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.
# Contents

<table>
<thead>
<tr>
<th>Section</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>Foreword</td>
<td>i</td>
</tr>
<tr>
<td>Acknowledgments</td>
<td>iii</td>
</tr>
<tr>
<td>Introduction</td>
<td>iv</td>
</tr>
<tr>
<td>Key messages</td>
<td>vi</td>
</tr>
<tr>
<td>Executive summary</td>
<td>vii</td>
</tr>
<tr>
<td><strong>Part I  Waterbird responses to flows and flooding in Australia</strong></td>
<td>1</td>
</tr>
<tr>
<td>1 Conceptual model: Drivers of waterbird species persistence and diversity</td>
<td>2</td>
</tr>
<tr>
<td>2 Summary of known relationships</td>
<td>1</td>
</tr>
<tr>
<td>3 Knowledge gaps</td>
<td>19</td>
</tr>
<tr>
<td><strong>Part II  Waterbird responses to stressors and threats</strong></td>
<td>25</td>
</tr>
<tr>
<td>4 Habitat loss, fragmentation, and change</td>
<td>26</td>
</tr>
<tr>
<td>5 Predation</td>
<td>32</td>
</tr>
<tr>
<td>6 Climate change</td>
<td>40</td>
</tr>
<tr>
<td>7 Pollution</td>
<td>46</td>
</tr>
<tr>
<td>8 Disease and parasites</td>
<td>49</td>
</tr>
<tr>
<td>9 Human disturbance</td>
<td>52</td>
</tr>
<tr>
<td>10 Competition</td>
<td>56</td>
</tr>
<tr>
<td>11 Interactions and links between flooding, stressors, threats and waterbird responses</td>
<td>58</td>
</tr>
<tr>
<td><strong>Appendix A  Part I Summary table of waterbird responses to flows and flooding in Australia</strong></td>
<td>64</td>
</tr>
<tr>
<td><strong>Appendix B  Part II Web of Science Advanced search string example</strong></td>
<td>65</td>
</tr>
<tr>
<td>Reference list Part I</td>
<td>66</td>
</tr>
<tr>
<td>Reference list Part II</td>
<td>71</td>
</tr>
</tbody>
</table>
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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 sufficient 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 Pelecaniformes (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 disadvantaged 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,
Affecting prey impacts and changes in predation, particularly waterbird habitats, predation, and breeding success have been documented. 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 waterbird predation 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 & Beversbergen 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 moult.

• 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 Schikert 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; Stirmeman 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.
Response variables
Diversity and richness
Species persistence

Driving and interacting variables
SOI, Rainfall, Evaporation, Temperature
River discharge/flow
Flood timing
Flood duration
Flood depth
Flood (wetland) extent/size/area
Flood frequency (+ antecedence, clustering)
Interflood period (dry period duration)
Rate of change in flood depth
Flood location

The above affect other drivers including:
Food abundance
Breeding habitat availability
Foraging habitat availability
Refuge/shelter/roosting habitat availability
Connectivity between habitats
Competition from native species
Competition from invasive/pest species
Predation from native species
Predation from invasive/pest species
Nutrient, toxin, salinity and sediment levels
Disturbance
Disease
Catchment condition

Humans change drivers and interactions through:
Dams and weirs
Levees and other flow obstructions
Channels
Pumping
Draining
Vegetation clearing
Grazing
Cropping
Irrigation
Hunting
Pollution (nutrients, toxins, sediments)
Tourism
Residential development
Introduction of pest/invasive species
Transmission of disease
Salinity changes
Climate change

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.

<table>
<thead>
<tr>
<th>Known relationships</th>
<th>References</th>
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<tbody>
<tr>
<td>Waterbird abundance at individual sites or wetland complexes is highly variable and often unpredictable both temporarily 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.</td>
<td>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</td>
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Species/group abundance changes in response to wetland availability/area/flood extent at continental, basin, catchment and local scales

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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.

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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.

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<tr>
<td>Kingsford et al. 2013</td>
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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.

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<tr>
<td>Kingsford et al. 2013</td>
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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).

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<td>Wen et al. 2011; Kingsford and Thomas 2004</td>
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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.

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<td>Roshier et al. 2002</td>
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<td>Known relationships</td>
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<tr>
<td>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.</td>
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<tr>
<td>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.</td>
</tr>
<tr>
<td>Species/group abundance changes in response to rainfall at local, catchment, and basin scales.</td>
</tr>
<tr>
<td>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.</td>
</tr>
<tr>
<td>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.</td>
</tr>
<tr>
<td>Species/group abundance at a site may be associated with temperature (season)</td>
</tr>
<tr>
<td>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.</td>
</tr>
<tr>
<td>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.</td>
</tr>
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</table>
### Known relationships

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<tr>
<th>Key individual wetlands are consistently important and should be prioritised for management of long-term waterbird abundance.</th>
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| These wetlands include the following 17 wetlands: Booligal wetlands, Currawinya Lakes, Cuttaburra channels, Gwydir wetlands, Kerang wetlands, Lake Buloke, Lake Cowal, Lake Brewster, Lindsay-Walpolia-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 |

<table>
<thead>
<tr>
<th>Species/group abundance is greater following clusters of floods over two to four years</th>
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<tr>
<td>Kingsford et al. 2013</td>
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| 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 and Norman 2002 citing Braithwaite 1981a, Fullagar et al. 1988; Cowan 1973 |

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<tr>
<th>Extended wet periods may allow waterbirds to produce more than one or two clutches successively, resulting in rapid increases in abundance.</th>
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<tr>
<th>Different functional groups and species differ significantly in abundance and can heavily influence overall abundance assessments (and community/media impressions of abundance)</th>
</tr>
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</table>

<p>| 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 |</p>
<table>
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<th>Known relationships</th>
<th>References</th>
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<tr>
<td>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%).</td>
<td>Kingsford et al. 2013</td>
</tr>
<tr>
<td>Species/group abundance may be partly determined by bird experience/memory/site attachment</td>
<td></td>
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<tr>
<td>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)</td>
<td>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?</td>
</tr>
<tr>
<td>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%.</td>
<td>(p.9 Kingsford et al. 2013).</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford et al. 2013</td>
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</table>
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.

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<tr>
<td>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.</td>
<td>Leslie 2001; Arthur 2011; Kingsford et al. 2013; Roshier et al. 2001; Scott 1997</td>
</tr>
<tr>
<td>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%.</td>
<td>Leslie 2001;</td>
</tr>
<tr>
<td>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.</td>
<td>Arthur 2011</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford et al. 1999</td>
</tr>
<tr>
<td>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).</td>
<td>Scott 1997; Braithwaite 1975; Maher 1991</td>
</tr>
<tr>
<td>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 goose and banded stilt), while others appear to have declined considerably (e.g. grey teal).</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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)</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>In 2008, banded stilt population size was estimated at 200,000, compared with past estimates of around 206,000 (in 2006)</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>In 2008, grey teal population size was estimated at 320,000, compared with past estimates of over 1 million</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford et al. 2013</td>
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2.3 Population boundaries / movements

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

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There is considerable variability in spatial and temporal movement patterns within and between species

| 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'. | Rosher 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. Moultng 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; Tulg 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

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<td>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.</td>
<td>al. 1998; Dorfman and Kingsford 2001; Roshier et al. 2001; Kingsford and Norman 2002</td>
</tr>
<tr>
<td>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).</td>
<td>Roshier et al. 2001</td>
</tr>
<tr>
<td>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).</td>
<td>Kingsford and Norman 2002 citing Carrick 1962; McKilligan 1975; Geering et al. 1998; Marchant and Higgins 1990</td>
</tr>
<tr>
<td>Movement patterns may vary with age in ibis, with young ibis moving further than older birds</td>
<td>McKilligan 1975</td>
</tr>
<tr>
<td>Cormorants bred in the lower Murray tend to disperse eastwards seeking permanent water; however there is variation in the direction of dispersal</td>
<td>Llewellyn 1983</td>
</tr>
<tr>
<td>Waterbird movements in the Murray-Darling Basin are influenced by flooding patterns in the Lake Eyre Basin and other inland areas</td>
<td>Roshier et al. 2002; Kingsford et al. 2010; Kingsford et al. 2013</td>
</tr>
<tr>
<td>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.</td>
<td>Roshier et al. 2002</td>
</tr>
<tr>
<td>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</td>
<td>Kingsford and Norman 2002; Woodall 1985; Drafan et al. 1983; Matheson 1978; Briggs 1990</td>
</tr>
<tr>
<td>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)</td>
<td>Kingsford and Norman 2002</td>
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</table>
### 2.4 Breeding initiation

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

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<td><strong>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.</strong></td>
<td>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</td>
</tr>
</tbody>
</table>

| 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. | Rosnier 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 Calbonna in 1931 resulted from local rainfall. | Kingsford et al. 1999 citing McClimp 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 |

<p>| Flood area/extent is a key driver of breeding initiation for many waterbirds, with positive | Reid et al. 2009; Brandis |</p>
<table>
<thead>
<tr>
<th>Known relationships</th>
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<tbody>
<tr>
<td>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.</td>
<td>2010; Frith 1959</td>
</tr>
<tr>
<td>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.</td>
<td></td>
</tr>
<tr>
<td>At a regional scale, the number of breeding waterbird species is strongly ‘explained’ by wetland area but less so by the number of wetlands</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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.</td>
<td>Arthur et al. 2012; Briggs and Thornton 1999; Scott 1997; Reid et al. 2009;</td>
</tr>
<tr>
<td>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.</td>
<td>Arthur et al. 2012</td>
</tr>
<tr>
<td>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.</td>
<td>Arthur et al. 2012</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford and Johnson 1998; Kingsford and Auld 2005</td>
</tr>
<tr>
<td>Ibis nesting at Lake Merreti is associated with a maximum flow threshold in September of ~25,000 ML/day</td>
<td>Arthur et al. 2012</td>
</tr>
<tr>
<td>Nankeen Night Heron breeding initiation is closely related to maximum daily flow in October in Barmah-Millewa Forest and the Macquarie Marshes</td>
<td>Arthur et al. 2012</td>
</tr>
<tr>
<td>Early modelling for the Barmah Forest found that the Algebaia 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)</td>
<td>Leslie 2001</td>
</tr>
<tr>
<td>At Narran Lakes, modelling found recorded ibis breeding events were associated with thresholds for three flow measures: total flow event volumes &gt;100,012 ML, mean daily flows &gt;1,552 ML, and flow durations &gt;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.</td>
<td>Brandis et al. 2010</td>
</tr>
<tr>
<td>Australian white ibis and straw-necked ibis breed 1 month and 2 months, respectively, after flooding begins</td>
<td>Carrick 1962</td>
</tr>
<tr>
<td>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 for flooding under nest trees to first broods of ducks observed were 1-2 months, regardless of season.</td>
<td>Briggs and Thornton 1999</td>
</tr>
<tr>
<td>In the Paroo system, waterbird breeding (egg laying) may not commence until 8 weeks following the beginning of flooding, or two weeks after 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.</td>
<td>Maher and Braithwaite 1992</td>
</tr>
<tr>
<td>For some species, threshold depths of inundation exist for breeding initiation</td>
<td>Reid et al. 2009</td>
</tr>
<tr>
<td>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.</td>
<td>Briggs and Thornton 1999; Briggs et al. 1997; Leslie 2001; Carrick 1962; Maher 1991; Maher and Braithwaite 1992</td>
</tr>
<tr>
<td>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</td>
<td>Brandis et al. 2011 citing Bren 1988; Brock and others 2006; Capon 2005; Briggs et al. 1997; Kingsford and</td>
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<td>Known relationships</td>
<td>References</td>
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<td>essential for breeding waterbirds and floodplain vegetation is significantly affected by changes in flood frequency, duration, depth, and interflood periods.</td>
<td>Norman 2002 citing Marchant and Higgins 1990, 1993, 1996;</td>
</tr>
<tr>
<td>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.</td>
<td>Briggs and Thornton 1999; Briggs et al. 1997; Leslie 2001; Carrick 1962; Maher 1991; Maher and Braithwaite 1992</td>
</tr>
<tr>
<td>Australian wood ducks are obligate tree hollow nesters, breeding on or near wetlands with trees of appropriate age to provide such hollows.</td>
<td>Kingsford and Norman 2002 citing Frith 1982; Kingsford 1992</td>
</tr>
<tr>
<td>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.</td>
<td>Brandis 2010</td>
</tr>
<tr>
<td>Dense wetland vegetation is essential for nesting of many herons, egrets, ibises and bitterns.</td>
<td>Kingsford and Norman 2002 citing Frith 1982; Marchant and Higgins 1990</td>
</tr>
<tr>
<td>Islands in lakes are preferred breeding habitat for banded stilt, Australian pelican, silver gull, caspian tern and gull-billed tern.</td>
<td>Kingsford and Norman 2002 citing Burbridge and Fuller 1982; Waterman and Read 1992; Kingsford and Porter 1993, 1994; Kingsford et al. 1999a</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford and Norman 2002 citing Braithwaite 1976a, 1982; Kingsford 1989b</td>
</tr>
<tr>
<td>Breeding initiation at a site can be partly determined by bird experience/memory/site attachment</td>
<td></td>
</tr>
<tr>
<td>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.</td>
<td>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</td>
</tr>
<tr>
<td>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)</td>
<td>Kingsford et al. 2012; Kingsford et al. 2013; Leslie 2001;</td>
</tr>
<tr>
<td>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 sufficient size or the majority of waterbird breeding events.</td>
<td>Brandis 2010</td>
</tr>
<tr>
<td>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%</td>
<td>Kingsford and Thomas 1995; Kingsford and Johnson 1998</td>
</tr>
<tr>
<td>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 &gt;10 breeding events - most (64%) wetlands had only one event recorded. Ibis breeding accounted for 36% of all records.</td>
<td>Brandis 2010</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford and Norman 2002</td>
</tr>
<tr>
<td>Conversely, prolonged inundation may kill or significantly alter nesting habitat (such as live</td>
<td>Kingsford and Norman 2002</td>
</tr>
<tr>
<td>Known relationships</td>
<td>References</td>
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<tr>
<td>trees) to the point that breeding is reduced or no longer occurs at a site</td>
<td>citing Briggs et al. 1994</td>
</tr>
<tr>
<td>Mean annual temperature was identified as an influencing variable for probability of</td>
<td></td>
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<tr>
<td>breeding of Australian Pelicans and Great Cormorants in modelling conducted by Brandis (2010), however its influence was outweighed significantly by wetland area</td>
<td>Brandis 2010</td>
</tr>
</tbody>
</table>
### 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.

<table>
<thead>
<tr>
<th>Known relationships</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>The number of breeding pairs/nests increases with increasing flood extent and duration</td>
<td>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;</td>
</tr>
<tr>
<td>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</td>
<td>Kingsford et al. 2012; Briggs et al. 1997</td>
</tr>
<tr>
<td>There is a positive relationship between flow in the Macquarie River (positively related to flood area/extents 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.</td>
<td>Kingsford and Johnson 1998; Kingsford and Auld 2005</td>
</tr>
<tr>
<td>The number of breeding pairs/nests increases following extended wet periods and/or clusters of floods over two to four years</td>
<td>Kingsford et al. 2013</td>
</tr>
<tr>
<td>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</td>
<td></td>
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<tr>
<td>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 successive 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.’</td>
<td>p.123 Reid et al. 2009 citing Puckridge et al. 2000</td>
</tr>
<tr>
<td>The number of breeding pairs/nests is typically greater in intermittently flooded wetlands that experience productivity booms than in permanently inundated wetlands</td>
<td>Kingsford et al. 2004; Briggs et al. 1985; Maher and Carpenter 1984; Maher 1984; Crome 1986, 1988; Kingsford and Thomas 2004;</td>
</tr>
<tr>
<td>Ducks breed most prolifically at sites that have reflooded after a dry period</td>
<td>Briggs and Thornton 1999; Maher and Carpenter 1984; Crome 1986; Briggs et al. 1997</td>
</tr>
<tr>
<td>Key locations are consistently important and should be prioritised for management of waterbird breeding</td>
<td>Kingsford et al. 2013; Brandis et al. 2009</td>
</tr>
<tr>
<td>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, Lowidgee 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.</td>
<td>Brandis et al. 2009</td>
</tr>
<tr>
<td>Large recorded breeding events (&gt;10,000 individuals) of colonial nesting waterbirds occurred on only 24 individual wetlands during the period 1899-2008</td>
<td>Brandis 2010</td>
</tr>
<tr>
<td>The impact of river regulation on waterbird breeding - at least colonial-nesting waterbirds -</td>
<td>Kingsford et al. 2013;</td>
</tr>
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<td>Known relationships</td>
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<tr>
<td>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</td>
<td>Brandis et al. 2009; Kingsford and Johnson 1990</td>
</tr>
<tr>
<td>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 significantly in the forest.</td>
<td>Leslie 2001</td>
</tr>
<tr>
<td>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.</td>
<td>Briggs et al. 1997</td>
</tr>
<tr>
<td>In coastal NSW, breeding populations of Intermediate Egrets declined by 98% between 1988 and 1998</td>
<td>Kingsford and Norman 2002 citing Maddock 2000</td>
</tr>
<tr>
<td>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</td>
<td>Kingsford and Johnson 1998</td>
</tr>
</tbody>
</table>
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.

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<tr>
<td><strong>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.</strong></td>
<td>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</td>
</tr>
<tr>
<td>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).)</td>
<td>Brandis et al. 2011</td>
</tr>
<tr>
<td>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.</td>
<td>Brandis 2010</td>
</tr>
<tr>
<td>Australian pelicans abandoned their colony on Lake Eyre in 1990 after eggs were laid because the flood was no longer sufficient</td>
<td>Kingsford et al. 1999 citing Kingsford and Porter 1993</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford and Auld 2005</td>
</tr>
<tr>
<td>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</td>
<td>Leslie 2001</td>
</tr>
<tr>
<td>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 Alboina Plain nest site because of rapid water level recession</td>
<td>Leslie 2001</td>
</tr>
<tr>
<td>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</td>
<td>Leslie 2001</td>
</tr>
<tr>
<td><strong>Temperature (and therefore season and flood timing) can also affect the success of nests.</strong></td>
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<tr>
<td>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.</td>
<td>Scott 1997 citing Briggs 1990, 1994, Maher 1991, Johnson pers.comm</td>
</tr>
<tr>
<td>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.</td>
<td>Brandis 2010 citing McCosker 1996; Taft et al. 2000; Scott 1997 citing B. Johnson pers. Comm.</td>
</tr>
</tbody>
</table>
### 2.7 Fledging rate

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

<table>
<thead>
<tr>
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<tr>
<td>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. &gt;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.</td>
<td>Reid et al. 2009; Brandis et al. 2011; Briggs and Thornton 1999; Leslie 2001; Marchant and Higgins 1990;</td>
</tr>
<tr>
<td>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.</td>
<td>Briggs and Thornton 1999</td>
</tr>
<tr>
<td>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 <strong>minimum</strong> 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).</td>
<td>Briggs et al. 1994; Briggs and Thornton 1999</td>
</tr>
<tr>
<td>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.</td>
<td>Leslie 2001 citing Butler 1994</td>
</tr>
<tr>
<td>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.</td>
<td>Brandis et al. 2011</td>
</tr>
<tr>
<td>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.</td>
<td>Baxter 1994; Arthur 2011 citing Maddock and Baxter 1991;</td>
</tr>
</tbody>
</table>
2.8 Juvenile survival

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

<table>
<thead>
<tr>
<th>Known relationships</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>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.</strong></td>
<td>Leslie 2001 citing Butler 1994; Marchant and Higgins 1990; Briggs and Thornton 1999</td>
</tr>
<tr>
<td>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.</td>
<td>Arthur 2011 and citations therein</td>
</tr>
</tbody>
</table>

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.

<table>
<thead>
<tr>
<th>Known relationships</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>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.</strong></td>
<td>Kingsford et al. 1999; Kingsford and Norman 2002</td>
</tr>
<tr>
<td>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.</td>
<td>Arthur 2011</td>
</tr>
<tr>
<td>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.</td>
<td>Scott 1997 citing Maher 1991</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford and Norman 2002 citing unpublished data and Barnard 1927; Kingsford et al. 1999</td>
</tr>
<tr>
<td>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 occurred on Lake Salisbury some 370 km to the east of Lake Eyre.</td>
<td>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</td>
</tr>
<tr>
<td><strong>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.</strong></td>
<td>Reid et al. 2009; Leslie 2001;</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford and Norman 2002</td>
</tr>
<tr>
<td>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 ABBBS scheme on pages 151-152 of her thesis: Herons and egrets: 7-14 years; Cormorants: 10-26 years; Pelicans: 15-20 years; Ibises and spoonbills: 7-12 years</td>
<td>Brandis 2010 citing ABBBS 2009, Marchant and Higgins 1990; McKilligan 2005</td>
</tr>
</tbody>
</table>
### 2.10 Diversity and richness

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

<table>
<thead>
<tr>
<th>Known relationships</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>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’).</strong></td>
<td>(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;</td>
</tr>
<tr>
<td>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.</td>
<td>Reid et al. 2013; Reid et al. 2009</td>
</tr>
<tr>
<td>Different locations support different species/guilds/communities at different times</td>
<td>Kingsford and Norman 2002</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford et al. 2013; Kingsford et al. 2012; Kingsford et al. 2012 citing Cale et al. 2004; Kingsford and Porter 1994</td>
</tr>
<tr>
<td>Species/group diversity is often greater in intermittently flooded or ephemeral wetlands that experience productivity booms than in static, permanently inundated wetlands</td>
<td>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</td>
</tr>
<tr>
<td>Species richness is significantly correlated with abundance, with high species numbers generally occurring on wetlands supporting high numbers of waterbirds</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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.</td>
<td>Reid and Jaensch 2004; Reid et al. 2009</td>
</tr>
<tr>
<td>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.</td>
<td>Brandis et al. 2009 citing Kingsford et al. 2008</td>
</tr>
<tr>
<td><strong>A diversity of breeding species requires a range of nesting habitats</strong></td>
<td>(see Reid et al. 2009 for a list and example species); Leslie 2001; Kingsford and Norman 2002</td>
</tr>
<tr>
<td>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.</td>
<td>Brandis 2010 (thesis); Leslie 2001; Briggs et al. 1997</td>
</tr>
<tr>
<td>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.</td>
<td>Briggs et al. 1997</td>
</tr>
<tr>
<td><strong>A diversity of feeding species requires a range of foraging habitats and foods</strong></td>
<td>Frith 1959; Reid et al. 2009; Kingsford and Norman 2002</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford and Norman 2002 citing Kingsford and Porter 1994; McDougall and Timms 2001</td>
</tr>
<tr>
<td>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</td>
<td>Kingsford et al. 2013; Frith 1959</td>
</tr>
</tbody>
</table>
## Known relationships

<table>
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<tr>
<th>Description</th>
<th>References</th>
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<tbody>
<tr>
<td>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</td>
<td>Frith 1959</td>
</tr>
<tr>
<td>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.</td>
<td>Kingsford et al. 2013</td>
</tr>
<tr>
<td>Species diversity/richness at a site can be partly determined by bird experience/memory/site attachment</td>
<td>Reid et al. 2009; Leslie 2001</td>
</tr>
<tr>
<td>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</td>
<td>Kingsford et al. 2012; Kingsford and Thomas 1995; 2004; Leslie 2001; Kingsford et al. 2004; Kingsford and Auld 2005; Kingsford and Porter 2009</td>
</tr>
<tr>
<td>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</td>
<td>Kingsford et al. 2013</td>
</tr>
<tr>
<td>In Barmah Forest, diversity of 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 noticeable during the 1950's.</td>
<td>Leslie 2001</td>
</tr>
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</table>
# Knowledge gaps

## 3.1 Abundance

<table>
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<tr>
<th>Knowledge gaps</th>
<th>References</th>
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<tbody>
<tr>
<td>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:</td>
<td></td>
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<tr>
<td>Have abundances of the more cryptic species changed over time and/or space? What are the drivers of abundance for these species?</td>
<td></td>
</tr>
<tr>
<td>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 rates the best spatial scale of data aggregation at which to analyse trends of whole or substantial proportions of populations?</td>
<td></td>
</tr>
<tr>
<td>Has there been a greater reduction in waterbird abundance than percent reductions in flows in regulated rivers?</td>
<td></td>
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<tr>
<td>Are there critical benchmarks or thresholds in abundance for particular species or guilds?</td>
<td></td>
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<tr>
<td>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?</td>
<td></td>
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<tr>
<td>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?</td>
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</table>

## 3.2 Population size and age structure

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<th>Knowledge gaps</th>
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<tbody>
<tr>
<td>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.</td>
<td></td>
</tr>
<tr>
<td>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.</td>
<td></td>
</tr>
<tr>
<td>The impacts of loss of habitat on population sizes and age structures of both resident and migratory waterbird species is poorly understood.</td>
<td></td>
</tr>
</tbody>
</table>


Reid et al. 2009; Maher and Braithwaite 1992; Roshier et al. 2002

Kingsford et al. 2013

Reid et al. 2009

Kingsford and Norman 2002

Kingsford et al. 2012; Kingsford and Norman 2002


Brandis et al. 2009
3.3 Population boundaries / movements

<table>
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<tr>
<th>Knowledge gaps</th>
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</tr>
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<tbody>
<tr>
<td>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:</td>
<td>Roshier et al. 2002; Reid et al. 2009; Kingsford et al. 2013; Arthur 2011; Kingsford and Norman 2002</td>
</tr>
<tr>
<td>Over what spatial scales do waterbird species populations actually function? Although sufficient 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.</td>
<td>Arthur 2011; McKilligan 2005; Reid et al. 2009</td>
</tr>
<tr>
<td>Exactly what cues are being used/detected to trigger long-distance movements? Are atmospheric cues required in addition to environmental flows?</td>
<td>Kingsford and Norman 2002</td>
</tr>
<tr>
<td>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?</td>
<td>Brandis et al. 2009</td>
</tr>
<tr>
<td>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?</td>
<td>Reid et al. 2009; Kingsford and Norman 2002</td>
</tr>
<tr>
<td>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.</td>
<td>Brandis et al. 2009</td>
</tr>
<tr>
<td>Are long-distance movements exceptional, or repeated frequently in a long-lived bird’s life?</td>
<td>Reid et al. 2009</td>
</tr>
<tr>
<td>Are tropical drainage divisions used as refuges by waterbird species from southern temperate drainage divisions during extended drought?</td>
<td>Brandis et al. 2009</td>
</tr>
<tr>
<td>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.</td>
<td>Brandis et al. 2009</td>
</tr>
<tr>
<td>What is the impact of changes in waterbird movements on plant and invertebrate dispersal?</td>
<td></td>
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</table>

3.4 Breeding initiation

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<th>Knowledge gaps</th>
<th>References</th>
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</thead>
<tbody>
<tr>
<td>Are patterns of breeding initiation at nesting sites changing over time? If so, why?</td>
<td>Brandis 2010</td>
</tr>
<tr>
<td>Do undocumented nesting areas exist in Australia?</td>
<td>Brandis 2010</td>
</tr>
<tr>
<td>What are the precise effects of seasonal changes in flood timing on waterbird breeding initiation?</td>
<td>Scott 1997</td>
</tr>
<tr>
<td>Has there been a greater reduction in breeding effort and success than in waterbird numbers per se? If so, for which species, and why?</td>
<td>Leslie 2001; Reid et al. 2009</td>
</tr>
<tr>
<td>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?</td>
<td></td>
</tr>
<tr>
<td>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?</td>
<td></td>
</tr>
<tr>
<td>For which other species do threshold durations of inundation exist for breeding initiation, and what are those thresholds?</td>
<td></td>
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<tr>
<td>For which species do threshold areas of inundation exist for breeding initiation, and what are those thresholds?</td>
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</table>
3.5 Number of breeding pairs / nests

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

3.6 Proportion of nests successful

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

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

<table>
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<tr>
<th>Knowledge gaps</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>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:</td>
<td>Roshier et al. 2002; Marchant and Higgins 1990, 1993; Briggs 1992; McKilligan 2005; Arthur 2011</td>
</tr>
<tr>
<td>What are the main drivers of juvenile survival/mortality, and do these vary spatially or temporally?</td>
<td>Arthur 2011</td>
</tr>
<tr>
<td>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?</td>
<td>Scott 1997</td>
</tr>
<tr>
<td>How important are small to medium sized floods for juvenile survival between the large breeding events triggered by large floods?</td>
<td></td>
</tr>
<tr>
<td>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?</td>
<td></td>
</tr>
<tr>
<td>Can co-ordinated broad scale surveys that include marking and tracking of juvenile cohorts provide sufficient data to estimate survival?</td>
<td></td>
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</tbody>
</table>

3.9 Adult survival and longevity

<table>
<thead>
<tr>
<th>Knowledge gaps</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>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.</td>
<td>Reid et al. 2009; Arthur 2011; also citing Cezilly 1997; Brandis 2010</td>
</tr>
<tr>
<td>No estimates of adult, subadult, or juvenile survival are available for egrets in Australia.</td>
<td>Arthur 2011 citing McKilligan 2005</td>
</tr>
<tr>
<td>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).</td>
<td>Reid et al. 2009</td>
</tr>
</tbody>
</table>

3.10 Diversity and richness

<table>
<thead>
<tr>
<th>Knowledge gaps</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>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)</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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?</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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?</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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?</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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?</td>
<td>Kingsford et al. 2012</td>
</tr>
<tr>
<td>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.</td>
<td>Scott 1997 citing Kingsford et al. 1989</td>
</tr>
<tr>
<td>Knowledge of the distributions or ranges of cryptic or rare species is poor</td>
<td>Kingsford and Norman 2002</td>
</tr>
</tbody>
</table>
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 achieved 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 (Dorffman 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 (Dorffman 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 nesting 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 Mediterranean 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 alterniflora* 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 Malchik 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 (Chlidonias 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 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, uncultivated 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 due 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 (Sternula 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).
Introducing predators, particularly mammals, may cause local extinction of breeding populations

Introducing 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 & Beyersbergen 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 = 21), 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 (Padyasakova 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 melanoleucus* in Sydney failed to raise any young in 1993, and this was attributed to egg predation by Australian Ravens and Little Crows (Dorffman 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 Dorffman 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 (Dorffman and Read 2000).
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; Lijstethrom 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 minimizes 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 to 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.

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**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 fledging 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).

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**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, Møller 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 fat 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 1.24 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 scap 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 Poyha 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.

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 Poyya 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 toxics 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 Washington 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, reviruses 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). Reviruses 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 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 gubberifrons 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 (e.g. 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 moult

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 Schriber and Risebrough 1972; Anderson and Keith 1980; Boellstorff et al. 1988; Bunnell 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 unguarded 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 nest 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 Collop 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 quary 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 et 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(5S) 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 (McKillogan 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 is 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 the 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.

![Search string example](image)

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 Pelecaniformes (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).
Reference list Part I


Reference list Part II


Waterbird responses to flooding stressors and threats

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