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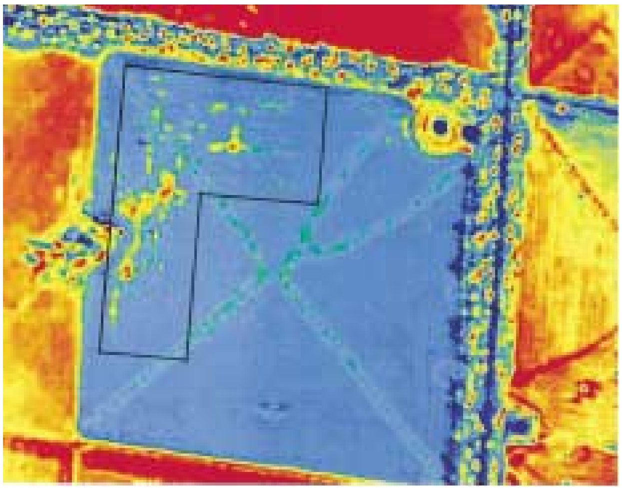

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Mortality of *Prosopis pallida* (mesquite) following burning

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Abstract. A 2-year study investigated fire as a potential control technique for the invasive woody weed *Prosopis pallida* (Willd.) Künth by comparing changes in plant density and stem basal area between burnt and unburnt plots. Three months after burning only 8% of the original 1760 plants per hectare remained alive, compared with 100% survival in control plots. Over the ensuing 21 months, a further 2.5 and 21% of the original plants died in burnt and control plots, respectively, and 90% of control plants that died were young plants (less than 0.5 m in height). Burning also reduced recruitment of *P. pallida* during the 2 post-fire years, although very few seedlings that emerged remained alive at the end of the study period in either control or burnt plots. Two years after treatments were applied, the population of *P. pallida* in control and burnt plots had decreased by 7 and 93% of the initial populations, respectively. Similarly, the total stem basal area of *P. pallida* in burnt plots after 2 years was 0.9 m^2/ha, 93% less than that in the unburnt controls. Seedling recruitment had a minimum impact on stem basal area in both control and burnt plots, contributing less than 0.005%. Subsequent control measures will be necessary to treat any plants not killed by fire and any new seedlings.

Introduction

Prosopis pallida (Willd.) Künth (Algaroba mesquite), a spiny shrub or small tree native to South America (Chile, Peru), infests an estimated 500000 ha of grazing land in north-west Queensland (Jeffrey and March 1995). Currently, most of this area consists of isolated plants or light infestations with only a few dense stands having developed (van Klinken and Campbell 2001). Control of these isolated infestations now will save the greater cost of eradication in the future.

Isolated mesquite plants probably have a minor impact on grazing productivity and may even enhance production in the short term due to the nutritious seed pods and shade they provide (Csurhes 1996). However, the inevitable thickening of these infestations with time can result in a decrease in carrying capacity through loss of grass cover caused by replacement and by competition for limited water (DeLoach 1985). Displacement of natural vegetation, stock handling difficulties, injury to livestock and damage to property can also occur (Jeffrey and March 1995).

The increased abundance of woody plants in savannas and grasslands is a worldwide occurrence. There is a plethora of examples in the literature, with equally many suggestions for why these changes have occurred (Johnson *et al*. 1993; Archer 1995; Scholes and Archer 1997; van Auken 2000). Archer (1995) recognised some 62 documented instances of increased abundance of woody plants in arid and semi-arid ecosystems in recent history.

Generally, it is accepted that the major factors influencing the distribution and relative proportion of grasses and woody plants are soil water availability (the seasonality and amount of rainfall and soil type), the frequency and magnitude of disturbances such as grazing, browsing and fire, and the worldwide increase in atmospheric carbon dioxide. Of these, a decrease in the incidence of fire is considered a primary factor and is often attributed to overgrazing, with grazing animals reducing available fuel loads and consequent reduction in fire frequency and intensity (Fisher 1977; Bahre and Shelton 1993; Archer 1995; Scholes and Archer 1997; van Auken 2000). Increasing atmospheric CO_2 confers a competitive advantage to plants with a C_4 metabolic carbon pathway (shrubs and cool season grasses for example) over plants with a C_3 pathway (warm-season grasses) and also increases drought tolerances of C_4 plants (Johnson *et al*. 1993).

The use of fire for controlling *Prosopis* spp. has met with limited success in areas where it has been tested as a control technique, the general conclusion being that mature plants of most species are tolerant of fire, and although defoliated, usually re-sprout (Glendening and Paulsen 1955; Wright *et al*. 1976; Hamilton and Scifres 1982; Martin 1983; Bock and Bock 1992; van Klinken and Campbell 2001). However, most of the literature reported is restricted to only a few of the 44 recognised species (Burkart 1976) and it is feasible that not all *Prosopis* spp. will respond to fire in the same manner.

To quantify the susceptibility of *P. pallida* to burning, a field study was undertaken in northern Queensland. Changes in plant density and stem basal area were compared over a 2-year period between burnt and unburnt plots.

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Materials and methods

Study site

The experiment was conducted near Hughenden (20°51′S, 144°12′E) in north-west Queensland between October 1996 and October 1998. Mean annual rainfall is 492 mm with 70% falling between December and March (Fig. 1). Mean daily maximum temperatures range from 24.7°C (July) to 36.3°C (December) and mean daily minimum temperatures range from 9.3°C (July) to 23.2°C (February) (Bureau of Meteorology 1995).

Based on comparisons with nearby areas not infested with *P. pallida*, the vegetation of the site before the spread of *P. pallida* would most likely have been an open woodland of *Lysiphyllum gilvum* (F. M. Bailey) Pedley, *Acacia shirleyi* Maiden and *Eucalyptus* spp. At the time of treatment, few native trees remained alive and the introduced grasses *Cenchrus ciliaris* L. and *Cenchrus setiger* Vahl dominated the ground cover. Domestic herbivores had been excluded from the site for a period of 12 months.

Soils ranged from texture contrast soils with sandy loam to clay loam A horizons over clay B horizons, to uniform non-cracking clays. Classifications include Lutic rudisols, Haplic chromosols and Haplic dermosols (Isbell 1996). Structure of A horizon soils is weak to moderate and there is some surface crusting. Some soils are underlain by unconsolidated sands and gravels at depths of as little as 90 cm.

Measurement of plant responses

Six areas containing relatively uniform stands of *P. pallida* and of sufficient size for 2 plots of 1 ha were selected. Five metre wide firebreaks were prepared around each plot by removing all trees and grass tussocks. From each pair of plots, one was randomly selected to be burnt and the other remained as an unburnt control.

A belt transect (50 m long and 10 m wide) was established within each plot 1 month before burning, and the location, stem basal diameter (at 20 cm above ground level for plants above 50 cm in height and at ground level for plants below 50 cm) and height of all *P. pallida* plants were recorded. Similar measurements were undertaken at 3-monthly intervals until October 1998, with seedling recruitment and adult plant mortalities recorded.

Within 2 days of burning, an assessment of fire damage was undertaken on all *P. pallida* plants located within transects, using the damage index outlined in Table 1.

Table 1. A damage score index for classifying fire damage to *Prosopis pallida* (adapted from R. Dyer pers. com.)

Damage score	Classification	Description of canopy damage
0	Unburnt	No physical damage
1	Very slight	Scorched with clusters of green leaf
2	Slight	Some leaves burnt but mainly scorched
3	Moderate	Most leaves burnt, but some scorching
4	Severe	All leaves burnt off
5	Very severe	All leaves and small twigs burnt off
6	Total destruction	Plant totally consumed to ground level

Fire characteristics

Fuel load and plant and soil moisture content were determined before burning. The Botanal procedure (Tothill *et al*. 1978) was used to measure fuel load, with 100, 50 by 50 cm quadrats recorded systematically in each plot. The moisture content of plant and litter material was determined immediately before burning by harvesting 5 quadrats (50 by 50 cm) in each plot and separating plant and litter material. Gravimetric moisture content within the top 5 cm of the soil profile was determined from 5 soil cores of 5 cm diameter, collected at random from within each plot. All plant, litter and soil samples were weighed, dried for 48 h (80°C for plant material and 105°C for soil samples), and re-weighed to calculate moisture content.

Burning was undertaken over 2 days (20–21 November 1996) in late spring when conditions were conducive to a ‘hot’ burn. Both litter and grass material were dry (Table 2) and environmental conditions on both days were characterised by hot temperatures, low humidity and a light but fluctuating wind (Table 2). All plots were burnt by head fires after an initial backburn phase with wind speed, ambient temperature and relative humidity recorded at the start and then periodically during burning. After the initial passage of the fire, remaining unburnt patches were manually ignited if sufficient fuel was available to carry a fire.

Temperatures experienced during the fires were recorded at 3 s intervals using type K steel encased thermocouples connected to data

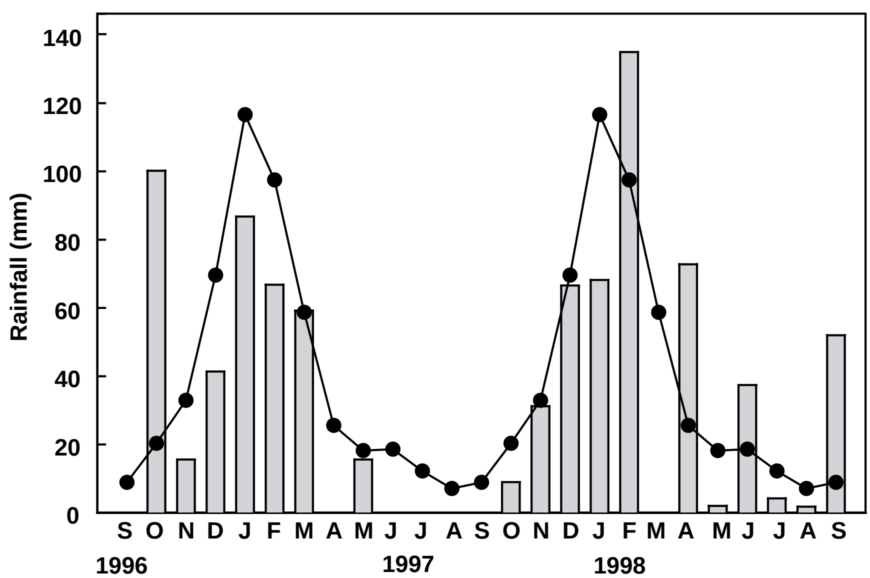

Figure 1. Monthly rainfall received at the experimental site (solid bars) and the long-term mean (●) for Hughenden.

Table 2. Fuel characteristics and environmental conditions experienced during burning of six experimental plots at Hughenden

Parameter	Mean ± s.e.	Range
Fuel load (kg/ha)	2110 ± 140	948–2626
Air temperature (°C)	36.85 ± 1.38	33.27–38.02
Relative humidity (%)	13.80 ± 1.64	10.53–18.26
Wind speed (m/s)	2.15 ± 0.38	1.04–2.70
Plant moisture content (%)	35.97 ± 3.80	28.77–45.87
Litter moisture content (%)	4.21 ± 0.65	2.63–5.82
Soil moisture content (%)	6.77 ± 1.14	3.71–9.20

loggers (Data Electronics Pty Ltd) buried 5 cm below the soil surface. At 4 locations within each plot temperatures were recorded at 0, 0.5, 1, 1.5 and 2 m above the soil surface. Two locations were within grassed areas and the other 2 in densely populated patches of *P. pallida* where litter was the only fuel.

On the day after burning the Botanal procedure was used to measure the unburnt fuel load and to determine the proportion of area not burnt. As for pre-burn measurements, 100, 50 by 50 cm quadrats were recorded systematically in each plot.

Data analysis

Paired *t*-tests (Steel and Torrie 1980) were used to compare plant density and basal area changes between burnt and unburnt plots. Analysis of variance was used to compare fire temperature variations between height levels and between fuel types.

Results

Fire behaviour

Plots contained a mean (± s.e.) fuel load of 2110 (± 140) kg/ha, with 61 (± 5)% of the ground area containing grassy fuel, 37 (± 5)% containing litter material only and 2 (± 1.5)% bare ground.

Burning consumed 80 (±10)% of the fuel present and 60 (±12)% of the ground surface area was burnt. Maximum temperatures reached during burning varied depending on the type of fuel present. Highest temperatures were recorded within thickets of *P. pallida* when *P. pallida* litter was the primary fuel source. Under these conditions, temperatures reached a maximum close to the ground surface (Fig. 2). Maximum temperatures in grassed areas averaged 135°C cooler than those in litter, but temperatures remained higher further above ground.

Impact of fire on P. pallida *density*

Original plants. A single spring burn significantly ($P<0.05$) decreased the density of *P. pallida*. Three months after treatment only 8% of the original plants (1760 plants/ha) remained alive in burnt plots, compared with 100% survival for control plants (Fig. 3). Over the ensuing 21 months, a further 2.5 and 21% of the original plants died in burnt and control plots, respectively. Of the control plants that died, 90% were small, less than 0.5 m in height.

All *P. pallida* plants were susceptible to fire irrespective of size (Fig. 4), with mortality dependent on the level of damage imposed. More than 95% mortality occurred across all size classes even where fire damage was only slight (rating 2). Where fire damage was moderate (rating 3) or greater, mortality was 100%. Of the 550 plants that received sufficient damage to cause at least top kill, only 2 regrew and this regrowth occurred from protected basal buds at the base of the plant. Most of the surviving plants in burnt plots were either unburnt or only very slightly damaged (rating 1). Larger plants had higher survival where only slight damage (rating 1) occurred.

Post fire recruitment

Average rainfall over the 2 summers encompassed by the study period was below the long-term mean of 396 mm (Fig. 1); averaging 370 and 310 mm for the 1996–97 and 1997–98 seasons, respectively. Nevertheless, for the 2 years following burning, 7 rainfall events produced germination of *P. pallida*: 3 during the 1996–97 summer and 4 during the 1997–98 summer. Maximum seedling density in 1996–97 seedlings was recorded following the first substantial (>25 mm) summer rainfall event in January 1997 where more than 200 mm of rainfall fell over a 9-day period. Burning did significantly ($P<0.05$) decrease the number of seedlings present, with 4000 and 2400/ha recorded in control and burnt plots (Fig. 3), respectively, 3 months after treatments were implemented. Similarly, during the 1997–98 summer, significantly more seedlings were recorded in control plots. In January 1998, 1735 seedlings/ha were present, more than 5 times the number in burnt plots. By October 1998, most seedlings that had emerged over the previous 2 years had died, with 254 and 16 seedlings/ha remaining alive in control and burnt plots, respectively.

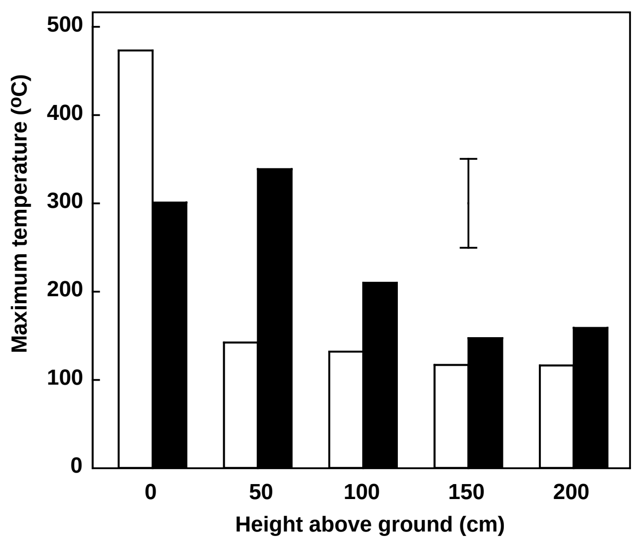

Figure 2. Maximum temperatures (°C) reached 0, 50, 100, 150 and 200 cm above ground in areas where the fuel load was dominated by either litter (□) or grass (■). Vertical bar represents least significant difference ($P = 0.05$).

Total P. pallida *population*

The density of *P. pallida* fluctuated markedly between the 3-monthly sampling times (Fig. 3), as new seedlings emerged after rainfall events and both fire induced and/or plant mortality due to other causes occurred. Nevertheless, at all post-fire sampling times the population of *P. pallida* was significantly ($P<0.05$) lower in burnt plots. Two years after treatments were applied, the populations in control and burnt plots had decreased by 7 and 93% of the original populations, respectively.

Impact on stem basal area

As for plant density, a single spring burn significantly ($P<0.05$) decreased the stem basal area of *P. pallida* (Fig. 5). Two years after treatment, the stem basal area of *P. pallida* in burnt plots was 0.9 m^2/ha, 93% less than that in unburnt controls. Seedling recruitment had a minimum impact on stem basal area in both burnt and control plots, contributing less than 0.005%.

Discussion

Fire is an effective control technique available to land managers for controlling the invasive *P. pallida* in North Queensland. Plants of *P. pallida* were highly susceptible, irrespective of their size. Post-fire recruitment of *P. pallida* in burnt plots was also less than that in unburnt controls. The reason for this in the first year after burning was possibly due to fire directly killing a percentage of the seed bank, most likely those seeds located on or close to the soil surface. A concurrent study (B. Lynes unpublished data) found that the number of viable seeds of *P. pallida* on the soil surface was reduced from 82 seeds/m^2 before a fire to 6 seeds/m^2 afterwards, with no difference in pre- and post-burn samples of buried seeds. Similarly, Cox *et al*. (1993) reported that, following fire, all seed of *P. velutina* Wooton failed to germinate if located on the soil surface, while seed buried 2 cm deep was unaffected.

The significant decrease in seedling numbers in the second year could have resulted from the fire killing a large number of reproductive trees. As such, there would have been fewer plants available to replenish the seed bank.

The effectiveness of fire on *P. pallida* contrasts with results for most other *Prosopis* spp. where, generally fire kills young plants but not older plants. For example, Wright *et al*. (1976) reported that *P. glandulosa* (honey mesquite)

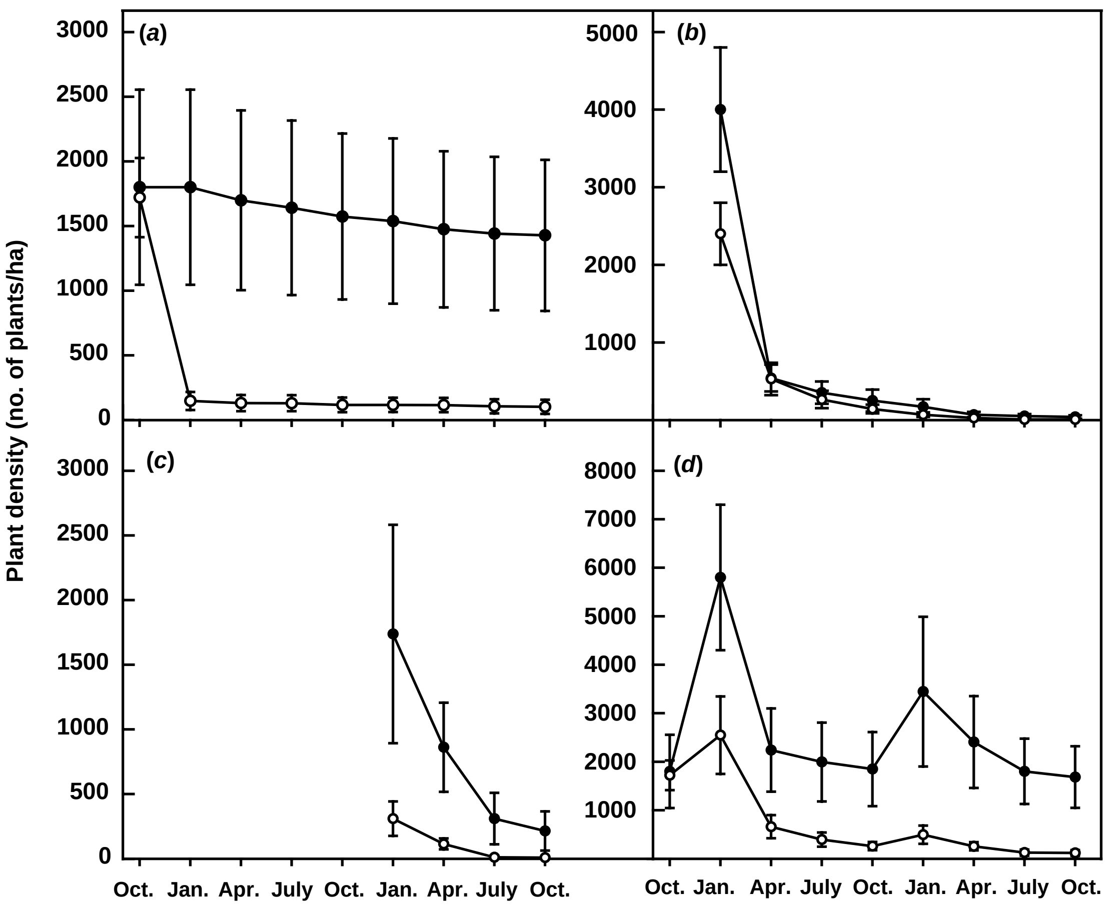

Figure 3. Changes in the density of (*a*) original, (*b*) 1996–97, (*c*) 1997–98 seedlings and (*d*) all *Prosopis pallida* plants over a 2-year period in burnt (○) and control (●) plots. Vertical bars represent standard errors.

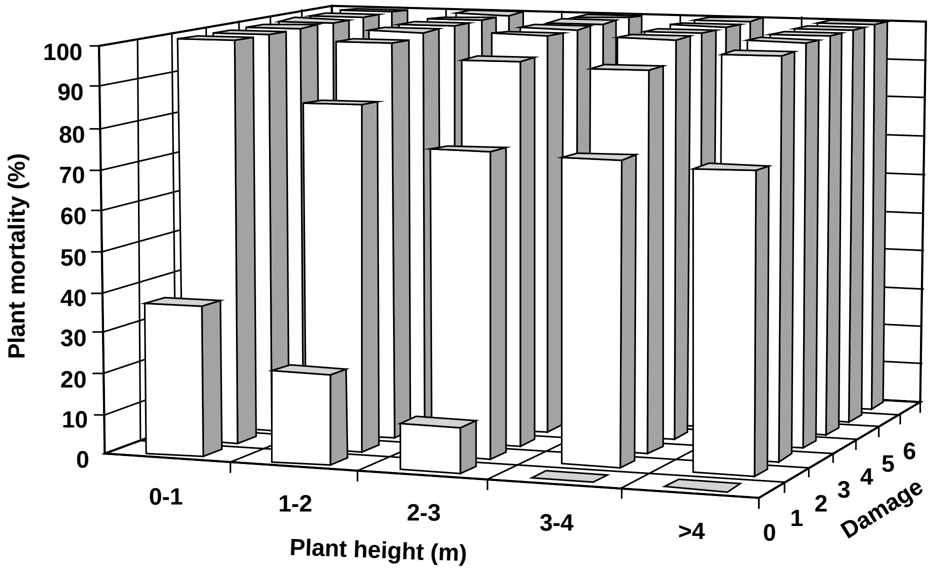

Figure 4. The effect of severity of fire damage on the mortality of different sized *Prosopis pallida* plants, using the damage score index (Table 1).

trees up to 1.5 years of age were readily killed by fire, those 2.5 years of age were severely harmed, but those older than 3.5 years were very tolerant of fire. They concluded that *P. glandulosa* trees older than 3 years were difficult to kill unless they had been previously top-killed by herbicide treatment. Similarly, Boo *et al*. (1997) concluded that both *P. flexuosa* DC. and *P. caldenia* Burk. exhibited minimal mortality (<10%) following controlled burns. McLaughlin and Bowers (1982) did, however, report a 50% mortality of *P. juliflora* (Sw.) DC. in the Sonoran desert following a

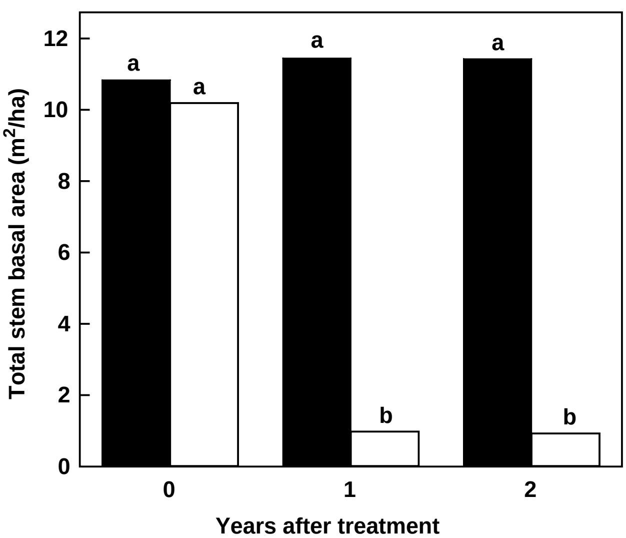

Figure 5. The stem basal area of *Prosopis pallida* in burnt (□) and control (■) plots, 1 month before, and 12 and 24 months after burning. Within years, bars with the same letters are not significantly different ($P<0.05$).

wildfire, but suggested that this result may have been due to an unusually high fuel load as a result of the previous 2 winters receiving above average rainfall.

Personal observations of other fires in northern Queensland concur with the findings of the current study. However, some of these did not appear to produce as high a mortality of *P. pallida*. Such differences could be associated with a number of factors, including environmental conditions at the time of burning, physiological status of the plants and the quantity and moisture status of the fuel load. More detailed studies on the timing and intensity of burning will improve management recommendations regarding the use of fire as a control technique for *P. pallida*.

If effective, using fire as a control technique for invasive woody weeds, such as *P. pallida*, offers some advantages over other available methods. In particular, it is relatively inexpensive (Burgess 1987; Hodgkinson and Beeston 1995) compared with chemical and mechanical control techniques and, furthermore, it can be applied over large areas. Even if trees are not killed by fire, top kill can occur, which may reduce seed production for a number of years and also cause a temporary reduction in woody cover. This could assist other control techniques and increase grass production. Effectiveness of the treatment depends on how long it takes for plants to return to their pre-burn size.

A major constraint to using fire is fuel availability and in pastoral areas any destocking period required to facilitate the accumulation of fuel will impose an economic cost (Grice 1997). Hodgkinson and Beeston (1995) quoted costs of A\$0.50–1.20/ha for burning of rangelands dominated by *Eremophila mitchellii* Benth. in Australia and Teague *et al*. (1997) estimated that burning for *P. glandulosa* control in

Texas would cost US$2.50–5.00/acre. However, because *P. pallida* appears so susceptible to fire, all attempts should be made to ensure that the initial burn is highly effective. The costs associated with locking up a paddock to achieve this are only small when compared with the alternate control options of chemicals or machinery (van Klinken and Campbell 2001).

The susceptibility of *P. pallida* suggests that fire should be reassessed for other *Prosopis* species posing problems in grasslands. Preliminary data from burning on Mardie station (Western Australia) found that mesquite taxa present there are also susceptible to fire (van Klinken and Campbell 2001). Mortality from a hot fire averaged 88%, with mortalities of 60% or more in all size classes. A further 9% of trees were top-killed and no seedlings were found, despite being common in nearby unburnt areas. Burning will also decrease other fire susceptible woody weeds that may be growing in association with mesquite such as *Cryptostegia grandiflora* (Roxb.) R. Br., but not *Ziziphus mauritiana* Lam. as it is able to survive (Grice 1997).

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