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Vegetation Biodiversity Response of *Eucalyptus* Regrowth Forest to Thinning and Grazing

By Ross J. Peacock

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Foreword

Forest managers are increasingly required to demonstrate to the community, government regulators and the purchasers of forest products, that their activities are environmentally sustainable. For forestry, thinning of *Eucalyptus* dominated regrowth forests is seen as integral to managing timber yield and quality in native forests, as well as wood and cash flows. At the same time landholders seek or are required by regulators to maintain or improve the biodiversity value of their forest. However very little information currently exists on the biodiversity response of forested vegetation in Australia to commercial thinning or the combination of selective logging or thinning and cattle grazing.

This report is of particular relevance to forest managers and landholders. It presents the results from three medium-term (14-51 year old) studies in managed forests in northern NSW, East Gippsland Victoria and southern Tasmania, where the vegetation response to thinning and grazing has been monitored. The key results were that initial forest vegetation composition, method and intensity of thinning, extent of soil disturbance and the timing of cattle grazing all had an effect on forest biodiversity and tree growth. Abrupt changes in vegetation composition and structure occurred in the first 1-3 years after thinning, but the medium term (12-17 years) composition and structure of the forest more closely resembled that of the untreated regrowth forest. Based on the recommendations in this study, forest managers, policy and landholders should consider the practices in their region, and whether some should be modified, for example delaying cattle grazing until four years after thinning in northern New South Wales spotted gum forests. Further research trials should be established on younger age forests, which are currently being thinned and may have different biodiversity responses.

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This report is an addition to RIRDC's diverse range of over 1800 research publications. It forms part of our Agroforestry and Farm Forestry R&D program, which aims to integrate sustainable and productive agroforestry within Australian farming systems. The JVAP, under this program, is managed by RIRDC.

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Peter O'Brien

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Executive Summary

What the report is about

This report examines the effects of forest thinning, or thinning and grazing, on vegetation biodiversity and structure in three long-term experiments in intensively managed forests. The experiments were established in 1954, 1988 and 1991 and repeatedly monitored until 2005. The report contrasts the initial trends available in the first years after treatment to the medium-term results evident today, and discusses their relevance to contemporary private native forest management.

Who is the report targeted at?

This report is of significance to policy makers and government regulators who need to assess competing arguments for active versus passive management of forest landscapes, if their regulatory regimes are to be defended. Forest managers, private landholders, forestry researchers and environmentalists will also find this study particularly relevant.

Background

Forest managers are increasingly required to demonstrate to the community, government regulators and the purchasers of forest products, that their activities are environmentally sustainable. For forestry, thinning of *Eucalyptus* dominated regrowth forests is seen as integral to managing wood flows and wood quality in native forests, as well as providing a short-term cash flow from small poles, mining and fencing timbers, salvage sawlogs or pulpwood. At the same time landholders seek or are required by regulators to maintain or improve the biodiversity value of their forest. However very little information currently exists on the biodiversity response of forested vegetation in Australia to commercial thinning or the combination of selective logging/thinning and cattle grazing. This makes it difficult to assess that management regimes are appropriate and will meet the varying landowner, site and community demands on forests.

Government reforms since the 1990s have sought to establish a clearer relationship between a secure native forest timber industry and improvements in landscape and site-specific biodiversity. One result of government-negotiated reductions in harvest from public native forests, can be increased intensity of management of regrowth forests – this is commonly a compensatory mechanism to maintain wood flows to industry despite the restriction in area used to calculate a sustainable yield. There is a need to improve understanding of sustainable forest management practices and biodiversity responses; this also applies if stewardship schemes are implemented. Alongside this, long term responses of forests to disturbance and different silvicultural management must be understood, in order to effect appropriate policy.

Aims/ Objectives

The objective of this study was to examine the medium-term effects of thinning or selective harvesting on forest vegetation structure and composition, and therefore the potential of the forest stand to regenerate following harvesting. It aims to assist private foresters and land management regulators to assess proposed regrowth thinning regimes in their region against their biodiversity objectives or regulatory standards.

Method used

Three long-term experiments were used to examine the response of vegetation to regrowth thinning or selective harvesting with cattle grazing. The three experiments all differ with respect to their location, climate, environmental characteristics, forest type, silviculture and management inputs. At each experiment, a variety of response variables were measured describing the forest structure, its vegetation composition and individual tree characteristics such as height, diameter and canopy dominance.

The experiment in the Florentine Valley of central Tasmania was initiated in 1991 to assess the response of vegetation to thinning treatments of varying intensity, implemented using either ground-based or cable harvesting techniques. The site was in even-aged 29 year old *Eucalyptus regnans* wet forest, originating from an intensive regeneration treatment. The climate is cool and wet; the site quality is high.

The experiment in East Gippsland Victoria was initiated in 1988 with the aim of assessing the vegetation response to regrowth thinning in a coastal lowland forest. The *Eucalyptus sieberi* (silvertop) – *Eucalyptus globiodes* (white stringbark) dry forest was un-even aged, originating from a combination of selective harvesting events, and both planned and un-planned fires. The climate is warm and sub-humid; the environment is fire prone and relatively low in nutrients.

The experiment near Grafton in northern NSW was initiated in 1954 with the aim of describing the effect of cattle grazing on the regenerative response in an un-even aged stand of *Corymbia henryi* (spotted gum) – *Eucalyptus moluccana* (grey box) dry open forest following selective harvesting. The site is typical of a relatively low site quality, coastal forest environment subject to long term grazing, repeated low intensity thinning and logging events, and low to moderate planned and un-planned fires. The climate is sub-tropical.

Results/Key findings

Thinning in a single species even-aged regrowth stand in southern Tasmania had an immediate effect on the vegetation and biodiversity, correlated with the intensity of the thinning operation and the degree of soil disturbance resulting from the application of either cable or ground-based harvesting techniques. Ground-based thinning created sufficient soil disturbance to encourage a new cohort of overstorey trees to appear and for ruderal and weed species invasion. Cable thinning, however, led to an expansion of clonal species such as rhizomatous ground ferns. Height growth was similar following either of the four thinning treatments, while the basal area increment varied widely across treatments. Natural self-thinning in the control treatment of the overstorey trees after fourteen years had equalled the stem density present on one of the thinning treatments. Both thinning techniques led to a significant reduction of the scattered tall shrub and small tree stratum as a result of log yarding and felling disturbance. Floristic change immediately post-thinning was of greater significance in the ground-based thinning treatment compared to the un-thinned control. Differences which were clearly evident between the thinning treatments in the initial period of monitoring proved to be of less significance after fourteen years of study.

Thinning in a mixed species un-even aged stand in East Gippsland led to a large increase in total stem density over the seventeen year period, as a result of the in-growth component, despite thinning removing just over half the pre-treatment basal area. In thinned stands the most numerous diameter class was sub-dominant and in-growth stems, while the larger diameter stems retained the dominant crown classification. Changes in the understorey vegetation composition and structure following thinning and its associated site disturbance were abrupt. There was however little evidence indicating that those short term effects would persist after 17 years – the vegetation composition and structure resembled the regrowth forest prior to treatment. Plant species lost in one plot would re-appear in another, indicating the rapid turnover of species in a relatively diverse and nutrient poor coastal lowland forest. Species richness, for example, declined immediately after thinning, but exceeded pre-treatment levels 17 years after thinning. Vegetation structure and complexity similarly increased post-thinning.

In northern New South Wales cattle grazing is commonly employed in open forest environments in association with timber harvesting, thinning and frequent low intensity fuel reduction burning. The effect, however, of ongoing cattle grazing following timber harvesting on seedling and coppice regeneration, survival and growth is poorly understood. A long term study examined the relationship between cattle grazing and density, composition and height growth of eleven overstorey tree species following harvesting in 1954. In the unfenced treatment, 65% of the low lignotuberous regeneration stems present after timber harvesting remained in the lowest height category for the first 2-4 years. In the fenced treatment however, these stems rapidly moved into the taller height categories. A lag of four years in tree growth rates in the unfenced treatment compared to the fenced treatment suggested grazing led to slower growth rates for stems under one metre tall. The differences between treatments became less apparent over time and were negligible after 12 years. Although the effect of grazing on forest structure does not appear to be long-term in this particular forest type, due to the lignotuberous regeneration strategy of the dominant tree species, excluding cattle for the first 2–10 years after

harvesting can ameliorate the short-term impacts of grazing on regeneration of forest trees and potentially of other forest biota.

Implications for relevant stakeholders

The results of these studies provide one of the few examples of medium-term studies examining the vegetation and plant biodiversity response to thinning in southern Australia. Results elsewhere from the literature, indicating short-term effects on species composition and structure, appear simply that – short term in the vicinity of years. The medium-term results demonstrate the resilience of the vegetation to this form of intermediate disturbance, assuming soil disturbance from thinning is kept to a minimum.

The biodiversity responses described differed between treatments, forest types and with time. The results presented here are considered localized to the vegetation types examined, and to some extent the nature of the thinning operations. The results provided should be assessed along with other influences on vegetation structure and composition, particularly repeated burning and the initial floristic composition at the time of treatment. Nonetheless several trends were apparent which are consistent with the broader literature on forest vegetation responses to harvesting disturbance. Private foresters and land management regulators can use this knowledge to help them assess proposed regrowth thinning regimes in their region against their biodiversity objectives or regulatory standards.

Recommendations

The following are the key recommendations provided for private forest managers:

Biodiversity, soil and logging and grazing management

- Thinning technologies which create a mosaic of treatment areas, for example patches of un-thinned vegetation, more open vegetation and variable retention densities should be encouraged.
- Legacies from the previous harvesting event such as large downed logs in various decay stages and large old trees should be retained during thinning.
- A management approach which seeks to enhance the biodiversity response post-thinning could apply a mosaic of disposal burning and non-burning treatments across the thinned area.
- Soil disturbance resulting from thinning operations within regrowth forest has a clear impact on the process of vegetation recovery. Where possible, low impact thinning machinery should be used, and if possible, cable technologies used during commercial thinning as they have a lower environmental impact.
- Where the overstorey trees rely largely on vegetative regeneration (ie coppice or lignotubers), excluding cattle grazing for a period of 3-9 years post-thinning is likely to minimise any loss in tree growth response post-thinning..
- The management of cattle grazing needs to be better integrated with the maintenance of a balanced forest structure designed to produce an acceptable growth increment; with suitable management of forest fuels and fire frequency; and with the broader ecological and biodiversity management objectives generally considered part of a sustainable native forest management system.

Further study

- The thinning studies reported here were based on forests 29-31 years old at the time of thinning. As our ability to predict the response of younger forests to non-commercial or commercial thinning may be limited, study of younger forests potentially subject to thinning is required.
- Biodiversity response to post-thinning vegetation with and without fuel-reduction burning should be investigated.
- Significant areas of forest which are managed jointly for wood and cattle production, also require their forest fuels to be managed to an acceptable level or risk. The interaction of these three objectives is poorly understood and requires examination using carefully designed manipulative experimental treatments.
- Future experimental designs need a more explicit consideration of scale, sampling, replication, silvicultural treatment and measurement issues.

1. Introduction

Thinning is a form of intermediate cutting aimed primarily at controlling the growth of stands by adjusting stand density or species composition (Nyland 1996, Smith *et al.* 1997, Cochran and Barrett 1998). Commonly associated with thinning is stand improvement, for example, the culling of non-merchantable stems to waste. The basic assumption when applying thinning to a forest stand is that the growing space on a particular site is limited by the available light, water and inorganic nutrients, and that thinning will promote the growth of desirable trees already present in the stand by releasing and re-allocating a greater proportion of those limiting resources to the retained stems. The general effect of thinning is crown and stem expansion. The typical growth response is an increase (but highly variable) in stem diameter, and a modest height increase (relative to controls). Stand biomass growth is directly proportional to light absorption by the forest canopy (West and Wells 1992), where light absorption is considered a surrogate for photosynthetic activity in the crown.

The natural application of thinning in a stand is generally termed self-thinning, a concept with a vast published literature from a theoretical perspective (Westoby 1984). Self-thinning is the basic concept that links the biological study of an individual tree with the aggregate properties of the forest patch. It has been described as the only law of plant ecology (Li *et al.* 2000). Self-thinning is also central to the understanding of the mechanisms in which forest systems stabilise (Barnes and Roderick 2004).

Regrowth thinning is most frequently applied with the objective of directing wood production and stand yield towards those forest products considered more desirable, for example, sawlog versus pulpwood production. Biologically, it represents a form of intervention to the existing self-thinning processes operating at the stand level, by transferring resources from uncompetitive individuals towards competitive ones. To be most effective, regrowth thinning should be applied to relatively young stands, ensuring growth concentration on fewer trees whilst the growth increment is relatively high (West 1991).

The response of regrowth forest to thinning can be divided into several components related to a) its initial composition and b) the nature of the disturbance to the overstorey, and c) indirect disturbance to the understorey and soil environment. Variation in the direction and rate of secondary succession post-thinning is likely to be related to the diversity and cover of plant groups with different regenerative traits, the presence of plant species susceptible to mechanical disturbance and the frequency with which late successional specialist species recover.

Disturbance to the soil environment is an unavoidable consequence of logging. Disturbance to soil organic matter, compaction, and exposing or displacing the soil horizons results in changed processes of soil nutrient, water and air availability, and at larger scales, mass movement and sediment transport (Halpern and McKenzie 2001). Soil disturbance will affect vegetation composition by its direct effects on site disturbance (Smith and Wass 1996) and indirectly by its effects on site productivity (Krag *et al.* 1991, Senyk and Craigdallie 1997). The process of yarding timber will affect the vegetation response by removing and damaging existing individual plants and uprooting or burying the above-ground stems of understorey species (Halpern and McKenzie 2001). While longer term vegetation responses are likely to be mediated by the nature of the overstorey development, shorter term responses are more likely to be mediated by levels of ground disturbance resulting from logging (Halpern and McKenzie 2001).

Most of our knowledge of the effects of thinning on vegetation is based on studies in north American and Canadian softwoods. These studies typically correlate changes in understorey vegetation after thinning to the changed light regimes following overstorey thinning (Young *et al.* 1967, Crawford 1976, Hill 1978, Metzger and Schultz 1984, Alaback and Herman 1988), and occasionally this is tested through experimentation (e.g. Riegel *et al.* 1995, Bauhus *et al.* 1999). More recent studies have argued that thinning-induced changes in vegetation composition are simply a response to transient increases in resource levels, and not the thinning-induced vegetation structural changes (Lindh and

Muir 2004). Studies which interpret the thinning effect as a function of soil disturbance consider the increased heterogeneity of soil conditions post-thinning to lead to a temporary increase in species richness, and even to be necessary for the maintenance of species diversity (Brunet *et al.* 1996). When soil disturbance in the context of ground-based thinning has been investigated, Reader (1987) reported that ground-based thinning led to a loss of woody species with increasing thinning intensity. In contrast to woody species, the loss of herbaceous species did not increase significantly with an increase in thinning intensity (Reader 1987). However, Young *et al.* (1967) observed that heavily disturbed areas in mixed coniferous forests following ground-based thinning in Oregon were easily invaded by pest species such as the Canadian thistle (*Cirsium arvense*).

There is a wide divergence of opinion from northern hemisphere studies on the effects of thinning on understorey vegetation. Some consider it has minimal effect on understorey vegetation (Sullivan *et al.* 2002, Sarkkola *et al.* 2005), the effect is short lived (Thysell and Carel 1999), the effect may persist only for a decade (Brockway and Lewis 2003), the effects occur in distinct pulses such as immediate, short term and long term gradual (Moore *et al.* 2006) or the longer term pattern of understorey succession is not altered (Crawford 1976, Metzger and Schultz 1984, Alaback and Herman 1988). At the landscape scale, thinning will lead to greater heterogeneity in the spatial structure of the forest stands, but within those stands, will lead to a more homogeneous microstructure (Montes *et al.* 2005). Others consider that thinning will alter the relationship between overstorey species (Ball and Walker 1997), stimulate understorey development (Tappeiner and Zasada 1993, Albrecht and McCarthy 2006) and alter species' relative covers (Ward 1992, Zenner *et al.* 2006). Species which responded to thinning in one study (Falkengren-Grerup and Tyler 1991) have been reported as being un-responsive in another study (Graae and Heskjaer 1997). More recent studies suggest, however, that ground-based thinning will increase species richness, increasing the cover of herbaceous species, and promote site colonisation by weeds (Schacht *et al.* 1988, Bailey *et al.* 1998, Zenner *et al.* 2006). A recent review of the Alaskan data has suggested the applicability of the existing research, conducted largely in young, even-aged stands may be more limited at the landscape scale than was originally thought (Hanley 2005).

Thinning has also been implicated in crushing large logs in advanced decay stages (McCarthy and Bailey 1994) and limiting the contribution of large coarse woody debris (Hayes *et al.* 1997, Prescott 1997, Fleming and Freedman 1998, Etcheverry *et al.* 2005) and litter (Roig *et al.* 2005) to the forest floor. The considerable volumes of logging debris generated by thinning can also kill existing vegetation and prevent its re-establishment (Metzger and Schultz 1984, Newton and Cole 2006). Thinning has also been viewed as a means of increasing the structural heterogeneity (Hunter 1999, Sullivan *et al.* 2005), floristic diversity (Cole 1996), old-growth characteristics (Bailey and Tappeiner 1998, Busing and Garman 2002, Homyack *et al.* 2004, Lindh and Muir 2004) or proportion of either mid to late successional understorey species (Chan *et al.* 1999, Homyack *et al.* 2005) or early successional species (Albrecht and McCarthy (2006) in young or intensively managed forest stands. Halpern *et al.* (1999) argued, however, that thinning will contribute little to enhancing understorey diversity in managed forests if it relies on the species response to occur from soil seed banks. Thinning will also increase levels of photosynthetically active radiation, decrease relative humidity, and increase air temperatures (Riegel *et al.* 1992).

Cable thinning of regrowth forests is a relatively new practice, and is still being evaluated globally (McConcachie 1993, Verani 1995, Visser and Stampfer 1998, Newton and Cole 2006) and in Tasmania (Cunningham 1997). Favourable results, especially on sensitive sites (Visser and Stampfer 1998) have led to predictions the practice will increase (McNeel and Dodd 1996, Cunningham 1997). Compared to ground-based systems, cable thinning can either result in less damage (Fairweather 1991, Youngblood 1990, Minato *et al.* 1997, Han *et al.* 2000), more damage (Nichols *et al.* 1994) or the same amount of damage to retained stems (Thompson 1996). In addition, cable thinning retains more advanced tree regeneration and understorey vegetation, and results in comparatively less site invasion by disturbance adapted woody and herbaceous species (Pominville 1993). While most disturbance during cable thinning occurs during the yarding of regrowth logs along the outcrops to the log dump or landing (Han *et al.* 2000), it effectively confines soil disturbance and tree damage into

these log extractions routes. Ground-based thinning, however, leads to increased soil compaction and mineralisation, surface disturbance and root damage, and reduces organic matter content (Wronski 1984, McIver 1988, Terlinden and Andre 1988, Han *et al.* 2000).

The cable thinning experimentation reported here formed part of the first operational trials of the technique in Australia. While other trials have been conducted in pine plantations in the ACT (Melmoth 1979), in New Zealand (McConachie 1993, Murphy 1982) and in oak (*Quercus cerris*, *Q. pubescens*) coppice forests in central Italy (Verani 1995), none have examined the native vegetation response. Additionally the incidence of thinning using cable yarders has been increasing (McNeel and Dodd 1996). In Tasmania it increased from virtually nil in 1993 to 1600 ha⁻¹/yr in 1998 (Forestry Tasmania 1998). In Victoria the introduction of more sophisticated feller-processors and forwarders is expected to continue to increase the available regrowth forest resource for thinning (Churton 1999).

The majority of Australian thinning studies have focussed on the response of the overstorey (e.g. Brown 1997, Connell and Raison 1996), an approach described by Reader (1987) as focusing on the remaining trees and not the component lost after treatment. In Australia, the ecological effects of native forest regrowth thinning are relatively poorly known (O'Connell 1997) and the complexities of multi-species interactions and responses in the understorey provide challenges to experimenters (Bailey and Tappeiner 1998). Studies in comparable East Gippsland forest communities describing the short term response (3-6 years) of the flora (Bauhus *et al.* 2001) and fauna (Kutt 1994) to regrowth thinning indicated a thinning led to a relatively minor change in plant species and a reduction in arboreal marsupials due to the reduction of structural features normally associated with mature forest.

Cattle grazing has been a traditional activity in the forests of northern NSW since the mid 1800's and generally involves a rotational system with no fixed stocking rates. Much of the private forest in this region has a long history of seasonal cattle grazing combined with opportunistic harvesting on a single tree selection basis, often with little attention being paid to the silvicultural condition of the forest post-harvesting (Combe *et al.* 1998). Cattle are normally agisted in the forests over winter to provide grazing relief to landholders' pastures. Grazing area and intensity will vary with time of the year, seasonal conditions and market forces. Cattle grazing relies upon a periodic burning of the ground cover to promote the growth of palatable grass species under the forest cover, generally at a frequency of every 2-5 years during the September-December (Spring) fire season. The combination of frequent burning and rotational cattle grazing has the potential to retard seedling recruitment, lignotuber release and the coppice growth response following timber harvesting unless cattle grazing and burning are removed from regeneration areas for a sufficient length of time. These forest stands have the potential to become overstocked with small diameter trees in the lignotuberous seedling and small advance regrowth stage, and potentially be maintained almost indefinitely in this state of disequilibrium (Henry and Florence 1966). The repeated imposition of thinning and single tree selective harvesting with forest grazing and burning could therefore lead to a decline in forest health and create a stand structure incompatible with the maintenance of a productive forest (Bauhus *et al.* 2002).

The interaction of forest grazing with forest establishment and growth is often unclear where the forest manager seeks an economic return from both the trees and the cattle and further complicated where biodiversity benefits are sought. While commercial forest thinning or diameter-limited cutting will provide a forest product, the immediate resumption of forest grazing has the potential to retard the post-thinning or logging regeneration and growth response through ongoing site disturbance. The repeated imposition of thinning and selective harvesting with forest grazing (and its associated burning practices) also has the potential to lead to a long term decline in the health and productivity of the forest stand.

The effects of cattle grazing on forest structure in eastern Australia have been difficult to quantify equivocally, as there is a paucity of published experimental studies, with the available literature frequently relying upon correlative approaches (with their inherent limitations, see Bennett and

Adams 2004). Such correlative approaches, based on surveying a range of sites of known or presumed management history, are constrained by a range of confounding factors such as variation in species responses across habitat types, the interaction of frequent burning and grazing, and by the treatment of grazing simply as an ordinal site variable, in the absence of any quantitative description of grazing seasonality, stocking rates or watering points. No published studies on the vegetation and biodiversity response to selective harvesting associated with continuous cattle grazing in eastern Australian are known to exist. Overseas studies examining the interaction of cattle grazing and post-logging or post-thinning regeneration have produced mixed results. Grazing ungulates can have a significant impact on forest regeneration and therefore forest development (Jorritsma 1999), or grazing by ungulates and cattle can have relatively little impact compared to the response of the vegetation to thinning and logging (Brockway and Lewis 2003). The majority of these overseas studies are also derived from even-aged single species stands and may have limited relevance to an Australian forest ecosystem.

In northern NSW, grazing cattle will prefer previously logged forest, where the resultant open vegetation structure promotes a palatable ground cover (York 1997, 1998, 1999, York and Tarnawski 2004). Cattle grazing will lead to a reduction in the cover of small shrub and herbaceous vegetation while taller shrubs are unaffected (York 1997, 1998, 1999, Henderson and Keith 2002). Cattle grazing also promotes open grassy understoreys (Tasker and Dickman 2004, Tasker and Bradstock 2006), reducing the richness of low shrubs and perennial herbs (Pettit *et al.* 1995, Abensperg-Traun *et al.* 1996). Studies overseas have described a range of vegetation responses to cattle grazing in forests, including reduced stem density (especially of lower diameter stems) and a shift from tree sexual reproduction to vegetative regeneration (Cierjacks and Hensen 2004). Other studies describe an effect where grazing led to the retardation of the height growth of regeneration and small trees (Jorritsma 1999, Gill 1992), suppressing the new age cohorts (the genet level response of Danell *et al.* 2003) thereby delaying the development of the stand (Bergquist *et al.* 1999; Joys *et al.* 2004) or tree reproduction (Zamora *et al.* 2001). Grazing will also stall and modify the direction of succession (Côté *et al.* 2004), or slow the successional progression of the understorey following clearcutting (Bergquist *et al.* 1999) or coppice regeneration (Joys *et al.* 2004).

1.1 Scope of this report

In this report, three long term experiments examining the response of vegetation to regrowth thinning or thinning with cattle grazing are presented. The three experiments all differ with respect to their location, climate, environmental characteristics, vegetation, silviculture and management inputs.

The first experiment was initiated in 1991 in the Florentine Valley of central Tasmania with the aim of describing the response of vegetation to thinning treatments of varying intensity implemented using either ground-based or cable harvesting techniques. The climate is cool and wet, the site quality is high; the *Eucalyptus regnans* wet forest present was even-aged, originating from an intensive regeneration treatment.

The second experiment was initiated in 1988 in East Gippsland Victoria with the aim of describing the vegetation response to regrowth thinning in a relatively low nutrient coastal lowland forest. The climate is warm and sub-humid, the environment is fire prone and relatively low in nutrients, and the *Eucalyptus sieberi* (silvertop) - *Eucalyptus globiodes* (white stringbark) dry forest was un-even aged, originating from a combination of selective harvesting events, and both planned and un-planned fires.

The third experiment was initiated in 1954 near Grafton in northern NSW with the aim of describing the effect of cattle grazing on the regenerative response in un-even aged stand of *Corymbia henryi* (spotted gum) - *Eucalyptus moluccana* (grey box) dry open forest following selective harvesting. The climate is sub-tropical, and the environment is typical of a relatively low site quality coastal forest environment subject to long term grazing, repeated low intensity thinning and logging events, and low to moderate intensity burning. The objective of this third case study was to examine the effects of

cattle grazing following selective harvesting on forest structure, and therefore the potential of the forest stand to regenerate and grow following harvesting. Tree species composition, density and growth were compared in fenced and unfenced plots for twelve years, and then for a further twenty nine years where all plots were subject to cattle grazing.

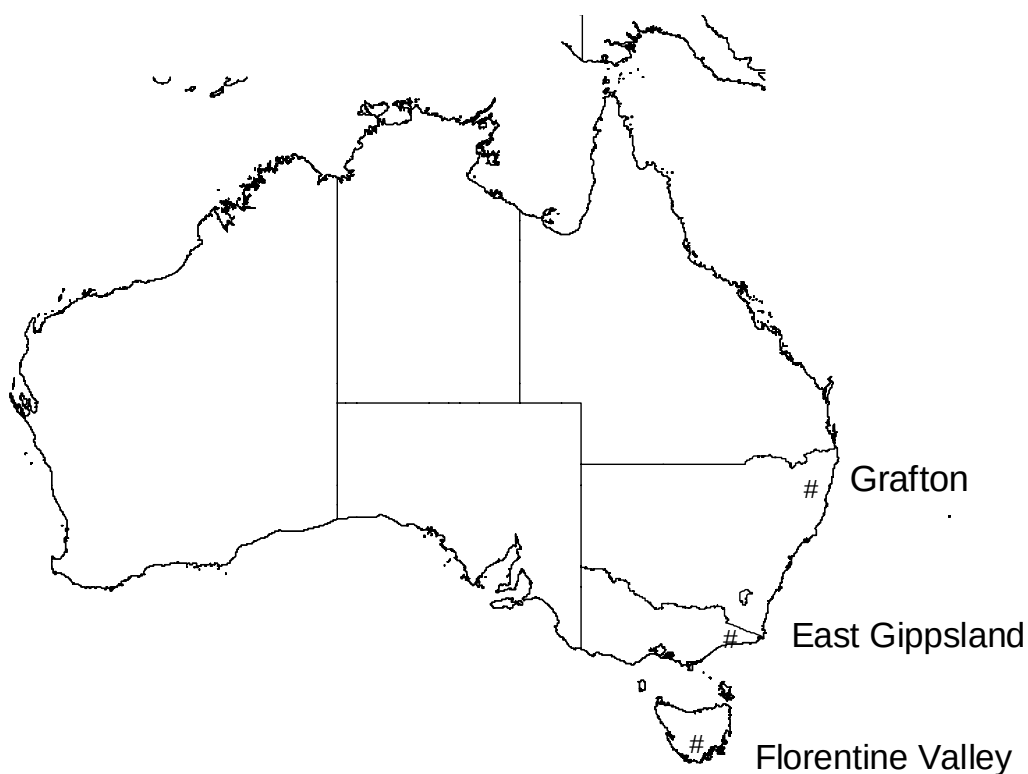


Figure 1.1 Location of the three case studies

The purpose of presenting the case studies is to extend the relevance of this report beyond a single forest environment and secondly to examine the extent to which patterns in the vegetation biodiversity response vary, or are common to, a range of forest environments.

2. Case Study 1. Thinning in an even aged single species forest stand in Tasmania

2.1 Objectives

The objectives of this case study were to examine the vegetation and structural response to thinning in an even aged 29 year old single species forest stand in southern Tasmania. The specific objectives were to examine the role of thinning method (cable or conventional ground-based) on the vegetation response, and secondly the significance of thinning intensity, or the proportion of the basal area removed, on the vegetation response.

2.2 Methods

2.2.1 Study Area

The study was located on Jungle Rd, Repulse area, Florentine Valley, southern Tasmania (grid reference E460500, N5297200, Table 2.1). The 18 ha management unit (known as Repulse 3) was 29 years old when subject to the thinning treatment in 1991. The vegetation consists of *Eucalyptus regnans* silvicultural regrowth originating from logging in 1961 and regeneration treatments (burning and sowing) between 1961 and 1964. The photo-interpreted forest type (as per Forestry Tasmania, Stone 1998) is E2cST.

Table 2.1 Environmental data for the southern Tasmania study area.

Mean annual rainfall (mm) ¹	Mean daily minimum of coldest month (July) ¹	Mean daily maximum of warmest month (February) ¹	Elevation, aspect and slope	Geology	Soils
1163 - 1214	0.3 - 1.3	19.8 - 21.6	460 - 490 m, north easterly aspect and slope of 10 degrees	Jurassic Dolerite	gritty brown clay loam with boulders comprising 5-20% of the surface

¹ www.bom.gov.au, from Tarraleah village and Maydena climate stations (BOM 2005)

Cable thinning was carried out with a purpose built skyline cable system using a gravity return carriage mounted on a Prentice excavator (Figure 2.1). The harvesting pattern consisted of 4 metre wide outrows at 15 m centres placed at right angles to the contour (Figure 2.2). Trees were felled manually and logs were secured to the choker lines and pulled laterally into the outrow and then upslope along the outrow to the cable yarder (Figure 2.2). Logs were partially suspended, allowing the load to clear obstacles on the ground and minimise soil and litter disturbance. The cable yarder remained on the road and logs were yarded along the outrows and stockpiled on the roadside. In the flatter ground-based treatment, thinning was undertaken using a Bell tracked feller-buncher mounted with a hydraulically operated chainsaw. Felled logs were placed in bundles within the treatment area and snigged to the roadside landing for stockpiling with a CAT rubber tired grapple-skidder. The thinning prescription for both the ground-based and cable treatments was form, vigour and spacing. The experiment aimed to limit damage to retained trees in both treatments to less than 10%.

2.2.2 Experimental Design

The Repulse 3 study area was divided into four treatment areas determined by ground slope and the position of the access roads. The steeper area (8-10°) was allocated to cable thinning and the less steep area (4-6°) to ground-based thinning (Figure 2.3).



Figure 2.1 Cable thinning at Repulse 3, southern Tasmania in 1991. The skyline cable system used a gravity return carriage mounted on a Prentice excavator to yard logs upslope to the roadside landing

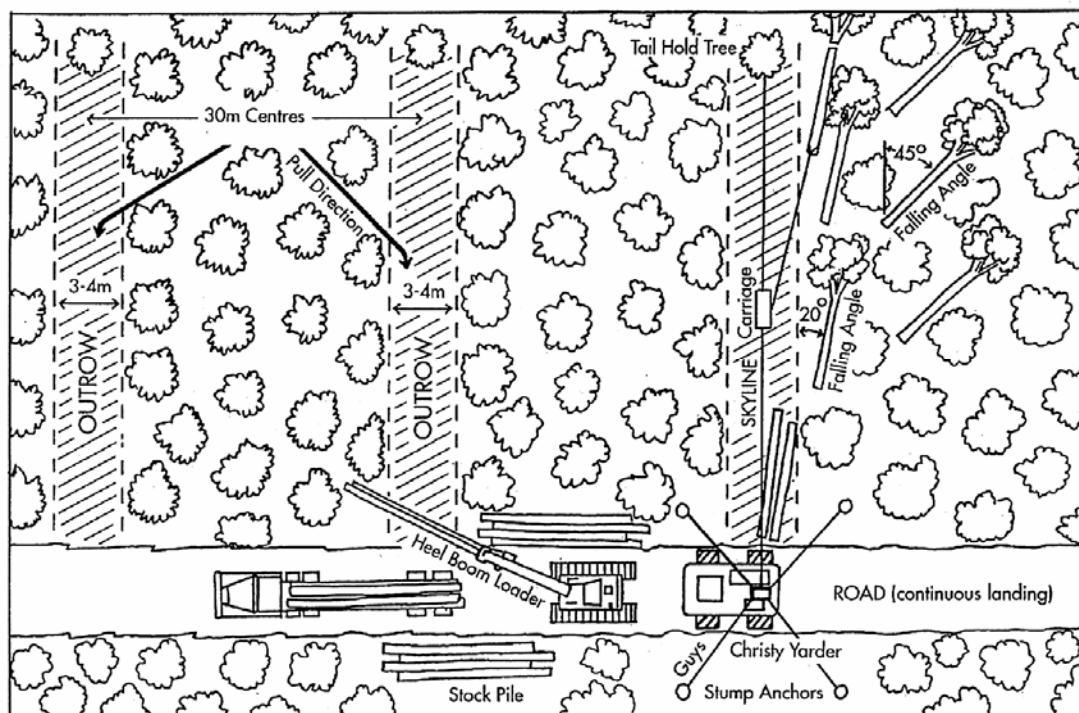


Figure 2.2 Diagram of the cable thinning operation at Repulse 3 southern Tasmania, vegetation surveys were undertaken in the sections between outrows (diagram from Cunningham 1997).

The cable thinning treatment area portion was further subdivided into two equal areas, one to be thinned to 150 stems/ha⁻¹, the other to 270 stems/ ha⁻¹. The ground-based area was to be thinned to 230 stems/ ha⁻¹. The treatments could not be randomly assigned due to the constraints of providing adequate machinery access for the cable yarder on the central access road and the gravity return requirements for operating the cable carriage (Cunningham 1997).

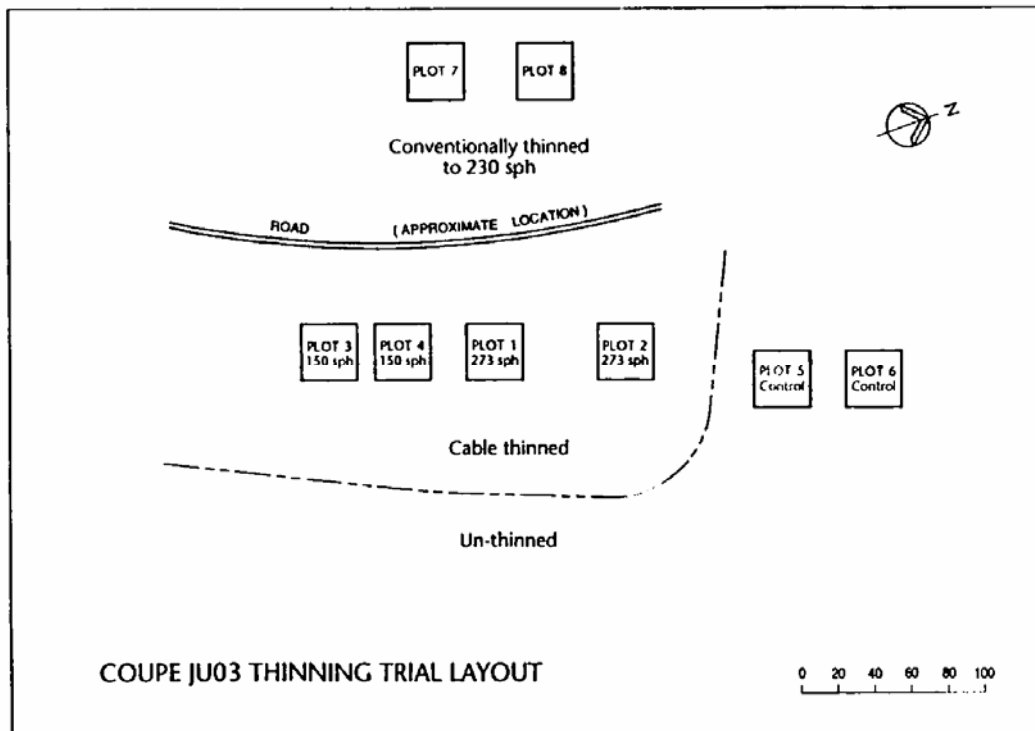


Figure 2.3 Distribution of vegetation quadrats (squares) within the three treatment areas (conventional ground-based thinning, cable thinning and unthinned control) at the Repulse 3. sph = stems per hectare.

Soil disturbance and canopy opening each have an effect on forest regeneration and these two factors probably interact. However, they were not considered separately in this experiment. It was not practical to mechanically thin the overstorey (and therefore change the light and temperate regime) without causing some soil disturbance.

2.2.3 Vegetation and Tree Sampling

For each of the four treatments two replicate 30 x 30 m vegetation quadrats were permanently established, nested within or overlapping the stand inventory plots. Within each 30 x 30 m quadrat, all vascular plant species growing within or projecting into or above the quadrat were recorded and assigned a visually assessed cover-abundance value modified from the Braun-Blanquet (Braun-Blanquet 1932, Poore 1955) scale. Cover was defined as foliage projective cover following Walker and Hopkins (1988). The height ranges, three dominant species and foliage cover of each recognisable vegetation stratum was recorded, in addition to the percentage cover of the ground layer covered by rocks, logs, branches, litter and bare ground.

Pre-treatment and immediately post-treatment tree inventory data was provided by Norske-Skog P/L and Forestry Tasmania. Prior to treatment, inventory plots were systematically established throughout the study area using a system of regrowth assessment strips with plots of either 0.02 or 0.045 ha. A range of tree and stand variables were scored using either the REGASS or CFI inventory systems (Forestry Tasmania 1988), of relevance to this study was species, tree diameter at 1.3 m, total height (m), and crown dominance (dominant, sub-dominant, co-dominant, suppressed, in-growth and dead). Pre-treatment (1991) tree heights in REGASS plots were measured using a clinometer for the four tallest trees per 0.02 ha plot or for a range of five dominant, five co-dominant, and five sub-dominant trees per 0.045 ha plots. Tree heights in 2005 were however measured for all trees regardless of height or dominance using a Vertex III laser height measuring instrument. The minimum tree

diameter measured was 1 cm. Following the thinning treatments in 1991, the inventory plots were re-configured with respect to their pre-treatment strip arrangement and dimensions. New inventory plots were re-established within the boundaries of the treatment areas which were not known with precision until thinning was completed. The dimensions of the re-configured inventory plots were either 25 x 25 m, 40 x 40 m or 35 x 50 m, with two plots per treatment. Vegetation quadrats (30 x 30 m) collected post-thinning in 1992 and 2005 were super-imposed over or within the re-configured inventory plots to ensure continuity of data collection.

The two replicate quadrats within the treatments were positioned at the same point on the contour and were approximately 40 m apart at their centroids (Figure 2.3). Vegetation surveying in the cable thinning experiment was undertaken in the sections between outrows (Figure 2.3 – 2.4) and in the ground-based thinning experiment in areas between the snig tracks created by the skidder. Post-thinning vegetation sampling and tree inventory was undertaken between April-May 1992 and in February 2005.



Figure 2.4 Cable outrows at Repulse 3 were clearly visible in 1992 (left) but were largely obscured in 2005 (right).

Litter displacement was defined as the removal of leaf litter and other fine woody debris sufficient to expose mineral soil. Soil gouging was defined as the displacement of the organic layer and A horizon to a depth >10 cm for > 1 m in length. Residue was defined as an area with accumulated residual of tree heads, branches and bark from the thinning operation. The proportion of the treatment areas subject to litter displacement, soil gouging and thinning residue was estimated with a series of random traverses using 1 m wide belt transects. For each treatment, a minimum of 10% of the treatment area was surveyed.

2.2.4 Data Analysis

Percentage cover scores derived from each 30 x 30 m quadrat were converted to the mid-point of the respective cover class. Species were grouped into the growth form categories of trees, shrubs, herbs, graminoids and ground ferns and their percent cover scores averaged by growth form, treatment and year. Mean species foliage cover scores then represented the mean value per species in that growth form per treatment or year. Association matrices based on the 1992 and 2005 measurements, and the combined data set, used the Bray-Curtis measure (Bray and Curtis 1957). Structural attributes (eg stratum cover) were averaged across treatments and measurement years.

The multivariate relationship between treatments was examined using ordination, based on the previous Bray-Curtis association matrix. Two ordination techniques were applied to the floristic percentage cover data, principal components analysis (PCA) and non-metric multidimensional scaling (NMS). Principal components analysis provided the most interpretable solution to the ordination of the year 1 and year 14 measurements, while non-metric multidimensional scaling provided the most interpretable solution to the pooled data set. NMS ordinations in the global form were derived from 50 multiple runs, using 10 different random starting configurations, with a maximum of 100 iterations. Ordinations were performed in 1-3 dimensions, and the minimum stress solutions for each dimension examined. Stress is “a measure of departure from monotonicity in the relationship between the dissimilarity (distance) in the original p-dimensional space and distance in the reduced k-dimensional ordination space” (McCune and Mefford 1999). The appropriate number of dimensions (axes) was chosen through an examination of the final stress versus the number of dimensions. The stability criterion for terminating successive iterations was 0.001. The axis scores from the 1-3 dimensional solutions were saved and examined graphically using 3-dimensional plots rotated on the x-axis and through graphical overlays. Both NMS and PCA ordinations were implemented in PC-ORD v. 4.35 (McCune and Mefford 1999). The significance of the cumulative percentage of variance explained by each eigenvalue in PCA is assessed using the Broken-stick method following Jackson (1993).

Hierarchical clustering was performed on percent cover floristic data using the unweighted pair-group method (UPGMA) (Sokal and Michener 1958). The distance measure was Bray-Curtis (Bray and 1957). Dendrograms were generated from the cluster analysis as a visual representation of the binary tree structure. Compositional differences between the four treatment groups were examined using the ANOSIM algorithm of Clarke and Green (1988). This method is effectively a distribution free one-way ANOVA (Clarke 1993). In this procedure, the test statistic was based on 10,000 randomisations, groups were defined by the four treatments, with two replicates per treatment. The test statistic r is constrained to values of -1 to 1. The value of r is indicative of differences among the treatment groups. Species fidelity with treatments and years was assessed using indicator species analysis (Dufrene and Legendre 1997). Relative abundance was defined as the average abundance of a given species in a treatment group divided by the average abundance of that species across all treatment groups, expressed as a percentage. It was implemented in PC-ORD v. 4.35 (McCune and Mefford 1999). Species abundance was derived from the midpoint percentages of the Braun-Blanquet foliage cover classes. Species richness is defined as the number of species per plot (0.09 ha) The index of species diversity used incorporates both species richness and abundance is Shannon's diversity index (H), (after Shannon and Weaver 1949). Evenness (E) of species abundances in a plot follows (Pielou 1966, 1969). It is the ratio of observed H to the maximum H if all species per plot were present in equal abundance. The index of species dominance per plot is the complement of Simpson's (D) diversity (Simpson 1949, McCune and Mefford 1999). Tree diameter distributions were analysed graphically by stem density in 10 cm classes, tree height distributions were analysed in 5 cm classes. Mean dominant height is defined as the height of the tallest stratum (Horne *et al.* 1986) per plot. It was assessed for 50 trees/ha⁻¹ and mean values presented graphically by measurement year. Crown dominance per tree was expressed as a proportion of the total number of stems/ha⁻¹ per treatment.

2.3 Results

Prior to thinning, the diameter distribution of the *E. regnans* overstorey was approximately normally distributed for the ground-based and control thinning treatments (Figures 2.6 and 2.8). In the two cable thinning treatments, the distributions prior to thinning either were skewed towards the 40-50 cm class or distributed evenly across the 20-30, 30-40 and 50-60 cm classes (Figures 2.5 and 2.7). In the first year following thinning, the distributions were flattened across the diameter classes, along with a corresponding reduction in stem density. Fourteen years after treatment, the classes have continued to flatten, in the ground-based treatment there is an increase in the smaller stems (in-growth) in addition to the migration of stems into larger size classes. The two cable thinning treatments have similar shaped distributions, and exhibit a reduced and delayed response for in-growth. In the control, the distribution after fourteen years similarly is flattened, with an overall movement of stems into larger size classes and a reduction in density. In the controls no in-growth is evident.

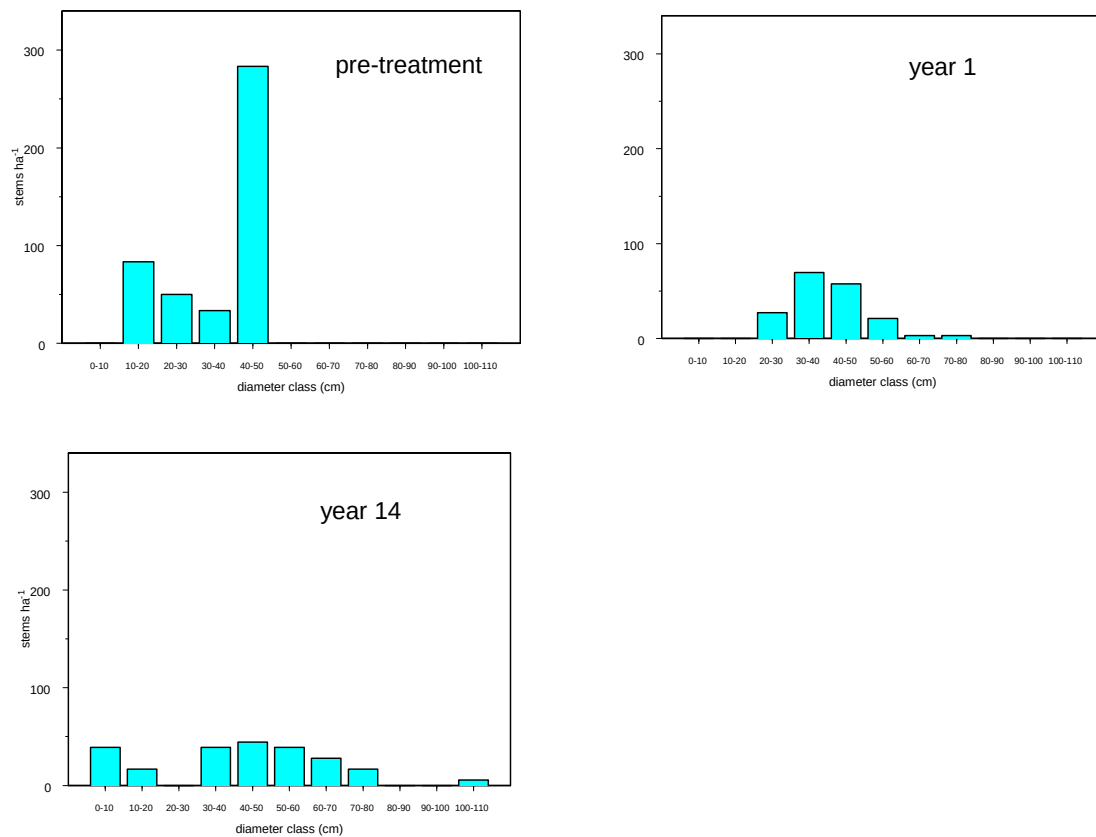


Figure 2.5 Diameter distributions prior to, and 1 to 14 years post-thinning on the 150 sph cable thinning treatment at Repulse 3.

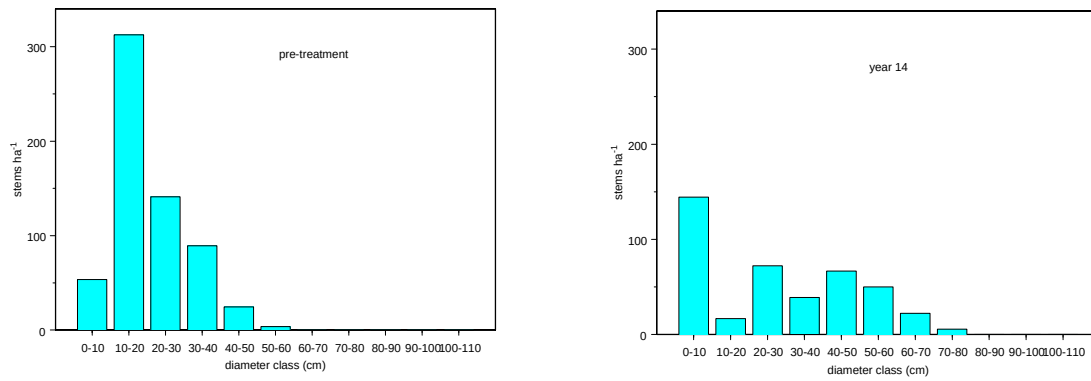


Figure 2.6 Diameter distributions prior to, and 14 years post-thinning on the ground-based 230 sph thinning treatment at Repulse 3. Data is not available for year 1.

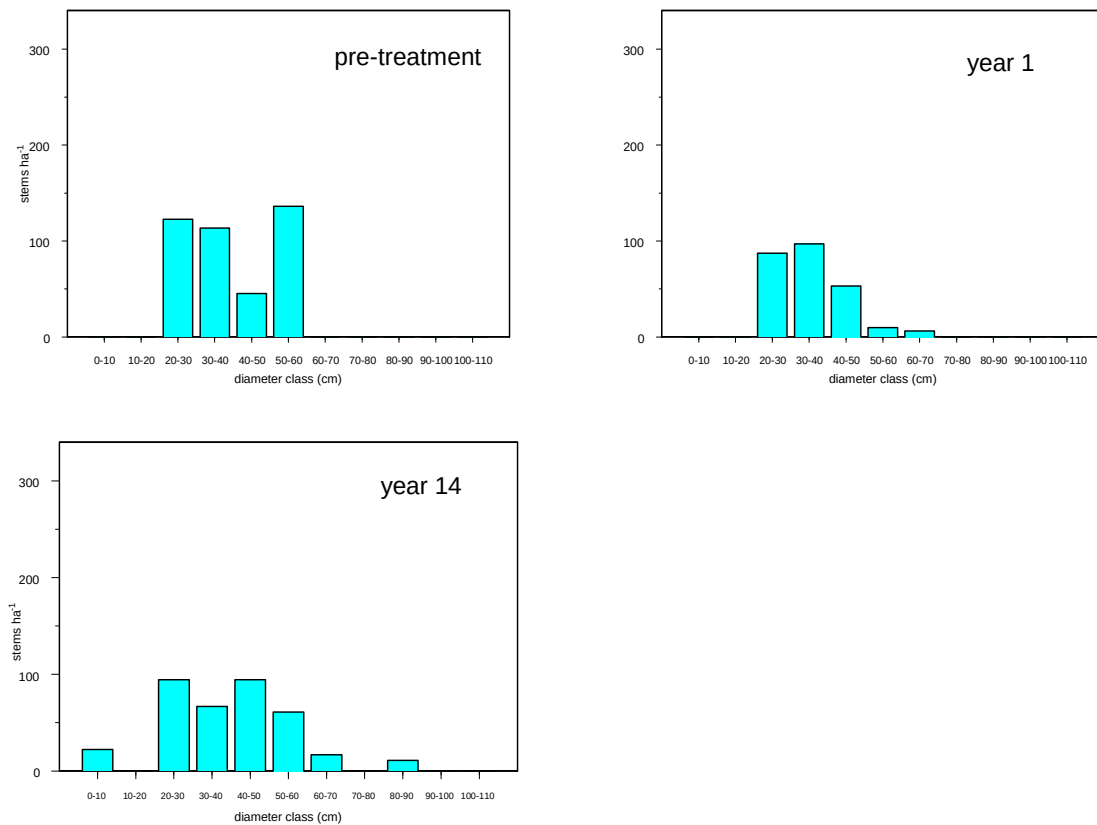


Figure 2.7 Diameter distributions prior to, and 1 to 14 years post-thinning on the 270 sph cable thinning treatment at Repulse 3.

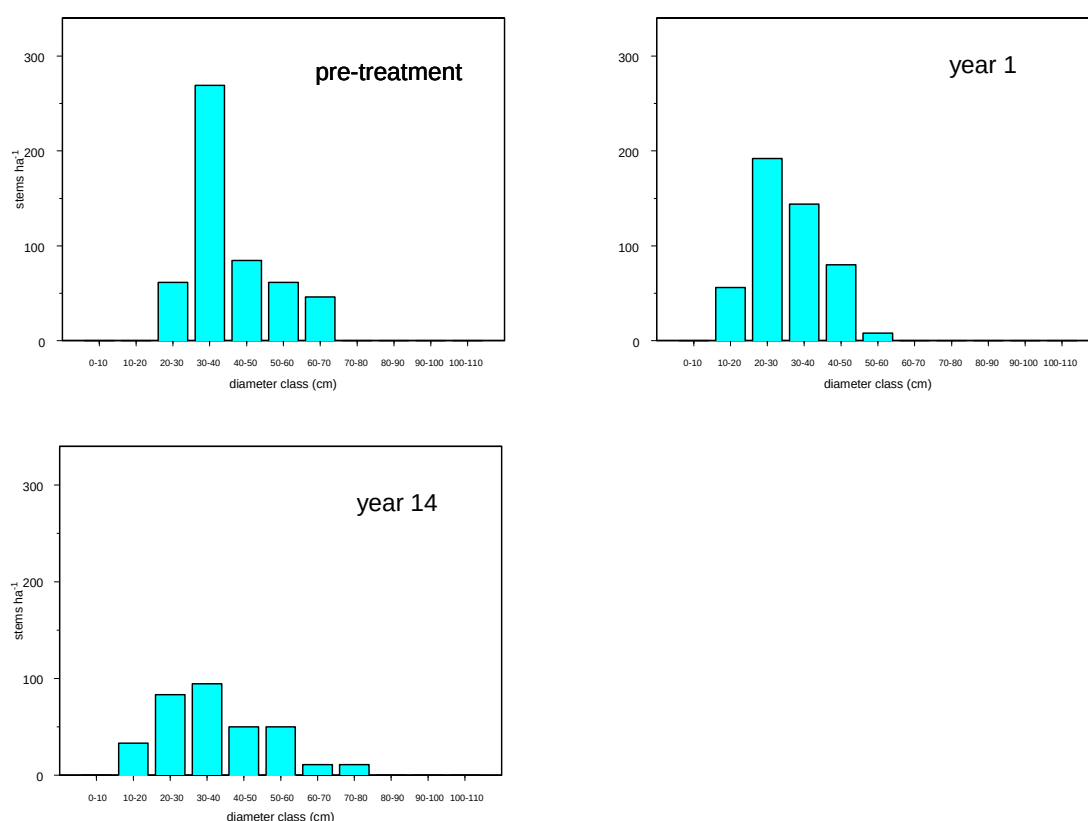


Figure 2.8 Diameter distributions on the control (un-thinned) treatments prior to, and 1 to 14 years post-thinning treatment at Repulse 3.

Height increment was broadly similar across all treatments for the fourteen year period (Figure 2.9), with a proportionally greater height increase for the most intense thinning treatment, cable thinning to 150 stems ha^{-1} . The total density of stems was reduced in proportion to the respective thinning treatment densities sought by the thinning schedules (Figure 2.10). After the initial decline in stem density related to treatment, all treatments exhibited a modest increase over the fourteen year period, except for the controls where stem densities continued to decline. After fourteen years, self-thinning in the control treatment equalled the stem density present in the cable 270 stems ha^{-1} treatment.

The proportional increase in basal area from year 1 to year 14 was greatest for the cable 270 stems ha^{-1} treatment (Table 2.2).

Table 2.2 Basal area ($\text{m}^2 \text{ha}^{-1}$) of *Eucalyptus regnans* stems at Repulse 3 prior to and post-thinning.

	Treatment			
	cable 150 sph	cable 270 sph	g-based 230 sph	control
Pre-treatment	32.8	36.0	34.4	35.4
Year 1	26.1	28.9	no data	39.4
Year 14	41.5	50.9	39.4	45.3
% increase (yr 14-yr 1/yr 1)	59.0	76.1	na	14.9

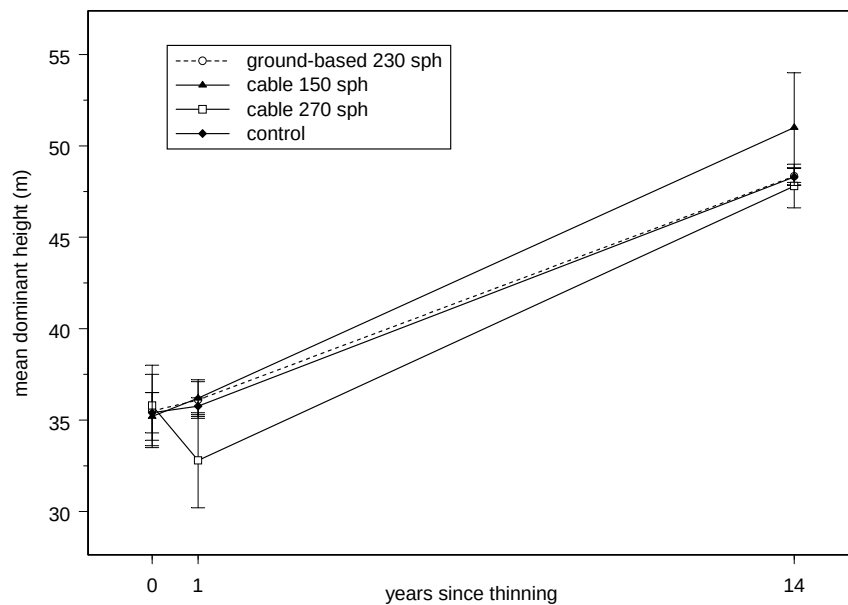


Figure 2.9 Mean ($\pm$ se) dominant height of 50 tallest trees per ha⁻¹ for each treatment following cable or ground-based thinning at Repulse 3.

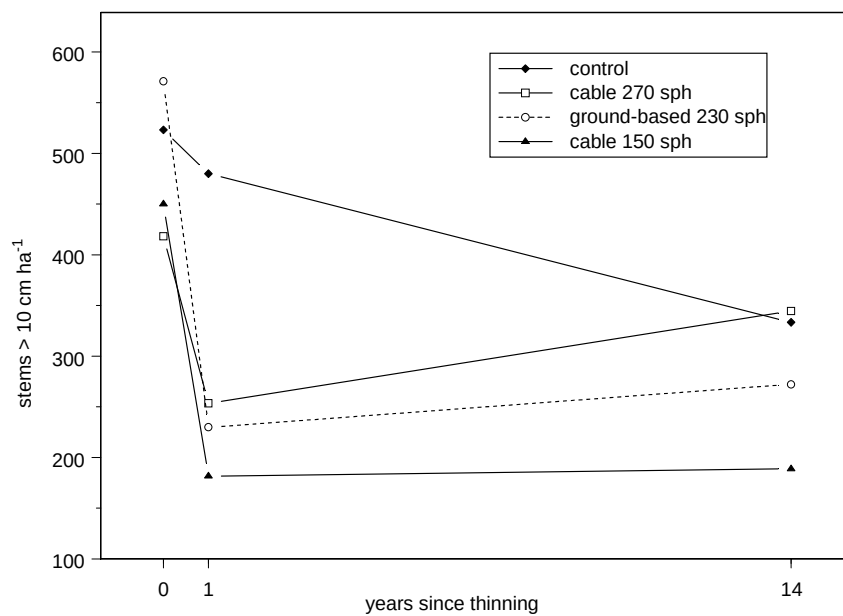


Figure 2.10 Density of all *Eucalyptus regnans* live stems following cable or ground-based thinning at Repulse 3.

Tree height distributions prior to treatment at Repulse 3 were approximately normal (Figure 2.11). Fourteen years after thinning the relative proportions of trees differed in all height classes. The ground-based treatment was notable for the proportion of in-growth in the 0-5 and 5-10 m height classes (Figure 2.12). The control site at fourteen years most closely approximated the distribution of height class values present prior to treatment.

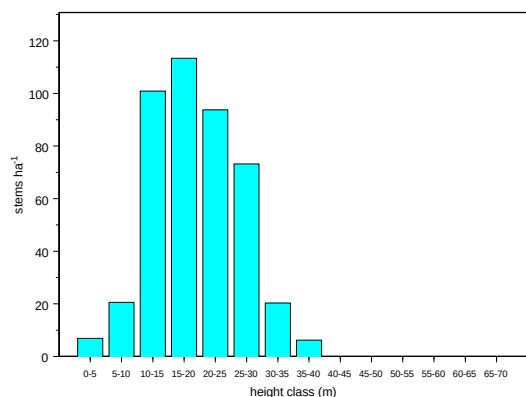


Figure 2.11 Tree height class distribution prior to thinning at Repulse 3. The pre-treatment data is not segregated by treatment as the initial assessment was done prior to the treatment areas being established.

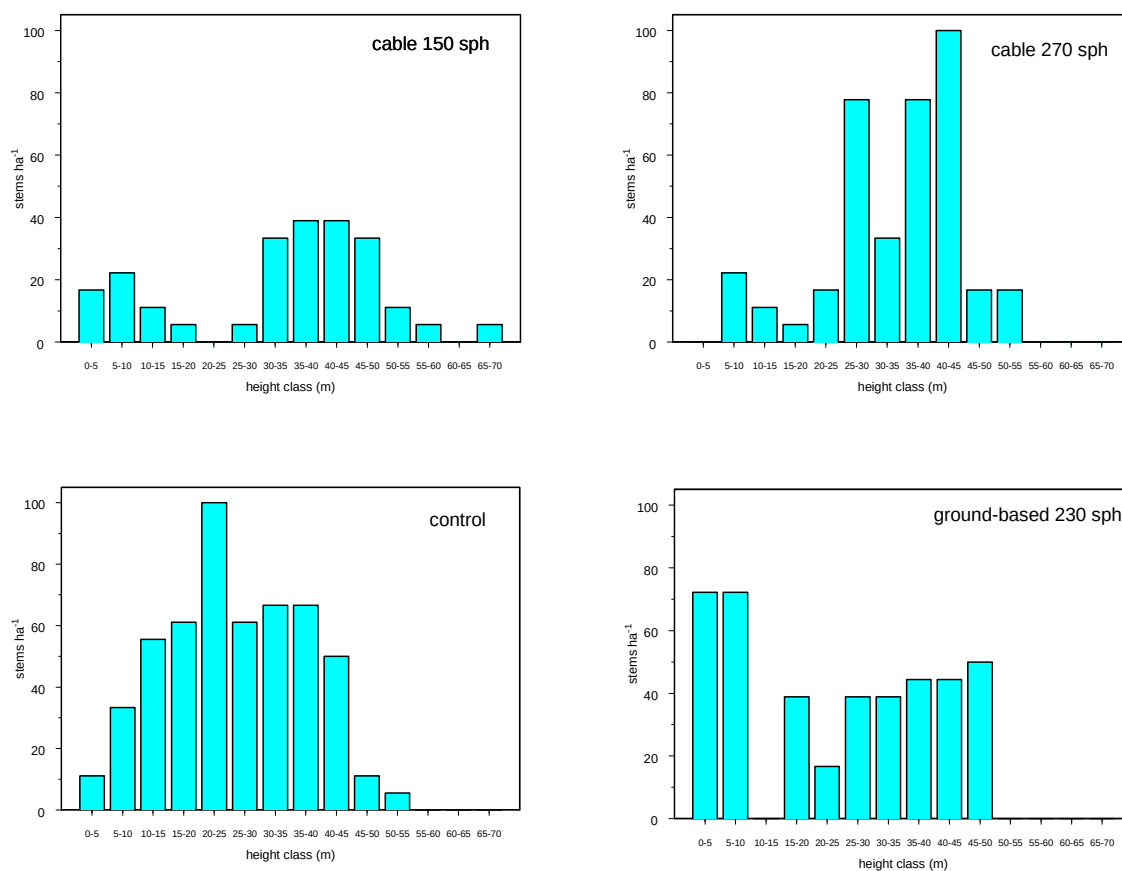


Figure 2.12 Tree height class distributions 14 years after cable or ground-based thinning at Repulse 3.

Crown dominance classes prior to thinning were differentiated into five of the six types, with no in-growth recorded (Figure 2.13). Most stems were considered co-dominant, then sub-dominant. Fourteen years following thinning, the distribution of classes differed markedly, with the cable thinned treatments having the greatest proportion of their stems in the dominant class, and the ground-based treatment having its greatest proportion in the in-growth class (Figure 2.14). For the control, most stems were classified as dead. The most intensive thinning treatment, cable 150 stems ha^{-1} , had the greatest proportion of stems in the dominant crown class.

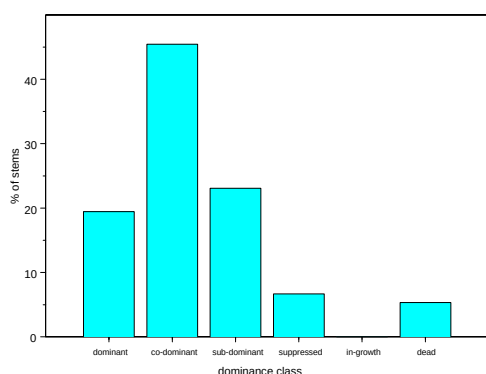


Figure 2.13 Initial crown dominance class differentiation for *Eucalyptus regnans* stems prior to thinning at Repulse 3. The data is not segregated by treatment as the initial assessment was done prior to the treatment areas being established.

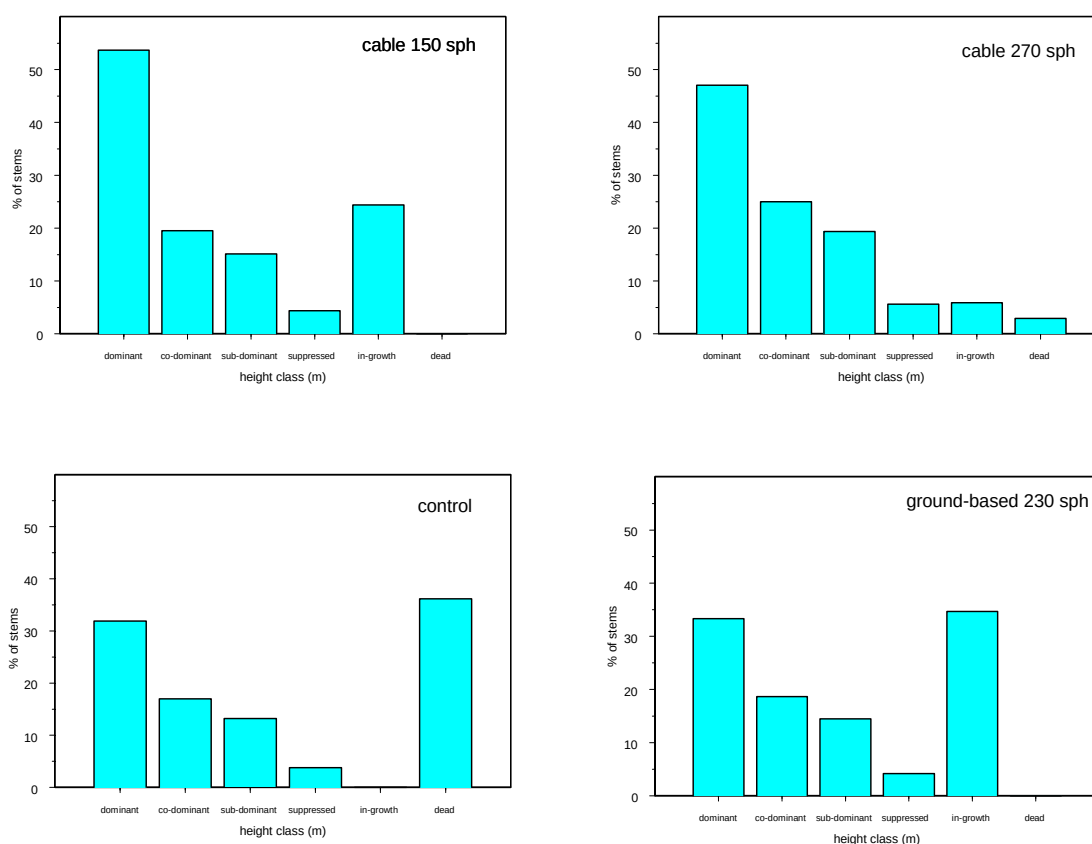


Figure 2.14 Crown dominance class differentiation 14 years after thinning for all *Eucalyptus regnans* stems at Repulse 3.

Thinning led to a sharp reduction in the density of woody understorey tall shrubs, after fourteen years this component of the vegetation was almost absent in the most intense thinning treatment (Table 2.3). The ground-based treatment exhibited the greatest density of residual or retained woody understorey species. Trends in individual species were highly variable.

Table 2.3 Density per ha⁻¹ of woody understorey species with stems >3 cm dbh by treatment 14 years after cable or ground-based thinning at Repulse3.

	Pre-treatment	Treatment			
		cable 150 sph	cable 273 sph	g-based 230 sph	control
<i>Acacia dealbata</i>	155	6	6	22	132
<i>Bedfordia salicina</i>	16	0	0	0	44
<i>Cassinia aculeata</i>	0	0	0	6	0
<i>Coprosma quadrifida</i>	25	0	0	6	0
<i>Nematolepis squamea</i>	36	0	0	0	99
<i>Olearia argophylla</i>	16	0	0	198	0
<i>Pomaderris apetala</i>	97	0	836	363	1601
<i>Zieria arborescens</i>	6	6	0	0	0



Figure 2.15 Ground-based soil disturbance at Repulse 3 in 1992 (left) and 2005 (right).

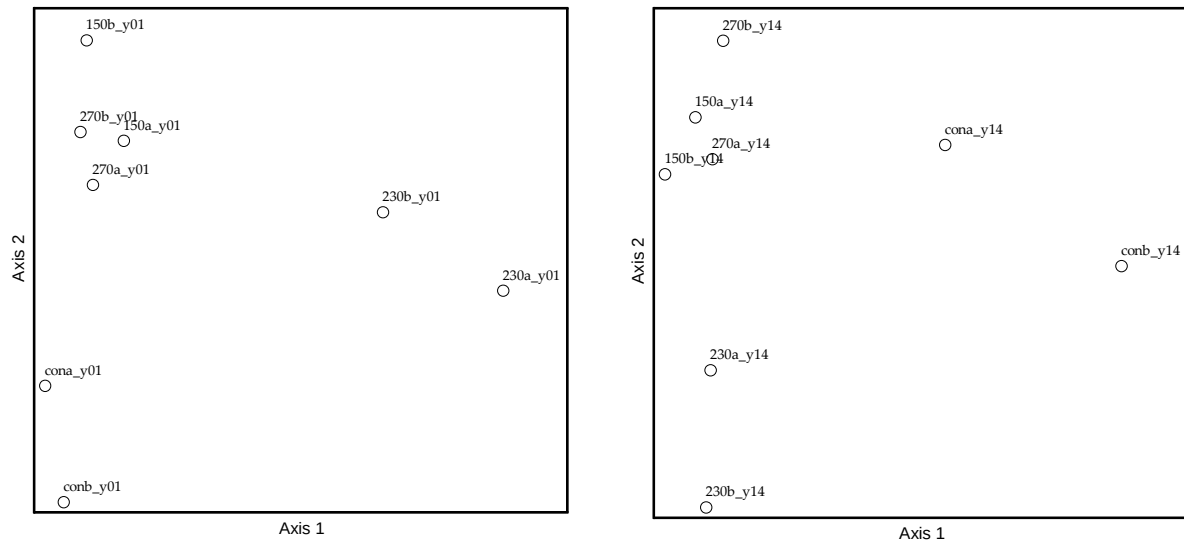


Figure 2.16 Principal components analysis ordination of floristic data for thinning treatments one (left) and fourteen years (right) following thinning at Repulse 3.

One year following cable or ground-based thinning, the distribution of plots in the ordination space closely resembles their respective treatment, with a cluster of cable logged plots clearly distinguishable from the controls and the ground-based thinned plots (Figure 2.16). The thinning method is distinguished by axis 1, whereas axis 2 distinguishes the controls from the thinned plots. Ground-based thinning results in a proportionally greater degree of change to the vegetation composition and biomass than the cable thinning treatment. Together axis 1 and 2 explain 50.7% of the variance in the multivariate space (Table 2.4). Fourteen years after thinning, a broadly similar pattern of dissimilarity is apparent, with the cable thinned plots remaining distinct from the ground-based thinned plots and the controls. Together axis 1 and 2 explain 44.8% of the variance in the multivariate space (Table 2.5).

Table 2.4 Axis eigenvalue scores and proportion of variance explained for principal components analysis ordination of treatments one year following thinning.

Axis	Eigenvalue	% of variance	Cumulative % of variance	Broken-stick Eigenvalue
1	16.931	28.697	28.697	4.663
2	12.993	22.021	50.718	3.663

Table 2.5 Axis eigenvalue scores and proportion of variance explained for principal components analysis ordination of treatments fourteen years following thinning.

Axis	Eigenvalue	% of variance	Cumulative % of variance	Broken-stick Eigenvalue
1	12.552	23.683	23.683	4.557
2	11.211	21.153	44.836	3.557

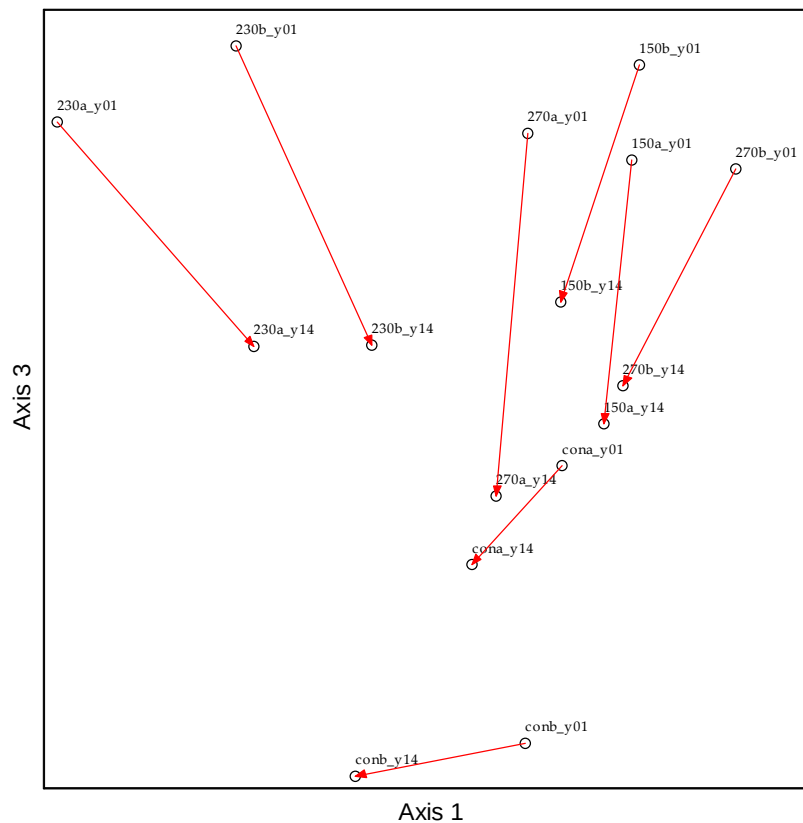


Figure 2.17 Three dimensional NMS ordination of thinning treatment plots at year 1 and year 14 at Repulse 3. Vectors link the same plots through time. Minimum stress for the 3-d solution was 8.4 (Table 2.6).

The pooled ordination of the plots at one and fourteen years after thinning indicates that there is a generalised trend in the direction of the successional vectors associated with axis 3 (Figure 2.17), and that the length of the vectors are greatest for both the 270 stems ha^{-1} and ground-based thinning treatments. The vector length is minimal for the controls, indicating a relatively modest change over the study period relative to the thinned plots. The successional vectors also suggest the direction of compositional change for the ground-based plots is in the direction of the cable thinned and control plots. The relative distinction between treatment groups is also apparent in the hierarchical classification dendrogram (Figure 2.18) in year 1, but this distinction is less clear in year 14 where groups are formed by a mixture of either the cable thinned plots or the ground-based thinned and control plots. Treatment effects, or overall floristic dissimilarity, is becoming less significant with time since thinning.

Table 2.6 Model fitting statistics for the NMS ordination in 1 to 3 dimensions and coefficients of determination for the correlations between ordination distances and distances in the original n-dimensional space. Increment and cumulative R-squared were adjusted for any lack of orthogonality of axes.

Axis	r-squared		Minimum stress in real data	r-squared and % orthogonality with	
	increment	cumulative		axis 1	axis2
1	0.143	0.143	26.6	-	0.245, 94%
2	0.422	0.565	13.9	-	-
3	0.348	0.912	8.4	-0.043, 99.8%	0.336, 88.7%

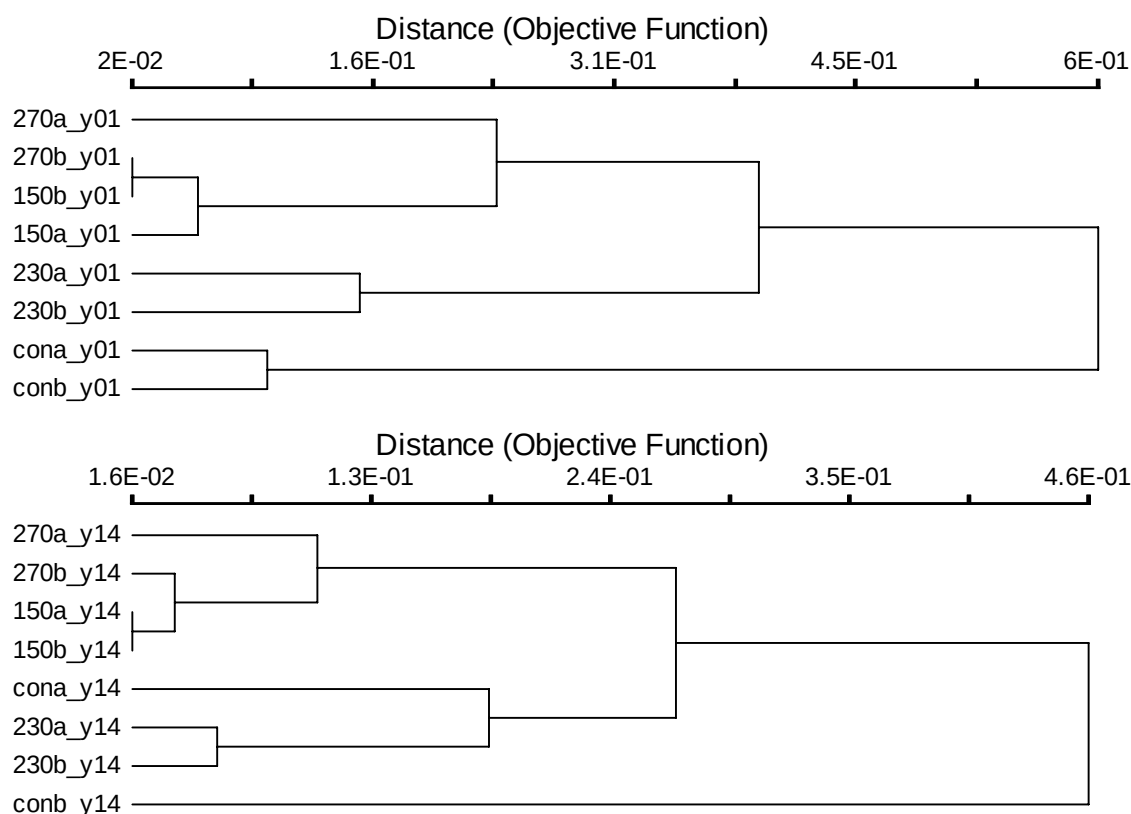


Figure 2.18 UPGMA cluster dendrogram of site floristics one year (top figure) and fourteen years (lower figure) after treatment at Repulse 3. The plots are identified by their treatment (eg 270 = 270 sph, con = control) and replicate number (a or b).

The strength of the treatment effect was examined using a ranked list of Bray-Curtis distance scores derived from the comparison of the differences between the year 1 and year 14 floristics (Table 2.7). The most dissimilar was the ground-based thinned vegetation, the most similar were the controls. The degree of floristic change over the fourteen year period was therefore most pronounced to vegetation subject to ground-based thinning.

Table 2.7 Ranked list of Bray-Curtis distance scores based on the comparison of the year 1 and year 14 species abundances per plot. Similarity is based on the relationship between shared abundance divided by total abundance and varies from 0 (identical) to 1 (no similarity).

Thinning Treatment	Distance score
ground-based 230 sph plot a year 1 versus year 14	0.5862
cable 270 sph plot a year 1 versus year 14	0.5116
ground-based 230 sph plot b year 1 versus year 14	0.4515
cable 270 sph plot b year 1 versus year 14	0.4348
cable 150 sph plot a year 1 versus year 14	0.4030
cable 150 sph plot b year 1 versus year 14	0.3807
control plot a year 1 versus year 14	0.3157
control plot b year 1 versus year 14	0.3135

The significance of the thinning treatment effect was greatest immediately following thinning (year 1), the treatment effect was reduced at year 14 (Table 2.8, ANOSIM test).

Table 2.8 Results of a test (ANOSIM) of within to between treatment group variation based on dissimilarity matrix of floristic cover scores. The value of *r* is indicative of differences among the treatment groups.

	<i>r</i> statistic	probability
Year 1 measurement	0.8333	p=2.697, ns
Year 14 measurement	0.6041	p=2.597, ns
Pooled data	0.8191	p=0.090, ns

The relationship between this treatment effect and species richness and diversity was less apparent, in some treatments such as the ground-based thinning, species richness fell over the measurement period, in others it was possible for one plot to increase and the other decrease (Table 2.9). The treatment response to species evenness was similarly variable, the ground-based plots were initially the highest at year 1, by year 14 however the cable 270 stems ha⁻¹ treatment exhibited the greatest evenness. Trends in species diversity and abundance were similarly variable, the ground-based thinned plots were initially the most diverse and also recorded the largest abundance score, at fourteen years however there was no clear trend with respect to these indices and the treatment applied (Table 2.10).

Table 2.9 Species richness and evenness per 0.09 ha plot by treatment and measurement year at Repulse 3.

	species richness		species evenness	
	year 1	year 14	year 1	year 14
cable 150 sph plot a	23	22	0.668	0.616
cable 150 sph plot b	26	24	0.698	0.669
ground-based 230 sph plot a	37	25	0.727	0.578
ground-based 230 sph plot b	34	28	0.743	0.629
cable 270 sph plot a	23	25	0.634	0.723
cable 270 sph plot b	24	28	0.725	0.569
control plot a	30	25	0.662	0.613
control plot b	22	23	0.546	0.667

Table 2.10 Species diversity and abundance index per 0.09 ha plot by treatment and measurement year at Repulse 3.

	Shannon`s diversity index		Simpson`s (D) abundance	
	year 1	year 14	year 1	year 14
cable 150 sph plot a	2.093	1.904	0.8501	0.7882
cable 150 sph plot b	2.275	2.127	0.8688	0.8528
ground-based 230 sph plot a	2.624	1.862	0.9049	0.7886
ground-based 230 sph plot b	2.621	2.094	0.9079	0.8259
cable 270 sph plot a	1.988	2.328	0.8036	0.8751
cable 270 sph plot b	2.306	1.896	0.8791	0.8051
control plot a	2.253	1.973	0.8444	0.8241
control plot b	1.689	2.090	0.7476	0.8253

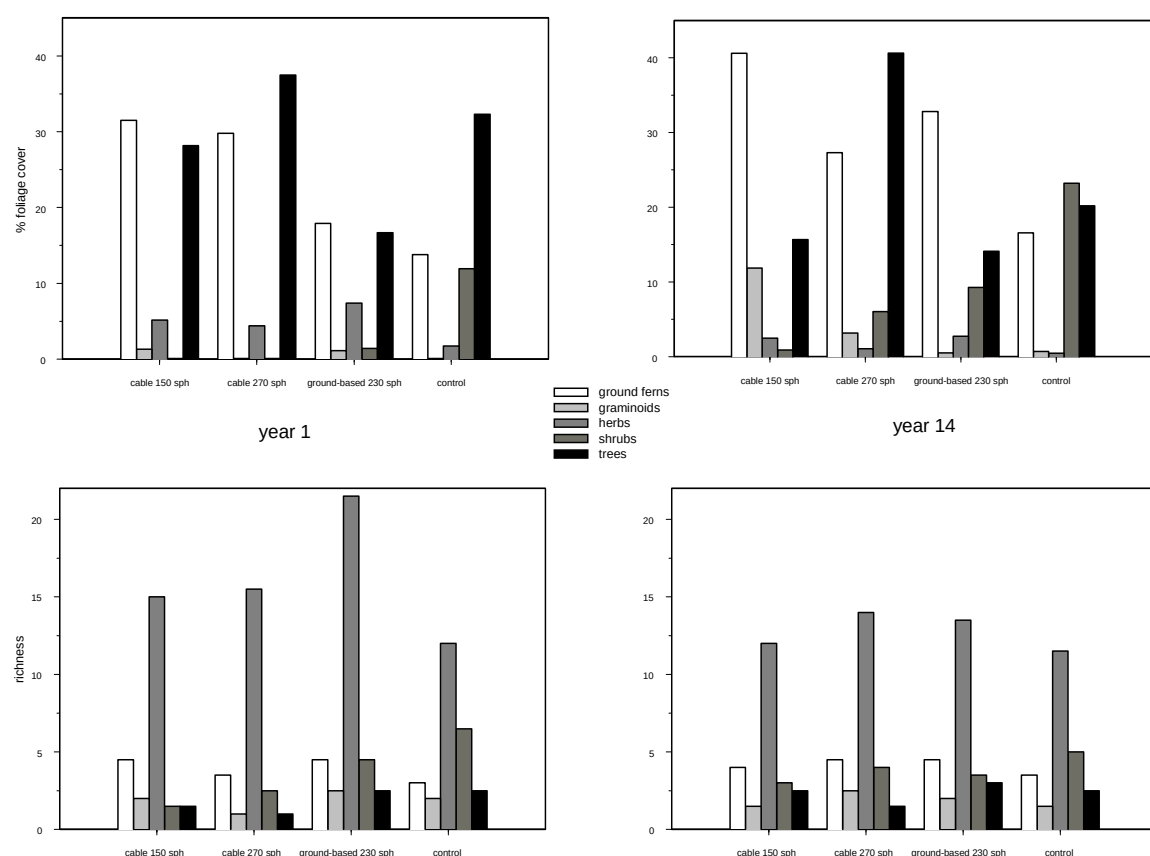


Figure 2.19 Species percent foliage cover and richness per 0.09 ha plot in five growth form groups one and fourteen years following thinning of 29 year old *Eucalyptus regnans* regrowth at Repulse 3.

The immediate response to thinning is apparent from the examination of growth forms by treatment and year. Ground-based thinning was associated with greater proportion of retained shrub cover (Figure 2.19) compared to the cable thinned areas (primarily of *Acacia dealbata*, *Pomaderris apetala* and *Olearia argophylla*, Table 2.12), especially in areas free of direct disturbance from the logging machine (a feller buncher). Ground-based thinning was also associated with a short term increase in the cover of herbaceous species relative to the cable thinned quadrats and the unthinned control. The increased cover and richness of herbaceous species following ground-based thinning was attributed predominantly to increases in *Acaena novae-zelandiae*, *Agrostis rudis*, *Echinopogon ovatus* and *Hydrocotyle hirta* (Table 2.12). Rhizomatous ground ferns including *Histiopteris incisa*, *Hypolepis rugosula* and *Polystichum proliferum* (Table 2.12) increased substantially following thinning, particularly in the cable thinned treatments. This increase in ground-fern abundance persisted to fourteen years post-thinning.

Table 2.11 Vegetation structural attributes at Repulse 3.

	150 sph year 1	150 sph year 14	270 sph year 1	270 sph year 14	230 sph year 1	230 sph year 14	control year 1	control year 14
Tree stratum >8 m. cover (%)	32.5	27.5	32.5	32.5	35	30	40	40
Shrub stratum 1-5 m. cover (%)	>1	12.5	>1	27.5	>1	20	67.5	40
Ground stratum 0-1 m cover (%)	60	70	40	70	25.0	80.0	20	22.5

Generalised trends in vegetation structure were also evident, particularly between year 1 and year 14 where much of the vertical stratification of vegetation evident prior to thinning began to re-establish (Table 2.11).

The degree of soil disturbance from gouging and from litter removal differed markedly between the cable and ground-based thinned areas, with 38% of the ground-based treatment area receiving ground disturbance from soil gouging, compared to <1% in the cable thinning treatments. The proportion of the treatment areas covered with thinning residue also varied with thinning method, with nearly twice the area in the ground-based treatment covered with residue compared to the cable thinning area.

Changes in species relative abundances by treatment and year are described in Table 2.12, with the species associated with the treatment column in bold. Herbaceous and weed species which were abundant immediately following ground-based thinning are almost absent by year 14. In the control plot several late successional colonist species (*Grammitis billardieri*, *Hymenophyllum australe*) are present at year 14 but absent from other treatments. In the most intense thinning treatment (cable, 150 stems ha⁻¹), *Polystichum proliferum* and *Dicksonia antarctica* are most abundant in year 14, along with other species spreading clonally such as *Dianella tasmanica* and *Gahnia grandis*.



Figure 2.20 Typical open fern dominated understorey in the cable thinning 150 sph treatment, Repulse 3, 2005

Table 2.12 Relative abundance of plant species for each combination of thinning treatment and year (1, 14) at Repulse 3. Each species appears in one row and is grouped (bold type) according to the treatment in which it reached its maximum relative abundance score (in the whole data set), the columns are the treatment and time category.

	150 sph year 1	150 sph year 14	270 sph year 1	270 sph year 14	230 sph year 1	230 sph year 14	control year 1	control year 14
<i>Cyathodes glauca</i>	100							
<i>Microsorium diversifolium</i>	100							
<i>Urtica incisa</i>	87	7	3		3			
<i>Uncinia tenella</i>	60		35				5	
<i>Hypolepis rugosula</i>	49	5	14	5	14		14	
<i>Histiopteris incisa</i>	35	18	18	10	6	6	5	1
<i>Deyeuxia contracta</i>	33				33		33	
<i>Atherosperma moschatum</i>	25	25			25	25		
<i>Dianella tasmanica</i>	4	96						
<i>Gahnia grandis</i>	9	54		31		4		
<i>Acianthus exsertus</i>		50				50		
<i>Dicksonia antarctica</i>	3	48	2	34	2	2		10
<i>Viola hederacea</i> ssp. <i>hederacea</i>	19	37	1	19	1	1	19	1
<i>Gonocarpus teucrioides</i>		33				33		33
<i>Polystichum proliferum</i>	16	23	9	16	9	16	2	9
<i>Ehrhartia stipoides</i>			88				13	
<i>Agrostis avenacea</i>	13		87					
<i>Senecio biseratus</i>	26		69				4	
<i>Australina pusilla</i> ssp. <i>muelleri</i>	43		43		6		6	
<i>Galium australe</i>	1	1	37	1	1	19	19	19
<i>Drymophila cyanocarpa</i>			33	33	33			
<i>Oxalis corniculata</i> ssp. <i>corniculata</i>	10	20	20	20	10	1	10	10
<i>Asperula gunnii</i>				100				
<i>Cassinia aculeata</i>		50		50				
<i>Deyeuxia frigida</i>				50		50		
<i>Juncus pallida</i>				50		50		
<i>Sonchus oleraceus</i> *				50			50	
<i>Coprosma quadrifida</i>		8		29		29	4	29
<i>Juncus pauciflorus</i>	25			25	25		25	
<i>Agrostis venusta</i>					100			
<i>Gnaphalium collinum</i>					100			
<i>Juncus bufonis</i>					100			
<i>Oxylobium arborescens</i>					100			
<i>Pimelea cinerea</i>					100			
<i>Senecio minimus</i>					100			
<i>Agrostis rudis</i>					97		3	
<i>Cirsium arvense</i> *			4		94		2	
<i>Hypericum gramineum</i>	4			4	89			4
<i>Agrostis</i> spp		3		3	86			7
<i>Hydrocotyle sibthorpioides</i>	13		1		86			
<i>Poa tenera</i>		5		5	78	5		5
<i>Mazus pumilio</i>				33	67			
<i>Hydrocotyle hirta</i>	19	5	5	3	57	5	3	3
<i>Echinopogon ovatus</i>	32		9		54		5	

* introduced species

Table 2.12 (cont.) Relative abundance of plant species for each combination of thinning treatment and year (1, 14) at Repulse 3. Each species appears in one row and is grouped (bold type) according to the treatment in which it reached its maximum relative abundance score (in the whole data set), the columns are the treatment and time category.

	150 sph	150 sph	270 sph	270 sph	230 sph	230 sph	control	control
	year 1	year 14	year 1	year 14	year 1	year 14	year 1	year 14
<i>Chiloglottis cornuta</i>						100		
<i>Deyeuxia quadriseta</i>						100		
<i>Luzula meridionalis</i>						100		
<i>Blechnum nudum</i>	4			4	4	89		
<i>Corybas diemenicus</i>		33				67		
<i>Olearia argophylla</i>				11	11	67	11	
<i>Clematis aristata</i>	2	1	2	1	2	58	30	2
<i>Acaena novae-zelandiae</i>	16	2	5	27	48	2	16	2
<i>Senecio linearifolius</i>		24		25	2	48		1
<i>Pteridium esculentum</i>	12	8	16	8	12	26	8	8
<i>Hydrocotyle pterocarpa</i>							100	
<i>Juncus australis</i>							100	
<i>Senecio</i> spp.							100	
<i>Acacia dealbata</i>					9		57	33
<i>Hypochoeris radicata</i>					50		50	
<i>Pimelea axiflora</i>					50		50	
<i>Eucalyptus regnans</i>	11	11	11	15	11	11	19	11
<i>Asplenium flabellifolium</i>								100
<i>Grammitis billardieri</i>								100
<i>Hymenophyllum australe</i>								100
<i>Juncus</i> spp.								100
<i>Carex appressa</i>				4			4	93
<i>Pittosporum bicolor</i>						4	4	93
<i>Zieria arborescens</i>		4					7	89
<i>Bedfordia salicina</i>							13	87
<i>Nematolepis squamea</i>							29	71
<i>Pimelea drupacea</i>	1	1	1	8	8	16	8	57
<i>Geranium potentilloides</i>					50			50
<i>Pomaderris apetala</i>				5	1	9	39	45

3. Case Study 2. Thinning in an un-even aged mixed species forest stand in East Gippsland.

3.1 Objectives

The objectives of this case study were to examine the vegetation and structural response to thinning in a complex un-even aged 31 year old mixed species forest stand in East Gippsland, Victoria.

3.2 Methods

3.2.2 Study Area

The study is located on Patrol Track, West Bemm Forest Block, east of Orbost in East Gippsland Victoria (Latitude 148° 51' 12'', Longitude 37° 38' 56''). The silvertop (*Eucalyptus sieberi*) - white stringbark (*Eucalyptus globiodes*) lowland sclerophyll regrowth forest was of mixed age classes, with the overstorey considered to have originated following a 1952 wildfire (Flinn and Hamers 1991), although subsequent fires from fuel reduction activities may also have burnt the area. While there was evidence of selective harvesting throughout the area, no discrete harvesting events prior to 1967 are recorded in management records. The majority of the regrowth overstorey subject to thinning is considered to be a result of the 1967 harvesting, following the seed tree silviculture method. Both the overwood basal area and volume of branch and old log material present prior to thinning in 1989 was considered relatively high at 234 t/ha⁻¹ (Buckley and Corkish 1991). Fire damaged stag trees were relatively common above the eucalypt regrowth, with a density of 48 stems/ha⁻¹ (Brown *et al.* 1991).

The environment at West Bemm is characteristic of the extensive areas of lowland forest on undulating terrain in East Gippsland (Table 3.1).

Table 3.1 Environmental data for the West Bemm study area

Mean annual rainfall (mm) ¹	Mean daily minimum of coldest month (July) ¹	Mean daily maximum of warmest month (February) ¹	Elevation, aspect and slope	Geology	Soils
1008	2.1	26.1	160 - 180 m, north westerly aspect, slope < 5 degrees	Tertiary sediments	silty grey clay loamy sand

¹ www.bom.gov.au from Cann River Forestry climate station

The regrowth forest at West Bemm prior to thinning consisted of a 20-25 m tall overstorey of *Eucalyptus baxteri*, *E. consideniana*, *E. globoidea* and *E. sieberi*, and the occasional *E. obliqua*. The tall shrub stratum was sparsely developed, consisting of scattered individuals of *Acacia terminalis* and *Banksia* species. The ground stratum was particularly well developed with a tangled mass of low shrubs and scrambling species. Conspicuous shrub and fern species included *Hibbertia empetrifolia*, *Gonocarpus teucriodes*, *Leucopogon lanceolatus*, *Platylobium formosum*, *Pteridium esculentum* and *Tetratheca pilosa*. Tussocks of *Gahnia sieberiana* were also common. Scrambling or climbing species, forming mats of vegetation linking these shrubs included *Billardiera scandens*, *Cassytha* spp. and *Tetrarrhena juncea*. Other common ground cover species included *Chionochloa pallida*, *Dianella caerulea* and *Lomandra longifolia*. Detailed regional floristic descriptions of lowland sclerophyll forest are provided by Mueck (1990) and the relationship of the flora and fauna of

regrowth and mature lowland sclerophyll forest in East Gippsland is discussed by Mueck and Peacock (1992), Loyn (1993) and Alexander *et al.* (2002).

3.2.3 Experimental Design

The study area at West Bemm forms part of a suite of study sites in East Gippsland established in the late 1980s to examine the feasibility of commercial thinning operations aimed at enhancing future sawlog production (Flinn and Hamers 1991). The West Bemm study area consisted of approximately 10 ha of relatively uniform regrowth forest in which a systematic grid was established using survey lines 100 m apart. Fourteen permanently marked vegetation quadrats were established prior to thinning, each of 0.09 ha on this survey grid. Due to logistical constraints only 4.1 ha of the study area was thinned (3 of the 14 plots), and no control area was established. This limited the experiment to only three replicate plots.

Conventional ground-based thinning was implemented between June and August 1989 using a Bell feller buncher, rubber tired ground skidder, and grapple excavator loader (Flinn and Hamers 1991). The thinning treatment aimed to reduce the *Eucalyptus* basal area to 15-18m²/ha⁻¹ and the stem density to 300 stems/ha⁻¹, provided no more than 60% of the original basal area (due to windthrow susceptibility) is removed (Flinn and Hamers 1991). Where the overwood basal area of mature trees was high, they were targeted for removal. Despite several attempted spot ignitions, no effective post-thinning fuel reduction burning took place at this study area. The study area had no recorded fires between 1988 and 2005.

3.2.4 Vegetation and Tree Sampling

For each 30 x 30 m vegetation quadrat all vascular plant species growing within or projecting into or above the quadrat were recorded and assigned a visually assessed cover-abundance value modified from the Braun-Blanquet (Braun-Blanquet 1932, Poore 1955) scale. Cover was defined as foliage projective cover following Walker and Hopkins (1981). The height ranges, three dominant species and foliage cover of each recognisable vegetation stratum was recorded, in addition to the percentage cover of the ground layer covered by rocks, logs, branches, litter and bare ground.

Pre-treatment and immediately post-treatment tree inventory data was sourced from the Department of Sustainability and Environment Victoria and various internal reports. While every attempt was made to match the location of inventory plots with the vegetation quadrats, the potential for intra-site variation is recognised. In 2005 tree diameter at 1.3 m, total height (m), canopy dominance class (dominant, sub-dominant, co-dominant, suppressed, in-growth and dead), and species was assessed for all overstorey tree and tall shrub species in the 0.09 ha vegetation plot. Overwood trees were those trees whose height, diameter and crown characteristics suggested had originated prior to the current regrowth cohort. Tree heights were recorded with a Vertex III laser height measuring instrument.

Fourteen vegetation quadrats were established in March 1988 (pre-treatment), and the three thinned plots subsequently re-measured in October 1989 (year 1), September 1999 (year 10) and March 2005 (year 17). The un-thinned plots were not re-measured. Floristic vegetation data collected pre-thinning and one-year post-thinning are from Mooress (1988) and Griffiths and Muir (1990) respectively.

3.2.5 Data Analysis

Percentage cover scores derived from each 30 x 30 m quadrat were converted to their mid-point of the respective cover class. Species were grouped into the growth form categories of trees, shrubs, herbs, graminoids, grasses, scramblers and ground ferns and their percent cover scores averaged by growth form, treatment and year. Mean species foliage cover scores then represented the mean value per species in that growth form per treatment or year. Association matrices used the Bray-Curtis measure

(Bray and Curtis 1957). Structural attributes (eg stratum cover) were average across treatments and measurement years.

The multivariate relationship between treatments was further examined using ordination, based again on the previous association matrix. The ordination technique was non-metric multidimensional scaling (NMS). NMS ordinations were performed as per the description in section 2.2.4. Hierarchical clustering was performed also using the techniques in section 2.2.4.

Compositional differences between the groups defined by measurement date were examined using Multi-Response Permutation Procedures (MRPP) as described by McCune and Mefford (1999). MRPP is a non-parametric procedure for testing the hypothesis that no multivariate difference exists between two or more groups. The *a priori* groups were defined as the pre-treatment, year 1 and year 17 measurements. The default weighting option was chosen and the distance measure was Bray-Curtis. MRPP was implemented in version PC-ORD version 4.35.

Species fidelity with measurement years was assessed using indicator species analysis (Dufrene and Legendre 1997) as described in section 2.2.4. Species abundance used the foliage cover class midpoint percentages previously described. Species richness, diversity, evenness and dominance follow the methods in section 2.2.4.

Tree height distributions were analysed graphically by stem density in 5 cm classes. Mean dominant height is defined as the height of the tallest stratum (Horne *et al.* 1986) per plot. It was assessed for 50 trees/ha⁻¹.

3.3 Results

From an initial density of 1424 stems ha⁻¹ prior to thinning in 1988, stem density increased to 1729 stems ha⁻¹ in 2005 despite being reduced to approximately 300 stems/ha⁻¹ (>10 cm dbh) immediately post-thinning. The majority of stems at year 17 in 2005 were in the 10-15 m class and the most frequent crown dominance class was in-growth and sub-dominant (Figure 3.1). The volume of basal area in the study area classified as overwood was relatively high at 9.9 m² ha⁻¹ in 2005 (Table 3.2).

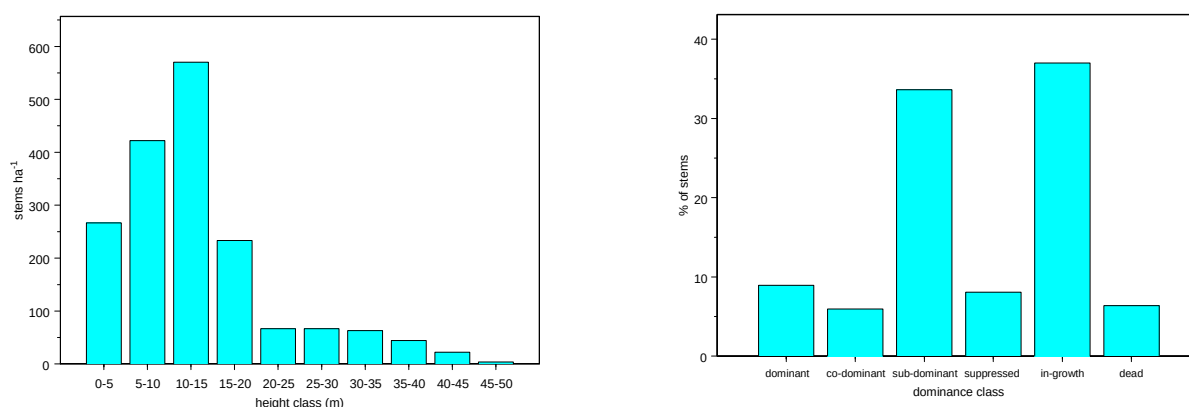


Figure 3.1 Tree height class distributions and crown dominance class differentiation for *Eucalyptus* species 17 years after thinning at West Bemm.

Table 3.2 Stand characteristics at West Bemm.

	pre-treatment	year 1	year 17
<i>Eucalyptus</i> stem >10 cm density (ha ⁻¹)	783	299	801
<i>Eucalyptus</i> basal area (m ² ha ⁻¹)	33.0	17.25	24.1
<i>Eucalyptus</i> overwood basal area (m ² ha ⁻¹)	10.0	3.29	9.9
<i>Eucalyptus</i> mean dominant height (m)	31.5	26.3	40.4

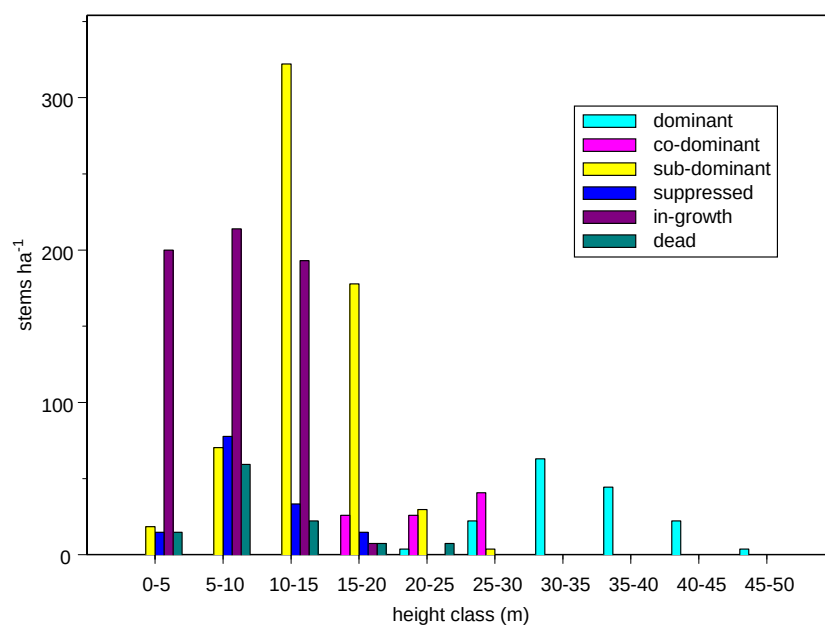


Figure 3.2 Crown dominance class differentiation 17 years after thinning for all *Eucalyptus* species at West Bemm.

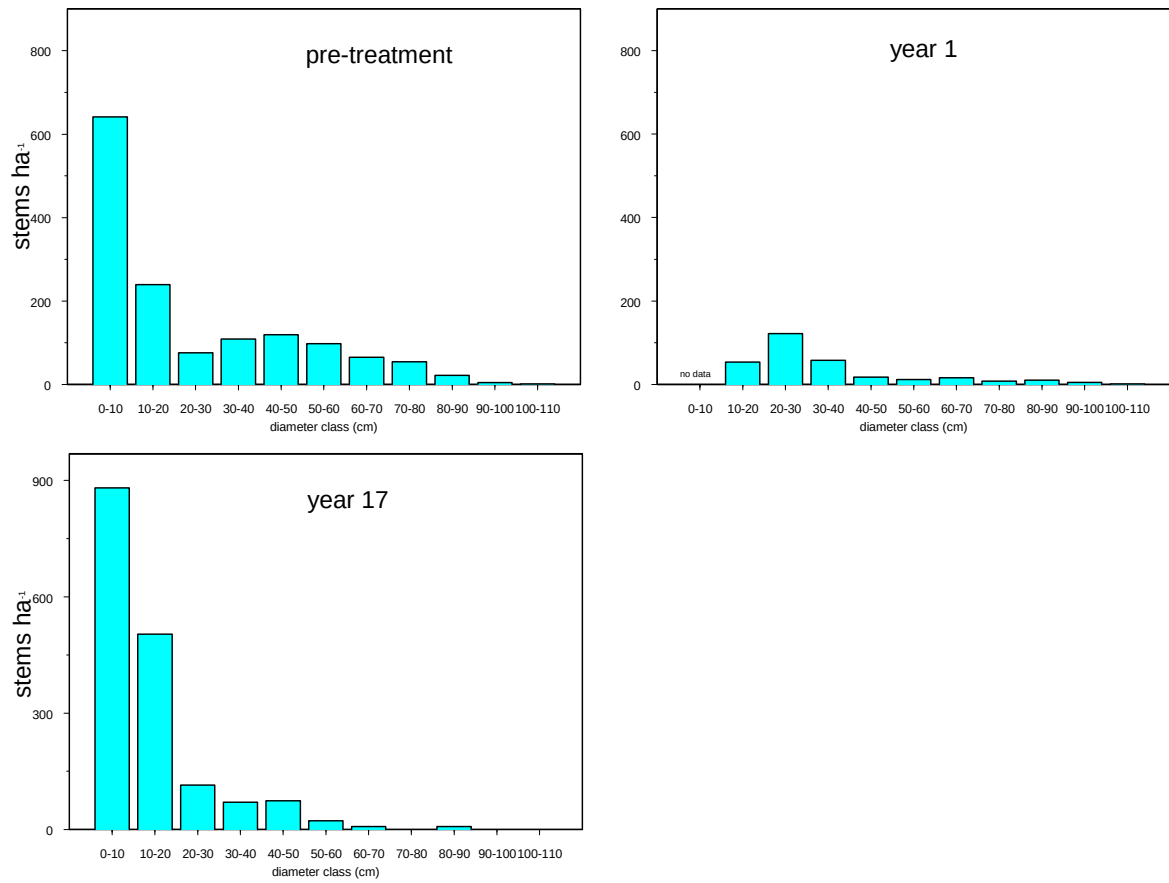


Figure 3.3 Diameter distributions at West Bemm prior to thinning and 1 to 17 years post-thinning.

The dominant class was restricted largely to the > 30 m height classes although they were relatively infrequent in terms of stems ha⁻¹ (Figure 3.2). Suppressed and dead stems were largely confined to the lower height classes < 15 m tall. The diameter distributions were highly skewed by the preponderance of stems < 10 cm dbh (Figure 3.3). While thinning flattened the overall diameter distributions, at year 17 the distribution was beginning to resemble the pre-treatment pattern, with the exception that many trees still had not moved into the larger diameter classes. The absence of a 0-10 cm class at year 1 was an artefact of the data collection method used.

The three plots in Figure 3.4 all exhibit a similar trend in the ordination space with time, from their initial position (eg p2_pre) close to axis 2 they all diverge significantly towards the lower end of the ordination space, then cluster towards the top left hand corner. Despite there only being three replicate plots measured pre-treatment, and at years 1, 10 and 17, they all appear to be exhibiting a broadly consistent response to thinning over the 17 year period.

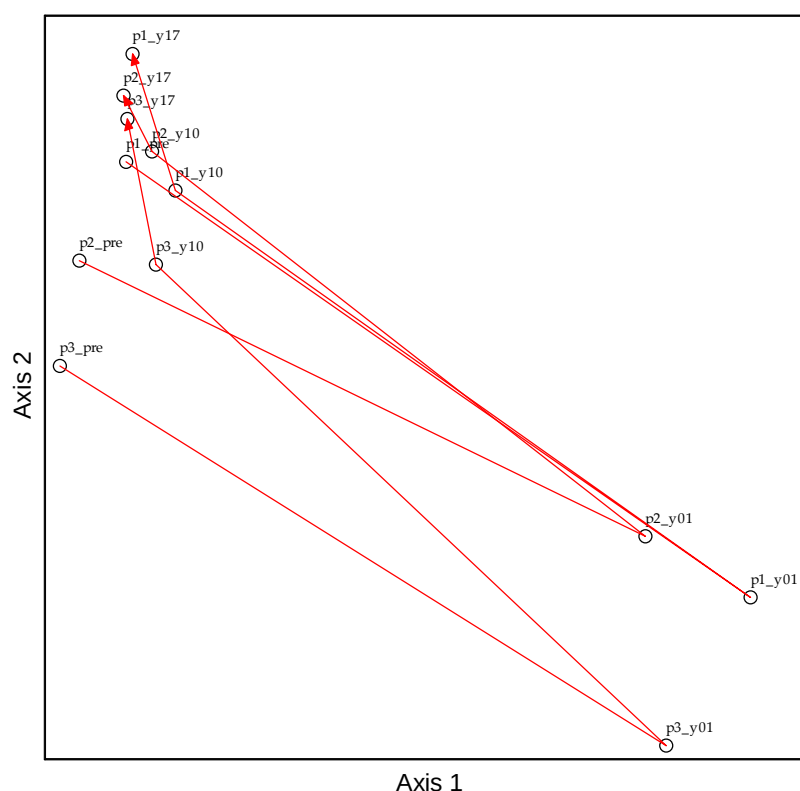


Figure 3.4 Three dimensional NMS ordination of thinning treatment plots measured pre-treatment and year 1 and year 17. Vectors link the same plots through time. Minimum stress for the 3-d solution was 1.2 (Table 3.3).

Table 3.3 Model fitting statistics for the NMS ordination in 1 to 3 dimensions and coefficients of determination for the correlations between ordination distances and distances in the original n-dimensional space. Increment and cumulative R-squared were adjusted for any lack of orthogonality of axes.

Axis	r-squared		Minimum stress in real data	r-squared and % orthogonality with	
	increment	cumulative		axis 1	axis2
1	0.835	0.834	20.1	-	-0.868, 24.6%
2	0.001	0.836	1.5	-	-
3	0.003	0.839	1.2	-0.961, 7.6%	0.896, 19.8%

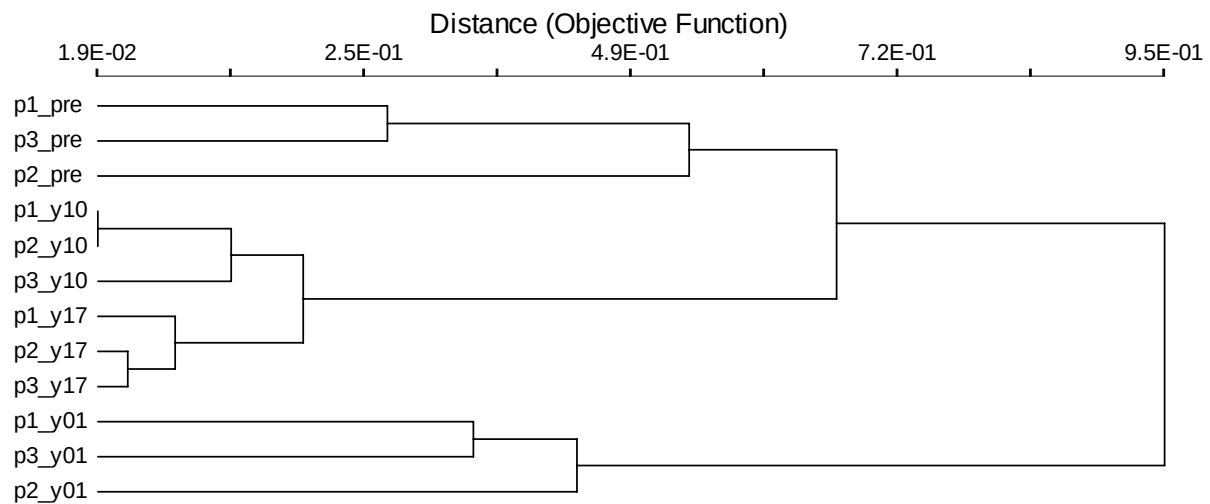


Figure 3.5 Cluster dendrogram of site floristics prior to thinning and one, ten and seventeen years after thinning at West Bemm. The plots are identified by their number and measurement year.

Cluster analysis indicated that the main dichotomy in the floristic data is between the year 1 measurement and the remaining observations, and that the year 10 and 17 measurements had greater similarity to the pre-treatment floristics than the immediate post-thinning floristics (Figure 3.5). The size of this age or treatment effect was tested directly using MRPP, and was found to be related in a linear manner to age since thinning. The average within-year weighted distance score was at its lowest value at year 17 indicating a relatively consistent level of homogeneity within the replicate plots (Figure 3.6).

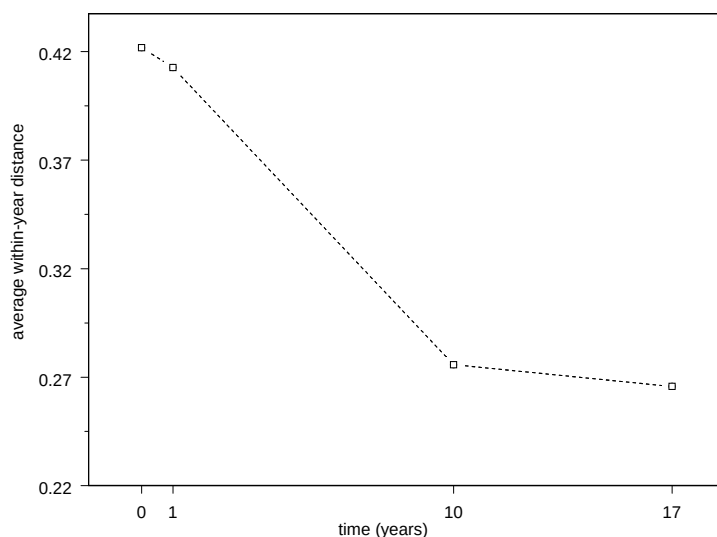


Figure 3.6 Weighted average within-year distance scores (from MRPP) for the three replicate plots per year following thinning at West Bemm. The probability of a smaller or equal delta was $p < 0.001$.

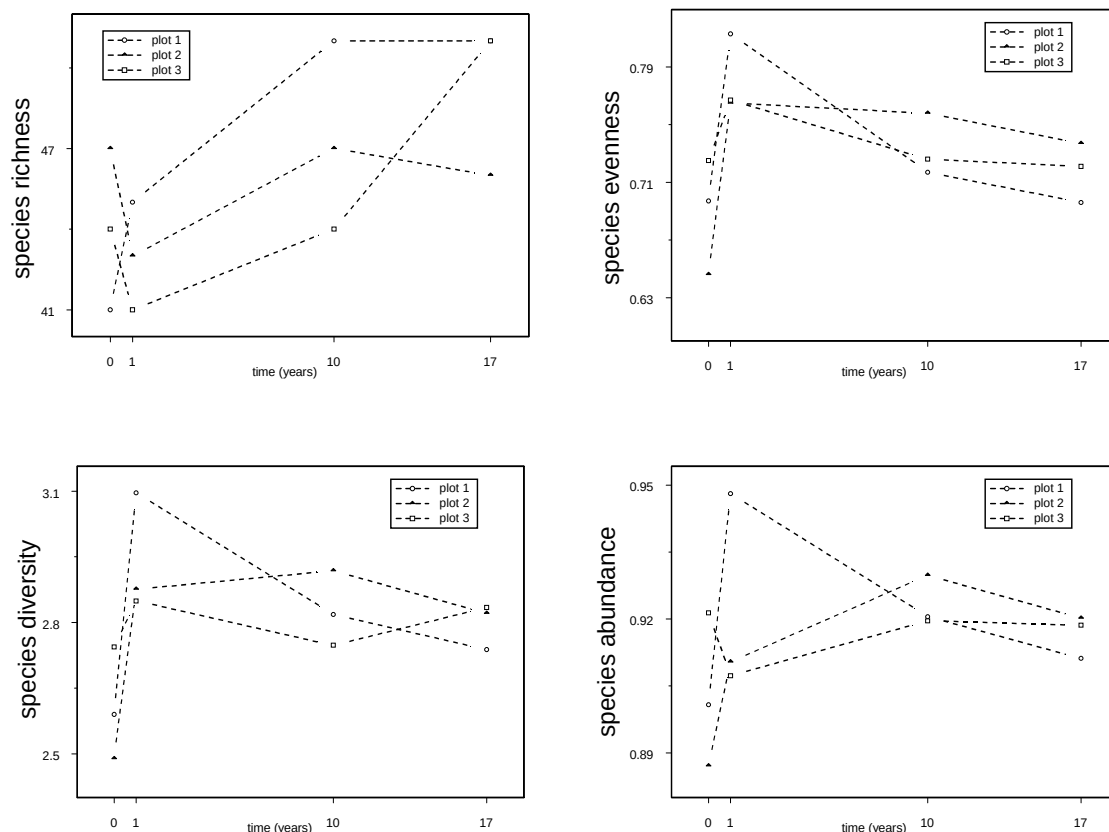


Table 3.7 Species richness, evenness, diversity and abundance per 0.09 ha plot by treatment and measurement year at West Bemm.

Species richness exhibits an overall increase with time since thinning, although in two of the three plots richness was reduced immediately post-thinning (Figure 3.7). Species evenness however increases immediately post-thinning, then exhibits a steady decline at year 10 and 17. In general species diversity and abundance respond similarly with a short term increase one year post-thinning and a decline or steady state until year 17.

Structural vegetation attributes exhibit a consistent increase in cover post-thinning, reflecting the gradual increase in biomass and vegetation complexity with time (Table 3.4). The cover of litter is increasing while the cover of bare ground is decreasing.

Table 3.4 Vegetation structural attributes at West Bemm.

	Pre-treatment	year 1	year 10	year 17
Tree stratum >8 m. cover (%)	73.3	10.0	36.6	41.7
Shrub stratum 1-5 m. cover (%)	20.0	8.3	30.0	10.0
Ground stratum 0-1 m cover (%)	56.7	5.0	68.3	73.3
Cover of bare ground (%)	21.7	13.3	6.6	2.6
Cover of litter (%)	78.3	30.0	41.6	50.0

Species richness and percent foliage cover within growth forms exhibit consistent patterns post-thinning, the relative proportion of species in growth form groups is stable with the exception of the increasing significance of shrubs post-thinning (Figure 3.8). The gradual recovery of foliage cover between growth form groups is a slower process, with most growth form categories increasing slowly compared to their pre-treatment values. Trends in species relative abundances with time since thinning are provided in Table 3.5.

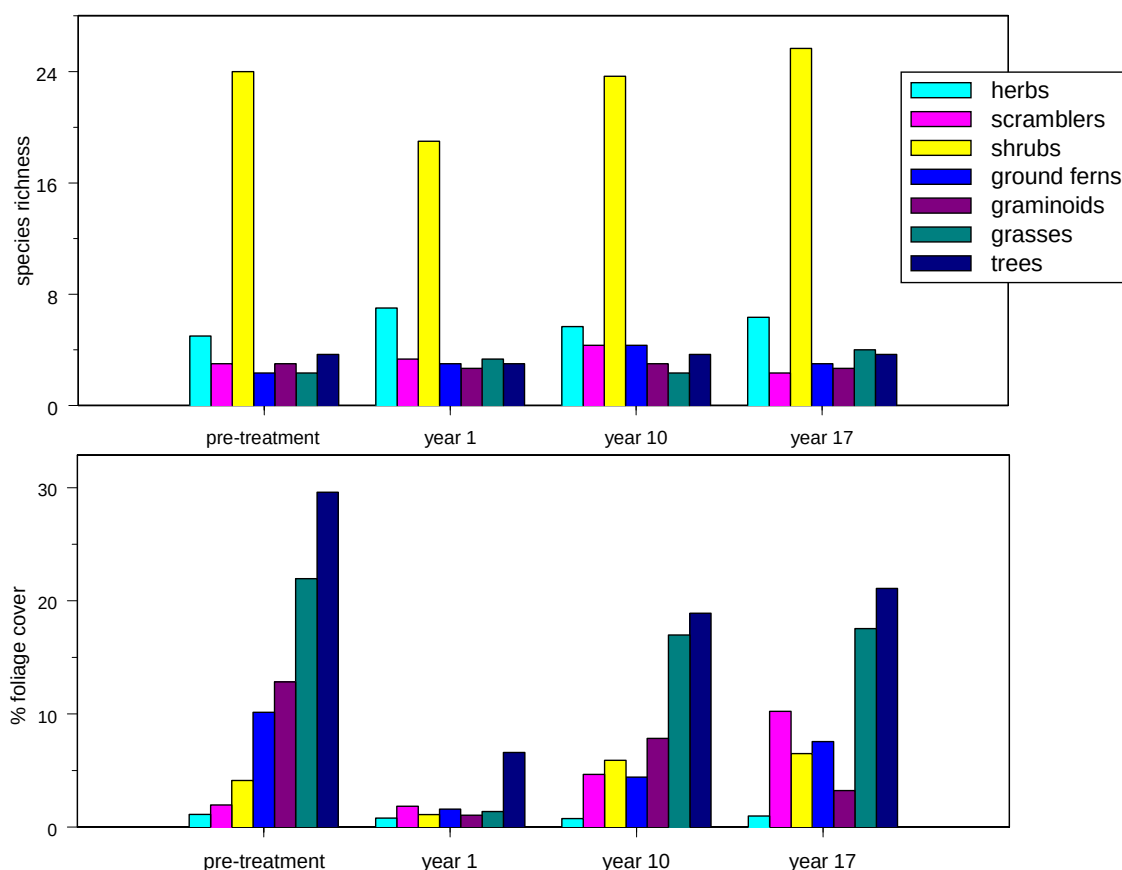


Figure 3.8 Species richness and mean species foliage cover in seven growth form groups per 0.09 ha plot prior to and following thinning at West Bemm.

Table 3.5 Relative abundance of plant species following thinning at West Bemm. Each species appears in one row and is grouped (bold type) according to the year in which it reached its maximum relative abundance score (in the whole data set).

	Pre-treatment	year 1	year 10	year 17
<i>Bedfordia arborescens</i>	100	0	0	0
<i>Hibbertia riparia</i>	100	0	0	0
<i>Hypericum gramineum</i>	100	0	0	0
<i>Persoonia confertiflora</i>	100	0	0	0
<i>Daviesia ulicifolia</i>	96	0	0	4
<i>Rhytidosporum procumbens</i>	90	0	10	0
<i>Acacia mucronata</i>	87	0	6	6
<i>Banksia spinulosa</i>	87	3	3	7
<i>Acacia sauveolens</i>	86	3	7	3
<i>Eucalyptus considaniana</i>	75	0	4	22
<i>Goodenia ovata</i>	73	13	13	1
<i>Lindsaea linearis</i>	71	25	2	2
<i>Leptospermum continentale</i>	66	11	22	1
<i>Joycea pallida</i>	64	4	0	32
<i>Melaleuca squarrosa</i>	60	8	28	4
<i>Eucalyptus globoidea</i>	58	4	13	25
<i>Gahnia sieberiana</i>	58	4	35	3
<i>Opercularia varia</i>	50	0	50	0
<i>Pultenaea retusa</i>	50	0	25	25
<i>Xanthosia dissecta</i>	50	0	0	50
<i>Lomandra longifolia</i>	45	5	5	45
<i>Billardiera scandens</i>	39	21	39	2
<i>Leucopogon lanceolatus</i>	37	1	37	25
<i>Eucalyptus baxteri</i>	36	10	30	24
<i>Pteridium esculentum</i>	34	5	28	34
<i>Viola hederacea</i>	33	33	12	22
<i>Eucalyptus sieberi</i>	31	9	30	30
<i>Lomatia ilicifolia</i>	29	14	29	29
<i>Banksia serrata</i>	0	100	0	0
<i>Blechnum cartilagineum</i>	0	100	0	0
<i>Diplarrena morea</i>	0	100	0	0
<i>Holcus lanatus*</i>	0	100	0	0
<i>Poranthera microphylla</i>	0	100	0	0
<i>Stackhousia monogyna</i>	0	100	0	0
<i>Cassinia aculeata</i>	0	93	7	0
<i>Drosera peltata</i>	0	75	25	0
<i>Schoenus opogon</i>	0	75	25	0
<i>Empodisma minus</i>	0	67	33	0
<i>Clematis aristata</i>	0	49	4	47
<i>Banksia marginata</i>	0	33	33	33
<i>Dianella caerulea</i>	18	27	27	27

* introduced species

Table 3.5 (cont.) Relative abundance of plant species following thinning at West Bemm. Each species appears in one row and is grouped (bold type) according to the year in which it reached its maximum relative abundance score (in the whole data set).

	Pre-treatment	year 1	year 10	year 17
<i>Comosperma volubile</i>	0	0	100	0
<i>Hypochaeris radicata</i> *	0	0	100	0
<i>Morinda jasminoides</i>	0	0	100	0
<i>Tylophora barbata</i>	0	0	100	0
<i>Opercularia aspera</i>	3	3	90	3
<i>Microlaena stipoides</i>	3	3	83	10
<i>Acacia genistifolia</i>	1	12	76	12
<i>Elaeocarpus reticulatus</i>	1	0	74	25
<i>Acacia myrtifolia</i>	13	13	73	0
<i>Baueria rubiodes</i>	0	0	72	28
<i>Cyathea australis</i>	0	0	67	33
<i>Gleichenia microphylla</i>	0	0	67	33
<i>Culcita dubia</i>	25	25	50	0
<i>Selaginella uliginosa</i>	0	0	50	50
<i>Xanthosia pilosa</i>	24	0	50	26
<i>Cassinia longifolia</i>	25	25	49	1
<i>Cassytha glabella</i>	5	5	45	45
<i>Hibbertia empetrifolia</i>	4	7	45	45
<i>Pultenaea scabra</i>	21	20	40	19
<i>Amperea xiphoclada</i>	1	21	39	39
<i>Acacia terminalis</i>	31	3	33	33
<i>Pimelea humilis</i>	0	0	0	100
<i>Poa sieberiana</i>	0	0	0	100
<i>Sticherus lobatus</i>	0	0	0	100
<i>Aotis ericoides</i>	0	0	4	96
<i>Ozothamnus cuneifolium</i>	0	4	0	96
<i>Comesperma ericinum</i>	7	0	0	93
<i>Eucalyptus obliqua</i>	7	0	0	93
<i>Lomandra filiformis</i>	4	2	2	93
<i>Patersonia glabrata</i>	0	7	0	93
<i>Hakea sericea</i>	7	0	3	90
<i>Xanthorrhoea minor</i>	5	5	2	88
<i>Lepidosperma laterale</i>	1	1	12	86
<i>Persoonia linearis</i>	6	3	6	84
<i>Lagenifera stipitata</i>	9	6	9	77
<i>Dampiera stricta</i>	11	12	11	66
<i>Tetratheca pilosa</i>	4	4	26	65
<i>Scaevola ramosissima</i>	20	20	0	60
<i>Gonocarpus teucrioides</i>	4	4	39	52
<i>Xanthosia tridentata</i>	45	2	4	49
<i>Eucalyptus botryoides</i>	24	0	28	48
<i>Tetrarrhena juncea</i>	16	2	33	48
<i>Epacris impressa</i>	25	25	3	47
<i>Cassytha phaeolasia</i>	20	20	20	40
<i>Platylobium formosum</i>	32	2	26	40

* introduced species



Figure 3.9 Thinned plot at West Bemm, East Gippsland 2005.



Figure 3.9 Un-thinned plot at West Bemm, East Gippsland 2005.

4. Case Study 3. Logging and grazing in an un-even aged mixed species forest stand in northern NSW

4.1 Objectives

The objective of this study in New South Wales was to examine the effect of seasonal cattle grazing in a spotted gum-grey box forest on the requirement for adequate overstorey regeneration and release of lignotuberous stems following logging.

4.2 Methods

4.2.1 Study Area

The study is located at Bom Bom and Divines State Forests, south of Grafton in northern NSW (Figure 4.1). Four plots were established in an area of similar forest types and management history, two on a broad ridge top (Bom Bom State Forest) and two on a lower slope (Divines State Forest). The ridge and lower slope plots are approximately three kilometres apart within the same contiguous section of State Forest. The geology and soils of the study area are typical of the Clarence Valley lowlands in New South Wales (Table 4.1).

Both Bom Bom and Divines State Forests have a long history of utilisation for both timber and rotational grazing, with early logging being so intensive that the forest was described as being effectively cleared in 1900. The forest was subsequently logged and silviculturally treated in 1911-1915, which led to the recruitment of a relatively even-aged cohort of pole and sawlog trees which were logged in 1954. Recovery rates from selective harvesting in this forest type following a coppice silvicultural system are generally low and in the vicinity of 2.0-3.0 m³/ha⁻¹, with the merchantable mean annual increment in the vicinity of 2.3 m³/ha⁻¹ (Combe *et al.* 1988). The 1954 logging was intensive with associated timber stand improvement including ring-barking of unwanted trees. Prolific flowering of *Corymbia henryi* in 1955 led to abundant seed crop in 1957. The current forest structure is assumed to have originated predominantly from lignotubers released by this logging event and a smaller contribution of new seedling recruits. The post-logging lignotuberous regrowth of *Corymbia henryi* present on these plots in 1957 was noted in management records as being lightly stocked.

Current silviculture in these forest types generally involves single tree selection systems and targeted gap creation, on a cutting cycle of approximately 15-40 years. Typical logs are of the tree species *Corymbia henryi* and in the 40-49 cm dbh under bark size class. Stand densities of trees > 10 cm dbh are in the vicinity of 300 stems/ha⁻¹. Harvesting residue is reduced in a broad area burn, while in areas subject to canopy gap treatments, the residue is heaped and burnt, and the seed bed is mechanically disturbed. The forests are predominately uneven-aged with a prolific component of regeneration (the lignotuber pool of Henry and Florence 1966), much of which is suppressed under the existing canopy. Management inventory in the surrounding forests indicates a large proportion (51%) of the canopy and sub-canopy trees exceeding basal diameter of 10 cm (dbhob) are unmerchantable, possibly as a result of long periods of growth suppression. Regeneration following harvesting largely relies upon the release of existing lignotuberous seedlings, advanced regrowth, coppice growth from cut stumps, and some additional seedling recruitment.

The available management records document a continuous history of cattle grazing from 1950 until the occupational permit was relinquished in 1997, although other historical sources indicate grazing has been continuous since European settlement in the 1840's. The approximate cattle stocking rate for the study area was 1 animal per 23 ha over an area of 2347 ha. Grazing without an occupational

permit can occur at any time. Fire, both managed and unplanned is a ubiquitous feature of this dry coastal forest environment, with wildfires recorded within the vicinity of the experiment in 1971, 1974, 1976, 1977, and 2001, and a hazard reduction burn in 1995. Fires are likely to have occurred at other times in the experiments history but are un-documented. The effects of these fires on the response variables are unknown however such regular fires are common feature of the local forest environment and fire protection of the experimental area may have been un-representative of the management conditions experienced or practically un-achievable. Current fuel management aims for broad area fuel reduction burning on a cycle of 2-5 years.

Rainfall data for the study area was obtained for the period 1872-1998 from the Grafton South Post Office composite rainfall record (Station no. 058025 and 058024), latitude 29° 42', longitude 152° 56' at an elevation of 6 m ASL. The long term average was calculated using a composite of the Grafton South Post Office and Grafton City Council climate stations. The climate in the study area is sub-tropical (Table 4.1), with rainfall peaking in summer (Figure 4.2). Compared to the long term average, rainfall during the study period was very similar to the long term pattern (Figure 4.3).

Table 4.1 Environmental data for the South Grafton study area. Climate data from the Grafton City Council climate station (www.bom.gov.au).

Mean annual rainfall (mm)	Mean daily minimum of coldest month (July)	Mean daily maximum of warmest month (January)	Elevation, aspect and slope	Geology	Soils
978	5.9	30.8	50 m, slope < 5 degrees, north westerly aspect	Clay rich sequence of lithic sandstone, siltstone and claystone	Clayey and well structured, with a greyish brown light clay topsoil over a stony porous lower topsoil

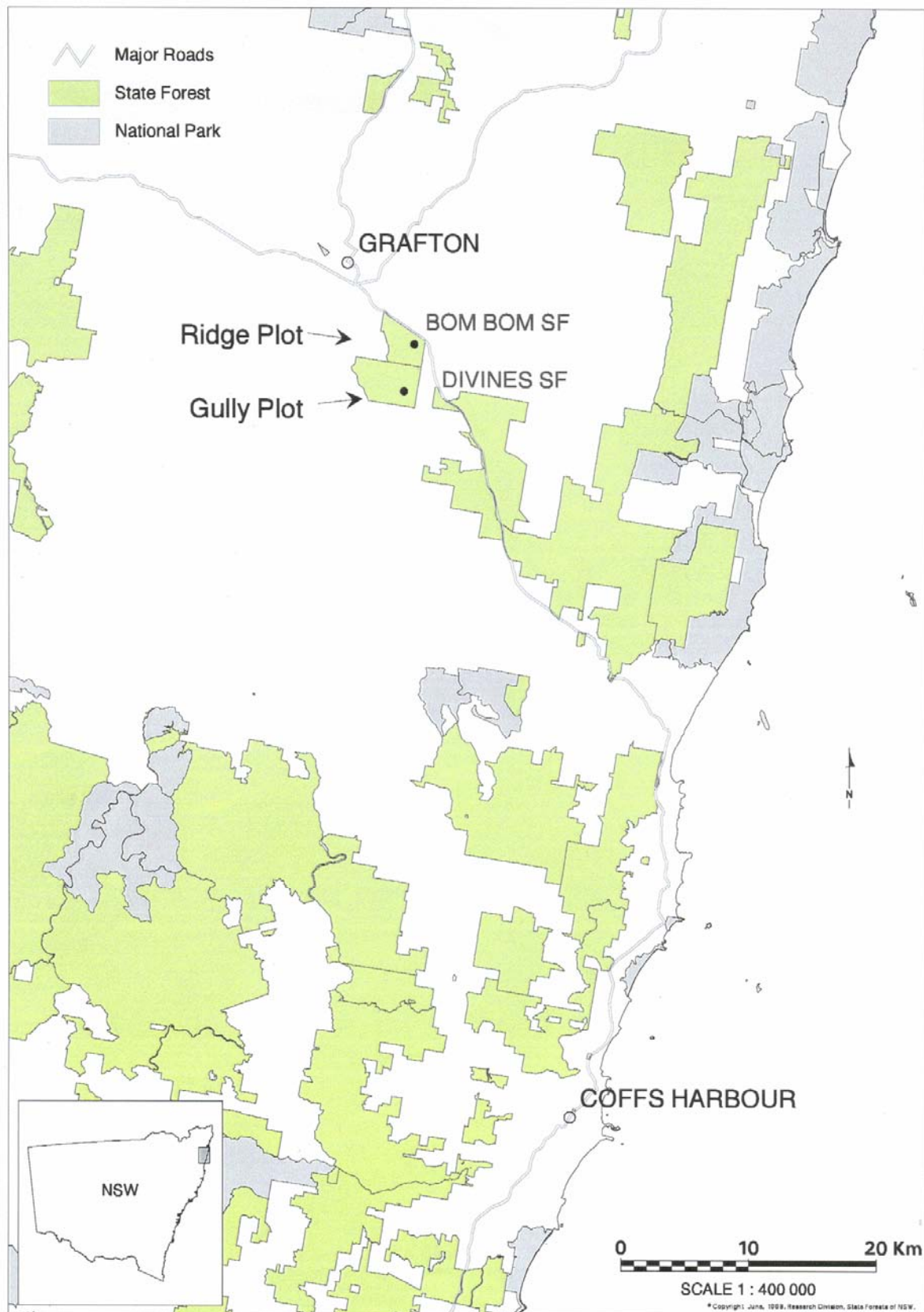


Figure 4.1 Location of Bom Bom and Divines State Forests, south of Grafton, northern NSW.

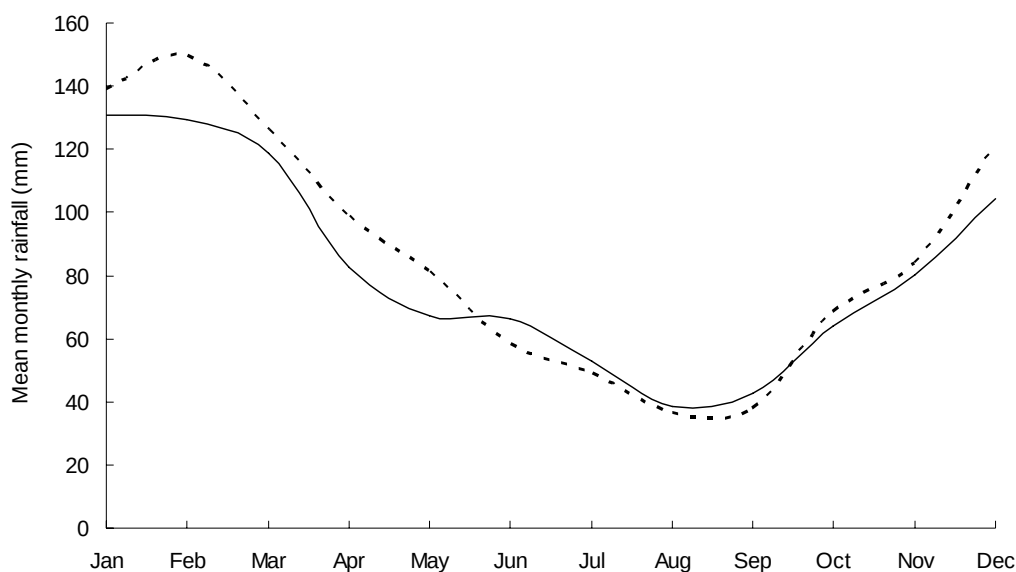


Figure 4.2 Relationship between the mean monthly rainfall (mm) for the study period 1954-1998 (continuous line) and the long term (1872-1998) average rainfall at Grafton, NSW (dotted line)

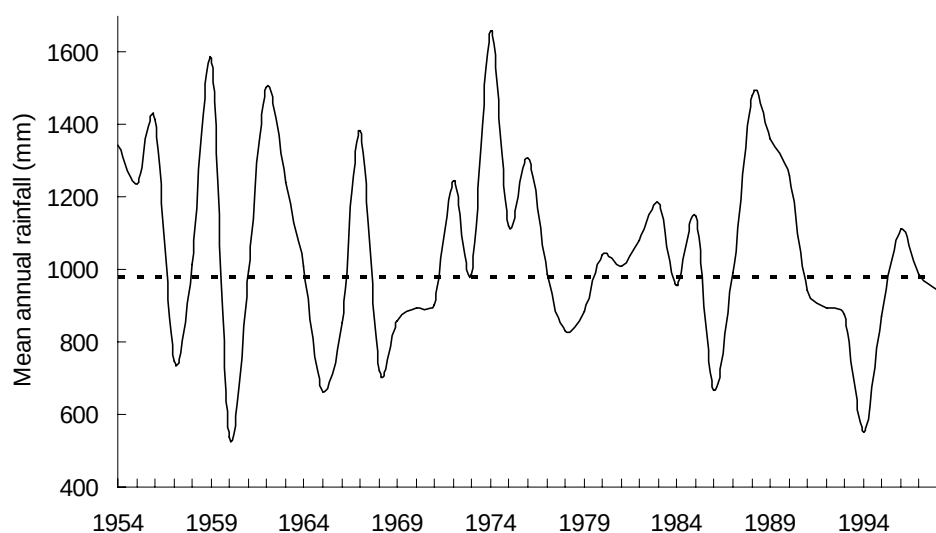


Figure 4.3 Relationship between mean annual rainfall (mm) for the study period 1954-1998 (continuous line) and the long term (1872-1998) average rainfall at Grafton, NSW (dotted line).

The ridge plots were rectangular in shape and 0.2 ha. in area and were located on a flat to very gently sloping ridge within Bom Bom State Forest compartment 78 (AMG E498650 N6708900, latitude $152^{\circ} 59' 10''$, longitude $29^{\circ} 45' 00''$, altitude 50 m ASL), south of Grafton, northern NSW. Plot size was 36x58m, placed within an area fenced with three stranded barb wire measuring 53x85m. The fencing was designed to be cattle proof but would not have excluded macropod grazing. The unfenced plot was placed adjacent to the fenced plot. The forest is Type 72a, Spotted Gum-Grey Box, with site height 30-40 m (Forestry Commission of NSW 1989). At the commencement of the experiment, the overstorey principally consisted of *Corymbia henryi* (S.T. Blake) K.D. Hill & L.A.S. Johnson (previously *Eucalyptus maculata* Hook.) and *Eucalyptus moluccana* (grey box). The understorey was dominated by native grasses including barbed-wire grass (*Cymbopogon refractus*), kangaroo grass (*Themeda australis*) and blady grass (*Imperata cylindrica*) and herbaceous perennials such as slender tick-trefoil (*Desmodium varians*) and Kidney Weed (*Dichondra repens*). A sparse low shrub layer of *Acacia* species, coffee bush (*Breynia oblongifolia*), and dogwood (*Jacksonia scoparia*) was also present.

The slope plots were rectangular in shape and 0.4 ha. in area and were located on a flat to very gently slope corresponding to the base of a sloping landform within Divines State Forest compartment 80 (AMG E497850 N6705200, latitude 152° 58' 40'', longitude 29° 47' 00'', altitude 60 m ASL), south of Grafton, northern NSW. Plot size was 50x80m, placed within an area fenced with three stranded barb wire measuring 53x85m. The unfenced plot was placed adjacent to the fenced plot. The forest is Type 74, Spotted Gum-Ironbark/Grey Gum, with site height 30-40 m (Forestry Commission of NSW 1989 and differs from the ridge plot with the additional occurrence of occasional *Eucalyptus tereticornis* and *E. tindilliae* and in its generally higher site quality. The understorey is dominated by native grasses (principally *Imperata cylindrica*) and a greater proportion of herbaceous perennials than in the ridge plots, with species including *Goodenia heterophylla* and *Glycine tabacina*. Apart from scattered low *Acacia* species, no shrub stratum was present. The sites logging and stand treatment is similar to the ridge plots, with the last logging event also being 1954.

4.2.2 Experimental Design

Plots were established in 1957 with the aim of assessing the growth of the mainly coppice-derived regeneration following timber harvesting and stand improvement in 1954. The plots were measured in 1959, 1961, 1963, 1965, 1967, 1969 and again in 1998. The treatments were maintained for 17 years (1957 to 1974). The 1974 measure was abandoned following a wildfire that year when the fencing was damaged. No attempt was made to repair the fencing and grazing occurred unhindered from 1974 to 1998. The 1998 re-measurement therefore occurred 24 years after the cessation of the experimental treatments and 44 years following disturbance by timber harvesting and stand improvement.

4.2.3 Vegetation and Tree Sampling

All measurements of tree regeneration recorded tree height and species, except for the 1998 measure which also recorded tree diameter at breast height over bark (dbhob). During 1957-1969, tree height was recorded in one foot height intervals. In 1998, tree height was measured with a Vertex III laser height measuring instrument. Diameter at breast height (dbhob) was measured in 1998 at 1.3 m above ground. For seedlings and coppice less than 1.3 m tall, stem diameter was measured with callipers, at the point of stem mid-height above ground. Data on the stand structure were not collected prior to timber harvesting in 1954. The 1967 re-measure of the lower-slope plots was missed.

4.2.4 Data Analysis

While the data are of a structured multivariate nature, consisting of grazing enclosure x site x species x height classes x time, limitations with the early measurements and a lack of replication precluded formal statistical analysis and hypothesis testing. Summary tables of stand height, density and basal area were compiled and 2-D and 3-D frequency histograms and age-profiles were created to assess the proportions of stems by treatment, plot and year. The lack of tree diameter measurements prior to 1998 precluded the use of basal area summaries for the experimental period.

The association between each proportional frequency histogram representing a combination of ridge or slope site, treatment and year was evaluated using two complementary approaches, the profile attributes measure (Faith *et al.* 1985), and the Bray and Curtis (1957) association measure. The purpose of applying the two techniques was to investigate the significance of the ordering of the plot attributes (stem counts in height categories) versus their un-ordered state where the progression of trees through the height classes is ignored in the measurement of plot associations. The profile attributes measure was implemented using the profile option in PATN (Belbin 2003b). Profile attributes is an association measure designed for the comparison of ordered profiles (Faith *et al.* 1985). The analysis was restricted to the data obtained between 1957 and 1965 where there were no gaps in the measurement program (ie 1967) or in the height categories (ie 1969). A profile is defined as ‘an attribute type where a single value is replaced by a set of values having some order dependency’; it is also referred to as a two dimensional variable (Belbin 2003a). The number of stems per hectare in ten height classes (0-0.3, 0.3-0.6, 0.6-0.9, 0.9-1.2, 1.2-1.5, 1.5-3.1, 3.1-4.6, 4.6-6.4, 6.1-9.1, 9.1-10.7 m) formed the profile. Examples such as these are referred to by Belbin (2003a) as a meristic ordinal profile. The integer constant applied was 1, the *p*-value for the neighbourhood shape value was 0.1, and the range standardisation for each class used actual values.

The results of the two association measures, profile attributes and Bray and Curtis, were examined graphically with a classification dendrogram and ordination. The hierarchical classification was performed using flexible pair group arithmetic averaging (UPGMA) with a beta parameter of -0.1 (Belbin and McDonald 1993). The ordination technique was semi-strong hybrid multidimensional scaling (Belbin 1991). The ordinations were derived from 50 multiple runs, using 10 different random starting configurations, with a maximum of 50 iterations. The stability criterion for terminating successive iterations was 0.005.

4.3 Results

In 1957, three years following timber harvesting, the ridge and lower-slope sites showed a broadly similar suite of regenerating overstorey species, with the exception of *Eucalyptus crebra* being absent from the slope plots and *E. tereticornis* and *E. tindaliae* being absent from the ridge site. Species common across both sites were *Corymbia henryi*, *E. fibrosa* and *Eucalyptus moluccana*. On the ridge plots, stem density did not differ between fenced and unfenced treatments. On the slope plots however, the grazed treatment had a higher stem density. The overall density of stems was higher on the lower-slope plots, reflecting initial inter-site differences. Despite the initial differences in the composition and density of tree species regenerating following timber harvesting (Table 4.2), in 1957 the proportions of lignotuberous regeneration in each height category were broadly consistent for the grazed and ungrazed treatments and for the ridge and lower-slope plots (Figure 4.4).

Table 4.2 Density of lignotuberous stems per ha⁻¹ at south Grafton in 1957 at the commencement of the fencing treatments, three years post-timber harvesting. Species nomenclature follows Harden (2002).

	Ridge site		Slope site	
	ungrazed	grazed	ungrazed	grazed
<i>Angophora subvelutina</i> F. Muell. (Broad-leaved Apple)			2	
<i>Corymbia henryi</i> (S.T. Blake) K.D. Hill & L.A.S. Johnson (Large-leaved Spotted Gum)	166	152	492	1282
<i>Eucalyptus crebra</i> F. Muell. (Narrow-leaved Ironbark)	170	133		
<i>Eucalyptus fibrosa</i> F. Muell. (Red Ironbark)	19	109	555	452
<i>Corymbia gummifera</i> (Red Bloodwood)				2
<i>Corymbia intermedia</i> ¹ (R.T. Baker) K.D. Hill & L.A.S. Johnson (Pink Bloodwood)				
<i>Eucalyptus moluccana</i> Roxb. (Grey Box)	251	203	240	188
<i>Eucalyptus siderophloia</i> ¹ Benth. (Grey Ironbark)				
<i>Eucalyptus tereticornis</i> Sm. (Forest Red Gum)			252	458
<i>Eucalyptus tindaliae</i> Blakely (Stringybark)			32	22
<i>Lophostemon sauevolens</i> (Sol. ex Gaertn. Peter G. Wilson & J.T. Waterh. (Swamp Turpentine)			2	
Sum	606	597	1577	2402

¹ Species recorded in later measurements (Table 4.4)

Despite the initial differences in the composition and density of tree species regenerating following timber harvesting (Table 4.2), the proportion of lignotuberous regeneration in each height category was broadly similar between the grazed and ungrazed treatments and the ridge and slope plots in 1957 (Figure 4.4). On the ridge plots, a slightly higher proportion of the lignotuberous regeneration was in the lowest height category, compared to the lower-slope plots. From 1959, these initial differences became increasingly pronounced, with the proportion of stems per plot in the smaller height classes being consistently greater in the ridge plots compared to the slope plots. In the slope plots however, a greater proportion of stems are progressing through the height classes, with a corresponding reduction in stems in the lower height categories. Treatments effects, assessed as the proportion of stems in each height class, are variable. In general, grazing results in a relative increase in stems in each height class on the ridge plots, and on the slope plots the reverse trend occurred. Total stem density across all height classes was increased by grazing on the ridge, and reduced by grazing on the slope.

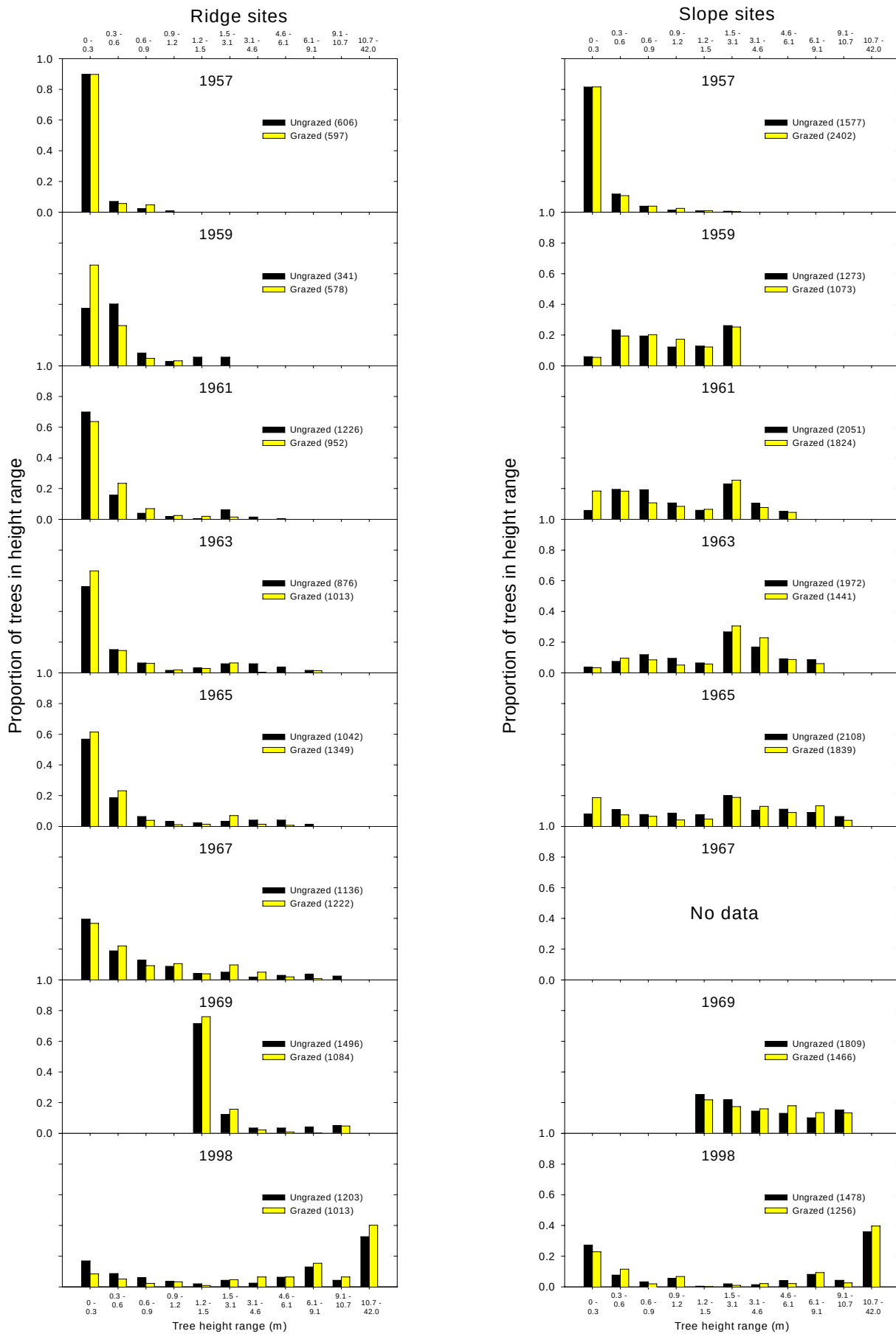


Figure 4.4 Histograms showing proportional frequency of all stems for each measurement and treatment at South Grafton, 1957-1998. The number of stems per hectare are given in parentheses.

In 1998, 24 years after the fencing exclosures were removed by fire, the proportional distribution of regenerating stems across the respective height categories is broadly similar for both the ridge and slope plots, despite the existence of clear inter-plot differences when the fencing exclosures were in place. The 3-D histograms provide a further indication of the treatment response in stem density and height (Figure 4.5 – 4.8). In the lower-slope plots, there is very little differentiation between the grazed and ungrazed treatment and stems moved through the size cohorts over the forty year period in a similar manner for both treatments (Figure 4.5 and Figure 4.6). The initially large proportion of stems in the lowest height category (0-0.3 m.) on the ridge plots persists with time in the plot subject to grazing (Figure 4.7). However, in the ungrazed plot more individuals advanced to the larger cohorts over time (Figure 4.8). The grazing effect is most evident in the first four years after the forest is regenerated and persists for approximately ten years. The grazing treatment effect is most pronounced in the smaller size classes, in the ridge plots where the proportion of stems exceeding 1.2 m height is reduced compared to ungrazed, for the first ten years following timber harvesting (Figure 4.9). The treatment effect was maximised two years following the erection of the grazing exclosures, when in the grazed understorey 65% of stems were less than 0.3 m tall and in the un-grazed treatment only 37% were less than 0.3 m tall (Table 4.3). By 1998, after forty-four years following disturbance, this trend is no longer apparent.

Table 4.3 The percentage of lignotuberous stems on ridge plots <0.3 m. tall that were effectively confined to the multi-stemmed low height growth increment stage of development.

	ungrazed	grazed
1957	89.8	89.7
1959	37.3	65.5
1961	69.9	63.7

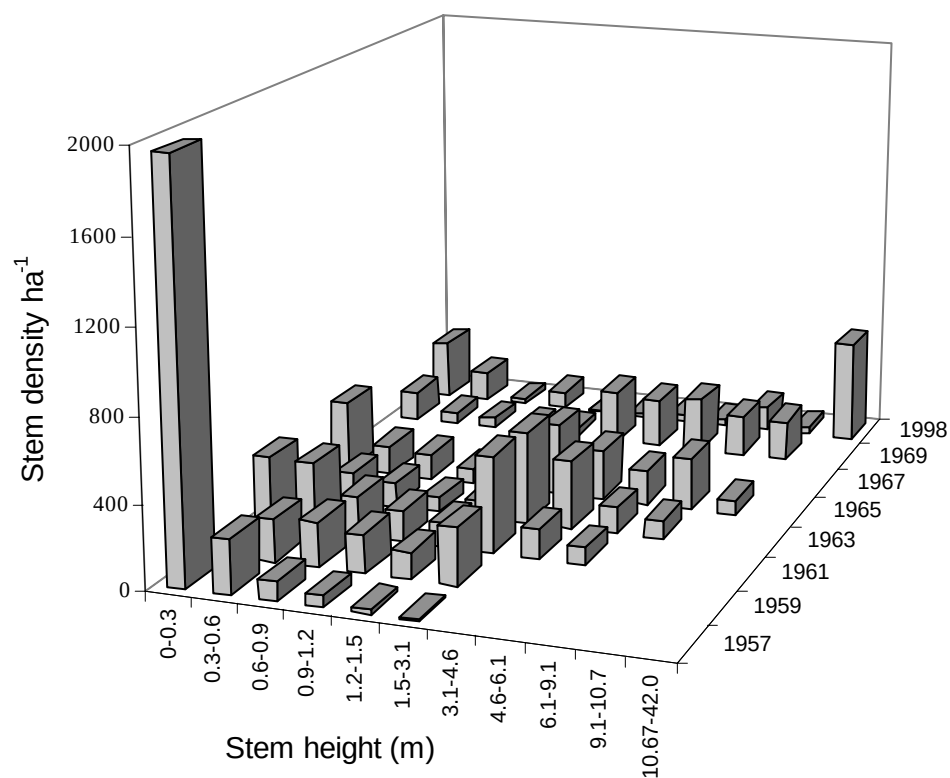


Figure 4.5 Density of all stems per year and treatment at the south Grafton sites, 1957-1998: grazed lower-slope plot

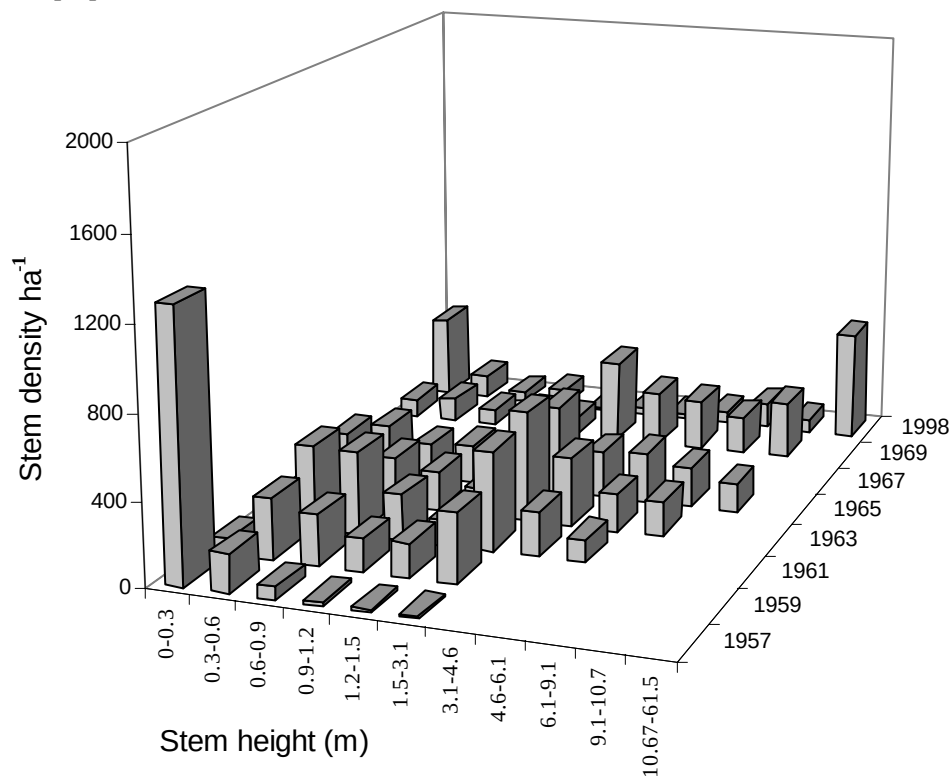


Figure 4.6 Density of all stems per year and treatment at the south Grafton sites, 1957-1998: ungrazed lower-slope plot

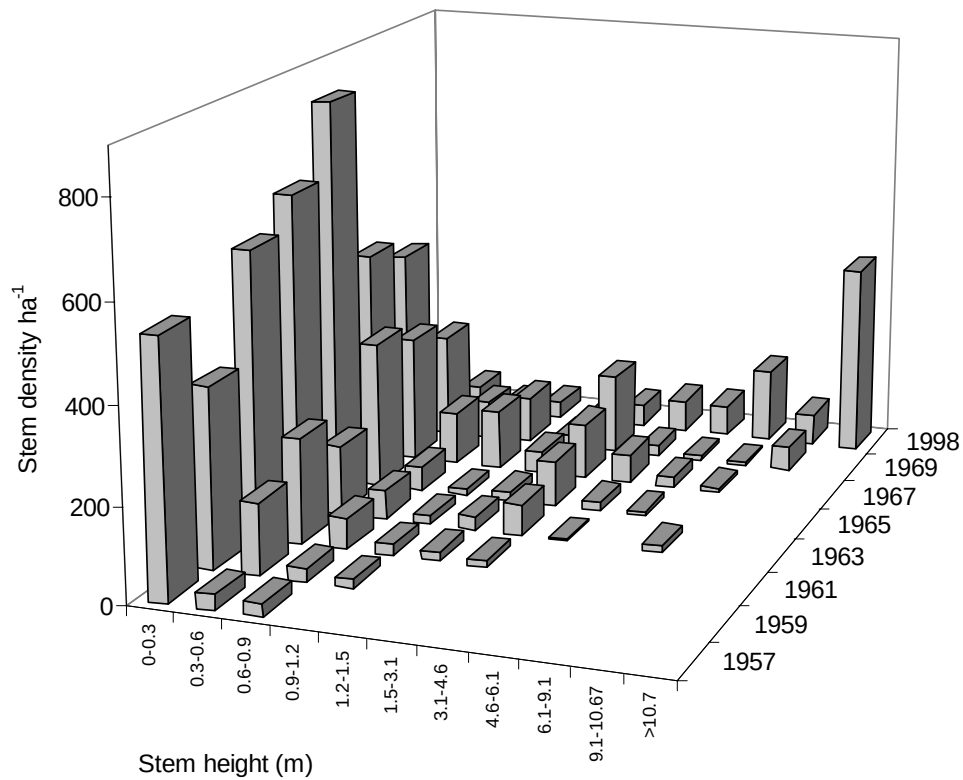


Figure 4.7 Density of all stems per year and treatment at the south Grafton sites, 1957-1998: grazed ridge plot.

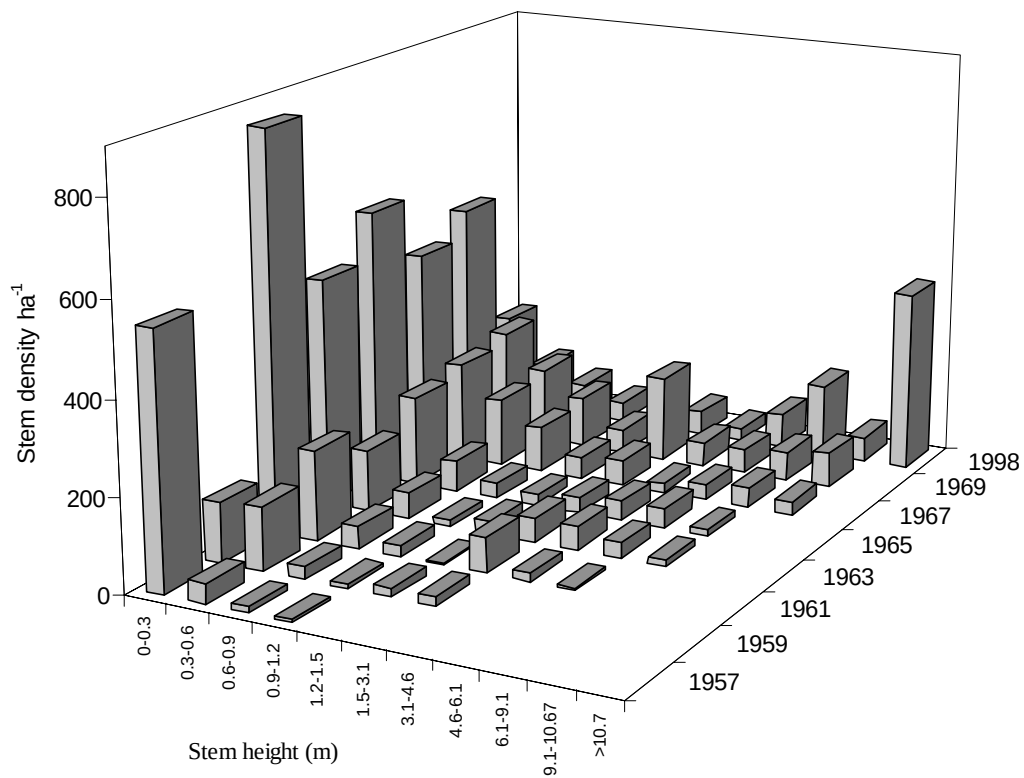


Figure 4.8 Density of all stems per year and treatment at the south Grafton sites, 1957-1998: ungrazed ridge plot.

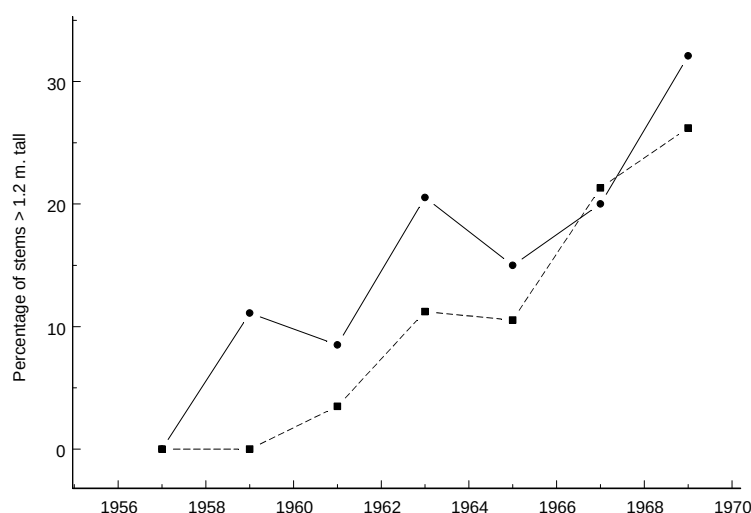


Figure 4.9 The percentage of lignotuberous stems > 1.2 m. tall on the ridge plots that were effectively released for further height growth. (grazed = broken line, ungrazed = continuous line)

The classification and ordination of association matrices based on stem counts data in height increments revealed several significant trends. Despite the initial differences between plots in 1957 due to inter plot variation in tree species composition and density (Table 4.2), these differences were marginal when the data was treated using a multivariate method based on the comparison of inter plot association scores. In both the ordination and classification analyses, using either the Bray-Curtis or profiles attributes association measures, the plots measured in 1957 are in close proximity (Figures 4.10-4.12). Vectors linking plots through time can be considered broadly indicative of the magnitude of temporal change in multivariate space due to the treatments applied. Compared to the ridge plots, the slope plots are changing more rapidly with time, with long vectors linking plots indicating an accelerated progression of stems through the height classes from 1957 to 1965. The slope plots paired in time also move in a similar direction in multivariate space irrespective of being grazed or ungrazed (Figure 4.9 and Figure 4.10) suggesting a lack of a treatment effect. The ridge plots exhibit a reduced magnitude of change with time, especially the grazed ridge plots in which the progression of stems through the height classes is retarded between 1957 and 1961. The length of the vectors between the plots paired in time differs markedly between the grazed and un-grazed treatments over this period.

The different trajectories taken by the ridge and slope plots are evident in the arrangement of plots in either the Bray-Curtis or profiles attributes association measure ordinations and classification dendrograms (Figures 4.11 – 4.13). The ordination based on the profiles attributes association measure, where the association scores are directly related to their ordered height categories, exhibits a tightly clustered pattern of ridge plots in which the developmental trajectory in the grazed plots is only half the length of the ungrazed plot. In the ordination based on the Bray-Curtis association measure, in which the stem counts in height classes are effectively un-ordered, the ridge plots are not tightly clustered, although the overall magnitude of change is still reduced compared to the slope plots. Similarly, the first division in the fusion history evident in the dendrogram, from either association measure, groups the plots into either ridge or slope environments (Figures 4.12 – 4.13).

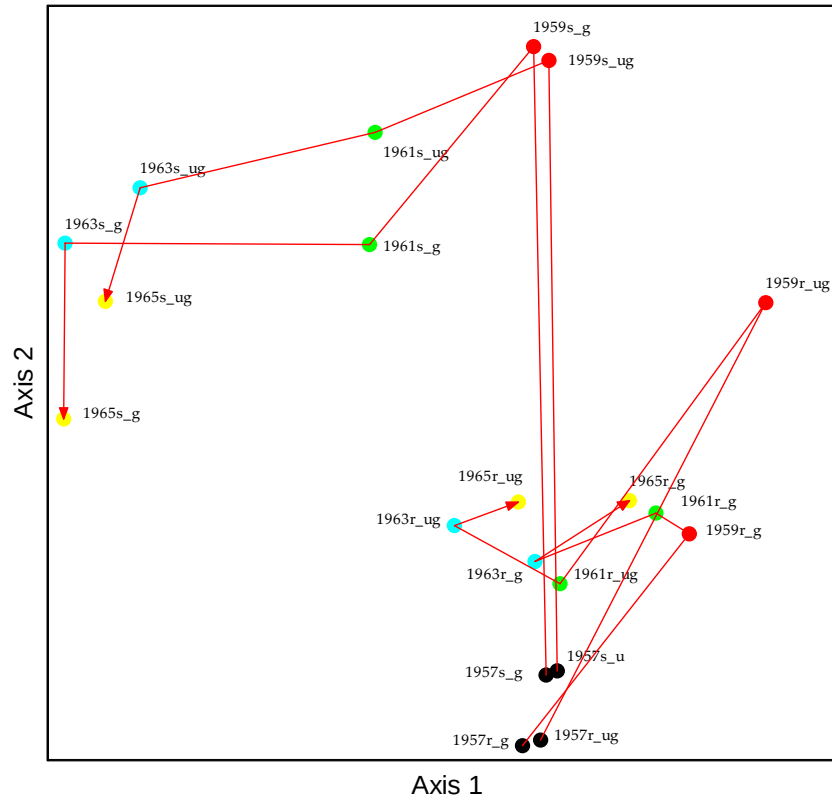


Figure 4.10 Two dimensional ordination of treatment plots for the period 1957 to 1965. Vectors link the same plots through time. The association matrix was based on the Bray-Curtis measure, minimum stress was 0.4701. r=ridge plot, s=slope plot, g=grazed, ug=un-grazed.

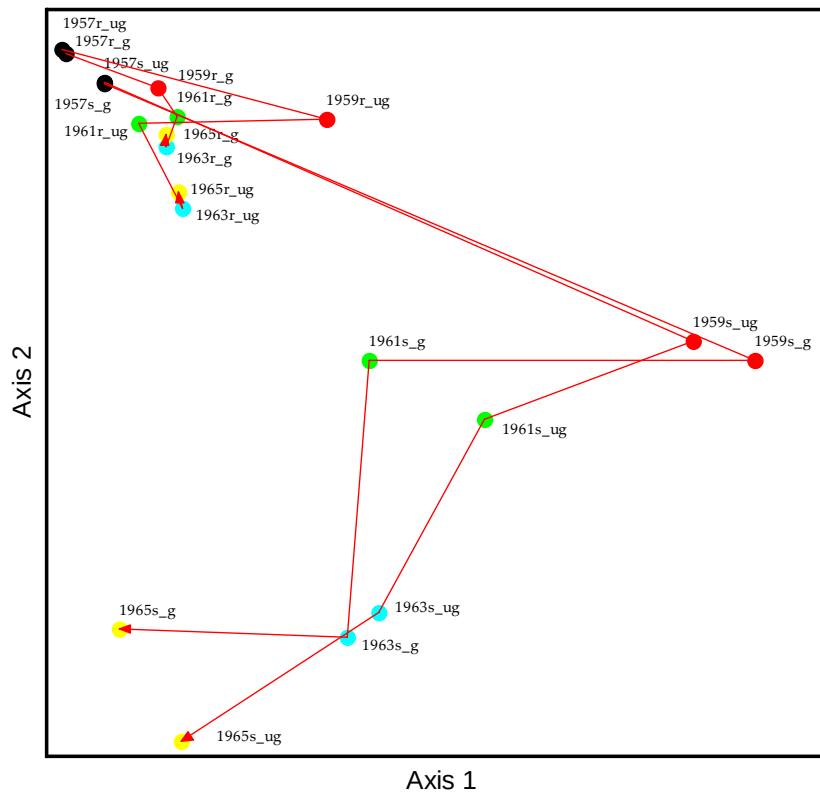


Figure 4.11 Two dimensional ordination of treatment plots for the period 1957 to 1965. The association matrix was based on the profiles attributes measure, minimum stress was 0.1339.

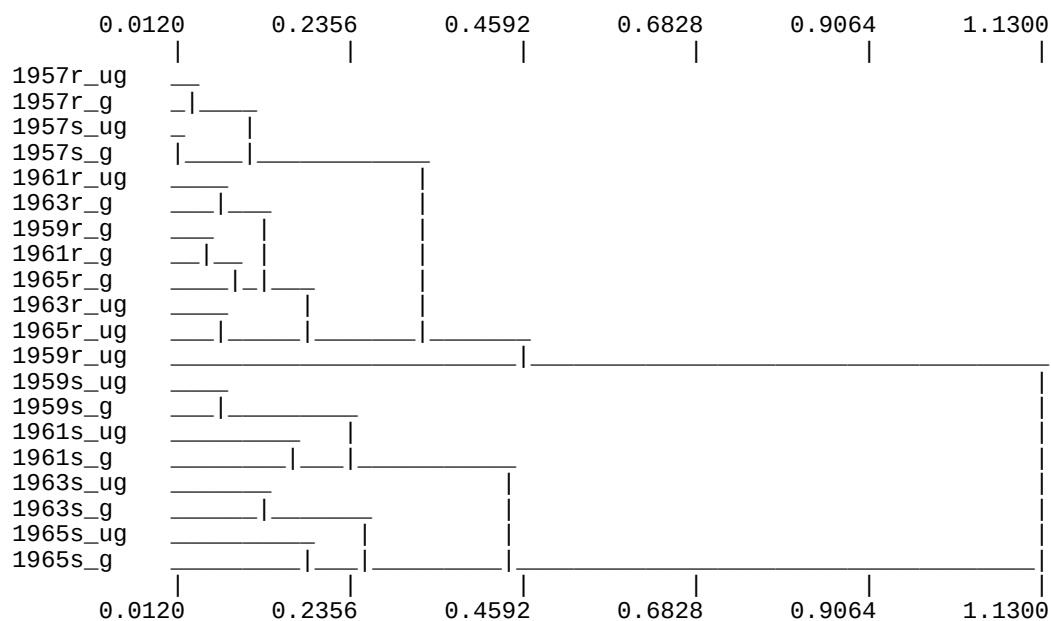


Figure 4.12 Classification dendrogram of treatment plots for the period 1957 to 1965. The association matrix was based on the Bray-Curtis measure. r=ridge plot, s=slope plot, g=grazed, ug=un-grazed.

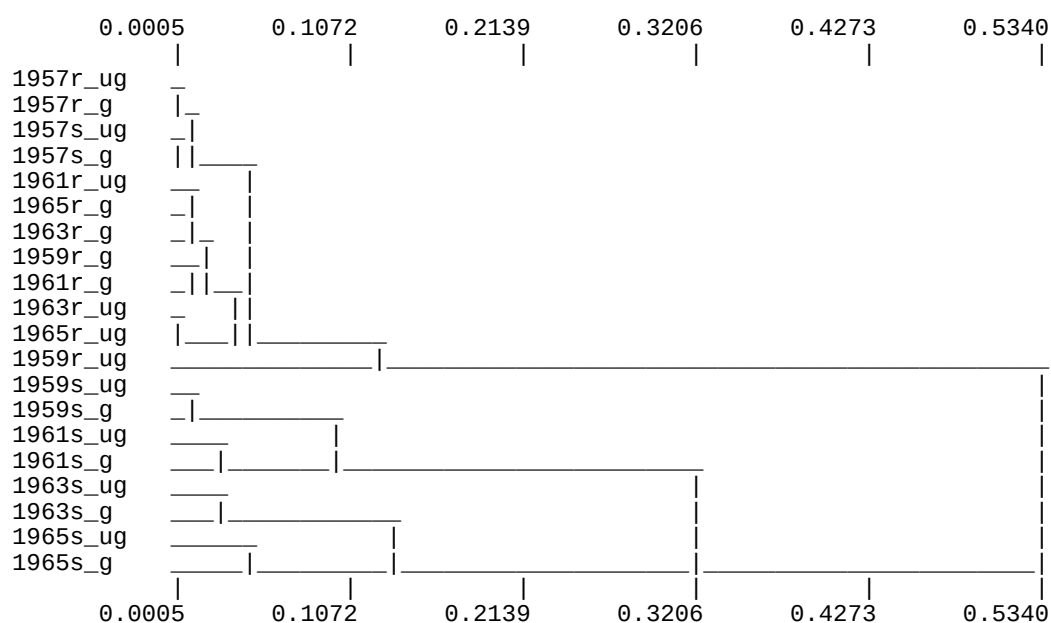


Figure 4.13 Classification dendrogram of treatment plots for the period 1957 to 1965. The association matrix was based on the profile attributes measure.

In 1998, following a period of 24 years since the fencing was removed in 1974, differences in stem density by species were of a lesser magnitude per plot than immediately following harvesting in 1957. The reduction in stem density of individual species on the ridge and slope sites were generally in proportion to their densities at the commencement of the experiment, except *Corymbia henryi* which exhibited a relative increase on the ridge and decline on the slope, and *Eucalyptus moluccana* which exhibited a larger proportional increase on the slope plots (Table 4.4). The density of stems > 10 cm dbh in 1998 was slightly higher than the regional average, however there was no consistent trend across the four plots in relation to the grazing treatments (Figure 4.14).

Table 4.4 Density of all stems per ha⁻¹ at south Grafton in 1998, following a period of 24 years since the fencing was removed in 1974.

	Ridge site		Slope site	
	ungrazed	grazed	ungrazed	grazed
<i>Angohora subvelutina</i>			22	7
<i>Corymbia henryi</i>	658	374	10	172
<i>Eucalyptus crebra</i> ¹				
<i>Eucalyptus fibrosa</i>	47		133	58
<i>Corymbia gummifera</i>			5	
<i>Corymbia intermedia</i> ²			7	
<i>Eucalyptus moluccana</i>	489	634	967	884
<i>Eucalyptus siderophloia</i>			60	2
<i>Eucalyptus tereticornis</i>	9	5	247	123
<i>Eucalyptus tindaliae</i>			27	5
<i>Lophostemon suaveolens</i>				5
total	1203	1013	1478	1256

¹Species recorded in 1957, ²species absent in 1957

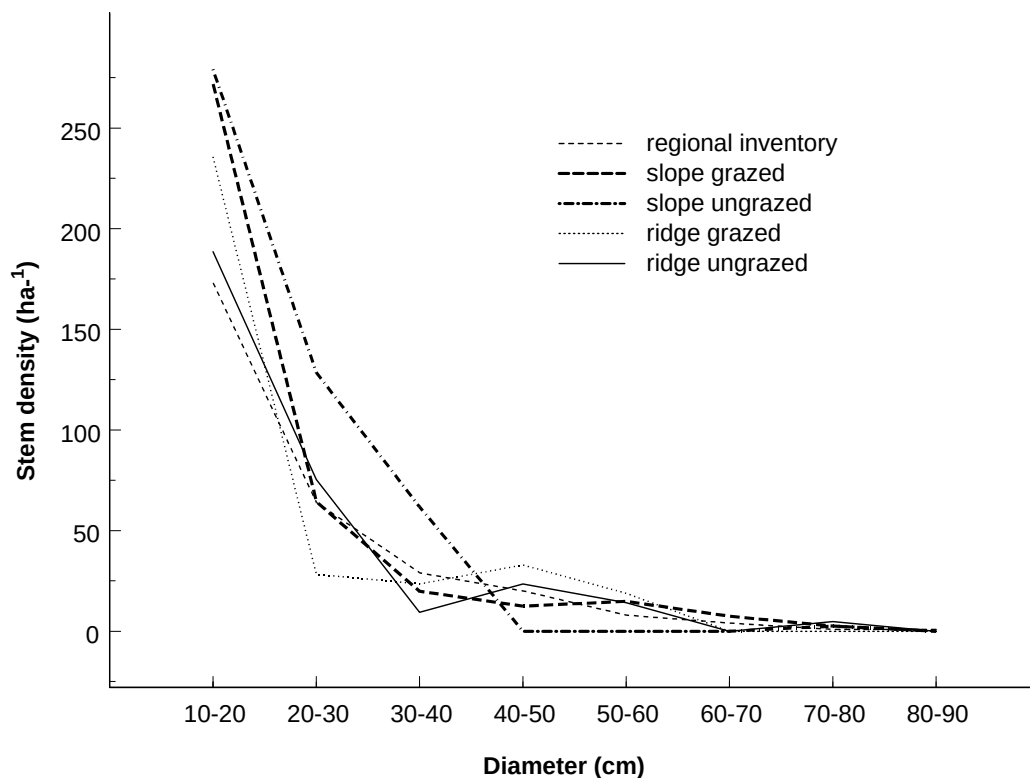


Figure 4.14 Density (stems/ha⁻¹) of overstorey tree species > 10 cm dbh by treatment at south Grafton in 1998. The inventory data is from a stand table developed for the Bom Bom and Glenugie forests by Forests NSW (Margules Groome Poyry Pty Ltd 1994b). Total tree density (stems/ha⁻¹ >10 cm dbh) was 300 (regional inventory), 393 (slope grazed), 472 (slope ungrazed), 339 (ridge grazed) and 316 (ridge ungrazed).

Treatment effects in total stem density by height category were not consistent among individual species (Figure 4.15). For *Corymbia henryi*, stem density declined rapidly after 1957, in both the grazed and ungrazed ridge plots, while in the lower-slope plots it increased. In *Eucalyptus moluccana*, stem density increased in both treatments, the increase being greatest in the lower-slope plots. The post-harvesting trends for *Eucalyptus fibrosa*, *E. tereticornis* and *E. crebra* were highly variable. *Angophora subvelutina* exhibited an initial increase under the grazing regime, however the longer term trends are less clear. For all species and treatments, stem densities of regeneration declined with time, and the differences between the ridge and slope environments, and the grazed and ungrazed treatments become less apparent.

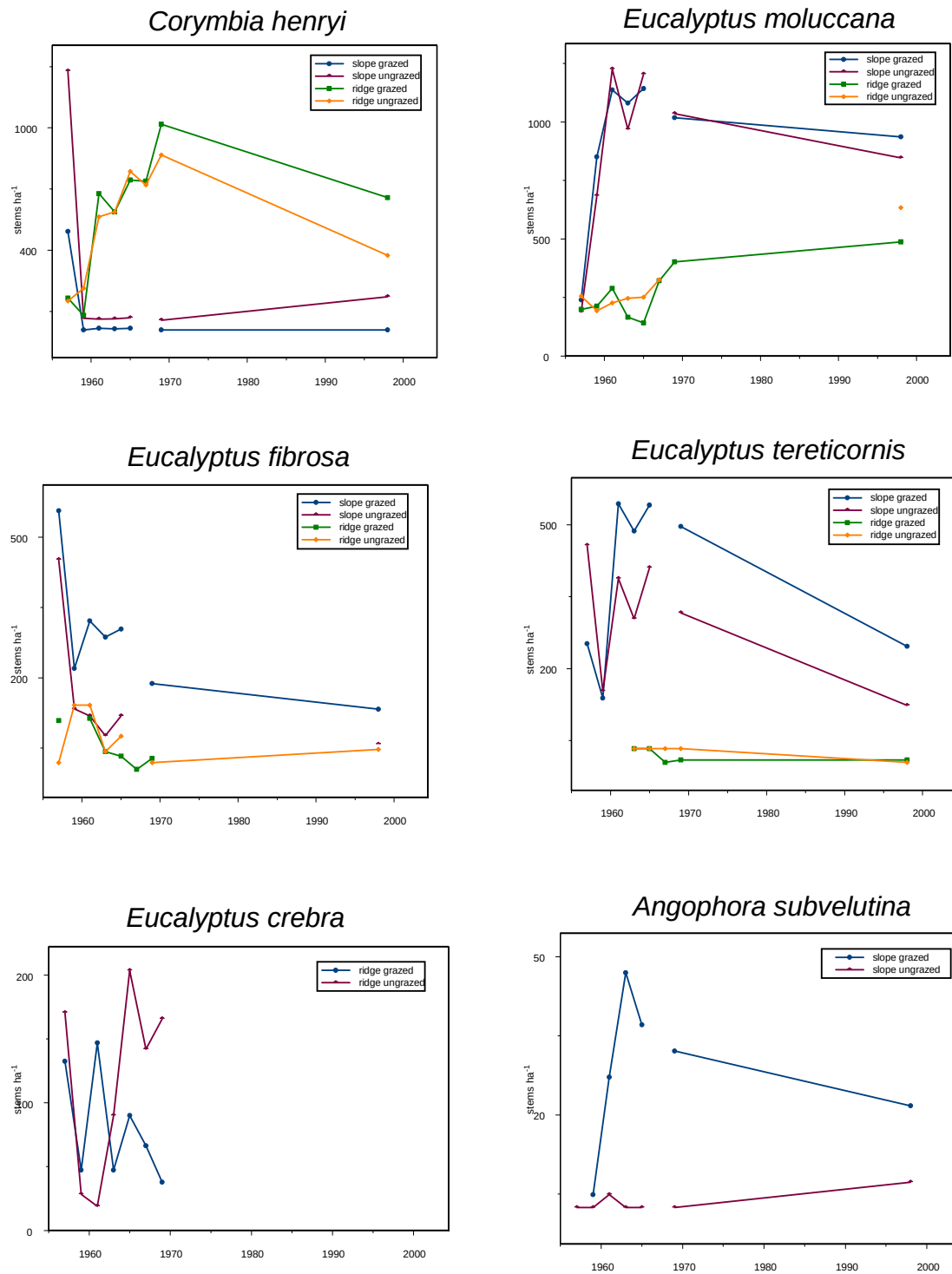


Figure 4.15 Density (stems/ha⁻¹) of overstorey tree species by year and treatment at South Grafton. A broken line indicates a species which was not recorded continuously on a particular plot.

The stand basal area of the two sites and treatments did not differ in 1998 (Table 4.5) and differences in the proportion of basal area by diameter class were poorly related to the treatments applied (Figure 4.16). The ridge grazed plot exhibited a bi-modal distribution of basal area across the diameter classes, indicating two size classes. The ridge ungrazed plot also exhibited a two size class distribution, although some size classes were absent. The slope grazed plot exhibited an irregular uneven-sized distribution while the slope ungrazed plot was essentially even sized. No consistent trends related to the grazing treatments were evident in the basal area of individual tree species in 1998. While some evidence exists from an adjacent State Forest that the dominant tree species can be segregated in terms of their immediate post-logging growth response, those differences decline with time (Figure 4.17).

Table 4.5 Basal area (m^2ha^{-1}) of all tree species present in 1998, following a period of 24 years since the fencing was removed in 1974.

	Ridge site		Slope site	
	ungrazed	grazed	ungrazed	grazed
<i>Angohora subvelutina</i>	-	-	0.029	0.013
<i>Corymbia henryi</i>	12.981	10.04	0.166	8.811
<i>Eucalyptus crebra</i> ¹				
<i>Eucalyptus fibrosa</i>	0.208	-	0.871	0.803
<i>Corymbia gummifera</i>	-	-	<0.001	-
<i>Corymbia intermedia</i> ²	-	-	0.032	-
<i>Eucalyptus moluccana</i>	4.983	8.436	14.312	8.267
<i>Eucalyptus siderophloia</i>	-	-	0.191	<0.001
<i>Eucalyptus tereticornis</i>	0.164	0.002	1.428	0.464
<i>Eucalyptus tindaliae</i>	-	-	1.066	0.041
<i>Lophostemon sauevolens</i>	-	-	-	0.039
total	18.336	18.478	18.096	18.439

¹Species recorded in 1957, ²species absent in 1957

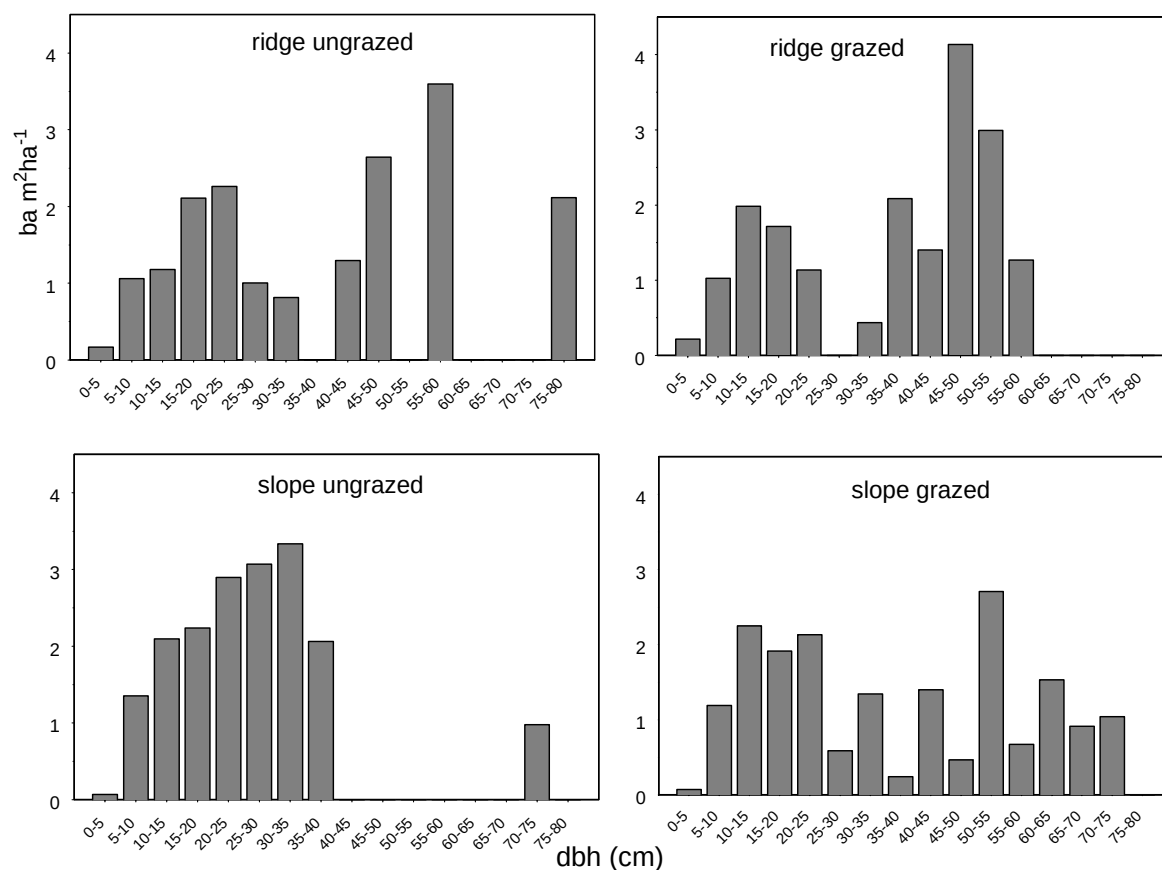


Figure 4.16 Basal area (m²/ha⁻¹) of overstorey tree species in 5 cm diameter classes at South Grafton in 1998.

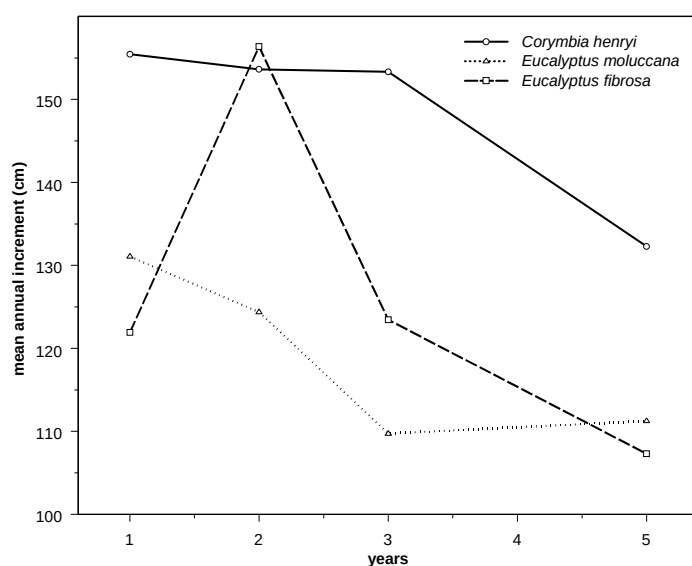


Figure 4.17 Mean annual increment of coppice growth following timber harvesting of *Corymbia henryi*, *Eucalyptus moluccana* and *E. fibrosa* at Glenugie State Forest, south of Grafton. Mean height after five years was 11.43, 12.7 and 8.38 m respectively. Data from Forests NSW, unpublished.



Figure 4.18 Bom Bom State Forest, northern NSW. a) dense regeneration of spotted gum and grey box on the ungrazed ridge and b) slope plots can lead to poor crown development and growth increment. Single tree selection logging (c) is a common method in un-even aged forest stands but it can result in poor regeneration (d) and open-grassy understoreys. Australian Group Selection logging (e) opens larger gaps in the forest and encourages crown expansion and a faster growth increment (f).

5. Discussion

The strongest influence on the composition of the ground flora in young even-aged forests and plantations is the nature and biomass of the overstorey (Young *et al.* 1967, Crawford 1976, Hill 1978, Metzger and Schultz 1984, Alaback and Herman 1988, Taylor *et al.* 1988, Fuhr *et al.* 2001, Moore *et al.* 2006). The forest overstorey influences light penetration (Hill 1978), soil moisture, litter fall, root competition and soil temperatures (Dryness *et al.* 1988). Thinning influences all forest structuring processes, including crown dominance and spatial heterogeneity (Carey 2003). The imposition of commercial thinning in the case studies presented occurred to forests which, in the case of *Eucalyptus regnans* and *E. sieberi* regrowth, exhibit rapid early growth in the first three decades following stand initiation, corresponding with a ruthlessly episodic and spatially clustered pattern of individual tree self-thinning mortality (Ashton 1975, Burgman *et al.* 1994, Florence 1996). In the case of the *Corymbia henryi* and *E. moluccana* forest studied, self-thinning is less intensive as both species will regenerate from lignotuberous stock. Commercial thinning, in the case of the *E. regnans* even-aged forest studied, effectively reduced the tree stocking to a level that, according to stand tables used by local forest managers, would have occurred in the following two decades through natural self-thinning when the less vigorous trees in the stand are becoming sub-dominant and suppressed (Opie *et al.* 1984). Thinning therefore has the potential to reduce rotation lengths in this forest environment by 5-20 years (Brown 1997). While the thinning responses reported here (in terms of height and diameter increment) are consistent with other studies in these forest types, the vegetation and biodiversity responses are less easily compared with the existing literature, which is dominated by studies in northern hemisphere even-aged stands.

In this study, differences in the response of the understorey vegetation to the two cable thinning intensities implemented in southern Tasmania were of less significance floristically than the differences due to thinning method and between thinned and un-thinned vegetation. An increase in thinning intensity can lead to a wide variety of vegetation changes, including an increase in ground fern cover (Alaback and Herman 1988), a reduction in woody species (Reader 1987), or an increase in woody species (Crawford 1976, Wetzel and Burgess 2001), a reduced cover of herbs, an increase in vines and broadleaved species (Shelton and Murphy 1994) or ground layer species (Ducrey and Boisserie 1992, Alaback and Herman 1988) or alternatively it may not effect composition and richness (Locasio *et al.* 1990), or produce a highly variable response (Alaback and Herman 1988, Puettmann and Berger 2006). Where compositional differences due to different thinning treatment intensities are reported, the responsible mechanisms have included differential shade tolerances (Deal 1999), the competitive interaction of woody and herbaceous species promoting shade tolerant species (Crawford 1976), and surface disturbance (Reader 1987). Brunet *et al.* (1996) suggested floristic change following thinning is minimal in the absence of soil disturbance.

A consistent theme from differing forest types (Horsley and Marquis 1983, Alaback and Herman 1988, Bailey and Tappeiner 1998) and both mechanical and chemical thinning (Pitt *et al.* 2000) in the understorey species response is a rapid increase in rhizomatous ground fern species (Crow *et al.* 1991, Harmer *et al.* 2005, Royo and Carson 2006). Rhizomatous species have the potential to store resources or acquire new ones through lateral spread in dense forests (Lezberg *et al.* 1999). An increase in the cover of ground ferns can also lead to a decrease in those species requiring germination spaces to reproduce (Horsley and Marquis 1983). Canopy reduction by cable thinning was directly correlated with an increase in the cover of ground ferns, with the most intensive cable thinning treatment being associated with a ground fern cover greater than 50%, compared to approximately 15% in the unthinned control. Minimal ground disturbance during the cable thinning treatment also resulted in minimal disturbance to their rhizomes, the effective removal of the competing shrub layer, and increased light availability, all of which provides a competitive advantage to rhizomatous ferns in comparison to shrub and herbaceous species relying on soil disturbance to regenerate.

The increased cover of ground ferns post-thinning in southern Tasmania may have retarded the development of other understorey species, similar to the trends observed after woodland thinning (Kirby 1990). Ground ferns will increase in frond cover, length and number following small gap formation in the absence of soil disturbance (Collins and Pickett 1987). Hughes and Fahey (1991) interpreted this type of post-thinning response in ground ferns as being the result of the survival of individuals and the expansion of existing patches. In other examples of clonal (O'Dea *et al.* 1995, Huffman *et al.* 1994), vegetative (Messier and Kimmins 1991, Messier and Mitchell 1994) or horizontal (Mallik *et al.* 1997) expansion following thinning, it was interpreted as a result of either increased light availability or thinning slash encouraging stem layering. Short term changes (after thinning with minimal soil disturbance) in species cover such as these (e.g. Riegel *et al.* 1995, Hughes and Fahey 1991) have been considered by some authors as insufficient to disrupt long term vegetation recolonisation patterns or reduce herbaceous species diversity (Reader and Bricker 1992), and by other authors, they have been implicated as potentially out-competing other ground flora over the longer term (Rooney and Dress 1997) and preventing the natural establishment of tree and shrub species in the ground layer (Huffman *et al.* 1994). In a broad review of the literature, Royo and Carson (2006) referred to this phenomena as the formation of recalcitrant understoreys.

Soil disturbance associated with thinning was an important mechanism for the response of understorey species, particularly herbs and other invasive species. During thinning, the severity of soil disturbance is a function of machine type, soil conditions and the number of load passes (Murphy 1982). The actual area affected is then dependent on the thinning intensity, the planning and location of skid routes, and the log bunching system used (Murphy 1982). Soil disturbance from gouging and litter removal in the cable thinned quadrats differed from soil disturbance caused by tracked machinery during ground-based thinning (0.2-0.45 and 38% respectively). Although compaction was not measured in this study, other authors comparing cable and ground-based thinning concluded ground-based techniques cause significantly more (23% in Wronski 1984) soil compaction than cable systems (McIver 1998), although in terms of the area of soil subject to rutting and water pooling, the difference may be less (Churton 1999). On the ground-based thinned area, rutting was observed where the thinning machinery followed snig tracks created in the 1960's during harvesting of the original old growth stand. These snig tracks were colonised by ruderal species while openings in the cable thinned portion were colonised by herbaceous genera relying on vegetative spread (e.g. *Oxalis*, *Viola*). Most of the herbaceous species common on the ground-based thinned quadrats were able to expand their cover on the more disturbed areas where the soil and litter layers were disturbed by the tracked machine. An interactive effect between ground disturbance and a lack of tree cover is likely; luxuriant herbaceous ground cover reported by Zoysa *et al.* (1986) on a skid trails was attributed to the lack of tree cover on these more open areas, not necessarily to the skid trail disturbance itself. The weed species *Cirsium vulgare* spread throughout the ground-based thinned area presumably because of the larger area of disturbed soil and open ground, or the absence of the previous shrub and herbaceous layer. Herbaceous species in the ground-based thinned area were also more frequently observed to adopt an epiphytic habit, the preferred substrate being large downed logs. These trends in increasing cover of weedy or invasive species following soil disturbance associated with ground-based thinning have been reported elsewhere (Young *et al.* 1967, Metzger and Schultz 1984). With increasing thinning intensity, the number of woody species lost can increase (Reader 1987), while changes in herbaceous species are not necessarily related to thinning intensity. In this study ground-based thinning caused proportionally more ground disturbance and this was correlated with an increase in the weed species *Cirsium vulgare*.

One of the legacies of cable and ground-based thinning is the volume of slash residue. Following cable thinning, up to half of the ground was covered by slash, rising to 70-80% in the ground-based quadrats, which is high compared to results reported elsewhere (e.g. 10-15% in Rosen *et al.* 1987). Additionally, the volume of woody residue persisting from logging in 1961 was approximately 120 tonnes/ha⁻¹. The type, depth and distribution of thinning slash also appeared to differ between the cable and ground-based areas, being distributed more evenly in the cable area, a trend recorded elsewhere (Hall 1991) and of greater volume in the cable area, probably due to its less impacted state (Woodward and Henry 1985). Slash can alter the micro-climate, degree of plant browsing protection,

suppress (Hill 1978) or kill established vegetation cover and prevent its re-establishment (Metzger and Schultz 1984), alter soil chemistry (Rosen *et al.* 1987) and suppress suckering (Bella 1986). It can favour species using clonal expansion (O'Dea *et al.* 1995) and reduce those using suckering (Bella 1986). At West Bemm, the volume of thinning slash present one year after treatment corresponded with the expansion of sprawling species such as *Tetrarrhena juncea* and *Cassytha phaeolasia*. These trends were relatively short lived, however, as thinning residue rapidly loses its volume and depth (Christiansen and Pickford 1991) with much of the smaller diameter material decaying within two to four years (Buckley and Corkish 1991, Christiansen and Pickford 1991, O'Connell 1997).

The vegetation responses from the thinning treatments in each of the three case studies had a common pattern, the initial short term response did not persist beyond the first year in the case of thinning only, or in the case of thinning with grazing, beyond the first decade. Where researchers have monitored thinning conducted 50 years ago (Metzger and Schultz 1981) it has provided new insights into the persistence of invasive species and of the effects of repeated thinning and associated disturbance on the vegetation composition. The case studies examined suggest that while treatment differences in species composition may be small, structural differences may be more persistent. The lack of recovery of the dense shrub stratum in the cable thinning treatments after fourteen years is potentially significant. One of the difficulties of attempting to extrapolate from short term results is that different species may respond to environmental change (e.g. thinning) at different rates. Reader (1988) suggested that variation in the response of herbaceous species to thinning was at least partly due to differing response times, and that by lengthening the study, species that showed no apparent short term response to thinning may do so. Moore *et al.* (2006) similarly described distinct pulses of response following thinning from the immediate increase in annual and biennial herbs at year one, to the short term five year pulse in forbs to other groups which show gradual increases in the first decade. While the short-term vegetation responses to thinning reported are likely to reflect the intensity of the thinning treatment, the longer-term trends are expected to reflect the effects of variation in overstorey retention in the even-aged forest (eg Halpern *et al.* 1999), but not in the uneven-aged forest or the drier sub-tropical forest subject to frequent low intensity burning.

The legacy of thinning in the even-aged forest in the longer term is likely to be evident primarily in the lack of a tall understorey stratum (e.g. *Pomaderris*, *Nematolepis*, *Acacia dealbata*). Patterns of secondary succession 20-30 years after logging in understorey species are likely to be mediated by micro environmental changes associated with the self-thinning of the dense tall shrub stratum, and micro-habitat features (Peacock 2003). Thinning may limit the contribution of large coarse woody debris to the forest floor (McCarthy and Bailey 1994, Hayes *et al.* 1997, Fleming and Freedman 1998) for a period of approximately 20 years (Prescott 1997). Thinning machinery may also crush large logs in advanced decay stages (McCarthy and Bailey 1994). Large diameter fallen logs are often a legacy of earlier stands (Hayes *et al.* 1997, Lindenmayer *et al.* 1999); in the southern Tasmanian study they were abundant from the 1961 harvest event. While an ample decaying log substrate was present to facilitate the re-establishment of vascular epiphytes and rainforest tree seedlings, other specialised substrates such as *Olearia argophylla* bark and *D. antarctica* caudexes were in low abundance but were increasing. The rapid removal of the understorey stratum during thinning, though an unintentional aspect of machinery movement, may therefore disrupt the normal pattern of secondary succession in these forests, removing future resources such as bark substrates for epiphytic establishment. The rapid response of ground ferns to canopy opening through cable thinning may also result in the competitive exclusion of herbaceous and shrub understorey species.

While thinning in the even-aged regrowth forest in southern Tasmania led to a series of vegetation responses broadly similar to those reported in the northern hemisphere examples, thinning in a mixed species un-even aged forest in East Gippsland led to a different suite of changes to the composition and structure of the vegetation. Thinning in this case led to an increase in total stem density over the seventeen year study period, as a result of significant input of in-growth stems, despite the fact that just over half the pre-treatment basal area had been removed. In contrast, in the southern Tasmanian study in-growth was relatively infrequent due to the regeneration requirements of the ash type

Eucalyptus present. While smaller stems (< 15 cm dbh) in East Gippsland of the sub-dominant and in-growth type were the most abundant in the stand, the total proportion of stems across all diameter size classes recovered with the exception of the continued shortage of larger (> 50 cm) diameter trees. The relative shortage of larger trees in East Gippsland post-thinning contrasts with the southern Tasmania study where large trees were absent at thinning due to the silvicultural system applied during the original stand initiating event in 1961.

Species richness, as a traditional measure of biodiversity response, was relatively un-informative in the case studies presented. In the southern Tasmanian study, trends in richness, evenness, diversity and abundance were highly variable with thinning method and time. In the East Gippsland study, species richness exhibited an initial decline, but recovered to exceed the values present prior to treatment. In this example, the short term decline was attributed to the effects of thinning slash, and the eventual recovery to the relatively rapid species turnover in this near coastal lowland forest. More informative trends were evident from an examination of species growth forms. In southern Tasmania herbaceous species increased in cover and richness where ground-based thinning was used and the canopy was less intensively thinned. Increases in herbaceous species are commonly reported following ground-based thinning (e.g. Young *et al.* 1967, Metzger and Schultz 1984, Brunet *et al.* 1996, Uresk and Severson 1998, Oliveira *et al.* 2005, Metlen and Fiedler 2006, Zenner *et al.* 2006) although other authors have reported more variable responses such as increased frequency, no change, or a decrease (Reader 1988), a short term increase (Reader and Bricker 1992) or an eventual decrease in herbaceous cover as other life forms, such as woody species increase in cover (Crawford 1976, Bailey and Tappeiner 1998). While changes in the herbaceous flora can be rapid following soil disturbance from thinning, the effects of canopy removal on herbaceous species may be more subtle (Frego 2000). Some of the variation between these studies in the response of the vegetation to thinning may be due to variability in stand age at treatment (Deal and Farr 1994).

Existing management of the extensive Spotted-Gum / Grey Box forest type in northern NSW and southern Queensland often uses episodic timber harvesting, rotational cattle grazing and frequent low intensity burning to maintain a continuous flow of products to the landholder. Such a management regime is unlikely to provide sufficient regeneration opportunities to release the lignotuberous regeneration, or provide an adequate stand density to maintain productive growing conditions (Bauhus *et al.* 2002). Stands in this condition, with a large component of lignotuberous seedlings and small advance regrowth can be maintained almost indefinitely (Henry and Florence 1966). The release of the lignotuberous seedlings and small advance regrowth following overwood removal is generally considered the primary regeneration mechanism in this dry forest type, with seedling recruitment making a relatively small contribution. In this study, lignotuber and seedling regeneration resulting from the 1954 harvesting event was abundant. Post-harvest grazing had the effect of lengthening the period in which the regeneration was confined to the smallest height classes thereby exacerbating the tendency for the regeneration to become overstocked and not achieve a useful volume growth increment. While the effect of grazing and the resultant slowing of stand growth could not be assessed during the early part of the experiment where stem diameter data were not collected, the 1998 basal area of 18 m²ha⁻¹ on all plots is in accordance with estimates elsewhere for spotted gum forests at the beginning of their cutting cycle (McElhinny 1999, cited in Bauhus *et al.* 2002). Forty-four years after stand disturbance through timber harvesting, productivity patterns did not differ between the grazed and ungrazed treatments which had been in place between 1957 and 1974.

While it is clear that grazing influences tree sapling growth in the years immediately following forest regeneration, its long-term consequences cannot be easily generalised without an understanding of other interacting factors operating at the local and landscape scale (Weisberg and Bugmann 2003). Trends in regeneration of overstorey tree species at south Grafton were highly variable and were possibly influenced by inter-site variation. The clear outcome for the grazing treatment was on the ridge plots, where a greater proportion of stems, principally *Corymbia henryi*, remained in the lower height classes for a longer period. In the ungrazed ridge plot, more individual stems moved through the measured size classes with time. Such an effect appears relatively short lived, in the order of

approximately ten years following timber harvest. On the higher site quality slope plots, treatment effects if any, were indistinct.

The slowing of stand development in the Spotted Gum / Grey Box forest is a common response following grazing, where forest succession can either be stalled, accelerated or deflected depending on the disturbance impacts on species colonisation, growth and survival (Reimoser and Gossow 1996, Cote *et al.* 2004). While these trends are characteristic of even-aged forests, the spotted gum-grey box forests examined frequently exhibit structures variously referred to as multi-aged, multi-sized or irregular uneven-aged, and the lignotuberous saplings and coppice stumps frequently maintain multiple shoots at low heights. At south Grafton, the upper height threshold for grazing damage was 120 cm. Above this height, saplings were effectively released for further growth, although this height effect was only evident for the first ten years following stand regeneration. Height growth was most impaired in the 30 cm or less height class, where in the grazed ridge plot, nearly twice as many stems (compared to ungrazed) were restricted from progressing to the next height class in the first two years following establishment. Regeneration in the spotted gum-grey box forest examined is achieved via a combination of the release of existing seedling and sapling stems, coppice regrowth from cut tree stumps and the recruitment of new seedlings. The ability of *Eucalyptus* species to regenerate following disturbance via seedling recruitment or lignotuberous sprouting can be severely constrained by grazing pressures (Barker 1988, Wimbush and Forrester 1988, Pickering and Barry 2005). Grazing impacts will differ for seedlings and lignotubers, seedling regeneration can be suppressed while the survival of lignotuberous multi-stemmed individual trees to maturity and old age is promoted (Barker 1988). Experimental clipping trials on the closely related *Corymbia variegata* (Walters *et al.* 2005) indicated that lignotuberous seedling survival was a function of age and lignotuber mass, and that by age four months all lignotubers survived clipping, compared to 70% of the non-lignotuberous *Eucalyptus siderophloia* seedlings of the same age. The coppice regrowth strategy is also potentially constrained by grazing which can lead to substantial stem damage to coppice stems (Gill and Beardall 2001). Damage to individual trees is maximised for the smallest or youngest stems (Kay 1993), leading to an increase in the density of small stems, an overall reduction in coppice stem height and understorey cover (Joys *et al.* 2004). The prevalence of lignotuberous saplings and their greater ability to resprout post-grazing from underground storage organs increased their tolerance of continuous grazing pressure compared to a seedling regeneration strategy (Pettit *et al.* 1995). Such a strategy involves trade-offs and allocation costs and probably leads to a delay in the maturity and growth rates of seedlings (Vesk and Westoby 2004 a,b).

The precise nature of the grazing effect is also possibly obscured by its inter-relationship with frequent low intensity burning in order to maintain palatable grazing species, such as *Themeda australis* and interstitial herbs. Cattle grazing will focus on this grassy layer, with other life forms such as tree seedlings and herbs being consumed largely due to the non-discriminatory nature of grazing behaviour (Gordon 2003). In the absence of further burning, low herbs and tussock grasses invade and the dense swards of *Imperata cylindrica* thin, exposing the tree seedlings to grazing damage and increasing their apparency (Rao 2003) or probability of being grazed. It is unclear whether the dense swards of *Imperata cylindrica* offer tree seedlings some protection from browsing damage. In the study of dry tropical savannah described by Pandey (1994) they did not, although in other ecosystems, tree seedlings have been observed to be protected from browsing damage by less palatable species (Rao 2003, Palmer *et al.* 2004 and Smit 2006).

The effect of cattle grazing on the post-harvesting regeneration appears relatively short lived. Excluding cattle for a period of 2-10 years from post-harvesting regeneration is likely to minimise any loss in growth response to canopy disturbance. Similar management policies exist or have been recommended elsewhere, e.g. for grazing exclusion periods of five to seven years (Zamora *et al.* 2001; Papachristou *et al.* 2005; and Teich *et al.* 2005). Current management intent in the dry Spotted Gum-Ironbark-Grey Box forest types on public land in northern NSW is to burn post-harvesting to reduce fine fuels and create a suitable seed bed, and to exclude cattle grazing for two years and fire for five years, until the desired regeneration is established. The results of this study suggest the cattle exclusion

period could be extended to four and ideally ten years in order to minimise any loss in growth response post-harvesting.

Defining a critical threshold for cattle grazing in the Spotted Gum/Grey Box forest is impractical without a clearer understanding of the role of the lignotuberos regeneration pool in stand dynamics, specifically in terms of the maintenance of an acceptable growth increment using a variable-diameter-limit selection cutting system. The results indicate however that forest development was influenced by cattle grazing on the ridge site, and that the influence was most pronounced for the first four years following harvesting, an effect which persisted for the first ten years. The variability of the results over the longer term and among overstorey tree species highlights the importance of maintaining consistent field measurement protocols throughout the experiment and an adequate experimental design and level of plot replication at commencement.

Trends in floristic composition or vegetation structure following thinning in the southern Tasmanian and East Gippsland case studies or logging and grazing in the northern NSW study had several common themes. In the first 1-3 years following thinning or logging and grazing, abrupt changes were observed and treatment effects were relatively consistent across plots despite the limited replication. In both regions a gradual change in composition towards pre-treatment or control plots with time occurred, where treatment effects were less evident as the vegetation developed. Generally the extent of floristic change post-thinning was directly related to the intensity of the thinning event; elsewhere thinning intensity can be directly correlated with changes in the relative cover of understorey life forms (Zenner *et al.* (2006). The extent of this treatment effect was reduced in the East Gippsland uneven-aged stand; Bauhus *et al.* (2001) interpreted their relative lack of a floristic response in terms of the naturally open canopy in lowland sclerophyll forest in East Gippsland.

6. Conclusions

6.1 Implications

The results of this study indicate that regrowth forest stands approximately 30 years old exhibit an immediate and distinct response to canopy thinning, which varies depending on the thinning method used, the nature of the vegetation structure at the time of thinning and its initial composition. Thinning is typically applied to young and fast growing stands where the gain, in terms of growth increment, can be maximised towards the stems considered most valuable at harvest. It is also applied to stands where the natural processes of self-thinning and self-organising at the stand level are resulting in the opening of the overstorey and a resulting increase in the complexity and diversity of vegetation structure and composition. Thinning effectively hastens the natural loss of the smaller tree size classes and promotes the diameter growth of the dominant trees. It also disrupts the normal sequence of secondary succession in understorey vegetation which typically occurs at stand initiation following timber harvesting.

The results obtained from a young even-aged stand comprising a single overstorey species (*Eucalyptus regnans*) differed from those obtained from a young un-even aged stand comprising several overstorey species. Some of the differences were a function of the different species present, others were a function of the silvicultural and environmental conditions present. Other results were interpreted in terms of the light environment, rate of species turnover, regeneration patterns of different life forms and the role of regular low intensity burning. The clear implication was that the vegetation and biodiversity response to thinning is largely a product of a multitude of local site factors and the nature of the thinning treatment applied, and the length of time in measuring the treatment effect. In all three case studies examined, the immediate short term vegetation response was an abrupt one, however the same response measured a decade later was more benign as the differences between the treatment and control sites became less significant.

Recently, forest management regulators in NSW and Victoria have attempted to codify thinning benchmarks which establish minimum stem densities or residual basal areas for different forest types (NSW Department of Natural Resources, 2006). The benchmarks attempt to represent a threshold of disturbance, which if exceeded, would require a more onerous environmental assessment by the landholder. They differ from the traditional thinning schedule approach, as they are attempting to consider not only silvicultural objectives, but environmental ones. Benchmarks also attempt to ensure that the forest structure is maintained at the landscape level, preventing permanent forest landscape change while also providing for silviculturally led improvement to the forest growing conditions. The results presented do not directly test the benchmarks proposed, which, in the case of the Grafton study, are expressed only in terms of a minimum basal area, not a stem density or size class range. They do however provide a clear statement of the vegetation response to a given suite of thinning approaches, and the rate at which the forest and vegetation structure recovers with time.

The three studies reported are all limited in their extent of treatment replication, the variables measured and the influence of possibly confounding factors such as unplanned fires. These limitations were either inherent in their initial study designs or were evident during the design phase and were considered an acceptable compromise to not undertaking the study at all. They do however span reasonably long periods (14-44 years) for ecological studies and represent actual field operations, not subsidised experimentation.

6.2 Recommendations

Biodiversity

The biodiversity responses described were highly variable between treatments, forest types and with time. Nonetheless several trends were apparent which are also consistent with the broader literature (eg Kozlowski 2002) on forest vegetation responses to harvesting disturbance. With this background the following recommendations can be provided to private forest managers.

- Thinning technologies which create a mosaic of treatment areas, for example patches of un-thinned vegetation, more open vegetation and variable retention densities should be encouraged. They are more likely to encourage a broader suite of plant species regeneration or persistence, and in turn, offer habitat resources to a more diverse range of fauna (Etcheverry *et al.* 2005, Metlen and Fiedler 2006).
- Legacies from the previous harvesting event such as large downed logs in various decay stages and large old trees should be retained during thinning, although the potential impact on tree diameter growth increment and machine movement is recognised.
- Disposal burning of thinning slash has the potential to remove its short term smothering effect on regenerating vegetation, although it is likely to induce a post-fire understorey response of a different nature to the thinning response. A management approach which sought to enhance the biodiversity response post-thinning could apply a mosaic of disposal burning and non-burning treatments across the thinned area.

Soil disturbance

- Soil disturbance resulting from thinning operations within regrowth forest has a clear impact on the process of vegetation recovery. Soil disturbance during ground-based thinning operations is potentially more significant than soil disturbance resulting from ground-based logging, as it provides a regenerating niche for ruderal and weed species which are normally competitively displaced by the closure of the shrub stratum in the first decade following logging. Where possible, low impact thinning machinery should be employed, and if feasible, cable technologies used during commercial thinning operations due to their reduced environmental impact.

Logging and grazing

- Existing management of the extensive Spotted-Gum Grey Box forest type in northern NSW and southern Queensland often uses episodic timber harvesting, rotational cattle grazing and frequent low intensity burning to maintain a continuous flow of products to the landholder. Such a management regime is unlikely to provide sufficient regeneration opportunities to release the lignotuberous regeneration, or provide an adequate stand density to maintain productive growing conditions (Bauhus *et al.* 2002).
- While the interaction of logging and cattle grazing was difficult to elucidate, the effect of grazing on the post-logging regeneration appears relatively short lived where the overstorey trees rely largely upon vegetative regeneration. Excluding cattle for a period of 3-9 years post-thinning is likely to minimise any loss in growth response post-thinning.
- The exclusion of cattle grazing, or burning, or both may be warranted after thinning for a part of this period as it is likely to minimise any loss in growth response post-thinning.
- Current management intent in the dry Spotted Gum-Ironbark-Grey Box forest types in northern NSW on public land is to burn post-thinning or logging to reduce fine fuels and create a suitable seed bed, and to exclude cattle grazing for two years and fire for five years until the regeneration is established. The results of this study to support this management intent, although the benefits of cattle exclusion would be enhanced if the two year period above was extended.

- The management of cattle grazing needs to be better integrated with the maintenance of a balanced forest structure designed to produce an acceptable growth increment; with suitable management of forest fuels and fire frequency; and with the broader ecological and biodiversity management objectives generally considered part of a sustainable native forest management system.

Further study

- The thinning studies reported here were based on forests 29-31 years old at the time of thinning, and are generally considered by silviculturists as being beyond the optimal age of thinning, based on the ideal growth increment response. The increasing interest shown by forest managers in non-commercial thinning in native forests (eg Homyack *et al.* 2005, Lasala 2006) requires examination. Commercial thinning of younger forests may elicit a different vegetation and biodiversity response, based on the different level of biological organisation present in younger forest stands, consequently our ability to predict the consequences of non-commercial thinning from published data on older stands may be limited. Younger forest stands potentially subject to commercial thinning require study.
- The interaction of thinning and residue disposal burning requires examination experimentally. Extensive areas of south eastern Australian *Eucalyptus* regrowth forest exist in fire prone environments where post-thinning slash represents a significant fire hazard. The interaction of the post-thinning vegetation and biodiversity response with and without fuel reduction burning should be investigated.
- Fire and cattle grazing have an interactive effect on vegetation (Noy-Meir 1995, Vulliamy *et al.* 2006) and this study was not able to examine the nature of that interaction. Such an interaction is particularly relevant where the overstorey tree species express adaptations to burning and moisture stress (lignotubers) which also confirm an advantage when cattle and macropod grazing is occurring. Future investigations would ideally examine the nature of this interaction via a carefully designed set of manipulative experimental treatments.
- Experimental designs in future need a more explicit consideration of scale, sampling, replication and treatment issues. While experiments conducted using actual forest operations almost inevitably compromise experimental control with operational convenience, the experiments reported would have benefited from additional replication, re-configured plot sizes and a better integration of vegetation and silvicultural measurements.



Figure 6.1 'Thinning for the future', trial thinning of *E. sieberi* regrowth, Nadgee Rd Eden southern NSW, 2005

7. References

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