

Pasture production and composition response after killing Eucalypt trees with herbicides in central Queensland

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Abstract. Clearing woodlands is practised worldwide to increase crop and livestock production, but can result in unintended consequences including woody regrowth and land degradation. The pasture response of two eucalypt woodlands in the central Queensland rangelands to killing trees with herbicides, in the presence or absence of grazing and regular spring burning, was recorded over 7 or 8 years to determine the long-term sustainability of these common practices. Herbage mass and species composition plus tree dynamics were monitored in two replicated experiments at each site. For 8 years following herbicide application, killing *Eucalyptus populnea* F. Muell. (poplar box) trees resulted in a doubling of native pasture herbage mass from that of the pre-existing woodland, with a tree basal area of 8.7 m² ha⁻¹. Conversely, over 7 years with a similar range of seasons, killing *E. melanophloia* F. Muell. (silver-leaved ironbark) trees of a similar tree basal area had little impact on herbage mass grown or on pasture composition for the first 4 years before production then increased. Few consistent changes in pasture composition were recorded after killing the trees, although there was an increase in the desirable grasses *Dichanthium sericeum* (R. Br.) A. Camus (Queensland bluegrass) and *Themeda triandra* Forssk. (kangaroo grass) when grazed conservatively. Excluding grazing allowed more palatable species of the major grasses to enhance their prominence, but seasonal conditions still had a major influence on their production in particular years. Pasture crown basal area was significantly higher where trees had been killed, especially in the poplar box woodland. Removing tree competition did not have a major effect on pasture composition that was independent of other management impositions or seasons, and it did not result in a rapid increase in herbage mass in both eucalypt communities. The slow pasture response to tree removal at one site indicates that regional models and economic projections relating to tree clearing require community-specific inputs.

Additional keywords: basal cover, bio-economic, *Dichanthium sericeum*, grazing, tree competition, woodland ecology.

Received 9 February 2016, accepted 27 June 2016, published online 4 August 2016

Introduction

In inland central Queensland, valuable native pastures for cattle production grow in the understorey of *Eucalyptus melanophloia* F. Muell. (silver-leaved ironbark) and *E. populnea* F. Muell. (poplar box) woodlands, known as forest country. These pastures, predominantly used for cattle breeding, extend either side of the Great Dividing Range. The detailed floristics of these eucalypt communities in the *Aristida/Bothriochloa* (A/B) native pasture land type (Weston and Harbison 1980; Weston *et al.* 1981) have been described by Silcock *et al.* (2015a), and their associated soil surface conditions are reported by Silcock *et al.* (2015b). The dominant perennial pasture forage grasses were *Bothriochloa ewartiana* (desert bluegrass) and *Heteropogon contortus* (black speargrass) in the northern areas and *Dichanthium sericeum* (Queensland bluegrass) with lesser areas of *Bothriochloa bladhii* (forest bluegrass) in the more southern areas. (Botanical names

in this paper are according to Henderson 1997). Though this understorey is mostly native pasture, *Cenchrus ciliaris* (buffel grass) or *Stylosanthes scabra* (shrubby stylo) have been introduced to increase potential livestock production in some areas. Besides introducing sown pastures, the main methods of improving the grazing potential of native pastures are via improving infrastructure, such as extra water points and fencing to manage grazing pressure, tree clearing, and improved grazing management, such as resting pastures or periodic burning to control woody regrowth.

Most world literature reports a curvilinear herbage mass decline by the herbaceous understorey to increasing density of trees (Walker *et al.* 1972), but some studies on this tree–grass balance resulted in an almost linear herbage yield decrease as tree density increased (Walker *et al.* 1986). Mott and Tothill (1984) suggest this tree–grass competition only occurs when

trees and herbage are actively growing, transpiring, during the wet season in the tropics, as during the dry season the herbaceous layer is dormant due to moisture deficiency and/or low minimum temperatures (in the sub-tropics). Burrows (2002) and Jackson and Ash (1998) propose that the nature of the ground layer response to altered tree density can be interpreted in terms of the fertility of the soil and the average annual rainfall. Hence there seem to be differing outcomes from changes to historical woodland densities depending on the environment, the soil types, the chance coincidence of biotic and abiotic events, and possibly the species of grazing animal. Measured data are required to address increasing concerns about poor or declining pasture condition and the potential for reduced grazing capacity and increased risk of erosion, dryland salinity and poor downstream water quality in the wooded grazing lands of Queensland. Some concern has also been raised about the impact that unreliable assumptions might have on broad regional models about the biological and economic impacts of tree clearing in Queensland (Star *et al.* 2010).

The research effort and costs are considerable to adequately document the changes in tree/pasture dynamics that occur over time, and there are only a few Australian examples with both longevity and rigour. For example, the Kunoth Paddock study in South Australia (Sinclair 2005), those of McIvor and Gardener (1995) in two eucalypt woodlands near Charters Towers, and the tree-thinning study by Back *et al.* (2008) near Dingo in Queensland are three documented studies. There are others that have not been formally published in detail (Tunstall and Cunningham 1990; Ash *et al.* 2001). However, if eucalypt trees in the A/B community are killed or thinned, the major visual response may be rapid and only after many years may more subtle changes in landscape function and vegetation communities, such as new springs, salinity, woody regrowth or herbaceous species change, become apparent. The landscape ecology of similar eucalypt communities in south-east Queensland have been discussed in detail by McIntyre *et al.* (2002), with an emphasis on optimising productivity without compromising ecological sustainability.

This study was established to quantify the effects on native pastures and landscape stability after killing the trees, in combination with cattle grazing intensity and regular spring burning, over a relatively long period in eucalypt woodlands in the north Australian rangelands. Two experimental sites were established in traditional native pasture grazing lands in different eucalypt communities of the *Aristida/Bothriochloa* land type in central Queensland to provide detailed data on the impact of removing the tree competition effect from the landscape. Trees were killed in mid-1994 using registered herbicides rather than mechanical methods to prevent any influence of surface soil disturbance by nitrogen release or on pasture density. The other treatments of varying grazing pressures and a burning regime were imposed thereafter. The subsequent effects on pasture growth, composition, tree seedling recruitment, surface soil condition, and rainfall runoff were recorded for the ensuing 7–8 years in both grazed and ungrazed paddocks. This paper reports the effects of tree competition, by killing the trees with herbicides, on the pasture production and composition responses under grazed and ungrazed management in those two communities.

Materials and methods

Similar experimental techniques were used to implement two replicated experiments in each of two open eucalypt woodland communities, a *Eucalyptus melanophloia* (silver-leaved ironbark) site near Rubyvale and a *E. populnea* (poplar box) site near Injune. Both research sites were on commercial cattle properties and each was spread over an area of ~2 km². The pastures at both sites are classed as *Aristida/Bothriochloa* (A/B) native pasture land type in eastern Queensland (Fig. 1).

Site descriptions

Silver-leaved ironbark site

The 'ironbark' site was 10 km NW of Rubyvale (50 km WNW of Emerald) in the Mackenzie River basin (Fig. 1). The experiment site, centred on lat. 23°22'30"S, long. 147°35'15"E, at 325 m elevation, is typical of the Peak Vale land system: undulating country with silver-leaved ironbark and texture contrast soils derived from granite (Galloway *et al.* 1967). This land system covers 185 000 ha between Rubyvale and Clermont and is coded as Queensland Regional Ecosystem (RE) 11.12.2 by Sattler and Williams (1999). It is not an RE of concern at present because over 30% of its original extent remains uncleared and is classified as remnant vegetation.

The site had a gentle undulation between each drainage line, with scattered *Corymbia erythrophloia* (bloodwood) and *Bursaria incana* (prickly pine) trees among the ironbark. *Corymbia dallachiana* (ghost gums) and *Petalostigma pubescens* (quinine bush) were common along waterways. The woodland was relatively open in comparison to other eucalypt woodland types in the district, for example poplar box and *Eucalyptus crebra* (narrow-leaved ironbark), but it had areas of dense, multi-stemmed ironbark seedling regrowth (tree densities of

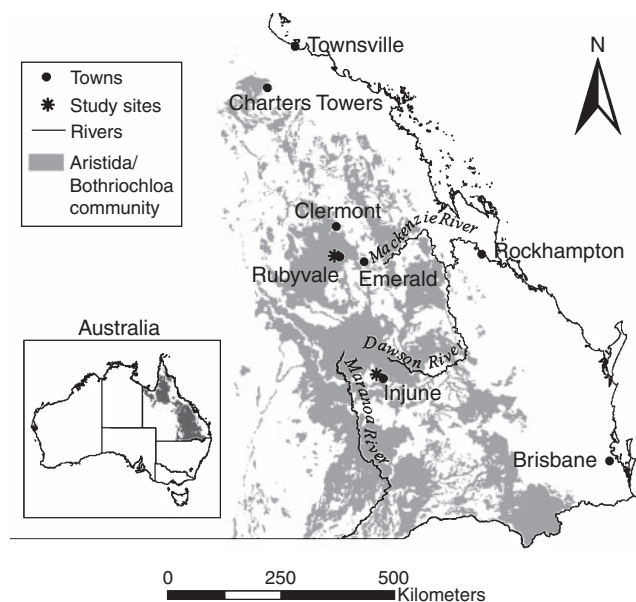


Fig. 1. Location of the two research sites, nearest towns and distribution of the *Aristida/Bothriochloa* native pasture community of eastern Queensland. Insert map shows the location of the *Aristida/Bothriochloa* community within Queensland and Australia.

>1000 stems ha⁻¹). The site average tree basal area at the start of the experiments was 7.7 m² ha⁻¹.

The uniform soil type across the site was classified as: Chromosol/red/eutrophic/haplic (Isbell 1996: Australian classification) or duplex non-calcic brown (Great Soil Group classification), or red duplex, Dr2.12 (Northcote 1979 classification). It had a 5–10-cm sandy-loam surface layer over a 50-cm-deep red saline/sodic medium clay B-horizon, overlying weathering granite.

Poplar box site

The 'poplar box' site was located 18 km NW of Injune at lat. 25°45'23"S, long. 148°24'56"E, at 480 m elevation in the Dawson River basin (Fig. 1). The undulating land was originally well timbered, but after axe-thinning the trees over 30 years ago, the large regrowth trees, predominantly poplar box, were ready for retreatment. The experimental site is part of the Montana Land System, low hills with texture contrast soils growing poplar box and silver-leaved ironbark woodland (Speck *et al.* 1968), and is equivalent to the Bymount Land Resource Area in the Maranoa River catchment to the south (Macnish 1987). These lands cover thousands of square kilometres and our site is RE 11.9.7 in the Sattler and Williams (1999) classification. This RE is rated as 'of concern' because less than 30% of its original vegetation remains remnant. Grazing management is critical on this land type because the labile sediments beneath the A-horizon are susceptible to erosion and have potential salinity inputs to downstream rivers (DNR 1998).

The trees present in 1994 at the start of the experiment, which ran from May 1994 to June 2002, represented small to mature poplar box regrowth up to 35 years old. Poplar box was dominant, but there were small patches of scattered *Allocasuarina luehmannii* (bullock), *Casuarina cristata* (belah) and *E. melanophloia* trees. There was a variable and patchy understorey of *Maytenus cunninghamii* (yellow-berry bush) and *Dodonaea* spp. (hopbush). *Acacia pendula* (myall) trees were common in patches on silty soils at the toe of slopes. A list of the diversity of 60 woody species recorded across the site is given in Silcock *et al.* (2005). The average tree basal area by the TRAPS methodology (Back *et al.* 1997) of all treed paddocks and plots was 8.72 m² ha⁻¹ at the start of the experiments.

There was a range of texture contrast soils, mostly with a shallow 10–15 cm deep loamy surface overlying grey, heavy clay subsoil and weathering labile sediments. The surface was generally hard-setting and slightly acid, but quickly changed to blocky structured, alkaline, saline-sodic clay up to 1 m deep above the parent rock. The major soil types were Sodosols and Dermosols (Isbell 1996), with Principal Profile Forms Db1.13, Db1.33, Dy2.33 and Dy2.43 (Northcote 1979). There is a varying degree of A2-horizon bleaching (due to lateral water flow above the B-horizon) and a consistent alkaline reaction trend with increasing soil depth.

Site layout

Both replicated experiments at each site compared killing trees with herbicides or leaving trees untreated: one experiment imposed three grazing pressures with continuous grazing by growing male cattle (two replications), and the second experiment was ungrazed with two burning regimes. At both

sites, the grazed experiment had 12 paddocks: six had the trees killed and six with the trees untreated. Paddock areas ranged from 3.5 to 21.5 ha at the ironbark site and 4 to 18 ha at the poplar box site, depending on the three grazing pressures.

The ungrazed experiment had 12 plots each of 1 ha; six with the trees killed by herbicide, and six burnt each spring when possible. There were three replications of the four treatments (+/- tree killing by +/- burning combinations). At the ironbark site, the ungrazed treatments were spread among the grazed paddocks, whereas at the poplar box site they were in a single block ~1 km from the grazed paddocks. Though fenced to control domestic livestock, marsupials had access to both experiments at the ironbark site and the grazing experiment at the poplar box site. The ungrazed poplar box site was surrounded by a netting fence, which prevented marsupials from affecting the pasture.

Herbicide treatment

Ironbark site

All trees in the 'Treeless' treatments at the ironbark site were treated in March 1994 by stem injection with the herbicide Velpar[®] (hexazinone; DuPont[™]) by commercial operators using recommended 'cut and squirt' application methods and concentrations (Back *et al.* 1993). Most small saplings and seedlings also received a herbicide squirt on the soil at the base of the stem. All tree species were readily killed by the chemical, except *B. incana*, which showed resistance.

Poplar box site

At the poplar box site, the trees, predominantly *E. populnea*, were stem injected with the herbicide Tordon 50D[®] (picloram and 2,4-D; DOW AgroSciences) herbicide in June–July 1994 using normal 'cut and squirt' commercial practices and concentrations. In the grazing paddocks, a few *A. pendula* trees were left for cattle shade and the rest were ringbarked because they are tolerant of Tordon. All established trees were killed by these methods. Trees in replicate 1 of the 'Treed'-high grazing pressure treatment were thinned by Tordon stem injection in June 1996 from a basal area of 14.5–7.9 m² ha⁻¹ (mean between 1997 and 2002) to make the tree competition in this paddock more closely represent that of the other treed paddocks.

Management after killing trees

Grazed experiment

The 12 grazed paddocks at each site were stocked continually with weaner Brahman-cross steers with a starting weight between 200 and 250 kg. The numbers were reset in response to pasture yield each autumn with new animals to retain either a heavy, moderate or light grazing pressure on the pasture over the next 12 months.

Ungrazed experiment

After rain each spring an attempt was made to conduct a controlled burn of half the plots: three timbered and three where the trees had been killed. Firebreaks were graded around each burning block before every burn. The same six plots were burnt on each occasion. This allowed dead pasture litter built up on the unburnt plots after years without grazing disturbance.

Burning was constrained by experiment 'rules' that aimed at a hot fire to cause damage to small trees and shrubs after at least 25 mm of rainfall had occurred. Atmospheric conditions did not permit us to burn safely every spring and sometimes there was an insufficient amount of grass fuel after a low rainfall summer. Ignition was by a drip fire-lighter.

At the ironbark site, spring burns were conducted in: July 1995, August 1997, October 1998 and November 1999. At the poplar box site, successful burns were implemented in: November 1994, October 1995, September 1996, October 1997, August 1999, October 2000 and October 2001.

Pasture measurements

Herbage mass (dry matter yield, kg ha⁻¹) of each paddock or plot was estimated each autumn using the Botanal procedure (Tothill *et al.* 1992). Twenty-five 0.25-m² quadrats were assessed per hectare to record the relative standing herbage mass and the proportion of the six most abundant plants in each, although the majority of quadrats at both sites had fewer than four plants recorded. During winter each year, crown basal area of pasture plants was measured along fixed transects, three transects each of 150 m in each grazed paddock, and two transects each of 100 m in each plot of the ungrazed experiment, using a 5-point frame with pins 15 cm apart (Bonham 2013). The ironbark site was recorded for 7 consecutive years and the poplar box site for 8 years, to 2002.

Statistical analyses

Statistical analysis of the results was undertaken using the GENSTAT package (GENSTAT 2015). Most were done via ANOVA with tree killing as the first order factor and sub-experiments using grazing pressure or spring burning were the first order interaction factor (not reported here). The grazed paddocks were analysed independently of the plots of the ungrazed experiment. The time-series nature of the data was taken into account by an ANOVA of repeated-measures (Rowell and Walters 1976), via the AREPMEASURES procedure of GENSTAT (2015). This forms an approximate split-plot ANOVA (split for time). The Greenhouse–Geisser epsilon estimates the degree of temporal autocorrelation, and adjusts the probability levels for this.

Significant differences ($P=0.05$) were determined by the l.s.d. process.

Results

Climate and seasons

Both sites have a similar historical mean annual rainfall of ~650 mm, but the proportion of winter rain is higher at the more southern poplar box site. The main period of pasture production is over summer, with potential growth from spring to early autumn, October to March, when 75% of the annual rainfall occurs and temperatures are optimal for C₄ grass species growth. Rainfall during the project was highly variable at both sites but not exceptional. Drought conditions commenced during the 2 years before the experiments commenced, and annual totals did not appreciably exceed decile 5 until late in the project at the ironbark site. Summarised 'wet' season (October–March) rainfall for both sites relative to long-term means, expressed as deciles, is shown in Table 1 along with the decile values for each pasture growth year (July–June) during the experiment from 1993–1994 to 2001–2002.

Pasture response to tree killing

Ironbark site

Herbage mass

Grazed experiment. Killing all the large ironbark trees produced no initial significant difference in annual pasture growth, measured as autumn standing herbage mass, compared to the untreated areas. Only by Year 5 was a significant ($P<0.05$) increase (25%) of standing autumn pasture dry matter yield due to tree killing recorded in the grazed paddocks (Fig. 2a). This yield difference increased to 35% after 7 years. The gradual increase in total pasture yield as seasons improved was mirrored by a greater amount being contributed by the three main perennial grasses, *B. ewartiana*, *H. contortus* and *Themeda triandra* (kangaroo grass) in the later years (Fig. 3a–c), but not from *Chrysopogon fallax* (golden-beard grass) (Fig. 3d). The treeless paddocks had developed slightly higher yields of these grasses after 1998 and that was maintained thereafter, although not always by a statistically significant amount ($P=0.05$).

Table 1. Summer 'wet' season (October–March) rainfall (mm) plus annual and wet season decile ratings at the *Eucalyptus melanophloia* (silver-leaved ironbark) and *Eucalyptus populnea* (poplar box) sites between 1993–1994 and 2001–2002
Project duration was July 1994 to June 2002. Values are means of eight and five rain gauges respectively spread across the sites

Site/Year	1993– 1994 ^A	1994– 1995	1995– 1996	1996– 1997	1997– 1998	1998– 1999	1999– 2000	2000– 2001	2001– 2002	Long-term mean
<i>Ironbark site</i>										
Wet season rainfall ^B	352	377	321	717	296	602	398	670	–	475
Wet season decile ^C	3	4	3	9	2	8	5	9	–	
Annual rainfall decile	6	1	3	7	2	8	6	7	–	659
<i>Poplar box site</i>										
Wet season rainfall	414	314	439	475	506	527	288	500	270	442
Wet season decile	5	2	6	7	8	8	2	7	2	
Annual rainfall decile	5	2	6	7	5	9	4	4	4	626

^A1993–94 data are partly from the homestead, 4 and 1 km from sites respectively.

^BWet season is October–March of Year 1–2.

^CDecile values are for July–June for the nearest long-term rainfall station.

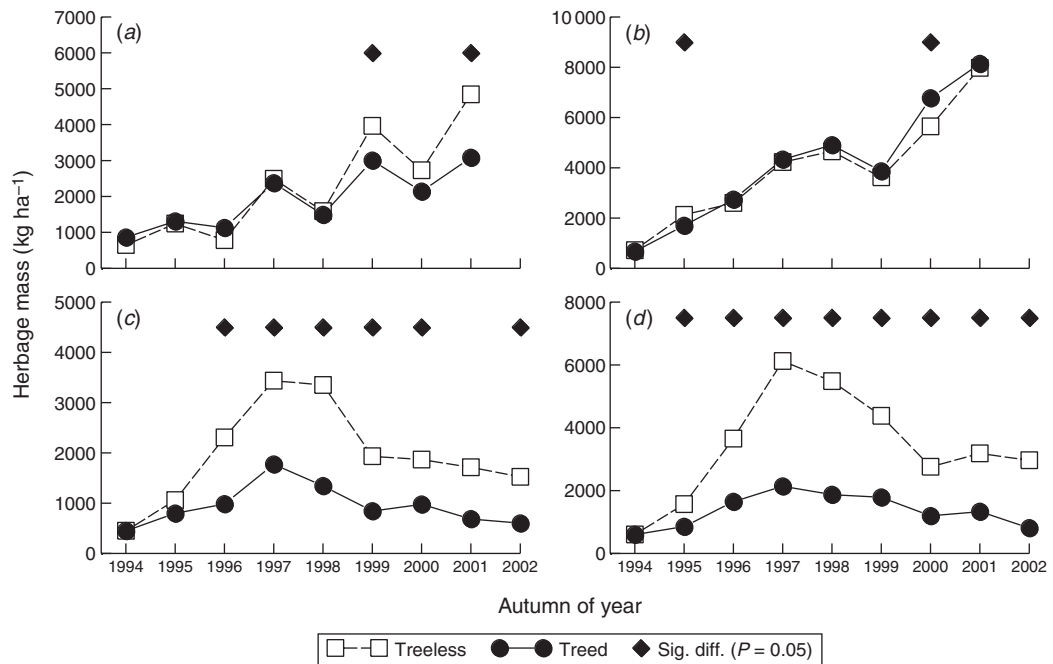


Fig. 2. Effect of tree killing in 1994 at the *Eucalyptus melanophloia* (silver-leaved ironbark) site (a, b) and the *Eucalyptus populnea* (poplar box) site (c, d) on pasture herbage mass (kg DM ha⁻¹) in grazed (a, c) and ungrazed (b, d) pastures in autumn between 1994 and 2002. Diamonds indicate a significant difference ($P < 0.05$) for that year. Note the different herbage mass scales.

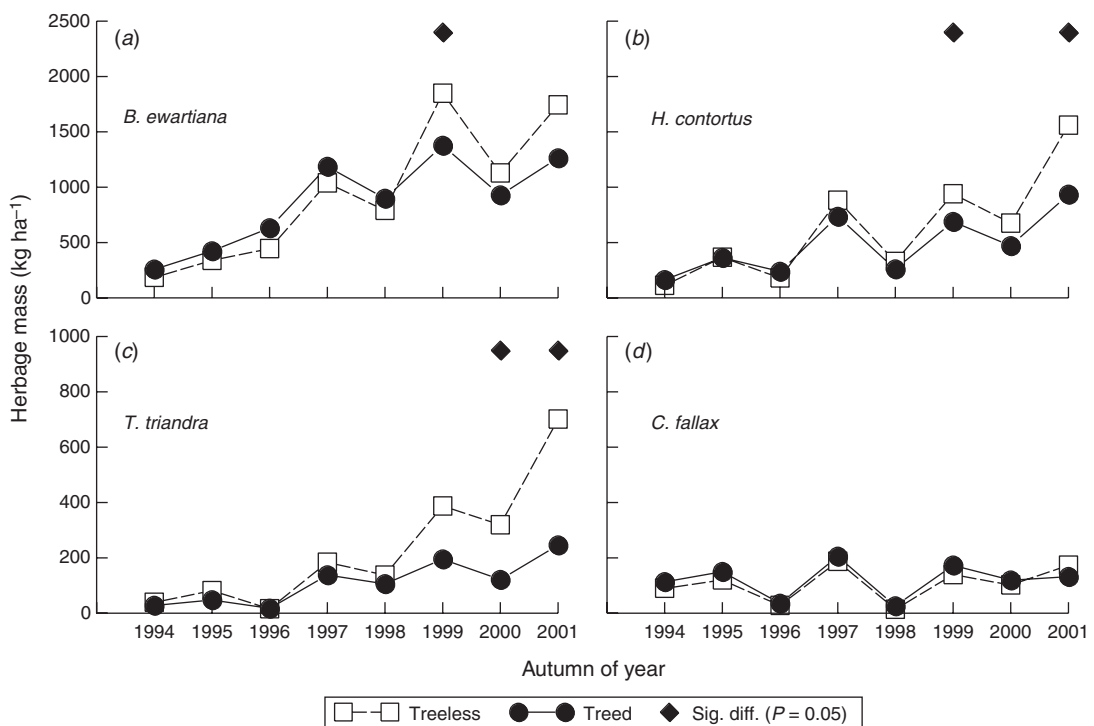


Fig. 3. *Eucalyptus melanophloia* (silver-leaved ironbark) tree competition effect on herbage mass (kg DM ha⁻¹) of four main pasture species (a) *Bothriochloa ewartiana*, (b) *Heteropogon contortus*, (c) *Themeda triandra*, and (d) *Chrysopogon fallax*, in grazed pastures in autumn between 1994 and 2001. Diamonds indicate a significant difference ($P < 0.05$) for that year. Note the different herbage mass scales.

Ungrazed experiment. In the ungrazed plots a significant tree effect on herbage mass took 6 years to develop and that significantly ($P < 0.05$) favoured the treed plots temporarily by 20% (Fig. 2b). The mean standing autumn yields on the ungrazed plots were greater, despite being periodically reduced by spring burns on half of the plots (data not shown). Note also that in the 1998 spring, the nominally unburnt treatment plots were also burnt to re-invigorate growth because up to 10 000 kg ha⁻¹ of moribund standing dry matter had accumulated after 4 years without defoliation and high 1996–1997 summer rainfall.

Pasture composition

Grazed experiment. At this ironbark site, the most abundant species at the start and end of the 7-year grazed experiment are shown in Table 2. The proportion of the herbage mass due to *B. ewartiana* increased from ~30% of the pasture in 1994, after the drought, to ~40% in 2001, although it was over 50% in individual years (Fig. 4a), whereas *H. contortus* increased from 20% to ~30% over this period (Fig. 4b). The proportion of *T. triandra* was consistently higher without tree competition (Fig. 4c) and the initial difference was magnified after later good summer rainfall seasons (Table 2, Fig. 4c). In contrast, the proportion of the autumn yield due to *C. fallax* decreased noticeably over the first 4 years, irrespective of tree competition (Fig. 4d).

The removal of tree competition had no major effect on the proportion of minor species, but the contribution of some species was more dynamic than others. Species that fluctuated most under changing seasonal conditions were native legumes

(mainly *Indigofera* spp. and *Desmodium* spp.), which increased to 3.5% in the higher rainfall years, although this was not consistently maintained thereafter (Table 2). *Aristida*

Table 2. *Eucalyptus melanophloia* (silver-leaved ironbark) tree competition effect on composition (%) of herbage mass in autumn of the main pasture species or taxon groups in grazed pastures in 1994 and 2001

Taxon or Group	1994	1994	2001	2001
	Treed	Treeless	Treed	Treeless
<i>Bothriochloa ewartiana</i> ^A	29.5	28.6	42.3	34.9
<i>Heteropogon contortus</i>	18.6	17.3	29.6	34.7
<i>Chrysopogon fallax</i>	13.1	14.0	4.7	3.5
<i>Eulalia aurea</i>	6.6	6.9	1.5	0.9
Non-legume forb species	3.5	3.5	0.3	0.4
<i>Themeda triandra</i>	3.3	6.2	5.6	12.8
<i>Panicum</i> spp.	2.3	2.5	0.3	0.4
Cyperaceae species	2.1	1.6	0.2	0.2
<i>Dichanthium sericeum</i>	2.0	1.4	0.9	0.8
Native legumes	2.0	2.0	0.3	0.4
<i>Eriochloa pseudoacrotricha</i>	1.2	0.5	0.3	0.0
<i>Chloris</i> spp.	1.1	0.5	0.7	0.7
<i>Enneapogon</i> spp.	0.9	1.1	0.3	0.2
<i>Aristida</i> spp.	0.7	0.9	0.4	0.4
<i>Bothriochloa bladhii</i>	0.6	0.0	1.8	0.2
<i>Eragrostis</i> spp.	0.4	0.4	0.2	0.2
<i>Tripogon loliiformis</i>	0.3	0.5	0.6	0.2
<i>Digitaria</i> spp.	0.2	0.1	0.0	0.0
<i>Melinis repens</i>	0.2	0.2	0.5	0.6
<i>Tragus australianus</i>	0.2	0.1	0.0	0.0

^AAuthors of plant names in all tables follow Henderson (1997).

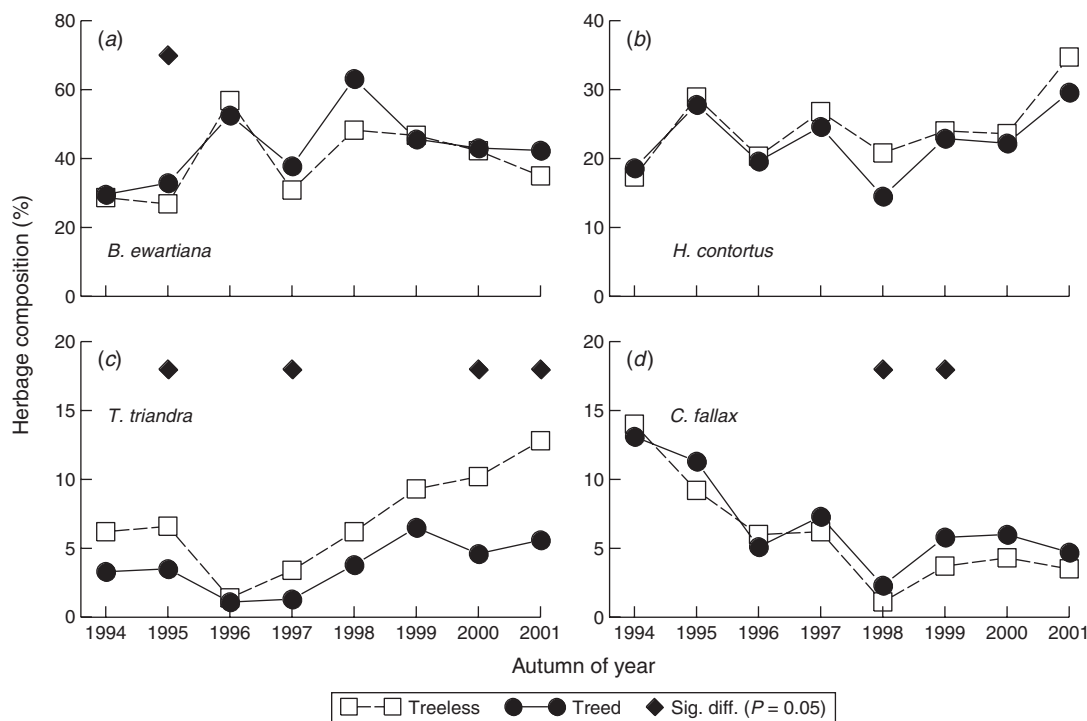


Fig. 4. *Eucalyptus melanophloia* (silver-leaved ironbark) tree competition effect on the proportion (%) of pasture herbage mass contributed by (a) *Bothriochloa ewartiana*, (b) *Heteropogon contortus*, (c) *Themeda triandra*, and (d) *Chrysopogon fallax*, in grazed pastures in autumn between 1994 and 2001. Diamonds indicate a significant difference ($P < 0.05$) for that year.

spp. (wiregrasses), an undesirable grazing component, did not show any yield response to tree killing (Table 2) and remained only a small proportion (mean 0.45% over 7 years) of the total herbage mass. Most minor species were less prominent in the pasture at the end of the experiment after drier seasons occurred, the exceptions being *B. bladhii* and *Melinis repens* (red Natal grass) (Table 2).

Ungrazed experiment. Similar changes in pasture composition occurred in the ungrazed plots (data not shown), but *T. triandra* increased from ~4% to 25% of the herbage mass, irrespective of tree competition.

Species frequency

Grazed experiment. These pastures have a range of minor forage species that vary in their proportions with seasons, whereas the dominant perennial grasses retain a consistent prominence (Table 3). Over the 7 years, average frequency of *B. ewartiana* under grazing was 50.1% (Treeless) and 47.1% (Treed). It began with an average 32% initial frequency and was relatively high at 57% in 2001 after good rainfall seasons, but treatment differences were inconsistent and non-significant ($P=0.05$) overall. The equivalent mean frequency for the effect of trees was 40.6% versus 36.8% for *H. contortus* and 15.5% versus 10.6% for *T. triandra*, and for both of those species the difference was consistent, if not statistically significant. Some *Aristida* spp. were grouped because of difficulties in positive identification to a species level, so changes are more difficult to identify by this frequency measure, but *Aristida leptopoda* (white speargrass) declined in frequency over time, while *A. calycina* (dark wiregrass) increased but without a significant tree killing effect. By the end of the experiment the average *Aristida* spp. frequency had increased from 17% to 28% under grazing (Table 3) but this was not correlated with tree competition.

There was an increase in *H. contortus* plant frequency in the higher rainfall years later in the experiment (Table 3) from

Table 3. *Eucalyptus melanophloia* (silver-leaved ironbark) tree competition effect on frequency (%) of the main pasture species and taxon groups in autumn in the grazed trial in 1994 and 2001

Taxon	1994	1994	2001	2001
	Treed	Treeless	Treed	Treeless
Non-legume forb species	66.4	75.8	63.6	60.7
Native legumes	54.3	60.1	43.4	51.8
<i>Chrysopogon fallax</i>	39.0	38.8	41.8	32.6
<i>Bothriochloa ewartiana</i>	29.8	33.6	57.6	63.3
<i>Panicum</i> spp.	23.3	28.5	12.9	14.8
<i>Heteropogon contortus</i>	25.5	20.1	50.5	57.5
<i>Enneapogon</i> spp.	17.4	22.3	22.5	22.3
Cyperaceae species	24.2	14.5	13.4	9.5
<i>Themeda triandra</i>	11.8	22.4	12.9	20.1
<i>Aristida</i> spp.	17.2	16.3	26.0	28.9
<i>Eulalia aurea</i>	13.6	14.0	4.9	3.5
<i>Tripogon loliiformis</i>	9.6	14.5	27.4	14.9
<i>Eragrostis</i> spp.	9.4	11.6	13.8	16.0
<i>Dichanthium sericeum</i>	7.3	0.0	6.6	4.1
<i>Tragus australianus</i>	0.8	0.4	1.7	1.4
<i>Chloris</i> spp.	0.0	0.0	24.7	28.1

~23% to 50%, and a similar large increase occurred in the ephemeral *Chloris* spp., particularly *C. divaricata* (slender chloris). Notable declines after 7 years were seen in *Eulalia aurea* (silky browntop) and in *Panicum* spp. such as *P. effusum* and *P. decompositum*. Some initial treatment differences in frequency for Cyperaceae spp. (sedges) and *T. triandra* due to paddock location remained after 7 years. The frequency of *Tripogon loliiformis* (five-minute grass) increased greatly under grazing in the treed plots (10–27%), but not where the trees had been killed (Table 3). Despite their low biomass, the frequency of occurrence of native legumes and dicotyledonous forb species was high (Table 2 vs Table 3). Some of the more ephemeral grasses such as *Chloris*, *Enneapogon* and *Eragrostis* species had low herbage mass (less than 5%, Table 2), but a high frequency of occurrence in these pastures, (10–25%, Table 3). Some individual species of these three genera are true annuals but many are short-lived perennials.

Ungrazed experiment. In the absence of grazing, frequency of *D. sericeum*, a preferred 3P species (3P=productive, palatable and perennial), increased from 4% to 13% where trees were killed, but not where they remained. The frequency of *T. triandra* also increased markedly with time from ~14% to 47% irrespective of tree competition and this is reflected in its herbage mass. Cyperaceae species fell markedly over the 7 years from ~35% to 7%, as did many other minor species, such as *T. loliiformis* (25–3%), *A. leptopoda* (16–2%), *P. effusum* (22–2%) and *P. decompositum* (12–0.5%), again irrespective of tree competition. This decline was associated with an increase in herbage mass of the more competitive, tall, dominant perennial grasses.

Crown basal area of pastures

Killing the trees produced a small, consistent, but generally non-significant increase in pasture crown basal area in both grazed (Fig. 8a) and ungrazed treatments. The mean pasture basal area over the experiment period was 2.3% (Treeless) versus 1.9% (Treed) in the grazed paddocks, compared to 2.6% and 2.2% respectively in the ungrazed plots. A short-term mean basal area difference in autumn 1999, after the October 1998 burn, was recorded in the ungrazed treeless plots, but was also not statistically significant (3.96% vs 2.25%).

Poplar box site

Herbage mass

Grazed experiment. The most significant and visually obvious effect of killing the trees in this community was the increase (doubling) in total standing herbage mass over summer each year (Fig. 2c, d), although it was two growing seasons before this difference (measured in autumn) was statistically significant ($P < 0.05$). The relative herbage mass difference in the treeless paddocks after the second year to 2002 was a mean increase of 130% over the treed paddocks across the range of three grazing pressures evaluated. This production increase by individual species, though not always significant, applied to most perennial grasses (Fig. 5). The preferred grass for this community, *D. sericeum*, increased rapidly after killing the trees to over 600 kg ha⁻¹, compared to a consistently lower production of <100 kg ha⁻¹ with grazing and tree competition (Fig. 5a).

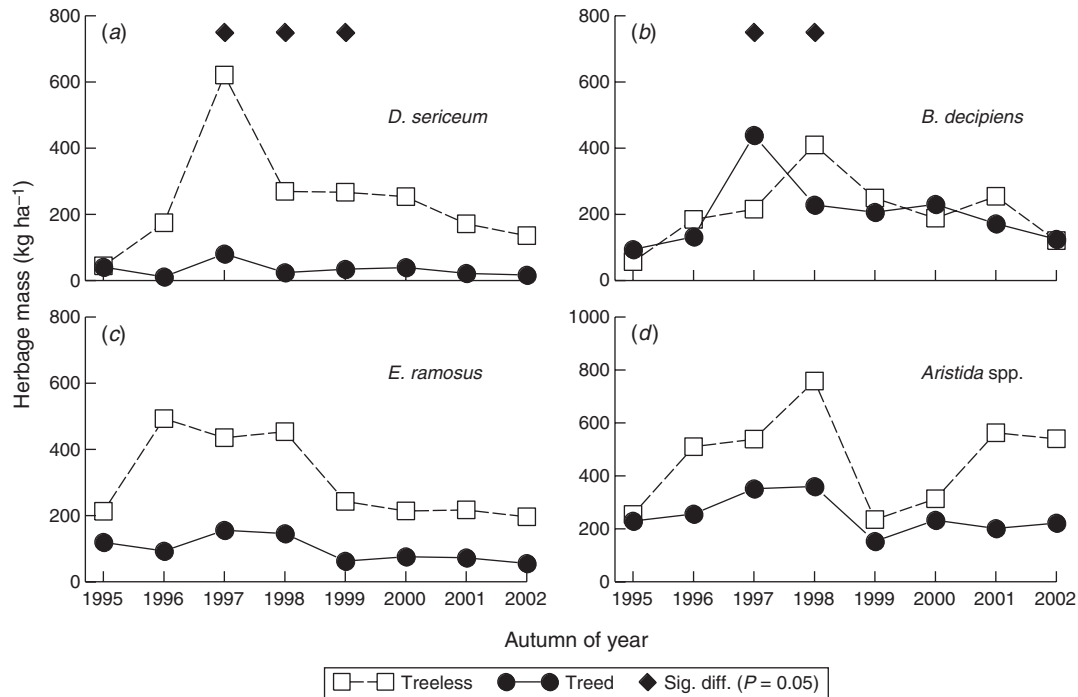


Fig. 5. *Eucalyptus populnea* (poplar box) tree competition effect on herbage mass (kg DM ha⁻¹) of four main pasture species (a) *Dichanthium sericeum*, (b) *Bothriochloa decipiens*, (c) *Enteropogon ramosus*, and (d) *Aristida* spp., in grazed pastures in autumn between 1995 and 2002. Diamonds indicate a significant difference ($P < 0.05$) for that year. Note the different herbage mass scales.

The increased production of *Bothriochloa decipiens* (pitted bluegrass) was not always consistent in its response to tree killing (Fig. 5b). There was an increase in total *Aristida* spp. contribution to herbage mass over time in the treeless treatments (Fig. 5d). The pasture mass contributed by *B. decipiens* and *Aristida* spp. in the treed paddocks was 55% of total production, compared with 36.1% in the treeless paddocks by 2002.

Ungrazed experiment. In the ungrazed experiment, the pasture response to tree competition was similar with an average 160% increase in herbage mass from killing the trees, and it was similar for both burnt and unburnt plots.

Pasture composition

On average, 24 common grass taxa and 18 common non-grass taxa were recorded at each autumn pasture sampling. There was a total of 173 herbaceous species recorded across the site (Silcock et al. 2005) during the two experiments.

Grazed experiment. Killing trees at the poplar box site changed the proportion of some important and common species, such as *D. sericeum*, *B. bladhii* and *B. decipiens*, but the broad floristics present in the pastures did not change over the ensuing 7 years. *Aristida* spp., *Enteropogon ramosus* (twirly windmill grass), *B. decipiens*, *D. sericeum* and *Enneapogon* spp. (bottle-washer grasses) were the major grass contributors to these pastures (Table 4).

Table 4. *Eucalyptus populnea* (poplar box) tree competition effect on composition (%) of herbage mass in autumn of the main pasture species or taxon groups in grazed pastures in 1995^A and 2002

Taxon	1995 Treed	1995 Treeless	2002 Treed	2002 Treeless
<i>Aristida</i> spp.	27.8	21.9	31.8	26.5
<i>Enteropogon ramosus</i>	15.4	21.4	8.1	19.5
<i>Bothriochloa decipiens</i>	12.6	5.5	24.2	9.6
<i>Sclerolaena</i> spp.	6.8	8.0	1.3	1.8
<i>Dichanthium sericeum</i>	5.4	4.6	2.6	7.2
<i>Chloris</i> spp.	2.1	4.2	1.5	2.4
Non-legume forb species	6.6	3.5	8.2	4.0
<i>Enneapogon</i> spp.	4.8	10.3	5.7	7.5
<i>Chrysopogon fallax</i>	2.9	2.0	4.8	1.2
<i>Eragrostis</i> spp.	2.2	1.7	2.0	1.5
Cyperaceae species	2.1	1.2	1.8	1.0
<i>Tripogon loliiformis</i>	1.9	1.9	1.4	1.6
<i>Sporobolus</i> spp.	1.6	3.1	0.7	3.8
<i>Tragus australianus</i>	1.0	2.0	0.3	0.1
<i>Bothriochloa bladhii</i>	0.9	1.3	0.4	1.4
<i>Themeda triandra</i>	0.4	0.1	0.0	0.1
<i>Heteropogon contortus</i>	0.4	0.1	1.1	1.5
Native legumes	0.2	0.1	0.1	0.3
<i>Panicum</i> spp.	1.5	0.4	1.5	0.4
<i>Eulalia aurea</i>	0.0	0.0	0.0	0.2

^ANote trees were killed (herbicide treated) in winter of 1994.

The proportion of some perennial grasses, including *D. sericeum* (Fig. 6a), increased under grazing after removal of the tree competition. The palatable grasses *C. divaricata* and *H. contortus* also responded positively. The exceptions were *B. decipiens* (Fig. 6b), *C. fallax* (Table 4) and *A. calycina* (branched wiregrass) (data not shown) that were relatively more productive in the presence of tree competition. Similarly, the total *Aristida* spp. contribution was consistently, but not significantly, greater in the treed paddocks in these grazed pastures (Fig. 6d). This tree competition effect was consistent across both experiments for *B. decipiens* in all years. During the grazed experiment to 2002, this low palatability 2P (perennial, productive) grass doubled its dry matter yield contribution to 24.2% in the treed treatments and to 9.6% where treeless (Table 4). Initial differences in the proportion of *E. ramosus* and *Aristida* spp. did not change after the first summer (1995) in response to killing the trees (Fig. 6c, d).

In the treeless treatments, *D. sericeum* increased from 4.6% to 7.2% of total herbage mass from 1995 to 2002 (Table 4), but it was significantly higher than that in some intervening years (Fig. 6a). The useful 2P (palatable, perennial) grass, *C. divaricata*, increased on average from 1.6% to 4.2% of total herbage mass over the same 7-year period. The contribution of the common 2P grass *Eragrostis lacunaria* (purple lovegrass) remained low and seasonally variable, and there was a decline by *Sclerolaena birchii* (galvanised burr) from 1% to almost zero of total herbage mass over this period (Table 5). This decline was not due to tree competition, but occurred after the

increase in annual rainfall and resultant increased grass competition.

Ungrazed experiment. Where ungrazed, *A. ramosa* retained its presence (5–10% of the herbage mass), as did other *Aristida* spp., and they were unaffected by tree competition, whereas the presence of *B. decipiens* declined in the treeless plots over 7 years, from 33% to 10% of the herbage mass (Table 5). This was due to a severe loss of the old moribund plants in the abnormally wet winter–spring period (342 mm between June and November) of 1998 when its crowns were covered in a high density of *Calotis* spp. (daisy burrs), especially *C. lappulacea*. Also, the accumulated decomposing dead pasture and litter seemed to promote fungal attack during this wet period. *Dichanthium sericeum* increased its total herbage mass (Table 5) where grazing and trees were removed, particularly in the wetter years (Fig. 7a). Growth of the bulky, tussock perennial grasses *B. bladhii* (Fig. 7b), *Cymbopogon* spp. and *E. ramosus* was also favoured by the removal of tree competition (Table 5). Most of the other pasture changes after 7 years were correlated more with recent seasonal conditions and were less affected by the presence of the mature trees (See Silcock *et al.* 2005 for the complete datasets). The increase in contribution of *C. ciliaris* over time in the absence of grazing was notable, but due primarily to its spread by the disturbance during the annual firebreak clearing.

Species frequency

Grazed experiment. At the poplar box site, similar to the ironbark community, many minor herbaceous species in terms

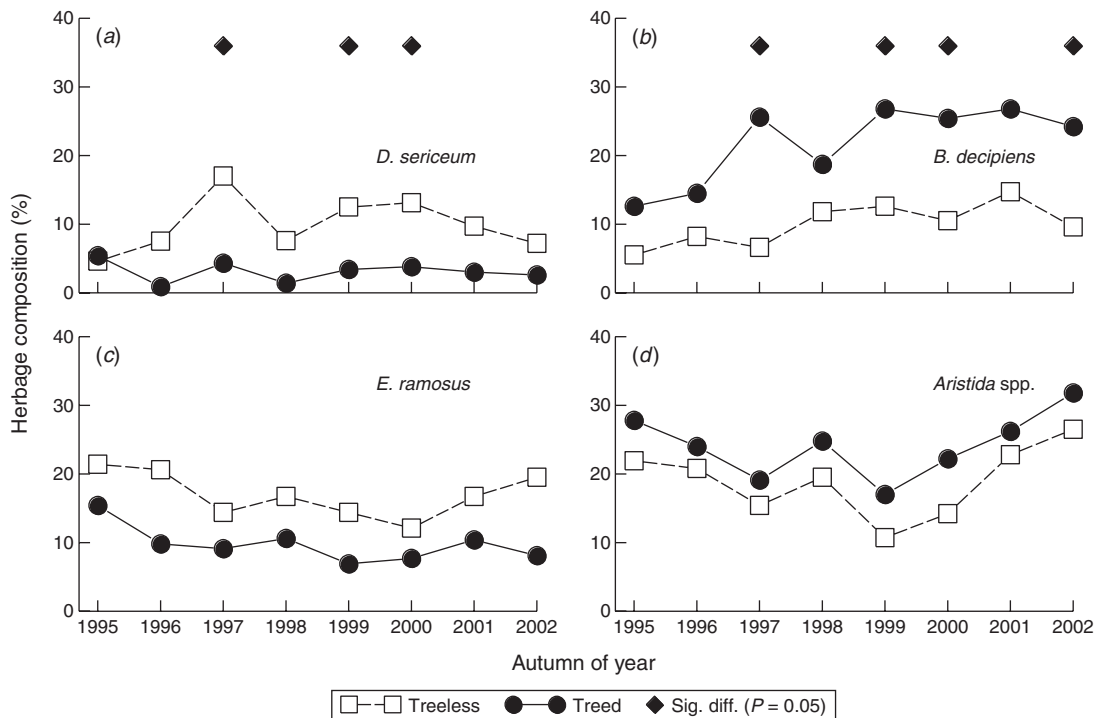


Fig. 6. *Eucalyptus populnea* (poplar box) tree competition effect on the proportion (%) of the autumn pasture herbage mass contributed by (a) *Dichanthium sericeum*, (b) *Bothriochloa decipiens*, (c) *Enteropogon ramosus*, and (d) *Aristida* spp., in grazed pastures in autumn between 1995 and 2002. Diamonds indicate a significant difference ($P < 0.05$) for that year.

of biomass (Table 4) were common and widespread when assessed by their frequency of occurrence (Table 6), such as *T. loliiformis* and *Tragus australianus* (small burrgrass). The widely distributed perennial forb *Brunoniella australis* (blue trumpet) maintained a moderate frequency in the grazed paddocks, irrespective of tree competition (Table 6). There were significant changes in the frequency of species between the treeless and treed treatments over 7 years, with the frequency of the main 3P grass at the site, *D. sericeum*, greatly reduced ($P < 0.05$) by the tree competition, from 26.6% to 8.5% by 2002 (Table 6). *Chloris divaricata* plant frequency was also significantly ($P < 0.05$) reduced by tree competition, 41–20%, but over all treatments it showed a large increase

over the experimental period from an average frequency of 9 to 30%.

Bothriochloa decipiens and *Aristida* spp. (Table 6) always had a higher frequency in the treed treatments. There were pre-existing areas of dense *Aristida ramosa* (purple wiregrass) on the creek flats of the treed and treeless paddocks that persisted and were not influenced by the tree killing treatment. Unchanged in their high frequency over the experimental period were *C. fallax*, *Enneapogon* spp., *T. loliiformis* and *E. ramosus*. However, large reductions were recorded for the annual grass *T. australianus* in all paddocks, and also for the sedge *Fimbristylis dichotoma* (common fringe-rush) where the trees were retained (Table 6).

Ungrazed experiment. In the ungrazed experiment, many species also retained their frequency in the pastures after 7 years (Table 7) irrespective of tree competition. Large forbs declined (8% to <1%) probably due to the spring fires regularly applied to some plots, but the forb *Verbena tenuisecta* (Mayne's pest) was more common by 2002, increasing from 0.4% to 8.1% frequency in the treeless plots, boosted mostly by the wet winter of 1998. A decline in presence in 2002 compared to 1995 was evident for *B. australis*, *T. australianus*, *S. birchii* and small *Sporobolus* species, like *S. caroli* (fairy grass), whereas an interacting change between seasons and tree competition was obvious for *Calotis* spp. and *B. decipiens*. Some perennial grasses increased their frequency markedly in the treeless plots where they had greater opportunity to express their seedling recruitment and growth potential. Notable examples were *H. contortus*, *C. divaricata*, *B. bladhii* and *D. sericeum* (Table 7). Under trees, there was a decline in the frequency of *T. loliiformis* in the absence of grazing that was independent of seasonal effects, whereas large opposing changes in *B. decipiens* frequency were recorded depending on the presence of trees (65% with trees vs 29.9% in treeless plots by 2002).

Table 5. *Eucalyptus populnea* (poplar box) tree competition effect on the composition (%) of herbage mass in autumn of the main pasture species and taxon groups in ungrazed pastures in 1995^A and 2002

Taxon	1995	1995	2002	2002
	Treed	Treeless	Treed	Treeless
<i>Bothriochloa decipiens</i>	28.1	32.6	36.3	9.5
<i>Chrysopogon fallax</i>	13.9	11.6	9.0	8.3
<i>Aristida ramosa</i>	11.8	8.6	10.5	12.1
<i>Aristida calycina</i>	9.9	8.5	4.0	7.2
Non-legume forb species	9.6	5.6	2.4	2.0
<i>Enneapogon</i> spp.	6.1	6.3	5.3	3.1
<i>Tripogon loliiformis</i>	2.3	0.5	0.0	0.3
<i>Dichanthium sericeum</i>	1.8	2.9	5.1	7.2
<i>Bothriochloa bladhii</i>	1.4	1.7	0.9	6.8
<i>Sclerolaena birchii</i>	1.4	0.5	0.0	0.0
<i>Cymbopogon</i> spp.	1.3	2.0	2.9	6.7
<i>Enteropogon ramosus</i>	1.0	1.7	2.4	5.4
<i>Tragus australianus</i>	0.9	0.6	0.1	0.0
<i>Eulalia aurea</i>	0.7	0.7	1.6	1.6
Native legumes	0.7	0.5	0.3	0.3
<i>Brunoniella australis</i>	0.6	0.3	0.0	0.2
<i>Chloris divaricata</i>	0.6	0.9	0.4	2.2
<i>Panicum effusum</i>	0.6	2.2	0.2	1.8
<i>Cenchrus ciliaris</i>	0.2	0.0	5.0	2.9

^ANote trees were killed (herbicide treated) in winter of 1994.

Crown basal area of pastures

On average, pasture crown basal area in the grazed paddocks increased 1–1.5% where tree competition was removed, which proportionally is an increase of 50%, compared to an increase

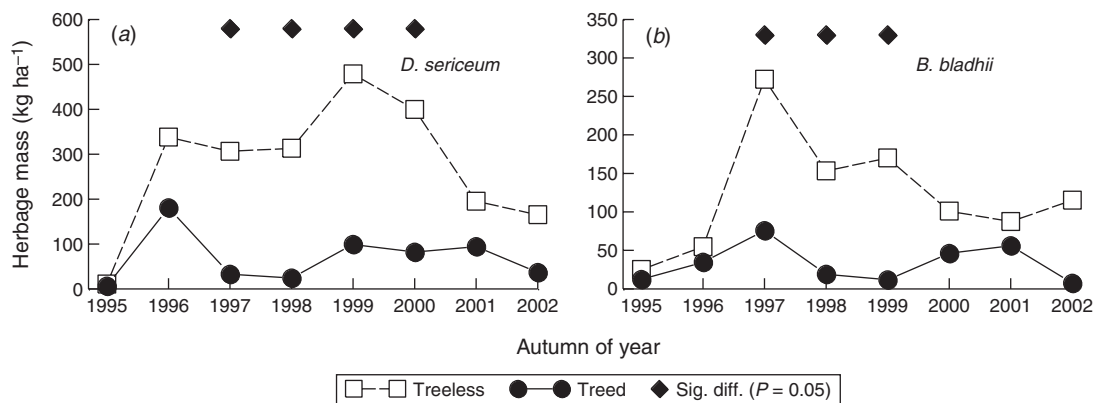


Fig. 7. *Eucalyptus populnea* (poplar box) tree competition effect on herbage mass (kg DM ha⁻¹) of (a) *Dichanthium sericeum* (Queensland bluegrass) and (b) *Bothriochloa bladhii* (forest bluegrass) each autumn in ungrazed pastures between 1995 and 2002. Diamonds indicate a significant difference ($P < 0.05$) for that year. Note the different herbage mass scales.

Table 6. *Eucalyptus populnea* (poplar box) tree competition effect on frequency (%) of the main pasture species and taxa groups in autumn in grazed pastures in 1995^A and 2002

Taxon	1995	1995	2002	2002
	Treed	Treeless	Treed	Treeless
<i>Aristida ramosa</i>	40.2	21.6	41.0	22.2
Forb species – small	36.9	53.4	22.5	18.8
<i>Bothriochloa decipiens</i>	33.0	16.3	46.6	21.5
<i>Brunoniella australis</i>	32.6	33.5	15.1	18.6
<i>Chrysopogon fallax</i>	29.1	23.7	30.3	20.7
<i>Enneapogon</i> spp.	23.4	54.8	32.0	54.4
<i>Fimbristylis dichotoma</i>	23.3	10.0	4.9	9.0
<i>Tripogon loliformis</i>	17.1	23.7	17.2	26.6
<i>Enteropogon ramosus</i>	17.0	17.8	13.2	16.9
<i>Tragus australianus</i>	13.8	33.3	1.8	2.2
<i>Sclerolaena birchii</i>	13.0	9.9	3.6	5.1
<i>Dichanthium sericeum</i>	8.9	13.1	8.5	26.6
<i>Chloris divaricata</i>	7.0	11.8	20.3	41.6
<i>Sporobolus</i> spp. (rats-tail)	3.3	3.9	2.3	8.9
<i>Heteropogon contortus</i>	2.2	0.6	5.2	2.6
<i>Aristida</i> spp. (minor)	1.3	0.6	2.8	1.5
<i>Panicum effusum</i>	1.2	2.0	0.6	0.8
<i>Rhynchosia minima</i>	0.9	1.0	4.9	5.2
<i>Themeda triandra</i>	0.3	0.6	0.2	0.6

^ANote trees were killed (herbicide treated) in winter of 1994.

of over 100% in herbage mass. The mean basal area of the pasture between 1995 and 2002 after killing trees was 4.10% compared with 2.69% with trees in the grazed paddocks, and 4.48% compared with 2.97% in the ungrazed treed plots. In both experiments, these changes were significantly different ($P < 0.01$). Annual differences tended to be higher in lower rainfall years and not always statistically significant (Fig. 8b). In the ungrazed plots, the significant increase in pasture basal area after killing the trees, persisted throughout the experiment, except after the wet winter–spring period of 1998 (Fig. 8c). There was an increasing trend in basal area in both treed and treeless plots over the 7 years. This crown cover reached high levels for such native pastures, to 7% in some treeless plots, in the continual absence of grazing.

Discussion

The different pasture yield and composition responses from removing tree competition by killing the trees with herbicides in these two eucalypt woodlands has confirmed the difficulty of reliably predicting the short- and medium-term responses that will follow such management changes, even in the same broad pasture type and climatic zone. This indicates that regional models that generate predicted outcomes of pasture management scenarios need to explicitly incorporate calculations that reflect the proportions of major soil and vegetation types as well as climate, tree cover and management. These results support the findings of McIvor and Gardener (1995) from two eucalypt communities in north Queensland. These authors found that tree clearing, averaged across fertiliser, sown pasture and stocking rate treatments, increased herbage mass, but that the magnitude of this response in a particular growing season was inconsistent across two eucalypt communities. Their pasture response varied

Table 7. *Eucalyptus populnea* (poplar box) tree competition effect on frequency (%) of the main pasture species and taxa groups in autumn in ungrazed pastures in 1995^A and 2002

Taxon	1995	1995	2002	2002
	Treed	Treeless	Treed	Treeless
<i>Chrysopogon fallax</i>	55.6	48.7	42.5	41.7
<i>Bothriochloa decipiens</i>	51.1	52.9	65.0	29.9
<i>Aristida ramosa</i>	28.7	26.7	37.2	31.1
<i>Brunoniella australis</i>	28.2	22.9	2.2	7.8
Legumes – palatable	25.0	20.4	14.0	15.2
<i>Enneapogon</i> spp.	21.1	22.5	28.9	20.2
<i>Aristida calycina</i>	20.4	29.6	21.6	19.8
Forb species – small	20.3	18.3	13.7	13.1
<i>Calotis</i> spp.	10.4	5.8	1.2	7.8
<i>Tripogon loliformis</i>	10.3	13.8	0.6	8.7
<i>Eragrostis molybdea</i>	9.5	11.7	3.8	6.7
<i>Sida</i> spp.	9.5	5.8	4.4	3.9
<i>Cyperus</i> spp.	8.7	9.6	13.4	6.2
Forb species – large	8.3	7.1	0.3	0.0
<i>Dichanthium sericeum</i>	5.8	10.0	12.6	19.0
<i>Eragrostis lacunaria</i>	5.8	7.9	3.5	0.6
<i>Sporobolus</i> spp. (small)	5.4	3.7	0.7	0.0
<i>Sclerolaena birchii</i>	4.9	2.5	0.6	0.7
<i>Tragus australianus</i>	4.9	7.9	0.6	0.0
<i>Eragrostis</i> spp.	4.6	3.7	4.7	9.7
<i>Chloris divaricata</i>	3.3	4.6	4.6	14.5
<i>Enteropogon ramosus</i>	3.3	4.6	4.7	6.5
<i>Bothriochloa bladhii</i>	2.5	3.7	2.5	6.4
<i>Panicum effusum</i>	2.1	9.2	1.5	6.5
<i>Sporobolus</i> spp. (rats-tail)	0.8	2.1	1.9	8.7
<i>Cenchrus ciliaris</i>	0.4	0.0	3.4	4.8
<i>Heteropogon contortus</i>	0.0	0.8	3.7	6.4
<i>Themeda triandra</i>	0.0	0.4	1.6	4.8
<i>Verbena tenuisecta</i>	0.0	0.4	5.6	8.1

^ANote trees were killed (herbicide treated) in winter of 1994.

on the more fertile basalt soil site, Hillgrove, from less than 10% to over 100%. At the granitic soil site, Cardigan, which is similar to our ironbark site, on average over 8 years the advantage from tree killing was 28% for native pastures compared to 100% at Hillgrove. Our ironbark site eventually achieved a 35% pasture production benefit after 7 years, but it was never near the 100% which was achieved at our poplar box site. However, these experiments produced valuable insights on pasture responses to management inputs for the sustainable management of these *Aristida/Bothriochloa* native pasture landscapes, particularly for scientists and land managers.

Herbage mass

At the ironbark site the negligible change in herbage growth for the first 4 years after the trees were poisoned was unexpected, and also in the absence of grazing, when higher herbage mass was sometimes recorded where the trees were retained (Fig. 2b). This contrasts with the immediate increased pasture growth response at the poplar box site in all paddocks and plots. There, killing the trees resulted in a rapid improvement in herbage growth that was sustained for the 8-year experiment duration (Fig. 2c, d) without any retreatment to control some regrowth of trees and shrub seedling regeneration. We have no obvious

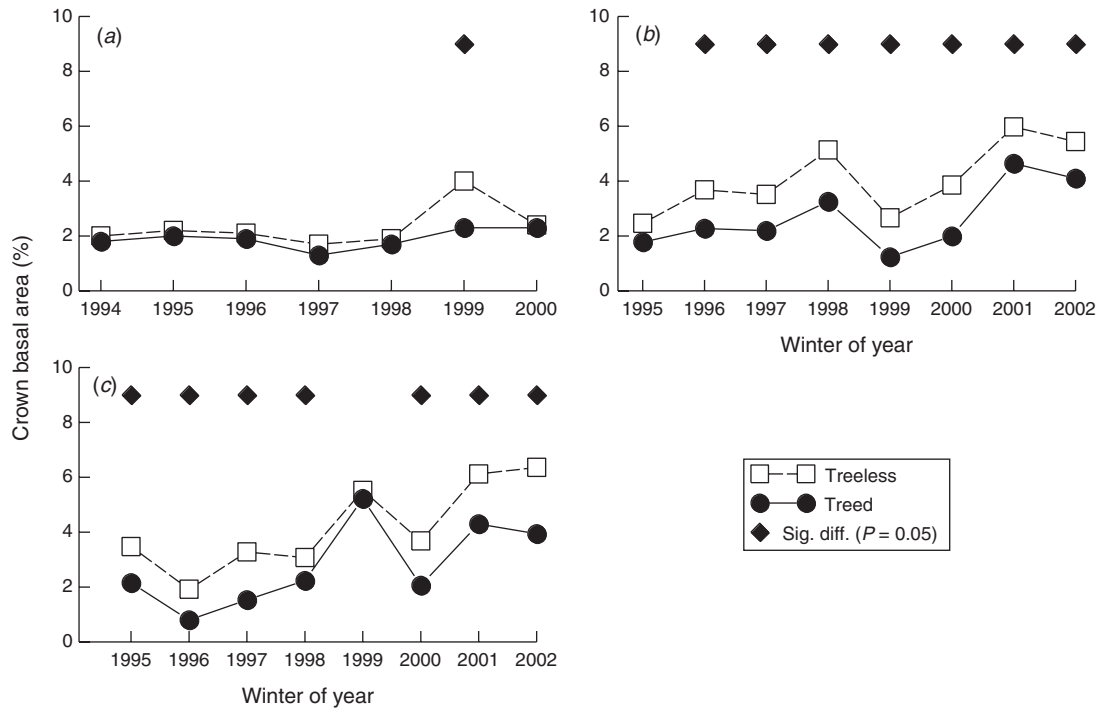


Fig. 8. Pasture crown basal cover (%) response to killing trees in grazed (a) *Eucalyptus melanophloia* (silver-leaved ironbark), and (b) *Eucalyptus populnea* (poplar box), and (c) in ungrazed *E. populnea* pastures, between 1994 and 2002. Diamonds indicate a significant difference ($P < 0.05$) for that year.

explanation for the early lack of initial pasture response at the ironbark site, which was confirmed with the detailed Swiftsynd methodology (Day and Philp 1997) measurements of inter-seasonal pasture growth. These were recorded at the same time in the same paddocks where the lack of pasture response to killing trees was recorded (Silcock *et al.* 1996). The explanation may involve soil fertility and water-holding capacity, tree rooting depth, nutrient cycling, soil fungal or insect pathogens or some weakness in the trees that had survived the preceding severe drought which then prevented them from competing with the pasture for several years. Death of savanna *Eucalyptus* trees in the Charters Towers region to the north during that drought was recorded by Fensham (1998).

Many other studies of tree density effects on pastures in Queensland show some decrease in pasture growth when the contrast is between a tree stem basal area of $\sim 5 \text{ m}^2 \text{ ha}^{-1}$ and virtually zero trees (Scanlan and Burrows 1990; Burrows 2002). Industry experience has expressed the view that, north of $\sim 20^\circ \text{S}$, tree clearing has not generally been an economic proposition for pastoralists (Burrows *et al.* 1988) and our ironbark site was at $23^\circ 22' \text{S}$. Economic studies place great emphasis on financial returns in the early years after large capital expenditure when trying to achieve a good Net Present Value and Internal Rate of Return over the expected payback period of an investment such as tree clearing (Star *et al.* 2010, 2015). Hence a lack of early potential improvement of pasture growth would significantly alter the outcome of a financial analysis of a tree clearing program in this community. It would also help to explain why commercial operations have indicated no benefits in tree clearing of natural woodlands in northern Australia, unless exotic

grasses and legumes are sown, or for particular purposes such as hay production.

These pasture responses occurred in the absence of significant early tree regeneration on most parts of both sites. That has not always been observed in this region, particularly over the long-term. Our poplar box trial site had been subjected to two cycles of tree killing in the past century because regrowth eventually reclaimed the landscape as woodland (property owners' records). Our data at that site has shown that the removal of trees had a dramatic and immediate effect on increasing herbage growth and consequently on stock carrying capacity. The ironbark site had never been cleared previously, although it was affected by periodic fires, and was experiencing major sapling recruitment, which was expected to eventually deplete herbage growth potential. Our data casts doubt on the universality of this assumed rapid pasture growth response to tree removal, although the tree competition effects that appeared by Year 5 may foretell such eventual suppression of the pastures by a dense mature woodland.

In general, herbage mass responded similarly to changing seasonal conditions when subjected to cattle grazing and to regular defoliation by way of fire after spring rain. The most extreme case of difference was at the ungrazed poplar box site, which after 4 years had accumulated a mass of moribund grass, and was then subjected to an unusually mild, wet winter. That led to widespread death of a dominant grass (*B. decipiens*) that appeared to be caused by significant fungal activity that rotted the crowns of many plants so that they did not regenerate during the following 1999 summer. Such ill-health was taken advantage of by the winter-active *C. lappulacea* (yellow

daisyburr), which grew a dense stand of large mature plants that survived into late 1999 before dying back to a normal population the following year.

Composition – herbage mass proportions

Generally, killing the trees did not alter the broad pasture composition greatly over 7–8 years. The exceptions were the significant increase in the important grazing species of *T. triandra* (Fig. 4c) at the ironbark site and *D. sericeum* at the poplar box site (Fig. 6a). These grasses are valued perennial forages (3P grasses), but neither persists well under grazing during poor seasons, compared to *B. ewartiana*. Scanlan and Burrows (1990) also reported an increase in the proportion of *T. triandra* in Queensland pastures as tree density decreased. They also support our findings at both sites that the removal of trees does not significantly alter the proportion of *Aristida* spp. in the pasture. Above average rainfall conditions produced major short-term changes in the growth of some perennial grasses such as *H. contortus* (Fig. 3b), and minor forb and legume species became prominent after above-average spring and autumn rainfall. Similar fluctuations in plant density and growth of *H. contortus* were reported by Orr *et al.* (2004) on a granitic soil supporting eucalypt woodlands in southern Queensland. This response was particularly noticeable at the poplar box site where a wider range of species responsive to winter rain, such as chenopods and *Calotis* spp., occur. Many are ephemeral, but *S. birchii* and *C. lappulacea* plants perennialize for several years and may dominate areas for some years after a favourable recruitment event.

The increase in standing herbage biomass at the poplar box site was provided by many species, but the major contributors were the previous dominant perennial tussock grasses, *A. ramosa*, *E. ramosus* and *D. sericeum* (Fig. 5). A similar differential response is found when the species data are grouped for intermediate forage value (2P) grasses, such as *E. ramosus*, or undesirable grasses, mainly *Aristida* species. Differences in the mean initial botanical composition among the various larger paddocks in the grazed experiments, measured after the first summer, tended to be maintained over time (Fig. 6), except for *D. sericeum*, which began with a similar mean percentage in the sward, but its herbage mass quickly more than doubled where the trees were killed (Fig. 7a). Such a change did not occur at the ironbark site (Tables 2 and 3), which re-emphasises the need for caution when extrapolating generally held wisdom to every environment within a pasture community in theoretical exercises or in the subject matter of grazing land management courses.

Dichanthium sericeum is known to be sensitive to extreme rainfall years and builds up in a run of wet summers and declines markedly during drought years (Blake 1938; Jacobsen 1981). It has a shorter average lifespan per plant (3–4 years) than most of the desirable 3P grasses of Australia's tropical savannas, although some of the *Aristida* spp. at the poplar box site are also relatively short-lived perennials, such as *A. latifolia* (Orr and Phelps 2013) and *A. calycina*. Hence, it would not have been unexpected if the production of *D. sericeum* in these pastures fluctuated markedly over time if special seasonal conditions were needed to recruit new seedlings as old plants died. As this

appeared not to happen, it indicates that the production of viable seed and seedling recruitment is consistently achieved by this valuable forage grass in that environment. This is despite a lack of evidence of a large viable soil seedbank for this grass at these sites (Silcock *et al.* 2005).

Under grazing without *E. populnea* tree competition, *C. divaricata* increased its production, especially on more open or bare surfaces of the medium and heavier textured soil types. This recruitment produced an increase in the standing autumn pasture from an initial 2.1% to 11.7% by 1999, before drier summers reduced its contribution back to 5%. As such, it was one of the common minor short-lived perennial forage species. In this woodland, its main values were that it initiated new growth rapidly after rain in early spring before the *Bothriochloa* species, *C. fallax* or *E. ramosus*, and it tolerated regular selective grazing at this time of the year. Its dense population of small plants provided cover and stability to the relatively bare soil surface at the end of winter. It is an increaser species at this site because of its ability to tolerate heavy grazing and was common in the patch-grazed areas in the treeless treatments along with *Enneapogon* spp. The herbaceous floristics of multiple sites in the *E. populnea* community of the Maranoa region of Queensland have been described by Scheffe *et al.* (1993) and show a similar species composition and variation across soil types as occurred across the Injune trial site.

Species frequency

Species frequency was more sensitive to our management treatments than was their percentage contribution to the total herbage mass. This reflects the small and seedling plants that are being recorded that contribute little biomass or seed compared to well grown plants that are usually more tolerant of defoliation and climatic stresses. Thus recording and monitoring systems based only on species frequency, such as GrassCheck (Forge 1994) and those that use point-centred recording methods such as a wheel-point apparatus (Tidmarsh and Havenga 1955), are likely to indicate appreciable temporal vegetation change that is illusory and potentially short-term, such as ground cover in more arid environments. The frequency measure is valuable for conducting a census of the vegetation, but is less able than percentage perennial herbage mass, to indicate stability and resilience of the pasture to management treatments, climatic extremes or of its grazing value.

Pasture crown basal area

Crown basal area of herbaceous species is a comparatively stable feature of perennial grass pastures and, as our data show, does not change markedly or rapidly in response to normal open woodland management shifts. The removal of all large trees produced an enhancement at both sites (Fig. 8), but larger changes arose from extremes of seasonal rainfall. An increase in basal cover due to tree removal was significantly higher at the poplar box site where that management also had the greater impact on herbage mass production. There the basal area remained well above 3% after the wet summers of 1997–1999, which is regarded by Silcock (1993) as indicative of healthy native pasture in southern inland Queensland. Peak values over 6% in treeless plots are unusual and were largely due to the prolific growth of

C. fallax in the absence of cattle grazing and *Macropus rufogriseus* (red-necked wallaby) digging. These marsupials fed on the moist, carbohydrate-rich rhizomes of this grass in dry winters when other suitable forage was scarce.

Where tree removal enhanced herbage growth, the area of perennial grass crowns also expanded, which substitutes in part for the loss of live tree stems. This contributes to greater landscape stability even if an appreciable proportion of the crowns come from small-statured perennial plants such as *C. divaricata*, *T. loliiformis*, *C. gracilis* and *F. dichotoma*. Though regarded as increaser species that are indicative of overgrazing, their cryptic presence in dry times belies their value in stabilising the soil surface against erosion from spring storms, even if they do not greatly impede runoff.

Conclusion

Removal of tree competition by herbicides without soil disturbance from open eucalypt woodlands in central Queensland does not always produce an immediate marked increase in the growth of native pasture, although it can double herbage growth in some woodland types, such as *E. populnea*. Also tree killing does not greatly alter the broad species composition of the pastures, but a small number of valuable forage grasses such as *D. sericeum* and *T. triandra* do benefit, with or without associated cattle grazing. Seasonal conditions can have a more immediate effect on individual species contribution to the pasture than either tree competition or the presence of cattle grazing. We conclude from this study, that killing blocks of eucalypt trees in these open woodlands on undulating country can significantly improve productivity, although the rate of change can vary between communities, and it does not adversely affect landscape stability, pasture composition or production potential. When all such biological parameters are factored into economic models and regional predictions of grazing land sustainability, the outcomes may be appreciably different from previous bio-economic studies.

Acknowledgements

We thank the Hawkins and Hicks families for the use of their land and supporting our research at Rubyvale. At Injune we also thank the Chandler family for their assistance and access to land. Joff Douglas, David Osten, Stephen Ginns, Jill Aisthorpe, Ann Sullivan, Melinda Cox, Scott Brady, Matthew Hall and Jodie Trace provided capable technical assistance during the experiments. Dr David Mayer, Ms Christine Playford and Ms Kerry Bell assisted with the statistical analyses and Ms Kathy Delaney prepared the figures for publication. A consultative committee of local producers at both locations provided valuable suggestions to the research team, especially about the practical effects of our findings in their environment. We thank the editors and two anonymous reviewers of this paper for their insightful and constructive recommendations. Operating funds were provided by the Australian Meat Research and Development Corporation and later by Meat and Livestock Australia. Their vision initiated the study, which the Queensland Government supported with its field staff.

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