Department of
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Environment

# How snow gum forests and sub-alpine peatlands recover after fire

Black Saturday Victoria 2009 – Natural values fire recovery program

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Parks Victoria

Victoria We will rebuild

How snow gum forests and sub-alpine peatlands recover after fire

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**Front Cover photograph**: Lake Mountain, resprouting snow gums, February 2011. (Fiona Coates)

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# Summary

## Snow gum forests and woodlands

The aim of this part of the project was to measure the impact of fires on *Eucalyptus pauciflora* (snow gum) under different fire frequencies and to address the current large gap in information on post-fire recovery of these forests and woodlands in Victoria. Pre- and post-fire snow gum stand structure was measured and compared at three mountains with comparable geology, altitude and topography but with different fire histories: recently-burnt forest at Lake Mountain, long-unburnt forest at Mt Baw Baw and Mt Buffalo, and forest burnt in one or more fires at Mt Buffalo.

While long-unburnt sites at Mt Baw Baw and Mt Buffalo were dominated by large trees with fewer than four stems, sites burnt in two or more fires at Mt Buffalo were characterised by multi-stemmed trees with numerous thin stems. A comparison of stand structure revealed three regeneration syndromes: (i) long-unburnt stands characterised by a single establishment phase in response to fire disturbance, most likely the 1939 fires; (ii) long-unburnt stands characterised by continuous regeneration and ongoing seedling recruitment independent of major disturbances, and (iii) multi-burnt stands with multiple stem establishment phases.

At Lake Mountain, stems which had originated after the 1939 fires were all killed in the severe fires of 2009, as were very large stems which had survived the 1939 fires. These were rarely recorded elsewhere in the study area, illustrating the widespread loss of old growth snow gum from these mountains under high fire frequencies. Tree mortality was 15%, which may be higher than expected after fire. However, even though post-fire seedling recruitment in snow gum forest is high, seedlings are likely to remain in a suppressed state and cannot compensate for the loss of mature adults. One site at Mt Buffalo, burnt four times since 1972 and with a structure typical of post-fire coppicing, may have reached a threshold where multi-stemmed architecture has become entrenched as numerous stems assume equal apical dominance. A return to a structure characterised by few, large stems seems remote and unlikely. Low total basal area at this site also suggests that capacity for above ground carbon storage may be constrained when there is investment in numerous small stems.

The size-class range of dead stems at multi-burnt sites demonstrated that stems of all sizes can be killed during a fire, implying that there is uncertainty around fuel reduction benefits of planned burning in snow gum forests. The multi-stemmed growth that results from fires may also increase flammability by producing numerous, thin stems (elevated and aerated fuel). In addition, repeat fires are likely to impact on the ability of snow gum forests to perform important landscape functions. Measurements of pre-fire and post-fire stand structure during this study provide a baseline for monitoring past and future regeneration. The results will facilitate better understanding of stem growth rates and stem thinning dynamics of this long-lived species and will inform the development of fire protection strategies which aim to preserve preferred structural types.

Although snow gum forests and woodlands comprise the most widespread vegetation type in the Victorian sub-alps, the recent frequency and extent of fires have led to a potentially irreversible degradation of stand structure, to the extent that old growth woodlands are now rare. Pre-fire forest structure at Lake Mountain will only be restored in the prolonged absence of future fires for at least 70 years and it is crucial that stands at Mt Baw Baw and Mt Buffalo are protected from fire as far as is practicable. Fire exclusion is imperative to preserve landscape quality and representation of unique, long-unburnt snow gums in the parks and reserve estate.

Recommendations are:

1. Manage fire in sub-alpine environments to:
   - foster the development of mature forest structure;
   - enable long-term regeneration of recently burnt stands;
   - preserve rare old growth stands;
   - promote above- and below-ground carbon storage.
2. Investigate the feasibility of a thinning program at selected sites within the study area by:
   - conducting workshops with relevant specialists to discuss the ecological merits of stand manipulation.
3. Conduct further surveys to:
   - measure stands at Lake Mountain five years after fire to monitor post-fire regeneration (2013–2015);
   - measure forest structure at a range of sites in Victoria and incorporate existing work from previous studies to provide an inventory of snow gum stand structure;
   - conduct vegetation surveys in snow gum forests to better understand post-fire vegetation dynamics and confirm or correct data sets of existing growth stages in vegetation recovery.

## Sub-alpine peatlands

This part of the project aimed to determine risks to, and rates of recovery of threatened peatland vegetation at Lake Mountain after the 2009 bushfires in comparison to long-unburnt peatlands at Mt Baw Baw 70 years after fire and frequently-burnt peatlands at Mt Buffalo two, five and seven years after fire. Mt Baw Baw sites were identified as reasonably undisturbed and the benchmark for full post-fire recovery, while Mt Buffalo sites were considered relatively disturbed and still in the early stages of regeneration.

Differences in peat properties and species evenness suggest that floristic composition and vegetation structure at Lake Mountain is likely to recover more rapidly than comparable vegetation at Mt Buffalo. Low species diversity and evenness seven years after fire at Mt Buffalo imply that recovery of these peatlands to a benchmark or near-benchmark state is likely to be significantly slower, or may never occur if recent high fire frequencies are maintained. There was no evidence of significant threats to vegetation recovery at Lake Mountain, other than the potential threat of another fire. The results of the study demonstrated that peatlands are able to regenerate after a single fire but frequent fire maintains peatland vegetation in an early successional state.

There were very few rare or threatened species recorded at Lake Mountain during the survey. It is not certain whether this is a result of fire severity in 2009, or may simply reflect the restricted area of available habitat, or is an artefact of environmental characteristics. In contrast, rare species were well represented at Mt Baw Baw and Mt Buffalo. Some species which were sparse or absent two years after the 2003 fires at Mt Buffalo were more frequent by 2011 or had remained stable. Given the slow rate of recovery of some rare or threatened species at frequently-burnt sites at Mt Buffalo and the high number recorded at long-unburnt sites at Mt Baw Baw, additional rare species might appear with increased time since fire at Lake Mountain. *Richea continentis* (candle Richea) was slow to recover at Lake Mountain and Mt Buffalo two years after fire, but seven years after fire was showing signs of good recovery at Mt Buffalo. *Sphagnum cristatum* (peat moss) was very slow to recover regardless of time since fire at these two mountains but was a major structural component of long-unburnt vegetation at Mt Baw Baw. Given the high levels of moisture and organic content of peat at Lake Mountain, which were comparable to the benchmark, it is likely that both of these keystone species will increase over time.

The outlook for vegetation recovery at Lake Mountain is positive. High soil moisture and organic content in peatlands and a history of infrequent fire suggest that the vegetation will eventually recover following a similar trajectory to Mt Baw Baw, provided there is no re-occurrence of fire. However, continued high fire frequency at Mt Buffalo and the threat of fire at Mt Baw Baw are of significant concern. Recommendations are:

1. Repeated surveys should be conducted at all three mountains within the next three to five years, to measure ongoing vegetation recovery.
2. Current fire and cattle exclusion policies at Lake Mountain, Mt Baw Baw and Mt Buffalo should remain in place to prevent damage to recovering and/or long-undisturbed peatland vegetation.
3. Active measures should be taken to exclude fire from peatlands at all three mountains, with high priority given to peatlands at Mt Baw Baw.
4. Long-term monitoring to detect changes to peat properties should be established, in conjunction with vegetation monitoring.

# 1 Introduction

A qualitative and quantitative understanding of the response of plant populations to disturbance informs management decisions and is important in distinguishing short-term changes from long-term changes. The response of species to repeated or infrequent perturbations, such as fire, poses a number of challenges for predicting the rate at which populations can regain their initial structure and function and thresholds beyond which recovery to a previous state may not fully occur (Westman 1986).

Recent high fire frequencies in mountainous terrain in south-eastern Australia have provided many opportunities to improve our understanding of post-fire vegetation recovery. However, opportunities to gather information on unburnt or rarely burnt sub-alpine vegetation are now exceedingly rare. High fire frequency is expected to continue into the future (Parry *et al.* 2007), including in areas with a history of rare or infrequent fire such as Lake Mountain and Mt Baw Baw in southern Victoria. Questions regarding the ability of these systems to withstand single or recurrent fires are now being raised. How soon can these landscapes sustain another fire without ecological degradation and loss of biodiversity? Which populations or vegetation communities are the most seriously impacted – those burnt by a single fire of high intensity, or by a regime of frequent fires? What short- and long-term changes can be expected?

This report examines two components of sub-alpine vegetation – snow gum forests and woodlands, and sub-alpine peatlands. The first part of the project will (i) compare snow gum stand structure at Lake Mountain after a single fire with stands burnt in repeated fires (Mt Buffalo) and with long-unburnt stands (Mt Baw Baw); (ii) identify any specific management needs; (iii) determine to what extent and over what timeframe snow gum forest at Lake Mountain is likely to recover, and (iv) identify recovery milestones and confirm or correct growth stages in vegetation recovery in existing data sets.

The second part of the project will predict risks to, and rates of recovery of sub-alpine peatland vegetation at Lake Mountain after the 2009 bushfires. Peatlands are listed under the *Victorian Flora and Fauna Guarantee Act 1988* and the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* as a threatened community and require particular attention with respect to management and rehabilitation. This part will (i) compare post-fire vegetation structure and composition at sites burnt in 2009 with frequently-burnt (1972–2003) vegetation at Mt Buffalo and with long-unburnt (post-1939) vegetation at Mt Baw Baw; (ii) identify peatland species that might be at risk after the 2009 fire at Lake Mountain; (iii) identify any specific management needs for significant species; (iv) determine to what extent the vegetation is likely to recover, the timeframe for recovery and recovery milestones, and (v) confirm or correct growth stages in vegetation recovery in existing data sets (Cheal 2010).

## 1.1 Study areas

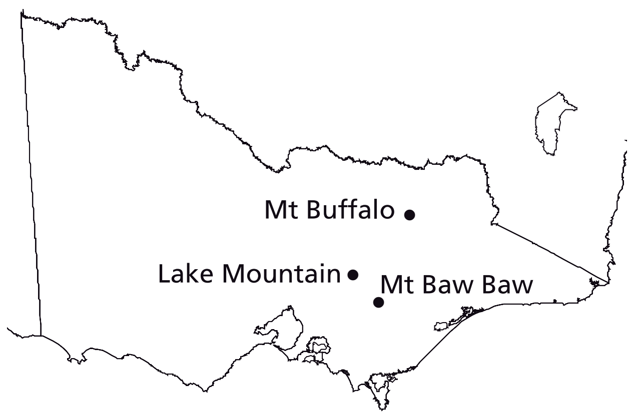
The three study areas, Lake Mountain, Mt Baw Baw and Mt Buffalo, include a range of sites which are similar in many respects (Figure 1). All three are isolated from the more contiguous and extensive alpine and sub-alpine areas of the Great Dividing Range (McDougall and Walsh 2007, Coates and Walsh 2010) and consist of mildly dissected plateaux which determine drainage patterns (Rowe 1970, Ashton and Hargreaves 1983). Plateaux of all three mountains are below 1,700 m ASL. The Lake Mountain plateau is the smallest of the three mountains and the Mt Baw Baw plateau the most extensive, although average plateau height is remarkably similar (Table 1). All three mountains are composed of granitic parent material: granite at Mt Buffalo, granodiorite at Mt Baw Baw, and rhyodacite at Lake Mountain.

Table 1. Maximum and average elevations and area of plateaux on each of the three mountains.

| Mountain | Max height (m ASL) | Average height (m ASL) | Plateau area (ha) |
|---|---|---|---|
| Lake Mountain | 1,482 | 1,348 | 2,352 |
| Mt Baw Baw | 1,565 | 1,378 | 6,941 |
| Mt Buffalo | 1,693 | 1,395 | 5,757 |

All three mountains were grazed by livestock from the early days of European settlement until leases were terminated in 1958 (Mt Buffalo), 1964 (Lake Mountain) and 1978 (Mt Baw Baw). Mt Buffalo has the longest and most intense history of land use by Europeans and various areas of the plateau have experienced a range of disturbances since the mid-19$^{th}$ century (Webb and Adams 1998). Weeds are relatively sparse at higher elevations on all three mountains, except at the margins of vehicle tracks and roads (McDougall and Walsh 2007, Coates and Walsh 2010).

Figure 1. Locations of Lake Mountain, Mt Baw Baw and Mt Buffalo.



## 1.2 Climate

The three mountains represent a climate gradient from southern areas subject to oceanic influences, to more continental northern and inland areas. All three mountains have high annual rainfall which persists as snow at upper elevations for up to about four months during winter and early spring.

Lake Mountain is the most westerly of the three mountains. Mean annual precipitation is around 1,600–1,900 mm (Ashton and Hargreaves 1983, McKenzie 1997). BIOCLIM estimates (Nix 1986) of mean annual temperatures indicate a range between 6.5°C and 10.4°C (McKenzie 1997).

At Mt Baw Baw, mean annual precipitation is around 1,540 mm (Kershaw *et al.* 1993) and mean annual temperatures are 5.7°C to 6.2°C (Kershaw *et al.* 1993). Mt Baw Baw is exposed to southerly, maritime weather and temperatures rarely exceed 30°C in summer (Tolsma and Shannon 2009).

Mean annual precipitation at Mt Buffalo is 1,856 mm (Coates and Walsh 2010) and strongly seasonal relative to the other two mountains. Mean annual minimum and maximum temperatures are 5°C and 11.7°C respectively, although lower temperatures would be expected in the high valley plains owing to cold air drainage from the adjacent slopes (Rowe 1970).

## 1.3 Fire history

Numerous bushfires have been recorded throughout the study area since 1850 (Zylstra 2006) but in many cases their precise localities are not known and it is not possible to reconstruct fire histories prior to 1939. However, parts of Mt Baw Baw National Park were probably burnt in 1923/24 and extensively burnt in 1939. Parts of Mt Buffalo appear to have been burnt in 1923/24 or 1926 and to a minor extent in 1939 (Hodgson 1927, Rowe 1970, Zylstra 2006). Lake Mountain plateau was burnt in 1939 (Ashton and Hargreaves 1983, Zylstra 2006, Shannon and Morgan 2007). Graziers are also believed to have lit fires to promote 'green pick' but these fires are not documented and their frequency, severity and extent is unknown.

The recent fire histories of the three mountains differ markedly since 1970. Lake Mountain was burnt at high fire intensity during the 2009 bushfires. The Mt Buffalo plateau was entirely burnt in 2003 and parts were burnt in 1972, 1985, and 2006 (Coates and Walsh 2010). Mt Baw Baw has remained unburnt since the 1939 bushfires (Shannon and Morgan 2007) and is currently the only sub-alpine area in Victoria that has not been burnt since 2003.

# 2 Snow gum forests and woodlands

Snow gum (*Eucalyptus pauciflora* Sieber ex Spreng) occurs in south Queensland, New South Wales (NSW), the Australian Capital Territory (ACT), South Australia, Tasmania and Victoria (Williams and Ladiges 1985). Snow gum forests occupy a range of altitudes between sea level and 2,000 m but are most typically associated with alpine and sub-alpine treelines on shallow rocky soils (Boland *et al.* 1984, Williams and Ladiges 1985, Brookhouse *et al.* 2008). Vegetation dominated by snow gum in Victoria is confined to mountains with elevations generally above about 1,300 m ASL. Since 2003, all except one of these mountains (Mt Baw Baw) have been burnt in bushfires and in the most recent fire in 2009, some of the last remaining long-unburnt sub-alpine vegetation at Lake Mountain was burnt at extremely high severity. This was the first fire to have reached the plateau since 1939 (Ashton and Hargreaves 1983). Peatlands, shrublands and extensive tracts of snow gum forest were all affected.

Snow gum is considered a resilient, long-lived tree which is able to persist *in situ* after disturbance by virtue of resprouting (Barker 1988, Barker 1989, Bond and Midgley 2001). After severe fires, stems are killed but trees rapidly resprout from epicormic shoots within well-developed lignotubers (Barker 1988, Noble 2001). Recurrent fire promotes multiple stem development, although resprouting ability appears to be weakened after low severity fires owing to relatively thin bark (Barker 1988, Lacey and Johnston 1990, Gill 1997, Noble 2001). Post-fire survival may also depend on the extent of the fire and on post-fire grazing levels (Good 1982 in Pickering and Barry 2005, Gill 1997).

In lignotuberous eucalypts, resprouting following fire-induced canopy removal is initially prolific (Noble 2001). Climate-related stress, such as drought or frost may induce low level resprouting at other times (Bell and Williams 1997). Stem thinning occurs from competition between stems (Barker 1988) as the upper branches extend and shade lower branches (Noble 2001). Adult mortality resulting from the death of the lignotuber after a single fire is rare but has been observed after repeated fires (Banks 1986, Noble 2001). Snow gums can also regenerate from seed following fire and after about six months of age, seedlings have well-developed lignotubers able to survive fires (Leigh and Holgate 1979, Carr *et al.* 1984, Noble 1984, Barker 1989). However, grazing by domestic livestock and feral herbivores has reduced the rate of seedling survival at many locations (Pickering and Barry 2005).

Since European occupation, widespread grazing, increased deliberate burning and extensive bushfires have affected much of the alpine and sub-alpine landscapes of south-eastern Australia (Rumpff 2008). At some localities, this has led to the conversion of mature open woodlands to dense stands of multi-stemmed mallees (Costin 1967, Good 1982 in Pickering and Barry 2005, Barker 1989, Pickering and Barry 2005, Rumpff 2008). In many cases, there has been insufficient time between fires (<100 years) to allow self-thinning of regrowth stems and a return to more open vegetation (Barker 1989).

Environmental factors (e.g., climatic stress, site productivity or soil depth) and land use history may interact to determine snow gum stand structure. The relative influence of these is difficult to determine in the absence of data on the degree and extent of impact of, for example, grazing intensity or fire severity (Rumpff 2008). The majority of studies which have examined the relationship between fire frequency and stem structure have focussed on alpine NSW and the ACT. In a recent Victorian study, Rumpff (2008) demonstrated that density of multi-stemmed trees at the treeline (1,700 m–1,850 m elevation) was positively correlated with increased fire frequency and grazing removal, as well as climate stress at higher elevations and sites with southerly aspects.

Snow gum demography and stand structure at topographically different sites below the treeline have not been studied in Victoria. In these landscapes, trees would be expected to tend toward a single-stemmed habit because multi-stemmed architecture in trees is less common at lower elevations (Lacey and Johnston 1990). However, multi-stemmed trees often dominate sites which have been burnt in one or more fires (Bond and van Wilgen 1996, Vesk and Westoby 2004). Sub-alpine snow gum forests occupy a range of topographies with varying environmental limitations (e.g. soil depth and rockiness), which also promote multi-stemmed architecture (Lacey and Johnston 1990). Hence, the interaction of factors which have influenced stand structure in these landscapes is not well understood. Within the study area, snow gum forests dominated by *E. pauciflora* subsp. *pauciflora* or *E. pauciflora* subsp. *acerina* Rule (endemic to Mt Baw Baw; Rule 1994) occur on upper slopes, crests and rocky rises. Understorey composition also varies within and between sites according to degree of rockiness and soil depth, aspect, cold air drainage and time since fire. On shallow soils, the understorey is generally dominated by shrubs, while grass cover increases on deeper soils. Growth form ranges from trees with about one to three or four stout stems to mallees with numerous stems.

It seems likely that the regime of widespread, high-intensity fire which has prevailed in south-eastern Victoria since 2003 has caused considerable changes to the structure of snow gum forest. As yet, the exact nature of these changes is uncertain. The general aims of this project are to predict risks, and examine potential changes, to snow gum forest structure after fire. We compare the demographic structure of stands on three sub-alpine mountains to provide baseline data against which future bushfire impacts can be assessed.

## 2.1 Methods

### 2.1.1 Study design and field methods

The Point-Centred Quarter method (Cottam and Curtis 1956) was used to estimate stem density, stem frequency and basal area of *E. pauciflora* trees at each mountain along 300 m transects which were either continuous or segmented depending on the terrain. Grid co-ordinates of transects at each mountain are provided in Appendices 1–3.

At Lake Mountain and Mt Buffalo, transects were located between 1,400 m and 1,440 m elevation. Sampling sites at Lake Mountain were located west of Triangle Junction (Table 1; Appendix 1). Sampling sites at Mt Buffalo were near the Parks Victoria Office (PVO) which were unburnt, Mt Dunn (burnt 2003, 2006), Mt McLeod (burnt 1972, 2003) and Five Acre Plain (burnt 1972, 1985, 2003, and 2006). Two sites with a fire frequency of 2 were sampled at Mt Buffalo because the inter-fire interval differed markedly and no suitable sites were located with a fire frequency of 1 or 3 (Table 1; Appendix 3). None of the sites selected at Mt Buffalo are known to have been burnt in 1939. The Mt Baw Baw transects were at higher elevations (1,500 m–1,550 m), located 0.5–1 km north-east of the Baw Baw Ski-village (Table 1; Appendix 2). Aspect varied at all sites as transects were selected randomly within burnt areas.

One hundred trees were sampled along each of two transects per fire frequency class, except at Mt Buffalo where one transect was sampled per fire frequency class owing to time constraints. Distances along each transect were derived by generating a list of random numbers. No two points were <10 m apart, to avoid the same tree being measured repeatedly.

The distance from the sample point centre to the closest tree and seedling in each of four quarters was measured to the nearest 0.1 m. Where trees were multi-stemmed, distance was measured from each point to the approximate centre of the lignotuber (Mitchell 2007). On each tree, stem diameter at breast height over bark ($D_{130}$ – approximately 130 cm above ground level) for all live and dead stems greater than 1 cm in diameter was recorded using a diameter tape measure. Dead stems which had broken away from the main clump and were lying on the ground were also measured. The diameter measures on dead stems were under bark diameters, as burnt bark had fallen from the trees after the fires. This is not expected to impact results to any serious extent as snow gum bark is relatively thin and the diameter

Figure 2. Stem clumps arising from a very large lignotuber at Mt McLeod. Some clumps are connected and others have become separated as sections of the lignotuber have rotted away.

classes used in the data analysis are relatively large. In order to establish that there is a positive relationship between stem diameter and age, previous studies have measured snow gum stems closer to the ground owing to the irregular growth habitat of some multi-stemmed *E. pauciflora* trees and their tendency to branch close to the base (Barker 1988, Rumpff 2008). A general relationship between diameter and age was assumed for the present study.

Some multi-stemmed trees at Mt Buffalo had arisen from extremely large lignotubers (>1–2 m diameter), or the remains of lignotubers where parts had decayed and divided into separate clumps (Figure 2). This has been previously recorded in trees which have repeatedly resprouted (Barker 1988, Lacey and Johnston 1990). In these cases, an attempt to detect whether there was a connection between lignotuber sections was made by excavation and the clump closest to the sample point centre was measured, although we acknowledge that in some cases this was not always accurate.

At Lake Mountain, 100% of trees had been severely burnt in 2009, so measurements were an estimate of pre-fire population structure. Trees were noted as resprouting or not because living stems were too small to measure (i.e., <$D_{130}$). Trees were unburnt at Mt Baw Baw since the last fire in 1939. Fifty points (=200 trees) were sampled at each of these mountains. Trees at Mt Buffalo were unburnt, or had resprouted after two or four fires since 1972. At Mt Buffalo 25 points (=100 trees) were sampled in each of the fire frequency classes, except at Mt McLeod (fire frequency class 2), where 20 points were sampled (=80 trees) owing to extremely dense vegetation (Table 2).

Seedlings were recorded as present if they occurred within twice the distance as the nearest tree in each quarter. Given time constraints and the density of vegetation at some sites it was not practical to search beyond this distance. It was not possible to estimate seedling density using the Point-Centre Quarter method as seedlings could not be found at each sample point. Hence, only the frequency of occurrence of seedlings per sample point was calculated. Presence or absence of fruits and resprouts were recorded for each tree if resprouts were <$D_{130}$.

### 2.1.2 Analytical methods

Stand structure and demography were analysed according to the methods of Cottam and Curtis (1956). To estimate the absolute density of trees at each site (number of trees/ha), the mean nearest neighbour distance (r) was obtained by summing the distances recorded in each of the quarters surveyed and dividing by the total number of quarters. Because r is actually an estimate of the square root of the mean area occupied by a single tree, individual density was calculated as $1/r^2$, and multiplied by 10,000 to give the number of trees per hectare. Relative tree density was calculated by dividing tree density at each site by the total tree density in all classes.

Trees were considered as having ‘mature stems’ if consisting of one to three main trunks and were more than 50 years old or likely to have arisen from the 1939 fires (Barker 1988, Pickering and Barry 2005) and ‘resprouts’ if they consisted of multiple stems less than 50 years old. At Mt Buffalo, multi-stemmed trees consisted of both dead and living stems. Dead stems were assumed to have arisen after one fire and then killed by a subsequent fire. Sites at

Table 2. Site names, fire history, number of transects and number of trees measured at each mountain.

Stem age is calculated from known fires, although other unknown fires may have occurred at sites.

LM1=Lake Mountain Site 1; LM2=Lake Mountain Site 2; BB1=Mt Baw Baw Site 1; BB2=Mt Baw Baw Site 2; PVO=Parks Victoria Office; Mt Dunn=Mt Dunn area; Mt McLeod=Mt McLeod area; 5 Acre Plain=Five Acre Plain area.

| Mountain | Site | Fire year(s) | Altitude (m) | Fire frequency post-1938 | Time since fire (yrs.) | Number of transects | Number of trees measured | Alive stem age (yrs.) | Dead stem age (yrs.) |
|---|---|---|---|---|---|---|---|---|---|
| Lake Mountain | LM1, LM2 | 1939, 2009 | 1440 | 2 | 2 | 2 | 200 | – | 70 |
| Mt Baw Baw | BB1, BB2 | 1939 | 1500–1500 | 1 | 72 | 2 | 200 | 72 | – |
| Mt Buffalo | PVO | unknown | 1400 | 0? | unknown | 1 | 100 | 70+ | – |
| | | | 1400 | | | | | | |
| Mt Buffalo | Mt Dunn | 2003, 2006 | 1410 | 2 | 5 | 1 | 100 | 5 | 3 |
| Mt Buffalo | Mt McLeod | 1972, 2003 | 1410 | 2 | 8 | 1 | 80 | 8 | 31 |
| Mt Buffalo | 5 Acre Plain | 1972. 1985, 2003, 2006 | 1430 | 4 | 5 | 1 | 100 | 5 | 13, 19, 3 |

Mt Baw Baw contained trees with living stems only and at Lake Mountain, dead stems only. The age of stems was calculated as the time between one fire and the next, or the time since the last fire. Separate calculations were carried out for dead and alive stems at each site to estimate pre- and post-fire structure, respectively.

The mean number of stems per tree (stem density) was derived by dividing the total number of stems by the number of trees measured; mean stem diameter was calculated by dividing the total diameter of all stems by the number of stems measured. Basal area for each stem was calculated by using the formula $A = \pi d^2/4$, where $A$ and $d$ denote basal area and diameter, respectively. The mean basal area was then derived from the sum of the basal area of each of the stems multiplied by tree density and divided by 10,000 (to convert to $m^2$/ha). Relative total basal area was calculated as the total basal area at each site divided by the sum of total basal areas at all sites multiplied by 100 (to give a percentage). The percent frequency of occurrence of seedlings and fruiting adults were the total number recorded divided by the number of quarters.

For each transect, stems were allocated to 10 cm $D_{130}$ size classes. Histograms of the frequency distributions of classes were compiled for each stand, representing pre-fire and post-fire population structure (as applicable).

The Kruskal-Wallis test was used to compare pre- and post-fire stem density, mean stem diameter and total basal area at each site, using Minitab® 16.1.0 Statistical Software (Minitab Inc. 2010).

## 2.2 Results

### 2.2.1 Tree density, stem number and stem diameter

Tree density at Lake Mountain (665 trees /ha; Table 3) was considerably lower in 2011 than the 1,072 trees/ha recorded 23 years after fire by Ashton and Hargreaves (1983). Pre-fire mean stem density measured on burnt trees at Lake Mountain was also lower ($n$ = 2 *cf*. $n$=2.9) than recorded 23 years after fire by Ashton and Hargreaves (1983), who also noted that many trees had thinned to single stems 43 years after fire (Figure 3; Table 3). With respect to post-fire mortality, <15% of trees failed to resprout, either as a result of fire severity or having been dead prior to the 2009 fires.

Mt Baw Baw sites (unburnt since 1939) were characterised by trees generally having three or four large stems but occasionally up to nine stems (Figure 4; Table 3). At the long-unburnt Mt Buffalo PVO site, trees were single-stemmed and occurred with *E. dalrympleana* as an occasional co-dominant (Figure 5; Table 3). Tree density was up to three times higher at the Mt Buffalo recently burnt sites compared to the long-unburnt sites and was particularly high at Five Acre Plain (Table 3). These multi-burnt stands ranged from four-stemmed trees on average at Mt Dunn (burnt twice since 2003; Figure 6; Table 3), to multi-stemmed mallees at Mt McLeod and Five Acre Plain with high (>9) but variable stem numbers (Figures 7 and 8; Table 3). Only three single-stemmed trees which clearly escaped the previous two fires in 2003 and 2006 were recorded at Mt Dunn (Table 3).

Of the two long-unburnt sites, stems at Mt Baw Baw were slightly larger than those at Mt Buffalo PVO (Table 3). The latter site is believed to have escaped the 1939 fire, although fire scars at the base of trees suggest that fire has occurred at some time in the past.

Pre-fire stem density was highest at Mt McLeod (Mt Buffalo) and lowest at Lake Mountain (Table 3). Pre-fire stem density was also relatively low at Mt Dunn and at the frequently-burnt site Five Acre Plain (Mt Buffalo; Table 3). Stems arising from previous fires at Five Acre Plain had probably been destroyed in successive fires since 1972 and pre-fire stem density is likely to have been underestimated at this site. In particular, the 2003 stem cohort is likely to be at least partly absent as these were only three years old and would have been destroyed in 2006. It is probable that the majority of dead stems at Five Acre Plain arose after the 1985 fires.

With respect to the multi-burnt Mt Buffalo sites, trees at Mt McLeod had the highest mean number of stems (17), followed by Five Acre Plain (9) but there was a high degree of variability at both sites, with stem numbers ranging from one to 46 or 49, respectively. Living stems at Mt McLeod were also larger, having had a few extra years growth (Table 2).

The number of post-fire stems was significantly higher than the number of pre-fire stems at both Mt McLeod and at Five Acre Plain ($P$ <0.0001, Kruskal-Wallace test). However, mean post-fire stem diameter was significantly lower at both sites ($P$ <0.0001 Kruskal-Wallace test). This trend would be expected to reverse with a reduction in intraspecific competition and stem-thinning over time.

Pre- and post-fire stem numbers at Mt Dunn were broadly similar ($P$ = 0.06 Kruskal-Wallace test). However, dead (pre-2006 fire) stem numbers at Mt Dunn ranged from one to 10 whilst live (post-2006 fire) stem numbers ranged from one to 21. Mean diameter of dead stems was higher (11.3 cm) than would be expected after three years growth in comparison to the other sites (Table 3), suggesting that dead stems at Mt Dunn are most likely to have originated prior to the 2003 fire. It is also likely that the three-year old stems which resprouted after 2003 were destroyed by the 2006 fire. Consequently, pre-fire structure at Mt Dunn probably relates to some previous (unknown) disturbance which occurred after 1939. Not surprisingly, mean stem diameter was significantly lower than pre-fire levels ($P$ <0.0001 in both cases).

Figure 3. Stand structure at Lake Mountain.
Time elapsed since the last fire in 2009 is two years.

Figure 4. Stand structure at Mt Baw Baw.
Time elapsed since the last fire is 72 years.

Figure 5. Stand structure at Mt Buffalo, Parks Victoria Office site.
There is no record of fire at the site, although fire scars at the base of trees imply that a fire has occurred at the site.

Figure 6. Stand structure at Mt Buffalo, Mt Dunn site.
Known fires occurred in 2003 and 2006; time elapsed since the last fire is five years.

Figure 7. Stand structure at Mt Buffalo, Mt McLeod site.
Known fires occurred in 1972 and 2003; time elapsed since the last fire is eight years.

Figure 8. Stand structure at Mt Buffalo, Five Acre Plain site.
Known fires occurred in 1972, 1985, 2003 and 2006; time elapsed since the last fire is five years.

Table 3. Tree density, mean number of stems and mean stem diameter of alive and dead stems as an estimate of pre-fire and post-fire stand structure at sites burnt since 2003 compared with unburnt sites.

Tree density is the number of trees per hectare. Unburnt sites occurred at Mt Baw Baw (BB1, BB2) and Mt Buffalo (PVO). Fire frequency is calculated since 1938 to include the 1939 fires.

LM1=Lake Mountain Site 1; LM2=Lake Mountain Site 2; BB1=Mt Baw Baw Site 1; BB2=Mt Baw Baw Site 2; PVO=Parks Victoria Office; Mt Dunn=Mt Dunn area; Mt McLeod=Mt McLeod area; 5 Acre Plain=Five Acre Plain area.

| Site | Fire frequency since 1938 | Fire history | Tree density (trees/ ha) | Mean number of pre-fire stems/ individual (dead stems) | Mean pre-fire stem diameter/ individual (dead stems) | Mean number of post-fire stems/ individual (alive stems) | Mean post-fire stem diameter/ individual (alive stems) | Mean number of mature stems/ individual | Mean mature stem diameter |
|---|---|---|---|---|---|---|---|---|---|
| LM1 | 2 | 1939, 2009 | 651 | 2±1 SD | 24.6±15.6 SD | – | – | – | – |
| LM2 | 2 | 1939, 2009 | 678 | 2±2 SD | 19.5±7.7 SD | – | – | – | – |
| BB1 | 1 | 1939 | 445 | – | – | – | – | 4±2 SD | 21.5±6.4 SD |
| BB2 | 1 | 1939 | 638 | – | – | – | – | 3±2 SD | 20.4±5.6 SD |
| PVO | 0? | unknown | 821 | – | – | – | – | 1±0.4 SD | 17.4±11.6 SD |
| Mt Dunn | 2 | 2003, 2006 | 1,468 | 3±2 SD | 11.3±5.1 | 4±4 SD | 2±1.1 | 1±0 SD | 27.3±1.5 SD |
| Mt McLeod | 2 | 1972, 2003 | 1,333 | 7±6 SD | 7.2±4.2 SD | 17±11 SD | 3.6±1.5 SD | – | – |
| 5 Acre Plain | 4 | 1972, 1985, 2003, 2006 | 2,246 | 4±4 SD | 5.4±4.3 SD | 9±8 SD | 2.6±1.4 SD | – | – |

## 2.2.2 Basal area

With respect to long-unburnt sites, total basal area of stands at Mt Baw Baw was roughly equivalent to pre-fire total basal area at Lake Mountain (Table 4). However, total basal area at Mt Buffalo PVO was about half that of stands on these other two mountains.

As expected in the short time that had elapsed since the last fire, total basal area was significantly lower than pre-fire levels at the three sites burnt since 2003 at Mt Buffalo ($P$ <0.05; Table 4). These had recovered to roughly half pre-fire levels at Mt McLeod and Five Acre Plain although these may have been under-estimated (as described above). However, post-fire total basal area was still very low at Mt Dunn. This was somewhat proportional to pre-fire stem densities (Table 3) but not pre-fire total basal area (Table 4).

## 2.2.3 Reproduction

Seedlings were most frequent at Lake Mountain and Mt Buffalo PVO (Table 5; Figures 9, 10 and 13). Continuous recruitment was occurring at Mt Buffalo in long-unburnt snow gum forest consisting of a high number of reproductive adults, whilst at Lake Mountain a pulse of recruitment had occurred following the 2009 fire. In spite of a high proportion of fruiting adults at Mt Baw Baw, seedling recruitment was very low (Figures 11 and 12; Table 5). The reproductive status of stands burnt in two or in four fires at Mt Buffalo was variable. Seedling recruitment was highest in stands with high stem numbers (Mt McLeod and Five Acre Plain; Figures 15 and 16; Table 3) and lowest at Mt Dunn (Figure 14; Table 5), even though the frequency of fruiting adults was substantial at the latter site.

Table 4. Total basal area and age of alive and dead stems indicating relative dominance of snow gum between sites.

LM1=Lake Mountain Site 1; LM2=Lake Mountain Site 2; BB1=Mt Baw Baw Site 1; BB2=Mt Baw Baw Site 2; PVO=Parks Victoria Office; Mt Dunn=Mt Dunn area; Mt McLeod=Mt McLeod area; 5 Acre Plain=Five Acre Plain area.

| Site | Fire frequency since 1938 | Fire history | Pre-fire total basal area (dead stems $m^2/ha$) | Relative pre-fire basal area (%) | Post-fire total basal area (alive stems $m^2/ha$) | Relative post-fire basal area (%) |
|---|---|---|---|---|---|---|
| LM1 | 2 | 1939, 2009 | 83 | 31 | 0 | 0 |
| LM2 | 2 | 1939, 2009 | 54 | 20 | 0 | 0 |
| BB1 | 1 | 1939 | 0 | 0 | 70 | 33 |
| BB2 | 1 | 1939 | 0 | 0 | 55 | 28 |
| PVO | 0? | unknown | 0 | 0 | 33 | 16 |
| Mt Dunn | 2 | 2003, 2006 | 48 | 18 | 2 | 1 |
| Mt McLeod | 2 | 1972, 2003 | 54 | 20 | 26 | 14 |
| 5 Acre Plain | 4 | 1972, 1985, 2003, 2006 | 32 | 14 | 13.7 | 7 |

Table 5. Frequency of seedlings and fruiting trees at each site.

LM1=Lake Mountain transect 1; LM2=Lake Mountain transect 2; BB1=Mt Baw Baw transect 1; BB2=Mt Baw Baw transect 2; PVO=Parks Victoria Office; Mt Dunn=Mt Dunn area; Mt McLeod=Mt McLeod area; 5 Acre Plain=Five Acre Plain area.

| Site | Fire frequency since 1938 | Last fire | Frequency of seedlings (%) | Frequency of fruiting adults (%) |
|---|---|---|---|---|
| LM1 | 2 | 1939, 2009 | 100 | 0 |
| LM2 | 2 | 1939, 2009 | 87 | 0 |
| BB1 | 1 | 1939 | 6 | 91 |
| BB2 | 1 | 1939 | 9 | 87 |
| PVO | 0? | unknown | 77 | 81 |
| Mt Dunn | 2 | 2003, 2006 | 3 | 44 |
| Mt McLeod | 2 | 1972, 2003 | 35 | 33 |
| 5 Acre Plain | 4 | 1972, 1985, 2003, 2006 | 56 | 65 |

### 2.2.4 Size class distributions

At Lake Mountain, no stems survived the 2009 fire (Figures 9 and 10). Pre-fire structure at LM1 consisted of a range of size classes including some very large trees (>40 cm $D_{130}$) which were largely absent from LM2. Most stems at both sites fell within classes 10–20 cm $D_{130}$ or 20–30 cm $D_{130}$.

Figure 9. Frequency of stem diameter size classes and seedlings at Lake Mountain Site 1 (LM1).

The site was burnt in 1939 and 2009 and all stems were dead. Estimated pre-fire stem age is 70 years.

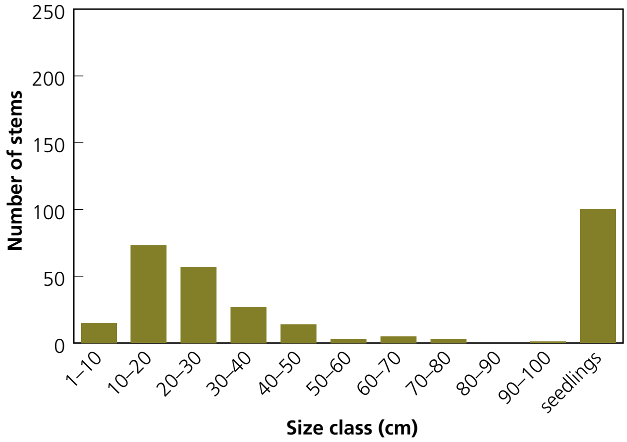
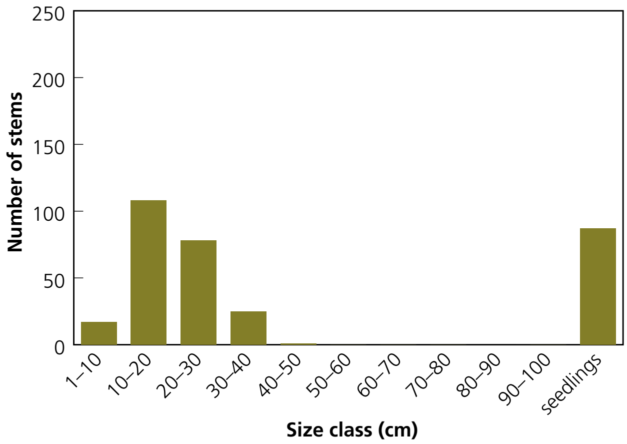



Figure 10. Frequency of stem diameter size classes and seedlings at Lake Mountain Site 2 (LM2).

The site was burnt in 1939 and 2009 and all stems were dead. Estimated pre-fire stem age is 70 years.




At Mt Baw Baw sites BB1 and BB2, stand structure indicated recruitment following a major disturbance. The majority of stems were 20–30 cm $D_{130}$ at BB1 (Figure 11) and in the 10–20 cm $D_{130}$ and 20–30 cm $D_{130}$ classes at BB2 (Figure 12). The results suggest that the disturbance history or resource availability of sites may differ. Large trees (30–40 cm $D_{130}$) existed at both sites with a few very large trees also recorded at BB1. There were a small number of stems below 10 cm $D_{130}$.

Figure 11. Frequency of stem diameter size classes and seedlings at Mt Baw Baw Site 1 (BB1).

The site was last burnt in 1939 and all stems were alive. Estimated stem age is 72 years.

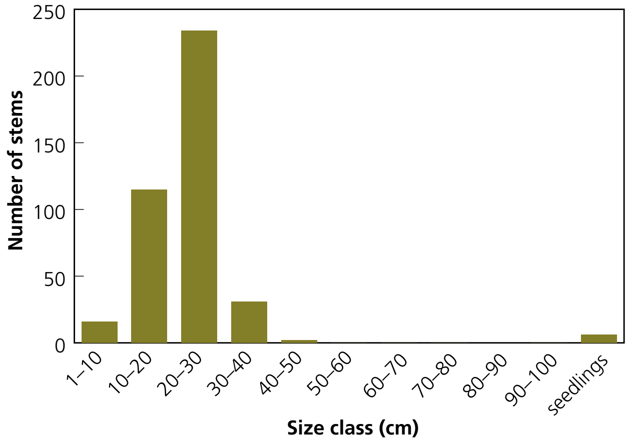
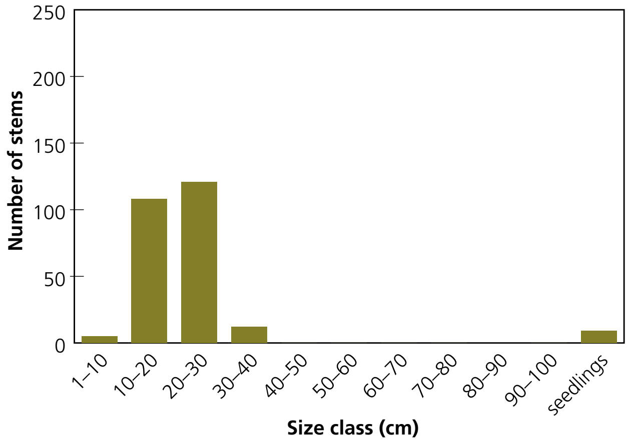



Figure 12. Frequency of stem diameter size classes and seedlings at Mt Baw Baw Site 2 (BB2).

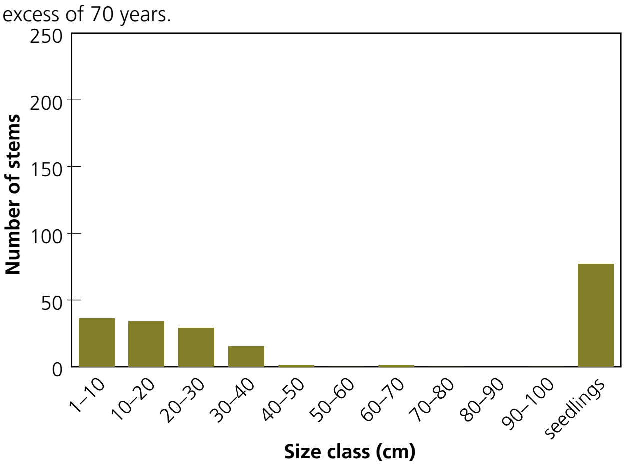
The site was last burnt in 1939 and all stems were alive. Estimated stem age is 72 years.




Stand structure at Mt Buffalo PVO comprised mostly single stemmed trees and was consistent with the assumed history of no recorded fire (Figure 13). Size classes were more or less evenly distributed. The majority of trees were allocated more or less evenly within size classes 1–40 cm $D_{130}$. There were a couple of very large trees (>40 cm $D_{130}$), suggesting that this stand has been established for a very long time.

Figure 13. Frequency of stem diameter size classes and seedlings at Mt Buffalo – Parks Victoria Office site.

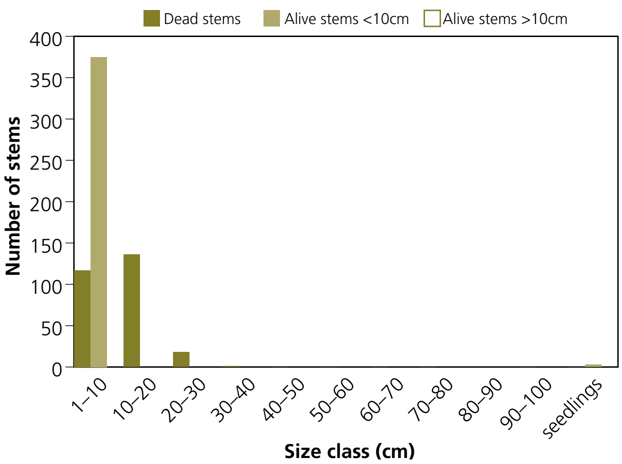
There was no record of fire for the site, which may not have been burnt in 1939. All stems were alive and estimated stem age is in excess of 70 years.




Stand structure at Mt Dunn, known to have been burnt in 2003 and 2006, comprised stems from three generations. A small number of large dead stems and living mature trees were recorded in size classes 20–30 and 30–40 cm $D_{130}$ (Figure 14). The large, living trees had survived both fires suggesting patchy fire coverage around the Mt Dunn area. These also suggested that pre-2003 fire structure probably consisted of single stemmed trees, which were replaced by a multi-stemmed growth form following 2003 (mean number of stems=3; Table 2). These resprouts were killed in 2006 and the current stand structure is almost entirely composed of numerous thin stems <10 cm $D_{130}$.

Figure 14. Frequency of stem diameter size classes and seedlings at Mt Buffalo – Mt Dunn site.

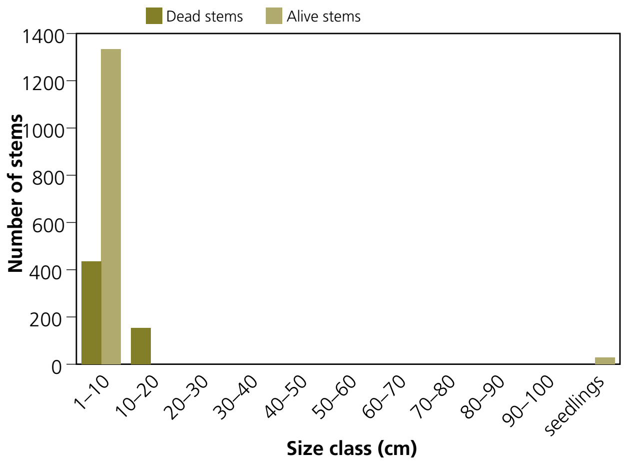
The site was burnt in 2003 and 2006. Stems were alive or dead. Stems which resprouted after the 2006 fire are 5 years old (alive stems, light grey bars). Estimated age of stems which resprouted after the 2003 fire and were subsequently killed in the 2006 fire is 3 years (dead stems, dark green bars). Some larger, living stems were also recorded ($D_{130}$ >20 cm) representing stems that survived both 2003 and 2006 fires but not represented in the scale of graph.




At Mt McLeod, stems which had resprouted after the 1972 fire had been killed in 2003 when the site was last burnt (Figure 15). The pre-fire structure was dominated by numerous small stems (<10 cm $D_{130}$) as well as larger stems (10-20 cm $D_{130}$). Stems which had resprouted after the 2003 fire were all <10 cm $D_{130}$ and occurred in extremely high numbers.

Figure 15. Frequency of stem diameter size classes and seedlings at Mt Buffalo – Mt McLeod site.

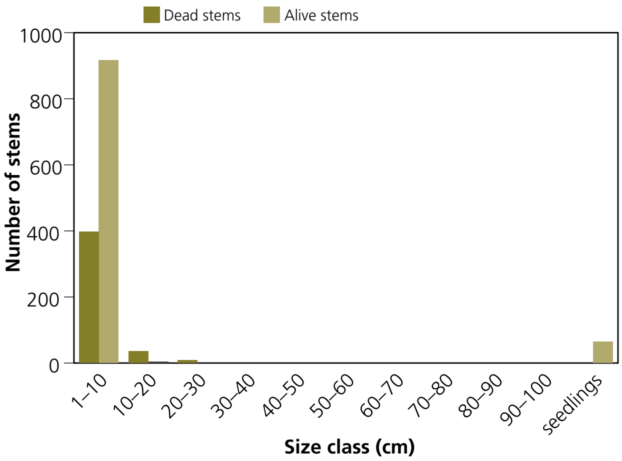
The site was burnt in 1972 and 2003. Stems were alive (light grey bars) or dead (dark green bars).




Stand structure at the most frequently-burnt site, Five Acre Plain, was characterised by numerous small stems (<10 cm $D_{130;}$ Figure 16). These smaller stems are most likely to a mix of post-2003 stems (dead) and post-2006 stems (alive), although this is difficult to confirm. It is likely that this size class also contains a proportion of stems killed by drought over the last 10 years as well as stems which have died as a result of natural thinning. There were very few dead stems measured >20 cm $D_{130}$ although where present, most likely represent regrowth after 1985 and/or 1972 fires. It is also likely that some stems killed in these earlier fires were since incinerated in 2003 or 2006 so that multiple regeneration events are not well represented in the overall stand structure.

Figure 16. Frequency of stem diameter size classes and seedlings at Mt Buffalo – Five Acre Plain site.

The site was burnt in 1972, 1985, 2003 and 2006. Stems were alive (light grey bars) or dead (dark green bars). Dead stems potentially included post-fire resprouts initiated prior to 2006 (i.e. 34 yrs., 19 yrs. and 3 yrs. old, respectively). Alive post-2006 fire stems were 5 years old.




## 2.3 Discussion

### 2.3.1 Stand structure

There was a noticeable difference in stand structure among all sites. While Lake Mountain (pre-fire structure), Mt Baw Baw and Mt Buffalo PVO were more or less similar, the multi-burnt stands at Mt Buffalo were all distinctly different from these sites and from each other.

The data show a trend toward increasing density of individuals with increased frequency of fire, both in terms of absolute numbers of stems per hectare and in relative terms among the various fire frequency classes. The number of stems per individual also increased with fire occurrence. There was also a general increase in the mean number of stems, and decreasing size, with higher fire frequency and shorter time between fires. These patterns were most pronounced at Mt McLeod and Five Acre Plain and least evident at Mt Dunn. Our finding, that fires promote an increase in tree and stem density, is consistent with other studies (Barker 1988, Barker 1989, Pickering and Barry 2005, Rumpff 2008). Recurrent fire in these snow gum stands in the future will promote a multi-stemmed habit and will further delay recovery of stand structure. This is not only because living stems are likely to be destroyed but also because natural thinning in stands with high stem densities and a mallee habit is thought to occur at a slower rate compared to rates within open forest (Barker 1988). As such, this study clearly demonstrates that an open forest structure cannot develop if recent fire frequencies persist.

Stem density at Lake Mountain was low prior to 2009 and consistent with the prediction of Ashton and Hargreaves (1983) that thinning would continue to occur in the absence of fire. Stem density was higher at Mt Baw Baw but still within expected ranges 70 years after fire. These stands will continue to thin over time if they remain unburnt. Of all the sites sampled, Mt Baw Baw has the most potential to develop the open structure characteristic of old growth woodland or open forest. Similar total basal areas at Lake Mountain and Mt Baw Baw indicate that snow gum regeneration after the 1939 fires was comparable and stand structure had developed along similar trajectories up until 2009. This also implies that snow gum forests at Lake Mountain will not return to their pre-fire structure for at least another 70 years. However, predicted climate change impacts of lower rainfall and higher fire frequency (Parry *et al.* 2007) may mean that recovery of pre-fire structure may take longer, or that these stands may never fully recover.

Stand structure at Mt Buffalo PVO was consistent with a long fire-free period. Trees were mostly single stemmed and seedling recruitment was high. Basal area was low compared to other mature stands, but this was expected because the PVO trees were slightly smaller. Given the assumed fire history, these trees are still relatively small, which may be a consequence of site conditions (e.g. water availability, soil depth or fertility), or inter-specific competition. *Eucalyptus dalrympleana* was an occasional co-dominant, indicating that frost and snow cover are less extreme relative to sites totally dominated by snow gum (Farrell and Ashton 1973), allowing co-existence between the two eucalypt species.

The other three sites at Mt Buffalo (Five Acre Plain, Mt McLeod, Mt Dunn) were burnt at moderate to high severity in 2003, during a period of extended drought. Whilst it is reasonable to expect some patchiness in fire coverage, the 2003 fire was of sufficient severity to burn peatlands and rocky outcrops across the entire plateau (e.g. Figure 17; Coates *et al.* 2006, Coates and Walsh 2010). This was rare in 2006; however, burnt peatlands and rocky outcrops occurred mostly at Five Acre Plain (Coates and Walsh 2010), where fire severity was high in some areas (e.g. Figure 18) although not in others (e.g. Figure 19). We expected that short inter-fire intervals at Mt Dunn and Five Acre Plain would have prevented stem establishment between 2003 and 2006, imposed a degree of stress on lignotuber carbon reserves from repeated resprouting, and/or destruction of meristematic tissue left unprotected between successive fires.

In spite of high post-fire stem numbers, total basal area was still relatively low at Five Acre Plain, suggesting that above ground carbon storage may be constrained when there is investment in numerous small stems (Figure 20). Since 1972, trees at this site have probably remained in the 'proliferative' growth phase characteristic of post-fire coppicing (Noble

2001). A risk of recurrent fire is that trees lose their capacity to develop a single stem and multi-stemmed architecture becomes entrenched as numerous stems assume equal apical dominance. As a consequence, trees will eventually become incapable of producing large stems and persist as clumps of small stems, even after a long fire-free period (Lacey and Johnston 1990). High rock cover at Five Acre Plain may protect lignotuber buds and reduce mortality rates within the population to some extent by disrupting fire coverage. Thus, it is possible that some trees were sampled that had not been burnt in the 2006 fire, resulting in an overestimate of post-fire basal area at this site. Seedlings were recorded in over half of the points sampled but given the dense canopy and understorey vegetation, these seedlings are likely to persist in a suppressed state or may die.

Cattle grazing has been credited with suppressing post-fire stem and seedling regeneration in alpine snow gum forests in the past in some more elevated locations on deeper soils with relatively herbaceous understoreys, for example Kosciusko NP and the Bogong High Plains (Rumpff 2008). Prior to 1958, it is probable that cattle occupied the area around Five Acre Plain, where salt was left for cattle before the autumn muster (Webb and Adams 1989). However, it is doubtful that cattle accessed the relatively dry, rocky terrain at the site, or preferentially grazed the shrubby understorey over adjacent herbaceous vegetation. It is more plausible that snow gum forest at this site tends toward a multi-stemmed habit owing to shallow soils and high fire frequency associated with bushfires and escaped fires from deliberate burning of vegetation by graziers in the past. It is also likely that there has been an increase in stem density since the 2006 fires but since some pre-fire structure has been destroyed (Figure 18) it was difficult to measure the magnitude of this increase.

We challenge the notion that removal of cattle promotes a multi-stemmed stand structure (Barker 1988, Rumpff 2008). Firstly, *Eucalyptus* is not a preferred food for cattle (van Rees 1984) and secondly, even if sites were grazed, multi-stemmed architecture is likely to have become more prevalent as a consequence of repeated defoliation (browsing), in much the same way as clipping or fire effects (Lacey and Johnston 1990, Vesk *et al.* 2004). Thus, we consider that the influence of cattle grazing on stand structure is likely to have been insignificant at Mt Buffalo and it is highly unlikely that current high stem densities have occurred due to a lack of grazing.

The structure of twice-burnt stands at Mt Dunn and Mt McLeod was markedly different. Tree density was comparable but pre- and post- fire stem densities were four times higher at Mt McLeod. Mean stem diameter and total basal area were also higher at Mt McLeod eight years after fire, compared to Mt Dunn stems at five years.

At Mt Dunn, there was no recorded fire prior to 2003 which also implies that stem growth is likely to have dominated over lignotuber growth until then (Carr *et al.* 1984). Pre-fire structure at Mt Dunn consisted of few-stemmed trees and only 10% of the trees measured had more than five stems. A few large trees were recorded but overall stand structure suggests that other factors (for example, drought or locally severe frost) may have killed seedlings and/or 2003 regrowth (Farrell and Ashton 1973). However, the mean diameter of dead stems was relatively large (11.3 cm) and their total basal area was comparable to LM2. The most likely explanation for stems of this size is that they had originated after an unrecorded fire around the time of cattle grazing, prior to removal in 1958.

The grassy understorey at this site is suggestive of deeper soils and higher site productivity, which also favour a shift away from multi-stemmed architecture. If this was the case, post-fire resprouting may have been constrained by lack of starch storage (Pate *et al.* 1990) and by two fires close together. The apparent anomalies at this site could be resolved using dendrochronology to establish the age of these stems and may shed light on the disturbance history of the site.

The mallee form and numerous thin stems at Mt McLeod suggest that this site was burnt in unrecorded fires prior to 1972. Its location on the northwest part of Mt Buffalo would also expose the site to bushfires, which typically occur on days of strong northerly winds. These may have led to dense stand structure after the severe fires of 1972 and 2003 (Dexter 1973, Coates *et al.* 2006). Lignotubers of most trees that were measured were very large, suggesting long-term dominance of lignotuber growth over stem growth, which typically occurs when existing shoots are replaced by other shoots and the secondary lignotubers fuse laterally (Carr *et al.* 1984). Soils are relatively shallow and rocky, which would also explain the mallee growth form which dominates this site. Regardless, multi-stemmed architecture and lack of apical dominance is likely to have been in place for a very long time at Mt McLeod. Lignotuber morphology consisted of stem clumps distributed in wide rings, consistent with descriptions in the literature of repeated resprouting after the upper lignotuber surface is killed and new growth is initiated at the periphery (Lacey and Johnston 1990).

Post-fire total basal area at Mt McLeod was high and comparable to PVO, suggesting that stem recovery at this site is reasonably good. There is no record in the available literature to confirm that Mt McLeod was ever grazed (e.g. Rowe 1970, Binder 1978, Stephenson 1980, Webb and Adams 1998) and it is unlikely that cattle occupied this remote site where access is difficult. Furthermore, the lack of stems in size classes greater than 10-20 cm $D_{130}$ adds further weight to the notion that multi-stemmed trees have prevailed at this site for a long time from continuous resprouting after fire and/or as a consequence of environmental stress.

Figure 17. Mt Dunn area in November 2004, 10 months after the 2003 fire. Grassland (foreground), peatlands (middle ground) and rocky rises (background) supporting shrubland or snow gum forest were all completely burnt.

Figure 18. Severely burnt area of snow gum woodland at Five Acre Plain in February 2007, 1 month after the 2006 fire.

Figure 19. Snow gum woodland at Five Acre Plain (see also Figure 18), showing regrowth from the 2003 fire. This patch escaped the 2006 fire which burnt much of the surrounding woodland one month before the photograph was taken in February 2007.

Figure 20. Regrowth from a large lignotuber at Five Acre Plain and the burnt remains of regrowth from previous fires.

### 2.3.2 Seedling recruitment

Seedling recruitment was prolific at Lake Mountain after the 2009 fire and throughout the subsequent period of average rainfall. This is consistent with the idea that vegetative reproduction prevails during periods of environmental stress and sexual reproduction occurs during more favourable periods (Lacey and Johnston 1990), such as after fires under conditions of increased resource availability (soil nutrients and light).

Ashton and Hargreaves (1983) also recorded high seedling densities 23 years after fire, which they attributed to a lack of understorey competition. Their results suggest that some thickening of the vegetation occurred after the 1939 fires which diminished over time with natural thinning. The high number of seedlings recorded in this study implies that vegetation thickening will occur again at Lake Mountain but will decline with restoration of the understorey and natural attrition. It will be interesting to monitor the fate of seedlings at Lake Mountain where there is a high incidence of vigorous lignotuber resprouting and relatively high adult survival.

At Mt Baw Baw, seedling occurrence was low in forest with a dense shrubby understorey. This result was expected and supports the notion that understorey competition suppresses seedling establishment (Ashton and Williams 1989). Alternatively, there was also a relatively high incidence of seedlings at PVO, demonstrating that seedling regeneration can also occur in the absence of fire, provided there is an open understorey. These seedlings are also likely to persist in a suppressed state until canopy disturbance provides them with growing space. This disturbance might take the form of fire or canopy damage resulting from snowfall or wind.

Seedling frequency varied among sites with recent fire histories. This might be attributed to variation in local environmental conditions rather than fire incidence alone. The very low seedling frequency at Mt Dunn is most likely a result of the short time between the two recent fires but may also indicate poor post-fire seedling recruitment with competition from post-fire stem and grass regrowth. Low seedling recruitment might also be expected during the drought that prevailed around the time of the two fires.

Seedling recruitment was moderate at Five Acre Plain and Mt McLeod. However, although fire in 2006 at Five Acre Plain probably destroyed seedlings which had originated from the earlier fire in 2003, seedlings were still more frequent than at Mt Dunn in 2011. Given the high rock cover at Five Acre Plain, fire coverage in 2006 is likely to have been patchy in places and seedlings better protected than on the deeper soils at Mt Dunn. Furthermore, trees with numerous, smaller stems at Five Acre Plain and Mt McLeod may be more productive than the larger but fewer-stemmed trees at Mt Dunn. We recorded only presence or absence of fruits per tree and a better estimate of stem fecundity may clarify this. However, dense understorey vegetation at both of these sites, for example compared to the PVO site, tends to suggest only limited opportunity for seedling establishment (Figure 21).

### 2.3.3 Size class distribution

Our data show that very large (likely pre-1939) trees have all but disappeared from the three mountains, with only a few remaining in isolated pockets at Mt Baw Baw (and none at Lake Mountain or Mt Buffalo).

With the exception of the stand PVO, all the stands measured show pre-fire size/age structures which indicate pulse regeneration following a major disturbance (in these cases fire). Stand PVO, in contrast, has a size/age structure which appears to indicate more or less continuous regeneration. The largest stem cohort is the smallest/youngest with decreasing numbers in larger/older stem cohorts and a few very large individuals. Seedling frequency supports this view. The relationship between stem diameter and age in trees can be variable. However, there was a clear grouping of stem size distributions and known fire histories and we do not consider the lack of known age classes a major impediment to the study.

At Lake Mountain and Mt Baw Baw the majority of dead stems were in size classes 10–20 and 20–30 cm $D_{130}$, which most likely represent regrowth after the 1939 fire. This pattern was consistent at both mountains. However, there was a number of very large/old individuals present at LM1 in size classes >40 cm $D_{130}$. This may point to better growing conditions or, more probably, that the previous fire was patchy and did not destroy the stems of very old trees. However, this was not the case in 2009 when 100% of trees were burnt at extremely high intensity. In contrast to relatively heterogeneous pre-fire structure, snow gum forest at Lake Mountain will comprise stands of even-aged stems for the foreseeable future.

At Mt Dunn, the few trees recorded in the 10–20 and 20–30 cm $D_{130}$ size classes imply that this site was burnt in 1939 but regeneration has since been poor and no trees have survived which pre-date the 1939 fire. At other multi-burnt Mt Buffalo sites, there were almost no stems >20 cm $D_{130}$.

The size-class range of dead stems at multi-burnt sites suggests that stems of all diameters can be killed during a fire so that stem size is unlikely to be useful for re-constructing fire severity. Large stems which had survived some fires were occasionally recorded at Mt Buffalo but these were rare. Snow gum bark is thin throughout the lifespan of the tree, a trend that differs from some other eucalypts where bark thickens with age, and resprouting occurs from lignotuber buds protected from fire by soil and rocks (Williams and Ashton 1988). Therefore, it is possible that even low severity fires, including planned burns, aimed at reducing fuel loads, might be equally likely to kill living stems as bushfires, leading to further degradation of snow gum forest structure.

Figure 21. Mallee-form snow gum woodland at (left) Mt McLeod, consisting of multi-stemmed regrowth, compared to (right) open forest at the Parks Victoria Office dominated by single-stemmed trees.

## 2.4 Recovery and management of snow gum forest and woodlands

### 2.4.1 How resilient are snow gum forests to recurrent fire?

This study found that snow gum is an extremely resilient tree able to survive disturbance by resprouting, fitting the category of 'niche persistor' (Bond and Midgley 2001). The main threat to forest structure is the conversion of open forest to dense, multi-stemmed stands following one or more fires. While this may not be detrimental to survival of the species, this stand structure is now becoming more dominant across south-eastern Australia. Recurrent fires will maintain these stands in this multi-stemmed regrowth state. Even though increased fire frequency appears to promote seedling recruitment, this cannot compensate for the observed loss of mature adults because seedlings typically persist in a suppressed state. Continuous regeneration was recorded at only one long-unburnt site (PVO) but was extremely slow and had likely occurred as a consequence of localised canopy disturbances.

While trees readily resprout after fire, at sites with very high fire frequency and/or short inter-fire intervals (Five Acre Plain, Mt Dunn), the results suggested that some trees may have reached, or are close to reaching a threshold beyond which recovery may not be possible if high fire frequency persists. Trees at Five Acre Plain and Mt Dunn consisted of very thin stems and at both sites, basal area was the lowest recorded during the study. This questions the degree to which snow gum can persist under particular disturbance regimes. Persistence may be enhanced under a regime of intermediate disturbance frequency (e.g. ~80-100 years; Cheal 2010) and eroded under more extreme disturbance regimes (e.g. very frequent fire < 35 year intervals, Cheal 2010; or long fire-free intervals followed by frequent fire). Under the latter scenario, we suggest that above and below-ground carbon reserves and lignotuber bud banks will decline; or after long periods of apical dominance may be capable of producing only depauperate trees. Further work is needed in this area to explore ecological amplitude, or the threshold of stress beyond which recovery to an initial state does not occur (Westman 1986). In particular, the relationship between the regenerative capacity of lignotubers and resprouting rates, including the ability of very old trees also subject to attack from insects and pathogens, to survive fire.

### 2.4.2 Recovery timelines and management

No specific management needs for regenerating snow gums at Lake Mountain were identified during the survey. Most trees were resprouting and seedlings were abundant, although some trees were killed. The few studies of post-fire mortality rates in eucalypts recorded 5% mortality rate in semi-arid mallees and 15% in forests and woodlands near Canberra (Wellington and Noble 1985, Gill 1997). Given that 15% of trees failed to resprout at Lake Mountain, mortality was within this range. We were unable to find any comparable data for *E. pauciflora* but observations at Mt Buffalo suggest that snow gum mortality is relatively rare after fire, in which case mortality at Lake Mountain may be higher than expected, assuming that these trees had been alive immediately prior to the fire. Even though their 'new' multi-stemmed growth form is unlikely to impact on fecundity levels, we estimate that it will take at least another five to 10 years before new stems become reproductive. This stage may occur more rapidly in sub-alpine areas than previously noted for high altitude woodland (Cheal 2010).

Pre-fire stand structure consisted of few-stemmed trees, 70 years after the last fire in 1939 (Figure 22). It is certain that trees will return as multi-stemmed individuals with a higher number of stems than before the 2009 fire (Figure 23) but these will thin over time in the absence of fire. Ashton and Hargreaves (1983) calculated a stem-thinning rate of about 25% between 1962 and 1982. Post-2009 regeneration was not measured during the current project (as it was typically <1 cm at $D_{130}$) but we recommend that these stems are measured within the next few years (2013–2015). It may be unrealistic to expect development of an open woodland structure characteristic of sites with no history of fire within a time frame of shorter than several centuries (Barker 1988). The time-frame for recovery of the pre-2009 fire forest structure at Lake Mountain is likely to be in the order of 70–100 years without fire. Therefore, to restore pre-2009 structure at Lake Mountain, the mountain needs to be protected from fire. Although this might not be achievable, we recommend that this aim be pursued as far as is practicable. A targeted stem thinning trial could be considered, as has been implemented for Box-ironbark forests in central Victoria (Pigott *et al.* 2010) and in river red gum floodplain forests to improve habitat quality and rates of above ground carbon storage (Horner *et al.* 2010). We recommend further investigations to determine whether this is a viable course of action and if so, over what time-frame.

### 2.4.3 Recovery milestones and growth stages

An objective of the project was to identify recovery milestones at Lake Mountain and confirm or correct growth stages in vegetation recovery in existing data sets. The three mountains were broadly comparable in terms of geology, topography and altitude. However, no specific recovery milestones could be identified, mainly owing to the differing fire histories at each of these sites and their influence on current stand structure, soil depth and rockiness at the site scale, the ability of snow gums to survive single or multiple fires by prolific resprouting and their lack of reliance on establishment of a seed bank. Thus, the length of inter-fire intervals and time to develop a few- or single-stemmed habit is more relevant at infrequently or long-unburnt sites (e.g. Lake Mountain, Mt Baw Baw, PVO and possibly Mt Dunn) but not necessarily all sites (e.g. Mt McLeod, Five Acre Plain). However, measurements of pre-fire stand structure have provided a baseline for future monitoring and understanding stem growth rates and natural stem thinning dynamics of this long-lived species which dominates the most widespread vegetation type in the sub-alps.

The results of this project have limited application to confirming or correcting data sets of existing growth stages of snow gum forests and woodlands *sensu* Cheal (2010), as no data were collected on co-occurring species. These data are needed to describe successional stages at the community level from early post-fire recovery ('renewal', Cheal 2010), to a structural stage characterised by trees tending toward a monopodial habit ('waning', Cheal 2010). At present, the floristic composition of understorey vegetation in snow gum forests has not been described in Victoria. This study has provided some insight into snow gum structure at a limited number of sub-alpine locations but should be extended in future to include other areas and incorporate floristic surveys.

### 2.4.4 Future outlook

This study has described stand structure of long-unburnt snow gum forest at Mt Baw Baw and the only long-unburnt site which exists at Mt Buffalo. However, as these sites support some of the only (relatively) long-unburnt snow gum forest in Victoria, excluding fire from these areas is also imperative to maintain these important stands and to preserve landscape quality (Leigh and Noble 1981 and references therein). Some very old, single-stemmed snow gums with large hollows still exist in a few pockets of the Mt Baw Baw plateau (Sue Berwick, DSE, pers. comm.) which presumably escaped the 1939 fires. Although a widespread vegetation type, the recent frequency and extent of fires have led to a potentially irreversible degradation of forest structure in many parts of the State and it is appropriate and indeed crucial, that long-unburnt stands are represented within Victoria's parks and reserves estate. In alpine environments, open snow gum stands have been shown to provide the most effective means of water collection and regulation and repeated fire may have considerable impact on the ability of these forests to sustain snow accumulation and persistence, retain soil moisture by intercepting rain and prevent erosion (Costin 1967).

Recent severe and widespread bushfires between 2003 and 2009 have seen the introduction of mandatory targets for fuel reduction burning across Victoria's public lands to mitigate the scale and impact of future unplanned fires. Low-severity fire has been associated with snow gum mortality (Good 1982 in Pickering and Barry 2005). Fire severity data were not available to our study and we did not investigate any stands burned by low-severity fuel reduction fires. The impact of such fires on stand structure may be similar to that of unplanned bushfires because thin-barked snow gum stems are readily killed regardless of their pre-fire diameter. Conversely, the impact of stand structure on fire severity has not been studied in sub-alpine forests in Victoria. However studies of other forests in the northern hemisphere have found that stands consisting of many small trees are more likely to be burnt at high severity than stands with fewer trees (Lentile *et al.* 2006).

Furthermore, fire is known to negatively impact on populations of fauna that rely on old growth forest, in particular the emblematic threatened species Leadbeater's Possum at Lake Mountain and potentially at Mt Baw Baw. Federally listed reptiles Alpine She-oak Skink and Guthega Skink are associated with open vegetation including old growth snow gum woodland (N. Clemann, ARI, pers. comm.). There have been no other studies on the importance of long-unburnt snow gum forests as habitat for other species, although it is reasonable to assume that they are important for a range of vertebrate and invertebrate fauna as is known to occur in old growth forests elsewhere in Victoria (Lindenmayer and Wood 2010).

Figure 22. Unburnt snow gum forest at Lake Mountain in 2006.

Figure 23. Burnt snow gum forest at Lake Mountain in 2011.

# 3 Sub-alpine peatlands

Alpine and sub-alpine treeless vegetation is relatively rare on mainland Australia, being confined to the Australian Alps in north-east Victoria, the Australian Capital Territory and south-eastern New South Wales. There is scant evidence of fire in these environments throughout the Holocene (Singh *et al.* 1981, Kershaw *et al.* 1993, Dodson *et al.* 1994, Kershaw *et al.* 2002) and little is known about the use of fire by humans in the high country prior to European settlement (Banks 1989). People are known to have seasonally used alpine regions, particularly to harvest Bogong Moths and probably used fire for land management and other cultural activities (Costin *et al.* 1979, Flood 1980). However, there is little or no evidence for extensive use of fire to manage vegetation as has been documented in some lowland forests and grasslands.

In the last 100 years, landscape scale fires have been rare because of the low incidence of extreme fire weather conditions at high altitudes, but have become increasingly common in south-eastern Australia in the past decade during periods of extended hot, dry weather. Large fires have also occurred as a direct result of human activities (Leigh *et al.* 1987, Banks 1989, Gill and Moore 1990, Dodson *et al.* 1994, McCarthy and Tolhurst 2000, Wahren *et al.* 2001). In spite of their scientific and cultural values (Kirkpatrick 1994, Wahren *et al.* 1999b), some treeless alpine and sub-alpine areas have been licensed for grazing by domestic stock, or are relatively close to commercial forests and farmland. Humans are also able to access these remote areas using off-road vehicles or on foot, and their neglected campfires, or arson, have been known to cause extensive fires (Wahren *et al.* 2001). Prescribed burns are also used in adjacent forest areas to reduce the likelihood of bushfire but occasionally escape.

The effects of fire and other disturbances are likely to have ongoing implications for alpine and sub-alpine ecosystems. Soil erosion, increased variability of sedimentation rates and drying of surface peat have been attributed to the effects of fire, grazing, and road and track construction (Rowe 1970, Wimbush and Costin 1983, Dodson *et al.* 1994, Lawrence 1995, O'Donnell 2008). Peatlands in particular are likely to have been severely impacted by the detrimental effects of cattle grazing, fire and drought.

The response of vegetation to fire is contingent on a number of influences. These include the nature and extent of the most recent fire, previous fire history and a range of environmental and biological factors (Whelan *et al.* 2002). Significant changes to alpine vegetation which are linked to fire frequency have rarely been recorded but many species can respond rapidly to fire and quickly return to their pre-fire cover or replace some fire sensitive species within a relatively short period of time (Costin 1983, Kirkpatrick and Dickinson 1984, Banks 1989, Wahren *et al.* 1999a).

In high altitude peatland, fire can potentially exert a significant influence over vegetation structure, function and floristic composition, depending on the severity of the burn and the moisture content of surface peat layers (Tallis 1983, Whinam *et al.* 2003). Resprouting species are dominant after fire and tend to maintain their dominance for a number of years, while many obligate seeders are slower to establish (Wahren and Walsh 2000, Walsh and McDougall 2004). Peatlands gradually recover after fire although some changes, including the loss of rare or keystone species such as *Sphagnum* may be permanent. Other effects of burning include disruption to hydroseral processes and the formation of hummocks and hollows (Ashton and Hargreaves 1983, Campbell 1983), increased sedimentation, loss of peat, long-term nutrient losses, and hydraulic changes and lowered water tables resulting from stream incisement (Tallis 1983, Whinam and Chilcott 2002). Invasion by shrubs and opportunistic species onto increasingly dry substrates (Costin 1954, Binder 1978, Tallis 1983, Wahren *et al.* 1999b; Wahren and Walsh 2000) increases the flammability of peatlands and the likelihood of recurrent fire.

Less well understood are changes in floristic composition, structure and function in alpine or sub-alpine vegetation subjected to recurrent fires (Kirkpatrick and Dickinson 1984). However, recurrent fire is considered a threat to conservation values and damaging to alpine and sub-alpine vegetation (Wahren *et al.* 1999a, Wahren and Walsh 2000). Studies to date are confined to Tasmania and have focussed on long-lived gymnosperms (Kirkpatrick and Dickinson 1984) which are largely absent from alpine and sub-alpine environments on the Australian mainland. The only other studies of recurrent fire in peatland addressed vegetation change at Mt Buffalo (Coates *et al.* 2006, Coates and Walsh 2010). These studies found that floristic composition was correlated with bulk density, moisture, and organic content of peat and time elapsed since the last fire. There was also a decline in species evenness and diversity between 1982 and 2008 and a trend toward compositional and structural simplification.

Two types of peatlands which occur in cold drainage depressions and valleys are common within the study area. "Hillside or Hillslope peatlands" (Shannon and Morgan 2007, Lawrence *et al.* 2009) occupy areas of drainage depressions at the confluence of unchannelled tributaries on slopes >5°. "Valley floor peatlands" (Shannon and Morgan 2007) occupy slopes <5° along defined streamlines (Lawrence *et al.* 2009). Channels may be either discontinuous with pools, or continuous and meandering (Lawrence *et al.* 2009). "Plateau peatlands" have also been mapped at Lake Mountain and Mt Baw Baw. These are essentially unchannelled peatlands on slopes <5° but are relatively rare (Lawrence *et al.* 2009). Peatlands are most extensive at Mt Baw Baw (~560 ha), with the largest of these covering ~35–40 ha (Lawrence *et al.* 2009, Tolsma and Shannon 2009) and are the least extensive at Lake Mountain (~32 ha). At Mt Buffalo, peatlands cover ~130 ha (Lawrence *et al.* 2009). The largest peatlands at Lake Mountain and Mt Buffalo are ~6–10 ha (Lawrence *et al.* 2009, Tolsma and Shannon 2009).

Species characteristic of hummocks (raised areas formed mainly from growth of *Sphagnum*) are *Baeckea gunniana, Empodisma minus, Richea continentis, Epacris paludosa, E. gunnii* (Mt Buffalo), *E. petrophila* (Mt Baw Baw), *Callistemon pityoides* (Mt Baw Baw, Lake Mountain), *Olearia algida* (Mt Baw Baw, Lake Mountain), *Ozothamnus hookeri* (Lake Mountain), *Poa costiniana, Astelia alpina, Erigeron paludicola, Gonocarpus micranthus, Asperula gunnii, Nertera granidensis* and *Carpha nivicola* (Ashton and Hargreaves 1983, Kershaw *et al.* 1993, McDougall and Walsh 2007, Shannon and Morgan 2007, Tolsma and Shannon 2009, Coates and Walsh 2010).

Hollows (inter-hummock depressions) are occupied by small species tolerant of inundation. Typical of these are *Isolepis aucklandicus, Myriophyllum pedunculatum, Oreobolus oxycarpus, O. distichophylla, Carex gaudicahaudiana, Coprosma moorei, Diplaspis nivis, Herpolirion novae-zelandiae* and *Sphagnum novozelandicum* (Ashton and Hargreaves 1983, Coates and Walsh 2010). *Empodisma minus* and *Baeckea gunniana* are more abundant at Mt Buffalo relative to the other two sites (Walsh *et al.* 1984). This difference is expected to be even more marked in the current study as both species have increased at Mt Buffalo between 1982 and 2008 (Coates and Walsh 2010).

## 3.1 Methods

### 3.1.1 Study design and field methods

The study used a space-for-time substitution design describing changes in vegetation at sites burnt at different times within the last 72 years but observed at the same time (Morrison *et al.* 1995; Watson and Wardell-Johnson 2004). We considered that peatlands across the three mountains, Lake Mountain, Mt Buffalo and Mt Baw Baw, were analogues in the environmental and species compositional space. A Geographic Information System (GIS, Arcview 3.3) was used to examine the extent of fire on all three mountains. At Mt Buffalo, sites burnt within the 2003 and 2006 fire perimeters were confirmed in previous studies as having been burnt or not (Coates *et al.* 2006, Coates and Walsh 2010; Table 6). The extent of the fire in 2009 at Lake Mountain was examined from aerial photography (Department of Sustainability and Environment unpub. data). Mt Baw Baw was considered fully recovered after the last fire in 1939 (Zylstra 2006) and was used as the benchmark for recovery.

Fieldwork was conducted in December 2010 at Mt Buffalo NP, in February/March 2011 at Lake Mountain NP, Mt Baw Baw NP and again at Mt Buffalo. Twenty quadrats were surveyed at Lake Mountain and Mt Baw Baw (Appendices 4 and 5). At Mt Buffalo, a subset of 22 peatland quadrats established in 2005 (Coates *et al.* 2006) and re-surveyed in March 2008 (Coates and Walsh 2010) were selected (Appendix 6). All Mt Buffalo quadrats surveyed were last burnt in 2003 but had experienced a range of fire frequencies. Few Mt Buffalo peatland quadrats were burnt in the 2006 fire, which generally lacked the severity to ignite wet areas. As far as practicable, quadrats were sampled at similar elevations on each mountain (Table 6).

Floristic data were recorded from 4 m × 5 m quadrats (Figures 24–26), consistent with previous surveys (Walsh *et al.* 1984, Coates *et al.* 2006, Coates and Walsh 2010). At Mt Buffalo, previously surveyed quadrats were re-located using GPS co-ordinates and from photographs (Walsh and Coates 2010). Quadrats on all three mountains were photographed/re-photographed in 2010/2011.

Percentage live cover was visually assessed and recorded for all vascular plant species and *Sphagnum* in each quadrat. Species diversity (species richness weighted by species abundance), species evenness [diversity/log(species richness)] and the relative mean abundance of species were also derived from floristic data.

Table 6. Mean elevation, fire history and number of quadrats surveyed on each mountain.

| Survey site | Mean elevation (m ± SD) | Fire history | Fire frequency | Number of quadrats surveyed |
|---|---|---|---|---|
| Lake Mountain | 1,403 ± 18 | Burnt 1939, 2009 | 2 | 20 |
| Mt Baw Baw | 1,468 ± 35 | Burnt 1939 | 1 | 20 |
| Mt Buffalo | 1,373 ± 54 | Burnt 2003 | 1 | 6 |
| Mt Buffalo | 1,449 ± 54 | Burnt 2003, 1972, or 2003, 1985 | 2 | 7, 6 |
| Mt Buffalo | 1,436 ± 47 | Burnt 2003, 1985, 1972 | 3 | 3 |

The following variables were collected in each quadrat:

1. Geographic position (Australian Map Grid co-ordinates, GDA 94) and elevation (m): measured using a Magellan Meridian or Garmin eTrex GPS.
2. Slope angle: measured using a Suunto clinometer.
3. Litter cover: percentage cover of fixed and loose litter estimated visually.
4. Exposed bare ground cover: percentage cover estimated visually.
5. Soil depth: mean depth measured at 5 points in each quadrat using a steel probe (3 mm × 1 m).
6. Aspect: 1 = NW; 2 = N and W; 3 = NE and SW; 4 = S and E; 5 = SE, representing the estimated evaporation gradient in south-eastern Australia (Kirkpatrick and Nunez 1980; Kirkpatrick and Bridle 1998).
7. Soil moisture and organic matter: estimated as percentage loss on ignition (LOI, Dean 1974). Soil samples were taken at 10 cm depth.

Keystone species were identified as peatland endemics always present in the vegetation and which control its structure and function (Hope *et al.* 2011). The mean height of keystone species in major life form groups was calculated by measuring 5–10 individuals common to all three mountains. These were *Baeckea gunniana* (shrub), *Epacris gunnii* or *E. paludosa* (shrub), *Empodisma minus* (graminoid) and *Poa costiniana* (perennial grass). Mean height of *Richea continentis* (shrub) and *Sphagnum cristatum* hummocks (moss) was also measured where present.

Percentage live cover and height of all life form groups [shrubs, perennial forbs, perennial grasses and graminoids (non-Poaceae)] were derived from floristic data.

Botanical nomenclature follows Walsh and Stajsic (2007) and conservation status for rare or threatened species follows the Department of Sustainability and Environment (2005).

Figure 24. Quadrat layout, Lake Mountain.

Figure 25. Quadrat layout, Mt Buffalo.


Figure 26. Quadrat layout, Mt Baw Baw.

### 3.1.2 Analytical methods

To examine compositional similarities between mountains, floristic data were ordinated using non-metric multidimensional scaling (NMDS) from several random starts. Percent cover of each species was allocated to one of seven cover classes and the mid-point of each class was used in the analyses, to overcome any bias that may have arisen from data having been collected by four recorders (Table 7). Compositional dissimilarities were calculated using the Bray-Curtis co-efficient (Faith *et al.* 1987). To determine an appropriate dimensionality, minimum stress values were evaluated and plotted against the number of scaling dimensions (Clarke 1993). The solution was chosen that provided the most reduction in stress (McCune and Grace 2002). Species that occurred in fewer than 5% of quadrats were excluded from the analysis.

Table 7. Cover classes and mid-points for floristic data.

Classes are derived from percentage live cover recorded in quadrats.

| Class | Range (%) | Mid-point |
|---|---|---|
| 0 | 0 | 0 |
| 1 | <1 | 0.5 |
| 2 | 1–5 | 3 |
| 3 | 5–25 | 15 |
| 4 | 30–50 | 40 |
| 5 | 55–75 | 65 |
| 6 | 80–90 | 85 |
| 7 | 95–100 | 97.5 |

The relationship between composition and environmental and/or structural variables was investigated by fitting directional vectors of maximum correlation through the ordination space (Kantvilas and Minchin 1989). Percent compositional similarity between post-fire stages was represented graphically by plotting the NMDS ordination scores and significant vectors.

Ecosystem properties common to all three mountains were identified and compared graphically (Westman 1986). These were species richness, diversity and evenness; height of keystone species, the cover of major life form groups and bare ground cover. These were compared to Mt Baw Baw to estimate the likely direction of recovery in the absence of fire and to the Mt Buffalo data to estimate likely trends in the event of recurrent fire, using interval plots.

Two-way factorial Generalized Linear Models (GLM) using a normal error structure with an identity link function were used to test for the influence of cover and height of dominant species *Baeckea gunniana* and *Empodisma minus* on species richness, diversity and evenness.

Data manipulation, statistical analyses and graphics were conducted using Minitab® 16.1.0 Statistical Software (Minitab Inc. 2010) and Program R Version 2.13.1 (R Development Core Team 2011) with packages MASS (Venables and Ripley 2002) and Vegan version 2.0-0 (Oksanen *et al.* 2011).

## 3.2 Results

### 3.2.1 Floristic composition

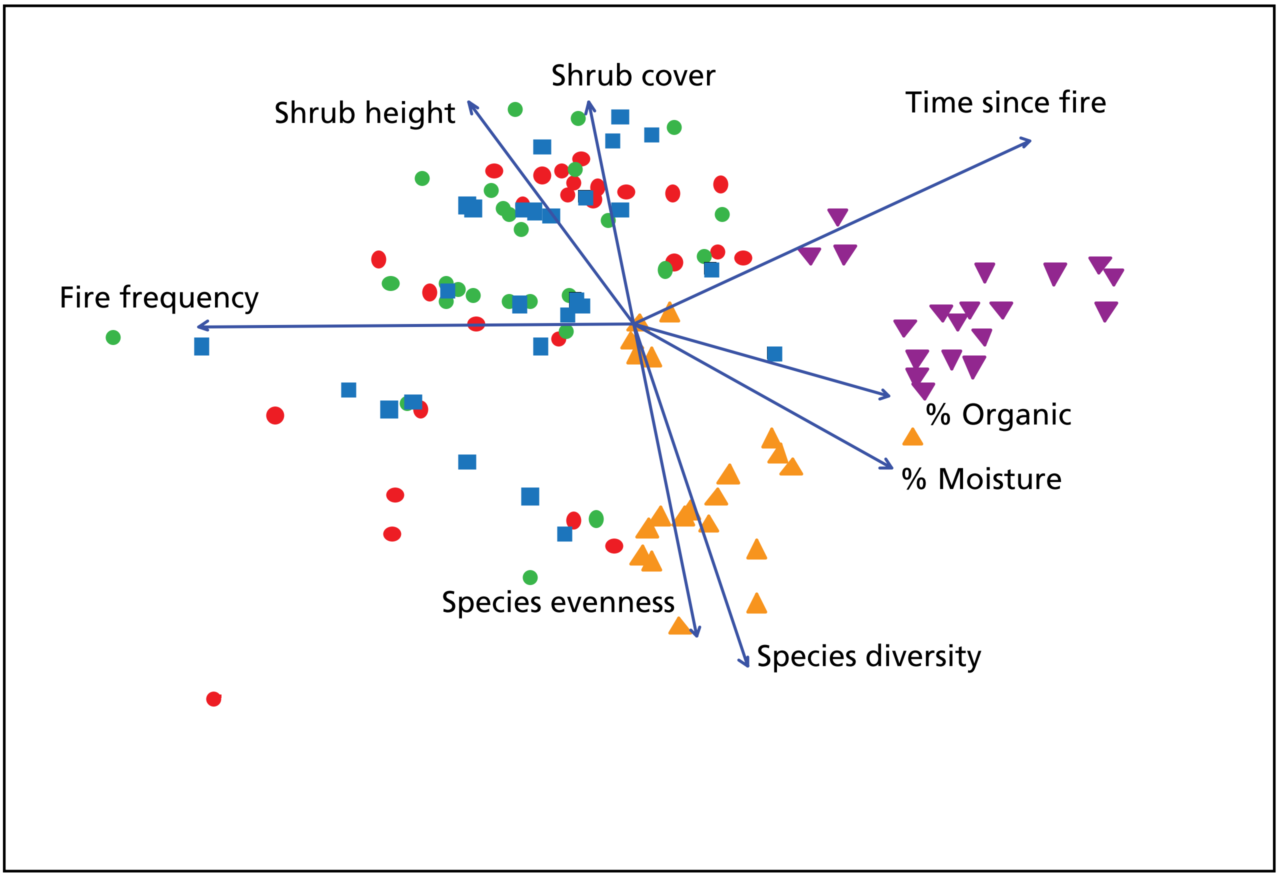
NMDS attained a minimum stress of 0.16 in two dimensions, decreasing to 0.12 in three dimensions but adding little to interpretation of the results. Eighty-two species which were recorded from fewer than 5% of quadrats were excluded from the ordination. Variation in floristic composition was correlated with vectors for time since fire, fire frequency, the height and cover of shrubs ($P$ <0.001), species diversity and evenness ($P$ <0.001) and moisture and organic content of peat ($P$ <0.05; Figure 27).

Baw Baw quadrats were separated from all others (Figure 27). As expected, these quadrats were positioned in the ordination space representing long unburnt vegetation and floristic composition was correlated with high peat organic and moisture content. Quadrats from Lake Mountain were positioned relatively closely to the Mt Baw Baw quadrats although formed a distinct group, with some overlap between these quadrats and some Mt Buffalo quadrats.

Two years after fire, peat at Lake Mountain had relatively high moisture and organic content, low shrub cover and low shrub height. Regeneration in the Lake Mountain peatlands was generally more diverse and species cover was more equitably distributed compared to two years after fire at Mt Buffalo (Figure 27). At Mt Buffalo, low species diversity and evenness were associated with vegetation dominated by shrubs on the driest and least organic soils. Floristic composition between survey years was similar at Mt Buffalo, indicated by poor separation of quadrats in the ordination space. However, quadrats were also widely dispersed indicating a degree of compositional and structural variation along gradients of fire frequency and time since fire (Figure 27).

Figure 27. Two-dimensional site ordination (NMDS) showing the distribution of quadrats in the ordination space and fitted vectors of maximum correlation.

Quadrats are labelled by site: ● Mt Buffalo 2005; ● Mt Buffalo 2008; ■ Mt Buffalo 2011; ▲ Lake Mountain; ▼ Baw Baw. Vector length is proportional to the strength of the correlation.




## 3.2.2 Species richness, diversity and evenness

Richness and diversity were highest two years after fire at Lake Mountain and in long-unburnt vegetation at Mt Baw Baw (Figures 28 and 29), although these differences were attributable to compositional and structural dissimilarities owing to different time since fire and are not directly comparable. Species richness and diversity were significantly lower at Mt Buffalo in all three survey years compared to the other two mountains ($P$ <0.05; Figures 28 and 29).

Vegetation was most even at Lake Mountain, significantly more so compared to Mt Buffalo at the same time since fire ($P$ <0.05). This difference became more pronounced at seven years after fire as evenness declined at Mt Buffalo (Figure 30).

Figure 28. Mean species richness with time since fire.

Error bars are 95% CI for the mean. LM=Lake Mountain; MB = Mt Buffalo; BB = Mt Baw Baw.

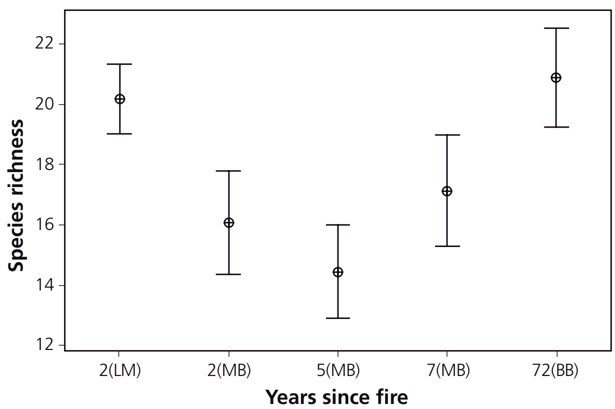


Figure 29. Mean species diversity with time since fire.

Error bars are 95% CI for the mean. LM=Lake Mountain; MB = Mt Buffalo; BB = Mt Baw Baw.

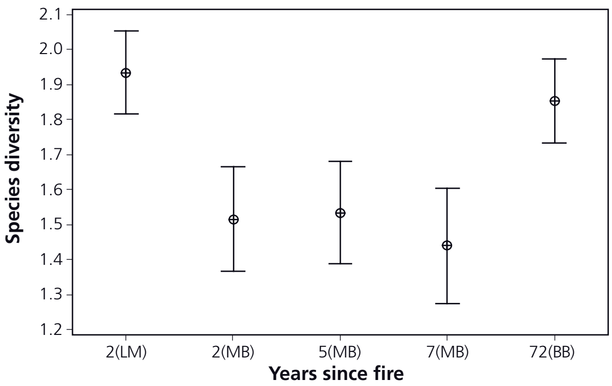
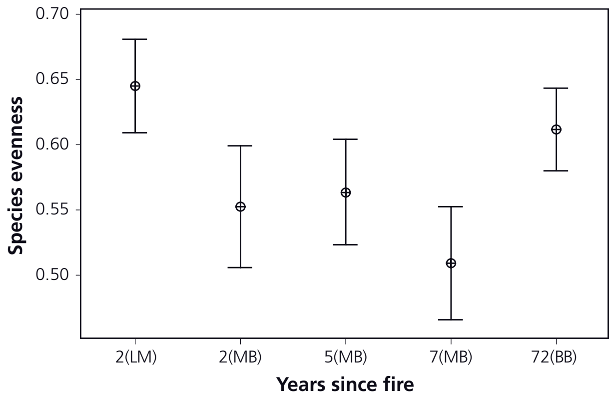



Figure 30. Mean species evenness with time since fire.

Error bars are 95% CI for the mean. LM=Lake Mountain; MB = Mt Buffalo; BB = Mt Baw Baw.




Cover of the graminoid *Empodisma minus*, the shrub *Baeckea gunniana,* the lily *Astelia alpina* and the moss *Sphagnum cristatum* explained most of the difference in species evenness between mountains (Table 3). *Empodisma minus* accounted for 50% of vegetation cover at Mt Buffalo and 40% at Lake Mountain but was minor (14%) in long-unburnt peatlands at Mt Baw Baw.

Two years after fire, cover of *Baeckea gunniana* accounted for 13% of vegetation cover at Mt Buffalo compared to 2% at Lake Mountain, and remained high (20%) at Mt Buffalo over the next six years. Cover of other shrubs, for example *Epacris gunnii* and *E. paludosa*, was only a small proportion of total vegetation cover at all three mountains. *Astelia alpina* was better represented in the vegetation at Lake Mountain two years after fire compared to either Mt Buffalo or Mt Baw Baw (Table 8). Other main differences were a higher representation of peatland endemics in long unburnt peatlands at Mt Baw Baw (e.g. *Caltha introloba*, *Carpha* spp. and *Richea continentis*) and proportionally higher cover of post-fire colonisers two years after fire at Mt Buffalo (e.g. *Asperula gunnii, Carex breviculmis, Gonocarpus micranthus*), although these differences were reasonably minor.

Table 8. Relative mean cover of some frequently recorded species.

MB 2005=Mt Buffalo 2005; MB 2008=Mt Buffalo 2008; MB 2011=Mt Buffalo 2010/2011; LM=Lake Mountain 2011; BB 2011 = Mt Baw Baw 2011.

| Species | MB 2005 | MB 2008 | MB 2011 | LM 2011 | BB 2011 |
|---|---|---|---|---|---|
| *Asperula gunnii* | 0.05 | 0.01 | 0.008 | 0.02 | 0.001 |
| *Astelia alpina* | 0.005 | 0.007 | 0.005 | 0.13 | 0.05 |
| *Baeckea gunniana* | 0.13 | 0.2 | 0.2 | 0.02 | 0.02 |
| *Caltha introloba* | 0.002 | 0.002 | 0.004 | 0 | 0.04 |
| *Carex breviculmis* | 0.004 | 0.004 | 0 | 0 | 0 |
| *Carpha alpina* | 0 | 0 | 0 | 0 | 0.03 |
| *Empodisma minus* | 0.5 | 0.5 | 0.5 | 0.4 | 0.14 |
| *Epacris gunnii* | 0.02 | 0.03 | 0.03 | 0 | 0 |
| *Epacris paludosa* | 0.008 | 0.01 | 0.01 | 0.02 | 0.09 |
| *Gonocarpus micranthus* | 0.04 | 0.008 | 0.006 | 0.01 | 0.002 |
| *Poa costiniana* | 0.05 | 0.06 | 0.06 | 0.06 | 0.01 |
| *Richea continentis* | 0.003 | 0.002 | 0.005 | 0.007 | 0.05 |
| *Sphagnum cristatum* | 0.08 | 0.05 | 0.06 | 0.1 | 0.5 |

### 3.2.3 Influence of dominant species

Cover and height of *Empodisma* had a significant influence on species diversity (Table 9). Diversity was predicted to decline with an increase in *Empodisma* cover and height. *Empodisma* height had only a weakly significant negative relationship with species richness (Table 10). Species evenness was also predicted to decline with an increase in *Empodisma* cover (Table 11).

*Baeckea gunniana* height was a weakly significant influence on evenness ($P$ <0.05) but *Baeckea* cover had no significant influence of species richness, diversity or evenness ($P$ >0.05; Table 12).

Table 9. Results for generalised linear modelling (GLM). Significance of *Empodisma minus* height and cover as predictors of species diversity.

| Coefficients: | Estimate | Std. Error | T value | P |
|---|---|---|---|---|
| (Intercept) | 2.32 | 0.15 | 15.8 | <0.0001 |
| *Empodisma* cover | 2.31 | 0.004 | -3.6 | <0.0006 |
| *Empodisma* height | 2.21 | 0.005 | -2.3 | 0.02 |
| *Empodisma* height*cover | 2.32 | 0.0001 | 1.2 | NS |

Table 10. Results for generalised linear modelling (GLM). Significance of *Empodisma minus* height and cover as predictors of species richness.

| Coefficients: | Estimate | Std. Error | T value | P |
|---|---|---|---|---|
| (Intercept) | 23.5 | 1.9 | 12.384 | <0.0001 |
| *Empodisma* cover | 23.4 | 0.05 | -1.713 | NS |
| *Empodisma* height | 23.3 | 0.06 | -2.315 | 0.02 |
| *Empodisma* height*cover | 23.5 | 0.002 | 0.818 | NS |

Table 11. Results for generalised linear modelling (GLM). Significance of *Empodisma minus* height and cover as predictors of species evenness.

| Coefficients: | Estimate | Std. Error | T value | P |
|---|---|---|---|---|
| (Intercept) | 0.75 | 0.04 | 18.9 | <0.0001 |
| *Empodisma* cover | 0.73 | 0.001 | -3.3 | 0.001 |
| *Empodisma* height | 0.73 | 0.001 | -1.5 | NS |
| *Empodisma* height*cover | 0.75 | 0.00003 | 0.5 | NS |

Table 12. Results for generalised linear modelling (GLM). Significance of *Baeckea gunniana* height and cover as predictors of species evenness.

| Coefficients: | Estimate | Std. Error | T value | P |
|---|---|---|---|---|
| (Intercept) | 0.07 | 0.04 | 16.2 | <0.0001 |
| *Baeckea* cover | 0.07 | 0.002 | -1.3 | NS |
| *Baeckea* height | 0.06 | 0.003 | -2.2 | 0.03 |
| *Baeckea* height*cover | 0.07 | 0.0001 | 0.9 | NS |

Table 13. Mean percent cover of *Epacris paludosa, Richea continentis* and *Baeckea gunniana.*

| | % Cover MB 2005 | % Cover MB 2008 | % Cover MB 2011 | % Cover LM 2011 | % Cover BB 2011 |
|---|---|---|---|---|---|
| *Epacris paludosa* | 1 | 1 | 1 | 1 | 12 |
| *Richea continentis* | <1 | <1 | <1 | <1 | 6 |
| *Baeckea gunniana* | 11 | 20 | 20 | 1 | 2 |

### 3.2.4 Influence of time since fire

Compared to Lake Mountain, overall shrub cover was higher at Mt Buffalo two years after fire (Figure 31). Five years after fire, shrubs attained the highest cover then appeared to decline over the next couple of years (Figure 31). Overall shrub cover in long-unburnt vegetation at Mt Baw Baw was slightly lower (Figure 31) but cover of *Epacris paludosa* and to a lesser extent *Richea continentis* at Mt Baw Baw was higher than at the other two mountains and cover of *B. gunniana* was substantially lower (Table 13). *Baeckea gunniana* resprouted after fire and grew substantially taller up to seven years post-fire (Mt Buffalo, Figure 32). Growth of *Epacris* spp. and *Richea continentis* were low in the years immediately after fire but grew substantially taller five to seven years after fire (Mt Buffalo, Figure 32). *Richea continentis* was particularly slow to recover at Mt Buffalo but once established was beginning to form dense thickets by 2011 (Figure 34). However, all of these species were significantly shorter at Mt Baw Baw (Figure 32).

Forb cover was generally low across the study area (~5-10%; Figure 31). At Mt Buffalo, these were most abundant two years after fire then decreased in the following years. The main driver of this trend was a decline in post-fire colonisers *Asperula gunnii, Gonocarpus micranthus* and *Scaevola hookeri.*

The most increase in graminoid and grass cover (mostly *Empodisma minus* and *Poa costiniana*) occurred up to two years post-fire (Figure 31). Perennial grass cover remained sparse across the study area regardless of time since fire (Figure 31). At Mt Buffalo, both species steadily increased in height over time (Figure 33). At Lake Mountain, *P. costiniana* was roughly in the range recorded at Mt Buffalo 1–2 years after fire but *E. minus* was taller than at Mt Buffalo (Figure 33). Graminoids and grasses were least dominant at Mt Baw Baw (Figure 31), where *E. minus* was a minor, low growing species and accounted for only a small proportion of the vegetation cover (<15%; Table 3, Figure 33).

Cover of *Sphagnum cristatum* was very low at Mt Buffalo and Lake Mountain two to seven years after fire but very high in long-unburnt vegetation at Mt Baw Baw (Figure 33). Mean hummock height of *Sphagnum cristatum* was low at Lake Mountain and at Mt Buffalo (5–6 cm), regardless of time since fire. *Sphagnum* hummocks were well developed in long-unburnt peatlands at Mt Baw Baw with a mean height of 60 cm.

Bare ground cover at Lake Mountain was similar to Mt Buffalo two years and five years after fire, although mean values were only around 10%. Bare ground cover declined at seven years post-fire to around 5% (Figure 33). Bare ground cover at Mt Baw Baw was only slightly lower, indicating good recovery in this respect at Mt Buffalo.

Figure 31. Mean cover of life form groups.

Error bars are 95% CI for the mean. LM=Lake Mountain; MB = Mt Buffalo; BB = Mt Baw Baw.

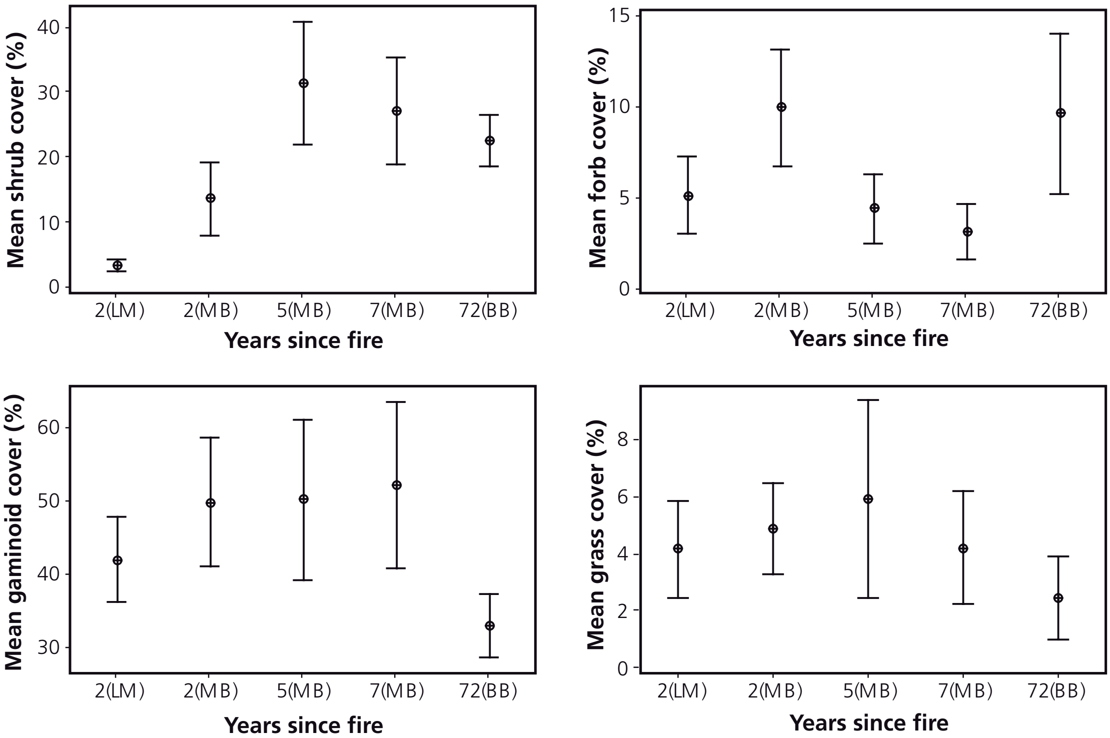



Figure 32. Mean height of shrubs with time since fire.

Error bars are 95% CI for the mean. LM=Lake Mountain; MB = Mt Buffalo; BB = Mt Baw Baw.

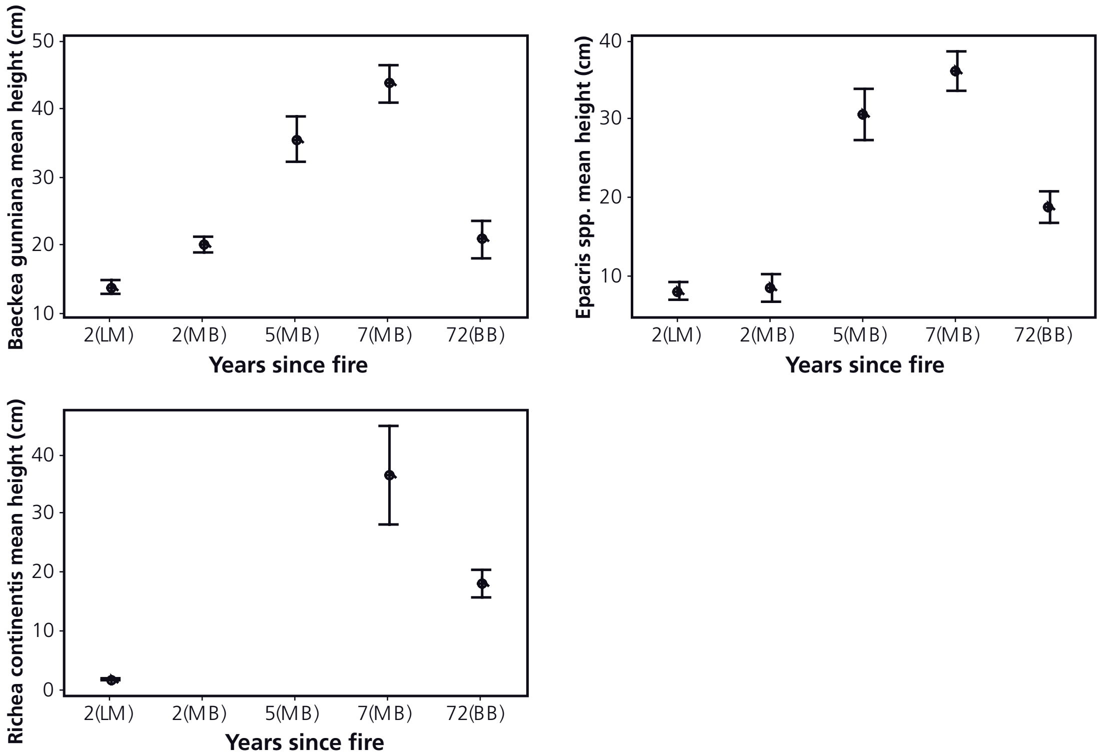
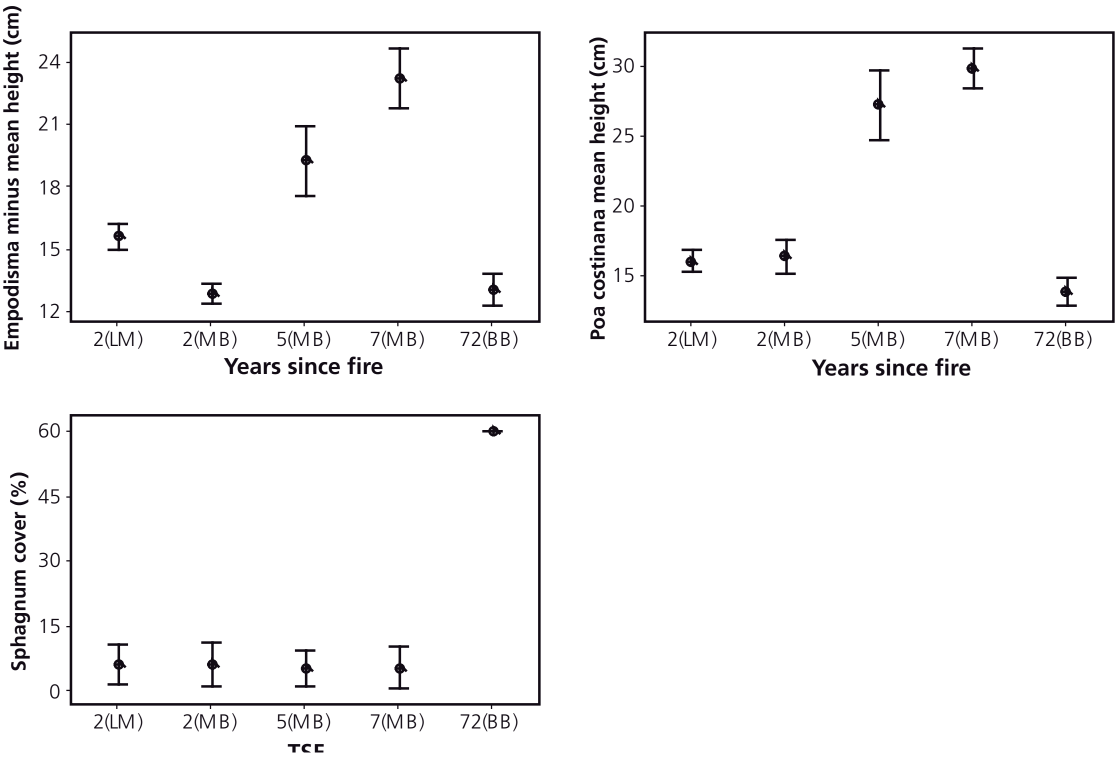


Figure 33. Mean heights of *Empodisma minus* and *Poa costiniana*, and mean cover of *Sphagnum cristatum* and bare ground with years since fire.

Error bars are 95% CI for the mean. LM=Lake Mountain; MB = Mt Buffalo; BB = Mt Baw Baw.




### 3.2.5 Rare or threatened species

Twenty-nine rare species and one threatened (vulnerable) species (*Isolepis gaudichaudiana*) were recorded in the five surveys at the three mountains. The most rare species were recorded at Mt Baw Baw (17) and the least at Lake Mountain (7). Species most frequently recorded at Lake Mountain were *Brachyscome obovata, Carex blakei* and *Oreobolus oxycarpus* (Table 14).

At Mt Baw Baw, species uniquely recorded for the mountain were *Carpha alpina, C. nivicola* and *Lycopodium scariosum*. Mean species cover was highest at Mt Baw Baw but still <0.5% overall. *Caltha introloba, Carpha alpina* and *Rytidosperma nivicolum* were recorded in most (75–90%) quadrats.

At Mt Buffalo, 13 rare species were recorded in 2005 and 2008 and 15 in 2011. The additional two species in 2011 were *Erigeron nitidus* and *Herpolirion novae-zelandiae*. Some species recorded in 2005 were not seen in 2008 but re-appeared in 2011. These were *Deyeuxia carinata, Ranunculus gunnianus, Rytidosperma nivicolum* and *Tetrarrhena turfosa.* After the 2006 fire, some species were less frequently recorded than after the 2003 fire but by 2010/2011 had re-established (e.g. *Aciphylla simplicifolia, Caltha introloba*). In general, the frequency with which rare species were recorded in quadrats at Mt Buffalo increased slightly or remained stable over time. Mean cover of rare or threatened species was very low (<0.2%) but remained roughly the same over the three surveys.

Figure 34. Recovery of *Richea continentis,* seven years after fire at Mt Buffalo.

Table 14. Percent frequency of occurrence of rare or threatened species.

| Species | Status | Frequency MB 2005 | Frequency MB 2008 | Frequency MB 2011 | Frequency LM 2011 | Frequency BB 2011 |
|---|---|---|---|---|---|---|
| *Aciphylla simplicifolia* | r | 42 | 27 | 42 | 0 | 0 |
| *Agrostis australiensis* | r | 0 | 4 | 0 | 0 | 0 |
| *Agrostis muelleriana* | r | 0 | 12 | 0 | 0 | 0 |
| *Brachyscome obovata* | r | 0 | 0 | 0 | 76 | 10 |
| *Caltha introloba* | r | 31 | 15 | 35 | 0 | 75 |
| *Carex blakei* | r | 69 | 77 | 81 | 57 | 0 |
| *Carex jackiana* | r | 4 | 8 | 4 | 0 | 0 |
| *Carpha alpina* | r | 0 | 0 | 0 | 0 | 90 |
| *Carpha nivicola* | r | 0 | 0 | 0 | 0 | 25 |
| *Celmisia tomentella* | r | 54 | 42 | 50 | 0 | 15 |
| *Coprosma moorei* | r | 23 | 23 | 23 | 14 | 10 |
| *Coprosma perpusilla* subsp. *perpusilla* | r | 0 | 0 | 0 | 0 | 35 |
| *Derwentia nivea* | r | 0 | 0 | 0 | 5 | 0 |
| *Deyeuxia carinata* | r | 0 | 12 | 0 | 0 | 5 |
| *Diplaspis nivis* | r | 15 | 19 | 31 | 0 | 5 |
| *Epacris petrophila* | r | 0 | 0 | 0 | 0 | 45 |
| *Erigeron nitidus* | r | 0 | 0 | 4 | 0 | 10 |
| *Erigeron tasmanicus* | r | 0 | 0 | 0 | 0 | 15 |
| *Herpolirion novae-zelandiae* | r | 0 | 0 | 8 | 5 | 10 |
| *Isolepis gaudichaudiana* | v | 0 | 0 | 0 | 33 | 0 |
| *Lycopodium scariosum* | r | 0 | 0 | 0 | 0 | 5 |
| *Oreobolus oxycarpus* subsp. *oxycarpus* | r | 15 | 19 | 27 | 62 | 0 |
| *Oreobolus pumilio* subsp. *pumilio* | r | 0 | 0 | 0 | 0 | 20 |
| *Plantago alpestris* | r | 0 | 0 | 0 | 0 | 20 |
| *Pultenaea tenella* | r | 46 | 50 | 46 | 0 | 0 |
| *Ranunculus gunnianus* | r | 4 | 0 | 4 | 0 | 5 |
| *Rhytidosporum inconspicuum* | r | 4 | 0 | 8 | 0 | 0 |
| *Rytidosperma nivicolum* | r | 0 | 0 | 0 | 0 | 85 |
| *Tetrarrhena turfosa* | r | 4 | 0 | 4 | 0 | 0 |
| *Viola fuscoviolacea* | r | 31 | 42 | 35 | 0 | 0 |
| **Total number of rare spp.** | | **13** | **13** | **15** | **7** | **18** |
| **Mean cover rare spp. (%)** | | **0.17** | **0.20** | **0.14** | **0.10** | **0.46** |

Table 15. Rare or threatened species common to the three mountains.

| | Lake Mountain | Mt Baw Baw |
|---|---|---|
| Mt Baw Baw | *Brachyscome obovata, Coprosma moorei, Herpolirion novae-zelandiae* | |
| Mt Buffalo | *Carex blakei, Coprosma moorei, Herpolirion novae-zelandiae, Oreobolus oxycarpus* | *Caltha introloba, Celmisia tomentella, Coprosma moorei, Deyeuxia carinata, Diplaspis nivis, Erigeron nitidus, Herpolirion novae-zelandiae, Ranunculus gunnianus,* |

*Coprosma moorei* was the only species common to all three mountains but was most frequently recorded at Mt Buffalo (Table 15). Mt Baw Baw and Mt Buffalo had the most species (8) in common (Table 15). There were only three species in common at the geographically closest mountains Lake Mountain and Mt Baw Baw (Table 15). *Brachyscome obovata* was more common at Lake Mountain, frequency of *Coprosma moorei* was roughly similar at these mountains but *H. novae-zelandiae* was more frequent in long-unburnt vegetation at Mt Baw Baw (Table 15). Four species were common to both Lake Mountain and Mt Buffalo (Table 15). *Carex blakei* and *Coprosma moorei* were recorded at roughly the same frequency two years after fire at both of these mountains whilst *H. novae-zelandiae* and *O. oxycarpus* were far more common at Lake Mountain.

## 3.3 Discussion

### 3.3.1 What is the rate and extent of post-fire vegetation recovery and the timeframe for recovery?

Vegetation recovery two years after fire was comparable between Lake Mountain and Mt Buffalo in many respects (Figures 35 and 38). However, differences in peat properties and species evenness suggest that floristic composition and vegetation structure at Lake Mountain is more likely to eventually approach long-unburnt vegetation, identified as the benchmark for recovery at Mt Baw Baw (Figure 39). In contrast, frequently-burnt vegetation at Mt Buffalo on drier, less organic soils was dominated by *Empodisma minus* and *Baeckea gunniana* under a regime of recurrent fire (e.g. Figures 35 to 37). Low species diversity and evenness seven years after fire and dominance of *Empodisma minus* and *Baeckea gunniana* at Mt Buffalo suggest that recovery of these peatlands to a benchmark or near benchmark state is likely to be slow, even in the absence of fires.

High growth of *Empodisma minus* was recorded in the early stages after fire at both Lake Mountain and Mt Buffalo. *Empodisma* is a highly competitive species (Wahren and Walsh 2000), even at sites that have been severely burnt. It produces extensive rhizomatous mats that resprout from perennating buds after fire and has an important role in stabilising burnt peat and impeding stream flow (Coates and Walsh 2010, Hope *et al.* 2011). After severe scorching, these buds are thinned so that there is reduced intra-specific competition for shoot growth. This may explain the high species evenness recorded at Lake Mountain, in comparison to Mt Buffalo. Cover of *Empodisma* was lower at Mt Buffalo two years after fire than at Lake Mountain but still accounted for the relatively highest proportion of species cover. This initially lower cover may be a consequence of the drought conditions which prevailed at the time of the 2005 survey. The amount of exposed bare ground also decreased during this period as *E. minus* cover increased. *Empodisma* continued to dominate the vegetation at Mt Buffalo over the next five years and the results from this and previous studies at Mt Buffalo (Coates and Walsh 2010) suggest that *E. minus* and the co-occurring shrub *Baeckea gunniana* are likely to increase their dominance over time with recurrent fire.

Vegetation began to open out five years after fire at Mt Buffalo. This was attributed initially to a decrease in the abundance of widespread shrub species *Baeckea gunniana*, which is favoured by drier soils (Costin 1954, Kershaw *et al.* 1993, Whinam and Chilcott 2002, Whinam and Hope 2005). However, shoot extension was substantial up to seven years post-fire during which time this species grew rapidly. Other shrubs which were initially slow to establish also grew quickly during this period but accounted for little of the overall vegetation cover (e.g. *Epacris* spp.). In long unburnt vegetation (Mt Baw Baw), shrub cover was slightly lower than at Mt Buffalo at seven years since fire, but was associated with well-developed *Sphagnum* hummocks, comparatively lower graminoid and grass cover and higher cover of forbs. This is undoubtedly a consequence of the long period that had elapsed since the last fire in 1939.

Fire history appears to have had a significant influence in determining current levels of species richness, diversity and evenness at Mt Buffalo. In comparison, there was little difference in these variables between Lake Mountain and Mt Baw Baw, although floristic composition and vegetation structure differed substantially and as such direct comparisons are not valid. Species have clearly accumulated in long-unburnt vegetation at Mt Baw Baw, but at Lake Mountain the currently high levels of species richness and diversity are a result of establishment of early post-fire colonisers and are likely to decline within the next few years. However, general similarities in floristic composition and peat properties between Mt Baw Baw and Lake Mountain suggest that if Lake Mountain remains unburnt over the coming decades, species richness and diversity can be expected to eventually increase.

Some of the vegetation patterns recorded at Mt Buffalo can probably be attributed to disturbance that occurred well before 1972. Evidence of soil erosion, road and track construction and past grazing impacts were recorded by Rowe (1970). However, he described far greater ecological changes in bogs, attributed to the detrimental effects of cattle grazing and fire. Fire control lines in the form of mineral earth breaks have been relatively extensive at Mt Buffalo at least since 1985 and these may also have contributed to the current condition of peatlands. The other two mountains have had a more or less similar grazing history but infrastructure development is less extensive.

It is likely that there will be an eventual increase in the abundance of woody perennials at Mt Buffalo as obligate seeders continue to establish (e.g. *Epacris* spp., *Comesperma retusum, Richea continentis*), provided there are no further fires. As a result, *Empodisma* may be less abundant in future. *Sphagnum* is relatively sparse at Mt Buffalo but is likely to slowly increase and add further competitive pressure on *Empodisma*.

Short-lived perennial forbs were prominent in the ground layer for up to two years at Mt Buffalo. The most common of these, *Asperula gunnii, Gonocarpus micranthus* and *Scaevola hookeri* are early post-disturbance colonisers associated with exposed bare ground. Cover would be expected to decline over time as a result of inter-specific competition (Carr & Turner 1959, Wimbush & Costin 1979a, 1979b; Wahren *et al.* 1994, Clarke & Martin 1999).

There were compositional differences between the three mountains but peat had higher moisture and organic content at Mt Baw Baw and at infrequently burnt sites at Lake Mountain. To some extent, this may also reflect differences between these mountains and the warmer, more seasonally drier climate of Mt Buffalo to the north. Previous work at Mt Buffalo demonstrated no apparent relationship between fire frequency and peat desiccation (Coates and Walsh 2010), raising the question of whether climate is an equal if not more important determinant of post-fire recovery. However, peatland endemics are most abundant on wetter soils at Mt Buffalo (Coates and Walsh 2010) implying that vegetation recovery at Lake Mountain has been relatively good to date and sites may be less vulnerable to dominance by one or two species than at Mt Buffalo. This notion is further supported by significantly higher species evenness at Lake Mountain two years post-fire, in spite of relatively high *Empodisma* cover.

The results suggest that frequent fire maintains peatland vegetation in an early post-fire successional stage. This is evidenced at Mt Buffalo where frequent fire combined with a decline in the condition of organic soils has promoted dominance of *Empodisma minus* and *Baeckea gunniana.* However, in the absence of fire at sites with wetter, more organic soils, dominance of these species appears to decline as slower growing peatland endemics increase in abundance. This seems to be the case at Mt Baw Baw which recorded high species diversity, richness and evenness. At present, given their floristic similarities and peat properties, as well as their geographic proximity, peatlands at Lake Mountain are likely to develop along a similar trajectory provided that there are no further bushfires in forthcoming decades. It is not possible to identify a precise rate of, or timeframe for recovery given the lack of comparable floristic data for Mt Baw Baw since the 1939 fires. Other influences on recovery include climate change and a return to the extended drought conditions which have impacted many wetland communities in south-eastern Australia. However, indications are promising for Lake Mountain largely owing to the apparently good condition of peat, in spite of having been burnt at high severity, a lack of recurrent fire in the past, the continued exclusion of grazing and the absence of significant levels of other disturbance close to peatlands.

Figure 35. Peatland at Mt McLeod (Mt Buffalo north), two years after fire.

Figure 36. Peatland at Mt McLeod (Mt Buffalo north), five years after fire.

Figure 37. Peatland at Mt McLeod (Mt Buffalo north), seven years after fire.

Figure 38. Echo Flat North, Lake Mountain, two years after fire.

Figure 39. Sandys Flat, Mt Baw Baw, 72 years after fire.

### 3.3.2 Are rare, threatened or significant species at risk?

There were very few rare or threatened species recorded at Lake Mountain during the survey but it is not certain whether this is a result of high fire severity in 2009. Most rare species were associated with hollows, which may be at additional risk after fire (Ashton and Hargreaves 1983) owing to a loss of organic substrate and an increase in silts and mud, increased surface radiation and soil temperatures and increased competition from vigorous post-fire resprouters (Norton and de Lange 2003). Hollows are also more desiccation-prone and combustible and may experience greater and more variable habitat changes after fire (Benscoter and Wieder 2003, Benscoter *et al.* 2005). Additional rare species might appear with increased time since fire at Lake Mountain, provided there is no further damage to peat or changes in hydrology.

Lake Mountain was also the smallest area sampled and the low number of rare or threatened species recorded may simply reflect the restricted area of available habitat. No pre-fire data existed for the Lake Mountain quadrats and it is possible that there were relatively few rare species prior to 2009. With one exception *(Epacris petrophila*), none of the rare species recorded at the other two mountains have been recorded at Lake Mountain (Walsh and Entwisle 1994, Walsh and Entwisle 1996, Walsh and Entwisle, 1999). The results suggest that the current low level of rarity at Lake Mountain is a consequence of climatic and geographical characteristics rather than high fire severity in 2009.

In contrast, rare species were well represented at Mt Baw Baw (17), implying that long-unburnt vegetation may provide more favourable habitat for rare species than recently-burnt vegetation. However, most species recorded at Mt Baw Baw only (i.e. and not at Mt Buffalo), also occur at higher elevations on the Bogong High Plains and Snowy Range, suggesting that their distribution is determined by climate.

In spite of recurrent fires and drier substrates, Mt Buffalo also recorded a high number of rare species (13; e.g. Figure 40), although two more quadrats (22 compared with 20) were surveyed in these peatlands. Mt Buffalo and Mt Baw Baw had the most species in common (8), suggesting that many rare species are relatively resilient to fire and/or there is suitable habitat available at both of these mountains. Previous work at Mt Buffalo demonstrated that some rare species were absent or uncommon in frequently burnt peatlands, or were slow to recover after fire (Coates *et al.* 2006, Coates and Walsh 2010). However, some species which were sparse or absent at Mt Buffalo after the 2003 fires were more frequent by 2011 or had remained stable, although still in low abundance, as expected given their natural rarity (e.g. *Caltha introloba, Coprosma moorei, Diplaspis nivis, Herpolirion novae-zelandiae, Oreobolus oxycarpus*).

Figure 40. Regeneration of *Caltha introloba* (A), *Oreobolus oxycarpus* (B) and *Richea continentis* (C), five years after fire at Lyrebird Plain, Mt Buffalo.


*Richea continentis* was slow to recover after the 2003 fire at Mt Buffalo. It was still scarce in 2008 but was showing signs of good recovery by 2011. Wahren and Walsh (2000) reported that *R. continentis* had still not regained its pre-fire cover 14 years after the 1985 fire. Similar slow recovery was observed at Kosciusko National Park, where seedlings were not recorded until three years after fire, in very low numbers from a single site (McDougall *et al.* 2009). *Richea* is an invader of later successional stages where it may co-dominate with *Baeckea gunniana* and *Epacris* spp. (Ashton and Hargreaves 1983, Kirkpatrick and Bridle 1999). It was also recorded more frequently in long-unburnt (36 years) vegetation in 1982 than in 2008 (Walsh *et al.* 1984, Coates and Walsh 2010) lending further support to previous observations that recovery of this species is slow. At Mt Buffalo, noticeable growth of other slow-growing woody species such as *Epacris gunnii, Epacris paludosa* and *Comesperma retusum* did not occur until five years post-fire and even then these species accounted for only a minor proportion of total vegetation abundance (Walsh and Coates 2010). Consequently, major changes in the abundance of woody species at Lake Mountain may not be detectable until roughly five years after fire.

*Sphagnum cristatum* was also very slow to recover after fire at Lake Mountain and Mt Buffalo but was a major structural component of long-unburnt vegetation at Mt Baw Baw. Pre-fire abundances are unknown at the former two mountains but it is unlikely that *S. cristatum* was previously particularly abundant at Mt Buffalo, at least since 1982 (Coates and Walsh 2010). *Sphagnum* cover was highly variable in all three surveys at Mt Buffalo, possibly reflecting patchy coverage of past fires, but was observed to be regenerating two years after fire (Coates *et al.* 2006). Given the higher levels of moisture and organic content of peat at Lake Mountain, it is likely that *S. cristatum* was previously more abundant and hummocks are expected to develop and expand over time.

### 3.3.3 Do species have specific management requirements?

We did not record any evidence of significant threats to vegetation recovery at Lake Mountain, other than the potential threat of another fire. The results of the study imply that peatlands are able to regenerate after fire but overall species diversity, richness and evenness may decline with frequent fire and/or a reduction in soil moisture and organic content.

It is widely accepted that severe fires can cause long term damage to peatlands by destroying their structure and initiating hydrological and hydraulic changes that affect surface flows, saturation, floristic composition and vegetation flammability. A major consequence of peat desiccation is an increase in sclerophyllous vegetation and invasion of species with a preference for drier soils (Costin 1954; Kershaw *et al.* 1993; Whinam & Chilcott 2002; Whinam and Hope 2005). In spite of this, the outlook for vegetation recovery at Lake Mountain appears to be positive. High soil moisture and organic content in peatlands at Lake Mountain suggest that the vegetation is likely to recover over the next 15 to 20 years in a similar trajectory to Mt Baw Baw, provided there is no re-occurrence of fire. At present, continued high fire frequency at Mt Buffalo, impairing the potential for peatlands to develop to an advanced successional stage, and the threat of fire at Mt Baw Baw are of significant concern. Whilst the results suggest that the three mountains may not be strictly comparable in terms of their floristic composition, some common patterns were evident. To better understand these at a site-specific scale, repeated surveys are recommended at all three mountains within the next three to five years.

Very few weeds were recorded during the study. The most potentially problematic of these was *Salix cinerea* (grey sallow wattle) at Mt Buffalo. Many individuals of this species were manually removed after the 2006 fires; however, there was some evidence that eradication measures will need to be repeated in the near future, particularly in the Five Acre Plain area.

### 3.3.4 Recovery milestones and growth stages

The response of plant species depends on life history strategies – recovery of resprouters is rapid whereas recovery of obligate seeders and *Sphagnum* is relatively slow (McDougall *et al.* 2009). Recovery of peatlands after fire is likely to be variable, as sites vary naturally (McDougall *et al.* 2009). Floristic composition also varies with altitude but in this study, it was demonstrated that the composition of peatland vegetation after fire is also likely to be dependent on previous fire frequency.

Growth stages for High Altitude Wetland are based on the premise that peat has been burnt during a fire (Cheal 2010). In the event of a less severe fire, the recovery period is likely to be shorter (Cheal 2010). There was very little evidence of completely burnt peat in the study area, even at Lake Mountain under high fire severity. This is probably because moisture content was sufficiently high enough to protect organic soils, and/or the fire moved at such a rapid pace that residency times were very short. Growth stages relevant to this report are the initial ‘renewal’, ‘founding’ and ‘maturity’ stages. We emphasise that the information below should be an addition to, rather than a correction, to Cheal (2010) because the peatlands in this study occurred at sub-alpine elevations where processes may differ from peatlands at higher elevations.

The following milestones/growth stages should be addressed in future vegetation surveys.

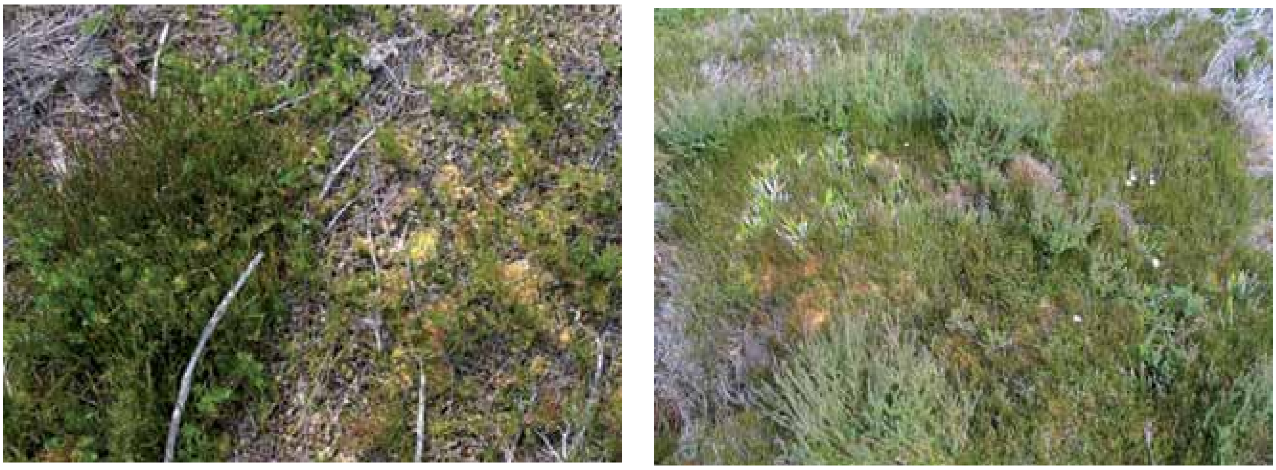
1. Renewal (2 years after fire, Figure 41). This stage is attributed with very low species diversity (Cheal 2010). However, the results of this study showed that species diversity is high in the two years after fire, owing to establishment of short-lived post-fire colonising forbs. Examples of species which are prominent in the ground layer for up to two years after fire are *Gonocarpus micranthus* and *Asperula gunnii.* These species regenerate from soil-stored seed, are short-lived and reproduce quickly. Species evenness will vary according to previous fire frequency. At lower sub-alpine elevations and depending disturbance history, *Sphagnum* may never have been particularly abundant, so may not feature as one of the major fire-killed taxa. Resprouting restiads, sedges and grasses also recover rapidly within this period with a critical function in stabilising bare ground The most important of these species is *Empodisma minus*. Bare ground cover of around 10% was also lower than indicated by Cheal (2010).

2. Founding (2–12 years after fire). The characteristics attributed to this stage were mostly observed at the study sites. However, some rapid changes occurred during this period at Mt Buffalo. The most noticeable of these was an opening out of the vegetation, so that there was a shift from lateral to vertical extension of keystone species at around five years since fire. Perennial forbs declined with increasing competition from resprouters. Obligate seeding shrubs (e.g. *Epacris* spp., *Comesperma retusum, Richea continentis*) were initially very slow to regenerate but began to grow rapidly five years post-fire, although they still accounted for only a minor proportion of total vegetation cover. Regeneration of *Sphagnum* was still very slow seven years after fire at Mt Buffalo but hummocks were beginning to re-establish at some sites. Areas of bare ground had decreased substantially and were approaching those measured at the benchmark site (Mt Baw Baw).

3. Maturity (60–150 years after fire). Species diversity was high at Mt Baw Baw, as predicted by Cheal (2010). Species evenness was also high at this relatively undisturbed site. Otherwise, our observations were in accordance with characteristics attributed to this growth stage. Sedges and restiads were dominant in the field layer but with low cover

Figure 41. Comparison of vegetation two years after fire (left) and 20 years after fire (right).

Two years after fire at Mt Buffalo (left) *Sphagnum* persists beneath an *Empodisma* root mat and litter. Early coloniser *Asperula gunnii* is prominent. Twenty years after fire at Mt Buffalo (right) there is well developed *Sphagnum* underlying a range of typical peatland endemics. The vegetation at this site represents a very small patch of unburnt peatland and was virtually unique on the mountain after the 2006 fires.



and stature (relative to Mt Buffalo). *Epacris* spp. and *R. continentis* were most abundant in the shrub layer, with *Baeckea gunniana* only a minor species in most quadrats. However, *B. gunniana* cannot be described as emergent and common in the canopy at Mt Baw Baw, where individuals were among the shortest in the study area (Cheal 2010). *Sphagnum* was well developed and rare peatland endemics were most frequent at Mt Baw Baw. However, we would suggest that peat accumulation rates referred to by Cheal (2010) are unknown and unlikely to be measurable over this time frame.

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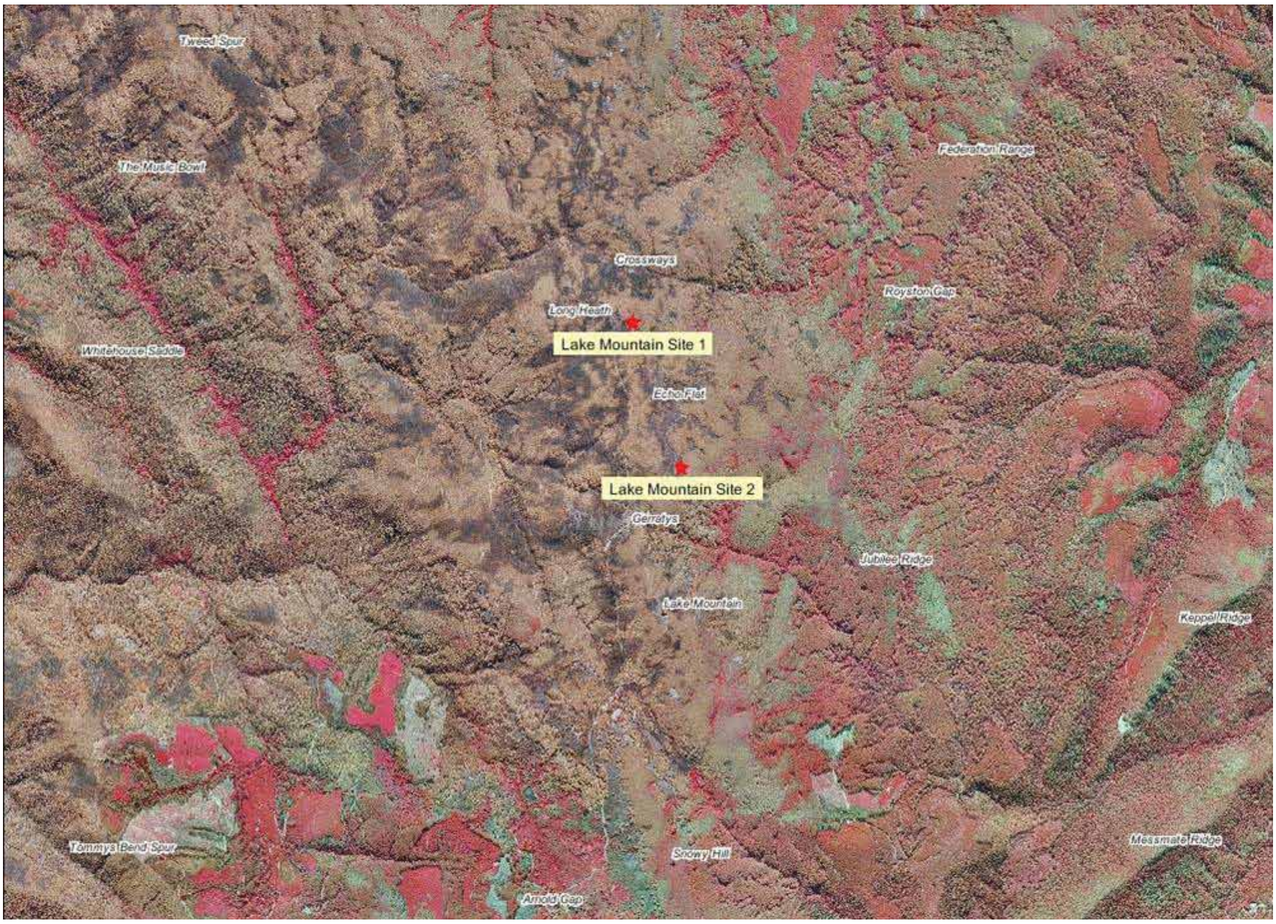
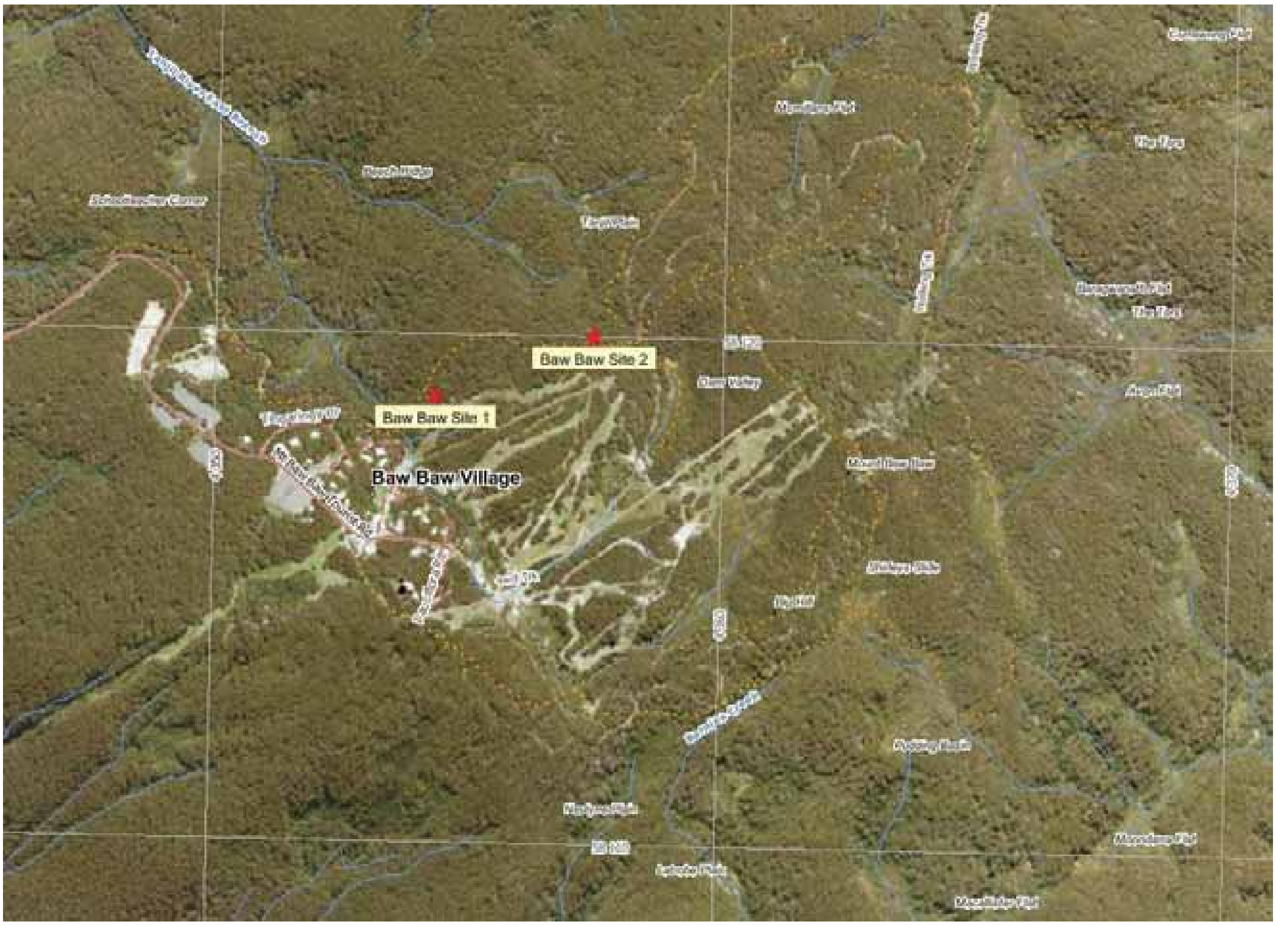
# Appendix 1 Transect grid references Lake Mountain

| Site | Start/finish | Easting (GDA 94) | Northing (GDA 94) | No. of points |
|---|---|---|---|---|
| *Triangle Junction 1 (1,452–1,445 m ASL)* | Start @200° | 400712 | 5851408 | 25 |
| | Finish | 400593 | 5851000 | |
| *Triangle Junction 2 (1,385 m ASL)* | Start 1 @0° | 401205 | 5850035 | 20 |
| | Stop 1 | 401246 | 5850422 | |
| | Start 2 @270° | 401216 | 5850431 | 5 |
| | Finish | 401134 | 5850433 | |



# Appendix 2 Transect grid references Mt Baw Baw

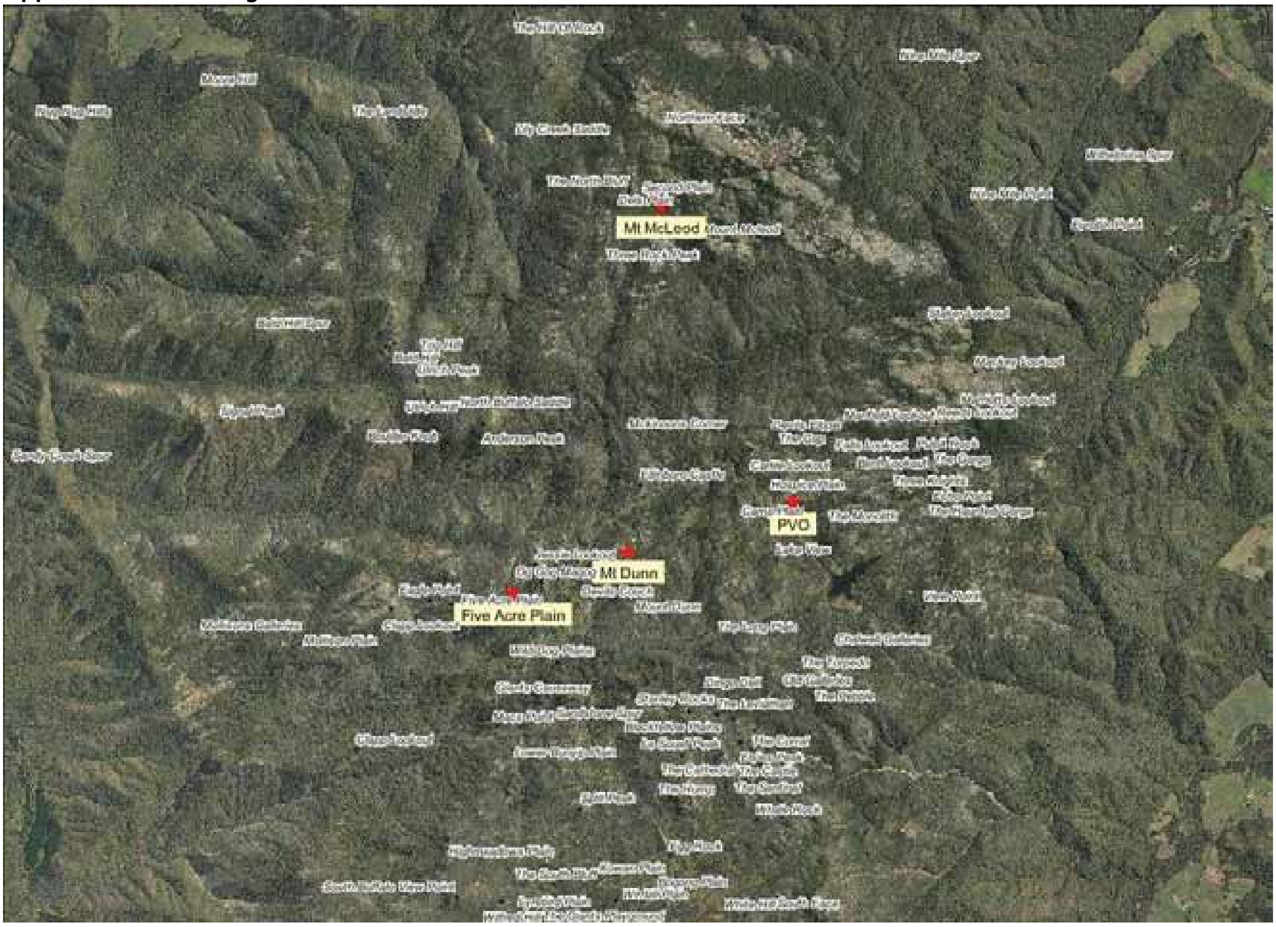
| Site | Start/finish | Easting (GDA 94) | Northing (GDA 94) | No. of points |
|---|---|---|---|---|
| *Tanjil Flat (1,505–1,498 m ASL)* | Start 1 @30° | 435433 | 5811876 | 10 |
| | Stop 1/Start 2 @60° | 435525 | 5812000 | 5 |
| | Stop 2/Start 3 @90° | 435604 | 5812024 | 7 |
| | Stop 3/Start 4 @180° | 435734 | 5812026 | 3 |
| | Finish | 435744 | 5811998 | |
| *McMillans Flat (1,544–1,490 m ASL)* | Start @350° | 435994 | 5812235 | 25 |
| | Finish | 435986 | 5812657 | |



# Appendix 3 Transect grid references Mt Buffalo

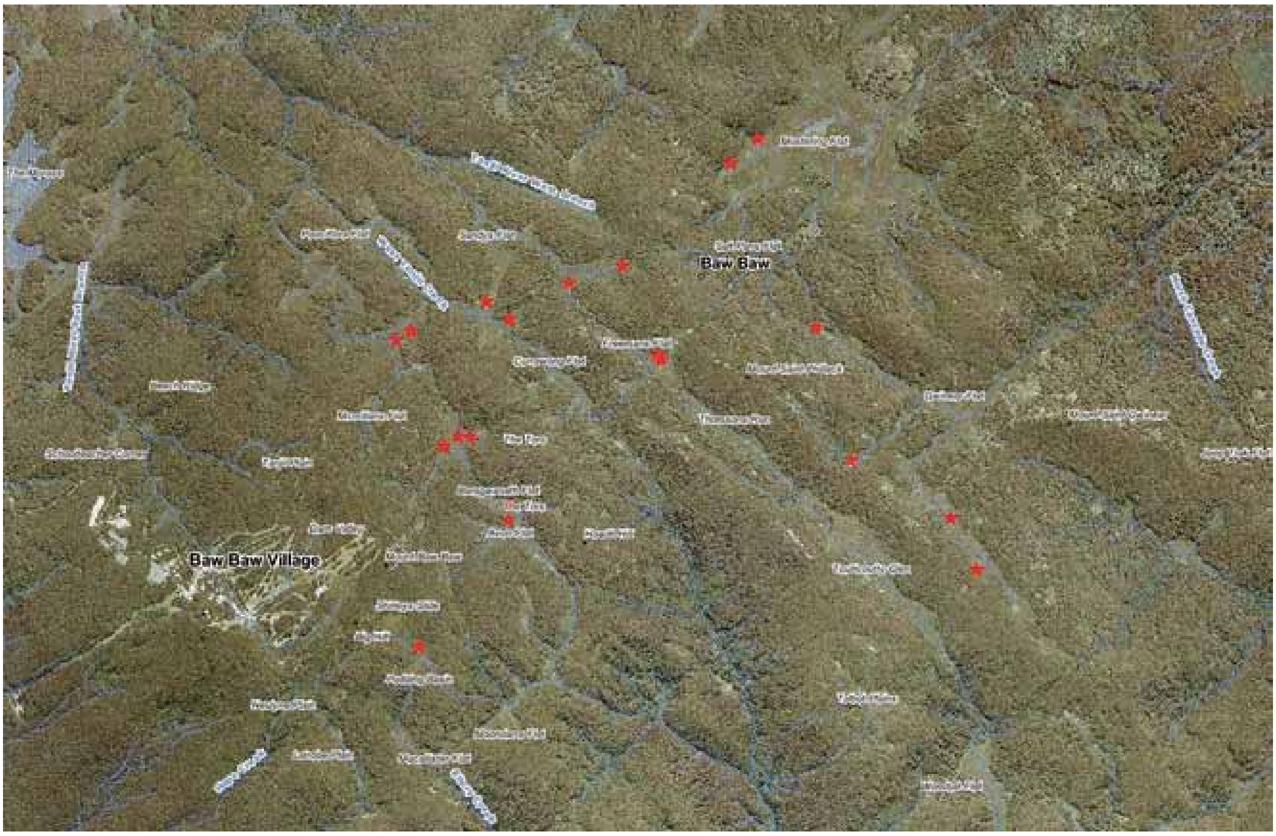
| Site | Start/stop/bearing | Easting (GDA 94) | Northing (GDA 94) | No. of points |
|---|---|---|---|---|
| *PV Office (1,401 m ASL);* Fire frequency = 0 (no record of fire) | Start 1 @315° | 482336 | 5935617 | 9 |
| | Stop 1/Start 2 @160° | 482218 | 5935561 | 8 |
| | Stop 2/Start 3 | 482246 | 5935497 | 8 |
| | Finish (approx) | 482396 | 5935647 | |
| *Mt Dunn (1,413 m ASL).* Fire frequency = 2 (2003, 2006) | Start 1 @230° | 480269 | 5934952 | 8 |
| | Stop 1/start 2 @135° | 480148 | 5934906 | 10 |
| | Stop 2/Start 3 @280° | 480215 | 5934820 | 7 |
| | Finish | 480157 | 5934789 | |
| *Mt McLeod (1,413–1,407 m ASL).* Fire frequency = 2 (1972, 2003) | Start 1 @60° | 480597 | 5939316 | 5 |
| | Stop 1 | 480645 | 5939331 | |
| | Start 2 @100° | 480649 | 5939316 | 4 |
| | Stop 2/Start 3@220° | 480707 | 5939288 | 5 |
| | Stop 3/Start 4@190° | 480748 | 5939250 | 6 |
| | Finish | 480717 | 5939197 | |
| *Five Acre Plain (1,452–1,420 m ASL).* Fire frequency = 4 (1972, 1985, 2003, 2006) | Start 1 @220° | 478830 | 5934401 | 7 |
| | Stop 1/Start 2 @290° | 478783 | 5934362 | 1 |
| | Stop 2 | 478764 | 5934373 | |
| | Start 3 @0° | 478737 | 5934418 | 6 |
| | Stop 3 | 478683 | 5934425 | |
| | Start 4 @20° | 478828 | 5934404 | 5 |
| | Stop 4 (approx) | 478903 | 5934479 | |
| | Start 5 @10° | 478845 | 5934470 | 6 |
| | Finish | 478843 | 5934512 | |

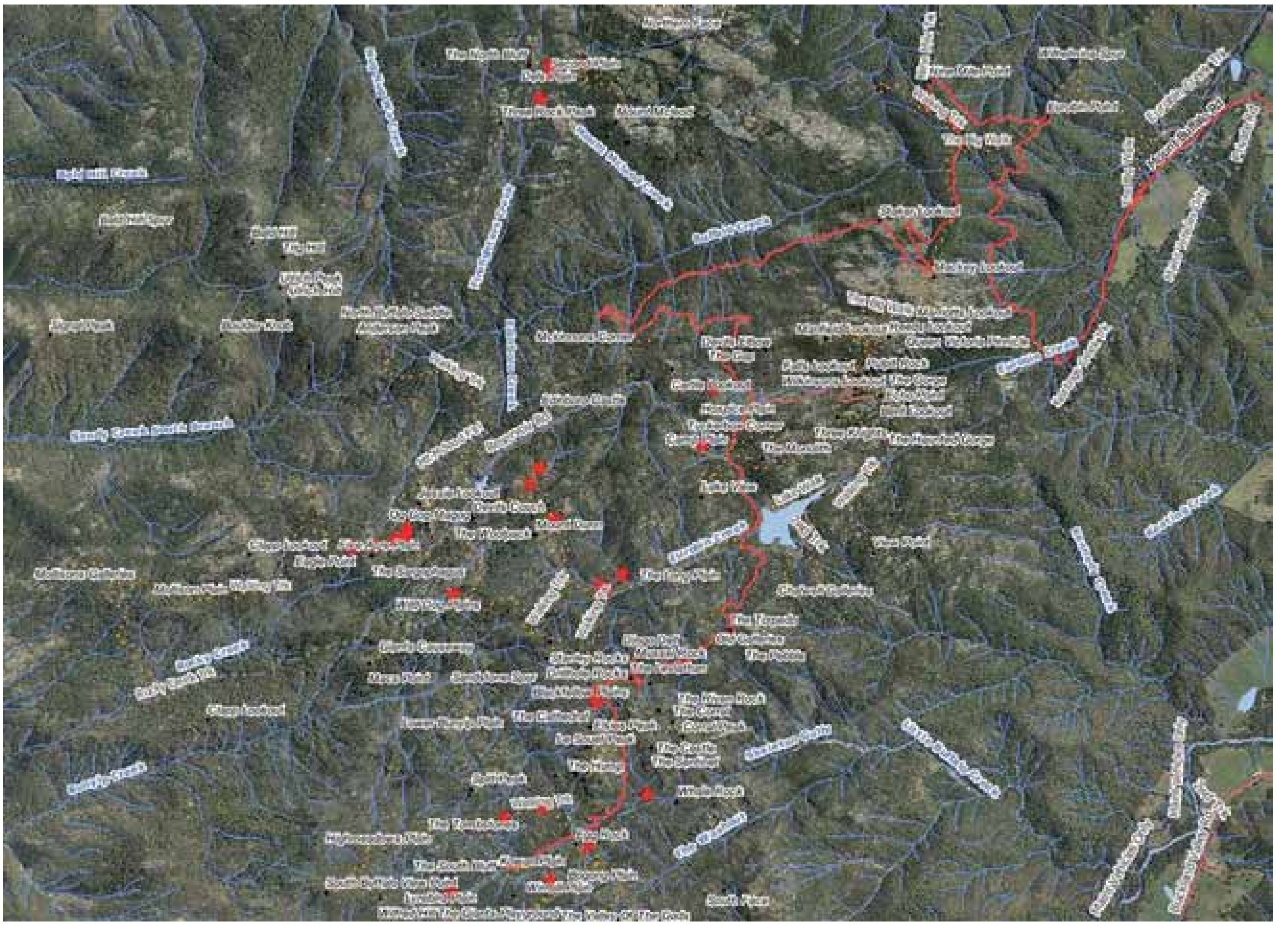
## Appendix 3. Transect grid references Mt Buffalo continued



# Appendix 4 Quadrat locations Lake Mountain

# Appendix 5 Quadrat locations Mt Baw Baw





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