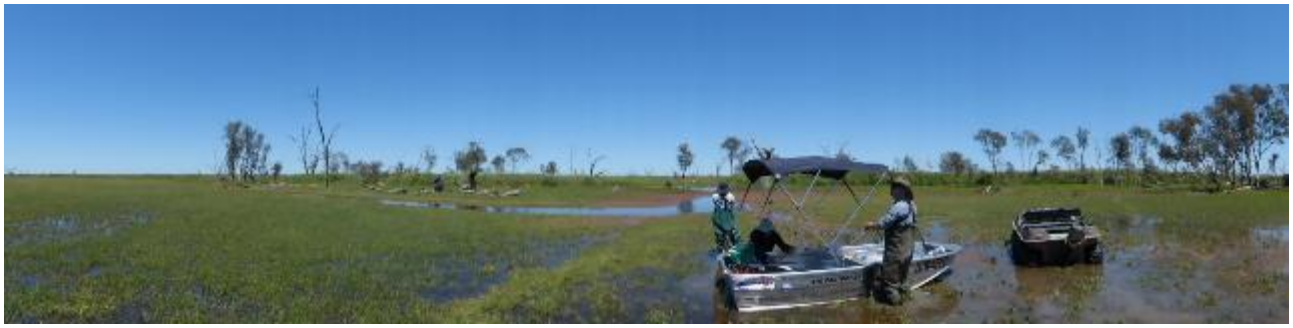
Date	Person(s)	Type	"Location"	Audience	Content / Activity
28/09 /2016	Heather McGinness	Conference	Australian Society for Limnology, Sydney	Academic	Waterbird recruitment: Responses to flooding, stressors and threats.
01/12 /2016	Heather McGinness	Factsheet (including web distribution)	MDFRC	General	MDB EWKR Waterbird Theme Factsheet: Background and Work Program, Wrote the text and supplied images.
01/02 /2017	Heather McGinness	Magazine	Wetlands Australia	General	McGinness, H. and Robinson, F. (2017) Monitoring waterbird recruitment to inform environmental flow management: A pilot study at Reed Beds wetland, Millewa Forest NSW. Wetlands Australia 29: 17-18.
14/11 /2016	Heather McGinness	Workshop	The Living Murray Icon Site Managers Forum, Moama	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
18/03 /2017	Heather McGinness	Webpage information	Australian River Restoration Centre	General	The importance of wetlands: Water => Plants => Animals.
01/02 /2017	Heather McGinness	Workshop	Macquarie Marshes Environmental Flows Reference Group, Dubbo	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
18/03 /2017	Heather McGinness	Workshop	Rivers of Carbon Community Conversation - Boosting Biodiversity in Breadalbane, Breadalbane	General	The importance of wetlands: Water => Plants => Animals.
01/09 /2017	Heather McGinness	Magazine	Riprap	General	McGinness, H., Brandis, K., Doerr, V., Kingsford, R., Mac Nally, R., Martin, J. (2017) Feathering their nests. Riprap Edition 40, 9-11
01/09 /2017	Heather McGinness	Magazine	Riprap	General	McGinness, H., Brandis, K., Doerr, V., Kingsford, R., Mac Nally, R., Martin, J. (2017) Waterbirds on the Wing. Riprap Edition 40, 6-8.
22/03 /2017	Heather McGinness	Workshop	NSW OEH Regional Operations Annual Environmental Water Management Workshop, Leeton	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
06/04 /2017	Heather McGinness	Seminar	Commonwealth Environmental Water Office, Canberra	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Responses to flooding, stressors and threats.

Date	Person(s)	Type	"Location"	Audience	Content / Activity
01/10 /2017		Facebook	na	General	Co-designed the content, co-wrote the text and supplied images.
01/10 /2017	Heather McGinness	News article	MDB EWKR Collaboration Space	General	Call for sightings of straw-necked ibis Australia-wide as part of Bird Week and in preparation for fieldtrips satellite-tagging ibis, Co-designed the content, co-wrote the text and supplied images.
18/10 /2017		Facebook	na	General	Please let Waterbirds Australia know of any straw-necked ibis you see! https://m.facebook.com/story.php?story_fbid=1567239326672105&id=1104133242982718 https://www.facebook.com/pg/TheMDFRC/posts/
14/08 /2017	Heather McGinness	Meeting	CSIRO Animal Ethics Committee, Canberra	General	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
20/09 /2017	Heather McGinness	Conference	River Symposium, Brisbane	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
25/09 /2017	Heather McGinness	Conference	Australian Society for Limnology, Sydney	Academic	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
30/03 /2018	Heather McGinness	Video	ABC Open, Facebook, Twitter, Australian River Restoration Centre blog	General	A short video explaining that Australia has three types of ibis https://open.abc.net.au/explore/254033
30/03 /2018	Sean Dooley	Magazine	Birdlife Australia magazine	General	Dooley, S. (2018) Ode to an Ibis. Australian Birdlife Magazine
30/03 /2018	Sean Dooley	Webpage information	Birdlife Australia magazine	General	Dooley, S. (2018) Ode to an Ibis. Australian Birdlife Magazine http://www.birdlife.org.au/australian-birdlife/detail/ode-to-an-ibis
15/04 /2018	Heather McGinness	News article	International Union for Conservation of Nature Species Survival Commission Stork, Ibis and Spoonbill Specialist Group	General	Australian project using remote nest cameras and GPS satellite tracking to study ibises and spoonbills http://storkibisspoonbill.org/news/australian-project-using-remote-nest-cameras-and-gps-satellite-tracking-to-study-ibises-and-spoonbills/
26/04 /2018	Heather McGinness	News article	ECOS Issue 243	General	The story of Australian waterbirds - the length and breadth of the Murray-Darling Basin. ECOS Issue 243 https://blogs.csiro.au/ecos/ibis/ https://blogs.csiro.au/ecos/ibis/
02/05 /2018	Heather McGinness	News article	Spatial Source	General	Waterbirds' stories revealed through satellite tracking (online magazine article) https://www.spatialsource.com.au/gps

Date	Person(s)	Type	"Location"	Audience	Content / Activity
					nav/waterbirds-stories-revealed-through-satellite-tracking
13/10 /2017	Heather McGinness	Workshop	North-Central CMA Brolga Round Table, Berrigan	General	The importance of wetlands: Water => Plants => Animals.
29/05 /2018		Facebook	na	General	Elvis continues his travels north! Share post, 87 Reach at 7/8/2018
31/05 /2018	Heather McGinness	Video	TBC	General	A short video describing the MDB EWKR Waterbird Theme research and preliminary results
31/05 /2018	Heather McGinness	Video	ABC Open	General	A 5-minute film/ video describing the MDB EWKR Waterbird Theme research and preliminary results https://open.abc.net.au/explore/276633
04/06 /2018		Facebook	na	General	Read about the latest story on Benji, a juvenile straw-necked ibis. Part of www.ewkr.com.au, 62 Reach at 7/8/2018
10/05 /2018	Heather McGinness	Workshop	Wetland Monitoring and Assessment Program for Environmental Flows - Waterbirds, Melbourne	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
26/06 /2018	*Heather McGinness*	*Workshop*	*Murray–Darling Basin Authority, Canberra*	*Stakeholders and/or Water Managers*	*MDBA Waterbird Expert Community of Practice Workshop*
06/06 /2018	Heather McGinness	Seminar	CEWO, Canberra	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
16/07 /2018	Heather McGinness	Newspaper	Numurkah Leader	General	Phone interview
23/07 /2018	Heather McGinness	Radio	ABC Radio	General	Recorded radio interview 19/7; aired 23.24/7
11/07 /2018	Heather McGinness	Seminar	Canberra Ornithologists Group	General	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
26/09 /2018	Heather McGinness	Conference	Australian Freshwater Sciences Society, Adelaide	Academic	Waterbird recruitment and movement: Responses to flooding, stressors and threats.
15/10 /2018	Heather McGinness	Workshop	Victorian Environmental Water Holder Prioritisation Project Workshop	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Informing prioritisation of environmental flows
16/10 /2018	Heather McGinness	Conference	River Symposium, Sydney	Stakeholders and/or Water Managers	Waterbird recruitment and movement: Responses to flooding, stressors and threats.

Date	Person(s)	Type	"Location"	Audience	Content / Activity
	and Nikki Thurgate				
04/12/2018	*Heather McGinness*	*Workshop*	*Murray–Darling Basin Authority, Canberra*	*Stakeholders and/or Water Managers*	*MDBA Waterbird Expert Community of Practice Workshop*
activity dependent	*Heather McGinness*	*Meeting*	*Fieldwork meetings with stakeholders and managers, Barmah-Millewa Forest, Macquarie Marshes, other sites*	*Stakeholders and/or Water Managers*	*Meetings with stakeholders and managers before, during and after fieldwork*
activity dependent	*Heather McGinness*	*News article*	*MDB EWKR Collaboration Space*	*General*	*Waterbird Theme information, Wrote the text and supplied images.*
activity dependent	*Heather McGinness*	*Webpage information*	*EWKR Story Space*	*General*	*Waterbird Theme information; Riprap articles as listed, Wrote the text and supplied images. http://ewkr.com.au/waterbirds-feathering-their-nests/; http://ewkr.com.au/waterbirds-on-the-wing/*
weekly updates	*Heather McGinness and Freya Robinson*	*Facebook*		*General*	*Waterbirds Australia: Science for Management Co-designed the content, co-wrote the text and supplied images.*
weekly updates	*Heather McGinness and Freya Robinson*	*Twitter*	*@AusWaterbirds*	*General*	*Waterbirds Australia: Science for Management Co-designed the content, co-wrote the text and supplied images. https://www.facebook.com/ColonialWaterbirdScience/*
weekly updates	*Heather McGinness*	*eMail*	*Broad distribution (100+ recipients)*	*Stakeholders and/or Water Managers*	*Update Emails: EWKR Waterbird Theme satellite tracking, Wrote the text and supplied images.*
weekly updates	*Heather McGinness*	*Webpage information*	*CSIRO EWKR Waterbirds*	*General*	*Waterbird breeding and movements: Knowledge for water managers, Co-designed the content, co-wrote the text and supplied images. https://research.csiro.au/ewkrwaterbirds/*
8/08/2018		Facebook	na	General	*CALLING ALL CITIZEN SCIENTISTS NEAR BARALABA, QLD! Help find ELF!*
16/08/2018		Facebook	na	General	This National Science Week get to know our project (and our waterbirds) by checking out the website! https://research.csiro.au/ewkrwaterbirds/ - Shared page
22/08/2018		Twitter	na	General	CALLING ALL CITIZEN SCIENTISTS NEAR NATHALIA, VICTORIA!

Date	Person(s)	Type	"Location"	Audience	Content / Activity
27/08/2018		Facebook	na	General	How do birds use their habitats? Read more on Waterbirds Gaia, Elvis and Elf who are sporting lightweight GPS mini-backpacks for the sake of science ! EWKR.COM.AU
29/08/2018		Twitter	na	General	Can you help find SUNNY ! Calling all citizen scientists
13/09/2018		Facebook	na	General	CALLING ALL CITIZEN SCIENTISTS NEAR KATANDRA WEST, VIC! Help needed to find SUNNY!
18/09/2018		Twitter	na	General	CALLING ALL CITIZEN SCIENTISTS NEAR AVENEL, VIC! We challenge you to find KEITH! ⁉️
27/11/2018	Heather McGinness	Conference	Ecological Society of Australia, Brisbane	Academic	Waterbird recruitment and movement: Responses to flooding, stressors and threats
19/03/2019	Heather McGinness	Workshop	MDB EWKR Forum	Stakeholders and Academic	Presentation: Waterbird recruitment and movement: Responses to flooding, stressors and threats
6/4/2019	Heather McGinness	Conference	Birdlife Australia BIGnet Conference, Southern Highlands	General	Presentation: Water, wetlands and waterbirds in the Murray-Darling Basin
2/07/2019	Heather McGinness	Conference	Australasian Ornithological Conference	General	Presentation: Ibis and spoonbill recruitment and movements



Waterbirds Australia

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The field guide to chick development produced by Lucy Wenger and Heather McGinness (CSIRO) ***Available for preview and purchase (at cost) here: http://au.blurb.com/bookstore/invited/7714996/2775e42c847608953b28c7020e788bd36b9f0770***

HOW TO DISTINGUISH BETWEEN MALE AND FEMALE STRAW-NECKED IBIS....

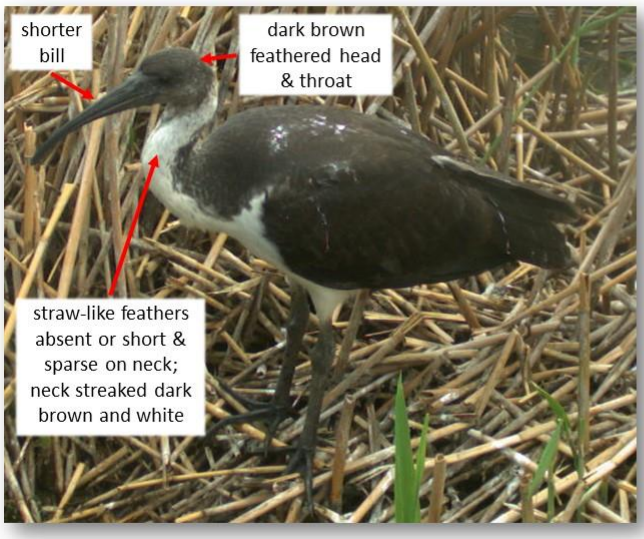


Female
Solid black band across chest

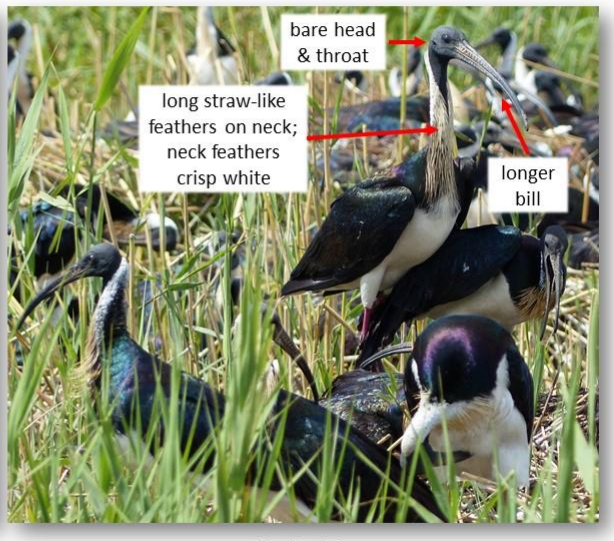
Male
Broken black band across chest



HOW TO DISTINGUISH BETWEEN JUVENILE AND ADULT STRAW-NECKED IBIS....



Juvenile



Adult

Pictorial guides for straw-necked ibis produced by Freya Robinson and Heather McGinness (CSIRO) and distributed through social media, email and websites.

PROFILE #1:

Australian White Ibis





Species name: *Threskiornis molucca*

Lives: Widespread throughout south-east, south-west and northern Australia. Resident in southern New Guinea and New Zealand.

Breeds: Largest breeding colonies in inland south-east Australia and east Australian coast.

Feeds: Often seen wading in wetlands and marine habitats with bill immersed probing for fish, frogs and aquatic invertebrates, or in flocks on grasslands and ovals looking for insects.

Did you know...

Young ibis like to hang out together?

Once chicks are a few weeks old they begin to group together with others from neighbouring nests, forming 'crèches'. It's not uncommon to have crèches with both Australian white ibis and straw-necked ibis chicks.

Species profile produced by Freya Robinson and Heather McGinness (CSIRO) and distributed through social media, email and websites.

PROFILE #2:

Straw-necked Ibis



Species name: *Threskiornis spinicollis*

Lives: Originally endemic to Australia however now occurs in southern New Guinea. Widespread throughout Australia, preferring drier inland to coastal habitats.

Breeds: Largest breeding colonies in inland south-east Australia.



Feeds: Feed more on inland grasslands and pasture than Australian white ibis, favouring locusts and other insects.

Did you know...

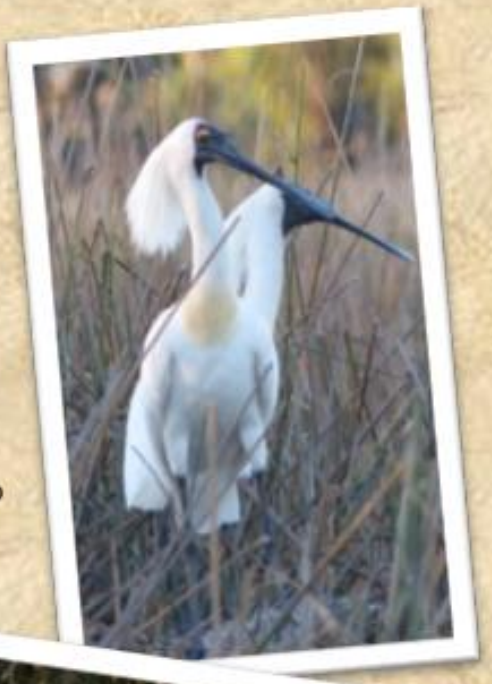
Straw-necked ibis parents are dedicated?

Both parents take it in turns to incubate the eggs and tend to young at the nest, continuing to feed their young until they can fly.

Species profile produced by Freya Robinson and Heather McGinness (CSIRO) and distributed through social media, email and websites.

PROFILE #3:

Royal Spoonbill



Species name: *Platalea regia*

Lives: Originally endemic to Australia with populations now in NZ. Occurs throughout eastern and northern Australia.

Breeds: Largest breeding colonies thought to be in inland and coastal south-east Australia.

Feeds: Feed in open shallow water of inland wetlands, sheltered marine habitats and submerged grasslands by wading and sweeping spatula-shaped bill through water to locate fish and aquatic invertebrates.



Did you know....

Spoonbills like to impress their partner?

Spoonbills use all kinds of display techniques to court potential mates and reaffirm partnerships throughout the breeding season. This includes head shaking and presenting their partner with sticks or nesting material.

Species profile produced by Freya Robinson and Heather McGinness (CSIRO) and distributed through social media, email and websites.

For further information

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