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D.S.L. Ramsey, C. Pacioni, S. McLeod and S. Dundas

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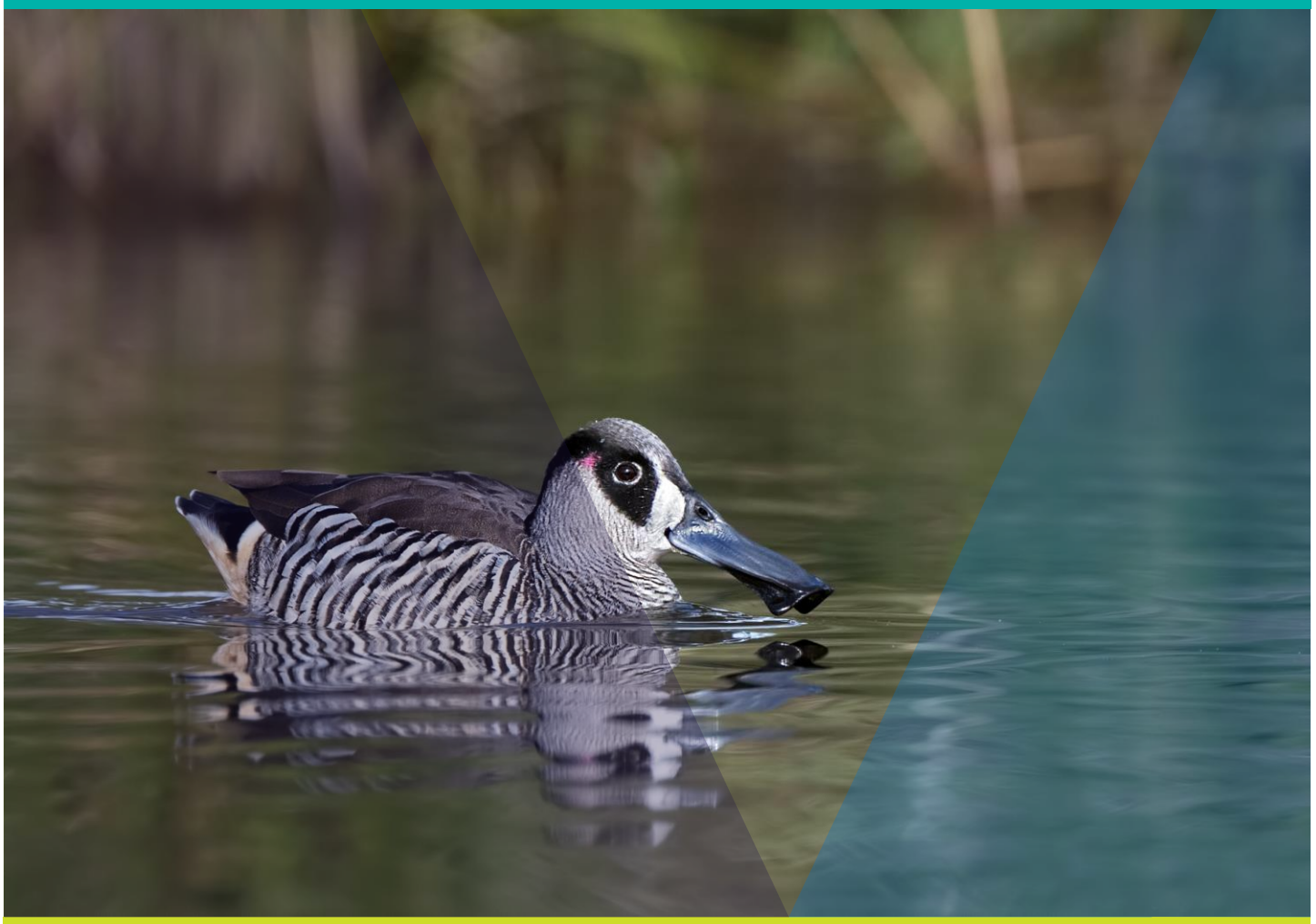

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Pink-eared Duck (*Malacorhynchus membranaceus*) (Photo: Craig Greer – http://craiggreer.com).

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Towards the implementation of adaptive harvest management of waterfowl in south-eastern Australia

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Contents

1. Summary

During 2008–2009, an expert panel was convened and tasked with developing a robust scientific approach to sustainable waterfowl harvesting in Victoria. This expert panel recommended an adaptive harvest management (AHM) approach be adopted, based on the development of a prototype model of the population dynamics of the various game species of waterfowl. The model was designed to be updated each year using monitoring data collected annually for the different game species, and then used to predict the optimal harvesting regime (bag limits and season length) for the upcoming season that would maximise harvesting opportunities while ensuring that populations remain within sustainable limits. The resulting model and monitoring recommendations were outlined in a report published in 2010 (Ramsey et al. 2010).

Since the publication of that report, almost 7 years have elapsed, and there is now a need to review the recommendations of that report to determine whether they are still fit for purpose. Additionally, there is interest in the AHM approach from other state agencies that undertake waterfowl harvesting and/or control [New South Wales (NSW), South Australia and Tasmania]. Hence, there is a need to revise the approach so that the different objectives, monitoring data and waterfowl/wetland dynamics in those states can be accommodated under an expanded AHM.

The objectives of this project are therefore:

1. Review the AHM modelling approach proposed for Victoria as well as the approach currently used in NSW, and propose a common AHM modelling approach that would be suitable for implementation by state agencies in south-eastern Australia.
2. Conduct investigations of the common modelling approach to determine its suitability for predicting the population dynamics of the various games species of waterfowl.
3. Review the AHM monitoring recommendations for Victoria and recommend a standard set of data collection protocols that would provide data relevant for informing the revised modelling approach. These data collection protocols would form the minimum necessary standards required for implementing AHM.

1.1 Recommendations

1.1.1 AHM framework

To help reduce contention around using AHM for the setting of harvest regulations, we recommend that ‘triple-loop’ learning be examined as a possible framework for conducting AHM. Hence, in addition to the annual update of the harvest model and regulations, there would also be an annual review and update of management objectives and alternative model structures, as well as of stakeholder ownership and governance processes. Hence, triple-loop learning would ensure that the most appropriate models were within the set examined by AHM, and also facilitate hunter and other stakeholder involvement in the AHM process.

1.1.2 Model for duck population dynamics

We recommend that a relatively simple, unstructured, Bayesian state–space model be adopted, to predict the outcome of environmental drivers and harvest on game species of ducks. Essentially, this model would be similar to that currently used by NSW for the setting of harvest quotas. Although existing data sources [principally the Eastern Australian Aerial Waterfowl Count (EAAWC)] can be used in this approach, we consider

that existing data is currently inadequate for effective AHM. Hence, additional data collection and/or modification of existing data sources will be required for effective implementation. These key data requirements/modifications are outlined in Table 1. If the full suite of these data collection requirements is implemented, the corresponding models used in AHM can be modified with additional structure, which should provide a greater degree of accuracy in model predictions.

1.1.3 Waterfowl monitoring data

This review has identified that the abundance of game duck species should be estimated within a set of predefined bioclimatic regions, to capture the likely regional variation in waterbird abundance at a tractable spatial scale. If this structure were adopted for AHM, suitable monitoring data would be required for each of the identified regions. We recommend that a desktop study be undertaken to identify the optimal combination of aerial (fixed-wing and/or helicopter) and ground-based surveys that would be required to provide estimates of waterfowl abundance for each region with sufficient precision, for the minimum cost. The desktop design should examine the utility of existing survey data sources (e.g. EAAWC, additional aerial transects, and citizen science bird surveys) for minimising the amount of extra survey effort required in each region. The design should be suitable for estimating the abundance of both mobile and sedentary game duck species.

In addition, ground counts and aerial surveys (fixed-wing and helicopter) need to be undertaken concurrently for a set of wetlands to determine visibility correction factors for the three different survey methods, so that counts from the different sources can be compared equivalently. This should be undertaken for a sample of 100–200 wetlands, stratified by wetland area/type (i.e. 100–200 wetlands per 3 – 4 strata). If correction factors were found to be reliable, then this would only need to be undertaken once, with the correction factors used to calibrate all subsequent survey data. However, if visibility correction factors are found to be unreliable, then monitoring should be undertaken in such a way that counts can be corrected for imperfect detection. This could be achieved using either distance sampling and/or double-observer sampling techniques (Koneff et al. 2008; Ridgway 2010). However, further work is required to overcome the practical challenges in implementing distance and/or double observer aerial surveys of waterfowl.

1.1.4 Water occurrence data

We recommend that accurate maps of water availability be assembled from satellite imagery annually to estimate wetland area/shoreline length available for breeding by ducks. Such data sources are now becoming more easily accessible from publicly available repositories (e.g. Google Earth Engine, Geosciences Australia). In addition, an easily calculated surrogate of water availability [e.g. the standardised precipitation evapotranspiration index (SPEI)] should also be investigated for possible use in AHM models and for estimating the occurrence of water in wetlands. (The SPEI can easily be calculated from gridded data on rainfall and evapotranspiration.)

1.1.5 Harvest data

We consider that the current arrangements for estimating harvest offtake for the game species of ducks in Victoria using telephone surveys is adequate for the purposes of AHM, as the current survey design is a representative sample of hunters and has enough resolution to apportion harvest by bioclimatic strata. Additional data are required on the harvest regulations (i.e. harvest quota/bag limits/season length) that were in operation during the years when telephone surveys were implemented. This would allow identification of any relationships between regulations and offtake for model predictions as part of AHM. We also recommend that a similar program, based on telephone surveys of hunters, be implemented in both NSW and South Australia to provide a representative

sample for estimating the size of the harvest. This could most easily be achieved by expanding the current Victorian program to those states. Further work is required to estimate the possible size of any reporting bias (i.e. over- or under-reporting of the number of birds harvested) present in the telephone survey.

1.1.6 Additional data required to increase predictive power of AHM models

The above recommendations are considered to be the minimum requirements for implementing effective AHM in south-eastern Australia. We also recommend that additional data collection be considered and gradually incorporated into the AHM program to increase the predictive accuracy of the models (Table 1).

Harvest bag surveys

Harvest bag surveys can provide data on age/sex ratios, which can be used to estimate juvenile recruitment/productivity. However, the current harvest bag surveys are unrepresentative and require a robust sampling design before that data can be used for AHM. We recommend that a new harvest bag survey be designed that uses stratified random sampling of wetlands within bioclimatic zones to provide unbiased and precise estimates of age ratios for each of the game species. We note that a guide to ageing birds more accurately is currently being produced, and the findings from that project should be incorporated into any future survey redesign.

Duck survival and movements

If AHM moves to a more structured model of duck population dynamics, additional data will be required for estimating the survival and movement of ducks. We recommend the implementation of studies involving the satellite/GPS telemetry of individual ducks, for estimating age- and sex-specific survival rates as well as movement between each of the bioclimatic zones. These could be supplemented by duck-banding programs. Telemetry/banding programs should be undertaken, ideally, in each bioclimatic zone, or at a minimum in zones with contrasting bioclimatic conditions.

Table 1: Monitoring data requirements for the effective implementation of AHM, subdivided into immediate, mid-term and long-term requirements

Immediate requirements
1. Survey design: A desktop study should be undertaken to determine the optimal mix of ground and aerial surveys (fixed-wing and/or helicopter) required for estimating game duck species abundances within each bioclimatic region in south-eastern Australia. Survey design should use the best mix of additional monitoring required for supplementing existing aerial and ground monitoring sources.
2. Monitoring data calibration: To calibrate monitoring data from different sources, aerial and ground counts should be undertaken concurrently for a stratified random sample of wetlands to estimate visibility correction factors. These would then allow monitoring data from different sources to be used in AHM models as if they were from a single source. However, if visibility correction factors prove to be unreliable, then, double-observer and/or distance sampling techniques should be investigated for both aerial and ground-based monitoring to correct for visibility bias.
3. Wetland availability: Assessment of wetland availability within each bioclimatic stratum is required to estimate total game duck population abundance from survey data. Accurate assessment will require the analysis of remotely sensed data (e.g. LandSat) and should be considered a high priority.
4. Harvest estimates: Current arrangements in Victoria are considered adequate. It is recommended that telephone surveys, similar to those undertaken in Victoria, be undertaken in other jurisdictions (e.g. NSW, South Australia) to provide a representative sample of hunters for estimating harvest offtake. This could be achieved by expanding the current Victorian program to those states.
Mid-term requirements
5. Redesign existing harvest bag surveys to provide a representative sample of the age/sex composition of the harvest.
Longer-term requirements
6. Implement satellite telemetry studies for the game species of ducks to provide high-resolution data on movement as well as age- and sex-specific estimates of survival. Ideally these should be undertaken across NSW, Victoria and South Australia.
7. Implement banding programs to estimate survival and movement of ducks as a complement to telemetry studies.

2. Adaptive harvest management: the ‘state of the art’

Adaptive management (AM) (Walters 1986; Walters and Holling 1990; Williams & Johnson 2013; Johnson et al. 2015) is a scientific approach to the design, implementation and evaluation of the effect of management decisions on natural resources that takes into account uncertainty in the decision process (Schreiber et al. 2004). When applied to management decisions around the regulation of harvesting of wild populations, adaptive harvest management (AHM) allows managers to learn about the potential of populations to support harvest, the ability of managers to regulate harvest, and the influence of environmental, social and economic factors on management decisions. However, despite being widely accepted as a suitable approach for learning how natural systems respond to management, there have been few successful implementations of AHM worldwide (Johnson et al. 2015). In the context of the management of recreational harvest of waterfowl, there is only a single example of AHM that has been undertaken over a sufficiently long period to enable some evaluation of the program. This is the program of AHM for waterfowl harvest in North America adopted by the U.S. Fish and Wildlife Service in 1995. Another more recent example of waterfowl AHM involves the program for the pink-footed goose (*Anser brachyrhynchus*) in Northern Europe, which has been running since 2012 (Madsen et al. 2017). We briefly review the 20+ year AHM program for mallards (*Anas platyrhynchos*) in North America in order to identify possible future issues for any implementation of AHM in south-eastern Australia.

AHM for midcontinent mallards was initiated in 1995 to replace the largely subjective decision-making process around the management of harvest regulations that had been used up to that point (Nichols et al. 2007). The key components of the AHM program included:

1. regulatory alternatives involving management settings for bag limits, season length and opening and closing dates
2. a set of population models describing how mallards respond to harvest regulations and environmental factors
3. an objective or ‘target’ describing the preferred state of the system (this allows the evaluation of management alternatives)
4. monitoring of both mallard populations and harvest offtake that allows a comparison of predicted and observed responses to management.

The key component of AHM is the explicit acknowledgement of alternative hypotheses about the effects of harvest and environmental variation on waterfowl populations. Each hypothesis is represented by a different population model and a model weight, reflecting the degree to which the model agrees with monitoring data. Each year the weights are updated by comparing the model predictions with monitoring data. Models that more closely predict the observed monitoring data have their weights increased, while those that agree less with the monitoring data have their weights decreased. This process continues each year, following a four-step process (Williams et al. 2002).

1. Each year the optimal harvest regulations are identified based on the current status of the waterfowl population and model weights.
2. Predictions are then made for the waterfowl population in the following year using the identified harvest regulations for each alternative model.

3. Once monitoring data become available, the predictions for each model are compared with the data, and model weights are updated based on the degree to which the models agree with the data.
4. The process begins again the following year, using the updated set of model weights.

This process should ensure that over time, the model that best reflects the population dynamics of waterfowl and the effects of harvest regulations will be identified and will become increasingly influential in the setting of harvest regulations. The alternative models for mallards expressed uncertainty about the extent to which harvest is additive to sources of natural mortality (i.e. additive vs compensatory harvest), as well as about the extent to which recruitment is density dependent (i.e. strong vs weak dependence). The model weights for these models have been updated over a 19-year period and currently suggest strong support for the weak density-dependent recruitment model (0.96 vs 0.04) and moderate support for the additive vs compensatory mortality effects of harvest (0.67 vs 0.33).

Although this updating process has reduced structural uncertainty by favouring a particular model over alternatives, it has also raised a number of issues related to the rate of learning, and whether the models are still an adequate description of waterfowl population dynamics. The first issue relates to the rate of learning represented by the movement in the model weights over time. During the first 3 years of the AHM program, there was little movement in the model weights because none of the models was a particularly better predictor of the observed monitoring data. However, in 1999, both of the weak density-dependent recruitment models showed a large increase in model weights compared with the alternatives. Thereafter, learning has been incremental, with increasing support for the additive vs compensatory mortality model emerging only in the last 5 years. This illustrates that the rate of learning about which of the models is a better representation of the system could be slow, requiring a long-term commitment to the AHM process. The second issue relates to the current suitability of the four candidate models. All four models have underpredicted the observed population of mallards in 6 of the last 8 years, suggesting that a key relationship is missing from the current model set (Johnson et al. 2015). This suggests that the existence of trends in long-term structural changes in the system (e.g. climate change, land-use changes) may need to be accounted for in the AHM process. These findings suggest that any future implementation of AHM in Australia should be closely linked to changing environmental conditions. Learning rates will also need to be monitored and model alternatives reviewed annually to ensure that the population dynamics of the main game species in Australia are adequately predicted by at least one of the candidate models.

The review by Johnson et al. (2015) identified four broad lessons from the North American AHM process.

(i) When extending the AHM to species other than mallards, the ability to tailor harvest regulations to each species was limited, with regulations set predominantly on what was required for mallard harvests. In effect, managers in many jurisdictions were reluctant to have species-specific regulations due to the increased complexity and cost. This usually meant that management opted for a more simplified set of regulations regarding harvest. It was noted that regulatory alternatives are basically social constructs that are usually designed to be acceptable to hunters, and that this restricts the designing of a systematic and biologically meaningful approach to defining a set of regulatory alternatives. A related problem was the fact that large differences in hunting regulations often resulted in small changes to average harvest rates. This presented a communication challenge for managers when frequent changes to regulations were recommended, because hunting groups prefer consistent regulations.

(ii) Management objectives for AHM need to be unambiguous, and they are usually crafted from a biological perspective. The mallard AHM management objectives were to maximise cumulative harvest while maintaining population size above a threshold. However, there is now a concerted effort to frame management objectives more in terms of hunter satisfaction, rather than to maximise harvest. While these complexities can be embraced by the AHM process, there would need to be considerable research into metrics that would be suitable for measuring hunter satisfaction and participation.

(iii) There is now concern over the growing divide between stakeholders in the AHM process. Due to the increased complexity of the technical aspects of AHM, involvement in and ownership of the process is now dominated by those biologists with a more numerical background. This has led to stakeholders concerned more about the social aspects of AHM being marginalised. This has led to some partners calling for harvest regulations to be set based on 'rules of thumb'.

(iv) The current AHM process needs to be modified to cope with long-term incremental environmental changes. Models developed to date for AHM assume that the environment, in the long term, is essentially stable (i.e. fluctuating around a constant long-term average). However, large-scale, global processes like climate change and land-use changes essentially result in average environmental conditions changing over the long term. This means that models that assume stable environmental conditions will essentially become outdated and will no longer reliably predict the population dynamics of waterfowl. Adapting the AHM modelling process to cope with long-term environmental change presents a considerable technical challenge.

In summary, after 20 years of AHM of mallards in North America, Johnson et al. (2015) conclude that while AHM has been successful in reducing the contentiousness around the setting of harvest regulations, and that much has been learned regarding the population dynamics of mallards and their relationships to the environment and hunting, there are now growing concerns around changes in the social and institutional aspects of hunting and whether the AHM process can rise to meet these challenges. This has led to an examination of so-called 'triple-loop learning'. The first loop involves the annual update of the AHM model and regulatory decisions around harvest. The second loop involves a re-examination of management objectives, regulations and alternative model structures, while the third loop involves an examination of stakeholder ownership and governance processes. It is recommended that triple-loop learning be examined to determine whether this would be a valuable framework for AHM for Australian waterfowl.

3. Models of waterfowl population dynamics suitable for AHM

Wildlife populations can be characterised as highly structured, stochastic, dynamic systems about which we usually have incomplete information. However, inferences about these systems are required, especially where species of economic and/or conservation importance are involved. Here, mathematical models of the system can play a vital role in the decision-making process by allowing hypotheses about the population to be tested against observations (Thomas et al. 2005). Models form an integral part of AHM because they allow the prediction of the likely response of a population to various levels of harvest. However, all models are approximations of real systems, with the utility of any model being measured by how close it approximates reality. In respect of AHM, one measure of utility would be how accurately the model reflects the dynamics of the waterfowl population. This, in turn, will depend on how well the model structure accounts for important differences between individuals, in terms of demographic rates such as survival, recruitment and movement. A second measure of model utility would be how accurately it

represents the relationships between management actions (through changing harvest regulations) and the response of the population to harvest (Cooch et al. 2014).

Model structures of the population dynamics of wild populations can be bracketed by two extremes (Cooch et al. 2014). At one extreme, scalar projection models assume that all individuals are subject to identical rates of survival and recruitment (i.e. are non-spatial). A further simplification occurs when survival and recruitment parameters are implicitly combined into a single parameter, the rate of increase. The scalar state–space models adopted by the NSW Department of Primary Industries (DPI) for setting damage mitigation culling quotas (Dundas et al. 2016), and those for mallards developed by Lillegård et al. (2008), are examples of these types of models. The advantages of these models are their modest data requirements (essentially just a time series of abundance estimates or indices), their ability to incorporate the uncertainty around the monitoring data (i.e. imperfect detection) and the relative ease with which the effects of environmental variables on the rate of population change can be assessed. These models have only one state parameter—total population abundance—and assume the modelled population is closed to movement. At the other extreme, models in which each individual has separate probabilities of survival, recruitment and movement (dispersal) are known as individual-based models (IBMs). These models allow the closest approximation to reality and accommodate any structure, including demographic, spatial and even genetic structures. However, the complexity of IBMs is also their Achilles' heel. IBMs are difficult to construct and analyse and may be intractable for very large populations, such as for regional populations of waterbirds (where numbers can run into the millions). In addition, IBMs are difficult to fit to data, as most data sources lack sufficient demographic, temporal and spatial resolution.

Hence, the most appropriate model structure for waterfowl AHM will necessarily be dictated by the demographic and spatial resolution/extent of existing and proposed future data sources. We reviewed the different approaches to modelling the dynamics of waterfowl populations that have been published in the recent literature (2007–2017). The aim of the modelling review was to canvas the different approaches to waterfowl modelling that have been undertaken since the publication of the waterfowl AHM model detailed in Ramsey et al. (2010). Based on this review (Appendix), we recommend a state–space modelling approach be adopted for modelling waterfowl population dynamics in south-eastern Australia. The advantage of the general state–space modelling approach over that recommended by Ramsey et al. (2010) is that the state–space approach can be implemented using more modest data requirements and hence can make the best use of existing data sources. Because the state–space approach recommended is a general one, it can easily be extended as more detailed data sources become available. This will be important for the initial implementation of AHM to make maximum use of existing data sources, which will be required until more detailed/higher-spatial-resolution data sources become available. Below we describe the generalised state–space model for a harvested waterfowl population. It would be suitable for describing the dynamics of game birds in south-eastern Australia. We also describe possible extensions to the model that could be implemented when higher-spatial-resolution and/or additional data sources become available.

3.1 A general state–space model of waterfowl subject to harvest

We propose a general state–space modelling framework (Clark and Bjørnstad 2004; Buckland et al. 2004) for the modelling of a waterfowl population subject to harvest that has the general form:

$$N_t = \lambda_t N_{t-1} - k_t,$$

where N_t is the population abundance of a game species at time *t*, λ_t is the (finite) rate of population growth between times *t* – 1 and *t*, and k_t is the number of animals harvested between times *t* – 1 and time *t*. Setting $\log(N_t) = x_t$ and including a stochastic error term (ε), gives the following linear model:

$$x_t = f(x_{t-1}) + \varepsilon_t \qquad \text{Eqn. 1}$$

$$y_t = g(x_t) + \upsilon_t,$$

where $f(x_{t-1})$ represents the process model for the population at time *t* – 1. The state–space formulation assumes that x_t represents the true, but unobserved (log) population size (the state process). Similarly, y_t is the observed (log) count of the population at time *t* that is linked to the state process through the observation process $g(x_t)$, that is also associated with stochastic error υ_t. Here, the state process $f(x_{t-1})$ can incorporate additional structure to that of harvest, including density dependence, environmental covariates, and movement. For example, if we assume one homogeneous population with no movement but density-dependent regulation and harvest, Equation 1 becomes:

$$x_t = \beta_0 + \beta_1 x_{t-1} + h_t, \qquad \text{Eqn. 2}$$

where $h_t = \log|1 - k_t/N_{t-1}|$ is the logarithmic expression for the harvest rate at time *t*, which is zero in the absence of harvest and negative otherwise (Colchero et al. 2009), and β_0 and β_1 are parameters relating to density dependence. This is the Gompertz model of population growth with harvest. The effects of environmental variables on the population can also be easily incorporated by including additional structure into Equation 2.

3.2 Spatial structure

If we wish to model the movement of birds between discrete patches or regions, then Equation 2 can be generalised by including a separate model for each patch, linked by dispersal:

$$x_t^i = \beta_0^i + \beta_1^i x_{t-1}^i + h_t^i + \textstyle\sum_{j=1}^{n} \omega_t^{i,j}, \qquad \text{Eqn. 3}$$

where x_t^i is the abundance of the populations in patch *i* (*i* = 1, .. , *n*) and $\omega_t^{i,j}$ is the dispersal rate of birds from patch *i* to patch *j*.

Further generalisation to multiple patches or regions is of course possible and would depend on the level of spatial resolution in the various sources of monitoring data. It is also possible to include additional structure on the dispersal rates by expressing them as linear models of potential explanatory variables (e.g. rainfall, wetland area).

3.3 Demographic structure

If one wishes to add additional demographic structure, then the population growth rate can be decomposed into survival and recruitment terms:

$$N_t = \left(\phi_t^a + \phi_t^j R_t\right) N_{t-1} - k_t, \qquad \text{Eqn. 4}$$

where ϕ_t^a and ϕ_t^j are the annual survival rates of adult and juvenile birds, respectively, and R_t is the productivity rate (juvenile birds per female) (Freeman and Besbeas 2012). On the log scale, Equation 4 becomes:

$$x_t = x_{t-1} + \log\left(\phi_t^a + \phi_t^j R_t\right) + h_t. \qquad \text{Eqn. 5}$$

The inclusion of additional demographic structure will become important when there are additional sources of data that can be used for estimating the survival rates (e.g. band return data) and productivity (e.g. age class ratios). Survival rates can include expressions for density dependence and/or incorporate components for natural and harvest mortality.

Equations 1–5 outline the general form of the state–space modelling approach (and possible extensions) proposed for modelling waterfowl population dynamics. The approach is flexible in that a simple model can be proposed if the only source of data is a time series of population counts for a single area. However, as additional data sources become available, or if improved spatial resolution is required, then the modelling approach can be easily expanded as necessary. We propose that the resulting model be fitted to the observed monitoring data using an integrated population modelling (IPM) Bayesian framework (Appendix) that can easily accommodate multiple sources of monitoring data to make model-based inferences.

4. Monitoring data required to inform the AHM model

The modelling framework proposed above requires information on waterfowl abundances and the size of the harvest to be collected annually, as a minimum requirement. In addition, if hypotheses regarding the effects of harvest on waterfowl are required (i.e. additive vs compensatory harvest mortality), then Equation 5 will need to be used and additional information required in order to estimate survival rates (i.e. band return data) and productivity (i.e. age ratios). If spatial heterogeneity is to be incorporated into the model to account for regional differences in waterfowl abundance/trends (Equation 3), then the spatial resolution of the data will also need to be considered. Here, adequate data would need to be collected for each proposed spatial unit (region). Below we review the minimum information requirements and associated monitoring data that will be required to inform the AHM model.

4.1 Waterfowl abundance estimates

As many species of game birds are highly mobile, the area subject to waterfowl monitoring necessarily needs to encompass a large portion of eastern Australia, roughly equivalent to the area of the Murray–Darling Basin. This is the spatial scale required by monitoring if we wish to understand the major drivers of the population dynamics for the highly mobile duck species [e.g. grey teal (*Anas gracilis*), pink-eared duck (*Malacorhynchus membranaceus*)]. Hence, the areas recommended for waterfowl monitoring by Ramsey et al. (2010) included 16 major wetland complexes encompassing Victoria, NSW, SA and Qld. The only current source of monitoring data that is carried out over this spatial scale is the EAAWC (Kingsford and Porter 2009). The EAAWC is flown annually in October and consists of 10 east–west transects (survey bands) spaced every 2° of latitude (Figure 1). Transects were approximately 30 km wide, with every wetland greater than 1 ha in area subject to survey using a fixed-wing aircraft flying at 46 m height at 167 km/h. In addition, an index of wetland area for the approximately 800 surveyed wetlands was also obtained (Kingsford and Porter 2009).

The advantages of the EAAWC are that it is the only spatially and temporally extensive source of monitoring data for waterbirds in Australia. It encompasses an area containing most of the important wetland regions in eastern Australia and includes historical information on waterbird populations and wetland areas stretching back to 1983. The

disadvantages of the EAAWC are that it has a relatively low spatial coverage of the total survey area (12%: Kingsford and Porter 2009); hence, survey coverage for smaller regions of significance can be low or absent. For example, the major wetland complexes in Victoria have a relatively low coverage by the EAAWC. As the EAAWC uses a systematic survey design, the low coverage means that many important wetland regions remain unsurveyed. Hence, important sources of spatial heterogeneity in waterfowl abundances may not be accounted for in the EAAWC. The implications of this are that the EAAWC data for particular regions may be unrepresentative, and hence may not adequately index the population abundance of some waterbird species in these areas. To obtain more accurate and precise estimates of waterfowl populations for regions such as Victoria would require additional aerial monitoring data to supplement that of the EAAWC. Additional aerial transects are already flown in the Riverina district in NSW to help inform decisions around waterfowl pest mitigation in NSW, and other transects have been flown over important wetland complexes in the Murray–Darling Basin elsewhere. Hence, it would also be desirable to include some additional aerial transects in Victoria to increase the spatial resolution of the monitoring data currently available for the state. This could incorporate some or most of the important wetland complexes identified in Ramsey et al. (2010).

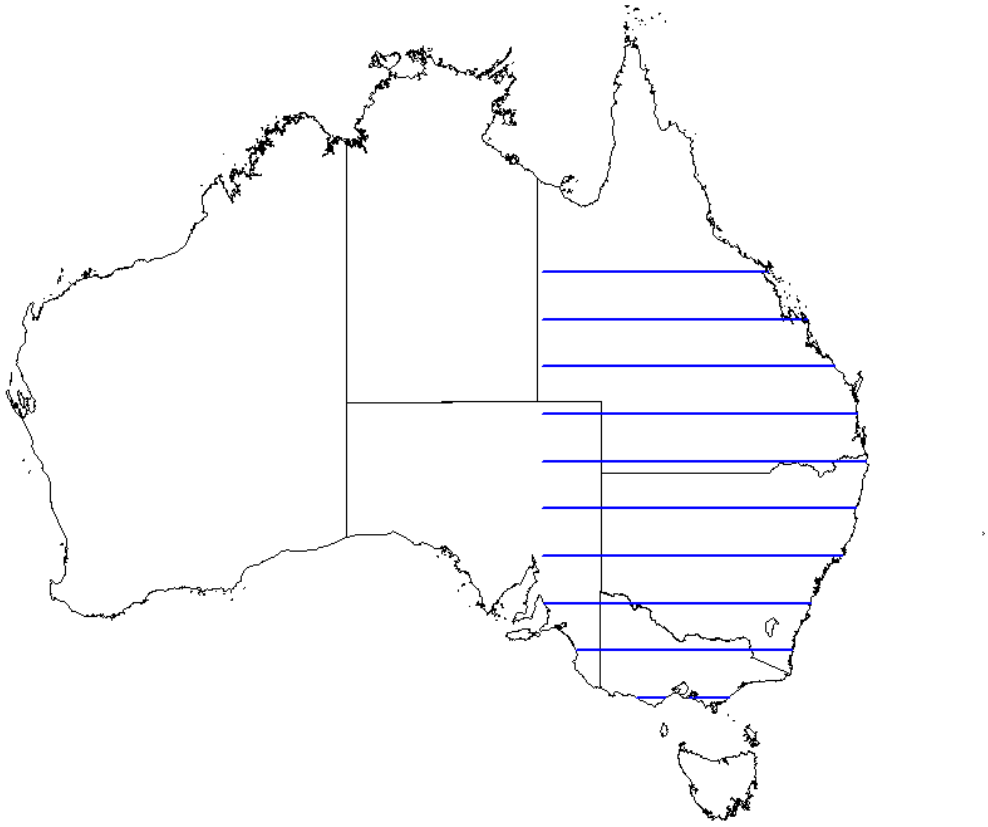

Figure 1. Location of the 10 survey transects for the waterfowl aerial surveys undertaken annually during the eastern Australian aerial waterbird count (Kingsford and Porter 2009).

4.2 Other sources of waterfowl survey data

4.2.1 Victorian waterfowl surveys

A ground-based waterfowl count has been conducted in Victoria each February, since 1987, averaging around 300 wetlands counted per year, and it is known as the Summer Waterfowl Count (SWC) (Menkhorst and Purdey 2016a). However, in response to resource constraints, the number of wetlands surveyed in recent years has been much

lower. The main purpose of the survey is to identify wetlands where threatened waterbird species occur [e.g. freckled duck (*Stictonetta naevosa*), blue-billed duck (*Oxyura australis*)] so that wetland closures could potentially occur prior to the start of the hunting season. The main issues with the data collected from the SWC is that the wetlands surveyed each year are a non-random sample of wetlands and, hence, the data are unrepresentative. Thus, no attempt has been made to estimate total populations of waterbird species using these data. As the purpose of these surveys is to identify wetlands containing threatened species prior to the start of the hunting season, it is unlikely that the structure of these surveys will change significantly.

4.2.2 South Australian waterfowl surveys

Ground-based surveys of waterfowl have also been undertaken in South Australia by the Department of Environment, Water and Natural Resources (DEWNR) since 2003 (Anon 2016). In 2016, 85 wetlands were surveyed in October and November across four districts of south-eastern South Australia: Murraylands, the Fleurieu Peninsula, the Coorong and the South-East. Surveys are carried out by staff of the DEWNR with the assistance of volunteers. In addition to the ground counts, aerial surveys are also conducted by the DEWNR on a limited number of wetlands (to calibrate the ground counts). However, no details about the calibration were provided (Anon 2016). In addition, although sampled wetlands have clearly been stratified by region, there is no other information about sampling design and, hence, whether the surveyed wetlands can be considered a representative sample. However, it is considered that these surveys could usefully inform any implementation of AHM in that state.

4.2.3 NSW waterfowl surveys

In NSW, the Riverina region has been the focus of waterfowl surveys, as pest mitigation harvests are carried out primarily in this region. Surveys of waterfowl were carried out in February 2017, using a range of survey designs and techniques, with the aim of identifying a survey methodology suitable for the region. Surveys were carried out by counting waterfowl using both aerial (helicopter and fixed-wing aircraft) and ground surveys. Survey transects were defined by targeting larger dams, natural wetlands and wastewater treatment ponds as the primary survey points, then surveying any small dams, rivers or irrigation canals encountered along these routes.

To estimate the detection probability of ducks during aerial surveys, double-observer counts were carried out in both the helicopter and fixed-wing aircraft. Two observers seated on the same side of the aircraft independently identified and counted all ducks of each species observed at each of the survey locations. Observations were recorded using voice recorders that had been synchronised to start at the same time. Recorders were left on for the entire survey. When large flocks were encountered (100s to 1000s), the total number of waterfowl seen was estimated using group blocks (generally groups of 20, 50 or 100), and a total proportion of each species present was recorded. In addition to the aerial surveys, concurrent ground-based counts were carried out as a complementary survey method. The aim of the ground surveys was to derive a total count that could be used to calibrate the aerial counts, so that aerial counts could be expressed as an equivalent ground count (or *vice versa*). One issue with such calibration exercises is that the number of ducks on a water body can potentially change from day to day, meaning that population size can change between aerial and ground survey times. Hence, any such calibration should attempt to synchronise the timing of aerial and ground counts to minimise the chances of population changes between surveys.

Analysis of the data from these surveys is still being undertaken. However, some preliminary conclusions can be drawn. From a logistical and practical perspective, helicopters offer the best solution with respect to covering all water body types where

waterfowl are likely to reside, because they are slow enough to give observers sufficient time to identify and count waterbirds and resurvey if necessary. Fixed-wing planes are much cheaper to operate, but speed, limited manoeuvrability and difficulties with surveying smaller dams make them less suited for targeted waterfowl surveys on smaller water bodies. However, it is likely that the optimal survey design for waterfowl will contain a mix of fixed-wing and helicopter aerial surveys, so that coverage can be maximised at a reasonable cost. Correction of aerial counts for imperfect detection is still a key element for any well-designed survey, and calibration surveys (using ground counts) or double-observer surveys are recommended in order to account for imperfect detection during aerial surveys.

An additional challenge is surveying less common species that are only ever present in small groups, or those easily missed when present in mixed flocks [e.g. Australasian Shoveler (*Anas rhynchotis*), Chestnut Teal (*Anas castanea*)]. For these species, a general survey may not have adequate detection rates, and more intensive, targeted surveys may be required. Another option would be to establish species-specific detection probabilities during a standard survey. This could be achieved in helicopter surveys if a third observer was assigned the task of simply identifying all species present; their data could then be compared with the identified species counts collected by the two primary observers.

5. Additional monitoring data required for AHM

To obtain more accurate and precise estimates of waterfowl abundances in different regions of Victoria and NSW, it will be necessary to implement additional aerial monitoring of wetlands to complement the monitoring of wetlands presently sampled by the EAAWC. Some additional aerial survey transects are already undertaken in NSW, so any further investment in aerial surveys should complement the existing effort. Any additional monitoring should also be examined to determine the level of coverage of the major wetland complexes identified in Ramsey et al. (2010), especially in Victoria. Current ground surveys of wetlands in Victoria surveyed as part of the SWC are unrepresentative, and hence cannot be used to make the unbiased estimates of waterfowl abundance required for AHM. As the SWC is undertaken for a specific purpose (locations of threatened species), it should not be considered further as part of any future implementation of AHM.

Any additional aerial or ground surveys contributing to AHM should be designed so as to be considered a representative sample of wetlands. By undertaking representative sampling of wetlands, we can ensure that the resulting estimates of waterfowl abundance are unbiased. The easiest way of ensuring representative sampling is by using random sampling of survey units. However, simple random sampling is usually inefficient, requiring a large sample for precise inference. This is because simple random sampling often results in sample locations that are in relatively close proximity to each other, providing redundant information if samples are spatially autocorrelated. One method for counteracting the inefficiency in simple random sampling is a spatially balanced survey design (Foster et al. 2017). A spatially balanced design is an extreme form of stratification that aims to reduce the frequency of samples located in close proximity to each other so that autocorrelation is reduced; hence, the greatest amount of spatial information is extracted from each sample. In addition, it should be recognised that the EAAWC contains important historical information for the wetlands it covers, and this information should be taken into account in the survey design. The spatially balanced survey design allows for the accommodation of legacy sites (i.e. existing EAAWC survey data) by reducing the likelihood of sampling in close proximity to existing legacy sites (Foster et al. 2017).

Hence, a spatially balanced survey design makes the best use of both existing and new monitoring information.

We recommend that a desktop study be undertaken to determine the optimal amount of new monitoring effort required in order to implement AHM in south-eastern Australia. This study would incorporate existing survey information collected as part of the EAAWC (and any additional aerial transects flown) as well as the ground and aerial waterbird monitoring undertaken in South Australia. The aim of the desktop study would be to find the optimal (i.e. cost-effective) amount of additional monitoring required for estimating abundances of the game species of ducks with sufficient precision, but minimum cost. The study should examine combinations of ground surveys and aerial surveys (fixed-wing and/or helicopter-based), in the optimisation. If possible, citizen science–based monitoring data (e.g. from Birdlife Australia) should also be examined for inclusion in the monitoring design. Monitoring strategies should be stratified by regions that might be expected to show broad differences in waterfowl abundance (i.e. due to bioclimatic factors). Regions should be defined based on catchment boundaries (i.e. aligned to CMA boundaries) to capture regional variation in hydrology, and hence variation in wetland availability. An example of possible regions across Victoria and NSW that could be used for bioclimatic stratification is provided in Figure 2. However, we recommend that expert opinion is sought before deciding on regional stratification boundaries. Monitoring designs should be optimal for both the highly mobile game species (e.g. grey teal) as well as for the more sedentary species [e.g. Australian wood duck (*Chenonetta jubata*)]. Hence, the design will need to consider coverage of both the major wetland complexes (e.g. Ramsey et al. 2010) and smaller water bodies that are likely to harbour the more sedentary species.

Figure 2. One possible example of bioclimatic regions that could be used to stratify waterfowl monitoring in Victoria and NSW. Boundaries are aligned with catchment management authority boundaries. The blue-shaded coastal region of NSW [including the Australian Capital Territory (ACT)] could be either excluded from, or included in, the waterfowl monitoring program.

One issue with designing surveys of wetlands is that many wetlands are ephemeral, due to the highly variable rainfall patterns in some parts of Australia. This means that surveys are encountering wetlands that are often dry. It is important to include ephemeral wetlands in the aerial surveys, because these are often highly productive when they contain water, leading to high abundances of waterbirds. On the other hand, the efficiency of surveys would be increased if surveys were more likely to encounter wetlands that contain water. Hence, this suggests that it may be useful to stratify survey effort in a region based on the likelihood that wetlands within aerial transects would contain water. Thus, more effort would be directed to areas where wetlands are more likely to contain water, and less effort directed to areas where wetlands are less likely to contain water. Here, historical satellite imagery of water availability could be used to estimate the likelihood of wetlands containing water based on historical climate patterns. The Global Surface Water project undertaken by the Joint Research Centre of the European Commission recently completed a global map of the occurrence and persistence of surface water (Pekel et al. 2016). Using 3 million Landsat images at 30 m resolution, the changes in surface water occurrence were quantified by recording the month and year when water was present at each pixel

location over the past 32 years (1984–2015). This dataset is now publicly available at https://global-surface-water.appspot.com/.

We downloaded data for the occurrence of surface water for Victoria and NSW for the period 1984–2015 from the Global Surface Water project made available through the Google Earth Engine website https://earthengine.google.com/. To reduce the size of the downloaded file, we decreased the spatial resolution to a pixel size of approximately 500 m. The occurrence of water for each pixel was summarised as the percentage of times over the 32-year period that the pixel contained surface water (Figure 3). We then classified this percentage into three possible strata, labelled as low, moderate and high water occurrence (low, 0–30%; moderate, 30–70%; high, 70–100%). Examples of this stratification are given for each of the suggested bioclimatic regions for Victoria (Figure 4). We recommend that information, such as presented in Figure 4 be the basis for stratification to guide the development of optimal, spatially balanced waterfowl survey designs. The design of an optimal monitoring strategy that makes the best use of existing survey data should be a high priority in any implementation of AHM.

One issue for a waterfowl monitoring design that uses both aerial and ground-based monitoring data is the fact that bird detection probabilities are likely to be very different between the two methods. Indeed, detection probabilities of birds are also likely to differ between fixed-wing and helicopter-based aerial surveys. There are two ways of dealing with this issue. The first method would be to conduct some aerial surveys and ground surveys concurrently to estimate visibility correction factors to express aerial counts as ground count equivalents (Lillegård et al. 2008). The advantage of this kind of calibration is that it would only have to be performed once, assuming the estimation of the calibration parameters was sufficiently precise and reliable. However, if visibility correction factors prove to be unreliable, then methods for estimating the detection probability of aerial and/or ground surveys, such as double-observer and/or distance sampling techniques, (e.g. Koneff et al. 2008; Ridgway 2010) should be investigated. In particular, the practical and logistical complications of distance and/or double observer sampling for aerial surveys would need further investigation.

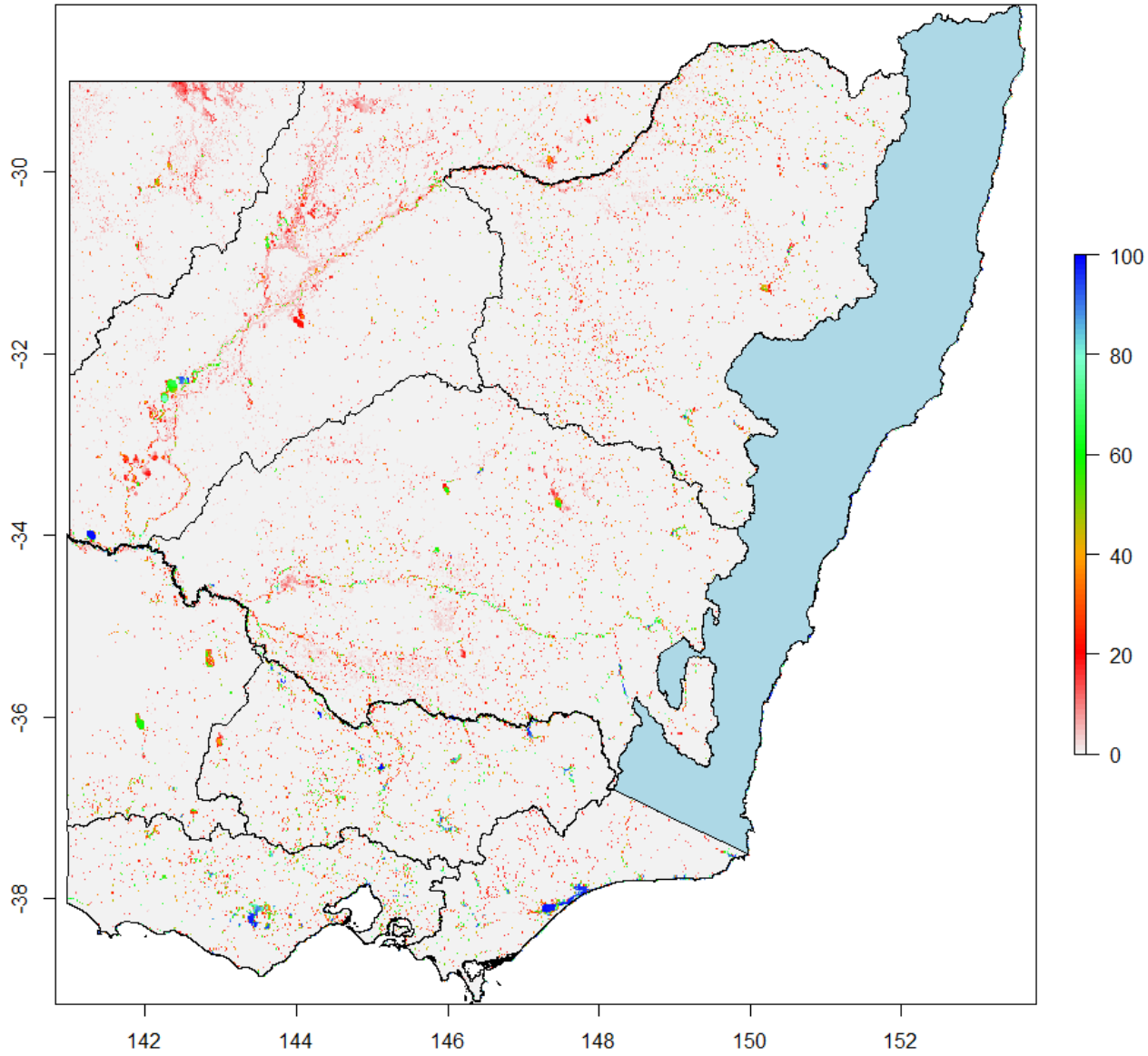

Figure 3. Wetlands in Victoria and NSW and the associated probability of them containing surface water (on a scale of 0–100) over the period 1984–2015, estimated from Landsat imagery from the Global Surface Water project.

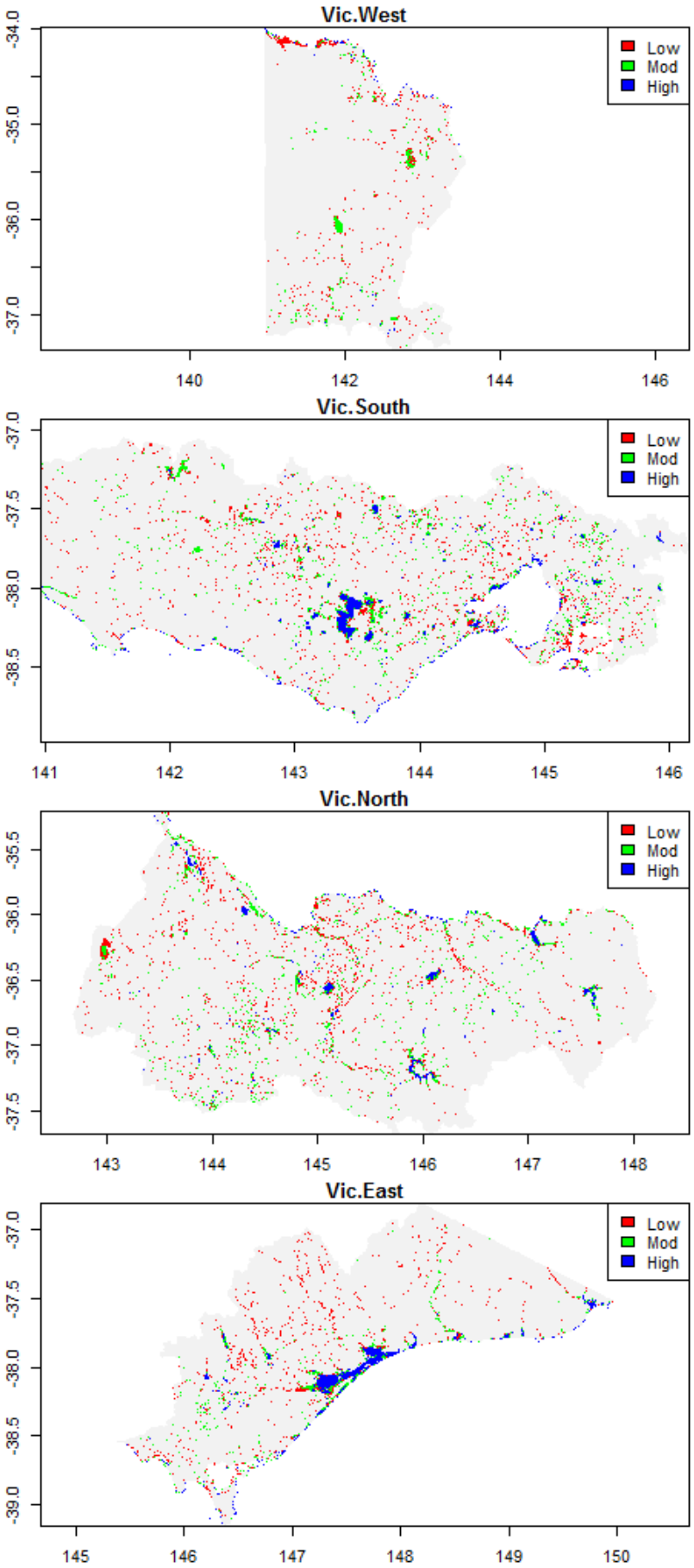

Figure 4. Wetlands in Victoria stratified by the probability they contained surface water over the period 1984–2015, estimated by Landsat imagery from the Global Surface Water project (Low, 0–30%; Mod, >30–70%; High, >70–100%).

5.1 Harvest estimates

Estimates of the total harvest of each of the game species of ducks is required for estimating the harvest rate for each species, each year (i.e. k_t/N_t), so that the effects of the harvest rate on the population dynamics of each species can be estimated. Harvest can interact with waterfowl populations in different ways, influenced by population abundance and age structure, which ultimately could affect the processes governing population regulation (Hauser et al. 2006). Moreover, restrictions on harvest imposed by managers are an attempt to regulate the harvest rate to bring about a desired outcome. Hence, knowledge of the relationship between management regulations (e.g. harvest quota, bag limit, season length) and the harvest rate are critical for understanding the effects of management of waterfowl populations. Here, we review the data currently collected by Victoria and NSW that are used for estimating the total harvest of hunted waterfowl to determine whether they are adequate for estimating harvest rates for AHM.

5.1.1 Victoria

The total harvest for each of the game species of ducks has been estimated each year since 2009 using phone surveys of hunters (Moloney and Turnbull 2016). Game licence holders are randomly sampled and interviewed by telephone at regular intervals during the hunting season and asked questions about the number of days they hunted, the number and species harvested, and the (approximate) location of hunting activities. An estimate of the total harvest of each game species is then made by correcting for the proportion of respondents who hunted and the average harvest per respondent [see Moloney and Turnbull (2016) for estimation details]. As the survey also provides hunting locations, it is relatively easy to apportion harvest by region. For example, the estimate of the total duck harvest by CMA for 2016 is given in Figure 5 (Moloney and Turnbull 2016). These data are critical for the development of AHM, and we recommend that no major changes be made to the current methodology.

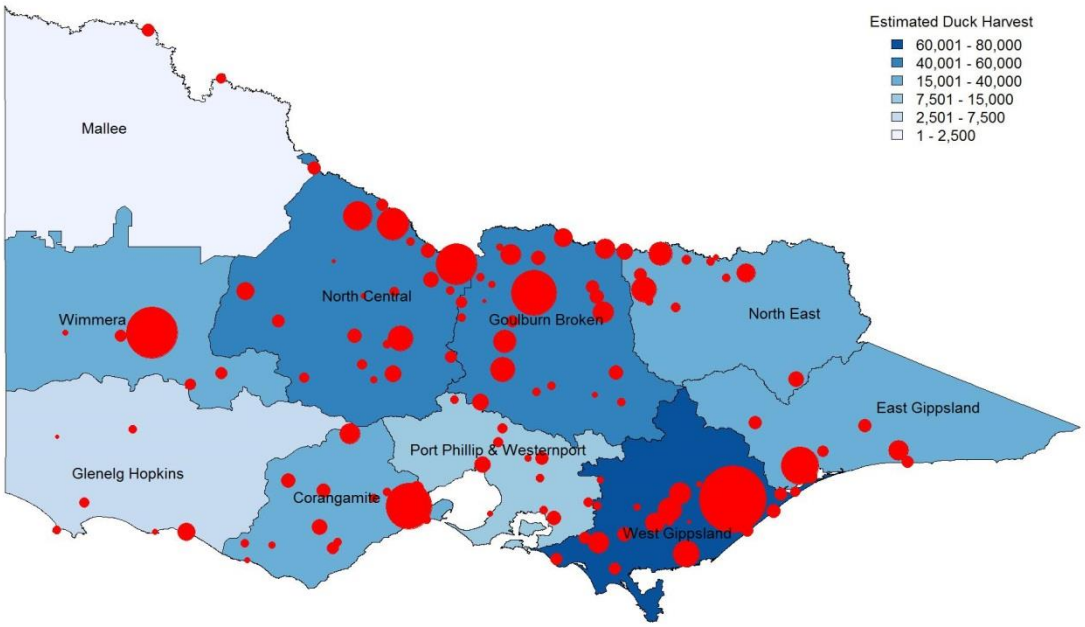

Figure 5. Estimates of the total harvest of game ducks in Victoria by CMA in 2016. Red circles indicate the nearest town to harvest locations, with symbol size proportional to reported harvest (Moloney and Turnbull 2016).

5.1.2 NSW

In NSW, 10 native duck species can legally be culled for the purposes of pest mitigation, primarily in the rice-growing region of the Riverina. The Game Licensing Unit within the NSW DPI manages the issuing of pest mitigation licences for hunters and property owners, following their successful completion of the Waterfowl Identification Test (WIT). Each year, an annual NSW-wide quota is allocated and, from this, each property is allocated a harvest quota. As a requirement of the licence to harvest waterfowl, hunters and property owners must report all waterfowl harvested on each property through the online native game bird management portal. However, since this is based on self-reporting, the harvest estimates may be biased if certain kinds of property owners are more likely to report than others. There is also an incentive for hunters and landowners to under-report harvest offtake to avoid exhausting the quota.

5.2 Estimates of demographic rates

If a modelling framework that contains demographic structure (e.g. Equation 5) is adopted for AHM, then additional information will be required on the survival rates of adult and juvenile ducks, as well as on productivity. Survival rates can be estimated using an analysis of band returns (White 1983) or telemetry data (Roshier et al. 2008a). Telemetry data collected from tagged individuals is the most robust way of estimating survival rates, but is relatively expensive, requiring a large number of individuals to be tagged. Band returns require a large initial investment in banding, but hunting can be used for recovering bands (White 1983). In addition to survival rates, analysis of band returns through hunting also allows estimates of recovery and harvest rates of ducks; hence, it can be used to apportion the various sources of mortality (Williams et al. 2002). Furthermore, telemetry can also be used to obtain high-resolution data on movement or dispersal rates of ducks (e.g. Roshier et al. 2008b), information that will be critical for informing the metapopulation model of ducks and their responses to regional environmental conditions. Therefore, we recommend that both telemetry studies and duck-banding programs be considered as an integral part of any implementation of AHM.

Estimates of productivity (i.e. juvenile recruitment rates) can be obtained from an analysis of the age and sex composition of the harvest. Usually, the ratio of juveniles to adult female birds of each species in a large, representative sample can be used as an estimate of the recruitment rate (Skalski et al. 2005). This is often obtained through surveys of hunter bags, usually on opening weekend. In Victoria, the hunter bag surveys (Menkhorst and Purdey 2016b) have been conducted during the season opening weekend since 1972. In 2016, surveys were conducted at 20 public wetlands over the opening weekend, and the composition of the take by individual hunters was assessed. For each hunter, the numbers and species composition of the bag was assessed, as was the age (juvenile/adult) of each bird (through an examination of wing or tail plumage) (Menkhorst and Purdey 2016b). Overall, in 2016 a total of 1042 individual bags were assessed, sampling 1843 game birds. However, only 219 of these were examined for age classification; furthermore, the sample of birds examined was not representative of the overall species composition of the bag surveys. Hence, the current data collection undertaken for the opening weekend bag surveys is not suitable to be used for AHM. We recommend that hunter bag surveys be redesigned to provide a larger, representative sample of hunter bag composition and age/sex classifications. It is noted that efforts for rectifying the deficiencies of the hunter bag surveys are currently underway.

5.3 Assessment of wetland status

Models developed for AHM should include the effects of environmental variation on population dynamics. Typically, the population dynamics of game birds are linked to

wetland availability, with the distribution of water responsible for large-scale movements of birds, resulting in large changes in abundance in response to flooding events (Roshier et al. 2008a, 2008b). Hence, any model used for AHM will need to be linked to the seasonal and annual variability in wetland water availability as potential predictors of change in waterfowl populations. Furthermore, estimates of the abundance of game birds will require some estimate of the total available wetlands for a region to derive a total regional abundance estimate.

Estimates of wetland availability (i.e. total surface water) can be derived from a number of sources. Direct estimates of surface water can be obtained from Landsat imagery, such as that used for the Global Surface Water project (Pekel et al. 2016). However, that project is complete and no further updates are expected to be forthcoming in future years. Estimates of surface water from remote sensing can be considered the 'gold standard', and development of capability and resources to undertake similar modelling in Australia should be pursued. One possible source of surface water estimates from satellite imagery for Australia is the 'water observations from space', developed by Geoscience Australia http://www.ga.gov.au/scientific-topics/hazards/flood/wofs. Estimates of water occurrence are updated every 3 months, but at the time of writing, only composite data depicting surface water over the entire time period (1987 – present) was available.

An alternative method of estimating regional surface water availability could be by using measurements of precipitation (rainfall). This is likely to be less accurate than direct observations, as many wetland may be replenished from underground aquifers, and thus are not reliant on rainfall. Nevertheless, estimates of total precipitation should be investigated to see how well they can index changes in the extent of surface water. A slightly more sophisticated measure of precipitation is the standardised precipitation evapotranspiration index (SPEI) http://spei.csic.es/. The SPEI is a multiscalar water balance model derived from the difference between precipitation (wetting) and evapotranspiration (drying) events. It has recently been shown that the SPEI index is a good predictor of population change for some game species of ducks (Dundas et al. 2016). Furthermore, the SPEI index can be easily calculated through both space and time, allowing regional and time-specific estimates of SPEI. For example, estimates of the SPEI index at 0.5° resolution were obtained from the global SPEI database http://spei.csic.es/ for the region encompassing the EAAWC. Results for each pixel accumulated over a 12-month period during 2013 revealed relatively dry conditions (SPEI < 0) in south-eastern Australia and parts of central Queensland (Qld) during this year (Figure 6). A composite of the SPEI index over the period 1983–2014 indicates the temporal fluctuations in the SPEI index, which correlated with the total game bird index derived from the EAAWC (Pearson's product moment correlation = 0.40, $P = 0.02$) (Figure 7). Hence, further investigation of the SPEI index as a predictor of wetland water availability is warranted.

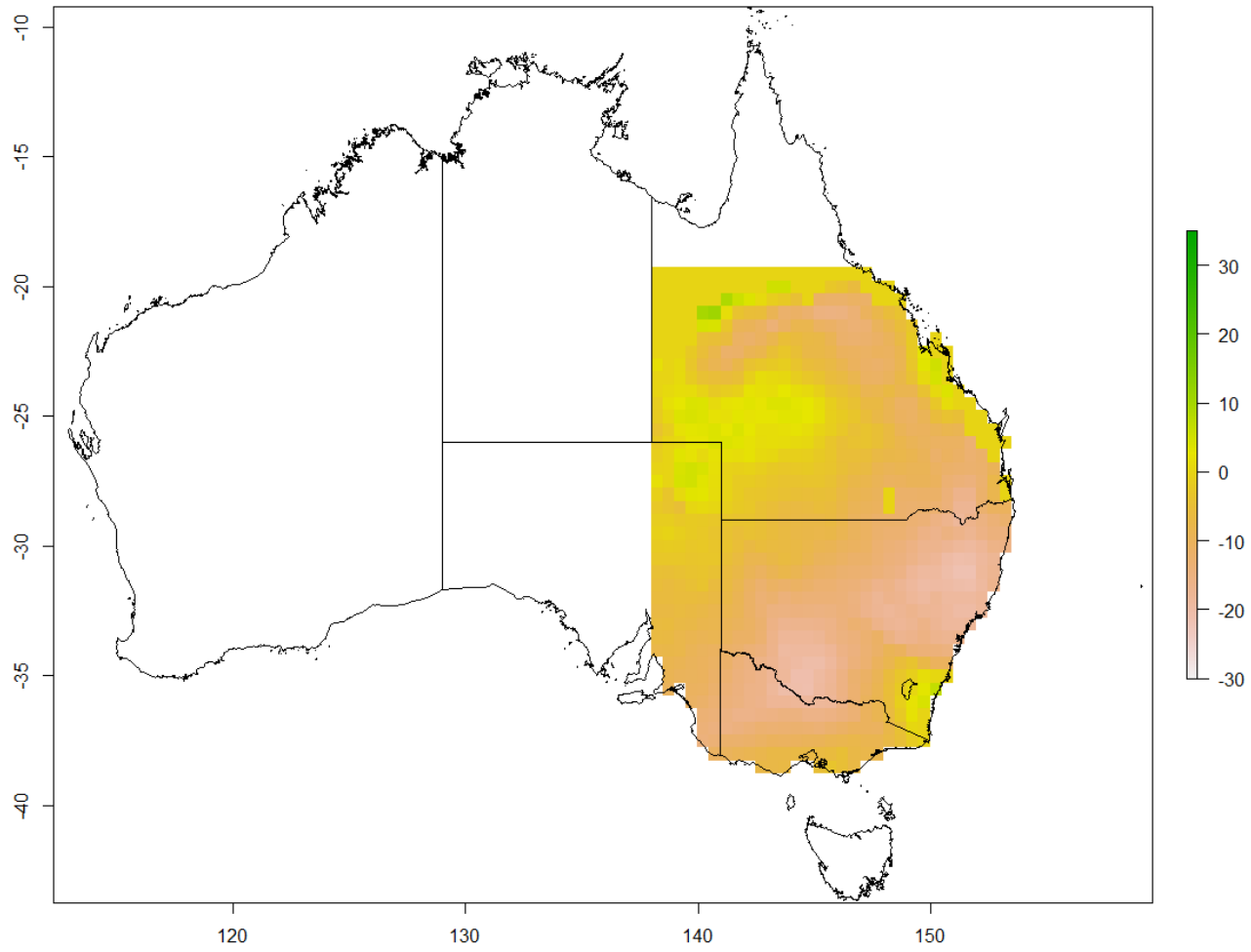

Figure 6. Estimates of the standardised precipitation evapotranspiration index (SPEI) at 0.5º resolution for the region of Australia sampled by the EAAWC during 2013. Values greater than zero indicate relatively wet conditions, and values less than zero indicate relatively dry conditions.

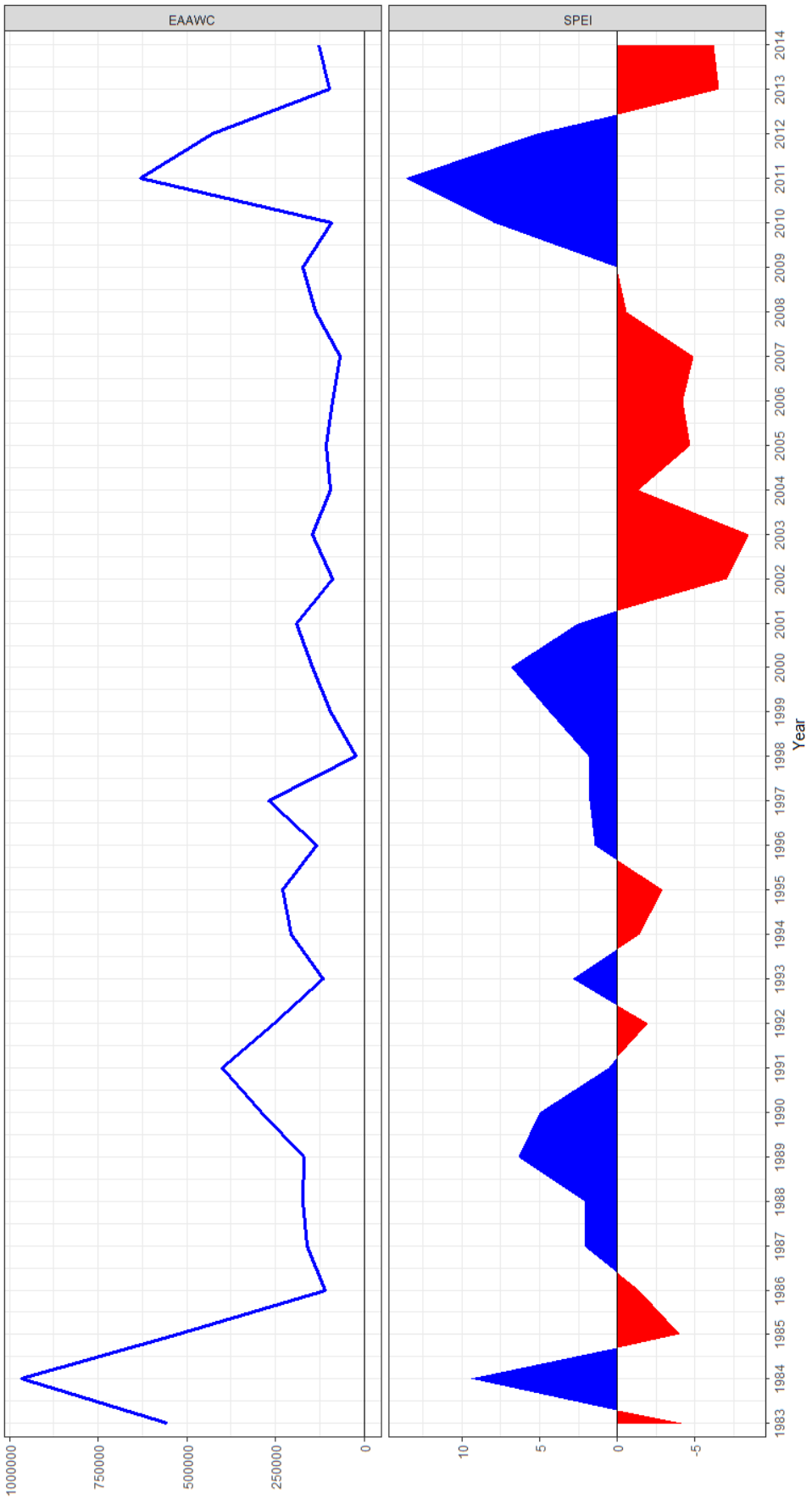

Figure 7. Time series of the index of game bird species sampled by the EAAWC, and the composite SPEI index for the years 1983–2014. Blue values indicate wet conditions and red values indicate dry conditions.

6. Discussion

Long-term declines in river flows in the Murray–Darling Basin (due to increasing regulation and extraction of water, as well as climate change) has seen decreasing trends in the abundances of many waterbird species, including game species. An analysis of 32 years of waterbird abundance data from south-eastern Australia (EAAWC) has shown that reductions in flows of the major river systems in the Murray–Darling Basin has resulted in around 40% less water reaching floodplains and wetlands in the basin, which form critical habitat for waterbirds (Kingsford et al. 2017). Against this background, hunting appears to have a relatively minor impact on most species. However, stronger effects of hunting were seen in some species, such as Australasian (blue-winged) shoveler (*Anas rhynchotis*), hardhead (*Aythya australis*), Australian wood duck (*Chenonetta jubata*) and plumed whistling-duck (*Dendrocygna eytoni*) (Kingsford et al. 2017).

The strong relationships between river flows and waterbird abundances, overlain with the variable impacts of hunting on game species, indicates that management of both environmental water and harvesting impacts should be an integral part of the long-term conservation of the game species of waterbirds. While management of water resources for floodplain and wetland restoration form a central component of a cross-jurisdictional plan for the Murray–Darling Basin (MDBA 2016), there is currently no similar cross-jurisdictional cooperation around the management of waterbird harvests.

AHM offers the potential for a scientific and logical approach to the regulation of waterbird harvests that considers both the direct and cumulative effects of environmental drivers as well as harvest offtake on the long-term sustainability of waterbird populations (Johnson et al. 2015; Johnson et al. 2016). Hence, this review aims to identify a suitable framework with which AHM could be implemented within the south-eastern Australian states that undertake waterfowl harvests. The core of AHM consists of the development of a credible model(s) for how game species of ducks respond to both environmental drivers and harvest offtake. We considered several modelling approaches proposed in the literature for modelling the population dynamics of waterfowl subject to harvest. The proposed approach differs somewhat to that proposed originally in the 2010 Victorian sustainable harvest model report (Ramsey et al. 2010) in that the recommended modelling approach now proposes an unstructured state–space model as the initial base model. Hence, the initial base model is closer to the modelling approach adopted by NSW (Dundas et al. 2016). However, by adopting an Integrated Modelling Approach, we will be able to flexibly adapt this base model to incorporate additional structure (e.g. species- and age-specific survival/recruitment and movement), when suitable data sources become available. The advantages of this approach are that it should be possible to develop a model suitable for AHM relatively quickly, solely with access to abundance monitoring data obtained using a similar approach to that currently used in NSW.

7. Recommendations

7.1 AHM framework

To help reduce contention around using AHM for the setting of harvest regulations, we recommend that ‘triple-loop’ learning be examined as a possible framework for conducting AHM. Hence, in addition to the annual update of the harvest model and regulations, there would also be an annual review and update of management objectives and alternative model structures, as well as of stakeholder ownership and governance processes. Hence, triple-loop learning would ensure that the most appropriate models were within the set

examined by AHM, and also ensure hunter and other stakeholder involvement in the AHM process.

7.2 Model for duck population dynamics

We recommend that a relatively simple, unstructured, Bayesian state–space model be adopted, initially, to predict the outcome of environmental drivers and harvest on game species of ducks. Essentially, this model would be similar to that currently used by NSW for the setting of harvest quotas. Although existing data sources (principally the EAAWC) can be used in this approach, we consider that existing data is currently inadequate for effective AHM, especially for Victoria. Hence, additional data collection and/or modification of existing data sources will be required for effective implementation. These key data requirements/modifications are outlined in Table 1. If the full suite of these data collection requirements is implemented, the corresponding models used in AHM can be modified with additional structure, which should provide a greater degree of accuracy in model predictions.

7.3 Waterfowl monitoring data

This review has identified that the abundance of game duck species should be estimated within a set of predefined bioclimatic regions (e.g. Figure 2), to capture the likely regional variation in waterbird abundance at a tractable spatial scale. If this structure were adopted for AHM, suitable monitoring data would be required for each of the identified regions. We recommend that a desktop study be undertaken to identify the optimal combination of aerial (fixed-wing and/or helicopter) and ground-based surveys that would be required to provide estimates of waterfowl abundance for each region with sufficient precision, for the minimum cost. The desktop design should examine the utility of existing survey data sources (e.g. EAAWC, additional aerial transects, and citizen science bird surveys) for minimising the amount of extra survey effort required in each region. The design should be suitable for estimating the abundance of both mobile and sedentary game duck species.

In addition, ground counts and aerial surveys (fixed-wing and helicopter) need to be undertaken concurrently for a set of wetlands to determine visibility correction factors for the three different survey methods, so that counts from the different sources can be compared equivalently. This should be undertaken for a sample of 100–200 wetlands, stratified by wetland area/type (i.e. 100–200 wetlands per 3 – 4 strata). If correction factors were found to be reliable, then this would only need to be undertaken once, with the correction factors used to calibrate all subsequent survey data. However, if visibility correction factors are found to be unreliable, then monitoring should be undertaken in such a way that counts can be corrected for imperfect detection. This could be achieved using either distance sampling and/or double-observer sampling techniques (Koneff et al. 2008; Ridgway 2010). However, further work is required to overcome the practical challenges in implementing distance and/or double observer aerial surveys of waterfowl.

7.4 Water occurrence data

We recommend that accurate maps of water availability be assembled from satellite imagery annually to estimate the wetland area available for breeding by ducks. Such data sources are now becoming more easily accessible from publicly available repositories (e.g. Google Earth Engine, Geosciences Australia). In addition, an easily calculated surrogate of water availability [e.g. the standardised precipitation evapotranspiration index (SPEI)] should also be investigated for possible use in AHM models and for estimating the occurrence of water in wetlands. (The SPEI can easily be calculated from gridded data on rainfall and evapotranspiration.)

7.5 Harvest data

We consider that the current arrangements for estimating harvest offtake for the game species of ducks in Victoria using telephone surveys is adequate for the purposes of AHM, as the current data collection design is representative of hunters and has enough resolution to apportion harvest by bioclimatic strata. Additional data are required on the harvest regulations (i.e. harvest quota/bag limits/season length) that were in operation during the years when telephone surveys were implemented. This would allow identification of any relationships between regulations and offtake for model predictions as part of AHM. We also recommend that a similar program, based on telephone surveys of hunters, be implemented in both NSW and South Australia to provide a representative sample for estimating the size of the harvest. This could most easily be achieved by expanding the current Victorian program to those states. Further work is required to estimate the possible size of any reporting bias (i.e. over- or under-reporting of the number of birds harvested) present in hunter surveys.

7.6 Additional data required to increase predictive power of AHM models

The above recommendations are the minimum requirements for implementing effective AHM in south-eastern Australia. We also recommend that additional data collection be considered and gradually incorporated into the AHM program to increase the predictive accuracy of the models (Table 1).

Harvest bag surveys

Harvest bag surveys can provide data on age/sex ratios, which can be used to estimate juvenile recruitment/productivity. However, the current harvest bag surveys are unrepresentative and require a robust sampling design before the data can be used for AHM. We recommend that a new harvest bag survey be designed that uses stratified random sampling of wetlands within bioclimatic zones to provide unbiased and precise estimates of age ratios for each of the game species. We note that a guide to ageing birds more accurately is currently being produced, and the findings from that project should be incorporated into any future survey redesign.

Duck survival and movements

If AHM moves to a more structured model of duck population dynamics, additional data will be required for estimating the survival and movement of ducks. We recommend the implementation of studies involving the satellite/GPS telemetry of individual ducks, for estimating age- and sex-specific survival rates as well as movement between each of the bioclimatic zones. These could be supplemented by duck-banding programs. Telemetry/banding programs should be undertaken, ideally, in each bioclimatic zone, or at a minimum in zones with contrasting bioclimatic conditions.

Table 1: Monitoring data requirements for the effective implementation of AHM, subdivided into immediate, mid-term and long-term requirements

Immediate requirements
1. Survey design: A desktop study should be undertaken to determine the optimal mix of ground and aerial surveys (fixed-wing and/or helicopter) required for estimating game duck species abundances within each bioclimatic region in south-eastern Australia. Survey design should use the best mix of additional monitoring required for supplementing existing aerial and ground monitoring sources.
2. Monitoring data calibration: To calibrate monitoring data from different sources, aerial and ground counts should be undertaken concurrently for a stratified random sample of wetlands to estimate visibility correction factors. These would then allow monitoring data from different sources to be used in AHM models as if they were from a single source. However, if visibility correction factors prove to be unreliable, then, double-observer and/or distance sampling techniques should be investigated for both aerial and ground-based monitoring to correct for visibility bias.
3. Wetland availability: Assessment of wetland availability within each bioclimatic stratum is required to estimate total game duck population abundance from survey data. Accurate assessment will require the analysis of remotely sensed data (e.g. LandSat) and should be considered a high priority.
4. Harvest estimates: Current arrangements in Victoria are considered adequate. It is recommended that telephone surveys, similar to those undertaken in Victoria, be undertaken in other jurisdictions (e.g. NSW, South Australia) to provide a representative sample of hunters for estimating harvest offtake. This could be achieved by expanding the current Victorian program to those states.
Mid-term requirements
5. Redesign existing harvest bag surveys to provide a representative sample of the age/sex composition of the harvest.
Longer-term requirements
6. Implement satellite telemetry studies for the game species of ducks to provide high-resolution data on movement as well as age- and sex-specific estimates of survival. Ideally these should be undertaken across NSW, Victoria and South Australia.
7. Implement banding programs to estimate survival and movement of ducks as a complement to telemetry studies.

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Appendix. Review of models of waterfowl population dynamics

Non-spatial models

Non-spatial, structured population models have been developed for egrets (*Ardea* spp.) in the Murray–Darling basin (Arthur 2011), greylag geese (*Anser anser*) in Scotland (Trinder et al. 2010) and blue ducks (*Hymenolaimus malacorhynchos*) in the North Island of New Zealand (Simpkins et al. 2015). These models were structured models, typically describing two to three age/stage classes (i.e. juveniles, subadults, adults) with survival rates for the different age classes usually estimated from banding data. Density dependence in the population vital rates (productivity/survival) was assumed only for the greylag goose population. Both the egret and blue duck models were developed primarily to determine the effects of environmental variation on populations [through effects on survival (blue ducks) or productivity (egrets)] to conduct population viability analyses. In general, the modelled populations were assumed to be closed to movements.

Non-structured population models have been developed for North American mallards (Lillegård et al. 2008) using a state–space approach. These models assume that all individuals are subject to the same probabilities of survival and reproduction, and hence that there is no age or stage structure. Moreover, these scalar models usually do not separate survival and productivity parameters, but are used to estimate the composite of these, the intrinsic rate of increase. Hence, these models are primarily used for estimating the effects of density dependence and of other extrinsic environmental variables (including harvest) on the population growth rate. They are usually fitted to time series of population count data and assume that the counts arise from an imperfect observation process, and are then used to infer the underlying (unobserved) state process (i.e. the true population abundance). State–space models were fitted to the time series of aerial count data for North American mallards over a 45-year period for 50 aerial segments using a Bayesian approach (Lillegård et al. 2008). The aerial counts were assumed to be subject to imperfect detection (negative bias) and were corrected using an observation model involving concurrent ground counts. The ground counts were assumed to be a precise estimate of the true population size, and an observation model was developed using the relationship between the aerial counts and the concurrent ground counts. This relationship took the form:

$$\sqrt{Y} \sim N\left(a + b\sqrt{N}, \sigma\right),$$

where $\sqrt{Y}$ is the square root of the aerial counts, $\sqrt{N}$ is the square root of the ground counts, and a, b and σ are the estimated parameters. This observation model was estimated from the subset of aerial counts that had concurrent ground counts, and was then used to correct the entire set of aerial counts.

Spatial models

A spatially explicit, stage-structured model of the double-crested cormorant (*Phalacrocorax auritus*) in North America was developed to explore the efficacy of different management strategies (Guillaumet et al. 2012). The area of interest was divided into discrete cells, with each assumed to be occupied by a single breeding colony. The dynamics of each breeding population consisted of three age classes and contained expressions for density-dependent recruitment. Individuals within each local colony could potentially disperse to other colonies, with the rate of dispersal dependent on both the direction and the distance between colonies. Colonies could also be subject to management, principally by

destroying eggs (egg oiling), rather than culling individuals directly. No explicit links between local demography or dispersal and environmental variables were included in the model.

A similar spatially explicit, stage-structured model was developed to examine the possible effects of future climate change on the population dynamics of magpie geese (*Anseranas semipalmata*) in the Northern Territory (Traill et al. 2010). It was hypothesised that future climate change would result in sea level rises leading to the inundation of key wetland habitats critical for magpie geese. Thus, the objective of the modelling was to estimate the effects of habitat loss due to sea-level rise on the population viability of the geese. The model included four age classes, and density-dependent survival and fecundity rates. Hence, it was assumed that the population would follow a logistic growth model. Spatial structure was included by modelling local populations distributed over 12 major river–wetland systems of the Northern Territory, with dispersal between wetlands being a function of distance. Abundance and carrying capacity for each wetland used data from previous aerial surveys, corrected for imperfect detection using aerial photography (Brook and Whitehead 2005). Recreational harvest of magpie geese was also included in the model, with 30,000–150,000 birds being harvested per year (harvest rates varying from wetland to wetland). The model was constructed in RAMAS Metapop v5, a general-purpose, flexible ecological modelling platform (Lindenmayer et al. 1995).

Models incorporating harvest as part of AHM

North American mallards

The AHM program for mallards contained four alternative models that shared a common underlying structure

$$N_{t+1} = N_t\{pS_t^{AM} + (1-p)[S_t^{AF} + R_t(S_t^{JF} + S_t^{JM})]\}, \qquad \text{Eqn. 6.}$$

where N_t is the population size at time *t*, p is the proportion of males in the population, and $S_t^{AM}, S_t^{AF}, S_t^{JM}$ and S_t^{JF} are the adult male, adult female, juvenile male and juvenile female annual survival rates at time *t*, respectively. The time-specific parameters S_t and R_t are, in turn, functions of harvest and environmental variation, respectively. For example, the annual survival of adult male ducks during year *t* is

$$S_t^{AM} = \phi\beta_t,$$

where ϕ is the natural mortality rate and β_t is the adult male harvest rate, which is given by:

$$\beta_t = 1 - \frac{h_t}{1-c},$$

where h_t is the adult male harvest rate and *c* is the crippling loss. This formulation assumes that mortality due to harvest is additional to natural mortality. Compensatory mortality models are formulated by making β_t a function of the ratio of harvest to natural mortality, with harvest mortality being completely compensated for if it is less than the natural mortality rate (Johnson et al. 1997). Similarly, the time-specific recruitment rate R_t is a function of both mallard abundance at time *t* (N_t), but also the number of ponds in the Prairie Pothole Region of North America during the breeding season:

$$R_t = (a_0 + \alpha N_t) + \gamma E_t,$$

where α_0, α and γ are the effects of mallard abundance and pond abundance (E_t) on the recruitment rate. It is of note that Equation 6 only has one state variable (*N*), and hence does not have any additional age or spatial structure, apart from the use of female

juvenile/adult age ratios required for recruitment estimation. Hence, waterfowl for the entire Prairie Pothole Region are assumed to be one homogeneous population. Since no surveys are conducted outside the Prairie Pothole Region of North America, AHM is assumed to apply to the breeding population only, even though harvest can occur outside this region.

Svalbard pink-footed goose

In 2012, AHM was formally adopted for populations of migratory pink-footed goose (*Anser brachyrhynchus*) subject to harvest in Norway, Sweden, Denmark and the Netherlands (Madsen et al. 2017). This was the first implementation of AHM for a migratory waterbird population in Europe. The move to AHM for this species was mainly driven by the need to mitigate agricultural impacts driven by an increase in goose numbers from around 10,000 in the 1960s to around 80,000 birds currently (Johnson et al. 2013). Due to the longevity of geese and the fact that sexual maturity usually does not occur until 3 years of age, a more structured population model was used, containing three age classes; juvenile (0.5 years), subadult (1.5 years) and adult (2.5 years). In most other respects, the base model was similar to that used for North American mallards. A suite of nine alternative models was used to characterise the structural uncertainty around the effects of population density and environmental variation on the population dynamics of the geese. Three models were used to describe variation in natural mortality rates, including density dependence and the effects of environmental variables. No compensatory harvest mortality model was developed for pink-footed geese because examination of existing data suggested a lack of evidence for compensation. Three models for the recruitment of geese were also developed, in which recruitment was a function of either temperature (i.e. number of days above freezing during May), density dependence or density independence (Johnson et al. 2013). As for the mallard model, there was no spatial structuring of the population, with a single abundance figure being estimated for the population in each year.

NSW waterfowl quota

The Game Licensing Unit of the NSW Department of Primary Industries (DPI) manages the culling of native game birds in order to help manage the impacts of waterfowl on agriculture, principally rice crops. Up to ten species of ducks are authorised for culling under the damage mitigation scheme. Each year an index of the duck numbers of each species is analysed and a quota set. For species that respond predictably to environmental conditions, and whose predicted population size is greater than 50,000 birds, an upper harvest quota of 10% is set. For species that do not respond predictably to environmental conditions, and which have a lower predicted population abundance, a reactive quota is used. Species on the reactive quota are only harvested on those properties deemed to be at significant risk of damage from those species.

Population estimates used to define quotas for each species are derived from a Bayesian state–space model fitted to data from the EAAWC (Kingsford and Porter 2009). The model takes the following form:

$$N_{t+1} = N_t \lambda_t \qquad \text{Eqn. 7}$$

$$\log(\lambda_t) = \beta_0 + \beta_1 \log(N_t) + \beta_2 R_t + \beta_3 SOI_t + \beta_4 SPEI_t,$$

where N_t is population size at time t, β_0 is the intrinsic rate of population increase, β_1 is the effect of (log) density on the population growth rate, β_2 is the effect of rainfall in the previous 12 months, β_3 is the effect of the average southern oscillation index over the previous 12 months and β_4 is the effect of the standardised precipitation evapotranspiration index (SPEI) over the previous 12 months. Equation 7 essentially describes a Gompertz model of population growth, where the population growth rate

depends on the (log) previous population size (Dennis et al. 2006). To complete the estimation of the harvest quota, the estimated abundance derived from the count data are scaled up to an estimate for the entire NSW region by using an estimate of the total wetland area and the proportion of wetlands sampled by the EAAWC. The best estimates of future population size were predicted by the SPEI. The SPEI is an index of drought conditions calculated from the difference between the total precipitation (rainfall) and the potential evapotranspiration to form a simple water balance model (http://sac.csic.es/spei/home.html). The model developed for NSW waterfowl has a simple structure, with both survival and recruitment incorporated into a single parameter, the rate of increase (λ). In addition, there was no spatial structure to the model, with a single abundance figure estimated for each species for all of NSW.

Victorian waterfowl model

The model for AHM that was proposed for Victorian waterfowl was developed by Ramsey et al. (2010). The main issues that influenced the development of the modelling approach included:

1. the high mobility of some game species: some species of game duck are highly mobile [e.g. grey teal (*Anas gracilis*)], sometimes travelling hundreds of kilometres in a few days to exploit ephemeral water sources. This is challenging from a population dynamics perspective, because the spatial scale of any model must include extensive areas outside Victoria, essentially including most of eastern Australia and parts of central Australia.
2. Waterfowl population dynamics are primarily driven by the availability of wetlands. Hence, the filling and drying of major wetland complexes (or a proxy thereof) would need to be a key component of any model.

Given these considerations, the model proposed for the Victorian AHM was a stage-structured, spatial metapopulation model. The proposed model essentially divided up the area of interest (i.e. eastern and central Australia) into spatial units. Within each spatial unit, a structured population 'base' model would be used to describe the population dynamics (Figure 1). This base model had two stage classes (juveniles and adults) for both males and females, and parameters describing annual survival and recruitment for each stage/sex class. Base models for each spatial unit would be linked to other spatial units by dispersal, with the magnitude and direction of dispersal in any year being driven by environmental conditions related to the wetland area within each unit (or an index thereof).

The life cycle for the base model can be written mathematically as in Figure 8:

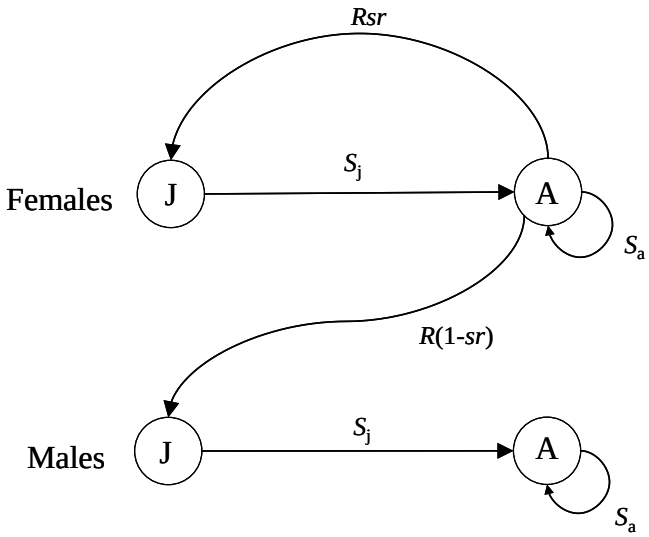

Figure 8. Life cycle diagram describing the simple stage-structured base model for a waterfowl population inhabiting a single wetland 'node'. S_j = annual juvenile survival rate; S_a = annual adult survival rate; R = recruitment rate (juveniles per adult female); sr = sex ratio of recruits.

$$\mathbf{n}_{(t+1)} = \mathbf{A}\mathbf{n}_{(t)} \,, \qquad \text{Eqn. 8}$$

$$\mathbf{n} = \begin{bmatrix} n_{jf} \\ n_{af} \\ n_{jm} \\ n_{am} \end{bmatrix}; \qquad \mathbf{A} = \begin{bmatrix} 0 & Rsr & 0 & 0 \\ S_j & S_a & 0 & 0 \\ 0 & R(1-sr) & 0 & 0 \\ 0 & 0 & S_j & S_a \end{bmatrix},$$

where $\mathbf{n}$ represents the vector of abundances for each stage/sex class and $\mathbf{A}$ is the projection matrix containing the transition rates between stages. Equation 8 can be generalised to multiple spatial units by incorporating the dispersal probabilities between each unit. For example, if there were two units, then the state of the metapopulation can be described by the following vector:

$$\mathbf{N}_t = \begin{pmatrix} \text{unit 1} \\ \text{unit 2} \end{pmatrix} = \begin{pmatrix} n_{11} \\ n_{21} \\ n_{31} \\ n_{41} \\ n_{12} \\ n_{22} \\ n_{32} \\ n_{42} \end{pmatrix},$$

where the $n_{i,j}$ are the abundances of stage i in unit j at time t. The transition matrix $\mathbf{B}$ is then constructed from the transition matrices for each unit and the corresponding dispersal matrix $\mathbf{M}$:

$$\mathbf{B} = \left(\begin{array}{c|c} \mathbf{A_1} & \mathbf{M_{2\to 1}} \\ \hline \mathbf{M_{1\to 2}} & \mathbf{A_2} \end{array} \right),$$

where $\mathbf{A}_1$ and $\mathbf{A}_2$ are the transition matrices for units 1 and 2, respectively, and $\mathbf{M}$ is the dispersal matrix (Ramsey et al. 2010). Harvest is incorporated into the model separately for each unit, expressed as the proportional offtake for each stage/sex class:

$$\mathbf{n}_{(t+1)} = \mathbf{A}\mathbf{n}_{(t)} - \mathbf{H}\mathbf{n}_{(t)},$$

where $\mathbf{H}$ is a matrix giving the harvest rate for each sex/stage class on the diagonal and zero elsewhere. For spatial units where no harvesting occurs (e.g. refuges), the elements of $\mathbf{H}$ are all set to 0.

Integrated population models

Traditional approaches to modelling wildlife populations usually involve constructing a model based on assumptions about the system, followed by fixing model parameters by analysing available data, usually piecemeal. A major limitation of that approach is that there is usually no formal way of accounting for uncertainty in all the various inputs (Thomas et al. 2005). A relatively new modelling approach for estimating population dynamics that makes use of multiple, independent data sources in the one unified framework has emerged in recent years. Integrated population models (IPMs) (Schaub and Abadi 2010) utilise population count (survey) data as well as demographic data (e.g. band recoveries) that have been collected independently in the one model to estimate their joint likelihood. The central component of an IPM is a structured population model (i.e. a projection matrix—Equation 8), where the demographic parameters of the model are estimated from the joint likelihood from the combined data sources. Traditionally, parameters for population models (e.g. structured population models) have been analysed separately, with the population count data used to estimate population growth rates, and the demographic data used to estimate survival rates for different age/stage classes. Other data on productivity/recruitment are also collected and analysed separately. For example, the AHM model for North American mallards was parameterised using separate analyses of independent data sources (e.g. survey data, band return data and hunter bag data for age ratios). This traditional method of model building has several drawbacks: (i) it requires separate analyses, and does not use all the available information efficiently, (ii) each separate analysis has associated uncertainty and there is no formal way to account for this in the final model and (iii) if some of the information on demographic rates is incomplete, estimates of population growth can be biased (Schaub and Abadi 2010). IPMs overcome these drawbacks because each separate data source is included in the one model; hence, the uncertainty associated with each source is properly accounted for. Demographic parameters are also usually estimated with greater precision compared with when estimation is undertaken on each data source separately. Additionally, demographic parameters for which there is no associated data can often be estimated using an IPM. This is because each data source is exploited more efficiently using a joint analysis as population count data contain information not only on population growth, but also on demographic rates (survival/recruitment). In turn, mark–recapture and band recovery data also contain information on population growth rates. Hence, combining both of these data sources in the one analysis leads to more efficient and precise estimation of all demographic parameters (Tavecchia et al. 2009; Abadi et al. 2010, 2012).

Another advantage of IPMs are their ability to model population processes over large spatial scales (Robinson et al. 2014; Chandler and Clark 2014). Capture–recapture studies are often expensive and hence are usually conducted over small spatial scales. In contrast, population count or presence/absence data are often collected over large spatial scales. Combining both data sources allows the results of small-scale studies to be ‘scaled up’ to make inferences at large spatial scales. An example of this comes from a study of data collected from large-scale surveys of birds in the United Kingdom (Robinson et al. 2014). Using population count data collected by volunteers, as well as smaller-scale observations of nest/fledging success along with some band recoveries, the population dynamics of 17 passerines could be reconstructed over periods ranging from 18 to 46

years. In addition, rigorous inferences could be made on unobserved quantities, such as unobserved age/stage classes or immigration rates (Schaub and Fletcher 2015).

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