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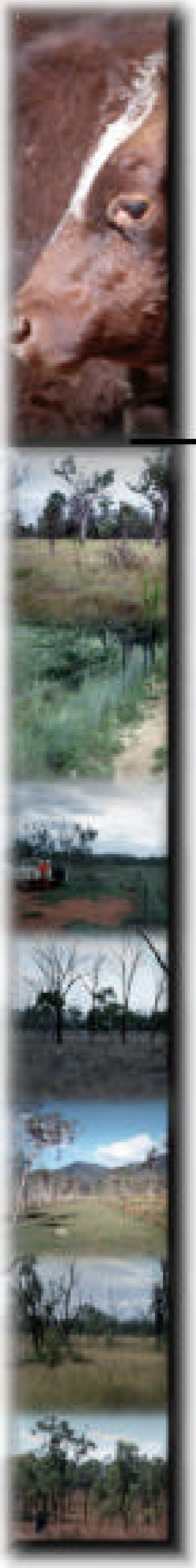

THE EFFECTS OF CATTLE GRAZING ON VEGETATION DIVERSITY AND STRUCTURAL CHARACTERISTICS IN THE SEMI-ARID RANGELANDS OF NORTH QUEENSLAND

THESIS SUBMITTED BY
GREGOR ALAN CALVERT BSc (Hons) JCU
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James Cook University

“IT IS NOT THE CRITIC WHO COUNTS, NOR THE MAN WHO POINTS OUT HOW THE STRONG MAN STUMBLED, OR WHERE THE DOER OF DEEDS COULD HAVE DONE BETTER. THE CREDIT BELONGS TO THE MAN WHO IS ACTUALLY IN THE ARENA; WHOSE FACE IS MARRED BY DUST AND SWEAT AND BLOOD; WHO STRIVES VALIANTLY; WHO ERRS AND COMES SHORT AGAIN AND AGAIN; WHO KNOWS GREAT ENTHUSIASMS, GREAT DEVOTIONS, WHO SPENDS HIMSELF IN A WORTHY CAUSE; WHO, AT THE BEST, KNOWS IN THE END THE TRIUMPH OF HIGH ACHIEVEMENT; AND WHO, AT THE WORST; IF HE FAILS AT LEAST FAILS WHILE DARING GREATLY, SO THAT HIS PLACE SHALL NEVER BE WITH THOSE COLD AND TIMID SOULS WHO KNOW NEITHER VICTORY NOR DEFEAT”

- Attributed to Theodore Roosevelt

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ABSTRACT

The semi-arid rangelands of tropical North Queensland have suffered a major decline in land condition since arrival of Europeans. This includes erosion and soil loss, the widespread loss of native perennial tussock grasses and the widespread invasion of exotic plant species; some accidental, others introduced to help stem the process of land degradation. It has often been stated or implied that cattle grazing is an important factor in the land degradation process; a suggestion supported by various research projects. The present research examined impacts of grazing on various characteristics of plant communities in the semi-arid rangelands of tropical north Queensland. Characteristics examined included diversity, functional groups, ground cover and tree dieback.

The diversity and composition of a pasture is usually determined by abiotic effects such as soil and climate, and secondarily by the nature of grazing. However, the present study demonstrated that, in some cases, grazing played an equally important role in determining species assemblages. Grazing generally resulted in a decline in the abundance of:

- native perennial tussock grasses
- exotic pasture legumes, and
- palatable species

Grazing caused an increase in:

- exotic grasses
- forbs
- native legumes, and
- unpalatable species.

Impacts of grazing on diversity were dependant on the dominant grass species and its palatability. When native palatable and perennial grasses such as kangaroo grass (*Themeda triandra*) and black spear grass (*Heteropogon contortus*) dominated, intermediate levels of grazing resulted in an increase in diversity since the grazing released other plant species from competition. Where the dominant grass was a less-palatable exotic grass species such as Indian couch (*Bothriochloa pertusa)* or buffel grass (*Cenchrus ciliaris*), grazing reduced diversity since grazing reinforced the dominance of those grass species. *Cenchrus ciliaris* itself was identified as having a deleterious effect on species diversity, independent of grazing pressure.

The responses of functional groups such as annual grasses were dependant on levels of palatability, which may have varied from site to site. The effects of cattle grazing on many functional groups were less pronounced in areas grazed only during the dry season, in contrast to areas grazed continuously throughout the year.

Grazing had a deleterious influence on ground cover. While intermediate grazing caused a level of decline in ground cover that was beneficial to many species, heavy grazing may result in scalding and erosion. An exception to this is where grazing reinforced the dominance of *Bothriochloa pertusa*; the spreading stoloniferous habit of which can result in increased ground cover.

During this research, widespread dieback of ironbarks (*Eucalyptus crebra* sensu lat.) was observed throughout the semi-arid rangelands on a range of soil types and grazing regimes. In contrast to previous research, the present study found a correlation between cattle grazing and the dieback of *Eucalyptus crebra*, although dieback occurred to some degree even in the absence of grazing. Large trees were more susceptible to dieback than small saplings, which, in some cases, may have benefited from grazing by the removal of competing herbaceous species from their proximity.

The present research showed that with prolonged heavy grazing, transitions in states of land condition might occur which would be irreversible without major inputs. It was recommended that achieving sustainability of the grazing industry in both economic and conservation terms would involve the regular monitoring of several land condition parameters. This would identify economically feasible opportunities for pasture rehabilitation from opportunistic de-stocking or changing the seasons of cattle grazing. The present study noted that diversity and land condition were optimal under a regime of intermediate disturbance, and that this level of disturbance occurred with macropod grazing. Likewise, the provision of cattle exclosures adjacent to pastures allowed a source of seed for recolonisation of native perennial tussock grasses where those species had been otherwise eliminated by the excessive overuse of grazing.

Limitations in this study were discussed and recommendations for future research priorities were made.

ACKNOWLEDGEMENTS

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Now I can get a life!

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GLOSSARY OF ABBREVIATIONS AND ACRONYMS

- *: Introduced species
- ANOVA: Analysis of Variance
- BLURNG: Blue Range study site
- CSIRO: Commonwealth Scientific Industry Research Organisation
- DBH: Diameter at Breast Height
- EPP1: Epping Forest National Park study site #1
- EPP2: Epping Forest National Park study site #2
- JCU: James Cook University
- JERVOIS: Jervoise study site
- KHILEA: Kangaroo Hills East study site
- KHILWT: Kangaroo Hills West study site
- KIRK: Kirk River study site
- LCKDAM: Lucky Downs dam study site
- LYSHN: Leyshon View study site
- MTLEY: Mt Leyshon mine study site
- PAJING: Pajingo mine study site
- QDPI: Queensland Department of Primary Industries
- RSHTN: Rishton mine study site
- TABTOP: Table Top study site

CHAPTER 1: GENERAL INTRODUCTION

1.1. AIMS AND SCOPE OF THESIS

Tropical savannas cover more than 80% of northern Australia (Stewart 1996), encompassing most of the coastal and sub-coastal lands of northern and north-eastern Australia north of the Tropic of Capricorn (Holmes 1996). Mott & Tothill (1994) describe these savannas as sharing the common characteristic of having a *Eucalyptus* dominated overstorey with a tall tropical grass understorey and an average rainfall over 650mm/yr, most of which falls in the summer months.

Grazing by beef cattle has been the predominant land use on these savannas for over a century. This has expanded to the extent that, by 1993, the cattle grazing industry had grown to include 5.5 million head of cattle on the Queensland tropical savannas (Stewart 1996). Unfortunately, pressure from overgrazing and poor land management has caused considerable land degradation over the last 50 years. These degraded rangelands exhibit symptoms such as pasture deterioration, soil loss, degradation of soil structure, tree loss and weed invasion (Mott & Tothill 1994). Such degradation not only reduces the productive capability of the land, but also usually requires high inputs to arrest or prevent further degradation (Williams *et al.* 1993).

Quirk *et al.* (1997) observed that cattle properties with low economic performance tended to be those with greater levels of land degradation. However, attitudes towards land management are changing. Producers are more aware than ever of the need to monitor production in terms of sustainability with the need for ecologically sustainable development becoming one of the major concerns of the beef grazing industry (Stewart 1996). The term ‘sustainable use’ is often poorly understood, but was defined by Williams *et al.* (1993) as being “...whether the current or changed land uses to which land is subjected can be maintained”. In North America, research assisting in the rehabilitation and management of degraded rangelands includes evaluating the roles of herbivores on the structure and function of those rangelands (Ryerson & Parmenter 2001). However, one of the difficulties impeding an understanding of herbivore and land management impacts is the large variation of response exhibited by different areas of the savanna zone (Ryerson & Parmenter 2001, Mott & Tothill 1994). Trying to extrapolate results from one study site to another often results in confusion since responses of plant communities to herbivory vary according to differences in soil, climate, evolutionary history and the composition of those plant communities (Ryserson & Parmenter 2001).

Although research has played an important part in improving productivity of the grazing industry (Stewart 1996), a considerable amount of this research has focussed on obtaining an enhanced knowledge of natural systems and their responses to human intervention (Holmes 1996). From an examination of the literature, it appears that large amounts of research have been focussed on improving pasture productivity through plant introductions or on enhancing the productivity of a small number of native perennial grass species of greatest economic importance to the grazing industry. Although a decline in land condition has raised serious concerns over the loss of diversity in rangeland flora and fauna, virtually no research has focussed on grazing impacts on diversity as a whole in these tropical savannas (Quirk *et al.* 1997). Hill (1973) notes that diversity is important conceptually since it can be related to stability, maturity, productivity, evolutionary time, predation pressure and spatial heterogeneity. Although there have been studies from temperate regions examining the need to identify species useful as indicators of pasture degradation, these principles have not been adequately applied to the semi-arid tropical savannas of North Queensland. Holmes (1996) expects to see more research focussing on the protection of biodiversity and of target species and habitats.

The present research addresses a number of issues. Firstly, by understanding the changes that occur to savanna pastures grazed by cattle, pasture management techniques that enhance pasture composition for increased productivity and commercial gain can be recommended. Secondly, there is a need to understand the conservation implications of cattle grazing on plant species diversity and thereby assist in striving towards an environmentally sustainable industry. Finally, an understanding of the changes that cattle grazing causes to plant communities helps to determine the complex ecological process involved in savanna dynamics and to test the applicability of various ecological theories and models as proposed by various authors.

The present research concentrated on the impacts that grazing regimes have on plant species diversity and plant community structure and composition.

The aim of the present research was to answer the primary question:

- What was the importance of grazing in influencing patterns in plant species diversity, compared to the background landscape heterogeneity, and are there predictable patterns in community changes with increased grazing?

This question may be answered by breaking it down into several sub-sections:

- What was the effect of grazing on dominant grass species?
- How did grazing alter the dominance of a particular species?
- What was the effect of grazing on plant species diversity, richness and evenness?
- What was the variation in response to grazing by different functional groups?
- How did this variation alter plant community composition under varying grazing regimes?
- What relationship existed between grazing and ground cover?
- What influence did grazing have on dieback of the ironbark *Eucalyptus crebra*?

The present research aimed to address these questions using appropriate statistical methods to examine the original data, and to compare these results with the published literature.

1.2 THESIS LAYOUT AND DESIGN

In the present research, the second chapter examined literature relevant to understanding the historical background of the study area in relation to the grazing and degradation of plant communities, and then explored theories relating to the function and importance of diversity to ecosystem function. An appraisal was made of previous research using grazing exclosures and other methods to elucidate the changes that grazing imposes on a plant community in contrast to the normal level of community heterogeneity imposed by the variability of the landscape itself.

In Chapter 3, the methodology used for data collection in this research was explained, incorporating a map of the sites and brief explanations of statistical procedures. An overview of the statistical methods employed to analyse the data was also presented.

The sites were initially examined in Chapter 4 by looking at the influence that soil type and site location had on community composition. Domination of the sites by a select number of grass species was also examined to allow results from following chapters to be examined in light of those grass domination results.

Chapter 5 explored the question of the impact of grazing on plant diversity, richness and evenness, and examined various measures and indices that have been used to describe these concepts. These results were examined in light of the results from Chapter 4 and relevant literature that explain how such changes may occur.

Chapter 6 investigated how species can be separated into functional groups using taxonomy, naturalisation status (native or introduced), gross morphology and life history strategy. Differences in abundance of these functional groups between grazing treatments were examined to identify trends in community composition that were not obvious when dealing either with individual species or diversity as a whole. There has been a considerable amount of literature published on selecting these functional groups and how those groups might be expected to respond to disturbance. Much of this literature was examined in light of the data presented.

Examining attributes of rangelands other than floristic composition can also assist in predicting or indicating degradation. Chapter 7 examined the effects of grazing on pasture degradation in terms of ground cover and its potential for soil erosion. The relevant literature was examined to explain these results.

Tree dieback was widespread throughout the Dalrymple and Belyando shires at the time of this research, most of which has been attributed to drought. In Chapter 8, the potential link between grazing and tree dieback is examined. The results of the data analysis are presented in conjunction with literature examining the possible causality of such a relationship.

Various authors have used ‘State and Transition’ models to organise and present existing knowledge. In Chapter 9, these models were used to allow a synthesis of the results and conclusions of the various chapters into a broad overview, summarising the results and testing for the applicability of State and Transition models.

Finally, in Chapter 10, the results of the present research were summarised and suggestions were made for future pasture management in the context of the results presented. Recommendations and suggestions for future research were given.

An appendix is included at the end of this thesis, including detailed descriptions of the sites examined, data collection pro-formas and a complete list of the plant species and functional groups that were identified in this research.

CHAPTER 2: SPECIES DIVERSITY IN TROPICAL RANGELANDS

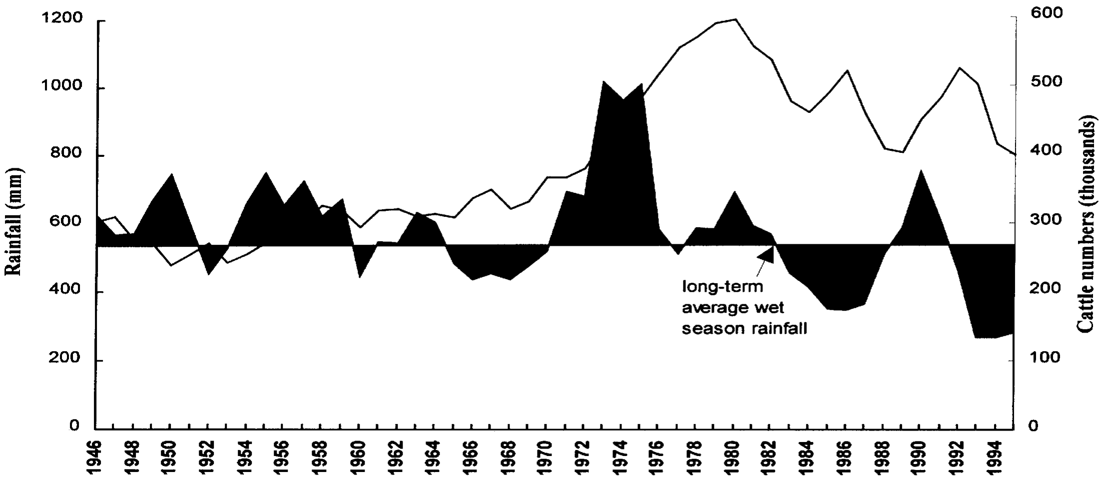
2.1 CHANGES TO RANGELAND DIVERSITY

2.1.1 Features and history of grazing in Australian tropical rangelands

Grazing by domesticated stock is the major land use over 60% of Australia. In 1990, these lands supported a grazing community consisting of 50 million sheep, 15 million cattle, 20 million kangaroos and over 100 million rabbits (Wilson 1990). The North Queensland savannas in the Burdekin River catchment have been grazed since the early 1860s (Landsberg *et al.* 1998), however, cattle impacts on the land were relatively low for the first 100 years (Gardener *et al.* 1990). This limited impact may have been due to poor quality feed, poor breeding performance and high stock mortality during drought years (Gardener *et al.* 1990). The drought-induced mortality was exacerbated by poorly adapted british breeds of cattle (Landsberg *et al.* 1998) and low accessibility to watering points (Gardener *et al.* 1990), while during wetter periods, high numbers of parasitic cattle ticks also kept cattle numbers low (Quirk *et al.* 1997).

The modern grazing industry has circumvented many of these limitations. Artesian water was discovered in 1880, which led to increased numbers of watering points and therefore greater access of stock to pastures (Landsberg *et al.* 1997a). Graziers also increase the carrying capacity of their land with ‘improved pastures’ using introduced grass and legume species. Walker & Weston (1990) estimate that by 1990, 9.4 million hectares of pastoral land had either been sown with or colonised by introduced improved pasture species. Graziers have also increased livestock survival with the use of feed supplements and through the introduction of hardier Zebu cattle (*Bos indicus*) (Gardener *et al*. 1990). Feed supplements allow greater utilisation of pastures by cattle and greater survival rates of stock during droughts, thereby increasing the possibility of overgrazing-related land degradation (O’Reagain P. pers. comm.). By 1983, over 90% of cattle in North Queensland had Zebu genes, since those genes have allowed enhanced animal production, greater resistance to heat stress and cattle ticks, increased foraging ability and a consequently higher forage intake (Gardener *et al.* 1990). In addition, nutritious and persistent legumes such as *Stylosanthes hamata*, *S. scabra* and *S. humilis* were introduced to grasslands to promote higher pasture yields amd higher animal production (Jones *et al*. 1997, McIvor & Gardener 1998). The cattle population of the Dalrymple Shire increased significantly in the 1970s in response to a crash in beef prices which saw producers keeping cattle rather than selling them at a loss (Landsberg *et al.* 1998, Quirk *et al.* 1997) Cattle numbers also increased with above average rainfall, development of stock waters and pasture improvement with the exotic legume *Stylosanthes humilis* (Quirk *et al.* 1997). This enhanced cattle survival, growth and retention saw cattle numbers in North Queensland double in the years between 1961 and 1978 from 300,000 to 600,000 head, resulting in severe overgrazing throughout the region (Gardener *et al.* 1990). Despite subsequent droughts and low rainfall periods since then (Quirk *et al.* 1997) and a widespread disease-induced collapse of *Stylosanthes humilis* (Walker & Weston 1990), cattle numbers remained relatively high in the region and are only gradually declining (Quirk *et al.* 1997). Even as late as 1997, many herd sizes in the region were still well in excess of carrying capacity of the land (Quirk *et al.* 1997). Figure 2.1 below illustrates changes in cattle numbers in the Dalrymple Shire in relation to fluctuating wet season rainfall.

Figure 2.1: Wet season rainfall for Charters Towers (relative to long term average) and fluctuating cattle numbers in Dalrymple Shire from 1946 to 1995.
(reproduced with permission Quirk *et al.* 1997)



While there is a general belief amongst many graziers that pastures “bounce back” following these severe periods of drought and overstocking, Landsberg *et al* (1998) observe that the botanical composition does not fully recover and that the resultant decline of perennial tussock grasses has a significant impact on pasture productivity.

2.1.2. Cattle-related land degradation

(Gardener *et al.* 1990) interpreted land degradation in the Burdekin catchment as manifesting itself in five ways:

- Increased invasion by shrubs, both introduced and native species
- Reduction in vigour in grasses
- Reduction of vegetative cover resulting in increased run-off (high loss of soil & nutrients)
- Gully erosion with increased sediment transport along waterways
- Change in species composition (especially more *Bothriochloa pertusa*)

A number of these effects have been verified. Quirk *et al.* (1997) state that the observed increase in woody plant density is related to reduced fire frequency, which in turn is a consequence of a grazing-related reduction in fuel loads. In relation to erosion, Woods (1983) defines 'seriously degraded' lands as those suffering soil loss in excess of 20t/ha/year and, using that definition, notes that 43.2 million hectares (13%) of Australia's rangelands are seriously degraded. In the Dalrymple Shire, a third of the shire has severe soil erosion, with scald and sheet erosion and exotic weed invasion also common (DeCorte *et al.* 1994). Gardener *et al.* (1990) estimated that the amount of soil lost from the Burdekin River catchment exceeded 20 million tonnes per year, equivalent to a cost of $90 million in replacing nutrients with fertilisers. This may have increased since the 1992-1994 drought.

Changes in species composition include a decline in abundance of desirable perennial grasses as a common feature of declining pasture condition (Ash & Corfield (1998), Quirk *et al.* (1997), McIvor & Gardener (1995), Mwalyosi (1992), Shaw & 't Mannetje (1970)). Walker & Weston (1990) report that, in 1978, 29.1 million hectares of Queensland's pastoral lands had been affected by vegetation degradation, however, the magnitude of this figure may differ from the opinion of others, as Walker & Weston (1990) do not consider invasion by exotic palatable plants as degradation. In regards to abundance of important grass species, 21% of the Dalrymple Shire was poor, 26% fair and 53% good (DeCorte *et al.* 1994), however, this gives no real understanding of the impact of grazing on numers of plant species overall. Models described by Quirk *et al.* (1997) showed fluctuating abundances of perennial grasses in the Dalrymple Shire up until the 1990s when catastrophic losses occurred, leading to the local extinction of preferred species in some areas. It was noted that the decline in abundance of those grasses has led to increases in less palatable species and/or invasion by exotic grasses (Quirk *et al.* 1997).

2.1.3. Historical changes in species composition

There has been a documented transition in the dominant grass species in the northern rangelands. Kangaroo grass (*Themeda triandra*) was far more prominent prior to European colonisation, while black speargrass (*Heteropogon contortus*) has become increasingly prevalent due to altered burning and grazing regimes (McIvor & Scanlan 1994). Mott & Andrew (1985) record *Heteropogon contortus* increasing in establishment 80 fold after fire and that fire also accelerates its development. Lacey *et al.* (1982) also note that fire benefits *Heteropogon contortus* over *Themeda triandra*, but add that this only seems to be the case in conjunction with grazing. Isbell (1969) believes that due to the frequency of fire prior to the arrival of Europeans, it is doubtful that fire alone has caused the floristic shift from *Themeda triandra* to *Heteropogon contortus*. In comparison to *Themeda triandra*, *Heteropogon contortus* has a higher photosynthetic rate, higher specific leaf area and greater leaf area remaining after defoliation, giving it a competitive advantage following heavy grazing (Grice & McIntyre 1995). It has been noted that grazing is of great importance to the spread of *Heteropogon contortus* and that more palatable species such as *Themeda triandra* are preferentially grazed by cattle, whilst *Heteropogon contortus* is, by comparison, relatively ignored (Isbell 1969). Although Tothill & Gillies (1992) see the transition from *Themeda* to *Heteropogon* as being an acceptable change from a pasture productivity viewpoint, it is obviously a symptom of changing ecological processes.

The introduction of Indian couch (*Bothriochloa pertusa* – formerly *Andropogon pertusus*) into some areas of the northern rangelands followed the transition from *Themeda triandra* to *Heteropogon contortus*. Maiden (1889) noted that Indian Couch was present in Queensland in the 19th century, though Partridge (1995) suggests that current populations spread from a possible wartime import into the Bowen area of North Queensland. Walker & Weston (1990) also note that it is an important species in deliberately sown introduced pastures. While Quirk *et al.* (1997) note that *Bothriochloa pertusa* has become the dominant grass on granodiorite-derived soils in the Dalrymple Shire, its potential for spread may ultimately be confined to soils of high fertility (O'Reagain P. pers. comm.). The rate of spread throughout Queensland may be indicated by estimates of cover. Walker & Weston (1990) observed the extent of *Bothriochloa pertusa* to be 0.8 million hectares (0.2 million in the Dalrymple Shire), while Partridge (1995) states that *Bothriochloa pertusa* had increased to more than one million hectares, having taken over largely from the native perennial *Heteropogon contortus*. *Bothriochloa pertusa* is an aggressive coloniser, which spreads rapidly into bare areas where native grasses have been grazed out (Rolfe *et al.* 1997). Its ability to invade *Heteropogon* pastures is due to lower palatability, higher tolerance to grazing, higher seed production, vigorous stoloniferous runners and adaptability to a wide range of soil types and nutrient levels (Partridge 1995). *Bothriochloa pertusa* has also been reported to exhibit allelopathic effects, however, it is unclear as to whether this is chemical or competitive in nature (Hu 1995).

As stated previously, 'pasture improvement' through the introduction of exotic pasture species has been widespread in Queensland, and this has significantly altered the botanical composition of native pasture communities over large areas of Queensland. Lonsdale (1994) reports that between 1947 and 1985, 463 exotic grasses and legume species in at least 2033 accessions were introduced to northern Australia for pasture improvement. Leslie *et al.* (1987) argue that such introductions are necessary since native fodder species are susceptible to defoliation by grazing, have restricted periods of acceptable nutritive value and have seed production features that render them unsuitable for domestication. Lonsdale (1994) notes that very few pasture plant introductions have actually been useful, while many have become weeds and only four species (<1%) have been useful without becoming a weed. Although 70% of improved pastures have been improved with grasses only, there have also been significant releases of legumes, primarily to improve productivity of less fertile soils and to help restore run-down grass pastures (Walker & Weston 1990). Most significant of the introduced legumes have been the several species and cultivars of the genus *Stylosanthes* in the family Fabaceae, particularly Caribbean stylo (*Stylosanthes hamata* cv. Verano) and shrubby stylo (*S. scabra* cv. Seca) (McIvor & Gardener 1998, Jones *et al.* 1997, Walker & Weston 1990). Anecdotal evidence suggests that the Townsville stylos (*S. humilis*) dominated much of Dalrymple Shire following the widespread loss of speargrass in the shire following droughts (O'Reagain P. pers. comm.). It should be noted that the use of *Stylosanthes* has resulted in soil acidification in some areas due to leaching of fixed nitrates from their roots (Jones *et al.* 1997, Noble *et al.* 1997). *Stylosanthes humilis* has now declined significantly in importance since the period 1975-1980, which saw extensive areas of this exotic legume devastated by the fungal anthracnose disease *Colletotrichum gloeosporioides* (Walker & Weston 1990).

Fairfax & Fensham (2000) note that many of the exotic grasses introduced for pasture improvement are highly efficient competitors in some situations with the potential to have a detrimental impact on numbers of native plant species. The introduction of South African buffel grass (*Cenchrus ciliaris*) for pasture improvement has also altered the composition of many northern pastures. Pastoralists favour *Cenchrus ciliaris* because of its palatability, responsiveness to limited rainfall, ability to colonise and its tolerance to drought and heavy grazing (Fairfax & Fensham 2000). Conversely, Landsberg *et al.* (1997a) describe *Cenchrus ciliaris* as a critically invasive species with the potential to displace native herbaceous species and alter fire regimes, causing it to be listed amongst the top environmental weeds in Australia. McIvor (1998) and Fairfax & Fensham (2000) record a decline in numbers of native plant species following the introduction of *Cenchrus ciliaris* into a North Queensland savanna woodland, with Fairfax & Fensham (2000) observing that the spread of *Cenchrus ciliaris* is likely to be having a more detrimental impact on species diversity than tree clearing. Sattler & Williams (1999) list invasion of *Cenchrus ciliaris* as being a threatening process for several 'of concern' regional ecosystem types in Queensland. Walker & Weston (1990) noted that three cultivars of *Cenchrus ciliaris* (Gayndah, American and Biloela) had covered 0.2 million hectares of Queensland.

Every site examined in this research project contained introduced species to some degree. Bridgewater (1990) defines such a mixture of native, naturalised and exotic species as 'synthetic vegetation'. The question posed by Bridgewater (1990) is whether such communities are transitional phases in the degeneration of native plant communities or metastable communities capable of persisting as such even through disturbance cycles. Similarly, Grice & Campbell (2000) ask whether we should consider introduced weeds as a disease causing pasture degradation or as a symptom of other underlying forces.

2.1.4. Research into grazing-related changes to plant species composition

It has long been understood that under certain grazing regimes, there is an increase or invasion of unpalatable species into rangelands, and that taller, more productive grasses are replaced by shorter, less productive species (Dyksterhuis 1958).

Elucidating the effects of cattle grazing on numbers of plant species has been difficult. Landsberg *et al.* (1997b) suggest that this is because there is no single, unified response of plant species to grazing, and that the response of a species is dependant upon characteristics such as habitat requirement and life history attributes (e.g. plant size and shape, position of growing buds and regeneration traits). McIvor (1998) adds that the impact of grazing on a species depends on its palatability and competitive ability. For instance, when a dominant is palatable, grazing will increase species numbers through removal of competitive exclusion, but when a dominant is unpalatable, then grazing reinforces their dominance (McIvor 1998). Wilson (1990) recognises this pattern when he observes that, following heavy over-grazing, the only perennial species remaining are unpalatable.

Ingestion of plant material is only one aspect of the impact of cattle since Bryant *et al.* (1972) and McIvor *et al.* (1995b) note that mowing and clipping have different impacts to grazing on grass communities. Although Brown & Evans (1973) suggest that the mechanism causing a loss of species richness in grazed pastures is due to the differing sensitivities of the plants themselves to treading, other authors believe the relationship is soil related. Savory & Parsons (1980) suggest that hooves improve soil hydrological properties by breaking up crusts and increasing infiltration. However, Weigel *et al.* (1990) and McIvor *et al.* (1995a) observed that heavy grazing reduced soil infiltration rate and porosity, increased soil compaction and soil strength and increased sediment production and runoff. Animal treading also increased bulk density and surface roughness, and decreased aggregate stability, soil penetrability and herbage production (Mullen *et al.* 1974). Bryant *et al.* (1972) explained their observed differences in plant growth by the impact of cattle trampling on soil compaction, partially attributed to the resistance of the soil to root growth. Supporting this assumption is research that records high mechanical strength of soil as being a factor in the persistence of bare scalds in degraded rangelands (Williams *et al.* 1993). In addition to these physical impacts on soil structure, abundance of vesicular-arbuscular mycorrhizal fungi is greatly reduced in overgrazed pastures, and this in turn reduces the growth rate of perennial grass species because of reduced uptake of phosphorous (Ash & McIvor 1995).

Rather than looking at the persistence or disappearance of individual species under grazing, some authors have tried to group species into functional groups; a concept that will be explored in greater detail in Chapter 6. Dyksterhuis (1949) found that certain groups of plants could be defined by the way they responded to grazing, and he recorded which species decreased, increased or invaded under grazing in a variety of sites. Although McIvor (1998) advises caution about the use of these grazing-response groups due to the wide variation in response between different sites, those broad groups are still in wide use. (The pros and cons of various attempts to cluster species into such functional groups will be expanded on further in Chapter 6). In a research program detailed in Landsberg *et al.* (1997b), plants and animals identified along a grazing gradient showed that 10-33% were increasers, 15-38% were decreasers and other species were neutral. From the same research, James *et al.* (1996) noted that 20% of species were only found in areas so far from water that they could be regarded as ungrazed. This suggests that 20% of species are intolerant of and threatened by grazing, and are dependant for their continued survival on having areas free of grazing (James *et al.* 1996). There is no mention in the literature of precise figures of loss of biodiversity from our rangelands, although Leigh & Briggs (1992) report that 0.2% of plants that have recently become extinct have become so as a consequence of grazing.

The history of a site being examined is of vital importance, since the longer that an area has been grazed, the more likely it is that remaining plants will be those that are adapted to such disturbances (Landsberg *et al.* 1997b). In experiments conducted at Hillgrove and Cardigan Stations near Charters Towers, species richness was highest on heavily grazed sites (McIvor 1998). This was due to the fact that these areas have had a long grazing history and that most of the species recorded were increaser species (McIvor 1998). It is worthwhile noting these results used quadrats on transect lines with no timed searches, which would have underestimated species richness, especially amongst rarer decreaser species. In some cases, species richness would not be a good indicator of biological integrity since species introductions (usually increaser species) may actually increase the number of species, therefore a more appropriate scale of measurement would be one of intactness or integrity of the original biota (Landsberg *et al.* 1997b). Wilson (1990) sees changes in species composition from a production point of view when he states that a change from one palatable species to another should not be seen as degradation, however, this does not acknowledge the role of a species in ecosystem processes apart from being cattle fodder.

Numerous authors citing a variety of experiments and observations in both Australia and Africa have reached fairly similar conclusions in terms of cattle-induced changes in plant community composition. The conclusion appears to be that heavy grazing results in a decrease in palatable perennial grasses, with an increased abundance in unpalatable grasses, annuals and forbs (Ash & Corfield 1998, McIvor & Gardener 1995, Mwalyosi 1992, Shaw & ‘t Mannetje 1970).

2.1.5 How does grazing facilitate change?

The frequent statement that grazing impacts upon range ‘condition’ needs quantifying. ‘Condition’ could be considered to be the ability of the land to absorb rainfall and produce useful feed (Rolfe *et al.* 1997). These factors are influenced by the degree of ground cover (minimum of 30-40%) and the degree of domination by perennial, palatable and productive grasses (Rolfe *et al.* 1997). Ash & Corfield (1998) defined pasture condition by the broad groups of plants that dominated, such as palatable perennial tussock grasses versus less palatable perennial grasses, annual grasses and forbs.

Cattle are selective grazers and specifically target certain palatable plant species while ignoring less palatable species (Ash & Corfield 1998). As a particular palatable species becomes more rare, the more grazing selectivity increases until that plant has considerably declined in abundance (Ash & Corfield 1998). Less palatable species may not be grazed until other more palatable species have declined in abundance (Ash & Corfield 1998). Selective grazing has long been considered to be one of the major cause of shifts in rangeland plant communities (Hart 1999, Ash & Corfield 1998, Orr *et al.* 1991). Examples of this may be seen in the decline and/or elimination of palatable perennial grasses such as *Themeda triandra* and *Heteropogon contortus* in Tanzania and the Eastern Transvaal in Africa (Mwalyosi 1992, O’Connor & Pickett 1992) and in central Queensland, especially after drought (Shaw & ‘t Mannetje 1970). Remaining clumps of palatable perennial grasses show a decrease in basal circumference, possibly leading to decreased seed production (O’Connor & Pickett 1992). Because of selective grazing behaviour, Ash & Corfield (1998) believe that reducing stocking rates may not necessarily allow the recovery of a targeted species since the remaining cattle will continue to selectively graze that species. Regular ‘spelling’ is therefore required to allow recovery.

Selective grazing pressure favours numerous less palatable species. The increased abundance of *Stylosanthes* under grazing has been interpreted as a consequence of cattle favouring competing grasses in the early wet season and a higher tolerance of *Stylosanthes* to poor conditions (McIvor & Gardener 1995). Jones *et al.* (1997) also note that Stylos often require the death of competing perennial grasses to get established and that dominance often increases under higher stocking rates.

Exotic pasture grasses such as *Cenchrus ciliaris* and *Urochloa mosambicensis* may increase in abundance after heavy grazing because they have a higher tolerance to defoliation than many native perennial grasses (McIvor & Gardener 1995). From trials, Hodgkinson *et al.* (1989) record that the introduced *Cenchrus ciliaris* produced horizontal nodal tillers below the defoliation level, allowing it to have a 10-fold higher leaf surface area after grazing than the native *Themeda triandra*. They also predicted that if *Themeda triandra* was defoliated too regularly, as occurs under heavy grazing, that it would die whereas the *Cenchrus ciliaris* would survive (Hodgkinson *et al.* 1989). The low growth form of *Bothriochloa pertusa*, and its low horizontal rhizomes also assist this species to be favoured by grazing (Partridge 1995).

An example of grazing related changes to species composition can be seen on pastures monitored at Virginia Park near Charters Towers. Under grazing, pastures changed from tall perennial grasses with numerous legumes and other dicots, to a pasture dominated by Indian couch (*Bothriochloa pertusa*) with much bare soil and xerophytic forb species such as *Portulaca* (Gardener *et al.* 1990). Generally, this type of grazing-induced transition in pasture composition appears to be a recurring theme in much of the literature on the subject.

2.2 IMPORTANCE OF DIVERSITY TO ECOSYSTEM PROCESSES

2.2.1 What roles do species perform?

There is an unquestionable value of biodiversity for our economic future, products and pharmaceuticals, regulating climate and weather conditions and our moral obligations to protect biodiversity. Of particular interest here, however, is the value of the diversity of species to ecosystem function at the ecosystem level. While it has been a popular notion that ecosystems rely upon species diversity for proper functioning, there is considerable dissent and controversy surrounding that view (Tilman *et al.* 1996). Ecosystem functions involve the movement of energy and nutrients through air, water, land and food chains (Pimm 1994), and include productivity, carbon sequestration, water relations, nutrient cycling and storage, litter quality and stability

(Grime 1998). Ecosystem stability is a measure of resilience (elasticity or the ability of the system to return to normal after disturbance) and resistance (inertia: ability of the ecosystem to avoid displacement during a perturbation) (Leps *et al.* 1982). Different species have different roles to play in ecosystem function. Species vary and complement each other due to differences in phenology, photosynthetic mechanism, rooting depth and reproductive biology (Grime 1998).

2.2.2 Is there species redundancy?

A major question is how many species are actually required to maintain ecosystem functions at natural levels? Rather than examining total species numbers, many authors prefer to examine trophic levels or functional groups. Within a particular functional group, there may be several species performing the same function, suggesting a high degree of redundancy (Pimm 1994). Ecosystem processes should conceivably be able to function just as well with single representatives from each functional group if this was the only consideration. One problem encountered with discussing functional groups is that of ‘keystone species’. These are functional species without redundancy, the presence or absence of which changes the properties of the whole ecosystem (Schulze & Mooney 1994). Lawton (1994) asks us whether it matters if ecosystem functions are divided amongst few, or many different species and proposes four alternative hypotheses for consideration:

- Redundant species hypothesis: A few species are necessary for ecosystem functioning, but beyond that, many species are redundant
- Rivet hypothesis: Like the rivets that hold together an aircraft, there is a degree of redundancy but only so many rivets can be removed before the wing falls off or the ecosystem collapses
- Idiosyncratic response hypothesis: As the number of species changes, so does ecosystem function, however, the size and direction of the change would be impossible to predict due to the complexity of function of individual species
- Null hypothesis: There is no relationship between the number of species present and ecosystem function

2.2.3. Biodiversity and productivity

Several authors concentrate on the effect of diversity on different ecosystem functions. Although the species richness of natural communities is often strongly related to their productivity, the direction of this relationship is known to vary widely (Kassen *et al.* 2000). McNaughton (1994) presents data indicating that primary productivity (expressed as biomass above and below ground) is not related to species richness. In contrast, Tilman *et al.* (1996) present results from a replicated experiment with 147 plots to show that productivity does indeed increase with plant species richness. Due to intrinsic differences between species in terms of nutrient uptake, then nutrients such as nitrogen tends to be utilised more completely, and therefore there is reduced leaching of nitrogen from the environment (Tilman *et al.* 1996). Lowering the number of species present then increases the rate of loss of soil nutrients, ultimately reducing soil fertility and thus lowering productivity (Tilman 2000). In addition, a European study showed that the removal of half the species in a plot reduced the productivity by 10-20%, and that a plot with one species is less than half as productive as a plot with 24-32 species (Tilman 2000). The nature of the relationship between species richness and productivity may, however, be dependant on the ecosystem under examination. Schulze & Mooney (1994) note that increasing species richness has been shown to either increase or decrease ecosystem productivity, biomass and stability depending on which system you happen to be looking at. Research using microcosm ecosystems suggests that the relationship between species richness and productivity is a bell-shaped curve. Species richness initially increases with productivity, and then declines in highly productive systems, due to changes in niche availability (Kassen *et al.* 2000). In conclusion, although there is ample evidence showing a strong correlation between species richness and productivity, there are other conflicting results that indicate the nature of the relationship varies between different ecosystems.

2.2.4. Biodiversity and stability

As with the relationship between species richness and productivity, there is also considerable debate on the relationship between species richness and stability. Most definitions of stability concentrate on either the dynamic stability of populations or an ecosystems resistance and resilience to change (McCann 2000). Most authors accept the fact that at any given point in time, many species are redundant in terms of their role in ecosystem functioning, as long as their functional groups are preserved (Grime 1998, Pimm 1994). However, several arguments revolve around the notion that it is in times of perturbation that species richness shows the greatest benefit in maintaining ecosystem stability. Functional groups are much more likely to survive a perturbation if they are diverse (with a high level of redundancy) (Chapin *et al.* 1992). A higher degree of redundancy within a functional group presents a greater chance that one species will be present that responds differently to a perturbation (McCann 2000). Therefore, increased species richness gives rise to greater stability by the increased ability of a community to contain and retain species or functional groups that exhibit certain responses (McCann 2000). Like an investment portfolio, species exhibit variation in response, and the more species that variability is spread over; the less variable is their total sum (Tilman 2000).

Conversely, Pimm (1994) suggests that it is simple, low diversity systems that show the greatest level of resilience, since the resilience of a community is dependent upon the response of the least resilient species. Observations by Pimm (1994) and Elton (1958) that plant communities are more susceptible to invasion by exotic species if they have few species and simple patterns of interspecific interactions suggest a relationship between diversity and resistance. When a community has certain species removed, this causes the community to become simplified and thus allows a quicker invasion of new species, which may in turn accelerate the loss of further original species (Pimm 1994). Schulze & Mooney (1994) concluded that despite the fact that there is species redundancy within functional groups, the benefits of species richness to resistance means that there is no actual redundancy in the long term.

Grime (1998) states that it is not plant species richness *per se* that is of importance, but that the influence a plant has in its environment is proportional to its biomass. On that basis, plant species can be divided into three groups: dominants, subordinates and transients (Grime 1998). It is the abundance of the dominants (defined by their higher biomass) that regulate ecosystem processes. The main role of subordinate species is to regulate or ‘filter’ the recruitment of dominant species, while transient species are an indicator of the complexity of the seed bank and therefore the resilience of that ecosystem (Grime 1998). The conclusion to that argument is that the loss of plant species richness has its greatest impact on the recruitment of dominants, rather than any intrinsic ecological effects of species richness *per se.*

In conclusion, the research results presented in the literature vary. It appears that for some ecosystem processes, there may well be a degree of redundancy, while for others, such as resistance and stability, the rivet hypothesis appears likely. It is most likely that the effects of species richness on ecosystem processes may vary between different ecosystems.

2.3 INTERMEDIATE DISTURBANCE AND COMPETITIVE EXCLUSION.

2.3.1 Disturbance

Numerous definitions of disturbance have been presented in the literature. Sousa (1984) provides the most useful definition as it relates to grazing impacts on tropical graslands:

“A disturbance is a discrete, punctuated killing, displacement, or damaging of one or more individuals (or colonies) that directly or indirectly creates an opportunity for new individuals (or colonies) to become established”

In tropical grasslands, there are three major forms of natural disturbance: climatic, fire and biotic (e.g. grazing, burrowing, excavating or wallowing) (Collins & Barber 1985).

In a model presented by Grime (1977), different plants respond differently to disturbance, depending on their particular speciality towards competition, disturbance or stress. A summary of this model may be that a particular species either resists competitors or resists disturbance, but it can’t excel at both (Petraitis *et al.* 1989). If a particular species is a superior competitor, then it is usually susceptible to disturbance or predation. According to this model therefore, the species richness of any ecosystem represents a balance between the rate of disturbance and the rate of competitive exclusion (Petraitis *et al.* 1989).

2.3.2 Competition and Disturbance

The ‘Principle of Competitive Exclusion’ suggests that complete competitors cannot coexist and that there must be some niche separation between species for coexistence to occur (Gotelli 1995). Where resources are limited, the more similar species are in their use of shared resources, then the more precarious their coexistence (Gotelli 1995). Petraitis *et al.* (1989) assert that the idea that species richness is regulated by competition, is central to ecology, and that the role of disturbance as a mechanism for increasing the number of existing species comes from its impact on competitive interactions.

The ‘Intermediate Disturbance Hypothesis’ predicts that the greatest number of species should occur at intermediate levels of disturbance (Petraitis *et al.* 1989). The relationship between species richness and grazing intensity is often represented as a bell-shaped curve. At very low grazing levels, species richness is low because the vegetation is dominated by a few competitive dominants (Petraitis *et al.* 1989, McIvor 1998). In contrast to the view by Milchunas *et al* (1988) that grazing exerts a uniform impact on all individuals of all species, cattle are often highly selective in the selection of preferred species (Ash & Corfield 1998). If a grazing disturbance targets and thus excudes the competitively dominant species, then competition amongst remaining species is less and so recruitment levels are higher (Petraitis *et al.* 1989). At very high grazing levels, species selection is less specific, so an equally low number of species occur as few can survive the high stress and disturbance levels (McIvor 1998). As disturbance levels increase, fewer competitors and more colonising species survive, but at even greater disturbance, then local extinctions caused by the disturbance are no longer matched by immigration and so species richness declines (Petraitis *et al.* 1989). The majority of species would therefore be found at an intermediate level of disturbance, as it is easy to survive the lower levels of stress while still escaping competitive exclusion by the aggressive dominants.

The 'Intermediate Disturbance Hypothesis' is a non-equilibrium model, in that it assumes that species composition is rarely in equilibrium and that high species richness is maintained only when the composition is constantly changing (Petraitis *et al.* 1989). In some scenarios, competitive exclusion is not reached due to the high frequency of disturbances. Collins & Barber (1985) suggest that the intermediate disturbance hypothesis is simplistic, and that species richness is not simply a function of the frequency, intensity or magnitude of a single disturbance, but is a result of the responses of species to several concurrent disturbances (e.g. climatic and grazing).

Milchunas *et al*. (1988) also notes that the intermediate disturbance hypothesis lumps all disturbances in together on a single axis, and that evolutionary history of exposure to grazing is important to determine whether grazing is actually a disturbance to a particular plant community. This would be of greater significance in comparing Australian sites to sites in Africa.

While the concept of a bell-shaped curve to explain the relationship between grazing pressure and species richness seems logical and alluring, it is not always supported by empirical studies. In examining the relationship between grazing pressure and species richness along a grazing gradient, Landsberg *et al.* (1997a) found a series of trends that were not consistent between sites. Some disturbance-richness curves were U-shaped, while others were increasing or hump-shaped (∩) curves. While not discounting the possibility of a sampling artefact, Landsberg *et al.* (1997a) suggest that the diversity of curves represents a real diversity of responses to disturbance.

The effects of the disturbances will vary depending on variables such as the size, frequency and timing of the events. When that disturbance mostly affects the superior competitor through selective grazing, then that will increase species richness in what is known as ‘predator-mediated co-existence’ (Petraitis *et al.* 1989). Different disturbances will impact on species richness in different ways. For example, grazing may reduce the competitive superiority of a species, while wallowing may increase habitat heterogeneity (Collins & Barber 1985). The effects of disturbance are also dependent upon the scale of the impacts being examined. In a study by Chaneton & Facelli (1991), they found that species richness was higher in grazed than ungrazed plots at the point scale (quadrat size) but that the opposite was true at the community scale. The reason for this is that at the fine scale, resource availability, resource partitioning and competitive interactions regulate species diversity, while at the community level, species richness may be more dependant upon habitat heterogeneity (Chaneton & Facelli 1991). Grazing increases species richness at the fine scale because of its impact on competitive exclusion, while on the larger scale, grazing reduces heterogeneity and causes smaller patches to resemble each other in terms of species composition and dominance relations.

2.4 RESEARCH INTO GRAZING IMPACTS USING EXCLUSION CONTRASTS

The potential for using fenceline contrasts to examine the impact of disturbance on plant communities was recognised by Dyksterhuis (1958) who noted that “Communities, as distinguished from community types, are sharply defined where there are abrupt changes in physical environmental factors or disturbances”. Such abrupt changes in disturbance can be easily located along certain fencelines separating sites with different management regimes. The use of exclosures can therefore allow for the collection of empirical and objective rather than anecdotal evidence for changes to species richness by grazing. The use of exclosures may also be used to test the practicality of de-stocking as a tool for pasture regeneration (McIvor & Gardener 1990). It should be noted that, in the present research, the term exclosure does not refer only to experiments specifically designed to exclude grazing, but to any fenceline or feature that has a similar effect, for whatever purpose.

Exclosures have been used to evaluate herbivore impacts in a variety of vegetation communities in western North America, Argentina and Africa (Milchunas *et al* 1988, Ryerson & Parmenter 2001). Milchunas *et al* (1988) report on results into the impacts of grazing on grassland communities from several different regions of the world with different evolutionary histories of grazing. This is an important factor in the response of vegetation communities to grazing. In summary,

- Western North America (short evolutionary history of grazing): Moderate grazing did not produce any large, immediate effects on community composition or physiognomy compared to exclosures, however, no impacts on diversity were recorded
- Patagonian Argentina (short evolutionary history of grazing): A clear decrease in species number, diversity and equitability occurred with increased grazing pressure.
- Argentian flooding pampas (subhumid grasslands with a short evolutionary history of grazing): Grazing increased diversity through an increase in forbs. Exclosures resulted in increased cover of grasses and sedges with an increase in tussock size, an increase in perennials and decrease in annuals and exotic species.
- Serengeti grasslands of South Africa (long evolutionary history of grazing): A higher plant diversity was recorded in grazed areas compared to exclosures.

In contrast, there have been few Australian rangeland exclosure experiments published. McIvor (1998) undertook research at Hillgove and Cardigan Stations near Charters Towers between 1982 and 1992, examining effects of pasture management on diversity by measuring the number of species per plot and number of species per quadrat. McIvor & Gardener (1990) set up experiments on 20 pastures near Collinsville, North Queensland, but only monitored them for a year. The objective in this case was to determine what soil and pasture characteristics might be useful in predicting the capacity of a degraded pasture to regenerate. They were also interested in knowing whether overgrazed paddocks can regenerate by themselves or whether they need inputs such as seed, cultivation or other soil disturbances for regeneration to occur. Orr *et al.*

(1991) set up exclosures specifically to see whether burning and excluding stock could reverse degradation of pastures of speargrass (*Heteropogon contortus*). While it was found that those treatments increased spear grass from 20% to 70% (Orr *et al.*1991), no assessment was made on effects of overall species richness. McIvor & Gardener (1990) examined the floristics of their plots, but combined it with pasture growth measurements to get a 'site condition value'. It was found that the basal area of perennial grasses was the best predictor of pasture improvement, since there was little seed in the soil bank and regeneration was particularly slow when existing clumps of grass were absent (McIvor & Gardener 1990).

Of the greatest relevance to this research is the study by Scanlan *et al.* (1996a,b), which utilised some of the same exclosures as those used in the present research, although macropod exclosures were not monitored. Scanlan *et al.* (1996a,b) examined these sites from 1985 to 1990, prior to the severe drought of 1992-1994 reported by Fensham & Skull (1999) as having the lowest rainfall recorded in 113 years at Charters Towers. Several parameters such as the cover of the dominant species (speargrass and Indian couch), grass basal area, yield, and recruitment and mortality of the woody plant component were measured (Scanlan *et al.* 1996a). The study showed declines in vegetative cover, yields and grass basal area with increased grazing and an increase in woody plant recruitment with reduced mortality in the exclosure areas. Although that study yielded valuable information of the condition of plots prior to this study commencing, there were, however, no measures made of species richness as a whole.

A number of exclosure experiments have been reported for areas outside Australia. Stohlgren *et al* (1999) measured 26 exclosures in Colorado, Wyoming, Montana and South Dakota USA with a mean age of 31.2 yrs. Although anticipating lower species richness in the exclosures due to competitive exclusion, results showed nearly identical species richness in all treatments. Although Stohlgren *et al* (1999) found no significant difference in species diversity, eveness or cover of various life forms (eg grasses, forbs & shrubs), they noted that there was only an overlap of species composition of 57.9 ± 2.8% between grazing treatments. The significance of this was diminished by results showing an even smaller overlap between grazed plots. They concluded that grazing does affect local plant species and life form composition but the spatial variation is considerable, and that few plant species show a consistent directional response to grazing or grazing exclusion. They concede that the low number of consistent results may be due to the areas sampled having a long evolutionary history of grazing and recommend further work be conducted in habitats with a different evolutionary history.

In Argentina, a cattle exclosure experiment reported by Facelli (1988) involved three treatments. One plot was grazed by cattle, the other had cattle excluded for nine years and the third had cattle excluded for nine years then reintroduced for three years. The plots were compared in terms of species richness and composition. After two years of reintroduction, the third plot was most similar in composition to the ungrazed plot, however, after the third year it was most similar to the grazed plot. Facelli (1988) suggested that the species richness was highest in the reintroduced grazing plot after two years since increaser and decreaser species from the other treatments were able to co-exist; an observation that supports the Intermediate Disturbance Hypothesis.

An experiment in England by Putman *et al.* (1989) examined the effects of exclosure of fallow deer on plant species composition. After 25 years of exclosure, they found only a small change in species composition, which they suggested might be an artefact of the fact that the area has had a past grazing history of centuries. Putman *et al.* (1989) concluded that the speed and extent of recovery is affected by the degree of isolation from the potential colonisers, in effect mirroring the principles of island biogeography.

McIvor & Gardener (1990) also suggested that pasture recovery would depend on immigration from outside areas when certain desirable perennial grasses had been totally eradicated. Many decreaser species are not entirely absent from the landscape but still persist along roadsides or other ungrazed areas (Orr *et al.* 1991). It is likely that pasture recovery in some areas may be dependant on immigration from these roadside reservoirs of decreaser species.

In conclusion, an examination of the literature revealed that there has been relatively little research conducted in tropical Australia that have attempted to elucidate the relationship between grazing and species richness. The rangelands of north Queensland may be expected to show different results to several studies from other countries due to differences in evolutionary histories of grazing.

2.5 SEARCHING FOR PREVIOUSLY UNGRAZED AREAS

While comparing adjacent grazed and ungrazed sites is useful, it is also useful to examine the effects of grazing on previously ungrazed sites (Facelli 1988). Fenced exclosures may merely document recovery from pre-existing grazed states and therefore only highlight ecosystem functions that are reversible (Fensham & Skull 1999). However, it is generally accepted that that areas that are truly ungrazed by cattle are either rare or restricted to specialised habitats (Fensham & Skull 1999).

Debrot & Defreitas (1993) examined grazing impacts at Curacao, Netherlands Antilles, by comparing livestock-inaccessible rock vegetation with lower altitude grazed vegetation, however, sites such as these are uncommon in North Queensland. A rare opportunity exists in the site selected by Fensham & Skull (1999), which consists of pockets of grassland surrounded by braids of broad walls of rough basalt flows inaccessible to cattle, thereby ensuring that those grasslands do not have a history of cattle grazing. Their findings highlight the complexity of the issue of cattle impacts on species richness. Plots that were grazed had lower values of species richness and Shannon-Weiner diversity index values for perennial herbs but a higher species richness and abundance of annual grasses. Two species (*Capillipedium parviflorum* and *Sehima nervosa*) were found only in the cattle exclosure plots and possibly represent species highly intolerant to grazing. In general, Fensham & Skull (1999) recognised a broad shift from perennial species to a mixed annual-perennial grass flora under grazing; a trend recognised in the State and Transition model for the northern speargrass zone produced by McIvor & Scanlan (1994). Aspects of the research by Fensham & Skull (1999) that may give confounding results include the exclusion of fire as well as grazing and the possibility that the results may be specific to that particular land and soil type. Grazing intensities in the grazed plots were moderate to the extent that it was not enough to completely eliminate a dominant life form (Fensham & Skull 1999). Some models described earlier in this chapter may suggest that this needs to happen for a significant change to occur in vegetation succession.

Locating ungrazed areas for the present research proved problematical. While some studies have utilised “accidental reserves” such as ungrazed roadsides, cemeteries and railway reserves, it was found that these areas were generally too narrow, subject to other disturbances such as mowing and soil compaction, frequent fire, and, in the case of roadsides, subject to water enrichment from runoff. Small areas have large edge to area ratios, which increases the problems of edge effects altering the vegetaion within the protected area. Long, narrow exclosures exacerbate this problem. Since stock frequently graze through wire fences, a fence boundary cannot always be guaranteed to be the precise edge of the grazing effect.

It was the original intent to examine grazing impacts along a grazing gradient such as occurs with increasing distance from water. These gradients produce circular zones of grazing intensities around water holes known as ‘piospheres’. Although gradients of ever-decreasing grazing pressure occur from 1-2km from water, cattle generally travel up to 10km between watering points and will travel further during drought (Landsberg *et al.* 1997a). As a result, rangelands could not be assumed to be ‘ungrazed’ unless they are more than 18km from water (Landsberg *et al.* 1997a). After extensive examination of topographic maps of the Dalrymple Shire, it had not been possible to locate any suitable grazing areas further than five km from water, excluding any possibility of locating sites remote enough from water to be considered to be lightly grazed or ungrazed.

In conclusion, locating ungrazed areas in the semi-arid tropical savannas of north Queensland that have been heavily grazed for over a century is difficult. There are certain specialised habitats that have been free from grazing, such as steep and rocky hillsides or areas with an abundance of the toxic poison heart-leaf (*Gastrolobium grandiflorum*). Trying to locate areas of the same habitat that have been grazed so that comparisons can be made is more difficult, since areas that have not been grazed previously are usually unsuitable for such a purpose. The use of exclosures therefore presents the only practical tool for the study of cattle grazing impacts in the semi-arid rangelands of North Queensland and has been shown to yield interesting results in several overseas studies.

CHAPTER 3: **GENERAL METHODOLOGY**

3.1 SITE SELECTION

A range of sites on different soil types was considered desirable for the present research, since a common criticism of many research projects is that results cannot be extrapolated to other areas (Grice & McIntyre 1995). It was also considered necessary to have, where possible, several sites representing a single soil type to examine whether a particular site is showing a typical response for that soil type. Five other sites were visited but were rejected because of confounding factors such as different application of fire management and tree clearing between grazed and ungrazed plots. For selection, each site had to fit the following criteria:

1) That fences existed and were placed in such a way as to result in different grazing histories and effects (treatments).

2) That both sides of the fence (treatments) were as similar as possible with respect to climate, topography, soil, dominant vegetation, pasture improvement and other confounding influences such as fire management.

3) That they were within reasonable distance from James Cook University in Townsville, North Queensland. All but two of the 13 sites were located in the Dalrymple Shire.

Also avoided were sites that had extreme slopes ($>10^0$), strongly contrasting slope or aspect between treatments, areas of significant rock outcropping, wetlands or dissecting watercourses or had been cleared or cultivated. Some areas may have been sown with introduced pasture species or were adjacent to such areas, but this was not considered to be an excluding condition so were included in the samples. It was not possible to determine if any such practices had actually been carried out within any of the research plots. Introduced pasture grasses such as buffel grass (*Cenchrus ciliaris*) and Indian couch (*Bothriochloa pertusa*) dominated some sites. Since this is a widespread phenomenon throughout the northern rangelands, it was considered useful to include these sites, since differences in plant composition between plots may be an interactive effect of grazing on the abundances and competitive exclusion of these grasses. It was assumed that all sites were grazed by macropods.

In consideration of these numerous factors, a total of 13 separate sites on eight soil types were examined for the purposes of the present research.

3.2 TYPES OF EXPERIMENTS

3.2.1 Cattle and macropod exclosure sites

The five sites referred to collectively in this thesis as 'QDPI Exclosures' were established by the Queensland Department of Primary Industries (QDPI) as cattle grazing experiments in 1986. Each of these experiments was set up on operating cattle properties within cattle grazing paddocks. They consisted of a single four hectare paddock fenced off to exclude cattle and horses, but not macropods and other such grazing animals. In addition to this, another exclosure had been erected to prevent access to all grazing animals including macropods. This exclosure is only one quarter of a hectare (50m X 50m). Several other sites had also been prepared in an identical fashion, however, in the intervening years, fences were removed allowing grazing to occur within the excluded areas, thus erasing any useful data that could be collected. One site that was rejected as a study site had the fences removed, the vegetation burnt and cattle allowed to graze the regrowth. In addition to the five sites prepared by the QDPI in Charters Towers, another similar site was located at Epping Forest National Park. At this site in 1989, QDPI staff from Rockhampton had erected a 50 X 50m grazing exclosure within the boundaries of the National Park. The park itself is a cattle exclusion zone, but is bordered by cattle grazing properties. This site was of particular interest since the cattle exclosure area was grazed by the Northern Hairy Nosed Wombat (*Lasiorhinus krefftii*); a highly endangered marsupial with a total population size of approximately 100 individuals (Horsup A. pers. comm.).

3.2.2 Fence line Comparisons

Seven locations were used in which cattle were either excluded or rarely grazed on one side of a fence but were able to freely graze the other side. Plots labelled as 'ungrazed' had cattle excluded, but were still accessible to macropods, rabbits and other grazing animals. Plots where both cattle and other grazing species graze freely throughout the year are labelled as 'grazed' plots. On one site (Mt Leyshon mine), cattle were allowed to graze certain plots during the dry season but were excluded during the wet season. At Lucky Downs dam, cattle infrequently gained access to an otherwise ungrazed plot. For both sites, these plots are hereafter referred to as 'intermittently grazed' plots. In each situation, the sites were carefully assessed to ensure that the criteria listed in 3.1 were met. Numerous sites had to be excluded due to differences in tree clearing histories, burning, slope and aspect, and soil differences. Many of the sites from which cattle were excluded were mine sites bordered by cattle properties.

The relative distribution of plots showing the different grazing intensities are summarised in Table 3.1 below.

Table 3.1: Number of plots representing grazing treatments at each study site.

Site	**Ungrazed**	**Intermittently grazed**	**Grazed**	**Macropod exclosure**
K.Hills East	3		3	1
K.Hills West	3		3	1
Blue Range	3		3	1
Leyshon View	3		3	1
Kirk River	3		3	1
Mt Leyshon (1 & 2)	3	6	3	
Pajingo Mine	3		3	
Jervoise Stn	3		3	
Lucky Downs		3	3	
Rishton Mine	3		3	
Epping Site 1	3		3	
Epping Site 2	3		3	1
Tabletop	3		3	
Total:	36	9	39	6

Total number of sites examined = 13

Total number 50 X 50 m plots examined = 90

90 plots @ 30 quadrats/plot = 2,700 quadrats + 90 timed searches

3.2.3 Limitations of experimental design

A number of factors limited the number of plots that could be sampled. Exclosure plots were established in many cases more than a decade before the current research commenced. The addition of new ungrazed plots to these experiments was therefore not possible. Fenceline contrasts suffered confounding factors such as different landforms, soil types, vegetation types and distance from water that eliminated the possibility of being able to add more comparable plots to each site. This limited number of plots meant that replicates are not statistically independent of each other, and thus the experimental design is pseudoreplicated. This in turn limits the choice of statistical analyses that can be used to explore the data and reduces the robustness of those analyses. James Cook University statisticians Mike Steel and Tim Hancock were consulted and it was agreed that the design did not lend itself towards a robust statistical analysis. It was advised that although the design was pseudoreplicated, this only tends to have a major impact on analysis with much larger data sets than the one presented in the current thesis. The analyses used in this thesis, as recommended by statisticians, are essentially one of data exploration, not hard statistical tests. Nevertheless, the opportunity to examine and document the changes that have occurred as a result of grazing exclusion far outweighed the negative aspects of the experimental design.

Although the presence of aggressive exotic grasses such as buffel grass (*Cenchrus ciliaris*) was expected to have a significant impact on plant communities through competitive exclusion, it was expected that its impact on those communities will be regulated by grazing pressure. Thus, although sites dominated by such grasses are not native pastures in the strictest sense, they are an unavoidable element of rangelands in north Queensland and their interaction with grazing to influence the many aspects of plant diversity was considered to be a worthwhile field of study.

3.3. SITE DESCRIPTIONS

Photographs illustrating four of the thirteen sites are shown in Plate 1 (a-d) below.

Plate 1(a-d). Appearance of pastures at four representative sampling sites

a) Kangaroo Hills East (ungrazed plot dominated by *Heteropogon contortus*).

b) Mt Leyshon (fence-line effect between intermittently grazed and ungrazed plots).

c) Epping Forest National Park (site 2): ungrazed plots dominated by buffel grass (*Cenchrus ciliaris*).

d) Tabletop (fence-line effect between grazing treatments).

Descriptions of each of the study sites are provided in Appendix A using the following descriptions. A summary of site characteristics is shown in Table 3.2 and a map showing the locations of the various sites is provided in Figure 3.1.

3.3.1 Explanations of site descriptions

SITE NUMBER: Sites are ordered by the way in which they are grouped for analysis, not in the order in which they were surveyed. These numbers are used instead of site names on graphs where space does not permit use of the full name or abbreviation.

SITE LOCATION: The local government shire in which the site is located is given, as well as coordinates given as latitude/ longitude and UTM/UPS. Locations were determined using a hand-held Garmin 12 Global Positioning System (GPS).

DATE SURVEYED: Timing of plant collection was vital. In very dry seasons, short-lived plants may not be represented in the vegetation at all (Landsberg *et al*. 1997a). Assessing vegetation only after rainfall reduces the seasonal variability on a site and, by assessing the Wet Period Average Cover (WPAC), it is possible to gain an indication of land degradation as shown by the degree of wet season recovery (Bastin *et al*. 1993). Although the QDPI sites were initially visited during July or September 1998, sampling of all sites took place only during the ‘wet’ season. This was to ensure that the greatest proportion of species richness would be sampled. During the dry season, annual plants die off and so sampling during these times would lead to a significant underestimate of total plant species richness. In addition to this, the majority of plants collected during the wet season bore fertile parts such as flowers or fruit, which are necessary for the correct identification of most species. Collecting dry, dead and infertile specimens during the dry season would have resulted in most plants being unidentifiable. Hence any assessment of species richness in the field would have been ineffectual. It can be noted from Appendix A that the Tabletop site was sampled in early June 2000; a time usually considered being well within the ‘dry’ season. Fortunately, the 1999-2000 wet season extended considerably into the year, allowing sampling to continue into this period. Such late season data collection was necessary since sampling during the wet season can be a precarious pursuit, due to accessibility problems on some properties. It was therefore necessary to collect data over two consecutive wet seasons. To avoid any possible bias this might have had on results, each individual site was completely surveyed within two days. None of the sites was half-surveyed one wet season, then completed the following wet season. Sampling a particular site several times in different years or at different times of the year would doubtless give different results. Likely differences would include changes in suites and abundances of species, especially annuals. Since the comparison was focussed more on treatments rather than with different properties, the value of this approach would be doubtful.

<u>SOILS</u>: Soils types were determined using soil maps (Isbell & Hubble 1967, Rogers *et al.* 1999). The descriptions of the colour and texture of the soils presented in Appendix A are usually self-explanatory. Descriptive soil terms used are defined below:

- Chromosols: Soils with a clear or abrupt textural B horizon which is *not* sodic and not strongly acidic (Isbell 1996)
- Sodosols: Soils with a clear or abrupt textural B horizon which *is* sodic and not strongly acidic (Isbell 1996)
- Dermosols: Soils that do not have clear or abrupt textural B horizons but have a B2 horizon with a structure that is more developed than weak (Isbell 1996)
- Duplex soil (texture contrast soils): Soil in which there is a sharp difference in clay content between the surface and subsoil over a boundary of less than 5cm (Rogers *et al.* 1999)
- Kandosols: Soils that have a B horizon that is massive, has more than 15% clay, is not clear or abrupt and are not calcareous (Isbell 1996)
- Tenosols: Soils with a distinct A horizon but otherwise with weak structural organisation (Isbell 1996)

<u>CLIMATE</u>: Average yearly rainfall is given as an indication of the relative ability of sites to react and recover quickly from disturbance. Large differences in mean annual rainfall between sites allow predictions to be made about differences in vegetation composition. Rainfall records are derived either from published information for that site or from records for the nearest weather station or rain gauge. In some cases, rainfall records were derived from the property owners. It should be noted that there is often considerable deviation from the mean in some years (see Figure 2.1), and the scale of this deviation varies considerably between sites.

VEGETATION TYPE: Vegetation type is a description of the dominant upper strata, giving an indication of the spacing between trees and the dominant tree species, following the classification scheme used by Specht & Specht (1979). Epping site 2 occurs on a vegetation type considered to be 'of concern' (Sattler & Williams 1999).

DESCRIPTION OF SITE: A description of the site is given in relation to its general layout and recent history of grazing and grazing exclusion. Much of this information is anecdotal and unpublished, and is usually derived from the property owners themselves.

DESCRIPTION OF PLOTS: A description is given of the floristic components of the three dominant strata: tree layer, shrub layer and grass/forb layer. Where data is available, the species dominating that strata was identified and its relative contribution to the total was expressed as a percentage of the number of quadrats in which it occurred. The relative contribution of common tree species to the total number of individual trees is also given. Sub-dominant and commonly occurring species are listed. Species listed as 'common' were usually those occurring in at least 5 of the 30 quadrats (17%). All non-indigenous species are denoted with an asterisk (*). The occurrence of scat types was noted as this indicates the type of grazing occurring in the plot, but not the grazing intensity. A herd of some 40 cattle was observed moving and feeding in a plot, however, upon later examination, very few droppings were left behind.

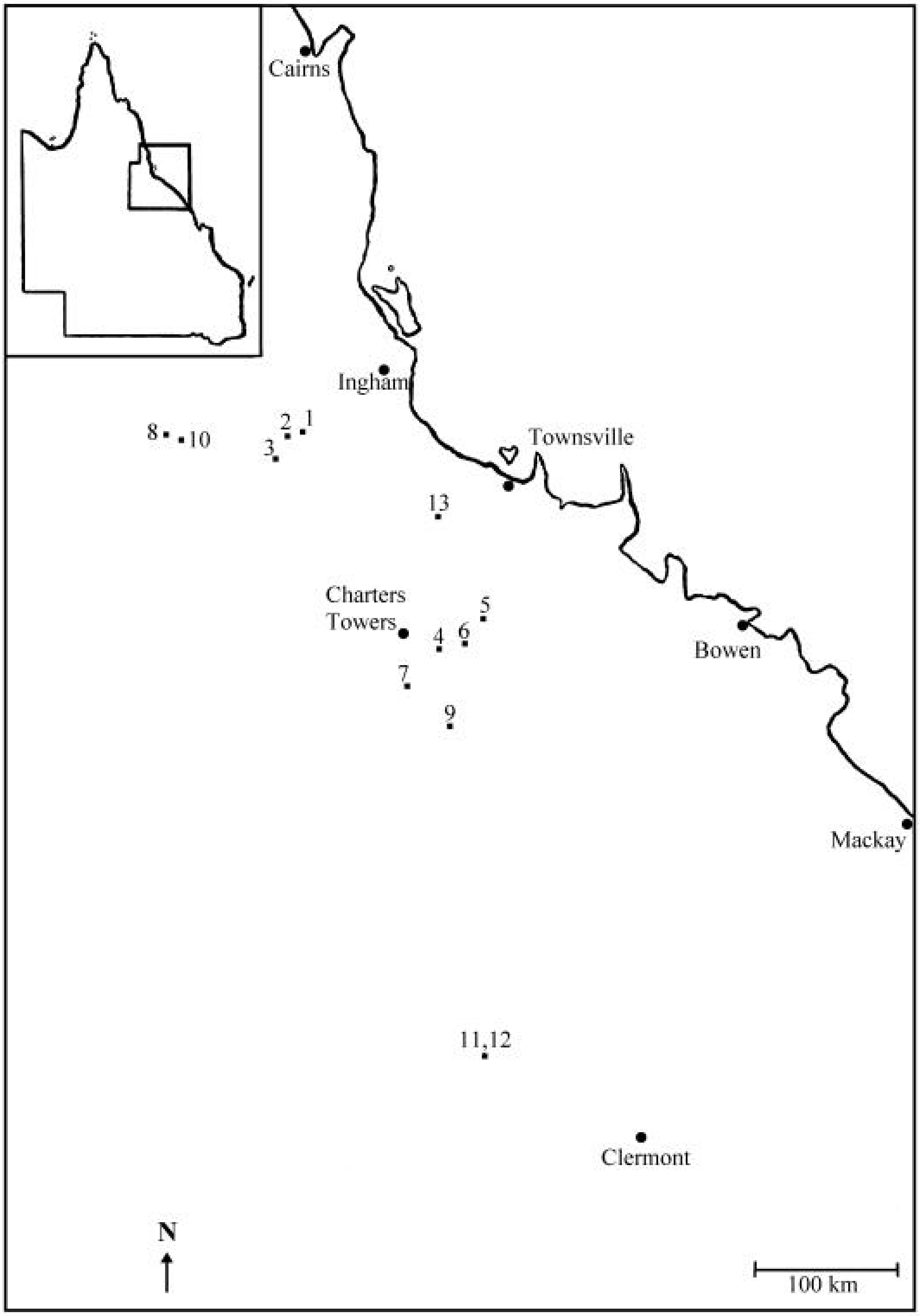
GRAZING PRESSURE: Grazing pressure should be considered to be minimum in macropod exclosures, medium in cattle exclosures and intermittently grazed plots, and highest in grazed plots, relative to each other at a particular site. It should be noted that a heavily grazed plot on one site may not have the same level of disturbance as a heavily grazed plot on another site due to different stocking rates between properties. Differences in grazing pressure between sites were not possible to quantify accurately due to the anecdotal nature of those records. Anecdotes on the different grazing pressures at various sites are included in the vegetation descriptions in Appendix A.

NOTES: Special notes were made where they were deemed useful to understanding the appearance of the site and historical factors that may have influenced its current state.

Table 3.2: Summary of site features for sites used in cattle grazing research (* = introduced species)

Site Name	Site No.	Experiment type	Soil Type	Vegetation type	Dominant Grass	Mean rainfall	Exclosure age
Kangaroo Hills East	1	Cattle & macropod exclosure	Greenvale (GR) Brown/ Yellow Chromosols	*Eucalyptus crebra* open woodland	*Heteropogon contortus* , *Bothriochloa bladei*	640 mm/ yr	13 years
Kangaroo Hills West	2	Cattle & macropod exclosure	Greenvale (GR) Brown/ Yellow Chromosols	*Eucalyptus crebra* open woodland	*Themeda triandra/ Heteropogon contortus*	640 mm/ yr	13 years
Blue Range	3	Cattle & macropod exclosure	Greenvale (GR) Brown/ Yellow Chromosols	*Eucalyptus crebra* open woodland	*Heteropogon contortus, Panicum larcomianum*	640 mm/ yr	13 years
Leyshon View	4	Cattle & macropod exclosure	Dalrymple (DA) Red Chromosols	*Eucalyptus crebra/ Corymbia erythrophloia* open woodland	*Chrysopogon fallax, Melinus repens**, *Bothriochloa pertusa**	659.5mm/ yr	12 years
Kirk River	5	Cattle & macropod exclosure	Dalrymple (DA) Red Chromosols	*Eucalyptus crebra* open woodland	*Bothriochloa pertusa**	650 mm/ yr	13 years
Rishton	6	Fence-line comparison	Dalrymple (DA) Red Chromosols	*Eucalyptus crebra* open woodland	*Bothriochloa pertusa**/ *Heteropogon contortus*	659.5 mm/ yr	11-12 years
Mt Leyshon (1)	7	Fence-line comparison	Rangeview (RA) Red Chromosols	*Eucalyptus crebra* open woodland	*Cenchrus ciliaris**, *Bothriochloa pertusa**	660 mm/ yr	5-6 years
Mt Leyshon (2)		Fence-line comparison	Rangeview (RA) Red Chromosols	*Eucalyptus crebra* open woodland	*Cenchrus ciliaris**, *Bothriochloa pertusa**	660 mm/ yr	5-6 years
Jervoise	8	Fence-line comparison	Rangeview (RA) Red Chromosols	*Eucalyptus crebra* open woodland	*Themeda triandra*/ *Bothriochloa pertusa**	711 mm/ yr	?
Pajingo	9	Fence-line comparison	Pentland (PE) Red Kandosols	*Acacia shirleyi* woodland	*Chrysopogon fallax*/ *Aristida calycina*	600 mm/ yr	13 years
LuckyDowns Dam	10	Fence-line comparison	Wairuna (WR) Red Dermosols	*Eucalyptus crebra* open woodland	*Heteropogon contortus*	711 mm/ yr	10 years
Epping site 1	11	Fence-line comparison	Epping Chromosols	*Eucalyptus brownii* woodland	*Cenchrus ciliaris**	671 mm/ yr	19 years
Epping site 2	12	Cattle & macropod exclosure	Walthum Tenosols (Siliceous sands)	Mixed Eucalypt woodland	*Cenchrus ciliaris**	671 mm/ yr	19 yrs (12 for exclosure)
Tabletop	13	Fence-line comparison	Bluff: Brown Sodosols	*Melaleuca viridiflora* woodland	*Heteropogon contortus*	750 mm/ yr	~ 12 yrs

Figure 3.1: Map of northeastern Queensland showing location of study sites. (site numbers are provided in Table 3.2 on previous page).

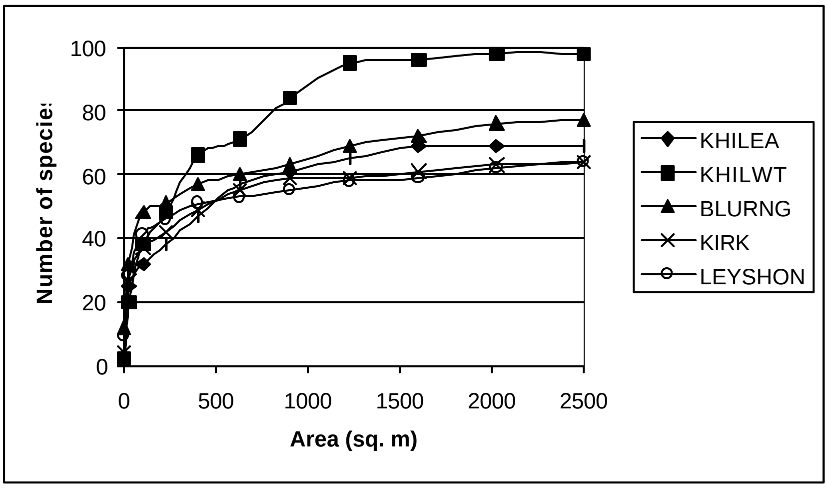


3.4 SELECTING PLOT SIZES

The long-term nature of the QDPI exclosure experiments ensured that no alterations could be made to the experimental design. Although the grazing exclosure on those sites was four hectares in size, the macropod exclosure measured only a quarter hectare. Comparing the species richness of a large area with a small area is invalid as species richness is proportional to the size of an area being investigated. Any comparison between the macropod exclosure and the other treatments has to be made with a plot of an equal size.

To investigate whether a plot measuring 50 X 50m contained a representative diversity of plants indicative of the site (and would be a valid scale to measure plant species richness), species area curves were constructed for the five QDPI plots, using a single randomly-selected 50 X 50m plot within the ungrazed (cattle exclosure) area. Squares of incrementally increasing size were marked with surveyors tape, with an edge length of 1, 5, 10, 15, 20, 25, 30, 35, 40, 45 and 50 metres. The strip between the square being measured and the previous square was intensively searched and every new species was recorded. In Figure 3.2 below, it can be seen that in each of the five sites examined, the species-area curve plateaus close to the 2,500m2 (50 X 50-metre mark), indicating that a quarter hectare plot is a suitable replicate size. The 50 X 50m replicate size is used throughout this research.

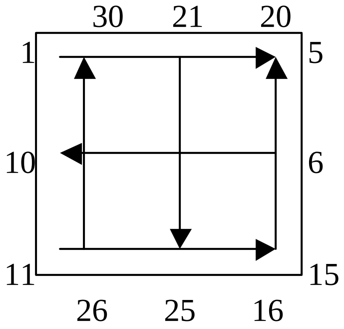
Figure 3.2: Plant species -area curves of the QDPI cattle exclosures examined in the present research.



3.5 COLLECTION OF DATA WITH QUADRATS

Although some estimates assume a random sample had been taken from a population, Zahl (1977) notes that field sampling is rarely conducted in such a fashion and that systematic sampling is more commonly practiced. A systematic sampling technique was employed in the present research. Within each of the 50 X 50m replicates, a total of 30 quadrats were thrown. Each quadrat was square in size with a side measuring 50cm (quadrat area = 0.25m^2). These quadrats were placed systematically within the plot along six transect lines. The first three transect lines were at right angles (90^0) to the last three transect lines. An illustration of the transect line directions can be seen in Figure 3.3 below.

Figure 3.3: Transect lines walked to for collect quadrat data
(numbers indicate quadrat number at the end of that line)



Data was recorded onto a pro-forma data sheet, consisting of a table for 30 quadrats and space for up to eight species per quadrat (See Appendix B). Within each quadrat, the total percentage cover was estimated and placed into a ranked category. The results of the cover data are presented in Chapter 7.

Data collected was in the form of species frequency. Friedel & Shaw (1987) note that frequency data is responsive to long-term shifts in management effects since it is a more stable measure than species weight or cover. Limitations of frequency data include the inability to derive herbage amount since it gives equal value to small and large plants, and the integration of patchiness and density (Friedel & Shaw 1987). To collect this data, all plants found to be growing in the quadrat were identified and noted. On the data pro-forma sheet, a number was written that corresponded with a unique species. A list of the species with their corresponding identification number was carried into the field. A copy of this list in various formats can be seen in Appendix C (a-c). Stohlgren *et al.* (1998) made comparisons of various sampling techniques and found that transect-based sampling techniques were generally reliable for estimating foliar cover and soil cover, but such techniques under-estimated species richness, especially in terms of numbers of rare species (<1% cover). Suggestions that searches be undertaken to "...assure a record of rare but important indicator species which may not otherwise be encountered on the transect" have been made in the past (Parker 1951). Stohlgren *et al.* (1998) found that conducting a search of a plot 1000m^2 in size greatly enhanced plant species detection and measurement. For that reason, the present research incorporated a timed search component into the survey, to identify and record species that were present in the replicate plot, but were too spatially rare to be encountered in a quadrat. In many of the open *Eucalyptus* savannas, these timed searches tended to include most of the tree species. Generally speaking, most of these habitats had very sparse woody vegetation (<10% foliage cover) that was rarely encountered with quadrats, although making the quadrats of an open-ended design did not exclude the possibility of a tree being recorded in a quadrat.

When species were located that could not be satisfactorily identified, a voucher specimen was taken and given either a new unique number or, alternatively, a code number (eg. GC21, BF18) to allow retrospective identification. Once the species was properly identified later in the laboratory, the pro-forma sheets were corrected so that they would display the correct identification number. It was found that both systems had their advantages and disadvantages. Obviously, all code numbers required correction, which was a slow and laborious task. Species that had numbers assigned to them were far easier to cross-reference. In some cases, species would be assigned a new unique number and then later identified as a species that already had a unique number. Correcting these numbers was far more difficult and numerous errors were encountered in setting up the database when incorrect numbers turned up in the data set.

In addition to collecting data relating to herbaceous species richness and cover, trees were also measured in the five QDPI exclosures. In these few sites, trees were identified; their diameter at breast height (DBH) recorded and note was taken as to whether the tree was alive or dead. This data was not collected in other sites, as trees were often either absent or present in such low numbers that data analysis would not have achieved any meaningful result. The results of this data are presented in Chapter 8.

3.6 PLANT IDENTIFICATION

Plant specimens were labelled according to the site, date, treatment and unique identification number or code. The specimens were then pressed and dried in a drying oven at 40^0C before being removed for identification. Specimens were mounted on stiff A5 card, labelled, sealed and sterilised by freezing to produce an extensive field herbarium, numbering over 350 specimen cards. While a large number of these species were readily identifiable using a broad assortment of field guides, many species required the use of plant keys involving careful dissection of reproductive structures and examination of indumentum under a dissecting microscope. Where possible, use was made of reference material housed in the herbarium at Tropical Plant Sciences, JCU, Townsville. Assistance in plant identification was provided by Assoc. Prof. Betsy Jackes (JCU), Russell Cumming (Senior botanist, EPA, Townsville), Bryan Simon (QLD Herbarium) and Dr Peter O'Reagain (QDPI Charters Towers).

While McIntyre & Filet (1997) highlight the importance of proper identification, they also recognise the need to lump certain species that are particularly difficult to discriminate either in the field or laboratory. In some cases, there was considerable confusion over plant identification due to obscure taxonomic boundaries or insufficient plant material to make an identification, so, to avoid confusion, it was deemed necessary to clump certain plant species together under a single species name. For example:

- *Eucalyptus crebra* may potentially contain individuals of *E crebra*, *E. drepanophylla, E. xanthoclada* and *E. whitei* as well as possible hybrids
- *Dicanthium annulatum* may potentially contain individuals of *D. annulatum*, *D. aristatum, D. fecundum* and *D. sericeum*

In other cases, species could not be satisfactorily identified as separate species due to a lack of keys and other reference material. In some groups of plants, enough specimens were available to make artificial groups of species. Problematic genera were *Cyperus*, *Eragrostis* and *Phyllanthus*. In the genus *Phyllanthus* in the family Euphorbiaceae, nine morpho-types were recognised from the collected specimens according to gross morphology, leaf venation, seed-coat architecture and indumentum. These species were labelled simply as *Phyllanthus* sp A, *Phyllanthus* sp B, *Phyllanthus* sp C etc. The grass genus *Eragrostis* was also particularly difficult to identify to species. Examination of collected specimens under the microscope revealed eight morpho-types that could not be taxonomically labelled.

3.7 DATA ENTRY AND ANALYSIS

The final data set consisted of quadrat data for 319 species in 90 plots, with corresponding ground cover data. Another 118 species were located during timed searches. A database of this size was too large for a single conventional spreadsheet. A data entry form using tick boxes for each species was constructed using the statistical program ACCESS. Once entered, the data was split into separate sites and imported into EXCEL for manipulation. Totals were calculated for rows and columns, enabling a database to be constructed using multiple sites. From this database, the number of species rather than the frequency of each species then used in calculating various diversity indices (see Chapter 5). While most calculations were performed using EXCEL, Simpson's Index was calculated using the software package 'Species Diversity and Richness' (Henderson & Seaby 1997). This program was also used in calculation of species accumulation curves (see section 5.5). Diversity indices are defined in Section 5.2.2 and the data used to calculate each diversity index is provided below the formulas. Since a diversity index is, mathematically speaking, a data transformation, and there is no guarantee that the data have a normal distribution, they are not comparable with univariate analysis (Steele M. pers. comm.). Diversity indices were instead ranked using a Mann-Whitney U-test, which allows for significant differences between treatments to be calculated on ranking alone. In determining significant differences between treatments, all probabilities were calculated at $P<0.1$, as a standard significance level of 0.05 would eliminate the opportunity to discuss results that appear interesting and potentially enlightening. This same probability level is applied to all statistical analyses throughout the present study.

For the analyses in Chapters 4 and 6 comparing selected species and functional groups, data relating to the timed search was then excluded, as there was no quantifiable index of abundance for these species. Calculations were then made to determine the number of quadrats in which each species or group occurred, and then transformed into proportions (% quadrats occupied) allowing for tests of abundance to be made using a Chi Square Test for multiple proportions.The best measure of abundance is the proportion observed (Steele M. pers. comm.). This test is also known as the "Chi Square test of Independence". This analysis is conducted using the statistical software package

Statistix version 7 using a two-sample test of proportions in calculations. The null hypothesis is that the two proportions measured are equal.

Analyses of functional groups excluded species that had less than 5% (less than 3 individuals) in at least one plot. This was based on recommendations by McIntyre & Filet (1997) that the inclusion of rare species can obscure patterns. The functional group data was then transformed into proportions (% quadrats occupied).

For each soil type, a Principal Components Analysis (PCA) was conducted using the multivariate statistical package CANOCO (version 4) and CANODRAW., with each PCA representing all the plots from a particular soil group. This technique is described in greater detail in Section 6.2.3. Hierarchical dendrograms of similarity were constructed using the statistics program SPSS (version 10) and used to help interpret Principal Component Analysis (PCA) triplots. Justification for clustering sites by soil is provided by the cluster analysis presented as an average linkage hierarchical dendrogram in Figure 4.1.

The ground cover data examined in Chapter 7 was collected as the frequency of occurrence of 6 different categories of percentage cover. These were analysed with a Monte-Carlo Chi^2 test calculated using the STATISTIX software package. Differences in the rate of dieback of Ironbarks (Chapter 8) were determined using the proportion of the total ironbarks The resultant proportional data were then transformed by arcsine {√ proportion trees dead} to satisfy the assumption of normal data distribution, and analysed using a nested ANOVA.Details of these various analyses and justification for their use are given in greater depth in the relevant chapters.

CHAPTER 4: NON-GRAZING INFLUENCES ON PLANT SPECIES RICHNESS

4.1 INTRODUCTION

Before undertaking any statistical analyses of species richness or other measure of ecosystem composition, it is necessary to first examine other factors potentially influencing vegetation community development. The results of this chapter allowed a more intuitive approach to be taken in deciphering patterns revealed in the following chapters.

4.1.1 Grazing versus abiotic constraints on community composition

The first consideration to be examined is the role of soil and climate influences, in determining community composition, in comparison to grazing. There are a number of important factors to be considered here in examining the composition of a community. At the largest scale, the presence of certain species and the exclusion of others depend on biogeographic distribution (Putman 1994). Secondly, the ability of a particular species to disperse to that community, and thirdly, the ability of that species to persist and reproduce will determine if a species can be encountered in a particular community (Putman 1994).

Dispersal ability is of some interest in this research. It should be noted that all plots, even those currently excluded from any type of mammalian herbivore, were subject to grazing in the past. Extremely grazing-sensitive species may have been eliminated from grazing paddocks in historical times. When an exclosure is erected, those areas become virtual islands in a sea of grazed pastures, which may be difficult for grazing-sensitive species to disperse across. Although numerous species known to decline under cattle grazing were recorded in this research, there is no way of determining if any species have previously disappeared and were unable to recolonise the exclosures from grazing refuges.

Once a species has dispersed to a particular community, it must survive the prevailing environmental conditions, both biotic and abiotic. Every organism has a restricted range of physio-chemical conditions over which it operates; each having lethal limits, a preferred range and an optimal range (Putman 1994). To cope with the abiotic environmental conditions of soil and climate, a species must possess the appropriate physiological machinery to cope. Microhabitats within a site allow for the co-existence of numerous species, which vary slightly in terms of their tolerance and their physical mechanisms (Barbour *et al*. 1987).

While Table 3.2 indicates average yearly rainfall for the various sites, it should be noted that actual recorded rainfall might deviate significantly from that average in any given year. The magnitude of such deviation is illustrated in Figure 2.1. For a plant species to have long-term persistence in the sites measured, they must possess the mechanisms to survive both climatic extremes of floods and prolonged drought.

Rainfall also has an interactive effect with grazing. Illius & O'Conner (1999) observe that high variability in rainfall prevents plant populations reaching equilibrium, and also influences the impact of herbivores due to changes in their population densities. A further factor is the change in herbivore movements and concentrations between periods of drought and good rainfall. In particular, areas distant from water may receive episodic intense grazing pressure during droughts (Illius & O'Conner 1999).

Differences in climate also affect the rate of soil formation, the rate of leaching and decomposition of organic material and the type of soil profile (Barbour *et al*. 1987). Sites examined in this research were distributed over eight different soil types (Rogers *et al*. 1999, Isbell & Hubble 1967). These vary in terms of the parent material they were derived from, their texture, chemistry, colour, profile and depth (Rogers *et al*. 1999, Isbell & Hubble 1967). Plant communities on these different soil types may have to cope with many soil differences, including organic matter content, water storage capacity, hydraulic conductivity (capacity of soil to transport water), porosity, permeability, cation exchange capacity, pH (affects plant nutrient availability) and, ultimately, fertility (Barbour *et al*. 1987).

Williams *et al*. (1993) summarise by stating that plant communities are highly diverse and are determined primarily by soil and climate, overriding the other influences of land use such as grazing. The present chapter tests the validity of that statement as it relates to the sites examined.

4.1.2 The role of dominant species and their interaction with grazing

The second consideration is the role that the dominant species has on the plant communities. The role of dominant species, as defined by their higher biomass and frequency of occurrence, is of vital importance in plant communities. The theory of intermediate disturbance (Chapter 2) states that species richness comes from a balance between the rate of disturbance and the rate of competitive exclusion (Petraitis *et al.* 1989). If grazing is seen as the source of disturbance, then the abundance or frequency of the dominant grasses fits comfortably into the role of regulating competition. Grime (1998) notes that it is the dominant species that regulate ecosystem processes, with the main role of subordinate species being to regulate the recruitment of those dominant species. It is likely then, that the behaviour of the dominant grass species in response to cattle grazing is crucial to the understanding of cattle grazing effects on the ecosystem as a whole. Considering the significance of dominant species in an ecosystem, they may well be defined as keystone species, where the presence of that species changes the properties of the whole ecosystem (Schulze & Mooney 1994). Depending on the species of dominant grass present, and its response to grazing, the presence of that species may have a positive or negative influence on the vegetation community. The palatability of the dominant grass is important since, when a dominant is palatable, grazing increases species richness through removal of competitive exclusion, but when a dominant is unpalatable, then grazing reinforces its dominance (McIvor 1998).

Since the degree of domination by perennial, palatable and productive grasses is considered a pre-requisite in determining land ‘condition’ (Rolfe *et al.* 1997, Ash & Corfield 1998), then determining the relative abundance of those species is of particular interest to both the grazier and the plant ecologist. The abundance of these species also gives some indication of the degree of selective browsing being undertaken by cattle. Since cattle are selective grazers and specifically target certain palatable plant species while ignoring less palatable species (Ash & Corfield 1998), then it is reasonable to presume that those palatable species are in the highest risk group for being overgrazed. It is likely that some herbaceous legumes and forbs should be included as being highly palatable, and this may be a contributing factor to the scarcity of some species. The impact of such selective grazing is a well known contributing factor to the decline of the grasses *Themeda triandra* and *Heteropogon contortus*; two significant species examined in this thesis. Monitoring these well-known decreaser grasses is thus a requirement for any study examining grazing effects on plant communities and pasture condition.

In contrast, many other grass species known to dominate areas of the semi-arid rangelands of North Queensland are either unpalatable or are of reduced palatability relative to co-occurring native perennial tussock grasses. Low levels of palatability can be caused by numerous factors, including the presence of spines, increased fibre content with age or the presence of bitter or aromatic compounds such as essential oils, tannins and alkaloids (Wilson & Harrington 1984). In some cases, the difference in palatability may be slight, however, there may be considerable differences in their ability to withstand grazing pressure. For example, the low horizontal tillers and stolons present in *Cenchrus ciliaris* and *Bothriochloa pertusa* give those plants a competitive edge over *Themeda triandra* and *Heteropogon contortus* when subjected to heavy grazing (Hodgkinson *et al.* 1989, Partridge 1995).

There is a degree of confusion in the literature relating to the responses of several abundant grass species to grazing pressure. For example, red Natal grass (*Melinus repens**) is labelled as both an increaser (Rolfe *et al*. 1997, Henry *et al* 1995) and a decreaser (Roberts & Silcock 1982). Both Noy-Meir *et al* (1989) and Landsberg *et al* (1999) recorded numerous species with inconsistent population trends under grazing regimes. Several factors may explain these paradoxical results.

1/ There may be differences in site conditions such as soil attributes or rainfall between the sites (Noy-Meir *et al* 1989). This may give a particular species a relative advantage at one site and a disadvantage at another site (Landsberg *et al* 1999).
2/ Although grazing is selective, resulting environmental change can alter plant establishment and growth rates (Landsberg *et al* 1999).
3/ Different ecotypes of the same species may vary greatly in grazing related attributes and, therefore, grazing response (Noy-Meir *et al* 1989).
4/ Different site histories may result in the stabilisation of alternative vegetation composition at sites, resulting in inconsistent effects such as a discontinuous response to herbivory (Noy-Meir *et al* 1989).

By monitoring the abundance of ‘increaser’ grasses, especially relative to the abundance of ‘decreaser’ grasses, a strong indication may be obtained of changes to the dominant grass strata and the likely effects that this may have on species richness and ecosystem function.

Grass species selected for individual analyses are listed in the methods (4.2) below. In some cases, the identity of a dominant grass may have ramifications for the plant community, even in the absence of grazing. Buffel grass (*Cenchrus ciliaris**) is regarded as having the ability to alter vegetation community structure (Fairfax & Fensham 2000), so is analysed separately to test this assumption.

4.2 METHODS

At each site, two treatments (grazed and ungrazed) had three replicate plots measuring 50 X 50 metres. Macropod exclosures of the same size were not present at each site and were not replicated where they were present. Within each plot, 30 0.25m^2 quadrats were systematically thrown and the species present within each quadrat recorded.

Analysis and discussion of the results was conducted under three sub-topics:

- Abiotic influences vs. grazing impacts on plant communities
- Impact of grazing on select grass species
- Influence of buffel grass on community structure

All sites and species records were initially combined to examine the influence of cattle grazing over the entire study area. By combining the frequency data for all individual species recorded within quadrats, a hierarchical dendrogram was constructed in SPSS (version 10) to group sites, treatments and replicates by their of similarity of community composition and structure. This allows both abiotic effects and grazing to be put into perspective as ecological forces at the regional level, and tests the validity of analysing separate sites together clustered by soil type.

Since certain trends in the impacts of grazing may be related to the identity of the dominant grass species (Scanlan *et al.* 1996a), seven grass species were initially examined to determine if a significant difference was detectable between treatments. The abundance of these grass species (measured by frequency of occurrence within quadrats) was analysed using a Chi Square Test for multiple proportions, the results of which were used in following chapters to assist in interpreting other results in terms of their impact on other trends. The “Chi Square Test for multiple proportionsis also known as the “Chi Square test of Independence”. This analysis is conducted using the statistical software package Statistix version 7 using a two-sample test of proportions in calculations. The null hypothesis is that the two proportions measured are equal.

One of these grass species (*Cenchrus ciliaris*) was individually tested to determine its influence on community structure at Epping Forest National Park. The two sites on the park were selected as a test case due to the concern of national parks rangers that changes to forage opportunities for the endangered northern hairy-nosed wombat (*Lasiorhinus krefftii*) were being influenced by *Cenchrus ciliaris*. For the two Epping Forest sites, regressions were made between *Cenchrus ciliaris* and species richness using Statistix version 7. Regressions were also conducted between *Cenchrus ciliaris* and the grasses *Aristida* spp. and *Enneapogon* spp., previously identified as being preferred food items of the wombat (Woolnough & Johnson 2000).

4.3 RESULTS

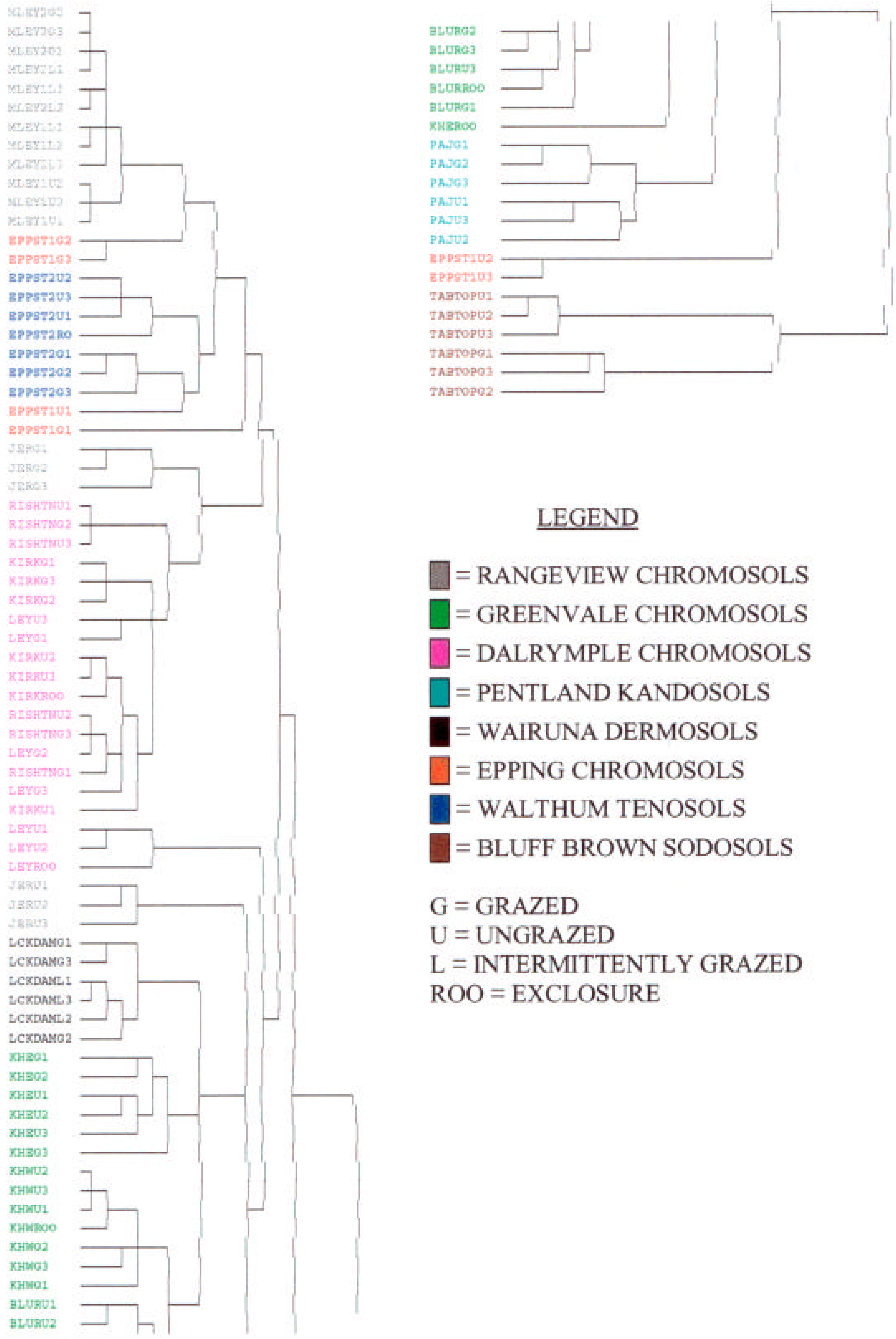
4.3.1 Abiotic vs. grazing influences on community structure

Cluster analysis was conducted on all sites to assess their similarity and dissimilarity based on the presence, absence and abundance of all species. This is presented as an average linkage hierarchical dendrogram in Figure 4.1 below.

Clusters produced using a hierarchical approach can be considered eurytopic (derived from higher level clusters) or stenotopic (derived from lower level clusters). In the dendrogram below, it can be seen that, largely, plots clustered within the large eurytopic branches of the dendrogram belong to the same soil type. Exceptions were plots in Epping Chromosols (Epping Forest 1), and Rangeview Chromosols. In the latter case, plots from Jervoise were separated from each other on the basis of grazing treatment, and were clearly separated from the other Rangeview plots (Mt Leyshon). In the Dalrymple soils, the plots Leyshon ungrazed 1 & 2 and Leyshon exclosure are clustered under a different branch than other plots on Dalrymple soils.

Stenotopic clusters indicate separation of plots on the basis of grazing treatment for several sites. At both Epping site 2 (Walthum Tenosols) and Kangaroo Hills West (Greenvale chromosols), ungrazed plots and the exclosure plot are combined together and separated from grazed plots. Pajingo (Pentland Kandosols) and Tabletop (Bluff Brown Sodosols) showed clear separation of grazed and ungrazed plots. At Mt Leyshon (Rangeview chromosols), there is clear separation of ungrazed plots, however, grazed and intermittently grazed plots are mixed. Plots on Dalrymple chromosols tend to cluster together under eurytopic branches. Within that group, however, there was no sort of stenotopic clustering, either by site or grazing treatment.

Figure 4.1: An average linkage dendrogram of all study plots, separated by relative abundance of all species recorded in quadrats.



LEGEND

▨ = RANGEVIEW CHROMOSOLS
🟩 = GREENVALE CHROMOSOLS
■ = DALRYMPLE CHROMOSOLS
■ = PENTLAND KANDOSOLS
■ = WAIRUNA DERMOSOLS
🟧 = EPPING CHROMOSOLS
🟦 = WALTHUM TENOSOLS
🟫 = BLUFF BROWN SODOSOLS

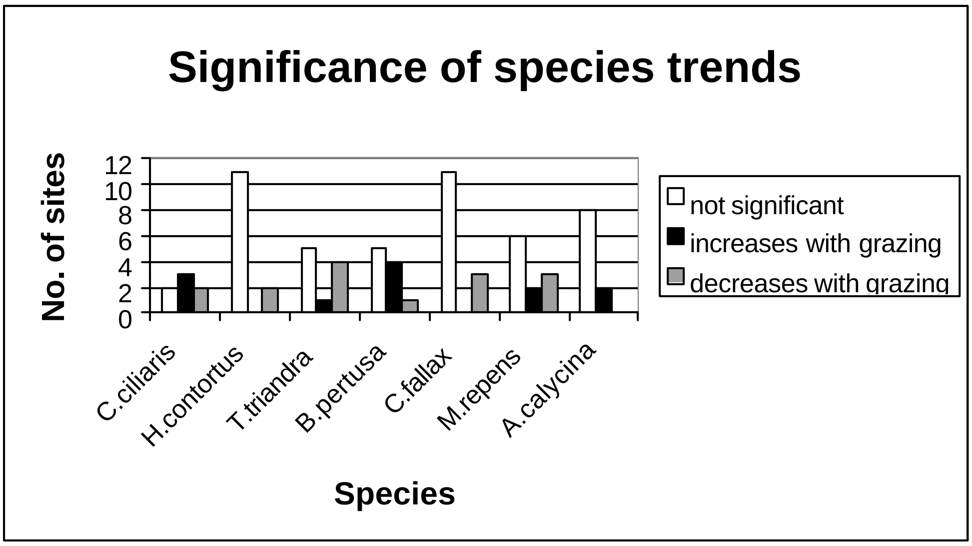
G = GRAZED
U = UNGRAZED
L = INTERMITTENTLY GRAZED
ROO = EXCLOSURE

4.3.2 Impact of grazing on select grass species

A selection of abundant grass species was analysed individually to allow analyses in other chapters to be interpreted by relating back to these species. The species included in the analysis were: buffel grass (*Cenchrus ciliaris** - #7), spear grass (*Heteropogon contortus* - 31), kangaroo grass (*Themeda triandra* - 36), Indian couch (*Bothriochloa pertusa** - 41), golden beard grass (*Chrysopogon fallax* - 43), red Natal grass (*Melinus repens** - 76) and dark wire grass (*Aristida calycina* - 109).

In Figure 4.2, the number of sites showing a significant difference in the frequency of these seven commonly abundant species is shown. Note that macropod exclosure plots are excluded due to lack of replication, and that abundance does not necessarily equate with biomass. Actual frequencies for the seven grasses can be found in Appendix E.

Figure 4.2: Histogram showing the number of sites with a significant change in frequency of seven grass species with grazing.



probability level: P <0.1

Figure 4.2 showed that most grass species showed a large proportion of non-significant results, with the exception of buffel grass (*Cenchrus ciliaris*). This result shows that, at the majority of sites, grazing had no significant impact on the abundances of the other six grass species. Where a significant result was obtained, these were largely divided between increases and decreases in abundance. Specific description of the results for each species is provided below.

Buffel grass (*Cenchrus ciliaris**): three sites showed a significant increase with grazing and two sites showed a decrease. At Mt Leyshon, both the ungrazed and grazed plots had a higher abundance than intermittent grazed. In a comparison between ungrazed and grazed at Mt Leyshon, there were significantly higher abundances in the ungrazed plots.

Spear grass (*Heteropogon contortus*): two sites had significantly higher values in ungrazed plots, while eleven sites showed no significant difference. No sites showed a significant increase with grazing.

Kangaroo grass (*Themeda triandra*): four sites had significantly higher values in ungrazed plots, while only one site had higher values in the grazed plots.

Indian couch (*Bothriochloa pertusa**): four sites had a significant increase in abundance in grazed plots, while one site showed an increase in ungrazed plots. There were three significant results on Rangeview soils, but none on Greenvale soils. At Lucky Downs Dam, it was favoured by light, intermittent grazing as opposed to continuous grazing.

Golden beard grass (*Chrysopogon fallax*): three sites had significantly higher values in ungrazed plots; no sites had higher values in the grazed plots.

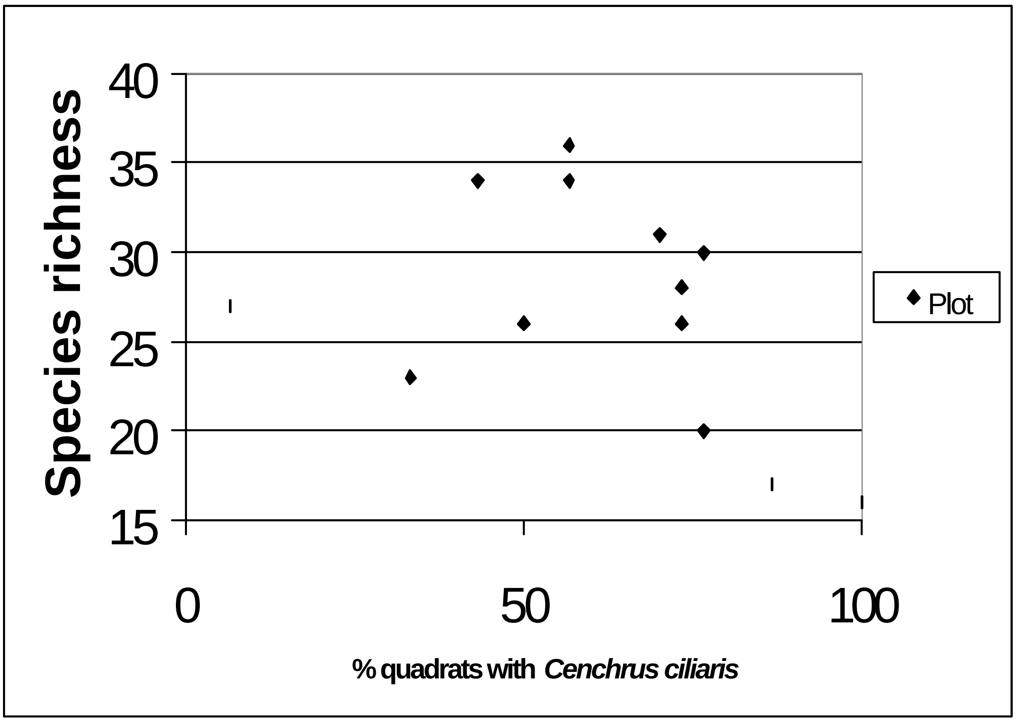
Red Natal grass (*Melinus repens**): five sites showed significant differences in abundance. On three sites there was a higher abundance in ungrazed plots, while grazed plots had higher abundance in two sites. Greenvale soils showed an increase in abundance with grazing, while Dalrymple and Rangeview soils showed a decrease.

Dark wiregrass (*Aristida calycina*): two sites showed significantly higher abundances in grazed plots (Blue Range and Pajingo). All other eight occurrences were non-significant.

4.3.3 Influence of buffel grass (*Cenchrus ciliaris*) and Indian couch (*Bothriochloa pertusa*) on species richness

After examination of the interaction between grazing and *Cenchrus ciliaris*, the interaction between buffel grass and species richness was tested for the two sites at Epping Forest. Initially, a scatter plot of *Cenchrus ciliaris* abundance and species richness for individual plots was presented, which was then tested with a regression. A regression is also conducted against the grass genera *Aristida* and *Enneapogon*, since these genera were subdominant and were identified by Woolnough & Johnson (2000) as being an important component of the diet of the northern hairy-nosed wombat (*Lasiorhinus krefftii*). The relationship between the dominance of *Cenchrus ciliaris* and species richness, as measured in the quadrat transects, is shown in Figure 4.3 below.

Figure 4.3: Scatterplot of species richness and buffel grass (*Cenchrus ciliaris*) abundance at Epping Forest National Park (sites 1 & 2).



The scatterplot in Figure 4.3 shows an erratic interaction between species richness and *Cenchrus* at lower levels of *Cenchrus* abundance. However, for plots where *Cenchrus* occurred in over 50% of quadrats, there is an apparent decline in species richness. This relationship is further clarified by conducting a regression on the 9 plots with over 50% buffel frequency. The data appeared to have a normal distribution and a graph of standardised residuals against fitted y-values indicated that variance in richness was homogeneous across the observed range of buffel abundance (homoscedastic). No transformations of the data was therefore considered necessary. Table 4.1 below shows results of regressions for species richness and frequency of selected grass species, versus *Cenchrus ciliaris* frequency (in plots where frequency of occurrence is more than 50%).

Table 4.1: Regression values comparing buffel grass (*Cenchrus ciliaris*) frequency with species richness and frequency of selected species at Epping Forest National Park. Only plots with over 50% frequency of *Cenchrus* were included in this analysis. (significant probability level: $P<0.1$)

Species	Slope (regression coefficient)	Regression P value (significance)	Adjusted R^2 (variation explained)
All (richness)	-1.62373	**0.0008**	0.7947
Aristida	-0.0146	0.3781	-0.0146
Enneapogon	-0.02777	0.7661	-0.1275

The regression P value showed a significant relationship between buffel grass frequency and species richness, in plots where buffel grass occurred in more than 50% of quadrats. The regression coefficient (slope) indicated that this was a negative interaction (richness decreased with increasing buffel grass frequency). Impacts of buffel grass abundance on the frequency of occurrence of *Aristida* and *Enneapogon* were not significant.

After noting the inverse relationship between *Cenchrus ciliaris* and species richness, it was decided to examine the interaction between *Bothriochloa pertusa* and species richness. Sites dominated by *Bothriochloa pertusa* were Leyshon View, Kirk River, Rishton and Jervoise. Scatter plots of *Bothriochloa pertusa* abundance and species richness for individual plots as measured in the quadrat transects, are presented below.

Figure 4.4.(a): Scatterplots of species richness and Bothriochloa pertusa abundance at the Mt Leyshon experimental grazing site.

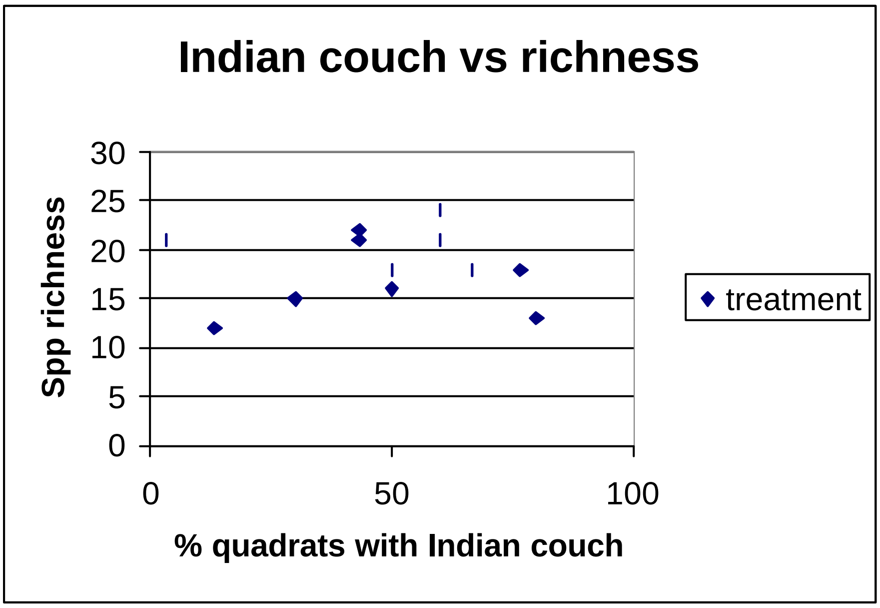


Figure 4.4.(b): Scatterplots of species richness and *Bothriochloa pertusa* abundance at the Kirk River experimental grazing site.

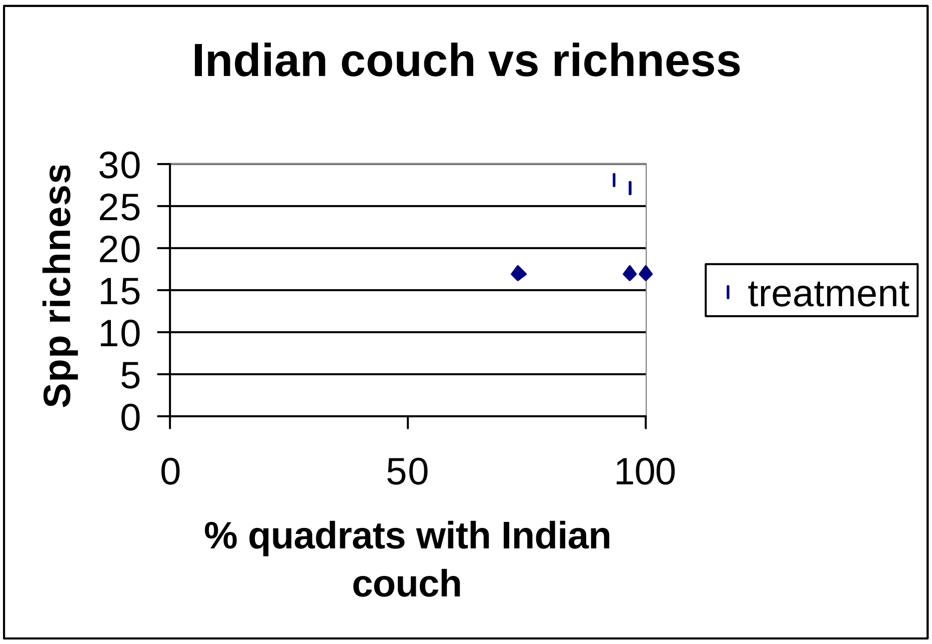


Figure 4.4.(c): Scatterplots of species richness and *Bothriochloa pertusa* abundance at the Rishton experimental grazing site**.**

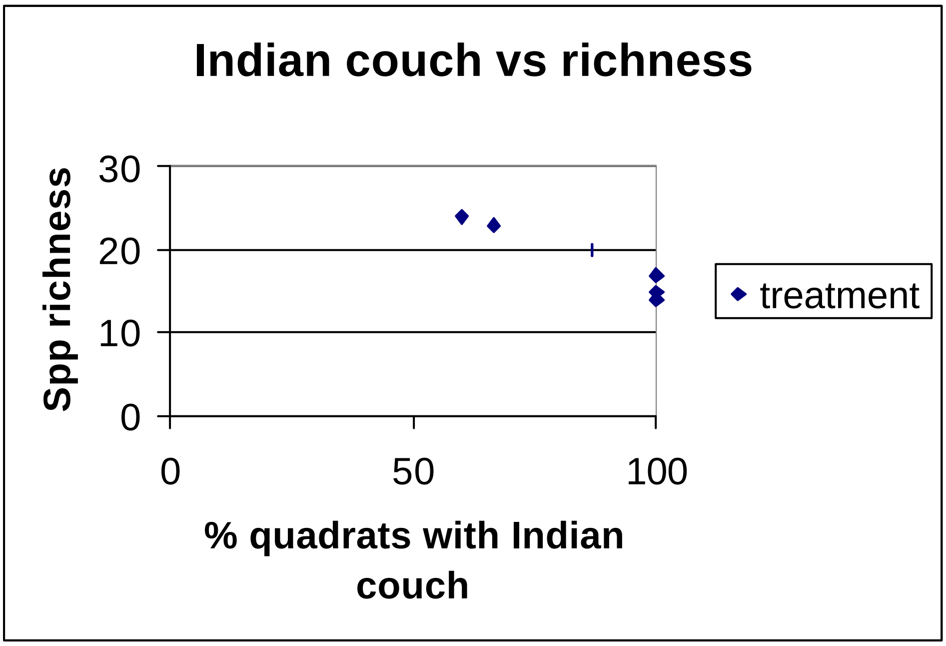
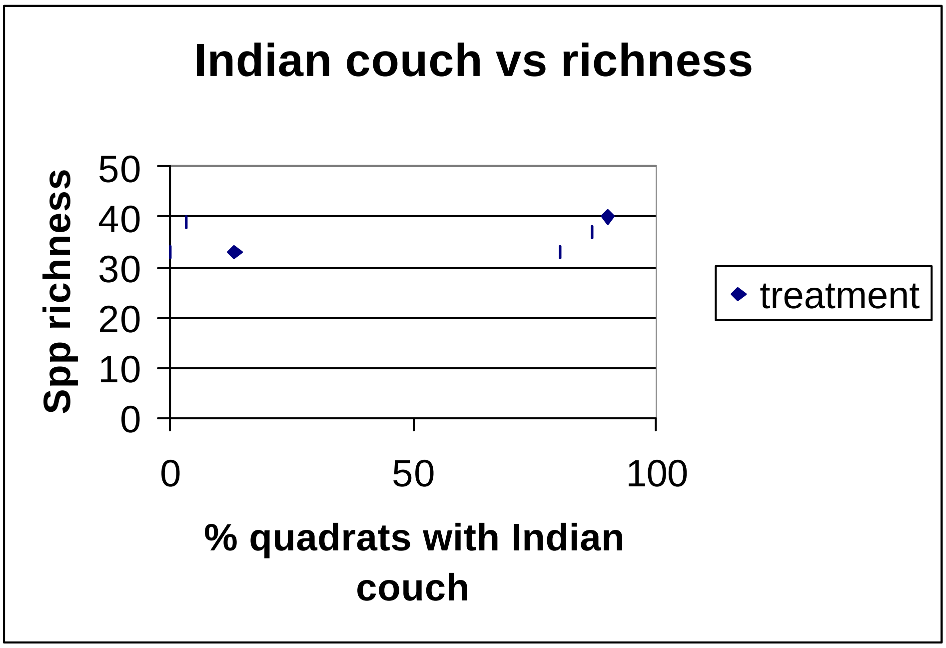


Figure 4.4.(d): Scatterplots of species richness and *Bothriochloa pertusa* abundance at the Jervoise experimental grazing site.



4.4 DISCUSSION

4.4.1. Grazing versus abiotic constraints

The dendrogram presented in Figure 4.1 examines the extent to which grazing treatment and site location determine the community composition of a pasture. It can be seen from Figure 4.1 that the majority of plots generally cluster together on the basis of soil type. Plots on Walthum Tenosols, Wairuna dermosols, Greenvale chromosols soils, Pentland kandosols and Bluff Brown sodosols also clustered by site. Where such clustering occurs, this could be taken as evidence that abiotic effects such as soil and climate are primarily responsible for determining the suite of species comprising the pastures. In addition, plots on Dalrymple chromosols clustered closely together by soil; however, there was no stenotypic clustering by either site or treatment. This indicates that site and treatment are having a relatively low influence on pasture composition compared to the role of the abiotic environment. Results showing that grazing treatment has a secondary stenotopic influence on pasture composition concur with statements by Williams *et al*. (1993) that plant communities are highly diverse and are determined primarily by soil and climate. These plant communities and the relative abundances of their constituent species are secondarily altered by other influences of land use such as grazing within the constraint envelope of abiotic and historical factors.

On a number of sites, grazing did exert a strong influence, where sampled plots of the same treatment generally clustered close together. The patterns and relationships between plots show a broad range of responses to grazing influence. Plots at Jervoise (Rangeview chromosols) were widely separated on the dendrogram on the basis of treatment, indicating the strong influence of grazing treatment compared to other factors. Plots at Epping Forest 1 (Epping Chromosols) were also widely separated, however, the division was not clearly on the basis of grazing treatment. One ungrazed plot at Epping Forest 1 clustered together with plots at Epping Forest 2 on Walthum Tenosols (deep siliceous sands). This may be due to abundance patterns in *Cenchrus ciliaris*; which were abundant on both Epping Forest sites despite differences in soil type. Grazing showed a strong influence at Mt Leyshon (Rangeview chromosols) where ungrazed plots, intermittently grazed plots and grazed plots all separated. Although such stenotypic separation is not considered as reliable as eurytopic divisions (Sheaves M. pers. comm.), it should be noted that intermittently grazed plots clustered as an intermediate state between grazed and ungrazed plots. The arrangement of these plots indicates a transition between extremes of grazing disturbance. At Epping Forest 2 (Walthum Tenosols) and Kangaroo Hills West (Greenvale chromosols), the exclosure plots clustered together with ungrazed plots; these clusters being distinct from grazed plots. This indicates that grazing by macropods resulted in little alteration to composition and structure to a community completely ungrazed by mammals. On Dalrymple chromosols, grazed plots at Leyshon View were distinctly different in their community composition from ungrazed plots, although the ungrazed plots themselves showed a range of variation. The ungrazed plots 1 & 2, and the macropod exclosure separated from the other Leyshon View plots at the eurytopic level, indicating a considerable difference being exerted on those plots. The causes for such differences are examined in greater depth in following chapters. Kangaroo Hills East (Greenvale chromosols) showed little variation in community composition between grazed and ungrazed plots, however, both of these treatments vary considerably from the macropod exclosure. These results indicate that on that on this site, both macropods and cattle were able to force a transition to a different vegetation community than that which existed in the total absence of grazing. Finally, both Pajingo (Pentland kandosols) and Tabletop (bluff brown sodosols) showed clear eurytopic separation of plots by grazing treatment. Although their clustering on the dendrogram shows that abiotic factors were obviously a strong influence, their division by treatment also highlights the important influence of grazing disturbance on species assemblages and abundances.

In conclusion, the majority of cases showed that abiotic site conditions had an overwhelming effect on determining community composition beyond grazing treatment effects. Most sites showed that grazing had a secondary effect on vegetation communities, although some sites indicate that grazing impacts can be significant enough to alter species. The influence of macropod grazing varied between soil types. Whether or not this was a difference in resilience of those communities to withstand macropod grazing, or simply a difference in macropod numbers and grazing pressure could not be determined.

4.4.2 Grazing and dominant grasses

The responses of the seven selected grass species to grazing treatment were varied. This was an expected outcome since the group of grasses analysed was comprised of natives and exotics, palatable and unpalatable species, and recognised increaser and decreaser species. Identifying the different trends on the different sites was considered a prerequisite to interpreting other results relating to plant composition since different trends in the impacts of grazing may be related to the identity of the dominant grass species (Scanlan *et al.* 1996a). This is because grazing will increase species richness through removal of competitive exclusion when a dominant is palatable, but when a dominant is unpalatable; grazing reinforces their dominance (McIvor 1998). It should, however, be noted that palatability may not be the only single trait which determines how a plant species will react to grazing. Landsberg *et al.* (1999) and Noy-Meir *et al.* (1989) list several other contributing factors that may influence a plants grazing response. These factors were described previously in Section 4.1.2.

Numerous authors focussed on the abundances of certain species that are believed to be good indicators of pasture condition or that may be a driving influence in the direction of ecosystem processes under grazing (Shaw & ‘t Mannetje 1970, McIvor & Gardener 1995, Scanlan *et al.* 1996a, Fairfax & Fensham 2000). All the grasses tested in this chapter are species that have described in the literature as being indicative of a particular grazing regime. From examination of Figure 4.2, there were a large number of non-significant results considering the weight placed upon these species as indicators by various authors. Species are individually discussed below:

Buffel grass (*Cenchrus ciliaris*)*: Results presented in Figure 4.2 showed that *Cenchrus ciliaris* can respond both as an increaser or decreaser species with grazing pressure. Results showed a significant increase in abundance with grazing in three out of five significant results, indicating that this species was an increaser at those sites. Particularly at Epping Forest (site 2), there was 80% more *Cenchrus* in one of the grazed plots than in the macropod exclusion plot. There was also considerably less *Cenchrus* in the exclosure than in the cattle ungrazed plots, indicating that *Cenchrus ciliaris* abundance may also have been positively influenced by macropod and wombat grazing. Woolnough & Johnson (2000) note that although both macropods and wombats regularly eat *Cenchrus ciliaris,* there is evidence to suggest that this is not a preferred food item, so those native marsupials may be exhibiting selective grazing pressure on species that compete with *Cenchrus*. In the Mt Leyshon experimental plots, *Cenchrus ciliaris* had higher abundance in ungrazed plots than in either intermittently grazed or grazed plots, a result which concurs with the observations of Rolfe *et al*. (1997) that *Cenchrus ciliaris* is a decreaser species.

It is possible that the discrepancy in *Cenchrus ciliaris* trends between sites indicates soil-related differences in palatability and therefore grazing response, or that difference in abundance relate to stocking rate. Although many graziers regard *Cenchrus ciliaris* as being palatable, the results presented here showing it as an increaser at three out of five sites would indicate its palatability may be lower than other co-occurring species. Casual observations made during the course of this study revealed tussocks of native grasses were heavily cropped wheras surrounding tussocks of *Cenchrus ciliaris* were untouched by cattle. Where those more desirable grasses have been greatly reduced by grazing pressure, grazing pressure would then focus on less desirable species. Wilson & Harrington (1984) commented that grazing impacts on a particular species is dependant on where that species sits in the hierarchy of palatability Therefore, the more intense the grazing pressure, the greater the impact on species with moderate to low palatability.

Spear grass (*Heteropogon contortus*): In both sites showing significant results, there was a significant decline in abundance of *Heteropogon contortus* with grazing. These results agree with comments by Rolfe *et al*. (1997), Grice & McIntyre (1995), Gardener *et al*. (1990) and Shaw and 't Mannetje (1970) who suggest that *Heteropogon contortus* would be expected to decline in abundance under heavy grazing. Heavy grazing of *Heteropogon contortus* is known to cause a reduction in seed production, seedling survival, plant survival, basal area and recruitment (Grice & McIntyre 1995). All other sites examined showed no significant change in *Heteropogon contortus* abundance with grazing, although at six sites, this was due to very low numbers of individuals. One site, Tabletop, is worth discussion. Although there were marginally more quadrats containing *Heteropogon contortus* in the grazed plot than in the ungrazed plot, most of those recorded there were small seedlings, in sharp contrast to the large, sexually mature specimens in the ungrazed plot. No older cohorts or mature specimens were located in grazed plots. This indicates that grazing hampered the long-term survival of *Heteropogon contortus* on this site; however, grazing pressure increased opportunities for seedling establishment. This result may suggest that removal of the top hamper by cattle grazing reduced competitive exclusion to the benefit of *Heteropogon contortus* seedlings. Strategic destocking of the grazed pasture at this time would allow these seedlings to mature, greatly enhancing the biomass of productive and palatable perennial grasses on that site, which would be to the advantage of the grazier.

Kangaroo Grass (*Themeda triandra*): Results showed *Themeda triandra* declining with cattle grazing pressure at four sites. At Tabletop, this species had been completely eliminated from grazed plots while it still persisted in a large proportion of quadrats in the ungrazed plots. On three other sites, *Themeda triandra* was so rarely recorded that no significant result could be expected. These results concur with comments by Milson (2000), McIvor (1998) and Rolfe *et al*. (1997) that *Themeda triandra* is a decreaser species that can be lost from heavily grazed pastures. At Lucky Downs Dam, a significantly higher abundance was recorded in grazed plots. Upon examination of the raw data, one grazed plot had a high proportion of *Themeda triandra*, while all other plots of both treatments had very low values. Although this shows that *Themeda triandra* was able to persist under the grazing regime imposed, more extensive sampling at this site may be required to see if this single plot was an anomaly. It should be noted that, similar to *Heteropogon contortus* at Tabletop, it was mostly seedlings that were recorded in grazed plots. This indicates that the prevailing level of grazing supported recruitment, but the rarity of individuals from previous cohorts indicated that survival rates were low.

Indian Couch (*Bothriochloa pertusa*)*: This species did not occur in five sites examined, supporting the observations by O'Reagain (pers.comm.) that this species has not spread across the entire area. At four sites, it was found to significantly increase with cattle grazing. At Mt Leyshon, it also increased with intermittent (dry season) grazing. These results support comments by Milson (2000) McIvor (1998) and Rolfe *et al.* (1997) that *Bothriochloa pertusa* is an increaser species. *Bothriochloa pertusa* was a relatively rare species at Blue Range. Although there was no significant difference between grazed and ungrazed plots at Blue Range, there was significantly higher abundance in the macropod exclosure than in the other treatments. Scanlan *et al.* (1996a) observes that *Bothriochloa pertusa* will continue to increase even in the absence of cattle grazing. These results indicate that *Bothriochloa pertusa* is well adapted to take advantage of grazing disturbance, but that grazing is not a pre-requisite for its spread and growth.

Golden beard grass (*Chrysopogon fallax*): This research showed that all three significant results obtained showed *Chrysopogon fallax* to be a decreaser species, having significantly higher abundance in ungrazed plots than in grazed plots. In several of the sites where no significant result could be obtained, this was due to the very low abundance of that species overall. Where macropod exclosures were available for comparison, abundances in those plots were generally intermediate between those for the grazed and ungrazed plots. This indicates that, while *Chrysopogon fallax* is sensitive to grazing pressure, it can also decrease to a lesser degree when grazing is entirely excluded. In the macropod exclosure at Leyshon View, clumps of *Chrysopogon fallax* were frequently very close together and appeared to be shading neighbouring individuals. It is likely that light macropod grazing in the ungrazed plots helped reduce this competitive effect, possibly resulting in an increased overall abundance in 'ungrazed' plots. The results showing *Chrysopogon fallax* to be a decreaser species agree with observations by Milson (2000), McIvor (1998), Rolfe *et al.* (1997) and Roberts and Silcock (1982), but differ with Grice & McIntyre (1995) and Henry *et al.* (1995) who state that this grass is an indication of heavy grazing pressure.

Red Natal grass (*Melinus repens*)*: Five sites in this study gave significant results; two indicating *Melinus repens* was an increaser and three a decreaser. In the conservatively stocked sites at Kangaroo Hills East and West, it was significantly more common in grazed than ungrazed plots, however, it reached its highest abundance in macropod exclosures on both these sites. At Leyshon View, *Melinus repens* behaved as a classic decreaser with highest abundance in the macropod exclosure, lower abundances in the ungrazed plots (indicating herbivory by macropods) and was entirely absent from grazed plots. At Mt Leyshon, it was significantly more abundant in ungrazed plots, even in comparison to intermittently grazed plots. *Melinus repens* has been labelled as both an increaser (Rolfe *et al.* (1997), Henry *et al.* (1995)) and a decreaser (Roberts and Silcock 1982). Results presented here support the observation that although *Melinus repens* is a pioneer species (Henry *et al.* 1995), it has low resistance to continuous grazing and is rarely seen in a well-grazed pasture (Roberts and Silcock 1982). Assuming a constant response to grazing at all sites, these results would then indicate that Leyshon View has higher grazing pressure than Kangaroo Hills East and West.

Dark wire grass (*Aristida calycina*): This species was found to be significantly more abundant in grazed plots on two sites. At Leyshon View, there was no significant difference in the abundance of *Aristida calycina* between grazed and ungrazed, however, it was entirely absent from the macropod exclosure, indicating that macropod grazing alone can be sufficient disturbance for this species to increase. In the other seven sites for which there was no significant result, *Aristida calycina* was a relatively rare species and abundance values were too sparse and low for an accurate assessment to be made. These results indicate that although *Aristida calycina* is an increaser species that indicates heavy grazing pressure where it dominates (Milson 2000, Rolfe *et al.* 1997), it will not always show that level of dominance under grazing across all sites examined. This may possibly be due to competition from other increaser species that do dominate those sites.

Notes on Lucky Downs Dam: Trends in the abundance of common grass species at Lucky Downs dam appeared to be consistently opposite to those expected. Continuously grazed plots had an insignificantly higher abundance of the decreaser grasses *Heteropogon contortus* and *Themeda triandra*; yet had significantly lower abundance of the increaser species *Bothriochloa pertusa*. This site was not a standard grazed-ungrazed experiment, but instead contrasts between continuously grazed plots (grazed) and intermittently grazed plots (ungrazed). There was a noticeable difference between the plots, with the taller and denser grass occurring in the ‘ungrazed’ plots. Petraitis *et al.* (1989) notes that if a disturbance is strong enough to exclude the competitively dominant species, then competition amongst remaining species is less and so recruitment levels are higher. As noted previously for *Themeda triandra,* it is possible that recruitment of decreaser species was higher in the grazed plots due to the absence of high levels of competition. There was little evidence of those grass seedlings surviving to maturity, suggesting that, while recruitment was enhanced under those particular circumstances, their persistence was diminished. No hypothesis is presented to explain the higher abundance of *Bothriochloa pertusa* in the intermittently grazed plots, except to note that that the expansion of this species is obviously not reliant on heavy overgrazing.

4.4.3 Influence of buffel grass (*Cenchrus ciliaris*) and Indian couch (*Bothriochloa pertusa*) on species richness

The relationship between *Cenchrus ciliaris* and the species richness of a site was tested using study plots at Epping Forest National Park and adjacent properties. Figure 4.3 and Table 4.1 shows the relationship between the dominance of *Cenchrus ciliaris* and species richness, as measured in the quadrat transects. It is evident that at lower densities of *Cenchrus ciliaris,* a lot of variation in species richness was present, however, at higher densities (frequencies over 50%), there appeared to be a notable decline in species richness. The significant regression values appeared to confirm a relationship between *Cenchrus ciliaris* and a decline in species richness. These results corroborate with those of Fairfax & Fensham (2000) that *Cenchrus ciliaris* has the potential to have a negative impact on species richness. Landsberg *et al.* (1997a) also describe *Cenchrus ciliaris* as a critically invasive species with the potential to displace native herbaceous species.

These results also suggested a negative impact of *Cenchrus ciliaris* on the diversity of diet of the critically endangered northern hairy-nosed wombat at Epping Forest National Park. Diversity of diet is considered to be of importance to the conservation of the northern hairy-nosed wombat; since numerous species implies stability through a higher probability that at least one of those grass species would survive a perturbation (Johnson C. pers. comm.). The previous section of the present chapter indicated an increase in abundance of *Cenchrus ciliaris* with grazing. The combined effect of grazing and *Cenchrus* introduction is likely to have been one of reduced species richness.

The relationship between abundance of Indian couch (*Bothriochloa pertusa*) and species richness was investigated at four sites (Leyshon View, Kirk River, Rishton and Jervoise) that were dominated by this species. Figures 4.4 (a-d) showed scatterplots of of *Bothriochloa pertusa* abundance and species richness for individual plots as measured in the quadrat transects. Unlike the result presented for *Cenchrus ciliaris*, no clear trend was identifiable from these plots apart from Rishton (Figure 4.4c), which shows a slight decline in species richness with increased cover of Indian couch and Mt Leyshon (Figure 4.4a), which showed highest species diversity at 50% cover of Indian couch. Kirk River (Figure 4.4b) shows wide variation in species richness at very high cover of Indian couch. Jervoise (Figure 4) has very different levels of Indian couch cover between treatments but there is no significant difference in species richness. This obvious contrast in the influence of Indian couch compared to buffel grass is not unexpected given their very different growth habits. While buffel grass is a tall clumping species that easily out-competes other species with smaller stature, Indian couch tends to be a low, prostrate species unlikely to generate much in the way of a shading effect. The contrasting results would suggest that while buffel may be encroaching into native pastures and displacing diversity through heavy competition, Indian couch is not so much of a severe competitor but is symptomatic of extreme events of drought and heavy grazing that may have reduced much of the species richness anyway.

4.5 CONCLUSION

The present research indicates that different sites supported vegetation communities with different floristic components and, in most cases; these were correlated primarily with differences in soil type. Sites on the same soil type clustered together, irrespective of grazing impacts. Within the constraints imposed by the abiotic nature of the site, grazing treatment had a secondary influence on separating plots into sub-groups. In a few cases, the influence of grazing treatment had a greater influence on community composition than did soil type. It can be concluded, therefore, that there is a broad diversity of floristic communities within the semi-arid rangelands and that these communities correspond largely to the heterogeneity of the landscape itself in terms of soil, climate and other abiotic effects. It must also be concluded that, while abiotic effects are important in determining which species may be present on a particular site, grazing plays a significant role in altering those communities.

Numerous grass species were commonly encountered throughout the course of the present research. These species vary in their palatability and desirability to both graziers and conservationists. Different grass species exhibited a broad range of responses to grazing pressure. These may have value in interpreting both level of degradation and grazing history. Differences in the response of the grass species examined may be due to the following reasons:

1/ In most sites that showed no significant result, the particular species examined occurred at frequencies too low for statistical analyses to show trends.

2/ Differences in site conditions such as soil or rainfall may give a particular species a relative advantage at one site and a disadvantage at another site (Landsberg *et al* 1999).

3/ Different ecotypes of the same species may vary greatly in grazing related attributes and, therefore, grazing response (Noy-Meir *et al* 1989).

4/ Different site histories such as stocking rates may have resulted in different frequencies of co-occurring palatable grass species, which may influence selective grazing behaviour.

5/ Different site histories resulting in the stabilisation of alternative vegetation composition at sites may result in inconsistent effects such as a discontinuous response to herbivory (Noy-Meir *et al* 1989).

Despite the apparent inconsistencies in results, once mitigating effects have been taken into consideration, some tentative generalisations can be made. The native perennial grasses considered to be palatable and productive were generally shown to decline in abundance under grazing pressure, suggesting a decline in the long-term productivity of those pastures for grazing. There was a general increase in abundance of most exotic or unpalatable grass species examined. This may have been due to lower relative palatability and release from competition as cattle preferentially graze competing native grasses. Observations that, in the presence of a reservoir of native perennial grasses, those grass species can re-colonise pastures from which they have declined supports comments by Dyksterhuis (1958) who noted that when given relief from grazing, taller grasses could again utilise their competitive advantages, thus restoring potential productivity. Several State and Transition models also indicate that a reverse transition back to a state dominated by native perennial grasses is possible (Ash *et al.* 1994, McIvor & Scanlan 1994). Relief from grazing has the greatest benefit if it follows a germination event. Experiments by Suijdendorp (1969) showed that by deferring grazing on a recently burnt pasture, germination and survival allowed a transition from a Spinifex dominated pasture to one dominated by more palatable species.

Some exotic grass species, such as *Cenchrus ciliaris*, may have the potential to dominate a site and prevent a return to a native grass community due to high levels of competitive exclusion. The role of the other common grass species in influencing community structure and composition is examined in the following chapters.

CHAPTER 5: GRAZING IMPACTS ON DIVERSITY

5.1 USE OF MEASURES OF DIVERSITY, RICHNESS AND EVENNESS

Diversity is important conceptually since it can be related to stability, maturity, productivity and spatial heterogeneity (Hill 1973). There has been considerable debate in the literature about how diversity should be defined. Although species richness is certainly an important element of diversity, the two terms are not synonymous. Washington (1984) defines diversity as a function of the number of species present (richness) and the evenness with which the individuals are distributed amongst these species. In Chapter 2, published literature relating to the value of diversity and the impact that cattle grazing has on diversity was reviewed. The present chapter aims to contribute the results of the present research to the discussion of grazing-diversity interactions by presenting original empirical data. It should be noted, however, that for the range of opinions on grazing- diversity impacts, there are also a number of ways in which diversity and its components can be measured. The literature revealed considerable debate on what constitutes the most useful measure.

5.1.1 Measures of species richness

Hill (1973) believes that most measures of diversity are an indication of how many species were present when sampling was undertaken to a certain depth of its rarities. For example, a small number of samples are only likely to contain common species, while rare species may not be encountered unless many samples are taken. The simplest measure is the comparison of numbers of species, generally regarded as ‘richness’. As illustrated in Figure 3.2, species richness increases in proportion to the size of an area sampled, so this measure should only be used when comparing equal sized plots (Ludwig & Reynolds 1988). Although a comparison of the direct count of species is simple and unambiguous (Ludwig & Reynolds 1988), Landsberg *et al*. (1997b) suggest that species richness can be a poor indicator of biological integrity since species introductions (usually increaser species) may actually increase species richness. Both Fensham & Skull (1999) and Stohlgren *et al*. (1999) measured total species richness first, before examining the richness of particular functional groups such as native and exotic. This division of species richness into functional groups is undertaken in the next chapter. Despite criticisms, the measure of species richness and derivations thereof are widely used in ecological research and were examined for their usefulness in the present chapter. In the present study, five direct measures of diversity were used.

5.1.2 Diversity indices

The term 'diversity' often combines a measure of richness (total number of species) and evenness (relative abundance) (Ludwig & Reynolds 1988). Diversity indices were created because the combination of the number of species and their relative abundances in an index relates better to other properties in the plant community than does species richness by itself (Hulbert 1971, Washington 1984).

Interpretation of diversity indices is the source of debate. For example, it is possible for two sites to have the same index value where one site has low richness and high evenness, while the other has high richness and low evenness (Hulbert 1971, Ludwig & Reynolds 1988). By combining the two measures together into a single index, it is impossible to determine the relative importance of one over the other (Ludwig & Reynolds 1988). Smith & Wilson (1996) note that the most important criteria for an evenness index is that it should be independent of species richness. Diversity indices also vary in their relative sensitivity to rare species. Many diversity measures differ only in their tendency to include or ignore relatively rare species.

5.2 METHODS

5.2.1 Measures of species richness

The five direct measures of richness used to compare grazing treatments were:

- Total number of species (richness): All species located in a 50 X 50m plot, combining quadrat data with the five minute timed search. Using this measurement, Fensham & Skull (1999) found no significant difference between grazing treatments. Stohlgren *et al.* (1999) did find a significant difference in their grazing research in the United States, however, the direction of the significant result varied between sites.
- Total number of species in quadrats: All species identified within the quadrats. This measurement excludes rare species only located during timed searches. Most researchers do not include an additional search for rare species, and so the

total number of species recorded within quadrats is their only measurement of species richness.

- Mean number of species per quadrat: Mean number of species found in a single $0.25m^2$ quadrat. This is a measurement of species richness at the finer scale of community structure. McIvor (1998) found average species per quadrat increased with increased stocking rate, but not significantly. Chaneton & Facelli (1991) found grazing increased species richness at these finer scales but reduced richness at the larger (plot) scale. Stohlgren *et al.* (1999) found ungrazed plots had a lower richness than grazed plots, although it should be noted that the quadrats used in that study were four times as large as those used in the present research.
- Number of rare species: Total number of species observed within the quadrats that had an abundance of less than 2 records. Rare species are often excluded from analyses due to the influence they can have on various diversity indices and statistical tests. For example, Fensham & Skull (1999) only analysed species with a minimum frequency of three records. In contrast, Stohlgren *et al.* (1998) compared species that had less than 1% cover, stating that rare species were an important component of plant diversity and biodiversity.
- % of total site richness: The contribution a single 50 X 50m plot makes to the combined species richness of all plots from all treatments at that site. Although no example of this measurement being used was found in the literature, it was considerd interesting to see how different grazing treatments resulted in different percentages of the total site species richness being expressed in an individual plot.

These direct measures of richness were analysed using t-tests.

5.2.2 Diversity indices

Washington (1984) gives a useful overview of numerous diversity and similaritity indices, including rarely used measures such as Kothe's "species defecit" index, Menhinick's index, Motomura's geometric progression and Yule's characteristic. An examination of the literature resulted in six diversity indices being selected to explore trends in differences of diversity between grazing treatments. These formulas generally explore trends in number of species rather than the frequency of each species.

Descriptions of the use and derivation of these diversity indices are presented below. A summary of the diversity indices, their formulas and interpretation is provided in Table 5.1. The diversity indices were compared between grazing treatments using a Mann-Whitney U test to establish significance of differences.

Shannon-Wiener's H': The Shannon-Weiner diversity index (H') is the most widely used index in community ecology (Ludwig & Reynolds 1988). Authors who include a measurement of the Shannon-Weiner diversity index include Kwiatkowska & Symonides (1986), Putman *et al*. (1989), Mwalyosi (1992), Chaneton & Facelli (1991), Fensham & Skull (1999) and Stohlgren *et al*. (1999). This index is a measure of variability or uncertainty in predicting to what species an individual plant chosen at random from a population will belong (Ludwig & Reynolds 1988). In a perfectly 'even' community where the number of individuals per species is equal, the evenness index (H_{max}) would be 1 (Barbour *et al*. 1987). The formula for the Shannon-Weaver diversity index is:

$$H' = -\sum_{i=1}^{S} (p_i \ln p_i)$$

where p_I = relative cover of species$_I$, s= total number of species

Brillouin Index (HB): Ganter (2000) regards the Brillouin Index as being similar to Shannon-Wiener's H', however, the Shannon-Wiener index assumes that samples were collected at random so the Brillouin Index is used when the randomness of the sample is suspect. The value HB can increase as the total number of individuals in the sample increases, even if the number of species or the their relative proportions remain unchanged (Ganter 2000). The formula for the Brillouin Index is:

$$HB = \frac{\ln(N!) - \sum_{i=1}^{S} \ln(n_i!)}{N}$$

where: N= total abundance of all species, s= total number of species, n_i= proportion of total sample belong to the i^{th} species.

Simpson's Index ($^1/_D$): Simpson's Index has been used to test for site diversity. It tests similarities between sites by calculating the probability that any two species sampled from that community would be the same (Henderson & Seaby 1997). The Simpson's Index reflects dominance because it rests more heavily on abundant species than rare species (Barbour *et al.* 1987, Hill 1973). Washington (1984) believes that Simpson' Index is one of the best diversity indices available due to its applicability to biological systems.

Simpson's Index is usually calculated as:

$$D = \sum_{i=1}^{S} \left(\frac{n_i(n_i - 1)}{N(N-1)} \right)$$

where: s= total number of species, N= total abundance of all species, n_i= proportion of total sample belong to the i^{th} species,

There are several versions of Simpson's Index. In the present study, the index is expressed as $^1/_D$, so that the index would increase as diversity increased (Ganter 2000). In application to grazing research, neither Chaneton & Facelli (1991) nor Stohlgren *et al.* (1999) found any significant difference between grazing treatments using Simpson's Index.

Wilson's E_{var}: Smith & Wilson (1996) tested numerous indices of diversity and concluded that E_{var} was the only one without severe problems. They particularly recommend the index E_{var} where sensitivity to rare species is required. No examples in the literature could be found of grazing experiments using E_{var} to compare treatments.

The formula E_{var} is expressed as:

$$\mathbf{E_{var} = 1\text{-}2/p\ arctan\ \{ \overset{s}{\underset{s=1}{S}} (ln\ (x_s) - \overset{s}{\underset{t=1}{S}}\ ln(x_t)/\ S)^2\ /S\}}$$

Where: S= total number of species, x_s = abundance of the s^th^ species

Pielou's *J'* (Evenness/ Equitability): Equitability or evenness refers to the pattern of distribution of the individuals between the species (Henderson & Seaby 1997), determined by the ratio of observed diversity to maximum diversity (Hulbert 1971). Evenness is an important component of diversity, since Hulbert (1971) notes that an increase in evenness can lead to an increase in diversity, even if species richness declines. Pielou's *J'* index is the commonly used measure of evenness (Washington

1984) and compares the observed Shannon-Wiener index against that of a hypothetical community with the same number of species but with perfect evenness (Pielou 1975). The formula for evenness is therefore expressed as:

$$J' = \frac{H'}{H_{\max}}$$

Where: H'= Shannon-Wiener's index, H'_{MAX}= Maximum value of H' (H'_{MAX} =ln S where S = total number of species).
Although Kwiatkowska & Symonides (1986) note that Pielou's J' index is sensitive to quadrat size in a species-poor community with clumped distribution, there was no reason to believe that such a scenario existed in the present research.

Dominance: The Community Dominance Index (McNaughton & Wolf 1970) is the percentage contribution made by the two most common species to the total observed frequency of that plot. Intuitively, the greater a proportion of a community is comprised of few dominating species, the lower the diversity will be for that community. The Community Dominance Index (McNaughton and Wolf 1970) is expressed as:

$$\textbf{Dominance } (C) = 100 \times \frac{(Y_1 + Y_2)}{Y}$$

where: Y_1 = abundance of most abundant species, Y_2 = abundance of second most abundant species, Y = total abundances for all species.

A summary of these diversity indices, their formulas and interpretation is provided in Table 5.1 below.

Table 5.1: Summary of diversity indices, mathematical formulas and interpretation.

INDEX	FORMULA	Summary description
Shannon-Wiener's H'	$H' = -\sum_{i=1}^{S} \left(p_i \ln p_i \right)$	High *H*' values indicate a high sample uncertainty and therefore a high diversity (random samples).
Brillouin Index (HB)	$HB = \dfrac{\ln(N!) - \sum_{i=1}^{S} \ln(n_i!)}{N}$	High *HB* values indicate a high sample uncertainty and therefore a high diversity (non-random samples).
Simpson's Index ($^1/_D$)	$D = \sum_{i=1}^{S} \left(\dfrac{n_i(n_i - 1)}{N(N-1)} \right)$	Probability of equal samples: calculated as $^1/_D$ so the index will increase as diversity increases
Wilson's E_{var}	$E_{var} = 1-2/\pi \arctan\{\sum_{s=1}^{s} (\ln(x_s) - \sum_{t=1}^{s} \ln(x_t)/S)^2/S\}$	Best diversity index sensitive to rare species. Value of 0 is minimum evenness and 1 is the maximum.
Evenness (Pielou's J')	$J' = \dfrac{H'}{H_{max}}$	Compares *H*' index to value for a hypothetically even community. High values are maximum evenness
Dominance (McNaughton's C)	$\text{Dominance } (C) = 100 \times \dfrac{(Y_1 + Y_2)}{Y}$	Higher values indicates more of the community is dominated by 2 most common species, plot is less diverse.

5.2.3 Species accumulation curves

Species richness is related to the size of the area being examined. The number of species encountered is proportional to a power of the area sampled (Hill 1973), described by the equation

$$S \propto A^{z}$$

Where S = no. spp, A = area sampled, z= an empirical constant between 0.1 and 0.4.

Collins & Barber (1985) suggest using quadrat data to compare species-accumulation curves between plots, hence species recorded in the timed searches were excluded. The species accumulation curves were calculated using the software package 'Species Diversity and Richness' (Henderson & Seaby 1997) and show the relative value for the measure 'Total number of species in quadrats' changing with increasing number of quadrats examined.

This program allows for the accumulation of species with increased sampling effort to be randomly shuffled between runs. If the number of runs is set to 1 then this produces the most simple species accumulation curve for the data, however, if the number of runs is increased to 10, then the samples are shuffled 10 times and the average calculated to remove the effect of sample order on the species accumulation curve and to produce a smoothed curve (Henderson & Seaby 1997).

5.3 RESULTS: EFFECTS OF GRAZING ON SPECIES DIVERSITY

5.3.1 Measures of Species Richness

The mean values and significance levels are given in Tables 5.2 a-e below. These results and the trends they represent are summarised in Table 5.3 below. In all the following tables, sites from the same soil type are grouped together in the same row.

Table 5.2.a: Comparison of grazing treatments showing mean values of total plot richness and significant values for t- tests (P).

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	50	50.7	48	NS
Khilwt	56.7	45.3	47	NS
Bluerng	50	40.3	53	**0.0054**
Leyshn,	32. 7	40. 7	39	NS
Kirk	30.3	38	35	**0.0483**
Rshtn	30. 7	38. 7	NA	NS
MtLey	31.66	29	NA	NS
Jervoise	48.66	44. 7	NA	NS
Pajing	42. 7	487	NA	NS
Lckdam	48.66	44.3	NA	NS
Epp1	42	38. 7	NA	NS
Epp2	38	50. 7	33	**0.0856**
Tabtop	70.3	45. 7	NA	**0.0000**

probability level: P <0.1

Table 5.2.b: Comparison of grazing treatments showing mean values of species richness (observed within quadrats) and significant values for t- tests (P).

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	39.3	37. 7	41	NS
Khilwt	36.3	24. 7	36	**0.0142**
Bluerng	37.3	27.3	41	NS
Leyshn,	21	31	25	NS
Kirk	18. 7	28	27	**0.0002**
Rshtn	17. 7	20	NA	NS
MtLey	20	16	NA	NS
Jervoise	37. 7	35.3	NA	NS
Pajing	27.3	38. 7	NA	**0.0884**
Lckdam	42.3	31.3	NA	**0.0738**
Epp1	21. 7	26	NA	NS
Epp2	27.3	32	27	NS
Tabtop	57	33.3	NA	**0.0007**

probability level: P <0.1

Table 5.2.c: Comparison of grazing treatments showing values of mean number of species per quadrat and significant values for t- tests (P).

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	5.4	4.4	6.7	NS
Khilwt	4.2	2.4	3.9	**0.0076**
Bluerng	3.7	2.8	4.8	**0.0352**
Leyshn,	2.8	4	4	**0.0344**
Kirk	3.1	3.5	3	NS
Rshtn	2.5	2.7	NA	NS
MtLey	6.6	1.9	NA	**0.0276**
Jervoise	5	4.2	NA	NS
Pajing	3.1	5	NA	**0.0387**
Lckdam	4.6	3.76	NA	NS
Epp1	3.4	5.2	NA	**0.0420**
Epp2	4.1	4.3	4.5	NS
Tabtop	9.7	5.78	NA	**0.0024**

probability level: P <0.1

Table 5.2.d: Comparison of grazing treatments showing mean values of number of rare species and significant values for t- tests (P).

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	21.3	21	20	NS
Khilwt	22. 7	18	25	NS
Bluerng	22	17. 7	24	NS
Leyshn,	12.3	16	14	NS
Kirk	11	17	17	**0.0499**
Rshtn	12.3	14	NA	NS
MtLey	15.3	12	NA	NS
Jervoise	21.3	22	NA	NS
Pajing	17. 7	21. 7	NA	NS
Lckdam	28.3	17.3	NA	**0.0185**
Epp1	12	12.3	NA	NS
Epp2	14.3	18. 7	11	NS
Tabtop	29.3	15.3	NA	**0.0106**

probability level: P <0.1

Table 5.2.e: Comparison of grazing treatments showing mean values of '% of total site richness' and significant values for t- tests (P).

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	47.2	48.7	45.3	NS
Khilwt	49.3	39.4	40.9	NS
Bluerng	49	39.5	52	NS
Leyshn,	23.2	34.5	28.1	**0.097**
Kirk	20.6	32.5	32.1	NS
Rshtn	24	26.7	NA	NS
MtLey	196	16.5	NA	NS
Jervoise	38.5	36.5	NA	NS
Pajing	27	38.3	NA	NS
Lckdam	39	28.9	NA	NS
Epp1	23.4	27.7	NA	NS
Epp2	26	30.5	25.7	NS
Tabtop	56.5	33.7	NA	**0.001**

probability level: P <0.1

The observed trends derived from the significant t-tests in Tables 5.2 (a-e) are summarised in Table 5.3 below.

Table 5.3: Trends of significant t-test results for direct measures of plant species richness.

(+ = higher value in grazed plot, - = higher value in ungrazed plot).

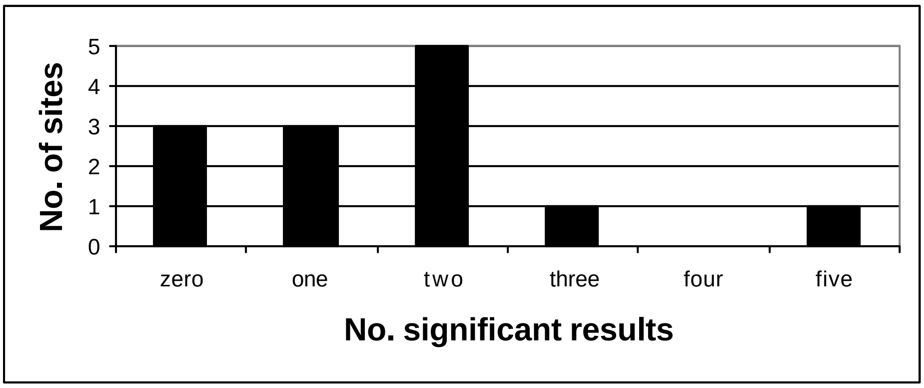
Site	Total spp No.	Total spp in quadrats	Mean spp per quadrat	Number of rare spp	% of total site richness
Khilea					
Khilwt		+	+		
Bluerng	+		+		
Leyshn,			-		-
Kirk	-	-		-	
Rshtn					
MtLey			+		
Jervoise					
Pajing		-	-		
Lckdam		+		+	
Epp1			-		
Epp2	-				
Tabtop	+	+	+	+	+

probability level: P <0.1

Table 5.3 above showed that all direct measurements of species richness gave a significant result in at least two sites and that each measurement gave a different suite of results. Results indicated that at some sites, grazing improved species richness (sites marked with a +), while on other sites, grazing caused a decline (sites marked with a -). Not all sites showed a significant result. All sites and soil groups were consistent in that they all recorded either positive or negative effects, without contradicting results. Where there were several sites representing a soil type, all significant results for those various sites were consistent with each other. Trends indicated in the table above are illustrated in greater detail in the histograms below.

The number and variety of richness measurements for which a significant result was recorded may suggest the strength of those trends. There was a broad range in the number of significant results recorded for each site. Figure 5.1 below shows the number of sites that recorded significant results from the five direct measures of species richness.

Figure 5.1: Frequency of significant results recorded for measures of species richness.

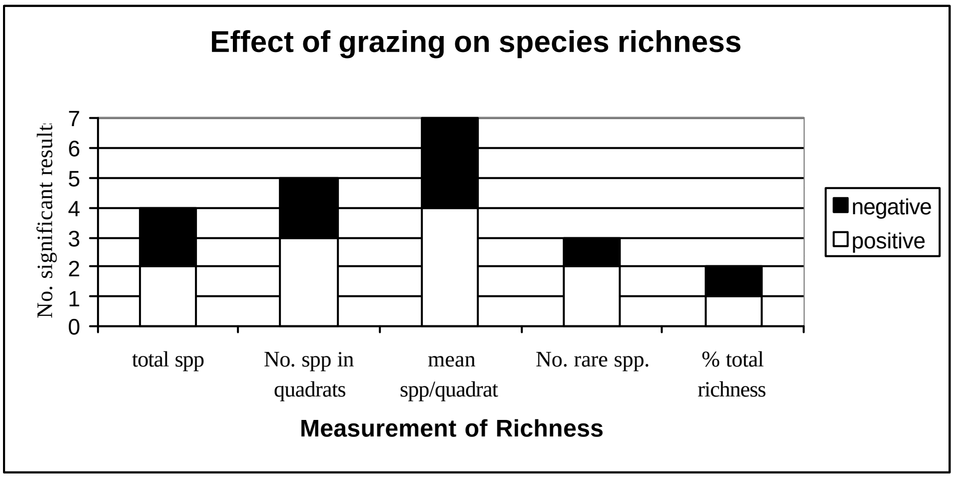


probability level: P <0.1

It is evident from Figure 5.1 above that three sites showed no significant results, three sites had one result, five sites had two results, one site had three results and one site had a significant result for all five direct diversity measurements.

Different measurements of species richness had more significant results than others, and each contributed differently towards trends on different sites. Figure 5.2 below shows the number of significant results obtained for the different measures, and the relative direction of the observed trends.

Figure 5.2: Significant results obtained for measures of species richness and the effect of grazing on richness indicated by those values.
(negative= diversity declines with grazing, positive= diversity increases with grazing)



probability level: P <0.1

Of the 13 sites examined, it can be seen that comparing the mean number of species per quadrat gave the highest number of significant results (7 sites). Measurements of ‘total number of species’ and ‘% of total richness’ were equally divided between positive and negative results, while other measurements recorded more positive results than negative.

Total species richness: The total number of species located within a 50 X 50m plot varied between 24 and 72 species. Grazed plots had significantly higher richness than ungrazed plots at Blue Range and Tabletop, while ungrazed plots had higher values at Kirk River and Epping (2). The values for macropod exclosures were usually intermediate between the means for the other treatments.

Total species in quadrats: The number of species recorded within quadrats thrown in a 50 X 50m plot varied from 12 to 61 species. Values were significantly higher in grazed plots at Kangaroo Hills West, Lucky Downs dam and Tabletop, while ungrazed plots had higher values at Kirk River and Pajingo. Values for the macropod exclosures at Kangaroo Hills West and Kirk River confirmed the trend on those sites.

Mean species per quadrat: The mean number of species recorded per $0.25m^2$ quadrat varied from 1.83 to 10.7. Values were significantly higher in grazed plots at Kangaroo Hills West, Blue Range, Mt Leyshon and Tabletop, while ungrazed plots had higher values at Leyshon View, Pajingo and Epping (1). At Mt Leyshon, there were significantly higher values in grazed plots than ungrazed, and higher values in intermittently grazed plots than ungrazed.

Number of rare species: The number of species recorded in the 30 quadrats with an observed frequency of less than two records ranged from 8 to 29 species. Values were significantly higher in grazed plots at Lucky Downs dam and Tabletop, while ungrazed plots had significantly higher values at Kirk River.

Percentage of total site richness: Individual research plots contained between 12.37 % and 61.74 % of the total site richness. Significantly higher percentages were found in grazed plots at Tabletop, while ungrazed plots had higher values at Leyshon View.

5.3.2 Diversity Indices

A total of six different indices were used to measure community diversity at the 13 sites. Comparisons between grazing treatments were made using a Mann-Whitney U test. The mean values of those indices and their respective significance levels are given in Tables 5.4 (a-f) below. These results and the trends they represent are summarised in Table 5.5 below.

Table 5.4.a: Comparison of diversity of grazing treatments showing mean values of Shannon-Wiener diversity index (H') and significant values for Mann-Whitney U tests. (**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	1.4076	1.372	1.443	NS
Khilwt	1.3496	1.1403	1.287	**0.0000**
Bluerng	1.4173	1.238	1.458	NS
Leyshn,	1.043	1.301	1.135	NS
Kirk	0.9506	1.18	1.225	**0.0000**
Rshtn	0.896	0.9823	NA	NS
MtLey	0.974	0.8633	NA	NS
Jervoise	1.3386	1.333	NA	NS
Pajing	1.1896	1.3706	NA	NS
Lckdam	1.4196	1.2996	NA	**0.1090**
Epp1	1.12	1.2033	NA	NS
Epp2	1.2286	1.379	1.316	NS
Tabtop	1.528	1.265	NA	**0.0000**

probability level: P <0.1

Table 5.4.b: Comparison of diversity within grazing treatments showing mean values of Simpson's Index ($^1/_D$) and significant values for Mann-Whitney U tests.

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	21.29	18.52	25	NS
Khilwt	17.82	9.53	13.95	**0.0000**
Bluerng	26.78	14.56	25.17	NS
Leyshn,	8.44	15.33	9.68	NS
Kirk	6.0	9.55	11.67	**0.0000**
Rshtn	5.58	6.76	NA	NS
MtLey	6.02	4.61	NA	NS
Jervoise	15.76	16.51	NA	0.8149
Pajing	12.11	17.95	NA	NS
Lckdam	19.93	16.74	NA	0.2489
Epp1	10.12	13.81	NA	NS
Epp2	13.80	20.30	17.39	NS
Tabtop	25.48	13.25	NA	**0.000**

probability level: $P < 0.1$

Table 5.4.c: Comparison of diversity within grazing treatments showing mean values of Wilson's E_{VAR} and significant values for Mann-Whitney U tests.

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	0.555	0.602	0.506	NS
Khilwt	0.596	0.614	0.542	NS
Bluerng	0.632	0.598	0.631	NS
Leyshn,	0.559	0.588	0.491	NS
Kirk	0.477	0.570	0.642	**0.000**
Rshtn	0.494	0.500	NA	NS
MtLey	0.566	0.546	NA	NS
Jervoise	0.556	0.588	NA	NS
Pajing	0.583	0.550	NA	NS
Lckdam	0.598	0.575	NA	NS
Epp1	0.539	0.439	NA	NS
Epp2	0.523	0.602	0.536	**0.000**
Tabtop	0.505	0.502	NA	NS

probability level: $P < 0.1$

Table 5.4.d: Comparison of diversity within grazing treatments showing mean values of Brillouin Index and significant values for Mann-Whitney U tests.

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	2.91	2.79	3.03	NS
Khilwt	2.74	2.23	2.59	**0.000**
Bluerng	2.84	2.46	2.98	NS
Leyshn,	2.09	2.66	2.34	NS
Kirk	1.94	2.39	2.45	**0.000**
Rshtn	1.79	1.97	NA	NS
MtLey	1.9	1.68	NA	NS
Jervoise	2.75	2.70	NA	NS
Pajing	2.38	2.81	NA	NS
Lckdam	2.88	2.64	NA	NS
Epp1	2.28	2.53	NA	NS
Epp2	2.52	2.79	2.66	NS
Tabtop	3.24	2.65	NA	**0.000**

probability level: $P < 0.1$

Table 5.4.e: Comparison of diversity within grazing treatments showing mean values of Pielou’s Evenness Index (J’) and significant values for Mann-Whitney U tests.

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	0.883	0.871	0.895	NS
Khilwt	0.8653	0.8216	0.827	**0.000**
Bluerng	0.9046	0.862	0.904	NS
Leyshn,	0.7943	0.8733	0.812	NS
Kirk	0.7476	0.8153	0.856	**0.000**
Rshtn	0.722	0.7553	NA	NS
MtLey	0.7483	0.7210	NA	NS
Jervoise	0.850	0.8613	NA	NS
Pajing	0.8323	0.8636	NA	NS
Lckdam	0.873	0.8733	NA	NS
Epp1	0.8166	0.8456	NA	NS
Epp2	0.8473	0.903	0.919	NS
Tabtop	0.883	0.8306	NA	NS

probability level: $P < 0.1$

Table 5.4.f: Comparison of diversity within grazing treatments showing mean values of McNaughton's Dominance Index and significant values for Mann-Whitney U tests.

(**Bold** = significant value) (NS= Not Significant, NA= Not Applicable)

Site	Grazed	Ungrazed	Exclosure	P
Khilea	21.53	27.2	14.9	NS
Khilwt	24.96	42.5	30.2	**0.000**
Bluerng	18.53	30.8	18.75	**0.000**
Leyshn,	45.3	29.8	36.9	NS
Kirk	55.47	39.3	35.2	**0.000**
Rshtn	55.9	46.97	NA	NS
MtLey	60.17	60.2	NA	**0.000**
Jervoise	29.3	28.8	NA	NS
Pajing	38.2	25.02	NA	NS
Lckdam	23.8	28.2	NA	NS
Epp1	41.8	27.1	NA	NS
Epp2	33.2	22.2	21.5	NS
Tabtop	18.7	29.9	NA	**0.000**

probability level: P <0.1

Results of these tests are summarised in Table 5.5 below. Symbols (+ or -) are used to show where the highest values occur. In each of the indices used, a higher value indicates a higher diversity, with the exception of dominance where the opposite is true.

<u>Table 5.5</u>: Trends observed using significant Mann-Whitney U-test results for differences of diversity indices

(+ = higher value in grazed plot, - = higher value in ungrazed plot).

Site	Shannon-Wiener's H'	Simpson's Index ($^1/_D$)	Brillouin Index	Evenness (Pielou's J')	Dominance	E_{Var}
Khilea						
Khilwt	+	+	+	+	-	
Bluerng					-	
Leyshn						
	-	-	-	-	+	-
Kirk						
Rshtn						
MtLey						
Jervoise						
Pajing						
Lckdam						
Epp1						
Epp2						-
Tabtop	+	+	+	+	-	

probability level: P <0.1

Table 5.5 above indicates that there were fewer significant results obtained using diversity indices than with direct measures of diversity. As with Table 5.3 previously, there were no contradictory results for any of the sites, considering that a high dominance value in grazed plots is entirely consistent with a high diversity index value in ungrazed plots. These results are further illustrated in Figures 5.3 and 5.4 below.

Figure 5.3: Number of significant results recorded using t-tests of direct measures of diversity.

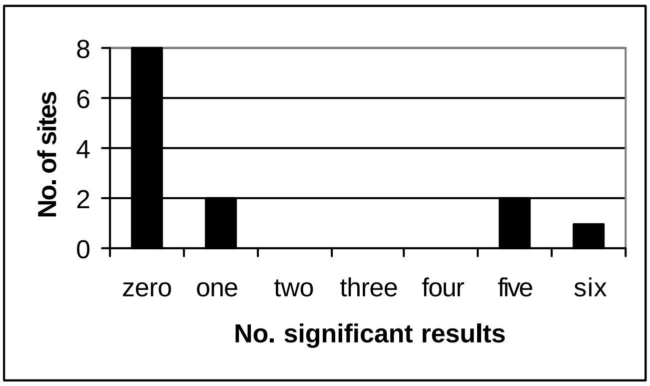


Figure 5.3 above reveals that, using t-tests of direct measures of diversity, eight sites showed no significant effect of grazing on diversity. Of the six measures of diversity used, two sites had a significant result with one aspect of diversity, two sites had five significant results and one site had six significant results.

Figure 5.4: Significant results obtained using diversity indices and the effect of grazing on diversity indicated by those values.
(negative: grazing reduced diversity, positive: grazing increased diversity)

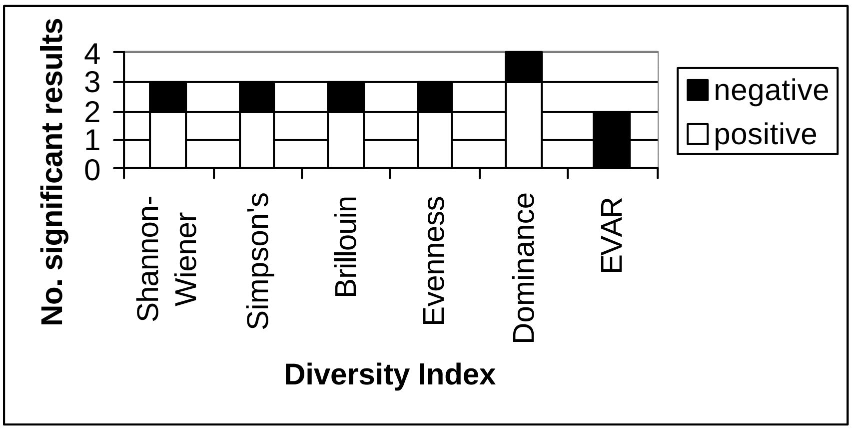


Figure 5.4 above shows the variance of the six diversity indices over the range of sites. This variance ranged from two sites having a significant difference in E $_{VAR}$ index values to four sites with a significant difference in dominance values. The E $_{VAR}$ index

only showed results indicating a negative response of diversity to grazing, while the other five indices all showed mixed responses. Five indices showed more sites to have an increased (positive) diversity under cattle grazing.

Mt Leyshon sites contrasted intermittent (dry season) grazing with the treatments of grazed and ungrazed. At Mt Leyshon (1), three indices (Simpson's, Brillouin & Pielou's evenness) showed that intermittent grazing had higher diversity than ungrazed plots. McNaughton's Dominance index values were significantly higher in ungrazed plots than in intermittently grazed plots.

Results for the individual diversity indices are discussed below.

Shannon- Wiener's H': Values for *H*' ranged from 0.753 to 1.543, with mean values ranging from 0.896 to 1.528. Significantly higher values (indicating highest diversity) were found in grazed plots at Kangaroo Hills West and Tabletop, while Kirk River had higher *H*' values in ungrazed plots.

Simpson's Index ($^1/_D$): Values for Simpson's Index ranged from 3.10 to 32.78, with mean values ranging from 4.61 to 26.78. Significantly higher values (indicating highest diversity) were found in grazed plots at Kangaroo Hills West and Tabletop, while Kirk River had higher values in ungrazed plots. At Mt Leyshon, intermittently grazed plots had higher values than ungrazed plots.

Brillouin Index (*HB*): Values for the Brillouin Index ranged from 1.37 to 3.27, with mean values ranging from 1.68 to 3.24. Significantly higher values (indicating highest diversity) were found in grazed plots at Kangaroo Hills West and Tabletop, while Kirk River had higher values in ungrazed plots. At Mt Leyshon, intermittently grazed plots had higher values than ungrazed plots.

Evenness (Pielou's *J'*): Values for Evenness ranged from 0.657 to 0.933, with mean values ranging from 0.722 to 0.919. Significantly higher values (indicating highest diversity) were found in grazed plots at Kangaroo Hills West and Tabletop, while Kirk River had higher values in ungrazed plots. At Mt Leyshon, intermittently grazed plots had significantly higher values than ungrazed plots.

McNaughton's Dominance (*C*): Values for Dominance ranged from 14.9 to 69.1, with mean values ranging from 14.9 to 60.2. Significantly higher values (indicating highest dominance and potentially lower diversity) were found in grazed plots at Kirk River, while Kangaroo Hills West, Blue Range and Tabletop had higher values in ungrazed plots. Ungrazed plots at Mt Leyshon had higher dominance than intermittently grazed plots.

Wilson's E_{VAR}: Values for E_{VAR} ranged from 0.409 to 0.707, with mean values ranging from 0.439 to 0.642. Significantly higher values (indicating highest diversity) were found in ungrazed plots at Kirk River and Epping (2). No sites had significantly higher values in grazed plots.

5.3.3 Species-accumulation curves

The species accumulation curves shown in Figures 5.5 (a-d) represent sites that showed a significant difference in the total number of species recorded within the 30 quadrats. Figures 5.5 (a -c) show changes in species diversity with increasing area where richness is significantly higher in grazed plots. Figures 5.5 (d & e) show sites where higher richness values occurred in ungrazed plots.

Figure 5.5.a: Plant species accumulation curve for plots measured at Kangaroo Hills West

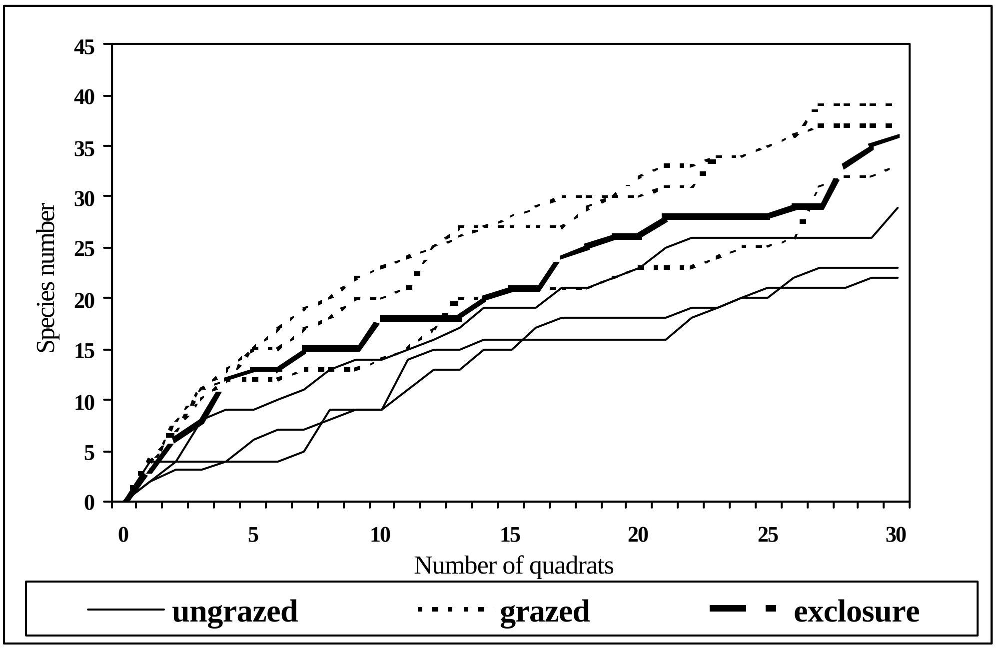


Figure 5.5.b: Plant species accumulation curve for plots measured at Tabletop

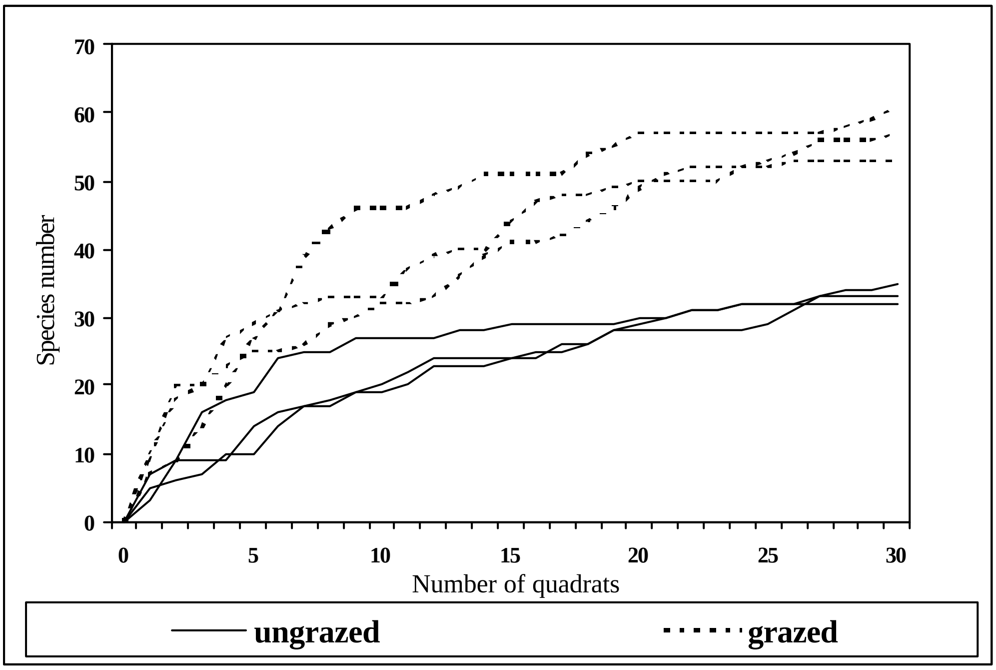


Figure 5.5.c: Plant species accumulation curve for plots measured at Lucky Downs dam

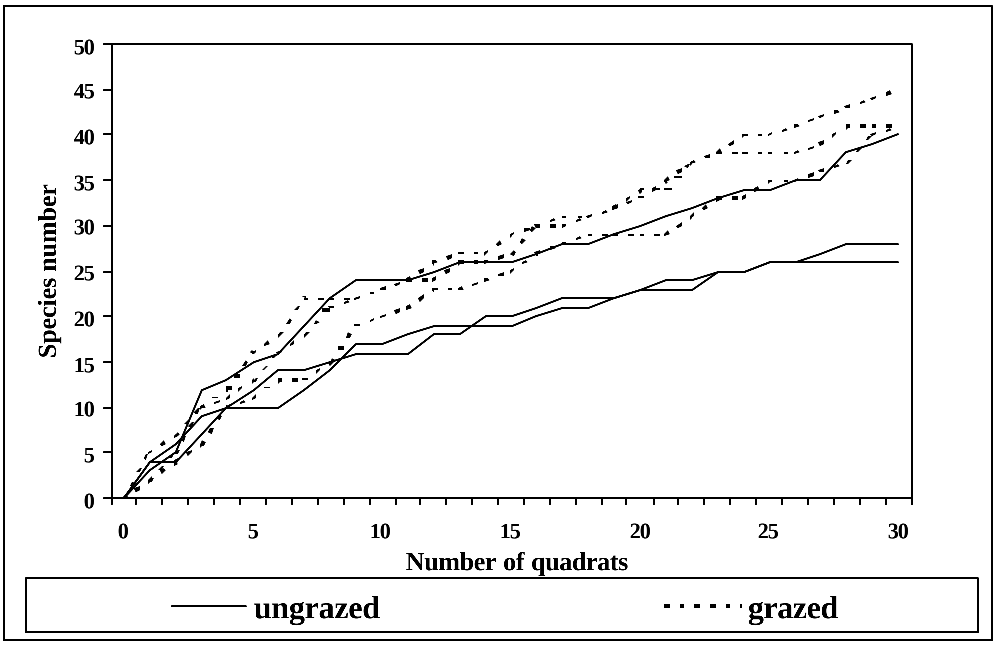


Figure 5.5.d: Plant species accumulation curve for plots measured at Kirk River

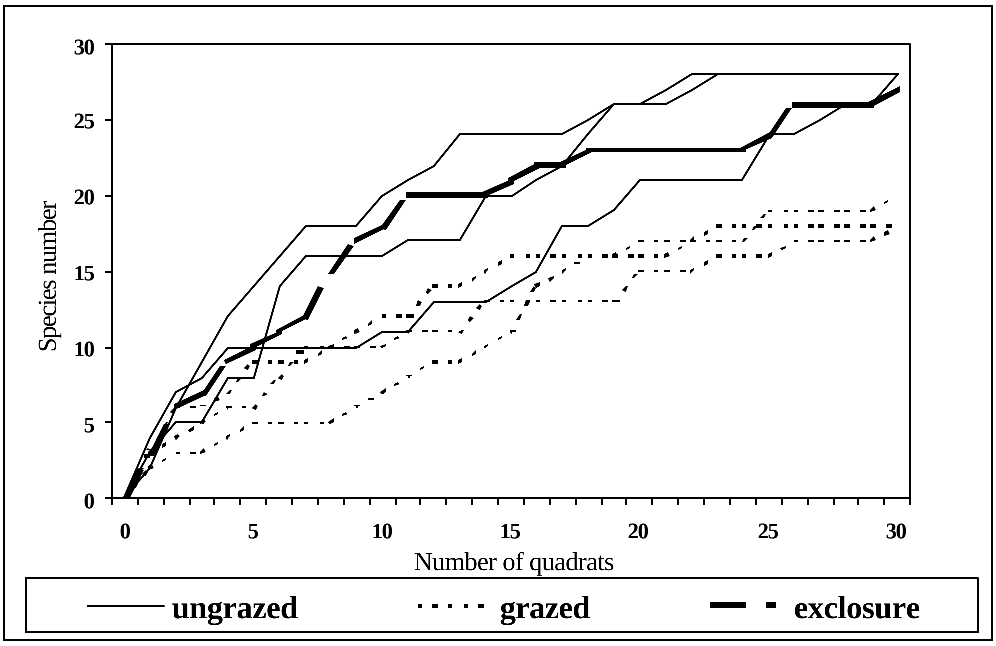
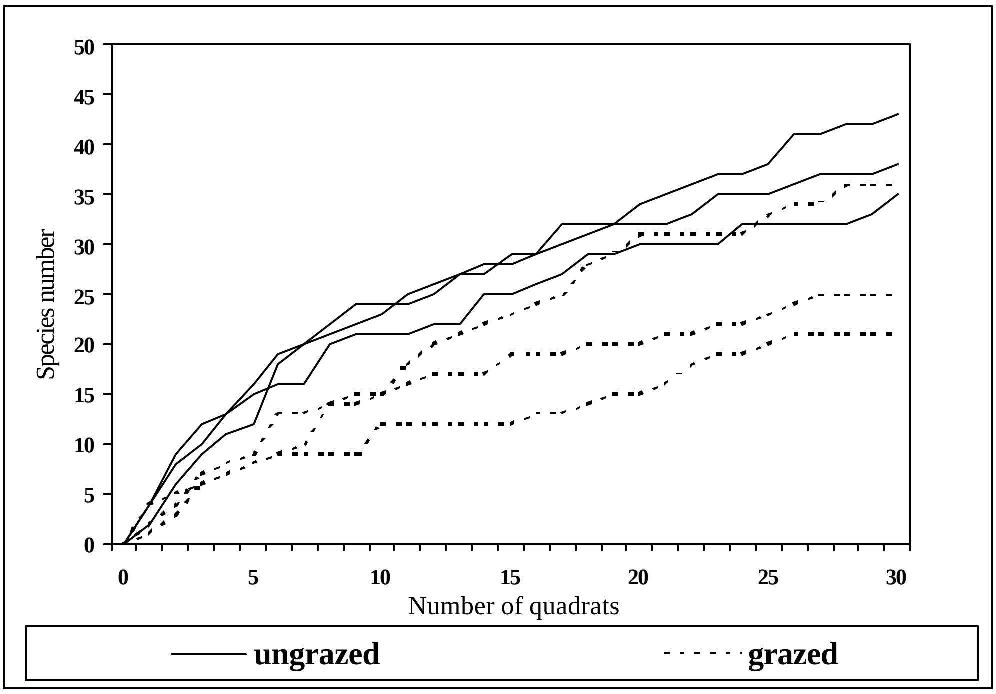


Figure 5.5.e: Plant species accumulation curve for plots measured at Pajingo



Figures 5.5 (a-e) illustrate the two significant trends observed in species richness data. By looking at the overlap between curves of different plots, it can be seen that the differences in species richness seen on Lucky Downs dam (Figure 5.5 c) were significant using 30 quadrats, however, if a smaller sample size was used (eg. <10 quadrats), such trends would not have been significant. In contrast, the species richness

values seen at Tabletop (Figure 5.5 b) and Pajingo (Figure 5.5 e) were significantly different between treatments independent of scale. At Kirk River (Figure 5.5 d), the macropod exclosure was most similar to ungrazed plots. At Kangaroo Hills West (Figure 5.5 a), the macropod exclosure had consistently higher values than ungrazed plots but at the 26th quadrat, diversity in that plot suddenly jumps to give a value more consistent with grazed plots.

5.3.4 Summary of results

The relationship between grazing treatment and simple measurements of community diversity has been thoroughly explored in the present chapter, using direct measures of diversity, diversity indices and the relationship between species richness and area sampled. In Table 5.6 below, a summary is provided comparing the results from the different measures. Those results were compared to the prediction by Harper (1969) that diversity would increase if the dominant grass was palatable, but would decrease if the dominant were relatively unpalatable. From Table 5.6, it can be seen that more sites had significant results using direct measures of diversity than with diversity indices. Some sites recorded large numbers of significant results (10), while three other sites showed no significant results. The majority of sites showed significance with only one or two measures of diversity. This possibly indicates that the impacts of grazing on diversity at these sites were small.

Table 5.6: Frequency of the significant results of comparison of direct measures of diversity and diversity indices, compared to predictions of diversity- grazing response based on Harper (1969).

(+ = higher diversity in grazed plot, - = higher diversity in ungrazed plot, / = not significant, C = correct prediction, X = incorrect prediction)

SITE	DIRECT MEASURES	DIVERSITY INDICES	PREDICTED RESULT	PREDICTION CORRECT?
K. HILL EAST	**/**	**/**	+	/
K. HILL WEST	**+ (2)**	**+ (5)**	+	C
BLUE RANGE	**+ (2)**	**+ (1)**	+	C
LEYSHON	**- (2)**	**/**	-	C
KIRK RIVER	**- (3)**	**- (6)**	-	C
RISHTON	**/**	**/**	-	/
MT LEYSHON	**+ (1)**	**/**	-	X
JERVOISE	**/**	**/**	-	/
PAJINGO	**- (2)**	**/**	-	C
LUCKY DAM	**+ (2)**	**/**	+	C
EPPING 1	**- (1)**	**/**	-	C
EPPING 2	**- (1)**	**- (1)**	-	C
TABLETOP	**+ (5)**	**+ (5)**	+	C

Five sites showed an increase in diversity with grazing by one or more measures while five sites showed a decrease. Three sites showed no significant change in diversity with grazing treatment.

Where multiple significant results were obtained for a particular site, the trends they indicated complimented each other. Note that where multiple sites on a single soil type show significant results, those sites all show the same trends as each other.

5.4 DISCUSSION

It is evident from the data presented that there was no single unifying statement that could be made on the effects of cattle grazing on species diversity. On some soil types, diversity increased, on others it decreased, while on other sites there was no significant difference. By comparing observed results with the type of grasses dominating that site (*Heteropogon contortus*, *Bothriochloa pertusa* or *Cenchrus ciliaris*), it can be seen that in nine of the thirteen sites, the response of diversity to grazing could be accurately predicted using the theory by Harper (1969) that the palatability of the dominant grass dictates community response to grazing. One site (Mt Leyshon) showed a result that contradicted the prediction when grazed and ungrazed plots were compared. It should be noted that there was only a single result obtained from the eleven measures of diversity, so the accuracy of that conflicting trend could be doubtful, and may be due to a type 1 error (due to chance).

Three sites (Kangaroo Hills East, Jervoise and Rishton) showed no significant result in diversity. It may be possible that declines in some species or functional groups were countered with an increase of other species. This possibility is explored in greater detail in Chapter 6. An alternative explanation for the result at Kangaroo Hills East was that grazing pressure was not sufficient to cause a significant alteration to community diversity, since that site did not record any significant differences in the abundance of any of the dominant grasses explored in Chapter 4 previously. In finding no significant difference in species richness between grazing treatments, Fensham & Skull (1999) also suggested that grazing pressure had not been strong enough to significantly alter abundance of the native grasses that dominate the ecosystem. At Jervoise, grazing changed the composition of the dominating grass from *Themeda triandra* in the ungrazed plots to *Bothriochloa pertusa* in the grazed plots. Although the appearance of these two treatments were visually very different, it is possible that the replacement of one diversity-suppressing dominant grass with another one may have resulted in no net increase in richness or diversity. In ungrazed plots, the dominant but palatable *Themeda triandra* may be suppressing other species through intense competition for light and other resources, while in the grazed areas; cattle may be suppressing diversity through selective grazing on species that are rarer but more palatable than the dominating *Bothriochloa pertusa*. Thus, conditions exist in both grazing treatments for the suppression of co-occurring species. Such a scenario is supported by comments by Harper (1969), McIvor (1998) and Olff & Ritchie (1999).

Plots at Mt Leyshon clearly show that both continuous and intermittent grazing had higher diversity than ungrazed plots. These results contrast with results from North America (Hart 1999), which showed that species richness was the same in ungrazed and lightly grazed, higher in moderately grazed pastures and declined at the heaviest stocking rates. This contrast may be explained by differences in community composition and structure. Of particular interest is the significantly higher value for dominance in the ungrazed plots at Mt Leyshon. In this instance, *Cenchrus ciliaris* was the dominating species. In light of these results, it could be suggested that *Cenchrus ciliaris* dominance had a negative impact on species diversity. This conclusion concurs with the relationship between *Cenchrus ciliaris* and species richness that was explored in depth in Chapter 4.

Usefulness of measures of diversity: Since researchers could justify the use of any of these numerous measures of diversity alone, it was a valid and interesting approach to see which of these indices would be of greatest use to a researcher attempting to elucidate trends and changes to pasture diversity. It is likely that those measures that consistently gave a result while others did not possibly have a greater sensitivity to pasture change due to differences in the parameters used in those formulas. An examination of the results shows that, generally, more significant results were obtained using direct measures of diversity than with diversity indices. Measuring the mean number of species per quadrat gave more significant results than any other direct measure of diversity. In decreasing usefulness (number of significant results), species per quadrat was followed by number of species in all quadrats, total number of species per plot and the representation of total site diversity in that plot. Decreasing usefulness occurred with the increasing scale being measured. This may indicate that grazing had a greater impact on diversity at the quadrat level than at the plot or site level.

Of the diversity indices selected, measuring dominance (*C*) gave the highest number of significant results, while the formula E_{VAR} gave the smallest number of significant results. The diversity indices Shannon- Wiener's *H'*, Simpson's Index ($^1/_D$), Brillouin Index (*HB*) and Evenness (Pielou's *J'*) all showed identical trends of changing diversity at every site, so it was not possible to distinguish between the relative benefits of using those indices. Sites such as Tabletop showed such a strong contrast between grazing treatments that most of the measures of diversity yielded significant results, with the exception of the index E_{VAR}.

It should be noted that simply providing an accurate description of the vegetation present in the various plots may be a useful tool for long term monitoring of sites as this gives impressions of the appearance of a site that may be used to indicate changing pasture conditions. Note that overall vegetation descriptions for each of the sites measured in the present research can be seen in Appendix A. While it was not possible to test the usefulness of this approach given the short term nature of this research, it is hoped that these descriptions may be used by future researchers and their usefulness in understanding grazing impacts at these sites would be assessed.

Species accumulation curves: Species accumulation curves showed that the rate of discovery of new species increased with sampling effort. The curves showed that direct measurements of species richness may, on some sites, be variable depending on the number of quadrats thrown, and, almost certainly, on the size of the quadrat. Sites that did not show a significant difference in species richness between treatments may have shown significant results with more or less quadrats. Since the number and size of quadrats used may simply be a matter of logistics, the value of the measure of species richness may be questionable. Species accumulation curves may be used to confirm the trends identified by using the measure of species richness, but ultimately convey similar information.

Differences in observed trends: Although there were three sites for which there was no significant trend, it can be seen that trends on the other sites were equally divided between grazing increasing diversity (five sites) and decreasing diversity (five sites). It should be noted that these trends were faithful to soil type. Using examples where multiple sites represented a particular soil type, it can be seen that grazing on Dalrymple soils decreased diversity, while diversity increased on Greenvale soils. Using the definitions by Scanlan *et al*. (1996a), Dalrymple soil sites were *Bothriochloa pertusa* dominated sites, while Greenvale sites were *Heteropogon* dominated. It should be noted that the aromatic nature of *Bothriochloa pertusa* makes it less palatable than *Heteropogon contortus* (O’Reagain P. pers. comm.). These results therefore confirm observations by Harper (1969) that when a dominant is palatable, grazing will increase diversity through removal of competitive exclusion, but when a dominant is unpalatable, then grazing reinforces their dominance, thus reducing diversity (Harper 1969). Two sites (Kangaroo Hills East and Rishton) that showed no significant difference in richness or other measures of diversity also showed no significant change in abundance of the dominating grasses, therefore strengthening the argument that grazing effects on dominant grasses determines ecosystem response to the grazing disturbance.

CHAPTER 6: EFFECTS OF CATTLE GRAZING ON FUNCTIONAL GROUPS

Plant communities can be examined by looking at their richness, evenness or diversity; however, examination of the species composition of those communities can reveal numerous underlying processes and changes. In Chapter 2, the concept of numerous species fulfilling similar functions was discussed. It was therefore possible to reduce the 319 species recorded within quadrats into a smaller number of functional groups that might be expected to show similar results. The present chapter investigated how species could be separated into functional groups and identified trends that were not obvious when dealing with diversity as a whole. There has been a considerable amount of literature published on selecting plant functional groups and how those groups might be expected to respond to disturbance. This literature was examined in light of the data presented in the present research.

6.1 DEFINING FUNCTIONAL GROUPS

Attempting to elucidate the complex patterns of ecological responses to grazing from a large and complex data set is difficult to say the least. Hence Friedel (1997) notes that it is "…desirable to devise functional groups to reduce the apparent complexities of fluctuating species composition". Utilising all species in an analysis can lead to confounding results, since the presence of numerous zeros in the data set can mislead certain statistical packages into assuming that sites are similar based on those absences (Sheaves M. pers. comm). From an ecological point of view, examining representativeness or representative diversity would be more satisfactory than examining diversity *per se*, hence the need for the identification of characteristic or indicator species (Dufrene & Legendre 1997). To reduce the size of the data set, eliminate zeroes and simplify the ecological understanding of trends, it was considered appropriate to categorise species into functional groups and then to analyse the data using those functional groups.

Despite the obvious advantages of classifying species into functional groups, McIntyre *et al*. (1995) warn that any generalised classification using few functional groups "…omits relevant details for the interpretation of specific patterns of diversity", so this must be taken into account in interpretation of the data. For example, simply listing all dicots as 'weeds' is inappropriate since only a few of these are actually weeds in an ecological sense (McIntyre & Filet 1997). In consideration of that cautionary advice, it was decided that the potential of using functional groups to assist in understanding the results of this research could be enhanced by looking at which species made the greatest contribution to the functional groups that showed a significant result. Prior to separating species into functional groups, an examination of the relevant literature was made, which revealed a varied approach to selecting functional groups or guilds.

The selection of traits to use in separating functional groups is not an easy one. Landsberg *et al*. (1999) note that numerous environmental filters have given rise to many different species with different attribute combinations, although few of these attributes may be associated with grazing response. Physical attributes recognised by Landsberg *et al*. (1999) include leaf density, leaf size, leaf coating (indumentum), regrowth potential, flowering period, seed size, seed dispersal traits, fecundity and plasticity (levels of trait variation). Although Landsberg *et al*. (1999) was able to describe some response groups by the presence or absence of several of these traits, it was not found to be possible to describe all these attributes for all 437 species recorded in the present research. A different approach has therefore been taken with the present research.

To pick functional groups, certain attributes are required to assign species by, but which attributes should be used? Lavorel *et al.* (1997) identified four approaches to selecting functional groups:

a) Emergent groups: groups of species that reflect natural correlations of biological attributes. This essentially relates to the major life forms (eg. trees, shrubs, grasses and forbs)
b) Strategies: species using a similar strategy that have similar attributes/adaptations to particular patterns of resource use
c) Functional types: species having similar roles in ecosystem processes by responding similarly to multiple environmental factors
d) Specific response groups: species responding similarly to specific environmental factors

Literature relating to these approaches is reviewed below.

<u>a) Emergent groups</u> Raunkiaer (1934) developed a system of defining life forms based on the stature of the plant, the position above the ground where persistent buds are

present, and life history strategy. Those functional groups were assigned labels such as Therophyte, Phanerophyte and Hemicryptophyte. McIntyre *et al.* (1995) combined Raunkiaer's life form groups with other aspects such as seed dispersal, seed morphology and vegetative reproduction to conduct their analyses. The result of those analyses showed that those reproductive traits were not relevant to grazing disturbances and that a classification based on life form was most useful (McIntyre *et al*. 1995). Landsberg *et al*. (1999) also tested the usefulness of a broad range of plant physical and reproductive traits to identify those related to grazing syndromes, but also found life form to be the best feature on which to classify grazing response groups. Life form and the position of regenerating buds may directly correlate with ability to withstand herbivory. For example, plants of higher stature have increased sensitivity to defoliation, while plants with smaller stature not only avoided herbivory but also benefited from grazing through a release from competition (Lavorel *et al*. 1997, McIntyre *et al*. 1995). To illustrate this point, Landsberg *et al*. (1999) found that for grassy communities, grasses classified as “large erect tussocks branching above ground” were responsive to light grazing, while grasses that were “small, sprawling basal tussocks” indicated heavier grazing.

<u>b) Strategies</u> The most widely publicised theoretical identification of functional groups based on strategies is the model proposed by Grime (1977), previously discussed in section 2.3.1. This model suggests that there are three primary strategies for plants, depending on their adaptation towards establishment, growth and survival under competition, stress or disturbance. Plants that are specifically resistant to disturbance are known as ‘ruderals’ (Grime 1977), hence the model is often referred to as the C-S-R model. If pasture management can be seen as the manipulation of these three factors, then Grimes C-S-R model is useful for initial predictions when there is insufficient knowledge to design detailed management strategies (McIvor 1993).

There has, however, been criticism of Grimes model, ranging from criticism of the terminology used, to claims that the whole model is misleading (McIvor 1993). Oksanen & Ranta (1992) discuss the possibilities of using Grimes C-S-R model by altering it to replace the ‘stress-tolerant’ dimension with ‘grazing tolerance’, despite the fact that, using Grimes’ definitions, grazing is a disturbance, not stress. McIvor (1993) proposes that grazing pastures should be in the centre of the C-S-R triangle, since all factors are present but none dominate, while Lavorel *et al*. (1997) see that plants with resistance to grazing require both S and R strategies.

Although Petraitis *et al*. (1989) agree that increased diversity is a result of a balance between the rate of disturbance and the rate of competitive exclusion, Westoby (1979/1980) does these are entirely opposing forces, and states that a reduction in one will not necessarily lead to an increase in the other due to deflection caused by changes in range condition such as soil structure. Secondly, while Grime (1977) sees stress tolerators as being a particular strategy group, Westoby (1979/1980) notes that there are many different growth forms and life histories that are all effective at allowing a plant species to persist under stressful conditions such as aridity. These different strategies can result in a broad range of responses to competition and biomass removal (disturbance).

c) Functional types. For plants to have a similar role in ecosystem processes and/or response to disturbance, they must share common biological attributes. Lavorel *et al*. (1997) state, “subjective growth form classifications have been repeatedly found to reflect broad ecosystem functions along environmental gradients”. Different sets of traits are needed that explore the details of the interactions between the plants and their biotic and abiotic environment (Lavorel *et al*. 1997). Such traits may include mechanisms by which species persist in the environment, including their seed bank, dispersal mechanisms and the ability to regenerate vegetatively (Lavorel *et al*. 1997).

An example of basing functional groups on persistence traits can be seen in Gill’s (1981) classification of plant functional groups based on response to fire. Although no equivalent model for the persistence of plants subject to grazing has been constructed, Landsberg *et al*. (1999) were able to identify certain physical traits associated with grazing response and persistence. They were not, however, able to find groups of traits that could successfully classify plants on their grazing response. Lavorel *et al*. (1997) note that separating functional groups on persistence traits is rarely used in grazing ecology as the various attributes necessary to make the separation are difficult to measure.

Another biological attribute that can be considered here is life history. Life history strategies can be categorised into annual, perennial and weakly perennial, which tend to respond differently to disturbance. Symonides (1988) suggests that the short life cycle of many annual species has probably formed in response to disturbance and is then favoured by natural selection. Lavorel *et al.* (1997) note that annuals and short-lived perennials are favoured by disturbance events such as grazing that open gaps in a matrix dominated by perennial species (refer back to comments on the Intermediate Disturbance Hypothesis in section 2.3). Because of the attributes possessed by annual plants, annuals generally tolerate or avoid unfavourable conditions but have a low competitive ability, so often specialise in colonising frequently disturbed areas (Symonides 1988). It is no surprise, therefore, that numerous studies have identified an increase in annuals with heavy grazing (eg. Landsberg *et al.* 1999, Fensham & Skull 1999, Friedel *et al.* 1988, Symonides 1988 and McIvor *et al.* 1995a).

<u>d) Specific response groups</u> A functional group could be defined as being comprised of species that respond similarly to a perturbation such as grazing (Friedel *et al.* 1988 and Dufrene & Legendre 1997). However, it is difficult to characterise the ecological responses of rare species due to small sample sizes and so characterising functional groups by response is difficult with rare species (McIntyre & Filet 1997). The most popular functional groups based on response to grazing are those of increaser, decreaser or neutral species. Dividing species into ‘increasers’, ‘decreasers’ or ‘invaders’ depending on whether the population of a species decreased, increased or invaded under grazing regimes was a concept first introduced by Dyksterhuis (1949). More recent literature appears to discard the ‘invader’ functional group, incorporating it into ‘increasers’, while proposing a third group, ‘neutral’. Landsberg *et al.* (1997b) defined these three major response groups:

1/ Decreasers (Intolerant): generally native species intolerant to disturbance and comprising much of the species richness of ecosystems
2/ Neutral (tolerant): native and exotic species in disturbed and undisturbed habitats, not showing any significant response to grazing
3/ Increasers (Disturbance specialists): mostly exotic species correlated with high disturbance levels

McIvor (1998) advises caution about the use of these groups since he notes wide variation in the response of species to grazing between different sites. For example, flinders grass (*Iseilema vaginiflorum*) is an increaser species in north-western Queensland but is a decreaser species in southern areas (Henry *et al.* 1995). In the present chapter, the increaser, neutral and decreaser response groups were derived from the analysis *post hoc*, rather than used to classify groups prior to analysis.

One aspect of plant biology not considered above was palatability and the way that palatability influences herbivores to preferentially target some species in preference of others. Palatability has been listed as a factor in the decline of some plant species in favour of others (McIvor 1998, Partridge 1995, Harper 1969, Isbell 1969), however, there are surprisingly few authors who comment on its use in selecting functional groups. In assessing land condition, McIvor *et al*. (1995a) clustered species into three categories of desirability to the grazing industry, though, using their criteria, species with different levels of palatability were included into each category. Noy-Meir *et al*. (1989) found that plant growth form was more important than palatability in determining if a species was an increaser or decreaser. Friedel (1997) also commented that palatability was not correlated with a grazing gradient as species compositions may have been influenced by changes in soil conditions. It should be noted that perhaps the term “palatable” could as easily be replaced by the term “preferable” to cattle. Westoby (1979/1980) not that preference for a particular species over another may also be due to:

- Preference of species with little dead foliage in proportion to live (eg the preference for tussocks that have already been grazed compared ungrazed individuals)
- Preference of species that yield the most material from a single bite (lower preference for small ephemeral plants with little biomass)
- Preference for species that continue to photosynthesise into periods of drought (eg perennials are eaten while annuals are ignored)
- Avoidance of woody species with the foliage 2 metres over the ground

Additionally, it is possible that certain species will decline, not just because they are particularly sought after or preferred by grazers but because they may be exceptionally sensitive to even light grazing

Lavorel *et al*. (1997) note that several studies combined several approaches such as growth form and function-oriented classifications. For example, ruderal plants (as defined by Grime 1977) can be characterised by a range of attributes such as short life span, small stature, abundant and widely dispersed seeds and/or a dormant seed bank. In the ‘arid rangelands model’ described by Lavorel *et al* (1997), a hierarchical approach was taken, separating plants first on life form (forbs, grasses and shrubs), then subdivided on palatability.

6.2 METHODOLOGY

6.2.1 Aims

The overall aim of the present chapter was to determine if broad generalisations could be made about the influence of cattle grazing on community composition. To analyse the impact of grazing on functional groups, species were firstly assigned into smaller assemblages of functional groups based on techniques and recommendations in the literature. Trends in the response of these groups to grazing were then examined. Significant results were examined further to determine which species made the greatest contribution, especially where the trend appears contradictory to other results. Reflective comments were made about the selection of those functional groups and advice given for further research.

6.2.2 Description of the functional groups used in the present research

In the analysis of the present research, functional groups were assigned using two approaches.

The first approach divided plant species into broad, overlapping functional groups. A similar technique was used by Friedel (1997) who assigned species into overlapping functional groups (grass, forb, sub-shrub, ephemeral, short-lived, perennial, palatable, not preferred or unpalatable), whereby a particular species could belong to 2 or more of these functional groups. In the present research, the overlapping functional groups examined were exotic species, annuals, perennials, palatable species and unpalatable species. In many cases, there was insufficient information about a species to assign it to either the category of palatable or unpalatable, although every species recorded was marked as either an annual or perennial. Weak perennials were assigned to the annual category. The factors influencing a grazers preference for a particular food are numerous and complex, ranging from growth form, chemical composition, physical defences, palatability and previous experience of the grazing animal (Malechek & Balph 1987). In these analyses, only the aspect of palatability was used to separate groups of plants. Palatable and unpalatable plant species were those identified as such in the literature. Species that could not be assigned to either category of palatability were excluded from the analysis. Other aspects of preference to grazers were not included as they were generally unavailable from the literature. While it may have been possible to group species with obvious features such as spines, Belovsky *et al*. (1991) found that spines are not necessarily a significant deterrent against herbivores. Univariate analyses (t-tests) were conducted on the abundances of these overlapping functional groups as given by frequency of occurrence within quadrats.

In the second approach, a more detailed hybrid method of classifying functional groups was taken. This approach incorporated and combined aspects of growth form, life history strategy and taxonomy with the broader categories of ‘native’ or ‘exotic’ to form 15 non-overlapping functional groups that could be examined individually using univariate analysis or, simultaneously, in a multivariate ordination approach. Growth form has been shown by McIntyre *et al*. (1995) and Landsberg *et al.* (1999) to be correlated with numerous other traits and they note that this can play a major role in community response to disturbance. Vines were separated from other life forms as it was hypothesised that this growth form may make them more susceptible to mechanical damage by trampling. In regards to life history strategy, using the primary and secondary literature describing the species collected, it was possible to assign every species to a category of annual or perennial, although there could be some possible minor errors as some species will fall into different groups depending on the prevailing conditions (Rolfe *et al*. 1997). McIntyre & Filet (1997) comment that functional groups do not assume phylogenetic relatedness of the merged taxa, however, in this research it was decided to use taxonomy in a minor role in the group separation. Examples include grasses and sedges, which are obviously all in the families Poaceae and Cyperaceae respectively, and which are considered separately, despite the fact that other authors (eg Friedel *et al*. 1988) lumped them together. Noy-Meir *et al.* (1989) noted that the only plant families that showed grazing response along taxonomic lines were grasses and legumes. Legumes were targeted as the basis for three functional groups because of their importance to the grazing industry, but were separated into exotic versus native to distinguish between the native component and those introduced for pasture improvement.

The representative species of each of the functional groups can be seen in Appendix E. Short comments on the functional groups analysed are provided below:

- Exotic Grasses (EG): Mostly perennial, this group contains 3 species that are annual, but were rarely recorded so are unlikely to affect results (9 species)
- Exotic forbs (EF): Non-woody herbaceous species (excluding legumes) that are not indigenous to QLD (19 species)
- Exotic legumes (EL): Legume species not indigenous to QLD, mostly introduced for pasture improvement (14 species)
- Exotic vines (EV): Trailing or climbing plants that are not indigenous to QLD (2 species)
- Exotic woody plants (EW): Shrubs and trees that are not indigenous to QLD. All species are declared noxious weeds. One species (*Cryptostegia grandiflora*) is a vine but grows as a shrub without support (4 species)
- 3P Native Grasses (NG3P): Indigenous grasses that are palatable, perennial and productive (16 species)
- Native grasses-perennial (NGP): This category contains all native perennial grasses other than those specifically listed in the literature as being 3P grasses. Many of these are generally regarded as unpalatable (50 species)
- Native grasses- annual (NGA): Native grasses completing their life cycle from germination to death in one season or year (37 species)
- Native forbs- annual (NFA): Native herbaceous species (excluding legumes, grasses and sedges) usually completing their life cycle from germination to death in one season or year (58 species)
- Native forbs – perennial (NFP): Native herbaceous species (excluding legumes, grasses and sedges) that live for more than one year (62 species)
- Native legumes – erect or prostrate (NL): Native legumes except trees, shrubs or vines (35 species)
- Native legumes- trailing or vine species (NLV): Native legumes that normally exhibit a trailing or climbing habit (14 species)
- Native vines (NV): Native plants (except legumes) that normally exhibit a trailing or climbing habit (16 species)
- Native woody plants (NWP): Woody plants native to QLD (79 species)
- Sedges (S): Plants of the sedge family Cyperaceae (23 species)

6.2.3 Methodology for analysis of data

Data was collected from the 13 sites, each of which had contrasting grazing treatments. ‘Grazed’ refers to plots that were continuously grazed by cattle and macropods throughout the year, while cattle grazed ‘intermittently grazed’ plots at Mt Leyshon during the dry season only. ‘Ungrazed’ plots were fenced to exclude cattle but were open to macropod grazing. At Epping Forest, the endangered Northern Hairy-nosed Wombat also grazed ‘ungrazed’ plots. The ‘macropod exclosure’ was an area fenced to exclude grazing vertebrates of all species. At each site, two treatments (grazed and ungrazed) had three replicate plots measuring 50 X 50 metres. Macropod exclosures were not present at each site and were not replicated where they were present. Within each plot, 30 0.25m^2 quadrats were systematically thrown and the species present within each quadrat recorded.

Analysis and discussion of the results are conducted under two sub-topics:

- Impact of grazing on broad plant types
- Impact of grazing on functional groups

Analysis of functional groups was undertaken with abundance data indicating the proportion of the 30 quadrats in which a functional group was observed. In assembling functional groups, species that did not achieve an abundance of 5% (3 records) in at least one of the plots were excluded in accordance with recommendations by McIntyre & Filet (1997) and Friedel *et al.* (1988). The relevant proportions of the broad functional groups were analysed using a Chi squared Multiple Proportions Test, conducted using the statistical software package Statistix version 7 using a two-sample test of proportions in calculations. The null hypothesis is that the two proportions measured are equal.

For each soil type, a Principal Components Analysis (PCA) was conducted using the multivariate statistical package CANOCO (version 4), with each PCA representing all the plots from a particular soil group. The PCAs used the indirect gradient analysis with untransformed proportional data. Species were centred and standardised. A triplot was then constructed. Ter Braak & Smilauer (1998) define a triplot as being “..an ordination diagram with 3 kinds of entities, of which all pairs form biplots”. In this case, those three entities are 1) plot 2) functional group and 3) treatment. The distance between the points on the triplot is measured in Eigenvalues, which is a scale of similarity. Hence a PCA ordination triplot is essentially a scatter plot of the eigenvalue scores for each plot, based on their similarities and differences on presence and abundance of functional groups, the influence of which is also indicated. It must be remembered that the PCA triplots are a 2-dimensional representation of a 3-dimensional data cloud. The axes shown explain part of the variance of the data cloud. Usually, the first two axes will explain the greatest part of the variance, however, there are times that other axes need to be examined to help interpret a pattern. In each of the PCA triplots presented, the first two axes were found to yield the most information, and so only those axes are illustrated.

Although the PCAs show the difference in response of the various functional groups, most importantly they illustrate the differences that exist between plots based on their floristic components. The differences between plots are based on all functional groups, not only those that yielded a significant t-test result.

Ordination methods are usually used in conjunction with a clustering method to produce clusters of species or sites. Clusters of similarly related plots were identified using hierarchical dendrograms of the plots, which were also separated by the relative abundances of the functional groups. These hierarchical dendrograms were generated using the statistical software package SPSS version 10. Interpretation of the dendrograms can be subjective in regards to where a line is drawn for separating clusters. Clusters produced using a hierarchical approach can be considered eurytopic (derived from higher level clusters) or stenotopic (derived from lower level clusters). It is considered that eurytopic clusters are more reliable and significant than stenotopic clusters (Sheaves M. pers. comm.). Deciding where to separate clusters is relatively arbitrary. In dendrograms used to identify clusters for interpretation of the PCAs, an average linkage score of 11 was used as this separated larger stenotopic branches while including enough information to make the clustering useful. The use of the PCAs and dendrograms in what is essentially data exploration also allowed inclusion of the macropod exclosures, which had to be excluded from the t-tests due to a lack of replication.

6.3. RESULTS OF ANALYSIS

6.3.1 Impact of grazing on broad plant types

Initially, the broad plant categories of exotic (introduced), annual, perennial, palatable and unpalatable were analysed using a test of multiple proportions. The results of these analyses are shown in Table 6.1 below.

Table 6.1: Results of ‘Chi^2 tests for multiple proportions’ of broad functional groups between grazing treatments ‘grazed’ and ‘ungrazed’ on the 13 sites.

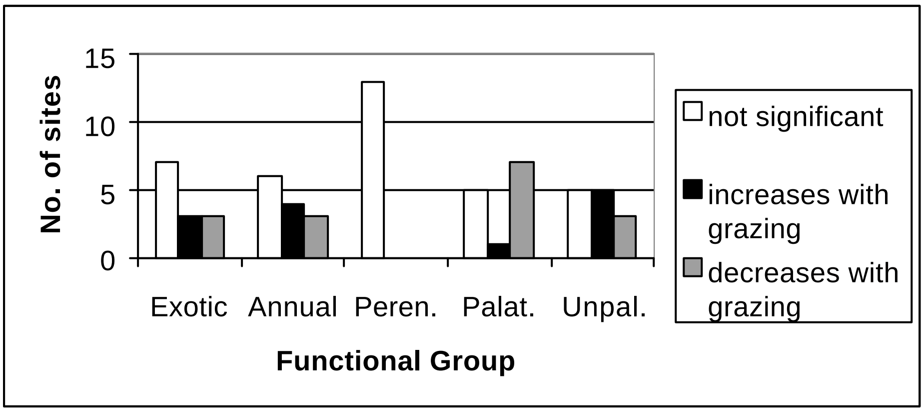
(NS = Not Significant).

SITE	Exotic	Annual	Perennial	Palatable	Unpalatable
K.HILL EAST	NS	NS	NS	NS	NS
K.HILL WEST	0. 00001	0. 0001	NS	NS	NS
BLUERANGE	0. 00001	NS	NS	0. 0003	0. 0057
LEYSHON V.	0. 0038	0. 0505	NS	0. 00001	0. 032
KIRK RIVER	NS	NS	NS	0. 0704	0. 007
RISHTON	NS	NS	NS	0. 0147	NS
MT LEYSHN	NS	0. 0012	NS	0. 00001	NS
JERVOISE	0. 00001	NS	NS	0. 00001	0. 0981
PAJINGO	NS	0. 00001	NS	0. 0002	0. 0001
LUCKYDAM	NS	NS	NS	NS	0. 0445
EPPING 1	0. 016	0. 0002	NS	NS	0. 0002
EPPING 2	NS	0. 0003	NS	0. 0002	NS
TABLETOP	0. 00001	0. 0004	NS	NS	0. 0606

probability level: $P < 0.1$

By examining Table 6.1 and Figure 6.1, it can be seen that several sites showed non-significant results for several broad functional groups and that there was a wide diversity of responses of these groups to grazing.

Figure 6.1: Histogram showing the number of sites with different abundances of broad functional groups between grazing treatments.



probability level: P <0.1

- Exotic: Three sites showed an increase in exotic species with grazing (Kangaroo Hills West, Leyshon View and Jervoise), while three sites showed a decrease (Blue Range, Epping Forest 1 and Tabletop). At Mt Leyshon (2), grazed plots had a higher abundance of exotics than did intermittently grazed, due mostly to the dominance of *Cenchrus ciliaris*. However, there was no significant difference in abundance of exotics between grazed and ungrazed plots. There did not appear to be any soil-related trends.
- Annual: Four sites showed an increase in abundance of annuals with grazing (Kangaroo Hills West, Mt Leyshon, Epping Forest 2 and Tabletop), while three sites showed a decrease (Leyshon View, Pajingo, Epping Forest 1).
- Perennial: No significant results were found comparing grazed and ungrazed plots. At Mt Leyshon (1), ungrazed plots had a higher frequency of perennial species than intermittently grazed plots.
- Palatable: Seven plots showed a decrease in abundance of palatable species with grazing (Blue Range, Leyshon View, Kirk River, Mt Leyshon, Pajingo, Jervoise and Rishton). Epping Forest (2) showed an increase in palatable species with grazing.
- Unpalatable: Five sites showed an increase in abundance of unpalatable species with grazing (Blue Range, Kirk River, Pajingo, Jervoise, Tabletop), while three sites showed a decrease (Leyshon View, Lucky Dam, Epping Forest 1). At Mt Leyshon 1, intermittently grazed plots had a significantly higher abundance of unpalatable species than did ungrazed plots.

6.3.2 Impact of grazing on functional groups

Table 6.2 presents the results of the tests for multiple proportions for all 15 functional groups in each of the 13 sites.

Table 6.2: Results of ‘Chi^2 tests for multiple proportions’ for differences in abundance of discrete functional groups between ‘grazed’ and ‘ungrazed’ treatments on the 13 study sites.

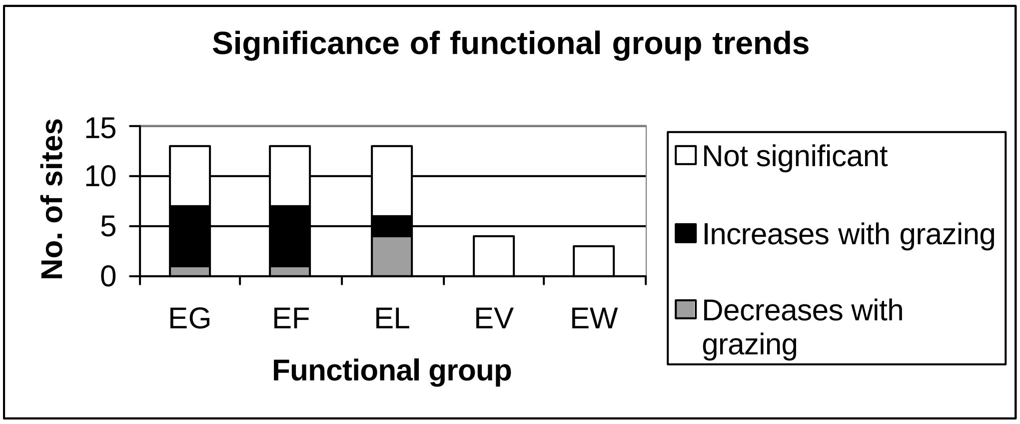
(NA = Not Applicable, NS = Not Significant, **Bold** = significant value).

SITE	EG	EF	EL	EV	EW	NG3P	NGP	NGA	NFA	NFP	NL	NLV	NV	NWP	S
KHILEA	NS	NS	**0.0008**	NS	NA	NS	NS	NS	**0.089**	NS	**0.0003**	NS	NS	NS	**0.0006**
KHILWT	**0.034**	**0.0253**	**0.0574**	NA	NA	NS	**0.0009**	**0.001**	NS	NS	**0.00001**	**0.025**	NS	NS	**0.00001**
BLURNG	NS	NS	NA	NA	NA	**0.0113**	NS	NS	NS	**0.014**	**0.00001**	**0.02**	NS	**0.067**	NS
LYSHN	**0.00001**	**0.0086**	NS	NA	NS	**0.00001**	**0.0052**	NS	NS	NS	**0.0052**	NS	NS	NS	NS
KIRK	NS	**0.016**	NS	NA	NA	**0.0057**	NS	NS	NS	NS	**0.0004**	**0.00001**	NS	NS	NS
RISHTN	NS	NS	**0.0526**	NS	NA	NS	NS	NS	NS	NS	NS	NS	NA	NS	NS
MTLEY	**0.00001**	NS	NS	NS	NA	NS	NS	**0.0022**	**0.0227**	**0.066**	NS	**0.0123**	NS	NS	NS
JERVOIS	**0.00001**	**0.0001**	NS	NA	NA	**0.00001**	NS	**0.0075**	NS	NS	NS	NS	NS	**0.0365**	NS
PAJING	NS	NA	NS	NA	NA	NS	**0.00001**	**0.0009**	**0.00001**	**0.00001**	NS	NA	NS	**0.074**	NS
LCKDM	**0.03**	**0.0004**	NS	NA	NS	NS	NS	NS	NS	NS	NS	NS	NS	**0.0253**	NS
EPP1	NS	NS	**0.00001**	NA	NA	NS	NS	**0.0227**	**0.0026**	NS	**0.00001**	NS	NS	NS	**0.017**
EPP2	**0.006**	**0.031**	**0.0002**	NA	NA	NS	**0.012**	NS	NS	**0.013**	**0.00001**	NS	**0.0012**	NS	**0.0016**
TABTOP	NA	**0.028**	**0.00001**	NA	NA	NS	NS	NS	**0.0001**	**0.00001**	**0.0033**	NS	NA	NS	NS

Probability level P<0.1

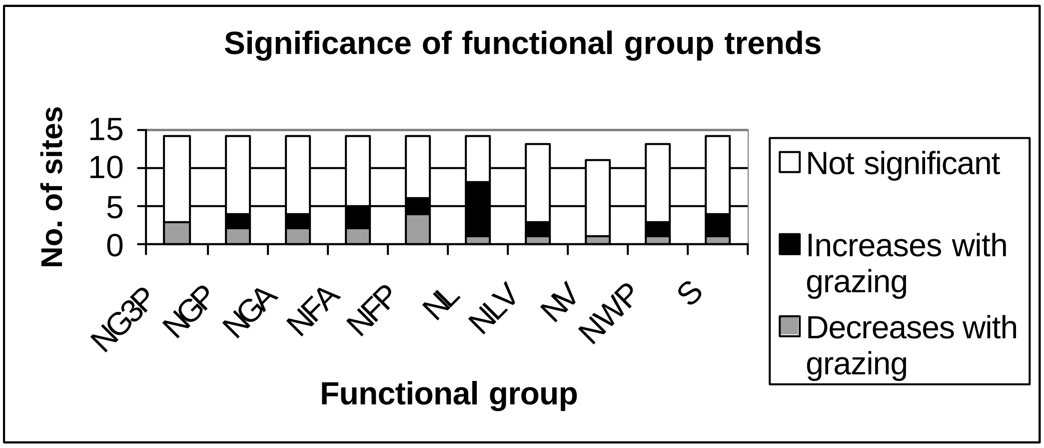
When the abundances of the 15 mutually exclusive functional groups were tested for their response to grazing, a similar pattern to that found with the broader groups was also seen. An examination of Figures 6.2 and 6.3 below showed that there was wide variation in grazing response and there were no universal trends in response to grazing that could be applied across the entire range of soil types and sites sampled. Figures 6.2 and 6.3 are separated into 'exotic' and 'native' and show trends for the functional groups that make up those two categories. Macropod exclosure plots were excluded from this analysis.

Figure 6.2: Stacked histogram showing the number of sites with different abundance of exotic functional groups between grazing treatments.
(EG – exotic grasses, EF- Exotic forbs, EL- exotic legumes, EV- exotic vines, EW- exotic woody plants).



probability level: P <0.1

Figure 6.3: Stacked histogram showing the number of sites with a different abundance of native functional groups between grazing treatments.
(NG3P- native grasses (3P), NGP- native perennial grasses, NGA-native annual grasses, NFA-native annual forbs, NFP-native perennial forbs, NL-native legumes, NLV- native legume vines, NV-native vines, NWP-native woody plants, S-sedges).



probability level: P <0.1

From the figures above, it can be seen that in a considerable proportion of sites examined, there was no significant result recorded. In two functional groups, EV (exotic vines) and EW (exotic woody plants), no significant values were obtained for any of the sites examined. The only functional group with several significant results that never showed a contradiction in response to grazing was the NG3P grasses. In each of the five plots showing a significant result, there was a lower abundance of NG3P grasses in grazed plots. Although the reduction in abundance of native vines (NV) with grazing was without contradiction, only one site had a significant result, so it is difficult to draw definite conclusions from this result.

The results of the 'tests for multiple proportions' of the functional groups is summarised in Table 6.3 below. From an examination of Table 6.3, it can be seen that the distribution and nature of significant results may be aligned with the major soil groups.

Table 6.3: Summary of trends for the relative abundance of functional groups between grazing treatments (determined from significant Chi^2 tests for multiple proportions).

(+ = Increases with grazing/ significantly higher abundance in grazed plots)

(- = Decreases with grazing / significantly higher abundance in ungrazed plots)

Site	EG	EF	EL	EV	EW	NG3P	NGP	NGA	NFA	NFP	NL	NLV	NV	NWP	S
Khilea			-						-		+				+
Khilwt	+	+	+				+	+			+	+			+
Bluerng						-				+	+	+		+	
Leyshn,	+	-				-	-				+				
Kirk		+				-					+	-			
Rshtn			-												
MtLey	+							+	+	+		-			
Jervoise	+	+				-		+							
Pajing							+	-	-	-				-	
Lckdam	-	+												+	
Epp1			-					-	+		-				+
Epp2	+	+	+				-			-	+		-		-
Tabtop		+	-						+	+	+				

The relative abundance of the various functional groups between plots can be difficult to visualise. Multivariate analyses of sites on the same soil group were therefore carried out and are presented below. An ordination of the magnitude of functional group abundance clearly distinguished between grazing treatments on several sites, while on other sites, such separation was not clearly recognisable. An examination of the PCAs and their respective dendrograms (used to identify clusters) show a range of responses. For brevity, only a selection of these has been presented. The PCA triplots for all study sites are included in Appendix F.

In Figure 6.4 (a) below, a PCA triplot for Mt Leyshon (Rangeview chromosols) is presented. Clusters of closely related plots were derived from the dendrogram in Figure 6.4 b. It was evident from the dendrogram presented previously in Figure 4.1 that Rangeview chromosols gave clear separation to plots on Jervoise and Mt Leyshon, so it was not considered useful to combine those two sites in this analysis.

Figure 6.4.a. PCA triplot of grazing treatment plots on Mt Leyshon (Rangeview chromosols), separated by abundance of functional groups (vectors).
(= grazed, O= intermittently grazed, ∇ = ungrazed) (PCA of Axes I & II explain 51.1 % of variation)

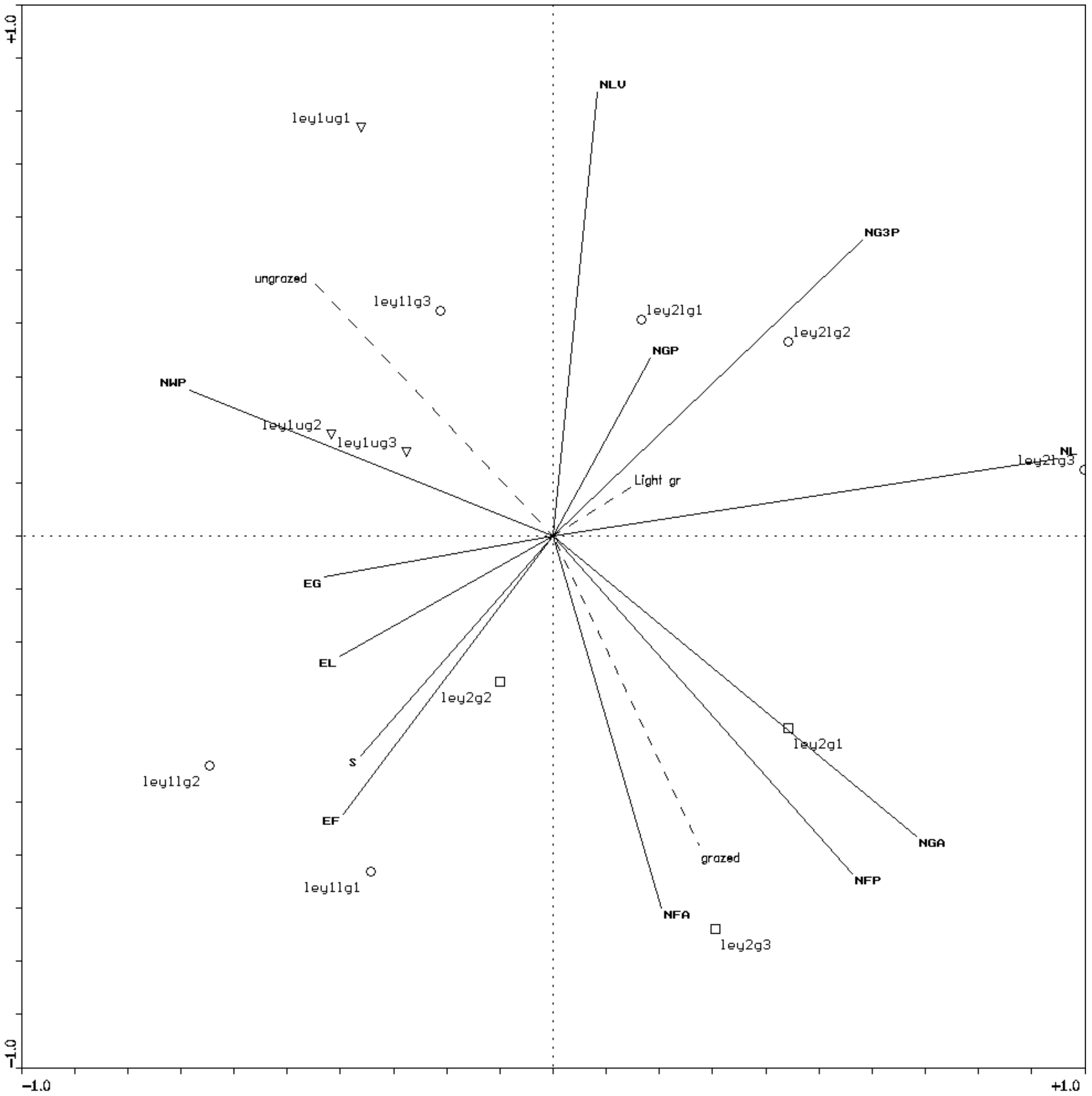
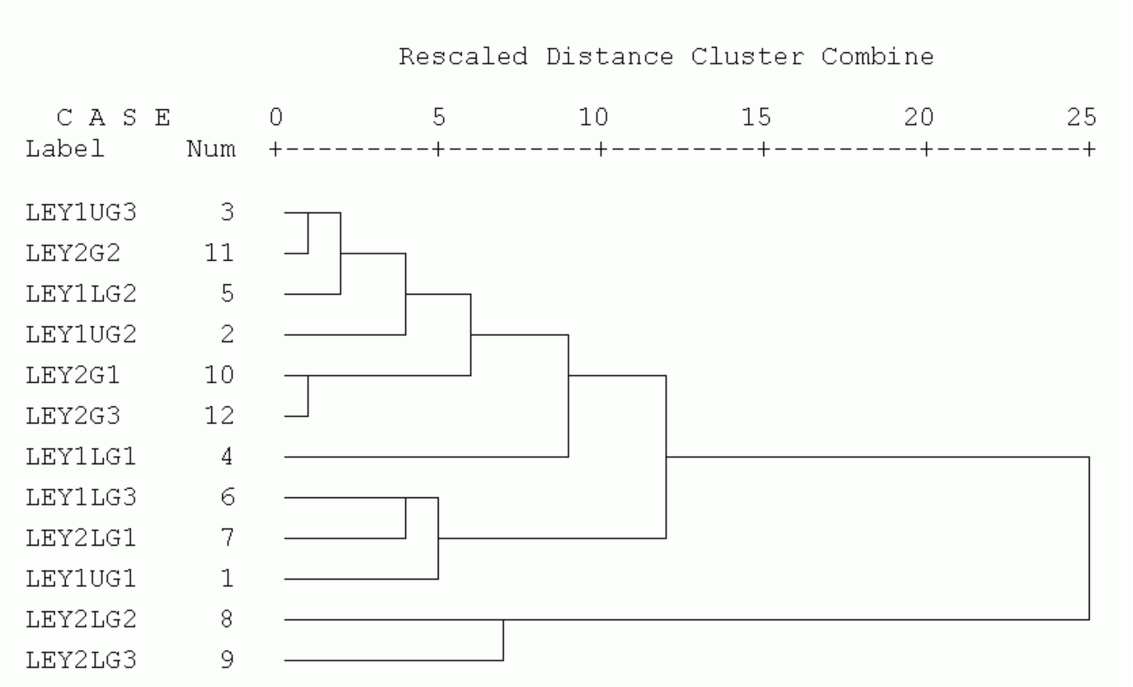


Figure 6.4.b. An average linkage dendrogram of grazing treatment plots on Mt Leyshon (Rangeview chromosols), indicating clustering of plots based on abundances of functional groups.



The hierarchical dendrogram in Figure 6.4 b indicates the degree of similarity of plots to each other based on the abundance of functional groups using species with at least 5% abundance in a single plot. Using an average linkage score of 11 to define the separate branches of the dendrogram, it can be seen that three clusters were identified. The first cluster contained plots representing all three treatments; the second combined an ungrazed plot with two intermittently grazed plots and the third cluster contained two intermittently grazed plots. It should be noted that this dendrogram used the frequency of abundance of functional groups and did not give the clear separation between treatments that was indicated in the dendrogram in Figure 4.1, which used abundances of all individual species.

However, from the PCA (Fig. 6.4a), it was evident that a grade exists from ungrazed plots in the upper left hand corner, through intermittently grazed plots to continuously grazed plots in the lower right hand corner. This grade involved several functional groups, all of which showed a significant difference between grazing treatments using ‘tests of multiple proportions’. The grade from ungrazed to grazed plots existed along a gradient of increasing NGA (P=0.0022), NFP (P=0.0667) and NFA (P=0.0227), with decreasing NLV (P=0.0123). Intermittently grazed plots in site 2 had significantly higher abundances of NG3P than continuously grazed plots (P= 0.00001).

Figure 6.4.c: Principal Component Analyses (PCA) triplot of grazing treatment plots on Dalrymple chromosols, clustered by presence/absence of functional groups.
(PCA of Axes I & II explain 55% of variation) (= grazed, ∇ = ungrazed, o = macropod exclosure)

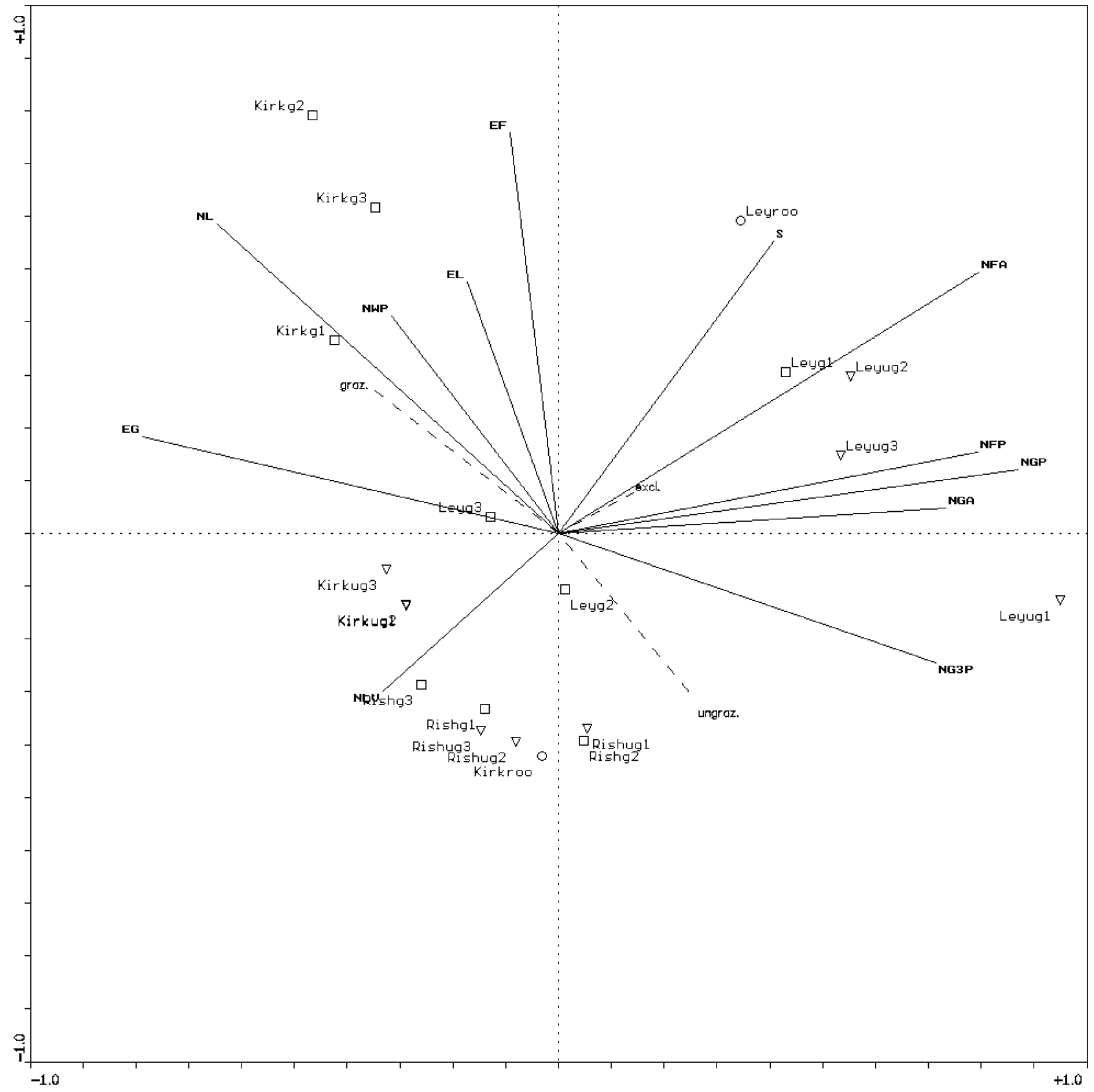
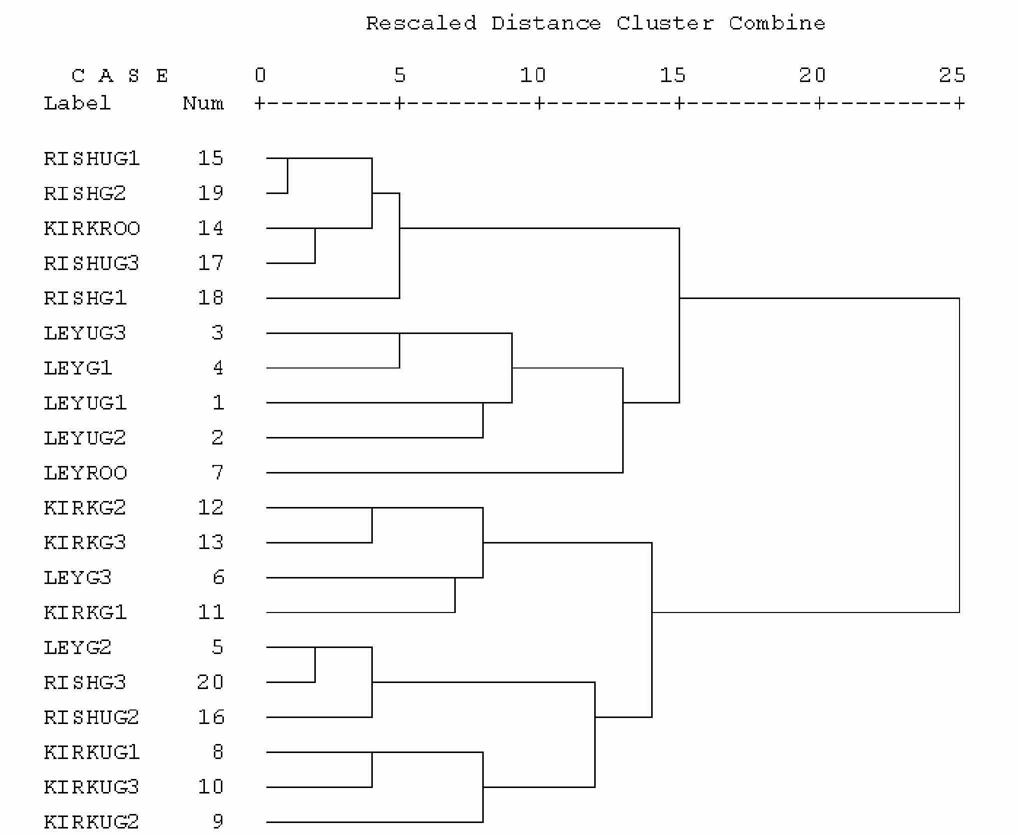


Figure 6.4.d: An average linkage dendrogram of grazing treatment plots on Dalrymple Chromosols, indicating clustering of plots based on abundances of functional groups (excluding species with less than 3 individuals in any plot).



Using an average linkage score of 11 to define the separate branches of the dendrogram, six clusters were identified in Figure 6.4 (d). It is evident from Figures 6.4 (c & d) that plots on Leyshon View and Kirk River are not clearly separated. Although axes I and II of the PCA (Figure 6.4 c) account for only 55% of the total variance, an examination of axes III and IV added little to the interpretation. Leyshon View grazed plot #3 (Leyg3) was more similar to grazed plots of Kirk River than to other grazed plots from the same site, while the Kirk River macropod exclosure (kirkroo) and another Leyshon grazed plot (Leyg2) clustered together with plots from Rishton. At the Kirk River site (Figure 6.4 (c)), there was a gradient of functional group abundance. Many of those functional groups had significantly different abundances between grazing treatments when compared using a test for multiple proportions (see Table 6.2). At Kirk River, a gradient of reduced grazing disturbance from 'grazed' to 'ungrazed' to 'roo' (exclosure) shows a decreasing abundance in EF (P=0.016), NL (P=0.0004) and NWP, with grazed plots having a low abundance of NLV (P=0.00001) and NG3P (P=0.0057). A similar directional gradient in Leyshon View shows a significant increase in EF (P=0.0086), NG3P (P= 0.00001) and NGP (P=0.0052), although, the NGP was absent in the exclosure plot. Figures 6.4 (c) and 6.4 (d) show that grazing treatments at Rishton do not clearly cluster apart. Tables 6.2 and Table 6.3 shows that most functional groups had no significant difference between treatments at Rishton, except for EL, which declined with grazing (P=0.0526).

Figure 6.4.e: Principal Component Analyses (PCA) triplot of grazing treatment plots on Bluff Brown sodosols, clustered by presence/absence of functional groups
(PCA of Axes I & II explain 76% of variation)

(= grazed, ∇ = ungrazed, o = macropod exclosure)

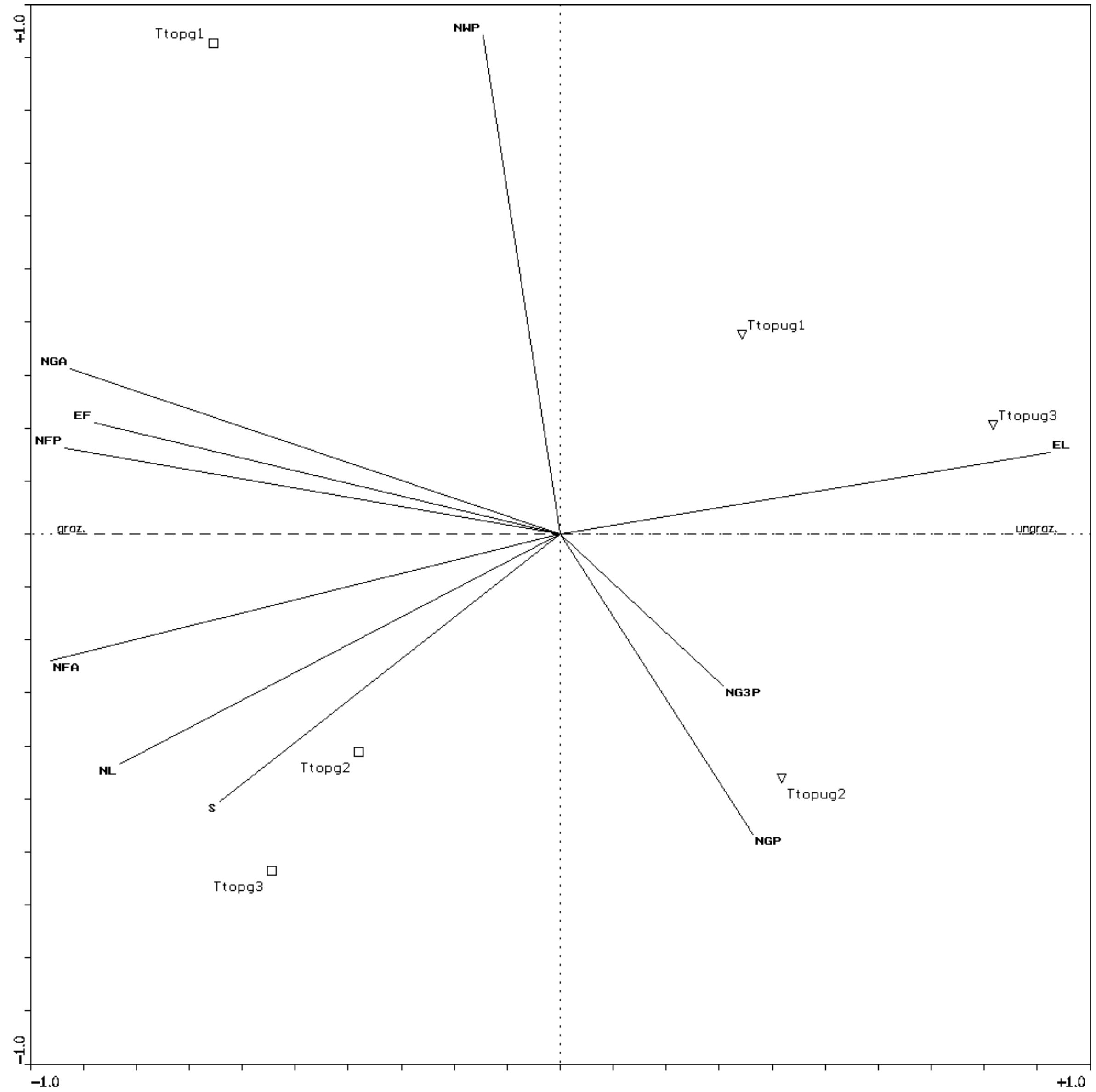
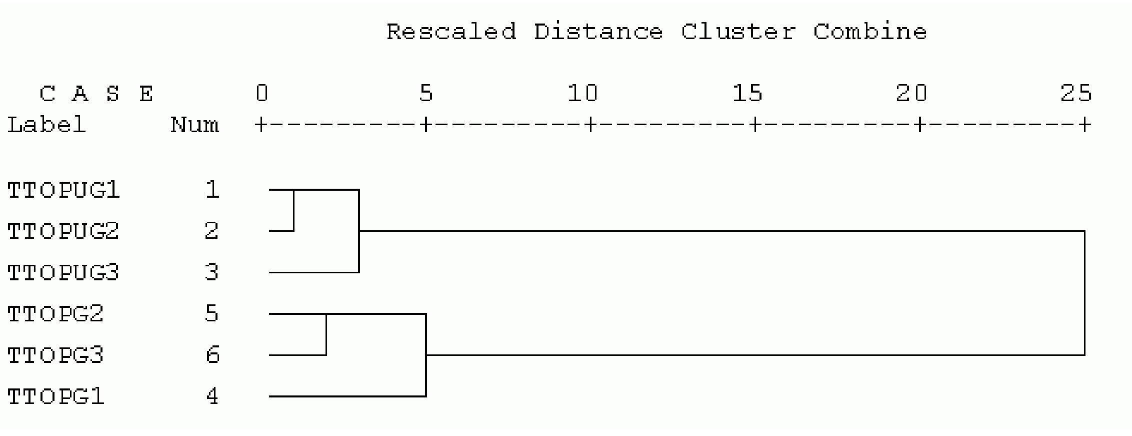


Figure 6.4.f: An average linkage dendrogram of grazing treatment plots on Bluff Brown Sodosols, indicating clustering of plots based on abundances of functional groups.



The clusters of closely related plots shown in Figure 6.4 (e) were derived from the dendrogram in Figure 6.4 (f). Using an average linkage score of 11 to define the separate branches of the dendrogram, it can be seen that two clusters were identified. In both Figure 6.4 (e) and Figure 6.4 (f), plots on Tabletop were clearly separated by grazing treatment. Axes I and II shown on the PCA in Figure 6.4 (e) explained 76% of the total variance. Although both treatments have plots spread along Axis II (vertical axis), they are widely separated along Axis I (horizontal axis). This separation between grazing treatments on Tabletop was based on grazed plots having significantly higher abundances of EF (P=0.028), NFP (P=0.00001), NFA (P=0.0001) and NL (P=0. 0033), and a trend towards more NGA, while ungrazed plots had significantly higher values of EL (P=0.00001). The vertical spread of plots within the two treatments was due to widely varying abundances of NWP and, to a lesser extent for grazed plots, the abundances of NGP.

Figures 6.4 (a-f) above are examples of PCAs and dendrograms. Trends in the responses of functional groups to grazing were identified using the PCAs, dendrograms, tests for multiple proportions (Table 6.2) and the original raw data (Appendix E). These trends are summarised in Table 6.4 below for all sites. This list of trends includes all those found to be significant (see Table 6.3 previously) and adds those indicated on the PCAs as having an influence on the data cloud.

Table 6.4: Summary of trends in functional group abundance between grazing treatments derived from ‘Chi^2 tests for multiple proportions’, PCAs and the raw data.

(+ = Increases with grazing, - = Decreases with grazing, ∧ = Increases with intermediate disturbance)

Site	EG	EF	EL	NG3P	NGP	NGA	NFA	NFP	NL	NLV	NWP	S
Khilea		-	-			+	-		+			+
Khilwt	+	+	+		+		+		+	+	+	+
Bluerng				-				+	+	+	+	
Leyshn,	+	-		-	∧		-	∧	+			-
Kirk		+		-					+	-	+	
Rshtn			-									
MtLey	+	∧		∧		+	+	+	∧	-	-	∧
Jervoise	+	+		-		+						
Pajing				-	+	-	-	-			-	
Lckdam	-	+	-		+	-		+			+	
Epp1			-		-	-	+	-	-	-		+
Epp2	+	+	+		-			-	+			-
Tabtop		+	-			+	+	+	+			

Table 6.5: Contribution of macropod exclosure data to explaining trends in abundance of functional groups.

(+ = Increases with grazing, - = Decreases with grazing, Ù = Increases with intermediate disturbance, Ú = Decreases with intermediate disturbance)

Site	EG	EF	EL	NG3P	NGP	NFA	NFP	NL	NLV	NWP	S
Khilea		-	Ù			-		Ú			+
Khilwt	Ú	Ú				Ú		Ú	+	Ú	Ú
Bluerng				Ù				Ú	+	Ú	
Leyshn,	Ú			-	Ù	-	Ù	+			-
Kirk		+		-				+		+	
Epp2	+	+	+		-		-				-

6.4 DISCUSSION

6.4.1 Impact of grazing on broad plant types

The groups analysed for this discussion were not exclusive of each other, in that a single plant species could belong to several of the groups discussed. Most of these groups showed very different trends between sites. This variation in response within certain functional groups may be related to the differences that soil and community composition had on determining which particular plant species dominates the influence over these functional groups at a particular site.

Introduced Species: Each site examined in this research project contained some introduced species. Of the 437 species recorded in this research, 47 were introduced. This broad functional group contained a broad spectrum of plant morphological types, some of which were palatable (eg *Alysicarpus bupleurifolius, Stylosanthes hamata*), while others were toxic or armed with anti-herbivore defences (eg *Cryptostegia grandiflora, Opuntia stricta*). Nevertheless, Pimm (1994) used invasion by exotic species as a sign of reduced community diversity, complexity and resistance, so an examination of this broad group was justified.

Review of the literature predicts that introduced plants should increase in abundance under grazing pressure. This is likely due to the characteristics they possess for selection for introduction (eg drought tolerance, high fecundity) also make those plants effective weeds. Species that are toxic or unpalatable have been shown to increase (Ash & Corfield 1998, McIvor & Gardener 1995), however, deliberately introduced fodder species such as *Cenchrus ciliaris* and *Stylosanthes hamata* are also known to increase under moderate grazing pressure (McIvor & Gardener 1995, McIvor & Gardener 1998). Results from Chapter 4 showed a variation in the responses of the exotic species *Cenchrus ciliaris, Bothriochloa pertusa* and *Melinus repens* to grazing.

Results from the present chapter showed a balanced response to grazing by introduced plants; the interpretation of which is assisted by the results presented in Chapter 4. Introduced plants had significantly higher abundance with grazing in three sites. *Bothriochloa pertusa* was responsible for that increase in two of those sites (Leyshon View and Jervoise), while *Melinus repens* had the greatest influence on the result for introduced plants at Kangaroo Hills West. At Mt Leyshon, continuously grazed plots had significantly more introduced plants (mostly *Bothriochloa pertusa* and *Cenchrus ciliaris*) than intermittently grazed plots.

Three sites showed a significantly lower abundance of introduced plants in grazed plots. At Blue Range, the grass species *Dicanthium annulatum, Bothriochloa pertusa* and *Melinus repens* were the greatest contributors to the decrease in exotic plants, although numbers of all these species were individually generally low. At Epping Forest (1), *Cenchrus ciliaris* and the legume *Chamaecrista absus* were the most common exotic species. Although there were no significant differences in *Cenchrus ciliaris* between treatments, there was significantly more *C. absus* in ungrazed plots. The negative response of the unpalatable *C. absus* to grazing treatment was difficult to explain solely in grazing terms. Populations of this species may have responded negatively to an increased abundance of the population of an increaser species, however, determining the exact nature of such an interaction was outside the scope of the present study. *Stylosanthes scabra* was the most abundant exotic species at Tabletop, and showed a significant decline in abundance with grazing. This agrees with comments by McIvor and Gardener (1998) that indicate *S. scabra* declines with high stocking rates.

In each case, the primary species contributing the most to the result was an edible species with a low resistance to heavy grazing. All, with the exception of *C. absus,* were listed in the literature as being palatable.

In conclusion, the response of the 'introduced plants' functional group to grazing was usually dependant on the palatability and grazing response of the most abundant species of introduced plant at that site; a variable that differed between sites and soil types. Suggestions that introduced plants in general are increasers in grazing-disturbed pastures often fail to take this into account.

Life History Strategy: Four sites (Kangaroo Hills West, Mt Leyshon, Epping Forest 2 and Tabletop) had significantly greater abundance of annuals in grazed plots, while three sites (Leyshon View, Pajingo and Epping Forest 1) had significantly less. Results that showed an increase of annuals in grazed plots concur with results of Landsberg *et al.* (1999), Friedel *et al*. (1988), McIvor *et al*. (1995a) and the comments by Symonides (1988) noting that annuals are generally poor competitor species, and so are most likely being released from the competitive exclusion of perennial species by cattle grazing. However, three sites showed significantly fewer annuals in grazed plots. The latter sites had their annual plant functional group dominated by a variety of species. The dominant annual species that significantly declined in response to grazing at Pajingo and Epping Forest (1) were species recorded in the literature as being grazed by cattle. These species were *Schizachryium fragile*, *Setaria surgens*, (Pajingo) and *Indigofera colutea* (Epping Forest). Despite the annual life history being adapted for disturbance, the relative desirability of these select species to cattle has obviously resulted in their decline. However, the most common annual at Leyshon View was the native forb *Spermacoce brachystema*, a species with unknown grazing significance (Milson 2000). This species was totally absent in grazed plots. Whether or not this was from grazing or from a low tolerance to damage by trampling could not be determined.

In conclusion, the prediction that annual plants are increasers that become abundant with grazing (Symonides 1988) was not confirmed for every site, since this research shows that some annual plants can be decreasers, and that palatability of some species to cattle may play a role in determining the total abundance of annuals.

Perennial plants showed fewer significant results than did annual plants. One site (Mt Leyshon) had significantly fewer perennial plants in intermittently grazed plots than ungrazed plots. All other sites showed a non-significant result. The difference in the number of significant results between annuals and perennials was most likely due to ‘perennials’ containing a broader range of life forms such as herbs, grasses and woody plants, all of which would be expected to have a broad range of responses to grazing. It is therefore more likely to see significant results if these broad functional groups could be examined in finer scale, separated into life form groups that are likely to reflect response to perturbations.

Palatability: Seven sites (Blue Range, Leyshon View, Mt Leyshon, Kirk River, Jervoise, Epping Forest 1 and Tabletop) had significantly lower abundance of palatable species in grazed plots, while one site (Epping Forest 2) had significantly greater abundance. In the latter case, the palatable species group was dominated by exotic *Cenchrus ciliaris*. *Cenchrus* individually showed a significantly higher abundance in grazed plots at that site (see Chapter 4). It should be noted that there is no clear line distinguishing palatability of species, with relative palatability being the key issue. Wilson & Harrington (1984) observe that in vegetation communities there are a range of levels of palatability. Selective grazing pressure impacts are transferred from one level of palatability to the next level down as availability progresses from high to low (Wilson

& Harrington 1984). The impact of grazing on a particular species therefore depends on where that species sits in that hierarchy of palatability. In the case of *Cenchrus ciliaris*, although many graziers regard it as being palatable, the response of increased abundance with grazing may indicate it was less palatable than co-occurring species, occupied a lower level in the palatability hierarchy and that selective grazing pressure on those more palatable species was responsible for the increased domination of *Cenchrus ciliaris*.

Five sites (Blue Range, Kirk River, Pajingo, Jervoise and Tabletop) had a significantly greater abundance of unpalatable species in grazed plots, while three sites (Leyshon View, Lucky Downs dam and Epping Forest 1) had significantly lower abundance. In contrast, however, there were more unpalatable plants in ungrazed plots at Leyshon View (Dalrymple chromosols), Lucky Dam (Wairuna dermosols) and Epping site 1 (Epping chromosols). At Leyshon View, the two most abundant unpalatable species showed no significant difference when analysed individually. Lucky Down Dam and Epping Site 1 both had a high abundance of the unpalatable exotic forb *Sida rhombifolia*, noted by Milson (2000) as being an increaser species. At Lucky Downs Dam, *Sida rhombifolia* was more abundant in grazed areas, so went against the overall trend for unpalatable species for that site. In contrast, *Sida rhombifolia* was marginally more abundant in ungrazed areas at Epping Site 1, contrary to the expected trend for that species. Such variation in grazing response within a single species may be due to genotypic variation of grazing-related attributes, history-related differences in plant species interactions, or variation in its palatability depending on soil characteristics or co-habiting species (Noy-Meir *et al.* 1989). In light of the comments by Wilson & Harrington (1984) relating to relative palatability, it is possible that *Sida rhombifolia* may have occupied a higher relative position of palatability at Epping site 1 than it did at Lucky Downs Dam, due to the relative palatability of co-habiting species.

Four sites (Blue Range, Kirk River, Pajingo and Jervoise) showed both a decrease in palatable species with a corresponding increase in unpalatable species. These results support observations by Ash & Corfield (1998) that cattle selectively graze palatable species while ignoring relatively unpalatable species.

6.4.2 Impact of grazing on functional groups

It is evident from Figures 6.2 and 6.3 that in a considerable proportion of sites examined, there was no significant difference in abundance between grazing treatments for many functional groups. In the two functional groups EV (exotic vines) and EW (exotic woody plants), no significant differences were observed for any of the sites examined. This is most likely due to the low values obtained for these groups (2 and 4 species respectively). Although numerous declared noxious woody plants were encountered in the timed searches, they were rarely encountered in the quadrats, as they were widely spaced and therefore rare at the spatial scale being considered.

Listed below are discussions for the discrete functional groups that gave significant results when tested for differences between treatments using tests for multiple proportions.

<u>Exotic Grasses (EG)</u>: Nine species of grasses are combined here, including *Cenchrus ciliaris*, *Bothriochloa pertusa* and *Melinus repens*, which have been individually analysed and discussed previously in Chapter 4. At five sites there was a significant increase in exotic grasses with grazing. At Mt Leyshon, there was a higher abundance of exotic grasses in intermittently grazed than ungrazed plots, and more in continuously grazed than in intermittently grazed plots. Only at Lucky Downs Dam were there significantly fewer exotic grasses in grazed plots. This was primarily due to the response of *Bothriochloa pertusa*, which was previously discussed in Chapter 4.

In conclusion, exotic grasses as a group tended to respond as increasers, and that increase occurred with intermittent grazing as well as continuous grazing. There was evidence to suggest that an increase in *Bothriochloa pertusa* can occasionally occur in the absence of heavy grazing since values were, in one instance, higher in ungrazed plots.

<u>Exotic Forbs (EF)</u>: Few of the species in this group were introduced for the pastoral industry and many are described in the literature as unpalatable species of no grazing significance, so it is expected that with grazing pressure, their abundance would either increase or show no response. Six sites showed significantly greater abundance of exotic forbs with grazing, while one site (Leyshon View) showed significantly less. The two most common species contributing to this result were basil (*Ocimum basilicum*) and

Sida rhombifolia, both of which individually showed lower abundance with grazing. *Sida* has been discussed previously as decreasing under cattle grazing at Epping Forest, supporting observations that cattle graze certain *Sida* species (Henry *et al*. (1995) and O'Reagain (pers. comm.)). There is no literature that would explain a decline of *Ocimum* under grazing.

In conclusion, exotic forbs, as a functional group, were generally increasers, however, a small number of species did respond as decreasers under certain conditions.

Exotic Legumes (EL): Most of these species (eg. *Alysicarpus* spp, *Macroptilium* spp and *Stylosanthes* spp) were recorded as being deliberately introduced for the purpose of improving cattle grazing pastures (Hacker 1990). The present research found exotic legumes had significantly lower abundances in grazed plots at four sites, while two sites had significantly higher abundance in grazed plots. Species that behaved as decreasers were *Crotalaria juncea* (Kangaroo Hills East), *Stylosanthes scabra* (Rishton and Tabletop) and *Chamaechrista absus* (Epping Forest 1), while *Chamaechrista absus* increased with grazing at Epping Forest (2).

Crotalaria juncea is regarded as toxic (Hacker 1990), so it can be assumed that this is not a pasture improvement species. *C. juncea* had higher abundance in plots with intermediate disturbance. It is possible that in the macropod exclosure, it faced heavy competition while its slender, erect habit may have made it susceptible to trampling damage in the grazed plots. The present results showing *Stylosanthes scabra* as a decreaser supports observations by McIvor and Gardener (1998). Observations made throughout the Dalrymple Shire also show a strong fenceline effect on *Stylosanthes scabra* along roadsides, with *S. scabra* apparently dominating in the absence of grazing but almost entirely disappearing under heavy grazing. However, most *Stylosanthes* species have a persistent seed bank (McIvor 1987), which may allow rapid regeneration if relief from grazing pressure is applied. No clear individual results were obtained for *Stylosanthes hamata*, reported by McIvor & Gardener (1998) as benefiting from intermediate stocking rates. The results for *Chamaechrista absus* are apparently contradictory. An explanatory hypothesis may be that this species has higher palatability or desirability to cattle on the Epping Chromosol soils (Epping 1) where it decreases under grazing, than it does on the sandy Walthum Tenosols soils (Epping 2) where it increases in abundance. Alternatively, differences in grazing response on different soil types may also be due either genotypic variation in grazing-related physical attributes or to some complex interaction with another plant or herbivore species that was itself influenced by differences in historical cattle grazing (Noy-Meir *et al* 1989). The testing of either hypothesis was, however, outside the scope of the present research.

In conclusion, exotic legumes as a functional group were generally decreasers or benefited from intermediate disturbance, although response to grazing may have varied depending on soil-related palatability or other grazing response mechanisms.

3P Native Grasses (NG3P): These are the palatable, perennial and productive native grasses frequently described as consistently decreasing under grazing pressure (Ash & Corfield (1998), Rolfe *et al*. (1997), McIvor & Gardener (1995)). The present research confirmed this functional group as consistent decreasers, and did not show a contradiction in response to grazing. Results from the present research showed that four sites had significantly higher abundances of NG3P grasses in ungrazed plots. These results indicate a susceptibility to grazing. At Mt Leyshon, there was no significant difference between ungrazed and grazed plots; although there was higher abundance in intermittently grazed than in ungrazed plots. Further examination of macropod exclosures added evidence that this functional group achieved highest abundance in areas of intermediate disturbance (ungrazed plots subject to macropod grazing) since abundance was lower in both grazed and exclosure plots.

In conclusion, this functional group was a strong decreaser, declining with cattle grazing but increasing with intermediate disturbance (macropod grazing).

Native Grasses- other Perennials (NGP): This functional group consists of all the native perennial grasses that are not both palatable and productive. This functional group consequently contains many species described in the literature as unpalatable increasers, hence it could be predicted that greater abundance with grazing would be observed. Results from this research showed significantly greater abundance in grazed plots on two sites (Kangaroo Hills West and Pajingo), which supported the predicted increase. However, there was also a significantly lower abundance of native perennial grasses with grazing on two sites (Leyshon View and Epping Forest 2). Examination of the macropod exclosure shows evidence that these perennial grasses may have greater abundance with intermediate disturbance since there is a decline in abundance in both grazed and exclosure treatments.

The unexpected lower abundances were influenced by *Aristida contorta* (Leyshon View) and *Enneapogon flavescens* (Epping Forest 2). Many authors refer to all *Aristida* species as unpalatable increasers (Milson (2000), Rolfe *et al*. (1997), Henry *et al*. (1995)), however, McIntyre and Filet (1997) observe that a broad range of ecological responses occur within this genus and that *A. contorta* is valued as a fodder species. Tothill and Hacker (1996) state that the genus *Enneapogon* is regarded as a valuable group of forage grasses and Milson (2000) records other species as being palatable fodder species. It is apparent that the reverse trend was observed at Leyshon View and Epping Forest (2) because the dominant NGP grasses were palatable decreasers, in contrast to sites where the functional group increased with grazing.

In conclusion, this group showed a broad range of responses, increasing or decreasing with grazing, or increasing at intermediate levels of grazing. Differences in the palatability of the most common species appeared to be the source of this variation, while productivity potential seemed to be of little consequence.

Native Grasses – annual (NGA): Results presented in the previous section showed that, while many annuals were increasers, some annual plants responded as decreasers. It was decided that separating annuals by life form may assist in elucidating their response to grazing, however, results for NGA were mixed. Four sites showed the expected trend of increasing abundance with grazing pressure, three of which gave significant results with the test for multiple proportions. Two sites (Pajingo and Epping Forest 1) had significantly lower abundance of native annual grasses in grazed plots. *Schizachyrium fragile* and *Setaria surgens* influenced the Pajingo result, as they did previously in the broad group 'annuals'. At Epping Forest (1), comet grass (*Perotis rara*) was the largest contributor to the trend of more NGA on ungrazed plots. Comet grass is known to be a coloniser of disturbed ground (Tothill and Hacker 1996), so would be expected to be a increaser, however, Milson (2000) and Henry *et al.* (1995) report that it is palatable to stock despite having very little bulk. It is possible that this species may have been opportunistically grazed to a state of lower abundance in cattle grazed plots or that macropod grazing in particular promotes higher abundances. Other examples of palatable annual grasses declining under heavy grazing have previously been reported (Suijdendorp 1969). Due to the age of the exclosure (19 years), it was unlikely that the current patterns of comet grass abundance still reflect pre-exclosure grazing impacts.

In conclusion, even when separated from other life forms, the native grass component of annual plants still showed wide variation in response, and it is likely that palatability was the source of this variance.

Native Forbs – annual (NFA): As with other annual groups, it was considered probable that native annual forbs would have a higher abundance in disturbed plots. Three sites showed a significantly higher abundance of this functional group with grazing. At Mt Leyshon, intermittently grazed plots had a higher abundance than ungrazed plots, with only marginally higher values in grazed plots. These results support the prediction of native annual forbs being an increaser group. However, two sites (Kangaroo Hills East and Pajingo) had lower abundance in grazed plots. The result from Kangaroo Hills East appears to be the cumulative effect of several non-significant species. Conversely, Pajingo had three species that, individually, were significant decreasers. These were *Phyllanthus* type 3, *Oldenlandia mitrasacmoides* and *Goodenia hirsuta*. No information was available on the ecology of these species in relation to grazing. These plants are generally small and delicate and may be susceptible to treading.

Results presented here indicate that while there was an expected increase on some sites, some small annual forb species of no grazing significance can decline under grazing.

Native Forbs – Perennial (NFP): This group contained numerous species for which little information on palatability or response to grazing could be gathered. No predictions could be made about an expected response of this group to grazing. Three sites (Blue Range, Mt Leyshon and Tabletop) showed a significant increase in native perennial forbs with grazing, while the PCA triplots suggested an increase with grazing at Lucky Downs Dam and with intermittent grazing at Mt Leyshon. Two sites (Pajingo and Epping Forest 2) showed a decrease in the abundance of this functional group with grazing. Abundant species that responded as increasers at different sites included *Boerhavia paludosa, Brunoniella acaulis* subsp. *acaulis, Drosera indica, Gratiola pedunculata* and *Polymeria longifolia,* while decreasers included *Evolvulus alsinoides* and *Waltheria indica*. The result for Mt Leyshon appeared to be the net effect of several species that gave no significant result individually. At Epping Forest (2) there was significantly higher abundance of *Waltheria indica* in ungrazed plots than in grazed plots. A high abundance of *Waltheria* in the adjacent macropod exclosure confirmed this trend. This result was unexpected since Milson (2000) records that this species is an unpalatable increaser.

The 'native perennial forb' functional group obviously consisted of more than one grazing response group, so further splitting of this group by palatability may yield more significant results; however, such information was not available for many species.

Native legumes (NL): This group consisted of erect or prostrate native leguminous plants but excludes those species with trailing or climbing stems. Many legume species are nutritious but may vary in palatability (Hacker 1990), which may be a source of variation in grazing response. One site (Epping Forest 1) showed a decline in NL, due to a decline in the palatable species *Indigofera colutea*, which was discussed previously in 'annuals'.

Seven sites had a significantly higher abundance of native legumes in grazed plots. Mt Leyshon showed a slight increase with intermittent grazing in comparison to continuously grazed sites. There were no significant results from sites on Dalrymple chromosols, while Greenvale chromosols had a significant increase at each representative site.

By examining the PCA triplots and results from exclosure plots, it appears that abundances for native legumes on Greenvale chromosols were highest in both grazed and total exclosure plots and that values were lowest in intermediate disturbance plots (ungrazed), effectively creating a 'U' shaped curve against an x axis of increasing disturbance. Close examination of the raw data showed that a combined response of several species caused this effect. The legumes *Indigofera linnaei* and *Zornia muriculata* were the greatest contributors to the native legume functional group on those sites. *Indigofera* was an increaser at Kangaroo Hills East and Blue Range but showed no evidence of the 'U' shaped response curve. *Zornia muriculata* was an increaser in each of the Greenvale sites and showed a 'U' shaped curve at Blue Range. While bell-shaped curves for species benefited by intermediate disturbance are well known in ecology (Petraitis *et al*. 1989), 'U' shaped curves are not. It should be noted, however, that Landsberg *et al*. (1997a) found a variety of shaped curves (including 'U' shaped) in their examination of species richness along a grazing gradient and concluded that the diversity of curve shapes represented a real diversity of responses to disturbance.

In the case of native legumes in the present study, it is possible that *Zornia* was being out-competed by another species that had itself benefited from intermediate disturbance, or that this was a chance distribution pattern of what were essentially small figures of abundance.

In conclusion, however, it was evident that this functional group generally responded as increasers under grazing.

Native Legumes – vines (NLV): This functional group contained 14 native legume species with a vine or scrambler life form. Hacker (1990) describes most of these species as being palatable to stock, though not necessarily very productive. Two sites (Kangaroo Hills West and Blue Range) had significantly higher abundance of native legume vines in grazed plots. *Galactia muelleri* showed a noticeable increase in abundance with grazing at Kangaroo Hills West, suggesting a low grazing preference by cattle. Hacker (1990) notes that there is no information available on the acceptability of this species to stock. Two sites (Kirk River and Mt Leyshon) had significantly lower abundance of native legume vines in grazed plots. Blue Range had a marginal increase in *Rhynchosia minima* with grazing, however, the values were too low for any measure of significance. This same species was a significant decreaser at Kirk River and Mt Leyshon, as was *Glycine clandestina* and *Glycine* sp. A. These results counter those of Rolfe *et al*. (1997) that *Rhynchosia* abundance increases with deterioration of pastures. In most instances, native legume vines indicated a susceptibility to cattle grazing.

Native Vines (NV): There was only one significant result for this functional group. *Jaquemontia paniculata* decreased significantly with grazing at Epping Forest site 2; a trend supported by a large abundance in the macropod exclosure. This vine species has been observed as being relatively palatable to cattle in other sites (O'Reagain P. pers. comm.). Although 16 species of native vines were recorded in this study, most sites gave only non-significant results, indicating that this functional group type had a low response to grazing disturbance.

Native Woody plants (NWP): Numerous authors predict an increase in woody plant abundance under increasing grazing pressure (Quirk *et al*. 1997, Scanlan *et al*.1996a, Friedel *et al*. 1988). Generally, numbers of woody plants encountered within quadrats in the present study were low. The most abundant species were woody sub-shrubs. Trees were mostly encountered as seedlings and small saplings and were not encountered with a high enough frequency to yield significant results. Tree communities are examined in greater depth in Chapter 8. Two sites (Blue Range and Lucky Downs Dam) showed significantly higher abundance of woody plants in grazed plots. Blue Range was influenced by the net effect of several species that showed no significant results individually, while Lucky Downs showed an increase in the sub-shrub *Sida subspicata.* Milson (2000) records that this species specialises in disturbed and degraded pastures. One site (Pajingo) showed a decrease of woody plant abundance with grazing, mostly due to an absence of *Hibiscus meraukensis* in grazed plots, although only low numbers were present in ungrazed plots.

While there may be a trend towards an increase in woody plants with grazing, confirmation of this trend was hampered by generally low abundances of representative species within the quadrats.

Sedges (S): Although Friedel *et al*. (1988) lumped sedges together with grasses, they are unrelated and possess certain morphological traits such as tubers, which are not present in grasses. McIvor (1987) notes that although sedges are often an insignificant component of grasslands, they contribute greatly to seed banks and can dominate pastures after their destruction by disturbances such as cultivation. They may therefore be expected to exhibit different responses to grasses in reaction to perturbations such as grazing. Three sites (Kangaroo Hills East/ West and Epping Forest 1) had a significantly higher abundance of sedges in grazed plots, while one site (Epping Forest 2) had significantly fewer sedges in grazed plots. Most sedge species were relatively rare. The perennial sedge *Fimbristylis dichotoma* was the most commonly encountered sedge. At Mt Leyshon, this species increased with intermittent grazing but was virtually absent in other treatments. At Kangaroo Hills West it was a significant increaser, while at Epping Forest 2 it was a clear decreaser. In both cases, examination of the macropod exclosure confirmed these trends. This discrepancy may result from differences in soil nutrients influencing levels of palatability and/or acceptance to grazing animals. The increase in abundance of sedges at Epping Forest (1) was a function of a weak increase in *Fimbristylis*, combined with a weak increase in the perennial *Cyperus betchei*. The overall decrease in sedges at Epping 2 was a combination of a decrease in *Fimbristylis* and the annual *Cyperus* sp. D.

In conclusion, most sedge species were too rarely encountered to yield significant results, and there seemed to be a wide variation in grazing response of the most commonly encountered species *Fimbristylis dichotoma*. Therefore, no generalisations could be made about the response of sedges to grazing from the present research.

6.4.3 Trends in species composition

Research from Africa and Australia has led numerous authors to make some generalisations about the impact of cattle grazing on species composition. The conclusion of that research appears to be that heavy grazing results in:

- A decrease in palatable perennial grasses, and
- An increased abundance in unpalatable grasses, annuals, forbs and woody plants

(Ash & Corfield (1998), McIvor & Gardener (1995), McIvor *et al*. (1995a), Mwalyosi (1992), Shaw & 'tMannetje (1970)).

The present research concurs with many of these expected trends. All but two sites showed at least some of the trends in changes to pasture composition identified in the literature. These trends include:

- A decline of palatable species and a corresponding increase of unpalatable species
- A reduction in native perennial, palatable and productive grasses (NG3P)
- A decrease in exotic legumes and native legume vines (EL, NLV)
- An increase in exotic grasses (EG)
- An increase in unpalatable annuals with grazing (NGA, NFA)
- An increase in forbs (EF, NFA)
- An increase in non-vine native legumes (NL)

A range of palatability within several functional groups reduced the ability to recognise directional response of those groups. Several individual plant species showed significantly different responses to grazing on different sites, indicating a possible change in palatability or complex species interactions depending on soil type. These findings support comments by McIvor (1998) that there is wide variation in response to grazing between different sites.

The results from Kangaroo Hills East and Tabletop showed an interesting trend with legumes in that exotic legumes introduced for pasture improvement achieved a higher abundance with cattle exclusion, whereas many native legumes had a significantly higher abundance in grazed plots. Native legume vines showed an increase with grazing at two sites, despite the fact this functional group was elsewhere a decreaser. Although some of these native legume species may be unpalatable, it still indicates that many native legumes are capable of persisting, even thriving, under cattle grazing regimes, whereas their exotic counterparts are more susceptible to being grazed out. The lower abundance of exotic legumes located in grazed plots is contrary to the general belief (eg. Leslie *et al*. 1987) that exotic legumes have a greater ability to persist under heavy grazing. The present research suggests that native grasslands still retained a substantial abundance of native legumes, in contrast to assertions by Leslie *et al*. (1987) that native grasslands have long since lost the significant dietary contribution made by native legumes. It was, however, impossible to calculate if a permanent loss of some native legumes had occurred in the 100 years of grazing prior to the present research being undertaken.

Grasses showed the opposite trends. Both exotic (EG) and native grasses (NG3P, NGP) are of economic importance to the grazing industry (Rolfe *et al*. 1997). The native perennial, palatable and productive grasses declined in abundance under heavy cattle grazing but increased under an intermediate level of grazing that was generated by macropods. Other native perennial grasses (NGP) that are palatable were decreasers, while unpalatable species were increasers. Exotic pasture grasses such as *Cenchrus ciliaris* and *Bothriochloa pertusa* were generally found to be increasers. Although *Melinus repens* showed a range of responses, Tothill and Hacker (1996) regard this grass as unproductive. None of the graziers involved with the present research regarded *Melinus repens* as having any grazing significance. Results from the present research indicated that, on some sites, productive exotic pasture grasses had a higher level of persistence under grazing than do their native equivalents.

A note of caution must be applied in recommending the use of certain exotic pasture grasses. In particular, *Cenchrus ciliaris* has been widely criticised for its potential to invade and diminish pasture diversity (Fairfax and Fensham 2000, McIvor 1998, Landsberg *et al*. 1997a). The present research suggests that increased domination by *Cenchrus ciliaris* led to a decline in species richness and the abundance of certain native grass species that were of dietary significance to the endangered northern hairy-nosed wombat. It is not possible to make any unifying comments about the impacts of all native grasses due to their broad ranges of grazing tolerance and impacts on species richness, as was previously explored in Sections 4.3.3 and 4.4.3.

6.4.4 Use of Functional Groups

Using the terminology suggested by Lavorel *et al.* (1997), this research combined emergent groups (life forms) with strategies (annual/ perennial) to locate specific response groups (increasers/decreasers). As predicted, there was a range of responses from the functional groups assembled for this research.

Seven of the 15 specific functional groups showed no identifiable response to grazing. Two groups (Exotic Vines –EV, and Exotic Woody Plants - EW) had sample sizes too low for accurate statistical analysis. The use of these functional groups would have greater application in areas where densities of these plants are obviously more prominent. Ecological patterns are undetectable in very rare species as their occurrence may simply be a matter of chance (McIntyre & Filet 1997).

Using growth form to cluster species as recommended by McIntyre *et al.* (1995) allowed for species to be easily assigned to groups, with the exception of some species that could be described as being either taller woody forbs or smaller woody shrubs. However, comparing functional groups by life form showed no consistent trends. For example, if growth form alone were used, this would combine the grass groups EG, NG3P, NGP and NGA, all of which showed different responses. Even by adding the further attribute of life history (annual and perennial), there still remained a high degree of variation in response to grazing. When examining the raw data to locate the source of this variation, it became clear that variations in palatability were the likely cause. This supports statements by McIvor (1998), Partridge (1995), Harper (1969) and Isbell (1969) that grazing impacts on plant species are partially related to palatability. The present research disagrees with Noy-Meir *et al.* (1989) that plant growth form was more important than palatability in determining if a species was an increaser or decreaser. Despite the fact that palatability is recognised as an important factor in plant community dynamics under grazing, it was not included in the division of functional groups in the present research due to a lack of palatability information for most species. McIntyre *et al.* (1995) also cited the paucity of palatability information as being a hindrance to developing palatability-based functional groups.

The varied responses of the functional groups identified from this research (increasers, decreasers all apparently neutral) support some of the concepts of competition and disturbance specialists. Decreasers may often equate to competition specialists, able to persist at highest densities in undisturbed areas where competition from other plants is the main influencing factor on plant growth. Increasers are disturbance specialists, benefiting from grazing that open gaps in a matrix dominated by perennial competition specialists. If the relationship between competition and disturbance are seen as opposing forces shaping plant communities, then several functional groups should be placed mid-way since they were benefited by intermediate disturbance (macropod grazing) but decline at either ends of the competition-disturbance spectrum. Results indicating that many species are advantaged by intermediate disturbance concur with comments by Petraitis *et al*. (1989) that increased diversity is a result of a balance between the rate of disturbance and the rate of competitive exclusion.

Landsberg *et al*. (1997a) looked for trends in abundance between native and introduced species. Their results were that most decreaser species were native and that increasers included both natives and introduced species (Landsberg *et al*. 1997a). In the present research, ‘introduced’ plants showed no consistent response to grazing due to differences in palatability within that group. While exotic grasses (EG) and exotic forbs (EF) were generally increasers, most exotic legumes tended to decrease under grazing. Although Landsberg *et al*. (1997a) found very few introduced species that behaved as decreasers, this is possibly due to the fact that most of the introduced plants recorded in that research were deliberate introductions for the grazing industry, in contrast with the present research which recorded numerous pasture improvement plants in the “Introduced plants” category. Amongst native plants, some groups increased (NFA, NL, NWP) while others decreased (NG3P, NLV). Native plant functional groups such as NGA and NFP showed conflicting trends on different sites depending on the palatability (or other aspect of preference by grazers) of the most common representatives.

More precise functional groups could be constructed with greater information relating to the palatability or preference of many species to grazers. Understanding this aspect of plant ecology may be essential in predicting the population dynamics of those species. For future research using functional groups, it is recommended that a hierarchical approach be taken, separating plants first on life form (forbs, grasses and shrubs), secondly on life history strategy, and finally on palatability/ preference to grazers.

CHAPTER 7: IMPACTS OF GRAZING ON GROUND COVER

As noted in Chapter 6, grazing results in certain changes to plant community composition, impacting on the carrying capacity of a particular pasture. In the present chapter, changes in the amount of plant cover and bare soil will be examined.

7.1 IMPORTANCE OF GROUND COVER AND EVIDENCE OF DECLINE UNDER CATTLE GRAZING

Apart from the obvious benefits of high ground cover to cattle production, there are also many other effects of ground cover on ecosystem processes such as water infiltration, erosion prevention and microclimate. Ground cover is considered to be an integral part of land condition. McIvor (2001) used a site condition value to compare sites, calculated from herbage yields and botanical composition, but recommends using an index based on the proportion of desirable species combined with level of ground cover.

The most clearly recognisable impact of grazing by cattle is that they reduce herbage mass through the ingestion of plant material, which may reduce ground cover (Fensham 1998). Scanlan *et al.* (1996b) observed that perennial tussock grasses were not removed *per se*, but declined in basal area leading to an increase in bare patches. Vegetation cover prevents erosion by protecting the soil from raindrop impact, increasing surface roughness (in turn reducing volume and velocity of runoff) and increasing organic matter in the soil (Williams *et al.* 1993). A high soil organic content affects cation exchange capacity, improves structural stability, provides a substrate for soil dwelling organisms, and influences nutrient availability (Williams *et al.* 1993). These factors promote soil stability through the direct effects of increased water infiltration on plant growth and productivity (Williams *et al.* 1993). Infiltration of water into the soil is higher in the presence of perennial grasses than with bare soil due to channelling of water into the soil by stem flow and macropores in the soil (Williams *et al.* 1993). A loss of these perennial grasses will therefore reduce infiltration and increase runoff (Williams *et al.* 1993). The loss of ground cover and topsoil may also lead to salt accumulation (Cisneros *et al.* 1999).

Chapter 2 previously discussed the effect that cattle have on community composition through selective grazing behaviour and through the different coping mechanisms of the various plant functional groups to that disturbance. Treading or hoof impact by cattle also has an impact, in terms of the physical properties of soil and pasture production. Livestock treading increases bulk density and surface roughness of soil, with a related decline in aggregate stability, soil penetrability and herbage production (Mullen *et al.* 1974). Brown & Evans (1973) assert that “All treading damages pasture irrespective of soil type, soil moisture level, plant species or kind of animals”, and also comment that the different levels of treading sensitivity between plant species result in plant community simplification under treading.

Although McIvor *et al.* (1995a) suggests that herbage production may increase, decrease or remain constant despite declining pasture condition, Ash & Brown (1997) report that reduced herbage led to low ground cover and to soil loss, especially when ground cover fell to below 40% or when surface sealing occurs. Extreme situations occur when bare areas known as scalds develop. Scalds result in a tenfold reduction in water infiltration, leading to runoff as high as 30% from heavily grazed pastures (Mott & Tothill 1994). This is attributable to poor water entry into the soil (Williams *et al.* 1993). Dry conditions persist under this sealed surface, making conditions unfavourable for soil organisms, while the high mechanical strength of the surface prevents seedling emergence (Williams *et al.* 1993). There is little evidence of plant establishment within scalded areas and so such areas are naturally reduced in size only through the spread of vegetative cover from their edges (Scanlan *et al.* 1996a). Low rates of seedling establishment on scalds can be attributed to seed removal during periods of high water runoff (Williams *et al.* 1993). These scalds can be abundant in heavily utilised pastures. DeCorte *et al.* (1994) recorded that in the Dalrymple Shires, scalds occurred in 37% of inspection sites on Dalrymple (gold field) soils and 40% of sites on black soils (vertosols). The worst-case scenario with scalding and erosion is desertification. A reduction in ground cover by cattle grazing during droughts, which exposes topsoil to erosion by wind and water, is considered to be a leading cause of desertification (Ludwig & Tongway 1995).

Soil loss is widespread in the Dalrymple Shire, with sheet, rill and gully erosion observed in 40% of sites examined by Rogers *et al.* (1999). Erosion is particularly bad since it dramatically reduces the resilience or ability of an ecosystem to return to equilibrium following a disturbance event (Williams *et al.* 1993). On several soil types, the potential for soil erosion to develop is dependant of the exposure of the highly dispersive subsoil layers, so maintaining a stable surface is essential to preventing erosion (Rogers *et al.* 1999). A high ground cover of perennial plants assists in maintaining the integrity of a stable soil surface. The removal of that plant material by grazing leads to a decreased ability to capture local runoff, water entrained sediments and nutrients (Freudenberger & Palmer 1996, Gardener *et al.* 1990). Exposure of the subsoil by cattle grazing and cattle pads will often lead to gully erosion (Rogers *et al.* 1999). Numerous scalds and erosion gullies were encountered and recorded in the present research project.

There have been numerous studies examining changes in ground cover in relation to cattle grazing. Studies from several states in the USA found that some sites had increased cover with cattle exclusion; while on other sites cover decreased (Ryerson & Parmenter 2001). Popolizio *et al.* (1994) found that research plots in Colorado that had been protected from grazing for 30 years then exposed to short term grazing, had increased foliar cover due to stimulation of growth of forbs, grasses and sedges. In an experiment in the arid Negev Desert in Israel, Saltz *et al.* (1999) used Landsat 5 satellite imagery to determine that grazing by Asiatic wild asses (*Equus hemionus*) led to a 30% decline plant cover.

Despite a number of international research projects into the impacts of grazing on ground cover, it may not be valid, however, to extrapolate results such as these to Australia, since the response of vegetation communities to grazing differs depending on the soil, climate and evolutionary history (Ryerson & Parmenter 2001).

Research into grazing impacts on ground cover in north Queensland has not been as extensive. McIvor (2002) noted different responses of ground cover to grazing depended upon pasture yield. At sites with pasture yields below 3000 kg/ha, cover levels increased with stocking rate, but on sites with higher pasture yields there was little difference between stocking rates (McIvor 2002). Remote sensors of land degradation from grazing obviously recognise the sometimes enigmatic responses of ground cover to grazing. Pickup *et al.* (1994) instead combine several parameters including ground cover, trends in cover variance with distance from water, and the size of ground cover response at individual locations compared with what would be expected from an entirely ungrazed plant community.

Scanlan *et al.* (1996a&b) examined a number of the same exclosure sites used in the present research project. They recorded that standing dry matter declined in all treatments including exclosures during the drought of 1985 to 1988, but that following good seasonal conditions from 1987 to 1990, rapid increases in yield occurred. Over the period from 1985 to 1990, Scanlan *et al.* (1996a) noted higher yield and percentage vegetative cover in ungrazed plots than in the grazed plots. An increase in bare ground was partially attributable to a grazing–related decline in basal area of perennial grasses; however, those impacts were more noticeable where the dominant grass was *Heteropogon contortus* than in sites dominated by *Bothriochloa pertusa* (Scanlan *et al.* 1996b). It was also noted that there were significantly more scalds in grazed areas than ungrazed, and that there were more scalds in 1988 than there were in 1990 (Scanlan *et al.* 1996a). Significant loss of vegetative cover occurred during the drought of 1992 to 1994. Photographs taken during this drought indicate an almost complete absence of herbaceous cover in areas adjacent to some of the QDPI exclosures (O'Reagain P. pers. comm.).

Scalding and erosion can occur as a result of increasing runoff, which, in turn, is reduced by high ground cover. Numerous factors influence water infiltration and runoff, including slope, soil saturation levels, microtopography, storm intensity and ground cover (Scanlan *et al.* 1996a). Intuitively, if rainfall exceeds the infiltration rate, for whatever reason, then runoff will occur. On an even slope during a high rainfall event, runoff decreases linearly with increasing cover, while low cover was associated with high soil loss and sediment concentration (Scanlan *et al.* 1996a). Monitoring of runoff in several of the QDPI exclosures found higher runoff from grazed than ungrazed plots (Scanlan *et al.* 1996a). McIvor & Gardener (1995) note that simply removing cover doesn't account for observed effects since there was a significant difference in the effects resulting from removal of cover by grazing or mowing. It is likely then that the impacts of cattle grazing on increasing runoff and erosion is the interaction of cover removal and treading impacts on soil structure.

The measurement of grass basal area (of perennial tussock or hummock grasses) as an index to assess pasture condition is frequently used (eg. Rogers *et al.*1999, Scanlan *et al.* 1996a). Considerable changes in grass basal area occur under different stocking regimes. For example, an increase in stocking rate from 0.2 head of cattle/ha^{-1} to 1 head of cattle/ha^{-1} reduced the grass basal area by 72% (Gardener *et al.* 1990). A reduction in basal area of perennial grasses does not necessarily mean a reduction in ground cover. Scanlan *et al.* (1996b) found that where perennial tussock grasses had been replaced by *Bothriochloa pertusa*, there was actually an increase in ground cover due to the low, spreading stoloniferous habit of that introduced grass, compared to the discrete upright clumps of *Heteropogon contortus* (Scanlan *et al.* 1996b).

7.2. METHODS

While numerous studies collected data on the basal area of tussock grasses, McIvor *et al* (1995a) suggest that measuring ground cover is a suitable substitute. This technique has also been found to be a sensitive early warning indicator of desertification Desoyza *et al*. 1998). Although the use of transects may underestimate plant diversity, Stohlgren *et al*. (1998) recommend transects as being ideal for assessing foliar and soil cover.

Data was collected from 13 sites, each of which had contrasting grazing treatments. Field trips were conducted during the wet season in accordance with Pickup *et al*. (1994). At each site, two treatments (grazed and ungrazed) had three replicate plots measuring 50 X 50 metres. On some sites, a third treatment (macropod exclosure) with a single plot was included. Macropod exclosures were not present at each site and were not replicated where they were present. Within each plot, 30 0.25m^2 quadrats were systematically thrown and the projected vegetative cover was estimated for each of the quadrats. The methodology for data collection was described in Chapter 3.

Rogers *et al*. (1999) measured only the basal area of perennial tussock or hummock grasses in assessing ground cover. This considers the most productive fodder species but does not take into consideration the value of other life forms in protecting soil from erosion. In this study, ground cover as a whole was estimated without discriminating the life forms or functional groups that comprised that cover. Projected cover included both those plants rooted within the quadrat and those projecting or laying over the quadrat. It did not include projected cover from overhanging trees. Data used in the analysis was the frequency of the different ground cover categories in each plot.

Within each quadrat, the total percentage projected cover was estimated and placed into a ranked category. These categories were:

1	<5% (scalds)
2	5-15%
3	15-30%
4	30-50%
5	50-90%
6	>90% (total cover)

Due to time limitations, actual yields were not measured for each of the 2,700 quadrats examined. At several sites, contrasts in pasture yield were obvious.

Although the dataset structure was suitable for a Chi^2 analysis, the assumption that there was an expected value of at least 5 for all categories could not be met as lower cover categories frequently had very low values. Although categories of lower ground cover could be combined to increase frequency values, this removed a considerable amount of detail of ground cover, including the presence of scalds. The small expectation values are, however, allowable in ‘exact tests’. These involve calculating the probability of every alternative table of values that have the same outcome as the observed table. In this case, a Monte-Carlo exact Chi^2 was conducted using SPSS version 10 was considered to be most appropriate (Steele M. pers comm.). Due to the lack of replication in the macropod exclosures, these analyses could only be made for grazed and ungrazed treatments. The null hypothesis in each of the Monte-Carlo exact Chi^2 tests was that grazing treatment has no influence on the frequency distributions of cover.

7.3. RESULTS

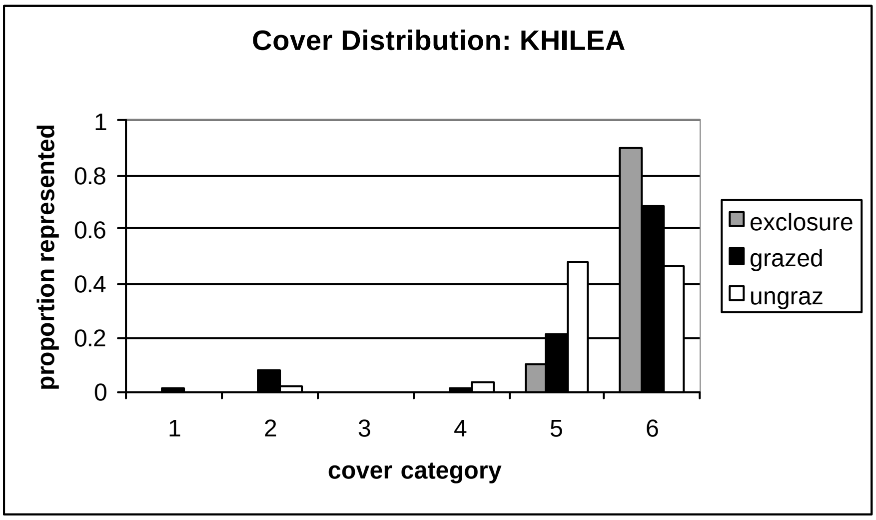
In ten of the thirteen sites analysed, there was a significant influence of grazing on the distribution of frequencies of ground cover categories. Histograms were prepared for each site illustrating the distribution of these categories. These histograms also allow comparisons to be made with macropod exclosures where relevant. These histograms are presented in Figure 7.1 (a-m) and exhibited a range of responses that are discussed below.

Of the ten sites that showed a significant difference between grazed and ungrazed plots, eight of these indicated to varying degrees that grazed plots had a higher proportion of quadrats with low ground cover and scalds. Ungrazed plots had a higher proportion of quadrats with high ground cover, while macropod exclosures, where applicable, tended to have the highest ground cover of all treatments. An example of this trend can be seen in Figure 7.1(c), showing the results for Blue Range.

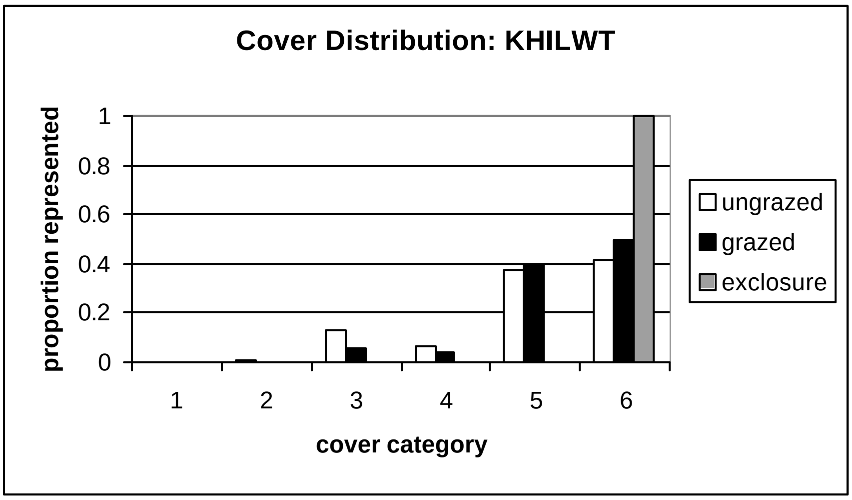
At Jervoise, (Figure 7.1.h.), grazed plots represented the broad spectrum of ground cover. Grazed plots had the greatest proportion of quadrats with the maximum cover, but also had the only quadrats with very low cover recorded. Ungrazed plots had no quadrats with low ground cover recorded.

Figure 7.1 a-m: Proportions of total quadrats with different ground cover categories for grazing treatment plots at all sites.
Significance levels (P) are provided for each histogram.

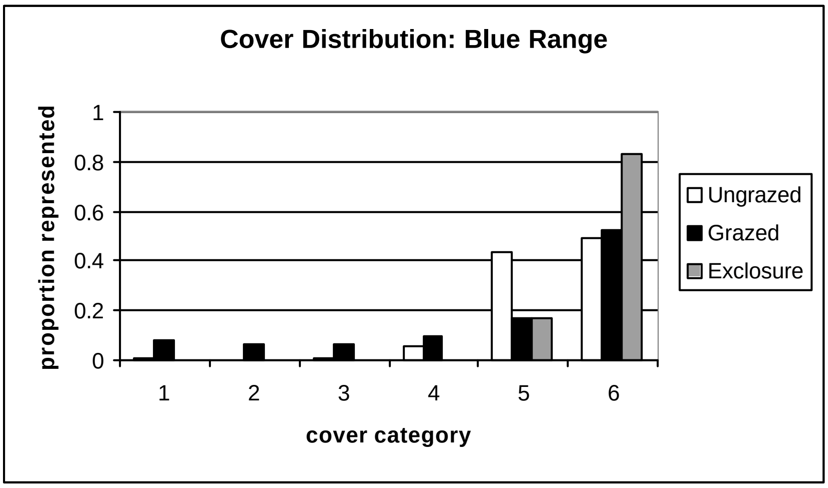
7.1.a: KHILEA (P= 0.0000)



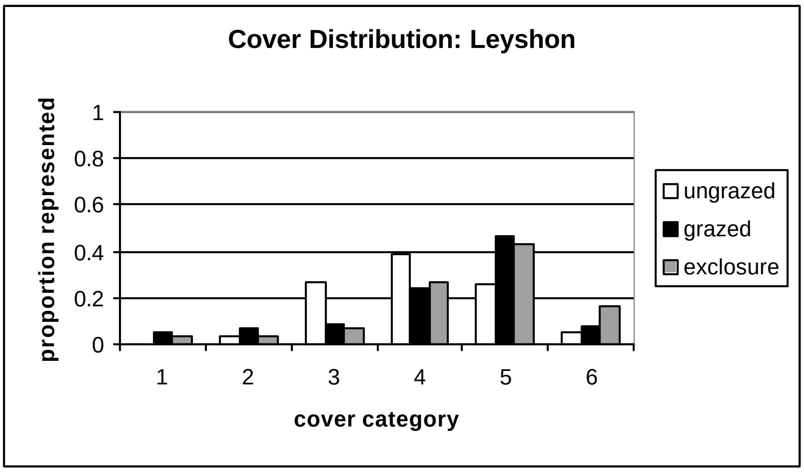
7.1.b. KHILWT (P= 0.0000)



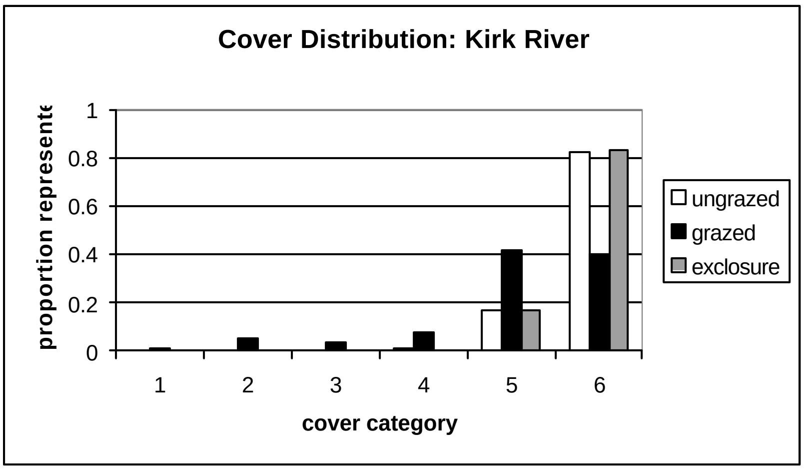
7.1.c. Blue Range (P= 0.0000)



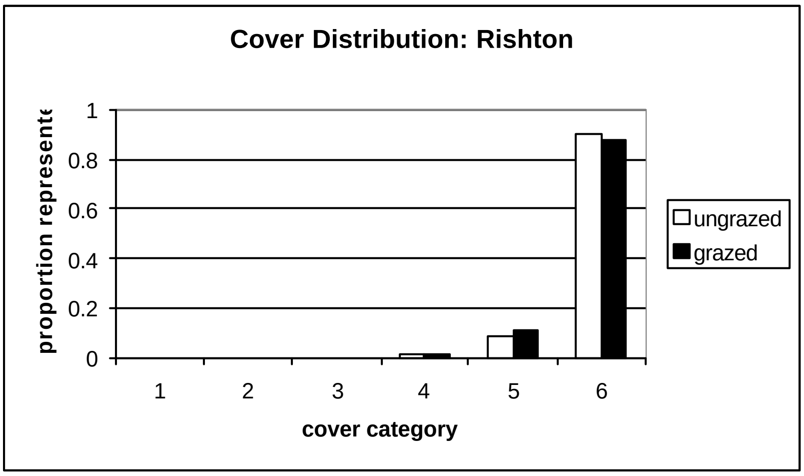
7.1.d. Leyshon View (P= 0.001)



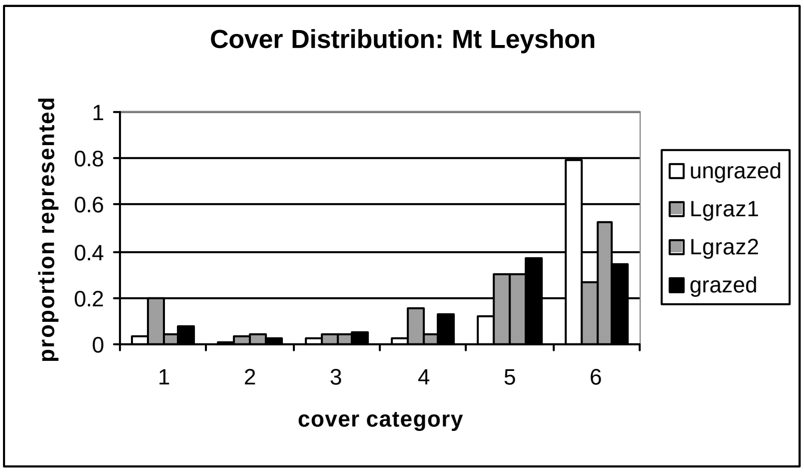
7.1.e: Kirk River (P= 0.0000)



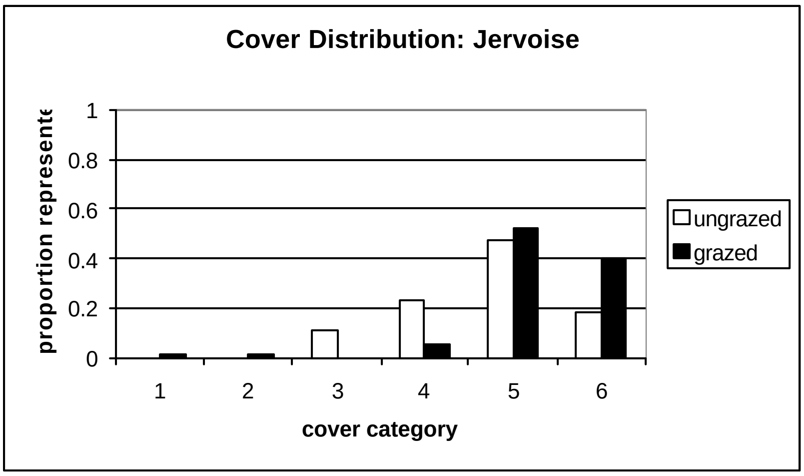
7.1.f: Rishton (P= 0.9 - not sig.)



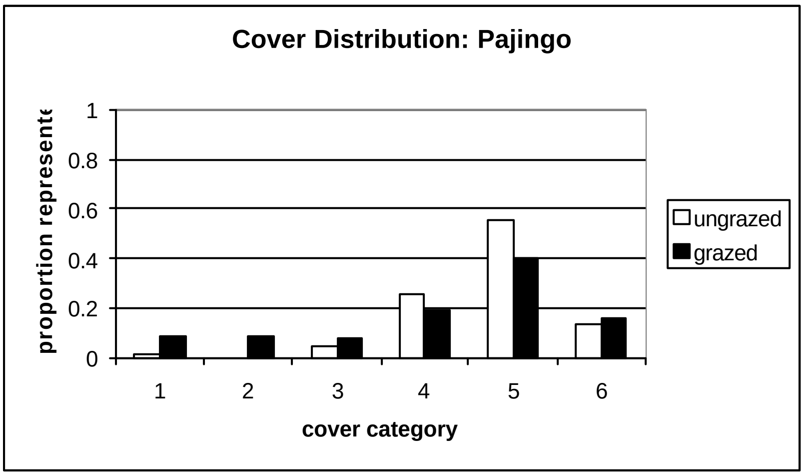
7.1.g: Mt Leyshon (P= 0.0000)



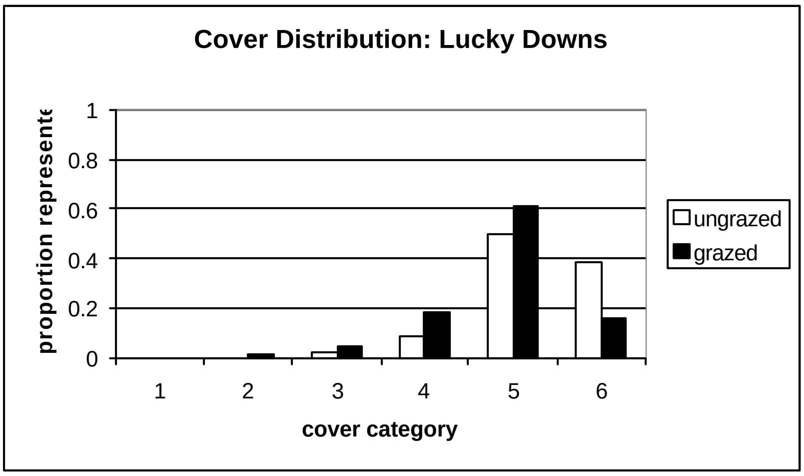
7.1.h: Jervoise (P= 0.0000)



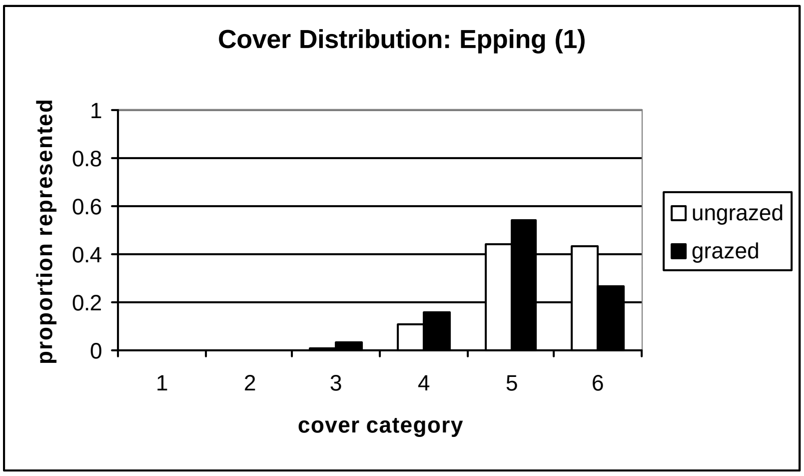
7.1.i: Pajingo (P= 0.003)



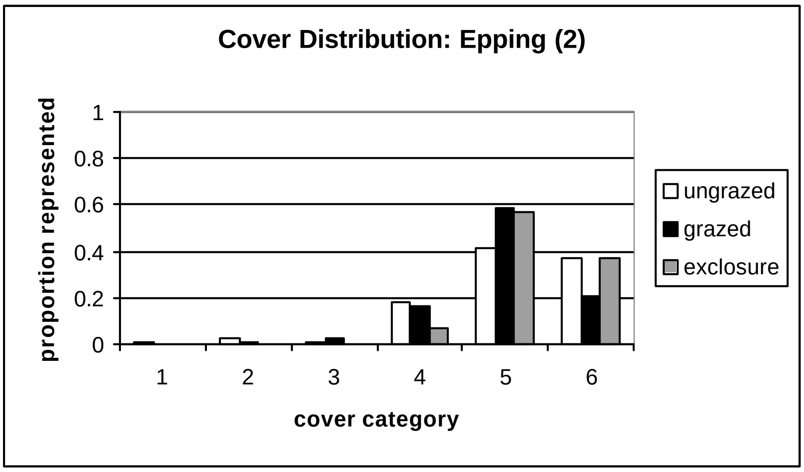
7.1.j: Lucky Downs (P=0.003)



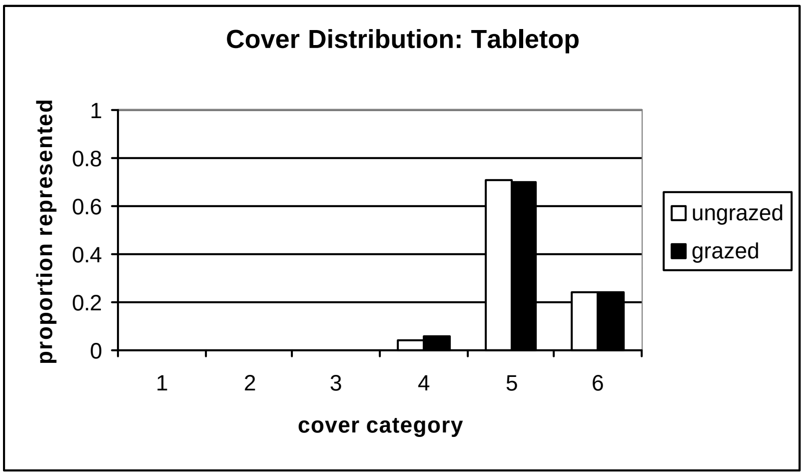
7.1.k: Epping 1 (P=0.097)



7.1.l: Epping 2 (P= 0.276 - not sig.)



7.1.m: Tabletop (P= 1.00 - not sig.)



At Mt Leyshon Mine, ungrazed plots had the greatest proportion of high ground cover and the lowest level of bare areas and scalds. Examination of Figure 7.1.g showed that continuously grazed plots had lower cover than ungrazed plots and that intermittently (dry season) grazed plots were comparable to grazed sites. At the lowest scale of cover, lightly grazed (1) plots had more scalds than did grazed plots.

At two sites (Kangaroo Hills West and Leyshon View), the trend was for ungrazed plots to have the lowest ground cover and for grazed plots to have a higher representation of high ground cover. Relative distribution of ground cover for these two sites is shown in Figures 7.1.b and 7.1.d below. At Kangaroo Hills, ungrazed plots had more quadrats with low ground cover. However, it should be noted that the macropod exclosure at this site had only the maximum ground cover recorded. At the Leyshon View study site, there was generally low ground cover overall. Grazed plots tended to have higher cover than ungrazed, and the peak of cover distribution for ungrazed plots was lower than for either grazed or exclosure plots. Although the macropod exclosure had the largest proportion of quadrats with the maximum ground cover, it also had representation of scalds; the only site in which this occurs.

Three sites (Rishton, Epping Forest 2 and Tabletop) showed no significant difference in ground cover between treatments. The Rishton (Fig. 7.f) and Tabletop (Fig. 7.m) sites had only high ground cover in both treatments. While there were no significant differences between grazed and ungrazed plots at Epping Forest 2 (Fig. 7l), the exclosure plot appeared noticeably different. Both grazed and ungrazed plots had noticeably more areas with low ground cover than in the exclosure plot.

7.4. DISCUSSION

Results presented in section 7.3 indicate that a significant loss of ground cover with grazing had occurred in several sites and that even dry season grazing had a deleterious impact on ground cover. Scalds were predominantly located in grazed and lightly grazed plots. This comparative loss of ground cover under cattle grazing confirms observations and statements made in a number of previous studies (eg. McIvor 2001, Fensham 1998 and Scanlan *et al*. 1996a).

A comparison between ungrazed plots and macropod exclosures showed that grazing pressure by macropods also reduced ground cover, and increased areas of bare ground. Grazing by macropods and wombats at Epping Forest (2) altered ungrazed plots so that they resembled grazed plots in respect to ground cover.

A loss of some ground cover in ungrazed plots was expected in light of the response of certain functional groups to that treatment (Chapter 6). The high levels of ground cover that occurred in macropod exclosure plots confirmed those treatments as zones of high competition. It was noticed that in these plots, there was a tendency towards fewer individual grass tussocks, but with much larger basal areas. A similar situation had been reported in north Queensland (McIvor 2001) and for the subhumid grasslands of Argentina (Milchunas *et al*. 1988). Although grass basal area was not recorded on the data sheet for the present research, site descriptions (Appendix A) show that grass basal area was higher in ‘ungrazed’ plots at Leyshon View, Kirk River, Jervoise and Tabletop. The ‘ungrazed’ plots had a level of ground cover intermediate between the total cover of exclosures and the frequent scalds of grazed plots, so confirmed their definition as areas of intermediate disturbance. It should be noted that in some instances a decline in ground cover can be associated with increase frequencies of certain plants that make little contribution to ground cover. For example, domination of a pasture by *Stylosanthes* species can lead to low rates of ground cover (Jones *et al*. 1997), however, this was not the case in any of the plots examined in the present thesis.

The levels of high ground cover observed at Rishton and Tabletop indicate that the predicted trend of declining cover with cattle grazing is not universal. The Rishton study site had very few significant results for functional groups in Chapter 6. The non-significant result for ground cover at this site may indicate that there was no significant release from competition in grazed plots. This suggests that ground cover had not been reduced sufficiently to benefit disturbance specialists like annuals. Conversely, comparison between grazing treatments at the Tabletop study site produced ten significant results for functional groups, despite showing no significant difference in ground cover. Although both grazed and ungrazed treatments had similar ground cover, there was a noticeable difference in height of herbaceous vegetation (Plate 1.d). Ungrazed plots had thick swards of *Heteropogon contortus* 1.5 metres high that heavily shaded the soil, while grazed plots resembled a mown lawn rarely taller than 10cm in height, dominated by forbs and smaller grasses. It is likely then, that competition is an expression of vegetation height and shading as well as the percentage of ground covered.

The higher levels of ground cover observed in grazed plots at Leyshon View can be attributed to a higher abundance of Indian couch (*Bothriochloa pertusa*). In Chapter 4, it was shown that grazed plots had a significantly higher abundance of *Bothriochloa pertusa* than did ungrazed plots. This result supports observations by Scanlan *et al.* (1996b) that an increase in ground cover can occur under grazing due to the low, spreading stoloniferous habit of *Bothriochloa pertusa*, compared to the upright tussock habit of decreaser grasses. This particular spreading characteristic is common in grasses from areas with a long evolutionary history of grazing and is known to result in increased ground cover in numerous overseas grazing studies (Milchunas *et al.* 1988). In terms of reducing scalding and erosion, exotic *Bothriochloa pertusa* has obvious benefits since it reduces the rate of runoff (Scanlan *et al.* 1996a) and increases under certain levels of grazing pressure.

While an increase in *Bothriochloa pertusa* was responsible for an increase in ground cover with grazing at Leyshon View, this species was absent from the Kangaroo Hills West study site, so could not be used to interpret the patterns of ground cover at that site. It is possible that this result was related to patterns of abundance of red Natal grass (*Melinus repens*). This species had its highest abundance in the exclosure plot, followed by grazed plots, with ungrazed plots having the lowest abundance. This pattern of *Melinus repens* abundance matched the pattern of cover abundance.

In conclusion, these results showed that grazing reduced ground cover except where that grazing encouraged the growth of *Bothriochloa pertusa*, which, because of its morphology and tillering habit, actually increases ground cover. Ground cover could only be used as a predictor of grazing pressure and pasture condition if the abundance of these species is taken into consideration.

CHAPTER 8: GRAZING IMPACTS ON TREE DIEBACK

8.1 THE VALUE OF TREES IN SEMI-ARID RANGELANDS

Although spatially rare in terms of frequency of occurrence in quadrats, trees were nevertheless a significant component in most of the sites examined in the present research. It was noted, however, that in many areas, large numbers of trees had recently died in a mass mortality event commonly referred to in the literature as 'die-back'. Dieback is a term used to describe the premature decline and death of trees usually as a consequence of environmental stress, both biotic and physiochemical (Wylie & Bevege 1980). In particular, it was noted during the present research that large numbers of narrow-leaved ironbark (*Eucalyptus crebra sensu lat*.), silver-leaved ironbark (*Eucalyptus shirleyi*) and Reid River box (*Eucalyptus brownii*) had succumbed.

Large-scale dieback events should be regarded as a source of concern. Fensham & Holman (1999) note that the woody plant component of our ecosystems forms a major part of Australia's carbon sink, and that accounting for changes in that sink may soon become an international obligation under guidelines to ameliorate the impacts of the greenhouse effect. Extensive tree dieback means less sequestration of carbon in long-lasting sinks (Burrows 1993).

Biologically, the woody plant component of savannas contributes significantly to the biodiversity, structural complexity and spatial heterogeneity of those ecosystems (Belsky 1994). Grime (1998) suggests that the influence a plant has in its environment is proportional to its biomass. Although trees were found to be spatially rare (in relation to the probability of encountering a trunk within *x* number of quadrats), they account for a considerable proportion of biomass in typical savanna woodlands. Thus, according to the theory by Grime (1998), they must exert a major influence over the ecosystem processes of that community. For land managers, the loss of tree cover also has numerous disadvantages. These include a loss of drought and browse reserves, increasing salinisation risks (in salt prone areas), increased erosion hazards, reduced access to soil nutrients at depth, promotion of understorey woody weeds or undesirable regrowth, reduced shade and shelter for stock, fragmentation of wildlife habitat (with associated impacts on biodiversity), and the loss of usable timber for building (Burrows 1993). Although they note that deliberate tree clearing is a common practice of graziers to promote a greater production of grasses and increase animal production, Anderson & Dowling (1987) recommended leaving strips or clumps of trees to prevent the negative impacts often associated with over-clearing. The widespread dieback of trees also reduces tree cover, but it does so in such a way as to reduce the potential for strategic retention of trees.

Any suggestion of a link between this dieback event and cattle grazing is considered controversial to say the least amongst the cattle grazing fraternity. In the present chapter, the potential relationship between grazing impacts and tree dieback are examined.

8.2 PREVIOUS RESEARCH INTO CAUSES OF TREE DIEBACK

The phenomenon known as ‘dieback’ is widespread throughout the world, including Australia. Landsberg *et al*. (1990) note that Eucalypt dieback is widespread along the east coast of Australia from Queensland to Tasmania and also in southwest Western Australia. Numerous authors have attempted to establish cause and effect and had found that the causes for the observed mass mortality are diverse. Although Fensham (1998) views dieback as a natural pre-European phenomena, Landsberg *et al*. (1990) believed that other forces are partially responsible. In some overseas examples, dieback may be attributed to pathogens such as Dutch elm disease and white pine blister rust (Mueller-Dombois 1986). In Western Australia, large stands of jarrah (*Eucalyptus marginata*) forest have been devastated by the fungal pathogen *Phytophthora cinnamomi* (Boland *et al*. 1984). This disease occurs in all states of Australia and causes the death of plants in all vegetation strata over a large area (Boland *et al*. 1984), however, there has not been any evidence to suggest that such a pathogen could be responsible for the Eucalypt dieback in north Queensland. Eucalypts suffering from dieback at Princess Hills and Mt Spurgeon have been tested for *Phytophthora cinnamomi* but these tests proved negative (P.Gadek pers.comm.).

Insect damage (phytophagy) is the cause most frequently implicated in dieback in Australia (Anderson & Dowling 1987, Heatwole & Lowman 1986, Landsberg & Wylie 1983). Repeated defoliation by insects exhausts starch / carbohydrate reserves in the roots, reducing the ability of the tree to produce new leaves, and finally causing death (Heatwole & Lowman 1986, Landsberg & Wylie 1983). Rural trees live in a delicate and precarious balance with insect herbivores, which can be upset by either changes in insect populations or by enhanced nutritional quality of eucalypt leaves (Landsberg & Wylie 1983). Landsberg *et al*. (1990) examined dieback on temperate

Eucalypt woodlands in New South Wales and found that both defoliation by native insects and associated dieback were more severe in heavily grazed pastures than in ungrazed areas. Testing several hypotheses, Landsberg *et al.* (1990) concluded that pasture improvement or uneven redistribution of nutrients by grazing animals under trees increased soil fertility, thereby increasing the nutritive value of eucalypt foliage, leading to an increase in defoliating insects. Numerous insects have been implicated as causing severe defoliation, including psyllids, scarab beetles, chrysomelid beetles, leaf hoppers, phasmids (stick insects), leaf hoppers, sawfly larvae, scale, gall-forming insects and skeletonising caterpillars (Nadolny 1995). In the Dalrymple Shire of North Queensland, *Eucalyptus crebra* can be severely defoliated by Christmas beetles (*Anoplognathus* spp – Coleoptera: Scarabaeidae), which occur in large numbers on the trees in November (Jackes B. pers. comm.). The defoliating effect of this genus on eucalypts have also been noted by Lawrence and Britton (1991), Nadolny (1995), and Landsberg & Wylie (1983).

One of the most commonly suggested causes of dieback for the tropical semi-arid woodlands has been the influence of drought (Fensham & Holman 1999, Fensham 1998, Whalley 1973). Droughts of varying severity are a recurrent and dominating force in the ecology of the semi-arid rangelands, driven to a large degree by the El Nino Southern Oscillation (Fensham & Holman 1999). Drought-induced dieback has been recorded in southeast Queensland (Bell 1999), from southeast Queensland to the Victorian border (Pook *et al.* 1965), Victoria (Ashton *et al.* 1975), Tasmania (Kirkpatrick & Marks 1985) and South Africa (Scholes 1985). Heathcote (1973) recorded numerous droughts impacting on the area examined in this research occurring in the years 1888, 1902, 1905, 1915, 1919, 1923, 1926 and 1961, interspersed with another seven periods of lower than average rainfall. Scanlan (1996a) notes that drought again hit the Charters Towers area in the years 1985-1987, while Fensham & Skull (1999) described the drought of 1992-1994 as having the lowest rainfall recorded in 113 years at Charters Towers.

Fensham & Holman (1999) offered evidence that extensive tree dieback had occurred following many of these droughts, both in Australia and Africa, with records of mass tree mortality following drought extending as far back as the 1850's in Queensland. Fensham & Holman (1999) observed that, as a result of the 1992-1994 drought, 29% of trees were either dead or nearly so over an area of 55,000 km^2. In particular, Fensham (1998) demonstrated that considerable mortality of ironbarks (more than 25%) occurred as a result of the extensive 1992- 1994 drought, regardless of grazing pressure. After revisiting that site, Fensham & Holman (1999) subsequently upgraded that dieback figure to 69% for ungrazed areas as many moribund trees had since died.

Whalley (1973) explains the mechanisms by which drought kills trees. Absorption of water by tree roots depends on the hydraulic conductivity of the soil and the amount of water present. As the soil dries, both of these characteristics are reduced and the tree’s transpiration-driven water deficit cannot be replenished from the soil. As a result, permanent wilting occurs, leading to tissue damage and finally death (Whalley 1973). Plant species that occur naturally in drought-prone areas usually possess mechanisms to either increase their water uptake mechanisms or to reduce the transpirational water loss through their leaves (Whalley 1973). Despite the species constituting these semi-arid plant communities being well adapted to that region, there is still a marked differential susceptibility between species. Closely related Eucalypt species show a similar susceptibility to drought, while there are noticeable differences with other distantly related species (Whalley 1973). Alternatively, while Fensham (1998) noted that the frequency of dieback in the ironbark *Eucalyptus xanthoclada* was higher (29%) than in *Corymbia dallachiana* (4%), he noted that the difference was not statistically significant. As suggested by Kirkpatrick & Marks (1985), it may be that the distribution ranges for plant species are limited by these uncommon, rather than common environmental events.

The role of drought in causing dieback is not entirely agreed upon in the literature. Fensham & Holman (1999) note that dieback of ironbarks in North Queensland was substantially patchy, but this was poorly correlated to local rainfall patterns. However, Pook *et al*. (1966), Scholes (1985) and Kirkpatrick and Marks (1985) all note that dieback patchiness following drought is common and generally related to soil type, specifically the influence of soil volume (depth) on soil moisture content. Clearly, drought cannot be the prime factor responsible in all cases of dieback. Landsberg & Wylie (1983) measured xylem pressure potential levels (Ψ) in trees affected and unaffected by dieback in the Brisbane hinterland. Although Ψ levels are used to determine the water status of a tree, they found no correlation between Ψ levels and dieback of eucalypts.

In some cases, a perturbation (such as drought) may simply be the *coup-de-grace* for a weakened stand of trees (Mueller-Dombois 1986). The theory of cohort senescence as defined by Mueller-Dombois (1986) suggests that a cohort of trees progresses through various life stages until senescence causes a susceptibility to an abiotic perturbation, triggering a stand level dieback. Although Fensham (1998) only divided his trees into two size classes, he found no significant difference in mortality between the two classes. Conversely, Scanlan *et al*. (1996a) found significantly higher mortality in *Eucalyptus* saplings less than 1 metre in height than for taller specimens. Fensham & Holman (1999) examined several species and found that only the *Eucalyptus crebra/ E. xanthoclada* complex showed a significant difference in mortality between size classes, with significantly more large trees dying than smaller sized trees.

Although ample evidence exists for a relationship between drought and dieback, it remains to be shown whether grazing management may or may not exacerbate the extent of such mass mortality. Fensham & Holman (1999) observed that although dieback across the north was substantial, it was not uniform, however, they attributed this to differences in soil type and also suggested the density-dependant competitive effects of neighbouring trees may have had an influence. Although rejecting it as a possible cause, Fensham (1998) did explore the possible combination of grazing and drought as having an increased impact. It has been suggested that the most likely contribution of cattle grazing towards tree dieback is through effects on soil and water infiltration. Fensham (1998) notes that cattle grazing reduce ground cover, which decreases water infiltration, so therefore less water reaches the roots of the trees. Williams *et al*. (1993) state that reducing plant cover results in a decreased retention of surface water and decreased infiltration rates, so that volume and velocity of runoff is increased. Infiltration of water into the soil is higher with perennial grasses than bare soil due to channelling of water into the soil by stem flow and macropores in the soil (Williams *et al*. 1993). Mott & Tothill (1994) also warn of the deleterious effects of scald formation following heavy grazing. These scalds can form in as little as two years with a ten-fold reduction in water infiltration resulting (Mott & Tothill 1994). They note that up to 30% of rainfall can run off heavily grazed pastures, and this must have an impact on the vegetation of that pasture. Photographs taken by Peter O'Reagain (QDPI) indicate an almost complete absence of herbaceous cover in areas adjacent to the research exclosures; however, Scanlan *et al*. (1996a) observed that during the drought years 1985-1987, standing dry matter was low even in the grazing exclosures.

Fensham (1998) conversely noted that grazing may also increase available moisture by reducing competition from the herbaceous layer. Considerable research has been conducted into the competitive interactions between trees and grass, a selection of which is reported here. Anderson & Dowling (1987) see this competition between grass and trees as being important, noting that only a few trees per hectare can significantly depress grass production. Burrows (1993) observes that competition for water between grass and trees occurs mostly during the wet season as herbaceous plants are generally dormant through their natural senescence in the dry season. Although tropical savanna trees were thought to reduce understorey plant productivity (through competition for light, water and nutrients), Belsky (1994) reports that numerous other studies indicate an increase in understorey productivity around isolated trees. Winter *et al*. (1989) concluded that the impact of tree killing on pasture production was a very minor one in comparison to managing stocking rates and was of value only when feed was scarce. It should be noted, however, that savanna response to tree clearing shows considerable site-specific variation (Mott & Tothill 1994).

The observation by Scanlan *et al*. (1996a) that a significantly higher number of trees died in the grazed plots than in the ungrazed plots contrasts with the findings by Fensham (1998) that there was no significant difference. Fensham (1998) does, however, note that his comparison was done between lightly and moderately utilised pastures and so cannot rule out the possibility of heavy grazing having an impact on dieback. Scanlan *et al*. (1996a) actually recommend heavy grazing to increase woody plant mortality to decrease competition with grass production. Fensham & Holman (1999) conclude their research by stating that there is no evidence to suggest that dieback is exacerbated by cattle grazing, and that even in the absence of soil compaction or grazing pressure, substantial dieback still occurred. It was evident from an examination of the literature that conflicting ideas about the role of cattle grazing in tree dieback exist, and the present research attempts to provide further empirical evidence to help resolve the debate.

8.3 METHODOLOGY FOR COLLECTING AND ANALYSING DATA

Several approaches have been taken in collecting data on sparse woody plant vegetation. In terms of assessing woody plant cover, Landsberg *et al.* (1997a) used a rapid assessment with Bitterlich wedges, counting all species with a height over 50cm. However, to expand on the work conducted by Fensham (1998), it would be necessary to use a more precise method than Bitterlich wedges, such as recording all individual trees by species, trunk Diameter at Breast Height (DBH) and mortality ranking (alive or dead). In the present research, populations of trees were examined in 50 X 50m plots at five sites with exclosures erected by QDPI in 1986. These sites were Kangaroo Hills (east), Kangaroo Hills (west), Blue Range, Leyshon View and Kirk River and were sampled at the same time as the herbaceous layer was sampled. The five sites examined had a high occurrence of dead ironbark trees, which local knowledge suggests died during the 1992-1994 drought. Plots at these sites represent the treatments ‘grazed’ (continuous grazing by cattle and other herbivores), ‘ungrazed’ (cattle were excluded, but macropods and other herbivores could still enter), and ‘exclosure’ (all large herbivores were excluded). All trees within these plots with a height exceeding two metres were identified, noted as dead or alive and had their DBH measured. Due to the relatively tall height of the grass sward, it was not considered feasible to locate and measure small saplings less than 1 metre in height, as did Scanlan *et al.* (1996a) during periods of less grass growth. It was noted, however, that in none of the plots examined were dead small saplings obvious. Note that other sites excluded from this analysis were not used as they were only very lightly wooded and no substantial data set could be obtained from a very limited number of trees.

The raw data was sorted into ironbarks and ‘other species’ for analysis. It is possible that the category ‘ironbarks’ is a composite of three species: *Eucalyptus crebra, Eucalyptus whitei* and *Eucalyptus xanthoclada*. These species often looked very similar, especially when dead, and were all noted as being susceptible to dieback, supporting the notion by Whalley (1973) that closely related Eucalypt species show a similar susceptibility to drought. Fensham & Holman (1999) also combined the Ironbark species *Eucalyptus crebra* and *Eucalyptus xanthoclada* in their research on dieback. Trees commonly encountered and included into the category ‘other’ include *Corymbia dallachiana, Maytenus cunninghamii* and *Zizyphus mauritiana*.

Trees that had multiple trunks had their DBH calculated by:
$\mathrm{DBH} = \surd \mathrm{dbh}_1^2 + \mathrm{dbh}_2^2 + \mathrm{dbh}_3^2 \ldots$

Although Fensham (1998) treated multi-stemmed trees as a single individual and assigned the specimen to the class size of the largest trunk diameter, the use of the formula above gives an approximation closer to the real cross-section area of trunk (Congdon R. pers. comm.).

For the statistical analysis of treatment and size effects, trees were divided into two size classes <11 cm DBH and >11cm DBH, in accord with Fensham (1998) and Fensham & Holman (1999) and suggestions by a statistician (Steele M. pers. comm.). The proportion of the total ironbarks that were dead was calculated for each plot. Plots were deleted from the analysis where a division by zero error occurred. The resultant proportional data were then transformed by arcsine {√ proportion trees dead} to satisfy the assumption of normal data distribution, and analysed using a nested ANOVA. The treatments ‘grazed’ and ‘ungrazed’ were compared, with the exclusion of trees in the exclosure from the analysis being necessary due to a lack of replication. Results from the exclosure are compared descriptively.

For the generation of size class histograms, trees were sorted into five size classes. The following size class categories were used for construction of histograms:

Category 1	0-10 cm DBH
Category 2	11-<21 cm DBH
Category 3	21-<31 cm DBH
Category 4	31-<41 cm DBH
Category 5	>41 cm DBH

Histograms were generated for dieback patterns in ironbarks, combining all plots for a single treatment together and transforming the size class frequency data into a percentage of the total for that grazing treatment.

8.4 RESULTS OF DATA ANALYSIS

Tree species other than ironbarks were encountered less frequently and showed comparatively low levels of mortality. Results of the nested ANOVA of transformed data for ironbarks are shown in Table 8.1below.

Table 8.1: Interaction effects of grazing treatment and trunk diameter on dieback of *Eucalyptus crebra* at five study sites
(NS = Not significant).

	K.Hills East	K.Hills West	BlueRange	Leyshon V.	Kirk R.
Grazing	NS	P=0.0422	NS	P=0.0203	P= 0.0116
Size (DBH)	P=0.0000	NS	NS	NS	P= 0.0003
Grazing*size	NS	P=0.0700	NS	NS	NS

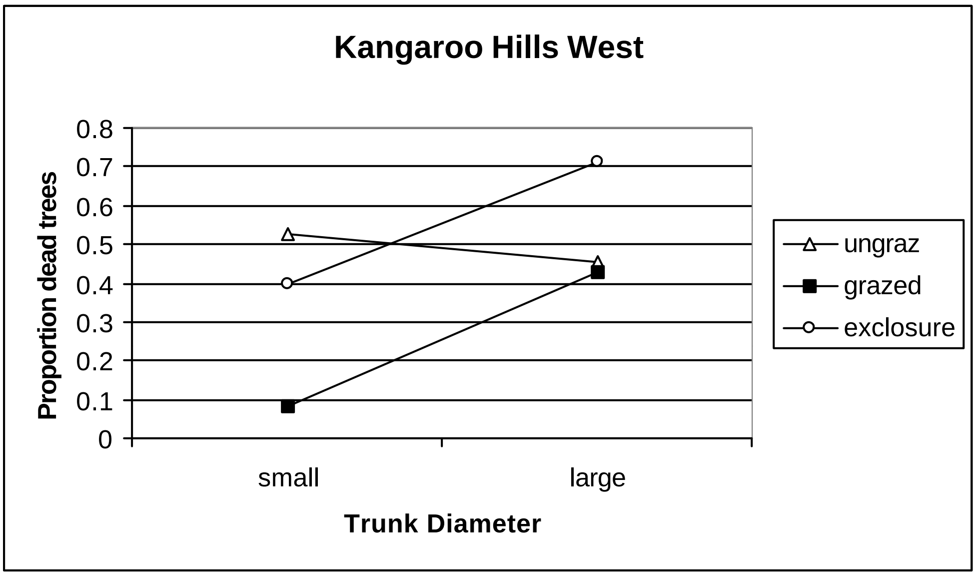
probability level: P <0.1

Table 8.1 above shows six significant results. Three sites (Kangaroo Hills West, Leyshon View and Kirk River) showed a significant difference in tree dieback between grazing treatments. At Kangaroo Hills West, ungrazed plots had a higher proportion of dead ironbarks than grazed plots. The exclosure plot had the highest rate of mortality for all three treatments. At Leyshon View and Kirk River, grazed plots had significantly higher ironbark mortality than ungrazed plots, with the exclosure plot showing the lowest proportion of tree mortality of the three treatments.

Two sites (Kangaroo Hills East and Kirk River) showed that a significant difference in tree dieback occurred between size classes. In these two sites, there were a significantly greater proportion of large trees that were dead than small trees. The same trend was also evident in the macropod exclosures.

In Table 8.1 above, it can be seen that Blue Range had no significant results for any of the treatments, while Kangaroo Hills West showed a significant interaction between grazing treatment and size class. Figure 8.1 below shows the relationship between trunk size and (transformed) proportions of dead ironbark trees. The values for grazed and ungrazed treatments represent the average value for all replicated plots. Although the significant result given in Table 8.1 was for comparison between grazed and ungrazed plots, results from the macropod exclosure have been added to Figure 8.1 for comparison, however, due to a lack of replication, these points are not an average. Figure 8.1 below shows that in grazed plots and the exclosure plot, there was a noticeable increase in the proportion of trees that were dead in the larger size classes. In ungrazed plots, there was an average trend of decreased mortality with increased trunk diameter. In comparison between grazed and ungrazed plots, ungrazed plots had the highest proportion of saplings that were dead, while at larger tree sizes; grazed and ungrazed plots had similar proportions that had died. The highest mortality of large trees was in the exclosure plot.

Figure 8.1: Two-way interaction of trunk diameter and grazing treatment in influencing the proportion of dead ironbark trees at Kangaroo Hills West study site.



Although the tree sizes were separated into just two categories for the ANOVA, a greater amount of detail was collected for the distribution of tree dieback by size class. Figures 8.2 to 8.6 show the percentage of dead and alive ironbarks in five discrete size classes. Results from replicate plots have been combined to show the total result from a particular grazing treatment. On each graph, the number of individuals (n) is indicated.

At Kangaroo Hills East (Figure 8.2), it was evident that there was no mortality at either the smallest or largest size classes, and that the size classes of 21-<31cm DBH and 31-<41cm DBH were the most commonly encountered. All observed mortality occurred in trees with a DBH between 11-41cm. There was no significant difference between treatments.

At Kangaroo Hills West (Figure 8.3), saplings (DBH <11cm) were the most abundant size class, with abundance declining with increasing trunk diameter. There were considerably more saplings in the ungrazed plots (n=50) than in the grazed plots (n=36). In ungrazed plots (Figure 8.3 b), the percentage of trees that died decreased with size, whereas grazed plots (Figure 8.3 a) showed mortality in all but the largest size class. The exclosure plot (Figure 8.3 c) showed particularly high mortality in the 11-<21cm DBH size class.

At Blue Range (Figure 8.4), there was no tree mortality observed in the exclosure plot, however, the total number of trees was considerably lower than in either the grazed or ungrazed plots. Across the site, the most abundant size class was in the 21-<31 cm DBH range, with the smallest and largest size classes having the least representation. There were no significant differences between grazed and ungrazed plots.

At Leyshon View (Figure 8.5), the total numbers of ironbarks were quite small, with the exclosure plot having only two specimens (Figure 8.5 c). In the other treatments, mortality was particularly high in the 11-<21cm DBH size range, however, it is evident that rates of mortality were significantly higher in grazed plots than ungrazed. There was no mortality in the largest size class of >41cm DBH.

At Kirk River (Figure 8.6), saplings (DBH <11cm) were the most abundant size class. Saplings were more abundant in ungrazed plots (n= 177) than in grazed plots (n=27). Mortality of saplings was the highest in the exclosure plot (Figure 8.6 c). In comparison between grazed and ungrazed plots, there were significantly higher rates of mortality with increasing trunk diameter and there were significantly more dead trees in grazed than ungrazed plots.

Figure 8.2 (a-c): Percentage of dead or alive *Eucalyptus crebra* within grazing treatment plots at Kangaroo Hills (East)
(Values expressed as percentages of the total found in each treatment)

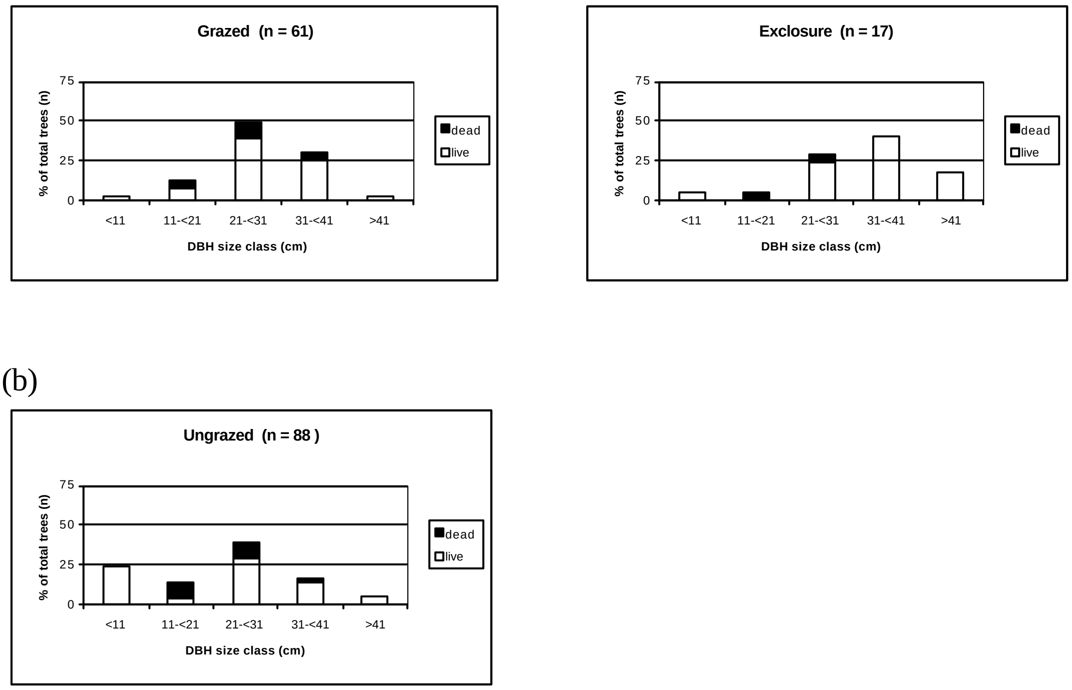

(a) (c) (b)

Figure 8.3 (a-c): Percentage of dead or alive *Eucalyptus crebra* within grazing treatment plots at Kangaroo Hills (West)
(Values expressed as percentages of the total found in each treatment)

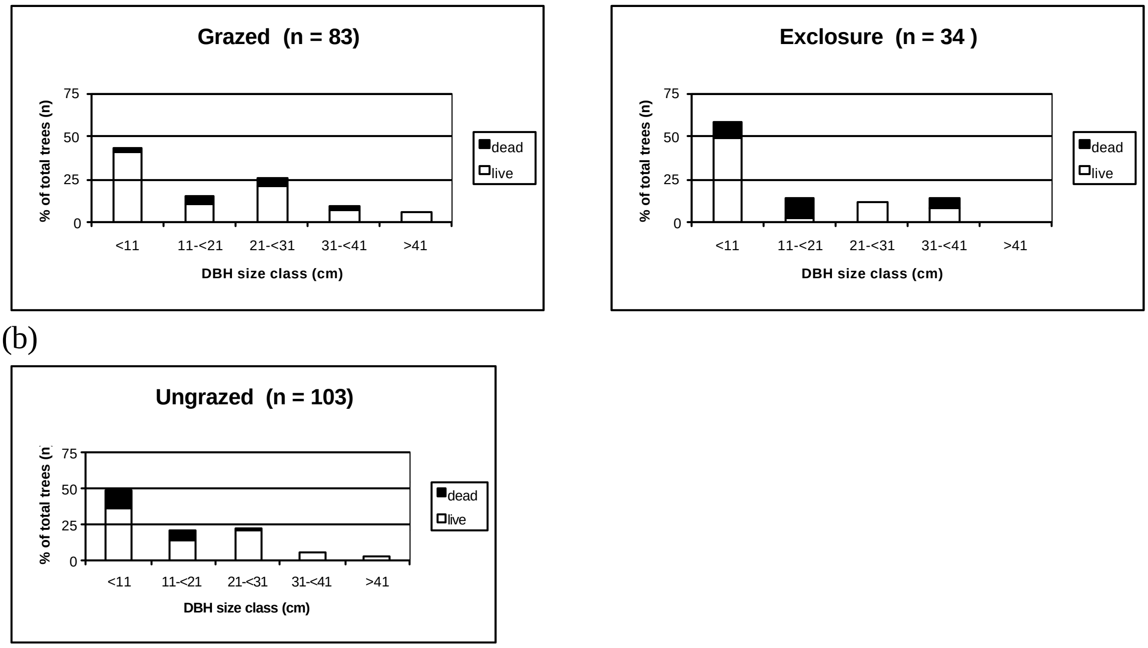

(a) (c) (b)

Figure 8.4 (a-c): Percentage of dead or alive *Eucalyptus crebra* within grazing treatment plots at Blue Range

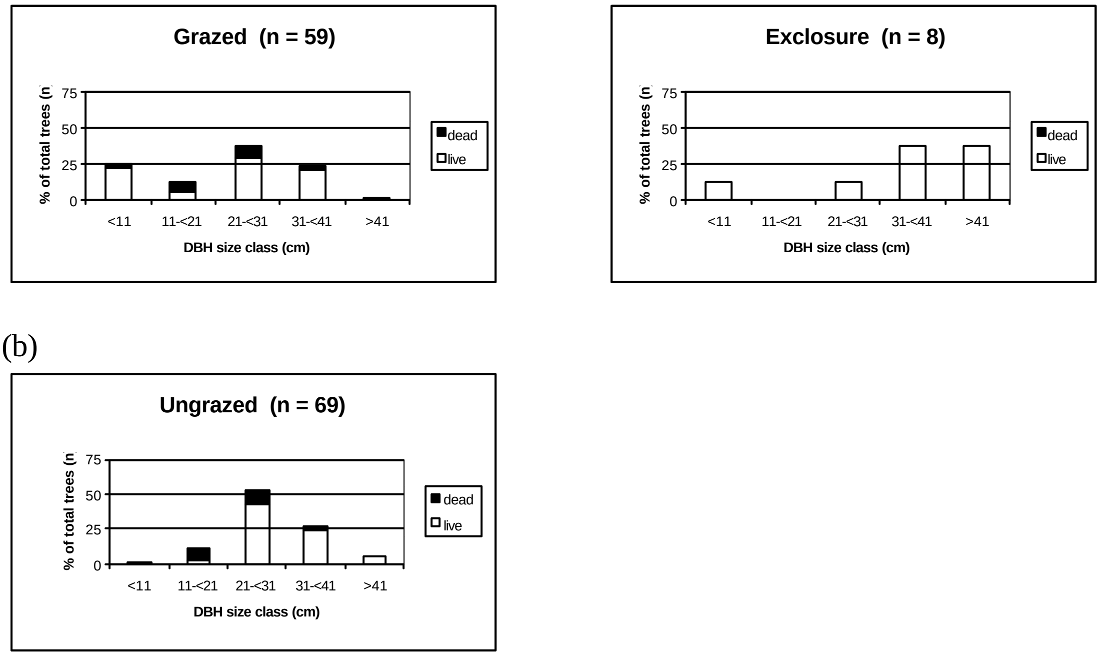

(a) (c) (b)

Figure 8.5 (a-c): Percentage of dead or alive *Eucalyptus crebra* within grazing treatment plots at Leyshon View

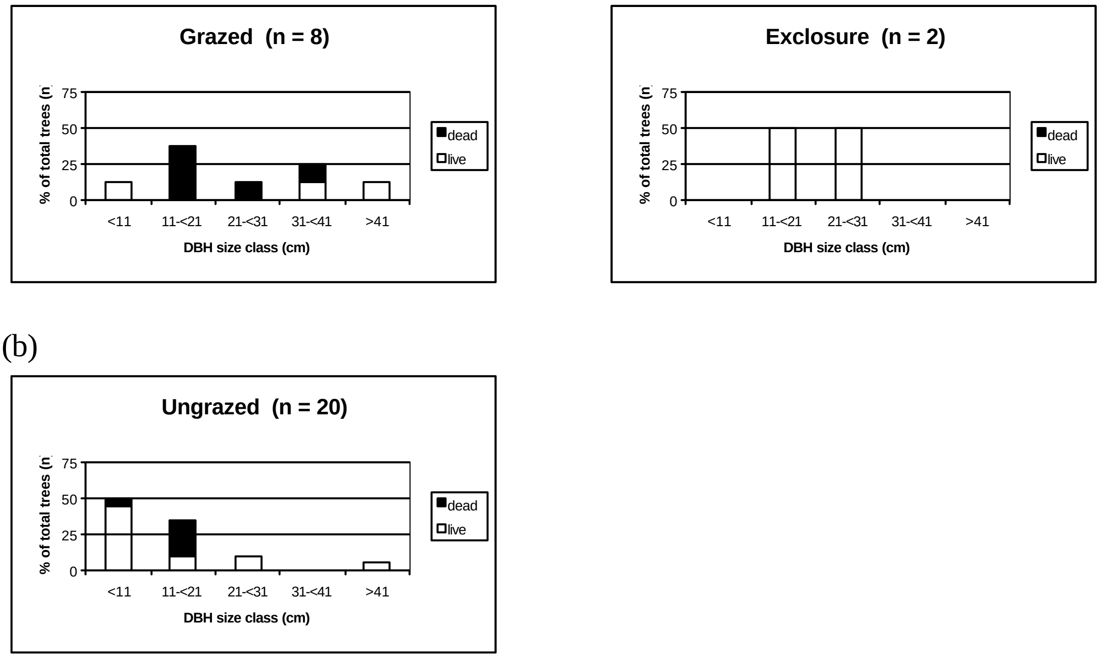
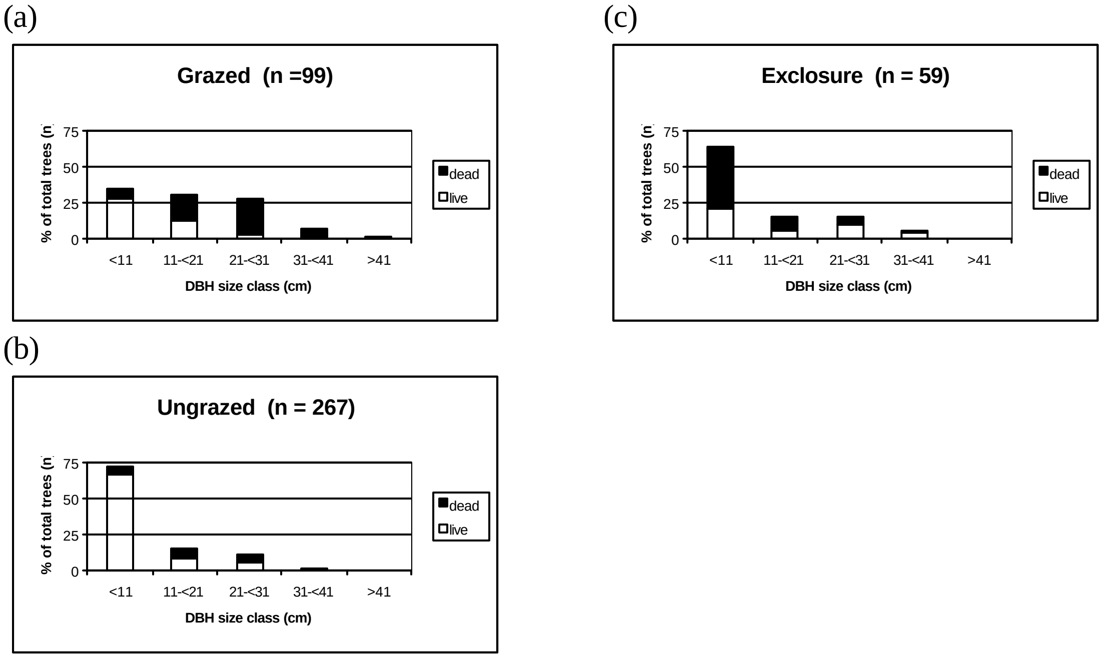

(a) (c) (b)

Figure 8.6 (a-c): Percentage of dead or alive *Eucalyptus crebra* within grazing treatment plots at Kirk River


(a)
(c)
(b)

8.5 CONCLUSIONS

A number of significant results were obtained and these are discussed in context with the literature reviewed in sections 8.1 and 8.2.

Although no mathematical analyses were conducted, there was a considerably higher mortality amongst ironbarks than in any other species. There was zero mortality amongst *Maytenus cunninghamii* and the exotic weed *Zizyphus mauritiana,* whilst mortality levels were so low amongst the bloodwoods *Corymbia dallachiana* and *C. erythrophloia* that numbers were too low for comparison. In contrast, every site examined suffered relatively high levels of dieback amongst ironbarks. This observation contrasts with those by Fensham (1998) who noted that different rates of dieback between ironbarks and *Corymbia dallachiana* were not significant.

While no soil samples were taken to attempt to culture possible soil pathogens, it is not believed that the root-rot fungus *Phytophthora cinnamomi* was responsible for the dieback event described in the present research. Several aspects of the dieback event tend to discount *Phytophthora cinnamomi* as the agent responsible:

- Dieback in sites inspected was predominately confined to specimens of ironbark trees. Cahill (1999) notes that death of understorey plants is symptomatic of *Phytophthora* infection.

- *Phytophthora* infection often results in lesion formation and the destruction of above-ground parts including fruit (Cahill 1999), however, no evidence of these signs were noted in the present research.
- Most areas known to suffer dieback from *Phytophthora cinnamomi* are those with Mediterranean climates in Tasmania, Victoria, South Australia and Western Australia (Environment Australia 2001). While *Phytophthora cinnamomi* has been recorded from coastal Queensland, it is usually confined to wet coastal forests (Environment Australia 2001).
- The fact that a very large area was affected in a very short time, immedialty manifesting itself over that area rather than spreading across a definable front discounts *Phytophthora cinnamomi* as the causal factor (P.Gadek pers. comm.).

The suggestion of the dieback being the result of a stand-level dieback caused by an even aged stand of trees reaching senescence must also be discounted in this particular case. Unlike the example described by Mueller-Dombois (1986), results show that dieback occurred across a wide range of age classes, as approximated by trunk diameter. A wide variation in tree height and trunk width was noted at the various sites, and there is no evidence to suggest that any of the woodlands examined were an even-aged cohort.

Ironbarks suffered high rates of mortality on both the grazed and ungrazed plots at each of the five sites examined, and in three sites; there was also ironbark mortality within exclosures. This observation concurs with those by Fensham & Holman (1999) who observed that substantial dieback still occurred even in the absence of soil compaction or grazing pressure. The cause of dieback was then neither specific to a site nor a grazing treatment and it must be concluded that the 1992-1994 drought, as suggested by numerous graziers and authors, was the causal factor. However, the question remains as to whether grazing or other external forces may have increased susceptibility.

Results from the present research did not concur with the conclusions by Fensham & Holman (1999) and Fensham (1998) that cattle grazing had no influence of tree dieback. Table 8.1 showed three sites with a significant effect of grazing treatment on tree dieback. Two sites (Leyshon View and Kirk River) had highest proportional mortality in grazed plots and lowest mortality in exclosures, showing a clear trend of grazing intensity and dieback. These results concur with those of Scanlan *et al*. (1996a) who also found a significantly higher number of trees died in the grazed plots than in the ungrazed plots. The contrast with the findings by Fensham (1998) may be attributed to the extent of the contrast between treatments. Fensham (1998) suggests his result may be due to a comparison between lightly and moderately utilised pastures and so could not dismiss the possibility of heavy grazing having an impact on dieback. In the present research, two sites (Kangaroo Hills East and Blue Range) had no significant difference between grazed and ungrazed treatments. These sites have been described anecdotally as having very conservative stocking rates, and therefore little contrast between treatments. It should be noted, however, that the respective exclosure for these two sites showed considerably less mortality.

Due to the constraints of the present research and the fact that the dieback event occurred prior to the research commencing, it was not possible to determine the exact cause of the grazing-dieback interaction. It may be, as suggested by Landsberg & Wylie (1983), Landsberg *et al.* (1990) and Jackes (pers. comm.), that the impacts of grazing animals were influencing the feeding behaviour of defoliating insects. Although most of the dead trees had succumbed some time previously and had lost all of their foliage, it should be noted that none of the sick or surviving trees showed a noticeable difference in insect-induced herbivory. It may even be that the drought reduced insect numbers, since Nadolny (1985) notes that populations of herbivorous scarab beetles underwent drastic declines on the New England Tablelands during a severe drought. It is also possible that increased dieback was due to the effects of cattle grazing on decreasing retention of surface water and infiltration rates and increasing runoff and topsoil loss, as suggested by Fensham (1998) and Williams *et al.* (1993). It can be noted from Chapter 7 that the sites at Leyshon View and Kirk River showed significant differences in ground cover between treatments, however, that was also the case for other sites that did not show a significant difference in tree dieback between treatments.

Two sites showed a significant difference in the proportion of dead individuals between the two size classes. This finding contrasted with that of Fensham (1998) that there was no significant difference in dieback between size classes but agreed with Fensham & Holman (1999) that, amongst ironbarks, significantly more larger diameter trees died than smaller sized trees.

Results of the data show that mortality occurred across a range of stem diameters, and in many cases, smaller trees died while larger trees survived. This indicates that the theory of Mueller-Dombois (1986) of cohort senescence causing dieback is not applicable in this particular case. Indeed, the small number of individual trees alive or dead actually represented in the largest size class indicates that most individuals of that age cohort have succumbed to some perturbation(s) long before reaching the age of senescence.

In contrast to all the results discussed above was the pattern observed from Kangaroo Hills West. This site showed an increased rate of mortality with grazing exclusion. Grazed plots had the lowest proportion of dead saplings, while the exclosure had the highest proportion. In comparison between the grazed and ungrazed plots, Table 8.1 shows that there was a significant interaction between grazing treatment and trunk diameter in influencing dieback. Examination of Figure 8.3 above shows that the proportion of trees that died increased with trunk diameter in both the grazed and exclosure plots, and decreased slightly in ungrazed plots. The cause of higher mortality in larger trees has already been discussed above. In the ungrazed plots, however, there was a markedly higher mortality amongst saplings (DBH <11cm). This result concurs with those by Scanlan *et al.* (1996a) who found significantly higher mortality in *Eucalyptus* saplings than for taller specimens. It is suggested here that the shallower root systems of saplings place them in greater competition with grasses and herbaceous species. Smaller saplings also have less developed lignotubers and other regenerative bud development those more mature individuals. Retention of competing species during periods of extreme stress such a drought may have made them more susceptible to a dieback event, to which they possess fewer mechanisms to survive. This scenario is most likely to develop with the complete exclusion of herbivores (exclosure plot) and is less likely to develop where competition is being removed (grazed and ungrazed plots). This hypothesis is supported by results from Kangaroo Hills West (Figure 8.3 b & c) and Kirk River (Figure 8.6 c).

Results presented in the present research support comments by Tothill (1971) and Scanlan *et al.* (1996a) that heavy grazing increases woody plant mortality. However, as previously stated, the absence of grazing may exacerbate dieback of saplings during drought years (see Figure 2.1).

In conclusion, the present research has shown that ironbarks are relatively sensitive to dieback. That this dieback occurred over a range of soils, conditions and grazing regimes following a severe drought suggests that drought is the primary cause of death. On sites regarded as heavily grazed (as noted in Appendix A), grazing treatment had a significant effect on dieback of ironbarks, but on sites regarded as having conservative stocking rates, dieback rates were similar between treatments. Trees of a moderate size (11-41cm DBH) were generally the most susceptible to dieback, more so than saplings; however, under some conditions, saplings may have had increased susceptibility to dieback in the presence of competition from the herbaceous layer, which arose from the absence of grazing. Age distribution patterns indicate periods of tree recruitment fluctuated with episodic climatic events such as above-average wet seasons and droughts.

CHAPTER 9. STATE AND TRANSITIONAL MODELS

9.1. WHAT ARE 'STATE AND TRANSITION' MODELS?

To understand the impact of grazing on rangelands, there are numerous vegetation community attributes that can be examined. In the present chapter, the various community attributes measured in the present research were united to develop an all-encompassing model of grazing changes.

Numerous theories have been presented to explain observed changes in vegetation composition in response to perturbations such as drought and grazing. The Clementsian model of rangeland dynamics suggests that rangelands undergo a succession towards a climax community in the absence of grazing and that grazing causes a successional shift in the opposite direction from the climax community (Westoby *et al.* 1989). This model suggests that all states of land condition exist along a continuum, with drought having a similar effect to grazing while periods of above-average rainfall accelerate the progression towards a climax community (Westoby *et al.* 1989). Sustainable grazing therefore depends on finding the equilibrium point along the spectrum and adjusting cattle numbers to compensate for climatic fluctuations (Westoby *et al.* 1989). An example of successional theory is proposed by Gibson & Brown (1992) who identify two possible effects of cattle grazing on succession:

1/ Cattle grazing creates a sub-climax community, by effectively halting succession before a full climax community is achieved

2/ Succession continues under cattle grazing but is deflected along a different route to a 'plagioclimax'.

Although successional theory is still recognised as valid for mesic and wetter habitats, it has been criticised for its validity in explaining the dynamics of other ecosystems such as semi-arid rangelands (Westoby *et al.* 1989, Whalley 1994). It is recognised that several alternative stable states of condition can exist, the response of vegetation to grazing is not continuous and some transitions to states of land degradation are not reversible by removal of grazing pressure (Westoby *et al.* 1989).

In response to changing theoretical approaches to rangeland ecology and management, 'State and Transition' (S &T) models were developed. S & T models consist of a catalogue of alternative states of condition and the possible transitions that exist between those states (Westoby *et al.* 1989). A theoretical advantage of the S & T model over the Clementsian successional approach is that, by discarding the assumption that a pasture would return back to the most productive state in the absence of disturbance, a pro-active approach to land management is encouraged rather than taking a passive fatalistic view of community dynamics (Whalley 1994). State & Transition models were developed to be ‘decision support systems’ for graziers; collating previous knowledge into an improved format, focusing new research, enhancing an understanding of vegetation changes and their underlying causes and improving the general understanding of ecological principles (Filet 1994).

The catalogue of states of land condition is defined by looking at the differences in vegetation composition under various management regimes (McArthur *et al*. 1994). The various S&T models provide lists of the states, the important factors that influence the transition between those states, the stage at which those influencing factors are active and which states are affected by which factors (Filet 1994). While vegetation communities are in a continuous state of flux, transitions between identifiable states are often driven by discrete and identifiable disturbance events such as fires and droughts (Whalley 1994). Grazing is also recognised as a major driving force for transitions to occur in all S & T models (Grice & McIntyre 1995). Transitions can be seen as a catalogue of opportunities and hazards to the condition of pasture composition and structure. Opportunities can be seen as circumstances favouring a transition to a state of improved condition, and hazards exist where a transition would lead to a more degraded state (Westoby *et al.* 1989). Although disturbances are generally followed by successional change (Whalley 1994), S & T models do not favour the equilibrium approach to sustainable cattle grazing, but rather encourage the seizing of favourable opportunities and avoidance of hazards by recognising the potential for a transition and acting appropriately (Westoby *et al.* 1989, Whalley 1994). The S & T model is therefore a highly flexible multi-state approach to community ecology, where the state of the pasture is related directly to the nature of the transitions that are driven, to a large degree, by land management.

Transitions are commonly caused by a combination of several different factors such as climate and manipulation of grazing pressure and fire (Filet 1994). In many cases, transitions between states represent hazards involving shifts from a productive pasture dominated by native, palatable perennial tussock grasses to a less productive, more degraded system with less palatable perennials. The transition between some states of degradation is generally not reversible in a practical time scale without some degree of substantial intervention (Filet 1994). McArthur *et al.* (1994) separates transitions into three categories depending on their ability to be reversed:

1) Strategic: low input required for a better state (low hazard, high opportunity)
2) Critical: A decision is critical to either prevent further decline or return to a better state
3) Major: High input required for a better state (high hazard, low opportunity)

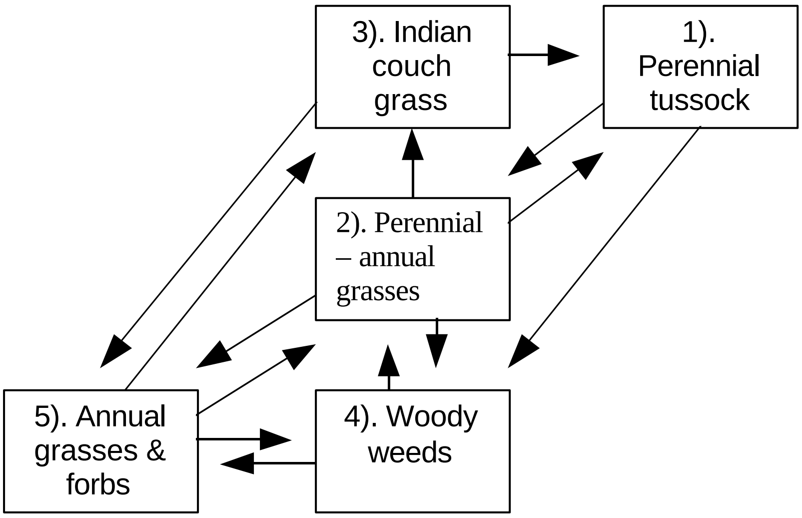
Transitions between states may be rapid. In experiments reported by Lodge & Whalley (1985), a transition from *Aristida* domination to *Danthonia* domination occurred rapidly through manipulation of stocking intensity, and by matching the grazing and rest intervals with the phenology of the grass species being manipulated. In many cases, however, the time taken for transitions to occur may depend on the previous history of the site (Filet 1994). For example, the longer an area has been heavily grazed, the lower the rate of recovery becomes. After 10 years of heavy grazing, reversed transitions may no longer be possible. Filet (1994) also sees the critical loss of soil as being another example of an irreversible transition, though he concedes that soil parameters are not directly included in S&T models. Several models do, however, incorporate ground cover as a characteristic of certain states.

9.2. USE OF STATE AND TRANSITION MODELS

Numerous S&T models have been produced for various pasture zones in Australia's northern rangelands. Pasture zones examined include the tropical tallgrass savannas, *Aristida-Bothriochloa* pastures, northern and southern speargrass zones, and the pasture zone dominated by mitchell grass, bluegrass-browntop and Queensland bluegrass. Of greatest relevance to this study was the S&T model proposed by McIvor & Scanlan (1994) for the northern speargrass zone; a region covering 17 million hectares and encompassing many of the sites examined in the present study.

McIvor & Scanlan (1994) identified five states of pasture condition for the northern speargrass zone, defined by the dominating species. These five states are: 1) perennial tussock grasses; 2) perennial– annual grasses (mix of perennials, annuals and forbs); 3) Indian couch (*Bothriochloa pertusa*) dominated; 4) woody weeds (both native and introduced) and 5) annual grasses, forbs and unpalatable perennial grasses. Figure 9.1 represents these five states and the direction of transitions between those states.

Figure 9.1: A general state and transition model for pastures in the northern speargrass zone of Queensland (from McIvor & Scanlan 1994).



In this model, state 1) was considered to be the most desirable and productive for grazing purposes, while state 5) was considered to be the most degraded. A transition between states 1) and 5) (via state 2) is accompanied by a decline in pasture production, soil stability, ground cover, infiltration and water quality, and higher runoff and soil loss (McIvor & Scanlan 1994). Note that this model also incorporates opportunities for a return from degraded states to a state dominated by perennial tussock grasses.

It should be noted that S & T models contain many generalities that have not been tested experimentally and so an objective assessment of these models is required for their further development (Grice & McIntyre 1995). For example, suggestions that forbs and annuals increase in dominance in heavily grazed pastures do not distinguish between increases in total biomass or their abundance relative to perennial grasses (Grice & McIntyre 1995). Nonetheless, S & T models provide a convenient mechanism to convey the results of research, observations and ecological theory, in spite of the limitations imposed by their necessarily reductionist approach. In the present research, statistically significant changes to community characteristics were compiled to assist in progressing the general understanding of states and transitions. These observations relate only to differences between heavily grazed and lightly grazed plots. While it has been widely acknowledged that fire is also a major ecological force driving transitions between states, this factor was not incorporated into the present research due to a total absence of fire within the recent history of any of the 13 sites examined.

9.3. TESTING STATE AND TRANSITIONAL MODELS

'State and Transition' models have been popular in promoting awareness of ecological processes and assisting graziers in the decision-making process to maximise their land's potential for sustainable cattle grazing. However, it remains a problem that many of the characteristics of S&T models are based on subjective, rather than empirical data. In the present study, it has been possible to examine the apparent transition of states between ungrazed and grazed plots.

None of the plots examined in the present research include areas untouched by cattle grazing, however, it is probable that the only rangeland areas untouched by cattle grazing are those unsuitable for grazing. It may be possible that even within exclosures, species that are extremely susceptible to cattle grazing had disappeared long before the present study was commenced. The present study compared grazed plots with plots given long-term protection from grazing, as opposed to a comparison of grazed plots with pastures that are in a pre-grazing condition.

Despite the obvious filtering effect that prior grazing influence may have had on plant diversity and community structure, most of the ungrazed and exclosure plots represent states described by McIvor & Scanlan (1994) as 1) Perennial tussock grasses and 2) Perennial – annual grasses (mix of perennials, annuals and forbs). The exact floristic composition of these plots varies with site, depending on soil type and climatic regime. In many areas, native perennial grasses (NG3P) such as *Heteropogon contortus, Themeda triandra* and *Chrysopogon fallax* dominated, whilst in other sites, those native perennials had been replaced by exotic pasture species such as *Cenchrus ciliaris*. It should be noted that although the present research indicated that *Chrysopogon fallax* was a decreaser species, there are suggestions that the state dominated by that species is the result of a transition of degradation from a previous state dominated by *Themeda triandra* and *Heteropogon contortus* (O'Reagain P. pers. comm.). Supporting this assumption are observations by McIvor *et al*. (1995a) who recorded that *Chrysopogon fallax* had the highest abundance at levels of intermediate disturbance; that it first increased with light grazing but then declined again as grazing intensity was increased. (See site descriptions in Appendix A).

The series of states that follows as a result of applied grazing pressure varied from site to site, and was dependant on vegetation type and community composition. These in turn are dependant on the soil and climate since different soil types support different species. Table 9.1 below summarises the transitions between ungrazed and grazed land condition states.

On the lightly grazed Greenvale chromosols (Kangaroo Hills East, Kangaroo Hills West & Blue Range), grazing pressure resulted in a shift away from a state dominated by native perennial grasses (predominantly *Heteropogon contortus*) to one with more annual grasses, legumes, forbs, woody species and legumes, probably equivalent with 'state 2' described by McIvor & Scanlan (1994). There was a decrease in ground cover and a corresponding increase in species diversity as numerous species were released from suppression by competition. These sites indicated that good populations of native perennial tussock grasses could be maintained under light grazing pressure. However, total reliance on these grasses as indicator species of pasture change may overlook the fact that grazing induced changes are occurring in the pastures and that regular pasture monitoring may be required to advise if a major transition in species composition is occurring.

On the heavier grazed Dalrymple chromosol sites (Leyshon View and Kirk River), ungrazed plots were dominated by perennial tussock grasses such as *Chrysopogon fallax*, but had a high abundance of undesirable *Bothriochloa pertusa* even in the absence of grazing disturbance. Under increasing grazing pressure, however, perennial tussock grasses declined in abundance, while domination by *Bothriochloa pertusa* increased and diversity declined. It is also possible that these sites had already suffered a decline in condition before the exclosures were erected. This prior transition may have involved a replacement of *Themeda triandra* and *Heteropogon contortus* with *Chrysopogon fallax*, and the colonisation of *Bothriochloa pertusa*. It is possible that *Bothriochloa pertusa* had not necessarily invaded into ungrazed plots, but was persisting in those plots from an earlier colonisation event prior to removal of grazing disturbance.

The responses of the various functional groups varied between sites. This was usually dependant on the palatability of those particular species that made the greatest contribution towards the abundance of that functional group at that particular site. Ground cover increased or decreased depending on the degree of domination by *Bothriochloa pertusa*. Whether it was the degree of grazing intensity or the susceptibility of vegetation communities on this particular soils type is not clear, however, it appears that major transitions have occurred on these sites, resulting in a more degraded condition. Using the categories provided by McArthur *et al.* (1994), reversal of these transitions would be major (high input required for a better state, representing a high hazard and low opportunity).

In contrast, there were few statistically significant results at the other Dalrymple soil site (Rishton). It was evident at the site that decreaser perennial grasses such as *Heteropogon contortus* were re-establishing the grazed plots, where *Bothriochloa pertusa* dominated. This indicates that if reservoirs of valuable decreaser species can be retained on site through the strategic exclusion of cattle, the transition from valuable perennial grass communities to a less productive *Bothriochloa pertusa* community would be reversible, presumably as long as no serious loss of topsoil had occurred in the meantime. This reverse transition to a more productive state also appeared to have halted the decline in diversity and ground cover seen on the other sites on Dalrymple soils. It may be possible, therefore, that on heavily altered pastures such as Kirk River and Leyshon View, de-stocking may allow favourable transitions to occur in pastures adjacent to seed reservoirs, by allowing the succession and reinvasion by more productive grass species.

It is possible, however, that growth rates of colonising perennial grass species may be lower than in adjacent ungrazed areas due to the inhibiting effect of reduced mycorrhizal fungi populations (Ash & McIvor 1995) and the altered state of soil hydrological properties (Weigel *et al.* 1990).

Table 9.1: Summary of observed trends in land condition between heavily grazed and lightly grazed plots (NS- no significant results). (Note that grazing pressure is not equal on each site)

SITE	Grazing induced transition of condition state				
	Diversity	Dominant grasses	Annuals/ Palatable	Functional Groups	cover/ dieback
Kangaroo Hills East	increases	more *Melinus repens*	more annuals	less forbs & exotic legumes, more native legumes	cover decreases
Kangaroo Hills West	increases	less *Themeda*, more *Melinus*	more annuals	more woody, more forbs, more legumes	cover increases
Blue Range	increases	less NG3P: less *Heteropogon*, more *Bothriochloa pertusa* & *Aristida*	more annuals, more unpalatable species	more woody, forbs & native legumes	cover decreases
Leyshon View	decreases	less NG3P: more *Bothriochloa pertusa*, less *Chrysopogon fallax* & *Melinus repens*	less annuals, less unpalatable & palatable species	less forbs, more native legumes	cover increases, dieback increases
Kirk River	decreases	less NG3P: more *Bothriochloa pertusa*, less *Chrysopogon fallax*	more unpalatable species	more woody, more forbs, more native legumes	cover decreases, dieback increases
Rishton	NS	NS	NS	less exotic legumes	dieback increases
Mt Leyshon	increases	NG3P intermediate, less *Cenchrus*, *Melinus* & *Heteropogon*, more *Bothriochloa pertusa*	more annuals, more unpalatable species	less woody, more forbs, native legumes intermediate	cover decreases
Jervoise	NS	less NG3P: less *Themeda*, more *Bothriochloa pertusa*	more unpalatable	more forbs	cover decreases
Pajingo	decreases	less NG3P: less *Chrysopogon fallax*	less annuals, more unpalatable species	less woody, less forbs	cover decreases
Lucky Downs Dam	NS	more *Heteropogon* seedlings & Themeda, less *Bothriochloa pertusa* & *Chrysopogon fallax*	less unpalatable, more palatable species	more woody, more forbs, less exotic legumes	cover decreases
Epping 1	decreases	less *Heteropogon*, *Themeda* & *Chrysopogon fallax*	less annuals, less palatable & unpalatable	less legumes	cover decreases
Epping 2	decreases	more *Cenchrus ciliaris*	more annuals & unpalatable species	more legumes	NS
Tabletop	increases	more *Heteropogon* seedlings, less *Themeda*	more exotics, annuals & unpalatable species	more forbs, annual grasses & native legumes	NS

An examination of sites on Rangeview soil sites allowed a contrast between a site dominated by *Themeda triandra* (Jervoise) and one dominated by the exotic pasture species *Cenchrus ciliaris* (Mt Leyshon). At Mt Leyshon, ungrazed sites were heavily dominated by *Cenchrus ciliaris*, which appeared to suppress other species resulting in an overall low level of diversity. This state is presumably incorporated into the S & T model by McIvor & Scanlan (1994) as being one dominated by perennial tussock grasses (state 1). However, the nature of the *Cenchrus* dominated state, and the limitations that *Cenchrus* imposes on possible transitions indicates that this state should be regarded as being completely separate from those dominated by native perennial tussock grasses. At Mt Leyshon, *Cenchrus ciliaris* dominance was reduced under a regime of dry season grazing, and there was an associated increase in native perennial tussock grasses and legumes and overall diversity. Under a regime of continuous grazing, however, there was a decline in both *Cenchrus ciliaris* and the native *Heteropogon*, while *Bothriochloa pertusa*, annuals, forbs and unpalatable species increased in abundance. This transition also involved a reduction in ground cover. This site highlighted the difference between season-based grazing regimes and how changing the timing of grazing pressure can allow favourable transitions in land condition to occur. This site also highlighted the problems represented by the exotic *Cenchrus ciliaris* where that species is able to escape into areas ungrazed by cattle, such as reserves and national parks. Aggressive dominating grasses such as *Cenchrus ciliaris* that reduce the diversity and complexity of grazing pastures should also be of concern to the grazing industry. Reliance on monocultures can be economically unwise, especially in light of the problems resulting from the decimation of Townsville Stylo (*Stylosanthes humilis*) by Anthracnose disease in the 1980’s (Quirk *et al*. 1997).

On Jervoise, where *Themeda triandra* dominated the ungrazed plots, grazing pressure resulted in a significant reduction in *Themeda triandra* and a significant increase in *Bothriochloa pertusa*, forbs and unpalatable species, with an associated decline in ground cover. There was no overall change in diversity as decreaser species were replaced by increaser species. The fact that *Themeda triandra* still persisted in grazed plots implies that this transition may be reversible if grazing pressure was reduced. The decline in ground cover also suggests that *Bothriochloa pertusa* has not completely dominated the site, since there is often an increase in cover with *Bothriochloa pertusa* domination (Scanlan *et al*. 1996b). Jervoise, therefore, represented a site at an intermediate stage between states identified by McIvor & Scanlan (1994).

Evidence suggests that a transition was occurring between state 1 (perennial tussock grasses) and state 3 (*Bothriochloa pertusa* dominated) and that conditions existed for a favourable reverse transition through grazing management to prevent the total transition to the more degraded state. Using the terminology of McArthur *et al.* (1994), a reverse transition would be both strategic (low input required for a better state - low hazard, high opportunity) and critical (a decision is critical to either prevent further decline or return to a better state).

In the *Acacia shirleyi* scrubs on Pentland kandosols (Pajingo), grazing resulted in a loss of native perennial tussock grasses such as *Chrysopogon fallax*, an increase in unpalatable species such as *Aristida calycina* and a decrease in overall diversity. This loss of diversity includes a decrease in the abundance of annuals, forbs and woody plants; functional groups usually predicted to increase under grazing. It is possible that the dynamics of some woody plant species show different responses when factors such as slope and aspect are changed. Some areas of Pajingo adjacent to the research plots had a considerably higher abundance of the shrub *Erythroxylum australe* in grazed areas than in ungrazed. The grazed community states identified at Pajingo were not easily defined using the states identified by McIvor & Scanlan (1994), so may represent either a transition in progress or a state not recognised in their State and Transition model. It may be that some transitions involve a general decline in palatable representatives of most plant groups with a corresponding increase in unpalatable representatives. A loss of ground cover with increasing grazing should be of particular concern on these areas, since there was abundant evidence of gully erosion forming. Hence, regular monitoring of ground cover would be extremely valuable in this case to arrest erosion before it becomes too extensive to be able to manage the problem practically and economically.

The Epping Forest (1) site on Epping chromosols was another area dominated by the exotic grass *Cenchrus ciliaris*. Independent of grazing pressure, this species was associated with an overall decrease in species diversity. Grazing had no significant impact on abundance of *Cenchrus ciliaris*. However, under grazing pressure, there was a decline in the native perennial tussock grasses such as *Heteropogon contortus*, *Themeda triandra* and *Chrysopogon fallax*. The observed differences in response may have been due either to cattle feeding on these native perennial species in preference over *Cenchrus ciliaris*, or the relatively higher level of grazing tolerance by *Cenchrus ciliaris*. There was a further decline in diversity under grazing, including a significant reduction in legumes and annuals, a reduction in both palatable and unpalatable species, and a corresponding reduction in ground cover. Increases were recorded for sedges and native annual forbs. This transition resulted in a broad change in species composition that contradicted expectations of an increased abundance of most annuals and unpalatable species. The differences observed with annuals may be attributed to differences in palatability to cattle; however, it is also possible that overall declines in many functional groups may be related to interactions with species such as *Cenchrus ciliaris*. Following the example of Mt Leyshon, it would be interesting to see if a favourable reverse transition to a state more diverse and more dominated by native perennial tussock grasses would occur if grazing was applied only during the dry season. The site at Epping Forest (1) illustrates how cattle grazing can detrimentally affect the populations of desirable native tussock grasses through preferential feeding behaviour without having any significant impact on the exotic pasture species *Cenchrus ciliaris*. If this is indeed the case, then this result suggests the high fodder value of native grasses. Where mixed pastures of *Cenchrus* and native tussock grasses are grazed, then selective grazing behaviour by cattle may lead to reduced abundance of the native tussock grasses and increased dominance of *Cenchrus ciliaris*, possibly resulting in a transition from a state of mixed grass species to a state dominated by *Cenchrus ciliaris*.

Epping Forest (2) occurs on very deep beds of alluvial sand and is considered of very high conservation significance due to it being the habitat for the critically endangered Northern Hairy-nosed Wombat. *Cenchrus ciliaris* also dominated this site. An increase in cattle grazing increased *Cenchrus* domination, causing a decline in native grass species that wombats favour. There was an increase in most exotic plant groups, annual grasses, legumes, and unpalatable species. These transitions begin with grazing by kangaroos and wombats, but were more extreme under cattle grazing. The desirability of one state over another may depend on the favoured land use. Although graziers may benefit temporarily from an increase in *Cenchrus*, the long-term transition to a state with *Cenchrus* dominance and a host of unproductive species is of concern from both a grazing and conservation perspective. Intuitively, such simplified landscapes must have a reduced number of options for transitions to different states, and the possibility for a transition to a more productive state is far less likely than a transition to a more degraded state.

The Tabletop site on Bluff Brown sodosols occurred in the highest rainfall zone and was characterised to a large degree by plant species not seen elsewhere. Ungrazed plots were characterised by tall *Heteropogon contortus* and the exotic legume *Stylosanthes scabra*. Under cattle grazing, the reduction in competition from tall grass allowed for a much higher diversity of small annual herbaceous forbs and legumes. Although *Heteropogon* germination was high, survival to sexual maturity was not observed to have occurred, and *Themeda triandra* had disappeared completely. The exotic pasture legumes also disappeared. While species diversity was much higher under grazing, from a grazing perspective, the botanically diverse ‘lawn’ in the grazed pastures would be very unproductive. The large number of *Heteropogon* seedlings do show, however, that recovery of the native perennial tussock grass component would rapidly follow the de-stocking of cattle, and that their competitive superiority would likely cause diversity to fall sharply.

Transitions in the species composition at Lucky Downs dam on Wairuna dermosols appear enigmatic, since intermittently grazed plots had lower abundance of the decreaser grasses *Heteropogon contortus* and *Themeda triandra*, yet had higher abundance of the increaser species *Bothriochloa pertusa*. In addition, higher grazing pressure resulted in less unpalatable species while abundance of palatable species increased. Those observations appear opposite to those expected, and occurred despite an obviously lower grazing disturbance in those plots, based on the difference in height and density of the dominant grasses. However, increased grazing also increased abundance of woody plants and forbs, decreased exotic pasture legumes and caused a decline in ground cover. This site was similar to Tabletop in that many of the native perennial tussock grasses observed in grazed plots were seedlings, with few mature specimens seen. This indicates that this site was undergoing colonisation by these native grasses. Although germination from a pre-existing seed bank is possible, it is not likely since seed banks of these native perennial grasses tend to be very small even under existing stands (McIvor & Gardener 1994). Presumably the seed originated from the adjacent and lighter grazed plots, and the resultant seedlings accounted for the high abundance of palatable species. These seedlings were able to geminate and establish readily in grazed plots where competition is lower. Assuming a frequent immigration of seeds, the low number of adult plants indicate that there was a high attrition of seedlings and that a return of the pasture state to one dominated by perennial tussock grasses may be dependant of partial de-stocking or spelling during the wet season, as previously seen at Mt Leyshon.

9.4. IRREVERSIBLE TRANSITIONS

Under prolonged heavy grazing, transitions may occur which are irreversible without major inputs of time and money. Features of these irreversible transitions may include scalding and erosion. During drought years, increased rates of dieback of ironbark trees occurred in grazed areas, especially amongst larger specimens, although dieback still occurred to some degree regardless of grazing pressure. Results showing that heavy grazing exacerbated dieback support observations by Scanlan *et al*. (1996a) but contradict those by Fensham & Skull (1999). From both an economic and conservation perspective, it is desirable to use strategic grazing management to halt transitions to degraded states before they become irreversible, rather than trying to repair highly degraded areas at huge expense later on. If sustainable land use is defined by “not reaching the threshold for recovery” (Williams *et al*. 1993), then the cattle grazing industry can achieve sustainability only by avoiding irreversible transitions to degraded states of land condition.

In the present research, several of the sites examined had undergone transitions to states that may be irreversible, maybe however, many transitions ought to be reversible with proper land management. As previously stated in Section 9.3 and shown in Table 9.1, many of the sites had undergone a transition to a more degraded state; however, these transitions mostly involved shifts in floristics from desirable to less desirable grassland communities. While the frequency of several desirable species had, in several instances, undergone considerable reduction, their continued presence at low abundances should allow for favourable reverse transitions to occur if subject to appropriate land management techniques such as strategic de-stocking and/or application of an appropriate fire regime.

One apparently irreversible transition is the loss of *Themeda triandra* from Leyshon View, Kirk River and Mt Leyshon, and its nearly total loss at Kangaroo Hills East. These sites all occur on soil types that supported populations of *Themeda* at other representative sites. (Note that although *Themeda* was also absent at Pajingo, there were no other sites on Pentland Kandosols to determine if *Themeda* should have been expected). This irreversible transition has occurred over a broad area of Australia’s tropical rangelands (McIvor & Scanlan 1994). The replacement of this species by other productive pasture grasses such as *Heteropogon contortus* and *Cenchrus ciliaris** may allow for these areas to remain productive from a grazing viewpoint, hence this transition may not be regarded as significant degradation by some graziers. From a conservation viewpoint, however, the loss of such an important dominant grass species from large areas of our tropical rangelands should be of concern, due to the obvious role that dominant grasses have on influencing grassland species dynamics.

While a return to native grasslands dominated by native perennial tussock grasses may be technically, if not practically feasible at many of the sites examined, those dominated by *Cenchrus ciliaris* (eg. Mt Leyshon, Epping Forest) are a different matter. The highly competitive nature of this species excludes the possibility of a reverse transition, hence domination of a pasture by this species should also be regarded as an irreversible transition. From discussions with managers of protected areas, a feasible method by which to reverse this transition is yet to be discovered.

Although erosion and scalding were evident at some sites (eg. Leyshon View, Mt Leyshon), there was not the extensive scalding and gully erosion recorded in other areas of the Dalrymple Shire (DeCorte *et al*. 1994, Gardener *et al.* 1990). Although the gully erosion observed in the present research may take considerable time to recover, the vast amount of topsoil loss reported by Gardener *et al.* (1990) would require extensive injection of labour and money to halt, let alone repair.

CHAPTER 10: DISCUSSION AND RECOMMENDATIONS

10.1 DISCUSSION

The primary aim of the present research was to determine what the importance of grazing was in influencing patterns in species diversity, and determining if there were any predictable patterns in community changes with increased grazing. This question was answered in several sub-sections:

- What was the effect of grazing on species diversity, richness and evenness?
- What was the variation in response to grazing by different functional groups?
- How did this variation alter plant community composition under varying grazing regimes?
- What was the effect of grazing on dominant grass species?
- How did grazing alter the dominance of a particular species?
- What relationship existed between grazing and ground cover?
- What influence did grazing have on rates of dieback of the ironbark *Eucalyptus crebra*?

These attributes were examined in context with ‘state and transition’ models that have been developed previously for comparable pasture zones.

Using a broad range of criteria to measure diversity, the results showed that changes to species diversity varied between sites and soil types. The diversity and composition of a pasture was determined primarily by soil and climate, and secondarily by grazing pressure and timing. The results indicated that variation in the changes to diversity under grazing was due to the domination of different plant species at different sites. When palatable native grasses dominated, grazing increased diversity, confirming predictions by McIvor (1998) and Harper (1969). Intermediate grazing levels reduced competition by preferentially targeting the palatable dominant species, thus allowing opportunities for the germination and establishment of other species through ‘predator-mediated co-existence’ (Petraitis *et al*. 1989). This state of high diversity declined as grazing pressure increased, as competition pressure was replaced with disturbance pressure. At either end of the competition – disturbance gradient, there was a decline in diversity due to fewer species being adapted to those conditions.

Less palatable exotic grasses, especially *Cenchrus ciliaris* and *Bothriochloa pertusa*, are not reliant on grazing to persist but can completely dominate a site under heavy grazing. Although selective grazing behaviour appeared to reduce the abundance of the native perennial grasses in favour of *Cenchrus ciliaris*, *Cenchrus* also has a high seed production compared to native perennial grasses and horizontal nodal tillers below the defoliation level, thereby allowing it to gain a competitive advantage under the same grazing pressure (Hodgkinson *et al.* 1989). *Bothriochloa pertusa* also has a low stoloniferous habit, high defoliation tolerance, rapid growth rates and high seed production (Grice & McIntyre 1995), allowing it to take advantage of any reduction in ground cover by grazing (McIvor & Gardener 1985). The reduction in diversity observed in grazed pastures dominated by *Cenchrus ciliaris* and *Bothriochloa pertusa* may be due to an increase in their domination, causing an increase in competition, rather than a reduction. These results confirm comments by McIvor (1998).

Examining overall diversity can mask underlying changes in composition, if a decline in one species is matched with an increase in another species. The persistence and abundance of five broad and overlapping functional groups and fifteen discrete functional groups were therefore examined for their response to grazing. These groups exhibited a broad range of responses between sites and other groups. In some cases, certain functional groups responded in different ways to grazing on different sites. The responses of functional groups such as annual grasses depended on the palatability of the particular species that contributed most to the abundance of that functional group at that site. The effects of cattle grazing on many functional groups were less pronounced in areas grazed only during the dry season, in contrast to areas grazed throughout the year.

Grazing generally resulted in a decline in frequency of:

- native perennial tussock grasses
- exotic pasture legumes, and
- palatable species

Grazing caused an increase in:

- exotic grasses
- forbs
- native legumes, and
- unpalatable species.

Although Pimm (1994) suggests that ecological species redundancy means that diversity of functional groups is of greater importance than species diversity, functional groups are much more likely to survive a perturbation if they are diverse (with a high level of redundancy) (Chapin *et al.* 1992). A decline in diversity of both species and functional groups may result in a loss of stability and resilience of the vegetation community to perturbations.

In addition to impacts on diversity and abundance of species and functional groups, grazing also resulted in a reduction of ground cover. While intermediate grazing caused a level of decline in ground cover that was of benefit to many other species through a reduction in competition and increased opportunities for colonisation, heavy grazing may result in scalding and erosion. An exception to the trend in declining ground cover were the sites where grazing reinforced the dominance of *Bothriochloa pertusa*; the spreading stoloniferous habit of which can result in increased ground cover.

The present research found widespread dieback of ironbarks (*Eucalyptus crebra* sensu lat.) throughout the study region on a range of soil types and grazing regimes. In contrast to previous research, this study found a correlation between cattle grazing and the dieback of *Eucalyptus crebra* on sites noted to be heavily grazed, although dieback did occur to some degree even in the absence of grazing. On heavily grazed sites, larger trees were more susceptible to dieback than small saplings, which, in some cases, may have benefited from grazing by the removal of competing herbaceous species from their proximity.

The various attributes of community structure and composition were compared and contrasted with states of pasture condition presented in the State and Transition model for the northern speargrass pasture zone (McIvor & Scanlan 1994). This research confirmed the existence, with further clarification, of a number of the states recognised by McIvor & Scanlan (1994) for the northern speargrass zone and added several other recognizable states. In most sites, native perennial tussock grasses dominated the original and potentially most productive state. These grass species were often the first to decline with increasing grazing pressure and were generally replaced by exotic grasses and unpalatable species such as forbs and woody plants.

While the present study included examples of native perennial tussock grass species re-colonising *Bothriochloa pertusa* pastures, the same could not be said of *Cenchrus ciliaris* pastures. Confirming observations by Fairfax & Fensham (2000), *Cenchrus ciliaris* tended towards forming a monoculture under increasing grazing pressure, possibly resulting in a reduction in diversity. This may lead to a reduction in stability and resilience to perturbations, while options for favourable transitions would be reduced should the *Cenchrus ciliaris* populations be decimated as has occurred with other exotic pasture species in the past. The introduction of exotic pasture species should not be seen as a substitute for sensible land management, especially since research has indicated that careful management of native pastures can produce weight gain rates in cattle approaching that of improved pastures (Lonsdale 1994).

10.2 MANAGEMENT RECOMMENDATIONS

The examination of a broad range of vegetation community components indicated that different sites showed significant results for different attributes. While some attributes, such as ground cover, may be a reliable indicator of over-utilisation on some pastures, it may not necessarily provide a clear indication of pasture decline on others. Similarly, the use of certain species such as *Aristida calycina* as biological indicators of over-utilisation may also not provide ample warnings of transitions between states of pasture condition. It is therefore recommended that pastures be regularly assessed by the land manager using a broad range of attributes to obtain a concise and accurate view of the current pasture condition and to assess the opportunities or hazards for a transition between states. These attributes would include comparative abundances of particular functional groups, to recognise a shift from productive pastures dominated by palatable perennial native grasses to unproductive pastures dominated by unpalatable or annual species. The results of the present research suggests that cattle grazing and retention of such productive pastures are non mutually exclusive. Such responsible cattle grazing have positive environmental as well as economic benefits.

Avoiding hazardous transitions to increasingly degraded pastures would include de-stocking during severe drought periods. Allowing cattle to remain on drought-affected native pastures through the provision of feed supplements allows over-utilisation of pastures to occur. Results from the present research also showed that drought-affected trees become increasingly susceptible to dieback in grazed pastures, and that this affects larger specimens of high conservation value, while allowing for the increased survival of young saplings.

There are a number of management techniques that would allow favourable transitions to occur. Results from Rishton, Lucky Downs dam and Tabletop indicated that retention of areas of native pastures within fenced off areas might be of benefit to graziers as a seed source to allow recruitment of productive and palatable decreaser species. Positive transitions to a more productive condition would then still be possible even if the surrounding pastures deteriorated to the point that those valuable species were lost. Such valuable seed reserves currently exist along fenced road verges and railway lines. Opportunistic de-stocking of pastures when recruitment events of decreaser pasture species are occurring may potentially result in a rapid transition to a more diverse and productive pasture.

Regular monitoring of ground cover, the proportions of known increaser and decreaser species and functional groups, and abundance of undesirable species can forewarn graziers of impending deliterious changes to pastures. With this information, strategic spelling of pastures can be used to allow for improvement of pasture condition.

In conclusion, an increase in grazing pressure results in changes to plant communities that are undesirable from both a cattle production and conservation perspective. In contrast to the commonly held belief that overgrazed pastures recover after good rain events, cattle grazing cause significant changes in the botanical composition of pastures. Grazing strategies that incorporate monitoring of pasture condition, and strategies to allow for retention of biodiversity allows for a higher degree of sustainability of the cattle grazing industry in the semi-arid rangelands of North Queensland.

10.3 LIMITATIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH

The long- term sustainable future of the cattle grazing industry is dependant on the wise utilisation of existing resources, not on achieving higher short-term productivity through the introduction of more exotic pasture species. The semi-arid rangelands of North Queensland contain a relatively high level of diversity. This is a valuable resource that we are obliged to manage wisely for the prosperity of future generations.

Research should be focussed on issues of sustainability; determining impacts of grazing on community structure and determining the thresholds for pasture recovery. Future research should be expanded to include impacts on fauna as well as flora. It was not been possible in the present research to define grazing levels in terms of actual stocking rates. Being able to define sustainable grazing in terms of actual cattle numbers would be an invaluable tool for graziers, although there could never be any substitute for regular on-site monitoring of pasture condition by the land manager.

The QDPI exclosures used in this research proved to be a valuable tool, however, improvements could be made to the macropod exclosures. It would be advantageous to establish a greater area for total herbivore exclusion. This proposal is obviously limited by funding availability, and it may be some time before these areas could be used as replicates of the current exclosure. However, the immediate and repeated sampling of a heavily grazed area suddenly excluded from all forms of grazing would present an excellent opportunity to examine in minute detail the expected transition of the pasture to a less degraded state.

For the Dalrymple Shire, I would recommend that a formal long-term grazing-gradient experiment be established. Grazing gradients occur with increasing distance from water points, however, all water points examined in the Dalrymple Shire were located too close to each other for gradients of any magnitude to develop. An experiment using closure of water points, strategic placements of fences and exclosures to allow a statistically balanced and meaningful data set to be established, would allow a finer detail of our understanding of grazing impact on plant communities. Allowing space for more plots and quadrats on a single soil type would increase the robustness and diversity of statistical analyses. With a larger area, it may be possible to incorporate research into impacts of fauna as well, since the current exclosure sizes are probably too small to support viable populations of decreaser animal species.

For future research examining grazing impacts on functional groups of plants, it is recommended that a hierarchical approach be taken, separating plants first on life form (forbs, grasses and shrubs), secondly on life history strategy, and finally on palatability. For example, such functional groups would include ‘annual palatable grasses’, ‘perennial unpalatable forbs’, ‘perennial palatable grasses’ etc. The further use of dendrograms and Principal Component Analyses to analyse the data is recommended.

In regards to tree deaths, although this study has shown a link between grazing and dieback, there is the opportunity for physiological and ecological studies to examine the exact cause and effect, although such research may have to be carefully timed to coincide with severe droughts. This may be difficult to achieve within the time constraints of the university research system.

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APPENDICES

APPENDIX A: DESCRIPTION OF STUDY SITES

(asterisk * = introduced species)

SITE NUMBER: 1
SITE NAME: Kangaroo Hills (East) (KHILEA)
SITE LOCATION: Dalrymple Shire, Nth Queensland
S $18^0$54'14.2" E $145^0$37'56.9" 55 K 0355974 UTM 79O9237
DATE SURVEYED: 16th March 1999
SOILS: Greenvale (GR) Brown/ Yellow Chromosols: Dark greyish-brown sand to sandy loam over structured dark yellowish brown to brownish yellow clay. (Rogers *et al.* 1999)
CLIMATE: Average yearly rainfall 640mm (Gillard 1983)
VEGETATION TYPE: *Eucalyptus crebra* open woodland
EXPERIMENT TYPE: Cattle and macropod exclosure
DESCRIPTION OF SITE: This site is located on an open woodland site on a slight slope. Anecdotal evidence suggests level of grazing is quite low, and this is supported by the general land condition. Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.
DESCRIPTION OF UNGRAZED PLOTS: Vegetation is open *Eucalyptus crebra* woodland with some *Carissa lanceolata, Erythroxlum australe, Petalostigma pubescens* and *Maytenus cunninghamii* in the shrub layer. The grassy understorey consists mostly of *Heteropogon contortus,* which occupied between 56 and 90% of quadrats. A subdominant grass was *Panicum larcomianum.* Numerous herbaceous plants were very common here, particularly *Crotalaria juncea* (up to 43% of quadrats), and also *Cyanthillium cinereum* and *Pratia concolor.* Overall diversity measured in plots varied between 44 and 55 species.
DESCRIPTION OF GRAZED PLOTS: Vegetation is open spaced woodland dominated by *Eucalyptus crebra* with some individuals of *Corymbia dallachiana* and *Corymbia erythrophloia.* Other woody plants include *Carissa lanceolata, Petalostigma pubescens* and *Grevillea striata*. Charcoal was present on the ground, however, no trees showed fire scars on their trunks. The grassy understorey consists mostly of *Heteropogon contortus,* which occupied between 36 and 73% of quadrats. A subdominant grass was *Panicum larcomianum.* The grass is tall (~1.5m) and rank. The understorey was highly diverse, with *Eragrostis brownii, Fimbristylis dichotoma*, *Brunoniella acaulis* subsp. *acaulis* and *Indigofera linnaei* all having a high abundance in the plots. Overall diversity measured in plots varied between 47 and 52 species.
DESCRIPTION OF MACROPOD EXCLOSURE: Vegetation is open woodland dominated by *Eucalyptus crebra* (89% of trees), with some *Corymbia dallachiana* present. A shrub layer was indistinct, comprised of *Breynia obtusifolia* and *Grewia retusifolia.* Abundant grasses were *Heteropogon contortus* (53.3% of quadrats) and *Bothriochloa bladei* (43% of quadrats). Also abundant in the understorey was *Emelia sonchifolia, Brunoniella acaulis* subsp. *acaulis, Cyanthillium cinereum*, and *Pratia concolor*; each occurring in over a third of quadrats. Overall diversity measured in the plot was 48 species.
NOTES: Gillard (1983) notes that this area was originally dominated by *Themeda triandra,* which has been largely replaced by *Heteropogon contortus.* Pasture development experiments using the exotic African grass *Urochloa mosambicensis* and the exotic legume *Stylosanthes hamata* were undertaken on this property in 1965 (Gillard 1983).

SITE NUMBER: 2
SITE NAME: Kangaroo Hills (West) (KHILWT)
SITE LOCATION: Dalrymple Shire, Nth Queensland
S 18^{0}54’21.2” E 145^{0}32’36.9” 55 K 0346613 UTM 7908945
DATE SURVEYED: 18th March 1999

SOILS: Greenvale (GR) Brown/ Yellow Chromosols: Dark greyish-brown sand to sandy loam over structured dark yellowish brown to brownish yellow clay. (Rogers *et al.* 1999)

CLIMATE: Average yearly rainfall 640mm (Gillard 1983)

VEGETATION TYPE: *Eucalyptus crebra* open woodland

EXPERIMENT TYPE: Cattle and macropod exclosure

DESCRIPTION OF SITE: This site is located in an open woodland site on a slight slope. Anecdotal evidence suggests the level of grazing is quite low, and the general land condition supports this suggestion. Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.

DESCRIPTION OF UNGRAZED PLOTS: Vegetation is *Eucalyptus crebra* (71% of trees) to a height of 15 metres, with scattered *Corymbia dallachiana* and *Corymbia erythrophloia*. A shrub layer consisted of *Petalostigma pubescens,* and *Maytenus cunninghamii*. The grassy understorey consists mostly of *Themeda triandra,* which occupied 50-70% of quadrats. A co-dominant grass was *Heteropogon contortus,* which occupied between 36 and 50% of quadrats. Other common plants in the herb/grass layer included *Melinus repens**, *Eragrostis brownii* and *Cyanthillium cinereum*. Overall diversity measured in plots varied between 40 and 52 species. Macropod scats were common.

DESCRIPTION OF GRAZED PLOTS: Vegetation is dominated by *Eucalyptus crebra* (86% of trees) with some individuals of *Corymbia dallachiana* and *Corymbia erythrophloia*. A shrub layer consisted of *Acacia bidwillii, Bursaria tenuifolia, Geijera salicifolia Melaleuca viridiflora* and *Petalostigma pubescens*. The grassy understorey consists mostly of *Heteropogon contortus,* which occupied between 30 - 80% of quadrats. Co-dominant grasses were *Melinus repens** (30-40%) and *Sorghum plumosum* (16-53%). The understorey was highly diverse, with *Emilia sonchifolia*, *Evolvulus alsinoides, Fimbristylis dichotoma, Glycine clandestina* and *Eragrostis brownii* all having a high abundance in the plots. Overall diversity measured in plots varied between 47 and 71 species. Clumps of Kangaroo Grass had been selectively grazed. Scats of macropods and cattle were noted.

DESCRIPTION OF MACROPOD EXCLOSURE: Vegetation is dominated by sparse *Eucalyptus crebra* (71% of trees), with some *Corymbia erythrophloia* present. A shrub layer comprised of *Acacia bidwillii, Petalostigma pubescens* and *Dolichandrone heterophylla.* Abundant grasses were *Heteropogon contortus* (63% of quadrats) and *Themeda triandra* (53% of quadrats). Also abundant in the understorey was *Melinus repens** occurring in 46% of quadrats. Overall diversity measured in the plot was 47 species.

NOTES: Gillard (1983) notes that this area was originally dominated by *Themeda triandra,* but had been largely replaced by *Heteropogon contortus.* The exotic pasture grass *Urochloa mosambicensis* and the exotic legume *Stylosanthes hamata* were released on the Kangaroo Hills property in 1965 (Gillard 1983).

SITE NUMBER: 3
SITE NAME: Blue Range (BLURNG)
SITE LOCATION: Dalrymple Shire, Nth Queensland
S 19^{0}00'31.3" E 145^{0}28'52.3" 55 K 0340139 UTM 7897514
DATE SURVEYED: 17th March 1999

SOILS: Greenvale (GR) Brown/ Yellow Chromosols: Dark greyish-brown sand to sandy loam over structured dark yellowish brown to brownish yellow clay. (Rogers *et al.* 1999)

CLIMATE: Average yearly rainfall 640mm (Gillard 1983)

VEGETATION TYPE: *Eucalyptus crebra* open woodland

EXPERIMENT TYPE: Cattle and macropod exclosure

DESCRIPTION OF SITE: Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.

DESCRIPTION OF UNGRAZED PLOTS: Vegetation is open *Eucalyptus crebra* woodland to 15 metres (84 % of woody plants) with *Petalostigma pubescens* and some *Dolichandrone heterophylla* in the shrub layer. The thick grassy understorey chiefly consists of *Heteropogon contortus*, which occupied between 33-53% of quadrats. A subdominant grass was *Panicum larcomianum* (16-53%). Other common grasses included *Eragrostis brownii* and *Chrysopogon fallax*. Overall diversity measured in plots varied between 39 and 43 species.

DESCRIPTION OF GRAZED PLOTS: This plot is bordered to the north by an erosion gully, surrounded by a zone of increasingly bare ground. Otherwise, the vegetation is dominated by *Eucalyptus crebra* (78% of trees) with some individuals of *Corymbia dallachiana* and a single *Corymbia erythrophloia* tree. A shrub layer consisted of *Petalostigma pubescens, Ventilago viminalis* and *Eremophila mitchelli.* The grassy understorey was mixed, with no clear dominants in all 3 plots. Common species included *Panicum larcomianum* (up to 53%), *Heteropogon contortus* (up to 30%) and *Panicum effusum (up to 43%).* Other common herbaceous plants included *Indigofera linnaei* and *Polymeria longifolia.* Overall diversity measured in plots varied between 48 and 52 species. Scats of macropods were noted and the area was criss-crossed with cattle tracks.

DESCRIPTION OF MACROPOD EXCLOSURE: Vegetation is dominated by *Eucalyptus crebra* (62% of trees), with some *Corymbia dallachiana* present. A shrub layer consisted of *Acacia bidwillii* and *Dolichandrone heterophylla.* Abundant grasses were *Heteropogon contortus* and *Panicum effusum* (each 43% of quadrats) and the sedge *Fimbristylis dichotoma* (46% of quadrats). Also abundant in the understorey was *Chrysopogon fallax, Eragrostis brownii* and *Zornia muriculata* each occurring in nearly a quarter of quadrats. Overall diversity measured in the plot was 53 species.

NOTES Anecdotal evidence suggests that this area has a history of low grazing pressure.

SITE NUMBER: 4
SITE NAME: Leyshon View (LYSHN)
SITE LOCATION: Dalrymple Shire, Nth Queensland
S 20^{0}08'56.0" E 146^{0}28'26.1" 55 K 0445021 UTM 7771949
DATE SURVEYED: 24th November 1998

SOILS: Dalrymple (DA) Red Chromosols: Reddish brown to dark brown loamy sand to sandy loam over structured red to reddish brown clay. (Rogers *et al.* 1999)

CLIMATE: Average yearly rainfall is 659mm/yr (Charters Towers PO)

VEGETATION TYPE: *Eucalyptus crebra/ E. xanthoclada* and *Corymbia erythrophloia* open woodland

EXPERIMENT TYPE: Cattle and macropod exclosure

DESCRIPTION OF SITE: Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.

DESCRIPTION OF UNGRAZED PLOTS: Canopy vegetation is mixed with an ironbark complex (*Eucalyptus crebra/ xanthoclada*) comprising 31% of trees, with the bulk of the remainder consisting of *Corymbia erythrophloia*. Many of the latter had the epiphyte *Cymbidium canaliculatum* growing on them. Specimens of the exotic *Zizyphus mauritiana* also contributed to both the tree and shrub layer. The shrub layer was relatively diverse and contained *Acacia bidwillii, A. farnesiana*, A. julifera, Carissa lanceolata, Dolichandrone heterophylla* and *Eremophila mitchelli*. The grassy understorey consists mostly of *Chrysopogon fallax*, which occupied between 53-83 % of quadrats. A subdominant grass was *Melinus repens** (23-53%). Exotic *Bothriochloa pertusa** was insignificant in 2 plots but was found in 53% of quadrats in plot 3. Common herbaceous plants included several species of *Sida*. Overall diversity measured in plots varied between 36 and 48 species.

DESCRIPTION OF GRAZED PLOTS: Vegetation is dominated by mixed Eucalypts, with *Eucalyptus crebra/ xanthoclada* making up only 19%, while *Corymbia erythrophloia* and *Corymbia dallachiana* contributed significantly. Other woody plants include *Acacia farnesiana*, Atalaya hemiglauca, Archidendropsis basaltica, Carissa lanceolata* and *Zizyphus mauritianum**. The grassy understorey was patchy and consists mostly of, *Bothriochloa pertusa**, which occupied between 70 - 100 % of quadrats. *Chrysopogon fallax* was insignificant in 2 plots and was found in 30% of quadrats in another. Common herbaceous plants included *Indigofera linifolia* and *I. linnaei*. Overall diversity measured in plots varied between 24 and 39 species.

DESCRIPTION OF MACROPOD EXCLOSURE: Vegetation is comprised of a mixed selection of 10 species, *Eucalyptus crebra/ xanthoclada* making up only 11% while exotic *Zizyphus mauritiana* made up 27%. Other common trees included *Corymbia erythrophloia, Corymbia dallachiana* and *Dolichandrone heterophylla*. A shrub layer comprised of *Carissa lanceolata* with exotic *Zizyphus* and *Acacia farnesiana**. Abundant grasses were *Melinus repens** (93% of quadrats) and *Chrysopogon fallax* (53% of quadrats). Also abundant in the understorey was *Spermacoce brachystema* and *Fimbristylis dichotoma*, each occurring in over a third of quadrats. Overall diversity measured in the plot was 39 species.

NOTES: There was a noticeable difference in bulk and average height of the grass layer between treatments, indicating a strong grazing influence on the site. During a previous visit on 16th September 1998, it was noted that erosion on the hill adjoining the site was carrying large quantities of sand and depositing it over parts of the macropod exclosure. These areas had low percentage cover of vegetation, however, a massive germination event of grasses was taking place. These were covered in grass by 24th November 1998. Macropod scats and a Spectacled Hare-wallaby were observed inside the cattle exclosure during sampling.

SITE NUMBER: 5
SITE NAME: Kirk River (KIRKR)
SITE LOCATION: Dalrymple Shire, Nth Queensland
S 20^{0}00'23.3" E 146^{0}43'47.5" 55 K 0471744 UTM 7787773
DATE SURVEYED: 19th January 1999
SOILS: Dalrymple (DA) Red Chromosols: Reddish brown to dark brown loamy sand to sandy loam over structured red to reddish brown clay. (Rogers *et al.* 1999)
CLIMATE: Average yearly rainfall 650mm
VEGETATION TYPE: *Eucalyptus crebra* open woodland
EXPERIMENT TYPE: Cattle and macropod exclosure
DESCRIPTION OF SITE: Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.
DESCRIPTION OF UNGRAZED PLOTS: Woody vegetation is dominated by *Eucalyptus crebra* (84% of total), with 45% (121 of 267) of that species occurring as saplings. Other tree species include *Corymbia erythrophloia* and *Corymbia dallachiana. Eucalyptus crebra* saplings and *Maytenus cunninghamii* dominated the shrub layer. The grassy understorey consists mostly of *Bothriochloa pertusa** which occupied between 93-96% of quadrats. A subdominant grass in one plot was *Chrysopogon fallax* with 36%. Common herbaceous plants included *Indigofera linifolia* (33-43%) and *Glycine* spp. Overall diversity measured in plots varied between 36 and 42 species. (Two Eastern Grey Kangaroos were seen in this exclosure.)
DESCRIPTION OF GRAZED PLOTS: Vegetation is dominated by *Eucalyptus crebra* (72%) with some individuals of *Corymbia erythrophloia* and *Corymbia dallachiana* present. Other woody plants include *Maytenus cunninghamii* and *Zizyphus mauritiana*.* The grassy understorey consists mostly of *Bothriochloa pertusa*,* which occupied between 96-100% of quadrats. The next most common understorey plant was *Indigofera linifolia*, represented in 60 – 93% of quadrats. Overall diversity measured in plots varied between 28 and 34 species. The plot was criss-crossed by cattle tracks and cattle scats were observed. There was no evidence of fire.
DESCRIPTION OF MACROPOD EXCLOSURE: A gully runs through the centre of this plot. Vegetation is dominated by *Eucalyptus crebra* (70% of trees), with some *Corymbia erythrophloia* and *Corymbia dallachiana* present. A shrub layer was composed of *Maytenus cunninghamii and Zizyphus mauritiana*.* Abundant grasses were *Bothriochloa pertusa* (73% of quadrats) and *Dicanthium annulatum* (33% of quadrats). Also common in the understorey was *Chrysopogon fallax* in 26% of quadrats. Overall diversity measured in the plot was 35 species.
NOTES: There was a noticeable difference in bulk and average height of the grass layer between treatments, indicating a strong grazing influence on the site. Specimens of *Chrysopogon fallax* and *Dicanthium* spp were far bigger and more robust in the macropod exclosure than in the cattle exclosure.

SITE NUMBER: 6
SITE NAME: Rishton (RSHTN)
SITE LOCATION: Dalrymple Shire, Nth Queensland
S 20^{0}07’36.2” E 146^{0}32’24.8” 55 K 0451768 UTM 7773500
DATE SURVEYED: 11th April 2000
SOILS: Dalrymple (DA) Red Chromosols: Reddish brown to dark brown loamy sand to sandy loam over structured red to reddish brown clay. (Rogers *et al.* 1999)
CLIMATE: Average yearly rainfall 659.5mm/yr
VEGETATION TYPE: Open *Eucalyptus crebra* woodland
EXPERIMENT TYPE: Fence-line comparison

DESCRIPTION OF SITE: This site represents a mine site with an ungrazed buffer zone between the mine and the adjoining cattle property. Rishton Mine erected their fence in 1988-89, giving an exclosure age of 11-12 years. Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.

DESCRIPTION OF UNGRAZED PLOTS: The tree canopy is dominated by *Eucalyptus crebra*, with some *Corymbia erythrophloia, Bauhinia carronii* and *Pleiogynium timorense* present. A shrub layer was present consisting of *Maytenus cunninghamii, Zizyphus mauritiana**, *Larsenaikia ochreata* and *Petalostigma banksii*. The grassy understorey consists mostly of, *Bothriochloa pertusa** which occupied between 66 - 100% of quadrats. Subdominant grasses were *Heteropogon contortus* (up to 56% in plot 1) and *Chrysopogon fallax* (up to 36% in plot 3). Common herbaceous plants included *Evolvulus alsinoides* and *Stylosanthes scabra**. Overall diversity measured in plots varied between 33 and 47 species.

DESCRIPTION OF GRAZED PLOTS: The tree canopy is dominated by *Eucalyptus crebra*, with some *Corymbia erythrophloia*. A shrub layer was present consisting of *Atalaya hemiglauca, Bursaria incana, Maytenus cunninghamii, Ventilago viminalis* and *Zizyphus mauritiana**. The grassy understorey consists mostly of *Bothriochloa pertusa** which occupied between 60 - 100% of quadrats. Subdominant grasses were *Heteropogon contortus* (up to 60% in plot 2) and *Chrysopogon fallax* (up to 50% in plots 1 & 2). Common herbaceous plants included *Indigofera linnaei* and *Stylosanthes scabra**. Overall diversity measured in plots varied between 29 and 34 species.

NOTES Anecdotal evidence reports that this site has remained unburnt for 20 years.

SITE NUMBER: 7
SITE NAME: Mt Leyshon Mine (site 1 & 2)
SITE LOCATION: Dalrymple Shire, Nth Queensland
SITE 1: S 20^{0}18’26.6” E 146^{0}16’46.3” 55 K 0424780 UTM 7754331, SITE 2: S 20^{0}18’41.9” E 146^{0}17’35.2” 55 K 0426201 UTM 7753866
DATE SURVEYED: 27th January 2000
SOILS: Rangeview (RA) Red Chromosols: Dark reddish brown loam over structured red to yellowish red clay (Rogers *et al.* 1999).
CLIMATE: Average yearly rainfall 660mm (Ken Ramsay pers. comm.).
VEGETATION TYPE: Open *Eucalyptus crebra* woodland
EXPERIMENT TYPE: Fence-line comparison
DESCRIPTION OF SITE: The site represents a mine with an ungrazed and intermittently grazed buffer between the mine and adjoining grazing property. The intermittently grazed buffer zone is only grazed during the months of April/May to November, whilst no grazing occurs during the wet season months of December to March (Ken Ramsay pers. comm.). This buffer has been in place since 1994 - 1995, giving an exclosure age of 5 -6 years. Three replicate 50 X 50m plots were sampled in both the ungrazed and intermittently grazed areas (site 1) and another 3 plots in the lightly grazed and grazed plots (site 2).
DIAGRAM OF SITE LAYOUT

(lines represent fencelines)

Site 1 Ungrazed Mine site ⇐	Site 1 intermittently grazed	Site 2 intermittently grazed	Site 2 grazed

DESCRIPTION OF UNGRAZED PLOTS: The tree canopy was dominated by *Eucalyptus crebra* with some *Corymbia erythrophloia*, *Corymbia dallachiana* and *Eucalyptus brownii.* A shrub layer composed of *Acacia bidwillii, Atalaya hemiglauca, Erythroxlum australe, Maytenus cunninghamii* and *Santalum lanceolatum* The grassy understorey consists mostly of *Cenchrus ciliaris* * (73-100%). Subdominant grasses were *Bothriochloa pertusa** (up to 30% in plot 1) and *Melinus repens** (up to 30% in plot 2). Common herbaceous plants included *Rhynchosia minima*. Overall diversity measured in plots varied between 27and 32 species. No dung was seen.
DESCRIPTION OF INTERMITTENTLY GRAZED PLOTS (sites 1 & 2 combined): Tree vegetation was very sparse, co-dominated by *Eucalyptus crebra* and *Corymbia erythrophloia.* A shrub layer was composed of *Acacia bidwillii, Carissa lanceolata, Maytenus cunninghamii* and *Senna planiticola*. The grassy understorey consists mostly of *Cenchrus ciliaris** (50-66% in site 1, 40-50% in site 2), and *Bothriochloa pertusa** (50 - 60% in site 1, 43-80% in site 2). Subdominant grasses were *Chloris virgata* (site 1) and *Bothriochloa decepiens* (site 2). Common understorey species include *Indigofera linnaei* and *Fimbristylis dichotoma.* Overall diversity in plots varied between 22 and 43 species (site 1) and 34 – 39 species (site 2). Cattle and macropod dung were recorded.
DESCRIPTION OF GRAZED PLOTS: Vegetation is dominated by *Eucalyptus crebra* and *Corymbia erythrophloia*. A shrub layer was composed of *Atalaya hemiglauca, Bursaria incana, Carissa lanceolata* and *Maytenus cunninghamii.* Abundant grasses were *Bothriochloa pertusa** (60-76% of quadrats) and *Cenchrus ciliaris** (56-60% of quadrats). Also common was *Enneapogon polyphyllus* and *Tragus australianus.* Overall diversity in the plots was 27-39 species.
NOTES: Exotic pasture species have been introduced by the mine for rehabilitation (*Cenchrus ciliaris**) and by the graziers for improved pastures (*Cenchrus ciliaris*, Stylosanthes scabra**).

SITE NUMBER: 8
SITE NAME: Jervoise
SITE LOCATION: Dalrymple Shire, Nth Queensland
S 18^{0}59’28.9” E 144^{0}41’18.7” 55 K 0256654 UTM 7898524
DATE SURVEYED: 10th March 2000

SOILS: Rangeview (RA) Red Chromosols: Dark reddish brown loam over structured red to yellowish red clay (Rogers *et al.* 1999).

CLIMATE: Average yearly rainfall 711mm
VEGETATION TYPE: Open *Eucalyptus crebra* woodland
EXPERIMENT TYPE: Fence-line comparison

DESCRIPTION OF SITE: The Gregory Developmental Highway west of Greenvale splits this site into treatments. The grazed plot is located on the western side of the highway and is bordered by a fence. The ungrazed site is on the eastern side of the highway. It is protected from grazing on the highway side by the grazed plot fenceline, and is protected from grazing from the east by a rugged “U” shaped hill approximately 150 metres high. A precise age of the fence could not be determined, but is thought to exceed 20 years. Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.

DESCRIPTION OF UNGRAZED PLOTS: Vegetation is dominated by *Eucalyptus crebra*, with some *Corymbia erythrophloia* and *Corymbia dallachiana* present. A sparse shrub layer was evident and was mostly composed of *Maytenus cunninghamii*, with few *Hakea lorea*. The grassy understorey consists mostly of *Themeda triandra,* which occupied between 56 - 76% of quadrats. A subdominant grass was *Dicanthium annulatum*, located in 30-50% of quadrats. Common herbaceous plants included *Chamaesyce macgillivrayi* and *Indigofera linnaei*. Overall diversity measured in plots varied between 42 and 49 species. No dung was noted.

DESCRIPTION OF GRAZED PLOTS: Vegetation is dominated by open *Eucalyptus crebra*, with some *Corymbia dallachiana* present. A sparse shrub layer was evident and was mostly composed of *Acacia bidwillii, Carissa lanceolata* and *Maytenus cunninghamii*. The grassy understorey consists mostly of *Bothriochloa pertusa**, which occupied between 80-90% of quadrats. A subdominant grass was Themeda triandra (26-53%). Common herbaceous species included *Indigofera colutea, Indigofera linnaei* and *Sida rhombifolia**. Overall diversity measured in plots varied between 44 and 52 species. Cattle dung was noted.

NOTES: Specimens of *Themeda triandra* were noticeably smaller in the grazed plots than in the ungrazed

SITE NUMBER: 9
SITE NAME: Pajingo Mine (PAJ)
SITE LOCATION: Dalrymple Shire, Nth Queensland
S $20^0$31’54.5” E $146^0$25’23.7” 55 K 0439873 UTM 7729553
DATE SURVEYED: 31st March 2000
SOILS: Pentland (PE) Red Kandosols: Dark reddish brown to dark brown sandy clay loam grading to earthy red light clay (Rogers *et al.* 1999).
CLIMATE: Average yearly rainfall 600mm (Stuart Roseby pers. comm.).
VEGETATION TYPE: Mid-high woodland of *Acacia shirleyi*.
EXPERIMENT TYPE: Fence-line comparison
DESCRIPTION OF SITE: Pajingo is a gold mine with a fenceline separating the mine and Pallamanna cattle station, excluding stock from a buffer zone around the mine. Cattle were excluded in 1987; giving an exclosure age of 13 years (Stuart Roseby pers. comm.). Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.
DESCRIPTION OF UNGRAZED PLOTS: Vegetation is *Acacia shirleyi* to 15 metres, often with connecting canopies. Some *Corymbia clarksoniana* and *Eucalyptus crebra* were also represented in the upper canopy. A diverse and distinct shrub layer consisted of *Acacia leptostachya, Erythroxylon australe, Grevillea decora, Larsenaikia ochreata, Petalostigma banksii* and numerous saplings of *Acacia shirleyi.* The grassy understorey consists mostly of *Chrysopogon fallax,* (23-76%) and *Schizachyrium fragile* (30-76%). Common herbaceous plants included *Evolvulus alsinoides, Goodenia cycloptera* and *Setaria surgens*. Overall diversity measured in plots varied between 46 and 54 species.
DESCRIPTION OF GRAZED PLOTS: Vegetation is *Acacia shirleyi* to 15 metres, often with connecting canopies. Some *Corymbia dallachiana, Corymbia erythrophloia* and *Eucalyptus crebra* were also represented in the upper canopy. A diverse and distinct shrub layer consisted of *Erythroxylon australe, Everistia vacciniifolia, Grevillea dryandri, Larsenaikia ochreata, Petalostigma banksii* and numerous saplings of *Acacia shirleyi.* The grassy understorey consists mostly of *Aristida calycina,* which occupied between 43 - 73% of quadrats. Other common grasses were *Schizachyrium fragile, Thyridolepis mitchelliana* and *Chrysopogon fallax.* Herbaceous plants were common, although most species existed at low densities. Overall diversity measured in plots varied between 35 and 49 species.
NOTES: The presence of carnivorous plants may suggest low soil fertility.

SITE NUMBER: 10
SITE NAME: Lucky Downs Dam (LCKDAM)
SITE LOCATION: Dalrymple Shire, Nth Queensland
S 19^{0}02'11.5" E 144^{0}50'07.0" 55 K 0272170 UTM 7893718
DATE SURVEYED: 11th March 2000
SOILS: Wairuna (WR) Red Dermosols: Dark reddish brown to dark brown clay loam over structured dark red to yellowish red clay (Rogers *et al.* 1999).
CLIMATE: Average yearly rainfall 711mm/yr
VEGETATION TYPE: Open *Eucalyptus crebra* woodland
EXPERIMENT TYPE: Fence-line comparison
DESCRIPTION OF SITE: This site is a large fenced-off triangle surrounded by freely grazed woodland. Age of the exclosure is 10 years. Cattle have made infrequent incursions into the fenced off triangle (Atkinson H. pers. comm.), so those plots are described as "lightly grazed". Three replicate 50 X 50m plots were sampled in both the grazed and lightly grazed areas.
DESCRIPTION OF LIGHTLY GRAZED PLOTS: The tree stratum consists of open *Eucalyptus crebra* woodland to 15-20 metres. A shrub layer consisted of *Indigofera pratensis, Maytenus cunninghamii*, *Petalostigma banksii* and *Petalostigma pubescens*. The grassy understorey consists mostly of *Heteropogon contortus*, (occupied between 30-53% of quadrats) and *Chrysopogon fallax* (46-60%). A sub-dominant grass was *Bothriochloa ewartiana* (23-50% of quadrats). Common herbaceous plants included *Brunoniella acaulis* subsp. *acaulis* and *Stylosanthes scabra**. Overall diversity measured in plots varied between 39 and 54 species. Cattle and macropod dung was noted
DESCRIPTION OF GRAZED PLOTS: The tree stratum consists of open *Eucalyptus crebra* woodland to 15-20 metres. A shrub layer consisted of *Atalaya hemiglauca, Carissa lanceolata* and *Dolichandrone heterophylla*. The grassy understorey consists mostly of *Heteropogon contortus*, which occupied between 40-80% of quadrats. Other common grasses were *Aristida calycina, Bothriochloa ewartiana, Chrysopogon fallax* and *Digitaria brownii*. Common herbaceous plants included *Brunoniella acaulis* subsp. *acaulis* and *Sida rhombifolia*. Overall diversity measured in plots varied between 44 and 56 species.
NOTES Limited tree clearing was conducted circa 1966; however, areas obviously affected by that event were avoided. Deliberate introductions of the exotic pasture legume *Stylosanthes scabra** have been made in the past (Atkinson H.pers. comm).

SITE NUMBER: 11
SITE NAME: Epping Forest (Site 1)
SITE LOCATION: Belyando Shire, North Queensland
S 22^{0}22’40.6” E 146^{0}41’15.6” 55K 0467846 UTM 7525306
DATE SURVEYED: 17th May 2000

SOILS: Epping Chromosols (informal name): Neutral red duplex soils – A horizon soft and loamy with red clay subsoils (Isbell & Hubble 1967).

CLIMATE: Average yearly rainfall 671mm (Woolnough & Johnson 2000)

VEGETATION TYPE: *Eucalyptus brownii* woodland

EXPERIMENT TYPE: Fence-line comparison

DESCRIPTION OF SITE: This site is the boundary between Epping Forest National Park and an adjoining cattle property. The park has been fenced since 1981, giving an exclusion age of 19 years. Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.

DESCRIPTION OF UNGRAZED PLOTS: Vegetation is dominated by *Eucalyptus brownii* with some *Eucalyptus crebra, Eucalyptus coolabah* ssp. *coolabah* and *Grevillea striata*. The shrub layer consists of *Acacia excelsa, Carissa lanceolata, Maytenus cunninghamii* and *Opuntia stricta**. The grassy understorey consists mostly of *Cenchrus ciliaris**, which occupied between 73-76% of quadrats. A subdominant grass was *Chrysopogon fallax* in 26-46% of quadrats. Common herbaceous plants included *Chaemaecrista absus*, Fimbristylis dichotoma, Glycine tomentella* and *Indigofera colutea*. Overall diversity measured in plots varied between 32 and 43 species. Kangaroo dung was noted.

DESCRIPTION OF GRAZED PLOTS: Vegetation is dominated by *Eucalyptus brownii* with some *Eucalyptus coolabah* ssp. *coolabah* and *Terminalia oblongata*. The shrub layer includes *Acacia excelsa, Carissa lanceolata, Denhamia oleaster, Eremophila mitchelli* and *Maytenus cunninghamii*. The grassy understorey consists mostly of *Cenchrus ciliaris**, which occupied between 33-100% of quadrats. A co-dominant grass was *Bothriochloa ewartiana* in 23-70% of quadrats. Common herbaceous plants included *Fimbristylis dichotoma, Glycine tomentella* and *Phyllanthus* type 7. Overall diversity measured in plots varied between 40 and 44 species. Dung of cattle, kangaroos and rabbits were noted.

NOTES: Large numbers of *Eucalyptus brownii* were dead.

SITE NUMBER: 12
SITE NAME: Epping Forest (Site 2)
SITE LOCATION: Belyando Shire, North Queensland
S 22^{0}22’17.6” E 146^{0}41’32.5” 55K 0468328 UTM 7526013
DATE SURVEYED: 18th May 2000

SOILS: Walthum Tenosols (informal name): Siliceous Sands: Deep sand with uniform texture profile – very deep sands derived from sandy plains and slightly elevated sand-filled prior stream channels (Isbell & Hubble 1967).

CLIMATE: Average yearly rainfall 671mm (Woolnough & Johnson 2000)

VEGETATION TYPE: Mixed Eucalypt woodland. Classified by Sattler & Williams (1999) as “Regional ecosystem 11.3.7: Tall woodland of *Corymbia clarksoniana, C. tessellaris* and *C. dallachiana* on Cainozoic alluvial sandy soils”, it is listed as “of concern”.

EXPERIMENT TYPE: Cattle and macropod exclosure site

DESCRIPTION OF SITE: A fence enclosing the National Park was erected in 1981, giving an exclosure age of 19 years. This allows access to kangaroos, wombats and smaller grazing animals, but prevents cattle entry. Immediately adjacent to this is a macropod exclosure measuring 50 X 50m erected circa 1986 (exclosure age of 14 years). This exclosure prevents grazing by any mammals. Outside cattle and macropods freely graze outside these exclosures. Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.

DESCRIPTION OF UNGRAZED PLOTS: Vegetation is mixed open forest composed of *Corymbia dolichocarpa*, *Eucalyptus coolabah* ssp. *coolabah* and *Bauhinia carronii*. The shrub layer includes *Acacia excelsa, Alectryon diversifolius, Maytenus cunninghamii* and *Opuntia stricta**. The grassy understorey consists mostly of *Cenchrus ciliaris**, which occupied between 43-56% of quadrats. Other common grasses were *Perotis rara* and *Enneapogon flavescens*. Common herbaceous plants included *Evolvulus alsinoides* and *Jacquemontia paniculata*. Overall diversity measured in plots varied between 46 and 57 species. Dung of kangaroos, wombats and rabbits were noted.

DESCRIPTION OF GRAZED PLOTS This site was located close to a cattle dam. Vegetation is mixed open forest composed of *Corymbia tessellaris, Eucalyptus brownii, Eucalyptus coolabah* ssp. *coolabah* and *Eucalyptus crebra*. The shrub layer included *Alphitonia exclesa, Erythroxylon australe* and *Petalostigma pubescens*. The grassy understorey consists mostly of *Cenchrus ciliaris**, which occupied between 56-86% of quadrats. Other common grasses were *Perotis rara* and *Aristida* spp. Common herbaceous plants included *Indigofera colutea, Indigofera hirsuta* and *Sida rhombifolia**. Overall diversity measured in plots varied between 29 and 43 species. Dung of cattle and kangaroos were noted.

DESCRIPTION OF MACROPOD EXCLOSURE: Trees are completely absent from this plot, although no clearing has been undertaken. A shrub layer was nearly absent and comprised of *Owenia acidula* and *Ventilago viminalis*. The dominant grass was *Enneapogon flavescens* (46% of quadrats), with *Cenchrus ciliaris** in only 6% of quadrats. Common herbaceous plants included *Waltheria indica* and *Jacquemontia paniculata*. Overall diversity measured in the plot was 33 species.

NOTES: The macropod exclosure was originally erected in 1986 for pasture modelling purposes. Burning and mowing experiments were undertaken in September 1987 and October 1988, causing changes to species composition. By 1994, these effects had virtually disappeared (Day K. pers. comm). The age of the exclosure should be regarded as 12 years (1988-2000).

SITE NUMBER: 13
SITE NAME: Tabletop
SITE LOCATION: S 19^{0}24'00.3" E146^{0}26'08.3" 55K 0440745 UTM 7854800
DATE SURVEYED: 6th June 2000
SOILS: Bluff: Brown Sodosols: Greyish brown sand to sandy loam over alkaline structured yellowish brown clay (Rogers *et al*. 1999).
CLIMATE: Average yearly rainfall
VEGETATION TYPE: Mid-high *Melaleuca viridiflora* and mixed Eucalypt woodland
EXPERIMENT TYPE: Fence-line comparison

DESCRIPTION OF SITE: This site is comprised of openly grazed woodland, bordered by a rubbish dump surrounded by a fence to exclude stock. Three replicate 50 X 50m plots were sampled in both the grazed and ungrazed areas.

DESCRIPTION OF UNGRAZED PLOTS: Vegetation is predominantly *Melaleuca viridiflora* to 10 metres with some specimens of *Corymbia clarksoniana*, *Corymbia dallachiana* and *Eucalyptus platyphylla* in the tree strata. The grassy understorey consists mostly of *Heteropogon contortus,* which occupied between 70-93% of quadrats. A co-dominant was the introduced legume *Stylosanthes scabra* (53-83%). The sedge *Fimbristylis dichotoma* was also very common (73-93% of quadrats). Common herbaceous plants included *Epaltes australis, Mitrasacme* spp, and *Hybanthus enneaspermus*. Overall diversity measured in plots varied between 45 and 46 species.

DESCRIPTION OF GRAZED PLOTS: Vegetation is predominantly *Melaleuca viridiflora* to 10 metres with some specimens of *Corymbia clarksoniana, Corymbia dallachiana* and occasional *Eucalyptus shirleyi* in the tree strata. The short, cropped grassy understorey consists mostly of *Heteropogon contortus,* which occupied between 90-100% of quadrats. The understorey was highly diverse, with many species having a high abundance in the plots. Common species include *Cyperus squarrosus, Drosera indica, Epaltes australis, Mitrasacme* spp*, Selaginella gracillima* and numerous species of Scrophulariaceae. Overall diversity measured in plots varied between 68 and 72 species. Cattle dung was noted.

NOTES: There was a noticeable difference in bulk and average height of the grass layer between treatments, indicating a strong grazing influence on the site. In particular, specimens of *Heteropogon contortus* were much larger in the ungrazed sites. The abundance of carnivorous plants (*Drosera*) may suggest low soil fertility.

APPENDIX B: EXAMPLE PRO-FORMA DATA COLLECTION SHEET

SITE						DATE			
PLOT / TREATMENT									
SPECIES									
QUADRAT	COVER	1	2	3	4	5	6	7	8
1									
2									
3									
4									
5									
6									
7									
8									
9									
10									
11									
12									
13									
14									
15									
16									
17									
18									
19									
20									
21									
22									
23									
24									
25									
26									
27									
28									
29									
30									

APPENDIX C: CATALOGUES OF SPECIES OBSERVED IN STUDY SITES

Appendix C (a): List of species sorted by family, and their respective numerical codes for use on pro-forma data sheets
(asterisk * = introduced species, hash # = species represented in field herbarium)

No.	TAXON
	Acanthaceae
421	*Brunoniella acaulis #*
848	*Brunoniella australis*
719	*Pseuderanthemum variable*
522	*Rostellularia adscendens #*
	Adiantaceae
426	*Cheilanthes sieberi #*
	Amaranthaceae
404	*Achyranthes aspera*
409	*Alternanthera ficoidea* #*
2	*Alternanthera denticulata #*
831	*Alternanthera nodiflora*
408	*Alternanthera pungens**
412	*Amaranthus viridis*
461	*Gomphrena celosioides* #*
792	*Gomphrena lanata #*
	Anacardiaceae
3	*Pleiogynium timorense*
	Apocynaceae
4	*Carissa lanceolata*
5	*Parsonsia lanceolata #*
341	*Wrightia saligna*
	Asclepiadaceae
9	*Cryptostegia grandiflora**
	Asteraceae
345	*Bidens pilosa #**
12	*Blumea saxatilis #*
13	*Blumea (tenella?) #*
834	*Blumea sp. A #*
716	*Calotis cuneifolia #*
852	*Camptacra barbata #*
15	*Chrysocephalum apiculatum #*
16	*Conyza canadensis var.pusilla*#*
130	*Cyanthillium cinereum #*
17	*Emelia sonchifolia* #*
18	*Epaltes australis #*
459	*Gnaphalium polycaulon #*
20	*Peripleura hispidula #*
21	*Peripleura sp. #*
23	*Pterocaulon serrulatum #*
514	*Pterocaulon sphacelatum #*
853	*Sigesbeckia orientalis #*
383	*Sonchus oleraceus**
610	*Tridax procumbens**
130	*Vernonia cinerea #*
27	*Wedelia spilanthoides #*
572	*Xanthium pungens**
	Bignoniaceae
34	*Dolichandrone heterophylla*
	Boraginaceae

No.	TAXON
783	*Ehretia membranifolia #*
465	*Heliotropium pauciflorum #*
35	*Heliotropium ventricosum #*
37	*Trichodesma zeylanicum #*
	Brassicaceae
786	*Rorippa sp #*
	Cactaceae
338	*Opuntia stricta **
	Caesalpiniaceae
342	*Bauhinia carronii*
754	*Chamaecrista absus*#*
38	*Chamaecrista mimosoides #*
884	*Chamaecrista rotundifolia#*
39	*Senna planitiicola #*
	Campanulaceae
45	*Pratia concolor #*
571	*Wahlenbergia caryophylloides #*
46	*Wahlenbergia gracilis #*
47	*Wahlenbergia graniticola #*
	Capparaceae
798	*Capparis lasiantha*
611	*Capparis mitchelli*
48	*Capparis umbonata #*
612	*Cleome viscosa*
	Caryophyllaceae
49	*Polycarpaea corymbosa #*
	Celastraceae
343	*Denhamia oleaster*
50	*Maytenus cunninghamii #*
	Chenopodiaceae
774	*Chenopodium carinatum #*
51	*Dysphania glomulifera #*
779	*Dysphania littoralis #*
445	*Einadia nutans ssp. nutans #*
525	*Salsola kali*
371	*Sclerolaena calcarata #*
52	*Sclerolaena lanicuspis #*
	Clusiaceae
53	*Hypericum gramineum #*
	Combretaceae
185	*Terminalia oblongata*
	Commelinaceae
431	*Commelina (cyanea)? #*
54	*Murdannia gigantea #*
493	*Murdannia graminea #*
	Convolvulaceae
818	*Bonamia sp. A #*
452	*Evolvulus alsinoides #*
367	*Ipomoea eriocarpa #*
613	*Ipomoea gracilis #*

No.	TAXON
863	*Ipomoea muelleri #*
614	*Ipomoea plebia #*
615	*Ipomoea polymorpha #*
55	*Jaquemontia paniculata #*
56	*Polymeria longifolia*
57	*Xenostegia tridentata #*
	Cucurbitaceae
58	*Cucumis anguria* #*
59	*Mukia maderaspatana #*
	Cyperaceae
805	*Bulbostylis barbata #*
775	*Cyperus betchei #*
60	*Cyperus concinnus #*
785	*Cyperus difformis #*
366	*Cyperus fulvus #*
365	*Cyperus perangustus #*
369	*Cyperus rotundus*
873	*Cyperus squarrosus #*
826	*Cyperus sp. A*
877	*Cyperus sp. B*
377	*Cyperus sp C*
378	*Cyperus sp D*
379	*Cyperus sp E*
827	*Cyperus sp F*
916	*Fimbristylis acicularis #*
454	*Fimbristylis dichotoma #*
914	*Fimbristylis microcarya #*
897	*Fimbristylis schoenoides #*
905	*Fuirena ciliaris #*
908	*Lipocarpha microcephala #*
881	*Rhynchospora pterochaeta #*
61	*Scleria brownii #*
364	*Scleria sphacelata #*
889	*Scleria sp A*
	Droseraceae
890	*Drosera indica #*
705	*Drosera spatulata*
	Erythroxylaceae
62	*Erythroxlum australe*
	Euphorbiaceae
64	*Breynia obtusifolia #*
449	*Chamaesyce drummondii #*
450	*Chamaesyce hirta* #*
65	*Chamaesyce macgillivrayi #*
67	*Chamaesyce mitchelliana #*
761	*Euphorbia tannensis #*
751	*Flueggea leucopyrus #*
69	*Petalostigma banksii #*
335	*Petalostigma pubescens*
68	*Phyllanthus fuernrohrii #*
845	*Phyllanthus type 1*
500	*Phyllanthus* type 2 #

No.	TAXON
702	_Phyllanthus_ type 3 #
811	_Phyllanthus_ type 4 #
166	_Phyllanthus_ type 5 #
854	_Phyllanthus_ type 6 #
767	_Phyllanthus_ type 7 #
919	_Phyllanthus_ type 9 #
855	_Phyllanthus_ type 10 #
856	_Phyllanthus_ type 11 #
857	_Phyllanthus_ type 12 #
370	_Poranthera microphylla_ #
70	_Sauropus trachyspermus_ #
	Fabaceae
616	_Aeschynomene brevifolia_ *#
405	_Aeschynomene indica_* #
375	_Alysicarpus bupleurifolius_
71	_Alysicarpus glumoceus_ #
72	_Alysicarpus ovalifolius_ #
814	_Alysicarpus rugosus_ #
794	_Aphyllodium biarticulatum_ #
73	_Cajanus scarabaeoides_ #
753	_Canavalia papuana_
74	_Crotalaria brevis_ #
75	_Crotalaria calycina_ #
77	_Crotalaria goreensis_* #
79	_Crotalaria juncea_* #
80	_Crotalaria lanceolata_ #
81	_Crotalaria medicaginea_ #
376	_Crotalaria montana_
82	_Crotalaria novaehollandae_ #
83	_Crotalaria verrucosa_ #
789	_Desmodium filiforme_ #
934	_Desmodium macrocarpum_ #
84	_Desmodium (muelleri)_ #
85	_Desmodium nemorosum_ #
882	_Desmodium trichostachyum_ #
816	_Desmodium sp. aff. gunnii_ #
363	_Galactia muelleri_ #
362	_Galactia tenuiflora_ #
88	_Glycine clandestina_ #
763	_Glycine tabacina_ #
458	_Glycine tomentella_ #
799	_Glycine sp. A_
90	_Indigofera colutea_ #
91	_Indigofera hirsuta_ #
471	_Indigofera linifolia_ #
470	_Indigofera linnaei_ #
617	_Indigofera parviflora_ #
92	_Indigofera pratensis_ #
823	_Indigofera sp. nov. 2_ #
521	_Rhynchosia minima_ #
815	_Rhynchosia sp. nov._ #
550	_Stylosanthes hamata_
551	_Stylosanthes humilis_* #
549	_Stylosanthes scabra_ #
618	_Tephrosia dietrichiae_ #
619	_Tephrosia filipes_
94	_Tephrosia flagellaris_ #
554	_Tephrosia juncea_ #
352	_Tephrosia leptoclada_ #
164	_Uraria cylindrica_
351	_Uraria lagopodioides_ #
567	_Vigna lanceolata_ #

No.	TAXON
95	_Vigna radiata_ #
96	_Zornia adenophora_ #
97	_Zornia areolata_ #
98	_Zornia dyctiocarpa_ #
99	_Zornia muelleriana_ #
574	_Zornia muriculata_ #
	Goodeniaceae
182	_Calogyne pilosa_ #
361	_Dampiera adpressa_ #
360	_Goodenia armitiana_ #
100	_Goodenia cycloptera_ #
706	_Goodenia disperma_
359	_Goodenia hirsuta_ #
357	_Goodenia purpurascens_ #
	Haloragaceae
810	_Gonocarpus humilis_ #
	Lamiaceae
469	_Hyptis suaveolens_ *
101	_Ocimum basilicum_* #
	Lauraceae
620	_Cassytha pubescens_
	Liliaceae
128	_Crinum flaccidum_ #
103	_Dianella crinoides_ #
887	_Dianella rara_ #
891	_Thysanotus tuberosus_
	Loganiaceae
104	_Mitrasacme connata_ #
903	_Mitrasacme micrantha_ #
932	_Mitrasacme nummularia_ #
105	_Mitrasacme pygmaea_ #
	Lythraceae
930	_Ammannia multiflora_ #
	Malvaceae
401	_Abutilon otocarpum_ #
107	_Gossypium australe_ #
468	_Hibiscus meraukensis_ #
709	_Hibiscus sturtii_ #
828	_Hibiscus trionum_ #
486	_Malvastrum americanum_*
536	_Sida cordifolia_ #
108	_Sida fibulifera/filiformis?_ #
621	_Sida phaeotricha_ #
538	_Sida rhombifolia_ #
110	_Sida rohlenae_ #
539	_Sida spinosa_ #
541	_Sida subspicata_ #
858	_Sida trichopoda_ #
711	_Sida_ sp. nov "Pajingo" #
	Marsileaceae
487	_Marsilea drummondii_
	Meliaceae
340	_Owenia acidula_
	Mimosaceae
636	_Acacia bidwillii_
339	_Acacia calycina_
190	_Acacia coriacea_
112	_Acacia excelsa_ #
113	_Acacia farnesiana_*

No.	TAXON
188	_Acacia harpophylla_
114	_Acacia julifera_ #
622	_Acacia leptostachya_
356	_Acacia oligophleba_ #
115	_Acacia shirleyi_
116	_Archidendropsis basaltica_ #
496	_Neptunia gracilis_
	Molluginaceae
117	_Mollugo cerviana_ #
	Myoporaceae
189	_Eremophila bignoniiflora_
118	_Eremophila mitchelli_
	Myrtaceae
346	_Corymbia clarksoniana_
120	_Corymbia dallachiana_
344	_Corymbia dolichocarpa_
121	_Corymbia erythrophloia_
193	_Corymbia terminalis_
336	_Corymbia tessellaris_
122	_Eucalyptus brownii_
772	_Eucalyptus coolabah_ #
123	_Eucalyptus crebra_ #
197	_Eucalyptus platyphylla_
892	_Eucalyptus shirleyi_ #
347	_Lithomyrtus microphylla_
194	_Eucalyptus whitei_
187	_Lysicarpus angustifolius_
186	_Melaleuca nervosa_
878	_Melaleuca viridiflora_
	Nyctaginaceae
354	_Boerhavia dominii_ #
420	_Boerhavia paludosa_ #
	Oleaceae
125	_Jasmimum didymum_
	Onagraceae
368	_Ludwigia octovalvis_
	Orchidaceae
126	_Cymbidium canaliculatum_
	Passifloraceae
750	_Passiflora foetida_*
	Pittosporaceae
127	_Bursaria incana_
	Plumbaginaceae
755	_Plumbago zeylanica_ #
	Poaceae
129	_Allopteropsis cimicina_ #
102	_Allopteropsis semialata_ #
109	_Aristida calycina_ #
111	_Aristida contorta_ #
793	_Aristida holathera_ #
116	_Aristida hygrometrica_ #
119	_Aristida jerichoensis_ #
66	_Aristida latifolia_ #
844	_Aristida (macroclada?)_ #
124	_Aristida pernicosa_ #
128	_Aristida queenslandica_ #
159	_Aristida sp trailing_ #
838	_Aristida sp. A_ #
389	_Aristida sp. B_ #

No.	TAXON
24	*Bothriochloa bladhii #*
40	*Bothriochloa decepiens #*
25	*Bothriochloa ewartiana #*
41	*Bothriochloa pertusa**
145	*Brachiaria gilesii #*
624	*Brachiaria holosericea #*
802	*Brachiaria piligera*
178	*Brachiaria subquadripara #*
800	*Brachiaria sp. A #*
181	*Brachyachne convergens #*
809	*Capillipedium parviflorum #*
7	*Cenchrus ciliaris**
829	*Chionachne hubbardiana #*
158	*Chloris pectinata*
161	*Chloris virgata* #*
43	*Chrysopogon fallax #*
162	*Cleistochloa subjuncea #*
859	*Cymbopogon ambiguus #*
26	*Cymbopogon bombycinus #*
169	*Dactyloctenium aegyptii*
14	*Dactyloctenium radulans*
170	*Dicanthium annulatum #*
832	*Dicanthium aristatum*
177	*Digitaria ammophila #*
179	*Digitaria brownii #*
180	*Digitaria ciliaris #*
184	*Digitaria parviflora #*
381	*Digitaria* type B #
382	*Digitaria* type C #
712	*Digitaria* type D #
176	*Echinochloa colona**
195	*Enneapogon avenaceus*
202	*Enneapogon polyphyllus #*
200	*Enneapogon nigricans #*
208	*Eragrostis brownii #*
212	*Eragrostis elongata*
215	*Eragrostis leptostachya*
221	*Eragrostis sororia #*
224	*Eragrostis tenellula*
384	*Eragrostis type A #*
860	*Eragrostis type C #*
883	*Eragrostis type E #*
713	*Eragrostis type F #*
773	*Eragrostis type G #*
708	*Eragrostis type H #*
386	*Eragrostis type I #*
387	*Eragrostis type J #*
388	*Eragrostis type K #*
625	*Eriachne ciliata #*
921	*Eriachne pallescens #*
237	*Eriochloa pseudoacrotricha #*
817	*Eriochloa* sp. *A*
30	*Eulalia aurea*
165	*Hackelochloa granularis #*
31	*Heteropogon contortus*
32	*Heteropogon triticeus*
240	*Ischaemum fragile #*
243	*Iseilema vaginiflorum#*
76	*Melinus repens* #*
252	*Mnesithea formosa*
813	*Mnesithea rottboellioides #*
257	*Panicum decompositum #*
258	*Panicum effusum #*
260	*Panicum larcomianum #*
131	*Panicum mitchellii #*
262	*Panicum queenslandicum*
266	*Paspalidium distans #*
820	*Paspalidium flavidum #*
191	*Paspalidium gracile #*
825	*Paspalidium sp. A*
273	*Paspalum scrobiculatum #*
19	*Perotis rara #*
840	*Pseudo-Aristida sp.*
879	*Pseudopogonatherum irritans #*
285	*Schizachyrium fragile #*
921	*Schizachyrium pseudeulalia#*
286	*Sehima nervosum*
290	*Setaria pumila* #*
33	*Sorghum plumosum #*
132	*Sporobolus australasicus #*
874	*Sporobolus diandrus #*
307	*Sporobolus indicus* #*
78	*Sporobolus caroli*
316	*Thaumastochloa pubescens #*
36	*Themeda triandra*
317	*Thyridolepis mitchelliana #*
22	*Tragus australianus*
721	*Triodia mitchelli*
63	*Tripogon loliiformis #*
329	*Urochloa mosambicensis* #*
	Polygalaceae
862	*Polygala exsquarrosa #*
133	*Polygala linariifolia #*
	Portulacaceae
507	*Portulaca oleracea*
508	*Portulaca pilosa #*
	Primulaceae
89	*Anagallis pumila*#*
	Proteaceae
875	*Grevillea dryandri*
134	*Grevillea paralella #*
135	*Grevillea striata*
136	*Hakea lorea #*
851	*Persoonia falcata*
	Rhamnaceae
137	*Alphitonia excelsa*
138	*Ventilago viminalis #*
139	*Zizyphus mauritiana**
	Rubiaceae
140	*Canthium coprosmoides*
141	*Canthium oleifolium*
349	*Everistia vacciniifolia #*
374	*Larsenaikia ochreata*
894	*Mitracarpus hirtus * #*
833	*Oldenlandia argillacea #*
106	*Oldenlandia mitrasacmoides #*
142	*Spermacoce brachystema*
350	*Synaptantha tillaeacea*
	Rutaceae
143	*Geijera salicifolia #*
	Santalaceae
372	*Santalum lanceolatum*
	Sapindaceae
808	*Alectryon diversifolius #*
144	*Atalaya hemiglauca*
628	*Distichostemon hispidulus#*
	Scrophulariaceae
812	*Buchnera tetragona #*
898	*Centranthera cochinchinensis #*
872	*Gratiola pedunculata #*
899	*Lindernia crustacea #*
192	*Lindernia sp. A*
907	*Mimulus uvedaliae #*
145	*Striga curviflora #*
	Selaginellaceae
880	*Selaginella gracillima* #
	Solanaceae
843	*Physalis minima*
146	*Solanum dianthophorum #*
543	*Solanum ellipticum*
723	*Solanum ferrocissimum #*
806	*Solanum semiarmatum #*
	Sterculiaceae
491	*Melhania oblongifolia*
147	*Waltheria indica*
	Stylidiaceae
821	*Stylidium alsinoides #*
148	*Stylidium eriorrhizum #*
	Thymelaeaceae
149	*Thecanthes corncopiae #*
	Tiliaceae
895	*Corchorus trilocularis*
463	*Grewia retusifolia*
	Typhaceae
561	*Typha domingensis*
	Verbenaceae
373	*Clerodendrum floribundum*
	Violaceae
150	*Hybanthus enneaspermus #*
	Vitaceae
358	*Cayratia trifolia #*
627	*Cissus opaca #*
	Xanthorrhoeaceae
151	*Lomandra (filiformis)*
152	*Lomandra multiflora #.*
	Zygophyllaceae
557	*Tribulus terrestris*
	Family Unknown
837	Unknown vine sp.
	Other features
153	Termite mound
154	Cattle Dung
760	Wombat Dung
156	Rabbit Dung
155	Kangaroo Dung

Appendix C (b): Plant species sorted by the numerical codes used in the present research.

code	species	code	species	code	species
1	*Brunoniella acaulis*	55	*Jaquemontia paniculata*	105	*Mitrasacme pygmaea*
2	*Alternanthera denticulata*	56	*Polymeria longifolia*	106	*Oldenlandia mitrasacmoides*
3	*Pleiogynium timorense*	57	*Xenostegia tridentata*	107	*Gossypium australe*
4	*Carissa lanceolata*	58	*Cucumis anguria**	108	*Sida fibulifera*
5	*Parsonsia lanceolata*	59	*Mukia maderaspatana*	109	*Aristida calycina*
7	*Cenchrus ciliaris*	60	*Cyperus concinnus*	110	*Sida rohlenae*
8	*Aristolochia thozetii*	61	*Scleria brownii*	111	*Aristida contorta*
9	*Cryptostegia grandiflora**	62	*Erythroxlum australe*	112	*Acacia excelsa*
10	*Panicum maximum**	63	*Tripogon loliiformis*	113	*Acacia farnesiana**
11	*Aristida spp.* (generic)	64	*Breynia obtusifolia*	114	*Acacia julifera*
12	*Blumea saxatilis*	65	*Chamaesyce macgillivrayi*	115	*Acacia shirleyi*
13	*Blumea (tenella?)*	66	*Aristida latifolia*	116	*Aristida hygrometrica*
14	*Dactyloctenium radulans*	67	*Chaemaesyce mitchelliana*	117	*Mullugo cerviana*
15	*Chrysocephalum apiculatum*	68	*Phyllanthus fuernrohrii*	118	*Eremophila mitchelli*
16	*Conyza canadensis**	69	*Petalostigma banksii*	119	*Aristida jerichoensis*
17	*Emelia sonchifolia**	70	*Sauropus trachyspermus*	120	*Corymbia dallachiana*
18	*Epaltes australis*	71	*Alysicarpus glumoceus*	121	*Corymbia erythrophloia*
19	*Perotis rara*	72	*Alysicarpus ovalifolius*	122	*Eucalyptus brownii*
20	*Peripleura hispidula*	73	*Cajanus scarabaeoides*	123	*Eucalyptus crebra*
21	*Peripleura sp.*	74	*Crotalaria brevis*	124	*Aristida (pernicosa)*
22	*Tragus australianus*	75	*Crotalaria calycina*	125	*Jasmimum didymum*
23	*Pterocaulon serrulatum*	76	*Melinus repens*	126	*Cymbidium canaliculatum*
24	*Bothriochloa bladhii*	77	*Crotalaria goreensis**	127	*Bursaria incana*
25	*Bothriochloa ewartiana*	78	*Sporobolus caroli*	128	*Aristida queenslandica*
26	*Cymbopogon bombycinus*	79	*Crotalaria juncea**	129	*Allopteropsis cimicina*
27	*Wedelia spilanthoides*	80	*Crotalaria lanceolata*	130	*Cyanthillium cinereum*
30	*Eulalia aurea*	81	*Crotalaria medicaginea*	131	*Panicum mitchelii*
31	*Heteropogon contortus*	82	*Crotalaria novaehollandae*	132	*Sporobolus australasicus*
32	*Heteropogon triticeus*	83	*Crotalaria verrucosa*	133	*Polygala linariifolia*
33	*Sorghum plumosum*	84	*Desmodium (muelleri)*	134	*Grevillea paralella*
34	*Dolichandrone heterophylla*	85	*Desmodium nemorosum*	135	*Grevillea striata*
35	*Heliotropium ventricosum*	86	*Desmodium rhytidophyllum*	136	*Hakea lorea*
36	*Themeda triandra*	87	*Desmodium clandestina v. sericea*	137	*Alphitonia excelsa*
37	*Trichodesma zeylanicum*			138	*Ventilago viminalis*
38	*Chamaecrista mimosoides*	88	*Glycine clandestina*	139	*Zizyphus mauritiana**
39	*Senna planiticola*	89	*Anagallis pumila*	140	*Canthium coprosmoides*
40	*Bothriochloa decepiens*	90	*Indigofera colutea*	141	*Canthium oleifolium*
41	*Bothriochloa pertusa*	91	*Indigofera hirsuta*	142	*Spermacoce brachystema*
42	*Brachiaria* spp.	92	*Indigofera pratensis*	143	*Geijera salicifolia*
43	*Chrysopogon fallax*	93	*Indigofera* sp nov?	144	*Atalaya hemiglauca*
44	*Cynodon dactylon*	94	*Tephrosia flagellaris*	145	*Striga curviflora*
45	*Pratia concolor*	95	*Vigna radiata*	146	*Solanum dianthophorum*
46	*Wahlenbergia (gracilis)*	96	*Zornia adenophora*	147	*Waltheria indica*
47	*Wahlenbergia graniticola*	97	*Zornia areolata*	148	*Stylidium eriorrhizum*
48	*Capparis umbonata*	98	*Zornia dyctiocarpa*	149	*Thecanthes corncopiae*
49	*Polycarpaea corymbosa*	99	*Zornia muelleriana*	150	*Hybanthus enneaspermus*
50	*Maytenus cunninghamii*	100	*Goodenia cycloptera*	151	*Lomandra (filiformis?)*
51	*Dysphania glomulifera*	101	*Ocimum basilicum**	152	*Lomandra multiflora*
52	*Sclerolaena lanicuspis*	102	*Allopteropsis semialata*	158	*Chloris pectinata*
53	*Hypericum gramineum*	103	*Dianella crinoides*	159	*Aristida schultzii*
54	*Murdannia gigantea*	104	*Mitrasacme connata*	161	*Chloris virgata*

code	**species**
162	*Cleistochloa subjuncea*
164	*Uraria cylindrica*
165	*Hackelochloa granularis*
166	*Phyllanthus type 5*
169	*Dactyloctenium aegyptii **
170	*Dicanthium annulatum*
172	*Dicanthium sp.*
176	*Echinochloa colona**
177	*Digitaria ammophylla*
178	*Brachiaria subquadripara*
179	*Digitaria brownii*
180	*Digitaria ciliaris*
181	*Brachyachne convergens*
182	*Calogyne pilosa*
184	*Digitaria parviflora*
185	*Terminalia oblongata*
186	*Melaleuca nervosa*
187	*Lysicarpus angustifolius*
188	*Acacia harpophylla*
189	*Eremophila bignoniflora*
190	*Acacia coriacea*
191	*Paspalidium gracile*
192	*Lindernia sp. A*
193	*Corymbia terminalis*
194	*Eucalyptus whitei*
195	*Enneapogon avenaceus*
196	*Enneapogon flavescens*
197	*Eucalyptus platyphylla*
199	*Enneapogon lindleyanus*
200	*Enneapogon nigricans*
202	*Enneapogon polyphyllus*
208	*Eragrostis brownii*
212	*Eragrostis elongata*
214	*Earagrostis lacunaria*
221	*Eragrostis sororia*
224	*Eragrostis tenellula*
229	*Eriachne mucronata*
236	*Eriochloa procera*
237	*Eriochloa pseudoacrotricha*
240	*Ischaemum fragile*
243	*Iseilema vaginiflorum*
252	*Mnesithea formosa*
257	*Panicum decompositum*
258	*Panicum effusum*
260	*Panicum larcomianum*
262	*Panicum queenslandicum*
266	*Paspalidium distans*
273	*Paspalum scrobiculatum*
285	*Schizachyrium fragile*
286	*Sehima nervosum*
290	*Setaria pumila ssp pallide-fusca*
292	*Setaria surgens*
302	*Sporobolus actinocladus*
307	*Sporobolus indicus var. capensis*
316	*Thaumastochloa pubescens*
328	*Triraphis mollis*
329	*Urochloa mosambicus*
335	*Petalostigma pubescens*
336	*Corymbia tessellaris*
338	*Opuntia stricta**
339	*Acacia calycina*
340	*Owenia acidula*
341	*Wrightia saligna*
342	*Bauhinia carronii*
343	*Denhamia oleaster*
344	*Corymbia dolichocarpa*
345	*Bidens pilosa**
346	*Corymbia clarksoniana*
347	*Lithomyrtus microphylla*
349	*Everistia vacciniifolia*
350	*Synaptantha tillaeacea*
351	*Uraria lagopodioides*
352	*Tephrosia leptoclada*
354	*Boerhavia dominii*
356	*Acacia oligophleba*
357	*Goodenia purpurascens*
358	*Cayratia trifolia*
359	*Goodenia hirsuta*
360	*Goodenia armitiana*
361	*Dampiera adpressa*
362	*Galactia tenuifolia*
363	*Galactia muelleri*
364	*Scleria sphacelata*
365	*Cyperus perangustus*
366	*Cyperus fulvus*
367	*Ipomoea eriocarpa*
368	*Ludwigia octovalvis*
369	*Cyperus rotundus*
370	*Poranthera microphylla*
371	*Sclerolaena calcarata*
372	*Santalum lanceolatum*
373	*Clerodendrum floribundum*
374	*Larsenaikia ochreata*
375	*Alysicarpus bupleurifolius*
376	*Crotalaria montana*
377	*Cyperus sp. C*
378	*Cyperus sp. D*
379	*Cyperus sp. E*
380	*Fimbristylis sp. A*
381	*Digitaria type B*
382	*Digitaria type C*
383	*Sonchus oleraceus**
384	*Eragrostis type A*
385	*Eragrostis type B*
386	*Eragrostis type I*
387	*Eragrostis type J*
388	*Eragrostis type K*
389	*Aristida type B*
401	*Abutilon otocarpum*
404	*Achyranthes aspera*
405	*Aeschynomene indica*
408	*Alternanthera pungens*
409	*Alternanthera ficoidea**
412	*Amaranthus viridis*
420	*Boerhavia paludosa*
421	*Brunoniella acaulis subsp. acaulis*
426	*Cheilanthes sieberi*
445	*Einadia nutans ssp. nutans*
449	*Chamaesyce hirta*
452	*Evolvulus alsinoides*
454	*Glycine tomentella*
458	*Glycine tomentella*
459	*Gnaphalium polycaulon*
461	*Gomphrena celestoides*
463	*Grewia retusifolia*
465	*Heliotropium pauciflorum*
468	*Hibiscus meraukensis*
470	*Indigofera linnaei*
471	*Indigofera linifolia*
474	*Ipomoea sp.*
480	*Lomandra filiformis*
482	*Macroptilium lathyroides**
483	*Marsilea drummondii*
491	*Melhania oblongifolia*
493	*Neptunia gracilis*
498	*Oxalis radicosa*
500	*Portulaca oleracea*
507	*Portulaca pilosa*
508	*Pterocaulon sphacelatum*
521	*Rhynchosia minima*
522	*Rostellularia adscendens*
525	*Salsola kali*
536	*Sida cordifolia*
538	*Sida rhombifolia*
539	*Sida spinosa*
541	*Sida subspicata*
543	*Solanum ellipticum*
548	*Stachytarpheta jamaicensis*
549	*Stylosanthes scabra*
550	*Stylosanthes hamata*
551	*Stylosanthes humilis*
554	*Tephrosia juncea*
563	*Verbena officinalis*
567	*Hibiscus meraukensis*
568	*Wahlenbergia caryophylloides*
572	*Xanthium pungens**
610	*Tridax procumbens**
611	*Capparis mitchelli*

code	**species**
612	*Cleome viscosa*
613	*Ipomoea gracilis*
614	*Ipomoea plebia*
615	*Ipomoea polymorpha*
616	*Aeschynomene brevifolia**
617	*Indigofera parviflora*
618	*Tephrosia dietrichiae*
619	*Tephrosia filipes*
620	*Cassytha pubescens*
621	*Sida phaeotricha*
622	*Acacia leptostachya*
623	*Astrebla squarrosa*
624	*Brachiaria holosericea*
625	*Eriachne ciliata*
626	*Lantana camara**
627	*Cissus opaca*
702	*Phyllanthus type 3*
705	*Drosera spatulata*
708	*Eragrostis type H (8)*
709	*Hibiscus sturtii*
710	*Thyridolepis mitchelliana*
711	*Sida sp. nov. "Pajingo"*
712	*Digitaria type D (4)*
713	*Eragrostis type F(6)*
716	*Calotis cuneifolia*
718	*Paspalidium gracile*
719	*Pseuderanthemum variable*
721	*Triodia mitchelli*
723	*Solanum ferrocissimum*
751	*Passiflora foetida**
753	*Flueggea leucopyrus*
754	*Canavalia papuana*
755	*Chamaecrista absus**
756	*Zornia prostrata*
757	*Aristida sciuroides*
758	*Enneapogon robustissimus*
759	*Panicum whitei*
761	*Euphorbia tannensis*
763	*Glycine tabacina*
766	*Brachiaria subquadriparia*
767	*Phyllanthus type 7*
772	*Eucalyptus coolabah ssp. coolabah*
773	*Eragrostis type G (7)*
774	*Chenopodium carinatum*
775	*Cyperus betchei*
778	*Brachiaria sp*
779	*Dysphania littoralis*
783	*Ehretia membranifolia*
785	*Cyperus difformis*

code	**species**
786	*Rorippa sp.*
789	*Desmodium filiforme*
791	*Eragrostis type D (4)*
792	*Gomphrena lanata*
793	*Aristida holathera.*
794	*Aphyllodium biarticulatum*
798	*Capparis lasiantha*
799	*Glycine sp. A*
800	*Brachiaria sp. A*
802	*Brachiaria piligera*
805	*Bulbostylis barbata*
806	*Solanum semiarmatum*
808	*Alectryon diversifolius*
809	*Capillipedium parviflorum*
810	*Gonocarpus humilis*
811	*Phyllanthus type 4*
813	*Buchnera tetragona*
814	*Alysicarpus rugosus*
815	*Rhynchosia sp. nov.*
816	*Desmodium* sp. aff. *gunnii*
818	*Bonamia sp. A*
820	*Paspalidium flavidum*
821	*Stylidium alsinoides*
823	*Indigofera sp nov #2 "Emu Valley"*
826	*Cyperus sp. A*
827	*Cyperus sp. F*
828	*Hibiscus trionum*
829	*Chionachne hubbardiana*
830	*Glycine sp.*
831	*Alternanthera nodiflora*
832	*Dicanthium aristatum*
833	*Oldenlandia argillacea*
834	*Blumea sp. A*
835	*Digitaria type E (5)*
838	*Aristida sp. A*
840	*Pseudoaristida grass sp.*
843	*Physalis minima*
844	*Aristida (macroclada?)*
845	*Phyllanthus type 1*
848	*Brunoniella australis*
851	*Persoonia falcata*
852	*Camptacra barbata*
853	*Sigesbeckia orientalis**

code	**species**
854	*Phyllanthus type 6*
855	*Phyllanthus type 10*
856	*Phyllanthus type 11*
857	*Phyllanthus type 12*
858	*Sida trichopoda*
859	*Cymbopogon ambiguus*
860	*Eragrostis type C (3)*
862	*Polygala exaquarrosa*
863	*Ipomoea muelleri (?)*
872	*Gratiola pedunculata*
873	*Cyperus squarrosus*
874	*Sporobolus (diandrus?)*
875	*Grevillea dryandri*
877	*Cyperus sp. B (2)*
878	*Melaleuca viridiflora*
879	*Pseudopogonatherum irritans*
880	*Selaginella gracillima*
881	*Rhynchospora pterochaeta*
882	*Desmodium trichostachyum*
883	*Eragrostis type E (5)*
884	*Chamaecrista rotundifolia*
887	*Mitrasacme nummularia*
889	*Scleria type A (1)*
890	*Drosera indica*
891	*Thysanotus tuberosus*
892	*Eucalyptus shirleyi*
894	*Mitracarpus hirtus**
895	*Corchorus trilocularis*
896	*Eriachne glauca var. barbinodis*
897	*Fimbristylis schoenoides*
898	*Centranthera cochinchinensis*
899	*Lindernia crustacea*
903	*Mitrasacme micrantha*
905	*Fuirena ciliaris*
907	*Mimulus uvedaliae*
908	*Lipocarpha microcephala*
914	*Fimbristylis microcarya*
916	*Fimbristylis acicularis*
917	*Schizachyrium pseueulalia*
919	*Phyllanthus type 9*
921	*Eriachne pallescens var gracilis*
930	*Ammannia multiflora*

Appendix C (c): Plant species sorted by alphabetical order

No.	Species
401	Abutilon otocarpum
636	Acacia bidwillii
339	Acacia calycina
190	Acacia coriacea
112	Acacia excelsa
113	Acacia farnesiana
188	Acacia harpophylla
114	Acacia julifera
622	Acacia leptostachya
356	Acacia oligophleba
115	Acacia shirleyi
404	Achyranthes aspera
616	Aeschynomene brevifolia
405	Aeschynomene indica
808	Alectryon diversifolius
129	Allopteropsis cimicina
102	Allopteropsis semialata
137	Alphitonia excelsa
409	Alternanthera ficoidea
2	Alternanthera denticulata
831	Alternanthera nodiflora
408	Alternanthera pungens
375	Alysicarpus bupleurifolius
71	Alysicarpus glumoceus
72	Alysicarpus ovalifolius
814	Alysicarpus rugosus
412	Amaranthus viridis
89	Anagallis pumila
794	Aphyllodium biarticulatum
629	Archidendropsis basaltica
844	Aristida (macroclada?)
124	Aristida (pernicosa)
109	Aristida calycina
111	Aristida contorta
793	Aristida holathera
116	Aristida hygrometrica
119	Aristida jerichoensis
66	Aristida latifolia
128	Aristida queenslandica
757	Aristida sciuroides
159	Aristida sp (trailing)
838	Aristida sp. A
11	Aristida spp. (generic)
389	Aristida type B
144	Atalaya hemiglauca
342	Bauhinia carronii
345	Bidens pilosa
13	Blumea (tenella?)
12	Blumea saxatilis
834	Blumea sp. A
354	Boerhavia dominii
420	Boerhavia paludosa
818	Bonamia sp. A
24	Bothriochloa bladhii
40	Bothriochloa decepiens
25	Bothriochloa ewartiana
41	Bothriochloa pertusa
624	Brachiaria holosericea
802	Brachiaria piligera
778	Brachiaria sp
800	Brachiaria sp. A
178	Brachiaria subquadripara
181	Brachyachne convergens
64	Breynia obtusifolia
421	Brunoniella acaulis subsp. acaulis
848	Brunoniella australis
812	Buchnera tetragona
805	Bulbostylis barbata
127	Bursaria incana
73	Cajanus scarabaeoides
182	Calogyne pilosa
716	Calotis cuneifolia
852	Camptacra barbata
753	Canavalia papuana
140	Canthium coprosmoides
141	Canthium oleifolium
809	Capillipedium parviflorum
798	Capparis lasiantha
611	Capparis mitchelli
48	Capparis umbonata
4	Carissa lanceolata
620	Cassytha pubescens
154	Cattle dung
358	Cayratia trifolia
7	Cenchrus ciliaris
898	Centranthera cochinchinensis
754	Chaemaecrista absus
38	Chaemaecrista mimosoides
884	Chaemaecrista rotundifolia
67	Chaemaesyce mitchelliana
449	Chamaesyce drummondii
450	Chamaesyce hirta
65	Chamaesyce macgillivrayi
426	Cheilanthes sieberi
774	Chenopodium carinatum
829	Chionachne hubbardiana
158	Chloris pectinata
161	Chloris virgata
15	Chrysocephalum apiculatum
43	Chrysopogon fallax
627	Cissus opaca
162	Cleistochloa subjuncea
612	Cleome viscosa
373	Clerodendrum floribundum
431	Commelina (cyanea?)
16	Conyza canadensis
895	Corchorus trilocularis
346	Corymbia clarksoniana
120	Corymbia dallachiana
344	Corymbia dolichocarpa
121	Corymbia erythrophloia
193	Corymbia terminalis
336	Corymbia tessellaris
74	Crotalaria brevis
75	Crotalaria calycina
77	Crotalaria goreensis
79	Crotalaria juncea
80	Crotalaria lanceolata
81	Crotalaria medicaginea
376	Crotalaria montana
82	Crotalaria novaehollandae
83	Crotalaria verrucosa
9	Cryptostegia grandiflora
58	Cucumis anguria
130	Cyanthillium cinereum
126	Cymbidium canaliculatum
26	Cymbopogon bombycinus
44	Cynodon dactylon
775	Cyperus betchei
60	Cyperus concinnus
785	Cyperus difformis
366	Cyperus fulvus
365	Cyperus perangustus
369	Cyperus rotundus
826	Cyperus sp. A
877	Cyperus sp. B (2)
377	Cyperus sp. C
378	Cyperus sp. D
379	Cyperus sp. E
827	Cyperus sp. F
873	Cyperus squarrosus
169	Dactyloctenium aegyptii
14	Dactyloctenium radulans
361	Dampiera adpressa
343	Denhamia oleaster
84	Desmodium (muelleri)
789	Desmodium filiforme
934	Desmodium macrocarpum
85	Desmodium nemorosum
816	Desmodium sp. aff. gunnii
882	Desmodium trichostachyum
103	Dianella crinoides
887	Dianella rara
170	Dicanthium annulatum
832	Dicanthium aristatum
703	Digitaria (minima)?
177	Digitaria ammophylla
179	Digitaria brownii
180	Digitaria ciliaris
184	Digitaria parviflora
381	Digitaria type B
382	Digitaria type C
835	Digitaria type E (5)
628	Distichostemon hispidulus
34	Dolichandrone heterophylla
705	Drosera burmanii
890	Drosera indica
51	Dysphania glomulifera
779	Dysphania littoralis
176	Echinochloa colona
783	Ehretia membranifolia
445	Einadia nutans ssp. nutans
17	Emelia sonchifolia
195	Enneapogon avenaceus
196	Enneapogon flavescens
200	Enneapogon nigricans
202	Enneapogon polyphyllus
758	Enneapogon robustissimus
18	Epaltes australis
208	Eragrostis brownii
214	Eragrostis lacunaria
221	Eragrostis sororia
224	Eragrostis tenellula
384	Eragrostis type A
860	Eragrostis type C (3)
883	Eragrostis type E (5)
713	Eragrostis type F(6)

No.	Species
773	Eragrostis type G (7)
708	Eragrostis type H (8)
386	Eragrostis type I
387	Eragrostis type J
388	Eragrostis type K
189	Eremophila bignoniflora
118	Eremophila mitchelli
625	Eriachne ciliata
896	Eriachne glauca var. barbinodis
229	Eriachne mucronata
921	Eriachne pallescens var gracilis
237	Eriochloa pseudoacrotricha
62	Erythroxlum australe
122	Eucalyptus brownii
772	Eucalyptus coolabah ssp. coolabah
123	Eucalyptus crebra
197	Eucalyptus platyphylla
892	Eucalyptus shirleyi
194	Eucalyptus whitei
30	Eulalia aurea
761	Euphorbia tannensis
349	Everistia vacciniifolia
452	Evolvulus alsinoides
916	Fimbristylis acicularis
454	Fimbristylis dichotoma
914	Fimbristylis microcarya
897	Fimbristylis schoenoides
751	Flueggea leucopyrus
905	Fuirena ciliaris
363	Galactia muelleri
362	Galactia tenuifolia
143	Geijera salicifolia
88	Glycine clandestina
799	Glycine sp. A
763	Glycine tabacina
458	Glycine tomentella
459	Gnaphalium polycaulon
461	Gomphrena celesioides
792	Gomphrena lanata
810	Gonocarpus humilis
360	Goodenia armitiana
100	Goodenia cycloptera
706	Goodenia disperma
359	Goodenia hirsuta
357	Goodenia purpurascens
107	Gossypium australe
872	Gratiola pedunculata
875	Grevillea dryandri
134	Grevillea paralella
135	Grevillea striata
463	Grewia retusifolia
165	Hackelochloa granularis
136	Hakea lorea
465	Heliotropium pauciflorum
35	Heliotropium ventricosum
31	Heteropogon contortus
32	Heteropogon triticeus
468	Hibiscus meraukensis
709	Hibiscus sturtii
828	Hibiscus trionum
150	Hybanthus enneaspermus
53	Hypericum gramineum
90	Indigofera colutea
91	Indigofera hirsuta
471	Indigofera linifolia
470	Indigofera linnaei
617	Indigofera parviflora
92	Indigofera pratensis
823	Indigofera sp nov #2 “Emu Valley”
367	Ipomoea eriocarpa
613	Ipomoea gracilis
863	Ipomoea muelleri (?)
614	Ipomoea plebia
615	Ipomoea polymorpha
474	Ipomoea sp.
240	Ischaemum fragile
243	Iseilema vaginiflorum
55	Jaquemontia paniculata
125	Jasmimum didymum
155	Kangaroo dung
374	Larsenaikia ochreata
899	Lindernia crustacea
192	Lindernia sp. A
908	Lipocarpha microcephala
347	Lithomyrtus microphylla
151	Lomandra (filiformis?)
152	Lomandra multiflora
368	Ludwigia octovalvis
187	Lysicarpus angustifolius
482	Macroptilium atropurpureum
483	Macroptilium lathyroides
487	Marsilea drummondii
50	Maytenus cunninghamii
186	Melaleuca nervosa
878	Melaleuca viridiflora
491	Melhania oblongifolia
76	Melinus repens
907	Mimulus uvedaliae
894	Mitracarpus hirtus
104	Mitrasacme connata
903	Mitrasacme micrantha
885	Mitrasacme nummularia
105	Mitrasacme pygmaea
252	Mnesithea formosa
813	Mnesithea rottboeliodes
59	Mukia maderaspatana
117	Mullugo cerviana
54	Murdannia gigantea
493	Murdannia graminea
496	Neptunia gracilis
101	Ocimum basilicum
833	Oldenlandia argillacea
106	Oldenlandia mitrasacmoides
338	Opuntia stricta
340	Owenia acidula
498	Oxalis radicosa
257	Panicum decompositum
258	Panicum effusum
260	Panicum larcomianum
131	Panicum mitchelii
262	Panicum queenslandicum
5	Parsonsia lanceolata
266	Paspalidium distans
820	Paspalidium flavidum
191	Paspalidium gracile
718	Paspalidium gracile
825	Paspalidium sp. A
273	Paspalum scrobiculatum
750	Passiflora foetida
20	Peripleura hispidula
21	Peripleura sp.
19	Perotis rara
851	Persoonia falcata
69	Petalostigma banksii
335	Petalostigma pubescens
68	Phyllanthus fuernrohrii
845	Phyllanthus type 1
855	Phyllanthus type 10
856	Phyllanthus type 11
857	Phyllanthus type 12
500	Phyllanthus type 2
702	Phyllanthus type 3
811	Phyllanthus type 4
166	Phyllanthus type 5
854	Phyllanthus type 6
767	Phyllanthus type 7
919	Phyllanthus type 9
843	Physalis minima
3	Pleiogynium timorense
755	Plumbago zeylanica
49	Polycarpaea corymbosa
862	Polygala exaquarrosa
133	Polygala linariifolia
56	Polymeria longifolia
370	Poranthera microphylla
507	Portulaca oleracea
508	Portulaca pilosa
45	Pratia concolor
719	Pseuderanthemum variable
840	Pseudoaristida grass sp.
879	Pseudopogonatherum irritans
23	Pterocaulon serrulatum
514	Pterocaulon sphacelatum
156	Rabbit dung
766	Brachiaria subquadriparia
521	Rhynchosia minima
815	Rhynchosia sp. nov.
881	Rhynchospora pterochaeta
786	Rorippa sp.
522	Rostellularia adscendens
525	Salsola kali
372	Santalum lanceolatum
70	Sauropus trachyspermus
285	Schizachyrium fragile
917	Schizachyrium pseueulalia
61	Scleria brownii
364	Scleria sphacelata
889	Scleria type A (1)
371	Sclerolaena calcarata
52	Sclerolaena lanicuspis
286	Sehima nervosum
880	Selaginella gracillima
39	Senna planiticola
290	Setaria pumila ssp pallide-fusca
292	Setaria surgens
536	Sida cordifolia
108	Sida fibulifera
621	Sida phaeotricha
538	Sida rhombifolia
110	Sida rohlenae
711	Sida sp. nov. “Pajingo”
539	Sida spinosa
541	Sida subspicata

No.	Species
858	Sida trichopoda
853	Sigesbeckia orientalis
146	Solanum dianthophorum
543	Solanum ellipticum
723	Solanum ferrocissimum
806	Solanum semiarmatum
383	Sonchus oleraceus
33	Sorghum plumosum
142	Spermacoce brachystema
874	Sporobolus (diandrus?)
132	Sporobolus australasicus
78	Sporobolus caroli
307	Sporobolus indicus var. capensis
548	Stachytarpheta jamaicensis
145	Striga curviflora
821	Stylidium alsinoides
148	Stylidium eriorrhizum
550	Stylosanthes hamata
551	Stylosanthes humilis
549	Stylosanthes scabra
350	Synaptantha tillaeacea
817	Tall Brachiaria
618	Tephrosia dietrichiae
619	Tephrosia filipes
94	Tephrosia flagellaris
554	Tephrosia juncea
352	Tephrosia leptoclada
185	Terminalia oblongata
153	Termite mound
316	Thaumastochloa pubescens
149	Thecanthes corncopiae
36	Themeda triandra
710	Thyridolepis mitchelliana
891	Thysanotus tuberosus
22	Tragus australianus
557	Tribulus terrestris
37	Trichodesma zeylanicum
610	Tridax procumbens
721	Triodia mitchelli
63	Tripogon loliiformis
561	Typha domingensis
837	unknown vine species
164	Uraria cylindrica
351	Uraria lagopodioides
329	Urochloa mosambicus
138	Ventilago viminalis
567	Vigna lanceolata
95	Vigna radiata
46	Wahlenbergia (gracilis)
571	Wahlenbergia caryophylloides
47	Wahlenbergia graniticola
147	Waltheria indica
27	Wedelia spilanthoides
760	Wombat dung
341	Wrightia saligna
572	Xanthium pungens
57	Xenostegia tridentata
139	Zizyphus mauritiana
96	Zornia adenophora
97	Zornia areolata
98	Zomia dyctiocarpa
99	Zornia muelleriana
574	Zornia muriculata
756	Zornia prostrata

APPENDIX D: COMPARISON BETWEEN GRAZING TREATMENT OF MEAN NUMBER OF QUADRATS OCCUPIED BYSEVEN COMMON AND DOMINANT GRASS SPECIES

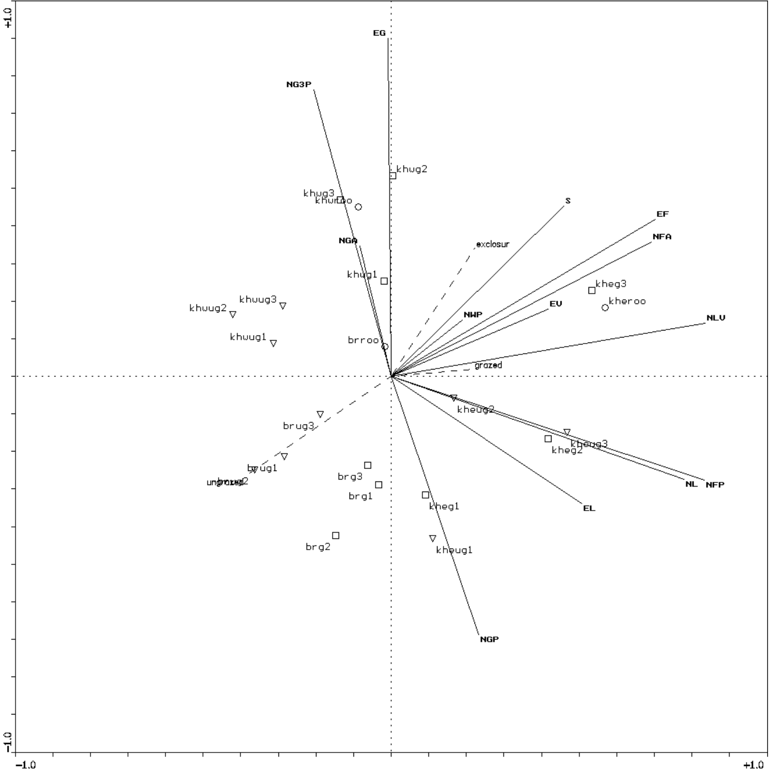
Site	grazing treatment	*Cenchrus ciliaris*	*Bothriochloa pertusa*	*Melinus repens*	*Aristida calycina*	*Heteropogon contortus*	*Chrysopogon fallax*	*Themeda triandra*
Kangaroo Hills East	ungrazed	0	1	0.33	1.33	20.33	2.66	0.66
	grazed	0	0.33	2.66	0.66	17.66	3	0.33
	exclosure	0	0	8	0	16	4	2
Kangaroo Hills West	ungrazed	0	0	6.66	0	13.33	0.33	17.33
	grazed	0	0	11.33	0	16.33	0.33	9
	exclosure	0	0	14	0	19	0	16
Bluerange	ungrazed	0	0	0.33	1.33	12	5.66	5.66
	grazed	0	0.66	1.66	5	6.66	4.33	4.66
	exclosure	0	5	5	2	13	7	2
Leyshon View	ungrazed	0	8.33	11	2.66	3.33	22	0
	grazed	0	26.33	0	3	0.66	3.66	0
	exclosure	0	1	28	0	5	16	0
Kirk River	ungrazed	0	28.66	0	0	2.33	4.66	0
	grazed	0	29.33	0	0	0	0.66	0
	exclosure	0	22	3	1	4	8	0
Mt Leyshon	ungrazed	25	4.66	6.33	0.33	0.33	0	0.66
	Int.grazed	18	16	0.66	1	0.66	0.33	0
	Int.grazed	13.33	16.66	2.66	1	0.33	0.66	0
	grazed	17.66	20.33	0.66	0.33	0.33	0.66	0
Pajingo	ungrazed	0	0	0	4	0	14.66	0
	grazed	0	0	0.33	17.66	0	8.33	0
Jervoise	ungrazed	0	1.66	0.33	0	1.33	2	19.66
	grazed	0	25.66	0	0	0.33	1	12
Lucky Downs Dam	ungrazed	0	8	0	4.66	13.33	15.66	1
	grazed	0	3.33	0	5.33	20	7.66	3.33
Rishton	ungrazed	0	25.33	1	0.33	10.33	8.66	0.33
	grazed	0.66	26	1	0	7.66	11.33	0
Epping Forest 1	ungrazed	22.66	0	0	0.33	1.66	11.66	2
	grazed	20.66	0	0	0.33	0	2	0
Epping Forest 2	ungrazed	15	0	0	0.33	0.33	0.66	0
	grazed	21.33	0	0	0.33	0.66	1	0
	exclosure	2	0	0	0	0	0	0
Tabletop	ungrazed	0	0	0	0	25.33	4	7.33
	grazed	0	0	0	0	28.33	4.33	0

APPENDIX E: NUMBER OF QUADRATS CONTAINING INDIVIDUAL SPECIES AND FUNCTIONAL GROUPS IN EACH OF THE STUDY PLOTS.

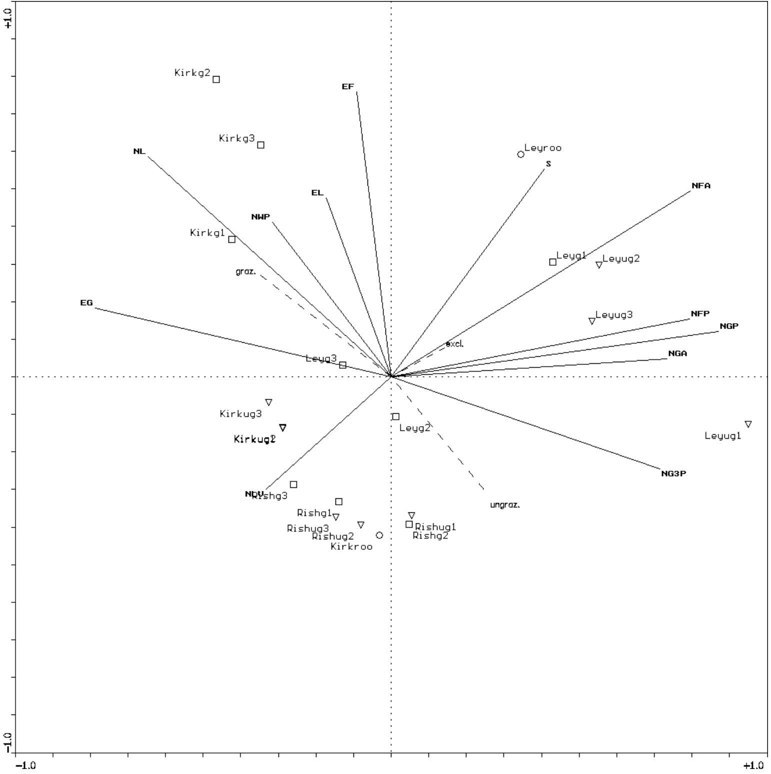
APPENDIX F: PRINCIPAL COMPONENT ANALYSES TRIPLOTS SHOWING THE RELATIONSHIP BETWEEN GRAZING TREATMENT AND ABUNDANCE OF FUNCTIONAL GROUPS

(please refer to attached CD-ROM)

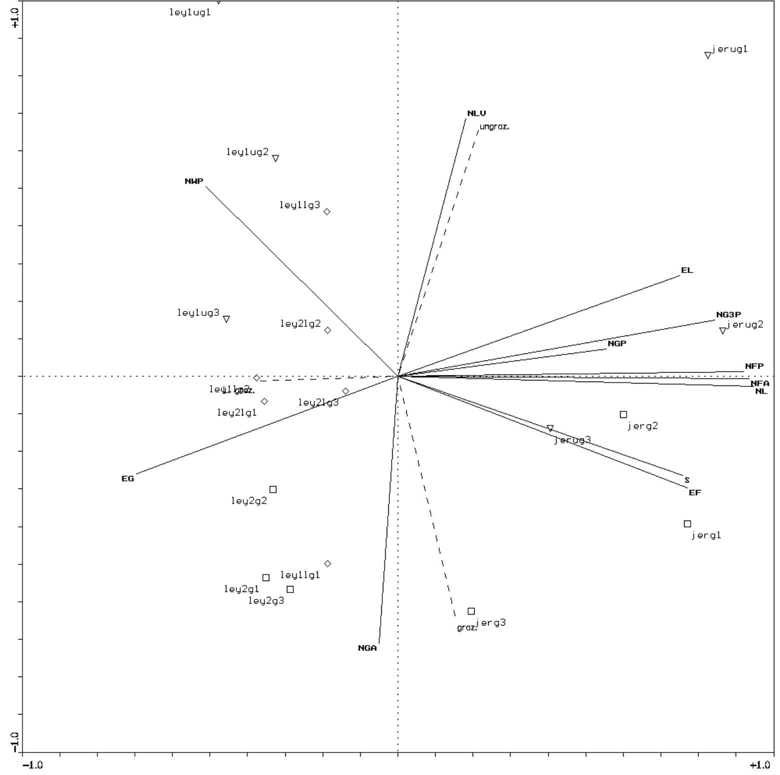
Appendix F (a) Principal Component Analyses (PCA) triplot of sites on Greenvale Chromosols, clustered by presence/absence of functional groups (PCA of Axes I & II).
(= grazed, ∇ = ungrazed, o = macropod exclosure)



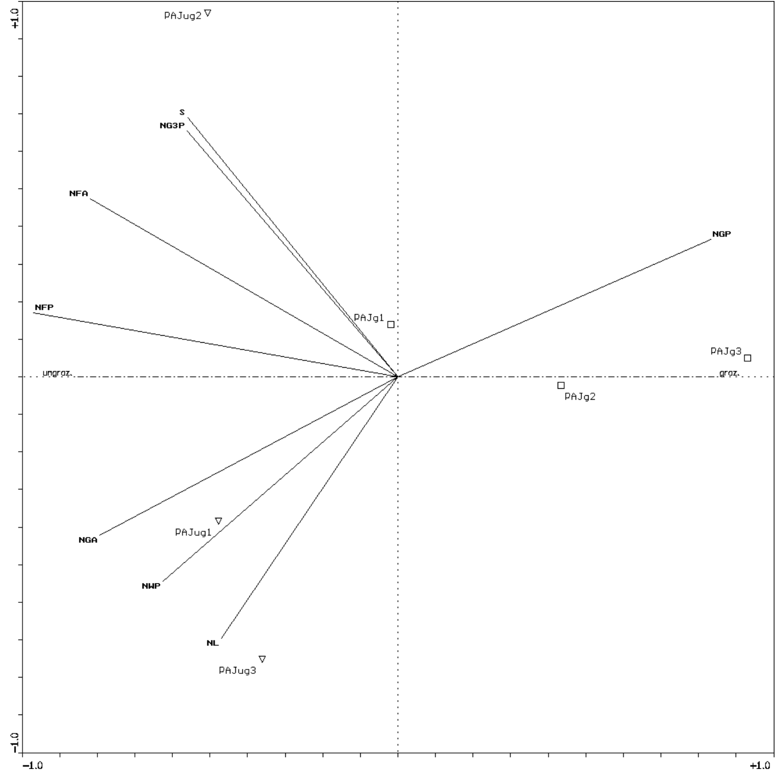
Appendix F (b): Principal Component Analyses (PCA) triplot of sites on Dalrymple Chromosols, clustered by presence/absence of functional groups (PCA of Axes I & II).
(= grazed, ∇ = ungrazed, o = macropod exclosure)



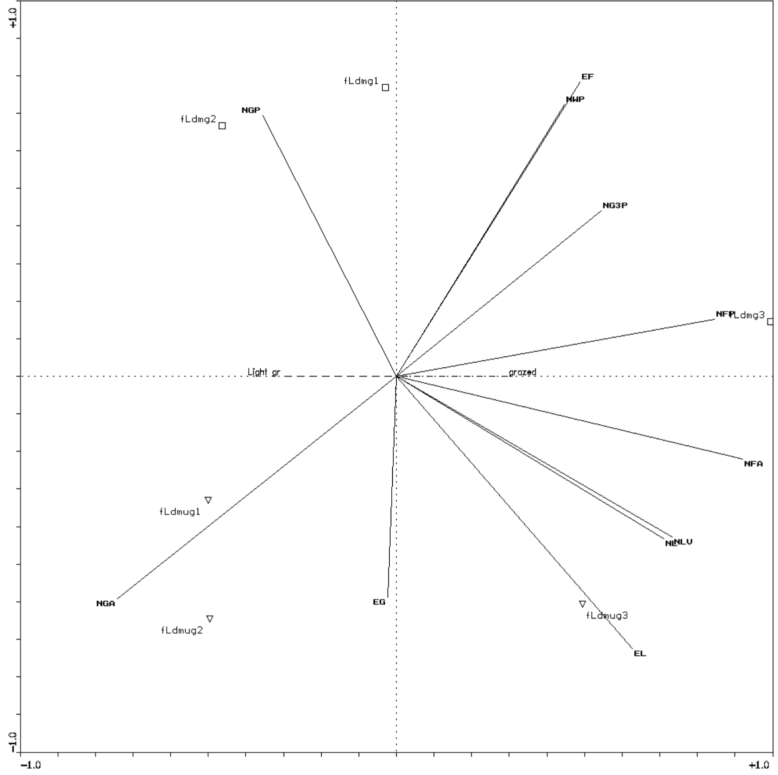
Appendix F (c): Principal Component Analyses (PCA) triplot of sites on Rangeview Chromosols, clustered by presence/absence of functional groups (PCA of Axes I & II).
(= grazed, O= intermittently grazed, ∇ = ungrazed)



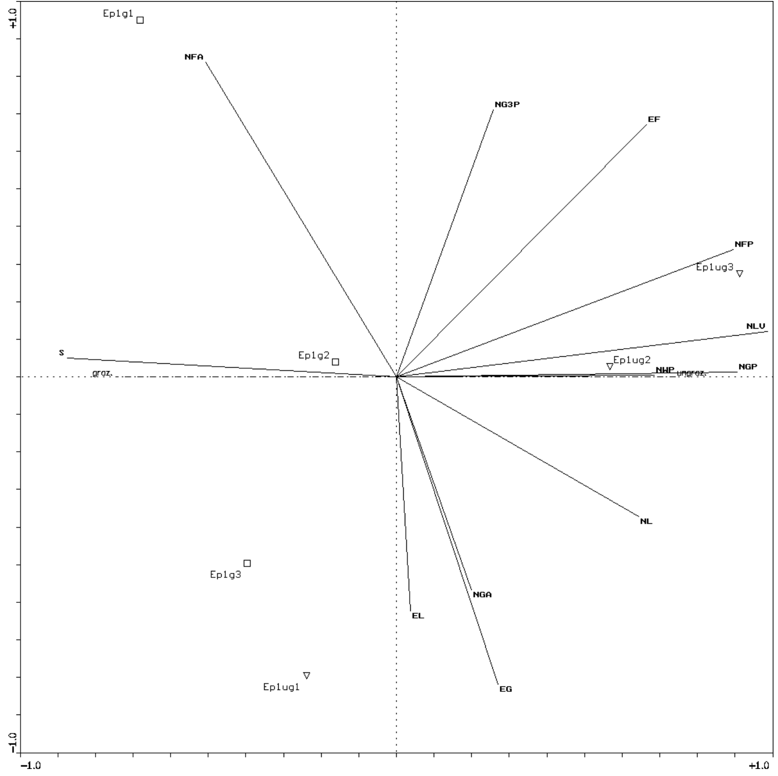
Appendix F (d): Principal Component Analyses (PCA) triplot of sites on Pentland Kandosols, clustered by presence/absence of functional groups (PCA of Axes I & II).
(= grazed, ∇ = ungrazed)



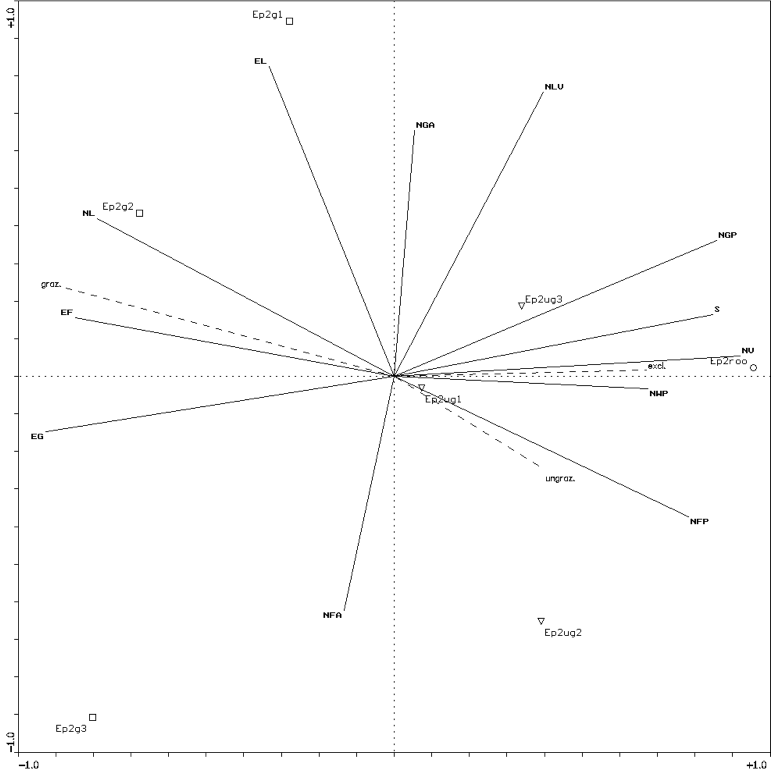
Appendix F (e): Principal Component Analyses (PCA) triplot of sites on Wairuna Dermosols, clustered by presence/absence of functional groups (PCA of Axes I & II).
(= grazed, ∇ = ungrazed)



Appendix F (f): Principal Component Analyses (PCA) triplot of sites on Epping Chromosols, clustered by presence/absence of functional groups (PCA of Axes I & II).
(= grazed, ∇ = ungrazed)



Appendix F (g): Principal Component Analyses (PCA) triplot of sites on Walthum Tenosols, clustered by presence/absence of functional groups (PCA of Axes I & II). (= grazed, ∇ = ungrazed, o = macropod exclosure)



Appendix F (h): Principal Component Analyses (PCA) triplot of sites on Bluff Brown Sodosols, clustered by presence/absence of functional groups (PCA of Axes I & II). (= grazed, ∇ = ungrazed)