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DEMOGRAPHIC EFFECTS OF SEVERE FIRES IN MONTANE SHRUBS ON TASMANIA'S CENTRAL PLATEAU

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Cover: Clockwise from top left) Burnt transect with resprouting *Orites revoluta*; resprouting *Orites revoluta*; unburnt*Orites acicularis;* burnt transect, mixed species. Source: Judy Foulkes.

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ABSTRACT

Australian montane sclerophyll shrubland vegetation is considered to be resilient to infrequent severe fire but this may not be the case in Tasmanian shrublands. Our research reports on the regeneration response of a Tasmanian non-coniferous woody montane shrubland following a severe fire. The 2019 Great Pine Tier fire in the Central Plateau Conservation Area was a severe crown fire that killed all above ground vegetation in the shrubland. Our field survey revealed that less than 1% of the burnt plants were not top-killed by the fire, and only 5% of the burnt plants were observed to be resprouting one year following the fire. Such a low resprouting rate means the resilience of the shrubland depends on seedling regeneration from aerial and soil seedbanks or colonisation from plants outside the fire ground. The low number of resprouters within the shrubland suggest that it may not be as resilient to fire as mainland Australian montane shrubland. As a consequence, our research highlights the fragility of the shrubland under a warming climate and potential increase in fire frequency.

INTRODUCTION

Shrubland or woody treeless vegetation may not be as economically important as forest vegetation but it is still an important vegetation type. However, shrubs have often been considered as invaders and indicative of ecosystem decline; a nuisance because they are unapalatable to livestock; or low in feed value and therefore not compatible with productive land (McKell 1975, McArthur & Kitchen 2007). But, shrubland provide important ecosystem functions and services such as wildlife habitat, sizeable carbon sinks, prevention of soil erosion and hydrological regulation (McKell 1975, Eldridge & Soliveres 2014, Kröel-Dulay *et al.* 2015). The relative importance given to these two opposing points of view will impact on how particular disturbance events (such as wildfire) are viewed. Wildfires are a natural phenomenon and with a warming climate severe wildfires are expected to become more frequent (Flannigan *et al.* 2013, Fox-Hughes *et al.* 2014). Therefore, if shrubland is considered to be an important component of the landscape, understanding the resilience (capacity to return to its pre-fire state (Holling 1973)) of the shrubland to fire will be critical. Alternatively, if shrubs are viewed as nuisance invaders then fire can be used as an important management tool in their control (Hodgkinson & Harrington 1985, Hodgkinson 1991, Joubert, Smit & Hoffman 2012, Lohmann *et al.* 2014), although removal of encroaching shrubs may not result in an improvement in land condition (Eldridge & Soliveres 2014). Within a montane landscape shrubs can be one of the most flammable components, and under a warming climate shrub encroachment may increase resulting in strong positive feedback effects (Camac *et al.* 2017).

TASMANIAN MONTANE WOODY SHRUBLAND AND FIRE

Tasmanian montane treeless woody shrub vegetation occurs in Tasmania above the climatic treeline from an altitude of 750 m in the south-west to 1400 m in the north-east of the State (Kirkpatrick 1983, Kitchener & Harris 2013). The most prominent taller woody shrub species within the Central Plateau Conservation Area (CPCA) are *Orites revoluta*, *Orites acicularis*, *Richea sprengelioides*, *Ozothamnus hookeri*. Other species that may be present include *Grevillea australis*, *Richea acerosa*, *Leptecophylla juniperina*, *Cyathodes straminea*, *Boronia citriodora*, *Leptospermum rupestre*, *Baeckea gunniana*, *Monotoca empetrifolia* and *Epacris serpyllifolia*.

Tasmanian montane shrubland is considered to have fewer resprouter species than mainland Australia, and therefore may be more fire sensitive than comparable mainland communities (Kirkpatrick & Bridle 2013). With the exception of deciduous heath and coniferous heath communities, many Tasmanian montane species are able to regenerate following widely spaced fires (Kirkpatrick & Dickinson 1984). Historically, wildfires have been very infrequent in alpine Tasmania, but with a warming climate, fire frequency could increase, posing a significant threat to its plant communities.

PLANT RESPONSE STRATEGIES – RESPROUTERS VS OBLIGATE SEEDERS

Certain life-history traits are important to the long-term survival of plant and vegetation communities following disturbances such as fire (Noble & Slatyer

1980, Pausas *et al.* 2004, Clarke, Lawes & Midgley 2010, Clarke *et al.* 2015). In the case of plant populations top-killed by fire, regeneration can occur in three ways: resprouting from dormant buds (apical, epicormic or basal); by seed germination; or colonisation from elsewhere. A trade-off exists between resprouting and investment in seedling regeneration (Bellingham & Sparrow 2000, Bond & Midgley 2001). For instance, juveniles of resprouters must allocate resources to storage tissues whereas non-sprouters can allocate more resources to shoot growth (Pausas *et al.* 2004). Combinations of these traits give rise to five potential post-fire responses to fire (Table 1) (Pausas *et al.* 2004, Pausas & Keeley 2014, Clarke *et al.* 2015, Prior & Bowman 2020). These responses to fire will determine the impact of fire on a vegetation community and are therefore an important influence on the resilience of a system to fire.

TABLE 1 POST FIRE RESPONSES OF PLANTS FOLLOWING FIRE FOLLOWING PRIOR AND BOWMAN (2020).

Trait	Post-fire response
Post-fire obligate seeders	Killed by fire, no ability to resprout, but regenerate from seedlings.
Post-fire obligate resprouters	Have ability to resprout following fire, but do not regenerate from seedlings.
Post-fire facultative resprouter/seeder	Capacity to both resprout and regenerate from seedlings.
Post-fire obligate coloniser	No capacity to resprout or regenerate from seedlings, but can disperse into burnt area from outside.
Fire intolerant	Killed by fire and no capacity regenerate from resprouting, seedlings or colonisation.

FIRE CHARACTERISTICS

Plant responses to fire can be impacted by the fire regime. Fire severity measures the degree to which organic material both above and below ground has been consumed, and indicates the ecological effect of the fire. Within a plant community, it is correlated with fire intensity, which is the amount of energy released during a fire per unit time (Keeley 2009). In the case of large fires, the fire will not necessarily be uniformly severe across the entire fire ground, with patches of less severe and unburnt vegetation occurring (Bradstock 2008). Fire severity can affect the success or otherwise of plant regeneration strategies with some species requiring fire to release seeds (Lamont *et al.* 1991), although low intensity fires may not trigger fire-cued seed germination (Clarke *et al.* 2015). In some ecosystems resprouting and seedling recruitment are minimally impacted by variation in fire severity (Knox & Clarke 2004, Camac *et al.* 2013, French *et al.* 2016), whereas other ecosystems may tolerate mild surface fires but take centuries to recover from high severity crown fire (Kirkpatrick, Bridle & Dickinson 2010, Bowman *et al.* 2019). Maximum rates of resprouting may occur at medium fire severity (Nicholson *et al.* 2017) or high fire severity (Rodriguez-Cubillo *et al.*. 2020). Fire severity can also impact on seedling recruitment with higher seedling density occurring in higher fire severity sites (Vivian *et al.* 2008). Following a severe fire, post-fire obligate seeders require time for seed to germinate, grow and mature and produce the next batch of seed. If fire intervals are reduced to such an extent that this cycle does not have time to complete before a subsequent fire, these species face a significant extirpation risk (Enright *et al.* 2015).

AIMS OF THIS STUDY

In February 2019, a wildfire resulted in some areas of the eastern CPCA being severely burnt with all above ground biomass being destroyed. This provided an excellent opportunity to increase knowledge of the resilience to fire of the CPCA shrubland. Our field research focused on the resprouting response of woody shrubs to fire and in particular, whether an increase in fire severity results in a decrease in resprouting. Second, we wanted to measure postfire seedling recruitment, and determine whether an increase in fire severity decreases the density of overall seedling recruitment, one year post-fire. It is an opportune time to consider the on-going management requirements of the Central Plateau shrubland. Historical records suggest that 'fire-stick' farming practices were adopted to maintain grassy vegetation for animal feed (Jackson 1973, Johnson & Marsden-Smedley 2002), and there is anecdotal evidence that the taller woody shrubs have increased in density following the cessation of this practice (Harold Riley, personal communication). If the Central Plateau shrublands are in fact invaded grassland, then follow-up management burning may be warranted to control the shrubs and allow the grasslands to re-establish.

BACKGROUND

STUDY REGION

Our study area was the shrublands near Lake Augusta, to the west of the Great Lake, on Tasmania's Central Plateau (Fig. 1). The study area is at approximately 1100 m elevation. The climate of the Central Plateau region is perhumid cold to humid cold, with heavy rainfall in the Western Plateau region becoming lighter towards the east (Gentilli 1972). Snow falls frequently during the winter months but rarely lies on the ground for extended periods (Gentilli 1972, Kirkpatrick 1983). At Liawenee, the nearest weather station to Lake Augusta, mean monthly maximum temperatures range from 5.6 °C in July to 19.1 °C in January, and the mean minimum monthly temperature ranges from -1.6 °C in July to 5.5 °C in January, but minimum temperatures below zero can occur during any month of the year (Bureau of Meteorology 2020). The mean annual precipitation is 923 mm, with the wettest months occurring during winter. In the twelve months before the fire, precipitation was 984 mm, but the month immediately preceding the fire was very dry, with only 4 mm rain, compared with an average for January of 50 mm. The twelve month period following the 2019 fire was slightly drier than average, with a total 771 mm precipitation.

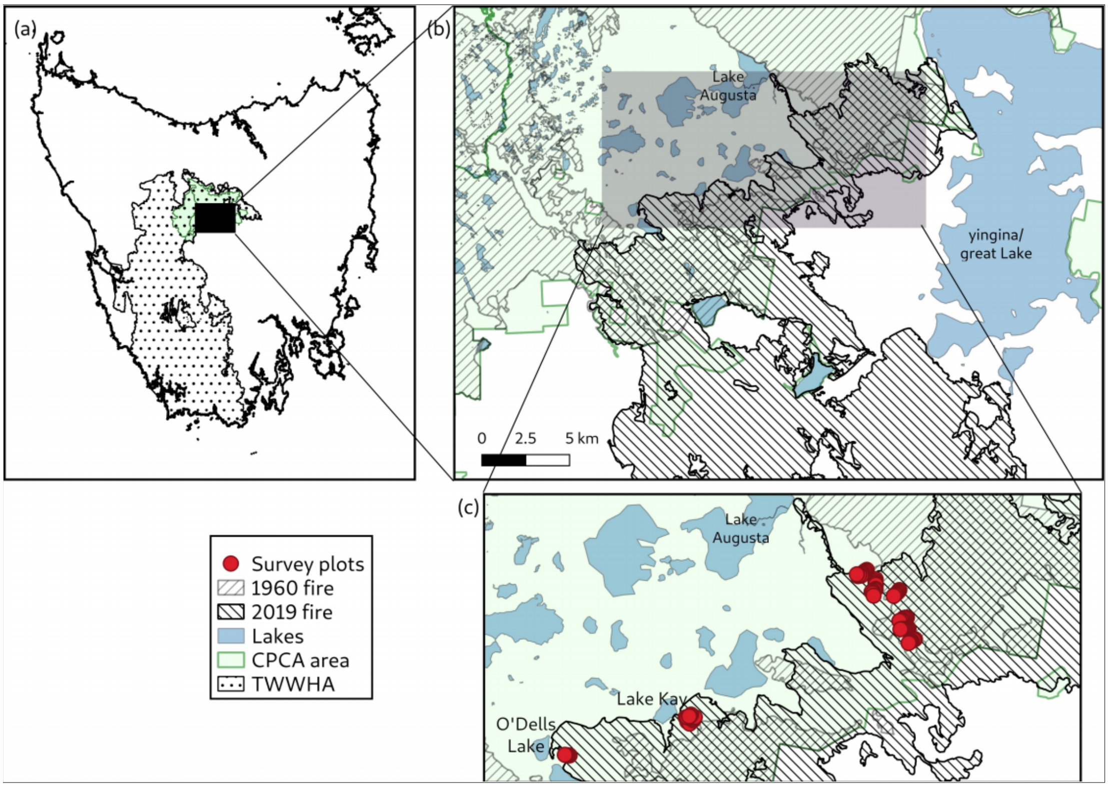

FIG. 1 (A) MAP OF TASMANIA SHOWING LOCATION OF THE CENTRAL PLATEAU AND TASMANIAN WILDERNESS WORLD HERITAGE AREA (B) THE SURVEY AREA IN THE CENTRAL PLATEAU CONSERVATION AREA (CPCA), SHOWING THE AREA BURNT BY THE 1960/61 AND THE 2019 FIRES. (C) LOCATION OF FIELD SURVEY PLOTS.

FIRE HISTORY

On 15 January 2019 dry lightning ignited several small fires which combined to form the Great Pine Tier fire. Over a period of 3 weeks this fire burnt 51,224 ha, with the CPCA shrubland burnt on 3 and 4 February 2019 (AFAC 2019) (Fig. 1(b) & Fig. 2).

Prior to 2019, the last time the CPCA shrubland had been burnt was in 1960/61, purportedly by highland graziers to promote green-pick for stock grazing (Johnson & Marsden-Smedley 2002). The fire history prior to the 1960/61 fire is unclear. Although the region is likely to have experienced many small fires from burning grasslands for stock grazing, there are no reports of wide-scale fires occurring in the vicinity of the study region prior to the 1960/61 fires (Johnson & Marsden-Smedley 2002).

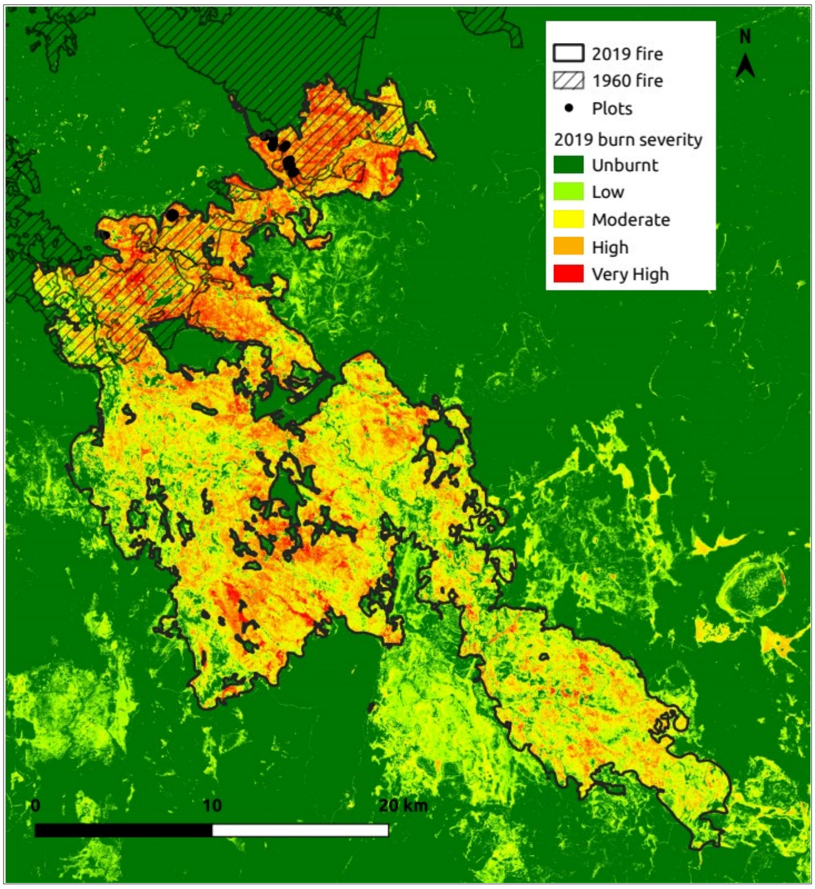

FIG. 2 2019 PINE TIER FIRE BURN SEVERITY AS SHOWN BY THE DIFFERENCED NORMALISED BURN RATION (DNBR) FOR PRE AND POST FIRE SENTINEL 2 IMAGERY. IMAGE DATES PRE-FIRE: 31/12/2018 & 13/1/2019; POST-FIRE: 1/3/2019.

SHRUBLANDS

Alpine and subalpine or montane regions (areas that are above the climatic treeline) comprise approximately 10% of the total land area of Tasmania. The Central Plateau is the largest of these regions, with the CPCA occupying 921 km^2 (Costin 1973). The Central Plateau is extensively glaciated and is an important water catchment area for hydroelectricity and irrigated agriculture (Misiak 2007). The geology of the region is predominantly Jurassic dolerite (Banks 1973). The CPCA is part of the Tasmanian Wilderness World Heritage Area (TWWHA) and is important for its scleromorphic montane vegetation, in contrast to many other alpine and subalpine areas that are dominated by grasses and herbs (Balmer *et al.* 2004). The TWWHA contains approximately 37,700 ha of montane heath/shrubland vegetation, which represents approximately 58% of the total for Tasmania (Balmer *et al.* 2004).

Prior to European settlement, Aborigines from the Big River tribe occupied the region during the warmer months and historical records suggest that they used fire to promote grassy areas to encourage wallaby grazing (Shepherd 1973, Misiak 2007). At that time, the region was thought to be predominantly a shrub/grass mosaic, with bolster plants, bog and heath in poorly drained depressions and woodland on the more protected slopes (Jackson 1973). The introduction of sheep and cattle grazing during the warmer months (December to May) together with regular burning (to encourage grass growth) severely degraded the landscape and resulted in much bare ground and erosion (Jackson 1973, Shepherd 1973, Johnson & Marsden-Smedley 2002, Misiak 2007). Following the declaration of the Tasmanian Wilderness World Heritage area and exclusion of stock grazing from the CPCA, vegetation cover has been slowly increasing (Bridle & Kirkpatrick 1999, Bridle *et al.* 2001, Kirkpatrick & Bridle 2013). Anecdotal evidence also suggests that with the cessation of regular burning, woody shrubs have invaded native grasslands (Harold Riley, personal communication).

METHODS

FIELD SURVEY

The field survey took place during February and March 2020, 12 months after the 2019 Pine Tier fire. Eleven traverses were made of the burnt shrubland with measurements made within variable length belt plots established at approximately 50 m intervals along each traverse (Fig. 1). In total, 55 plots were established in the burnt area. In addition, we established 5 plots in nearby unburnt shrubland.

All plots were placed at least 50 m from the Lake Augusta Road. Each plot was at a right angle to the main traverse and incorporated up to 20 woody shrubs. GPS coordinates of the start of each plot were recorded using the Handy GPS Application, Version 34.4, operating on an android smart phone (www.binaryearth.net). Initially all woody stems were recorded, but for the later plots, only those plants with at least one stem with a basal diameter ≥ 2 cm were recorded. Within each plot, all woody shrubs were identified to species level where possible. Where a group of stems were growing close together, they were considered to be the same plant if they appeared to be connected underground. For each plant we recorded whether it was burnt or not, condition (alive, dead or resprouting), number of stems and width and length of the lignotuber/basal area. As a proxy for fire severity we measured the diameter (to the nearest mm) of the smallest burnt twig. One measurement was recorded for each plant within the plot. Larger minimum burnt twig measurements indicate a more severe fire (Moreno & Oechel 1989, Whight & Bradstock 1999). Minimum twig measurement was recorded as '0' for all plants that had an unburnt canopy, including the occasional unburnt plant within the burnt plots. For those plants resprouting, the number of resprouting shoots, and height of tallest resprout was recorded. For plants within the unburnt plots, the presence or absence of a canopy seedbank was noted.

Three 50 x 50 cm quadrats were established at the beginning, centre and end of each belt plot. We counted the number of seedlings present in each quadrat. We were not able to identify species due to the very small size of the seedlings. Within each quadrat, the total number of scats and type of animal (macropod, wombat, rabbit, deer) were also recorded. Plots were characterised by visually estimating the percentage of rock, forbs, grasses and bare soil for a 1 m^2 quadrat located at the centre of the belt plot. Soil condition on a scale of 1 (burnt but soil surface undamaged with an intact thatch, litter or moss layer) to 5 (cavities burnt into soil with distinct scarps) was also noted for each plot. Due to the very open and flat terrain of the study site, we did not record slope or aspect.

DATA ANALYSIS

All data exploration and statistical analysis was undertaken using R statistical software (version 3.6.3) (R Core Team 2020).

Size class distributions

In the burnt plots, most stems were largely consumed by the fire. Therefore, we used the shrub basal area (calculated as length x width occupied by the shrub stems at ground level) and number of stems per plant as indicators of plant size, which would be loosely correlated with shrub age. We compared the basal area and the number of stems per plant between the plots that had been burnt in the 1960 fire and those that had not.

Resprouting in *Orites revoluta*

The resprouting response (whether dead or resprouting) of burnt *O. revoluta* shrubs was analysed using a binomial generalised linear model (glm) with resprouting as a binary response variable and fire severity (represented by burnt twig diameter at the plant level) and basal area as explanatory variables.

Seedlings

Because seedling data was recorded at the plot level, we calculated the mean burnt twig measurement for all burnt plants within the plot as a proxy for fire severity at the plot level.

We investigated the relationship between seedling density and fire severity by comparing linear Poisson, zero-inflated Poisson and negative binomial glms using average minimum twig diameter per plot as a continuous predictor variable. We also tested whether there was a humped response to fire severity by comparing linear models with those that also included a quadratic (twig diameter squared) term. Vuong tests and AIC were used to select the best model.

RESULTS

SITE CONDITIONS

The percentage of bare ground, rock and vegetation cover varied considerably between the plots. Bare ground ranged from 0% to 80% with a mean of 28%, and rock cover ranged from 0% to 90% with a mean of 18%. Forbs and grass cover ranged from 0% to 80% and 70% respectively (Fig. 3).

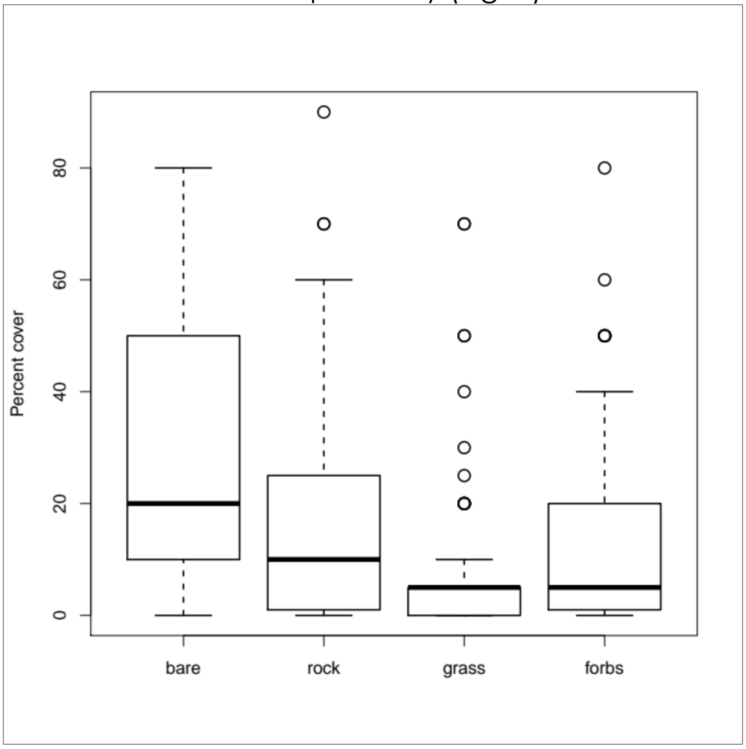

FIG. 3 VARIATION IN SITE CONDITIONS ACROSS ALL SURVEYED PLOTS. THE THICK HORIZONTAL BAR IS THE MEDIAN, UPPER AND LOWER SIDES OF THE BOX ARE THE 25% AND 75% PERCENTILES. WHISKERS ARE 1.5 X THE INTERQUARTILE DISTANCE AND CIRCLES ARE OUTLIERS.

PLANT RESPONSE

Of the 1036 woody shrubs surveyed in total, 92% (952) had been burnt, with more than 99% (946) of the burnt plants top-killed (Table 2). By contrast, most plants (94%) in the unburnt plots were alive (Table 2). Within the burnt plots, the level of resprouting was low with only 5% (43) of the burnt shrubs observed to be resprouting from basal buds. Of the species observed to resprout (90 plants), 48% (43) were resprouting, and 93% (40) of the resprouters were *O. revoluta* (Table 2). The other resprouting species were *Leptospermum rupestre* (2 resprouters representing 17% of burnt *L. rupestre*) (Fig. 4(h)) and *Bauera rubioides* (1, 100%) (Fig. 4(g)). Resprouting *Olearia myrsinoides* were observed at the end of the survey period when Covid restrictions were introduced. Time prevented a survey being undertaken of the population, therefore their presence was noted but not quantified or included in the results.

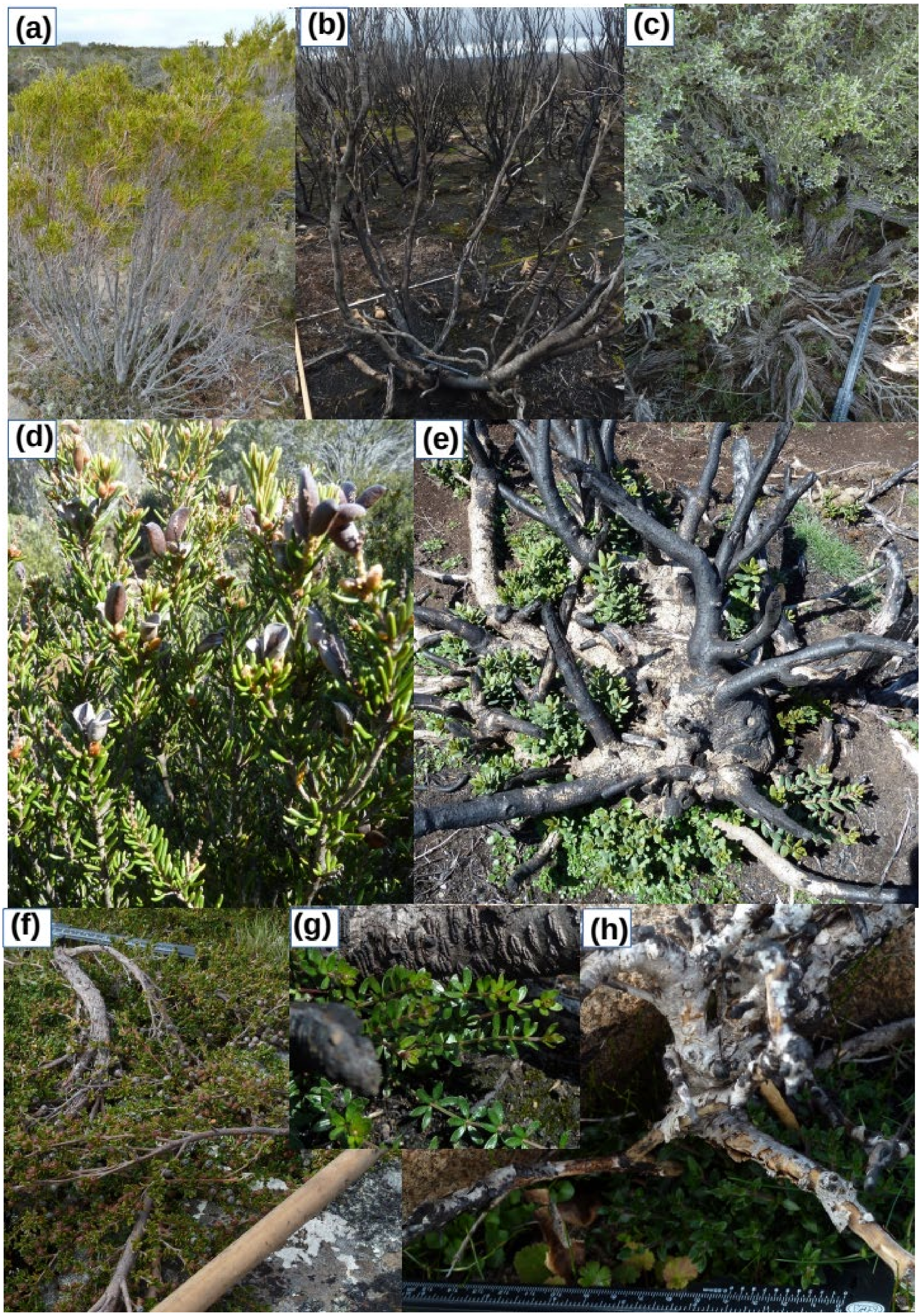

FIG 4 OBSERVATIONS FROM THE FIELD SURVEY. *ORITES ACICULARIS* (A) ALIVE AND (B) BURNT. *OZOTHAMNUS HOOKERI* (C). *ORITES REVOLUTA* (D) ALIVE WITH OPEN AND CLOSED SEED FOLLLICLES PRESENT, (E) BURNT SHOWING RESPROUTING FROM LARGE BASAL LIGNOTUBER. *LEPTOSPERMUM RUPESTRE* (F) ALIVE WITH SEED CAPSULES PRESENT, (H) BURNT WITH RESPROUTING SHOOTS. *BAUERA RUBIOIDES* (G) RESPROUTING.

TABLE 2 NUMBER OF SHRUBS IN THE UNBURNT AND BURNT PLOTS, CONDITION OF SHRUBS IN THE BURNT PLOTS AND MEAN BASAL AREA OF SHRUBS IN THE UNBURNT AND BURNT PLOTS

Species	Unburnt	Burnt	Condition of burnt shrubs			Mean basal area (cm^2) ± s.d.	
			alive	dead	Resp.	unburnt	burnt
Acrothamnus montanus	0	1	1				250
Bauera rubioides Fig.4(g)	0	1			1		135
Leptospermum rupestre Fig.4(f) & (h)	0	15	1	12	2		169 ± 256
Orites acicularis Fig.4(a) & (b)	14	40		40		303 ± 394	225 ± 216
Orites revoluta Fig.4(d) & (e)	9	77	2	35	40	223 ± 294	528 ± 1130
Ozothamnus hookeri Fig.4(c)	30	13	0	13		141 ± 223	108 ± 101
Richea acerosa	24	54	9	45		224 ± 306	178 ± 228
Richea sprengeloides	0	43		43			65 ± 72
unknown	0	715		715			72 ± 97
Total	**77**	**959**	**8**	**901**	**43**		

Note: Only 5 of the unburnt shrubs were dead, and none was resprouting.

STAND STRUCTURE OF *ORITES REVOLUTA* IN RELATION TO THE 1960 FIRES

Orites revoluta was the only species recorded in sufficient numbers to investigate the legacy effects on stand structure of the 1960 fires. Although the differences were only marginally significant, the data suggests that plots not burnt in 1960 contained larger shrubs, with a higher basal area, than those burnt in that year (Fig. 5).

RESPROUTING *ORITES REVOLUTA*

Of the 75 *O. revoluta* plants that were burnt, 53% (40) were resprouting (Table 2). There was no significant relationship between fire severity and resprouting of *O. revoluta* (Fig. 5), although this may be due to an insufficient sample size at the higher fire severity. There was a significant positive relationship ($p < 0.01$) between the proportion of *O. revoluta* plants resprouting and basal area (Fig. 5 (b) & (d)). The median number of resprout shoots per resprouting plant was 21, with three plants having at least 100 resprout shoots. The maximum height of the tallest resprout shoot ranged from 3 cm to 90 cm with a median of 14.0 cm.

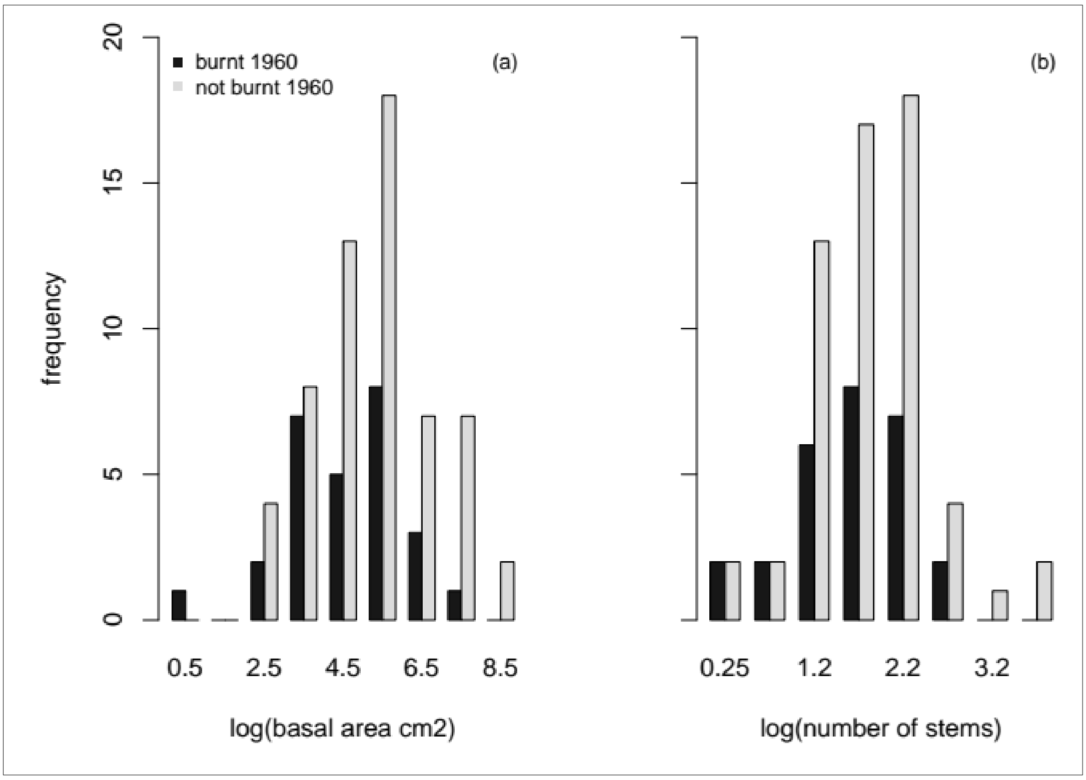

FIG. 5 STAND STRUCTURE COMPARISON OF *ORITES REVOLUTA* BETWEEN PLOTS THAT HAD BEEN BURNT OR UNBURNT DURING THE 1960 FIRES. (A) BASAL AREA (LOG_{10} SCALE) (MARGINALLY SIGNIFICANT, P=0.067) AND (B) NUMBER OF STEMS (LOG_{10} SCALE) (NOT SIGNIFICANT).

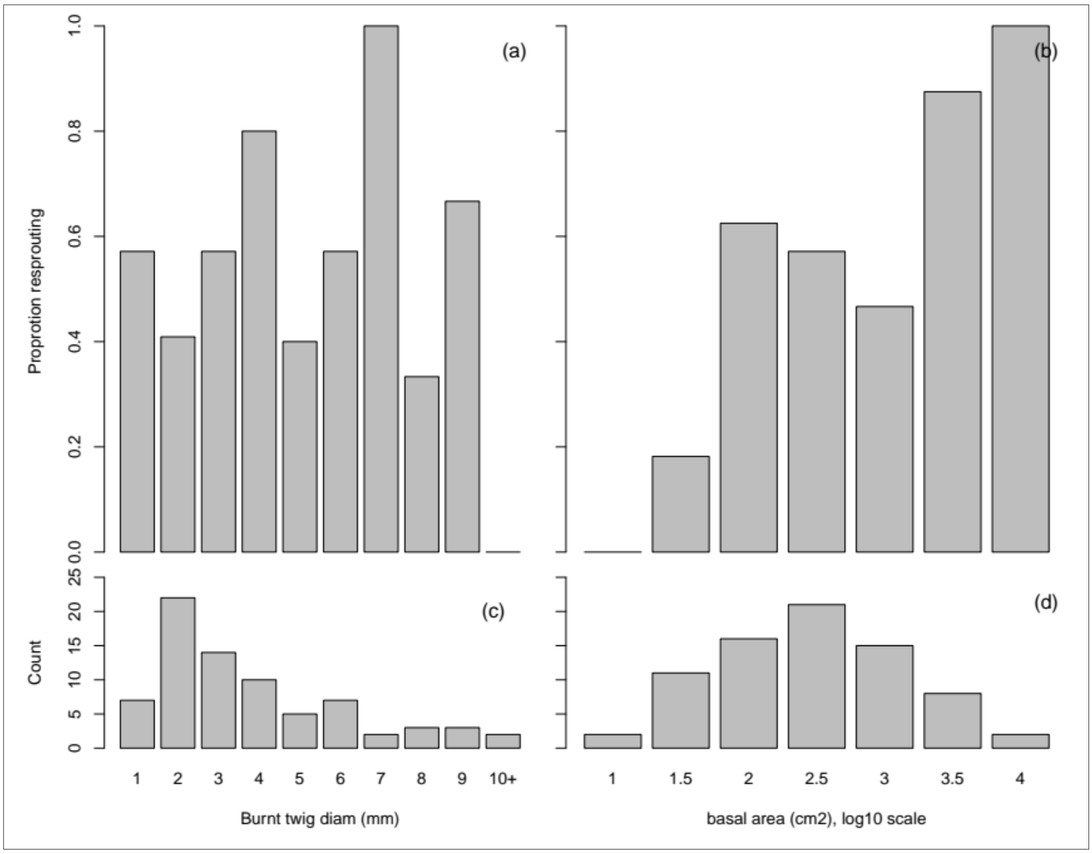

FIG. 6 PROPORTION OF *ORITES REVOLUTA* RESPROUTING IN RELATION TO BURNT TWIG DIAMETER, AND BASAL AREA. (A) & (B) PROPORTION OF PLANTS RESPROUTING; (C) NUMBER OF PLANTS PER BURNT TWIG DIAMETER; (D) NUMBER OF PLANTS PER BASAL AREA. NOTE: SCALE FOR BASAL AREA IS SHOWN IN LOG_{10}.

SEEDLINGS RECRUITMENT

Seedlings were present in all unburnt plots and most unburnt plants had a canopy seedbank at the time of the survey. Seedlings were absent in nine of the burnt plots (15%), and most of these were at lower than average fire severity as measured by the mean minimum burnt twig diameter of all plants within the plot (Fig. 7). Across all plots, seedling density was very variable ranging up to a maximum density of 242 seedlings/m^2,with a mean density of 25 seedlings/m^2. The best model between seedling density and fire severity (as measured by mean minimum burnt twig diameter per plot) using the AIC statistic was the negative binomial quadratic (AIC = 465.2) followed by zero-inflated poisson quadratic (AIC = 1255.0) and poisson quadratic regression (AIC = 1680.4). This shows a humped relationship with higher seedling density occurring under intermediate fire severity conditions (Fig. 7).

SCATS

Scats (macropod, rabbit, wombat and deer) were present in most plots (n=49; 82%). Most scats were macropod, with feral deer and rabbit scats recorded in a minority of plots (9 plots and 1 plot respectively) (Table 3).

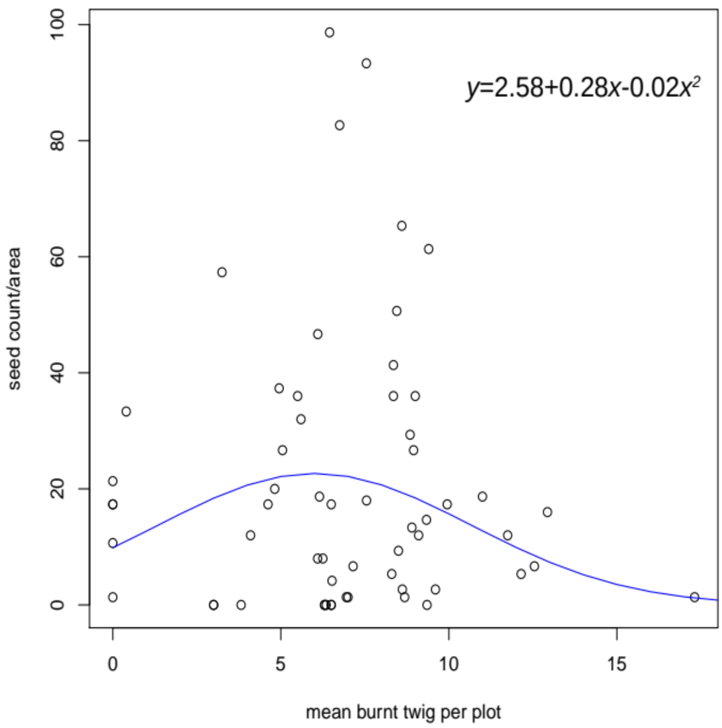

FIG. 7 SEEDLING REGENERATION RESPONSE AS A FUNCTION OF MEAN BURNT TWIG DIAMETER FOR EACH PLOT. NOTE, ONE OUTLIER DATA POINT REMOVED FOR ILLUSTRATION PURPOSES.

TABLE 3 DETAILS OF SCAT PRESENCE WITHIN SURVEY PLOTS

Animal	Number of plots	Number of scats
Macropod	44	322
Wombat	3	16
Deer	9	18
Rabbit	1	21
Total		**377**

DISCUSSION

RESPROUTING RESPONSE

The Central Plateau shrubland is a crown fire system comprised largely of obligate seeders. Overall resprouting response was very low with only 3 species observed to be resprouting, and an overall 5% of all burnt plants resprouting. Such a low resprouting rate is similar to that found by Nicholson *et al.* (2017) following a severe fire on Clarke Island off the north east coast of Tasmania. However, in the Nicholson *et al.* (2017) study, approximately 60% of species encountered had at least one plant resprouting compared to only 25% of species on the Central Plateau. French *et al.* (2016) also observed a low resprouting rate in south-west Tasmania. Again, their study included many more species than the present study. The 40% of species resprouting in the current study was less than observed in subapline shrubland in Kosciuszko National Park, New South Wales following the 2003 fires (Walsh & McDougall 2004).

O. revoluta did not display the apparent humped relationship between resprouting rates and fire severity reported by Nicholson *et al.* (2017), but we encountered too few individuals that experienced high severity fire to be confident that resprouting rates were maintained at these severities. It is also possible that the shape of the relationship between resprouting and fire severity differs among species, or alternatively other factors (such as climate or soil) influence the resprouting success on the Central Plateau. There was a higher proportion of *O. revoluta* individuals resprouting than observed overall by Nicholson *et al.* (2017) in the resprouting species on Clarke Island. This highlights the variability among species in resprouting response (Vesk & Westoby 2004).

Interestingly, we found no instances of *O. hookeri* resprouting. *O. hookeri*, is reported to be an early successional resprouter that produces large quantities of wind-dispersed seed (Kirkpatrick, Bridle & Wild 2002). One possibility for the lack of resprouting is that the 2019 fire was too severe and the species only resprouts under less severe fire conditions. Additionally, pre-fire drought combined with a high severity fire may significantly reduce resprouting success (Twidwell *et al.* 2016). The rainfall records for Liawenee do not show an unusually prolonged dry period immediately prior to the 2019 fire, therefore, short-term drought stress is unlikely to explain the lack of resprouting, but rainfall was very much below average during the millennium drought (1997 to 2009) (Bureau of Meteorology 2020). Kirkpatrick, Bridle & Wild (2002) also reported that on Mt Wellington (near Hobart Tasmania, maximum elevation 1270 m) *O. hookeri* does not establish in long unburned vegetation. There were insufficient *O. hookeri* measured in our 1960 burnt plots to determine if this is the case within the CPCA. Further study is warranted to better understand the response of *O. hookeri* to fire.

SIZE CLASS DISTRIBUTION

The data suggest that *O. revoluta* shrubs in areas burnt in 1960 were smaller overall than those in unburnt areas, but the effect was statistically marginal. From our observations, it is likely that the 1960 fires top-killed many shrubs, and it would take years for basally resprouting shrubs to regain their former size, given the slow growth rates in this harsh environment. With a median resprout height of 14 cm attained 12 months following the fire, it will be at least 8 years before those plants will reach a similar height to the unburnt plants measured.

SEEDLING RESPONSE

Given the weak resprouting response following the 2019 fire, recovery of the pre-fire shrubland will depend on regeneration from seed. Due to time constraints for this study, and the small size of seedlings present, no attempt was made to identify seedlings, therefore, additional field work is clearly warranted in order to gain an understanding of the post-fire seedling response of the shrubland. The main germination period for seedlings is autumn, however the harsh winter environment, particularly periods of frost heave, result in substantial seedling losses (Misiak 2007). These harsh winter conditions mean that on the Central Plateau seedlings tend to do best in sheltered microsites, such as under existing vegetation cover (Misiak 2007, McPhail & Kirkpatrick 2016). Along with seedling losses due to weather conditions, the other main threat to seedling survival is herbivory (Bridle & Kirkpatrick 1999, Bridle *et al.* 2001, Kirkpatrick & Bridle 2016, Kirkpatrick *et al.* 2016). Fire can result in increased macropod densities (Styger *et al.* 2011). In addition to native hervbivores, other herbivory pressures come from rabbits (Shepherd 1973, Cullen 1995, Bridle *et al.* 2001), and fallow deer (Locke 2007). We found minimal evidence of rabbit presence and only minor evidence of deer presence, but macropod scats were present in most survey plots, suggesting that the dominant herbivory pressure post-fire comes from native herbivores, particularly macropods.

RESILIENCE OF THE SHRUBLAND

The low percentage of species that we found resprouting supports the finding of Kirkpatrick, Bridle & Dickinson (2010) that Tasmanian montane flora may be less resilient to infrequent severe fire than mainland Australian montane flora. The low rate of resprouting, together with the patchy seedling regeneration observed, indicates that it is likely that the shrubland will take a lot longer than the 8 years postulated by Williams *et al.* (2012) in relation to the recovery of the alpine Bogong High Plains in Victoria, Australia. With one resprouting species (*O. revoluta*) currently dominating the regeneration, it will be interesting to see whether this has the effect of decreasing the woody shrub diversity of the shrubland in the future. In montane areas woody shrubs tend to establish in bare ground in the less exposed sites (Bridle *et al.* 2001). Additionally, the size of the bare ground gaps can influence regeneration with smaller gaps favouring grasses and larger gaps favouring woody shrub seedlings (Williams 1992). A longer term study focusing on the regeneration success of species over time would improve our understanding of the impact of the 2019 fire on the vegetation dynamics of the shrubland.

The resilience of the shrubland may be threatened by a warmer, drier climate leading to a risk of more frequent and more severe fires (Barbero *et al.* 2015). Under warmer, drier climate conditions plants may experience decreased seed production and slower growth rates requiring longer time periods between fires to allow for growth and reproduction. Additionally, unfavourable weather conditions following a severe fire may reduce post-fire recruitment. At the same time, a drier, warmer climate is likely to lead to increased fire frequency. The conflict between an increased fire frequency and longer growth period is likely to result in significant pressures on some ecosystems (the interval squeeze concept of Enright *et al.* (2014)). This poses a particular risk for obligate seeder systems, such as the CPCA shrubland (Fairman, Nitschke & Bennett 2016).

CONCLUSION

The 2019 Great Pine Tier fire that swept through the Central Plateau Conservation Area in February 2019 was a high severity crown fire that killed all above ground vegetation in the shrubland. Only a minority of species present prior to the fire have resprouted in the 12 months following the fire, and none exhibited a strong resprouting response. Regeneration of the shrubland therefore relies on seedling recruitment or colonisation.

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