

Grazing in remnant woodland vegetation: changes in species composition and life form groups

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Abstract. Grazing by domestic livestock in native woodlands can have major effects on ecosystem functioning by the removal of plant species that form important functional groups. This paper documents the changes in floristics in a large group of remnants of native woodland left after agricultural clearing in southwestern Australia. Species richness and diversity were significantly reduced in remnants and the proportion of exotic species increased. Detrended Correspondence Analysis (DCA) was used to identify floristic and environmental patterns among plots and identified two distinct groups based on grazing intensity. This indicated that the significance of the relationship between grazing effects and DCA floristic axes was greater than edaphic characteristics that normally influence floristic patterns. Floristic characteristics of sites that were influencing the position of plots on the ordination diagram included proportion of exotic species and proportion of native perennial shrubs and herbs. Numbers of species of native shrubs and perennial herbs were significantly reduced in grazed plots and numbers of exotic annual grasses and herbs were significantly higher. Other life form groups such as native perennial grasses and geophytes were not significantly affected by grazing. Reproductive strategies of perennial species showed a significant decrease in numbers of resprouters and a significant increase in numbers of facultative seeder/sprouters. Exclosure plots showed increases in number and cover of perennial shrubs and herbs after three years whereas number and cover of exotic species did not change. Time series DCA showed that the floristic composition of exclosure plots in grazed sites became closer to that of the ungrazed sites.

Keywords: Detrended Correspondence Analysis; Exotic species; Grazing; Life form; Native species; Livestock; Reproductive strategy; Woodland.

Nomenclature: Green (1985).

Introduction

The most obvious effects of grazing disturbance in native plant communities include changes to vegetation structure and composition as certain species (such as

exotic annuals) are favoured by grazing so that their numbers and/or cover will increase, while other species are disadvantaged and will reduce in number and/or cover (Williams 1969; Hacker 1984; Belsky 1992; Milchunas & Lauenroth 1993). Depending on the environment and the major land use, grazing by domestic livestock can dramatically change the structure of the vegetation by preventing recruitment of trees and shrubs, thus transforming woodlands into grasslands (Chesterfield & Parsons 1985; Gibson & Kirkpatrick 1989; Cheal 1993). In some semi-arid regions of Australia the opposite problem has emerged with invasion by unpalatable woody shrubs adversely modifying native grasslands (Hodgkinson & Harrington 1985; Williams 1990). In some pastoral areas of Australia such as chenopod shrublands, palatable species may become locally extinct under continued grazing pressure from domestic stock (Wilson 1990). In environments that have had a long history of grazing disturbance herbivore grazing is considered essential to maintaining species diversity, and in many communities species diversity has decreased when grazing has stopped (Carr & Turner 1959; McDonald et al. 1984; McNaughton 1985; Noy-Meir et al. 1989; Smith & Rushton 1994).

Grazing disturbance in many Australian habitats can lead to the complete loss of particular functional types of plants, and may have important implications for ecological functioning (Hobbs 1992). For example the loss of perennial woody shrubs may increase the risk of soil erosion, disrupt nutrient cycling, reduce habitat for animals and reduce the food source for pollinators thus leading to major changes in habitat functioning. Alternatively, grazing can contribute to the success of other groups such as annual grasses or annual herbs which are well adapted to high levels of disturbance (Newsome & Noble 1986). Identifying species which are essential to ecosystem functioning is difficult because of the complex interactions involved. To overcome this difficulty Walker (1992) proposed a classification based on functional groups. Several such groups have been suggested for terrestrial plants based on phenology, life form,

nitrogen fixing ability or life history (Armstrong 1993).

Species response to grazing varies according to growth form and reproductive strategy. For example, annual species are more tolerant of disturbance, due to their fast growth rates and early and prolific seed set, than perennial species which tend to be comparatively slow growing and usually require several years to reach reproductive maturity (Grime 1974). Other groups such as geophytes can avoid heavy grazing pressure by having above ground growth occurring in winter and spring, and dying back to an underground storage organ over summer (Pate & Dixon 1981) when grazing pressures are usually heaviest.

Plant species of the fire-prone mediterranean-type climates of Australia exhibit several strategies in response to fire (Gill 1981). The defoliation of plants by grazing can in some ways be compared to the effects of fire in terms of plant reproductive/regenerative responses. For perennial species Bell et al. (1984) recognised three categories of reproductive response after fire: resprouters, obligate seeders and facultative seeder/sprouters. Species which have the ability to resprout from an underground storage organ after defoliation (resprouters) are more likely to be tolerant of continuous livestock grazing than species which are killed by defoliation and rely on seed for regeneration (obligate seeders). Species which have an ability to resprout after defoliation as well as to reproduce well from seed (facultative seeder/sprouter) may not be as greatly affected by grazing pressure.

This paper documents the changes to the vegetation brought about by livestock grazing in a large number of remnants of native woodland left after agricultural clearing in south-west Western Australia. The identification of floristic differences between remnants exposed to different degrees of grazing intensity, and the corresponding changes to different groups of plant species that are important to ecosystem functioning was the principal aim of the study. In addition the floristic changes over time, as determined by enclosure experiments in degraded remnants was also assessed.

Methods

Study sites were located within remnants of native vegetation left after clearing for agriculture in the eastern portion of the Wellington Catchment (33° 30' S, 116° 30' E) in the south-west of Western Australia where approximately 67% of the native forest has been cleared for farmland (Loh et al. 1989). The landscape and soils of these sites are typical of the Beraking valley form with slopes dominated by gravelly sandy soils with some ferruginous duricrust on the hilltops (Mulcahy &

Bettenay 1972). The sites are at the eastern extent of the northern jarrah forest where *Eucalyptus marginata* (jarrah) and to a lesser extent *E. calophylla* (marri) and *E. wandoo* (wandoo) form a woodland or open forest (Dell & Havel 1989). There is a mid storey of small trees such as *Allocasuarina fraseriana*, *Banksia grandis* and *Persoonia longifolia*, and the understorey is dominated by perennial woody shrubs and herbs with the most abundant families being *Proteaceae*, *Papilionaceae*, *Myrtaceae* and *Cyperaceae* (Bell & Heddle 1989). The climate of the study region is Mediterranean-type with average annual rainfall of between 650 - 700 mm/yr of which ca. 80% falls between the months of May and October.

At 27 sites 56 permanent plots of 10 m × 10 m were established. Of the 27 sites 10 were determined to be paired, where a grazed site occurred directly adjacent to an ungrazed site. The sites selected were considered to be representative of the range of remnants that occur in the eastern Wellington Catchment. Remnants in which sites were located varied in size and shape with sizes ranging from 8 - 100 ha. The selection was based on the overstorey species, composition of the understorey, soil types, landscape position and degree of grazing disturbance. Each 10 m × 10 m plot was pegged out and in the case of currently grazed sites a sheep proof ringlock fence was erected and an adjacent unfenced plot was also pegged. This gave a total of 24 enclosure plots and 21 open plots in grazed remnants and 11 open plots in ungrazed remnants. While fencing excluded domestic livestock, rabbits and native herbivores could gain access. Plots were placed at least 50 m from the boundary of the remnant to minimize edge effects. All sites were in woodland and were characterized by the relative dominance of the overstorey species and included 16 jarrah/marri, 8 jarrah/wandoo and 3 wandoo/jarrah sites. All sites were located on lateritic uplands ranging from mid to upper slope positions with gravelly loam sandy soils, with the exception of site No. 7 which was lower slope with a duplex sand over clay soil.

Grazing intensity at each site was given a subjective ranking based on past and present grazing history, soil disturbance, erosion and condition of vegetation at the site:

- (1) never grazed - some past disturbance such as logging;
- (2) some past light grazing - no obvious signs of grazing, known light grazing in the past.
- (3) current light grazing - obvious signs of livestock grazing such as droppings, low stocking rates (~ < 0.5 sheep/ha).
- (4) moderate grazing - obvious signs of livestock grazing such as droppings, moderate stocking rates (~ 0.5 - 2 sheep/ha), short history of grazing.

(5) current heavy grazing - obvious signs of livestock, large areas of bare ground, heavy stocking rates ($\sim \geq 5$ sheep/ha), short history of grazing.

(6) severe grazing - obvious signs of livestock, large areas of bare ground, heavy stocking rates ($\sim \geq 5$ sheep/ha), long history of grazing.

The initial vegetation survey of all 10 m \times 10 m enclosure and adjacent open plots was completed in November 1991 with subsequent surveys being carried out in November 1992 and 1993. For each survey all understorey species occurring within a plot were recorded and relative cover abundance estimated using the Domin - Krajina scale (Mueller-Dombois & Ellenberg 1974). The presence and cover abundance of only the seedlings of overstorey species were recorded and utilised in further analysis.

Soil samples from all locations were gathered in December 1991 and samples were taken over a depth of 0 to 5 cm at three points located around and close to the enclosure and open plots at each site. The soil analyses performed on all samples were total N and P, organic carbon, exchangeable cations (Na^+ , K^+ , Ca^{++} , Mg^{++}), pH, total soluble salts and particle size analysis. Descriptions of the methods of soil analyses used are given in Klute (1986) for soil physical properties and Page et al. (1986) for soil chemical properties. The soil analyses carried out included total nitrogen using the Kjeldahl digestion method, total phosphorus using the Olsen method, organic carbon using the heat of dilution method, pH using 1:5 soil to deionised water solution, total soluble salts calculated from electrical conductivity of 1:5 soil de-ionized water solution and exchangeable cations (K, Na, Ca, Mg) by ammonium chloride method. Soil particle size analysis was also done using the pipette method.

Detrended Correspondence Analysis (DCA), detrending by second order polynomials, (ter Braak 1987) was performed to enable a quantitative comparison of plots so that floristic and environmental patterns could be identified. For time series data, survey results for each plot for each year were analysed in a single DCA analysis, allowing a visual representation of changes in plot floristics over time and the movement of plots in ordination space depicting changes in species composition and abundance. Two way indicator species analysis (TWINSPAN) (Gauch & Whitaker 1981) was used to classify plots.

To identify the effects of grazing on functional types, species were grouped according to life form (i.e. native shrubs, native perennial herbs, native perennial grasses, native geophytes, native annual herbs, exotic annual herbs and exotic annual grasses). Perennial species were also grouped according to reproductive strategy using the groups defined by Bell et al. (1984) (i.e. seeders,

resprouters and facultative seeder/sprouters).

For the purpose of comparing grazed and ungrazed and open and enclosure plots cover abundance values were transformed to give equal weighting to each Domin scale class so that plot averages could be calculated. This was done using the method described by Bannister (1966) where the mean of the combined cover and frequency values can be used to give a transformed linear scale according to $y = 0.0428x$.

Results

In the 27 sites a total of 210 species were recorded which included 177 native species and 33 exotics. Of the native species, the most common families with perennial shrubs included:

<i>Papilionaceae</i>	(19 spp.),
<i>Epacridaceae</i>	(9 spp.),
<i>Proteaceae</i>	(8 spp.),
<i>Mimosaceae</i>	(8 spp.).

The most common families of

perennial herbs were:	<i>Cyperaceae</i>	(8 spp.),
	<i>Dasyogonaceae</i>	(6 spp.),
	<i>Restionaceae</i>	(4 spp.).

Families with a high share of other life forms were

geophytes:	<i>Orchidaceae</i>	(9 spp.),
annual herbs:	<i>Asteraceae</i>	(14 spp.),
native perennial grasses:	<i>Poaceae</i>	(7 spp.).

Exotic species consisted entirely of annual grasses and herbs:

	<i>Asteraceae</i>	(5 spp.).
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The most obvious change in species composition at the sites disturbed by grazing was the increase in introduced species and decline in the number of native species (Fig. 1). Average species richness was 36 % less in the heavily grazed plots with the number of species in one 100 m² plot as low as 9. The highest species richness for a 100 m² plot (41) was recorded at an ungrazed site. Species diversity was also affected by grazing pressure (Fig. 1). This is most noticeable in the more severe grazing categories (groups 5 and 6). These reductions in species richness and species diversity with increased grazing disturbance are reflected in the proportion of introduced species, with species diversity decreasing as the proportion of introduced species in a plot increases ($r = -0.8049$; $p < 0.0001$).

Site ordination of the vegetation using Detrended Correspondence Analysis revealed a separation of plots along the first floristic axis into two distinct groups (Fig. 2). The first two axes accounted for 80 % of the variability of the data with axis 1 making up 50 %. The other two floristic axes revealed relatively little information. The separation of plots into two groups is confirmed by TWINSPAN analysis which splits the sites at the first cut level into two groups corresponding to those seen in

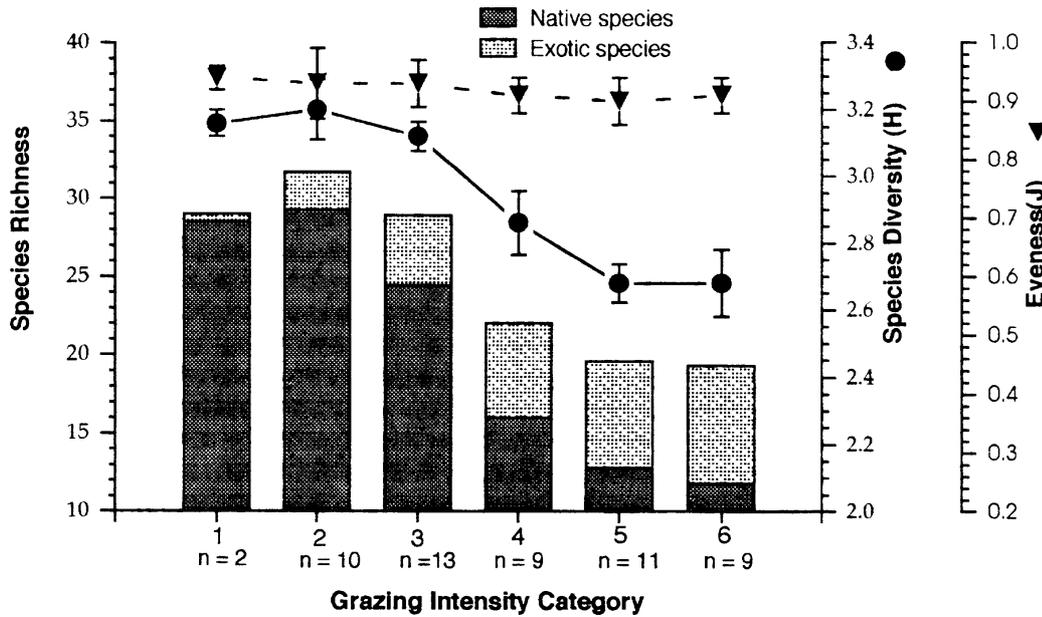


Fig. 1. The effect of grazing intensity on species richness, diversity and evenness and the proportion of exotic species. Grazing intensity categories: 1 = no grazing; 2 = past light grazing; 3 = current light grazing; 4 = moderate grazing; 5 = heavy grazing; 6 = severe grazing.

Fig. 2. The two groups reflect the grazing history of the particular sites. Plots in the right hand group have all had moderate to heavy grazing whereas the plots in the left hand group have either never been grazed or have had only some light grazing.

By correlating different floristic attributes, functional groups and soil parameters with floristic axes 1 and 2 values for each plot, an insight is gained into the attributes that influence the position of each plot along the axes. There were no significant correlations between soil variables and axis 1 or between any variable and axis 2 (Table 1). Along DCA axis 1 however, species diversity tended to decrease with lower species diversity corresponding to lower axis 1 values (grazed plots). The proportion of introduced species increased along axis 1 indicating increasing numbers of introduced species in the grazed plots. The spread of plots along floristic axis 1 also reflects the change in life form groups among the plots with a decreasing proportion of native shrubs and native perennial herbs along axis 1.

Other life form components of the vegetation such as native geophytes, native perennial grasses and native annual herbs showed no significant correlation with axis 1, although the proportion of native annuals and native perennial grasses tended to increase along axis 1. Grouping perennial species according to their reproductive strategies and correlating these with DCA axis 1 showed that the proportion of resprouters decreases along axis 1 while the proportion of facultative seeder/sprouters increased.

Table 1. Correlations between DCA Axis 1 and various floristic and environmental (soil) attributes of the 56 plots.

	<i>r</i> -values	<i>p</i> -value
Floristic attributes		
Species diversity	-0.7272	< 0.0001
Species richness	-0.7301	< 0.0001
Proportion life form groups		
Exotic species	0.8539	< 0.0001
Shrubs	-0.8141	< 0.0001
Native perennial herbs	-0.7680	< 0.0001
Native annual herbs	0.4108	0.150
Native perennial grasses	0.3659	0.152
Native geophytes	0.0812	0.552
Proportion reproductive strategy perennial species		
Obligate seeder species	-0.2701	0.154
Resprouter species	-0.4818	0.01
Facultative seeders/sprouters	0.5322	0.006
Soil attributes		
Total nitrogen	0.2594	0.134
Phosphorus	0.0984	0.576
Organic carbon	0.3078	0.173
pH	-0.0173	0.923
Total soluble salts	0.1367	0.438
Exchangeable Na	0.2433	0.160
Exchangeable K	0.1204	0.496
Exchangeable Ca	0.2583	0.135
Exchangeable Mg	0.3149	0.165
Proportion sand	-0.4347	0.085
Proportion clay	0.2009	0.249

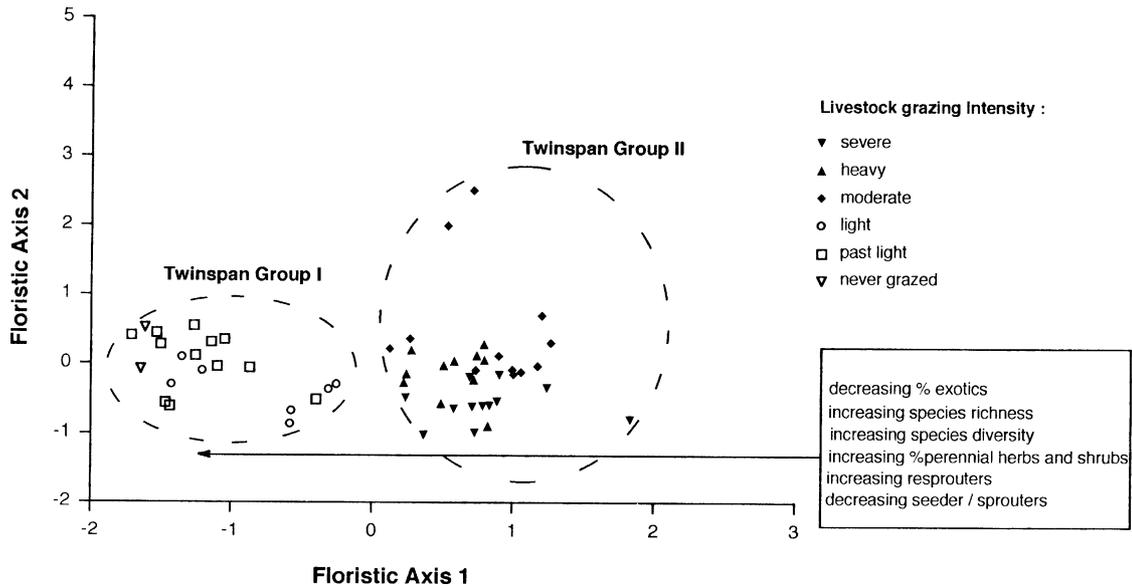


Fig. 2. Detrended Correspondence Analysis of floristic data. Plots grouped in two TWINSPAN groups (grazed and ungrazed). Floristic attributes significantly correlated with DCA axis 1 are shown.

For comparison of the effects of grazing on the soil attributes listed in Table 1, plots were grouped as either grazed or ungrazed. Ten plots were omitted from this analysis as the status of these plots in terms of grazing regime could not be determined precisely. This comparison was also done for the different functional types of species. There were no significant differences between mean values of the grazed and ungrazed plots for any of the soil attributes. The effects of livestock grazing differed for different life form groups with numbers of native perennial shrubs and herbs being significantly reduced in the grazed plots compared with ungrazed plots, whereas the numbers of native annual herbs, native grasses and geophytes were not significantly different (Fig. 3a). There was a significantly greater number of exotic herb and grass species at the grazed plots. Among the perennial species the proportion of resprouters decreased significantly in the grazed plots while there was a significant increase in the proportion of facultative seeders/sprouters (Fig. 3b). The proportion of obligate seeder species between the grazed and ungrazed plots was similar.

In subsequent years (1992, 1993), plots were generally clustered in the same groups as the ordination for the initial 1991 survey, with ungrazed plots in a group to the left - low values on axis 1 - and the grazed plots to the right - higher values on axis 1 - (Fig. 2). After 24 months 88 % of sites had moved further to the left on DCA axis 1 and therefore showing closer floristic similarity to the ungrazed plots. 90 % of enclosure plots had a > 5 %

decrease in axis 1 value (i.e. moved further to the left on axis 1) than the adjacent open plots. Movement of ungrazed plots was small and tended to show no trend. Movement of enclosure plots was consistently to the left although the magnitude of the move varied between plots. Change in position of open plots was also to the left but this was not consistent. This is probably due to variations in grazing intensity and climatic conditions between years. Differences in values on DCA axis 1 for each plot for each year are obtainable upon request from the corresponding author. Eight sites were selected as showing responses typical of those recorded at the 56 plots over the 24 months (Fig. 4). These were the paired sites (i.e. grazed adjacent to an ungrazed site) 5 and 6, 15 and 16, and 26 and 27 as well as two unpaired sites. Differences in movement at individual sites may reflect the different grazing intensity between sites and between years. For example Site 6 was in an area where trees were planted in winter 1991 so that stock was excluded for 2 yr and were re-introduced in early 1993. Similarly at Site 23 a crop was sown in the adjacent paddocks with sheep being re-introduced in 1993 and this is reflected in the change in position of the unfenced plots on the ordination diagram. At Sites 3 and 26, open plots have moved to the left but to a lesser extent than the enclosure plot. The open plot at Site 15 has moved very little where grazing has constantly been at a high rate over the study period. In contrast the enclosure plot had moved considerably to the left. Change of position of the ungrazed Sites 5 and 16 has been relatively small

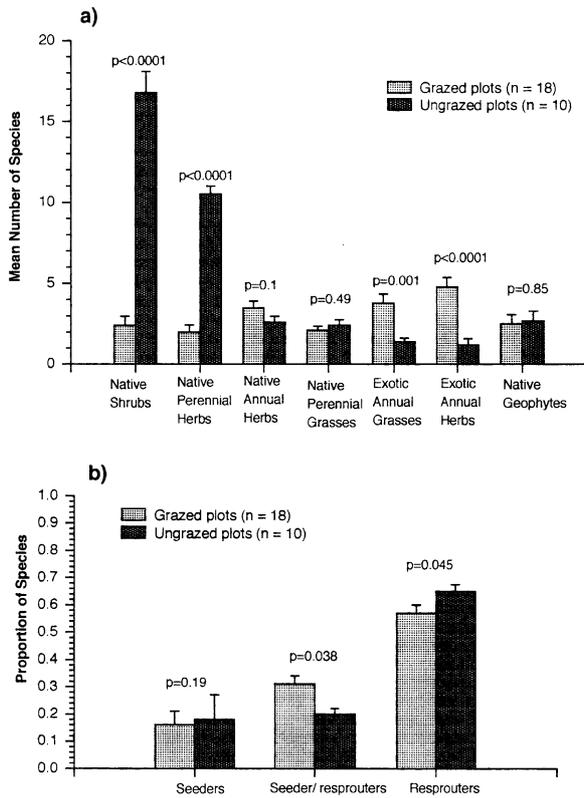


Fig. 3. a. Mean number of species per life form group. **b.** Mean proportion of species in reproductive groups in plots in grazed and ungrazed remnants. *P*-values are for *t*-tests comparing means for grazed and ungrazed plots for each of the groups.

and may reflect variations in rainfall in each year. At Site 27 light grazing took place in the first half of 1992 but ceased in 1993.

The nature of the floristic shifts discussed above is identified by examining changes in species composition and abundance of combined sites. Mean number of species of native perennial shrubs occurring in the enclosure plots increased significantly after 12 ($p < 0.009$) and 24 months ($p < 0.001$) from 4.7 to 7.2 to 8.8. Numbers in the open plots also increased after 24 months (4.1 to 5.2 to 5.8) but these increases were not significant ($p > 0.09$). Numbers were significantly higher in enclosure plots compared with open plots after 12 months ($p < 0.0001$) and 24 months ($p < 0.0001$). The same trends were seen for the mean number of native perennial herbs which increased from 3.5 to 5.3 to 7.9 in the enclosure plots and 3.7 to 4.6 to 4.3 in the open plots. No significant change was recorded in numbers of exotic herbs and grasses and in native perennial grasses, annual herbs and geophytes.

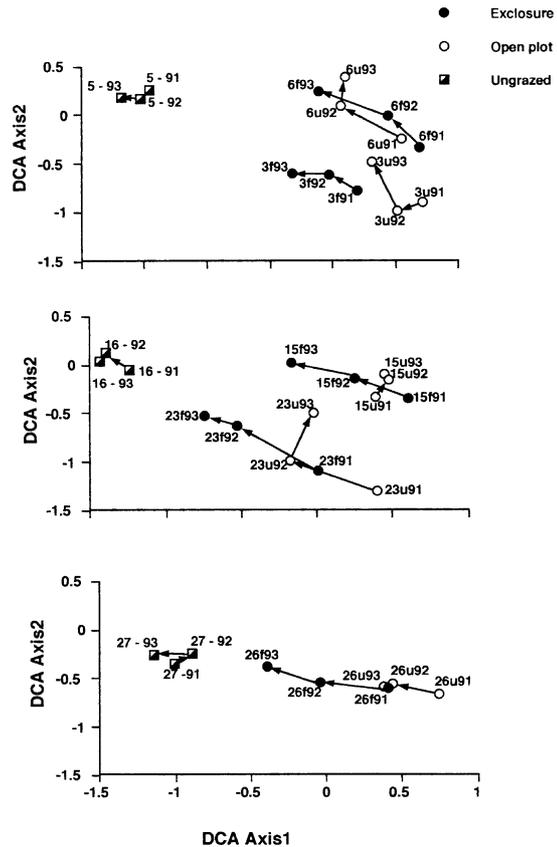


Fig. 4. Shift of selected plots in the time-series along DCA axes 1 and 2 over a 3-yr period. Labels refer to site number and year; f = fenced (closed circles); u = unfenced (open circles).

The number of species within a plot with cover values in the lower cover classes (+, 1, 2 on the Domin scale) increased from 1991 to 1993 in both open and enclosure plots for both native shrubs and perennial herbs. However, in the higher cover classes (3, 4, 5, 6, Domin scale) frequency of scores increased from 12 to 18 in the enclosure plots while decreasing from 14 to 4 in the open plots. By transforming the Domin scale cover abundance values using the method of Bannister (1966) equal weighting is given to each cover class allowing more reliable averaging of cover values. Trends in changes in vegetation cover of the different life form groups showed a 30 % increase in mean cover of native shrubs in enclosure plots after 24 months while decreasing by 3 % in open plots. Similarly, cover of native perennial herbs increased by 20 % in enclosures while decreasing by 25 % in open plots. Cover of exotic grasses in the enclosure and open plots showed a similar trend after 24 months with a 30 % and 32 % decrease in cover respectively. Exotic herbs also showed

a similar trend between enclosure and open plots with cover decreasing by 22 % and 20 % respectively. This probably reflects slightly different climatic conditions between years affecting the germination of annual species. There was also a 28 % increase in mean cover of native grasses in enclosure plots after 24 months while open plots decreased by 4 %. Mean total cover of native grasses was also between 25 % to 30 % higher in the grazed plots compared with the ungrazed plots.

Discussion

The results presented here confirm initial observations that the major effects of grazing disturbance on the vegetation in remnants on farmland has been the loss of native species and the invasion of exotic species. Species richness and diversity has been reduced as many native perennial species have been lost and replaced by fewer exotic annual species. Disturbances such as grazing and soil disturbance have also led to loss of native species with partial replacement by exotics in temperate Australian grasslands (McIntyre & Lavorel 1994). The fact that species evenness has not been affected by grazing (Fig. 1) indicates that no exotic species have become dominant. This is typical of the ecology of grazed environments as grazing pressure prevents the dominance of a few species (Grime 1979; Shugart 1984). The exotic species recorded here are all annual grasses and herbs most of which are ruderal species with characteristics which allow them to thrive in highly disturbed environments. These attributes include seed dormancy, short life histories and early flowering (Grime 1979). The invasion by exotic species may also be enhanced by the reduction of cover of perennial species creating gaps which these ruderal species are best able to exploit. Results here show that invasion of introduced annual species is mainly restricted to grazed (disturbed) sites. Results in lower rainfall woodlands have shown that removal of the native perennial canopy can lead to an increase in abundance of exotic annuals (Hobbs & Atkins 1991).

The positioning of plots in the DCA ordination indicates the fundamental differences between grazed and ungrazed plots in terms of floristics. The dominant influences on the floristics of the surveyed sites were effects of grazing disturbance such as proportion of introduced species, reduction in species richness and diversity and loss of native perennial species. These factors have tended to override the importance of edaphic characteristics such as soil texture, soil fertility, topography, fire and moisture availability (Bell & Heddle 1989; Bell et al. 1989) in determining the floristic composition of jarrah forest communities. Other authors

have reported increases in nutrient levels in remnants in agricultural areas (Muir 1979; Scougall et al. 1993) especially phosphorus from fertiliser drift and nitrogen from livestock excreta. That there was no significant correlation between sites and nutrient levels may have been due to the small number of samples taken from each site as soil nutrient levels in the jarrah forest tend to vary spatially, especially nitrogen which can vary with the distribution of leguminous plants (Hingston et al. 1981).

Time series ordination using DCA shows the effects of the exclusion of grazing in degraded remnants. The floristics of the enclosure plots have shifted closer to that of the ungrazed plots. Similar time series trends in ordination diagrams have been documented for ant species data in rehabilitated mine sites (Majer 1992; Majer & de Kock 1992). The movement of the open plot also towards the ungrazed sites may indicate that there is a seasonal response influencing the floristic changes in the plots. This is borne out by the data which showed a reduction in exotic annual grasses and herbs in both enclosure and open plots. The closer similarity of enclosure plots to the ungrazed plots is also indicated by the data with significant increases in the cover and number of species of perennial herbs and shrubs in the enclosure plots. The general trend of the movement of all grazed sites to the left on DCA axis 1 may reflect lower grazing levels in remnants in 1993 due to changed management practices.

Native perennial shrubs and herbs have suffered the most significant loss of species and cover in grazed remnants, and are usually the life form groups which make up the greatest proportion of the understorey species in the jarrah forest (Bell & Heddle 1989). These species are generally slow growing stress tolerators which are particularly susceptible to grazing pressure in the late summer and early autumn when annual pasture species have died and perennial species provide the only source of green feed for livestock.

Of the different reproductive strategies of perennial species, the resprouters seem to be the most susceptible to grazing pressure. Sustained grazing pressure would tend to exhaust starch reserves without allowing time for replenishment of the rootstock. Starch reserves have been shown to be almost totally eliminated in the resprouter *Stirlingia latifolia* after continual defoliation over a 12 month period, preventing the input of photosynthates (Bowen & Pate 1993). Resprouter species also tend to take a longer period to reach reproductive maturity when establishing from seed (Keeley 1986), and have much slower growth rates than seeder species (Pate et al. 1990). Resprouters therefore require a longer period free from grazing. The fact that facultative seeder/sprouters actually increased in proportion to resprouters

in grazed plots, indicates that this ability to either resprout or reproduce from seed following defoliation makes them better adapted to sustained grazing. The proportion of seeder species has not changed significantly in grazed sites. This may be due to the naturally low proportion of these species in the jarrah forest (Bell & Koch 1980) so that any significant changes are difficult to detect. The most common of the seeder species, *Acacia pulchella*, has spines which would act as a deterrent to sheep thus giving them some protection from grazing.

Other life form groups not greatly affected by grazing include the native perennial grasses. The increase in biomass of native grasses in grazed sites compared with ungrazed sites indicates that they have been able to exploit the reduction in cover of the shrubs and perennial herbs. Native grasses such as *Stipa semibarbata* are not generally grazed by sheep and the above ground parts die back in summer to a below ground rhizome so that they can avoid high grazing pressures. Similarly, geophytes such as the *Orchidaceae* complete the above ground part of their life cycle in winter and spring before dying back to an underground storage organ over summer. The native annual herbs are able to complete their life cycle before summer thus avoiding the season of highest grazing pressure. This lack of response to livestock grazing of native annuals was also reported for lower rainfall woodland remnants in the wheatbelt of Western Australia (Hester & Hobbs 1992).

If the purpose of management of these remnants is their maintenance and /or rehabilitation, the first requirement is fencing so that stock grazing can be controlled. Results from enclosure plots after 3 years indicate that many native perennial species can regenerate with minimal intervention. In grazing exclusion experiments in the Western Australian wheatbelt some large increases in exotic weeds have occurred (Hobbs 1989) which can effectively prevent regeneration of native perennial species (Panetta & Groves 1990; Hobbs & Atkins 1991). There has been no large increase or dominance of exotic grasses or herbs in the enclosure plots in this study. This could be due to possible climatic influences on annual species. The fact that many of the regenerating native species have arisen from underground storage organs may also make them better able to compete with weed species than would seedlings. This would also be influenced by the length of time that grazing has been taking place at a site. Continued monitoring of the sites is required to assess these aspects and the long term prospects of regeneration in enclosures.

The loss of perennial shrubs and herbs and invasion of exotic species seen here can have serious implications for the long term survival of these remnants.

Modification to ecosystem functions in natural systems can arise from loss of species and/or functional groups and the addition of others. The loss of groups of species which may be important as functional groups in such areas as nutrient recycling (e.g. legumes and *Proteaceae*) or restoring hydrological balance within an ecosystem may lead to the degradation of the system. Lack of recruitment of overstorey as well as understorey species through grazing can result in a significant reduction in the ability of remnants to be self sustaining (Hobbs 1992).

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