Animal Conservation

The response of arboreal marsupials to long-term changes in forest disturbance

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ABSTRACT

Quantifying the long-term population trajectory of species and the factors affecting these trends is a fundamental part of animal conservation. We describe the results of a long-term investigation of temporal changes in the occurrence of arboreal marsupials in the wet eucalypt forests of south-eastern Australia. The assemblage includes habitat specialists such as the vulnerable greater glider (*Petauroides volans*) and the critically endangered Leadbeater's possum (*Gymnobelideus leadbeateri*), as well as common and widespread taxa. Using data gathered between 1997 and 2018, we quantified relationships between site occupancy of four marsupial species and spatio-temporal site and landscape-level variables, including the number of hollow-bearing trees at a site, and the extent of fire and logging in the surrounding landscape. We found evidence that: **(1)** The number of hollow-bearing trees (which are critical den sites for arboreal marsupials) has declined substantially in the past two decades. **(2)** There was a decline in all species of arboreal marsupials. **(3)** The presence of all species of arboreal marsupials was positively linked to the number of large old hollow-bearing trees at a site. **(4)** The extent of logging disturbance in the landscape surrounding a site had a positive impact on the sugar glider (*Petaurus breviceps*) but a negative effect on Leadbeater's possum, suggesting that ongoing logging will have further negative impacts on the Leadbeater's possum. (**5)** The presence of the greater glider and sugar glider declined with increasing amounts of fire in the landscape. Negative fire effects are a concern as montane ash forests are increasingly susceptible to high-severity wildfires. Stronger efforts are needed to reduce the extent and frequency of logging and fire disturbance in mountain ash forests to protect arboreal marsupial populations.

KEYWORDS: Arboreal marsupials, logging, clearcutting, wildfire, landscape ecology, south-eastern Australia, Mountain Ash forest, biotic homogenization, long-term studies

INTRODUCTION

The world is experiencing a biodiversity crisis, with large numbers of species at risk of decline and extinction (Ceballos, Ehrlich & Dirzo, 2017; IPBES, 2019). While global assessments of biodiversity show major declines (e.g. Maxwell *et al.*, 2016), some species are increasing at regional and local levels, while others are declining (Inger *et al.*, 2014; Lindenmayer *et al.*, 2018b; Nielsen *et al.*, 2019). Quantifying and understanding such variation in the trajectories of species is dependent on long-term population data. Analyses of long-term population trajectories can be particularly powerful when they are linked with drivers of change such as the impacts of invasive species (e.g. Savidge, 1987), the spread of disease (e.g. Scheele *et al.*, 2019), modifications to local habitat suitability (Morrison, Marcot & Mannan, 2006), or changes in landscape cover (Fahrig, 2017; Tscharntke *et al.*, 2012).

Investigations that couple patterns of temporal change in populations of species with potential drivers of those changes are critical for guiding effective conservation management (e.g. Betts *et al.*, 2019; Haddad *et al.*, 2015). In the study reported here, we use a dataset gathered between 1997 and 2018 on Australian arboreal marsupials, to quantify temporal changes in animal occurrence at 158 long-term sites. We examined these changes in the context of spatio-temporal changes in key habitat and landscape attributes. Our work focused on the montane ash forests of the Central Highlands of Victoria, which is a heavily disturbed native forest environment that supports several species of arboreal marsupials (and other plant and animal taxa) of conservation concern (Taylor & Lindenmayer, 2019). The arboreal marsupial assemblage includes specialist species such as the vulnerable folivore, the greater glider (*Petauroides volans*), the vulnerable exudivore, the yellow-bellied glider (*Petaurus australis*), and the range-restricted and Critically Endangered Leadbeater's possum (*Gymnobelideus leadbeateri*). The assemblage also includes widespread generalist taxa such as the mountain brushtail possum (*Trichosurus cunninghami*) and sugar glider (*Petaurus *breviceps*) (see Appendix Table S1). Our overarching aim was to quantify patterns of temporal change in these species and determine which site and landscape-level factors were associated with such temporal changes. Our study tested four inter-related questions:

Q1. What are the temporal trends in critical denning resources for arboreal marsupials?

Almost all species of arboreal marsupials in montane ash forests are cavity-dependent and require large old hollow-bearing trees for shelter and reproduction (Lindenmayer *et al.*, 2017a). We sought to determine if the trend for declines in populations of these trees documented in past studies (e.g. see Lindenmayer *et al.*, 2011) has continued.

Q2. What are the temporal trends in the occurrence of arboreal marsupials?

We predicted a decline in the occurrence of all species of arboreal marsupials, in line with an expected decline in denning resources. However, we anticipated that declines would be most pronounced in range-restricted species such as Leadbeater's possum and the dietary specialist, the greater glider (which consumes only eucalypt leaves).

Q3. Is the number of hollow-bearing trees related to land tenure and landscape levels of wildfire and logging?

The landscapes surrounding our long-term sites have been subject to extensive disturbance as a result of clearcutting (VicForests, 2019) as well as a major wildfire in 2009. These disturbances can alter patterns of wind movement leading to elevated tree fall (e.g. Savill, 1993). We sought to quantify relationships between the abundance of hollow-bearing trees on sites and the amount of logging and fire in the surrounding landscape. We predicted there would be negative relationships between these measures; that is, fewer trees on sites where more of the surrounding landscape had been logged or burnt.

Q4. Is the occurrence of arboreal marsupials related to the number of hollow-bearing trees, land tenure, and landscape-levels of wildfire and logging?

Past investigations have established strong statistical relationships between the number of hollow-bearing trees at a site and the occurrence of arboreal marsupials (Lindenmayer *et al.*, 2017a). We predicted that such relationships would persist and, hence, that changes in the number of hollow-bearing trees (see Q1) would underpin changes in the occurrence of arboreal marsupials.

Animal species may be adapted to the fire regimes with which they have co-evolved (Frelich, 2005; Whelan, 1995). Montane ash forests and associated animal species have evolved under a fire regime characterized by rare, high-severity wildfire (Ashton, 1981). High-severity wildfire can affect habitat structure and food resources and we therefore anticipated that the occurrence of arboreal marsupials would be negatively related to the amount of fire that occurred in the landscape during the 2009 wildfires. Clearcutting can substantially modify forest and landscape structure and, in turn, reduce habitat suitability for many species, including cavity-dependent taxa. We predicted that the occurrence of animals at a site would be negatively associated with an increasing amount of logging in the surrounding landscape.

Quantifying temporal patterns of change in animal occurrence, and the factors influencing those changes, is fundamental to the development of informed strategies for effective biodiversity conservation (Scheele *et al.*, 2018). The results of the work reported here are therefore important for guiding forest management strategies that aim to conserve communities of arboreal marsupials.

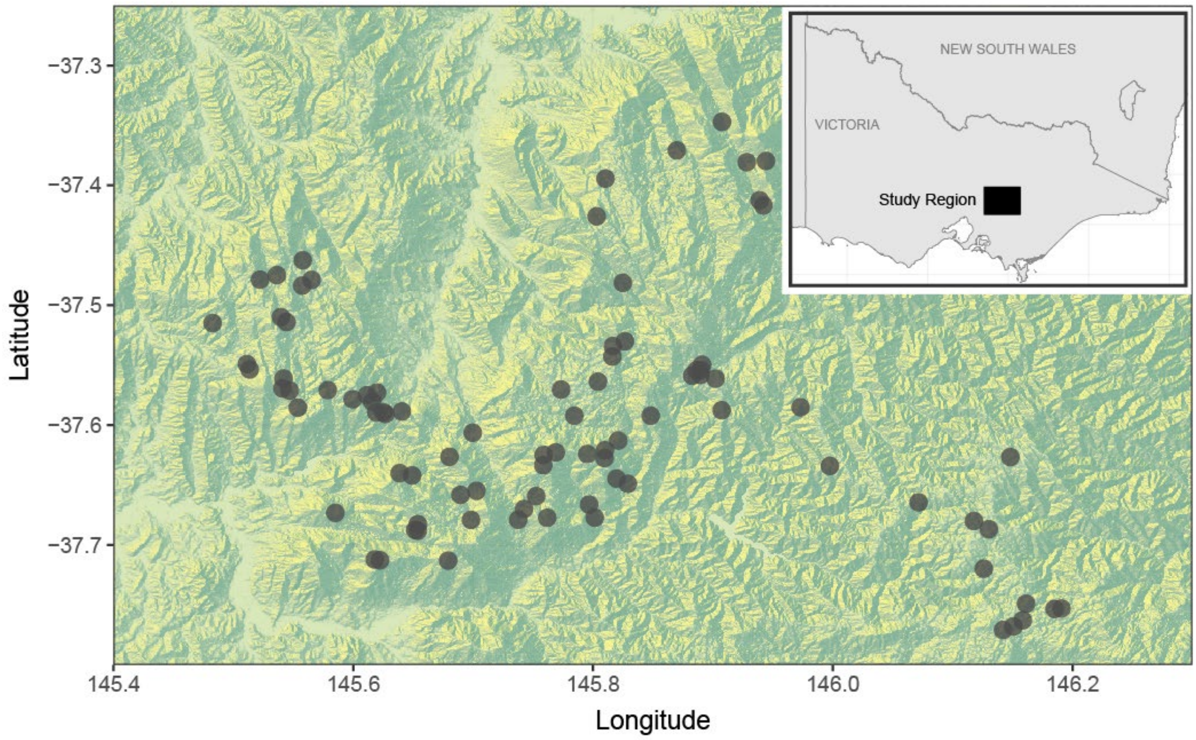
METHODS

Study area

This study was focused on the Mountain Ash (*Eucalyptus regnans*), Alpine Ash (*E. delegatensis*) and Shining Gum (*E. nitens*) forest ecosystems in the Central Highlands of

Victoria, south-eastern Australia (Fig. 1). Forests dominated by these three tree species are collectively termed montane ash forest. We have established 164 long-term monitoring sites, each measuring 1 ha in the Central Highlands region. These sites have been surveyed on a repeated basis for arboreal marsupials and vegetation attributes since 1997.

Our long-term field sites encompass a wide range of environmental conditions including the age of stands, slope, aspect, and whether a site was burned in the 2009 Black Saturday fires (the only wildfire to occur in the region during our study). Our 164 sites spanned two key forms of land tenure in approximately equal proportion. These were areas broadly designated for pulpwood and timber production (hereafter 'wood production forests') and reserves and closed water catchments where logging is excluded (hereafter called 'protected forests'). This large dataset on hollow-bearing trees was representative of broader montane ash environment in our study region.



<u>Fig. 1. The location of the study area in the Central Highlands of Victoria, south-eastern Australia. The black dots show the location of field survey sites.</u>

Site-level surveys of hollow-bearing trees

We mapped the location of all hollow-bearing trees at each of our 164 long-term sites in 1997, 2003, 2009, 2011, 2012, 2015 and 2017. We defined a hollow-bearing tree as any tree > 80 cm DBH and containing obvious hollows as determined by scanning using binoculars. During each survey, we conducted a full reconnaissance of each field site and assessed the condition of each hollow-bearing tree, including whether it had collapsed or remained standing. Importantly, there has been very little recruitment of hollow-bearing trees over the 20+ years of repeated surveys at our long-term sites (Lindenmayer *et al.*, 2018a). Limited recruitment was expected given that the forest at most sites is dominated by trees that are 80 years or younger and it will be at least another 40 years before we would expect these trees to develop cavities (Ambrose, 1982).

Site-level survey of arboreal marsupials

We surveyed arboreal marsupials on 158 of our 164 field sites using the stagwatching method (sensu Lindenmayer *et al.*, 1991a). We did not survey six sites with this method because of logistical constraints in accessing parts of the forest. The stagwatching survey approach entails observers scanning each hollow-bearing tree on a given site for the emergence of arboreal marsupials for an hour before and after dusk. Thus, we documented the number of individuals of each species that was recorded emerging from a hollow-bearing tree on a given long-term monitoring site.

Stagwatching surveys are labour-intensive because at least one observer is required to verify animal emergence from each hollow-bearing tree on each site. This is because all species of arboreal marsupials exhibit den-swapping behaviour whereby animals move regularly between cavities in different hollow-bearing trees (Gibbons & Lindenmayer, 2002). This demands that all hollow-bearing trees on a given site are watched simultaneously by experienced volunteers (Lindenmayer *et al.*, 1991a). Given the logistical demands of our field survey method, in any given year we completed stagwatching surveys at an average of 35-45 of the 158 long-term field sites (see Lindenmayer *et al.*, 2003). We ensured that each year, the sites that we surveyed spanned a range of site and landscape conditions to avoid confounding year and other effects. We conducted stagwatching surveys in the following years: 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2006, 2007, 2009, 2010, 2011, 2012, 2013, 2014, 2015, 2016, 2017, and 2018.

As outlined above, observers are required to document the emergence of animals from each hollow-bearing tree on each site. However, since the 2009 wildfire, the number of sites with at least one hollow-bearing tree has been decreasing. When a site has no hollow-bearing trees, it was removed from the suite of sites targeted for surveys of arboreal marsupials (although it was still surveyed~~s~~ for the presence of hollow-bearing trees). ~~We subsequently made allowances for this procedure by using the predicted number of hollow-bearing trees for the sites that were unobserved.~~ Importantly, in past studies, we have used other methods to survey sites with no hollow-bearing trees and found that cavity-dependent arboreal marsupials are absent from these places (Lindenmayer *et al.*, 1991b).

Measurement of landscape disturbance

None of our 1 ha sites was logged, but the landscape surrounding some of our sites has been subject to clearcutting. The amount of logged forest in the landscape increased over time (Department of Environment, 2019). The exception was deep within closed water catchments and reserves (i.e. protected forests) where logging is excluded. We calculated a spatially-weighted proportion of 25 m x 25 m pixels logged within a 2500 m x 2500 m square surrounding each survey site in the previous 20 years. We selected landscapes of this size for two reasons. First, radio-tracking studies indicate that animals such as Leadbeater's possum may move at least 600 m from their nest sites (Lindenmayer *et al.*, 2017b) with other species also making occasional long distance movements (e.g. the mountain brushtail possum; (How, 1972)). Second, multiple interconnected colonies and groups of animals with home ranges spanning 1-60 ha in size are likely to respond to forest conditions in the landscape surrounding our sites.

For each site, we defined the following spatially weighted proportion of a square subject to logging:

$$L_{it} = \frac{\sum_k \sum_l w_{kl} lp_{itkl}}{\sum_k \sum_l w_{kl}}$$

where L_{it} is the calculated amount of logging at site *i* in survey year *t*, lp_{itkl} is 1 if the $(k,l)^{th}$ 25 m x 25 m pixel of site *i* in year *t* is logged and 0 if not, and the weights, w_{kl}, are constructed from two different Gaussian kernel weight functions, that is,

$$w_{kl} = e^{-\phi d_{kl}} e^{-\tau \Delta_{kl}}$$

where d_{kl} is the distance between pixel (*k,l*) and the origin (i.e. the centre of the 2500 m x 2500 m square) with spatial kernel scale parameter, ϕ, and Δ_{kl} is the time lag in years between the current survey and the logging that occurred in pixel (k,l) with associated temporal kernel scale parameter, τ. Details of our choice of scale parameters are given in Appendix 2, and they were set based on the assumption that logged areas close to our long-term sites would have a stronger effect on animals than more distant logged places. We set the scale parameters so that effects would diminish over time (as logged stands regenerated) with advanced regrowth forests potentially supporting animals or facilitating their movement through the landscape. We therefore set the temporal weighting factor to be 1 in the year after an area was logged and 0.01 30 years after a site had been harvested. Notably, we used different scale parameters for the hollow bearing tree analysis compared to that for the occurrence of animals.

The wildfire that burnt during February-March 2009 was the only major fire that occurred during our study. Using spatial data obtained from the Government of Victoria on forest cover following the 2009 fires (Department of Environment and Primary Industries, 2014), we calculated a spatially-weighted proportion of 25 m x 25 m pixels burned in a 2500 m x 2500 m square surrounding each survey site in a broadly analogous fashion to that described above for logging with similar Gaussian kernel scale parameters (see Appendix 2). Specifically, the definition is as follows:

$$F_{it} = \frac{\sum_k \sum_l w_{kl} fp_{itkl}}{\sum_k \sum_l w_{kl}}$$

where F_{it} is the calculated amount of fire in the landsacpe at site *i* in survey year *t*, fp_{itkl} is 1 if the $(k,l)^{th}$ 25 m x 25 m pixel of site *i* in year *t* was burned and 0 if not, and the weights, w_{kl} are defined in an analogous fashion to the logging weights.

Prior to 2009, all sites were assigned a value of zero for this variable. Although we measured whether each 1 ha site had been burned (or not) in the 2009 wildfires, we did not include this covariate in subsequent statistical analyses. This was because it was highly correlated with the extent of fire in the surrounding landscape.

STATISTICAL ANALYSES

We describe the analysis of two-distinct, but inter-related processes; **(1)** the factors associated with the number of hollow bearing trees at a site (Q1 and Q3), and **(2)** how the number of hollow bearing trees and other factors influenced the occurrence of arboreal marsupials (Q2 and Q4). We conducted these analyses separately due to differences in the sampling regime for hollow-bearing trees and marsupial occurrence. These differences in sampling were unavoidable because **(1)** the labour-intensive nature of stagwatching means it was not possible to undertake marsupial surveys at all sites in each year (see above for more details), and **(2)** stagwatching surveys cannot be conducted at sites that no longer support at least one hollow bearing tree. The removal of sites lacking hollow-bearing trees from the sampling regime for marsupials introduced a slight, but unavoidable, positive bias to the marsupial occurrence probability of occurrence through time. For this reason, we retained all sites in our analysis of hollow-bearing trees, which allows us to draw inference across the broader landscape and remove the ~~basis~~ bias induced by our method of sampling arboreal marsupials. As a final step in our analysis we combined estimates from both the hollow-bearing tree and marsupial occurrence analyses in order to draw a landscape-level inference about marsupial occurrence.

Full model for count of hollow-bearing trees (Q1 and Q3)

The count of the number of hollow bearing trees on sites occurred in 1997, 2005, 2009 (post fire), 2011, 2012, 2015 and 2017. Each site was surveyed on average 6.8 times with a low of two visits and a high of seven visits. Let HBT_{it} represent the number of hollow bearing trees occurs on site *i* ($i = 1, \dots, 164$) in year *t*. We modelled this process with a Bayesian Poisson regression with the following predictors: land tenure (1 if protected, 0 if wood production), survey year, amount of fire in the surrounding landscape and the amount of harvesting in the surrounding landscape. Thus, our model can be expressed as a generalized linear model as follows: Let:

$$HBT_{it} \sim Poisson(\mu_{it})$$

$$\eta_{it} = \beta_0 + u_i + \beta_1 F_{it} + \beta_2 L_{it} + \beta_3 LT_{it} + \beta_4 rs^1(SY_{it}) + \beta_5 rs^2(SY_{it}) + \beta_6 rs^3(SY_{it}) + \beta_7 rs^4(SY_{it})$$

$$u_i \sim N(0, \sigma_u)$$

$$log(\mu_{it}) \ = \eta_{it}$$

where, F_{it} is the fraction of forest burned in 2009, L_{it} is the fraction of forest that was logged, LT_t is the land tenure of site *I*, SY_{it} is the survey year, $rs^1(SY_{it})$ to $rs^4(SY_{it})$ are the basis functions for a cubic regression spline of survey year with four degrees of freedom, u_i is the site level random intercept with standard deviation σ_u and η_{it} is the linear predictor. We note that F_{it} was zero prior to the 2009 wildfire. For the models that included a survey year effect (see section on Model fitting & selection), we also looked at whether the spline could be simplified to a linear function of time. In addition, for the models with linear year effects, we also examined whether or not the addition of a random slope effect for year improved the model, with the slope of year allowed to vary according to site. Specifically, our linear predictor for this model is given by:

$$\eta_{it} = \beta_0 + \ u_i + \beta_1 F_{it} + \ \beta_2 L_{it} + \ \beta_3 LT_{it} \ + \ (\beta_4 + \ b_{i4})\ SY_{it}$$

$$u_i \sim N(0, \sigma_u)$$

$$b_{i4} \sim N(0, \sigma_{SY})$$

where u_i is the site level random intercept with standard deviation σ_u and b_{i4} is the site level random slope for survey year with standard deviation σ_{SY}.

Full model for marsupial occurrence (Q2 and Q4)

Conditional on their being at least one hollow bearing tree on a site (necessary to conduct stag-watching), we modelled the presence of each individual species of arboreal marsupial with a Bayesian logistic regression model. Specifically, let y_{it} represent the presence of the species on site *i* ($i = 1, \dots, 158$) in year *t* (*t=1997, 1998, 1999, 2000, 2001, 2002, 2003, 2006, 2007, 2009, 2010, 2011, 2012, 2013, 2014, 2015, 2016, 2017, 2018*) and let $\pi_{it} = Prob(y_{it} = 1)$. Each site was surveyed on average 4.8 times with a low of one and a high of ten visits. Our full logistic regression model is specified as follows:

$$y_{it} \sim Bernoulli(\pi_{it})$$

$$\psi_{it} = \beta_0 + v_i + \beta_1 F_{it} + \beta_2 L_{it} + \beta_3 LT_{it} + \beta_4 \log(HBT_{it}) + \beta_5 rs^1(SY_{it}) + \beta_6 rs^2(SY_{it}) + \beta_7 rs^3(SY_{it}) + \beta_8 rs^4(SY_{it})$$

$$v_i \sim N(0, \sigma_v)$$

$$logit(\pi_{it}) = \left(\frac{\pi_{it}}{1 - \pi_{it}}\right) = \psi_{it}$$

where HBT_{it} is the number of hollow bearing trees, v_i is site level random intercept with standard deviation σ_v. Note we are using $\log(HBT_{it})$ instead of HBT_{it} as preliminary analysis showed a better fit to the presence of arboreal marsupials. We note that F_{it} was zero prior to the 2009 wildfire. For the models that included a survey year effect, we also examined whether the spline could be simplified to a linear function of time.

Model fitting & selection

As the landscape factors we were interested in were time-varying, each analysis included two steps: **(1)** Fitting of a model including time as the only fixed effect (i.e. excluding landscape variables), to describe the temporal trends in HBT (Q1) and arboreal marsupials (Q2). And; **(2)** A model selection process to choose the best-fitting model of the effects of landscape factors on numbers of hollow-bearing trees (Q3) and arboreal marsupial occurrence (Q4). We completed model selection for each stage of our analysis using the widely applicable information criteria (WAIC) (Vehtari, Gelman & Gabry, 2017) to choose the best fitting model for each response variable. For the analysis of temporal trends in the number of hollow bearing trees and the occurrence of each species of arboreal marsupial (Q1 and Q2), we performed a model selection amongst various combinations of year discussed above, i.e. none, linear, linear with random slopes (number of hollow bearing trees only) and cubic regression spline of year with four degrees of freedom. For the full analysis (Q3 and Q4), we performed model selection on a set of 32 candidate models for the Poisson regression for the number of hollow bearing trees on a site (see Table S6) and 48 models for the occurrence of each species of arboreal marsupial (see Tables S8-S11). We elected to interpret the best fitting model (i.e. the one with the lowest WAIC) and the most parsimonious model (the simplest model within two WAIC units of the best fitting model) in each case.

All continuous variables were standardized to have zero mean and standard deviation one, this was done to aid with convergence of Markov Chain Monte Carlo (MCMC) algorithms and to aid with prior specification. We performed a prior sensitivity analysis for Bayesian logistic regression parameters on the logit scale using the following priors: 1) Student-t distribution with 7 degrees of freedom, zero mean and scale parameter of 2. (https://github.com/stan-dev/stan/wiki/Prior-Choice-Recommendations); 2) Student-t distribution with 7 degrees of freedom, zero mean and scale parameter of 1.414 (Northrup & Gerber, 2018); 3) Gaussian (normal) distribution with zero mean and standard deviation 1.414 (Northrup & Gerber, 2018); 4) Logistic distributions with zero mean and scale parameter of 1 (Northrup & Gerber, 2018); and 5) "flat" or non-informative priors. We used a half student-t distribution with three degrees of freedom, centred at zero and a scale parameter of ten, for the random effect standard deviation. The results of the prior sensitivity analysis are reported in Figs. S1-S4 and we proceeded with our model selection procedure with the "flat" or non-informative priors.

We constructed all models using the brms package (version 0.10.0) (Bürkner, 2017; Bürkner, 2018) in R 3.6.1 (R Core Team, 2018). We ran four Markov chains for 10,000 iterations, discarding a warmup of 2,000 with a thinning factor of eight giving a total 4,000 samples for posterior inference. We assessed convergence of the chains using the Rhat statistic (Gelman & Rubin, 1992), values less than 1.01 were deemed to have been adequately converged.

The best fitting logistic regression model for each species was used to predict the occurrence of each species for all years and sites. ~~In order to~~To account for the sites that were unobserved (for a given year site combination), we used the predicted number of hollow bearing trees from the best fitting Poisson regression model.

Model synthesis

We combined the hollow bearing tree analyses and the animals species occurrence analyses to quantify the overall effect of temporal changes in the hollow bearing tree resource on the occurrence of individual species of arboreal marsupials. Combining these analyses allowed us to account for the fact that not all sites were stag-watched in every year and also the conditional nature of the stagwatch sampling (i.e. if a site no longer supports any hollow-bearing trees, then we no longer conducted stagwatching surveys). Combining these analyses required that we assume sites with zero hollow-bearing trees supported zero arboreal marsupials.

To combine these analyses, we first generated posterior samples of the annual numbers of hollow-bearing trees for each of the 164 sites that were stagwatched between 1997 and 2018. These were generated from the posterior samples of the best fitting model for the number of hollow-bearing trees. Thus, we developed a complete, albeit model-based, data set of the number of hollow-bearing trees at each site from 1997-2018. We obtained values for tenure, amount of fire in the surrounding landscape, and amount of logging in the surrounding landscape for each site in each year; using the same methods as for the original models. We then derived posterior samples of the probability of occurrence for each species of arboreal marsupial for each posterior sample of the number of hollow bearing trees for each site and year combination. If the estimated number of hollow bearing trees at a site was zero, we assumed that site had a zero probability of animal occurrence. For computational reasons, we used 500 posterior samples for each stage and we present the annual estimate of occupancy for each species by averaging across the 158 sites.

RESULTS

We detected eight species of arboreal marsupials over the duration of this study (Table S2). There were sufficient data available for detailed analyses of four species; Leadbeater's possum, greater glider, mountain brushtail possum, and sugar glider. The yellow-bellied glider, common ringtail possum, eastern pygmy possum, and feathertail glider were too rarely recorded to allow statistical analyses. Descriptive information on the species of arboreal marsupials and the associated covariates is given in Tables S2 and S3. We provide descriptive information on the number of sites at which enumeration of hollow bearing trees was performed as well as basic information on covariates in Table S4 and Figs. S5-S7.

Q1 and Q2: What are the temporal trends in hollow-bearing trees and arboreal marsupials?

Analysis of time effects only revealed that the expected number of hollow bearing trees per site declined between 1997 and 2018 (Fig. 2a), as did the expected probability of occurrence of the four individual species of arboreal marsupials (Fig. 2b-2e), although the effect of time was marginal for the mountain brushtail possum (as measured by WAIC, Table S5).

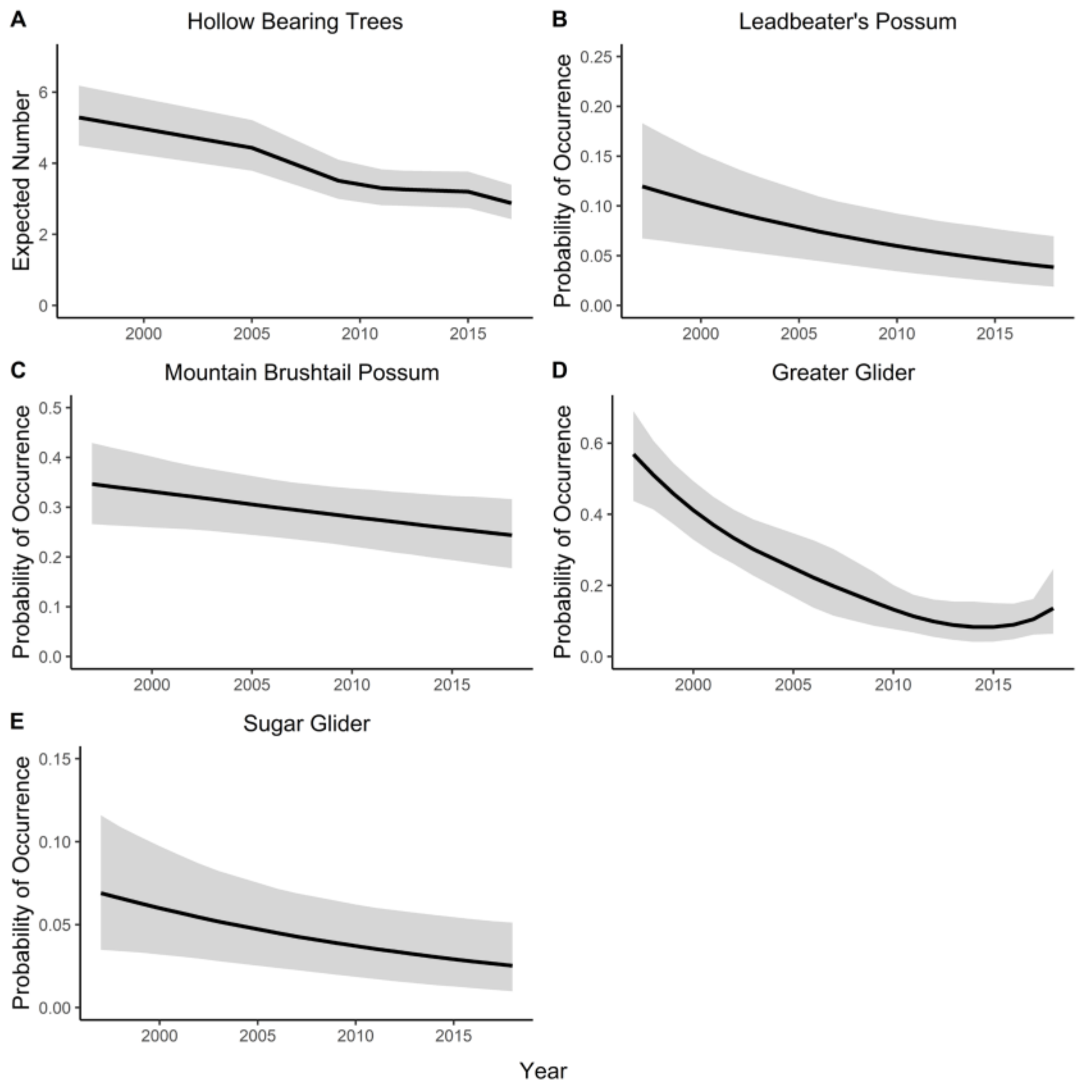


Fig. 2: Estimated temporal trends in the number of hollow bearing trees, and the occurrence of each species of arboreal marsupial. The grey shaded areas represent 95% credible intervals.

Q3: Is the number of hollow-bearing trees related to land tenure and landscape levels of wildfire and logging?

When landscape factors were included in the model (Table S7), the best-fitting model still included a downward temporal trend in the number of hollow-bearing trees (Fig. 3a). This model also revealed fewer hollow-bearing trees on sites where high proportions of the surrounding landscape had been burned (Fig. 3b), and on sites located in wood production forest (Fig. 3c).

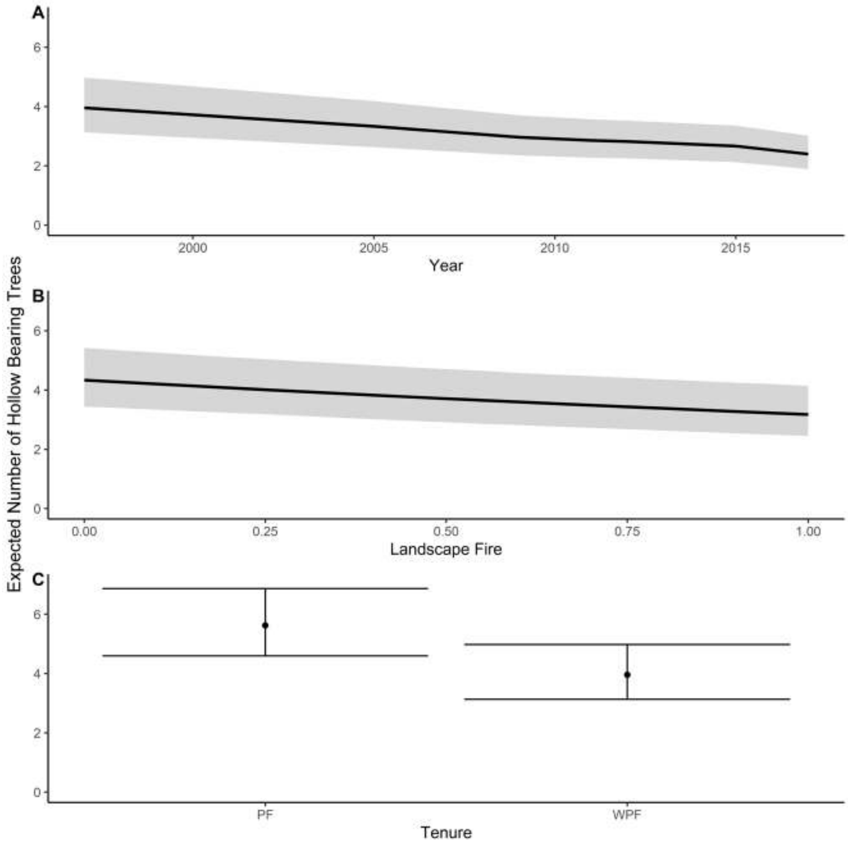


Fig. 3. Factors influencing the number of hollow-bearing tree (HBT) in the montane ash forests of the Central Highlands of Victoria. Values for other covariates in each of the models are held to the mean value. PF = protected forest. WPF = wood production forest. 95% credible intervals are indicated by the grey shaded areas and errors bars where appropriate.~~The grey shaded areas represent 95% credible intervals and error bars.~~

Q4: ***Is the occurrence of arboreal marsupials related to the number of hollow-bearing trees, land tenure, and landscape-levels of wildfire and logging?***

We found evidence of a strong positive relationship between the occurrence of all species of arboreal marsupials and the number of hollow-bearing trees at a site (Fig. 4; Tables

S8-S15). The probability of occurrence of Leadbeater's possum declined with increasing amounts of logging in the surrounding landscape, whereas we found the opposite effect for the sugar glider and the mountain brushtail possum (Fig. 4). The presence of Leadbeater's possum was higher on sites in wood production forests compared to protected forests (Fig. 4). Finally, the probability of occurrence of the two glider species was negatively associated with increasing amounts of fire in the landscape (Fig. 4). Top-ranked models for each species indicated that temporal trends in mountain brushtail possums and sugar gliders observed under Q2 (Fig. 2) were no longer important once landscape factors were included in the model, but remained important for Leadbeater's possum and the greater glider.

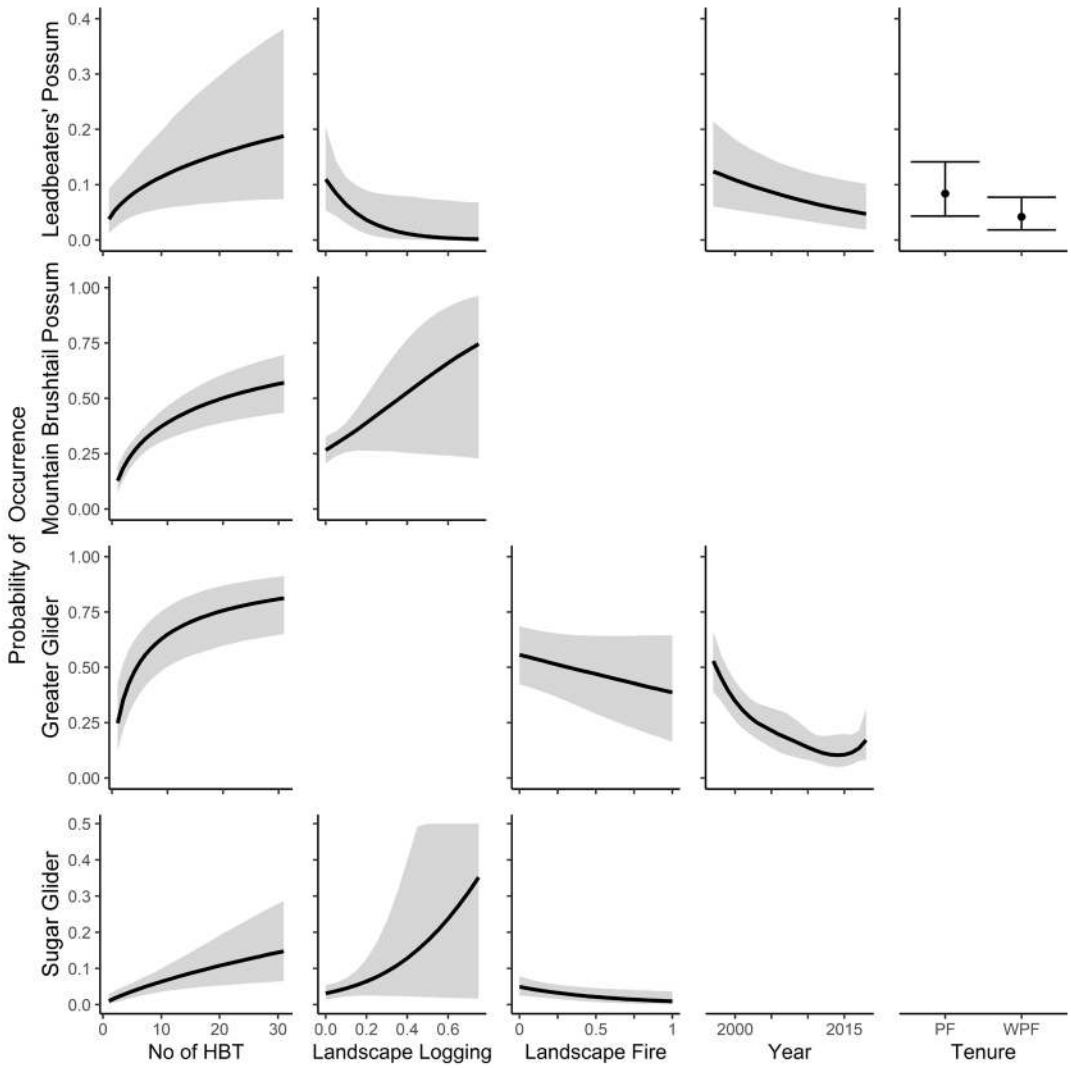


Fig. 4. The effects of the number of hollow bearing trees, landscape logging, landscape fire and year on the occurrence of four species of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria. The analysis for each species was conditional on the presence of at least one hollow bearing tree at the site. Values for other covariates in each of the models are held to the mean value. Blank plots in the grid correspond to where a particular covariate was not included in the top-ranked model for that species. Note that we have used different y-axis scales for each species. 95% credible intervals are indicated by the grey shaded areas and errors bars where appropriate.

~~**The grey shaded areas represent 95% credible intervals and error bars.**~~

Model synthesis

The final part of our analysis combined the model for the number of hollow-bearing trees with the analysis of the factors influencing the occurrence of each species of arboreal marsupial to estimate marsupial occurrence at all sites in all years of the study to estimate each species' temporal trend at the landscape scale. The results of this combined analysis closely resembled results from analysis of temporal trends in the marsupial occurrence data alone, and shows a decline in the occurrence of each of the four species of arboreal marsupial (Fig. 5). The combined analysis indicated a slightly stronger decline for the Leadbeater's possum than the occurrence only analysis, and also highlighted marked declines in the occurrence of both the sugar glider and greater glider, coincident with the 2009 wildfire.

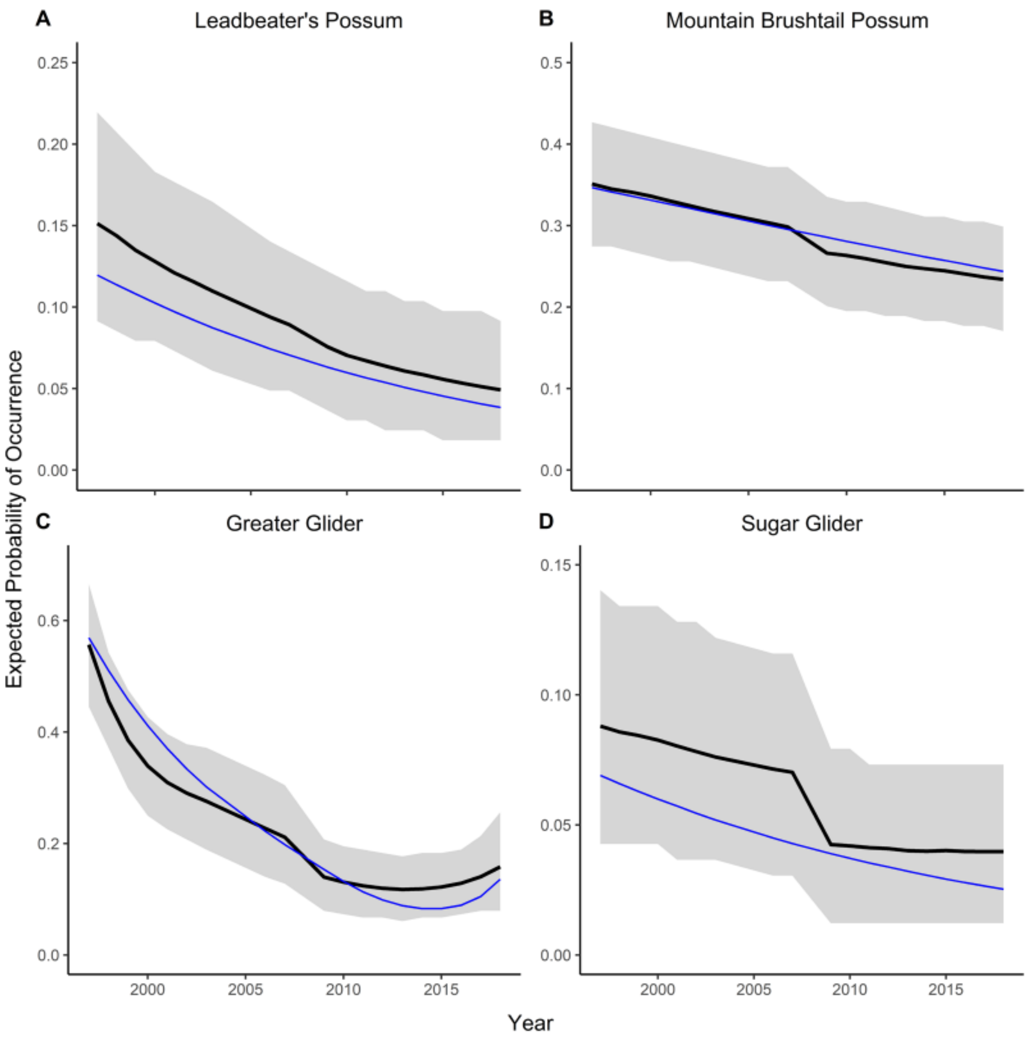


Fig. 5. The combined effects (shown by the black line) of the Poisson regression model for the number of hollow bearing trees at a site and the logistic regression model for the occurrence of each individual species of arboreal marsupial (see Methods). We also include the temporal trend from Fig. 2 (shown by the ~~blue~~ dashed line) for each species. 95% credible intervals are indicated by the grey shaded areas.

~~**The grey shaded areas represent 95% credible intervals.**~~

DISCUSSION

We examined long-term, large-scale temporal changes in the occurrence of arboreal marsupials and how they are associated with site-level and landscape-level variables. Our analyses revealed that: **(1)** The number of hollow-bearing trees on sites has declined since 1997 (Fig. 2). The number of these trees was lower on sites located in wood production areas compared to sites in water catchment areas. It was also lower where a large proportion of the surrounding landscape had burnt. **(2)** Populations of arboreal marsupials have declined since 1997 (Fig. 2) and also after accounting for other factors (e.g. the decline in hollow-bearing trees and the amount of disturbance in the landscape) as well as the inherent biases associated with the stagwatching method (Fig. 5). **(3)** The occurrence of all species of arboreal marsupials was strongly, positively linked to the number of large old hollow-bearing trees at a site. **(4)** There were marked inter-specific differences in species responses to landscape disturbance such as the amount of logging and the amount of fire surrounding a site. We further describe these findings in the remainder of this paper and conclude with commentary on the implications of our work for ~~of~~ the conservation of arboreal marsupials.

Q1 and Q3. What are the temporal trends in critical denning resources for arboreal marsupials? And: Is the number of hollow-bearing trees related to land tenure and landscape levels of wildfire and logging?

Our analyses contained strong evidence of a decline in the abundance of hollow-bearing trees (Fig. 2). This is a particular concern for the long-term persistence of arboreal marsupials, given the strong positive relationships between the number of hollow-bearing trees and the occurrence of all species that we modelled (see below). Previous work has shown that the fastest losses of hollow-bearing trees occur in regrowth stands aged 80 years or younger (Lindenmayer *et al.*, 2018a), where such trees are legacies of past old growth stands that were disturbed by wildfires (Lindenmayer *et al.*, 2019). Conversely, the slowest rate of decline is in old growth forests, but these comprise only 1.16% of the Mountain Ash estate and 0.47% of the Alpine Ash estate in the Central Highlands region.

We found that the decline in numbers of hollow-bearing trees was associated with the amount of fire in the landscape and land tenure. These results were expected given that fires can consume a large amount of forest, leading to altered microclimatic conditions (such as elevated windspeeds) which can increase the loss of large old trees (Lindenmayer *et al.* 2011). We recorded fewer large old trees on sites within wood production forests. Timber harvesting in logged landscapes intersperses cutblocks with uncut stands (including areas supporting our long-term sites). This can lead to landscape-level changes in windspeeds and increases windthrow (Lindenmayer, Cunningham & Donnelly, 1997), which is likely to be one of the key reasons for reduced numbers of large trees in such areas.

Q2. What are the temporal trends in the occurrence of arboreal marsupials?

We found evidence for a decline in the occurrence of all species of arboreal marsupials (Figs. 2 and 5). The effects remained after accounting for other factors (e.g. the decline in hollow-bearing trees and the amount of disturbance in the landscape) as well as the inherent biases associated with the stagwatching method (Fig. 5). The inter-specific patterns observed were broadly consistent with our predictions at the outset of this investigation. That is, the feeding specialist greater glider and range-restricted Leadbeater's possum exhibited the most marked declines (Figs. 2 and 5), whereas there was a marginal effect of time in the best fitting model for the widespread generalist, the mountain brushtail possum (Fig. 2). The most pronounced decline we observed was for the greater glider (Figs. 2 and 5). This trend is of considerable concern as it is mirrored in other parts of eastern Australia (Lindenmayer *et al.*, 2018c; Smith & Smith, 2018). Indeed, the greater glider used to be one of the most commonly detected species in field surveys in many areas of wet forest in eastern Australia. Its decline appears to be an example of a formerly common – but specialist species – rapidly becoming rare (Lindenmayer *et al.*, 2011). Losses in specialist species suggest that not only will the future arboreal marsupial assemblage in montane ash forests be characterized by fewer species, but also that it will be simplified with key niches vacated, such as those for a specialist arboreal folivore (greater glider). Other niches may have already been vacated, such as the large exudivore niche that would formerly have been occupied by the yellow-bellied glider, a species which is now rarely recorded (with only 59 animals detected in the past 20 years of surveys; see Table S5). There may well be cascading impacts of these losses on other species, such as large forest owls for which arboreal marsupials can be important prey items (Debus, Davies & Hollands, 2009).

Q4. Is the occurrence of arboreal marsupials related to the number of hollow-bearing trees, land tenure, and landscape-levels of wildfire and logging?

Our analyses contained evidence of a positive relationship between the number of hollow-bearing trees at a site and the occurrence of all four species of arboreal marsupials (Fig. 4). This result was expected given that arboreal marsupials are cavity-dependent and cannot persist in areas where hollow-bearing trees are absent.

The landscape surrounding our long-term field sites has undergone considerable change in the past 20 years as a result of wildfire (primarily in 2009) and ongoing clearcutting. Our analyses revealed that temporal patterns of site occurrence by some species of arboreal marsupials have been associated with these spatio-temporal changes in forest cover. We found evidence of a negative association between the occurrence of Leadbeater's possum at a site and the amount of logged forest in the surrounding landscape (Fig. 4). Such relationships are of considerable concern. Recent studies have indicated that proposed future logging in Victoria over the next 5-10 years (VicForests, 2019) will be focused disproportionately in high conservation value forests (Taylor & Lindenmayer, 2019). Therefore, additional planned logging will add to the extent of logged forest in the landscape and hence magnify logging-induced disturbance impacts on species such as Leadbeater's possum.

Whilst there was a negative relationship between the amount of logging in the landscape and Leadbeater's possum, both the mountain brushtail possum and the sugar glider exhibited a positive association with this covariate (Fig. 4). Indeed, a potential reason for the negative relationship between Leadbeater's possum and the amount of logging in the surrounding landscape may be competition with the sugar glider. Leadbeater's possum and the sugar glider are functionally similar species and have even been known to co-occupy the same nest trees (Lindenmayer & Meggs, 1996). However, the sugar glider is one of the most widely distributed marsupials globally, including being introduced to a number of areas outside its natural range (Lindenmayer, 2002). It is possible that negative responses of Leadbeater's possum to the amount of logged forest is linked with the positive response of the sugar glider to the same landscape attribute. Indeed, work in logged forests on the Australian island of Tasmania (where the sugar glider is an introduced species) has shown it often colonizes logged and regenerated forests (Allen *et al.*, 2018). A detailed co-occurrence analysis, coupled with radio-tracking and behavioural studies, would be required to determine if there are negative relationships between the two species in Victorian forests, although data for both species in montane ash forests were too sparse to allow such work to be completed for this investigation.

Our analyses indicated that sites in wood production forests were more likely to support Leadbeater's possum than sites in protected areas. This was an unexpected outcome, especially as wood production forests support fewer hollow-bearing trees than protected areas (Fig. 3) and such areas are subject to more logging. It is possible that this result occurred because, relative to protected areas, wood production forests tend to be located in more productive environments (Braithwaite, Turner & Key, 1984). Moreover, sites in wood production forests earmarked for future logging are also those which have high conservation value, including for species such as Leadbeater's possum (see Taylor & Lindenmayer, 2019).

We identified a negative relationship between the extent of fire in the landscape and the occurrence of the greater glider and sugar glider (Fig. 3). We suggest that this effect may be related to the high likelihood that animals are killed on-site by the high-intensity conflagrations that typically occur in montane ash forests, or indirectly through the loss of feeding resources in canopy consuming fire. In addition, increasing rates of collapse of hollow-bearing trees are associated with increasing areas of burned forest in the landscape (Fig. 3) and this may deplete nesting and den sites for cavity-dependent animals.

We found evidence of a decline for the mountain brushtail possum and the sugar glider (Fig 2), but the effects of time no longer remained in the best fitting model once other covariates were fitted. Hence, declines in the number of hollow-bearing trees and the amount of fire and logging in the landscape explained temporal declines in these species. Conversely, in the case of Leadbeater's possum and the greater glider, an effect of time was retained in the best fitting model after other covariates were fitted. This suggests that other factors that were not modelled are contributing to the ongoing decline of these species. For example, the greater glider is known to be heat-sensitive (Rubsamen *et al.*, 1984) and elevated temperatures in recent decades in this region may be contributing to declines (Lindenmayer *et al.*, 2011). Similarly, recent studies have indicated that Leadbeater's possum may be preyed upon by feral predators like cats (McComb *et al.*, 2018) although whether this threat has increased in recent decades within the study region remains unclear.

Management implications

The results of this study have important implications for conservation. First, our data highlight the major declines occurring in key species of conservation concern such as

Leadbeater's possum and the greater glider. The strength and consistency of these declines demonstrate that it is important to take effective conservation action now and not simply monitor these species until they suffer regional or even global extinction (see Lindenmayer, Piggott & Wintle, 2013). In the case of Leadbeater's possum, the importance of the amount of logged forest in the landscape indicates that ongoing logging will have further negative impacts on the species, as does our finding that this species is more likely to occur in areas earmarked for future logging. A moratorium on logging in landscapes where the species occurs is urgently required. The recent decision by the Victorian Government to halt logging by 2030 (Office of the Premier of Victoria, 2019) is a proactive step. However, given the trajectory of declines in these species of conservation concern, a more rapid cessation of logging is needed.

The negative effects of the extent of fire in the landscape on numbers of hollow-bearing trees and the occurrence of arboreal marsupials is a second issue of concern arising from our analyses. Montane ash forests are increasingly susceptible to high-severity wildfires, with five major conflagrations in the region in the past century. This frequency has been elevated relative to the historical record (which was previously an average of one major fire every 75-150 years (McCarthy, Gill & Lindenmayer, 1999)). We suggest that greater efforts are needed to reduce the occurrence of wildfire in montane ash forests. This may entail both: **(1)** reducing the spatial extent of logging operations which creates large areas of flammable young forest (Zylstra, 2018) and can lead to regenerating stands being more prone to crown-scorching fires (Taylor, McCarthy & Lindenmayer, 2014) and **(2)** protecting existing advanced regrowth forest and allowing it to mature through to old growth, as fire severity is lower in such stands (Lindenmayer & McCarthy, 1998).

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SUPPLEMENTARY MATERIALS

Methods S1: Spatial and temporal weighting functions

To anchor our spatio-temporal weighting scheme, we used the following points which relate directly to animal home range and temporal effects of fire/logging.

Spatial animal weighting functio

Distance (km)	Weight
0	1.00
0.1	0.99
0.2	0.95
1	0.5
2	0.01

Our spatial weighting function for animal occurrence on sites was developed using insights from radio-tracking and other field-based studies of animal movements, including inter-den movements (e.g. see Lindenmayer, McBurney, Blair *et al.*, 2017; Lindenmayer, Welsh & Donnelly, 1997b; Pope, Lindenmayer & Cunningham, 2004) as well as known home range sizes (see Table S1) and previous studies of the spatial effects of fire-derived disturbance on animal occurrence (Lindenmayer, Blanchard, McBurney *et al.*, 2013). The weighting function was set so that logged or burned areas close to our long-term sites had more pronounced effects on animals than more distant logged or burned places.

Temporal animal weighting function

Time Difference (Yrs)	Weight
0	1.0
15	0.5
30	0.01

We set the scale parameters so that effects of logging or fire in the landscape around our long-term sites would dissipate over time (as stands regenerated after logging or fire). There would be immediate effects of disturbance (such as clearcut logging) in adjacent areas, as documented for some species of arboreal marsupials in montane ash forests (Lindenmayer, Cunningham & Donnelly, 1993). However, over time, regrowth forests could potentially support animals or facilitate their movement through the landscape such as ~15 years after disturbance (provided that hollow-bearing trees are present) (Smith, Lindenmayer & Suckling, 1985; Lindenmayer & Possingham, 1996). Based on this understanding of species responses to disturbance we therefore set the temporal weighting factor to be 1 in the year after an area was logged and 0.01 30 years after a site had been harvested.

Spatial hollow-bearing trees weighting function

Distance (km)	Weight
0	1.00
0.1	0.99
0.2	0.95
1	0.5
2	0.2

We used different scale parameters for the hollow bearing tree analysis compared to that for the occurrence of animals. However, some of the key principles were similar to those underpinned our analysis for animals. That is, disturbed areas close to our sites were considered likely to have a more marked effect on temporal changes in the abundance of hollow-bearing trees than disturbed areas further away. We based on our spatial weighting function on work showing that elevated rates of collapse of large old hollow-bearing trees can occur as a result of areas being clearcut close to uncut forest (Lindenmayer, Cunningham & Donnelly, 1997a) and increasing amounts of disturbance (logging and/or fire) in the landscape (Lindenmayer, Blanchard, Blair et al., 2018).

Temporal hollow-bearing trees weighting function

Time Difference (Yrs)	Weight
0	1.0
1	0.99
10	0.75
30	0.01

We set a temporal weighting function for hollow-bearing trees so that the effects of landscape disturbance on the risk of collapse dissipated over time as the forest regenerated. That is, hollow-bearing trees on sites adjacent to areas recently disturbed would be at high risk of collapse (Lindenmayer *et al.*, 1997a) with negligible effects after 30 years (when stands of montane ash trees would be approximately 40 m tall).

We fitted a Gaussian Kernel to the weights outlined above (by least squares) which produced the following estimates of ϕ using the relationship:

$$e^{-\phi}d^2$$

The estimates are given in the following table:

Case	Estimate
Animal spatial weighting	1.243
Animal temporal weighting	0.0050
Hollow-bearing tree spatial weighting	0.268
Hollow-bearing tree temporal weighting	0.0051

We note that the phi's are on different scales, i.e. spatial versus temporal weighting functions. Graphical representation of the fits are given in the following figure with the response curves for animals shown in the upper two diagrams and those for hollow-bearing trees in the lower two diagrams.

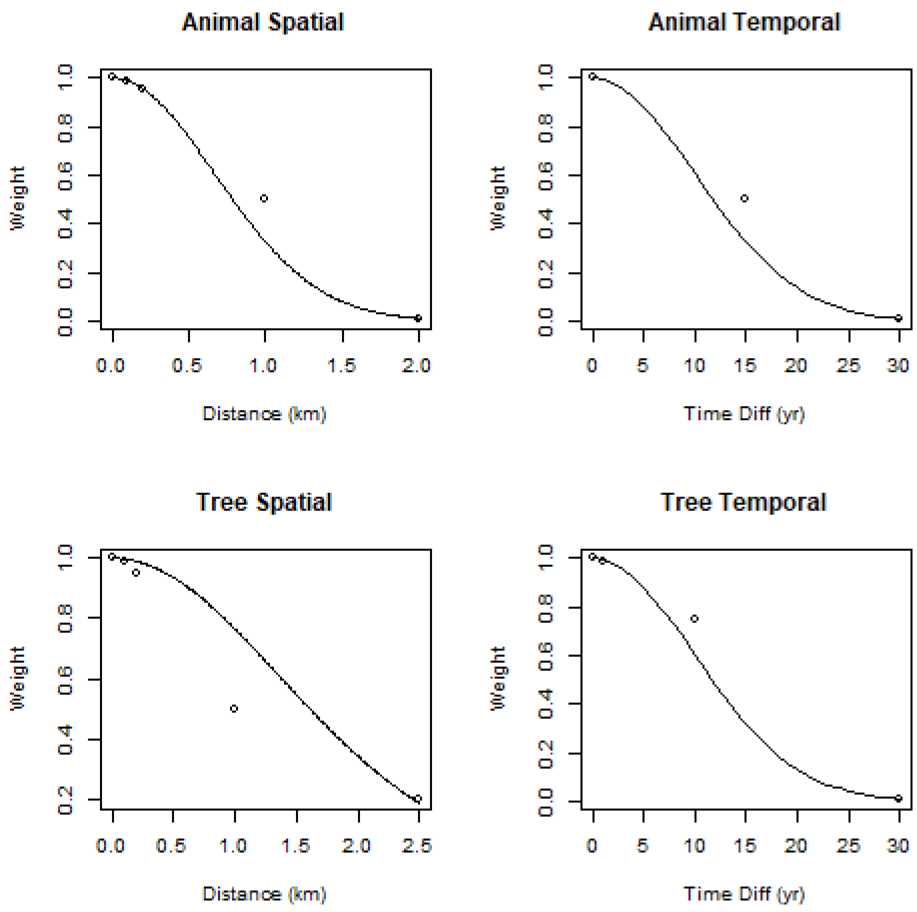

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Table S1. Life history attributes of the eight species of arboreal marsupials recorded in the montane ash forests of the Central Highlands of Victoria, south-eastern Australia. Summarized from Lindenmayer (1997).

Species	Mean body size (grams)	Home range (ha)	Social system	Litter size
Leadbeater's Possum	140	1-3	Colonial	2
Sugar Glider	130	1-5	Colonial	2
Yellow-bellied Glider	650	30-60	Colonial	1
Greater Glider	1200	1-2	Solitary	1
Common Ringtail Possum	800	0.8	Colonial	1-4
Mountain Brushtail Possum	3000	4-6	Solitary or pairs	1
Feathertail Glider	15	0.8	Colonial	4
Eastern Pygmy Possum	25	1	Colonial	2-4

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Table S2: Number of sites and the total number of animals emerging from hollow bearing trees by year and species. Note the species analysed are highlighted in bold. Note, NSD= number of sites the at least one animal was seen emerging from a hollow bearing tree and TD = the total number of animals emerging from all hollow bearing trees across all sites in a given year.

	No Sites	**Leadbeater's Possum**		**Mountain Brushtail Possum**		Common Ringtail Possum		Eastern Pygmy Possum		**Greater Glider**		**Sugar Glider**		Yellow-Bellied Glider		Feathertail Glider	
	Surveyed	NSD	TD	NSD	TD	NSD	TD	NSD	TD	NSD	TD	NSD	TD	NSD	TD	NSD	TD
1997	74	10	33	27	40	1	2	1	1	45	98	8	17	0	0	0	0
1998	84	20	46	25	41	3	3	0	0	31	46	6	11	6	12	1	1
1999	50	4	7	19	30	0	0	0	0	28	53	8	9	1	3	0	0
2000	52	8	13	24	52	2	3	0	0	22	42	5	5	2	2	1	1
2001	55	8	15	23	57	0	0	0	0	20	37	2	2	4	13	0	0
2002	38	4	16	16	34	2	2	0	0	9	13	2	2	2	3	0	0
2003	43	10	15	15	31	3	4	0	0	10	16	1	1	2	5	0	0
2006	21	4	14	10	21	1	1	0	0	8	13	3	8	2	4	1	1
2007	19	3	10	9	16	0	0	0	0	2	2	2	4	0	0	1	1
2009	41	6	16	15	26	0	0	0	0	8	11	1	2	1	1	0	0
2010	31	2	6	8	11	0	0	0	0	2	4	0	0	0	0	0	0
2011	34	3	7	13	21	0	0	0	0	7	12	4	8	1	1	1	1
2012	22	1	2	4	7	0	0	0	0	5	7	2	4	2	2	2	5
2013	31	3	13	11	24	0	0	0	0	2	5	2	4	1	2	1	3
2014	30	1	2	7	12	1	1	0	0	5	10	0	0	0	0	0	0
2015	39	3	5	15	21	0	0	0	0	7	9	1	2	0	0	0	0
2016	23	1	2	8	13	1	2	0	0	4	5	1	2	1	2	0	0
2017	26	3	4	7	16	0	0	0	0	4	5	1	3	2	6	1	1
2018	40	2	6	7	13	0	0	0	0	8	10	3	8	1	3	1	1
Total	753	96	232	263	486	14	18	1	1	227	398	52	92	28	59	10	15

Table S3: Descriptive statistics for the number of sites surveyed in each year with the corresponding covariates (Tenure, number of hollow bearing trees (HBT), landscape fire, and landscape logging). Note, the value for the landscape fire variable is zero prior to the 2009 wildfire and constant thereafter.

Survey	**Number of**	**Tenure:**	**HBT**				**Landscape Fire**				**Landscape Logging**			
Year	Sites	No Protected	Min	Median	Mean	Max	Min	Median	Mean	Max	Min	Median	Mean	Max
1997	74	36	1	6	6.96	19	0	0	0	0	0	0.05	0.1	0.75
1998	84	51	1	6.5	8.32	31	0	0	0	0	0	0	0.05	0.31
1999	50	29	2	6	7.66	24	0	0	0	0	0	0.02	0.07	0.32
2000	52	28	1	6.5	7.65	19	0	0	0	0	0	0.02	0.07	0.38
2001	55	31	2	7	7.87	19	0	0	0	0	0	0.02	0.06	0.28
2002	38	21	1	7	8	19	0	0	0	0	0	0.03	0.07	0.59
2003	43	21	1	6	7.44	19	0	0	0	0	0	0.03	0.07	0.21
2006	21	13	1	5	7.05	20	0	0	0	0	0	0.05	0.07	0.2
2007	19	13	2	7	7.84	19	0	0	0	0	0	0.01	0.04	0.19
2009	41	27	1	4	5.68	27	0	0.58	0.53	1	0	0.02	0.06	0.24
2010	31	13	1	3	3.39	13	0	0	0.31	1	0	0.09	0.11	0.41
2011	34	19	1	5	6.44	21	0	0.33	0.42	1	0	0	0.05	0.27
2012	22	13	1	8.5	9.27	22	0	0.27	0.39	1	0	0.05	0.08	0.29
2013	31	18	1	8	9.74	30	0	0.66	0.55	1	0	0.03	0.08	0.34
2014	30	8	1	2	3.2	13	0	0.01	0.28	1	0	0.11	0.1	0.33
2015	39	27	1	4	5.54	29	0	0.45	0.47	1	0	0	0.03	0.2
2016	23	14	1	5	6.13	13	0	0	0.29	1	0	0.03	0.07	0.26
2017	26	12	1	3.5	4.81	19	0	0.55	0.49	1	0	0.06	0.09	0.31
2018	40	21	1	3.5	5.2	19	0	0.49	0.48	1	0	0.03	0.08	0.25

Table S4: Descriptive statistics for the number of hollow bearing trees (HBT) and the amount of landscape logging and fire by year. Of the 164 sites, 73 are in timber production areas and 91 are in protected areas. Note, the amount of landscape fire was zero prior to 2009.

			Landscape Logging				**Landscape Fire**			
Year	No of Sites	No Sites with Zero HBT	Min	Mean	Median	Max	Min	Mean	Median	Max
1997	150	0	0	0.052	0.019	0.552	0	0	0	0
2005	153	1	0	0.053	0.02	0.298	0	0	0	0
2009	161	19	0	0.054	0.024	0.343	0	0.459	0.407	1
2011	163	20	0	0.058	0.027	0.402	0	0.444	0.379	0.98
2012	160	22	0	0.056	0.023	0.388	0	0.435	0.379	0.955
2015	162	26	0	0.059	0.027	0.382	0	0.375	0.321	0.833
2017	163	34	0	0.055	0.019	0.416	0	0.327	0.28	0.722

Table S5: Model selection results WAIC (widely applicable information criteria) for choosing the temporal trend for the number of hollow bearing (HBT) trees and the temporal trend for each Arboreal marsupial species. Where LBP=Leadbeater's Possum, MBP=Mountain Brushtail Possum, GG=Greater Glider, SG=Sugar Glider. Note all models have a random effect of site, and RY indicates the model has random slopes for the linear effect of year and random intercepts depending on site. The best fitting model is indicated in bold.

Model	HBT	LBP	MBP	GG	SG
Intercept	4572.08	536.40	896.51	874.84	371.88
YearLinear	4305.53	**528.69**	**895.15**	794.51	**367.78**
YearLinear + RY	**4041.08**	NA	NA	NA	NA
YearSpline	4299.12	533.33	898.26	**788.51**	374.47

Table S6: Model selection results using the widely applicable information criteria (WAIC) for the Bayesian Poisson regression model for the number of hollow bearing trees (HBT). Where YearSpline is a cubic regression with 4 degrees of freedom, YearLinear is the linear effect of year, tenure is land tenure (categorical with two levels: wood production and protected), Fire is landscape fire, Logging is landscape logging, RI indicates the model has random intercepts which vary by site and RY indicates the model has random slopes which vary by site. The best fitting model is in **bold** and the most parsimonious model is indicated in *italics.*

No	Model	WAIC	Delta WAIC
31	**YearLinear + Tenure + Fire + RI + RY**	**4014.34**	**0**
32	YearSpline + Tenure + Fire + Logging + RI + RY	4014.95	0.61
29	YearLinear + Fire + Logging + RI + RY	4015.22	0.88
27	*YearLinear + Fire + RI + RY*	*4015.45*	*1.11*
28	YearLinear + Tenure + RI + RY	4040.23	25.89
30	YearLinear + Tenure + Logging + RI + RY	4041	26.67
25	YearLinear + RI + RY	4041.8	27.46
26	YearLinear + Logging + RI + RY	4042.25	27.91
23	YearLinear + Tenure + Fire + RI	4275.76	261.42
19	YearLinear + Fire + RI	4276.07	261.73
24	YearSpline + Tenure + Fire + Logging + RI	4277.6	263.26
15	YearSpline + Tenure + Fire + RI	4277.96	263.63
21	YearLinear + Fire + Logging + RI	4279.34	265
9	YearSpline + Fire + RI	4279.84	265.5
16	YearSpline + Tenure + Fire + Logging + RI	4280.12	265.78
12	YearSpline + Fire + Logging + RI	4283.12	268.78
11	YearSpline + Tenure + RI	4298.32	283.98
4	YearSpline + RI	4299.12	284.78
14	YearSpline + Tenure + Logging + RI	4300.33	285.99
7	YearSpline + Logging + RI	4301.1	286.76
20	YearLinear + Tenure + RI	4304.35	290.01
17	YearLinear + RI	4305.53	291.19
18	YearLinear + Logging + RI	4306.4	292.06
22	YearLinear + Tenure + Logging + RI	4306.8	292.47
10	Tenure + Fire + RI	4383.76	369.42
13	Tenure + Fire + Logging + RI	4386.16	371.82
3	Fire + RI	4386.2	371.86
6	Fire + Logging + RI	4387.95	373.61
8	Tenure + Logging + RI	4567.94	553.6
2	Logging + RI	4569.75	555.41
5	Tenure + RI	4570.74	556.4
1	Intercept + RI	4572.08	557.74

Table S7: Model coefficients for the best fitting and the most parsimonious logistic regression models for the probability of having at least one hollow bearing tree on a site post the 2009 wild fire. Note (S) means the variable was standardized to mean 0 and standard deviation 1 before modelling. Where year is the survey year, tenure is land tenure (categorical with two levels: wood production and protected), fire is landscape fire and logging is landscape logging. Note the best fitting and most parsimonious model coincide.

	Estimate	Lower 95 CI	Upper 95 CI	Rhat
Best Fitting				
Fixed Effects:				
Intercept	1.11	0.91	1.32	1
Tenure	0.16	-0.07	0.38	1
Year (S)	-0.27	-0.32	-0.22	1
Fire (S)	-0.15	-0.2	-0.09	1
Random Effects:				
Sd(Intercept)	1.01	0.89	1.15	1
Sd(Year)	0.23	0.19	0.28	1
Cor (Intercept, Year)	0.88	0.79	0.96	1
Most Parsimonious (Delta WAIC = 1.11)				
Fixed Effects:				
Intercept	1.2	1.03	1.37	1
Year (S)	-0.27	-0.33	-0.22	1
Fire (S)	-0.15	-0.2	-0.1	1
Random Effects:				
Sd(Intercept)	1.02	0.9	1.15	1
Sd(Year)	0.23	0.18	0.28	1
Cor (Intercept, Year)	0.89	0.8	0.96	1

Table S8: Model selection results using the widely applicable information criteria (WAIC) for the Bayesian logistic regression model for the conditional occurrence, on at least one hollow bearing tree being presence of Leadbeater's Possum. The model with the lowest WAIC is highlighted in **bold** and the most parsimonious model is given in *italics*. Where LogNoHBT = Log of the number of hollow bearing trees. YearSpline is the cubic regression spline with four degrees of freedom for year and fire and logging are refer to landscape fire and logging respectively. Note, in this case, the most parsimonious model and the best fitting model coincide, though LogNoHBT + Fire + Logging is very close, but it is equal in complexity. YearSpline + LogNoHBT+ Logging and YearSpline + Tenure + Logging are also quite close but are more complex due to the inclusion of the spline for year.

No	Model	WAIC	Delta WAIC
46	**YearLinear + LogNoHBT+ Tenure + Logging**	**520.28**	**0**
39	*YearLinear + Tenure + Logging*	*521.61*	*1.33*
40	*YearLinear + LogNoHBT+ Logging*	*521.96*	*1.69*
20	LogNoHBT+ Tenure + Logging	523.17	2.89
30	YearSpine + LogNoHBT+ Tenure + Logging	523.17	2.9
48	YearLinear + LogNoHBT+ Tenure + Fire + Logging	523.17	2.9
34	YearLinear + Logging	523.61	3.33
27	LogNoHBT+ Tenure + Fire + Logging	524.1	3.83
22	YearSpine + LogNoHBT+ Logging	524.46	4.18
18	LogNoHBT+ Fire + Logging	524.53	4.25
45	YearLinear + LogNoHBT+ Fire + Logging	524.59	4.32
21	YearSpine + Tenure + Logging	524.73	4.46
44	YearLinear + Tenure + Fire + Logging	524.81	4.54
32	YearSpine + LogNoHBT+ Tenure + Fire + Logging	524.94	4.66
38	YearLinear + Fire + Logging	525.2	4.92
9	LogNoHBT + Logging	525.4	5.13
10	YearSpine + Logging	525.82	5.55
29	YearSpine + LogNoHBT+ Fire + Logging	526.07	5.79
37	YearLinear + LogNoHBT	526.24	5.97
28	YearSpine + Tenure + Fire + Logging	526.35	6.08
12	LogNoHBT+ Fire	527.29	7.02
43	YearLinear + LogNoHBT+ Tenure	527.48	7.2
17	Tenure + Fire + Logging	527.53	7.26
42	YearLinear + LogNoHBT+ Fire	527.58	7.3
19	YearSpine + Fire + Logging	527.9	7.62
5	LogNoHBT	528.44	8.17
7	Fire + Logging	528.49	8.21
33	YearLinear	528.69	8.41
47	YearLinear + LogNoHBT+ Tenure + Fire	528.78	8.5
23	LogNoHBT+ Tenure + Fire	528.97	8.7
16	YearSpine + LogNoHBT	529.14	8.86
36	YearLinear + Tenure	529.62	9.34
14	LogNoHBT+ Tenure	529.67	9.4
8	Tenure + Logging	529.76	9.49
25	YearSpine + LogNoHBT+ Fire	529.79	9.51
35	YearLinear + Fire	529.96	9.68
41	YearLinear + Tenure + Fire	530.39	10.12
26	YearSpine + LogNoHBT+ Tenure	530.68	10.41
31	YearSpine + LogNoHBT+ Tenure + Fire	531.4	11.13
3	Fire	531.71	11.43
11	Tenure + Fire	532.02	11.75
2	Logging	532.12	11.85
13	YearSpine + Fire	532.86	12.59
6	YearSpine	533.33	13.05
24	YearSpine + Tenure + Fire	533.47	13.19
15	YearSpine + Tenure	534.31	14.04
1	Intercept	536.4	16.12
4	Tenure	537.87	17.59

Table S9. Model selection results using the widely applicable information criteria (WAIC) for the Bayesian logistic regression model for the occurrence, on at least one hollow bearing tree being presence, of Mountain Brushtail Possum. The model with the lowest WAIC is highlighted in **bold** and the most parsimonious model is given in *italics*. Where LogNoHBT = Log of the number of hollow bearing trees, tenure refers to land tenure, YearSpline is the cubic regression spline with four degrees of freedom for year and fire and logging are refer to landscape fire and logging respectively.

No	Model	WAIC	Delta WAIC
9	**LogNoHBT + Logging**	**885.53**	**0**
20	LogNoHBT+ Tenure + Logging	885.78	0.25
5	*LogNoHBT*	*886.19*	*0.66*
14	LogNoHBT+ Tenure	886.53	1
18	LogNoHBT+ Fire + Logging	886.53	1
27	LogNoHBT+ Tenure + Fire + Logging	887.45	1.92
12	LogNoHBT+ Fire	887.69	2.16
40	YearLinear + LogNoHBT+ Logging	887.87	2.33
23	LogNoHBT+ Tenure + Fire	887.88	2.35
37	YearLinear + LogNoHBT	888.1	2.57
45	YearLinear + LogNoHBT+ Fire + Logging	888.13	2.6
43	YearLinear + LogNoHBT+ Tenure	888.31	2.78
42	YearLinear + LogNoHBT+ Fire	889.24	3.71
46	YearLinear + LogNoHBT+ Tenure + Logging	889.31	3.78
47	YearLinear + LogNoHBT+ Tenure + Fire	889.52	3.99
48	YearLinear + LogNoHBT+ Tenure + Fire + Logging	889.6	4.07
22	YearSpine + LogNoHBT+ Logging	891.05	5.52
26	YearSpine + LogNoHBT+ Tenure	891.85	6.32
29	YearSpine + LogNoHBT+ Fire + Logging	891.89	6.36
16	YearSpine + LogNoHBT	891.91	6.38
32	YearSpine + LogNoHBT+ Tenure + Fire + Logging	892.53	7
25	YearSpine + LogNoHBT+ Fire	892.63	7.1
30	YearSpine + LogNoHBT+ Tenure + Logging	892.83	7.3
31	YearSpine + LogNoHBT+ Tenure + Fire	892.87	7.34
3	Fire	894.26	8.73
11	Tenure + Fire	894.74	9.21
7	Fire + Logging	894.75	9.22
17	Tenure + Fire + Logging	894.9	9.37
35	YearLinear + Fire	895.38	9.85
33	YearLinear	895.45	9.92
38	YearLinear + Fire + Logging	895.59	10.06
34	YearLinear + Logging	895.86	10.33
41	YearLinear + Tenure + Fire	896.26	10.73
1	Intercept	896.51	10.97
36	YearLinear + Tenure	896.56	11.03
44	YearLinear + Tenure + Fire + Logging	896.78	11.25
4	Tenure	897.16	11.63
39	YearLinear + Tenure + Logging	897.33	11.8
19	YearSpine + Fire + Logging	897.44	11.91
2	Logging	897.62	12.09
13	YearSpine + Fire	897.96	12.43
8	Tenure + Logging	898.21	12.68
6	YearSpine	898.26	12.73
24	YearSpine + Tenure + Fire	898.59	13.06
28	YearSpine + Tenure + Fire + Logging	898.77	13.24
15	YearSpine + Tenure	899.25	13.72
10	YearSpine + Logging	899.34	13.81
21	YearSpine + Tenure + Logging	900.01	14.48

Table S10: Model selection results using the widely applicable information criteria (WAIC) for the Bayesian logistic regression model for the occurrence, on at least one hollow bearing tree being presence, of Greater Glider. The model with the lowest WAIC is highlighted in **bold** and the most parsimonious model is given in *italics*. Where LogNoHBT = Log of the number of hollow bearing trees, tenure refers to land tenure, YearSpline is the cubic regression spline with four degrees of freedom for year and fire and logging are refer to landscape fire and logging respectively.

No	Model	WAIC	Delta WAIC
25	**YearSpine + LogNoHBT+ Fire**	**771.7**	**0**
16	*YearSpine + LogNoHBT*	*772.95*	*1.25*
31	YearSpine + LogNoHBT+ Tenure + Fire	773.15	1.44
29	YearSpine + LogNoHBT+ Fire + Logging	773.47	1.77
26	YearSpine + LogNoHBT+ Tenure	773.59	1.89
22	YearSpine + LogNoHBT+ Logging	774.38	2.68
32	YearSpine + LogNoHBT+ Tenure + Fire + Logging	774.43	2.72
30	YearSpine + LogNoHBT+ Tenure + Logging	775.85	4.15
42	YearLinear + LogNoHBT+ Fire	777.02	5.31
45	YearLinear + LogNoHBT+ Fire + Logging	778.73	7.03
47	YearLinear + LogNoHBT+ Tenure + Fire	779.02	7.31
37	YearLinear + LogNoHBT	779.18	7.48
48	YearLinear + LogNoHBT+ Tenure + Fire + Logging	779.94	8.24
43	YearLinear + LogNoHBT+ Tenure	780.61	8.91
46	YearLinear + LogNoHBT+ Tenure + Logging	781.33	9.62
40	YearLinear + LogNoHBT+ Logging	781.36	9.66
24	YearSpine + Tenure + Fire	784.24	12.54
13	YearSpine + Fire	784.79	13.09
28	YearSpine + Tenure + Fire + Logging	786.4	14.7
19	YearSpine + Fire + Logging	787.06	15.36
6	YearSpine	788.93	17.23
35	YearLinear + Fire	789.08	17.38
41	YearLinear + Tenure + Fire	789.5	17.8
44	YearLinear + Tenure + Fire + Logging	790.16	18.45
38	YearLinear + Fire + Logging	790.25	18.54
15	YearSpine + Tenure	790.5	18.8
10	YearSpine + Logging	790.64	18.94
21	YearSpine + Tenure + Logging	791.99	20.28
33	YearLinear	794.51	22.81
36	YearLinear + Tenure	794.57	22.87
34	YearLinear + Logging	795.92	24.22
39	YearLinear + Tenure + Logging	796.96	25.26
12	LogNoHBT+ Fire	799.61	27.9
18	LogNoHBT+ Fire + Logging	799.72	28.02
23	LogNoHBT+ Tenure + Fire	800.34	28.64
27	LogNoHBT+ Tenure + Fire + Logging	801.06	29.36
11	Tenure + Fire	820.09	48.39
17	Tenure + Fire + Logging	820.84	49.14
3	Fire	821.26	49.55
7	Fire + Logging	822.91	51.21
5	LogNoHBT	829.36	57.66
9	LogNoHBT + Logging	830.32	58.62
14	LogNoHBT+ Tenure	830.97	59.27
20	LogNoHBT+ Tenure + Logging	832.02	60.31
1	Intercept	874.84	103.13
4	Tenure	876.28	104.57
2	Logging	876.29	104.59
8	Tenure + Logging	877.86	106.16

Table S11: Model selection results using the widely applicable information criteria (WAIC) for the Bayesian logistic regression model for the occurrence, on at least one hollow bearing tree being presence, of Sugar Glider. The model with the lowest WAIC is highlighted in **bold** and the most parsimonious model is given in *italics*. Where LogNoHBT = Log of the number of hollow bearing trees, tenure refers to land tenure, YearSpline is the cubic regression spline with four degrees of freedom for year and fire and logging are refer to landscape fire and logging respectively.

No	Model	WAIC	Delta WAIC
18	**LogNoHBT+ Fire + Logging**	**359.24**	**0**
37	*YearLinear + LogNoHBT*	*359.66*	*0.43*
12	*LogNoHBT+ Fire*	*359.76*	*0.52*
23	LogNoHBT+ Tenure + Fire	360.19	0.95
43	YearLinear + LogNoHBT+ Tenure	360.52	1.28
9	*LogNoHBT + Logging*	*360.57*	*1.34*
40	YearLinear + LogNoHBT+ Logging	360.61	1.37
5	*LogNoHBT*	*360.68*	*1.44*
27	LogNoHBT+ Tenure + Fire + Logging	360.81	1.57
45	YearLinear + LogNoHBT+ Fire + Logging	361.06	1.82
14	LogNoHBT+ Tenure	361.15	1.91
46	YearLinear + LogNoHBT+ Tenure + Logging	361.22	1.98
42	YearLinear + LogNoHBT+ Fire	361.78	2.54
20	LogNoHBT+ Tenure + Logging	361.91	2.67
47	YearLinear + LogNoHBT+ Tenure + Fire	362.47	3.23
48	YearLinear + LogNoHBT+ Tenure + Fire + Logging	363.23	3.99
16	YearSpine + LogNoHBT	366.32	7.08
26	YearSpine + LogNoHBT+ Tenure	366.51	7.27
22	YearSpine + LogNoHBT+ Logging	366.92	7.68
30	YearSpine + LogNoHBT+ Tenure + Logging	367.28	8.04
25	YearSpine + LogNoHBT+ Fire	367.41	8.17
3	Fire	367.6	8.36
33	YearLinear	367.78	8.54
36	YearLinear + Tenure	368.24	9
29	YearSpine + LogNoHBT+ Fire + Logging	368.29	9.05
35	YearLinear + Fire	368.83	9.59
34	YearLinear + Logging	368.88	9.64
7	Fire + Logging	368.98	9.74
32	YearSpine + LogNoHBT+ Tenure + Fire + Logging	369.2	9.97
31	YearSpine + LogNoHBT+ Tenure + Fire	369.22	9.98
38	YearLinear + Fire + Logging	369.23	9.99
11	Tenure + Fire	369.49	10.25
41	YearLinear + Tenure + Fire	369.71	10.47
39	YearLinear + Tenure + Logging	370.08	10.84
17	Tenure + Fire + Logging	370.55	11.31
44	YearLinear + Tenure + Fire + Logging	370.85	11.61
1	Intercept	371.88	12.64
4	Tenure	372.59	13.35
2	Logging	372.91	13.67
8	Tenure + Logging	373.59	14.35
6	YearSpine	374.47	15.23
15	YearSpine + Tenure	375.07	15.83
10	YearSpine + Logging	375.22	15.98
13	YearSpine + Fire	375.35	16.11
19	YearSpine + Fire + Logging	375.99	16.76
21	YearSpine + Tenure + Logging	376.12	16.88
24	YearSpine + Tenure + Fire	376.33	17.09
28	YearSpine + Tenure + Fire + Logging	377.66	18.42

Table S12: Model coefficients for the best fitting and most parsimonious model for the logistic regression model for the conditional occurrence (on being at least one hollow bearing tree) Leadbeater's Possum. Note (S) means the variable was standardized to mean 0 and standard deviation 1 before modelling. Where LogNoHBT = Log of the number of hollow bearing trees, tenure refers to land tenure (0=wood production forests and 1=protected forests) and Logging is the amount of landscape logging surrounding the site.

	Estimate	Lower 95 CI	Upper 95 CI	Rhat
Best Fitting				
Fixed Effects:				
Intercept	-3.26	-4.37	-2.3	1
Year (S)	-0.36	-0.65	-0.07	1
LogNoHBT	0.52	0.08	0.99	1
Tenure	-0.75	-1.64	0.09	1
Logging (S)	-0.43	-0.89	-0.02	1
Random Effects:				
Sd (Intercept)	1.35	0.87	1.92	1
Most Parsimonious (Delta WAIC = 1.69)				
Fixed Effects:				
Intercept	-3.55	-4.6	-2.64	1
Year (S)	-0.35	-0.64	-0.07	1
LogNoHBT	0.48	0.05	0.93	1
Logging (S)	-0.24	-0.63	0.11	1
Random Effects:				
Sd (Intercept)	1.32	0.84	1.9	1
Most Parsimonious (Delta WAIC = 1.33)				
Fixed Effects:				
Intercept	-2.45	-3.16	-1.83	1
Year (S)	-0.45	-0.74	-0.17	1
Tenure	-0.65	-1.51	0.19	1
Logging (S)	-0.45	-0.91	-0.05	1
Random Effects:				
Sd (Intercept)	1.37	0.91	1.92	1

Table S13: Model coefficients for the best fitting and most parsimonious model for the logistic regression model for the conditional occurrence (on being at least one hollow bearing tree) Mountain Brushtail Possum. Note (S) means the variable was standardized to mean 0 and standard deviation 1 before modelling. Where LogNoHBT = Log of the number of hollow bearing trees, tenure refers to land tenure, and logging are refers to landscape logging.

	Estimate	Lower 95 CI	Upper 95 CI	Rhat
Best Fitting				
Fixed Effects:				
Intercept	-1.93	-2.51	-1.41	1
logNoOfHBT	0.65	0.37	0.94	1
Logging (S)	0.2	-0.03	0.42	1
Random Effects:				
Sd(intercept)	1.07	0.78	1.4	1
Most Parsimonious (Delta WAIC = 0.66)				
Fixed Effects:				
Intercept	-1.87	-2.44	-1.33	1
logNoOfHBT	0.61	0.33	0.89	1
Random Effects:				
Sd(intercept)	1.09	0.79	1.41	1

Table S14. Model coefficients for the best fitting and most parsimonious model for the logistic regression model for the conditional occurrence (on being at least one hollow bearing tree) Greater Glider. Note (S) means the variable was standardized to mean 0 and standard deviation 1 before modelling. Where LogNoHBT = Log of the number of hollow bearing trees, tenure refers to land tenure, YearSpline is the cubic regression spline with four degrees of freedom for year and fire refers to landscape fire.

	Estimate	Lower 95 CI	Upper 95 CI	Rhat
Best fitting				
Intercept	-1.12	-1.97	-0.28	1
YearSplineT1	-1.26	-2.37	-0.13	1
YearSplineT2	-1.45	-3.34	0.44	1
YearSplineT3	-2.72	-4.47	-0.93	1
YearSplineT4	-1.71	-2.64	-0.78	1
logNoOfHBT	0.75	0.4	1.14	1
Fire (S)	-0.22	-0.55	0.08	1
Sd(Intercept)	1.29	0.95	1.7	1
Most parsimonious (Delta WAIC = 1.25)				
Intercept	-1.04	-1.9	-0.23	1
YearSplineT1	-1.24	-2.38	-0.13	1
YearSplineT2	-1.37	-3.26	0.52	1
YearSplineT3	-3.14	-4.83	-1.47	1
YearSplineT4	-1.84	-2.72	-0.95	1
logNoOfHBT	0.77	0.41	1.13	1
Sd(Intercept)	1.25	0.91	1.65	1

Table S15: Model coefficients for the best fitting and most parsimonious model for the logistic regression model for the conditional occurrence (on being at least one hollow bearing tree) Sugar Glider. Note (S) means the variable was standardized to mean 0 and standard deviation 1 before modelling. Where LogNoHBT = Log of the number of hollow bearing trees, tenure refers to land tenure, and fire and logging are refer to landscape fire and logging respectively.

	Estimate	Lower 95 CI	Upper 95 CI	Rhat
Best fitting				
Intercept	-4.61	-5.96	-3.5	1
logNoOfHBT	0.83	0.33	1.39	1
Fire (S)	-0.54	-1.12	-0.08	1
Logging (S)	0.26	-0.08	0.58	1
Sd(Intercept)	0.89	0.17	1.55	1
Most parsimonious 1 (Delta WAIC = 0.43)				
Intercept	-4.59	-5.96	-3.44	1
Year (S)	-0.26	-0.61	0.06	1
logNoOfHBT	0.81	0.29	1.39	1
Sd(Intercept)	1.09	0.42	1.75	1
Most parsimonious 2 (Delta WAIC = 0.52)				
Intercept	-4.54	-5.87	-3.38	1
logNoOfHBT	0.78	0.27	1.33	1
Fire (S)	-0.55	-1.19	-0.08	1
Sd(Intercept)	0.96	0.19	1.61	1
Most parsimonious 3 (Delta WAIC = 1.34)				
Intercept	-4.68	-6.05	-3.55	1
logNoOfHBT	0.91	0.42	1.48	1
Logging (S)	0.27	-0.06	0.58	1
Sd(Intercept)	0.97	0.25	1.65	1
Most parsimonious 4 (Delta WAIC = 1.44)				
Intercept	-4.64	-6.07	-3.48	1
logNoOfHBT	0.87	0.37	1.44	1
Sd(Intercept)	1.05	0.37	1.72	1

Figure S1: Prior sensitivity of the full model for Leadbeater's Possum to the choice of prior distribution. We plot posterior density estimates for each of the model parameters including the random effects variance for each of 5 prior distributions: 1) Student-t with 7 degrees of freedom and scale parameter 2.5, 2) Student-t with 7 degrees of freedom and scale parameter 1.414, 3) Normal distribution with standard deviation 1.414, 4) Logistic distribution with scale parameter 1 and 5) a "flat" prior. Where, log(HBT) is the log of the number of hollow bearing trees, LT is land tenure (1=protected, 0=wood production), Fire is the amount of fire in the surrounding landscape, Logging is the amount of logging in the surrounding landscape, RS1(SY) through RS4(SY) are regression spline basis functions and sd(Int) is the standard deviation of the random site effect.

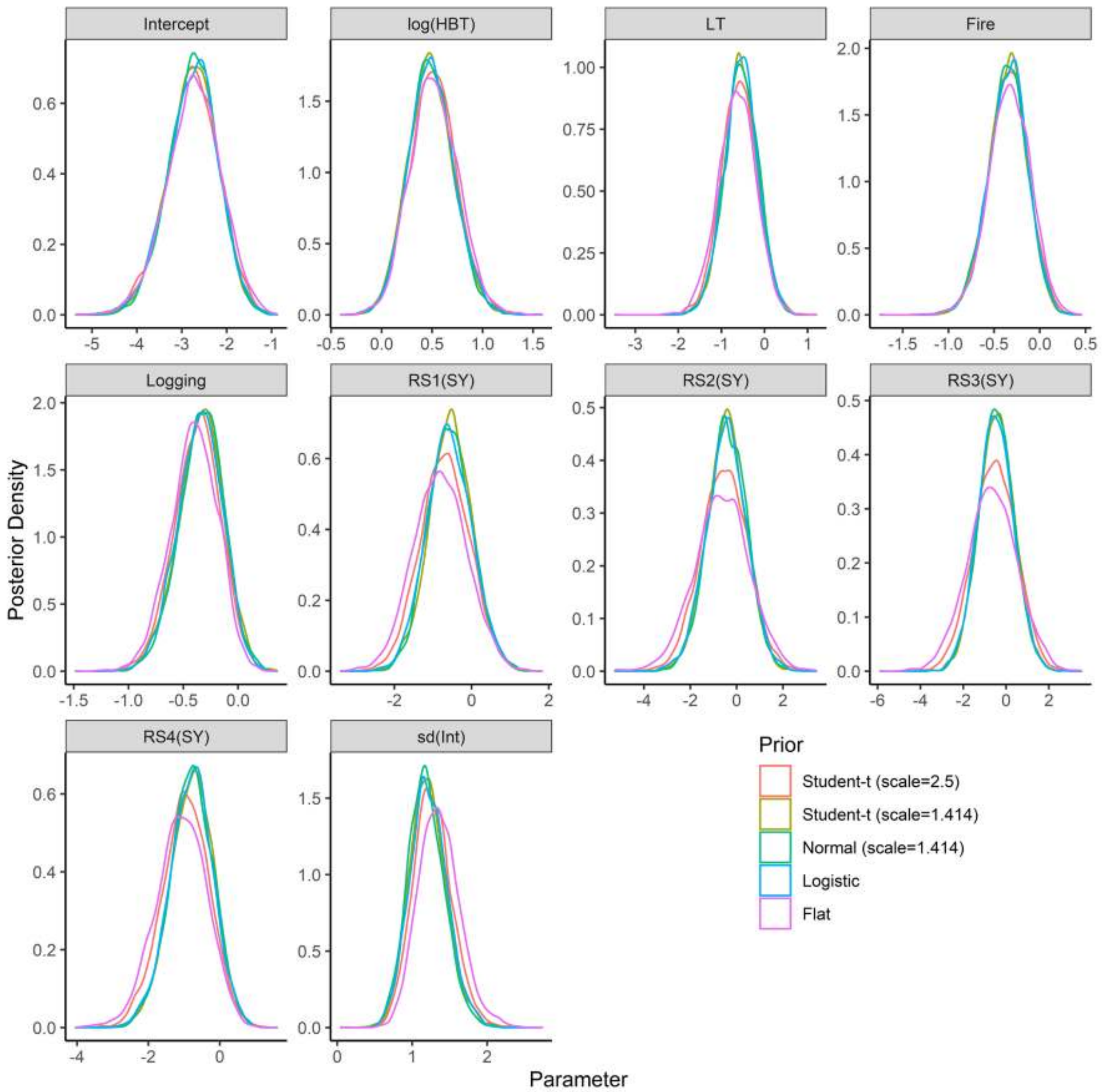

Figure S2: Prior sensitivity of the full model for Mountain Brushtail Possum to the choice of prior distribution. We plot posterior density estimates for each of the model parameters including the random effects variance for each of 5 prior distributions: 1) Student-t with 7 degrees of freedom and scale parameter 2.5, 2) Student-t with 7 degrees of freedom and scale parameter 1.414, 3) Normal distribution with standard deviation 1.414, 4) Logistic distribution with scale parameter 1 and 5) a "flat" prior. Where, log(HBT) is the log of the number of hollow bearing trees, LT is land tenure (1=protected, 0=wood production), Fire is the amount of fire in the surrounding landscape, Logging is the amount of logging in the surrounding landscape, RS1(SY) through RS4(SY) are regression spline basis functions and sd(Int) is the standard deviation of the random site effect.

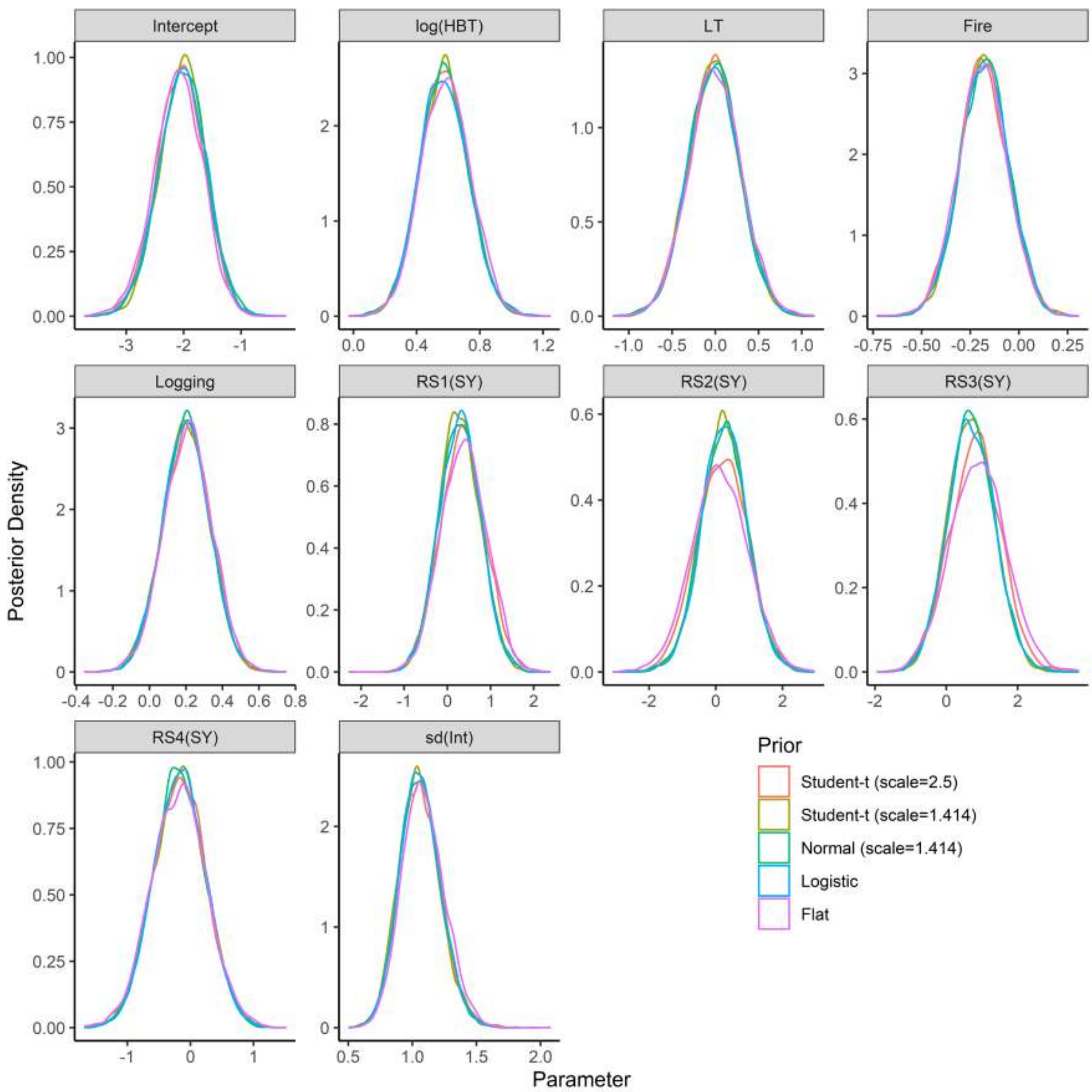

Figure S3: Prior sensitivity of the full model for Greater Glider to the choice of prior distribution. We plot posterior density estimates for each of the model parameters including the random effects variance for each of 5 prior distributions: 1) Student-t with 7 degrees of freedom and scale parameter 2.5, 2) Student-t with 7 degrees of freedom and scale parameter 1.414, 3) Normal distribution with standard deviation 1.414, 4) Logistic distribution with scale parameter 1 and 5) a "flat" prior. Where, log(HBT) is the log of the number of hollow bearing trees, LT is land tenure (1=protected, 0=wood production), Fire is the amount of fire in the surrounding landscape, Logging is the amount of logging in the surrounding landscape, RS1(SY) through RS4(SY) are regression spline basis functions and sd(Int) is the standard deviation of the random site effect.

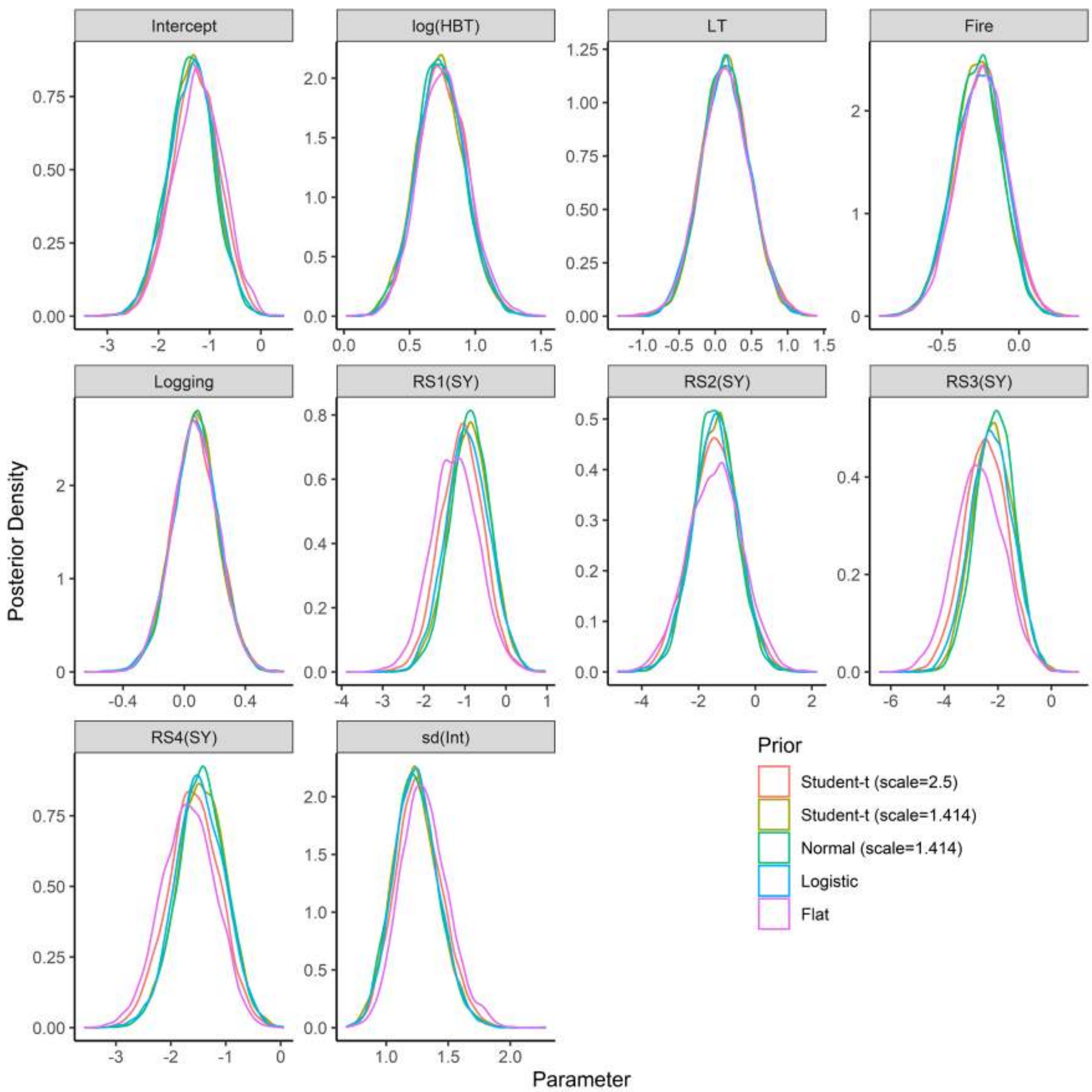

Figure S4: Prior sensitivity of the full model for Sugar Glider to the choice of prior distribution. We plot posterior density estimates for each of the model parameters including the random effects variance for each of 5 prior distributions: 1) Student-t with 7 degrees of freedom and scale parameter 2.5, 2) Student-t with 7 degrees of freedom and scale parameter 1.414, 3) Normal distribution with standard deviation 1.414, 4) Logistic distribution with scale parameter 1 and 5) a "flat" prior. Where, log(HBT) is the log of the number of hollow bearing trees, LT is land tenure (1=protected, 0=wood production), Fire is the amount of fire in the surrounding landscape, Logging is the amount of logging in the surrounding landscape, RS1(SY) through RS4(SY) are regression spline basis functions and sd(Int) is the standard deviation of the random site effect.

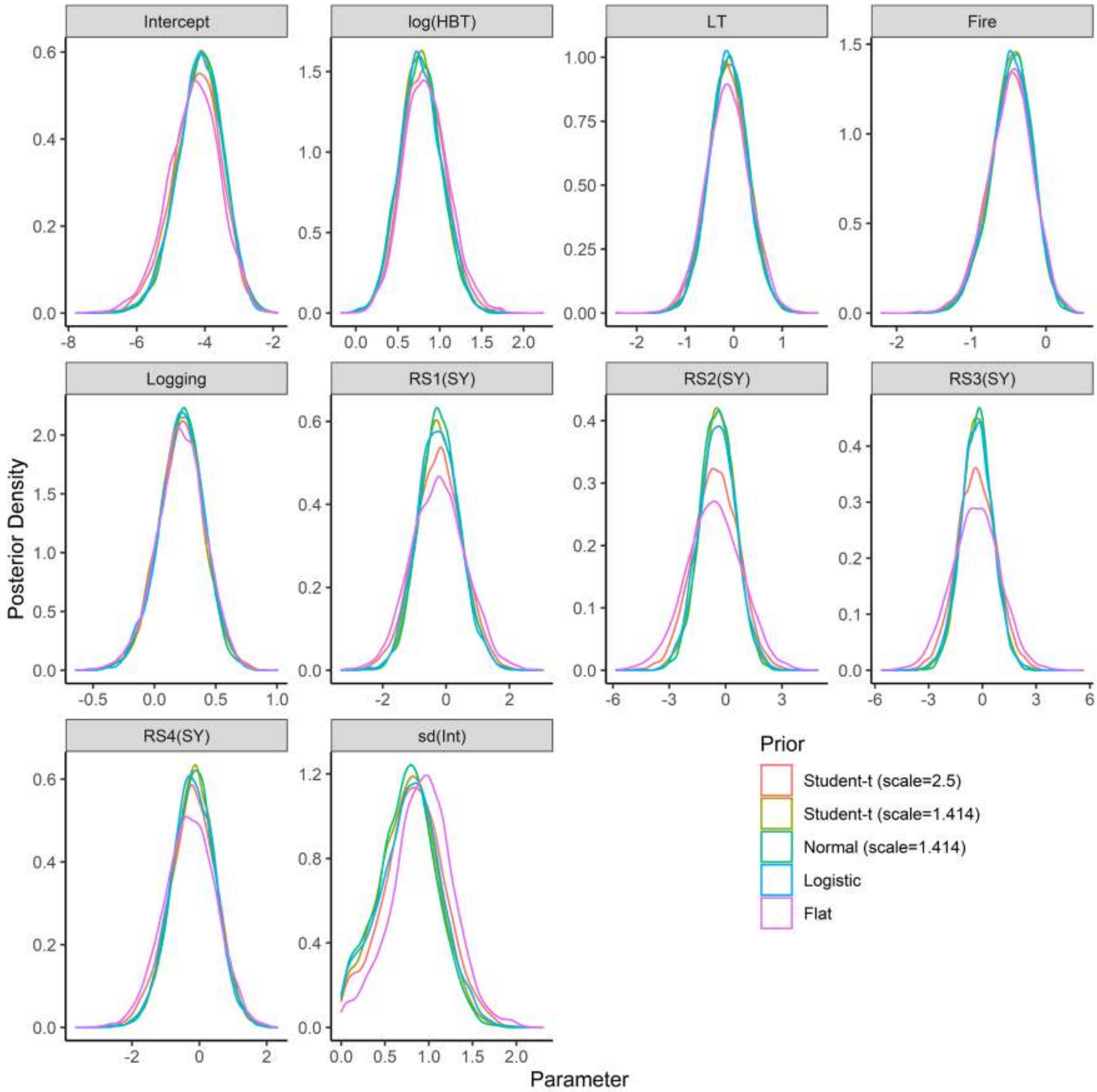

Figure S5: Boxplots of the number of hollow bearing trees(HBT) per site by year. The number of sites with zero HBTs are 0, 1, 19, 20, 22, 26, 34 for 1997, 2005, 2009, 2011, 2012, 2015 and 2017, respectively.

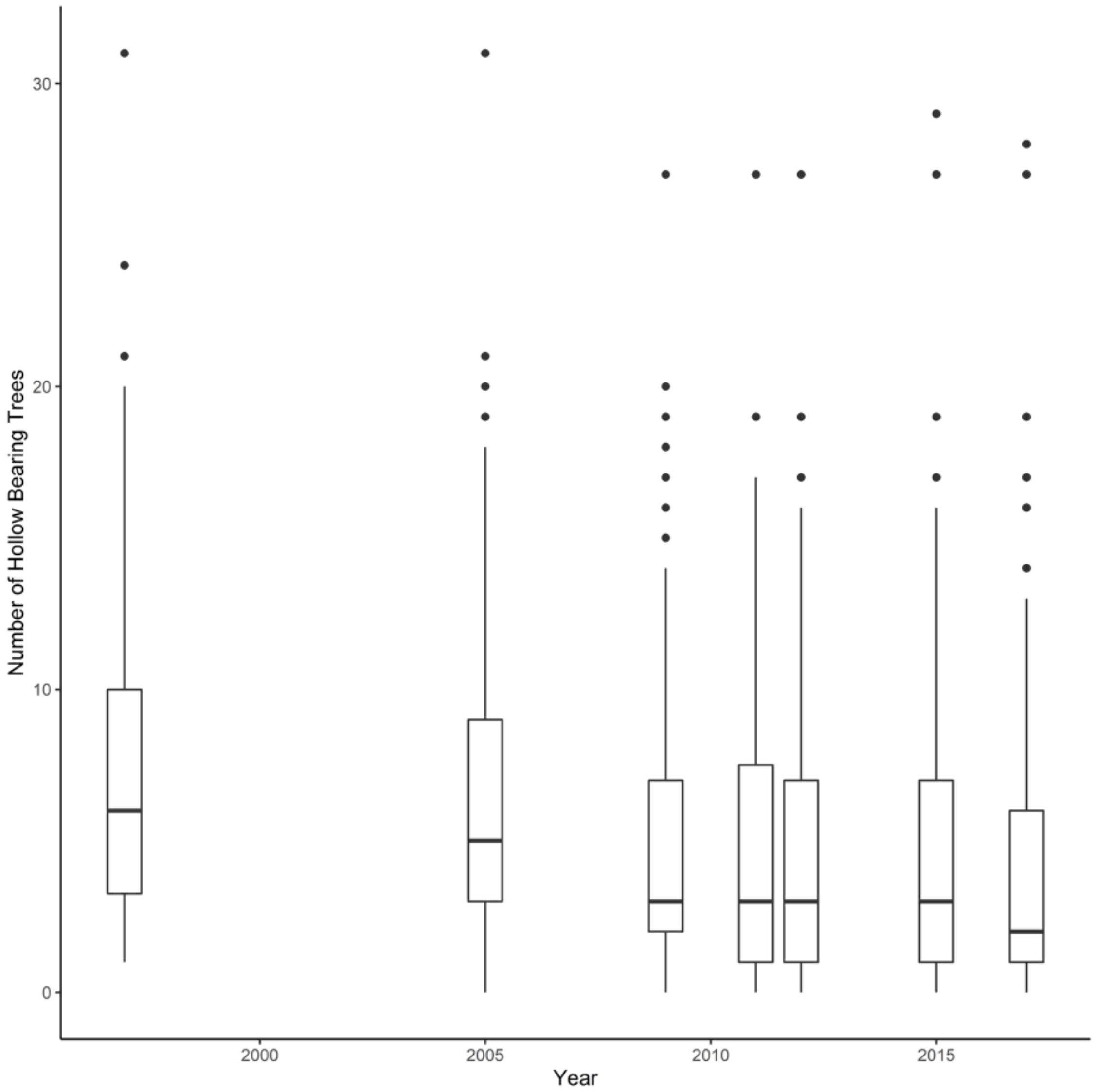

Figure S6: Boxplots of the distribution of logging in the landscape by year.

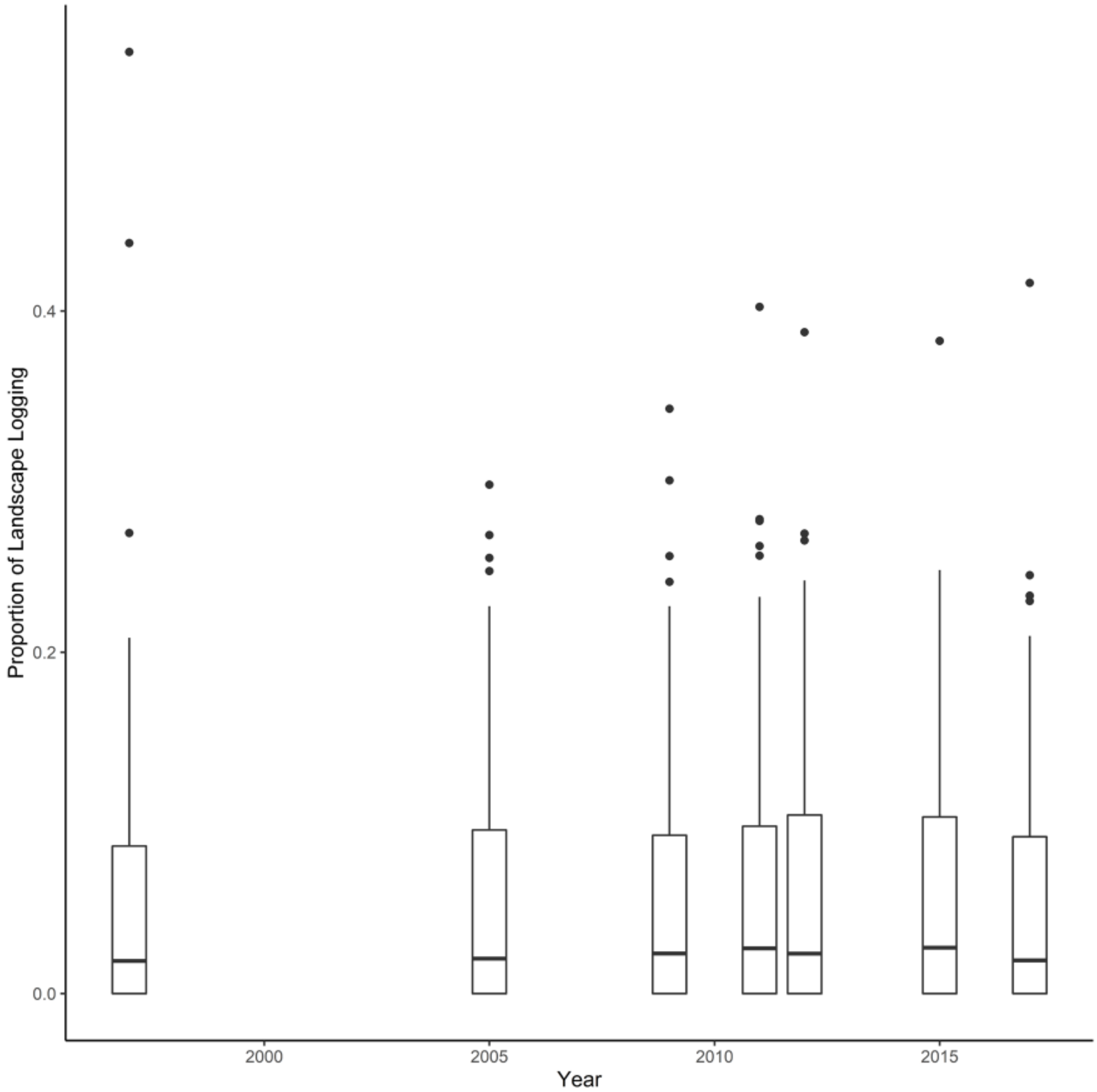

Figure S7: Boxplots of the distribution of fire in the landscape by year.