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Assessing the Vulnerability of Victoria's Central Highlands Forests to Climate Change

Technical Report 1/2007

Prepared for the Department of Sustainability and Environment

By: Craig R. Nitschke and Gordon M. Hickey
School of Forest and Ecosystem Science

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School of Forest and Ecosystem Science
Faculty of Land and Food Resources
The University of Melbourne

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Executive Summary

The ability to conserve ecosystems and ecological services in the face of climate change will rely heavily on our understanding of ecosystem vulnerability during the regeneration phase of species. It will also require an understanding of the degree of resilience that exists within species and ecosystems that will allow for natural or anthropogenic-aided adaptation. For the Department of Sustainability and Environment, climate change-related adaptation actions that begin from a foundation of understanding will allow decisions to be made based on risk instead of uncertainty. According to the Intergovernmental Panel on Climate Change (IPCC) (1998), vulnerability assessments are the best method for assessing potential climate change impacts (see also Lemmen and Warren 2004). Using this approach, we have analysed the possible effects that predicted climate change will have on the resilience of the dominant tree species in the native forests of the Central Highlands. The Central Highlands provides an excellent example of an area where forests are being actively managed for a wide variety of values, including commercial forestry operations, biodiversity conservation, recreation and water for the city of Melbourne.

This study represents a first step towards improving the understanding of tree species vulnerability to climate change in Victoria. The specific objectives were as follows:

1) To investigate the extent to which species and ecosystems in the Central Highlands study area are vulnerable to climate change;
2) To explore the whether climate thresholds are trigger points that may cause a decrease in ecological resilience; and
3) To identify whether current forest management practices can foster ecological resilience by expanding the coping range of ecosystems in the Central Highlands study area.

The research followed the approach used by Nitschke (2006) and Nitschke and Innes (2008) to model species and ecosystem resilience to changes in phenology, frost damage, drought risk, and heat stress in the temperate forests of British Columbia, Canada. For the Central Highlands study area, 22 tree species of the dominant Ecological Vegetation Classes (EVC) described in the Central Highlands Forest Management Plan (Department of Natural Resources and Environment 1998) were selected for analysis. These species

represent two broad regeneration strategies: 1) species that are entirely reliant on seedling regeneration; and 2) regeneration based on true and lignotuberous seedlings.

Our analysis focussed on the regeneration niche of these species because this is where they are considered to be most susceptible to climatic variation. In each case, species responses under three microclimatic treatments were analysed: 1) open conditions following a stand-replacing disturbance (e.g., high intensity fire, clearfell operations, etc.); 2) partially modified forest canopy following a stand modifying disturbance (e.g., thinning operations, low intensity ground fire, etc.); and 3) intact forest canopy (no-or canopy-maintaining disturbance) (e.g., protected forests, fire suppression, etc.).

Historically, climate change modelling in Australia has utilised BIOCLIM/ ANUCLIM climate envelope models. In our study, mechanistic modelling was used to assess the vulnerability of the selected tree species in their regeneration niche to predicted climate change. This approach is new to climate change modelling in Australia and has demonstrated itself to be insightful. Principal findings were as follows:

- Significant resilience was found to exist in the Central Highlands until 2040 when a threshold was reached. An increase in mean annual temperature of 1.4 °C and a decline in annual precipitation by 5 % were found to cause a significant contraction in 20 species regeneration niches and significant changes in the size and location of species potential ranges.
- The narrow climatic breadth of Australian species identified by Hughes et al. (1996) was observed in this study. Across all treatments, all the selected tree species exhibited significant vulnerability to predicted climate change. Despite this, the stand modifying treatment was found to mediate species response, resulting in lower vulnerability classifications for all but nine species when compared to the other two treatments. The stand modifying treatment moderated the interaction between temperature and precipitation to reduce the impact of climate change on species regeneration potential. It is, however, important to note that stand modifying treatments are not the “silver bullet” for adapting forests to predicted climate change. Importantly, our analysis did not consider light or competition, only climatic conditions, and these will have important influences on successful regeneration under stand modifying treatments.

- A significant increase in the climatic optimality of damping-off fungus was detected in all treatments but was most prevalent in the stand modifying treatment. This could represent a possible positive feedback loop.
- All 22 species were found to contract from lower elevations with each species able to regenerate above 1400 m in elevation by 2085, although regeneration potential declined for the majority of the species. Species responded in different ways to climate change, which may cause the eventual disassociation of current ecosystem assemblages and the creation of new ecosystems.
- Ancillary to the identification of thresholds is the discovery of areas that could act as potential climatic refugia. The higher elevation areas of the Central Highlands provide the chance to manage vulnerable species with reversible and adaptive practices. Areas where species resilience is overcome by climate change and regeneration potential is negligible will require preventative management that mediates the direct or indirect impact of stressors. This will be essential if the sustainable management of species and ecosystems are to be achieved in the long-term.

The Central Highlands forests are vulnerable to predicted climate change, but management options exist that can reduce this vulnerability. In particular, adaptation actions that focus on fostering the ecological resilience of all species will, in turn, maintain their regeneration niche across a wider range than would otherwise be the case. Our study has provided insight into the issues that will need to be considered when developing management plans and policies designed to diversify the vulnerability that species and ecosystems may face in the future. Further research needs have also been identified as part of this study. In particular, studies investigating species-specific responses to changes in environmental conditions are required to improve model calibration and validate the modelled responses.

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Acronyms

Basal Temperature	T_{base}
Bud Break	BB
Combination Sprouter	CS
Department of Sustainability and Environment	DSE
Ecological Vegetation Class	EVC
Chilling Requirement	CR
Confidence Interval	CI
Coppice	C
Environmental Systems Research Institute	ESRI
Forest Management Area	FMA
Geographic Information System	GIS
Global Circulation Model	GCM
Growing Degree Days	GDD
Intergovernmental Panel on Climate Change	IPCC
Maximum Temperature	Max T
Minimum Temperature	Min T
Obligate Seeder	OS
Precipitation	Precip
Royal Botanical Gardens	RBG
Secondary Dormancy	SD
Seed Break	SB
Statewide Forest Resource Inventory	SFRI
Stem Sprouter	SS
The World Conservation Union	IUCN
Weakly Lignotuberous	WL

1. Introduction

Compared to many northern hemisphere regions, Australia has a highly variable natural climate. In addition, Australia is likely experiencing the impacts of human-induced climate change. Since the 1950's the mean annual temperature has increased by between 0.4 and 0.7°C and annual precipitation has declined causing increased stress on water supply and agriculture, changes in natural ecosystems, and reduced seasonal snow cover (Hennessy et al. 2007a). The 2007 Intergovernmental Panel on Climate Change's (IPCC) Fourth Assessment Report predicts that mean annual temperatures in south-eastern Australia may increase from 0.6 to 2 °C by 2030 and 1 to 6 °C by 2070 with a corresponding decrease in annual precipitation from 15 to 20 % (Christensen et al. 2007). The median warming predicted for Australia is 2.6 °C by 2100. In southern Victoria, mean annual temperatures are predicted to increase from 0.2 to 1.4 °C by 2030 and 0.7 to 4.3 °C by 2070 with a corresponding change in annual precipitation ranging from + 9 to - 25 % (Suppiah et al. 2004). This predicted climate change is expected to have a significant impact on Victoria's natural ecosystems and water supplies due to the relatively narrow coping range of the system. According to Hennessy et al. (2007a), a change in mean temperature that is greater than 1.5°C will significantly increase the vulnerability of Australian species and ecosystems.

The IPCC Fourth Assessment Report stated the need for research into understanding the mechanisms that predispose physical, biological and human systems to irreversible changes as a result of exposure to climate and other stresses (Parry et al. 2007). Parry et al. (2007) go on to argue the need for scientists to identify how close natural ecosystems are to ecological thresholds and what positive feedback loops might occur if these thresholds are exceeded. As a result, research needs to focus on the mechanisms that enhance system resilience or vulnerability so that the risk of irreversible change can be diversified through an understanding of ecosystem response to these thresholds and feedback loops.

In the Australian context, Hennessy et al. (2007a) identified that water security and natural ecosystems should be the main research priorities for agencies wishing to assess ecosystem vulnerability to climate change. They note the need to improve our understanding of the impacts of climate change on droughts, floods and groundwater-levels in order to develop optimum adaptation strategies. For natural forest ecosystems, they stress the need to begin long-term monitoring, undertake modelling to assess the vulnerability of key ecosystems, develop management actions that can reduce ecosystem vulnerability and to identify thresholds, vulnerable and indicator species, and the rates at which autonomous adaptation may occur (Hennessy et al. 2007a).

The Victorian Government's ability to conserve ecosystems and maintain ecological services will rely on its understanding of ecosystem vulnerability to climate change. Through the Victorian Greenhouse Strategy (Department of Natural Resources and Environment 2002) and the Sustainability Charter for Victoria's State Forests (Department of Sustainability and Environment 2006) the State government has outlined its commitment to sustainable forest management and mitigating and adapting to the impacts of climate change. At the federal level, the National Biodiversity and Climate Change Action Plan 2004–2007 (Natural Resource Management Ministerial Council 2004) sets objectives for government to focus research on improving their understanding of the impacts of climate change on biodiversity, natural ecosystems and water resources. These policies are directly in line with the research requirements outlined by the IPCC's Fourth Assessment Report.

1.1 Climate Change and Understanding Ecosystem Vulnerability

Climate change is a stressor that will directly or indirectly influence the processes that impact ecosystems. The ecosystem concept has been a powerful tool for understanding the interaction between living organisms and the abiotic components of the environment (Tansley 1935). Changes in any biophysical component can alter the stable dynamic equilibrium that exists between biotic and abiotic components leading to the creation of new ecosystems (Tansley 1935). The maintenance of ecosystem health and vitality are a foundation of sustainable management. The ecosystem concept therefore provides a basis for developing sustainable management. Any process that results in a restructuring of controlling variables and ecological processes can potentially affect the capacity of the system to provide ecosystem services to society (Chapin et al. 2000). Significant restructuring of controlling variables and processes can shift an ecosystem to a new stable state (Gunderson et al. 2002) with different levels of service provision. The ability of an ecosystem to recover from natural disturbances and management actions or persist under changes in climate is referred to as ecological resilience (Holling 1996). Management generally aim to maintain or increase ecosystem resilience if the ecosystem and the provision of ecosystem services are to be sustained.

The resilience of an ecosystem is driven by species-level responses to change in the environmental factors that determine a species' distribution and abundance. The sum of environmental factors that are described by a species' abundance and distribution is referred to as the fundamental niche (Hutchinson 1957; Schoener 1989). The fundamental niche of a species is determined by a combination of environmental variables and processes. The extent to which this niche is realised in any situation is governed by the competition for the same niche space with other species. Realised niches are not static; they shift in time and space due to changes in resource availability resulting from changing climate, soil processes or biotic or abiotic disturbances (Schoener 1989). Niche breadth is also affected by species' phenology (seasonality of flowering, fruiting, germination, growth and mortality) and its regenerative niche (Grubb 1977).

Fagerström and Ågren (1999) showed that the temporal differences in phenology and productivity during regeneration allowed species to coexist that would otherwise competitively exclude each other. The regeneration niche is narrower than the fundamental niche of mature trees; typically reflecting the optimal portion of the fundamental niche (McKenzie et al. 2003). However, the dynamic nature of the niche usually means that the current species distribution often dominates the suboptimal portion of its niche over time in the presence of competition (Rehfeldt et al. 1999). Events occurring during the regeneration phase of natural communities can play a key role in community composition and may affect species diversity and promote species coexistence in environments that are homogeneous at the adult plant scale (Grubb 1977). Florence (1964) stated that an ecosystem is an expression of environmental pressures and that change in communities are sensitive and predictable to changes in the edaphic environment. Consequently, environments that are effectively homogeneous at the scale of the adult can be patchy at the seed or seedling scale (Battaglia 1997). Thus the presence/absence of species reflects the breadth of a species regeneration niche and environmental conditions at the time of establishment.

To establish the vulnerability of ecosystems to climate change we need to consider the resistance and resilience of individual species. Organisms in assemblages can have differential responses to the same disturbances (Walker 1989). The magnitude and/or frequency of a disturbance interacting with each species' unique physiology, demographics and life-cycle characteristics can cause divergent responses (Walker 1989). The divergent response of species within a community suggests that an ecosystem can be composed of species that are resilient to environmental change and species that are not. Thus, a species can be resilient, but the community (ecosystem) may not, and vice versa. Species are most vulnerable to changes in environmental conditions in the regeneration phase since it is the most critical phase for their survival (Bell 1999). Understanding species vulnerability at this stage is therefore an important step if we are to determine where, and what, adaptation strategies are to be incorporated into long-term forest planning and risk management in relation to climate change (Nitschke and Innes 2006).

The ability of Victoria to conserve ecosystems and ecosystem services in the face of climate change will rely heavily on our understanding of ecosystem vulnerability during the regeneration phase of species and the degree of resilience that exists within species and ecosystems that could allow for natural or anthropogenic-aided adaptation. This study represents a first step towards improving the understanding of tree species vulnerability to climate change in Victoria. The objectives of this study were as follows:

1) To investigate the extent to which species and ecosystems in the Central Highlands Forest Management Area (FMA) are vulnerable to climate change;
2) To explore the whether climate thresholds are trigger points that may cause a decrease in ecological resilience; and
3) To identify whether current or possible future forest management practices can foster ecological resilience by expanding the coping range of ecosystems in the Central Highlands FMA.

1.2 Study Area

The Central Highlands contains approximately 2 million hectares of land under various management tenures and consists of 10 major ecosystems (Figure 1). State forests comprise 512, 850 ha of the study area, national parks and reserves: 195, 330 ha, forest plantations: 8520 ha and private land 1, 223, 300 ha. The major ecosystems are primarily dominated by eucalypts (*Eucalyptus* spp.) with diverse understorey communities frequently dominated by wattle species (*Acacia* spp.). In cool, wet areas that have not had significant fires for 200-300 years, Cool Temperate Rainforest occurs. Arguably the most iconic and economically important ecosystems are the Wet Forests, dominated by Mountain Ash (*Eucalyptus regnans*), Montane Wet Forest, dominated by Alpine Ash (*E. delegatensis*), and Damp Forest, dominated by Messmate Stringybark (*E. obliqua*). This area has been used extensively for various research studies by DSE (and its predecessors) over the last 50 years, and is the focus of significant monitoring and research efforts.

The Central Highlands provides an excellent example of an area where forests are being actively managed for a wide variety of values, including commercial forestry operations, biodiversity conservation, recreation and water for the city of Melbourne. It is regarded as a highly significant area of forest biodiversity in Australia, home to iconic ecosystems and species, such as the endangered Leadbeater's possum (*Gymnobelideus leadbeateri*).

The study area ranges from 200 to 1600m above sea level and has an annual rainfall of between 600-2000 mm and a mean annual temperature range between 5.4 °C and 14.2 °C. The area typically experiences mild, humid winters with occasional periods of snow and cool summers. Howe et al. (2005) found that under the various IPCC climate change scenarios there could be an 8 to 20 % reduction in the water supply to Melbourne by 2050 due to decreases in streamflows and an increased risk of bushfires in catchment areas. Mackey et al. (2002) suggested that fire regimes in the Central Highlands are vulnerable to climate change and fires will likely be more frequent and severe in the future leading to a reduction in fire refugia. This increase in fire frequency under predicted climate change is supported by Hennessy et al. (2007b) who predict an increase in fire weather severity for south-eastern Australia.

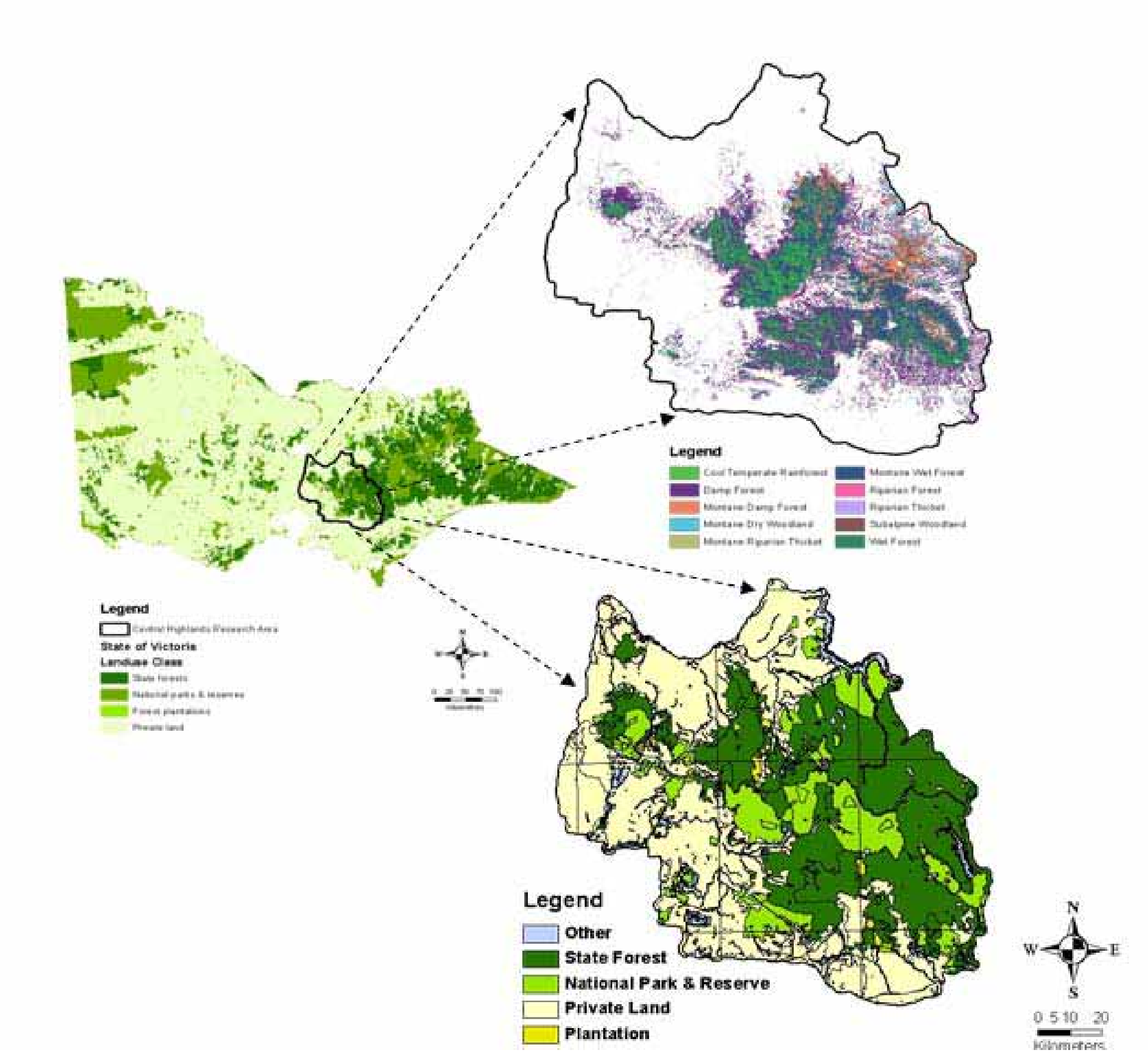


Fig. 1: Central Highlands study area: Land use and ten major ecosystems

2. Tree Species Characteristics in Relation to Potential Impacts of Future Climate Change in the Central Highlands

The species in the Central Highland ecosystems will face many threats from predicted climate change. Changes in climate will cause changes in biophysical conditions (Florence 1964) which, in turn, will interact with the physiological mechanisms and tolerances that will allow for successful regeneration and, therefore, long-term integrity of these systems. Species and ecosystems of Australia are potentially vulnerable to climatic change because they have narrow ecological ranges (Hughes et al 1996; Hennessy et al. 2007a, Hughes

2003). Hughes et al. (1996) found that 434 of 819 *Eucalyptus* species have a mean annual temperature range of less than 3°C, 335 of <2°C and 205 of <1°C, while 188 species have ranges that span less than a 20 % variation in mean annual precipitation. This makes many eucalyptus species vulnerable to predicted changes in temperature and precipitation, and may prevent species from adjusting their ranges to cope with the rapid rate of warming and drying predicted for the future. *Eucalyptus* is not the only susceptible genus. Hughes (2003) highlighted that Australia's *Acacia* species were also likely to experience significant range reductions if there was a 2°C warming in the climate, due to their inability to track shifting climatic zones. Busby (1988) found that Cool Temperate Rainforest species are also vulnerable to climate change.

Changes in the distribution of optimal temperature ranges will impact the ability of a species to successfully regenerate and compete under warmer conditions. For example, higher temperatures can increase the rate of germination but also induce secondary dormancy or cause direct mortality (Stoneman 1994). Change in annual precipitation also impacts regeneration success. Increases in precipitation may allow species to expand their ranges while decreases can force some species to contract to areas with more suitable climatic/ edaphic conditions (Ashton and Spalding 2001; Morgan 2004). Predicted changes in climate will likely provide opportunities for some species to expand from their core ranges within the Central Highlands while forcing others to contract within. A more dynamic pattern may also occur where species can expand at one ecotone (due to temperature) while contracting or remaining stationary at another (due to rainfall), thereby creating opportunity for new species compositions to form new ecosystems, ecoclines and landscape patterns.

Some future climate scenarios predict an increase in the frequency and severity of drought events caused by the el nino- southern oscillation. This may result in more severe and frequent fires and increase mortality through drought stress. Because they have small seeds and limited reserves, eucalypts are generally more sensitive to drought and soil moisture deficits in the regeneration phase than within their adult stage (Ashton et al. 1974). Mortality of germinants and seedlings tends be highest during the first growing season due to soil water deficits caused by drought conditions (Ferrar et al. 1988; Stoneman 1994). Battaglia (1996) identified summer drought stress as a major cause of seedling mortality for Alpine Ash, while Howard (1973b) identified drought stress as a major limiting factor in the survival of Myrtle Beech (*Nothofagus cunninghamii*) seedlings. Drought has been found to prevent the regeneration of mesic species and thus favour establishment of xeric species (Allen and Breshears 1998). Ashton and Martin (1996) found that drought acted to inhibit the germination of Mountain Ash seeds by inducing secondary dormancy, but did not prevent Silver Wattle (*Acacia dealbata*) and Hazel Pomaderris (*Pomaderris aspera*) from germinating. Cunningham (1960), Grose (1963), Gibson and Bachelard (1986) and Stoneman (1994) have found that eucalypts are sensitive to high temperatures, atmospheric vapour pressure and soil moisture during germination. This can lead to the initiation of secondary dormancy that will prevent germination until conditions become favourable. Alternatively it will eventually cause seed mortality. Drought can also affect inflorescence bud development and flowering in eucalypts which in turn can lead to insufficient seed crops for natural regeneration (Ashton 1975; Keatly et al. 2002; Flint and Fagg 2007). As a result, many tree species of the Central Highlands can be considered vulnerable to increases in drought, particularly in their regeneration niche. Increases in summer moisture deficits, combined with higher temperatures is also likely to impact on the growth, health and mortality of mature stands.

Frost damage is another important mechanism that can prevent the regeneration of tree species in the Central Highlands through direct mortality or its impact on growth and vigour (Ashton 1958; Battaglia 1996). Stoneman (1994) noted that frost is second only to drought as a cause of seedling/ germinant mortality in eucalypts. The degree of frost damage/ mortality is related to the ability of eucalypts to achieve a degree of hardening prior to frost events. The lack of hardening or premature dehardening of trees under climate change has been predicted to result in an increase in frost damage and mortality. For example, Nitschke (2006) and Nitschke and Innes (2008) found that climate change is likely to result in increased occurrence of spring frosts in the temperate forests of British Columbia, Canada. In Victoria, some future climate scenarios suggest that the number of frost days will be reduced, but in some areas a significant number of frost events may still occur. This could be a limiting factor in the establishment of species at higher elevations and an important feedback preventing species from moving to higher elevations in response to changing temperature or precipitation patterns. Therefore, the *Eucalyptus* species of Victoria's Central Highlands may be vulnerable to an increase in both winter and spring frost events, particularly in the regeneration niche of the species. Further, Read and Hill (1998) stated that frost is a limiting factor that prevents Southern Sassafras (*Atherosperma* *moschatum*) from regenerating at higher elevations. Ashton (1958) and Battaglia (1996) also stated that frost limits the expansion of mountain and alpine ash into higher elevation grasslands. Within mature forest communities, frost damage may predispose stands to attacks by defoliators such as the Spurlegged Phasmatid (*Didymuria violescens*) (DSE 2006b).

Chilling requirements for seed germination are generally not important for the tree species in the Central Highlands area, with four exceptions. Alpine Ash (*E. delegatensis*), Shining Gum (*E. nitens*), Snow Gum (*E. pauciflora*) and Tingaringy Gum (*E. glaucesens*) all require cold temperatures and moist conditions for a few weeks to break primary dormancy of their seeds and allow for successful germination (Ferrar et al. 1988; Battaglia 1993; Close and Wilson 2002). Increases in winter and early spring temperatures may not provide the chilling requirements for these species and thus prevent regeneration within their current ranges.

The tree species in the Central Highlands grow in areas of high rainfall and are considered to be fire adapted, in that they can regenerate after fire given the right conditions, but also fire sensitive, in that they are generally killed by moderate to intense fire (Department of Sustainability and Environment [DSE] 2003). Frequent fires (< 20 years return interval), or poor climatic conditions can prevent these eucalypt forests from regenerating and encourage the development of forests dominated by *Acacia* and shrub communities. For example, in Wet Forests, frequent fires can lead to the development of a community dominated by Silver Wattle and Hazel Pomaderris (DSE 2003). Over the last four years, frequent fires coupled with poor seed crops have resulted in failed regeneration in many areas of Wet and Montane Wet Forest adjacent to the Central Highlands with the Torongo Plateau providing a contemporary example. Conversely, where fires do not occur for 200 to 300 years, forest composition will change from eucalypts to Myrtle Beech, Southern Sassafras and Blackwood Wattle (*Acacia melanoxylon*) (DSE 2003). Sheltered gullies within the Highlands that are classified as Cool Temperate Rainforest provide a contemporary example of this successional path. Under predicted climate change, an increase in fire severity and frequency is expected to occur in areas that experience an increase in temperature along with marginal changes in precipitation (Veblen et al. 2003; Nitschke and Innes 2006). Studies by Mackey et al. (2002) and Hennessy et al. (2007) both predict this to be the case for the Central Highlands. Such a change in fire regime will increase the vulnerability of fire-sensitive eucalypt and rainforest species while favouring others, such as wattles and more fire-tolerant eucalypts that are better adapted to shorter fire intervals.

For many of the eucalypt species, the degree of vulnerability that they may exhibit to climate change within their regeneration niche needs to be considered in two separate ways: 1) seed-based regeneration; and 2) lignotuber sprouting-based regeneration. This is because the vulnerability of a species in its seed-based regeneration niche is potentially far different than its vulnerability in its sprouting-based regeneration niche.

Lignotubers are wood swellings containing dormant vegetative buds. They form in the axils of the cotyledons and successive early leaf nodes of the majority of eucalypt species within a few weeks of seedling germination (Kerr 1925; Nicolle 2006). Within four to six months the developed lignotuber can enhance survival following damage to or removal of the main shoot/ stem through vegetative sprouting (Walters et al. 2005b). As seedlings develop into saplings the lignotuber is occluded by the stem, eventually forming a buried or partially buried structure in the main stem (Jacobs 1955; Carrodus and Blake 1970; Whittock et al. 2003). Lignotubers contain non-structural carbohydrates that are used to support new sprouts following disturbance (Canadell and Lopez-Soria 1998). However, lignotubers are no more efficient than roots or stems at storing carbohydrates (Carrodus and Blake 1970).

The majority of eucalypt species produce lignotubers, with the exception of a few species. Jacobs (1955) listed 11 eucalypt species that do not produce lignotubers which Nicolle (2006) defined as either obligate seeders or sprouters. The eucalypts that produce lignotubers in the Central Highlands of Victoria are defined as combinational sprouters because they can coppice from epicormic buds and sprout from dormant lignotuberous buds (Nicolle 2006). Lignotubers enhance the regeneration success of the majority of eucalypt species by increasing the chance of individual survival from catastrophic events such as fire, drought and frost (Whittock et al. 2003). In harsh environments, the lignotuber provides a protected storage organ of buds that allows regeneration to occur after death or damage to the main stem (Blake and Carrodus 1970; Mullette 1978). As a result, the survival ability of many eucalypt species is dependent on the presence of vegetative buds in the lignotuber (Noble 2001). Noble (2001) described lignotubers as a ‘bud bank’ that can grow over time, particularly after disturbances where the number of buds multiplies with successful sprouting. The lignotuber structure itself is very long-lived with an apparent unlimited ability to vegetatively regenerate following disturbance (Nicolle 2006). For example, radio carbon dating of an isolated population of *Eucalyptus globulus ssp. bicostata* in South Australia identified that one stand in the population originated from a single lignotuber genet that was between 1000 to 4000 years old (Vaillancourt et al. 2001). Forest stands that originate from lignotubers may, therefore, have a very young aboveground part that is no older than the last fire or disturbance event and an extremely old lignotuber and rooting structure (Nicolle 2006). The long-lived nature and unlimited ability of mature lignotubers to produce vegetative sprouts imparts a degree of resistance within species as regeneration can still occur after repeated disturbance when suitable conditions for seed based germination are infrequent or absent. Noble (2001) highlighted that although the ‘bud bank’ provided by lignotubers will help perpetuate a species occurrence, the maintenance of genetic diversity requires periodic events that coincide with large-scale seedling-based regeneration. Seedling-based regeneration is therefore very important in the succession of many lignotuberous species-dominated stands since regeneration is primarily from a pool of lignotuberous seedlings that establish from seed and remain suppressed under the overstorey until released by a disturbance (Walters et al. 2005b). The presence and density of lignotuberous seedlings is dependent on the ability of seed to germinate and grow under established forest conditions (Walters and Bell 2005).

After a disturbance the regeneration of a stand by lignotuber seedlings or lignotuber sprouts is not guaranteed. Successful sprouting is determined by plant vigour prior to damage and the supply of carbohydrate reserves (Whittock et al. 2003; Walters et al. 2005a). Under sub-optimal conditions, limitations by low carbohydrate reserves may not be critical for success in sprouting from mature individuals but in seedlings the lack of carbohydrates can prevent successful sprouting (Walters et al. 2005a). This is supported by Fensham and Bowman (1992) who suggested that plants with small lignotubers may not be able to develop a sufficient carbohydrate reserve to produce successful saplings. This indicates that species or genotypes of species that have larger lignotubers are more likely to survive multiple disturbance events before sprouting becomes limited by carbohydrate reserves (Walters et al. 2005a; 2005b).

The edaphic conditions of the site also influence the vigour of lignotuberous seedlings and sprouts. Neave and Florence (1998) found that the vigour of lignotuberous seedlings/ sprouts was not related to the size of the lignotuber but to a preconditioning effect of environmental stress with drier, less fertile edaphic sites having more vigorous and resilient lignotubers than individuals found on mesic to moist, fertile sites. Neave (1987) also found that the number of lignotuberous seedlings declined as the edaphic gradient shifts from xeric to mesic to hygric with few seedlings found on mesic sites.

The vigour and available carbohydrate reserves following a disturbance are very important for the success of lignotuberous seedlings/ sprouts since little growth is directed to vertical growth following release. In the early stages of development, meristematic effort is focussed on maximising photosynthetic area to optimise nutrient uptake and carbon assimilation (Noble 2001). During this initial phase of development, young lignotuberous seedlings are vulnerable to reserve exhaustion which can lead to regeneration failure (Walters et al. 2005a). Under suboptimal environmental conditions, lignotuber-based regeneration can fail due to the exhaustion of carbohydrate reserves by the young lignotuberous seedlings; successful regeneration requires a well developed root system under these circumstances (Fensham and Bowman 1992). However, even with a well-developed root system regeneration may not be successful as mortality of the lignotuber and sprouts may occur due to an imbalance between photosynthesis and respiration (Noble 2001). Success of lignotuber regeneration from mature individuals with extensive root structures relies on the ability of the sprouts to produce enough photosynthetic surface area to offset a large respiratory imbalance (Noble 2001).

Under environmental conditions that prevent seed-based regeneration, the ability to regenerate vegetatively through lignotubers is an important mechanism that provides a large degree of resistance and resilience to species that occupy dry, fire prone environments. Despite this increased resilience, survival after disturbance is not guaranteed because sprouting success for different species ranges from 30 to 80 % (Strasser et al. 1996). The presence of lignotuberous seedlings also does not guarantee success because where there are optimal conditions for ‘large wave seedling regeneration’ (Florence 1996), lignotuberous seedlings and sprouts can be out competed by faster growing true (seed-based) seedlings (Walters and Bell 2005; Lutze and Faunt 2006). However, under adverse climatic or edaphic conditions lignotuberous seedlings will out-compete true seedlings (Carrodus and Blake 1970).

It is important to note that lignotuberous species must rely on seedling based regeneration in the wet sclerophyll forests of the Central Highlands since advanced growth is generally inadequate in these areas (Florence 1996). Advanced lignotuberous growth must be supplemented with new seedlings for regeneration of the forest to be successful (Florence 1996). In both instances the environmental and edaphic conditions along with the regeneration strategy that optimises a species regeneration niche can cause shifts in ecosystem composition from being dominated by seed-based to lignotuber-based regeneration and vice versa (Kellas 1994; Lutze and Faunt 2006). Another important caveat is the persistence of seed on a site. Eucalypts, beech and sassafras have short seed viability that typically does not persist for greater than 1 year in the soil (Cremer 1965; Howard 1973a; Hickey et al. 1982). In stark contrast, *Acacia* seeds can remain dormant and viable in a soil seed bank for decades, or even centuries, until released by disturbance (Gilbert 1959; Jennings 1998; Lynch et al. 1999). Therefore, the persistent seed bank of *Acacia* species will increase the seed capital of this genus versus which may expand the fundamental and realised regeneration niche of these species (Brown et al. 2003). The same can be assumed for species that provide a viable, persistent bud bank from a lignotuber.

Under the context of predicted climate change in Victoria, the presence of lignotubers could impart a level of resilience within the regeneration ranges of species where climatic conditions become unfavourable for seed-based regeneration. However, the ability of a species to shift its range in response to changes in climate will still require seed-based germination to ensure resilience over the long-term. Therefore, lignotubers can be seen as expanding the regeneration niche of species once a seed-based seedling becomes established.

3 Methods for Assessing Potential Impacts of Climate Change

Vulnerability assessments are recommended as the best method for assessing potential climate change impacts (IPCC 1998; Lemmen and Warren 2004). Using this approach we have analysed the possible effects that predicted climate change will have on the resilience of the dominant tree species in the Central Highlands. The research presented in this report follows the approach used by Nitschke (2006) and Nitschke and Innes (2008) to model species and ecosystem resilience to changes in phenology, frost damage, drought risk, and heat stress in the temperate forests of British Columbia, Canada.

Tree species of the dominant Ecological Vegetation Classes (EVC) described in the Central Highlands Forest Management Plan (Department of Natural Resources and Environment 1998) were selected for analysis. Twenty-two species were selected [see Table 1 - binomials follow Boland et al. (1992) and Costermans (1994)], representing two broad regeneration strategies: 1) species that are entirely reliant on seedling regeneration; and 2) those with regeneration based on mixed seedling and advanced growth (Florence 1996). Nicolle (2006) classified eucalypt species into finer-scale regeneration classes: 1) lignotuber sprouter; 2) stem sprouter; 3) combination sprouter; and 4) obligate seeder. Table 1 presents the regeneration strategy classification of each selected species.

3.1 The Ecological Model

The ecological model, TACA (Tree And Climate Assessment) (Nitschke and Innes 2008), was modified and parameterised for use in the ecosystems of the Central Highlands. TACA is a mechanistic model programmed in Microsoft Excel (Microsoft 2002). The model analyses the response of trees in their fundamental regeneration niche to climate-driven phenological and biophysical variables. It provides a vulnerability analysis that uses driving variables to determine the probability of species presence/absence. The modelling of presence/absence reflects the regeneration niche of a species, because presence is directly related to establishment, providing a modelling approach that is robust to life-history changes in species (McKenzie et al. 2003). A literature review on the autecology of selected species identified that the majority of existing TACA variables were portable to Australia; the exceptions being bud break and chilling requirements. Also the presence of

Table 1: Central Highland tree species assessed in study

Common Name	Scientific Name	Regen Strat.*
Silver Wattle	Acacia dealbata Link	OS
Montane Wattle	Acacia frigescens J.H. Willis	OS
Black Wattle	Acacia mearnsii De Wild.	OS
Blackwood Wattle	Acacia melanoxylon R. Br.	Seed (C)
Southern Sassafras	Atherosperma moschatum Labill.	Seed (C)
Mountain Grey Gum	Eucalyptus cypellocarpa L. Johnson	Seed & CS
Mountain Gum	Eucalyptus dalrympleana Maiden ssp. dalrympleana	Seed & CS
Alpine Ash	Eucalyptus delegatensis R. Baker	OS
Broad-leaved Peppermint	Eucalyptus dives Schauer	Seed & CS
Tingaringy Gum	Eucalyptus glaucescens Maiden & Blakely	Seed & CS
Victorian Blue Gum	Eucalyptus globulus Labill. ssp. bicostata (Maiden et al.) Kirkpatr.	Seed & CS
Red Stringybark	Eucalyptus macrorhyncha F. Muell. ex Benth. ssp. macrorhyncha	Seed & CS
Shining Gum	Eucalyptus nitens (Deane & Maiden) Maiden	OS
Messmate Stringybark	Eucalyptus obliqua L'Herit.	Seed & CS (WL)
Swamp Gum	Eucalyptus ovata Labill.	Seed & CS
Snow Gum	Eucalyptus pauciflora Sieb. ex Spreng. ssp. pauciflora	Seed & CS
Narrow-leaved Peppermint	Eucalyptus radiata Sieb. ex DC. ssp radiata	Seed & CS
Mountain Ash	Eucalyptus regnans F. Muell.	OS
Candlebark Gum	Eucalyptus rubida Deane & Maiden	Seed & CS
Silvertop Ash	Eucalyptus sieberi L. Johnson	Seed (SS)
Manna Gum	Eucalyptus viminalis Labill. ssp. viminalis	Seed & CS
Myrtle Beech	Nothofagus cunninghamii (Hook.) Oersted	Seed (C)

* Regeneration Strategy: Seed (regeneration by seed), SS (Stem Sprouter), CS (Combination Sprouter: lignotuber sprouting and stem coppice), C [Coppice; vegetative regeneration by root suckering but not defined by Nicolle (2006)]; OS (Obligate Seeder: regeneration only by seed); WL [Weakly Lignotuberous (Florence 1996)]. Strategy in brackets () indicates secondary regeneration strategy.

lignotubers in most eucalypt species is a regeneration mechanism that is absent from Northern Hemisphere species. Therefore, to model the selected tree species for the Central Highlands, TACA was modified to calibrate the model to the ecological regeneration requirements of the selected species. This included the development of a sub module that focused on lignotuber sprouting based regeneration. This calibration lead to the development of TACA-version Australia, abbreviated to TACA-OZ. The determination of species response to climate is described below and illustrated in Figures 2 and 3. The driving variables for TACA-OZ are:

- Growing Degree Day thresholds (GDD Min and GDD Max);
- Species-specific threshold Temperature (T_{base});
- Minimum Temperature (Min T);

- Chilling Requirement (CR);
- Seed Break (SB);
- Secondary Dormancy (SD)
- Drought;
- Number of frost days;
- Growing season frost;
- Lignotuber sprouting.

3.1.1 Growing Degree Days

The determination of presence/absence was based on the following steps outlined by Nitschke and Innes (2008). Minimum and maximum GDD thresholds are used to determine the lower and upper relationship limits between temperature and growth (Shugart and Noble 1981). If the maximum and minimum requirements are not met, minimum growth rates occur that can result in species mortality (He et al. 1999), while increases in temperatures can restrict and prevent species from re-establishing on a site (Franklin et al. 1992; Dale et al. 2001). In TACA-OZ, if the GDD thresholds were not met, or the maximum threshold was exceeded, it was assumed that the regeneration niche of a species was exceeded and that the species was unable to regenerate.

3.1.2 Basal Temperature

Species-specific basal temperatures are used to initiate physiological activity (Fuchigama et al. 1982). The accumulation of degree days above the basal temperature threshold occurs until a species-specific heat sum is reached, which then initiates bud break (Fuchigama et al. 1982). The timing of bud break is expected to occur at earlier dates due to climatic change, which may increase the risk of damage by growing season frosts (Cannell and Smith 1986, Lavender 1989).

3.1.3 Minimum Temperature (Killing Frost)

Minimum temperature thresholds are used to determine if killing frosts occur. In TACA-OZ, if a minimum temperature was reached or exceeded then tree fatality occurred (Nitschke and Innes 2008).

3.1.4 Seed Break and *Chilling Requirements* for Stratification

Climate change may interfere with the ability of trees to meet their chilling requirements for bud break, flowering and germination (Loehle and LeBlanc 1996). The inability of a species to obtain its chilling requirement can seriously affect the species' ability to re-establish after disturbance, facilitating changes in community composition (Shafer et al. 2001). The bud break and chilling mechanisms in Australia do not pertain to the breaking of winter dormancy as they do in the North Hemisphere. In the Northern Hemisphere some species are very sensitive to late spring frosts, resulting in very high chilling requirements being required before bud break will occur (Sykes and Prentice 1995). The native flora of Australia typically lack specialised resting buds, instead they have unprotected terminal and auxiliary buds (Sakai et al. 1981). The absence of dormant buds and their coniferous habit are adaptations to allow for small and variable photoperiodic responses to unfavourable conditions (Sakai et al. 1981). Cremer (1975) suggested that the dormancy of eucalypts during winter is due to quiescence, [a form of dormancy that is imposed by an unfavourable environment (Romberger 1963)], not rest. This is supported by Sakai et al. (1981) and illustrates the difference between dormancy in Australian plants and the plants of the Northern Hemisphere. Germination is the first of the physiological responses influenced by the environment, and correlation has been found between environmental cues and germination success (Bell 1994). Here, seed break and chilling mechanisms play a role in the germination physiology of the selected species through the breaking of primary seed dormancy. In TACA-OZ, these mechanisms were incorporated by utilising the "seed break" variable to model heat sum accumulation for "seed break" and the "chilling requirement" variable to calculate the chilling weeks needed for the breaking of primary seed dormancy. For germination to take place a species must achieve its heat sum and mean temperature must be above a species-specific threshold. In TACA-OZ, if the chilling requirement and/ or germination conditions were not met then a species was assumed to be unable to regenerate.

3.1.5 Secondary Dormancy

The presence of a secondary seed dormancy mechanism for some of the selected species needs to be considered. Secondary dormancy is initiated by high temperatures and low soil moisture conditions and can prevent regeneration (Cunningham 1960; Grose 1963; Gibson and Bachelard 1986; Stoneman 1994). This mechanism ran as a subroutine that tracked temperature and soil moisture conditions that impart secondary dormancy in order to determine if germination conditions are met. If mean daily temperature exceeds a species' temperature threshold and/ or soil moisture was below its specified threshold than secondary dormancy would be invoked and germination prevented at that time. If germination conditions are never met then a species is assumed to be to be unable to regenerate.

3.1.6 Frost Days

The number of frost days is also an important mechanism that can limit the upper elevation distribution of species (Shugart and Noble 1981). Therefore, a “number of frost day” threshold was utilised in TACA-OZ as a driving variable. A frost day is considered to occur when minimum temperatures are <= 0 °C. If the number of frost days exceeds a species threshold then a species is assumed to be unable to regenerate.

3.1.7 Growing Season Frosts

Species whose chilling requirements are satisfied, even under increased temperatures due to climate change, will have the date of mean seed break occurring at a lower mean temperature than is presently the case, thereby increasing the risk of frost damage. Species are most susceptible to frost damage and mortality during the regeneration phase (Murray et al. 1994). Therefore, frost modifiers were used in the TACA-OZ model to limit the probability of species presence. Growing season frosts can also kill buds, terminal twigs or the entire plant. In addition, plants damaged by frost are more susceptible to damage by disease and insects (Murray et al. 1994; Dale et al. 2001). For these reasons, species had their probability of presence reduced in TACA-OZ if they were subjected to a growing season frost event. A frost event is considered to occur when minimum temperatures are <= 0 °C.

3.1.8 Drought

Drought plays a major and direct role in shaping species distributions (Swetnam and Betancourt 1998; Aber et al. 2001; Hannah et al. 2002). It is regarded as a major limiting factor for determining tree species range limits (Sykes and Prentice 1995). Hogg and Wein (2005) identified that forests are very sensitive to drought during the regeneration phase and state that drier conditions due to climatic change could exacerbate this vulnerability and restrict or prevent regeneration. Drought conditions have, and will, prevent establishment of species on a site and cause mortality of established seedlings (Spittlehouse and Childs 1990; Whitmore 1998; Midgley et al. 2002). Drought in TACA was defined by the number of months where rooting-zone groundwater is absent during the growing season (Klinka et al. 2000). Drought was calculated based on the ratio of actual evapotranspiration (AET) to potential evapotranspiration (PET) which is determined by the annual water balance

(Thornwaite and Mather 1955; Thornwaite et al. 1957; Oke 1987). In TACA-OZ, presence/absence under drought was determined by species-specific thresholds related to the proportion of the growing season that can be survived under a water deficit. If the threshold is exceeded, then a species is assumed to be absent from the site.

3.1.9 Lignotuber Sprouting

Sprouting from lignotuber and epicormic buds is an important mechanism that allows the majority of eucalyptus species to recover following disturbance; particularly fire (Gill 1981). Sprouting, particularly form lignotubers is a primary or secondary regeneration mechanism in many eucalyptus species (Florence 1996). Sprouting from lignotubers was utilised by Shugart and Noble (1981) in their BRIND model to model forest succession in the Brindabella Ranges, ACT. Their sprouting mechanism was later modified by Strasser et al. (1996) to improve the realism of this mechanism by including species-specific sprouting probabilities following disturbance. In TACA-OZ species-specific sprouting probabilities were used to modify the probability of a species regenerating by sprouting. This was done through the assumption that if drought thresholds are exceeded, mortality of the main stem will occur allowing for the release of lignotuberous buds. Sprouting from epicormic buds were not modelled and were assumed to represent the survival of an individual following disturbance, not regeneration. This assumption is supported by Ashton and Spalding (2001) who found that coppicing from short-term droughts may allow for recovery of an individual but severe or prolonged droughts can result in complete death of the stem with survival and regeneration only possible from a lignotuberous bud bank. Coppicing from epicormic buds in roots and stems was also not modelled in TACA-OZ because it is not considered a reliable regeneration mechanism in many of the selected species (Florence 1996; Walters and Bell 2005). For example, Silvertop Ash does coppice from epicormic buds, but its primary regeneration mechanism is large wave seed regeneration (Florence 1996; Connell et al. 2004; Bassett et al. 2006). For this study, sprouting regeneration was modelled to occur only from vegetative buds within lignotubers.

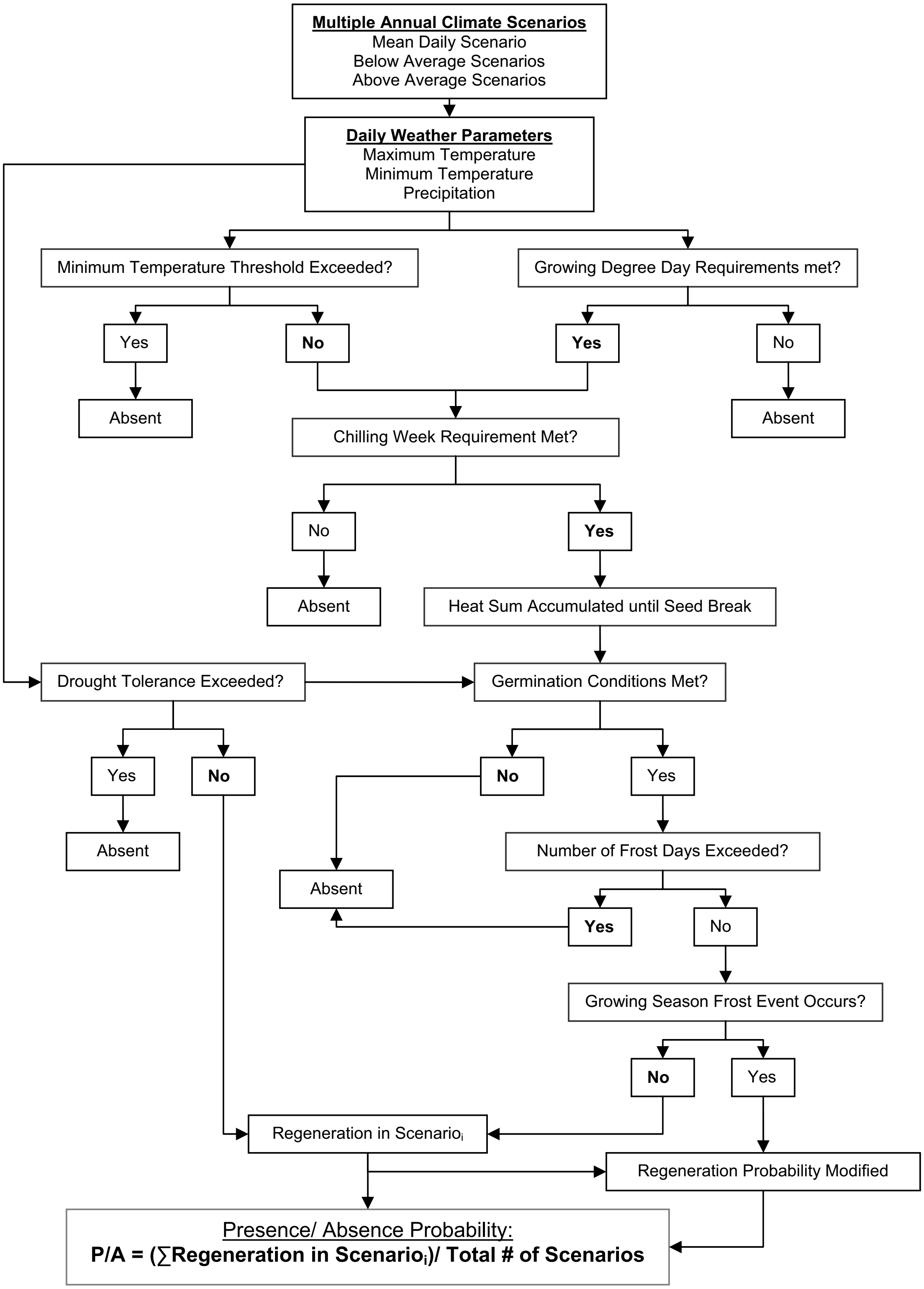


Fig. 2: Diagram of model components and information flow in TACA-OZ

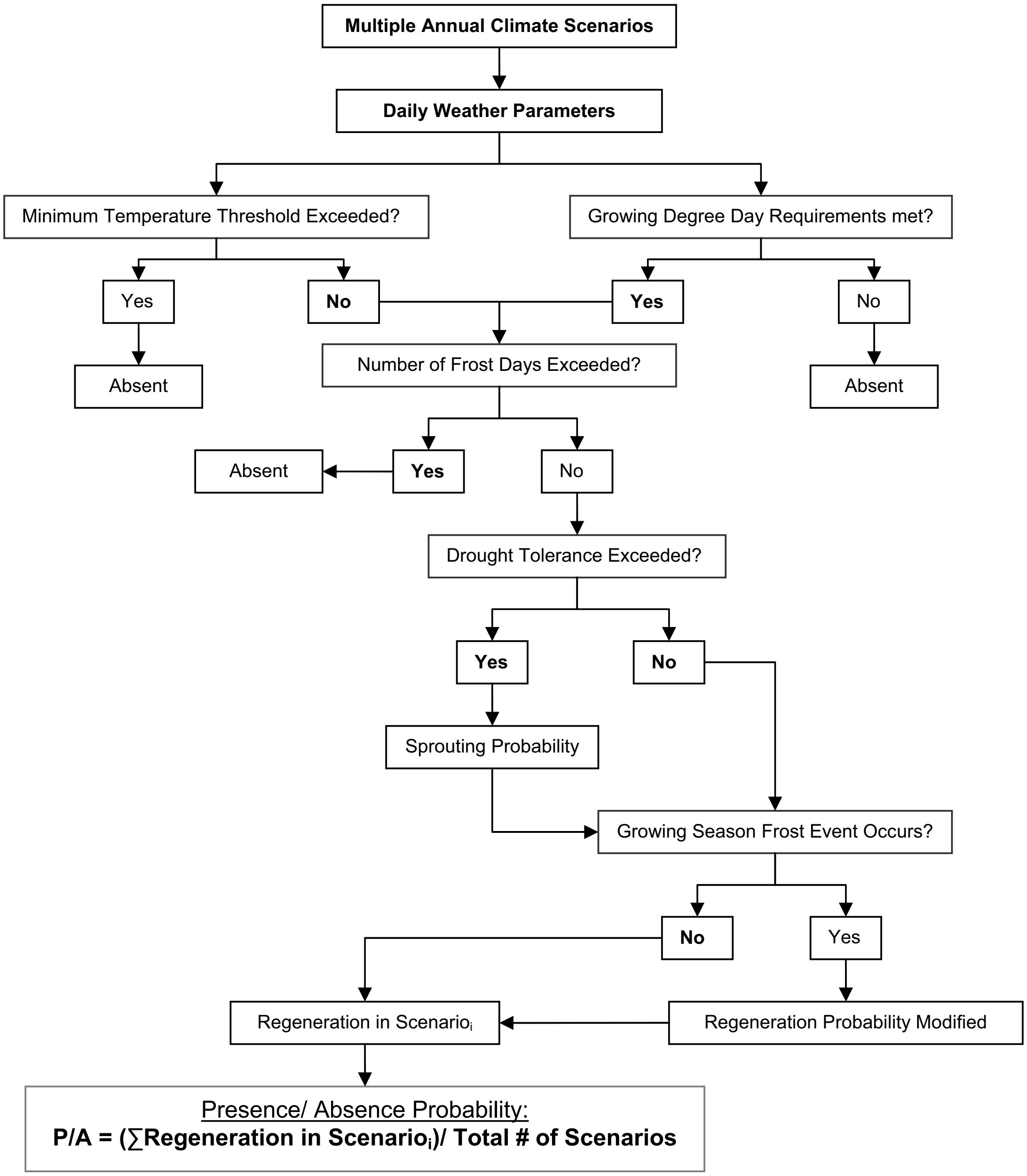


Fig. 3: Diagram of model components and information flow in the lignotuber sub-model in TACA-OZ

3.1.10 Model Limitations

A detailed description of model limitations is provided by Nitschke and Innes (2008).

3.2 Species Parameterisation

Species-specific parameters used to calibrate TACA-OZ were developed in one of four ways: 1) calculation; 2) literature-based 3) calculation with literature supported validation; and, 4) estimation. Species-specific parameters used in TACA-OZ are summarised in Table 2. Literature-based parameters are based on both peer-reviewed studies or databases and non-peer reviewed sources. Table 3 summarises the literature sources used for each species. The growing degree day; drought threshold, frost damage, seed break and soil moisture driving variables were calculated from literature based sources and/or species-climate analysis. Where available, calculated variables were validated using peer-reviewed literature.

3.2.1 Growing Degree Days (GDD)

Maximum and minimum degree days were calculated for the selected Central Highland species using two independent analyses. The Royal Botanical Garden's (RBG) Virtual Herbarium database (RBG, 2007) and a BIOCLIM-Statewide Forest Resource Inventory (SFRI) correlation analysis using ArcGIS 9.2 (Environmental Systems Research Institute (ESRI) 2006). BIOCLIM (Nix 1986; Busby 1991) is used to model long-term bioclimatic variables over a continuous surface gradient with estimates for locations varying with latitude, longitude and elevation (Garnier-Géré and Ades 2001). BIOCLIM has been widely used in studies of Australian flora and fauna (see Lindenmayer et al. 1996). In this study, 109 m resolution BIOCLIM grids for the Central Highlands were used in conjunction with SFRI data (10 m resolution) for each of the dominant eucalypt species. A GIS overlay analysis was then used to identify the relationship between species presence-absence and BIOCLIM temperature and precipitation variables. The results from the analysis were then used to calculate minimum and maximum GDD thresholds for all eucalyptus species. GDD were calculated following the approach used by Noble and Shugart (1981) and Pausas et al. (1997). The minimum and maximum GDD thresholds were then calculated using the lower and upper temperature variables that correlated with species presence from the BIOCLIM-SFRI overlay analysis. Data for the *Acacia* species, Myrtle Beech and Southern Sassafras were not available in the SFRI data.

The second method used to calculate GDD for all selected species was the Royal Botanical Garden's (RBG) Virtual Herbarium database (RBG, 2007). Species presence-absence data and coarse temperature and precipitation data are provided in the Virtual Herbarium database for each species across Australia. This database has previously been used by Mathews and Bonser (2005) to calculate species range size based on correlations between species presence-absence and mean annual precipitation and maximum temperature. In the present study, the Herbarium data were used to calculate minimum and maximum GDD using the lower and upper temperature variables that correlated with species presence. Because the Herbarium dataset only provided maximum temperature, minimum temperature was calculated by using the average reported difference between maximum and minimum temperature for each month from studies within the region [see Ashton (1986), Kellas (1994), Saveneh et al. (1996), Lutze (1998); Walters and Bell (2005), Flint and Fagg (2007) and the Bureau of Meteorology (2007)]. This approach allowed for proxy scenarios to be created that spread the annual maximum and minimum temperatures on a weighted basis across each month resulting in a more detailed calculation of GDD than would have otherwise been possible from the coarse Herbarium data.

GDD parameters for selected species were also available from Shugart and Noble (1981). The Shugart and Noble (1981) parameters were calibrated for the selected species in the Brindabella Ranges, ACT and were therefore integrated with the GDD thresholds calculated from the BIOCLIM and Herbarium analyses. Figures 4 and 5 illustrate the results from the analyses used to calculate GDD thresholds.

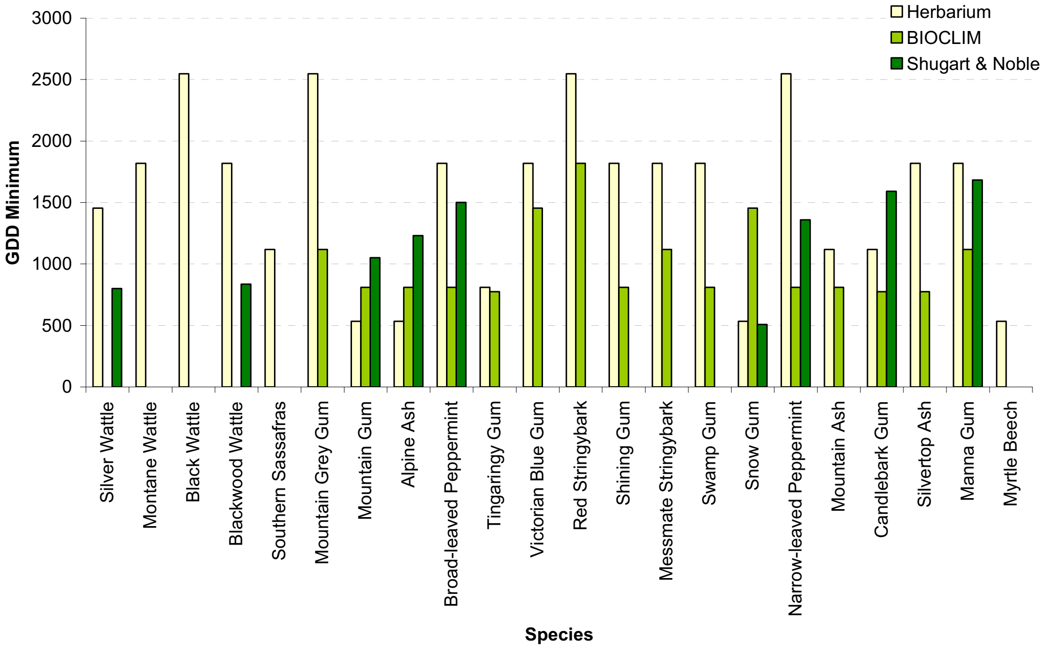


Fig. 4: GDD minimums from calculated from each method

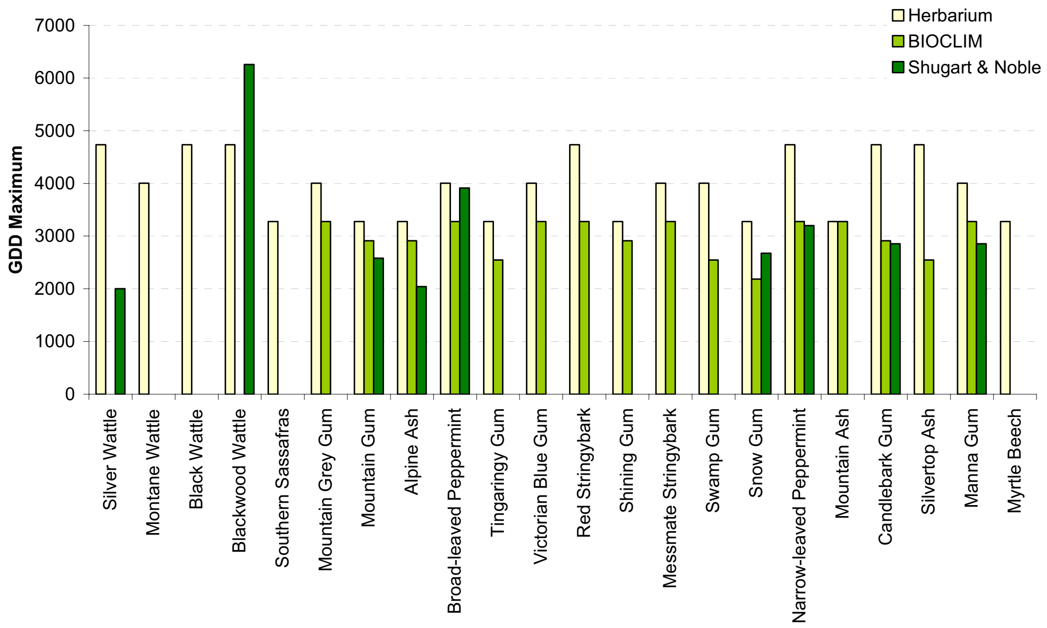


Fig. 5: GDD maximums from calculated from each method

The final GDD thresholds utilised the average threshold from all analyses, except where the averages did not make logical sense or were not supported by Shugart and Noble (1981). The minimum GDD requirements for Silver Wattle, Blackwood Wattle and Snow Gum are from Shugart and Noble (1981) as is the maximum GDD threshold for Candlebark Gum. The minimum GDD threshold calculated by the BIOCLIM analysis for Shining Gum was used because the average did not make logical sense in terms of the minimum GDD requirements for commonly associated high elevation species (Boland et al. 1992). Shining Gum was represented by few data points in the Herbarium data (36 points for all of Australia) compared to the BIOCLIM-SFRI data (occurs on 11176 ha in study area) and this likely resulted in a biased response. The Herbarium data were used to calculate the maximum GDD threshold for Red Stringybark to match this species with its common associate, Black Wattle (Boland et al. 1992). This was done because Red Stringybark was poorly represented in the SFRI data (reported on only 0.2 % of study area) leading to a biased representation of the species. By expanding the GDD range for these species in a manner that makes ecological sense, the estimated GDD thresholds are conservative. This approach is supported by the literature (see Shugart and Noble 1981; Shugart 1984).

3.2.2 Base temperature

Base temperatures were calculated for each species using the GDD minimum threshold and, where possible, validated by peer-reviewed literature. One GDD is assumed to equal 4.44 °C (Shugart and Noble 1981; Pausas et al. 1997). The base temperature, the temperature from which a species become physiologically active, was calculated using the following equation:

$$T_{base} = (\text{GDD Minimum}/365) + 4.44\ ^{\circ}\text{C}$$

The calculated base temperatures for six species were then validated with peer-reviewed literature. Southern Sassafras is described by Read and Busby (1990) as having a physiological base temperature between 7 and 8 °C; the method used in the present study calculated a base temperature for Southern Sassafras of 7.5 °C. Read and Busby (1990) identified that Myrtle Beech becomes physiologically active at 6 °C; in our study the base temperature was calculated as 5.9 °C. Battaglia (1996) identified that Alpine Ash becomes physiologically active between 6 and 7 °C; in our study 6.8 °C was calculated to be the base temperature. Ashton (1975) found that Mountain Ash becomes physiologically active between 5 and 7.5 °C, while Cremer (1975) stated its base temperature is below 10 °C; in our study 7.1 °C was calculated. Ashton (1975) also reported that Silver and Blackwood Wattle become physiologically active between 5 and 7.5 °C; in our study 6.6 °C and 6.7 °C were calculated for each respective species. Keatley and Hudson (2000) calculated the base temperatures for *Eucalyptus* species in the hotter and drier climate occurring Ironbark-Boxwood forests of Victoria as being between 9.9 – 14.1 °C. By way of comparison, the two selected species that are found in the hottest and driest areas of the Central Highlands study area, Red Stringybark and Black Wattle, both had calculated base temperatures over 10 °C. A comprehensive and rigorous validation of species base temperatures was not possible in this study. However, the congruency between calculated and reported base temperatures does provide a degree of independent validation for these parameters (Shugart 1984). Nevertheless, additional research is required to further validate the attributes of all selected species.

3.2.3 Drought Threshold

The drought threshold was calculated based on the relationship between mean annual precipitation and species presence-absence. The lowest precipitation value in which a species was recorded as being present was then used to calculate the drought threshold using the drought sub-model in TACA (Nitschke and Innes 2008). To calculate the drought threshold, proxy climate scenarios were developed to model soil moisture response (measured by the AET/PET Ratio) on a daily time step for one year. The temperature-species presence data derived from the BIOCLIM and Herbarium analyses where also used to provide the temperature component. The proxy climate scenarios were created by spreading annual precipitation, maximum and minimum temperatures on a weighted basis across each month. The proportion that each month contributed to the annual average was calculated from climate data used in previous studies with the region [Ashton (1986), Kellas (1994), Saveneh et al. (1996), Lutze (1998); Walters and Bell (2005), Flint and Fagg (2007)], and from historical climate data provided by the Bureau of Meteorology (2007).

For precipitation, the average number of rain days for each month from the Healesville region, located at the centre of the Central Highlands (Bureau of Meteorology 2007), was used to represent wet and dry periods. The "number of rain days" were then systematically spread across each month. The model was run with three soil types representing the two most common types found in the study region, skeletal soils and mountain soils (Attiwill and Leeper 1987). Three common textures, loam, loam-clay loam and clay loam were used along with three depths 60 cm, 120 cm and 180 cm (Ashton 1976; Ashton 1986; Attiwill and Leeper 1987; Bryant et al. 1992; Squire et al. 2006). These soil characteristics represent common soil textures and depths that are found across topographic gradients in the study area, and also represent the maximum root depth achieved by eucalypts during the first and second year following successful regeneration (Ashton 1975b). Where possible, the results were validated with literature-based data.

The results of the analysis are presented in Fig. 6. As was the case with GDD threshold calculations, the BIOCLIM-SFRI analysis was only used to calculate the drought threshold for the eucalyptus species. Species response to the different approaches resulted in congruent responses for some species and divergent responses for others. Snow Gum (*E. pauciflora*), for example, was calculated as having a threshold of two months by the BIOCLIM-SFRI approach versus five months by the Herbarium approach. The poor representation of Snow Gum in the BIOCLIM-SFRI data was the likely cause of this. In the Herbarium data, Snow Gum had 80 data points while in the BIOCLIM-SFRI there were only 32, representing 0.0003 % of the total study area. This is contradictory to the proportion of the study area that is occupied by the subalpine woodland ecosystem (6.2 %)

which is dominated by Snow Gum. This discrepancy was due to the fact that the SFRI data focussed predominantly on forest with relatively high productivity (assessed in terms of stand height) and composed of important timber species (Hamilton and Brack 1999).

The five species that could be validated by independent literature sources indicated that the approach used in the present study provided realistic estimates of species drought thresholds. In particular, the results for Manna Gum (*E. viminalis*) were very encouraging because both the BIOCLIM and Herbarium approaches provided the same estimate, which was also the threshold reported by Jovanovic and Booth (2002). For Tasmanian Blue Gum (*E. globulus ssp. globulus*), Jovanovic and Booth (2002) reported a drought threshold of five months, and this was also calculated by the BIOCLIM approach. In comparison, the Herbarium approach calculated a threshold of 6 months, however *E. globulus ssp. bicostata* has been found to be more drought tolerant than *E. globulus ssp. globulus* and is found in hotter and drier areas (Costermans 1981; 1994). The Herbarium dataset has a greater geographic range than the BIOCLIM-SFRI dataset and this may account for the small divergent response. Pook et al. (1966) identified that mortality of Red Stringybark (*E. macrorhyncha*) occurred on sites that experienced soil water deficits, AET/PET ratio < 0.30, for greater than 6-7 months. In both the BIOCLIM and Herbarium approaches, a threshold of 6 to 7 months was calculated for this species, providing a point of validation. Shining Gum (*E. nitens*) was calculated as having a threshold of 4 months by the Herbarium approach, and this is supported by White et al. (1996) and Jovanovic and Booth (2002). Although not directly related, Florence (1964) found that warm temperate rainforest species, which included a *Nothofagus* species, could survive water deficits for up to three months in New South Wales. In our study, Southern Sassafras and Myrtle Beech were calculated to have drought threshold of three and two months respectively. Howard (1973c) reported that myrtle beech seedlings survived 7 weeks under soil-water deficits followed by mortality in week 8 – findings that support the threshold calculated in our study. The species-specific drought thresholds are presented in Table 2. These were calculated by dividing the average “threshold months” across all approaches by 12 months to indicate one year.

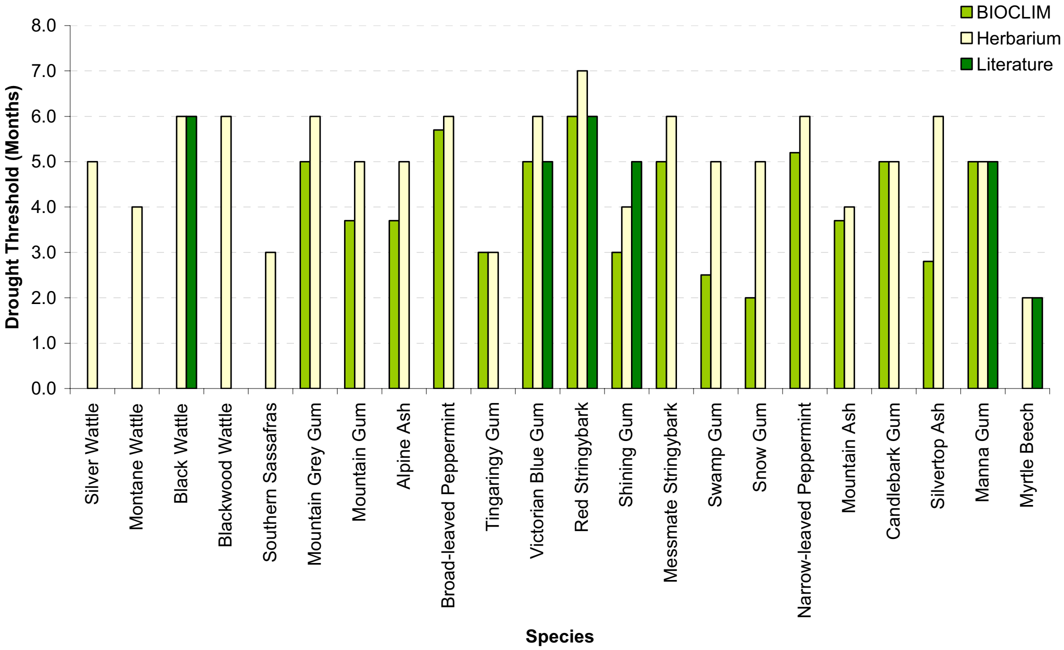


Fig. 6: Drought thresholds for each species from calculated from each method. Results are presented as the maximum number of months that a species can survive with "available water" absent from the rooting zone

Table 2: TACA-OZ Model Driving Variables and Species-specific Parameters

Species	Growing Degree Days			Drought**	Frost Variables			Germination & 1° Dormancy		Germination & 2° Dormancy			Lignotuber
	Min^{1}	Max^{1}	Base* Temp^{1}	Threshold^{1}	Min* Temp	Frost^{2} Days	Frost$^{\#}$ Damage^{1}	Chilling Weeks^{2}	Seed$^{\#\#}$ Break	Min* Temp	Max* Temp	Soil*** Moisture^{1}	Sprouting Probability
Silver Wattle	800^{3}	3367^{3}	6.6^{3}	0.42	-8^{2}	100	0.534	0	67^{3}	5^{2}	35^{4}	0.30^{4}	0^{2}
Montane Wattle	1819	4004	9.4	0.33	-10^{4}	100	0.612	0	70^{3}	5^{4}	35^{4}	0.30^{4}	0^{2}
Black Wattle	2547^{3}	4733^{3}	11.4	0.50^{3}	-8^{2}	40	0.418^{3}	0	60^{3}	5^{4}	35^{4}	0.30^{4}	0^{2}
Blackwood Wattle	835^{3}	5495	6.7^{3}	0.42	-7^{2}	40	0.301	0	68^{3}	5^{2}	35^{2}	0.30^{4}	0^{2}
Southern Sassafras	1118	3276	7.5^{3}	0.25	-10^{2}	50	0.418	0	70^{4}	3^{4}	32^{2}	0.90^{4}	0^{2}
Mountain Grey Gum	1833	3640	9.5	0.46	-10^{4}	100	0.616^{3}	0	43^{3}	4^{2}	25^{2}	0.47^{4}	0.5^{4}
Mountain Gum	798^{3}	2924^{3}	6.6	0.36	-13^{2}	100	0.612^{3}	0	39^{3}	4^{4}	35^{4}	0.47^{4}	0.469^{2}
Alpine Ash	859^{3}	2743^{3}	6.8^{3}	0.36	-14^{2}	100	0.651^{3}	6	33^{3}	6^{2}	35^{2}	0.73^{2}	0^{2}
Broad-leaved Peppermint	1377^{3}	3731^{3}	8.2	0.49	-9^{2}	80	0.573	0	58^{3}	4^{4}	35^{4}	0.77^{4}	0.479^{2}
Tingaringy Gum	793	2912	6.6	0.25	-12^{2}	150	0.441^{3}	4	26^{3}	5^{4}	29^{4}	0.77^{4}	0.5^{4}
Victorian Blue Gum	1637	3640	8.9	0.46^{3}	-7^{2}	70	0.534	0	44^{3}	4^{4}	35^{4}	0.3^{2}	0.5^{4}
Red Stringybark	2183	4733	10.4	0.54^{3}	-7^{4}	70	0.534	0	45^{3}	4^{4}	35^{4}	0.3^{4}	0.5^{4}
Shining Gum	811	3094	6.7	0.33^{3}	-15^{4}	150	0.689	3	37^{3}	5^{4}	33^{4}	0.47^{4}	0^{2}
Messmate Stringybark	1469	3640	8.5	0.46	-13^{4}	100	0.616^{3}	0	42^{3}	4^{2}	25^{2}	0.47^{4}	$0.11^{1,4}$
Swamp Gum	1315	3276	8.0	0.31	-7^{4}	70	0.534	0	30^{3}	4^{4}	34^{4}	0.47^{4}	0.5^{2}
Snow Gum	509^{3}	2711^{3}	5.8	0.29	-16^{2}	150	0.775^{3}	3	33^{3}	6^{2}	33^{2}	0.54^{2}	0.819^{2}
Narrow-leaved Peppermint	1572^{3}	3737^{3}	8.7	0.47	-9^{2}	100	0.573	0	46^{3}	4^{4}	34^{4}	0.47^{4}	0.333^{2}
Mountain Ash	965	3276	7.1^{3}	0.32	-10^{2}	80	0.588^{3}	0	46^{3}	4^{2}	32^{2}	0.69^{4}	0^{2}
Candlebark Gum	1162^{3}	2854^{3}	7.6	0.42	-13^{4}	100	0.612	0	60^{3}	4^{4}	35^{4}	0.47^{4}	0.371^{2}
Silvertop Ash	1297	3640	8.0	0.41	-7^{4}	70	0.534	0	49^{3}	4^{4}	32^{4}	0.73^{2}	0^{2}
Manna Gum	1540^{3}	3378^{3}	8.7	0.42^{3}	-13^{2}	100	0.612^{3}	0	49^{3}	4^{2}	25^{2}	0.47^{4}	0.553^{2}
Myrtle Beech	534	3276	5.9^{3}	0.17^{3}	-17^{2}	100^{4}	0.612^{3}	0	71^{3}	2^{2}	35^{2}	0.90^{4}	0^{2}

1: Calculated parameter
2: Literature based parameter
3: Calculated and validated by literature
4: Estimated parameter
*: °C; **: proportion of year with AET/PET ratio < 0.30; ***: AET/PET Ratio threshold; #: presence modifying probability; $^{\#\#}$: Heat sum required for seed break

Table 3: Literature used to parameterise TACA-OZ Species

Species	Parameter References
Silver Wattle	Ashton (1975a); Boland (1992); Ensis (2007); Pollock et al. (1986); Royal Botanical Gardens Melbourne (2007); Shugart and Noble (1981)
Montane Wattle	Boland (1992); Ensis (2007); Royal Botanical Gardens Melbourne (2007)
Black Wattle	Boland (1992); Ensis (2007); Jovanovic and Booth (2002); Royal Botanical Gardens Melbourne (2007); Searle et al. (1998)
Blackwood Wattle	Ashton (1975a); Boland (1992); Ensis (2007); Moore (1983); Royal Botanical Gardens Melbourne (2007); Shugart and Noble (1981)
Southern Sassafras	Boland (1992); Hickey et al. (1982); Read and Busby (1990); Read and Hill (1988); Royal Botanical Gardens Melbourne (2007)
Mountain Grey Gum	Boland et al. (1980; 1992); Lutze et al. (1998a; 1998b); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002)
Mountain Gum	Boland et al. (1980; 1992); Moore (1983); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Shugart and Noble (1981); Strasser et al. (1996); Varelides and Bofas (2000)
Alpine Ash	Battaglia (1993; 1996; 1997); Battaglia and Reid (1993); Bell (1999); Boland et al. (1980; 1992); Davidson and Read (1985); Geary and Fryer (1999); Grose (1963); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Shugart and Noble (1981); Stoneman (1994); Varelides and Bofas (2000)
Broad-leaved Peppermint	Boland et al. (1980; 1992); Doran et al. (2005); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Shugart and Noble (1981); Strasser et al. (1996)
Tingaringy Gum	Boland et al. (1980; 1992); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Varelides and Bofas (2000)
Victorian Blue Gum	Boland et al. (1980; 1992); Humara et al. (2002); Jovanovic and Booth (2002); Moore (1983); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Stoneman (1994); Strasser et al. (1996)

Red Stringybark	Boland et al. (1980; 1992); Nicolle (2006); Pook et al. (1966); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Strasser et al. (1996)
Shining Gum	Boland et al. (1980; 1992); Jovanovic and Booth (2002); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Tibbits (1987); White et al. (1996)
Messmate Stringybark	Battaglia and Williams (1996); Boland et al. (1980; 1992); Kellas (1994); Lutze et al. (1998a; 1998b); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Walters and Bell (2005); Walters et al. (2005)
Swamp Gum	Boland et al. (1980; 1992); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002)
Snow Gum	Abrect (1985); Boland et al. (1980; 1992); Close and Wilson (2002); Ferrar et al. (1988); Nicolle (2006); Patton (1980; 1981); Royal Botanical Gardens Melbourne (2007); Sakai et al. (1981); Schütz et al. (2002); Shugart and Noble (1981); Strasser et al. (1996); Varelides and Bofas (2000)
Narrow-leaved Peppermint	Boland et al. (1980; 1992); Doran et al. (2005); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Shugart and Noble (1981); Strasser et al. (1996)
Mountain Ash	Ashton (1958; 1975a); Ashton and Chinner (1999); Boland et al. (1980; 1992); Close and Wilson (2002); Cremer (1965; 1975); Cunningham (1960); Nicolle (2006); Schütz et al. (2002); Stoneman (1994)
Candlebark Gum	Boland et al. (1980; 1992); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Shugart and Noble (1981); Strasser et al. (1996)
Silvertop Ash	Bell (1999); Boland et al. (1980; 1992); Gibson (1985); Gibson and Bachelard (1986a; 1986b; 1988); Nicolle (2006); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Stoneman (1994)
Manna Gum	Boland et al. (1980; 1992); Jovanovic and Booth (2002); Ladiges (1974; 1976); Ladiges and Ashton (1974); Nicolle (2006); Patton (1980; 1981); Royal Botanical Gardens Melbourne (2007); Schütz et al. (2002); Shugart and Noble (1981); Strasser et al. (1996)
Myrtle Beech	Hickey et al. (1982); Howard (1973a; 1973b; 1973c); Read and Busby (1990); Read and Hill (1988); Royal Botanical Gardens Melbourne (2007); Sakai et al. (1981)

3.2.4 Secondary Dormancy & Soil Moisture

Schütz et al. (2002) reported that drought induced secondary dormancy for the majority of eucalypts occurs at soil matric potentials between -0.5 and -1.0 MPa. TACA represents soil moisture using an actual to potential evapotranspiration ratio. To calibrate the observed and hypothesised species response to soil matric potentials a relationship between soil matric potential and AET/ PET ratio was required. In studies conducted across Europe and Asia, a linear response between soil matric potential and AET/PET ratio has been found (Anderson and Harding 1991; Aydin et al. 2005; Henson et al. 2005). Using the data reported by Henson et al. (2005), we used a linear regression to determine if AET/PET ratio could be used as a predictor for soil matric potential (see Fig. 7). Subsequently, a significant relationship was found to exist ($P < 0.001$) with the AET/ PET explaining 90.2 % of the variation. The predicted model has a calculate Power of 1.00. The predictive model has the form of:

$$Y = -1.0603X + 1.2713$$

This predictive model was used to determine the AET/PET ratio that would result in a soil matric potential that would invoke secondary dormancy.

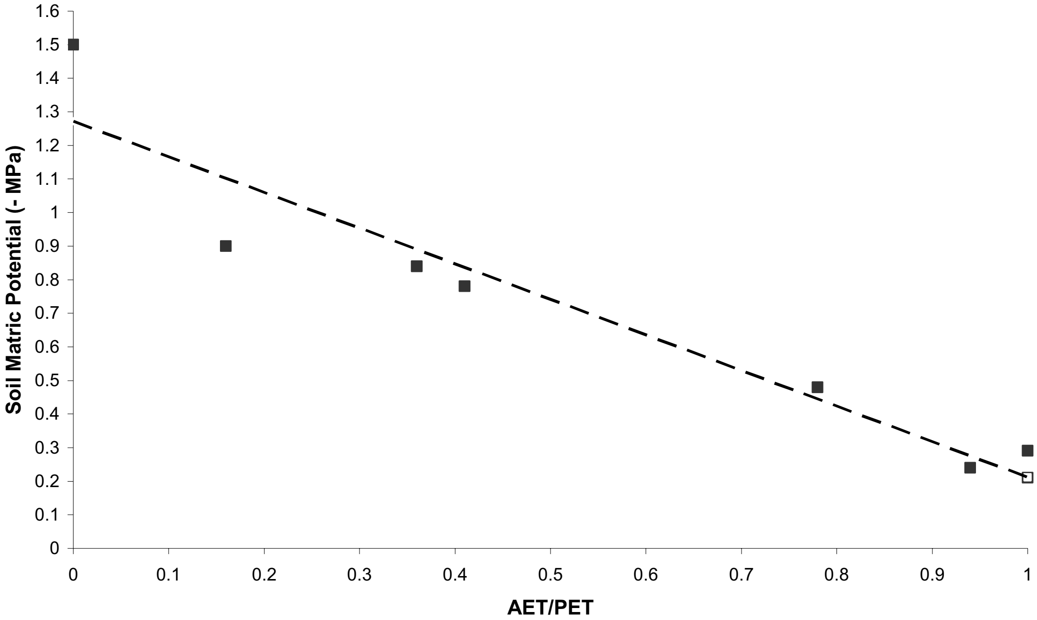


Fig. 7: Soil Matric Potential vs. AET/PET Ratio from Henson et al. (2005). -1.5 MPa = permanent wilting point in soil; 0 = field capacity Significant relationship exists between AET/PET ratio and soil matric potential; $P < 0.001$; adjusted $R^2 = 0.907$; Standard error 0.1337; Power = 1.00

3.2.5 Frost Damage

The tree species located in and adjacent to the Central Highlands have a wide regeneration window, where germination can occur throughout the year if conditions are optimal (Faunt et al. 2006). However, this large regeneration window predisposes different cohorts of germinants to different mortality agents throughout a year (Lutze et al. 1998b), particularly frost and drought (Faunt et al. 2006). Lutze et al. (1998b) found that frost caused the highest degree of mortality to cohorts that germinated in late autumn, winter and early spring. Both Faunt et al. (2006) and Lutze et al. (1998b) reported that survivors of this cohort generally dominated after the first two years following regeneration. To calculate the frost damage modifier for each species in our study, the reported proportion of frost damage a species experiences needed to be weighted to reflect the portion of mortality that frost is responsible for. Battaglia (1996) and Lutze et al. (1998b) both reported the total amount of regeneration mortality of eucalypt species during regeneration and the proportion attributed to frost. Based in these studies, we used a weight of 0.777 to modify the observed degree of frost mortality within individual species over the period of a year where other mortality factors agents such as fungus infections also cause mortality. The weight of 0.777 reflects the proportion of the population that suffer frost damage and do not recover. For example, Ashton (1958) reported that, on average, 50 % of Mountain Ash incurred mortality following a frost event. When the weight of 0.777 was applied to Mountain Ash regeneration in our study, 39 % of the year's regenerating population were assumed to incur mortality if a frost event occurred following germination. In this case, the frost modifier is one minus this mortality, which means 61 % of the regeneration niche would remain available to the population.

3.2.6 Estimated Parameters

Due to a lack of data on many species, it was impossible to calibrate or validate all of the driving variables for the majority of the species. The number of species that required parameters to be calculated without validation, varied from one out of 13 for Alpine Ash to nine of 13 for Montane Wattle. For variables that could not be empirically calculated, the variable was estimated based on associated species that had validated parameters. These species associations were based on Boland et al. (1992) and on a Pearson's correlation analysis of species based on changes in species abundance in response to change in elevation within the Central Highlands (see Table 4).

Table 4: Correlation of species in response to changes in abundance across elevational gradient in the Central Highlands

Species	Mountain Ash	Alpine Ash	Mountain Grey Gum	Mountain Gum	Broad-leaved Peppermint	Tingaringy Gum	Victorian Blue Gum	Red Stringybark	Shining Gum	Messmate Stingybark	Swamp Gum	Manna Gum	Silvertop Ash	Candlebark Gum	Narrow-leaved Peppermint	Snow gum
Mountain Ash	1.00															
Alpine Ash	-0.004	1.00														
Mountain Grey Gum	0.97	0.12	1.00													
Mountain Gum	0.83	0.42	0.87	1.00												
Broad-leaved Peppermint	0.84	-0.08	0.88	0.59	1.00											
Tingaringy Gum	0.18	-0.09	0.27	0.08	0.55	1.00										
Victorian Blue Gum	0.53	-0.31	0.54	0.14	0.87	0.64	1.00									
Red Stringybark	0.23	-0.32	0.31	-0.12	0.61	0.34	0.75	1.00								
Shining Gum	0.22	0.87	0.36	0.66	0.11	-0.04	-0.23	-0.27	1.00							
Messmate	0.66	-0.31	0.69	0.28	0.91	0.54	0.90	0.81	-0.16	1.00						
Swamp Gum	0.03	0.93	0.12	0.43	-0.12	-0.14	-0.36	-0.39	0.74	-0.36	1.00					
Manna Gum	0.82	-0.20	0.83	0.50	0.98	0.57	0.91	0.60	-0.03	0.92	-0.22	1.00				
Silvertop	0.47	-0.23	0.43	0.13	0.75	0.59	0.92	0.49	-0.17	0.70	-0.28	0.82	1.00			
Candlebark Gum	0.12	-0.21	0.21	-0.05	0.56	0.97	0.70	0.50	-0.18	0.62	-0.26	0.58	0.61	1.00		
Narrow-leaved Peppermint	0.69	-0.29	0.71	0.31	0.95	0.57	0.95	0.78	-0.15	0.98	-0.33	0.96	0.80	0.64	1.00	
Snow Gum	0.43	0.16	0.50	0.62	0.33	-0.09	0.03	0.06	0.33	0.16	0.20	0.24	-0.03	-0.08	0.14	1.00

3.3 Climate Parameters

TACA-OZ utilised minimum temperature, maximum temperature, and precipitation on a daily time step for a period of one year. Each year represented one climatic scenario with TACA able to analyse multiple climate scenarios at one time. The multiple scenario approach is required to determine species presence/absence probability under historic, current, and/or future climate scenarios and provides a form of sensitivity analysis to identify the mechanism(s) that drive species response (Nitschke and Innes 2008). The multiple scenario approach also addresses the issues of uncertainty, interdependence and complexity that can confound single scenario analysis (Schoemaker 1993).

Six weather stations were used to develop the multiple climate scenarios for the study. The geographic locations of the six stations are illustrated in Fig. 8 and the characteristics of each station summarised in Table 5. Climate data were obtained for the local weather stations from the Commonwealth of Australia’s Bureau of Meteorology.

Table 5: Summary of weather station characteristics used in the study

Station	Mt. Baw Baw	Woods Point	Noojee	Toolangi	Coldstream	Lake Eildon
Operator	B.O.M	B.O.M	B.O.M	B.O.M	B.O.M	B.O.M
Latitude	-37.8383	-37.5696	-37.9039	-37.5708	-37.7258	-37.2311
Longitude	146.2747	146.2541	145.9719	145.5047	145.4072	145.9122
Elevation	1561 m	680 m	275 m	620 m	89 m	230 m
Aspect	North	North	North	South	South	South
Record	1997-2007	1957-2007*	1981-2007	1965-2006	1994-2007	1970-2007
Average Max Temperature	9.5 °C	17.6 °C	16.9 °C	15.7 °C	20.1 °C	20.2 °C
Average Min Temperature	3.0 °C	3.0 °C	10.6 °C	7.7 °C	7.1 °C	7.8 °C
Average Precipitation	1612 mm	1462 mm	1120 mm	1440 mm	755 mm	872 mm
Ecosystem	Subalpine Woodland	Damp – Montane Damp Forest	Damp Forest	Damp - Wet Forest	Not Defined	Not Defined

* Temperature records from 1957-1969 and precipitation records completely missing for years 1972 and 2000-2003

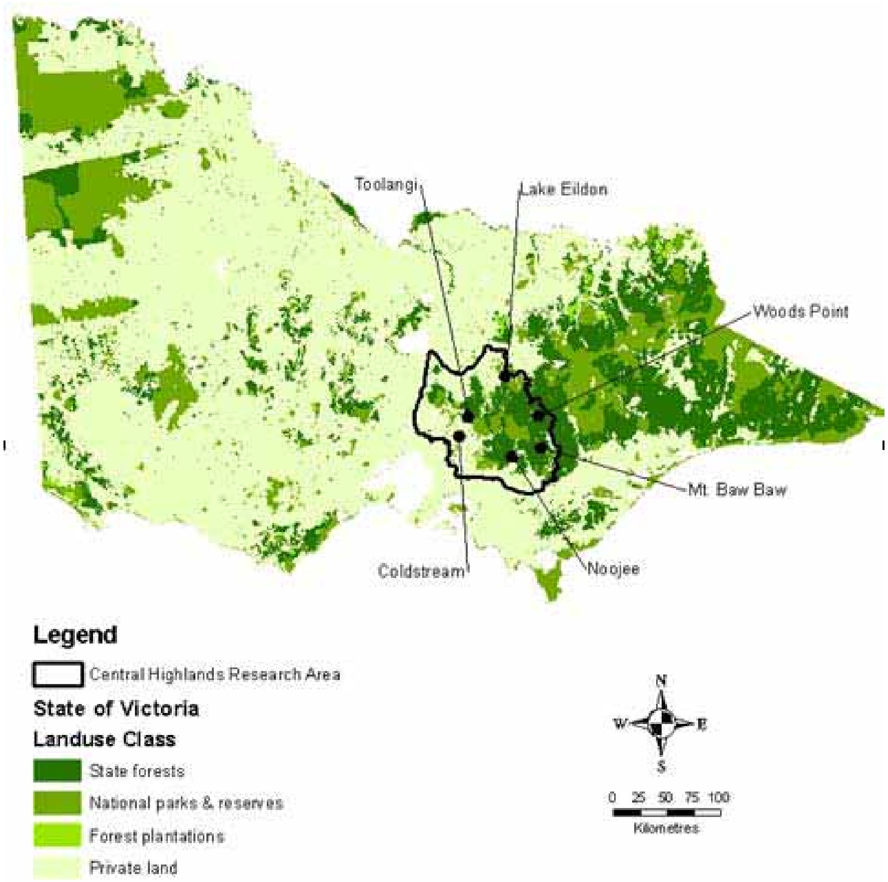


Fig. 8: Central Highlands study area and location of weather stations used in analysis

3.4 Climate Scenarios

Multiple scenarios of current and future climates were used to test the range of species' responses. In modelling climate, we utilised local climate data and global climate change model (GCM) predictions. Three different GCM's were used, CSIRO Mark 2, CSIRO DARLAM 125 km and HadleyCM3 models. The regional climate change predictions for the Central Highlands from each GCM were obtained using CSIRO's OzClim database (Jones et al. 2001). The IPCC's SRES emission scenario, A2 (Nakicenovic et al. 2000), was selected. The A2 SRES emission scenario models climatic change under both increased economic and population growth at regional scales. This storyline was chosen because of its regional representation, a scenario justified by the observed differential change in regional emissions since 2000 between both developed and developing countries (Raupach et al. 2007). The A2 scenario also represents the middle ground between fossil fuel intensive and extensive scenarios. An important caveat to note, observed emissions since 2000 have increased at a greater rate than predicted by the most fossil fuel intensive IPCC SRES scenario (Raupach et al. 2007). Multiple climate scenarios were generated following Nakicenovic et al. (2000), who argued that due to the large amount of uncertainty regarding future climate change, multiple scenarios that span a range of possible future climates should be adopted. Not surprisingly, each of the four models predicted a different future in terms of potential changes in temperature and precipitation in the Central Highlands. The average annual predicted changes for the region, based on an ensemble of the scenarios, are provided in Table 6.

Table 6: Average Annual predicted climate change in Victoria's Central Highlands

Climate Period	Min Temp. (°C)	Mean Temp. (°C)	Max Temp. (°C)	Precipitation (%)
2025 (2010-2039)	0.4	0.5	0.6	-1.8
2055 (2040-2069)	1.2	1.4	1.5	-4.7
2085 (2070-2100)	2.2	2.5	2.8	-8.9

3.4.1 Generation of Climate Scenarios

The development of multiple climate change scenarios for the area required the generation of local weather scenarios to incorporate the recorded variation in daily weather patterns. The methodology used followed Nitschke and Innes (in press). Daily data were used following the protocols of Bürger (1996). Six weather stations were used to provide an integrated representation of regional climate with a microclimatic component. The microclimatic component was incorporated by generating multiple weather scenarios for each station. Error analysis was then used to estimate the sampling distributions for minimum temperature, maximum temperature and precipitation (Pacala et al. 1996) while Monte Carlo techniques were used to sample the probability distributions of parameters to select outcomes (Luo et al. 2005). For each day, August 1 to July 31, the average and standard deviation of each variable was calculated. For days that had less than two weather observations, the sampling distribution of the daily observations from the preceding and following days were used to create the sampling distribution. The arithmetic average for each variable, for each day, was then used to represent the average daily climate for each weather station and to form the average weather scenario for each weather station (Table 5 illustrates the annual average climate).

Monte Carlo techniques were used to sample the probability distributions of each daily variable to create ranges of change. Following Luo et al. (2005), the sampling distribution created by the error analysis was assumed to be a prior distribution for each variable. The sampling method also enabled the maintenance of the same level of independence between each variable. It was assumed that: (1) precipitation ≥ zero and (2) minimum temperature is ≤ maximum temperature. Four of the generated weather scenarios from each station were then randomly selected with two representing a combination of variables below the annual averages (Table 5), and two representing above-average conditions. The scenario combination was chosen to provide an equal representation of below- and above-average conditions, while maintaining high variability between scenarios. Combining the average weather scenario with the four scenarios generated through the use of error analysis and Monte Carlo sampling resulted in the creation of five weather scenarios for each local weather station in the study. This permitted the incorporation of a more realistic level of variability for each local variable than would have been the case if a classic sensitivity analysis that looked at the average, lower and upper bounds for each variable, respectively, had been applied (Pacala et al. 1996; Zorita and von Storch 1999).

A direct adjustment approach was then employed to integrate climate change scenarios into the local weather scenarios generated for each station. This involved adjusting the weather station records using the predicted outputs from a GCM (Wilks 1999; Wang et al. 2006). The predictions for changes in temperature and precipitation for each month from the four climate change scenarios were then applied to the five local weather stations. Changes in temperature were increased by the predicted amount while changes in precipitation were multiplied by a factor that represented a percent change in that variable (ex. 0.92 or 1.12). These changes were applied on a month-by-month basis to create a daily time series of weather that represented the local variation, along with monthly variation of the GCM predictions. By applying four GCM scenarios, broken into 2025 (2010-2039), 2055 (2040-2069), and 2085 (2070-2100) outputs, to five local weather scenarios, a total of 65 scenarios were generated for each station. Multiplied by the number of weather stations, a total of 390 individual scenarios were created that integrated local variation and climate change predictions for the region at six different locations and elevations. These scenarios were used to evaluate the vulnerability of the selected tree species to climate change. Nitschke and Innes (2008) successfully used this approach to model the vulnerability of tree species to climatic change in south-central British Columbia, Canada.

3.5 Study Design: Synthetic Climate Transects

To model the response of tree species across the topographic diversity of the Central Highlands, synthetic climate transects were used. Using environmental lapse rates for temperature and precipitation (Saveneh et al. 1996) and for aspect (Ashton 1976) the climate scenarios where modified to represent climatic conditions at 100 m intervals between 200m and 1600 m in elevation on two aspects. The environmental lapse rates are provided in Table 7. A synthetic climate transect was developed for each weather station. Thirty theoretical plots were established at 100 m intervals along each transect, 15 on exposed sites (north aspect) and 15 on sheltered sites (south aspects). Figure 9 illustrates the synthetic climate gradient design. This approach has previously been used by Zolbrod and Peterson (1999) to model the response of tree species in the Pacific Northwest of the United States of America.

3.6.1 Microclimatic Treatments

At each plot, species responses on four site types under three microclimatic treatments were analysed. Figure 10 illustrates the three treatments with nested site types. Treatment one (T1) represents microclimatic conditions under open conditions following a stand-replacing disturbance; treatment two (T2) represents microclimatic conditions under a partially modified forest canopy following a stand modifying disturbance; and treatment three (T3) represents microclimatic conditions under an intact forest canopy (no-or canopy-maintaining disturbance). The microclimatic treatments represent changes in temperature and precipitation at the forest floor as a result of canopy interception of precipitation and shortwave radiation.

Table 8 summarises the microclimatic modifiers used to developed T2 and T3 from the climate scenarios designed for evaluating T1. Site A represents a skeletal soil profile, with a loam texture and 60 cm depth; Site B represents a mountain soil profile, with a loam-clay loam texture and 120 cm depth; and, Site C represents a mountain soil profile, with a clay loam texture and 180 cm depth. A fourth site with soil excluded was also included to represent non-water limiting sites. The nested site types represent common soil types found in the Central Highlands (Attiwill and Leeper 1987) and represent the edaphic gradient that exists at finer topographic scales as you move from ridge lines to gullies. The assessment of both exposed and sheltered sites, north and south aspects, incorporates an important determinant of species distributions in Victoria [Ashton (1976), Costermans (1994) and Ashton and Spalding (2001)]. The nested split-plot design was designed to incorporate the topographic and edaphic variability that exists in the Highlands and that has a mediating and controlling impact on species distribution and niche breadth.

Table 7: Environmental lapse models used to create synthetic climate transects

Model	Min Temp. (°C)	Max Temp. (°C)	Precipitation (%)
Increase in Elevation	-0.373 /100 m	-0.774 / 100 m	0.053/ 100m
North Aspect to South Aspect	-0.6	-2.8	0

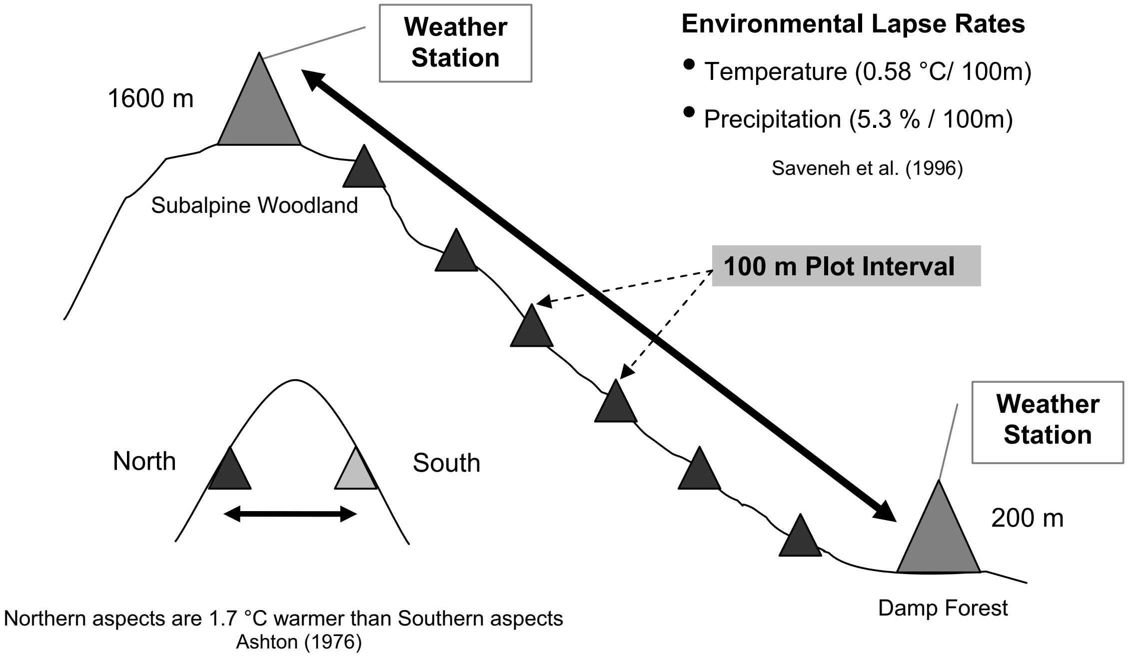


Fig. 9: Synthetic Climatic Gradients. Species response at climate plots at 100m interval from 200 m to 1600m, representing both North and South aspects, were modelled using multiple scenario analysis

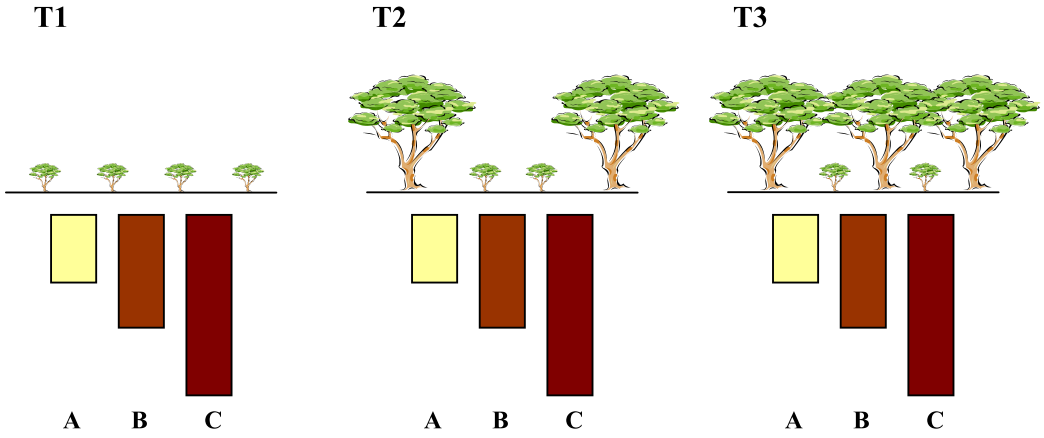


Fig. 10: Forest microclimatic scenarios with nested site types used in analysis. T1 represents microclimatic conditions under an open conditions following a stand-replacing disturbance; T2 represents microclimatic conditions under a partially modified forest canopy following a stand modifying disturbance; and T3 represents microclimatic conditions under an intact forest canopy (no-or canopy-maintaining disturbance). Site A represents a skeletal soil profile, with a loam texture and 60 cm depth; Site B represents a mountain soil profile, with a loam-clay loam texture and 120 cm depth; and, Site C represents a mountain soil profile, with a clay loam texture and 180 cm depth. A fourth site with soil excluded was also included to represent non-water limiting sites

Table 8: Microclimatic modifiers used to create the stand modifying and stand maintaining scenarios from the generated climate scenarios

Treatment	Stand Modifying			Stand Maintaining		
Month	Min Temp. (°C)	Max Temp. (°C)	Precip (%)	Min Temp. (°C)	Max Temp. (°C)	Precip (%)
August	3.5	-2.1	0.88	4.2	-3.5	0.76
September	0.7	-3.1	0.88	1.4	-4.5	0.76
October	0	-4.1	0.88	0.2	-6.9	0.76
November	0	-4.9	0.88	0	-8	0.76
December	1.4	-3.1	0.88	-0.7	-8.3	0.76
January	0	-2.8	0.88	-1.4	-7.6	0.76
February	-0.3	-2.7	0.88	-1	-6.9	0.76
March	0.7	-2.7	0.88	0	-6.2	0.76
April	2.1	-2	0.88	2.1	-4.8	0.76
May	3.5	-2.8	0.88	3.5	-3.5	0.76
June	-0.7	-1.4	0.88	4.2	-1.4	0.76
July	4.1	0	0.88	4.8	0	0.76

3.6 Determining the Probability of Presence/Absence

Presence/absence was determined at each elevation plot by amalgamating species response on the four evaluated site types. This was done to represent species response across the range of edaphic sites that occur at finer topographic scales. The probability of presence/absence was determined based on the average probability of a species meeting all phenological and biophysical criteria in all scenarios. If a species did not meet growing degree, minimum temperature, chilling, frost day, germination and/or drought parameters in any scenario, the species was determined to be absent from the site. If a species met all of these requirements in at least one scenario then the presence probability was modified by the probability of frost damage from growing season frosts. For the soil excluded site, the drought parameter was excluded and the same procedure followed. A species that met all criteria received a presence score of one and the climate conditions were assumed to be in the optimal range of the species regeneration niche. A score of zero meant that species never achieved a combination of required parameters, and that climate conditions were outside a species' regeneration niche. Probabilities between one and zero were a result of parameters being met in a proportion of the scenarios and/or the species being subject to frost damage.

3.7 Replication and Statistical Analysis Methods

Replication was achieved through the use of multiple scenario analysis. Statistical analyses of modelling results were used to determine if differences exist between the response of measured variables under current scenarios and future climate change scenarios. Two-sample Student's t-test and Analysis of Variance (ANOVA) were performed to test for differences in variable responses (Sokal and Rohlf 1995; Tabachnick and Fidell 2001). ANOVA was used to test if the mean niche breadth and mean range size of each species across the elevational gradient of the Central Highlands in each climate period were different. Where the final F-test in the ANOVA detected a significant difference in means, the Tukey test was used to identify which climate periods were significantly different. The Tukey test was used because each sample had an equal number of observations (Fowler and Cohen 1990). The Student's t-test was used to test if a difference existed in the mean number of frost days, germination days, fungus days and annual soil moisture between the current climates and predicted future climates. To provide support to the t-test, confidence intervals (CI) were calculated to provide both an estimate of the effect and a measure of uncertainty. It is important to note that it is the estimates of the magnitudes of effect with associated errors that are important in these types of studies rather than the statistical significance of a test (Johnson 1999).

3.8 Vulnerability to Climate Change

Species' vulnerability to climate change was categorised based on The World Conservation Union's (IUCN) risk categories and criteria. The IUCN (2001) classifies species at extreme risk if they have or are predicted to suffer a reduction in range ≥ 90 % within 100 years, very high risk if range reduction ≥ 70 % occurs, high risk if a range reduction of ≥ 30%, and moderate if a reduction occurs below the latter threshold. Species that exhibit no change are classified as being of "least concern", the category used to reflect low risk in this study. Therefore, we used the following classification in our analysis:

0. No change or increase: no vulnerability
1. < 30 % change: low vulnerability
2. < 50 % change: medium vulnerability
3. < 70 % change: high vulnerability
4. <= 90% change: very high vulnerability
5. > 90% change: extreme vulnerability

3.9 Aspatial to Spatial

One of the limitations of the TACA model is the aspatial design. To help overcome this limitation, the results for each species were integrated in ArcGIS 9.2 (ESRI 2006) using a digital elevation model for the Central Highlands. Using the results from TACA, the current and future regeneration potential for each species at each 100 m elevation band were predicted. This step enabled changes in regeneration niche and range to be mapped simultaneously in order to consider species vulnerability in a more holistic context. The classification of regeneration potential was based on the probability of regeneration under the multiple climate scenarios used in our study, as follows:

1. Nil: regeneration probability = 0
2. Very Low: regeneration probability < 10 %
3. Low: regeneration probability 10 % to < 20 %
4. Medium: regeneration probability 20 % to < 40 %
5. High: regeneration probability 40 % to < 50 %
6. Very High: regeneration probability >= 50 %

4. Results

4.1 Species Response to Climate Change: Regeneration Niche

In all treatments, no significant difference existed in the mean regeneration potential for all species in the 2025 climate period on both sheltered and exposed sites compared to the current time period. By 2055 a significant difference was identified for the majority of species across all treatments. In the stand replacing treatment, 20 of 22 species on north aspects and 18 of 22 on south aspects experienced a significant change in the breadth of their regeneration niche. By 2085, 21 of 22 species on north aspects and 20 of 22 on south aspects experienced a significant change in niche size. In the stand modifying treatment, 14 of 22 species on north aspects and 11 of 22 on south aspects experienced a significant change in the breadth of their regeneration niche. By 2085, 16 of the 22 species on north aspects and 21 of 22 on south aspects experienced a significant change in niche size. In the stand maintaining treatment, 20 of the 22 species on north aspects and 18 of 22 on south aspects experienced a significant change in the breadth of their regeneration niche. By 2085, 21 of 22 on north aspects and 20 of 22 on south aspects experienced a significant change in niche size. All significant changes were detected at $P < 0.05$ using the Tukey test when a significant difference was indicated by the ANOVA ($P < 0.05$). Table 9 summarises the species that did not suffer a significant change in their niche size.

Table 9: Species that did not exhibit a significant change in regeneration niche size between current modelled and 2055-2085 modelled responses

Stand Replacing Treatment			
2055		2085	
North Aspect	*South Aspect*	*North Aspect*	*South Aspect*
Black Wattle[+] Red Stringybark	Montane Wattle Black Wattle Mountain Grey Gum Red Stringybark[+]	Black Wattle[+]	Black Wattle Red Stringybark[+]

Stand Modifying Treatment			
2055		2085	
North Aspect	*South Aspect*	*North Aspect*	*South Aspect*
Montane Wattle[+] Black Wattle Blackwood Wattle Mountain Grey Gum[+] Broad-leaved Peppermint Victorian Blue Gum Red Stringybark[+] Narrow-leaved Peppermint	Montane Wattle Black Wattle Blackwood Wattle[+] Mountain Grey Gum Broad-leaved Peppermint Victorian Blue Gum[+] Red Stringybark Messmate Stringybark Narrow-leaved Peppermint[+] Silvertop Ash Manna Gum*	Montane Wattle[+] Black Wattle Blackwood Wattle Mountain Grey Gum[+] Victorian Blue Gum Red Stringybark[+]	Montane Wattle Black Wattle Blackwood Wattle[+] Mountain Grey Gum Broad-leaved Peppermint Victorian Blue Gum[+] Red Stringybark Messmate Stringybark Narrow-leaved Peppermint[+] Silvertop Ash Manna Gum[+]

Stand Maintaining Treatment			
2055		2085	
North Aspect	*South Aspect*	*North Aspect*	*South Aspect*
Black Wattle[+] Red Stringybark	Montane Wattle Black Wattle Mountain Grey Gum Red Stringybark[+]	Black Wattle[+]	Black Wattle Red Stringybark[+]

+ No significant difference in niche size detected by ANOVA between all climate periods

4.2 Species Response to Climate Change: Range Size

In all treatments, no significance difference was found to exist in the potential area available for regeneration (regeneration probability > 0) for all species in the 2025 climate period, on both sheltered and exposed sites, compared to the current time period. By 2055 a significant difference was identified for the majority of species across all treatments. In the stand replacing treatment, 18 of 22 species exhibited a change in area available for regeneration. By 2085, 19 of 22 species experienced a significant change in range size. For the stand modifying treatment, 10 of 22 species exhibited a change in area available for regeneration by 2055. By 2085, 14 of 22 experienced a significant change in range size. In the stand maintaining treatment, 18 of 22 species exhibited a change in area available for regeneration by 2055. By 2085, 19 of 22 experienced a significant change in range size. All significant changes were detected at $P < 0.05$ using the Tukey test when a significant difference was indicated by the ANOVA ($P < 0.05$). Table 10 summarises the species that did not suffer a significant change in niche size.

Table 10: Species that did not exhibit a significant change in range size between current modelled and 2055-2085 modelled responses

Stand Replacing Treatment	
2055	2085
Montane Wattle Black Wattle[+] Blackwood Wattle[+] Red Stringybark[+]	Black Wattle[+] Blackwood Wattle[+] Red Stringybark[+]

Stand Modifying Treatment	
2055	2085
Montane Wattle[+] Black Wattle[+] Blackwood Wattle[+] Mountain Grey Gum[+] Broad-leaved Peppermint[+] Victorian Blue Gum[+] Red Stringybark[+] Messmate Stringybark Narrow-leaved Peppermint[+] Silvertop Ash Manna Gum Myrtle Beech	Montane Wattle[+] Black Wattle* Blackwood Wattle[+] Mountain Grey Gum[+] Broad-leaved Peppermint[+] Victorian Blue Gum[+] Red Stringybark[+] Narrow-leaved Peppermint[+]

Stand Maintaining Treatment	
2055	2085
Montane Wattle Black Wattle[+] Blackwood Wattle[+] Red Stringybark[+]	Black Wattle[+] Blackwood Wattle[+] Red Stringybark[+]

+: No significant difference in range size detected by ANOVA between all climate periods

4.3 Species Response to Climate Change: Vulnerability Classification

The statistical analyses of potential changes in the size of species regeneration niches and potential range sizes suggests that large-scale changes may occur in the Central Highlands as a result of climate change. Even for species that exhibited a non significant change the future potential niche and range do not necessarily reflect the current. To classify the vulnerability of each species, the proportion of change in the size of the regeneration niche and range size were calculated and the species classified using the vulnerability classification scheme outlined in the methods. Figures 11 to 16 summarise the classification of species vulnerability to predicted climate change.

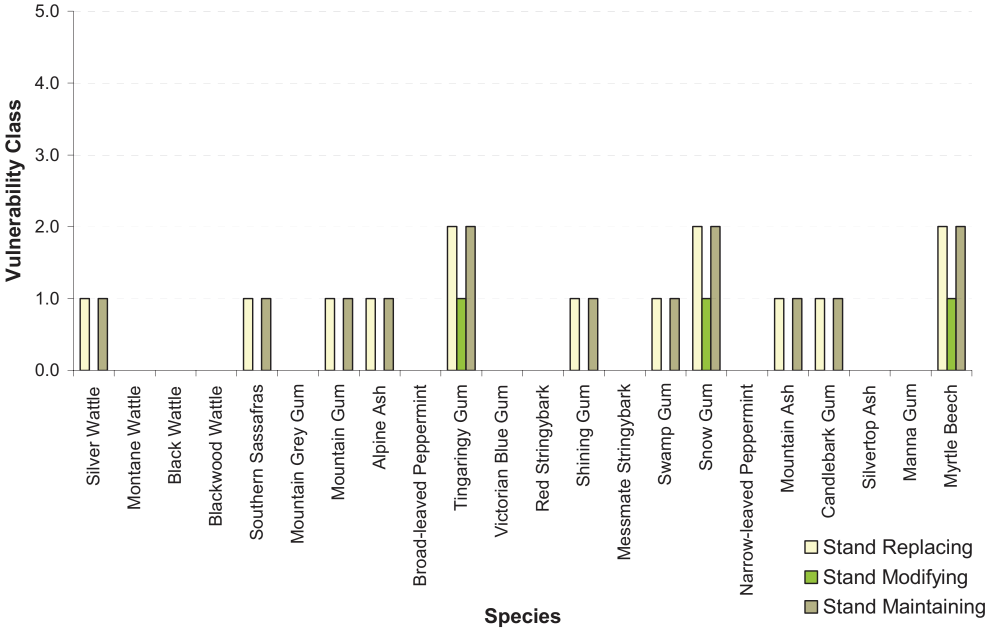


Fig. 11: Species vulnerability rating to regeneration niche contraction by 2025 by treatment type. 0 = Nil: no or positive change; 1 = Low: < 30 % contraction; 2 = Medium: < 50 % contraction; 3 = High: < 70 % contraction; 4 = Very High: < = 90% contraction; 5 = Extreme: > 90 % contraction

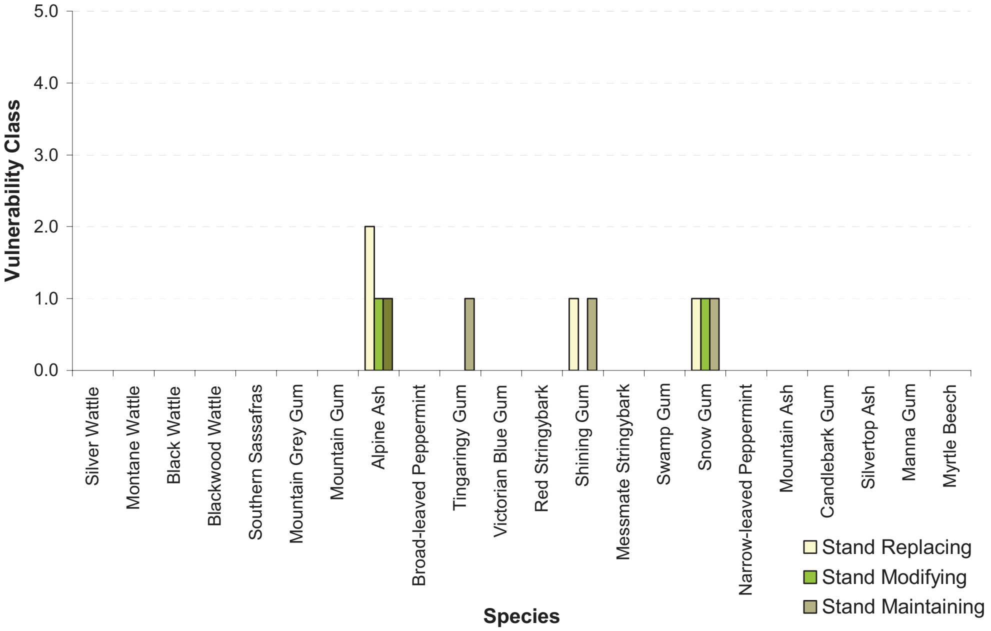


Fig. 12: Species vulnerability rating to range contraction by 2025 by treatment type. 0 = Nil: no or positive change; 1 = Low: < 30 % contraction; 2 = Medium: < 50 % contraction; 3 = High: < 70 % contraction; 4 = Very High: < = 90% contraction; 5 = Extreme: > 90 % contraction

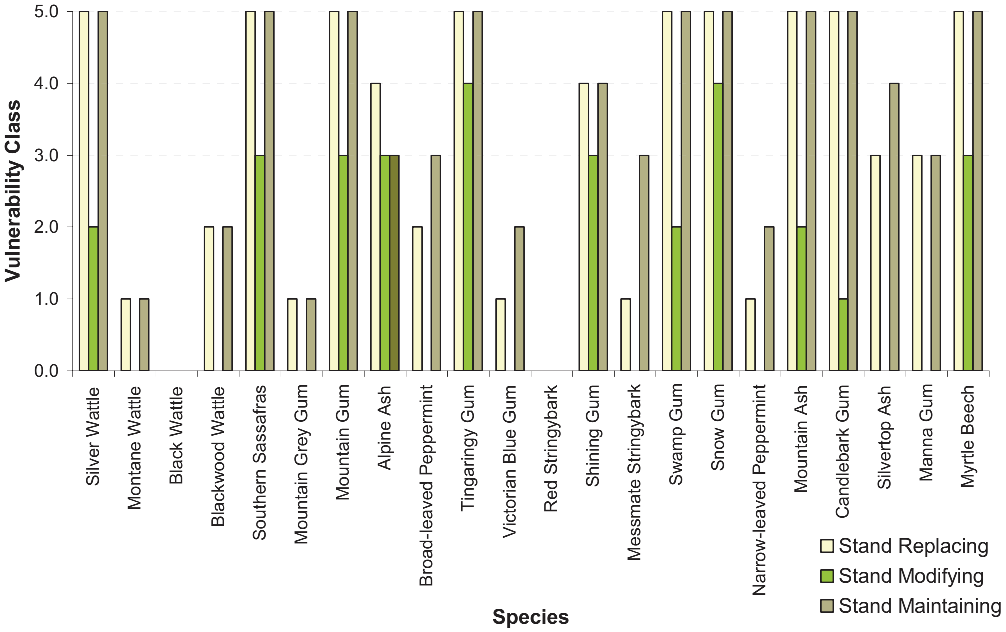


Fig. 13: Species vulnerability rating to regeneration niche contraction by 2055 by treatment type. 0 = Nil: no or positive change; 1 = Low: < 30 % contraction; 2 = Medium: < 50 % contraction; 3 = High: < 70 % contraction; 4 = Very High: < = 90% contraction; 5 = Extreme: > 90 % contraction

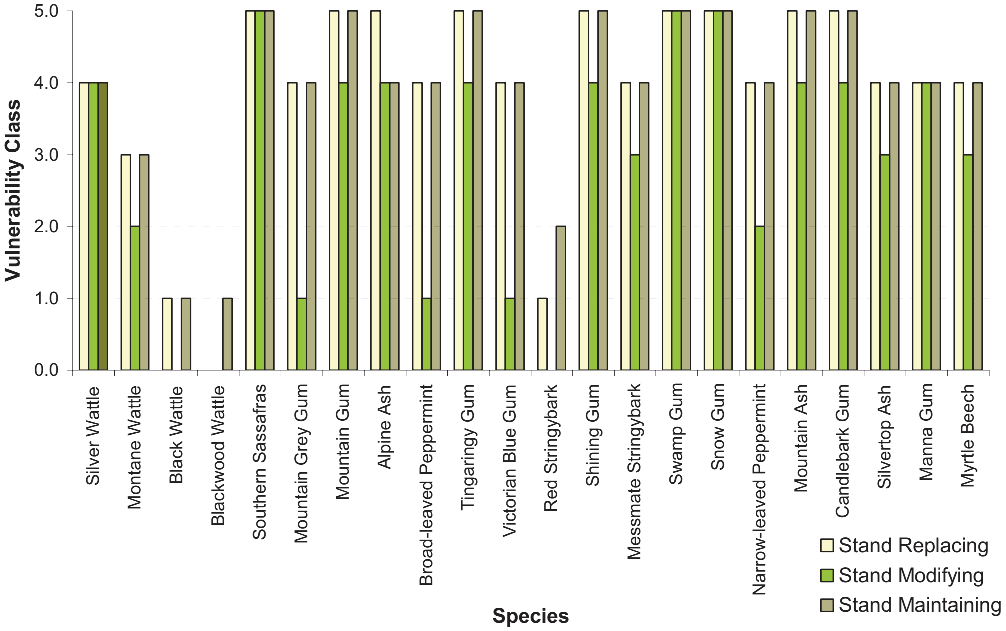


Fig. 14: Species vulnerability rating to range contraction by 2055 by treatment type. 0 = Nil: no or positive change; 1 = Low: < 30 % contraction; 2 = Medium: < 50 % contraction; 3 = High: < 70 % contraction; 4 = Very High: < = 90% contraction; 5 = Extreme: > 90 % contraction

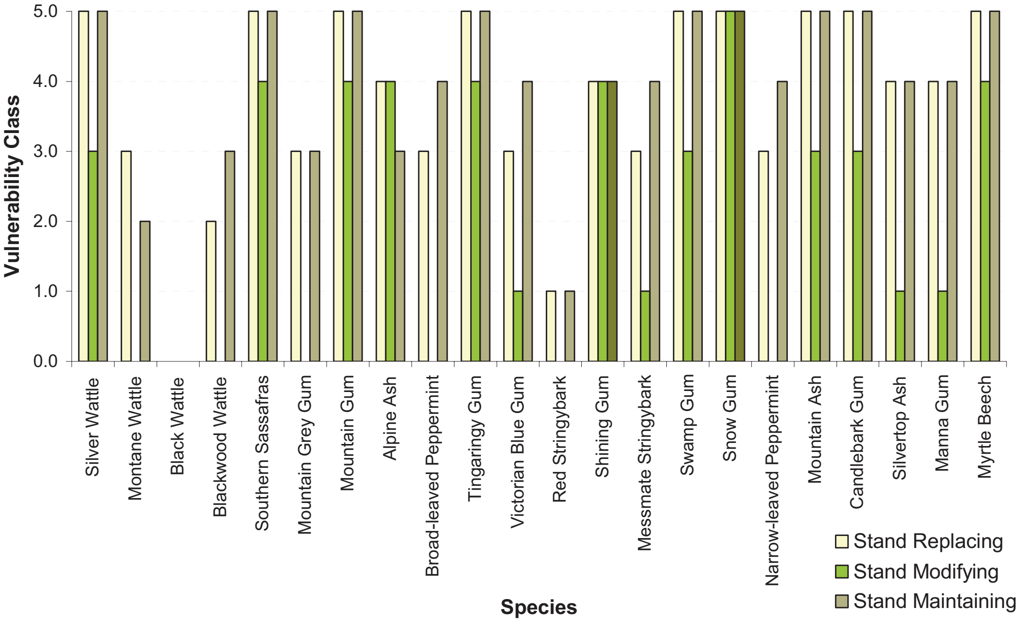


Fig. 15: Species vulnerability rating to regeneration niche contraction by 2085 by treatment type. 0 = Nil: no or positive change; 1 = Low: < 30 % contraction; 2 = Medium: < 50 % contraction; 3 = High: < 70 % contraction; 4 = Very High: < = 90% contraction; 5 = Extreme: > 90 % contraction

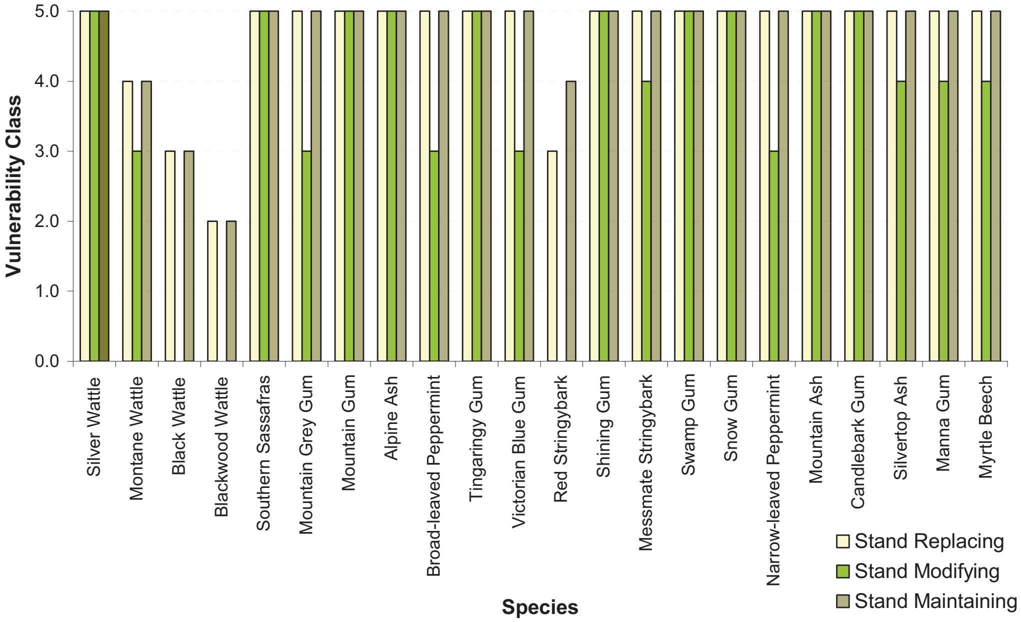


Fig. 16: Species vulnerability rating to range contraction by 2085 by treatment type. 0 = Nil: no or positive change; 1 = Low: < 30 % contraction; 2 = Medium: < 50 % contraction; 3 = High: < 70 % contraction; 4 = Very High: < = 90% contraction; 5 = Extreme: > 90 % contraction

The classification of species vulnerability indicates that changes in the regeneration niche and range size of species may occur in 2025, whereas the statistical analysis reported a non-significant response. In 2025, the potential change in regeneration niche size is more vulnerable than a contraction in range size with 11 of 22 species exhibiting a low to medium vulnerability versus only four species when range was considered. The stand modifying treatment had a mediating influence on vulnerability with only three (niche) and two (range) species being classified with low vulnerability. The results highlight that in 2025 species are able to regenerate across the same amount of area as the current but that the probability of successful regeneration is declining for some species. By 2055 and 2085, the inverse relationship occurs between niche vulnerability and range vulnerability. By 2055, the vulnerability of species suffering a contraction in range is greater than the number suffering a contraction in regeneration niche size. The classification results confirm the statistical analysis results which suggested a significant response by 2055 for the majority of the species. By 2055, 20 of the 22 species are classified as vulnerable in their regeneration niche and all species are vulnerable to range contraction. By 2085, only one species (black wattle) is not considered vulnerable in its regeneration niche but in range contraction all species are considered to have medium vulnerability or greater. The classification of species vulnerability to both contractions in the regeneration niche and range size illustrates that by 2055, species vulnerability increases significantly from 2025. These results suggest that species may suffer a significant decline in regeneration potential along with a greater contraction in future range size. This would further limit the area available for regeneration, constraining a species regeneration niche to more climatically restrictive areas within the Central Highlands. The results also highlight that the stand modifying treatment had a mediating influence on species vulnerability. Figure 17 summarises the number of species in each vulnerability class by treatment type in 2085. Under the stand modifying treatment resulted in 10 species were classified as extremely vulnerable to range contraction versus 18 in the other two treatments. Tables I-1 to I-3 in Appendix I summarise the change in fundamental range size for each species between the current and 2085.

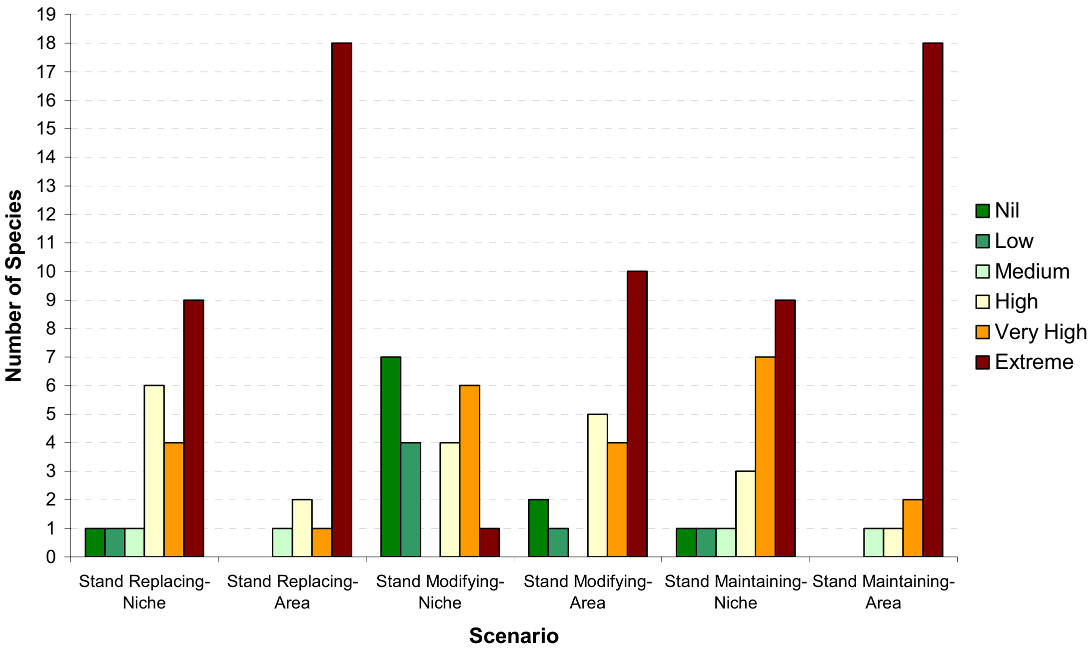


Fig. 17: Number of species in reach vulnerability class in 2085 by treatment type. "Niche" refers to vulnerability of the regeneration niche contracting and "Area" refers to the vulnerability of species' ranges contracting. Nil: no or positive change; Low: < 30 % contraction; Medium: < 50 % contraction; High: < 70 % contraction; Very High: < = 90% contraction; Extreme: > 90 % contraction

4.4 Species Response to Climate Change: Species Diversity

Figures 18 to 23 summarise the number of species that were modelled as being able to regenerate in at least one climate scenario at each elevation plot. The results show that under the current climate conditions the number of species that could regenerate at mid and lower elevations is greater than at higher elevations on both exposed and sheltered sites. In the 2025 period an increase in the number of species able to regenerate at higher elevations is observed, with little or no change at lower elevations across all treatments. By 2055, a marked increase in the number of species able to regenerate at higher elevations occurs along with a decrease in species at lower elevations. By 2085, the trend is exacerbated with more species able to regenerate at higher elevations than lower elevations. Exposed sites showed a decline in the number of species at all elevations in response to climate change, while sheltered sites maintained all species above 1600 m in the stand replacing treatment, and above 1400 m in the stand modifying treatment. The results illustrate the contraction of species to higher elevations under predicted climate change.

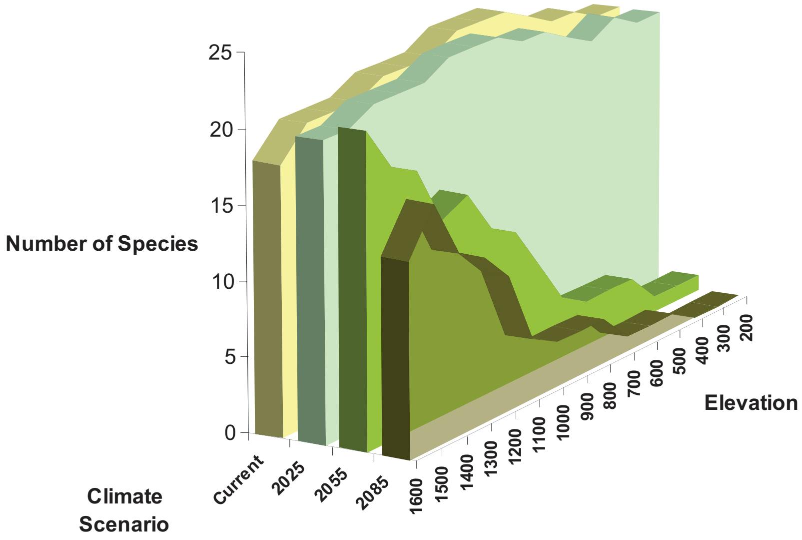


Fig. 18: Change in potential species diversity on exposed sites subjected to a <u>stand replacing treatment</u> across and elevation gradient under predicted climate change in the Central Highlands

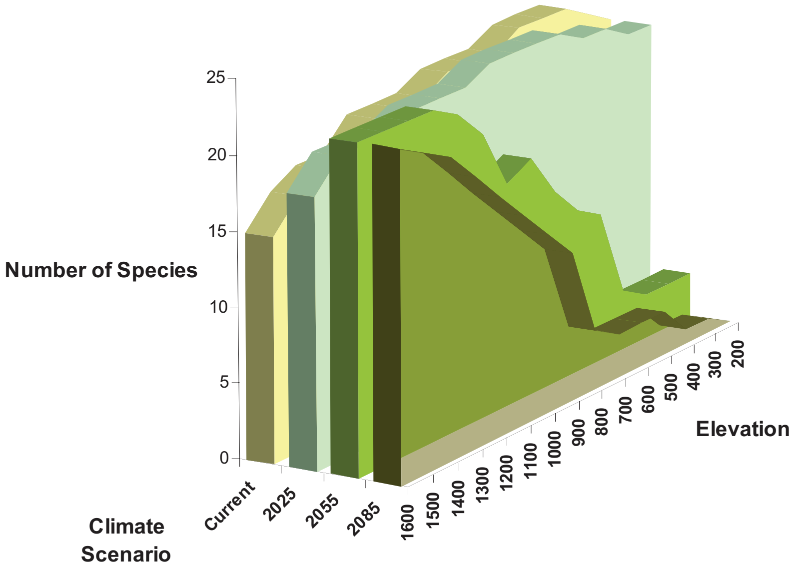


Fig. 19: Change in potential species diversity on sheltered sites subjected to a <u>stand replacement treatment</u> across and elevation gradient under predicted climate change in the Central Highlands

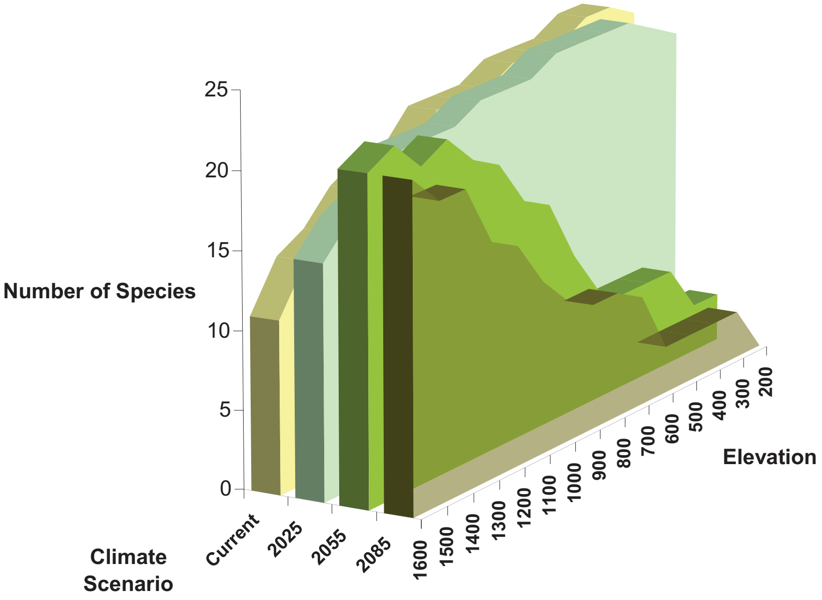


Fig. 20: Change in potential species diversity on exposed sites subjected to a <u>stand modifying treatment</u> across and elevation gradient under predicted climate change in the Central Highlands

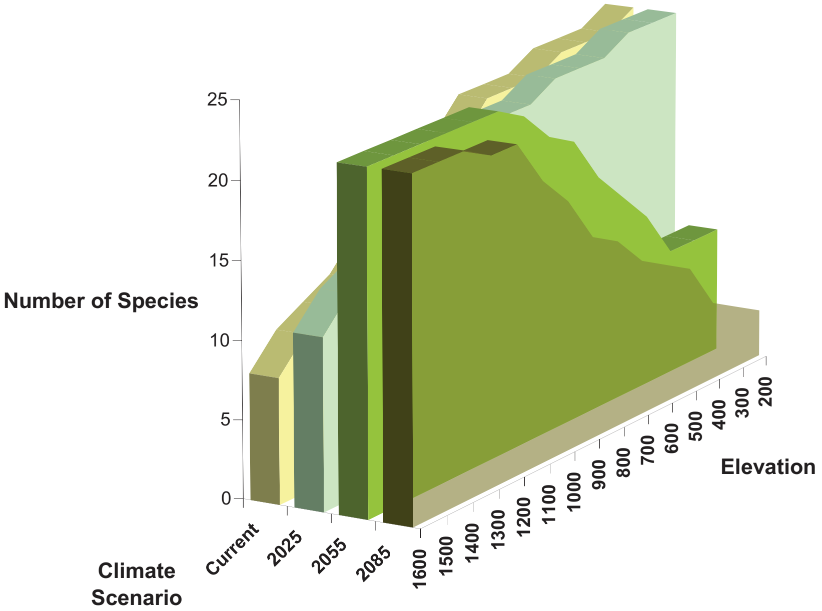


Fig. 21: Change in potential species diversity on sheltered sites subjected to a <u>stand modifying treatment</u> across and elevation gradient under predicted climate change in the Central Highlands

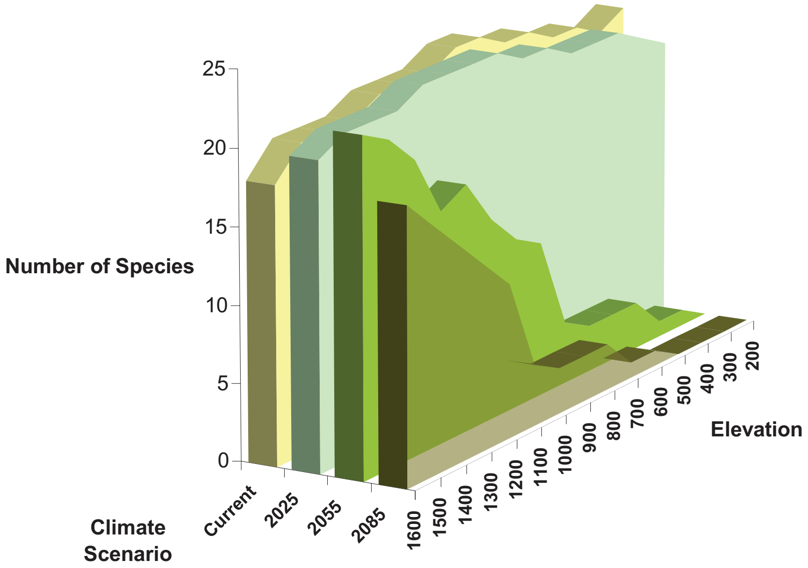


Fig. 22: Change in potential species diversity on exposed sites subjected to a <u>stand maintaining treatment</u> across and elevation gradient under predicted climate change in the Central Highlands

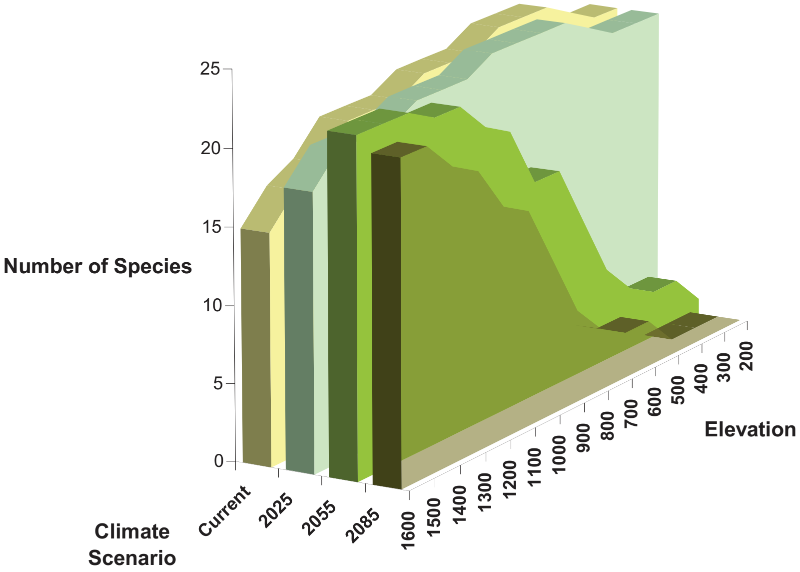
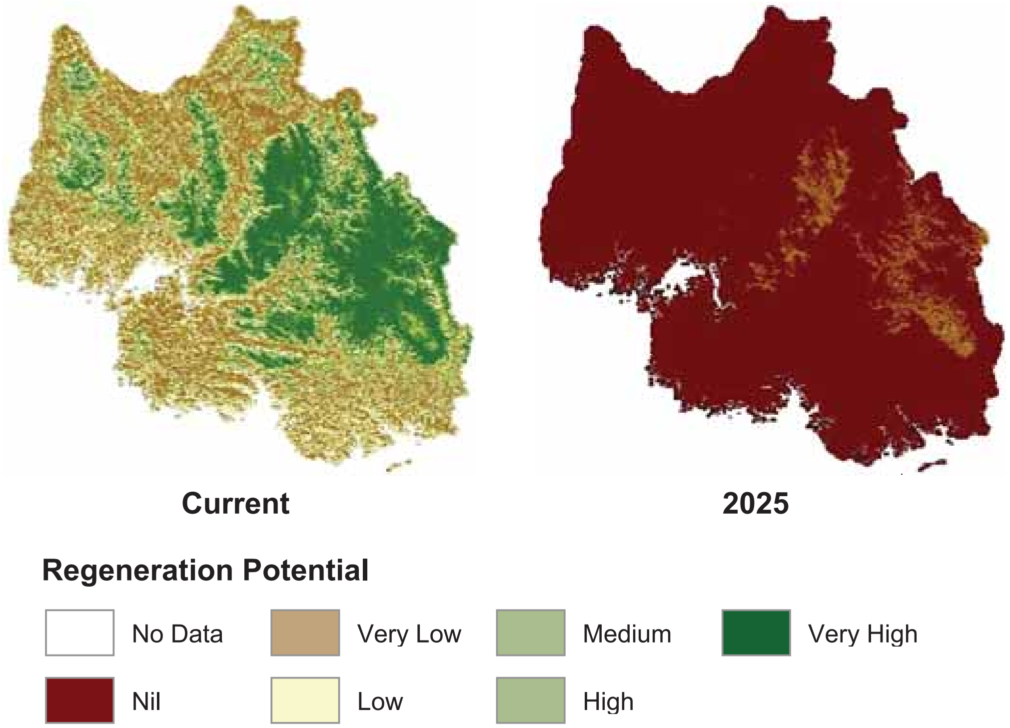


Fig. 23: Change in potential species diversity on sheltered sites subjected to a <u>stand maintaining treatment</u> across and elevation gradient under predicted climate change in the Central Highlands

4.5 Species Vulnerability to Climate Change in the Central Highlands

The preceding results of species response to climate change in their regeneration niche and potential range size is based on two separate aspatial analyses. To get a better understanding of why species were classified as so vulnerable to predicted climate change, we combined both the influence of changes in regeneration niche (regeneration potential) with changes in range size in relation to the geographic context of the Central Highlands. Using the regeneration potential classification scheme, species response to changes in their regeneration niche and range were combined with a digital elevation model of the study area using ArcGIS 9.2 (ESRI 2006). The species-specific maps of response in the stand replacing and stand modifying treatments to climate change were generated to illustrate the combined interaction between changes in regeneration potential and range. The stand maintaining treatment was not illustrated because of the similar response species exhibited between this scenario and the stand replacing scenario. The stand modifying treatment mediated species response and is provide to demonstrate the influence that microclimatic conditions may have on the distribution of species' regeneration niches. In both the stand modifying and stand maintaining treatments the influence of light was not consider only the influence of climate. This limits the assessment because some species require light to regenerate or have their regeneration success dampened. This does not make the assessment unsound but likely represents and underestimation for some species. These treatments illustrate the influence of microclimate only. The stand replacing treatment represents a scenario where clearfell harvesting or large stand replacing fires occur removing the mature forest communities and exposing the landscape to future climatic conditions. Figures 24-27 present the response of the four *Acacia* species, Figures 28 to 29 the two rainforest species and Figures 30 to 45 the 16 *Eucalyptus* species. An important caveat to note, the maps of species response illustrates the vulnerability of species in their regeneration niche; they do not represent the loss of established forest communities or mortality. Also, species response has been spread over each elevation and aspect in a coarse manner meaning that regeneration will not be evenly distributed within each elevation band or aspect represented. Edaphic and topographic mediated effects will lead to disjunctive distributions, especially in areas rated as having low to medium regeneration potentials. The following figures represent the fundamental regeneration niche of species and do take into consideration competition, light and mortality from diseases, insects, herbivory, etc.



Current

2025

Fig 24: Change in the regeneration potential of Silver Wattle following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

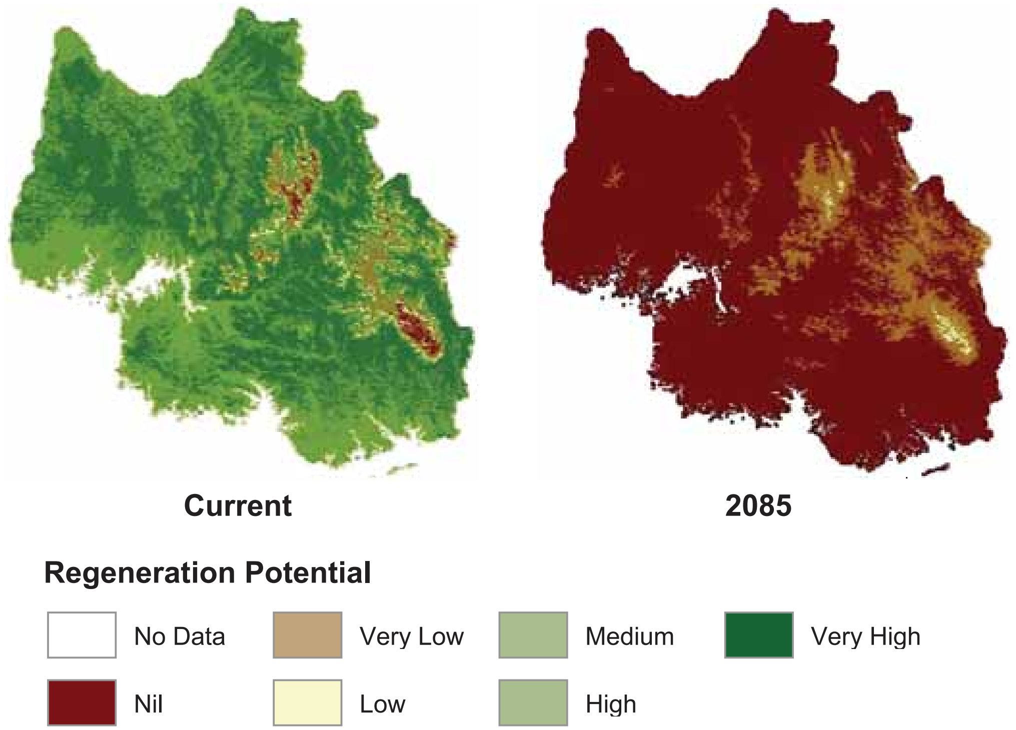
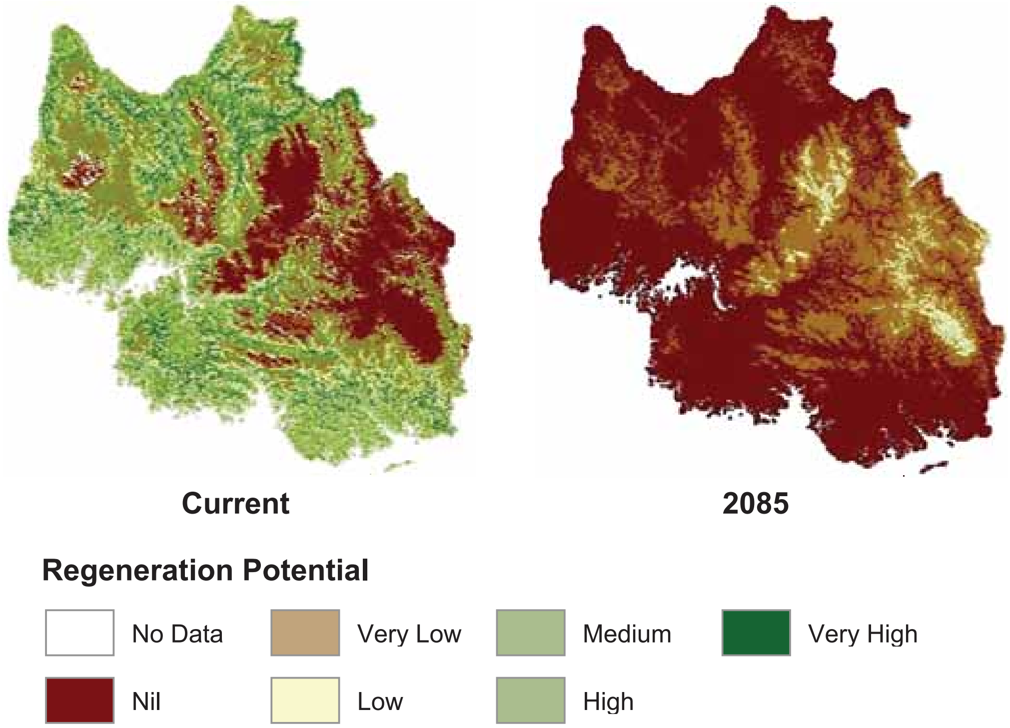


Fig. 25: Change in the regeneration potential of Montane Wattle following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area



Current

2085

Fig. 26: Change in the regeneration potential of Black Wattle following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

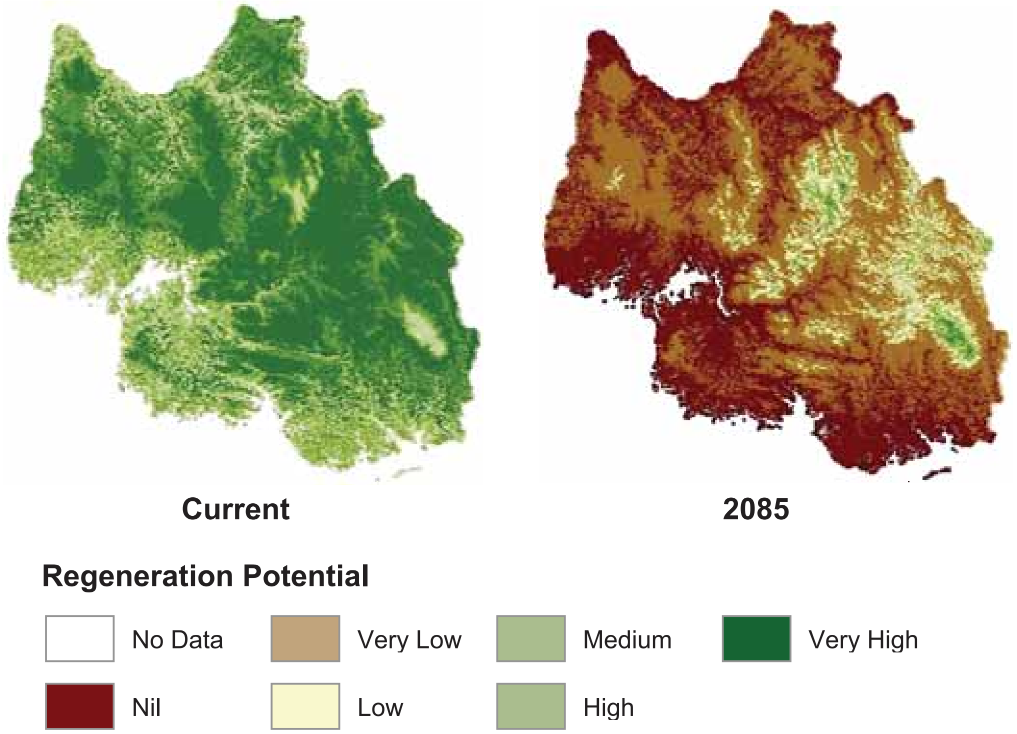


Fig. 27: Change in the regeneration potential of Blackwood Wattle following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

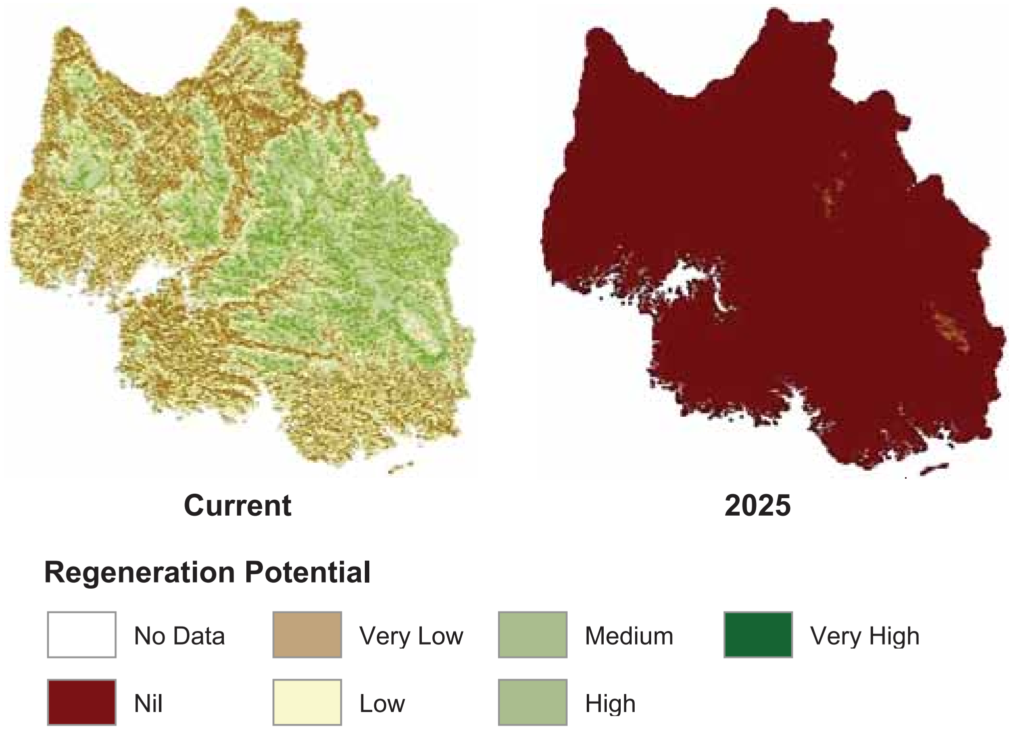


Fig. 28: Change in the regeneration potential of Southern Sassafras following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

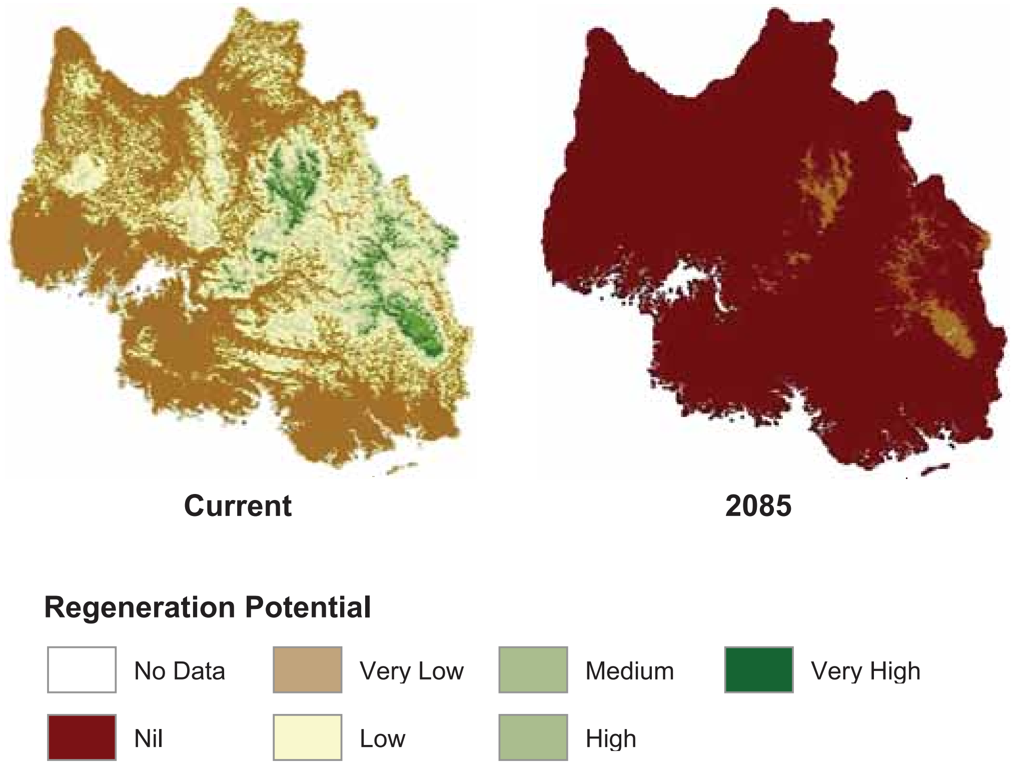


Fig. 29: Change in the regeneration potential of Myrtle Beech following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

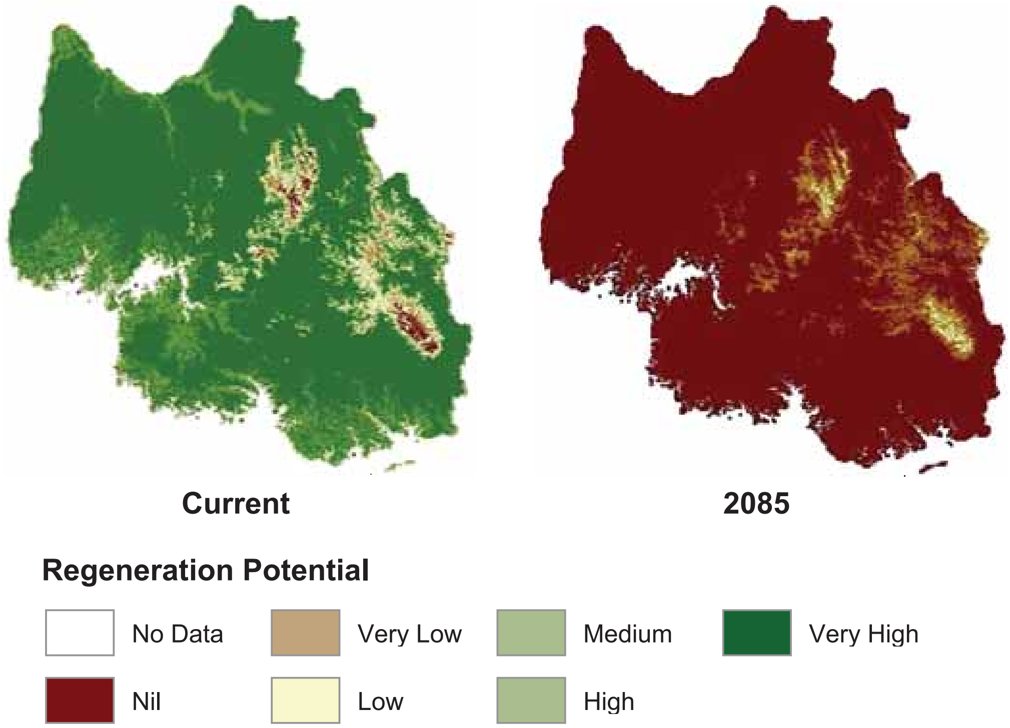
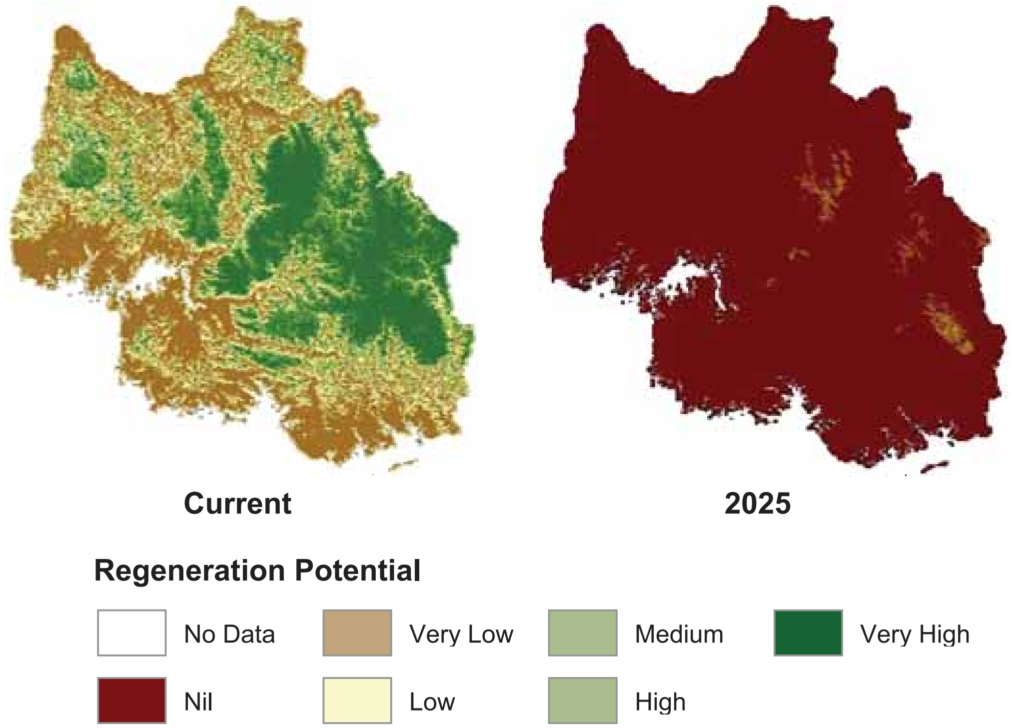


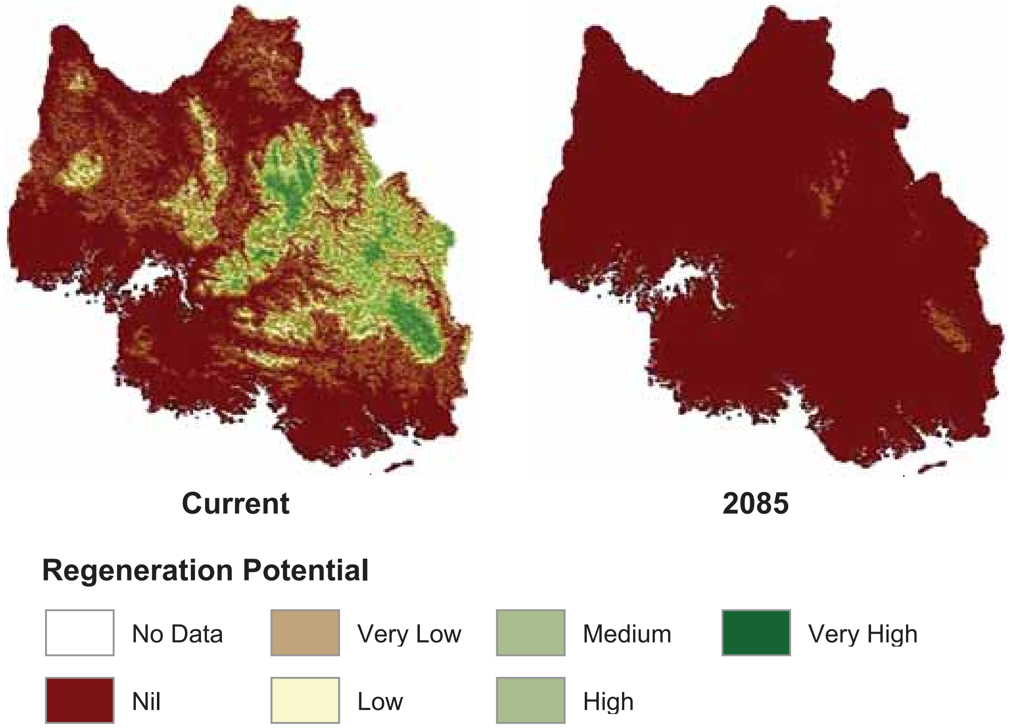
Fig. 30: Change in the regeneration potential of Mountain Grey Gum following stand replacing treatment under predicted climate change in Victoria’s Central Highlands Forest Management Area



Current

2025

Fig. 31: Change in the regeneration potential of Mountain Gum following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area


Current

2085

Fig. 32: Change in the regeneration potential of Alpine Ash following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

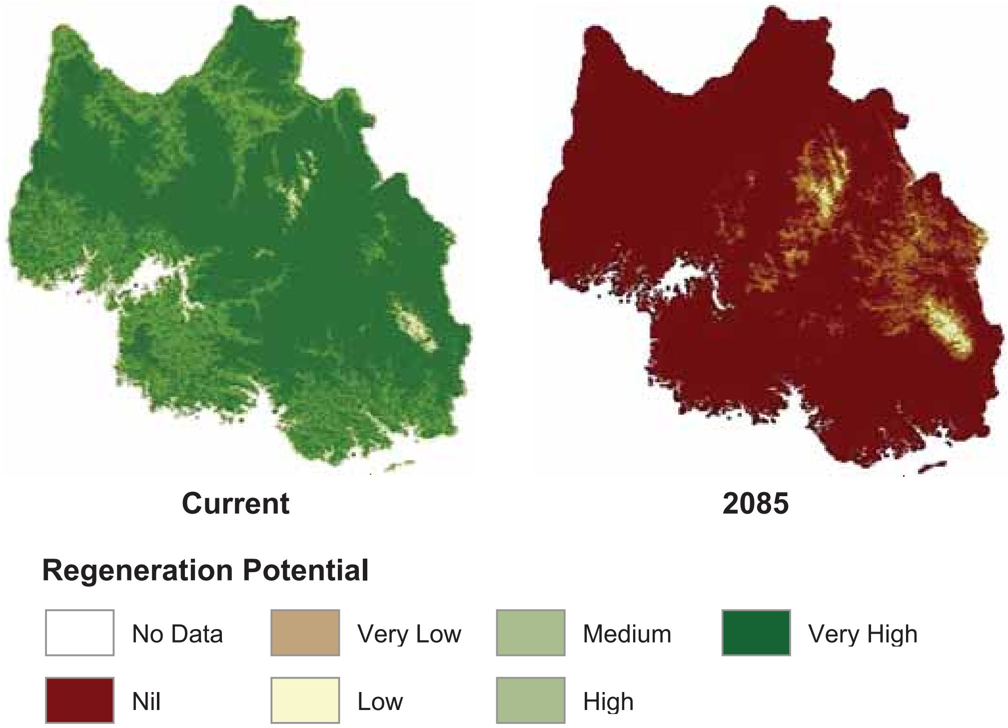


Fig. 33: Change in the regeneration potential of Broad-leaved Peppermint following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

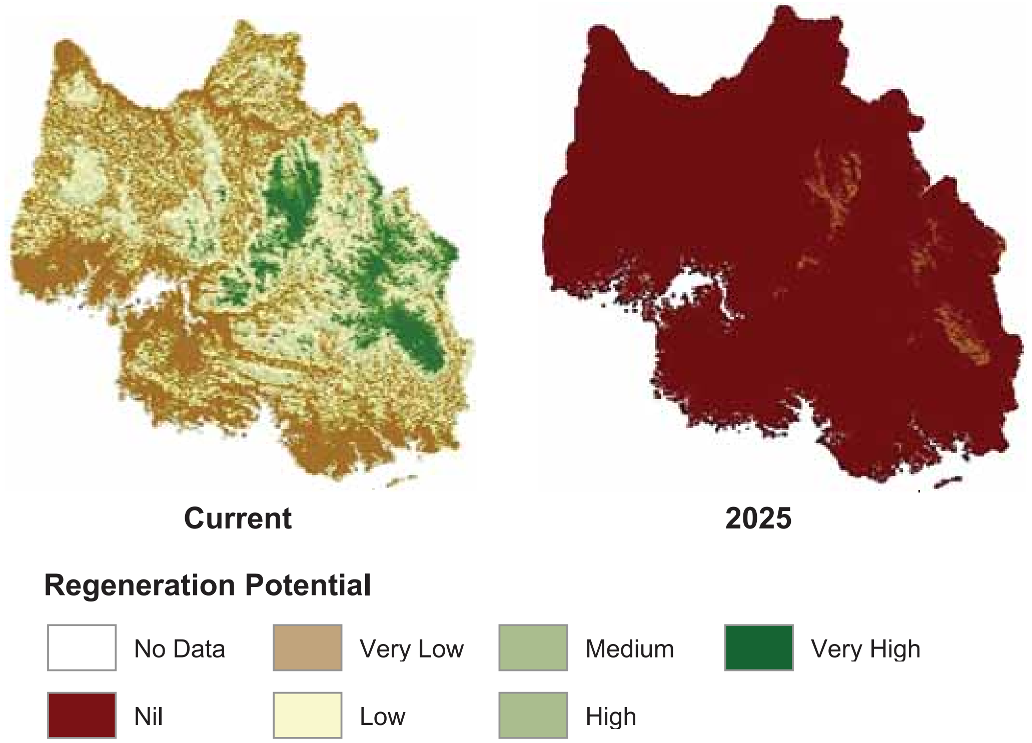


Fig. 34: Change in the regeneration potential of Tingaringy Gum following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

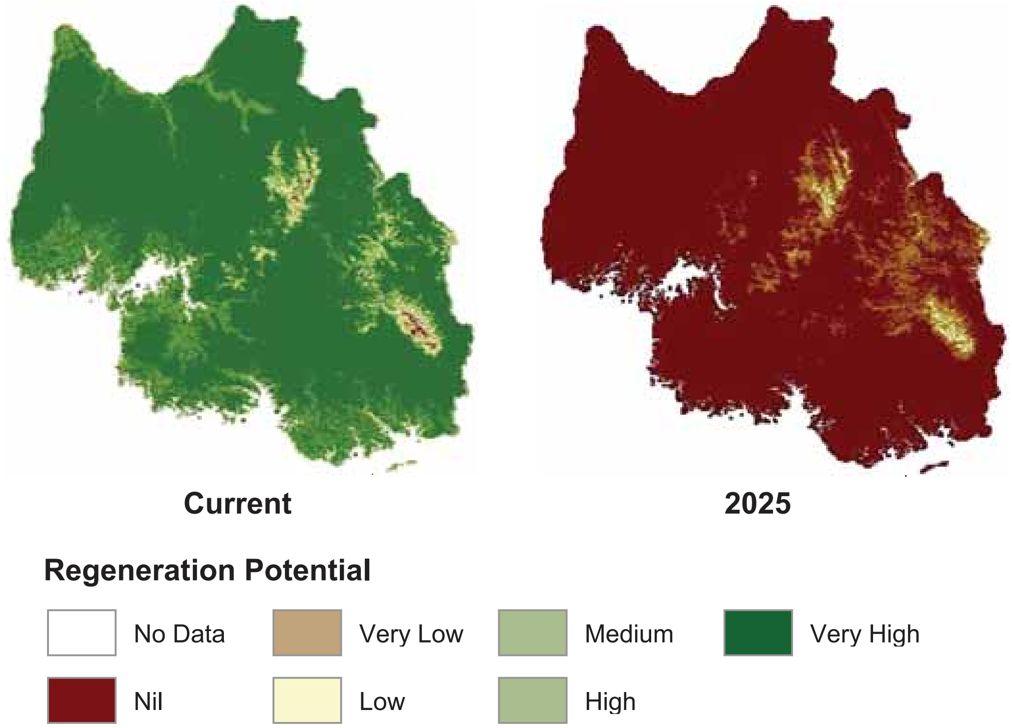


Fig. 35: Change in the regeneration potential of Victorian Blue Gum following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

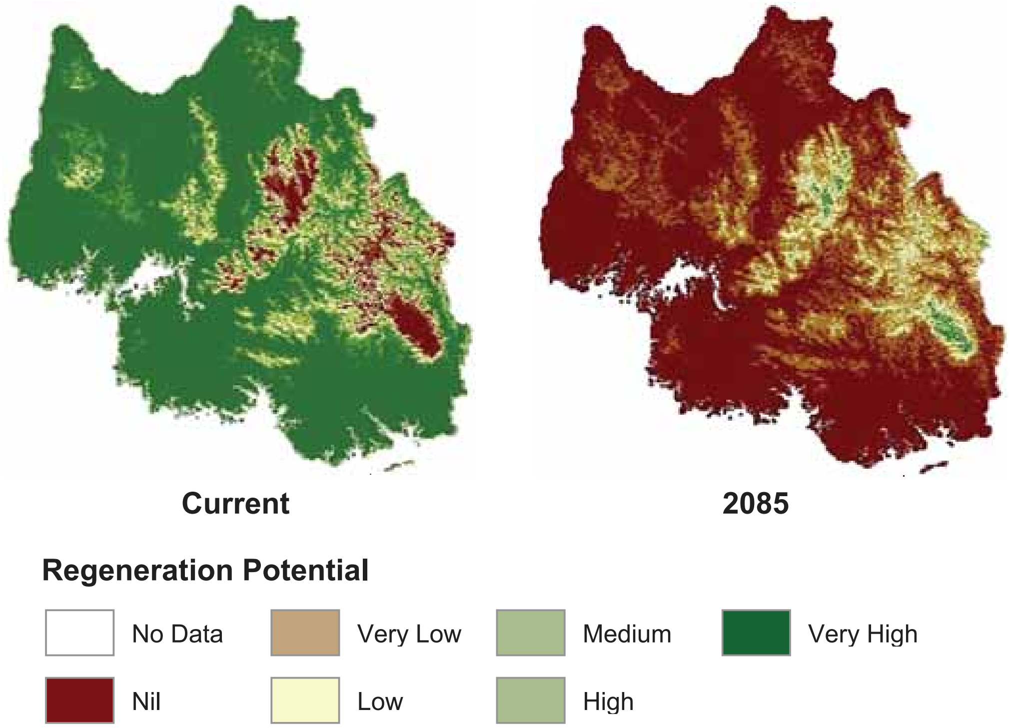


Fig. 36: Change in the regeneration potential of Red Stringybark following stand replacing treatment under predicted climate change in Victoria’s Central Highlands Forest Management Area

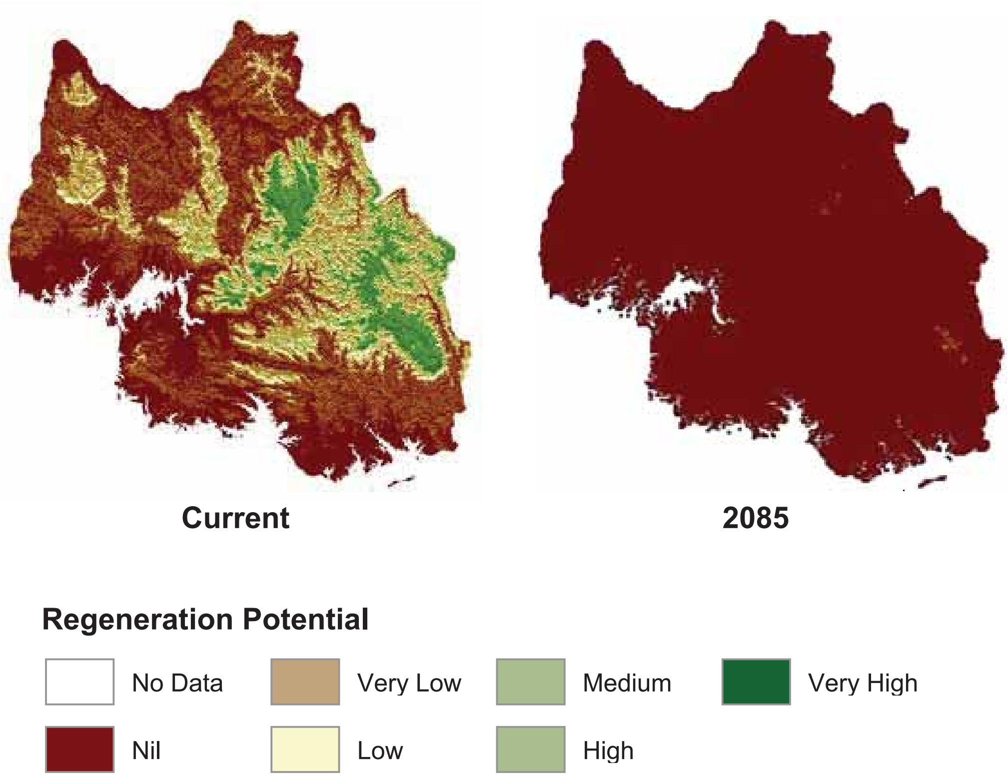


Fig. 37: Change in the regeneration potential of Shining Gum following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

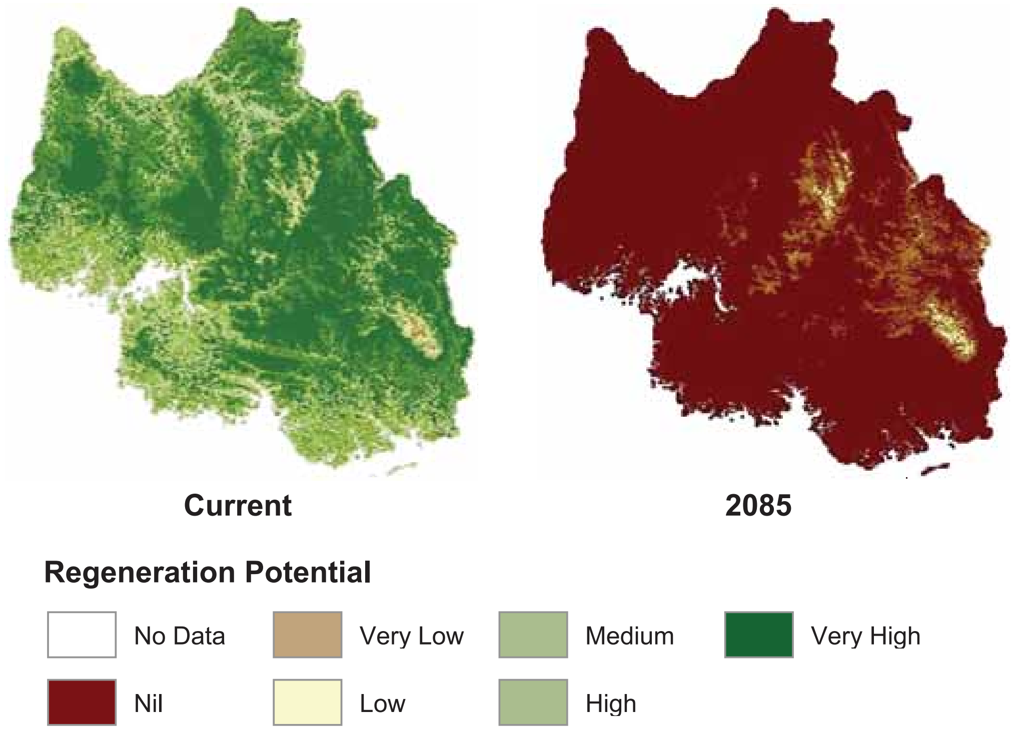


Fig. 38: Change in the regeneration potential of Messmate Stringybark following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

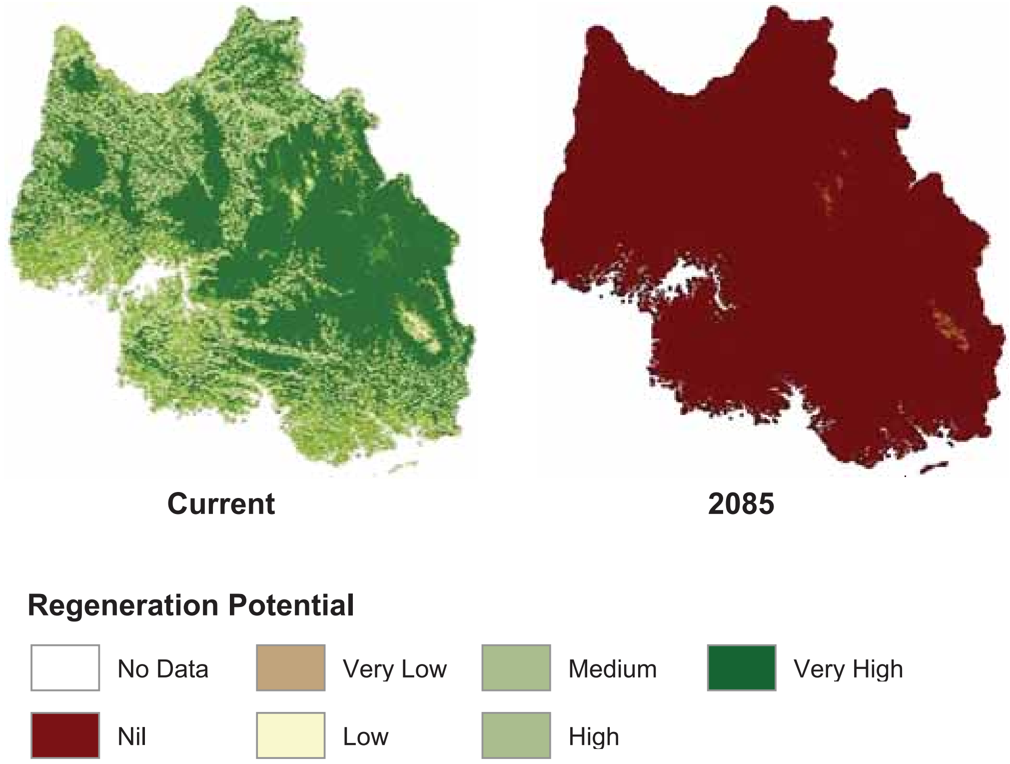


Fig. 39: Change in the regeneration potential of Swamp Gum following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

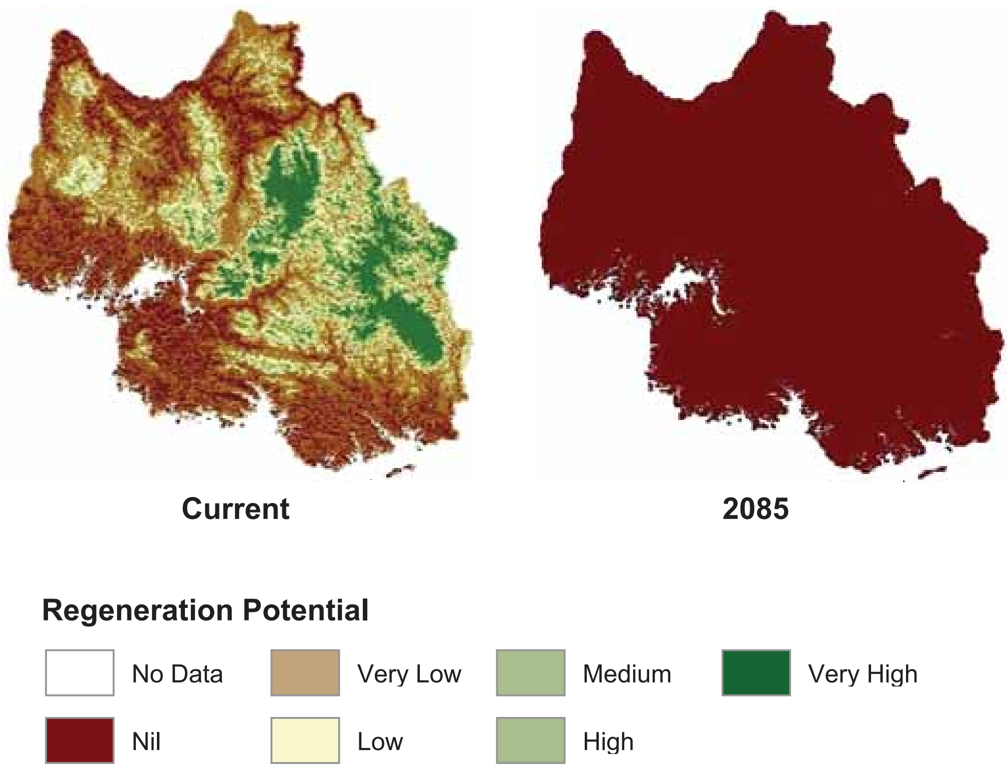
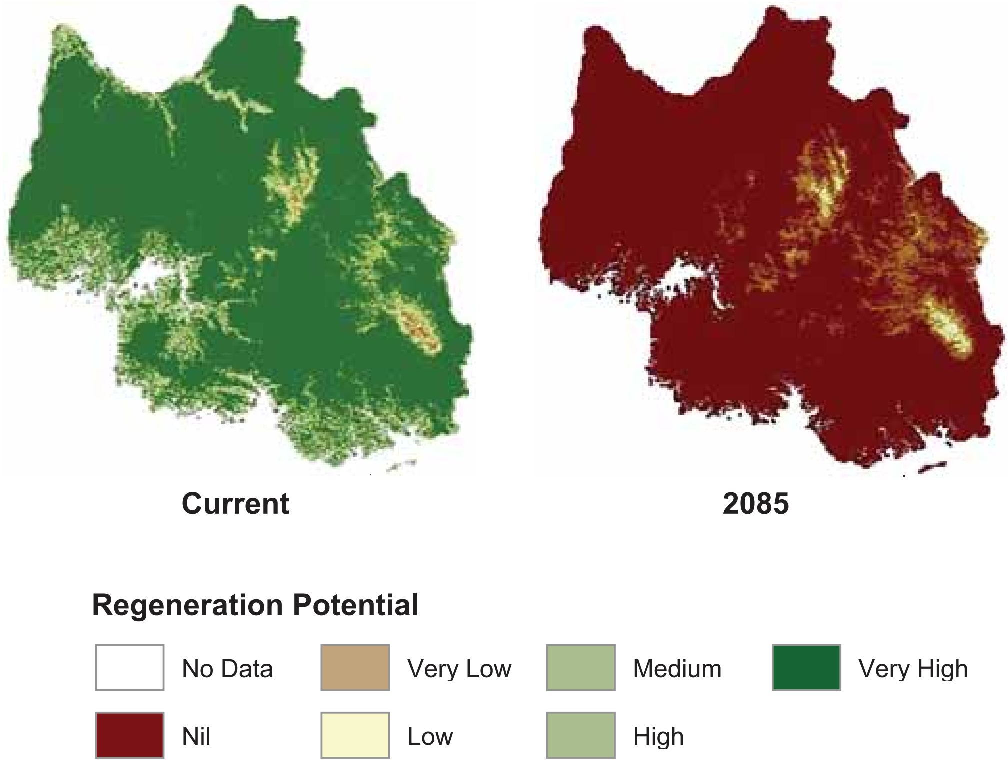


Fig. 40: Change in the regeneration potential of Snow Gum following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area



Current

2085

Fig. 41: Change in the regeneration potential of Narrow-leaved Peppermint following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

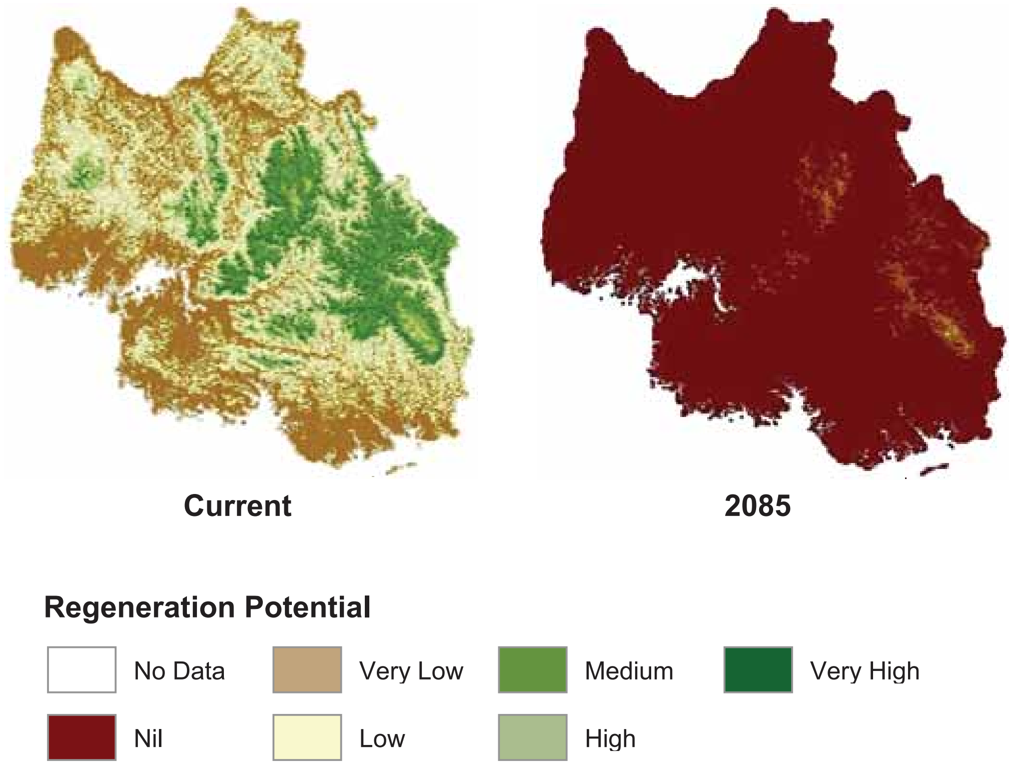


Fig. 42: Change in the regeneration potential of Mountain Ash following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

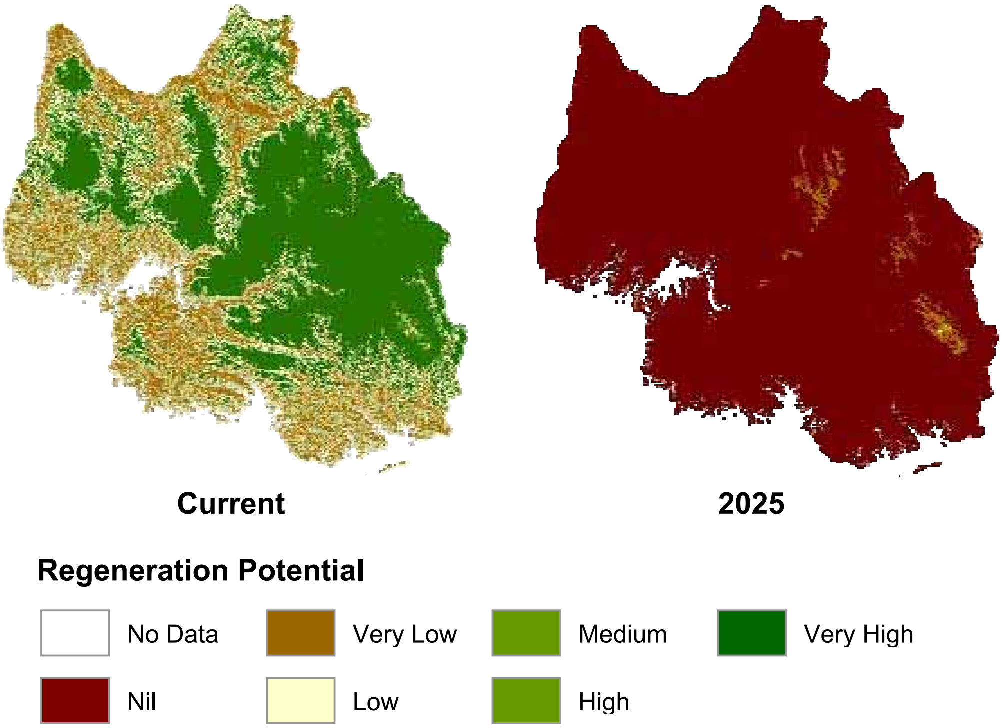


Fig. 43: Change in the regeneration potential of Candlebark Gum following stand replacing treatment under predicted climate change in Victoria’s Central Highlands Forest Management Area

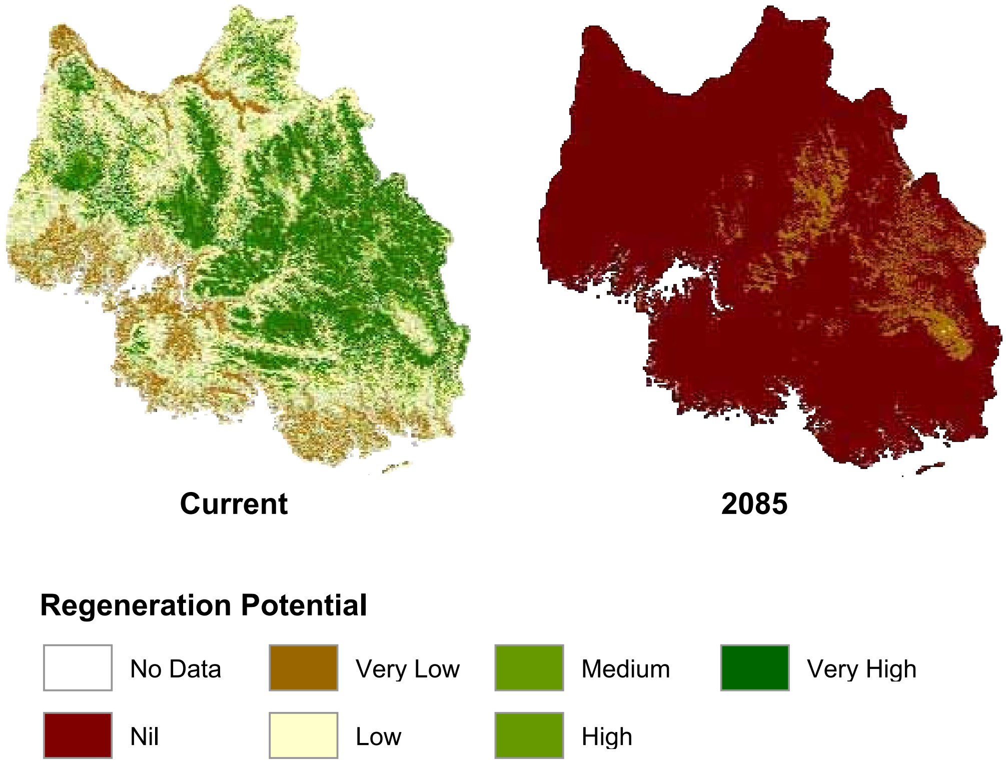
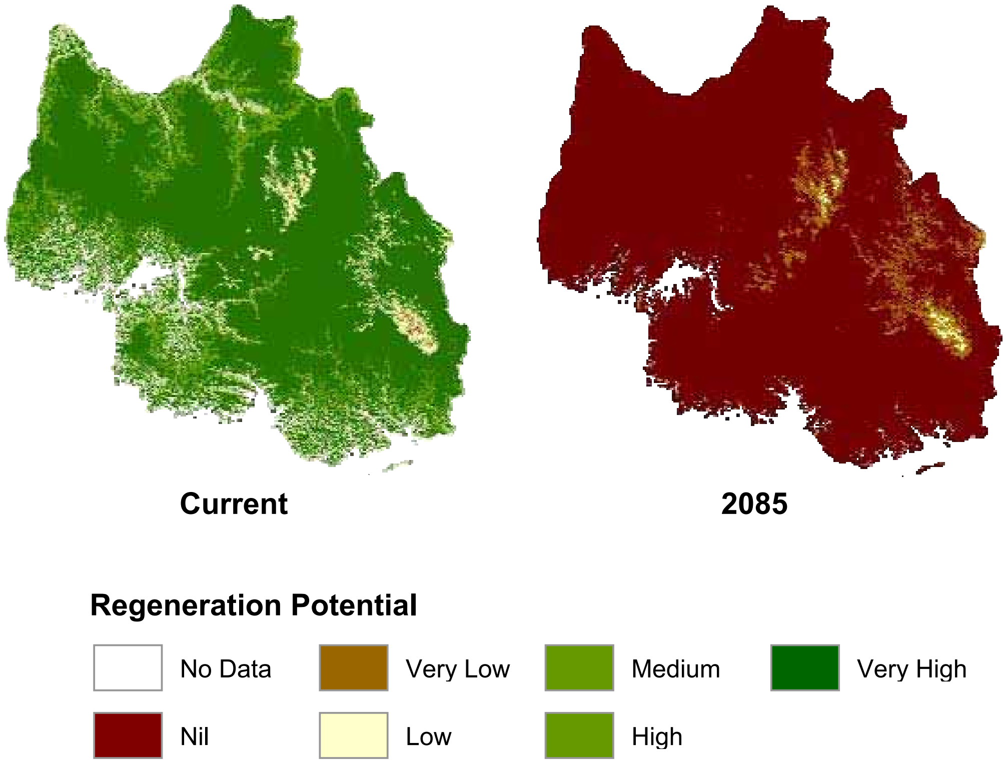


Fig. 44: Change in the regeneration potential of Silvertop Ash following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area



Current

2085

Fig. 45: Change in the regeneration potential of Manna Gum following stand replacing treatment under predicted climate change in Victoria's Central Highlands Forest Management Area

4.6 Driving Variables of Species Response

Figures 24 to 45 illustrate the spatial vulnerability that the species of the Central Highlands have to climatic change following a stand replacing disturbance. To understand which driving variables influence species, the probability of a driving variable being used was tracked in TACA. Figure 46 summarises the influence of each of the driving variables used by TACA. The Growing Degree Day variable had the largest influence on species response in both the current and future scenarios. Drought was the second largest limiting factor. Chilling requirements for species needing stratification of seeds increased in influence with increases in future temperatures, while the number of frost days and occurrence of frost damage declined. It should be noted that the occurrence of frost damage was nearly five times higher than the occurrence of frost days by 2085. This is a result of an increase in "germination days" during late autumn-winter-early spring conditions which still predispose germinants to frost events despite the decline in the number of frost days. Minimum temperature and germination were minor variables overall, but important for a few species. Lignotuber sprouting increased in 2025, but declined by 2055 and 2085 as the growing degree day threshold was commonly exceeded. Species-specific variable sensitivity analyses are presented in Appendix II.

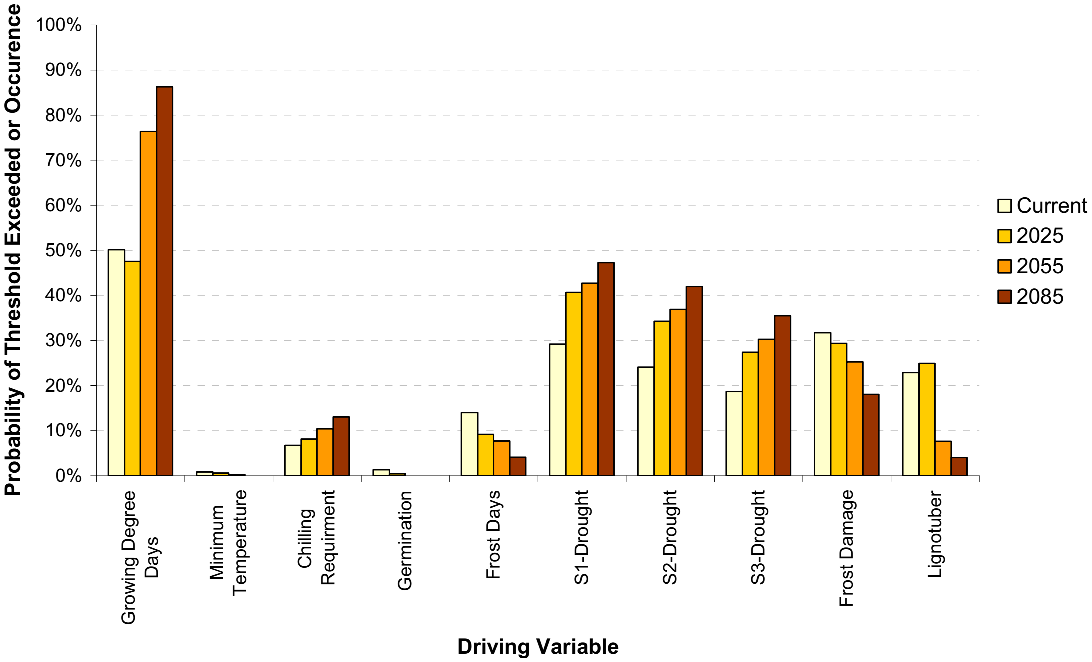


Fig. 46: Sensitivity analysis of driving variables for all species. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

4.7 Response of Selected Variables to Climate Change

Significant increases in the number of germination days were detected under climatic change scenarios across all treatments. Table 11 summarises these findings. The stand replacing treatment had the highest number of germination days and the stand modifying treatment the least. The results suggest that the number of days on which climatic conditions maintain primary dormancy and invoke secondary dormancy will decline under predicted future climates. The average number of frost days was also modelled to decrease under predicted climate change, with a significant difference in the average number of frost days from 2025 onwards across all treatments. Table 12 summarises the change in frost days under climate change. The significant change in average number of frost days and maximum number of frost days could allow species that are currently restricted to lower elevations by frost events to regenerate at higher elevations. The modelled species responses showed this to be the case for many of the species currently excluded from higher elevation sites.

Table 11: Change in number of "Germination Days" under predicted climate change across microclimatic treatments versus current climate period

		Stand Replacing		
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	143	2.1	4	304
2025	157^{***}	2.0	6	308
2055	195^{***}	1.9	40	336
2085	201^{***}	2.0	39	347
		Stand Modifying		
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	89	2.0	0	257
2025	107^{***}	2.0	1	269
2055	156^{***}	1.9	36	301
2085	171^{***}	1.9	39	317
		Stand Maintaining		
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	143	2.2	4	316
2025	156^{***}	2.1	7	319
2055	191^{***}	2.1	39	345
2085	196^{***}	2.2	37	354

n = 3960; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

Table 12: Change in number of "Frost Days" under predicted climate change across microclimatic treatments versus current climate period

		Stand Replacing		
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	40	11.0	0	232
2025	30^{***}	8.6	0	186
2055	20^{***}	6.3	0	153
2085	8^{***}	3.9	0	111
		Stand Modifying		
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	101	16.0	0	306
2025	81^{***}	14.3	0	266
2055	59^{***}	11.6	1	235
2085	33^{***}	9.0	0	193
		Stand Maintaining		
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	31	9.5	0	204
2025	22^{***}	7.2	0	166
2055	15^{***}	5.2	0	136
2085	6^{***}	3.0	0	91

n = 90; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

4.8 Damping-Off Fungus Risk under Climate Change

Damping-off fungus (*Pythium* spp.) can cause significant mortality to young eucalypt germinants under warm and wet conditions (Brown and Ferreira 2000). Increases in fungus-based diseases are predicted to increase under climate change and have already been documented (see Woods et al. 2005). To track any potential changes in fungus risk the number of days with climatic conditions conducive to damping-off fungus (*Pythium* spp.) occurrences was tracked in TACA-OZ. In southern Victoria, Pythium spp. favours wet conditions with soil temperatures from 18 to 25 °C (air temperatures from 10.3 to 14.2 °C) (Neumann and Marks 1989; Flint and Fagg 2007). Tables 13 to 15 summarise the response of "fungus days" for each treatment. A significant increase in the mean number of fungus days was found on sheltered aspects (south facing) only in the stand replacing (Table 13) and stand maintaining (Table 14) treatments. In the stand modifying treatment significant changes in the mean number of "fungus days" were detected on both exposed and sheltered sites. The changes in stand replacing and stand maintaining treatments, though significant, were marginal. The changes in the number of "fungus days" in the stand modifying treatment suggest a 30.3 % increase in risk by 2085 on exposed sites and a 40.6 % increase on sheltered sites. A general decrease in the maximum number of "fungus days" and general increase in the number of minimum “fungus days” indicate a possible increase in the risk of fungus outbreaks occurring from year to year under future predicted climates.

Table 13: Change in number of “Fungus Days” under predicted climate change within the stand replacing treatment on exposed (north aspect) and sheltered (south aspect) sites versus current climate period

North Aspect				
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	48	2.5	3	95
2025	49	1.9	14	81
2055	48	1.7	11	76
2085	47	1.8	1	75
South Aspect				
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	48	2.5	2	94
2025	51^{***}	2.0	8	82
2055	51^{***}	1.6	16	77
2085	52^{***}	1.3	15	75

n = 360; * P < 0.05; ** P < 0.01; *** P < 0.001

Table 14: Change in number of “Fungus Days” under predicted climate change within the stand maintaining treatment on exposed (north aspect) and sheltered (south aspect) sites versus current climate period

North Aspect				
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	47	2.6	5	94
2025	47	2.1	13	82
2055	46	1.8	9	76
2085	45	2.0	0	75
South Aspect				
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	48	2.5	2	93
2025	50^{***}	2.0	8	82
2055	50^{***}	1.6	16	76
2085	52^{***}	1.4	10	75

n = 360; * P < 0.05; ** P < 0.01; *** P < 0.001

Table 15: Change in number of "Fungus Days" under predicted climate change within the <u>stand modifying treatment</u> on exposed (north aspect) and sheltered (south aspect) sites versus current climate period

North Aspect				
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	33	2.8	0	89
2025	37^{***}	2.6	1	82
2055	39^{***}	2.4	3	79
2085	43^{***}	2.0	11	75
South Aspect				
Climate Period	Mean	95 % C.I.	Minimum	Maximum
Current	32	2.7	0	80
2025	38^{***}	2.6	1	79
2055	40^{***}	2.4	1	79
2085	45^{***}	2.0	7	81

n = 360; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

4.9 Change in Soil Moisture under Climate change

A significant decrease in soil moisture occurred between the current modelled period and the 2025 to 2085 climate periods across all treatments and aspects (n = 270; $P < 0.001$). Figure 47 presents the mean annual change in the soil AET/PET ratio. Figure 48 shows the proportion of change in potential soil moisture based on the change in AET/PET ratio. The exposed, north aspects were drier than the sheltered south aspects, as expected, with the relationship holding under climatic change. The stand maintaining site experienced the largest decline in potential soil moisture by 2085. The decline in soil moisture correlates with the increasing influence that drought had on species response under predicted climate change (see Fig 46). The lower soil moisture potential modelled for the stand maintaining treatment reflects the interaction between the smaller diurnal range in temperature and canopy interception of rainfall on microclimatic conditions in the understorey. The changes in the annual AET/PET ratio are subtle; however, many species are currently at the margin of their drought thresholds in much of the Central Highlands and these margins are exacerbated by the predicted warmer and drier conditions. The subtle decrease in soil moisture in the stand maintaining treatment is enough to predispose a species to more frequent drought events compared to the other two treatments.

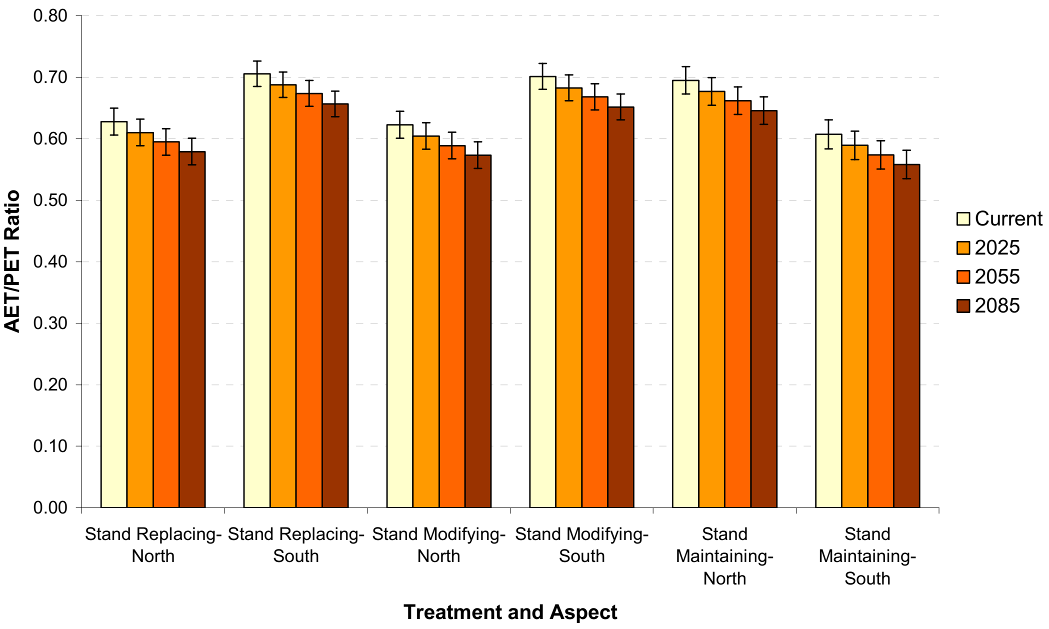


Fig. 47: Change in mean annual soil AET/PET Ratio under predicted climate change across all treatments and sites with 95 % Confidence Intervals

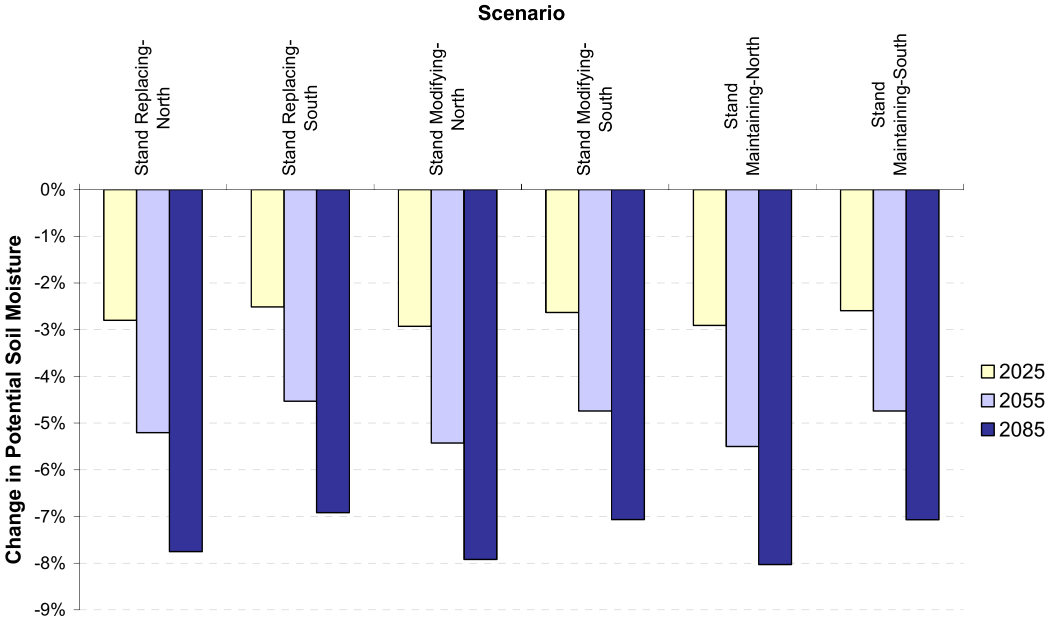


Fig. 48: Change in available soil moisture under predicted climate change across all treatments and sites

4.10 Model Performance

The performance of TACA in modeling the response of species to biophysical mechanisms under the current climate scenario is important for establishing a credible assessment of species vulnerability under future climate change scenarios. Table 16 summarises the modelled elevations where the TACA model identified the regeneration probability of a species to be greater than zero. The modelled responses were then compared to a literature-based source, Boland et al. (1992) and to the SFRI database, which has inventoried species presence in the Central Highlands, to validate TACA's ability to model species presence and absence. When compared to Boland et al. (1992), TACA modelled species presence within the reported lower elevation limit for all but four of the species. The upper elevation range was overestimated for all but three species, but was within 100 m for three other species. When compared to the SFRI data, TACA performed much better at the higher elevation range, over-estimating by 100 m for 16 of 18 species. No data were available for the *Acacia* species in the inventory and no data exists for Montane Wattle, so validation was not possible. TACA was able to model the observed 300 m difference in species presence between exposed, north facing sites and sheltered, south facing sites as described by Costermans (1994). The modelled presence of Alpine Ash was identical to inventoried distribution in the Central Highlands, which is much wider than that reported by Boland et al. (1992). The same applies for many species where the SFRI inventory data has identified species being present outside of the range described in the literature. Some species that are poorly represented in the inventory data are underestimated when compared to TACA and Boland et al. (1992). Based on a combination of the two independent sources, TACA was able to model species presence within the reported ranges for over 90 % of the species. TACA did overestimate the majority of species by 100 to 150 m on both north and south aspects. An explanation for this could be because the majority of forests in the Central Highlands are over 60 years old and the climate in Australia has risen by 0.4 to 0.7 °C over the last 50 years. The majority of the weather stations records are pre-1950. Over the weather record period, a measured environmental lapse rate in the Highlands of 0.53 °C/ 100 m exists. Therefore, the 100 m overestimation by TACA likely reflects the observed increase in temperature over the last 50 years in Australia.

Table 16: Modelled species presence along elevation gradient compared to report species elevation limits and inventoried presence of species at elevation in the Central Highlands

Species	TACA-North	TACA-South	Boland et al. 1992	SFRI GIS Data
Silver Wattle	200 - 1600	200 - 1600	50 - 1000	No Data
Montane Wattle	200 - 1500	200 - 1200	No Data	No Data
Black Wattle	200 - 900	200 - 600	0 - 850	No Data
Blackwood Wattle	200 - 1600	200 - 1600	0 - 1500	No Data
Southern Sassafras	200 - 1600	200 - 1600	200 - 1400	200 - 1500
Mountain Grey Gum	200 - 1500	200 - 1200	0 - 1200	200 - 1500
Mountain Gum	200 - 1600	200 - 1600	300 - 1700	400 - 1500
Alpine Ash	700 - 1600	400 - 1600	900 - 1500	400 - 1600
Broad-leaved Peppermint	200 - 1600	200 - 1600	150 - 1400	200 - 1500
Tingaringy Gum	200 - 1600	200 - 1600	800 - 1600	500 - 1400
Victorian Blue Gum	200 - 1600	200 - 1400	0 - 1050	200 - 1200
Red Stringybark	200 - 1200	200 - 900	150 - 1000	200 - 1100
Shining Gum	600 - 1600	300 - 1600	1000 - 1300	200 - 1500
Messmate Stringybark	200 - 1600	200 - 1600	0 - 1000	200 - 1400
Swamp Gum	200 - 1600	200 - 1600	0 - 1100	400 - 1600
Snow Gum	400 - 1600	200 - 1600	0 - 1500	500 - 1100
Narrow-leaved Peppermint	200 - 1600	200 - 1500	50 - 1200	200 - 1500
Mountain Ash	200 - 1600	200 - 1600	150 - 1100	200 - 1500
Candlebark Gum	200 - 1600	200 - 1600	75 - 1400	300 - 800
Silvertop Ash	200 - 1600	200 - 1600	0 - 1100	600 - 800
Manna Gum	200 - 1600	200 - 1500	0 - 1400	200 - 1500
Myrtle Beech	200 - 1600	200 - 1600	0 - 1600	200 - 1500

The final validation of TACA was its ability to model change in soil moisture. Since TACA used a potential evaporation model to model change in soil moisture over a year, potential evaporation was compared to the estimated change in potential evaporation for the Central Highlands region conducted by CSIRO. Howe et al. (2005) reported a 3 % increase in potential evaporation by 2020 and an 8 % increase by 2050 in the Central Highlands region. TACA modelled a 3.3% increase by 2020 and 8.1% by 2050 (weighted mean from 2040 and 2100). Figure 49 shows the modelled potential evaporation from TACA compared to Howe et al. (2005). TACA did not model the minimum and maximum values reported by Howe et al. (2005); however, the average was very similar. Howe et al. (2005) reported the output of 13 GCMs, while outputs of only three were used in our study. This likely explains the reduced variability. The similarity in potential evaporation modelled by TACA and reported by Howe et al. (2005) provides a degree of independent validation of TACA's ability to realistically model changes in potential evaporation and therefore potential changes in soil moisture.

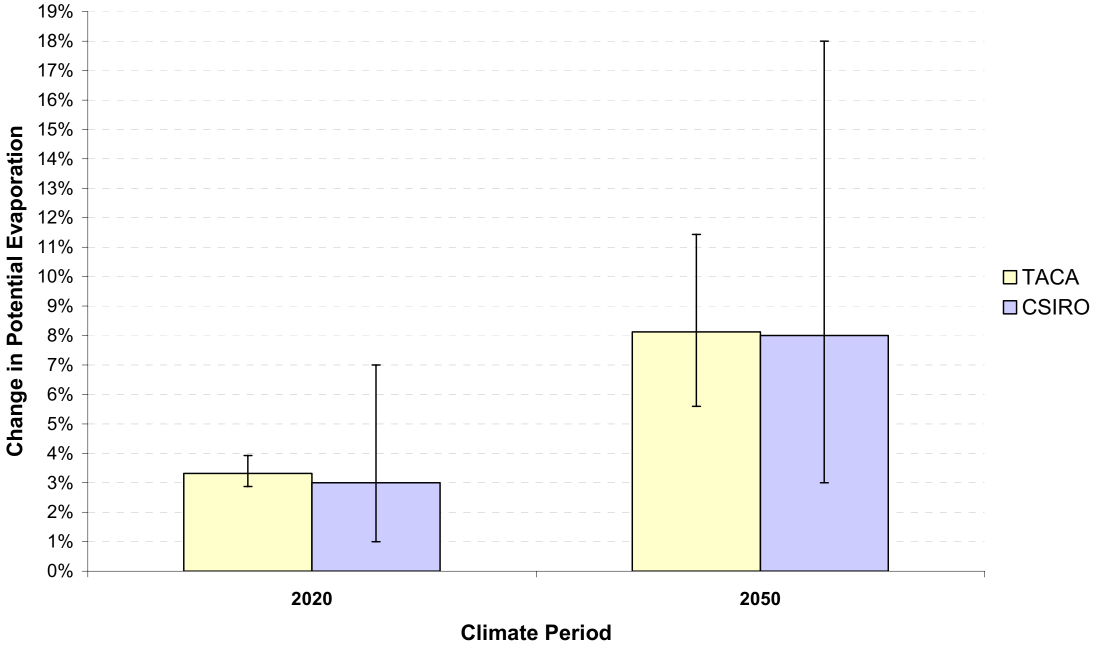


Fig. 49: Modelled change in potential evaporation under climatic change from TACA versus CSIRO (Howe et al. 2005) modelled change. Error bars represent the maximum and minimum modelled values

5. Discussion

Understanding species' regeneration responses can provide insight to how a species or ecosystem may respond to future perturbations (Carpenter et al. 2001). Understanding potential future responses can be useful for developing adaptation strategies that are both robust to uncertainties and reversible (Carpenter et al. 2001). Holling (1996) defined the magnitude of disturbance required to flip a system to a new stable state as ecological resistance. Ecological resilience refers to the magnitude of disturbance that can be absorbed before the system is restructured with different controlling variables and processes (Gunderson et al. 2002). The objective of this study was to investigate the vulnerability of forest trees in the Central Highlands of Victoria to predicted climate change by examining resilience in their regeneration niche.

The four-phase model of ecosystem dynamics (Holling 1986; Gunderson and Holling 2001) (see Fig. 50) is one useful method for understanding the resilience and resistance of a system to climate change (Hansell and Bass 1998). An ecosystem is postulated to pass through four phases: exploitation, conservation, creative destruction and reorganisation (Carpenter et al. 2001; Gunderson and Holling 2001). The influence of climate change is important in both the creative destruction and renewal phases (Hansell and Bass 1998). The creative destruction phase is related to disturbances that are directly linked to, if not caused by, climate, such as fire and drought. The reorganisation phase is associated with the renewal of resources that are released through the creative destructive phase. In this stage, changes in temperature and precipitation may favour different species (Hansell and Bass 1998) and cause the system to shift to a new stable state (Gunderson et al. 2002). Our study focused on the reorganisation phase, under the assumption that creative destruction by natural or anthropogenic disturbance has released the resources that occupied the site in the conservation stage. The reorganisation or renewal stage is synonymous with the regeneration niche of a species, as modelled with TACA-OZ.

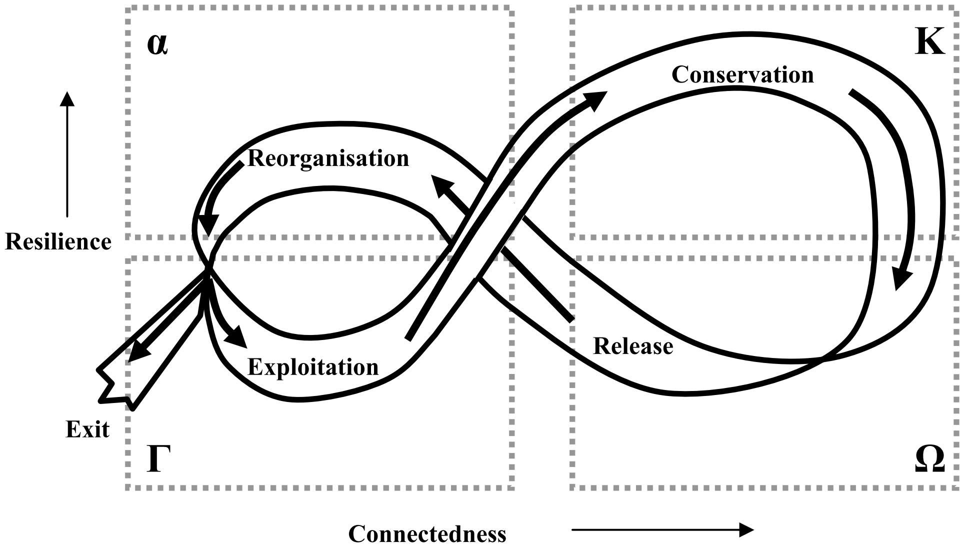


Figure 50: Holling's figure-eight model and a representation of the flow of events between ecosystem functions (Ω, Γ, K, α) (adapted from Gunderson and Holling 2001)

5.1 Species Response to Climate Change in Victoria's Central Highlands

The modeled responses of species are not predictions but instead illustrate the vulnerability of a species to predicted changes in climate. For example, species modelled as being extremely vulnerable will not necessarily disappear from the landscape, but will likely decrease in abundance over time and be restricted to sites where topographic and edaphic conditions mediate the direct effects of climate change. This is highlighted in Fig 46 where drought thresholds are exceeded less often on sites with deeper, finer textured soils. Also, species that were modelled to have nil or low vulnerability will not necessarily remain static because the rating of low or nil vulnerability points to the resilience of a species not the resistance. The change in regeneration potential on sites also shows the future resilience of a species to future disturbances in the system. The change in the regeneration potential reflects an increase in climatic optimality and vice versa. A decrease in optimality can reduce functionality within the system, which in turn will increase the vulnerability of species to future change (Gunderson et al. 2002).

Our analysis of species response suggests that the majority of species in the Central Highlands are both resistant and resilient to predicted climate change in the 2025 period. A non-significant change in mean regeneration niche and in fundamental range size highlights this resilience and resistance. Despite this, the analysis identified that species did not necessarily exhibit a static response in terms of regeneration potential. Figures 24 to 45 illustrate this point with regeneration potential declining while range size expands or remains static under the 2025 climate scenario. These findings highlight an increase in sub-optimal conditions at lower elevations as climate warms and dries. The increase in sub-optimal conditions at lower-elevations is counterbalanced with an increase in optimal conditions at higher elevations. Here, warming opens higher elevation sites to regeneration by species formerly constrained by frosts and cooler temperatures. Three species remained resilient to predicted 2025 climate change but not resistant under the stand replacing treatment. Alpine Ash, Shining Gum and Snow Gum all suffered minor range contractions at lower elevations with Alpine Ash suffering the greatest range contraction. Under the stand modifying treatment, only Alpine Ash and Snow Gum suffered a contraction in range. The stand maintaining treatment caused four species to suffer minor range contraction; the three species mentioned above and Tingaringy Gum. A reduction in chilling weeks is responsible for this loss of resistance at lower elevations. At higher elevations, where these species dominate, resistance and resilience was maintained. These findings are supported by Cumming and Burton (1996) who modelled a lack of winter cooling for forest regeneration to be a driving factor for species contraction to higher elevations.

By the 2055 climate period, the majority species suffered a significant contraction in range size in the stand replacing and stand maintaining treatments, with the exception of

Montane, Black, and Blackwood Wattle and Red Stringybark. By 2085, Montane Wattle had a significant range contraction. Despite the lack of significant difference in range size, these four species did exhibit a loss of resistance in their regeneration niche as shown in Figures 25-27 and 36. For all species, the resilience and resistance in their regeneration niche is overcome by 2055 at lower elevations, which forces a large contraction in species' ranges to higher elevations and causes a flip in their regeneration niche from lower elevation to higher elevation while still maintaining their fundamental range size. A change in regeneration potential was also observed with the breadth of species' regeneration niches declining under the stress of the 2055 climate conditions. The 2085 climate conditions exacerbated this contraction in range and niche, forcing species to higher and higher elevations. The stand modifying treatment mediated species response with all species suffering less contraction in range and having a higher regeneration potential. This treatment fostered the resilience of each species regeneration niche. Despite this, significant changes still occurred for over half of the species, especially on exposed, north facing aspects. Species responded much more negatively on sites with northern aspects versus sheltered sites on southerly aspects in all treatments. Tables 10 and 11 highlight the increase in resilience to predicted climate change associated with sheltered sites. Figures 24-45 also illustrate the influence of aspect, with species showing greater resistance to range contraction at lower elevations and increased resilience in their regeneration potential at higher elevations. Microclimate variation due to topography causes local and regional differentiations in environmental conditions, which can result in greater differentiation in species response in heterogeneous environments. This enables some species to potentially cope with change at the local-scale, but not the landscape-scale (Theurillat and Guisan 2001). The results of this study support this hypothesis, as species were found to respond differently on sheltered versus exposed sites in all time periods, with sheltered sites increasing species resilience through their mediating influence on temperature. No species were able to regenerate below 500 m by 2085 in the stand replacing treatment, 600 m in the stand maintaining treatments and 300 m in the stand modifying treatment on exposed aspects, while on sheltered slopes 300 m was the threshold in stand replacing and maintaining treatments and 200 m for the stand modifying treatment. At the higher elevations, all species were still able to regenerate above 1500 m on sheltered sites under the stand replacing treatment, above 1300 m under the stand modifying treatment, while in the stand maintaining treatment 21 versus 18 species were able to regenerate on the exposed sites.

Species regeneration niches were found to shift upwards in elevation, primarily due to increases in temperature and droughts. The significant decrease in frost days and reduction in the occurrence of frost damage allowed many species that are presently constrained by these mechanisms to move to higher elevations. The modelled decline in soil moisture increased the incidence of droughts, which in turn caused the lower-elevation range of many species to contract to areas were their drought tolerance was not exceeded. Edaphically, species suffered a greater incidence of drought on sites with shallower, coarser textured soils than sites with deeper, finer textured soils. For some drought sensitive species, such as Myrtle Beech and Mountain Ash, this allowed them to regenerate on the mesic to non-water limiting portions of a plot but forced them to contract from more xeric sites due to increased soil moisture deficits. This mechanism is reflected in the decline n regeneration potential observed within the modelled future ranges. In our study, species were found to respond to climate change in the three directions identified by Aber et al. (2001) as the expected responses of forest to climate change. At the broader topographic-scale of the Highlands, species' range contracted at lower elevations as drought and temperature increased, while for some species, ranges expanded as temperature increases made higher elevation sights optimal for regeneration. At the finer topographical scale, aspect mediated species response on sheltered sites while exacerbating their response on exposed sites. Edaphic conditions forced species to contract from xeric sites to mesic or non-water deficit sites, which in turn reduced species regeneration potential at a given elevation. The predicted contraction of Myrtle Beech to higher elevations in the Central Highlands is consistent with the findings of Busby (1988). Busby (1988) utilised the BIOCLIM model which predicts species' response based on a coarse and static species-climate envelope relationship. This study used a different approach and yielded congruent results.

Under climate change, it is hypothesised that species will respond individually as climate envelopes that govern habitat and eco-physiological thresholds deform (Bush 2002). The species' responses in our study support this hypothesis; with the majority of species responding to climatic change with a contraction in range, or with a contraction in one part of the range and expansion into a new range (see Appendix I). A change in the optimality of areas within the Central Highlands for regeneration was also driven by individualistic species response. The possible future ranges and regeneration potentials illustrated in

Figures 24 to 45 highlight this point (). The climatic thresholds that drove these changes varied between species, highlighted by the driving variable analysis presented in Appendix II. The change in possible species diversity at each elevation also suggests that species move as individuals and not as communities, with possible species diversity at each elevation changing with predicted climate change. This would result in a diversity distribution in 2085 that is much different than the one that exists in the current and 2025 modelled periods. Our results also suggest that current ecological vegetation communities in the Central Highlands will eventually disaggregate, if subjected to disturbance, as species' regeneration niches shift around the landscape under future climate change. The disaggregation of the current EVC's will bring into question the future utility of the regeneration and reforestation policies that are currently based on the species compositions that exist in the current EVCs (DSE 2007).

Results from this study suggest that the predicted changes in climate for the Central Highlands will not impact greatly on species and ecosystems through to 2025, but by 2055 the core range of each species distribution may be fragmented (Barrio et al. 2006). In the transient period between the present and 2025, species and ecosystems may also exhibit a temporal overlap in response to climate change. In our study, increases in temperatures between the present and 2025 enabled the potential expansion of species to higher elevations while still allowing current species to persist in their current locations. This could result in stands that have higher diversity than is presently the case. However, the temporary climatic window that would enable increases in diversity will close between 2055 and 2085, as higher temperatures and drought contracts the ranges of persisting species, thus reducing diversity at lower elevations and on exposed sites at higher elevations. Figures 18-23, illustrate this potential response. This mechanism was identified by Shriner and Street (1998) as a transient response that may occur in forested ecosystems as climate increasingly warms. The species-specific thresholds identify the varied responses over time that triggers both increases in diversity by 2025 and decreases in diversity by 2055.

5.2 Are Victoria's Central Highland Forests Vulnerable to Climate Change?

Mansergh and Cheal (2007) identified ecosystem resilience as an important attribute that needs to be restored in Victoria's forest ecosystems through regeneration of forests and woodlands in the landscape matrix. To restore ecological resilience we first need to understand if species are vulnerable to predicted climate change. Based on the results of our study, Victoria's Central Highland forests should be considered resilient to the next 20-30 years of predicted climate change, but by the 2055 climate period (representing the period 2040 to 2069), the Central Highlands should be considered vulnerable to climate change. Based on the significant contraction in regeneration niche and fundamental range size, 18 tree species should be considered extremely vulnerable to 2055 climate change if there is a stand replacing disturbance. Our analysis also identified that 18 species are extremely vulnerable if only stand maintaining disturbance occurs. The stand modifying disturbance was found to reduce species vulnerability in 2055, but 10 species are still rated as extremely vulnerable and nine as high or very high. Therefore, no matter what the treatment or disturbance intensity, the Central Highlands forest trees are vulnerable to predicted climate change by 2055. This is a result of increased temperatures and a decline in precipitation that will drive species past a generic threshold in this period. It should be noted that the influence of interactions between biophysical variables was not considered which could lead to an underestimation of species vulnerability. Likewise, we utilised species' biophysical thresholds from both the Central Highland and other Australian populations thereby incorporating the widest range of genetic variability possible. The wide environmental amplitude of the thresholds may mean that species' response was underestimated for some populations but overestimated for others.

5.3 Are There any Feedback Loops and Thresholds?

Parry et al. (2007) argued for the need to identify how close natural ecosystems are to tipping points/ thresholds and what feedback loops exist if these points are reached. This study identified four feedback loops, as well as a threshold where species vulnerability to climate change was magnified. An unexpected feedback was the significant increase in the mean number of days available for germination under predicted climate change. The increase in temperatures expanded the germination window for all species by 2025 into the late autumn, winter and early spring months. The reduction in annual precipitation masked an average increase in summer precipitation, which likely helped to moderate occurrences of drought-induced secondary dormancy. A significant decline in the number of frost days was also modelled. This allowed for the germination window to be expanded but did predispose species to frost damage. The occurrence of frost damage declined but by a lesser magnitude than the number of frost days. Under a warmer future, germination can be expected to occur more often with frost still being a major mortality agent for species germinating from late autumn to early spring. The significant decline in soil moisture also means that soil moisture conditions that invoke secondary dormancy will still occur in the hotter and drier months.

Another feedback that was identified was the significant increase in the number of days that were climatically suitable for damping-off fungus (*Pythium* spp.) to develop. This was particularly the case in the stand modifying treatment that incurred a significant increase in mean fungus days on both exposed and sheltered sites. Under the stand modifying treatment there was a 40 to 60% increase in the number of fungus days. Woods et al. (2005) have already documented that a Dothistroma needle blight (*Dothistroma septosporum*) outbreak in northwestern British Columbia, Canada is the result of recent climate change. Under predicted climate change, the number of fungus and disease outbreaks in forests are expected to increase (Volney and Hirsch. 2005; Woods et al. 2005). Our study supports this hypothesis with an increase in fungus risk modelled on sheltered sites in all treatments and on exposed sites in the stand modifying treatment. This finding constitutes a significant positive feedback because damping-off fungi can cause high mortality to young eucalypt germinants (Kellas 1994; Ashton and Chinner 1999; Brown and Ferreira 2000; Walters and Bell 2005; Flint and Fagg 2007). It is important to note that drought and frost are the largest causes of mortality to germinants and seedlings in Victoria. The increase in damping fungus risk provides an additional threat to the future regeneration potential of species on sheltered sites, particularly following a stand modifying event. The stand modifying treatment mediated species vulnerability to climate change but increased the risk of fungus outbreaks occurring. This feedback could reduce the regeneration potential of species even further, in combination with drought and frost, when climatic conditions are favourable for regeneration in the wetter and warm portions of a year.

Natural systems can experience complex responses, characterised by chaotic behaviour and reorganisation, when thresholds are crossed (Thomas 2001). The identification of thresholds is important if resilience and sustainability are to be fostered within systems (Gunderson and Holling 2001). Hennessy et al. (2007a) stated that the species and ecosystems of Australia are very vulnerable to climatic change because they have a narrow coping range and that a change in temperature greater than 1.5 °C would be a threshold for increased vulnerability. This vulnerability is a result of the majority of species in Australia having a narrow climatic breadth of tolerance (Hughes et al. 1996; Hughes 2003). Our study found a significant change in species vulnerability with a 1.4 °C increase in annual temperature coupled with a 5 % decline in annual precipitation in 2055. This threshold was the same for the eucalypt, rainforest and acacia species. Hughes et al. (1996) identified that 41 % of *Eucalyptus* species have a temperature range < 2 °C, while Hughes (2003) stated that a 2 °C in warming would cause significant range reductions in *Acacia* species. Blackwood Wattle which has the largest climatic breadth of all the acacia's had the lowest vulnerability while Silver Wattle, a more restricted species, suffered a greater range contraction. The 2055 threshold of 1.4 °C strengthens the argument that the majority of Australian species have a narrow coping range and are vulnerable to a 1.5 °C increase in temperature due to climate change. The reason why the threshold of 1.4 °C found in this study is less than the 1.5 °C identified by Hennessy et al. (2007a) can be explained by the climate records used in this study. Hennessy et al. (2007a) based there threshold on a 1990s baseline while in this study climate records up to 2007 were utilised. The last 10 years have been the warmest ever recorded and this effect is likely the cause of the lower threshold found in this study. Even the species that are rated with lower vulnerability in this study, (Black Wattle, Blackwood Wattle and Red Stringybark), still showed major shifts and contractions in their regeneration niches and fundamental ranges. The threshold of a 1.4 °C increase in annual temperature coupled with a 5 % decline in annual precipitation will likely cause species to exceed biophysical thresholds that allow regeneration and subsequent growth. This threshold will result in a significant contraction in regeneration niches and range size as species resilience is overcome by changes in climate. The threshold should be considered an inflection point where ecosystem function and structure will be significantly vulnerable. This threshold may be reached before 2055 since greenhouse gas emissions are currently increasing at a greater rate than assumed in all IPCC climate change scenarios (Raupach et al. 2007). The species-specific response to predicted climate change in Victoria's Central Highlands illustrates the impact that a narrow coping range has on a species' ability to adapt. This will be a major challenge for land managers trying to adapt forest ecosystems to rapid climate change.

5.4 Species Vulnerability as a Proxy of Ecosystem Vulnerability

From an ecosystem perspective, the vulnerability of the Central Highlands tree species to rapid climate change suggests that many of the ecosystems that exist in the current landscape will also vulnerable. For example, Myrtle Beech and Southern Sassafras are the dominant species in the Cool Temperate Rainforest ecosystem in the highlands. Both species have been rated as extremely vulnerable in our study, and this would indicate the Cool Temperate Rainforest is also vulnerable. The lower vulnerability of Blackwood Wattle, a tertiary species in current Cool Temperate Rainforest, could elevate it to dominance on these sites creating an ecosystem that more closely represent Warm Temperate Rainforest (e.g., found on sheltered sites in Wilson's Promontory). The Wet and Montane Wet Forest EVCs are dominated by Mountain Ash and Alpine Ash, respectively. Both of these species are classified as extremely vulnerable. Damp Forest is also vulnerable as the species that dominate, particularly Messmate Stringybark, contract to higher elevations. Based on our results, the ecosystem that can be seen to have the highest vulnerability is the Subalpine Woodland EVC. Our results indicate that Snow Gum suffers the greatest range contraction of all species, whilst the area that Snow Gum dominated Subalpine Woodland EVC occupies becomes increasingly optimal for all modelled species from 2025 to 2085. Williams (1991) suggested that Snow Gum is likely to be occupying most of its potential range already. Our results reflect this observation.

The ecosystems that were found to have the lowest vulnerability are the mixed species Montane Damp Forest and Montane Dry Woodland. The mixed species nature of these ecosystems may provide a good indication of what the Central Highlands may look like in the future above 500 m elevation. The relative abundance and composition of species will likely differ, but the mixed nature and presence of all current species, at some abundance, may occur. At lower elevations, it is possible that ecosystems resembling the Box-Ironbark forests may emerge as the warmer, drier climate expands the regeneration niches of these more heat and drought tolerant species. Species will shift and ecosystems will transform as climate changes - it always has (Bartlein et al., 1997). In the future, we can expect the shifting biotic associations in the Central Highland, as species lose their resilience and contract to areas where climate permits regeneration. These areas should be regarded as bastions of biodiversity and climatic refugia for species and ecosystems (Kirkpatrick and Fowler 1998; Theurillat and Guisan 2001; Burke 2002; Rouget et al. 2003).

5.5 Can Current Forest Management Foster Ecological Resilience?

5.5.1 Climate Change and Forest Management: Fostering ecological resilience

Across all treatments, the Central Highlands forest trees exhibited significant vulnerability to predicted future climate change beyond 2055. The stand modifying treatment was found to have reduced impact compared to the others resulting in lower vulnerability classifications for all but nine species while the other two treatments resulted in 18 species being classified as extremely vulnerable. The stand modifying treatment was based on microclimatic differences that were recorded between a clearfell and a shelterwood treatment with > 30 % overstorey retention (Kellas 1994). This silvicultural treatment moderated the interaction between temperature and precipitation to reduce the response that species exhibited in the stand replacing and stand modifying treatments. This treatment scenario does not suggest that the shelterwood silvicultural systems are the only management response that might be used in adapting management to minimise the predicted impacts of climate change on forests, but it does provide some insight into how to foster species resilience. Importantly, our analysis did not consider light or competition, only climatic conditions, and these will have important influences on successful regeneration under any silvicultural system. For example, under the historical climate regime, the type of silvicultural system used has had an influence on the composition of regenerating stands. Lutze and Faunt (2006) found that harvesting treatments, ranging from selection harvesting to 10 ha clearfells, caused a significant shift in species mix following regeneration in low elevation mixed forests, with the more intensive clearfell and 7 % retention treatments being the best systems for maintaining species composition. Faunt et al. (2006) found that all species were able to successfully regenerate after three years irrespective of silvicultural treatment, but that growth was significantly lower in the less intensively harvested sites. Kellas (1994) identified a shift in species dominance following shelterwood harvesting of Messmate Stringybark dominated forest. Following treatment, the proportion of Messmate Stringybark declined while the proportion of Broad- and Narrow-leaved Peppermints increased. Interestingly, in our study both Broad- and Narrow-leaved Peppermints exhibited increased resilience under the stand modifying treatment, resulting in a non-significant change in both niche size and potential range by 2055 and 2085. Messmate Stringybark maintained its mean niche size but not is range.

Cool Temperate Rainforest is protected from harvesting in most situations in Victoria. A Tasmanian study showed that the intensity of selection harvesting in Cool Temperate Rainforest also has an impact on stand composition, with an increase in Myrtle Beech and Southern Sassafras mortality when greater than 50 % of the overstorey is removed (Elliot et al. 2005). In our study, both Myrtle Beech and Southern Sassafras suffered less range contraction under the stand modifying treatment, with Myrtle Beech being classified in a lower vulnerability class. It should be noted that Elliot et al. (2005) also identified an increase in Myrtle Wilt disease caused by the fungus (*Chalara australis* Kile & Walker) following selective harvesting, leading to the increased mortality of Myrtle Beech. Though Myrtle Wilt was not considered in our study, a significant increase in climatic optimality for damping-off fungus was identified in the stand modifying treatment. This may also occur for Myrtle Wilt because an increase in "fungus days", along with the modelled increased incidence of drought, increases the susceptibility of Myrtle Beech under stand modifying treatments (Elliot et al. 2005).

In general, regeneration of the eucalypt dominant ecosystems after harvesting in the Central Highlands rely on stand replacing treatments, particularly clearfell or seed tree silvicultural systems. In both the lowland forest types and the high elevation mixed species forests, a seed tree system is used in conjunction with an intensive slashburn to produce a suitable seed bed and induce seedfall (Lutze et al. 1998b; Bassett et al. 2006). In the Wet Forests of the Central Highlands, clear-fell harvesting is the most commonly used silvicultural system, while the seed tree system is used infrequently (Flint and Fagg 2007). Shelterwood, group selection, and variable retention harvesting have been trialled but are not used operationally (Flint and Fagg 2007). The shelterwood and selection systems are not recommended for managing Mountain Ash dominated forests (Campbell 1997). This is supported by Ashton and Chinner (1999) who stated that regeneration of Mountain Ash will "almost certainly fail" under a mature canopy unless accompanied by exceptional circumstances or following a stand replacing disturbance. Dignan et al. (1998) found that Mountain Ash could regenerate under a shelterwood system, but that growth was reduced by 96 % after 3 years with regeneration ultimately failing. Utilisation of an overwood retention system that retains 10-15 % of the canopy has been found to successful in Mountain Ash forests and offers intermediate option between clearfell/ seed tree and shelterwood/ selection systems (Campbell 1997). We did not consider the microclimatic conditions offered by this treatment; further research is required to investigate this potential management option.

Stand modifying treatments such as shelterwood harvesting has been used extensively in Victoria's low elevation mixed species foothill forests (Walters and Bell 2005) while in the Cool Temperate Rainforests of Tasmania, selective harvesting is used to harvest Myrtle Beech and Southern Sassafras for speciality wood products (Elliot et al. 2005). Cool and Warm Temperate Rainforest are protected from harvesting in Victoria. Higher elevation populations of Alpine Ash in Tasmanian are harvested using the shelterwood system but this technique is not suitable for the Victoria populations of Alpine Ash (Hickey and Wilkinson 1999; Lutze et al. 1999).

The reliance on stand replacing management treatments in the State forests of Victoria's Central Highlands and on stand maintaining management treatments (ex. fire suppression) in many protected areas and national parks, presents a conundrum for forest and ecosystem management aimed at adapting the Highland's forests to climate change. Based on the results of our study, the high degree of vulnerability that was recorded under both the stand replacing and maintaining treatments indicate that these management approaches will not foster the resilience of the modelled tree species. Instead, these treatments may perpetuate the narrow coping range that the various species exhibit. Our results indicate that the broader use of stand modifying treatments may provide a means for forest managers to foster species resilience by expanding their coping range through altered microclimatic conditions. This approach could ameliorate the adverse climatic conditions that regenerating species will face in the future. However, stand modification is not, by itself, a definitive solution. Species were found to still exhibit a high degree of vulnerability and will likely face the increased occurrence of drought and damping-off fungus if they successfully germinate. Also, a generic treatment will not suit all species. Bassett (2002) identified that in mixed species stands each species silvicultural requirements need to be managed individually. Even today, we see differential responses in species occupying the same stand following harvesting. Based on the findings of our study, this phenomenon is likely to increase under predicted climate change. It is important to note that interannual and interdecadal variation in climate will likely provide optimal conditions for regeneration which can and will need to be exploited by forest managers. This study identified a decline in regeneration potential which reflects a decline in the regeneration window of a species. Thus, regeneration could still occur where species are modelled to be able to regenerate at very low probabilities; however, the number of years that are suitable for regeneration will be reduced. This means that current management practices may still work as long as they are coordinated with years with suitable climatic conditions. For species with a low regeneration potential this reflects a regeneration window potentially opening from once every 15 years to once every 100 years. The understanding developed in this study and tools like TACA-OZ can be used to inform the timing of these actions. This type of management will be difficult and will likely be unable to achieve the sustainable management of these forest ecosystems when and where disturbances occur and species' regeneration windows are closed. The broad decline in regeneration potential illustrated in Figures 24-45 and A-1 to A-22 show the increased risk associated with pursuing adaptation based solely on changing the timing of current forest practices, not the practices themselves.

Perhaps the best hope for adapting the Central Highlands to predicted climate change is to classify the landscape into management zones that are designed to reflect increases in, or maintenance of, ecological resilience. Areas above 1300 m in elevation provide an ideal starting point for conducting more robust adaptive management actions, since species at this elevation are more resilient to climatic change. These areas provide the opportunity for the conservation of high-risk species, such as the rainforest species, through stand maintaining actions while allowing more intensive management actions, such as stand replacing treatments for species that have this ecological requirement (e.g., Mountain Ash). Such areas can be regarded as robust, where actions are made based on an understanding of ecosystem function under uncertain futures. Management actions in these areas will likely be robust to uncertainty and reversible if actions are misguided (Carpenter et al. 2001).

In areas that species and ecosystems are considered highly vulnerable (e.g., a regeneration potential of "Nil"), adaptation strategies should focus on actions that reduce disturbances and promote healthy ecosystem functions in order to exploit the biological inertia that exists within these systems (Franklin et al. 1992; Noss 2001). Maintenance of this inertia will assist the ability of the systems to cope with rapid environmental change (Brereton et al. 1995). Forest types where the dominant trees are protected from catastrophic disturbance may persist through periods of unfavourable climate and reproduce when more favourable conditions return (Noss 2001). Arguably, the best example of this is the Wollemi Pine (*Wollemia noblis*) in Australia. Managing areas where species lose their resilience in their regeneration niche may require management actions that reduce the chance of stand replacing disturbances [e.g., through fuel reduction or ecological thinning that alleviates competitive stresses for reduced resources (like soil moisture)]. The use of thinning has been recommended elsewhere in late-successional stands to decrease fire risk by reducing ladder and surface fuels (Spies et al. 2006). The reduction in competition for site resources will reduce the vulnerability of trees, particularly established understorey seedlings, to drought (Man and Lieffers 1999). Despite the presence of inertia, gradual shifts will still occur with local shifts in elevation and aspect (Franklin et al. 1992). This is particularly true in areas that have expanding regeneration niches from lower elevation species and contracting niches from current occupants. In Victoria, this process is already underway with certain trees invading Victoria's alpine areas (Hennessey et al. 2007a). This is projected to increase from 2025 onwards due to the increased diversity of species that can regenerate at higher elevations under predicted climate change.

Ecosystems are spatially heterogeneous due to variations in microclimate, edaphic properties and disturbance regimes. As a result, research outcomes from larger spatial scales may not be representative of smaller areas (and vice versa) (Bugmann et al. 2000; Barrio et al. 2006). The same applies for broad management actions. In our study, edaphic and microclimatic variation was incorporated, suggesting that sheltered sites and/ or sites with deep fine-textured soils respond differently than more exposed and xeric sites. This will likely result in increased heterogeneity in the landscape because species contract and expand from lower elevations while others contract to sites with reduced moisture deficits. Regeneration potential may also be poor across large portions of a species edaphic gradient and abundant in sheltered sites with favourable soil moisture conditions. The rainforest species are examples of species that are currently confined to sites with no or limited soil water deficits. Management actions that create or maintain heterogeneous landscapes will help facilitate the maintenance of ecosystem structure and function by increasing resilience at larger scales while fostering fine scale resilience (Halpin 1997; Theurillat and Guisan 2001; Bush 2002). Edaphic and topographic conditions will need to be taken into account to help foster future resilience of tree species in the Central Highlands.

Genetic variability also needs to be taken into account to help foster species resilience. The response of species will be influenced by genetic variation in growth and adaptation traits that influence how they will respond to climatic change (Rehfeldt et al. 1999; Hamann and Wang 2005). Broad geographic and climatic patterns may not account for local climatic and genetic variation; this can lead to both over and underestimates of a species' response and ability to persist (Rehfeldt et al. 1999; Epperson 2003). For the eucalypts assessed in this study we know that the level of genetic variation between populations is more than twice as great as northern hemisphere trees and that variations between populations is often continuous and clinal (Potts and Pederick 2000). In areas with steep environmental gradients, which the Central Highlands most certainly is, rapid changes in phenotypes can occur over hundreds of meters resulting in dramatic clinal variation that influences traits associated with the ability of the progeny of species to adapt to different climates (Potts and Pederick 2000). In this study, the parameters utilised were not specific to the populations of species within the Central Highlands. The parameters were derived from the maximum and minimum limits of both *ex situ* and *in situ* populations in relation to regional climatic patterns. The parameters used thus incorporate genetic variation from multiple populations which may mean that we have over or underestimated species response in this study. More research is needed to investigate the role genetic variability will have on species and ecosystem resilience. The identification of genotypes that can cope with the degree of predicted climate change may be important for conserving species within their current range. This may require the use of “human-assisted” migration as the degree of predicted climate change will require species to be able to migrate at a rate 10 to 15 times higher (300-500 km/ century) than observed migration rates (20-40 km/ century) and three times higher (100-150 km/ century) than anything recorded in the fossil record (Davis and Shaw 2001). Adaptation of local populations is also not guaranteed as genetic selection will need to be dramatic with multiple traits potentially under selection which may be impeded by genetic correlations adverse to the direction of selection necessary (Davis and Shaw 2001). This means that local populations of species may not be able to adapt and regional populations may not be able to migrate fast enough to provide the genes required for a species to adapt to the rapidly changing climate. This being said, genetic variability is an important tool that will need to be considered when undertaking adaptation to help foster future resilience of tree species in the Central Highlands. Maintaining the genetic variability within populations of different species will need to be an important management response.

The potential contraction of a species regeneration niche is a barrier that reduces the ability of a species to persist, although the change in abundance within its regeneration niche does not constitute an obstacle for maintaining species or ecosystem diversity (Bush 2002). The scale-based difference in the perceived vulnerability of a system creates a challenge for the managers of forest ecosystems. The fundamental decision that needs to be made is whether to protect the current mix of species or manage areas simply for the maintenance of diversity. Management actions that enable a species to move through environmental gradients may provide a viable means of protecting specific groups of species, while actions that seek to maintain species within heterogeneous sites (i.e., reserves) may or may not (Halpin 1997). The individualistic response of species suggests the need to manage at the niche-level rather than the community-level if species are to be conserved (Bush 2002). Management actions that are made based on the climatic optima of species are therefore recommended by Peters (1992) as a means of achieving this, so long as the management actions are flexible. Bush (2002) recommends basing management actions on the identification of climatic refugia; climatically suitable areas that provide a resource (niche space) and buffer habitat for species. The higher elevation areas of the Central Highlands fit this definition, as does the need to manage species individually. The modelled responses of species in this study are not endpoints as climate will continue to change well past the time period used in this study; as such, adaptation strategies should not focus on creating or maintaining ecosystems in new stable states but to foster species resilience and resistance so that important ecological structures and functions can be maintained under changing climatic conditions.

5.5.2 Regeneration: Natural or Artificial; Single or Mixed Species?

Regeneration of Victoria's native forests frequently rely on natural or induced seedfall from retained trees, or the direct sowing of seed into harvested coupes (Lutze et al. 1999; Basset et al. 2006; Flint and Fagg 2007). These are not the only options available to land managers. Following a disturbance (natural or anthropogenic), there is the option to plant seedlings. The use of artificial planting usually involves the establishment of single-species stands that are then augmented by natural regeneration from adjacent stands. In highly resilient systems, the use of natural regeneration or planting will enable the ecosystem to follow a successional pathway toward one of many possible states (Gunderson et al. 2002). In ecosystems with low resilience to climatic change, the use of natural regeneration alone may not guarantee that the ecosystem will develop along a traditional pathway - a new stable state may develop. The results from our study highlight the vulnerability that species may face in their regeneration potential in the future, with resilience lost by 2055. The utilisation of mixed-species planting is one method of diversifying the risk that ecosystems face and will lessen the risks posed by disturbance and biodiversity loss (Franklin et al.

1992; Whitehead et al. 2004). The modelled resilience in the 2025 period presents an opportunity for managers to engage in adaptive management actions that can diversify risks based on an improved understanding of species vulnerability offered through this study. The planting of mono-specific stands should be avoided because it will increase the vulnerability in systems with low resilience (Noss 2001). Artificial regeneration can also be used to facilitate the persistence of species and ecosystems through "human-assisted migration". This may become an important management strategy to deal with climate change (Hogg and Bernier 2005). Such a strategy is particularly important for species that require a chilling period for seed stratification to allow regeneration. Alpine Ash, Snow Gum, Shining Gum and Tingaringy Gum are examples of species that will be affected by climate change in this manner, particularly Alpine Ash. Artificial regeneration provides a management option that will reduce the degree of range contraction that these species could face. The planting of seedlings will also reduce the incidence of drought tolerance being exceeded. This is because seedlings have a much larger root system that will provide a greater chance of reaching soil moisture at greater depths. In our study, species on Site 3 (S3) exceeded their drought thresholds less often due to a larger available water supply. The use of artificial planting will also reduce the occurrence of damping-off fungus outbreaks since germinants, not seedlings, are susceptible (Brown and Ferreira 2000). The use of artificial planting will also benefit lignotuberous species. Despite the increases resilience offered by lignotuberous seedlings (Kellas 1994; Walters and Bell 2005), they must first develop from seed-based seedlings. Therefore, a species may be able to resist contraction of its range but it may not be able to expand its range if the seed-based portion of its regeneration niche does not materialise or become available at higher elevations. Planting seedlings of lignotuberous species can work to reduce this vulnerability.

For species that are not lignotuberous, enrichment planting under shelterwood or selective harvesting may be required. Enrichment planting involves the planting of seedlings within a forest where natural regeneration is poor or non-existent (d'Oliveira 2000; Parsons 2004). Typically, enrichment planting is utilised in sub-tropical and tropical forests to restore species that have been exploited by selective logging or grazing (Montagnini et al. 1997; Parsons 2004). It has been used successfully to establish desired timber and non-timber species in combination with shelterwood, nurse-tree, and selective harvesting systems (d'Oliveira 2000; Lozada et al. 2003). Enrichment planting has also been successfully used after single and group-tree selection harvesting in the temperate Rimu (*Dacrydium* *cupressinum*) forests of New Zealand. Enrichment planting can also be used following artificial or natural regeneration planting to fill in the gaps that result from disturbance or climate-based mortality. Enrichment planting may become a requirement in forest reserves and after fuel-reduction treatments or selective harvesting in vulnerable ecosystems. Enrichment planting can also be used to facilitate the persistence of species and ecosystems through "human-assisted migration", and be used to plant new species that are better adapted to the altered climate (Hogg and Bernier 2005). This approach can allow for a gradual and controlled transition to a more climatically adapted ecosystem.

One risk associated with both planting and seed-based regeneration that must be considered is the potential loss of seedlings due to browsing. The native swamp (black) wallaby (*Wallabia bicolour*) and the introduced European rabbit (*Oryctolagus cuniculus*), the most prevalent browsers of seedlings among others, can cause significant damage to regenerating eucalypt forests, with control and mitigation measures costing $800 to $1500/ ha (Poynter and Fagg 2005). These costs can be accrued in both planted and seeded areas; however the planting of seedlings is four times more expensive than seed-based regeneration (King 1997).

Our study has identified that stand modifying treatments offer reduced species vulnerability in their regeneration phase. By combining silvicultural treatments that ameliorate microclimatic conditions with the planting of seedlings that have an increased coping range, species vulnerability can be further reduced and resilience fostered.

5.6 Further Research and Development Needs

The parameterisation of the Central Highland species identified that significant knowledge gaps exist for the majority of the 22 species studied. Studies on species-specific responses are required to identify robust empirical biophysical thresholds that can be used to calibrate and model species with a greater degree of confidence and to validate the modelled species' responses. A balance of nursery and field based studies on species regeneration potential under different climatic conditions is a necessary requirement for increasing the confidence around the modelled species response to climate change. Further research will allow for both empirical model calibration and validation, which are necessary steps if we are to turn our understanding of species vulnerability into knowledge.

Possible areas for further investigation are as follows:

1. Primary and Secondary Dormancy:
    - Investigation of soil matric potentials and high temperatures that invoke secondary dormancy.
    - Determining the cold temperatures that prevent primary dormancy from being broken.
2. Drought and Temperature Thresholds:
    - Field-based studies that involve research plots being established along elevation transects. These would be used to test both germination and seedling responses to changes in temperature and precipitation at broad topographic-scales and at finer edaphic scales.
    - Field-based studies that allow species response to be tested under different stand/ disturbance treatments that have mediating influence on climatic conditions. This would allow for the identification of biophysical thresholds that prevent regeneration or reduce regeneration potential.
3. Lignotubers:
    - Field-based studies that investigate the potential of lignotubers to increase species resilience to climate change across both elevation and edaphic gradients.
    - Investigate sprouting potential and lignotuber viability.
4. Model Validation:
    - Field and laboratory testing will allow for TACA-OZ to be validated and refined in order to provide modeling outputs with higher degree of confidence.
5. Forest Succession:
    - Investigate species-specific competition under different climatic conditions to calibrate forest succession models. This will improve our understanding of what ecosystems may look like under the combined influence of climate, light and competition (fundamental versus realised distribution).
6. Policy and Societal Impacts:
    - Further consideration of the impacts of ecosystem change on the species

preferences in the timber industry, with greater exploration of commercialisation opportunities in mixed species forests.

- Evaluation of ways in which policies and regulatory tools can foster ecosystem resilience and adaptation.
- Further exploration of alternative silvicultural strategies in both protected and multiple-use forests.
- Evaluation of the impacts of future climate scenarios on sustainable timber harvesting levels.

6. Conclusion

A central component of sustainable forest management relates to our level of understanding regarding species and ecosystem vulnerability. In particular, it is not possible for forest management to adapt to new challenges if there is a poor understanding of what needs to be adapted and where. This research represents a first step towards understanding the implications of climate change for native forest management in Victoria. Mechanistic modelling was used to assess the vulnerability of 22 tree species in Victoria's Central Highlands in their regeneration niche to predicted climate change. Significant resilience was found to exist in the Central Highlands until 2040 when a threshold was reached. An increase in mean annual temperature of 1.4 °C and a decline in annual precipitation by 5 % were found to cause a significant contraction in 20 species regeneration niches and significant changes in the size and location of species potential ranges. The narrow climatic breadth of Australian species identified by Hughes et al. (1996) was realised in this study. Eighteen species were classified as extremely vulnerable to predicted climate change under a stand-replacing and maintaining treatment with the stand modifying treatment reducing species vulnerability. A significant increase in the climatic optimality of the damping-off fungus caused by *Pythium* spp. was detected in all treatments but was most prevalent in the stand modifying treatment. This could represent a possible feedback loop. All species were found to contract from lower elevations with all species able to regenerate above 1400 m in elevation by 2085, although regeneration potential declined for the majority of the species. Species responded in different ways to climate change, which may cause the eventual disassociation of current ecosystem assemblages and the creation of new ecosystems. The predicted climate change by 2040 may represent a threshold for loss of species and/or community resilience and ability to recover from future disturbances.

Management agencies can use this knowledge to incorporate adaptive and flexible actions into their regulatory framework to reduce species vulnerability and promote robust adaptation strategies based on understanding versus uncertainty. Ancillary to the identification of thresholds is the discovery of areas that could act as potential climatic refugia. The higher elevation areas of the Central Highlands provide the chance to manage vulnerable species with reversible and adaptive practices.

Our study was based on the premise that species are most vulnerable to changing climate in the regeneration phase. It has identified critical thresholds that may limit natural regeneration in the future. Under future climate scenarios, artificial regeneration following disturbances (in both protected areas and areas managed for timber production) may assist in maintaining species in their current locations.

Climate change is a stressor that threatens the persistence of mature communities through its influence on disturbance regimes. The renewal and sustainability of these communities can only be achieved by integrating an understanding of their vulnerabilities and knowledge of the thresholds where management actions can mediate or exacerbate species resilience. The Central Highlands are vulnerable to predicted climate change but management options exist that can reduce this vulnerability. In particular, adaptation actions that focus on fostering the ecological resilience of all species will, in turn, maintain their regeneration niche across a wider range than would otherwise be the case. Of course, many of the worst impacts identified in this study may not be realised if the global community can act to reduce greenhouse gas emissions and mitigate future changes in climate. We sincerely hope that this can be achieved.

7. References

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Appendix I

Summary of Change in Species Range Size by 2085

Table I-1: Change in species fundamental range size between current modelled period and 2085 modelled climate period under stand replacing treatment

Species	Current Niche (ha)	2085 Niche (ha)	Change (%)	Vulnerability
Silver Wattle	1110212	59683	-94.6 [I]	Extreme
Montane Wattle	1101709	145530	-86.8 [II]	Very High
Black Wattle	906961	369754	-59.2 [II]	High
Blackwood Wattle	1110212	698104	-37.1 [I]	Medium
Southern Sassafras	1110212	4464	-99.6 [I]	Extreme
Myrtle Beech	1110212	42601	-96.2 [I]	Extreme
Mountain Grey Gum	1101709	96664	-91.2 [II]	Extreme
Mountain Gum	1110212	19114	-98.3 [I]	Extreme
Alpine Ash	488127	8377	-98.3 [I]	Extreme
Broad-leaved Peppermint	1110212	96664	-91.3 [I]	Extreme
Tingaringy Gum	1110212	17561	-98.4 [I]	Extreme
Victorian Blue Gum	1108526	96664	-91.3 [II]	Extreme
Red Stringybark	1047541	369754	-64.7 [II]	High
Shining Gum	636498	1686	-99.7 [I]	Extreme
Messmate Stringybark	1110212	92646	-91.7 [II]	Extreme
Swamp Gum	1110212	4464	-99.6 [II]	Extreme
Snow Gum	885696	140	-99.98	Extreme
Narrow-leaved Peppermint	1110073	96664	-91.3 [II]	Extreme
Mountain Ash	1110212	26410	-97.6 [I]	Extreme
Candle Bark Gum	1110212	19240	-98.3 [I]	Extreme
Silvertop Ash	1110212	83570	-92.5 [I]	Extreme
Manna Gum	1110073	62672	-94.4 [II]	Extreme

I: range contracts at lower elevations
II: range contracts at lower elevations and expands at higher elevations
III: range disappears

Table I-2: Change in species fundamental range size between current modelled period and 2085 modelled climate period under stand modifying treatment

Species	Current Niche (ha)	2085 Niche (ha)	Change (%)	Vulnerability
Silver Wattle	1110212	96664	-91.3[I]	Extreme
Montane Wattle	996937	362183	-63.7[II]	High
Black Wattle	412108	999896	142.6[II]	Nil
Blackwood Wattle	1110212	999896	-9.1[I]	Low
Southern Sassafras	1110072	17688	-99.6[II]	Extreme
Myrtle Beech	1110212	257850	-76.8[I]	Very High
Mountain Grey Gum	996937	408558	-59.0[II]	High
Mountain Gum	1110212	38582	-96.5[I]	Extreme
Alpine Ash	999896	22102	-97.8[I]	Extreme
Broad-leaved Peppermint	1101709	488127	-55.7[II]	High
Tingaringy Gum	1110212	38582	-96.5[I]	Extreme
Victorian Blue Gum	1047541	408558	-61.0[II]	High
Red Stringybark	800680	999896	24.9[II]	Nil
Shining Gum	1110212	62672	-94.4[I]	Extreme
Messmate Stringybark	1090972	145530	-86.7[II]	Very High
Swamp Gum	1101709	17688	-98.4[I]	Extreme
Snow Gum	1110073	4464	-99.6	Extreme
Narrow-leaved Peppermint	1071630	488127	-54.5[II]	High
Mountain Ash	1110212	71747	-93.5[I]	Extreme
Candle Bark Gum	1108526	58653	-94.7[II]	Extreme
Silvertop Ash	1105622	145530	-86.8[II]	Very High
Manna Gum	1071630	268096	-75.0[II]	Very High

I: range contracts at lower elevations
II: range contracts at lower elevations and expands at higher elevations
III: range disappears

Table I-3: Change in species fundamental range size between current modelled period and 2085 modelled climate period under stand maintaining treatment

Species	Current Niche (ha)	2085 Niche (ha)	Change (%)	Vulnerability
Silver Wattle	1110212	42601	-96.2 [I]	Extreme
Montane Wattle	1105622	145530	-86.8 [II]	Very High
Black Wattle	945150	309533	-67.5 [II]	High
Blackwood Wattle	1110212	636498	-42.7 [I]	Medium
Southern Sassafras	1110212	1686	-99.8 [I]	Extreme
Myrtle Beech	1110212	38582	-96.5 [I]	Extreme
Mountain Grey Gum	1105622	71747	-93.5 [II]	Extreme
Mountain Gum	1110212	10057	-99.1 [I]	Extreme
Alpine Ash	369754	8504	-97.7 [I]	Extreme
Broad-leaved Peppermint	1110212	96664	-91.3 [I]	Extreme
Tingaringy Gum	1110212	10057	-99.1 [I]	Extreme
Victorian Blue Gum	1108526	71747	-93.5 [II]	Extreme
Red Stringybark	1067612	309533	-71.0 [II]	Very High
Shining Gum	537788	13647	-97.5 [I]	Extreme
Messmate Stringybark	1110212	71747	-93.5 [I]	Very High
Swamp Gum	1110212	1686	-99.8 [I]	Extreme
Snow Gum	996012	0	-100 [III]	Extreme
Narrow-leaved Peppermint	1110073	96664	-91.3 [II]	Extreme
Mountain Ash	1110212	37029	-96.7 [I]	Extreme
Candle Bark Gum	1110212	10057	-99.1 [II]	Extreme
Silvertop Ash	1110212	71747	-93.5 [II]	Extreme
Manna Gum	1110073	42601	-96.2 [II]	Extreme

I: range contracts at lower elevations

II: range contracts at lower elevations and expands at higher elevations

III: range disappears

Appendix II

Driving Variable Sensitivity Analyses for Individual Species

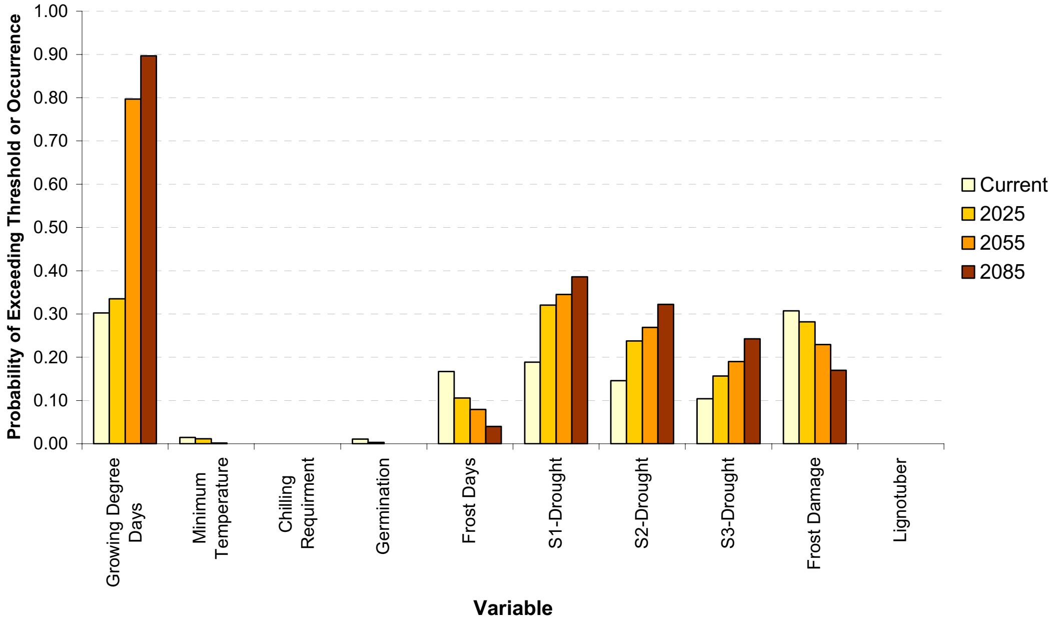

Fig. II-1: Sensitivity analysis of driving variables for Silver Wattle. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

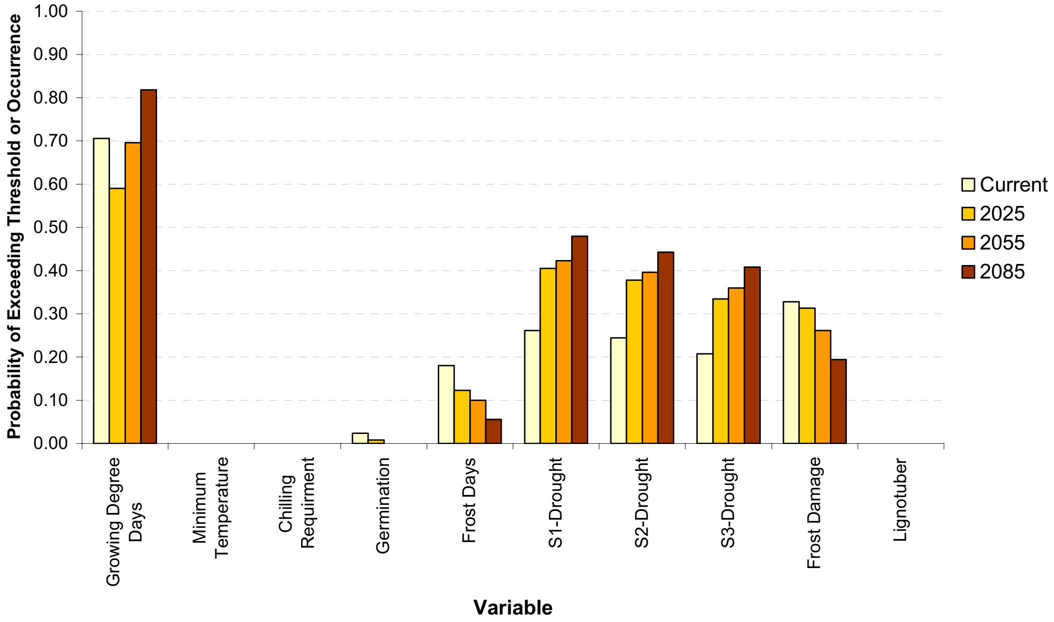

Fig. II-2: Sensitivity analysis of driving variables for Montane Wattle. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

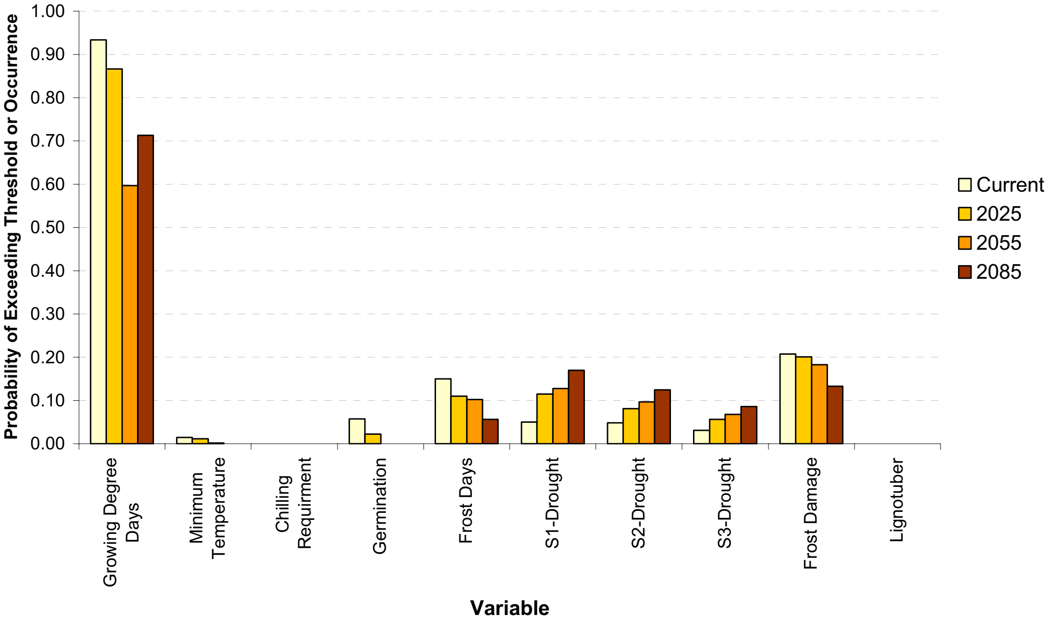

Fig. II-3: Sensitivity analysis of driving variables for Black Wattle. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

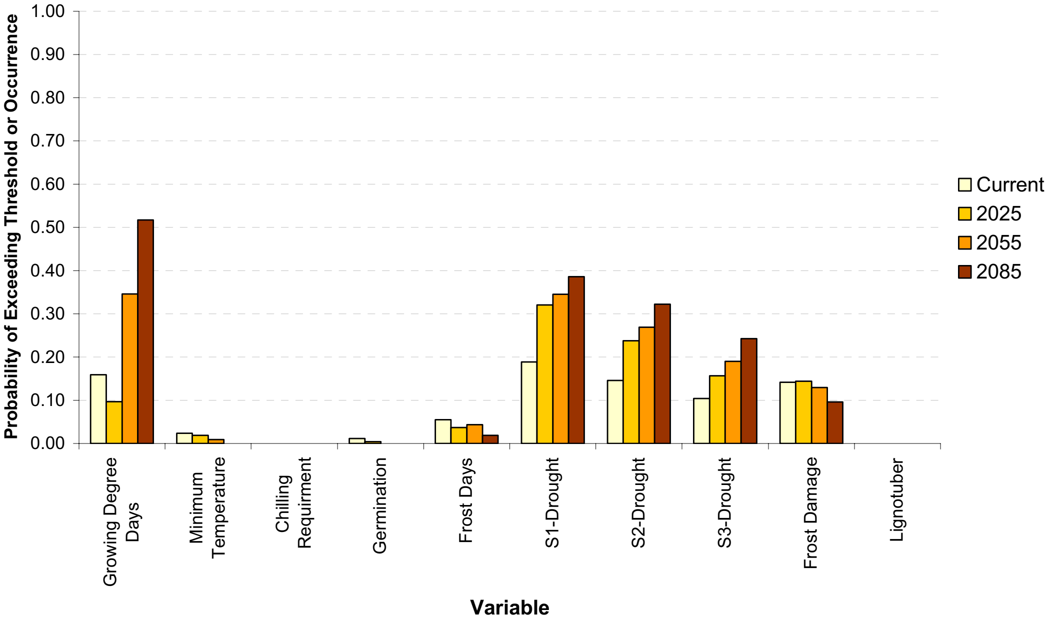

Fig. II-4: Sensitivity analysis of driving variables for Blackwood Wattle. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

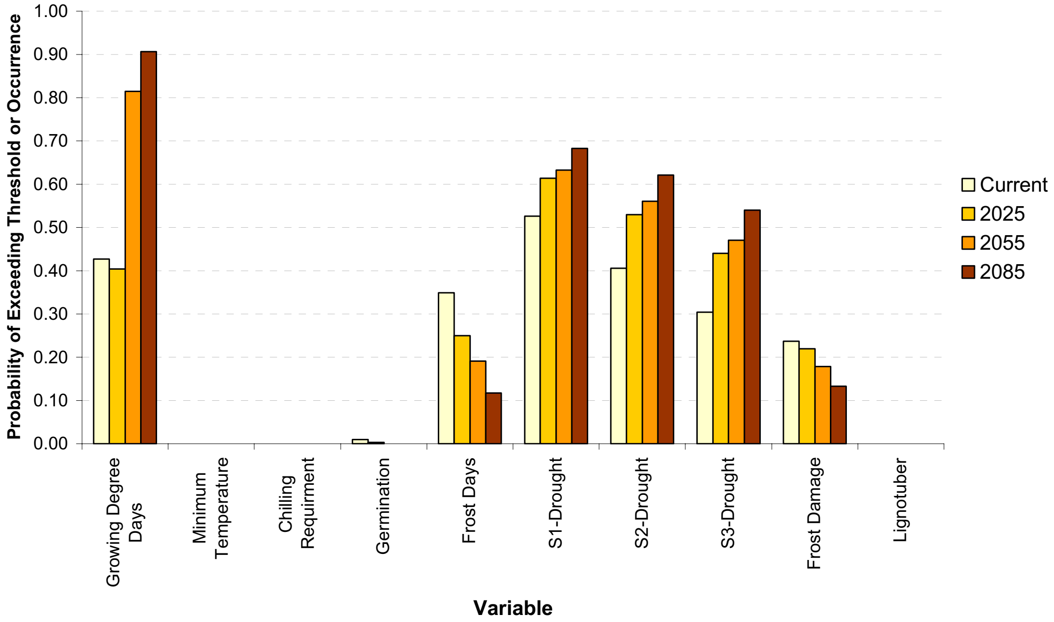

Fig. II-5: Sensitivity analysis of driving variables for Southern Sassafras. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

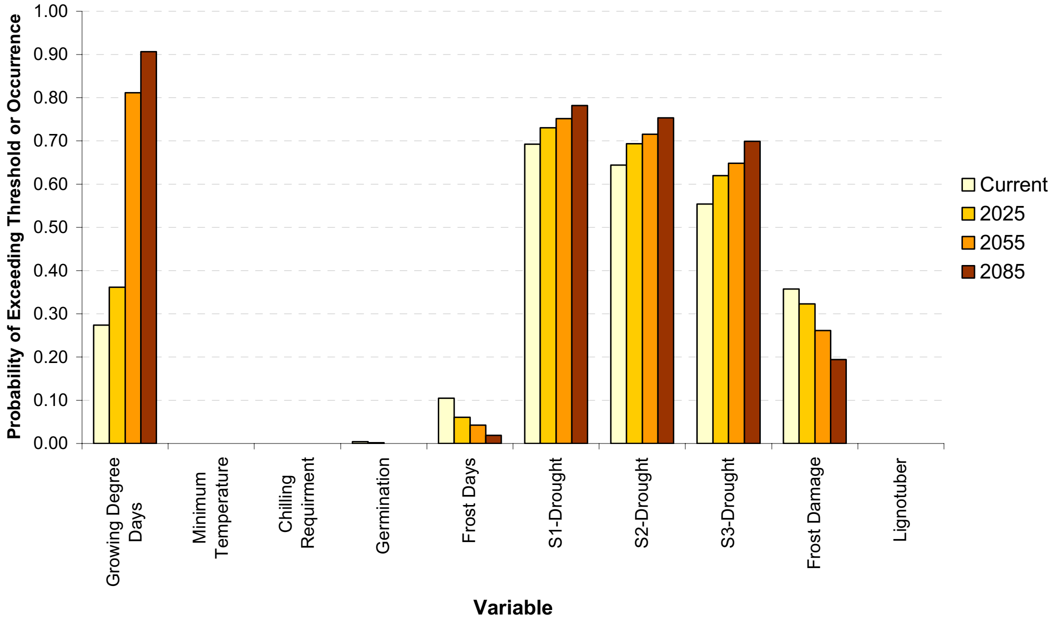

Fig. II-6: Sensitivity analysis of driving variables for Myrtle Beech. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

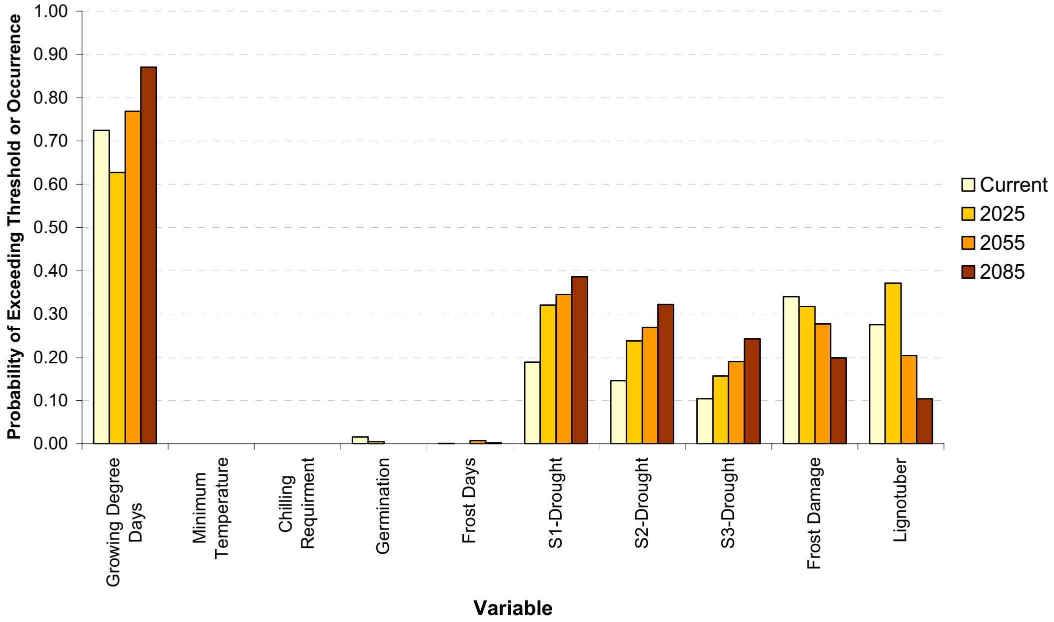

Fig. II-7: Sensitivity analysis of driving variables for Mountain Grey Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

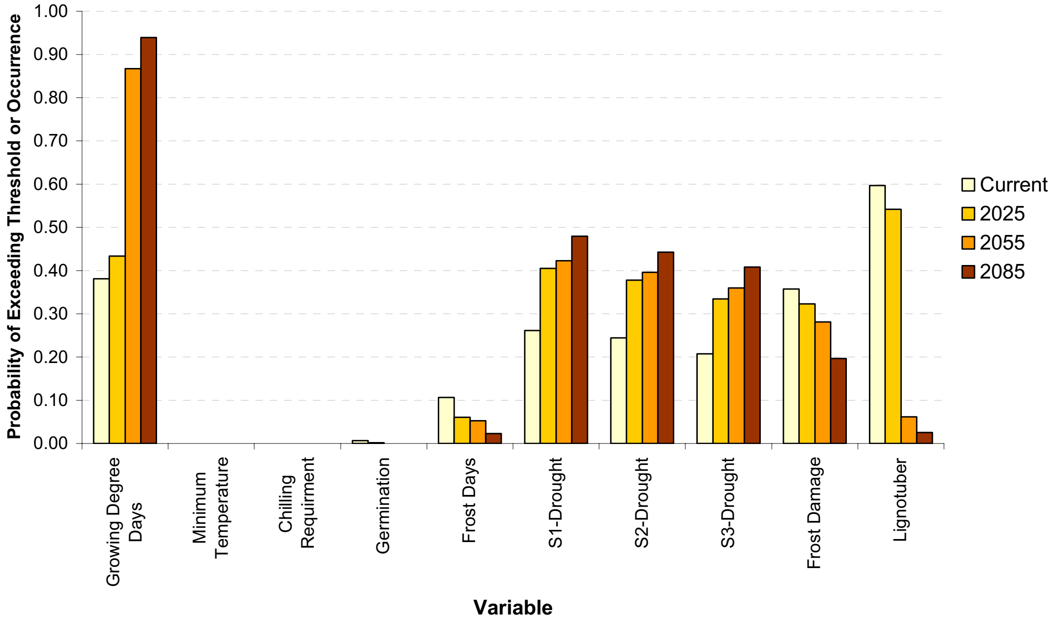

Fig. II-8: Sensitivity analysis of driving variables for Mountain Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

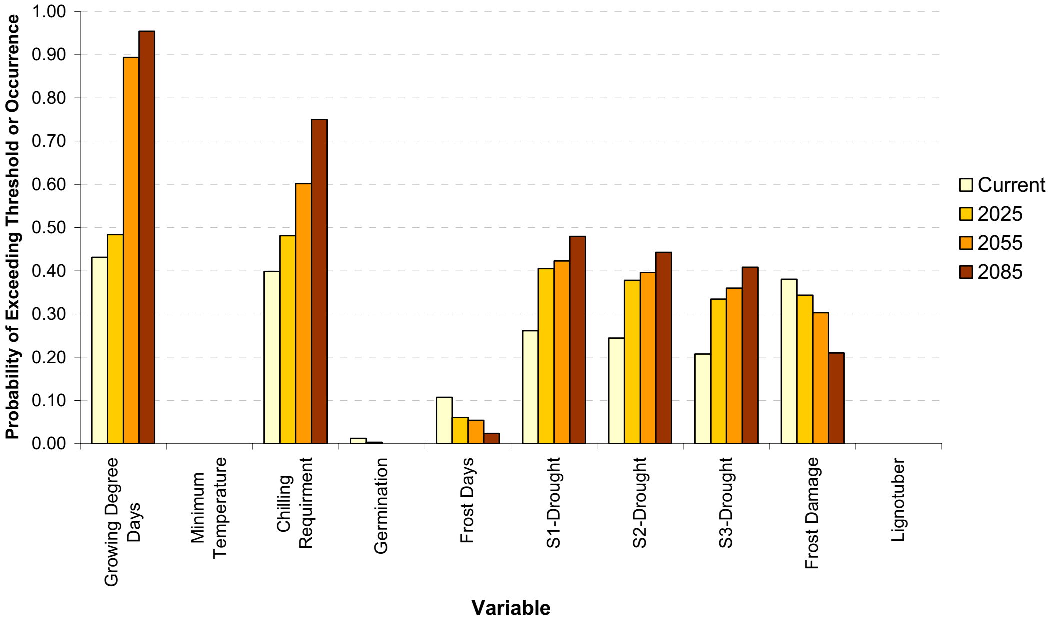

Fig. II-9: Sensitivity analysis of driving variables for Alpine Ash. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

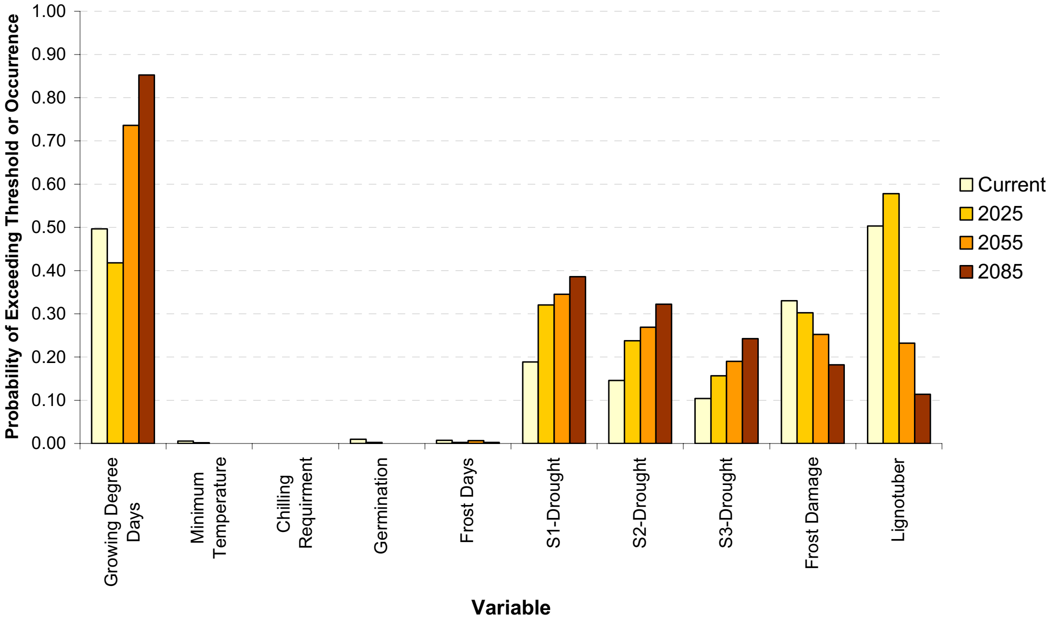

Fig. II-10: Sensitivity analysis of driving variables for Broad-leaved Peppermint. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

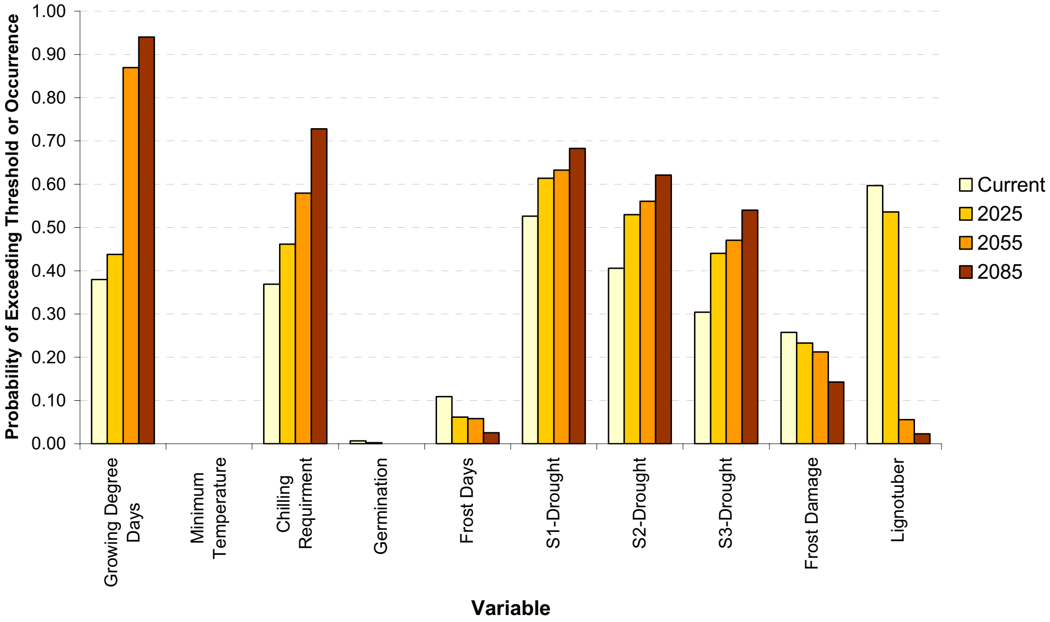

Fig. II-11: Sensitivity analysis of driving variables for Tingaringy Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

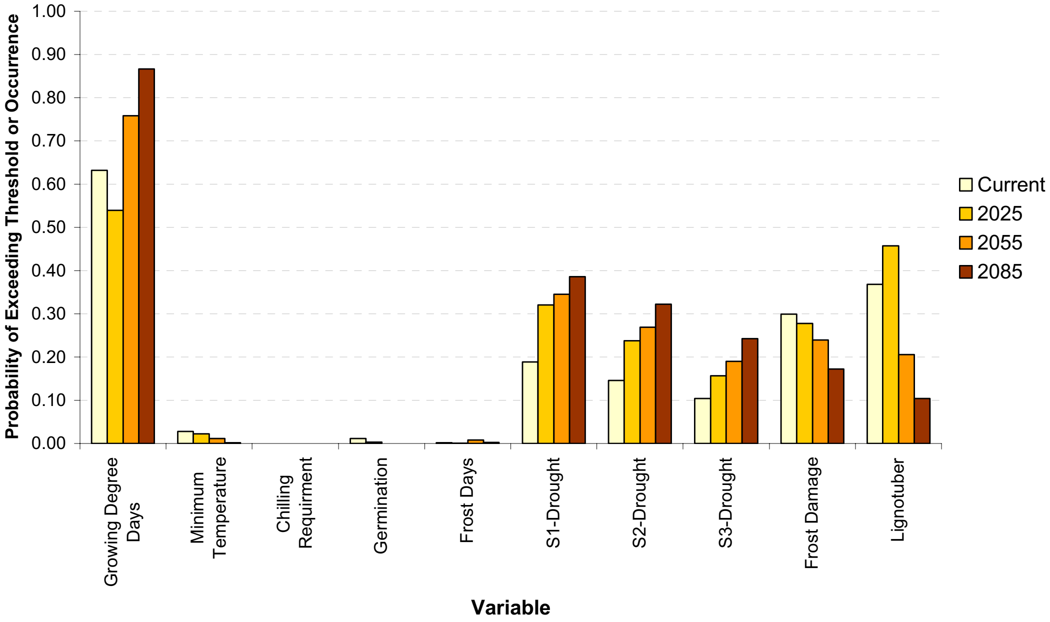

Fig. II-12: Sensitivity analysis of driving variables for Victorian Blue Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

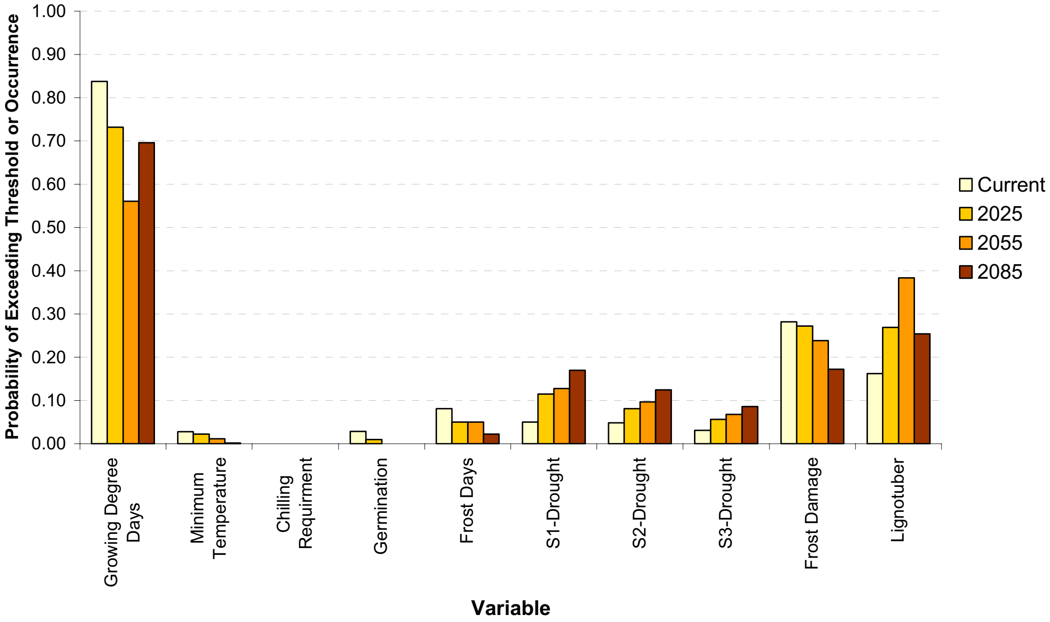

Fig. II-13: Sensitivity analysis of driving variables for Red Stringybark. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

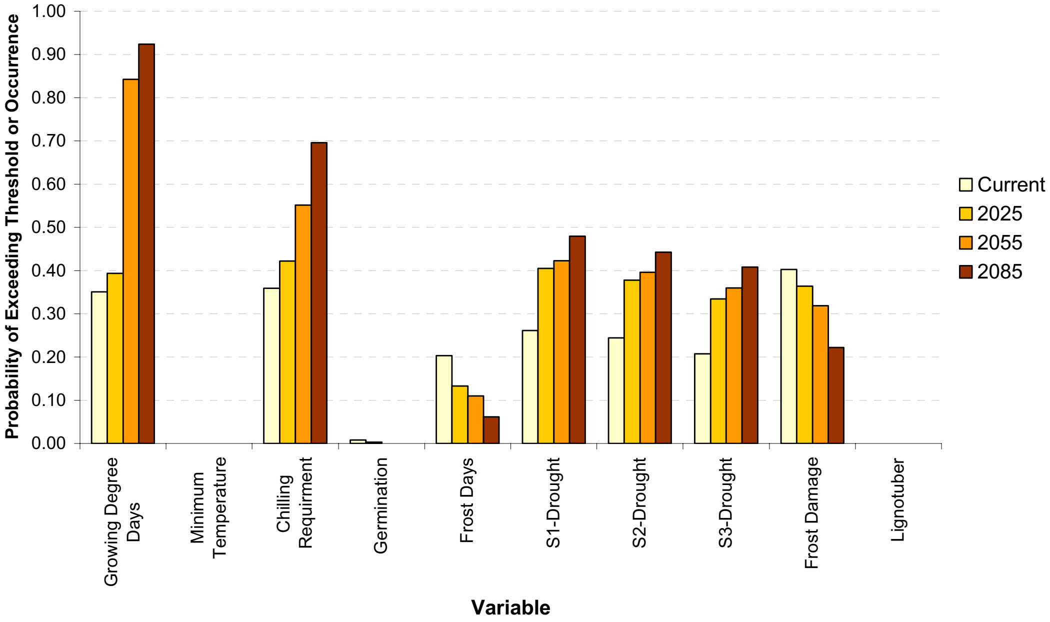

Fig. II-14: Sensitivity analysis of driving variables for Shining Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

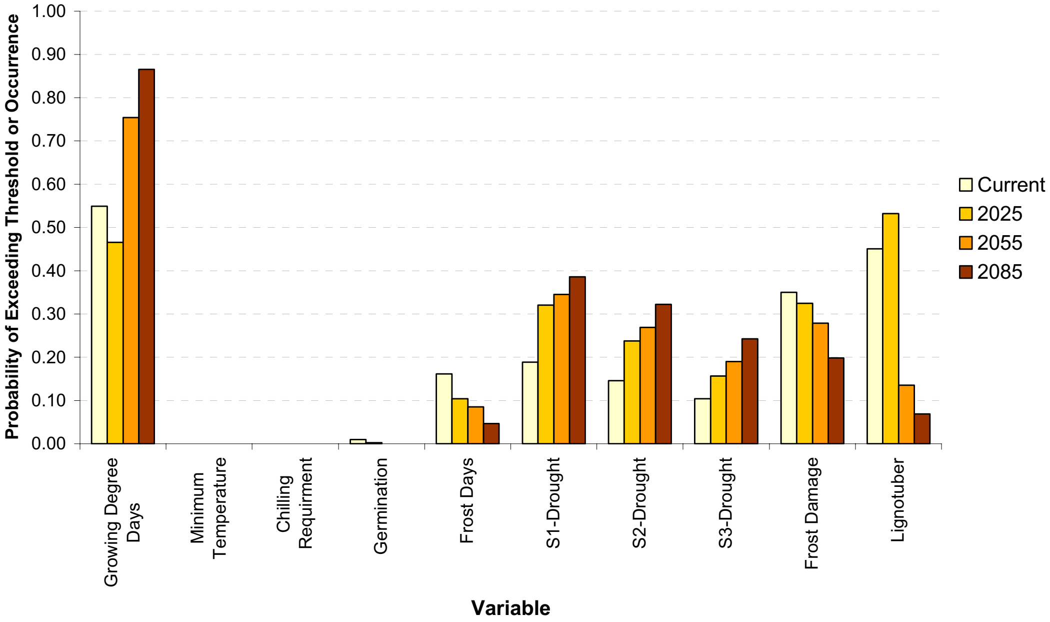

Fig. II-15: Sensitivity analysis of driving variables for Messmate Stringybark. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

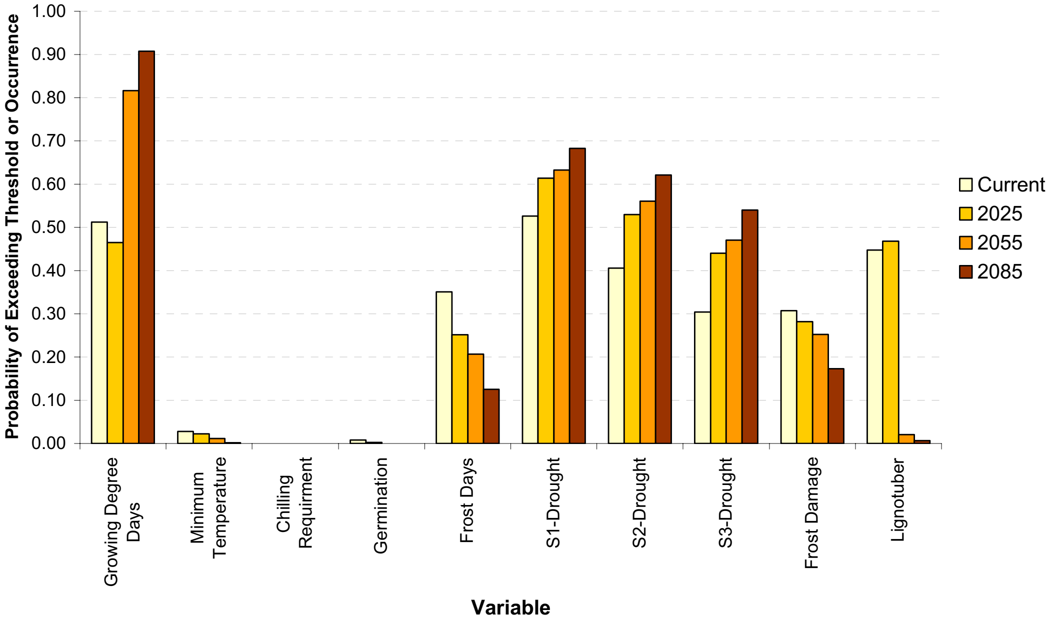

Fig. II-16: Sensitivity analysis of driving variables for Swamp Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

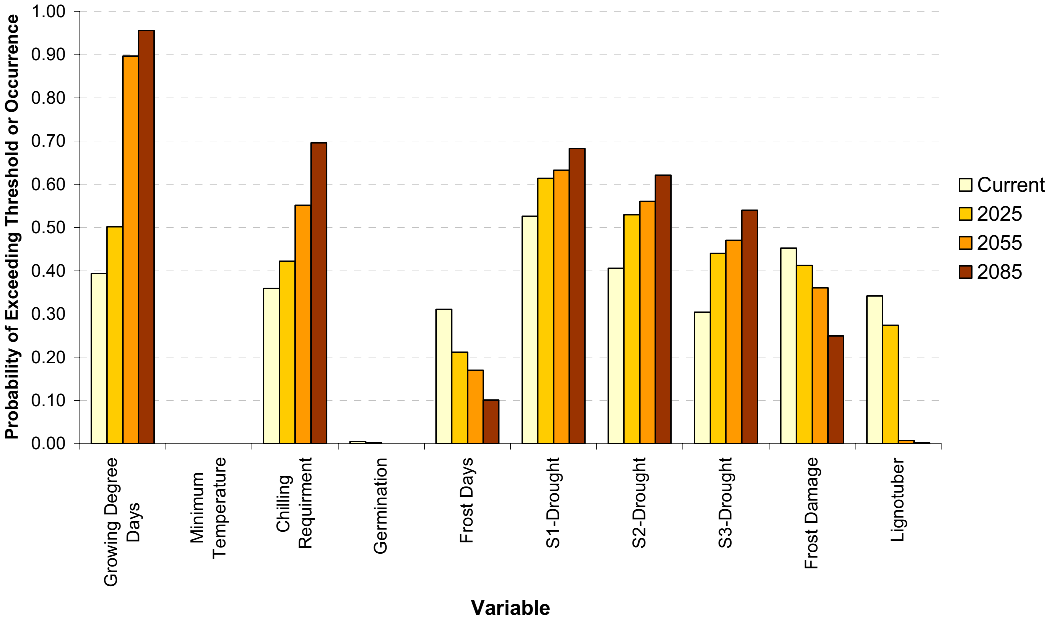

Fig. II-17: Sensitivity analysis of driving variables for Snow Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

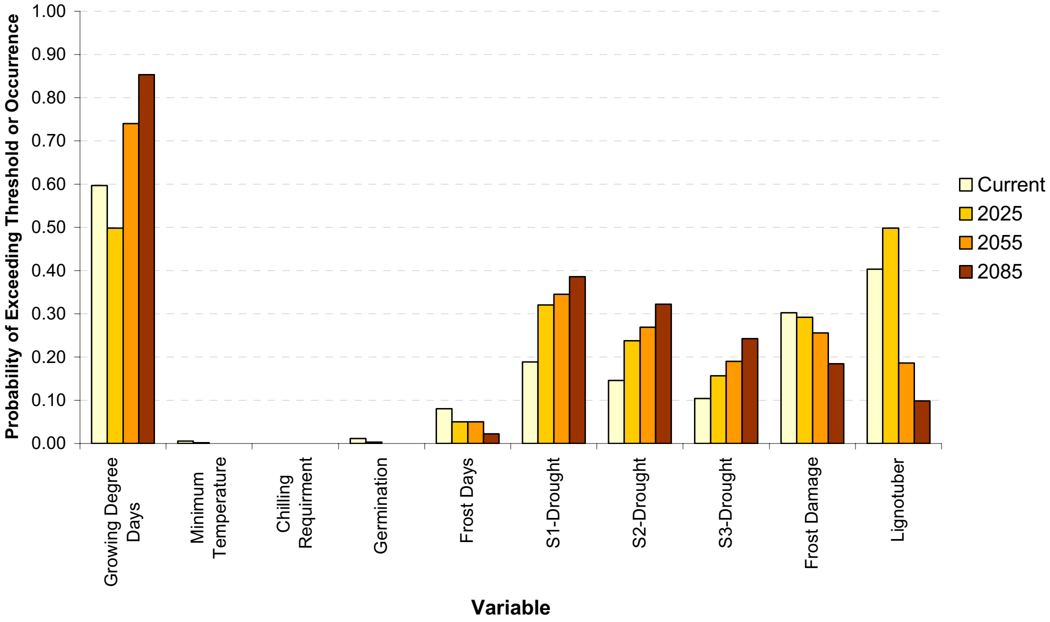

Fig. II-18: Sensitivity analysis of driving variables for Narrow-leaved Peppermint. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

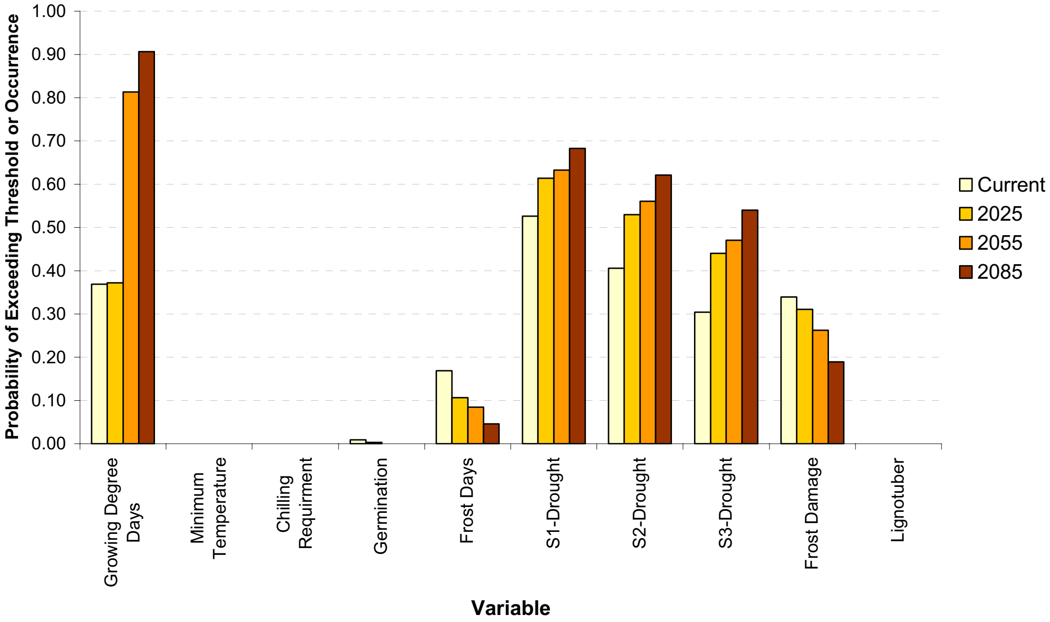

Fig. II-19: Sensitivity analysis of driving variables for Mountain Ash. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

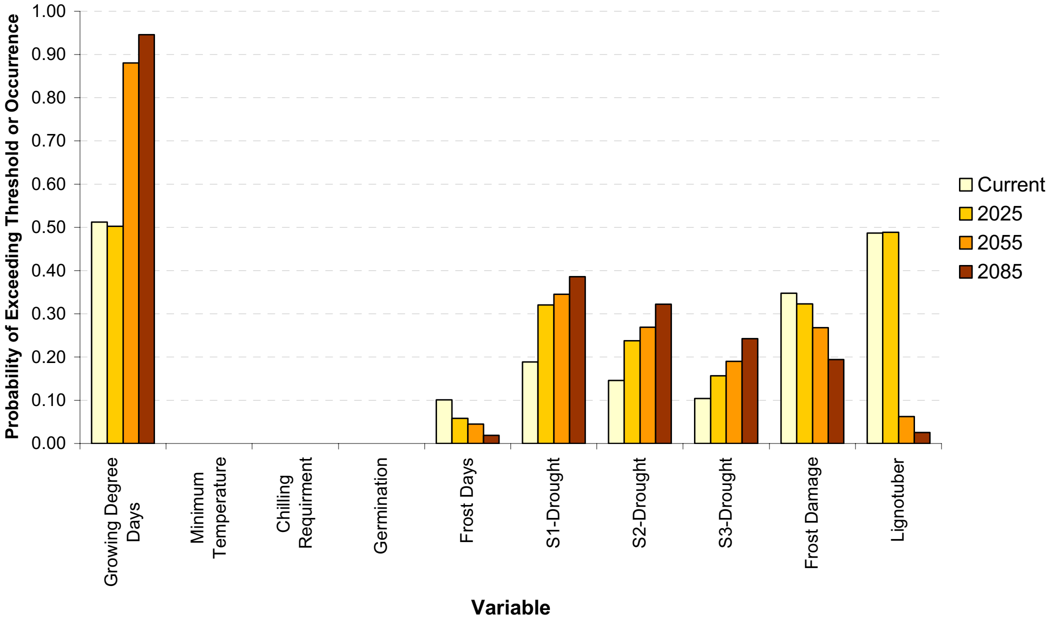

Fig. II-20: Sensitivity analysis of driving variables for Candlebark Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

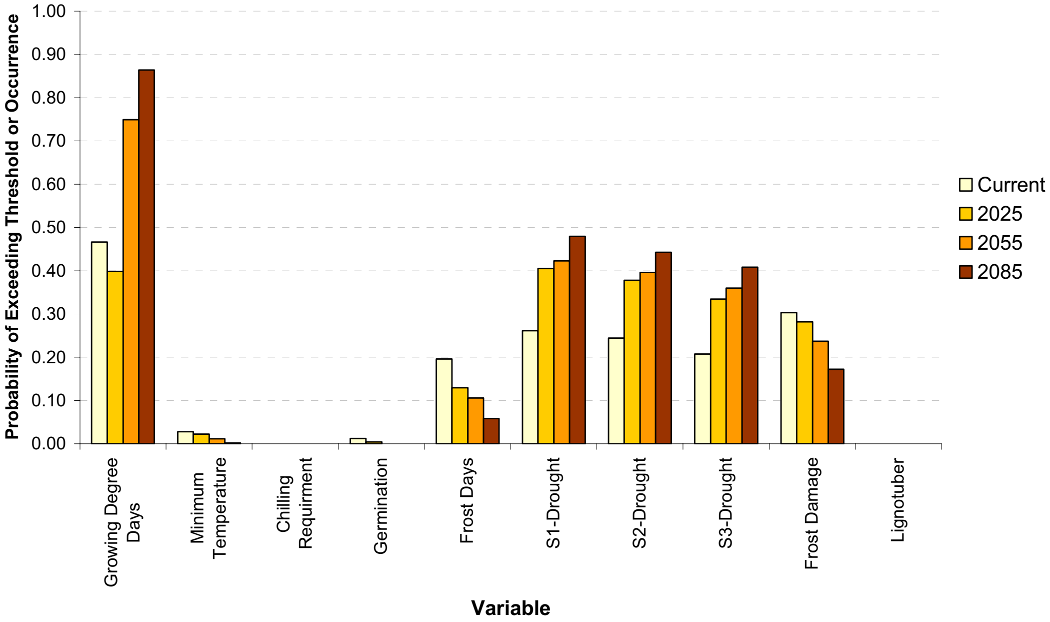

Fig. II-21: Sensitivity analysis of driving variables for Silvertop Ash. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

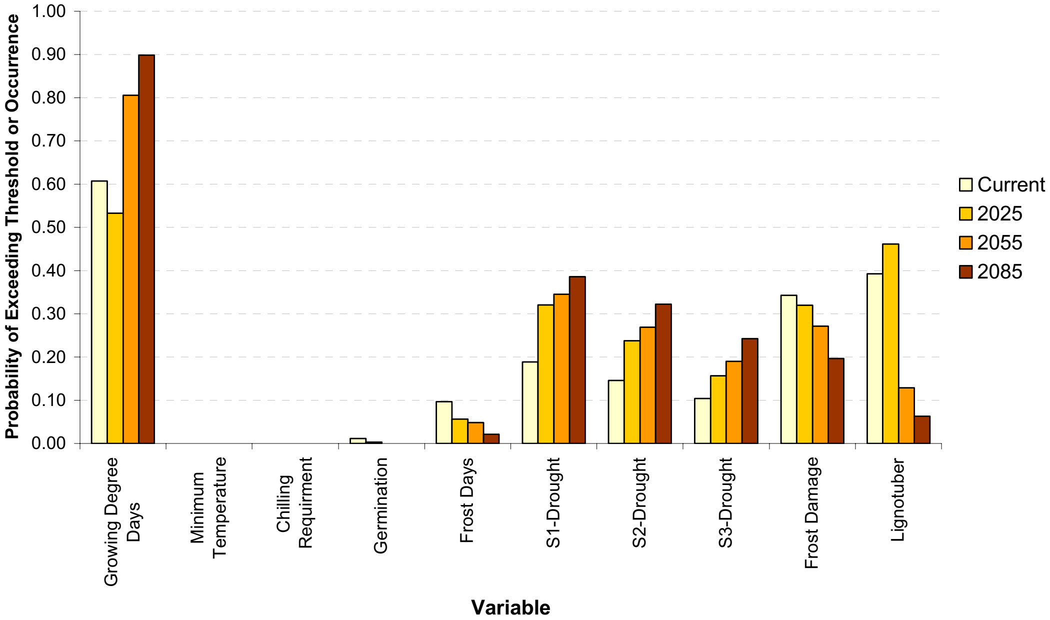

Fig. II-22: Sensitivity analysis of driving variables for Manna Gum. Frost Damage and Lignotuber illustrate probability of occurrence, the remaining variables illustrate probability of threshold being exceeded across all scenarios

Glossary

Biodiversity	Encompasses the diversity of indigenous species and communities occurring in a given region; includes genetic (genes/ genotypes within each species) diversity, species (variety of living species) diversity and ecosystem (different types of communities formed by living organisms and the relations between them) diversity.
Community	An ecological community is defined as a group of actually or potentially interacting species living in the same physical place. A community formed and maintained by the influences that species exert on one another.
Competition	The relative growth of trees as a consequence of limited availability of water, nutrient and light due to other neighbouring vegetation.
Cool Temperate Rainforest	An EVC dominated by myrtle beech (Nothofagus cunninghamii) and southern sassafras (Atherosperma moschatum) which form a continuous dense canopy up to 40m tall. Blackwood (Acacia melanoxylon) forms part of the canopy in some areas. Scattered emergent eucalypts may also be present. Occur in protected gullies, on sheltered slopes and along streams in the wetter, mountainous parts of the Central Highlands. Moisture and virtual absence of fire are key determinants in the distribution of cool temperate rainforest.
Damp Forest	A widespread EVC that occupies a range of sites on a variety of soils and aspects from 200 m to 1000 m in elevation. Understorey structure dominated by small shrubs, herbs and grasses and without a distinct cover of tree ferns. Messmate (Eucalyptus obliqua) and mountain grey gum (E. cypellocarpa) are the widespread dominants, although mountain ash (E. regnans), manna gum (E. viminalis), Silvertop ash (E. sieberi) and Eurabbie (E. globulus ssp. Bicostata) may dominate locally.
Ecological Vegetation Class (EVC)	The components of a vegetation classification system. They are groupings of vegetation communities based on floristic, structural and ecological features.
Ecocline	A shifting association of species or biotic communities along a climatic and/ or topographical gradient.
Ecosystem	All the organisms (including plants and animals) present in a particular area together with the physical environment with which they interact.
Ecotone	The transition zone between two adjoining communities.
Edaphic	Refers to plant communities that are distinguished by soil conditions rather than by the climate.
Fire Regime	The frequency, intensity, season and scale of fire in a given area over a period of time.
Hygric	A type of habitat characterized by decidedly moist or humid conditions.
Mesic	A type of habitat with a moderate or well-balanced supply of moisture.
Montane	The biogeographical zone in mountain regions located below the tree line with relatively moist, cool temperatures and dominated by evergreen trees.
Montane Wet Forest	An EVC that occupies the most protected, usually south-facing slopes and gullies where soils are deep and fertile and well drained. The canopy may grow more than 60 m, and consists of pure or mixed stands of alpine ash (E. delegatnesis) and shining gum (E. nitens). Manna gum (E. viminalis) and Tingaringy gum (E. glaucescens) may co-dominate in certain areas.
Niche	The ultimate unit of the habitat, i.e., the specific spot occupied by an individual organism. The ecological niche of a species is the functional role of the species in a community; the fundamental niche is the totality of environmental variables and functional roles to which a species is adapted; the realised niche is the niche which a species normally occupies, the regeneration niche is the optimal range of the fundamental niche.
Regeneration (noun)	The young regrowth of forest plants following a disturbance of the forest such as timber harvesting or fire.

Regeneration (verb)	The renewal of the forest by natural or artificial means
Resilience	The ability of a species/community to returns to its former state after disturbance.
Resistance	The ability of a species/community to avoid alteration of its present state by a disturbance.
Sclerophyll	Of trees, hard leaved (e.g. Members of the genus *eucalyptus* and *acacia*)
SFRI	Statewide Forest Resource Inventory program. A strategic level inventory of forest resources on State forests of Victoria
Shelterwood system	Used for harvesting and regenerating particular forest types that may not be suited to clearfell regime. It consists of the removal of a proportion of mature trees to allow establishment of essentially even-aged regeneration under sheltered conditions, followed by later felling of the remainder mature (seed) trees.
Silviculture	The theory and practice of managing forest establishment, composition and growth to achieve specified objectives.
State forest	As defined in section 3 of the Forests Act 1958. State forest comprises publicly owned land which is managed for the conservation of flora and fauna; for the protection of water catchments and water quality; for the provision of timber and other forest products on a sustainable basis; for the protection of the landscape, archaeological and historical values; and to provide recreational and educational issues.
Subalpine Woodland	An EVC which is occurs on slopes above 1200m with relatively free draining soils. This community forms a woodland or forest dominated by snow gum (E. pauciflora).
Succession	The progressive change of species composition within a stand over time. If left undisturbed this succession will continue to a climax where the species composition will remain largely unchanged.
Vulnerability	In the context of climate change, vulnerability can be characterised as biophysical vulnerability. The IPCC defines vulnerability as the degree to which a system is susceptible to, or unable to cope with the adverse effects of climate change. It is a function of the character, magnitude, and rate of climate variation to which a system is exposed, its sensitivity, and its adaptive capacity.
Wet Forest	An EVC dominated by the mountain ash (E. regnans), forming tallest forest in the Central Highlands. Occurring on the protected slopes of ranges, plateaux, and outlying hills, these sites tend to have abundant rainfall, deep, rich, well drained soils. Distinct cover of tree ferns in understorey.
Xeric	A type of habitat with a strong moisture deficit and where annual potential loss of moisture from evapotranspiration well exceeds the moisture received as rainfall.