Yellow-bellied Glider Occupancy Trends on the Bago Plateau (1995 - 2023)

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Summary

This report updates a previous analysis of yellow-bellied glider occupancy on the Bago plateau between 1995 and 2020; this earlier report highlighted a substantial decline in the first year after the black summer fire (Bilney et al. 2022). Here we include a further three years of primarily acoustic monitoring to assess the extent of recovery after the fires. Initial occupancy was positively associated with the extent of montane gum and wet peppermint gum, but negatively associated with the extent of western forest types. Rainfall in the 12-months preceding sampling was positively associated with site colonisation probability. The extent of high severity fire was positively associated with extinction probability leading to 10 % decrease in occupancy observed immediately post-fire. The long-term trend has been characterised by generally low occupancy in the first 25 years of monitoring with small increases observed between 1995 and 2019, followed by a 10 % reduction post-fire in 2020/21. Occupancy then increased dramatically in 2022 and this continued in 2023. Thus the influence of high severity fire did not persist, as substantial recovery followed with years of above average rainfall. Harvesting was not associated with the trend in occupancy. Passive acoustic monitoring has resulted in greater levels of detection and a more precise estimate of occupancy than spotlighting used in the early part of the study. Given the current high level of occupancy and that there has been no decline in occupancy other than immediately post-fire, annual sampling could be scaled back to sampling every two years to continue tracking this long-term trend.

Background

A population of the yellow-bellied glider on the Bago Plateau, adjacent to the southwestern edge of Kosciuszko National Park, is listed as an Endangered Population (under the Biodiversity and Conservation Act 2016) and was predicted to decline based on an expected reduction in habitat quality due to timber harvesting. The population's susceptibility is further threatened by the risk of large stochastic events, such as severe wildfire, which impacted the Bago Plateau during the 'Black Summer' fires of 2019-2020.

Long-term monitoring forms the basis of this study commenced in 2013 (Bilney et al. 2022) and built on the baseline survey of 126 monitoring sites established in 1995 by Kavanagh and Stanton (1998). The aim of the monitoring program was to establish a trend for yellow-bellied glider occupancy and to assess whether changes in the trend were associated with disturbances from wildfire and timber harvesting across the Bago Plateau while also accounting for imperfect detection during surveys and changes to detectability associated with evolving sampling methods.

This report updates the previous analysis, reported in Bilney et al. (2022), which reported the trends up until 2020 and highlighted a 26 % decline in yellow-bellied glider occupancy in the first year after the black summer fire. This analysis includes a further three years of data to assess the extent of recovery after the fires.

Methods

Study area

The Bago Plateau is in the south-western slopes region of New South Wales' Snowy Mountains, immediately adjacent to Kosciuszko National Park. The study area encompasses Bago State Forest (31,681 ha), Maragle State Forest (12,332 ha) and a section of Kosciuszko National Park west of the Tumut River. Most of the Bago Plateau exceeds 1000 m in elevation (950-1350 m) and is bounded by the Tumut River valley to the east with its streams contributing to both the upper reaches of the Murray and Murrumbidgee River catchments.

Four forest type groups dominate across Bago and Maragle State Forests and these are aligned to Research Note 17 Forest Types (FT) (Anon. 1989) and include: "Montane Gums" (primarily FTs 138, 140) comprising Mountain Gum *Eucalyptus dalrympleana*, Snow Gum *E. pauciflora* and Manna Gum *E. viminalis*, but also including Mountain Swamp Gum *E. camphora*, and Black Sallee *E. stellulata*; "Wet Peppermint/Montane Gums" (FT 131) comprising Narrow-leaved Peppermint *E. robertsonii* mixed with montane gums; "Alpine Ash/ Montane Gums" (FT 148) comprising Alpine Ash *E. delegatensis* mixed with montane gums; and "Alpine Ash" (FT 147) being pure stands of *E. delegatensis* which is sought after as a commercial timber resource (Kavanagh and Stanton 1998; Kambouris et al. 2014). The drier forest types "Western Types" (FT 111,124) dominated by Red Stringybark *E. macrorhyncha*, Broad-leaved Peppermint *E. dives* and Brittle Gum *E. mannifera* occur at lower and drier elevations on the foothills surrounding the plateau.

Survey methods

A survey targeting arboreal marsupials and large forest owls was conducted on the Bago Plateau and surrounding foothills in 1995 where 126 survey sites were established and surveyed once (Kavanagh and Stanton 1998). All of the Kavanagh and Stanton (1998) sites were assigned to four forest type groupings (Alpine Ash, Alpine Ash/Montane Gums, Montane Gums, Wet Peppermint/Montane Gums) described above. A subset of 48 sites that represented 12 of each of these four forest types were resurveyed twice in 2010 (Kambouris et al. 2014). All 126 sites of Kavanagh and Stanton (1998) were adopted for a long-term landscape scale monitoring program for yellow-bellied glider on the Bago Plateau, which commenced in 2013 (Bilney et al. 2022). Sampling of the total population of sites was undertaken on a rotating panel basis such that the entire population of sites were sampled once (on a single visit) in a three-year period between 2013 and 2015. This sampling protocol was continued for the period 2016-2018. Sites were randomly selected across the four main forest type groups to ensure even representation and geographic distribution across the Bago Plateau within each annual subset of the 3-year survey cycle, with some clustering of sites to aid logistics in remote landscapes with limited track access. A subset (40) of the population of sites was sampled once in 2019 and a different subset (51) sampled in 2020/21 post-megafires. These sites were drawn from 124 of the 126 sites which had been burnt with varying severity to ensure recovery at sites affected by the 2019-20 megafires could be assessed. From 2022 onwards, the population of sites was modified such that sites that had never recorded yellow-bellied gliders were dropped from future monitoring. These were generally those sites characterised by greater coverage of western forest types that were less associated with yellow-bellied glider occupancy. These sites were substituted with 17 new sites that were located in forest types associated with a greater probability of occupancy by yellow-bellied gliders. In 2022 and 2023, 52 and 57 sites were sampled, respectively.

The survey procedure at each site between 1995 and 2020 used a combination of passive listening for 15 minutes, broadcasting owl calls (call playback interspersed with periods of listening for a response) for 15 minutes and 1 ha fixed radial spotlighting and listening for 15 minutes (45 minutes of total survey effort). The call playback was used to elicit responses from the yellow-bellied glider due to the species' propensity to respond vigorously (both vocally and in approach) to playback of calls of the owls (Kavanagh and Bamkin 1995). The call playback included five minutes of broadcasting calls of the Powerful Owl (*Ninox strenua*) and listening for five minutes, and then five minutes of broadcasting calls of the Masked Owl (*Tyto* *novaehollandiae*) and listening for five minutes. This technique was consistent with Kavanagh and Stanton (1998) and Kambouris et al. (2014), with the exception that sites were visited twice in the latter study and Sooty Owl (*Tyto tenebricosa*) calls were rarely broadcast post-1995. From 2022 onwards, passive acoustic recorders (Songmeter SM4; 2022 and Songmeter Mini; 2023) sampled for yellow-bellied glider vocalisations from sunset to sunrise over a period of at least 7 nights in autumn. Seven nights of recordings were scanned in AviaNZ using two recognisers developed by NSW DPI Forest Science for the call of yellow-bellied glider. The first recogniser was used to scan acoustic data from 2022 and was replaced by a new recogniser with higher recall to scan the 2023 data.

Weather details at the time of survey and site variables, including forest height, tree species composition, estimated time since disturbance to verify fire or timber harvesting history, and counts of the number of trees with large hollows (>10 cm openings), were recorded within the 1 ha plots centred on each site. As the exact survey location (and 1 ha survey plot) may have varied slightly over time and site attribute data not collected during every survey, changes in some site attributes were not assessed over time (e.g. number of hollow trees), including post-fire. For years without site attribute data, these variables were coded as missing data in the analysis. Surveys also avoided periods of heavy rainfall or if moderate winds were forecast.

Data analysis

Dynamic occupancy modelling (MacKenzie et al. 2003) was undertaken to estimate Yellow-bellied Glider occupancy on the Bago Plateau between 1995 and 2023. Detection data used in the modelling were derived from 126 sites sampled in 13 years of surveys between 1995 and 2023, with not all sites sampled in each year (Table 1). Repeat sampling in a given year was only undertaken for a subset of sites in 2010 (48) when each site was visited twice using the spotlight and call playback method and then for all sites sampled from 2021 onwards when passive acoustic recorders sampled for 7 nights. For all other years, a second visit was coded as missing data for modelling. As annual surveys are required to model dynamic parameters, colonisation and extinction, a three-year rotation (e.g. 2013, 2014 and 2015) was treated as a single survey period (e.g. 2013- 15) with data pooled across each rotation to allow for colonisation and extinction to be modelled.

A hierarchical approach was taken to modelling in order to reduce the total number of candidate models (Appendix 1). Detection probability was first modelled to account for imperfect detection associated with surveys while holding initial occupancy, colonisation and extinction constant. A range of weather covariates including temperature, rain (no evidence in last 24 h, evidence in last 24 h, raining during survey) and wind (calm, light, moderate, strong) recorded for the night of each survey were included as covariates for detection for all years except 2022 onwards as the sampling window for these years was not compatible with the weather metrics used for other years. Models that allowed detection probability to vary among survey periods and with survey method (spotlighting & call playback, passive acoustics with SM4 and recogniser version 1; passive acoustics with SM Mini and recogniser version 2) was included as well as a null model that assumed constant detection across all visits to a given site.

The top detection probability model was carried forward to model initial occupancy (i.e. occupancy in 1995). Initial occupancy was modelled while holding colonisation and extinction constant. Several site-based variables were included as covariates for occupancy including the number of observed hollow-bearing trees per 1 ha spotlight search area at each site (HT; Kambouris et al. 2014); elevation as metres above sea level based on Australian Height Datum (Ele); extent (%) of each forest type within a 450 m buffer of each site which represented an estimate of Yellow-bellied Glider home range size (e.g. Goldingay and Kavanagh 1993); and the extent (%) of harvesting of different age classes as at 1995 within this buffer. A null model that held initial occupancy constant across sites was also included in the set of candidate models. The influence (direction and magnitude) of a supported covariate was assessed by plotting occupancy estimates that were generated while holding all other supported covariates at the median value.

Colonisation and extinction parameters were then modelled separate using the top model for initial occupancy and holding the other dynamic parameter constant. Variables included as covariates for these parameters were the number of hollow-bearing trees per 1 ha site (HT), total rainfall for the 12 months preceding surveys (Tumbarumba Post Office), and the extent (%) of harvesting of different age classes within a 450 m buffer. The extent (%) and severity of wildfire, based on the Fire Extent and Severity Mapping (FESM – December 2020) was calculated from a 450 m buffer surrounding survey sites, with severity ratings pooled into three categories, being unburnt (category 0), low fire severity (categories 2 and 3) and high fire-severity (categories 4 and 5). These variables were considered as they either had the potential to or did vary among years and could influence the likelihood of an occupied site becoming unoccupied or an unoccupied site becoming occupied. For example, a reduction in the density of hollow-bearing trees could be associated with a lower likelihood of an unoccupied site becoming occupied. A null model where these parameters were held constant was also included.

The trend for yellow-bellied glider occupancy was estimated by first calculating initial occupancy in 1995 assuming median conditions for supported covariates among all sites. Occupancy estimates for subsequent years were derived based on relationships established between covariates and colonisation and extinction. For each subsequent year, the median value for supported covariates in that year was used to establish the proportion of occupied sites that became locally extinct and the proportion of unoccupied sites that were colonised. This provided a trend for the study area (i.e. across all sites) rather than for sites sampled in each year which may be biased if annual, but partial, sampling is not representative of the range of conditions in the study area. Prior to analysis, covariates were examined for collinearity with none considered highly correlated (r>0.7). Modelling and model selection were carried out in program R using the RPresence package (MacKenzie and Hines 2021).

Results

Environmental conditions during the study

Conditions across study sites were variable at the start of the study (Appendix 2). Sites were on average 1058 m ASL. A 450 m buffer around sites contained on average two hollow-bearing trees per ha. Approximately 15 %, 14 %, 25 %, 21 % and 3 % of this area was represented by Alpine Ash, Alpine Ash / Montaine Gum, Montane Gum, Wet Peppermint and Montane Gum and Western Type forest communities, respectively. The remaining extent of the buffer around sites was made up of a mix of different communities. On average, 28 % of the area around sites was unharvested at the start of the study. Recent (<5 years old), intermediate aged (5-15 years old), old (>15-30 years) and very old (>30 years old) harvesting occurred in 8 %, 20 %, 33 % and 10 %, respectively, of the buffer around sites. All sites were unburnt at the start of the study.

Some of these conditions changed over time as monitoring progressed (Appendix 3). For example, the extent of forest around each site that was affected by recent (<5 years old) fire increased from 0 % in 2009-10 to 96 % of the landscape in 2020 following the Black Summer fires. The extent of forest that was recently harvested initially reduced from 8 % to 1-2 % of the landscape, on average by 2018, before increasing to ~10 % of the landscape by 2023. Annual rainfall fluctuated among years, including low rainfall in the years immediately preceding the Black Summer fires and then highest rainfall post-fire (Fig. 1).

Naïve occupancy

Across all years, there were 440 yellow-bellied glider detections. Naïve occupancy, which does not account for imperfect detection or variation in habitat quality of sites sampled each year, fluctuated among years but was lowest in 2016-2018 and highest in 2023 (Fig. 1). The broad trend for naïve occupancy aligned closely with the trend for 12-monthly rainfall in the year preceding glider surveys (Fig. 1).

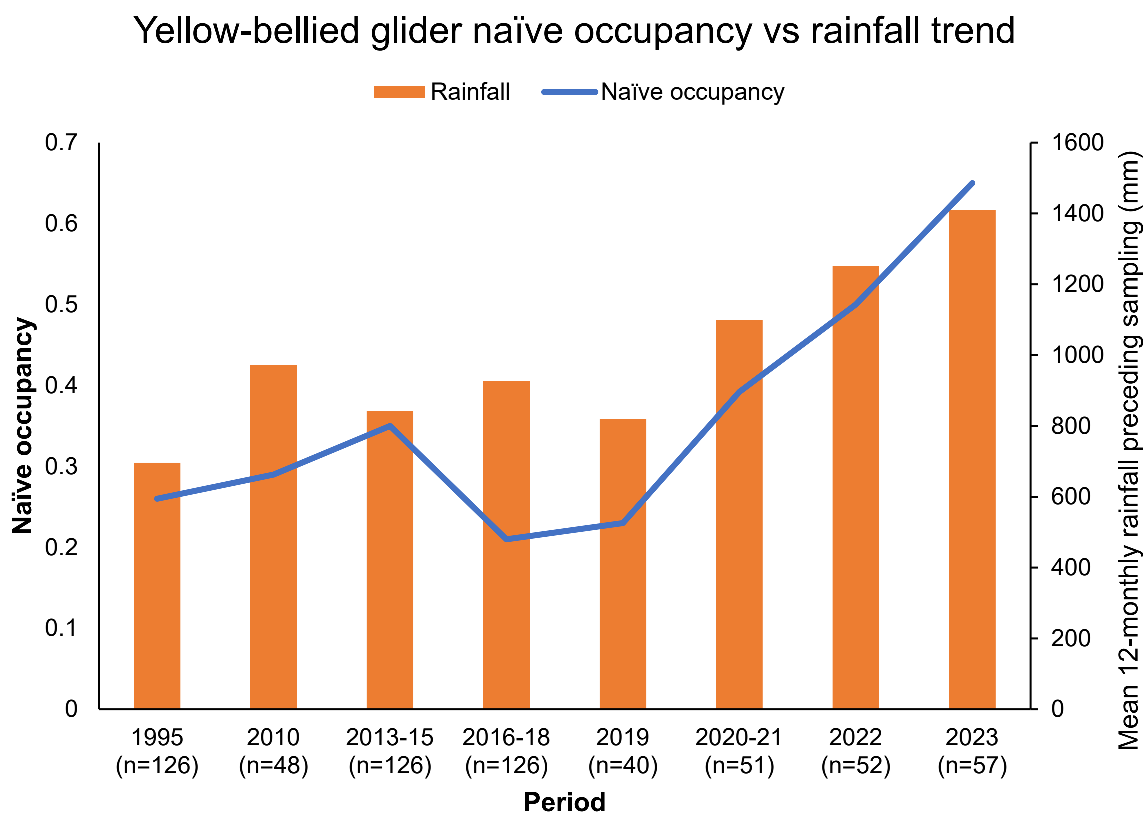

Fig. 1. Line graph illustrating variation in naïve occupancy among sampling periods.

Detection probability

One single covariate model for detection probability was supported (Table 1). This model allowed detection probability to vary with survey method. Additive models for detection probability did not converge, which is likely due to the small number of repeat visits in most years of monitoring). Detection probability per night was lowest in 2022 (0.41±0.04 per night) when Songmeter SM4 recorders were used to passively survey for yellow-bellied gliders along with an earlier version of call recogniser but highest in 2023 (0.61±0.03 per night) when Songmeter Mini recorders were used along with an updated call recogniser (Fig. 2). The updated call recogniser had higher recall and precision than the earlier version and this improved recall likely reflects the difference in detection probability between years when passive acoustics were used rather than the difference being associated with sensor type as detection probability for other vocalising mammals has been shown to be comparable using SM4 and SM Mini sensors (Law et al. 2024). Detection probability for yellow-bellied gliders using spotlighting & call play-back was intermediate (0.5±0.05 per visit). Given acoustic surveys sample over seven nights, cumulative detection probability is higher for this method (>0.98) compared to the spotlighting/call-playback method (0.75) used earlier in the monitoring program with up to two visits.

Table 1. Model summary for detection probability. Grey shading indicates models with support.

Model	DAIC	weight	npar	neg2ll
psi(.),gam(.),eps(.),p(Method)	0	0.9474	6	1414.02
psi(.),gam(.),eps(.),p(Period)	5.94	0.0486	11	1409.96
psi(.),gam(.),eps(.),p(.)	12.41	0.0019	4	1430.43
psi(.),gam(.),eps(.),p(Temp)	14.41	0.0007	5	1430.43
psi(.),gam(.),eps(.),p(Wind)	14.41	0.0007	5	1430.43
psi(.),gam(.),eps(.),p(Rain)	14.41	0.0007	5	1430.43

DAIC = delta AIC (difference in AIC score between the top model and other models) .

weight = model weight (explanatory power).

npar = number of parameters in the model.

neg2ll = negative 2 x log-likelihood.

psi = initial occupancy.

gam = colonisation

eps = extinction probability.

Method = survey method used (acoustic or spotlighting & call playback).

Period = Year or series of years (e.g., 1995, 2013-2015, etc.).

Temp = Temperature at time of survey.

Wind = Ordinal scale of wind speed.

Rain = Presence/Absence within 24 hrs of survey or during survey.

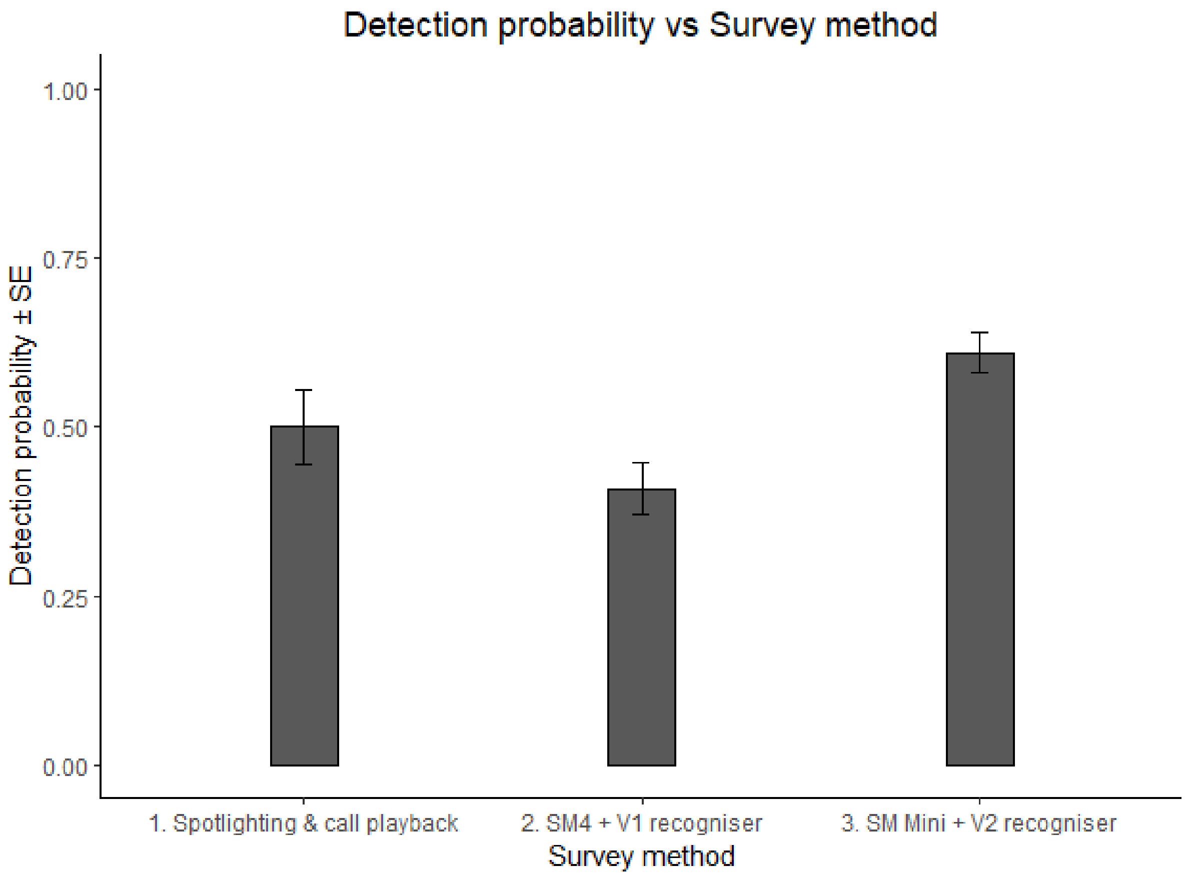

Fig. 2. Column graph illustrating variation in detection probability per night among survey methods.

Initial occupancy

A single 3-covariate model was supported for initial occupancy (Table 2). This model allowed occupancy to vary with the extent of montane gum, wet peppermint gum and western forest types. Occupancy was positively associated with the extent of montane gum and wet peppermint gum, but negatively associated with the extent of western forest types (Fig. 3). Under median conditions for these forest types at sites sampled in 1995, occupancy was 0.17 ± 0.10.

Table 2. Model summary for initial occupancy. Grey shading indicates models with support. See Appendix 2 for description of the covariates denoted in brackets after 'psi', 'gam' and 'eps'.

Model	DAIC	weight	npar	neg2ll
psi(Montane Gum+Wet_Peppermint_Gum+Western_Types),gam(.),eps(.),p(Method)	0	0.7	9	1370.93
psi(Montane Gum+Wet_Peppermint_Gum),gam(.),eps(.),p(Method)	3.88	0.101	8	1376.81
psi(Montane Gum+Wet_Peppermint_Gum+Ele),gam(.),eps(.),p(Method)	5.56	0.043	9	1376.49
psi(Montane Gum+Wet_Peppermint_Gum+HT),gam(.),eps(.),p(Method)	5.59	0.043	9	1376.53
psi(Montane Gum+Wet_Peppermint_Gum+Ash),gam(.),eps(.),p(Method)	5.8	0.038	9	1376.73
psi(Montane Gum+Wet_Peppermint_Gum+Ash_Gum),gam(.),eps(.),p(Method)	5.81	0.038	9	1376.74
psi(Montane Gum+Wet_Peppermint_Gum+harvesting),gam(.),eps(.),p(Method)	5.92	0.036	13	1368.86

DAIC = delta AIC (difference in AIC score between the top model and other models) .

weight = model weight (explanatory power).

npar = number of parameters in the model.

neg2ll = negative 2 x log-likelihood.

psi = initial occupancy.

gam = colonisation

eps = extinction probability.

Method = survey method used (acoustic or spotlighting & call playback)

See appendix 1 for description of model covariates that are denoted inside brackets.

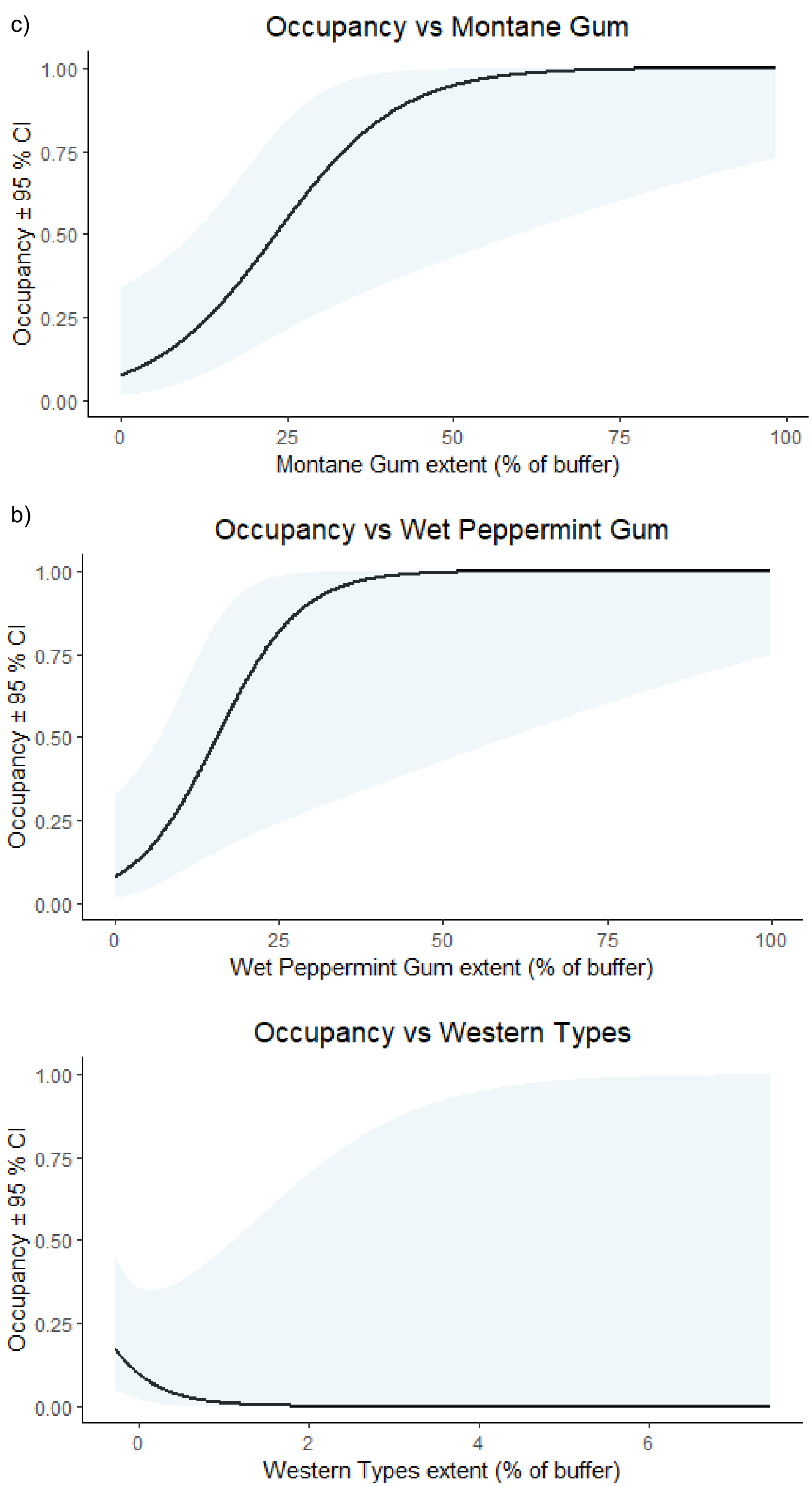

Fig. 3. response plots illustrating the association between initial occupancy and the extent of a) Montane Gum, b) Wet Peppermint Gum and c) Western Types.

Colonisation Probability

Two models were supported for colonisation probability (Table 3). The top model allowed colonisation probability to vary positively with the amount of rainfall in the 12 months preceding sampling (Fig. 4). The second supported model allowed colonisation to vary positively with the density of hollows at a site. It should be noted that this relationship is based on site dynamics between 1995 and 2020-21 as no hollow tree data were available for modelling 2022 and 2023. Under median conditions, colonisation probability was 0.16 ± 0.04.

Table 3. Model summary for colonisation probability. See Appendices 2 and 3 for description of the covariates denoted in brackets after 'psi', 'gam' and 'eps'.

Model	DAIC	weight	npar	neg2ll
psi(Forest type extent),gam(rain),eps(.),p(Method)	0	0.6150	10	1359.83
psi(Forest type extent),gam(HT),eps(.),p(Method)	1.17	0.3429	10	1361
psi(Forest type extent),gam(high fire severity extent),eps(.),p(Method)	6.83	0.0202	10	1366.66
psi(Forest type extent),gam(low fire severity extent),eps(.),p(Method)	8.82	0.0075	10	1368.65
psi(Forest type extent),gam(fire extent),eps(.),p(Method)	8.82	0.0075	10	1368.65
psi(Forest type extent),gam(.),eps(.),p(Method)	9.1	0.0065	9	1370.93
psi(Forest type extent),gam(harvesting),eps(.),p(Method)	14.48	0.0004	13	1368.31

DAIC = delta AIC (difference in AIC score between the top model and other models).

weight = model weight (explanatory power).

npar = number of parameters in the model.

neg2ll = negative 2 x log-likelihood.

psi = initial occupancy.

gam = colonisation.

eps = extinction probability.

Method = survey method used (acoustic or spotlighting & call playback).

See appendices 1 & 2 for description of model covariates that are denoted inside brackets.

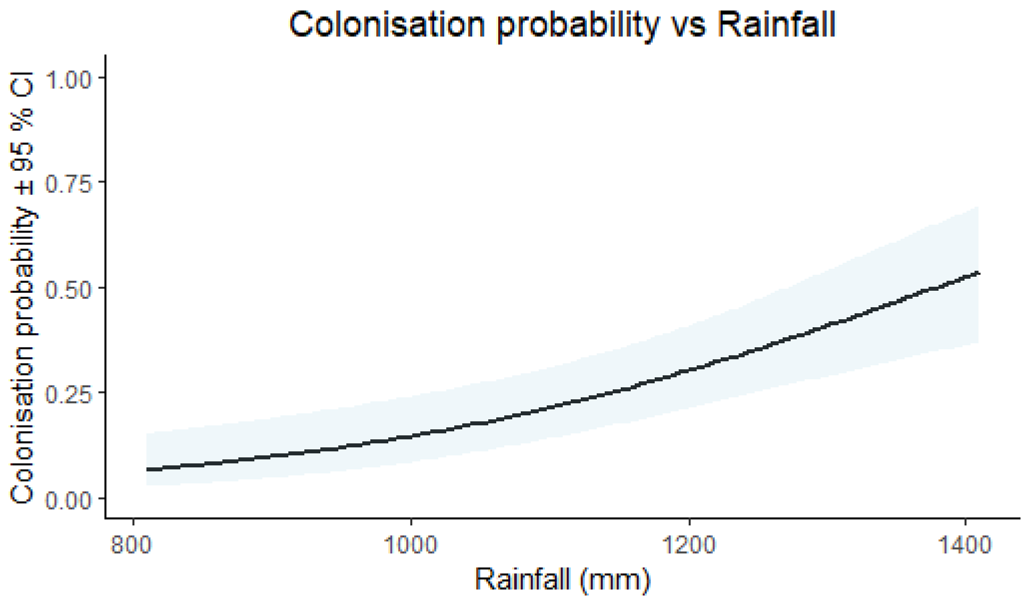

Fig. 4. Response plots illustrating the association between colonisation probability and the amount of rainfall in the 12 months preceding sampling.

Extinction Probability

A single model was supported for extinction probability (Table 4). This model allowed extinction probability to vary positively with the extent of high severity fire within 450 m of the site (Fig. 4). Under median conditions (no high severity fire), extinction probability was 0.13±0.04.

Table 4. Model summary for extinction probability. See Appendices 2 and 3 for description of the covariates denoted in brackets after 'psi', 'gam' and 'eps'.

Model	DAIC	weight	npar	neg2ll
psi(Forest type extent),gam(.),eps(high fire),p(Method)	0	0.852	10	1362.82
psi(Forest type extent),gam(.),eps(.),p(Method)	6.11	0.04	9	1370.93
psi(Forest type extent),gam(.),eps(rain),p(Method)	6.23	0.038	10	1369.05
psi(Forest type extent),gam(.),eps(low fire),p(Method)	6.68	0.03	10	1369.49
psi(Forest type extent),gam(.),eps(fire),p(Method)	6.7	0.03	10	1369.52
psi(Forest type extent),gam(.),eps(Period),p(Method)	8.85	0.01	15	1361.67
psi(Forest type extent),gam(.),eps(HT),p(Method)	29.97	0	10	1392.79
psi(Forest type extent),gam(.),eps(harvesting),p(Method)	54.13	0	13	1410.95

DAIC = delta AIC (difference in AIC score between the top model and other models).
weight = model weight (explanatory power).
npar = number of parameters in the model.
neg2ll = negative 2 x log-likelihood.
psi = initial occupancy.
gam = colonisation.
eps = extinction probability.
Method = survey method used (acoustic or spotlighting & call playback).
Period = Year or series of years (e.g., 1995, 2013-2015, etc.).
See appendices 1 & 2 for description of model covariates that are denoted inside brackets.

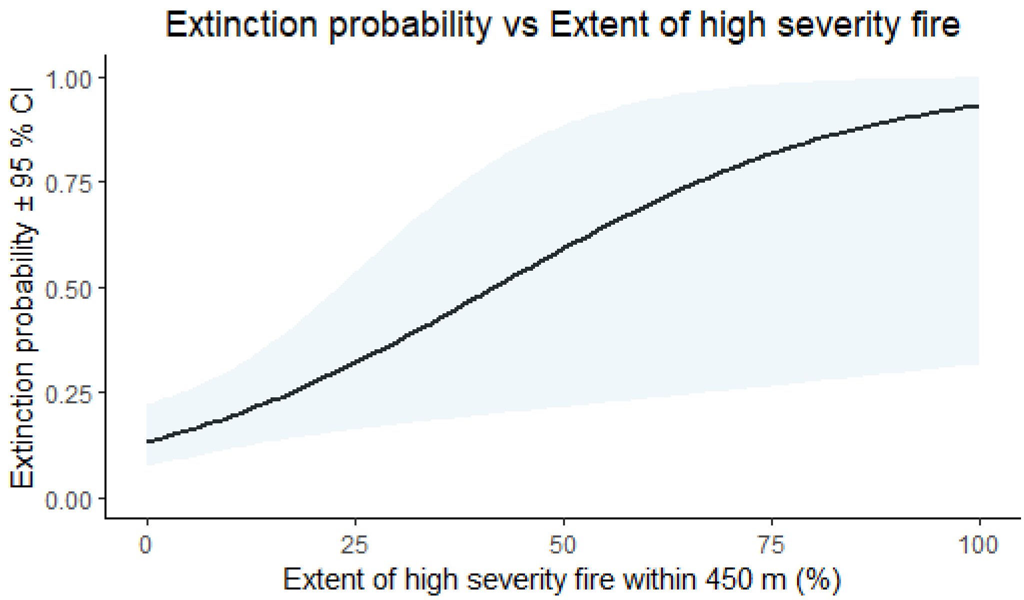

Fig. 5. Response plot showing the relationship between extinction probability and the extent of high severity fire.

Trend

Yellow-bellied glider occupancy on the Bago Plateau was lowest (~0.19) in 1995 and increased to 0.26 in 2010 which was maintained in 2013-15. Occupancy then increased to 0.3 in 2016-18 and remained stable in 2019, but decreased post- fire to 0.27 (10 % decline) in 2020/21. Occupancy increased by 81 % to 0.49 in 2022 and this continued in 2023 with a 43 % increase to 0.7 (Fig. 6).

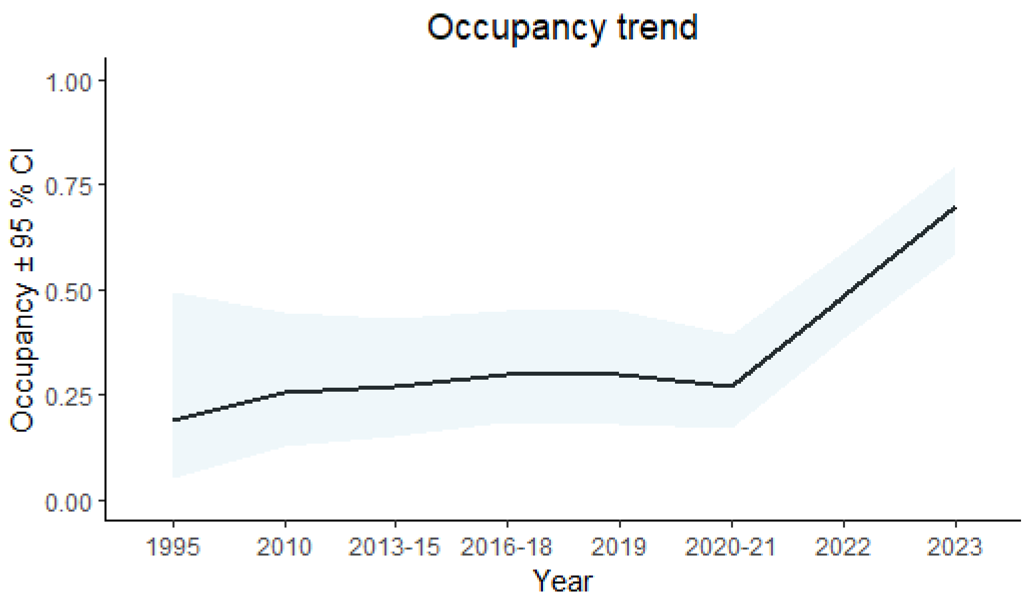

Fig. 6. Trend for yellow-bellied glider occupancy on the Bago Plateau between 1995 and 2023.

Discussion/interpretation of results

- Naïve occupancy fluctuated among years. This fluctuation likely reflects variable detection probability recorded among sampling periods that used different methods (spotlighting and acoustics with improvements made to call recognisers) and the range of habitat quality sampled in each period.

- Detection probability recorded pre-fire is based on a single year with repeat visits, albeit only two visits in that year (2010). Although detection probability can be estimated from two repeat visits to a site, estimates can be biased (MacKenzie et al. 2002). For example, with just two visits to a site, an occupied site will have a detection history of 01, 10 or 11 and this will provide a detection probability estimate that is >0.5, which may not accurately reflect detection probability using spotlighting & call playback. As such, caution should be used when interpreting modelled estimates of occupancy based on this part of the dataset.

- An assessment of detection probability for a subset of sites sampled with spotlighting & call playback and passive acoustics on the Bago Plateau revealed that single-visit detection probability was 0.50±0.05 for spotlighting & call playback and 0.41±0.04 and 0.61±0.03 per night for passive acoustics depending on the sensor type and recogniser version used. The different detection probability recorded in both years of passive acoustic monitoring likely reflects the change in recogniser version used to scan recordings, with version 2 used for 2023 data having significantly higher recall than version 1 (unpublished data – NSW DPI). Nevertheless, with two visits to a site with spotlighting & call playback, this equates to a detection probability of 0.75 whereas with 7 nights of passive acoustics, detection probability was ≥0.98 highlighting the value (i.e., greater detection probability and more precise estimates of detection probability than two nights of spotlighting & call playback) of passive acoustics as a monitoring method.

- Initial occupancy in 1995 closely reflected previous analyses (see Bilney et al. 2022) and highlighted the influence of forest type on probability of yellow-bellied glider occupancy. Yellow-bellied gliders were more likely to occupy sites with a greater extent of montane gum and wet peppermint/gum, but lower extent of western types. Given this variation, it is important to account for the habitat quality of each site sampled in a given year as the sites monitored in each period fluctuated during the monitoring program.

- Rainfall was a significant correlate with the dynamic parameter, colonisation. Rainfall in the 12-months preceding sampling was positively associated with the probability that unoccupied sites became occupied from one period to the next. In recent years following the 2019-2020 megafires, annual rainfall has been the highest recorded during the monitoring program and this level of rainfall was associated with a colonisation probability of 0.35-0.54.

- The extent of high severity fire was positively associated with extinction probability and the only decrease (10 %) in occupancy observed over the 25 years of monitoring occurred immediately post-fire, highlighting high severity fire as a threat to yellow-bellied gliders on the Bago Plateau.

- Yellow-bellied glider occupancy was generally low in the first 25 years of monitoring with small increases in occupancy observed between 1995 (0.19) and 2019 (0.3). The only decline in occupancy was observed immediately post-fire, with a 10 % reduction in 2020/21 from 0.3 to 0.27. A similar decline (13 %) was found for the species in the study area post-fire using glider sap tree surveys (Goldingay et al. 2024). The decline reported in our study is lower than previously reported because the relationships for colonisation and extinction in the current analysis differed to that previously found. The current estimated occupancy pre-fire (0.30) is lower than the previous estimate (0.34), albeit within the bounds of error. For example, colonisation probability was previously associated with hollow tree abundance and not the amount of rainfall in the 12-months preceding sampling. The strong influence of rainfall on colonisation may not have been apparent in the previous analysis given rainfall was generally low for the duration of monitoring at that point in time and without the extremely high levels seen in recent years. Furthermore, hollow tree abundance data was not available for modelling post 2020/21. Post-2020/21, occupancy increased sharply to 0.49 before continuing to rise to 0.7 in 2023. Although the increase in occupancy post-2020/21 also corresponded to a change in the sampling methods, the influence of variable sampling methods on detection probability was accounted for when estimating the occupancy trend. Modelling suggests the fire impacts had already occurred by 2020/2021, and the following year was dominated by colonisation rather than extinction, particularly as there were more sites at which colonisation could occur compared to the number of sites at which extinction processes applied.

- Harvesting was not associated with the trend in occupancy for yellow-bellied glider, whereas the extent of high severity fire was associated with higher probability of extinction and this was related to a 10 % decrease in occupancy in 2020/2021. However, the influence of high severity fire did not persist, as recovery followed with years of above average rainfall which was associated with higher probability of colonisation.

- There are limitations associated with this preliminary analysis and some of these have been described earlier (e.g., few years with repeat visits pre-fire). In addition to these, some covariates were not available for post-fire sampling. For example, counts of hollow trees are likely to have been affected by fires and this variable was previously found to be associated with colonisation probability (Bilney et al. 2022). With the transition to passive acoustics, weather variable data for survey nights is not easily applied to historic and more recent data given historical data were collected for a short period of the night when surveys were undertaken whereas passive acoustics samples for the entire night. Nevertheless, the inclusion of a model that accounts for yearly variation in detection probability accounts for the influence of variable weather and other factors (e.g., variable abundance of yellow-bellied gliders among years) on detection probability and in this study the model was not supported, indicating a stronger influence of survey method.

- Given the shift to passive acoustics from 2022 onwards and the complexity associated with modelling historical data with few repeat visits, we suggest that future analyses should seek to generate a new trend from 2024 onwards while still reporting the current trend for context. This will simplify interpretation of the trend given the methodological changes and changes to the population of sites being sampled in recent years and any future change to the population of sites (e.g., to incorporate CIFOA trend monitoring sites).

- The frequency of sampling should be re-assessed following analysis of the trend with data for 2024. Given the current high level of occupancy and that there has been no decline in yellow-bellied glider occupancy other than immediately post-fire, annual sampling could be scaled back to sampling every two years to continue tracking this long-term trend. More frequent sampling could be re-introduced following major disturbance, particularly high severity fire, to assess impacts and track recovery.

Conclusion

Trend analysis for yellow-bellied glider identified low occupancy across the Bago forests pre-fire, a small impact of the Black Summer fires in 2020, but then rapid recovery to high occupancy levels with high rainfall post-fire. Passive acoustic sampling has resulted in greater levels of detection and a more precise estimate of occupancy than spotlighting used in the early part of the study.

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Appendix 1. Candidate models for initial occupancy for sets of one, two and three covariates. The addition of a fourth covariate did not improve on the top three covariate model.

Model	DAIC	weight	npar	neg2ll
psi(Montane_gum),gam(.),eps(.),p(Method)	0	0.9917	7	1392.66
psi(Harvesting),gam(.),eps(.),p(Method)	10.38	0.0055	10	1397.05
psi(Western_Types),gam(.),eps(.),p(Method)	13.08	0.0014	7	1405.75
psi(Ash),gam(.),eps(.),p(Method)	14.1	0.0009	7	1406.76
psi(Ele),gam(.),eps(.),p(Method)	16.39	0.0003	7	1409.06
psi(Wet_Peppermint_Gum),gam(.),eps(.),p(Method)	18.85	0.0001	7	1411.52
psi(HT),gam(.),eps(.),p(Method)	19.22	0.0001	7	1411.88
psi(.),gam(.),eps(.),p(Acoustics)	19.36	0.0001	6	1414.02
psi(Ash_Gum),gam(.),eps(.),p(Method)	20.74	0	7	1413.4

Model	DAIC	weight	npar	neg2ll
psi(Montane Gum+Wet_Peppermint_Gum),gam(.),eps(.),p(Method)	0	0.9924	8	1376.81
psi(Montane Gum+Western_Types),gam(.),eps(.),p(Method)	11.65	0.0029	8	1388.47
psi(Montane Gum+Ash),gam(.),eps(.),p(Method)	13.29	0.0013	8	1390.11
psi(Montane Gum+HT),gam(.),eps(.),p(Method)	13.65	0.0011	8	1390.46
psi(Montane_gum),gam(.),eps(.),p(Method)	13.85	0.001	7	1392.66
psi(Montane Gum+Ele),gam(.),eps(.),p(Method)	15.3	0.0005	8	1392.12
psi(Montane Gum+harvesting),gam(.),eps(.),p(Method)	15.3	0.0005	12	1384.12
psi(Montane Gum+Ash_Gum),gam(.),eps(.),p(Method)	15.85	0.0004	8	1392.66

Model	DAIC	weight	npar	neg2ll
psi(Montane Gum+Wet_Peppermint_Gum+Western_Types),gam(.),eps(.),p(Method)	0	0.718	9	1370.93
psi(Montane Gum+Wet_Peppermint_Gum),gam(.),eps(.),p(Method)	3.88	0.103	8	1376.81
psi(Montane Gum+Wet_Peppermint_Gum+Ele),gam(.),eps(.),p(Method)	5.56	0.044	9	1376.49
psi(Montane Gum+Wet_Peppermint_Gum+HT),gam(.),eps(.),p(Method)	5.59	0.044	9	1376.53
psi(Montane Gum+Wet_Peppermint_Gum+Ash),gam(.),eps(.),p(Method)	5.8	0.04	9	1376.73
psi(Montane Gum+Wet_Peppermint_Gum+Ash_Gum),gam(.),eps(.),p(Method)	5.81	0.039	9	1376.74
psi(Montane Gum+Wet_Peppermint_Gum+harvesting),gam(.),eps(.),p(Method)	8.2	0.012	13	1371.13

Appendix 2. Environmental and disturbance site covariates used to model initial occupancy (1995).

Variable	Description	Units	Min	Max	Mean
HT_95	Density of hollow-bearing trees	No. of trees per ha	0	8	2
unharvested_per	Extent of unharvested forest	Proportion within 450 m	0.00	0.98	0.28
rec_har_per	Extent of recent (<5 years) harvesting	Proportion within 450 m	0.00	0.80	0.08
int_har_per	Extent of intermediate (5-15 years) harvesting	Proportion within 450 m	0.00	1.00	0.20
old_har_per	Extent of old (>15-30 years) harvesting	Proportion within 450 m	0.00	1.00	0.33
vold_har_per	Extent of old (>30 years) harvesting	Proportion within 450 m	0.00	1.00	0.10
Rain95	Annual rainfall in the 12-months preceding surveys	mm	696	696	696
Ash	Extent of Alpine Ash	Proportion within 450 m	0.00	1.00	0.15
Ash_Gum	Extent of Alpine Ash and Montane Gum	Proportion within 450 m	0.00	0.92	0.14
Montane_gum	Extent of Montane Gum	Proportion within 450 m	0.00	0.99	0.25
Wet_Peppermint_Gum	Extent of Wet Peppermint and Montane Gum	Proportion within 450 m	0.00	1.00	0.21
Western_Types	Extent of drier western types	Proportion within 450 m	0.00	0.87	0.03
Ele_standard	Elevation	m ASL	400	1330	1058

Appendix 3. Environmental and disturbance site covariates used to model colonisation and extinction (1995-2023).

Year	Metric	HT	rec_har_per	int_har_per	old_har_per	vold_har_per	Rain	unburnt	low_fire	high_fire
2010	Min	0	0.00	0.00	0.00	0.00	972	1.00	0.00	0.00
2010	Max	6	0.75	0.84	0.96	0.83	972	1.00	0.00	0.00
2010	Mean	2	0.02	0.06	0.08	0.09	972	1.00	0.00	0.00
2013-2015	Min	0	0.00	0.00	0.00	0.00	842	1.00	0.00	0.00
2013-2015	Max	18	0.41	0.75	0.87	1.00	842	1.00	0.00	0.00
2013-2015	Mean	3	0.01	0.10	0.15	0.25	842	1.00	0.00	0.00
2016-2018	Min	0	0.00	0.00	0.00	0.00	927	1.00	0.00	0.00
2016-2018	Max	16	0.74	0.75	0.87	1.00	927	1.00	0.00	0.00
2016-2018	Mean	1	0.02	0.08	0.16	0.25	927	1.00	0.00	0.00
2019	Min	0	0.00	0.00	0.00	0.00	820	1.00	0.00	0.00
2019	Max	13	0.91	1.00	0.84	0.69	820	1.00	0.00	0.00
2019	Mean	3	0.07	0.33	0.19	0.11	820	1.00	0.00	0.00
2020-2021	Min	0	0.00	0.00	0.00	0.00	1019	0.00	0.01	0.00
2020-2021	Max	8	0.53	0.57	1.10	1.00	1019	0.59	1.00	0.99
2020-2021	Mean	1	0.05	0.06	0.24	0.34	1019	0.04	0.62	0.34
2022	Min		0.00	0.00	0.00	0.00	1251	1.00	0.00	0.00
2022	Max		0.91	0.56	0.83	1.00	1251	1.00	0.00	0.00
2022	Mean		0.10	0.04	0.21	0.41	1251	1.00	0.00	0.00
2023	Min		0.00	0.00	0.00	0.00	1410	1.00	0.00	0.00
2023	Max		0.91	0.74	0.78	1.00	1410	1.00	0.00	0.00
2023	Mean		0.09	0.05	0.20	0.41	1410	1.00	0.00	0.00