Austral Ecology

Temporal fragmentation of a critically endangered forest ecosystem

Chris Taylor[1] and David B. Lindenmayer[1, 2]

[1]Fenner School of Environment & Society, The Australian National University, Canberra, ACT, 2601

[2]Threatened Species Recovery Hub, National Environmental Science Program, Fenner School of Environment & Society, The Australian National University, Canberra, ACT, 2601

Corresponding author

david.lindenmayer@anu.edu.au; Tel 02 61250654

Fenner School of Environment & Society, The Australian National University, 141 Linnaeus Way, Canberra ACT 2601

Running head: Critically endangered ecosystem fragmentation

ACKNOWLEDGEMENTS

The authors wish to acknowledge the Koori First Nations People of our study area, the GunaiKurnai, Taungurung and Wurundjeri People, upon whose respective lands this study was conducted and upon which their Sovereignty was never ceded. We wish to acknowledge their Elders past, present and emerging. This study was funded by the Caponero Grant; the Department of Environment, Land, Water and Planning; Parks Victoria; and donations from the public. Claire Shepherd assisted in editorial aspects of manuscript preparation. Comments by Professor Harry Recher and anonymous reviewer improved an earlier version of the manuscript. This paper is dedicated to the memory of our friend and colleague, Dr David Blair.

Temporal fragmentation of a critically endangered forest ecosystem

ABSTRACT

Landscape change and habitat fragmentation is increasingly affecting forests worldwide. Assessments of patterns of spatial cover in forests over time can be critical as they reveal important information about landscape condition. In this study, we assessed landscape patterns across the Mountain Ash (*Eucalyptus regnans*) and Alpine Ash (*Eucalyptus delegatensis*) forests in the Central Highlands of Victoria between 1999 and 2019. These forests have experienced major disturbance over the past 20 years through a major fire (in 2009) and extensive industrial logging. We found that around 70% and 65% of the Mountain Ash and Alpine Ash forest areas, respectively, were either disturbed or within 200 m of a disturbed area. Inclusion of planned logging increased these disturbance categories to 72% and 70%, respectively. We also found that the isolation of Mountain Ash core areas (patches of undisturbed forest > 1000 ha) increased significantly ($P<0.05$) over our study period, with the proximity between disturbed areas conversely increasing significantly ($P<0.05$). This means that continued and planned disturbance through industrial logging will have an amplified adverse effect on remaining undisturbed ash forest patches which will become smaller and more dispersed across the landscape.

Keywords: Mountain Ash forest, Alpine Ash forest, disturbance, logging, fire, fragmentation

INTRODUCTION

Landscape change and habitat fragmentation have been extensively studied (Saunders et al. 1991; Lindenmayer & Fischer 2006; Haddad et al. 2015; Fletcher et al. 2018) and identified as a major driver of species loss globally (Baillie et al. 2004; Betts et al. 2017). Habitat fragmentation is increasingly affecting forests worldwide (Kettle & Koh 2014; Watson et al. 2018) by reducing fragment size, increasing the isolation of patches and creating more edge environment (Ries et al. 2004; Lindenmayer et al. 2008). Haddad et al. (2015) found that nearly 20% of the world's remaining forests are within 100 m of an edge, and 70 % are within 1 km of an edge. These effects can lead to the decline of populations, restrict animal movement and disrupt gene flow (Crooks et al. 2017) as well as alter key ecosystem processes (Fischer & Lindenmayer 2007; Watson et al. 2018).

Much of the work on landscape change and habitat fragmentation has focused on agricultural landscapes where the original cover often has been removed and replaced by crops or pastures for domestic livestock (Saunders et al. 1991; IPBES 2019). Such a focus on agricultural landscapes is understandable given that agricultural development is a major driver of biodiversity loss globally (Maxwell et al. 2016; IPBES 2019). However, assessments of the effects of human modification of forest ecosystems is often more challenging than agricultural landscapes (Lindenmayer & Fischer 2006). This is because natural forests can regenerate after human and natural disturbance and there can be a less marked physical, structural and ecological contrast between human-modified areas and remaining undisturbed sites than where the surrounding landscape is cleared for crops and pastures (Harper et al. 2005; Lindenmayer 2016).

Assessments of patterns of spatial cover in forests can be critical as they reveal important information about landscape condition (Franklin & Forman 1987; Li et al. 1993) as well as their ability to support key elements of the biota (Phalan et al. 2019). In the study reported here, we completed a detailed spatial assessment of the Mountain Ash (*E. regnans*) and Alpine Ash (*E. delegatensis*) forests in the Central Highlands of Victoria (which we collectively refer to as 'ash' forests). We targeted these ecosystems for analysis for several reasons. First, the Mountain Ash ecosystem has been classified as Critically Endangered under the formal Red-listed Ecosystem approach developed by the IUCN (Burns et al. 2015) and understanding patterns of spatial cover is important for predicting their future ecological integrity (Lindenmayer & Sato 2018). Second, Mountain Ash and Alpine Ash forests support habitat for a range of high profile but rapidly declining populations of species of conservation concern such as the Critically Endangered Leadbeater's Possum (*Gymnobelideus leadbeateri*) and the vulnerable Greater Glider (*Petauroides volans*) (Lindenmayer & Sato 2018). Some of these species are potentially vulnerable to the spatial arrangement of suitable habitat patches in ash forests (Possingham et al. 1994; Taylor et al. 2017). Third, both the Mountain Ash and Alpine Ash forests have been targeted for intensive and extensive logging for many years by the native forest timber industry (DCFL 1986; Flint & Fagg 2007) and there are proposals to further expand the amount of forest that is clearcut over the next 5-10 years (VicForests 2019a). These same forests also have been subject to large-scale s in the past 35 years (Cruz et al. 2012; Lindenmayer et al. 2019b). Finally, the extent of disturbance in Mountain Ash and Alpine Ash forests has significant impacts on levels of carbon storage (Keith et al. 2014b; Keith et al. 2014a) as well as the production of water (Langford 1976; Langford et al. 1982; Taylor et al. 2019). Given these impacts, it is important to quantify how past disturbances have influenced spatial patterns of forest cover in the region and how additional human disturbances may further influence future patterns forest cover.

Specifically, we sought to answer several inter-related questions associated with the spatial cover in Mountain Ash and Alpine Ash ecosystems in the Central Highlands of Victoria using a number of landscape metrics over a time period of 20 years.

- How much of the Mountain Ash and Alpine Ash forests have been disturbed by logging and fire over the past 20 years?
- What is the spatial extent of Mountain Ash and Alpine Ash forest core areas?
- What is the proximity of Mountain Ash and Alpine Ash forest core areas to each other?

- How have disturbance patterns across these forest areas changed over the past 20 years?
- How will patterns of forest cover change if areas currently proposed for logging are in fact logged?

We focused on the amount of forest fragmentation across the landscape, specifically on changes in core area size, amount of edge created and the isolation of patches (Fahrig 2003) across Mountain Ash and Alpine Ash forest. Increased fragmentation across the landscape was indicated by low core area size and increasing isolation for those areas (Wang et al. 2014) as well as increasing size and proximity of disturbed patches. We selected a 20-year time period for our analysis because it provided a time period of 10 years before and 10 years after the February 2009 fires, which was an extensive disturbance across the region (Cruz et al. 2012). In addition, 20 years is approximately the period of enhanced flammability of regenerating vegetation that has been reported by Zylstra (2018). The landscape study presented here provided an opportunity to analyse the flow-on effects of cumulative disturbance on remaining patches of forest (Lindenmayer & Burgman 2005). These impacts can often be concealed when only the area disturbed in a single period (i.e. a given year) is reported (VAFI 2016), with the consequence that the cumulative impacts of past disturbance and levels of associated fragmentation are ignored.

METHODS

Study area

Our study area focused on the ash forests within the Central Highlands Regional Forest Agreement (RFA), located in the Australian state of Victoria between 40-130 km to the north and east of the city of Melbourne (Fig. 1). The Central Highlands RFA area covers an area of 1.13 million ha, with ash forests encompassing ~ 190,000 ha (Table 1). This includes the largest collective area of Mountain Ash forest remaining in mainland Australia (137,000 ha) (Fig. 2).

Land tenure analysis

The ash forests are covered by multiple land tenures, including protected areas (in the form of dedicated reserves) and areas available for resource extraction, such as State Forests. The datasets underpinning our land tenure analysis were the Collaborative Australian Protected Areas Database (CAPAD) and Forest Management Zones dataset (DEE 2016; DELWP 2019a). Protected areas of Mountain Ash and Alpine Ash forests were established in their current form when Australian and State governments agreed to a Comprehensive, Adequate and Representative (CAR) reserve system (Australian Government 1992). This system was to consist of dedicated reserves, informal reserves and other areas on public land that were protected by prescriptions for land use management.

The dedicated reserve system was informed by the IUCN Commission for National Parks and Protected Areas (CES 2018), which consists of strict nature reserves (Ia), wilderness areas (Ib), national parks (II), natural monuments or features (III), habitat or species management areas (IV), protected landscapes/seascapes (V) and protected areas with limited use of natural resources (VI).In Australia, a dedicated reserve is an area secured under parliamentary action, either by the Commonwealth Government or by a State Government (JANIS 1997).

The CAR reserve system outside the formal reserve system in State Forests is composed of informal protected areas and areas excluded from logging (DEPI 2014a). These areas were established under approved forest management plans as Special Protection Zones (SPZ) (DNRE 1998) and under the Code of Forest Practices for Timber Production (which is the regulatory document to which logging in native forests must comply) as Code of Forest

Practice Exclusion zones (DEPI 2014a)). However, these zones are not considered secure because they are not gazetted under legislation (JANIS 1997).

Forest across public land outside the CAR reserve system is where logging is permitted under the Code of Forest Practices for Timber Production (DEPI 2014a) and incorporated management standards (DEPI 2014b). This land area is also designated State Forest and it covers three zones: (i) General Management Zone (GMZ); (ii) Special Management Zone (SMZ); and (iii) Historical Reserves (DNRE 1998). Logging is generally prioritised in General Management Zones. Special Management Zones requires logging operations to be modified in an attempt to conserve areas of high landscape value. However, Special Management Zones do not constitute informal protected areas. Logging is also permitted in Historic Reserves (DNRE 1998).

Forest and disturbance input data

We used several datasets to calculate the extent of disturbance in Mountain Ash and Alpine Ash forests. We sourced data on the extent of ash forests from the State Forest Resource Inventory (SFRI) (DSE 2007b), Melbourne Water Vegetation mapping (Mackey et al. 2002) and Ecological Vegetation Class datasets (DELWP 2019b). The most detailed forest dataset is the SFRI, which mapped forest type at a stand level or at the scale of 1:25,000. The SFRI provided a standardised statement of the Victoria's State Forests and has been used for forecasting wood yields, strategic planning and a range of other investigations, such as old growth mapping (DSE 2007b). We extracted data on the extent of Mountain Ash, Alpine Ash and Shining Gum (*Eucalyptus nitens*) from this dataset. We combined Shining Gum with the Alpine Ash forest type, because this forest occurs as smaller patches adjacent to Alpine Ash forest.

SFRI data have been compiled only for State Forests and do not include dedicated reserves or private land. We sourced vegetation mapping data for the dedicated reserve system from Melbourne Water Vegetation mapping (Mackey et al. 2002) and the Ecological Vegetation Class dataset (DELWP 2019b). The Melbourne Water forest mapping focused on the Maroondah and O'Shannassy Water Catchments, which are located in the Yarra Ranges National Park. For remaining parts of the dedicated reserve system and areas of forest on private land, we sourced forest type data from the EVC dataset, which describes broad Ecological Vegetation Class groups and subgroups (DELWP 2019b). The layer was designed for use at a large scale (1:25,000 to 1:100,000). We used the EVC subgroups of Wet Forests and Montane Wet Forests because these EVCs aligned with the extent of Mountain Ash and Alpine Ash forests, respectively.

Disturbance data focused on clearfell logging, forests where high severity fires occurred, cleared areas designed to act as fire fuel breaks and main roads. We sourced data on the extent of roads and tracks from the VicMap Transport Road Network (DELWP 2019c). This dataset featured the state-wide road network, including roads, highways, freeways and tracks. We focused on major road networks and access roads, including highways, freeways, sealed roads, and unsealed access roads exceeding seven metres in width. We cross-checked each road and track against recent Landsat Satellite imagery (USGS 2019). We excluded smaller tracks, such as minor 4x4 tracks and walking tracks as these could not be mapped reliably using Landsat Satellite imagery. Around 600 km of fire fuel breaks were cut following the 2007 s (DSE 2007a). These were established around the water catchments for Melbourne and expanded a comparatively smaller existing network of fuel breaks previously cut following the 1939 wildfires. We sourced data on the extent of fuel breaks across the study area from a proposal published by the then Department of Sustainability and Environment in 2007 (DSE 2007a) and cross-validated this extent with Landsat Imagery dated 18 October 2018 (USGS 2019).

We sourced data on the areas disturbed by logging from a Logging History dataset, which represented the spatial extent of the most recent logging activity recorded for any given area within state forest (DJPR 2019). This dataset stored details of the last time a forest was known to be logged, the tree species logged, and the logging method used. It represented a consecutive overlay of all years, from 1961-62 to the most recent logging seasons. We focused our analysis on the most intensive logging methods of “Clearfelling”, “Clearfelling Salvage”, “Seed Tree” and “Regrowth Retention Harvesting”, of which the first three have been used extensively throughout the ash forests of Victoria (Squire et al. 1991; Lutze et al. 1999).

The clearfelling method involves the removal of almost all the commercial trees from a coupe in one integrated operation (Flint & Fagg 2007). Remaining forest debris is burnt in an intense planned fire and the seeds of the commercially preferred eucalypt trees are then dropped onto an ash bed (Florence 1996). Clearfelling with seed trees involves retaining a selected number of trees, around 10% of the total initial basal area, on a logging coupe to provide a seed source (Flint & Fagg 2007). This method has been used primarily where sufficient trees can be retained to reseed the entire coupe (Florence 1996). Similar to clearfelling, a high intensity planned burn is applied to the logged site (Flint & Fagg 2007). The sizes of clearfelled coupes and areas cut by clearfelling with seed tree retention are generally up to 40 hectares in size and can be aggregated up to 120 ha over five years (DEPI 2014b). Clearfelling salvage is conducted in forest previously burnt in a fire (VicForests 2018). Coupe sizes range up to 120 ha in Alpine Ash or Mountain Ash dominated forest and no size restrictions apply to aggregates (DEPI 2014b). “Regrowth Retention Harvesting” or variable retention is where patches of forest are retained within the forest area being logged (Lindenmayer et al. 2019a). The intent of variable retention across ash forests is to maintain an average of 30+% (by area) of tree cover across gross coupe area where possible, retain >

10 habitat trees per hectare where possible, and/or ensure gaps between retained vegetation do not exceed 150 metres (VicForests 2019b). We excluded less intensive logging methods from our analyses such as “single tree selection” and “thinning”. This was because they do not create edges in a manner similar to clearfell logging.

For ash forests burned by wildfires, we focused on the highest severity wildfire classes: crown consuming and crown scorching fires (Taylor et al. 2014). Crown consuming fires are those where 70–100% of the canopy is burnt and consumed in a fire. Crown scorching fires are those where 60–100% of eucalypt and non-eucalypt canopies are scorched, but the leaves remain on the branches immediately following the fire (DELWP 2019d). We deemed these two classes to be “high severity” fire as they often result in tree death in ash forests (see Smith & Woodgate 1985; Vivian et al. 2008; Bowman et al. 2016). These high fire severity impacts can create edges between areas of fire killed ash forest and those areas sustaining lower severity impacts or ash forest remaining unburnt. Similar impacts have been observed in North America across conifer forests burnt in mixed severity fires (Lentile et al. 2005; Donato et al. 2009). We extracted data on the spatial distribution of the high fire severity classes from Victorian Bushfires Severity Map 2009 (Taylor et al. 2014; DELWP 2019d). We did not include lower severity fires as trees can survive them.

Spatial analysis

We used the Euclidean Distance tool within ArcGIS 10 (ESRI 2011) to generate distance raster grids across the study region and calculate the distance of a cell from its nearest edge (Joppa et al. 2008; Crooks et al. 2017). We generated six Euclidean Distance rasters detailing the distance from disturbed area boundaries for the years 1999, 2004, 2009, 2014, 2019 and 2019 inclusive of the current Timber Release Plan (TRP) which details planned logging (VicForests 2019a). We generated the rasters beyond our study area boundary to ensure the edge of our analysis boundary did not influence distance from a disturbed area edge in the ash forests of our study area. We then clipped each Euclidean Distance raster to include only the Mountain Ash and Alpine Ash forests within the Central Highlands RFA area to create two separate input rasters for each forest type.

Our analysis of Mountain Ash and Alpine Ash forest fragmentation used the landscape metrics of core areas, disturbed area edge and proximity index. As the Euclidean Distance rasters generated were continuous, we grouped the distances into four proximity groups: 1) site of disturbance; 2) <200 m from disturbance; 3) <1000 m from disturbance; and 4) >1000m from disturbance. The 1000 m threshold was determined to be the minimum distance from disturbance needed for the persistence of the Critically Endangered Leadbeater's Possum (Lindenmayer et al. 1993; Lindenmayer et al. 2013). We classified locations >1000 m from a disturbance as "core areas". We selected a threshold distance of 200 m to reflect current government policy of excluding logging by 200 m from locations where Leadbeater's Possum had been detected (LPAG 2014).

We used the program FRAGSTATS (McGarigal & Marks 1995; McGarigal 2015) to calculate the length of disturbed forest edge and proximity index. FRAGSTATS provides a choice of landscape metrics to compute categorical map patterns. We calculated a disturbed forest edge length for locations where disturbed ash forest adjoined non-disturbed ash forest. We did not include boundary edges of the Mountain Ash and Alpine Ash forests extent. This was because these boundaries adjoined other forest types, such as mixed species forest and cool temperate rainforest (Lindenmayer et al. 2015), which were not part of our analysis. As the distribution of ash forest can occur in patches interspersed with other forest types, we used a Mean Proximity Index (Gustafson & Parker 1992) for each proximity group area to measure the relative isolation of patches over time in response to disturbance. The index was calculated using the relationship:

$$PX_i = \sum \frac{S_k}{n_k} \qquad 1$$

where PX_i is the proximity index for focal patch *i* within a specified search distance, s_k is the area of patch *k* within the search areas and n_k is the nearest neighbour distance between a grid cell of the focal patch and the nearest grid cell of patch *k* (Turner & Gardner 2015). The mean proximity metric provides a dimensionless index and it was used here as a comparison between analysis years. A low value for the index indicated ash forest patches of a specific proximity group were relatively isolated from other ash forest patches of the same group. This means that patches were comparatively distant from each other. High index values indicated that patches were relatively close to other similar patches within the specified search distance. We set the search radius to 1000m to align with the foraging range of the Leadbeater's Possum (Lindenmayer et al. 2013).

RESULTS

Land tenure allocation

Mountain Ash and Alpine Ash forest comprise ~137,000 and 55,000 hectares, respectively in our study area (Table 1) (Fig. 3). Approximately 44% and 45% of Mountain Ash and Alpine Ash forest, respectively, occurs in land tenures where logging is permitted (General Management Zones, Special Management Zones, and Historic Reserves). The area of Mountain Ash and Alpine Ash forest assigned to State Forests, which also includes informal protected areas, is 67% and 71%, respectively. The areas of Mountain Ash and Alpine Ash forest assigned dedicated reserves is 28% and 29%, respectively.

Disturbed areas and Edges

Clearfell logging across the ash forests of the Central Highlands RFA area steadily increased from 1999 to 2019 (Fig. 4). Between 1960 and 1999, 19,714 ha and 5,206 ha of Mountain Ash and Alpine Ash forest were clearfelled, respectively, equating to around 14% and 9% of the total respective forest areas. By 2019, 32,276 ha and 11,716 ha of Mountain Ash and Alpine Ash forest had been clearfell logged (Table 2), equating to around 24% and

22% of the total respective forest areas. For Mountain Ash, most of the logging occurred between 1982 and 2015. Annual areas of clearfell logging in Alpine Ash increased steadily from 1962, peaking at 789 ha being cut in 2015 and then declined.

A total of 21,132 ha of Mountain Ash and 7,969 ha of Alpine Ash forest was burned at high severity in the 2009 wildfires. This equated to around 16% and 15% of Mountain Ash and Alpine Ash forest areas, respectively. Overlapping areas of high severity wildfire and clearfell logging consisted of 5,313 ha and 2,552 ha for Mountain Ash and Alpine Ash forest, respectively. By 2019, around 48,095 ha and 17,133 ha or 35% and 31% of Mountain Ash and Alpine Ash forest, respectively, had been impacted by a high severity disturbance either through clearfell logging or by the 2009 wildfires (Table 2).

An extensive network of roads and fuel breaks has been constructed in Mountain Ash and Alpine Ash forests. We identified approximately 715 km of major roads, including highways, sealed roads, and unsealed roads exceeding seven metres in width across Mountain Ash and Alpine Ash forests. Some of these roads align with an extensive network of fuel breaks, most of which were cut following the 2007 and 2009 wildfires. These were between 20-40 metres wide and the combined total distance of these fuel breaks across the ash forest area was 185 km.

Inclusive of clearfell logging, roads, fuel breaks and fire, the length of edge between disturbed and non-disturbed areas across Mountain Ash and Alpine Ash forests doubled between 1999 and 2019 (Fig. 5). In 1999, the length of edge was 4,661 km and 1,360 km for Mountain Ash and Alpine Ash forest, respectively. By 2019, this length of edge had increased to 8,752 km and 3,190 km for Mountain Ash and Alpine Ash forest, respectively. Inclusive of the 2019 TRP, the length of edge will increase to 9,282 km and 3,441 km for Mountain Ash and Alpine Ash forest, respectively. The largest increase in the length of edge was a result of the 2009 wildfires, which along with continued logging, increased the length of edge by 2,697 km and 1,129 km between 2004 and 2009 for Mountain Ash and Alpine Ash forest, respectively.

Spatial and temporal distribution of fragmented forest areas

We calculated large areas of Mountain Ash forest were within close proximity of disturbed areas. In 1999, 60,958 ha or 44% of Mountain Ash forest was either disturbed or within 200m of a disturbed area. By 2019, this increased to 94,058 ha or nearly 70% of the total Mountain Ash forest area. Inclusive of the 2019 TRP, the amount of directly disturbed forest and forest within 200m of a disturbed area will increase to 98,590 ha or 72% of the total Mountain Ash forest area. Conversely, there was a decline in the core areas (i.e. those places >1000 m from a disturbed area), from 29,614 ha in 1999 to 9,382 ha by 2019, a decrease of 68%. Planned logging under the TRP will further decrease core areas of Alpine Ash forest to 8,000 ha.

In 1999, 18,475 ha or 34% of Alpine Ash forest was either disturbed or within 200m of a disturbed area (Fig 6). By 2019, this increased to 35,447 ha or 65% of the total Alpine Ash forest area. Inclusive of the 2019 TRP, the disturbed area and forest within 200m from a disturbed area will increase to 38,472 ha or 70% of the total Alpine Ash forest area. Conversely, there was a decline in the core areas, from 14,773 ha in 1999 to 4,830 ha by 2019, a decrease of 67%. Planned logging under the TRP will further decrease core areas of Alpine Ash forest to 4,287 ha.

We found evidence of significant ($P<0.05$) changes in the isolation and proximity across all patch types for both Mountain Ash and Alpine Ash forest (Table S1). In 1999, the mean proximity index was 1265 for core areas of Mountain Ash. This value had declined to 156 by 2019 (Fig. 8). This means that the isolation of remaining core areas of Mountain Ash forest has increased significantly ($P<0.05$) over the past 20 years. In contrast, the mean proximity index for disturbed areas increased significantly ($P<0.05$) (Table S2) from 52 in

1999 to 835 in 2019, and 1059 under logging associated with the 2019 TRP. This means that the proximity between logged and burnt patches of Mountain Ash has increased between 1999 and 2019.

For Alpine Ash forest, the mean proximity index for core areas declined from 385 in 1999 to 303 in 2019 (Fig. 8). We detected no significant changes ($P<0.05$) across this proximity group (Table S3). This means that the proximity between core areas of Alpine Ash has remained largely consistent. However, we detected significant changes ($P<0.05$) in mean proximity for the <200 m and <1000 m proximity groups from disturbed areas. For the <1000 m proximity group, the mean proximity index had decreased from 255 in 1999 to 103 by 2019. It further decreased to 90 with the inclusion of the 2019 TRP. We detected significant changes in the proximity index between 1999 and 2009 (Table S4), meaning that the isolation of patches of Alpine Ash <1000m from disturbance has increased. For the <200 m proximity group, the proximity index decreased from 680 in 1999 to 439 by 2019. It will further decrease to 196 with the inclusion of the 2019 TRP. We detected significant ($P<0.05$) changes between 1999 and 2009 (Table S5), meaning that the proximity between areas of Alpine Ash <200m from disturbed areas has increased significantly ($P<0.05$) over the past two decades. The mean proximity index for disturbed areas also increased significantly ($P<0.05$) (Table S6) for Alpine Ash forest from 1999 (when it was 30) to 772 by 2019. The inclusion of the 2019 TRP will result in a further increase to this index to 1020. This means that the proximity between disturbed patches across Alpine Ash forest will increase.

Disturbance of ash forest by tenure

We found that the evidence of disturbance across the ash forest was greatest in land tenures where logging is permitted (Fig. 9) and least within the dedicated reserve system. In 1999, disturbed areas across land tenure where logging is permitted was 20,371 ha or 24% of the total ash forest (i.e. for both Mountain Ash and Alpine Ash forests) for this tenure. For the same year, 47,871 ha or 56% of the total forest was disturbed or within 200 metres of disturbance. The extent of core areas was limited to 7,131 ha or 8% of the ash forest. By 2019, the area of disturbance increased to 46,344 ha or 51% of the ash forest where logging is permitted. Around 72,361 ha or 85% of the total forest was disturbed or within 200 metres of disturbance. The extent of core areas decreased to 1,529 hectares or 2% of the total ash forest within tenure where logging is permitted. Inclusive of the 2019 TRP, our analyses indicated that: **(1)** core areas will comprise just 583 ha or < 1% of the total ash forest, **(2)** the extent of disturbed areas will increase to 55,511 ha or 65% of the total ash forest, and **(3)** 76,792 ha or 90% of the total forest logging tenure area will be either disturbed or within 200 metres of a disturbance.

The area of disturbance across informal protected areas and dedicated reserves increased between 1999 and 2019. For dedicated reserves, the disturbed area increased from 611 ha to 10,215 ha between 1999 and 2019. This increase was a result of the 2009 wildfires. For informal protected areas, the area of disturbance increased from 4,839 ha in 1999 to 10,216 ha by 2019. This increase was a combination of the 2009 wildfires and previously logged areas being added to the informal protected area network as a result of 200 metre exclusion zones established following detections of Leadbeater's Possum across land tenures previously allocated to logging.

DISCUSSION

Several major studies have highlighted the critical importance of intact forests for biodiversity conservation and the maintenance of key ecological processes such as carbon storage and water production (Gibson et al. 2011; Watson et al. 2018). Conversely, biodiversity can be threatened and key ecological processes impaired in highly disturbed forests (Lindenmayer & Fischer 2006; Haddad et al. 2015; Phalan et al. 2019). Spatial analyses of patterns of natural and human disturbance can provide an indication of the extent to which forests remain intact or are disturbed (McGarigal 2015). In the study reported here, we quantified the extent of forest disturbance resulting from logging and fire in the ash forests of the Central Highlands of Victoria.

Our spatial analyses underscore the very high levels of disturbance in both Mountain Ash and Alpine Ash forests. Indeed, 70% of the Mountain Ash ecosystem was either disturbed or within 200 metres of a disturbed area. Similar patterns characterize the Alpine Ash ecosystem with 65% of the forest either disturbed or within 200 metres of a disturbed area (Fig. 7). This impact has been compounded by the isolation of remaining Mountain Ash forest core areas. As core areas of Mountain Ash forest have decreased in size, they have become increasingly isolated. Furthermore, proximity between disturbed areas has increased for both Mountain Ash and Alpine Ash forest, which means that disturbed areas are becoming more concentrated and extensive. Notably, our analyses also indicated that significant disturbance has occurred in the past ten years (Tables S2 and S6) and therefore since the 2009 wildfires. This is due to widespread industrial clearfelling. We discuss these and other findings in the remainder of this section and conclude with some key recommendations for management.

The extent of disturbance and forest fragmentation

The high levels of disturbance that characterize the Mountain Ash and Alpine Ash forests was expected given the extent of the 2009 wildfires (see Cruz et al. 2012; Taylor et al. 2014). What was unexpected, however, was the large amounts of logging-related disturbance that had occurred in the decade since the 2009 wildfires. These are primarily clearfelled logging coupes that have been planned and logged under successive Timber Release Plans (e.g. VicForests 2017; VicForests 2019a). These human-generated disturbances have meant that the levels of disturbance in forest areas within 200 metres of disturbance in Mountain Ash and Alpine Ash forest have increased by 9,014 ha and 7,309 ha for Mountain Ash and

Alpine Ash, respectively, since 2009. Notably, during this time there has been limited fire-related disturbance.

The extent of disturbance in Mountain Ash and Alpine Ash forest has important implications for forest biodiversity and ecological processes. First, additional logging and additional fire in Mountain Ash and Alpine Ash forest increases the landscape-level dominance of young regenerating forest which is, in turn, prone to additional high-severity disturbance such as crown-scorching (Taylor et al. 2014; Zylstra 2018). Fires can have significant negative impacts on a range of elements of the biota including arboreal marsupials (Lindenmayer et al. 2013) and birds (Lindenmayer et al. 2019b) as well as on populations of large old trees (Lindenmayer et al. 2018a) and soil microbiomes (Bowd et al. 2019). Second, further disturbances such as additional logging in Mountain Ash and Alpine Ash forests will drive a decline in ecosystem integrity; for example, additional logging coupes in wood production landscapes accelerate rates of decay and collapse of large old trees in remaining uncut areas (Lindenmayer et al. 2018b). This will, in turn, have negative effects on species that are dependent on such trees such as hollow-using vertebrates, many of which are already exhibiting marked patterns of population decline (Lindenmayer & Sato 2018).

Advocates for ongoing widespread logging of Mountain Ash and Alpine Ash forests claim that a large amount of the forest outside of the dedicated reserve system remains unlogged and therefore current off-reserve management is sufficient for conservation. Our analyses have empirically demonstrated the heavily disturbed and highly fragmented nature of the forest estate outside the dedicated reserve system. The wood production landscape is comprised primarily of relatively narrow filter strips and streamside reserves (typically 40 m in width) between otherwise clearfelled areas, as well as small patches of uncut forest on steep and rocky terrain. While it is important that these areas remain uncut, they are unlikely to support viable areas of habitat for some elements of the biota. For example, narrow retained linear strips are unsuitable habitats for some species of arboreal marsupials (Lindenmayer et al. 1993) and steep and rocky areas and gullies are avoided by some species of birds (Lindenmayer et al. 2009)). Conversely, recent analyses show that areas that are currently being logged, or proposed for logging in the next few years under the TRP, have high conservation value for the 70 threatened forest-dependent species in Victoria (Taylor & Lindenmayer 2019). Therefore, additional logging-related disturbances in wood production Mountain Ash and Alpine Ash forests will have amplified the negative impacts on biodiversity in these ecosystems (Taylor & Lindenmayer 2019). Clearfell logging not only directly increasing disturbed area, but also of increasing the likelihood of disturbance by fire. That is, it has a double disturbance effect. Moreover, the influence of more flammable young post-logging regenerating patches in the landscape (see Zylstra 2018) is non-additive. It is characterized by a threshold where fragmentation with flammable patches tips the entire landscape into a more flammable state (Tiribelli et al. 2018). An important caveat with the work reported in this study is that we have not connected the forest fragmentation metrics to biodiversity responses. That is, we have not quantified the responses of various elements of the biota to temporal changes in spatial patterns of forest landscape cover. Such work was beyond the scope of this study, but it will be an important complementary investigation to the one reported here.

The extent of the road network

Our spatial analyses have revealed that large parts of the Mountain Ash and Alpine Ash ecosystems are heavily roaded. The wood production forests were characterized by 715 km of primary roads and a further 1,418 km of smaller, secondary roads. Roads can have a range of negative impacts in forest environments (Forman 2002; Laurance & Arrea 2017), including acting as point sources of fire ignitions (Collins et al. 2015), providing a conduit for the movement of feral animals (such as introduced predators), and being a source of weeds ("the car-borne" flora" (Wace 1977)). Of course, a large road network is required to transport pulpwood and sawlogs from the forest to mills. In addition, there are extensive lengths of tracks created within logged areas such as snig trails and boundary tracks around cutblocks and these can lead to suppressed levels of growth of regenerating forest after logging operations have been completed (Rab 1998).

Management implications and recommendations

Our findings have at least three significant implications for forest management. First, given the high level of disturbance that has already occurred in the Mountain Ash and Alpine Ash ecosystems, it is critical to reduce any further disturbances. We argue there is strong evidence to remove any further logging in forests dominated by Mountain Ash and Alpine Ash. Notably, this recommendation is consistent with that made by the Australian Government's Threatened Species Scientific Committee that there should be no further logging in montane ash forests, given its impacts on the Critically Endangered Leadbeater's Possum. This is also consistent with recommendations about ecosystem vulnerability and boosted levels of protection, given a formal IUCN assessment of the Mountain Ash ecosystem as being Critically Endangered made through the Red Listed Ecosystem process (Burns et al. 2015). Removal of logging is also important given the high levels of impact proposed cutting under the Timber Release Plan would have on areas of high conservation value, including on threatened forest-dependent species (Taylor & Lindenmayer 2019).

A second important implication of our analyses is the need for strengthened protection of the Alpine Ash ecosystem in the Central Highlands of Victoria which is almost as heavily disturbed as the Mountain Ash ecosystem. The importance of greater protection is emphasized by the fact that many areas of Alpine Ash elsewhere in Victoria have been subject to repeated fires in recent decades (Bowman et al. 2014) and are at risk of collapse with further reburning (Zylstra 2018). Finally, given the extent of the road network in wood production areas, we argue that consideration is given to removing some roads and rehabilitating the forest. This will require trade-off analysis to determine the disadvantages of reduced access for fighting fires relative to the benefits of reduce ignition points for arson.

CONCLUSIONS

The Mountain Ash and Alpine Ash forests of the Central Highlands Regional Forest Agreement area have undergone significant disturbance and fragmentation in the past 20 years, with further disturbance and fragmentation inevitable under planned logging operations. By 2019, approximately 70% and 65% of the Mountain Ash and Alpine Ash forest areas, respectively, were either disturbed or within 200m of a disturbed area. Inclusion of planned logging increases these disturbance categories to 72% and 70%, respectively. Disturbance and proximity to disturbance increased significantly between 1999 and 2019. Core areas of Mountain Ash have become fragmented, as indicated by a significant increase ($P<0.05$) in the isolation of remaining patches over the study time period. In contrast, proximity between disturbed areas has increased significantly ($P<0.05$) for both Mountain Ash and Alpine Ash forests. The inevitable consequences of continued logging will amplify the adverse impact on the remaining undisturbed ash forests.

REFERENCES

Australian Government (1992) *National Forest Policy Statement*. Advance Press, Perth.

Baillie J.E., Hilton-Taylor C. & Stuart S.N. (2004) *A Global Species Assessment*. International Union for Conservation of Nature, Gland, Switzerland.

Betts M.G., Wolf C., Ripple W.J., *et al.* (2017) Global forest loss disproportionately erodes biodiversity in intact landscapes. *Nature* **547**, 441-444.

Bowd E.J., Banks S.C., Strong C.L. & Lindenmayer D.B. (2019) Long-term impacts of wildfire and logging on forest soils. *Nature Geosci.* **12**, 113-118.

Bowman D.M., Williamson G.J., Prior L.D. & Murphy B.P. (2016) The relative importance of intrinsic and extrinsic factors in the decline of obligate seeder forests. *Glob. Ecol. Biogeogr.* **25**, 1166-1172.

Bowman D.M.J.S., Murphy B.P., Neyland D.L.J., Williamson G.J. & Prior L.D. (2014) Abrupt fire regime change may cause landscape-wide loss of mature obligate seeder forests. *Glob. Ch. Biol.* **20**, 1008-1015.

Burns E.L., Lindenmayer D.B., Stein J.*, et al.* (2015) Ecosystem assessment of mountain ash forest in the Central Highlands of Victoria, south-eastern Australia. *Austral Ecol.* **40**, 386-399.

CES (2018) *State of the Forests 2018 Report*. Commissioner for Environmental Sustainability Victoria, Melbourne.

Collins K.M., Price O. & Penman T. (2015) Spatial patterns of wildfire ignitions in south-eastern Australia. *Int. J. Wildland Fire* **24**, 1098-1108.

Crooks K.R., Burdett C.L., Theobald D.M.*, et al.* (2017) Quantification of habitat fragmentation reveals extinction risk in terrestrial mammals. *Proc. Natl. Acad. Sci. USA* **114**, 7635-7640.

Cruz M.G., Sullivan A.L., Gould J.S.*, et al.* (2012) Anatomy of a catastrophic wildfire: The Black Saturday Kilmore East fire in Victoria, Australia. *Forest Ecol. Manage.* **284**, 269-285.

DCFL (1986) *Victoria Timber Industry Strategy*. Government Statement No. 9. Victorian Government Department of Conservation, Forests and Lands, Melbourne.

DEE (2016) Collaborative Australian Protected Area Database. Australian Government Department of the Environment and Energy, Canberra. Available from https://www.environment.gov.au/land/nrs/science/capad/2016. Accessed 29 April 2019.

DELWP (2019a) Forest Management Zones - Simplified View. Victorian Government Department of Environment, Land, Water and Planning, Melbourne. Available from https://discover.data.vic.gov.au/dataset/forest-management-zones-simplified-view. Accessed 30 April 2019.

DELWP (2019b) Native Vegetation - Modelled 2005 Ecological Vegetation Classes (with Bioregional Conservation Status). Victorian Government Department of Environment, Land, Water and Planning, Melbourne. Available from https://discover.data.vic.gov.au/dataset/native-vegetation-modelled-2005-ecological-vegetation-classes-with-bioregional-conservation-sta. Accessed 29 April 2019.

DELWP (2019c) Vicmap Transport - Road Network. Victorian Government Department of Environment, Land, Water and Planning, Melbourne. Available at https://discover.data.vic.gov.au/dataset/vicmap-transport-road-network. Accessed 25 June 2019.

DELWP (2019d) Victorian Bushfires Severity Map 2009 (Polygons). Victorian Government Department of Environment, Land, water and Planning, Melbourne. Available from https://discover.data.vic.gov.au/dataset/victorian-bushfires-severity-map-2009-polygons. Accessed 25 June 2019.

DEPI (2014a) Code of Practice for Timber Production. Victorian Government Department of Environment and Primary Industries, Melbourne.

DEPI (2014b) Management Standards and Procedures for timber harvesting operations in Victoria's State forests 2014. Victorian Government Department of Environment and Prmary Industries, Melbourne.

DJPR (2019) Logging history overlay of most recent harvesting activities. Victorian Government Department of Jobs, Precincts and Regions, Melbourne. Available at https://discover.data.vic.gov.au/dataset/logging-history-overlay-of-most-recent-harvesting-activities. Accessed 17 July 2019.

DNRE (1998) Forest Management Plan for the Central Highlands. Victorian Government Department of Natural Resources and Environment, Melbourne.

Donato D.C., Fontaine J.B., Campbell J.L., Robinson W.D., Kauffman J.B. & Law B.E. (2009) Conifer regeneration in stand-replacement portions of a large mixed-severity wildfire in the Klamath–Siskiyou Mountains. *Can. J. Forest Res.* **39**, 823-838.

DSE (2007a) Strategic Firebreaks: Protecting Melbourne's Precious Water Supply. Victorian Government Department of Sustainability and Environment, Melbourne.

DSE (2007b) Victoria's statewide forest resource inventory: Central, Dandenong and Central Gippsland forest management areas. Victorian Government Department of Sustainability and Environment, Melbourne

ESRI (2011) *ArcGIS Desktop: Release 10*. Environmental Systems Research Institute, Redlands, CA.

Fahrig L. (2003) Effects of habitat fragmentation on biodiversity. *Ann. Rev. Ecol. Evol. Syst.* **34**, 487-515.

Fischer J. & Lindenmayer D.B. (2007) Landscape modification and habitat fragmentation: a synthesis. *Glob. Ecol. Biogeogr.* **16**, 265-280.

Fletcher R.J., Didham R., Banks-Leite C., *et al.* (2018) Is habitat fragmentation good for biodiversity? *Biol. Conserv.* **226**, 9-15.

Flint A. & Fagg P. (2007) *Mountain Ash in Victoria's State Forests. Silviculture reference manual No. 1*. Department of Sustainability and Environment, Melbourne.

Florence R.G. (1996) *Ecology and Silviculture of Eucalypt Forests*. CSIRO Publishing, Melbourne.

Forman R.T. (Eds) (2002) *Road Ecology. Science and Solutions*. Island Press, Washington.

Franklin J.F. & Forman R.T. (1987) Creating landscape patterns by forest cutting: ecological consequences and principles. *Landscape Ecol.* **1**, 5-18.

Gibson L., Lee M.L., Koh L.P.*, et al.* (2011) Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature* **478**, 378-381.

Gustafson E.J. & Parker G.R. (1992) Relationships between landcover proportion and indices of landscape spatial pattern. *Landscape Ecol.* **7**, 101-110.

Haddad N.M., Brudvig L.A., Clobert J.*, et al.* (2015) Habitat fragmentation and its lasting impact on Earth's ecosystem. *Sci. Adv.* **1**, e1500052.

Harper K.A., Macdonald S.E., Burton P.J.*, et al.* (2005) Edge influence on forest structure and composition in fragmented landscapes. *Conserv. Biol.* **19**, 768-782.

IPBES (2019) *IPBES Global Assessment Summary for Policymakers*. Intergovernmental Science-policy Platform on Biodiversity and Ecosystem Services (IPBES), United Nations.

JANIS (1997) *Nationally agreed criteria for the establishment of a comprehensive, adequate and representative reserve system for forests in Australia*. Joint ANZECC/MCFFA National Forest Policy Statement Implementation Sub-committee, Government of Australia, Canberra.

Joppa L.N., Loarie S.R. & Pimm S.L. (2008) On the protection of "protected areas". *Proc. Natl. Acad. Sci. USA* **105**, 6673 6678.

Keith H., Lindenmayer D.B., Mackey B.G.*, et al.* (2014a) Accounting for biomass carbon stock change due to wildfire in temperate forest landscapes in Australia. *PLOS One* **9**, e107126.

Keith H., Lindenmayer D.B., Mackey B.G.*, et al.* (2014b) Managing temperate forests for carbon storage: impacts of logging versus forest protection on carbon stocks. *Ecosphere* **5(6)**, Art. 75. [online] http://dx.doi.org/10.1890/ES1814-00051.00051.

Kettle C.J. & Koh L.P. (Eds) (2014) *Global Forest Fragmentation*. CABi International, Wallingford, United Kingdom.

Langford K. (1976) Change in yield of water following a bushfire in a forest of *Eucalyptus regnans*. *J. Hydrol.* **29**, 87-114.

Langford K.J., Moran R.J. & O'Shaughnessy P.J. (1982) The Coranderrk Experiment - the effects of roading and timber harvesting in mature Mountain Ash forest on streamflow and quality. In: *The First National Symposium on Forest Hydrology*. (eds E.M. O'Loughlin and L.J. Bren), pp. 92-102. Institution of Engineers, Canberra.

Laurance W. & Arrea I.B. (2017) Roads to riches or ruin? Global infrastructure expansion must balance social benefits and environmental hazards. *Science* **358**, 442-445.

Lentile L.B., Smith F.W. & Shepperd W.D. (2005) Patch structure, fire-scar formation, and tree regeneration in a large mixed-severity fire in the South Dakota Black Hills, USA. *Can. J. Forest Res.* **35**, 2875-2885.

Li H., Franklin J.F., Swanson F.J. & Spies T.A. (1993) Developing alternative forest cutting patterns: a simulation approach. *Landscape Ecol.* **8**, 63-75.

Lindenmayer D.B. (2016) Interactions between forest resource management and landscape structure. *Curr. Landscape Ecol. Rep.* **1**, 10-18.

Lindenmayer D.B., Blair D. & McBurney L. (2019a) Variable retention harvesting in Victoria's Mountain Ash (Eucalyptus regnans) forests (southeastern Australia). *Ecol. Proc.* **8**, Art. 2.

Lindenmayer D.B., Blair D.P., McBurney L. & Banks S.C. (2015) *Mountain Ash: Fire, Logging and the Future of Victoria's Giant Forests*. CSIRO Publishing, Melbourne.

Lindenmayer D.B., Blanchard W., Blair D. & McBurney L. (2018a) The road to oblivion – quantifying pathways in the decline of large old trees *Forest Ecol. Manage.* **430**, 259-264.

Lindenmayer D.B., Blanchard W., Blair D., McBurney L., Stein J. & Banks S.C. (2018b) Empirical relationships between tree fall and landscape-level amounts of logging and fire *PLOS One* **13(2)**, e0193132.

Lindenmayer D.B., Blanchard W., Blair D., Westgate M.J. & Scheele B.C. (2019b) Spatio-temporal effects of logging and fire on forest birds. *Ecol. Appl.*, https://doi.org/10.1002/eap.1999.

Lindenmayer D.B., Blanchard W., McBurney L.*, et al.* (2013) Fire severity and landscape context effects on arboreal marsupials. *Biol. Conserv.* **167**, 137-148.

Lindenmayer D.B. & Burgman M.A. (2005) *Practical Conservation Biology*. CSIRO Publishing, Melbourne.

Lindenmayer D.B., Cunningham R.B. & Donnelly C.F. (1993) The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-east Australia: IV. The presence and abundance of arboreal marsupials in retained linear habitats (wildlife corridors) within logged forest. *Biol. Conserv.* **66**, 207-221.

Lindenmayer D.B. & Fischer J. (2006) *Habitat Fragmentation and Landscape Change*. Island Press, Washington, D.C.

Lindenmayer D.B., Hobbs R., Montague-Drake R.*, et al.* (2008) A checklist for ecological management of landscapes for conservation. *Ecol. Lett.* **11**, 78-91.

Lindenmayer D.B. & Sato C. (2018) Hidden collapse is driven by fire and logging in a socioecological forest ecosystem. *Proc. Natl. Acad. Sci. USA* **115**, 5181-5186.

Lindenmayer D.B., Wood J., Michael D.*, et al.* (2009) Are gullies best for biodiversity? An empirical examination of Australian wet forest types. *Forest Ecol. Manage.* **258**, 169-177.

LPAG (2014) *Leadbeater's Possum Technical Report*. Leadbeater's Possum Advisory Group. Report to the Minister for Environment and Climate Change and the Minister for Agriculture and Food Security, Melbourne.

Lutze M.T., Campbell R.G. & Fagg P.C. (1999) Development of silviculture in the native State forests of Victoria. *Aust. Forestry* **62**, 236-244.

Mackey B., Lindenmayer D.B., Gill A.M., McCarthy M.A. & Lindesay J.A. (2002) *Wildlife, Fire and Future Climate: A Forest Ecosystem Analysis*. CSIRO Publishing, Melbourne.

Maxwell S., Fuller R.A., Brooks T. & Watson J. (2016) Biodiversity: The ravages of guns, nets and bulldozers. *Nature* **536**, 143-145.

McGarigal K. (2015) FRAGSTATS. University of Massachusetts, Amherst, Massachusetts.

McGarigal K. & Marks B.J. (1995) *FRAGSTATS: Spatial Pattern Analysis Program for Quantifying Landscape Structure. General Technical Report PNW-GTR-351*. US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Phalan B., Northrup J.M., Yang Z*., et al.* (2019) Impacts of the Northwest Forest Plan on forest composition and bird populations. *Proc. Natl. Acad. Sci. USA* **116**, 3322-3327.

Possingham H.P., Lindenmayer D.B., Norton T.W. & Davies I. (1994) Metapopulation viability analysis of the Greater Glider *Petauroides volans* in a wood production area. *Biol. Conserv.* **70**, 227-236.

Rab M.A. (1998) Rehabilitation of snig tracks and landings following logging of *Eucalyptus regnans* forest in the Victorian Central Highlands - a review. *Aust. Forestry* **61**, 103-113.

Ries L., Fletcher R.J., Battin J. & Sisk T.D. (2004) Ecological responses to habitat edges: Mechanisms, models, and variability explained. *Annu. Rev. Ecol. Evol. Syst.* **35**, 491-522.

Saunders D.A., Hobbs R.J. & Margules C.R. (1991) Biological consequences of ecosystem fragmentation: a review. *Conserv. Biol.* **5**, 18-32.

Smith R.B. & Woodgate P. (1985) Appraisal of fire damage for timber salvage by remote sensing in mountain ash forests. *Aust. Forestry* **48**, 252-263.

Squire R.O., Campbell R.G., Wareing K.J. & Featherston G.R. (1991) The mountain ash forests of Victoria: Ecology, silviculture and management for wood production. In: *Forest Management in Australia*. (eds F.H. McKinnell, E.R. Hopkins and J.E.D. Fox), pp. 38-57. Surrey Beatty and Sons, Chipping Norton.

Taylor C., Blair D., Keith H. & Lindenmayer D.B. (2019) Modelling water yields in response to logging and Representative Climate Futures. *Sci. Total Environ.* **688**, 890-902.

Taylor C., Cadenhead N., Lindenmayer D.B. & Wintle B.A. (2017) Improving the design of a conservation reserve for a critically endangered species. *PLOS One* **12**, e0169629.

Taylor C. & Lindenmayer D.B. (2019) The adequacy of Victoria's protected areas to conserve its forest-dependent fauna. *Austral Ecol.* **44**, 1076-1090.

Taylor C., McCarthy M.A. & Lindenmayer D.B. (2014) Non-linear effects of stand age on fire severity. *Conserv. Lett.* **7**, 355-370.

Turner M.G. & Gardner R.H. (2015) *Landscape Ecology in Theory and Practice (Second Edition)*. Springer, New York.

USGS (2019) Landsat Sentinel 2 Standard Product. US Geological Survey, Available at https://landsatlook.usgs.gov/sentinel2/viewer.html. Accessed 25 April 2019.

VAFI (2016) Industry Review 2016. Victorian Association of Forest Industries, Melbourne.

VicForests (2017) Timber Release Plan (Including Approved Changes) - January 2017. VicForests, Melbourne.

VicForests (2018) VicForests Procedures Regulatory Handbook Version 3.1, February 2018. VicForests, Melbourne. Available at http://www.vicforests.com.au/static/uploads/files/20180228-vicforests-operating-procedures-regulatory-handbook-v3-1-wffobneovxgi.pdf. Accessed 1 September 2019.

VicForests (2019a) Approved Timber Release Plan 2019. VicForests, Melbourne. Available from http://www.vicforests.com.au/planning-1/timber-release-plan-1/approved-timber-release-plan-april-2019. Accessed 28 August 2019.

VicForests (2019b) Harvesting and Regeneration Systems Draft Version 1.1. VicForests, Melbourne. Available at http://www.vicforests.com.au/static/uploads/files/rft-2019-6-attachment-b3-vicforests-harvesting-and-regeneration-systems-wfnazhvhbqad.pdf. Accessed 1 September 2019.

Vivian L.M., Cary G.J., Bradstock R.A. & Gill A. (2008) Influence of fire severity on the regeneration recruitment and distribution of eucalypts in the Cotter River Catchment, Australian Capital Territory. *Austral Ecol.* **33**, 55-67.

Wace N. (1977) Assessment of dispersal of plant species - the car borne flora of Canberra. In: *Exotic Species in Australia - Their Establishment and Success*. (eds D.R. Anderson), pp. 166-186, Proceedings of the Ecological Society of Australia 10.

Wang X., Blanchet F.G. & Koper N. (2014) Measuring habitat fragmentation: an evaluation of landscape pattern metrics. *Methods Ecol. Evol.* **5**, 634-646.

Watson J.E., Evans T., Venter O.*, et al.* (2018) The exceptional value of intact forest ecosystems. *Nature Ecol. Evol.* **2**, 599-610.

Zylstra P. (2018) Flammability dynamics in the Australian Alps. *Austral Ecol.* **43**, 578-591.

SUPPORTING INFORMATION

Table S1. Tukey's HSD for Proximity Index in Mountain Ash Core Areas (>1000m from disturbed areas)

Table S2. Tukey's HSD for Proximity Index in disturbed Mountain Ash Areas

Table S3. Tukey's HSD for Proximity Index in Alpine Ash Core Areas (>1000m from disturbed areas)

Table S4. Tukey's HSD for Proximity Index in Alpine Ash Forests (<1000m from disturbed areas)

Table S5. Tukey's HSD for Proximity Index in Alpine Ash Forests (<200m from disturbed areas)

Table S6. Tukey's HSD for Proximity Index in disturbed Alpine Ash Forests

Table 1. Land Tenure breakdown of Mountain Ash and Alpine Ash forests across the Central Highlands Regional Forest Agreement area

Land Tenure	Mountain Ash	% of Subtotal	Alpine Ash	% of Subtotal	Total	% of Total
Dedicated Reserve	37,955	28%	15,676	29%	53,631	28%
Informal Protected Area	31,991	23%	14,120	26%	46,111	24%
Other Parks	13	0%	0	0%	13	0%
Private Land - Other	7,024	5%	454	1%	7,479	4%
Logging Permitted	60,290	44%	24,633	45%	84,923	44%
Total	137,273	100%	54,882	100%	192,156	100%

Table 2. Extent of high severity disturbance across Mountain Ash and Alpine Ash forest areas by 2019

Disturbance Category	Mountain Ash area (ha)	% of Mountain Ash Area	Alpine Ash Area (ha)	% of Alpine Ash Area	Total Area (ha)	% of Total Area
Clearfell Logging	26,963	20%	9,165	17%	36,127	19%
Clearfell Logging and High Severity Fire	5,313	4%	2,552	5%	7,865	4%
High Severity Fire	15,819	12%	5,417	10%	21,235	11%
Subtotal of High Severity Disturbance	*48,095*	*35%*	*17,133*	*31%*	*65,228*	*34%*
Remainder	89,178	65%	37,749	69%	126,928	66%
Total	137,273	100%	54,882	100%	192,155	100%

FIGURE LEGENDS

Fig. 1. Study area and location.

Fig. 2. Extent of Mountain Ash and Alpine Ash forest.

Fig. 3. Land tenure across the ash forests for the Central Highlands RFA area.

Fig. 4. Trends for areas clearfell logged across Mountain Ash and Alpine Ash forests in the study area since 1960.

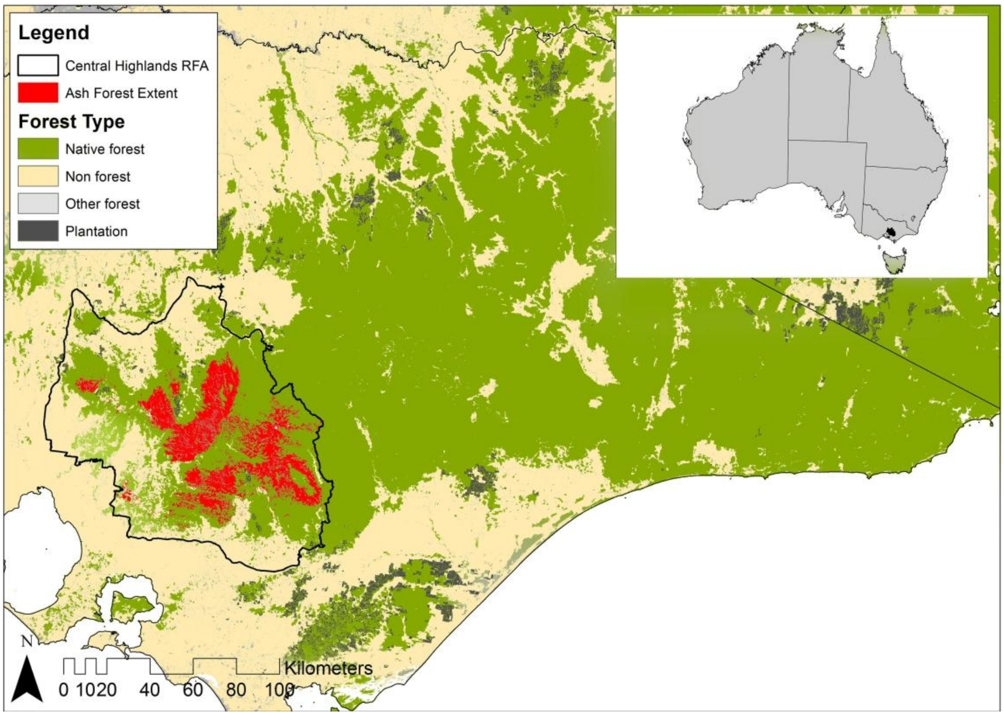
Fig. 5. Distance of disturbed area edge, including edge of high severity fire impact areas, roads, fuel breaks and clearfell logged areas.

Fig. 6. The location of disturbed areas and the proximity to disturbance in Mountain Ash and Alpine Ash forests.

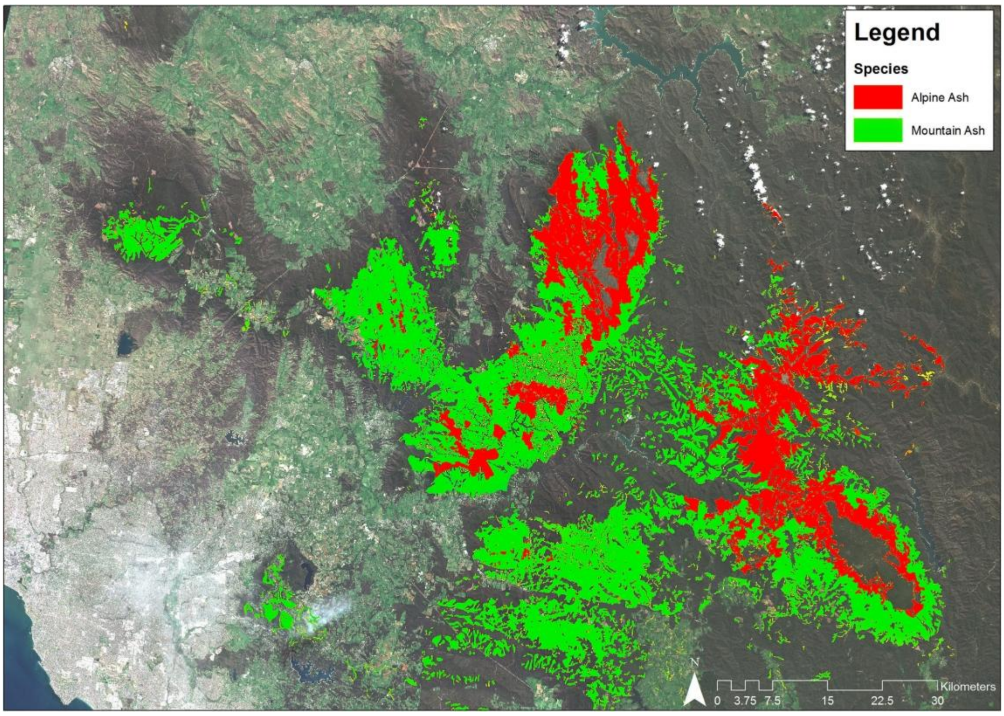
Fig. 7. Disturbance category distributions for Mountain Ash (left) and Alpine Ash (right) forests.

Fig. 8. Mean Proximity Indices for forest patches of Mountain Ash (left) and Alpine Ash (right) forests.

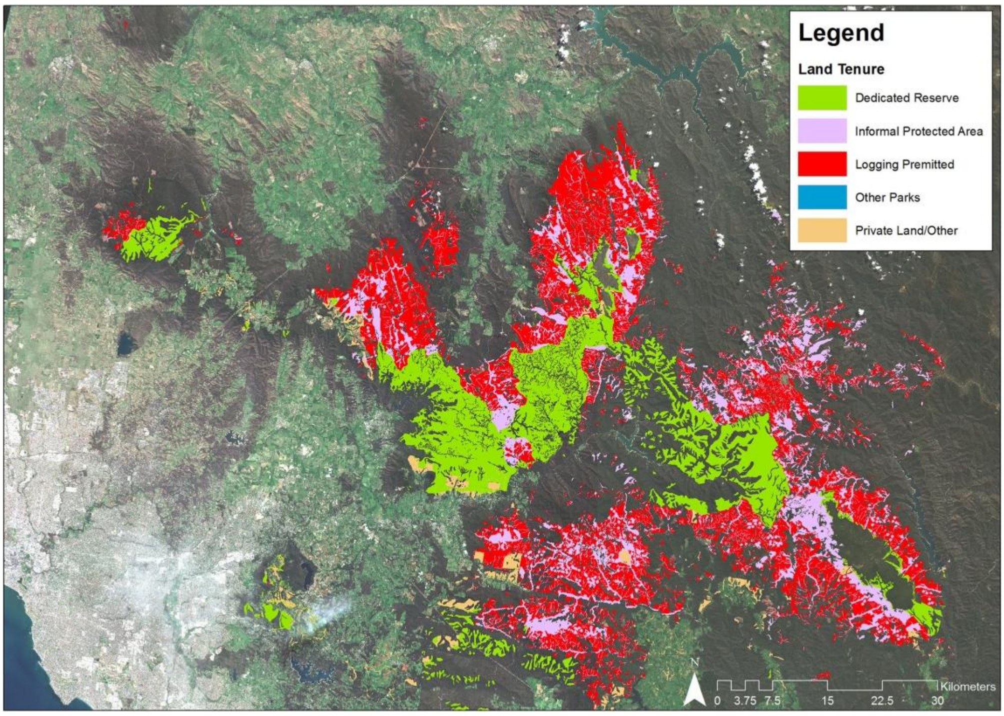
Fig. 9. Disturbance category for ash forest (Mountain and Alpine Ash forest) across dedicated reserves (top left), informal protected areas (top right) and where logging is permitted (bottom).



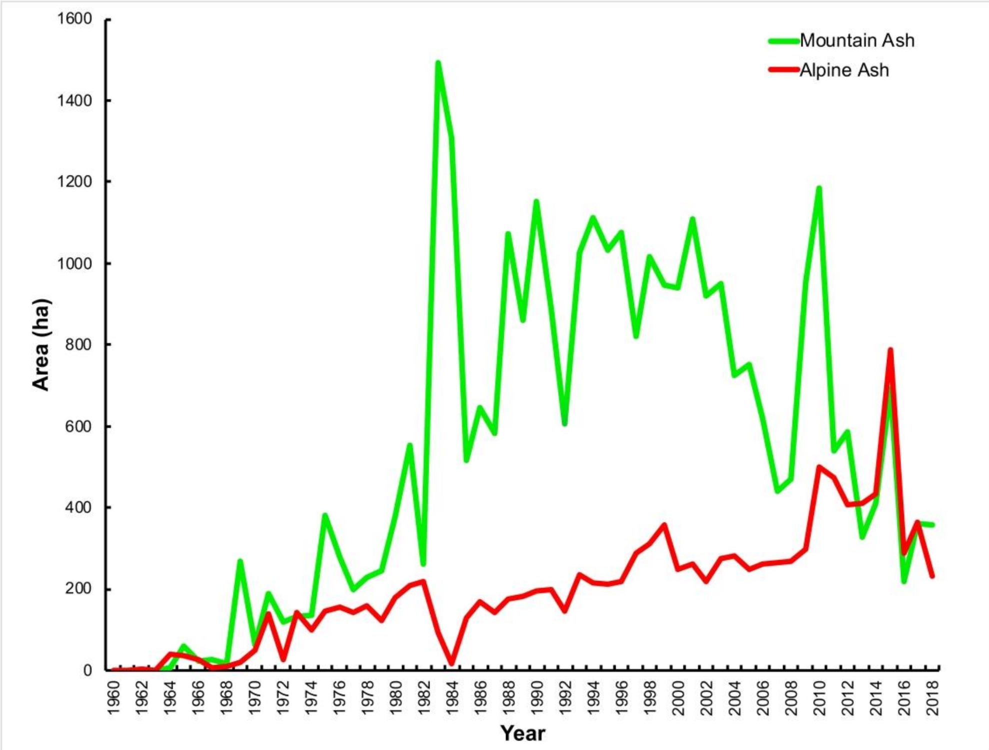
299x212mm (300 x 300 DPI)



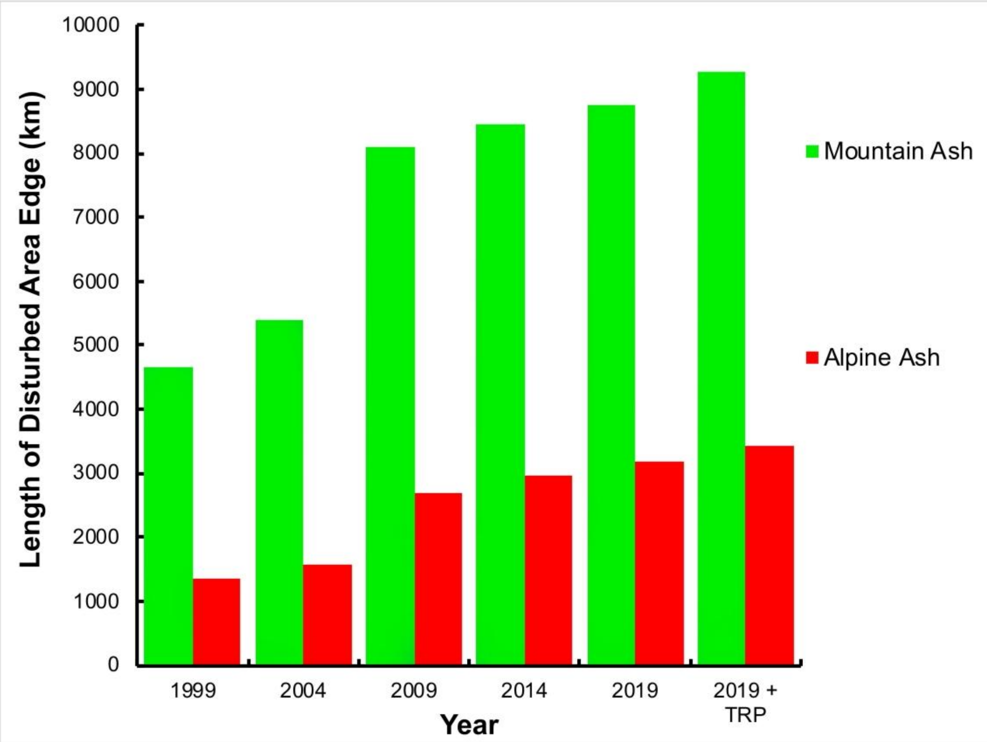
199x141mm (300 x 300 DPI)



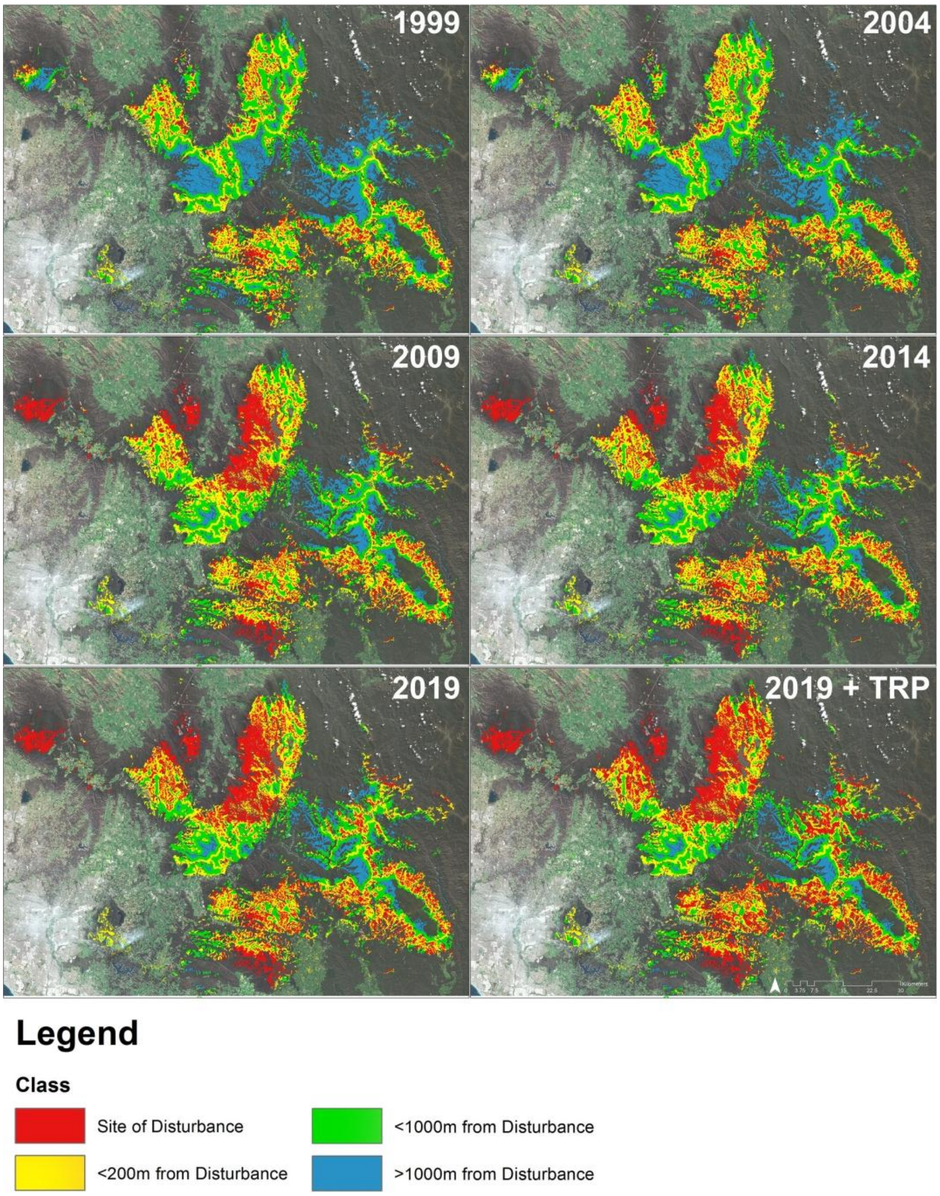
199x141mm (300 x 300 DPI)



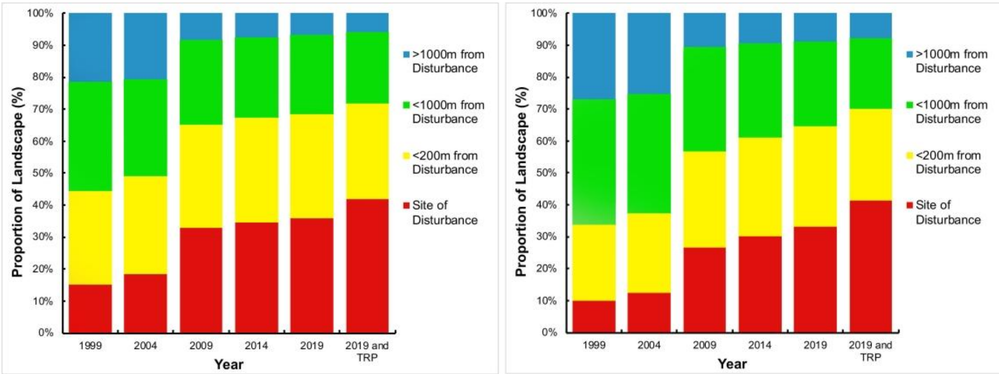
299x229mm (300 x 300 DPI)



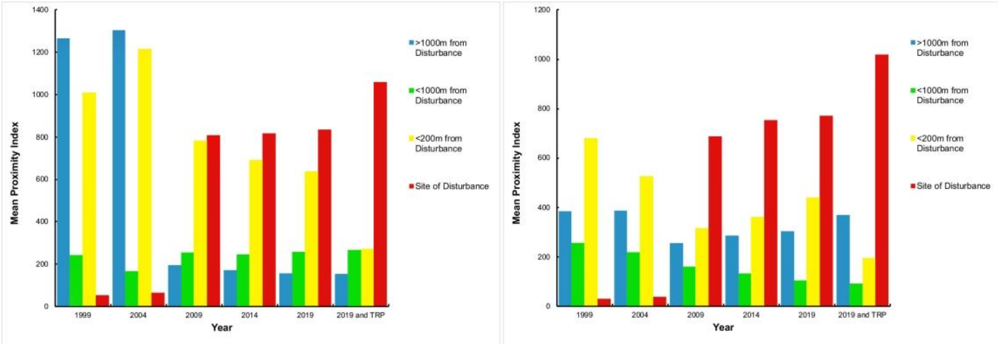
299x226mm (300 x 300 DPI)



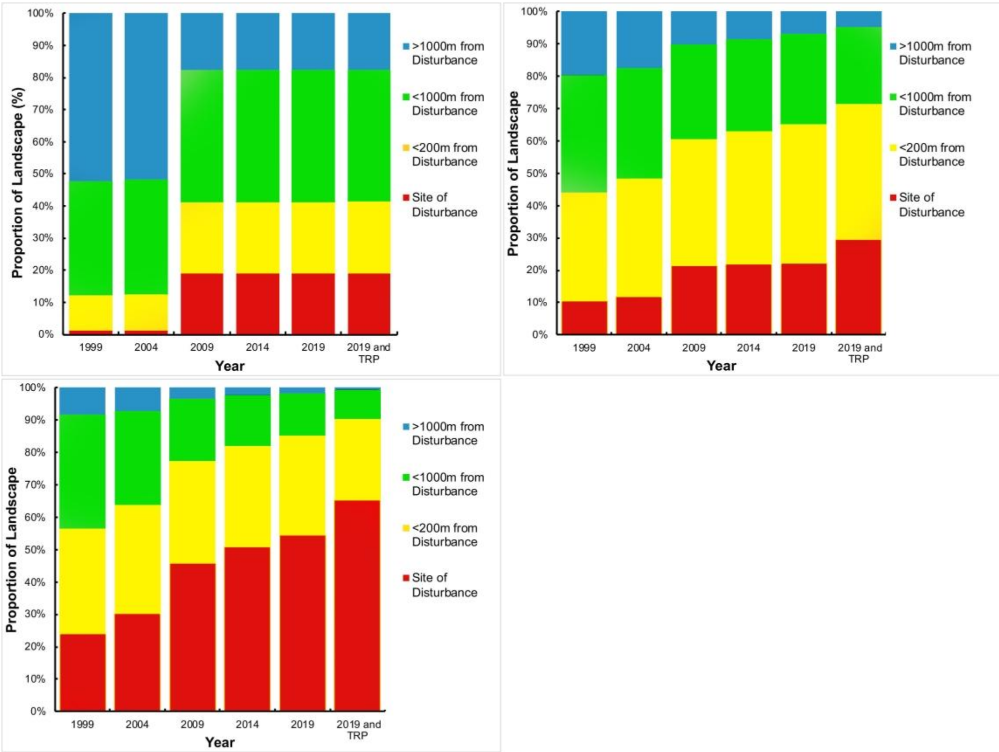
233x299mm (300 x 300 DPI)



299x111mm (300 x 300 DPI)



299x103mm (300 x 300 DPI)



299x225mm (300 x 300 DPI)

Temporal fragmentation of a critically endangered forest ecosystem

SUPPORTING INFORMATION

Table S1. Tukey's HSD for Proximity Index in Mountain Ash Core Areas (>1000m from disturbed areas)

Category	diff	lwr	upr	p adj
Year 2004-Year 1999	40.344	-427.928	508.615	1.000
Year 2009-Year 1999	-1069.699	-1627.507	-511.891	0.000
Year 2014-Year 1999	**-1094.990**	**-1655.760**	**-534.221**	**0.000**
Year 2019-Year 1999	**-1108.985**	**-1675.294**	**-542.677**	**0.000**
Year 2022-Year 1999	**-1111.004**	**-1702.584**	**-519.424**	**0.000**
Year 2009-Year 2004	**-1110.042**	**-1671.235**	**-548.850**	**0.000**
Year 2014-Year 2004	**-1135.334**	**-1699.470**	**-571.198**	**0.000**
Year 2019-Year 2004	**-1149.329**	**-1718.971**	**-579.686**	**0.000**
Year 2022-Year 2004	**-1151.347**	**-1746.120**	**-556.575**	**0.000**
Year 2014-Year 2009	-25.291	-665.698	615.115	1.000
Year 2019-Year 2009	-39.286	-684.549	605.976	1.000
Year 2022-Year 2009	-41.305	-708.857	626.247	1.000
Year 2019-Year 2014	-13.995	-661.820	633.830	1.000
Year 2022-Year 2014	-16.013	-686.042	654.015	1.000
Year 2022-Year 2019	-2.019	-676.690	672.653	1.000

Table S2. Tukey's HSD for Proximity Index in disturbed Mountain Ash Areas

Category	diff	lwr	upr	p adj
Year 2004-Year 1999	12.810	-150.556	176.176	1.000
Year 2009-Year 1999	**754.833**	**598.222**	**911.444**	**0.000**
Year 2014-Year 1999	**764.955**	**608.548**	**921.361**	**0.000**
Year 2019-Year 1999	**782.448**	**626.306**	**938.590**	**0.000**
Year 2022-Year 1999	**1007.089**	**847.899**	**1166.279**	**0.000**
Year 2009-Year 2004	**742.023**	**585.987**	**898.059**	**0.000**
Year 2014-Year 2004	**752.144**	**596.314**	**907.975**	**0.000**
Year 2019-Year 2004	**769.638**	**614.073**	**925.203**	**0.000**
Year 2022-Year 2004	**994.279**	**835.655**	**1152.903**	**0.000**
Year 2014-Year 2009	10.121	-138.612	158.855	1.000
Year 2019-Year 2009	27.615	-120.840	176.070	0.995
Year 2022-Year 2009	**252.256**	**100.598**	**403.914**	**0.000**
Year 2019-Year 2014	17.493	-130.746	165.733	0.999
Year 2022-Year 2014	**242.135**	**90.688**	**393.582**	**0.000**
Year 2022-Year 2019	**224.641**	**73.468**	**375.815**	**0.000**

Table S3. Tukey's HSD for Proximity Index in Alpine Ash Core Areas (>1000m from disturbed areas)

Category	diff	lwr	upr	p adj
Year 2004-Year 1999	2.033	-236.708	240.774	1.000
Year 2009-Year 1999	-129.957	-400.547	140.633	0.745
Year 2014-Year 1999	-99.950	-381.501	181.601	0.914
Year 2019-Year 1999	-81.675	-372.870	209.521	0.967
Year 2022-Year 1999	-16.890	-330.835	297.056	1.000
Year 2009-Year 2004	-131.990	-405.043	141.062	0.739
Year 2014-Year 2004	-101.984	-385.902	181.935	0.910
Year 2019-Year 2004	-83.708	-377.193	209.778	0.965
Year 2022-Year 2004	-18.923	-334.993	297.148	1.000
Year 2014-Year 2009	30.007	-281.171	341.185	1.000
Year 2019-Year 2009	48.283	-271.648	368.213	0.998
Year 2022-Year 2009	113.068	-227.699	453.835	0.934
Year 2019-Year 2014	18.276	-310.977	347.529	1.000
Year 2022-Year 2014	83.061	-266.474	432.595	0.984
Year 2022-Year 2019	64.785	-292.564	422.134	0.996

Table S4. Tukey's HSD for Proximity Index in Alpine Ash Forests (<1000m from disturbed areas)

Category	diff	lwr	upr	p adj
Year 2004-Year 1999	-38.609	-119.641	42.423	0.752
Year 2009-Year 1999	**-96.842**	**-177.530**	**-16.154**	**0.008**
Year 2014-Year 1999	**-123.563**	**-204.279**	**-42.847**	**0.000**
Year 2019-Year 1999	**-152.223**	**-233.431**	**-71.015**	**0.000**
Year 2022-Year 1999	**-165.344**	**-248.438**	**-82.251**	**0.000**
Year 2009-Year 2004	-58.233	-138.624	22.158	0.306
Year 2014-Year 2004	-84.954	-165.374	-4.535	0.031
Year 2019-Year 2004	**-113.614**	**-194.527**	**-32.701**	**0.001**
Year 2022-Year 2004	**-126.735**	**-209.541**	**-43.930**	**0.000**
Year 2014-Year 2009	-26.722	-106.794	53.351	0.933
Year 2019-Year 2009	-55.381	-135.949	25.187	0.366
Year 2022-Year 2009	-68.503	-150.971	13.966	0.168
Year 2019-Year 2014	-28.660	-109.256	51.937	0.914
Year 2022-Year 2014	-41.781	-124.277	40.715	0.700
Year 2022-Year 2019	-13.121	-96.099	69.856	0.998

Table S5. Tukey's HSD for Proximity Index in Alpine Ash Forests (<200m from disturbed areas)

Category	diff	lwr	upr	p adj
Year 2004-Year 1999	-155.242	-324.383	13.900	0.093
Year 2009-Year 1999	**-365.314**	**-509.796**	**-220.833**	**0.000**
Year 2014-Year 1999	**-319.832**	**-462.699**	**-176.965**	**0.000**
Year 2019-Year 1999	**-240.756**	**-382.357**	**-99.156**	**0.000**
Year 2022-Year 1999	**-484.885**	**-626.311**	**-343.460**	**0.000**
Year 2009-Year 2004	**-210.073**	**-346.711**	**-73.434**	**0.000**
Year 2014-Year 2004	**-164.590**	**-299.520**	**-29.661**	**0.007**
Year 2019-Year 2004	-85.515	-219.104	48.074	0.450
Year 2022-Year 2004	**-329.644**	**-463.047**	**-196.241**	**0.000**
Year 2014-Year 2009	45.483	-56.851	147.816	0.803
Year 2019-Year 2009	**124.558**	**24.000**	**225.116**	**0.006**
Year 2022-Year 2009	**-119.571**	**-219.882**	**-19.260**	**0.009**
Year 2019-Year 2014	79.075	-19.149	177.299	0.196
Year 2022-Year 2014	**-165.053**	**-263.024**	**-67.082**	**0.000**
Year 2022-Year 2019	**-244.129**	**-340.245**	**-148.013**	**0.000**

Table S6. Tukey's HSD for Proximity Index in disturbed Alpine Ash Forests

Category	diff	lwr	upr	p adj
Year 2004-Year 1999	7.394	-301.359	316.146	1.000
Year 2009-Year 1999	**656.547**	**366.234**	**946.860**	**0.000**
Year 2014-Year 1999	**723.518**	**432.409**	**1014.627**	**0.000**
Year 2019-Year 1999	**741.957**	**451.198**	**1032.715**	**0.000**
Year 2022-Year 1999	**989.824**	**689.493**	**1290.156**	**0.000**
Year 2009-Year 2004	**649.153**	**358.222**	**940.085**	**0.000**
Year 2014-Year 2004	**716.125**	**424.399**	**1007.850**	**0.000**
Year 2019-Year 2004	**734.563**	**443.187**	**1025.939**	**0.000**
Year 2022-Year 2004	**982.431**	**681.501**	**1283.360**	**0.000**
Year 2014-Year 2009	66.971	-205.163	339.106	0.982
Year 2019-Year 2009	85.410	-186.350	357.170	0.948
Year 2022-Year 2009	**333.277**	**51.299**	**615.256**	**0.010**
Year 2019-Year 2014	18.439	-254.171	291.048	1.000
Year 2022-Year 2014	266.306	-16.492	549.104	0.078
Year 2022-Year 2019	247.867	-34.570	530.304	0.124