An ecosystem assessment of mountain ash forest in the Central Highlands of Victoria, south-eastern Australia

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Abstract

We applied an ecosystem risk assessment to the mountain ash forest ecosystem of the Central Highlands of Victoria (hereafter "mountain ash forest"), south-eastern Australia, using the IUCN Red List of Ecosystems criteria. Using this methodology, we quantified: **(1)** key aspects of the ecosystem's historical, current and future decline in spatial distribution; **(2)** the extent of occurrence and area of occupancy for the mountain ash ecosystem; and **(3)** the decline in key abiotic and biotic processes and features for historical, current and future time periods. We developed a probabilistic model of tree growth stages to estimate the risk of ecosystem collapse within 50 to 100 years in the mountain ash forest.

There was uncertainty in our estimates of risk under the various IUCN criteria, with two sub-criteria being categorised as 'Data Deficient'. Our overall ranking of risk of collapse for the ecosystem was Critically Endangered. We are confident that this risk category is appropriate because all 39 scenarios modelled indicated a ≥ 92% chance of ecosystem collapse by 2067. Our findings highlight the important need for timely policy reform to facilitate improved management of the mountain ash ecosystem in Victoria. In particular, there needs to be greater protection of remaining areas of unburned forest, and restoration activities in parts of the forest estate. Implementation of these strategies will require a significant reduction in logging pressure on the mountain ash ecosystem.

Keywords: mountain ash, ecosystem collapse, climate, fire, logging

INTRODUCTION

The mountain ash forest is a globally iconic ecosystem. Mountain ash is the world's tallest flowering plant and individual trees may reach 50 m height within 35 years of germination (Ashton 1975). They can eventually exceed 90 m after several hundred years with some spectacular trees over 100 m tall having been documented (Beale 2007). The mountain ash forest is highly valued for its contributions to water and timber production (Flint & Fagg 2007; Viggers *et al.* 2013), its recreational and aesthetic values, and its unique biodiversity (see for example Mueck 1990; Lindenmayer 2009a). In addition to its unique species, the dynamics of the ecosystem are unusual among Australian forests because they can be subject to stand-replacing disturbance regimes such as wildfire (Lindenmayer *et al.* 2011a).

In this paper, we applied an ecosystem risk assessment to the mountain ash forest, using the International Union for the Conservation of Nature's (IUCN) Red List of Ecosystems criteria. The risk assessment requires an assessment of five criteria, A-E, each (except E) with three sub-criteria. For each criterion, numerical thresholds define ordinal categories of threat from Least Concern through to Critically Endangered (Table 1). The final, overall ranking is determined by the most severe ranking of the five criteria (refer to the Methods section for further information, and Appendix S2 for a full explanation). The IUCN developed these criteria to support a global Red List of Ecosystems, analogous to criteria that support the IUCN Red List of Threatened Species (IUCN 2001). The IUCN Council formally endorsed the Red List of Ecosystems criteria in mid-2014, which was the final step after resolutions passed at two successive World Conservation Congresses (in 2008 and 2012). The criteria are described by Keith *et al.* (2013), and an earlier paper by Rodríguez *et al.* (2011) provides further historical context. More recently, Nicholson *et al.* (2014) compared the IUCN's approach to that used in Australia to assess the threat status of ecosystems/ecological communities within different jurisdictions. Boitani *et al.* (2014) critiqued the criteria, arguing that species and ecosystems were not analogues and the criteria were open to ambiguous interpretations and uncertain outcomes. These are noteworthy concerns, and we expect testing and refinement will improve the criteria and how they are used. We apply the IUCN Red List of Ecosystems criteria here to inform this process.

In applying the IUCN criteria to the mountain ash forest, we aimed to: **(1)** identify the defining features of the system and the processes that threatened them, **(2)** evaluate trends in key variables relevant to the persistence of the ecosystem, and **(3)** assess the risk of ecosystem collapse in the 21st century.

Ecosystem description

Characteristic native biota

Many areas of mountain ash forest are generally dominated by a single overstorey tree species – *Eucalyptus regnans*. Mountain ash forest can also contain other overstorey tree species like alpine ash (*E. delegatensis*) and shining gum (*E. nitens*) at higher elevations (Lindenmayer *et al.* 1993) or messmate (*E. obliqua*), mountain grey gum (*E. cypellocarpa)* and red stringybark (*E. macrorhyncha)* at lower elevations (Campbell 1984).

Mountain ash forest supports a wide range of plant species in the midstorey tree layer and shrub layers (Supporting Information Appendix S1), and a rich array of native mammals (Lumsden *et al.* 1991). Native mammals include the endangered Leadbeater's possum (*Gymnobelideus leadbeateri*) which is virtually confined to the Central Highlands region (Lindenmayer *et al.* 2014a) and the vulnerable yellow-bellied glider (*Petaurus australis*) as well as six other species of arboreal marsupials and a diversity of forest bird species (Lindenmayer 2009a). Details of other native vertebrate species are outlined in Appendix S1.

Classifications

According to the Victorian vegetation classification system, mountain ash forest occurs within one Ecological Vegetation Class, 'EVC 30 Wet Forest' (http://www.dse.vic.gov.au/__data/assets/pdf_file/0010/98515/HFE_0030.pdf). Nationally, mountain ash forests are assigned to Major Vegetation Group 2, 'Eucalypt Tall Open Forests' (Department of the Environment and Water Resources 2007). Under the IUCN Habitats Classification Scheme (Version 3.1), mountain ash forests fall within habitat type 1.4, Temperate Forest.

Abiotic environment

Temperature and precipitation are the key abiotic variables in the mountain ash ecosystem and, in general, climatic conditions are the key determinant of its broad distribution patterns (Lindenmayer *et al.* 1996). The climate is typically characterised by mild, humid winters with occasional periods of snow, generally cool summers, mean annual temperature varying from 7.2° to 14.1°C and mean annual precipitation varying from 815 to 1775 mm (long-term mean climate variables estimated for the 1976 to 2005 period using ANUCLIM, Xu & Hutchinson 2013). Tree growth that is characteristic of the mountain ash forest can only occur within the ‘wet and cool’ environmental envelope defined by the Central Highlands of Victoria (Lindenmayer *et al.* 1996; Mackey *et al.* 2002). This ecosystem is therefore vulnerable to the effects of climate change, particularly higher temperatures and reduced rainfall (Nitschke & Hickey 2007; Wood *et al.* 2014).

At finer spatial scales, other environmental factors such as soil fertility, topography and natural disturbance also have an important influence on mountain ash forest (Florence 1996; Lindenmayer *et al.* 1999; Mackey *et al.* 2002). The ecosystem typically occurs at altitudes between 85 and 1380 m above sea level in mesic conditions favourable for tree growth (Mackey *et al.* 2002).

The deep, well drained soils typical of mountain ash forest are derived from Ordivician and Devonian sediments, extrusives, granitic soils and alluvium. Some of the most fertile soils are deep red earths which overlay igneous felsic intrusive parent material. These have a high soil water holding capacity and nutrient availability compared with most forest soils in Australia (McKenzie *et al.* 2004).

Distribution

As discussed above, factors at multiple spatial scales shape the distribution of the mountain ash forest. This ecosystem occurs in a region approximately 120 km north-east of Melbourne, south-eastern Australia (Figure 1). Forests dominated by mountain ash also occur in north-east Victoria, south-western Victoria (in the Otway Ranges) and throughout Tasmania (where it is also called swamp gum; Boland *et al.* (2006)). But several key aspects differentiate those forests from the mountain ash ecosystem, including distinctive vertebrate fauna (Lindenmayer 2009a) and flora (Mueck 1990).

The Central Highlands region supports approximately 157,000 ha of mountain ash forest. Approximately 20% is in closed water catchments, parts of which are also managed as the Yarra Ranges National Park (Viggers *et al.* 2013). The remaining 80% is located in areas broadly designated for paper pulp and timber production (Flint & Fagg 2007; Lindenmayer 2009a).

Key processes (threatening and natural) and their interactions

Fire is the primary form of natural disturbance in the mountain ash ecosystem (Ashton 1981), and overstorey trees are often killed in a high-severity conflagration (Smith *et al.* 2013). Young seedlings germinate from seed released from the crowns of burned mature trees to produce a new even-aged regrowth stand. Lower severity fires can result in multi-aged stands when the fire stimulates a regeneration cohort but is insufficiently intense to kill all overstorey trees (McCarthy & Lindenmayer 1998; Lindenmayer *et al.* 2014b). Following disturbance and stand initiation, young mountain ash trees exhibit rapid rates of growth.

The effects of fire on stand structure are linked to the age of a forest at the time it is burned. Fire in an old growth forest will produce a cohort of large dead trees and fire-scarred living old trees that can provide nesting habitat for a suite of cavity-dependent species such as Leadbeater's possum (Lindenmayer 2009b). Such habitat does not develop in young burned forest because small diameter trees do not remain standing long after they are burned, nor do they have the internal volume of wood capable of supporting suitably sized cavities (Gibbons & Lindenmayer 2002). When the interval between stand-replacing disturbances is < 20 years – the period required for trees to reach sexual maturity and begin producing seed (Ashton 1981) – stands of mountain ash will be replaced by other species, particularly wattle (*Acacia* spp.) (Lindenmayer *et al.* 2011a). Therefore, fires in rapid succession have the potential to eliminate populations of fauna as a result of the direct effect they can have on floristic composition (Lindenmayer *et al.* 2013).

Logging is the primary form of human disturbance in the ecosystem, and large areas have been subject to timber and pulpwood harvesting. Historically, more timber flowed through Port Melbourne than ports like Seattle in western North America, which have a far greater area than Melbourne from which to source timber (Dingle & Rasmussen 1991). In the past 40 years, the conventional method of logging has been clearfelling in which all merchantable trees within a 15-40 ha area are cut in a single operation. Following clearfelling, logging debris is burned to create a bed of ashes in which the regeneration of a new stand of mountain ash takes place, often by artificial reseeding (Flint & Fagg 2007).

There are two kinds of important interactions between natural disturbance (fires) and human disturbance (logging) in the ecosystem. First, burned forests are subject to salvage logging in which clearfelling operations are employed in an attempt to recover some of the economic value of fire-damaged trees (Lindenmayer *et al.* 2008). For example, widespread salvage logging occurred after wildfires in 1939, 1983 and 2009 (Noble 1977; Lindenmayer & Ough 2006; Lindenmayer *et al.* 2010). Second, artificial stand regeneration practices following conventional clearfelling operations in green forest produce young stands of dense regrowth forest. Recent quantitative analyses indicate that such stands are at risk of re-burning at high severity (Taylor *et al.* 2014).

The interacting effects of wildfire and logging (Figure 2) have the potential to create a landscape trap (Lindenmayer *et al.* 2011a) in which the mountain ash forest persists in short-interval disturbance dynamics. A landscape trap is defined as occurring where entire landscapes are shifted into, and then maintained (trapped) in, a highly compromised structural and functional state as the result of multiple temporal and spatial feedbacks between human and natural disturbance regimes. The core process underlying this 'landscape trap' is a positive feedback loop where logging leads to dense, young post-harvest regeneration that is at risk of re-burning at high severity – resulting in reduced forest age at the stand and landscape levels (Taylor *et al.* 2014). This can create irreversible changes in disturbance dynamics, forest cover, landscape pattern, and vegetation structure – thereby leading to collapse or a major regime shift to an alternative state (Lindenmayer *et al.* 2011a). This is because if fires occur at intervals of less than 20-30 years (Ashton 1981) the re-burning of young overstorey trees replaces mountain ash forest with *Acacia* spp. thickets.

The mountain ash forest has been found to be the most carbon-dense forest on earth (Keith *et al.* 2009). Old growth stands in which there has been no human disturbance can support more than 1800 tonnes of carbon biomass per ha. Conversely, logging operations can significantly deplete carbon stocks (Dean *et al.* 2012) and result in carbon storage levels approximately half that of unlogged old growth forests (Keith *et al.* 2014). Notably, unlike logging, fire (including high-severity wildfire) does not lead to large losses of carbon biomass (Keith *et al.* 2014).

For many years, the widespread application of industrial clearfell logging has been justified, in part, on the premise that it mimics high-severity wildfire (Attiwill 1994). This is an important assumption, for if there are strong similarities between human and natural disturbance, then forest values like the conservation of biodiversity are likely to be little affected under clearfelling regimes. However, several lines of empirical evidence indicate that there are substantial differences between these two forms of disturbance on the structure and composition of mountain ash forests. First, key elements of stand structure, particularly the abundance of large old hollow-bearing trees are far less abundant in forests subject to clearfelling than those that are burned. Second, data show that populations of many species of plants (especially re-sprouters) are significantly reduced in clearfelled sites in comparison to burned areas. Third, carbon stocks in burned forests are depleted by 8-14% after fire but carbon stock reduction is up to five times greater in clearfelled areas (Keith *et al.* 2014). There are other major differences between clearfelling and wildfire in mountain ash forests. Perhaps one of the most important is the overall age structure of the forest. Under a rotation of 50-80 years on cut sites, the mean age of the forest is significantly younger than the overall mean age of a forest estate subject only to wildfire (McCarthy & Burgman 1995). Perhaps most critical now is the realisation that mountain ash forests are not subject to **either** clearfell logging or fire, but rather **both** kinds of disturbance. That is, large parts of mountain ash forest have experienced not only fire or clearfelling, but a combination of both in rapid succession (*viz*: salvage logging). Some elements of the biota like Leadbeater's possum are highly sensitive to all three kinds of disturbance – fire, clearfelling and salvage logging. They are also sensitive to the substantial changes to landscapes associated with these disturbances, such as the widespread diminution of old growth cover. Finally, there is an increasing realisation that human disturbance and natural disturbance are not necessarily independent processes, as logged forests can be more prone to high severity fire for prolonged (decadal) periods (Taylor *et al.* 2014).

METHODS

A comprehensive account of our interpretation and application of the IUCN Red List of Ecosystems criteria is provided at Appendix S2, which also provides details on the datasets named below, and a more thorough explanation of our approach. Here we provide a brief overview.

Criterion A (Decline in distribution)

This criterion requires an assessment of the amount of change in spatial distribution for each of the three time periods. (**A1**) A current decline ≥ 80% equates to a category of Critically Endangered; ≥ 50% equates to Endangered, and ≥ 30% equates to Vulnerable. If these are not met, then a category of Least Concern applies. (**A2**) The same levels relate to future decline. (**A3**) For historic decline ≥ 90% equates to a category of Critically Endangered; ≥ 70% equates to Endangered and ≥ 50% equates to Vulnerable.

A1. Current decline: the decline in distribution over the past 50 years (i.e. since 1964)

An assessment of the amount of change in spatial distribution over the past 50 years is required. We first estimated the current distribution (i.e. in 2014) of the mountain ash forest by interrogating three Victorian Government spatial layers: 1) the Statewide Forest Resource Inventory dataset; 2) the Logging History dataset; and 3) the Ecological Vegetation Classes dataset. We then examined the Victorian Government's Forest Management Zone dataset to determine the proportion of the ecosystem which occurred on public land. We made our determination based on this proportion, and a review of the literature.

A2. Future decline: the predicted decline in spatial distribution over the next 50 years (i.e. to 2064)

Similar to A1, we predicted future geographic distribution (in 2064) based on land tenure. This was a highly conservative approach as we did not consider areas planned to be logged under the Timber Release Plan (Government of Victoria 2011). The was primarily because these areas regenerate after logging and are therefore arguably still part of the geographic distribution, albeit in a modified condition.

A3. Historical decline (sub-criterion 3): the decline in spatial distribution since 1750

Our estimate of decline here needed to differ from A1 because the mapped classes employed in A1 were not available from 1750. We therefore used the Australian Government dataset 'NVIS Major Vegetation Subgroups' which provided modelled vegetation cover pre-1750.

Criterion B (Restricted geographic distribution)

Criterion B addresses **(1)** extent of occurrence; **(2)** area of occupancy; and **(3)** the number of 'locations' occupied relative to the most serious plausible threat (where the number of locations within an ecosystem is greater than 1, if the impact of threats will affect spatially discrete areas within the ecosystem differently).

(1) To quantify the extent of occurrence, we created a minimum convex polygon that encompassed mapped distribution of the mountain ash ecosystem. **(2)** Area of occupancy was based on 10 km^2 grid cells. **(3)** The number of locations was based on wildfire as we consider this to be the most extensive, unpredictable threat of varying severity. We obtained information on the time and location of historical fires from the Victorian Government dataset 'FIRE100_YEAR'. To check the accuracy, we then overlayed the mapping for the most extensive fire with our long-term field sites distributed throughout the Central Highlands of Victoria (see Lindenmayer *et al.* 2012).

Assessment for Criterion C, D and E

Three key concepts underpinned assessments for Criteria C, D and E: **(1)** ecosystem collapse; **(2)** environmental degradation; and **(3)** relative severity (see Appendix S2 for an explanation). For Criteria C and D, we evaluated degradation of the abiotic environment and disruption to biotic processes and interactions, respectively. We did this by estimating the relative extent and severity of declines in the relevant variables. The severity estimates were assumed to be an average across 100% of the extent of the ecosystem. Therefore, only relative extent thresholds were used to differentiate between the three categories of threat: Critically Endangered (CR), Endangered (EN), and Vulnerable (VU). For Criterion C1, C2, D1 and D2 the thresholds were CR ≥ 80%, EN ≥ 50% and VU ≥ 30%. For Criterion C3 and D3 they were CR ≥ 90%, EN ≥ 70% and VU ≥ 50%. Various other comparisons are possible when severity estimates do not apply to 100% of the extent (see Table 1 of Appendix S2).

Collapse thresholds

Ecosystem collapse was considered to have occurred when:

(i) 100% of the area where the ecosystem currently occurs was no longer bioclimatically suitable (see Criterion C methods below). The climatic envelope range limits defined by the profile of the current distribution are shown Appendix S2. These thresholds were employed for Criterion C2 analyses.

Or

(ii) The abundance of hollow-bearing trees dropped below 1 per hectare averaged across the entire mountain ash ecosystem. This threshold was employed for sub-criterion D2, and Criterion E analyses.

Or

(iii) There was less than 1% of old-growth forest remaining in the ecosystem. This threshold was employed for sub-criterion D1 and D3.

Our three forms of ecosystem collapse were defined as ecosystem-wide averages. Therefore our assignment to risk categories was dependent on the relative severity percentages only.

Criterion C

Criterion C entails an assessment of the ecosystem in response to the most important abiotic variable over three time periods – since 1750 (C3), the past 50 years (since 1964; C1), and in the future (until 2064; C2). Temperature and precipitation and other climate-related variables were used for Criterion C.

Criterion D

Criterion D requires an assessment for the same three time periods but in response to hollow-bearing trees, the principal biotic variable. For Criterion D we drew on long-term field survey data, fire-history records and mapped old-growth forest. For D2, we investigated 39 scenarios based on varying harvesting and fire regimes. To investigate the impact of uncertainty in our definition of collapse, we varied the point of collapse from 0.5 to 1.5 tree-hollows per ha at 0.1 intervals.

Criterion E

Criterion E is an overarching analysis of the impacts of biotic variables on the probability of ecosystem collapse within 50-100 years. We ran simulations to investigate the future resource of hollow-bearing trees by systematically exploring scenarios that differed in harvesting and fire regime and the density level of hollow-bearing trees that defined ecosystem collapse. We ran 10,000 stochastic simulations for 39 scenarios (the same scenarios as used for D2). Each simulation consisted of a random draw from the appropriate distribution for each parameter. We then projected the results forward in the same manner as for criterion D2. That is, we have 10,000 projections for each of the 39 scenarios which we then used to calculate the probability of collapse.

Similar to D2, to investigate the sensitivity of the probability of collapse, we varied the definition of collapse for the number of hollow-bearing trees per ha. Additionally, we computed several percentiles of the simulations for each of the 39 scenarios, which we present graphically and in tabular form in Appendix S4. The 10^{th}, 20^{th} and 50^{th} percentiles of the simulations gave an estimate of what number of hollow bearing trees per hectare we would have to set as a definition of collapse, to result in a different endangerment category (i.e. Vulnerable vs Endangered vs Critically Endangered respectively). We present a graphical summary of the 2.5th, the 50th and the 97.5th percentiles for each of the 39 scenarios in Appendix S4.

RESULTS

The mountain ash forest is among one of the best studied Australian ecosystems. Despite this, two of the sub-criteria were Data Deficient (Table 1). Overall, our allocated rankings ranged from Least Concern to Critically Endangered thereby leading to an overall risk category of Critically Endangered because, as per the Keith *et al.* (2013) protocol, the highest level of risk determines the final overall rating.

Criterion A - Decline in distribution

A1. Current decline (sub-criterion 1): the decline in distribution over the past 50 years (i.e. since 1964)

We estimated the current distribution of the mountain ash ecosystem to be 156,700 ha, 96.4% of which is on public land. We assumed, due to stable land tenure (see below), that there had been virtually no change in distribution since 1964. The status of the mountain ash ecosystem is therefore **Least Concern** (LC).

Approximately 20% of the mountain ash forest occurs in closed water catchments, parts of which are also managed as the Yarra Ranges National Park. Areas of forest were closed for water supply purposes starting in the 1890s with several closed catchments added progressively after that time (Viggers *et al.* 2013). The remaining 80% of mountain ash forest is located in areas broadly designated for paper pulp and timber production (Flint & Fagg 2007; Lindenmayer 2009a). Agreements regarding logging practices between water catchment management authorities and the (then) Forests Commission of Victoria for the Central Highlands pre-date 1928 (Viggers *et al.* 2013).

A2. Future decline (sub-criterion 2): the predicted decline in spatial distribution over the next 50 years (i.e. to 2064)

Almost the entire mountain ash forest occurs on public land. We therefore assumed that the distribution would remain largely unchanged from 2014 until 2064. The status of the ecosystem is therefore **Least Concern** (LC).

A3. Historical decline (sub-criterion 3): the decline in spatial distribution since 1750

The amount of wet sclerophyll forest dominated by mountain ash has decreased from a modelled pre-1750 area of 183,000 ha to an extant area of 180,000 ha, a reduction of approximately 2% (see Figure 3). The status of the ecosystem is therefore **Least Concern** (LC).

Criterion B - Distribution size: extent of occurrence and area of occupancy

B1- Extent of occurrence

The area of the minimum convex polygon enclosing all mapped occurrences (see Figure 4) was 11,000 km^2. However, before making a final determination under Criterion B1 and B2, we assessed the number of locations within the ecosystem (see Appendix S2). In 1939, a single wildfire affected between approximately 74% and 96% of the ecosystem. Our 'best-estimate' is that 85% of the distribution of the ecosystem was burnt (see Methods). This indicates that a single threatening event can affect almost the entire distribution of the mountain ash ecosystem. Based on this evidence, we concluded there are ≤ 2 locations within the ecosystem. The ecosystem is therefore **Endangered** (EN).We acknowledge that this approach does not account for regeneration following disturbances such as wildfire (Smith *et al.* 2013). However, the simplification of the ecosystem through one such event is in itself sufficient to meet the endangered criterion for Criterion B1.

B2- Area of occupancy

We superimposed a 10 km^2 grid over the mapped polygons of the mountain ash forest (see Figure 4) and calculated that the ecosystem was present within 96 (of 140) grid cells. Of these, 23 grid cells contained < 1 km^2 of the ecosystem. After excluding cells with limited occurrence, we estimated that the ecosystem occupied 73 of the 10 x10 km grid cells (i.e. more than 50 cells). Based on the number of occupied cells, the ranking of the ecosystem is therefore **Least Concern** (LC).

B3 - Number of locations

Sub-criterion 3 requires a determination of whether there are a small number of locations (typically fewer than five) **and** if these are prone to the effects of human activities or stochastic events within a short period of time, thereby making the ecosystem susceptible to collapse or becoming critically endangered within a short period of time. Based on our assessment of there being ≤ 2 locations, and our results for Criterion D and E (see below), our judgement was that the mountain ash ecosystem was indeed prone to collapse within a short period. It therefore met the requirements for **Vulnerable** (VU).

Criterion C - Decline in abiotic processes

C1. Current decline (sub-criterion 1): Environmental degradation through changes in precipitation and temperature over the past 50 years (i.e. since 1964)

There were insufficient data to complete the assessment (see Appendix S2 for an explanation). The ecosystem was therefore **Data Deficient** (DD) for this criterion.

C2. Future decline (sub-criterion 2): an estimated assessment of environmental degradation through climate change over the next 50 years (i.e. to 2064)

<u>Environmental degradation</u>: we used the Intergovernmental Panel on Climate Change (IPCC) emission scenarios A2, A1b and A1F1 to calculate the predicted losses of extent relative to climatic suitability under these scenarios. From the current distribution of 156,700 ha (see Table 2), the predicted losses at the 50th percentile were 36,350 ha (23%), 29 900 ha (19%)

and 70,500 ha (45%) respectively. Lower bounds at the 10th percentile were 17%, 14% and 47%, while upper bounds at the 90th percentile were 97%, 98% and 100% respectively.

Relative severity: the amount of loss needed to trigger collapse is 156,700 ha; that is, the total current extent of the mountain ash forest. Relative severity ranged from 14% to 100%. This ranks the system between Least Concern and Critically-Endangered (LC-CR).

Based on the 50th percentile predictions, and given that functional collapse will be realised prior to 100% physical loss of the ecosystem, we ranked the ecosystem as **Vulnerable (VU)**.

C3. Historical decline (sub-criterion 3): an assessment of environmental degradation through changes in precipitation and temperature since 1750

There were insufficient data to complete the assessment (see Methods and Appendix S2). The ecosystem was therefore **Data Deficient** (DD) for this criterion.

Criterion D - Decline in biotic processes and interactions

D1. Current decline (sub-criterion 1): Environmental degradation resulting from the loss of hollow-bearing trees (using old-growth as a surrogate) over the past 50 years (i.e. since 1964)

Environmental degradation: there has been a significant reduction in the amount of old-growth since 1964 (Table 2). We estimated that the area of mountain ash forest that was unlogged and unburnt by wildfire in 1964 was 6,300 ha (4% of the estate). This had been reduced by 4,600 ha (i.e. the observed decline) to 1,700 ha (1% of the estate) by 2011. Note that the estimate of 6,300 ha was derived using the 1939 fire extent mapping layer which we considered to be an over-estimate by approximately 11%. Because we cannot determine the spatial interaction with logging while accounting for this error, we do not provide bounds on our estimate.

Relative severity: to trigger ecosystem collapse, the amount of old growth forest would need to decline to 1,400 ha (which equals 0.9% of 156,700 ha an approximation for < 1% old-growth forest remaining in the current ecosystem; see Methods). Therefore, the loss needed to achieve ecosystem collapse was: approximately 6,300 ha (Total predicted area of old growth in the absence of logging in 1964) –1,400 (ecosystem collapse level) = **4,900 ha**, yielding a relative severity of approximately 94% (i.e. 4,600/4,900 * 100). We concluded that the disruption of biotic processes over the past 50 years, based on a change in the number of hollow-bearing trees (using old-growth as a surrogate), indicated a decline with ≥ 80% relative severity (average across 100% extent; see Appendix S2). The ecosystem is therefore **Critically Endangered (CR)**. To change this ranking would require a relative severity of less than 90%, requiring the 1964 area of old growth to be < 4,400 ha (an error of 70%) which is improbable.

D2*. *Future decline (sub-criterion 2): Environmental degradation resulting from the projected loss of hollow-bearing trees over the next 50 years

For projections of future decline in the abundance of hollow-bearing trees, we utilised empirical research data from long-term field sites. We could not do this for the other two time periods because data collection on the condition and abundance of hollow-bearing trees commenced in 1997. Our projections included provision for ongoing clearfell logging under the Victorian Government's Timber Release Plan (Government of Victoria 2011) and future fires, both of which can accelerate the loss of large hollow-bearing trees in mountain ash ecosystems (see Appendix S2).

Our modelling projected a severe decline in the average number of large old hollow-bearing trees across the mountain ash forest from approximately 3.77 ha^{-1} in 2011 to approximately 0.29-0.82 ha^{-1} by 2067 (Table 3, and Appendix S3 for full data). The lower bound in the range of values was from the scenario based on simulating the occurrence of a fire before 2067 that was equal in extent to the 1939 fire (1939+1983 re-growth harvesting scenario, large fire extent; see Appendix S2 and S3). The highest values in the range for the abundance of hollow-bearing trees result from the scenario with no future fire or logging (Table 3, and Appendix S2). Therefore, our ‘observed estimate of decline’ in the best case scenario was 78% and the worst case scenario was 92% (Table 3).

We defined ecosystem collapse as occurring when the average abundance of hollow-bearing trees declined to < 1 hollow-bearing tree per ha (see Appendix S2). Modelling in all 39 scenarios (3 harvesting scenarios by 13 fire regimes) projected a decline of ≥ 78% with ≥ 100% relative severity (averaged across 100% extent of the ecosystem) (see Appendix S3). The ecosystem is therefore **Critically Endangered** (CR), with plausible bounds also within this category.

Interrogation of the sensitivity of the relative severity and ecosystem ranking to variation in the definition of collapse between 0.5 to 1.5 hollow-bearing trees per ha showed that all relative severities remained above 90% (see Table 3 and Appendix S3). This means that the ecosystem would remain ranked as CR under D2 even if the definition of collapse was set at 0.5 hollow-bearing trees per ha (half what we consider to be the appropriate level).

D3. Historical decline (sub-criterion 3): an assessment of environmental degradation and relative severity through a loss of hollow-bearing trees (using old-growth as a surrogate) since 1750

<u>Environmental degradation</u>: it is not known how much old-growth forest was present within the mountain ash ecosystem in 1750. However, Lindenmayer *et al.* (2014a) suggest that between 30-60% of the forest was formerly old-growth. The current extent of the mountain ash forest is 156,700 ha, of which a minimum of 47,000 ha (30%) is estimated to have been old-growth in 1750 (see Table 2). The current area of old-growth forest is 1,700 ha. Therefore, our lower bound ‘observed estimate of decline’ in the amount of old growth is **45,300** ha.

Relative severity: we defined ecosystem collapse for Criterion D as occurring when the area of old growth forest was < 1,400 ha (see Appendix S2). The level of loss needed to trigger ecosystem collapse was: 47,000 (Total pre-1750 old growth) – 1,400 (ecosystem collapse level) = 45,600 ha, yielding a relative severity of 99% (i.e. 45,300/45,600 * 100). We therefore concluded (based on the lower bound of the range in old growth cover) that the disruption to biotic process over the past 50 years has led to a decline with ≥ 90% relative severity. The ecosystem is therefore **Critically-Endangered** (CR), with lower and upper plausible bounds within this ranking.

Criterion E - Quantitative assessment of the probability of ecosystem collapse within 100 years

Table 4 and Appendix S4 contain estimates of the probability of ecosystem collapse derived from 10,000 simulations generated by varying the input parameters with a coefficient of variation of 17.3%. All scenarios indicated a ≥ 92% chance of reaching a collapsed state (less than 1 hollow bearing tree per hectare) by 2067, although there was some variation in the trajectory of collapse depending on which previous fire (1939, 1983 or 2009) was used as a template to simulate the future fire and the timing of that simulated fire event between 2014 and 2067. Therefore, for Criterion E, we ranked the ecosystem as being **Critically Endangered** (CR), with plausible bounds within this category because it has a ≥ 50% probability of collapse within 50 years (although our projections were made to 2067 because of the 14-year time steps used in the simulations; see Methods or Appendix S2).

Table 4 and Appendix S4 also show the sensitivity of the probability of collapse and rating category to variation in the definition of collapse between 0.5 to 1.5 hollow-bearing trees per hectare. The results indicate that for the majority of the 39 scenarios we would have to change the definition of collapse to 0.7 to change the probability of collapse from the Critically Endangered level of 50%. In the case of a large fire event, the definition of collapse would have to change to less than 0.5 hollow bearing trees per hectare. This also happens for a scenario characterised by a fire of medium spatial extent and current logging practice, that is, logging both 1939 and 1983 regrowth stands.

DISCUSSION

We have used the protocol developed by Keith *et al.* (2013) to complete a detailed assessment of the mountain ash forest ecosystem. A notable outcome of our investigation was that we identified markedly different estimates of risk – ranging from Least Concern through to Critically Endangered – depending on which aspects of the ecosystems were under consideration. For example, the distribution of the ecosystem (Criterion A) has undergone remarkably little reduction over the past 50 years and was classified as Least Concern. However, examination of the disruption to the biotic process of hollow-bearing trees for Criterion D showed the ecosystem to be Critically Endangered for all time periods examined. This is because of a positive feedback between logging and fire (Lindenmayer *et al.* 2011b; Taylor *et al.* 2014) which significantly impairs the recruitment of large old trees and old growth forest in the mountain ash ecosystem (Lindenmayer *et al.* 2012; Lindenmayer *et al.* 2014a). Under Criterion E, interactions such as these can be explicitly assessed through stochastic modelling. Ideally, for Criterion E, in our stochastic model which predicts the probability of ecosystem collapse within 50 to 100 years, we would also have incorporated the various climate change scenarios considered under Criterion C. However, we did not have sufficient data to compute collapse rates of hollow-bearing trees under the various climate change regimes. We therefore utilised only available data on the decline of hollow-bearing trees (see Criterion D) to complete the assessment. Nevertheless, our modelling highlighted the risk of collapse of the ecosystem within a very short time. The rank of Critically Endangered under Criterion E is a result of the rapidly declining abundance of large old hollow-bearing trees and the limited current area of old growth forest in the ecosystem.

Although there were marked differences in the assessment outcome, depending on which criterion was being examined (see Table 1), we suggest that this is a strength and not a weakness of the approach. Indeed, we argue there is considerable merit in an ecosystem assessment process that is underpinned by a range of criteria. This is because a single metric may not provide a detailed picture of the status of a given ecosystem. This was highlighted in our case study in which current area of occupancy had changed little relative to historical extent, but major changes in biotic variables suggested that the ecosystem is at high risk of collapse within the next 50 years (Table 4). The lack of ranking among the different criteria allows a level of flexibility or pragmatism within the assessment framework. This is because different ecosystems will be data deficient in different respects and this should not unduly influence the overall rating. This does, however, raise interesting questions as to whether a minimum set of criteria should be addressed and, if so, which criteria should comprise this minimum set? However, prescribing such an approach might generate perverse outcomes rather than introduce increased robustness, and we would want to examine multiple systems and determine the effects of different combinations of criteria before advocating any given approach that ranks certain criteria over others.

The application of the protocol developed by Keith *et al.* (2013) provided a platform on which to assemble current knowledge of the status of, and threats to, the mountain ash forest. It also helped identify key knowledge gaps such as: **(1)** the paucity of data on pre-1939 logging operations and post-fire salvage harvesting, both of which are likely to have had profound impacts on the ecosystem (Lindenmayer 2009a); and **(2)** the lack of long-term climatic data for the study region, which led to two categories of Criterion C being ranked as Data Deficient (see Appendix S2). However, we also found that the protocol was difficult to use because the mountain ash ecosystem is so well understood as a result of several decades of intensive research. That is, the availability of data on the ecosystem meant that there were difficult choices to be made about which sub-criteria within particular criterion, and which variables, were the most suitable for the purposes of the assessment (e.g. for Criterion B). The selection of different options within sub-criterion B2 may have led to different outcomes for categories of threat. We therefore suggest an important future step will be to complete a sensitivity analysis to determine the robustness of assessment outcomes to the different options chosen for assessment. In part, we tried to include this in our approach (see Table 3 and 4) by examining the sensitivity of our informed 'choice' of collapse being 1 hollow-bearing tree per hectare, and by taking an alternative perspective and asking: What would the number of hollow-bearing trees per hectare need to be to alter our risk assessment from Critically Endangered under Criterion E? We would encourage the incorporation of these types of analyses in other applications of the criterion in different ecosystems in the future. Sensitivity analyses are also common in other kinds of risk assessment approaches like Population Viability Analysis (PVA). Like the IUCN Criteria, PVA can be data intensive and complex in nature. PVA has been used in the past within the mountain ash ecosystem, but primarily to explore the impacts of fire, logging, management actions and other drivers on the possible persistence of populations of various individual species of arboreal marsupials (e.g. Lindenmayer & Lacy 1995; Lindenmayer & Possingham 1996; Lindenmayer & McCarthy 2006). We did not use PVA here because we did not focus on individual populations within the ecosystem, but it would be interesting to compare the outcomes of a series of PVAs of specific populations within the ecosystem, which are dependent on the processes examined here. Similar to Criterion E, PVAs generate predictions of the relative risk of extinction in terms of a probability of extinction or quasi-extinction (Possingham *et al.* 2013). But unlike the IUCN Criteria, PVAs are not employed at an ecosystem level, and estimates are not bounded within pre-defined time periods. Both approaches have their utility but the IUCN Criteria is a single application that can examine many processes relevant to multiple populations within an ecosystem. A focus on key ecosystem processes can help improve understanding of other environments where similar kinds of processes may be important. As an example, understanding of the combined effects of fire and logging in the mountain ash forest has general lessons for other logged tall wet forested ecosystems. Similarly, the disruption of age classes and how this undermines an ecosystem's ability to support hollow-bearing trees has general applicability to other ecosystems such as temperate eucalypt woodlands which also support populations of cavity-dependent arboreal marsupials and birds. We provide discussion on the management implications of our assessment in Appendix S5.

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Supporting Information

Appendix S1: Information on flora and fauna in the mountain ash ecosystem

Appendix S2: Detailed Methods for the Ecosystem Assessment

Appendix S3: Results - Supplementary material for Criterion D2: Projections of hollow bearing trees per hectare in 2067 by fire/harvesting scenario; and Sensivity of relative severity to changing the definition of collapse by fire/harvesting scenario

Appendix S4: Results - Supplementary material for Criterion E: Probability of collapse by fire/harvesting scenario, tree hollow/hectare density, and year; and Sensitivity of the probability of collapse to varying the definition of collapse by fire/harvesting scenario

Appendix S5: Management Implications - Supplementary material for the Discussion.

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Table 1. IUCN Red List ecosystem criterion for the mountain ash ecosystem in the Central Highlands of Victoria. LC is Least Concern, VU is Vulnerable, EN is Endangered, CR is Critically Endangered and DD is Data Deficient. Sub-criteria with brackets indicate assessments undertaken using upper and lower bounds.

Criterion	**A**	**B**	**C**	**D**	**E**
Sub-criterion 1	LC	EN	DD	CR	CR(CR-CR)
Sub-criterion 2	LC	LC	VU (LC-CR)	CR (CR-CR)	
Sub-criterion 3	LC	VU	DD	CR (CR-CR)	

Table 2. Summary of mountain ash extent (hectares) logged or burnt by large wildfires

Threat	Last 50 years			Since 1750			Next 50 years		
	All	Old-Growth	Mature	All	Old-Growth	Mature	All	Old-Growth	Mature
Logging									
Total logging	55,300	1,000	42,300	>62,800	>1,000	>49,300	113,600 (unless capped at 83,750)	0	33,000 (unless capped at 21,700)
Percentage	35%	0.6%	27%	>40%	>0.6%	>32%	73%or 54% if capped		21%; or 14% if capped
Salvage logging	7,900	600	4,800	>7,900	>600	>4,800	Depends on fire regime	Depends on fire regime	Depends on fire regime
Percentage	5%	0.4%	3%	>5%	>0.4%	>3%			
Large wildfires									
1939				150,400*	>45,100	unknown	80,400	1,600	49,500
Percentage				96%*	>29%		51%	1%	32%
1983	17,250	1,100	14,900	17,250	1,100	14,900	9,200	200	5,700
Percentage	11%	0.7%	9.5%	11%	0.7%	9.5%	6%	0.1%	4%
2009	53,400	3,100	35,350	53,400	3,100	35,350	28,500	600	17,500
Percentage	34%	2%	23%	34%	2%	23%	18%	0.4%	11%

Estimated pre-1750 extent < 183,000 ha (note this is based on wet sclerophyll forest that is predominantly mountain ash – see Appendix S2); Estimated pre-1964 extent = 156,700 ha; Estimated current extant = 156,700 ha; Old-growth defined as forest with no wildfire or logging since records began (1903 for fire and 1932 for logging); Mature defined as 1939 regrowth; Note. Some areas of forest burnt in the wildfires were subject to salvage logging (as estimated above) but we do not know how much was salvage logged after the 1939 fire. The estimates of area logged or burnt are not necessarily mutually exclusive, but logged and burnt = salvage logging.

* This value was derived as per the values shown in this Table for the 1983 and 2009 fires. However, see Methods and Appendix S2, for our revised estimate of 85% extent, which is approximately 133,200 ha.

Table 3. Projections of hollow bearing trees per hectare in 2067, change relative to 2011, relative severity and its sensitivity to varying the definition of collapse (the number of hollow bearing trees per hectare) by fire/harvesting scenario. We considered three different harvesting scenarios (no harvesting, 1983 regrowth only, and 1939&1983 regrowth), four fire regimes (no fire in the next 56 years and a small, medium and large fire extent). We report results for 2067, the final interval (the additional intervals are given in the Appendix S3). We calculated the projected number of hollow bearing trees for each of the above scenarios and computed the percentage decline in 2067 relative to 2011 and the relative severity using 1.5, 1.0 and 0.5 hollow bearing tree per hectare as our definitions of collapse to investigate the sensitivity to varying the definition of collapse. Values >100% indicate collapse and replacement by a novel ecosystem with reduced diversity of vertebrate fauna and few large trees.

Harvesting	**Fire**	**2067 Projected**	**Percentage decline**	**Relative Severity, Sensitivity**		
Scenario	**Regime**	**hbt/ha**	**from 2011**	**1.5 htb/ha**	**1.0 hbt/ha**	**0.5 hbt/ha**
No Harvesting	No Fire	0.82	78.36	130.1	106.7	90.3
	Small	0.77	79.62	146.3	108.4	91.8
	Medium	0.67	82.26	132.2	112.0	94.8
	Large	0.45	88.11	136.6	119.9	101.6
1939+1983	No Fire	0.58	84.67	140.6	115.2	97.6
	Small	0.56	85.26	152.1	116.0	98.3
	Medium	0.48	87.23	141.6	118.7	100.6
	Large	0.32	91.56	144.9	124.6	105.6
1983	No Fire	0.77	79.48	132.0	108.2	91.6

	Small	0.73	80.71	147.7	109.8	93.0
	Medium	0.63	83.27	134.0	113.3	96.0

Table 4. The probability of collapse in 2067 and its sensitivity to definition of collapse by fire/harvesting and percentiles of the simulated number of hollow bearing trees per hectare distribution. We considered three different harvesting scenarios (no harvesting, 1983 regrowth only, and 1939&1983 regrowth), four fire regimes (no fire in the next 56 years and a small, medium and large fire extent). We report results for 2067, the final interval (the additional intervals are given in the Appendix S4). The probability of collapse (expressed as a percentage) is estimated from 10,000 simulations by varying the input parameters discussed in the methods using a coefficient of variation of 17.3% (see Methods for more details). The percentiles give the number of hollow bearing trees per hectare necessary to meet the Vulnerable (10%), Endangered (20%) and Critically Endangered (50%) thresholds given in the IUCN Red List Criteria. The percentiles indicate at what level we would need to set the collapse rate for the particular scenario to meet the criteria given above. Note a value of < 0.01 means that none of the 10,000 simulations meet the criteria for collapse similarly, and a value of >99.99 means that all the values meet the criteria for collapse.

Harvesting	**Fire**	**Probability of Collapse in 2067**			**Percentiles of the simulated hbt/ha distribution**		
Scenario	**Regime**	**1.5 hbt/ha**	**1.0 hbt/ha**	**0.5 hbt/ha**	**VU-10%**	**EN-20%**	**CR-50%**
No Harvesting	No Fire	>99.99	92.04	0.04	0.65	0.70	0.81
	Small	>99.99	97.19	0.16	0.62	0.66	0.77
	Medium	>99.99	99.92	1.37	0.56	0.60	0.68
	Large	>99.99	>99.99	53.41	0.40	0.43	0.49
1939+1983	No Fire	>99.99	>99.99	20.97	0.46	0.49	0.58
	Small	>99.99	>99.99	22.07	0.46	0.49	0.57
	Medium	>99.99	>99.99	53.15	0.41	0.43	0.49
	Large	>99.99	>99.99	99.13	0.26	0.28	0.33

1983	No Fire	>99.99	96.77	0.21	0.62	0.66	0.77
	Small	>99.99	99.14	0.59	0.59	0.63	0.73
	Medium	>99.99	>99.99	4.28	0.53	0.56	0.64
	Large	>99.99	>99.99	70.81	0.37	0.40	0.46

Figure 1. The distribution of mountain ash forests in the Central Highlands of Victoria (see Appendix S2 for data sources).

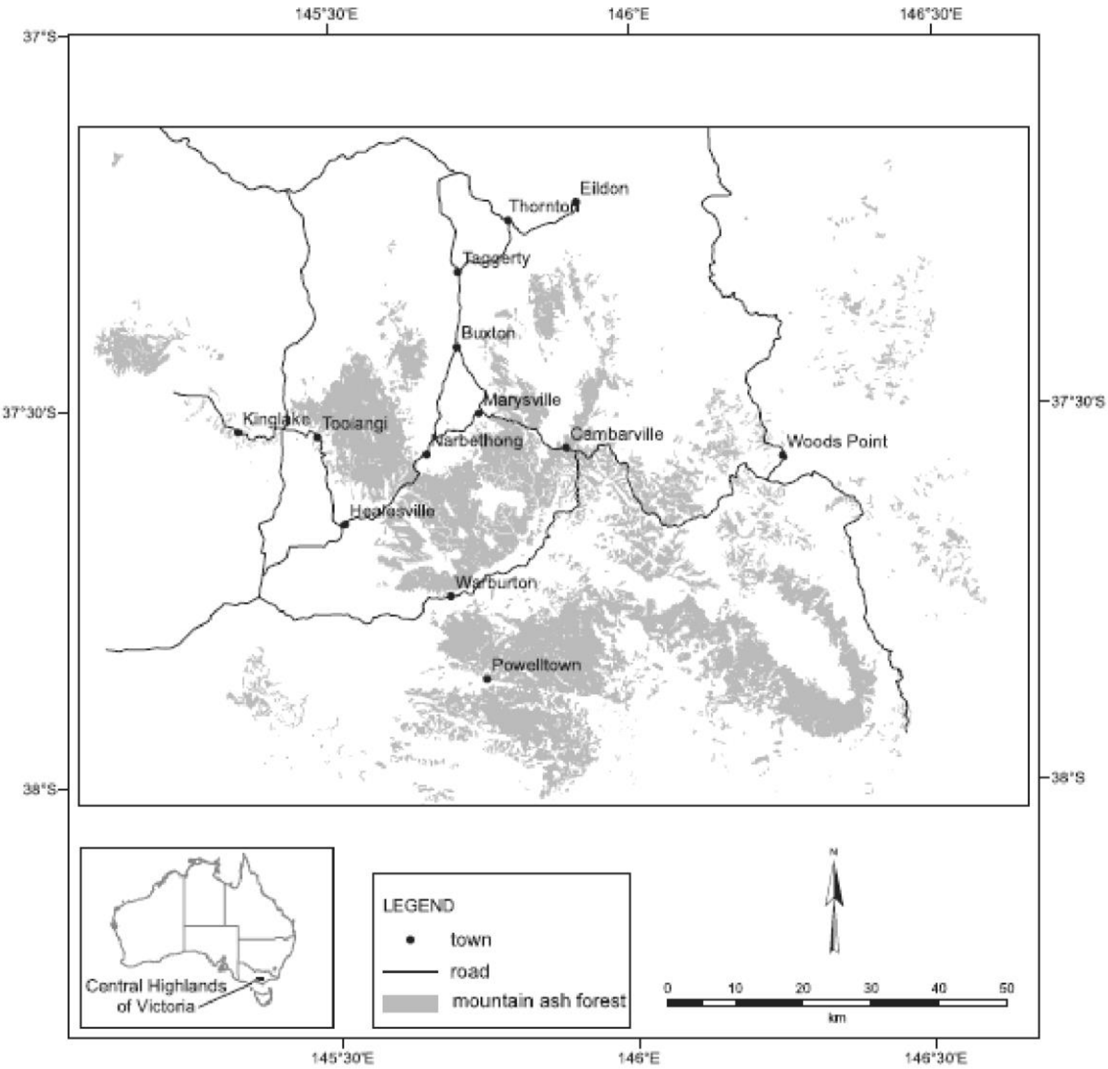

Figure 2: Conceptual model of salient ecological processes relevant to the assessment of the mountain ash ecosystem.

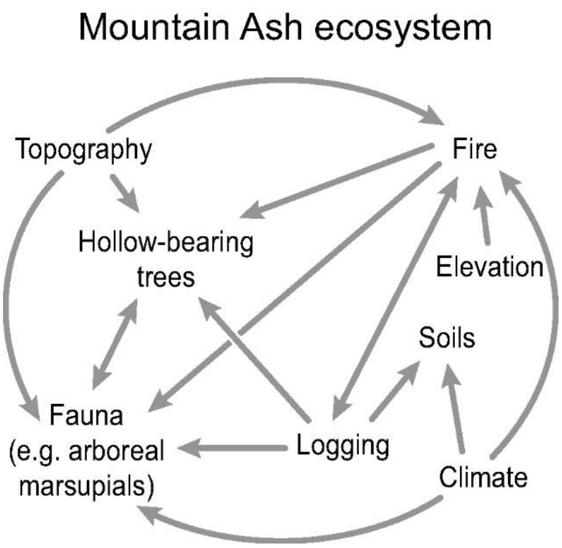

Figure 3. The estimated change in extent of the mountain ash ecosystem in the Central Highlands of Victoria since 1750.

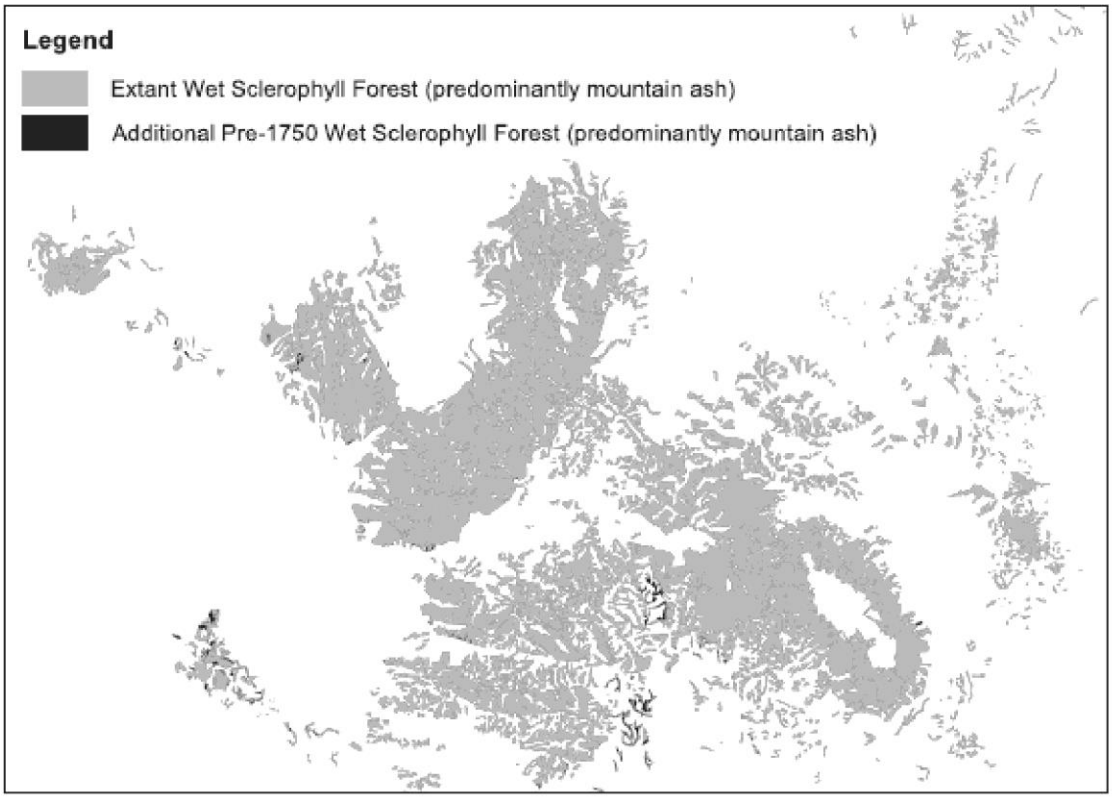

Figure 4. Mapped distribution of mountain ash ecosystem showing minimum convex polygon enclosing all occurrences (extent of occurrence) and occupied 10 x 10 km grid cells (area of occupancy). Light grey – all occupied cells. Dark grey – cells with ≥ 1% of cell area occupied.