

Open woodland tree and shrub dynamics and landscape function in central Queensland after killing the trees with herbicide

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Abstract. Herbicides are used in savanna to control tree and shrub density, primarily to maintain the value of the country for pastoral enterprises. However, the concomitant effects on biodiversity and landscape functioning need to be recognised and better understood. This study monitored tree and shrub dynamics and eventual landscape functionality in response to tree-killing over 7–8 years at two open eucalypt woodland sites in central Queensland. Paddocks denuded of trees using herbicide or not so treated were subject to three differing grazing pressures by cattle. Similarly treated but ungrazed sets of plots were subjected to either regular spring burns or were rarely burnt.

Tree and shrub growth and seedling recruitment were slightly affected by grazing pressure but regular spring burns minimised recruitment of minor woodland species and reduced the population of original saplings and seedlings that survived the herbicide. Few eucalypt seedlings emerged from soil surface samples taken each spring in any treatment, despite the presence of flowering trees in half the treatments. Capture and retention of resources, particularly rainfall and nutrients, were slightly improved by killing the trees, and worsened by grazing. We conclude that killing trees with herbicide at these sites did not adversely affect landscape function and that woody species regeneration was almost inevitable on these open eucalypt woodland native pastures.

Additional keywords: ground cover, LFA, seedlings, TRAPS.

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Introduction

Clearing or thinning of trees is used globally to increase agricultural production albeit with mixed results (Archer *et al.* 2017). In Australia, eucalypt (*Eucalyptus* L'Hér. and *Corymbia* K.D.Hill and L.A.S.Johnson species) forests and woodlands, and *Acacia* Mill. shrublands are still being cleared at steady rates (SLATS 2000, 2017). In many drier parts of northern Australia, pastoralism has been the main agricultural industry. In these areas there was initially no significant attempt to thin woodland trees, which were seen as beneficial for shade. Native pastures there were regarded to be as productive as most exotic pastures tested over the long term (Silcock 1983), and tree retention, with some loss of pasture production (Scanlan 2002), was economically logical.

In the past 50 years increases in woody cover in some grazed woodlands has reduced the forage available to livestock (Burrows 2002) and possibly reduced wildlife biodiversity in an increasingly dense, low-diversity understorey (Verdú *et al.* 2000; Askins 2001; Archer *et al.* 2017). The reported increase in tree and shrub density has been ascribed to a lack of fires due to low fuel loads because of heavy forage utilisation by livestock

and sometimes by feral goats and marsupials (Tunstall *et al.* 1981; Witt *et al.* 2009; Witt 2013). However, woody thickening and shrub encroachment is a global phenomenon and may be driven by increasing carbon dioxide (Archer 1995; Archer *et al.* 2017). Fluctuations in rainfall with climate cycles have been implicated in woody thickening in the mulga lands following above-average rainfall periods in south-west Queensland and thinning of woodlands after extreme droughts in northern Queensland (Fensham *et al.* 2009).

Trees have important ecological roles in reducing erosion and improving downstream water quality as the litter increases infiltration rates (Roth 2004) and roots help reduce streambank erosion (Anon. 2006). Grazed savanna woodlands contain large stocks of carbon in vegetation that provide a potential sink, especially through storing carbon in the regrowth areas following clearing, so there is potential for producers to earn additional income as well as reduce atmospheric greenhouse gases by letting cleared areas regrow (Whish *et al.* 2016). Trees also extract water from significant depth in the regolith (Thorburn and Walker 1993; Silburn *et al.* 2011) and there are concerns over this extraction ability and the intake of water

into the important Great Artesian Basin aquifers (Williams and Coventry 1980). Conversely, dryland salinity in Australia is sometimes due to tree clearing (Land and Water Audit 2000) including in north Queensland (Bui 1997; Williams *et al.* 1997). Thus major changes in tree coverage of landscapes can potentially induce appreciable changes in landscape functioning that need to be understood at a local level.

Major thickening of mulga (*Acacia aneura* F.Muell. ex Benth.) shrublands (Booth and Barker 1981; Noble 1997) has been documented although Queensland woodlands have not encroached appreciably into open grassy pastoral lands over the past century due to forest clearance (Fensham *et al.* 2011). Hall (pers. obs.) recently documented a marked visual increase in tree and shrub density in western and northern Queensland rangelands since the 1970s. Selective thinning to control woody thickening has been shown to be uneconomical (Burrows *et al.* 1999) with a short-term benefit (Burrows *et al.* 1988; Scanlan 2002) and is rarely practised on a commercial scale by livestock industries in Queensland. The majority of timber clearing research from Queensland prior to our studies dealt with *Acacia* woodlands – gidyea (*Acacia cambagei* R.T.Baker), brigalow (*A. harpophylla* F.Muell. ex Benth.) and mulga (*A. aneura*) that have very different biology and recruitment methods from eucalypts (Johnson 1964; Purcell 1964; Coaldrake *et al.* 1969; Beale 1973). Tree and shrub removal studies on southern Queensland eucalypt woodlands have demonstrated a marked increase in pasture growth in the absence of grazing (Walker *et al.* 1972, 1986; Hall *et al.* 2016). Pastoral productivity of eucalypt woodlands in north Queensland increased significantly after tree clearing with larger increases and more exotic grasses on the more fertile soil type (McIvor and Gardener 1995; Jackson and Ash 1998). However, pasture quality was less after tree clearing and may offset the increased pasture productivity and have longer-term implications for soil nutrient dynamics. In central Queensland, major concerns often centre on the proliferation of understorey shrubs such as native currant bushes (*Carissa* L.), false sandalwood (*Eremophila mitchellii* Benth.), hopbushes (*Dodonaea* Mill.) and wattles (*Acacia* Mill.), in addition to the regrowth and thickening of the dominant eucalypts (Burrows *et al.* 1988; Back 2005). Our studies focussed on central Queensland eucalypt woodlands within the *Aristida/Bothriochloa* native pasture land type (Weston *et al.* 1981), which cover greater than 10 million hectares.

Two experimental sites were established in grazing woodlands of central Queensland to provide detailed information about the impact of the killing of trees by herbicide on the landscape and the vegetation. The effect of tree-killing and grazing pressure on pasture growth and composition is reported in Hall *et al.* (2016) whereas this paper reports on tree and shrub seedling recruitment and growth, ground cover changes and surface soil condition after 7 years in both grazed paddocks and ungrazed plots. The interaction of regular spring fires on ungrazed areas after killing the trees and the impact of widely differing grazing pressure on changes to woody vegetation and landscape function are also reported. Well-researched data and results were needed and these three hypotheses were tested:

- (1) Spring fires will decrease tree and shrub density and basal area;
- (2) Grazing pressure will not directly affect woody plant density and basal area; and
- (3) Spring fire and/or tree-killing will not adversely affect soil surface condition.

Methods

Two sites were established to assess the impact of tree-killing on subsequent tree and shrub regrowth and landscape functioning. Two experiments at each site further examined the interactions of tree-killing with grazing pressure (Grazing experiment) and burning (Burning experiment). Experimental techniques were described in detail in Hall *et al.* (2016). The northern 'Ironbark' site on a gritty red chromosol near Rubyvale was an open, silver-leaved ironbark (*Eucalyptus melanophloia* F.Muell.) woodland with an understorey pasture dominated by the perennial grasses *Bothriochloa ewartiana* (Domin) C.E.Hubb. (desert bluegrass), *Heteropogon contortus* (L.) P.Beauv. ex Roem. and Schult. (black speargrass), *Themeda triandra* Forssk. (kangaroo grass) and *Chrysopogon fallax* S.T.Blake (golden-beard grass). The southern 'Poplar box' site near Injune was a regrowing open poplar box (*E. populnea* F.Muell.) woodland on a mix of silty sodosols and dermosols (Isbell 1996) with the perennial understorey pasture dominated by *Aristida* L. (wiregrasses), *Bothriochloa decipiens* (Hack.) C.E.Hubb (pitted bluegrass) and *Dichanthium sericeum* (R.Br.) A.Camus (Queensland bluegrass). The trees were axe-thinned more than 30 years before the trial began and the regrowth comprised mainly large poplar box trees. Botanical names in this paper accord with the Australian National Plant Index web site (ANPI 2016). Herbicide was applied via stem injection by commercial operators in March 1994 to the ironbark site (hexazinone) and July 1994 to the poplar box site (picloram plus 2,4-D). Because myall (*Acacia pendula* Sweet) is tolerant of picloram, the few scattered large trees in some paddocks at the poplar box site were ringbarked to kill them. Grazing commenced by weaner Brahman-cross steers in November 1994 at both sites.

Interacting with tree-killing at both sites were grazing pressure treatments and a spring burning regime on ungrazed areas. The grazing experiment examined heavy, medium or low grazing pressure (75%, 50% and 25% utilisation) by resetting the number of steers each year in response to pasture yield in autumn. Trees-killed and treed treatments were applied across 12 paddocks with two replicates of each treatment. The burning experiment had spring burns that were conducted in most years when atmospheric conditions permitted. Trees-killed and treed treatments were applied across 12 plots with and without burning and replicated three times. One replicate of the treed treatment poplar box woodland with high grazing pressure had an abnormally high tree density, which was reduced by selective stem-injected herbicide application in early 1996 from a basal area of 14.1 m² ha⁻¹ to 9.1 m² ha⁻¹. A range of tree, pasture and soil surface measurements were taken at both sites (Silcock *et al.* 1996).

Tree dynamics

Tree basal area was assessed using the Bitterlich technique (Grosenbraugh 1952) at 30 cm above ground level prior to herbicide application for both grazing and burning experiments. A recording was taken at every point on a 50 × 50-m grid in every paddock. Following herbicide application tree and shrub growth was recorded about every 2 years from 1995, using the TRAPS technique (Back *et al.* 1997): a 4-m-wide, fixed transect along which every woody plant was identified, its location recorded, its stem diameter or circumference at 30 cm above ground and height measured, and canopy cover extent estimated. Three fixed transects each of 150-m length were recorded in every grazed paddock (3.5–21 ha). In each of the 12 1-ha burning experiment plots two fixed transects of 100-m length were recorded. Recently killed trees were identified and their basal area calculated in the 1995 recordings as well as that of the live woody plants.

Shrubs that were less than 30 cm tall had the stem diameter measured at around half their height. However, their contribution to total basal area was insignificant. TRAPS recordings were conducted three times at the ironbark site in July 1995, October 1997 and September 1999, and five times at the poplar box site in March 1995, September 1996, September 1997, August 1999 and April 2002. At the burnt plots, the degree of fire scorch was categorised using the methodology of Back *et al.* (1997). Basal area was calculated from the stem diameter of small trees and shrubs and the stem circumference measurement of larger trees.

Soil surface cover

Projected ground cover by all organic (non-rock) materials was estimated during the Botanal pasture recordings each autumn (see Hall *et al.* 2016 for details) and included live and dead plant stems, litter, standing pasture and dung.

Landscape function analysis

At the conclusion of the research in 2001, the condition of the soil surface was assessed by the Landscape Function Analysis (LFA) method of Tongway and Hindley (1995) along three 50-m transects in each of the 12 grazed paddocks and on one 50-m transect in each of the 12 ungrazed burning experiment plots. The land surface was categorised into various cover types and an index computed for the stability, infiltration and nutrient retention potential of that area of land. A lack of slope across the

1-ha plots, as required by the LFA theory, meant that only four plots were assessed at the ironbark ungrazed site.

Germinable soil seed loads

The soil was sampled for potentially germinable tree and shrub seeds in spring each year. Four 5-cm-diameter cores to 5-cm depth were taken, and combined, at the centre of each burning experiment plot (48 samples, 0.11-m² sampled area) and at three locations on each TRAPS transect in each of the 12 grazed paddocks (144 samples, 0.33-m² sampled area). Early the following summer these were spread out 1-cm deep on the surface of pots of sand and watered persistently for several weeks in a glasshouse or shadehouse. Emerging seedlings were counted and identified to either a species or genus level. It was not feasible to grow tree and shrub seedlings out to reproductive maturity to confirm the identity of some species. Nonetheless, seedlings of *Eucalyptus* species could be distinguished from their close *Corymbia* relatives and the limited range of other woody species present permitted most to be identified with confidence by the time they were 20–30-cm tall.

Statistical analysis

Statistical analysis of the results was undertaken using the GENSTAT package (GENSTAT 2015). Most was done via analysis of variance (ANOVA) with tree-killing as the first order factor and grazing pressure or spring burning as secondary factors. The grazed paddocks were analysed independently of the ungrazed burning trial plots. The time-series nature of the data was taken into account by an analysis of variance of repeated measures (Rowell and Walters 1976), via the AREPMEASURES procedure of GENSTAT. This forms an approximate split-plot ANOVA (split for time). Greenhouse-Geisser epsilon estimates of the degree of temporal autocorrelation were used to adjust the probability levels returned. Significant differences were determined by the l.s.d. process with $P < 0.05$ taken throughout to indicate a significant difference.

Results

Seasons following herbicide application

Details of the seasonal and annual site rainfall throughout the experiments are given in Table 1. Both sites have a similar historical mean annual rainfall of ~640 mm, but the amount of winter rain was greater at the more southern poplar box site. The

Table 1. Annual and seasonal rainfall (mm) at the ironbark and poplar box sites during the study
Project duration was July 1994 to June 2002. Values are means of 8 and 5 rain gauges respectively spread across the sites

Site/Year	1994–95	1995–96	1996–97	1997–98	1998–99	1999–2000	2000–01	2001–02	Long-term mean
<i>Ironbark</i>									
Warm-season rainfall ^A	378	321	717	296	602	398	670	–	475
Cool-season rainfall	5	80	156	95	376	47	262	–	177
Annual rainfall decile ^B	1	3	7	2	8	6	7	–	659
<i>Poplar box</i>									
Warm-season rainfall	314	440	475	506	527	288	501	270	442
Cool-season rainfall	16	102	218	76	402	122	139	92	188
Annual rainfall decile	2	6	7	5	9	4	4	4	626

^AWarm season is Oct. –Mar. of Year 1–Year 2; cool season is Apr. –Sept. of Year 1.

^BDecile values taken from nearest long-term rainfall station for July of Year 1 to June of Year 2.

first 2 years following herbicide application were relatively dry at both sites followed by some good to excellent seasonal rainfall, apart from 1999 to 2000 and for 2001 to 2002 at the poplar box site. The 1994 winter was especially dry (Table 1).

Tree and shrub density and basal area

Ironbark site

There was a very high survival rate of the very small plants from the initial tree-killing in both the grazed and ungrazed trials. Tiny ironbark seedlings were nearly impossible to kill with this method of tree-killing or were overlooked among the pasture.

Grazing experiment: The Bitterlich-derived tree and shrub basal area of the ironbark site was $6.6 \text{ m}^2 \text{ ha}^{-1}$ in 1994 as compared with $7.7 \text{ m}^2 \text{ ha}^{-1}$ in 1994 using the TRAPS procedure. In 1995 silver-leaved ironbark trees accounted for 81% of the total tree density and 81% of the total basal area (Silcock *et al.* 1996). An appreciable number of small plants (544–760 ha^{-1} , or one-quarter of the density of the treed plots in the trees-killed treatments initially survived the herbicide application (Fig. 1). Thereafter the mean tree data varied greatly among treatments due to large initial replicate differences and the continual ‘death’ and resprouting of lignotubers exposed to herbicide and/or the initial dry seasons (Table 1). The basal area of the surviving seedlings and saplings mirrored their initial density. In the trees-killed high grazing pressure paddocks, basal area of remaining saplings steadily, but non-significantly, increased over the ensuing 4 years but not under low to medium grazing pressure (Fig. 1).

Between mid-1995 and mid-1999, live woody plant density in the treed grazed paddocks fell by 8% and basal area declined by 15% due mainly to the death of ironbark trees. The density decrease occurred mostly in plants less than 0.5-m tall, whereas the basal area decrease was recorded for all height classes, except 4–7-m tall. Deaths occurred mostly in plants less than 20-cm tall (204 plants ha^{-1}). After 5 years, grazing pressure had no significant effect on total tree and shrub density or basal area whether trees were killed or not (Fig. 1). There was a recruitment of 145 woody plants ha^{-1} between 1995 and 1999, of which 64 were silver-leaved ironbark. Maintenance of density and basal area in many treatments from 1997 to 1999 (Fig. 1) was associated with improving seasonal conditions.

Prickly pine (*Bursaria incana* Lindl.) was the second most common woody species and was tolerant of the herbicide.

In 1995, 72% of prickly pine plants (178 ha^{-1}) were less than 0.5-m tall and 65% of their basal area was contributed by the height class 7–10-m tall. Its abundance was more stable through dry conditions than that of the ironbark population. On average there were only 90 bloodwood (*Corymbia* spp.) plants ha^{-1} originally, most as seedlings, with a total basal area of $0.83 \text{ m}^2 \text{ ha}^{-1}$ in 1995. Their numbers and basal area were also not significantly affected by grazing pressure over 5 years.

Burning experiment: After the spring fires, most trees and shrubs regained their pre-fire foliage within 5 months. A weak trickle fire in 1995 left many grass tussocks unburnt and only the low tree foliage was scorched, despite $\sim 1600 \text{ kg ha}^{-1}$ of pasture fuel (Hall *et al.* 2016). In 1997, 70% of the foliage of trees over 10-m tall was scorched with most scorching near ground level. The 1998 and 1999 burns were satisfactory with appreciable scorching of even the tallest canopies. By mid-1999 after three burns, there was no significant fire effect on either tree and shrub density or basal area in the treed or trees-killed plots, despite the apparent difference in surviving basal area, $0.04 \text{ m}^2 \text{ ha}^{-1}$ for the burnt compared with $0.25 \text{ m}^2 \text{ ha}^{-1}$ in the unburnt trees-killed plots (Fig. 2).

Despite the excellent kill of large trees by herbicide and ongoing signs of toxicity on many initially surviving seedlings, an appreciable number of small ironbark and bloodwood plants (367–842 plants ha^{-1} depending on the recording year) survived the herbicide and burning treatments in the ungrazed plots (Fig. 2). By 2001 as the trial ended, this was visually apparent and photographically recorded, with scattered silvery-blue foliage of ironbark saplings clearly showing above dry winter pastures (Silcock *et al.* 2005, Fig. 5.2).

Poplar box site

Grazing experiment: Species distribution at the more sloping grazed site was more variable such that poplar box provided between 12% and 83% of the total tree plus shrub numbers in different treatments with a mean of 42%. However, poplar box in the treed treatment always contributed over 90% of the basal area measured, and often over 98%. The original mean tree and shrub density before treatments were applied was 377 ha^{-1} , with basal area of $3.8 \text{ m}^2 \text{ ha}^{-1}$ using the Bitterlich method. This was much lower than the TRAPS methodology (mean $7.7 \text{ m}^2 \text{ ha}^{-1}$) on the permanent transects for several reasons – including site variability, and edge effects from

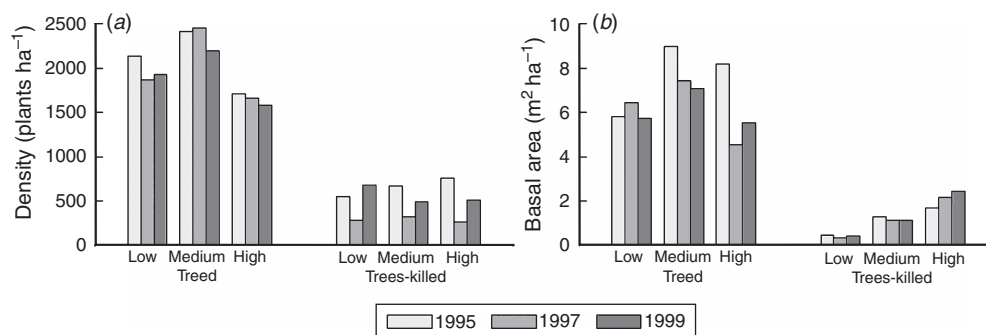


Fig. 1. Effect of grazing pressure on (a) density, and (b) basal area of live trees and shrubs in grazed pastures in ironbark woodland between 1995 and 1999. The interaction was not significant for any of the variables tested for any year.

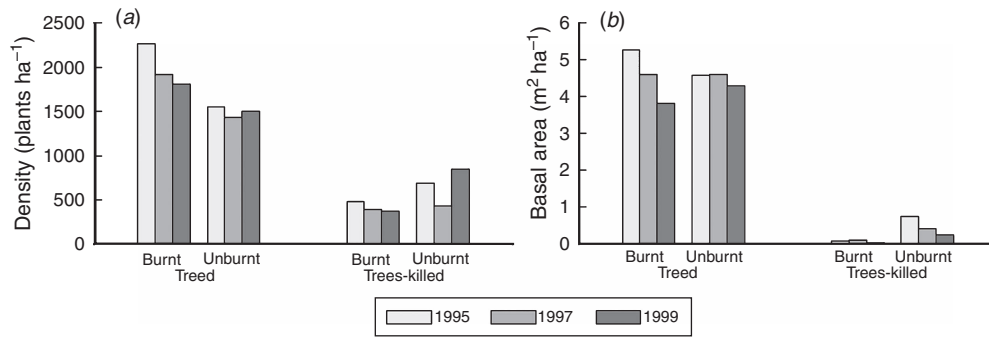


Fig. 2. Effect of spring fires ('Burnt') on (a) density, and (b) basal area of live trees and shrubs in the absence of grazing in ironbark woodland between 1995 and 1999. The interaction was not significant for any of the variables tested for any year. There was a cool spring burn in all plots in September 1998, to reduce excessive moribund pasture prior to this recording.

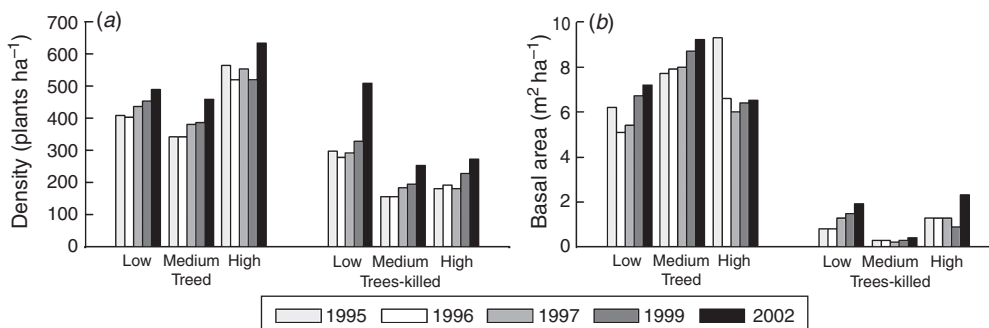


Fig. 3. Effect of grazing pressure on (a) density, and (b) basal area of live trees and shrubs in grazed pastures in poplar box woodland between 1995 and 2002. The interaction was not significant for any of the variables tested for any year.

narrow transects in open woodlands. After the herbicide had killed the treated trees and shrubs, although basal area was much lower on tree-killed sites, the density of plants was still half to two-thirds of the density of the treed plots. Poplar box in mid-1995 made up between 6% and 70% of the remnant population of small trees and shrubs and over 60% of the surviving basal area.

Over the next 7 years (1995–2002), woody plant density increased slowly but non-significantly ($P=0.061$) in most treatments, by an average of 81 and 89 plants ha⁻¹ in the trees-killed and treed treatments respectively, and at the three grazing pressures, of between 81 and 146 plants ha⁻¹ (Fig. 3).

By 2002 there was a trend for increasing numbers and basal area in most paddocks. Recruiting species included poplar box, belah (*Casuarina cristata* Miq.), bull-oak (*Allocasuarina luehmannii* (R.T.Baker) L.A.S.Johnson), false sandalwood, myall and hopbush, all apparently from seed, during the consecutive higher rainfall years.

Canopy cover increased slightly (from 487 to 638 m² ha⁻¹) in the trees-killed pastures under low grazing pressure between 1995 and 2002, whereas there was a small decline of 5 m² ha⁻¹ in the medium grazing pressure treatment and a larger decline (from 584 to 161 m² ha⁻¹) under high grazing pressure.

Burning experiment: The ungrazed burning experiment contained far fewer woody species than the more extensive grazing trial with 19 woody species compared with 60. Of those

19, poplar box (112 plants ha⁻¹) originally contributed 87% of all plants with a mean stem basal area of 6.2 m² ha⁻¹ (range 4.0 to 11.8 m² ha⁻¹). The main minor woody species surviving after the herbicide took full effect include *Eremophila* R.Br. (18 plants ha⁻¹) and the non-natives *Opuntia stricta* (Haw.) Haw. and *Opuntia tomentosa* Salm-Dyck with a combined density of 18 plants ha⁻¹.

Woody plant recruitment was lower than in the grazed treatments (37 ha⁻¹ vs 85 ha⁻¹ over 7 years). Poplar box declined as a proportion of the total woody plant population by 2002, from 87% to 74%, due to appreciable numbers of *Eremophila* and *Jasminum* L. plants recruiting. There was a small increase in average woody plant density over the ensuing 7 years (Fig. 4), from a range of shrubs and also poplar box seedlings.

After 8 years, the basal area in the unburnt plots was marginally higher than in the burnt ones. There was no significant increase in basal area of the small remnant poplar box trees in the trees-killed plots, but the basal area of the treed plots increased by 1.8 m² ha⁻¹ during the experimental period (Fig. 4), mostly due to growth of the existing poplar box trees. Most of the increase occurred after the above-average rainfall in 1998–99.

The canopy cover was consistently higher in the unburnt treatments than in the regularly burnt treatments irrespective of the tree-killing treatments (Fig. 5). The mean winter canopy cover over the four recording times in the treed treatments

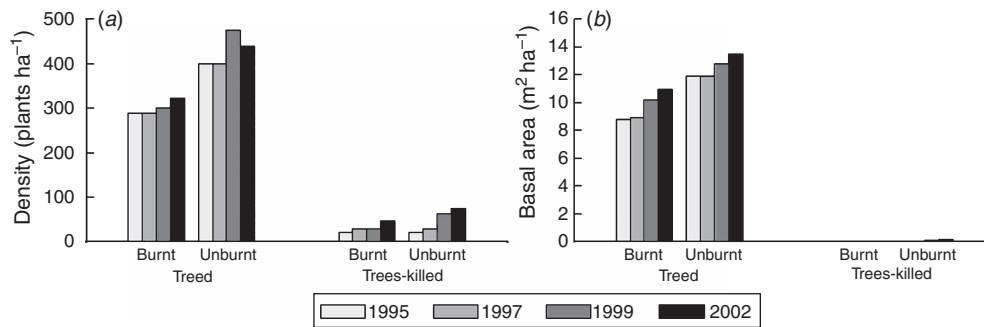


Fig. 4. Effect of spring fires ('Burnt') on (a) density, and (b) basal area of live trees and shrubs in the absence of grazing in poplar box woodland between 1995 and 2002. The interaction was not significant for any of the variables tested for any year.

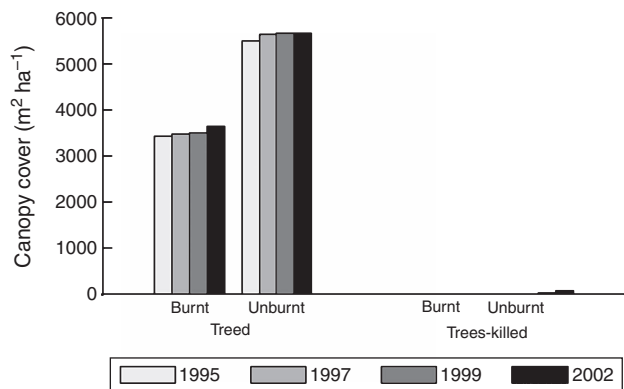


Fig. 5. Effect of spring fires ('Burnt') on canopy cover of live trees and shrubs in the absence of grazing in poplar box woodland between 1995 and 2002. The interaction was not significant for any of the variables tested for any year.

was 3510 m² ha⁻¹ with regular spring burning compared with 5630 m² ha⁻¹ without burning. By comparison, in the trees-killed treatments, there was a mean canopy cover of only 5 m² ha⁻¹ with regular spring burning compared with 33 m² ha⁻¹ without burning.

Germinable soil seed loads

Ironbark site

Surface soil samples taken every spring between 1994 and 2000 produced no germinating seeds of any tree or shrub species from either the grazed or ungrazed areas, despite the presence of mature, flowering specimens in the treed paddocks and plots. Seed-set by silver-leaved ironbark, bloodwoods and prickly pine was noted in the field during the experiments, particularly in higher rainfall summers.

Poplar box site

Germination of woody species seeds from surface soil samples was extremely low. From the grazing experiment, two woody plant seedlings emerged in most years, with none in 1994 and three in 1998. Of these, eight were poplar box, all from paddocks where trees had not been killed. No *Acacia* species germinated and there was a single belah and hophbush seedling recorded. Profuse seed set was noted on several occasions on

poplar box, belah and bull-oak trees, particularly in the spring-early summer period after good rainfall in the previous summer-autumn period.

Soils from the ungrazed plots produced similarly low germination of woody plant seeds, with two seedlings germinating over 7 years. They were poplar box seedlings from the unburnt plots that had trees retained on them, i.e. where there was a reliable potential source of flowering plants.

Soil surface condition

Ironbark site

Throughout the experiment, tree-killing via herbicide resulted in a minor increase in ground cover in the grazed paddocks. There was a small significant improvement due to tree-killing in 1995, 1998 and 1999. However, significantly greater cover (>10% more) was recorded in the autumn of 1994 and 2001 ($P < 0.001$, Fig. 6a). The increase in ground cover in 2001 from 63% to 73% in the trees-killed paddocks occurred when there was a significantly greater pasture yield in those paddocks (Hall *et al.* 2016).

In the ungrazed plots, ground cover was not significantly affected by the killing of trees although the average level of cover increased from about 50% initially to 80% late in the experiment after a series of good seasons. The level of cover was regularly about 10% greater than under grazing (mean of 70% vs 60% over eight recordings), but still varied markedly in response to seasonal rainfall.

Landscape functionality assessments in 2001 using the Tongway and Hindley (1995) methodology did not find any statistically significant differences due to herbicide use 7 years after the trees were killed (Table 2) and the slight difference was towards poorer landscape surface indices where the woodland remained intact (treed). Under grazing, the stability index and the landscape organisation index were noticeably greater on average where the trees had been killed although, in the absence of grazing for 7 years the infiltration index was appreciably greater (58.6 vs 51.9) where the trees had been killed. Overall, landscape functionality was slightly higher without grazing.

Poplar box site

There was significantly greater ground cover in the grazed trees-killed paddocks than in the treed paddocks in nearly

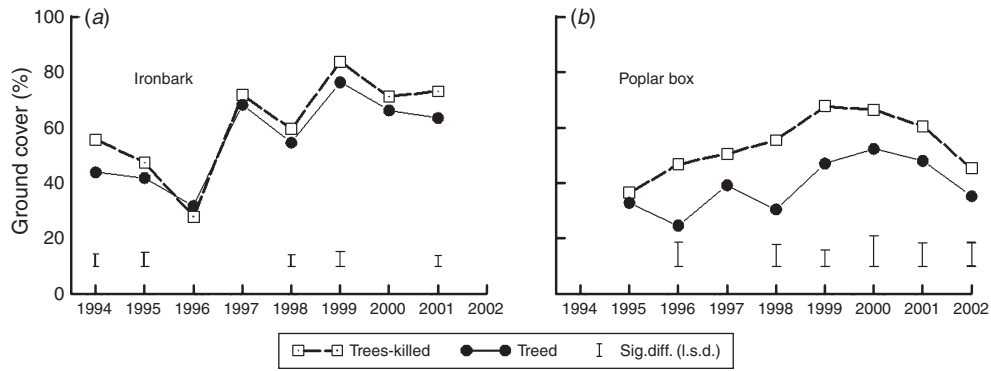


Fig. 6. Total autumn ground cover (%) response to tree competition on grazed (a) ironbark woodland, and (b) poplar box woodland between 1994 and 2002. Bars show l.s.d. values where differences were significant ($P < 0.05$) for that year. (Means of three grazing pressures, after trees were killed in 1994.)

Table 2. Mean soil surface condition response by 2001, without grazing (burning experiment mean) and with continual grazing (mean of three grazing intensities) at the ironbark site

There are no significant differences in any of the variables tested. Note: only 4 of the 12 ungrazed plots were sampled because of negligible slope on other plots

Soil surface attribute	Ungrazed		Grazed	
	Trees-killed	Treed	Trees-killed	Treed
Protected soil (patch) (%)	71	61	56	50
No. of grassy patches/10 m	5.6	6.9	5.9	6.3
Average fetch ^A length (m)	0.5	0.5	0.8	0.8
Stability index	72.0	67.8	64.2	61.9
Infiltration index	58.6	51.9	51.4	50.3
Nutrient cycling index	25.8	25.9	26.4	26.3
Landscape organisation index	0.7	0.6	0.6	0.5

^AFetch is the upslope zone above a grassy patch from which incident rain would flow to the patch.

Table 3. Mean soil surface condition response by 2002, without grazing (burning experiment mean) and with continual grazing (mean of three grazing intensities) at the poplar box site

Soil surface attribute	Ungrazed			Grazed		
	Trees-killed	Treed	l.s.d. $P < 0.05$	Trees-killed	Treed	l.s.d.
Protected soil (patch) (%)	86	69	5	29	38	n.s.
No. of grassy patches/10 m	2.6	3.9	n.s.	0.9	1.0	n.s.
Average fetch ^A length	1.3	1.3	n.s.	7.5	6.0	n.s.
Stability index	66.6	62.8	3.3	59.3	57.9	n.s.
Infiltration index	35.1	32.4	2.7	32.5	32.2	n.s.
Nutrient cycling index	28.1	24.7	n.s.	23.5	21.5	n.s.
Landscape organisation index	0.9	0.7	0.1	0.3	0.4	n.s.

^AFetch is the upslope zone above a grassy patch from which incident rain would flow to the patch.

every year after Year 1 (Fig. 6b). The mean ground cover over the experimental period in the treed paddocks under contrasting grazing pressure was about 70% of that in the trees-killed paddocks, averaging 39% versus 54% over 8 years.

In the ungrazed plots, a similar response existed with the treed plots averaging 41% cover, significantly less compared

with 64% on the trees-killed plots. Overall, ground cover was 13% greater than on the grazed paddocks, 52% versus 46%, and was also strongly influenced by rainfall.

Continuous grazing on the poplar box country also produced no statistically significant effect on LFA indices whether the trees had been killed or not (Table 3) but again the assessment protocol pointed to an improved landscape surface condition where the trees had been killed. At the ungrazed site, tree-killing resulted in a significant improvement in the LFA parameters (Table 3), landscape organisation indices and also on the percentage of the soil surface protected by surface cover (86 vs 69%). Similar to the ironbark site, landscape functionality was higher without grazing.

Discussion

Landscape functionality benefits were anticipated from the near complete killing of mature trees and large saplings at both sites due to increased pasture growth and ground cover. Regular burning and low grazing pressure was also expected to maintain low tree and shrub densities and enhance ground cover. The expected pasture yield response occurred rapidly at the poplar box site but was unexpectedly slow and muted at the ironbark site (Hall *et al.* 2016). A doubling of pasture yield could be expected at both sites (Jackson and Ash 1998). Increasing pasture growth with decreasing tree competition is the most common response however tree species, rainfall, soil and grazing management all interact (Scholes and Archer 1997). Woody plant regrowth rate after tree-killing was reduced by spring fires and low cattle grazing pressure.

Tree and shrub recruitment and growth

Neither woody seedling establishment nor drought-induced death was apparent at either site. Seasonal conditions during the trial were not conducive to either response. Fensham *et al.* (2017) have shown that woody competition in Eucalypt savanna can be significantly reduced during drought about three times per century and that *Eucalyptus* seedlings up to 2 years old are easily killed by fire. Due to the short term nature of the trial we were not able to integrate either with our burning management to achieve a significant reduction in woody populations. Scholes and Archer (1997) have also shown that in the absence of

disturbance, such as fire, tree cover will increase at the expense of grass production until it is limited by tree–tree competition. Forecast climate change may well enhance woody dieback due to increased drought intensity and daytime temperatures (Fensham 2012) offering an opportunity to increase pasture yield and more readily manage fire to reduce woody cover and/or achieve high mortalities in woody seedling pulses. Although not specifically demonstrated by our study, the best outcome for ecological and production values would be the maintenance of low woody populations by prescribed fire.

Tree and shrub recruitment in both experiments at both sites was small in the first 7–8 years after herbicide treatment. The lack of statistically significant changes is understandable given the low number of germinable seeds of woody plants in the soil in spring in many years. Although there was a paucity of samples, the survival rates of seedlings of all species is typically low and therefore, large numbers or exceptionally good seasonal conditions are required for seedlings to establish. The trend at the treed ironbark site was towards fewer woody plants from 1995 to 1999. This possibly was a continuation of the death and ill-thrift of ironbark trees in central Queensland reported by Fensham *et al.* (2009) after the 1994–97 drought. It is possible the ironbark canopy cover, (which is a better measure of water use and hence competitive potential with the understorey than basal area) was unusually low due to drought stress, which may have contributed to the initial lack of pasture response after killing all the trees (Hall *et al.* 2016). McDowell *et al.* (2008) has quoted Manion (1991) who suggested a slow-decline hypothesis for causes of mature tree death in which plants experience a three-stage decline over many years, driven initially by long-term stress, for example poor edaphic location, followed by a severe short-term stress, for example drought, and finally death occurring via a contributing factor, for example pathogen. Therefore, the reduced competitive potential of the trees initially could have existed for several years prior to the trial starting.

There was a very high survival rate of the very small plants from the initial tree-killing in both the grazed and ungrazed ironbark trials. Tiny ironbark seedlings were nearly impossible to kill with the method of tree-killing used or were overlooked among the pasture. The trees-killed high grazing pressure paddocks had a steady, increasing trend in basal area of surviving ironbark trees and shrubs likely due to the much-reduced pasture competition on young woody plants. Also, some surviving saplings were in the 4–7-m tall class and thus had not reached a fully mature stage where canopy losses associated with old age might occur. They had a strong potential for basal area increase when the overstorey and pasture competition was reduced. By comparison, trends at the poplar box site were generally of slowly increasing numbers and basal area of trees and shrubs in all treatments. This is consistent with the 30-year-old regrowth still having potential to increase basal area and canopy cover, albeit at reduced rates with regular spring fires. The increasing woody basal area of 1.2% per year was consistent with findings of Burrows (2002) in central Queensland eucalypt woodlands that had an annual increase of 2.1% per year. Differences in the rate of basal area increase of heavily grazed treed plots in the grazing trial versus the burning trial is unlikely due to grazing as any consumption by cattle is only of the

tips of regrowth whereas that from wallabies and insects is unknown.

A small number of woody plants of diverse species remained at the poplar box site after tree-killing (21 ha^{-1}) and this was to be expected from a commercial application operation. All of the poplar box recruits occurred in the latter years of the trial when seasonal conditions were improving and they could have come from seedset in nearby treed areas. As poplar box seeds have no significant dormancy mechanism (R. G. Silcock, pers. obs.) they are unlikely to have come from seed that existed in the soil at the start of the trial. Their ingress from the adjacent trees-killed paddocks could have been via strong winds, surface rainwater flow or mislaid, ant-harvested seeds. The higher density of diverse species such as myall and yellow-berry bush (*Maytenus cunninghamii* (Hook.) Loes) in the grazed versus ungrazed burnt poplar box experiments (85 ha^{-1} vs 37 ha^{-1}) facilitated the higher average woody plant density increase on the grazed sites. This is in contrast to Scanlan *et al.* (1996) who reported a higher recruitment of eucalypts in enclosed plots than grazed plots over 6 years (270 ha^{-1} vs 160 ha^{-1}) but similar to the 64 ha^{-1} recruited in the treed plots of the grazing trial at the ironbark site.

Seed in the soil

The lack of germinable seeds of tree and shrub species in the soil over many years indicates that woodland regeneration from seed is dependent on the chance combination of viable seeds, appropriate rainfall and then favourable subsequent conditions for seedling survival. Woody plant seedlings do not flower for many years and have to survive numerous threats beforehand including fire, drought, grazers, pests and disease. Woody plant seedlings are most susceptible during their first year and, if they survive that long, they will have established a sturdy taproot and be well placed to survive subsequent severe foliage damage. By then most eucalypts will have developed lignotubers and all would have buds low down on the stem from which they can reshoot after top damage. Hence, on an annual basis, successful seedling recruitment does not need to be significant but on a decadal scale it is probably vital. Common understorey shrubs like *E. mitchellii* and *Carissa ovata* R.Br are rarely observed to germinate in the field, yet they are very common in the woodland understorey in central Queensland (R. G. Silcock, pers. obs.). Back (2005) also made this observation for *C. ovata*. This would indicate that once woody plant seedlings do survive their seedling year they are resilient and their rate of growth is controlled mainly by overstorey competition, annual rainfall and fire. Nonetheless, a lack of detailed, long-term field observations of tree seedling dynamics means that episodic widespread losses due to insect pests and diseases cannot be excluded as a common factor in recruitment levels in Queensland woodlands. The relative proportion of seedling recruits in this study is possibly artificially high compared with those that would occur where much greater distances exist to the nearest untreated woodland area or strip.

Dormancy in seed of species like *Acacia* and wilga (*Geijera parviflora* Lindl.) is well documented (Ralph 2003; ANPS 2016) whereby viable, dormant seeds are blocked from germinating by a hard seed coat restricting water entry in the former species and a

complex physiological control in the latter. Soil from beneath most mature trees and shrubs usually produces some germinable seeds of that species (R. G. Silcock, pers. obs.). There was no difference between the burning and grazing trials for woody seed in the soil indicating that fire did not enhance germination. Hence a lack of seed germination at our sample sites was almost certainly due to a lack of germinable seed of trees and shrubs over those years. This applies particularly to the grazed poplar box site where some paddocks had adult myall trees for shade and many small saplings yet no myall seedlings germinated from our samples. Wind dispersal is feasible for species with winged seeds such as belah, false sandalwood and whitewood (*Atalaya hemiglauca* (F. Muell.) F. Muell. ex Benth.) but did not seem to occur during our trial period.

Ground cover changes

Throughout the experiment, tree-killing via herbicide resulted in a minor increase in ground cover at the ironbark site (Fig. 6a), which is not surprising given that standing pasture yield (Hall *et al.* 2016) was largely unaffected by tree-killing. The stem injection treatment was applied in late March 1994 so that the early significant increase in ground cover by July 1994 could only have been the result of leaf drop from the dying trees. The increase in ground cover due to tree-killing in 2001 (73% vs 63%) was associated with a significantly greater pasture yield (4840 vs 3080 kg ha⁻¹), but only after six previous years with no yield response. Rainfall patterns could not explain the absence of an effect of tree-killing on ground cover.

The ground cover at the poplar box site showed similar yearly trends to that of standing pasture yields. The decline in cover in the last 2 years follows the declining rainfall during this period. A proportion of this cover in treed areas was from the regular poplar box leaf litter fall in most winters.

Landscape functionality shifts

The LFA protocol was developed in rangeland environments where tree density is usually low. Hence we are uncertain how readily it extrapolates to our more densely treed paddocks as tree trunks reduce the area of capture patches and of potential 'standard' soil surface. In many ways tree trunks act as run-on areas due to stemflow during storms (Pressland 1976) and their canopy provides litter and pod fall and protection from incident raindrops and, for some eucalypt species, enhanced growing conditions for some pasture grasses (Ebersohn and Lucas 1965; Silcock 1980). Tongway and Hindley (1995) include the effect of tree and shrub canopies into their indices but make no overt mention of the possible role of tree trunks.

At our sites tree-killing generally resulted in better LFA indices but the improvement was mostly non-significant. Only at the ungrazed poplar box plots were significantly greater indices recorded, 8 years after killing the trees (Tables 2 and 3). Here the proportion of ground covered by patches of litter plus grass tussocks was 17% higher where the trees had been killed compared with land beneath trees. This translated into a 6% greater stability index and an 8% greater infiltration index based on the calculation protocols of Tongway and Hindley (1995). The nutrient cycling index was not significantly affected although the trend was similar with a notional 13% improvement with tree-

killing. That is somewhat counter-intuitive because nutritional and microbial enhancement produced in soil by long-term tree litterfall can be dramatic for species with specific nutritional needs, such as exotic grasses and weeds (Silcock 1980; Jackson and Ash 1998).

The herbicide application via stem injection is a method that creates minimal soil disturbance while ensuring a large reduction in woody basal area. It is very suited to *Eucalypt* woodlands with native pastures where introduced pastures are not being established. Tree-killing by chaining is required to disturb the soil surface for successful introduced pasture establishment (FutureBeef 2018). Chaining would quite likely reduce some of the LFA indices as the soil surface is disturbed, dispersive sub-soils can be exposed and soil particles can be transported more readily.

Management lessons

Post-treatment management has been shown to gradually influence the rate of woody plant regrowth and the general level of ground cover. The speed of such regrowth of eucalypt woodlands in our study was shown to be strongly influenced by seasonal rainfall such that the excellent winter rains of 1998, following a good summer, accelerated growth of existing trees and shrubs as well as establishing new seedlings. New tree and shrub plants were more frequent where the adult woody plants had not been killed (Fig. 3) and where grazing pressure was high. Thus, our second hypothesis that grazing pressure would not directly affect woody plant density and basal area was shown not to be universally valid – compare Figs 1 and 3. Woody basal area tended to increase proportionally more than density and canopy cover increase. Increased basal area usually correlates closely with a reduction in pasture growth (Burrows *et al.* 1988; Scanlan 2002), which is a concern to many pastoralists in the study area. Contrary to our first working hypothesis, regular spring fires did not reduce woody plant density and basal area, merely slowing the increase of both the canopy cover and basal area of trees and shrubs and with an inconsistent effect on recruitment of new woody seedlings (Figs 2 and 5). The best outcome for ecological and production values would be the maintenance of low woody populations by prescribed fire. Low grazing pressure is necessary to ensure adequate fuel loads for frequent burns if needed. It will also enable the adaptive management required to burn opportunistically within 2 years of a woody seedling pulse to obtain high mortalities and prevent future woody thickening problems. However, excessive use of prescribed fire can lead to poorer soil surface condition and needs to be avoided (Silcock *et al.* 2005).

Summary

Tree-killing from appreciable areas of natural eucalypt woodland in central Queensland was followed by a slow but steady regeneration of the trees and a range of shrubs from seed and tiny survivors of the original herbicide application at both locations. Woody plant regrowth post tree-killing was reduced by spring fires and often by cattle grazing pressure. Hence, land management after tree-killing can influence the rate at which eucalypt woodlands inevitably regenerate. The presence of isolated mature, seeding trees would provide greater

recruitment potential within cleared areas because the seeds of many species are not wind dispersed. Tree-killing tended to improve landscape processes for a significant number of years, in terms of soil surface condition, and on some criteria improved the state of the land surface.

Conflicts of interest

The authors declare no conflicts of interest.

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