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Spatial pattern and severity of fire in areas with and without buffel grass (*Cenchrus ciliaris*) and effects on native vegetation in central Australia

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Abstract The spread of buffel grass (*Cenchrus ciliaris*) in semi-arid Australia in recent decades has substantially increased ground cover and fuel loads, particularly in open woodland vegetation communities. The resulting alteration of fire regimes may be the most significant impact of buffel invasion on ecological communities in these areas. Broad scale management of buffel grass is currently not an option in Australia but it is becoming increasingly relevant to assess the benefits of restoring areas of native vegetation where preventing buffel grass invasion is no-longer possible. We managed buffel grass in a series of experimental plots from 2008–2012. In June and August 2011, two unplanned fires burnt through the plots providing a unique opportunity to compare the outcome of wildfire, including the spatial pattern of fire, and the effect on ground vegetation and on a long-lived, perennial overstorey species, in replicated managed and unmanaged plots. The area of ground that remained unburnt was much greater in managed plots (with predominantly native vegetation) than unmanaged (predominantly buffel grass) plots and where the managed plots did burn the fire was more patchy. This had direct implications for the richness of ground layer plant taxa following fire and the extent to which overstorey trees were exposed to fire. Fire increased pre-existing differences in the number of taxa in the ground level vegetation, an effect that persisted for the duration of our study, suggesting that fire accelerates direct negative competitive effects between buffel grass and native grasses and forbs. *Hakea divaricata* (fork-leafed corkwood) trees in unmanaged buffel grass sites suffered higher burn intensities, and their long-term viability at this location is likely to be threatened if fires fuelled by buffel grass continue. Our results demonstrate clear benefits of removing fire-enhancing invasive plants from areas of high conservation value.

Key words: buffel grass, *Cenchrus ciliaris*, fire, *Hakea divaricata*, invasive grass, restoration.

INTRODUCTION

Buffel grass (*Cenchrus ciliaris* (L.) Link) is an invasive grass native to east Africa and southern Asia, and is considered to be among the most serious environmental weeds in arid and semi-arid areas where it has been introduced (Low 1997). A recent review by Marshall *et al.* (2012) outlined the ecology, distribution and current knowledge of biodiversity impacts of buffel grass in areas where is has invaded. Buffel grass is highly successful in low rainfall environments where relatively sparse vegetation is the norm. These areas can be transformed into dense grasslands which exclude many native grasses and forbs, transform the structural habitat for fauna and, perhaps most significantly, carry greatly increased and more continuous fuel loads compared to the pre-invasion state. This increase in fuels, in combination with the tendency for weather conditions in arid regions to promote intense fires, has led to more hazardous and more frequent or extensive fires in areas where, historically, fire has been relatively infrequent. In addition to immediate and ongoing risks to humans and property the alteration of fire regimes may be the most significant ecological impact of buffel grass invasion (Humphries *et al.* 1991; Butler & Fairfax 2003; Miller *et al.* 2010; McDonald & McPherson 2011; Brooks 2012). Interactions between introduced grasses and fire are of serious concern worldwide (D'Antonio & Vitousek 1992), and there is increasing evidence that positive feedback relationships between buffel grass and fire are leading to long-term changes in vegetation and transformation of ecosystems (Butler & Fairfax 2003; Miller *et al.* 2010; Olsson *et al.* 2012).

Impacts of buffel grass on the richness of native grasses and forbs have been documented (Franks 2002; Jackson 2005; Olsson *et al.* 2012), but there is

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uncertainty about the importance of fire in these interactions. There is particular concern that changed fire regimes caused by the presence of buffel grass are having serious impacts on long lived species, particularly trees occurring in open woodland habitats in semi-arid Australia (Butler & Fairfax 2003; Clarke *et al.* 2005; Miller *et al.* 2010) and cacti in Sonora and Mexico (McDonald & McPherson 2011; Olsson *et al.* 2012). Despite concern about the effects of fire, fuelled by buffel grass, on plant and animal communities, there is still only limited empirical evidence to guide our understanding of the nature and extent of this impact. This is in large part due to the difficultly of setting up suitable studies to investigate interactions between fire and invasive species, particularly in areas where fire is relatively infrequent.

In native vegetation communities in central Australia fire occurrence is highly correlated with rainfall such that high rainfall years, particularly two or more consecutive years of above-average rainfall, result in high risk of fire in the subsequent warm season (Allan 2008; Edwards *et al.* 2008; Griffin *et al.* 1983). In low rainfall years, growth of native grasses and forbs is limited and generally does not result in sufficient fuel to carry widespread fire. As high rainfall years are relatively infrequent, fire return time can be many years, particularly in open woodland and mulga communities, although this may be changing now that buffel grass has colonized extensive areas. In 2010 and 2011, above-average rainfall was experienced in a broad band across the central Australian region, from the Tanami Desert in the north-west to the Simpson Desert in the south-east, in association with one of the strongest La Niña events on record (Bureau of Meteorology 2012). High rainfall was experienced in two consecutive warm seasons and during the intervening cool season, producing excellent conditions for plant growth and associated increases in fuel loads. As a consequence extensive fires were experienced throughout northern central Australia during 2011 across a wide range of native plant communities including areas with buffel grass (see Appendix S1 for a map of the area burnt). Whereas fire in buffel grass is not necessarily limited to above-average rainfall periods, the risk of fire inevitably increases under conditions promoting unusually high fuel loads and when fire is ubiquitous across the landscape.

In those extensive areas of central Australia where buffel grass is well established, reduction of fuel loads around infrastructure is necessary and routine. Most commonly this is achieved by broad scale slashing or herbicide application, or by bulldozing fire breaks, and native vegetation is necessarily reduced or killed along with the buffel grass. Increasingly, the selective removal of buffel grass and restoration of native vegetation in small areas is being attempted for primarily conservation and aesthetic reasons. As more land managers consider whether it is cost-effective to undertake this more labour intensive management, it is becoming increasingly important to understand the benefits for ecological communities.

We have been managing buffel grass in a series of experimental plots, situated within an area heavily infested with buffel grass, since 2008 as part of ongoing research to monitor the effect of removing buffel grass and the subsequent reestablishment of native vegetation on faunal communities. In 2011, two separate fires burnt through six plots (three managed and three unmanaged) at Simpsons Gap in the West MacDonnell National Park. The first fire, in June 2011, which burnt through four plots, was the unplanned result of a control burn. Two other plots burnt in August 2011 as a result of a second, more extensive wildfire. These un-planned fires provided a unique opportunity to compare the outcome of wildfire in areas with and without buffel grass in replicated experimental plots and to evaluate the benefits of removing an invasive grass and restoring native plant communities in relation to the impacts of fire.

Our focus was to report on the spatial pattern of the fires, the impacts on the ground vegetation and on *Hakea divaricata* (fork-leafed corkwood), which is the only long-lived woody perennial occurring within the experimental plots. Corkwood trees (*Hakea* sp.) occur primarily on alluvial flats where buffel grass tends to be most successful and in central Australia, along with *Eucalyptus camaldulensis* (river red gum), are considered among the main local tree species that are directly threatened by increases in intensity and frequency of fire in areas where buffel grass has invaded.

Our specific aim was to compare our pre-existing managed (native plant) and unmanaged (buffel grass) experimental plots in relation to (i) the spatial pattern of the two unplanned fires, (ii) the effect of the fires on ground vegetation, and (iii) the burn severity of *H. divaricata*.

METHODS

Sites

Our study was located at Simpsons Gap in the West MacDonnell National Park (23°43′S, 133°43′E) near Alice Springs in central Australia. Prior to the 1970s the area was extensively used for cattle grazing and was considered to be in a severely degraded state (Clarke *et al.* 2005). The area has been free from grazing for the last four decades during which time buffel grass has progressively colonized. Buffel grass was already present at low abundance (<5%) in the mid 1970s and had reached abundances that regularly exceeded 80% by 2004 (Clarke *et al.* 2005).

The experimental sites were located on a rich alluvial plain adjacent to a significant drainage line dominated by

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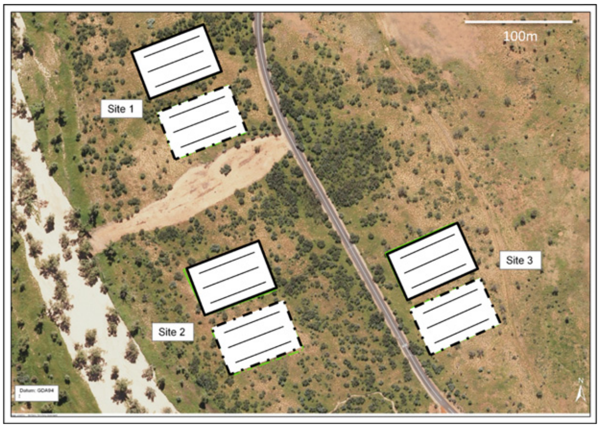

Fig. 1. Layout of experimental plots at Simpsons Gap showing paired managed (solid outline) and unmanaged (dashed outline) plots. The position of three permanently located transects used to survey ground vegetation are shown within each plot. A fire break is visible between sites 1 and 2 running between the road and the dry river bed to the west. Imagery sourced from NT Land Information System, Department of Lands and Planning, 2011.

E. camaldulensis. Woody perennial vegetation at the sites consisted primarily of *H. divaricata* and *Acacia victoriae* (elegant wattle), a relatively short-lived (10–15 years) shrubby *Acacia*. The experimental layout comprised three pairs of plots with each plot measuring 50 m × 70 m. Three plots were managed to remove buffel grass and three were left unmanaged as controls (Fig. 1).

The most recent previous burn at the sites was in 2001, 10 years prior to the fires that we report on here. Local accounts suggest that buffel grass was at a very high density prior to the 2001 fire and that the burn was intense, and many *H. divaricata* were severely burnt as a result (C. Day, pers. comm., 2008).

Management of buffel grass and restoration of native vegetation

The first stage of management, undertaken in February 2008, was by mechanical means. Ground vegetation in managed plots was mown using a slasher attached to a small tractor. Cut vegetation was predominantly buffel grass interspersed with a variety of native grasses and forbs. Trees and shrubs (including seedlings, if detected) were avoided if possible, and large logs and substantial quantities of dead wood lying on the ground were also circumnavigated as they were considered to be valuable habitat for native fauna.

The second, chemical, stage of management was commenced in December 2008 after 2 months of above-average rainfall resulted in sufficient regrowth of buffel grass at the slashed sites for herbicide application to be effective. Spot application of herbicide (glycophosphate 10% solution) with backpack spray kits enabled the specific targeting of buffel grass and minimized any effect on regenerating native species.

Spot spraying of buffel grass at managed sites continued in an opportunistic manner from 2009 to 2012 with the aim of maintaining sites in as buffel-free a state as possible. The ongoing herbicide treatment was conducted by the park rangers at Simpsons Gap who monitored the sites and made decisions about when to apply treatment. Generally this was after bouts of substantial rain but, during 2011, when grass was in active growth most of the time, treatment was applied at intervals and when the Park staff could fit the work into their schedules.

We relied on the *in situ* seed bank for restoration of native vegetation. The high rainfall in 2010 and 2011 resulted in considerable regeneration of native plants at the managed sites.

Ground vegetation

The cover and composition of ground vegetation was measured at managed and unmanaged plots pre fire (November 2010) and approximately 3 and 16 months post fire (October 2011 and November 2012). Above-average rainfall occurred in each month after the fire until March 2012 stimulating a post-fire response and some recovery of vegetation before our initial post-fire survey in Oct 2011 and for several months afterward. Very dry conditions from May–October 2011 probably limited the opportunity for further growth over the six months prior to our final survey. Surveys were conducted using three permanently located 50 m transects in each plot (Fig. 1) using the line intersection method. All plants intercepted by the transect were identified to genus level (grasses) or species level (most other species) and percentage cover for each taxon was calculated.

We used PERMANOVA to test for pre-fire differences in the cover of buffel grass, the cover of native plant species and total plant cover between managed and unmanaged experimental plots to determine the effectiveness of management and provide an indirect comparison of fuel loads. To determine how the fires affected the richness of the ground vegetation over the short and medium term we used two separate PERMANOVAs to compare managed and unmanaged sites before and after the fire using data from surveys in October 2011 and November 2011, approximately 3 and 16 months after the fires. In PERMANOVA probabilities are generated by permutation and normality of data is not assumed or required (Anderson *et al.* 2008). The analyses were conducted with 999 permutations in PERMANOVA+ add-in for PRIMER (Anderson *et al.* 2008). Results were regarded as significant for $P < 0.05$.

Hakea divaricata

In 2007, all *H. divaricata* present within the experimental plots were located and mapped using a GPS. All *H. divaricata* within the plots were identified within two months after each of the 2011 fires to determine whether any trees had died and whether any new trees were present.

The impact of the fires on individual *H. divaricata* was assessed in March (sites 1 and 2) and June 2012 (site 3) approximately 9 months after each burn. Re-sprouting stimulated by post-fire rainfall was evident on most burnt trees after this time interval. For each tree we noted whether

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Table 1. Visual burn severity index, adapted from Miller *et al.* (2010)

0	No apparent recent burn
1	<25% leaf scorching or, if tree <1 m, some burn damage
2	>25% leaf scorching or, if tree <1 m, all foliage scorched
3	All foliage scorched and much foliage missing; some fine fuel <3 mm burnt above 1 m height; bark markedly charred
4	Fuel 3 mm to 1 cm consumed below 1 m height; most fine fuel burnt above 1 m height; bark partially consumed
5	Heavy burn; most minor stems <2 cm consumed below 1 m height; some minor stems consumed above 1 m height; holes burnt in wood
6	Very intense burn; some major stems >2 cm above 1 m consumed; severe burning of major trunk
7	>25% of major trunk consumed but some re-sprouting evident
8	>25% of major trunk consumed and no re-sprouting evident or entire trunk consumed

the tree appeared dead (no regrowth or completely consumed) or alive (regrowth apparent). If alive the mode of post-fire persistence was categorized as (i) original foliage present, (ii) basal and epicormic re-sprouting only, and (iii) basal re-sprouting only (following Miller *et al.* 2010). We used the assessment criteria developed by Miller *et al.* (2010) to further categorize the severity of burn on individual trees from a scale of 0–7 (no burn to >25% of main trunk burnt) but we added a further category (8), where there was no resprouting or the entire trunk was consumed by fire (Table 1).

Mapping of burnt and unburnt areas

The June fire was small in extent, approximately 2km^2, and we were able to map the entire fire scar. Mapping was conducted in August 2011, 2 months after the fire using a Trimble ProXT receiver, connected via Bluetooth to a Trimble Ranger handheld computer. The total area of the fire scar was mapped by walking around the scar perimeter with the GPS logging locations at one second intervals. All unburnt patches within the main fire scar were mapped. The second fire burnt a much larger area (approx. 1100 km^2) so, instead of mapping the entire burnt area, we mapped unburnt patches in the two plots at site 3. This mapping was undertaken in September 2011, several weeks after the fire.

GPS field data collection files were post-processed using correction files from the Continually Operating Reference System, managed by Geoscience Australia, located in Alice Springs, to correct for atmospheric disturbances and produce maps with an accuracy of approximately 50 cm. Mapping data were then imported into ArcGIS (ESRI) for further analysis. The unburnt areas were clipped to the extents of the survey plots and the burnt area in each plot was calculated as a percentage of total plot area.

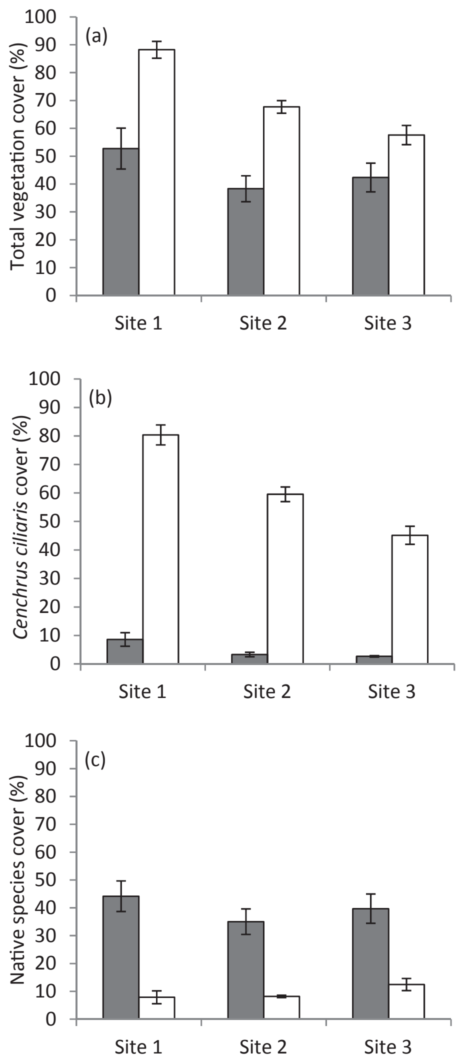

Fig. 2. Mean ± SE of (a) cover of all ground vegetation, (b) buffel grass cover and (c) cover of native species in managed (grey) and unmanaged (white) sites in November 2010, prior to the fires.

RESULTS

Ground vegetation prior to fires

Ground vegetation cover in November 2010, approximately 6 months before the first fire, was about 30% higher at unmanaged plots compared to managed plots (Fig. 2a, Table 2). There were also differences in cover among replicate sites, with site 1 having higher cover than sites 2 and 3 (PERMANOVA pairwise tests, $P < 0.01$ for both).

As expected, buffel grass cover was very low on managed plots but was the dominant cover type at unmanaged plots (Fig. 2b, Table 2). A small amount of buffel grass remained at the managed plots, particularly at site 1, comprising either tussocks that had

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Table 2. Effects of management on cover of all ground vegetation, *Cenchrus ciliaris* and native plants prior to fires (in November 2010) as identified by permutational ANOVA with management and site as fixed effects

		All ground vegetation		*Cenchrus ciliaris*		Native plants	
	d.f.	Pseudo-F	P	Pseudo-F	P	Pseudo-F	P
Management	1, 24	51.96	0.001**	707.3	0.001**	108.4	0.001**
Site	2, 24	13.40	0.003**	32.28	0.001**	0.9531	NS
Management × site	2, 24	3.469	NS	7.564	0.007**	1.201	NS

**$P < 0.01$. NS, not significant.

already been treated with herbicide and were dead or dying, or tussocks that were missed during herbicide application. There were also differences in buffel grass cover between each of the three sites (Fig. 2b, PERMANOVA pairwise tests $P < 0.01$ for all three site comparisons), and there was a significant management × site interaction (Table 2), indicating that site differences were not consistent among treatments and reflecting the relatively high variation in buffel grass cover, from 45% to 80%, among unmanaged plots and less variation among the three managed plots. When buffel grass was excluded, the cover of all other species combined (native species) was significantly higher at managed sites than at unmanaged sites (Fig. 2c, Table 2).

As well as differences in cover, there were differences in structure between native species and buffel grass, which we did not measure quantitatively (see Appendix S2 for details).

Fire characteristics

On the 15th and 16th June 2011 a series of small prescribed patch burns were lit in the vicinity of the Simpsons Gap Ranger Station buildings (just south of sites 1 and 2). One of these patch burns, lit in the late afternoon on the 16th June, extended down into the dry creek bed, to the west of the sites, through heavy buffel and couch grass (*Cynodon dactylon*) and was contained by a wet line on the northern edge. It is believed that this fire smouldered overnight in heavy litter at the edge of the creek bed and then burnt to the north and through sites 1 and 2 the following afternoon. Dense grass along the river bank provided connectivity across the fire break between sites 1 and 2 (Fig. 1).

The second fire which burnt through site 3 started on 9 August 2011. This fire was an extensive, unplanned wildfire, which continued burning through the next two weeks (Grant Allan, pers. comm., 2012). On 10 August the fire travelled north along the eastern side of the road (Fig. 1). Comparative weather conditions (data from Bureau of Meteorology) for the days when each fire occurred are given Table 3. Compared with conditions for the June fire, temperature and wind speed were higher in August while relative humidity was lower (Table 3).

Table 3. Weather conditions for fires at Simpsons Gap in 2011

	Fire 1 (June)	Fire 2 (August)
Min Temperature (°C)	0.9	9.7
Max Temperature (°C)	20.7	21.5
Median wind speed AM (km h^{-1})	7.4	16.7
Median wind speed PM (km h^{-1})	17.6	20.38
Predominant wind direction	W tending S to SE	SW
Max gusts (km h^{-1})	32	35
RH (%) 9.00 hours	58	35
RH (%) 15.00 hours	21	20

Median wind speeds calculated from data from 6.00–11.30 hours (AM) and 12.00–17.30 hours (PM). RH is relative humidity.

Area burnt and patchiness of fire

The June fire burnt the area between the road and the river fairly continuously except for the managed plots which remained largely unburnt (Fig. 3), the fire break, and a small number of other patches. Eighty-eight per cent of the managed plot at site 1 and 80% of the managed plot at site 2 remained unburnt. In contrast the two unmanaged plots were heavily burnt with only 14% of the unmanaged plot at site 1 remaining unburnt and there were no unburnt areas in the unmanaged plot at site 2. Within the broader burnt area there was no evidence of small unburnt patches other than those mapped (Fig. 3).

The August fire burned through both the managed and unmanaged plots at site 3, but there were small patches of unburnt ground within the broader burnt area, mainly in the managed plot. These unburnt patches comprised 17% of the total ground area in the managed plot, compared with only 1% of the unmanaged plot which had fewer and much smaller unburnt patches (Fig. 4).

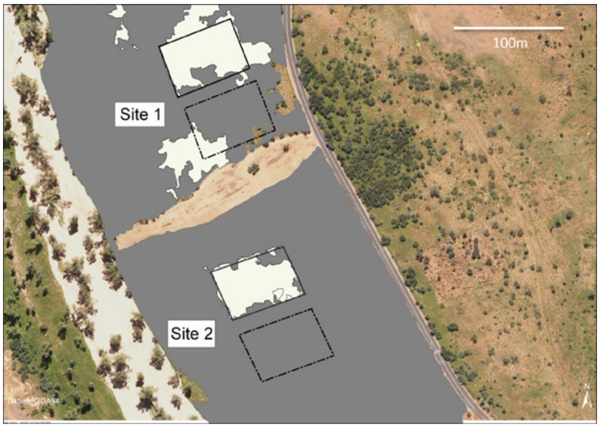

Fig. 3. Area burnt in June 2011 (dark shading) and unburnt patches (light shading) in the vicinity of the experimental plots. Large areas of managed plots (solid outline) remained unburnt compared to unmanaged plots (dashed outline) and the surrounding area. Imagery sourced from the NT Land Information System, Department of Lands and Planning, 2011.

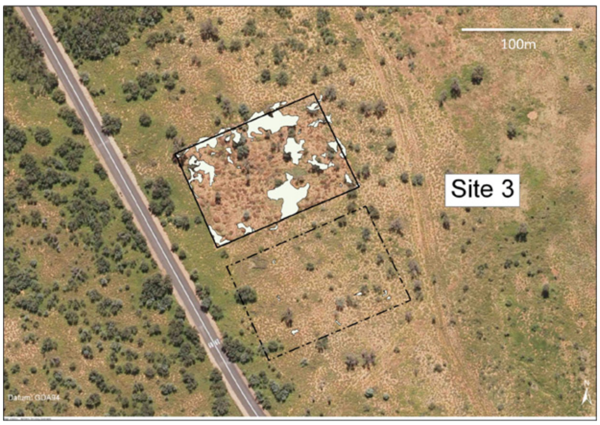

Fig. 4. Managed (solid outline) and unmanaged plots (dashed outline) at site 3 showing patches of ground within the plots that were not burnt (light shading). Imagery sourced from NT Land Information System, Department of Lands and Planning, 2011.

Effects of fire on richness of taxa and cover within ground layer vegetation

In both the short and medium term, management and fire had significant effects on the richness of taxa in the ground layer vegetation with higher richness at managed plots and lower richness after the fires (Table 4, Fig. 5). The significant interaction between management and fire in both analyses indicated that the effect of fire on taxa richness differed in managed and unmanaged sites. Pairwise comparisons showed that there were significant differences in richness between managed and unmanaged sites both before and 3 months after the fire ($t = 4.8856$, $P < 0.01$; $t = 6.8802$, $P < 0.01$) and after 16 months ($t = 9.6383$, $P < 0.01$), but the fires caused a greater reduction in species richness at unmanaged sites (Fig. 5). In the two managed plots that remained largely unburnt (sites 1 and 2 managed) the richness of taxa remained relatively stable over time whereas there was an initial drop in richness at all burnt sites, including the managed plot at site 3 (Fig. 5b). However, 13 taxa were still recorded at the managed plot at site 3 after the fire compared to a maximum of 6 taxa on plots with buffel grass (Fig. 5b). Approximately 16 months after the fire, richness at the managed plot that burnt (site 3) had increased whereas richness at the unmanaged sites remained low (Fig. 5c).

Effect of fire on *Hakea divaricata*

The majority of *H. divaricata* in managed plots retained some or all of their original foliage after the fires whereas very few *H. divaricata* at unmanaged plots retained original foliage and a large proportion only persisted through basal re-sprouting or died (Table 5). Visual assessments of burn severity highlighted the contrasting effect of the fires in managed and unmanaged plots (Fig. 6), with *H. divaricata* in managed plots either less severely burnt compared with unmanaged plots (August fire) or not burnt at all (June fire). At unmanaged plots trees were more severely burnt in the June fire compared to the August fire (Fig. 6).

Because there were only three *H. divaricata* present at site 1 and larger numbers present at sites 2 and 3 and because the conditions differed between fires, formal analyses were not undertaken on the burn severity indices. Only one new *H. divaricata* was detected at the sites in 2011 when compared to the trees originally mapped in 2007.

DISCUSSION

Buffel grass is now widely distributed in semi-arid Australia and it is becoming increasingly relevant to assess the benefits of restoring habitat, in the extensive areas where preventing invasion is no longer an option. In this context manipulative weed removal studies are particularly useful, because they enable attribution of observed effects directly to weed control. The differences in the extent of fire and patchiness of burn at different plots in our study can be directly attributed to buffel grass management. Prior to the fires reported on here, management at our sites had substantially reduced the cover of buffel grass and enabled the cover of native species to increase, but with overall cover at managed plots remaining significantly less than in surrounding areas.

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Table 4. Effects of management and fire on the richness of plant taxa approximately 3 and 16 months after fire as identified by permutational ANOVA with management, site and fire as fixed effects

		Approx. 3 months post fire		Approx. 16 months post fire	
	d.f.	Pseudo-F	P	Pseudo-F	P
Management	1	71.13	0.001**	106.1	0.001**
Site	2	4.030	0.023*	6.403	0.006**
Fire	1	58.45	0.001**	96.46	0.001**
Management × site	2	3.284	NS	1.338	NS
Management × fire	1	6.328	0.020*	11.85	0.002**
Site × fire	1	2.418	NS	2.244	NS
Management × site × fire	1	0.9755	NS	0.9750	NS

*$P < 0.05$, **$P < 0.01$. NS, not significant.

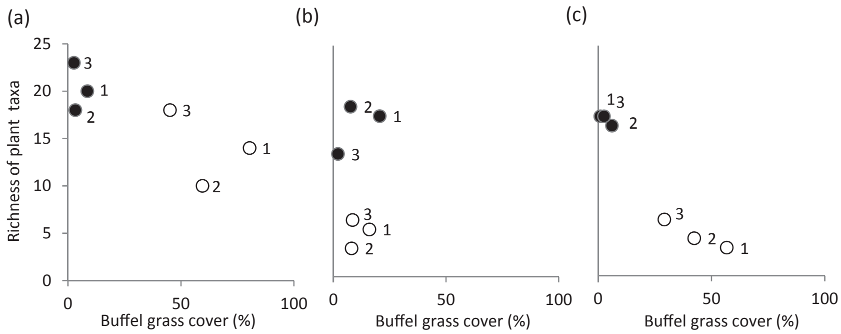

Fig. 5. Relationship between the number of taxa recorded within each plot and average buffel grass cover (a) before fire, (b) approximately 3 months after fire, and (c) approximately 16 months post fire (data from Nov 2010, Oct 2011 and Nov 2012 respectively). Site numbers are labelled and colours differentiate managed (black) and unmanaged (white) sites.

Table 5. Mode of persistence of individual *Hakea divaricata* after the 2011 fires

	O	EB	B	NE
S1 managed	2			
S2 managed	8			
S3 managed	7		1	3
S1 unmanaged	–			1
S2 unmanaged	–	3	7	5
S3 unmanaged	2	1	12	1

Categories are: O – retained original foliage, EB – epicormic and basal re-sprouting, B – basal re-sprouting only, NE – no live foliage evident (includes individuals that were entirely combusted).

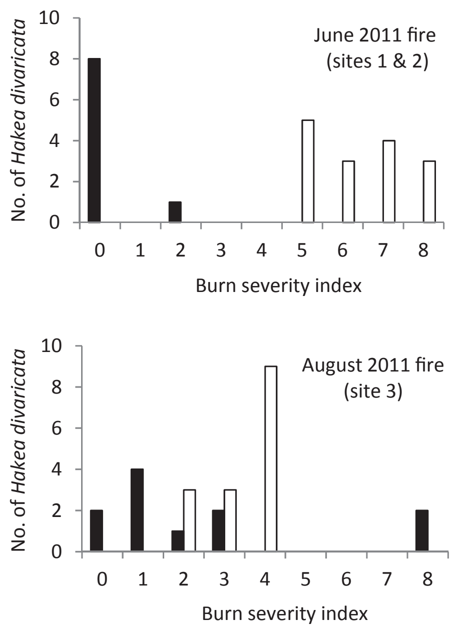

Fig. 6. Burn severity index for *Hakea divaricata* at managed (black) and unmanaged (white) sites after the June and August fires based on a visual assessments of burn severity (ranging from 0 = no burn to 8 = complete combustion) 9 months after each burn.

The lower cover of ground vegetation overall at managed sites prior to the fires together with differences in density of fuel contained in buffel grass tussocks compared with native grass tussocks, most probably account for the observed differences in how managed and unmanaged plots burnt. Low rainfall and cool temperatures would have promoted further curing of fuel between the two fire events and, in combination with higher night-time temperatures and slightly higher wind speeds on the day of the August

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fire, this probably explains why the managed plot exposed to the August fire burnt and the managed plots exposed to the June fire remained largely unburnt. The lower buffel grass cover at the unmanaged plot at site 3 compared to the other unmanaged plots may explain why *H. divaricata* in this plot suffered lower burn severity compared to trees burnt in June at sites 1 and 2, and reduced residence times associated with higher winds speeds (McDonald & McPherson 2011) may also have been a factor. Our study was not designed to enable detailed comparisons of fire characteristics as the occurrence of fire at the experimental plots was opportunistic; instead it provides two examples of the effects of fire, under different conditions, in managed (native vegetation) and unmanaged (buffel dominated) areas. Based on weather conditions we can assume that both of the fires at our study sites had a fairly low intensity compared to what could have happened, under hotter, dryer and windier conditions.

Effects of buffel grass and fire on plant communities

Pre-fire comparisons of the richness of ground layer plant taxa at our managed and unmanaged sites are consistent with strong negative relationships between buffel grass cover and plant species richness that have been previously documented (Franks 2002; Jackson 2005; Eyre *et al.* 2009; McDonald & McPherson 2011; Olsson *et al.* 2012). Buffel grass invasion has been shown to cause the decline of native species richness, even in the absence of fire (Olsson *et al.* 2012), with initial decrease in richness of native grasses and forbs likely to be due to competition (Clarke *et al.* 2005) and changed grass fire cycles becoming progressively more important at later stages of invasion (Olsson *et al.* 2012). McDonald and McPherson (2011) recorded a large drop in species richness after fires in areas with high buffel grass, from 26 to 5 species, but it is difficult to interpret these results without comparative data for burnt areas without buffel grass or information on longer term effects after rainfall has stimulated post-fire recovery. At our sites there was a distinct drop in the number of taxa recorded in unmanaged (buffel grass) plots after the fire and pre-existing differences in the richness of taxa between managed and unmanaged sites were amplified (Fig. 5). These negative effects of fire in areas with buffel grass had persisted 16 months after the fires. Above-average rainfall post fire, until March 2011, enabled substantial recovery of native vegetation at burnt areas in managed sites but not at unmanaged sites. Meanwhile, buffel grass at unmanaged sites had attained cover levels almost as high as those recorded before the fires. Our data provide evidence that fire accelerates direct negative competitive effects of buffel grass on native grass and herb species at least in the short term. Continued monitoring at the sites is required to determine whether this effect persists in the long term and to better understand the positive feedback relationships between buffel grass and fire.

Buffel grass fire interactions also pose a serious threat to longer lived perennial vegetation in the overstorey. The fires at our experimental sites had a greater effect on *H. divaricata* in unmanaged sites, firstly because many of the individuals in the managed sites were not subjected to fire at all, and secondly because the damage to individual trees as indicated by their mode of recovery post fire and visually estimated severity of burn was less in sites with predominantly native vegetation cover. Miller *et al.* (2010) also reported increased burn severity of woodland overstorey species with increasing buffel grass. As burn intensity is correlated with the mortality of woodland overstorey species (Miller *et al.* 2010) buffel grass fuelled fire is likely to lead to increased mortality of trees in open woodland habitat. Over 20% of trees at our sites that were subject to any burn, whether in managed or unmanaged sites, were completely burnt or, if sections of trunk remained, showed no evidence of re-sprouting and we assume these trees died during the 2011 fires. This is in contrast to the high (>94%) post-fire survivorship of *Hakea*s reported by Miller *et al.* (2010). Other studies have shown that that future recruitment of woody canopy species can be hindered by areas of dense buffel grass (Fairfax & Fensham 2000; Franks 2002). In such areas, it is likely that the long-term viability of some woodland overstorey species that are not able to tolerate frequent fire, including *H. divaricata*, is threatened, unless buffel grass is managed to reduce fuel loads.

In Arizona and Sonora there are parallel issues as fire is introduced to environments from which is has been virtually absent. For example McDonald and McPherson (2011) observed extensive damage to cacti tissue as a result of buffel grass fuelled fires and Olsson *et al.* (2012) found that saguaro (*Carnegiea gigantean*) in areas with dense buffel grass had an age structure skewed toward adults, which may relate to inhibition of recruitment or increased mortality of juveniles post invasion. Similar effects of buffel grass on size distribution of columnar cactus (*Pachycereus pectin-aboriginum*) have been reported (Morales-Romero & Molina-Freaner 2008). In north-western Mexico, large areas of desert scrub and thorn scrub are being converted to buffel grass pastures for cattle production and recent practice is to leave some trees and columnar cacti intact during the clearing process, but Morales-Romero and Molina-Freaner (2008) found that juvenile cacti

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are not able to survive in dense pasture, suggesting populations in these areas will not be able to persist long term.

Management of buffel grass

Biological invasions by exotic grasses present issues of global significance as major drivers of ecosystem change, particularly through alteration of fire regimes (D'Antonio & Vitousek 1992). Buffel grass is among the most widespread and problematic of these species, and options for management of the grass at both local and landscape scales need to be considered. However, there continue to be considerable challenges involved with considering management at a broad scale, or even preventing further deliberate spread, because of the value of buffel grass to the pastoral industry (Marshall *et al*. 2012). These controversies, characteristic of many deliberately introduced grasses that have subsequently become invasive (Grice *et al*. 2012), are common to regions worldwide where buffel grass has been introduced. For example, in Mexico areas converted to buffel grass pasture are important for cattle ranching but the grass has spread over extensive areas beyond the converted areas, creating direct conflict between economic and conservation interests (De la Barrera 2008). In Australia, as elsewhere, contention about management is most likely in areas with high value for both grazing and conservation (Friedel *et al*. 2011). Marshall *et al*. (2011) emphasize that the social dimensions of buffel grass management, particularly effects on pastoralists, need to be considered. However, the social implications of increasingly hazardous fires and the costs of fire mitigation are becoming increasingly important considerations in some areas. Pastoralists are among the stakeholders at risk from these changes in fire regimes; in terms of potential loss of life or property, resources required to fight fires that are more extensive, more frequent, and more difficult to fight than previously, and from the temporary loss of pasture that may result from extensive uncontrollable fires. The need for a national approach to address the issues surrounding buffel grass in Australia is discussed by Grice *et al*. (2012). We suggest that interactions between buffel grass and fire, which are becoming much more widely recognized within the general community, may facilitate a new level of consensus among stakeholders giving impetus to the management of buffel grass to simultaneously achieve economic, social and conservation benefits.

Management to protect infrastructure and, in some cases, to increase biodiversity is currently being undertaken on small scales (often only a few hectares) across all major land tenures in central Australia including privately owned semi-urban land, in Parks and Reserves, on Aboriginal land and to a lesser extent on pastoral leases. Our research will better inform land managers by providing experimental evidence of the benefits of undertaking management in relation to how fire is likely to behave, and of associated positive effects on native plant communities.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article at the publisher's web-site:

Appendix S1. Areas of northern Australia burnt in 2011.

Appendix S2. Photos of managed and unmanaged plots at site 3.

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