

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

## Technical Report: Vegetation changes following the short-term exclusion of grazing in the Traprock region



Biodiversity component of Traprock (USQ5) LWW/AWI project  
May, 2006



*A collaborative research project involving:*



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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: Vegetation changes following the exclusion of grazing in the Traprock region*

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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, 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 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 *Vegetation changes following the exclusion of grazing 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.

This technical report outlines the patterns in vegetation, including floristic composition, stand structure and species richness, in response to the short-term exclusion of grazing under different tree densities.

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## Summary

This study is the continuation of earlier research examining the effects of vegetation management on woodland communities in the Traprock region and will contribute to an understanding of the 'biodiversity potential' of managed land units on properties (including open paddocks, scattered treed areas, and remnant woodland). A number of grazing exclosures were established on selected properties with the aim to monitor biodiversity changes over-time following the removal of grazing.

Eighteen study sites across 10 properties in the Traprock region were selected for this study. The experimental design consisted of: 2 vegetation types (grassy box (*Eucalyptus melliodora*, *E. microcarpa*, or *E. moluccana*) woodland (L) and ironbark (*E. crebra*) /gum (*E. dealbata*) (U) woodland); 3 mature (overstorey) tree densities (<6 trees/ha [low](L); 6-20 trees/ha [medium](M); >20 trees/ha [high](H), and; 3 exclosures (full exclosure (1) [2.5m complete fence], partial exclosure (2) [1.5m three-wire fence], open (3) [corner makers]). Exclosure plots were erected in January/February 2005 and sampled in April 2005 (two months after exclosure establishment) and again in February 2006 (12 months after exclosure establishment). Within each 6 x 6 m exclosure plot, a central 2 x 2 m quadrat was sampled for plant species cover (determined subjectively). Above-ground vegetation ("biomass") was clipped in a 0.25m<sup>2</sup> sample and dry weight determined. Overstorey cover and recruitment were determined within each 6 x 6 m exclosure plot. Stand structural characteristics, including foliage projective cover of distinct strata, and cover of litter, logs and rocks, and general habitat condition were also determined at each site.

Patterns in floristic composition were determined using non-metric Multidimensional Scaling (nMDS). Two-way crossed Analysis of Similarity (ANOSIM) determined whether there were significant differences in floristic composition between exclosure types and mature tree density classes. nMDS was also used to assess patterns in cover data for growth forms. Two-way crossed Analysis of Variance (ANOVA) was used to determine if groups (density class, exclosure type and the interaction of density and exclosure) differed significantly for total, native, exotic, growth-form, perennial, and annual species richness and above-ground biomass. In addition, Spearman-rank correlations were performed to determine if biomass and estimates of ground cover covaried.

A total of 151 plant species was recorded across all treatments with an average richness across treatments of 17 species per 4m<sup>2</sup>. No differences were apparent in overall plant composition (cover) between the enclosure treatments one year following grazing exclusion (ANOSIM,  $p > 0.05$ ). nMDS ordinations show no distinction between enclosure treatments, but patterns were observed in mature tree density treatments within vegetation types as found in previous research. Stand structure (cover of strata) showed much the same pattern as floristic composition.

There were generally no detectable differences ( $p > 0.05$ ) in plant above-ground biomass between enclosure treatments, although significant differences between tree density classes was indicated with a significantly higher plant biomass in low density treatments compared to high density for both vegetation types. Correlation results showed that estimates of grass cover provide a good indication of above-ground biomass ( $p < 0.05$ ). There were some differences between density classes for growth-form species richness, exotic species richness and annual species richness, however enclosure treatments did not differ. Overall, there were no differences in total or native species richness between groups.

While the distinction between vegetation type and mature tree density is observed in species composition, plant biomass and species richness, the exclusion of grazing (native and exotic) has not significantly altered composition after 12 months. The patterns in floristic composition are associated with different mature tree densities and vegetation type, which are consistent with earlier findings. There is some evidence to suggest that plant above-ground biomass has responded to the removal of grazing in open paddock areas, although this is not consistent across mature tree density treatments. It is suggested that a longer period of exclusion will be necessary to detect changes (if any) in plant species composition.



# 1. Introduction

## 1.1 General Introduction

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Woodland ecosystems can provide a number of important services for agricultural production such as 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, birds, 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).

Unfortunately, much of Australia's woodland ecosystems have been cleared for agricultural production. It is estimated that 500 000 km<sup>2</sup> of woodlands have been cleared since European settlement (AUSLIG 1990). As a result, some woodland communities are amongst the most poorly conserved ecosystems in Australia (Yates and Hobbs 1997). It is now recognised that where woodlands have little or no representation in nature reserves, biodiversity conservation may best be achieved if combined with current production systems (McIntyre 1994; Chilcott *et al.* 1997). Pastoral grazing lands that contain semi-intact woodlands, termed variegated landscapes, may be compatible with conservation outside reserves if managed appropriately (McIntyre and Barrett 1992; McIntyre 1994; Chilcott *et al.* 1997). This requires understanding how management practices within pastoral grazing lands affect vegetation characteristics in woodland communities.

## 1.2 Land management for livestock grazing

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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).

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 the 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) found that livestock 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).

In addition to livestock grazing, 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).

### 1.3 Project Aims

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An important part of the Biodiversity Component of the *Integrating paddock and catchment planning: a wool producer-driven approach to sustainable landscape management* project is to ascertain the biodiversity *potential* of managed land units on properties (including open paddocks, scattered treed areas, and remnant woodland). To gauge this, a number of grazing exclusions were established on selected properties with the aim to monitor biodiversity changes over time following the removal of grazing.

Studies that have examined the response of native vegetation to grazing exclusion elsewhere in Australia have reported diverse results. For example, in Western Australia the floristic composition and species richness of exclusion plots had become more floristically similar to ungrazed woodland remnants after 7 years of recovery from livestock grazing (Pettit and Froend 2001), and in southern NSW greater tree and shrub recruitment and lower soil compaction were observed after the short-term (2-4 years) exclusion of livestock grazing in remnant grassy woodlands (Spooner *et al.* 2002). Conversely, in the subalpine grasslands in eastern Australia, species richness was reported to decrease in exclusion plots (Gibson and Kirkpatrick 1989 cited in Pettit and Froend 2001), and in the grassy communities of northern NSW, areas left ungrazed for 16 years were densely vegetated and relatively species-poor compared to adjacent areas that had been grazed for 16 years (Tremont 1994).

These studies demonstrate that it is often hard to predict the response of vegetation to grazing exclusion and results may largely depend on differences in environment, grazing history (Milchunas and Lauenroth 1993 cited in Pettit and Froend 2001) and length of grazing exclusion.

The objectives of this study were to:

1. Determine the response of vegetation to the short-term exclusion of grazing in the Traprock region by examining the floristic composition, species richness and above-ground biomass of exclusion plots (complete, partial and open/control).
2. Determine if tree density and/or vegetation type are important in explaining patterns in plant communities.

3. Determine if tree density, vegetation type or enclosure type (complete, partial and open/control) interact to influence floristic patterns.

This research is a significant component of an ongoing evaluation of sustainable land management in the Traprock region. 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

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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 (Figure 1).

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 - 30.2°C for Warwick (28°22'S, 152°03'E) and 0.9 - 27.4°C for Stanthorpe (28°66'S, 151°93'E) (Bureau of Meteorology 2005). Rainfall is generally higher in summer months, although the winter proportion can be significant (Wills 1976).

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 vegetation of the Traprock region is predominately grassy eucalypt woodland mainly comprised of narrow-leaved ironbark (*Eucalyptus crebra*), tumbledown gum (*Eucalyptus dealbata*), and white box (*E. albens*) and, on the lower slopes, yellow box (*E. melliodora*), grey box (*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 on an average Traprock property of approximately 2000 ha, the amount of timber killed ranged between 60-100%. Natural grasslands are not considered to have been a common component before settlement (Wills 1976).

Regional ecosystem mapping by the Queensland Herbarium shows approximately 22 % of remnant vegetation remains in the Traprock region (Queensland Murray Darling Committee, 2004). The 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. camaldulensis*, *E. tereticornis* and *E. blakeyi*. *Callitris* spp. also occurs occasionally in this vegetation type. While *E. melliodora* and *E. macrocarpa* are of value from grassy box woodlands.

## 2.2 Study Sites & Experimental Design

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To ensure the major community types were sampled, site selection was stratified across the study area according to vegetation type and density of mature trees. 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. Only sites with an absence of woody regrowth in the understorey were included. 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 experimental design consisted of:

- two vegetation types (ironbark/gum woodlands; box woodlands); and
- three mature tree densities (<6 trees/ha [low]; 6-20 trees/ha [medium]; >20 trees/ha [high]).

Six treatment combinations were recognised (Table 1) and a total of 18 sites across ten properties were selected (Figure 1). Three 6x6 metre exclosure plots were erected in January/February 2005 at each of the eighteen sites (totalling 54 exclosure plots). The exclosure plots consist of:

1. **complete exclosure:** 2.5 metre fence to exclude both sheep and large herbivores (e.g. kangaroos);
2. **partial exclosure:** 1.5 metre three-wire fence to exclude sheep only;
3. **open (control):** corner markers, no fence (to allow grazing).

Exclosure plots were erected at locations representative of the vegetation at each site. Plots were a minimum distance of 20 m apart and a minimum of 20 m from the boundary of the patch to minimise edge effects.

**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; ironbark/gum woodland	LU	3	1, 2, 3,
Low density; box woodland	LL	3	4, 5, 6
Medium density; ironbark/gum woodland	MU	3	7, 8, 9
Medium density; box woodland	ML	3	10, 11, 12
High density; ironbark/gum woodland	HU	3	13, 14, 15
High density; box woodland	HL	3	16, 17, 18

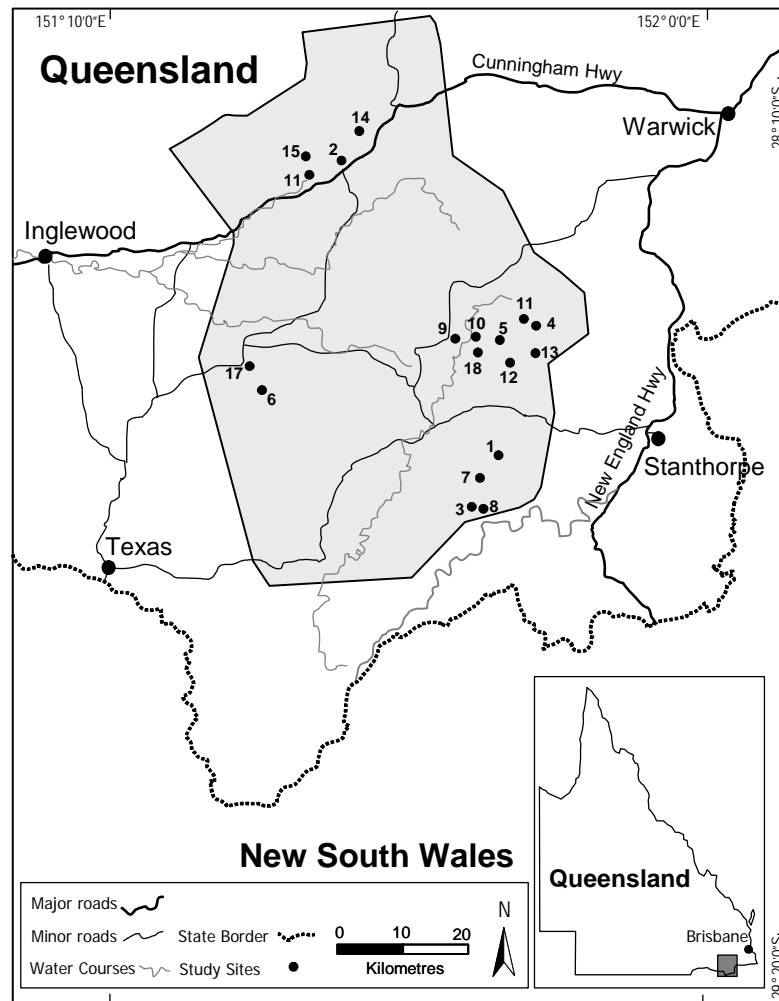


Figure 1. Map of Traprock study area showing approximate location of enclosure sites.

## 3. Methods

### 3.1 Data Acquisition

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Sites were sampled in April 2005 (two months after exclosure establishment) and in February 2006 (12 months after exclosure establishment). Within each 6 x 6 m exclosure plot, a central 2 x 2 m quadrat was sampled for ground cover and vascular plant species composition (Figure 3). Ground cover was determined by subjectively estimating the percent cover of forbs/herbs/other (non-woody), graminoids (grasses/sedges), logs and branches >10cm in circumference, coarse litter (twigs and branches, 4-10 cm circumference), fine litter (leaf and twigs, < 4 cm circumference), rock cover, bare ground and cryptogams within each 2 x 2 m quadrat. The average heights (centimetres) of grasses were also recorded within each 2 x 2 m quadrat by measuring the height of four different grasses with a ruler and taking the mean.

The composition of plant species was determined by estimating the percent cover of vascular plants within each 2x2 metre 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 Henry *et al.* (1995). Exotic species were distinguished as any plant species that has been introduced into Australia and identified using nomenclature of Stanley & Ross (1983, 1986, 1989) and Auld & Medd (1987).

Overstorey cover and recruitment were also recorded for each exclosure plot. Overstorey cover was determined by subjectively estimating the percentage foliage cover of trees >10 m, trees <10 m, shrubs >2 m and shrubs <2 m within each 6x6 m exclosure plot. Recruitment was determined as the number of juvenile trees (> 1 m, <1 m) and shrubs, and by scoring epicormic regrowth and suckering from 0 (none) to 3 (high) according to their percent cover within each 6 x 6 m quadrat (where 0=0%; 1=<10%; 2=10-20%; 3=>20%).

Above-ground vegetation was clipped in a separate 0.25 m<sup>2</sup> quadrat within each 6 x 6 m exclosure plot (Figure 2). Plant biomass (gm/0.25 m<sup>2</sup>) was determined by drying the samples in paper bags in an oven at 50°C for 4 days in 2005, and at 60°C for 48 hours in 2006.

At each 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 (>20m, 10-20 m, <10 m), shrubs (>2 m, <2 m), forbs and herbs (non-woody species), and graminoids (including grasses and sedges). The percentage foliage cover of each stratum was subjectively estimated at each site. In addition, the cover of logs >100 cm in circumference, rock cover, and the cover of coarse litter (twigs and branches, 4-10 cm circ.) and fine litter (leaf and twigs, < 4 cm circumference) were estimated and recorded for each site. The condition of the site was determined by recording evidence of disturbance. 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 at each site.

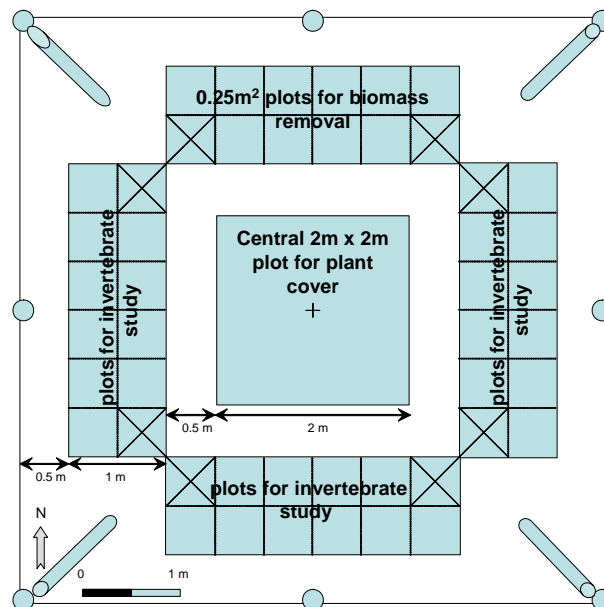


Figure 2. Diagram of enclosure plot design.

### 3.2 Statistical analyses

Non-metric multidimensional scaling (nMDS) was performed on plant cover data and stand structure 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). nMDS was performed on all enclosure and site plant cover data, as well as on a subset to determine the relationship between sites in ironbark/gum woodlands and box woodlands. 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 cover data to allow sites to be represented graphically and to discriminate sites from each other (Clarke and Warwick 2001). Bray-Curtis similarity coefficient is widely accepted as a satisfactory coefficient for biological data on community structure (Clarke and Warwick 2001).

Two-way crossed analysis of similarity (ANOSIM) was performed on plant cover data for ironbark/gum woodlands and box woodlands using the Primer v.5.2.9 for Windows computer program (Primer-E Ltd 2001). A Bray-Curtis similarity matrix was computed and the two-way crossed ANOSIM test performed on the cover data to determine if there were differences between enclosure plots, allowing for the fact that there may be density differences or (vice versa) to determine if there were differences between tree densities (Clarke and Gorley 2001).

SPSS® for Windows version 11.5 (SPSS Inc. 2002) was used to perform two-way crossed ANOVA (Analysis of Variance) to determine if there were differences in total species richness, shrub species richness, graminoid species richness, herb/forb species richness, native species richness, exotic species richness, perennial species richness, annual species richness and plant biomass for each vegetation type due to density class (low, medium, high) or enclosure type (complete, partial, open) or the interaction of density and enclosure (all species richness data were log transformed to improve normality and reduce the influence of any outliers).

Prior to performing two-way crossed ANOVA, the Levene's statistic and residual plots were used to test homogeneity of variances in SPSS® for Windows version 11.5 (SPSS Inc. 2002). In addition, Spearman-rank correlations were performed using SPSS® for Windows version 11.5 (SPSS Inc. 2002) to determine whether biomass and estimates of ground cover were related and the degree to which they vary together (Ashcroft and Pereira 2003).

## 4. Results

A total of 151 plant taxa (135 native, 16 exotic) were recorded across the study area. Total species richness ranged from 2 to 29 species, with an average richness over all samples of 17 species per 4 m<sup>2</sup>.

### 4.1 Floristic Composition

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Non-metric multi-dimensional scaling (nMDS) ordination of floristic composition data (Figure 3) (stress=0.21) shows a clustering of sites to the left of the diagram, with one high density ironbark/gum woodland site (site 15) separated from this major cluster. The ordination shows a clustering of samples according to site rather than enclosure type. The large cluster shows that sites with similar mature tree densities tend to cluster together (with some exceptions). Low tree density sites (LU and LL) are largely grouped to the left, with medium density ironbark/gum woodlands (MU) immediately to the right. Medium and high density box woodlands (ML and HL) are slightly separated from this cluster with high density ironbark/gum woodland sites (HU) separated towards the end of the major cluster. Based on previous evidence of differences in vegetation types (see Goodhew 2005), floristic data were separated by vegetation type and a nMDS ordination performed on each data set.

nMDS ordination of floristic composition data for ironbark/gum woodland sites (Figure 4) (stress=0.17) shows a gradient of increasing mature tree density from left to right across the diagram. Low density sites (LU), regardless of enclosure type were shown to cluster to the left of the diagram, with medium density sites (MU) in a loose cluster immediately to the right. High density sites (HU) were reasonably separated from low and medium density sites, representing the other end of the cluster.

nMDS ordination of floristic composition data for box woodland sites (Figure 5) (stress=0.23) shows a closer clustering of samples. With some exceptions, the ordination shows low density sites (LL) clustering towards the left of centre with medium (ML) and high density sites (HL) in a separate cluster towards the right of centre.

Two-way crossed ANOSIM of floristic composition data for each vegetation type (Table 2) shows that the test for differences between mature tree densities was significant for both ironbark/gum woodlands ( $p=0.007$ ) and box woodlands ( $p=0.037$ ). However, the enclosure differences overall were not significant (see Table 2). For ironbark/gum woodlands, high mature tree density sites (HU) were significantly different in floristic composition to low (LU) and medium (MU) tree density sites. For box woodlands, high mature tree density sites (HL) were significantly different in floristic composition to low tree density sites (LL). Medium density box woodland sites (ML) were not significantly different in floristic composition to either high or low tree density sites.



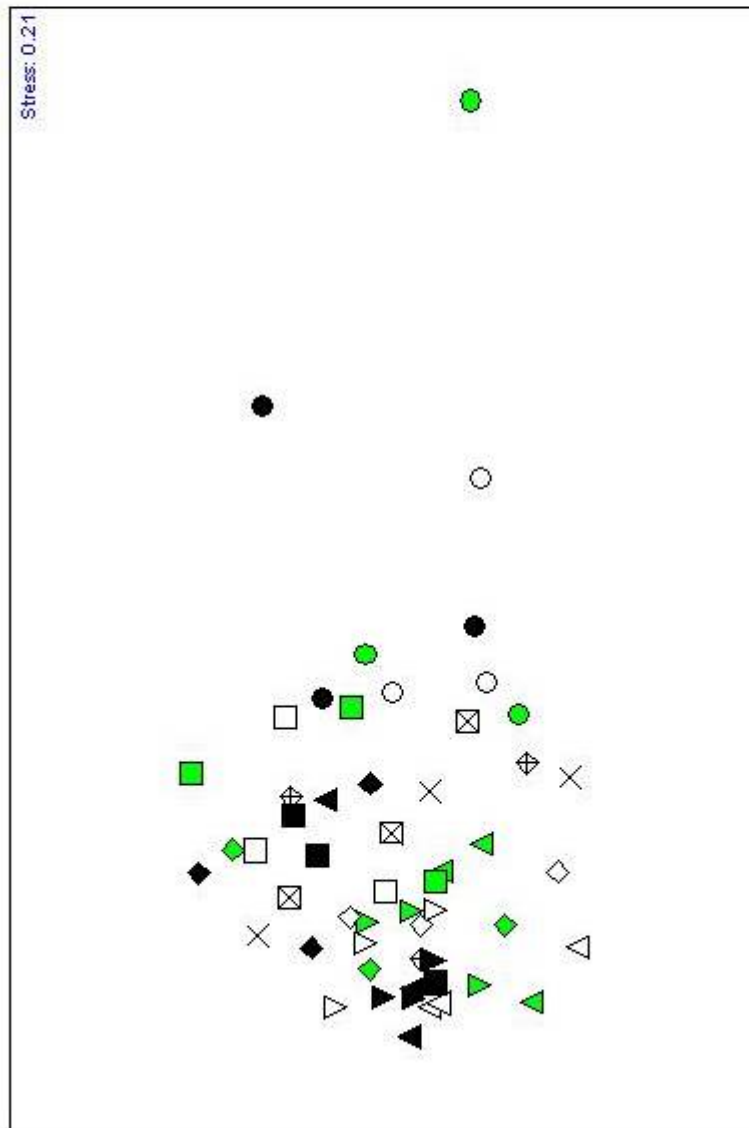
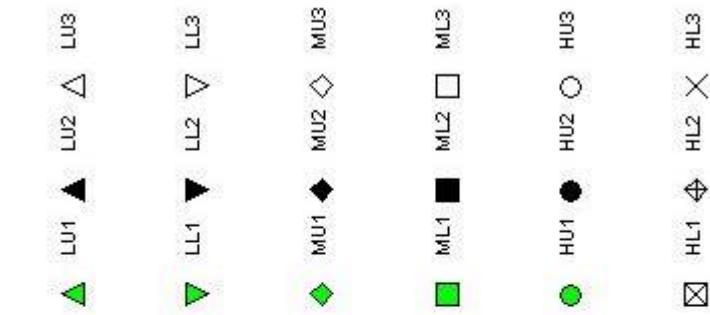


Figure 3. Non-metric multi-dimensional scaling ordination of floristic data. See Table 1 for descriptions of treatment codes.

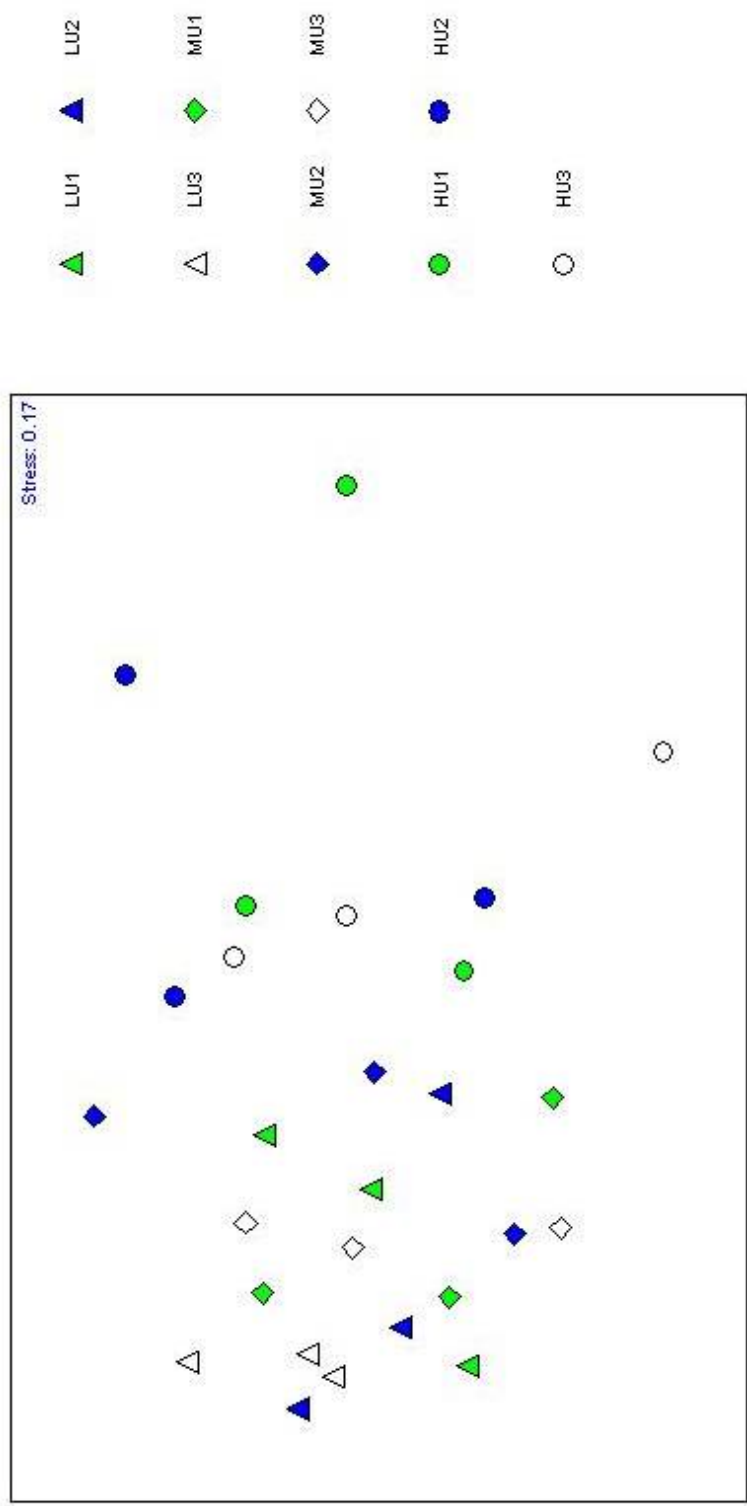


Figure 4. Non-metric multi-dimensional ordination of floristic data for ironbark/gum woodland sites.

See Table 1 for descriptions of treatment codes.

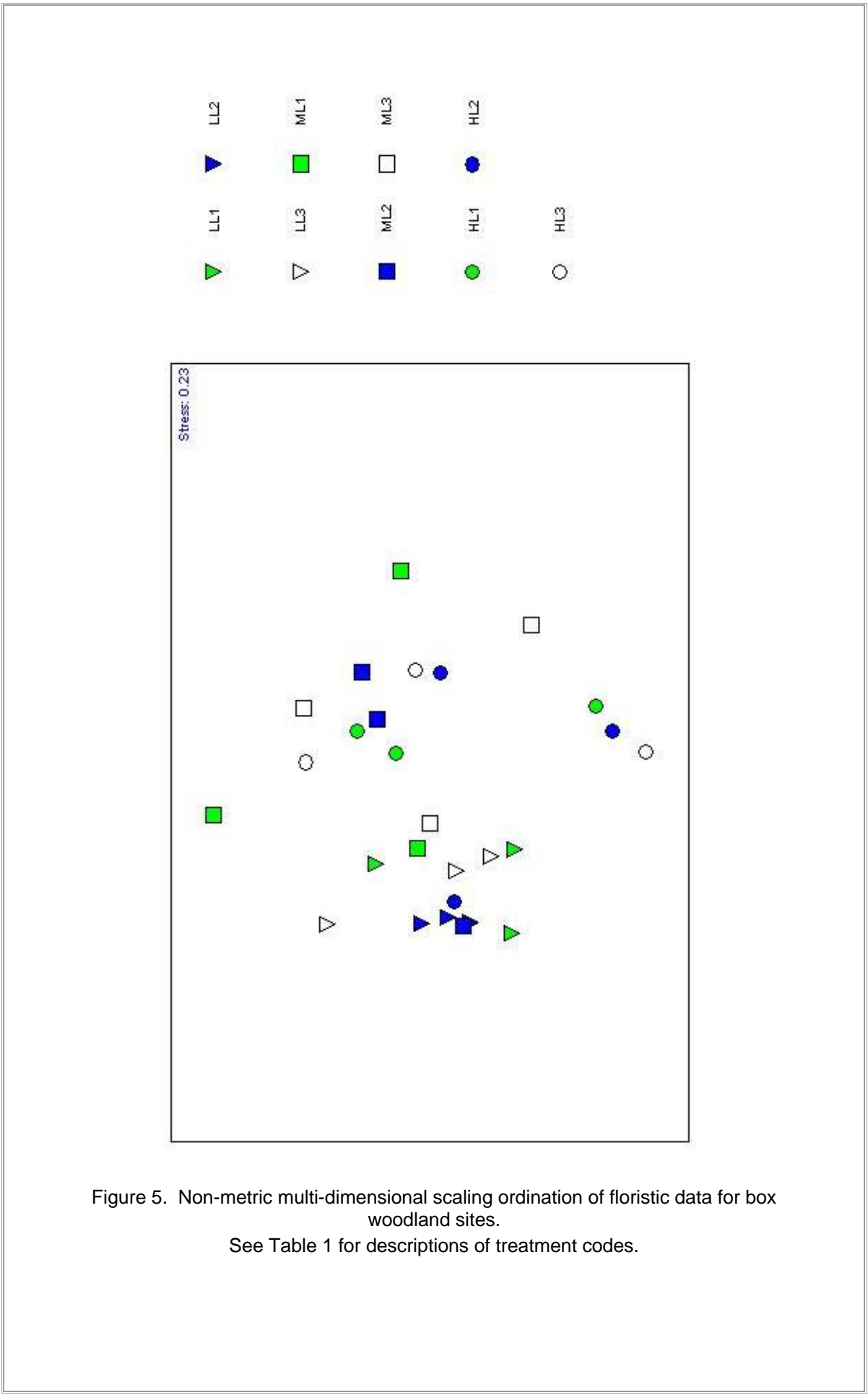


Figure 5. Non-metric multi-dimensional scaling ordination of floristic data for box woodland sites.

See Table 1 for descriptions of treatment codes.

Table 2. Two-way crossed ANOSIM of floristic data for ironbark/gum woodland sites and box woodland sites.

Global R value and significance level shown (groups sharing the same superscript letter are not significantly different).

	R value	Significance (p-value)	Pairwise tests		
<b>Ironbark woodland</b>					
Density	0.262	0.007*	Low <sup>a</sup>	Medium <sup>a</sup>	High <sup>b</sup>
Exclosure	-0.192	0.953	Complete <sup>a</sup>	Partial <sup>a</sup>	Open <sup>a</sup>
<b>Box woodland</b>					
Density	0.193	0.037*	Low <sup>a</sup>	Medium <sup>ab</sup>	High <sup>b</sup>
Exclosure	-0.147	0.894	Complete <sup>a</sup>	Partial <sup>a</sup>	Open <sup>a</sup>

\*results are significant at the 0.05 level

## 4.2 Species richness and plant biomass

Two-factor ANOVA results for total plant species richness and growth-form species richness are shown in Table 3. The results indicate some significant differences in shrub species richness, graminoid species richness and herb/forb species richness between mature tree density classes (although variances are unequal). Exclosure type and the interaction of exclosure type and tree density did not contribute to differences between groups.

Two-factor ANOVA results for native and exotic species richness, perennial and annual species richness and plant biomass are shown in Table 4. The results indicate some significant differences in exotic species richness, annual species richness and plant biomass between mature tree density classes. In particular, for ironbark/gum woodlands annual species richness was significantly lower in high density sites compared to low and medium density sites. Exclosure type and the interaction of exclosure type and tree density did not contribute to differences between groups.

Table 3. Two-factor ANOVA of total species richness and growth-form richness for each vegetation type.

Homogeneity of variances indicated by N (no) or Y (yes). [Degrees of freedom: 2=density; 2=exclosure; 4=density\*exclosure.] Post-hoc results where applicable include means and standard errors in parentheses. Means sharing the same superscript letter are not significantly different (L=low, M=medium, H=high tree density).

		Homogeneity	F	P-value	Post-hoc		
<b>Total species richness (no.sp/4m<sup>2</sup>)</b>							
Ironbark woodland	Density	N	2.76	0.090			
	Exclosure		0.01	0.991			
	Density*Exclosure		0.10	0.981			
Box woodland	Density	Y	0.90	0.423			
	Exclosure		0.33	0.723			
	Density*Exclosure		0.95	0.945			
<b>Shrub species richness (no.sp/4m<sup>2</sup>)</b>							
Ironbark woodland	Density	N	5.90	0.011*	L 0.0 <sup>a</sup> (0.0)	M 0.7 <sup>ab</sup> (0.2)	H 2.4 <sup>b</sup> (0.9)
	Exclosure		0.19	0.830			
	Density*Exclosure		0.09	0.984			
Box woodland	Density	N	5.26	0.016*	L 0.1 <sup>a</sup> (0.1)	M 0.8 <sup>b</sup> (0.2)	H 0.1 <sup>a</sup> (0.1)
	Exclosure		0.53	0.597			
	Density*Exclosure		0.33	0.855			
<b>Graminoid species richness (no.sp/4m<sup>2</sup>)</b>							
Ironbark woodland	Density	N	2.04	0.159			
	Exclosure		0.26	0.777			
	Density*Exclosure		0.08	0.989			
Box woodland	Density	N	5.60	0.013*	L 9.6 <sup>a</sup> (0.5)	M 8.6 <sup>ab</sup> (0.8)	H 6.2 <sup>b</sup> (0.8)
	Exclosure		0.23	0.795			
	Density*Exclosure		0.39	0.810			
<b>Herb/forb species richness (no.sp/4m<sup>2</sup>)</b>							
Ironbark woodland	Density	N	6.26	0.009*	L 7.1 <sup>ab</sup> (0.8)	M 10.7 <sup>b</sup> (1.0)	H 4.2 <sup>a</sup> (1.0)
	Exclosure		0.12	0.887			
	Density*Exclosure		0.18	0.947			
Box woodland	Density	N	0.95	0.405			
	Exclosure		0.26	0.775			
	Density*Exclosure		0.06	0.993			

\*significant at the 0.05 level

Table 4. Two-factor ANOVA of native, exotic, perennial and annual species richness and above-ground plant biomass for each vegetation type.

Homogeneity of variances indicated by N (no) or Y (yes). [Degrees of freedom: 2=density; 2=exclosure; 4=density\*exclosure.] Post-hoc results where applicable include means and standard errors in parentheses. Means sharing the same superscript letter are not significantly different (L=low, M=medium, H=high tree density).

		Homogeneity	F	P-value	Post-hoc		
<b>Native species richness (no.sp/4m<sup>2</sup>)</b>							
Ironbark woodland	Density	N	2.25	0.134			
	Exclosure		0.00	0.997			
	Density*Exclosure		0.01	0.985			
Box woodland	Density	Y	0.94	0.408			
	Exclosure		0.35	0.712			
	Density*Exclosure		0.04	0.997			
<b>Exotic species richness (no.sp/4m<sup>2</sup>)</b>							
Ironbark woodland	Density	N	5.34	0.015*	L 1.8 <sup>a</sup> (0.3)	M 1.4 <sup>ab</sup> (0.5)	H 0.2 <sup>b</sup> (0.2)
	Exclosure		1.14	0.341			
	Density*Exclosure		0.44	0.782			
Box woodland	Density	Y	0.20	0.817			
	Exclosure		1.41	0.270			
	Density*Exclosure		1.78	0.177			
<b>Perennial species richness (no.sp/4m<sup>2</sup>)</b>							
Ironbark woodland	Density	N	2.17	0.143			
	Exclosure		0.00	0.996			
	Density*Exclosure		0.12	0.972			
Box woodland	Density	Y	1.17	0.332			
	Exclosure		0.39	0.686			
	Density*Exclosure		0.12	0.974			
<b>Annual species richness (no.sp/4m<sup>2</sup>)</b>							
Ironbark woodland	Density	Y	18.00	0.000*	L 1.4 <sup>a</sup> (0.2)	M 2.0 <sup>a</sup> (0.3)	H 0.3 <sup>b</sup> (0.2)
	Exclosure		0.96	0.401			
	Density*Exclosure		0.46	0.765			
Box woodland	Density	Y	0.22	0.806			
	Exclosure		0.69	0.516			
	Density*Exclosure		0.61	0.661			
<b>Plant biomass (gm/0.25 m<sup>2</sup>)</b>							
Ironbark woodland	Density	N	3.83	0.041*	L 40.1 <sup>a</sup> (10.4)	M 16.7 <sup>ab</sup> (2.9)	H 2.1 <sup>b</sup> (0.9)
	Exclosure		0.18	0.840			
	Density*Exclosure		0.12	0.972			
Box woodland	Density	N	5.75	0.012*	L 47.3 <sup>b</sup> (13.2)	M 26.2 <sup>a</sup> (4.1)	H 12.2 <sup>a</sup> (4.9)
	Exclosure		0.20	0.819			
	Density*Exclosure		0.50	0.737			

\*significant at the 0.05 level

Mean above-ground plant biomass ( $\text{gm}/0.25\text{m}^2$ ) per enclosure type for each mature tree density class and vegetation type across years are shown in Figures 6 to 11. An increase in mean above-ground biomass from April 2005 to February 2006 was indicated for low tree density treatments for both vegetation types, and medium and high tree density box woodlands sites. In the low density ironbark/gum woodland sites, mean above-ground biomass increased from an average of  $28 \text{ gm}/0.25\text{m}^2$  (SE  $\pm 13.7$ ) in 2005 to an average of  $73.2 \text{ gm}/0.25\text{m}^2$  (SE  $\pm 11.5$ ) in the complete enclosure (Figure 6). In the low density box woodland sites, the most notable increase in mean above-ground biomass was in the partial enclosure with an average of  $14.8 \text{ gm}/0.25\text{m}^2$  (SE  $\pm 5.7$ ) recorded in 2005 and an average of  $62.3 \text{ gm}/0.25\text{m}^2$  (SE  $\pm 32.5$ ) recorded in 2006 (Figure 7). There was no notable increase in average above-ground biomass between years for medium (Figure 8) and high density (Figure 10) ironbark/gum woodland sites.

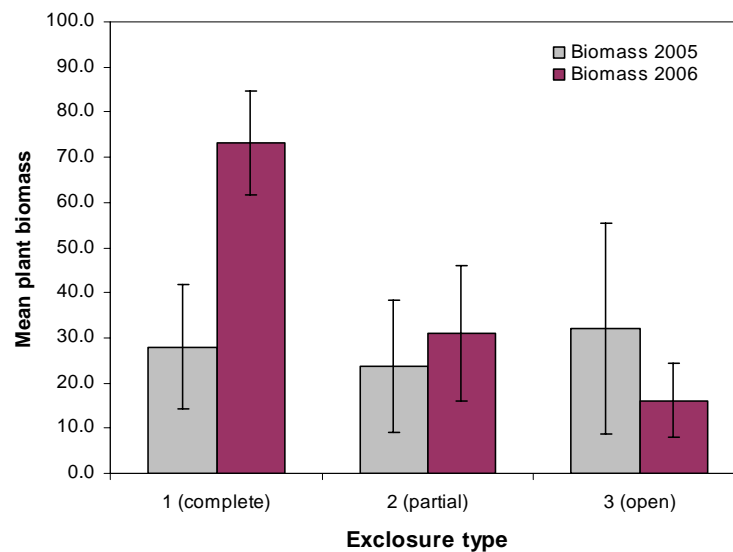


Figure 6. Mean above-ground plant biomass ( $\text{gm}/0.25\text{m}^2$ ) for low density ironbark/gum woodland (LU) across years (error bars represent standard error)

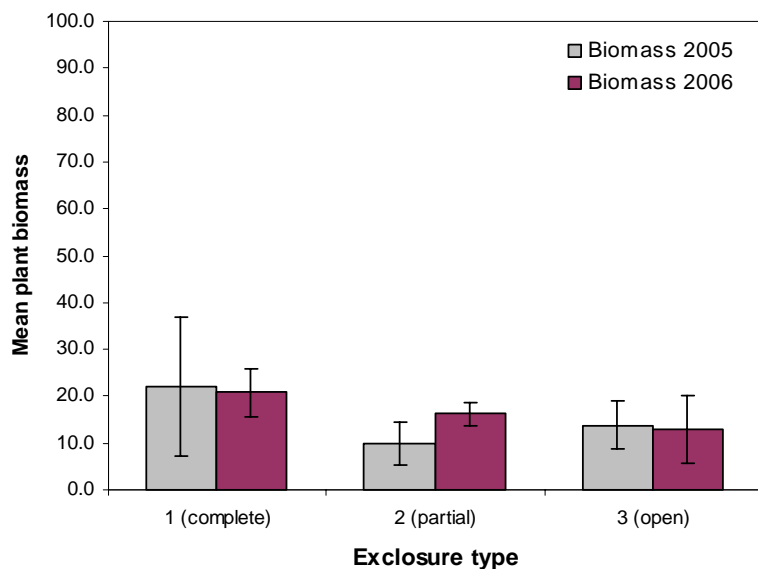


Figure 7. Mean above-ground plant biomass (gm/0.25m<sup>2</sup>) for low density box woodland (LL) across years (error bars represent standard error)

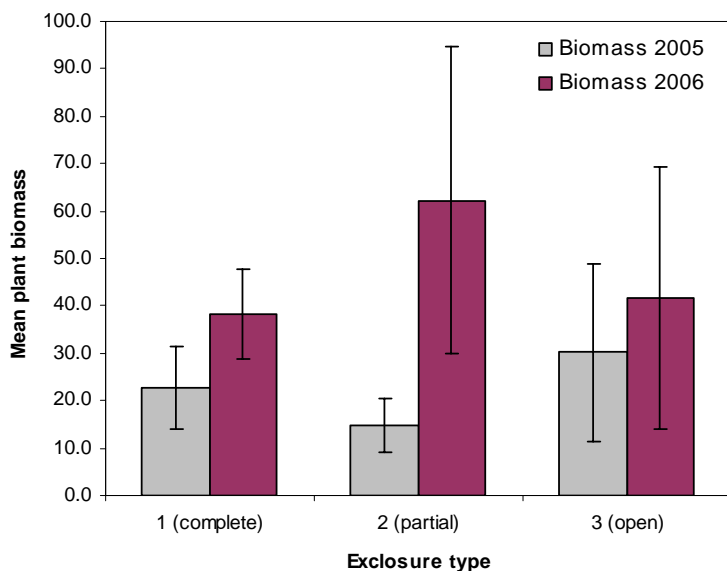


Figure 8. Mean above-ground plant biomass (gm/0.25m<sup>2</sup>) for medium density ironbark/gum woodland (MU) across years (error bars represent standard error)



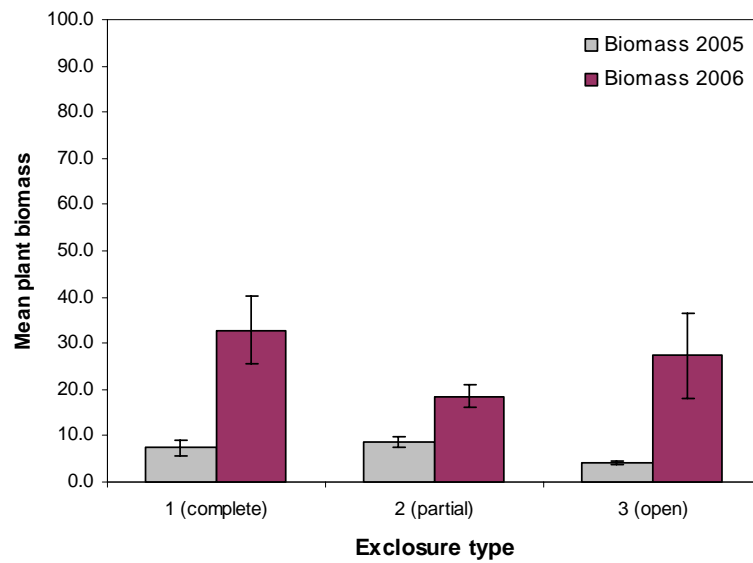


Figure 9. Mean above-ground plant biomass (gm/0.25m<sup>2</sup>) for medium density box woodland (ML) across years (error bars represent standard error)

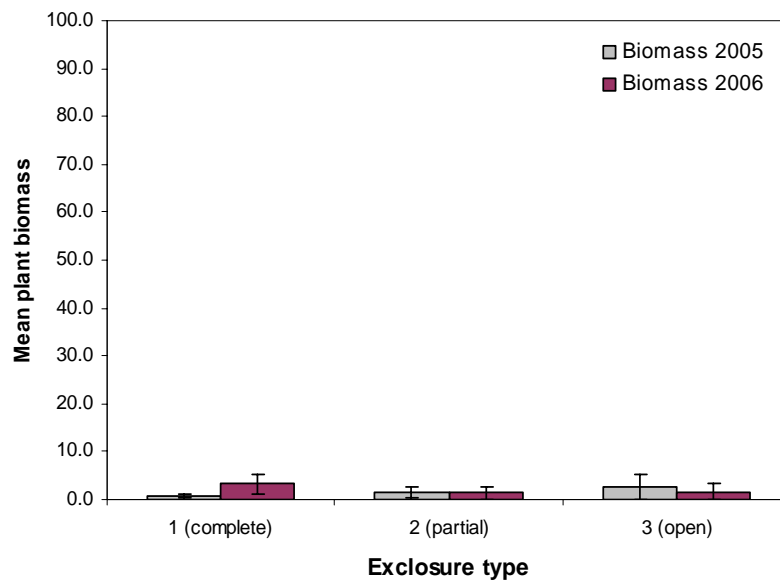


Figure 10. Mean above-ground plant biomass (gm/0.25m<sup>2</sup>) for high density ironbark/gum woodland (HU) across years (error bars represent standard error)

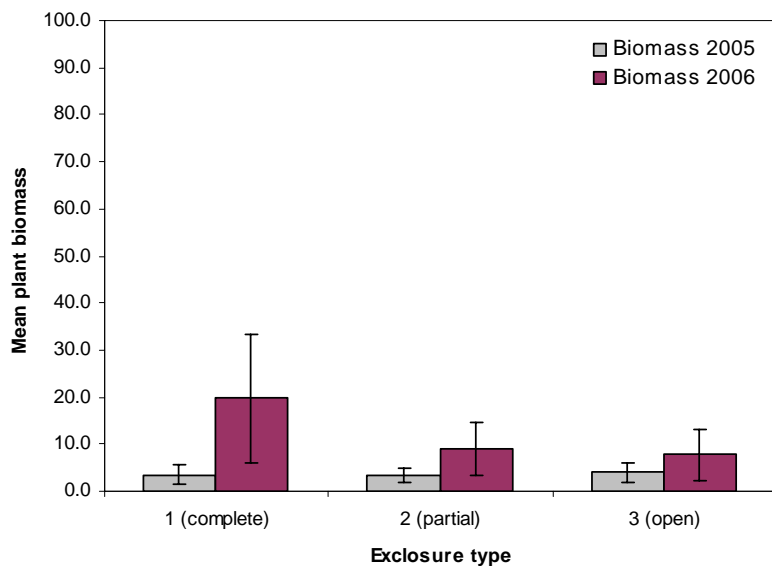


Figure 11. Mean above-ground plant biomass (gm/0.25m<sup>2</sup>) for high density box woodland (HL) across years (error bars represent standard error)

Estimates of ground cover (per 4m<sup>2</sup>) and total ground cover (per 4m<sup>2</sup>) were positively correlated with above-ground plant biomass (gm/0.25m<sup>2</sup>) for both sampling years (Table 5). Estimates of herb/forb cover (per 4m<sup>2</sup>) were not correlated with either above-ground biomass (gm/0.25m<sup>2</sup>) or estimates of grass cover and total ground cover (per 4m<sup>2</sup>).

nMDS ordination of stand structure data revealed that the structure of treatments were reasonably consistent across sites (Figure 12; stress 0.07). With a few exceptions, high tree density ironbark/gum woodlands were well separated from high and medium density box woodlands. Medium density ironbark/gum woodlands were separated from this cluster, while low tree density sites clustered together regardless of vegetation type, and were well separated from all other clusters.

Table 5. Spearman rank correlation results showing the relationship between ground cover biomass (gm/0.25m<sup>2</sup>) and estimates of ground cover (per 4m<sup>2</sup>). Data are for both sampling years (2005 and 2006).

	<b>Biomass (gm/0.25m<sup>2</sup>)</b>	<b>Grass cover (per 4m<sup>2</sup>)</b>	<b>Herb/forb cover (per 4m<sup>2</sup>)</b>
<b>2005</b>			
Grass cover	0.696*		
Herb/forb cover	0.173	0.114	
Total ground cover	0.786*	0.910*	0.264
<b>2006</b>			
Grass cover	0.702*		
Herb/forb cover	0.132	-0.035	
Total ground cover	0.697*	0.969*	0.130

\*significant at the 0.01 level

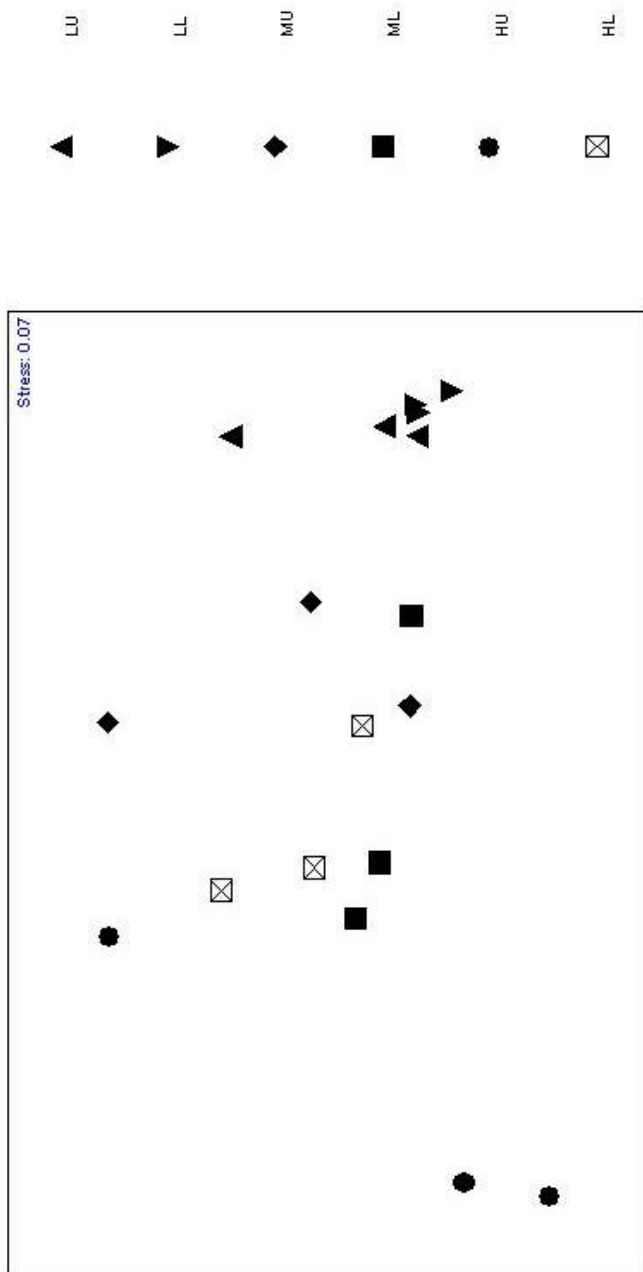


Figure 12. Non-metric multidimensional-scaling ordination of stand structure data. See Table 1 for descriptions of treatment codes.

## 5. Discussion

While the distinction between vegetation type and tree density is observed in plant species composition, the exclusion of grazing has not significantly altered composition after 12 months. This suggests that the vegetation has had little time to recover after 12 months of grazing exclusion, which agrees with results reported from other studies in Australia (Pettit and Froend 2001; Spooner *et al.* 2002). For instance, Pettit and Froend (2001) concluded that initial patterns in floristic composition after 3 years of grazing exclusion were only temporary and highly influenced by fluctuations in annual climatic patterns when exclosure plots were sampled again after 7 years (Pettit and Froend 2001). Pettit and Froend (2001) recommended that long-term monitoring is needed to detect long-term trends in floristic composition after grazing is excluded.

There was some evidence to suggest that above-ground plant biomass has responded to the removal of grazing in open paddock areas, in particular there was an increase in biomass from 2005 to 2006 in the complete exclosure for low density ironbark/gum woodland sites. However, no difference in above-ground biomass between exclosure plots was found for ironbark/gum woodland sites or box woodland sites, and there was no interaction between exclosure plot and tree density. Significant differences in above-ground biomass between tree density classes was indicated with a significantly higher plant biomass in low density box woodlands compared to medium and high density treatments, and a significantly higher plant biomass in low density ironbark/gum woodlands compared to high density treatments (although there were unequal variances).

This result does confirm conclusions made in earlier research (Goodhew 2005) and reported elsewhere in Australia, that grass cover (or biomass) declines significantly with increasing total tree cover (e.g. Walker *et al.* 1986; Harrington and Johns 1990; Scanlan and Burrows 1990; McIvor and Gardener 1995; McIvor 2001). Interestingly, the results from the correlation of grass cover (per 4 m<sup>2</sup>) and above-ground biomass (gm/0.25 m<sup>2</sup>) indicate that subjectively estimating the cover of grasses within exclosure plots will provide a good indication of plant above-ground biomass. This result is important in that more confidence can be given to subjective estimates of grass cover which are often less time consuming and costly. If landholders monitor exclosure plots in the future this result is particularly relevant.

The number of plant species recorded during this study is comparable to previous studies on woodland communities in Australia. Mean species richness has been reported as 20 species per 25 m<sup>2</sup> for woodlands in Western Australia (Yates and Hobbs 1997), 28 per 30 m<sup>2</sup> for grazed temperate grassy woodlands in New South Wales (McIntyre and Martin 2001) and 25 per 20 m<sup>2</sup> for grazed hill woodlands in south-eastern Queensland (Fensham 1998). In this study, mean species richness was 17 per 4 m<sup>2</sup> indicating the Traprock region has a significant biodiversity component (in terms of plant species). However, little evidence of overstorey species recruitment suggests that the wooded sites sampled in this study may not be viable over-time.

The recruitment of overstorey species depends on a number of factors such as suitable environmental conditions, adequate seed supply, and absence of herbivory (Clarke 2002). Possible explanations for the lack of recruitment are that there may not have been suitable environmental conditions for the germination of overstorey species since exclosure establishment (Clarke 2002), and the fact that exclosure plots were only established 12 months ago. For example, Spooner *et al.* (2002) found a significant positive correlation between tree recruitment densities and time since fencing in woodland remnants of northern NSW. However, another possibility is that disturbance (e.g. fire) may be necessary to enhance germination. Clarke (2002) found that no natural recruitment of native shrub species had occurred after 5 years in the grassy woodlands of NSW and concluded that recruitment is episodic and disturbance driven. Experiments that examine mechanisms of shrub and tree recruitment could be important for implementing management practices specific for tree and shrub establishment in the Traprock region.

The patterns in floristic composition found in this study are similar to those reported in earlier research (Goodhew 2005) where sites were separated according to vegetation type (either ironbark/gum or box woodland) and tree density (low, medium or high). As reported previously (Goodhew 2005), high density ironbark/gum woodlands are floristically and structurally different to both low and medium density ironbark/gum woodlands, yet low and medium density ironbark/gum woodlands do not differ floristically (yet are different structurally). This result suggests that both grazing and tree clearing have resulted in a vastly different understorey in low and medium density patches, with a higher annual species richness (in low and medium density treatments), higher herb/forb species richness (in medium density treatments), and lower shrub species richness (in low density treatments) compared to that of more 'natural' ironbark/gum woodlands. However, it should be noted that there were no differences in total or native species richness between groups.

The higher annual species richness in low and medium density treatments is possibly related to the more intense grazing and tree removal practices that have historically occurred in these areas. Annuals are reported to be more tolerant of disturbance than perennials due to their fast growth rates and early and prolific seed set (Grimes 1974 cited in Pettit *et al.* 1995) and many studies within Australia have reported an increase in annual species in areas that were frequently grazed by livestock (Pettit *et al.* 1995; Prober and Thiele 1995; Clarke 2003). Similarly, the higher herb/forb richness in medium density treatments compared to high density treatments may be a response to reduced tree density/cover. Walker *et al.* (1986) reported that forb density, while not influenced by cattle grazing did increase with increasing tree thinning approximately 3 years after the experimental manipulation of original tree densities in the *E. crebra* woodlands of south-east Queensland. However, due to unequal variances caution must be applied when making assumptions relating to differences in growth-form richness between density treatments.

For the box woodlands, the results show that medium density patches are floristically similar to both low and high density patches. This result is different to that found previously where medium density patches were floristically similar to high density but dissimilar to low density (Goodhew 2005). This result may be attributable to site 11 which was shown to be more floristically similar to low density sites in ordination space than the other two medium density sites. Site 11 differs from the other medium density sites in that the native perennial grass *Bothriochloa decipiens* largely dominates the ground layer, similar to low density sites. *Bothriochloa decipiens* is recognised as a widespread grass species and reported to be very resistant to heavy grazing (Henry *et al.* 1995). However, a high proportion of *Bothriochloa decipiens* can indicate a drop in condition on more fertile soils. Therefore, the similarity between these treatments may be the result of a historically similar intensity of grazing.

## 5.1 Conclusion

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After 12 months, the exclusion of large herbivores and livestock from grazing has resulted in little change in the understorey of woodland communities in the Traprock region. The patterns in floristic composition are associated with different mature tree densities and vegetation type, which are consistent with earlier findings (Goodhew 2005). It is suggested that a longer period of exclusion may result in changes in floristic composition. Long-term

monitoring of enclosure plots is essential to determining the biodiversity ‘potential’ of management units.

A long history of livestock grazing and tree removal in the region has contributed to significant differences in floristic composition between management units, yet this is not reflected in differences in total or native species richness between groups. As concluded in earlier research (Goodhew 2005), the heterogenous nature of vegetation management practices in the Traprock region maintains different communities of plant species and therefore, each management unit (or treatment combination) has value in contributing to regional plant diversity in the Traprock region.



## 7. References

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