Spring fire effects on two *Aristida/Bothriochloa* native pastures in central Queensland, Australia

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Abstract. Controlled burns are commonly used to suppress woody plant regrowth and to remove accumulated unpalatable pasture from rangelands and occasionally to alter pasture composition in native pastures in central Queensland, Australia. Outcomes can be somewhat unpredictable and short-term, and reliable evidence is needed to confirm the likely long-term efficacy of such fires. We imposed a regime of repeated spring burns on native *Aristida/ Bothriochloa* pastures growing in two contrasting eucalypt woodlands of central Queensland to determine the effects on pasture composition, ground cover, landscape stability and woody plant recruitment, all in the absence of grazing. The sites selected were a silver-leaved ironbark (*Eucalyptus melanophloia* F.Muell.) woodland and a poplar box (*E. populnea* F.Muell.) woodland.

Weather conditions precluded spring burns in 3 years out of 7 at the silver-leaved ironbark site and in 2 years out of 8 at the poplar box site. The burn intensity was variable, and frequent fires produced a marked change in abundance of only a few pasture species. Depending on the site, fires significantly increased the frequency of *Enneapogon* spp., *Bothriochloa bladhii* (Retz.) S.T.Blake and *Dichanthium sericeum* (R.Br.) A.Camus and reduced the frequency of some minor components such as *Cymbopogon* spp., *Panicum effusum* R.Br., *Cenchrus ciliaris* L. and, ephemerally, that of some forbs. Contrary to expectation, only *Aristida calycina* R.Br. declined in abundance among the many *Aristida* species present, and the abundance of *Heteropogon contortus* (L.) P.Beauv. ex Roem. & Schult. barely increased under regular spring fires. The total germinable seeds of herbaceous species in the soil each spring was significantly reduced by burning in the previous spring.

Repeated spring fires rarely reinforced any initial change induced by burning, and slightly lowered average ground cover as well as various indices of landscape stability and ecosystem functionality. Changes produced were not always consistent across the two communities. Though prescribed burning is often important for maintaining grazing productivity and landscape values, very regular use is not recommended.

Additional keywords: *Eucalyptus melanophloia*, *Eucalyptus populnea*, ground cover, landscape ecology, pasture composition, seed banks.

Introduction

Grassy woodlands and open grasslands in rangelands commonly experience fires (Trollope and Trollope 2004; Scheintaub *et al.* 2009; Cowley *et al.* 2014), and many native species are adapted to this threat (Hodgkinson and Harrington 1985; Engle and Bidwell 2001; Snyman 2003). Some species tolerate fire well once well grown, e.g. most *Eucalyptus* spp. and perennial grasses, but most *Acacia* species, Chenopodiaceae species (saltbushes) and annual plants are fire sensitive and persist via their seedbanks (Western Australian Government 2018). The result of irregular fires is sometimes a shifting interface

between adjacent vegetation communities with differing wildfire tolerance, such as grassland and fire-sensitive *Acacia aneura* (mulga) woodlands or rainforest (Blake 1938). Botanical nomenclature throughout follows that of Henderson (1997).

Mass recruitment of some eucalypts after fires is common (Wellington and Noble 1985; Peeters and Butler 2014; Gillen 2017). Cool spring burns are potentially beneficial to tree and shrub recruitment, because grass competition is temporarily reduced and soil and litter fungal spores, which can attack their seedlings, are probably fewer after the fire (Facelli *et al.* 1999). Control of small eucalypt seedlings that germinate in autumn is

also possible by intense burns in the recruitment year before they have gained resprouting ability from lignotubers (Williams 2009). Most woody weed control by fire depends on a hot burn and is normally carried out as late in the spring as possible, before the wildfire risk becomes high but rapid pasture regrowth is likely. Eucalypt woodlands are very well adapted to fires that mainly act to suppress or reduce growth of woody plants (Williams 2009; Jacobs 2018).

Sustained heavy grazing of grassy pastures increases the proportion of unpalatable biomass and reduces the severity and frequency of fires to the extent that certain fire-sensitive or unpalatable species proliferate while others decline in abundance (Leach and Givnish 1996; Vermeire *et al.* 2014). To manage such potential shifts and reduce wildfire risk, pastoralists and local councils carry out strategic burns to shift the vegetation balance to a less threatening state (Mapiye *et al.* 2008). Some pastoralists regularly burn dry, rank tropical pastures to stimulate new green growth while there is still sufficient deep soil moisture to elicit a green growth response from which their livestock can benefit, although Trollope (2004) strongly disapproves of this tactic because it places severe stress on regrowing plant crowns and roots. Many prescribed burns aim primarily to destroy woody undergrowth and seedling trees that reduce grazing value (Jones and Burrows 1994; Burrows 2002) and make livestock difficult to muster.

Some prescribed burning aims to alter pasture composition away from increasing proportions of undesirable grasses, such as *Aristida*, *Elionurus* and *Cymbopogon* spp. (Orr *et al.* 1991; Engle and Bidwell 2001; Snyman 2015). However, fire can stimulate the germination of many species such as *Heteropogon contortus* (black speargrass) (Campbell 1996; Snyman 2005) and legumes (Keeley and Fotheringham 2000; Williams *et al.* 2003*b*). Are these changes in pasture composition largely due to death of old plants, death of seedlings, a reduction in flowering and seed-set, or the destruction of most of the recent seedbank (Phelps 2006)? Recently shed fluffy seeds of many rangeland grasses lie loosely low in the pasture canopy, in fresh litter or on the soil surface for many months and would presumably be easily killed by even a mild fire. Conversely, some species have awned seeds that promote seed burial, which may protect seedbanks from fire (Campbell 1996; Snyman *et al.* 2013).

We aimed to document the effects of regular spring burns on the composition of two contrasting *Aristida*/*Bothriochloa* pastures (Weston *et al.* 1981), on ground cover and on woody plant recruitment. Springtime was chosen because it is the most common time of year used in commercial practice for prescribed burning in subtropical woodlands. We hoped to achieve an appreciable reduction of unpalatable *Aristida* spp. (wiregrasses), akin to those reported by Orr and Paton (1997) and Cowley *et al.* (2014). Regular burns were implemented because that would presumably hasten any potential changes mediated via altered recruitment and death rates of plants. Many changes were possible, but the main hypotheses investigated were:
(1) Spring fires can reduce the proportion of *Aristida* spp. and other low palatability grasses;
(2) Spring fires do not adversely affect the important 3P (productive, palatable, perennial) grasses in the pasture; and
(3) Regular spring fires after rain do not, in the absence of grazing, damage landscape functioning.

Materials and methods

Site locations

Ironbark site

An ironbark site was chosen near Rubyvale (23°22.37′S, 147°35.56′E) on the gently undulating Peakvale land system (Galloway *et al.* 1967) in a *Eucalyptus melanophloia* (silver-leaved ironbark) woodland with scattered *Bursaria incana* (prickly pine) and *Corymbia erythrophloia* (variable-barked bloodwood) in the tree overstorey. It was in A⁻ (good) grazing land condition (Karfs *et al.* 2009) with good pasture composition, slight streambank erosion and negligible shrubby understorey. Mean tree basal area was ~6.5 m² ha⁻¹, but tree density was ~2000 stems ha⁻¹ due to a large number of ironbark saplings. This site lies within Regional Ecosystem 11.12.2 (Sattler and Williams 1999) and has a gritty, hard-setting, reddish, texture-contrast soil derived from granite – a haplic, eutrophic, red Chromosol in the Australian Soil Classification system (Isbell 1996). The pasture layer was dominated by *H. contortus* and *Bothriochloa ewartiana* (forest Mitchell grass) plus appreciable contributions from *Chrysopogon fallax* (golden-beard grass) and *Themeda triandra* (kangaroo grass) (Hall *et al.* 2016), while *Aristida* spp. were a minor component.

Poplar box site

A poplar box site was chosen near Injune (25°44.36′S, 148°25.71′E) in the Bymount land system (Macnish 1987) in a *Eucalyptus populnea* (poplar box) woodland with scattered *Eremophila mitchellii* (false sandalwood), *Myoporum acuminatum* (boobialla) and *Callitris glaucophylla* (white cypress pine) in its understorey. It was on almost level terrain and was old regrowth in an area that had been ringbarked decades before and now had a tree basal area of 10.4 m² ha⁻¹ and 344 stems ha⁻¹. The area was in B⁻ (fair to poor) grazing land condition (Karfs *et al.* 2009) with poor pasture composition. This site, within Regional Ecosystem 11.9.7, had a hard-setting, shallow, grey, loamy-surfaced soil over a blocky, alkaline saline-sodic clay subsoil – a calcic, brown to grey Sodosol with variable subsoil salinity. The understorey pasture was dominated by *Aristida* spp., mostly *A. ramosa* (purple wiregrass) and *A. calycina* (dark wiregrass), *Bothriochloa decipiens* (pitted bluegrass), *C. fallax* and *Dichanthium sericeum* (Queensland bluegrass).

The two sites have a similar climate. Most pasture growth occurs between October and March, when 75% of the annual rain falls and hotter temperatures are optimal for growth of the C₄ grasses that dominate these pastures. However, both sites can receive appreciable rainfall at almost any time of year, which can quickly change pasture greenness, ambient temperature, humidity and fire hazard.

Experiment layout

Each experimental site consisted of 12 square or rectangular 1-ha plots, with the trees on six of them recently killed by herbicide applied by commercial operators using the 'knick and squirt' stem-injection method (Kyser *et al.* 2015). The numerous small silver-leaved ironbark saplings at the ironbark site were treated with a squirt of chemical to the soil at their stem base. The ironbark site was treated in March 1994 using hexazinone (Velpar®), while picloram + 2,4-D (Tordon 50-D®)

was used at the poplar box site in July 1994. The remaining six plots were left untouched to assess the interaction of burning with live tree removal on pasture composition and ground cover. Both sites were treated near the end of a significant drought period.

At the ironbark site, the plots were scattered within a 200-ha grazing management research site, whereas they were closely but randomly grouped in a single block at the poplar box site (Hall *et al*. 2016). Plots to be burnt were encircled by a 3-m-wide graded firebreak. Each of the four treatments, ± trees × ± burning, was replicated three times. The first attempted burn was in spring 1994 at both sites.

Burning protocol

The intention was to burn the same six plots each spring, three with the woody plants untouched and three where herbicide had been used. The other six plots, three treed and three treeless, were left unburnt throughout the study. However, at the ironbark site a scheduled burn in 1998 escaped, burning some of the 'non-burnt' plots. Consequently, all the other plots were burnt that spring with a 'cool' fire so as to standardise the overall comparisons.

Timing of a burn was to be during warm weather in spring (September–November) when wind conditions were favourable, shortly after a fall of 25–50 mm of rain, and where sufficient standing fuel existed (>1200 kg ha^{-1}). As all plots were ungrazed, fuel load was not expected to be a problem, especially at the poplar box site, where fencing excluded marsupials. However, weather conditions in some springs were not conducive to a burn. Dates of burns and basic environmental

conditions at the time are recorded in Table 1. Flame heights were recorded, but fire intensity was not quantified. Scorch ratings and refoliation extent of tree canopies after a burn were sometimes recorded.

Sampling techniques

Pasture biomass (total and main species), frequency (percentage of quadrats possessing each recorded species) and projected cover of the surface soil were recorded each autumn via the Botanal technique (Tothill *et al*. 1992). At least 40×0.25 m^2 quadrats were assessed in each plot at each sampling, with the estimated percentage of the biomass of the six most abundant species in a quadrat recorded. Early each winter, the crown basal cover by living plants was recorded using a 5-pin frame with pins 25 cm apart to record the percentage that struck a living tussock at ground level (Bonham 2013). This was assessed along the entire length of two fixed 100-m-long transects in each plot.

Germinable seedloads were estimated by collecting four cores 5 cm in diameter and 5 cm deep from the centre of each plot in early spring, when the soil was very dry before the planned burns. The soil and litter were sieved and spread thinly on the surface of sand-filled pots in a shadehouse in late spring/ early summer. The pots were then kept well watered with town water by overhead sprays for 4–8 weeks, until the rate of new seedling appearance was negligible. All emerging seedlings were counted and identified to species where possible.

At the end of the experiments (in 2001 and 2002 for the ironbark and poplar box sites respectively), the Landscape Function Analysis (LFA) procedure of Tongway and Hindley

Table 1.   Burn dates, ambient conditions and flame height at the two experimental sites
Curing describes the percentage of the standing pasture biomass that was dead

| Year | Ironbark site | | Poplar box site | |
	Weather conditions	Burning effect	Weather conditions	Burning effect
1994	very dry, low fuel, high risk of fire escaping	no safe or effective burn possible	3 Nov. – warm, dry, no wind; little grass fuel but plenty of tree leaf litter	very slow, cool burn; 2 m max flame height
1995	12 Oct. – cool, moist, low wind	slow trickle burn; negligible scorch of tree foliage	5 Oct. – cool, no wind; greenness in grass; abandoned attempt	fire would not run, especially under trees; only two treeless plots burnt;
1996	cool, moist weather	no burn possible after August	19 Sep. – dry, warm, gusty wind	good burn where treeless, slow under trees; 4 m flame height;
1997	27 Aug. – 70% grass curing; tricky weather, high fuel load (>4000 kg ha^{-1})	mild burn but some tree foliage scorch	14 Oct. – dry, warm, gusty wind; some greenness in grass	good burn where no trees, patchy but fair burn under trees
1998	18–24 Sep. – 50% grass curing; mild wind and weather; ALL PLOTS BURNT	trickle burn needing regular relighting	cool, green wet grass and dense forbs	No burn possible
1999	9 Nov. – 80% grass curing; weather favourable	headfire with 3-m flame height	24 Aug. – cool, dry fuel, low humidity	good burn, 4-m flame where treeless but <1 m and patchy under trees
2000	pasture too green to burn in spring	no burn	26 Oct. – cool, cloudy, windy; green bases to grass; 50 mm rain after	good burn where treeless, 4-m flame height; low slow burn under trees
2001	experimental site closed down	no burn	16 Oct. – mild, cloudy, gusty wind; some green in grass crowns	excellent burn, 4-m flame where treeless; slower, low burn under trees; dead Tordoned trunks burning

(1995) was applied along the transect more closely aligned with the slope of each plot. Three indices of landscape functionality – Stability, Nutrient Cycling and Infiltration – were then calculated, using the Tongway and Hindley LFA formulae in a similar manner to that of Maestre and Puche (2009).

The ironbark site experiment ran for 7 years, until April 2001, and the poplar box one for 8 years, until June 2002.

Statistical analysis

Statistical analyses of the results were undertaken using the GENSTAT package (GENSTAT 2015). 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. This forms an approximate split-plot analysis of variance (split for time). A Greenhouse–Geisser epsilon value was calculated, which estimates the degree of temporal autocorrelation and adjusts the probability levels for this effect. Analyses included all main effects of tree killing, spring burning and year and their interactions, but the data presented in this paper about spring burns has the tree effect aggregated within it. Interactions between tree presence and burning were rarely statistically significant. Significant differences were determined by the l.s.d. process, and $P < 0.05$ is reported as a significant effect. Data were transformed if mean values were not independent of variance.

Results

Climate and seasons

Seasonal conditions in 1994, 1996 and 2000 at the ironbark site, and in 1995 and 1998 at the poplar box site were such that pasture fuel loads on burnt plots were too low or too green to carry a spring fire. The 1998 winter plus spring was exceptionally wet at both sites, as was the 1996–1997 summer and the 2000 spring at the ironbark site (Table 2). See Hall *et al.* (2016) for other seasonal rainfall details.

Fire

The poplar box site achieved more fires than the ironbark site (6 versus 4, Table 1). Fire intensity was variable due to the

variation in the prevailing weather after sufficient rain had fallen to allow a prescribed burn. Appreciable scorch of tree foliage was often achieved, and thus complete consumption of most grass tussocks and the leaves of small saplings. Beneath untreated trees at the poplar box site, where pasture biomass was sometimes suboptimal for a good burn, carry of fires was improved by the tree leaf litter. More details about individual burns are given in Table 1.

Pasture biomass changes

The general nature of the pastures at both sites was not changed by repeated spring burns, although the abundance of some annual and weakly perennial species did vary considerably in the short-term, and the abundance of some true perennials did change slowly.

Ironbark site

Biomass increased in nearly every year, with yields in 2001 of between 7000 and 9000 kg ha^{-1} (Fig. 1*a*). This reflects the many average to above-average summer growing seasons between 1995 and 2000 (Table 2). After the unplanned burning in September 1998, both treatments at the end of the ensuing summer had an average biomass of 4000 kg ha^{-1}, but thereafter the regularly burnt plots had a significantly lower biomass.

Table 2. Rainfall during the experiments (spring is Sept–Nov, summer is the Dec–Apr period preceding the year listed)

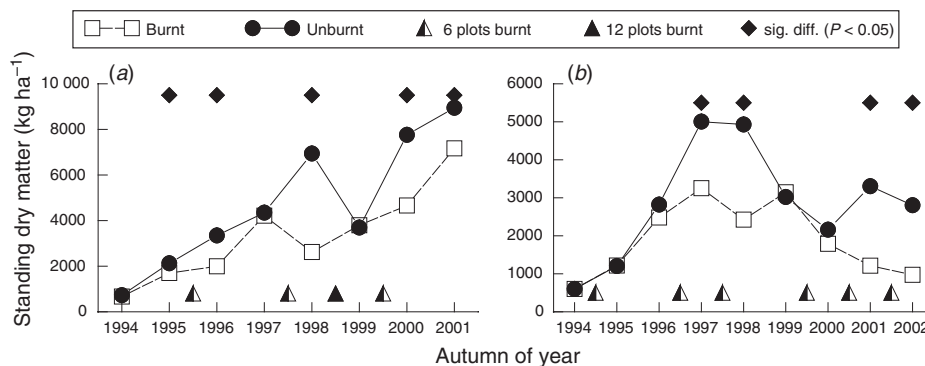| Year	Ironbark site		Poplar box site	
	Spring of year	Prior summer	Spring of year	Prior summer
1994	21	196	120	319
1995	133	410	167	228
1996	189	239	185	337
1997	119	552	173	348
1998	250	281	240	453
1999	143	392	63	462
2000	314	359	183	269
2001	–	–	133	331
125-year median	104	391	122	336

Fig. 1. Standing autumn pasture biomass (kg ha^{-1}) of ungrazed pasture with and without burns the previous spring. (*a*) Ironbark site (1994–2001): no burns in 1994, 1996 and 2000 and all plots burnt in spring 1998. (*b*) Poplar box site (1994–2002): no effective spring burn in 1995 and 1998. A '◆' indicates significant differences within years ($P < 0.05$); a ▲ indicates when designated plots were burnt and ▲ when all plots were burnt in 1998 at the ironbark site.

On average across all years, the autumn pasture biomass was dominated by *B. ewartiana* (35%), *H. contortus* (22%), *T. triandra* (16%) and *C. fallax* (9%), with *Enneapogon* spp. (bottlewasher grasses, 3%), native legumes (1%) and *Aristida* spp. (0.6%) as minor components.

Poplar box site

Pasture biomass at the ungrazed poplar box site increased steadily for 2 years without any significant effect of burns until autumn 1997, aided by the lack of a spring fire in most plots in 1995. Thereafter, standing pasture each autumn after a burn was significantly reduced (Fig. 1*b*), except in autumn 1999, when no prior spring burn was possible, and in 2000, when the fire under treed plots the previous spring was patchy (Table 1). There was a large accumulation of dead grass in the unburnt plots after 4–5 years that restricted new pasture growth, and there was a steady autumn biomass decline in the burnt plots in the drier years after 2000. Averaged over 8 years, the dominant biomass species were *B. decipiens* (28%), *Aristida* spp. (15%), *C. fallax* (11%), *D. sericeum* (6.5%) and *Enneapogon* spp. (4.5%). Native legumes usually provided <1% of pasture biomass, although their frequency in quadrats often exceeded 15%.

Changes in abundance of low palatability grasses

Ironbark site

There was a low proportion of unpalatable grasses at this site, and spring burning did not alter their biomass or frequency. Many of the nine species of *Aristida* here were of the finer-stemmed, more palatable type. They occurred with decreasing frequency over time, regardless of burn treatment (initially 21%, Table 3), and contributed only a small proportion to the total pasture biomass (mean 0.6%) and to the spring seedbank (6% of all grasses). However, many *Aristida* seeds did germinate

every year from spring soil samples, but there was no effect of fire (Table 4).

Poplar box site

A much higher proportion of poplar box pasture was relatively unpalatable compared with the ironbark site, notably *A. ramosa* and *A. calycina* among the six *Aristida* species recorded. Standing biomass after the year 2000 of *Aristida* spp. and *Cymbopogon* spp. (Fig. 2*b, d*) was much reduced by regular burning, but that of *B. decipiens* was generally not (Fig. 2*a*). The *Cymbopogon* spp. effect failed to be statistically different because of a most uneven distribution of tussocks that grew very large if not burnt. Burning prevented an increase through time in *A. ramosa* (from 30 to 43% frequency by 2002 without fires) and led to a significant decrease in frequency of *A. calycina* (from 21 to 11% over the same period) (Table 5). This opposite trend in frequency was supported by demography data reported in Silcock *et al*. (2005). *Bothriochloa decipiens* contributed a large proportion of the germinable spring soil seedload (13.5%), much greater than the 2.3% from *Aristida* spp. The burning regime did not significantly affect the spring soil seed reserves of either taxon (Table 4), both of which exhibited great spatial variability of germinable seeds.

Changes in abundance of important perennial grasses

Ironbark site

Regular spring burning after rain decreased the autumn biomass of *B. ewartiana* significantly in 1997, 1998 and 2000 (Fig. 3*a*), and that of *T. triandra* in 1998 and 2000 (Fig. 3*c*). The reduction of *B. ewartiana* had a major influence on current total pasture biomass. *Themeda* was unusual by exhibiting a significant interaction ($P < 0.05$) in biomass in some years between spring burning and tree killing. Where trees remained, its proportion of pasture biomass by 2001 was greater after regular spring burning (32% vs 19%), but less after regular fires where trees had been poisoned (17% vs 34%). Burning in the absence of grazing did not stimulate the growth of *H. contortus* (Fig. 3*b*), but total biomass rose in parallel with increasing plant frequency (Fig. 4*d*). Spring burns had minimal impact on the frequency of most perennial grasses by 2001 (Table 3), but

Table 3. Cumulative effect of the spring burning regime on the frequency (%) in the ungrazed pasture of major pasture species at the ironbark site between 1994 and 2001
Data combines treed and herbicide-treated plots. *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$

Taxon	2001		Change 1994–2001		Signif. diff.[A]
	Burnt	Unburnt	Burnt	Unburnt	
Aristida spp.	17.5	11.1	–4.3	–9.7	
Bothriochloa ewartiana	59.4	63.9	18.1	21.5	
Chrysopogon fallax	45.6	35.7	–3.7	–12.8	
Cyperaceae species	8.4	7.0	–20.1	–35.0	
Dichanthium sericeum	8.4	5.7	3.5	3.8	
Enneapogon spp.	26.9	11.8	14.4	–4.2	**
Eragrostis spp.	2.7	3.4	–2.7	–5.5	
Eulalia aurea	1.6	2.0	–3.4	–9.5	*
Forbs	49.6	45.5	–28.0	–37.4	
Heteropogon contortus	47.2	46.4	26.4	18.0	
Native legumes	37.5	57.8	–20.6	–13.6	***
Panicum effusum	1.2	1.7	–16.9	–24.4	
Themeda triandra	42.7	51.9	29.6	37.7	
Tripogon loliiformis	5.3	1.6	–19.5	–19.8	

[A]The difference relates to the % frequency in the treatments in 2001.

Table 4. Number of seedlings (m^{-2}) of major species emerging from the spring soil samples, summed for all years and aggregated for tree cover at the ironbark and poplar box sites

Taxon	Ironbark site		Poplar box site	
	Burnt	Unburnt	Burnt	Unburnt
All grasses	9728	10374	3267	4052
Aristida spp.	532	646	509	424
Bothriochloa decipiens	0	0	1676	1782
Bothriochloa ewartiana	1596	1064	0	0
Cyperaceae species	760	2204	743	2482
Dichanthium sericeum	38	190	488	467
Enneapogon spp.	950	798	85	64
Enteropogon ramosus	0	0	64	382
Heteropogon contortus	4142	5168	0	0
Wahlenbergia spp.[A]	11058	23598	21	361

[A]Fires significantly reduced ($P < 0.05$) *Wahlenbergia* spp. total seedbank numbers at the ironbark site.
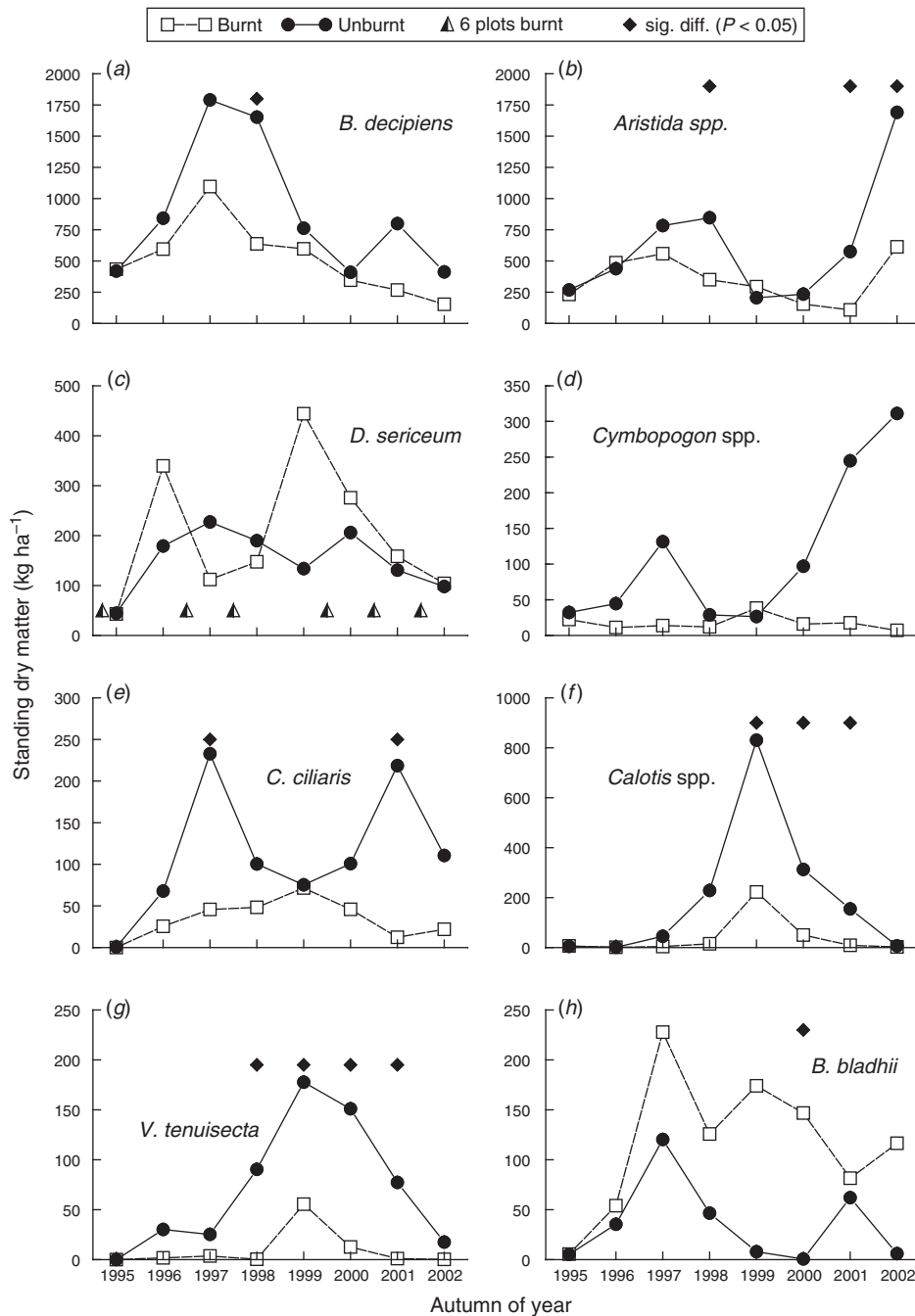
Fig. 2. Effect of the spring burning regime on the standing autumn pasture yields (kg ha^{-1}) of major and some minor species at the poplar box site over time – (*a*) *Bothriochloa decipiens*, (*b*) *Aristida* spp., (*c*) *Dichanthium sericeum*, (*d*) *Cymbopogon* spp., (*e*) *Cenchrus ciliaris*, (*f*) *Calotis* spp., (*g*) *Verbena tenuisecta* and (*h*) *Bothriochloa bladhii*. A '◆' indicates significant differences within years ($P < 0.05$); a ▲ indicates when designated plots were burnt.

they significantly decreased the short-term frequency of *B. ewartiana* compared with that in unburnt plots in the autumn of 1998 and 2000 (Fig. 4*a*). *Chrysopogon fallax* had a consistently high frequency (mean 40%, Table 3) under regular burning, similar to that of the other main perennial grasses, but a smaller aboveground biomass (mean 374 kg ha^{-1}).

The biomass of major grasses was not closely reflected in their spring soil seedload, with *B. ewartiana* (5.1%) having far fewer germinable seeds than *H. contortus* (12.8%) and only double that of the *Aristida* spp. from all years (Table 4). In individual years, *B. ewartiana* had highly variable numbers, whereas *H. contortus* germinated in appreciable numbers each

Table 5. Effect of the spring burning regime in changing the frequency (%) of common pasture species in ungrazed pastures at the poplar box site between 1995 and 2002

Data combines treed and herbicide-treated plots. *$P<0.05$; **$P<0.01$

Taxon	2002		Change 1995–2002		Signif. diff.[A]
	Burnt	Unburnt	Burnt	Unburnt	
Aristida calycina	10.6	24.0	−10.7	2.1	**
Aristida ramosa	25.2	43.1	0.2	12.7	*
Bothriochloa bladhii	7.1	1.9	2.9	0.2	**
Bothriochloa decipiens	47.6	47.2	−7.4	−1.8	
Brunoniella australis	6.1	3.9	−26.8	−14.3	*
Cenchrus ciliaris	2.2	6.1	2.2	5.7	**
Chloris divaricata	11.9	7.3	8.2	3.2	
Chrysopogon fallax	53.5	30.7	1.8	−22.0	*
Cymbopogon spp.	2.7	6.4	−0.6	1.0	*
Dichanthium sericeum	20.7	10.9	13.2	2.6	
Enneapogon spp.	26.2	22.9	4.9	0.5	
Enteropogon ramosus	1.5	6.4	0.3	4.9	
Eragrostis spp.	3.5	1.2	1.1	−2.5	
Heteropogon contortus	7.5	2.7	7.5	1.7	
Legumes – palatable	15.0	14.2	−10.8	−5.4	
Legumes – poisonous	4.2	0.0	0.5	2.1	**
Panicum effusum	2.8	5.3	−2.2	0.9	*
Themeda triandra	3.4	2.9	3.0	2.9	
Verbena tenuisecta	1.9	11.8	1.5	11.8	*

[A]The difference relates to the percentage frequency in the treatments in 2002.

year. Burning had no significant effect on the number of germinants of either grass. Only three *C. fallax* and no *T. triandra* seeds germinated, despite a major presence in the sward (Table 3).

Poplar box site

Spring burns had no marked effect on autumn biomass of *D. sericeum* (Fig. 2c) or *H. contortus*, which was a minor species (2.5% of biomass and frequency). However, burning increased the frequency of *D. sericeum* by 2002 from an initial 8% to 21%, compared with reaching only 11% in unburnt plots (Table 5, Fig. 5e). That difference was non-significant in 2002, but the effect was consistent in the intervening years (Fig. 5e), with an 8-year average of 21% versus 12% respectively. Behind that trend were fluctuations in density and mean plant crown size of the charted plants of this species, particularly in treeless pasture (unpresented data; see Silcock *et al.* 2005). Regular burning had no significant effect on the biomass of *C. fallax*, but its frequency steadily and significantly decreased through time in unburnt plots from an initial 53% to 31% by 2002 (Fig. 5f). *Bothriochloa bladhii* (forest bluegrass) biomass increased from a very small initial presence, but increased more with regular spring burns (Fig. 2h). *Cenchrus ciliaris* (buffel grass) biomass also increased from a very low base to over 200 kg ha^{-1}, which was better maintained (Fig. 2e) if spring burns did not occur. By 2002, its frequency was significantly less ($P<0.05$) if pastures were regularly burnt, 2.2 versus 6.1% (Table 5). Germinable
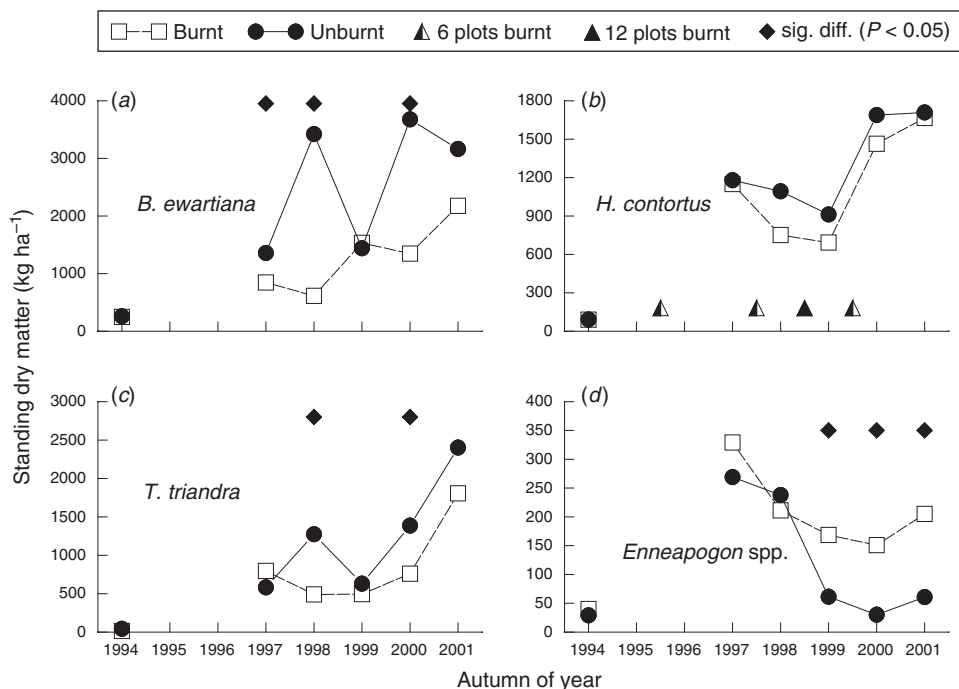

Fig. 3. Spring burning effects on standing autumn pasture biomass (kg ha^{-1}) of major perennial grasses and *Enneapogon* spp. at the ironbark site. No data available for 1995 and 1996. No burn was possible in spring of 1994, 1996 and 2000. A '◆' indicates significant differences within years ($P<0.05$); a △ indicates when designated plots were burnt and ▲ when all plots were burnt in 1998.

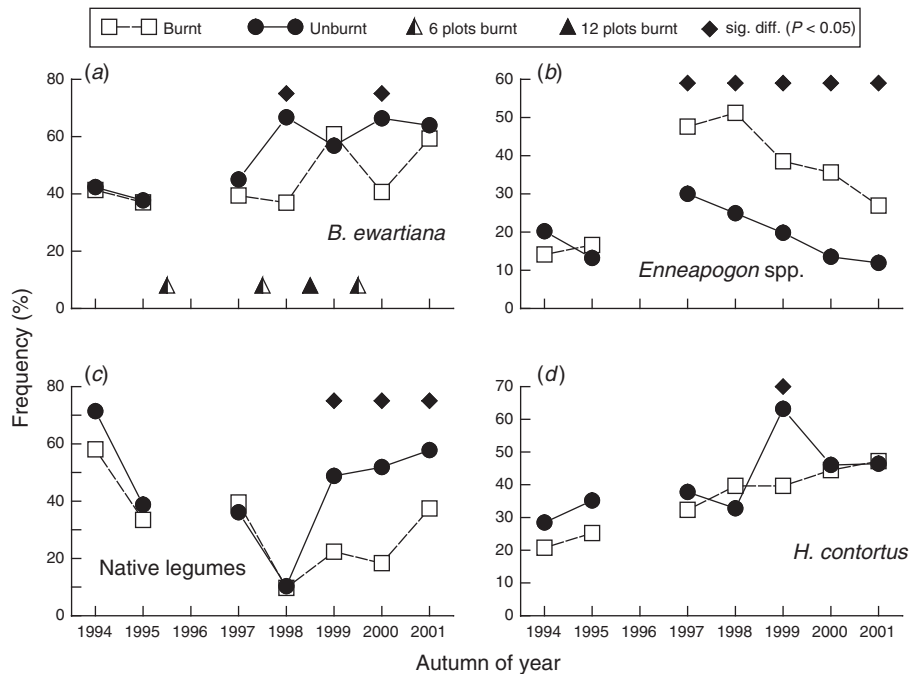**Fig. 4.** Changes in autumn frequency (%) associated with spring burning at the ironbark site of (*a*) *Bothriochloa ewartiana*, (*b*) *Enneapogon* spp., (*c*) native legumes and (*d*) *Heteropogon contortus*. A '◆' indicates significant differences within years (*P* < 0.05); a △ indicates when designated plots were burnt and ▲ when all plots were burnt in 1998.

seed loads of the key palatable grasses such as *D. sericeum* were unaffected by spring burning (Table 4).

Changes in abundance of non-dominant species

Ironbark site

Regular burning had minimal impact on the biomass of minor components of the pasture, except for *Enneapogon* spp., which had a consistently higher yield (*P* < 0.01) in burnt plots from 1999 onwards (Fig. 3*d*). Native legume biomass, always low and averaging 33 kg ha⁻¹, was generally decreased after spring fires, but only significantly decreased in autumn 2000 (*P* < 0.01) after a dry late summer.

When abundance was measured as frequency, the outcome of burning was similar but the numerical variability was much greater. The extreme example was the native legumes, which had a high mean frequency of 47% (Table 3), compared with negligible biomass that averaged <1% of the total pasture. Their frequency also varied greatly over years, with an initial level of 60–70% declining to 10% in 1998 before increasing again, but to a far greater extent (*P* < 0.01) in the unburnt plots (Fig. 4*c*). Spring fires significantly increased the frequency of *Enneapogon* spp., from ~15% initially to over 50% in 1998 after several fires, but their frequency in unburnt plots rose only to 30% (Fig. 4*b*). Thereafter, their frequency dropped appreciably but the significant treatment difference persisted (Table 3).

Over all years, burning significantly (*P* < 0.05) reduced the seed numbers potentially germinating the next spring (Fig. 6*a*), driven by a significant effect (*P* < 0.05) on *Wahlenbergia* spp. (bluebells) (Table 4). Sedges (Cyperaceae) germinated in large numbers (Table 4), despite providing <1% of pasture biomass.

The clear effect of burning on *Enneapogon* spp. biomass and frequency was not evident in the numbers of its fluffy seeds germinating (Table 4), which in later years generally decreased. No germination by seeds of any woody species was recorded.

Poplar box site

Autumn biomass yields of minor species varied greatly over the 8 years (Fig. 2). Growth of the short-lived perennial forbs *Calotis* spp. (daisy-burrs) and *Verbena tenuisecta* (Mayne's pest), which flourished for 1–2 years during abnormally wet winters and springs, was reduced by spring burning (Fig. 2*f, g*), as was their frequency (Fig. 5*c, d*). As in the ironbark site results, the frequency of *Enneapogon* spp. was differentially affected by the fires after 1997, but the effect, though consistent, was not statistically significant (Fig. 5*g*). Spring fires also resulted in a consistent but non-significantly greater frequency of occurrence of *Chloris divaricata* (slender chloris) (Fig. 5*a*).

Though not evident in their biomass, the frequency of occurrence of some minor species was significantly affected by regular spring burning. Burning increased the frequency of *Brunoniella australis* (blue trumpet) and unpalatable (poisonous) legumes, mainly *Indigofera* spp. (Table 5). Conversely, spring burns significantly suppressed the frequency of *Cymbopogon* spp. (Fig. 5*b*) and *Panicum effusum* (hairy panic) (Table 5) by 2002.

Spring burning significantly reduced the total germinable soil seedloads in most years (Fig. 6*b*), but not significantly for any single species. Reductions were most apparent for *Centaurium* spp. (69 versus 15 total germinants over all years), *Wahlenbergia* spp. (17 versus 1) and *Calotis lappulacea* (yellow daisy-burr,
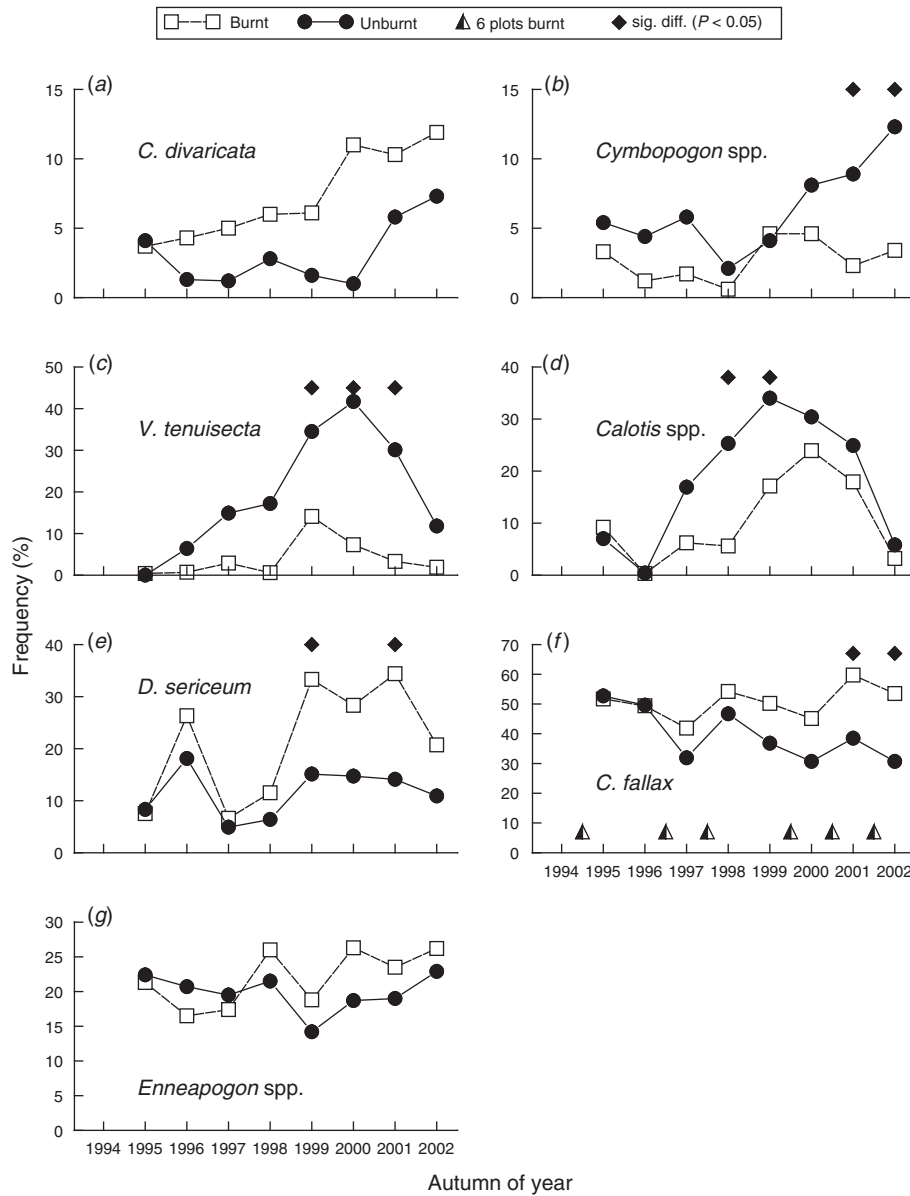
Fig. 5. Variation in autumn frequency (%) over time for (*a*) *Chloris divaricata*, (*b*) *Cymbopogon* spp., (*c*) *Verbena tenuisecta*, (*d*) *Calotis* spp., (*e*) *Dichanthium sericeum*, (*f*) *Chrysopogon fallax* and (*g*) *Enneapogon* spp. at the poplar box site. A '◆' indicates significant differences within years ($P < 0.05$); a ▲ indicates when designated plots were burnt.

8 versus 0). Only three tree seedlings emerged, each in a different year, two from unburnt plots and two from beneath poplar box trees that often set fruit.

Trends in surface soil protection

Ground cover and pasture basal area

Ironbark site. Exclusion of grazing animals eventually produced the expected increase in autumn ground cover, but it happened slowly (Fig. 7*a*). After 1996, cover rose to 80–90% where unburnt in response to better summer rainfall. The difference between regimes depended on whether a burn was achieved the previous spring, but even after a spring burn, ground

cover levels after 1996 returned to ~60% or more by the ensuing autumn.

At this site, the pasture basal area in winter, mostly provided by perennial grasses, was ~2% in unburnt pastures and was not significantly affected by spring burning until the year 2000 (Fig. 8*a*). Then the burnt plots had a greater pasture basal area because *H. contortus* plants had increased in density and tussock size over the past year (data not presented; see Silcock *et al.* 2005), whereas that of dense, undefoliated plots remained unchanged.

Poplar box site. The first three spring burns did not produce any marked relative change in ground cover next autumn, but thereafter spring burning resulted in significantly lower cover
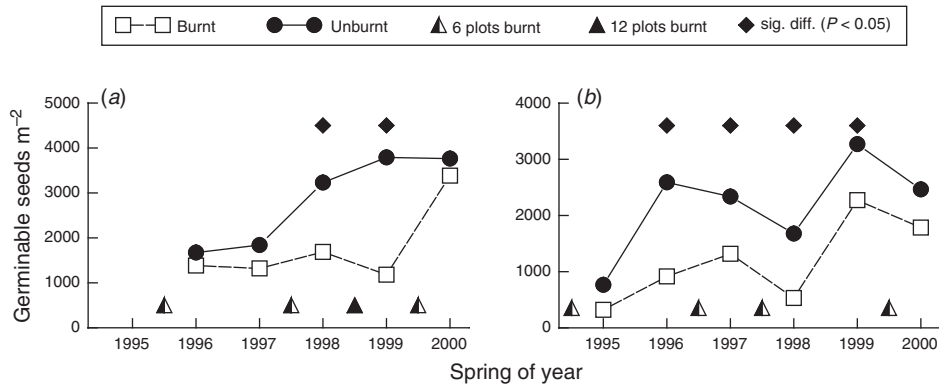
Fig. 6.   The impact of regular spring burns on total germinating seeds (no. m^{-2}) from spring soil samples of ungrazed (*a*) ironbark and (*b*) poplar box pasture. A '◆' indicates significant differences within years (*P* < 0.05); a △ indicates when designated plots were burnt and ▲ when all plots were burnt in 1998 at the ironbark site.
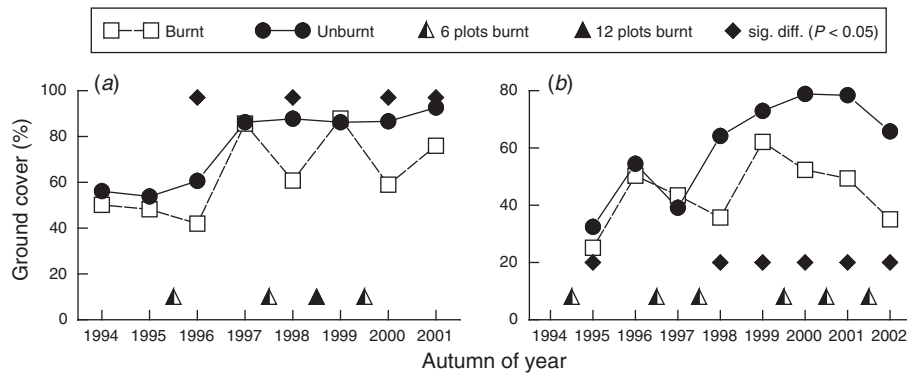

Fig. 7.   Changes in total autumn ground cover (%) in ungrazed (*a*) ironbark pastures, and (*b*) poplar box pastures, with or without repeated spring burns. A '◆' indicates significant differences within years (*P* < 0.05); a △ indicates when designated plots were burnt and ▲ when all plots were burnt in 1998 at the ironbark site.
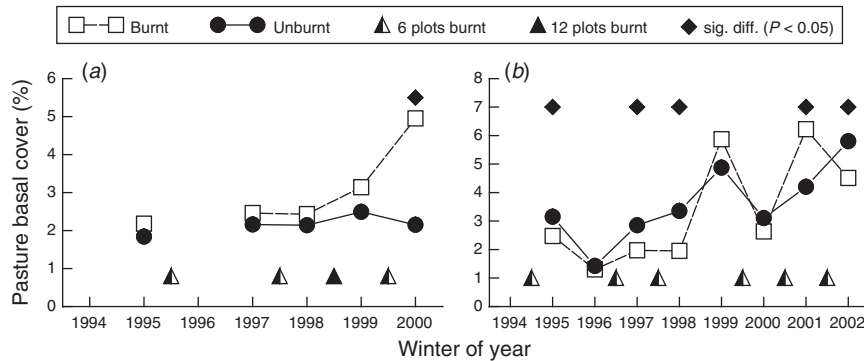

Fig. 8.   Changes in pasture basal cover (%) over time due to spring burning of ungrazed pastures at (*a*) ironbark site, and (*b*) poplar box site. A '◆' indicates significant differences within years (*P* < 0.05); a △ indicates when designated plots were burnt and ▲ when all plots were burnt in 1998 at the ironbark site.

after the ensuing summer (Fig. 7*b*). Cover levels were low initially after the prior drought, but increased to at least 40% for the next 2 years. It then improved markedly in unburnt plots as good rains occurred, yet plots burnt in spring ended most summers with significantly less cover (35–62%) than the unburnt

plots (65–80%) in later years. Dry years in 2000 and 2001 saw cover decline generally, but this was most marked in burnt plots, where it again fell to below 40% in autumn 2002 (Fig. 7*b*).

Pasture basal area at this site showed an increasing trend, irrespective of spring burning, but also varied between years

and burning treatment, often significantly, under the combination of growing conditions during the recent summer and whether a burn occurred the previous spring (Fig. 8*b*). The fires had no consistent significant impact on pasture basal area.

Cumulative fire effects on landscape functionality indices

Ironbark site

Four spring fires in 7 years produced negligible differences in the LFA stability and infiltration indices (Table 6), but there was a significant reduction in the nutrient cycling index. Overall, the indices were consistently better where burning was not regularly practised. Big differences between treatments were recorded in the downhill length of 'fertile' patches and interpatch zones. This translated into a significantly poorer Landscape Organisation Index (L.O.I.) of 0.45 for the regularly burnt plots, versus 0.89 where mostly unburnt.

Poplar box site

Six spring burns in 8 years resulted in greater amounts of relatively bare runoff or interpatch zones, and the size of individual zones was more variable in the burnt plots. All indices were significantly greater in the unburnt treatment (Table 6), and all indices other than the L.O.I. differed to a greater degree due to burning than occurred at the ironbark site. However, the L.O.I. for the unburnt plots at both sites was similar, at 0.86 and 0.89.

Discussion

Overall in the burnt plots, 1500–2000 kg ha^{-1} of pasture dry matter was often sacrificed for a small, often temporary effect on pasture composition, a transient reduction in litter and ground cover, and a small but cumulative effect on tree and shrub competition (Jones *et al.* 2018). Much of the pasture loss would have been old, dead leaf and stem, which would not be attractive to cattle, but as litter it contributes to improved landscape functioning. The pasture species most affected by regular spring fires were from diverse plant types – legumes (mostly perennial), short-lived, low biomass species (*Enneapogon* spp., *P. effusum*, *V. tenuisecta*), erect perennials (*Cymbopogon* spp., *Eulalia aurea*, *B. bladhii*, *C. ciliaris*), and *C. divaricata* (low biomass, low flammability, semi-prostrate) (Tables 3 and 5). Our results accord with those of Snyman (2015) in South Africa, who

recorded significant temporary reductions in available forage but few marked shifts in the composition of burnt grassy pastures that contained small amounts of forbs and legumes.

No noticeable increase in the abundance of annual grasses occurred as a result of spring fires, unlike the increase recorded by Cowley *et al.* (2014) and Lebbink *et al.* (2018). Any early changes were not magnified dramatically as new burns took place, e.g. *Enneapogon* spp. (Figs 3*d*, 5*g*) and *C. divaricata* (Fig. 5*a*).

Hypothesis 1. Spring fires can reduce the proportion of *Aristida* spp. and other low palatability grasses

The reduction in abundance of unpalatable grasses from regular spring fires was not as great as expected, but the frequency of *A. calycina* was significantly reduced by spring fires at the poplar box site. The increase in *Aristida* spp. biomass at the poplar box site in the absence of spring fires (Fig. 2*b*) matches the findings of Orr *et al.* (1991) in the black speargrass zone and Cowley *et al.* (2014) in the tropical grasslands of northern Australia. It also illustrates why graziers in the Maranoa region habitually use spring fires to remove coarse unpalatable wiregrasses in places where they can be dense, such as *A. ramosa* on flats. The greater impact of fires on *A. calycina* seems related to the higher, more exposed crown of this species compared with that of *A. ramosa*, which can be several centimetres below ground. Our relatively low soil seed numbers for *Aristida* spp. are comparable with the results of Orr *et al.* (2004) in more coastal pastures, and fires did not significantly alter the size of *Aristida* seed reserves at either site.

The biomass of other low-palatability grasses such as *Enteropogon ramosus*, *A. ramosa*, *B. decipiens* and *Cymbopogon* spp. was often temporarily reduced, in line with the general short-term reduction produced by burning. Thus, the results of this study do not support Hypothesis 1 well.

Hypothesis 2. Spring fires do not adversely affect the important 3P (productive, palatable, perennial) grasses in the pasture

Desirable 3P grasses did not generally have their presence reduced by fires in the absence of grazing, as was also recorded by Phelps (2006) and Lebbink *et al.* (2018). Some, such as *B. bladhii*, increased their frequency (from 4.2% to 7%, Table 5) and biomass (Fig. 2*h*) at the poplar box site. The data are inconclusive as to the cause, but it has a low, sturdy crown and is regarded as tolerant of burning (Orr *et al.* 1999). However, when grazing occurs shortly after fires, some studies report a small decline in abundance of some palatable perennial grasses (Cowley *et al.* 2014; Orr *et al.* 2010). There were indications that *B. ewartiana* was adversely affected by regular spring burning (Fig. 3*a*) and that it replaced lost biomass more slowly than other 3P grasses during the summer after a spring burn. Significant enhancement of *H. contortus* abundance did not occur, as was predicted from studies in the Burnett region (Orr and Paton 1997), either by the biomass measure (Fig. 3*b*) or in its frequency (Tables 3 and 5). Germinable seed of *H. contortus* and *B. ewartiana* was always present in spring to facilitate recruitment at the ironbark site (Table 4), but never for *T. triandra*.

Table 6. Differences in Landscape Function Analysis indices due to contrasting spring burning regimes, after 7 years at the ironbark site and 8 years at the poplar box site
Data combines treed and herbicide-treated plots. *$P<0.05$; **$P<0.01$; ***$P<0.001$; n.s., not significant

Index	Ironbark site			Poplar box site		
	Spring burns	Unburnt	Signif. diff.	Spring burns	Unburnt	Signif. diff.
Stability	67.7	72.1	n.s.	60.1	69.3	***
Infiltration	51.1	59.4	n.s.	28.2	39.2	***
Nutrient cycling	23.1	28.7	*	20.6	31.9	***
Landscape organisation	0.45	0.89	**	0.71	0.86	**

The complete absence of germination by *T. triandra*, which was abundant at the ironbark site, is unusual and contrasts with its regular emergence from the soil seedbank in South Africa (Snyman 2005) and the finding of Mott and Andrew (1985) that viable seed persists through the dry season in northern Australia, even if lost during the next wet season. A lack of flowering by the dominant grasses in the high biomass unburnt plots was noted in later years of the trial, particularly for *T. triandra*. The cause may be nutrient stress or soil moisture deficit at flowering time, because flowering of many tropical grasses responds strongly to added nitrogen (Cameron and Mullaly 1969; FutureBeef 2018). Lack of defoliation suppresses flowering of *Astrebla* spp. (Orr and Evenson 1991). The main grasses could be affected differentially because they have different flowering rhythms in Central Queensland, perhaps differing nutrient requirements for floral development, or a differing need for periodic defoliation. *Themeda* flowers in late spring (Scattini pers. comm.) and *H. contortus* in late summer (Campbell 1996), whereas *B. ewartiana* flowers in all warmer months in response to soil moisture. Hence different limiting factors at different times during the warmer months will influence the relative degree of flowering and seed set and could be the basis for the contrasting soil seedloads recorded for these three grasses. Currently, we lack sufficient information to reach any firm conclusion about the reason for this contrast.

Nonetheless, Hypothesis 2 was generally supported for these ungrazed pastures.

Hypothesis 3. Regular spring fires after rain do not, in the absence of grazing, damage landscape functioning

Landscape functionality (assessed at the end of the experiments) found that, in the absence of grazing, regular spring fires after rain affected the LFA indices differently between sites. The L.O.I. at both sites was significantly poorer ($P < 0.01$) after years of regular moderate spring fires. Significantly poorer results ($P < 0.001$) for all 4 LFA indices were recorded at the poplar box site (Table 6), but at the ironbark site the nutrient cycling index was the only other index significantly poorer ($P < 0.05$) due to spring fires. It is possible that the vegetation community may influence the sensitivity to change of different components of the LFA indices, as Holm *et al*. (2002) found in Western Australia.

Lack of an initial assessment of landscape functionality was unfortunate, but given the length of time over which the differing treatments were imposed and the level of significant difference recorded from a randomised block experiment with tree cover interaction, we think that these differences were the result of the fires and not pre-trial site variability. Jones *et al*. (2018) conducted a similar analysis of the tree effect under grazed and ungrazed conditions and, contrary to our fire effects, found no significant impact on landscape functionality after many years.

Serious negative consequences can result from inappropriate removal of mature forests (McIntyre *et al*. 2002; Williams and Saunders 2005). Our study was conducted on gently sloping land and done without soil disturbance, yet there was still evidence of significant damage to landscape functioning from fairly regular spring fires. Our burns were deliberately carried out after a significant fall of rain, whereby the surviving herbage and trees could renew growth immediately and thus return soil cover quickly. Also, there was no grazing of reshooting palatable plants, which is detrimental to their persistence (Orr and Paton 1997; Cowley *et al*. 2014), particularly the strongly perennial grasses that are pivotal to erosion control.

Ground cover was not always adequate

Ground cover after the summer growing season at the ironbark site was mostly above the 50% identified by Silburn *et al*. (2011) as critical for minimising erosion of similar land types. Regular spring fires reduced cover below that crucial level at the poplar box site, and was not a desirable outcome. However, even after a spring burn, cover levels in wetter years after 1996 returned to acceptable levels of ~50% by the end of the next summer. Thus, a good rainy summer after a spring burn will best minimise detrimental effects on ground cover.

Pasture basal area was not consistently affected by regular spring fires

Crown basal area values of just over 2% at the ironbark site in the early years of the experiment were low by Queensland standards for an A⁻ condition pasture (Orr *et al*. 1991; Silcock 1993) and reflect the dry years that preceded this study. The big increase in basal area in the burnt ironbark pastures in later years was due to recent recruitment of perennial grass seedlings after 1998 (Silcock *et al*. 2005) and may not have been sustainable in the long term. Conversely, the unburnt plots had rank foliage that did not favour seedling recruitment or crown expansion.

In contrast, the crown basal area of 5% or more in the later years is good for poplar box native pastures in its region (Silcock 1993). Initial values of 2–3% are indicative of a pasture in only fair condition. Prolonged removal of grazing contributed to the increasing basal area over time, regardless of fires, with variability between years possibly being due to difficulty in defining a 'strike' on a living crown with partially dead areas in a rank pasture (Lodge *et al*. 1981).

Hypothesis 3 was not generally supported by our data.

Notable impacts of spring burns on other herbaceous species

Regular spring burns had an obvious effect on some species that, while relatively minor in their individual biomass, are important to pasture production, animal nutrition and land condition. Native legumes had their biomass more than halved (20 vs 50 kg ha⁻¹) and frequency reduced by burning at the ironbark site (Fig. 4*c*, Table 3), which is undesirable for the pasture's nitrogen economy. However, it is somewhat desirable in the case of poisonous legumes such as *Indigofera* and *Crotalaria* species, which poison livestock if eaten in large amounts (Fletcher *et al*. 2015). In contrast, at the poplar box site, the frequency of poisonous legumes was significantly increased by regular burning, whereas that of palatable, desirable legumes was unchanged (Table 5).

The effect of the reduced legume biomass in this ecosystem cannot be readily quantified because, though nitrogen fixation is proportional to biomass growth within a species under the same conditions (Vallis and Gardener 1985), rates of fixation by Australian native legumes are generally variable and low

(Lawrie 1981). The latter seems due to low soil fertility and limited effective growing periods, particularly as root nodule formation and persistence is very sensitive to moisture stress (Silcock 1998). The flow-on effects into carbon : nitrogen ratios in soil organic matter, nitrogen volatilisation under hot, wet conditions and loss during fires can only be speculative (Peoples *et al*. 1995). Suffice it to assume that a reduced legume biomass after a fire could slightly affect the ecosystem nitrogen economy, probably negatively in the short-term, independent of the direct effect of the fire itself.

Regular burning increased the standing biomass and frequency of *Enneapogon* spp. at the ironbark site (Fig. 4*b*, Table 3), but only their frequency at the poplar box site. This could be due to a difference in the suite of species at each site, but the high-level skills required to differentiate within this genus in the field meant recordings were only made to genus level. These plants are reasonably palatable to stock, but they are only short-term perennials and cannot be relied upon for landscape stability or forage during prolonged droughts. Cowley *et al*. (2014) reported that frequent fires at a red earth woodland site encouraged *E. polyphyllus* to proliferate, and that species was a common component of the *Enneapogon* suite at our ironbark site. The biomass differences were not related to any difference in germinating seed numbers (Table 4) and thus were most likely related to enhanced seedling recruitment and reduced competition on small *Enneapogon* plants from existing perennial grass tussocks in a more open sward in early summer after fires.

At the poplar box site, regular spring fires doubled the frequency of *C. divaricata*, which has mixed pastoral values. The species is weakly perennial and has low biomass, but is highly palatable, weakly sward-forming and grows actively in spring. It is common in heavily grazed pasture and thus provides important ground cover to vulnerable parts of the landscape in spring, when cover is usually at its lowest. Further research is required to determine whether the same result occurs in a grazed pasture after burning. Orr and Paton (1997) found its frequency declined under repeated burning in grazed pasture – another example of how responses to fire can depend on pasture type (*sensu* Cowley *et al*. 2014) and be confounded by grazing pressure.

It is speculated that the conditions before, during and shortly after a burn can produce particular outcomes that are episodic and difficult to predict with our current knowledge of the biology of most rangeland species. Our data show sudden marked changes linked to the fire regime, at different times, in the relative frequency of several species during the trial that were not repeated or markedly reinforced by later spring burns, such as by *Enneapogon* spp. (Figs 3*d* and 5*g*), *Cymbopogon* spp. (Fig. 2*d*), native legumes (Fig. 4*c*) and *Chloris divaricata* (Fig. 5*a*). Our ancillary charting data did not involve these minor species, so it cannot be determined whether the relative change in frequency was due to death of existing plants or recruitment of new ones or both. Germinable soil seedloads were not significantly different due to spring fires for these species individually, but germination is but one stage in the recruitment process.

Spring germinable seed population

Burning regularly in spring reduced the total germinable seedbanks significantly in many years (Fig. 6), but few individual grass species or guilds of related species had seedloads strongly advantaged or disadvantaged by spring fires after rain, with the fewer *Wahlenbergia* spp. at the ironbark site the exception (Table 4). The short viability lifespan of eucalypt seeds in the field may have contributed to the negligible numbers of their seeds germinating at both sites in the spring. Because our sampling was done almost a year after each burn, our differences may be conservative, because seeds of hard-seeded species should be more germinable immediately following the fires (Keeley and Fotheringham 2000). The low sampling intensity used may also have hindered detection of significant fire effects when combined with the natural huge spatial variability commonly evident in rangeland seed distribution. However, for pasture species, the germinable seed density recorded at the nearby grazed poplar box site, which was more intensively sampled, averaged over grazing pressure, was 20% lower over the last 2 years than at the ungrazed burning trial site (Silcock *et al*. 2005). Thus spring fires do not seem to be a serious threat to recruitment from seed of native rangeland plants, and benefit some forbs and annual species (Williams *et al*. 2003*a*; Cowley *et al*. 2014).

Would the results have been different if the pastures had been grazed after the fires?

Perhaps yes – the answer would depend on how soon animals grazed the burnt pastures afterwards, how heavy that grazing pressure was, how favourable the weather conditions were in the growing season following the burn, and many other potential factors. Research specifically to study the interaction of prescribed fire and grazing is rare (Kutt and Woinarski 2007; Vermeire *et al*. 2014), and even then the experimental design is easily upset by unintended wildfires or change in animal presence (Hill and French 2004; Werner 2005). Empirical evidence suggests that, if good pasture composition is to be maintained, rest from grazing after a burn should be as long as possible, preferably a whole growing season (Snyman 2003), but at least until pasture regrowth of desirable forage species is appreciable. That said, heavy grazing pressure on a particular plant as it resprouts may be the intended objective, e.g. on a fast-maturing, stalky grass or a re-sprouter shrub. In such cases, immediate heavy grazing may be employed for a short time, but then prolonged rest afterwards would be mandatory. Our objective was to identify herbaceous species whose population dynamics showed great sensitivity to fire itself in order to understand what role prescribed burning may have in their management. This we have done.

Management principles arising

Post-burning management is more critical than the burning strategy (O'Reagain *et al*. 2009). If a burn is patchy, the green shoot on burnt areas will be patch-grazed down to the crown after each small fall of rain until autumn. Detrimental outcomes of burning have been demonstrated by Orr and Paton (1997) due to excessive grazing on recruiting 3P grass seedlings. Trollope (1989, 2004) has denounced regular burning to induce a 'green pick' as 'an invalid use of fire'. However, in extensive rangelands patchy annual burns in widely separated locations can shift

grazing pressure away from last year's burnt, albeit more heavily grazed, places (Andrew 1986).

If a large woody species seed set is observed on grazing country in summer, a burn next spring should be considered and, if feasible, be left as late as possible to enhance the killing of any major recruitment that may follow. Deliberate spelling to accumulate fuel for a hotter fire should cause a higher mortality rate among tree and shrub seedlings (Fensham and Fairfax 2006) and minimal damage to the perennial pasture grasses. A delayed burn, after rain, would also minimise the period during which bare ground existed for recruitment from germinable seeds. However, this logic would not always work reliably for trees and shrubs that have a persistent seedbank, such as many *Acacia* spp. (Hodgkinson and Harrington 1985). In those cases, a burn may stimulate mass recruitment that would require herbicide use (or a follow-up burn within a few years for fire-susceptible species) before those seedlings reached flowering stage. If they are not very fire-susceptible then a species-specific solution devised in consultation with experts would be needed. Our sites had no major shrub understorey, but other eucalypt woodlands in the region do, such as those with *Carissa* spp. (currant bush), which is not killed by burning (O'Reagain and Bushell 2011). Our fires did not produce a mass germination of seed of any woody species, but that is not universally the case (Thomas *et al.* 2003).

Unfortunately, the risk of missing a planned burn exists every spring, and the nature of the seedbank of major pasture species is also a complex issue. Thus managers of *Aristida/Bothriochloa* land types will do best by managing the existing pasture to achieve desired outcomes and reserve prescribed fire for managing woody plants. Grazing management following a fire is a subject in need of more research globally (Snyman 2004; Hunt *et al.* 2014).

Conclusion

Regular spring burns after rain in *Aristida/Bothriochloa* communities in central Queensland in the absence of grazing did not adversely affect the abundance of important perennial grasses and increased that of *Enneapogon* spp. in the pasture. The fires increased the abundance of *Bothriochloa bladhii* and *Dichanthium sericeum* at the poplar box site, and reduced the proportion of *Aristida calycina*, but had no big effect on the abundance of other *Aristida* species, nor on that of other low-palatability grasses.

Repeated spring fires lowered the average ground cover as well as some indices of landscape stability and ecosystem functionality. These changes were not always consistent across the two communities. Less frequent fire targeted to years that will be most effective for woody plant management will minimise potential landscape ill-effects.

The effect of spring fires on our pastures may be affected by the grazing management that precedes and follows them, but the degree to which any effect would alter our scientific conclusions is speculative, particularly in the absence of specific details about the grazing management envisaged.

We recommend that strategic spring burning be used in these woodland communities to address specific issues, but that annual fires should not be practised on the same piece of land.

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