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THE MAINTENANCE OF *POA CITA* GRASSLAND BY GRAZING

Summary: The effect of protection from sheep grazing on indigenous plant species in relatively mesic, strongly modified *Poa cita* grassland, was examined using paired grazed and ungrazed sites. Ungrazed sites, fenced at times ranging from four to more than 24 years ago, had fewer species, lower indigenous cover and lower tussock density than adjacent grazed sites. The dominance of adventive species, particularly cocksfoot *(Dactylis glomerata),* in ungrazed sites appeared to be important to these differences. Grazing would appear to be necessary for the maintenance of indigenous species in this strongly modified short tussock grassland.

Keywords: short tussock grassland; grazing; *Poa cita;* silver tussock; *Dactylis glomerata;* cocksfoot; Port Hills, Canterbury, New Zealand.

Introduction

Short tussock grassland is widespread in eastern South Island, occupying the equivalent of 7% of the total land area of New Zealand (Newsome, 1987). Despite this, short tussock grassland is under-represented in New Zealand's reserve system (Mark, 1985) and therefore the appropriate management of those areas of grassland that are protected is important.

Before the arrival of humans in New Zealand, short tussock grassland would have been restricted to frost-flats and riparian sites such as river flood-plains (Cockayne, 1928; Connor, 1969). Polynesian and European deforestation led to the rapid expansion of short tussock grassland which was then substantially modified in many areas by the invasion of adventive species. Nowadays, much short tussock grassland must be regarded as semi-natural in character because of its modified nature and anthropogenic origin.

It has already been recognised that active management such as grazing or burning may be necessary for the preservation of much short tussock grassland (Molloy, 1971; O'Connor, 1982; Meurk, 1987; Norton, 1988). However, because short tussock grassland is largely an induced association of both adventive and indigenous species, it differs substantially from overseas examples of managed grassland systems. In British managed grasslands, indigenous species that have developed under mammalian herbivory dominate the vegetation. If these grasslands are not grazed, tall coarse grasses and scrub dominate at the expense of annuals and low-growing perennials (Wells, 1969; Gibson *et al.,* 1987). As grasses and grassland species indigenous to New Zealand evolved without mammalian herbivory, grazing by stock may be detrimental.

Previous work in New Zealand short-tussock grassland provides conflicting evidence as to the impact of grazing on indigenous species abundance, with results ranging from an improvement after protection from grazing (e.g., Malcolm, 1925) to a decline (e.g., Sewell, 1952; Barker, 1953). Often, however, no clear directional change in indigenous species abundance is evident, even within the scope of a single study (e.g., Moffat, 1957; Rose, 1983; Scott *et al.,* 1988).

Agriculturally orientated studies on the impact of management practices (e.g., Wraight, 1964; Edge, 1979; Allan, 1985) are of little value in elucidating the role of grazing in the conservation of short tussock grassland, as the main concern is with pasture productivity and little attention is paid to the agriculturally less important indigenous species.

The emphasis of the present study was on the role of grazing in the conservation of indigenous species, with the aim to test the hypothesis that in relatively mesic, strongly modified short tussock grassland, the cessation of grazing has no effect on the number or abundance of indigenous species, all other factors being constant.

Nomenclature follows Cheeseman (1925) and Allan (1961), and taxonomic changes listed in Connor and Edgar (1987), unless otherwise stated.

Study area

The Port Hills, on the eastern edge of the Canterbury Plains (Fig. 1), are part of the eroded remnant of an ancient volcano centered on Lyttelton Harbour. Soils are predominantly of the Stewart-Summit set, with Rapaki Hill complex soils occurring on strongly rolling slopes along the summit and Takahe silt loams on the rolling tops of spurs. All of these soils are derived from varying amounts of basalt and loess and are well drained to seasonally moisture deficient with a moderate natural fertility (Fitzgerald, 1966; Griffiths, 1974).

A rainfall gradient exists along the east-west axis of the Port Hills ranging from 600 mm in the north-

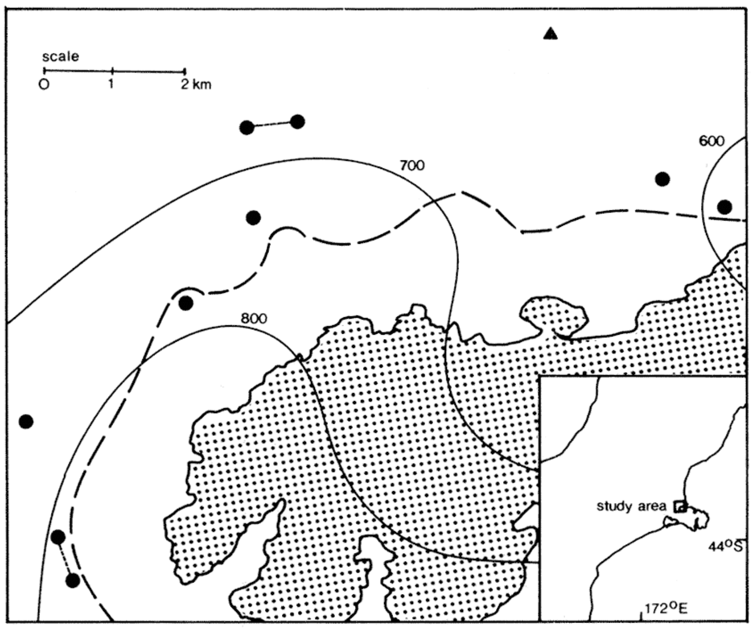

Figure 1: *Location map of Port Hills showing Mt. Pleasant climate station (triangle), rainfall isohyets (in mm) and study sites (circles). Joined circles indicate where adjacent ridges were used for the grazed/ungrazed comparison.*

east to 800 mm in the south-west. Mean annual temperature for the Mount Pleasant climate station, which is situated on the lower slopes of the Port Hills (Fig. 1), is 12,3 ∞C, with a January mean temperature of 16.4∞C and a July mean of 7.6∞C (NZ Meteorological Service, 1981).

The area was forested in pre-human times but Polynesian fires resulted in the expansion of grassland (Molloy, 1969). European settlers further modified the vegetation by burning, grazing and oversowing with pasture grasses. Low-intensity sheep grazing has since been the dominant land use with cattle grazing of minor importance. Hares and rabbits occur throughout. The production of cocksfoot *(Dactylis glomerata* L.) as a seed-crop was also of importance to the area from the late 1800's to the mid 1900's (Coulson, 1979).

Present-day cover of the Port Hills is predominantly modified *Poa cita* short-tussock grassland with *Festuca novae-zelandiae* a minor component in some areas. Other common indigenous species are *Acaena anserinifolia, Dichondra repens, Elymus rectisetus, Helichrysum filicaule, Hydrocotyle novae-zealandiae, Rytidosperma* species and *Wahlenbergia gracilis*. The inter-tussock matrix is dominated by adventive species, particularly cocks foot and *Anthoxanthum odoratum* L., with *Holcus lanatus* L., *Hypochoeris radicata* L. and to a lesser extent, *Agrostis capillaris* L. throughout.

Methods

Seven pairs of sites (Fig. 1) were selected on the basis of having adjacent grazed and ungrazed areas of similar slope, aspect and altitude. Because of the requirement for ungrazed sites to have been consistently without stock, site selection was limited and often coincided with areas within the Summit Road suite of reserves (Kelly, 1972). Site pairs were situated across a fenceline within twenty metres of each other for all but two sites, where the lack of a grazed/ungrazed pair of sites separated by a fenceline necessitated the use of adjacent grazed and ungrazed ridges.

The sites ranged in altitude from 160 m to 546 m. Aspects were from south-west to north-east, but predominantly northerly. All grazed areas were set stocked with sheep, with stocking rates decreasing from 1.4 to 0.6 sheep per hectare towards the lower rainfall end of the Port Hills. All ungrazed areas were formerly grazed by sheep, but were fenced at different times, ranging from about four to more than 24 years ago. Grazing by rabbits and hares was assumed to be continuous throughout.

Sampling technique

Field work was carried out between late February and April, 1988. Species composition at each site was sampled by means of 4.5 m by 0.5 m belt transects. These were located using random numbers to select a fence-post and to determine the number of metres from the fenceline to the transect origin. No transect was located closer than two metres to a fenceline, shrub-edge or another transect. The number of transects at a site was consistent within the grazed/ungrazed pair but differed between pairs due to differences in available sampling area. The area of ungrazed grassland was usually the limiting factor.

Percentage cover was estimated by eye for every species in five alternate 0.5 m by 0.5 m plots along each transect. This was done for a total of 260 plots in 52 transects. Soil depth to the 'C' horizon was measured at three random points along each transect to give a mean value for the transect. Slope and aspect were measured for each transect.

Soil samples were collected from all sites on the same day and analysed for percent moisture content (by weighing, oven-drying overnight at 110°C, then re-weighing) and percent organic matter content (by weighing, igniting at 500°C overnight, then reweighing). The pH of a slurry of equal volumes of soil and distilled water was measured with a standardised Solstat FET pH meter (EPM-310) and a sealed combination electrode with silver/silver-chloride reference.

At each site, tussock density and size were measured by extending a randomly selected transect to a 4.5 m by 4.5 m plot, counting the number of discrete clumps in the plot and recording average extended leaf length (assessed visually) and basal diameter. For irregularly shaped clumps, two measures of basal diameter were taken and the mean value used.

Analysis techniques

Indirect ordination of the transects, utilising species cover values, was performed using detrended correspondence analysis (Hill and Gauch, 1980) as implemented by DECORANA (Hill, 1979). This technique produces abstract axes summarising floristic change which mayor may not correspond to environmental gradients. Correlation analysis using Pearson product-moment correlation coefficients was used to compare the ordination axes with measured variables.

The physical comparability of grazed and ungrazed sites was tested by means of one-way Analysis of Variance using the environmental factors measured. The total number of species per transect was compared for grazed and ungrazed transects using one-way Analysis of Variance. Mean indigenous and mean adventive percentage cover (which were arcsin square-root transformed before analysis to improve normality) were compared for grazed and ungrazed transects using paired t-tests. Mean tussock height and diameter were also compared for grazed and ungrazed sites using paired t-tests. Tussock density was compared between grazed and ungrazed sites using one-way Analysis of Variance.

Results

Tests for physical comparability show, for all environmental factors measured, no significant difference between grazed and ungrazed sites (Table 1).

When the sites were initially ordinated one transect, Kennedys Bush (ungrazed) transect 2, proved to be responsible for three-quarters of the spread

Table 1: *Comparison of environmental factors for grazed and ungrazed short tussock grassland, using one-way ANOVA. Soil % moisture and organic content measurements were log transformed before analysis. Significance levels: ***p<0.001; *p<0.05; NS=not significant.*

	Grazed mean (SD)	Ungrazed mean (SD)	ANOVA F
soil depth	17.30	15.92	0.52^{ns}
(cm)	(6.96)	(6.78)	
soil moisture	33.63	38.95	0.68^{ns}
(0/0)	(9.01)	(11.93)	
soil organic	17.80	16.47	0.00^{ns}
content	(9.13)	(5.70)	
soil pH	5.000	5.100	0.28^{ns}
	(0.283)	(0.132)	
slope	8.519	8.000	0.13^{ns}
(degrees)	(5.393)	(5.026)	
aspect	46.54	51.81	0.15^{ns}
(degrees from N)	(49.87)	(46.88)	

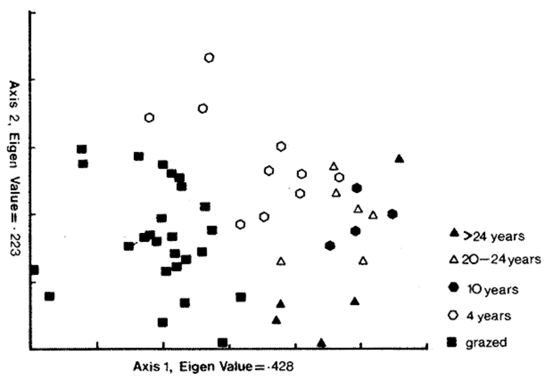

Figure 2: *The first two axes from a DCA ordination of transects using species cover values. showing ungrazed (with number of years ungrazed) and grazed transects.*

along axis 1. This transect was composed almost entirely of bracken *(Pteridium esculentum)* of which no other transect had more than 2% cover abundance. The ordination was recalculated omitting this transect and the resulting spread overlaid with grazed and ungrazed designations (Fig. 2).

The first axis of this ordination (axis 1) separated grazed and ungrazed transects (Fig. 2). Correlation analysis showed a significant relationship between axis 1 and both the abundance of cocksfoot and the number of years a site had been without stock ($r = 0.923$, $p<0.001$; $r = 0.763$, $p<0.001$ respectively). Cocksfoot abundance and number of years without stock were themselves highly correlated ($r = 0.868$, $p<0.001$). The abundance of *Rytidosperma* species showed a significant negative correlation with axis 1 ($r = -0.615$, $p<0.01$). No other species or environmental factor was significantly correlated with the first axis.

Altitude and the abundance of *Poa cita* were negatively correlated with the second ordination axis (axis 2) ($r=0.535$, $p<0.05$; $r= -0.639$, $p<0.01$ respectively) and also significantly correlated with each other ($r=0.585$, $p<0.05$). The abundance of *Holcus lanatus* showed a significant positive correlation with axis 2 ($r = 0.597$, $p<0.01$). *Poa cita* abundance was also negatively correlated with cocks foot abundance ($r= -0.65$, $p<0.05$).

A total of 63 species were identified at the sites surveyed, of which 23 were indigenous. However, because some indigenous taxa were not distinguished beyond the generic level (e.g., *Rytidosperma* species). the data-set underestimated indigenous species richness. Despite this, the number of both indigenous and adventive species was significantly greater in grazed transects (Table 2). Grazed transects also had a higher mean percentage cover of indigenous species and a lower mean percentage cover of adventive species than ungrazed transects (Table 3). Tussock density was significantly higher at grazed sites (Table 4); however, the tussocks were significantly shorter (Table 5). There was no significant difference in tussock diameters between grazed and ungrazed sites (Table 5).

Table 2: *Comparison of the number of adventive and indigenous species per transect, for grazed and ungrazed short tussock grassland, using one-way ANOVA. Significance levels as for Table* 1.

	Grazed mean (SD)	Ungrazed mean (SD)	ANOVA F
no. adventive species	10.69 (2.346)	7.231 (2.160)	30.64***
no. indigenous species	5.731 (2.325)	2.615 (1.699)	30.44***

Table 3: *Comparison of mean indigenous and mean adventive percentage cover per transect, for grazed and ungrazed short tussock grassland, using paired t-tests. Cover values were arcsin square-root transformed before analysis. Significance levels as for Table* 1.

	Grazed mean (SD)	Ungrazed mean (SD)	T
indigenous cover (%)	32.58 (14.67)	13.48 (14.87)	-5.28***
adventive cover (%)	56.97 (17.91)	84.05 (16.30)	6.63***

Table 4: *Comparison of tussock density in 4.5 m x 4.5 m plots, for grazed and ungrazed short tussock grassland, using one-way ANOVA. Significance levels as for Table 1.*

	Grazed mean (SD)	Ungrazed mean (SD)	ANOVA F
tussock density	32.43 (17.39)	9.857 (11.57)	8.18.

Table 5: *Comparison of mean tussock diameter and height in 4.5 m x 4.5 m plots, for grazed and ungrazed short tussock grassland, using paired t-tests. Significance levels as for Table 1.*

	Grazed mean (SD)	Ungrazed mean (SD)	T
mean tussock diameter (cm)	14.11 (3.609)	13.62 (4.864)	0.58^{ns}
mean tussock height (cm)	54.95 (3.898)	68.43 (11.4)	- 3.52*

Discussion

The differences in vegetation at paired sites with and without grazing show that the cessation of grazing has an appreciable effect on both the number of species present and floristic composition in this grassland. The trend of fewer adventive species (Table 2) but higher adventive cover (Table 3) in ungrazed sites indicates the dominance of just a few species when grazing pressure is removed. This parallels findings in British chalk grassland where selective control by grazing of more vigorous species is important for the maintenance of species-richness (Grubb, 1986; Gibson *et al.,* 1987).

The increase in cocksfoot following the removal of grazing appears to be a major influence in the changes that have occurred in the Port Hills sites. Cocksfoot is a tall, vigorous, adventive grass with a high relative growth rate compared with other grasses, both indigenous and adventive (Scott, 1970). It is also known to be preferred by sheep (Hughes, 1975). As a result, it commonly increases when protected from grazing, as has been found in an earlier Port Hills study (Boyce, 1939), in the Hunter Hills (Barker, 1953), on Mana Island (Timmins *et al.,* 1987), and also in British studies (Gibson *et al.,* 1987).

Although correlation by no means implies causality, the significant relationships between cocksfoot abundance, the time a site had been without stock, and the first ordination axis summarising floristic similarity, indicate that changes in vegetation composition subsequent to the cessation of grazing may be, at least in part, a result of the increasing dominance of cocksfoot.

The hypothesis of this study was that the cessation of grazing has no effect on the number or abundance of indigenous species in strongly modified grassland. However, the results presented here indicate a decrease in both the number and abundance of indigenous species following protection from grazing. As many of the indigenous forbs in Port Hills grassland are low-growing (e.g., *Dichondra repens, Hydrocotyle novae-zelandiae, Leptinella* species, *Nertera ciliata),* the decrease could well be due to competitive exclusion by vigorous adventive grasses in a rank ungrazed sward, as suggested by Wraight (1964). However, grazing may be detrimental for relatively palatable species such as *Dichelachne crinita* and *Elymus rectisetus*. It was observed during this study that these species occurred almost exclusively within tussock clumps, presumably protected to a certain extent from grazing, as suggested by Barker (1953). If feed was scarce these refugia would be unlikely to prevent grazing completely.

Tussocks also appear to have been adversely affected by the removal of grazing pressure. Tussock density was markedly lower in ungrazed sites (Table 4), probably because of suppression in a rank ungrazed sward as indicated by the negative correlation between the abundance of *P. cita* and cocks foot. The suppression of indigenous tussock species by ungrazed cocksfoot and browntop *(Agrostis capillaris)* has also been noted on the Hunter Hills by Barker (1953). The difference between grazed and ungrazed sites in mean tussock height is most probably due to grazing of tussocks when feed is scarce, indicating the need to keep grazing intensity at a low level where it is used as a management tool. A detrimental effect of heavy grazing on *F. novae-zelandiae* tussocks has also been reported by O'Connor (1966).

In all the sites surveyed, no obvious seedlings of either *P. cita* or *F. novae-zelandiae* were encountered. In ungrazed areas the density and depth of the sward could prevent seedling establishment but in grazed areas the apparent lack of regeneration could be due to selective elimination by sheep (Boyce, 1939). Because tussocks are relatively long-lived individuals, a failure to regenerate from seed could easily go unnoticed. However, for the long-term survival of tussock grassland, seedling recruitment is necessary. If grazing does prevent seedlings from becoming established, it may be a useful management option to remove stock from all or part of the area for a few years.

The effect of protection from grazing on different types of short tussock grassland may be seen as a function of both the physical environment and degree of modification of the site. When the apparently conflicting results of studies in New Zealand grassland that have involved the effect of grazing are examined in the light of these two factors, some general trends are obvious.

In unmodified or virtually unmodified grassland, protection from grazing is likely to result in an improvement in indigenous species abundance, as is the case in alpine grasslands (Evans, 1980; Rose & Platt, 1987). The scarcity of short tussock grassland that has not been modified to some extent may explain why this result is seldom reported (Malcolm, 1925 is an exception).

In slightly- to moderately-modified short tussock grassland where some environmental factor limits growth, there may be little or no change in species composition (e.g., Meurk *et al.,* 1989). However, in areas where growth is not severely limited, the effect of protection from grazing appears to depend on the initial condition of the vegetation. In depleted grassland or where invasive adventive species such as *Hieracium pilosella* L. are present, indigenous species tend to decrease in abundance, while showing a more positive response in areas with a higher tussock cover (Rose, 1983; Scott *et al.,* 1988).

In comparison with other areas of short tussock grassland, Port Hills grassland is strongly modified and is also on moderately fertile soils. In this grassland, protection from grazing appears to result in adventive species dominating the sward at the expense of low-growing or less vigorous indigenous species. This does not necessarily mean that Port Hills grassland is at floristic equilibrium under the present regime of low-intensity sheep grazing. In fact, early botanical records (Laing, 1918; Wall, 1922) indicate the possibility that some indigenous species have been lost from the area. However, in the short term, when facing the question of whether to graze or not to graze strongly modified short tussock grassland, low-intensity grazing appears to be the best option.

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