

# Yellow-bellied Glider PMP data from Bago-Maragle Forests (1995-2019)

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## **Introduction**

Surveys for the Yellow-bellied Glider in the Bago-Maragle forests have been undertaken over a 25 year period for research and more recently as part of monitoring specified in a population management plan.

DPI Forest Science was requested to investigate whether data from surveys could be used in an occupancy modelling framework to account for imperfect detection associated with surveys and to assess potential drivers of trends in occupancy. Furthermore, recommendations to improve sampling effort for future surveys were requested.

## **Methods**

A dynamic occupancy modelling framework was used to estimate Yellow-bellied Glider occupancy in Bago-Maragle forests between 1995 and 2019 based on call playback and spotlighting surveys (Kavanagh and Stanton 1988; Kambouris et al. 2014). Detection data used in modelling were obtained across 126 sites that were sampled in nine years between 1995 and 2019, though not all sites were sampled in each year (Table 1). This period also included a hiatus in harvesting between 2009 and 2013. Repeat sampling in a given year was only undertaken in 2010, with each site visited twice. For all other years, a second visit was coded as missing data for modelling. A rotating panel design was implemented from 2013 onwards such that 3 panels of ~40 sites were sampled each year, with no subset of sites sampled in all three rotations (i.e., annual sites). As annual sites 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.

**Table 1. Summary of sampling effort and season of surveys among years.**

Year of survey	Number of sites	Number of site repeats	Season of survey	Naïve Occupancy	Mean % of non-preferred forest types sampled within 450 m buffer	Mean % of recent (<5 years) harvesting within 450 m buffer
1995	126	1	Autumn	0.26	30.9	3.0
2010	48	2	Autumn	0.29	41.9	1.9
2013	42	1	Spring	0.36	30.2	0.5
2014	44	1	Spring	0.43	28.2	0.3
2015	38	1	Spring	0.26	34.6	0.1
2016	43	1	Spring	0.21	31.4	0.6
2017	41	1	Spring	0.24	30.0	0.6
2018	37	1	Spring	0.22	34.6	1.1
2019	40	1	Spring	0.23	31.7	8.5

A hierarchical approach was taken to modelling in order to reduce the total number of candidate models. We first modelled detection probability to account for imperfect detection associated with surveys and held initial occupancy, colonisation and extinction constant. A range of weather covariates (temperature, rain and wind) for the night of each survey were included as covariates for detection as well as a null model that assumed constant detection across all visits to a given site. Supported models were model-averaged to provide estimates of detection, but only the top 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 – the number of hollow trees per 1-ha site (Kambouris et al. 2014; HT), elevation as metres above sea level (Ele), forest type category as assigned by Kavanagh and Stanton (1998), forest type extent within a 450 m buffer (average home range of YBG in the study area) of each site and the extent of harvesting of different age classes - <5 years, 5-15 years, 16-30 years and >30 years within this buffer (Table 2). 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 (proportion of unoccupied sites where the species was detected in the following season) and extinction (proportion of occupied sites where the species was not detected in the following season) parameters were then modelled using the top model for initial occupancy. Variables included as covariates for these parameters were number of hollow trees per 1-ha site (HT), annual rainfall for the calendar year preceding surveys (ann\_rain; obtained from Batlow weather station: 72082; Fig. 1.) and the extent of harvesting of different age classes within a 450 m buffer - <5 years, 5-15 years, 16-30 years and >30 years. The latter was used to account for the hiatus in harvesting between 2009 and 2013. A null model where these parameters were held constant was also included.

Prior to analysis, covariates were examined for collinearity. None of the covariates considered were highly correlated ( $r > 0.7$ ).

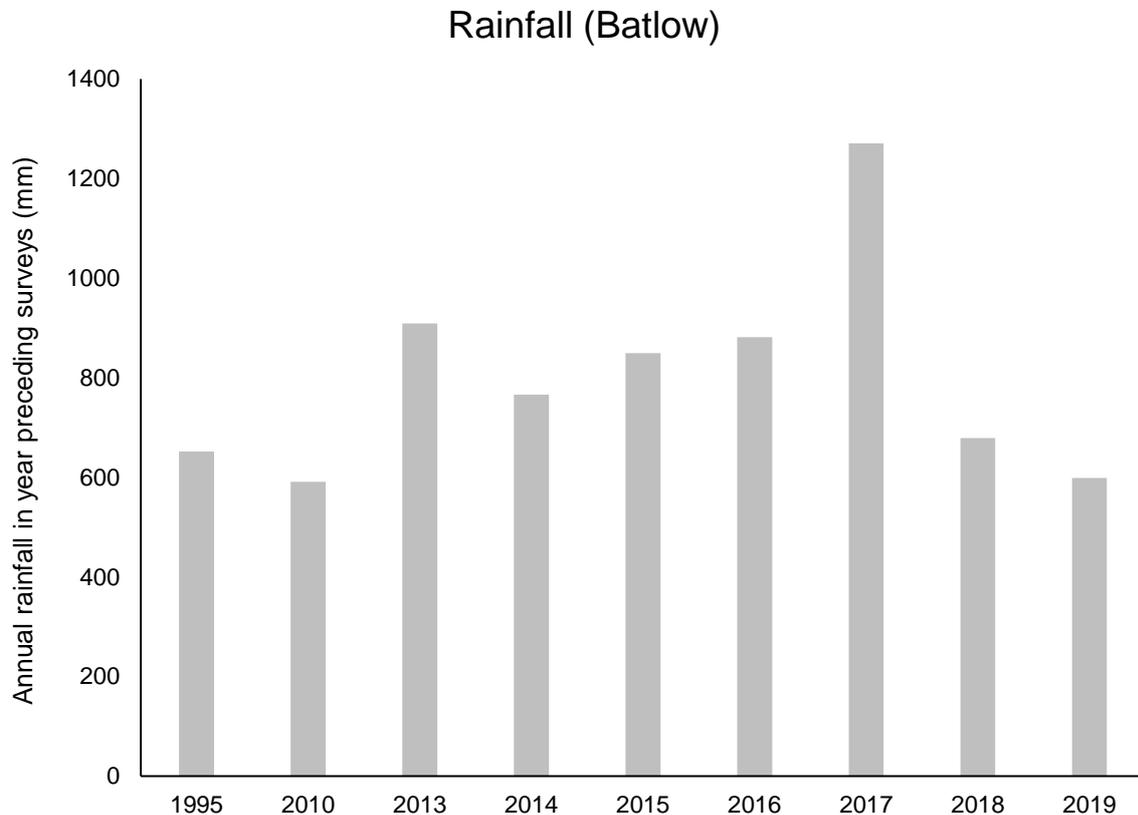
**Table 2. Summary statistics (minimum, maximum and mean) for covariates used in modelling of occupancy.**

Covariate	Hollow trees per 1 ha plot	Elevation (m ASL)	Extent (%) of recent (<5 yrs) harvesting	Extent (%) of intermediate (5-15 yrs) harvesting	Extent (%) of old (16-30 yrs) harvesting	Extent (%) of very old (>30 yrs) harvesting	Annual rainfall preceding surveys	Extent (%) of Alpine Ash	Extent (%) of Alpine Gum	Extent (%) of Gum/Peppermint	Extent (%) of Low Peppermint	Extent (%) of Western Types
Min	0	400	0	0	0	0	591.2	0	0	0	0	0
Max	18	1330	94.77484	100.0544	100.0644	100.0644	1270.8	100	98.59502	99.99988	82.17262	39.35582
Mean	2.312088	1058.095	2.05566	4.053009	6.466546	7.208681	799.8333	28.69578	24.78183	20.89299	2.198507	0.859261

## Results

### *Annual rainfall*

Annual rainfall in the calendar year preceding surveys was variable but generally below the long-term average for Batlow (939 mm) in all years of monitoring with the exception of 2017 (Fig. 1).



**Fig. 1. Column graph illustrating annual calendar year rainfall in the year preceding surveys.**

### *Naïve occupancy*

Naïve occupancy, which does not account for imperfect detection associated with surveys, ranged between 0.21 and 0.43 and fluctuated between years. The year (2014) with greatest naïve occupancy coincided with the presence of an observer that contributed 37.5 % of survey effort but targeted a more remote region which had more preferred forest types and recorded 57 % of YBG records (25/44) in that year. This highlights the potential for observer bias which is not accounted for in naïve occupancy. Furthermore, surveys in 2014 included sites with on average the lowest extent of non-preferred forest types sampled among all years (Table 1).

### *Detection*

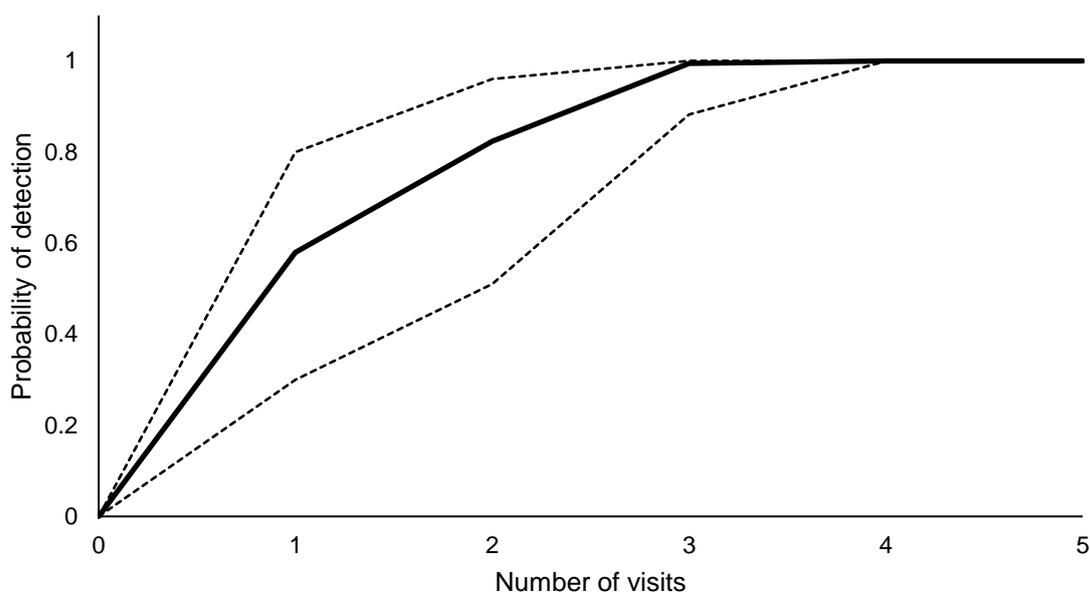
Three of the four candidate models for detection (~0.58 per night) were supported, though the beta coefficients for each covariate were small (<0.001) indicating only minor positive

influences on detection (Table 3). Furthermore, surveys were not carried out on very wet or windy nights. Only the top model using rain as a covariate for detection was used in the next step to model occupancy.

**Table 3. List of models for Yellow-bellied Glider detection probability. Supported models are shaded in grey.**

Model	dAIC	wgt	Npar	neg2ll
psi(.),gam(.),eps(.),p(Rain)	0	0.315	5	558.18
psi(.),gam(.),eps(.),p(Wind)	0.015	0.313	5	558.19
psi(.),gam(.),eps(.),p(Temp)	0.033	0.31	5	558.21
psi(.),gam(.),eps(.),p(.)	3.236	0.062	4	563.41

Accounting for confidence intervals, at least 3 repeat visits are required to be >90 % confident of detecting a species at an occupied site (Fig. 2).



**Fig. 2. Plot illustrating the probability of detecting a Yellow-bellied Glider at an occupied site with increasing number of visits.**

Although not specifically examined as part of the occupancy modelling framework, a crude comparison was made to identify whether initial detections favoured a particular aspect of the sampling protocol (i.e., passive listening, active call playback and spotlighting). Across survey periods,  $37 \pm 12$  % of detections were made during active listening, whereas  $53 \pm 16$  % of detections were made during call playback. Only  $9 \pm 8$  % of detections were made with spotlighting.

### *Initial (1995) occupancy*

In all, six candidate models assessed initial occupancy for Yellow-bellied Gliders (Table 4). A single model was supported. This model allowed initial occupancy to vary with the extent of forest types within a 450 m buffer of each site.

**Table 4. List of models for Yellow-bellied Glider initial occupancy. Supported models are shaded in grey.**

Model	dAIC	wgt	npar	neg2ll
psi(Forest type extent),gam(.),eps(.),p(Rain)	0	1	10	506.52
psi(Ele),gam(.),eps(.),p(Rain)	22.03	0	6	536.54
psi(Harvesting),gam(.),eps(.),p(Rain)	32.99	0	9	541.51
psi(HT),gam(.),eps(.),p(Rain)	34.3	0	6	548.82
psi(.),gam(.),eps(.),p(Rain)	41.66	0	5	558.18
psi(Rain),gam(.),eps(.),p(Rain)	43.66	0	6	558.18

Forest type extent = % of Alpine Ash + % Alpine Gum + % Gum/Peppermint + % Low Peppermint + % Western Types.

Ele = Elevation (m ASL).

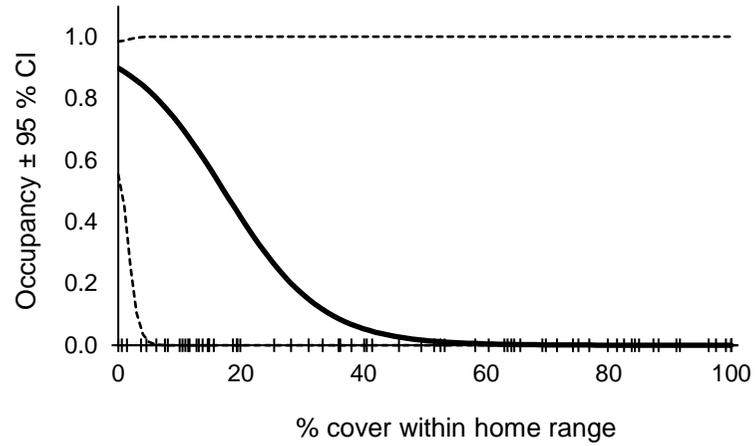
Harvesting = % recent (<5 yrs) harvesting + % intermediate (5-15 yrs) harvesting + % old (>16-30 yrs) harvesting + % very old (>30 yrs) harvesting.

HT = Hollow tree abundance per 1 ha plot.

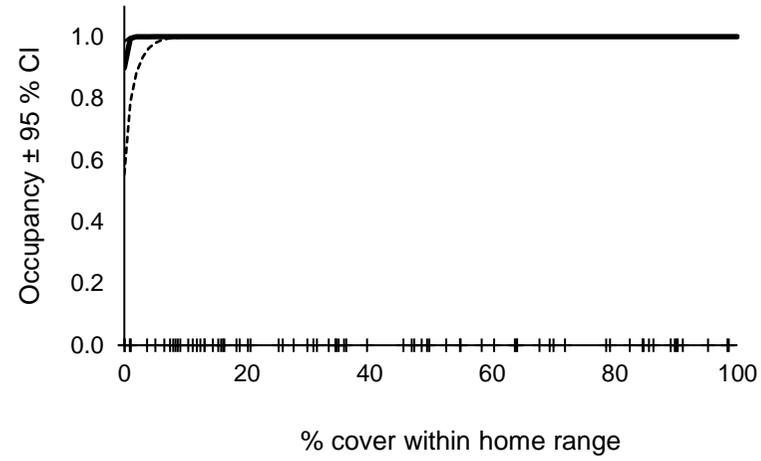
Rain = Annual rainfall preceding survey.

Initial occupancy was influenced by the extent of five forest types within a 450 m buffer off each site (Fig. 3). However, the influence of each forest type was variable, with occupancy most strongly influenced (-ve) by the extent of Alpine Ash, Low Peppermint and western types. For other forest types, initial occupancy was high irrespective of their extent within the 450 m buffer.

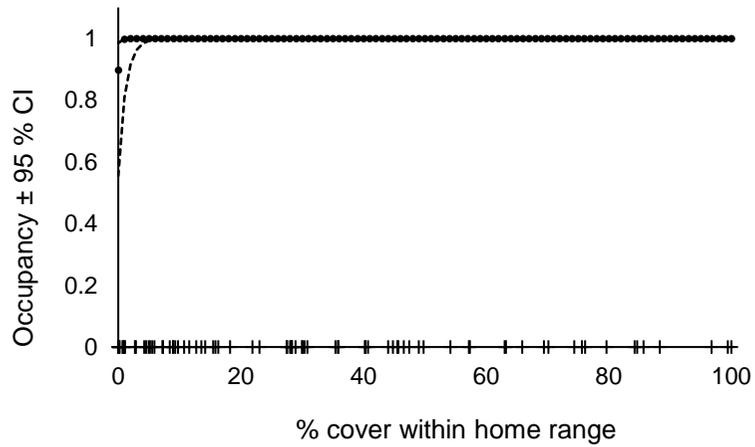
Occupancy vs Alpine Ash



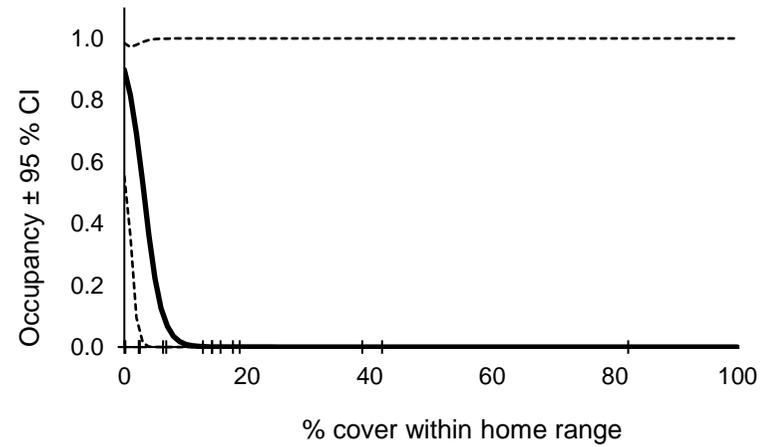
Occupancy vs Alpine Gum



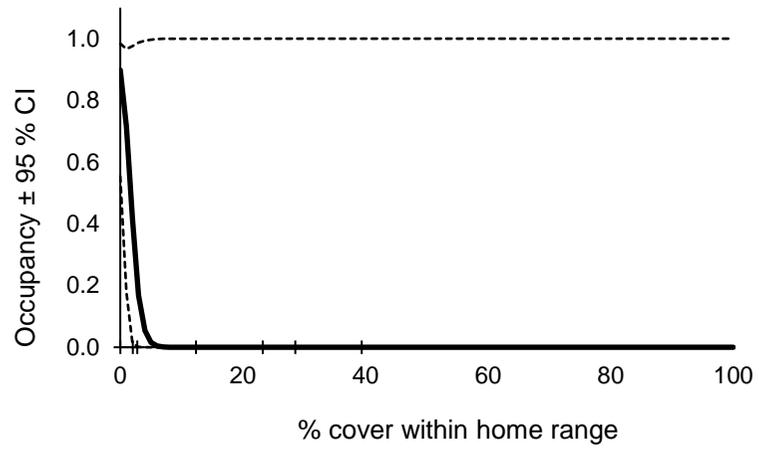
Occupancy vs Gum/Peppermint



Occupancy vs Low Peppermint



### Occupancy vs Western Types



**Fig. 3.** The relationship between initial occupancy and the extent of five forest types within a 450 m buffer of each site.

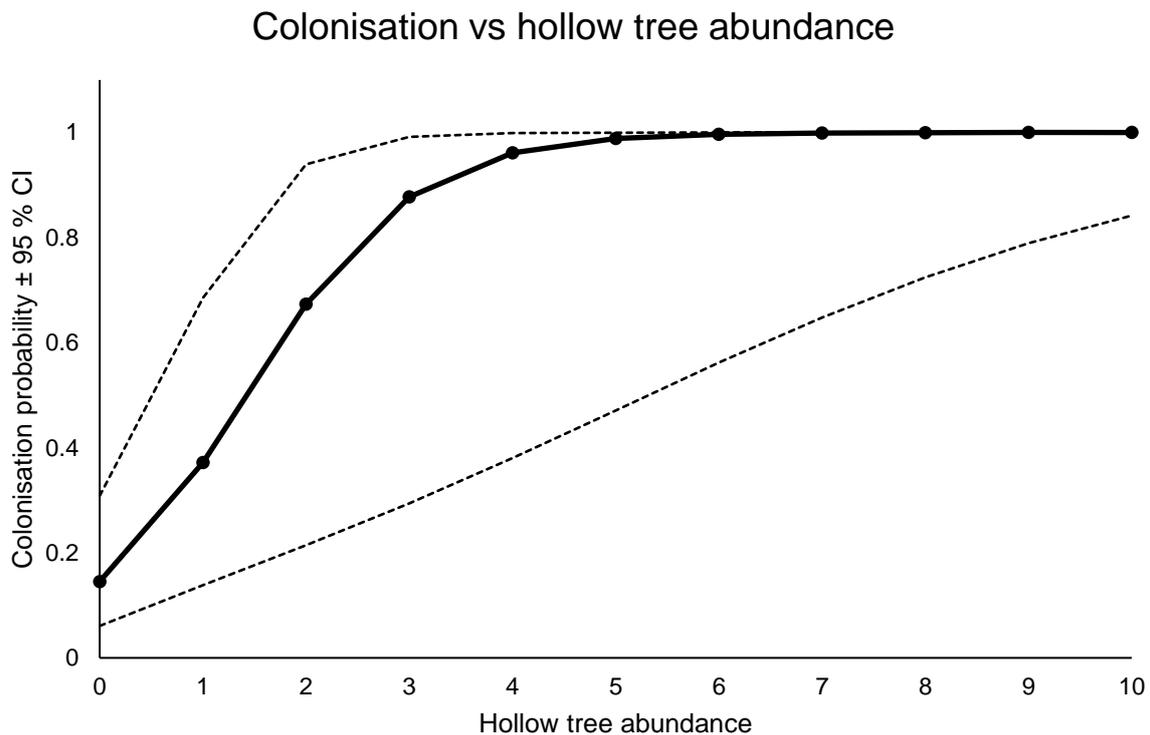
### Colonisation and Extinction

In all, four candidate models assessed colonisation probability for Yellow-bellied Gliders (Table 5). A single model was supported. This model allowed colonisation probability to vary with the number of hollow trees at each site. Colonisation probability increased from ~0.15 at sites without hollow trees to >0.9 at sites with  $\geq 4$  hollow trees (Fig. 4).

**Table 5. List of models for Yellow-bellied Glider colonisation probability. Supported models are shaded in grey.**

Model	dAIC	weight	npar	neg2ll
psi(Forest type extent),gam(HT),eps(.),p(Rain)	0	0.788	11	500.74
psi(Forest type extent),gam(.),eps(.),p(Rain)	3.78	0.119	10	506.52
psi(Forest type extent),gam(Rain),eps(.),p(Rain)	5.35	0.054	11	506.08
psi(Forest type extent),gam(Harvesting),eps(.),p(Rain)	6.03	0.039	14	500.77

Forest type extent = % of Alpine Ash + % Alpine Gum + % Gum/Peppermint + % Low Peppermint + % Western Types.  
 Harvesting = % recent (<5 yrs) harvesting + % intermediate (5-15 yrs) harvesting + % old (>16-30 yrs) harvesting + % very old (>30 yrs) harvesting.  
 HT = Hollow tree abundance per 1 ha plot.  
 Rain = Annual rainfall preceding survey.



**Fig. 4. Relationship between colonisation probability and the abundance of hollow trees.**

In all, four candidate models assessed extinction probability for Yellow-bellied Gliders (Table 6). Two models were supported, including the null model. The top model allowed extinction probability (~0.19) to vary with the number of hollow trees at each site, though the beta

coefficient was weakly negative ( $<0.001$ ) indicating only a very minor influence on extinction probability. For example, extinction probability varied between 0.194 at sites without hollow trees and 0.195 at sites with 10 hollow trees. Extent of harvesting was not supported as influencing extinction.

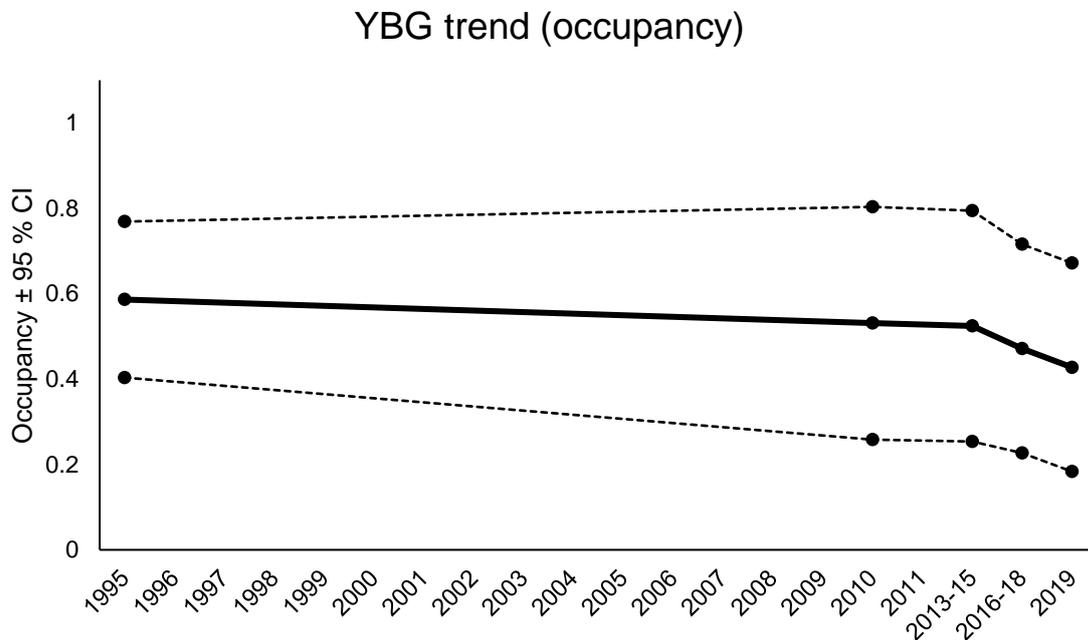
**Table 6. List of models for Yellow-bellied Glider extinction probability. Supported models are shaded in grey.**

Model	dAIC	weight	npar	neg2ll
psi(Forest type extent),gam(HT),eps(HT),p(Rain)	0	0.482	12	498.05
psi(Forest type extent),gam(HT),eps(.),p(Rain)	0.69	0.342	11	500.74
psi(Forest type extent),gam(HT),eps(Rain),p(Rain)	2.32	0.151	12	500.37
psi(Forest type extent),gam(HT),eps(Harvesting),p(Rain)	5.91	0.025	15	497.95

Forest type extent = % of Alpine Ash + % Alpine Gum + % Gum/Peppermint + % Low Peppermint + % Western Types.  
 Harvesting = % recent (<5 yrs) harvesting + % intermediate (5-15 yrs) harvesting + % old (>16-30 yrs) harvesting + % very old (>30 yrs) harvesting.  
 HT = Hollow tree abundance per 1 ha plot.  
 Rain = Annual rainfall preceding survey.

*Trend*

Yellow-bellied Glider occupancy decreased by 16 % between 1995 and 2019, though confidence intervals were relatively wide (Fig. 5).



**Fig. 5. The trend for Yellow-bellied Glider occupancy between 1995 and 2019. Dashed lines are 95 % confidence intervals.**

## **Limitations**

Although estimates of detection and occupancy were able to be made for the Yellow-bellied Glider dataset, there are a number of limitations that should be acknowledged. The maximum number of repeat visits to sites was two, with this restricted to a single year (2010). This means accounting for the effects of imperfect detection will be relatively imprecise for this data set. Nevertheless, our estimate of 0.58 per visit is similar to estimates at Eden using a greater number of repeat visits (0.51 per visit – Wintle et al. 2005). The imprecision associated with few repeat visits in part may be a reason why the effects of covariates on detection were very minor. Covariate data were unavailable for a small subset of sites and this may also have contributed to the very minor effects of covariates. Furthermore, restrictions imposed on when surveys were undertaken (i.e., trying to avoid periods of heavy rainfall, windy nights, etc.) meant that there was little variation in these covariates among surveys. It is worth noting that Wintle et al. (2005) found that rain had a negative effect on detection at Eden. Estimates of occupancy with just two repeat visits can be biased (overestimate moderate occupancy levels, ~0.5; underestimate high occupancy levels, ~0.9), particularly when detection probability is <0.5, though the degree of bias increases as detection probability decreases (MacKenzie et al. 2002).

Another limitation of the analysis is that seasonal variability in detection was unable to be reliably modelled given there were no repeat visits to a site for spring surveys (2013-2019), whereas repeat visits were only available for one of the autumn surveys (i.e., 2010). It is unclear whether detection varies seasonally. Preliminary assessments suggested a potential for differences in detection between spring and autumn, but this could not be verified given the lack of repeat visits in spring. Elsewhere, calling varies due to food type (Goldingay 1994).

## **Conclusions**

Detection probability (~0.58 per visit) for Yellow-bellied Gliders was weakly associated with survey covariates examined.

Occupancy in year 1 (1995) averaged 0.59 (0.4-0.77, 95% CI) per site, but was strongly, negatively associated with the extent of Alpine Ash, Low Peppermint and Western Forest types, with the forest type preference consistent with analyses by Kavanagh and Stanton (1998) for Bago-Maragle. In comparison, at Eden occupancy was 0.94 (0.84-0.98, 95 % CIs; Wintle et al. 2005), indicating suitable forest types for Yellow-bellied Gliders were more widespread than at Bago-Maragle.

Modelling of dynamic parameters, colonisation and extinction, revealed that colonisation probability was strongly, positively associated with the abundance of hollow trees at a site, whereas extinction probability was relatively stable (~0.19). The probability that an unoccupied site became colonised between surveys increased from ~0.15 at sites without hollow trees to >0.9 at sites with  $\geq 4$  hollow trees. This result makes sense in that gliders are unlikely to colonise sites with no or very few tree hollows.

The trend for Yellow-bellied Glider occupancy in Bago-Maragle forests showed a 16 % decline in occupancy between 1995 and 2019. The rate of decline was strongest between the 2013-15 sampling period and 2019, when annual rainfall was low (2018 and 2019). Only one third of sites were sampled in 2019, indicating estimates for this year were less robust and may have been biased by the extent of forest types at these sites (e.g.,  $29 \pm 6$  % Alpine Ash within 450 buffer of sampled sites). There was no evidence that the decline in occupancy was related to the extent of harvesting surrounding sites. For example, the recent decline in Yellow-bellied Gliders has occurred despite a significant reduction in the area harvested and the additional protective measures being implemented (Fig 6). We note that the hiatus in logging between 2009 and 2013 was associated with higher naïve occupancy, but as we have pointed out naïve occupancy is strongly affected by detectability. It should also be highlighted that there was no change in occupancy between 1995 and 2010 when harvesting continued at “typical” levels.

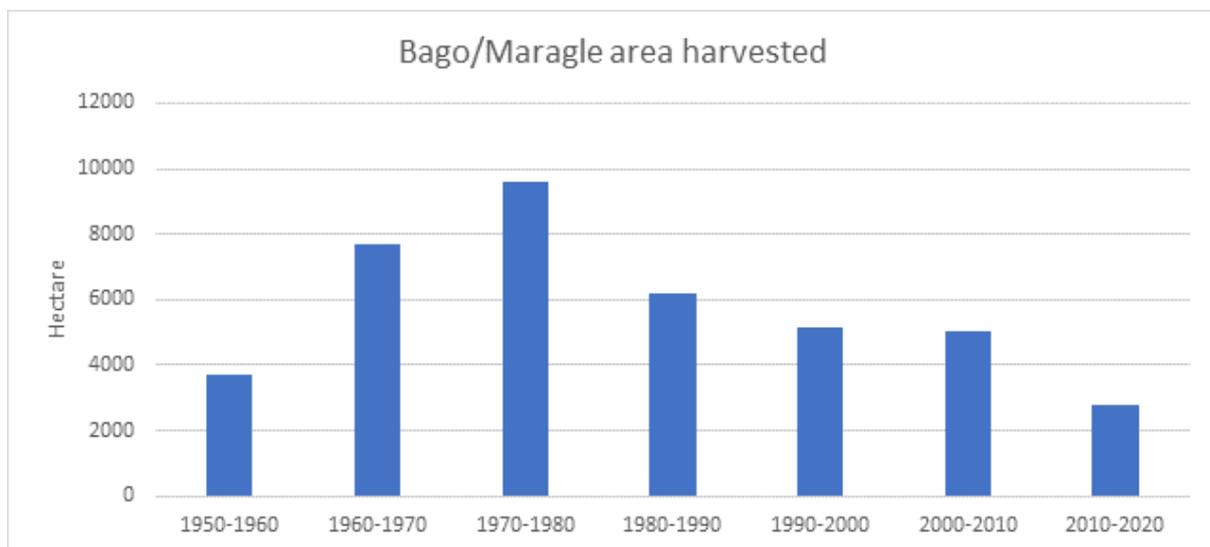


Fig 6: The gross area harvested per decade across Bago & Maragle SF. Source: R. Bilney

Finally, confidence intervals (95 %) for estimates of occupancy were relatively wide and this is likely to be due to a combination of the imprecise estimates of detection, as well as the variability in extent of different forest types at each of the sampling sites.

## **Recommendations**

Going forward, there are several recommendations to improve monitoring and management of the Yellow-bellied Glider population in Bago-Maragle forests.

1. Repeat visit sampling should be undertaken each year, with visits to be undertaken within a period in which occupancy is unlikely to change (i.e., no colonisation or extinction of sites within that period). Based on the estimate of detection in this assessment, at least 3 visits would be needed to have 90 % probability of detecting an animal at an occupied site (similar to the recommendation by Wintle et al. (2005) based on ideal survey conditions). Given this, remote recording devices (e.g., SM4) would be well-suited as a sampling method for Yellow-bellied Gliders, though this will be dependent on the availability of an acoustic recogniser for the calls of the species.
2. In one year, repeat visits should be carried out in multiple seasons to assess which season is best to carry out surveys.
3. Covariates that may influence detection (e.g., temperature, rainfall and wind during surveys) should continue to be recorded.
4. Based on initial occupancy (~0.59) in 1995 and assuming a detection probability of 0.58 per visit (using the current survey methodology), a 30 % decline in occupancy in 10 years would require five visits to achieve 0.9 power at an alpha of 0.1. Given current occupancy levels (0.43), a further 135 sites would be required to detect the same trend going forward. The high level of temporal replication required within survey periods at over 100 sites is unlikely to be practical using existing survey protocols. Passive acoustic sampling over 7 consecutive nights may yield a higher detection probability given entire nights are sampled using this approach.
5. The existing rotating panel design undertaken from 2013 onwards does not include annual sites that are monitored in each year. Sampling of some number of annual sites is required to adequately model colonisation and extinction parameters. Preliminary assessment suggests that 20-30 annual sites would provide reasonable estimates and confidence intervals for these dynamic parameters. This would require 60-70 sites to be sampled per year (20-30 annual sites and ~40 non-annual sites) in a three-year

rotating panel design (i.e., ~150 sites in total). If all sites can be sampled annually, this would be the preferred approach.

6. Future monitoring should focus on preferred forest types, which are now well-established for Yellow-bellied Gliders in Bago-Maragle forests.
7. The abundance of hollow trees is a strong predictor of colonisation probability for Yellow-bellied Gliders, with colonisation of unoccupied sites approaching 1 when at least four hollow trees per ha are present. As such, retention of hollow trees is clearly an important management action for Yellow-bellied Gliders. At sites where hollow trees are absent, targeted research could investigate the value of artificial hollows (e.g., cut-in/chainsaw hollows), but should account for the influence of forest types on Yellow-bellied Glider occupancy.

## References

Goldingay, R. L. (1994). Loud Calls of the Yellow-Bellied Glider, *Petaurus australis* Territorial Behaviour by an Arboreal Marsupial. *Australian Journal of Zoology*, **42**, 279-293.

Kambouris, P. J., Kavanagh, R. P., & Rowley, K. A. (2014). Distribution, habitat preferences and management of the yellow-bellied glider, *Petaurus australis*, on the Bago Plateau, New South Wales: a reassessment of the population and its status. *Wildlife Research*, **40**, 599-614.

Kavanagh, R., & Stanton, M. (1998). Nocturnal forest birds and arboreal marsupials of the southwestern slopes, New South Wales. *Australian Zoologist*, **30**, 449-466.

MacKenzie, D. I., Nichols, J. D., Lachman, G. B., Droege, S., Andrew Royle, J., & Langtimm, C. A. (2002). Estimating site occupancy rates when detection probabilities are less than one. *Ecology*, **83**, 2248-2255.

Wintle, B. A., Kavanagh, R. P., McCarthy, M. A., & Burgman, M. A. (2005). Estimating and dealing with detectability in occupancy surveys for forest owls and arboreal marsupials. *The Journal of Wildlife Management*, **69**, 905-917.