

Influence of pastoral management on plant biodiversity in a depleted short tussock grassland, Mackenzie Basin

David A. Norton^{1*}, Peter R. Espie², Will Murray³ and Jim Murray³

¹School of Forestry, University of Canterbury, Private Bag 4800, Christchurch, New Zealand

²AgScience Ltd., PO Box 1475, Dunedin, New Zealand

³Glenmore Station Ltd., PO Box 97, Lake Tekapo, New Zealand

*Author for correspondence (E-mail: david.norton@canterbury.ac.nz)

Published on-line: 9 October 2006

Abstract: This study investigated the effects of different management inputs (fertiliser and seed) and grazing patterns on plant biodiversity in a short tussock grassland with a strong *Hieracium pilosella* component. Cover abundance of vascular and non-vascular plants and environmental variables were measured in 32 10×10-m plots located in five blocks with different management treatments. Ordination of the floristic data separated the block with the highest management inputs from other blocks. Several adventive species were significantly more abundant in this block, while several native species were either absent or uncommon, but were significantly more abundant in other study blocks. *H. pilosella* was significantly more abundant in blocks with lower management inputs. Diversity was significantly higher in the block with the highest management inputs. The native tussock *Poa colensoi* had significantly greater cover abundance while *Festuca novae-zelandiae* tussocks were significantly taller in this block. Our results suggest that high management inputs reduce the abundance of *H. pilosella* and diversity of native species, but increase the abundance of other adventive species and the cover and vigour of native tussocks. Our results highlight an interesting management conundrum for short tussock grasslands. No-input management is likely to result in a decline in native biodiversity, as well as production values, as *H. pilosella* mats deplete soil nutrients and restrict regeneration of native species. However, input of fertiliser and adventive seeds to enhance production values, although resulting in an increase in the vigour and abundance of some native species (mainly tussocks) and a reduction in *H. pilosella* abundance, will also result in a decline in overall native species richness.

Keywords: short-tussock grassland; biodiversity conservation; farm management; *Hieracium pilosella*; high country

Introduction

Grasslands dominated by native tussock forming grasses from the genera *Chionochloa*, *Festuca* and *Poa* are the main vegetation cover of the inter-montane basins and ranges of the eastern South Island high country (Mark and McLennan, 2005). Although much of this area was forested prior to human settlement (McGlone, 2001), dramatic ecological transformations have occurred with both Polynesian and European settlement (Molloy et al., 1963; Connor, 1964; Scott, 1984; McGlone and Moar, 1998; Ledgard, 2001; O'Connor, 2003) due to human induced fires, grazing by sheep and cattle, and through the deliberate and accidental introduction of adventive species, resulting in large areas of induced grassland. As a result of these changes it is possible that some of these high country ecosystems are now crossing ecological thresholds that are unlikely to be readily reversed (Hobbs and Norton, 2004). These

induced high country grasslands are used primarily for pastoral fine-wool production, but have recreation, conservation and other values and there is considerable debate over their appropriate long-term management, particularly the interaction between pastoralism and native biodiversity conservation (Swaffield and Hughey, 2001; O'Connor, 2003; Mark and Dickinson, 2004; Mark and McLennan, 2005).

Despite considerable ecological research on the interaction between pastoral farming and native biodiversity conservation in New Zealand tussock grasslands, most of this has focused solely on the effects on native biodiversity of excluding domestic stock with little consideration of other management effects (Walker, 2000; Meurk et al., 2002; Mark and Dickinson, 2003). It is, however, clear that the impacts of grazing on these grasslands is compounded by the influence of invasive species, especially herbs from the genus *Hieracium* (Treskonova, 1991; Rose et al., 1995;

Duncan *et al.*, 1997). These and other studies strongly suggest that *Hieracium* species are spreading through tussock grasslands irrespective of the occurrence of grazing by domestic stock. Furthermore, there is evidence that more fundamental changes are occurring in these grasslands (e.g. declining plant species richness) independent of either grazing or *Hieracium* invasion (Duncan *et al.*, 2001).

There have been only a few substantive studies that have considered the broader effects of farm management on native biodiversity (Scott, 1999, 2001; Espie and Barratt, 2006), although several studies have addressed the effects of farm management on adventive biodiversity (e.g. Allan and Chapman, 1987). Scott (1999, 2001) in two grazed multi-species trials investigated the influence of nutrient additions, stocking rates and watering on a 25-species pasture mix that was direct drilled into depleted short tussock grassland with a strong *Hieracium pilosella* component. Espie and Barratt (in press) investigated the effects of three levels of development (oversowing and fertiliser addition) in short and tall tussock grasslands. These studies highlight the complex response of both native species and pasture species to management inputs and suggest that it is not possible to consider the sustainable management of tussock grasslands without consideration of the full range of management inputs.

The present study sought to better understand the effects of pastoral management on plant biodiversity in depleted (mainly through invasion of *Hieracium pilosella*) short tussock grassland on one high country farm in the Mackenzie Basin. Specifically we were interested in the effects of different fertiliser and seed inputs, and grazing patterns. We investigated an area of short tussock grassland that had been subjected to essentially the same management regime prior to 1980 when it was subdivided and a number of new management regimes imposed. Although it was not possible to obtain true replication for the different management 'effects', the presence of different management treatments on the same vegetation type provided the opportunity to explore how native plant biodiversity responded to these treatments in a field situation.

Study area

This study was undertaken on Glenmore Station on the western side of Lake Tekapo, Mackenzie Basin (Tekapo Ecological District, 43° 54' S, 170° 26' E, c. 900 m a.s.l.). The study area comprises greywacke sandstone till deposited by the Tekapo Glacier during the last glaciation (Mt John formation, c. 16–17 000 years old: Gair, 1967; Maizels, 1989). The landforms are a mixture of hilly "kettle hole" moraine, extensive

planner moraine and intervening glacial outwash channels. Soils have developed from loess that has been deposited on these moraines and belong to the Tekapo Series (Anon, 1968), corresponding to upland Humose, Orthic Brown Soils in the current New Zealand Soil Classification (Hewitt, 1998; Scott, 1999).

The climate is semi-continental, with warm dry summers and cold winters. No detailed climate data is available for the study area, but data from adjacent stations suggests that annual rainfall is c. 750–800 mm with considerable temporal variation. Mean annual temperature at Godley Peaks (762 m; 1951–1980 normals), 5 km to the north, is 8.5 °C, with a January mean daily maximum of 21.0 °C and a July mean daily minimum of -3.3 °C (Anon. 1983). Frosts are frequent in winter (annual mean of 154 days at Lake Tekapo: Scott, 1999) and snow falls several times each winter.

The current vegetation is depleted short tussock grassland with the adventive herb *Hieracium pilosella* the dominant ground cover species. The native tussock forming grasses *Festuca novae-zelandiae* and *Poa colensoi* are conspicuous, although their total cover is low. A variety of other native and adventive herbs and grasses are present, although woody species are rare.

Management history

Glenmore Station, which has been managed by the Murray family since 1918, was initially stocked with half-breed sheep and then since 1940 with merino sheep. The study area was initially run as part of an extensive ewe, wether and hogget block with the stocking rate being similar across the whole block. The first subdivision fencing in 1890 resulted in the formation of two large blocks, Little Downs (664 ha) and Big Downs (1515 ha). In the 1960s and 1970s Little Downs was grazed by merino sheep from October to January (250 ewes and lambs), and then from February to mid March (350 hoggets). It was spelled (i.e. retired from grazing) in April, then grazed from mid-May to end June during mating (250 ewes), before being spelled again from July to the end of September. Big Downs was grazed from February through July (800–1500 hoggets) and June and July (1200–1400 wethers), and then spelled from August through January. No other stock types were grazed and no over-sowing or fertiliser additions were made in either block.

In 1980 both blocks were subdivided to allow better control of *Hieracium pilosella* and rabbits through closer fencing and more tightly controlled grazing. Little Downs was split into Little Downs (516 ha) and Stud blocks (148 ha) while Big Downs was split into four blocks: (Hartleys 148 ha), Fishing Hut (369 ha), Sunday (747 ha), and Peters Patch (248 ha). A further change occurred in the 1990s, when Fishing Hut and

Hartleys were deer fenced and deer introduced (1996 in Hartleys, 1998 in Fishing Hut). These four blocks plus Little Downs were chosen to investigate the effect of management practices as they contained similar landforms, soils and altitudes. The management history of each block is now reviewed with the key points summarised in Table 1. Data for current management is based on 2000–2005. Unless otherwise stated all grazing involves merino sheep.

Hartleys

Direct drill seeding (3 kg ha⁻¹ of 90:10 white clover (*Trifolium repens*) – suckling clover (*Trifolium dubium*) mix and 1 kg ha⁻¹ alsike clover (*Trifolium hybridum*)) and superphosphate fertiliser addition (100 kg ha⁻¹) started as soon as this block was subdivided. Subsequently the block has been aerial top dressed with 150 kg ha⁻¹ superphosphate (sulphur max) every second year. This block was aerial over-sown twice initially, and then once since 2000. This involved a seed mix comprising a predominantly white clover– suckling clover mix, and was sown at 4 kg ha⁻¹. Typically 175–200 ewes (and lambs) were grazed on this block from October (lambing) to February (weaning). The block was then spelled from March to September, except for 450 ewes carried for three weeks in June. With the erection of the deer fence, 220 red deer hinds are now grazed from June to February, with the hinds fed out from June through September. Some 60 stud ewes (and lambs) are also carried from October to February. The block is spelled from March to May.

Fishing Hut

The part of this block that we studied was aerial top-dressed and over-sown in 1999 and 2001 with the same fertiliser and seed mix application as in Hartleys, but no direct drilling was undertaken. Typically 250–300 ewes (and lambs) were grazed in this block from October (lambing) to February (weaning). The block was then spelled from March to June, with 1000 ewes then carried through July and August with feeding out. The block was again spelled in September. With the erection of the deer fence, 300 ewes are now grazed from October to February, and 220 red deer stags run from late February through to August, with silage fed out daily from June to August. The block is again spelled in September.

Sunday

This block has had no improvements and has been subjected to the same management regime since the early 1980s which has involved 100 ewes (and lambs) from October (lambing) to December (tailing), with the block spelled for the rest of the year (January to September). A Queen Elizabeth the Second National

Trust (QEII) open space covenant was gazetted over this block in 2003 to protect nationally significant kettle-hole tarns.

Peter Patch

This block was top dressed in 1996, 1998 and 2000 at a rate of 150 kg ha⁻¹ superphosphate. It was direct drilled in 2002 (same seed mix as in Hartleys, but no fertiliser added). One hundred and fifty ewes (and lambs) are grazed from October (lambing) to December (tailing). The block is then spelled from January to September. A QEII open space covenant was also gazetted over this block in 2003 to protect nationally significant kettle-hole tarns.

Little Downs

Although this block is grazed as one unit, one half of the block has been subjected to regular aerial fertiliser and over-sowing as well as drilling while the other has had no improvements. Our research was undertaken in the unimproved part of the block. Three hundred and fifty ewes (and lambs) are grazed from October (lambing) to February (weaning). The block is then spelled in March and April. Between 450 and 500 ewes are carried from May to October and are fed out from June to September, although this is all on the developed part of the block. Thirty cows (and calves) are also carried from August to March and are fed out in August and September (again on the developed part of the block). Personal observation (W. Murray) and the lack of cattle sign suggests that the unimproved part of the block that we worked in is only subjected to light sheep grazing. Data presented in Table 1 is for the undeveloped part of the block only.

Methods

We used a stratified random design to position 32 10×10-m plots within the five blocks. Potential locations were restricted by landform type (level to gently rolling moraine) and altitude (850–900 m), and stratified to give an approximately regular spacing through each block with the number of plots proportional to the available block area. Plot locations were finalised by random numbers determining direction and distance from the initial potential position.

The cover abundance of all vascular and non-vascular (bryophyte and lichen) plant species within plots was visually estimated using a seven-point scale (r , $\leq 0.1\%$; 1, 0.1–0.9%; 2, 1–5%; 3, 5–25%; 4, 26–50%; 5, 51–75%; 6, 76–100%). Cover was independently estimated by two observers (DN & PE), and re-scored if estimates differed by more than one point. Field assessments were made during February 2005.

Table 1. Management inputs (2000–2005), soil chemistry, environmental attributes and vegetation attributes of the study blocks, with ANOVA *P* values for site and vegetation attributes. Means with the same superscript letter are not significantly different from each other ($\alpha = 0.05$) for significant variables ($P \leq 0.003$; Bonferroni correction). Block names (ordered as along axis 1 of the DCA ordination; Fig. 1): H, Hartleys ($n = 5$ plots); PP, Peters Patch ($n = 7$); FH, Fishing Hut ($n = 9$); LD, Little Downs ($n = 5$); S, Sunday ($n = 6$). Hiepil, *Hieracium pilosella*. Fesnov, *Festuca novae-zelandiae*.

	H	PP	FH	LD	S	<i>P</i>
Area (ha)	148	248	369	263	747	
<i>Management inputs</i>						
Stock type	Ewes & hinds	Ewes	Ewes & stags	Ewes & cows	Ewes	
Stock units ha ⁻¹ yr ⁻¹	2.07	0.10	0.62	0.08	0.02	
Fertiliser	Regular	Once	Regular	No	No	
Over sown	Once	No	Once	No	No	
Drilling	Yes	Yes	No	No	No	
<i>Soil chemistry</i>						
pH	5.2	5.5	5.3	5.4	5.5	
Ca (MAF QT)	4	3	3	2	4	
Olsen P (µg/ml)	32	15	28	10	19	
K (MAF QT)	6	5	7	5	7	
S(SO ₄) (ppm)	21	14	14	3	3	
Mg (MAF QT)	8	10	13	11	16	
Na (MAF QT)	2	1	1	1	2	
<i>Site attributes</i>						
Altitude (m)	877±3.0	897±1.4	883±2.2	890±4.6	885±5.4	0.0089
Slope (°)	1.0±0.5	0.6±0.2	0.2±0.2	0.4±0.2	0.8±0.7	0.5158
Rock (%)	0.9±0.5	1.7±0.8	1.6±0.6	1.5±0.5	1.6±0.5	0.9544
Stone (%)	0.2±0.1	1.1±0.5	2.1±1.1	0.9±0.4	0.9±0.4	0.7471
Bare (%)	9.3±6.7	12.3±2.5	18.9±4.5	18.0±6.4	8.7±2.3	0.2685
<i>Vegetation attributes</i>						
Total cover (%)	102.4±7.9	88.5±1.9	81.2±5.6	83.9±8.1	92.9±1.7	0.0913
Hiepil / total cover	0.39±0.07 ^a	0.82±0.04 ^b	0.88±0.03 ^b	0.86±0.02 ^b	0.94±0.01 ^b	<0.001
Adventive cover (%)	85.4±8.9	81.5±1.6	72.4±4.1	74.8±7.2	87.2±1.7	0.1435
Native cover (%)	17.0±6.7	7.0±0.8	8.8±2.4	9.2±2.1	5.7±0.7	0.1352
Total species richness (S)	27.2±1.1	31.9±1.9	29.2±0.9	29.0±1.9	30.3±1.3	0.2788
Native species richness (S)	17.4±1.0 ^a	23.7±1.4 ^b	21.7±1.0 ^b	21.8.0±1.6 ^b	25.3±1.1 ^b	0.0030
Adventive sp. richness (S)	9.8±0.6 ^a	8.1±0.9 ^{ab}	7.4±0.4 ^b	7.2±0.5 ^b	5.0±0.6 ^c	0.0004
Simpson's diversity (D)	0.69±0.04 ^a	0.29±0.06 ^b	0.20±0.04 ^b	0.24±0.03 ^b	0.11±0.01 ^b	<0.0001
D – Hiepil	0.65±0.08	0.66±0.04	0.73±0.03	0.64±0.04	0.77±0.03	0.1705
Fesnov leaf height (cm)	52.8±2.9 ^a	33.7±1.8 ^b	33.8±2.3 ^b	33.4±3.8 ^b	28.0±1.0 ^b	<0.0001
Fesnov inflor. height (cm)	68.8±3.4 ^a	46.6±2.7 ^b	52.7±4.4 ^b	45.6±4.5 ^b	50.2±2.3 ^b	0.0020

Plot coordinates (NZ Map Grid) and altitude were determined by Global Positioning System. Landform, slope and aspect, and percentage rock cover (≥ 5 cm across), stone cover (< 5 cm across) and bare ground (including litter), were also recorded. We also noted evidence of management inputs (e.g. presence of elemental sulphur from top-dressing and drill lines), and measured leaf and inflorescence length of the five tallest *Festuca novae-zelandiae* tussocks in each plot.

Field data were entered onto a spreadsheet with cover abundance scores converted to the mid-points of their respective cover classes. We used analysis of variance (ANOVA) to compare individual floristic and environmental variables between the five study block and ordination analysis to identify differences in community composition.

From the floristic data we calculated vegetation cover (total, native, adventive, ratio of *Hieracium pilosella*/total cover), species richness (total, native, adventive), Simpson's diversity (Magurran, 2004), and Simpson's diversity minus *H. pilosella*. We then compared these indices, the measures of *Festuca novae-zelandiae* leaf and inflorescence length, environmental variables and individual plant species cover abundances between the five study blocks using ANOVA as implemented using SAS version 8. Because multiple contrasts were made between the five study blocks, Bonferroni corrections were applied to determine an appropriate alpha level for determining significance. *A posteriori* Student-Newman-Keuls (SNK) tests (Underwood 1997) were used to test for significant differences between individual block means

for those ANOVAs that were significant after applying the Bonferroni correction. Proportional data were arcsine(sqrt) transformed prior to analysis. Means are presented ± 1 standard error.

Ordination was undertaken using Detrended Correspondence Analysis (DCA) and Detrended Canonical Correspondence Analysis (DCCA) to detect the underlying structure of the data and to assess relationships with environmental variables as implemented in CANOCO (ter Braak and Smilauer, 1998). DCA provides an indirect ordination of the species by plot data matrix, identifying the dominant floristic compositional gradients independent of other factors. In contrast, DCCA, a form of direct gradient analysis, extracts the dominant gradients with the constraint that they must be orthogonal linear combinations of independent environmental variables. Comparison of DCA and DCCA ordinations assists interpretation of the relative importance of the measured environmental variables in explaining the underlying floristic gradients. To facilitate interpretation of the dominant floristic gradients we removed *Hieracium pilosella* from the ordination data set prior to analysis as this species is so dominant in all sample plots and its inclusion masked more subtle floristic patterns. The ordinations were undertaken using default options in CANOCO.

Standard agricultural soil nutrient tests were undertaken in the five blocks by Ballance Agri-Nutrients Ltd. in May 2005. Soil samples were collected from each of our sample plots and bulked to form one sample per block. These were then tested for pH (1:2.1 v/v water slurry), Ca (Ammonium Acetate extraction: AA determination), P (Olsen extraction: colorimetry), K (Ammonium Acetate extraction: AA determination), S (SO_4) (Potassium Phosphate extraction: IC), Mg (Ammonium Acetate extraction: AA determination), and Na (Ammonium Acetate extraction: AA determination).

For each block we also calculated the stocking rate (stock units $\text{ha}^{-1} \text{yr}^{-1}$) based on the average stocking data for the period 2000-2005. We treated ewes (including lambs) as 1.0 stock units, red deer hinds as 1.9 stock units, red deer stags as 2.1 stock units, and cows as 5.3 stock units (Fleming, 1996). For Little Downs the data are only for the undeveloped part of the block where we sampled. Based on personal observation of stock distribution in Little Downs (W. Murray pers. obs.) we assumed that only 20% of stock use was in the undeveloped area.

Results

The five blocks vary considerably in size and management history as described above (Table 1). There

is also two orders of magnitude variation in stocking rate which ranged from 0.02 stock units $\text{ha}^{-1} \text{yr}^{-1}$ in Sunday to 2.07 stock units $\text{ha}^{-1} \text{yr}^{-1}$ in Hartleys. Soil nutrient tests suggested that all blocks had a similar pH (Table 1), but that there were differences in soil fertility, although on an agricultural scale all soils are relatively infertile. Hartleys had the highest value for Ca, Olsen P, S(SO_4) and Na, and second highest value of K of the five blocks, although it had the lowest Mg value. In contrast Little Downs had the lowest values for Ca, Olsen P, K, S(SO_4) and Na. The other blocks were intermediate in their values.

Reflecting the restrictions imposed in our experimental design, there were no significant differences ($P \leq 0.003$) in sample plot slope or altitude among the five study blocks. There were also no significant differences in the cover of rocks, stones or bare ground between study blocks (Table 1). There was no significant difference ($P \leq 0.003$) in total plant cover abundance, or the abundance of adventive and native cover individually, between the five study blocks (Table 1), although the proportion of total cover comprising *Hieracium pilosella* was significantly less ($P \leq 0.003$) in Hartleys than in the other four blocks. While there was no significant difference in total and native species richness between the study blocks, there was a significant difference in exotic species richness which was highest in Hartleys and lowest in Sunday. There was also significant difference in Simpsons diversity reflecting the dominance of *Hieracium pilosella*, with diversity significantly higher in Hartleys than in the other blocks (Table 1). However, removal of *Hieracium pilosella* from the data set resulted in no significant differences in this index. While there was no significant difference in the abundance of *Festuca novae-zelandiae* plants among study blocks ($P=0.238$), tussocks of this species had significantly longer leaves and inflorescences in Hartleys than in the other four blocks (Table 1).

Only 10/65 species recorded had statistically significant ANOVA models ($P \leq 0.008$; Table 2). Two main patterns were apparent in the distribution of these species among study blocks. The first were species that were either far more abundant in Hartleys (and occasionally in either Peters Patch or Little Downs) than in the other study blocks. These were mainly adventive species (e.g. *Agrostis capillaris*, *Bromus tectorum*, *Trifolium* species, *Veronica arvensis*), but the native grass *Poa colensoi* also showed this pattern (Table 2). The second pattern was shown by species that were either absent or uncommon in Hartleys, but were significantly more abundant in other study blocks (and always including Sunday). This group mainly comprised native species (e.g. *Brachyglottis haastii*, *Stereocaulon ramulosum*, *Wahlenbergia albomarginata*) but also included the adventive herb *Hieracium pilosella*.

Table 2. Summary statistics for species abundances for those species with a significant ($P \leq 0.0008$; Bonferroni correction) ANOVA test statistic. Means with the same superscript letter are not significantly different from each other ($\alpha = 0.05$). Block names as in Table 1.

	H	PP	FH	LD	S	P
<i>Agrostis capillaris</i>	4.56±1.68 ^a	0.16±0.05 ^b	0.11±0.04 ^b	2.32±0.90 ^a	0.10±0.04 ^b	<0.0001
<i>Brachyglottis haastii</i>	0.01±0.01 ^a	0.07±0.02 ^b	0.04±0.02 ^{ab}	0.01±0.01 ^a	0.09±0.02 ^b	0.0004
<i>Bromus tectorum</i>	0.26±0.13 ^a	0 ^b	0 ^b	0 ^b	0 ^b	0.0003
<i>Hieracium pilosella</i>	39.0±8.1 ^a	72.9±4.5 ^b	70.7±3.9 ^b	71.5±7.1 ^b	86.7±1.7 ^b	<0.0001
<i>Poa colensoi</i>	1.38±0.29 ^a	0.287±0.129 ^b	0.2±0.04 ^b	0.18±0.073 ^b	0.133±0.076 ^b	<0.0001
<i>Stereocaulon ramulosum</i>	0 ^a	0 ^a	0 ^a	0 ^a	0.007±0.002 ^b	<0.0001
<i>Trifolium arvense</i>	23.8±7.611 ^a	3.486±2.030 ^b	0.59±0.323 ^b	0.002±0.002 ^b	0 ^b	<0.0001
<i>Trifolium repens</i>	8.2±2.332 ^a	4.529±2.802 ^b	0.059±0.016 ^c	0.002±0.002 ^c	0 ^c	<0.0001
<i>Veronica arvensis</i>	0.028±0.018 ^a	0.003±0.002 ^b	0.001±0.001 ^b	0.002±0.002 ^b	0 ^b	0.0005
<i>Wahlenbergia albomarginata</i>	0.01±0 ^a	0.023±0.013 ^a	0.08±0.013 ^b	0.01±0 ^a	0.1±0 ^b	<0.0001

The first axis of the DCA ordination explained 17% of the total floristic variance in the data set, and the second axis 8.8%, suggesting that the first axis is the most important ecologically. There was considerable species turnover along both axes (3.11 and 2.85 SD units) highlighting the considerable floristic variation between sample plots (with *Hieracium pilosella* removed). The study blocks are best separated along ordination axis one (Fig. 1), with sample plots from Hartleys having the lowest values, those from Peters Patch intermediate values, and sample plots from the other three study blocks having the highest values. Species with low values on axis one of the species ordination included the adventives *Bromus tectorum*, *Poa pratensis*, *Trifolium arvense*, *Trifolium repens* and *Veronica arvensis*, all species that were either restricted to or most abundant in Hartleys. In contrast species with high values on axis one were native species typical of more-intact short tussock grassland (e.g. *Pimelia pulvinaris*, *Leucopogon fraseri*, *Raoulia hookerii*, *Prasophyllum colensoi* and *Carex mulleri*). Vectors for the five environmental variables with r values > 0.5 in the DCA ordination (Fig. 1; Table 3) are aligned along ordination axis one, with stocking rate, Ca, S and Na negatively correlated and Mg positively correlated with this axis. Correlations of the measured environmental variables with the second axis are weak.

Explained variance and gradient length for the first DCCA ordination axis is similar to those for the DCA (14.8 cf. 17.0% variance explained, 2.79 cf. 3.11 SD units) and Spearman's rank correlation coefficients show that the sample scores are almost identical ($r = 0.98$). These results indicate that the measured environmental variables account for much of the dominant floristic compositional variation identified in the DCA along the first axis. The correlation is, however, much weaker between the second axis DCCA and DCA sample scores ($r = -0.33$), and the variance explained and gradient length for this axis is also lower than in the

DCA (4.6 cf. 8.8% variance explained, 1.88 cf. 2.85 SD units). However, the much lower variances explained by the second axis in both ordinations suggest that this difference is of little ecological importance.

Discussion

Of the five blocks sampled, Hartleys stood out as the most distinct floristically and structurally. Our data suggest that while there was little difference in the broad environmental attributes of the Hartleys plots compared to those in the other blocks, there were marked differences in soil fertility. Hartleys has had the most intensive management of the five blocks, including regular fertiliser and seed application, and, at least for the last 10 years, the highest stocking rate of all the blocks. It would therefore seem that the most likely driver of the observed floristic differences were these management inputs.

Hartleys was the only block to have a substantial cover abundance of adventive species other than *Hieracium pilosella*, with the cover of clovers (*Trifolium arvense*, *T. dubium* and *T. repens*) and adventive grasses (*Agrostis capillaris*, *Bromus tectorum* and *Poa pratensis*) all significantly higher here than in any other block. Scott (2001) also noted a significant positive response of some of these species to fertiliser addition and grazing in his fertiliser and fertiliser×grazing trials in similar depleted short tussock grassland. In particular he observed higher abundances of *T. repens*, *A. capillaris* and *B. tectorum* with either increased P and/or increased S fertiliser application. He also observed higher abundances of *T. repens* and *B. tectorum* at moderate and high grazing rates, but no effect for *A. capillaris*. Scott (2001) also observed an increase in species richness and diversity with an increase in fertiliser application from low to medium, although the trend was reversed in his high fertiliser treatment, and

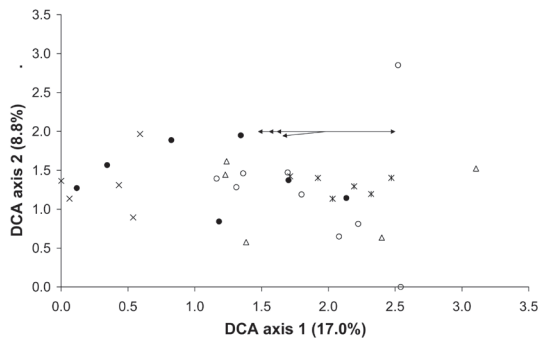


Figure 1. Detrended correspondence analysis of plots for the all species minus *Hieracium pilosella* ordination. Plot codes: × Hartleys. • Peters Patch. ○ Fishing Hut. ∗ Sunday. △ Little Downs. The environmental vector pointing to the right is Mg, while those pointing to the left are stocking rate, Ca, S and Na.

from low to high stocking, similar to the differences observed here.

In contrast the species richness of native species was lower in Hartleys than in the other blocks. Furthermore, tussock cover abundance (*Poa colensoi*) and heights (*Festuca novae-zelandiae*) were also greater in Hartleys than in the other blocks. Scott (2001) observed that native species richness was highest in his zero fertiliser treatment and low across all other fertiliser treatments, while Espie and Barratt (2006) found a reduction in native species richness with the addition of fertiliser and pasture species in four tussock grassland communities. The addition of fertilizer has also been shown to significantly increase the size of *Festuca novae-zelandiae* and *Chionochloa* tussocks in other studies (Molloy and Connor, 1970; Espie, 1990).

Perhaps the most dramatic vegetation difference observed among our five study blocks was the significantly lower cover abundance of *Hieracium pilosella* in Hartleys. Not only was the cover abundance of *H. pilosella* significantly lower (39% cf. 71–87%) but the ratio of *H. pilosella* cover abundance to total cover abundance was even more markedly different (0.39 cf. 0.82–0.94). Clearly, the abundance of *H. pilosella* is having a major influence on the overall ecology of these depleted short tussock grasslands. Sunday, the block with no fertiliser input and the lowest stocking rate, was the most different from Hartleys floristically, and had the highest *H. pilosella* cover abundance and the highest richness of native species (although the lowest native cover abundance).

The differences observed here are consistent with patterns observed in other studies, with diversity decreasing when fertiliser is added to a plant community

Table 3. Correlations between the first two DCA ordination axes and environmental variables of the 'All species – *Hieracium pilosella*' ordination.

Variables	Axis 1	Axis 2
Altitude (m)	0.076	-0.019
Slope (°)	-0.294	-0.071
Rock (%)	0.304	-0.465
Stone (%)	0.464	-0.434
Bare (%)	0.249	-0.213
Stocking (stock units ha ⁻¹ yr ⁻¹)	-0.661	0.058
pH	0.477	0.008
Ca (MAF QT)	-0.571	0.082
Olsen P (µg/ml)	-0.491	0.011
K (MAF QT)	0.232	-0.125
S(SO ₄) (ppm)	-0.750	0.108
Mg (MAF QT)	0.782	-0.139
Na (MAF QT)	-0.522	0.058

(DiTommaso and Aarssen, 1989; Gough *et al.*, 2000). These patterns are caused by competition for both below-ground and above-ground resources (Tilman, 1988), with the relative importance of these in grassland ecosystems dependent on productivity and structure. Root competition is usually considered more important in less productive and simply structured grasslands while light competition is more important in more productive and complex structured grasslands (Rajaniemi, 2002). The vigorous growth of adventive grasses and legumes after fertiliser addition and over-sowing in depleted short tussock grasslands (Scott, 2001) suggests that competition for light might be a serious limitation to native species recruitment in this situation (cf. Henry *et al.*, 2004).

The role of *Hieracium pilosella* in the ecology of New Zealand tussock grasslands has been extensively discussed and it is clear that this invasive species has had and continues to have a major impact in these grasslands, especially through a decline in the abundance and richness of many native species (Espie, 2001). While there has been debate about the model(s) that might explain the success of *H. pilosella* in invading these grasslands, and Rose *et al.* (1998) caution against single-factor explanations for this success, it does seem clear that at least in degraded short tussock grasslands similar to those in our study site that there is an association between fertiliser inputs and reduced cover of *H. pilosella* (Scott *et al.*, 1990; Scott, 1993, 2001; Duncan *et al.*, 1997; Rose *et al.*, 1998).

While the Sunday block, which had no fertiliser or seed inputs and the lowest stocking rate, had the highest richness of native species it also had the highest abundance of *Hieracium pilosella* and the lowest

abundance of native species, although none of these differences were statistically significant. The literature strongly implicates the expansion of *H. pilosella* with a decline in the abundance of native species (Treskonova, 1991; Rose *et al.*, 1995; Espie, 2001). Given the limited dispersal distances of many tussock grassland plants, including several of the species in our study (Spence, 1990), it would seem likely that as plant abundance and especially frequency declines (Rose *et al.*, 1995), the chances of recolonisation will also decline thus exacerbating the effect of *H. pilosella* invasion on native biodiversity.

The results of our research together with the results of other studies of short tussock grasslands highlight an interesting management conundrum if biological control fails to significantly reduce *Hieracium pilosella* abundance. No-input management (as has occurred in Sunday block) is likely to result in a decline of conservation values (native biodiversity), as well as production values, as *H. pilosella* mats both deplete soil nutrients and restrict regeneration of native species. However, management input of fertiliser and adventive seeds to increase the abundance and enhance the vigour and persistence of dominant species (as has occurred in Hartleys), although resulting in an increase in the vigour and abundance of some native species (mainly tussocks), will also result in a decline in overall native species richness as a few, mainly adventive legume and grass species, dominate.

It is obvious that the type of management input required in short tussock grasslands will depend on the management goals for the grassland concerned. Fertiliser can be used to enhance the vigour and abundance of native tussocks, but will most-likely result in the loss of other native grassland species, especially if applied in conjunction with the sowing of adventive grassland species, although it is less clear what the effect of fertiliser addition without adventive seed addition will be on native biodiversity. Where the management goals are pastoral production, then it seems clear that the only viable management option is to maintain fertiliser and adventive seed inputs, otherwise *H. pilosella* mats will continue to deplete soil nutrients resulting in the declines in soil and vegetation condition that have been well documented in other studies (Martin, 1994). At the whole-property scale it is probable that active management inputs will be required to maintain areas of short tussock grassland where the specific management goal is maintaining high native species diversity.

Acknowledgements

This paper is dedicated to the memory of Rodney Patterson (1955–2006) who inspired this research. We

thank Maria Bosman and Tammy McMahon for field assistance, and Ben Todhunter, Richard Duncan and two anonymous referees for constructive comments on the manuscript. This research was supported by funding from the Ministry of Agriculture and Forestry Sustainable Farming Fund (Grant 04/063), High Country Accord Trust, High Country Committee of Federated Farmers New Zealand, and Merino Inc.

References

- Anon. 1968. *General survey of the soils of South Island, New Zealand*. Soil Bureau, Department of Scientific and Industrial Research, Wellington, N.Z. 404 pp.
- Anon. 1983. *Temperature normals 1951 to 1980*. New Zealand Meteorological Service, Wellington, N.Z. 25 pp.
- Allan, B.E., Chapman, H.M. 1987. Over sown tussock country; lessons from 30 years improvement and management on Tara Hills. *Proceedings of the New Zealand Grassland Association* 48: 77-81.
- Connor, H.E. 1964. Tussock grassland communities in the Mackenzie country, south Canterbury, New Zealand. *New Zealand Journal of Botany* 2: 325-351.
- DiTommaso, A., Aarssen, L.W. 1989. Resource manipulation in natural vegetation: a review. *Vegetatio* 84: 9-29.
- Duncan, R.P., Colhoun, K.M., Foran, B.D. 1997. The distribution and abundance of *Hieracium* species (hawkweeds) in the dry grasslands of Canterbury and Otago. *New Zealand Journal of Ecology* 21: 51-62.
- Duncan, R.P., Webster, R.J., Jensen, C.A. 2001. Declining plant species richness in the tussock grasslands of Canterbury and Otago, South Island, New Zealand. *New Zealand Journal of Ecology* 25: 35-47.
- Espie P.R. 1990. The impact of nutrients on fescue tussock grasslands. *What's New in Forest Research* 189: 1-4.
- Espie, P. 2001. *Hieracium in New Zealand: ecology and management*. AgResearch Ltd., Mosgiel, N.Z. 66 pp.
- Espie, P., Barratt, B.I.P. 2006 Biodiversity of indigenous tussock grassland communities in Otago, Canterbury and the Central North Island of New Zealand IV. Vegetation and the effect of disturbance by agricultural development and fire. *Journal of the Royal Society of New Zealand* 36: 69-82.
- Fleming, P. 1996. *Farm technical manual*. Department of Farm and Horticultural Management, Lincoln University, Canterbury, New Zealand, N.Z.

- Gair, H.S. 1967. *Sheet 20. Mt Cook*. Geological Map of New Zealand 1:250,000. Department of Scientific and Industrial Research, Wellington, N.Z.
- Gough, L., Osenberg, C.W., Gross, K.L., Collins, S.L. 2000. Fertilization effects on species density and primary productivity in several herbaceous plant communities. *Oikos* 89: 428-439.
- Henry, M., Stevens, H., Bunker, D.E., Schnitzer, S.A., Carson, W.P. 2004. Establishment limitation reduces species recruitment and species richness as soil resources rise. *Journal of Ecology* 92: 339-347.
- Hewitt, A.E. 1998. *New Zealand soil classification*. Manaaki Whenua Press, Lincoln, N.Z. 133 pp.
- Hobbs, R.J., Norton, D.A. 2004. Ecological filters, thresholds, and gradients in resistance to ecosystem reassembly. In: Temperton, V.M., Hobbs, R.J., Nuttle, T.; Halle, S. (Editors), *Assembly rules and restoration ecology*. Pp 72-95. Island Press, Washington D.C., U.S.A.. 439 pp.
- Ledgard, N. 2001. The spread of lodgepole pine (*Pinus contorta*, Dougl.) in New Zealand. *Forest Ecology and Management* 141: 43-57.
- Magurran, A.E. 2004. *Measuring biological diversity*. Blackwell Publishing, Oxford, U.K. 256 pp.
- Maizels, J.K. 1989. Differentiation of late Pleistocene terrace outwash deposits using geomorphic criteria: Tekapo valley, South Island, New Zealand. *New Zealand Journal of Geology and Geophysics* 32: 225-241.
- Mark, A.F., Dickinson, K.J.M. 2003. Temporal responses over 30 years to removal of grazing from a mid-altitude snow tussock grassland reserve, Lammerlaw Ecological Region, New Zealand. *New Zealand Journal of Botany* 41: 655-668.
- Mark, A.F., Dickinson, K.J.M. 2004. South Island high country in transition: issues, options and initial outcomes with the tenure review process. In: Kearsley, G.; Fitzharris B. (Editors), *Glimpses of a Gaian world*, pp. 285-308. School of Social Science, University of Otago, Dunedin, N.Z. 400 pp.
- Mark, A.F., McLennan, B. 2005. The conservation status of New Zealand's native grasslands. *New Zealand Journal of Botany* 43: 245-270.
- Martin, G. 1994. *South Island high country review*. Final report from the Working Party on Sustainable Land Management, Wellington, N.Z. 184 pp.
- McGlone, M.S. 2001. The origin of the native grasslands of southeastern South Island in relation to pre-human woody ecosystems. *New Zealand Journal of Ecology* 25: 1-15.
- McGlone, M.S., Moar, N.T. 1998. Dryland Holocene vegetation history, Central Otago and the Mackenzie Basin, South Island, New Zealand. *New Zealand Journal of Botany* 36: 91-111.
- Meurk, C. D., Walker, S., Gibson, R.S., Espie, P. 2002. Changes in vegetation states in grazed and ungrazed Mackenzie Basin grasslands, New Zealand, 1990-2000. *New Zealand Journal of Ecology* 26: 95-106.
- Molloy, B.P.J., Burrows, C.J., Cox, C.J., Johnston, J.A., Wardle, P. 1963. Distribution of subfossil forest remains, eastern South Island, New Zealand. *New Zealand Journal of Botany* 1: 68-77.
- Molloy, B.P.J., Connor, H.E. 1970. Seedling growth in four species of *Chionochloa* (Gramineae). *New Zealand Journal of Botany* 8: 132-152.
- O'Connor, K.F. 2003. Conflicting innovations: A problem for sustainable development of New Zealand high-country grasslands. *Mountain Research and Development* 23: 104-109.
- Rajaniemi, T.K. 2002. Why does fertilization reduce plant species diversity: testing three competition-based hypotheses. *Journal of Ecology* 90: 316-324.
- Rose, A.B., Basher, L.R., Wiser, S.K., Platt, K.H., Lynn, I.H. 1998. Factors predisposing short-tussock grasslands to *Hieracium* invasion in Marlborough, New Zealand. *New Zealand Journal of Botany* 22: 121-140.
- Rose, A.B., Platt, K.H., Frampton, C.M. 1995. Vegetation change over 25 years in a New Zealand short-tussock grassland: effects of sheep grazing and adventive invasions. *New Zealand Journal of Ecology* 19: 163-174.
- Scott, D. 1984. Hawkweeds in run country. *Tussock Grasslands and Mountain Lands Institute Review* 42: 33-48.
- Scott, D. 1993. Response of *Hieracium* in long term manipulative agricultural trials. *New Zealand Journal of Ecology* 17: 41-46.
- Scott, D. 1999. Sustainability of New Zealand high-country pastures under contrasting development inputs. 1. Site, and shoot nutrients. *New Zealand Journal of Agricultural Research* 42: 365-383.
- Scott, D. 2001. Sustainability of New Zealand high-country pastures under contrasting development inputs. 7. Environmental gradients, plant species selection, and diversity. *New Zealand Journal of Agricultural Research* 44: 59-90.
- Scott, D., Robertson, J.S., Archie, W.J. 1990. Plant dynamics of New Zealand tussock grasslands infested with *Hieracium pilosella* l. Effects of seasonal grazing, fertiliser and overdrilling. *Journal of Applied Ecology* 27: 224-234.
- Spence, J.R. 1990. Seed rain in grassland, herbfield, snowbank, and fellfield in the alpine zone, Craigieburn Range, South Island, New Zealand. *New Zealand Journal of Botany* 28: 439-450.
- Swaffield, S., Hughey, K. 2001. The South Island high country of New Zealand, landscape challenges

- and future management. *Mountain Research and Development* 21: 320-326.
- ter Braak, C.J.F., Smilauer, P. 1998. *CANOCO reference manual and user's guide to canoco for Windows: software for canonical community ordination (version 4)*. Microcomputer Power, Ithaca, U.S.A.
- Tilman, D. 1988. *Plant strategies and the dynamics and structure of plant communities*. Princeton University Press, Princeton, U.S.A.
- Treskonova, M. 1991. Changes in the structure of tall tussock grasslands and infestation by species of *Hieracium* in the Mackenzie country, New Zealand. *New Zealand Journal of Ecology* 15: 65-78.
- Underwood, A.J. 1997. *Experiments in ecology*. Cambridge University Press, Cambridge, U.K., 504 pp.
- Walker, S. 2000. Post-pastoral changes in composition and guilds in a semi-arid conservation area, Central Otago, New Zealand. *New Zealand Journal of Ecology* 24: 123-137.

Editorial Board member: Richard Duncan