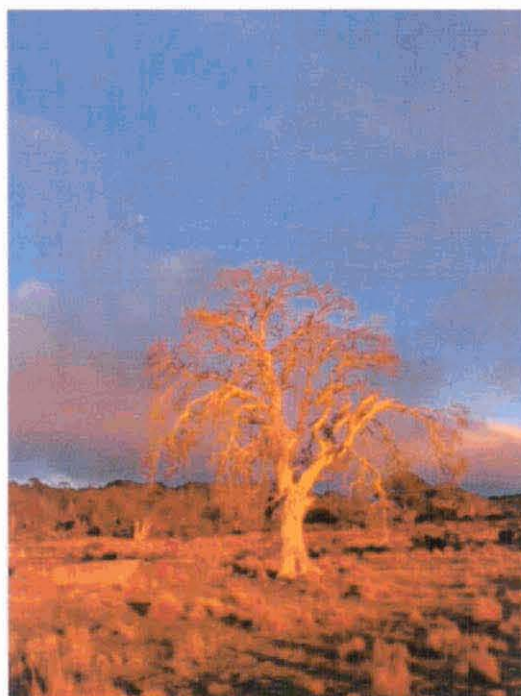


Spatial and temporal variation in the health of an endemic Tasmanian tree, *Eucalyptus gunnii* Hook.f.

By

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Bachelor of Science with Honours

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Declaration

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Signed

A handwritten signature in black ink, appearing to read 'jennycalder', written in a cursive style.

Jenny Calder Bsc.

Abstract

Eucalyptus gunnii Hook.f. is a tree endemic to the subalpine regions of Tasmania. A subspecies, *E. gunnii* spp. *divaricata* (McAulay & Brett) B.M. Potts, has recently been listed as endangered, due to extensive dieback over the last few decades. It has been suggested that the dieback may be in part due to increased intensity and frequency of droughts brought about by climate change. Examination of weather records indicated that a prolonged dry period has occurred across in much of the species' range. Mean annual temperatures have also risen. The timing of dieback, observed through a time series of aerial photographs, coincided with the timing of the worst droughts in some regions. However, field surveys of the health of *E. gunnii* across its range indicated a stronger negative relationship between livestock grazing and tree health than between rainfall and tree health. This may have been partly due to the long term suppression of regeneration in these populations, and therefore the predominantly old age of many of the trees, and their lower resilience to stress. Possum defoliation was also implicated as a major agent of decline. Furthermore, the possibility was raised that herbivorous insects are having a greater impact at higher altitudes than previously, which could be due to warmer winter temperatures. Although it was difficult to unravel the effects of climate and land management changes, the results suggested that we may already be seeing the impacts of climate change in Tasmania, especially in those environments which have been most heavily modified by land use.

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Chapter 1: Introduction

Large stands of dead and drying trees are a stark sight along the road to Cradle Mountain on the Middlesex Plains, and along the Lakes Highway in the Central Plateau of Tasmania. While dieback has affected numerous tree species across Tasmania, some of the worst deterioration in the subalpine environment has been in *Eucalyptus gunnii* Hook.f., a species commonly known as the cider gum. A recently defined subspecies, *Eucalyptus gunnii* ssp. *divaricata* (McAulay & Brett) B.M.Potts, (The Miena cider gum) has recently been listed as endangered under both State and Commonwealth legislation, due to significant dieback over the last few decades. In 2001 it was estimated that over 60% of the mature *divaricata* trees have died since the start of the 1990s (Potts *et al.* 2001), a figure which may have increased since then. Dieback of eucalypts has been recognised as a serious problem in Australia since around the early 1970s (Heatwole and Lowman 1986). Whilst dieback can have a complex aetiology, involving the interaction of many different stress factors, several authors have suggested that long-term climatic changes may be a contributing factor (e.g. Hepting 1963, Gardner 1996, Doyle 2003, Close and Davidson 2004).

The cider gum is endemic to Tasmania, and a distinctive and impressive-looking tree. The sweet sap produced by the species is a food source for native insects, birds and marsupials. Hollows in the mature trees also provide habitat for tree-dwelling mammals and birds (Potts *et al.* 2001). There is a recognised need to maintain genetic diversity across the range of a species (Duncan 1989). *E. gunnii* ssp. *divaricata* populations are some of the most frost-resistant eucalypts in the world and the taxon has been the subject of ongoing scientific study of evolutionary processes in Tasmania (Potts *et al.* 2001, Potts and Read 2003). The Threatened Species unit of the Department of Primary Industries and Water, Tasmania, the Bothwell Landcare Group and landowners in the Central Plateau

are implementing a recovery plan for the endangered subspecies.

This research project is an investigation into whether the dieback of *Eucalyptus gunnii* can realistically be attributed, at least in part, to climate change. The study aims to contribute information to aid in nature conservation planning for the endangered subspecies, and to provide further information on the implications of drought and climate change in Tasmania. If we are already seeing the impacts of climate change on Tasmanian species and communities, this has major implications for future vegetation management planning.

In order to investigate the relationship between climate change and dieback, chapter 2 will describe the environment and nature of the species, review the previous research on tree dieback in Australia, and discuss other studies which have linked tree decline to climate change. Chapter 3 will examine the climatic record from weather stations in the range of *E. gunnii*, to determine if the climate has changed in the species' habitat. Chapter 4 will discuss the spatial pattern and environmental correlates of tree health from 53 sites across the range of the *E. gunnii*. Chapter 5 will analyse the environmental correlates of regeneration success, and the implications of the size structure of populations. Chapter 6 will use a time series of aerial photographs to establish the timing of dieback in several populations. Finally, chapter 7 is a general discussion of the implications of the results, including conclusions and management recommendations.

Chapter 2: Background

2.1 *Eucalyptus gunnii* – natural and cultural significance

Eucalyptus gunnii is endemic to Tasmania, and was one of first eucalypts from the island to be described, by Sir Joseph Dalton Hooker. Sir Joseph, the director of the Kew Gardens from 1865, visited the Central Plateau during his 6-month stay in Tasmania, specifically to view the “cider tree” (Potts and Read 2003). The species gains its common name from reports that both Aborigines and stockmen would collect and drink sap, which, if left to ferment, had an intoxicating effect (Jetson 1989, Potts *et al.* 2001). It was described in 1858 as being 'amongst the hardiest as well as the handsomest of Tasmanian trees' (*The Mercury* 10.7.1858, cited in Jetson 1989).

Eucalyptus gunnii has a wide distribution across the State (figure 2.1). It grows in the north-western, central and north-eastern highland regions, and in several disjunct habitats in the Eastern Tiers and in the south (Williams and Potts 1996). The species' high tolerance to both frost and water logging, and its relatively slow growth rates, means it is mainly found in cold, waterlogged habitats, such as lake edges, or poorly drained valley flats. In the Central Plateau *Eucalyptus gunnii* is mainly found on the edges of frost hollows, between 900 and 1200 m in altitude. These depressions store cold air drainage, and thus experience intense frosts. The species' superior frost resistance has long been recognised. After Sir Joseph Hooker's visit to the Central Plateau, he reported on the “great frost of 1837”, which had killed large stands of other eucalypt species on flat ground, yet stands of *Eucalyptus gunnii* were almost uninjured (Potts and Read 2003).

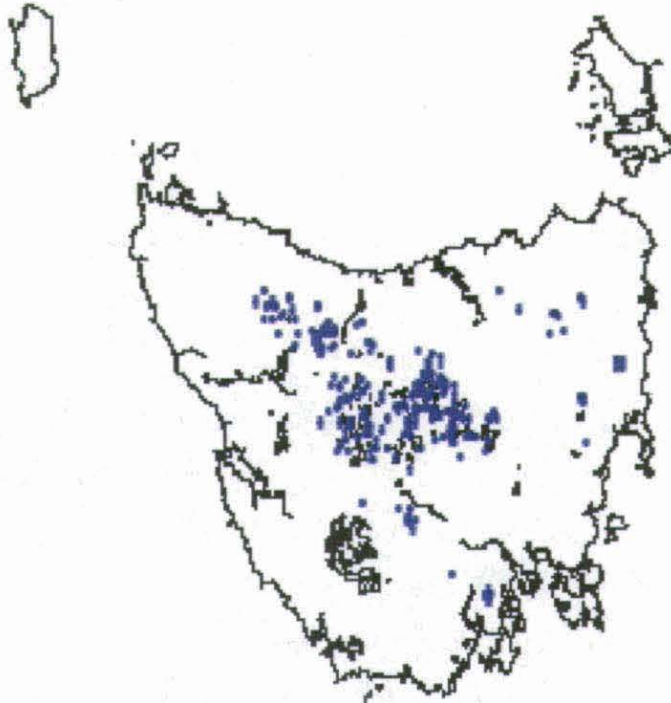


Figure 2.1: Distribution of *Eucalyptus gunnii* in Tasmania (from Tasmanian herbarium records).

E. gunnii most typically dominates grassy and sedgy open forest and woodlands. The small southern and far north-western populations occur at relatively low altitudes where they are associated with buttongrass plains and tussock grassland respectively. In restricted situations at lower altitudes *E. gunnii* can dominate tall montane forests with various shrub and rainforest understories, or co-exist with other subalpine eucalypts, such as *E. delegatensis*, *E. coccifera* and *E. dalrympleana* (Williams and Potts 1996). *Eucalyptus gunnii* primarily grows in the relatively fertile soils developed on an igneous parent material of basalt or dolerite. However it is quite capable of growing on peaty podzols, as evidenced by its inhabiting the edge of buttongrass plains and marshes (Potts 1977).

The taxa described as *E. gunnii* varies both phenotypically and genetically across the state. In the more southern populations the species is a small multi-stemmed

mallee, found on the edges of buttongrass plains (plate 2.1). In contrast, it occurs as a tall tree in the Central Plateau and northwest, and previously formed extensive woodlands (plate 2.2). A study conducted by Potts and Reid (1985) found that *E. gunnii* and the closely related species *E. archeri* represent extremes of a phenetic continuum, that, "internally encompass a diverse assemblage of morphologically differentiated populations". Potts and Read (1985a) recognised a total of five clinal morphs in the *E. gunnii* – *archeri* complex. These are: *E. archeri*, southern *E. gunnii* ssp. *gunnii*, north-western *E. gunnii* ssp. *gunnii*, *E. gunnii* ssp. *divaricata*, and localised intermediates (figure 2.2). Clines and intergradation between species are common in *Eucalyptus*. These continua are often interpreted as the result of primary intergradation and strong environmental gradients (Potts and Reid 1985). *E. gunnii* and *E. archeri* clinally intergrade on the Central Plateau along the north-south gradient in rainfall and frost severity. The main taxonomic character used to differentiate the two species is the degree of waxy glaucousness, a character that is associated with frost tolerance (Williams and Potts 1996). There is also an east-west gradient in exposure, which correlates with the clinal transition between *divaricata* and southern *gunnii* (Williams and Potts 1996).

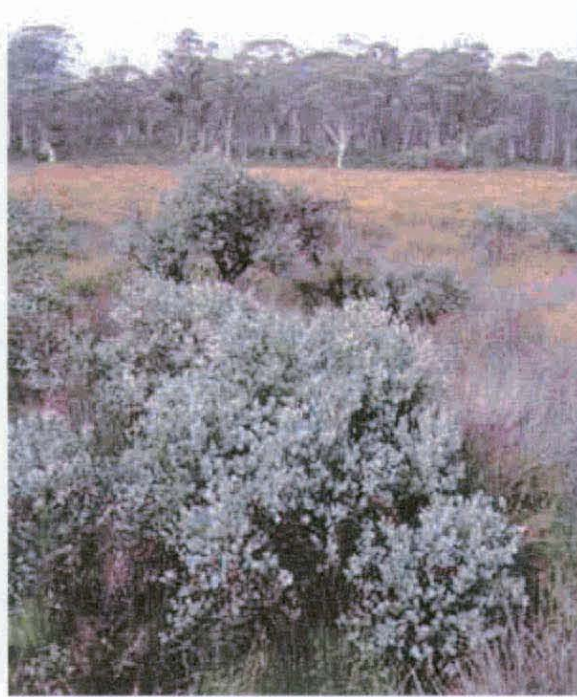


Plate 2.1: Small multi-stemmed mallee form of *E. gunni* growing near the Navarre River, near Lake St. Clair

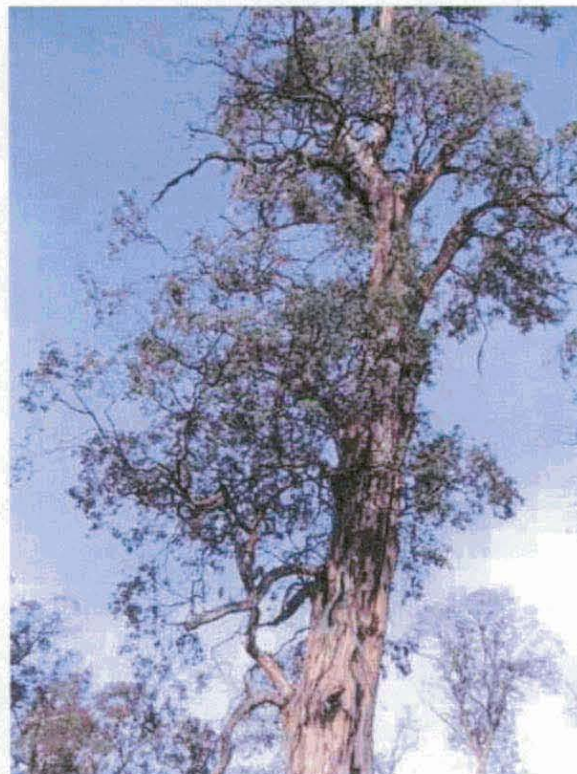


Plate 2.2: Largest tree found during field work, at Jack's Marsh in the Eastern Central Plateau

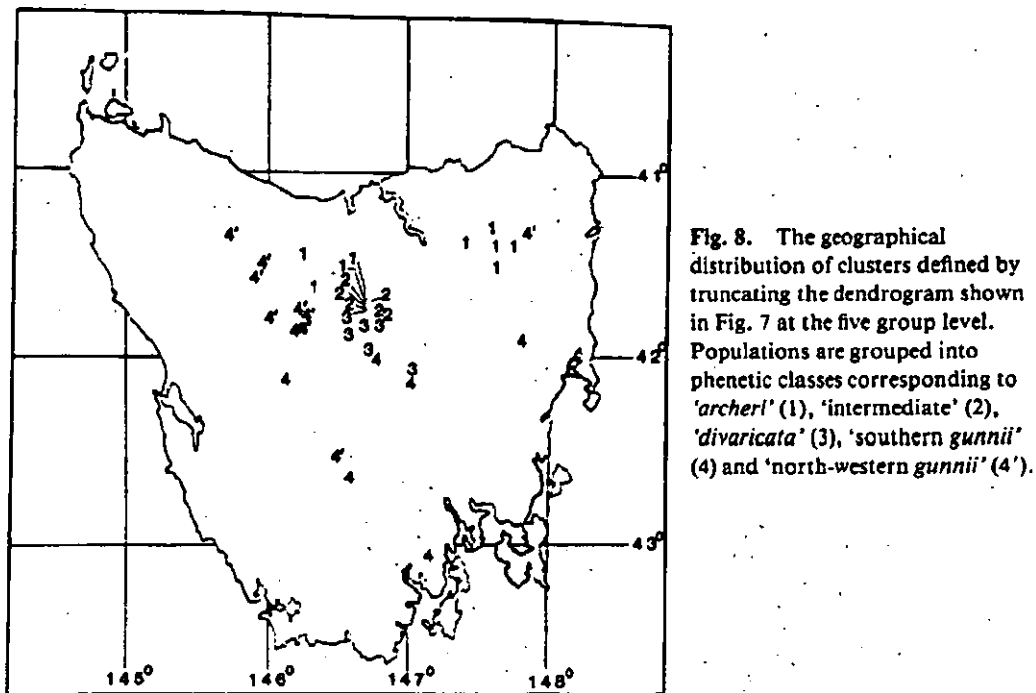


Figure 2.2: Distribution of clinal morphs in the *E. gunnii* – *E. archeri* complex identified by Potts and Read (1985a). (From Potts and Read 1985a).

Hardest hit by the dieback is the Miena Cider gum, *Eucalyptus gunnii* ssp. *divaricata*, a recently recognised taxon. Originally these populations were recognised as a separate species, *E. divaricata* by Brett (1938) then later subsumed into *E. gunnii* (Potts 1985). Its distinguishing features include smaller, broader leaves and a greater degree of glaucousness in the leaves and stems. Mature trees usually have more branched and rounder crowns than other forms of *E. gunnii*, hence the name, '*divaricata*'. This feature is likely to be partly due to the more exposed habitat. Core populations of the subspecies are also differentiated by up to 1.5 months in peak flowering time, which would act as a barrier to gene flow between the subspecies when they are in close geographical proximity. Shannon Lagoon, in the Eastern Central Plateau, is both the type location of the Miena Cider gum and the epicentre of the recent dieback (Potts *et al.* 2001) (plate 2.3):

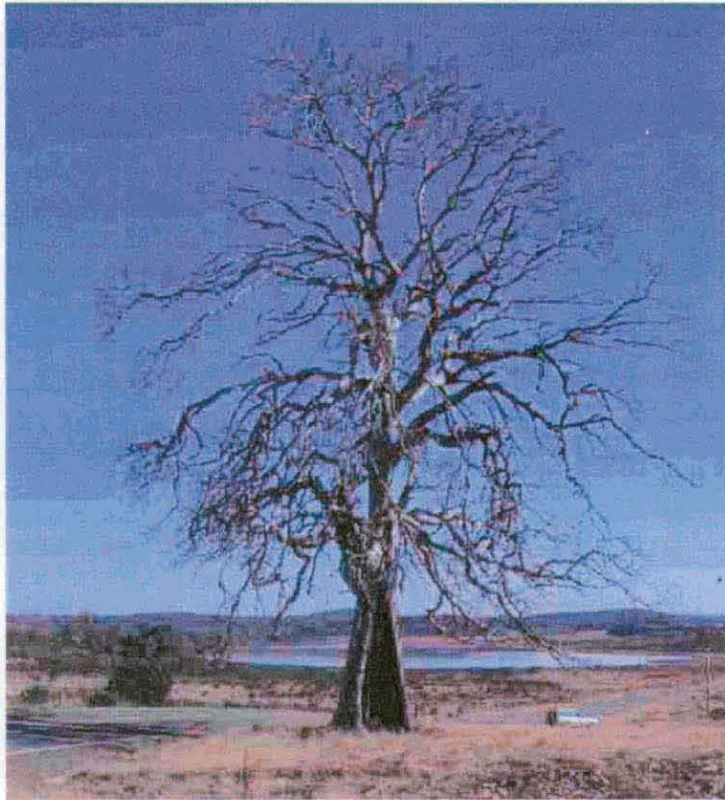


Plate 2.3: Typical *divaricata* form (and condition), at Shannon Lagoon.

2.2 Eucalypt dieback in Australia

“Sorting out the various causes is one of the most challenging ecological detective tasks of the century” (Hinrichsen 1987, on dieback).

Dieback, the term used to describe the apparently abnormal death of trees, has occurred in Australia for over a hundred years (Heatwole and Lowman 1986, Wylie *et al.* 1993). Some degree of dieback is normal, even in the healthiest forests, being “an adaptive response to the vagaries of weather and the activities of indigenous pests and diseases” (Podger 1973). However, since approximately the early 1970s, the extent, rate and severity of dieback have increased

dramatically across Australia. The decline has been most noticeable on private landholdings in open grazing country (Heatwole and Lowman 1986). Initially the problem was viewed as one of aesthetics or of primarily academic interest, but there has been a gradual realisation of the ecological and economic consequences of widespread tree decline, such as the loss of shelter trees in paddocks and the loss of habitat for native animals (Wylie *et al.* 1993, Heatwole and Lowman 1986).

Much research has been done in Australia to identify the causes of dieback. Some of the factors that have been investigated (Heatwole and Lowman 1986, Wylie *et al.* 1993, Kirkpatrick *et al.* 2000, Neyland 1999, Jurskis and Turner 2002, Low 2003, Doyle 2003, Close and Davidson 2004, Jurskis 2005) include:

- drought;
- long term climate change;
- herbivorous insect outbreaks;
- fungal disease such as *Phytophthora* and *Armillaria*;
- increased livestock numbers around paddock trees, which can compact the soil, reducing aeration and water penetration;
- stress on tree roots from increased nutrients from pasture fertilisation and livestock;
- competition for water with introduced pasture grasses;
- decreased diversity of mycorrhizal soil fungi, many of which facilitate the uptake of water and nutrients by the tree roots;
- waterlogging;
- salinity;
- land clearing, leaving remnant trees exposed to greater intensity of the elements, including wind and frost and soil drying, which can accelerate their decline;
- changed fire regimes;

- damage from increased vertebrate herbivore numbers, such as brush tail possums (*Trichosurus vulpecula*) and koalas (*Phascolarctos cinereus*);
- noisy miners (*Manorina melanocephala*) excluding other insectivorous bird species, thus increasing insectivory on the trees.

Nearly all studies have found that dieback is an exceedingly complex problem, with hardly ever a single cause. Many of the above factors can interact. Dieback can be progressive or sudden, and it is often the persistent action of not one, but a combination of factors that cause trees to malfunction in vital physiological processes beyond their recovery capacity (Podger 1981, Spencer 1984). Manion (1991) proposed a framework for understanding the decline of trees involving a combination of stress factors. “Predisposing factors” are long term stresses such as competition or air pollution that weaken a tree, and increase its susceptibility to subsequent stressors. An “inciting stress” is a relatively short duration stress such as drought or insect defoliation that substantially reduces tree vigour. Depending on the condition of a tree prior to the stress and the severity of that stress, a tree may or may not regain its prior vigour. Trees that do not recover are susceptible to “contributing factors” such as weak pathogens that further decrease vigour until death occurs.

2.3 Theories on the decline of *E. gunnii*

Potts *et al.* (2001) identified a series of factors believed to be contributing to the decline of *E. gunnii* ssp. *divaricata*. An obvious cause was the submersion of several populations when the levels of Great Lake were raised, and due to the clearing of trees on roadsides. Other factors include the old age of many trees and the lack of a younger cohort available to replace these trees. There has been low seedling recruitment in some populations due to grazing pressure from sheep, rabbits, native marsupials, and defoliation pressure from invertebrates. Drought

and frost in the open woodland environment have also contributed to a harsh environment for seedling establishment, and the increased isolation of flowering trees is likely to have increased the rate of self-fertilisation and hence reduced the fitness of the progeny. Increases in the numbers of brush tail possums (*Trichosurus vulpecula*) in highland areas is also implicated in the decline. It is further believed that the rapid decline of *Eucalyptus gunnii* ssp. *divaricata* is likely have been brought to head by severe droughts in the region over the last two decades. The decline of *E. gunnii* coincides with severe decline of other eucalypt species in the dry Midlands region of Tasmania. Extensive areas of subalpine forest in the Central Plateau dominated by *E. delegatensis* and *E. coccifera* have also suffered from significant crown decline, and are now recovering from epicormics. Potts *et al.* (2001) further suggests that long-term climate change may be responsible for the increased intensity and frequency of droughts in the area, contributing to the decline of *E. gunnii*.

2.4 Drought, dieback and climate change

Climate has long been recognised as one of the major agents determining the geographical distribution of species and communities on the planet, from a regional to a global scale (McCarty 2001, Hansen *et al.* 2001, Bakkenes 2002, MacIver and Wheaton 2005). The fossil record provides many examples of shifts in the geographical distributions of plant species in response to changing climates (Davis 1990). It is now generally agreed that the increase in carbon dioxide and other heat-trapping 'greenhouse gases' in the atmosphere, derived primarily from the combustion of fossil fuels, will lead to significant changes in the energy balance of the earth and, hence, regional and seasonal climatic patterns (Bakkenes 2002). Evidence that supports this prediction continues to mount. The average annual temperature has risen approximately 0.9°C in Australia since 1919 and the twelve warmest years on record in Australia have all been since 1990 (Bureau of Meteorology 2006) (figure 2.3). Other observed trends in Australia include

precipitation reductions in much of the eastern and south-western part of the country since 1950. However, these statistics are less convincing than those for temperature, partly due to the much greater temporal and spatial variability of rainfall (Pearman and Hennessy 2003). Perhaps most relevant is the combined effect of changes in precipitation and evaporation on the soil water supply (Hughes *et al.* 1996).

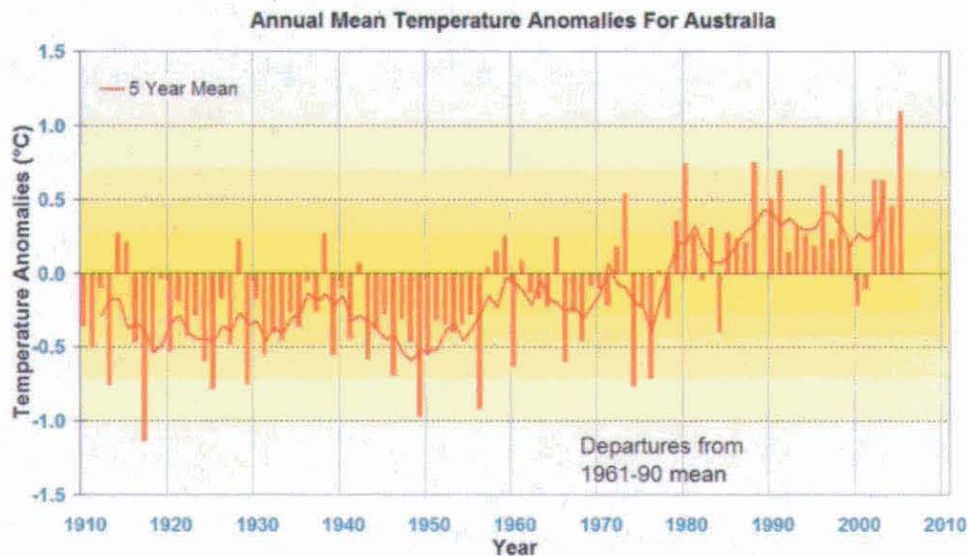


Figure 2.3. Annual Mean Temperature Anomalies for Australia 1910-2005. (From Bureau of Meteorology 2006).

Although a changing climate is not unprecedented in the history of the earth, and indeed in the history of the present biota, the rate at which the global mean temperature is projected to rise under future atmospheric conditions is faster than any known natural warming in the past (Kappelle *et al.* 1999). Additionally, the impacts of global warming will be imposed upon landscapes that have been dramatically altered by human activities (Kappelle *et al.* 1999). Under projected climatic change, it is likely that species ranges will shift. Extinctions will occur when species are unable to keep pace with the rate of climate change or are unable to migrate across highly fragmented landscapes (Hebda 1998, Noss 2001). The main concern is for those species which have 'nowhere to go'; for which

migration to a more amenable climate would mean migrating upwards into thin air or polewards into the ocean.

Several recent papers suggest that the warmer climate of the past century is already affecting some species in ways consistent with theoretical predictions. For example, Walther *et al.* (2005) has documented the coherent and synchronous shift both in increasing local winter temperatures and the northward shift of holly (*Ilex aquifolium*) in Europe. Other studies have documented recent changes in the distribution and the timing of events for plants; migratory birds and temperate-zone butterflies (Battisti *et al.* 2005). McCarty (2001) believes that the world is seeing remarkable consistency in the types and magnitude of changes observed across multiple studies. Although natural climatic variation or non-climatic factors might be responsible for some of these trends, human-induced climate change is the most parsimonious explanation for many (Hughes 2000).

Boardman (1994) predicts that trees, the tallest elements in an ecosystem, will be the first to show the impact of climate change, being the organisms with the most biomass to sustain. Tree life cycles take several decades to complete, and their dispersal rate is often slow, which limits their potential for short-term adaptation, and means they will be slow to respond to shifting climate zones by migration (Hughes *et al.* 1996, Solomon and Kirilenko 1997, Alewell and Bredemeier 2005). Solomon and Kirilenko (1997) predict that while little measurable tree migration and forest development will take place over the next 70 - 80 years due to reasons such as dispersal barriers, significant 'climatic obsolescence' and mortality of trees will take place. Drought-induced water stress can be responsible for some of the fastest vegetation responses (Martin 1996). Increased drought frequency and intensity may cause rapid, profound and long-lasting effects on forest and woodland ecosystems, especially if large numbers of dominant or key-structural species are killed (Suarez *et al.* 2004). Climate change can also have an indirect effect on species by influencing the distribution and abundance of pest

species, and changing fire regimes (Saxe *et al.* 2001).

At least two cases of forest dieback in Europe have been attributed to increased intensity of drought. Rebetez (2004) has monitored the dieback of large numbers of Scots pine (*Pinus sylvestris* L.) in the European Alps since 1995. Switzerland's temperature has increased at more than twice the global average in the 20th century, with most of this increase in the last 20 years. Tree mortality of Scots pine was highest following 1998, the driest, hottest year. Tree defoliation also showed a strong correlation with the previous year's precipitation. Climate change in southern Germany has also been greater than the global average. European beech (*Fagus sylvatica* L.), one of the region's most important forest trees, appears to have been affected by increases in the periods of drought in summer and heavy rain in autumn and spring. Reduced growth and competitiveness of the trees have been recorded in recent years, especially of seedlings at extreme sites (Rennenberg *et al.* 2005).

Differential responses of species to a long-term drying trend have been observed in patches of old-growth tropical forest in Panama (Condit 1998). The area has experienced an abrupt decline in rainfall of about 14% that began around 1966. Concurrently, a subset of species defined as 'moisture demanding' has declined markedly in abundance. One of these moisture specialists is large, prominent canopy species that had undergone a striking population crash of about 50%. The decline of moisture demanding species has been mirrored by an increase in the number of drought tolerant trees and shrubs.

There is also evidence in Canada that trees may be being affected by pathogens that have previously been unimportant (Woods *et al.* 2005). In British Columbia, decreased winter mortality due to warmer minimum temperatures has been implicated in the catastrophic outbreaks of the native mountain pine beetles.

Outbreaks occur almost exclusively in regions where it is warm enough for large numbers of larvae to survive the winter. They have had an enormous effect on the health of large forests of native pine, killing an estimated 300 million trees in British Columbia over the last 20 years. It is also possible that the pest will spread northwards with warmer temperatures (Dale *et al.* 2001, Carroll *et al.* 2004).

Climate change has been partly implicated in several cases of eucalypt decline in Australia. There are many cases where periods of below-average rainfall have coincided with increased incidence of tree decline (Neyland 1999, Close and Davidson 2002, Fensham *et al.* 2005). Decline of *Eucalyptus wandoo* woodlands in south-west Western Australia was attributed to a period of below average rainfall from 1997 to 2000, and drought stress was associated with a marked increase in the extent and severity of chronic eucalypt decline in coastal New South Wales during late 2002 (Close and Davidson 2004). Fensham and Holman (1999) documented extensive eucalypt death in northern Queensland savannah following three severe droughts in the 20th century. Other studies in the New England Tablelands have concluded that changed rainfall patterns have induced water stress in rural trees and increased their susceptibility to insect attack, leading to their decline. In the Midlands of Tasmania, the worst affected areas of rural tree dieback are those with the lowest precipitation (Neyland 1999, Close and Davidson 2004). Data from several stations in Tasmania indicate that there has been a sustained dry period in the state since the late 1970s. Growth rings of felled eucalypts from the Midlands also indicate declining growth rates since the early 1980s, which has been attributed to drought (Neyland 1999). Close and Davidson (2004) argue that these investigations support the contention that increased rural tree decline over the past two to three decades is related to climate change. However it is very difficult to singularly prove that climate change is the cause behind a documented biological trend, as many other natural and human-induced changes have occurred in these ecosystems.

Chapter 3: Has the climate changed in the range of *Eucalyptus gunnii*?

3.1 Introduction

There is strong evidence that climate in Australia has changed in the past century (Australian Bureau of Meteorology 2006). However, changes in temperature and rainfall patterns have not been uniform across the country. Some areas have become wetter, others drier, and some have undergone a greater degree of temperature change than others (Pearman and Hennessy 2003). To determine whether climate change can be implicated in the dieback of *Eucalyptus gunnii*, there must first be evidence for climate change in the region in which it is declining.

The research of Kirkpatrick and Gibson (1999) and Harris *et al.* (1988) would suggest that the warming in the Central Plateau of Tasmania has been greater than the Australian average. They reported a 1.5°C increase in mean maximum temperatures at the Shannon weather station in the Central Plateau between 1945 and 1995. This would have placed significantly higher transpiration stress on *Eucalyptus gunnii*, and has provided weight to the theory that climate change is contributing to the decline of the species (Potts *et al.* 2001). Work undertaken by other researchers does not contradict this. There is evidence of increased sea surface temperatures around Tasmania, of approximately 0.2°C over the last 50 years (McInnes *et al.* 2004). Tree ring records from Huon Pine (*Lagarostrobos franklinii*) on the west coast of Tasmania also indicate that the period since 1965 has been the most consistently warm of any period in the last 700 years (Cook 1991). The increased average annual temperature in Australia has mainly been influenced by rising minimum temperatures in the northern

half of the continent (Collins and Della-Marta 1999). A report undertaken for Hydro Tasmania by McInnes *et al.* (2004) found that in the period 1950 to 2002, Tasmanian trends in minimum and maximum temperatures have been the opposite to the Australian average. In Tasmania, maximum temperatures were found to have increased at about twice the rate of minimum temperatures (an average of 0.2 °C per decade compared to 0.1°C per decade). Allen (1998) found that the largest increase in temperatures has been across the north of the state. She found no evidence of temperature change in the south.

There has also been rainfall decline in some parts of Tasmania. McInnes *et al.* (2004) found that, since 1950, rainfall had decreased by 0 to 20 mm in the south-east and north-west of Tasmania, and increased by 0 to 30 mm in the south-west. However most of these trends were weak relative to natural decadal scale fluctuations (McInnes *et al.* 2004). Additionally, Allen (1998) identified a changing relationship between temperature and precipitation in Tasmania in the 20th century, similar to the Australia-wide trend detected by Nicholls (1996) and Collins and Della-Marta (1999). There has been less precipitation and higher temperatures. There is also evidence of a sustained dry period in the Midlands area of Tasmania, which has been implicated in rural tree decline (Neyland 1999, Close and Davidson 2002). Neyland (1999) combined data from stations at Avoca, Longford, Oatlands, Ouse, Swansea and Kempton, and found there had been at least twenty years of average and sub average rains, and no years of significantly above average rainfall since the mid 1970s. Kirkpatrick *et al.* (2000) also note that in the Midlands there has been a change from evenly distributed rainfall throughout the year, to a pronounced dry period in the later summer and early autumn. The increased definition of a dry season during the period of warmest temperatures would presumably increase the incidence of intense, short-term stress on trees.

This chapter will examine the long-term rainfall and temperature records for stations representative of the range of *Eucalyptus gunnii*. If climate change has contributed to the dieback, a long term drying trend should be detectable, either in decreased rainfall, increased temperature, or both. It will also examine the timing of the most intense short-term drought periods, to see if these may have been important in the recent dieback of the species.

3.2 Methods

Information on the location, altitude and period of record of all Tasmanian weather stations was obtained from the Bureau of Meteorology. All stations with less than 15 years of records were excluded. ARCVIEW GIS software was used to detect the closest weather stations to each field site (chapter 4). Daily rainfall and temperature records from the identified stations were obtained from the Bureau of Meteorology. After examination, further stations were excluded due to incomplete or discontinuous data sets (i.e. stations with no data or incomplete data for over a 10 year period within the record). The stations which were included are listed in table 3.1. Figure 3.1 is a map of the station locations.

For each rainfall station, the total rainfall was calculated for each month over the period of the record. Months in which any days had missing records were excluded. In order to smooth the inherent variation in the data, five year running means were calculated, using the average monthly rainfall for the five years leading up to each month. Line graphs were produced, which compared the five year running means to the long-term average monthly rainfall for each station. The total rainfall in the driest 3 months of the year (January to March), was calculated for each year, and plotted in a bar chart. The total rainfall of each season was also calculated and plotted. For each station, the total rainfall

of all consecutive 3-month periods was calculated. The years in which the ten driest 3-month periods on record occurred were identified.

For each temperature station, the average monthly minimum and maximum temperature were calculated. Months which had more than 5 days missing of missing data were excluded. Five year running means were calculated for most stations. At stations with only short periods of continuous data, 1-year running means were used. A record of at least 50 years is needed to confidently differentiate long-term trends from short-term fluctuations in climate (M. Nunez *pers. comm.* 2006). There are very few long-term temperature records with reliable data in the region where *E. gunnii* grows. The only reliable long-term temperature data sets that spanned more than 50 years were from the Hobart, Launceston and Low Head stations. Grove and Erriba also provided 48 years of records. However, temperature trends are less variable over space than rainfall (Pearman and Hennessy 2003). Due to the short period of record at most temperature stations, the trends from each station were compared to the trends from the Hobart station, which had the longest continuous period of record. If the pattern of increasing and decreasing temperature for that period was similar to that of Hobart, it was assumed that the long-term temperature trends that occurred at Hobart may also have occurred at that station.

Station name	Station number	Period of rainfall record	Mean annual rainfall	Period of temperature record	Mean annual maximum temperature	Mean annual minimum temperature
Hobart	94030	1885 - 2005	617mm	-	-	-
Victoria Valley	95021	1934 - 2005	838mm	-	-	-
Lake Leake	92019	1890 - 2005	833mm	-	-	-
Fingal	92012	1888 - 2005	609mm	-	-	-
Waddamana	96023	1925 - 2005	813mm	-	-	-
Steppes	96042	1965 - 2004	657mm	-	-	-
Miena Dam	96046	1889 - 2005 (but record incomplete until 1970)	826mm	-	-	-
Tarraleah Villa	95018	1935 - 2005	1145mm	-	-	-
Liawenee HEC	96004	1919 - 1991	1006mm	-	-	-
Liawenee comparison	96065	1984 - 2005	1050mm	1985 - 2003	12°C	1.2 °C
Butlers Gorge	96003	1941 - 1998	1688mm	1957 - 1993	12.8 °C	2.9 °C
Erriba	91119	1957 - 2000	1595mm	1962 - 2000	13.1 °C	4.7 °C
Shannon HEC	96021	-	-	1957 - 1985	10.8 °C	2.1 °C
Sheffield	91091	-	-	1965 - 1985	16 °C	6.4 °C
Hobart	94029	-	-	1882 - 2005	16.8 °C	8.3 °C
Grove	94069	-	-	1957 - 2005	17.1 °C	5.9 °C
Launceston	91104	-	-	1939 - 2005	16.9 °C	6.2 °C
Low Head	91057	-	-	1895 - 2001	16.2 °C	9.5 °C

Table 3.1: Duration of record and average values from rainfall and temperature stations

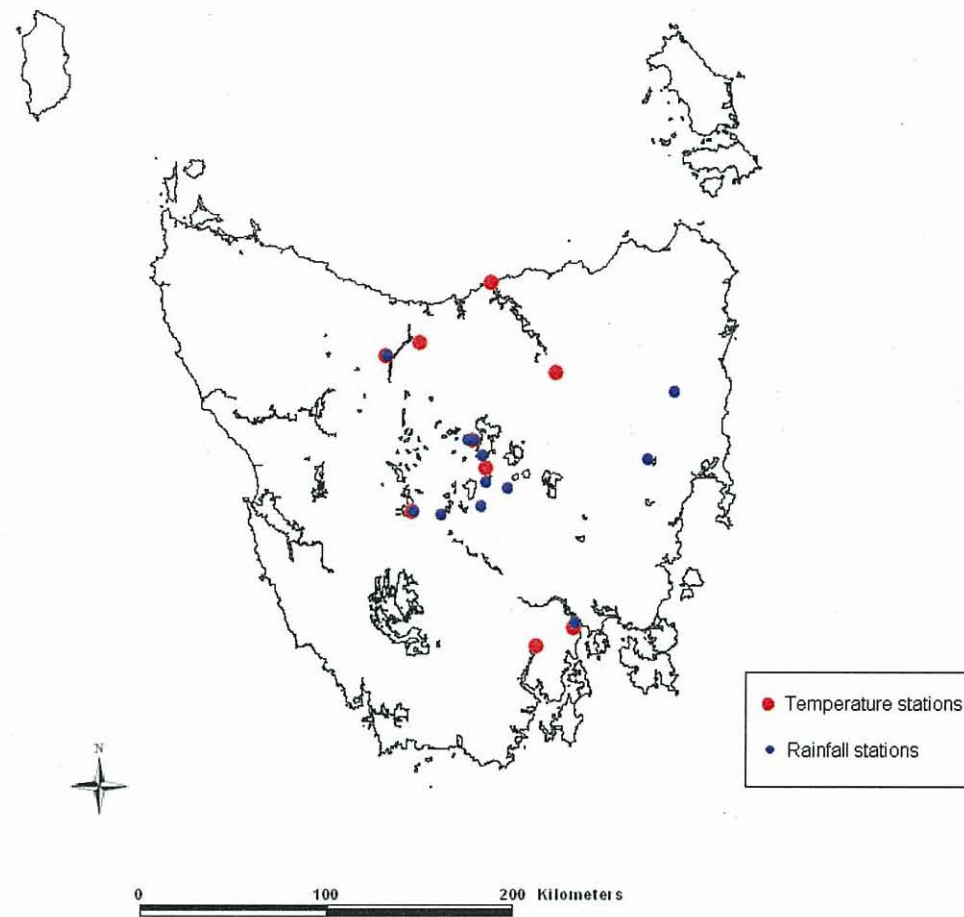


Figure 3.1: Locations of rainfall and temperature stations

3.3 Results:

3.3.1 Rainfall

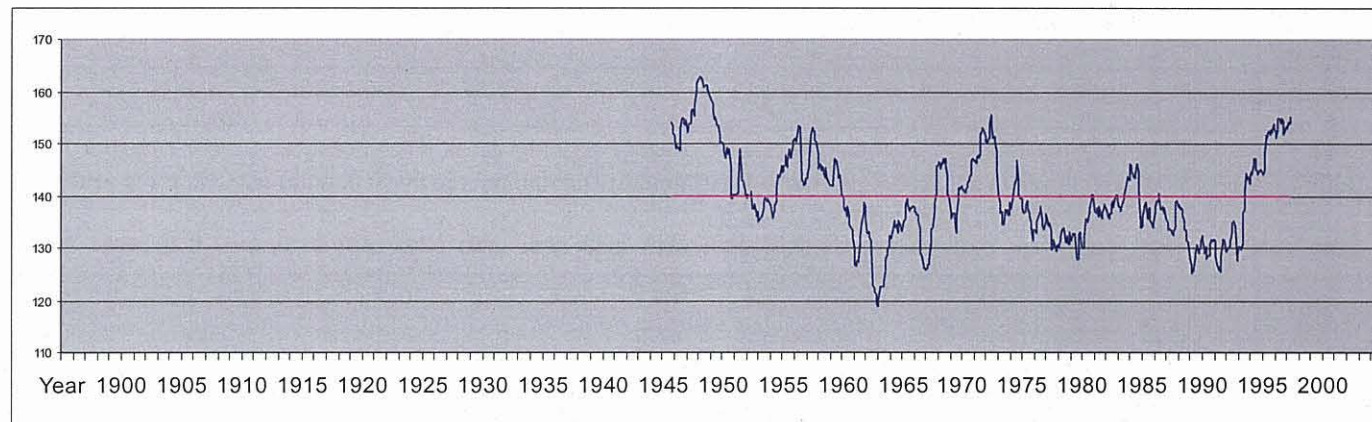


Figure 3.2: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Butlers Gorge

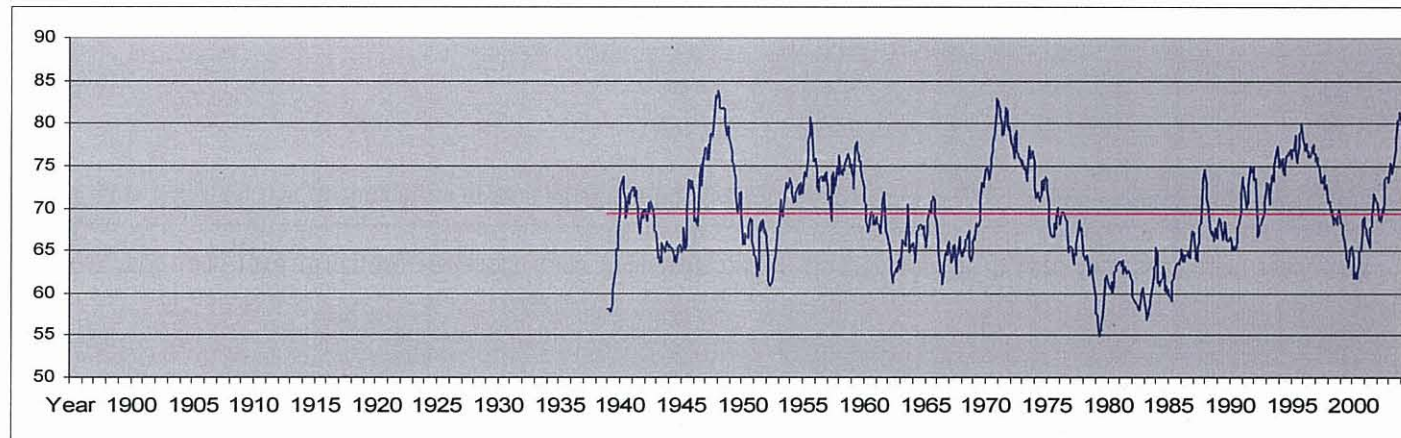


Figure 3.3: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Victoria Valley

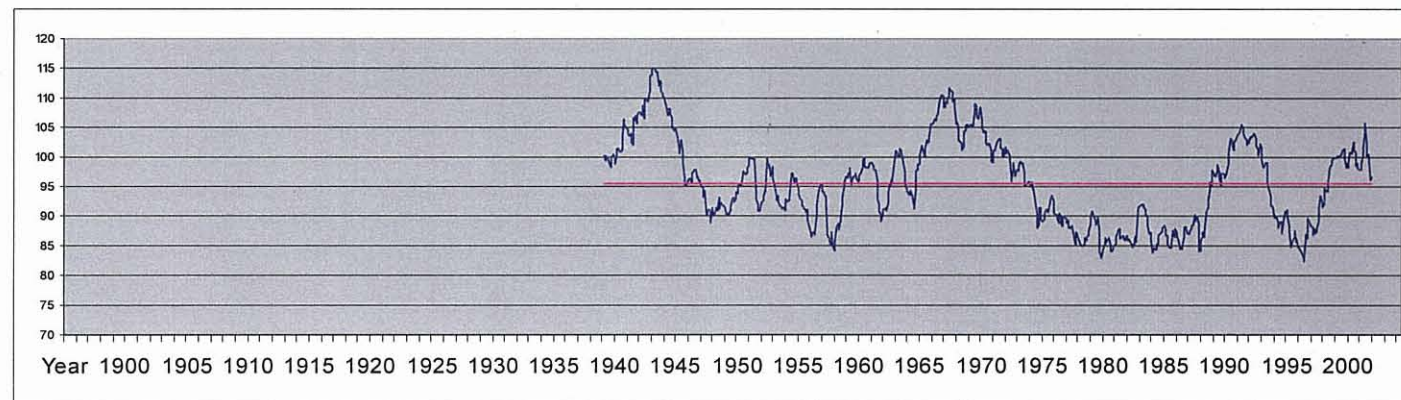


Figure 3.4: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Tarraleah

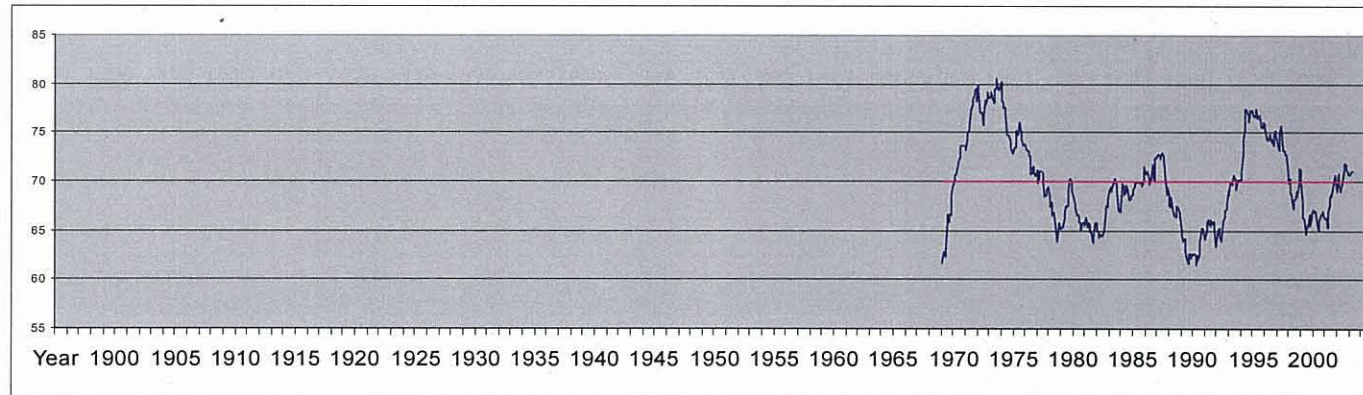


Figure 3.5: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Miena Dam

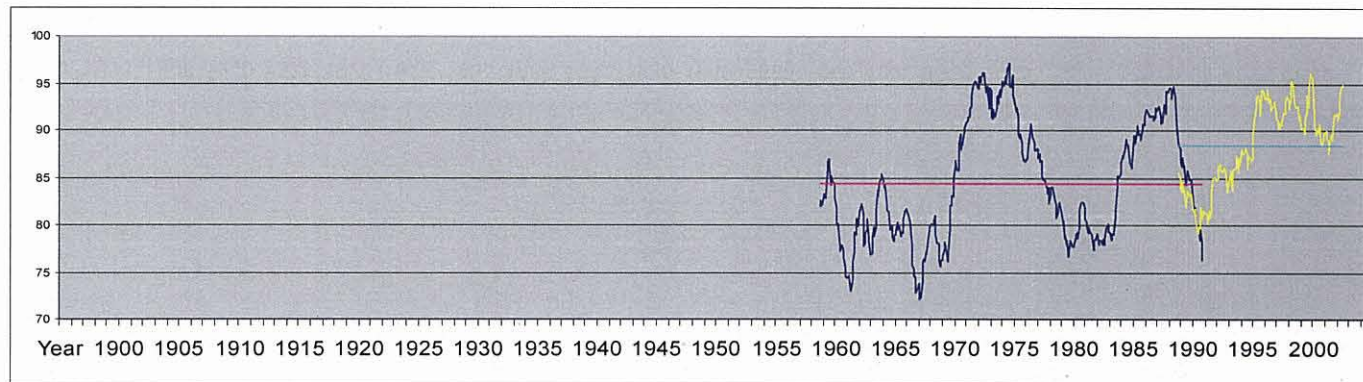


Figure 3.6: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall, Liawenee HEC (1959-1991) and Liawenee comparison (1989-2005).

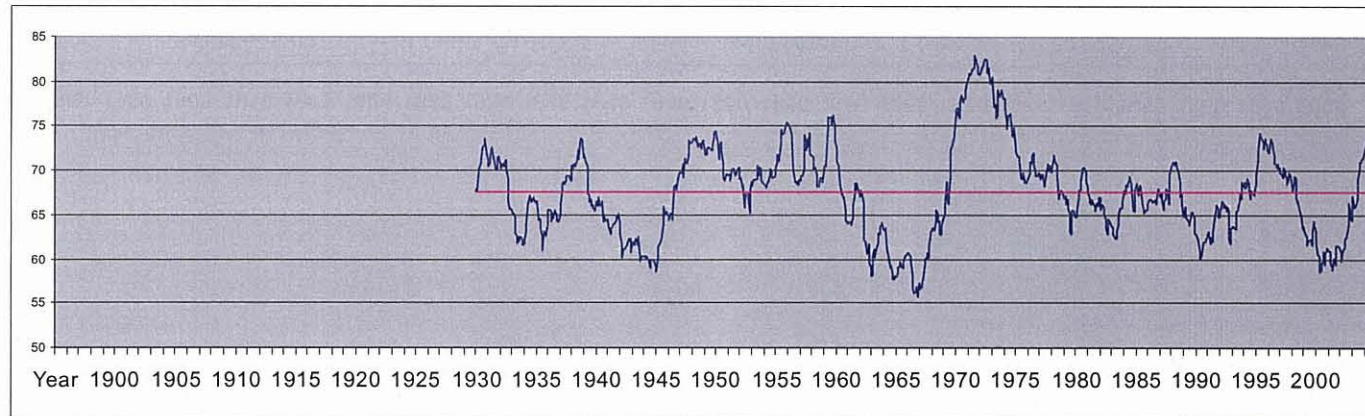


Figure 3.7: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Waddamana

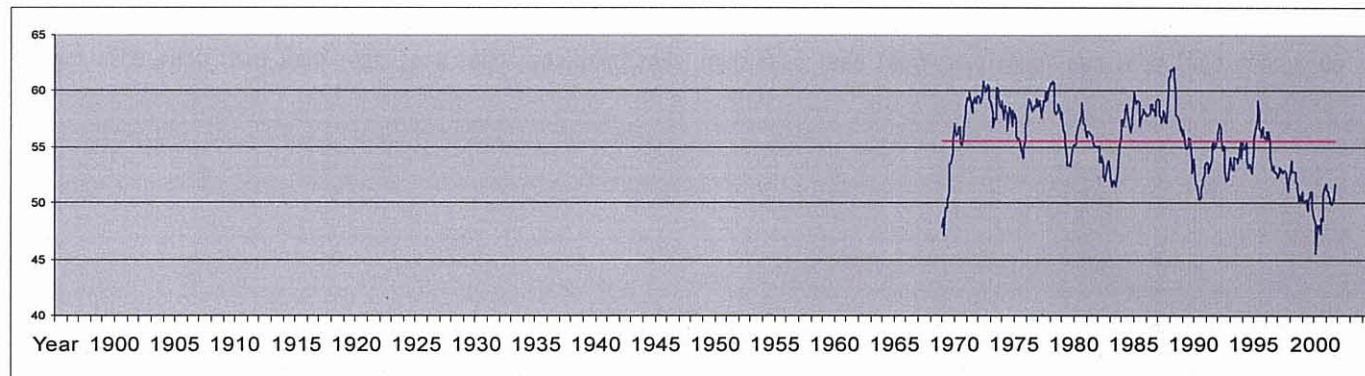


Figure 3.8: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Steppes

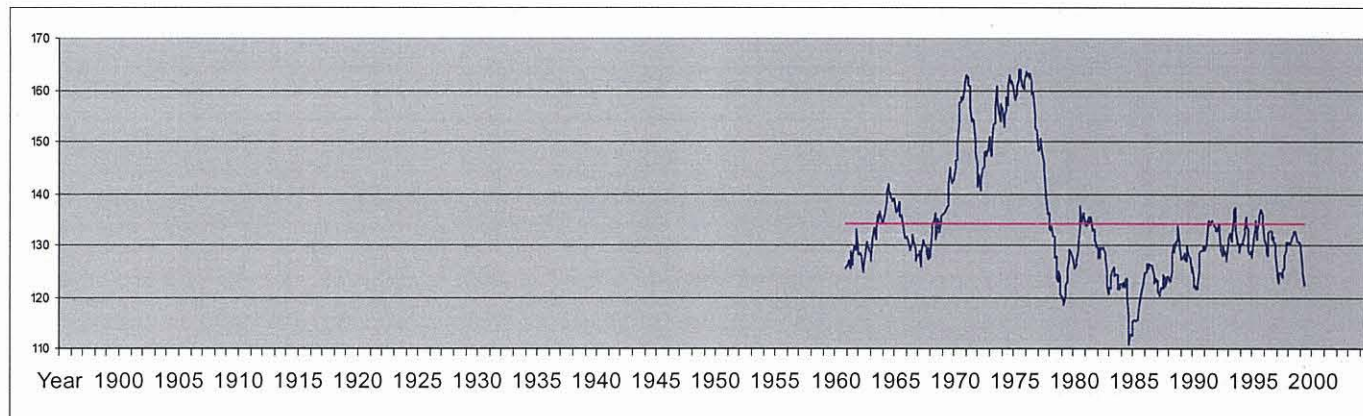


Figure 3.9: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Erriba

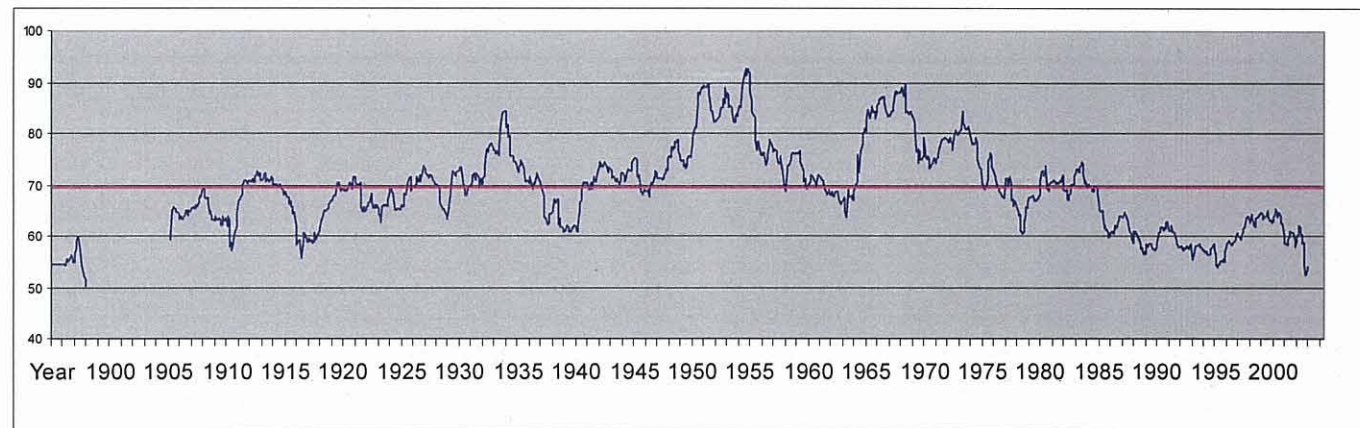


Figure 3.10: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Lake Leake

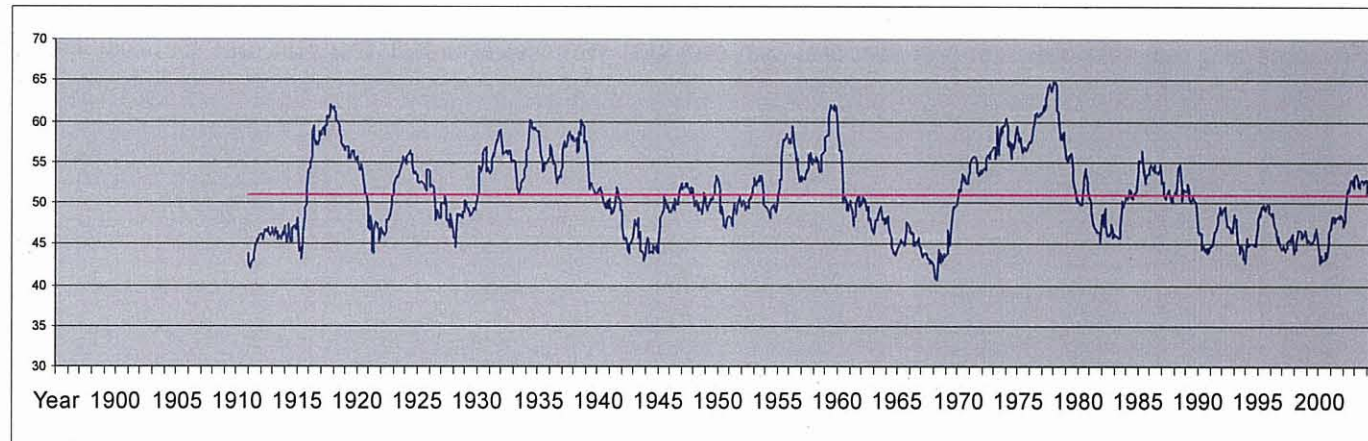


Figure 3.11: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Fingal

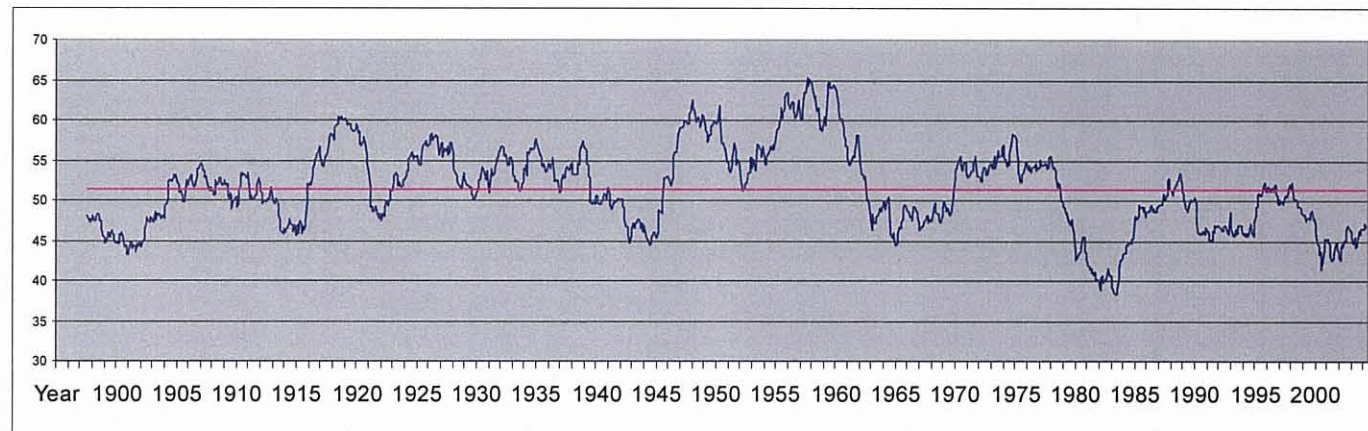


Figure 3.12: Five year running mean of total monthly rainfall (mm) compared to average monthly rainfall at Hobart

	Butlers	Waddamana	Victoria Valley	Tarraleah	Miena	Steppes	Erriba	Lake Leake	Fingal	Hobart
Early 1910s								1		10
Late 1910s										
Early 1920s								9		
Late 1920s									5	
Early 1930s										
Late 1930s								2		
Early 1940s										
Late 1940s										
Early 1950s	9		5					10	6, 7, 10	
Late 1950s		4, 6	6	6						
Early 1960s	1, 2, 10	2, 9		1, 9			1, 2	4	1	
Late 1960s		5, 7, 10	10		1, 2, 3, 5, 10	2, 4, 5, 7, 8	6, 7, 9	3	2, 4	5, 9
Early 1970s	3								8	
Late 1970s	4, 5		4, 7, 8	2, 10		10				2
Early 1980s	8		2, 3, 9	4				5	9	4, 9
Late 1980s	6, 7			3, 7	8, 9		3, 10	6	3	
Early 1990s							4, 5			
Late 1990s				5			8			
Early 2000s		1, 3, 8	1	8	4, 6, 7	1, 3, 6, 9		7, 8		1, 3, 6, 7

Table 3.2: Weather stations, showing the timing of their most prolonged period of average or below-average rainfall (grey), and the 10 years in which the driest 3 consecutive months on record occurred (numbered 1 – 10, with 1 being the year in which the driest 3 consecutive months on record occurred). (Crossed cells = no record, or patchy record)

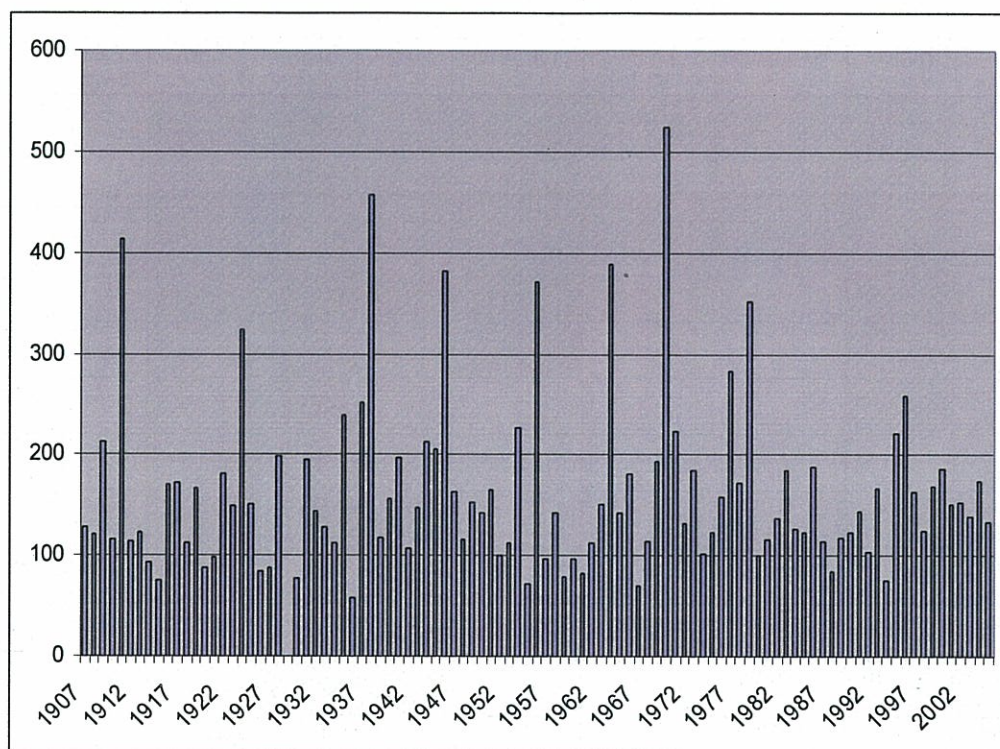


Figure 3.13: Total rainfall (mm) from January to March at Lake Leake

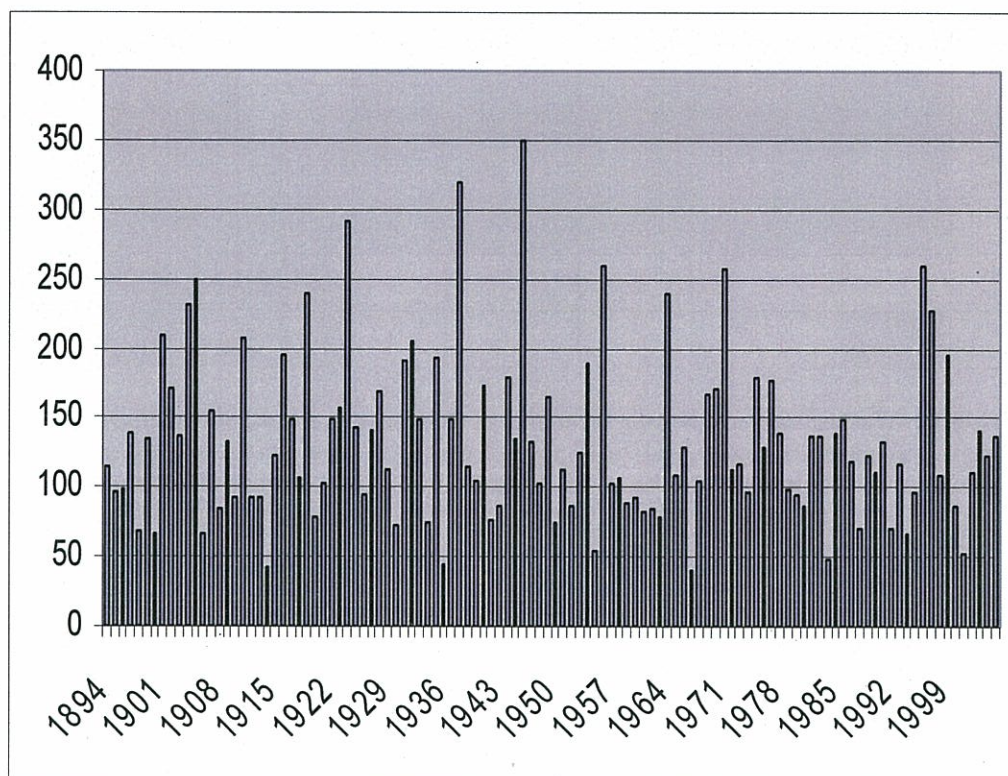


Figure 3.14: Total rainfall (mm) from January to March at Hobart

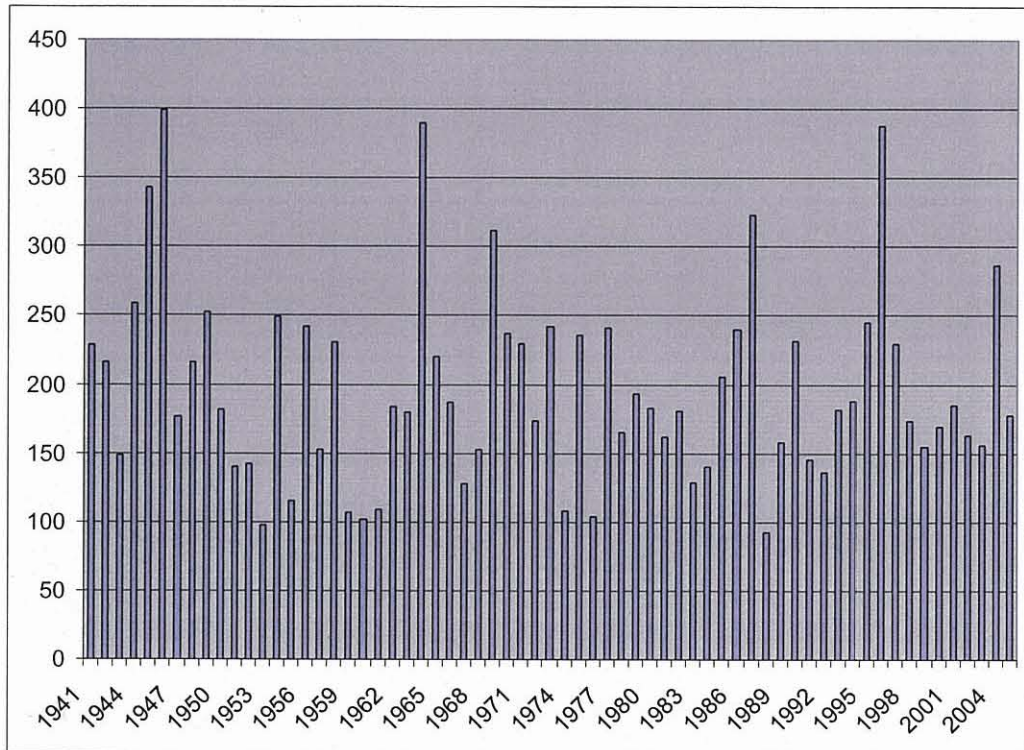


Figure 3.15: Total rainfall (mm) from January to March at Tarraleah

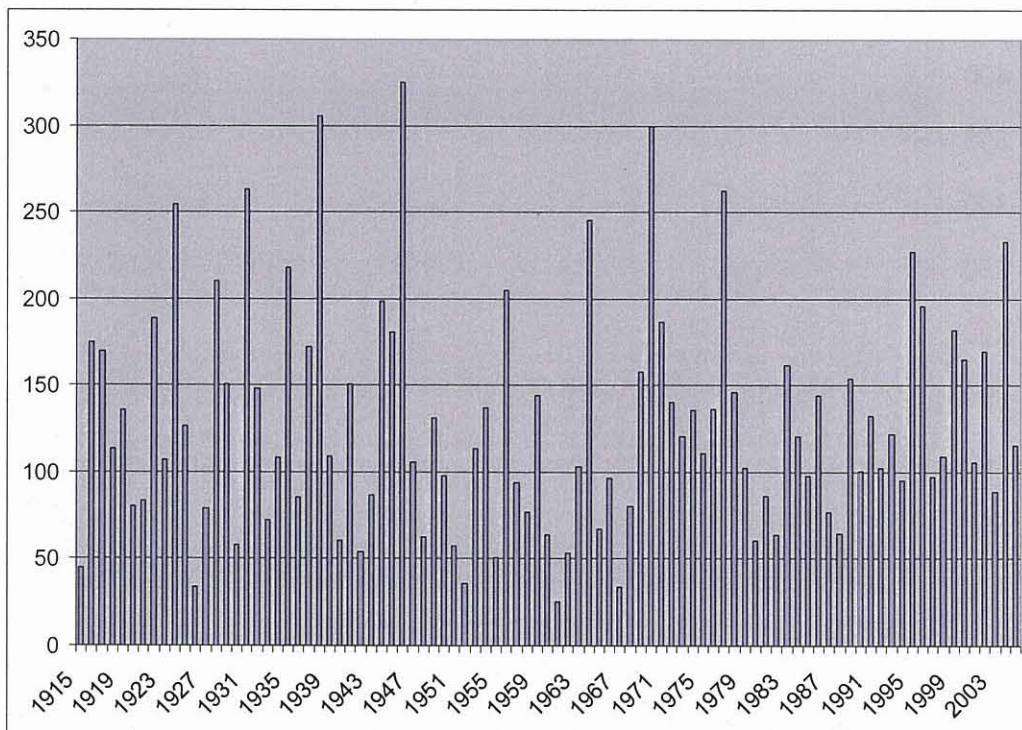


Figure 3.16: Total rainfall (mm) from January to March at Fingal

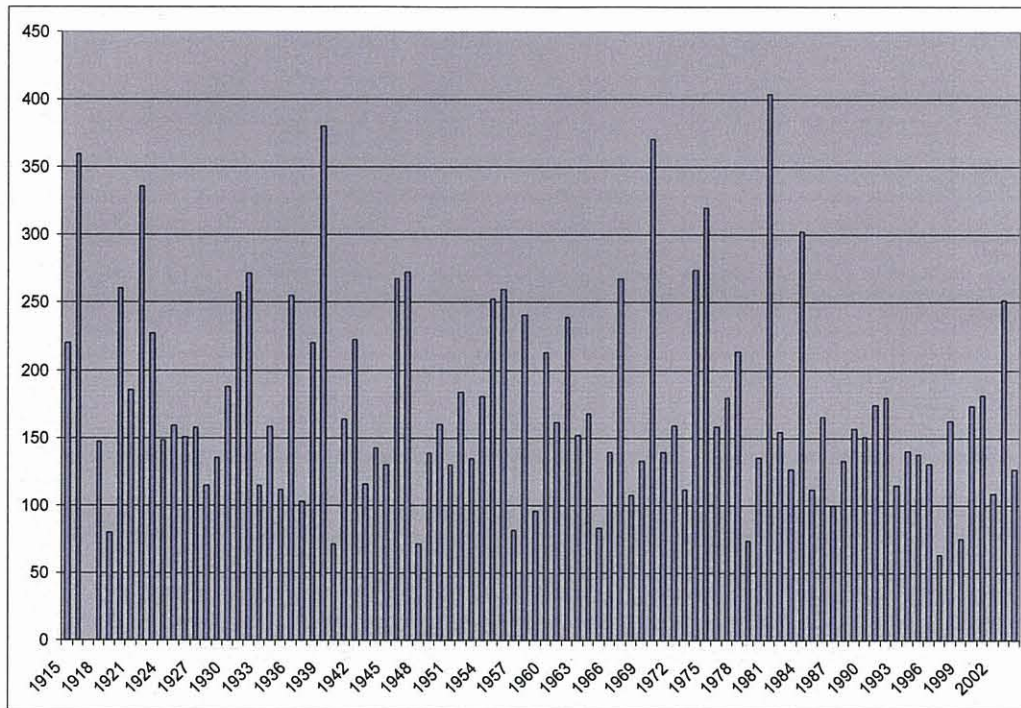


Figure 3.17: Total rainfall (mm) from June to August at Fingal

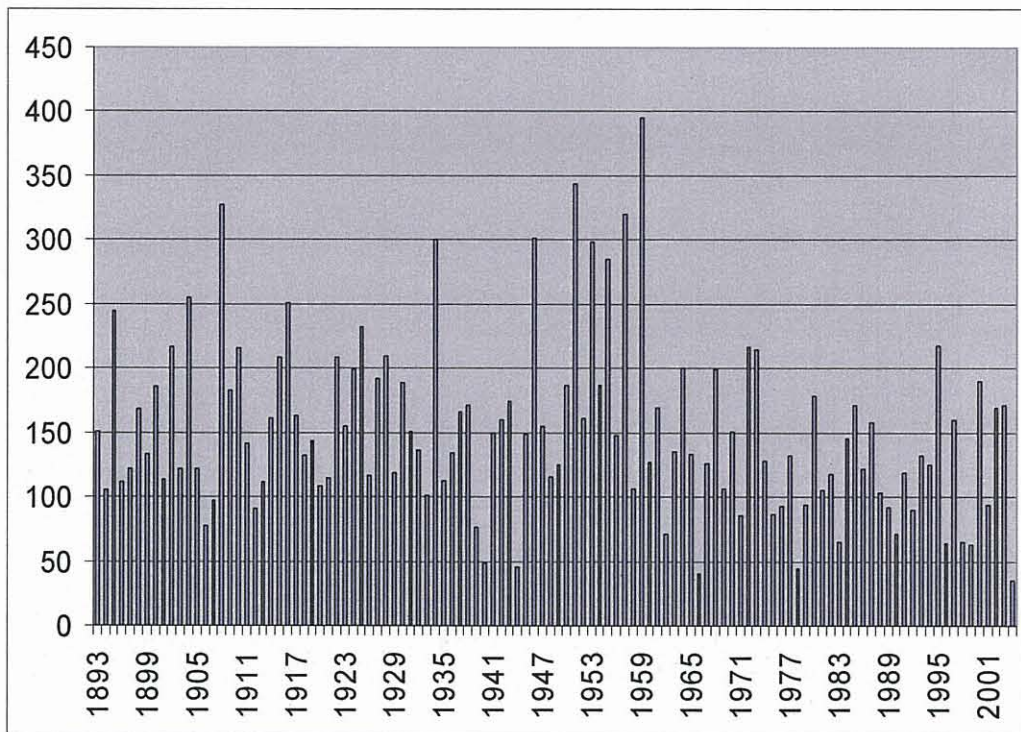


Figure 3.18: Total rainfall (mm) September to November at Hobart

Four of the eleven rainfall stations recorded their most consistent period of at or below average rainfall from the mid to late 1970s until the end of the record. These stations were Erriba (figure 3.9), Lake Leake (figure 3.10), Fingal (figure 3.11) and Hobart (figure 3.12). There were other periods of low rainfall at these stations, some of which were more severe, but the more recent lows have been the most consistent and prolonged. Butlers Gorge (figure 3.2), Victoria Valley (figure 3.3), Tarraleah (figure 3.4), Waddamana (figure 3.7) and Miena Dam (figure 3.5) also entered their most prolonged period of low rainfall on record in the late 1970s, but this was reprieved by several years of above-average rainfall in the early to mid 1990s. Steppes (figure 3.8) entered a period of below average rainfall in the early 1990s, which has continued until the end of the record. The severity of the dry periods over time is difficult to compare at Liawenee as the location of the rainfall station was moved after 1990 (figure 3.6), but it does not appear to have suffered from a period of consistently below average rainfall.

The years with the driest 3-month periods did not necessarily occur within the most prolonged period of low rainfall. Stations in relatively close geographic proximity experienced their 3 consecutive months of lowest rainfall in different years. For example Victoria Valley and Waddamana had their driest 3 month period in 2003 but Tarraleah and Butlers Gorge recorded their driest 3 month period in 1961. The driest 3 months at Lake Leake were in 1914, but in 1967 at Fingal. Despite this, there is evidence that the highest concentration of short-term droughts across Tasmania was the 1960s, 1980s and early part of the 21st century (table 3.2).

The total rainfall of the late summer – early autumn period at Hobart and Lake Leake (figures 3.13 and 3.14) has undergone a noticeable change since the late 1970s. There appears to be fewer years with significantly high rainfall in the late summer and early autumns. The trend towards a drier late summer – early

autumn period seems less pronounced at Fingal (figure 3.16). However, Fingal does appear to show a more pronounced trend of decline in winter rainfall (figure 3.17), with less wet winters since the mid 1980s. Spring rainfall also appears to have declined more significantly in Hobart than the late summer – early autumn rainfall (figure 3.18). At other stations, such as Butlers Gorge, Tarraleah, Waddamana and Victoria Valley, little change in the rainfall of any particular season was visible (e.g. figure 3.15).

3.3.2 Temperature

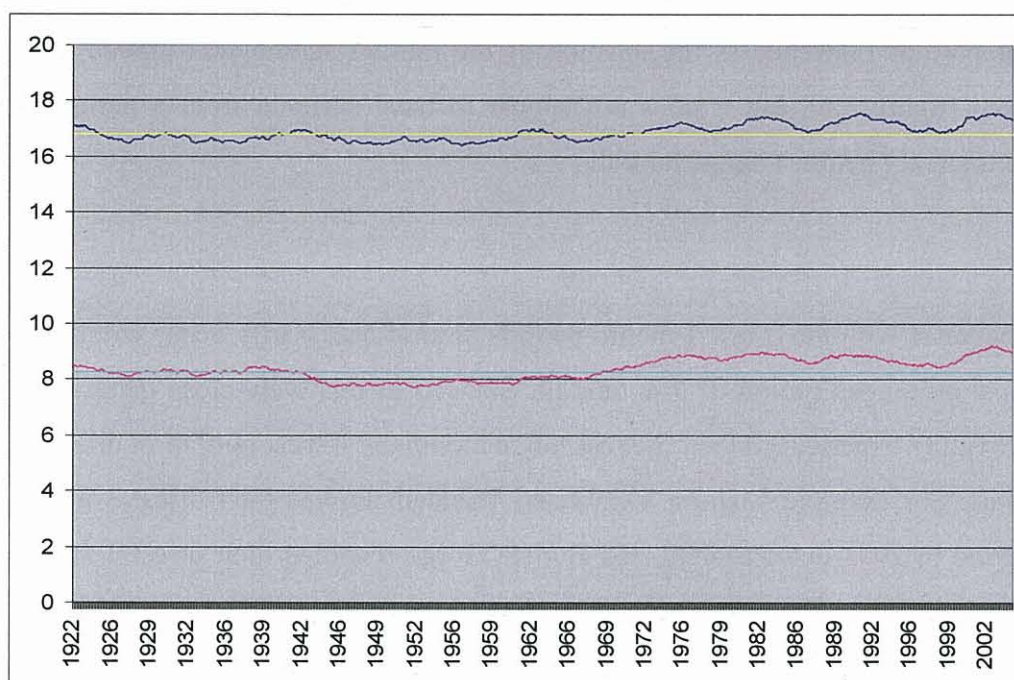


Figure 3.19: Five year running means of average monthly maximum and minimum temperatures at Hobart (°C)

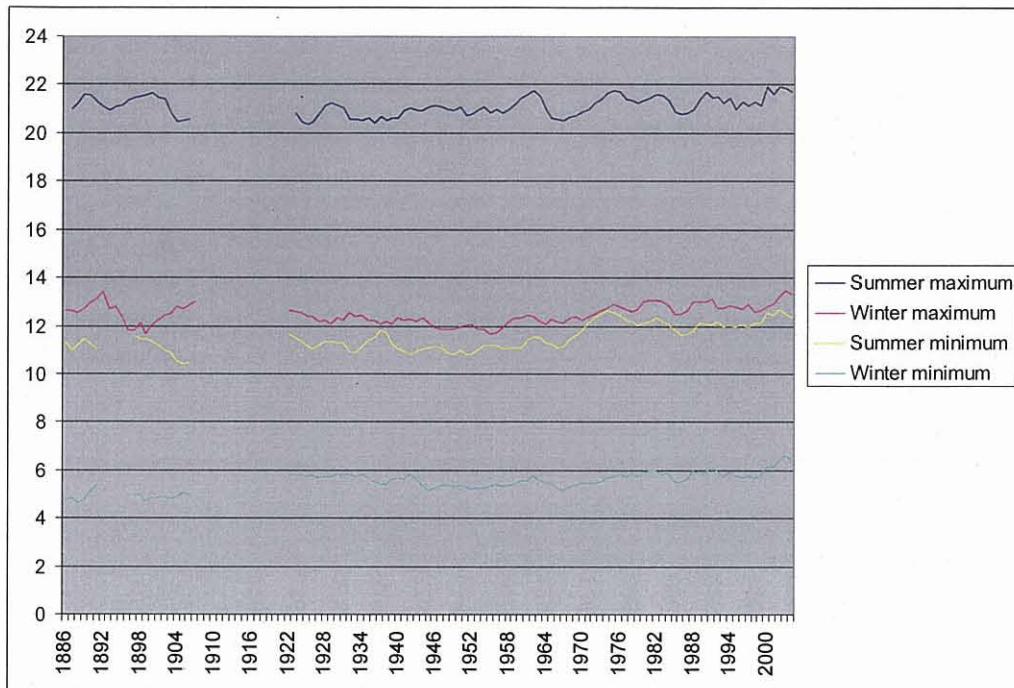


Figure 3.20: Five year running means of average summer and winter maximum and minimum temperatures at Hobart (°C)

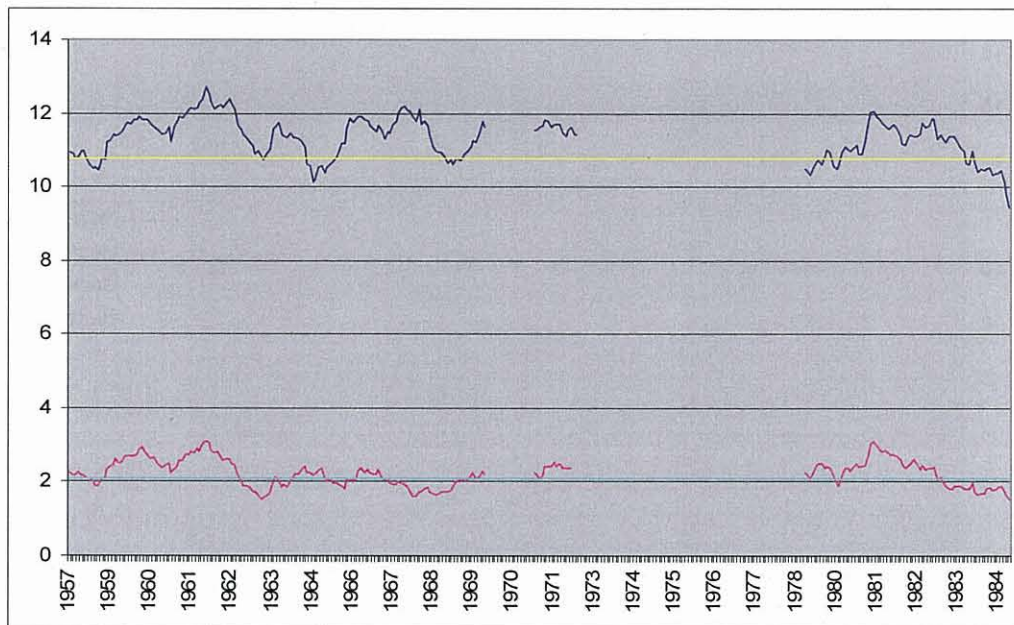


Figure 3.21: One year running means of average monthly maximum and minimum temperatures at Shannon (°C). (Note: One year running means were used for Shannon because there was not enough continuous data to generate 5-year running means)

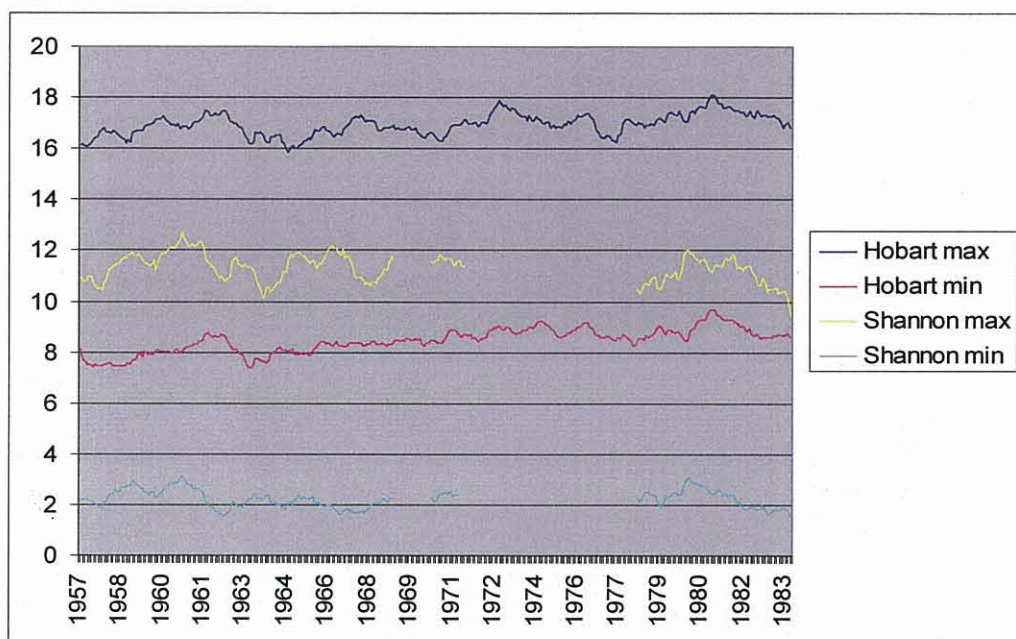


Figure 3.22: Comparison of one year running means of average monthly maximum and minimum temperatures at Hobart and Shannon (°C)

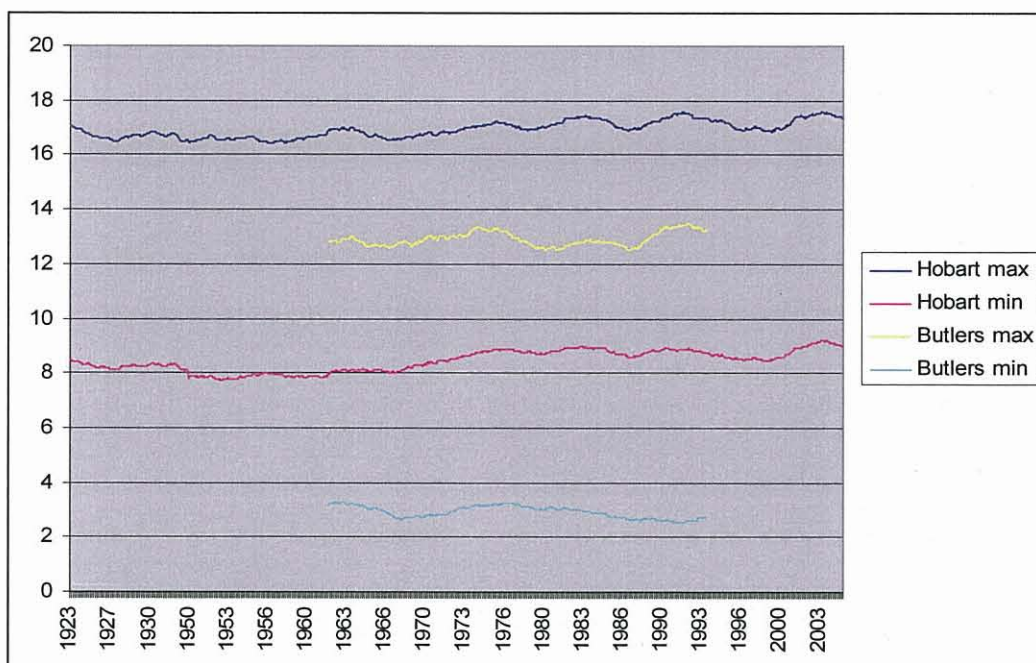


Figure 3.23: Comparison of five year running means of average monthly maximum and minimum temperatures at Hobart and Butlers Gorge (°C)

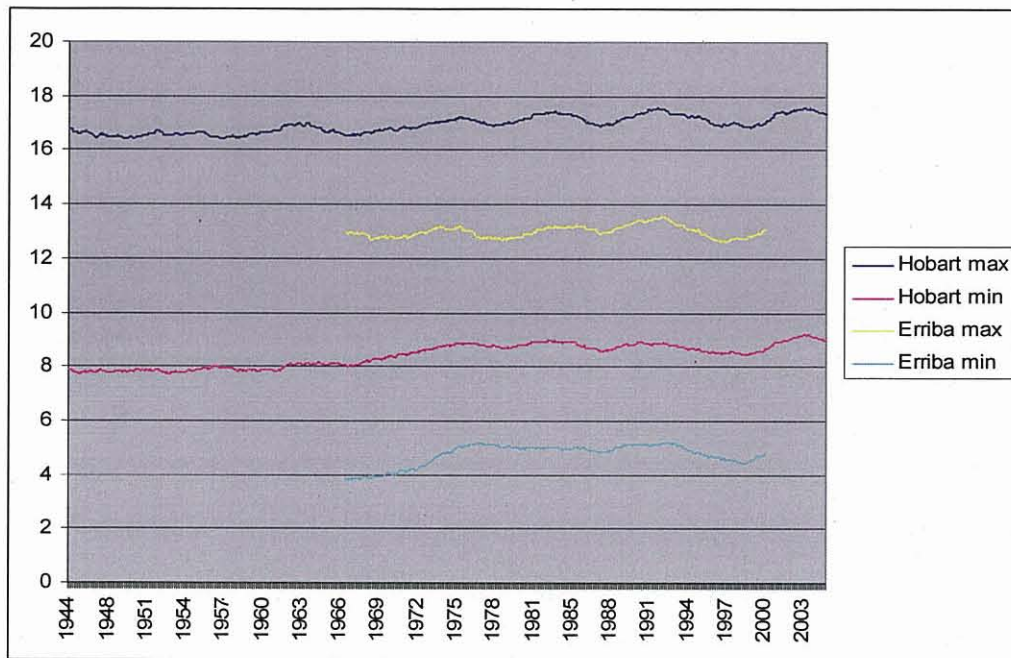


Figure 3.24: Comparison of five year running means of average monthly maximum and minimum temperatures at Hobart and Erriba (°C)

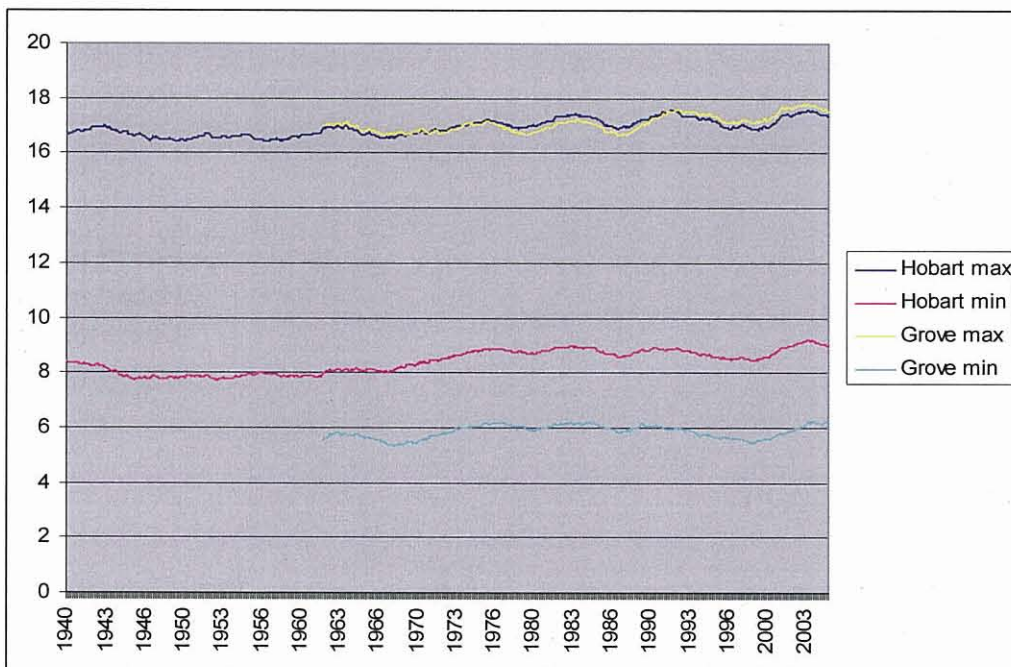


Figure 3.25: Comparison of five year running means of average monthly maximum and minimum temperatures at Hobart and Grove (°C)

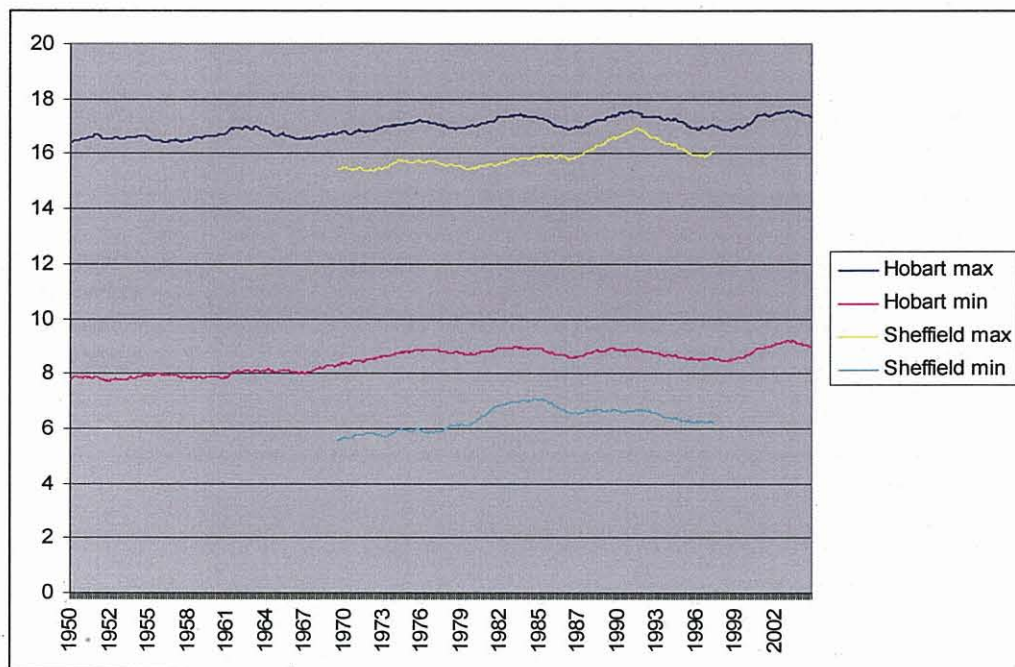


Figure 3.26: Comparison of five year running means of average monthly maximum and minimum temperatures at Hobart and Sheffield (°C)

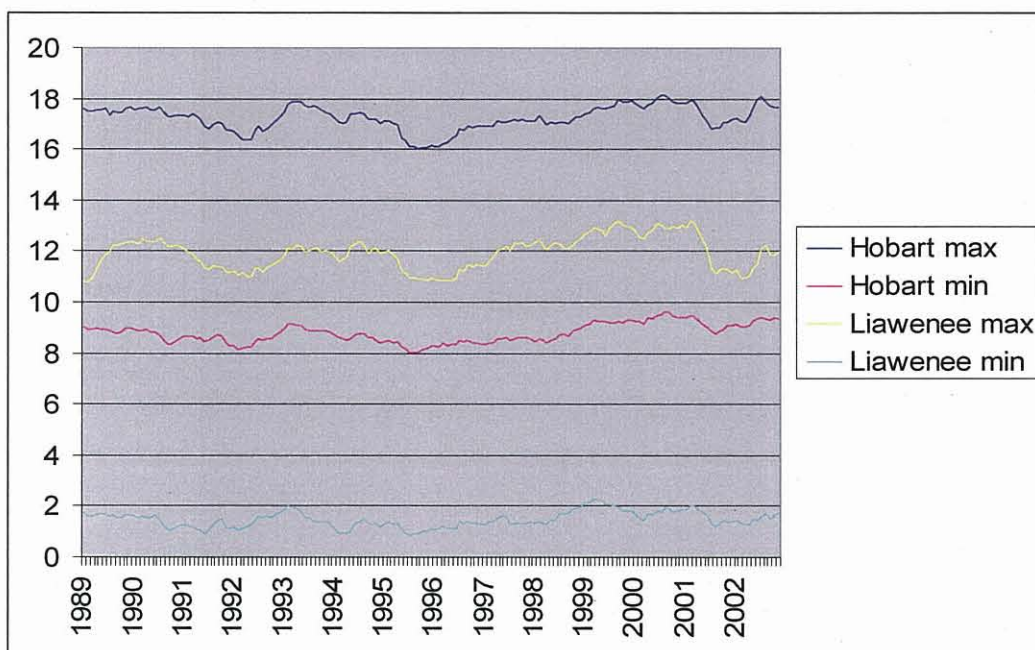


Figure 3.27: Comparison of one year running means of average monthly maximum and minimum temperatures at Hobart and Liawenee (°C)

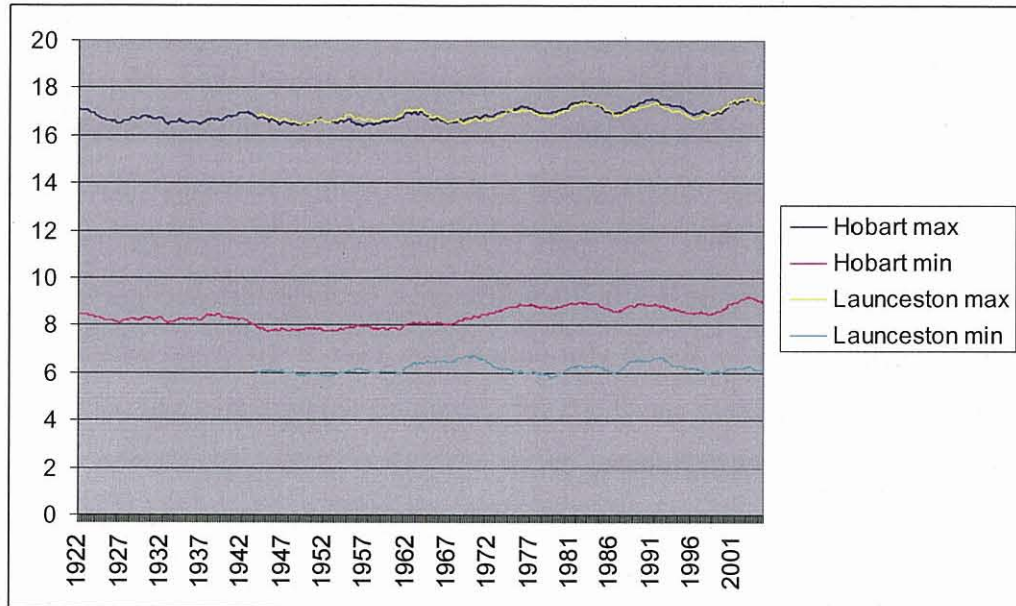


Figure 3.28: Comparison of five year running means of average monthly maximum and minimum temperatures at Hobart and Launceston (°C)

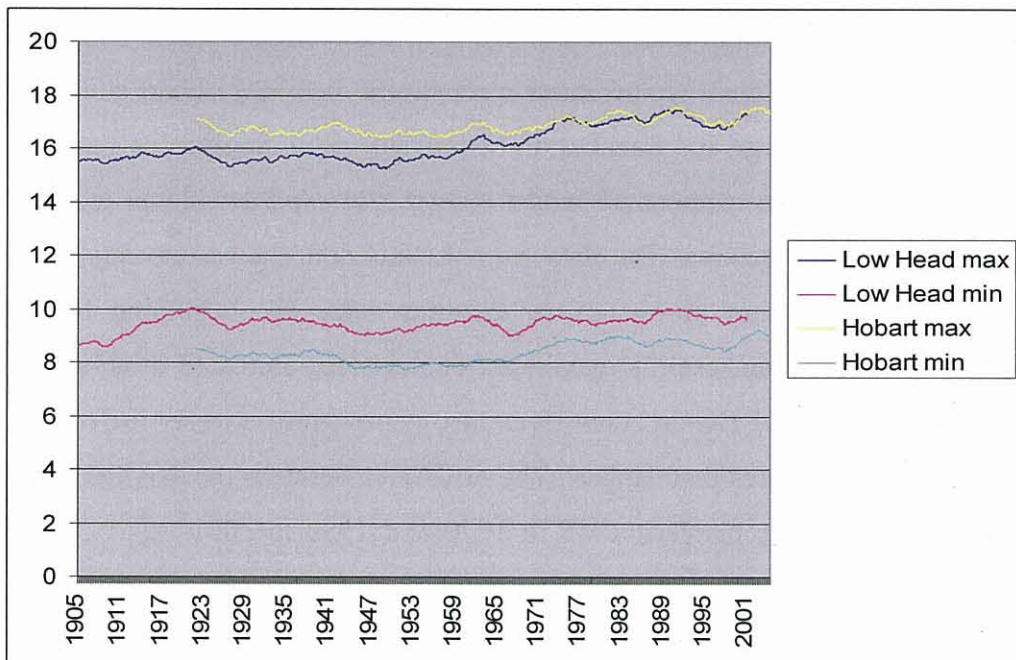


Figure 3.29: Comparison of five year running means of average monthly maximum and minimum temperatures at Hobart and Low Head (°C)

The data from the Hobart station (figure 3.19) indicates that there was a relatively sudden shift to higher average temperatures around 1973. As with the rainfall at Hobart, there has not been a continuous increase in temperature, but rather a shift to higher minimum and maximum temperatures that has been relatively stable since then. This same trend has occurred in both the summer and winter months (figure 3.20). The difference between the average of the monthly temperatures for the 25 year period from 1945 to 1970 and the 25 year period from 1980 to 2005 was 0.6°C for maximum temperatures and 0.8°C for minimum temperatures in Hobart.

Most stations showed the same or very similar trends in maximum and minimum temperatures to those recorded at Hobart during the period of their record (figures 3.24 to 3.27), except for Shannon (figure 3.22), Butlers Gorge (figure 3.23), Launceston (figure 3.28) and Low Head (figure 3.29). While the trends in mean maximum temperatures were similar between Hobart and most stations, Butlers Gorge and Launceston underwent certain periods of stable or decreasing minimum temperature whilst Hobart was experiencing an increase in minimum temperatures. The data from Launceston showed no sign of a significant increase in minimum temperatures since the beginning of the record. However Launceston's maximum temperatures increased at about the same rate as those of Hobart. Low Head also experienced a lesser increase in minimum temperatures than Hobart. The difference between the average mean monthly temperature between 1945 to 1970 and 1980 to 2001 was 0.3°C at Low Head. However Low Head experienced a greater increase in maximum temperatures than any other station, with the average temperature being 1.3 °C warmer in the period after 1980 than in the 25 years before 1970.

Although most stations experienced similar trends to Hobart in mean maximum temperatures, the rates of change were sometimes different. For example, although mean maximum temperatures at Hobart and Shannon both decreased

in the early 1980s, those at Shannon fell to a greater extent than those at Hobart (figure 3.22). This also occurred in the early 1960s. Conversely, the mean maximum temperatures rose at a greater rate at Sheffield during the early 1990s than they did at Hobart (figure 3.26).

3.4 Discussion

3.4.1 Evidence for climate change

Long term temperature and rainfall records from several stations indicate that there was a pronounced change in the Tasmanian climate in the 1970s. The Hobart station clearly shows that the average minimum and maximum temperatures rose quite abruptly in the early 1970s, and have remained at that level since then (figure 3.19). Rainfall has also remained consistently below average since the late 1970s (figure 3.12). This supports the observations of Nicholls (1996), Allen (1998) and Collins and Della-Marta (1999), that there was a shift in the relationship between temperature and rainfall in Australia in the 1970s. The same trend in rainfall was also visible in the north-west at Erriba (figure 3.9) and in the north-east at Lake Leake and Fingal (figures 3.10 and 3.11). There has also been a consistent dry period for at least 15 years since the early 1990s at Steppes, which is the most easterly station in the Central Plateau (figure 3.8). The remaining stations in the central region, Butlers Gorge, Victoria Valley, Tarraleah, Miena, Liawenee and Waddamana (figures 3.2, 3.3, 3.4, 3.5 and 3.7), also suffered approximately 15 years of below-average rainfall from the late 1970s to the early 1990s, but had above average rainfall in the late 1990s. However, there must be caution in interpreting the results from Steppes and Miena, which only had reliable rainfall data set since the late 1960s. This is a marginal time period for interpreting long term trends (M. Nunez *pers. comm.* 2006).

Although the timing of short-term drought periods is not consistent between stations, the trend to below average rainfall since the late 1970s is seen in several major stations. This consistency is good evidence that the changes are real, as it is possible to discount irregularities in the climate record of an observing station, which can result from changes in site location, exposure, instrumentation or observation practice (Collins and Della-Marta 1999).

Cores taken from long-lived King Billy pine trees (*Athrotaxis selaginoides*) in the mountains of Tasmania support the observation of a less significant rainfall decline in the south west of the state than the east. Pines from the eastern mountains show a decreased rate of annual growth since 1978, which coincides with beginning of the period of low rainfall recorded at the Hobart station. However, the cores taken from trees on the west coast did not show this, only an array of good and bad years (Dwiputranto 1990).

Temperature change has also been variable across the state. For example, Low Head experienced a greater and more continuous upward trend in maximum temperatures than Hobart (figure 3.29). Several stations, including Launceston, Butlers Gorge and Low Head did not experience the same increase in minimum temperatures as Hobart. This may be due to local topography and patterns of cold-air drainage, to which both Launceston and Butlers Gorge are susceptible. The proximity of the sea may also have had an affect on the temperatures at Low Head. The temperature trends seen at the Hobart and Grove stations contradict the findings of Allen (1998), that there has been no temperature increase across the south of the state. McInnes *et al.* (2004) found that maximum temperatures in Tasmania have increased at about twice the rate of minimum temperatures. This was observed in the Launceston and Low Head data, but not at Hobart.

Due to the lack of long term of records with continuous data sets, temperature change was more difficult to observe in the stations that occurred close to the habitat of *E. gunnii*. However the temperature trends for most stations roughly match those of Hobart for the same period. Therefore it can be assumed that similar temperature increases may have taken place at those stations, at least in mean maximum temperatures. Further evidence that warm season temperatures have increased across most of Tasmania comes from Huon Pine (*Lagarostrobos franklinii*) tree rings from the west coast of Tasmania, which indicate that the growth of 700 year old trees has been unusually rapid since the mid 1960s. Changing warm season temperatures appear to be largely responsible for observed Huon Pine growth fluctuations, and the period of rapid growth correlates with a period of warmer sea surface temperatures around Tasmania (Cook 1991). However, it cannot be assumed that mean maximum temperatures have warmed to the same degree across the whole of Tasmania. Although the pattern of temperature rise and fall at most stations was similar to Hobart, the rate of change during those periods was not necessarily the same.

It is important to note that there was no evidence of a 1.5°C increase in mean maximum temperatures at the Shannon station, as reported by Harris *et al.* (1988) and Kirkpatrick and Gibson (1999). Both sources reported using temperature data from Shannon from 1945. However, according to the Bureau of Meteorology (*pers. comm.* 2006), temperature records before 1957 at the Shannon station were not quality controlled and therefore unreliable. Kirkpatrick and Gibson (1999) reportedly used data from Shannon until 1995. This is an error in their paper, as the Shannon station was closed in 1985 (Bureau of Meteorology *pers. comm.* 2006). The supposed 1.5 °C increase in mean maximum temperatures at Shannon has been cited in later papers as evidence for climate change in the Central Plateau region, and to support claims that the decline of *E. gunnii* is due to climate change (e.g. Potts *et al.* 2001, Mocatta 2006). Such an increase would surely have a dramatic effect on the transpiration demands on the vegetation. Although the post 1957

temperature data obtained for from the Shannon station showed no evidence of increasing temperature (figure 3.21), the mean maximum temperatures trends at Shannon did show similar patterns, if not rates of change, to those that occurred at Hobart (figure 3.22). The same is true for the temperature at Liawenee between 1989 and 2003 (figure 27). It is therefore possible that average maximum temperatures in the Central Plateau have become warmer than before 1970, but there is no simple way to tell at what rate and extent they may have increased.

3.4.2 Implications for the health of *Eucalyptus gunnii*

Changed rainfall patterns have occurred in several stations which are more or less representative of the geographic range of *E. gunnii*. However it can be difficult to extrapolate the trends from stations to some field sites, as the nature of topography and altitude can cause precipitation stations in close proximity to vary greatly in recorded rainfall (Allen 1998). This is evident in the variation in the timing of severe drought years across the state, and consistent with what was noted by Pemberton (1986), that droughts in Tasmania are seldom widespread and are usually restricted to a certain area. Therefore it is likely that rainfall at stations such as Erriba may not be so representative of rainfall at the field site at Leary's Corner (chapter 4), which is 18 kilometres away to the south west, 200 m higher in altitude, and receives an extra 1200 mm of average annual rainfall than Erriba. Unfortunately, this is unavoidable. Allen (1998) also found that most of the weather stations in Tasmania are concentrated at low altitude and around populated regions. Most others are of short duration or suffer from large amounts of missing data.

Despite this, the rainfall data from several stations do indicate that many stands of *E. gunnii* are likely to have undergone a 15 to 35 year period of

predominantly below average rainfall. Increased mean maximum temperatures across the state may also have contributed to decreased soil moisture availability by increasing evaporation and transpiration demand. This long period of below-average rainfall and above average temperatures would have put considerable stress on the trees, by reducing their potential growth rates, and their capacity to recover from disturbance. Within Manion's (1991) dieback model, this would be known as a 'predisposing factor' – a long term stress that weakens the tree and increases its susceptibility to later stresses. The timing of the most severe short-term droughts in relation to the long-term periods of rainfall deficit is therefore very important. Eucalypts can be quick to recover if stress is not prolonged and conditions return to normal soon after disturbance. Therefore it is likely that the short-term droughts which occurred at several stations in the 1960s did not have a significant long term effect, as they were followed by a period of above average rainfall at most stations in the early 1970s. However if the short-term droughts occurred within the recent prolonged period of rainfall deficiency, as has happened at most stations (table 3.2), the trees would not have been given the option to recover. In Manion's (1991) framework, these short-term 3-month droughts would be known as 'inciting stresses' – those of relatively short duration that can substantially reduce tree vigour.

Few stations show distinct patterns in the amount of late summer – early autumn rainfall. This suggests that it has not been these months specifically that are responsible for overall decline in rainfall, and that increasing severity of the 'dry season' is not to blame for increased stress on *E. gunnii*. However rainfall decline in any season can be important. Less rainfall in winter and spring means the soil will not accumulate and sustain as much moisture over summer (Hutchinson 1983).

A further consideration is that rainfall trends, although undoubtedly the most important variable, are also the least sophisticated measure of moisture deficiency. For example, a one-off high rainfall event can distort what may have otherwise been a very dry year. A more realistic measure would be gained from including a combination of climatic factors, such as temperature, wind speed and humidity (Lindesay 2003). The timing and intensity of rainfall events, hydrogeological position and the moisture holding capacity of the soil would also contribute to the accumulated rainfall deficit of a particular site over time (McCutchan 1976). A possible model that would be more relevant to vegetation is the Soil Dryness Index, developed by Mount (1972). However such models have a large demand for data and need careful checking that the assumptions involved are valid under local conditions (McCutchan 1976). Applying the SDI to the stations in the region of *E. gunnii* for a period enough to detect long term trends, was not possible with the paucity of the meteorological data available. A spatial interpolation technique would be required to achieve this, under which accuracy might suffer.

In conclusion, there is good evidence that the period from the late 1970s has been the most consistently warm and dry period over the last 100 years over much of Tasmania. It is very likely that this prolonged dry period has weakened the trees and compromised their ability to recover from short term stresses, such as severe droughts, which have also been common in the second half of the century.

Chapter 4: The spatial pattern of dieback and its correlates

4.1 Introduction

Spatial patterns and correlates of dieback have been documented by many researchers, as a means of generating hypotheses of the causes of tree decline (e.g. Podger 1981, Manion 1991, Eldridge *et al.* 1993, Wylie *et al.* 1993, Gardner and Fisher 1996, Fensham and Holman 1999, Macgregor and O'Connor 2002). Distinct patterns of dieback would be expected to result from different damaging factors. For example, if decline were due to climatic drying, the poorest tree health would be expected to be associated with the spatial and temporal patterns of rainfall and of soil moisture accumulation in the landscape (Kirkpatrick and Nunez 1980, Doyle 2003). However, if an unusually severe frost were to blame for tree death, the trees closer to the edges of frost hollows would be expected to be the most damaged (Paton 1988, Kirkpatrick 1999). The negative factors associated with grazing, such as soil compaction and nutrient addition (Neyland 1999, Close and Davidson 2004) would be expected to have a localized impact on the health of individual stands of trees; whereas other changes, such as altered fire regimes (Jurskis 2005) would have an impact depending on fire history.

Species and communities are predicted to be most susceptible to climate change at their range limits (Lavoie and Payette 1992, Harte and Shaw 1995, Gardner and Fisher 1996, Gottfried *et al.* 1999, Hughes 2000, Leohle 2001, Walter *et al.* 2002, Jia *et al.* 2006). *Eucalyptus gunnii* is adapted to high levels of moisture, as it grows

in wet, cold conditions on valley bottoms, or the edges of waterlogged plains (Williams and Potts 1996). Due to the prevailing westerly winds, there is a marked rainfall gradient across Tasmania, on an axis that is roughly west to east (Jackson 1999) (figure 4.1). Therefore, *E. gunnii* populations that occur further east experience lower rainfall, and presumably are closer to the climatic range limit of the species. Drying in areas which already receive low rainfall may sooner exceed the physiological tolerances to drought stress of these trees than trees in wetter areas.

This chapter will present and discuss the spatial patterns of mature tree health that were observed through field surveys of *E. gunnii* across its range. It will explore the spatial correlates associated with mature tree health, and discuss whether the results favour particular explanations for variation in the health of *E. gunnii*.

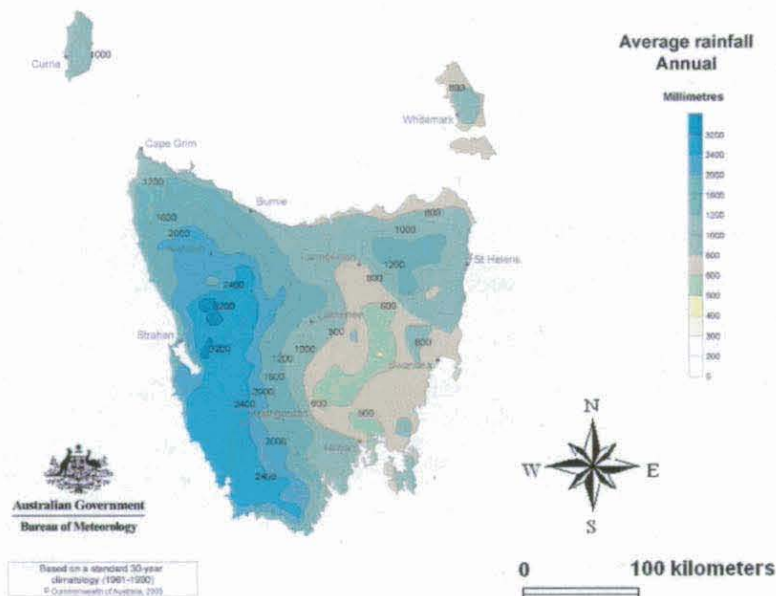


Figure 4.1: Rainfall gradient across Tasmania (From Bureau of Meteorology 2006).

4.2 Methods

4.2.1 Field methods

Field surveys of 53 stands of *E. gunnii* were undertaken across Tasmania. Field sites were chosen to cover the range of the species across the west to east rainfall gradient. The driest sites sampled were in the 600-800 mm region of average annual rainfall, and the wettest site in the 2400-3200 mm region (figures 4.1 and 4.2). For ease of accessibility, all sites were within 1 km of a road. Sites were identified by observation of the species from the road, and from the coordinates stored with records from the Tasmanian herbarium. Fourteen sites were sampled in the north-west region; 29 in the Central Plateau; 7 along the Lyell Highway in the Lake St. Clair region; 2 in the north-east and; 1 in the south-east at Snug Tiers (figure 4.2 and appendix 1).

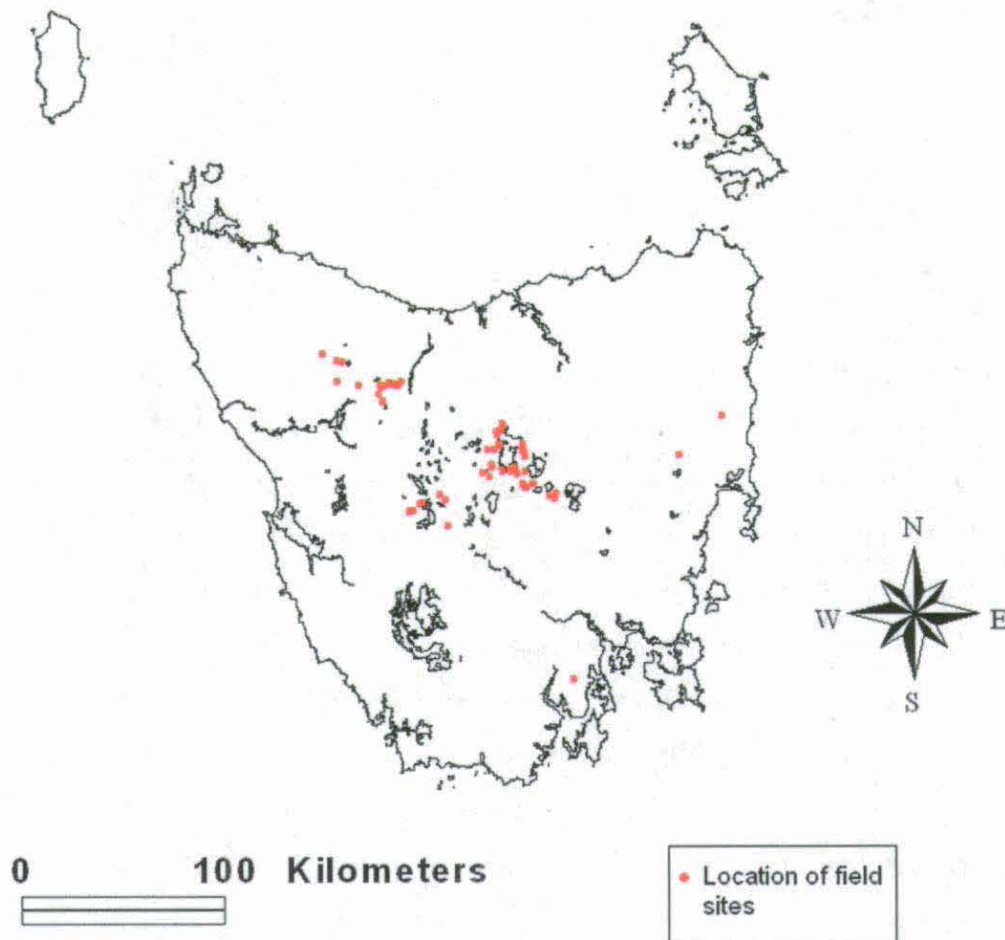


Figure 4.2: Location of the 53 field sites

Due to the large variation in the form of *E. gunnii* across the State (chapter 2), a plotless sampling technique was employed. At each site a minimum of 50 plants were sampled. The first tree was a randomly chosen mature individual. From here, the next tree sampled was the nearest neighbour to the previous. In situations of high juvenile density, sampling of juveniles was ceased at 30 individuals, and additional mature trees were sampled regardless of the interstitial juveniles, until at least 20 mature trees had been measured. Juveniles were only sampled if they were above 20 cm in height. The distance to the nearest metre between each individual

was estimated.

Each tree was measured for circumference at breast height, and this measure was later converted to basal area. For smaller or juvenile trees, circumference was measured from just below where the branches split into more than two. The basal area of *E. gunnii* per hectare was approximated for each site by calculating the average basal area of all individuals, and dividing this by the average area occupied by a single tree, which was calculated using the formula:

$$(\text{Average distance between mature trees})^2 / 4.$$

Each tree was given a crown health rating from 0 to 5. Crown density and the presence of epicormic shoots were used as visual indicators. Dead trees received a score of 0, and a score of 5 was given to trees with a dense and healthy crown. Plates 4.1 – 4.6 show typical *E. gunnii* specimens which would have received crown health ratings from 0 to 5.

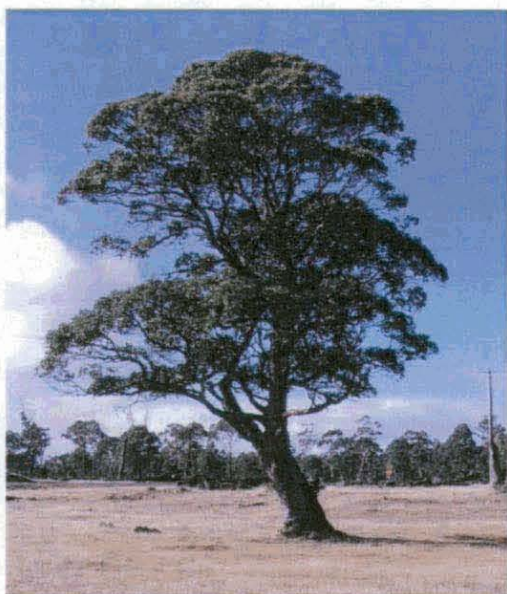


Plate 4.1: Crown health 5



Plate 4.2: Crown health 4

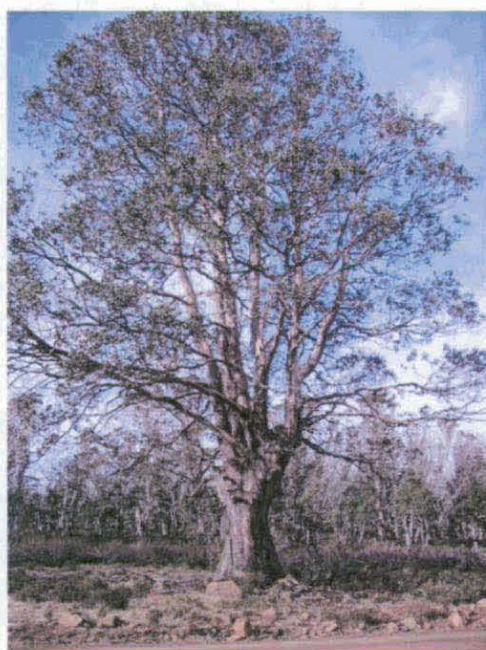


Plate 4.3: Crown health 3

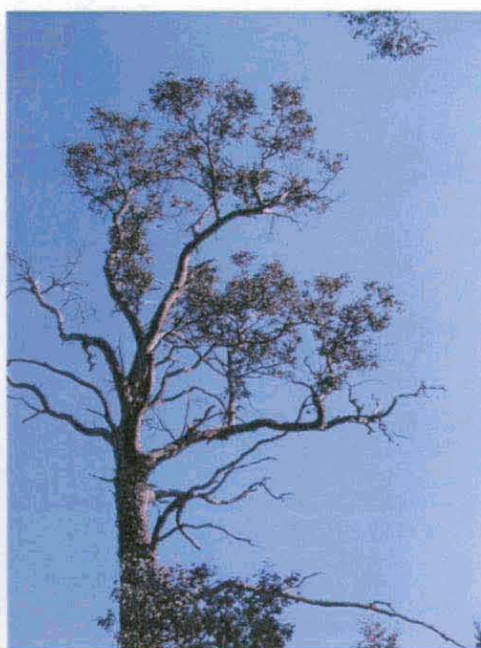


Plate 4.4: Crown health 2

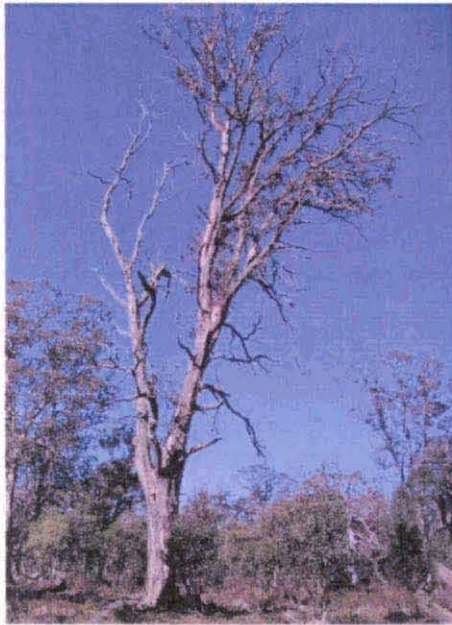


Plate 4.5: Crown health 1

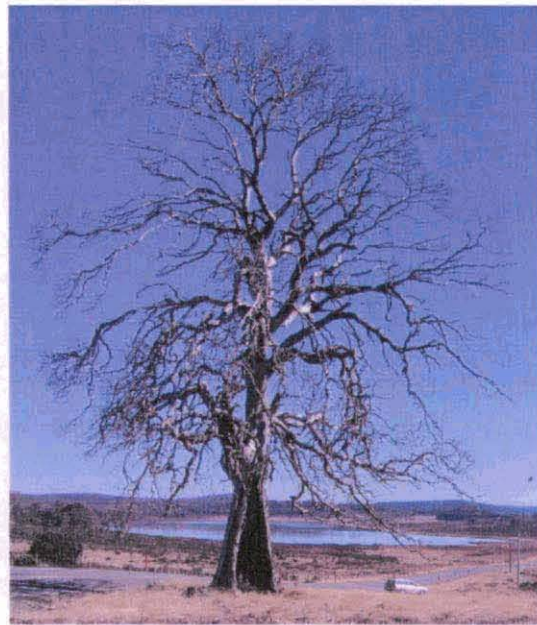


Plate 4.6: Crown health 0

For each population, the percentage of mature trees in different health classes was calculated. Trees in health classes 0-2 were combined and defined as 'poor health', and health classes 3-5 as 'good health'. The percentage of mature trees with poor and good health in a population was calculated.

The crown health rating did not include leaf health. However, if insect herbivory was noticeable, this was noted as "little", "moderate" or "severe". Plates 4.7 and 4.8 show typical leaves which would have been placed in the 'little' and 'severe' classes.

The form of the mature trees was also noted as either 'tall and straight', 'divaricate' (divaricate branching), or 'mallee', (multistemmed from the base)

(plates 4.9 - 4.11).



Plate 4.7: Little insect defoliation



Plate 4.8: Severe insect defoliation

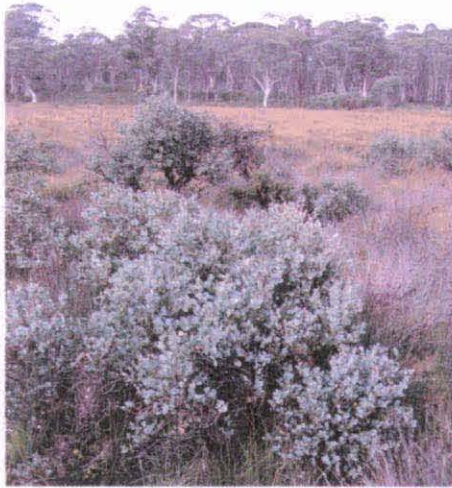


Plate 4.9: Mallee form



Plate 4.10: Divaricate form



Plate 4.11: Tall straight form

The environmental characteristics noted for each site were:

- altitude (from 1:100 000 topographic *Tasmaps*);
- aspect, which was measured with a compass where applicable, and given a score from 1 to 9 based on the expected intensity of solar radiation and the influence of drying winds for that aspect (Kirkpatrick and Nunez 1980) (table 4.1);

North west	1	ENE or SSW	6
NNW or WNW	2	E or S	7
N or W	3	ESE or SSE	8
NNE or WSW	4	SE	9
NE or SW or flat	5		

Table 4.1. Aspect classes based on the intensity of solar radiation and influence of drying winds. Number 1 is the driest aspect and number 9 the wettest.

- topographical position, which was noted as either “upper slope”, “mid slope”, “lower slope” or “flats”;
- the average slope of the site, which was measured with a clinometer;
- a visual estimate of the percentage of bare ground at the site, to the closest 10%;
- a visual estimate of the percentage of exposed rock at the site to the closest 10%;
- a measure of grazing pressure: (1 = livestock or livestock scats present; 2 = livestock not seen but evidence of the land having been used for grazing in the past in the form of fencing; 3 = no evidence of grazing);
- tenure, which was determined from *The List* (<http://www.thelist.tas.gov.au>), and was classified as either government-owned conservation area, or privately-owned land;
- the degree of waterlogging of the ground surface, which was divided into 3 categories (1 = dry underfoot, 2 = a little wet underfoot, 3 = very wet

underfoot), sites that were 'very wet underfoot' typically had streams flowing through them or the ground was spongy and wet;

- evidence of recent fire (1=evidence of recent fire in charred or dead plants, 2=evidence of fire in the past in charred older trees, 3= no evidence of fire);
- a list of the vascular species at each site, gathered from an approximately 20 minute search;
- the structure of the vegetation at each site, which was classified into 3 types, broadly analogous to those identified by Duncan (1999) for dry sclerophyll woodlands: 1 = shrubby forest or woodland, 2 = grassy woodland, 3 = sedgy woodland or buttongrass plain bordered by mallee;
- the geology of each site, which was determined from 1: 63360 and 1: 50 000 geology maps (Tasmania Department of Mines 1954, 1963, 1966, 1984 and 1986 and Tasmania Department of Resource and Energy 1991);
- the average rainfall and maximum and minimum temperatures for each month for each site, which were interpolated using a model developed with the 'thin plate smoothing spline' technique developed by Hutchinson (1998), using data from the Bureau of Meteorology from all Tasmanian stations from the beginning of their records until 2005.

4.2.2 Data analysis

Four dimensional ordination solutions for presence/absence data of species were derived using the default options for non-metric multidimensional scaling in DECODA (Minchin 1990). The four dimensional solution had a minimum stress of 0.124542. The axis scores were used as input variables in MINITAB, to derive an agglomeration classification of floristic plant communities, using Wards method and Euclidean distance. A dendrogram was produced, from which

communities were selected. The percentage frequency of species in each community was calculated using DECODA.

MINITAB was used to perform ANOVA for analysis between class and continuous variables, and to calculate the Pearson's product moment correlation coefficient to test the significance of relationships between continuous variables. Pearson's product moment correlation was used to test for relationships between the percentage of mature trees in poor health categories, and the continuous variables of rainfall, temperature, altitude, percentage rock and percentage bare ground. 1-way ANOVA was performed to test the significance of relationships between the percent of mature trees in poor health categories and the categorical variables of tree form, topographic position, aspect, land tenure, evidence of grazing, evidence of recent fire, degree of water logging and geology. A 1-way ANOVA was also used to test the significance of relationships between tree health variables and the NMDS scores of the floristic communities detected by DECODA. Cross-tabulation and the Chi-squared test were performed in MINITAB to test for relationships between the categorical variables of crown health class and size class, and form and grazing. A general linear model was used to test for an interactive effect between rainfall and grazing on tree health.

The same analyses were done separately within the 24 protected and the 29 private sites, and also performed separately on the subset of 29 sites within the Central Plateau region.

4.3 Results

4.3.1. Spatial patterns of population health

Figures 4.3a and 4.3b and appendix 1 show that the majority of the populations with over 60% mature poor health were found on the Central Plateau, with a concentration of very unhealthy populations to the south east of Great Lake. There were also two populations with 80-100% unhealthy mature trees in the North West: at Leary's Corner and by the Murchison Highway. Populations with 60-80% poor health were found at two of the higher altitude sites in the Central Plateau (Half Moon Creek and the Liawenee garden), and at Cider Gum Road in Miena. The concentration of populations with poor health to the south and east of Great Lake had three notable exceptions: the Ellis Plains, Sandbanks Tier and the site on Interlaken Road.

Populations of intermediate health were found in the north-west at the two Guildford sites, at the Hatfield River, and on the upper side of the road near the Iris River on the Middlesex Plains. In the Central Plateau populations of intermediate health were found along Poatina Road to the east of Great Lake. Relatively healthy populations, where 20-40% of the trees were in poor health were found in the north-west at Mount Kate, Sunshine Creek and the most easterly site just out of Moina. On the Central Plateau, they occurred near the Ouse River on the Marlborough Highway, and at the Ellis Plains (Appendix 1).

The Snug Tiers population and the two populations in the north east were among the healthiest populations, with only 20% or less of their mature trees suffering

from poor health. The populations along the Lyell Highway also had predominantly healthy adult trees. Several other very healthy populations occurred in the north-west near Catley Road, outside the Visitor Centre at Cradle Mountain, Weaning Paddock Creek, and the population a little further to the east of this (cow-sign). There were several very healthy populations on the western and south western sides of Great Lake: on the Lake Augusta Road, Monpeelyata Road and at Reynolds Neck (Appendix 1).

Figure 4.4 shows the variability within populations with over 60% mature poor health in the Central Plateau. The populations which had suffered 100% death were Cider Marsh and Shannon Lagoon. The next worse in health, in order, were: Pensford, Barren Tier, Todd's Corner, Barren Tier Road, Jack's Marsh, Jimmy's Marsh, Half Moon Creek, Cider Gum Road, Scrummies Marsh, Liawenee Garden and Poatina 2. Three other sites (with less than 60% mature poor health) in the region to the south-east of Great Lake were also included in figure 4.4. Arthurs Lake Road was on average healthier than the Ellis site, but this was a much smaller population. The Monpeelyata Road site was in turn healthier than the site at Arthurs Lake Road. The Ellis Plains population is of particular interest. There was no notable topographic difference, or difference in slope, aspect or noticeable ground moisture between this population and other neighbouring populations of unhealthy and dead *divaricata*. The area was currently grazed, and a recreational shooter's camp existed at the site.

There were also examples of poor health in areas of otherwise healthy populations. These included Leary's Corner and the Murchison Highway. The Murchison Highway site was a roadside population adjacent to *E. nitens* plantation, and appeared to have been recently burned. The area around Leary's Corner had also been logged in the past (Kirkpatrick *pers comm.* 2006).

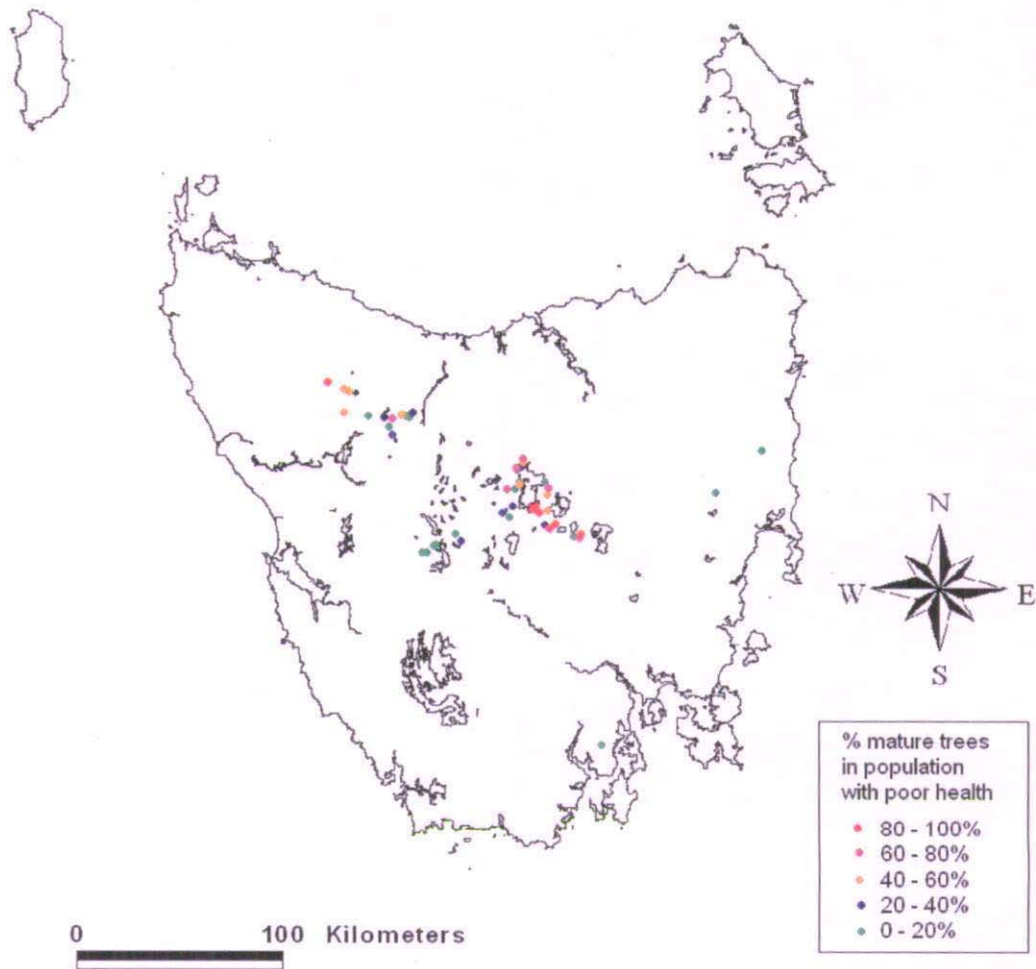


Figure 4.3a: Health status of *E. gunnii* across Tasmania

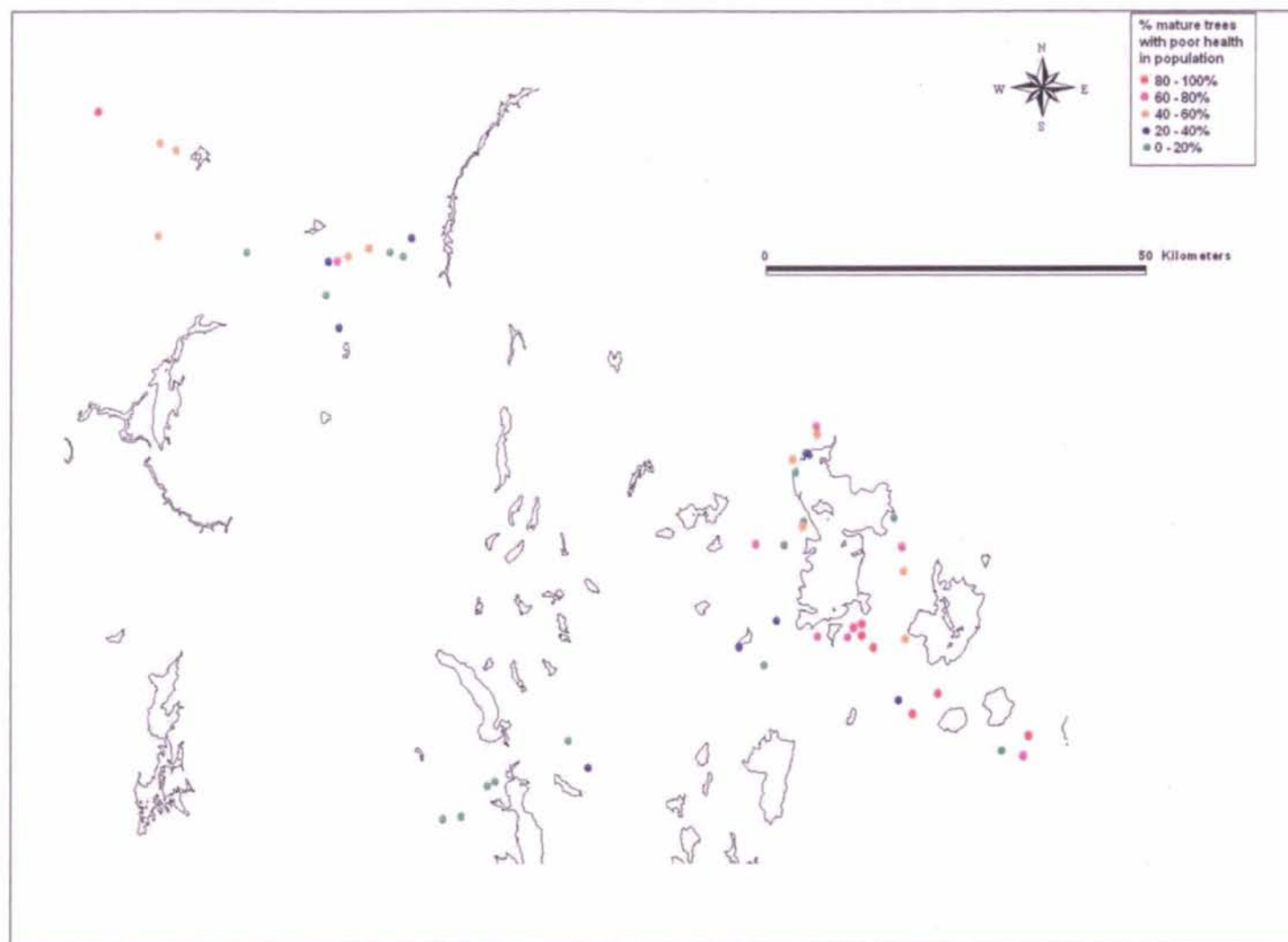


Figure 4.3b: Health status of *E. gunnii* in the central region of Tasmania

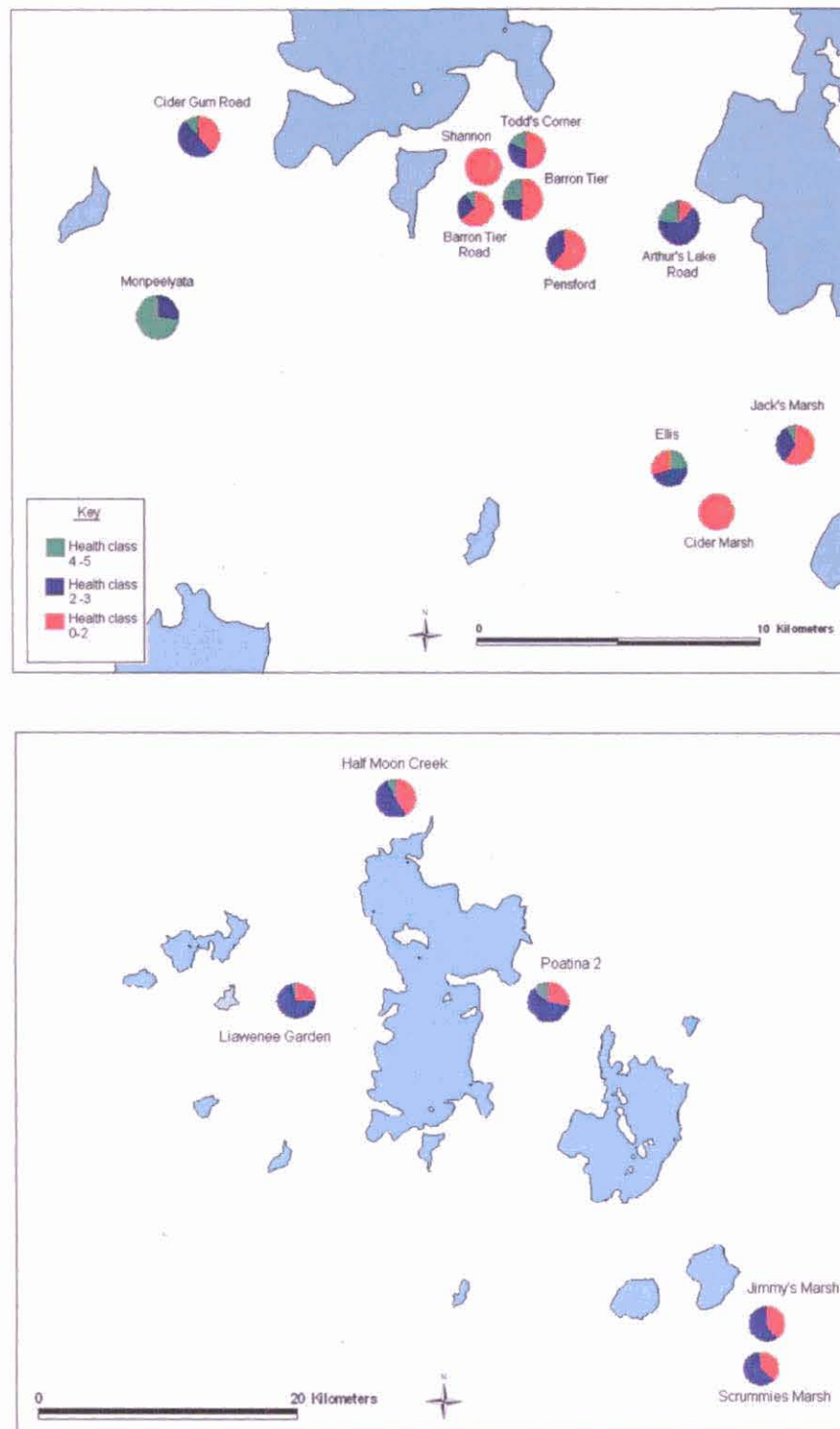


Figure 4.4: Variation in health in the region south-east of Great Lake where the worst dieback has occurred (top), and variation in the health of other Central Plateau sites where over 60% of the mature trees had health ratings less than 2 (bottom).

4.3.2. Tree and population factors related to mature tree health

Several characteristics of the trees themselves were related to poor health. Populations of a divaricate form were more likely to be unhealthy than the trees with a tall, straight form, or specimens of the mallee form (ANOVA: $F = 12.54$, $df = 2$, $P = 0.000$) (figure 4.5). Populations with a higher average basal area of mature trees were more likely to be unhealthy ($r = 0.581$, $df = 51$, $P = 0.000$) (figure 4.6). Populations with high average basal areas were also more likely to occur on private land (ANOVA: $F = 26.42$, $df = 1$, $P = 0.000$), on currently grazed land (ANOVA: $F = 66.49$, $df = 2$, $P = 0.000$) (figure 4.7) and to be of the divaricate form (ANOVA: $F = 40.92$, $df = 2$, $P = 0.000$) (figure 4.8). The estimate of basal area per hectare of *E. gunnii* was not significantly correlated with mature poor health. There were several exceptions to the general pattern of dieback discussed above. Although populations of the divaricate form were more likely to be unhealthy, 3 of these populations were in relatively good health. These were Arthurs Lake Road (plate 4.12), Monpeelyata Road (85% mature good health) (plate 4.13) and the Ellis Plains (65% mature good health) (plate 4.14).

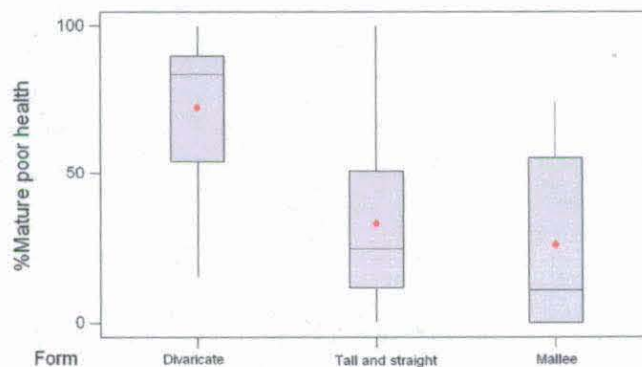


Figure 4.5: Box plots of the percentage of mature trees in poor health by tree form. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

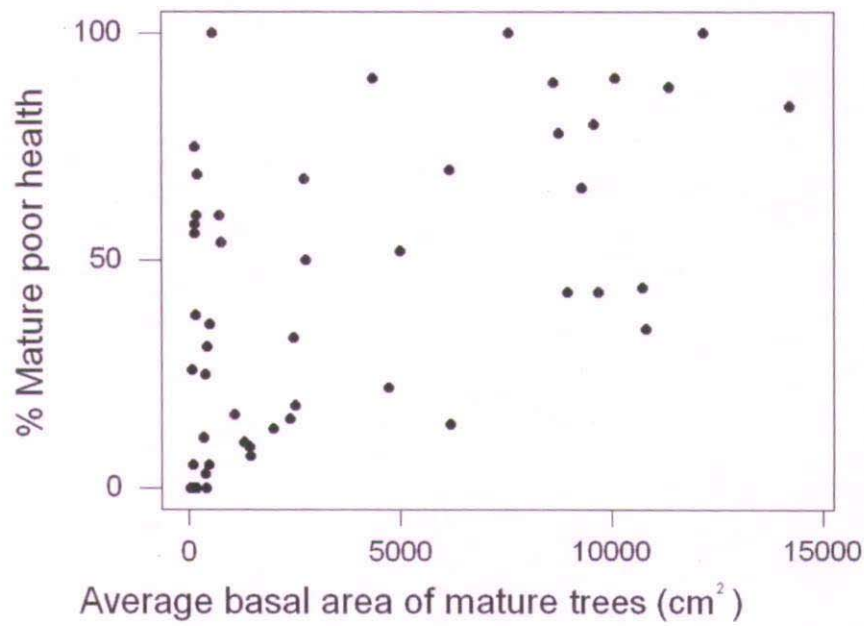


Figure 4.6: Scatter plot of the percentage of mature trees in poor health by the average basal area of the mature trees.

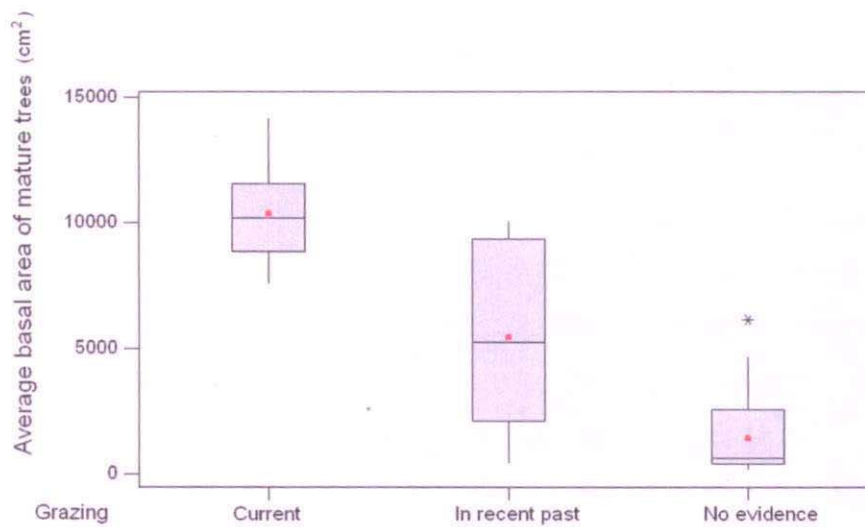


Figure 4.7: Box plots of grazing by average basal area of mature trees. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value. The asterisk represents an unusual observation.

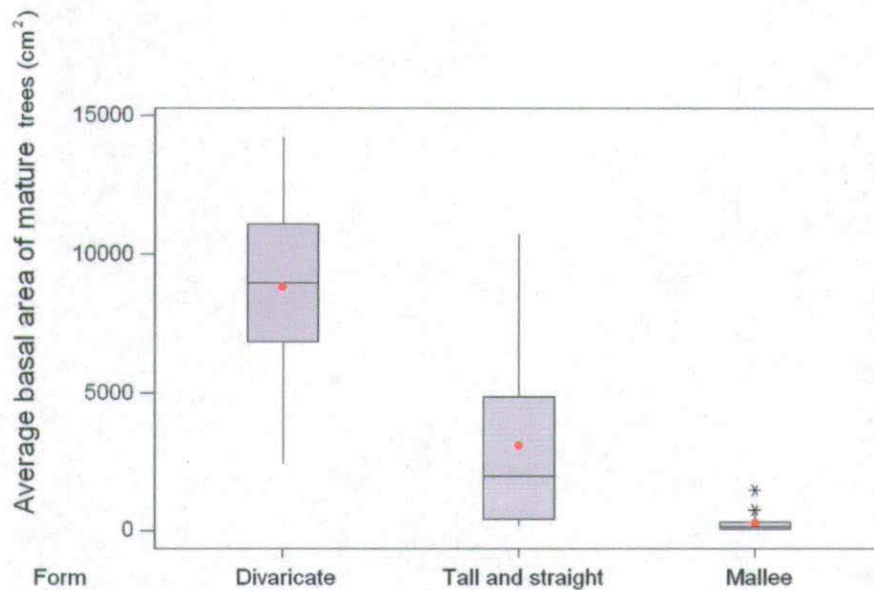


Figure 4.8. Box plots of form by average basal area of mature trees. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value. The asterisks represent unusual observations.

4.3.3 Environmental variables related to mature tree health

Populations of *E. gunnii* on private land were more likely to be unhealthy than populations in public conservation areas (ANOVA: $F = 13.91$, $df = 1$, $P = 0.000$) (figure 4.9). Additionally, populations in which there was current evidence of grazing were more likely to have unhealthy mature trees than populations where there was no evidence of grazing, or populations where grazing may have occurred in the past (ANOVA: $F = 7.89$, $df = 2$, $P = 0.001$) (figure 4.10). All sites currently or previously grazed were on private land. Populations of the divaricate form were also more likely to occur on land that was or had been used for grazing, and populations of the mallee form were more likely to occur on land that had no evidence of previous grazing ($\chi^2 = 26.231$, $df = 2$, $P = 0.000$). The divaricate form was also more likely to occur in low rainfall areas than the mallee or tall-straight form (ANOVA: $df = 2$, $F = 16.71$, $P = 0.000$) (figure 4.11).

There was a slight tendency for populations of *E. gunnii* occurring in areas of lower average annual rainfall to be unhealthy ($r = -0.296$, $df = 51$, $P = 0.031$) (figure 4.12). Removing the two most unhealthy populations in the north-west (Leary's Corner and Murchison) improved the relationship ($r = -0.401$, $df = 49$, $P = 0.004$). When average annual rainfall was divided into two categories (over 1000 mm a year and under 1000 mm a year), the populations in the wetter regions were found to be significantly healthier than those in the drier regions (ANOVA: $F = 15.16$, $df = 1$, $P = 0.000$).

A general linear model did not detect any significant interactive effect between grazing and average annual rainfall category (interactive effect in the general linear model: $F = 0.58$, $df = 2$, $P = 0.566$).

Topographic position, aspect, slope or geology did not explain any State-wide variation in the health of *E. gunnii*.

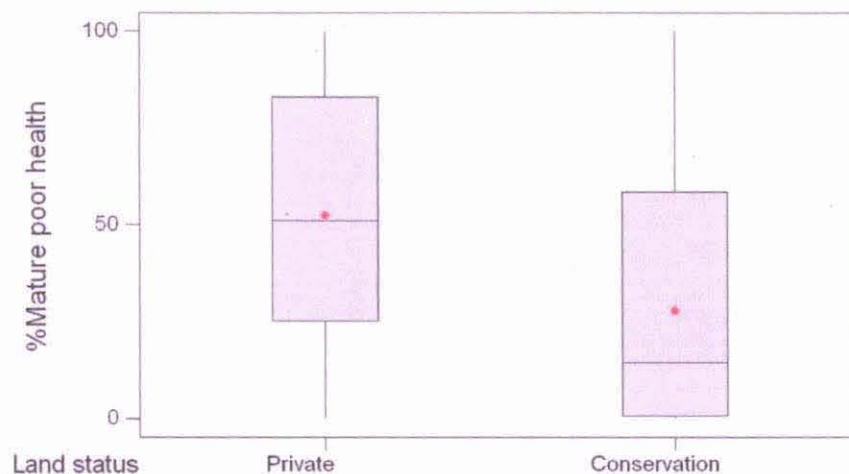


Figure 4.9: Box plots of the percentage of mature trees in poor health by land tenure. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

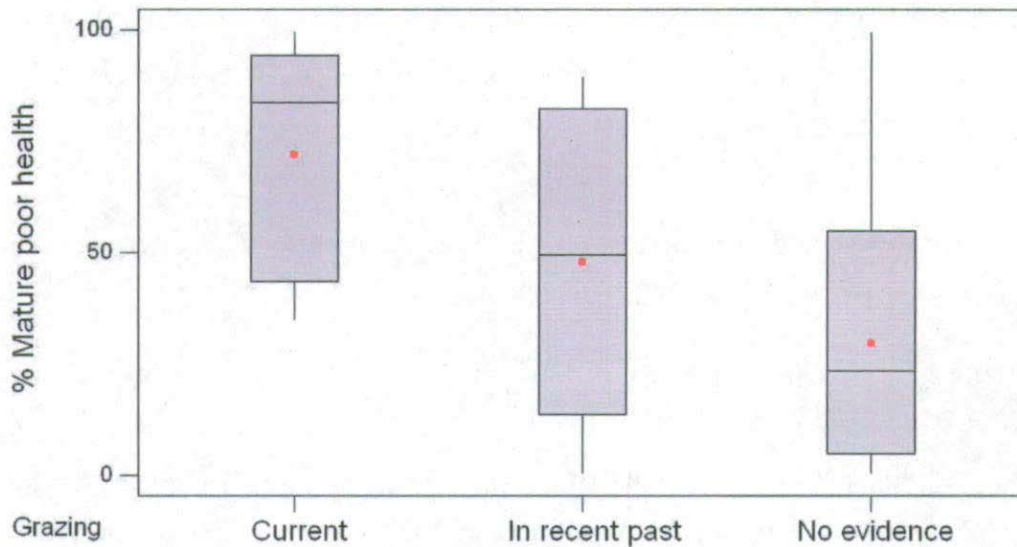


Figure 4.10: Box plots of the percentage of mature trees in poor health by grazing. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

Rows = Form (1 = divaricate, 2 = tall and straight, 3 = mallee) Columns = Grazing (1 = currently or previously grazed, 2 = no evidence of grazing).				
		1	2	All
1	Actual	12	1	13
	Expected	4.66	8.34	
2	Actual	6	15	21
	Expected	7.53	13.47	
3	Actual	1	18	19
	Expected	6.81	12.19	
	All	19	34	53
Chi-Square = 26.231, DF = 2, P-Value = 0.000				

Table 4.2: Results of chi-squared test "Form" by "Grazing".

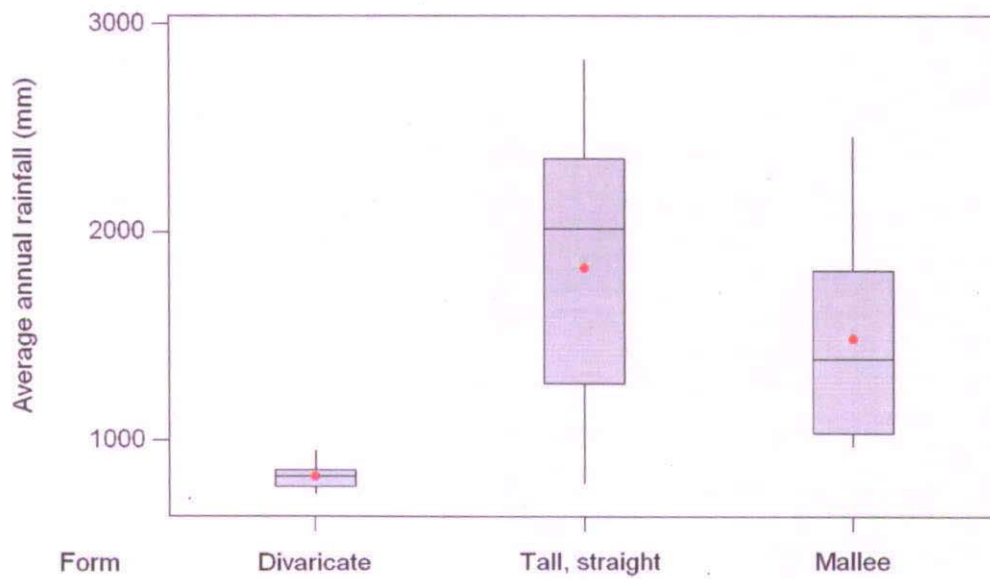


Figure 4.11: Box plots of “form” by average annual rainfall (mm). Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

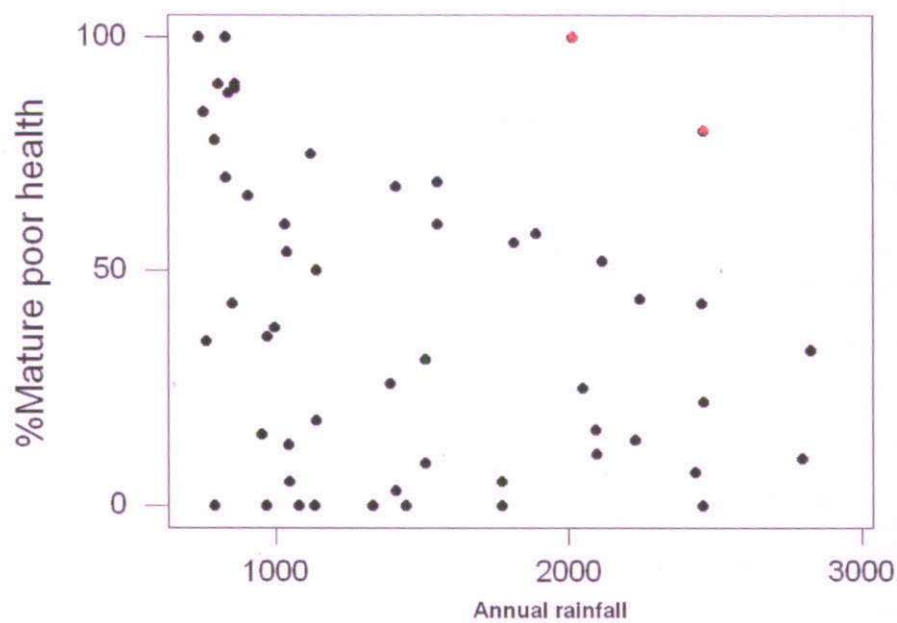


Figure 4.12: Percentage of mature trees in poor health by average annual rainfall. Red dots represent the two unusual observations at Murchison and Leary's Corner which were removed for the 2nd analysis.

4.3.4 Plant communities associated with *E. gunnii*

Five plant communities associated with *E. gunnii* were identified (figure 4.13). Community 1 was best identified by the co-dominant eucalypt *E. coccifera*, woody understorey shrubs such as *Orites revoluta*, *Grevillea australis* and *Almaleea subumbellata*, and an herbaceous layer including *Lycopodium fastigiatum*, *Veronica calycina* and *Helichrysum rutidolepsis*. Community 2 was best identified by woody shrubs *Hakea microcarpa*, *Richea acerosa*, *Hovea montana* and the cushion herb *Abrotanella forsteroides*, and, herbs such as *Senecio gunnii*, *Euchiton* sp., *Wahlenbergia* sp., and the grass *Dichelachne* sp. Community 3 was best identified by the codominant tree *Eucalyptus delegatensis*, the shrubs *Tasmannia lanceolata*, *Lomatia tinctoria* and *Bellendenia montana*, ferns such as *Blechnum pennamarina* and *Polystichum proliferum*, and the herb *Stylidium graminifolium*. Community 4 was best identified by several exotics such as *Holcus lanatus* and *Acetosella vulgaris*. This community was also identified by *Eucalyptus dalrympleana*, the grass *Austrodanthonia* sp., the herb *Veronica gracilis* and the creeping shrub *Bossiaea cordigera*. Finally, community 5 was characterised by two sedges, buttongrass (*Gymnoschoenus sphaerocephalus*) and cutting grass (*Gahnia grandis*), the shrubs *Baeckea gunniana*, *Callistemon viridiflorus* and *Bauera rubioides*, the fern *Gleichenia alpina* and; the herb *Rubus gunnianus* and the graminoid *Astelia alpina* (see Appendix 2). With some exceptions, these communities broadly separated the sites into geographical regions. Community 1 included the high altitude Central Plateau sites such as Half Moon Creek; Community 2 included the lower altitude Central Plateau sites; Community 3 were the sites in the North West; Community 4 comprised the most easterly Central Plateau sites and; Community 5 represented the sites found in the south east and north east. A 1-way ANOVA of the classificatory classes with “% mature poor health” revealed that community 5 was less likely to have unhealthy trees than any other community (ANOVA: $F = 6.02$, $df = 4$, $P = 0.001$) (figure 4.14).

The structural vegetation communities identified in the field were also significantly correlated with mature poor health (ANOVA: $F = 9$, $df = 2$, $P = 0.000$). Again the 'sedgeland understorey' communities (distinguished by *Gymnoschoenus sphaerocephalus*) had significantly healthier plants than the populations with either the grassy or shrubby understoreys, between which there was no statistically significant difference.

The health of other eucalypt species was also significantly correlated with the health of *E. gunnii* (ANOVA: $F = 10.55$, $df = 2$, $P = 0.000$) (figure 4.15). *E. gunnii* tended to have the worst health where other species were also suffering from dieback. The severity of the dieback in other eucalypt species was significantly related to annual average rainfall, with lower rainfall sites having poorer health than higher rainfall sites (ANOVA: $F = 6.48$, $df = 2$, $P = 0.003$).

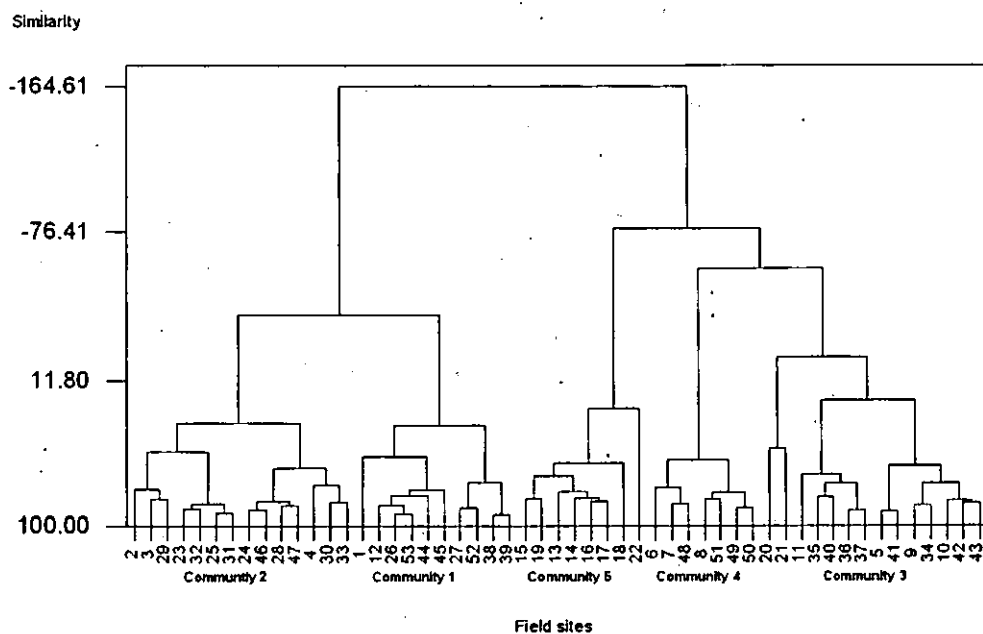


Figure 4.13 Dendrogram of floristic composition of sites. See Appendix 1 for code for site numbers.

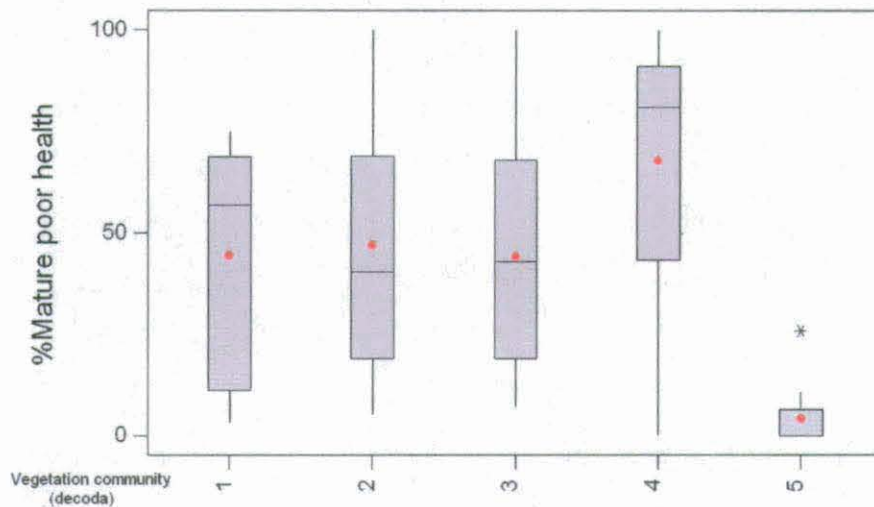


Figure 4.14: Box plots of the percentage of mature trees in poor health by floristic community. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value. The asterisk represents an unusual observation.

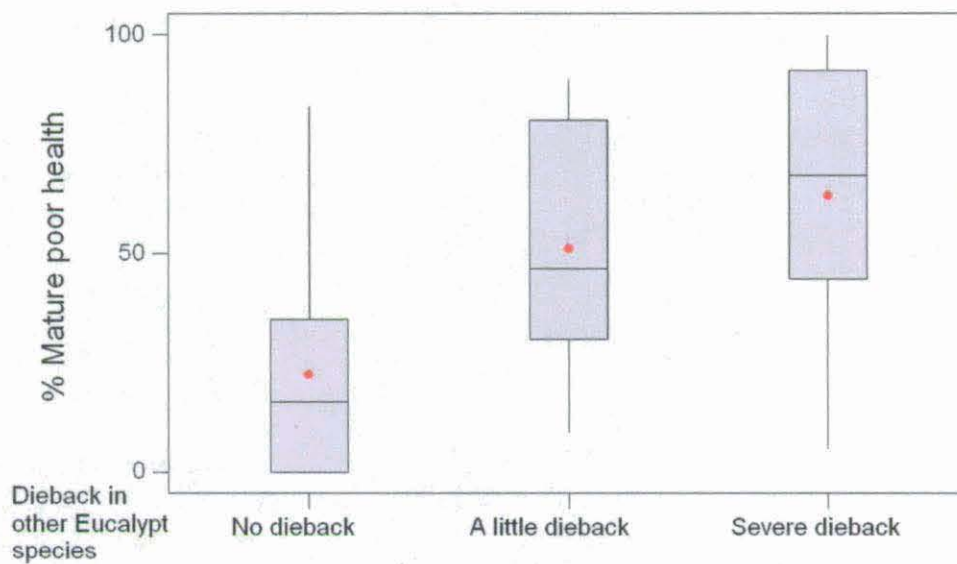


Figure 4.15: Box plots of the percentage of mature *E. gunnii* trees in poor health by dieback in other eucalypt species. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

4.3.5 Analyses within protected areas

Within protected areas, the higher altitude populations had a greater percentage of unhealthy trees ($r = 0.710$, $df = 23$, $P = 0.000$). The climatic factor most closely correlated with altitude was minimum average spring temperatures ($r = -0.779$, $df = 23$, $P = 0.000$).

Figure 4.17 shows that average minimum spring temperatures were also correlated with mature poor health ($r = -0.548$, $df = 23$, $P = 0.006$). Mature poor health was also significantly related to the degree of insect defoliation (ANOVA: $F = 5.29$, $df = 2$, $P = 0.014$) (Figure 4.18). Sites which had suffered medium to severe herbivore defoliation were more likely to be in sites which had colder spring temperatures (ANOVA: $F = 4.88$, $df = 2$, $P = 0.018$) (figure 4.19).

Moisture underfoot was significantly related to mature poor health within protected areas (ANOVA: $F = 4.46$, $df = 2$, $P = 0.024$). 'Very wet underfoot' sites tended to be healthier than sites that were a little wet, which in turn tended to be healthier than sites that were dry underfoot (figure 4.20). Sites which were wet underfoot were also more likely to have a sedgy understory than other sites ($\chi^2 = 12.185$, $df = 1$, $P = 0.000$).

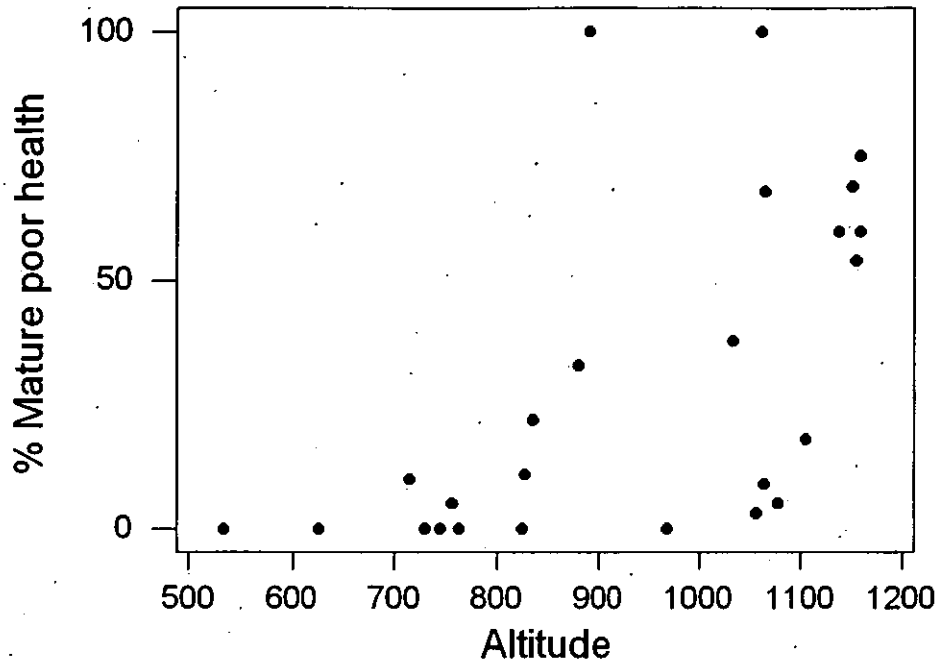


Figure 4.16: Scatter plot of the percentage of mature *E. gunnii* trees in poor health by altitude within protected areas.

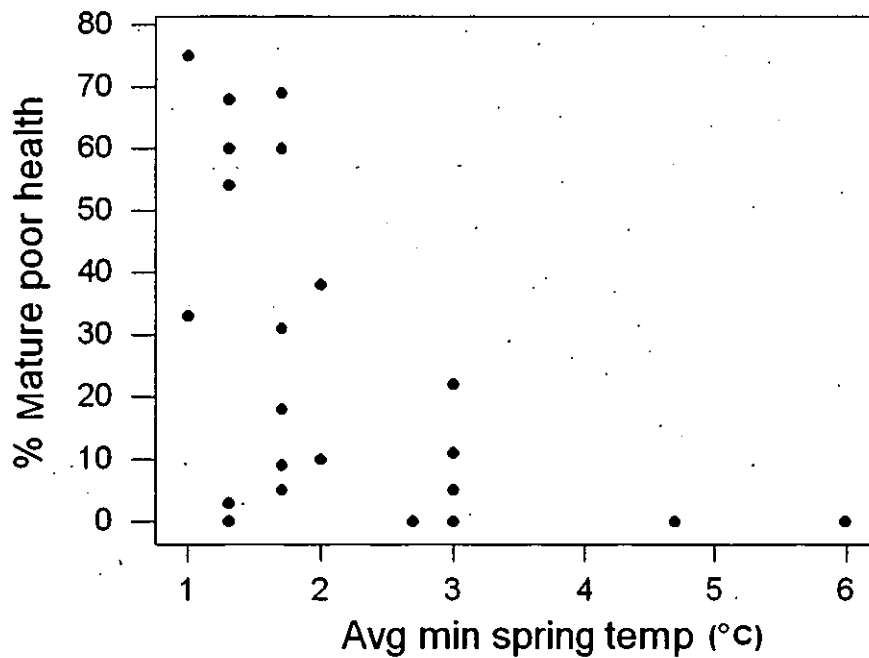


Figure 4.17: Scatter plot of the percentage of mature *E. gunnii* trees in poor health by average minimum spring temperatures, within protected areas.

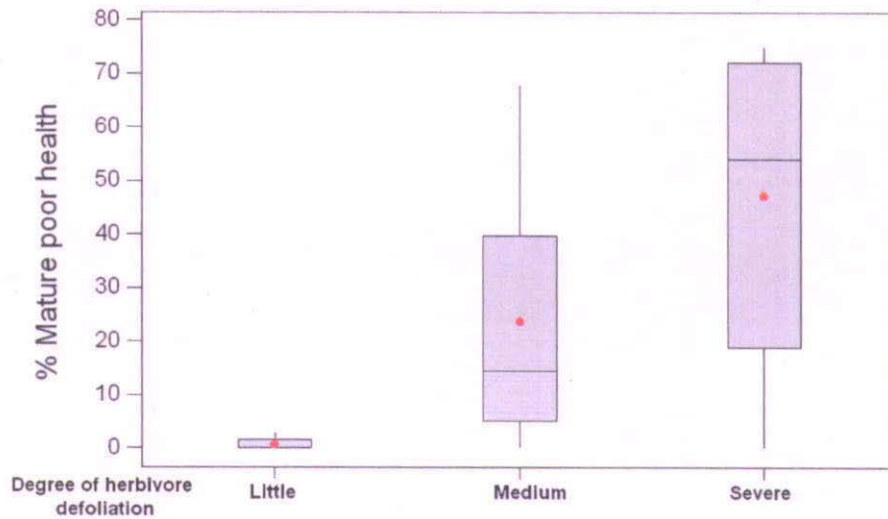


Figure 4.18: Box plots of the percentage of mature *E. gunnii* trees in poor health by the degree of herbivore defoliation, within protected areas. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

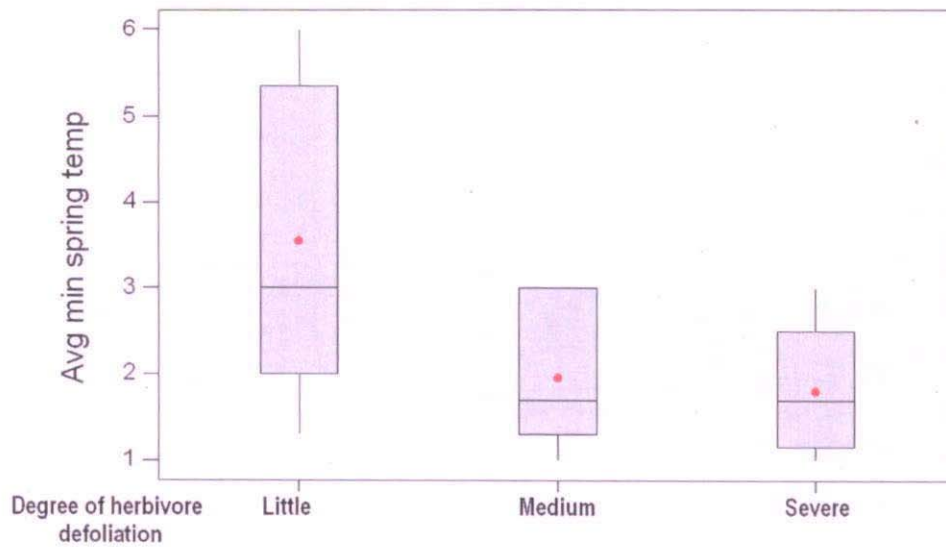


Figure 4.19. Box plots of 'minimum spring temperature' by 'degree of herbivore defoliation'. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

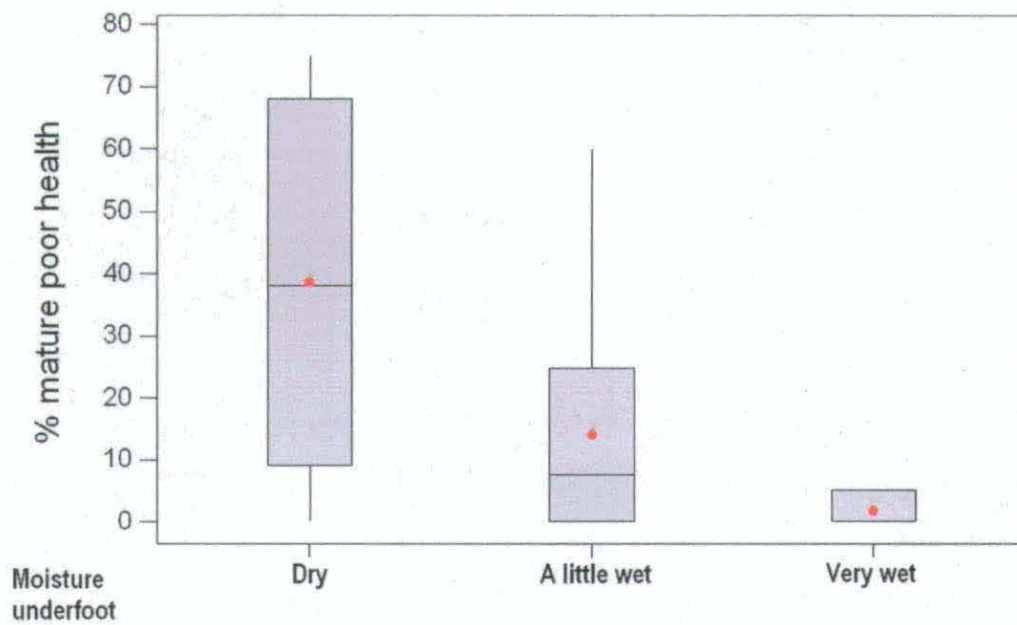


Figure 4.20. Box plots of the percentage of mature *E. gunnii* trees in poor health by 'moisture underfoot', within protected areas. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

4.3.6 Analyses within non-protected areas

Within non-protected areas, poor health was significantly correlated with slope. The gentler the slope, the greater tendency for unhealthy trees ($r = -0.444$, $df = 28$, $P = 0.016$) (figure 4.21).

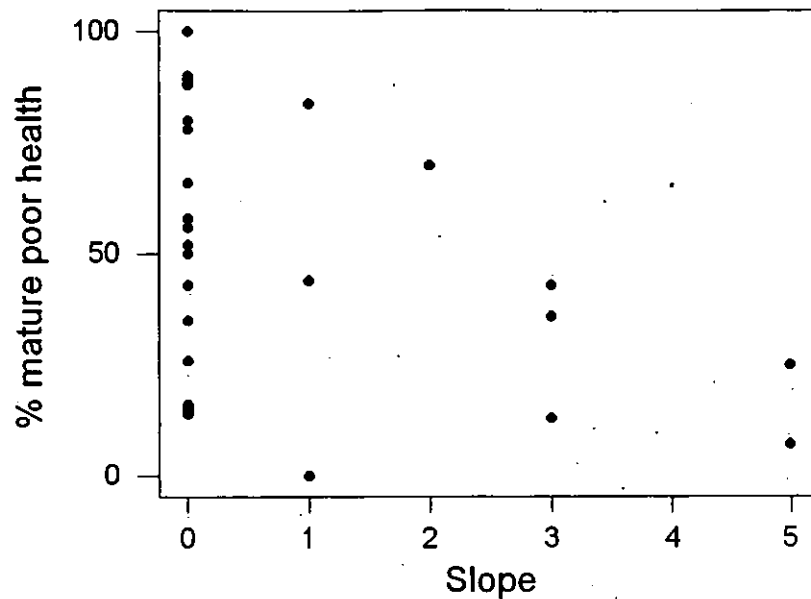


Figure 4.21: Scatter plot of the percentage of mature *E. gunnii* trees in poor health by slope, within non-protected areas.

4.3.7 Analyses within the Central Plateau

Within the Central Plateau sites, unhealthy populations were just as likely to occur on government conservation area as privately owned land. Sites which were currently grazed had a tendency to have poorer mature health than those which were not, but unlike the results of the State-wide analysis, this was not significant (ANOVA, $F = 3.35$, $df = 2$, $P = 0.051$).

There was a tendency for sites which received lower spring rainfall to have unhealthier trees in the Central Plateau ($r = -0.379$, $df = 27$, $P = 0.043$). Sites with lower average rainfall were also more likely to be grazed, or have evidence of past grazing (ANOVA: $F = 14.47$, $df = 2$, $P = 0.000$).

Unlike the results of the overall analysis, the health of other eucalypt species was not significantly correlated with the poor health of *E. gunnii* in the Central Plateau. There were sites where *E. coccifera* and *E. delegatensis* had suffered severe dieback, yet neighbouring trees of *E. gunnii* were not so affected. These sites included Monpeelyata Road (plate 4.15), Sandbanks Tier, and to a lesser extent, Doctor's Creek.

4.4 Discussion

4.4.1 Evidence for land management as the cause of dieback

The spatial pattern of dieback suggests that grazing may have had a major effect on the health of *E. gunnii*. Across all 53 sites, unhealthy stands of trees were more likely to occur on privately owned land than in public conservation areas, and stands with a higher percentage of mature poor health were more likely to occur in sites that were currently grazed. This is consistent with the findings of Gilfedder and Kirkpatrick (1998), who found that bush remnants suffering from tree decline were more likely to occur where there was also a high number of exotic species, which in turn was strongly associated with heavy and prolonged grazing pressure. Grazing of sheep and cattle on native pastures, and associated rangeland burning has occurred on most areas below 1100 m on the Central Plateau since approximately the 1820s, with the greatest stocking rates in the regions to the south and east of Great Lake (Pemberton 1986, Jetson 1989).

There is much literature on the direct affects that stock may have on tree health (Neyland 1999, Close and Davidson 2002, Jurskis 2005). Livestock tend to gather beneath paddock trees for shelter. The resulting soil compaction can cause reduced water infiltration into the soil and also damage the fine feeder roots of the trees, causing reduced water uptake and accentuating drought stress (Neyland 1999, Kirkpatrick *et al.* 2000). Exotic pastures can also restrict the water available to eucalypt roots, as they rapidly take advantage of light rainfalls and limit soil moisture recharge (Doyle 2001). If this was so in the case of *E. gunnii*, there would be expected to be an interactive effect between grazing and climate on tree health, with drier sites suffering more where

stocking rates were the highest. The analysis did not reveal this interaction.

Another possible direct effect of stock on tree health is the accumulation of livestock droppings around the base of trees, and the artificial fertilisation of pastures, which has been shown to lead to increased nutrient levels in tree foliage, resulting in higher browsing damage from possums and insects (Neyland 1999, Old 2000). The present study did not find that the degree of insect damage was correlated with grazing pressure. Instead, trees in ungrazed, protected areas had suffered some of the worst insect defoliation. However, the impact of insect defoliation was not able to be observed in trees which were already dead, and was more easily noticed in smaller trees than tall ones. Smaller trees were also more likely to have younger foliage, which is more attractive to herbivores (Landsberg 1989).

Increased nutrient levels in the soil may also have a detrimental effect on mycorrhizal fungi, which act in a symbiotic association with the tree, increasing its water uptake (Close and Davidson 2002). However, this too would be expected to produce an interaction between grazing and climate.

The theory that increased possum herbivory has contributed to severe tree decline holds greater weight. Pressure from brush-tail possum (*Trichosurus vulpecula*) browsing has been strongly implicated in eucalypt decline in the Midlands of Tasmania (Neyland 1999, Edwards 1999). Additionally, *E. gunnii* is known to be a highly palatable species (Dungey and Potts 2002). Evidence suggests that brush-tail possum numbers have undergone huge increases in rural areas in Tasmania since the collapse of the fur trade in the early 1980s (Jetson 1989, Neyland 1999). Possums feed both on eucalypt foliage and on herbs, particularly clover. The mixture of improved pasture and remnant bush areas provide possums with nutritious feed and nearby nesting sites (Neyland

1999). In the Midlands it has been noted that brush-tail possums, on a diet of clover, are now breeding twice a year where traditionally they only bred once. It has been reported that at one property on the Midlands, 13,500 possums were shot over a three month period, after which trees which had been significantly defoliated began to recover (Neyland 1999). Kirkpatrick *et al.* (2000) also showed that possum proofing stabilised or slowed the rate of deterioration of eucalypts in the Midlands. No data was collected on brush-tail possum numbers at the stands of *E. gunnii* in the present study, although possum densities would be expected to be higher in areas with agricultural land in close proximity. Possum density is also known to be higher in the low rainfall districts than elsewhere in the State (Neyland 1999). One possible explanation for the relative good health of the Ellis population is the location of a recreational shooter's camp within the population. Prior to this, there was a resident in the hut on the edge of the Ellis population who used to shoot possums for their skins (Shaw *pers. comm.* 2006). Therefore there has been a consistent effort to hunt possums over the last 25 years at least on the property. However, the possum control measures undertaken by landowners at the other sites was not investigated. Doyle (2001) also questions possums as a prime causal agent in tree decline in Tasmania, citing cases in the Midlands where dry northerly aspects suffered complete tree death, while adjacent easterly aspects were in good health. This occurred despite unrestricted possum movement. However, possums can cause a negative feedback effect on drought stressed trees, as resprouted epicormic foliage is more nutritious and attractive (Potts *et al.* 2001).

Privately owned land is also more likely to have been partially cleared, which could explain the tendency for unhealthy mature trees to occur on private land. The extent of land clearing was not included in the analysis, due to difficulty distinguishing artificial clearing from natural tree sparseness in the Central Plateau region. Some areas can be naturally open due to limiting factors in the environment such as moisture, or the rare occurrence of events such as extreme frosts (Kirkpatrick 1999). Land clearing has been implicated in dieback of

remnant paddock trees, due to changes in microclimate, particularly in extreme conditions. For example, land clearing may cause summer soil temperatures to increase, frost effects to be greater and wind speeds at ground level to increase. These can dramatically increase evaporation and transpiration rates, again accentuating drought stress (Neyland 1999). No data were collected that supported or rejected this theory for *E. gunnii*.

The populations with the largest average basal area of mature trees were more likely to be unhealthy. This may be due to higher soil fertility. While this allowed the growth of tall trees, these sites have also been used more intensively for agriculture. Sites with high average basal area of mature trees were generally on privately-owned land. Another possibility is that the correlation between average basal area of mature trees and poor health may be due to the absence of younger, more vigorous mature trees in these populations. This is discussed in chapter 5.

There was some evidence that does not support the hypothesis that land-use alone has caused the dieback of *E. gunnii*. Health in some protected areas was also quite poor, particularly in the higher altitude sites in the Central Plateau. However, due to the multiple causes of dieback, some of which may also be operating in protected areas, this evidence does not invalidate the land use theory. Changed land use and its effects may well have increased the susceptibility of *E. gunnii* to dieback, but not have been the sole cause.

4.4.2 Evidence for climate change as cause of dieback

There was a weak relationship across Tasmania between annual rainfall and health of mature trees. Sites which received lower rainfall were more likely to have unhealthy mature trees, especially the sites which received less than 1000 mm of average annual rainfall. This is consistent with the pattern of dieback than would be expected under broad-scale long-term drying of the climate. However, it cannot be said that low rainfall is directly related to dieback severity, as sites which received lower rainfall were more likely to be currently grazed, or to have evidence of past grazing. They were also more likely to have a higher numbers of brush tail possums (Neyland 1999). However, Manion (1991) explains that there is usually only a weak correlation between climate and tree health, but the relationship is nevertheless real. The reason an absolute correlation does not occur is because of the importance of sequential timing of the other contributing factors associated with tree decline. The timing of dieback will be investigated in chapter 6, which may provide insight into this.

The analyses done within protected areas did provide some possible evidence for the climate change hypothesis. Although there was no relationship between average rainfall and dieback within protected areas, sites in protected areas with a higher degree of moisture underfoot were more likely to be healthy than sites that were dry underfoot. Sites in wet microsites would be expected to be less affected by climate change, as they receive adequate supplies of ground moisture. However, this may be an auto-correlation, as sedgy understoreys often indicate poor drainage, and low nutrients (Duncan 1999). The populations with sedgy understoreys were more likely to be healthy than sites with grassy or shrubby understoreys. Wet, sedgy populations occur in low nutrient environments, and thus the foliage of the trees is not as palatable to herbivores, and the environment is unable to sustain large numbers of possums.

Microclimate and moisture accumulation vary considerably across a landscape with topography, slope and aspect (Kirkpatrick and Nunez 1980). Research and aerial observations by Doyle and Rees (2000) and Doyle (2001, 2003) suggest that variation in soil moisture with geology and soil depth in the Midlands has a significant effect on tree survival during times of drought. They observed that tree decline is most rapid where soils are shallow, where wind exposure is high, on columnar rock types such as dolerite and basalt (where the fractured nature of the bedrock leads to rapid vertical drainage of ground water), and on northerly (sunny) and westerly (windy) aspects. The present study did not detect any influence of slope, topographical position, or aspect on the health of *E. gunnii*. This may be due to the lack of variation in these factors across the range of the species. Thirty of the 50 sites sampled occurred on a more-or-less flat ground, and only 10 sites had a slope of above 2 degrees. However, to fully investigate the effect of soil moisture it would also be necessary to undertake a detailed survey to determine soil depth, the locations of geological boundaries and analyse soil drainage patterns in the landscape. Despite that fact that observations during field surveys detected no obvious topographical advantage, it is possible that the Ellis Plains population does occupy a favourable hydro-geological position (Doyle *pers comm.* 2006), or is sheltered from the most drying north-west winds (B.Potts *pers comm.* 2006). Additionally, the Shannon site, where 100% death occurred, may have been predisposed to moisture stress. The pattern of basal flow is visible in the aerial photograph of the site (see plate 6.7), indicating very shallow soils above the bedrock (Doyle *pers comm.* 2006).

The fact that flatter sites were more likely to have unhealthy trees on private land is interesting. There was no evidence that flatter sites were more heavily grazed, as might be expected due to their higher productivity. However, flatter sites would be expected to accumulate more ground moisture and thus be less susceptible to climate drying. The worse health of the *E. gunnii* stands on

private land on gentle slopes could possibly be explained by the 'gully dieback' theory (Roberts 1993). A reduction in significant rainfall events, and thus a decline in downslope run off and through-flow, can lead to a proportionally greater reduction in soil moisture lower in the landscape than higher up. Additionally, Jurskis (2005) suggests that trees growing on soils with poor drainage and aeration in rural areas are especially predisposed to decline, due to nutrient accumulation and subsequent defoliation pressures. These lower slope sites may also be more susceptible to frost. There appeared to be a pattern of dieback that could be attributable to frost-kill at Jack's Marsh. The individual trees closer to the edge of the woodland were more likely to be dead or very unhealthy than trees further up the slope. The same was true at the Iris River, where the proportion of dead trees increased downslope (plate 4.16).

Sites with a higher density of vegetation might be assumed to suffer greater water stress in times of drought, and undergo a higher degree of 'self thinning'. There was no evidence for this, as the basal area per hectare of *E. gunnii* at each site was not significantly correlated with mature poor health. However, the estimate of basal area per hectare used on this study was not a realistic measure of competition between adult trees and other vegetation as it did not include other species of eucalypts, which at some sites were co-dominant with *E. gunnii*.

Within protected areas, sites with lower average minimum temperatures (i.e. those at high altitude) were more likely to have unhealthy mature trees (plate 4.17). Dieback of other eucalypt species was also likely to be greater at higher altitude. There was also a relationship between tree health, minimum spring temperatures and herbivore defoliation. Because of their short life cycles, high reproduction and physiological sensitivity to temperature, it has been predicted that even modest climate change may have rapid impact on the distribution and abundance of many forest insects and pathogens (Ayres and Lombardero 2000, Battisti *et al.* 2005). Invertebrate pests have been shown to have a large

impact on the health of eucalypts elsewhere in Australia (Morgan and Bungey 1981, Elliot *et al.* 1981, Kile 1981, Neyland 1999), and their interaction with climatic factors has been implicated in several cases of dieback. For example, drought-stressed trees have been shown to be more susceptible to insect attack (Landsberg 1985, Lowman and Heatwole 1992, Marsh and Adams 1995). Potts (1985) found that the death of main parts of the stem of *E. gunnii* seedlings in experimental gardens mainly resulted from the effects of frost and insect predation. It is possible that insects are having a greater effect in populations of *E. gunnii* where they were previously unimportant, due to amelioration of minimum temperatures at higher altitudes. In parts of Canada, decreased winter mortality due to warmer minimum temperatures has been implicated in the catastrophic outbreaks of the native mountain pine beetles (Dale *et al.* 2001, Carroll *et al.* 2004). As well as changes in abundance, climate change may also facilitate changes in the local community composition of insects, which was found to be highly differentiated between the four stands of *E. gunnii* studied by Potts (1985). However, there is no evidence that the high degree of insect defoliation is a factor that has changed in the environment of high altitude *E. gunnii* populations. Twenty years ago, Potts (1985) also found the activity of leaf eating insects on *E. gunnii* was greater at higher altitudes, in terms of both insect abundance and damage to seedlings (figure 4.22). He also found that leaf eating insect activity was low at the sites at Pensford and Shannon Lagoon, two of the sites which have since suffered from the most severe dieback. A problem with the results of the present study is that the protected areas sites do not represent a controlled altitudinal transect, but come several different regions in Tasmania.

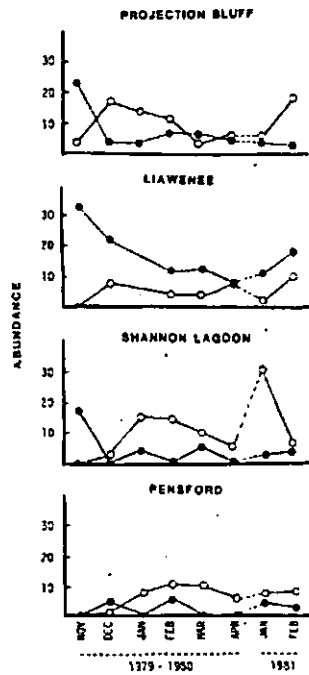


Fig. 9. The abundance (number) of Coleoptera (● leaf-eating) and Homoptera (○ sap-sucking) insects in samples collected from adult foliage at each site.

Figure 4.22: Abundance of leaf eating and sap sucking insects on *E. gunnii* adults at four different sites in the Central Plateau: Projection Bluff (at an altitude of 1100 m), Liawenee (1150 m), Shannon Lagoon (1050 m) and Pensford (960 m) (from Potts 1985).

Additional to the effects of drying, climate change may also increase in the competitive ability of other species. Warming of the climate has been postulated as being able to facilitate the upward movement in altitude of *E. rodwayi*, a species that usually replaces *E. gunnii* on the margins of poorly drained flats at lower altitudes (Kirkpatrick and Gibson 1999). Loehle (1998) points out that tree species are generally not limited by warmth, but by superior performance of competing species in warmer environments. However, *E. gunnii* was more likely to be healthy where other species of eucalypt were also healthy, and to be suffering from dieback where other species of eucalypt were also suffering from dieback. There were also some cases where *E. gunnii* was healthier than its neighbours in the Central Plateau (e.g. Monpeelyata Road, Sandbanks Tier). It is possible that landscape position may have played a role in the relative health of *E. gunnii* at Sandbanks and Monpeelyata. At both sites, *E. gunnii* occupied a distinct drainage hollow in the landscape and would have

received more soil moisture than the surrounding species of eucalypts, growing on steeper, rockier slopes. Replacement of one eucalypt species with another is likely to be a very slow process, and out-competition of one species by another will occur in the seedling stage. Worth (2002) found that at the Shannon Lagoon population, at the boundary between *E. gunnii* and *E. pauciflora*, the more drought-tolerant cabbage gum seedlings were in far greater numbers than the cider gum, suggesting that over time the stand may come to be dominated by *E. pauciflora*.

The fact that dieback of neighbouring eucalypt species generally co-occurred with dieback of *E. gunnii*, suggests that across Tasmania, the stress factors that are operating on *E. gunnii* have also had an affect on the other species. The dieback of other species across Tasmania was significantly correlated with average annual rainfall, with drier sites suffering greater dieback. This strongly implicates the effect of localised and severe droughts.

4.4.3 Genetic variation within the species and inherent susceptibility of *E. gunnii* spp. *divaricata*.

Eucalyptus gunnii has a relatively wide climatic range that spans approximately 5°C of annual average temperature and 2000 mm of average annual rainfall. The hypothesis examined in this chapter is that climatic drying would have a greater effect on the populations at the driest edge of the species' climatic range. Under this hypothesis it is unlikely that the decline in the wetter parts of the species' range is due to climate change. However, this does not account for intraspecific variation in drought tolerance within *E. gunnii*. Variation in the drought tolerance of *Eucalyptus viminalis* has been demonstrated by seedling trials (Ladiges 1974). Seedlings from low rainfall populations were more resistant to desiccation than those from the high rainfall

populations. Potts (1985) found that *Eucalyptus gunnii* varies both phenotypically and genetically across its range, and the clinal variation observed in the Central Plateau may be at least partially due to differential response to drought. If so, it is likely that the populations growing in the Eastern Central Plateau belong to the most drought-tolerant ecotype. Therefore populations in wetter areas could also suffer from precipitation reductions below the average for that area.

The populations informally identified as *E. gunnii* spp. *divaricata* occurred within a much narrower climatic envelope of 1°C in annual average temperature, and 25 mm of annual average rainfall. In theory, this makes the subspecies highly susceptible to climate change (Hughes *et al.* 1996). However, it must be noted that the climatic envelope may not accurately represent the species' thermal or rain tolerances, as trees can survive in places that would be not be expected from the low rainfall, relying on groundwater supplies (Boardman 1994). Nevertheless, the results show that the epicentre of mature-tree dieback is concentrated at the type location of the endangered *divaricata* subspecies. Populations of the divaricate form were also more likely to be unhealthy. The present study did not formally differentiate between the subspecies in terms of the phenetic classes identified by Potts and Read (1985), which included multi-character features such as glaucousness, leaf shape and size and oil composition in the juvenile leaves. However, the form of the trees is likely to reflect both genetic differences and its response to the environment (Potts 1985). The fact that the divaricate form was more likely to be suffering from poor health may reflect two possibilities. The first is that the subspecies *divaricata* may be genetically less able to cope with drought stress than *E. gunnii* spp. *gunnii*. However, trees of the divaricate form were also more likely to occur in areas on the eastern side of the Central Plateau that had been heavily used for stock grazing, had lower average annual rainfall and were in more exposed environments. Therefore the more likely explanation is that these populations have been exposed to greater stress, in the form of rural land

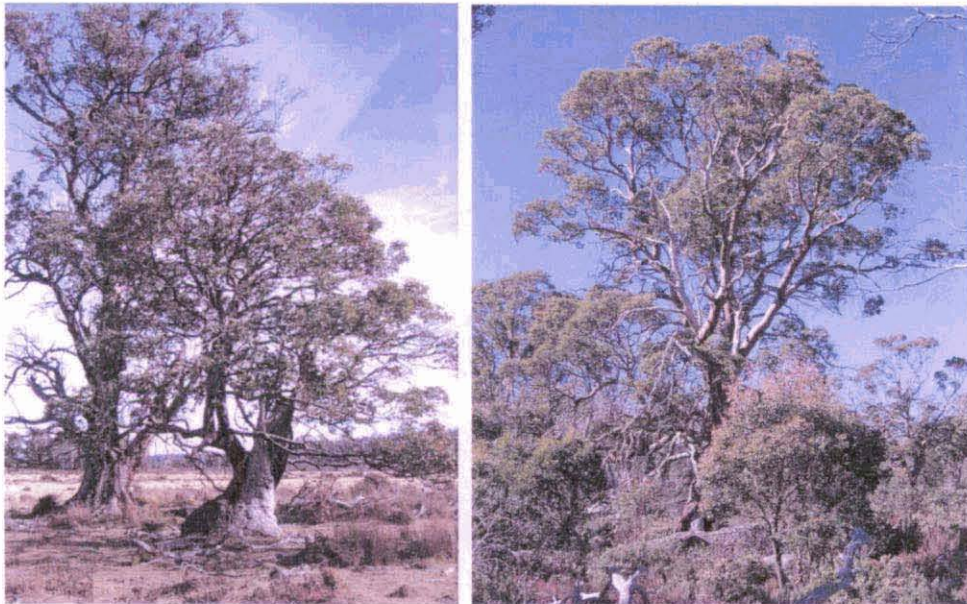
use, drought and frost than the forms growing in higher rainfall regions.

4.4.4. Conclusions

The results of the field surveys demonstrate that the dieback of *Eucalyptus gunnii* is a complex problem, most likely to do with the additive effect of various stress factors. This is consistent with the majority of studies into dieback. Several land-use and climatic factors were significantly related to the health status of *E. gunnii*. However, most of these variables were highly cross-correlated, making it difficult to distinguish whether climatic or land-use variables have had a greater effect. For example, sites closer to the precipitation range limit of *E. gunnii* generally did have worse health, yet this was just as likely to be due to the higher intensity of land use in these areas as climate change. Several high rainfall sites also had severe dieback, suggesting that land use or other changes have been the major cause here. The importance of the influence of local factors is demonstrated by the fact that the dieback was not distributed evenly over the landscape. The intensity and type of land use varies across the landscape, as does the genetic make up of *E. gunnii*, the community structure, the insect fauna, climate, and moisture accumulation. Each population has a different ecological history and local landscape position, which may have either buffered them from, or exposed them to the various stress factors.

A likely explanation is that the pressures associated with stock grazing have increased the susceptibility of the trees to decline. However, as demonstrated by the dieback in the protected areas, and the relative health of some populations in grazed areas, the negative effects of grazing are not the primary cause of dieback. It remains very likely that the increased intensity and frequency of droughts, due to climate change, may have put considerable stress

on trees. Additionally, the fact that the main differentiating factor at the Ellis population seems to be the history of possum control, gives strong evidence that increased possum numbers have played a large role in the decline of trees in the Eastern Central Plateau. However, this hypothesis is only based on anecdotal evidence and would require further research. It is also possible that the population occupies an advantageous hydrogeological position. There is also a possibility that other facet of climate change, temperature increase, may be the reason behind the high levels of insect defoliation at high altitudes in protected areas. Evidence would be needed to prove that this has changed in the last 20 years.



Plates 4.12 and 4.13 Healthy mature trees at Arthurs Lake Road (left) and Monpeelyata Road (right)



Plate 4.14: Healthy trees at the Ellis Plains

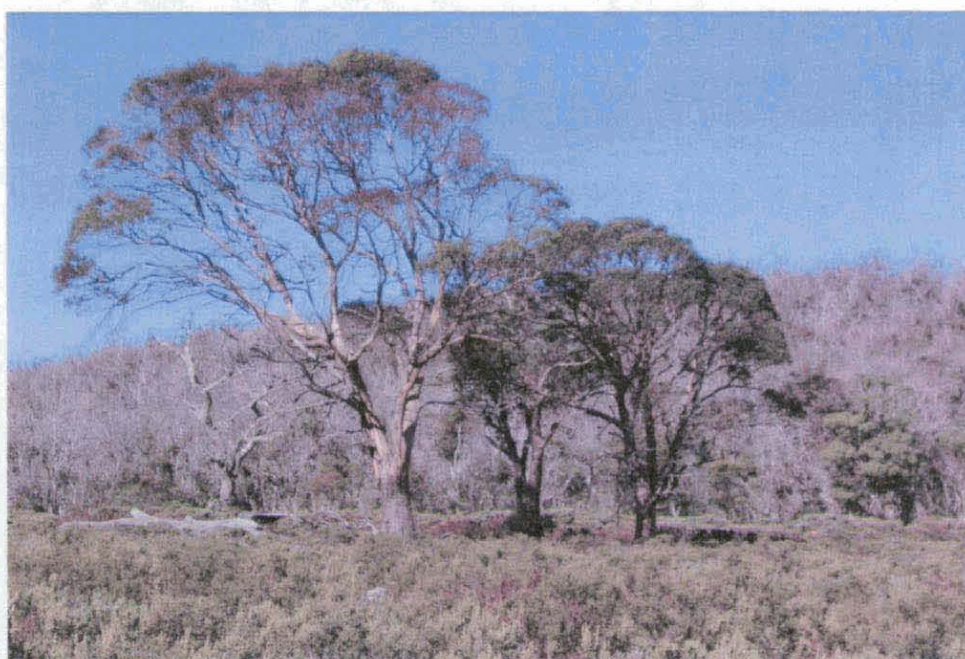


Plate 4.15: Dieback of other eucalypts and relative health of *E. gunnii* at Monpeelyata Road



Plate 4.16: Differential dieback of *E. gunnii* on either side of the road at the Iris River, Middlesex Plains



Plate 4.17: Poor health at high altitude in the Central Plateau (1160 m)

Chapter 5: The spatial pattern and environmental correlates of regeneration success



5.1 Introduction

The long term future of a population is affected by its birth, growth and death rates (Schemske 1994). The recent dieback of *Eucalyptus gunnii* spp. *divaricata* has been noted as particularly significant, as many populations have no or very few juveniles available to replace the senescing older cohort (Potts *et al.* 2001). Seeds of *Eucalyptus* are typically short-lived and dispersed over only short distances (Reid and Potts 1999). Therefore, if young trees are unable to establish and grow to reproductive maturity in a population, the species may become locally extinct when the mature trees die. The only way to re-establish is by gradual encroachment from surrounding populations.

The presence of juveniles is firstly influenced by the amount and viability of seed produced by the mature trees. This can depend on their age and health status. If there is a viable source of seed, and germination is not inhibited, the successful growth of seedlings can be influenced by both land use factors such as grazing (Fensham and Kirkpatrick 1992), and climatic factors such as drought and frost. Several authors predict that climate change may have a pronounced effect on the regeneration success of trees in forests and woodlands, even if it does not have an immediate effect on mature trees with well established root systems. This is because seedlings can have radically different requirements for establishment and growth than mature trees (Walker 1991, Brown and Wu 1998, Hansen *et al.* 2001, Alewell and Bredemeier 2005), and the seedling stage is often the most vulnerable stage of growth (Ashton 2000). This is highly relevant in Australia, as the successful regeneration of eucalypts is an episodic event over much of the country, and can be driven either directly or indirectly by climate. The timing and amount of rainfall, or extreme temperature events, can be critical to seedling survival (Ashton 2000, Sutherst and Floyd 1999). Thus, although the mature trees may survive, climate and land use changes may decrease regeneration success, leaving behind only a few 'senescing veterans' (Hebda 1998).

The demographic structure of a population may also provide some insight into the cause of dieback. Dieback often affects small trees more than large ones (Roberts 1993, Kirkpatrick *et al.* 2000). This is because larger trees have generally gained a superior position in the local landscape, and have a more extensive and thus competitive root system. In some cases, dieback of smaller trees may indicate a natural process of 'self-thinning', especially in forests and woodlands where there appears to have been an increase in tree density because of changes in disturbance regimes (Kirkpatrick *et al.* 2000).

This chapter will present and discuss the spatial patterns of the demographic structure of 53 stands of *E. gunnii*. It will explore the environmental correlates associated with juvenile success, the demographic structure of populations and the reproductive vigour of the mature trees. It will then discuss whether the results provide evidence that climate change or land use changes have had a greater impact on the regeneration success of *E. gunnii*.

5.2 Methods

Field surveys of the health of 53 stands of *E. gunnii* were undertaken across Tasmania. Fifty trees were sampled at each site, using the 'nearest neighbor' method, as described in chapter 4. Each tree was measured for basal area at breast height, except for short or multi-branched trees, which were measured below the level where the branches split into more than two. Beginning with a mature tree, the first 30 trees sampled were of any age. Juveniles were only recorded if they were above approximately 20 cm in height, as smaller plants may have been obscured by shrubs and frequently missed. In order to include at least 20 mature trees at each site, sampling of interstitial juveniles was abandoned at 30. From the first 30 trees sampled, an estimate of the ratio of mature individuals to juveniles in a population was calculated (% juveniles). The full details of the field sites and of the sampling methods used are presented in chapter 4.

In the following analyses, the populations with an average of over 1.5 stems per plant were excluded (mainly the mallee populations). This is because size is not an appropriate proxy for age in these populations. For the remaining 35 populations, the basal area of the mature trees was averaged. The coefficient of variation for mature tree size was also calculated, to give an indication of the diversity in population

size structure. Tree size classes were defined, to compare the health of individual trees with their size, and to determine the size class distributions within populations. Trees with a basal area below 20 cm² were put in the juvenile age class. A basal area of 20 cm² roughly coincides to a 5 cm diameter. The other four size classes were basal areas 200 – 800 cm² (roughly 5 to 30 cm in diameter), 800 to 3200 cm² (roughly 30 to 60 cm diameter) and over 3200 cm² (roughly over 100 cm in diameter).

Using binoculars, a rough estimate of the percentage of branches in the mature trees with buds and/or capsules was recorded, to give an indication of reproductive vigor. However, due to the tendency of binoculars to fog up in wet or cloudy weather, the estimates gained for the taller trees in poorer weather were less reliable than for trees under 2 m in height, or trees that were observed in fair weather. A regression analysis was performed between individual tree health and reproductive vigour.

The environmental characteristics that were noted at each site are presented in chapter 4. MINITAB was used to perform 1-way ANOVA and to calculate the Pearson's product moment correlation coefficient to test the significance of relationships between the percent of juveniles in a population, the coefficient of variation in mature tree size, and environmental variables.

5.3 Results

5.3.1. Spatial pattern of regeneration success of *E. gunnii*

Figure 5.1 shows the distribution of populations with different proportions of juveniles. Coinciding with the dieback epicentre (chapter 4), there were several populations with no or very low numbers of juveniles to the south east of Great Lake. There was also one population with very few juveniles to the north-west of Great Lake (Half Moon Creek). Most populations in the north-west of the state had high numbers of juveniles, except for the population between Daisy Dell and Iris, which had no juveniles. This population was currently grazed by cattle.

At some sites there were unhealthy older trees, but many healthy juveniles. These sites included Leary's Corner, the Iris River, Barren Tier Road and Barren Tier (Appendix 1).

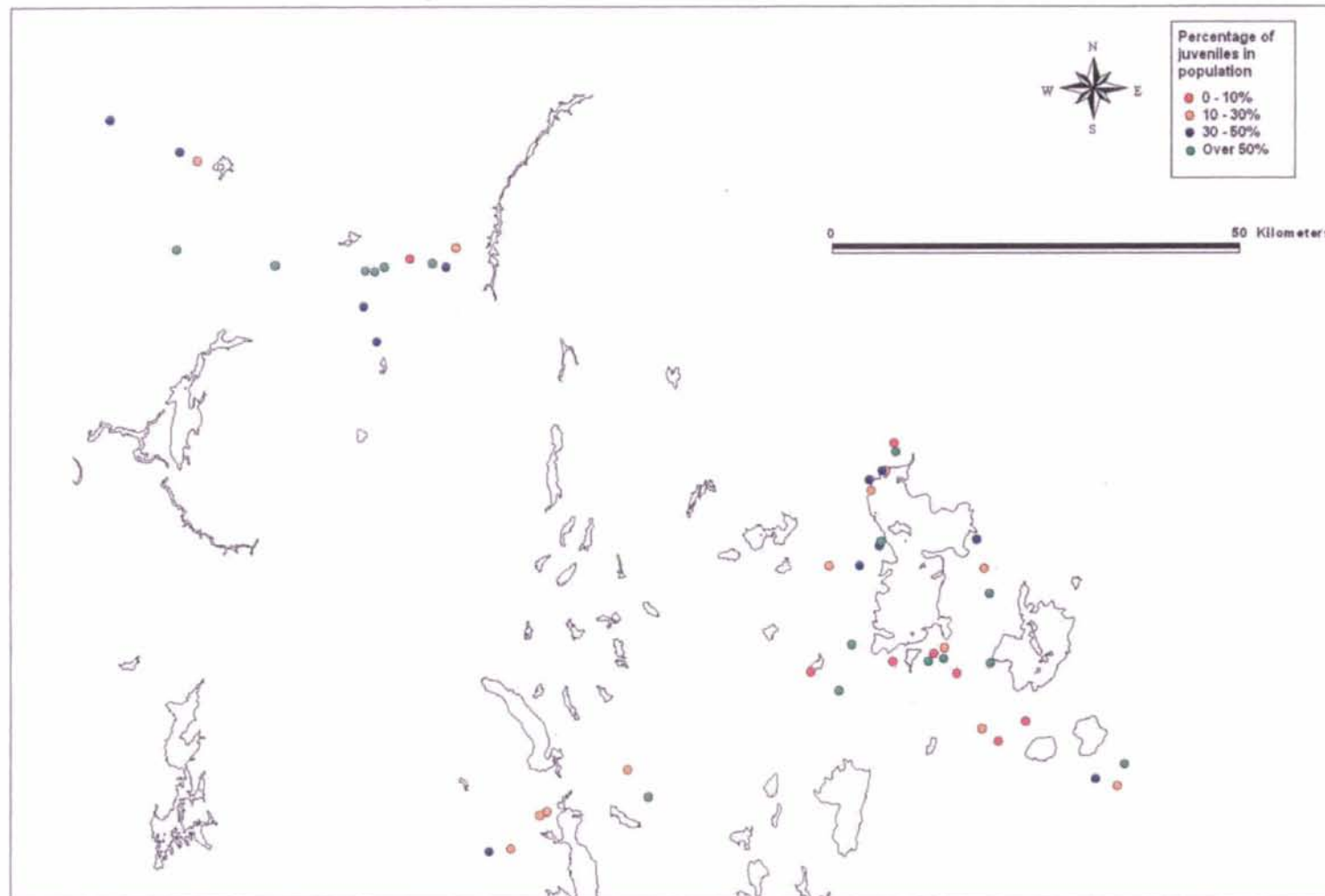


Figure 5.1: The percent of juveniles in populations of *E. gunnii* in the central region of Tasmania

5.3.2 Environmental correlates with juvenile success

There were significantly less juveniles in sites that were currently grazed, than in sites that were not (ANOVA: $F = 10.1$, $df = 2$, $P = 0.000$) (figure 5.2). The negative impact of grazing on juvenile numbers was further demonstrated at Arthur's Lake Road, where very few juveniles occurred inside a paddock fence, yet there were large numbers of juveniles outside the fence. In four populations (Cider Marsh, Shannon Lagoon, Between Daisy Dell and Iris River and Jack's Marsh) no juveniles were recorded. All of these four sites were currently used for stock grazing. Several juveniles were seen at Shannon Lagoon, but these had all been grazed back to below 20 cm in height and therefore not recorded. One small juvenile was seen but unrecorded at Cider Marsh, and no juveniles were seen at Jack's Marsh or the site between Daisy Dell and the Iris River on the Middlesex Plains. The majority of sites with very low numbers of juveniles were also in grazed areas, such as Cider Gum Road, the Ellis Plains, Pensford, Scrummies Marsh and Todd's Corner (appendix 1). Additionally, vegetation community 4 (see chapter 4) had the lowest number of juveniles (ANOVA: $F = 2.82$, $df = 4$, $P = 0.035$). This vegetation community had the most exotic species.

There was no significant relationship between the percentage of mature trees suffering from poor health, and the number of juveniles in a population ($P = 0.319$). There was a significant relationship between individual tree health and reproductive status ($r = 0.392$, $df = 1336$, $P = 0.000$). Below a health rating of 2.5, mature trees were likely to have lower reproductive vigour (figure 5.3). However, the proportion of mature trees reproducing was not significantly correlated with the percentage of juveniles in a population ($P = 0.663$).

No climatic variables were significantly correlated with % juveniles in a population.

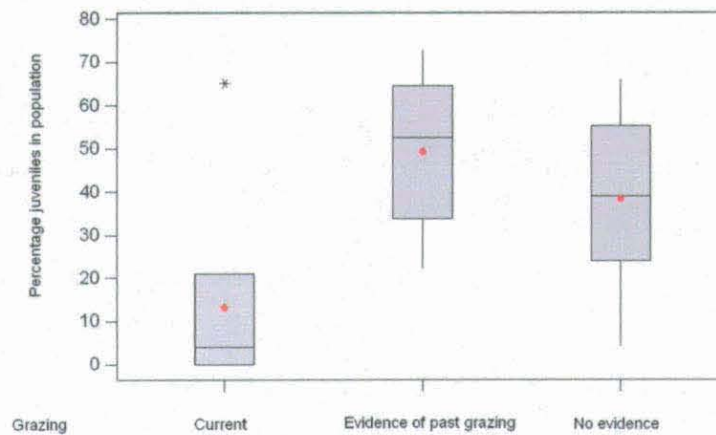


Figure 5.2: Box plots of the percentage of juveniles in a population by grazing. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

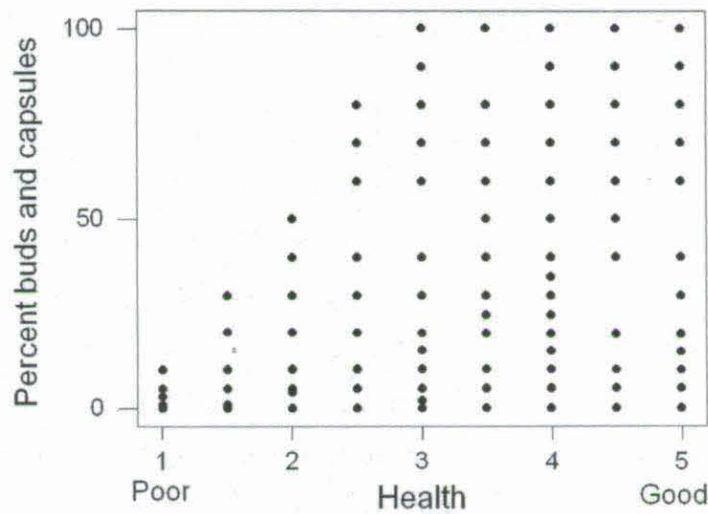


Figure 5.3: Scatter plot of the percentage of branches with buds and capsules in individual mature trees, compared to their crown health rating.

5.3.3. Population age structure and health

Populations with low variability in age structure tended to have poor mature health (figure 5.6) ($r = 0.681$, $df = 34$, $P = 0.000$) (excluding the mallee populations). The populations with the 5 largest average basal areas (Jack's Marsh, Cider Marsh, Between Daisy Dell and Iris, Pensford and the Ellis Plains) all had predominantly large mature trees, and very few mature trees of smaller size. Populations with lower average basal area tended to show a different pattern of distribution between size classes to those with higher average basal areas, with a higher proportion of smaller and medium sized trees (see figures 5.4 and 5.5). The coefficient of variation in mature basal area was also related to the percent of mature trees with poor health ($r = .0.535$, $df = 34$, $P = 0.001$) (figure 5.6). Populations with low variation in mature tree size were more likely to occur on private land (ANOVA: $F = 14.76$, $df = 1$, $P = 0.001$), and on currently grazed land (ANOVA: $F = 10.19$, $df = 2$, $P = 0.001$). Figure 5.7 shows that sites which were currently grazed were more likely to have lower variation in size structure than sites which were grazed in the past, which had lower variation than sites with no evidence of grazing. Populations with low variation in age structure were also more likely occur in grassy rather than shrubby vegetation communities (ANOVA: $F = 9.05$, $df = 1$, $P = 0.004$) (figure 5.8), and to be of the divaricate form (ANOVA: $F = 7.99$, $df = 1$, $P = 0.008$).

The larger the individual tree, the more likely it was to be unhealthy. Table 5.1 shows that there were fewer small trees and more large trees in poor health classes than expected ($\chi^2 = 311.965$, $df = 20$, $P = 0.000$).

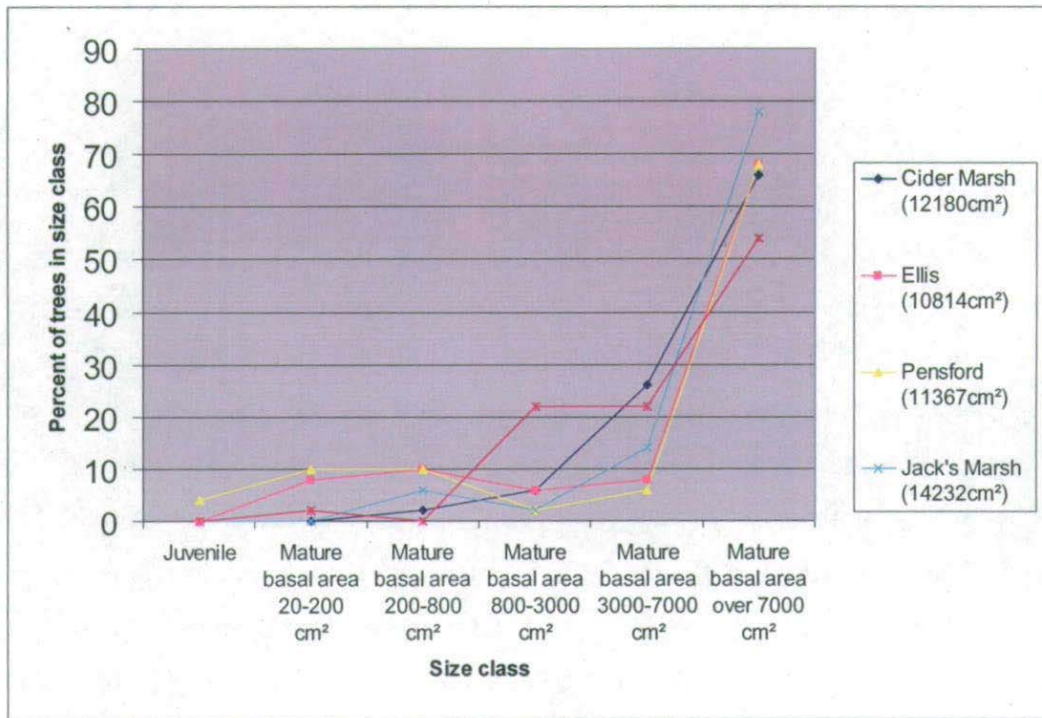


Figure 5.4: The percentage of juvenile trees and of mature trees in different size classes, in 5 populations with high average basal area of mature trees (cm²).

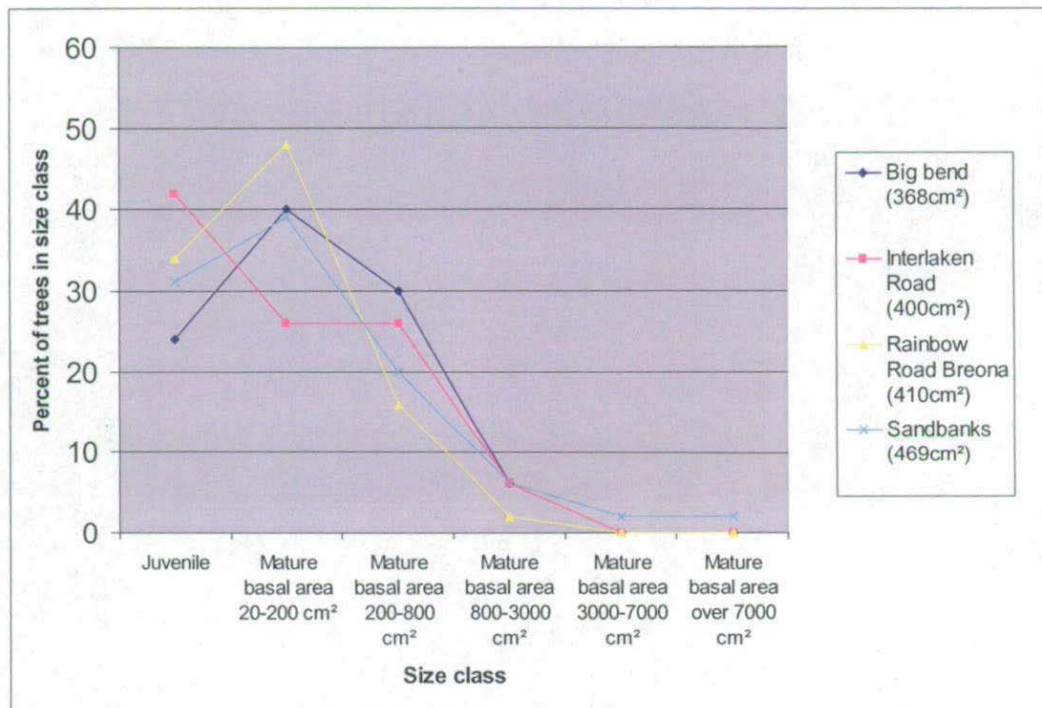


Figure 5.5: The percentage of juvenile trees and of mature trees in different size classes, in 5 populations with low average basal area of mature trees (cm²).

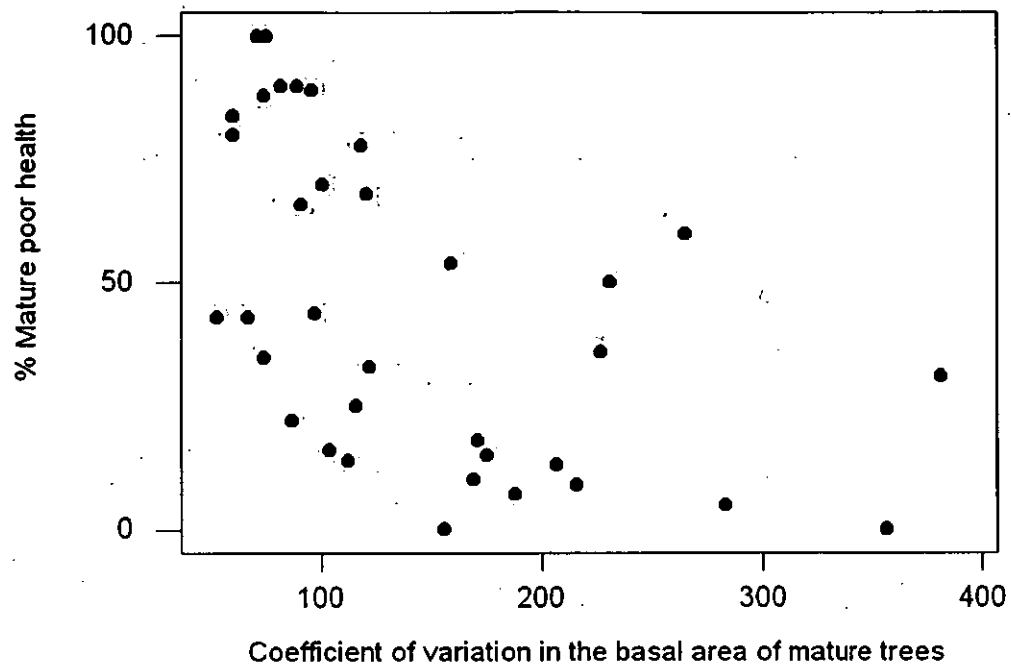


Figure 5.6: Scatter plot of the percentage of mature *E. gunnii* trees in poor health by the coefficient of variation in basal area of mature trees within a population (excluding the mallee populations).

Columns = Health class (0-5), Rows = Size class								
		0	1	2	3	4	5	All
20 – 200 cm ²	Actual	31	73	172	232	232	68	808
	Expected	98.66	95.92	167.63	190.47	199.6	55.72	
	Difference	-67.66	-22.92	4.37	41.53	32.4	12.28	
200 – 800 cm ²	Actual	16	39	37	54	90	40	276
	Expected	33.7	32.76	57.26	65.06	68.18	19.03	
	Difference	-17.7	6.24	-20.26	-11.06	21.82	20.97	
800 – 3000 cm ²	Actual	14	20	28	33	43	9	147
	Expected	17.95	17.45	30.5	34.65	36.31	10.14	
	Difference	-3.95	2.55	-2.5	-1.65	6.69	-1.14	
3000 – 7000 cm ²	Actual	51	24	39	30	23	1	168
	Expected	20.51	19.94	34.85	39.6	41.5	11.59	
	Difference	30.49	4.06	4.15	-9.6	-18.5	-10.59	
7000 cm ² +	Actual	104	54	91	68	49	4	370
	Expected	45.18	43.92	76.76	87.22	91.4	25.52	
	Difference	58.82	10.08	14.24	-19.22	-42.4	-21.52	
All		216	210	367	417	437	122	1769
Chi-Square = 311.965, DF = 20, P=Value = 0.000								

Table 5.1: Chi-squared test for relationship between size class and health class of individual *Eucalyptus gunnii* trees.

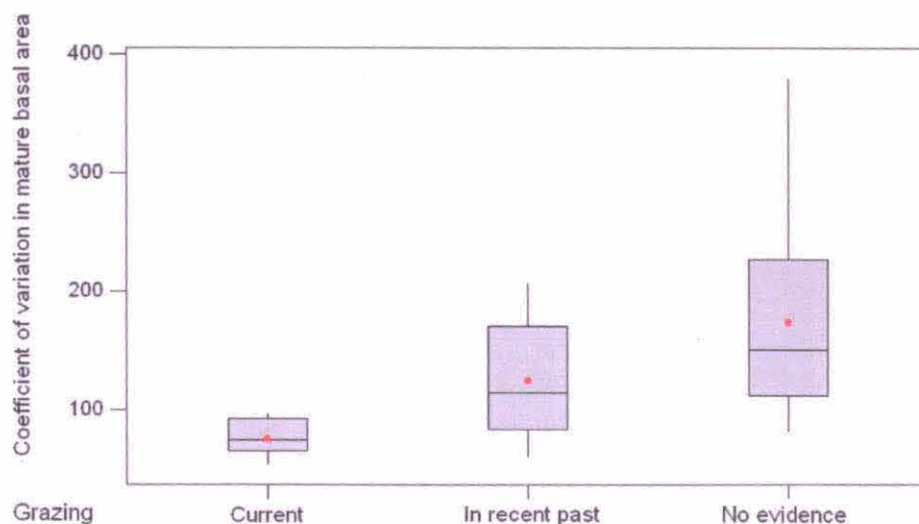


Figure 5.7: Box plots of the coefficient of variation in mature tree basal area size in populations and the level of grazing. Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value.

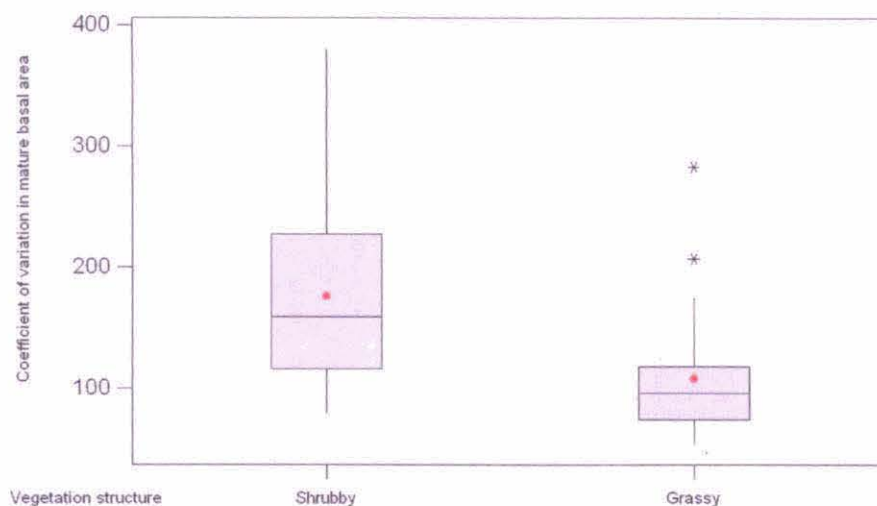


Figure 5.8: Box plots of the coefficient of variation in mature tree basal area size in populations and the structural vegetation community in which populations were found (excluding the mallee populations with sedgey understoreys). Means are indicated by dots, boxes show the interquartile range and the central lines indicate the median value. Asterisks represent unusual observations.

5.4 Discussion

5.4.1 Juveniles, grazing and climate change

There was strong evidence that stock grazing is the primary reason behind the lack of regeneration success in many populations. Sites which were currently grazed had the fewest juveniles, and the juveniles that were present at these sites were generally stunted and had resprouted from lignotubers (plate 5.1), and were mainly associated with shrubs (plate 5.2). The strong association of juveniles with shrubs at Shannon Lagoon was also found by Gibson and Kirkpatrick (1989) and Worth (2002). The most likely explanation is that the often rigid and prickly foliage of the shrubs (e.g. *Hakea microcarpa*) provide protection from grazing. Further support for the argument that grazing has the greatest impact on juvenile survival, is that the number of juveniles was unrelated to the number of mature trees that were reproductively active. This suggests their presence has been more inhibited by grazing than by lack of seed. The influence of grazing on eucalypt regeneration has been previously demonstrated by Fensham and Kirkpatrick (1992). They found that in grassy woodlands grazing increases the probability of seedling establishment by reducing the grass sward. However, the subsequent constant pruning greatly increases the probability of seedling mortality.

The present study found that sites which had been grazed in the past had on average a higher percent of juveniles than those which had no evidence of grazing, although this was not statistically significant. However, this could demonstrate that the disturbance created by past grazing has created sites suitable for early establishment, and the discontinuation of grazing enabled the seedlings to persist.

There was no correlation between percent of juveniles and climatic variables. However, microclimatic conditions and extreme events such as frosts are more likely to be more important for seedling survival than the average conditions (Kirkpatrick and Gibson 1999). In 1979, *E. gunnii* seedlings were transplanted in fenced experimental gardens at four different sites in the Central Plateau (Potts 1985). The high seedling mortality rates recorded by Worth (2002) at the Shannon garden, (figure 5.9), demonstrate that agents other than vertebrate grazing have contributed to juvenile mortality. Potts (1985) found the Shannon Lagoon site to have higher maxima, lower minima, and more damaging frosts than any of his other sites. The high rates of seedling mortality in the 1979-1981 and 1981-1982 periods (5.9) correspond with episodes of drought, which is thought to be the major cause of mortality at the Shannon site (Potts 1985). Another theory proposed by Worth (2002), is that the association of juveniles with shrubs may also be due to the greater water availability in the shrub habitat (Worth 2002). The amount of water that penetrates below the root system of a dense grass sward can be very limited. Therefore, even in areas of high annual rainfall, seasonal drought may cause seedling mortality in a grassy understorey. Insects were also found to be a significant factor in the defoliation of seedlings in the higher altitude sites (Potts 1985).

It would also be expected that the likelihood of seedling establishment in populations would decrease as the dieback of the mature trees progresses (Potts *et al.* 2001). As with clearing, the environment would be subject to more extremes of temperature, wind and frost, and there would also be a reduced supply of seed. It was shown in the present study that once trees reach a certain dieback stage, their reproductive output begins to decline. The increased isolation of mature flowering trees also results in reduced outcrossing rates, and thus inbreeding depression (Hardner *et al.* 1996, Potts *et al.* 2001, Worth 2002). It has been shown that *E. gunnii* seedlings derived from self-fertilisation have poorer vigour and survival than seedling derived from unrelated crossing (Potts *et al.* 1987). The mean number of

seeds per capsule, mean seed weight, and mean percentage of seed germination is also likely to be lower as a result of self-pollination (e.g. Burrows 2000). However, juvenile numbers were not shown to be related to mature tree health, suggesting different factors have operated on mature and juvenile plants.

The strong association between juvenile numbers and stock grazing suggests that grazing by uncontained animals, such as wallabies, possums, rabbits and deer has not had the same impact as sheep. However it is also likely that the highest concentrations of most grazing animals also occur in the regions of the most intense agricultural use, due to higher fertility and the ability of improved pastures to support larger populations.

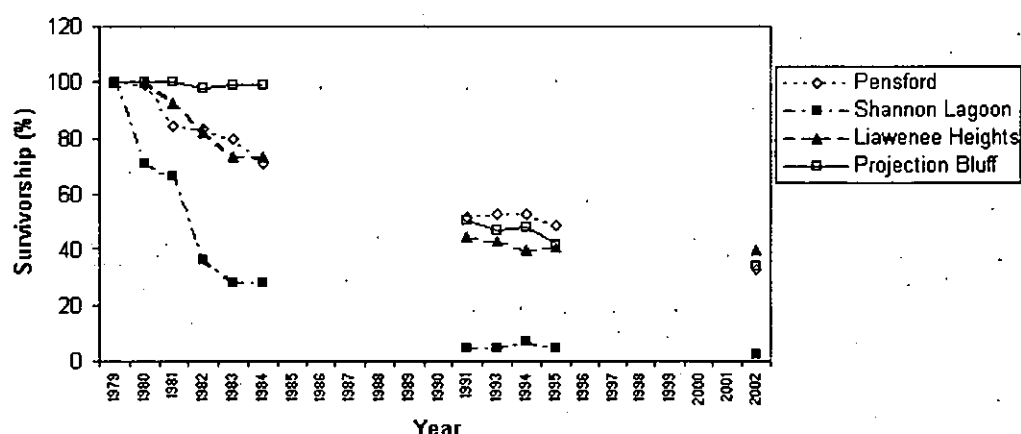


Figure 5.9: The percentage of all seedlings surviving between 1979 and 2002 in four gardens (Shannon Lagoon, Liawenee High, Projection Bluff and Pensford). No records of survival were undertaken between 1985- 1990 and 1996-2001, respectively. Note that slight increases in survival are due to the regrowth of individuals back from lignotuber. (Source, Worth, 2002).

5.4.2 Population age structure and health

Sites which were currently grazed had significantly higher average basal areas of mature trees, and a lower variation in mature tree size than sites which were grazed in the past, or had no evidence of past grazing. Three of the sites identified as examples of relative good health in the dieback region in chapter 4 (Sandbanks Tier, Monpeelyata Road and Interlaken Road), were all populations with greater mature size diversity, having higher numbers of smaller and medium sized trees than other populations. The lack of a younger cohort available to replace ageing mature trees, which may be nearing the end of their natural lifespan, was mentioned by Potts *et al* (2001), as a characteristic of the sites worst affected by dieback. Gibson and Kirkpatrick (1989) also predicted that heavily grazed *E. gunnii* woodlands would inevitably change to grasslands at the end of the natural lifespan of the existing trees, if grazing was not removed for a period sufficient for seedlings grow large enough to escape grazing. The Central Plateau region has been subjected to grazing of sheep and cattle on native pastures since approximately the 1820s, particularly in the region to the south and east of Great Lake (Pemberton 1986, Jetson 1989). Many of the older *divaricata* trees were cited by Potts *et al* (2001) as being overmature in the 1970s. Therefore the decline of some populations may be due to the natural senescence of the old trees, and the lack of replacement with a younger generation. Sites with low diversity in age structure, and more large trees were more likely to be found on grazed land, and in grassy habitats. This may be a case of cross-correlation. Grassy woodland communities often indicate relatively high soil fertility (Kirkpatrick 1999), and are thus likely to be more intensively used for agricultural purposes. Seedling establishment can also be naturally low in grassy environments, as the dense sward of grass roots impedes seedling root penetration and the amount of water that infiltrates below the grass roots can be very low. Environments can generally support either a large number of smaller trees, or a small number of larger trees (i.e. a thicket or a woodland). Fire, grazing and disturbance history are

important in determining which of these two alternatives occupies a particular site (Kirkpatrick 1999). The reason behind the relationship between low diversity in age structure and poor health may therefore be due to several interactive factors. Firstly, the low diversity in age structure of the populations may be a natural characteristic of a grassy woodland community. Secondly, the relative fertility of the grassy woodland environments is more attractive to livestock graziers and hence the sites have suffered from more intense and prolonged grazing pressure, which may have had a direct effect on mature tree health. Thirdly, the prolonged grazing in the region may have lead to the absence of a younger cohort available to replace the aging trees.

5.4.3 Distribution of dieback between size-classes

The results suggest that the current dieback of *E. gunnii* has affected the larger sized trees more than the small ones. This is contradictory to what has been found by other studies into dieback (e.g. Roberts 1993, Kirkpatrick *et al.* 2000). Bigger trees are usually growing in the best micro-sites and have better developed root systems, so are at a competitive advantage to younger trees during times of drought (Pook and Forrester 1984, Bowman and Kirkpatrick 1985). However, there may be several reasons for the concentration of large trees in the poor health class. Firstly, large dead trees in the landscape can be more persistent than dead-saplings. This may have skewed the results to present a proportionally greater mortality of old trees to young trees than actually occurred. Some of the populations with poor mature health had no small trees. These may have died previously, or they may never have existed due to the long-term suppression of juveniles by grazing pressure. Additionally, the populations with trees in the largest size class were predominantly in the privately owned and grazed areas, which have suffered from the worst dieback. It is also possible that the stands which had a large number of young healthy trees (e.g.

Interlaken, Lake Augusta Road, Sandbanks), have not yet begun to exert competitive stress on each other. This does not generally happen until about 30 years of age in eucalypts (Davidson *pers. comm.* 2006).

Nevertheless, the greater proportion of large trees to small trees in poor health classes, and the fact that the proportion of juveniles in a population was not correlated with the percentage of mature trees suffering from poor health, may indicate that different stresses have operated upon young trees and adults. Factors that could have differentially affected adults include: logging of large trees, therefore allowing the proliferation in juveniles compared to adults in some environments and; the increased susceptibility of old trees to decline. Large trees are more likely to be healthy until they reach the age of natural decline, when their youthful recuperative vigour is lost (Manion 1991). Old age may well be a significant factor in the decline, if grazing has inhibited the long-term regeneration success of the woodlands, leaving nothing to replace the 'senescing veterans'. However, it must also be noted that the analysis between the size class and health class included every mature tree sampled during field work. Within this, each population may have its own story, which may have been masked by the overall trend.

5.4.4 Conclusions

There is little doubt that grazing by stock has the greatest effect on the poor regeneration success of *E. gunnii*. However, other environmental stresses such as drought and frost may have combined to decrease the survival of juveniles in some populations. It is also possible that in some populations, the hindrance to regeneration over two hundred years of stock grazing has lead to the absence of a younger cohort to replace the old, naturally senescing trees. Overall, the findings of

this chapter do not contradict the recommendations of Worth (2002), that seedlings be protected from grazing, and that the development of the shrub layer be promoted.



Plate 5.1: An isolated, multi-stemmed seedling found at the Shannon population



Plate 5.2: Many seedlings growing amongst a shrubby understory protected from stock grazing at Barren Tier

Chapter 6: Timing of the onset and rate of dieback

6.1 Introduction

Aerial photographs have been used by many researchers to study changes in the extent and severity of tree dieback over time (e.g. Clark *et al.* 1981, Myers 1981, Spencer 1984, Dezzio *et al.* 1997, Herwitz *et al.* 2000). Determining the timing of onset and the rate of tree loss can provide clues as to its cause.

This chapter will review historical aerial photographs to determine the approximate timing of the onset of dieback at different sites, and the rate of decline. It will then explore whether the timing of decline coincides with events such as the droughts identified in chapter 3.

6.2 Methods

A time series of aerial photographs were obtained from 10 locations (figure 6.1, appendix 3). These were:

- Cider Marsh: 1968, 1992, 1997 and 2002
- Jack's Marsh: 1992, 1997 and 2002
- Near Jack's Marsh: 1968, 1992, 1997 and 2002
- Ellis: 1968, 1992, 1997 and 2002
- Shannon: Unidentified year in the 1950s, 1984, 1989, 1993, 1997 and 2006
- Todd's Corner: Unidentified year in the 1950s, 1984, 1989, 1993, 1997 and 2006
- Barren Tier: Unidentified year in the 1950s, 1984, 1989, 1993, 1997 and 2006
- Iris River: 1953, 1985, 1992, 2006
- Middlesex Plains: 1953, 1985, 1992, 2006
- Leary's Corner: 1985, 1992, 2006

Each photograph was scanned to digital format at a resolution of 800 dots per inch. The photographs were cropped to include the same area in each photograph over the time series. The earliest photograph was printed, and the location of all living trees was copied to tracing paper. This sketch map was compared to the next photograph in the time series, and trees which had died or were no longer present since the first photograph were marked. The living trees in each photograph in the time series were counted. Scatter plots combining the sites in closest geographical proximity were produced.

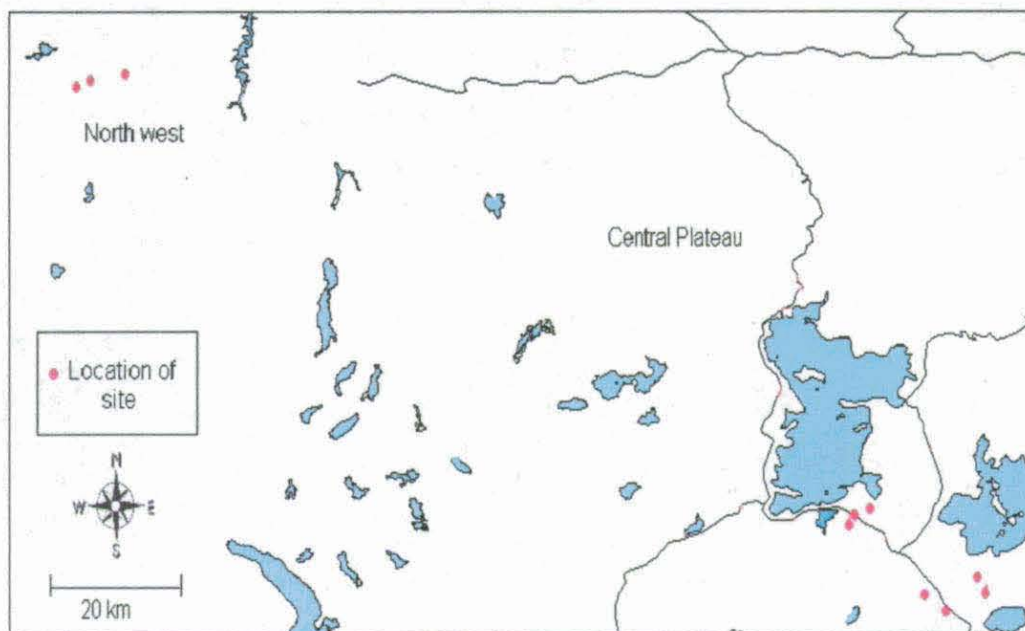


Figure 6.1: Location of sites for which aerial photographs were obtained: In North West from the left: Leary's Corner, Iris River and Middlesex Plains; At Central Plateau from most westerly site in clockwise direction: Barren Tier Road, Shannon Lagoon, Todd's Corner, Near Jack's Marsh, Jack's Marsh, Cider Marsh and Ellis.

6.3 Results

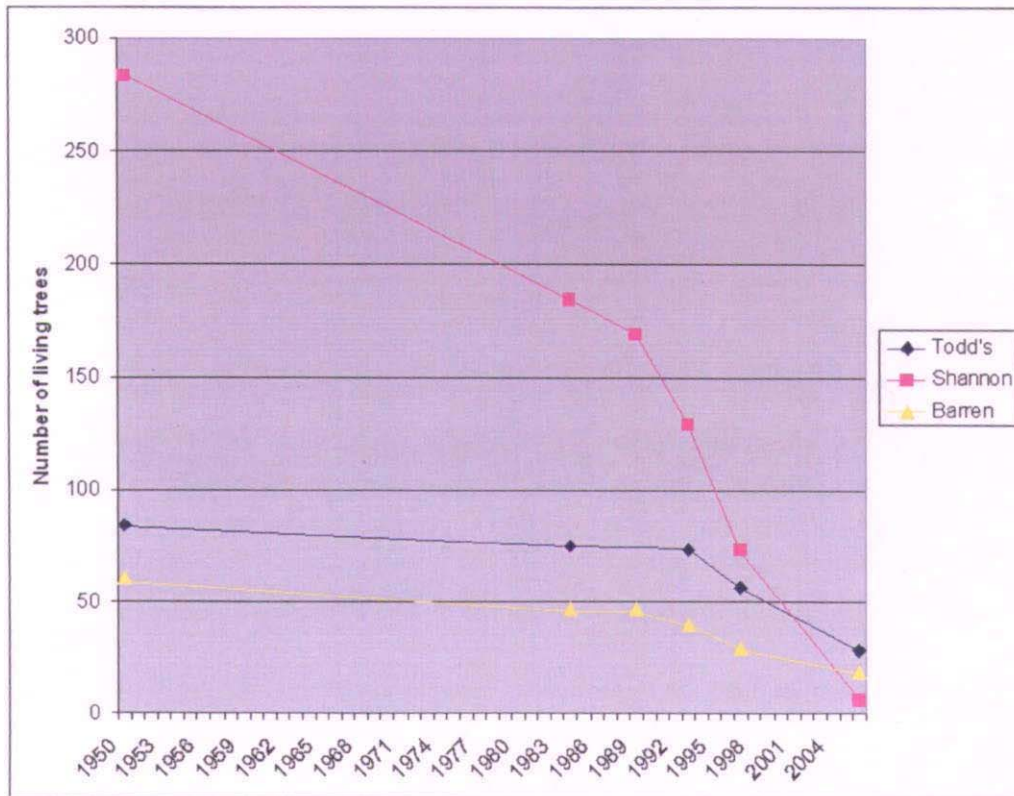


Figure 6.2: Number of live trees by time at Todd's Corner, Shannon Lagoon and Barren Tier

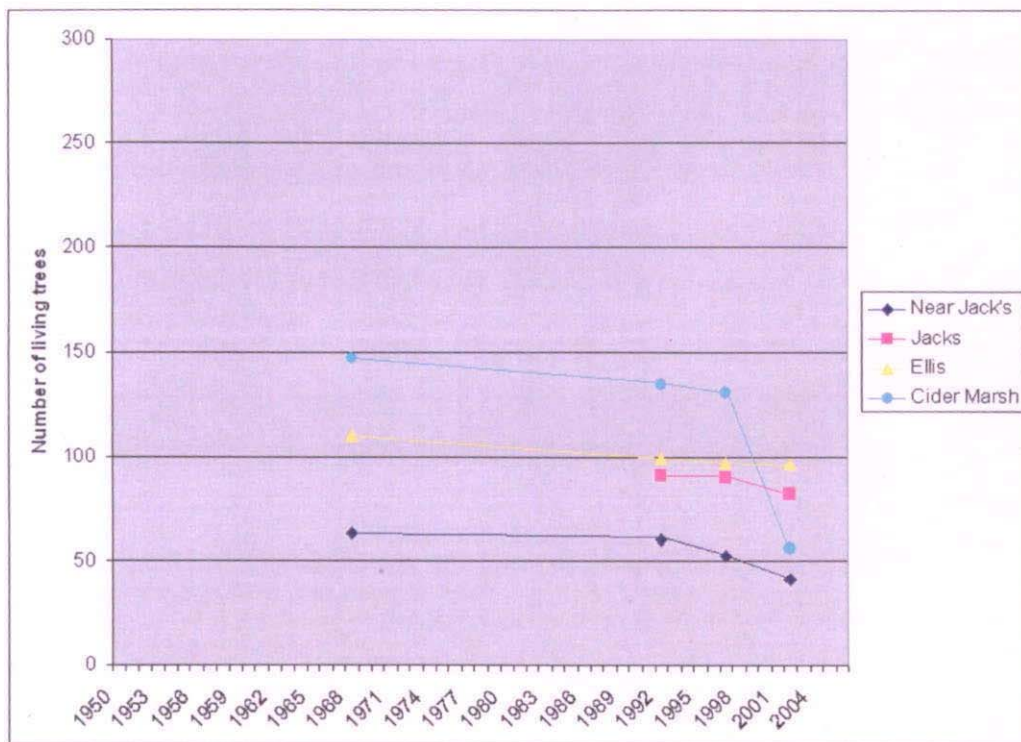


Figure 6.3: Number of live trees lost by time at Near Jack's Marsh, Jack's Marsh, Ellis Plains and Cider Marsh.

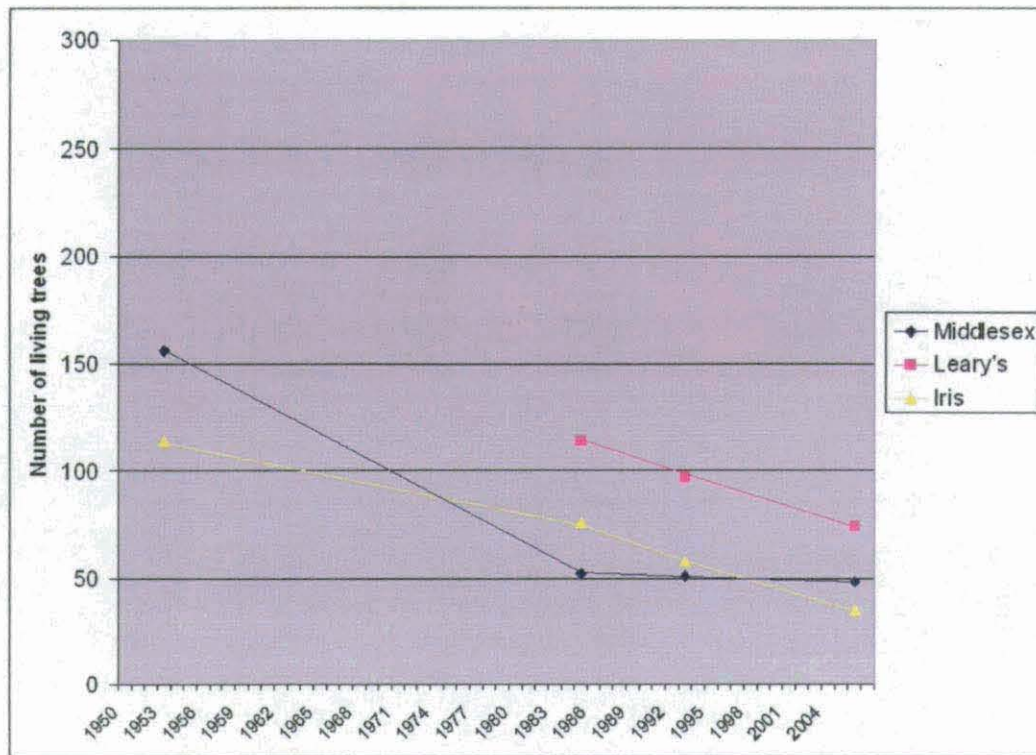


Figure 6.4: Number live trees lost by time at Iris River, Leary's Corner and Middlesex Plains.

The populations which suffered the steepest periods of decline over the record of photographs were Cider Marsh (plates 6.1 – 6.4) and Shannon Lagoon (plates 6.5 – 6.7) (figures 6.1 and 6.2). These were the populations for which 100% mortality was recorded during field surveys in 2006 (chapter 4). The steepest period of decline at Shannon Lagoon occurred after 1989 and after 1997 at Cider Marsh. Before 1992, the population at Cider Marsh underwent only a small decline. The photograph of Cider Marsh in 1992 (plate 6.1) shows that the trees still had dense and healthy crowns. Some thinning occurred by 1997 (plate 6.2), and the populations crashed after this. However the Shannon population has been in decline since the 1950s. There were 283 trees in the section of photograph analyzed at Shannon Lagoon in the 1950s, and 184 in 1984 (figure 6.2, plates 6.5 – 6.7).

Steep rates of decline also occurred Near Jack's Marsh (plates 6.8 – 6.10) and at Jack's Marsh between 1992 and 2002; and at Todd's Corner (plates 6.11 – 6.14) and Barren Tier (plates 6.15 – 6.18) between 1993 and 2006 (figures 6.1 and 6.2). Shannon, Barren Tier and Todd's Corner all suffered their steepest period of decline between 1993 and 1997, but unlike Shannon, Todd's' Corner

and Barren Tier did not suffer a high rate of decline before the 1990s. The rate of decline near Jack's Marsh and at Jack's Marsh was similar to that of Cider Marsh before 1997, after which the population at Cider Marsh underwent more rapid decline. The Ellis Plains population (plates 6.19 – 6.20) underwent a small and constant rate of decline between 1968 and 2002 (figure 6.3). Most of the trees were still healthy when surveyed in 2006 (chapter 4).

In the north-west (figure 6.13), Leary's Corner (plates 6.21 – 6.23) and the Iris River (plates 6.24 – 6.26) have undergone consistent decline since the earliest photographs. The rate of decline at Leary's Corner and the Iris River was very similar between 1985 and 2006. However the site at the Middlesex Plains (plates 6.27 – 6.30) underwent its steepest period of decline in the three decades before this. There was little decline at the Middlesex Plains site after 1985. Many fallen trees were visible on the ground in the north-west (e.g. plates 6.9, 6.12 and 6.21), which was not so in the Central Plateau photographs.

6.4 Discussion

At Leary's Corner and the Iris River in the north-west, the decline since 1985 has been at a similar rate, suggesting a similar factor, or factors operating at both sites. There were major road works upslope of these populations between 1985 and 1992 (plates 6.21, 6.22, 6.23 and 6.24). Many of the trees lost during this period were close to the roadside, and presumably felled. It seems unlikely that water deficiency would be the cause of death of trees in this area, which receives over 2000 mm of rainfall on average each year. However, the proximity of these populations to the road may explain some of the dieback. Changed drainage from roads can intercept and redistribute run off and may result in excesses or deficiencies of available moisture (Macgregor and O'Connor 2002). The lower side of the road has been far more affected than the upper side at the Iris River (plate 4.16 and 6.26). It is possible that the trees on the lower side of the road are receiving reduced down slope run off, due to the diversion of water flow into drainage channels beside the road. The station at Erriba has also recorded below average rainfall since the early 1980s (chapter 3). It is also possible that the populations of *E. gunnii* growing in the

north-west have a lower drought tolerance than the Eastern Central Plateau populations, due to both genetic and developmental differences. Whether the deleterious factor was due to road works or something else, it did not affect the Middlesex Plains population after 1985. This population underwent a large decline in tree numbers between 1953 and 1985. There is a good chance that much of this tree loss was due to felling. Many more dead trees were seen lying on the ground in the north-west photographs than in the Central Plateau. The high rate of decline at Shannon Lagoon in the three decades before the 1980s could also be partly due to felling, for pasture or firewood. Natural population thinning is another possible explanation, but it is difficult to tell without a higher temporal resolution of photographs for this period.

The most severe period of decline in all the Central Plateau sites, except the Ellis Plains, occurred after 1992/1993. This again suggests there was a similar factor, or factors, operating on most populations within the area, yet the Ellis Plains site was somehow protected. Jack's Marsh, Near Jack's Marsh, Todd's Corner and Barren Tier were affected to a lesser extent than Shannon or Cider Marsh. Landscape-wide processes that may have contributed to the dieback include increases in herbivore numbers and drought. The intensity of these would vary across the landscape, with localized possum control, the west-east rainfall gradient, and local topography and drainage. The rates of decline in the Central Plateau support the drought and climate change hypothesis. Steppes, the rainfall station closest to Cider Marsh, Ellis and Jack's Marsh, recorded its most consistently dry period from the early 1990s, and its most severe drought was in 2002 (chapter 3). This is consistent with the timing of the dieback at Cider Marsh and Jack's Marsh. However, the nearby Ellis Plains site remained unaffected, suggesting that drought has not been the sole contributing factor. The combination of drought and possum browsing seems a likely explanation (chapter 4).

Miena, the station closest to Shannon, Todd's and Barren Tier, recorded its most consistently dry period between the late 1970s and mid 1990s, and had its worst droughts in the late 1980s and early 2000s (chapter 3). This is consistent with the greater rate of dieback that occurred in the more western Central

Plateau sites before 1992. However, there is difficulty in comparing the pre 1992 rate of decline between the four more eastern, and three more western Central Plateau sites (figure 6.1), due to different photograph timing.

In the photographs with low resolution and a high degree of shadow it was difficult to distinguish between standing stags and living trees. For example, the 1992 black and white photograph from the Middlesex Plains (plate 6.29) had several trees which were casting a very thin pole-like shadow and were therefore assumed to be dead. However, there were signs of green on these trees in the 2006 colour photograph (plate 6.30). This may either represent recovery or misinterpretation. The problems identified by Fensham and Fairfax (2002), as limiting the interpretability of a long series of aerial photographs, were all present. These were variation in scale, contrast, colour and shadow. Therefore the estimate of tree decline gained from the aerial photographs is reasonable, but not precise. Additionally, where photographs were widely spaced in time it was difficult to tell whether tree loss was due to a sudden or gradual process. Nevertheless, the results support the hypotheses that that felling, road disturbance, drought and herbivore numbers may each have had a strong influence on tree decline.

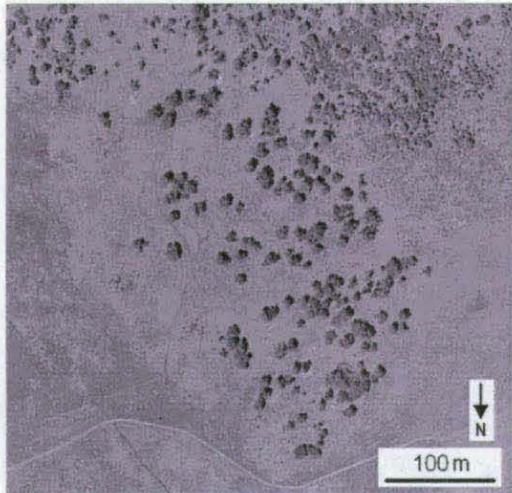


Plate 6.1: Cider Marsh 1992



Plate 6.2: Cider Marsh 1997



Plate 6.3: Cider Marsh, 2002



Plate 6.4: Cider Marsh from the ground, 2006

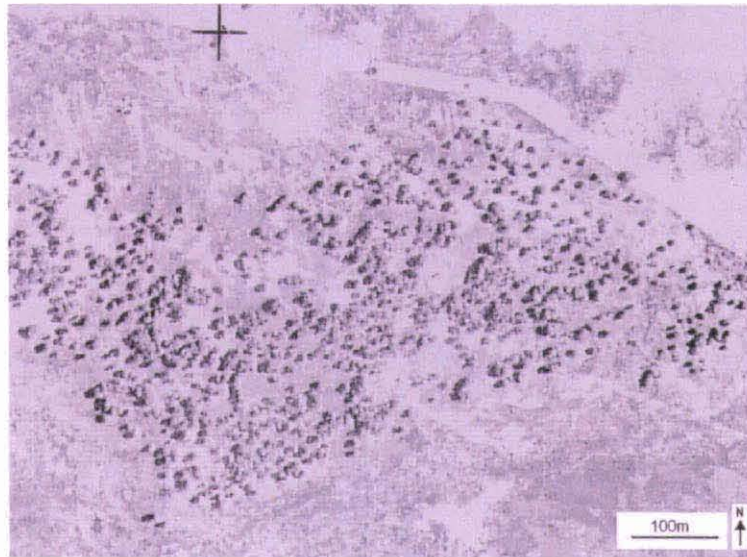


Plate 6.5: Shannon Lagoon in the 1950s

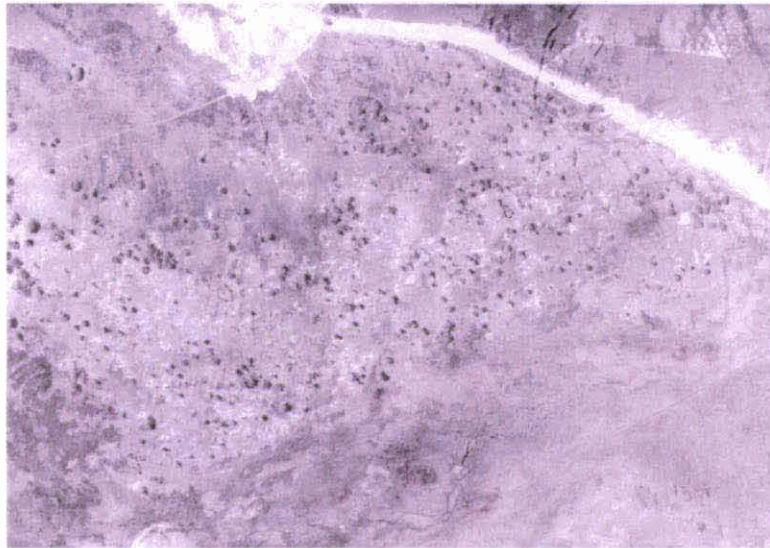


Plate 6.6: Shannon Lagoon 1989



Plate 6.7: Shannon, 2006

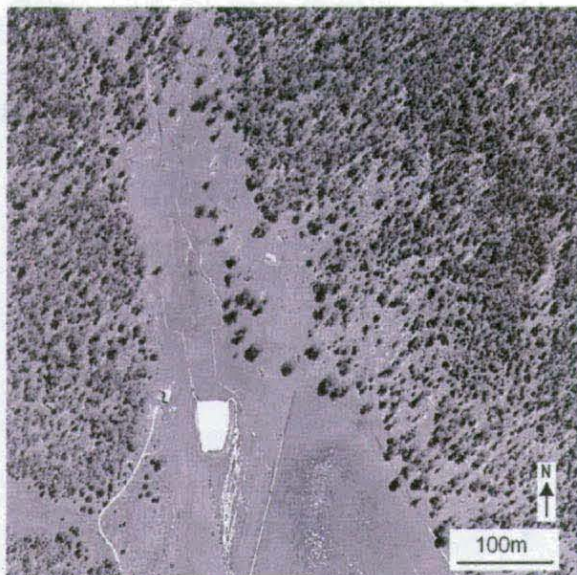


Plate 6.8: Near Jack's Marsh, 1992

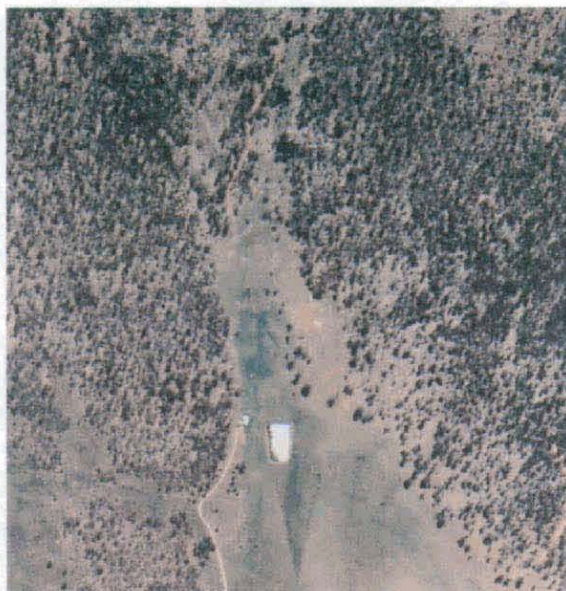


Plate 6.9: Near Jack's 1997



Plate 6.10: Near Jack's Marsh, 2002

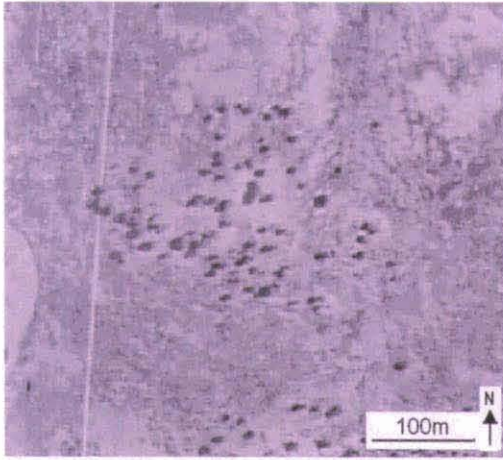


Plate 6.11: Todd's Corner 1950s

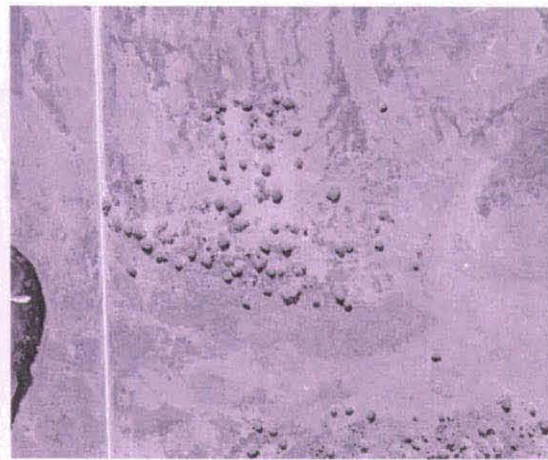


Plate 6.12: Todd's Corner 1992

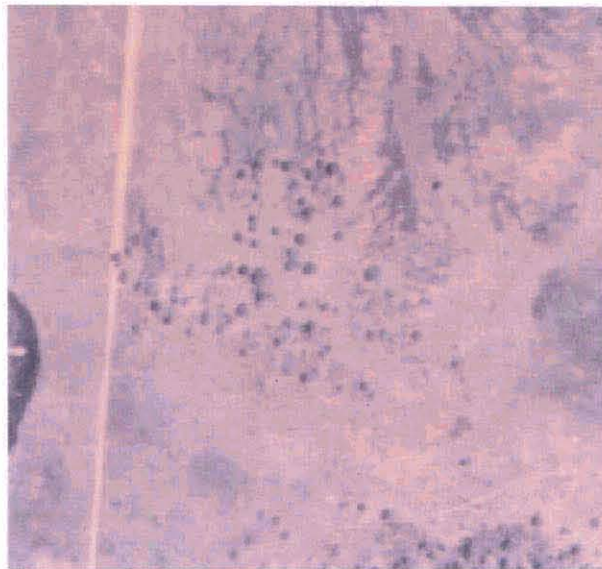


Plate 6.13: Todd's Corner 1997



Plate 6.14: Todd's Corner 2006

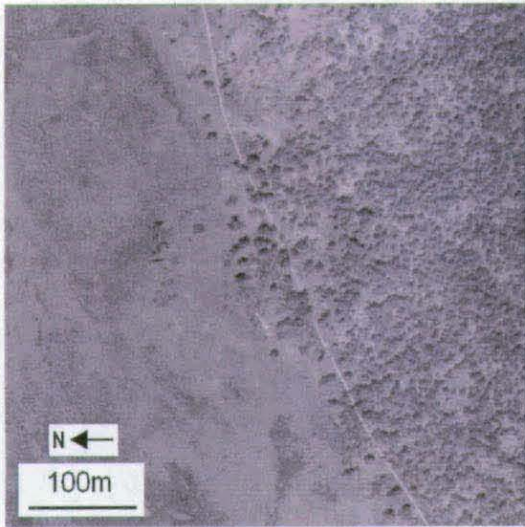


Plate 6.15: Barron Tier 1989



Plate 6.16: Barron Tier 1992



Plate 6.17: Barron Tier 1997



Plate 6.18: Barron Tier 2006

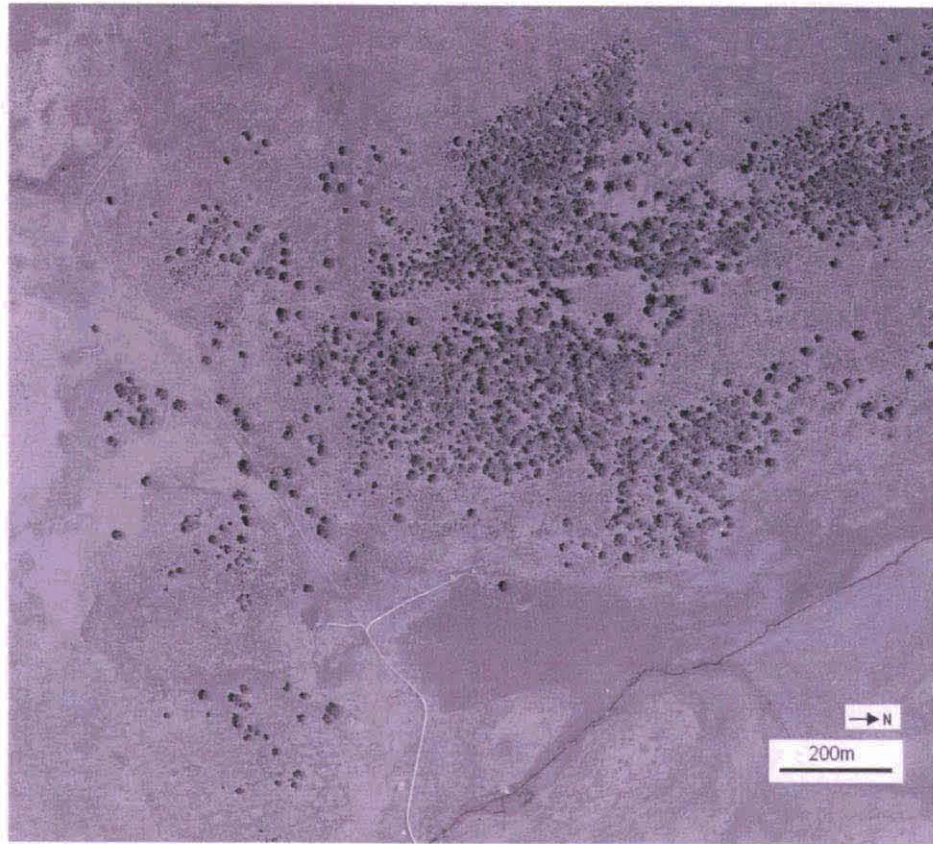


Plate 6.19: Ellis Plains 1992

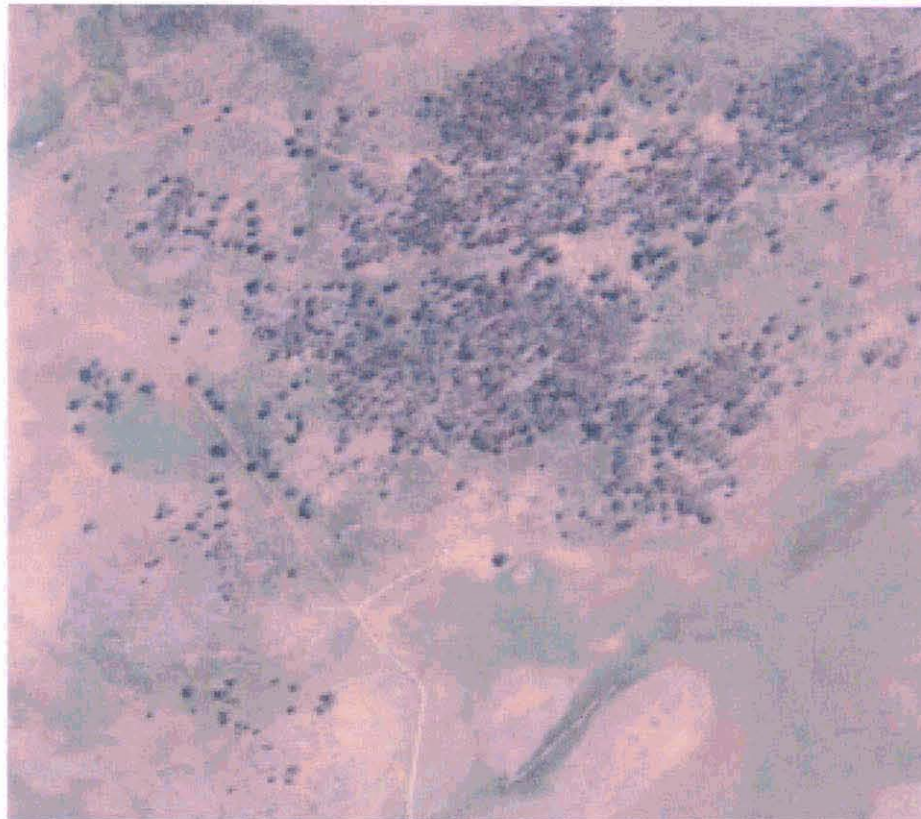


Plate 6.20: Ellis Plains 2002

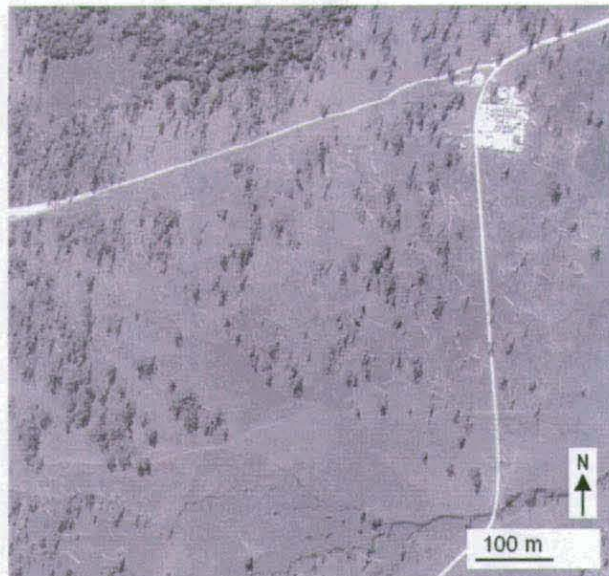


Plate 6.21: Leary's Corner 1985

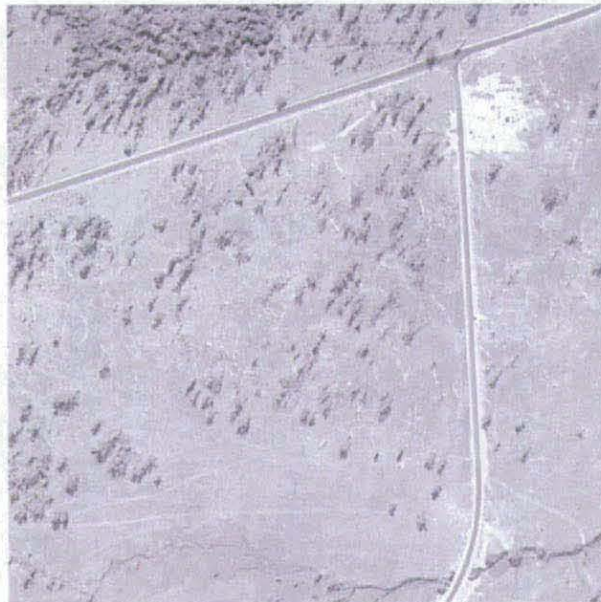


Plate 6.22: Leary's Corner 1992



Plate 6.23: Leary's Corner 2006

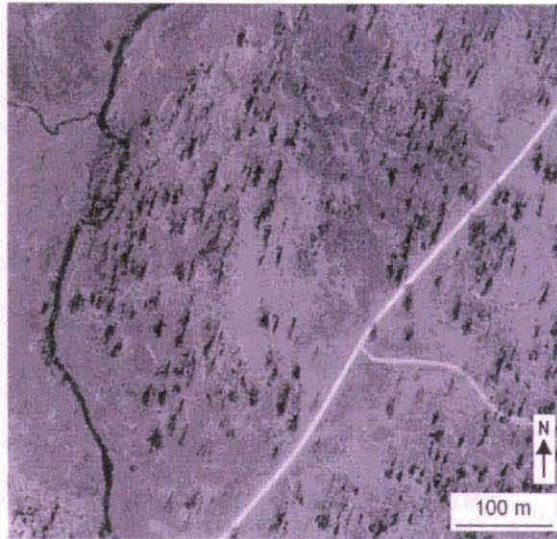


Plate 6.24: Iris River 1985

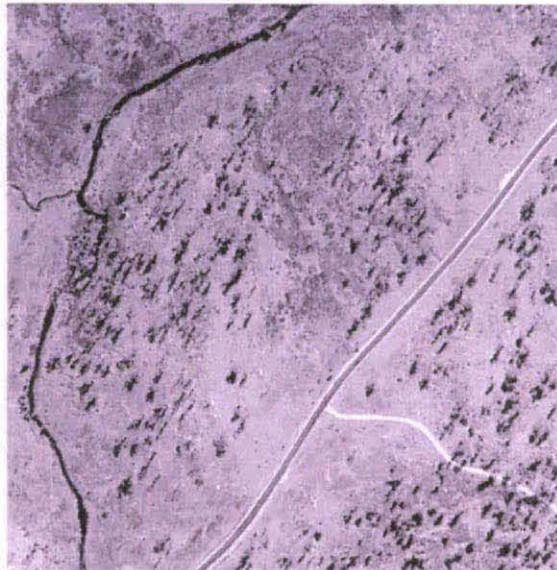


Plate 6.25: Iris River 1992



Plate 6.26: Iris River 2006



Plate 6.27: Middlesex Plains 1953



Plate 6.28: Middlesex Plains 1985



Plate 6.29: Middlesex Plains 1992



Plate 6.30: Middlesex Plains 2006

Chapter 7: Overall discussion, conclusions and management recommendations

7.1 Discussion

It has been predicted that significant mortality of forest trees will take place worldwide, due to increased intensity and frequency of droughts brought about by climate change (Martin 1996, Solomon and Kirilenko 1997). Species in high altitude environments and at the edges of their climatic range limits are expected to be the first to show the effects of global warming (Lavoie and Payette 1992, Harte and Shaw 1995, Gardner and Fisher 1996, Gottfried *et al.* 1999, Hughes 2000, Leohle 2001, Walter *et al.* 2002, Jia *et al.* 2006).

Several lines of evidence support the hypothesis introduced by Potts *et al.* (2001) that the dieback of *E. gunnii* is partly due to climate change. Changes in rainfall and temperature have occurred in the habitat of *E. gunnii*, particularly since the late 1970s (chapter 3), and the worst dieback has generally been in the driest part of the species' range (chapter 4). Most weather stations recorded a prolonged period of below average rainfall between the 1970s and the 1990s. Additionally, the timing of the most rapid decline of several of the eastern Central Plateau populations coincides with the worst drought on record in the region, in the period 2001-2002 (chapters 3 and 6). Moisture stress therefore appears a likely explanation for tree death. It would be possible to further examine this hypothesis by detailed investigations into the depth, drainage and moisture-holding capacity of the soil across the landscape (e.g. Rees 2000, Doyle 2003). These soil characteristics can vary over short distances (Davidson and Reid 1989), and trees can survive where they otherwise would not due to an advantageous hydrogeological position (Boardman 1994, Doyle 2003).

Despite the strong circumstantial evidence, it is not possible to prove that climate change is the sole cause of the dieback of *E. gunnii*. Most researchers have found it difficult to separate the effects of climate change and other stress factors on biological systems (Kapelle *et al.* 1999, Stohlgren *et al.* 2000,

McCarty 2001). Fragmentary evidence exists that suggests other causal agents may have had an impact on tree health, either in combination or independent of climate change. For example, local changes in hydrology associated with the road may have contributed to the decline at the Iris River and Leary's Corner (chapter 6). The incidence of stock grazing also has a negative relationship with the health of mature trees (chapter 4). Several authors have researched the direct effects that livestock grazing may have on tree health, with most explanations involving negative impacts on water relations (Close and Davidson 2004). However, the direct effects of stock do not seem a major a cause of decline in the case of *E. gunnii*, as no interactive effect between grazing and climate was detected (chapter 4). Further research, such as measuring the severity of soil compaction around the base of paddock trees, or investigations into mycorrhizal relationships would be needed to dismiss this hypothesis. The results of the present study provide greater support for the theory that livestock grazing has suppressed regeneration (chapter 5). This means that the dominant cohort in many populations is composed of trees which may be nearing the end of their natural life spans. Privately-owned land is also more likely to have had trees removed for pasture expansion, firewood or seed collection. This may have lead to increased exposure of the remaining trees, accelerating their decline. Investigations of historical land management records could elucidate the extent of these practices. It would also be useful to determine historical stocking rates.

There is support for the theory that possum grazing has had a significant effect on tree health in some areas, particularly the eastern Central Plateau (chapter 4). However, this is mainly based on anecdotal evidence. To further test this hypothesis it would be useful to investigate possum population density across the range of *E. gunnii*, and undertaken possum removal or exclusion experiments.

It is also possible that higher minimum temperatures in the region have had an impact on the herbivorous insect fauna and increased defoliation pressure in high altitude populations (chapter 4). To further investigate this hypothesis, a more precise measure of insect defoliation than the rough visual estimate used

in this study would be needed, as would sampling over a controlled altitudinal transect. It would also be useful to determine if the community composition and impact of insects has changed over time, and what climatic factors are correlated with insect outbreaks.

Conclusions

The decline of *Eucalyptus gunnii* has been shown to be a severe problem, particularly in the populations to the south-east of Great Lake (chapter 4). The results provide support for most of the causes identified by Potts *et al.* (2001) as possible contributors to the dieback of *E. gunnii* spp. *divaricata* (chapter 2), especially old age, lack of population replacement and possum defoliation (figure 7). It appears very likely that the recent prolonged dry period and severe droughts have significantly contributed to the decline of the species (chapter 3 and 6). Warmer temperatures may also have influenced folivorous insect populations at higher altitudes (chapter 4). It is also highly probable that the observed climatic phenomena (chapter 3) have been brought about by changes in the earth's energy balance due to increased carbon dioxide in the atmosphere (Bakkenes 2002). However, as discussed in chapter 2, dieback seldom has a simple aetiology, and this complicated story resembles most other studies related to the phenomenon.

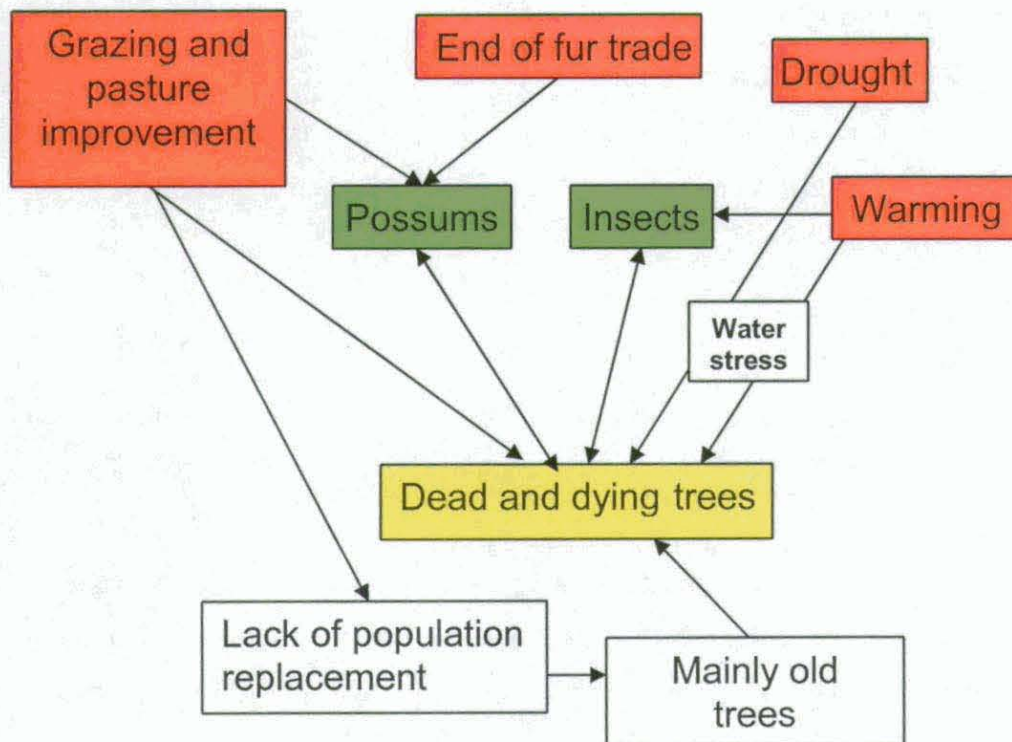


Figure 7.1: Factors that may be contributing to decline of *Eucalyptus gunnii* identified by the present study

7.3 Management recommendations

The present study does provide support for the hypothesis that the dieback of *E. gunnii* may be partly due to climate change. Therefore there should be much concern for the species, as well as other drought-intolerant and cold-adapted species in Tasmania. Even if migration rates could keep pace with the changing climate across a fragmented landscape, many subalpine species have essentially 'nowhere to go'. Planning for climate change is not an easy task (Haplin 1997). Droughts, temperature extremes and insect defoliation are all factors which are likely to be further exacerbated by the continuation of long term climatic changes in the region. As well as direct effects, climate change can also affect other aspects of a species' environment, such as soil organisms, fire regimes and carbon dioxide levels (Saxe *et al.* 2001). As more evidence of biological trends that are consistent with the predictions of climate change is found, perhaps governments, higher planning bodies and the general public can

be further convinced of the reality of this significant global trend and its impacts on the world's biodiversity.

In the meantime, the relative health of several populations within the dieback region (chapter 4), suggests that it has not been drought alone that has caused the complete death of stands. The decline may be ameliorated by protecting the trees from some of the other contributing stresses. For example, Kirkpatrick *et al.* (2000) showed that trees in the Midlands which were possum-proofed showed less decline than those which were not. Evidence suggests that the resistance and resilience of vegetation to drought and climate change is reduced in intensely stressed ecosystems compared with lightly stressed ecosystems (Noss 2001). This is worth testing in the case of *E. gunnii*. Therefore, key management recommendations for the species are:

- Possum removal or exclusion, which may have a positive effect on tree health, particularly on their capacity to recover after stress.
- The protection and encouragement of juveniles, which is critical. This would be best achieved by livestock and native animal exclusion, and encouragement of the shrub layer in the understorey.
- Further research into the changes in herbivorous insect population numbers and community composition with warming temperatures.
- Ongoing monitoring of the successes of conservation measures. If these relatively simple management techniques do not significantly improve the health of populations, and the present climatic trends continue, the artificial watering of surviving *divaricata* trees may be necessary to maintain their health. It may also be necessary to focus replanting efforts in regions and topographical positions that are less likely to be subject to moisture stress.

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Appendices

Appendix 1:

Field site names, date visited, code number, eastings and northings (GDA 1994), altitude, % mature poor health and % juveniles

Site name	Date visited	Code number	Easting	Northing	Altitude	% Mature poor health	% Juvenile
Central Plateau Sites							
Jimmy's Marsh	04/04/06	5	504463	5337424	1025	90	53
Scrummies Marsh	04/04/06	6	503615	5334690	930	78	22
Interlaken Road	18/05/06	50	500905	5335562	955	0	40
Jack's Marsh	16/05/06	49	492304	5342680	888	84	0
Cider Marsh	18/05/06	51	489021	5340223	890	100	0
Ellis Plains	30/03/06	4	487099	5341877	900	35	21
Arthurs Lake Road	16/05/06	49	488070	5349893	1005	43	66
Poatina Road 2	18/05/06	52	487882	5358640	1156	54	66
Poatina Road 1	24/03/06	2	491455	3762346	1160	60	24
Sandbanks	24/03/06	3	486355	5365339	1078	5	35
Pensford	5/04/06	7	483941	5348763	909	88	8
Barren Tier	25/04/06	23	482248	5350426	1089	90	64
Barren Tier Road	16/04/06	47	480447	5350165	1074	70	58
Todds Corner	27/04/06	30	482281	5351856	1063	89	21
Shannon	28/04/06	33	481050	5351157	1080	100	0

Cider Gum Road	15/05/06	48	476082	5350153	1060	66	4
Near Ouse River	25/04/06	24	471068	5352318	1039	36	63
Pine Lagoon	27/04/06	28	466023	5348893	1035	38	6
Monpeelyata Road	27/04/06	29	469379	5346526	1064	15	57
Liawenee fence	27/04/06	31	474431	5364520	1079	50	46
Lake Augusta Road	28/04/06	32	472013	5362078	1082	13	34
Liawenee garden	27/03/06	1	468212	5362072	1160	75	30
Reynolds	15/05/06	45	474663	5365106	1106	18	56
Brandum Bay	26/04/06	27	473404	5371391	1058	3	10
Pine tree rivulet	26/04/06	26	473235	5372820	1066	68	37
Doctor's Creek	26/04/06	53	474836	5373956	1065	9	47
Rainbow Road	26/04/06	25	475226	5373864	1062	31	28
Mickey's Creek	09/04/06	12	476412	5376291	1139	60	55
Half Moon Creek	15/5/06	44	476207	5377377	1153	69	4
North West sites							
Guildford 1	07/04/06	8	390838	5412776	631	58	22
Guildford 2	08/04/06	11	388835	5413978	617	56	33
Murchison	07/04/06	9	380557	5417798	600	100	45
Hatfield River	08/04/06	10	388492	5401807	660	52	65
Catley Road	03/05/06	36	400241	5399756	746	7	62
Sunshine Creek	03/05/06	37	411078	5398935	835	22	62
Leary's Corner	03/05/06	39	412399	5398818	800	80	73
Near Parks Office	02/05/06	35	410866	5394537	715	10	40
Mount Kate	04/05/06	40	412621	5390189	880	33	48
Iris River	03/05/06	38	413549	5399544	802	43	65
Between Daisy Dell and Iris River	04/05/06	43	416690	5400460	814	44	0

Weaning Paddock Creek	04/05/06	41	419338	5400000	777	14	52
Cow sign	04/05/06	42	421111	5399484	766	16	40
Big bend	02/05/06	34	422269	5401864	740	25	30
Lyell Highway sites							
Butlers	12/04/06	13	447367	5318038	730	0	45
Clarence River	14/04/06	19	445880	5333386	728	26	59
Before Derwent Bridge	12/04/06	15	443268	5336762	763	0	24
Coates Creek	12/04/06	18	432465	5331031	745	0	10
Navarre River	13/04/06	14	433511	5331542	756	5	22
King William Creek	13/04/06	16	429044	5326993	825	0	24
Beehive Canal	13/04/06	17	426381	5326546	828	11	45
South east and North east sites							
Snow hill	18/04/06	20	569348	5359300	969	0	38
St Pauls River	19/04/06	21	591863	5382210	533	0	30
Snug Tiers	23/04/06	22	513981	5224449	626	0	28

Appendix 2:

Percentage frequency of each species in the five vegetation communities associated with *E. gunnii*

Species	Community 1 (10 sites)	Community 2 (14 sites)	Community 3 (14 sites)	Community 4 (7 sites)	Community 5 (8 sites)
<i>Abrotanella forsteroides</i> (Hook.f.) Benth.	10	35.71	-	-	-
<i>Acaena novae-zelandiae</i> Kirk	90	100	92.86	100	75
<i>Acetosella vulgaris</i> Fourr.	10	21.43	-	28.57	-
<i>Agrostis</i> sp. L.	30	35.71	35.71	28.57	50
<i>Aira</i> sp. L.	10	28.57	14.29	57.14	-
<i>Ajuga australis</i> R.Br.	-	35.71	-	14.29	-
<i>Almaleea subumbellata</i> (Hook.) Crisp & P.H.Weston.	40	21.43	-	-	25
<i>Asperula</i> sp. L.	30	14.29	-	14.29	12.5
<i>Astelia alpina</i> R.Br.	20	28.57	7.14	-	87.5
<i>Athrotaxis cupressoides</i> D.Don.	10	-	7.14	-	-
<i>Austrodanthonia</i> sp. H.P.Linder	20	64.29	14.29	100	50
<i>Baeckea gunniana</i> Schauer	40	7.14	7.14	-	100
<i>Banksia marginata</i> Cav.	-	-	7.14	-	12.5
<i>Bauera rubioides</i> Andrews	-	-	7.14	-	62.5
<i>Bedfordia linearis</i> (Labill.) D	-	7.14	7.14	-	-
<i>Bellenden montana</i> R.Br.	20	-	35.71	-	-
<i>Billardiera longiflora</i> Labill.	20	-	7.14	-	37.5
<i>Blechnum pennamarina</i> (Poir.) Kuhn	60	21.43	71.43	-	62.5
<i>Baloskion australe</i> (R.Br.) B.G.Briggs &	80	57.14	78.57	100	100

L.A.S.Johnson					
<i>Boronia citriodora</i> Gunn ex Hook.f.	10	-	-	-	12.5
<i>Bossiaea cordigera / foliosa</i>	-	7.14	28.57	42.86	12.5
<i>Brachyscome</i> sp. Cass.	20	7.14	-	-	-
<i>Bracteantha subundulata</i> (Sch.Bip.) Paul G.Wilson	-	-	7.14	14.29	-
<i>Callistemon viridiflorus</i> (Sol. ex Gaertn.) G.Don	-	-	7.14	-	87.5
<i>Carex</i> sp. L.	20	21.43	28.57	14.29	12.5
<i>Carpha alpina</i> R.Br.	10	21.43	-	14.29	25
<i>Centaureum</i> sp. Hill.	-	7.14	7.14	-	-
<i>Cirsium</i> sp. Mill.	10	14.29	14.29	57.14	-
<i>Clematis aristata</i> R.Br. ex Ker Gawl.	-	-	14.29	14.29	-
<i>Coprosma nitida</i> Hook.f.	100	100	85.71	71.43	62.5
<i>Cotula alpina</i> (Hook.f.) Hook.f.	10	-	21.43	14.29	12.5
<i>Cyathodes parvifolia</i> R.Br.	100	85.71	92.86	100	50
<i>Deyeuxia quadriseta</i> (Labill.) Benth.	-	7.14	-	-	25
<i>Dichelachne</i> sp. Endl.	40	78.57	14.29	28.57	62.5
<i>Diplarrena</i> sp. Labill.	40	21.43	64.29	42.86	75
<i>Drymophila cyanocarpa</i> R.Br.	-	7.14	21.43	-	12.5
<i>Ehrharta stipoides</i> Labill.	-	-	28.57	14.29	25
<i>Elymus scabrus</i> (R.Br.) A.Love	-	28.57	-	28.57	12.5
<i>Empodisma minus</i> (Hook.f.) L.A.S. Johnson & Cutler	70	35.71	42.86	28.57	75
<i>Epacris gunnii</i> Hook.f.	80	78.57	78.57	71.43	62.5
<i>Epacris petrophila</i> Hook.f.	20	-	7.14	14.29	-
<i>Epilobium</i> sp. L.	10	7.14	14.29	-	12.5
<i>Erigeron</i> sp. L.	10	7.14	28.57	14.29	12.5

<i>Eucalyptus coccifera</i> Hook.f.	80	28.57	28.57	14.29	25
<i>Eucalyptus dalrympleana</i> Maiden	-	-	21.43	71.43	37.5
<i>Eucalyptus delegatensis</i> R.T.Baker	30	28.57	64.29	57.14	37.5
<i>Eucalyptus nitens</i> (H.Deane & Maiden) Maiden	-	-	14.29	14.29	-
<i>Eucalyptus nitida</i> Hook.f.	-	-	7.14	-	25
<i>Eucalyptus pauciflora</i> Sieber ex Spreng.	-	64.29	-	42.86	75
<i>Eucalyptus rodwayi</i> R.T.Baker & H.G.Sm.	-	-	14.29	14.29	-
<i>Euchiton</i> sp. Cass.	30	78.57	35.71	57.14	-
<i>Euphrasia</i> sp. L.	10	-	14.29	-	-
<i>Gahnia grandis</i> (Labill.) S.T.Blake	-	-	35.71	-	50
<i>Galium</i> sp. L.	30	21.43	7.14	14.29	37.5
<i>Geranium</i> sp. L.	80	100	85.71	100	75
<i>Gleichenia alpina</i> R.Br.	20	-	14.29	-	37.5
<i>Gleichenia dicarpa</i> R.Br.	-	-	7.14	-	25
<i>Gonocarpus humilis</i> Orchard	30	78.57	42.86	57.14	62.5
<i>Gonocarpus micranthus</i> Thunb.	-	7.14	21.43	-	-
<i>Grevillea australis</i> R.Br.	70	57.14	21.43	-	12.5
<i>Gunnera cordifolia</i> (Hook.f.) Hook.f.	10	-	28.57	-	-
<i>Gymnoschoenus sphaerocephalus</i> (R.Br.) Hook.f.	-	-	21.43	-	87.5
<i>Hakea lissosperma</i> R.Br.	20	7.14	7.14	28.57	-
<i>Hakea microcarpa</i> R.Br.	40	100	21.43	57.14	37.5
<i>Helichrysum rutidolepis</i> DC.	50	21.43	28.57	14.29	25
<i>Holcus lanatus</i> L.	30	50	42.86	100	37.5
<i>Hovea montana</i> (Hook.f.) J.H.Ross	-	28.57	7.14	-	-
<i>Hydrocotyle</i> sp. L.	80	57.14	78.57	42.86	75
<i>Hymenanthera dentata</i> R.Br. ex DC.	-	14.29	-	28.57	-

<i>Lagenifera stipitata</i> (Labill.) Druce	20	14.29	7.14	14.29	-
<i>Leptorhyncos squamatus</i> (Labill.) Less.	50	57.14	21.43	57.14	-
<i>Leptospermum lanigerum</i> (Sol. ex Aiton) Sm.	30	14.29	57.14	57.14	62.5
<i>Leptospermum rupestre</i> Hook.f.	30	7.14	21.43	-	12.5
<i>Leptospermum scoparium</i> J.R.Forst. & G.Forst.	-	-	14.29	-	12.5
<i>Leucopogon hookeri</i> Sond.	90	85.71	78.57	85.71	12.5
<i>Lomatia tinctoria</i> (Labill.) R.Br.	-	-	35.71	28.57	12.5
<i>Lotus</i> sp. L.	-	-	7.14	14.29	-
<i>Luzula</i> sp. DC.	10	28.57	7.14	14.29	-
<i>Lycopodium fastigiatum</i> R.Br.	80	14.29	57.14	-	25
<i>Monotoca empetrifolia</i> R.Br.	20	-	7.14	-	-
<i>Nothofagus cunninghami</i> (Hook.) Oersted	10	-	50	-	-
<i>Olearia algida</i> N.A.Wakef.	-	7.14	-	14.29	-
<i>Olearia erubescens</i> (DC.) Dippel	80	35.71	28.57	-	37.5
<i>Olearia ledifolia</i> (DC.) Benth.	10	7.14	-	-	-
<i>Olearia phlogopappa</i> (Labill.) DC.	20	14.29	50	-	37.5
<i>Olearia tasmanica</i> (Hook.f.) W.M. Curtis	10	-	7.14	-	12.5
<i>Oreomyrrhis eriopoda</i> (DC.) Hook.f.	20	21.43	14.29	14.29	12.5
<i>Orites revoluta</i> R.Br.	80	28.57	7.14	-	-
<i>Oxylobium ellipticum</i> (Labill.) R.Br.	70	28.57	-	-	37.5
<i>Ozothamnus ericeteus</i> (W.M.Curtis) Anderb.	-	28.57	7.14	-	-
<i>Ozothamnus hookeri</i> Sonder	80	71.43	7.14	-	37.5
<i>Ozothamnus thyrsoides</i> DC.	-	-	7.14	-	12.5
<i>Pentachondra pumila</i> (Forst & Forst.f.) R.Br.	40	42.86	-	-	-
<i>Phyllocladus aspleniifolius</i> (Labill.) Hook.f.	10	-	21.43	-	-
<i>Pittosporum bicolor</i> Hook.	20	-	28.57	-	25
<i>Plantago</i> sp. L.	10	7.14	7.14	14.29	-

<i>Poa</i> sp. L.	100	100	100	100	87.5
<i>Polystichum proliferum</i> (R.Br.) C.Presl	30	7.14	50	14.29	12.5
<i>Pratia pedunculata</i> (R.Br.) Benth.	-	-	7.14	14.29	-
<i>Pultenaea juniperina</i> Labill.	10	50	50	28.57	12.5
<i>Ranunculus</i> spp. L.	70	64.29	50	57.14	12.5
<i>Richea acerosa</i> (Lindl.) F.Muell.	60	85.71	14.29	14.29	-
<i>Richea gunnii</i> Hook.f.	30	21.43	14.29	-	25
<i>Richea procera</i> (F.Muell.) F.Muell.	20	7.14	-	-	-
<i>Richea scoparia</i> Hook.f.	20	-	21.43	-	-
<i>Richea sprengelioides</i> (R.Br.) F.Muell.	20	7.14	-	-	12.5
<i>Rubus gunnianus</i> Hook.	60	21.43	57.14	-	75
<i>Scleranthus singuliflorus</i> (F.Muell.) Mattf.	-	7.14	-	14.29	-
<i>Senecio gunnii</i> (Hook.f.) Belcher	50	78.57	7.14	-	-
<i>Senecio</i> sp. L.	-	-	21.43	-	12.5
<i>Solenogyne gunnii</i> (Hook.f.) Cabrera	-	7.14	7.14	14.29	-
<i>Sphagnum</i> spp.	-	-	28.57	-	25
<i>Stylidium graminifolium</i> Sw. ex Willd.	30	21.43	35.71	-	25
<i>Tasmannia lanceolata</i> (Poir.) A.C.Sm.	70	7.14	85.71	14.29	62.5
<i>Telopea truncata</i> (Labill.) R.Br.	20	-	7.14	-	25
<i>Veronica calycina</i> R.Br.	50	35.71	35.71	28.57	-
<i>Veronica gracilis</i> R.Br.	10	-	21.43	42.86	12.5
<i>Viola betonicifolia</i> Sm.	10	57.14	14.29	14.29	-
<i>Viola hederacea</i> Labill.	50	14.29	42.86	-	-
<i>Wahlenbergia</i> spp. Schrad. ex Roth	10	35.71	7.14	-	12.5

Appendix 3:

Details of aerial photographs

Area	Project number	Film number	Frame number	Run	Scale	Date flown	Sites covered
Shannon L Sorrel	1591	504	301	3	1:14 000	22.2.1968	Cider Marsh. Near Jack's Marsh, Ellis
Eastern Revision	A003	1194	41	24W	1:42 000	17.12.1992	Cider Marsh. Near Jack's Marsh, Jack's Marsh, Ellis
Central East	A118	1279	63	24	1:42 000	11.12.1997	Cider Marsh. Near Jack's Marsh, Jack's Marsh, Ellis
Revision							
Central East	A118	1355	39	24C	1:42 000	15.02.2002	Cider Marsh. Near Jack's Marsh, Jack's Marsh, Ellis
Central Plateau	-	309	8	5	-	1950s	Shannon, Todd's, Barron
Central East	M456	1004	208	22	1:42 000	31.10.1984	Shannon, Todd's, Barron
-	M443	1010	092	22	-	7.11.1989	Shannon, Barron
Eastern Revision	A003	1202	10	22	1:42 000	21.3.1993	Shannon, Todd's, Barron
Northern	A111	1278	133	22W	1:42 000	11.12.1997	Shannon, Todd's, Barron
Revision							
Central East	A118	1407	163	22	1:42 000	2.03.2006	Shannon, Todd's, Barron
Revision							
Mersey	-	304	112	7	-	28.1.1953	Iris, Middlesex
North West	M548	1030	95	15	1:42 000	19.3.1985	Iris, Middlesex, Leary's
Sophia	M913	1185	181	15	1:42 000	8.3. 1992	Iris, Middlesex, Leary's
North West	A143	1405	204	15E	1:42 000	23.01.2006	Iris, Middlesex, Leary's