www.csiro.au

After the fence: condition of fenced, unfenced and reference York gum – jam woodlands in the Avon Catchment, Western Australia

Suzanne M. Prober, Rachel J. Standish and Georg Wiehl

December 2009

A project funded by Wheatbelt NRM Inc., in collaboration with WWF-Australia and the Department of Environment and Conservation (WA)

The University of Western Australia
Achieving International Excellence

Enquiries should be addressed to:

Suzanne Prober, Sustainable Agriculture Flagship and CSIRO Sustainable Ecosystems, Private Bag 5, Wembley WA 6913, suzanne.prober@csiro.au;
Rachel Standish, School of Plant Biology, University of Western Australia, 35 Stirling Hwy, Crawley, WA 6009, standish@cyllene.uwa.edu.au;
Georg Wiehl, CSIRO Sustainable Ecosystems, Private Bag 5, Wembley WA 6913, georg.wiehl@csiro.au

Citation: Prober SM, Standish RJ, Wiehl G (2009) After the fence: condition of fenced, unfenced and reference York gum – jam woodlands in the Avon Catchment, Western Australia. CSIRO Sustainable Ecosystems and Sustainable Agriculture Flagship.

Cover image: A fenced remnant of York gum-jam woodland showing significant York gum recruitment. This site was fenced and burnt in c. 2003. No recruitment was evident in the adjacent unfenced plot, which was burnt at the same time. © 2009 CSIRO. Photographer: Suzanne Prober.

Contents

List of Figures

List of Tables

1. SUMMARY

Considerable investment through natural resource management programs supports restoration of biodiversity in intensively used landscapes such as the Western Australian wheatbelt. One of the most common interventions is fencing of remnant vegetation to facilitate recovery from degradation caused by livestock grazing, yet the effectiveness of this intervention for enhancing biodiversity has only rarely been quantified. We compared 69 unfenced, fenced and reference sites in widespread but highly threatened herb-rich York gum (*Eucalyptus loxophleba* subsp. *loxophleba*) – jam (*Acacia acuminata*) woodlands of the Avon catchment, Western Australia, to explore two hypotheses: (1) that fencing facilitates recovery of degraded York gum – jam woodlands towards conditions of little-degraded 'reference' woodlands, and (2) that after fencing, recovery of degraded woodlands to reference condition is constrained by ecological or other limits. Fenced and unfenced sites were selected as adjacent, matched pairs, and fenced sites had been fenced for an average of 11 (2-22) years.

For measures of vegetation condition, our first hypothesis was supported by several lines of evidence (noting our assumption that prior to fencing, fenced sites were similar to their unfenced pairs). Fenced sites differed significantly from unfenced sites in species richness of most groups of native plants, native cover, exotic cover and tree recruitment, and the direction of these differences was generally towards reference conditions. Further, frequency of jam increased with increasing time since fencing, exotic cover decreased with time since fencing, and fenced plots were more similar to reference plots in floristic composition than unfenced plots were. However we found no evidence for recovery of nutrient-enriched woodland topsoils due to fencing.

Our second hypothesis was also supported for most condition measures. On average, soil nutrients were elevated, exotic cover was higher, and native richness was lower in fenced compared with reference sites, and ordination analyses suggested soils and understorey of fenced sites reached reference condition in only a small subset of cases. This may be partly due to lack of sufficient time since fencing, but regression analysis suggested that recovery from exotic invasion is more limited at higher soil nutrient levels, and that recovery of native species richness is constrained by the persistence of exotics. On the other hand, recruitment of jam in fenced sites was similar to that in reference sites. York gum recruits were absent from 81% of all sites (including reference sites), but there was a pulse of recruitment in some fenced plots that suggested fencing can enable episodic recruitment in conjunction with other disturbances.

We conclude that fencing to exclude livestock grazing is often effective for enhancing the biodiversity conservation values of remnant York gum – jam woodlands. However, additional interventions are likely to be necessary to achieve particular conservation goals or to promote recovery in some types of degraded sites. We develop an indicative framework to guide setting of conservation targets and to clarify where additional interventions may be of highest priority for York gum – jam woodlands.

2. INTRODUCTION

The wheatbelt of Western Australia is renowned for the richness and endemism of its flora, but this diversity is threatened due to fragmentation, weed invasion, salinization and degradation associated with widespread land clearing and intensive agricultural use (Prober and Smith 2009). To ameliorate these threats and enhance biodiversity conservation, considerable investment through natural resource management programs supports restoration activities in these landscapes. Fencing to exclude or reduce livestock grazing is one of the most common restoration activities in the Western Australian (WA) wheatbelt, consistent with other temperate agricultural landscapes in Australia (Spooner and Briggs, 2008). It is generally assumed that fencing will lead to positive outcomes for biodiversity. Despite 20 years of fencing programs in the Western Australian wheatbelt however, these assumptions remain poorly tested.

Only two studies, one in jarrah (*Eucalyptus marginata*) – marri (*Corymbia calophylla*) woodlands (Pettit and Froend, 2001) and one in salmon gum (*Eucalyptus salmonophloia*) woodlands (Fox, 2001), have measured outcomes of exclusion from livestock grazing in the WA wheatbelt. Both of these recorded benefits for biodiversity and ecosystem function, including increased native plant cover in both communities, and decreased abundance of exotic annuals in jarrah–marri but not salmon gum woodland (Pettit and Froend, 2001; Fox 2001). Effects of fencing on overstorey recruitment have not been systematically studied in the WA wheatbelt, although studies in eastern Australian have shown that fencing can promote tree regeneration in some but not other native tree species (Spooner and Briggs, 2008; Briggs et al. 2008). A series of studies in the WA wheatbelt to investigate poor recruitment in salmon gum (*Eucalyptus salmonophloia*) woodland similarly showed that interventions other than fencing (e.g. weed control and deep ripping) may be needed to reverse the impacts of grazing and enhance recruitment of woodland trees (Yates et al. 2000a,b).

Herb-rich York gum (*Eucalyptus loxophleba* subsp. *loxophleba*) – jam (*Acacia acuminata*) woodlands are one of the most common ecological communities of the central WA wheatbelt (Beard 1990), occurring in semi-arid to temperate Mediterranean rainfall zones with c. 300–450 mm mean annual rainfall. By contrast with related, shrubby York gum woodlands that are common in other parts of the WA wheatbelt, the understorey of herb-rich York gum – jam woodlands is naturally dominated by a diverse array of herbaceous species. These include a diversity of native grasses, interspersed with many annual and perennial forb species, and with scattered or patchy shrubs (Prober and Wiehl unpub. data).

Woodlands mapped by NVIS (Shepherd et al. 2002) as containing York gum (*Eucalyptus loxophleba* subsp. *loxophleba*) as a dominant species once occupied an estimated 3.7 million hectares in the WA wheatbelt (Fig. 1). Because they occur on some of the most profitable agricultural land however, about 90% of these woodlands have been cleared (Hobbs and Saunders, 1993; Shepherd et al. 2002), and the remaining 10% is typically heavily impacted by altered fire regimes, livestock grazing, and nutrient enrichment (Prober and Smith 2009). Consequently, the native understorey has frequently become heavily invaded by exotic species, especially annual grasses such as wild oats (*Avena barbata*) (Hobbs and Atkins 1988, Prober and Smith 2009), native plant diversity has declined (Prober and Wiehl unpub. data), and tree recruitment processes have been modified (Hobbs and Atkins 1991). Unlike many other ecological communities of the WA wheatbelt, herb-rich York gum woodlands do not extend significantly into uncleared areas to the east of the WA wheatbelt (a region increasingly known as the 'Great Western Woodlands', Watson et al. 2008). Consequently, conservation of these woodlands is largely reliant on effective conservation management and restoration within the intensive agricultural zone, raising them as a particularly high priority for monitoring and evaluation programs.

This project, funded by Wheatbelt NRM Inc. (formerly Avon Catchment Council) in partnership with World Wide Fund for Nature and the Department of Environment and Conservation WA, aimed to evaluate outcomes of fencing to control grazing by domestic livestock in York gum – jam woodlands. Considerable fencing of remnant vegetation in the region has been undertaken since the establishment of the Remnant Vegetation Protection Scheme by the Department of Agriculture WA in 1989, followed by later programs supported, for example, by WWF, Greening Australia and Wheatbelt NRM Inc. We aimed to evaluate whether fencing during the past 20 years has promoted recovery of native understorey, ameliorated altered soil conditions, or facilitated recruitment of trees and shrubs. In relation to these ecosystem components, we used comparisons of unfenced, fenced and reference sites to explore two hypotheses:

(1) Fencing facilitates recovery of degraded York gum – jam woodlands towards conditions of reference sites.

(2) After fencing, full recovery of degraded woodlands to reference condition is constrained by ecological or other limits, potentially including time since fencing, livestock grazing, propagule limitation, nutrient enrichment and weed invasion.

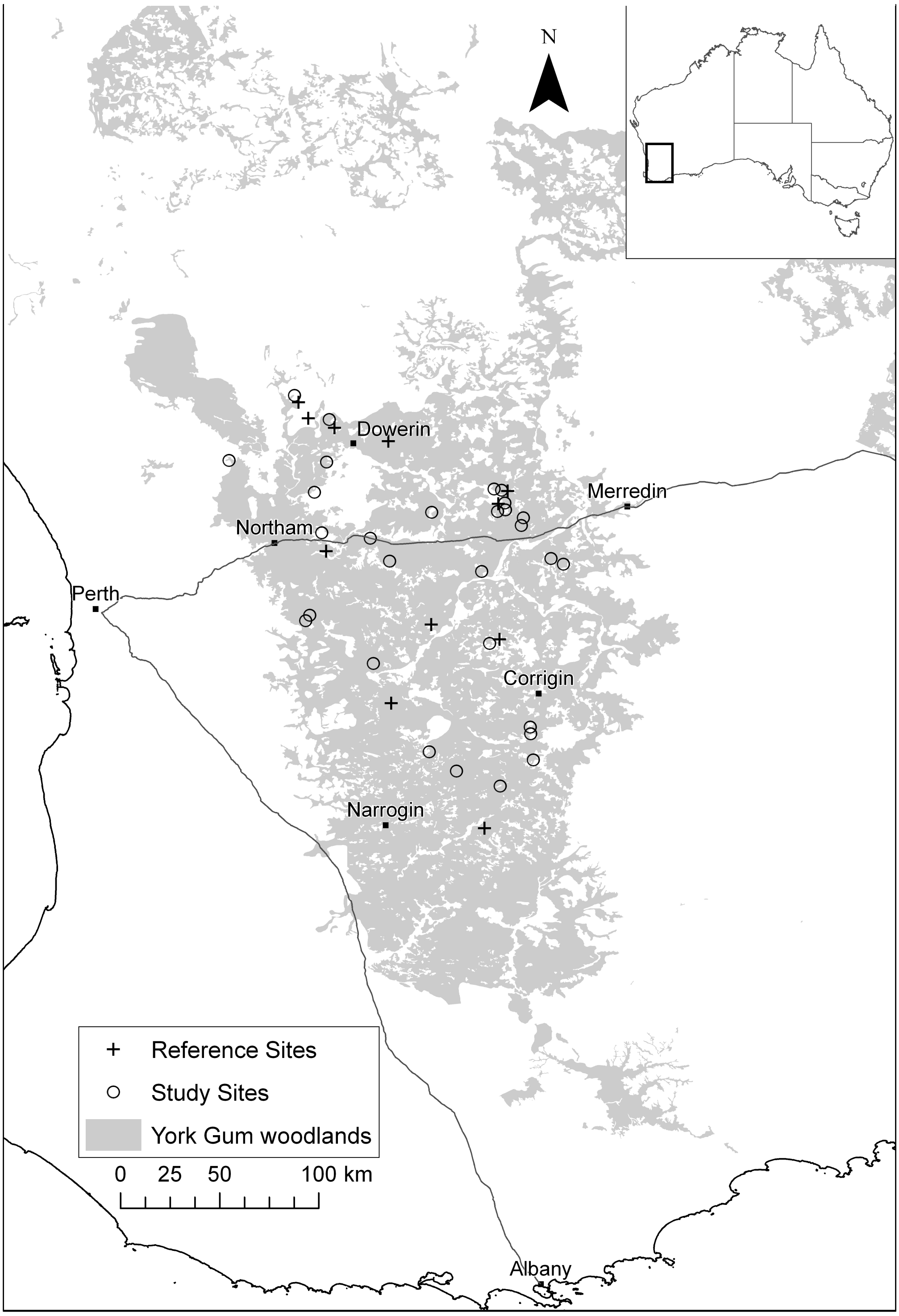
3. MATERIALS AND METHODS

3.1 Survey design

The study spanned the distribution of herb-rich York gum – jam woodland within the Avon catchment of south-western Australia (Fig. 1), and aimed to capture gradients in natural environmental parameters (topographic position, rainfall) and vegetation condition. To estimate changes in native vegetation associated with fencing, we compared plots placed within fenced York gum - jam woodlands with plots placed in adjacent parts of the remnant that were outside the fence. We assumed that prior to fencing, fenced sites were similar to their unfenced pairs.

To identify potential survey sites, databases of fencing programs over the past 20 years were supplied by WWF-Australia, Greening Australia (WA) and the Department

Figure 1. Distribution of study sites (unfenced/fenced pairs) and reference sites sampled in this study, overlaid on the distribution of York gum – jam woodlands (Shepherd et al 2002).



of Agriculture and Food (WA), providing details of over 600 sites where fencing had been undertaken in the Avon catchment. Approximately 150 sites were selected from these databases as likely to contain York Gum woodland, based on associated information. Landholders of these sites were contacted by telephone to assess whether they would like to participate in the project, and whether their sites might meet the following criteria: (1) the fenced remnant contained York gum woodland, (2) the fence passed through the remnant leaving a suitable, environmentally similar ‘control’ site on the other side of the fence, (3) prior to fencing, patches now on different sides of the fence had received the same management (almost always sheep grazing), and (4) management of the unfenced site was currently similar (although not always identical) to what it was prior to fencing.

Based on results of telephone surveys, we visited 134 potential sample sites on 61 farms across the Avon catchment. Of these, 29 sites adequately met the above criteria and were used to establish 58 permanently marked monitoring plots (one fenced and one unfenced plot per site). Remnants ranged in area from 1-511 ha (mean 66.6 ha), and had been fenced for between 2 and 22 years (mean 10.9 years). Sites spanned a rainfall gradient of c. 320-470 mm (mean 360 mm), with mean annual temperature ranging from 15.9–17.7˚C (mean 16.9˚C). When visiting the site, landholders were also asked to indicate grazing levels before and after fencing (scored using a scale of 0-5, nil to very high), type of grazing animal, and the year fences were erected.

In addition to paired fenced/unfenced plots, we surveyed eleven ‘reference’ York gum – jam woodlands distributed across the same region (Fig. 1), to facilitate assessment of condition of fenced sites. These represented the highest-quality remnants of this vegetation type that we were able to identify through local knowledge and reconnaissance surveys. They included five sites on private land, identified opportunistically when speaking with landholders, and accepted as reference sites owing to a history of minimal livestock grazing (usually associated with the presence of ‘poison pea’, *Gastrolobium* spp. that are highly toxic to livestock). Six reference sites were selected within relatively weed-free sections of Nature Reserves or town reserves. Although now long-ungrazed, most of these have a history of intermittent livestock grazing early in the 20th century, so some modification due to grazing cannot be excluded.

Consistent with the distribution of York gum – jam woodlands, most sites occurred on soils derived from granite or granitic gneiss, varying from sites with abundant exposed granite outcrops to sites with deeper soils and colluvial or alluvial influences (Department of Industry and Resources 2001). A dolerite dyke was noted passing through one fenced/unfenced pair.

3.2 Monitoring

At each of the 29 monitoring sites we established two 20 x 50 m quadrats, approximately adjacent to each other but on the ungrazed vs. grazed sides of the fence. All plots were at least 10 m from remnant edges and 1 m from internal fences (i.e. fenced and unfenced pairs were at least 2 m apart, or rarely up to 200 m to allow for intervening tracks or other anomalies). Within these plots, we assessed the following ecological characteristics:

1. Recruitment and mortality of native trees and shrubs were estimated by recording the size frequency distribution of live and dead plants. Diameter at breast height (DBH) was measured if live or dead plants were taller than 1.4 m and a nominal DBH of 0.5 cm was allocated to plants less than or equal to 1.4 m tall (defined as 'recruits'). For multi-stemmed plants, an averaged DBH was calculated as the square root of the sum of the squares of the DBH of individual stems. Calculations included dead stems if the plants were alive.

2. Abundance of pre-defined native and exotic plant groups was estimated in September-October 2008 using a line-intercept technique (see Prober et al. 2005). An 8 mm dowel was placed vertically at each of 50 points on an approximate grid across each plot; the relative abundance ("percent points") for any group was the percentage of points at which any leaves, stems or inflorescences of species from that group intercepted the dowel. Groups that were present but did not intercept the dowel at any point were allocated a nominal abundance of 0.5. This technique provided an objective measure of abundance reflecting but not equivalent to projective cover, and is hereafter referred to as cover.

3. Bare ground, native and exotic plant litter and log abundance were estimated using the same line intercept technique. In addition we measured the cumulative length of all logs of >5 cm diameter at their widest point as a measure of potential fauna habitat.

4. All plant species occurring within a 10 x 10 m subplot nested within each 20 x 50 m plot were recorded. Sub-plots were systematically placed with one edge centred along the 20 x 50 m plot edge parallel with and closest to the fence whenever possible, or occasionally in a more appropriate position to achieve a comparable pair with similar canopy cover and landscape position. Nomenclature follows WA Herbarium (2009). Abundance of each species was estimated using the line intercept technique described above (within 10 x 10 m plots), and cumulative scores for native and exotic plant cover were calculated.

5. Soil nutrient levels were estimated by collecting thirty 2 cm diameter, 10 cm deep soil cores from a grid pattern across the 10 x 10 m subplots, during September to October 2008. Samples for each plot were bulked, stored in sealed plastic bags in a refrigerator at ~4℃, and analysed at CSBP Futurefarm analytical laboratories (Bibra Lake, WA). Samples for each plot were thoroughly mixed, air dried at 40 ℃, and ground to pass through a 2 mm sieve. Analyses were undertaken on each bulked sample as follows (where given, method numbers apply to Rayment and Higginson 1992): available phosphorus (Colwell method, bicarbonate-extractable phosphorus- manual colour, 9B1), potassium (Colwell method, bicarbonate-extractable potassium, 18A1), ammonium and nitrate (measured simultaneously using Lachat Flow Injection Analyser, soil:solution ratio 1:5, 1M KCl, indophenol blue, Searle 1984, and with copperized-cadmium column reduction), pH (1:5 soil/0.01M $CaCl_2$, 4B2), conductivity (1:5 soil:water extract, 3A1), organic carbon (Walkley and Black method, 6A1), extractable sulphur (40℃ for 3 hours , 0.25M KCl, measured by

ICP, Blair et al. 1991), and total nitrogen (oxygen combustion, 950℃ with Leco FP-428 analyser).

6. Soil physical properties were measured in the 10 x 10 m subplots. Soil surface compaction was measured at 30 random positions using a calibrated 0-5 MPa pocket penetrometer (6.4 mm needle diameter). Bulk density was estimated by weighing dried soil from each of five soil cores (55 mm diameter and 60 mm depth) per plot, and dividing by the volume of each core. Instantaneous volumetric soil moisture content was measured using a MPM406 soil moisture probe, with fifteen measurements averaged across each plot.

We also measured or calculated a range of associated environmental variables that might influence recovery or apparent recovery of fenced sites. We scored topographic position as an ordinal variable (1, hill top through to 5, drainage lines), distance from crop to nearest plot edge, and landscape integrity (% area of remnant vegetation within a 100 m and a 1000 m radius, calculated using remnant vegetation extent layers in ArcGIS (Shepherd *et al.* 2002). Mean annual rainfall and temperature for each plot were estimated using BIOCLIM, a component of ANUCLIM version 5.1 (Houlder *et al.* 2001).

In reference sites, we measured the same variables described above, except that we did not have time to measure abundances at the 20 x 50 m scale (2 and 3). We measured floristic composition, and soil properties in each of two 10 m x 10 m subplots rather than one subplot per reference site. Other studies have shown that soil properties in eucalypt woodlands vary beneath trees compared with gaps (e.g. Prober et al. 2002a). Hence to maximize the range in soil properties sampled in reference sites, we placed one plot beneath York gum canopy and one in a gap.

3.3 Data analyses

3.3.1 Comparisons of fenced, unfenced and reference plots

Univariate statistical analyses were conducted using Genstat 12.1 (VSN International Ltd, 2009). Paired t-tests were used to test for overall differences in floristic and soil characteristics between fenced and unfenced plots. Soil chemical and plant cover variables required log transformation (ln (x + 1)) to satisfy the assumptions of parametric analysis. We also used permutational tests (using 4999 random permutations) to obtain significance values. These were generally identical or similar to values from parametric tests, so are presented only for analyses with more than 20% zeros. Means and standard errors for reference sites were also calculated, and were compared with fenced and unfenced sites using independent groups t-tests.

Of the trees and tall shrubs we surveyed, jam wattles were the most frequent, followed by York gums and the needle tree *Hakea preissi*. Other species were infrequent so were grouped together as ‘other trees’ or ‘other shrubs’ for analyses. We estimated the effect of fencing on frequency of all individuals (all size classes), frequency of individuals in the <0.5cm DBH class (recruits), and the frequency of dead plants, using two analyses. First, for each species or group we used generalised linear regression with a Poisson distribution and a log-link function to compare fenced and unfenced plots. We assumed a Poisson distribution was appropriate because these data were skewed towards zero i.e., for any one species there were a small to large number of plots with no individuals. Then, generalised linear mixed models were fitted to the full dataset to determine if the effects of fencing were consistent among species/groups, and to test for species x fencing interactions. Interaction terms were not statistically significant and so only the results of the regression analyses for individual species/ groups are presented.

A similar set of analyses were applied to mean DBH of live jam, York gum and needle tree as response variables. We compared the mean DBH of live plants in fenced and unfenced plots using linear regressions fitted to the data for each species. Then, a linear mixed model was fitted to data from all three species to test the effect of fencing over all species and to test the species x fencing interaction. None of these analyses yielded statistically significant results, so only the results of the linear regressions for individual species are presented.

3.3.2 Which variables predict benefits of fencing?

To facilitate prediction of the likely effectiveness of fencing in remnants of differing condition or environmental contexts, we calculated differences between individual fenced and unfenced pairs for key variables (exotic annual cover, native richness, soil nutrients, number of York gum and jam individuals, number of York gum and jam recruits). We then used general linear regression to explore how the extent of change at each site was associated with time since fencing and the range of environmental variables collected. For each response variable, the most informative explanatory variables were identified using all-subsets regression of up to 16 explanatory variables at a time (permitting a maximum of three to six variables in any one model). Stepwise general linear regression was then applied to determine optimal combinations of these variables and selected interactions and quadratic terms. Final models were selected on the basis of maximum adjusted R^2 values. We also used this regression approach to investigate drivers of some explanatory variables.

We were concerned that our sampling strategy would lead to a bias associated with greater exposure to cleared cropland in unfenced sites, because unfenced areas were more likely to be at the edge of remnants and may be more exposed to nutrient enrichment, grazing or weed seed rain. We actively attempted to avoid this by seeking pairs where unfenced plots were not directly associated with edges, but our final sampling confirmed a small bias, with unfenced sites an average of 42.6 m from crop edges compared with 48.8 m for fenced sites (p=0.045). We thus calculated the difference in distance from paddock edges and difference in landscape integrity at the 100 and 1000 m scales, between plots in each pair, and included this as a candidate variable in general linear regressions. These variables did not significantly contribute to any models produced in all-subsets regressions, providing some confidence that slightly greater exposure to edges did not unduly influence results.

3.3.3 Multivariate analyses

Ordinations were used to explore differences in the soil and floristic properties of unfenced, fenced and reference plots. Ordinations are used to uncover the structure or dimensions of a dataset of multiple variables. At best, we could expect our plots to cluster according to whether or not they were unfenced, fenced or within reference sites, and for fenced plots to cluster more closely to reference plots than to unfenced plots.

For the soil data, we used Principal Components Analysis (PCA) to analyse soil chemical variables plus bulk density and surface hardness. PCA seeks a linear combination of variables such that the maximum variance is extracted from the variables. It then removes this variance and seeks a second linear combination which explains the maximum proportion of the remaining variance, and so on. PC-ORD was used for this analysis (McCune and Mefford 1999), using the correlation matrix (data centred and standardized by standard deviation, Greig-Smith 1983) and Euclidean distance.

For floristic data we used non-metric multidimensional scaling (nMDS) analysis to produce ordinations. nMDS uses ranked distances to arrange plots along a number of axes based on the abundance and composition of the vegetation within them. The placement of plots and number of axes in nMDS ordination are calculated as the solution minimizing the final 'stress' between the dissimilarity in the original data matrix and that in the reduced ordination matrix. Unlike PCA, this ordination method is well-suited to floristic data that are non-normally distributed. Quantitative floristic data (excluding tree species) were square root transformed (to reduce the influence of dominant species) and used to produce a distance matrix using the Bray-Curtis coefficient of dissimilarity (Faith *et al.* 1987). nMDS was performed on the distance matrix using the software package DECODA (Minchin 1989). Preliminary analyses were performed in one to four dimensions using 10 random starts; these indicated that the three dimensional solution was optimal, and the solution with lowest stress value (0.15, a measure of poorness-of-fit that varies from 0 to 1) was selected.

Direct overlays and biplots were produced using PC-ORD to examine relationships between the ordinations and environmental variables. For overlays, larger symbols indicate larger values; for the biplots, the angle and length of the line indicate the direction and strength of the linear relationship between sample plots and response or explanatory variables (McCune and Mefford, 1999). We also used PC-ORD to perform a non-parametric test similar to discriminant analysis (MRPP), to test the significance of soil and floristic differentiation between reference and other plots. To test for apparent recovery of floristic composition towards reference sites due to fencing, we rotated the ordination on the vector best separating reference sites, using the vector-fitting procedure of DECODA (Minchin 1989). Scores for the position of plots on this vector (axis 1) were extracted and further analysed using paired t-tests as described above, and used to order sites and species to produce two way tables indicating species contributing to the difference between reference and other plots.

4. RESULTS

Managers of almost all sites indicated that sheep grazing was the major type of livestock grazing prior to fencing. Current grazing levels in unfenced sites were similar to (or sometimes lower than) levels prior to fencing of their adjacent pair, and had increased in only one case. There was rarely a substantial amount of livestock grazing in fenced sites after fencing. Notwithstanding, moderate to high levels of rabbit or kangaroo grazing were noted by managers of nine pairs of plots, three pairs had been burnt within the past ten years and one pair had been flooded.

4.1 Soil attributes

There were no indications of recovery of soil chemical properties associated with fencing. On average, both fenced and unfenced plots had higher soil nutrient levels than reference sites, which were significant for all measured soil properties other than ammonium and pH (Table 1). Further, there were no significant differences between fenced and unfenced plots that might suggest a consistent trend in soil chemical properties towards or away from reference sites (Table 1). Even for a subset of 16 plots that had been fenced for nine or more years ('long-fenced plots'), most nutrients were on average, higher than reference levels and not significantly different from unfenced plots. Notwithstanding, the extent of differences between means were not particularly large, ranging between 1.3 and 1.9 times greater on fenced compared with reference sites. Further investigation of the difference in soil nutrient levels between each fenced and unfenced pair revealed no significant linear or non-linear relationships between potential explanatory environmental variables, including the number of years fenced.

Some differences between fenced and unfenced plots were evident in soil physical properties and surface conditions. Fenced plots had on average less bare ground, fallen logs and weed litter, lower topsoil bulk density, and a tendency to lower surface hardness ($p=0.064$, Table 1) than unfenced plots. Notably, the shift in bulk density and bare ground was away from rather than towards reference levels, which were more similar to unfenced sites in these characteristics. There were no differences between fenced and unfenced plots in instantaneous soil moisture content, native litter or cover of soil cryptogams, although there was a tendency towards higher cover of foliose lichens in fenced plots ($p=0.076$, Table 1).

Principle coordinates analysis of soil properties showed a strong gradient along axis 1, which explained 48% of the variance in the data (Fig. 2a). This axis was parallel to the maximum separation between reference and other plots (MRPP $p<0.001$), and was most strongly related to Colwell phosphorus, total nitrogen and organic carbon. Fenced plots showed no consistent shift from their unfenced pair towards reference sites along this axis (Fig. 2b, $p=0.86$), and there was no apparent relationship between position of fenced sites on this axis and years since fencing (Fig. 2c). Sites separated to a lesser extent on axis 2 compared with their separation along axis 1; axis 2 explained 13% of the variance in the data and was most strongly related to pH and potassium (Fig. 2a), but was unrelated to fencing.

Table 1. Means for soil properties at reference (ref), long-fenced (9-22 years), all fenced (2-22 years) and unfenced plots. ***P<0.001, **P<0.01, *P<0.05, ns=not significant for comparisons as indicated, na=not available ref=reference. Means for are back-transformed where appropriate.

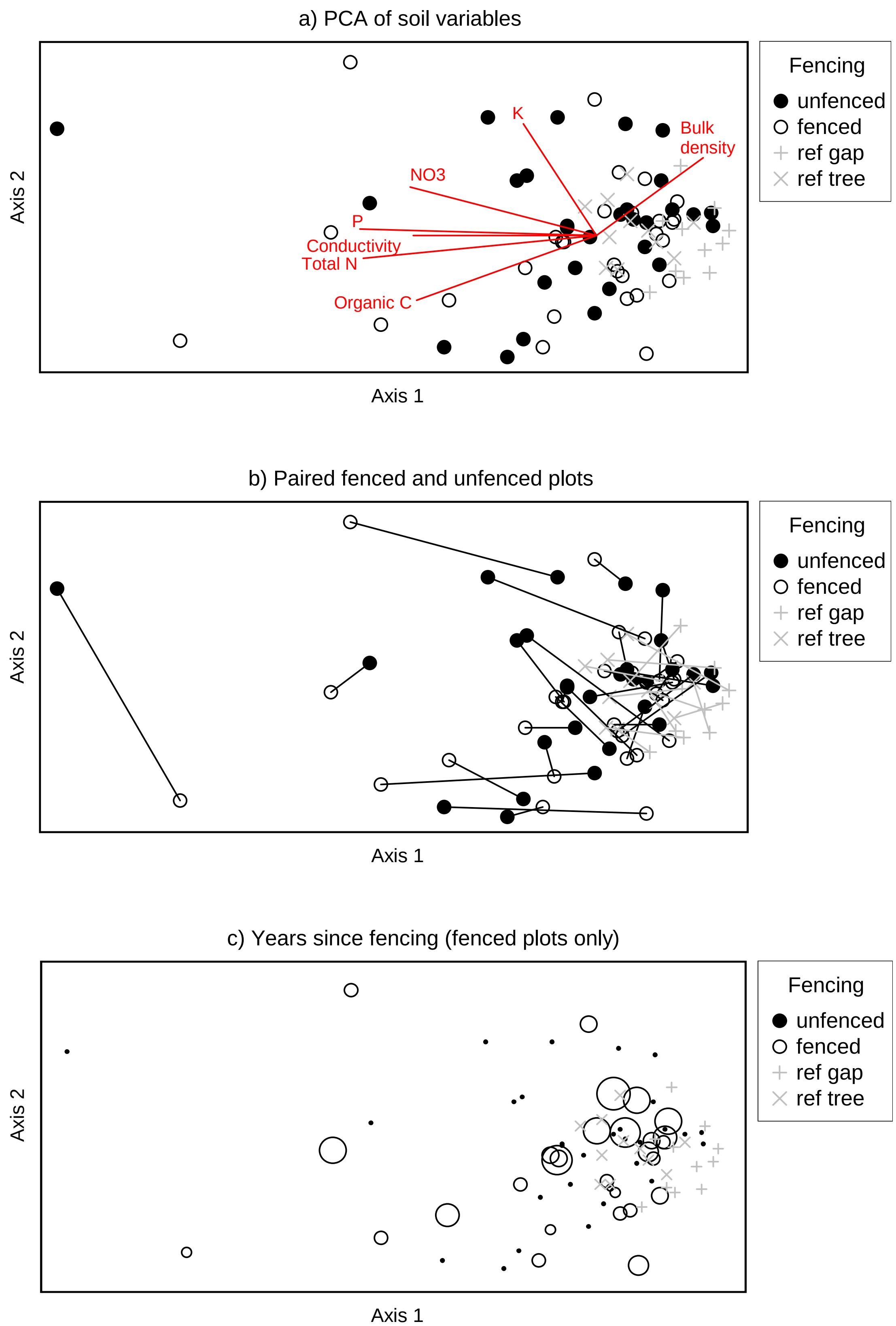
		ref	ref v long-fenced	long-fenced	ref v fenced	fenced	ref v unfenced	unfenced	fenced v unfenced
		n=22	P	n=16	P	n=29	P	n=29	P
Soil chemistry									
Ammonium	mg/kg	2.32	ns	2.61	ns	3.32	**	4.14	ns
Conductivity	dS/m	0.04	*	0.07	***	0.08	***	0.08	ns
Nitrate	mg/kg	3.19	**	6.33	**	5.92	**	5.95	ns
Organic carbon	%	1.13	***	1.61	***	1.83	***	1.76	ns
pH		5.12	ns	5.36	ns	5.22	ns	5.26	ns
Phosphorus	mg/kg	2.43	***	4.38	***	4.66	***	5.04	ns
Potassium	mg/kg	84.11	***	160.74	***	148.61	***	152.24	ns
Sulphur	mg/kg	3.78	ns	4.23	*	5.06	*	5.11	ns
Total nitrogen	%	0.08	***	0.12	***	0.15	***	0.14	ns
Soil physical and surface properties									
Bulk density	g/cm3	1.32	ns	1.23	***	1.21	ns	1.31	***
Moisture	%vol	na				2.78		2.70	ns
Hardness	MPa	2.80	*	3.33	ns	3.00	*	3.44	0.064
Bare ground	% points	25.82	ns	20.62	**	15.70	ns	26.70	***
Native litter	% points	47.90	***	29.78	**	34.40	***	30.20	ns
Weed litter	% points	0.46	***	4.32	***	4.84	***	7.65	*
Length of logs	m	78.76	ns	75.78	ns	68.97	ns	85.31	*
Cryptogam crust									
All+	% points	na		27.40		23.60		19.10	ns
Leafy lichens	% points	na		3.22		3.26		2.31	0.076
Mosses	% points	na		6.70		4.16		2.88	ns
Other	% points	na		17.60		16.20		13.90	ns

+ Cumulative score derived by summing % points of relevant sub-classes

Although fenced and unfenced plots were higher on average in soil nutrients than reference sites, there was still notable overlap among these groups. This is illustrated by the PCA, which suggests that about 10 fenced and unfenced pairs were comparable with reference sites in soil properties, and about five sites were within the range of reference soil conditions for their respective fenced but not unfenced plots. Closer examination of two important nutrients, total nitrogen and Colwell phosphorus, confirmed a similar pattern, with eight fenced and nine unfenced plots (29% of plots) within the ranges of total nitrogen and Colwell phosphorus measured at most reference plots.

Figure 2 (over). Results of PCA of soil variables showing separation among fenced, unfenced and reference plots (each symbol corresponds to one sampled plot), and (a) relationship with soil variables, (b) unfenced vs fenced pairs (black lines) and pairs in gaps (gap) vs beneath trees (tree; grey lines) for reference sites, (c) years since fencing.

Figure 2. (see caption previous page)



4.2 Floristic composition and diversity

There was strong evidence that vegetation condition was better in fenced compared with unfenced plots (Table 2). In particular, mean native plant species richness was on average four species greater in fenced plots, contributed mostly by herbaceous groundcover species (especially native perennial and native annual forbs). Richness of native shrub species was generally very low (mean 0.9 species per plot), but was marginally higher in fenced plots (p=0.077). Despite better condition of fenced plots, native plant species richness (especially for herbaceous species) was significantly lower than in reference sites, by an average of six species. This difference from reference sites remained at an average of six species even for the subset of 16 sites fenced for nine years or more (Table 2).

Table 2. Means for floristic characteristics at reference (ref), long-fenced (9-22 years), all fenced (2-22 years) and unfenced plots. ***P<0.001, **P<0.01, *P<0.05, ns=not significant for comparisons as indicated, na=not available, ref=reference. Richness in 10 x 10 m plots, cover (% points) based on 50 x 20 m plots unless 10 x 10 m indicated.

	ref	ref v long-fenced	long-fenced	ref v fenced	fenced	ref v unfenced	unfenced	fenced v unfenced
	n=22	P	n=16	P	n=29	P	n=29	P
Native cover								
Trees	na		23.39		26.97		23.09	ns
Understorey+ (50x20)	na		40.72		43.66		34.73	*
Understorey+ (10x10)	148.75	***	96.91	***	106.45	***	95.25	0.13
Shrubs	na		1.33		1.36		0.95	ns
Ground layer+	na		37.21		40.68		31.62	*
Grasses	na		21.53		19.39		16.27	ns
Perennial forbs	na		3.22		3.18		1.61	*
Annual forbs	na		7.07		10.88		9.00	ns
Native richness								
Total	28.23	***	21.81	***	22.41	***	18.28	***
Shrubs	1.50	ns	1.19	ns	1.10	**	0.69	0.077
Ground layer	24.95	***	18.88	***	19.38	***	15.97	***
Grasses	4.82	ns	4.19	ns	4.17	*	3.86	ns
Perennial forbs	7.68	ns	5.81	*	5.86	***	4.24	**
Annual forbs	12.45	***	8.88	***	9.34	***	7.86	**
Exotic cover								
Total+ (50x20)	na		29.23		34.23		54.37	***
Total+ (10x10)	12.61	***	38.02	***	41.69	***	53.49	0.078
Annuals+ (50x20)	na		25.84		31.79		50.52	***
Annuals+ (10x10)	11.77	***	31.46	***	35.74	***	50.11	*
Annual grasses	na		15.01		19.33		32.45	**
Annual forbs	na		7.93		8.01		12.24	*
Perennial forbs	na		1.75		1.10		1.16	ns
Exotic richness								
Total	1.06	***	8.86	***	9.51	***	9.07	ns

+ Cumulative score derived by summing % points of relevant sub-classes

Table 3. Summary of significant regression models. F=fenced, U=unfenced, ln=natural log. No significant models were found for soil nutrients (ammonium, organic carbon, phosphorus, potassium, nitrate, sulphur, total nitrogen), length of fallen logs, jam or York gum recruits, or York gum frequency, so these are not shown.

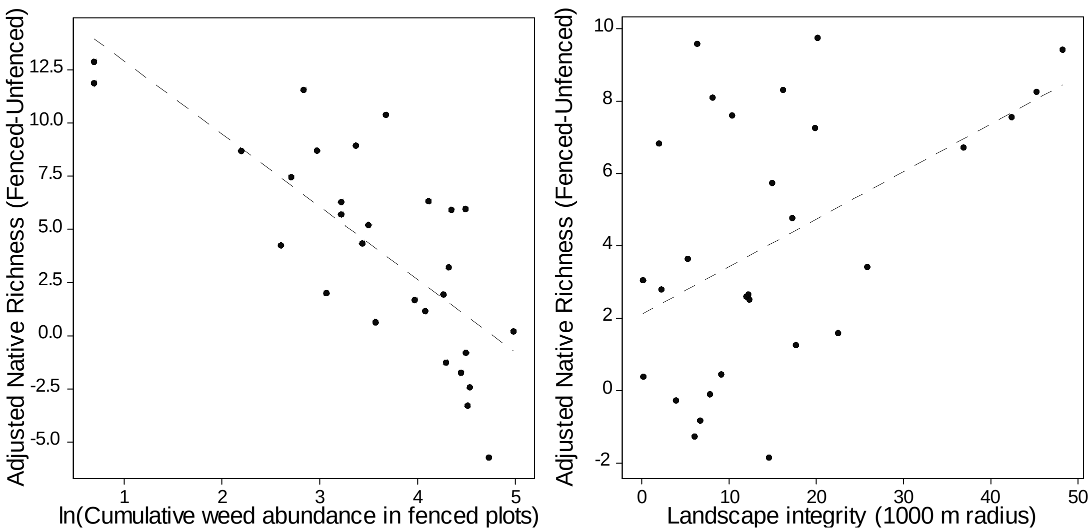
Response variable	Form of response variable	n	Best models	Adjusted R^2 (%)
Native richness	F-U	29	+Native richness unfenced -Native richness unfenced^2 -Exotic cover fenced +Landscape integrity 1000m / -Prior grazing	46.0/45.3
Exotic annual cover	ln (F+1) – ln (U+1)	29	-Years fenced -ln Total N unfenced +Years fenced*ln Total N unfenced (interaction)	41.6
Jam frequency	ln (F+1) – ln (U+1)	29	-ln Jam frequency unfenced +Landscape integrity 1000 m radius +Years fenced +ln K unfenced	60.5
Relationships among predictor variables				
Landscape integrity (1000 m radius)	Untransformed	58	-Prior grazing level -ln S / -ln P -Surface compaction	57.5/53.8
		58	-Topographic class -ln S -Surface compaction +Rock cover (constrained to exclude prior grazing level)	37.1
Total N	ln (X+1)	58	+ln Organic C +ln Colwell P	69.5
		58	+ln Colwell P +Rainfall +ln S (constrained to exclude organic C)	53.3

Cumulative cover of native understorey groups showed patterns similar to but weaker than patterns for species richness. Native cover in fenced plots was significantly greater than in unfenced plots as measured at the 20 x 50 m scale, by an average of 10%. Measurements at the 10 x 10 m scale resulted in higher cumulative cover values than at the 20 x 50 m scale, because we summed scores for species rather than species groups. At the 10 x 10 m scale, fenced plots again had higher native cover than unfenced plots, although the difference was not significant. At this scale, cumulative native cover in fenced plots (106%) and in the 16 long-fenced plots (97%), were still significantly and substantially lower than in reference plots (149%, Table 2).

Cumulative cover of exotic species was 20% lower on fenced compared with unfenced plots, although fenced plots still had significantly higher average exotic cover (34%) than optimum levels (0%) or levels on reference sites (12%). This trend was consistent and significant for cumulative cover of exotic annual grasses and exotic annual forbs, but not for exotic perennial forbs (Table 2).

General linear regressions highlighted that differences in floristic characteristics between fenced and unfenced pairs were not consistent from site to site. For native species richness, the best-fit model (R^2=46%) suggested a quadratic (curvilinear) relationship with richness of unfenced plots. This result suggested that if species richness was high at the time of fencing, then fencing was less likely to result in the appearance of new species, perhaps not surprising. Once this pattern was accounted for, the model suggested differences in native richness declined with increasing residual weed cover (weed cover of fenced plots) and decreasing landscape integrity at the 1000 m scale (Fig. 3, Table 3). That is, where fenced plots had a high cover of weeds remaining, native richness appeared less likely to recover, and where there was a high vegetation cover in the landscape, native richness appeared more likely to recover. If landscape integrity is replaced with prior grazing levels in this model, only slightly less variation was accounted for, which is consistent with a significant correlation between these two variables (Table 3). However, no significant relationships with years since fencing were detected.

Figure 3. Effects of residual weed cover and landscape integrity (%) on differences between fenced and unfenced plots in native richness, after adjustment for after adjustment for richness in unfenced plots.



For exotic annuals, the difference between fenced and unfenced sites became more negative (suggesting greater decline in exotics) with time since fencing, but the extent of the difference was dependent on soil total nitrogen (or to a lesser extent with organic carbon or mean annual rainfall, noting that these variables were related, Fig. 4, Table 3, Plate 2). That is, where total nitrogen or related variables were high, the decline in exotic annuals associated with fencing was slower.

Figure 4. Difference in cover of exotic annuals between fenced and unfenced plots in relation to time since fencing and total nitrogen, 0.08–0.32 % represented by the size of circles.

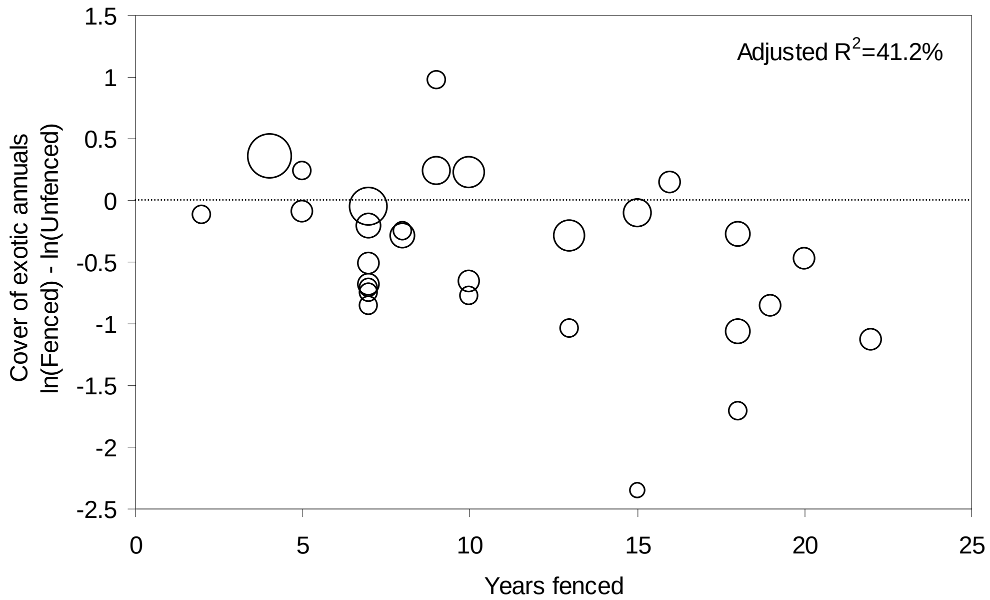


Plate 1. Nutrient rich sites such as this one showed little difference between fenced (right) and unfenced (left) plots. This site had been fenced for ten years. © 2009 CSIRO. Photographer: Suzanne Prober.

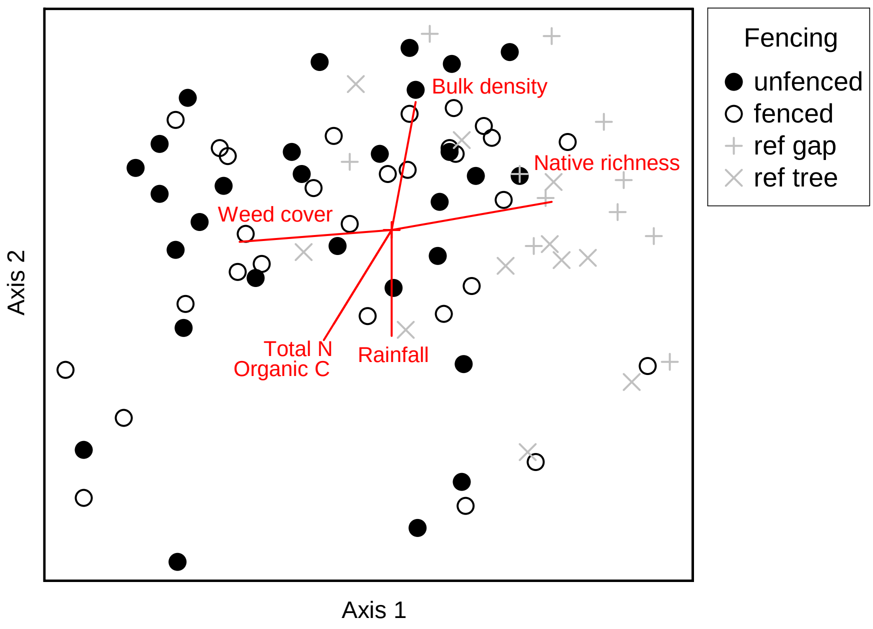
Similar to PCA analysis, nMDS ordination of floristic data showed reference plots clustered at one extreme of axis 1 of the ordination, overlapping with some fenced and unfenced plots (Fig. 5). These trends were strongly correlated with native plant and exotic cover (Fig. 6). They illustrate that, although reference plots were on average higher in native richness, lower in weeds, and different in species composition, some fenced and unfenced plots were indistinguishable from the core group of reference sites. The nMDS also suggests that several reference plots were "outliers", falling more generally amongst unfenced and fenced plots on the ordination. It is likely that this reflects some historical degradation in these reference sites; hence it is not possible to delineate exactly which sites match reference conditions. If a core group of 16 reference plots is considered (i.e. excluding six apparently outlying plots), only two pairs were within the range of reference soil conditions for their respective fenced but not unfenced plots (and for one pair, both fenced and unfenced plots occurred within the envelope of reference plots).

Despite this uncertainty with regard to reference conditions, paired t-tests clearly indicated that the fenced plots generally occurred significantly closer to reference plots on axis 1 (the axis best distinguishing reference sites) compared with their unfenced pair ($P=0.035$, Fig. 5).

Species contributing to trends along axis 1 (reflecting condition in relation to reference sites) included a predominance of exotic annuals (e.g. *Hordeum leporinum, Erodium botrys, Bromus rubens, Avena barbata*) at the greatest distance from reference sites, and at the other extreme, a suite of native species most frequent in reference plots (Table 4). These included the native annuals *Gilberta tenuifolia, Lawrencella rosea* and *Gnephosis tenuissima,* and the native perennial forbs *Thysanotus patersonii* and *Dampiera lavandulacea.* Many other native species were absent from the most species-poor plots, but occurred in fenced and unfenced plots as well as reference plots, with increasing cover along axis 1 (e.g. the native perennial grass *Neurachne alopecuroidea* and the native annuals *Waitzia acuminata* and *Trachymene cyanopetala*).

Axis 2 of the ordination correlated most strongly with mean annual rainfall and soil bulk density (which tended to increase with decreasing rainfall) (Figs. 5,6). Organic carbon and total nitrogen increased at higher rainfall and decreased towards reference sites, resulting in a diagonal trend on the ordination (Figs. 5,6). There was no apparent relationship between position of fenced plots on the ordination and years since fencing (Fig. 6).

Figure 5. Non-metric multi-dimensional scaling analysis of fenced, unfenced and reference plots based on floristic data showing (a) relationship with variables correlating with these axes at R^2>0.35, (b) unfenced vs fenced pairs (black lines) and plots beneath trees vs gaps for reference (ref) sites (grey lines).



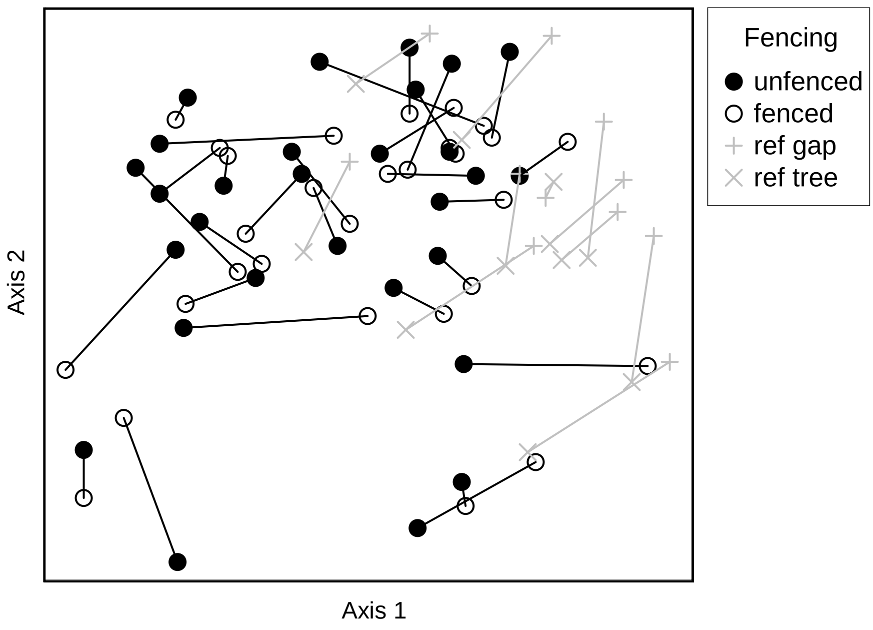


Figure 6. Relative magnitude (indicated by circle size) of floristic and environmental variables for each plot on the nMDS ordination of floristic composition (see Fig. 5 for symbol codes).

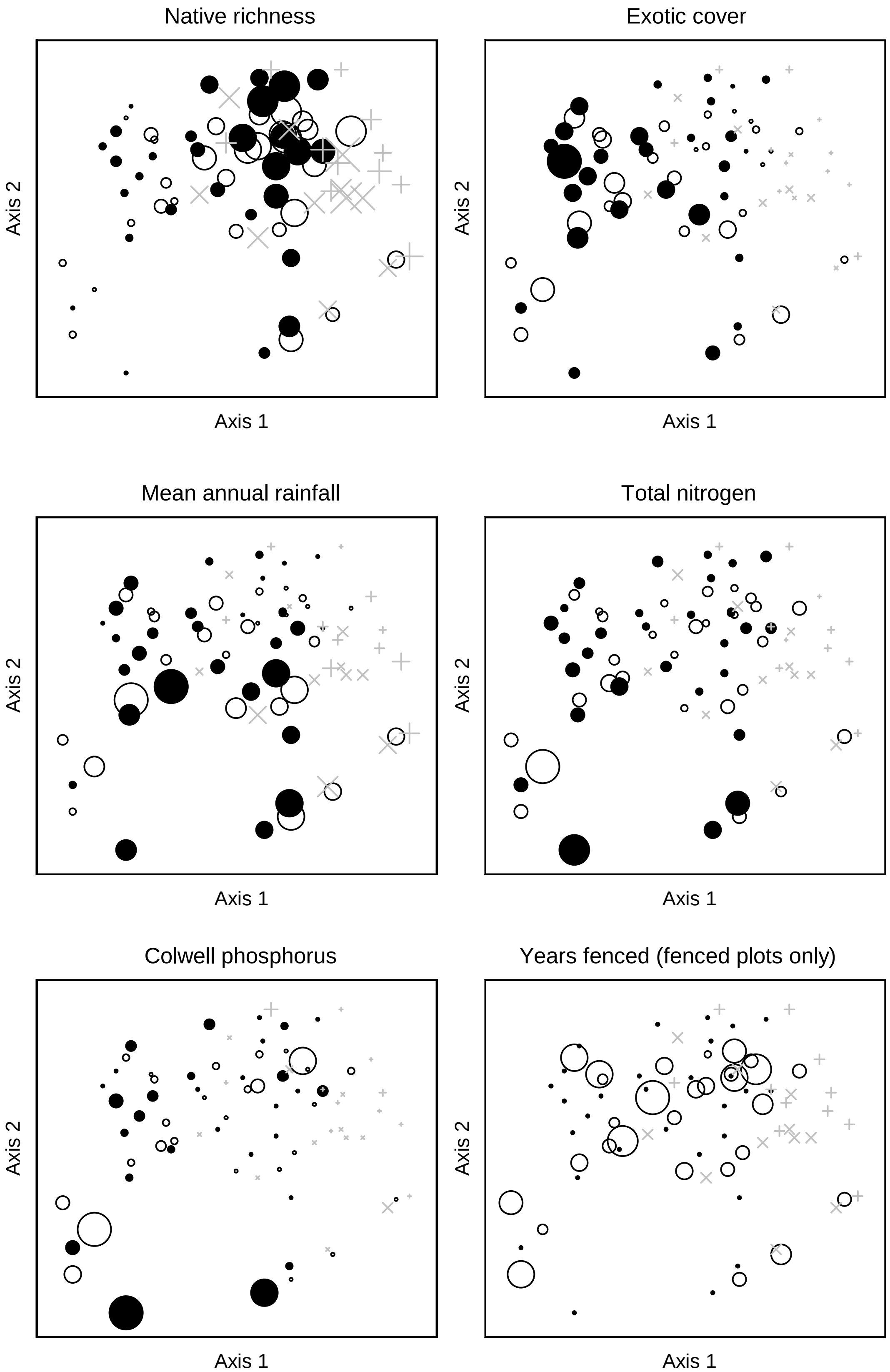


Table 4. Two-way table ordered on axis 1 of the nMDS ordination, showing change in cover of plant species in unfenced (U), fenced (F) and reference (**R**) plots. All understorey species occurring in > 16 plots are shown, as well as selected others. Abundance codes 1-4 low-high. *denotes exotic species.

	FFUUUUUUFUUFUUFUFFFUFUURFUFUFRRFUFU**R**FFUUU**R**UUFFUUFFU**R**UFFUFFF**R**U**R**U**RR**F**RRRRR**F**RRRRR**F**RR**
Bromus diandrus	1--3-----2-----2--1---------1------1--
Hordeum leporinum	1-22-3-3-322-3-11-121-2--1----1---1---1-----1-------------------------------
Lolium perenne	1111-3-2---3-3132112232-----2---114111------3-------1-------------------------
Erodium botrys	2---322-3-31322-22211-2-1-321-2-11---112--12-2-112-----1------2----------------
Ehrharta longiflora	23332-----33-11--2323--1---211-212-2----2112------2---1--112-1-21-----1---------
Bromus rubens	2-1223232-3212122-22232111--12122-2-21----2-12121--21--1---1-------1-----1-1----
Enchylaena tomentosa	-1----2-------1-1121---11----1------2--1---------1----------------1--1----------
Trifolium subterraneum	2-----1---22---11-22--2--22-1-111---111---2222----11-11--1---------------------
Avena barbata	3--3223-3-233-2-2333-1123-2311131112-222-1-2-12--1-1--11-222-21-21-2---12----1--
Romulea rosea	2311-3----31--3---1---2-----1-2---------3-1-------2-232--1-----1-311----------2
Arctotheca calendula	-111222211-22213313212312232221122--22-11-22111212-121112111-11--1---11-111-----
Crassula colorata	----2212---1-1211221121-122-211212-21122-111212-21----1---1-12----1---11--------
Ptilotus spathulatus	------2------1-----1-1-111-11111-1--2--1-1-----111-1----2----------------------
Austrostipa nitida	--1-33--3-3-32313-32123-----2-221-2-2331-1-2-223---23-21-3-2322----23--222---2--
Eucalyptus loxophleba	34433--3-433-3-333-433-331-3323334243--33-333--3333413122-33--24-2-3-4423-2-43--
Brassica tournefortii	-22----------1---1--11-111--1---21--2--1---1---11--2-1-11------1---1-11-----2---
Pentaschistis airoides	----22-23-213222312-22212--2322-21-22232-222-23112-2---122--321-1-1-21-2-112--2-
Vulpia myuros	---1-2223--222-23--22322----22-2-131-221-222221---1-2------11121---221-111111-12
Hypochaeris glabra	----1-1-1-1-2-1-11212121111211-21-22122-122-112-1--1112-12-1-111111-1-11----1111
Calotis hispidula	-----1--------1------1-11-----1-12-12--1-----1-111-1--121---1------1-1----------
Austrodanthonia caespitosa	-----------1-------21--1-----21--1221-------2-1--2-1---1-------1---2-11-----1---
Aristida contorta	------3---------------1--33--31-1----232-213-1-12-----11212-12------2--2-1------
Austrostipa trichophylla	2------2---------------23221-1---2--1---1--1----22122--2221-112---1--2--1----2--
Arthropodium curvipes	-----------------1--1-1-1211--2-11--21----11-22211-12-111-111-2---1--1----------
Austrostipa tenuifolia	-1----2----2-----22----22222-2--13-23322--22-321122---223-22--21212--212--1-2-1-
Calandrinia eremaea	-----1-------1112121-112-2-1-12-22-232211122-1211112-11121-3222----21-222--13-1-
Dichopogon preissii	--------------2--------1-----1-112-----2--2----1123--211--2----1--1------------
Acacia acuminata	2-------1----23--32--23-21222-3222323322222212-22--33233232222321213-2232-222321
Anagallis arvensis	--------------------1--11--221-2---------1---------3--21--------1-212-1--1----1--
Actinobole uliginosum	----------------------1--1--11-1-1----1----11---1--------2--1-------2--1-1------
Goodenia berardiana	----------------------21111-12---1--2--2--21-1-2-2-1-1211-1--1--1-1-112---2-----
Erymophyllum tenellum	------3-------1---------323-----12-21222-322-22211----12212-1--12--2-21-12-21-12
Erodium cygnorum	------------------1------11---1--1---221-2---12----11---2--1111---1--1-2--1-----
Austrostipa scabra	--------------------1--2---1-23--2--22-----1---121-2---212-212----2--231-----2--
Austrostipa elegantissima	------1---22-1----22-1--2--22--22213----1-22121-112122-122-2---22211-12221213222
Podolepis capillaris	------------------2---2-------1---12---1-2------12-1--1--2--21----2---11122-----
Siloxerus multiflorus	-------------------------12-----1---1--1--11-211-1----1-1--------------------1-2
Podolepis lessonii	-----------------------11-22121-21-11222--22-12212----112-1-12122-22-2--11-1--12
Calandrinia granulifera	--------------------------------1--1-12------1111--1-----2111---------1--1------
Ursinia anthemoides	-----------------221--12---21212--3211--311221111112-22-1112-112131121211-122212
Drosera macrantha	-------------------------11---------1---1-221--1--2--2----11-11---------11----1--
Velleia cycnopotamica	-------------------------121-21-22--2-12-222122211-1--211212121-2-2222111122----
Podotheca gnaphalioides	-----------------------1-1---2-------33------11-1--2-------1--1-----2--1----2-3-
Waitzia acuminata	--------------1--11--1-111-2221-2---233--1-1-2222--2----2212233---1232132312133-
Caesia micrantha	--------------------------1------1------2--1---1--1-111---2-----1----1----1----1
Neurachne alopecuroidea	-----------------1-11-21221-11-21212222221132122212112222222232322232232222232212
Rhodanthe manglesii	------------------------1-----32---21--2-11--22--11-3-2123122-1-1112--111-111--2
Dichopogon capillipes	----------1---------1-------11---------1----1----2-----1-32-21111-12-2121-12----
Borya sphaerocephala	--------------1----------121-121-1---122-232-221-2----212221132-3-3222231222--13
Hydrocotyle piliferavglab	--------------------------------1--11--1--2--1--122--111--11-11------1111---12--
Trachymene cyanopetala	------------------------1--11-1-11-11--2--11-22122---1121-2222112212-1221112--12
Podolepis canescens	--------------1---------------3----------3-2-1--1--2---1-21----1-2--2----1--2-22
Chamaescilla corymbosa	--------------------------------1---1--1---1---2-1--11--2-2--------1--12---2-1--
Trachymene ornata	--------------------------------11-112-2--1--21111----221122111-1111-1121221-1--
Ptilotus declinatus	--------------------------1---111--------111---------11-------111---21---11--11
Amphipogon caricinus	---------------------------------11-2--1---22--2-2--2--2--3-----3---1212221---2
Briza maxima	--------------------------12-------1----2-3-------2--22---12-22--2-2-1112--11212
Gilberta tenuifolia	---------------------------------1-----------2-1-------2-3--111-3--2--132211----
Thysanotus patersonii	--1--1-1-------------2-2-2-11---2------111
Dampiera lavandulacea	-----------------1-------------------------1-----2-1---2---2-123-12--212222-2-22
Lawrencella rosea	---11-2-------11---2--112-121-1--
Gnephosis tenuissima	--------------------------------1---------------1-----------11-----2--112----22

4.3 Tree and shrub demography

The frequency and recruitment of trees and tall shrubs were generally higher in fenced compared with unfenced plots (Fig. 7a,b). This trend was significant for frequency of Jam, York gum, and other trees and shrubs, and for frequency of jam and York gum recruits (Fig. 7a,b). Only for Needle tree were there no effects of fencing. For jam, 72% of fenced plots had recruits compared with 38% of unfenced plots, and fencing effects were readily apparent during the survey (Plate 5). For York gum, differences were less apparent in the field because only 8 fenced plots had any York gum recruits (28%, compared with 10% of unfenced plots), and there were usually fewer recruits in any one site (Fig. 7b). Notably the three fenced plots with the highest number of York gum recruits (4, 9 and 18 individuals) had undergone natural disturbances (fire, flood) within the past ten years. Recruitment of other tree and shrub species was low across all fenced, unfenced and reference plots (Fig. 7b).

We did not detect an effect of fencing on the mean DBH of any species (Fig. 7c). The frequency of standing dead jam, other trees and other tall shrubs was greater in fenced compared with unfenced plots (Fig 7d). However, we note this was compensated by the greater length of fallen logs in unfenced compared with fenced plots (Table 1).

The difference between the frequency of jam individuals in fenced vs unfenced pairs increased as the number of individuals in unfenced plots decreased (implying jam recovered more when plots were more open to begin with). Once this had been accounted for, the difference between fenced and unfenced plots was best explained by a positive relationship with landscape integrity (within a 1000 m radius), the number of years since fencing and soil potassium levels (R^2=60.5%, Table 3, Fig. 8). No linear regression models significantly explained the occurrence of recent jam recruits, but there was an apparent flush in this size class at about seven years post-fencing (Fig. 9). The models we explored did not significantly predict the difference in frequency of York gum or occurrence of York gum recruits between fenced and unfenced pairs.

Only York gum and jam frequency (all size classes) and jam recruitment could be compared with reference sites using t-tests, owing to high numbers of zeros for other variables. York gum frequency in reference plots did not differ significantly from fenced or unfenced plots. Jam frequency and recruitment were higher in reference plots than unfenced plots, but not significantly different from fenced plots.

Figure 7. Demography of trees and shrubs recorded in fenced, unfenced and reference plots: a) frequency of live individuals (all size classes); b) frequency of recruits (i.e., live plants with ≤ 0.5 cm DBH or < 1.4 m tall); c) mean size of three most frequent species and; d) frequency of dead individuals (all size classes). Jam = *Acacia acuminata*, York gum = *Eucalyptus loxophleba* subsp. *loxophleba*, Needle tree = *Hakea preissii*; Other trees = *Allocasuarina campestris, A. huegeliana, Eucalyptus salmonophloia, E. wandoo*; Other shrubs = *Acacia acuaria, A. microbotrya, Exocarpos aphyllus, Grevillea paniculata, Santalum spicatum, Senna artemisioides.* Frequencies are mean number of individuals per 1000 m^2; ***P<0.001 and **P<0.01 indicate results of comparisons between fenced and unfenced plots, determined by GLM models of frequency data. There were no significant differences in mean DBH of three species between fenced and unfenced plots.

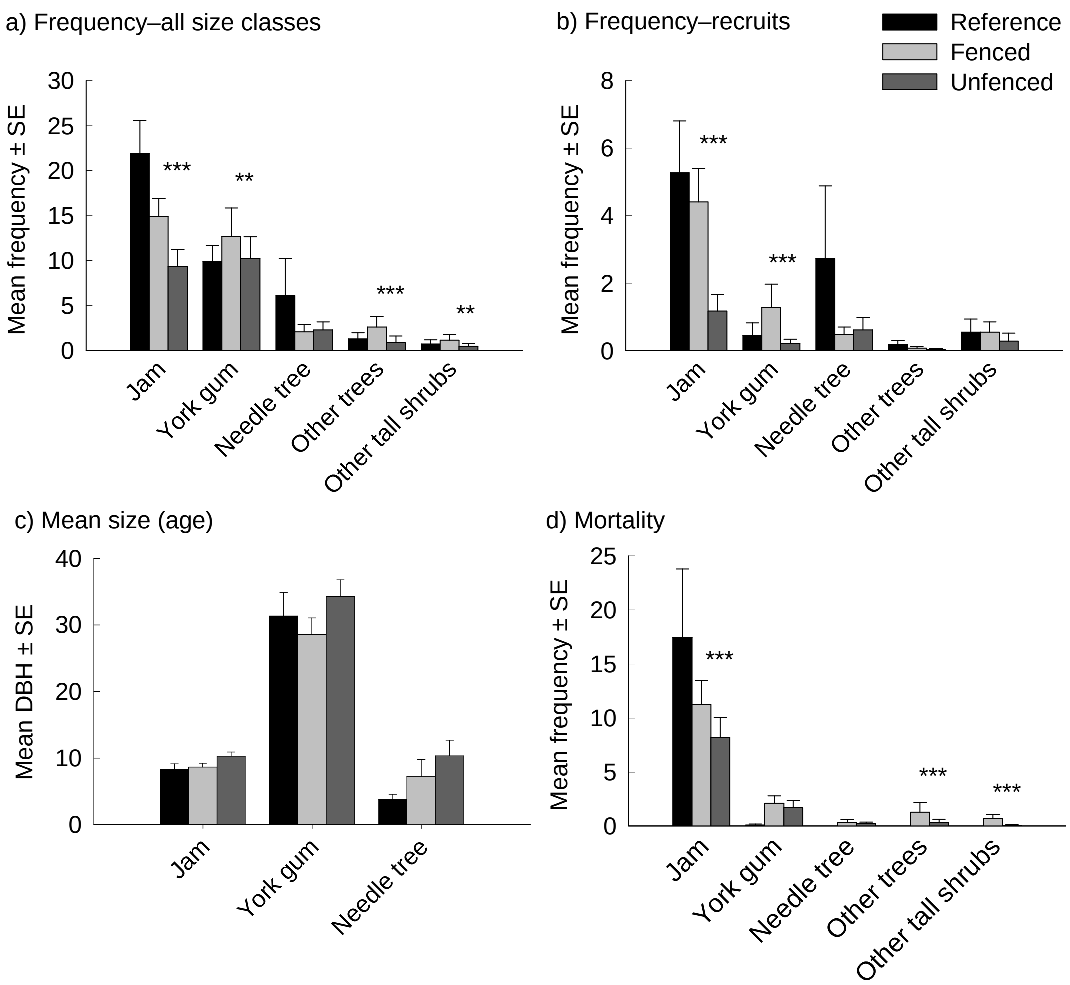


Figure 8. Effects of time since fencing and landscape integrity on differences between fenced and unfenced plots in frequency of jam individuals, after adjustment for frequency in unfenced plots.

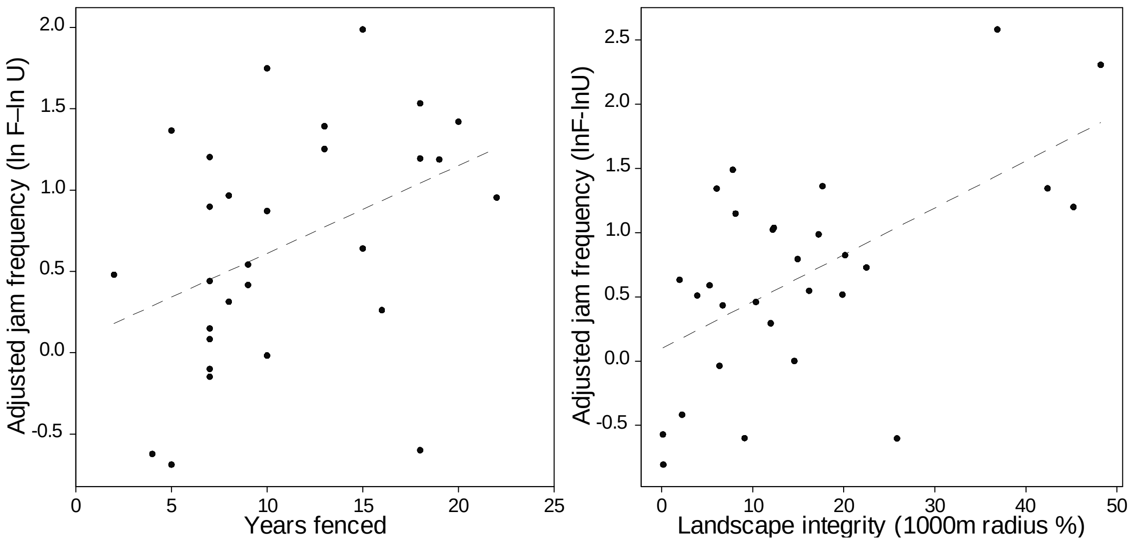


Figure 9. Difference in frequency of jam recruits between fenced and unfenced plots, in relation to years since fencing.

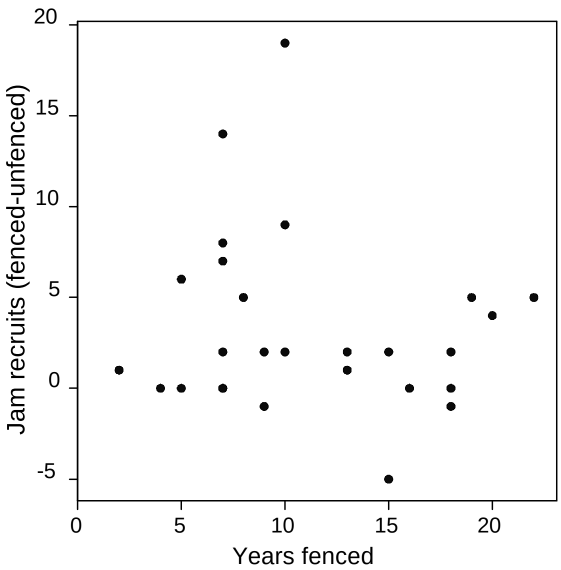


Plate 2. Fenced sites often had abundant recruitment of jam wattle, evident in the area fenced since 1998 (right), compared with little recruitment in the unfenced area (left). © 2009 CSIRO. Photographer: Suzanne Prober.

5. DISCUSSION

5.1 Benefits of fencing

Within the constraints of our sampling strategy, our results suggest that if appropriately targeted, fencing is a valuable tool for enhancing vegetation condition in degraded herb-rich York gum – jam woodlands. Our first hypothesis, that fencing facilitates recovery towards reference conditions, was supported by several lines of evidence. First, fenced sites were on average, significantly different from unfenced sites for most vegetation measures. Secondly, the direction of these differences was usually towards reference conditions. These patterns were consistent for native richness of most plant groups, native cover, exotic cover, and recruitment of jam and York gum (but not needle tree). Thirdly, fenced plots were more similar to reference plots in floristic composition than unfenced plots were.

These benefits of fencing for native biodiversity are consistent with earlier studies in temperate eucalypt woodlands (Fox 2001; Pettit and Froend 2001; Duncan et al. 2007; Briggs et al. 2008; Spooner and Briggs 2008). Increased native plant cover was observed in all earlier studies, and native plant richness increased in the two studies that measured this (Pettit and Froend 2001; Briggs et al. 2008), although to a greater extent in shrubby than grassy ecosystems. Decreased exotic cover was clearly apparent in only two of these other studies (Duncan et al. 2007; Spooner and Briggs, 2008); the reasons for differences between studies are not clear and are worthy of further investigation.

The effectiveness of fencing for promoting tree recruitment has been variable across studies, suggesting that it is species dependent (Spooner and Briggs 2008, Briggs et al. 2008). This is consistent with our study, where fenced plots had significantly more recruitment of jam than unfenced plots, a smaller proportion of fenced sites had significant York gum recruitment, and we did not detect consistent differences in recruitment or frequency of needle tree. Effective recruitment of jam after fencing concurs with casual observations by many landholders involved in this study. As a palatable legume, it is not surprising that jam recruits are suppressed by livestock grazing, but with a long-lived soil seed store can recover rapidly after exclusion of grazing. Conversely, as its name implies, needle tree has pungent spines that may lead to greater natural defences against grazing.

For York Gum, it is notable that while there was a pulse of recruitment at some fenced plots, the frequency of individuals and recruits did not differ significantly between reference and unfenced plots. Hence ‘recovery’ owing to release from grazing would not necessarily be expected. Low levels of recruitment across all sites do however, raise broader issues regarding the long term dynamics of York gum populations that are worthy of further investigation. In particular, Hobbs and Atkins (1991) suggested that recruitment of York gum is limited by lack of fire, consistent with our observation that the three fenced plots where most York gum recruitment was recorded had been burnt or flooded within the past ten years. It is thus likely that fencing is a necessary but not sufficient condition for promoting recruitment of York gum.

Although we recorded generally superior vegetation condition in fenced compared with unfenced plots, our first hypothesis was not supported for soil conditions, where we found no evidence for recovery of enriched woodland topsoils after fencing. Topsoil enrichment is a well-established consequence of livestock grazing and adjacent cropping in remnant vegetation in southern Australia (Scougall et al. 1993; Petit et al. 1995; Yates et al. 2000b; Fox 2001; Prober et al. 2002b), and our analyses confirmed that a proportion of the fenced and unfenced plots in our study were nutrient enriched. Few studies have directly evaluated whether recovery of soil chemical properties is promoted by cessation of livestock grazing, but a lack of recovery is not unexpected. For example, Duncan et al. (2007) concluded that time since fencing was not a strong driver of soil phosphorus levels in Victorian woodlands, although they had no data on levels prior to fencing. Studies of old fields have similarly shown that altered soil conditions, including elevated soil phosphorus, can persist for decades after cultivation has ceased in woodland soils in WA (Standish et al. 2006). Most studies have focused on soil phosphorus, but our data indicated that other nutrients such as total nitrogen, organic carbon and potassium were also higher in fenced compared with reference plots. Notwithstanding, the extent of the differences was not large for some nutrients. For example, Colwell phosphorus in fenced plots was on average less than 5 mg/kg, which is not much greater than in our reference sites (2.4 mg/kg) or in other little-modified eucalypt woodlands of eastern Australia (1-5 mg/kg, Prober et al. 2002a).

Soil physical conditions did not follow hypothesized patterns based on expectations drawn from the literature. In particular, other Western Australian studies have suggested that soil bulk density and levels of bare ground increase with livestock grazing or cultivation (Yates et al. 2000b; Standish et al. 2006), whereas in our study, grazed, unfenced sites were similar to reference sites in these characteristics. Lower bulk density and bare ground associated with fencing might generally be seen as positive outcomes for plant growth and soil health, but given the deviation from reference sites it is difficult to interpret whether such outcomes are favourable for biodiversity conservation. Exotic annuals are also associated with decreasing soil bulk density and bare ground (Prober et al. 2002, Prober and Wiehl unpub. data), so our results could reflect an interaction between release from grazing and a greater cover of exotic annuals in fenced compared with reference sites. Soil surface hardness on the other hand, followed more expected patterns, being lowest on reference sites and highest on unfenced plots, with ambiguous suggestions of improvement in fenced but not long-fenced sites.

Our study did not directly address benefits of fencing for woodland fauna. Differences in vegetation condition between fenced and unfenced plots observed in our study that might enhance native fauna habitat include higher tree densities, greater structural diversity associated with higher tree recruitment and shrub frequency, and higher native ground cover and forb richness (Barrett et al. 2008; Montague-Drake et al. 2009). Fallen logs are often cited as an important element of fauna habitat (e.g. Mac Nally 2006), but in our study we recorded lower lengths of fallen logs in fenced compared with unfenced plots. This was compensated however, by higher numbers of standing dead trees in fenced plots (suggesting that dead trees were more likely to have fallen over in grazed plots). Similarly few data are available to evaluate outcomes of fencing for fauna in other eucalypt woodlands. Briggs et al. (2008) detected few improvements in habitat variables such as tree health, tree hollows and fallen logs in eastern Australian, and concluded that recovery of fauna habitat was slow.

5.2 Limits to recovery

Our second hypothesis, that recovery of York gum – jam woodlands to reference conditions is limited by ecological or other constraints, was supported for most condition measures (except jam recruitment). First, average soil and floristic parameters of fenced plots were significantly different from averages for reference sites: soil nutrients were elevated, exotic cover was higher, and native richness was lower. Secondly, recovery to reference condition, as inferred from differences between unfenced and fenced plots on floristic and soil ordinations, was suggested in only a small subset of cases. Thirdly, a number of environmental variables were significantly associated with the degree of difference between fenced and unfenced pairs, suggesting they limited recovery (see below). Few previous studies have explicitly assessed recovery of fenced sites in relation to reference conditions. Exceptions are

Pettit and Froend (2001), who observed incomplete recovery to reference floristic composition in jarrah woodland after seven years of exclosure from livestock grazing, and Fox (2001), who found that Salmon gum woodlands fenced for up to 30 years had not returned to reference conditions.

It is possible that for some fenced plots in our study, differences between fenced and reference sites reflect a lack of sufficient time for recovery. For example, change in exotic cover was significantly associated with time since fencing, and it will be of particular interest to see whether these decline further over the next 5-10 years. However, time since fencing was not significantly associated with differences in soil nutrients or native richness, and regression models for all vegetation measures suggested environmentally-driven limits to recovery.

We hypothesized that such limits would include ongoing livestock grazing within fenced sites, propagule limitation, nutrient enrichment and weed invasion. Briggs et al. (2008) found that in eastern Australia, native richness was higher in fenced sites without livestock grazing than those with ongoing, lower levels of livestock grazing. However in our study, few fenced sites were grazed by livestock, and so effects of grazing on recovery could not be evaluated.

Regression analyses pointed to total nitrogen as a limit to recovery from exotic invasion. Models containing various combinations of the variables organic carbon, mean annual rainfall, sulphur and Colwell phosphorus explained up to 69% of the variation in total N. Hence we interpret these regression results more broadly as indicating that exotics are more persistent in higher productivity environments, consistent with related studies that emphasize attention to soil nutrient levels for restoration of temperate eucalypt woodlands (Prober and Thiele 2005; Dorrough et al. 2006; McIntyre and Lavorel 2007; Standish et al. 2009; Prober et al. 2009).

In turn, exotic cover in fenced plots was the variable most strongly associated with the extent of the difference in native species richness between fenced and unfenced plots. Because exotic cover in fenced plots showed a stronger relationship than exotic cover in unfenced plots, we hypothesize that recovery of species richness is dependent more on capacity for recovery from exotic invasion (as driven by productivity variables) than by initial exotic cover alone. Dependence of native richness on exotic cover is consistent with a strong quadratic relationship between exotic cover and native richness across the 29 pairs of plots in this study (R^2=76%) and a related study (R^2=66%, Prober and Wiehl unpub. data) of York gum – jam woodlands, and with studies in other ecological communities that indicate that exotic annuals limit recruitment and growth of native herbaceous species (Alvarez and Cushman 2002; Lenz and Facelli 2005; Prober et al. 2005; Smallbone et al. 2007; Standish et al. 2008).

One further variable, landscape integrity at the 1 km scale, provided significant additional contribution to best models of change in native species richness and jam recruitment. This variable was intended to address the hypothesis that recovery of native plants species is limited in isolated remnants due to lack of propagules (e.g. Standish et al. 2007; Prober and Smith 2009). However, up to 57.5% of the variation in landscape integrity could be explained by variables reflecting landuse and productivity. This included increasing landscape integrity with decreasing historical grazing levels or higher topographic positions and rock cover; lower soil surface compaction; and lower soil available sulphur or phosphorus (likely to reflect the amount of cropping in the landscape). Indeed, when we replaced landscape integrity with historical grazing levels in the regression for native richness, total variance explained declined by only 1%. By contrast, historical grazing levels did not significantly contribute to models for explaining change in the frequency of jam individuals. Jam naturally becomes a more prominent component of York gum – jam woodlands in poorer (less productive) parts of the landscape (S. Prober, pers. obs.), providing an alternative explanation for greater change in jam frequency in areas with greater total native vegetation cover. Contribution of potassium to the model for jam may similarly reflect significant negative correlations of potassium with productivity measures such as total nitrogen and Colwell phosphorus, but we are uncertain of the relevance of this variable.

We conclude that, while our measure of landscape integrity is a potentially simple objective predictor of potential for vegetation recovery, it is a complex variable that can be difficult to relate directly to underlying ecological drivers. In particular, it is not possible to determine whether higher propagule availability played any role in its relationship with species richness. This covariance between measures of landscape integrity, productivity and landuse should be carefully considered in future studies analyzing effects of landscape-scale vegetation measures on biodiversity. Further, although better vegetation recovery might be achieved in landscapes with higher existing native vegetation cover, it may be more critical to increase representation of native species and communities in landscapes with lower existing vegetation cover.

A number of ecological models have been used to describe ecological degradation and recovery. One common approach describes restoration as a more or less ordered and gradual change along a desired ecological trajectory (Luken 1990). A second model, the state and transition approach, suggests that multiple alternative pathways are possible and that different metastable states can exist under similar environmental conditions. Transitions between states can be rapid and non-linear, and reversing transitions is dependent on overcoming ecological thresholds through management inputs (Westoby et al*.* 1989). While our data suggest limits to recovery, we have insufficient evidence to test whether factors such as productivity and exotic cover simply slow rates of recovery, or whether sites are likely to remain in alternative stable states. Relationships between exotic cover and time since fencing allow for the possibility that sites would recover unaided given sufficient time, although this is not consistent with the persistence of elevated soil nutrients that are known to promote these weeds (Prober and Smith 2009). Lack of relationships with time since fencing for other variables do suggest more insurmountable barriers to recovery, consistent with studies in wheatbelt and other eucalypt woodlands (Prober et al. 2009; Standish et al. 2009), and suggesting a more pressing need for additional management interventions. However it is also possible that our relatively small dataset was not sufficiently powerful to detect relationships with time since fencing amongst a number of other variables. Indeed, although 40-60% of variance in apparent recovery of ecological variables could be attributed to explanatory variables, a large amount of variance remained unexplained.

Finally, it is important to emphasize that because our study required *post-hoc* selection of fenced and unfenced pairs, we cannot preclude the possibility that a systematic bias influenced our results. We took all possible measures to ensure fenced and unfenced plots within pairs reflected similar starting conditions. Some confidence in our assumptions is provided by the lack of significant contribution to regressions of variables such as the difference in distance to paddock edges or difference in landscape integrity between fenced and unfenced pairs. Similarly, we detected no differences in soil nutrients between unfenced and fenced plots that might be attributed to differing distance to crop edges. Further, we emphasize that comparisons of fenced sites with reference sites are free from these potential biases. Nonetheless, future coordinated investments in fencing programs (such as those established by WWF) would benefit from establishing appropriate monitoring pairs (fenced and unfenced plots) at strategically selected sites, representing a range of initial vegetation states and productivity levels. Pre-treatment data would be valuable for ensuring no initial systematic differences occurred between these fenced and unfenced pairs. This study has provided baseline data along these lines, which will be valuable for indicating whether ongoing recovery occurs at these permanently marked plots.

5.3 Management implications

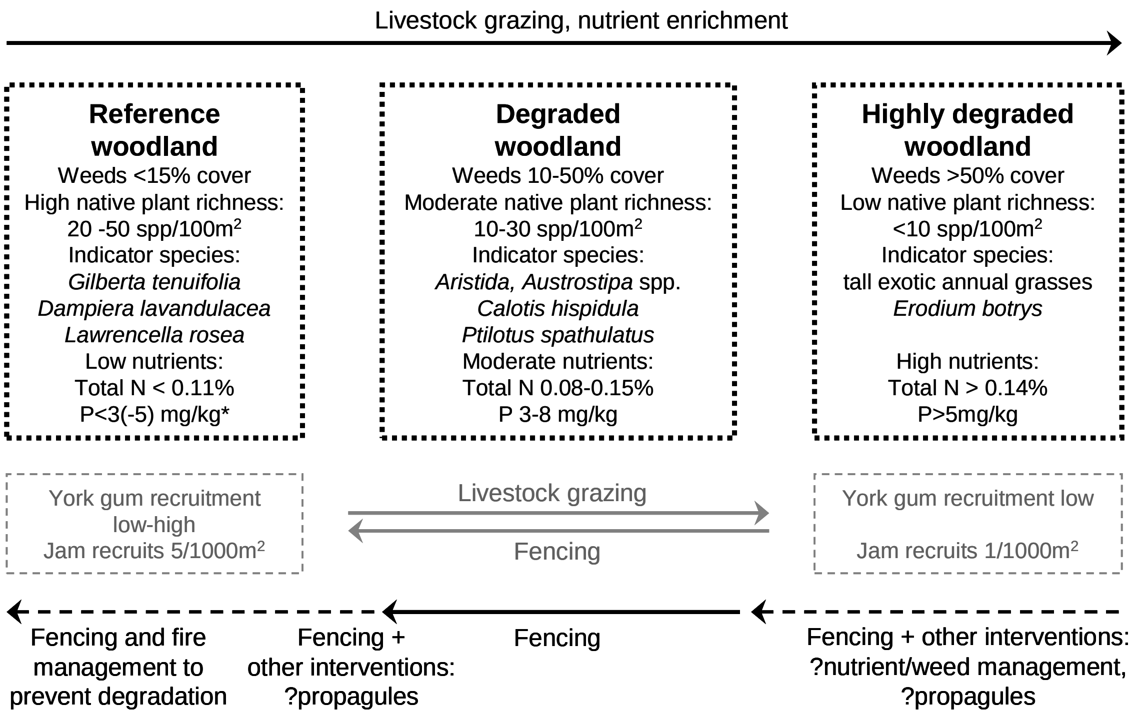
Our data support ongoing investment in fencing to exclude livestock grazing for enhancing biodiversity conservation values in remnant York gum – jam woodlands. Likely benefits include increased native richness and cover, reduced exotic abundance and enhanced recruitment of jam and York gum (Table 5), as well as prevention of further degradation.

Table 5. In a nutshell: summary of key similarities and differences between unfenced, fenced and reference woodlands of this study.

Unfenced better than fenced	Fenced same as unfenced	Fenced better than unfenced	Fenced poorer than reference
		Higher native richness	Lower native richness
		Higher native cover	Lower native cover
		Lower exotic cover	Higher exotic cover
	All soil nutrients and pH		Higher soil nutrients (N,P,K,S,C)
	Soil cryptogam cover		
More fallen logs (but fewer standing dead trees)	Needle tree frequency and recruitment	Higher frequency of overstorey species	
	Mean DBH of jam, York gum and needle tree	Higher recruitment of jam & York gum	

However, all of these benefits will not occur at all sites within medium (20 year) time-frames, and full recovery to reference condition will not necessarily occur due to fencing alone, even over long time frames (Table 5). Rather, additional interventions may be needed to achieve some conservation goals. Figure 10 proposes an indicative framework to guide setting of conservation targets and to clarify where additional interventions may be of highest priority. The framework focuses on three generalized woodland states in a degradation sequence from high quality reference woodlands (e.g. Plate 3), through degraded woodlands with low-moderate levels of exotic invasion and nutrient enrichment (e.g. Plate 2), to highly degraded woodlands with high levels of exotic invasion and nutrient enrichment (e.g. Plate 1). Each state captures considerable ecological variation, and as indicated by solid arrows on the restoration (reverse) axis (Fig. 10), we suggest that fencing alone is likely to be most effective for promoting jam recruitment and for enhancing condition of moderately degraded woodlands.

Figure 10. Framework for guiding restoration decisions in York gum – jam woodlands, based on three generalized woodland states (see text for further detail). Values provided are indicative only. Dashed arrows indicate uncertainty regarding capacity of the ecosystem to recover without additional interventions; solid arrows represent greater confidence in ecosystem transitions. *some exceptions noted e.g. on dolerite



Improvements in understorey condition in highly degraded remnants are likely to be slower, constrained by nutrient enrichment and exotic invasion. Indeed as indicated by dashed arrows, it is possible that significant recovery would not occur in the longer term without additional interventions. Similarly, we did not find strong evidence for recovery of degraded woodlands to reference condition, hence additional interventions may also be needed to achieve this peak level of improvement.

Recruitment of York gum appeared to be inherently uncommon, although we detected some difference between fenced and unfenced sites. Managers should not become dissolute about not seeing immediate results of their actions for this overstorey species, especially where densities and sizes of York gum are similar to reference sites. However a better understanding of episodic recruitment in would be beneficial for enhancing recruitment in sites with sparse or overmature York gum.

Options for additional management interventions to enhance vegetation condition are not well established and require further investigation (Prober and Smith 2009; Standish and Hobbs 2009). Restoration principles suggest attention to underlying ecological barriers including soil nutrient enrichment and limited soil seed banks is likely to be needed (Prober and Smith 2009), and this is reinforced by the lack of recovery of soil nutrient levels indicated by our data. Studies in other ecological communities to address invasion by exotic annuals demonstrate some success with management of weed seed banks through spring burning or strategic grazing (Menke 1992; Prober et al. 2005), and manipulation of soil nitrogen cycling. The latter involves addition of carbon (typically in the form of sugar) to the soil surface, followed by re-establishment of dense native cover through propagule addition (Blumenthal et al. 2003; Prober et al. 2005; Prober and Lunt 2009), but the success of these methods is dependent on the capacity of the restored native understorey to outcompete weeds in the long term. Techniques for sequestering soil phosphorus are poorly established, with efforts to date focusing on nutrient stripping through cropping or scalping of topsoil (e.g. Walker et al. 2004), which would be difficult to implement in woodland remnants. An alternative approach would be to match the vegetation to the nutrient regime by identifying and restoring native species that are able to compete in the nutrient-enriched conditions (Standish et al. 2009).

Similarly, although not confirmed by our data, recovery of native species richness to reference levels is likely to be constrained by lack of native propagules (e.g. Standish et al. 2007). For example, many perennial forb species lack long-lived seed banks and are poorly dispersed (Lunt 1997). Hence targeted addition of native seed may facilitate recovery, and seed technologies to enhance success of seeding operations is a promising avenue for future development. We note also that other types of degradation not encountered in our study (hence not shown in Fig. 10) may also occur in York gum – jam woodlands. For example, soil compaction and/or depletion of soil carbon has been shown to occur in other types of temperate eucalypt woodlands (Prober et al. 2002b, Yates et al. 2000b, Standish et al. 2006), and this would require different types of interventions to stimulate soil processes (e.g. Yates et al. 2000a).

Finally, Figure 10 focuses on site-scale factors, but we emphasize that the landscape context of remnant woodlands should also be considered in the establishment of conservation targets in fencing programs. For example, the degree of investment required to restore highly degraded remnants may be higher than for moderately degraded woodlands. However where these highly degraded woodlands represent the only remnants in an intensively-used landscape, they may still be of considerable value to landscape processes and to the local community.

Plate 3. An example of a reference herb-rich York gum – Jam woodland sampled in this study, with native perennial grasses and annual forbs of the daisy family (Asteraceae) dominating the understorey. © 2009 CSIRO. Photographer: Suzanne Prober.

6. ACKNOWLEDGEMENTS

We thank the many landholders who generously gave time and information for this study, or allowed us to survey their remnant vegetation. Damian Shepherd (Department of Agriculture and Food WA), Helena Mills, Mike Griffiths and Mick Davis (World Wide Fund for Nature WA), and Anne Smith (Greening Australia) assisted with data for site selections, Steve Zabar (CSIRO) assisted with data collection, and Jane Speijers (Department of Agriculture and Food WA) provided statistical advice. This project would not have been possible without the support of Chris Curnow (WWF-Australia), Neil Riches (Caring for our Country facilitator WA) and Jeff Richardson (Department of Environment and Conservation WA).

REFERENCES

Alvarez ME, Cushman JH (2002) Community-level consequences of a plant invasion: effects on three habitats in coastal California. *Ecological Applications* 12, 1434–1444.

Barrett GW, Freudenberger D, Drew A, Stol J, Nicholls AO, Cawsey EM (2008) Colonisation of native tree and shrub plantings by woodland birds in an agricultural landscape. *Wildlife Research* 35, 19-32.

Beard JS (1990) Plant life of Western Australia. Kangaroo Press: Sydney.

Blair GJ, Chinoim N, Lefroy R B, Anderson G C and Crocker G J (1991) A soil sulfur test for pasture and crops. *Australian Journal of Soil Research* 29, 619–626.

Blumenthal DM, Jordan NR, Ruselle MP (2003) Soil carbon addition controls weeds and facilitates prairie restoration. *Ecological Applications* 13, 605–615.

Briggs SV, Taws NM, Seddon JA, Vanzella B (2008) Condition of fenced and unfenced remnant vegetation in inland catchments in south-eastern Australia. *Australian Journal of Botany* 56, 590-599.

Department of Industry and Resources (2001) Atlas of Mineral Deposits and Petroleum Fields. Geological Survey of Western Australia.

Dorrough J, Moxham C, Turner V, Sutter G (2006) Soil phosphorus and tree cover modify the effects of livestock grazing on plant species richness in Australian grassy woodland. *Biological Conservation* 130, 394–405.

Duncan D, Moxham C, Read C (2007) Effect of stock removal on woodlands in the Murray Mallee and Wimmera bioregions of Victoria. Department of Sustainability and Environment, Victoria.

Faith DP, Minchin PR, Belbin L (1987) Compositional dissimilarity as a robust measure of ecological distance. *Vegetatio* 69, 57-68.

Fox P (2001) Conservation of salmon gum (*Eucalyptus salmonophloia* F. Muell.) woodlands in the Western Australian wheatbelt: a consilience of biology, culture and perception. PhD Thesis, Murdoch University: Perth, Australia.

Greig-Smith P (1983) Quantitative Plant Ecology. Third edition. Blackwell Scientific Publications: Oxford. 359 pp.

Hobbs RJ (2003) How fire regimes interact with other forms of ecosystem disturbance and modification. In: Abbot, I, Burrows, ND (Eds.) Fire in ecosystems of south-west Western Australia: impacts and management. Backhuys Publishers: Leiden, pp. 421-436.

Hobbs RJ, Atkins L (1988) Effects of disturbance and nutrient addition on native and introduced annuals in plant communities in the Western Australian wheatbelt. *Australian Journal of Ecology* 13, 171-179.

Hobbs RJ, Atkins L (1991) Interactions between annuals and woody perennials in a Western Australian nature reserve. *Journal of Vegetation Science* 2, 643-654.

Hobbs R J and Saunders D A (1993) Re-integrating fragmented landscapes. Towards sustainable production and nature conservation. Springer-Verlag: New York.

Houlder DH, Hutchinson MF, Nix HA, McMahon JP (2001) ANUCLIM user guide. Version 5.1. Centre for Resource and Environmental Studies, Australian National University, Canberra.

Lenz TI, Facelli JM (2005) The role of seed limitation and resource availability in the recruitment of native perennial grasses and exotics in a South Australian grassland. *Austral Ecology* 30, 684–694.

Luken JO (1990) Directing ecological succession. Chapman and Hall: New York.

Lunt I (1997) Germinable soil seed banks of anthropogenic native grasslands and grassy forest remnants in temperate south-eastern Australia. *Plant Ecology* 130, 21-34.

Mac Nally R (2006) Longer-term response to experimental manipulation of fallen timber on forest floors of floodplain forest in south-eastern Australia. *Forest Ecology and Management* 229, 155-160.

McCune B, Mefford MJ (1999) PC-ORD. Multivariate analysis of ecological data, Version 4. MjM Software Design: Oregon, USA.

McIntyre S, Lavorel S (2007) A conceptual model of land use effects on the structure and function of herbaceous vegetation. *Agriculture Ecosystems and Environment* 119, 11-21.

Minchin PR (1989) DECODA user's manual. Australian National University: Canberra.

Montague-Drake RM, Lindenmayer DB, Cunningham RB (2009) Factors affecting site occupancy by woodland bird species of conservation concern. *Biological Conservation* 142, 2896–2903.

Pettit NE, Froend RH (2001) Long-term changes in the vegetation after the cessation of livestock grazing in *Eucalyptus marginata* (jarrah) woodland remnants. *Austral Ecology* 26, 22-31.

Pettit NE, Froend RH, Ladd PG (1995) Grazing in remnant woodland vegetation: changes in species composition and life form groups. *Journal of Vegetation Science* 6, 121-130.

Prober SM, Smith FP (2009) Enhancing biodiversity persistence in intensively-used agricultural landscapes: a synthesis of 30 years of research in the Western Australian wheatbelt. *Agriculture Ecosystems and Environment* 132, 173-191.

Prober SM, Lunt I, Thiele KR (2002a) Determining reference conditions for management and restoration of temperate grassy woodlands: relationships among trees, topsoils and understorey flora in little-grazed remnants. *Australian Journal of Botany* 50, 687-697.

Prober SM, Thiele KR, Lunt I (2002b) Identifying ecological barriers to restoration in temperate grassy woodlands: soil changes associated with different degradation states. *Australian Journal of Botany* 50, 699-712.

Prober SM, Lunt I (2009) Restoration of *Themeda australis* swards suppresses soil nitrate and enhances ecological resistance to invasion by exotic annuals. *Biological Invasions* 11, 171-181.

Prober SM, Lunt I, Morgan J (2009) Rapid internal plant-soil feedbacks lead to alternative stable states in temperate Australian grassy woodlands. In: Suding K, Hobbs RJ (Eds.) New Models for Ecosystem Dynamics and Restoration. Island Press: Washington, USA, pp. 156-168.

Prober SM, Thiele KR, Lunt I, Koen TB (2005) Restoring ecological function in temperate grassy woodlands - manipulating soil nutrients, annual exotics and native perennial grasses through carbon supplements and spring burns. *Journal of Applied Ecology* 42, 1073-1085.

Rayment GE, Higginson FR (1992) Australian laboratory handbook of soil and water chemical methods. Inkata Press: Melbourne.

Scougall SA, Majer JD, Hobbs RJ (1993) Edge effects in grazed and ungrazed Western Australian wheatbelt remnants in relation to ecosystem reconstruction. In: Saunders DA, Hobbs RJ, Ehrlich PR (Eds.) Nature Conservation 3: Reconstruction of Fragmented Ecosystems. Surrey Beatty and Sons: Sydney, Australia, pp. 163-178.

Searle PL (1984) The Berthelot or indophenol reaction and its use in the analytical chemistry of nitrogen. *Analyst* 109, 549–568.

Shepherd DP, Beeston GR, Hopkins AJM (2002) Native vegetation in Western Australia: Extent, type and status. Department of Agriculture, Western Australia, Perth.

Smallbone L, Prober SM, Lunt ID (2007) Restoration treatments enhance early establishment of native forbs in a degraded grassy woodland. *Australian Journal of Botany* 55, 818-830.

Spooner PV, Briggs SV (2008) Woodlands on farms in southern NSW: a longer-term assessment of vegetation changes after fencing. *Ecological Management and Restoration* 9, 33-41.

Standish RJ, Cramer VA, Hobbs RJ (2008) Land-use legacy and the persistence of invasive *Avena barbata* on abandoned farmland. *Journal of Applied Ecology* 45, 1576–1583.

Standish RJ, Cramer VA, Hobbs RJ, Kobryn HT (2006) Legacy of land-use evident in soils of Western Australia's wheatbelt. *Plant and Soil* 280, 189-207.

Standish RJ, Cramer VA, Wild SL, Hobbs RJ (2007) Seed dispersal and recruitment limitation are barriers to native recolonization of old-fields in Western Australia. *Journal of Applied Ecology* 44, 435-445.

Standish RJ, Cramer VA, Yates CJ (2009) A revised state-and-transition model for the restoration of eucalypt woodlands in Western Australia. In: Suding K, Hobbs RJ (Eds.) New Models for Ecosystem Dynamics and Restoration. Island Press: Washington, pp. 169-188.

Standish RJ, Hobbs (2009) Restoration of OCBILs in south-western Australia: Response to Hopper. Plant Soil DOI 10.1007/s11104-009-0182-z

VSN International Ltd (2009) Genstat Release 12 Reference Manual. www.vsni.co.uk

WA Herbarium (2009) FloraBase - The Western Australian Flora. Department of Environment and Conservation. http://florabase.calm.wa.gov.au/

Walker KJ, Stevens PA, Stevens DP, Mountford JO, Manchester SJ, Pywell RF (2004) The restoration and re-creation of species-rich lowland grassland on land formerly managed for intensive agriculture in the UK. *Biological Conservation* 119, 1-18.

Watson A, Judd S, Watson J, Lam A, Mackenzie D (2008) The Extraordinary Nature of the Great Western Woodlands. The Wilderness Society, Perth WA.

Westoby M, Walker B, Noy-Meir I (1989) Opportunistic management for rangelands not at equilibrium. *Journal of Rangeland Management* 42, 266-274.

Yates CJ, Hobbs RJ, Atkins L (2000a) Establishment of perennial shrub and tree species in degraded *Eucalyptus salmonophloia* (salmon gum) remnant woodlands: Effects of restoration treatments. *Restoration Ecology* 8, 135-143.

Yates CJ, Norton DA, Hobbs RJ (2000b) Grazing effects on plant cover, soil, and microclimate in fragmented woodlands in south-western Australia: implications for restoration. *Austral Ecology* 25, 36–47.

APPENDIX A

Floristic, soil and demographic data, along with site location details for plots sampled in this study, are provided in the disc accompanying this report.

Contact Us

Phone: 1300 363 400
+61 3 9545 2176

Email: enquiries@csiro.au

Web: www.csiro.au

Your CSIRO

Australia is founding its future on science and innovation. Its national science agency, CSIRO, is a powerhouse of ideas, technologies and skills for building prosperity, growth, health and sustainability. It serves governments, industries, business and communities across the nation.