



Effects of local-scale management on herbaceous plant communities in Brigalow (*Acacia harpophylla*) agroecosystems of southern Queensland, Australia

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ABSTRACT

Remnant vegetation in agricultural landscapes is assumed to contain the majority of biodiversity, although few Australian studies have considered the contribution made by the surrounding production matrix. In this study, herbaceous plant communities from Brigalow (*Acacia harpophylla*) remnants and adjoining areas of the matrix were sampled to determine differences in plant species composition at the core and edge of four distinct land management categories. Nineteen sites consisting of Brigalow remnant and adjacent matrix were selected in the Brigalow Belt Bioregion of southern Queensland, Australia. This region has undergone recent agricultural development relative to the older, well studied landscapes of western Europe. Edge and core transects in four different land management categories were sampled for herbaceous plant species richness, composition and cover. Brigalow remnants contained significantly higher mean herbaceous richness than all other land management categories, and cultivated areas contained the lowest. Species richness, cover and composition did not differ between previously cultivated and uncultivated grasslands. Unlike other studies, there were no detectable edge effects for plant species richness, herbaceous cover and composition in any of the land management categories. Biodiversity conservation in agricultural landscapes in Australia should maintain a focus on the remnant vegetation components; however, our results suggest that there is the potential for enhancing plant biodiversity and ecological functioning by targeted management in modified grassland communities. Management and restoration efforts to conserve agroecosystem plant biodiversity should therefore integrate a range of different landscape elements.

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1. Introduction

Modern agricultural landscapes in Australia, and in many other parts of the world, typically evolve along a gradient of increasing anthropogenic modification and structural simplification, from natural vegetation, through pastoralism, to an intensively managed mosaic of monocultures (Matson et al., 1997; McIntyre and Hobbs, 1999). Such agricultural intensification is often accompanied by declining native plant species diversity and changes in community composition (McLaughlin and Mineau, 1995; McIntyre and Martin, 2001; Fedoroff et al., 2005) and associated impacts on ecosystem functioning. These changes may be caused by the combined effects of local-scale disturbances such as vegetation clearing and recurrent soil disturbance (Chalmers et al., 2005), and by broader-scale processes such as habitat fragmentation and consequent population isolation and edge effects (Dauber et al., 2003; Fahrig, 2003;

Tscharntke et al., 2005). Local plant assemblages are thus influenced by both current and historic land use across a range of spatial and temporal scales (Koerner et al., 1997; Lunt and Spooner, 2005; Reitalu et al., 2009).

The broad-scale clearing and alteration of native vegetation to support agricultural expansion has been linked to local and regional declines in native biodiversity (Saunders et al., 1991; Bennett and Ford, 1997; Fahrig, 2003). In Australia, such transformation has occurred relatively recently, compared with other systems (e.g. Western Europe), where the extent and intensity of modification is typically higher and where it is recognised that the resulting agricultural matrix plays a greater role in biodiversity conservation (e.g. Attwood et al., 2009). The majority of Australian ecological studies in fragmented agricultural landscapes have been confined to patches of relatively undisturbed, native remnant (pre-European) vegetation. Comparatively few studies in Australia have examined the predominant matrix of cleared and disturbed cropping and semi-native and derived grazing lands typical of agroecosystems. Those that have examined the 'production matrix', focus mainly on plant community responses to grazing in native grasslands and grassy woodlands (e.g. Lunt, 1997; Fensham, 1998; McIntyre

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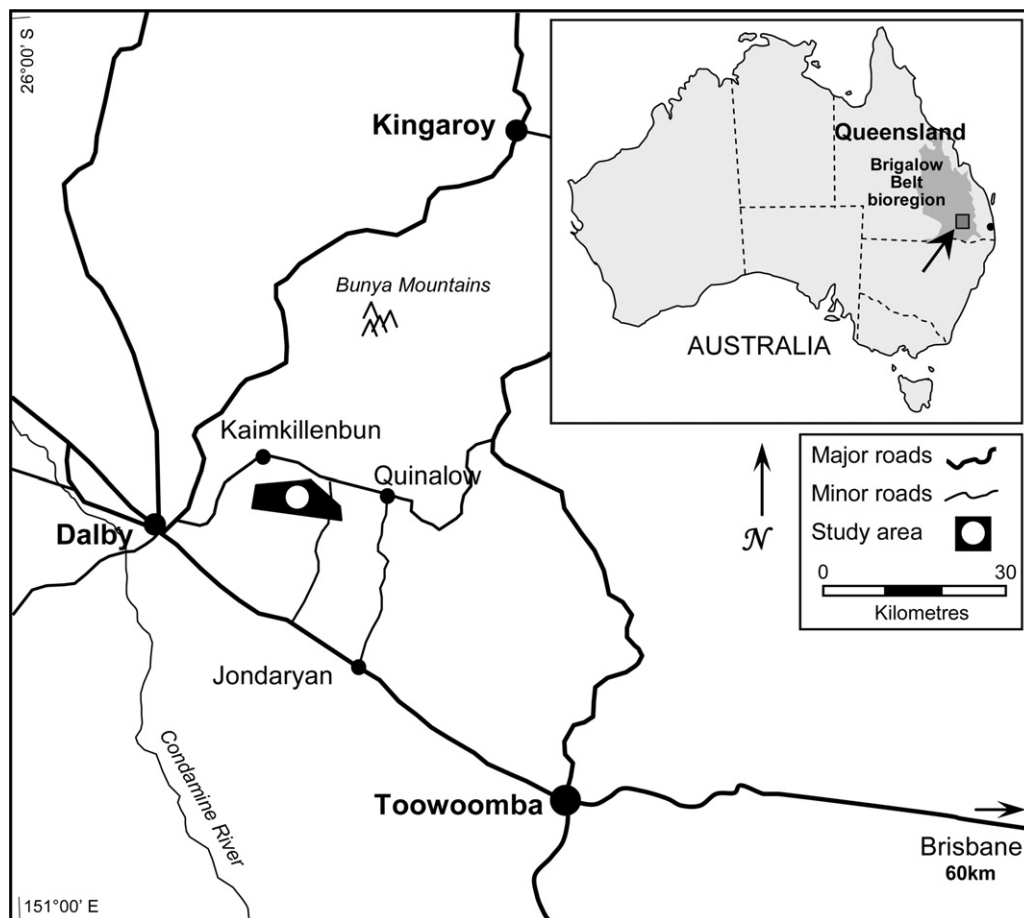


Fig. 1. Map of Queensland, Australia, showing the Brigalow Belt Bioregion and location of the study area in south eastern Queensland.

and Martin, 2002; Lewis et al., 2008). Despite the predominance of more intensively managed cropping and grazing components in these landscapes, very few ecological studies have considered their potential contribution to agroecosystem biodiversity and ecosystem functioning (although see McIntyre and Martin, 2001; Chalmers et al., 2005; Fischer et al., 2005; McIntyre et al., 2005; Collard et al., 2009; Le Brocq et al., 2009).

Changes in land use create new edges between native and managed ecosystems (Saunders et al., 1991; Forman, 1995). These edges are typically abrupt, although they may exhibit environmental and compositional gradients on either side of the edge (Turton and Sexton, 1996; Fox et al., 1997). Edges of native vegetation remnants may be adversely affected by a range of factors including, micro-climate effects, such as exposure to higher levels of wind and solar radiation (Matlack, 1993), competition from ruderal plant species (Murcia, 1995; Fox et al., 1997), and fertiliser runoff and herbicide drift (Kleijn and Snoeijs, 1997; Boutin and Jobin, 1998). As patches of vegetation reduce in size, external disturbances are more likely to affect a greater proportion of the patch (Forman, 1995). Hence, the plant species composition in small remnant patches may be influenced by their proximity to and the nature of the surrounding agricultural matrix (i.e. landscape context) (Usher, 1987). In small linear patches typical of agricultural landscapes, this can have a significant impact on the overall remnant integrity and species assemblages (Forman, 1995).

In this study, we compare herbaceous plant communities (forbs and grasses) in four different landscape elements (land management categories) along a gradient of agricultural intensification in the Brigalow Belt Bioregion of southern Queensland. Land management categories comprise continuous cropping and cultivation

(Cultivation), regenerating grassland that has previously been cultivated (Grass_p), uncultivated grassland (Grass_u) and largely intact remnant brigalow vegetation (Brigalow). In addition, we compare plant communities sampled at the core and edge of patches of remnant Brigalow vegetation with those from the core and edge of adjacent managed grasslands and cultivation (the matrix) to determine whether there are differences in herbaceous plant communities both between and within land management categories. Research findings are discussed in the broader context of biodiversity conservation and ecological functioning of agroecosystems.

2. Methods

2.1. Site and vegetation description

Field sampling was conducted in the Eastern Darling Downs province of the Brigalow Belt Bioregion of southern Queensland, Australia. The nearest major town to the study sites is Dalby (Lat. 27°18'S, Long. 151°26'E) which lies 340 m above sea level (Fig. 1) and receives an average annual rainfall of 676 mm, mostly during the summer months (Bureau of Meteorology, 2005). Mixed farming systems with summer and winter cropping and limited cattle grazing are the primary land uses (Harris et al., 1999).

Prior to European settlement, the landscape was largely covered with mixed Brigalow (*Acacia harpophylla*) vegetation communities (Sattler and Williams, 1999). This vegetation occurs naturally in deep, fertile and well structured cracking clays (vertisols), usually in lowland areas (Sattler and Williams, 1999). Remnant brigalow vegetation in the study area typically consists of a mix of *A. harpophylla* (brigalow) and *Casuarina cristata* (belah) as canopy species,



Fig. 2. Aerial photograph of the study landscape taken in 2001. Brigalow study sites are outlined and numbered.

Geijera parviflora (wilga) in the understorey, and a mixed assemblage of mainly perennial grasses and forbs as groundcover (e.g. *Einadia nutans*, *Enchylaena tomentosa* and *Paspalidium criniforme*) (Sattler and Williams, 1999). Brigalow lands were favoured by European settlers in the late 1800s/early 1900s for clearing and conversion to crops and pasture (Johnson, 1964).

Extensive areas of the Brigalow Belt Bioregion have undergone recent land clearing (Wilson et al., 2000). However, the amount of original Brigalow vegetation in the study area has remained virtually unchanged at about 6% since the 1970s. Most of the original native woody vegetation in the study area was cleared by European settlers in the 1940s, often by deep ripping and burning to overcome the problem of persistent Brigalow resprouting. Some of the cleared land was immediately used for cropping; however, large areas have regenerated naturally to form derived semi-native grasslands (Johnson, 1964). Since the 1940s, much of this derived grassland has been gradually converted to intensively managed cropping lands and today, only 26% of non-cropping land (consisting of grassland and Brigalow vegetation) remains in the landscape (Fig. 2).

2.2. Study design

Study sites were selected within and surrounding 19 remnant patches of Brigalow (*A. harpophylla*) and Belah (*C. cristata*) shrubby open forest on cracking Cainozoic clay soils (Regional Ecosystem 11.9.5, Sattler and Williams, 1999). The size of Brigalow patches sampled ranged from 2 to 33 hectares, however, most were smaller than 10 ha. Adjacent areas of grassland were typically less than 5 ha. Cropping was often continuous across large areas of the landscape and therefore patch sizes were not determined.

Adjacent matrix sites were further divided into three categories according to past and current land use practices, derived from aerial photography, landholder accounts of management history and field observations. The resulting four land management categories were:

- (1) Remnant Brigalow vegetation that has never been cleared ('Brigalow', $n = 19$). Vegetation was often subjected to sporadic, low intensity grazing and occasional timber harvesting
- (2) Grassland that has not been cultivated following clearing of the original Brigalow vegetation in the 1940s, with no deliberate addition of improved pasture species or fertiliser ('Grass_U', $n = 3$). Includes grassland subjected to frequent (8–12 months/year) medium intensity cattle grazing and occasional spot spraying for woody weeds such as boxthorn
- (3) Grassland that has been previously cultivated but left fallow for 3–15 years ('Grass_P', $n = 4$). Includes regenerating grassland grazed frequently (8–12 months/year), usually at medium intensity by cattle, and occasionally fertilised and sprayed with herbicide to control woody weeds.

- (4) Recently (<2 yrs) cultivated land ('Cultivation', $n = 12$). The majority of cultivated sites were managed using zero or minimum tillage, with frequent (>twice/year) herbicide application and annual fertiliser application. The most commonly grown crops were sorghum, millet and wheat with occasional, low intensity grazing of stubble.

Both grassland categories were uncommon in the landscape, particularly in areas adjoining Brigalow, leading to a small number of available samples and high internal variation in management regimes of selected grassland sites. Where present, these grassland sites were preferentially selected over the more common cropping elements. Landholder surveys revealed that grazing was generally infrequent and at low intensity across all land management categories, with grassland areas potentially subjected to more intensive grazing by cattle at certain times, relative to adjacent brigalow and cropping areas.

Treatments at each study site were: Brigalow remnant core (BC), Brigalow remnant edge (BE), Matrix edge (ME), and Matrix core (MC). BC was located at the centroid of each of the 19 Brigalow patches and the other treatments (BE, ME and MC) were positioned along a transect that extended from BC into the adjacent production matrix (Fig. 3). The distance between ME to MC was either proportional to that between BC and BE, or a minimum of 100 m.

2.3. Vegetation sampling

Vegetation sampling was conducted once at each site between 25th February and 7th April 2003 at the core and edge of each of the 19 Brigalow patches and at the core and edge of the adjacent grassland or cultivated area. Belt transects (10 m × 50 m) were established at the core and edge of the Brigalow and in the adjacent matrix, parallel to the linear edge of each remnant. Ten (1 m × 1 m) quadrats were positioned at 5 m intervals along the transect, at a randomly generated distance (up to 5 m) on either side of the centre line of the transect. Ground cover species (woody and herbaceous species <50 cm in height) within the quadrats were identified to species where possible, and classified as native or non-native following the nomenclature of Stanley and Ross (1983). In each quadrat, the percent projected foliage cover of each herbaceous plant species was estimated—this estimate included plants that were rooted outside the quadrat, but were projecting over the quadrat area.

Each plant species was categorized using the following life history attributes according to the literature (Stanley and Ross, 1983; Henderson, 1997) and from field observations: (1) herb type – monocots (grasses and sedges) or dicots (herbaceous forbs and woody dicotyledons) <50 cm; (2) origin – exotic (species introduced from an area outside its natural range) or native and (3) annual (including biennial) or perennial species. The average per-

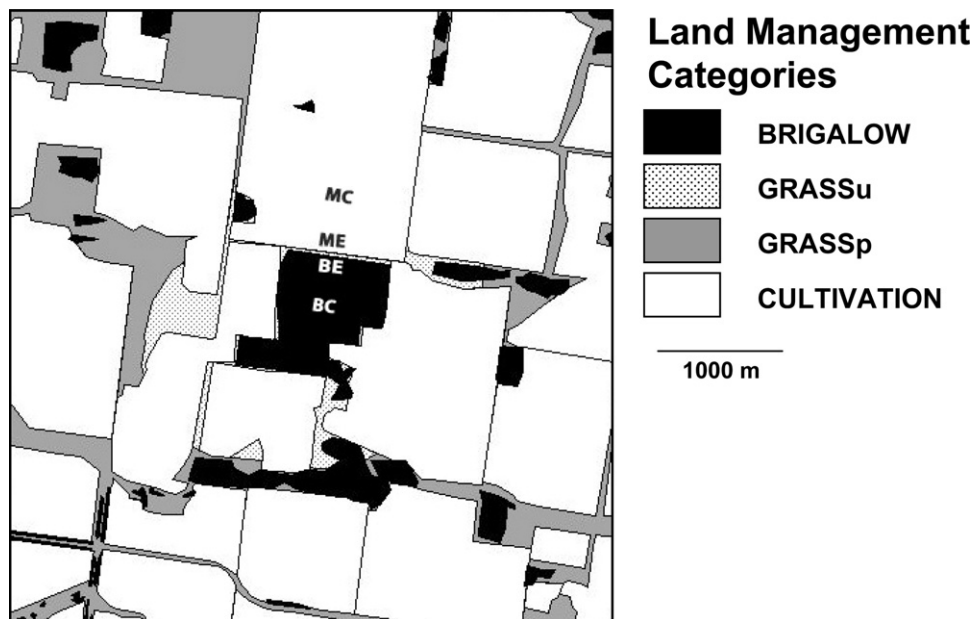


Fig. 3. Schematic of a study site and the surrounding landscape showing treatments at Brigalow core (BC), Brigalow edge (BE), matrix core (MC) and matrix edge (ME).

centage cover of each species across the 10 quadrats in the belt transect was used in all analyses.

2.4. Statistical analyses

Two-factor crossed analysis of variance (ANOVA) using SPSS version 12.0.1 (SPSS Inc., 2003) was used to compare means of total, native and exotic species richness (cumulative across the 10 quadrats in each site and log transformed) and percentage cover (averaged across ten 1 m × 1 m quadrats and arc-sine root transformed) across core and edge samples and land management category (Sokal and Rohlf, 1995). Residual plots and Levene's test (Levene, 1960) indicated homogeneity of variance in all tested variables. Differences within factors that were significant in the ANOVA were tested using *a priori* contrasts with Bonferroni adjustments (Bonferroni, 1936). Species richness and cover data from the matrix core sample in site 9 (a Grass_p site) were identified as outliers in residual plots (both before and after transformation) and were omitted from all further analyses.

Patterns in herbaceous community composition were investigated using PRIMER for Windows Version 5.2 (PRIME-E Ltd, 2002). The frequency, out of 10 quadrats, and mean percentage cover of each plant species were used for these analyses. Similarity matrices were created using the Bray–Curtis similarity coefficient on square root transformed data. Non-metric multi-dimensional scaling ordinations (Kruskal and Wish, 1978) and two-way crossed Analysis of Similarity (ANOSIM; Clarke and Warwick, 2001) were used to determine differences in plant community composition between core and edge and between land management categories. Similarity percentage analyses (SIMPER; Clarke and Warwick, 2001) were used to determine the contribution of particular species to patterns of similarity within, and dissimilarity between, land management categories. Species that contributed up to 20% of the similarity are presented here.

3. Results

3.1. General floristic patterns

A total of 148 herbaceous plant species was recorded across all study sites. Of these, 29 species were annual dicots, 14 species

were annual monocots, 68 species were perennial dicots and 37 species were perennial monocots (Appendix A). Forty-four species (28 dicots and 16 monocots) were exotic species. One grass species, *Homophilis belsonii*, listed as rare and threatened in the eastern Darling Downs province by Sattler and Williams (1999), was recorded in 11 Brigalow sites (Appendix A).

Brigalow remnants contained 118 (80%) of the total species recorded, with 36 of these species unique to this land management category. Two species, *Bothriochloa decipiens* (native) and *Zinnia peruviana* (exotic), were recorded in only Grass_u sites and 8 species (7 native and 1 exotic, *Digitaria eriantha*) were restricted to Grass_p sites. Fifty-one species were recorded in Cultivation sites, 15 of which (4 native and 11 exotic) were unique to this land management category. Thirteen species (9 native and 4 exotic) were common to all four land management categories. Sixty-three species (43%) were recorded only in sites with no history of cultivation (i.e. in Brigalow or Grass_u sites).

3.2. Edge effects and land management categories

Mean native species richness ranged from 25 species per 10 m² at Brigalow edge and Brigalow core to 3 species in the same area at Cultivation core. Mean exotic species richness ranged from 6 at Grass_p core to 3 at Brigalow core. Mean total species richness ranged from 30 at Brigalow edge to 8 at Cultivation core. Native cover ranged from 64% at Grass_p core to 6% at Cultivation core. Exotic cover ranged from 29% at Grass_u core to 3% at Brigalow core. Total cover ranged from 75% at Grass_p core to 19% at Cultivation core.

There were no significant differences (2-factor ANOVA, $p > 0.05$, d.f. 1, 75) between core and edge samples across all land management categories for total ($F=1.6$), native ($F=1.1$) and exotic ($F=0.6$) species richness and total ($F=0.1$), native ($F=0.3$) and exotic ($F=0.6$) herbaceous cover. No interaction between main effects was detected (2-factor ANOVA, $p > 0.05$, d.f. = 3, 75) for total ($F=0.2$), native ($F=0.3$) and exotic ($F=0.7$) herbaceous cover.

Significant differences existed between land management categories for total, native and exotic species richness and for total, native and exotic percent cover (Fig. 4). Mean total and native species richness were significantly higher in Brigalow and significantly lower in cultivation than for the other land management

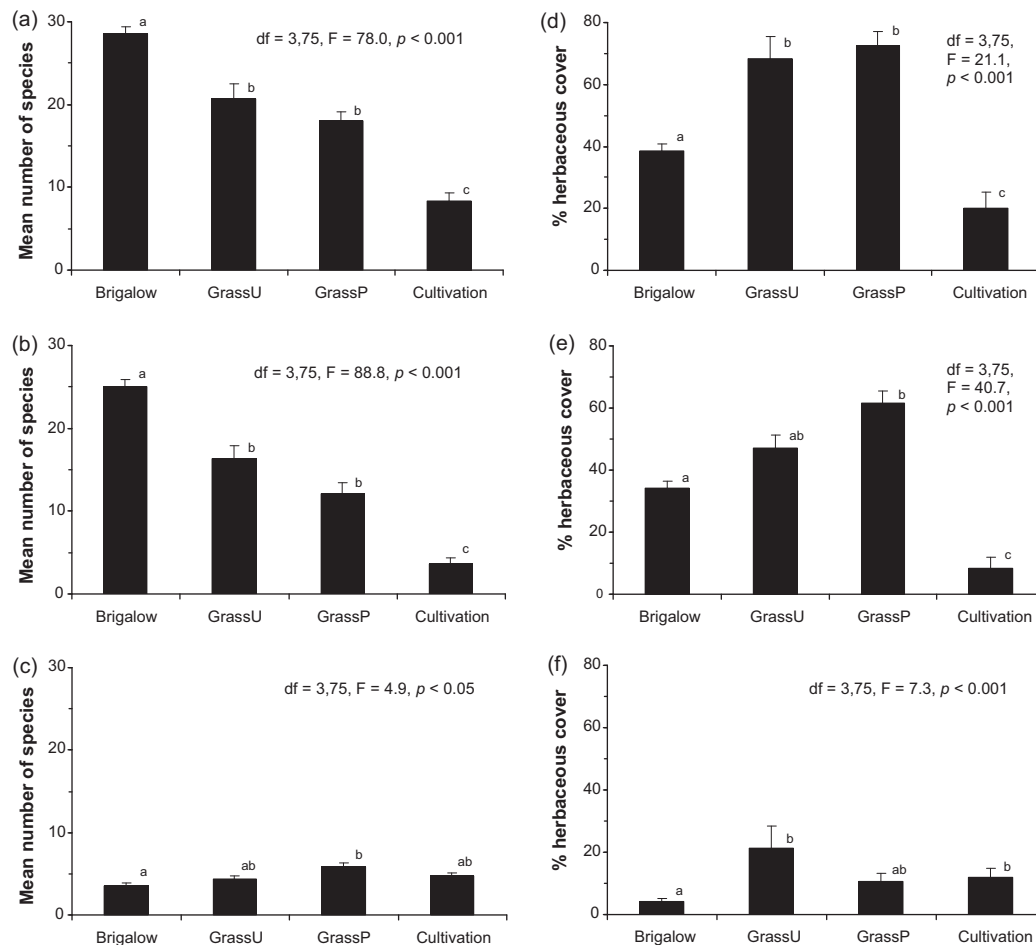


Fig. 4. Mean plant species richness (left) and percentage herbaceous plant cover (right) for total (a and d), native (b and e) and exotic (c and f) species in Brigalow, uncultivated grassland (Grass_U), previously cultivated grassland (Grass_P) and current cultivation (Cultivation). Means sharing the same letter are not significantly different (*a priori* contrasts $p > 0.05$). Species richness data are edge and core data pooled and untransformed. Cover data are edge and core data pooled and arcsine-root transformed. Error bars are standard errors.

categories (*a priori* contrasts, $p < 0.05$, Fig. 4a and b). There were no differences in total, native or exotic species richness between Grass_U and Grass_P ($p > 0.05$). Mean exotic species richness was significantly higher in Grass_P sites than in Brigalow but was not significantly different between other land management categories (Fig. 4c).

Mean total and native groundcover were significantly lower in Cultivation than for the other land management categories, and significantly higher in Grass_U and Grass_P than in Brigalow or Cultivation (*a priori* contrasts $p < 0.05$; Fig. 4d and e). Exotic cover was significantly lower in Brigalow than in Grass_U and Cultivation sites, but not significantly different from exotic cover in Grass_P sites. There were no significant differences in exotic cover between any of the matrix land management categories (Grass_U, Grass_P and Cultivation; $p > 0.05$; Fig. 4f).

An nMDS ordination of plant frequency data showed distinct separation between land management categories. A gradient in floristic composition from Brigalow sites (on the right of the diagram) through grassland (Grass_U and Grass_P) to cultivation (on the left) is evident on the ordination (Fig. 5). There was considerable overlap of core and edge samples within all land management categories, while cultivation samples were highly variable compositionally, despite similar management practices across sites (Fig. 5).

Two-way crossed Analysis of Similarity (ANOSIM) confirmed no significant differences in herbaceous plant community composition between core and edge across all land management

categories (Global $R = 0.056$, $p = 0.05$). However, there were significant differences between land management categories (Global $R = 0.776$, $p = 0.001$). Pairwise tests showed significant differences ($p < 0.05$) between all land management category combinations except Grass_U and Grass_P (Table 1).

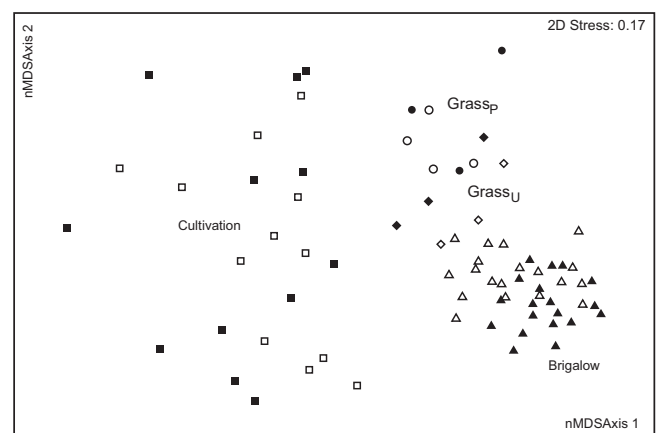


Fig. 5. Two-dimensional ordination (nMDS) of plant frequency data showing core (shaded) and edge (open) within each land management category. (▲) Brigalow core, (△) Brigalow edge, (◆) Grass_U core, (◇) Grass_U edge, (●) Grass_P core, (○) Grass_P edge, (■) Cultivation core, (□) Cultivation edge.

Table 1

Summary of Analysis of Similarity (ANOSIM) on plant frequency data for land management categories main effect (Global $R=0.776$, Sign. level = 0.001).

	Brigalow	GRASS _U	GRASS _P
GRASS _U	0.741**	–	–
GRASS _P	0.940**	0.123 ^{ns}	–
CULT	0.877**	0.265*	0.398**

Notes: Values in table are R values (from pairwise tests) with significance level indicated. ANOSIM R statistics fall between –1 and 1, with $R=0$ indicating completely random grouping and $R=1$ indicates that all replicates of a category are more similar to each other than to any replicates of another category. A significant Global R indicates that there are differences between categories somewhere in the analysis. Significant R values indicate that the R value is significantly different from zero (** $p < 0.001$, * $p < 0.05$, ^{ns} = not significant). Significant R values for individual category comparisons indicate that the compared categories are significantly dissimilar from one another.

Average Bray–Curtis dissimilarities and the average abundance of species that contributed up to 20% of the dissimilarity between land management categories are provided in Table 2. Differences were caused largely by the high abundance of species in one land management category and the low abundance or absence of these species in the other land management category (Table 2).

Dichanthium sericeum (Queensland Bluegrass) was mostly recorded in Grass_P sites. The annual exotic grass *Urochloa panicoides* (Liverseed Grass) showed high abundance in uncultivated grassland (Grass_U) sites and had a large influence on dissimilarity in all Grass_U and Grass_P comparisons. *Cynodon dactylon* (Couch grass) had a strong influence on dissimilarity in all comparisons and was found in high abundance in Brigalow, Grass_U and Grass_P sites (Table 2).

4. Discussion

4.1. Responses of plant communities to land management

Differences in herbaceous plant species richness and composition between land management categories were evident along a gradient of intensification from remnant brigalow, through grassland, to recent cultivation. Consistent with other studies (e.g. Tilman, 1997; McIntyre and Martin, 2001), total and native plant species richness decreased with increasing intensity of agricultural management. As expected, cultivated areas that were subjected to continual disturbances such as tillage and herbicide application, had significantly lower total and native richness than any of the other land management categories, and made only a small relative contribution to overall native plant diversity. Similarly, Chalmers et al. (2005) found that total plant species richness was lower under high levels of soil disturbance than moderate and low levels, and that the abundance of most native perennial species decreased with high soil disturbance. Other studies also show that soil disturbance has measurable negative effects on native plant species richness and cover, and leads to decreases in native perennial species and increases in the richness and cover of exotic annuals (Fensham, 1998; McIntyre and Lavorel, 1994). In particular, Fensham (1998) noted a transition from native to exotic colonisation (e.g. *Chloris virgata* and *Eragrostis cilianensis*) with increasing mechanical disturbance in roadside strips. In our study, highly disturbed Cultivation areas were characterised by the abundant annual exotic monocot *Urochloa panicoides*. However, exotic species richness did not vary significantly across three of the four land management categories (Brigalow, Grass_U and Cultivation), a similar finding to that of McIntyre and Martin (2001).

Remnant brigalow vegetation in the studied agricultural landscape contained higher herbaceous plant species richness than cleared/uncultivated grassland (Grass_U). This is contrary to other Australian studies in grassy eucalypt woodlands in the trop-

Table 2

Average dissimilarity between land management categories (LMC) and the average abundance of plant species that contributed up to 20% of the dissimilarity between LMCs.

Species ^a	Average abundance ^b		Cumulative (%) ^c
	Brigalow	Cultivation	
Average dissimilarity = 94.22			
<i>Einadia nutans</i>	4.74	0	4.72
<i>Tetragonia tetragonioides</i>	5.47	0.96	9.42
<i>Paspalidium criniforme</i>	4.71	0	14.05
<i>Sclerolaena tetracuspis</i>	3.71	0	17.73
<i>Cynodon dactylon</i>	3.39	0.67	21.24
Species	Average abundance		Cumulative (%)
	Brigalow	Grass _U	
Average dissimilarity = 76.21			
<i>Urochloa panicoides</i>	0.87	5.83	5.11
<i>Cynodon dactylon</i>	3.39	7.67	9.98
<i>Tetragonia tetragonioides</i>	5.47	0.83	14.64
<i>Paspalidium criniforme</i>	4.71	0.33	18.99
<i>Sclerolaena tetracuspis</i>	3.71	1.33	22.24
Species	Average abundance		Cumulative (%)
	Cultivation	Grass _U	
Average dissimilarity = 87.69			
<i>Cynodon dactylon</i>	0.67	7.67	9.91
<i>Urochloa panicoides</i>	2.58	5.83	16.31
<i>Portulaca oleracea</i>	2.88	3.5	20.95
Species	Average abundance		Cumulative (%)
	Brigalow	Grass _P	
Average dissimilarity = 85.43			
<i>Dichanthium sericeum</i>	0	8.29	7.02
<i>Paspalidium criniforme</i>	4.71	0	10.94
<i>Tetragonia tetragonioides</i>	5.47	1.29	14.83
<i>Urochloa panicoides</i>	0.87	4.57	18.31
<i>Cynodon dactylon</i>	3.39	4.43	21.59
Species	Average abundance		Cumulative (%)
	Cultivation	Grass _P	
Average dissimilarity = 91.20			
<i>Dichanthium sericeum</i>	0.04	8.29	10.38
<i>Cynodon dactylon</i>	0.67	4.43	15.93
<i>Urochloa panicoides</i>	2.58	4.57	21.05

^a Species are listed in decreasing order of their importance in discriminating the two sets of samples.

^b Average abundance represents the average presence of species per plot within each land management category.

^c Cumulative % represents the cumulative influence of the species to the overall Bray–Curtis dissimilarity.

ics and sub-tropics in Queensland, which have shown that tree removal without cultivation, can enhance (McIvor, 1998), or not affect, species richness (Fairfax and Fensham, 2000; McIntyre and Martin, 2001; Le Brocque et al., 2009). However, the lower herbaceous species richness in our grassland study sites may be due to soil disturbances associated with historical tree clearing practices such as deep ripping and burning to suppress persistent brigalow regrowth (Johnson, 1964). This clearing may have caused the reversion of plant communities to early stages of succession, allowing competitive exotic species to colonise and potentially affecting shade-tolerant species of the original brigalow ecosystems (Scanlan, 1991) and thus, long-term species composition. However, other studies in the Darling Downs have noted that threats to native grassy communities caused by competition with exotic species may not be as severe as in temperate systems elsewhere in Australia (Fensham, 1998; McIntyre and Martin, 2001).

Agricultural 'de-intensification' (*sensu* Diamond, 2005) of ex-arable lands provides opportunities for the regeneration of semi-native ('novel') plant communities (Hobbs et al., 2006). Results from previously cultivated grasslands (Grass_P) provide clear evidence of regeneration of herbaceous semi-native plant communities following cultivation. These novel communities showed no detectable differences in species richness or community composition when compared with uncultivated grasslands (Grass_U). This rapid floristic convergence contrasts with the lack of recovery shown by soil carbon and bird communities in the same landscape (Collard and Zammit, 2006; Collard et al., 2009) and suggests a general floristic self-assembly to a native grassland, and potential restoration of ecosystem function. Although long-term agricultural abandonment is unlikely in this landscape, the formation of temporary novel assemblages of native plant species is likely to enhance overall plant diversity and ecosystem heterogeneity at a range of spatial scales.

In the previously cultivated grassland (Grass_P), the dominant exotic annual colonisers (e.g. *U. panicoides*) associated with cultivation had been largely replaced by the native perennial grass *D. sericeum* (Queensland bluegrass) and by a range of other native perennials. Other studies have shown a similar transition from exotic to native, as well as an increase in perennial species, with increasing time since disturbance (e.g. Corbet, 1995; Wilcox, 1998). Comparisons of species richness, percentage cover and community composition showed that there were no differences between previously cultivated (Grass_P) and uncultivated (Grass_U) grasslands. However, the SIMPER analyses support field observations that *D. sericeum* was found in much higher abundance in the more disturbed Grass_P sites. Other studies have also shown that native perennial grasses colonise rapidly following cultivation (Robinson et al., 1993; Garden et al., 2001; O'Connor, 2005). For example, Fensham (1998) reported that disturbed ground on basalt and alluvial soils in the Darling Downs is commonly colonised by *D. sericeum*. This species is regarded by graziers as a beneficial grass species, valuable for production (Roberts and Silcock, 1993). The successful regeneration of this grass in ex-arable farmlands may provide farmers with a viable alternative to intensive cropping practices. Further study on the apparent dominance of the native grass *D. sericeum* and potential impacts on the functioning in these novel systems and interactions with other components of biodiversity would be beneficial.

4.2. Edge effects

Many ecological studies have shown patterns of increased species richness at habitat edges (Angelstam, 1992; Risser, 1995). Edges often contain higher species diversity than either of the two adjacent habitats because of the composite nature of the ecotonal assemblage (Murcia, 1995). However, in this study, there were no detectable differences between core and edge for plant species richness and cover in any of the land management categories. Unlike the higher plant species richness at remnant rainforest edges (Fox et al., 1997; Beer and Fox, 2000), the brigalow remnants had similar plant species richness and community composition at both core and edge.

The observed homogeneity within patches was unexpected, particularly for brigalow edges, where differences in community composition were predicted due to the movement of plant propagules from the matrix across land management boundaries (Fox et al., 1997). An explanation for this may be that the small, disturbed brigalow patches contain 'porous' plant assemblages with similar levels of exotic plant colonisation from edge to core. Proximate factors that might contribute to this include cattle grazing, herbicide drift and the open canopy structure allowing sufficient light to enable ruderal weed colonisation. The alternative hypothesis – that

these patches are resilient to edge effects – is not supported by our evidence. However, due to limited information on the original plant assemblages in the study area and no available undisturbed 'reference' site in the study landscape, the extent of modification of the groundlayer vegetation in study sites compared with the original plant community could not be tested.

5. Conclusions

Agricultural landscapes in many parts of Australia have a recent history of intensification compared with those in other countries, where elements of the agricultural matrix make a proportionately greater contribution to biodiversity. Despite the low sample size for grassland elements, this study clearly demonstrates significant patterns in herbaceous plant communities across a gradient of agricultural intensification. The major contribution to plant biodiversity was made by remnant Brigalow. However, the less intensively managed grassland components can also contribute significantly to plant biodiversity. Homogeneity of plant communities between the core and edge of the main land management categories, particularly remnant vegetation is an atypical feature of this recently developed landscape. To optimise biodiversity outcomes in complex agroecosystems, consideration should be given to both remnant and matrix landscape components.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.agee.2011.05.001.

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