Integrating Paddock and Catchment Planning: A Wool Producer-Driven Approach to Sustainable Landscape Management

Technical Report: Effect of Vegetation Management on Woodland Communities in the Traprock region

Biodiversity component of Traprock (USQ5) LWW/AWI project

November, 2005

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Integrating Paddock and Catchment Planning: A Wool Producer-Driven Approach to Sustainable Landscape Management (Biodiversity component of Traprock (USQ5) LWW/AWI project): Technical Report: Effect of Vegetation Management on Woodland Communities in the Traprock region

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A research collaboration involving: University of Southern Queensland, Queensland Murray-Darling Committee Inc., Traprock Wool Association Inc. and funded by Land, Water and Wool, a joint initiative of Land and Water Australia and Australian Wool Innovation P/L

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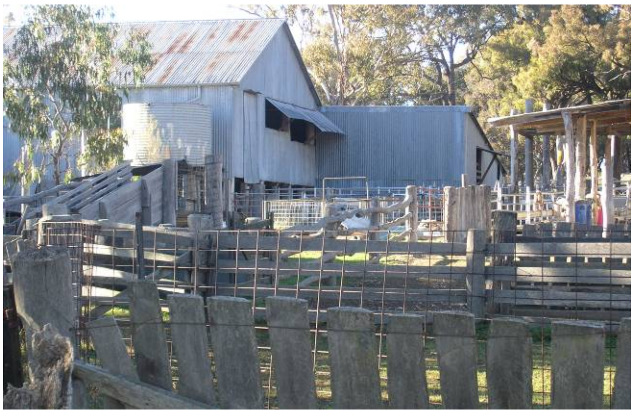

Executive Summary

Integrating Paddock and Catchment Planning: A Wool Producer-Driven Approach to Sustainable Landscape Management (referred to as 'Traprock USQ5 project') is a multi-faceted two-year research collaboration between the University of Southern Queensland, Queensland Murray-Darling Committee Inc., and Traprock Wool Association Inc. The research is funded through Land, Water and Wool, a joint initiative of Land and Water Australia and Australian Wool Innovation P/L as part of the ***Native Vegetation and Biodiversity*** Sub-Program.

The Traprock Association is a proactive group of fine woolgrowers from the high country of south-east Queensland, which has established a voluntary quality assurance system for wool production and marketing. The group aims to link wool production to integrated farm management and landscape planning throughout the region and has detailed property mapping and planning underway to identify biodiversity assets.

The research, initiated by wool producers through the Traprock Wool Association, was a response to a need to identify biodiversity values within the production landscape and link these to socio-economic and production components. The project specifically aims to develop a toolkit that will assist woolgrowers meet national and regional biodiversity objectives. The toolkit will enable woolgrowers to reliably assess and monitor native habitats, identify biodiversity values and maintain profitable and productive land management practices. The toolkit will include management principles, monitoring procedures and guidelines, and protocols for data reporting and management.

The ***Biodiversity Component*** of the Traprock USQ5 project consists of a number of distinct, but related studies, including the examination of the *Effects of Vegetation Management on Woodland Communities in the Traprock region,* which aims to provide a sound scientific assessment of the patterns in biodiversity in response to the broad management practices employed in wool growing properties in the region. This component will contribute directly to the development of biodiversity monitoring procedures and guidelines, particularly through an enhanced understanding of the biodiversity value and potential of vegetation (land) types within the landscape.

Executive Summary (cont.) …

This technical report outlines the patterns in vegetation, including floristic composition, stand structure and species richness, in response to the landholder practise of retaining or removing trees from the landscape. Tree density is used here as a simple surrogate for a complex of livestock grazing practises, which includes grazing intensity and tree (re-growth) removal.

Contents

A separate volume entitled "Appendices to Technical Report" accompanies this report. These appendices contain raw data, descriptive information and some output from statistical analyses. A list of appendices is given below.

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Summary

The modification of natural woodland tree densities through tree removal or clearing is one management option used by landholders to increase native grass production for livestock grazing. These variegated landscapes are found in many eastern states where both livestock grazing and tree removal represent significant anthropogenic disturbances on natural woodlands. An understanding of the effects of vegetation management on native vegetation characteristics is an important first step to ensure appropriate management of woodland communities.

The broad aim of this research project was to determine if vegetation management has an effect on floristic composition, species richness and plant cover in the Traprock wool-producing region of southern Queensland. The following specific questions were examined: Is there a difference in floristic composition, species richness and plant cover between mature tree density classes?; Do woody regrowth areas have a different floristic composition and lower species richness than areas without woody regrowth?; and Are there differences in vegetation responses across vegetation types?

Forty-seven sites were sampled across the study area according to vegetation type (ironbark/gum woodland and box woodland), density of mature trees (low [<6 trees/ha], medium [6-20 trees/ha] and high [>20 trees/ha]), and the presence or absence of woody regrowth. Sample sites were established randomly within patches and the composition and relative abundance (frequency) of plant species were determined in a 500 m² quadrat. Stand structural characteristics, including foliage projective cover of distinct strata, and cover of litters, logs and rocks were also determined subjectively within each quadrat. Tree recruitment was determined by measuring the density of juvenile trees in three height classes < 1m, 1-3m and > 3m within each 500 m² quadrat. Management history (from landholder questionnaires), landscape context, the perimeter and area of sampled vegetation patches, and the level of disturbance at each site were also determined for all sites sampled.

Patterns in floristic composition were determined using Indicator Species Analysis (ISA) and non-metric Multidimensional Scaling (nMDS). Analysis of Similarity (ANOSIM) determined whether there were significant differences in floristic composition between mature tree density classes, with or without woody regrowth. nMDS was also used to assess patterns in cover data for growth forms. The relationship between floristic composition and environment was examined by Canonical Correspondence Analysis (CCA). In addition, analysis of variance was used to determine if groups differed significantly for species richness, percentage cover of selected plant functional groups and tree and shrub recruitment.

Distinct patterns in floristic composition were detected by multivariate analysis. The results show that vegetation management has an effect on floristic composition and plant cover. nMDS showed a major gradient in floristic composition from low and medium density no regrowth sites, through mostly low and medium density regrowth, and medium and high density box woodlands to high density ironbark/gum woodlands. Stand structure (cover of strata) showed much the same pattern as floristic composition.

Low density (open paddock) areas were indistinguishable in terms of pre-European vegetation types, although distinct from other treatments in terms of floristic composition. At higher tree densities, a difference in composition between vegetation types was evident. Woody regrowth areas were shown to have a similar floristic composition to that of high density no regrowth areas. While some patterns in species richness (and components) were evident, richness did not reflect patterns exhibited by floristic composition and cover.

While differences in grass cover exist between low and high density treatments, no difference in grass cover was evident between low and medium density no regrowth treatments, suggesting that intermediate tree densities may not adversely impact on pasture production. Tree recruitment was higher in those areas where the understorey was allowed to regrow, and notably highest in low density plots regardless of vegetation type, indicating that the presence of mature trees may inhibit the excessive regrowth by eucalypt seedlings.

This research shows that retaining trees in variegated landscapes provides significant landscape heterogeneity and important habitats for plant species that may be largely excluded from open grassland habitats. Importantly, there is some support for maintaining a medium density of trees in box woodlands that can potentially provide both production and biodiversity benefits. Future studies should separate the effects of sheep grazing from that of tree density to increase understanding of the ecological changes undergone by vegetation in response to management.

1. Introduction

1.1 General Introduction

Sheep and wool production in Queensland were established as early as 1840 at Toolburra near Warwick on the Darling Downs and today it is the tenth largest agricultural industry in Queensland (Department of Primary Industries and Fisheries 2005). The majority of the yearly wool clip is exported overseas following early stage processing. Leading buyers of Queensland wool include China, Taiwan, Italy, the Czech Republic, France and Germany.

Estimates of the sheep population across Queensland show a decline from approximately 18 million in 1990-91 to 4.4 million in 2002-03, with prolonged severe drought and falling prices seen as contributing factors (Department of Primary Industries and Fisheries 2005). The projected estimate of Queensland's 2003-2004 wool production was 15 million kg greasy wool, down from 36 million kg in 2001-2002 (Department of Primary Industries and Fisheries 2005).

The Traprock Wool Association in southern Queensland is recognised both nationally and internationally for producing high quality fine wool. Annually, approximately 6000 bales of wool ranging from 15.5 to 18.5 micron are produced in the Traprock region (Mills, D. 2005, pers comm., 27 October). Since 2001, an average of 400 000 kg of branded Traprock wool has been sold annually (Mills, D. 2005, pers. comm., 27 October). A majority of the wool produced in the region is exported overseas to China and Italy (Mills, D. 2005, pers. comm., 27 October).

1.2 Land management for livestock grazing

Pastoral land management in Queensland often involves removing or reducing the tree layer to increase native grass production for livestock grazing (McIvor and McIntyre 2002). Pastoral landscapes may be described as variegated, where the landscape matrix is predominately native pastures with varying densities of trees (McIntyre and Barrett 1992). Two major anthropogenic disturbances influencing woodland communities in variegated landscapes are livestock grazing and the associated modification of natural tree densities (McIntyre and Barrett 1992; Clarke 2003). The combination of both livestock grazing & clearing in grassy woodlands can be detrimental to many native plant species (Prober and Thiele 1995; Clarke 2003). For example, in the grassy woodlands of New South Wales, Clarke (2003) found that native forb, shrub and twiner species richness was negatively affected by frequent grazing and the absence of a canopy.

1.2.1 Effects of livestock grazing

Grazing by sheep and cattle represents the single greatest pressure on two-thirds of Australia's agricultural land (Hamblin 2001). In pre-European times, herbivore numbers would have been fairly low and grazing was only likely to be intense on rare occasions or in isolated patches due to scarcity of water and control by predators (Wilson 1990; Tremont and McIntyre 1994). Since the introduction of livestock and permanent water points there has been a major increase in the rate of defoliation of native vegetation and consequently changes in floristic composition and diversity of plant communities (Wilson 1990).

Grazing by domestic stock has altered the composition of understorey species (Prober and Thiele 1995; Clarke 2003), prevented seedling recruitment (Tothill 1971; McIntyre and Lavorel 1994), contributed to soil erosion and compaction (Wahren *et al.* 1994; Yates and Hobbs 1997) and enhanced the invasion of exotic species (Prober and Thiele 1995; Clarke 2003). Furthermore, significant changes in fire frequency have occurred as fire has often been excluded from pastoral lands (Wilson 1990). The exclusion of fire can have important consequences for plant species that require fire to enhance germination (Clarke 2002). For example, Clarke proposes that fire may be required to break the dormancy of hard-seeded shrubs (legumes and epacrids) and stimulate flowering of shrubs (*Lomatia* and *Xanthorrhoea*) prior to a rainfall event.

In the subalpine grasslands of Victoria, the composition of native plant species has been altered due to the selective grazing by livestock of taller forbs and short, palatable shrubs (Wahren *et al.* 1994). In the south-west of Western Australia, livestock grazing has resulted in the loss of native perennial species and subsequent replacement by fewer exotic annual species (Pettit *et al.* 1995). Similarly, Clarke (2003) has found that grazing in the pastoral lands of eastern Australia results in a change in the dominant species of the herbaceous layer, from native warm-season perennial grasses to short-lived exotic cool-season grasses. This also represents a change in the predominant lifecycles in native vegetation from mainly perennial natives to exotic annual or biennial species (Prober and Thiele 1995).

Grazing can also contribute to soil deterioration due to soil compaction and trampling of vegetation (Yates and Hobbs 1997). Soil compaction can impede root growth reducing the ability of roots to provide plants with water and nutrients (Willatt and Pullar 1983; Yates and Hobbs 1997). In heavily grazed areas, water infiltration may be decreased, which has important implications for plant growth, reproduction and seedling establishment (Willatt and Pullar 1983; Yates and Hobbs 1997). Grazing also creates greater areas of bare ground that are susceptible to soil erosion (Wahren *et al.* 1994).

1.2.2 Effects of clearing

In the last 200 years, clearing native vegetation on more productive soils has resulted in the loss of large areas of woodlands in eastern Australia (AUSLIG 1990; Hobbs and Hopkins 1990; McIvor and McIntyre 2002). For example, grassy temperate woodlands that once covered millions of hectares in south-eastern Australia, have largely been cleared for cropping or modified for sheep and cattle grazing (Prober and Thiele 1995; Prober *et al.* 2002). Remnants with near natural understoreys are now rare, with biodiversity often restricted to patches that vary considerably in size, quality and isolation (Prober and Thiele 1995; Yates and Hobbs 1997; Prober *et al.* 2002). Small woodland remnants are particularly vulnerable to loss of native species, changed soil conditions and additional disturbances, while the isolation of remnant patches can result in changes to the normal dispersal and reproductive success of both plants and animals (Hobbs 1987; Prober and Thiele 1995; Gilfedder and Kirkpatrick 1998; Ross *et al.* 2002; Godefroid and Koedam 2003).

In grazing landscapes, clearing trees often has a significant effect on grass production (McIvor and McIntyre 2002). Numerous studies have shown that tree density is inversely related to pasture yield in many Australian woodland communities, with often a significant increase in pasture yield when all trees are removed or killed (Walker *et al.* 1986; Harrington and Johns 1990; Scanlan and Burrows 1990; McIvor and Gardener 1995; McIvor 2001). While increased grass production is ideal for livestock grazing, the removal of trees from grazing landscapes can have negative impacts on original woodland understories. For example, Gibbs *et al.* (1999) report that clearing trees from grazing lands may result in a change in dominant grass species, from shade-tolerant grasses (e.g. *Microlaena*, *Danthonia*, *Poa*) to species that dominate long-cleared pastures (e.g. *Aristida ramosa*). The change in floristic composition has been attributed to altered microclimatic and competitive regimes, and lower soil fertility (Gibbs *et al.* 1999).

Tree removal, by ringbarking or mechanical techniques, has also been associated with an increase in woody regrowth (McIvor and Orr 1991; Eldridge *et al.* 2003). Woody regrowth in grazing lands is often viewed as a significant problem by landholders due to reduced stock carrying capacity and stock management problems (Eldridge *et al.* 2003). In central Queensland, *Acacia harpophylla* (brigalow) regrowth at densities of 3000-5000 stems ha^{-1}, were suggested to have a greater negative impact on pasture production than mature trees due to increased water intake owing to a higher leaf biomass (Scanlan 1991). In the semi-arid woodlands of New South Wales, the perception by landholders is that lower groundstorey cover in woody regrowth areas leads to an increase in soil erosion (Eldridge *et al.* 2003). However, there is limited information on the impacts of woody regrowth on understorey vegetation.

1.2.1 Retaining trees in pastoral lands

Trees have been historically viewed as having a negative impact on grass production (Scholes and Archer 1997); however, recent studies suggest that trees may not be incompatible with production. Retaining mature trees on grazing lands can provide shelter and shade for stock (Walpole 1999), prevent salinity and land deterioration (McIvor and McIntyre 2002), enhance soil nutrients (Gibbs *et al.* 1999) and potentially improve the quality of grasses for livestock (Jackson and Ash 2001). For example, a study by Jackson and Ash (2001) on two native grass species within the open forests of northern Queensland found that trees, through their ability to increase soil nutrient status, can positively influence the quality of grasses by enhancing dry matter digestibility and nitrogen content within leaves. The increase in the forage quality of grasses is proposed to lead to improved diet quality and feed intake for livestock (Jackson and Ash 2001).

In addition, a study by Walpole (1999) in northern New South Wales found that the value of pasture output may be increased by having a proportion of pasture area under dry sclerophyll or woodland vegetation. The gross value of pasture output (determined by the sale of livestock, biophysical characteristics [e.g. slope, soil type, tree cover] and costs associated with agricultural inputs [e.g. labour and fertiliser]) was found to be highest when the proportion of tree area was 34%.

Woodland ecosystems also provide a number of important services for agricultural production. For example, woodlands can contribute to soil formation and protection, nutrient storage and cycling, natural control of diseases and parasitic organisms, insect pollination for seed and fruit set, and the breakdown and absorption of pollutants (McIvor and McIntyre 2002). Woodlands also provide critical wildlife habitats (e.g. for possums, owls, and bats) (McIntyre 2002; Lumsden and Bennett 2005), and habitat diversity, which may facilitate the establishment of different native plant species (Chilcott *et al*. 1997). For instance, within the little-grazed *Eucalyptus albens* (white box) and *Eucalyptus melliodora* (yellow box) woodlands of New South Wales, trees were associated with high heterogeneity in floristic composition and soil fertility and higher species richness than open areas (Prober *et al*. 2002).

Potentially there are both biodiversity and production benefits if trees are retained in grazing landscapes. However, there is little information to suggest what tree density may be appropriate so that both production and conservation goals are met.

1.3 Project Aims

There are many factors that may potentially influence woodland communities within pastoral lands, including livestock grazing, tree removal, and an increase in woody regrowth. In this study, tree density was used as a surrogate of vegetation management practices for livestock grazing in the Traprock wool-producing region of southern Queensland.

Land management for livestock grazing within the Traprock region has resulted in a modification of natural woodland tree densities (Wills 1976). Across the landscape, tree density may range from areas of remnant native vegetation to scattered tree areas to open areas. In addition, previous clearing of woodlands has resulted in patches of regrowth that vary in age and size (Wills 1976).

The objective of this study was to determine the effects of vegetation management on the floristic composition, richness and plant cover of two dominant woodland communities within the study area. This research project examined the following specific questions:

1. *Is there a difference in floristic composition, species richness and plant cover between mature tree density classes?*

2. *Do woody regrowth areas have a different floristic composition and lower species richness than areas without woody regrowth?*

3. *Are there differences in vegetation responses across vegetation types?*

This research is a significant component of an ongoing evaluation of sustainable land management in the Traprock region. Specifically, the project aims to provide an increased understanding of the changes in ecological processes undergone by vegetation in response to management. It is intended that this information will be used for the development of guidelines and principles that support the integration of biodiversity and production objectives.

2. Study Area

2.1 Description of Study Area

The study was undertaken in the Traprock wool-growing region, west of the Stanthorpe-Wallangarra granite belt in southern Queensland. The region is approximately bounded by the major towns of Warwick and Stanthorpe to the east and Inglewood and Texas to the west.

The climate within the region is influenced by both tropical and temperate weather patterns (Queensland Murray Darling Committee 2004). The average minimum and maximum temperatures for the region range between 2.6 - 30.2ºC for Warwick (28º22´S, 152º03´E) and 0.9 - 27.4ºC for Stanthorpe (28º66´S, 151º93´E) (Bureau of Meteorology 2005). Rainfall is generally higher in summer months, although the winter proportion can be significant (Wills 1976). The average rainfall ranges from 701 mm at Warwick to 770 mm at Stanthorpe, but typically declines to the west of the study area (Wills 1976; Queensland Murray Darling Committee 2004).

The Traprock region was first settled over a century ago and since this time sheep have been grazed throughout the area (Wills 1976). Currently, the Traprock region supports approximately 300 000 hectares of sheep grazing country at a stocking rate of about 1-2 dse (dry sheep equivalent) per hectare (Queensland Murray Darling Committee 2004). Wool production is the dominant land-use, with limited winter and summer cropping and horticulture (Wills 1976; Queensland Murray Darling Committee 2004).

The soils developed on 'traprock' (metamorphic sedimentaries) are a complex pattern of shallow loams, shallow earths and shallow texture contrast soils characterised by a high content of angular parent rock (Wills 1976). The fertility of these soils ranges from low to fair and are associated with a generally high subsoil sodicity (Toohey 2004). The lower slopes and drainage lines are dominated by deeper texture contrast soils, while shallower soils occur on the upper slopes (Wills 1976).

The vegetation of the Traprock region is predominately grassy eucalypt woodland mainly comprised of narrow-leaved ironbark (*Eucalyptus crebra*), tumbledown gum (*Eucalyptus* *dealbata*), white box (*E. albens*) and, on the lower slopes, yellow box (*E. melliodora*), greybox (*Eucalyptus microcarpa*) or gum topped box (*Eucalyptus moluccana*). Remnant patches have been mapped by the Queensland Herbarium and classified as Regional Ecosystems 13.11.3/13.11.8, respectively (Environment Protection Agency 2003). Both regional ecosystems are listed as of concern as a result of both grazing and clearing within the region (Environment Protection Agency 2003).

Native pastures are the main source of forage for grazing livestock within the study area (Wills 1976). Past land management practices have largely consisted of ring-barking woody plants to encourage the growth of grasses and burning regularly to control woody regrowth and to remove unpalatable dry herbage (Wills 1976). Wills (1976) reported that between 60 to 100% of original timber on an average Traprock property (approximately 2000 ha) was often killed. Natural grasslands are not considered to have been a common component before settlement (Wills 1976).

Regional ecosystems mapping by the Queensland Herbarium shows approximately 22 % of remnant vegetation remains in the Traprock region (Queensland Murray Darling Committee, 2004), although considerable areas of regrowth and other unmapped vegetation are evident throughout the area. Regrowth areas are considered an important component of the landscape, contributing to higher connectivity for wildlife movement than would be the case if these areas were absent (Queensland Murray Darling Committee, 2004).

Remnant vegetation in the region has been subject to some degree of forestry pressure with a number of species of forestry value. In ironbark/gum woodlands species of forestry value include *Eucalyptus crebra*, *E. cammaldulensis*, *E. tereticornis* and *E. blakelyi*. *Callitris* spp. also occur occasionally in this vegetation type. While *E. melliodora* and *E. macrocarpa* are of value from grassy box woodlands.

Figure 1 shows a diagrammatic representation of some of the main vegetation types in the study area and their approximate position in the landscape.

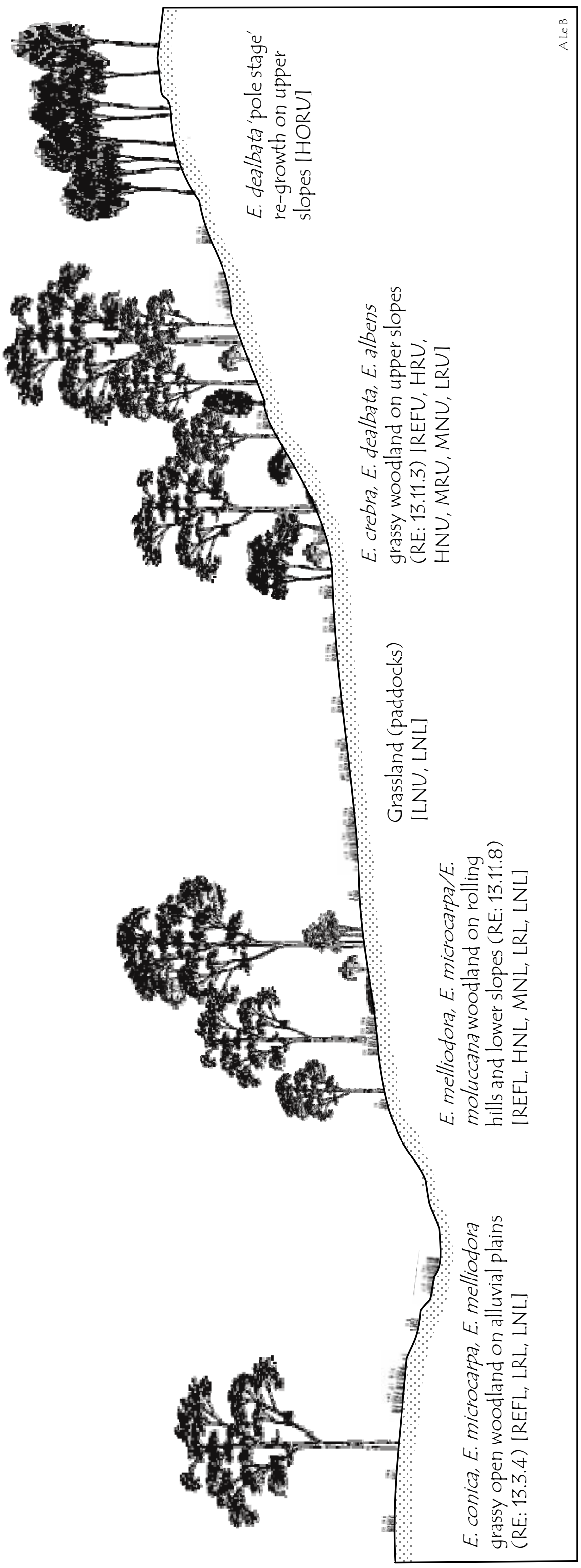

Figure 1. Diagrammatic representation some of the main vegetation types in the study area and their approximate position in the landscape

An explanation of treatment codes is given in Table 1.

2.2 Study Sites & Experimental Design

To ensure the major community types were sampled, site selection was stratified across the study area according to vegetation type, density of mature trees, and presence/absence of woody regrowth. Satellite imagery was used to select potential sample sites and actual sites were chosen in the field if all criteria were met.

Two dominant woodland communities were recognised using the regional ecosystem mapping by the Queensland Herbarium. The grassy box woodlands (RE 13.11.8) typically occur on lower slopes and are distinguished from ironbark/gum woodlands (RE 13.11.3) that typically occupy upper slopes and ridge lines. The box woodlands are dominated by *E. melliodora* (yellow box) and *E. microcarpa* (grey box), while *E. tereticornis*, *E. albens* and *Angophora floribunda* are occasional components of the community (Wills 1976). The box woodland community generally lacks a well developed shrub layer, but occasionally *Acacia* spp. and *Cassinia* spp. may form a dense shrub layer to two metres tall (Wills 1976). The ground layer component is moderately dense and dominated by *Cymbopogon* spp., *Bothriochloa* spp., *Austrodanthonia* spp., *Dichelachne* spp., *Stipa* spp. and *Aristida* spp. (Wills 1976). For the purposes of this study, the box woodlands are labelled as lower (L) slope vegetation.

The ironbark/gum woodlands are dominated by *E. crebra* and *E. dealbata* and occasionally *Angophora costata* (Wills 1976). *E. crebra* may be replaced by *E. sideroxylon* in some areas. The ground layer is typically sparse, but has a well developed shrub layer consisting of *Acacia* spp., *Jacksonia* spp., *Leucopogon* spp., *Daviesia* spp. and *Olearia* spp. (Wills 1976). For the purposes of this study the ironbark/gum woodlands are labelled as upper (U) slope vegetation.

Within each of these vegetation types, sites were assigned to one of three mature tree density classes: low (<6 trees/ha), medium (6-20 trees/ha) or high (>20 trees/ha) based on the number and cover of mature trees observed on the satellite imagery. Sites were then further stratified based on the presence or absence of woody regrowth (<30 years old) and distinguished as either shrub regrowth (e.g. *Cassinia* spp., *Dodonaea* spp.) or tree regrowth (e.g. *E. dealbata*, *E. melliodora*). Areas with tree regrowth greater than 30 years old were also recognised and included as a separate type of regrowth in site selection. These areas are referred to as pole stage regrowth (after McIntyre and Martin 2001).

Potential sites were excluded if the vegetation patch was less than 5 ha in size, if fence lines and water points were less than 250 m from the patch, and if areas were recently cleared (<5 years ago) or burnt (< 10 years ago). Ease of access to sites, spatial spread of sites across the study area, and landholder's permission to access properties were also considered in site selection.

The final uneven factorial design (Quinn and Keough 2002) included 3 main factors:

1. vegetation type (ironbark/gum woodlands; box woodlands);
2. mature tree density (<6 trees/ha [low]; 6-20 trees/ha [medium]; >20 trees/ha [high]); and,
3. woody regrowth (present; absent).

At least 4 replicates of each 'treatment' combination were chosen as potential sites. In addition, replicates of pole stage regrowth sites and reference sites were chosen for both woodland types. The aim was to include the two major vegetation types and associated treatment variables with adequate replication of each. However, equal replication was not possible due to limited representation (and absence) of some treatment combinations within the study area.

Thirteen treatment combinations were recognised (Table 1) and a total of 47 sites were sampled, including 4 reference sites (Figure 2). Three reference sites were located within road reserves, and one within the State Forest. These sites were chosen to represent woodlands with minimal grazing impact; however, they may still be considered disturbed systems.

Table 1. Description of treatment combinations.

Abbreviated description (label), the number (n) of replicates for each treatment combination and site numbers are indicated.

Site description	Label	n	Site numbers
Low density; no regrowth; ironbark/gum woodland	LNU	5	1, 2, 3, 4, 5
Low density; regrowth; ironbark/gum woodland	LRU	5	6, 7, 8, 9, 10
Low density; no regrowth; box woodland	LNL	5	11, 12, 13, 14, 15
Low density; regrowth; box woodland	LRL	4	16, 17, 18, 19
Medium density; no regrowth, ironbark/gum woodland	MNU	4	20, 21, 22, 23
Medium density; regrowth, ironbark/gum woodland	MRU	4	24, 25, 26, 27
Medium density; no regrowth, box woodland	MNL	3	28, 29, 30
High density; no regrowth; ironbark/gum woodland	HNU	4	31, 32, 33, 34
High density; regrowth; ironbark/gum woodland	HRU	2	35, 36
High density; pole stage regrowth; ironbark/gum woodland	HORU	4	37, 38, 39, 40
High density; no regrowth; box woodland	HNL	3	41, 42, 43
Reference; ironbark/gum woodland	REFU	2	44, 45
Reference; box woodland	REFL	2	46, 47

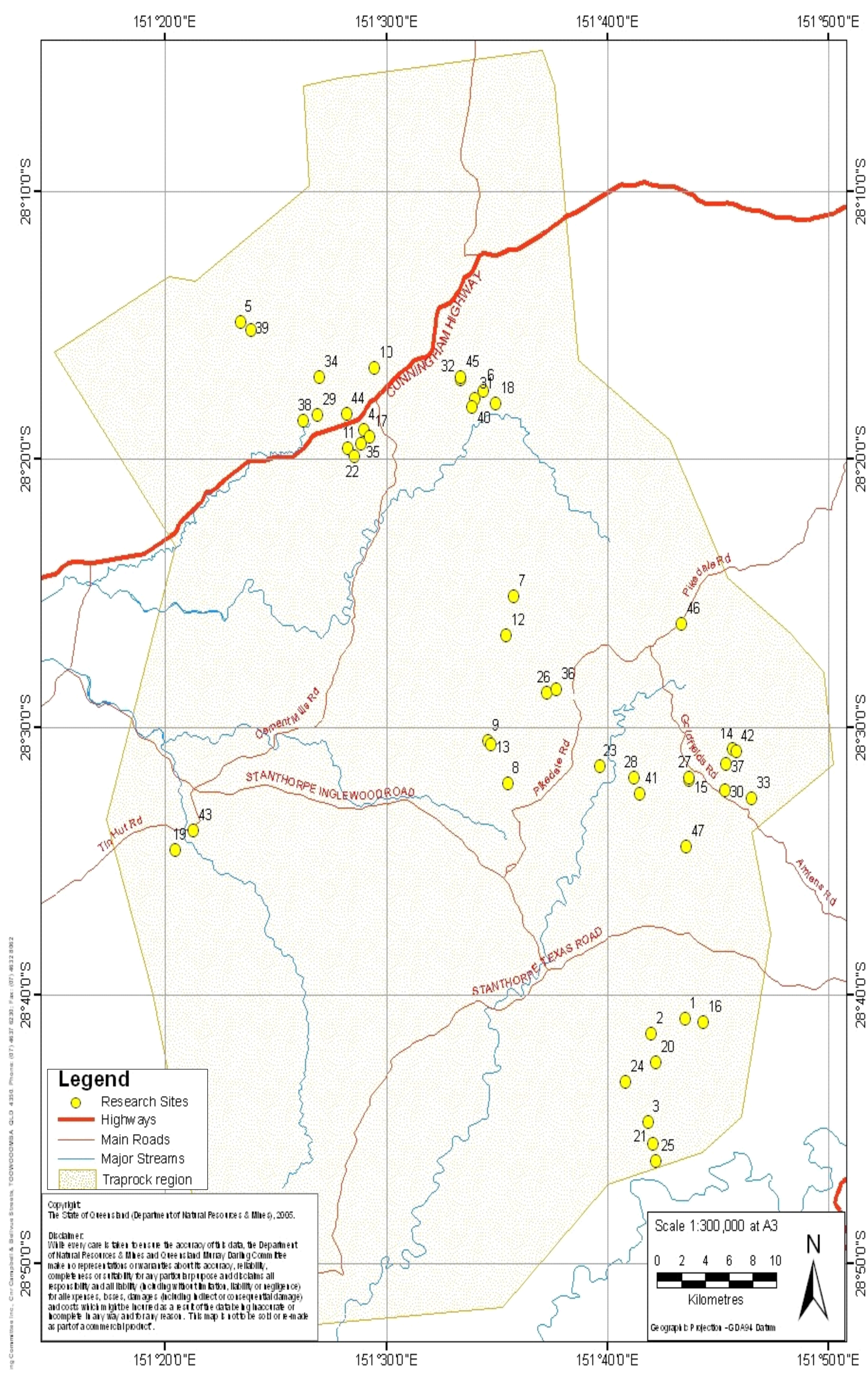

Figure 2. Map of study area showing the location of study sites.

3. Methods

3.1 Data Acquisition

The majority of sites were sampled from September to November, 2004 with two additional sites sampled in December 2004. At each site, a 500 m² quadrat was randomly established by choosing a point close to the centre of the patch and using a list of random numbers to determine distance (number of paces) and direction (compass bearing) from that point. The 500 m² quadrat was laid out from the centre pole, with markers indicating sub-quadrat sizes of 1, 2, 5, 10, 20, 100, 200 and 500 m².

Quadrats were sampled at each site to determine vascular plant species composition, site stand structure, site condition, site physical attributes and the diameter at breast height (DBH) of tree species. The composition and relative abundance of plant species was determined using the frequency-score method (after Morrison *et al.* 1995). Each sub-quadrat, starting with the smallest area was searched for the presence of an individual and assigned a frequency score. An organism that was first encountered in the smallest sub-quadrat was given a maximum score of 9, with lower scores given to species encountered in a larger sub-quadrat.

Plant species that could not be identified in the field were collected and later identified using the nomenclature of Harden (1991a; 1991b; 1991c; 1991d), Stanley and Ross (1983, 1986, 1989) and Auld and Medd (1987). Some grass and forb species that were not flowering or seeding at the time of sampling were identified to genus level. Species unable to be identified, due to lack of any distinguishing features, were labelled as 'unknown' and included in the final plant list. Exotic species were distinguished as any plant species that has been introduced into Australia and identified using the nomenclature of Stanley and Ross (1983, 1986, 1989) and Auld and Medd (1987).

Site stand structure was determined by using the modified Specht (1981) structural classification scheme (after Le Brocque and Buckney 1997). Based on the vegetation of the study area, seven strata were pre-defined: trees >30m, trees 10-30 m, trees <10 m, shrubs >2 m, shrubs <2 m, Forbs/Herbs/Other (non-woody species), and graminoids (including grasses, sedges and others) (Le Brocque and Buckney 1997). As per Le Brocque and Buckney (1997), trees are defined as single or multi-stemmed woody plants greater than 3 m in height and shrubs are defined as multi-stemmed plants less than 3 m in height. The percentage foliage cover of each stratum was subjectively estimated within the 500 m^2 quadrat. In addition, the cover of logs >20 cm and < 20 cm in circumference, rock cover, and the cover of course litter (twigs and branches, 4-10 cm circumference) and fine litter (leaf and twigs, < 4 cm circumference) were estimated and recorded within each quadrat.

Vegetation and habitat condition were also determined by recording evidence of disturbance, recruitment, presence and extent of hides and roosts. Disturbances included grazing, clearing, logging, erosion, weeds, feral animals, soil compaction, bare ground and canopy death and were subjectively scored from 0 (no evidence) to 3 (high) depending on the level of impact within each quadrat (grazing scores were derived through a combination of landholder information and subjective site estimates). Resilience was rated from 0 (none) to 3 (high) and determined by recording the recruitment of trees (> 3 m, 1 - 3m, <1 m), shrubs and grasses, and regrowth (suckering and epicormic) within each quadrat. Structural attributes recorded for hides/roosts included standing stags (>60 cm circumference, <60 cm circumference), tree hollows (>30 cm circumference, <30 cm circumference), log hollows (>30 cm circumference, <30 cm circumference), defoliating/ribbed/creviced bark, other crevices and stumps (>60 cm circumference, <60 cm circumference). These were given a score from 0 (none) to 3 (>10) based on the number of individual attributes within each quadrat. A modified habitat complexity score was developed, which combined both stand structure cover values and hides/roosts data. Cover values were scored as 0 (=0% cover), 1 (=1-9% cover), 2 (=10-19% cover), 3 (=20-30% cover), 4 (=31-50% cover) or 5 (=>50% cover).

Other information recorded within each quadrat included the diameter at breast height (DBH) of tree species and general site physical attributes. The DBH of each tree species within 500 m^2 was recorded to determine the size and density of trees within each quadrat. Site physical attributes recorded at each site included GPS location, slope, aspect, altitude and soil characteristics including soil drainage, texture and colour, and relative soil depth. Site slope was recorded in degrees using a clinometer, aspect was determined using a compass and altitude determined using a GPS receiver (see Appendix 1 for general site details).

Additional information gathered for each site included management history (from landholder questionnaires), landscape context, and the perimeter and area of sampled vegetation patches. A questionnaire for landholders was developed to gather important information on the management practices associated with each sample site. It was recognized that past and current management practices have an important role in determining floristic patterns at any one site. In August 2005, landholders were asked a series of questions relating to each site sampled on their property. Quantitative questions asked for details on patch stocking rate, time since patch was cleared, method of clearing, date of last fire and length of ownership, while qualitative questions asked landholders to describe the patch, provide details on current management practices and score the condition of the patch in terms of biodiversity and production (see Appendix 2A for a copy of the questionnaire and Appendix 2B for a summary of landholder responses).

The landscape context of each site sampled was determined by creating 100 ha and 900 ha grids in Microsoft® PowerPoint® (Microsoft Corporation 2001). The grids were created based on 1 km intervals as seen on satellite imagery maps. Both grids (100 ha and 900 ha) contained 49 points each (see Figure 3) and were arranged so that site position was roughly in the middle. The density of mature trees (either low, medium or high) at each point was determined to give a score out of 49 for each tree density. The score was then given a percentage value out of 100 to determine whether sites were largely surrounded by low, medium or high tree density, giving the landscape context of each site.

The area and perimeter of each patch was calculated in ArcGIS 9 Software package (ESRI 2005). A patch was defined as continuous vegetation of the same tree density. Any evidence of barriers (such as major roads or train lines) and disturbance (such as cleared vegetation), which could be seen on hardcopy satellite imagery maps, reduced the size of the patch. The perimeter to area ratio was determined by multiplying area (in hectares) by 10 000 (to obtain metres squared) and then dividing perimeter by area.

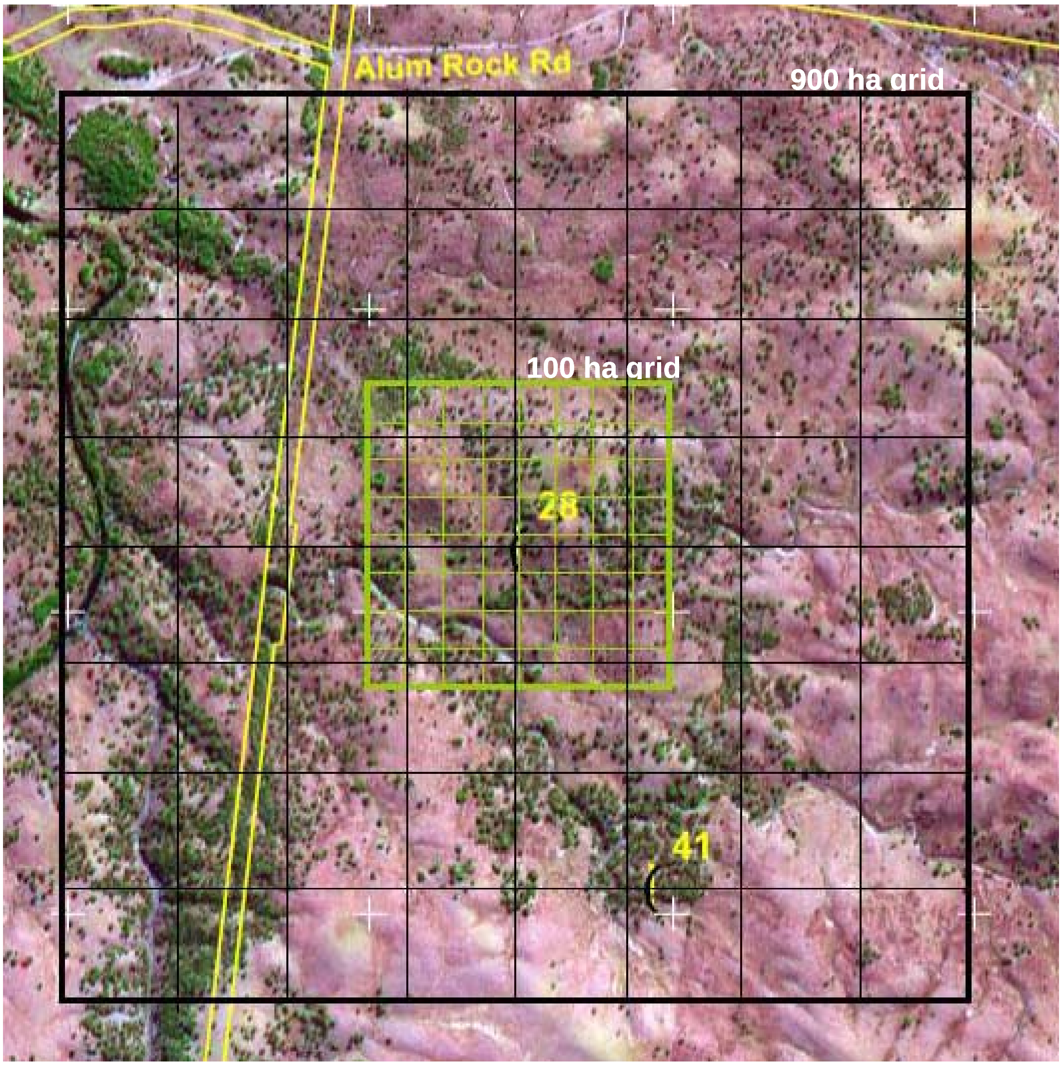

Figure 3. An example of the method used to determine the landscape context of a site (site 28).

This figure shows the satellite imagery and landscape grids used to determine the density of trees surrounding each site at 100 ha and 900 ha.

3.2 Statistical analyses

Two-way indicator species analysis (TWINSPAN) (Hill 1979) was performed on the frequency-score data and the output displayed in a constructed dendrogram to determine whether *a priori* groups (13 treatment combinations; Table 1) were similar in terms of species composition, and which species may be indicative of these groups. Indicator species analysis (ISA) is a divisive hierarchical clustering analysis that divides the first axis for both sampling units and species into smaller groups, combining similar objects (sites) into clusters (Quinn and Keough 2002). While ISA is not intended to find indicator species for predefined groups (McGune and Grace 2002), the results may be examined in combination with an ordination to fully understand group (dis)similarities (Clarke and Gorley 2001). UPGMA clustering analysis was also performed using the Primer v.5.2.9 for Windows computer program to confirm clustering of groups (Primer-E Ltd 2001).

Non-metric multidimensional scaling (nMDS) was performed on the frequency-score data and cover data using the Primer v.5.2.9 for Windows computer program (Primer-E Ltd 2001) to determine the dissimilarity relationship between sites (Clarke and Warwick 2001). nMDS is an ordination method that constructs a map or configuration of the sites in a specified number of dimensions, with sites closer together more similar (eg. in species composition) than those further apart (Clarke and Warwick 2001; Quinn and Keough 2002). The stress value provided with the ordination indicates how well the ordination shows the relationship between sites (McGune and Grace 2002). Higher dimensions (e.g. 3-D and 4-D) tend to decrease stress; however, a 2-dimensional ordination allows a reasonable visual summary of site relationships if the stress value is around 0.2 or less (Clarke and Warwick 2001).

Prior to performing nMDS, the Primer v.5.2.9 for Windows computer program (Primer-E Ltd 2001) was used to compute Bray-Curtis similarity matrix on species data and cover data to allow sites to be represented graphically and to discriminate sites from each other (Clarke and Warwick 2001). Species data were log transformed to allow mid-range and rarer species to exert some influence on the calculation of similarity, by down-weighting more abundant species, and cover data were arcsine transformed to improve normality and reduce the influence of large values (McGune and Grace 2002). Bray-Curtis similarity coefficient is widely accepted as a satisfactory coefficient for biological data on community structure (Clarke and Warwick 2001).

nMDS was also performed on a subset of the species data to determine the relationship between sites in ironbark/gum woodlands and box woodlands. A centroid plot was constructed from each nMDS ordination (all sites, ironbark/gum woodland sites, box woodland sites) by taking the mean of site positions for the first two axes of the ordination for each group.

Analysis of similarity (ANOSIM) was performed using the Primer v.5.2.9 for Windows computer program (Primer-E Ltd 2001). A Bray-Curtis similarity matrix (log transformed) was computed and the ANOSIM test performed on the frequency-score data to determine if there were differences between *a priori* groups (Clarke and Gorley 2001). ANOSIM is a hypothesis testing procedure comparing between-group and within-group variation using rank similarities (Quinn and Keough 2002). The test statistic (R) is scaled to be within the range +1 to -1 and is not overly affected by the number of replicates in the two groups being compared, whereas the statistical significance is dominated by group sizes (Clarke and Warwick 2001). ANOSIM provides a global R value for overall differences between groups, with large values (close to 1) indicating complete separation of groups (Clarke and Warwick 2001). Pairwise R values give an absolute measure of how separated the groups are. An R value >0.75 indicates groups are well separated, R >0.5 indicates groups are overlapping but clearly different, and $R<0.25$ indicates groups are barely separable (Clarke and Gorley 2001).

SPSS® for Windows version 12.0.1 (SPSS Inc. 2003) was used to perform one-way ANOVA (Analysis of Variance) to determine if there were group differences in total species richness, native species richness, exotic species richness, percent cover of stand structure, and total tree recruitment. Total species richness included all taxa recorded, while native species richness excluded unknown taxa and exotic taxa. Total tree recruitment included all tree individuals recorded (>3 m, 1-3 m and <1 m) at each site.

All species richness data and total tree recruitment data were log transformed to improve normality and reduce the influence of any outliers. Cover data were arcsine transformed to improve normality and reduce the influence of large values (McGune and Grace 2002).

Prior to performing one-way ANOVA, the Levene's statistic and residual plots were used to test homogeneity of variances in SPSS® for Windows version 12.0.1 (SPSS Inc. 2003). Where there were unequal variances, groups with high standard errors and/or low sample size ($n=2$) were excluded and the homogeneity test applied to a subset of the data. A Tukey test was performed using SPSS® for Windows version 12.0.1 (SPSS Inc. 2003) when significant differences were indicated by one-way ANOVA. In addition, Spearman-rank correlations were performed using SPSS® for Windows version 12.0.1 (SPSS Inc. 2003) to determine whether cover variables (e.g. tree and grass cover) were related and the degree to which they vary together (Ashcroft and Pereira 2003).

The relationship between multivariate vegetation patterns and environmental variables was examined by canonical correspondence analysis (CCA; Ter Braak 1986; CANOCO program - Ter Braak 1987). Canonical correspondence analysis is a multivariate direct gradient technique (ter Braak 1995) in which the axes of a vegetation ordination are constrained to be linear combinations of environmental variables. Environmental data, consisting of site physical data (slope, aspect, altitude etc), site history and condition data (degree of grazing, logging, weed invasion, time since fire etc), habitat data (litter, tree and log cover), landscape data (patch area), were range-standardised prior to analysis (see Appendix 3 for condition and landscape data).

4. Results

4.1 Site history

The results from landholder questionnaires indicate that land management for livestock grazing is fairly consistent across all properties. The stocking rate for open paddocks (LNU and LNL), and medium tree density patches (MNU and MNL) is similar at 1 sheep to 1.5 acres on average. Commonly, for both regrowth (LRU, LRL, MRU, HRU and HORU) and high tree density patches (HNU and HNL), stocking rate is close to nil with landholders indicating that there is often little forage value in these areas.

Most patches sampled (86% of sites) had been cleared by ring-barking more than 30 years ago with further clearing (often by selective poisoning) taking place in several low and medium density patches (63% of sites). High mature tree density patches (HNU, HRU, HORU and HNL) were not re-cleared. Just over half (52%) of all property sites sampled had been subject to timber harvesting, mainly for commercial use. Commercial timber harvesting generally (63% of sites) occurred between 10 to 30 years ago. Ironbarks are the most commercially harvestable timber in the region with little tree harvesting in box woodland communities.

Fire is used as a vegetation management tool by half of the property owners, often to control regrowth or manage unpalatable grasses, yet most (84%) patches had not experienced fire in the last 20 years. Many landholders were unsure when a fire may have taken place.

Landholders indicated that the main management problems were with woody regrowth and high tree density areas due to the increased mustering effort involved in these areas, lost production as a result of low grass density and cover, and the extra costs associated with woody regrowth control or erecting fences around high mature tree density areas. More than half (63%) of the property owners (11 in total) had owned or managed the land for over 40 years with 23% managing for less than 20 years.

Appendix 4 provides a summary description for each treatment combination. The description includes details on management history, the dominant plant species for that group and the DBH of tree species.

4.2 General Results

A total of 202 plant taxa (171 natives, 16 exotics, 15 'unknowns') from 53 families were recorded (Appendix 5 & 6). Total species richness ranged from 18 to 67 species with an average richness over all sites of 39 species per 500 m². Over all sites, richness of tree species ranged from 0 to 8 species, shrub richness from 0 to 11 species, forb/herb richness from 7 to 36 species, and graminoid richness from 3 to 24 species (Appendix 7).

4.2.1 Sampling Efficiency

A plot of the cumulative number of species against quadrat area showed a levelling of the species-area curve for total species richness (Figure 4). Quadrat size was sufficient to ensure complete sampling of all treatment combinations with an area of 200 m² sampling 88% of all sites. All plots of the cumulative number of species against quadrat area (Appendix 8) showed a levelling of the species-area curve for total species richness.

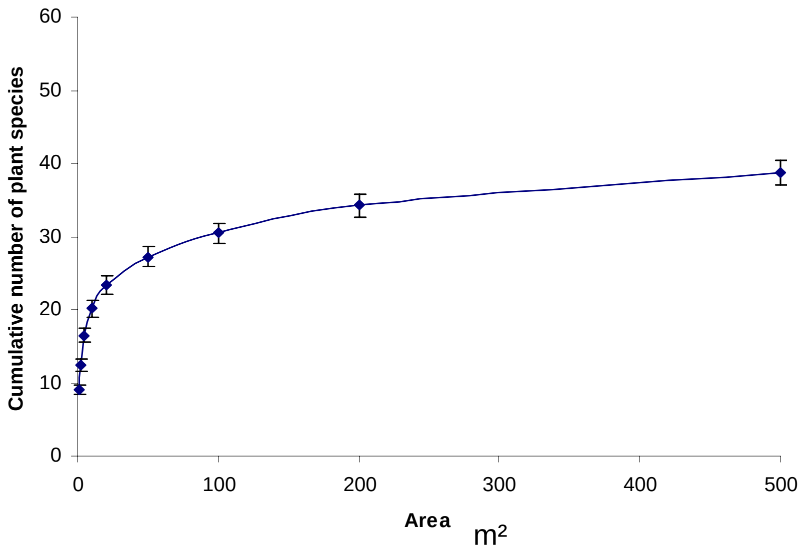

Figure 4. Species-area curve for total species richness from concentric nested sub-quadrats for all treatment combinations.
Symbols are means, error bars represent standard errors.

4.3 Floristic Composition

Indicator Species Analysis (ISA) of floristic composition (frequency) data clustered the 47 samples into 4 main groups that corresponded to low, medium and high mature tree density, and woody regrowth (Figure 5). Groups defined by ISA generally matched *a priori* treatment types. Exceptions included sites within groups LNU, LRU, MRU, HRU and REFL (Figure 5).

ISA shows a broad separation of low and medium tree density sites from regrowth and high tree density sites (Figure 5). Sites with a low tree density and no woody regrowth (LNU and LNL), regardless of vegetation type, were similar in floristic composition to each other, with the exception of site 1 (LNU) which was more similar to regrowth sites (Figure 5). Indicator species for low density no regrowth treatments (LNU and LNL) include *Sisyrinchium micranthum* (exotic annual forb), *Gnaphalium sphaericum* and *Crassula sieberiana* (both native annual herbs). Indicator species are defined as plant species that are common to a particular group (by occurring in more than 75% of sites), but largely absent from another group (by occurring in less than 20% of sites).

Medium tree density sites with no regrowth for both vegetation types (MNU and MNL), and high tree density no regrowth box woodlands sites (HNL) clustered together (Figure 5). However, two sites (8, 27) did not fit this description and were sampled as LRU (low density, regrowth ironbark/gum woodlands) and MRU (medium density, regrowth ironbark/gum woodlands), respectively. The native perennial graminoid, *Cyperus gracilis* was an indicator species of medium and high density no regrowth sites (including reference box woodland sites).

Woody regrowth sites (LRU, LRL, MRU, and HRU) were similar to each other with the exception of sites 8 (LRU), 9 (LRU), 10 (LRU), 27 (MRU), and 35 (HRU) which were placed in different groups (Figure 5). High tree density and no woody regrowth sites in the ironbark/gum woodlands were similar to each other, and included both reference sites (REFU), all HNU sites and three HORU sites. One high density, pole stage regrowth ironbark/gum woodland (HORU) site (40) was more similar to other woody regrowth sites. The native shrub, *Melichrus urceolatus* was an indicator species for regrowth and high density sites (including all reference sites).

The dendrogram from UPGMA clustering analysis is included in Appendix 9. The UPGMA dendrogram shows a clear separation of low density, no regrowth sites (LNU and LNL) from high density ironbark/gum woodland sites (HNU, HORU and REFU). Similarities between medium density, no regrowth sites (MNU and MNL), regrowth sites (LRU, LRL, MRU and HRU), high density, no regrowth box woodland sites (HNL) and reference box woodland sites (REFL) were also indicated.

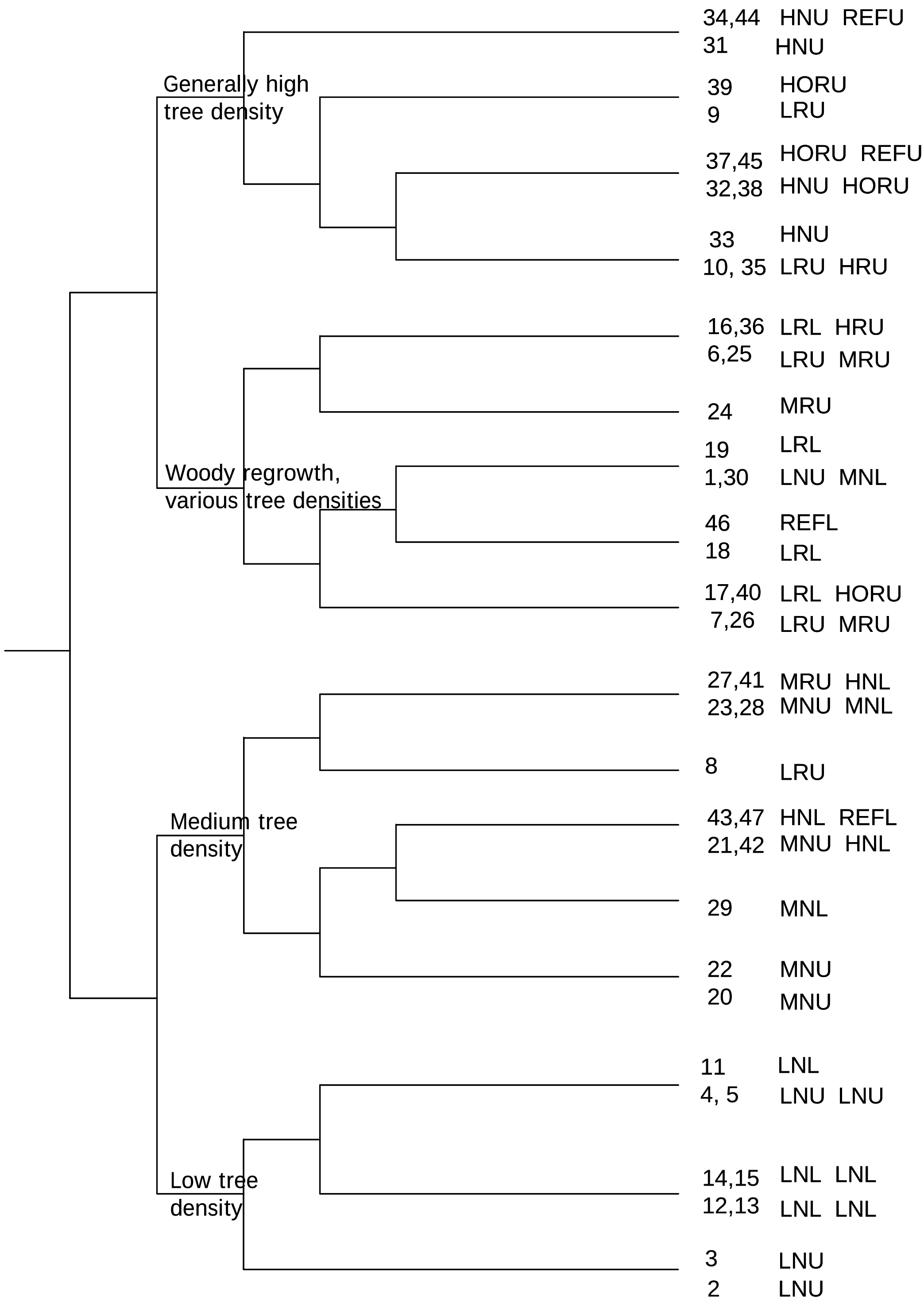

Figure 5. Dendrogram showing results from ISA analysis of frequency-score data.
Site numbers and corresponding descriptive labels are shown. See Table 1 for descriptions of treatment codes.

Non-metric multi-dimensional scaling (nMDS) ordination of floristic composition data (Figure 6) indicates a gradient of increasing mature tree density from left to right across the diagram. The stress value for the 2-dimensional ordination (0.22) indicates that the ordination reasonably shows the relationship between sites. Low tree density no regrowth sites (LNU and LNL) were well separated from a cluster of low density regrowth and medium density treatments (LRL, LRU, MRU, MNL, MNU) and high tree density box woodlands (HNL, REFL). High density ironbark/gum woodlands (HNU, HRU, HORU, REFU) are also reasonably well separated from this cluster, representing the other end of the cluster.

The centroids of each *a priori* treatment combination from the nMDS ordination of floristic data are shown in Figure 7. The centroid plot shows the overall gradient of treatments across the two-dimensional ordination. The gradient in floristic composition (from left to right of the ordination) is from low and medium density no regrowth sites (LNU, LNL and MNU), through mostly low and medium density regrowth (LRU, LRL and MRU) and medium and high density box woodlands (MNL, HNL and REFL) to high density ironbark/gum woodlands (HRU, HNU, HORU and REFU).

ANOSIM of all floristic data (Table 2) found no significant differences in floristic composition between low density no regrowth ironbark/gum woodlands (LNU) and low density no regrowth box woodlands (LNL) (R=0.176). However, both treatments were significantly different to all other treatment combinations, including regrowth, medium and high density groups, and reference groups. The exception was no significant difference between low density no regrowth ironbark/gum woodland (LNU) and medium density no regrowth ironbark/gum woodland (MNU) (R=0.231) (Table 2).

High density, no regrowth ironbark/gum woodland treatment combination (HNU) was not significantly different to high density regrowth ironbark/gum woodlands (HRU) (R=-0.036), high density pole stage ironbark/gum woodlands (HORU) (R=0.104), reference ironbark/gum woodlands (REFU) (R=-0.25), and reference box woodlands (REFL) (R=0.179), but was significantly different to high density, no regrowth box woodland treatment combination (HNL) (Table 2).

With a few exceptions (see Table 2), ANOSIM results show that there were significant differences in floristic composition between box woodlands and ironbark/gum woodlands. On this basis species data were divided into ironbark/gum woodland sites and box woodland sites and nMDS ordinations performed on each subset.

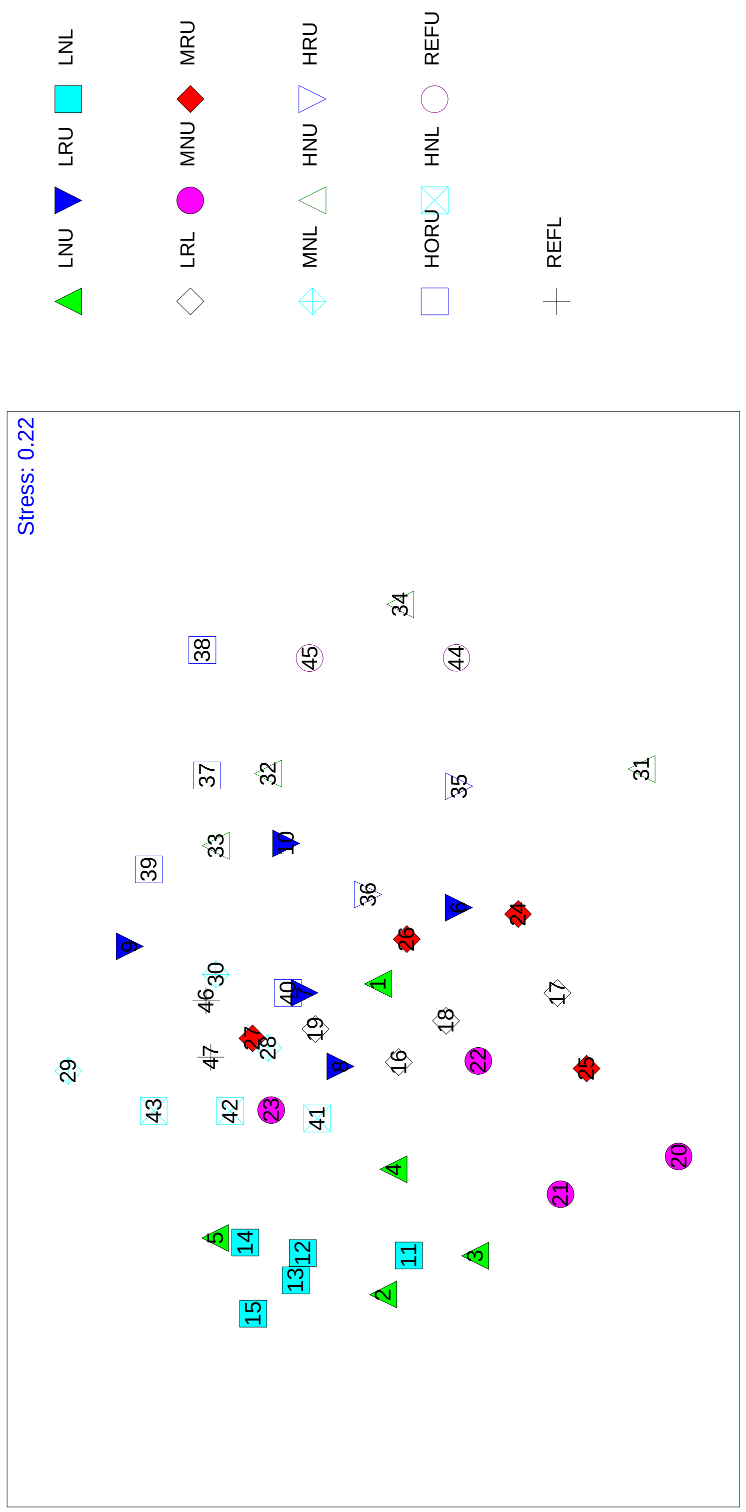

Figure 6. Non-metric multi-dimensional scaling ordination of floristic data.
Ordination shows the clear distinction between samples and the (dis)similarity within and between treatment groups. See Table 1 for descriptions of treatment codes.

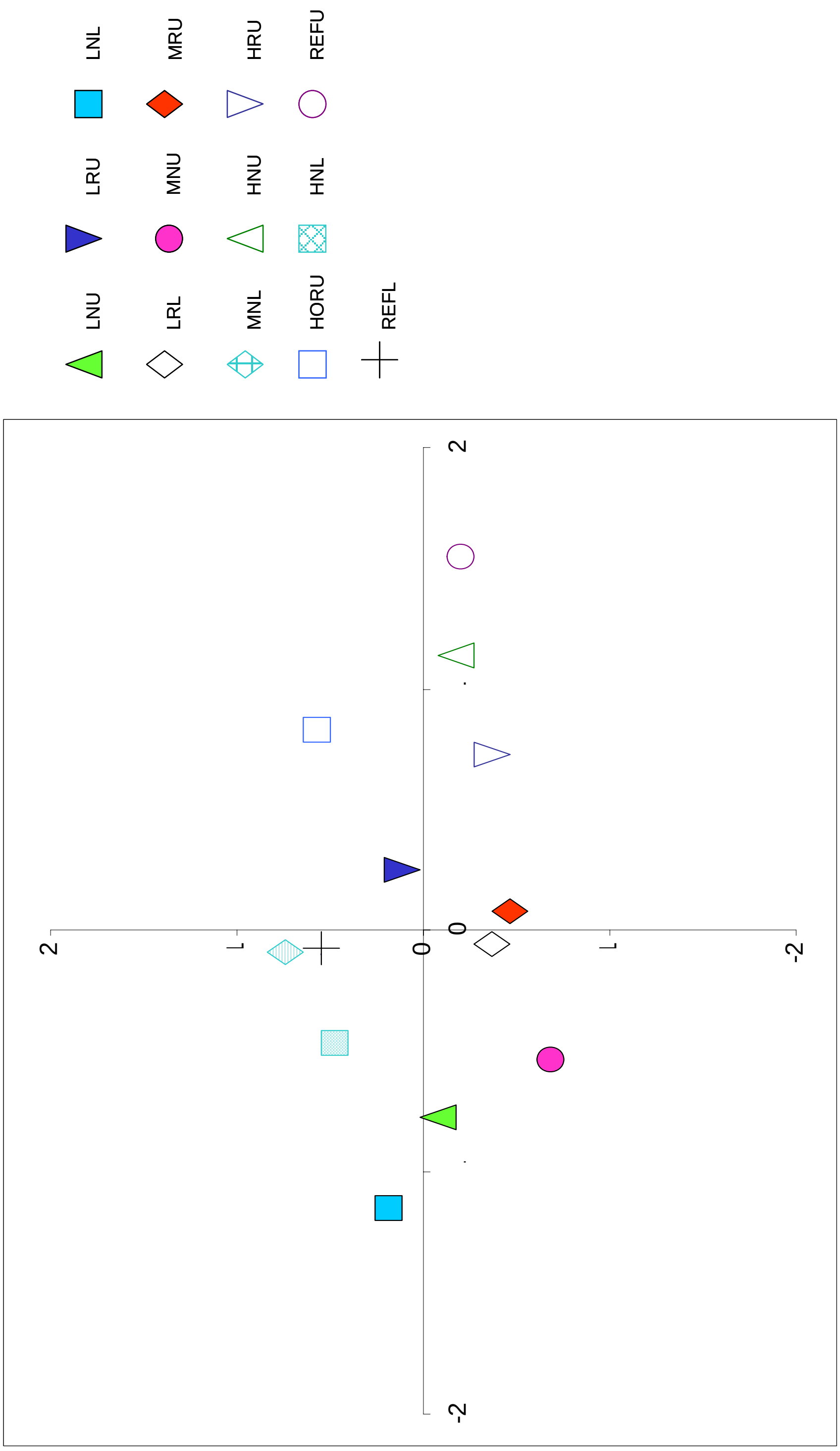

Figure 7. Centroid plot of nMDS ordination of floristic data from all sites.
See Table 1 for descriptions of treatment codes.

Table 2. ANOSIM R values for each pairwise group comparison.
* R values are significant (R>0.25)

	LNU	LRU	LNL	LRL	MNU	MRU	MNL	HNU	HRU	HORU	HNL	REFU
LRU	0.46*											
LNL	0.18	0.86*										
LRL	0.29*	0.31*	0.87*									
MNU	0.23	0.51*	0.54*	0.28*								
MRU	0.31*	0.11	0.74*	0.02	-0.05							
MNL	0.49*	0.29*	0.85*	0.20	0.28*	0.32*						
HNU	0.77*	0.50*	0.93*	0.62*	0.54*	0.28*	0.35*					
HRU	0.64*	0.46*	1.00*	0.43*	0.36*	-0.21	0.50*	-0.04				
HORU	0.65*	0.34*	0.91*	0.63*	0.67*	0.46*	0.48*	0.10	0.46*			
HNL	0.40*	0.59*	0.87*	0.61*	0.22	0.41*	0.15	0.37*	0.92*	0.72*		
REFU	0.89*	0.78*	1.00*	0.96*	0.96*	0.93*	1.00*	-0.25	0.50*	0.68*	1.00*	
REFL	0.64*	0.44*	0.89*	0.32*	0.07	0.00	0.00	0.18	0.50*	0.61*	0.67*	1.00*

An nMDS ordination of floristic data for ironbark/gum woodland sites (Figure 8) revealed a similar relationship between sites and groups, as shown by ISA and nMDS ordination of all sites. The stress value of the 2-dimensional ordination is below that of the previous ordination at 0.19 indicating a better representation of the relationship between samples. The ordination shows a gradient of increasing mature tree density from right to left.

A centroid plot of each ironbark/gum woodland treatment type (Figure 9) shows the mean position of groups and includes ANOSIM R values (at $R<0.25$). High density no regrowth treatment combination (HNU) was not significantly different to other high tree density groups (HRU, HORU, and REFU); however, significant differences in floristic composition existed between HRU, HORU and REFU. Low tree density treatments (LNU and LRU) were significantly different, but were not significantly different to medium tree density treatment combinations with the same regrowth treatment ($R= 0.231$ for LNU and MNU; $R= 0.106$ for LRU and MRU). High density and medium density regrowth treatment combinations were not significantly different ($R= -0.214$), and were similar to no regrowth treatments with the same tree density ($R= -0.036$ and $R= -0.052$, respectively).

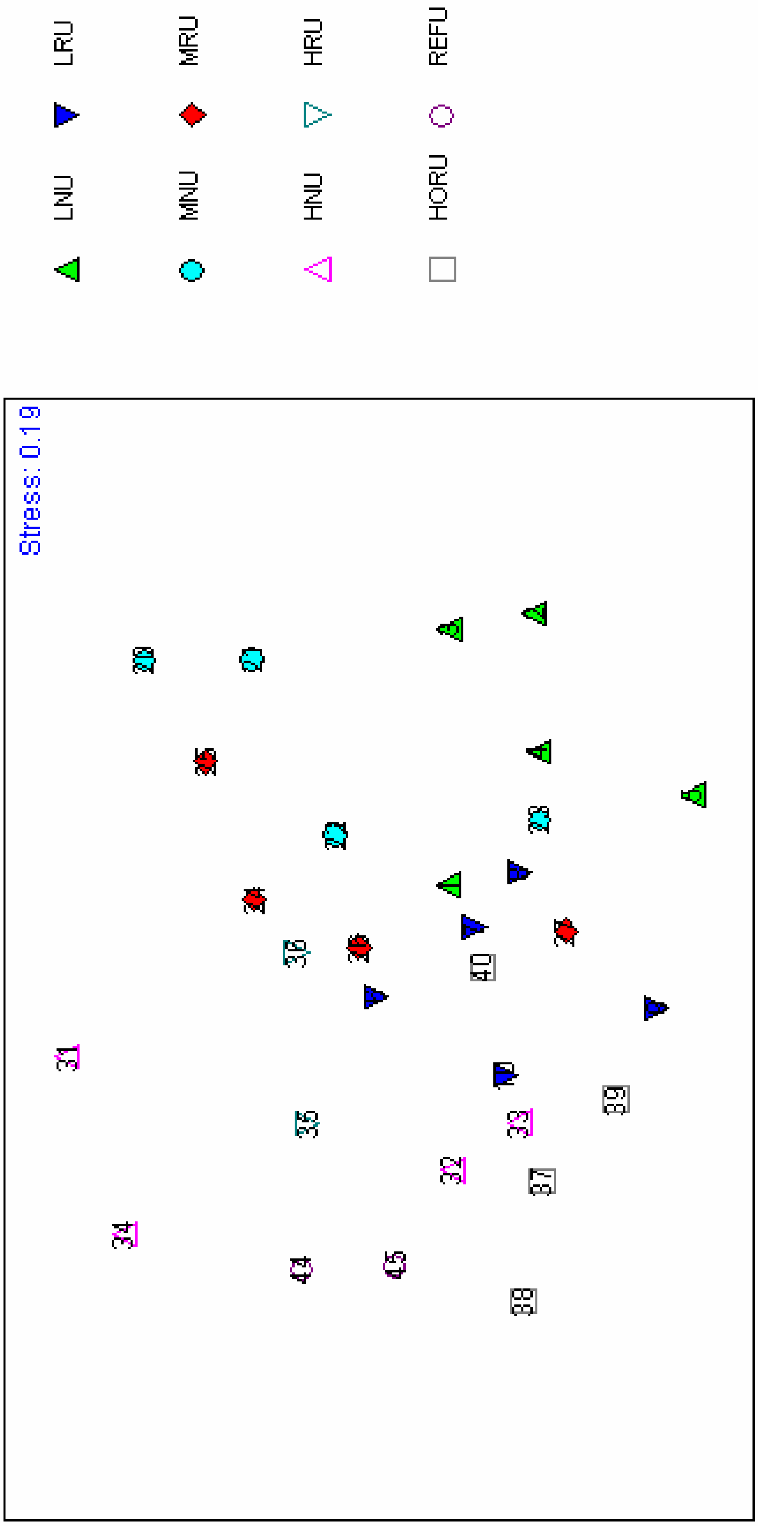

Figure 8. Non-metric multi-dimensional scaling ordination of floristic data for ironbark/gum woodland sites.
See Table 1 for descriptions of treatment codes.

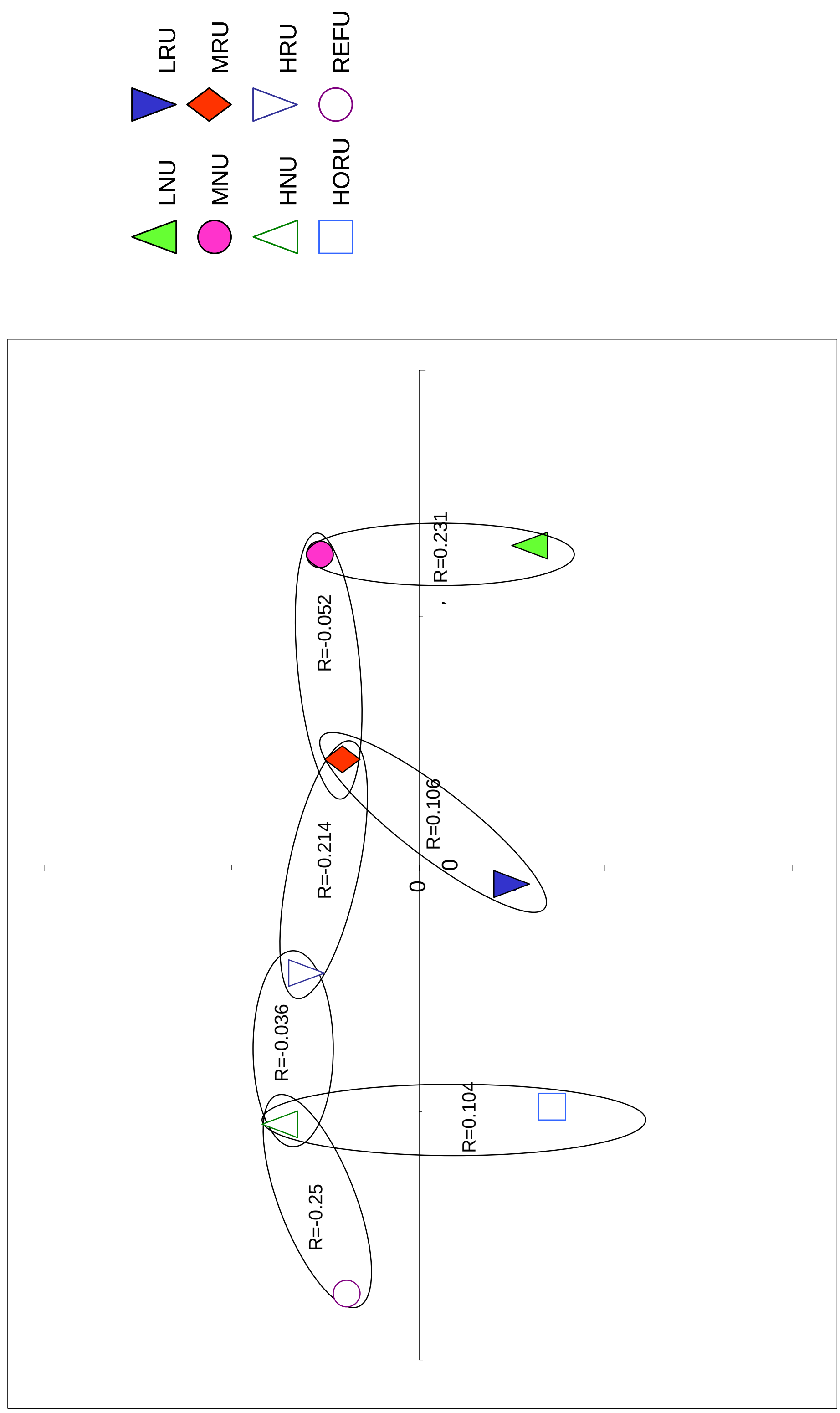

Figure 9. Centroids of all ironbark/gum woodland treatment combinations from nMDS ordination of floristic data.

Treatment combinations circled are not significantly different (ANOSIM; R<0.25). See Table 1 for descriptions of treatment codes.

An nMDS ordination of floristic data for box woodlands (Figure 10; stress = 0.17) shows a clear separation of low density sites with no regrowth from all other sites. The other four box woodland groups (LRL, MNL, HNL and REFL) are clustered together.

The centroid plot of box woodland treatments (Figure 11) with ANOSIM R values shows that low density, no regrowth treatment combination (LNL) was significantly different to other box woodland treatments. While medium density, no regrowth treatment combination (MNL) was not significantly different to any other group, there were significant differences in floristic composition between low density regrowth treatment combination (LRL), high density no regrowth treatment combination (HNL) and reference box woodlands (REFL).

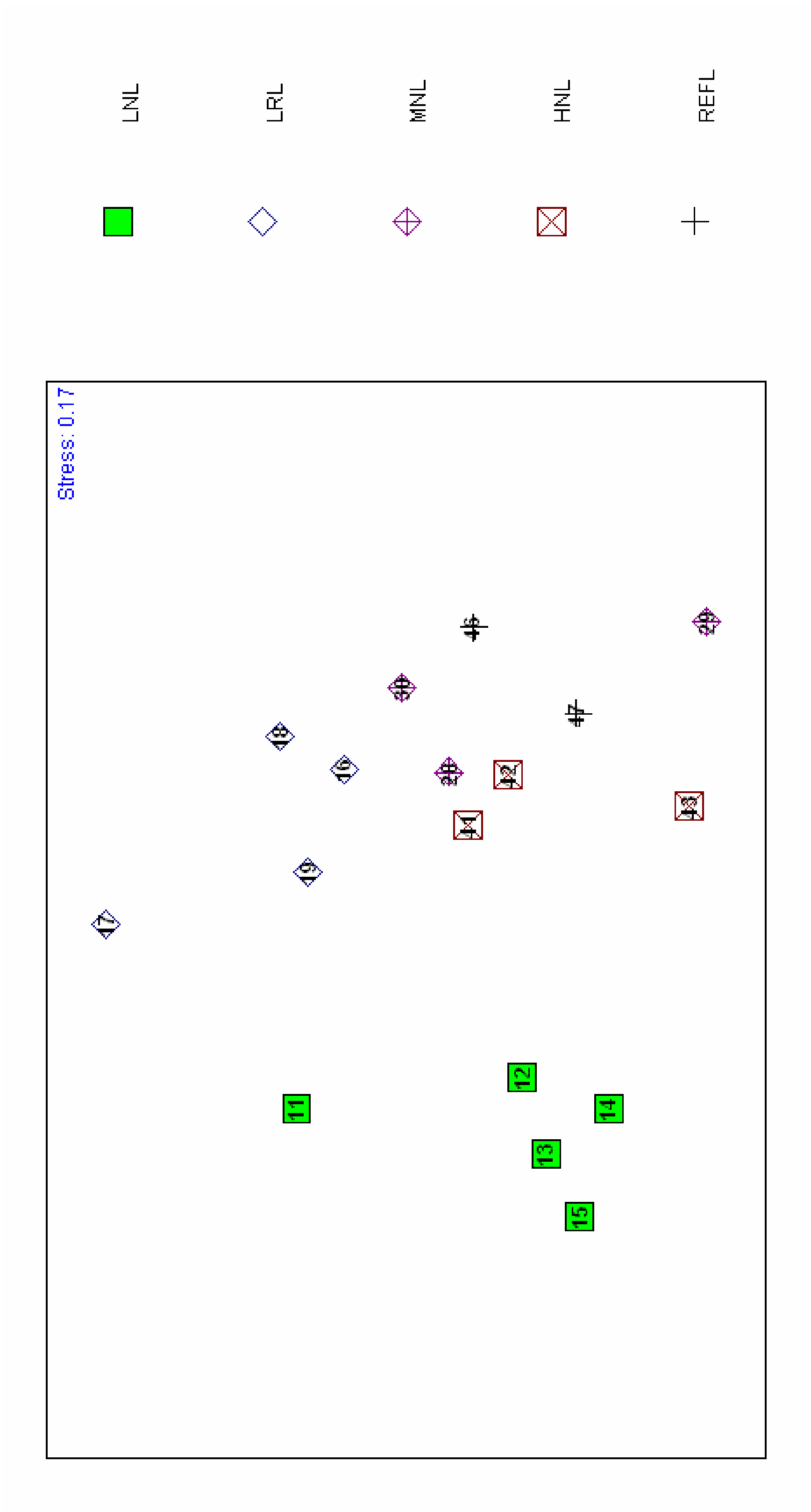

Figure 10. Non-metric multi-dimensional scaling ordination of floristic data for box woodland sites.
See Table 1 for descriptions of treatment codes.

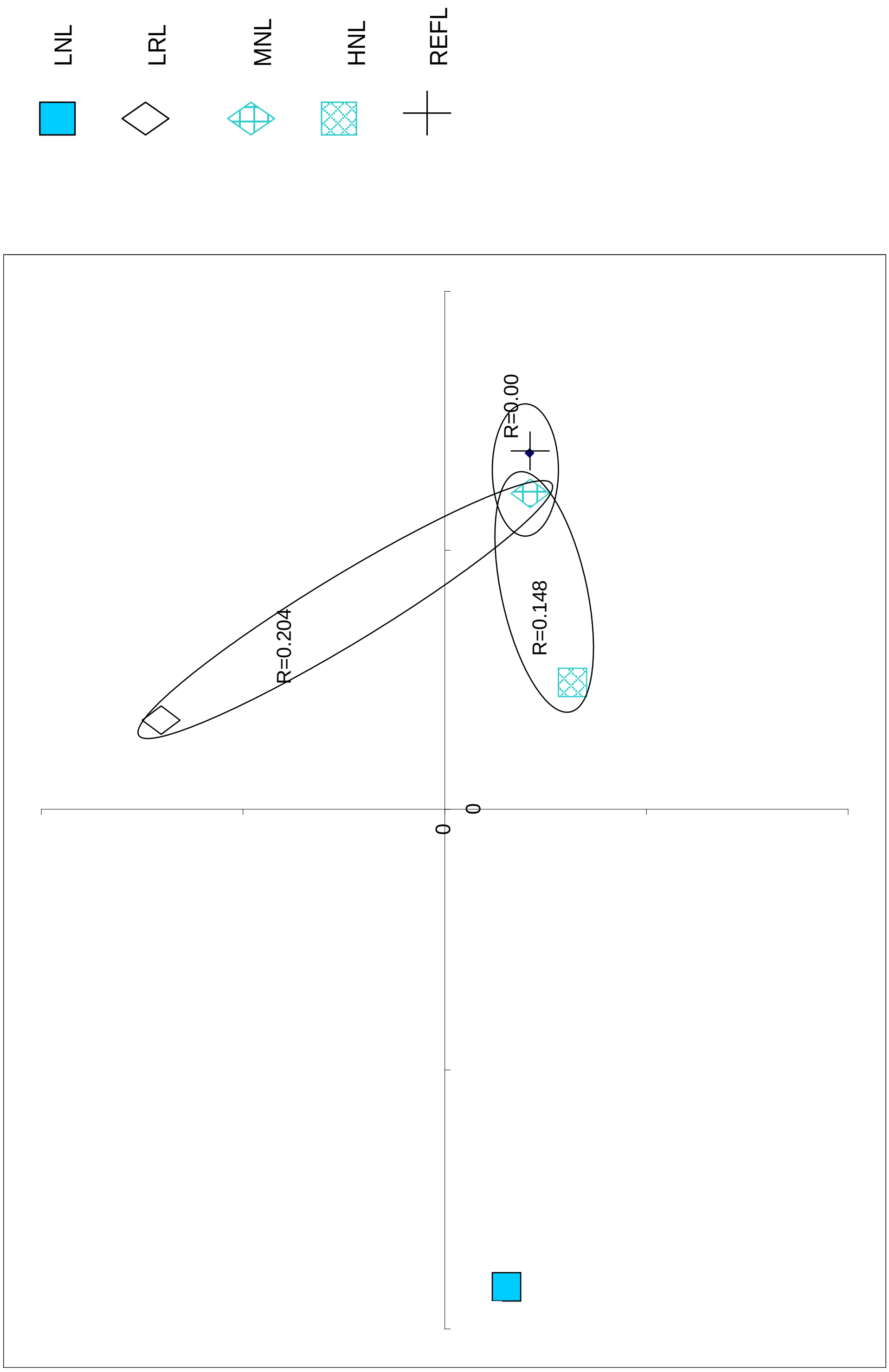

Figure 11. Centroids of all box woodland treatment combinations from nMDS ordination of floristic data.

Treatment combinations circled are not significantly different (ANOSIM; R <0.25). See Table 1 for descriptions of treatment codes.

4.4 Environmental Variation & Vegetation Patterns

Canonical Correspondence Analysis of floristic composition expressed as the closest fit (constrained) to the measured environment/landscape variables (Figure 12) shows a very similar gradient of treatments/sites to that of the nMDS ordination of floristic composition data (Figure 6).

Although there is a close similarity in the gradients shown in the two ordinations (relative positions of samples in figures are almost identical), only 16% of the variation in floristic composition is explained by the measured environmental variables in the two-dimensional ordination (Eigenvalues: axis 1 = 0.317; axis 2 = 0.214). Each arrow on the ordination from CCA represents an environmental axis (the axis is not bound by the ends of the arrow but extends from either end). The length of an arrow representing an environmental variable is equal to the rate of change of that variable across the ordination diagram, which thus allows an interpretation of the relative strength (or importance) of that variable. Descriptions of environmental variables used in CCA are shown in Table 3.

The corresponding gradient in floristic composition constrained to the environmental variables from CCA (from left to right of ordination) is from low density no regrowth sites (LNU and LNL), through low and medium density regrowth and no regrowth sites (LRU, LRL, MRU, MNU and MNL) and high density predominantly box woodland sites (HNL and REFL) although high density regrowth ironbark/gum woodland (HRU) is indistinguishable from these sites to high density ironbark/gum woodlands (HNU, HORU and REFU).

This major gradient in floristic composition is best explained, albeit weakly, by a range of environmental variables that correlate with the broad grouping of treatments. High grazing pressure, clearing and soil compaction distinguish open areas (low density no regrowth sites) from others. These sites were located well within this type, as indicted by a high proportion of low density vegetation in the immediate (100 ha) area, although at a larger scale (900 ha), the landscape matrix is predominantly medium density mature trees (Figure 12). Medium density no regrowth ironbark/gum woodlands (MNU) and high density no regrowth box woodlands (HNL) are distinguished by high weed disturbance, high perimeter:area ratio and predominantly medium density mature trees in the immediate (100 ha) area. These environmental variables also contribute to the separation of the box woodland reference sites

(REFL), which are also characterised by low patch size (area). Medium density no regrowth box woodlands (MNL) and regrowth ironbark/gum woodlands (MRU, HRU) are distinguished by high percentage course litter cover, high cover of mature trees and logs > 20 cm circumference, and lower grazing, clearing and logging pressure. High density ironbark/gum woodlands (HORU, HNU and REFL) are distinguished by high fine litter cover, and high proportion of high density vegetation at both the 100 ha and 900 ha scale, and to a lesser degree high cover of trees and high slope areas. These treatments are also characterised by low grazing pressure, clearing and soil compaction. LRU and LRL are reasonably variable in terms of the measured environmental variables.

Table 3. Description of environmental variables used in CCA.

Only those variables shown in Figure 12 are indicated. Cat. = categorical variable; Quant. = quantitative variable

Description	Label	Variable type
Grazing pressure (ranked 0-3) based on subjective site estimate and landowner knowledge	Grazing	Quant.
Soil compaction (ranked 0-3) based on subjective site estimate	Soil compaction	Quant.
Evidence of past clearing (ranked 0-3) based on subjective site estimate	Clearing	Quant.
Evidence of past logging (ranked 0-3) based on subjective site estimate	Logging	Quant.
Evidence and extent of weed invasion (ranked 0-3) based on subjective site estimate	Weeds	Quant.
Proportion of surrounding 100 ha comprised of low density mature trees (%)	Low 100ha	Quant.
Proportion of surrounding 100 ha comprised of medium density mature trees (%)	Med 100ha	Quant.
Proportion of surrounding 900 ha comprised of medium density mature trees (%)	Med 900ha	Quant.
Proportion of surrounding 100 ha comprised of high density mature trees (%)	High 100ha	Quant.
Proportion of surrounding 900 ha comprised of high density mature trees (%)	High 900ha	Quant.
Time since vegetation was first cleared (years)	TSFC	Quant.
Site slope (degrees)	Slope	Quant.
Patch perimeter (distance) (m)	Perimeter	Quant.
Patch perimeter:area ratio	P:A ratio	Quant.
Patch area (ha)	Area	Quant.
North-north-west aspect	NNW aspect	Cat.
West-south-west aspect	WSW aspect	Cat.
North-north-east aspect	NNE aspect	Cat.
Percentage bare ground cover within 500 m^2 plot (%)	Bareground	Quant.
Percentage cover of logs > 20cm circumference within 500 m^2 plot (%)	Logs > 20cm	Quant.
Percentage cover of trees 10-20m height within 500 m^2 plot (%)	%Trees10-20m	Quant.
Percentage cover of trees < 10 m height within 500 m^2 plot (%)	%Trees <10m	Quant.
Percentage cover of course litter within 500 m^2 plot (%)	%Course litter	Quant.
Percentage cover of fine litter within 500 m^2 plot (%)	%Fine litter	Quant.

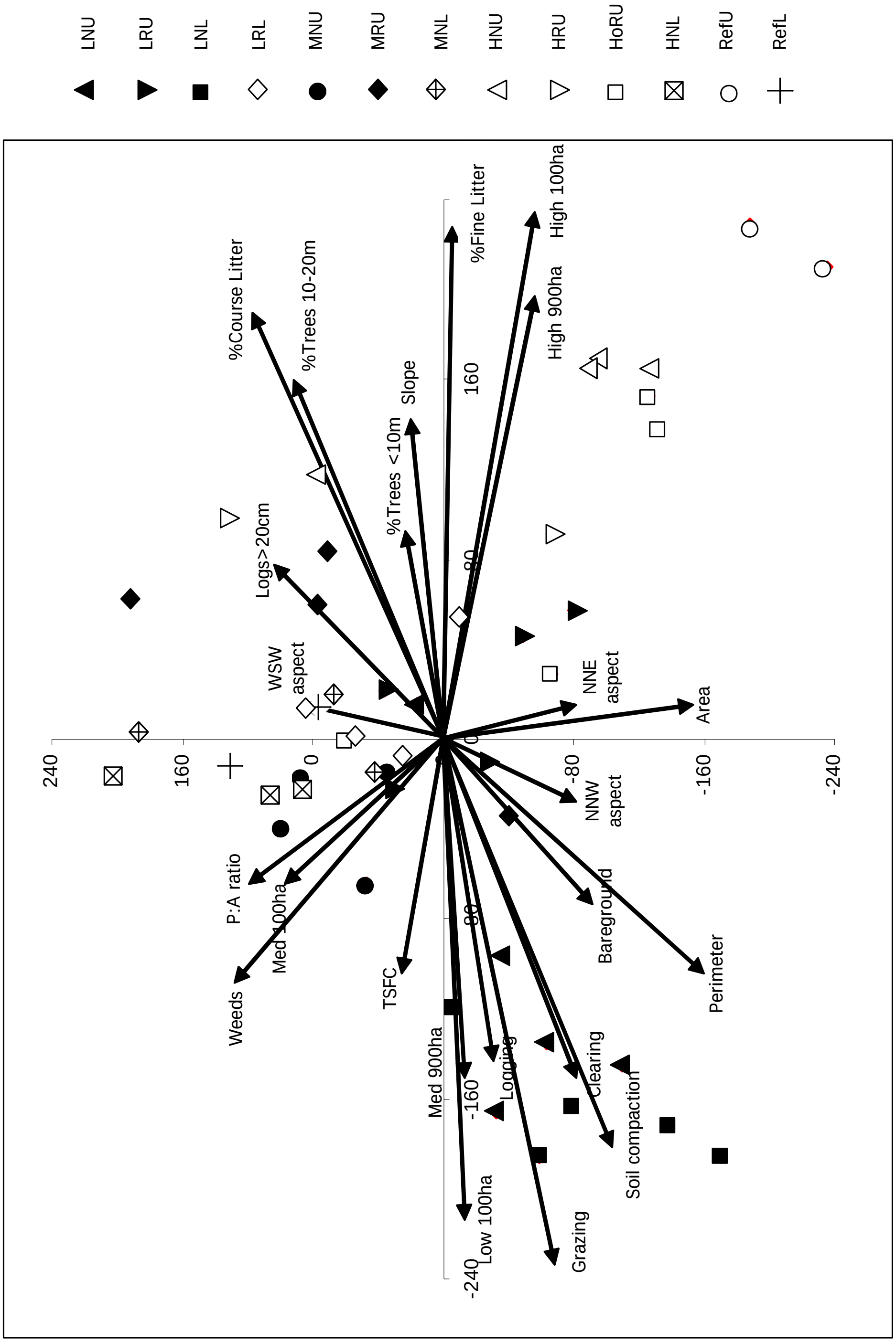

Figure 12. Canonical correspondence analysis (CCA) ordination of floristic composition and environmental variables.

See Table 1 for a description of treatment codes and Table 3 for description of environmental variables. Samples are represented by symbols, environmental variables are represented by arrows.

4.5 Vegetation Cover

4.5.1 Broad Patterns in Vegetation Cover

Non-metric multi-dimensional scaling (nMDS) ordination of cover data (Figure 13) also indicates a gradient of increasing mature tree density from left to right. The stress value for the 2-dimensional ordination (0.15) indicates that the ordination closely shows the relationship between sites. Sites classified *a priori* as low tree density and no regrowth (LNU and LNL) tended to cluster together and were well separated from sites with a high tree density for both woodland communities (HNU, HRU, HORU, HNL, REFU and REFL). Sites with a medium density of trees and no regrowth (MNU and MNL), and low density regrowth sites (LRU and LRL) tended to cluster in the middle of the ordination (see Appendix 10 for cover data).

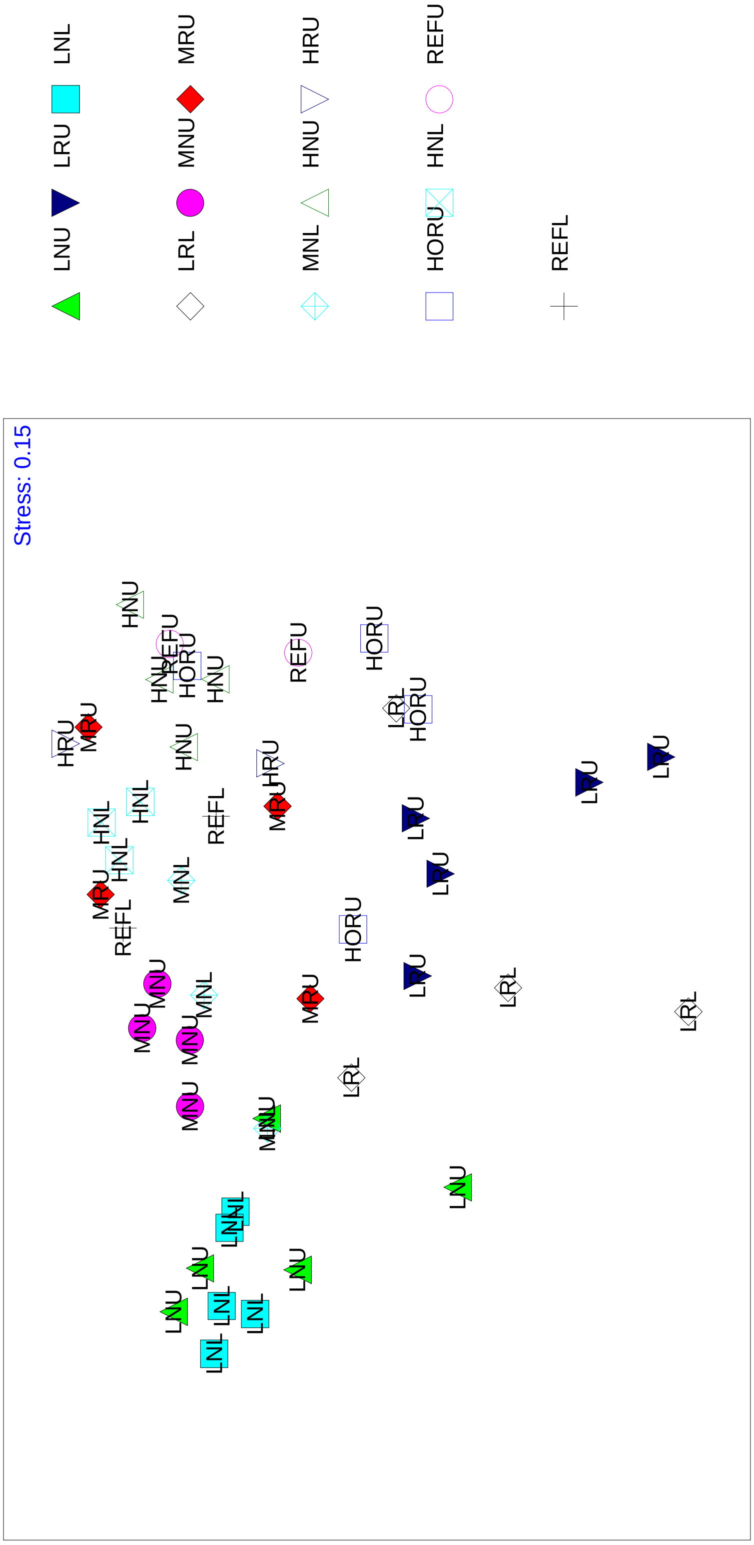

Figure 13. Non-metric multi-dimensional scaling ordination of strata percentage cover**.**
See Table 1 for descriptions of treatment codes.

4.5.2 Tree and Shrub Cover

The cover of trees 10-30 m in height was shown to decline as tree density ranged from high to low (as expected). High density no regrowth box woodland treatment (HNL) had a significantly higher ($p<0.05$) tree (>10 m) cover than all other groups except HNU (and possibly REFL and REFU). High density no regrowth ironbark gum woodland treatment (HNU) was significantly different ($p<0.05$) to low mature tree density treatments (LRL, LRU, LNU and LNL).

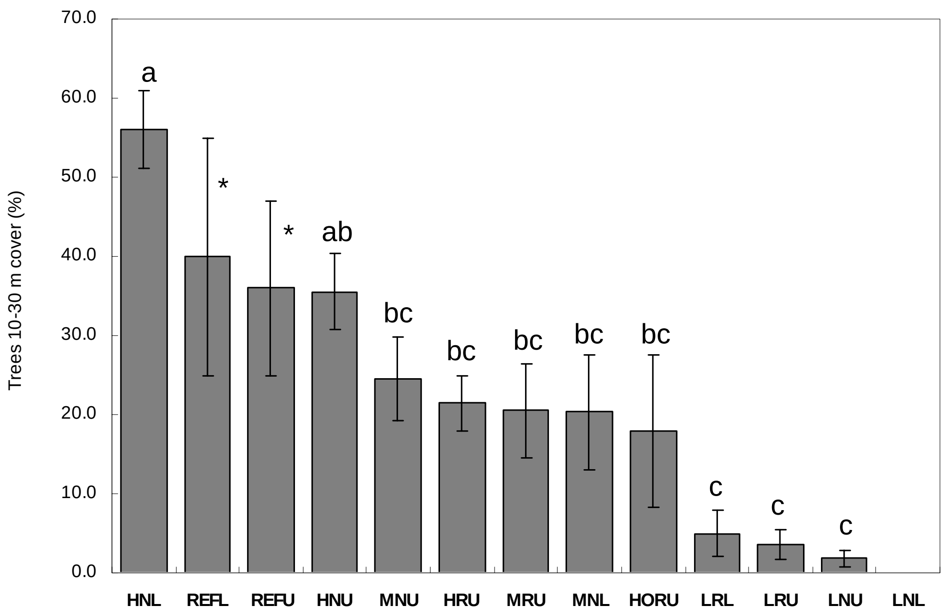

Figure 14. Mean percentage cover of overstorey trees (>10 m height) across treatments.
Treatments with same letter are not significantly different (Tukey's test, $p>0.05$); those indicated by an asterisk (*) were excluded due to unequal variances. Error bars are standard errors.

Mean total tree cover across treatments (Figure 15) shows a slightly different pattern to that of the previous cover graph (see Figure 14), as it includes mature trees and saplings. Low density no regrowth treatment combinations (LNU and LNL) had little tree cover and were significantly different ($p<0.05$) to most other treatments (with exception of MNU and MNL).

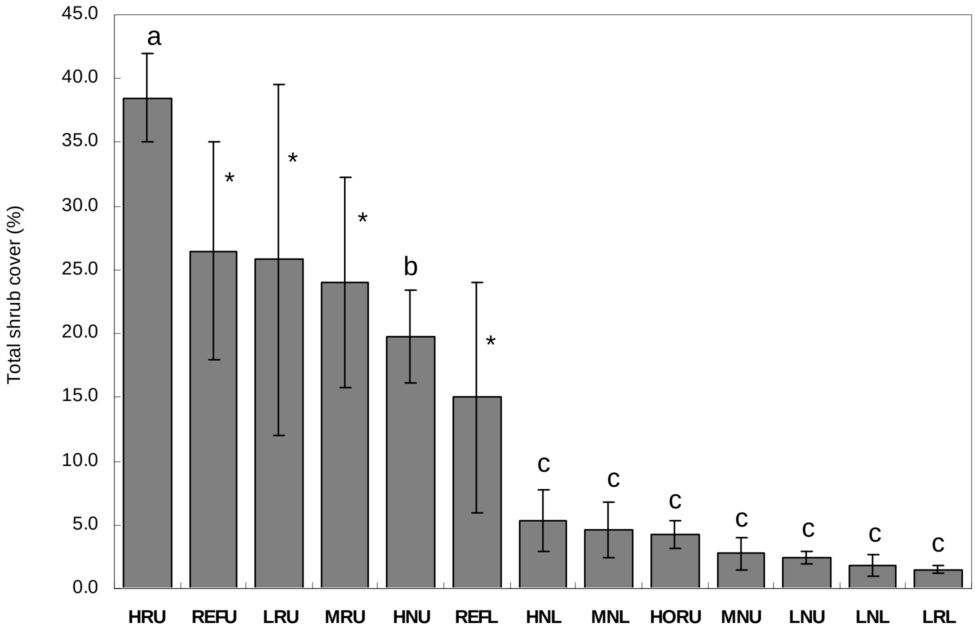

Figure 15. Mean percentage cover of all tree strata across treatments.
Treatments with the same letter are not significantly different (ANOVA, Tukey's test, p>0.05); treatments indicated by an asterisk (*) were excluded from analysis due to unequal variances; treatment indicated by a hash (#) were excluded from analysis due to small sample size. Error bars represent standard errors.

Mean total shrub cover across treatments (Figure 16) shows that shrub cover is generally higher for ironbark/gum woodland groups. High density regrowth ironbark/gum woodland (HRU) had a significantly higher total shrub cover ($p<0.05$) than all other groups (included in the analysis). High density no regrowth ironbark/gum woodland (HNU) had a significantly higher ($p<0.05$) total shrub cover than most box woodland treatment combinations.

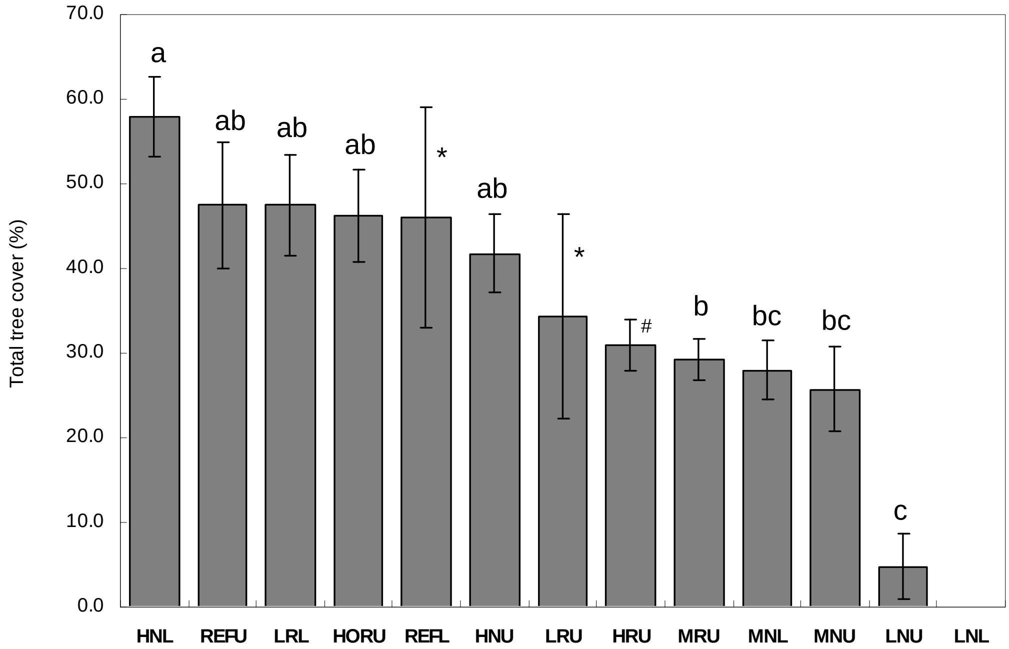

Figure 16. Mean percentage cover of shrubs across treatments

Treatments with the same letter are not significantly different (ANOVA, Tukey's test, p>0.05); treatments indicated by an asterisk (*) were excluded from analysis due to unequal variances. Values are means, error bars represent standard error.

4.5.3 Ground Cover

The change in mean grass cover across treatments is shown in Figure 17. The results show that low density no regrowth box woodland treatment (LNL) had a significantly higher grass cover than most other treatments ($p<0.05$), except for low density no regrowth ironbark/gum woodland treatment (LNU) and medium density no regrowth treatments (MNU and MNL).

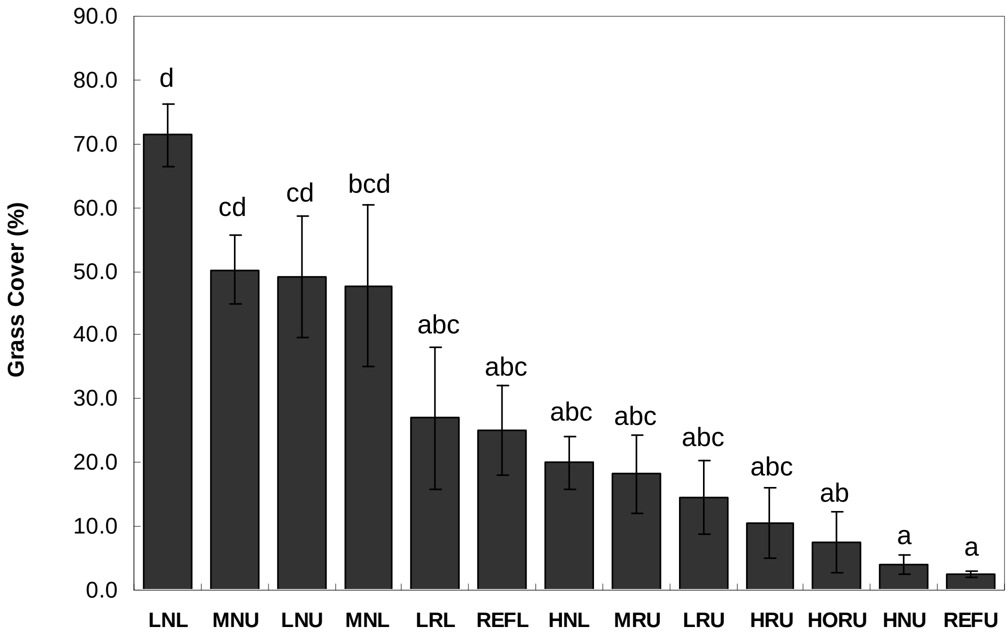

Figure 17. Mean percentage cover of grasses (%) across treatments
Treatments with the same letter are not significantly different (ANOVA; Tukey's test, $p>0.05$).
Values are means, error bars represent standard error.

Grass cover declines significantly with increasing total tree cover (Fig. 18; $p<0.001$) and increasing cover of trees 10-30m height (Fig. 19, $p<0.01$). Similarly, forb and herb cover decline with increasing total tree cover (Fig. 20; $p<0.001$).

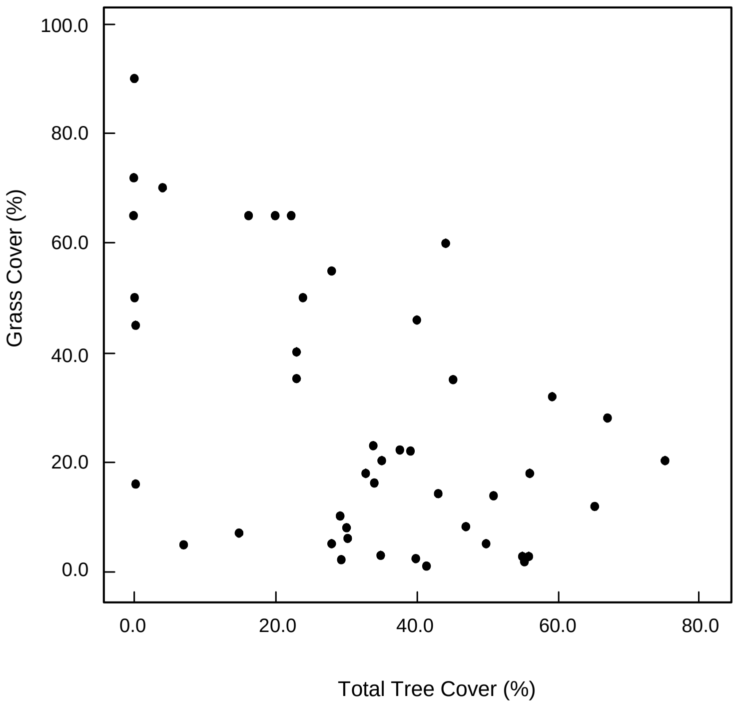

Figure 18. Scatterplot showing relationship between grass and total tree cover.
(Correlation: R = -0.59; p<0.001)

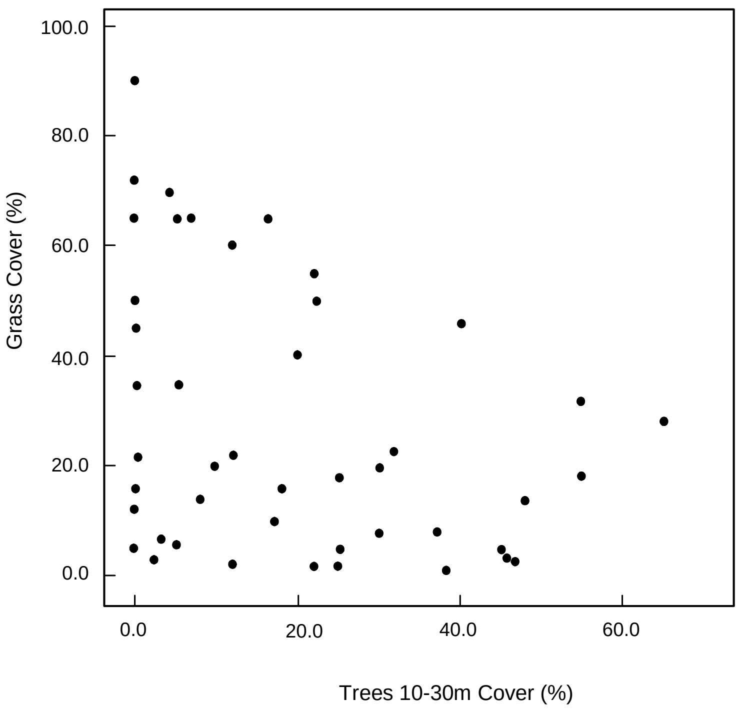

Figure 19. Scatterplot showing relationship between grass and tree 10-30m cover
(Correlation: R = -0.39; p<0.001)

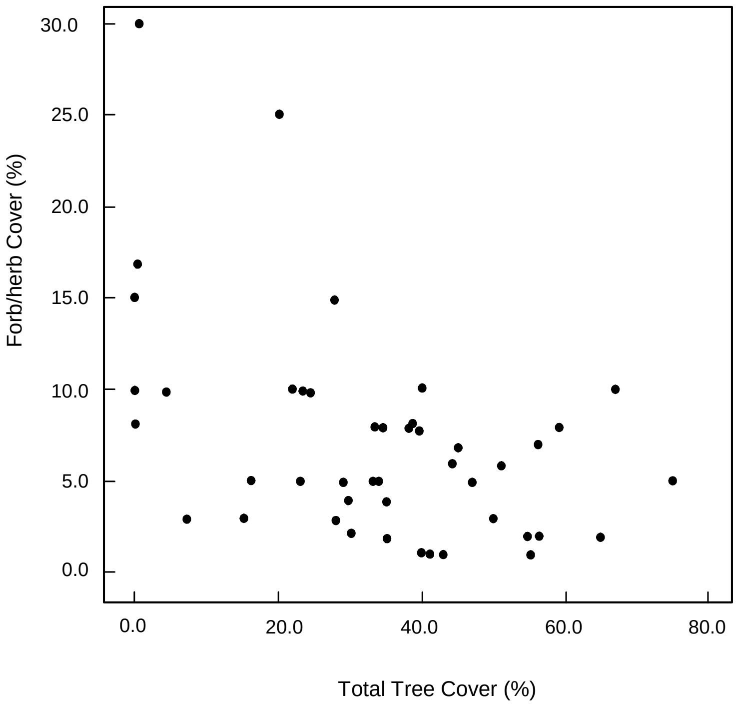

Figure 20. Scatterplot showing relationship between forb/herb cover and total tree cover
(Correlation: R = -0.48; p<0.001)

4.6 Species Richness

Comparison of mean total species richness (Figure 21) shows that there were no significant differences between groups. Similarly, mean native species richness was not significantly different across treatments (Figure 22). However, there were differences in mean exotic species richness between groups (Figure 23). Box woodland treatments tended to have a higher exotic richness component. Low density no regrowth box woodland treatment (LNL) had a significantly higher ($p<0.05$) exotic species richness than most ironbark/gum woodland treatments.

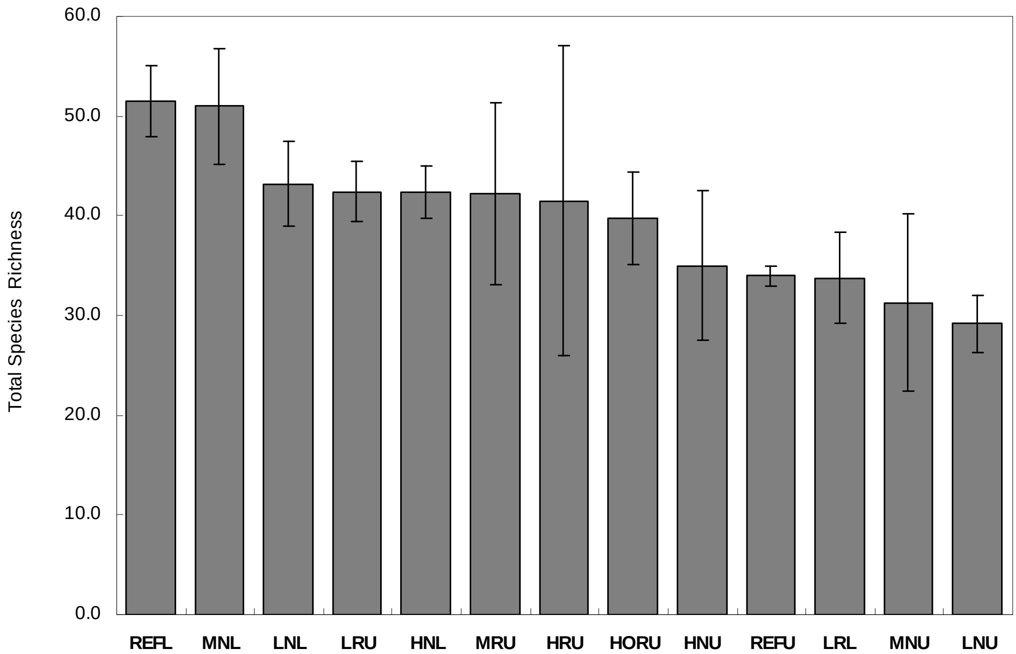

Figure 21. Mean total species richness across treatments.
Error bars are standard errors. Means are per 500 m^2 and are not significantly different (ANOVA, $p>0.05$)

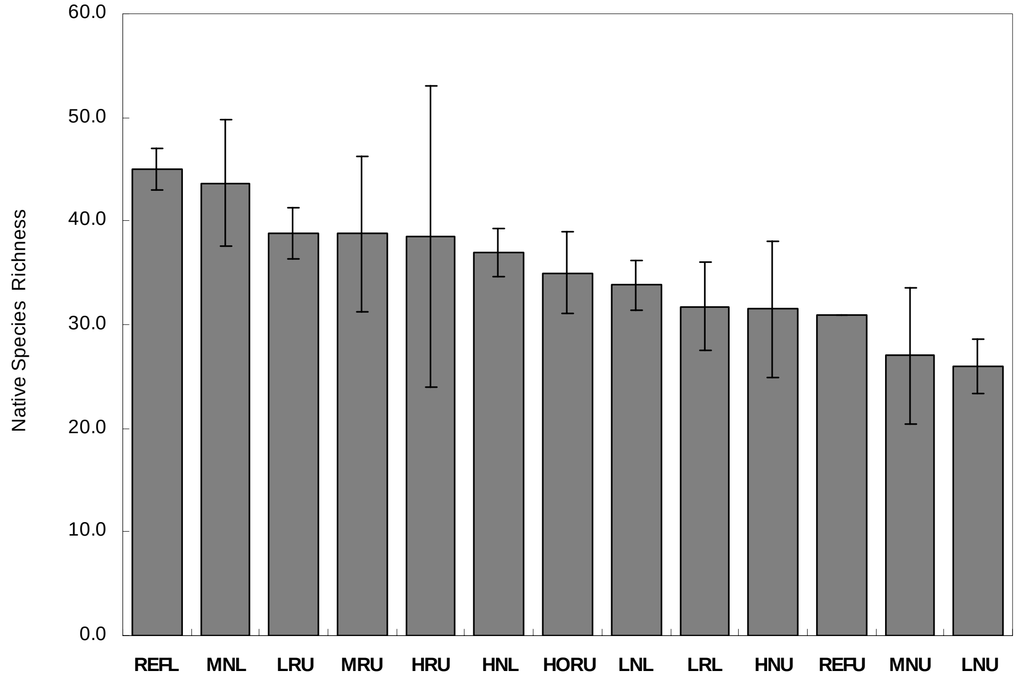

Figure 22. Mean native species richness across treatments.
Error bars are standard errors. Means are per 500 m^2 and are not significantly different (ANOVA, p>0.05)

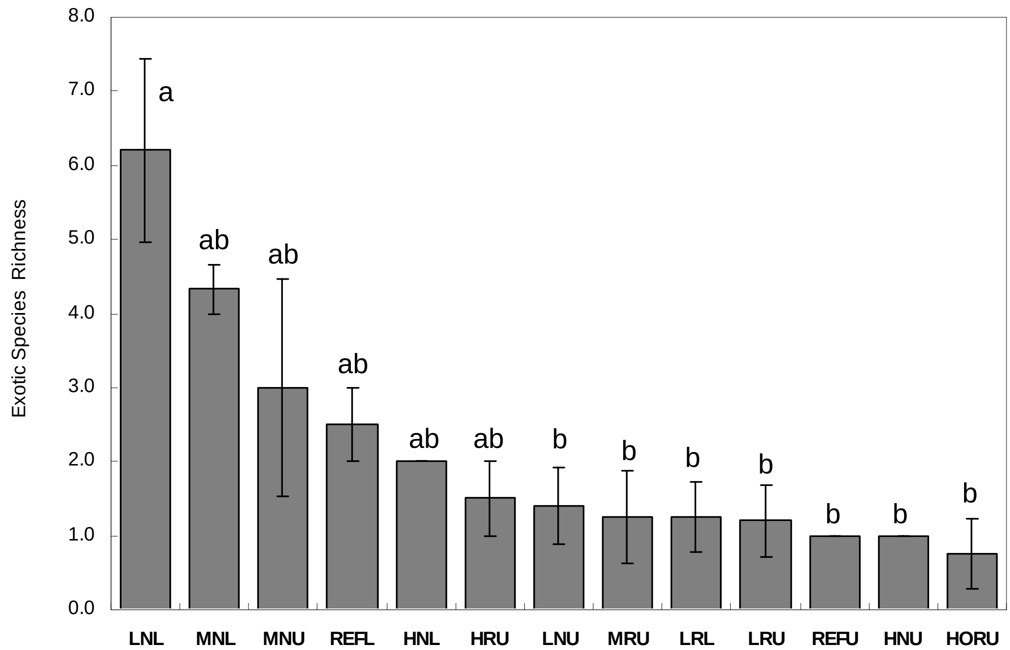

Figure 23. Mean exotic species richness across treatments.
Error bars are standard errors. Means are per 500 m^2 and means sharing same letter are not significantly different (ANOVA; Tukey's p>0.05)

4.7 Tree Recruitment

There was little tree recruitment in low density no regrowth treatments (LNU and LNL) (Figure 24), and no tree recruitment for medium density no regrowth ironbark/gum woodland (MNU). Low density no regrowth treatments (LNU and LNL) had significantly lower ($p<0.05$) total tree recruitment than regrowth treatment combinations (LRU, LRL, MRU, HRU) and reference ironbark/gum woodland (REFU). The mean recruitment of trees for each height category (> 3m, 1-3 m and <1 m) are shown in the following graphs.

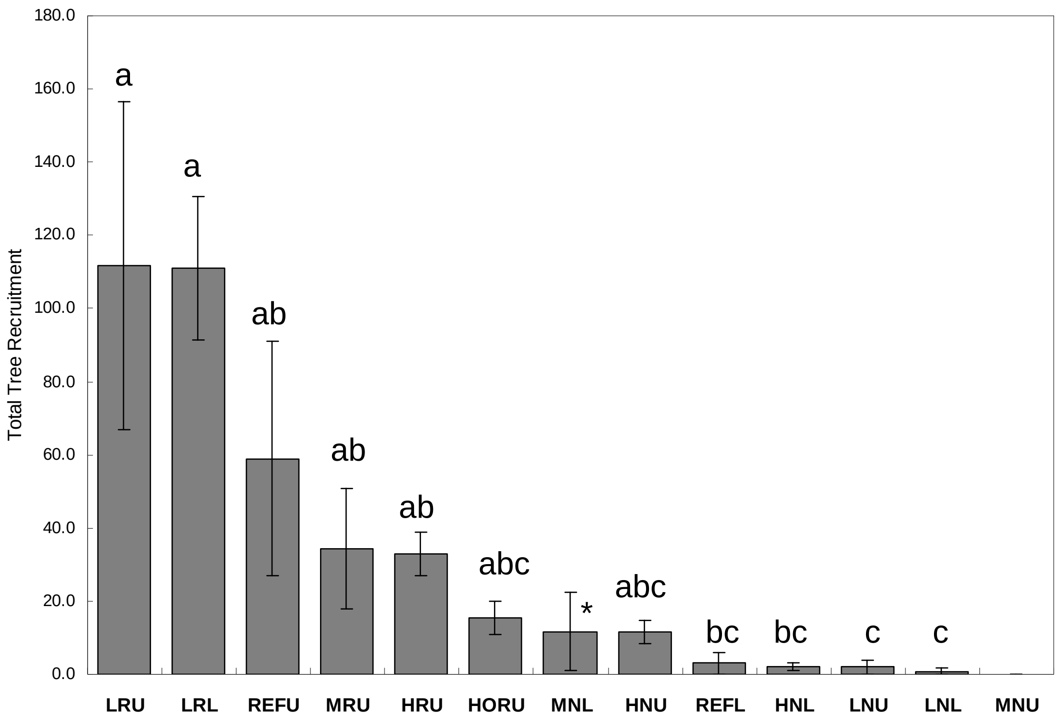

Figure 24. Mean total tree recruitment across treatments.
Error bars are standard errors. Means are per 500 m^2 and means sharing same letter are not significantly different (ANOVA; Tukey's $p>0.05$). Treatment indicated by an asterisk (*) was excluded from analysis due to unequal variances.

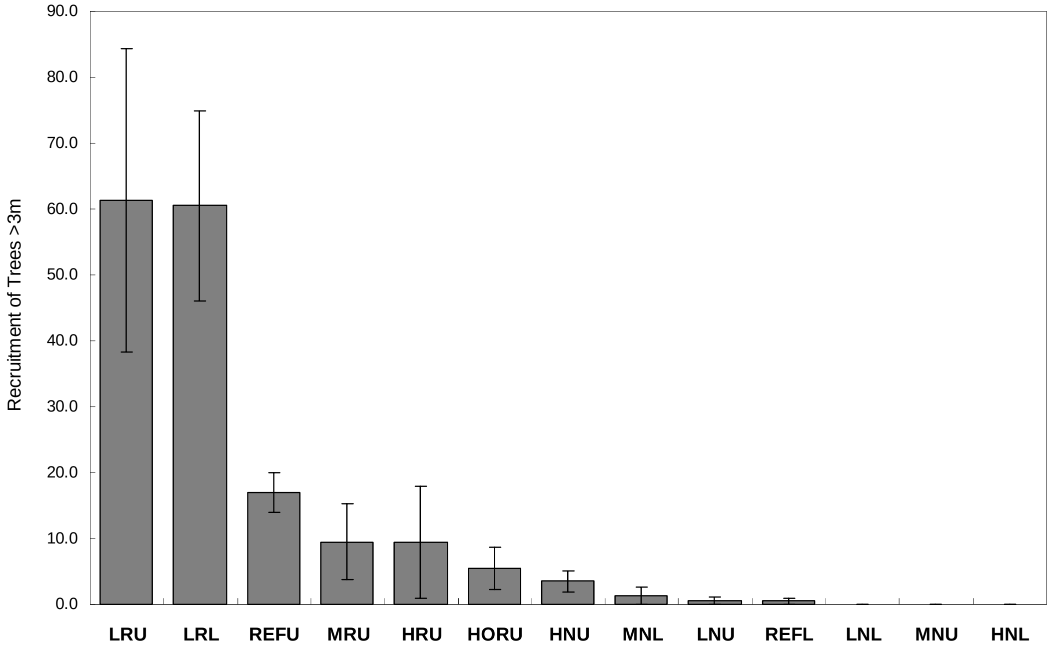

Figure 25. Mean recruitment of trees >3 m across treatments.
Error bars are standard errors. Means are per 500 m^2.

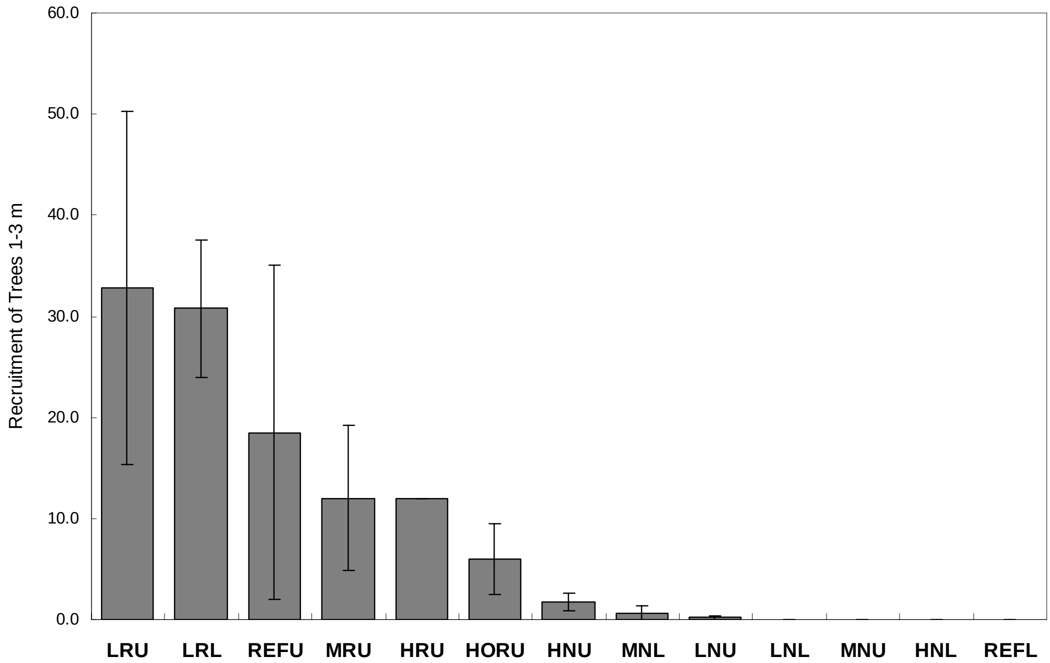

Figure 26. Mean recruitment of trees 1-3 m across treatments.
Error bars are standard errors. Means are per 500 m^2.

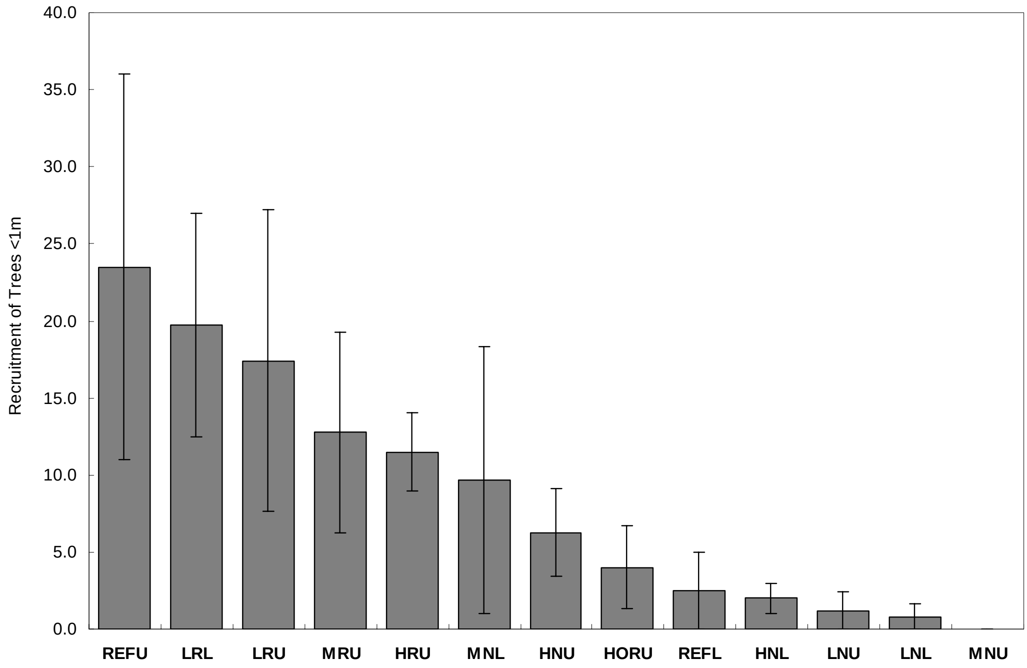

Figure 27. Mean recruitment of trees <1 m across treatments.
Error bars are standard errors. Means are per 500 m^2.

4.8 Habitat Complexity

The habitat complexity score for each treatment combination is shown in Figure 28. Scores are similar across most treatments, with the exception of low density no regrowth treatments (LNU and LNL). The mean habitat complexity scores ranged from 15.6 (LNU) to 30.5 (REFU).

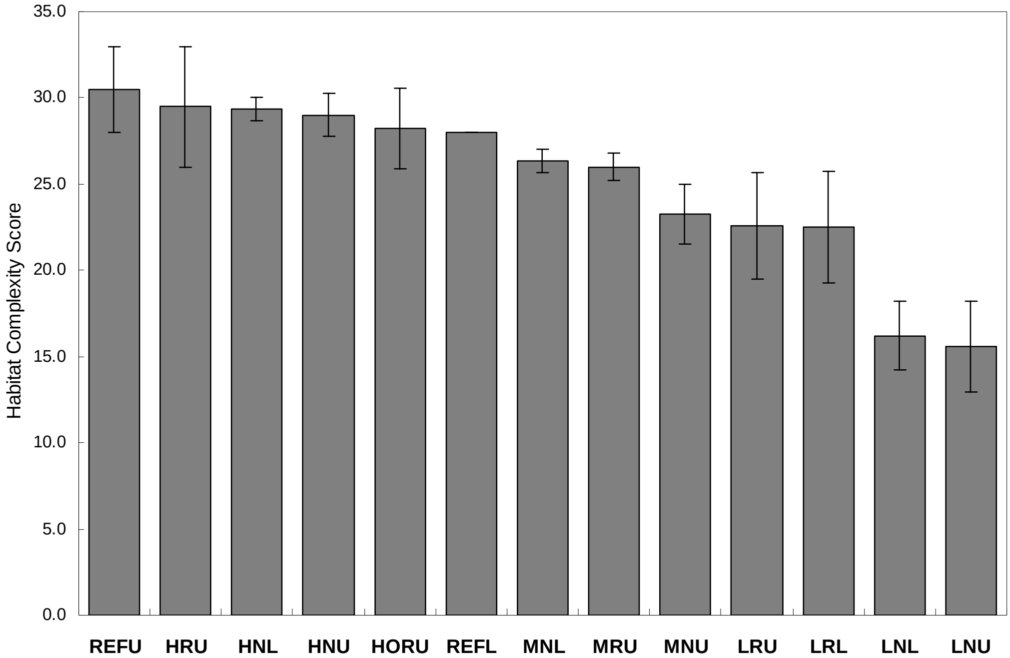

Figure 28. Mean Habitat Complexity Score across treatments.
Error bars are standard errors. Habitat Complexity Score is based on a modified scoring method (see Methods).

5. Discussion

5.1 General Patterns

The number of plant species recorded during the study is comparable to previous studies on woodland communities in Australia. Mean species richness has been reported as 20 species per 25 m² for woodlands in Western Australia (Yates and Hobbs 1997), 28 per 30 m² for grazed temperate grassy woodlands in New South Wales (McIntyre and Martin 2001) and 25 per 20 m² for grazed hill woodlands in south-eastern Queensland (Fensham 1998). In this study, mean species richness was 39 per 500 m² (equivalent to about 23 species per 20 m²) (Goodhew 2005) (Appendix 8).

The relationship between total richness and sampling area (Figure 4; Appendix 8), showed that plot area (500 m²) was sufficient to ensure sampling of the full complement of plant species present within sites.

This richness is despite a degree of taxonomic lumping (20% of taxa were identified to genus level, including 9 grass genera) in the present study. A further 4% of taxa were classified simply as 'unknown grass', and 3% classified into Family or higher taxonomic levels. The taxonomic lumping may be attributed to the lack of distinguishing features on some specimens, possibly due to sampling period and prolonged drought over the region. While the sampling period (September – November) was chosen to ensure sampling of spring flowering plants (mostly perennials), it may have missed summer flowering (and seeding) grasses (hence, difficulties in separating within grass genera). Continued drought in the region over the sampling period may have also meant that rain-response species were not sampled altogether.

Despite these potential limitations to the complement of plant species, distinct patterns in the vegetation were recovered. However, the vegetation patterns discussed here must be interpreted in the temporal context outlined.

5.2 Management History & Landscape Context

Since settlement one hundred years ago, the vegetation of the Traprock region has been influenced by a variety of anthropogenic disturbances such as sheep grazing, clearing, tree harvesting, and fertilizer application. Considerable social oral history is revealed by landowner responses to the site-specific questionnaire (see Appendix 2B for a summary of responses to the questionnaire), which provides insight into past management practices.

Original woodland communities were frequently subject to clearing by ring-barking more than 80 years ago. Many of the sites were re-cleared (almost continuously until about 30 years ago) to control woody regrowth. While less intense, clearing of regrowth vegetation remains a component of many management practices employed by landowners in the region.

As noted in the Introduction, tree density was used in this study as a surrogate for broader vegetation management practices (or *'management units'*) in relation to livestock grazing in the Traprock wool-producing region. Canonical correspondence analysis showed that the broad gradient in floristic composition was correlated with a range of environmental variables that included tree density (cover), grazing pressure, clearing, habitat structural components, patch size and landscape matrix.

It is evident that open paddock areas (LNU, LNL) and to a lesser degree, medium density tree areas (MNU, MNL), are subject to greater grazing pressure and more continued re-clearing than more densely wooded or regrowth areas: open areas and medium density patches currently sustain a heavier stocking rate than regrowth (regardless of mature tree density) or high mature tree density patches. Despite a nominal grazing intensity of 1-2 dse (dry sheep equivalent) per hectare (Queensland Murray Darling Committee 2004, and landowner survey) across the region, a more heterogeneous grazing pressure is indicated at the paddock scale.

Low density treatments were closely cropped areas that resemble 'grazing lawns' described by McIntyre *et al.* (2003) as areas that are repeatedly grazed. High density and regrowth treatments may be less heavily impacted due to steeper slopes (Appendix 1) and the presence of rocky outcrops. Study design constraints limited the availability of sites with a similar grazing intensity and therefore differences in floristic composition could possibly be a result of both grazing disturbance and changes to tree density. Hence, it is not possible to separate the effects of grazing (and other past history) from that of overstorey tree density. However, regardless of this, the differing management units (treatments) represented in the two major vegetation types occurring within the Traprock region show distinct and consistent patterns.

5.3 Broad Vegetation Patterns

5.3.1 Low density treatments

The results from this study suggest that vegetation management practices, particularly in the control of woody regrowth, significantly influence floristic patterns within the Traprock region of southern Queensland. The low density no regrowth treatments (LNU and LNL), regardless of pre-European vegetation type, were found to be similar to each other and consistently different to other management units in terms of floristic composition (Figures 5 and 6) and cover (Figure 12), and generally lower in tree and shrub recruitment (Figures 23, 24, 25 and 26) and habitat complexity (Figure 27). These areas are structurally very simple systems, with an absence of shrub and tree strata, and in the case of low density box woodland sites, have a generally significantly higher richness of weed species. Recruitment of shrub and tree species is very low, presumably due to high grazing pressure, continued re-clearing, or both.

The lack of any floristic difference between open areas (low density), indicates that in these highly modified systems, disturbances (such as sheep grazing and tree removal) can result in an indistinguishable floristic composition regardless of pre-clearing vegetation types. Similar results have been reported for other studies in Australia, where exogenous disturbances (such as livestock grazing) have been shown to be more important in determining floristic composition than biophysical factors such as lithology, slope and altitude (Pettit *et al.* 1995; Chilcott *et al.* 1997; McIntyre and Martin 2001; Clarke 2003; McIntyre *et al.* 2003).

The similarity in floristic composition between low tree density no regrowth treatments (LNU and LNL) may be a result of both sheep grazing and tree removal. Species that were common to both treatments included *Sisyrinchium micranthum* (exotic annual herb), *Gnaphalium sphaericum* (annual native herb) and *Crassula sieberana* (annual native herb). *Sisyrinchium micranthum* and *Gnaphalium sphaericum* are widespread herbaceous plants that are often weeds of disturbed areas within south-east Queensland, while *Crassula sieberana* prefers open dry places, which suggests some preference for areas without trees (Stanley and Ross 1983). McIntyre *et al.* (2003) reported that *Sisyrinchium micranthum* and *Crassula sieberana* respond positively to cattle grazing. In addition, all three species are annual plants and it has been suggested that annuals may be more tolerant of disturbance due to their fast growth rates and early and prolific seed set than perennial species, which tend to be comparatively slow growing and usually require years to reach reproductive maturity (Grimes 1974 cited Pettit *et al.* 1995).

Similarly, the density and richness of shrubs has been shown to be influenced by the combination of grazing and tree removal. Clarke (2003) reported that shrub species richness was enhanced where grazing intensity was low and canopy cover was present in the temperate woodlands of northern New South Wales. Pettit *et al.* (1995) reported that the exclusion of livestock grazing from woodlands in Western Australia resulted in a significantly higher number of native shrubs than adjacent grazed plots. In this study, the lower shrub cover in low density no regrowth treatments (LNU and LNL) may reflect a response to both grazing disturbance and the absence of trees.

Other species that were found predominantly (although not exclusively) in low density no regrowth treatments included *Paronychia brasiliana, Richardia stellaris, Vittadina sulcata, Solenogyne bellioides, Rumex* spp., *Euphobia* spp. These species are all low growing forbs, some exotic, that tend to dominate paddock areas (Harden 1991).

Low density no regrowth treatments had a significantly higher grass cover than high density treatments for the same vegetation type. The higher grass cover in open (low density) areas compared to high density treatments is consistent with results from a previous study by Walker *et al.* (1986) in the grassy woodlands of southern Queensland. Walker *et al.* (1986) report that when no trees were present herbage biomass (primarily native grasses) was significantly higher. However, somewhat surprisingly, while grass cover declined overall with increasing tree cover (Figures 17 and 18), there was no difference in grass cover between low and medium density no regrowth treatments for the same vegetation type (Figure 16). Grass cover was also up to half of that of low and medium density no regrowth treatments in treatments where there was woody regrowth or at high mature tree densities (Figure 16).

5.3.2 Medium and high density treatments

Differences in floristic composition between vegetation types were indicated for medium and high density treatments. The nMDS ordination of all sites showed that box woodland sites clustered together with a clear separation of high density box woodlands from high density ironbark/gum woodlands. The difference between vegetation types may be related to Wills' (1976) description of box and ironbark/gum woodlands within the study area. According to Wills (1976), box woodlands are described as consisting of a dense ground layer and sparse shrub layer, while ironbark/gum woodlands tend to have a well-developed shrub layer and sparse ground layer. Additionally, lower slopes are reported to have deeper texture contrast soils compared to ironbark/gum woodlands with shallower soils (Wills 1976).

The significant differences in floristic composition also suggest that vegetation types respond differently to management practices where trees are retained. For example, medium density no regrowth box woodland treatments (MNL) have a similar floristic composition to high density box woodland treatments (HNL and REFL), yet these results were not consistent for ironbark/gum woodlands where medium density no regrowth treatment combination (MNU) was more similar to low density no regrowth treatment combination (LNU). There were significant differences in shrub and grass cover between medium density (MNU) and high density (HNU) ironbark/gum woodlands not found for medium (MNL) and high density (HNL) box woodlands. It is suggested that a history of grazing and tree removal in medium density ironbark/gum woodlands results in the loss of native shrub species associated with high tree density communities (as reported previously, see Clarke 2003) and an increase in herbaceous cover (e.g. Walker *et al.* 1986). A vastly different understorey from that described by Wills (1976) develops as a result of these management practices in medium density ironbark/gum woodlands.

While there were strong floristic patterns associated with different tree densities, this was not reflected in significant differences in total species richness or native species richness between treatment combinations. This contrasts with a study by Prober and Thiele (1995) in the white box woodlands of New South Wales, where the combination of livestock grazing and tree clearing were shown to decrease native species richness. However, the results are consistent with a study by McIntyre and Martin (2001) where species richness within 30 m^2 plots was unaffected by lithology, slope, or tree density in the subtropical woodlands of southern Queensland.

The lack of significant differences in species richness between groups for this study could be a result of multiple factors, such as the spatial spread of sites resulting in a large variability of species richness, grazing disturbance, previous land management practices or heterogeneity within sample sites (structural attributes). For instance, logs may create opportunities for increased recruitment at sites where there is a low density of trees. It was observed that where logs have been retained on site after tree removal, a number of plants species are able to reach maturity possibly due to escaped herbivory by sheep. Logs have been reported to act as 'sinks' within landscapes allowing the accumulation of water, soil and litter as they flow or blow across the landscape (Tongway and Ludwig 1997; Yates *et al*. 2000). Therefore, the retention of logs may be important for maintaining species richness within open areas which are devoid of many other structural attributes. However, species richness as a measure of community change has been criticised as not adequately reflecting dynamic processes within communities, such as competition and succession (Steinberg and Geller 1994).

As previously noted, in high mature tree densities, grass cover was as low as half that of low and medium density no regrowth treatments in treatments (Figure 16). However, medium mature tree density box woodlands were not floristically dissimilar to high density and reference box woodlands, suggesting that maintaining a medium density of mature trees in box woodlands can potentially satisfy both production (in terms of grass cover) and biodiversity (floristic composition) goals in variegated and modified landscapes.

It has been suggested a landscape consisting of an overstorey of mature trees at a medium density may facilitate the establishment of native plant species (Chilcott *et al*. 1997), increase soil nutrients (Jackson and Ash 2001) and provide critical wildlife habitats (Lumsden and Bennett 2005). This finding in significant for land management practices in the box woodlands of Traprock region in that while there may only be a minimal increase in grass production in the very open areas, there is a significant decline in biodiversity value, at least in terms of floristic composition as determined here, and other ecosystem services provided by more structurally complex vegetation.

5.4 Patterns associated with woody regrowth

5.4.1 Understorey regrowth/tree recruitment

There were significant differences in floristic composition between regrowth and no regrowth treatment combinations for low density treatments. Low density woody regrowth treatments in both pre-European vegetation types (LRU and LRL) were significantly different in overall floristic composition to no regrowth sites in the same low density class (LNU and LNL) (Figure 7, Table 2). These regrowth sites were also similar to more wooded vegetation (Figure 7).

The difference in floristic composition may be attributed to tree or shrub establishment which can alter site conditions to suit the facilitation of species that prefer the higher soil nutrients and shade provided by overstorey species. It has been reported by Gibbs *et al.* (1999) that the establishment of trees during forest succession resulted in a change in the composition of dominant native grass species. Callaway (1995) provides a number of examples where the microenvironment provided by overstorey species favours plants that are excluded from open grassland habitat.

Species found predominantly in regrowth treatments, regardless of vegetation type or mature tree density class include *Desmodium varians, Desmodium brachypodum, Sida filiformis, Cymbopogon* spp., *Opercularia diphylla, Lomandra multiflora, Lomandra filiformis, Hibbertia obtusifolia, Glycine clandestina, Monotoca scoparia,* and *Melichrus urceolatus.* These are a mix of forb, grass and shrub understorey species.

Interestingly, low density regrowth box woodlands were indistinguishable from medium density no regrowth sites. This suggests that, at least for the box woodlands, allowing some degree of understorey regrowth in open areas can provide a community composition similar to more wooded vegetation lacking recruitment (no regrowth). A lack of available medium and high density sites containing woody regrowth within the study area prevents any conclusions regarding relationships between regrowth and no regrowth at higher tree densities within this vegetation type.

Within medium and high tree density classes, ironbark/gum woodlands with woody regrowth were not different floristically to no regrowth sites. As in the case for low and medium density box woodlands, low density regrowth ironbark/gum woodlands were indistinguishable from medium density regrowth sites and relatively similar to high density regrowth sites (Figure 9).

Tree recruitment, although variable, was higher in those areas where the understorey was allowed to regrow, and notably highest in low tree density treatments regardless of vegetation type (Figure 23). Recruitment in the 1-2 m and > 3m height classes was considerably higher in low tree density treatments than other treatments. Less separation of treatments was evident for tree recruitment in the < 1m height class, suggesting seed fall and germination may not be overly different across treatments compared to seedling survival.

The presence of mature trees may inhibit the excessive regrowth of eucalypts by causing mortality among saplings. Intra-specific competition for limited resources (nutrients, water, light etc.) has been long recognised as a mechanism of self-thinning within plants (e.g. Silvertown & Bullock 2003). Litterfall from existing vegetation has also been suggested as a possible mechanism controlling recruitment (Seiwa & Kikuzawa 1996). Yet other studies have suggested that herbivory and drought may be more important in affecting eucalypt seedling survival (e.g. Moles & Westoby 2004). Densities of eucalypt seedlings are probably insufficient in these landscapes to lead to seedling-seedling competition (Leishman 2001).

However, competition with existing vegetation may explain the considerably higher number of older juveniles in the >3 m height class in low mature tree density regrowth sites for both box woodlands and ironbark/gum woodlands than was recorded in the <1 m and, to a lessor extent, 1-3 m size classes. This suggests that a mass recruitment event in these sites some several years ago and the resultant competition for resources/space or other forms of interference (e.g. litterfall) by this cohort, reducing subsequent recruitment of overstorey species, may result in a fairly even-aged regenerating tree layer. It is possible that this may ultimately lead to the development of 'pole stage' regrowth (HORU) in the absence of natural thinning processes or management intervention.

5.4.2 Pole stage regrowth

High density pole stage regrowth ironbark/gum woodlands (HORU) were indistinguishable from high density no regrowth ironbark/gum woodlands in terms of floristic composition, although distinct from high density regrowth and reference ironbark/gum woodlands (Figures 6 and 7; Table 2). Tree and shrub recruitment shows a similar general pattern within the size (age) classes, although total recruitment was not significantly different across these types (Figure 23). It is likely that pole stage regrowth developed in areas where there was complete removal of the overstorey, preventing natural thinning processes and decreasing heterogeneity of recruitment ages within sites.

The notion that regrowth vegetation, particularly dense woodland regrowth, has little or no value for biodiversity seems widespread, although a number of authors have suggested that even dense regrowth can sustain a number of important ecological services (e.g. Kirkpatrick & Gilfedder 1999; Eldridge *et al.* 2003). William (2001) identifies a number of contributions to biodiversity of regrowth vegetation:

- regrowth may provide habitat for key elements of biodiversity that have been affected by vegetation clearance and fragmentation.

- regrowth may support and sustain biophysical and ecological processes. For example, as vegetation regrows, the structural, floristic and biological composition of areas change and may reduce the extreme nature of habitat fragmentation for resident species, and favourably modify local climate and environmental regimes (e.g. energy, radiation, light and exposure to extremes in ambient temperature).

- regrowth, although often less optimum, may reduce the impact of habitat fragmentation by providing corridors linking remnants or buffer areas between remnant and surrounding matrix (Saunders & Hobbs 1991).

However, our results would suggest that, while floristic composition of pole stage regrowth area is very different to open areas, they are also significantly different to high density and reference ironbark/gum woodlands. This would suggest that the pole stage regrowth is of less biodiversity value (at least in terms of floristic composition) than more natural wooded vegetation.

While natural vegetation thinning processes, such as tree dieback drought (Fensham 2000) or fire (Eldridge *et al.* 2003) may operate in some areas to reduce woody regrowth of overstorey species, a more active approach to controlling regrowth may be required in the Traprock region. It is particularly the case where self-thinning processes in this landscape are operating at longer timescales than can be easily managed by landowners (we note that some pole stage regrowth areas are greater than eighty years old).

However, we note that some caution is necessary as to how thinning of pole stage regrowth should proceed. Past practices have often involved considerable soil disturbance that can interfere with many ecological processes. Selective thinning through tree removal may not be economically viable in large vegetated areas, such as found in the Traprock region The careful and selective use of fire as a management tool is one option that would need further investigation in the region.

A second important question becomes to what level should pole stage regrowth be thinned? Watson & Reid (2001) suggest actively thinning areas of extreme woody eucalypt regrowth, although they point out that maintaining at least 10% of the regrowth provides greater increases in groundcover (grass herbage) than complete removal of overstorey. Clearly, this target may not necessarily be appropriate for the ironbark/gum woodlands of the Traprock region. Further research is needed to establish optimum densities of regrowth vegetation in relation to both re-establishment of ecological functioning (biodiversity) and production benefits.

5.5 Conclusions

A long history of pastoral management practices (including grazing pressure) in the region has resulted in open (paddock) areas which develop a unique composition of plant species and which are indistinguishable in terms of pre-European vegetation types. It is suggested that the combination of grazing and continued clearing results in the replacement of perennial native plant species with grazing specialist plant species, particularly annual native and exotic herbs and grasses.

At medium and high mature tree density classes, differences in floristic composition between vegetation types are evident. This finding is consonant with general ecological understanding that underlying environmental variation (e.g. soil type) determines local patterns of vegetation (Beadle 1981).

The two vegetation types sampled in this study respond differently to management practices which may be a result of natural environmental differences. For instance, ironbark/gum woodlands were reported to originally support a more shrubby understorey compared to box woodlands. The loss of shrub species either through livestock grazing on seedlings or increased grass competition as a result of tree removal (e.g. Clarke 2002), may have altered original ironbark/gum woodland understoreys significantly, while the natural grassy understorey of box woodlands may not have been considerably altered by grazing and tree removal (however these practices do result in a change in understorey composition when both tree removal and grazing intensity are increased (e.g. LNL)).

While some differences in plant species richness (and components) were evident, richness did not generally reflect patterns exhibited by floristic composition. We recommend caution when using plant species richness as an indicator of biodiversity patterns as it does not necessarily provide the complete story.

At low tree densities the presence of regrowth results in a very different plant species composition to areas lacking regrowth in the understorey, with low density regrowth sites more similar to more wooded (higher density) vegetation. These results suggest that native overstorey species may play a part in facilitating the establishment of different plant species as suggested by Gibbs *et al.* (1999).

While differences in grass cover (a crude measure of pasture production) exist between low and high density treatments, there was no difference in grass cover between low and medium density no regrowth treatments. This suggests that the effect of mature trees on grass cover may be marginal or absent at lower tree densities.

5.6 Limitations and Future Directions

As noted previously, this study was undertaken over one growing season during an extended period of drought, therefore the number of plant species recorded may not represent the true complement of species present within the region and longer-term studies combined with soil seed bank experiments will help to ensure the full suite of plant species are recognised. In addition, grazing intensity was possibly a confounding variable and owing to the lack of suitable reference sites, it is unclear how sheep grazing or other disturbances have influenced floristic composition. While this study aimed to determine the influence of vegetation management practices on woodland communities, examination of plant composition and cover within established exclosures will help to separate the effects of livestock grazing from that of tree density.

Future studies may examine the effects of competition, fire and soil disturbance on species composition and diversity which may lead to an increased understanding of the changes undergone by vegetation in response to management practices. For example, Belsky (1992) found that plant species within the Serengeti National Park have differential responses to disturbances and competition with both negative and positive responses recorded. Additionally, fire could be necessary for the reproductive success of some plant species (Belsky 1992; Clarke 2002).

6. Implications for Management

Lunt (2005) summarized a number of 'best practice' principles for managing grazing stock to conserve grassy ecosystems in Australia, of which the following may be of relevance in the Traprock region:

1. grazing managers should aim to promote a spatially variable, structurally complex understorey (grassland) structure – uniformly short, closely cropped 'grazing lawns' are undesirable;

2. continuous grazing should be avoided wherever possible – intermittent grazing, interspersed with rest periods, is preferred;

3. within seasonal or annual periods, the longer the rest period the better – intensive grazing over short periods interspersed by lengthy rest periods is commonly advocated – although Lunt (2005) points out that there is little scientific data to support this practice;

4. sites should be rested when desired native plants are flowering and setting seed in spring and early summer.

While these principles would seem rather intuitive, there are numerous economic pressures on land managers that often prevent all these principles to be enacted at the property level at any specific time. In addition to these principles being applied where feasible, the results of this study suggest a number of management implications.

6.1 Tree Cover

The results from this study show that vegetation management practices within the Traprock region have influenced the floristic composition and plant cover of woodland communities. The differences in floristic composition may have resulted from the interaction of multiple factors (such as livestock grazing and past management practices) and possibly from changes associated with tree removal. However, the strong floristic patterns associated with mature tree density classes are not reflected in significant differences in total species richness between groups. These results seem to suggest that the heterogenous nature of vegetation management practices in the Traprock region maintains different communities of plant species. Similarly, Lunt (1997) found that in the grassy forest remnants of south-eastern Australia maintaining different management regimes results in different suites of plant species. Each management unit has value in contributing to regional plant diversity, and landowners may consider maintaining all management units on their property.

Recommendation 1: Maintain a diversity of management practices at the property scale

In the grazed landscapes of the Traprock region maintaining high density, regrowth, medium density and low density patches will be important for the persistence of native species associated with each of these communities. However, an increase in the extent of open areas (either through tree death, lack of tree recruitment or by removing/clearing trees) may result in the loss of many native plant species found in more 'natural' communities. Differences in vegetation types as a result of natural environmental variation (seen where trees are retained at a medium and high tree densities) may no longer exist, resulting in floristically similar plant communities. This is highlighted by the similarity in composition between low tree density communities of different vegetation types.

In addition, it is important to consider original vegetation types when managing vegetation for livestock grazing. Applying one management regime to both ironbark/gum woodlands and box woodlands may be to the detriment of native plant species diversity. For example, while a medium density of trees in box woodlands has a similar floristic composition to high density and reference woodlands, this result is not consistent for ironbark/gum woodlands where a medium density of trees has a significantly different floristic composition to high density and reference ironbark/gum woodlands.

The impact of tree removal and livestock grazing on understorey plant composition has been discussed previously. In this instance the difference in medium density and high density ironbark/gum woodlands may be due to the loss of shrub species and the increase in herbaceous cover. While the same grazing and clearing disturbances may take place in box woodland communities, the effect may be less severe in box woodland communities as the understorey was probably originally largely herbaceous.

Recommendation 2: Consider natural environmental variation when managing vegetation for livestock grazing

6.1.1 Box Woodlands

In grassy box woodlands the results suggest that a medium density of trees may be adequate to ensure that the floristic composition of this vegetation type is maintained to that of 'natural' high density areas. The higher grass cover provided by medium density no regrowth box woodlands also suggests that this density may be compatible with grass production within the region. As noted, trees at a medium density may facilitate the establishment of native plant species (Chilcott *et al.* 1997), increase soil nutrients (Jackson and Ash 2001) and provide critical wildlife habitats (Lumsden and Bennett 2005), while maintaining a similar production value to open areas.

While further research may be needed to determine the dry matter digestibility and palatability of grass species associated with a medium density of trees, this provides a baseline for increasing plant diversity in a production landscape. This management option may also be applied to ironbark/gum woodlands as there are benefits associated with retaining trees for overall biodiversity, even though the composition of plant species in these communities was more similar to open (low density) areas.

6.1.2 Ironbark/gum Woodlands

For the ironbark/gum woodlands the results indicate that a high density of trees is necessary to maintain a distinct floristic composition. Landowners should ensure that there is a good representation of high tree density patches within their property. These areas may be important refuges for plant species that may not be able to establish successfully in areas with a higher grazing disturbance or lower tree cover. For example, the shrub *Melichrus urceolatus* was more common to areas with a higher cover of trees and lower grazing disturbance. The low abundance of many shrub species across the landscape highlights the importance of retaining high mature tree density areas for biodiversity conservation.

The lack of suitable 'undisturbed' reference sites in the region made it difficult to determine if the floristic composition of high density ironbark/gum and box woodland communities resemble that of 'natural' communities. It is unclear how previous disturbances may have influenced the composition of woodland communities. For instance, species sensitive or intolerant to disturbances may have already been lost and the remaining community now represented by tolerant species (McIntyre 1994). Therefore, maintaining representative areas of both vegetation types will help to provide a 'benchmark' for which to compare changes in communities over time.

Recommendation 4: As a region, maintain areas representative of 'natural' communities

6.1.3 Woody Regrowth

The regeneration of native woody species in the landscape has a similar effect on floristic composition regardless of mature tree densities. In particular, the regrowth of woody species in open areas significantly changes the composition of plants in the understorey. Possibly after clearing, the re-establishment of trees and shrubs facilitates the establishment of plant species that may prefer the increased soil nutrients associated with trees, and changes to the microclimate of these areas (e.g. through the provision of shade). Allowing some open areas to regenerate will provide a number of biodiversity benefits as the results suggest that the floristic composition of woody regrowth areas overtime may tend towards high density sites. Chilcott *et al.* (1997) reported similar results where pasture and soil mesofauna assemblages associated with an early (1 and 6-year old plantings) stage of reforestation were tending towards mature forest communities. The regrowth of eucalypts will be important for the future conservation of woodlands, especially as older paddock trees die or where there is little recruitment in medium or high density areas.

Recommendation 5: Allow the natural regeneration of native woody species

6.2 Options for Vegetation of the Traprock Region

Figure 29 provides a diagrammatic representation of the possible options for the future mix of vegetation conditions for the two major vegetation types, ironbark/gum woodlands and box woodlands occurring in the Traprock region to maximize biodiversity.

While there are arguments for maintaining a wide diversity of vegetation elements in any one landscape in order to maximize biodiversity (e.g. Lunt 1997), this study would suggest that open areas, areas with only scattered trees and pole stage regrowth may not be as preferable as medium to high density vegetation elements (Figure 29, types (ii) and (iii)). Clearly, large areas lacking understorey and some regeneration of overstorey tree species in a landscape are undesirable from a biodiversity perspective. Equally, large areas of heavily wooded (high density) vegetation are not desirable from a production (grass cover) perspective. We suggest that, while all elements should be maintained in the landscape, parts of the Traprock landscape should be either thinned or allowed to regenerate to achieve medium density woodlands containing either regenerating understorey (for biodiversity) or grassy understorey (for production) (Figure 29).

However, a remaining question is, how much of the various vegetation elements (regrowth, open areas, medium and higher density areas) should be contained within any landscape (sub-catchment/catchment)?

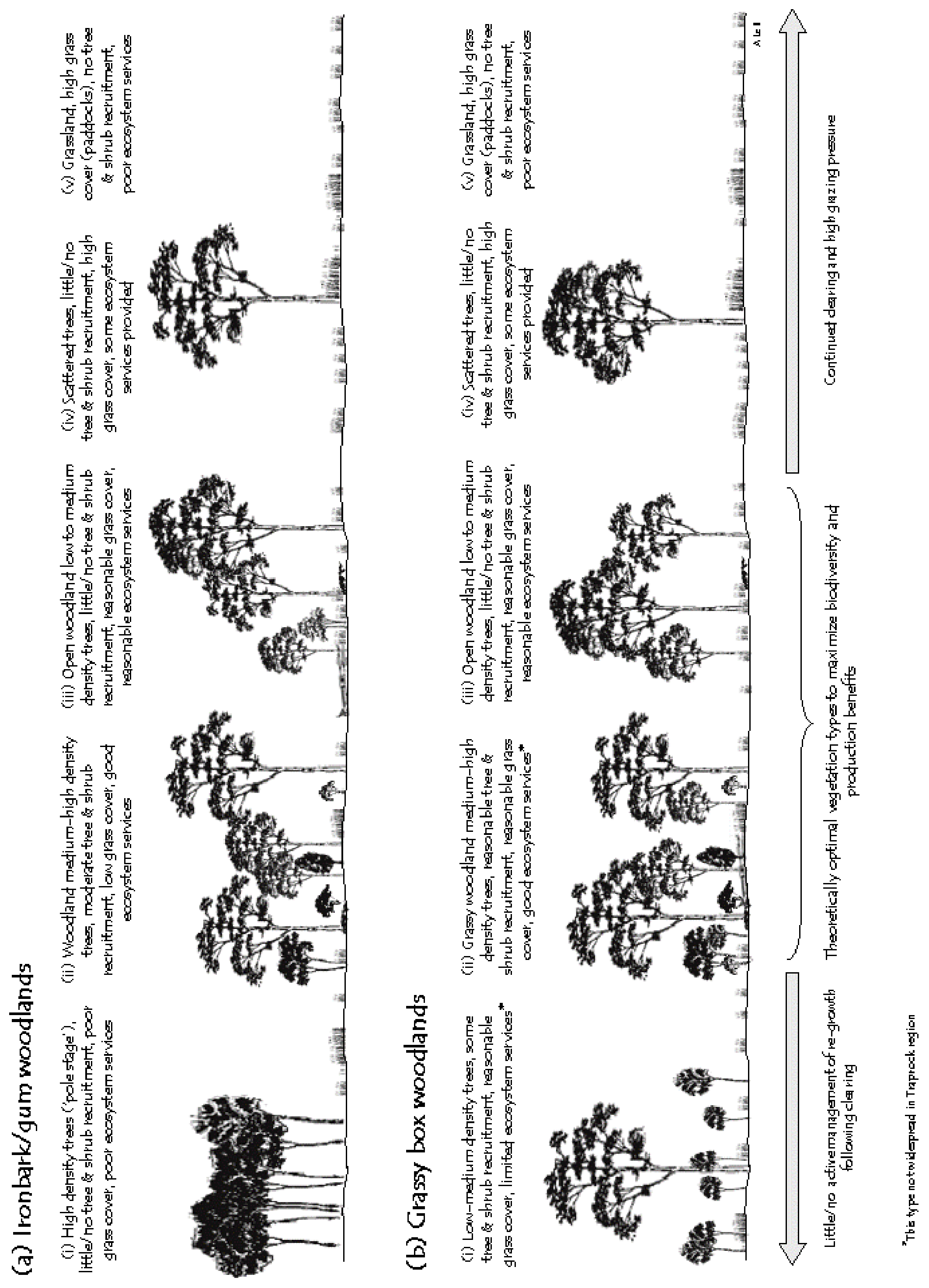

Figure 29. Options for landscape mix of vegetation conditions for (a) ironbark/gum woodlands and (b) grassy box woodlands in Traprock region.

6.3 Final Remarks

There have been few studies documenting changes in floristic composition due to vegetation management practices within pastoral grazing lands in Australia. This study has shown that a diversity of management practices is important for native plant species and heterogeneity within variegated landscapes. These results are consistent with previous studies that have examined the effects of anthropogenic disturbances on woodland communities within variegated landscapes (McIntyre 1994; Chilcott *et al.* 1997; Lunt 1997). Importantly, the results show that, at least for box woodland communities, restoring trees at a medium density to open areas may lead to more sustainable grazing practices.

The results of this study also confirm initial assumptions regarding three main factors impacting upon vegetation patterns: vegetation type, mature tree density and presence/absence of woody regrowth in the understorey. All three factors have been shown to influence vegetation patterns in the Traprock region. These effects are largely inseparable from grazing intensity which is identified as a spatial-autocorrelate of vegetation patterns observed; however, in these productive agro-ecosystems grazing is a constant, somewhat spatially variable, pressure on vegetation and one that is likely to remain.

The results from this study suggest that the Traprock region contains landscape elements with high relative biodiversity value and that sheep grazing for wool production in the region can exert significantly less impact on biodiversity through some simple, although long term, changes in management practices.

The significance of these results for the management of woodland communities for biodiversity suggests that a diversity of management practices will maintain a diversity of native plant species.

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