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Citizen scientists reveal nationwide trends and drivers in the breeding activity of a threatened bird, the malleefowl (*Leipoa ocellata*)

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ABSTRACT

Citizen scientists regularly collect monitoring data for threatened species to improve the spatial and temporal resolution of sampling. Such programs should adopt robust data assurance measures and statistical approaches to reduce observer bias and better inform uncertainty estimates while supporting management decisions. In this study, we estimated trends and drivers of malleefowl (*Leipoa ocellata*) breeding activity within a Bayesian hierarchical modelling framework using 1823 site × years of nest count data collected by volunteers in Australia. Our modelling suggests malleefowl breeding activity decreased 4.8% annually in South Australia (−0.050; 95%CIs −0.062, −0.037), decreased 2.1% annually in Western Australia (−0.022; 95%CI −0.040, −0.004), was stable in Victoria (−0.001; 95%CI −0.010, 0.009) and increased 4.8% annually in New South Wales (0.047; 95%CI 0.009, 0.086). We found strong evidence for positive associations between winter rainfall (0.084; 95%CI 0.004, 0.165), time since fire (0.288; 95%CI 0.179, 0.399) and an interaction between time since fire and the proportion of a site burnt (0.292; 95%CI 0.173, 0.410). Malleefowl breeding activity was negatively associated with patch size (−0.255; 95% CI −0.642, 0.020) and the proportion of a site burnt (−0.191; 95% CI −0.363, −0.030), suggesting small reserves are important for conservation and the extent and frequency of fire should be managed cautiously. While our index of fox abundance decreased as baiting effort increased (−0.484; 95%CI −0.640, −0.317), there was little evidence for this benefiting malleefowl. This study demonstrates how volunteers can play a vital role understanding population trends and informing conservation of a threatened species at a national scale.

1. Introduction

Monitoring is crucial to conservation because it informs the status and trends of populations, determines whether management interventions are effective, and can raise much needed political and community support (Possingham et al., 2012). However, effective monitoring should: have clear, well-defined objectives (Scheele et al., 2018); be designed with adequate statistical power to detect levels of change that are significant from a conservation standpoint (Southwell et al., 2019); provide appropriate data consistently through space and time (Likens, 1989); be sustained long-term and over an appropriate geographic scale (Lindenmayer and Likens, 2018); have the capacity to store and analyse the data collected (Robinson et al., 2018), and; produce results that inform management decisions (Lindenmayer and Likens, 2018). Although these steps are well-recognised, many species are not monitored at all, and for those that are, the quality of monitoring is often poor (Legge et al., 2018; Scheele et al., 2019).

Incorporating citizen scientists into biodiversity monitoring programs is an increasingly popular way to overcome scale and cost barriers to successful monitoring. Citizen scientist programs are generally designed to use simplified, standardised protocols repeated across many monitoring sites to gather data across large temporal and/or spatial scales (Dickinson et al., 2010). This allows data to be collected at reduced cost, vastly improving power to detect ecological patterns at large-scales, such as broad-scale population trends (Barlow et al., 2015), species range shifts (van Strien et al., 2013) and patterns of migration (Hurlbert and Liang, 2012). As a result, the number of citizen science programs has increased substantially in recent decades, with hundreds now operating around the globe to track the status and trends of a range of taxa, including butterflies, reptiles, mammals and birds

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(Dickinson et al., 2010; Penone et al., 2013).

However, citizen scientist monitoring programs do not overcome all barriers to effective monitoring. Relying on volunteers with no specialist training or experience can introduce additional challenges that must be considered for reliable inference and informed decision-making (Bird et al., 2014). For example, the population metric measured is usually limited to something that does not require specialised training, knowledge, equipment or time (Snall et al., 2011); monitoring can be biased towards more convenient locations, times or where biodiversity is known to be high (Betts et al., 2007), and; the influence of detectability, measurement errors and uncertainty can be inflated (van Strien et al., 2013). When designing and implementing citizen science monitoring programs, such issues must be considered, yet few citizen science programs have rigorous data quality assurance measures and robust statistical methods to appropriately deal with observer error, incomplete data, and uncertainty (Bird et al., 2014).

The National Malleefowl Monitoring Program (NMMP) has been specifically designed with these issues in mind so that reliable data may be collected entirely by a large number of citizen scientists across the species range. Malleefowl (*Leipoa ocellata*) is a threatened ground-dwelling bird, distributed in low densities across vast areas of semi-arid Australia (Benshemesh et al., 2018). As shy, elusive birds, malleefowl are cryptic and difficult to observe in their environment, making direct monitoring of their population size or abundance difficult. However, malleefowl are megapodes and build large conspicuous mounds (up to 90 cm high and typically 4 m wide) composed of sand and organic matter to incubate their eggs (Jones et al., 1985). The NMMP has been purposely designed to measure mound activity as a surrogate of the breeding population size, as well as the presence or absence of predator signs, so that monitoring is relatively cheap, easily repeatable and does not require specialist skills or experience (Benshemesh et al., 2018).

Under the guidance of the National Malleefowl Recovery Team, the NMMP has grown from only a few monitoring sites in the late 1980s to over one hundred sites in 2017 sampling much of the species' range across southern Australia (Benshemesh et al., 2018). Most of the early monitoring was conducted by consultants and government agencies; however, it is now maintained almost entirely by citizen scientists who collectively undertake all aspects of the program, from organising and collecting field data to vetting and validating these data in preparation for analysis. This expansion has fostered a national collaboration among citizen scientists, land management agencies, and academic researchers, making it now one of the most comprehensive citizen science monitoring programs in Australia, and one of only a few to provide nationwide population trends for a threatened species spanning multiple decades (Legge et al., 2018).

The standardised collection and flow of mound activity data by citizen scientists has previously provided valuable insight into malleefowl population status and trends. For example, an analysis of NMMP data from 1990 to 2005 suggested breeding activity was in decline nationally, with winter rainfall and time since fire positively associated with breeding activity (Benshemesh et al., 2007; Walsh et al., 2012). These studies also found no evidence that 1080 baiting (sodium fluoroacetate) for foxes benefited breeding activity, even though baiting suppressed indices of fox abundance. This result is surprising given that baiting is the most common management strategy for the species, and because experimental research has shown that predator baiting can increase the survival of captive-reared malleefowl in the younger life-stages (Priddel and Wheeler, 1990, 1997). Such insights have often challenged assumed knowledge and provided impetus to continue monitoring and to resolve management uncertainties.

The NMMP is also closely tied with policy and is informing active conservation of the species. The NMMP has drawn local attention to the plight of malleefowl across its range and motivated land managers to address clear threats to the species, particularly in regard to habitat destruction, fire management, and predator and competitor control. Previous analyses of NMMP data, such as the studies by Benshemesh et al. (2007) and Walsh et al. (2012), have formed the basis for successive National Malleefowl Recovery Plans, which outline strategies and priorities for conserving the species (Benshemesh, 2007). Uncertainty about the effectiveness of predator control for malleefowl was also uncovered by these studies (Benshemesh et al., 2007; Walsh et al., 2012), and has been the primary motivation for a national adaptive management experiment that is currently being established alongside the NNMP. In this experiment, predators are being managed at spatially replicated control-treatment monitoring sites to learn more about relationships between predator control, predator activity and malleefowl (Bode et al., 2017; Hauser et al., 2019).

Given the threat status of malleefowl, uncertainty surrounding the species' response to management alternatives, and growth of the NMMP in recent decades, a revised national assessment of NMMP data is urgently required using methods that account for biases and uncertainty in mound counts. In this study, we assessed trends and drivers in malleefowl breeding activity using a 28-year mound activity dataset collected primarily by citizen scientists across four states of Australia. Firstly, we quantified the growing contribution of citizen scientists to the NMMP by reporting the number of mounds monitored over time in each of the four states. Secondly, we quantified state-wide trends in malleefowl breeding activity while accounting for uncertainty in mound activity data using interval censoring methods in a Bayesian hierarchical framework. Finally, we quantified the influence of site characteristics and predator baiting on population trends to inform future management decisions. In doing so, we demonstrate how citizen scientists can contribute to comprehensive assessments of the status and trends of sparsely distributed threatened species. We provide insight into a successful monitoring program where data are collated, processed and analysed within a robust statistical framework for informed monitoring decisions.

2. Materials and methods

2.1. Study species

The malleefowl (*Leipoa ocellata*) is an iconic, ground-dwelling bird, sparsely occupying the dry eucalypt and acacia shrublands of semi-arid Australia. Malleefowl are mound-builders (megapodes) and reuse old mounds; typically about 1 in 10 mounds are used for breeding in a given year. Its geographic range has contracted considerably since the arrival of Europeans due to clearing of the wheat belts for agriculture (Benshemesh, 2007). Malleefowl continue to be threatened by introduced foxes and cats, further habitat loss, changed fire regimes, and over grazing by introduced and/or native herbivores (Benshemesh, 2007; Hauser et al., 2019). The species is listed as threatened by state and federal governments (Department of the Environment, 2010) and is one of 20 priority birds identified in the Australian Government's Threatened Species Strategy (Department of the Environment and Energy, 2015).

2.2. Data collection

Mound data were collected by pairs or small groups of volunteers between October and January each year when breeding birds tended their mounds. Within each site (typically 2 × 2 km), volunteers navigated to all known mounds using a system of gridded markers (pre-2001) or GPS (post-2001) and classified each as being active or not based on evidence of use that year. Given that foxes were introduced to Australia and are the cause of many declines in native species (Dickman, 1996), the presence or absence of fox scats on mounds was recorded as an index of fox abundance (Webbon et al., 2004). Since the early 2000s, most data were recorded on digital devices using the CyberTracker application (http://www.cybertracker.org) and digital photos were taken of each mound to aid data quality processing and analysis.

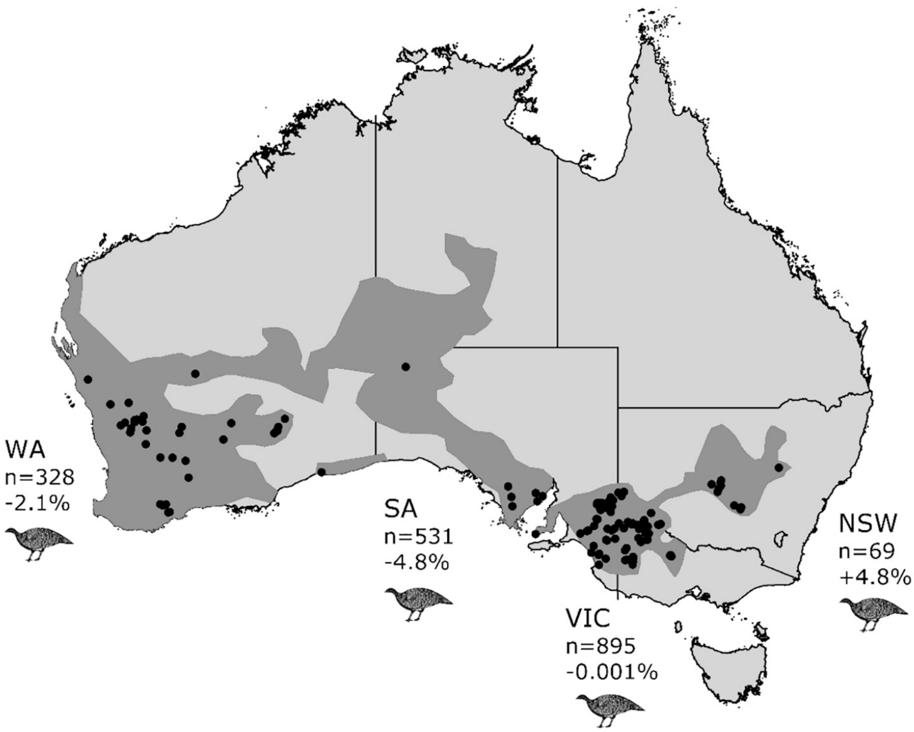

Fig. 1. Location of the long-term malleefowl monitoring sites (black dots) distributed across the species' historic range (dark grey shading) in Western Australia, South Australia, Victoria and New South Wales with the estimated trend and number of site-years of monitoring data for each state. The species is considered extinct in the Northern Territory.

Table 1
Description of covariates used in the Malleefowl trend analysis and reference to previous studies where they have been found to be important drivers of breeding activity.

Covariate	Definition	Reference
Time since fire	The number of years since the last fire at a site. Many studies suggest malleefowl prefer long-unburnt vegetation.	Benshemesh et al. (2007) Clarke (2005) Connell et al. (2017)
Proportion of area burnt	The proportion of a site burnt in the last known fire. Large fires are thought to have a greater effect of malleefowl than smaller fires	Benshemesh et al. (2007) Clarke (2005) Connell et al. (2017)
Patch size	The size of the habitat patch containing the monitoring site	Benshemesh et al. (2007)
Rainfall	Winter rainfall has a positive effect on Malleefowl breeding activity. Low rainfall during the winter period can reduce body condition and egg production, as well as influence the rate of decaying matter in mounds.	Benshemesh et al. (2007) Frith (1959) Priddel and Wheeler (2003)
Bait intensity	Baiting with 1080 is a common method for conserving Malleefowl in Australia. However, the effect of baiting on malleefowl population dynamics is highly uncertainty. The number of baits set over a 100 km^2 area was the measure of bait intensity.	Benshemesh et al. (2007) Walsh et al. (2012)
Fox scats	Foxes are known to prey on malleefowl, particularly juveniles, and to defecate on mounds for territorial marking. We used the proportion of inactive nests with scats as an index of fox abundance at sites. Our index only included inactive nests because active nests attract foxes confounding the index.	Benshemesh et al. (2007) (Priddel and Wheeler, 1990, 1997, 2003)

2.3. Data collation

We collated mound activity data from the National Malleefowl Monitoring Database (NMMD) (Benshemesh et al., 2018) for 127 sites collected between 1989 and 2017 in Victoria (895 site-years), South Australia (531 site-years), Western Australia (328 site-years) and New South Wales (69 site-years) (Fig. 1). We only included sites that had been monitored on at least two occasions. We extracted the total number of people or organisations who uploaded mound monitoring data to the program since it started in 1989 and the number of mounds surveyed each year.

2.4. Site characteristics

We collated information on the site covariates thought to influence malleefowl breeding activity. We calculated the cumulative winter rainfall at sites for each year of monitoring by obtaining 5 km gridded mean monthly rainfall estimates from the Australian Water Availability Project (Jones et al., 2009). Monthly rainfall estimates were summed from May to August as this period precedes the malleefowl egg laying season (influencing whether individuals decide to breed) and effects the abundance of herbaceous resources needed for egg production (Frith, 1959).

We calculated the size of the vegetation patch and the proportion of cleared land within a 5 km radius of monitoring sites using Google Earth or ArcGIS software. A radius of 5 km was selected to match a similar study by Benshemesh et al. (2007) and the reported home range of malleefowl (Booth, 1987). We recorded two descriptors of the fire history at sites from 1980 to 2017: the year each site burnt (i.e. time since fire), and; the proportion of the site that burnt. We calculated both time since fire and the proportion burnt by inspecting Google Earth historical imagery, and where possible, screened our fire history estimates with on-ground observations by land managers.

We collated information from government agencies and community groups on the baiting effort implemented at sites to control for foxes. There was considerable variation in the extent, frequency, timing, intensity and bait type over space and time. To collapse these variables into a single measure, we estimated the density of baiting within a 100 km^2 area centred over monitoring sites (Benshemesh et al., 2007). This area was chosen to cover the home range of foxes, which is needed for effective fox control (Saunders and McLeod, 2007). Further details and justification for our choice of site covariates is presented in Table 1.

2.5. Data processing and censoring

All mound descriptions and activity classifications were inspected by experienced personnel overseeing the NMMP after each year of monitoring. Where mound descriptors were not in complete agreement with the activity classification, past, current and future photographic records (available since 2006) were examined to determine whether it was likely that a mound had been misclassified by volunteers. Occasionally, the activity status of mounds was corrected if it was clear from their description and photo record that the original classification was in error.

In some cases, the number of active mounds was not recorded with certainty because either: 1) a site was not monitored that year; 2) some mounds were not visited, or; 3) data were otherwise incomplete. We included uncertain counts in our analysis using interval censoring (Fox, 2015). This required selecting a plausible lower bound for an uncertain site based conservatively on the known minimum number of active mounds, and an upper bound by adding the number of mounds that may have been active to the lower bound based on past and subsequent photo data. When a censoring interval could not be selected with confidence, we set the lower bound to zero and the upper bound to 100, well above the plausible upper limit.

2.6. Analysis

We modelled the number of active mounds using a Poisson regression model within a Bayesian hierarchical framework. The observed number of active mounds y_{ij} in site i on yearly visit j was modelled as independent Poisson random variables conditional on their parameter λ_{ij}:

$$y_{ij} \sim Pois(\lambda_{ij}) \tag{1}$$

where λ_{ij} represents the expected count of active mounds. For site-years where mound counts were not observed with certainty, a lower and upper bound of the number of active mounds was given, and the true number of active mounds was assumed to lie within that censoring interval (Fox, 2015). The effect of covariates on mound activity was expressed by modelling the natural log of the expected mound count as a linear function of effects:

$$\begin{aligned} ln(\lambda_{ij}) = {} & \alpha S_i + \beta_{1-4} S_i \mathrm{j} + \sum_{h=0}^{4} \beta_{5+h} \ln\left(\frac{\pi_{ij-h}}{1-\pi_{ij-h}}\right) + \sum_{h=0}^{4} \beta_{10+h} R_{i,j-h} \\ & + \beta_{15} P_i + \beta_{16} C_i + M_{i,j}(\beta_{17} TF_{i,j} + \beta_{18} PB_{i,j} + \beta_{19} TF_{i,j} PB_{i,j}) \\ & + reS_i + reY_{i,j} \end{aligned} \tag{2}$$

where α is a random intercept for sites by state; β_{1-4} is the temporal trend in breeding activity by state S; β_{5-9} is the effect of 0–4 year lags in the logit probability of foxes being present; $\pi_{ij}$is the probability of foxes being present at a site for a given year; β_{10-14} is the effect of 0–4 year lags in winter rainfall $R_{i,j}$; β_{15} is the effect of patch size P_i; β_{16} is the effect of the proportion of cleared land C_i; M_i indicates whether the fire history of site i in year j is known ($M_i = 1$) or not ($M_i = 0$); β_{17} is the effect of time since fire $TF_{i,j}$; β_{18} is the proportion of the site burnt in the last fire $PB_{i,j}$, and; β_{19} is the effect of an interaction between time since fire and the proportion of the site burnt.

All covariates were scaled to have a mean of zero and standard deviation of one. We assigned vague normal priors $N(0,10^{-6})$ (mean and precision parameterisation) for parameters α_{1-4} and β_{1-19}. The term reS_i is a site random effect, while $reY_{i,j}$ accounts for first-order temporal correlation among mound counts in successive years. The mean of these random effects were set at zero, and the standard deviation for the temporal variation (reS) was assigned a uniform distribution $U(0,100)$ for the prior.

In Eq. (2), our index of fox abundance per site-year $\pi_{i,\ j}$ was modelled as a binomial random variable:

$$f_{i,j} \sim Bin(\pi_{i,j}, n_{i,j}) \tag{3}$$

where $f_{i,j}$ is the observed number of inactive mounds with fox scats and $n_{i,j}$ is the total number of inactive mounds at a site. We included fox scats from inactive mounds in our index of fox abundance because active mounds are thought to attract foxes, potentially confounding any index. We modelled missing fox scat data using imputation (Gelman et al., 2014) given information on fox baiting, according to:

$$ln\left(\frac{\pi_{ij}}{1-\pi_{ij}}\right) = \gamma_{0i} + \gamma_1 Bait_{i,j-1} + \gamma_2 Bait_{ij} + re_i + rey_j \tag{4}$$

where γ_{0i} is a site-level intercept: γ_1 is the effect of baiting in the previous year; γ_2 is the effect of baiting in the current year; $Bait_{i,j}$ is the density of baits laid within 100 km^2 for each site-year; re_i is a site random effect; and rey_j is a temporal random effect. We modelled standardised bait as a normal random variable with mean 0 and variance σ^2 estimated from the available baiting data to account for missing values. We assigned a vague normal prior $N(0,10^{-6})$ for parameters γ_{0-2}.

2.7. Model fitting

We fitted models in R (v3.5) using Markov chain Monte Carlo sampling (MCMC) in the *R2OpenBUGS* package for Bayesian inference (Sturtz et al., 2010) (Appendix S1). We ran two parallel chains for the models for 120,000 iterations with the burn-in set to 60,000 and thinned the remaining samples by 4 to reduce memory requirements. We assessed model convergence using the *coda* package (Plummer et al., 2006) by evaluating the Gelman-Rubin statistic of the posterior distribution for all estimated values, ensuring that R-hat values did not exceed 1.1 (Brooks and Gelman, 1998).

3. Results

3.1. Monitoring effort and citizen science

The NMMD holds about 60,000 records of mound activity uploaded by 492 separate volunteers or land management agencies of which approximately 84% of the data set was collected by citizen scientists. Government agencies were directly responsible for 11% of records, mostly from New South Wales and South Australia, with the remaining 5% of monitoring conducted by mining companies and NGOs. Large increases in mound monitoring occurred in Victoria in the mid-1990s and in South and Western Australia in the mid-2000s, but not in New South Wales (Figs. 2, 3).

3.2. Trends in breeding activity

Trends in malleefowl breeding activity were inconsistent across the four states (Figs. 1, 4). Our model suggests South Australia has had the greatest rate of decline, with breeding activity decreasing 4.8% per year (−0.050; 95% CI −0.062, −0.037) between 1989 and 2017. We found evidence of a decline in breeding activity in Western Australia at a rate of 2.1% per year (−0.022; 95% CI −0.040, −0.004) and that breeding activity in Victoria is stable (−0.001; 95% CI −0.010, 0.009). In contrast, our results suggest that breeding activity has increased at 4.8% per year in New South Wales (0.047; 95% CI 0.009, 0.086), although considerable uncertainty surrounds this estimate (0.9–8.9%)

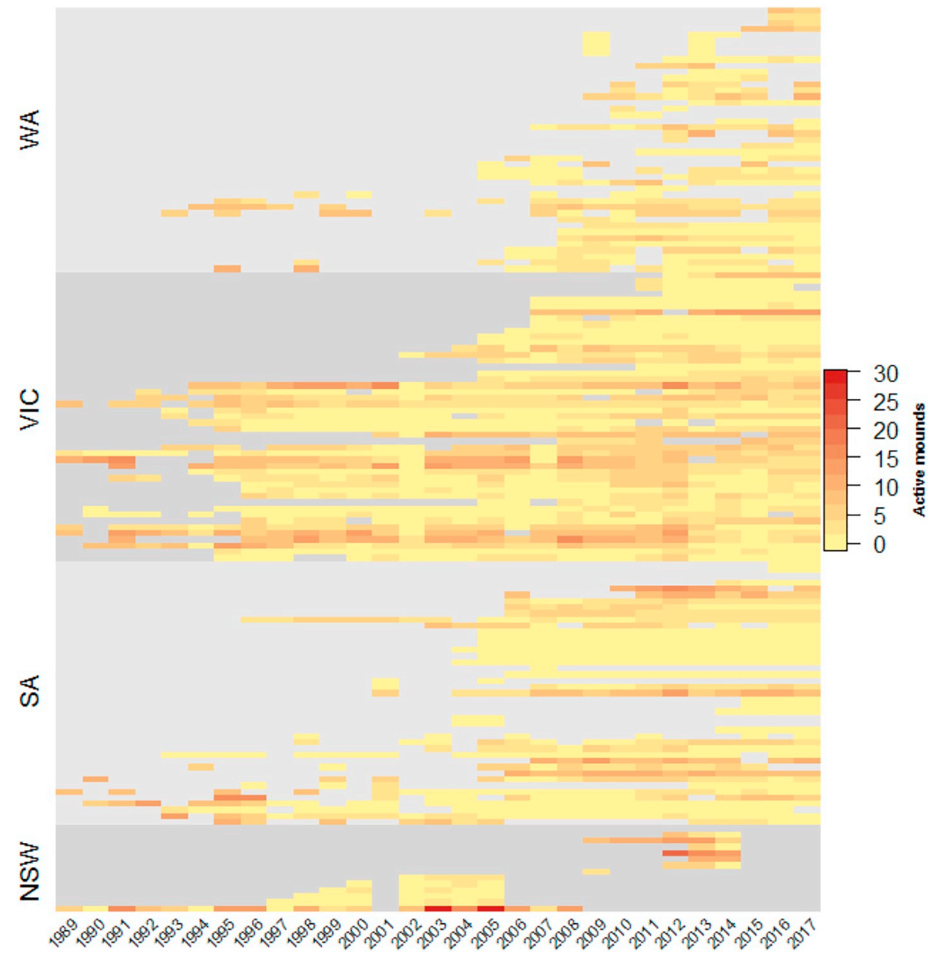

Fig. 2. Number of active malleefowl mounds observed each year at a site grouped by state. Grey shading indicates years when monitoring did not occur.

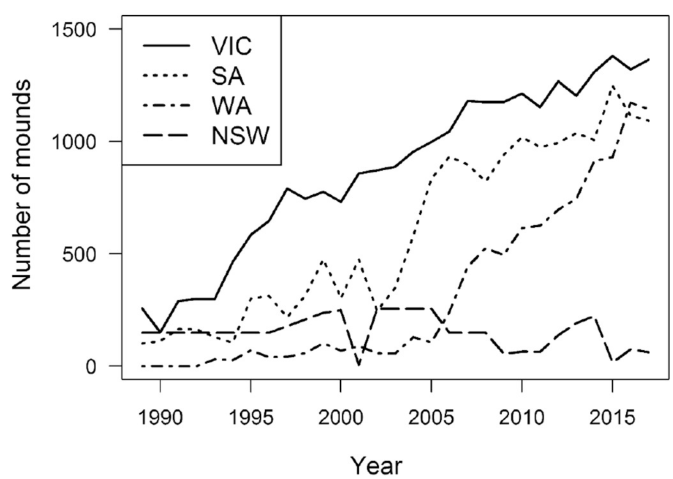

Fig. 3. Number of Malleefowl mounds monitored in each state and records entered on to the National Malleefowl Monitoring Database between 1989 and 2017.

compared to other states due to limited data.

3.3. Influence of site characteristics

There was strong evidence that winter rainfall had a positive influence on malleefowl breeding activity in that year (0.084; 95% CI 0.004, 0.165). Our model suggests a one standard deviation increase in average winter rainfall (60 mm) increased breeding activity by 8.7%. In general, the time-lagged effects of winter rainfall decreased with the length of the lag. For example, a one standard deviation increase in winter rainfall resulted in a 3.1% and 1.9% increase in mound activity in the following one (0.031; 95% CI −0.048, 0.110) and two years (0.019; 95% CI −0.058, 0.096), respectively. There was no evidence that 3 or 4-year lags had either positive or negative effects on breeding activity.

We found strong evidence for a positive effect of time since fire (0.298; 95% CI 0.179, 0.399), and an interaction between time since fire and the proportion of a site burnt (0.292; 95% CI 0.173, 0.410) on malleefowl breeding activity. A standard deviation increase in the number of years since a fire (17 years) increased breeding activity on average by 33.1%. The proportion of a site burnt was negatively associated with breeding activity (−0.191; 95% CI −0.363, −0.030). Our results suggest a negative effect of patch size on breeding activity (−0.255; 95% CI −0.642, 0.020). The effect of the proportion of cleared land surrounding sites was highly uncertain: the 95% credible intervals around our estimate suggest breeding activity could respond either positively or negatively to changes in the proportion of cleared land.

3.4. Effect of fox baiting

There was strong evidence that fox baiting reduced the index of fox abundance in the year of baiting (−0.484; 95% CI −0.640, −0.317), as well as in the following year (−0.259; 95% CI −0.443, −0.074). A one standard deviation increase in the density of baits per square kilometre (10.06) decreased the logit probability of inactive mounds

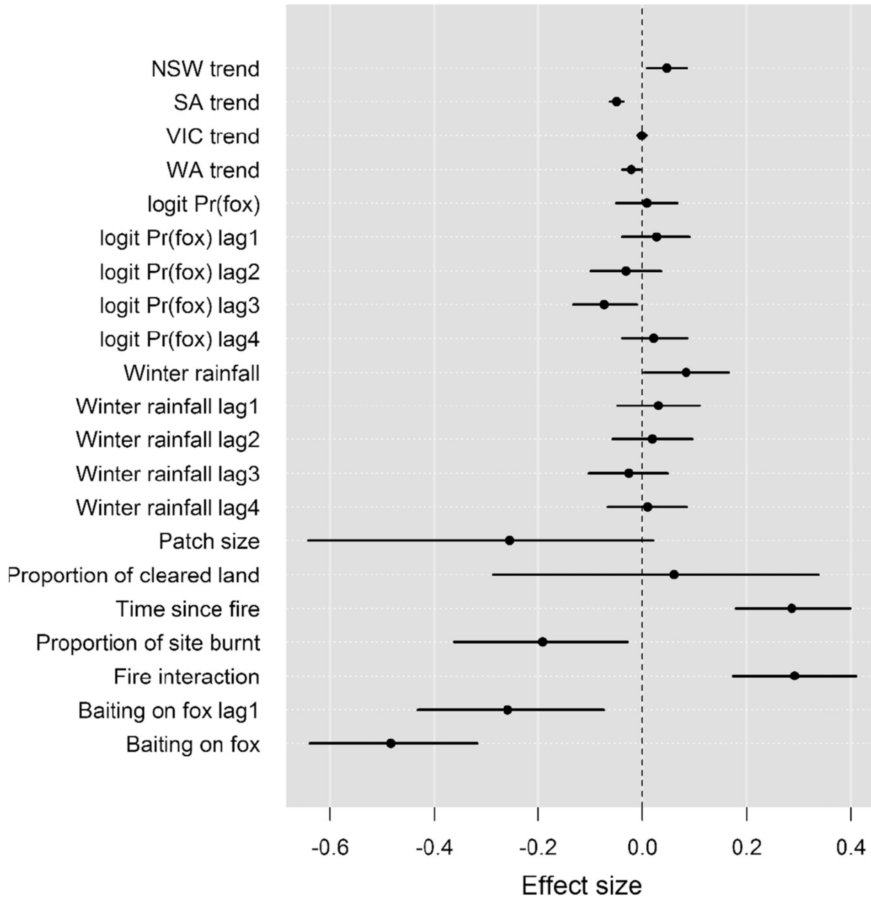

Fig. 4. The strength and direction of the effect of covariates on malleefowl breeding activity. Black circles represent the median of the posterior distribution, black bars represent 95% credible intervals.

having fox scats by 38.3% in the year of baiting and 22.8% the following year. Although fox baiting reduced the logit probability of fox scats on mounds, there was little evidence to suggest this had an immediate effect on malleefowl breeding activity in the same year: there was a very weak positive, but uncertain association between the logit probability of foxes and breeding activity in the year of baiting (0.009; 95%CI −0.051, 0.067). For example, increasing bait density from 3.4 to 10 baits/km^2 decreased the probability of fox scats from 25% to 14%, but had a negligible impact on the number of active mounds that year (1.62 to 1.61). This result was similar for all time-lagged effects of fox abundance on malleefowl, except our analysis suggests a possible negative association between foxes and a 3-year lag in malleefowl breeding activity (−0.074; 95%CI −0.134, −0.010).

4. Discussion

The NMMP has evolved and grown over a period of 30 years to be one of the largest threatened species monitoring programs in Australia both in terms of spatial and temporal extent (Legge et al., 2018). It is maintained almost entirely by citizen scientists who collectively undertake all aspects of the program, from organising and collecting field data to vetting and validating these data in preparation for analysis. In this study, we demonstrated how data collected by citizen scientists using a consistent protocol can be combined to generate statistically robust estimates of population trends at a national scale over a relatively long timeframe. The involvement of citizen scientists has imbued the NMMP with resilience (Couvet et al., 2008) that has sustained it over three decades of uncertain funding. This has allowed the program to expand to most states of Australia where the species is currently found, except for New South Wales, where monitoring has until recently been conducted by government agencies (usually by helicopter) rather than by citizen scientists.

Our analysis provides strong evidence that malleefowl breeding activity has decreased in South Australia and Western Australia and remains relatively stable in Victoria. This result is consistent with a previous study that analysed the first 15 years of monitoring data (Benshemesh et al., 2007), suggesting malleefowl conservation over the last decade in these states has had little effect at improving population trajectories. In contrast, our modelling suggests breeding activity has increased in New South Wales. This is likely to be partly true, at least for the key sites included in this study that were recovering from past wildfire. However, this result is unlikely to be representative of the state given that only 22 site-years spread across 7 sites were added to the NMMD in the last 12 years and included in this analysis. Declines, and in some cases local extinction, have been documented in other regions of New South Wales, especially small habitat patches (Brickhill, 1987; Priddel and Wheeler, 2003). Monitoring more consistently and at more sites in this state, and uploading these data to the NMMD, would improve our understanding of these population trends.

There was strong evidence that winter rainfall had an immediate positive effect on malleefowl breeding activity. This is consistent with our understanding of malleefowl breeding biology: rainfall in the months prior to breeding is thought to benefit body condition and egg production in breeders and enables the decomposition of organic matter to produce heat for incubation (Frith, 1959). This has significant implications for the species given predictions of a drier climate in semi-arid Australia under climate change (Anwar et al., 2007); at our study sites winter rainfall has declined by an average 37% over the past 30 years. Surprisingly, we found no evidence for time-lagged effects of winter rainfall on breeding activity, which is in contrast to Benshemesh et al. (2007) who found strong support for a 2–4 year lag. Such lag effects were expected given it likely takes 3–4 years for juveniles to begin breeding and thus become visible to monitoring. Why this effect was not visible in our study using the full dataset is unclear but suggests that juvenile survival is not simply related to past winter rainfall as previously supposed, but instead might be influenced by a combination of factors over consecutive years.

Our model provided support for a negative association between patch size and malleefowl breeding activity. This result was surprising given ecological theory suggests small patches have higher extinction rates than large patches due to limited connectivity and greater susceptibility to stochastic events (Hanski, 1994). One potential explanation for the negative effect is that sites in larger remnant vegetation patches tend to be found in low productivity areas (i.e. low fertility and rainfall) compared to smaller patches that are remnant from clearing within more productive agricultural landscapes. Although the proportion of cleared land had little influence on breeding activity, agricultural land surrounding small patches might also benefit malleefowl populations by providing additional food sources: malleefowl are often seen foraging for grain and herbs at the edge of cropping fields or on road-sides. Despite the challenges faced by malleefowl in small patches, our research suggests that populations appear resilient in the short to medium term and thus provide potential benefits to the conservation of the species (Wintle et al., 2019).

We found strong support for a positive association between time since fire and malleefowl breeding activity and a negative association between the proportion of a site burnt and breeding activity. There is relatively strong evidence to suggest that malleefowl prefer long unburnt areas and are more susceptible to large intense fires. For example, malleefowl breeding densities were reported to be lower in mallee burnt within a few decades compared to old growth vegetation (Clarke, 2005; Connell et al., 2017). Earlier research also suggests birds return to burnt areas to breed after around 15 years since fire (Cowley et al., 1969), although there is evidence to suggest that birds sometimes return as early as 6–8 years post-fire (Benshemesh et al., 2007). These results have implications for malleefowl conservation: decision-makers should carefully consider both the extent, frequency and timing of fire when managing known malleefowl habitat.

While fox baiting significantly reduced our index of fox abundance, our model suggests that baiting had little immediate benefit to malleefowl breeding activity. This was also a key finding in studies by Benshemesh et al. (2007) and Walsh et al. (2012), but unlike these studies we found weak evidence for a negative association between the log-odds of our fox abundance index and a 3-year lag in malleefowl breeding activity. Malleefowl are most susceptible to fox predation immediately following hatching (Priddel and Wheeler, 1990), so this effect might be a delayed signal of increased juvenile survival due to a reduction in fox abundance. However, this result should be treated with caution because there was no support for a 4 year lagged effect and because there is likely to be considerable variation in the detectability of fox scats by citizen scientists. The malleefowl adaptive management (AM) experiment aims to resolve this issue by deploying camera-traps at spatially replicated control (unbaited)-treatment (baited) sites that will be monitored alongside the NMMP (Hauser et al., 2019). The camera traps will provide more detailed information about the relationship between baiting, predator activity and malleefowl, as well as how competitors respond to changes in predator abundance. When operating in parallel, the NMMP will provide valuable information on malleefowl trends while the AM experiment will improve what can be learnt about the benefits of baiting for malleefowl conservation.

4.1. Dealing with uncertain mound counts

We screened mound activity data to minimise the chance of bias or classification errors by volunteers. Although this took considerable effort, post-data collection quality control is a critical component of citizen science programs (Crall et al., 2011). Once screened, we utilised interval censoring and imputation techniques within a Bayesian hierarchical framework to account for the remaining uncertainty in mound counts and baiting data (Gelman et al., 2014). Interval censoring is commonly applied to survival studies where observations cannot be made beyond the duration of an experiment (Fox, 2015), but it is perhaps under-utilised in ecological studies: uncertain counts are usually removed from datasets, which potentially biases parameter estimates. We demonstrate how interval censoring can deal with uncertainty in observations, which is particularly relevant to citizen science programs.

4.2. Limitations and assumptions

Our study made many important decisions and assumptions that warrant further attention. Firstly, while monitoring mound activity is cheap and achievable by citizen scientists, measuring abundance or age-class survival rates would provide more direct measures of population dynamics, but would come at the cost of massively reducing the spatial and temporal resolution of sampling because it requires specialist skills and experience, and considerable cost. Secondly, although birds usually return to the same mound each year, they can move between existing mounds or build new ones that remain undetected, giving the false impression of a declining trend. The NMMP tries to reduce the impact of this uncertainty by re-surveying sites for new mounds every 5–10 years, although this is rarely achieved due to large effort and/or costs involved. Thirdly, rainfall data at sites was interpolated from the nearest weather station while accounting for local topography. Comparing the predictive performance of rainfall with finer-scaled remotely sensed vegetation condition indices, such as NDVI, might better capture site-level conditions and improve the fit of our model.

4.3. Value of citizen scientists

By providing citizen scientists with the training required to undertake monitoring, the NMMP can cost-efficiently track changes in malleefowl breeding activity and indices of fox abundance across large spatial and temporal scales. The motivation for citizen scientists to participate in the NMMP has not been formally elicited; however, monitoring provides information learning that improves science literacy (Evans et al., 2005), improves critical thinking and provides an opportunity to reconnect with nature (Miller, 2005). While the time a volunteer spends monitoring every year is relatively small – most can monitor their share of mounds in a few days – the combined contribution of efforts across Australia is substantial. For example, 93 volunteers surveyed 1349 mounds at 47 sites in Victoria in 2017 alone. This amounts to an estimated 1429 h in the field, 673 h travelling to and from sites, and a further 280 h managing the ensuing data. Assuming a volunteer labour value of $54.72 per hour (research assistant grade 1) and accounting for the average travel expenses to and from sites, we estimate that Victorian volunteers contributed approximately AUS $146,496 in monitoring in 2017. This equates to a contribution of about AUS $397,443 annually of citizen effort across Australia each year, and a total of AUS $5,179,557 across the history of the program.

Despite their advantages, not all citizen science monitoring programs are successful or lead to useful management outcomes (Devictor et al., 2010). A key ingredient to the success of the NMMP has been the direction provided by the Recovery Team, which consists of federal government agency staff, academics, citizen scientists and land managers from each state in Australia where the species is found (Gillam et al., 2018). Success of the program has also been due to a small group of leaders, some of whom were partly funded by the Recovery Team, who have championed the program, guided others, and lobbied for widespread participation. These leaders have played a critical role recruiting the initial volunteers, training new ones and developing standardised methods across political boundaries to ensure citizen scientists have the skill level required to conduct monitoring and minimise biases and measurement error during data collection (Benshemesh et al., 2018). A key step towards maintaining participation has been regular ‘reporting back’ weekends, run once a year in some states to communicate back to citizen scientists the results of monitoring (McKinley et al., 2017). At these events, a simple report summarising the yearly status of malleefowl breeding activity is prepared for volunteers, which provides an understanding of how the species is trending at their site compared to their state and nationally. More importantly, these events build a sense of community and give volunteers ownership of the national monitoring project. Finally, the National Malleefowl Monitoring Database (NMMD) is crucial for storing all data collected by volunteers, and ensuring that these data are in an appropriate format to be analysed using robust statistical approaches. Maintaining the flow of data from citizen scientists, to the database, to analysts, and to the broad community, has been crucial to ensure the program influences conservation management decisions.

5. Conclusion

Our analysis suggests that malleefowl breeding activity has declined in all states except for Victoria and New South Wales. While fox baiting reduced our fox abundance index, there was no conclusive evidence to suggest malleefowl breeding activity benefited from the apparent reduction in foxes. We demonstrate how citizen scientists can collect valuable and centralised information on population trends for a widely dispersed threatened species. The temporal and spatial resolution required to monitor malleefowl across Australia is perhaps only achievable by the dedication and commitment of citizen scientists, the involvement of which has vastly improved our ability to document ecological patterns at large-scales, and has provided the program with the resilience needed to sustain it in the long-term.

CRediT authorship contribution statement

Joe Benshemesh:Conceptualization, Data curation, Writing - review & editing.**Darren Southwell**:Formal analysis, Writing - original draft, Writing - review & editing.**Richard Barker**: Methodology.**Michael McCarthy**:Methodology, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data and R code

Data and R code to run the trend analysis is provided in the online supporting information (Appendix S1). Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2020.108573.

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