

Chapter 4

EFFECT OF EXCLOSURE, BURNING AND CULTIVATION ON A LONG-GRAZED SPECIES-RICH REMNANT GRASSLAND ON VICTORIA'S NORTHERN RIVERINE PLAIN

Abstract

A species-rich riverine grassland, long-grazed at low stocking rates, was selected to monitor the impact of a range of disturbance regimes on the vegetation to try to develop an appropriate conservation management strategy. Ten treatments (cultivation once, cultivation annually, burning once, burning annually and control, all with and without sheep grazing) in 100 plots were established and monitored for three years. The grazing response of the flora was closely associated with plant size, habit, life-form and relative abundance in relation to the abundance of interstitial space. Only the diminutive annuals were small enough to avoid significant grazing and were able to increase in abundance under grazing by exploiting the greater abundance of gaps. Rainfall proved to have a very significant effect on flora with a substantial drop in species richness and abundance observed under drought. Fluctuations in the abundance of native annuals were closely linked to winter rainfall. Cultivation increased the richness and abundance of the predominantly annual exotic flora whilst reducing that of the natives. Some native annuals were encouraged by soil disturbance probably because they were represented in the soil seed bank which is likely to be predominantly exotic. Burning generally reduced the abundance of annual species, presumably by destroying seed at or near the soil surface, and had the most significant impact on annual exotics. Recovery from disturbance was slow and associated with the success of dispersal mechanisms linked to plant size/height. A conservative grazing regime developed on the principle of maintaining the "status quo" was found to be the preferred mode of disturbance for conservation management on the basis of current knowledge.

Introduction

Indigenous lowland grasslands of south-eastern Australia (NSW and Victoria), have recently been recognised as one of Australia's most endangered ecosystems (McDougall and Kirkpatrick 1994; Kirkpatrick *et.al.* 1995). The few large (privately owned) species-rich

remnants that remain have been subject to abnormally low stocking rates from a regional perspective and contain many threatened species (Foreman 1993; Maher and Baker-Gabb 1993; Diez and Foreman 1995). Appropriate conservation management of such remnants has been severely hampered by the lack of reliable experimental data.

One of the key findings emerging from study of remnant vegetation in the region (Chapter 2) is community divergence as a response to post-European settlement land management. This observation is based exclusively on the study of grassland vegetation, but probably also applies to grassy woodlands and similar grassy ecosystems elsewhere in south-eastern Australia (Lunt 1995a). To summarise, roadsides and rail reserves, which are largely protected from stock grazing, tend to support a significantly different flora to immediately adjacent freehold paddocks that have long been subject to frequent (although light) grazing. To explain this, a community divergence model was proposed which characterised species by their tolerance or intolerance of stock grazing and burning. According to this framework, the future composition of grasslands can be predicted under different disturbance regimes. Aspects of this model can be readily tested using stock exclosures in historically grazed species-rich remnants. The role of burning in the model, however, is not as straight-forward and requires clarification.

Of further relevance to this experiment is the intermediate disturbance hypothesis first proposed by Connell (1978) (also see Sousa (1984); Hobbs and Huenneke (1992)). This asserts that a clear relationship exists between the level of disturbance and the species diversity of a community and complements the community divergence model proposed (Chapter 2). At some intermediate level of grazing, it is predicted that species diversity (or richness) will be maximised and presumably sustained. In Australia, McIntyre and Lavorel (1995a, 1995b) suggest that native species richness may well respond positively when intermediate levels of endogenous (e.g. grazing and burning) rather than exogenous disturbances (e.g. soil disturbance, fertilisation and irrigation) are imposed on native vegetation. The experiment will test the use of the intermediate disturbance hypothesis in the conservation management of this type of grassland vegetation.

It is necessary to burn *Themeda triandra* dominated grassland remnants in southern Victoria to maintain community species richness by maintaining an open sward (Stuwe and Parsons 1977; Stuwe 1986; Lunt 1990; Lunt 1991; Department of Conservation and Environment 1992; Lunt 1994). It is presumed that disturbance is similarly required to maintain species richness in remnant grasslands of the Northern Plains, especially long-grazed paddocks (Diez and Foreman 1995), although this remains to be demonstrated. This chapter describes an experiment addressing these issues.

The key questions the experiment was designed to answer are (a) does the exclusion of grazing from long-grazed species-rich grasslands result in a reduction in floristic richness due to the unrestrained competitive influence of perennial grasses?, and (b) to sustain richness, can burning replace grazing in long-grazed species-rich grasslands, are there any grazing/burning interactive effects, and is burning useful as a conservation management tool? (c) can grasslands recover from mechanical soil disturbance or are they “permanently” damaged?, and (d) how do individual native and exotic species respond to the imposed disturbances and climatic variations? (i.e. drought).

Methods

Site Description

The study site is located on freehold land in the Northern Plain (Conn 1993), 80 kilometres due north of Bendigo (36°05'34"S, 144°16'00"E; 100 m above sea level), near Mitiamo and Pyramid Hill. The average annual precipitation recorded in nearby Pyramid Hill is 398 mm, 45% of which falls during irregular thunderstorms in summer and autumn. Summers are typically hot (average daily maximum temperature exceeding 30°C), winters are mild with few frosts (8 severe and 21 light frosts annually) and drought (< 200 mm rainfall between April and October) may occur one in five years with the last occurring in 1994 (LCC 1983).

Geologically, the site is on the Shepparton Formation, a thick sequence of flood-created sediments, and bounded to the west by a small range of outcropping Palaeozoic intrusive granites and granodiorites; the Terrick Terrick Range (Macumber 1991). Soils have deep, heavily textured uniform or gradational profiles with either a clay loam or friable clay upper

horizon over a paler clay subsoil (Skene 1971). The heavier textured soils exist as gilgai complexes due to seasonal water-logging. The clay loam soil exists at a slightly higher elevation, is better drained and has a more homogeneous microtopography.

The property has been managed by the same family since the 1860's, primarily for sheep pastoralism (using a regionally conservative grazing regime). The experimental site was not known to have ever been cropped or otherwise subject to broadscale soil disturbance (Foreman 1992).

Experimental Design

A 150 m transect consisting of 100 staggered 2 x 2 m permanently marked plots was established in March 1993 across a suitably homogeneous (floristic and microtopographical) section of grassland vegetation within the 108 ha paddock. The plots consisted of 10 treatments in a randomised block design and the first data (spring 1993) was collected after application of all treatments. These were: control (undisturbed), cultivation once (first year only), cultivation annually, burning once (first year only) and burning annually, all with and without stock grazing (Merino crossbred ewes and wethers). Grazing was applied across the entire paddock with exclosure achieved by 1 m high stock-proof fences. Rabbits, hares and kangaroos were observed both within the experimental paddock and in adjacent areas, but numbers were very low and impact on the vegetation was considered negligible.

A rotary hoe was used to disturb the surface soil to a depth of approximately 10 cm in May 1993-95 immediately following the first significant autumn rains ("autumn break" usually > 25 mm). After cultivation the surface was raked to reinstate microtopographical homogeneity. Burning was applied using an oxygen/acetylene torch aiming to consume all above-ground vegetation in either March or April at least 2 or 3 weeks since prior rains and before the "autumn break".

During the experiment, traditional management regimes were deliberately preserved within the selected paddock (Fig. 4.1 a). Whilst stocking rates remained between 1.8 and 2.6 Dry Sheep Equivalent (DSE)/ha, levels (as reflected in the stocking calendar) have varied as dictated by

environmental conditions. Features of note include resting after severe disturbance (overgrazing in 1991 and drought in 1994) and regular removal of stock during spring, especially in recent years (Fig. 4.1*b*). Exclusion during spring allowed flowering and seed production both for conservation management and ease of plant identification. The experiment concluded in the middle of October 1995.

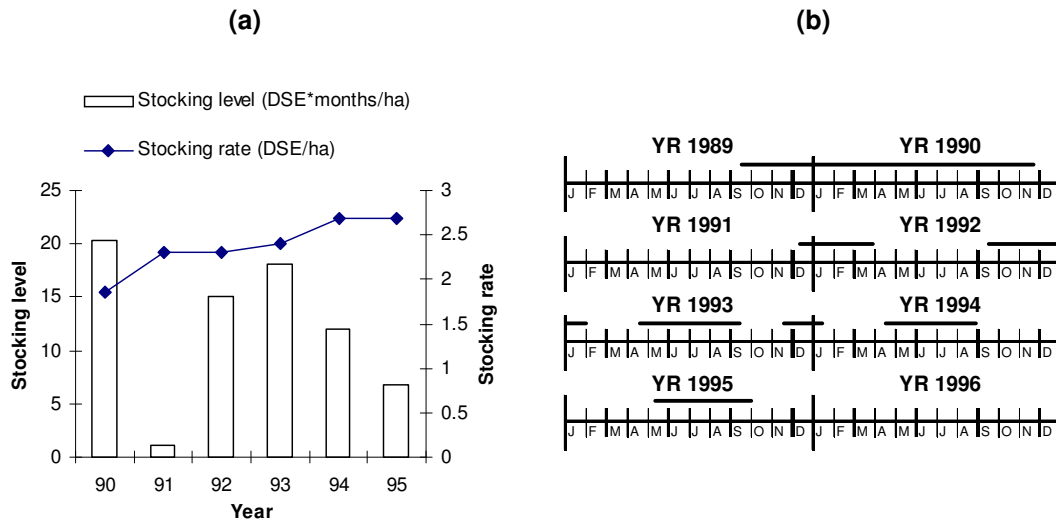


Fig. 4.1. (a) Stocking rates and levels and (b) grazing calendar between 1989 and 1996 in the experimental paddock. Note: stocking rate is the average number of stock per hectare present in a paddock across the whole year and stocking level is stocking rate multiplied by the number of months in the year they are present.

Variable Sampling and Data Collection

The primary variables measured were percentage overlapping cover (% OC) and density per species, from which the secondary variables, frequency per species, total richness and % OC per life-form group (6 in total) were derived. Sampling was undertaken annually in spring, consistently during the first two weeks of October, after stock had been removed for at least one month.

Each 2 x 2 m plot was subdivided into 100, 20 x 20 cm subplots projected onto the vegetation using a single 1 x 1 m aluminium frame (25 subplots each) fixed by two permanent wooden pegs. % OC was measured using the point quadrat method whereby a 50 cm long steel pin (2 mm thick) was lowered into the vegetation at the centre of each subplot, and all species contacted ('scores') were recorded (Mueller-Dombois and Ellenberg 1974). Total % OC per species, per plot was later calculated by adding up the total number of 'scores' and expressed as a percentage. Bare ground and litter were regarded as additional species.

Density per plot was measured by recording the total number of individuals of each species present in 10 randomly selected subplots. The rhizomic, stoloniferous or tussocky growth habit of some species warranted a consistently applied definition of a single individual. Frequency per species was calculated by counting the number of subplots in which at least one individual of that species was recorded and expressed as frequency per plot (a number out of ten).

Richness variables were derived directly from the density data by counting the total number of species recorded in any of the subplots and were expressed as richness per plot (note that this variable differs from richness per 2 x 2 m plot because only a tenth of that area was thoroughly searched). Richness per life-form group was also calculated.

% OC per life-form group was calculated by pooling the data for each species into six groups according to their life-form group allocation listed and described under site description (Table 4.2). These data were further pooled into total exotic and native % OC.

The flora of the site is typical of species-rich, grazed, annual Northern Plain grasslands (Foreman 1992; see Chapter 2).

Data Analysis

Statistical analyses were done with REML (Residual Maximum Likelihood), using GENSTAT 5.3 (Lawes Agricultural Trust, Rothamsted Experimental Station). This procedure is demonstrated for replicated field experiments in Gleeson and Cullis (1987), and Cullis *et al.* (1989), and first used by Patterson and Thompson (1971). ANOVA was not used because treatments were not balanced within blocks due to mistakes in treatment allocation in 1994 (plot no. 48 was rejected from the data analysis, reducing the total number of plots analysed under treatment 9 to nine and in total to 99). In some cases data had to be log (natural) transformed and/or treatments had to be removed in order to obtain homoskedasticity (homogeneity of variance). Preliminary investigation of the data set utilising spatial analysis found no significant autocorrelation between treatments.

REML provided the capacity to compare means and grand means of variables through the calculation of Least Significant Difference (LSD) values ($P \leq 0.05$). In total 292 separate statistical analyses were performed: 24 species x richness x % OC x density x 4 years (1993, 1994, 1995 and 1995-1993); plus 11 community descriptors x % OC x 4 years. The results were produced in the form of the example in Table 4.1 and summarised in Appendices 4.1 (Effect of Exclosure), 4.2 (effect of cultivation) and 4.3 (effect of burning).

Table 4.1. Example of REML analyses utilised: *Maireana excavata* density in 1995. G = grazing, U = grazing enclosure, CON = control, C1 = cultivation once, CC = cultivation annually, B1 = burn once and BC = burn annually. Values with the same superscript within each of the three sections of the table are not significantly different ($P \leq 0.05$). Values in parentheses have been excluded due to low variances in order to obtain homoskedasticity. Results presented within the text in graphical form have error bars which represent $\pm 1/2 * \text{LSD}$ values

		GRAZ		Mean	
		G	U		
CULT	CON	18.19 ^a	13.41 ^b	15.80^a	Degrees of freedom = 62 (2) LSD _{cult} = (1.472 x 2) = 2.94 LSD _{graze} = (1.079 x 2) = 2.16 LSD _{cultxgraze} = (2.113 x 2) = 4.23 N = 10
	C1	5.99 ^c	7.49 ^c	6.74^b	
	CC	(0.30) ^d	(0.60) ^c	(0.45)^c	
	B1	13.61 ^b	13.80 ^b	13.71^a	
	BC	12.02 ^b	16.06 ^{ab}	14.04^a	
	Mean	12.50^a	12.69^a		

Vegetation Composition

Sixty five native and 45 exotic species were recorded within the 100 plots. These were divided into six primary life-form groups based on Friedel *et al.* (1988): native annual forbs (NAF), native perennial grasses (NPG), native perennial forbs (NPF), exotic annual forbs (EAF), exotic annual grasses (EAG), and exotic perennial forbs (EPF). Twenty four of these were present sufficiently frequent (> 70%) that behavioural observations could be made with confidence (Table 4.2). Foreman (1992) presents a detailed description of the vegetation of this site.

Table 4.2. Description of common species in each of the six life-form groups present.
Species in bold are the 24 most frequent within the experimental plots.

Group	Description	Species
native annual forbs	Diminutive, erect forbs (5 to 20 cm tall), often with poorly developed fibrous roots. Germinating in most years during autumn and winter, then flowering and producing prolific quantities of seed in late spring, although at least one species persists well into summer.	<i>Crassula</i> spp. (<i>C. decumbens</i> and <i>C. sieberiana</i>), <i>Goodenia pusilliflora</i> , <i>Hyalosperma glutinosum</i> , <i>Eriochlamys behrii</i> , <i>Daucus glochidiatus</i> , <i>Isoetopsis graminifolia</i> , <i>Leptorhynchos scabrus</i> , <i>Levenhookia dubia</i> , <i>Rhodanthe pygmaea</i> , <i>Triptilodiscus pygmaeus</i> and <i>Wahlenbergia gracilentia</i> .
native perennial forbs	Large perennial (rarely biennial) forbs (10 to 50 cm tall). Resprouting in most years consistently in autumn or winter from well developed fibrous roots, subterranean taproot, tubers or other perennating organs. Flowering and producing modest quantities of seed in either spring or summer.	<i>Goodenia elongata</i> , <i>Leptorhynchos squamatus</i> , <i>Maireana excavata</i> , <i>Arthropodium minus</i> , <i>Bulbine bulbosa</i> , <i>Calotis scabiosifolia</i> , <i>Cheilanthes sieberi</i> , <i>Sida corrugata</i> , <i>Swainsona procumbens</i> , <i>Swainsona murrayana</i> and <i>Wurmbea latifolia</i> .
native perennial grasses	Large perennial graminoids (to 1 m tall) capable of growth at any time of the year depending on rainfall. Typically resprouting in autumn and winter from an extensive fibrous root system. Maximum growth in spring (C3 grasses), however, one species obtains maximum growth in summer (C4 grass).	<i>Danthonia</i> spp. (<i>D. caespitosa</i> and <i>D. setacea</i>), <i>Stipa</i> spp. (<i>S. gibbosa</i> and <i>S. scabra</i>) and <i>Enteropogon acicularis</i> .
exotic annual forbs	Diminutive, prostrate to erect naturalised forbs (to 10 cm tall), often with a well developed taproot. Germinating in most years during autumn and winter, and obtaining maximum growth in spring before producing prolific seed in late spring or early summer.	<i>*Erodium botrys</i> , <i>*Arctotheca calendula</i> , <i>*Cicendia quadrangularis</i> , <i>*Hedypnois cretica</i> , <i>*Hypochoeris glabra</i> , <i>*Parentucellia latifolia</i> and <i>*Plantago coronopus</i>
exotic annual grasses	Diminutive to large, erect naturalised graminoids (to 50 cm tall) with poorly developed fibrous roots. Germinating in most years during autumn and winter, and obtaining maximum growth in spring before flowering and producing prolific quantities of seed in late spring.	<i>*Aira</i> spp. (<i>*A. caryophyllea</i> , <i>*A. elegans</i> and <i>*A. cupaniana</i>), <i>*Lolium</i> spp. (<i>*L. rigidum</i> and <i>*L. perenne</i>), <i>*Rostraria cristata</i> and <i>*Vulpia</i> spp. (<i>*V. myuros</i> , <i>*V. muralis</i> and <i>*V. bromoides</i>)
exotic perennial forbs	Diminutive, erect to spreading biennial or perennial naturalised forbs (to 10 cm tall) originating from a subterranean corm. Germinating in most years during autumn and winter when maximum growth is obtained. Flowering and producing prolific quantities of seed during early spring.	<i>*Romulea minutiflora</i>

Results

Rainfall

Rainfall varied considerably during the experiment, ranging from a wet year in 1993 (28.6% above average), a drought in 1994 (27.1% below average) and an average season in the final year (Fig. 4.2a). Above average rainfall was recorded in 1993 because of very high falls in summer and spring and below average rainfall (-11.6%) in autumn and winter. In contrast, during the average 1995 season, well below average levels of rain fell in the warmer months and above average falls were received in autumn and winter (+25.6%) (Fig. 4.2b).

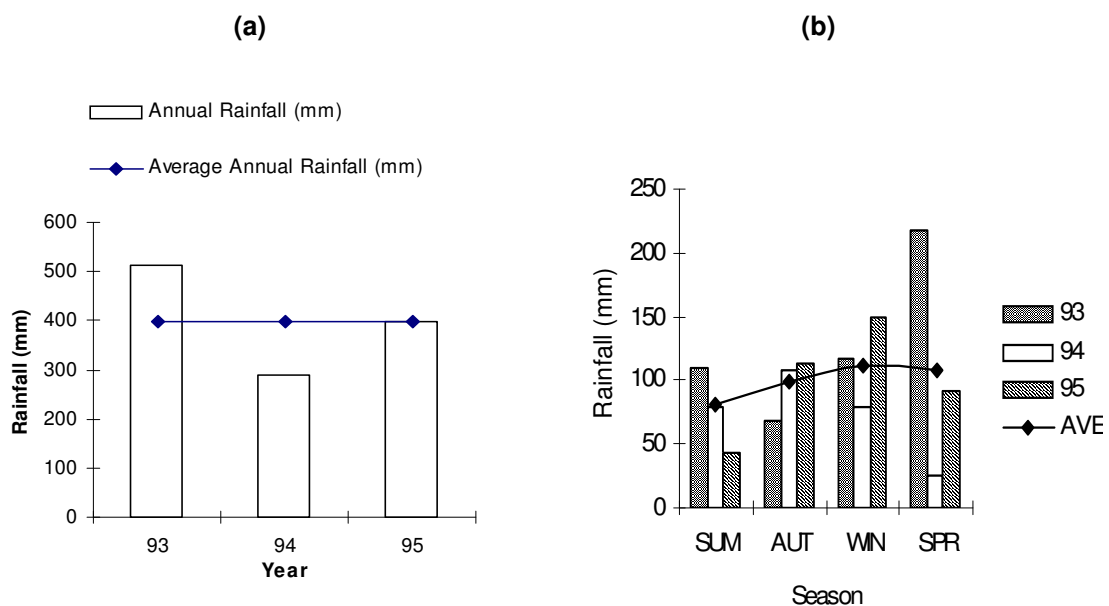


Fig. 4.2. (a) annual and (b) seasonal rainfall at Pyramid Hill - 1993 to 1995 versus long term average.

Effect of Exclosure

Community Level - Richness and Life-form group Cover

For both the exotic and native components of the flora, grazing and exclosure (control) were at no time significantly different throughout the experiment on the basis of the density derived richness data (Table 4.3). Whilst richness levels did drop significantly during the 1994 drought (74% drop in exotics and 29% drop in natives, which respectively represents losses of 5.6 and 6.5 species), they returned almost exactly to pre-drought levels in 1995 (Table 4.3).

The decline in exotic richness in 1994 is the consequence of drops in both annual grasses and herbs, and attributable to the predominantly annual nature of the exotic flora. The less substantial decline in the richness of the native flora is a combination of a 24% and 37% drop in the richness of the perennial forbs and annual forbs respectively (Table 4.3).

Table 4.3. Estimated species richness per plot (0.4m²) of grassland vegetation, with (G) and without (U) domestic stock grazing, over three years from 1993 to 1995 based on density techniques. All cover values are mean % OC per treatment (n = 10) which are either significantly (*) or not significantly (NS) different at P<=0.05 based on LSD values.

	1993			1994			1995			1993 to 1995	
	G	U		G	U		G	U		G	U
Exotic richness	7.51	7.29	NS	1.90	2.00	NS	7.71	7.09	NS	NS	NS
Native richness	20.90	20.20	NS	14.45	14.77	NS	20.30	21.30	NS	NS	NS

In the absence of grazing (and all other disturbance), bare ground quickly dropped to near zero in two years where it remained (Fig. 4.3a), while under grazing it increased, reaching nearly 15% in 1995 (Fig. 4.3a).

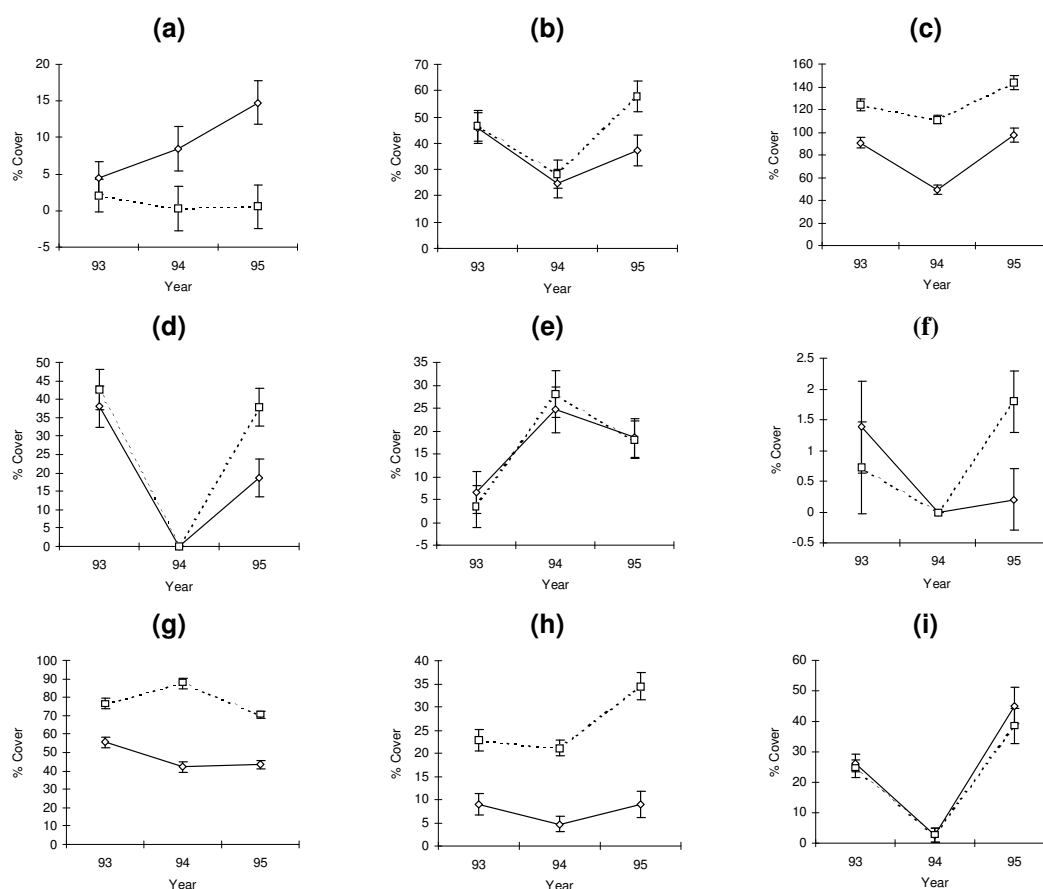


Fig. 4.3. Effect of Exclosure on grassland vegetation between 1993 and 1995 as measured by the mean (n = 10) % OC of (a) bare ground, (b) total exotic species, (c) total native species, (d) exotic annual grasses, (e) exotic perennial forbs, (f) exotic annual forbs, (g) native perennial grasses, (h) native perennial forbs, and (i) native annual forbs in control plots only. Error bars represent $\pm 1/2 * LSD$ at $P \leq 0.05$. Grazing = diamonds and solid lines and grazing exclusion = squares and broken lines.

Total exotic cover dropped sharply in 1994 due to the effect of drought conditions on the dominant grass, *Aira* spp. (reduced to near zero) (Fig. 4.3b, 4.3d). This effect is buffered to some extent by the corresponding increase in the cover of *Romulea minutiflora* in the same year (Fig. 4.3e). Despite these fluctuations, significant divergence between the two control treatments does not occur until year 3 (Fig. 4.3b) and is exclusively attributable to grazing reducing the % OC (rather than the density or frequency) of *Aira* spp. (Fig. 4.3d). Exotic annual forbs have a very low % OC under these treatments (Fig. 4.3f).

Under grazing, total native % OC shows a similar trend to the exotic element with a substantial dip in cover in the second year (Fig. 4.3c). This decline is attributed to a combination of all three life-form groups, but primarily the annual forbs which fell to near zero in drought

conditions (Fig. 4.3i). Under exclosure the annual forbs similarly declined, but this was offset by no change in and a rise in the % OC of perennial forbs and grasses respectively (Fig. 4.3g, 4.3h). In all three years total native % OC under exclosure was maintained at significantly higher levels than under grazing (Fig. 4.3c). This trend was maintained for both the perennial elements but not the annual component, although by the third year the % OC of annuals was slightly lower under exclosure. Overall % OC of annual forbs was 65% higher in 1995 compared with 1993 (Fig. 4.3j).

Species Level - % OC, Density and Frequency

The reduction in the % OC of *Aira* spp. combined with no change in density in 1995 indicates that individual plants tended to be smaller under grazing (Fig. 4.3d, Appendix 4.1). A similar situation was found in the same year for *Romulea minutiflora* as a significant rise in density accompanied by no shift in % OC was recorded under grazing (Fig. 4.3e, Appendix 4.1).

The dominant native annual forb species (*Triptilodiscus pygmaeus*, *Goodenia pusilliflora*, *Eriochlamys behrii*, *Daucus glochidiatus*, *Wahlenbergia gracilentia* and *Leptorhynchos scabrus* see Appendix 4.1) all behave similarly to the overall trend of the life-form group (Fig. 4.3i - greatest abundance in 1995 and lowest abundance in 1994). By the third year, the majority of these species were found to have highest abundance under grazing, two of which were significantly higher in both % OC and density (*Goodenia*, Fig. 4.4 and *Triptilodiscus*, Appendix 4.1). For some species this trend was further reflected in frequency data (Appendix 4.1).

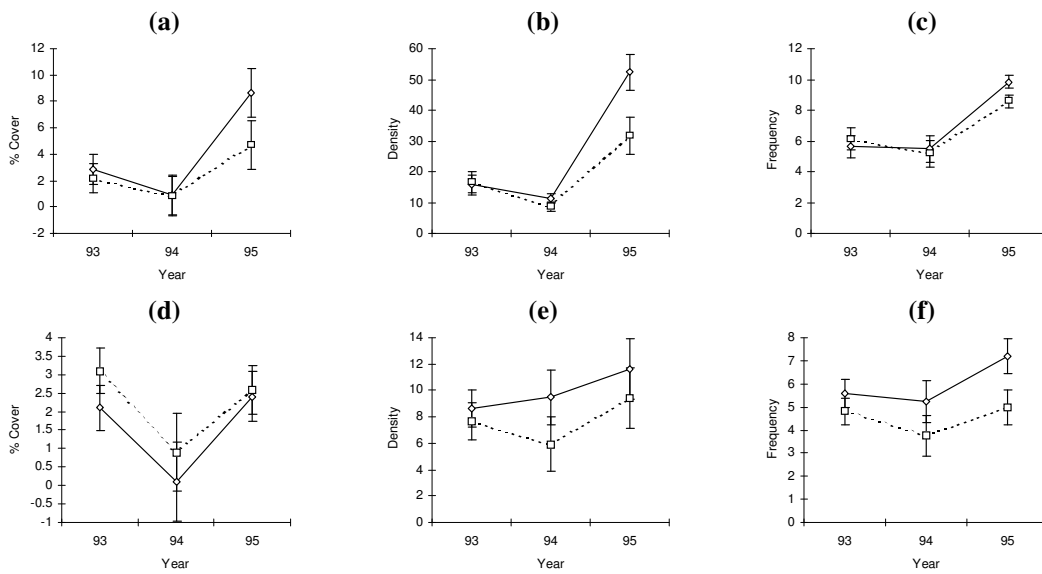


Fig. 4.4. Response of two native annual forbs, *Goodenia pusilliflora* (a) % OC, (b) density and (c) frequency and *Leptorhynchos scabrus* (d) % OC, (e) density and (f) frequency, to grazing (diamonds and solid lines) and exclusion (squares and broken lines) between 1993 and 1995. Error bars represent $\pm 1/2 * \text{LSD}$ values.

The close relationships between % OC and density indicate the changes observed for annual forbs are generally not size related but rather a consequence of fluctuations in population numbers (Appendix 4.1). There were, however, exceptions to this: *Leptorhynchos scabrus* for instance consistently supported a higher % OC (although not significantly so) when excluded from grazing, whilst the density data showed the reverse, suggesting that exclusion generally increased the size of a reduced number of plants (Fig. 4.4d, 4.4e). This response is further confirmed by the lower frequency of *Leptorhynchos* in the absence of grazing (Fig. 4.4f). This differential reaction to grazing amongst the annual forbs was closely associated with plant size where generally only the shortest species responded positively to grazing (Table 4.4).

Table 4.4. Relationship between size and habit of native annual forbs and their respective behaviour under grazing (G) and exclosure (U) as measured by % OC and density in 1995. Grazed and ungrazed treatments were significantly (*) or not significantly (NS) different based on LSD values ($P \leq 0.05$). The + or - symbol denotes the general trend of difference - grazing increases (+) or decreases (-) variable relative to the ungrazed treatment.

Species	Height (cm)	Habit	Density 1995 (individuals/10 subplots)			% Cover 1995		
			G	U		G	U	
<i>Goodenia pusilliflora</i>	< 5	Rosette	52.3	31.7	*+	8.68	4.72	*+
<i>Triptilodiscus pygmaeus</i>	< 5	Erect	193.7	120.8	*+	14.54	10.26	*+
<i>Eriochlamys behrii</i>	to 10	Erect	121.2	103.4	NS+	5.31	3.79	NS+
<i>Daucus glochidiatus</i>	to 10	Erect	41.5	37.0	NS-	4.90	5.50	NS+
<i>Wahlenbergia gracilentia</i>	10 - 15	Erect	121.6	118.2	NS-	7.45	10.46	NS-
<i>Leptorhynchos scabrus</i>	to 20	Erect	11.6	9.4	NS-	2.41	2.59	NS+

Overall trends in the % OC of vegetation were also closely linked to climate, primarily rainfall (Fig. 4.5). Only the native annual forbs show a clear positive correlation with rainfall, which is reflected in % OC of native species in total (Fig. 4.5a). The massive increases in abundance of this group in 1995 compared with the previous two years (especially 1993) is the result of above average rainfall during winter and autumn (+ 25.6%), despite the fact that total annual precipitation in 1995 was exactly average and in 1993 it was well above average (28.6%) (Fig. 4.2, 4.5a). No clear overall trend exists between rainfall and % OC of exotic plants because the increases of the annual grass *Aira* spp. cancel out the decreases of the perennial forb *Romulea minutiflora* (Fig. 4.5b).

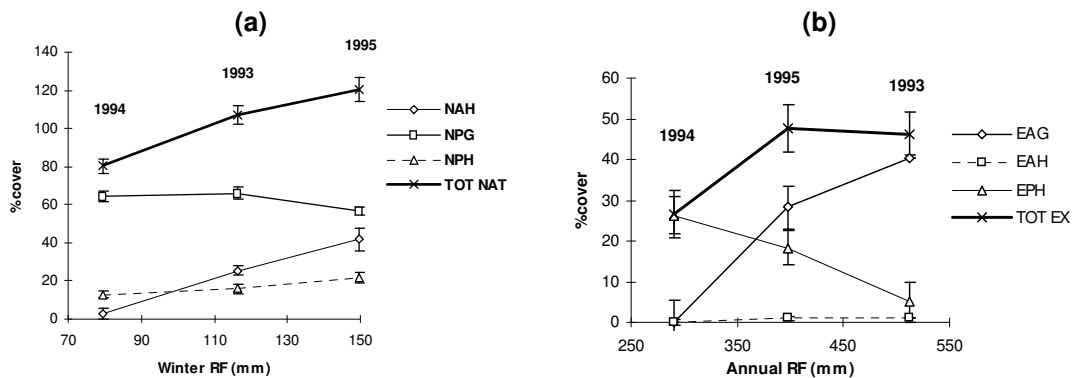


Fig. 4.5. Relationship between (a) winter rainfall and the % OC of native species and (b) annual rainfall and the % OC of exotic species under control conditions. Grazing and exclosure data pooled. Error bars represent $\pm 1/2 * LSD$ values. RF = rainfall. See Table 4.2 for explanation of native and exotic life-form groups.

The clearly dominant native perennial grasses present in the vegetation are *Danthonia* spp. (Appendix 4.1). Abundance diverged sharply in 1994: under exclosure % OC reached a peak of 81% as a result of the previous spring's record rainfall (Fig. 4.2b), and under grazing it

dropped to 41% due to relatively higher grazing pressure (17.8 DSE*months/ha between September 1993 and 1994, Fig 4.1a). This trend of delayed response continued into 1995 with % OC under exclosure dropping back to 67% due to the previous year's drought conditions (Fig. 4.2b, Appendix 4.1). Despite the fact that it was often difficult to consistently identify discrete individuals, the density data confirmed that % OC fluctuations were primarily the product of more growth rather than more individuals (Appendix 4.1). The only other common group of perennial grasses present throughout the plots was *Stipa* spp. This genus also increased in % OC under exclosure (significantly higher in 1994), however, this trend was mirrored in both the density and frequency results, indicating that grazing exclosure increased population size and spatial distribution (Appendix 4.1).

The amount of interstitial space in the vegetation (calculated by subtracting the % OC of *Danthonia* spp. from 100), decreased when grazing was excluded (Fig. 4.6). However, this general trend was reversed in 1995 due to the delayed effect of the 1994 drought on the growth of the dominant grasses. This rise in the amount of interstitial space in 1995 under exclosure is thought to be responsible for a slight drop, rather than the expected significant drop, in % OC of native annual forbs compared with grazing (Fig. 4.6, see Discussion).

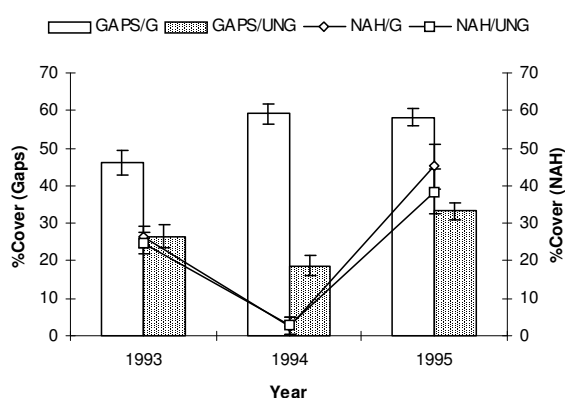


Fig. 4.6. Relationship between amount of available interstitial space (gaps) and the abundance of native annual forbs under control conditions. Error bars represent $\pm 1/2 * \text{LSD}$ values ($P \leq 0.05$). NAF = native annual forbs, G = grazed, UNG = ungrazed.

The dominant native perennial forbs present were *Arthropodium minus*, *Bulbine bulbosa*, *Maireana excavata*, *Leptorhynchos squamatus* and *Wurmbea latifolia* (Appendix 4.1). Overall, the abundance of these species tended to be less subject to annual variation and generally

significantly higher under exclosure than the native annual forbs (Fig. 4.3h, 4.3i). Whilst *Arthropodium* and *Bulbine* responded in a similar fashion to the annual forbs with a drop in % OC in 1994, the remaining species showed a general steady increase in % OC under exclosure (Appendix 4.1). Unlike the annuals, these fluctuations were largely a consequence of increased plant size as little variation was observed between the two treatments in both the density and frequency data. In fact, for *Leptorhynchos* and *Maireana*, the numbers and spatial distribution of populations remained remarkably stable (Fig. 4.7, Appendix 4.1).

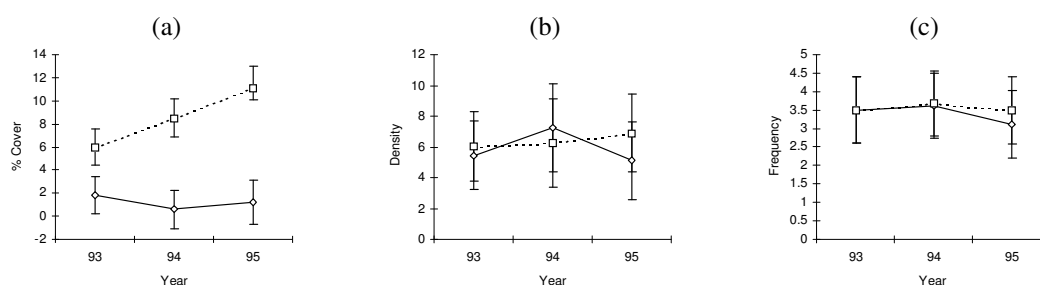


Fig. 4.7. Response of the native perennial forb, *Leptorhynchos squamatus* (a) % OC, (b) density and (c) frequency, to grazing (diamonds and solid lines) and exclosure (squares and broken lines) between 1993 and 1995. Error bars represent $\pm 1/2 * \text{LSD}$ values ($P \leq 0.05$).

Effect of Cultivation

Community Level - Richness and Life-form group Cover

After initial cultivation, exotic species richness increased significantly and although it remained above control throughout the duration of the experiment, it declined back to near control levels in 1995 (Table 4.5, Appendix 4.2). Under annual cultivation (CC), the effect on exotic richness was similar to that observed in C1 plots. However, by 1995 its level significantly fell below that of control, reflecting the severe effect CC had on all but a handful of annual exotic grasses.

In the first year, cultivation resulted in a significant decline in the richness of the native flora (Table 4.5). This low level was maintained throughout the experiment, and by 1995 no significant recovery was observed (Table 4.5). With continued cultivation (CC) the richness of native species declined to very low levels in 1994, and recovered only slightly in 1995 with less than half of the original flora still present and even these represented by very few individuals (Table 4.5).

Table 4.5. Estimated species richness per plot (0.4m²) of grassland vegetation, cultivated in the first year of the experiment (C1) and annually (CC) [with and without domestic stock grazing - data pooled], over three years from 1993 to 1995 based on density techniques. All values are mean richness of treatments (n = 10) which are either significantly (*) or not significantly (NS) different to control at P<=0.05 based on LSD values. The + or - symbol denotes the general trend of difference, cultivation increases (+) or decreases (-) variable relative to control.

	1993		1994			1995			1993 to 1995			
	C1		C1	CC		C1	CC		C1	CC		
Exotic rich	9.176	*+	4.32	3.74	*+	*+	7.80	4.70	NS+	*-	NS+	*-
Native rich	18.54	*-	12.32	5.80	*-	*-	18.57	9.73	S-	*-	*+	*-

The area of bare ground increased as a consequence of cultivation (Fig. 4.8a). Whilst high levels are maintained with continuous soil disturbance (CC), fluctuations occur under C1 due the effect of seasonal rainfall on the abundance of exotic annual grasses. With time to recover from cultivation (C1), bare ground levels fell to that of the control by 1995 (Fig. 4.8a).

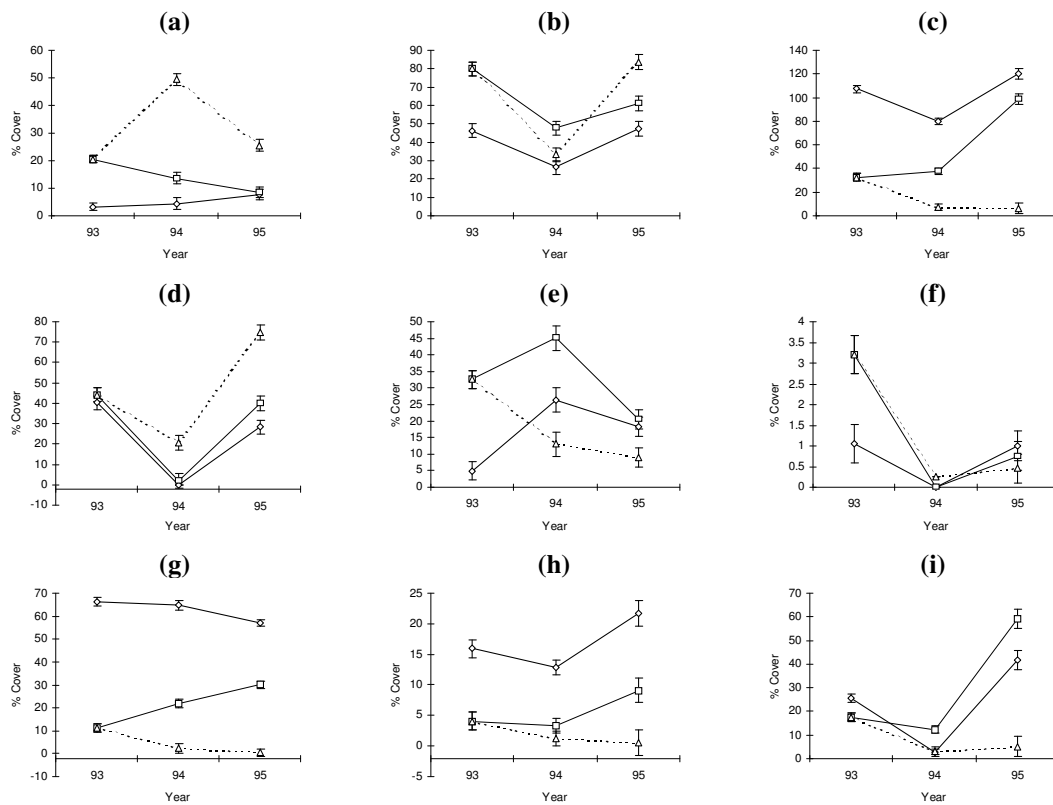


Fig. 4.8. Effect of cultivation on grassland vegetation between 1993 and 1995, with and without grazing (data pooled). Error bars represent $\pm 1/2 * LSD$ ($P \leq 0.05$). Control = diamonds and thick solid line, cultivation once = squares and light solid line, and cultivation annually = triangles and broken lines. See Fig. 4.3 for labels (a) to (i).

Total exotic cover predominantly consisted of the perennial forb, *Romulea minutiflora* and the annual grasses: *Aira* spp., *Lolium* spp. and *Vulpia* spp. Annual forbs were present at relatively low levels and contribute little to the overall exotic trends (Fig. 4.8f). The initial soil disturbance produced a dramatic rise in the % OC of exotic elements primarily because of the sudden appearance of *Lolium* spp. (Fig. 4.8b, Appendix 4.2). *Lolium* spp. were present at low levels under control conditions where *Aira* spp. were dominant (Appendix 4.2). Under C1, the exotic elements recovered over time, although by the final year they were still significantly more abundant than under control conditions (Fig. 4.8b). In contrast, under CC, exotics continued to increase because of the enormous growth of *Lolium* spp. in 1995 (Fig. 4.8e), despite a further decline in the abundance of the perennial, *Romulea minutiflora* (Fig 4.8d).

In comparison to the exotic element of the flora, cultivation generally had the opposite effect on total native cover. After the first cultivation treatment, native % OC dropped by 70%, being

effectively replaced by exotics (Fig. 4.8c). This decline was mainly due to the negative reaction of the perennial elements of the native flora (Fig. 4.8h, 4.8i). The perennial grasses and forbs dropped in % OC by 83% and 74% respectively, while the annuals dropped by only 31% (Fig. 4.8g, Appendix 4.2). With continued cultivation, the % OC of native plants continued to fall and by 1995 they had been virtually eliminated from the plots (Fig. 4.8c). In contrast, with resting after one cultivation, total native cover steadily climbed, and by 1995 its level was only marginally (although still significantly) lower than the control (Fig. 4.8c). This recovery was the result of both an increase in the % OC of the perennials (to about half control levels) and a significant rise above control for the annuals (Fig. 4.8g).

Species Level: % OC, Density and Frequency

Whilst both *Lolium* spp. and *Vulpia* spp. abundance increased after cultivation, *Aira* spp. were clearly discouraged, as they were almost eliminated under CC. Under C1, the density of *Aira* spp. remained significantly lower than the control in 1995 (Appendix 4.2). *Lolium* spp. increased in % OC, density and frequency when cultivated. Although densities continued to increase in the second year of cultivation (CC), the drier conditions reduced their size (i.e. lower % OC, see Fig. 4.8c, Appendix 4.2). Under the improved climatic conditions in 1995, similar numbers of plants obtained larger proportions and therefore unprecedented levels of cover (Fig. 4.8c, Appendix 4.2). In contrast, with no further soil disturbance (C1), *Lolium* spp. declined marginally in density and plant size during the 1994 drought. The improved conditions of 1995, however, resulted in higher germination, although without the additional soil disturbance, individuals remained small.

Vulpia spp. responded in a less clear fashion to cultivation than either *Lolium* spp. or *Aira* spp. In 1993, cultivation had no significant impact on *Vulpia* spp. densities, but did significantly increase plant size. With no further cultivation, this species rapidly declined in % OC to control levels in both 1994 and 1995 (Appendix 4.2). However, with continued disturbance, even though densities increased through the drought, *Vulpia* spp. virtually disappeared in 1995 (Appendix 4.2).

Of the exotic annual forbs, **Hypochoeris glabra* was common enough for some general observations to be made. Cultivation in the first year significantly increased the % OC of this species, although densities fell below that of the control (Appendix 4.2). Under continuous cultivation (CC), the abundance of **Hypochoeris* quickly dropped and by 1995 it had virtually disappeared. With no further soil disturbance (C1), relatively high densities were maintained through the 1994 drought and into 1995, but plant size steadily dropped down to control levels over the same period (Appendix 4.2).

The six dominant native annual forbs (page 4.13) can be divided into two groups by their reaction to cultivation: those encouraged and those discouraged (relative to the control) by once only disturbance. Like most other plants in the flora, under continuous cultivation, the abundance levels of all annual forbs dropped to near zero by 1995 (Fig. 4.8g). Those encouraged by once only cultivation (i.e. *Daucus glochidiatus*, *Eriochlamys behrii* and *Goodenia pusilliflora*) responded in the same characteristic fashion (Fig. 4.9a, 4.9b, 4.9c, Appendix 4.2). For these species, in the first year following cultivation, % OC was significantly higher than that of control, and densities were unchanged except in the case of *Eriochlamys* which initially had a significantly lower density (Appendix 4.2). In the following two years for all three species, densities and % OC steadily rose to unprecedented levels by 1995 (although % OC dipped marginally during 1994). The remaining three species, *Leptorhynchos scabrus*, *Wahlenbergia gracilentia* and *Triptilodiscus pygmaeus*, initially significantly fell below control levels from where they failed to fully recover over the duration of the experiment (Fig. 4.9d, 4.9e, 4.9f, Appendix 4.2). This drop was most dramatic in the density data where levels dropped to less than 20% of the control. The recovery was similar to that observed for perennial species, particularly for *Leptorhynchos*. For all six species, similar trends were apparent in the frequency data, although differences were generally not as great (Appendix 4.2).

(a)

(b)

(c)

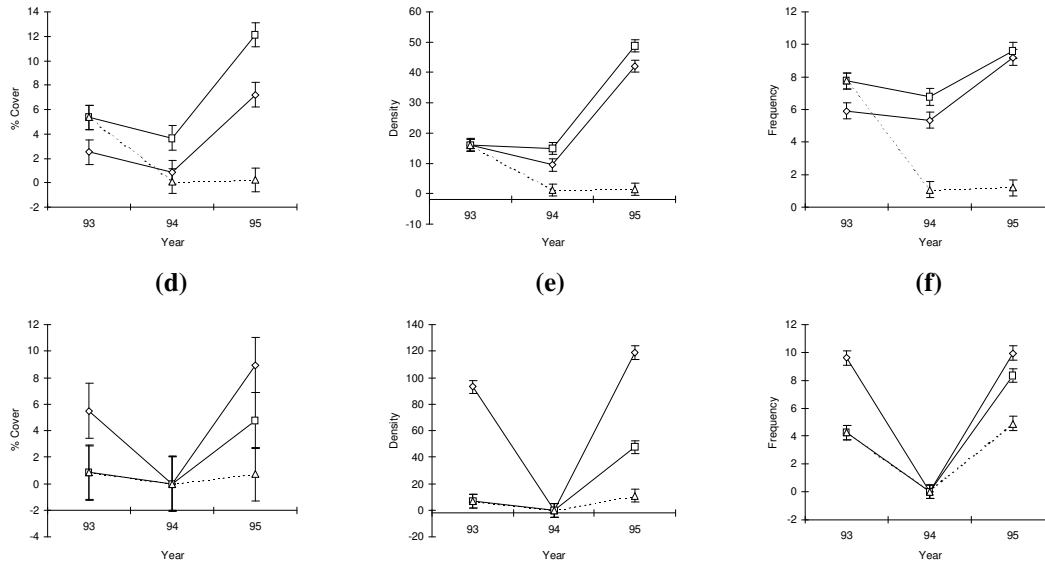


Fig. 4.9. Response of two native annual forbs, *Goodenia pusilliflora* (a) % OC, (b) density and (c) frequency and *Wahlenbergia gracilenta* (d) % OC, (e) density and (f) frequency to cultivation between 1993 and 1995. Control = diamonds and thick solid line, cultivation once = squares and light solid line, and cultivation annually = triangles and broken lines. Error bars represent $\pm 1/2 * \text{LSD}$ values ($P \leq 0.05$).

For native perennial species cultivation either once only or continuously, dramatically reduced abundance, from which recovery under C1 was very slow (Fig. 4.8). Of the four dominant perennial forbs, *Arthropodium minus*, *Bulbine bulbosa*, *Leptorhynchos squamatus* and *Maireana excavata*, two general responses were observed. The latter two steadily climbed to about 30 to 40% (% OC and density) of control levels by 1995, whilst the former two (Liliaceae) dropped to very low levels during 1994 (*Bulbine bulbosa* completely disappeared) before recovering in 1995 (Appendix 4.2).

Cultivation/Grazing Interactions

At the community level grazing/cultivation interactions were complex, but the general trends observed between grazing and exclosure under control conditions were in most cases maintained under cultivation, with either C1 or CC acting to slightly exaggerate or suppress the general trends (Table 4.6). Only rarely were the control/grazing trends clearly reversed under cultivation - with the most obvious example being the native annual forbs. Of these, significant reversal was observed in either % OC or density for *Eriochlamys behrii*, *Goodenia pusilliflora*, *Daucus glochidiatus* and *Triptilodiscus pygmaeus* (Table 4.6, Fig. 4.10).

Table 4.6. Grazing/cultivation and grazing/burning interactions in 1995: (a) % OC of life-form groups, (b) % OC of six major native annual forbs, and (c) density of six major native annual forbs. Treatments: CON = control, C1 = cultivation once, CC = cultivation continuously, B1 = burn once and BC - burn continuously. Grazed and ungrazed treatments were significantly (*) or not significantly (NS) different based on LSD values ($P \leq 0.05$). The + or - symbol denotes the general trend of difference, grazing increases (+) or decreases (-) variable relative to the ungrazed treatment.

	(a)	CON	C1	CC	B1	BC
Bare Ground		*+	*+	*+	*+	*+
Annual grass		*-	*-	*-	*-	NS-
Annual forb		*-	NS+	NS-	NS-	NS+
Perennial forb		NS+	*+	NS+	*+	NS+
Total exotic		*-	NS-	*-	NS-	NS-
Annual forb		NS+	*-	NS-	*-	NS-
Perennial grass		*-	*-	NS+	*-	*-
Perennial forb		*-	*-	NS-	*-	*-
Total native		*-	*-	*-	*-	*-

	(b)	CON	C1	CC	B1	BC
Eriochlamys behrii		NS+	*-	NS-	NS-	NS-
Goodenia pusilliflora		*+	NS+	NS-	*+	*+
Leptorhynchus scabrus		NS-	NS+	NS-	*-	*-
Triptilodiscus pygmaeus		NS-	*-	*-	*-	NS-
Daucus glochidiatus		*+	NS-	NS-	NS+	NS-
Wahlenbergia gracilentia		NS-	NS-	NS-	*-	*-

	(c)	CON	C1	CC	B1	BC
Eriochlamys behrii		NS+	*-	NS-	NS-	NS-
Goodenia pusilliflora		*+	*-	NS-	NS+	NS+
Leptorhynchus scabrus		NS+	NS-	NS-	*-	*-
Triptilodiscus pygmaeus		*+	NS+	NS-	NS+	NS+
Daucus glochidiatus		NS+	*-	*-	*-	NS-
Wahlenbergia gracilentia		NS-	*-	*-	NS-	*-

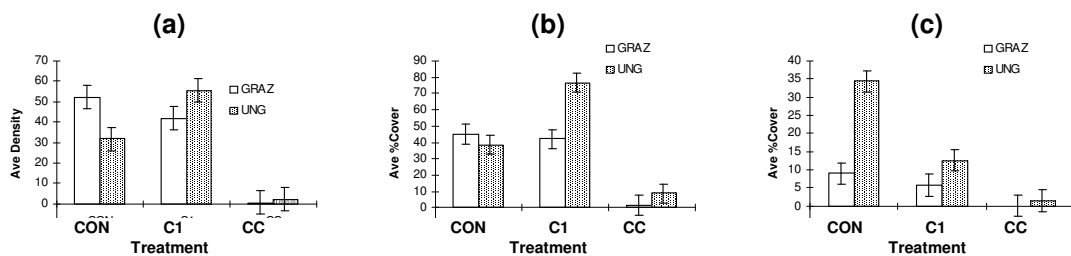


Fig. 4.10. The interaction between grazing and cultivation for (a) *Goodenia pusilliflora* (density), (b) native annual forbs (% OC) and (c) native perennial forbs (% OC). Error bars are $\pm 1/2 * \text{LSD}$ values ($P \leq 0.05$). CON = control, C1 = cultivation once and CC = cultivation annually. Graz = grazed, Ung = ungrazed.

Effect of Burning

Community Level - Richness and Life-form group Cover

The richness of the exotic flora was significantly reduced by burning in the first year (Table 4.7). Exotic richness failed to recover from this initial impact and under both burning treatments it remained significantly lower than the control throughout. Continuous burning (BC) further reduced richness, but not significantly below that of once only burning (B1) (Table 4.7).

Burning did not significantly reduce the richness of the native flora throughout the experiment. Both burning treatments equally conform to this observation, although continuous burning did result in the lowest richness levels (Table 4.7).

Table 4.7. Estimated species richness per plot (0.4m²) of grassland vegetation, burnt in the first year of experiment (B1) and annually (BC) [with and without domestic stock grazing - data pooled], over three years from 1993 to 1995 based on density techniques. All values are mean richness of treatments (n = 20) which are either significantly (*) or not significantly (NS) different to control at P<=0.05 based on LSD values. The + or - symbol denotes the general trend of difference, burning increases (+) or decreases (-) variable relative to control.

	1993		1994				1995		1993 to 1995			
	B1		B1	BC			B1	BC	B1	BC		
Exotic richness	5.80	*-	1.40	1.27	*-	*-	5.40	4.89	*-	*-	NS-	NS+
Native richness	19.53	NS-	14.53	13.95	NS-	NS-	20.35	19.85	NS-	NS-	NS+	NS+

Bare ground is significantly increased in the first year after burning (Fig. 4.11 a). With no further burning, the area of bare ground gradually dropped and in 1995 levels were only marginally above control. With annual burning however, the area of bare ground further increased in 1994 when the dry conditions exacerbated the impact of burning. Even with the prolific growth observed in 1995 (particularly the native annuals), the amount of bare ground dramatically dropped, but remained marginally above that of B1 and significantly above the control (Fig. 4.11 a).

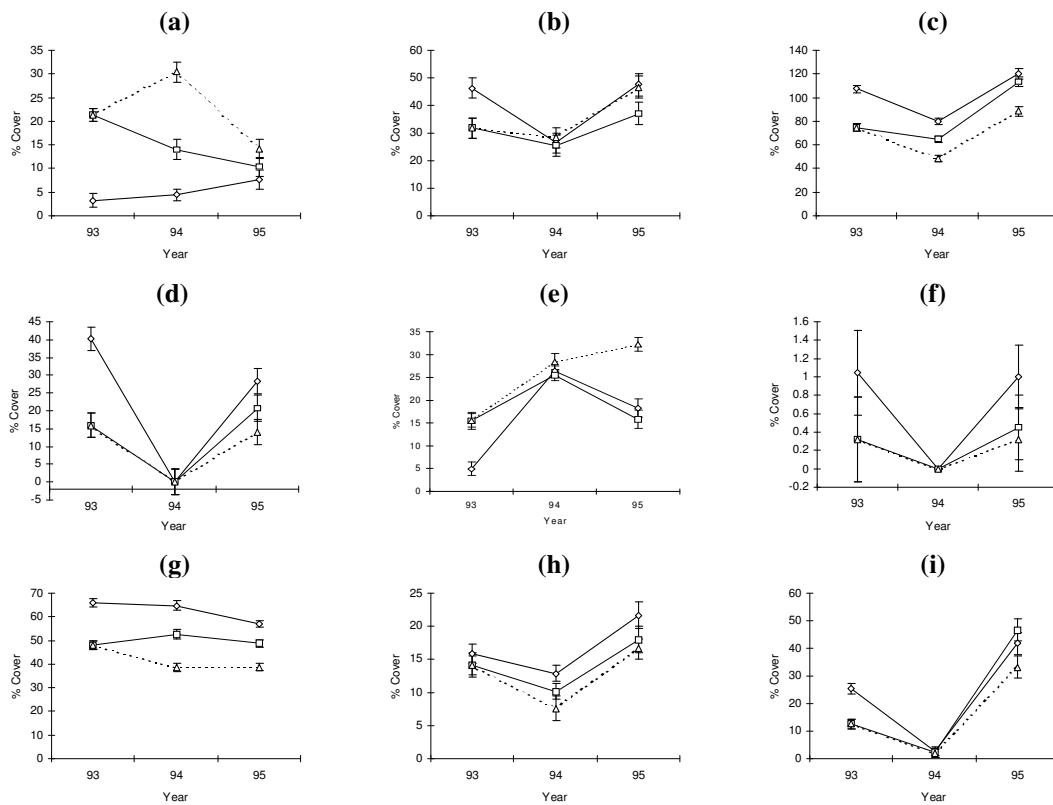


Fig. 4.11. Effect of burning on grassland vegetation between 1993 and 1995, with and without grazing (data pooled). Error bars represent + or - $\frac{1}{2}$ *LSD values ($P \leq 0.05$). Control = diamonds and thick solid line, burn once = squares and light solid line, and burn annually = triangles and broken lines. See Fig. 4.3 for labels (a) to (i).

Total exotic cover significantly fell in the first year following burning due to a very big decline in the annual, *Aira* spp. (Fig. 4.11b, Appendix 4.3), which was buffered to some extent by a significant increase in the perennial forb, *Romulea minutiflora* (Fig. 4.11b, Appendix 4.3). In 1994, total exotic cover fell slightly, being the combination of the complete disappearance of *Aira* spp. and a sharp rise in the abundance of *Romulea* in the dry conditions (Fig. 4.11d, 4.11e). In the final year under B1, exotic cover rose due to the return of *Aira* spp. which was again offset by a decline in *Romulea*, but still remained significantly lower than the control (Fig. 4.11d, 4.11e). Continuous burning (BC), encouraged the proliferation of *Romulea* and despite the lower % OC of *Aira* spp., exotics were just as abundant as under control conditions in 1995 (Fig. 4.11b). Annual forbs were also generally reduced in % OC by burning, but not significantly, although abundance was so low (<1%) that it contributed little to total exotic cover (Fig. 4.11f).

Total native cover, like the exotic component, was also reduced significantly in the first year following burning (Fig. 4.11c). This effect was the combination of a similar significant drop in the % OC of annual forbs and perennial grasses, and only a marginal (not significant) decline in the % OC of perennial forbs, some of which appeared to be stimulated by burning (Fig. 4.11g, 4.11h, 4.11i). With no further burning (B1), the total abundance of native plants gradually increased and by 1995 was only marginally below that of the control plots primarily due to the regrowth of the perennial grasses (*Danthonia* spp.) (Fig. 4.11h). The % OC of perennial forbs remained consistently (although not significantly) below control levels (Fig 18i), and that of the annual forbs actually rose slightly above the control due to the increase in % OC of *Wahlenbergia gracilenta*, *Eriochlamys behrii* and, to a lesser extent, *Leptorhynchos scabrus* in 1995 (Fig. 4.11g, Appendix 4.3). When continuously burnt, total native cover remained below control levels due to a combination of similar responses from all three primary life-form groups (Fig. 4.11c).

Species Level: % OC, Density and Frequency

**Aira* spp. were reduced in abundance by burning and exhibited no significant size variation throughout the burnt plots. In the same way, burning also significantly reduced the abundance of a number of other exotic annual grasses such as **Vulpia* spp. and **Rostraria cristata*. The dominant annual forb, **Hypochoeris glabra* was also reduced by burning (both density and frequency). Burning tended to encourage **Romulea minutiflora*, although this was to some extent rainfall dependent (Appendix 4.3).

In the year following the initial burning treatment, the abundance of most native annual forbs was significantly reduced (Fig. 4.11g, Appendix 4.3). In fact, this was the case in at least one of the three key abundance variables for all six main species, with the most consistent responses from *Wahlenbergia gracilenta*, *Triptilodiscus pygmaeus* and *Eriochlamys behrii* (Appendix 4.3). In the following two years however, reactions to the burning treatments varied considerably. In 1994 conditions were so dry that all annuals performed poorly and consequently the effects of the various treatments were not apparent (Fig. 4.11g). Under continuous burning (BC) by 1995, abundance was significantly lower than control levels for

Daucus glochidiatus, *Eriochlamys* and *Triptilodiscus* and higher for *Goodenia pusilliflora*, *Leptorhynchos scabrus* and *Wahlenbergia*, although only *Leptorhynchos* was significantly higher (density) (Appendix 4.3). With no further burning (B1) only *Daucus* and *Triptilodiscus* remained significantly less abundant than control by 1995 (although still at levels noticeably above those under BC). The remaining four species were either at (*Goodenia*) or significantly above (*Eriochlamys* - % OC, *Leptorhynchos* - density and *Wahlenbergia* - % OC) control levels in the final year (Appendix 4.3).

Burning only marginally influenced the abundance of some native perennial forbs. Whilst both *Bulbine bulbosa* and *Wurmbea latifolia* remained almost identical to the control throughout the experiment, the remaining three species experienced a reduction in abundance (Appendix 4.3). *Arthropodium minus* under both B1 and BC had a significantly lower % OC than that recorded in control plots because plant size was reduced. A similar effect was observed for both *Leptorhynchos squamatus* and *Maireana excavata*, although differences with the control were only significant for *Maireana* in the first two years (Appendix 4.3). No difference in levels of flower production was observed, although this information was not directly measured.

Burning/Grazing Interactions

At the community level, the grazing/burning interactions were just as complex as the grazing/cultivation interactions (Table 4.6). Burning variously exaggerated or reduced the grazing effects observed under control conditions and especially influenced the native annual forbs (Fig. 4.12). The grazing pressure for these species was increased under burning such that the positive grazing effect recorded for the diminutive annuals was lessened or even reversed. Furthermore, the negative grazing effect noted for the taller native annual forbs was exaggerated with the greatest shift observed for the tallest species; *Wahlenbergia gracilentia* and *Leptorhynchos scabrus*.

(a)

(b)

(c)

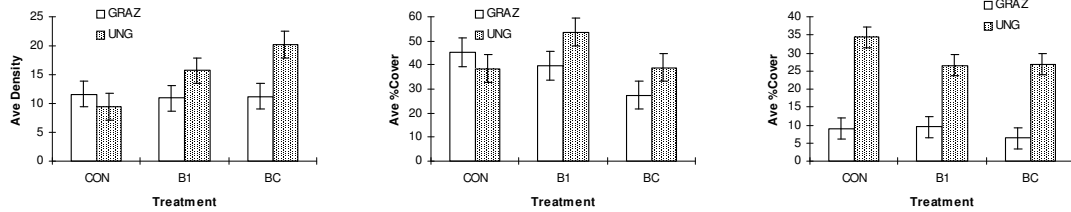


Fig. 4.12. The interaction between grazing and burning for (a) *Leptorhynchos scabrus* (density), (b) native annual forbs (% OC) and (c) native perennial forbs (% OC). Error bars are $\pm 1/2$ LSD values. CON = control, B1 = burn once and BC = burn annually. Graz = grazed, Ung = ungrazed.

Discussion

Grazing Trends

In discussing the community structure and function of long-grazed species-rich riverine grassland remnants, the general conceptual model developed by Grubb (1986) in chalk grasslands of Europe has proved appropriate and useful (also see Tremont and McIntyre 1994 for application in Australian grassy vegetation). According to this model, chalk grasslands consisted of a matrix of dominants amongst which interstices occurred. The dominants of the matrix consisted of various perennial species, whilst the interstitial plants were relatively smaller and short-lived species. The relative abundances of these two components was attributed to the competitive influence of the matrix-formers as influenced by a variety of disturbances.

This study has confirmed the close link between behaviour under grazing and species size, habit, life form and density in Mediterranean grasslands (Noy-Meir *et al.* 1989; Noy-Meir 1995) and has also identified some interesting interactions between grazing, and cultivation, burning and climate. Noy-Meir (1995) concluded that grazing was not significantly associated with family, which inferred chemical palatability, but in fact was significantly associated with plant growth form: protection increasers were mostly tall erect plants, grazing increasers mostly small, prostrate or rosette species, and species with an intermediate response were erect plants of medium height.

In the Riverina the dominant matrix plants are perennial and were significantly reduced in abundance by grazing consistently across all treatments because of their relatively large

stature. The less robust and generally diminutive interstitial native annual forbs in contrast experienced a mixed reaction to grazing primarily on the basis of size, habit and density. The taller annuals (10 to 20 cm) such as *Leptorhynchos scabrus* were more restrained showing only modest increases, no change or even slight decline when grazed. The smallest annuals (<5 cm) such as *Goodenia pusilliflora* and *Triptilodiscus pygmaeus* are inconspicuous and therefore less likely to be consumed by domestic stock than the dominant vegetation. Consequently, these species were virtually unrestrained by stock grazing when climatic conditions allowed them to proliferate in the interstitial space created under grazing.

Further support for this view of size dependent sensitivity to grazing was found under cultivation. With resting after this type of disturbance some of the smaller species were encouraged to unprecedented levels of abundance along with a handful of exotic annual grasses at the expense of the perennial vegetation. In this new context, the normally small annuals had become larger, denser and more conspicuous, and therefore subject to considerably more grazing. A similar shift in grazing pressure was also observed amongst the burning treatments for similar reasons. However, In this case the annuals became more conspicuous because of a significant lowering of both the perennial dominants and the exotic annual flora.

These data suggest that stock (at least under this conservative regime) are preferentially grazing the larger, more conspicuous and accessible elements of the vegetation rather than selecting particular species on the basis of chemical or physical palatability, although this may be happening to some extent (see Noy-Meir *et al.* 1989; Noy-Meir 1995).

Under less conservative grazing (an increase in either stocking rate and/or stocking level), particularly in combination with a lower annual rainfall, such as in 1990 (20 DSE*months*ha and 326 mm or 18% below average), the effect on the vegetation can be quite severe. Under these circumstances, the dominants are initially heavily grazed, opening or widening gaps which normally tends to favour the increase of the smaller annuals. However, because the grazing pressure is sustained, grazing is gradually redirected away from the dominants towards the interstitial annuals as they increase in relative abundance. Eventually this regime

will result in the almost complete loss of vegetation cover from which recovery can only occur with the removal of stock.

In 1994 the abundance of gaps in the exclosed vegetation was very low due to well above average spring rainfall and subsequent growth of the dominants in 1993. Despite this, no grazing effect was observed for the native annuals because soil moisture was clearly the limiting factor during the drought. In the following year when rainfall was not limiting, a grazing effect was observed but it was not as significant as expected because of the relatively high availability of gaps (Fig. 4.6). It would appear that the annuals were less able under exclosure to exploit the opportunity afforded by the favourable 1995 conditions.

Ecological work in grassy vegetation elsewhere in south-eastern Australia has linked a reduction of indigenous species richness to domestic stock grazing (Prober and Thiele 1995); greatest species richness was observed in remnants such as cemeteries never or rarely exposed to grazing. Whilst this conclusion probably applies throughout the country in the context of historical (post-settlement) vegetation change (see Chapter 3), it may not be relevant to species-rich remnants subject to conservative grazing regimes. In some regions, many of the richest remaining grassy remnants have been subject to stock grazing for long periods (Foreman 1993; Foreman and Westaway 1995; Diez and Foreman 1995; Foreman and Garner 1996). Tremont (1994) reported that when grazing was excluded from a paddock previously subject to intermittent sheep grazing, native species richness was reduced by about 50%. However, the direction of change in floristic diversity may well depend on past grazing regimes and will vary from site to site (Tremont 1994). In the Northern Plain grasslands, grazing *per se* is not necessarily a threat to conservation; practices such as cultivation, irrigation and other forms of soil disturbance are the key threatening processes (Diez and Foreman 1995) (see Discussion "Effects of cultivation"). By implication, in some circumstances, maintaining light grazing regimes would have clear application as a conservation management tool.

Several studies have stratified grassy flora into groups according to species response pattern to a range of grazing regimes; (a) intolerant species (only found in cemeteries etc.), (b)

tolerant species (a range of responses have been recognised under this group), and (c) disturbance specialists (favoured by grazing) (McIntyre and Lavorel 1994*b*; Prober and Thiele 1995). Whilst, in the *Eucalyptus albens* woodlands of the western slopes of NSW a small group of native species was identified as being favoured under light grazing, the majority of the natives were intolerant of grazing (Prober and Thiele 1995). This trend may be linked to the fact that the majority of the flora is also perennial. In the long-grazed, species-rich grasslands of the Northern Plain, the high proportion of diminutive annuals generally tolerant of or favoured under conservative grazing regimes could be responsible for the more favourable grazing-richness relationship observed.

In contrast to the native species, the predominantly annual exotic elements are generally enhanced under exclosure. Whilst it would be expected that *Aira* spp. would increase with exclosure because of their size, the enhancement of the more diminutive forbs such as *Hypochoeris glabra* in the absence of grazing was a less predictable result. In combination, the % OC and density data indicate that plant size increases by as much as 900% (Appendix 4.1) when grazing is removed. Under grazing, *Hypochoeris* is a diminutive rosetted forb often sheltered under grass tussocks where it readily persists and reproduces. But when the vegetation structure changes in the absence of grazing, this same species develops considerably enlarged basal leaves which obtain a decumbent to almost erect habit. This species and many other annual exotic forbs have the capacity to readily adapt to quickly changing environmental conditions through modifications in size and habit. This conclusion supports the generalised scheme of morphological plasticity in relation to disturbance developed for closely related cosmopolitan alien species in remnant grassy vegetation of the northern Tablelands of NSW (McIntyre and Lavorel 1994*a*, 1994*b*; McIntyre *et al.* 1995; Tremont 1994).

Effects of Cultivation

Cultivation produced a dramatic and significant effect on the structure and composition of grassland vegetation. This was characterised by (1) an increase in the abundance and richness of exotic species and (2) a decrease in the abundance and richness of native

species. The catastrophic effect of cultivation and similar soil disturbance on indigenous grassy vegetation has been previously recognised and discussed (Department of Conservation and Environment 1992; Baker-Gabb 1993; Maher and Baker-Gabb 1993; Diez and Foreman 1995).

The suite of exotic species present are ruderals (Grime 1979) that have advanced across the landscape primarily because cultivation and similar soil disturbance (plus domestic stock grazing) have created suitable environmental conditions. The prime example of this behaviour was shown by **Lolium* spp. (some species of which are very commonly sown as pasture) which was only very rarely detected under control conditions but very quickly assumed dominance when the soil was disturbed. This response suggests the soil seed bank is predominantly represented by exotic species, a conclusion which supports that of Lunt (1990) who examined the soil seed bank of a long-grazed (70 years) grassland in southern Victoria. With annual soil disturbance, the larger graminoid ruderals maintain dominance at the expense of everything else, particularly the native component which is very quickly eliminated.

Of the three native life-form groups, only the annual forbs did not all initially decline, with some species actually favoured by soil disturbance (i.e. *Daucus glochidiatus*, *Goodenia pusilliflora* and *Eriochlamys behrii*), although this effect was not clear until the second and third years following the treatment. This reaction is due either to the germination of a sizeable soil seed bank or to the movement of propagules from immediately adjacent, undisturbed vegetation. The small stature of these three (as will be further discussed under the effects of burning) limits their capacity to disperse large distances and quickly colonise new areas. Therefore the data suggest that these species may be represented in the soil seed bank which becomes depleted after only one disturbance event. The remaining species either have a very small soil seed bank or have no seed bank and are entirely sustained by seed moving in from undisturbed vegetation surrounding the plots.

After three years following a singular soil disturbance event, the richness and abundance of native species are still substantially lower than that prior to the disturbance. Although for most species there was a strong tendency for recovery, some species and total native richness

remained consistently at low levels. This indicates some possible permanent changes to the vegetation as a result of cultivation, although, over time, full recovery (to control levels) maybe possible under some circumstance. This conclusion is consistent with anecdotal comparisons between historically cultivated and uncultivated remnants throughout the Northern Plain (pers. obs.). Longer term vegetation dynamics following soil disturbance is an important conservation management issue for grasslands and should be the subject of future research.

Effects of Burning

The role of burning in south-eastern Australian grasslands in maintaining native species richness has long been recognised (Stuwe and Parsons 1977; Stuwe 1986; Lunt 1991; McDougall and Kirkpatrick 1994). Whilst fire is known to enhance the flowering and seed production of some perennial forbs (Lunt 1994), it is presumed that the disturbance of fire acts to restrain the growth of the dominant grass, *Themeda triandra*, and to maintain interstitial space occupied by a wide range of native species.

In the Northern Plain, burning tended to open up the vegetation in much the same way as it does elsewhere in grassy vegetation and in this respect mimicked the impact of grazing. However, burning also served to reduce the abundance of all components of the flora, both exotic and native. The perennial elements experienced only a minor drop in cover while maintaining densities. Because all perennials are capable of re-sprouting from tubers and buds, recovery after the applied burn was rapid. The annual species (native and exotic) responded quite differently to burning. In the first year following burning, the cover and density of most annuals was significantly reduced. Presumably the flames damaged or completely destroyed at least a proportion of seed lying on or near the soil surface - seed that would have germinated to produce the next season's population. This conclusion suggests long-burnt rail grassland remnants support few indigenous annuals because of frequent burning rather than the absence of grazing (personal observation, see Chapter 2). Longer term vegetation dynamics following burning is an important conservation management issue for grasslands and should be the subject of future research.

Since the vast majority of the exotic flora is annual, burning served to severely reduce the abundance and richness of the exotic flora. Although native richness was also marginally reduced, with subsequent resting it readily recovered to near pre-disturbance levels. Consequently, infrequent burning may have application as a conservation management tool useful for reducing the abundance and richness of weeds. In contrast, frequent (annual) burning has a greater toll on the native species and whilst it substantially reduces weeds such as *Aira* spp., it also tends to encourage the only exotic perennial species, *Romulea minutiflora*, although this effect depends on rainfall (i.e. in average rainfall years, burning encourages both germination and an increase in *Romulea* plant size, whilst in times of drought, burning prevents germination).

In the same way that the size of the annuals influenced their response under grazing, it is hypothesised that it also influenced behaviour under burning, in-so-much-as it influences seed dispersal capability. After a single burning treatment, the taller annuals were found to be most abundant in the following two years. Assuming germinable annual seed stores are equally depleted by burning, it is presumed that any subsequent recovery is the result of seed dispersing into the treated plots from outside. It is speculated that taller plants are able to disperse seed over greater distances and therefore are capable of depositing greater numbers of seeds across the burnt plots. Thus the taller annuals were able to reach unprecedented densities because of a combination of their superior dispersal capabilities combined with the increased area of interstitial space created by a reduction in the cover of perennials, particularly the native grasses.

Effects of Climate

Rainfall, particularly its seasonality, proved to be a very important factor in determining the behaviour of the vegetation over the duration of the experiment. It influenced rates of biomass production during the spring growing season and annual germination levels (native and exotic) during winter. In fact, a very strong relationship was observed between winter rainfall levels and the abundance of native annual forbs. The ecological significance of this for individual species is largely unknown. For perennial species, rainfall tended to affect plant size and hence cover, rather than population numbers.

In 1994, winter and spring falls were low enough for conditions to be defined as a drought (LCC 1983). Because water was the limiting factor, many annual species either did not emerge at all or struggled to obtain even the most modest levels of abundance. Under these conditions, treatment effects were difficult to detect. However, in 1995, when well above average rains fell and an explosion of germination occurred, space became the limiting factor and the effect of treatments on particular species emerged. Over time under exclosure, as gaps become less abundant, treatment differences will occur even in years of average or below average rainfall.

The generally better performance of the natives during drought conditions is the consequence of a greater proportion of perennial species and annual species capable of at least some germination and seed production during climatic adversity. This observation suggests native annual forbs generally germinate in autumn and winter and readily respond to increased precipitation during this period. Provided reasonable falls occur in the following spring (near or just below average) the earlier germinated populations will be adequately sustained until maturity. Above average spring rain will not compensate, however, for poor rains in autumn and winter. Irrespective of precipitation during autumn and winter, if spring rainfall is very low (drought conditions) only a small proportion of the earlier germinated annuals survive through spring.

In summary climate has three key influences on the vegetation: (1) it determines the germination (density) and abundance (% OC) of annuals in the year in question, (2) it determines the abundance (% OC) of the dominant perennial elements in the year in question, which (3) effects the area of gaps in the vegetation in the following year, which in turn influences the relative abundance of annuals irrespective of rainfall.

Reproductive Biology of Perennial Native Forbs

Work recently undertaken in south-eastern Australia on the reproductive biology of the threatened perennial grassland forb, *Rutidosia leptorrhynchoides* (Morgan 1995) has demonstrated the dependence of perennial species on gaps in the grassland matrix for

successful and sustained reproduction. Lunt (1994) has also linked perennial forb germination and recruitment to gap availability in frequently burnt remnant grasslands in Gippsland. In the Northern Plain, grazing exclosure reduced the availability of gaps and in the long term could result in the gradual loss of perennial forbs.

In a related field experiment involving the same grassland species (i.e. *Bulbine bulbosa*, *Arthropodium minus* and *Leptorhynchos squamatus*), Lunt (1995b) showed that these perennial forbs were only likely to form small, transient and short-term soil seed banks. Whilst some newly germinated seedlings of perennial forbs were observed over the duration of the experiment, the vast majority were ramets of unknown age persisting from previous years. In contrast to the wet conditions of southern Victoria, significant germination and recruitment in northern Victoria probably requires quite specific, episodic climatic conditions. Assuming similar soil seed bank behaviour, the provision of the opportunity to flower and set seed every spring is probably just as critical for perennials as it is for the annuals on the Northern Plain.

Conservation Management

Richness trends in this study have confirmed that exclosure from grazing does not increase species richness, at least in the short term (first three years). However, whilst all formerly present species have persisted under exclosure over the duration of this experiment, the abundance of many native annuals has significantly diminished along with the proportion of interstitial space. Trends suggest this response is likely to continue into the future as interstitial space declines, such that many or all native annuals will eventually disappear (a situation akin to infrequently disturbed roadsides and rail sites, see Chapter 2). On the basis of this response, it is predicted that a significant drop in species richness will be observed under grazing exclosure within the first decade and possibly even by year five. Whilst at the experimental scale, populations of annuals may be readily restored with the return of

disturbance, at the paddock level, recovery after long periods of disturbance exclusion may not be possible. In the context of grazing-modified, but never-the-less species-rich grassland refugia (see Chapters 2 and 3), maintenance of the historical (post-settlement) regime of disturbance will probably maintain species richness indefinitely - i.e. maintaining the "status quo". Whilst on occasions in the past, such remnants have been subject to over grazing, in general, the regionally conservative stocking level applied over the duration of this experiment, is presumably appropriate for conservation management. Longer term monitoring will be required to validate this conclusion.

Vegetation response to the range of disturbances applied will form the framework for developing methods which may improve the conservation values of such refugia, in particular, the use of fire to reduce the abundance of annual exotics and the knowledge that grazing exclosure does not result in species loss in the short term (i.e. first three years).

Furthermore, this study has served to describe the impact of soil disturbance in remnant Northern Plain grasslands, in particular its severe negative impact on native species. In addition, it has served to flag soil disturbance as one of the major immediate threats to the conservation of Northern Plain grasslands because most remaining remnants are privately owned (see Chapter 3).

Conclusions

Sustained conservative grazing regimes (light stocking rates, excluded during spring) did sustain vegetation richness. Both exotic and native richness was markedly reduced as a consequence of drought (i.e. 1994). Grazing exclosure resulted in no significant increase in exotic or native richness and vastly changed the vegetation structure by increasing the abundance of perennial species which reduced the area of interstices. Native annual species fluctuated widely in abundance as a consequence of rainfall, particularly winter rainfall when most germination occurs. Grazing response was determined by growth form and relative abundance, in relation to interstitial availability. The smaller annuals generally avoided grazing because of their stature and were therefore able to better exploit the increased space created/sustained under grazing. However, when relative abundance was modified by either cultivation, burning or over grazing (i.e. annuals become more prominent), the grazing

response was reversed. The greatest effect of treatments differing primarily in gap availability was observed in 1995 when winter rainfall was highest and not a limiting factor.

Cultivation favoured exotic annual grasses (thought to have a significant soil seed bank) at the expense of native perennial species. Although some native annuals were initially favoured by soil disturbance (species suspected of having a soil seed bank), repeated disturbance eliminated all natives and sustained the dominance of the ruderal annual exotic grasses. After a single cultivation treatment, the vegetation did show some signs of recovery, but much of this was probably occurring due to the movement of propagules from immediately adjacent undisturbed vegetation. Despite this, richness and abundance (native) were not restored to pre-disturbance levels by the third year. Whilst complete recovery would appear to be theoretically possible and likely at the experiment scale given time, a single broad acre cultivation (particularly of isolated remnants) would probably inflict permanent compositional and structural change.

Burning dramatically reduced the abundance of annuals. As the exotic component of the flora is predominantly annual, weed abundance and richness were significantly reduced by burning. After a single burning treatment, annuals did show some signs of recovery, but much of this was probably occurring due to the movement of propagules from immediately adjacent undisturbed vegetation, the success of which was probably dependent on seed dispersal capacity. Further experimental trials are recommended to test the usefulness and effectiveness of burning as a conservation management tool for reducing the abundance of exotic species. Conservative grazing is recommended as the preferred conservation management strategy for this vegetation, although the impacts of other disturbances have indicated the potential for management flexibility and potential avenues for vegetation enhancement.

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References

- Baker-Gabb, D. (1993). Managing native grasslands to maintain biodiversity and conserve the Plains-wanderer. RAOU Conservation Statement No. 8, supplement to *Wingspan* **10**.
- Conn, B. J. (1993). Natural regions and vegetation in Victoria. In 'Flora of Victoria Volume 1. Introduction'. (Eds. D.B. Foreman and N.G. Walsh.) pp. 79-158. (Inkata Press: Melbourne.)
- Connell, J.H. (1978). Diversity in tropical rain forests and coral reefs. *Science* **199**, 1320-1310.
- Cullis, B.R., Lill, W.J., Fisher, J.A., Read, B.J., and Gleeson, A.C. (1989). A new procedure for the analysis of early generation variety trials. *Applied Statistics* **38**, 361-375.
- Department of Conservation and Environment (1992). Draft conservation program for the native grasslands and grassy woodlands in Victoria. Department of Conservation and Environment, Melbourne.
- Diez, S., and Foreman, P.W. (1995). The management of natural grasslands on the Riverine Plain of south-eastern Australia. Australian Nature Conservation Agency, Canberra.
- Foreman, P.W. (1992). Floristic resource inventory, description and future management options for the Terrick Terrick Northern Plains property owned by D. Davies. A submission to the Victorian Conservation Trust, Department of Conservation and Environment, Bendigo.

- Foreman, P.W. (1993). The pre-settlement state of the lands of the Fernihurst district. In 'Reflections from the Kinypaniel - the Early Years of the Fernihurst District'. (Ed. M. Munro.) pp. 1-9. (Fernihurst District History Committee: Fernihurst.)
- Foreman, P.W., and Garner, S. (1996). Remnant vegetation survey and botanical inventory of the Shire of Gannawarra. Shire of Gannawarra, Barr Creek Tree Group and the Department of Conservation and Natural Resources, Bendigo.
- Foreman, P.W., and Westaway, J. (1994). Remnant vegetation survey and botanical inventory of the Shire of Gordon, northern Victoria - statement of results. Shire of Gordon, Mid Loddon Tree Group and the Department of Conservation and Natural Resources: Bendigo.
- Friedel, M.H., Bastin, G.N., and Griffin, G.F. (1988). Range assessment and monitoring in arid lands: the derivation of life-form groups to simplify vegetation data. *Journal of Environmental Management* **27**, 85-97.
- Gleeson, A.C., and Cullis, B.R. (1987). Residual maximum likelihood (REML) estimation of a neighbour model for field experiments. *Biometrics* **43**, 277-288.
- Grime, J.P. (1979). 'Plant Strategies and Vegetation Processes'. (John Wiley and Sons: Chichester.)
- Grubb, P.J. (1986). Problems posed by sparse and patchily distributed species in species-rich plant communities. In 'Community Ecology'. (Eds. J.M. Diamond and T.J. Case.) pp. 207-225. (Harper and Row: New York.)
- Hobbs, R.J., and Huenneke, L.F. (1992). Disturbance, diversity and invasion: implications for conservation. *Conservation Biology* **6**, 324-337.
- Kirkpatrick, J., McDougall, K., and Hyde, M. (1995). 'Australia's Most Threatened Ecosystems: the Southeastern Lowland Native Grasslands'. (Surrey Meatty and Sons: Norton NSW.)
- Land Conservation Council (1983). 'Report on the Murray Valley Area'. (Land Conservation Council Victoria: Melbourne.)
- Lunt, I.D. (1990). The soil seed bank of a long-grazed *Themeda triandra* grassland in Victoria. *Proceedings of the Royal Society of Victoria* **102**, 53-57.

- Lunt, I.D. (1991). Management of remnant lowland grasslands and grassy woodlands for nature conservation: a review. *Victorian Naturalist* **108**, 56-66.
- Lunt, I.D. (1994). Variation in flower production of nine grassland species with time since fire, and implications for grassland management and restoration. *Pacific Conservation Biology* **1**, 359-366.
- Lunt, I.D. (1995a). European management of remnant grassy forests and woodlands in south-eastern Australia - past, present and future. *Victorian Naturalist* **112**, 239-249.
- Lunt, I.D. (1995b). Seed longevity of six native forbs in a closed *Themeda triandra* grassland. *Australian Journal of Botany* **43**, 439-449.
- Macumber, P.G. (1991). Interaction between groundwater and surface water systems in Northern Victoria. Department of Conservation and Environment, Melbourne.
- Maher, P.N., and Baker-Gabb, D.J. (1993). Surveys and conservation of the Plains-wanderer in Northern Victoria. Department of Conservation and Environment, Arthur Rylah Institute for Environmental Research Technical Report Series No. 131, Melbourne.
- McDougall, K., and Kirkpatrick, J.B. (Eds.) (1994). 'Conservation of Lowland Native Grasslands in South Eastern Australia'. (World Wide Fund for Nature: Sydney.)
- McIntyre, S., and Lavorel, S. (1994a). Predicting richness of native rare and exotic plants in response to habitat disturbance variables across a variegated landscape. *Conservation Biology* **8**, 521-531.
- McIntyre, S., and Lavorel, S. (1994b). How environmental and disturbance factors influence species composition in temperate Australian grasslands. *Journal of Vegetation Science* **5**, 373-384.
- McIntyre, S., Lavorel, S., and Tremont, R.M. (1995). Plant life history attributes: their relationship to disturbance response in herbaceous vegetation. *Journal of Ecology* **83**, 31-44.
- Morgan, J.W. (1995). Ecological studies of the endangered *Rutidosia leptorrhynchoides* I. seed production, soil seed bank dynamics, population density and their effects on recruitment. *Australian Journal of Botany* **43**, 1-11.
- Mueller-Dombois, D., and Ellenberg, H. (1974). 'Aims and Methods of Vegetation Ecology' (Wiley: New York.)

- Noy-Meir, I. (1995). Interactive effects of fire and grazing on structure and diversity of Mediterranean grasslands. *Journal of Vegetation Science* **6**, 701-710.
- Noy-Meir, I., Gutman, M., and Kalpan, Y. (1989). Responses of Mediterranean grasslands plants to grazing and protection. *Journal of Ecology* **77**, 290-310.
- Patterson H.D., and Thompson, R. (1971). Recovery of interblock information when block sizes are unequal. *Biometrika* **58**, 545-554.
- Prober, S.M., and Thiele, K.R. (1995). Conservation of the grassy white box woodlands: relative contributions of size and disturbance to floristic composition and diversity of remnants. *Australian Journal of Botany* **43**, 349-366.
- Skene, J.K.M. (1971). Soils and land use in the Mid-Loddon valley, Victoria. Department of Agriculture Technical Bulletin No. 22, Melbourne.
- Sousa, W. P. (1984). The role of disturbance in natural communities. *Annual Review of Ecology and Systematics* **15**, 353-392.
- Specht, R.L. (1972). 'Vegetation of South Australia'. (South Australian Government Printer: Adelaide.)
- Stuwe, J. (1986). An assessment of the conservation status of native grasslands on the Western Plains, Victoria and sites of botanical significance. Department of Conservation, Forests and Lands, Arthur Rylah Institute for Environmental Research Technical Report Series No. 48, Melbourne.
- Stuwe, J., and Parsons, R.F. (1977). *Themeda australis* grasslands on the Basalt Plains, Victoria: floristics and management effects. *Australian Journal of Botany* **42**, 511-530.
- Tremont, R.M. (1994). Life history attributes of plants in grazed and ungrazed grasslands on the Northern Tablelands of NSW. *Australian Journal of Botany* **42**, 511-530.
- Tremont, R.M., and McIntyre, S. (1994). Natural grassy vegetation and native forbs in temperate Australia: structure, dynamics and life histories. *Australian Journal of Botany* **42**, 641-658.

Appendix 4.1. The effects of grazing. Cover (average % OC), density (average number of individuals per 10 subplots, 0.4m²) and frequency (average number on subplots in which a species occurs) of six exotic and 18 native species per plot of grassland vegetation, with (G) and without (U) domestic stock grazing, over three years from 1993 to 1995. All values are means of 10 replicates which are either significantly (*) or not significantly (NS) different based on the values of LSD (P<=0.05). T in superscript denotes the use of natural log transformations in analyses (note: a blank cell with - indicates LSD analysis was not performed). The +/- signs indicate direction of change when comparing 1993 to 1995.

COVER	1993			1994			1995			1993 to 1995	
	G	U		G	U		G	U		G	U
Aira spp.	32.91	38.09	NS	0.00	0.00	-	16.86	34.04	*	*-	NS-
Lolium spp.	0.60	0.10	NS ^T	0.00	0.10	-	0.47	0.83	NS	NS-	NS+
Rostraria cristata	2.43	1.87	NS	0.00	0.00	-	0.90	1.20	-	-	-
Vulpia spp.	1.70	2.50	NS ^T	0.00	0.00	-	0.50	1.40	*	NS-	NS-
Hypochoeris glabra	0.50	0.20	NS ^T	0.00	0.00	-	0.00	0.60	-	NS-	NS+
Romulea minutiflora	6.47	3.43	NS	24.62	28.08	NS	18.49	18.11	NS	*+	*+
Daucus glochidiatus	2.14	1.46	NS	0.30	0.10	NS ^T	4.90	5.50	NS ^T	NS+	*+
Eriochlamys behrii	3.37	3.14	NS	0.10	0.20	NS	5.31	3.79	NS	NS+	NS+
Goodenia pusilliflora	2.85	2.15	NS	0.90	0.80	NS ^T	8.68	4.72	*	*+	NS+
Isoetopsis graminifolia	0.40	0.30	-	0.00	0.00	-	0.00	0.00	-	-	-
Leptorhynchus scabrus	2.10	3.10	NS	0.10	0.90	* ^T	2.41	2.59	NS	NS+	NS+
Rhodanthe pygmaeus	0.10	0.30	-	0.00	0.00	-	1.00	0.20	-	*+	NS-
Triptilodiscus pygmaeus	8.62	6.68	NS	0.60	0.10	-	14.54	10.26	*	*+	*+
Wahlenbergia gracilentia	4.56	6.74	*	0.00	0.00	-	7.45	10.46	NS	NS+	NS+
Danthonia spp.	53.90	73.50	*	40.86	81.34	*	41.80	66.80	*	*-	NS-
Stipa spp.	1.51	2.69	NS	0.90	4.80	* ^T	1.30	2.60	NS ^T	NS-	NS-
Arthropodium minus	1.21	5.79	*	0.11	2.19	*	1.12	10.48	*	NS-	*+
Bulbine bulbosa	1.00	2.50	*	0.00	0.00	-	0.30	1.70	* ^T	NS-	NS-
Leptorhynchus squamatus	1.81	5.99	*	0.60	8.50	* ^T	1.23	11.1	*	NS-	*+
Maireana excavata	3.30	5.30	*	3.49	8.01	*	5.00	6.01	NS	NS+	NS+
Sida corrugata	0.00	0.10	-	0.00	0.20	-	0.00	0.20	-	NS	NS+
Swainsona murrayana	0.20	0.40	NS	0.00	0.50	-	0.40	0.90	NS ^T	NS+	NS+
Swainsona procumbens	0.50	0.70	NS	0.19	0.41	NS	0.30	0.90	NS ^T	NS	NS
Wurmbea latifolia	0.00	0.10	-	0.00	0.30	NS	0.20	2.00	*	NS	*

DENSITY	1993			1994			1995			1993 to 1995	
	G	U		G	U		G	U		G	U
Aira spp.	101.14	98.96	NS	0.00	0.00	-	62.44	66.36	NS	*-	*-
Lolium spp.	2.33	2.67	NS	0.00	0.30	-	4.22	4.58	NS	NS+	NS+
Rostraria cristata	32.05	18.35	*	0.00	0.00	-	18.60	14.70	NS ^T	NS-	NS-
Vulpia spp.	12.32	21.38	NS	0.3	0.2	-	3.10	5.20	NS	NS-	NS-
Hypochoeris glabra	5.96	1.65	*	1.38	0.62	NS	1.98	2.12	NS	*-	NS+
Romulea minutiflora	17.73	12.67	NS	43.14	46.16	NS	52.58	27.52	*	*+	NS+
Daucus glochidiatus	14.92	10.38	NS	9.20	6.30	NS ^T	41.50	37.00	NS ^T	* ^T	* ^T
Eriochlamys behrii	59.89	55.71	NS	46.80	40.90	NS ^T	121.21	103.49	NS	*+	*+
Goodenia pusilliflora	15.70	16.60	NS	11.20	8.80	NS ^T	52.29	31.71	*	*+	*+
Isoetopsis graminifolia	2.20	1.81	NS	0.61	0.19	NS	8.00	2.50	*	*+	NS+
Leptorhynchus scabrus	8.60	7.70	NS	9.48	5.92	NS	11.60	9.40	NS	NS+	NS+
Rhodanthe pygmaeus	1.35	0.95	NS	0.90	0.50	-	12.00	6.30	NS ^T	*+	*+
Triptilodiscus pygmaeus	83.93	60.67	*	33.60	38.50	NS	193.7	120.8	*	*+	*+
Wahlenbergia gracilentia	100.13	87.67	NS	0.00	0.00	-	121.60	118.20	NS ^T	* ^T	* ^T
Danthonia spp.	32.82	34.48	NS	39.00	38.60	NS	49.34	39.36	*	*+	*+
Stipa spp.	0.90	1.90	NS	0.77	2.23	*	1.07	2.14	NS	NS+	NS+
Arthropodium minus	5.09	8.39	NS	1.44	3.36	*	7.50	9.10	NS ^T	*+	NS+
Bulbine bulbosa	6.71	8.79	NS	0.00	0.20	-	3.66	6.74	*	*-	NS-
Leptorhynchus squamatus	5.47	6.03	NS	7.25	6.25	NS	5.11	6.89	NS	NS-	NS+
Maireana excavata	15.05	13.95	NS	15.01	13.19	NS	18.19	13.41	*	*+	NS-
Sida corrugata	0.59	0.50	NS	0.68	0.72	NS	0.87	0.63	NS	NS+	NS+
Swainsona murrayana	1.41	0.09	*	1.21	0.39	NS	1.40	0.90	NS	NS-	NS+
Swainsona procumbens	0.49	0.61	NS	0.39	0.31	NS	0.38	0.92	NS	NS-	NS+
Wurmbea latifolia	0.80	1.10	NS	1.22	4.09	*	6.64	13.76	*	*+	*+

FREQ	1993			1994			1995			1993 to 1995	
	G	U		G	U		G	U		G	U
Aira spp.	9.80	9.90	-	0.00	0.00	-	9.38	9.92	NS	NS-	NS+
Lolium spp.	1.20	1.80	-	0.00	0.20	-	2.70	2.20	NS	NS+	NS+
Rostraria cristata	7.17	6.23	NS	0.00	0.00	-	5.20	5.30	NS	NS-	NS-
Vulpia spp.	3.20	4.10	NS	0.20	0.20	-	2.10	2.51	NS	NS-	NS-
Hypochoeris glabra	3.72	1.42	*	0.90	0.60	-	1.73	1.57	NS	*-	NS+
Romulea minutiflora	5.92	4.88	NS	10.00	10.00	-	9.12	7.88	NS	*+	*+
Daucus glochidiatus	7.20	5.80	-	5.40	4.10	NS	9.28	8.92	NS	*+	*+
Eriochlamys behrii	9.80	9.80	-	9.50	8.00	NS	9.90	9.90	NS	NS+	NS+
Goodenia pusilliflora	5.65	6.15	NS	5.48	5.20	NS	9.84	8.57	*	*+	*+
Isoetopsis graminifolia	1.91	1.60	NS	0.61	0.20	NS	5.07	1.93	*	*+	NS+
Leptorhynchus scabrus	5.59	4.81	NS	5.25	3.75	NS	7.20	5.00	*	NS+	NS+
Rhodanthe pygmaeum	1.00	0.60	-	0.80	0.50	-	3.60	2.90	NS	*+	*+
Triptilodiscus pygmaeus	9.40	9.60	NS	7.80	8.20	NS	9.93	10.00	NS	NS+	NS+
Wahlenbergia gracilentia	9.86	9.34	NS	0.00	0.00	-	10.00	9.90	NS	NS+	NS+
Danthonia spp.	10.00	10.00	-	10.00	10.00	-	10.00	10.00	-	NS	NS
Stipa spp.	0.80	1.80	-	0.66	2.04	*	1.08	1.82	NS	NS+	NS+
Arthropodium minus	3.71	4.89	NS	1.31	2.79	*	4.45	5.15	NS	NS+	NS+
Bulbine bulbosa	4.25	5.15	NS	0.00	0.20	-	2.84	4.57	*	NS-	NS-
Leptorhynchus squamatus	3.50	3.50	NS ^T	3.62	3.68	NS	3.12	3.49	NS	NS-	NS-
Maireana excavata	7.50	7.60	NS	7.59	7.71	NS	8.20	7.40	NS	NS+	NS-
Sida corrugata	0.60	0.50	-	0.70	0.60	-	0.90	0.50	-	NS+	NS
Swainsona murrayana	1.10	0.10	-	1.10	0.40	-	1.30	0.80	-	NS+	NS+
Swainsona procumbens	0.50	0.50	-	0.40	0.30	-	0.40	0.90	-	NS-	NS+
Wurmbea latifolia	0.80	1.10	-	1.26	2.74	*	4.77	6.43	*	*+	*+

Appendix 4.2. The effects of cultivation. Cover (average % OC), density (average number of individuals per 10 subplots, 0.4m²) and frequency (average number of subplots in which a species occurs) of six exotic and 18 native species per plot of grassland vegetation, cultivated annually (CC) and in the first year only (C1) [with and without domestic stock grazing - data pooled], over three years from 1993 to 1995. All values are means of 10 replicates which are significantly (*) or not significantly (NS) different based on the values of LSD (P<=0.05). T in superscript denotes the use of natural log transformations in analyses (note: a blank cell with - indicates LSD analysis was not performed). The +/- signs indicate general trend of change in comparison with control (see Appendix 4.1) and between 1993 and 1995.

COVER	1993		1994			1995			1993 to 1995	
	C1		C1	CC		C1	CC		C1	CC
Aira spp.	11.85	*-	0.00	0.00	-	23.66	6.32	NS-	*-	*-
Lolium spp.	21.83	*+ ^T	2.55	17.40	NS+	13.36	67.07	*+	*+	*+
Rostraria cristata	0.91	*-	0.00	0.00	-	2.40	0.90	-	-	-
Vulpia spp.	9.38	*+ ^T	0.53	3.12	NS+	0.55	0.05	NS-	*-	*-
Hypochoeris glabra	2.10	*+ ^I	1.00	0.00	-	0.25	0.05	-	-	*-
Romulea minutiflora	32.54	*+	45.08	13.06	*+	20.45	8.85	NS+	*-	*-
Daucus glochidiatus	6.26	*+	6.00	2.30	*+ ^T	17.80	2.35	*+ ^T	*+ ^T	*-
Eriochlamys behrii	1.92	*-	1.90	0.55	*+	11.71	1.25	*+	*-	NS-
Goodenia pusilliflora	5.34	*+	3.65	0.10	*+ ^I	12.12	0.22	*+	*-	*-
Isoetopsis graminifolia	0.15	-	0.00	0.00	-	0.00	0.05	-	-	-
Leptorhynchus scabrus	0.92	*-	0.65	0.05	NS+	1.29	0.00	*-	*-	NS+
Rhodanthe pygmaeum	0.10	-	0.00	0.00	-	0.99	0.10	NS+	NS-	NS+
Triptilodiscus pygmaeus	1.18	*-	0.10	0.00	-	9.37	0.10	*-	*-	NS-
Wahlenbergia gracilentia	0.84	*-	0.00	0.00	-	4.77	0.71	NS-	*-	NS-
Danthonia spp.	10.88	*-	20.30	2.33	*-	27.27	0.43	*-	*+	*-
Stipa spp.	0.20	*-	1.40	0.15	*-	2.60	0.10	NS+ ^T	*+ ^T	NS-
Arthropodium minus	1.05	*-	0.38	0.22	*-	1.78	0.00	*-	*-	NS+
Bulbine bulbosa	0.85	*-	0.00	0.00	-	0.15	0.00	*+ ^I	*+ ^I	NS-
Leptorhynchus squamatus	0.95	*-	0.75	0.10	*-	1.56	0.15	*-	*-	NS+
Maireana excavata	0.30	*-	0.84	0.06	*-	1.38	0.00	*-	*-	NS+
Sida corrugata	0.00	-	0.10	0.00	-	0.25	0.00	-	-	-
Swainsona murrayana	0.13	NS-	0.20	0.15	-	0.85	0.15	NS+ ^I	*-	NS+
Swainsona procumbens	0.28	NS-	0.45	0.60	-	1.15	0.45	NS+ ^T	NS- ^T	NS+
Wurmbea latifolia	0.25	-	0.00	0.00	-	0.60	0.00	NS-	*-	NS-

DENSITY	1993		1994				1995				1993 to 1995	
	C1		C1	CC			C1	CC			C1	CC
Aira spp.	41.11	*-	0.00	0.00	-	-	51.31	13.76	NS-	*-	NS+	*-
Lolium spp.	22.56	*+	11.87	76.24	NS+	*+	75.58	74.83	*+	*+	*+	*+
Rostraria cristata	8.74	*-	0.00	0.00	-	-	36.20	8.75	*+ ^T	*+ ^T	*+	NS+
Vulpia spp.	18.98	NS+	4.22	23.94	NS+	*+	6.18	0.25	NS+	*-	NS-	*-
Hypochoeris glabra	2.26	NS-	3.05	1.09	*+	NS+	2.03	0.35	NS-	*-	NS-	*-
Romulea minutiflora	53.03	*+	62.10	15.51	*+	*-	43.53	12.41	NS+	*-	NS-	*-
Daucus glochidiatus	18.01	*+	3.65	2.21	*+	NS+	102.15	9.95	*+ ^T	*+ ^T	*+ ^T	*+ ^T
Eriochlamys behrii	27.60	*-	91.35	5.90	*+ ^T	*+ ^T	152.79	18.12	*+	*-	*+	NS-
Goodenia pusilliflora	15.95	NS-	14.85	1.15	*+ ^I	*+ ^I	48.75	1.40	NS+	*-	*+	*-
Isoetopsis graminifolia	1.87	NS-	1.11	0.00	*+	NS-	3.85	1.35	NS-	*-	NS+	NS-
Leptorhynchos scabrus	1.73	*-	4.71	0.05	*-	*-	4.33	0.15	*-	*-	NS+	NS-
Rhodanthe pygmaeus	1.14	NS-	1.95	0.10	-	-	9.10	0.80	NS- ^T	*+ ^T	*+	NS-
Triptilodiscus pygmaeus	8.96	*-	3.88	0.50	*-	*-	93.30	3.50	*-	*-	*+ ^T	NS- ^T
Wahlenbergia gracilentia	6.86	*-	0.00	0.00	-	-	47.60	11.00	*+ ^T	*+ ^T	*+ ^T	*+ ^T
Danthonia spp.	10.00	*-	22.35	4.50	*-	*-	26.30	0.87	*-	*-	*+	*-
Stipa spp.	0.40	*-	1.04	0.05	NS-	*-	1.55	0.08	NS-	*-	*+	NS-
Arthropodium minus	1.95	*-	1.31	0.26	*-	*-	3.20	0.35	*+ ^T	*+ ^T	NS+	NS-
Bulbine bulbosa	2.65	*-	0.00	0.00	-	-	2.10	0.65	*-	*-	NS-	*-
Leptorhynchos squamatus	1.29	*-	0.80	0.55	*-	*-	2.55	0.10	*-	*-	NS+	NS-
Maireana excavata	4.59	*-	6.30	0.75	*-	*-	6.74	0.45	*-	*-	NS+	*-
Sida corrugata	0.10	*-	0.92	0.02	NS+	*-	0.58	0.00	NS-	*-	*+	NS-
Swainsona murrayana	0.43	NS-	0.86	0.25	NS+	NS-	1.07	0.23	NS-	*-	*+	NS-
Swainsona procumbens	0.40	NS-	0.64	0.21	NS+	NS-	0.85	0.10	-	-	*+	NS-
Wurmbea latifolia	0.07	*-	0.40	0.10	*-	*-	5.44	0.30	*-	*-	*+	NS+

FREQ	1993		1994				1995				1993 to 1995	
	C1		C1	CC			C1	CC			C1	CC
Aira spp.	8.71	*-	0.00	0.05	-	-	9.43	4.17	NS-	*-	NS+	*-
Lolium spp.	5.83	*+	4.12	8.12	*+	*+	9.23	9.85	*+	*+	*+	*+
Rostraria cristata	4.02	*-	0.00	0.00	-	-	7.65	3.86	*+	NS-	*+	NS+
Vulpia spp.	4.90	*+	1.77	4.94	NS+	*-	3.15	0.20	NS+	*-	*-	*-
Hypochoeris glabra	2.02	NS-	1.90	0.90	-	-	1.83	0.35	NS+	*-	NS-	*-
Romulea minutiflora	9.40	*+	9.95	8.30	NS- ^I	NS- ^I	8.55	5.96	NS+	*-	NS-	*-
Daucus glochidiatus	7.75	*+	8.10	5.15	*+	NS+	9.68	4.57	NS+	*-	*+	*-
Eriochlamys behrii	8.95	*-	9.20	3.10	NS+	*-	9.90	4.90	NS+	*-	NS+	*-
Goodenia pusilliflora	7.76	*+	6.78	1.06	*+	*-	9.62	1.19	NS+	*-	*+	*-
Isoetopsis graminifolia	1.40	NS-	1.06	0.00	*-	*-	2.65	1.09	NS-	*-	NS+	NS-
Leptorhynchos scabrus	1.57	*-	3.48	0.00	NS-	*-	3.45	0.15	*-	*-	*+	*-
Rhodanthe pygmaeus	0.95	-	1.05	0.10	-	-	3.18	0.07	NS-	*-	*+	NS-
Triptilodiscus pygmaeus	5.12	*-	2.56	0.65	*-	*-	9.63	2.83	NS-	*-	*+	*-
Wahlenbergia gracilentia	4.24	*-	0.00	0.00	-	-	8.35	4.90	*-	*-	*+	NS+
Danthonia spp.	7.16	*-	9.57	3.91	NS-	*-	9.75	0.98	NS-	*-	*+	*-
Stipa spp.	0.40	-	0.93	0.00	NS-	*-	1.45	0.10	NS-	*-	*+	NS-
Arthropodium minus	1.58	*-	1.17	0.29	*-	*-	2.76	0.21	*-	*-	*+	*-
Bulbine bulbosa	2.19	*-	0.00	0.00	-	-	2.04	0.58	*-	*-	NS+	*-
Leptorhynchos squamatus	1.03	*- ^T	1.01	0.10	*-	*-	1.53	0.00	*-	*-	NS+	*-
Maireana excavata	3.73	*-	4.79	0.65	*-	*-	5.20	0.40	*-	*-	*+	*-
Sida corrugata	0.10	-	0.87	0.00	NS+	*-	0.58	0.00	NS-	*-	*+	NS+
Swainsona murrayana	0.40	-	0.87	0.23	NS+	NS-	1.08	0.25	NS+	*-	*+	NS-
Swainsona procumbens	0.40	-	0.65	0.20	-	-	0.84	0.10	NS+	*-	*+	*-
Wurmbea latifolia	0.08	-	0.30	0.10	*-	*-	4.10	0.30	*-	*-	*+	NS+

Appendix 4.3. The effects of burning. Cover (average % OC), density (average number of individuals per 10 subplots, 0.4m²) and frequency (average number of subplots in which a species occurs) of six exotic and 18 native species per plot of grassland vegetation, burnt annually (BC) and in the first year only (B1) [with and without domestic stock grazing - data pooled], over three years from 1993 to 1995. All values are means of 10 replicates which are significantly (*) or not significantly (NS) different based on the values of LSD (P<=0.05).). T in superscript denotes the use of natural log transformations in analyses (note: a blank cell with - indicates LSD analysis was not performed). The +/- signs indicate general trend of change in comparison with control (see Appendix 4.1) and between 1993 and 1995.

COVER	1993		1994				1995				1993 to 1995	
	B1	BC	B1	BC	B1	BC	B1	BC	B1	BC	B1	BC
Aira spp.	14.45	*-	0.00	0.00	-	-	19.32	13.72	NS-	*-	NS+	NS-
Lolium spp.	0.13	NS-T	0.00	0.00	NS-	NS-	0.17	0.07	NS-	NS-	NS+	NS+
Rostraria cristata	1.07	*-	0.00	0.00	-	-	0.75	0.30	-	-	-	-
Vulpia spp.	0.41	*-T	0.00	0.00	-	-	0.40	0.00	*-	*-	NS-	NS-
Hypochoeris glabra	0.26	NS-I	0.00	0.00	-	-	0.10	0.00	-	-	NS-	NS-
Romulea minutiflora	15.55	*+	25.46	28.34	NS-	NS+	15.86	32.24	NS-	*+	NS+	*+
Daucus glochidiatus	0.36	NS-	0.10	0.00	NS-T	NS-T	1.70	0.46	*-	*-	NS+	NS+
Eriochlamys behrii	1.67	*-	0.80	0.22	*+	NS+	6.69	3.46	*+	NS-	*+	*+
Goodenia pusilliflora	1.77	NS-	0.45	0.64	NS-T	NS-T	7.06	9.05	NS+	NS+	*+	*+
Isoetopsis graminifolia	0.15	-	0.00	0.00	-	-	0.20	0.05	-	-	-	-
Leptorhynchus scabrus	1.91	NS-	0.15	0.36	NS-T	NS-T	2.74	2.81	NS+	NS+	NS+	NS+
Rhodanthe pygmaeum	0.00	-	0.00	0.00	-	-	0.00	0.10	*-	NS-	NS+	NS+
Triptilodiscus pygmaeus	3.10	*-	0.00	0.05	-	-	11.23	4.71	NS-	*-	*+	NS+
Wahlenbergia gracilentia	2.23	*-	0.00	0.00	-	-	15.68	11.40	*+	NS+	*+	*+
Danthonia spp.	47.41	*-	50.67	36.72	*-	*-	47.35	37.89	*-	*-T	NS-	*-
Stipa spp.	0.43	*-	1.45	1.07	NS-	*-	0.80	0.46	NS-T	*-T	NS+	NS+
Arthropodium minus	2.54	NS-	0.70	0.54	NS-	*-	3.59	3.18	*-	*-	NS+	NS+
Bulbine bulbosa	1.38	NS-	0.00	0.00	-	-	1.10	0.80	NS+I	NS-I	NS-	NS-
Leptorhynchus squamatus	4.81	NS+	3.80	3.09	NS-T	NS-T	4.19	4.53	NS-	NS-	NS-	NS-
Maireana excavata	2.81	*-	4.10	2.79	*-	*-	5.50	4.81	NS+	NS-	*+	*+
Sida corrugata	0.23	-	0.15	0.10	-	-	0.30	0.17	NS+	NS+	-	-
Swainsona murrayana	0.38	NS+	0.00	0.10	-	-	0.30	0.87	NS-I	NS+I	NS+	NS+
Swainsona procumbens	0.74	NS+	0.35	0.20	-	-	1.20	0.76	NS+T	NS+T	NS+	NS+
Wurmbea latifolia	0.10	-	0.60	0.31	-	-	1.00	0.65	NS-	NS-	*+	NS+

DENSITY	1993		1994				1995				1993 to 1995	
	B1	BC	B1	BC	B1	BC	B1	BC	B1	BC	B1	BC
Aira spp.	46.14	*-	0.00	0.00	-	-	47.83	38.76	*-	*-	NS+	NS-
Lolium spp.	0.03	NS-	0.00	0.00	NS	NS	0.85	0.55	NS-	NS-	NS+	NS+
Rostraria cristata	18.53	*-	0.00	0.00	-	-	12.95	8.85	NS-	NS-	NS-	*-
Vulpia spp.	1.98	*-	0.10	0.10	NS-	NS-	1.80	0.20	NS-	*-	NS-	NS-
Hypochoeris glabra	2.38	NS-	0.25	0.06	NS-	NS-	0.25	0.35	*-	*-	*-	*-
Romulea minutiflora	45.74	*+	38.78	38.15	NS-	NS-T	41.96	86.99	NS+	*+	NS-	*+
Daucus glochidiatus	5.46	*-	1.75	0.64	*-T	*-T	15.10	9.71	*-T	*-T	*+T	*+T
Eriochlamys behrii	22.84	*-	55.55	34.24	NS+T	NS-T	114.89	54.00	NS+	*-	*+	*+
Goodenia pusilliflora	11.26	*-	12.00	9.38	NS+T	NS-T	35.25	44.07	NS-	NS+	*+	*+
Isoetopsis graminifolia	1.70	NS-	0.35	0.43	NS-	NS+	5.45	6.39	NS+	NS+	*+	*+
Leptorhynchus scabrus	5.43	*-	9.01	7.25	NS+	NS-	13.30	15.72	*+	*+	*+	*+
Rhodanthe pygmaeum	0.82	NS-	0.55	0.25	-	-	3.05	1.66	*-T	*-T	NS+	NS+
Triptilodiscus pygmaeus	30.83	*-	19.50	18.17	*-	*-	125.10	59.10	*-	*-	*+	*+
Wahlenbergia gracilentia	33.51	*-	0.00	0.00	-	-	156.80	127.56	NS+	NS-	*+T	*+T
Danthonia spp.	35.44	NS+	40.90	39.57	NS+	NS-	44.78	48.89	NS+	*+	*+	*+
Stipa spp.	1.19	NS-	1.46	0.96	NS-	NS-	1.97	1.31	NS+	NS-T	NS+	NS+
Arthropodium minus	6.39	NS-	2.53	2.74	NS+	NS+	6.85	6.71	NS-T	NS-T	NS+	NS-
Bulbine bulbosa	6.98	NS-	0.00	0.00	-	-	5.07	4.03	NS-	NS-	*-	*-
Leptorhynchus squamatus	6.55	NS+	6.37	8.40	NS-	NS+	6.44	8.11	NS+	NS+	NS+	NS+
Maireana excavata	13.01	NS-	13.64	12.95	NS-	NS-	13.71	14.04	NS-	NS-	NS+	NS+
Sida corrugata	0.72	NS+	1.11	0.42	NS+	NS-	0.81	0.47	NS+	NS-	NS-	NS-
Swainsona murrayana	0.37	NS-	0.60	0.72	NS-	NS-	0.85	0.67	NS-	NS-	NS+	NS+
Swainsona procumbens	0.80	NS+	0.70	0.75	NS+	NS+	1.05	1.00	-	-	NS+	NS+
Wurmbea latifolia	1.37	NS+	4.14	2.85	*+	NS+	9.28	9.35	NS-	NS-	*+	*+

FREQ	1993		1994				1995				1993 to 1995	
	B1		B1	BC			B1	BC			B1	BC
Aira spp.	8.29	*-	0.00	0.00	-	-	8.56	8.42	*-	*-	NS+	NS-
Lolium spp.	0.40	*-	0.00	0.00	-	-	0.65	0.53	*-	*-	NS+	NS+
Rostraria cristata	5.49	NS-	0.00	0.00			5.30	4.48	NS+	NS-	NS-	NS-
Vulpia spp.	1.08	*-	0.10	0.10	NS-	NS-	1.40	0.20	NS-	*-	NS+	NS-
Hypochoeris glabra	1.92	NS-	0.25	0.05	-	-	0.69	0.35	*-	*-	*-	*-
Romulea minutiflora	8.58	*+	10.00	9.65	NS- ^T	NS- ^T	7.89	9.64	NS-	NS+	NS-	NS+
Daucus glochidiatus	3.18	*-	1.35	0.59	*-	*-	6.71	5.42	*-	*-	*+	*+
Eriochlamys behrii	7.47	*-	8.90	8.55	NS+	NS-	9.75	9.28	NS-	NS-	*+	*+
Goodenia pusilliflora	6.00	NS+	6.16	5.40	NS+	NS-	9.54	9.80	NS+	*+	*+	*+
Isoetopsis graminifolia	1.44	NS-	0.35	0.43	NS-	NS+	4.02	4.01	NS+	NS+	*+	*+
Leptorhynchos scabrus	3.87	*-	4.54	5.00	NS+	NS+	6.70	7.47	NS+	*+	*+	*+
Rhodanthe pygmaeus	0.55	-	0.40	0.25	-	-	2.10	1.12	NS-	*-	*+	NS+
Triptilodiscus pygmaeus	8.22	*-	6.50	6.35	*-	*-	9.89	9.75	NS-	NS-	*+	*+
Wahlenbergia gracilentia	7.08	*-	-	-	-	-	9.85	9.95	NS-	NS	*+	*+
Danthonia spp.	9.95	NS-	10.00	10.00	NS	NS	10.00	10.00	NS	NS	NS	NS
Stipa spp.	1.10	-	1.32	0.97	NS-	NS-	1.76	1.26	NS+	NS-	NS+	NS+
Arthropodium minus	4.34	NS+	2.39	2.17	NS+	NS+	4.77	4.68	NS-	NS-	NS+	NS+
Bulbine bulbosa	4.50	NS-	0.10	0.10	-	-	3.53	2.98	NS-	NS-	*-	*-
Leptorhynchos squamatus	3.43	NS- ^T	3.79	3.97	NS+	NS+	3.49	4.07	NS+	NS+	NS+	NS+
Maireana excavata	6.99	NS-	7.55	7.13	NS-	NS-	7.50	7.49	NS-	NS-	NS+	NS+
Sida corrugata	0.53	-	1.06	0.42	NS+	NS-	0.76	0.41	NS+	NS-	NS+	NS-
Swainsona murrayana	0.35	-	0.50	0.67	NS-	NS-	0.79	0.67	NS-	NS-	NS+	NS+
Swainsona procumbens	0.75	-	0.65	0.65	-	-	1.11	0.96	NS+	NS+	NS+	NS+
Wurmbea latifolia	1.15	-	3.02	2.16	*+	NS+	5.31	4.67	NS-	NS-	*+	*+