Title: The impact of a fox- and cat free safe haven on the bird fauna of remnant vegetation in south-western Australia.

Running Head: Cat and fox predation on birds

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Manuscript preparation: MS prepared the manuscript; LR, BP, NP, GV, AT, NR organized the field work and collected the data;: GV, NR prepared and organized the data; MS analysed the data; LR, BP, NP, GV, AT, NR edited the manuscript.

Abstract

Introduced predators are a serious threat to Australian vertebrates. However, the consequences of predation for the avifauna have rarely been quantified. We took advantage of the establishment of a 7,832 ha fox- and cat-free safe haven at Mt Gibson, in Western Australia, to assess the consequences of excluding introduced mammal predators on the bird fauna. Bird surveys were conducted over six years, before and after the establishment of the introduced predator-free safe haven. After three years, half the sites were enclosed by the fence that excluded introduced predators, while the remainder of sites remained outside the fence and were exposed to fox and cat activity. The sites were stratified by four major vegetation types. A total of 91 bird species were variously detectable with the survey approach, but were typically more detectable during morning surveys. Site occupancy varied considerably among species, but overall, occupancy by all species was most likely to be either not impacted or positively impacted by the safe haven. The most notable change was that avifaunal richness appeared to increase in woodland and shrubland habitats within, as compared to outside, the safe haven. We conclude that: (1) the safe haven had an overall positive impact on bird occupancy; and (2) there were no consistent trends with respect to the kinds of species whose occupancy was positively impacted, beyond them all being small- to medium-sized birds and mostly insectivorous. However, these conclusions must be tempered by the poor detection probability of many species.

Implications for practice

• The creation of a safe haven free of introduced mammalian predators does not appear to have impacted negatively on occupancy by any bird species or on avifaunal richness

• Preventing predation by introduced mammalian predators in order to conserve threatened taxa (small mammals in this case) can have positive impacts upon non-target taxa, such as the ecosystem's avifauna — providing additional NRM benefit.

Introduction

The 'extinction crisis' (Ceballos & Ehrlich 2002) has seen many species disappear from the Australian landscape and many more threatened with extinction (Evans et al. 2011). Predation by introduced species, especially cats (*Felis catus*) and red foxes (*Vulpes vulpes*), is a major driver of extinctions in small- to medium-sized native Australian mammals (Woinarski et al. 2014), as well posing a serious threat to many native bird species (Woinarski et al. 2017; Woinarski et al. 2017). For native mammals, the creation of cat- and fox-free habitat inside fenced areas or islands (so-called safe havens; Legge et al. 2018), followed by reintroductions of threatened species, has been one of the few successful tactics (Woinarski et al. 2014; Legge et al. 2018). On first principles, the establishment of safe havens would also be expected to benefit those bird species preyed upon by foxes and cats; however, the consequences of cat and fox exclusion, followed by mammal reintroductions, for the extant avifauna at safe haven sites are poorly understood.

One of the difficulties in predicting the consequences of safe haven establishment for the native avifauna is that removing introduced mammalian predators is likely to result in a range of cascading system effects (Mittelbach et al. 1995; Ritchie et al. 2012). For example, with the removal of cats and foxes, herbivorous prey species may increase in density, with this having direct consequences for vegetation communities (possibly leading to changes in species composition and structure: e.g., Linley et al. 2017). In turn, changes in vegetation are likely to influence the availability of habitat and/or food for a range of fauna (as occurred

when key herbivore populations were impacted by the reintroduction of wolves to Yellowstone National Park, Ripple & Beschta 2012). These kinds of changes will likely also interact with a range of system processes such as fire (Fuhlendorf et al. 2009), competition from weed species (Cheal 1986), and hydrology (Eberbach 2003; Yates et al. 2000). Furthermore, the infrastructure required to exclude cats and foxes (~1.8 m high electrified mesh fence, extending over many kilometres for large safe havens) may have consequences for the management system. In particular, night-flying and ground-nesting bird species risk entanglement or collision with such fences (Long & Robley 2004). If mortality from this source is high, it could potentially negate the benefits of reducing predation by foxes and cats.

The Australian Wildlife Conservancy (AWC) is a leading proponent of the establishment of safe havens, managing five of the six largest such areas on the Australian mainland (Kanowski et al. 2018). In 2015, AWC completed a 7,832 ha cat and fox-free safe haven at its Mt Gibson Wildlife Sanctuary, located in the northern Wheatbelt of Western Australia (Figure 1). The safe haven was erected primarily to conserve threatened mammal fauna. From 2012 to 2017, occupancy of the terrestrial bird fauna in sites that remained outside the safe haven and sites that transitioned into the safe haven was monitored. In this paper, the initial impact of the cat and fox-free safe haven on occupancy by terrestrial birds is assessed.

Methods

The sampling design used to survey the bird fauna at Mt Gibson consisted of primary annual sampling periods (years) that incorporated repeat secondary surveys (days) which involved consecutive morning and afternoon visits over a two-day period. All primary survey periods were in October or November. Surveys were conducted over six years: 2012 – 2017. Surveys commenced three years prior to the final eradication of cats in May 2015 (no foxes were found inside the safe haven post-closure of the fence) until three years post-eradication (Ruykys & Carter 2019). Note that the surveys from 2013 to 2015 extended over the period of fence construction (September 2013 – June 2014) and eradication/post-eradication monitoring (June 2014 – August 2015).

The study consisted of 24 sites: of these, 12 transitioned into the safe haven, while 12 remained outside. One of the 24 sites was not surveyed in the first year and two were not surveyed in the first two years and these sites were not included in the analysis. The sites were also stratified by four broad vegetation types —a vegetation community dominated by *Callitris* species, a community characterised as a mixture of *Acacia* shrublands and *Eucalyptus* woodlands, a community dominated by *Acacia* shrublands, and a community characterised as *Eucalyptus* woodlands. Each vegetation type had six sites; of these, three remained outside of the safe haven and three transitioned into it.

The sites were set up such that, relative to each other, they formed a north-western group (outside the safe haven) and a south-eastern group (inside the safe haven; Figure 1) with a

distance of around 15 km between the two group centres. Consequently, any inferences from the study are confounded by location. A site longitude covariate was included in the modelling to account for some of the possible effects of this bias (i.e. Goijman et al. 2015); however, we consider any confounding factors unlikely to be as important as the main treatment effect, given that introduced foxes and cats are known to be major predators of the Australian avifauna (Woinarski, Woolley, et al. 2017; Dickman 1996).

Surveys were conducted by experienced volunteers from BirdLife Western Australia (http://www.birdlife.org.au/locations/birdlife-western-australia) and AWC ecologists. On any given survey day, surveys were conducted once in the morning (≈ 06:00 AWST) and once in the afternoon (≈ 16:00 AWST) and involved teams of three to five people undertaking systematic bird surveys of 2 ha, 20 minute centred on the site. This methodology is consistent with BirdLife Australia's Atlas survey methods (https://birdata.birdlife.org.au/) but used a circular area with a radius of 80 m such that surveyors always remained within 80 m of the central point. To minimise observer bias, groups of surveyors rotated through sites such that all teams visited all sites. Furthermore, to reduce any effects of time of day or temperature, the order of the sites surveyed was reversed on the second morning and second afternoon survey of each site. Whether seen (including flying overhead) or heard, birds were identified to species level whenever possible. Only presence information was recorded (i.e. not information on bird abundance).

Statistical analysis

Given the nature of the data — repeated secondary sampling of multiple species; within a series of primary periods; and with the implementation of a treatment on a subset of sites after three primary periods — the hierarchical multi-season, multi-species modelling formulation detailed by Goijman et al. (2015) is appropriate and was employed. Of note, we were particularly interested in alpha diversity not gamma diversity (Iknayan et al. 2014) and, as such, did not use the data augmentation approach (Royle et al. 2007). There are two major benefits to this method of statistical analysis. First, the approach easily and appropriately accounts for the non-balanced nature of the data: not every site was monitored in each primary period and the number of secondary surveys varied from two to four visits (Table S1). Second, it is likely that many species will be poorly detected. Given that multiple species will be linked together within the hierarchical formulation of the model, the data are used more efficiently and, as a consequence, there is improved precision in the estimates of occupancy (Goijman et al. 2015; Sauer & Link 2002; Royle & Dorazio 2008). The approach models detection/non-detection data from multiple species pre- and post-treatment (i.e. Goijman et al. 2015; Russell et al. 2009). Each species is modelled individually (with species-specific parameters) but also related to other species through the model's hierarchical formulation (Goijman et al. 2015). The modelling approach is formulated within a Bayesian framework, which incorporates binomial logistic regression sub-models (Kéry 2010). The two logistic sub-models account for: (1) the biological process (to describe true occupancy); and (2) the detection process given each species actually occurs at the site. The modelling code used is provided by Goijman et al. (2015).

For each species, the data $y_{i,j,k,t}$ represents the observed occupancy state (a binary variable) for species i at site j on secondary survey k and in year t. The basic form of the models is:

1) $z_{i,j,t} \sim$ Bernoulli ($\Psi_{i,j,t}$) — the ecological process that produces a '1' when the species is present and a '0' when it is absent. $\Psi_{i,j,t}$ is the probability that the species i occupies site j in year t;

2) $y_{j,k,i,t}|z_{i,j,t} \sim$ Bernoulli($z_{i,j,t}$, $\rho_{j,k,i,t}$) — the observation process, where $\rho_{j,k,i,t}$ is the probability that at site j on secondary survey k, species i is detected in year t.

Several covariates were included in the ecological process model: (1) a random variable for habitat type (indexed by 'h' for each of the four different habitat types); (2) a random site by time by habitat covariate for the completion of the safe haven; and (3) a longitude co-variate (standardised to a mean of 0 and standard deviation of 1). Thus, the process model was:

Logit ψ(j,i,t) = alpha.occ(i,t) + beta.occ.1(i, habitat[h]) + beta.occ.2 (i, habitat[h]) × safe haven(j,t) + beta.occ.3(i)× longitude(j). For the observation model, whether the site was in shrubland (or not) or mixed shrubland/woodland (or not) was included as habitat covariates, as was whether the site was visited in the morning (or not). Sites with shrubland components were thought to be more likely to decrease the observers' ability to visually detect birds. The observation model was:

Logit ρ (j,k,i,t) = alpha.det(i,t) + beta.det.1(i) ×shrubland (j) + beta.det.2(i) × mixed-woodland/shrubland(j) + beta.det.3(i) × morning(k)

As described by Goijman et al. (2015), the modelling approach allows for a "community hierarchical component" such that the individual species-level parameters are modelled as co-variate-driven random effects. Finally, following Goijman et al. (2015), we incorporated a random time effect on the intercept for occupancy and detection for each species at each site; this aimed to further control for among-year variation in factors such as observers, climate etc.

The statistical analyses was performed using the 'rjags' package (Plummer 2017) run within the R software environment (R Core Team 2013). In terms of model running and checking, the final model was run with three parallel chains of length 350,000, with the first 100,000 iterations discarded as burn-in. A thinning rate of 500 was used to reduce auto-correlation. Parameter estimates and their 95% Bayesian credible intervals (CI) are reported. The Gelman-Rubin statistic (Gelman & Rubin 1992) and a graphical assessment of the parallel chains was used to assess convergence. Non-informative (or vague) priors were used for all modelled parameters; however, model fit with alternate prior formulations was examined, exactly following the approach of Guillera-Arroita et al. (2019). Site-level species richness was calculated by summing 'z' at each site in each year. A Bayesian p-value was calculated to assess model fit. A Bayesian p-value compares a fit statistic calculated from a simulated dataset with a fit statistic calculated from the actual data (Kéry & Schaub 2011). Estimates close to 0.5 indicate a good fit, whereas estimates near to 1.0 or 0.0 indicate a poor fit (Kéry & Schaub 2011).

Results

Model performance

The Gelman-Rubin statistic values and the visual representation of the parallel chains confirmed convergence in all monitored parameters, and model fit was satisfactory (Bayesian p-value = 0.71). Consequently, we concluded that the model was suitable. Alternate prior specifications had little to no impact on the posterior estimates and, as such, standard uninformative priors were used.

Individual species

Ninety-one bird species were detected (Table S2). The probability of detection varied considerably among the different species (Fig S1) but was often very low, indicating that the survey approach had low power and high uncertainty for many species. Detection for most species was reduced in *Acacia* shrubland habitats but particularly for the Australian ringneck, black-faced cuckoo-shrike, brown goshawk, chestnut-rumped thornbill, common bronzewing, galah, Horsfield's bronze-cuckoo, inland thornbill, regent parrot, spiny-cheeked honeyeater, striated pardalote, tree martin, weebill and white-fronted honeyeater (Fig S2; species' scientific names in Table S2). Similarly, detection for most species was reduced in the mixed *Acacia* shrubland-*Eucalyptus* woodland habitats but particularly for the Australian ringneck, chestnut-rumped thornbill, common bronzewing, grey butcherbird, singing honeyeater, and

spiny-cheeked honeyeater (Fig S3). Detection increased in the *Acacia* shrubland and mixed

shrubland/woodland habitats for several species, but most notably for the golden whistler in

the *Acacia* shrubland habitat and the white-fronted honeyeater in the mixed habitat type (Fig

S2 and Fig S3). Finally, the majority of species were likely to be more detectable in the

morning surveys when compared with the afternoon surveys (Fig S4).

Mean probabilities of occupancy were often very uncertain and varied from one species to the

next. There was a slight tendency for species to be more likely to occupy the north-western

sites prior to fence construction (Fig S5). Nonetheless, occupancy by the vast majority of

species was likely to be either not affected or positively affected by the creation of the safe

haven (Fig S5 and Fig 2). Most notably, occupancy by the Australian ringneck, grey fantail,

grey-fronted honeyeater, rufous treecreeper, tree martin, and willie wagtail was significantly

increased inside the safe haven in one or more of the vegetation types when compared with

outside (i.e. 95% CI around mean probability of occupancy did not include zero; Fig 2).

Longitude typically had a negative effect on occupancy, but particularly for the singing

honeyeater (Fig. S6).

Avian community

More broadly, species richness across all sites oscillated in a similar manner — richness

gradually decreased over the first three primary periods, increased over the next two primary

periods, and then decreased over the last primary period (Fig 3). Other than the *Acacia*

shrubland sites, species richness was predicted to be higher in the north-western compared to the south-eastern (ultimately fenced) sites prior to the safe haven (Fig. 3). With the exception of the mixed woodland-shrubland sites, species richness generally increased in the safe haven sites after the creation of the safe haven, relative to the non-safe haven sites; however, this was particularly evident in the *Eucalyptus* woodland and *Acacia* shrubland sites (Fig 3). Post treatment, the *Eucalyptus* woodland sites within the safe haven tended to have the highest estimates of overall species richness (Fig 3).

Discussion

In Australia, when it comes to re-establishing threatened mammal species within a management system, cat- and fox-free safe havens are a proven conservation approach (Dickman 2012; Innes et al. 2012). Given Australia's wildlife extinctions and declines (Evans et al. 2011), any successful management approach is of the highest importance. However, there is less information available on the effects of safe havens on 'non-target' species (i.e. where threatened mammal species are re-established, they typically receive most of the monitoring attention). Here we show that the creation of a safe haven at Mt Gibson Sanctuary is likely to have had an overall positive impact on occupancy by avifauna, with no evidence of a decline in occupancy by any species. These conclusions must be tempered by the poor detection probability of many species. Furthermore, it is important to acknowledge that, even with the inclusion of spatial covariates to reduce any confounding issues related to site location (e.g., Goijman et al. 2015), some caution is still warranted. This is because we

cannot completely discount factors that may be unrelated to the creation of the safe haven. However, the impacts that introduced predators can have on birds is well-established (Woinarski, et al. 2017) and the spatial distance between sites in the different treatments is unlikely to have caused any major long-term environmental differences between the two areas. Additionally, the north-western sites (which remained outside of the safe haven) tended to be more species rich prior to the fence construction and less so after, suggesting that any more general differences between sites within and without the safe haven were more than counterbalanced by the removal of cat and fox predation. Consequently, our inference of an overall neutral or positive impact by the safe haven on occupancy by the terrestrial bird fauna appears reasonable.

With the exception of the Australian ringneck parrot, the species whose occupancy was predicted to be most positively impacted by the creation of the safe haven were all below the critical weight range for cat predation (60 to 300 g) proposed by Woinarski, Woolley et al. (2017) and were insectivorous. On the one hand, more work is warranted to better understand mechanisms that may underpin increased occupancy by bird species within a safe haven and whether it is the direct result of cat predation or indirectly through processes that relate to the presence of cats (e.g., Bonnington et al. 2013). For example, in addition to reducing habitat suitability for some species simply by their presence (Bonnington et al. 2013), cats are known to predate upon small mammals, reptiles, and insects (Kutt 2011) and, as such, their removal is likely to affect the availability of prey species to other predators. Additionally, a number of small mammals were in the process of being reintroduced to the safe haven over the study

period, also affecting prey availability. Either way, there is now little doubt about the involvement of cats in the extinction and decline of many bird species (Woinarski, Murphy, et al. 2017; Woinarski, Woolley, et al. 2017). Within that context, and given the results provided here, it is likely that cats have negatively impacted occupancy by many bird species in the Mt Gibson Sanctuary. Foxes may be more likely to impact larger bird species such as malleefowl (Dickman 1996), which were poorly detected in this study.

The detection probability for many species was low, reducing the power of the survey. Timed searches for birds within a spatial unit is a commonly-used survey methodology (Ellis & Taylor 2018; Loyn 1986; Slater 1994) and, as such, our results provide an opportunity to suggest potential improvements to the approach. At Mt Gibson, it would be prudent to determine whether increasing the number of repeat visits (especially morning visits), search time, and/or having additional people searching would improve detectability. An appropriate balance between use of resources and improving detectability would need to be reached; this may require a compromise such as conducting more labour-intensive (i.e. more people and/or longer search times) biennial or triennial surveys rather than the current annual visits. Technological improvements, such as the development of acoustic recorders and call recognisers, may allow more robust data to be obtained on the avifauna given available resources. As the reintroduced species increase in abundance and distribution within the safe haven, we can expect there to be concomitant changes to the system in terms of vegetation composition and vertebrate prey availability, in addition to significant change to the invertebrate fauna(Gibb 2012). It will thus be important to accurately monitor the response of the avifauna to any future changes to allow management to respond if required and feasible.

Here we conclude that safe havens can have a positive effect on bird occupancy, likely as a result of the removal of cat and fox predation. However, the detection probabilities for many species were low, indicating that monitoring efforts should be adjusted to improve detectability. With improved power, monitoring of site occupancy of the avifauna at Mt Gibson will allow AWC to maintain a suitably rigorous understanding of how species are 'tracking' over time and in response to system change and management. By doing so, management outcomes can be assessed and adapted as required. Additionally, if any bird species begin to show signs of decline in occupancy (beyond that expected with normal oscillations), management activities can be triggered, where appropriate.

Acknowledgements

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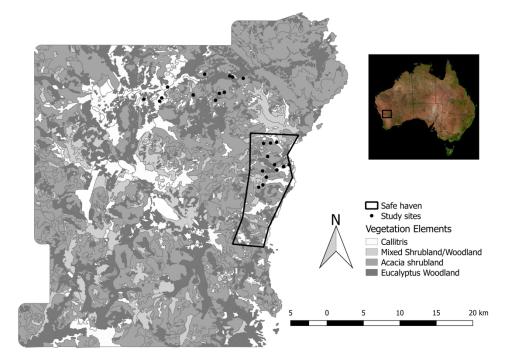1263

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Fig 1. Mt Gibson Wildlife Sanctuary and location of the safe haven and study sites used to assess changes in avian species' occupancy.
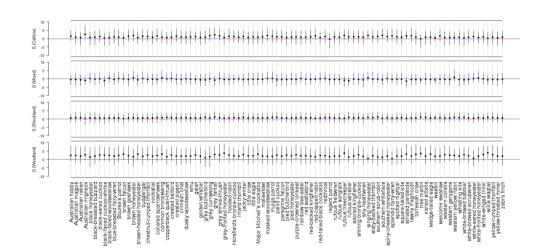
Fig. 2: Mean probability (± 95% CI; black bars) of the effect of the safe haven on occupancy rate (Beta) for each bird species within the four key habitat types. Where species are likely to have responded strongly to the effect of the safe haven, the 95% CI will not include zero (horizontal line). Grey vertical lines that connect each species to its data point have been included to facilitate interpretation. Bird species names are provided in Table S2 in the same order.

Fig 3. Predicted annual change in bird species richness (± 95% CI) at each site within each major vegetation element. Black lines indicate sites that transitioned into the safe haven; grey lines sites that did not. N.B.: values along the x-axis have been jittered to facilitate interpretation and a vertical dashed line has been included to indicate richness pre- (left of line) and post- (right of line) eradication of introduced predators from the safe haven. The actual location of the vertical line along the x-axis is not indicative of the exact time the eradication was completed. Dashed lines indicating the mean richness estimated across all sites in each year for those sites that were to be incorporated into the safe haven (and not) has also been included to facilitate interpretation of the results.

Calitris woodland

Mixed Eucalyptus woodland/Shrubland

Shrubland

Eucalyptus woodland