Is fencing enough? The short-term effects of stock exclusion in remnant grassy woodlands in southern NSW

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*This article was prepared by **Peter Spooner**, **Ian Lunt** and **Wayne Robinson** (The Johnstone Centre, Charles Sturt University, Albury, NSW 2640, Australia. Tel: + 61 2 6051 9620, Fax: + 61 2 6051 9897, Email: pspooner@csu.edu.au). It draws on data collected by Peter Spooner for his Honours thesis and was presented to the 2000 meeting of the Ecological Society of Australia, which was awarded the EMR Blackwell Prize for best student paper on a question of relevance to management and restoration.*

Summary Fencing remnant native vegetation has become a widespread activity for arresting declines in biodiversity in agricultural landscapes. However, few data are available on the effectiveness of this approach. The present study investigated the short-term effects of fencing to exclude livestock on dominant tree and shrub recruitment, plant species cover, litter and soil characteristics in remnant grassy woodlands in southern NSW. Vegetation and soil surveys were undertaken at 47 sites fenced by Greening Australia (NSW) for 2–4 years. Fenced and unfenced areas at each site were compared using split-plot sampling. Woodlands sampled were dominated by Yellow Box/Blakely's Red Gum (*Eucalyptus melliodora/Eucalyptus blakelyi*), Grey Box (*Eucalyptus microcarpa*) or White Cypress-pine (*Callitris glaucophylla*). Significantly higher numbers of tree recruits were found in the fenced sites, with tree recruitment found in 59% of fenced sites compared with 13% of unfenced sites. Fenced sites also had significantly greater cover of native perennial grasses, less cover of exotic annual species and less soil surface compaction. However, outcomes varied among woodland ecosystems and individual sites. Where tree recruitment occurred, there was significantly more tree recruitment where there was greater perennial grass cover and less regeneration where exotic annual grass cover or overstorey crown cover was dense. Few shrubs recruited in fenced or unfenced areas, reflecting the lack of mature shrubs in most sites. Fencing is an important first step for conserving threatened grassy woodlands, but more active management may be needed to enhance woodland recovery, particularly in sites where few or no recruits were found.

Key words *bush regeneration, fencing, grazing exclusion, rehabilitation, woodland restoration.*

Introduction

Before European settlement, south-eastern Australia supported extensive stands of grassy woodlands (DEST 1995). Most grassy woodlands occurred on relatively fertile soils and were quickly cleared and replaced with exotic crops and pastures (Robinson & Traill 1996). Remaining woodlands now occur as remnants of varying size, quality and isolation (Saunders *et al*. 1991). In addition to fragmentation effects, almost all woodlands in south-eastern Australia have been altered by stock grazing for up to 150 years, often resulting in substantial degradation (Wilson 1990). Livestock defoliate plants, prevent regeneration, introduce weeds and alter understorey composition and soil conditions (e.g. Pettit *et al*. 1995; Prober & Thiele 1995; Yates *et al*. 2000a). As a result, most woodland remnants are now of high conservation status, but many will continue to degrade unless management is improved (Yates & Hobbs 1997a).

The Greening Australia (NSW Riverina) Fencing Incentive Program (funded by the *Natural Heritage Trust* Program) is a typical example of current efforts to protect and enhance native vegetation on private property. This program provides landholders with up to $1200 per km to fence remnant vegetation, plus management advice (Driver *et al*. 2000). Since 1996, over 9500 hectares of native vegetation in the NSW Murray Catchment have been fenced under the program. While this program has been successful in terms of conservation outputs (e.g. total area secured and changed management practices), little is known of the ecological outcomes of fencing remnant native vegetation. Consequently, the fencing program has been likened to 'a huge experiment' (McDonald 2000; p. 8).

Fencing to exclude stock is widely regarded as a simple restoration method and that a fenced remnant will 'look after itself' (Lamb 1994; Reeves 2000). It is commonly assumed that plant populations will regenerate and remain viable after grazing is excluded. However, grazing exclusion has been found to have variable effects on tree and shrub recruitment, species richness, understorey composition and soil conditions (e.g. Curtis 1990; Scougall *et al*. 1993; Cluff & Semple 1994; Pettit *et al*. 1995; Prober & Thiele 1995). Consequently, the outcomes of excluding grazing from long-grazed woodlands are often difficult to predict (Yates & Hobbs 1997b). To address this uncertainty, the present study investigated the short-term effects of fencing grassy woodlands to exclude domestic stock, with an emphasis on tree and shrub regeneration, vegetation composition and changes in litter and soil conditions.

Materials and Methods

The present study was conducted in grassy woodlands in the southern Riverina and South-western Slopes, in the NSW Murray Catchment. The area has a cool temperate climate and mean annual rainfall ranges from 408 mm at Deniliquin in the west of the region, to 795 mm at Albury in the east. Altitude ranges from 70 m above sea level in the west to 350 m in the east (MCMC 2000).

Grassy woodlands occur across the region in flat to undulating areas on fertile soils, but have now been extensively cleared for agriculture. Four major woodland associations occur: (i) Yellow Box/Blakely's Red Gum (*Eucalyptus melliodora*/*Eucalyptus blakelyi*) woodlands on red or yellow podsolic soils, in areas receiving 480–700 mm mean annual rainfall (for brevity, these woodlands will subsequently be called Yellow Box woodlands; (ii) White Box (*Eucalyptus albens*) woodlands, on a variety of soil types receiving 500–800 mm mean annual rainfall; (iii) Grey Box (*Eucalyptus microcarpa*) woodlands on red-brown earths and heavy clays, with 400–580 mm mean annual rainfall; and (iv) White Cypress-pine (*Callitris glaucophylla*) woodlands, mostly on sandhills, footslopes or rocky outcrops, in areas receiving less than 480 mm mean annual rainfall. River Red Gum (*Eucalyptus camaldulensis*) forests also occur in areas that are regularly flooded and Black Box (*Eucalyptus largiflorens*) occurs in more western areas of the catchment (Moore 1953a; Porteners 1993).

The original composition of woodland ground-layer vegetation is now difficult to determine, but most woodlands are thought to have been dominated by native grasses (including Kangaroo Grass, *Themeda australis*; Poa tussock, *Poa sieberiana*; and spear-grasses, *Stipa* spp.) and herbs, with an open shrub layer (Moore 1953a; Prober & Thiele 1993). Few areas remain with an intact understorey and exotic annual species now dominate many remnants (Moore 1953b; Moore 1973). Little is known of the pre-European density of native shrubs, but shrub density is thought to have been naturally sparse, but with a tendency for increased density from east to west across the region (Moore 1953a). Native shrubs have been greatly depleted by past clearing, stock grazing, rabbits and fire and many species (e.g. *Acacia, Dodonaea, Pultenaea* and *Senna*) are now largely restricted to roadside remnants (Moore 1953a,b).

Site selection

Potential study sites were selected from a database of over 366 sites fenced under the Greening Australia (NSW Riverina) Fencing Incentive Program. No quantitative data were available on site conditions prior to fencing, so a split-plot design was used to assess the effects of stock exclusion, by comparing fenced and unfenced areas at each remnant. The underlying assumption was that the fenced area was in a comparable condition and under the same management regimen as the unfenced area when fences were erected. Fences excluded domestic stock, but not rabbits or native herbivores.

Sites were selected from the Greening Australia database if the following conditions were met: (i) the entire remnant had not been fenced due to various management reasons or site constraints (rather than differing quality of vegetation), thereby strengthening the assumption that the unfenced area represented a reliable control; (ii) the remnant was fenced for a minimum of 2 years (maximum duration possible was 4 years); (iii) vegetation in fenced areas was not disturbed since fencing, and was not regularly grazed by domestic stock; (iv) unfenced areas had been maintained under the same grazing regimen as when fencing was erected, and had not been mechanically disturbed or cropped; and (v) each pair of fenced and unfenced areas had similar vegetation condition, crown cover, size, slope and aspect.

This exhaustive process led to 70 potential study sites selected from a database of over 366 sites fenced by Greening Australia (NSW) in consultation with Greening Australia staff. Landholders were then contacted and a brief telephone questionnaire was carried out to determine management practices since fences were erected, to ensure comparisons could be made and to arrange for the inspection of sites prior to sampling.

Only 47 of the 366 sites fenced by Greening Australia (NSW) satisfied these criteria and were considered suitable for sampling. Of these, 15 were dominated by Yellow Box (totalling 120 ha), 15 by Grey Box (86 ha) and 17 by White Cypress-pine (74 ha). These three woodland associations were more commonly fenced under the Fencing Incentive Program (Driver *et al*. 2000). The selected sites were considered representative of remnants across the catchment and varied in size, shape and landscape position (Fig. 1). However, as Greening Australia has fenced sites of relatively high quality in the region, results from the present study cannot necessarily be interpolated to highly degraded sites.

Greening Australia prioritized high-quality sites (including those with abundant regeneration) over degraded areas when selecting sites for fencing (Driver *et al*. 2000). However, most of our contrasts were within (not between) sites that were selected for fencing by GA. In many study sites, straight fences were erected through irregularly shaped remnants, leaving a portion unfenced. These alignments were typically constrained by the landholder's desire to minimize the area fenced, rather than by the locality of regeneration within the remnant.

Sampling

Quantitative

At each site, a transect was placed in the middle of the fenced and unfenced areas on similar aspects (where possible) in June and July 2000. Transects were equidistant from remnant boundaries, and aligned parallel to the longer boundary. Where overstorey crown cover was spatially variable, transects were placed to be as representative as possible. Each transect was 200 m in length, except in small remnants (< 200 m in length), where a 100-m transect was

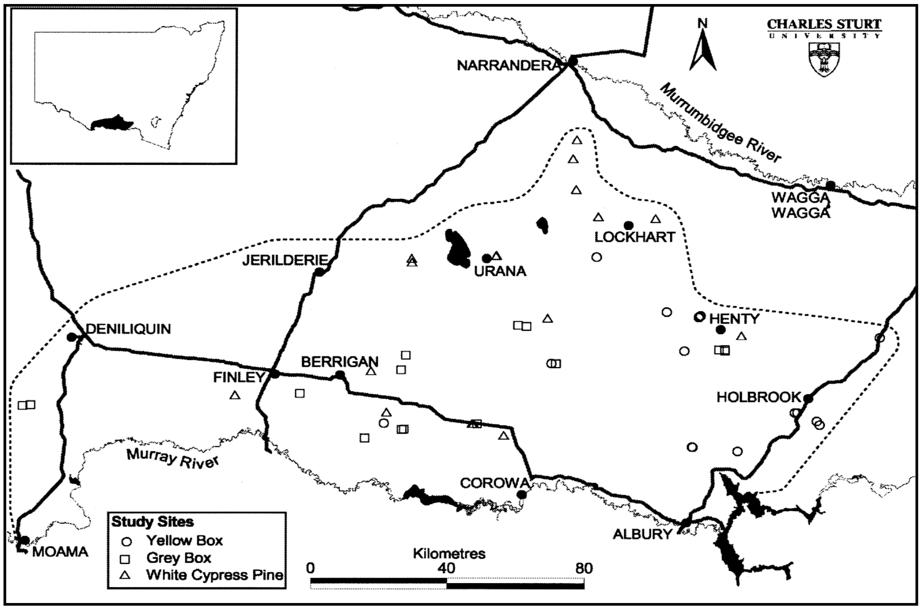

Figure 1. Location of 47 grassy woodland study sites within the NSW Murray catchment. The dashed line indicates the approximate boundary of the study area, with the Murray River forming the southern boundary.

used. A nested plot technique was used to measure tree, shrub and herb strata.

Ten quadrats were arranged at equal intervals along each transect. Within each of these 5 × 5 m quadrats, the percentage cover of trees and shrubs was visually estimated to the nearest 10%. The identity and height (to nearest 0.1 m) of all tree and shrub recruits were also recorded, as was the method of propagation (from seedling or lignotuber). To identify trees that recruited after fencing, only saplings less than 3 m in height were measured, as taller saplings would certainly have established before fences were erected (Curtis 1990). Very small seedlings (< 0.05 m) were ignored.

A 2 × 2 m quadrat was nested within each 5 × 5 m quadrat. Within each 2 × 2 m quadrat, the percentage cover of perennial grasses, annual grasses, non-leguminous herbs, legumes and the three ground-layer species with greatest cover were identified and visually estimated to the nearest 10 per cent, or to the nearest 1% where cover was less than 10%. Tree and shrub cover was also re-measured in 2 × 2 m quadrats (following the methods used in 5 × 5 m quadrats) to provide accurate data to correlate against ground-layer characteristics. The percentage cover of the following ecosystem attributes was also assessed in each 2 × 2 m quadrat: logs (> 10 cm diameter), branches (1–10 cm diameter), leaf litter (< 1 cm diameter), bare ground, exposed rock and undisturbed lichen crust.

At each quadrat, soil surface pH and moisture (1–8 ordinal scale) were recorded using a hand-held electronic pH-moisture meter. Soil surface compaction was assessed by averaging readings taken from the four corners of each 2 × 2 m quadrat using a calibrated 0–500 kPA pocket penetrometer.

Qualitative

Because of patchiness of regeneration within sites, tree and shrub recruitment across the entire remnant was also assessed using an additional qualitative method using three categories: none, scattered or abundant. This helped identify sparse recruitment, which would not have been sampled by the more restrictive quadrat method.

Data analysis

For most analyses, data from all quadrats in each fenced and unfenced area were averaged, to obtain a pair of mean values for each variable at each site. These values were statistically compared between fenced and unfenced areas using pooled data from all three woodland associations, and from each woodland association separately. Friedman's tests were used to perform the non-parametric equivalent of repeated-measures ANOVA based on ranked variables, using the SPSS package (Coakes & Steed 1996; Zar 1996). In most cases, the conservative Friedman's test, combined with many null values and tied rankings, was not sufficiently robust to partition between the many factors sampled. By applying individual Wilcoxon signed rank two-tailed tests to each woodland group and applying a Bonferroni correction factor to avoid Type I error, more powerful testing of pooled data was achieved (Zar 1996). In this manner, a critical value of $P < 0.017$ was considered significant ($P = 0.05/3$ woodlands) for analyses of tree recruitment within woodland associations.

To determine whether tree recruitment variables differed significantly between the three woodland associations (Yellow Box, Grey Box and White Cypress-pine), Kruskal–Wallis rank tests were used with the Bonferroni correction factor ($P < 0.017$). This test was also used to compare cover trends of individual plant species between the three associations.

To identify those ecosystem variables that differed significantly between fenced and unfenced areas in each of the three woodland associations, Wilcoxon signed rank tests were used and a significance value of $P < 0.01$ was used to minimize the chance of Type 1 errors. Correlations between site, tree recruitment and ecosystem attributes were explored using Spearman's non-parametric rank-order correlation coefficient and Bonferroni correction factors applied when all three woodlands were analysed (i.e. $P < 0.017$). Where a large number of variables and, hence, many analyses were conducted within each woodland association, a significance factor of $P < 0.01$ was used to minimize Type 1 errors.

Results

Tree recruitment

Using the central line transect method, tree recruitment was recorded in quadrats in 38% of fenced areas, compared with 11% of unfenced areas. Trees recruited more frequently in fenced Yellow Box woodlands (60% of sites) than fenced Grey Box (40%) or White Cypress-pine (18%) woodlands. In unfenced areas, recruitment was only recorded in quadrats in Yellow Box woodlands (at 33% of sites). In total, 94% of regeneration in fenced areas and 77% in unfenced areas was from seedlings rather than resprouting lignotubers.

On average, tree recruitment was significantly greater ($n = 47$, $z = -3.526$, $P < 0.001$) in fenced areas (82 recruits per ha) than in unfenced areas (11 recruits per ha) for all woodlands combined. There was no statistically significant difference ($P > 0.017$) in recruitment densities between fenced and unfenced areas when each of the three woodland associations was analysed separately. However, strong non-significant trends for increased recruitment were apparent in fenced Yellow Box and Grey Box woodlands, with few trees recruiting in White Cypress-pine woodlands (Fig. 2).

More tree recruitment was recorded using qualitative site assessments than by the central line transect method. Qualitative results showed tree recruitment in 59% of fenced areas and 13% of unfenced areas for all woodlands combined. The difference between the central line transect data and qualitative results was due to clumping of regeneration, especially along remnant boundaries, which were not sampled with the central line transect method (P. Spooner, pers. obs., 2000).

Tree recruitment was significantly greater in areas that had been fenced for longer periods ($n = 47$, $\chi^2 = 8.373$, $P = 0.015$): on average, 228 recruits per ha were recorded in sites fenced for 4 years, compared with 85 recruits per ha in sites fenced for 3 years and 12 recruits per ha in sites fenced for 2 years. There were slight differences between woodland associations in the mean duration of fencing: on average, Yellow Box woodlands had been fenced for 3.0 years, Grey Box woodlands for 2.9 years and White Cypress-pine woodlands for 2.3 years, although these differences were not significantly different ($n = 47$, $\chi^2 = 5.892$, $P = 0.057$).

Tree recruitment density was significantly positively correlated ($P < 0.01$) with perennial grass cover, and negatively correlated with tree crown cover and annual grass cover, when data from all quadrats was pooled (Table 1). Recruitment density was not significantly correlated with any other measured ecosystem variable.

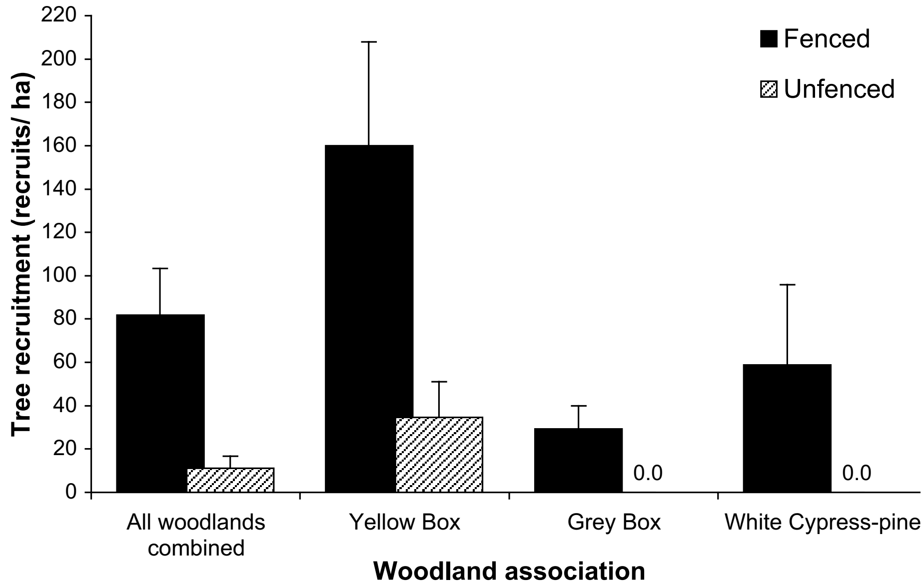

Figure 2. Density of tree recruitment (+ SEM) in fenced (■) and unfenced (▨) areas in the three woodland associations.

Table 1. Spearman's rank correlations between tree recruitment density and vegetation and ecosystem variables, for all fenced woodland quadrats sampled ($n = 470$)

	Tree recruitment density	
Attribute	**Rs**	**p**
Crown cover	– 0.201	< 0.001*
Perennial grass cover	0.208	< 0.001*
Annual grass cover	– 0.141	0.002*
Herb cover	0.014	0.759
Logs	0.008	0.856
Branches	0.031	0.503
Leaf litter	– 0.044	0.340
Bare ground	– 0.016	0.728
Lichen	0.075	0.453
Rocks	– 0.027	0.555
Soil pH	0.079	0.085
Soil moisture	– 0.014	0.756
Soil compaction	– 0.014	0.769

*Significant ($P < 0.01$).

Canopy gaps

Significant correlations were found between overstorey crown cover, tree recruitment and many ecosystem variables in fenced areas ($P < 0.01$; Fig. 3). The percentage cover of perennial grasses, herbs, bare ground and lichen crust, and levels of soil moisture and compaction were all significantly negatively correlated to crown cover. In contrast, the percentage cover of annual grasses, logs, branches and leaf litter were all significantly positively correlated to crown cover (Fig. 3).

Fenced sites where trees recruited had significantly less crown canopy cover ($n = 47$, $z = -2.429$, $p = 0.015$: Fig. 4) and significantly more perennial (native and exotic) grass cover ($n = 47$, $z = -2.743$, $P = 0.006$: Fig. 5) than fenced sites without tree recruitment. Tree recruitment tended to occur more frequently in sites where the average crown cover was < 30% (72% of sites) and no trees recruited in sites with

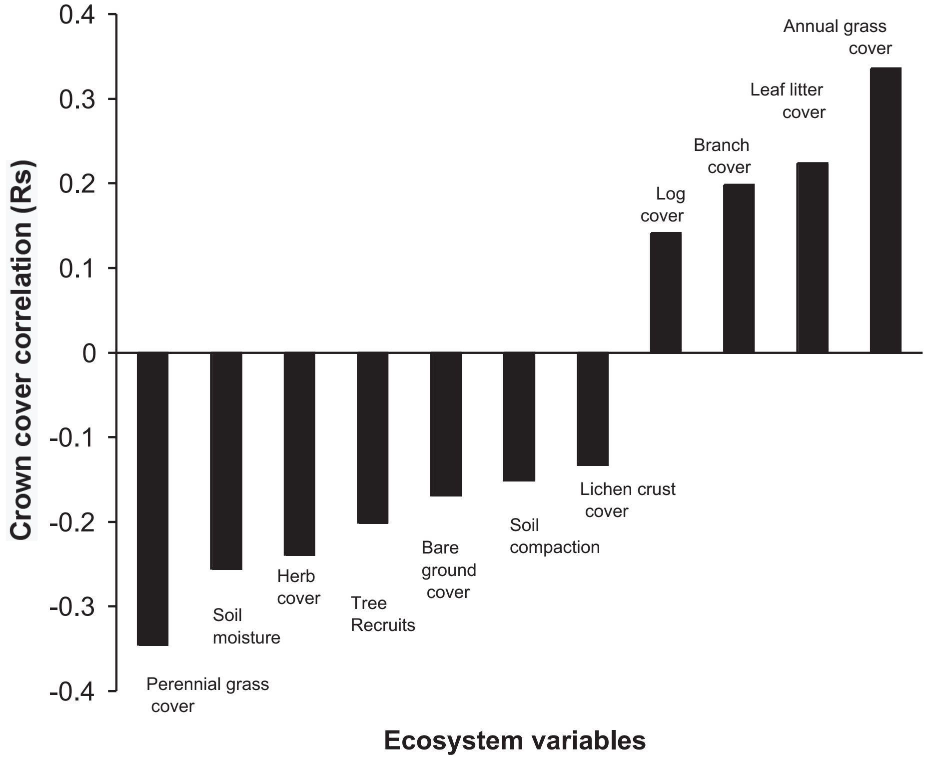

Figure 3. Ecosystem variables that were significantly correlated to percentage crown cover in fenced areas for all woodland associations. Increasing positive or negative values indicates stronger rank correlations (Spearman's Rs).

Figure 4. Mean percentage crown cover of canopy trees (+ SEM) at sites with and without tree recruitment for all woodland associations combined (based on qualitative site assessments).

Figure 5. Mean percentage cover of native and exotic perennial grasses (+ SEM) at sites with and without tree recruitment for all woodland associations combined (based on qualitative site assessments).

> 50% crown cover, although these differences were not statistically significant.

Changes in understorey condition

Scattered shrub recruitment was recorded in seven fenced areas (15% of sites), but no unfenced areas, using the qualitative site assessment method. Similarly, shrub recruitment (all *Acacia* species) was recorded in quadrats in just five fenced sites (11%) and no shrub recruitment was recorded in unfenced areas. Thus, there was a non-significant ($P > 0.017$) trend for greater shrub recruitment in fenced than in unfenced areas.

The cover of perennial grasses (native and exotic) was significantly greater in fenced than in unfenced areas ($n = 47$, $z = -5.026$, $P < 0.001$) for all woodlands combined and in Yellow Box ($n = 15$, $z = -3.010$, $P = 0.003$) and Grey Box woodlands ($n = 15$, $z = -3.107$, $P = 0.002$) when analysed separately (Fig. 6). Conversely, the cover of exotic annual grasses was significantly less in fenced than in unfenced areas ($z = -3.651$, $P < 0.001$) for all woodlands combined, and in Grey Box ($z = -2.669$, $P = 0.008$) woodlands when analysed separately (Fig. 7). The cover of bare ground was also significantly less ($z = -3.043$, $P = 0.002$) in fenced than in unfenced areas, but there was no significant difference in the cover of herbs, legumes, logs, branches, leaf litter, lichen crust and rocks, or soil moisture and pH (Table 2).

The dominant ground-layer species in fenced and unfenced areas were mostly native perennial grasses and exotic annual grasses and forbs (Table 3). Fenced Grey Box woodlands had significantly less cover of the exotic Barley Grass (*Hordeum* spp. $z = -2.480$, $P = 0.013$) and a strong non-significant trend ($P > 0.017$) for greater cover of native perennial Spear Grasses (*Stipa scabra*) and Wallaby Grasses (*Danthonia* spp.) than unfenced areas (Table 3). Soils were significantly less compacted ($z = -5.457$, $P < 0.001$) in fenced than in unfenced areas for all woodlands combined and in Grey Box ($z = -3.233$, $P = 0.001$) and White Cypress-pine woodlands when analysed separately ($n = 17$, $z = -3.621$, $P < 0.001$). A similar, but non-significant trend existed in fenced Yellow Box woodlands ($z = -2.358$, $P = 0.018$: Fig. 8).

Discussion

It is commonly argued that stock exclusion by fencing is a first requirement of remnant woodland management (e.g. Pettit *et al.* 1995; Prober & Thiele 1995). The results of the present study support previous studies that found benefits of grazing exclusion in woodlands on factors including: (i) enhanced tree and shrub regeneration (Curtis 1990; Cluff & Semple 1994; Windsor 1999); (ii) promotion of perennial

Table 2. Mean percentage cover of vegetation attributes and ecosystem attributes, and mean soil attribute values, in fenced and unfenced areas in all woodlands combined, and in the three individual woodland associations

	All woodlands combined			Yellow Box			Grey Box			White Cypress-pine		
	Fenced mean	Unfenced mean	P-value	Fenced mean	Unfenced mean	P-value	Fenced mean	Unfenced mean	P-value	Fenced mean	Unfenced mean	P-value
Perennial grasses	**32.9***	**17.4***	**< 0.001***	**43.9***	**24.1***	**0.003***	**30.3***	**15.5***	**0.002***	25.7	13.3	0.013
Annual grasses	**35.4***	**45.0***	**< 0.001***	24.5	32.4	0.041	**39.6***	**54.0***	**0.008***	41.3	48.3	0.049
Herbs (non-leguminous)	14.4	17.5	0.076	12.5	17.7	0.132	15.4	14.9	0.975	15.4	19.5	0.115
Legumes	0.7	1.5	0.096	1.0	0.9	0.432	0.3	0.7	0.893	0.9	2.9	0.068
Logs	0.4	0.4	0.968	0.9	0.3	0.310	0.4	0.2	1.000	0.1	0.6	0.141
Branches	0.7	1.2	0.047	0.7	1.5	0.018	1.0	0.9	0.726	0.5	1.2	0.173
Leaf litter	11.5	10.2	0.164	14.4	16.3	0.865	9.5	8.0	0.379	10.6	6.8	0.121
Bare ground	**4.5***	**8.0***	**0.002***	3.4	8.2	0.017	3.6	6.6	0.084	6.3	9.1	0.356
Lichen	0.5	0.1	0.036	0.5	0.1	0.108	0.8	0.1	0.176	0.1	0.1	0.581
Rocks	0.0	0.0	0.655	0.1	0.0	0.317	0.0	0.1	0.317	0.0	0.0	1.000
Soil pH	6.2	6.2	0.035	6.4	6.3	0.155	6.2	6.1	0.349	6.2	6.1	0.098
Soil moisture (1–9)	6.7	7.0	0.030	6.6	6.9	0.088	6.5	6.7	0.932	6.9	7.4	0.019
Soil compaction (kPa)	**243.5***	**264.4***	**< 0.001***	218.4	226.6	0.017	**298.0***	**340.3***	**0.001***	**217.8***	**230.7***	**< 0.001***

*$P < 0.01$ is significant difference between fenced and unfenced plots using Wilcoxon signed rank test.

Table 3. Differences in mean percentage cover between fenced and unfenced areas for the 10 most dominant plant species in three woodland associations

	Cover trend index			
Species‡	Yellow Box	Grey Box	White Cypress-pine	Combined woodlands
Hordeum species*	– 9.0	**– 15.0†**	– 4.7	– 9.1
Stipa scabra	4.4	4.4	10.0	5.4
Lolium species*	5.4	2.2	– 5.4	0.5
Danthonia species	0.9	7.5	0.9	3.1
Themeda australis	8.7	0	0	2.8
Stipa species	1.3	0.3	1.3	0.9
Microlaena stipoides	– 1.5	0	0	0.5
*Arctotheca calendula**	– 1.2	– 3.5	– 4.9	– 3.3
*Echium plantagineum**	– 1.8	0.6	– 0.5	– 0.6
Enteropogon acicularis	0.1	2.0	– 0.6	1.5

*Exotic species; †significant at $P < 0.017$ and ‡species listed in order of dominance in fenced areas.

The cover trend index shows the average difference in cover between fenced and unfenced areas. Positive values indicate a greater cover in fenced areas, negative values indicate greater cover in unfenced areas.

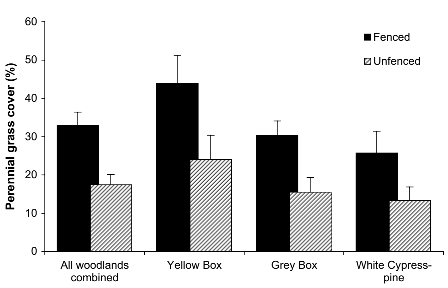

Figure 6. Mean percentage cover (+ SEM) of native and exotic perennial grasses in fenced (■) and unfenced (▨) areas in the three woodland associations.

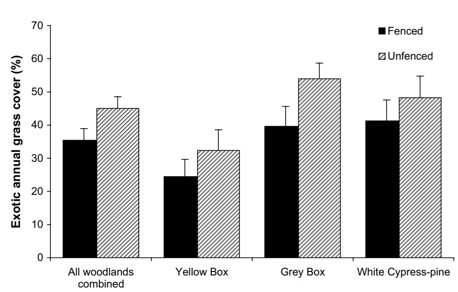

Figure 7. Mean percentage cover (+ SEM) of exotic annual grasses in fenced (■) and unfenced (▨) areas in the three woodland associations.

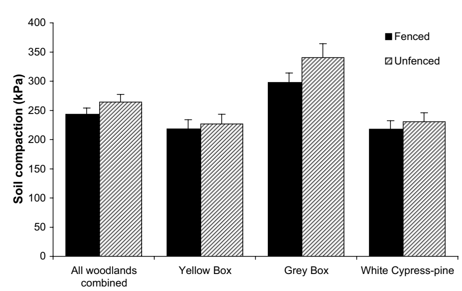

Figure 8. Mean compaction of the upper soil crust (+ SEM) in fenced (■) and unfenced (▨) areas in the three woodland associations.

grasses above exotic annual species and (iii) decreased surface soil compaction (e.g. Scougall *et al*. 1993; Pettit *et al*. 1995; Yates *et al*. 2000a). Thus, fencing appears to have rapidly led to positive environmental outcomes at many sites.

Unfortunately, quantitative regeneration data were not collected before the fences were erected. Consequently, we cannot unequivocally attribute these differences to the fencing program. An alternative explanation is that sites with abundant regeneration were selectively fenced and that abundant regeneration promoted fencing of remnants rather than vice versa. However, despite this design constraint, we are confident that the changes we have described were primarily a response to fencing. Our selection criteria (see Materials and Methods) were designed to minimize the possibility that fenced and unfenced areas were dissimilar before fences were erected. Furthermore, we sampled a large number of sites rather than gathering more detailed information from fewer sites, in order to minimize the possibility that unique site factors (including inappropriate fence-line contrasts) may have contributed to the outcomes.

Clearly, future assessments of fencing effects will be enhanced if detailed ecological data are gathered before fences are erected (e.g. using Before After Control Impact (BACI) designs). In practice, however, there are often insufficient resources to achieve this. Consequently, regional appraisals of fencing effects may continue to rely on careful comparisons between rigorously selected sites. In this case, we believe that our extensive site selection protocol has minimized the chances of confounding cause and effect, thereby allowing us to attribute the observed effects to the fencing treatment.

Regeneration of overstorey species

Despite the significantly higher results in the fenced areas overall, the magnitude and significance of responses varied greatly between the three woodland associations and individual sites. A number of explanations can be offered. First, insufficient time may have elapsed for significant ecological changes to occur (Lamb 1994). The maximum duration of fencing was just 4 years and greater changes might be expected in the future. This suggestion is supported by the significant positive correlation between tree recruitment densities and the time since fencing; the density of tree regeneration in sites fenced for 4 years was almost 20 times that in sites fenced for just 2 years. Furthermore, it is possible that differences in tree recruitment densities between the three woodland ecosystems (Yellow Box > Grey Box > White Cypress-pine) may partly be due to differences in the duration of fencing in each ecosystem (mean = 3.0, 2.9 and 2.3 years, respectively).

Successful tree recruitment depends on the chance occurrence of heavy seed-fall with a suitable seed-bed under amenable weather conditions (Cluff & Semple 1994; Lawrence *et al*. 1998). Consequently, regeneration events can be highly episodic (Lamb 1994; Yates & Hobbs 1997a). For example, Curtis (1990) estimated that suitable conditions for recruitment of woodland eucalypts occur only once every 10–20 years in the Northern Tablelands of NSW. White Cypress-pine regeneration is also markedly episodic, with mass regeneration occurring infrequently and only after successive years of high summer rainfall (Lacey 1972). Thus, the absence of tree recruitment in 40% of sites surveyed, within 4 years of fencing, may simply reflect unsuitable local weather conditions rather than intrinsic site problems.

It is also possible that ecological barriers may be preventing regeneration at some sites despite the removal of the major disturbance agent, domestic stock (Whalley 1994). A number of site-specific ecological factors (such as lack of canopy gaps, dominance of exotic species or compacted soils) are likely to create such barriers and influence potential recovery after stock removal (Whalley 1994; Yates & Hobbs 1997b; Windsor 1999; Yates *et al*. 2000a).

Canopy gaps

Overstorey crown cover was an important factor influencing tree recruitment and understorey composition. Tree recruitment was more common in gaps or near the edges of mature trees than beneath mature trees, as recorded in other woodland studies (Lacey 1972; Venning 1988; Curtis 1990). Tree recruitment in dense stands with > 50% crown cover appears to be a lengthy process, dependent on initial tree density, death of senescent trees and gap creation either by storm damage or human intervention (Lamb 1994).

Very few White Cypress-pine regenerated in fenced (or unfenced) plots. Most White Cypress-pine stands surveyed were dense even-age stands, which regenerated in the late 1800s and/or 1950s (Lacey 1972). While climatic conditions may not have been suitable for regeneration since fences were erected, White Cypress-pine regeneration is also strongly controlled by mature tree densities. Dense stands produce less viable seeds than more open stands and seedling recruitment in dense stands is inhibited, owing to strong competition from mature trees (Lacey 1972; Forestry Commission of NSW 1988). Thus, tree recruitment is unlikely to occur inside many White Cypress-pine stands because of the high density of existing mature trees.

Shrub recruitment

Little shrub regeneration was recorded, reflecting the paucity of surviving shrubs in most remnants. The results suggest that some localized shrub regeneration may occur after stock removal, but stock removal alone is unlikely to promote substantial shrub regeneration. This may be due to a number of factors. Many shrubs are hard-seeded (e.g. *Acacia, Pultenaea* and *Senna* spp.) and may require fire or soil disturbance to promote germination (e.g. Clemens *et al*. 1977; Clarke 1999). Therefore, shrub recruitment might be enhanced by burning or soil disturbance if viable seeds persist in the soil. However, there is little information on pre-settlement shrub distributions or densities and it is possible that shrubs were never abundant in many areas. Consequently, there may be little opportunity for shrubs to establish from persistent soil seed banks.

Despite uncertainties about original shrub densities, native shrubs provide important habitat for many bird species

(Gilmore 1985), and many land managers are attempting to promote shrub densities in woodland remnants. In degraded sites with few native ground plants, shrub establishment is unlikely to cause major management conflicts and could greatly enhance faunal diversity. Further studies are required on shrub distributions, population dynamics and soil seed banks in remnant woodlands, to provide practical suggestions for enhancing shrub populations where appropriate.

Soil compaction

Previous grazing history has led to high levels of soil compaction, which may restrict recruitment of native species (Windsor 1999), as compacted soils are more favourable to surface-rooted species (e.g. exotic annuals) than deep-rooted species, because of reduced moisture infiltration and root penetration (Moore & Biddiscombe 1964; Moran 1998). The observations made in the present study suggest that stock exclusion may have rapidly promoted a significant, although minor, reduction in soil compaction in most sites. Soil compaction is gradually alleviated by biological means (Bradford & Peterson 2000; Yates *et al*. 2000a) and further reductions in compaction may facilitate greater recovery of woodland species over time.

Ground-layer vegetation

Despite the fact that Greening Australia fenced many of the highest quality remnants in the region, the ground vegetation at most sites was substantially degraded by past grazing. Many grazing-sensitive native species (including Kangaroo Grass and Poa Tussock: Moore 1953a,b; Prober & Thiele 1993) were absent or very rare in most sites. Because many species that have been eliminated do not form persistent seed banks and have little potential to recolonize due to fragmentation and isolation (Lunt 1997; Morgan 1998) it would appear extremely improbable that fencing alone will restore the original ground-layer composition.

However, the results indicate that stock exclusion can lead to rapid improvements in native plant composition by promoting the growth (and possibly recruitment) of surviving grazing-tolerant plant species (e.g. Spear Grass and Wallaby Grass) and the depletion of exotic annual species such as Barley Grass (Moore 1973). The beneficial effects of native perennial grasses on tree and shrub recruitment have been observed elsewhere (e.g. Scougall *et al*. 1993; Cluff & Semple 1994; Pettit *et al*. 1995). Seedling survival is typically greater in bare ground between native grass tussocks, where seedlings are sheltered from severe weather conditions, but mortality is often high beneath dense grass swords (Curtis 1990; Kirkpatrick 1997). Therefore, it is possible that perennial grass cover helped promote tree recruitment in the present study.

Annual exotic plants, including annual grasses (e.g. Barley Grasses and Annual Rye Grass, *Lolium* spp.) and forbs (e.g. Capeweed, *Arctotheca calendula*) were abundant at most sites. Competition from annual exotics have been shown to restrict native plant regeneration in other Australian ecosystems (Hobbs & Atkins 1988; Scougall *et al*. 1993) and is likely to occur in these woodlands. However, perennial grasses can restrict the growth of many exotic annuals (Johnstone 2000). Although not significant, many common exotic species had less cover in fenced than in unfenced areas (e.g. Capeweed), which suggests the possibility that native perennial grasses that survived as grazed stubble may have recovered and out-competed exotic annuals since stock exclusion. Because we sampled functional plant groups and dominant species rather than total species composition, further changes might be detected from more detailed floristic studies.

Conclusions

The present study has documented positive ecological differences between fenced and unfenced remnant vegetation, including greater tree and shrub recruitment, increased perennial grass cover and a lower soil compaction. Approximately 26 000 recruits were found in the 47 fenced sites and it is likely that these recruited within 2–4 years of fencing. When compared to the $1.95 cost per seedling (tubestock, including labour), this equates to a saving in equivalent revegetation costs of nearly $69 000 (Schirmer & Field 2001). However, tree recruitment was largely restricted to Yellow Box and Grey Box woodlands and few trees recruited in White Cypress-pine woodlands. Despite described variability of results between sites, fencing to exclude stock, given appropriate time, is likely to provide strong financial and ecological benefits in terms of the challenge of rehabilitating remnant vegetation in Australia's agricultural landscape.

Refining management objectives

Increased tree and shrub recruitment is not necessarily needed in all sites and, therefore, will not always be an appropriate restoration goal or measure of success. For instance, few trees regenerated in sites with a dense tree canopy, but as there was no immediate need for more tree regeneration in these sites, enhanced understorey and soil conditions might have been more relevant goals and indicators of recovery.

In the future, a more refined appraisal of fencing outcomes might set desired goals according to initial conditions and appraise success in terms of these condition-dependent goals. Such an approach would be more useful than implying (as in the present study) that all sites should register an improvement on every ecological indicator. Furthermore, the aim of fencing is not always to enhance ecosystems in the short term, but rather to stabilize ecological systems and prevent further degradation (of vegetation, soils or streams) in the future. Thus, even where fencing leads to no apparent short-term enhancement, it may still be a useful strategy to prevent longer term degradation.

Ongoing management

A number of studies have stated that fencing alone will not rehabilitate remnant woodlands and that further inputs are required (e.g. Clarke 1999; Reeves 2000; Yates *et al*. 2000b; Clarke & Davison 2001). Assuming that the major emphasis is to enhance tree regeneration and native plant diversity, consideration could perhaps be given to the following: (i) where possible, placing fences further out from remnant edges to provide more gaps for recruitment (Venning 1988); (ii) conducting follow-up weed control, including spraying or crash grazing; (iii) applying disturbances such as ripping or scalping (particularly in heavily compacted soils) and patch burning to potentially trigger germination of native species; (iv) replanting shrubs and other native species; and (v) perhaps thinning dense overstoreys to provide gaps for regeneration and to improve structural complexity, especially in dense White Cypress-pine stands.

In reality, however, the potential for private landholders to undertake further works is limited by inadequate information to guide appropriate interventions and few resources (of time and money) to complete such tasks, regardless of landholder willingness (Lockwood *et al*. 1999). Unless there is a substantial injection of government resources (perhaps in the form of stewardship payments: Binning 1997; Miles *et al*. 1998), stock exclusion is the only management intervention that is likely to occur in most remnants.

Fencing is an important first step for conserving threatened woodland communities and achieving sustainable agricultural practices. However, little is known of the long-term outcomes of this management intervention. In effect, fencing to exclude stock is an abrupt ecological change that may trigger new and unexpected changes in woodland composition. Irrespective of this, the present results do suggest that regional fencing programs play an important role in helping to reverse woodland degradation. Fencing is clearly a critical first step for regional woodland rehabilitation. The question that is now posed is: what is the next step?

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