

# Overstorey tree density and understorey regrowth effects on plant composition, stand structure and floristic richness in grazed temperate woodlands in eastern Australia

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## Abstract

As natural woodlands decline in both extent and quality worldwide, there is an increasing recognition of the biodiversity conservation value of production landscapes. In low-input, low-productivity grazing systems in Australia, the modification of natural woodlands through overstorey tree and woody regrowth removal are vegetation management options used by landholders to increase native grass production for livestock grazing; however, there is little empirical evidence to indicate at what tree densities biodiversity attributes are compromised. We examined the effects of overstorey tree density and understorey regrowth on the floristic composition, stand structure and species richness of eucalypt woodlands in a grazing landscape in the Traprock region of southern Queensland, Australia. We sampled 47 sites stratified according to vegetation type (*Eucalyptus crebra*/*Eucalyptus dealbata* woodland; *Eucalyptus melliodora*/*Eucalyptus microcarpa* grassy woodland), density of mature trees (<6 trees/ha; 6-20 trees/ha; >20 trees/ha), and presence/absence of regrowth. Distinct patterns in composition were detected using indicator species analysis and non-metric multidimensional scaling, with low density areas compositionally indistinguishable, although distinct from other land management units. Within vegetation type, medium tree density woodlands were compositionally similar to high density and reference woodlands. Species richness ranged from 18 to 67 species per 500 m<sup>2</sup> across all sites. No differences in total or native species richness were detected across management units; however, some differences in exotic species richness were detected. Differences in grass cover existed between low and high density management units, yet no difference in grass cover was evident between low and medium density management units. Our results suggest that medium tree densities may provide biodiversity benefits concordant with more natural areas, yet not adversely impact on pasture production. Retaining trees in grazing landscapes provides significant landscape heterogeneity and important refuges for species that may be largely excluded from open grassland habitats. Maintaining a medium density of overstorey trees in grazed paddocks can provide both production and biodiversity benefits.

# 1. Introduction

Biodiversity research has over recent decades recognised the importance of landscape and regional scale conservation, especially in highly modified production landscapes (e.g. McIntyre and Lavorel, 1994; Fensham, 1998; Firbank, 2005; Kirkpatrick et al., 2005; Tschardt et al., 2005; Manning et al., 2006). In some agro-ecosystems where there has been a long history of grazing, landscape biodiversity may be enhanced by low levels of grazing disturbance (McIntyre et al., 2003; Tallowin et al., 2005; Guretzky et al., 2007). However, ecological studies assessing whether different grazing management strategies sustain biodiversity value while retaining long-term productivity are few (Dorrough et al., 2004). In contrast, the negative effects of grazing have been well documented. Grazing may alter the composition of under-storey species (Prober and Thiele, 1995; Clarke, 2003), prevent seedling recruitment (McIntyre and Lavorel, 1994), contribute to soil erosion and compaction (Yates and Hobbs, 1997) and enhance the invasion of exotic species (Prober and Thiele, 1995; Clarke, 2003).

In grazing landscapes, trees have been historically viewed as having negative impacts on grass production (Scholes and Archer, 1997). Trees may compete with grasses for water, soil nutrients, light or a combination of these factors (Scholes and Archer, 1997). Numerous studies (e.g. Walker et al., 1986; McIvor and Gardener, 1995; McIvor, 2001) have shown that tree density is inversely related to pasture yield in many woodland communities, with often a significant increase in pasture yield when all trees are killed (McIvor and Gardener, 1995; McIvor, 2001). However, the initial benefits of increased grass production for livestock grazing may be reduced by the invasion of woody shrubs and other undesirable species and, in some cases, the excessive regrowth of overstorey tree species following clearing (McIvor, 2001). Kaur et al. (2006) recently found that short-term gains from increased pasture production following tree clearing in semi-arid Queensland woodlands did not persist over time and suggested that tree clearing adversely impacted upon soil properties such as availability of nutrients and pasture growth.

Furthermore, some recent studies suggest that the presence of trees in the grazing landscape may not be completely incompatible with production (Tschardt et al., 2005). Retaining mature trees on grazing lands can provide shelter and shade for stock (Walpole, 1999), reduce salinity and land deterioration (McIvor and McIntyre, 2002), enhance soil nutrients and potentially improve the quality of grasses for livestock (Jackson and Ash, 2001). It is now recognised that even scattered paddock trees provide important ecological functions at disproportionate levels relative to their size (Manning et al., 2006). In spite of these benefits, the clearing of remnant vegetation and re-clearing of woody regrowth in grazing landscapes has resulted in the loss and severe modification of large areas of woodlands in eastern Australia (McIvor and McIntyre, 2002). In particular, grassy temperate woodlands that once covered millions of hectares in eastern and southern Australia have largely been cleared for cropping or modified for sheep and cattle grazing (Prober and Thiele, 1995; Prober et al., 2002). In addition, pastoral land management in many grazing systems often involves removing or reducing the tree layer in remaining wooded areas to increase native grass production for livestock grazing (McIvor and McIntyre, 2002). As a consequence, eucalypt woodland remnants with near natural understorey are now rare and often restricted to patches that vary considerably in size, quality and isolation (Prober and Thiele, 1995; Yates and Hobbs, 1997; Prober et al., 2002). It is now clear that in temperate grazing landscapes in Australia, where woodlands have inadequate representation in nature reserves, regional biodiversity conservation may best be achieved if combined with current production systems (Chilcott et al., 1997; Kirkpatrick et al., 2005; Dorrough et al., 2007).

Early evidence suggests there can be both biodiversity and production benefits if some tree cover is retained in grazing landscapes (e.g. Jackson and Ash, 2001). However, there is little empirical information to suggest what overstorey tree density may be appropriate so that both production and conservation goals may be achieved in these agricultural systems. Dorrough et al. (2004) suggested that compatibility between grazing management strategies and biodiversity conservation may be best achieved in low-input grazing systems in low-productivity landscapes. In this study, we use combinations of mature (overstorey) tree density and the presence or absence of woody regrowth in

the understorey as broad surrogates of past vegetation management practices in the Traprock wool-producing region of southern Queensland, Australia. The region is recognised for the production of fine wool fibres by predominantly unimproved native perennial pasture grazing and so represents a suitable landscape in which to explore the compatibility between grazing management and biodiversity conservation. The 'land management units' in the study landscape ('management cells', sensu Dorrough et al., 2004) from the perspective of landholders result from a number of practices which influence the overall structure and composition of vegetation: the broad-scale clearing of vegetation and continual grazing, clearing and burning regimes maintaining open structures (essentially open grassy paddocks with no or very few trees); broad-scale clearing of vegetation and continual re-clearing of trees through ring-barking and/or poisoning, grazing and burning regimes maintaining low shrub (regrowth) structures (no or few trees, high shrub regrowth); broad-scale clearing allowing woody tree regrowth to re-establish and developing into pole stage regrowth (high overstorey tree density, low understorey (shrub) regrowth); maintaining (semi-natural) remnants by minimal selective logging (occasionally for fence posts) and low intensity grazing and low frequency fire regimes (intermediate to high overstorey tree densities, presence of woody shrub regrowth dependent on grazing and fire regimes).

The objective of this study was to determine the effects of vegetation management for livestock grazing (altered overstorey tree density and understorey woody regrowth) on floristic composition, cover and plant species richness of two eucalypt woodland communities that previously dominated the landscape. Specifically, we examined the following questions: is there a difference in floristic composition, stand structure and species richness among mature tree density classes? Do woody regrowth areas have a different floristic composition, stand structure and species richness compared to areas without woody regrowth? Are there differences in vegetation responses across eucalypt woodland vegetation types?

## **2. Methods**

### ***2.1. Study area***

The study was undertaken in the Traprock wool-growing region of southern Queensland, in the northernmost part of the temperate zone in eastern Australia. The region lies between the major towns of Warwick and Stanthorpe to the east and Inglewood and Texas to the west (Fig. 1). The climate is influenced by both tropical and temperate weather patterns with mean annual minimum and maximum temperatures ranging between 2.6 and 30.2 °C for Warwick (28°22'S, 152°03'E) and 0.9 and 27.4 °C for Stanthorpe (28°66'S, 151°89'E) (Queensland Murray Darling Committee, 2004; Bureau of Meteorology, 2007). Rainfall is generally higher in summer months, although the winter proportion can be significant, with mean annual rainfall ranging from 701 to 770 mm (Bureau of Meteorology, 2007). The original vegetation of the region has been subjected to clearing by ring-barking more than 80 years ago and many sites were re-cleared (regularly until about 30 years ago) to control woody regrowth (Traprock Wool Association 2005, pers. comm.). While less intense than the broad scale clearing of mature trees, the clearing of woody regrowth vegetation remains a component of many management practices employed by landowners (Traprock Wool Association 2005, pers. comm.). The Traprock region supports approximately 300,000 ha of sheep grazing country at a stocking rate of about 1–2 DSE (dry sheep equivalent) per hectare (Queensland Murray Darling Committee, 2004). Native pastures are the main source of forage within the study area, with past land management practices also including regular burning to control woody regrowth and to remove unpalatable dry herbage (Wills, 1976), although preliminary landholder surveys suggested that the use of fire to reduce woody regrowth had not been a significant management practice in recent (~20) years. The

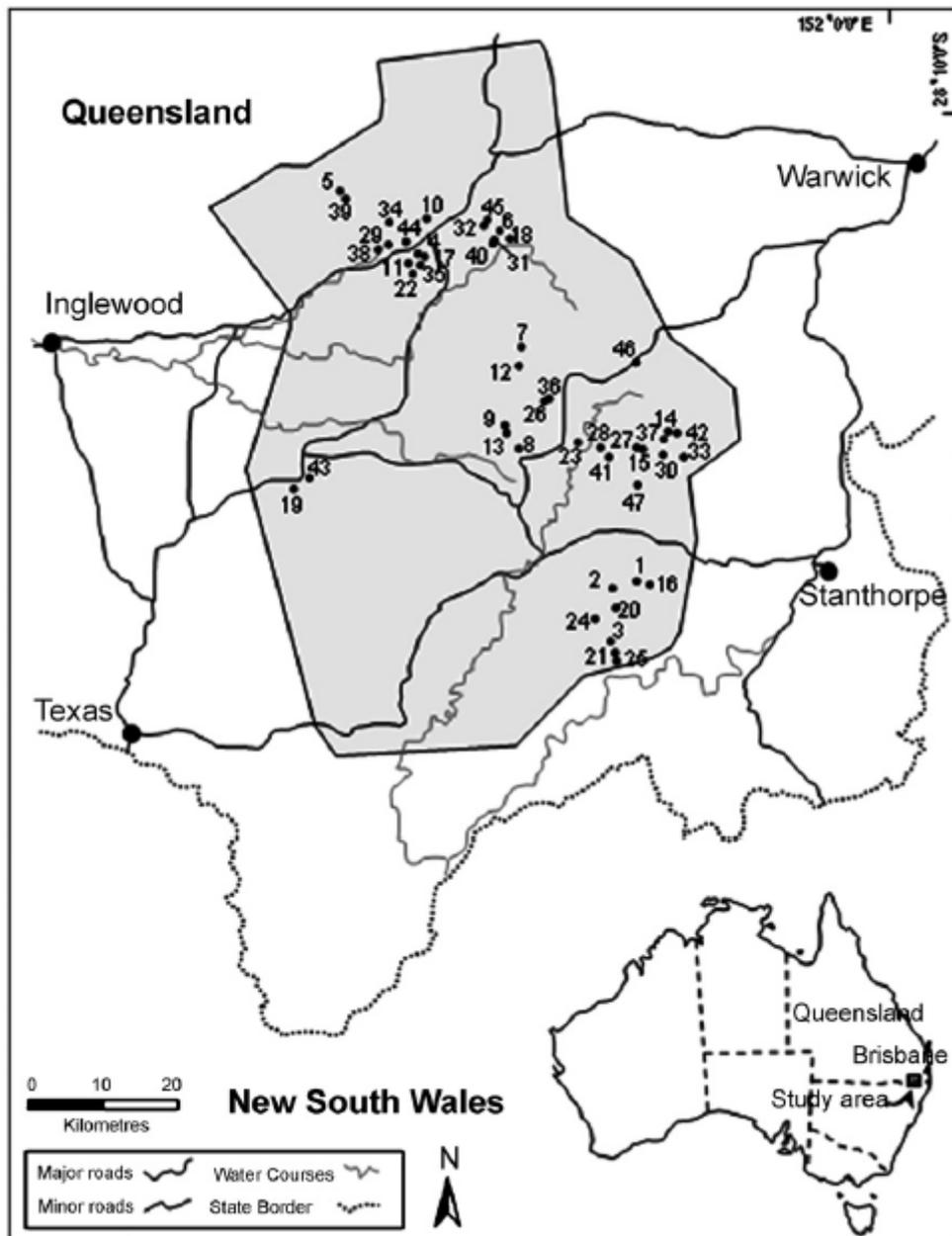


Fig. 1. Map of Traprock wool-growing region (shaded area) showing location of study sites.

vegetation is predominately grassy eucalypt woodland mainly comprised of narrow-leaved ironbark (*Eucalyptus crebra* F.Muell.), tumbledown gum (*Eucalyptus dealbata* A.Cunn. ex Schauer), and occasional white box (*Eucalyptus albens* Benth.) (Queensland Herbarium Regional Ecosystem type 13.11.3, Sattler and Williams, 1999) and, on the lower slopes, yellow box (*Eucalyptus melliodora* A.Cunn. ex Schauer), greybox (*Eucalyptus microcarpa* Maiden) or gum topped box (*Eucalyptus moluccana* Roxb.) (Regional Ecosystem type 13.11.8, Sattler and Williams, 1999).

Study sites were stratified in land management units across the study area according to dominant vegetation type (*E. crebra*/*E. dealbata* 'ironbark/gum' woodland; *E. melliodora*/*E. microcarpa*/*E. moluccana* grassy 'box' woodland), density of mature trees (low, medium or high), and presence/absence of woody regrowth in the understorey. Floristic differences between the two dominant vegetation types (ironbark/gum woodlands and box woodlands) reflect their relative position in the landscape (Sattler and Williams, 1999) and have been related to differences in soil drainage patterns (Wills, 1976). While some variation in over-storey tree density could be expected in these systems prior to European settlement, tree densities have been significantly modified by past practices. Wills (1976) reported that between 60 and 100% of original timber on an average Traprock

property (approximately 2000 ha) was often killed. Excessive tree and shrub woody regrowth in these grassy woodlands (particularly suckering of Eucalyptus species) generally reflects a response to past land clearing and grazing regime practices (Wills, 1976; Queensland Murray Darling Committee, 2004).

Mature tree density classes were derived subjectively from satellite (Spot V) imagery to sample the range of densities in the study landscape: low (<6 trees/ha), medium (6–20 trees/ha) or high (>20 trees/ha). Sites were 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 >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). Regrowth age was determined by landholder survey and confirmed with on-ground assessment. Sites within each of these management units were sampled from patches at least 5 ha in size and were located at least 250 m from stock watering points to minimise possible effects. Although at least two replicates of each management unit were sampled, a balanced design was not possible due to limited representation (and absence) of some management units within the study area. Thirteen management units were recognised (Table 1) and a total of 47 sites were sampled, including four reference sites (Fig. 1). Three reference sites were located within road reserves, and one within State Forest. These sites were chosen to represent woodlands with

Management unit description			Code	n	Site numbers
Overstorey tree density	Understorey regrowth	Vegetation type			
Low density	No regrowth	Ironbark/gum woodland	LN-IG	5	1, 2, 3, 4, 5
Low density	Regrowth	Ironbark/gum woodland	LR-IG	5	6, 7, 8, 9, 10
Low density	No regrowth	Box woodland	LN-B	5	11, 12, 13, 14, 15
Low density	Regrowth	Box woodland	LR-B	4	16, 17, 18, 19
Medium density	No regrowth	Ironbark/gum woodland	MN-IG	4	20, 21, 22, 23
Medium density	Regrowth	Ironbark/gum woodland	MR-IG	4	24, 25, 26, 27
Medium density	No regrowth	Box woodland	MN-B	3	28, 29, 30
High density	No regrowth	Ironbark/gum woodland	HN-IG	4	31, 32, 33, 34
High density	Regrowth	Ironbark/gum woodland	HR-IG	2	35, 36
High density	Pole stage regrowth	Ironbark/gum woodland	HRP-IG	4	37, 38, 39, 40
High density	No regrowth	Box woodland	HN-B	3	41, 42, 43
Reference	n/a	Ironbark/gum woodland	Ref-IG	2	44, 45
Reference	n/a	Box woodland	Ref-B	2	46, 47

*Table 1*

*Description of management units, abbreviated description (code), number (n) of replicates for each management unit and site numbers*

minimal grazing impact. However, they may still be considered disturbed as they have been subjected to some modification from grazing, altered fire regimes and, to a lesser extent, selective low-intensity logging, in the past.

## **2.2. Floristic composition, stand structure and cover**

Sites were sampled between September 2004 and March 2005. At each site, a 500 m<sup>2</sup> 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 composition and relative abundance of plant species was determined using the frequency score method (after Morrison et al., 1995). Each 500 m<sup>2</sup> quadrat was sub-divided into eight concentrically nested sub-quadrats of increasing area (1, 2, 5, 10, 50, 100, 200 and 500 m<sup>2</sup>) and the presence of all vascular plant species was used to determine a frequency score (number of sub-quadrats in which species occurs). Such data have been shown to be functionally equivalent to traditional frequency measures (Morrison et al., 1995). Plant species were identified using the nomenclature of Harden

(1991) and Stanley and Ross (1983, 1986, 1989). 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 (7% of total taxa), due to lack of any distinguishing features were excluded from analyses. Exotic species were distinguished as any plant species that has been introduced into Australia since European settlement and identified using the nomenclature of Stanley and Ross (1983, 1986, 1989).

Where possible, species were also classified according to the grazing response functional typology of McIntyre et al. (2003). This typology consisted of (a) grazing increasers: species exhibiting a positive (continuous) response to grazing; (b) low grazing intolerants: species that require grazing to the extent that its abundance is significantly reduced by low grazing pressure; (c) grazing generalist: no preferences across a grazing gradient exhibited; (d) intermediate response species: those species that require moderate levels of grazing; and (e) grazing decreaseers: species that decline in abundance with grazing. This information was included in the description of indicator species (defined here as species common to a particular group, by occurring in more than 60% of sites in that group, but largely absent from others by occurring in less than 20% of sites in other groups).

Site stand structure was determined using the modified Specht (1981) structural classification scheme (after Le Brocque and Buckney, 1997). Based on the vegetation of the study area, six strata were pre-defined: trees 10–30 m, trees <10 m, shrubs >2 m, shrubs <2 m, forbs/herbs/other (non-woody species), and grami-noids (including grasses, sedges and others). 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 estimated within the 500 m<sup>2</sup> quadrat. Data were pooled across some strata to derive total tree cover and total shrub cover.

### **2.3. Numerical analyses**

Indicator species analysis (ISA; TWINSPAN program, Hill, 1979) was performed on the frequency score data to determine whether a priori groups (13 management units; Table 1) were similar in terms of species composition. Non-metric multidimensional scaling (nMDS) was performed on Bray–Curtis similarities calculated from the frequency score data using the Primer v.5.2.9 for Windows program (Primer-E Ltd, 2001). Species data were log transformed to allow mid-range and rarer species to exert some influence on the calculation of similarity and cover data were arcsine transformed to improve normality and reduce the influence of large values (Clarke and Warwick, 2001). Analysis of similarity (ANOSIM) was performed on the frequency score data to determine if there were differences in similarity between a priori groups (Clarke and Gorley, 2001). SPSS<sup>1</sup> for Windows (Version 12.0.1, SPSS Inc., 2003) was used to perform one-way analysis of variance (ANOVA) to determine if there were group differences in total species richness, native species richness, and exotic species richness. All species richness data were log transformed to improve normality. Prior to performing one-way ANOVA, the Levene's statistic and residual plots were used to test for homogeneity of variances. 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 when significant differences were indicated by one-way ANOVA. Spearman-rank correlations were performed to determine whether cover variables (e.g. tree and grass cover) were related and the degree to which they vary together.

## **3. Results**

### **3.1. Floristic composition**

A total of 202 plant taxa from 53 families were recorded in the study. Indicator species analysis of floristic composition clustered the 47 samples into four main groups that generally matched a priori

management units, with a broad separation of low and medium tree density woodlands from regrowth and high tree density woodlands (Fig. 2). Woodlands with a low tree density and no woody regrowth (LN-IG and LN-B), regardless of vegetation type, were floristically similar. Medium tree density samples with no regrowth for both vegetation types (MN-IG and MN-B), and high tree density no regrowth box woodlands (HN-B) represented another cluster (Fig. 2). Woody regrowth samples (LR-IG, LR-B, MR-IG, and HR-IG) were generally similar in floristic composition, regardless of tree density. High tree density and no regrowth ironbark/gum woodlands (including Ref-IG, HN-IG and HRp-IG) were similar to each other, although this cluster also contained two low density regrowth ironbark/gum woodland (LR-IG) samples (Fig. 2).

nMDS ordination of floristic composition data (Fig. 3a) indicated a general gradient of increasing mature tree density (left to right across the ordination). Low tree density no regrowth samples were well separated from low density regrowth and medium density woodlands and high tree density woodlands (Fig. 3a). Analysis of similarity showed no significant differences in the similarity between low density no regrowth ironbark/gum woodlands and low density no regrowth box woodlands ( $R = 0.176$ ; Table 2). With a few exceptions, such as low tree density no regrowth samples, ANOSIM results reveal significant differences in similarity between box woodlands and ironbark/gum woodlands within any mature tree density or regrowth/no regrowth management units (Table 2). On this basis, the data were divided into ironbark/gum woodland samples and box woodland samples and nMDS ordinations were performed on each subset.

The nMDS ordination of the ironbark/gum woodland subset showed a general gradient in floristic composition in no regrowth woodlands with increasing mature tree density (Fig. 3b). Regrowth ironbark/gum woodlands cluster towards the centre of the ordination, regardless of tree density. There were no differences in similarity between high density woodlands and between these and medium density regrowth woodlands (Table 2). Medium density no regrowth ironbark/gum woodlands were similar to other medium and high density woodlands except for high density no growth and high density old regrowth ironbark/gum woodlands. Low tree ironbark/gum density woodlands were significantly different, but not different to medium tree density woodlands within the same regrowth/no regrowth management unit (Table 2).

The nMDS ordination of box woodlands showed a clear separation of low density no regrowth box woodland from all other woodlands (Fig. 3c; Table 2). While there was some separation of the remaining box woodlands, ANOSIM results showed low density regrowth box woodlands were not significantly different to medium density no regrowth or reference box woodlands, but different to high density no regrowth box woodlands (Table 2). High density no regrowth box woodlands were not different to medium density no regrowth box woodlands or reference woodlands (Table 2).

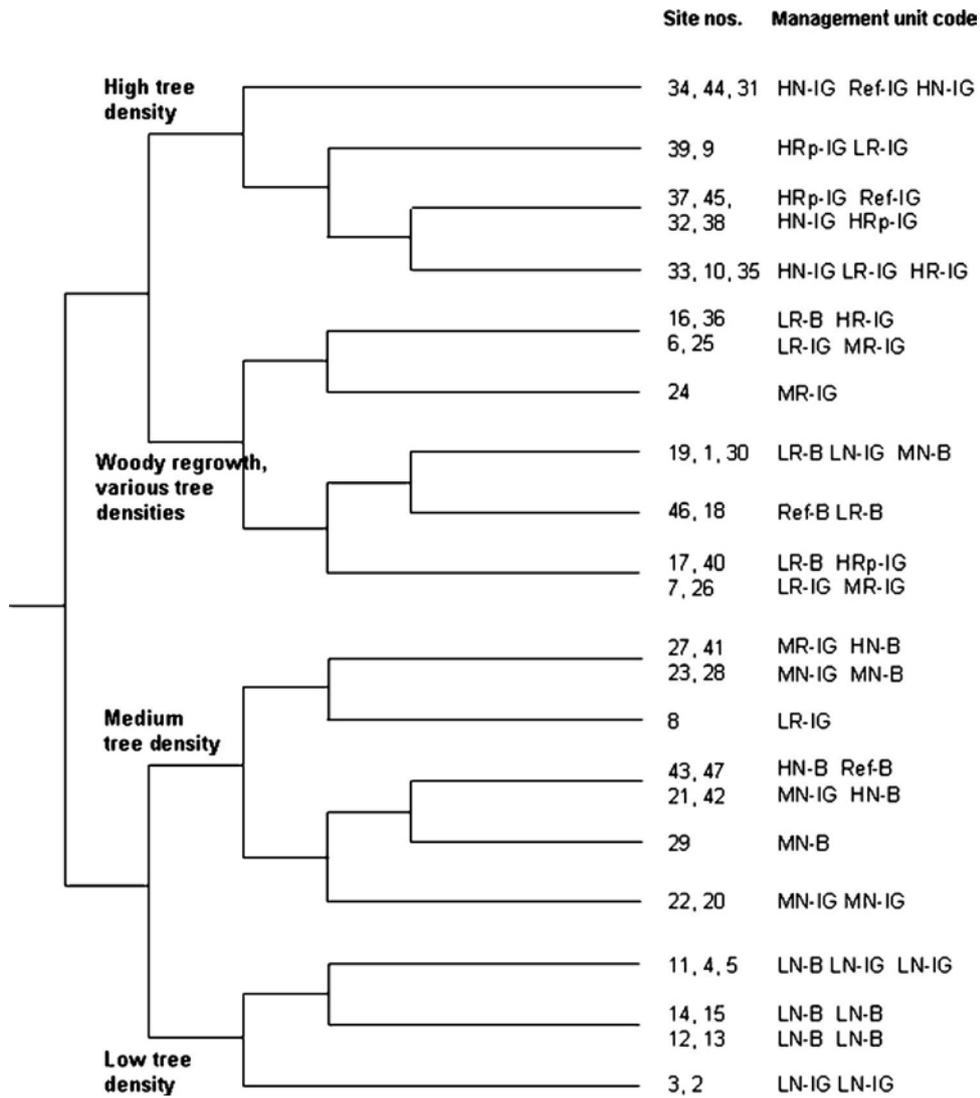


Fig. 2. Dendrogram showing results from ISA analysis of floristic composition data. Site numbers and corresponding descriptive labels are shown. See Table 1 for descriptions of management unit codes.

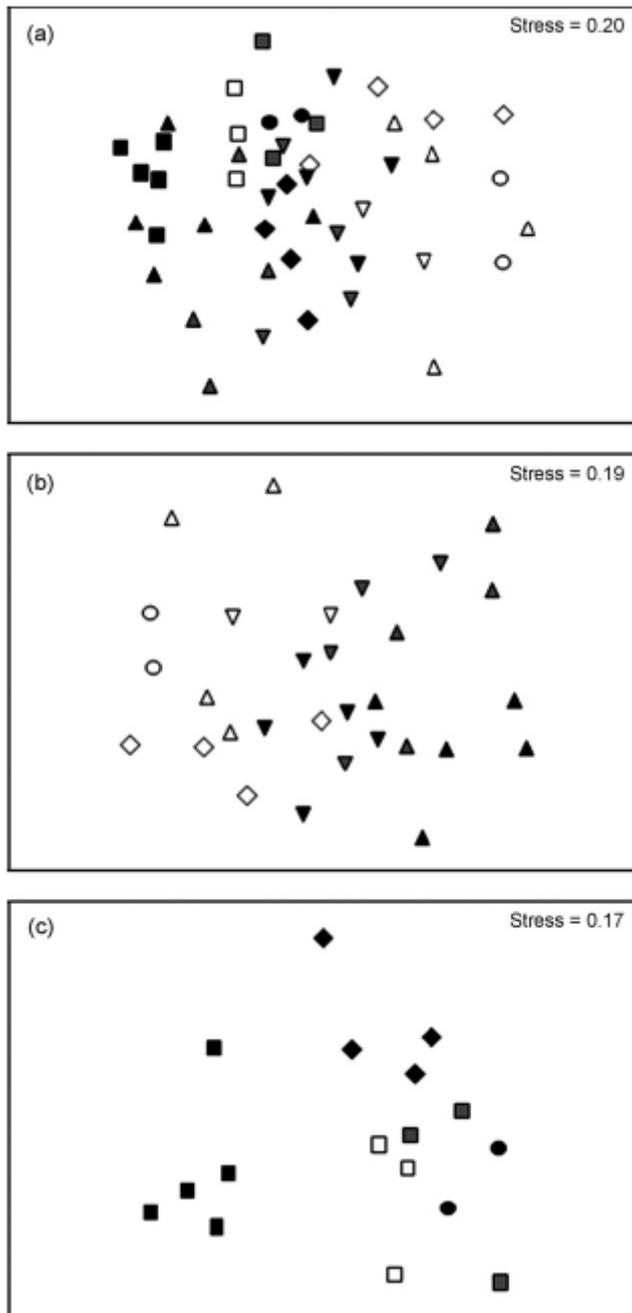


Fig. 3. Non-metric multi-dimensional scaling ordination of floristic data (a) across all sites; (b) ironbark/gum woodlands subset; and (c) box woodlands subset. Stress is a measure of the goodness-of-fit of the nMDS ordination; values of  $<0.20$  indicate that the two-dimensional summary is a good representation of sample relationships (Clarke and Warwick, 2001).

See Table 1 for descriptions of management unit codes: LN-IG (A), LN-B (■), LR-IG (▼), LR-B (◆), MN-IG (A.), MN-B (Q), MR-IG (^), HN-IG (A), HN-B(D), HR-IG (y), HRp-IG(O), Ref-B (#) and Ref-IG (o).

### 3.2. Grazing response

Following the grazing response functional typology of McIntyre et al. (2003), species characterising low density no regrowth samples in both ironbark/gum woodlands and box woodlands include grazing increasers and grazing generalists (Table 3). *Crassula sieberiana* (annual native forb), *Sisyrinchium micranthum* (exotic annual forb) and *Chloris divaricata* (native perennial grass) are taxa that increase in abundance with increasing grazing pressure (i.e. grazing increasers). The indicator

species for these samples also include grazing generalists: *Conyza sumatrensis* (exotic annual forb), *Fimbristylis dichotoma* (native perennial grass) and *Sporobolus creber* (native perennial grass) (Table 3). Low and medium density regrowth ironbark/gum woodlands were characterised by grazing generalist species: *Cheilanthes sieberi* (native perennial rhizomatous fern), *Calotis lappulacea* (native perennial forb) and *Lomandra filiformis* (native perennial forb), and the grazing increaser *Conyza bonariensis* (exotic annual forb) (Table 3). Medium density no regrowth ironbark/gum woodlands were characterised by the low grazing intolerant exotic perennial *Paronychia brasiliana* and tree species *E. albens*. Medium and high density no regrowth box woodlands were characterised by the grazing decreaser *Indigofera baileyi* and the native species *E. melliadora*, *Boerhavia dominii* (perennial forb), *Enneapogon gracilis* (perennial grass), *Rostellularia adscendens* (perennial forb), *Eremo-phila debilis* (perennial subshrub), and *Pratia concolor* (perennial forb). High density ironbark/gum woodlands were characterised by a number of native grazing decreaseers, *Jasminum suavisissimum* (perennial forb), *Laxmannia gracilis* (perennial forb) and *Lomandra longifolia* (perennial forb), grazing generalist *Panicum effusum* (perennial grass) and a number of tree and shrub species (Table 3).

### **3.3. Species and growth form richness**

Total species richness per sample ranged from 18 to 67 species with an average richness of 39 species per 500 m<sup>2</sup>. Comparison of mean total species richness showed no significant differences among groups (oneway ANOVA; d.f. = 12, 34;  $F = 1.36$ ;  $p > 0.05$ ; Fig. 4a). Similarly, mean native species richness was not significantly different across woodlands (d.f. = 12, 34;  $F = 1.42$ ;  $p > 0.05$ ; Fig. 4b). Low density no regrowth box woodland had a significantly higher exotic species richness than most ironbark/ gum woodlands and low density regrowth box woodland, although it was not different to that of other box woodlands (d.f. = 12, 34;  $F = 3.08$ ;  $p < 0.05$ ; Fig. 4c).

### **3.4. Stand structure**

Mean total tree cover generally increased as mature tree density ranged from low to high, ranging from 0% (LN-B) to 58% (HN-B) (Fig. 5a). Low density no regrowth woodlands (LN-IG and LN-B) had little tree cover and were significantly different to most other woodlands (with the exception of MN-IG and MN-B) (oneway ANOVA; d.f. = 8, 24;  $F = 11.45$ ;  $p < 0.05$ ). The mean cover of trees 10–30 m in height similarly ranged between 0% (LN-B) and 56% (HN-B), with overstorey tree cover higher in higher density woodlands (d.f. = 9, 28;  $F = 9.74$ ; Fig. 5b). Mean total shrub cover ranged from 2% (LR-B) to 39% (HR-IG) and was higher for regrowth and ironbark/gum woodlands (d.f. = 8, 25;  $F = 32.56$ ; Fig. 5c).

Mean grass cover ranged between 3% (Ref-IG) and 71% (LN-B) across woodlands (Fig. 5d). Low density no regrowth box woodland had a significantly higher grass cover than most other woodlands (d.f. = 12, 34;  $F=8.90$ ; Fig. 5d). Forb cover was generally low across all management units ranging from 2% (REF-B) to 15% (LN-IG) and was generally higher in no regrowth woodlands (d.f. = 12, 34;  $F = 3.47$ ; Fig. 5e).

Grass cover declined significantly with increasing total tree cover ( $r_s = -0.510$ ;  $p < 0.001$ ) and cover of trees 10-30 m height ( $r_s = -0.392$ ;  $p < 0.01$ ). Similarly, forb and herb cover declined with increasing total tree cover ( $r_s = -0.489$ ;  $p < 0.001$ ). Shrub cover was positively correlated with trees 10-30 m height ( $r_s = 0.450$ ;  $p < 0.001$ ), but not with total tree cover ( $p > 0.05$ ).

Summary of global ANOSIM tests												
Complete data set: global $R = 0.494$ (significant at $p = 0.001$ )												
Ironbark/gum woodland sub set: global $R = 0.423$ (significant at $p = 0.001$ )												
Box woodland sub set: global $R = 0.641$ (significant at $p = 0.001$ )												
Summary of pairwise comparisons ( $R$ values)												
	LN-IG	LR-IG	LN-B	LR-B	MN-IG	MR-IG	MN-B	HN-IG	HR-IG	HRP-IG	HN-B	Ref-IG
IR-IG	0.46**											
LN-B	0.18	0.86**										
LR-B	0.29	0.31	0.87**									
MN-IG	0.23	0.51**	0.54**	0.28								
MR-IG	0.31	0.11	0.74**	0.02	-0.05							
MN-B	0.49*	0.29*	0.85*	0.20	0.28	0.32						
HN-IG	0.77**	0.50**	0.93**	0.62 <sup>†</sup>	0.54*	0.28	0.35					
HR-IG	0.64*	0.46	1.00*	0.43	0.36	-0.21	0.50	-0.04				
HRP-IG	0.65*	0.34	0.91**	0.63*	0.67*	0.46	0.48	0.10	0.46			
HN-B	0.40*	0.59*	0.87*	0.61*	0.22	0.41	0.15	0.37	0.92	0.72		
Ref-IG	0.89*	0.78*	1.00*	0.96	0.96	0.93	1.00	-0.25	0.50	0.68	1.00	
Ref-B	0.64*	0.44	0.89*	0.32	0.07	<0.01	<0.01	0.18	0.50	0.61	0.67	1.00

See Table 1 for descriptions of management unit codes.  $R$  values are significant at \* $0.05 < p < 0.01$ ; \*\* $p < 0.01$ ).

**Table 2**  
Summary of results from analysis of similarity (ANOSIM) of floristic composition

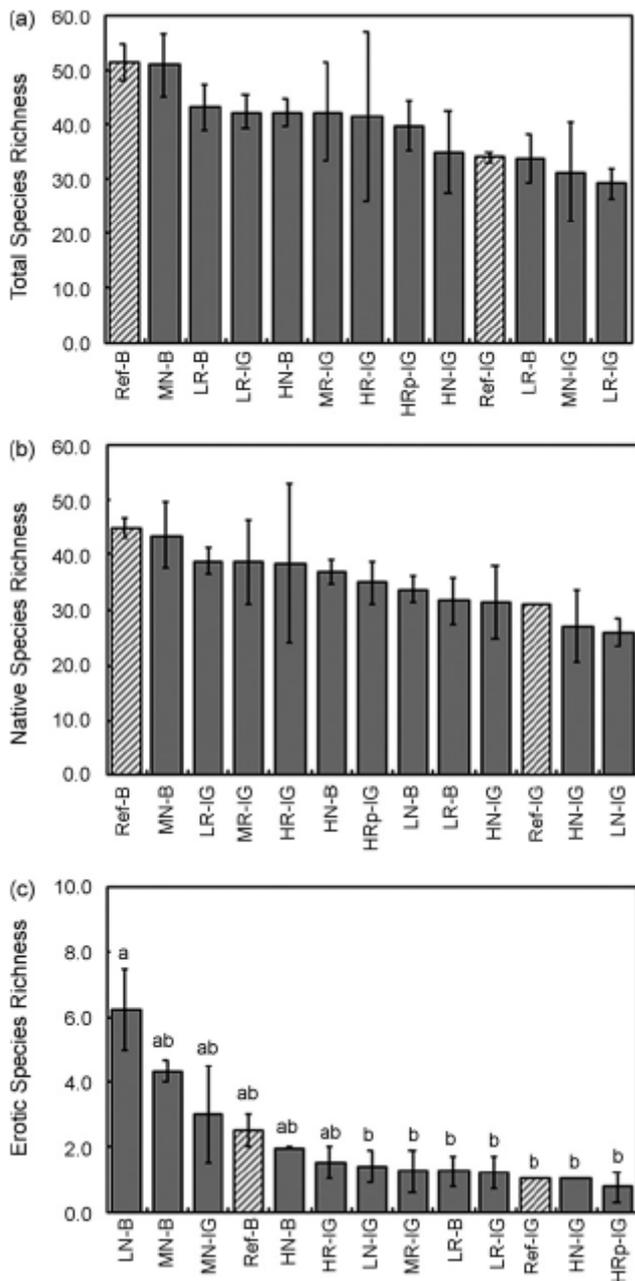
Group(s)	Indicator Species	Growth form	Grazing response
LN-B, LN-IG	<i>Chloris divaricata</i> R.Br.	Graminoid	Increaser
	<i>Fimbristylis dichotoma</i> (L.) Vahl	Graminoid	Generalist
	<i>Sporobolus creber</i> De Nardi	Graminoid	Generalist
	<i>Conyza sumatrensis</i> (Retz.) E.Walker <sup>a</sup>	Forb	Generalist
	<i>Crassula sieberiana</i> (Schult. and Schult.f.) Druce	Forb	Increaser
	<i>Gnaphalium</i> sp. J ( <i>aff. sphaericum</i> ) no auth.	Forb	
	<i>Richardia stellaris</i> Cham. & Schltdl.) Steud. <sup>a</sup>	Forb	Intolerant
	<i>Seyrinchium micranthum</i> Cav. <sup>a</sup>	Forb	Increaser
	<i>Vittadinia sulcata</i> N.T.Burb.	Forb	Intermediate
	LR-IG, MN-IG, MR-IG	<i>Lomandra filiformis</i> (Thunb.) Britten	Graminoid
<i>Calotis lappulacea</i> Benth.		Forb	Generalist
<i>Cheilanthes sieberi</i> Kunze		Forb	Generalist
<i>Conyza bonariensis</i> (L.) Cronquist <sup>a</sup>		Forb	Increaser
<i>Evolvulus alsinoides</i> var. <i>decumbens</i> (R.Br.) Ooststr.		Forb	
<i>Glycine clandestina</i> J.C.Wendl.		Forb	Intermediate
<i>Acacia blakei</i> Pedley		Shrub	
<i>Cassinia quinquefaria</i> R.Br.		Shrub	
<i>Melichrus urceolatus</i> R.Br.		Shrub	
<i>Notelaea microcarpa</i> R.Br.		Shrub	
HN-B, MN-B, Ref-B	<i>Enneapogon gracilis</i> (R.Br.) P.Beauv.	Graminoid	Increaser
	<i>Boerhavia dominii</i> Meikle & Hewson	Forb	
	<i>Indigofera baileyi</i> F.Muell.	Forb	Decreaser
	<i>Geranium solanderi</i> Carolin	Forb	Intermediate
	<i>Rostellularia adscendens</i> (R.Br.) R.M.Barker	Forb	
	<i>Pratia concolor</i> (R.Br.) Druce	Forb	
	<i>Eremophila debilis</i> (Andrews) Chinnock	Shrub	
	<i>Eucalyptus melliodora</i> A.Cunn. ex Schauer	Tree	
HN-IG, HRP-IG, HR-IG, Ref-IG	<i>Lomandra longifolia</i> Labill.	Graminoid	Decreaser
	<i>Panicum effusum</i> R.Br.	Graminoid	Generalist
	<i>Jasminum suavissimum</i> Lindl.) P.S.Green <sup>b</sup>	Forb	Decreaser
	<i>Laxmannia gracilis</i> R.Br.	Forb	Decreaser
	<i>Cassinia laevis</i> R.Br.	Shrub	
	<i>Leptospermum brevipes</i> F.Muell.	Shrub	
	<i>Acacia fimbriata</i> A.Cunn. ex G.Don	Shrub	
	<i>Acacia salicina</i> Lindl.	Tree	
MN-IG	<i>Paronychia brasiliiana</i> DC. <sup>a</sup>	Forb	Intolerant
	<i>Eucalyptus albens</i> Benth.	Tree	
LR-B	None		

Management units are grouped based on results of ISA and nMDS analyses. Grazing responses (where available) are taken from McIntyre et al. (2003) (see text for description of responses).

<sup>a</sup> Exotic species.

<sup>b</sup> Formerly *Jasminum simplicifolium* of McIntyre et al. (2003).

**Table 3**  
Summary of 'indicator' species, their growth form and previously reported grazing response



**Fig. 4.** Mean species richness across management units for (a) total species; (b) native species; and (c) exotic species. Management units are ordered from highest to lowest within each graph; reference woodlands are indicated by different shading. Means are species per 500 m<sup>2</sup> and are not significantly different for total ( $p > 0.05$ ) and native species ( $p > 0.05$ ), but different for exotic species ( $p < 0.05$ ). For exotic species richness, management units with same letter are not significantly different (Tukey's test,  $p > 0.05$ ). Error bars are standard errors.

## 4. Discussion

### 4.1. Effects on floristic composition and richness

The results from this study suggest that vegetation management practices, particularly the control of overstorey and under-storey regrowth, significantly influence floristic patterns within the sheep grazing region of southern Queensland. Low overstorey (mature) tree density no regrowth woodlands were similar to each other and largely different to other management units in terms of floristic

composition. Compositional differences were more pronounced for low density no regrowth box woodlands, which were different to all other box woodlands. These areas are structurally very simple systems, with a virtual absence of shrub and tree strata. Not surprisingly, the low density no regrowth woodlands were characterised by indicator species mostly represented by grazing tolerant species (grazing increasers and grazing generalists), high numbers of annual species and high exotic species richness (at least for low density no regrowth box woodland). These grazing tolerant species have been shown to respond positively to grazing in other systems (McIntyre et al., 2003). While some caution needs to be applied when using grazing responses due to inconsistencies in many species responses across regions (see Vesk and Westoby, 2001), such functional descriptions of components of plant communities may be useful to elucidate local scale compositional changes, particularly in response to modified management practices. Similar results have been reported for other studies in Australia, where continued 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 (Chilcott et al., 1997; McIntyre and Martin, 2001; Clarke, 2003; McIntyre et al., 2003). At medium and high mature tree densities, differences in floristic composition between the eucalypt woodland vegetation types became more evident. Our results suggest that the two vegetation types respond differently to management practices where trees are retained. For example, medium density no regrowth box woodlands had a similar floristic composition to high density box woodlands (HN-B and Ref-B), whereas the medium density no regrowth ironbark/gum woodlands were more similar to low density no regrowth ironbark/gum woodland, but different to high density and pole stage ironbark/gum woodlands.

A distinct gradient in floristic composition within ironbark/gum woodlands was evident from low density no regrowth through low and medium density regrowth and medium density no regrowth to high density woodlands. High density ironbark/gum woodlands were characterised by a number of grazing decreaseers and may represent important refuges for these species in this landscape. Numerous studies (e.g. Landsberg et al., 2002; McIntyre et al., 2003; Dorrough et al., 2004) have recognised the importance of less grazed, more wooded remnant areas for landscape-scale diversity. Medium density regrowth ironbark/gum woodlands and high density woodlands were also similar in terms of floristic composition, suggesting that allowing understorey regrowth in medium tree density areas can result in a floristic composition similar to that shown by areas with higher tree density in this vegetation type. A landscape consisting of an overstorey of mature trees at a medium density may facilitate the re-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). However, based on our results, this may require qualification in that medium mature tree densities, coupled with allowing some degree of understorey regrowth, may be necessary for the facilitation of plants species establishment and persistence in ironbark/gum woodland vegetation. The presence or absence of regrowth in the understorey clearly influences broad vegetation patterns in these woodlands.

Low density no regrowth box woodlands similarly exhibited strong differences in floristic composition to regrowth and higher mature tree density samples in the same vegetation type; however, unlike in the ironbark/gum woodlands, medium density no regrowth box woodlands essentially had the same composition as high density and reference box woodlands. Low density no regrowth box woodlands contained no tree species, very few shrub species and a generally higher proportion of forb species and higher exotic species richness (although the latter was not significantly different to medium and high density box woodlands).

While few previously published studies report grazing responses for indicator species of medium and high mature tree density box woodlands, these woodlands are characterised by species likely to be relatively sensitive to grazing, including *E. debilis* and *E. melliodora*, and the previously recognised grazing decreaseer *I. baileyi* (McIntyre et al., 2003). The lack of available regrowth woodlands in medium and high mature tree density box woodlands poses some difficulty in assessing whether allowing understorey regrowth in these woodlands would result in greater similarity to more 'natural' box woodlands. The natural grassy understorey of box woodlands (Wills, 1976) may not have been as considerably altered by grazing and tree removal compared to the ironbark/gum woodlands; however,

grazing and tree removal practices do result in a change in understorey composition when both tree removal and grazing intensity are increased in the box woodlands (i.e. low density no regrowth box woodlands).

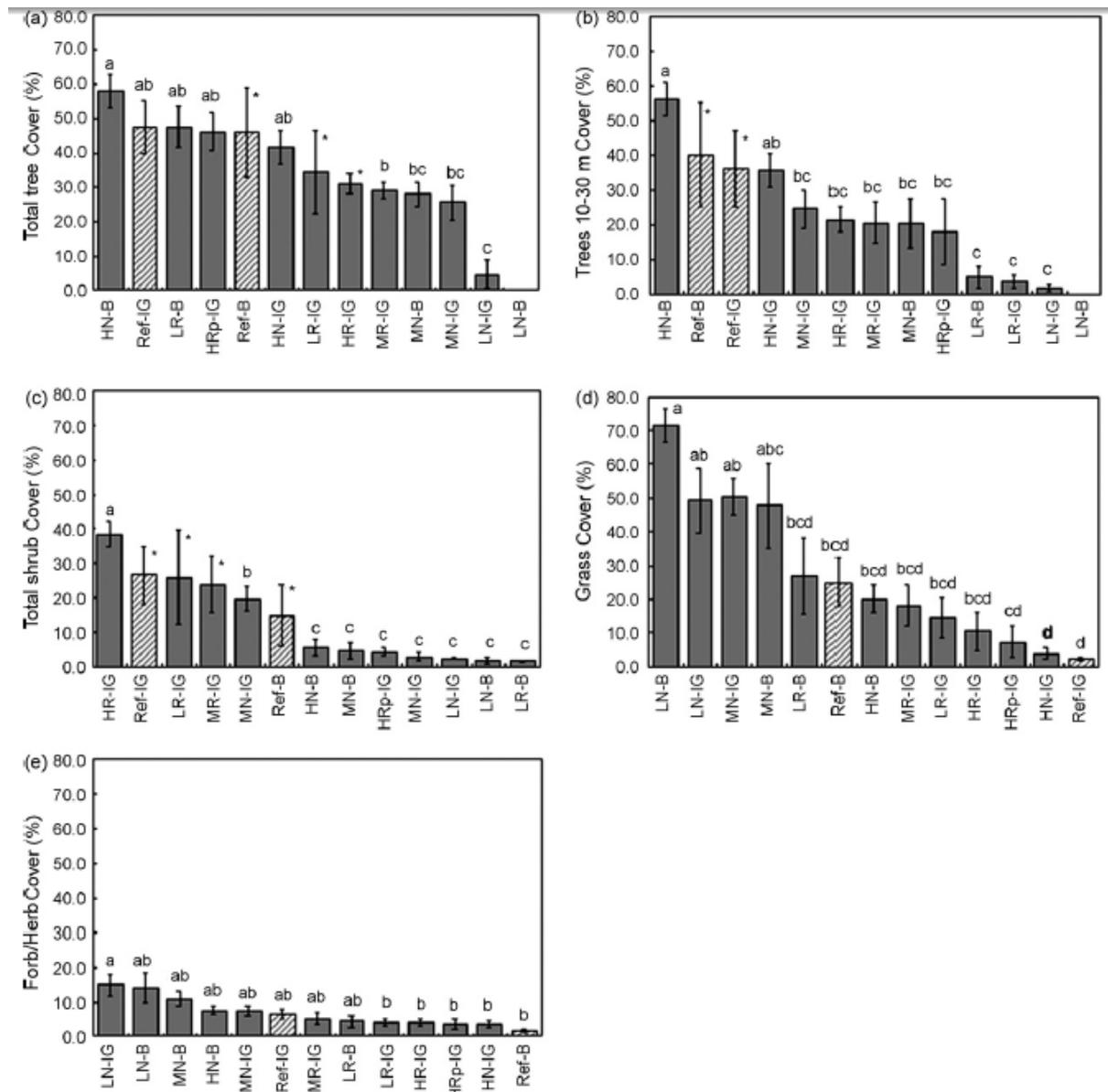


Fig. 5. Mean foliage projective cover (%) of (a) total trees; (b) trees 10–30 m height; (c) total shrubs; (d) grasses and (e) herbs/forbs across management units. Management units are ordered from highest to lowest within each graph; reference woodlands are indicated by different shading. Management units 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.

While there were clear floristic patterns associated with different tree densities and the presence/absence of understorey regrowth, these were not reflected in differences in total or native species richness between management units. 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, our results are similar to those of McIntyre and Martin (2001) where plant species richness was unaffected by lithology, slope, or tree density in the temperate woodlands of southern Queensland. The number of plant species recorded

during our 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 our study, mean species richness across all woodlands was 39 per 500 m<sup>2</sup>, equivalent to about 23 species per 20 m<sup>2</sup> as determined from species-area relationships for these woodlands (Goodhew, 2005). Lunt (1997) similarly found that maintaining different management regimes in the grassy forest remnants of southeastern Australia resulted in different suites of plant species, but little change in overall species richness.

## **4.2. *Effects on ground cover***

Our results accord with previous studies that have found a generally decreasing cover of grasses and forbs with increasing cover of trees and shrubs (e.g. Walker et al., 1986; McIvor and Gardener, 1995; McIvor, 2001; Peterson et al., 2007). Walker et al. (1986) reported that where trees were absent, herbage biomass (primarily native grasses) was significantly higher in the grassy woodlands of southern Queensland. Other studies have documented similar findings elsewhere (e.g. Scanlan, 2002). Correspondingly, vegetation management in temperate and semi-arid grazing landscapes in Australia has been centred on the premise that trees compete with palatable grasses and forbs (Scanlan, 2002). However, while grass and forb cover declined overall with increasing tree cover, there was no difference in cover between low and medium density no regrowth woodlands for the same vegetation type. In the absence of woody regrowth in the understorey, medium densities of mature trees (6–20 trees/ha) have the same cover of grass and forbs as open paddock areas. This finding is significant because maintaining a medium density of overstorey trees can provide the same production value (in terms of grass and forb biomass) as more open paddock areas.

Jackson and Ash (2001) found that trees, through their ability to increase soil nutrient status, can positively influence the quality of grasses by enhancing dry matter digestibility and leaf nitrogen content in open forests of northern Queensland. This increase in the forage quality of grasses was proposed to lead to improved diet quality and feed intake for livestock (Jackson and Ash, 2001). In addition, Walpole (1999) found that the value of pasture output may be increased by having a proportion of pasture area under dry sclerophyll or woodland vegetation in northern New South Wales. Woodlands provide a number of important ecosystem services for agricultural production, such as contribution to soil formation and protection, nutrient storage and cycling, natural control of diseases and parasites, and the breakdown and absorption of pollutants (McIvor and McIntyre, 2002). Woodlands also provide critical wildlife habitats (e.g. for possums, owls, and bats) (Lumsden and Bennett, 2005), and habitat diversity, which may facilitate the establishment of different native plant species (Chilcott et al., 1997; Prober et al., 2002).

## **4.3. *Implications for management***

Our results show that vegetation management practices within the Traprock wool-growing region have influenced the floristic composition and richness of the grassy woodland communities. Maintaining a medium density of mature trees (and some degree of regrowth in the understorey in the case of ironbark/gum woodlands) can potentially satisfy both production (in terms of grass and for b cover) and biodiversity (floristic composition) goals in low-input, low-intensity grazing landscapes. In both vegetation types examined here, a medium density of overstorey trees would provide for a broadly similar biodiversity value as that of high mature tree density areas and, importantly in a production landscape, maintain production value. This is significant for land management practices in the Traprock region and other perennial grazing systems in that while there is no significant increase in grass cover in the very open areas compared to medium mature tree density areas, there is a significant decline in biodiversity value, at least in terms of floristic composition, and potentially

other ecosystem services provided by more structurally complex vegetation. 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 maintaining and even increasing plant diversity in a production landscape.

However, there is an increasing body of evidence to suggest that a mosaic of land management units may be desirable to conserve biodiversity across agricultural landscapes (e.g. Alrababah et al., 2007; Billeter et al., 2008). A further question for the low-input, low-productivity grazing systems is, how much of the various vegetation elements (regrowth, open areas, medium and higher tree density areas) should be contained within a grazing landscape to maximize biodiversity values?

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