25. The Mallee Hawkeye Project: assessing most suitable habitat for threatened mallee birds using predictive fire history mapping

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Abstract

The Mallee Hawkeye Project is a collaborative research project between the Department of Environment and Primary Industries, La Trobe University and Deakin University, begun in 2011. It seeks to further understand factors affecting the response of species to fire, and to assess the impact of prescribed burning on biodiversity values. This presentation will provide an overview of the project's objectives and significant findings, with a special emphasis on our current research into the impact of prescribed burning on threatened mallee birds.

Fire has shaped the distribution of suitable habitat for many mallee bird species and inappropriate fire regimes may significantly threaten several endangered and iconic species such as the Malleefowl, Mallee Emu-wren, and Black-eared Miner. However, past research and management of threatened bird species in the Murray Mallee region has been constrained by scarce records and incomplete mapping of fire histories across political boundaries. There is urgent need to understand the effects of fire on this group of rare and declining species because current land management policy recommends historically unprecedented levels of fuel reduction burning. We collated a comprehensive dataset of species' occurrence for twelve key rare and threatened species in the region, including the Malleefowl, and used recent developments in fire history mapping to determine the extent to which their occurrences were driven by fire. We then assessed the regional distributions and identified areas of critical habitat for each species. These distribution models were then projected onto long-term future burning scenarios to evaluate how the extent of habitat was affected by proposed prescribed burning.

Part 1 – Distribution of threatened mallee birds with relation to post-fire age

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Introduction

Fire is a major component of disturbance regimes and shapes the distribution of suitable habitat for many species worldwide (e.g. Barro and Conard 1991, Woinarski and Recher 1997). Human-induced changes have altered historical fire regimes in a diversity of ecosystems including forests (e.g. Weber and Flannigan 1997, Gill 2012), shrublands (e.g. Conard and Weise 1998) and grasslands (e.g. van Wilgen, Govender *et al.* 2004). In Australia, inappropriate fire regimes, characterised by changes to the frequency, intensity, scale and seasonality of fire occurrence at a landscape level, have been identified as a major threat to the persistence of fifty-one terrestrial bird species (Woinarski and Recher 1997), and already responsible for the extinction of up to five (including three subspecies). In fire-prone regions, the effects of climate change are predicted to result in increased frequency and size of fires (Liu, Stanturf *et al.* 2010, Moritz, Parisien *et al.* 2012, Enright and Fontaine 2014) which will only exacerbate this risk. Ongoing fragmentation and loss of habitat have largely restricted many species to public reserve networks, further compounding the susceptibility of populations to both bushfire and planned burning management actions (Sandell, Tolhurst *et al.* 2006, Brown, Clarke *et al.* 2009).

Despite the need for urgent action, in many ecosystems knowledge of species' fire responses and even the nature of the historical fire regime itself are limited at best (e.g. Bradstock, Bedward *et al.* 2005, Driscoll, Lindenmayer *et al.* 2010). In the semi-arid mallee shrub environments of southeast Australia, there is a growing body of knowledge of both the landscape's fire history (Clarke, Avitabile *et al.* 2010, Avitabile, Callister *et al.* 2013, Callister, Griffioen *et al.* in prep.) and the association of birds with post-fire vegetation (e.g. Clarke 2005, Clarke, Boulton *et al*. 2005, Brown, Clarke *et al.* 2009). Novel landscape-scale studies have significantly advanced prior site-based understanding of those relationships (Taylor, Watson *et al.* 2012, Watson, Taylor *et al.* 2012). In spite of this, the capacity to investigate the effects of fire, at a landscape scale, on declining threatened and rare endemic bird species has been constrained by a scarcity of records and by the limited extent of reliable fire mapping to approximately forty years post-fire (Callister, Griffioen *et al.* in prep.). Yet it is many of these species, like the Malleefowl (*Leipoa ocellata*) and Black-eared Miner (*Manorina melanotis*), that are in fact considered most susceptible to altered fire regimes (Woinarski and Recher 1997) and that are believed to require long unburnt vegetation (Benshemesh 1990, Bradstock and Cohn 2002, Clarke 2005, Taylor, Watson *et al.* 2012). Thus, it is this suite of species that most urgently demand investigation.

The field of species distribution modelling has developed rapidly in recent years (Elith and Leathwick 2009). As better spatial layers have become more readily available due to advancements in satellite imaging technology, and as modelling capabilities have strengthened, it offers increasing application to species conservation management worldwide (e.g. Wintle, Elith *et al.* 2005, Addison, Rumpff *et al.* 2013). Within this field, Maxent is a widely used modelling method (Phillips, Anderson *et al.* 2006, Phillips and Dudík 2008) which is able to robustly predict species distributions at the landscape scale using presence-only datasets (e.g. Hernandez, Graham *et al.* 2006, Kaliontzopoulou, Brito *et al.* 2008, Kuemmerle, Perzanowski *et al.* 2010). This can lead to a significant expansion in the understanding of the requirements and distribution patterns of rare species. It has been found to perform favourably against a suite of established species distribution models including both presence-only and presence-absence methods (Hernandez, Graham *et al.* 2006, Phillips, Anderson *et al.* 2006, Dormann, Elith *et al.* 2013).

Critical to further informed study of threatened mallee bird distributions, predictive fire history mapping capable of differentiating old post-fire patches (Callister, Griffioen *et al.* in prep.) and predictive vegetation mapping (Haslem, Callister *et al.* 2010) that provides a uniform classification system have recently been developed for the Murray Mallee region. These predictive modelling tools provided the unique opportunity to obtain an understanding of the extent and location of suitable habitat available to threatened bird species across reserve and jurisdictional boundaries, and to determine to what extent their patterns of occurrence were driven by an association with fire.

We collated the most comprehensive presence-only historical datasets to date for twelve threatened and declining bird species of the region – the Malleefowl, Major Mitchell's Cockatoo (*Lophochroa leadbeateri*), Regent Parrot (*Polytelis anthopeplus*), Mallee Emu-wren (*Stipiturus mallee*), Striated Grass-wren (*Amytornis striatus*), Shy Heath-wren (*Calamanthus cautus*), Black-eared Miner, Southern Scrub-robin (*Drymodes brunniopygia*), Chestnut Quail-thrush (*Cinclosoma castanotum*), Red-lored Whistler (*Pachycephala rufogularis*), and Gilbert's Whistler (*Pachycephala inornata*). These datasets were used to develop species distribution models with Maxent, with the following objectives: (a) to determine to what extent species' occurrences were driven by post-fire vegetation age; (b) of those species with an association to post-fire vegetation age, to determine the nature of their response; (c) to predict the regional Murray Mallee distributions of all threatened bird species and identify where the most suitable habitat available to them was located; and (d) to identify whether any of these species were considered highly restricted within the region.

As discussed in the Malleefowl Forum presentation, models for the Malleefowl were not strong enough to discriminate areas of preferred habitat well, and so are not discussed further in this paper. Nonetheless, the approach and our findings have relevance within the wider context of threatened mallee bird species management, and presented here are results for those bird species for which strong models were developed.

Methods

Study area

Species distribution models were developed for the Murray Mallee study region which encompasses approximately 104,000 km^2 and covers areas of South Australia, New South Wales and Victoria.

Historical bird data

Distribution models were constructed for all threatened bird species occurring in the Murray Mallee region. This included all species identified under the Flora and Fauna Guarantee listed Victorian Mallee Bird Community: the Malleefowl, Mallee Emu-wren, Striated Grass-wren, Shy Heath-wren, Black-eared Miner, Southern Scrub-robin, Chestnut Quail-thrush, Red-lored Whistler, and Gilbert's Whistler. In addition, the following species were included: the Major Mitchell's Cockatoo, (listed as Threatened under Victorian legislation), Regent Parrot (listed as Threatened under Victorian legislation, and Vulnerable under Federal legislation), and Crested Bellbird (listed as Threatened under Victorian legislation).

All historical records for these species from 1999 to October 2010 were sought for the study region. Records were obtained from the New South Wales, South Australian and Victorian state agency databases, the Birds Australia Atlas, and data collected by Deakin, La Trobe and Monash University research groups. A presence-only record format was used for modelling, as the collection of datasets had comprised a range of methods including standardised transect and point count surveys, targeted threatened bird surveys with call playback and incidental records, which provided an incomplete record of absences.

Use of Maxent to develop species distribution models

Maximum entropy modelling, or Maxent, is a species distribution modelling (SDM) tool that operates within a defined area divided into a series of gridded cells, in which occurrence records for a focal species are compared with environmental variables at those localities to identify the species' distribution relative to those variables (Phillips, Anderson *et al*. 2006, Phillips and Dudík 2008). This information is used to build a model of species occurrence, as explained by those predictors found to be relevant. Maxent projects the predicted species occurrence to un-sampled cells within the defined area, based on the values of environmental predictors at those cells. Thus it generates a spatially explicit simulation of the species' predicted distribution in a given geographic space. Values of environmental predictor variables are compared between the occurrence localities for the given species and random 'background' points in the study region (where species presence is unknown). Maxent was chosen for this study because of its ability to robustly predict species distributions using presence-only datasets, and small sample datasets; typical of rare species (e.g. Phillips, Anderson *et al.* 2006, Anderson and Gonzalez Jr 2011).

Environmental variables

<u>Post-fire age class and vegetation type variables</u>

Our primary aim was to determine the distribution of threatened species with regard to seral class; accordingly, fire age class and vegetation type were selected as environmental predictor variables. Predictive vegetation class and fire-age class models developed by the Mallee Fire & Biodiversity Project (Watson, Taylor *et al*. 2012) were used because they represented the best spatially explicit knowledge available for the region. We amalgamated known fire scars with the distribution map of predictive post-fire vegetation age developed by Callister, Griffioen *et al.* (in prep.) for sites burnt prior to 1972 for the Murray Mallee region (Figure 1). Threatened bird records were assigned post-fire ages relevant to the year in which they were recorded. Age class intervals were assigned post-analysis and are defined by the following categories – very young (1-10 years), young (11-20 years), intermediate (21-40 years), late intermediate (41-50 years), old (51-70 years) and very old (71+ years).

The regional distribution of vegetation types was obtained from the predictive model developed by Haslem, Callister et al. (2010) (Figure 2). This model identified four distinct vegetation types - *Triodia*, chenopod, heathy and shrubby mallee. Species distribution models were developed only for those areas identified as either *Triodia* or chenopod mallee, because threatened bird sampling was not conducted across an adequate distribution of age classes within either heathy or shrubby mallee to permit their inclusion.

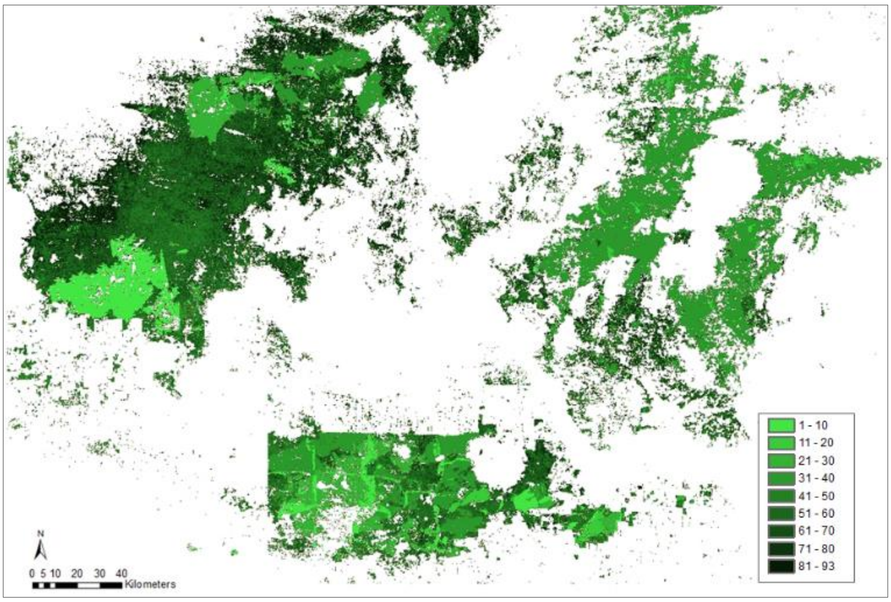

Figure 1. Distribution of post-fire age classes for mallee vegetation at 2011, Murray Mallee region.

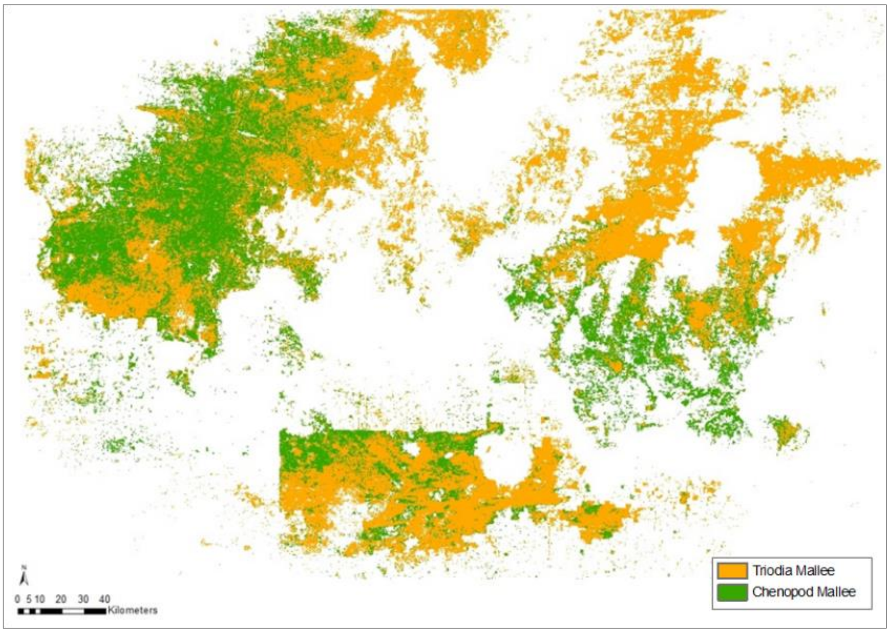

Figure 2. Distribution of Triodia and chenopod mallee vegetation classes across the Murray Mallee region.

Climatic zone variables

Rainfall and temperature gradients exist across the region and accordingly, the following predictor variables were considered biologically relevant to species distributions and included for contention in all species distribution models: variables relating to climatic zone (mean annual maximum and minimum temperatures and mean annual precipitation) and variables relating to climatic extremes (mean maximum temperature of the hottest month, January, and mean minimum temperature of the coldest month, July, and mean precipitation of the driest and wettest months, April and October respectively). Climatic spatial layers were obtained for the region from the Bureau of Meteorology (2014). Long-term data averages were calculated from records for the thirty year period 1961 – 1990 and depict regional climatic zones.

Estimation of sampling bias

We used target group sampling (Phillips and Dudík 2008) to contend with sampling biases likely to be present in our historical datasets. However, this was unable to negate the effect of biased sampling effort in intermediate post-fire age classes (20-60 years, Figure 3), with the greatest scarcity of records in very old age classes (>80 years) relative to their prevalence in the landscape at 2011. Given the perceived importance of older mallee to many of these threatened species (Clarke 2005, Watson, Taylor *et al.* 2012, Taylor, Watson *et al.* 2013), we remain cautious in providing any valuation of this age class beyond 80 years. Thus, individual species responses to post-fire age class were prepared for the age class distribution of 1-80 years only.

Model selection

For each species, outlier records were removed according to the protocol specified by Van Selst and Jolicoeur (1994), and raw models were then run using the full set of environmental predictors. Predictors were tested, by species, for collinearity using the Pearson's correlation coefficient. Any pairs of predictor variables with a coefficient $\geq$0.7 were deemed collinear (Dormann, Elith *et al.* 2013). The predictor of the pair found to have a lower explanatory effect on the test dataset gain was removed from the model (as done by Kuemmerle, Perzanowski *et al.* 2010). Models were then successively re-run until all predictors shown to detract from or contribute ~0% to model predictive performance and test gain were removed. This variable subset formed the final model.

Final models were screened and accepted for analysis on the basis of area under the curve, or AUC, performance. Models are typically evaluated using the strength of the AUC value (0-1) which is a rank-based statistic that provides a measure of model discrimination, or ability to rank cells in terms of predicted presences and background points (Yackulic, Chandler *et al.* 2013). Evaluation using the AUC provides a reliable ranking of areas in terms of relative habitat value (Wintle, Elith *et al.* 2005). Broadly, an AUC of $\leq$0.5 indicates a discriminatory ability of no better than random, 0.5-0.7 indicates some discriminatory ability, 0.7-0.9 indicates reasonable discrimination, and >0.9 indicates strong discrimination (Pearce and Ferrier 2000). Only models with AUCs >0.75 were accepted (Anderson, Dudík *et al.* 2006, Reside, VanDerWal *et al.* 2012), resulting in the exclusion of models developed for the Malleefowl, Major Mitchell's Cockatoo and Crested Bellbird.

Final species distribution models of relative predicted habitat suitability were projected onto the Murray Mallee study region as at 2011.

Results

Maxent distribution models with high performance (AUC values $\geq$0.75) were developed for nine of twelve threatened bird species (Table 1), but did not include the Malleefowl. Results for other species are discussed here.

Post-fire vegetation age was retained by all nine models and was a strong predictor for many species. Vegetation type showed relatively minor predictive power, and was retained in models for only three species. Nevertheless, its importance in predicting the occurrence of those three species was high, being the highest ranking predictor for the Mallee Emu-wren and the third highest ranked predictor for the Striated Grasswren and Red-lored Whistler.

Table 1. Final Maxent species distribution models with AUC $\geq$0.75, listing percent contributions of all predictors.

Species	Final Model AUC	Standard Deviation	Predictor 1	Predictor 1 contribution	Predictor 2	Predictor 2 contribution	Predictor 3	Predictor 3 contribution	Predictor 4	Predictor 4 contribution
Regent Parrot	0.844	0.039	Mean annual max temp	39.5	Highest temp hottest month	30.3	Post fire age	23.4	Mean annual min temp	5.4
Mallee Emu-wren	0.832	0.044	Vegetation type	36.2	Post fire age	35.9	Mean precip highest month	22.9	Mean annual total precip	5.1
Striated Grasswren	0.849	0.017	Mean precip driest month	31.7	Post fire age	24.8	Vegetation type	23.3	Mean annual total precip	12.6
Shy Heathwren	0.845	0.024	Mean annual min temp	41.4	Post fire age	32.0	Mean precip lowest month	13.1	Mean precip highest month	8.4
Black-eared Miner	0.873	0.025	Mean precip driest month	71.8	Post fire age	11.4	Mean annual max temp	9.9	Lowest temp coldest month	6.9
Southern Scrub-robin	0.806	0.024	Post fire age	30.6	Mean annual total precip	24.6	Mean precip lowest month	16.8	Mean annual min temp	15.2
Chestnut Quail-thrush	0.773	0.022	Mean annual min temp	42.3	Post fire age	38.2	Highest temp hottest month	7.6	Lowest temp coldest month	6.4
Red-lored Whistler	0.920	0.014	Post fire age class	41.0	Mean precip lowest month	39.0	Vegetation type	10.7	Mean annual max temp	9.3
Gilbert's Whistler	0.784	0.031	Mean annual total precip	45.9	Post fire age	29.8	Mean annual min temp	14.5	Mean precip lowest month	9.8

Species	Predictor 5	Predictor 5 contribution	Predictor 6	Predictor 6 contribution
Regent Parrot	Lowest temp coldest month	1.4	-	
Mallee Emu-wren	-		-	
Striated Grasswren	Mean annual min temp	7.5	-	
Shy Heathwren	Lowest temp coldest month	2.9	Mean annual max temp	2.3
Black-eared Miner	-		-	
Southern Scrub-robin	Lowest temp coldest month	12.8	-	
Chestnut Quail-thrush	Mean precip highest month	5.6	-	
Red-lored Whistler	-		-	
Gilbert's Whistler	-		-	

Variables relating to climatic zone provided high predictive power across all models; in particular measures of precipitation ranked highly. Whilst predictor variables relating to temperature were retained by a majority of models, most made only a small percent contribution.

Species predominantly showed maximum occurrence within intermediate, late intermediate and old post-fire vegetation classes (Figure 4). Maximum occurrence was highest in old post-fire age classes for the Striated Grass-wren (41-60 years), Black-eared Miner (41-60 years) and Gilbert's Whistler (41-60 years), highest in intermediate and late intermediate for the Regent Parrot (21-50 years), and highest in intermediate for the remaining species – Mallee Emu-wren (21-30 years), Shy Heathwren (21-40 years), Southern Scrub-robin (31-40 years), Chestnut Quail-thrush (31-40 years), and Red-lored Whistler (21-40 years). For all species except the Shy Heathwren, lowest occurrence was found in very young post-fire vegetation (1-10 years); young post-fire vegetation (11-20 years) similarly had lowest occurrence for all species except the Mallee Emu-wren. However, this species was not found to occupy any sites of <18 years post fire. Association with post-fire ages beyond 80 years remained undefined.

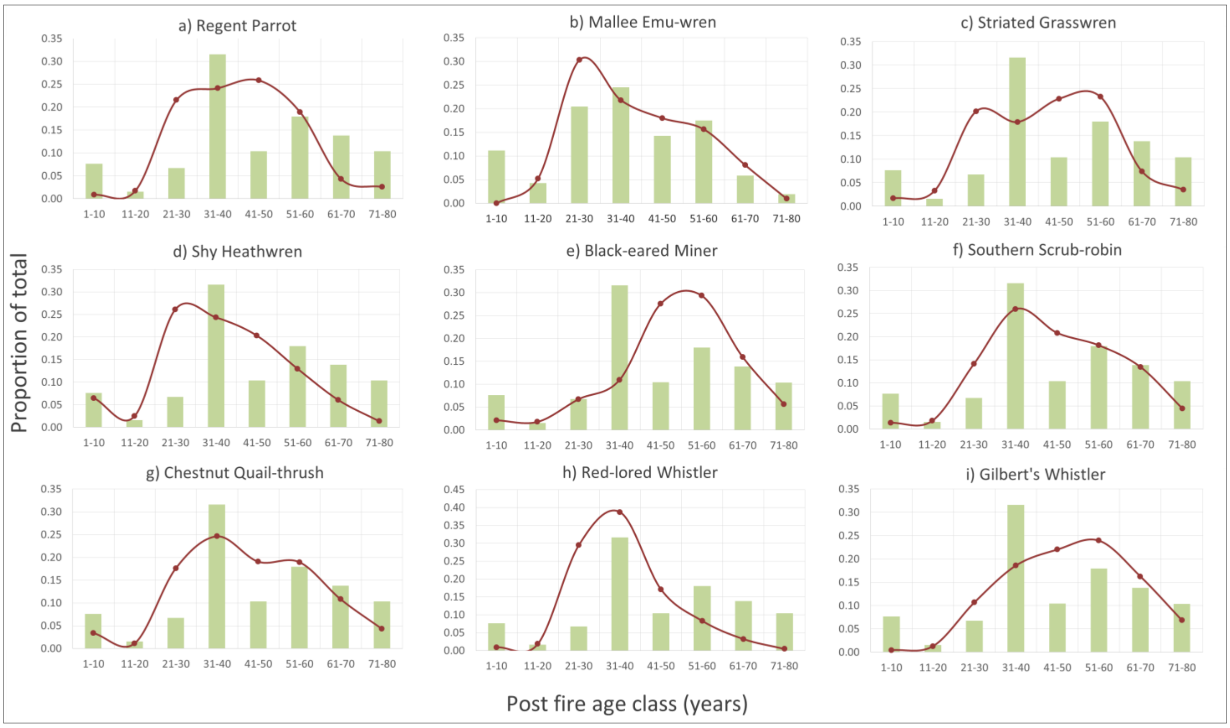

Figure 4. Proportion of total occurrence records for each species in each post-fire vegetation age class (red line), and available post-fire vegetation age classes as at 2011 for the Murray Mallee region (expressed as a proportion of total, green bars).

Of those three species which retained vegetation type as a predictor, response type was uniform; the Mallee Emu-wren, Red-lored Whistler and Striated Grasswren all demonstrated a strong preference for *Triodia* mallee (Figure 5).

Regional distribution of most suitable habitat differed between species, both in total patch extent and its configuration. Distribution models for many species predicted small and isolated patches, both within and off reserve, as highly suitable. Distribution of most suitable habitat was highly restricted within the study region for the Red-lored Whistler and Mallee Emu-wren (Figure 6). Small zones of suitable habitat were chiefly restricted to small areas within one or several connecting public reserves. Most suitable habitat for the Mallee Emu-wren was restricted *a priori* to the southeast of the study region, and here it was strongly associated with *Triodia* mallee, located in small, discontinuous patches of intermediate age vegetation located in the Hattah Kulkyne, Annuello and eastern Murray-Sunset Victorian reserves. Suitable habitat for the Red-lored Whistler was predominantly restricted to the southeast of the study region, and was located in a mostly connected series of small patches in the west of the Murray-Sunset

Victorian reserve, likewise chiefly in *Triodia* mallee. In contrast, only minor distinction in relative habitat suitability was seen across the region for both the Chestnut Quail-thrush and Gilbert’s Whistler.

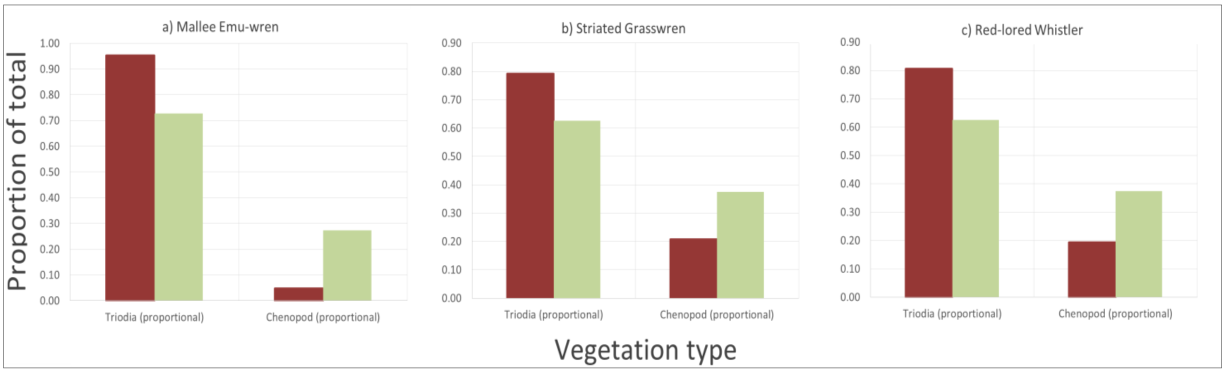

Figure 5. Proportion of total occurrence records for each species in each vegetation type (red column), and available vegetation types (expressed as a proportion of total, green bars; Mallee Emu-wren – for the Victorian mallee only, and Striated Grasswren, Red-lored Whistler - for the Murray Mallee region).

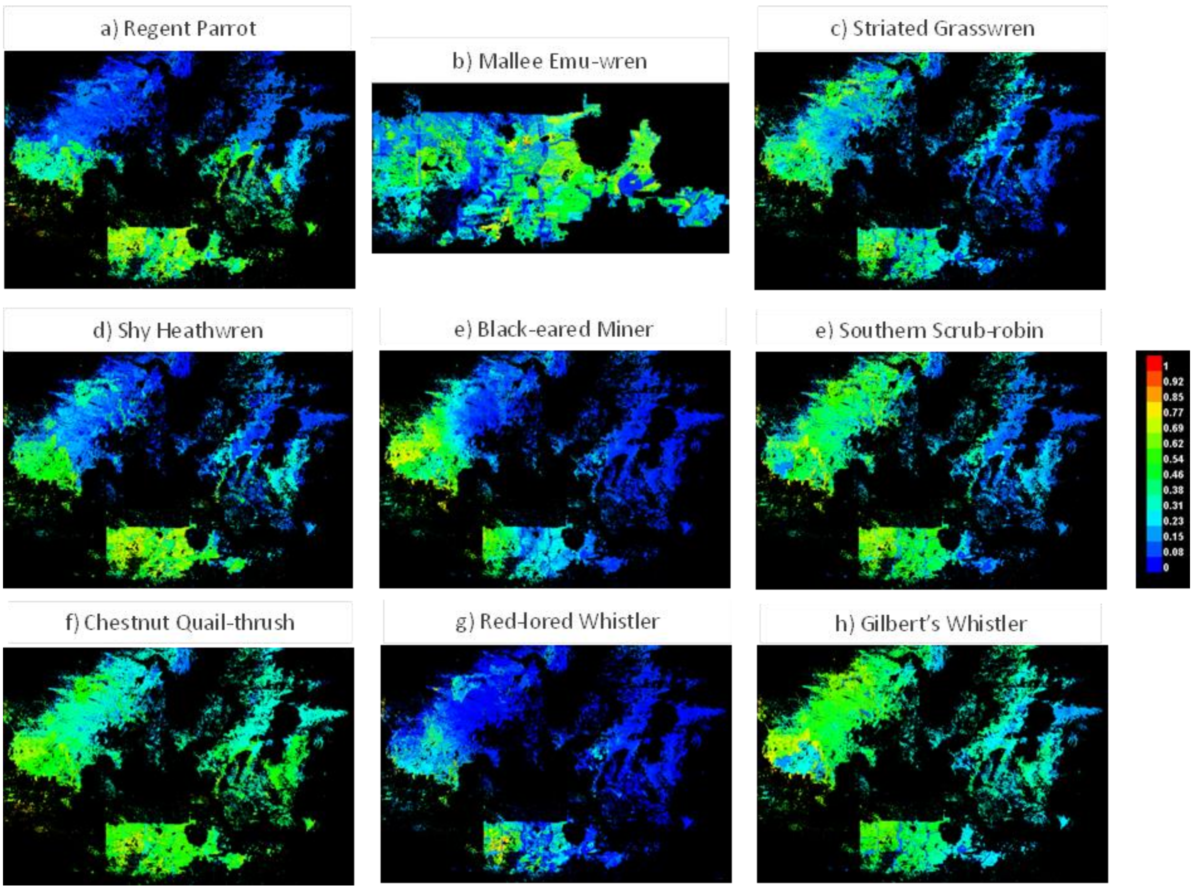

Figure 6. Maxent model distribution maps for the Murray Mallee region at 2011. Pixels of highest relative predicted occurrence (approaching 1) are indicated by red, intermediate (~0.5) green, and lowest (approaching 0) blue.

Discussion

These findings significantly advance our knowledge of the extent and distribution of suitable habitat available to declining mallee bird species, as we were able to associate historical threatened bird occurrences with the first spatially explicit distributions for both post fire age class (Callister, Griffioen *et al.* in prep.) and vegetation (Haslem, Callister *et al.* 2010), portrayed on a common scale across jurisdictional boundaries. Species' association with post fire age classes and vegetation types were broadly consistent with prior modelling (Clarke 2005, e.g. Clarke, Boulton *et al.* 2005, Brown, Clarke *et al.* 2009, Watson, Taylor *et al.* 2012). Our work identified that under prevalent conditions of low rainfall, all nine species for which strong predictive models were able to be developed showed an association with post-fire age class. Distinctly, almost all species showed a disproportionate association with vegetation >20 years. A number of species showed peaks of occurrence in intermediate age classes (21-40 years) which subsequently declined, likely to be affiliated with their requirements for structural elements and habitat resources shown to decline with time (Haslem, Kelly *et al.* 2011). Analogous to prior work, the Mallee Emu-wren, Red-lored Whistler and Striated Grasswren were found to be strongly associated with *Triodia* mallee. These species are widely recognised as *Triodia* specialists (Higgins and Peter 2002, Clarke 2005, Brown, Clarke *et al.* 2009). The Mallee Emu-wren for example utilises *Triodia* hummocks for nesting and refuge purposes (Brown, Clarke *et al.* 2009) and these begin to senesce approximately 40 years post fire (Haslem, Kelly *et al.* 2011), consistent with the birds' decline in occurrence. The Striated Grasswren, Black-eared Miner and Gilbert's Whistler were most strongly associated with late intermediate (41-50 years) and old (51-70 years) vegetation, consistent with prior studies (McLaughlin 1990, Clarke 2005, Clarke, Boulton *et al.* 2005, Watson, Taylor *et al.* 2012).

Clarke (2005) remarked that upper limits to preferred age class ranges for this suite of species had been identified by few studies due to the restrictions of satellite imagery (Callister, Griffioen *et al.* in prep.), which was limited to assigning anything above 35 years to 'old' (Clarke, Avitabile *et al.* 2010, Avitabile, Callister *et al.* 2013). Our work has more than doubled the length of 'known age' post fire age classes to 80 years. However, due to the scarcity of records available for very old post-fire age classes (>80 years), and constrained by our use of historical presence-only data, species use and requirement for resources affiliated with those still older age classes remain inconclusive. However, response curves developed by Watson, Taylor *et al.* (2012) using presence-absence data suggest that for species like the Gilbert's Whistler and Southern Scrub-robin, occurrence in very old vegetation (>70 years) continues to incline to at least 100 years post-fire. Evidence from prior studies indicates such older post-fire age classes are unequivocally important for the breeding requirements of species like the Black-eared Miner (Higgins, Peter *et al.* 2001) and others unable to be strongly modelled here, like the Major Mitchell's Cockatoo and Malleefowl (Benshemesh 1990). The Major Mitchell's Cockatoo, for example, requires hollows for breeding and has been found to occupy those of >15cm in dimension (Higgins 1999). These do not begin to develop in live stems until approximately 60 years post fire, and would require still many more decades for a tree to develop hollows large enough to accommodate this species (Higgins 1999, Haslem, Kelly *et al.* 2011). Thus, it must be emphasised that these distribution models may in fact underestimate the suitability of very old vegetation across the landscape.

Notably, limited preference for very young and young age classes (1-20 years) was exhibited. Only the Shy Heathwren showed any association with youngest vegetation (1-10 years), consistent with its habitat association with shrubs and the shrub-like coppicing of *Eucalyptus* spp. which occurs in early successional stages following fire (Higgins and Peter 2002).

Models showed an uneven distribution of most suitable habitat for species across reserve and regional boundaries. The highly restricted extent of suitable habitat identified for the Mallee Emu-wren and Red-lored Whistler reflect evidence from contracting occurrence records for these species in recent decades (Higgins, Peter *et al.* 2001, Clarke 2005, Brown, Clarke *et al.* 2009). For mallee endemics like these and the Black-eared Miner, these Murray Mallee distributions represent a significant extent, if not the entirety, of their global distributions and emphasise their vulnerability to stochastic events and land management actions at a single reserve scale (Brown, Clarke *et al.* 2009). This has been highlighted by recent population losses of the Mallee Emu-wren following bushfire events at Ngarkat Conservation Park and Bronzewing Flora and Fauna Reserve in January 2014.

Identification of suitable habitat by these distribution models is constrained by species requirements for minimum patch size and by their dispersal abilities, not just between reserves, but between suitable habitat patches within reserves. Firstly, we doubt off-reserve isolates (identified as highly suitable for several species, including the Black-eared Miner and Chestnut Quail-thrush by Maxent modelling) are of significant value for the species. The likelihood of species occupation and sustained use of such isolated or small fragments is unlikely (Luck, Possingham *et al.* 1999, Ford, Barrett *et al.* 2001) but remains to be examined. Secondly, within reserves there is an urgent need to better understand the effects of species dispersal capacity and minimum patch size needed to sustain both viable meta- and sub-populations. Radford and Bennett (2004) and Radford and Bennett (2006), for example, identified that the White-browed Treecreeper (*Climacteris affinis*) displayed evidence of a patch occupancy threshold, where it was limited in its ability to disperse between patches of no more than three kilometres apart. This threshold was exhibited even between suitable patches separated within natural vegetation matrices, although the threshold extended to eight kilometres. Such evidence of dispersal limitations highlights that ostensibly suitable habitat is not a guarantee of occupation. Similarly, minimum patch size requirements for a species are likely to render much purported suitable habitat to be of little use. Taylor, Watson *et al.* (2012) remarked that the response of species to fire-mediated spatial properties of landscapes (e.g. the size and configuration of patches) remains virtually unknown (Bradstock, Bedward *et al.* 2005, Driscoll, Lindenmayer *et al.* 2010), including for most of those species studied here. Preliminary assessments by Clarke, Boulton *et al.* (2005) speculated that greater than 13,000ha was required to sustain a viable Black-eared Miner population, and further analysis showed the species was positively associated with a low diversity of age classes (Taylor, Watson *et al.* 2013). Such findings would indicate that smaller and patchy pockets of suitable habitat identified by Maxent, like those in southeast South Australia, are in fact of little real use for the species. We require a better understanding of how such limiting factors affect these species to better interpret the spatial connectivity, and their likely use, of habitat.

These cautionary comments highlight that that use of species distribution models for conservation assessment purposes poses the danger of “adding up pixels”, whereby artificially high estimates of the extent of suitable habitat are generated which might not provide an immediate basis for registering concern. Nevertheless, to date ecological management of key species has chiefly been limited to the use of historical occurrence records to obtain an understanding of their distributions. However, catering for species’ needs must take into consideration the transient nature of post-fire vegetation age by managing for future age class distributions and ensuring its spatial connectivity over time to enable re-colonisation by species. These models significantly advance our understanding of threatened species relationships to fire age class and subsequent patterns of distribution at a landscape scale, thus providing a much greater capacity to do that.

Part 2 – Predicting the impact of future planned burning on a suite of threatened mallee bird species

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Introduction

Worldwide, fire-prone ecosystems are increasingly exposed to altered fire regimes resulting from changed land use practices (e.g. Westerling, Hidalgo *et al.* 2006, Liu, Stanturf *et al.* 2010, Keeley, Bond *et al.* 2012). Continued human population expansion at the wildland-urban interface generates increasing risk of such bushfires threatening human life and built assets (Gill and Stephens 2009, Penman, Bradstock *et al.* 2014). In response, planned fire has increasingly been employed as a management tool. Its objectives are two-fold – it is used widely in attempt to reduce the risk of bushfire causing loss to life and property (Reinhardt, Keane *et al.* 2008, Penman, Bradstock *et al.* 2014), and to better manage the ecological processes and values within remnant ecosystems (Morrison, Buckney *et al.* 1996, Burrows 2008, Penman, Christie *et al.* 2011). To marry these potentially conflicting goals is highly challenging. Questions remain both about the effectiveness of planned burns in mitigating the rate of spread and intensity of bushfire (Fernandes and Botelho 2003, Boer, Sadler *et al.* 2009, Penman, Bradstock *et al.* 2014), and how best to burn in order to maintain and promote ecological values (Driscoll, Lindenmayer *et al.* 2010, Pastro, Dickman *et al*. 2011). Increasing the capacity for fire management actions to prevent landscape scale bushfires is, however, often a key aim of both objectives (Penman, Bradstock *et al.* 2014).

Adequate knowledge of species' fire requirements is critical for their effective conservation management, but in many Australian systems such knowledge about historical disturbance regimes remains limited at best (Parr and Andersen 2006, Clarke 2008, Driscoll, Lindenmayer *et al.* 2010). In densely populated and fire-prone south-eastern Australia, state policy employs the use of ecological traits of plants to identify appropriate thresholds within which to apply planned burns to ecosystems (Cheal 2010, Cheal 2012). This assumes that by allowing for regeneration of key fire-sensitive plant species, the subsequent provision of a range of post-fire age classes in the landscape will be adequate for associated fauna. Faunal associations with post-fire age class suggest a much more complex range of responses even within single ecosystems. Landscape scale studies conducted in the semi-arid mallee (Taylor, Watson *et al.* 2012, Taylor, Watson *et al.* 2013) and work in other fire prone environments (Bradstock and Cohn 2002, Pastro, Dickman *et al.* 2011) has shown that there is no universal approach to fire management that is likely to benefit all species. Given this challenge, Pastro, Dickman *et al.* (2011) suggest that it would be most useful to focus fire management on those high priority, fire-sensitive species or species groups.

In the aftermath of recent bushfire disasters in southeast Australia, and with worsening climate change projected to significantly increase fire risk (Liu, Stanturf *et al.* 2010, Moritz, Parisien *et al.* 2012), the imperative to effectively manage fuel loads intensifies (Enright and Fontaine 2014). Following a Royal Commission into the Black Saturday fires of 2009, Victorian state land management policy now recommends a significant increase to the amount of planned burning on public land to facilitate protection of life and property (Teague, McLeod *et al.* 2010). This represents an historically unprecedented level of burning for ecosystems such as the semi-arid mallee in north western Victoria (Watson, Taylor *et al.* 2012, Avitabile, Callister *et al.* 2013). Planned fire acts in a similar manner to bushfire in this ecosystem, whereby mallee eucalypts stands die and stands are replaced by regenerating lignotubers. This allows the likely impact of planned fire on biota to be evaluated using our understanding of typical post-fire responses. Taylor, Watson *et al.* (2012) and Taylor, Watson *et al.* (2013) evaluated a range of likely fire management strategies for the region using landscape-scale fire mosaics and found a higher species richness of both common and rare birds in vegetation that had not been burnt for at least 35 years. Such evidence suggests increased fire frequency would be likely to jeopardise the status of some of its already threatened and endemic biota.

We recently identified the fire-responses and spatial distributions of a range of declining and fire-sensitive mallee bird species, including several endemics, across the Murray Mallee region. This suite of species has previously been recognised as most at-risk to inappropriate fire regimes (Woinarski and Recher 1997), and our models confirmed prior work showing that most exhibited preference for older post-fire vegetation (ranging from 20-60 years; Bradstock and Cohn 2002, Clarke 2005, Clarke, Boulton *et al.* 2005, Brown, Clarke *et al.* 2009, Watson, Taylor *et al.* 2012). This provided us with the unique opportunity to employ a spatially explicit approach to assess how the long term application of varying levels of planned burning would affect both bushfire occurrence and species persistence at a landscape scale, utilising a Victorian mallee Landscape Management Unit (LMU) as a case study. In doing this we sought to identify whether a maximum threshold to the extent of planned burning in this landscape could be determined, above which available habitat declined. Our objectives were: a) to develop a series of realistic, spatially explicit future planned burning scenarios, spanning two decades, which depicted varying levels of burning per annum (0, 1.5, 3 and 5%) and incorporated large wildfire events; b) to demonstrate changes to the distribution of post-fire age classes in the reserve, following application of these future planned burning scenarios for two decades; c) to quantitatively show how the extent of most suitable habitat available to those threatened species differed between scenarios and changed over the course of time, and d) from this, to identify those species most likely to be negatively impacted by planned burning management actions.

Methods

Study design

The semi-arid Victorian Mallee Landscape Management Unit (LMU), comprising Annuello Flora and Fauna Reserve, Hattah-Kulkyne and Murray-Sunset National Parks (excluding riverine vegetation and the Taparoo Grassland respectively), was selected as a case study region. Twenty-one years of spatially explicit planned burning scenarios were prepared for analysis to allow the cumulative, long-term impact of four different burning strategies to be assessed.

Development of pseudo-Fire Operations Plans

Staff from the Department of Environment and Primary Industries (DEPI) and Parks Victoria (PV) prepared the planned burning scenarios in a format consistent with current departmental protocol, to ensure they were of value for informing management approaches. The Fire Operations Plan (FOP) is the method currently employed to formulate the location, size and timing of planned burn treatments on Victorian public land (DEPI 2014). The twenty-one years of planned burn scenarios were prepared as spatially explicit 'pseudo' Fire Operation Plans (FOPs) for the LMU. Consistent with workplace FOPs, each plan was of three years' length which required the preparation of seven FOP scenarios for each treatment of planned burning, spanning twenty-one years in total (2011-13, 2014-16, 2017-19, 2020-22, 2023-25, 2026-28, 2029-31). Plans were prepared using predominantly a strategic corridor burning approach; repeated treatment of areas at short intervals was not employed. Rules governing the design of the planned burning scenarios were consistent with those already used by DEPI and PV for planning planned burns in the Victorian Mallee Fire District (Department of Sustainability and Environment 2008).

The pseudo-FOP scenarios were built upon the real fire history of the LMU as at 2011 using DEPI mapping records for the region. The preparation of pseudo-FOPs was led by the PV Fire and Environment Program Officer, Kathryn Schneider, with development carried out by the Mallee District planned burning team comprising both PV and DEPI staff, whilst the digitisation of pseudo-FOPs and application of burn coverage was performed by the DEPI Hawkeye Project Officer, Natasha Schedvin.

Burn percentage comparisons

Four levels of planned burning treatments per year were prepared, each of twenty-one years in scope. These were: no planned burning in the LMU (bushfires only, at levels simulating average area burnt *p.a.* over the past 35 years); 1.5% of planned burning *p.a.*; this approximates the long-term average of annual burning (bushfire plus planned burning) in the Murray Mallee over the last thirty years (Watson, Taylor *et al.* 2012, Avitabile, Callister *et al.* 2013); 3% of planned burning *p.a.* to represent a potential compromise target for this ecosystem, and which was consistent with the Victorian Mallee District target for 2011; and 5% planned burning *p.a.* which was consistent with the recommendation of the Victorian Bushfire Royal Commission from 2012 and onwards (Teague, McLeod *et al.* 2010).

Generation of simulated bushfires

It was considered imperative to incorporate stochastic and potentially significant events such as bushfire into the scenarios. The Victorian mallee is a fire-prone environment subjected to regular lightning ignited bushfires; large bushfires (of greater than 10,000 ha) have occurred approximately every ten years since the onset of satellite imagery in 1972 (Avitabile, Callister *et al.* 2013). Whilst fires of small size occur in any given year, large fire events are responsible for the majority of area burned by bushfire and substantially affect vegetation age-class distributions. Inclusion of simulated bushfires above a lower size limit was considered necessary to enable more relevant inferences to be made about the future suitability of species habitat following the two-decade projection.

The average historical size of large fires for the Victorian mallee was used to inform simulated fire size (36,000 ha for Murray-Sunset National Park, Avitabile *et al.* 2013). The bushfire simulation program *Phoenix Rapidfire V3.9.0.0* (developed by Tolhurst, Shields *et al.* 2008) was used to incorporate simulated bushfires into the pseudo-FOP scenarios. This program was developed in Victoria and is used by DEPI to model complex fire behaviour. The program is operated in a simulated environment which replicates the topographic features and fuel loads of a defined, real landscape. Spatially explicit simulations of bushfire direction of spread and extent are generated using fire behaviour models based on ignition location, fuel parameters and input of weather conditions. Simulated bushfires respond to changes in weather and fuel parameters as time progresses. Operation of Phoenix software was performed by the PV Fire and Environment Program Officer, who also undertakes the fire behaviour analyst role during bushfire suppression events.

Species distribution models

Maxent threatened bird species distribution models developed for the Murray Mallee region were projected onto the Victorian LMU future planned burning scenarios. Only species for which strong models (AUC $\geq$ 0.75) could be generated and which exhibited a strong response to post-fire age class were included in analyses. Distribution models for the Regent Parrot (*Polytelis anthopeplus*), Mallee Emu-wren (*Stipiturus mallee*), Striated Grass-wren (*Amytornis striatus*), Shy Heathwren (*Calamanthus cautus*), Black-eared Miner (*Manorina melanotis*), Southern Scrub-robin (*Drymodes brunniopygia*), Chestnut Quail-thrush (*Cinclosoma castanotum*), Red-lored Whistler (*Pachycephala rufogularis*) and Gilbert's Whistler (*Pachycephala inornata*) were projected onto scenarios.

Changes to extent of suitable habitat for each species were measured following the application of each three year planned burning treatment (0, 1.5, 3 and 5% *p.a.*), from 2011 to 2032. The application of a binary, non-species specific threshold using predicted occurrence rate to delineate species presence and absence was not considered suitable as it would likely mask potential loss or gain of suitable habitat for rare species, which have lower occurrence rates (Nenzén and Araújo 2011, Warren, Wright *et al.* 2014). The upper 20th percentile of a species' predicted occurrence rate was used instead. This index identifies those areas of the LMU in which the predicted occurrence rate was equal to or greater than the top 20th percentile of the predicted occurrence rates for that particular species in 2011. One could then examine how the availability (in hectares) of this most suitable habitat changed in extent and location under various burning scenarios. Being percentile based, this had the advantage of identifying occurrence rates specific to each species.

Results

True distribution of fire age classes in the Victorian reserve as at 2011 showed an approximately normal distribution, with spikes in the youngest and intermediate-older age classes (Figure 1). By 2032, following 21 years without planned burning the simulated landscape showed a considerable increase in the amount of very old vegetation. Bushfire events in each decade converted a small portion of the total reserve to the youngest age class (1-10 years). Application of 1.5% planned burning *p.a.* plus bushfire events resulted in a bimodal distribution by 2032. Application of 3% planned burning *p.a.* to 2032 plus bushfire events resulted in an age class distribution strongly dominated by vegetation of twenty years of age or less which comprised almost two-thirds of total reserve area. Application of 5% planned burning *p.a.* to 2032 plus bushfire events saw a rapid conversion of over eighty percent of the reserve to vegetation of 20 years of age or less. All older age classes lost substantial area to burning treatments. Under 3 and 5% *p.a.,* remaining remnant patches of very old vegetation were small and highly fragmented (Figure 2).

Simulated bushfire events in each decade made progressively smaller contributions to the total amount of vegetation in the youngest age class in 0, 1.5, 3 and 5% *p.a.* planned burning treatments (Figures 3, 4). However, despite the mitigating effect of higher planned burning on bushfire size (Figure 3), with increased planned burning the total extent of recently burnt vegetation in the reserve increased.

The amount of planned burning over two decades had a varied impact on the extent of most suitable habitat available in the reserve for each species. Following two decades of projections, a higher amount of planned burning was associated with substantial reductions in the extent of most suitable habitat for two of nine species, the Mallee Emu-wren and Black-eared Miner. The Mallee Emu-wren was one of few species to show any increase in total extent available relative to 2011, under any treatment type; in this case no planned burning resulted in a two percent expansion of most suitable habitat. Under 5% and 3% *p.a.* extent was thirteen and eight percent less, respectively, than extent available under no planned burning. Additionally, most suitable habitat for the Mallee Emu-wren was widely dispersed under all planned burning treatments. The configuration of most suitable habitat under 5% *p.a.* in particular comprised very small and isolated patches (Figure 6). For the Black-eared Miner a similar considerable decline was exhibited following higher amount of burning, where under 5 and 3% *p.a.,* extent of most suitable habitat was reduced by ten and eight percent relative to the extent available under no planned burning.

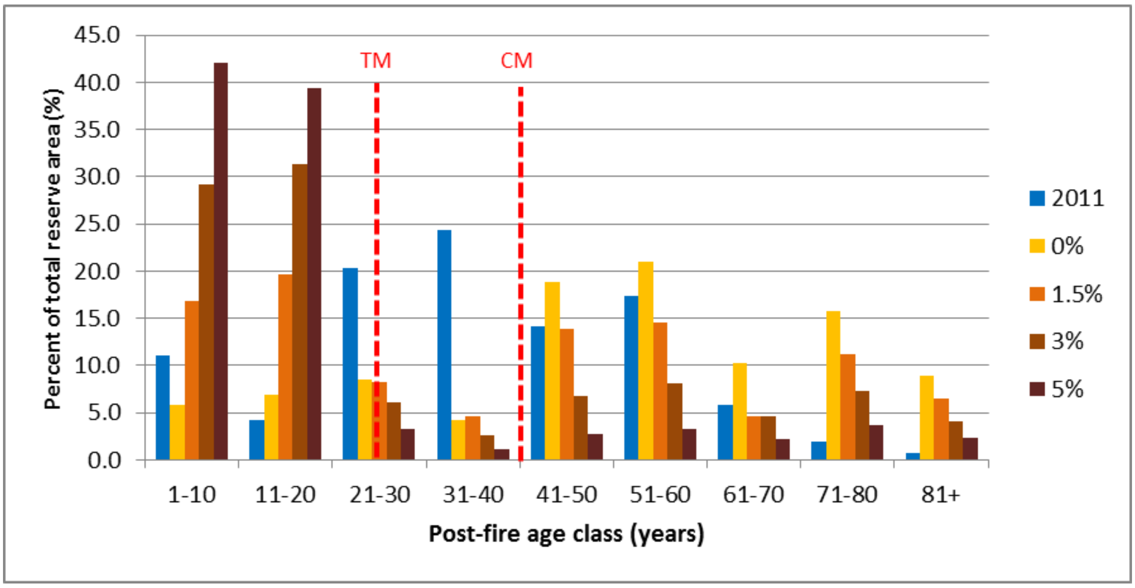


Figure 1. Post-fire age class distribution of reserve in 2032, following 21 years of planned burning (at 0% *p.a.* with bushfires only, and 1.5, 3, and 5% *p.a.* with bushfire). Blue indicates the 2011 distribution. Red dotted lines indicate the minimum tolerable fire interval for *Triodia* (TM, 25 years) and chenopod (CM 40 years) mallee.

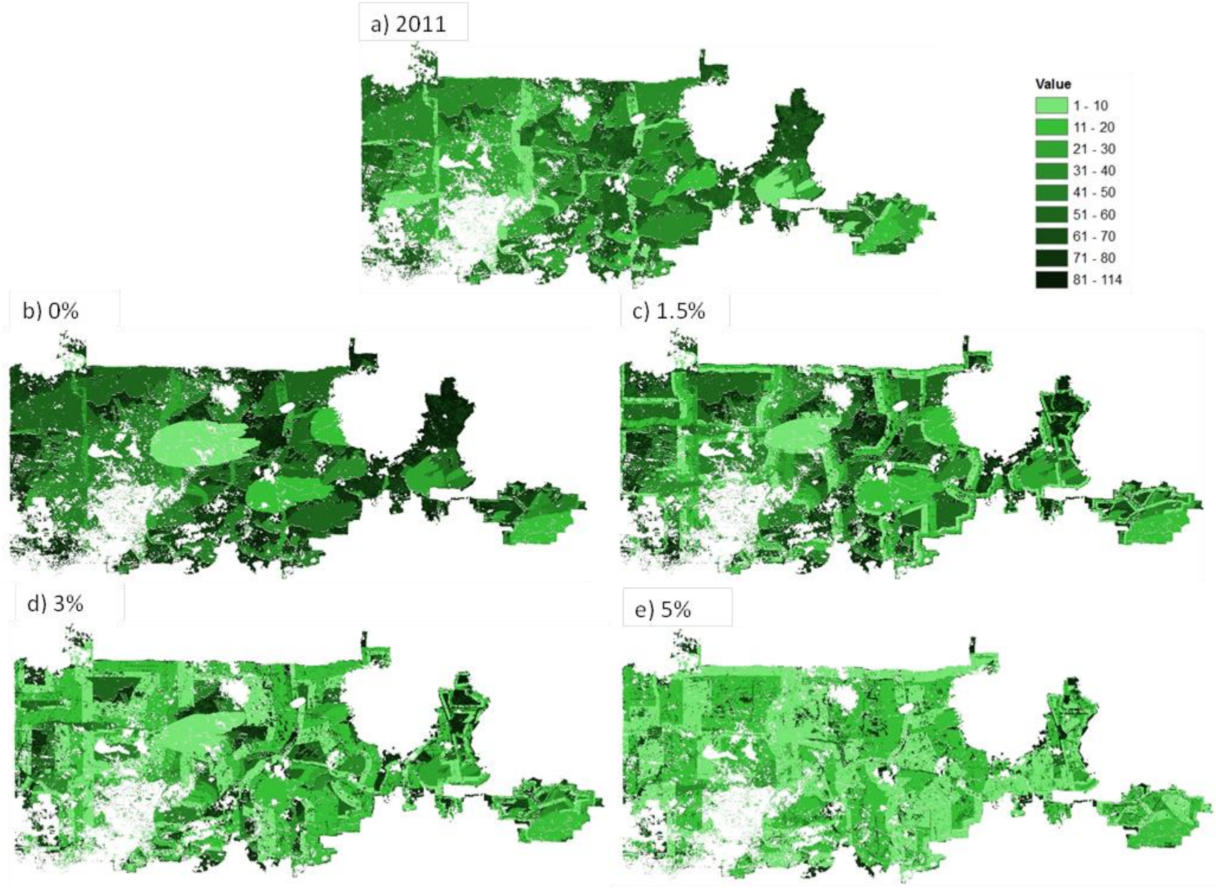


Figure 2. Post-fire age class distribution of Victorian reserve at 2011, and planned burning scenarios (0, 1.5, 3 and 5% *p.a.* plus bushfire) at 2032.

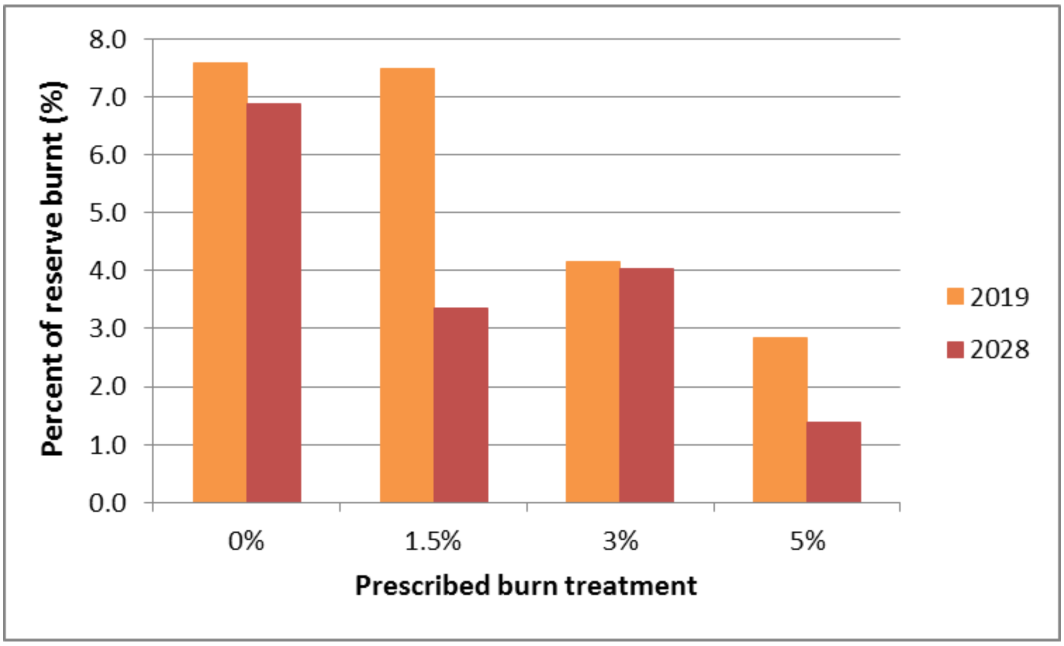


Figure 3. Bushfire complex size across treatments (0, 1.5, 3 and 5% *p.a.*), at 2019 and 2028.

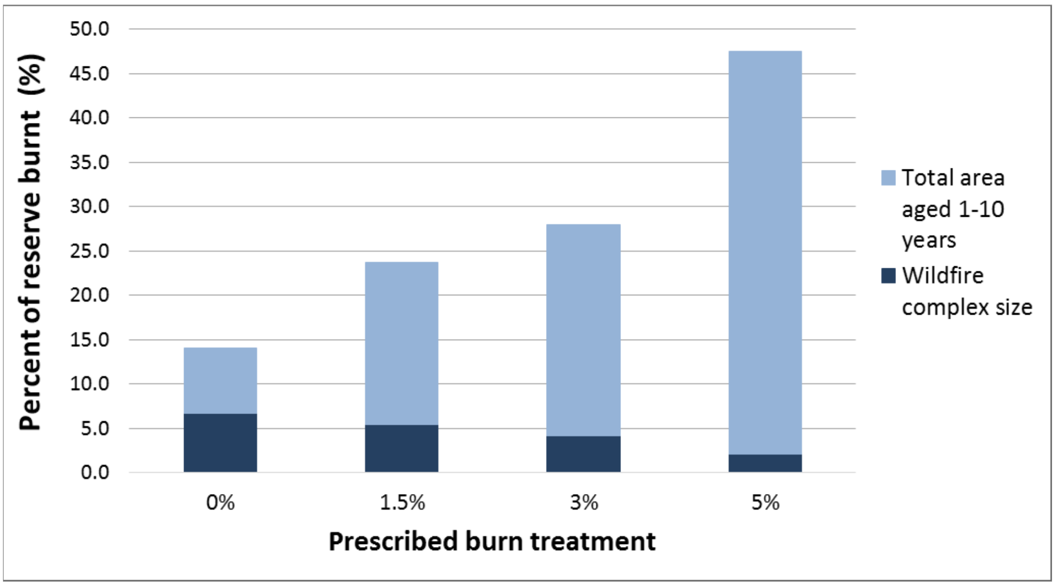


Figure 4. The average contribution of each bushfire complex (2018 and 2028) to the average total area burnt per decade (all vegetation aged 1-10 years, as at 2020 and 2032), per treatment (0, 1.5, 3 and 5% *p.a.*).

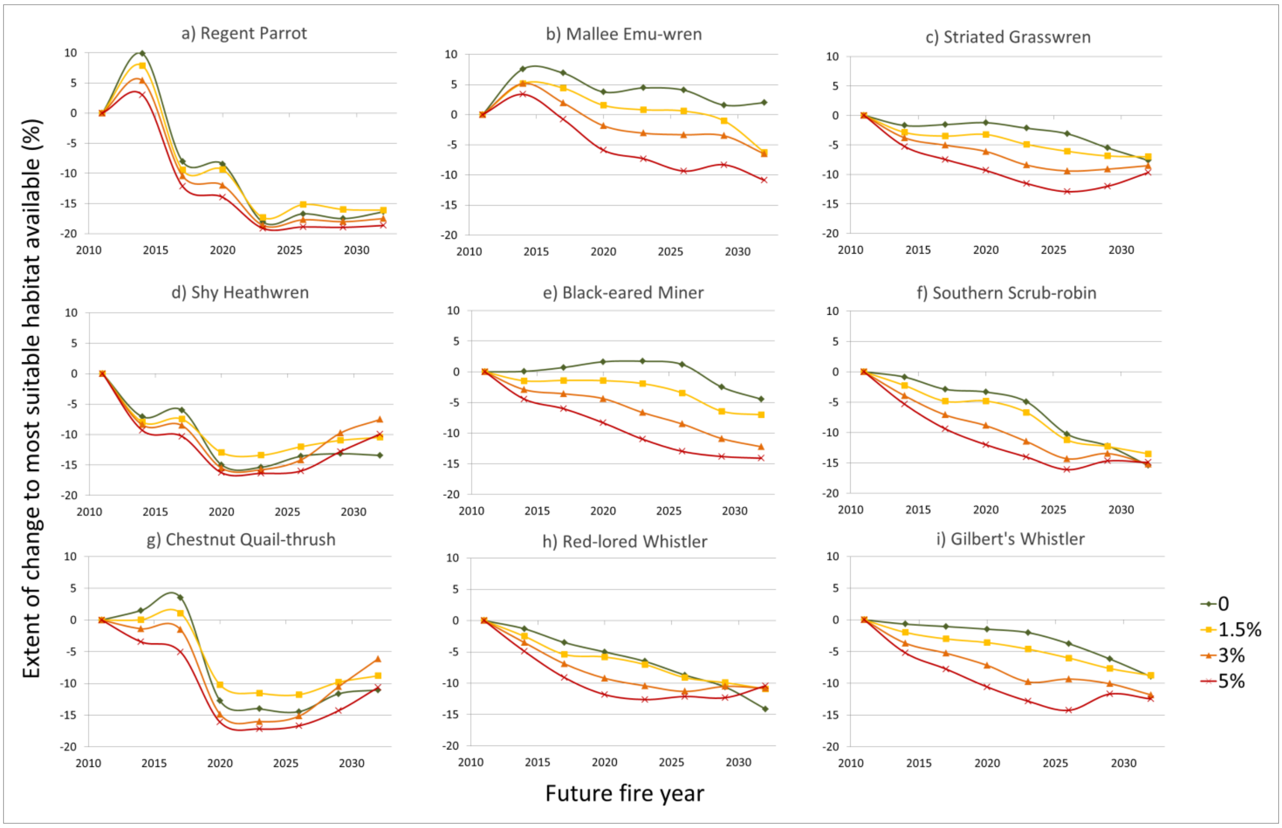
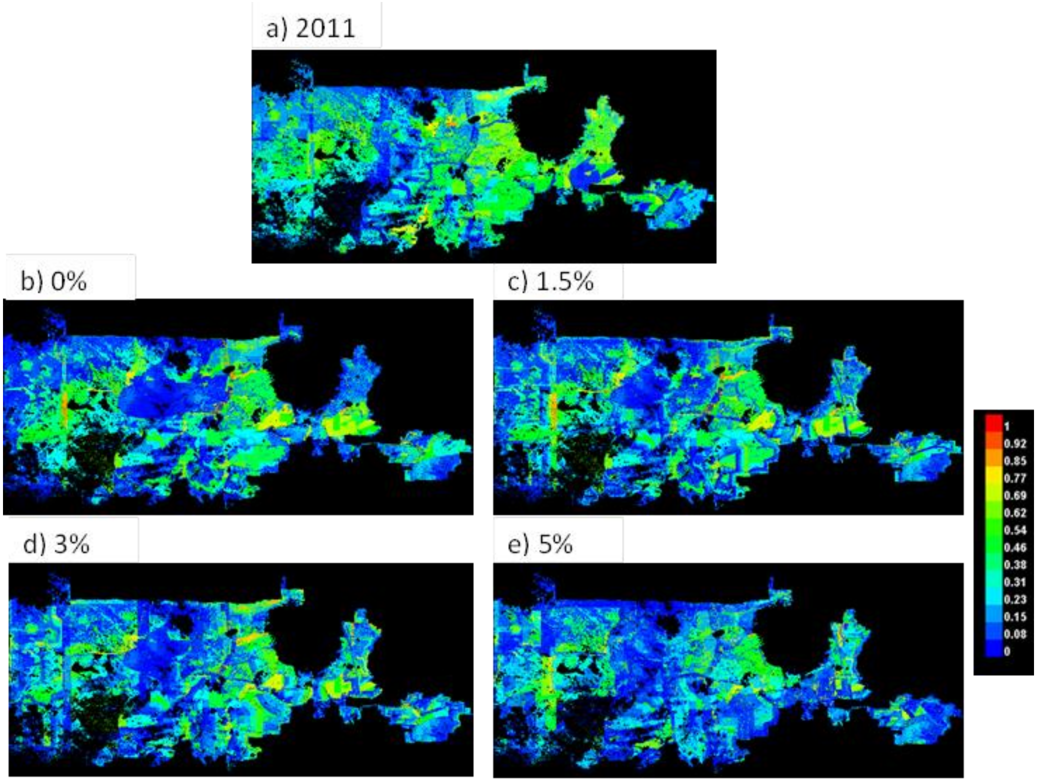
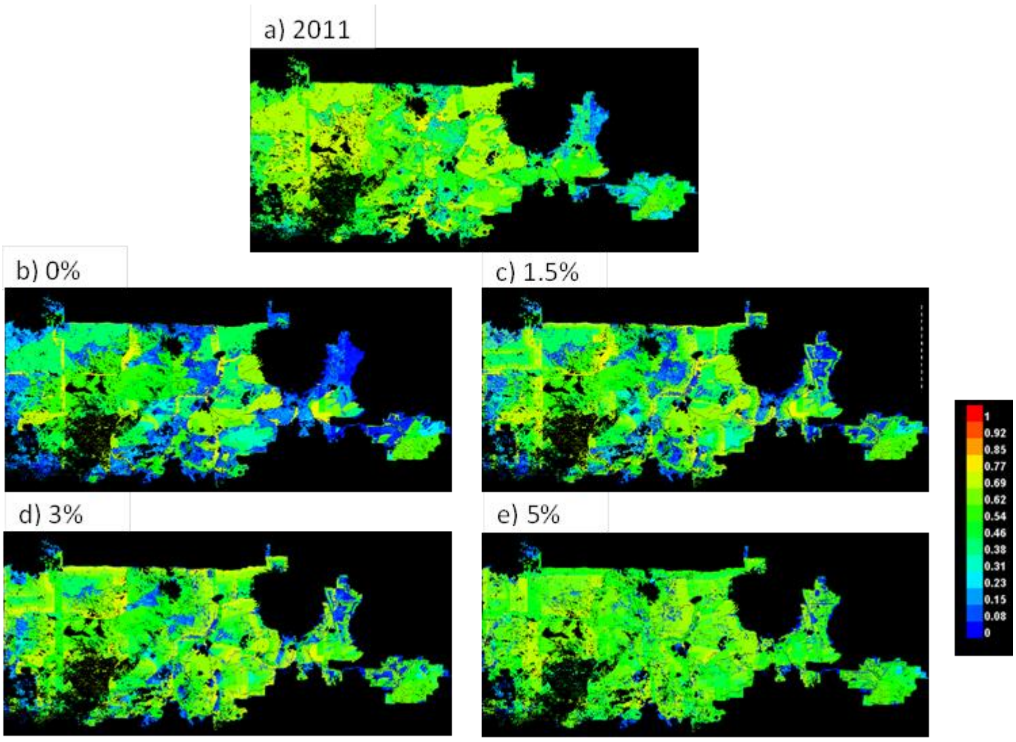


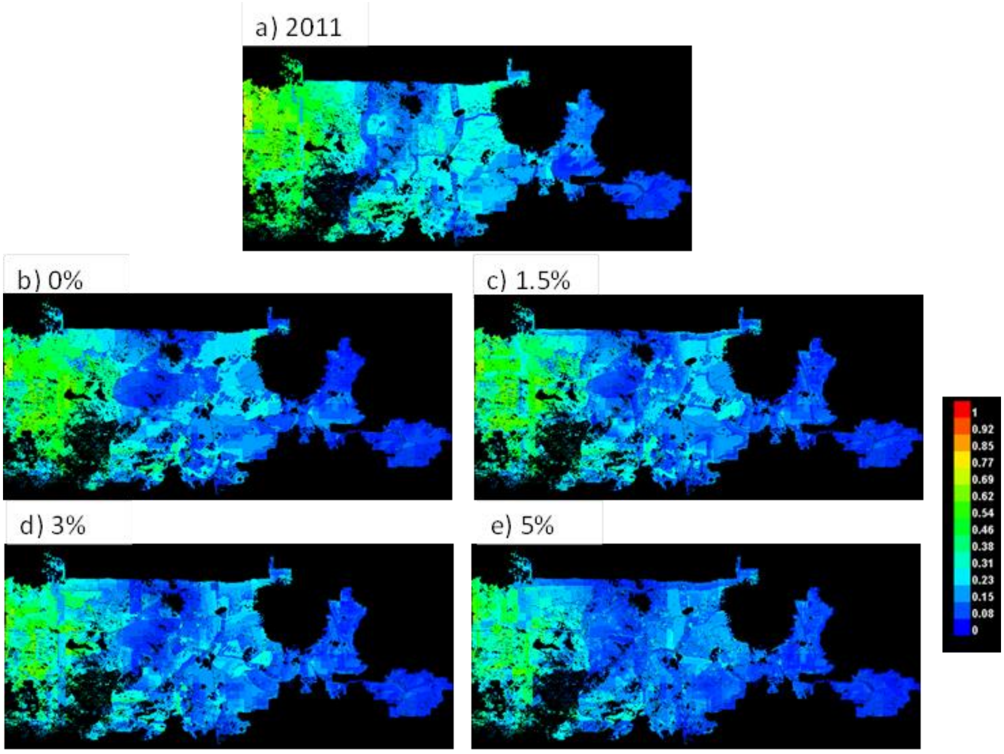
Figure 5. Changes to the extent of 'most suitable habitat' within the reserve over 21 years subjected to 0% (i.e. no planned burning but including bushfires), 1.5%, 3% and 5% *p.a.* planned burning. 'Most suitable habitat' is defined as those areas in which the predicted occurrence rate was estimated to be equal to or greater than the top 20th percentile of the predicted occurrence rates for each particular species in 2011. Changes are expressed relative to the total area of 'most suitable habitat' in 2011 in the reserve.



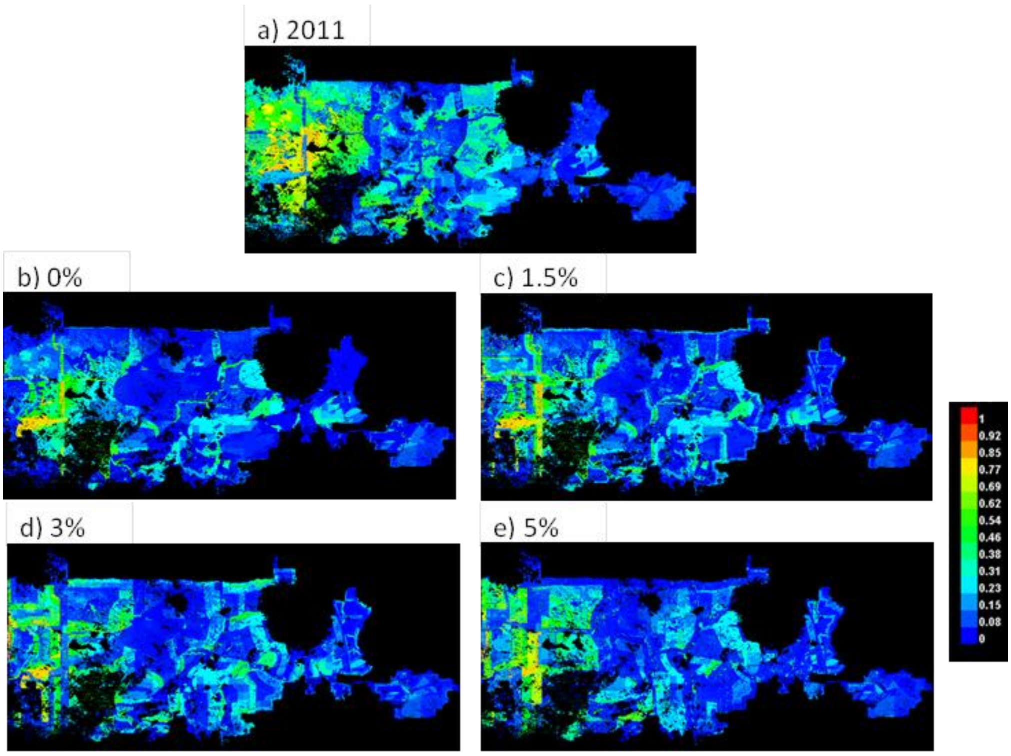
a) Mallee Emu-wren



b) Shy Heathwren



c) Black-eared Miner



d) Red-lored Whistler

Figure 6. Distribution maps at 2011 and 2032 (0, 1.5, 3 and 5% *p.a.*) depicting relative habitat suitability of the reserve for **a)** Mallee Emu-wren, **b)** Shy Heathwren, **c)** Black-eared Miner and **d)** Red-lored Whistler. Pixels of highest relative predicted occurrence (approaching 1) are indicated by red, intermediate (~0.5) green, and lowest (approaching 0) blue.

For remaining species, burning at 5% *p.a.* resulted in less marked differences in the final extent of most suitable habitat by 2032, nonetheless burning at 5% *p.a.* was not found to be most favourable for any species. The Regent Parrot, Striated Grasswren and Gilbert's Whistler each experienced a high reduction in extent of suitable habitat available under 5% *p.a.* relative to remaining treatments, although this difference was minimal.

Notably, for seven species (the Regent Parrot, Striated Grasswren, Shy Heathwren, Southern Scrub-robin, Chestnut Quail-thrush, Red-lored Whistler and Gilbert's Whistler) during the course of two decades of planned burning, the greatest extent of suitable habitat was lost under 5% planned burning *p.a.* (particularly 2023-26), but recovered by 2032 to an extent comparable, equal or higher than other treatment types. For the Striated Grasswren, Chestnut Quail-thrush, Southern Scrub-robin and Gilbert's Whistler these declines were most considerable, differing by up to ten percent relative to extent available under no planned burning in various years.

Mixed responses were shown to the remaining levels of planned burning, with little final distinction seen in the extent of suitable habitat for the Striated Grasswren, Southern Scrub-robin and Red-lored Whistler. Following no planned burning, the extent of suitable habitat showed greatest decline for four species. For the Shy Heathwren and Red-lored Whistler, under no planned burning extent of most suitable habitat was markedly lower than all other treatments, whilst for the Southern Scrub-robin and Chestnut Quail-thrush it was marginal.

Regardless of treatment type, the extent of most suitable habitat declined substantially over the two decades relative to 2011 for most species, by approximately five percent for the Black-eared Miner, to almost twenty percent for the Regent Parrot. Intermediate (21-40 years), late intermediate (41-50 years) and old (51-70 years) age classes were favoured habitat for most of these threatened bird species, and under all treatments vegetation either declined in value as it aged beyond a certain point (0% *p.a.*) or was burnt (1.5, 3 and 5% *p.a.*), resulting in substantial decreases in the percent contribution of intermediate vegetation across scenarios (Figure 1).

Isolated declines and inclines in extent of suitable habitat were observed across species over time, and were identified to the placement of planned burns or bushfire over patches of previously highly suitable habitat, and/or ageing of vegetation that led to in its inclusion or removal from a species' preferred post-fire age class (Figure 6). For example, for the Mallee Emu-wren, all treatments showed a slight increase in extent between 2011 and 2015, as more *Triodia* mallee vegetation entered the preferred age class of 18-28 years for the species. Subsequently, extent decreased across treatments as much of this vegetation was either 'reset' to younger age classes by planned burning, or aged beyond 28 years.

Discussion

We demonstrated the effects of the long-term application of varying levels of planned burning in a mallee reserve to post fire age class distribution, and the associated impact on available habitat for at-risk species. We showed that increased planned burning was associated with substantially higher proportion of very young and young vegetation, a higher loss of intermediate to very old post-fire age classes, and fragmentation of these older age classes. Incorporation of bushfire aimed to provide a more realistic assessment of likely risk in a mallee environment, and simulations provided an indication that a higher extent of planned strip burning was associated with reduced bushfire size. Whilst the age to which strip burns remain effective for mitigation purposes was not examined, land managers have reported this remains >10 years (Kathryn Schneider, *pers. comm.*). However, it has been found in other systems that effective mitigation of bushfire by planned burns was largely negated by severe fuel weather conditions (e.g. Fernandes and Botelho 2003) and thus the extent of bushfire control by scenario strip burns may have been underestimated. Interestingly, scenarios showed that the extent of reserve treated by planned burning, even at 1.5% *p.a.*, far exceeded the area subjected to bushfire when no planned burning was conducted, and these findings are consistent with formal investigations of fire behaviour in other ecosystems (Fernandes and Botelho 2003, King, Cary *et al.* 2006, Boer, Sadler *et al*. 2009, Penman, Bradstock *et al.* 2014). Given the low risk to loss of life and property in this remote region, it would be hoped that planned burning conducted would serve a beneficial ecological purpose to those species most at risk

However, our findings raise the concern that efforts to reduce bushfire risk to (ecological) assets may become counterproductive when the extent of planned burning required is higher than the likely size of bushfires (Bradstock, Bedward *et al.* 1998, Penman, Bradstock *et al.* 2014).

Following two decades of future fire, we showed that burning at 5% *p.a.* was found to have the most pronounced negative impact on available habitat for this suite of threatened and declining species. This finding is unsurprising given that almost all species had higher occurrence in vegetation >20 years, and is consistent with work by Taylor, Watson *et al.* (2013) who identified that, at a landscape scale, no common mallee bird species benefited from a high proportion of young post-fire vegetation. Two mallee endemics, the Mallee Emu-wren and Black-eared Miner, did however show strongest indication of being negatively impacted by higher burning, with greatest reduction in most suitable habitat evident under a 5% *p.a.* scenario. This is likely because both had highly localised distributions at the reserve level in 2011; suitable habitat for the Black-eared Miner was restricted to the far west of the reserve with preference for late intermediate and old post-fire vegetation, and the Mallee Emu-wren was tightly restricted to *Triodia* mallee only, and of intermediate age. These age classes both decreased in extent with increased planned burning. At the other end of the spectrum, no planned burning (but with continued occurrence of bushfire) in the landscape was associated with a declined extent of suitable habitat for a number of species, but most substantially for the Shy Heathwren and Red-lored Whistler. This is likely due to the Shy Heathwren's association with young post-fire ages (0-10 years), and the Red-lored Whistler with intermediate post-fire ages (20-40 years). With no planned burning, little vegetation remained which was of very young and young age (1-20 years), and in the Red-lored Whistler's stronghold in the west of the reserve, there was a declining amount of vegetation in the appropriate intermediate age range (21-40 years).

Higher levels of burning resulted in high fragmentation of suitable habitat for a number of species, particularly those with restricted distributions. As highlighted by our prior work, our ability to assess the value of small and isolated patches is limited until we better understand what patch sizes are required by species to sustain populations, and how readily they are able to move through unsuitable vegetation matrices (e.g. Radford and Bennett 2004, Radford and Bennett 2006). Such understanding of spatial connectivity is crucial to a proper evaluation of the long-term impact of high levels of planned burning.

Conclusions about the likely impact of planned burning cannot be based solely on final total quantity of projected suitable habitat. Whilst suitable habitat showed some recovery in extent by 2032 for a majority of species, projections showed most substantial slumps under 5% *p.a.* This demonstrated that final differences between treatments were not indicative of the full extent of the impact of planned burning on habitat. Recovery was likely seen because vegetation burnt by planned fire in the first years (2011-16) had, by 2032, aged to 16-20 years, and thus entered into a relatively more suitable post-fire age class for species like the Red-lored Whistler and Southern Scrub-robin. Despite this, our models cannot account for the likelihood that already at-risk populations would be sustained through those transitional bottleneck periods in which only less suitable habitat and associated resources would be available. Likewise, there must be connectivity of suitable habitat over time to permit species to disperse to newly emerging suitable habitat within the reserve.

Development of scenarios using both a realistic simulation of the landscape, and using a format consistent with current departmental protocol provided a novel approach with informative output, but was accompanied by a number of limitations. Concerning was the common trend of decline in most suitable habitat over time, regardless of burning treatment. We propose that this decline, however, is unlikely to have proceeded indefinitely. Whilst two decades of planned burning simulations provided a valuable preliminary indication of long term trajectories, anticipated successional cycles occur on a much greater time scale. Scenarios projected over a time frame which allowed for the full cycle of burning for all three treatments (with no repeat or sacrificial burn areas) would likely show a cyclical trend in changing habitat suitability based on post-fire vegetation age. Thus projections beyond approximately seventy years would ideally be developed (*i.e.* at 5% burning *p.a.*, most vegetation in the reserve was treated following twenty years and thus was mostly of twenty years of age or less; following on from this, at 3% *p.a.* all vegetation would be treated following approximately 34 years, and would be 34 years of age or less; and at 1.5% *p.a.* all vegetation would be treated following approximately 67 years, and would be 67 years of age or less).

Likewise, conclusions were restricted by the use of only one set of scenarios, which meant the results were susceptible to the chance placement of planned burns and bushfire occurrences. Use of emerging technology to automate FOP production would enable both much longer term projections to be developed, and a replicated design which would allow us to determine species average response and negate the significant effect of single burn placements. Nonetheless, as our method was specific to current land management practice within the Victorian mallee landscape, was conducted using simulations of the actual landscape, and incorporated a historically realistic assessment of bushfire threat, future projections of planned burning provide a preliminary and valuable indication of long term trends of risk.

Echoing prior conclusions, these results suggest that even within a suite of threatened and fire-sensitive species, it is challenging to identify a unified approach for appropriate fire management (Taylor, Watson *et al.* 2013). Burning at 5% *p.a.* was shown to convert too much vegetation to younger age classes. Given that most threatened bird species showed highest preference for intermediate to old post-fire vegetation classes, it would likely follow that even 1.5% *p.a.* planned burning would be too high to sustain suitably aged habitat for this suite of fire-sensitive species on an ongoing basis. However, simulations showed that an historically representative level of bushfire alone did not convert enough of the landscape to younger vegetation to allow for emerging intermediate age classes for two species. Additionally, the continued occurrence of bushfire in the system leaves highly localised populations of species like the Mallee Emu-wren, Black-eared Miner and Red-lored Whistler vulnerable to unbounded bushfire events if no or very little planned burning were to be conducted. Given that there is a distinction in the degree of vulnerability shown by species to population decline, based on population size, endemism and range restriction, it may be necessary to focus most concerted conservation action towards those most at-risk to fire management, like the Black-eared Miner and Mallee Emu-wren. Continued application of small amounts of planned burning would ideally be targeted to control the placement of emerging suitable habitat with connectivity in mind and, critically, to mitigate potential bushfire spread into areas identified as important for species of high conservation concern (as recommended by Sandell, Tolhurst *et al.* 2006, Brown, Clarke *et al.* 2009, Pastro, Dickman *et al.* 2011).

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Appendix 1

The natural patchiness of planned burns was simulated in ArcMap using the following procedure:

1. Mapped areas of previous planned burns were replicated and merged together to form a single layer that extended well beyond the boundaries of the LMU.
2. Once the pseudo-FOPs had been mapped they were clipped to the cover layer to produce a final area burnt.
3. The layer was created in such a way as to ensure that the coverage produced by clipping to the pseudo-FOP polygons produced burns with 50-80% coverage. This is the target specified for burns identified as Landscape Management Zone. Landscape Management Zone burns are those which aim to achieve bushfire protection by: reducing overall fuel hazard, promoting ecological resilience, and management of the land for particular environmental values (DSE 2012).
4. For each pseudo-FOP, the coverage layer was altered slighted by adjusting its extent and position in order to ensure clips with subsequent burn polygons produced differing patterns and extents of unburnt patches within the burn boundary.
5. The final planned burn polygons used for evaluation of fire impacts on threatened bird habitat were those with the simulated burn coverage applied.