CHAPTER ONE

Introduction and literature review

1.1 Introduction

Grasslands are among the most threatened vegetation types in Australia (Kirkpatrick 1993; Benson 1996), and their status is mirrored by natural grasslands in many regions of the world (Henwood 1998; White *et al.* 2000). Human-induced modifications to native grasslands involve cultivation, urbanisation, desertification, fire, livestock grazing, fragmentation and introduction of invasive species (White *et al.* 2000). Cultivation, intensive grazing and the associated fragmentation of the landscape, has led to a considerable loss of native biodiversity in the grasslands of Australia. Agricultural development is thought to have influenced the extinction of 55 plant species, endangered 172 species and constitutes a threat to approximately 110 species of Australian plants (Leigh *et al.* 1984). Hence, the impacts of further agricultural development need to be well understood and closely monitored. Land-uses such as grazing and cultivation are often in conflict with preservation of biodiversity (Hodgson *et al.* 2005) and, hence, many native grasslands are used primarily for purposes other than conservation of plants and animals and consequently often have lower diversity. A challenge facing scientists is to determine whether or not current strategies of land management for agricultural production are compatible with the maintenance or enhancement of native biodiversity.

The Queensland Bluegrass (*Dichanthium sericeum*) dominant grasslands of the Brigalow Belt Bioregions (North and South) have recently been recognised federally as a 'threatened ecological community' (*Environment Protection and Biodiversity Conservation Act 1999*). Mitchell grasses (*Astrebla* spp.) are commonly co-dominant with *D. sericeum* in grassland communities on the Moree Plains (Clarke *et al.* 1998). This community type has a greatly reduced geographic distribution (to approximately 10% of its former range) in the Brigalow Belt, and is under continued threat from ongoing conversion of native grasslands to crops (Anon. 2001). Other threats include over-grazing by livestock and weed invasion. These grassland communities are of considerable importance from a conservation perspective and to the pastoral industries of Australia. Mitchell grasslands provide high quality and a more reliable stock feed than other native grasslands in western New South Wales (NSW) (Orr 1975).

The native vegetation of the Moree Plains has been largely removed or modified because the soils are fertile and valuable for cultivation. The extent of removal and modification threatens the viability of both natural and agricultural systems (Hunter & Earl 1999). It is worsened by the fragmentation of remnants, which forces grassland-dependent species to survive in small populations in sub-optimal habitat (Osborne *et al.* 1993; McIntyre & Hobbs 1999). The remaining grasslands on the Moree Plains are also very poorly reserved and protected. Benson (1989) classified Mitchell grasslands as inadequately reserved in NSW. In fact, the *Dichanthium sericeum* and *Astrebla* spp. tussock grassland association is not known to be conserved in any reserve in NSW other than Kirramingly Nature Reserve on the Moree Plains (Clarke *et al.* 1998). Without active management native grassy ecosystems will continue to be degraded and disappear, but the management requirements for maintaining native diversity are still largely unclear in some regions. Ecological research is needed to determine the best management strategies for conservation of biodiversity.

Grassland research in Australia has concentrated on temperate grasslands and grassy woodlands, such as those occurring on the tablelands of NSW (e.g. Whalley *et al.* 1978; McIntyre *et al.* 1993; Trémont & Whalley 1995; McIntyre & Lavorel 1994b; Clarke 2003) and on the basalt-derived soils of Victoria (e.g. Stuwe & Parsons 1977; Lunt 1990a; Lunt & Morgan 1999; Morgan 1999). A substantial body of research has also been conducted in subtropical woodlands and grasslands of Queensland (e.g. Fensham 1998; Fensham *et al.* 1999; McIntyre & Lavorel 2001; McIntyre & Martin 2001) and on the Mitchell grasslands in semi-arid regions of Queensland (e.g. Orr 1980a,b; Orr 1981; Scanlan 1983; Orr & Holmes 1984; Roe & Davies 1985; Orr & Evenson 1991a,b). Unfortunately the majority of research carried out in Mitchell grasslands has focussed around their pastoral value, rather than their species composition and conservation value. Little research has been conducted on the native grassland communities of the Moree Plains, although recent surveys by Clarke *et al.* (1998) and Hunter and Earl (1999) have provided much-needed information on the grassland communities that occur in the region.

In the last decades there has been a considerable increase in the awareness of the need for appropriate management to conserve native grassland diversity (e.g. Lunt 1991, 1997; McIntyre 1994; Prober & Thiele 1995; Nadolny 1998; Eddy 2002; Davidson *et al.* 2005;

Kirkpatrick *et al.* 2005). Two important management issues in grasslands involve the use of grazing or fire as a disturbance. While we know the responses of some of the more common grasses to grazing and fire, very little is known about other species. Hence most of our knowledge about plant species management is based around species useful for grazing (e.g. work on *Astrebla* spp. by Orr 1975-2000). Additionally, the compositional changes brought about by other disturbances are largely unknown, especially for the Moree Plains. Therefore, there is a need to evaluate management techniques, including the use of fire and grazing as management tools, to help maintain biodiversity by encouraging recruitment and persistence of a full complement of species.

Clarke *et al.* (1998) made several research recommendations to assist the management of Kirramingly Nature Reserve and the native grasslands on the Moree Plains. Those that are relevant to this research include determining: (1) the long-term effects of livestock grazing removal; (2) the interactive effects of fire, rainfall and grazing; (3) the response of individual plants to fire (i.e. survival, germination cues and recruitment); (4) the effects of fire and other disturbances on endangered species; (5) the effects of fire on the recruitment of introduced species and Mimosa Bush (*Acacia farnesiana*); and (6) the changes in floristic composition of cultivated paddocks relative to uncultivated areas. Research into the recovery of native grasslands after cultivation was also recommended after preliminary research in the Walgett region (C. Nadolny, pers. comm., 2001). In addition, Hunter and Earl (1999) suggested further sampling on the Moree Plains over greater temporal periods and manipulative experiments to provide a better understanding of the dynamics of these grasslands. I have attempted to address these ideas in this research.

The current direction of grassland research and management is broadly to maintain areas that are productive and restore areas that have become degraded, using ecological principles (Michalk & Kemp 1994). Maintaining diverse native grasslands may be important for both nature conservation and agriculture, as research suggests that increasing diversity of grasslands can improve primary production (Tilman *et al.* 1996; Hector *et al.* 1999). While some predict that the goals of achieving high economic yields and high biodiversity are compatible (Tilman 1999; Tilman *et al.* 2001), others suggest they are not (Vitousek *et al.* 1997; Tilman *et al.* 2002). Current management of native grasslands may need to be modified in some cases so that diversity can be maintained. Given the spatial and temporal variability and dynamic nature of grassland communities it is unlikely that there will be one set management regime to conserve biodiversity. Management solutions may need to involve combinations of techniques, such as integrating burning and mowing or seasonal grazing (Lunt 1993).

This chapter will provide the aims and objectives of this thesis and a review of the literature pertinent to these aims and objectives. A general outline of the thesis structure is also provided.

1.2 Literature review

1.2.1 Terminology

Grassland can be defined as vegetation dominated by herbaceous species (grasses and forbs), containing <10% tree or shrub cover (Benson 1996). In this thesis 'native grassland' is where >50% of the vegetative ground cover is composed of native species, over half of the total number of species are native, and where there is at least 10% standing vegetation cover (Benson 1996). Native grassland may also occur as a ground cover in woodlands where tree canopy cover is >10%, in which case they may be referred to as grassy woodlands. Where native grasslands are primarily used for livestock grazing they may be referred to as native pastures. For consistency I will refer to native pastures as native grasslands throughout this thesis. Native grasslands vary considerably in their condition. Some contain a high number of native species with very few introduced species, and probably resemble their original state, while others may be dominated by only one species of native grass and may contain a high number of introduced weeds. Introduced (or exotic) species are denoted by an asterisk in front of the scientific name (e.g. **Rapistrum rugosum*) throughout this thesis. Grassland remnants of adequate size, with low weed abundance, relatively high native species richness, intact structural layers (e.g. a mixture of grasses, forbs, sub-shrubs and shrubs) and occurrence of rare or threatened species, or regionally significant species, may be considered high conservation value (Benson 1996; Benson *et al.* 1997; Davidson *et al.* 2005).

1.2.2 Disturbance and succession

Disturbance is a principal factor influencing species richness and composition in plant communities (Collins & Barber 1985; Collins *et al.* 1995; Mackey & Currie 2001). Disturbance can be defined as 'a force, often abrupt and unpredictable, with a duration shorter than the time between disturbance events, that kills or badly damages organisms and alters the availability of resources' (Mackey & Currie 2000). Grassland diversity is maintained by disturbances that assure heterogeneous mixtures of invaders, opportunistic pioneers, annuals, short-lived perennials and long-lived perennials (Vogl 1974). According to the intermediate disturbance hypothesis (Connell 1978), species richness immediately after disturbance is low, but increases with time until a maximal level is reached, and then decreases with time thereafter. Species richness at an intermediate time after disturbance is predicted to be greater due to the mixture of early and late successional species. According to this hypothesis species richness should be higher at sites with a history of intermediate disturbance, and lower at sites that are not disturbed or very frequently disturbed (Connell 1978; Collins *et al.* 1995).

Classical concepts of ecological succession (e.g. Clements 1916) assume that following a disturbance the community gradually resumes the structure and composition of the surrounding undisturbed area. Clements (1916) recognised successional pathways of pioneer, through subclimax to climax vegetation and the reversal of the pathways in response to disturbance (e.g. over-grazing), and such pathways are common in ecological texts (e.g. Begon *et al.* 1996). However, there are numerous problems with Clements' succession theory (Connell & Slatyer 1977; Burrows 1990). Vogl (1974) suggested that grasslands are maintained as vegetative cycles driven by fires (or their ecological equivalent) and, hence, succession may be considered a cyclic or circular phenomenon, rather than the traditional unidirectional approach (i.e. Clements 1916). More complex successional pathways have been recognised in which the system may cross into one or more stable or unstable states, depending on the type and intensity of disturbance and the state of the vegetation at the time (Walker & Noy-Meir 1982; Westoby *et al.* 1989; Tainton *et al.* 1996). The state and transition approach may be more useful in explaining responses in arid and semi-arid ecosystems where rainfall is less dependable (Westoby *et al.* 1989). A major advantage of the state and transition model is that it summarises substantive changes and lesser variation in vegetation states over space and time (Westoby *et al.* 1989) and may lead to a more pro-active approach to vegetation management (Whalley 1994).

Human-induced disturbances have significant effects on species richness (McIntyre & Lavorel 1994a). Disturbances increase spatial heterogeneity in communities by creating patches that develop assemblages of species that differ from the rest of the community (Belsky 1986b). Thus coexistence of species is promoted in patchy environments where species have life-history differences and variable responses to environmental fluctuations (Shmida & Ellner 1984). Different types of disturbance influence species in different ways. For example, native species may decline in response to soil disturbance, but some introduced species may benefit from it (McIntyre & Martin 2001). On the other hand, fire is unlikely to reduce native species richness (Morgan 1996, 1999). Some form of disturbance is considered important in grasslands to remove dense swards of vegetation and leaf litter, allowing light to penetrate to the soil surface and permitting photosynthesis and growth of emerging plants (Sydes & Grime 1981; Morgan 1998). Undisturbed grasslands can become less productive and may contain a lower number of species (Pandey & Singh 1992; Noy-Meir 1995; Blair 1997). Open conditions (i.e. gaps between grass tussocks) in grasslands are sustained through a combination of disturbances (natural and human-induced) such as fire, drought and grazing (Vogl 1974; Bazzaz & Parrish 1982; Dyer *et al.* 1982).

Trémont & McIntyre (1994) suggested native grasslands consist of a perennial matrix (composed mostly of dominant grasses) amongst which perennial and annual forbs and other grasses occur as interstitial components. The balance between matrix and interstitial species is affected by disturbances that influence plant size, litter accumulation, soil moisture and the amount of bare ground. McIntyre and Lavorel (1994b) defined three groups of species based on their response to disturbance: intolerant species, tolerant species and disturbance specialists. They used these groups in a model to show changes in community structure with increasing disturbance intensity. When anthropogenic disturbances are infrequent (i.e. under a natural disturbance regime) the native grassland consists of a matrix of dominant perennial grasses that coexist with smaller-statured interstitial species that form most of the species richness at the site. With increasing disturbance intensity, the tolerant species will persist and the intolerant species will decline, with a subsequent increase in the occurrence of disturbance specialists. At very high levels of disturbance, the matrix of native perennial grasses is affected. With the decline in matrix species, introduced species and disturbance specialists become dominant and only a few of the tolerant native species persist (McIntyre & Lavorel 1994b).

1.2.3 Fire and grasslands

General effects of fire on grasslands

It is not known whether Australian grasslands developed as a result of Aboriginal use of fire, or whether they formed earlier, possibly during arid climatic conditions (Nicholson 1981). However, there is evidence to suggest that, at least in some regions, Australian grasslands were maintained by fire (Nicholson 1981). It is not surprising therefore, that native grasslands are able to cope with fire. Many plant species can survive fire by sprouting from unburnt storage organs or by immediate and abundant germination of seeds after fire (Burrows 1990). However, the effects of fire on plant community composition are varied and dependent on several factors, such as fire intensity, fire frequency, fire season, soil fertility and the sensitivity of individual plants and populations to fire (Vogl 1974; Wright 1974; Gill 1981; Lunt 1990a, 1991; Morgan 1996; Peláez *et al.* 2003; Uys *et al.* 2004). In addition, the effects of fire on plants may be influenced by phenological stage, climatic conditions, grazing intensity post-fire and the strategies of persistence of individual species in response to fire (Wright & Klemmedson 1965; Scanlan 1980; Mulham 1985; Pfeiffer & Steuter 1994; Henderson & Keith 2002). Given the number of factors that potentially influence grassland response to fire, it is difficult to generalise. The time of burning and the frequency of fire can have very different effects on species composition in different grassland types (Vogl 1974).

Fire reduces the density of grass tussocks allowing regeneration of forbs, sub-shrubs and less dominant grasses (Vogl 1974; Lunt 1990a, 1994; Morgan 1998). Hence fire often results in an increase in the abundance or richness of native plants in Australia (Stuwe & Parsons 1977; Hester & Hobbs 1992; Morgan & Lunt 1999; Wright 2001) and elsewhere (Vogl 1965, 1974; Daubenmire 1968). While species richness may increase, plant density is sometimes reduced (e.g. Mulham 1985; Snyman 2004) and there are cases where productivity declines as a result of fire (Launchbaugh 1964; Sharrow & Wright 1977). Given that many grassland communities probably evolved with fire and have become adapted to fire, adverse effects of fire might be considered surprising. Vogl (1974) pointed out that studies reporting negative effects of fire on plants have usually been conducted in arid or semi-arid areas, or on marginal sites, often degraded to some extent by grazing or other means. When plants are subject to additional stresses (e.g. drought or grazing) even fire-resilient species may decline in abundance. In some cases, the density of certain species may be variously reduced or increased (Snyman 2004). Only in rare situations (e.g. Backlund 1956) does fire have virtually no impact on the composition of vegetation, possibly due to a long history of natural selection with repeated burning (Daubenmire 1968). A range of investigations of the effects of fire on grasslands report conflicting results (see reviews by Daubenmire 1968 and Vogl 1974). This is perhaps to be expected given the variable nature of fire and the environmental conditions in which they occur.

Fires create pioneer conditions that may allow invasion of opportunistic species. The invasion of remnant native vegetation by introduced plants is a serious problem (Groves & Burdon 1986; Hester & Hobbs 1992). The interaction of fire and the recruitment of introduced species and weeds requires investigation on the Moree Plains (Clarke *et al.* 1998), as fire could potentially result in mass recruitment of undesirable species under certain conditions. For example, a post-fire study by Lunt (1990a) found that introduced species benefited from a fire in Kangaroo Grass (*Themeda australis*) grassland in Victoria. Further, Lunt and Morgan (1999) reported that intermittent burning resulted in a composition that was not typical of species-rich temperate grasslands. Milberg and Lamont (1995) showed that introduced species increased at the expense of native species after a burn in south-western Australia. In central Queensland, Butler and Fairfax (2003) reported that two introduced species, Buffel Grass (**Cenchrus ciliaris*) and Parthenium Weed (**Parthenium hysterophorus*), increased in cover in burnt areas compared to unburnt areas. However, the susceptibility of plant communities to weed invasion after fire varies greatly (Hester & Hobbs 1992). By managing fire regime, it may even be possible to eliminate certain weeds or introduced species from the community.

Despite its potential importance, the use of fire as a management tool for manipulation of grassland species composition is not well developed in Australia. The impacts of prescribed burns may be important for the long-term management of conservation reserves like Kirramingly Nature Reserve on the Moree Plains. In grassland reserves that are no longer grazed by livestock, some form of disturbance may be required to maintain species diversity, and the use of fire is a potential option. Many fragments of native grassland, like Kirramingly Nature Reserve, have not been burnt since clearance of the surrounding land and the long-term effects of this are unknown (Hester & Hobbs 1992). In addition, the consequences of re-establishing a fire regime to such areas, after a history of grazing, are unknown for the Moree Plains. Given these knowledge gaps, land managers (e.g. pastoralists and conservation managers) are hesitant in their use of fire as a management tool. At present, the costs of burning to pastoralists, both in terms of spelling required to allow fuel levels to build up, as well as possible short term losses in animal production while grasses recover, remain unclear (Tropical Savannas CRC 2001). Burning regime (i.e. the frequency, intensity and season of fires) needs to be carefully considered if fire is to be used as an effective management tool.

Season and frequency of burning

While the effects of a single fire are easily observed, it is the more subtle effect of the fire regime that is more likely to impact on biodiversity (Watson 2001). Fire season, frequency and intensity can affect species in different ways, with particular intensities, seasons and frequencies favouring some species but not others (Auld & O'Connell 1991).

In Australia fire season varies with latitude and distance from the coast (Walker 1981). In the north of Australia the fire season begins at the end of summer while in southern Australia the fire season begins near the start of summer (Walker 1981). Based on the onset and duration of moisture stress (i.e. the bio-climatic regions of Walker 1981), the fire season for the Moree Plains extends from autumn to the end of spring. Unfortunately, the timing of Aboriginal burning in the region is largely unknown. However, a detailed review of explorers' records from Queensland suggests that burning may have occurred during autumn, winter and spring in *Astrebla* and *Dichanthium* communities (Fensham 1997). Prescribed fires in recent times are usually carried out early in the dry season when the fire intensity is low (Hodgkinson *et al.* 1984).

Burning usually stimulates immediate growth of herbaceous species and their subsequent flowering and seeding (Old 1969; Lloyd 1972; Vogl 1974; Lunt 1994; Morgan 1996). Further, the vegetative reproduction of perennial species on most post-burn sites is more rapid and vigorous than growth in unburnt areas (Vogl 1974). Therefore, burning prior to the growing season of most species should allow a rapid recovery of those species, but may be complicated where different species within a community grow and flower in different seasons. For example, burning during spring in southern temperate grasslands may be effective for decreasing the abundance of introduced, annual grasses (Robin 1991), but autumn burning should favour some native flora because it may promote their flowering (Lunt 1991, 1994; Pyrke 1993). Morgan's (1996) study in Victoria reported that late-spring burning is unlikely to harm grassland composition. However, Lunt (1990a) found that while fire in autumn had no impact on species richness, densities of certain plants did increase, including a number of introduced species.

Sometimes fires favour forbs over grasses, at least in the first two growing seasons after fire (e.g. Bock & Bock 1978; Lunt 1990a), but often the reverse is true (Vogl 1974). Uys *et al.* (2004) found that the dominant grasses were strongly influenced by season and frequency of fire, while forb diversity showed no consistent trends in southern African grasslands. In south-eastern Queensland, fire season can have a major impact on the dominant grasses (Walker & Tothill 1990). If fires occur every 1-3 years in the early dry season, *Themeda australis* dominates because the seeds of Bunch Speargrass (*Heteropogon contortus*) are killed by fire. However, if fires occur every 3 years late in the dry season, *Heteropogon* dominates because burning reduces *Themeda* biomass and *Heteropogon* has a greater ability to recruit from buried seed at this time.

Changes in fire frequency can lead to changes in vegetation structure and composition, which in turn can modify intensity of subsequent fires (Hudak *et al.* 2004). Conditions of high fire frequency can potentially eliminate certain plant species if their life-cycle processes are interrupted (Noble & Slatyer 1980; Henderson & Keith 2002). However, long-term exclusion of fire may also result in the loss of certain species from a community (Stuwe & Parsons 1977; Morgan 1999). For species that require bare ground for germination, seeds may die before a recruitment event occurs, and this could eventually lead to local extinction of the species. Repeated burning in native grasslands generally does not reduce the number of species present and may even increase diversity by promoting growth of additional grasses and forbs, including annual plants (Vogl 1974). However, there is evidence that repeated fires generally promote grasses at the expense of woody species (Kayll 1974; Vogl 1974; Hodgkinson *et al.* 1984).

Fire frequency varies greatly across Australia depending on litter accumulation (Walker 1981). The time it takes the herbaceous biomass and litter at a burnt site to return to pre-burn conditions varies between different grassland communities. Grasslands in northern Australia may be burnt almost annually, while those in semi-arid to arid regions generally support infrequent fires (Walker 1981). In *Themeda*-dominated grasslands in Victoria, Morgan (1998) found that gaps suitable for seedling recruitment had disappeared by the third year after fire. Hence, in these grasslands, a fire interval of 1-3 years is recommended (Morgan 1998, 1999). In savanna woodlands in the Northern Territory a fire frequency of 3-5 years may be more appropriate (Watson 2001), and a 3-6 year frequency may be appropriate for tall grass prairie (Hadley & Kieckhefer 1963; Vogl 1965). Partridge (1999) suggested a 4-6 year frequency for spinifex grasslands and a 5-6 year frequency for perennial short-grass grasslands. Fensham (1997) suggested that pre-European fire frequency was relatively low in the Mitchell grasslands of Queensland when compared to other vegetation types. However, I am unaware of any studies on the influence of fire frequency on Mitchell grasslands, or of any recommendations of appropriate fire frequency for these grasslands. This is because fire is generally not promoted in Mitchell grasslands because of their forage value and unreliable rainfall (Orr & Holmes 1984; Partridge 1996).

Fire and grazing interactions

There have been few studies on the impacts of post-fire grazing on vegetation recovery (Meers & Adams 2003), and in many cases the influence of grazing has been ignored (Vogl 1974). However, the interaction of fire and grazing can have a profound effect on vegetation health and composition (Bryant 1973; Kayll 1974; Leigh & Holgate 1979; Pfeiffer & Steuter 1994; Noy-Meir 1995) and may have long-term effects on ecosystem structure (Westoby *et al.* 1989). In fact, the presence of livestock or native herbivores before and, particularly, after burning can completely alter the vegetation response to fire (Vogl 1974). Kirkpatrick *et al.* (2005) reported significant interactive effects of fire and sheep grazing on vegetation attributes in Tasmania. In their study the combination of grazing and burning did not always have a negative impact on species richness. Some research suggests that certain grassland communities (e.g. tall grass prairie) may be more resilient to the interaction between fire and herbivory than earlier believed (Engle & Bidwell 2001), and that grazing after fire may be useful for generating heterogeneity in the landscape (Fuhlendorf & Engle 2004). Hence, if fire is to be used to promote the recruitment of species then the effects of post-fire grazing must also be monitored and managed.

Evidence suggests that herbivores preferentially graze areas regenerating after fire (e.g. Biondini *et al.* 1989; Letnic 2004). Post-burn plants are more palatable and apparently more nutritious (Vogl 1974; Snyman 2004), with higher protein and mineral content and, hence, are preferred by herbivores (Daubenmire 1968). Where animals have unrestricted access to new growth after a burn they tend to keep it grazed to ground level, which severely taxes root reserves and hinders recovery (Daubenmire 1968). This may be more obvious where small areas are burnt (e.g. small prescribed burns), given the potentially higher grazing pressure post-fire (Gill & Bradstock 1995; Meers & Adams 2003).

Fire and invasion of woody species

The establishment of large woody species in grass-dominated vegetation represents a significant change in structure, and is likely to cause substantial changes in the species composition of both fauna and flora. Trends towards increasing woody plant abundance in grasslands and savannas in recent times have been reported worldwide (Grover & Musick 1990; Brown & Archer 1999) and in the semi-arid grazing lands of Australia (Booth & Barker 1981; Hodgkinson *et al.* 1984; Orr & Holmes 1984; Booth 1986). Changes in fire regime (i.e. fire suppression) and over-grazing by domestic livestock are thought to be the two main factors responsible. Repeated fire has been used in the management of certain woody species, particularly in African and American grasslands and savannas, as fire often favours herbaceous species over shrubs (Daubenmire 1968; Kayll 1974; Vogl 1974; Partridge 1996; McPherson 1997; Uys *et al.* 2004). The interaction between fire and grazing is also important (Kayll 1974). In Africa, integrated rotational grazing and burning management is suggested by West (1971, cited in Kayll 1974). In some environments, however, frequent fire in the absence of grazing can increase the abundance of woody plants (Kirkpatrick *et al.* 2005). Clarke and Davison (2001) reported that fire was important for breaking the dormancy of many shrub species in temperate grassy woodlands, and Hill and French (2004) showed the importance of fire in promoting regeneration of shrubs in Cumberland Plain Woodland.

The invasion of woody vegetation in grasslands is a serious problem because herbaceous productivity is generally reduced under tree canopies (especially non-leguminous trees) (Beale 1973; Walker *et al.* 1986; Scanlan & Burrows 1990). Trees or shrubs create sub-habitats, which differ from open grassland and exert different influences on the herbaceous layer, often resulting in altered species composition (Archer *et al.* 1988; Belsky *et al.* 1989; Scanlan & Burrows 1990; Gibbs *et al.* 1999; Abule *et al.* 2005). *Acacia farnesiana* is one shrub species that may represent a threat to the native grassland communities in north-west NSW (Campbell 1989; Clarke *et al.* 1998). Harden (2002) reported *A. farnesiana* as a naturalised species native to tropical America, but the status of this species as an introduced plant is debatable, as reports suggest it was present in the region before European settlement (Clarke *et al.* 1998; Harden 2002). Anecdotal evidence from landholders in the region suggests that *A. farnesiana* has increased in abundance, especially on the travelling stock routes. However, surveys to date have shown that the presence or absence of *A. farnesiana,* even when it is very prominent at an individual site, does not affect the overall vegetation composition (Clarke *et al.* 1998; Hunter & Earl 1999).

Fire in Mitchell grasslands

Scanlan (1980) reported that burning Mitchell grasslands in Queensland did not increase pasture yield. Similarly, in central-western Queensland fire had little effect on the frequency of *Astrebla* spp. (Purcell & Lee 1970), and Everist (personal observations) observed no effect on botanical composition or pasture vigour some years after fire (Orr 1975). However, the effect of fire on Mitchell grasslands depends on when the burn takes place and the seasonal conditions, especially rainfall before and after the event (Scanlan 1980). For example, if soil moisture is low at the time of burning and after the burn, pasture yield may decline (Scanlan 1980; Mulham 1985). Four years after a burn in far north-western NSW, foliage cover of *Astrebla* spp. remained less than in unburnt areas (Mulham 1985). Even poorer recovery of *Astrebla* spp. occurred after a wildfire, and this was attributed to low summer rainfall (Mulham 1985).

Despite these results there may be benefits in burning Mitchell grasslands for new growth (new tillers) and seed production (Scanlan 1980; Orr & Holmes 1984). The production of new tillers is vital to the maintenance of *Astrebla* tussocks and new green tillers provide high quality forage. Scanlan (1980) suggested that pasture quality could be improved by controlled burning during growing seasons when soil moisture is high. While an occasional fire may have no overall effect on the density of *Astrebla* (Orr 1975), tussock numbers were reported to increase after a single burn, as large tussocks divided into smaller ones (Scanlan 1983). This may be important for maintaining species richness, as in temperate *Themeda* grasslands, by opening up inter-tussock spaces (Stuwe & Parsons 1977). The influence of fire on other species within *Astrebla* grasslands has received little attention and requires further examination.

The impact of fire on Mitchell grasslands from a biodiversity conservation perspective is largely unknown. No studies have investigated the influence of burning on species richness and composition in *Astrebla* grasslands using manipulative experiments. The lack of research is probably due to the economic importance of Mitchell grasslands, which play a major role in the pastoral industry of Australia. A recent post-fire study by Wright (2001) in central-western Queensland discovered that burning increased species richness, due to an increase in the numbers of annual grasses and forbs. It was argued that accumulation of litter and vegetation cover was the main reason for the differences in richness between burnt and unburnt sites. By decreasing cover of the dominant grasses, fire increased species evenness and subsequent diversity (Wright 2001).

1.2.4 Grazing and mowing

Grazing

Studies on the impacts of grazing have often involved exclosures to exclude grazing, manipulation of stocking rates, utilisation of land-use history records, or distances from watering points (Vesk & Westoby 2001). There is substantial evidence (e.g. Collins & Barber 1985; Belsky 1992; Díaz *et al.* 1994; Noy-Meir 1995; Bekker *et al.* 1997; Olff & Ritchie 1998; Bai *et al.* 2001) that plant species diversity and hence the conservation value of undisturbed grasslands will deteriorate. Research indicates that moderately grazed grasslands can have higher species diversity, produce more forage and have less standing dead material than grasslands where grazing is excluded (Everist & Webb 1975; Milchunas *et al.* 1988; Noy-Meir 1995; Trémont & Whalley 1995; Bai *et al.* 2001). There is also evidence that short periods of intensive grazing followed by long periods of rest, enhances species diversity in grassland vegetation (FitzGerald & Lodge 1997; Acocks 1966 cited in Hoffman 2003). Like fire, grazing reduces the competition from dominant tussock grasses that may otherwise exclude inter-tussock species by reducing their survival and reproductive capacity (Lunt 1993; Trémont & McIntyre 1994).

Moisture availability plays an important role in the effects of grazing on individual species and, hence, grassland diversity (Milchunas *et al.* 1988; Olff & Ritchie 1998; Vesk & Westoby 2001). Olff and Ritchie (1998) suggested that increases in diversity due to grazing occur in productive grasslands with non-limiting rainfall and fertile soils. For example, in the temperate grasslands of Europe and Australia, and in the tall grasslands in the Serengeti, some grazing usually results in an increase in diversity (Stuwe & Parsons

1977; Milchunas *et al.* 1988; Belsky 1992; Trémont & Whalley 1995; Van Wieren 1995; Bullock 1996; Murray & Illius 1996). However, increases in diversity due to livestock grazing are not always reported in semi-arid or arid environments (Milchunas *et al.* 1988; Brady *et al.*1989; Hobbs & Huenneke 1992; Olff & Ritchie 1998). In such environments where there is considerable variability in rainfall, grasslands are predicted to show little or no increase in diversity under intermediate grazing by large generalist herbivores, and a reduction in diversity with increasing grazing intensity (Milchunas *et al.* 1988). This is because species are more likely to show a negative response to grazing where environmental conditions are sub-optimal (Olff & Ritchie 1998). For example, Vesk and Westoby (2001) reported that species were more likely to respond positively to grazing at higher rainfall sites and negatively at lower rainfall sites.

Grasslands and populations of wild ungulates have coexisted for millions of years, which highlights the potential sustainability of grassland ecosystems (White *et al.* 2000). However, Fleischner (1994) has argued that total removal of livestock is necessary to restore health of grassland ecosystems. There is a surprising amount of evidence to support this idea. For example, Brady *et al.* (1989) reported that long-term (16 years) grazing exclusion increased species richness of semi-arid grassland, suggesting grazing is not necessary to prevent deterioration. In addition, Lacey and Van Poollen (1981) reported from published literature that annual herbage production is 68 ± 46% higher in plots protected from a moderate level of livestock grazing. In woodlands in south-western Australia, livestock grazing was associated with a decline in native perennial cover and an increase in exotic cover, and had negative effects on several soil properties (Yates *et al.* 2000). Spooner *et al.* (2002) also reported that sites excluded from livestock grazing had significantly greater cover of native perennial grasses and less cover of introduced annual species.

Landsberg *et al.* (1997) reported that in Australia's rangelands, 38% of understorey plants, 15% of overstorey plants and 23% of seed bank plants substantially decreased or were eliminated around watering points. Foran *et al.* (1985) reported that destocking allowed the recovery of biomass and species composition on calcareous red soils in semi-arid north-west Australia, and Cunningham and Milthorpe (1981) reported that in western NSW, several uncommon species, including Barley Mitchell Grass (*Astrebla pectinata*), increased in density in areas excluded from grazing. Several Australian studies also show that native species richness is similar in moderately stocked and ungrazed areas (e.g. Stuwe & Parsons 1977; McIntyre & Lavorel 1994a; Fensham 1998; McIntyre & Martin 2001).

Many studies have shown that grazing can influence vegetation structure and species composition (e.g. Whalley *et al.* 1978; Noy-Meir *et al.* 1989; Belsky 1992; Milchunas & Laurenroth 1993; Garden *et al.* 1996; Kemp *et al.* 2000; McIntyre & Lavorel 2001; Vesk & Westoby 2001). Variation in grazing intensity (i.e. stocking rate), duration and timing, the type of herbivore and the environment, can all have an impact on the composition of grassland vegetation. Interactions with site, seasonal conditions, history and type of disturbance are also important (Illius & Hodgson 1996; Smith *et al.* 2000). While some studies have demonstrated a unimodal species response to grazing intensity (Fensham *et al.* 1999), plant species richness along grazing gradients can trend monotonically in either direction or show no trend at all (e.g. Friedel 1997; Landsberg *et al.* 1997; Fensham 1998; Stohlgren *et al.* 1999; Vesk & Westoby 2001). In a study of grazing experiments in Australia, Vesk and Westoby (2001) found that 41% of 324 species responded inconsistently to grazing. Thus changes in abundance and community structure are often complex and it is not surprising that diversity will not be altered in one consistent direction as a result of grazing (Friedel & James 1995). The inconsistencies in grazing response highlight the importance of site and community-specific research and the need for a mechanistic understanding of plant responses to grazing.

The change in disturbance regime after European settlement and the onset of livestock grazing in Australia probably led to several changes in native grassland community composition, because, unlike fire, grazing disturbance can be highly selective (Whalley *et al.* 1978). Selective grazing can result in vegetation changes, through reducing or eliminating palatable species and allowing unpalatable species to increase in abundance (Whalley *et al.* 1978; Burrows 1990; Hodgkinson 1992; Murray & Illius 1996; Hoffman 2003). However, selective defoliation may also lead to an increase in landscape diversity by encouraging a range of species compositions (McIntyre *et al.* 2003). Differences in selectivity between domestic grazing animal species are related to body size (Murray & Illius 1996; Rook *et al.* 2004). For example, sheep tend to be more selective than cattle, preferring vegetation that is higher in protein and lower in grass (Wilson & Harrington 1984). Perhaps as a result of this selectivity, sheep grazing may have positive or negative effects on different native species. Kirkpatrick *et al.* (2005) reported that several plant species appeared dependent on sheep grazing for their persistence in Tasmanian grasslands, while others were intolerant of sheep grazing but required other types of disturbance.

The type of grazing animal may also influence grassland composition through trampling effects (Snaydon 1981; Sun & Liddle 1993), and trampling associated with grazing can reduce species richness (Ikeda 2003). Differences exist between grazing by native herbivores in Australia and grazing by domestic livestock. Native marsupial grazers have soft feet and probably have a lesser trampling effect than introduced livestock. Livestock can have a negative impact on soil structure and soil processes that influence plant water and nutrient availability (Yates *et al.* 2000). However, the hooves of livestock may be important for breaking up the soil surface and incorporating litter and seeds into the soil (Whalley 2005).

As with the use of fire, the timing and frequency of grazing is important if it is to be used effectively to maintain or promote species diversity. Rotational grazing may be more effective than continuous grazing in enhancing species richness, because the absence of grazing at strategic times (seed set, seedling emergence) may be necessary to allow recruitment and growth of certain species (Campbell 1989; Campbell *et al.* 1996). A study by Roe and Allen (1993) in Mitchell grasslands of Queensland highlights the importance of timing for rotational grazing. These authors reported that spelling from grazing did not result in improvements in pasture or animal production compared to continuous grazing at light (1 sheep to 3 ha) and medium (1 sheep to 2 ha) stocking rates. This was because the rotational system (summer grazing with a winter spell) adversely affected the response of pasture to summer rain and decreased the proportion of *Astrebla* in the pasture. While Roe and Allen (1993) did not measure species richness, it is likely that a decline in *Astrebla* abundance would have promoted higher species richness (Trémont & McIntyre 1994; McIntyre & Lavorel 2001; Wright 2001).

It is well known that frequent disturbances, such as continuous heavy livestock grazing, will produce changes in species composition, a reduction in herbage production and sometimes species diversity (Beadle 1948; Stoddart *et al.* 1975; Lacey & Van Poollen 1981; Milchunas *et al.* 1988; McIntyre & Lavorel 1994a,b). While perennial grasses may be favoured by light grazing, under heavy livestock grazing palatable, perennial grasses will decline and be replaced by annual grasses and forbs (Beadle 1948; Risser *et al.* 1981; Belsky 1986b; Noy-Meir *et al.* 1989; Lunt 1991; Trémont & McIntyre 1994; Briske 1996; McIntyre & Lavorel 2001). Distance from watering points can have a large influence on grazing intensity with a gradient of increasing grazing pressure towards watering points (Lange 1969; Andrew & Lange 1986; Wilson 1990). Increased grazing pressure around watering points may result in a decline in palatable species abundance, dominance of unpalatable species and potential invasion of introduced species (Wilson 1990; Friedel 1997; Friedel *et al.* 2003).

Beadle (1948) reported that under heavy stock grazing in north-western NSW the first plants to disappear were *Astrebla, Dichanthium* and Silky Browntop (*Eulalia aurea*). Neverfail (*Eragrostis setifolia*) and Fairy Grass (*Sporobolus caroli*) then dominated the pasture, and annuals became increasingly common as degeneration proceeded. There was a general trend from native grasses to introduced species (i.e. winter-growing annuals). Prober and Thiele (1995) reported a similar trend of enhanced introduced species richness in their study of temperate White Box woodlands. However, higher grazing intensities do not always result in increased richness of introduced species (McIntyre & Lavorel 1994a; Fensham 1998).

In some situations substantial changes in composition result in the same number of species colonising as become locally extinct in an area and, hence, there are no changes in species richness (Glenn & Collins 1992; Stohlgren *et al.* 1999; Bai *et al.* 2001; McIntyre *et al.* 2003). These extinctions and colonisations occur on a range of scales. One disadvantage of many grazing studies is that they are often carried out on a relatively small scale (e.g. paddock or plot scales). The impact of grazing on biodiversity needs to be integrated over regional scales so the spatial differences in the distribution of the biota, and the variations in management, can be taken into account (Friedel & James 1995). Stohlgren *et al.* (1999) reported that local species richness is poorly correlated with landscape-scale species richness. While grazing may increase species richness at small scales, it is possible that richness decreases at a landscape scale, because of the elimination of grazing-sensitive species (McIntyre *et al.* 2003).

The extent to which grazing by introduced animals has changed the flora of the Moree Plains will never be known, simply because there appear to be no areas that have not been grazed by livestock at some stage (Clarke *et al.* 1998). However, Mitchell grasslands have been labelled the most resilient of the rangeland pastures in Australia (Orr & Holmes 1984). Thus, permanent changes in composition and richness brought about by management (e.g. grazing or prescribed burning) may be unlikely (Campbell 1989), and some grazing is probably beneficial to *Astrebla* spp. populations, especially during wet years, as it encourages regrowth and seed production (Orr 1980b). Everist and Webb (1975) suggested that not grazing *Astrebla* dominated communities might ultimately lead to the development of different plant communities. Nevertheless, domestic stock were not present when explorer Sir Thomas Mitchell discovered these grasslands in 1835. As major recruitment events may be rare (i.e. perhaps every 40 years in south-west Queensland according to Roe (1987)), the continued survival of *Astrebla* populations depends on careful land management.

Mowing

Difficulties in experimentally controlling livestock grazing have persuaded many investigators to evaluate the effect of herbage removal by clipping at different intervals and plant heights (e.g. Neiland & Curtis 1956; Davis 1960; Smith 1960; Pond 1961; Tomlinson 1986; Hodgkinson *et al.* 1989; Orr & Evenson 1991b). These studies were mostly concerned with individual plant measurements, such as effects on biomass (below and above the ground), seed and tiller production. A review by Jameson (1963) outlined several problems with using clipping or mowing to simulate grazing. These include: the lack of selectivity; the lack of trampling effects; the lack of dung and urine input; and the different effects on litter accumulation. However, clipping studies have provided much useful information, especially from an agricultural perspective (Jameson 1963).

Management with light to moderate grazing or repeated mowing has replaced fires in some remnant grasslands (e.g. Lunt 1993; Eddy 2002), since these uses tend to produce many of the same results (Daubenmire 1968; Lunt 1991). Such management has probably helped maintain native diversity in some grassland ecosystems, however, relatively few studies have investigated the effects of mowing on species composition and richness in native grassland communities. Verrier & Kirkpatrick (2005) reported that removal of litter after frequent mowing (4-6 times per year) improved the conservation value of grassland when compared to moderately grazed areas in Tasmania. Frequent mowing increased native species cover, reduced introduced species cover and encouraged rare species. However, mowing does not always encourage native species and mowing frequency may have varied effects on species richness and composition (Risser *et al.* 1981; Kirkpatrick 1986; Zechmeister *et al.* 2003). High mowing frequencies (i.e. every 3 months) may result in a decline in native grasses and encourage introduced species, especially if mowing interferes with reproductive cycles (Lunt 1991). Some evidence suggests that broadleaf weeds and annual bromes (*Bromus* spp.) increase under increased frequency of clipping (Risser *et al.* 1981). If litter is not removed after mowing or slashing, a thick mulch layer can develop and this may prevent establishment and smother low statured plants (Lunt 1993). Mowing may also enhance the spread of introduced species if machinery is not carefully cleaned (Stuwe 1986).

1.2.5 Cultivation

The area dominated by *Astrebla* spp. in NSW has probably been reduced because of over-grazing (Beadle 1948; Bowman *et al.* 1996) and cropping in more recent years (Campbell 1989). There have been few Australian studies on the impacts of cultivation in native grassland communities (i.e. composition and diversity), despite the fact that cultivation is a continuing major threat to these communities. Several factors may influence the recovery of grassland after cultivation, such as the number of cultivations, grazing pressure in the recovery period, and the presence of nearby seed resources (i.e. remnants) (Montalvo *et al.* 1993; Robson 1995; Nadolny & Eldridge 1999; Fullbright 2004; Römermann *et al.* 2005). While it is generally accepted that the conservation value of cultivated areas is not the same as for land that has never been ploughed, there is little published information in Australia on how readily native grasslands re-establish after cropping ceases, or how long it takes for grasslands to return to the unploughed state, if ever.

Römermann *et al.* (2005) reported that regeneration of Mediterranean steppe on formerly cultivated land may take decades or centuries, if at all. Similarly, Graham and Hutchings (1988b) reported that there is little potential for re-establishment of chalk grasslands in Sussex from the seed bank and seed rain. However, response to cultivation is not always negative. Belsky (1986b) reported that small plots ploughed once in Serengeti grasslands returned to their previous compositions 1-3 years after disturbance and Fulbright (2004) reported an increase in plant diversity after certain disc ploughing treatments in semi-arid scrub in Texas. Lavorel *et al.* (1991) reported that species richness of herbaceous vegetation does increase with time since cultivation in abandoned Mediterranean grasslands of France. Austrheim and Olsson (1999) also reported an increase in species richness along a successional gradient with time since cultivation in subalpine grassland in Norway. It is generally accepted that annual plants appear first after cultivation and are replaced by perennials if subsequent conditions are favourable (Risser *et al.* 1981; Camill *et al.* 2004). Booth (1941) suggested that there are four successional stages after cultivation of grasslands in Oklahoma and Kansas: the first stage dominated by annual weeds, the second by annual grasses, the third by perennial bunchgrasses and the fourth by pre-existing dominant species of the tall grass prairies.

McIntyre *et al.* (2002) reported that repeated cultivation severely restricts the density of native grassland species, hence it is not surprising that intensive land-uses, such as cropping and sown pastures, are more of a threat to the conservation status of grasslands than livestock grazing. While these authors recognised a component of the flora that was sensitive to cultivation, they also noted that certain species tolerate or were favoured by this form of disturbance. Similar findings were reported by Scott and Whalley (1982) on the New England Tablelands of NSW. Hence, some native species can persist even in areas of intensive land-uses (Fairfax & Fensham 2000; Rew *et al.* 2005). Soil disturbance favours species that are able to disperse effectively (i.e. via the wind or livestock) and exploit resources rapidly (e.g. therophytes) (McIntyre *et al.* 1995). Unfortunately, this includes many weed species. An increase in weed species after cultivation may be partly due to reduced competition and higher levels of available nutrients (Wijesuriya & Hocking 1998), given that cultivation in many regions is accompanied by the addition of fertilisers. Graham and Hutchings (1988a) reported that seed banks of previously cultivated areas contained many annual weeds and very few seeds of species characteristic of chalk grasslands. There have been some attempts to restore grassland in previously cultivated land (e.g. Kindscher & Tieszen 1998; Sluis 2002; Camill *et al.* 2004). On the North American Great Plains, restoration of grassland vegetation in former cultivated areas has been attemped through replanting perennial grasses in the Conservation Reserve Program (Kindscher & Tieszen 1998; Sluis 2002).

Conversion of native grassland to cropping land leads to deterioration in soil properties (Eldridge 1999; Mikhailova *et al.* 2000; Saviozzi *et al.* 2001), such as loss of organic matter. Burke *et al.* (1995) proposed that in abandoned agicultural fields in Colorado, pools of total soil carbon change slowly and may require up to 100 years or more before returning to levels observed in native grasslands. However, some soil disturbance may be beneficial, especially where grazing intensity is high, and soils become compacted. Before European settlement, it is thought that the activities of rat kangaroos (fossorial marsupials) were important for incorporating litter into the soil, enhancing decomposition rates and rainfall infiltration, nutrient cycling and herbaceous plant growth (Martin 2003). In addition soil disturbance provides open space for colonisation through seedling establishment (McIntyre *et al.* 1995). In western NSW pasture biomass was significantly greater in plots that had been blade ploughed and ungrazed for a 30-month period following the treatment, compared to plots that were not ploughed (Robson 1995). However, ploughing initially killed a significant proportion of the pasture, and it took 14 months (with summer rainfall) before pasture biomass had recovered (Robson 1995).

There are no published results of investigations into the natural regeneration of Mitchell grasslands after cropping ceases on the Moree Plains, despite the fact that it is a major conservation issue in the region (C. Nadolny, pers. comm., 2001). However, Whalley (2000) suggested that grasslands of the Moree Shire have the capacity to regenerate to grasslands with high plant species diversity following cropping, given sufficient rest, favourable seasons and appropriate grazing management. Early results from a study in the Walgett area (west of the Moree Plains) by the Department of Infrastructure, Planning and Natural Resources suggested that a high diversity of perennial grasses may develop in as little as 5 years after cultivation (Nadolny & Eldridge 1999). However, species richness was higher at sites with a longer interval since cultivation and at sites with the lowest concentrations of soil nitrogen. In addition, sites with the greatest number of species tended to be near patches of intact vegetation, suggesting that undisturbed vegetation could be important in providing a seed source for re-colonisation (Nadolny & Eldridge 1999).

The ability of grassland to regenerate probably depends on the number of cultivations. This is because diversity may be hindered after a series of cultivations by a lack of propagules of new species in the seed bank or seed rain (Graham & Hutchings 1988b; Bullock *et al.* 1994; Buisson & Dutoit 2004). Many native grassland species have short-lived seed banks (Lunt 1990b, 1997; Morgan 1998; Clarke & Davison 2004) and, hence, could be completely removed after many years of consecutive cultivations. However, it is not known how many consecutive cultivations it takes to deplete the seed reserves in the soil, to the extent that native grasses can no longer re-establish. Given that seeds of *Astrebla* spp. remain viable for up to 5 years (Bowman 1992), continuous cultivation for a period of time greater than this may result in major changes in species composition. It is likely that after 10 years of cropping, the *Astrebla* spp. and associated plants will have disappeared, so recruitment from surrounding areas would be required (Bellotti *et al.* 1986).

1.2.6 Flooding

Flooding can affect herbaceous vegetation by submerging it for extended periods and potentially covering plants with silt at the same time (Boswell 1979). When the soil is saturated with water during a flood, insufficient oxygen is available to plant roots and this can lead to death of the root system and hence the plant (Boswell 1979). The response of grassland species to flooding and periods of inundation is not well documented in Australia, but flooding may have an important influence on community composition for grasslands occurring on floodplains. As with other forms of disturbance, flooding is likely to influence different species in different ways, hence it can alter plant community composition (Capon 2005). Interactions between flooding and grazing may also be very important. If grazing occurs soon after flooding there is a danger of uprooting of plants, and pugging (where livestock leave hoof indentations in the wet soil) may have a major effect on plant survival (Boswell 1979).

Flooding was reported as having an overriding effect on species richness and cover amongst most plant groups in central arid Australia (Capon 2005). Capon (2005) reported that landscape-scale effects of flood frequency have a stronger influence on plant community composition in frequently flooded areas, while local factors (e.g. soil type) are more important at low flood frequencies. In arid NSW, flooding had a significant influence on species composition and was considered to have a greater influence than fire and grazing (Westbrooke *et al.* 2005). Sporadic flooding events may facilitate the germination of large numbers of native annual and perennial species from the soil seed bank (Westbrooke *et al.* 2005). However, invasion of introduced species (e.g. Tree Tobacco (*Nicotiana glauca*)) may also be enhanced by flooding (Florentine & Westbrooke 2005). Variations in flood frequency, duration and depth create complex spatial gradients across floodplains, increasing spatial heterogeneity (Capon 2005).

Prolonged inundation of certain Mitchell grasses (e.g. Curly Mitchell Grass, *Astrebla lappacea*) can result in death of the tussock (Campbell 1989). A series of major floods in the 1970s (i.e. 1971, 1974, 1976) has been blamed for the decline in *Astrebla* spp. density in NSW (Campbell 1989). A study by Bowman *et al.* (1996) supported this hypothesis. These authors reported that *Astrebla* grasslands are prone to deterioration where inundation by flooding or waterlogging is common, as *Astrebla* spp. density was significantly lower in paddocks that were flooded. However, Campbell (1989) pointed out that only major floods have a direct deleterious effect on Mitchell grasslands and such events may be relatively infrequent.

1.2.7 Temporal variation

Understanding the environmental variability in ecosystems is important before making decisions on how to manage them. Environmental variables (e.g. rainfall) may influence the response of plant species to grazing or other disturbances (Scanlan 1980; Noy-Meir *et al.* 1989; Orr & Evenson 1991a; Stohlgren *et al.* 1999; Vesk & Westoby 2001).

The relationship between annual rainfall and above-ground grassland productivity has been reported as linear (Webb *et al.* 1978), which highlights the important role of rainfall in grassland ecosystems. Given the relatively low rainfall across most of Australia, and its seasonal nature, it is not surprising that soil moisture frequently limits primary production (Fitzpatrick & Nix 1970; Stafford Smith 1996). Several studies report increases in plant biomass and a reduction in bare ground stimulated by rainfall (e.g. Pandey & Singh 1992; Robson 1995; Walker *et al.* 1996; Teague *et al.* 2004). Large-scale trends in climatic variables affect water availability and, hence, are a major driving force in the structure and function of grassy ecosystems (Fowler 1986; Milchunas *et al.* 1988; Olff & Ritchie 1998; Oesterheld *et al.* 1999). However, there have been few detailed descriptions of changes in botanical species composition in native grasslands over different seasons of a year (J. Reseigh, unpublished data, 2004). In Australia, there is also little information on the responses of herbaceous species to drought (Whalley 1973). It is known, however, that native grasses are resistant to drought and that native grasslands are well adapted to the Australian environment. Many species can withstand long, hot, dry summers and droughts, far better than introduced species, by dying down or becoming dormant (Whalley & Davidson 1969; Burrows 1990).

Droughts are known to cause many changes in grasslands (Bazzaz & Parrish 1982). They can result in removal of species from the community, competitive release, thinning of the canopy, invasion of weeds, changes in the demographic structure of populations and shifts in phenological events (Bazzaz & Parrish 1982). To survive drought, many plant species have strategies which involve evading drought or resisting it (Everist 1964; Whalley 1973; Neldner 1991). Such adaptations support the idea that droughts had an important role in the evolution of grassland ecosystems. Drought evaders may be either annual, in which case they persist through a seed bank, or perennial, in which case they survive below the soil surface by means of storage roots or rhizomes and can recover from vegetative regrowth (Everist 1964; Whalley 1973; Neldner 1991). Drought resisters maintain above-ground foliage that is dormant throughout the drought (e.g. *Astrebla* spp.).

Despite adaptations to survive drought, some species are negatively affected (Tilman & El Haddi 1992). Mortality of grasses and forbs during drought has been reported in North American grasslands, and was related to the depth of root systems, with shallow-rooted species disappearing first (Bazzaz & Parrish 1982). On the Central Great Plains of North America, Tomanek and Hulett (1970) reported a decrease in basal cover from 85% before drought to less than 20% during drought. Tilman and El Haddi (1992) reported that above-ground biomass decreased by 47%, with a 37% reduction in species richness, in four different drought-affected grasslands in Minnesota. While biomass recovered with precipitation during the 2 years following the drought there was no significant recovery in species richness due to the local extinction of rare species.

Fluctuations caused by year-to-year, or longer-period, environmental variations are common in grasslands (Burrows 1990). Reseigh (unpublished data, 2004) reported significant fluctuations in native and introduced species richness due to climate over a two-year period. In Reseigh's (unpublished data, 2004) study on the Northern Tablelands of NSW, native species richness generally peaked during autumn while introduced species richness was highest in spring. In the Mitchell grasslands of Queensland, Fensham *et al.* (2000) found that floristic composition was greatly influenced by climatic variables. The main changes that occur in these grasslands, although usually temporary, are brought about by seasonal rainfall and drought (Williams & Roe 1975; Foran *et al.* 1982; Foran & Bastin 1984; Orr & Holmes 1984; Campbell 1989; Orr & Evenson 1991a; Bowman *et al.* 1996). In fact, Orr (1981) suggested that changes in botanical composition in *Astrebla* grasslands are influenced more by trends in seasonal rainfall than by grazing pressure. If this is the case, variability in rainfall may hide the effects of management in the short-term (Bellamy *et al.* 1996).

In Mitchell grasslands the effects of rainfall on botanical composition appear to be related to the competitive ability of perennial grasses, particularly *Astrebla* spp. (Orr 1981). The relative abundance of perennial grasses will be determined by recent rainfall history (1-3 years), while the relative abundance of annual grasses and forbs will be determined by the timing of immediate past rainfall (Orr 1981). Natural recruitment of many species relies on infrequent rainfall events or particularly wet years (Parsons & Brown 2000). The growth of grasses in *Astrebla* grasslands is generally promoted by rainfall in mid summer, while rain falling in early or late summer may encourage the growth of various forbs, and winter rainfall encourages winter-growing forbs (Orr & Holmes 1984). Previous surveys of the Moree Plains grasslands (Clarke *et al.* 1998; Hunter & Earl 1999) also suggested considerable seasonal variation in species composition. In a survey by Hunter and Earl (1999) sampling in winter and early spring increased the capture of species by around 20% compared to sampling in summer-autumn.

1.2.8 Grassland fauna

While the impacts of management on native grassland fauna are not a focus of this thesis, any management impacts that influence vegetation community composition are also likely to influence fauna. Native grasslands provide shelter and food for a wide variety of animals, both native and introduced (e.g. Duffey *et al.* 1974; Baker-Gabb *et al.* 1990; Osborne *et al.* 1993). Unfortunately, the remaining areas of native grassland have become increasingly fragmented and modified and a number of vertebrate species dependent on these habitats have disappeared, suffered extensive range contractions, or have declined within remaining patches (Osborne *et al.* 1993; Dickman 1994). Since European settlement, approximately 50% (27 species) of all native mammals that once inhabited the western plains of NSW are now extinct (Benson 1991; Dickman 1994).

There are few published studies on the fauna of Mitchell grasslands in NSW. However, the *Astrebla* grasslands on cracking clay soils in Queensland and the Northern Territory support a distinctive ground fauna, with a high diversity of elapid snakes, several endemic reptile species, high densities of several grassland birds and small marsupials (Tropical

Savannas CRC 2001). An extensive fauna survey in Mitchell grasslands in the Northern Territory by Fisher (2002) identified 10 vertebrate species and six ant species that were characteristic of these grasslands in the region. This included four birds (Singing Bushlark *Mirafra javanica*, Flock Bronzewing *Phaps histrionica*, Little Button Quail *Turnix velox* and Red-Chested Button Quail *Turnix pyrrhothorax*), three reptiles (*Tympanocryptis lineata, Ctenotus joanae* and *Proablepharus kinghorni*), and two mammals (*Planigale ingrami* and *Sminthopsis macroura*).

1.3 Thesis aims, objectives and structure

Previous studies on the Moree Plains (Clarke *et al.* 1998; Hunter & Earl 1999) and the above literature review identified several knowledge gaps regarding native grassland species composition and diversity in the region. There is a need to determine the effects of:

(1) Fire on the native grasslands of the Moree Plains and long-unburnt remnants like Kirramingly Nature Reserve, and the potential for its use in management of *Astrebla* grassland reserves (e.g. Kirramingly Nature Reserve);
(2) Livestock grazing;
(3) Cultivation;
(4) Flooding; and
(5) Seasonal variability, rainfall variability and drought.

Hence this thesis broadly aims to determine how the native grassland communities on the Moree Plains respond to different types of disturbance (both anthropogenic and natural), with a particular focus on the influences of grazing, cultivation, fire and flooding. This information is required by managers in making decisions about the most appropriate management regime(s) to conserve plant diversity in native grassland communities. The outcomes of this research might also be applicable to other grassy ecosystems with similar species and land practices.

There were two major components to this research, (i) a broad-scale regional vegetation survey collecting information over the Moree Plains, and (ii) experimental work at Kirramingly Nature Reserve, to provide more detailed information for a grassland community that was once typical of the study region. The thesis is structured to provide broad findings from exploratory analysis based on surveys, then more detailed findings to address objectives from experiments. An introduction to the study region and grasslands of the Moree Plains is provided in Chapter 2, followed by the first data chapter to report the findings of a regional vegetation survey. The major objectives of the regional survey (Chapter 3) were to determine:

- Changes in grassland community composition and abundance in different seasons and years.
- The relative importance of disturbance-related (grazing, cultivation, flooding and fire) and environmental variables (e.g. rainfall, soil type) in determining species composition, and the influence of these variables on grassland condition.
- The impacts of grazing, past cultivation and flooding on plant species richness and composition.

Chapter 4 provides further information on the impacts of grazing and cultivation on species richness and composition, based on surveys at Kirramingly Nature Reserve and the surrounding stock routes. As a number of surveys have been carried out over time at the Reserve temporal changes in composition and richness were investigated. Given that sites in this survey were not as widely dispersed as those in the regional survey, there were less confounding influences from environmental variables (e.g. rainfall). The influences of *Acacia farnesiana* abundance, presence of tree canopy and proximity to dams on community composition were also investigated.

Chapters 5 and 6 report findings from experiments at Kirramingly Nature Reserve to investigate the effects of fire, mowing and disc ploughing. The major objectives of the experiments at Kirramingly Nature Reserve were to determine:

- The impact of season of burning (autumn *vs* spring) and repeated burning (two burns *vs* one burn) on vegetation composition, richness and evenness.
- The effect of native herbivore grazing immediately after fire in autumn and spring on species composition, richness and evenness.
- How certain individual plant species (e.g. *Acacia farnesiana, Astrebla* spp.) respond to fire in terms of survival and seed production.
- The influence of mowing at two heights (low cut *vs* high cut) and mowing once or twice per year on species richness, evenness and composition.
- The influence of once-off disc ploughing on species richness, evenness and composition.

Chapter 7 provides a synthesis and general discussion based on findings from all data chapters. This chapter relates the findings of this research to other grassland ecosystems and existing ecological theory and discusses the applicability of this research to other grasslands. It also discusses the management of native grasslands on the Moree Plains and at Kirramingly Nature Reserve for conservation of plant diversity. A model of transitional stages is proposed to aid management for conservation of plant diversity. Avenues for further research are also recommended.

CHAPTER TWO

Study region

2.1 Introduction

The large variation in Australian native grasslands results from differences in climatic conditions, soil type and a range of other environmental factors (Moore 1970). There can be big differences in species composition, even within Mitchell grasslands in different regions, depending on the timing and amount of rainfall received (Bellotti *et al.* 1986). Hence it is important to provide some background on the environment and climatic conditions in which the native grasslands on the Moree Plains occur. This chapter aims to describe the study region so that the research in subsequent chapters of this thesis can be put into context with research elsewhere. More specifically, it aims to: (i) define the location of the study region; (ii) provide a description of the climate; (iii) describe the landform, soils and geology of the study region; (iv) provide a brief description of the Aboriginal and European land-use history; and (v) define the major plant communities within the study region, including some background on the Mitchell grassland communities of the Moree Plains.

2.2 Location

This study was carried out on the Moree Plains, which is situated in north-west NSW, west of the Great Dividing Range (Fig. 2.1). The Moree Shire covers an area of approximately 1 792 000 ha (Peasley & Walsh 2001). The study region extends from 50 km east of the Newell Highway to the Barwon River in the west, Macintyre River in the north and Bellata in the south. Regional survey sites were generally located within 110 km to the north and west of Moree and 50 km south and east of Moree.

Experimental and survey sites were located at Kirramingly Nature Reserve. This Reserve is situated 10 km west of Gurley, which is about 30 km south of Moree on the Newell Highway (Fig. 2.2). The Reserve covers an area of 1296 ha and was purchased by the NSW National Parks and Wildlife Service (NPWS) (now Department of Environment and Conservation) in 1996.

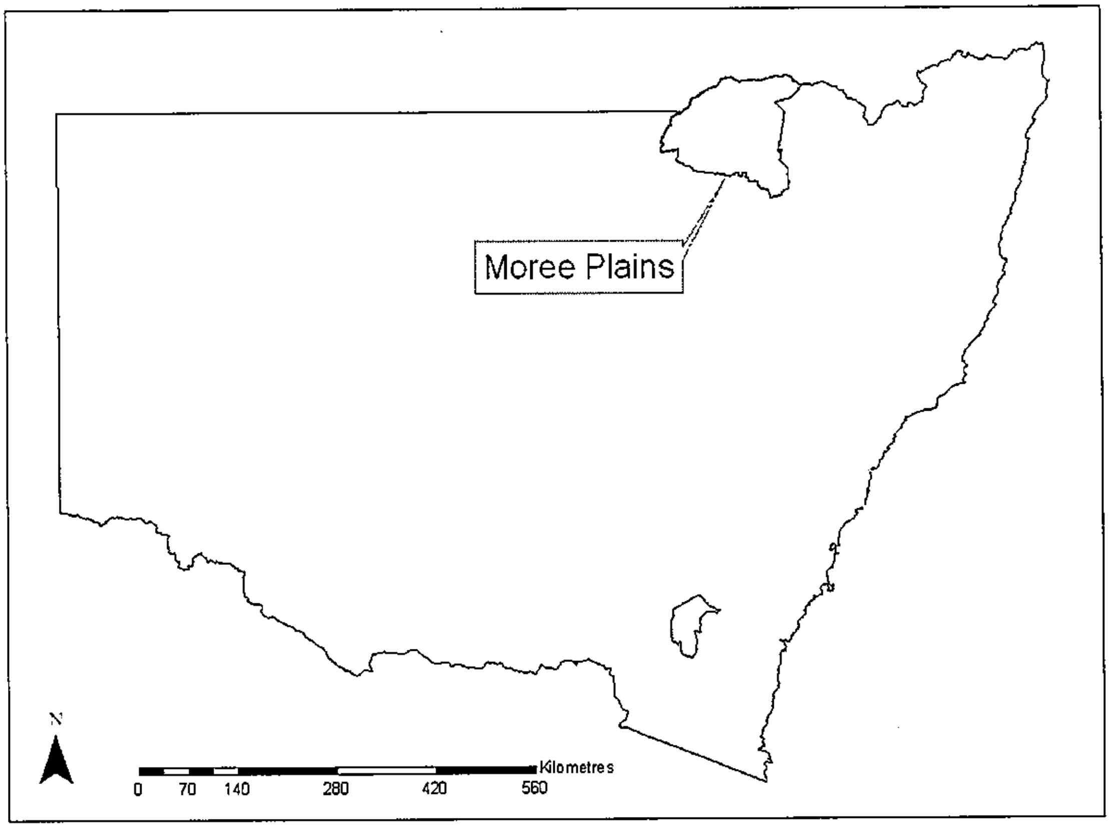


Fig. 2.1. Location of the Moree Plains in north-west NSW.

2.3 Climate

The climate of this region is semi-arid. Average annual rainfall increases from west to east across the study region, with an average of 503 mm at Collarenebri in the west, to 611 mm at Pallamallawa in the east (Commonwealth Bureau of Meteorology 2003). The average annual rainfall for Kirramingly Nature Reserve is 552 mm (Fig. 2.3). Rainfall is summer dominant (60%) but may occur at any time during the year (Fig. 2.4). As summer rainfall frequently falls as thunderstorms, it can be very localised. On average, Moree has 159 clear days and 88 cloudy days per year, with an average of 78 rain days per year (Commonwealth Bureau of Meteorology 2003).

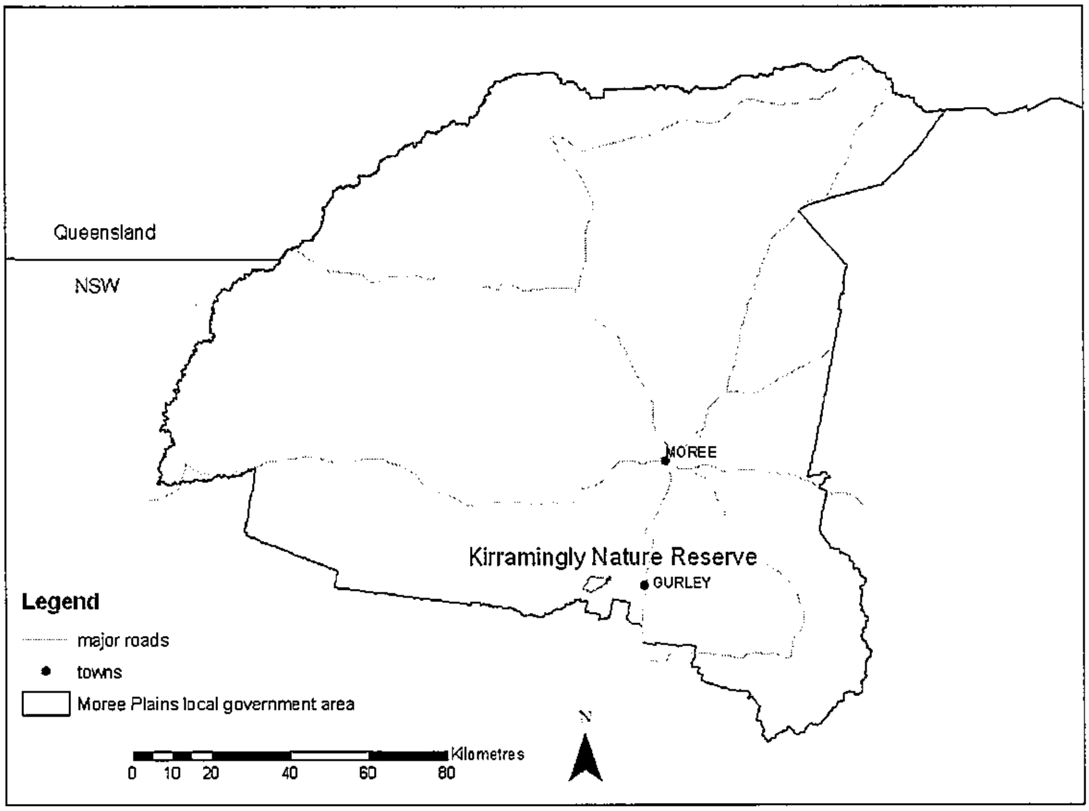


Fig. 2.2. Location of Kirramingly Nature Reserve within the Moree Plains local government area.

There were severe droughts in north-west NSW in the late 1890s (Gentilli 1971), but since 1947 rainfall has been substantially higher, with an increase in the amount of summer rain (Pickard & Norris 1994). During the current research, NSW and most of Australia suffered a major drought through most of 2002 and 2003. In an Australia-wide sense, this drought was one of the worst on record, being remarkable for the spatial extent of rainfall deficiencies and average levels of dryness (Anon. 2003). Drought is common in north-west NSW, with periods of drought lasting from a few weeks to several years (Campbell 1989). From 1900 to 1986, a total of eight major droughts (longer than 1 year) were recorded for western NSW (Campbell 1989). Short-term droughts occur more frequently, as Partridge (1996) suggested droughts occur irregularly every 5-7 years in Mitchell grasslands.

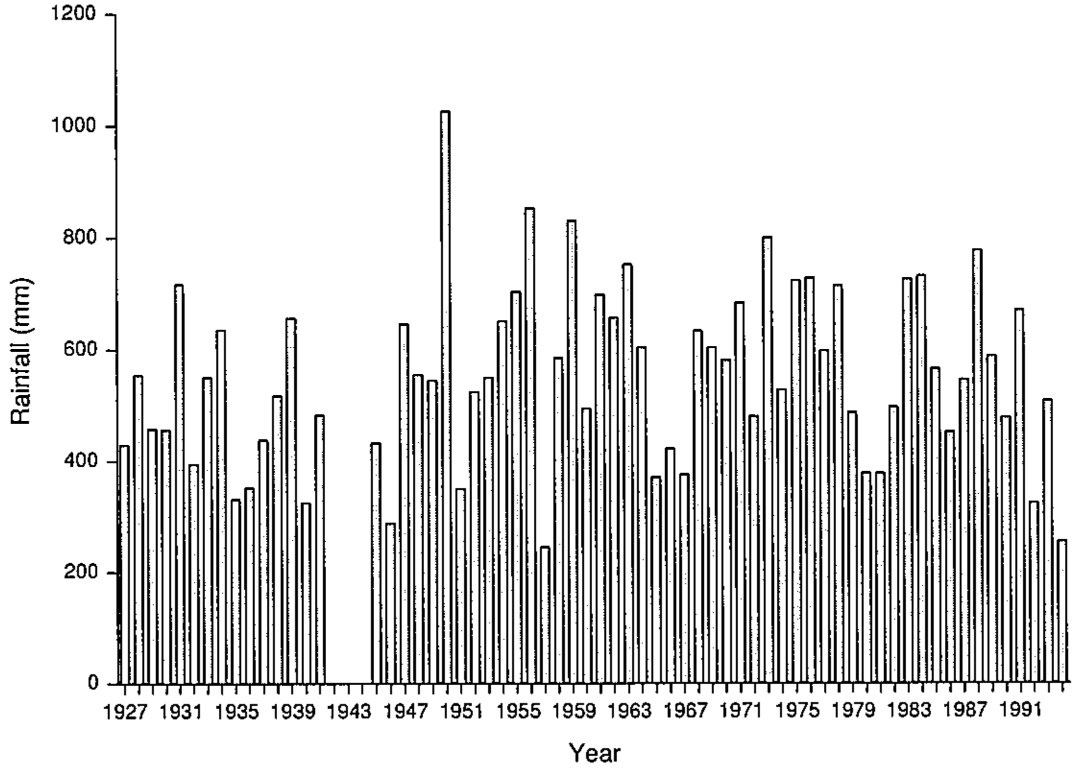


Fig. 2.3. Yearly rainfall (mm) for Kirramingly Nature Reserve from 1927 to 1994. No data were recorded in the years from 1942-1944.

The effectiveness of rainfall for plant growth is greatly influenced by temperature and evaporation (Moore 1970). Temperatures across the region range from an average maximum of 35°C in the summer months to 4°C average minimum temperatures in July (Hunter & Earl 1999). Moree has an average annual maximum of 26°C (Commonwealth Bureau of Meteorology 2003). Mean daily evaporation varies from 9.1 mm in summer to 2.6 mm in winter (Commonwealth Bureau of Meteorology 2003); hence winter rain can provide effective soil moisture. Evaporation rates are influenced by humidity and wind. Mean annual relative humidity for Moree is 63% at 9.00 a.m. and 39% at 3.00 p.m., and is highest during winter. Mean wind speed for Moree at 3.00 p.m. is 10.9 km/hr (varying from 8.5 km/hr in April to 13.4 km/hr in November) (Commonwealth Bureau of Meteorology 2003). Frosts occur in winter months, with Moree having 11 days where the minimum is $\leq$0°C on average each year.

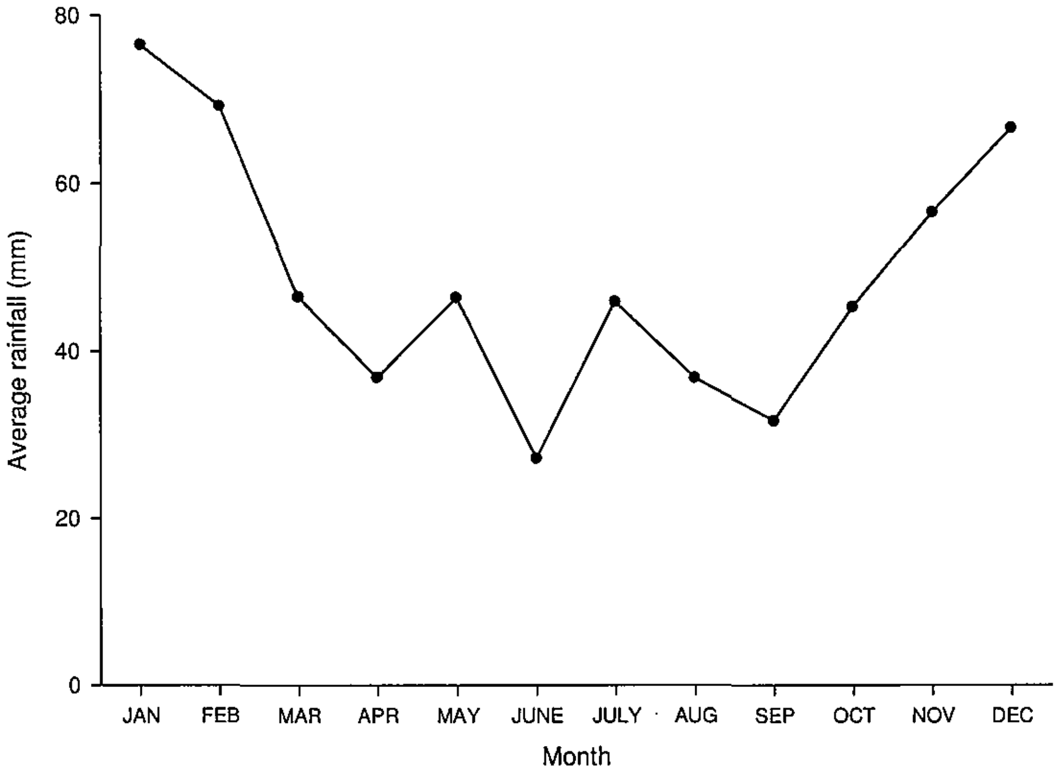


Fig. 2.4. Average monthly rainfall (mm) for Moree (1960 to 2003). Data were obtained from the Commonwealth Bureau of Meteorology (2003).

2.4 Landform

The majority of the Moree grasslands occur on floodplains associated with the Barwon, Gwydir and Mehi river systems and their tributaries (Hunter & Earl 1999). The Moree Plains are mostly gently sloping and undulating plains, falling away in elevation to the south-west (Hunter & Earl 1999).

Kirramingly Nature Reserve is located on a floodplain in the Gwydir catchment at an altitude of 200 m above sea level, with very little change in elevation over the Reserve (fall of 1.3 m over 1 km to the west). The Reserve has not been inundated by floodwaters in living memory (i.e. no water runs over the Reserve), although localised pooling of water can occur in shallow depressions after heavy rain (P. Wall, pers. comm., 2002). Certain areas can be under water for 3-4 days after heavy rains (P. Wall, pers. comm., 2002). There are no creeks within the Reserve, although Little Bumble Creek runs parallel to the eastern border of the Reserve, in the adjacent travelling stock route (TSR).

2.5 Soils and geology

The floodplains of the region are made up of Quaternary alluvium comprising a range of soil types, most of which are clays (Hunter & Earl 1999). The cracking clay soils have high fertility compared to most Australian soils, and this has been the reason for their agricultural development. Most of the grey and brown clay soils are sodic due to accumulation of soluble salts in depressions and gilgais (Thompson 1982). These soils are dynamic in response to moisture. When wet they swell impeding infiltration, and when dry they shrink, leaving deep soil cracks (up to 1 m deep). They are capable of storing large amounts of water, but significant rainfall events are required before plant growth can commence, because a high proportion of the moisture stored in the soil is unavailable to plants. The heavy clay soils are thought to be resistant to erosion because of their fine texture and strong structure and the lack of relief in the region (Campbell 1989), although some erosion can occur in floods (W. Yates, pers. comm., 2001).

Throughout the clay floodplains are ridges of red earths and sandy red earths. These are of generally low fertility (Boyland 1984) and rarely flood. Red and brown soils with a higher loam content are more common on the eastern boundary of the study region, where elevation is higher.

Kirramingly Nature Reserve is located on fine-grained, cracking, grey clay soil, free of sand or gravel, and typical of the floodplain soils in the region.

Most of the region is underlain by Cenozoic sedimentary parent materials. However, rocks in the area may be derived from the Lachlan Fold Belt (Devonian-Silurian rocks), the southern New England Fold Belt (Permian-Triassic), the Great Australian Basin (Jurassic-Cretaceous) and the Sydney-Bowen Basin (Permian-Triassic) (Hunter & Earl 1999). Clays and alluvium developed during the Quaternary due to weathering of basalt associated with the Nandewar Volcanics (Offenberg *et al.* 1968).

2.6 Aboriginal history and land-use

Archaeological studies suggest that Aboriginal people have occupied western NSW for over 40 000 years (Nicholson 1981). Prior to European settlement Aboriginal people used fire in their daily activities for a variety of purposes (Nicholson 1981). Fire stimulated the growth of new plants that were food for both Aboriginals and the animals they hunted

(Nicholson 1981). Unfortunately there is a lack of evidence on when and how much Aboriginal burning took place on the Moree Plains. Parbury (1988) states that annual burning off of grassland in NSW was often done at the end of summer, when the grains and fruits had been harvested and the grass was dry. However, Fensham (1997) suggests fires may have been less frequent in inland Queensland. The extent of Aboriginal fires is also largely unknown and probably varied greatly. Fires may have been more expansive in inland areas (e.g. expansive Mitchell grasslands of Queensland) due to the absence of barriers and the arid climate (Stocker & Mott 1981).

Kirramingly Nature Reserve is within the area occupied by the Kamilaroi Aboriginal nation. The area is currently administered by the Moree Local Aboriginal Lands Council (LALC) (Anon. 2002). There is evidence in the form of open campsites, carved and scarred trees and burials, that the area within 25 km of the Reserve was used by Aboriginals (Anon. 2002). Lou Swan of the Moree LALC is reported as saying that fires were used regularly across the Gallathera plain, including the area that is now Kirramingly Nature Reserve (T. Waters, pers. comm., 2002).

During the period of European settlement on the plains, there were several massacres of Aboriginal people. In 1837, local troopers and stockmen at Gravesend, near Moree, massacred over 200 Aboriginals (Parbury 1988). One can assume that shortly after European settlement and the onset of livestock grazing, the traditional Aboriginal methods of land-use (i.e. burning) were discouraged and greatly diminished. Hence, there was a massive shift in grassland management from disturbance with fire to disturbance with grazing. There is a body of circumstantial evidence which suggests that altered fire regimes following European settlement have resulted in substantial changes in the range and structure of many vegetation types (Bowman 1998), however, there is no evidence of this on the Moree Plains.

2.7 European history and land-use

European settlement in the region started soon after Sir Thomas Mitchell's expedition in 1831. The semi-arid region of western NSW has now been grazed for between 130-170 years (Cunningham & Milthorpe 1981). It is very likely that all of the grassland communities within the study region have had at least some history of livestock grazing.

The current main land-use is grazing with sheep or cattle, depending on market prices, and certain areas are used extensively for dryland cropping (Hunter & Earl 1999). Cultivation for cropping, however, has only become common in the last 50 years. Dryland cropping is usually continuous, and up until the last 10 years generally no fertiliser was applied (Bellotti *et al.* 1986). Fertilisers are now used for certain crops, such as wheat and barley (Edwards 2004). A range of crops is planted including wheat, barley, oats and sorghum. A relatively small amount of land has been sown to introduced pastures (e.g. Bambatsi (**Panicum coloratum* var. *makarikariense*), Purple Pigeon Grass (**Setaria incrassata*) and Buffel Grass (**Cenchrus ciliaris*)). According to recent vegetation mapping in the Moree Shire (Peasley & Walsh 2001), 43% of the Shire is used for dryland cropping and a further 9% is used for irrigation cropping (i.e. mostly cotton) (Table 2.1). Approximately 45% of the remaining land is grazed. Approximately 1.5% of the Shire is travelling stock route and <0.1% is reserved for conservation (A. Pittman, pers. comm., 2001).

Table 2.1. Percentage of land-use types for the Moree Shire (modified from Peasley & Walsh 2001)

Land-use	**Percentage**
Dryland cropping	43
Grassland	32
Timber / trees	12
Irrigation cropping	9
Drainage, creeks and rivers	3
Water storages	1
Wetlands	<1
Horticulture	<1
Other	<1

It is likely that the amount of cultivation in the region has increased in recent years, due to market prices, and SEPP46, *Native Vegetation Conservation Act 1997*. The Act requires that land that has not been cropped for more than 10 years be subject to a development application before it can be returned to cropping. In some cases this has forced landowners to cultivate land before the 10-year limit to retain a higher value for their land (cropping land is worth considerably more than land that cannot be cropped).

Given the extent of cultivation on the Moree Plains native grasslands are highly fragmented, and often surrounded by cultivation paddocks. This can lead to invasion of remnants by introduced species, and has important consequences for grassland seed dispersal due to increased isolation of remnants. Kirramingly Nature Reserve is surrounded by agricultural enterprises including cropping and grazing. The Rural Lands Protection Board (RLPB) manages travelling stock routes (TSRs) that border the Reserve to the west, south and east. The Reserve itself has a long history of pastoral use and is currently divided into a number of paddocks separated by fences. A minor public road also divides the Reserve. While most of the Reserve was managed as a grazing property, some of the paddocks in the north-eastern section were cultivated for fodder crops and wheat. The last cultivation occurred in 1991 and 1992 (P. Wall, pers. comm., 2002).

Kirramingly has been grazed by stock since the pastoral industry was established on the north-west plains of NSW. In the late 1920s, *Astrebla* spp. seed was spread over the Reserve and surrounding stock routes by Sid Nicholson (the owner of the property at the time) (P. Wall, pers. comm., 2002) to improve the grazing value of the land. It was initially a sheep pastoral property, and was apparently overstocked between 1929 and 1946 (P. Wall, pers. comm., 2002). No accurate grazing records are available, but it is likely that the Reserve was heavily grazed by sheep and cattle during periods of drought. For example, Peter Wall (pers. comm., 2003) recalls the pasture being in poor condition during the 1965 drought. The Reserve was destocked in 1996 and is currently grazed only by kangaroos.

2.8 Botanical exploration

Sir Thomas Mitchell led an expedition through the eastern part of the Moree Plains in 1831, from the present-day Narrabri to Moree and across to Mungindi and Collarenebri (Hunter & Earl 1999). Mitchell (1839) described the vegetation as being dominated by Weeping Myall (*Acacia pendula*) and Belah (*Casuarina cristata*), with 'rich grass'. The records from this expedition commonly remark on the productive grasslands. For example, 'We crossed … a line of ponds in a deep channel … and, beyond them, was a rich plain, with a few clumps of trees; where the grass was remarkably good …' (Mitchell 1839).

An exploration by Sturt in 1829, just west of the Moree Plains, provided quite different descriptions to those by Mitchell (1839). Sturt's (1833) description stated that grasslands were not a common feature, and that chenopod shrubs were commonly more abundant than grass. This is supported by Hunter and Earl (1999) who report that the vegetation generally grades from grassland-dominated communities in the east to open chenopod shrubland in the west.

2.9 Major plant communities

The pre-European vegetation of the Moree Plains probably consisted of a mixture of woodlands, grasslands and chenopod shrublands. The current major vegetation types of the Moree Plains are shown in Table 2.2. Grasslands with less than 5% tree canopy cover occupy approximately 19% of the Shire. Such grasslands are thought to occur naturally throughout much of the region because of the fine textured soils (i.e. cracking clays). The fine texture of these soils limits water availability, and in conjunction with the deep soil cracking and temporary seasonal waterlogging, limits tree seedling establishment (Beadle 1981). Despite this, there are certain tree and shrub species that grow well on the clay soils (e.g. Coolibah (*Eucalyptus coolabah*), River Cooba (*Acacia stenophylla*) and *Acacia pendula*). There are often patches or corridors of trees growing on the red earth ridges of lower soil fertility, and woodlands exist along the watercourses. Relatively open woodlands (with a canopy cover between 5 and 20%) are common, covering approximately 15% of the region (Table 2.3).

The grasslands of the Moree Plains have previously been referred to as a 'Mitchell grassland association' (e.g. Beadle 1948). Both *Astrebla lappacea* and *Astrebla elymoides* (Hoop Mitchell Grass) are widespread on the north-west plains of NSW (Cunningham *et al.* 1992; Harden 1993). *Astrebla* spp. were first collected from NSW in 1835 by Sir Thomas Mitchell (Orr & Holmes 1984). Mitchell grasslands occur in a discontinuous belt from northern NSW, through western Queensland and the Northern Territory to northern Western Australia (Campbell 1989), covering an area of 480 000 km^2 of which 328 000 km^2 is in Queensland (Orr 1975). Mitchell grasslands once occurred over about 14 000 km^2 of NSW (Orr & Holmes 1984), although this area has probably been greatly reduced. In NSW they generally extend south to a line between Bourke and Narrabri (Bellotti *et al.* 1986), although smaller patches do exist west of Wilcannia (R. Whalley, pers. comm., 2004). *Astrebla* spp. on the Moree Plains are at the eastern limits of their distribution (Bellotti *et al.* 1986).

Table 2.2. Current vegetation types and their estimated percentage distribution on the Moree Plains (modified from Peasley & Walsh 2001)

Vegetation type	% Distribution
Black Box (*Eucalyptus largiflorens*)	0.7
Brigalow (*Acacia harpophylla*)	0.3
Casuarina (*Casuarina* spp.)	2.1
Coolibah (*Eucalyptus coolabah*)	12.2
Cypress Pine (*Callitris* spp.)	1.8
Gum (*Eucalyptus* spp.)	2.4
Ironbark (*Eucalyptus* spp.)	0.5
Mallee (*Eucalyptus* spp.)	<0.1
Myall (*Acacia pendula*), Rosewood (*Alectryon oleifolius*), Whitewood (*Atalaya hemiglauca*)	0.4
Native Pasture (<5% tree / shrub cover)	19.3
Ooline (*Cadellia pentastylis*)	<0.1
Piliga Box (*Eucalyptus pilligaensis*)	0.1
Poplar Box (*Eucalyptus populnea*)	5.2
Sand Complex	0.3
Shrubland	1.0
Wetlands	0.3
White Box (*Eucalyptus albens*)	0.4
Wilga (*Geijera parviflora*) and Leopardwood (*Flindersia maculosa*)	0.2
Cultivation (<5% tree / shrub cover)	51.3
Water Storage / Ungrouped / Miscellaneous	1.4

Table 2.3. Percentage distribution of tree canopy cover over the Moree Plains (modified from Peasley & Walsh 2001)

Tree canopy cover	% Distribution
<5%	72
5-10%	8
10-20%	7
20-50%	10
50-80%	2
>80%	1

The long-lived perennial tussocks of *Astrebla* spp. provide a stable matrix in which a wide variety of other species may occur (Bellotti *et al.* 1986; Campbell 1989). Between the perennial grass tussocks, annual grasses and annual or short-lived perennial forbs frequently occur (Campbell 1989). Botanical composition at any point in time is dependent on a range of variables, such as location, seasonal rainfall and grazing history (Orr 1980a,b; Fensham *et al.* 2000). There are large differences in composition between states. In comparing Queensland and NSW, almost half of the total number of genera found in these grasslands are exclusive to either state (Bellotti *et al.* 1986). More annual winter herbage species occur in NSW, including annual medics and members of the Brassicaceae (Bellotti *et al.* 1986).

There has been recent concern about the decline of *Astrebla* spp. in NSW (e.g. Bowman *et al.* 1996). It is believed that large areas of Mitchell grassland have never recovered from flooding in the 1970s (Campbell 1989) and continuous overstocking for extended periods (Francis 1935). In a recent vegetation survey of the Moree Plains, Hunter and Earl (1999) suggested that 'Mitchell grassland' is not an appropriate name for the vegetation in this region, as *Astrebla* spp. were 'only dominant at a handful of sites'. In addition, some of the farmers and graziers around Moree have doubted the natural occurrence of *Astrebla* spp. on the Moree Plains (Clarke *et al.* 1998). This is based on reports that settlers spread *Astrebla* spp. seed from pack saddles in the early days of grazing (around 1930). While this is probably true, and may have resulted in increases in density and dominance of *Astrebla*, there is evidence to suggest that *Astrebla* spp. do occur naturally (Clarke *et al.* 1998). For example, herbarium specimens from the area date back to before the 1930s.

There have only been two site-based surveys of the grassland vegetation in the region, Hunter and Earl (1999) and Clarke *et al.* (1998). Hunter and Earl (1999) described ten grassland and shrubland communities that occur on the Moree and Walgett Plains. The most abundant grassland within the area surveyed was the 'Queensland Bluegrass Community', which occupied 38 of the 104 sites surveyed in Moree Shire (Hunter & Earl 1999). Species of *Astrebla* were common in this community, and were a major component at nine of the 38 sites. *Dichanthium sericeum* was a dominant species at another two of the ten communities (i.e. the 'Mixed Grassland' and 'Plains Grassland' communities). The vegetation of Kirramingly Nature Reserve was classified into four major vegetation associations by Clarke *et al.* (1998). Mitchell grasses (*A. elymoides* and *A. lappacea*) are common throughout most of the Reserve and were dominant species within two of the associations described by Clarke *et al.* (1998). The vegetation of the Reserve can be broadly described as *Astrebla elymoides* and *Dichanthium sericeum* tussock grassland.

Moore (1970) classified Mitchell grasslands as 'arid tussock grasslands' and *Dichanthium sericeum* grasslands (tropical grasslands) as the humid equivalent of *Astrebla* grasslands. According to Beadle (1981), *Astrebla* spp. dominated grasslands grade into *D. sericeum* dominated communities to the east. On the Moree Plains there is an overlap of these two dominant species as areas dominated by *Astrebla* spp. often have *D. sericeum* as a co-dominant (Clarke *et al.* 1998; Hunter & Earl 1999). There is also some overlap between *D. sericeum* grasslands and the drier temperate grasslands (Plains Grass, *Austrostipa aristiglumis* dominated) in the south (Moore 1970). Early reports (e.g. Francis 1935) suggest that *D. sericeum* was originally dominant in at least some areas of the arid tussock grasslands of NSW. However, the presence of *D. sericeum* may only be temporary in response to higher rainfall (Campbell 1989). Given the differing descriptions of grasslands on the Moree Plains and seasonal variation in dominance of *D. sericeum* and *Astrebla* spp. (Williams & Roe 1975; Roberts 1978; Groves & Williams 1981) further surveys are required to better understand the dynamics of these grasslands.

CHAPTER THREE

The regional vegetation survey: The effect of management, environmental variables and sampling time on grassland composition and richness

3.1 Introduction

The most extensive plot-based vegetation sampling carried out in the region was that by Hunter and Earl (1999). They surveyed 200 quadrats over the Moree and Walgett Plains and suggested that *Astrebla* spp. are naturally a minor component of the grasslands of the Moree Plains. However, it is possible that *Astrebla* spp. were only a minor component in the Hunter and Earl (1999) survey because of past management or seasonal fluctuations in the abundance of *Astrebla* (Campbell 1989; Orr & Evenson 1991a; Section 2.9). Hence there is a need for further work on the Moree Plains to determine changes in grassland community composition through time, and provide a better ecological understanding of the areas that are dominated by *Astrebla* spp., and those that are not. Ecological information is needed to determine how individual species and grassland communities respond to different management regimes. While there is a substantial body of research on Mitchell grasslands (e.g. work by Orr 1975-2000), there is relatively little information on how these grasslands should be managed from a conservation perspective. Hence the effects of management and environmental variables on the condition (from a plant diversity prespective) of the *Astrebla* and *Dichanthium sericeum* grassland remnants are largely unknown. Further surveys are also required to provide a detailed inventory of species that are currently present within the region, to help prevent or, at least recognise, local declines or extinctions of species that may be brought about by agricultural development.

There have been many studies investigating the influences of grazing on grassland composition and richness in Australia (e.g. Whalley *et al.* 1978; McIntyre & Lavorel 1994b; Fensham 1998; Fensham *et al.* 1999; McIntyre & Lavorel 2001; Clarke 2003; McIntyre *et al.* 2003) and worldwide (e.g. Collins & Barber 1985; Noy-Meir *et al.* 1989; Belsky 1992; Noy-Meir 1995; Milchunas *et al.* 1998; Olff & Ritchie 1998; Stohlgren *et al.* 1999). A review of the effects of livestock grazing on grassland composition and richness (Section 1.2.4) has recognised that many grazing studies report conflicting results, some arguing that certain grazing regimes are beneficial to native grasslands (e.g. Orr & Evenson 1991a; Trémont & Whalley 1995; Bai *et al.* 2001) and others suggesting removal of livestock grazing (e.g. Lacey & Van Poollen 1981; Fleischner 1994). The responses of individual species to grazing also vary greatly between different studies (Vesk & Westoby 2001). Livestock grazing is widespread over the Moree Plains, but there is a lack of published information on its effects on native grassland species composition and richness.

The effects of past cultivation on present day vegetation have received little attention in Australia, despite the large area that is used to grow crops. This is partly because many areas that have been cultivated remain in cultivation, with only short breaks and crop rotations. The ability to restore cultivated land to its pre-cultivation state is important in areas like the Moree Plains where cropping is one of the main land-uses. Some studies suggest it might take decades or even centuries for species richness and botanical composition to return to natural conditions (e.g. Kindscher & Tieszen 1998; Römermann *et al.* 2005). While the grasslands of the Moree Plains have some capacity to regenerate after cultivation (Whalley 2000), it is not known whether species richness and composition can recover or how long the recovery process takes. It is likely that the period of cultivation has an important influence on the ability of these grasslands to regenerate.

Environmental variables have an important influence on native grassland composition, and can potentially mask the effects of agricultural management (Bellamy *et al.* 1996). The importance of rainfall in determining species composition and biomass is well recognised for Mitchell grasslands (Roberts 1978; Foran & Bastin 1984; Orr & Holmes 1984; Campbell 1989; Orr & Evenson 1991a; Bowman *et al.* 1996; Fensham *et al.* 2000). In fact, seasonal influences can have more profound impacts on species composition and abundance than livestock grazing (e.g. Orr 1981; Foran & Bastin 1984; Orr & Evenson 1985, 1991a). Hence environmental variables (e.g. rainfall) may have a greater influence on species composition than disturbance-related variables (e.g. grazing and cultivation) in the grasslands of the Moree Plains. During this study NSW was under the influence of a severe drought (Section 2.3). Changes in grassland communities due to drought are varied (Section 1.2.7). Bazzaz and Parrish (1982) listed a number of potential impacts, including changes in species composition and a reduction in species diversity. While *Astrebla* spp. are able to survive droughts (Campbell 1989), the effects of drought on other herbaceous species and species richness have not been widely reported.

Other environmental variables also have an influence on species composition (McIntyre & Lavorel 1994b; Barbaro *et al.* 2004; Prober & Thiele 2004). The influence of soil type and soil chemistry on botanical composition of some Australian grassland communities has been well documented (e.g. Prober & Thiele 1995; Bowman *et al.* 1996; Bean & Whalley 2001; Prober *et al.* 2002). However, apart from Hunter and Earl (1999), there is little information regarding the effects of soil type on grassland composition on the Moree Plains. Other variables, such as flood frequency, are potentially important for floodplain environments like the Moree Plains (Hunter & Earl 1999; Capon 2005), but have not been thoroughly researched. Evidence suggests, however, that flooding has a negative influence on *Astrebla* spp. and hence composition (Bellotti *et al.* 1986; Bowman *et al.* 1996). Therefore, further investigation of its influence is necessary.

In this chapter, I report findings from a regional survey of grasslands with a range of management histories on the Moree Plains. The major objectives of this chapter are to summarise the floristic information from this survey and to determine:

1. Changes in grassland composition in different seasons and years.
2. The relative importance of disturbance-related variables (grazing, cultivation, flooding and fire) and environmental variables (e.g. rainfall, soil type) in determining species composition.
3. Which disturbance-related variables and environmental variables influence grassland condition.
4. The impact of grazing, type of grazing (periodic or continuous) and stocking rate on plant species richness and composition.
5. The impact of past cultivation, time since cultivation and the number of consecutive cultivations on plant species richness and composition.
6. The impact of flooding on plant species richness and composition.

3.2 Methods

3.2.1 Site selection and sampling methods

Vegetation survey sites were generally located in the Moree Plains local government area (Fig. 3.1). Survey sites were chosen based on management history, in particular the history of cultivation, grazing, flooding and fire. Hence sampling took place over a range of land tenures, including privately owned properties, reserves, travelling stock routes (TSRs) and other remnants (e.g. cemeteries). Sites were not evenly distributed over the study region due to the lack of rainfall in certain areas, particularly in the west (Fig. 3.1). Areas that had not received rainfall in the months prior to sampling were generally avoided because species richness is lower and plant identification is more difficult due to the lack of reproductive material. Sampling was carried out from late summer (February) to early winter (June) 2002 and 2003, and in spring (September and October) 2002 and 2003. The majority of sampling took place in late summer and autumn (hereafer referred to as autumn). This is thought to be the best time to sample these grasslands (Clarke *et al.* 1998; Hunter & Earl 1999), as the warm temperatures combined with rainfall provide good conditions for germination and growth. However, sampling at this time is unlikely to record species present as seeds or underground buds. Hence certain sites were revisited in spring to record species (i.e. winter-growing annuals) that may not have been detectable in summer or autumn.

The selection of an actual sampling location within a paddock or section of stock route was arbitrary, but sites were chosen within a homogeneous section of vegetation that was representative of the grassland in the surrounding management unit. Certain areas were avoided, such as stock camps, wetlands, areas dominated by introduced pasture (i.e. >50% cover), areas cultivated within the previous year and areas with a steep (>18°) slope. Up to four sites were surveyed on any one property in each year. In most cases no more than one site was placed in any given paddock. An exception to this occurred when two contrasting methods of management, frequencies of flooding, or soil types occurred within the one paddock, in which case a maximum of two sites was placed in the paddock.

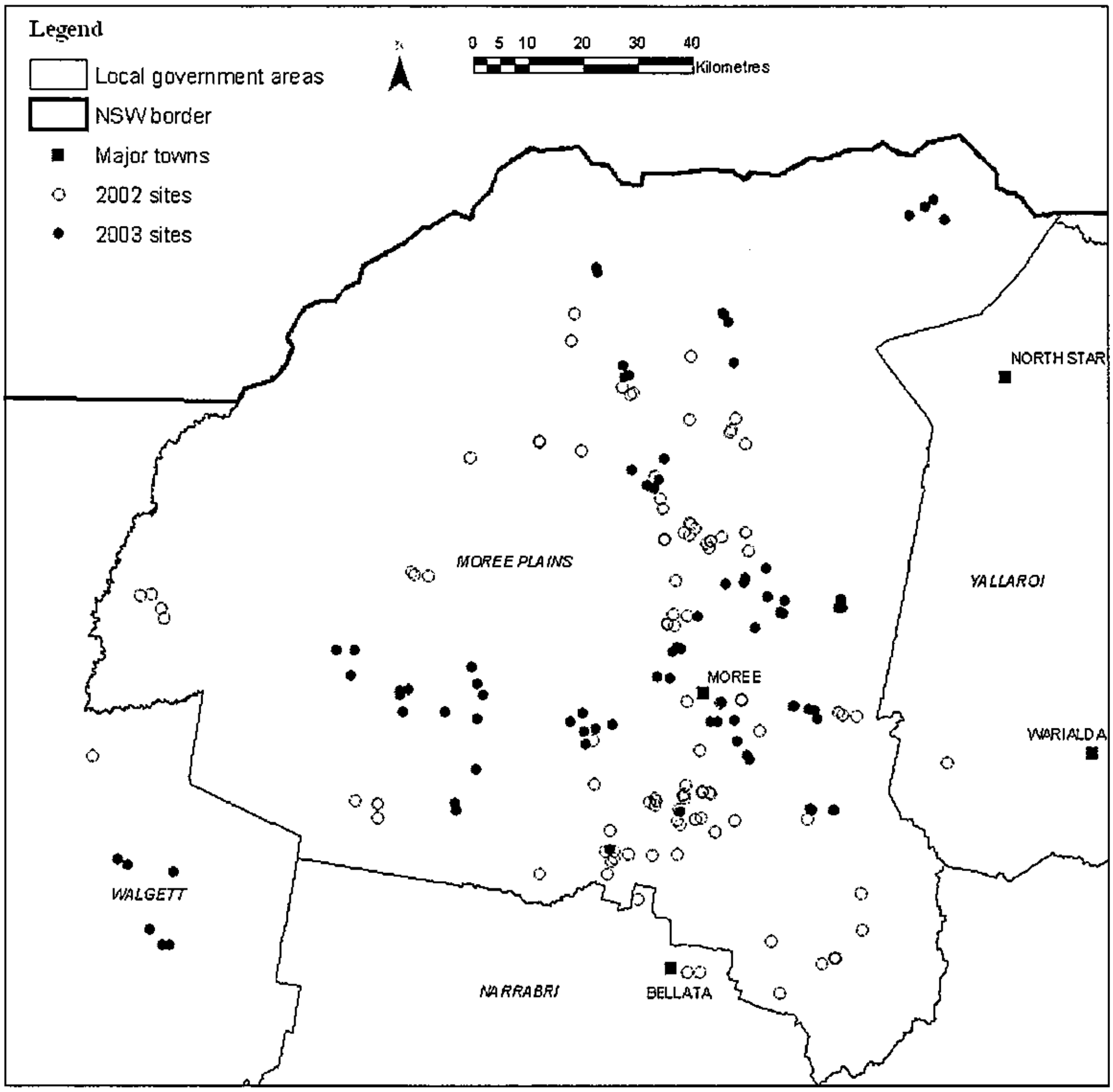


Fig. 3.1. Location of quadrats sampled in 2002 and 2003 within the Moree Plains local government area and neighbouring local government areas (italicised font).

A range of native grassland communities was sampled over the landscape, although most sites were on the clay soils (the most common soil type for the study region). Nine sites on non-clay soil types were sampled for comparative purposes. Areas that were dominated by, or that could potentially be dominated by, *Dichanthium sericeum* or *Astrebla* spp. were generally targeted in this survey, although not all sites fell into this broad category. Sites were chosen in areas of open grassland (sparse trees or shrubs, <5% cover of woody plants), shrubby grassland (5-25% cover of shrubs), and open grassy woodland (5-25%

cover of trees). Areas with a dense cover of trees were avoided, as this study was concerned with herbaceous vegetation.

An attempt was made to re-survey 32 grassland sites previously sampled by Hunter and Earl (1999) to investigate changes in composition over 3-4 years. Global Positioning System (GPS) coordinates for the Hunter and Earl (1999) sites were provided by the Department of Infrastructure, Planning and Natural Resources, but sites were not permanently marked during their survey. As sampling proceeded, difficulties in accurately locating the Hunter and Earl (1999) sites were encountered. Despite the use of a GPS it was unlikely that exactly the same areas could be re-sampled and GPS locations did not always agree with site descriptions and information provided by Hunter and Earl (1999). Given that changes in dominance and composition can vary considereably over a few metres in grasslands I decided not to make site comparisons between the two surveys.

I was able to investigate short-term changes in composition by selecting 17 of the sites sampled in 2002 on TSRs and Nature Reserves for re-sampling (Fig. 3.2). These sites were marked with a steel post in the north-western corner to eliminate problems due to GPS accuracy. The sites were re-surveyed during spring 2002, autumn 2003 and spring 2003. The spring of 2002 was influenced by severe drought, which provided an opportunity to record changes in composition and abundance associated with low rainfall. The regional survey sampled a total of 235 quadrats, including repeated sampling at the 17 sites that were revisited. A total of 184 individual sites were surveyed, 98 in autumn 2002 and 86 in autumn 2003. Including the 17 sites revisited in autumn 2003, a total of 103 sites were sampled in this period.

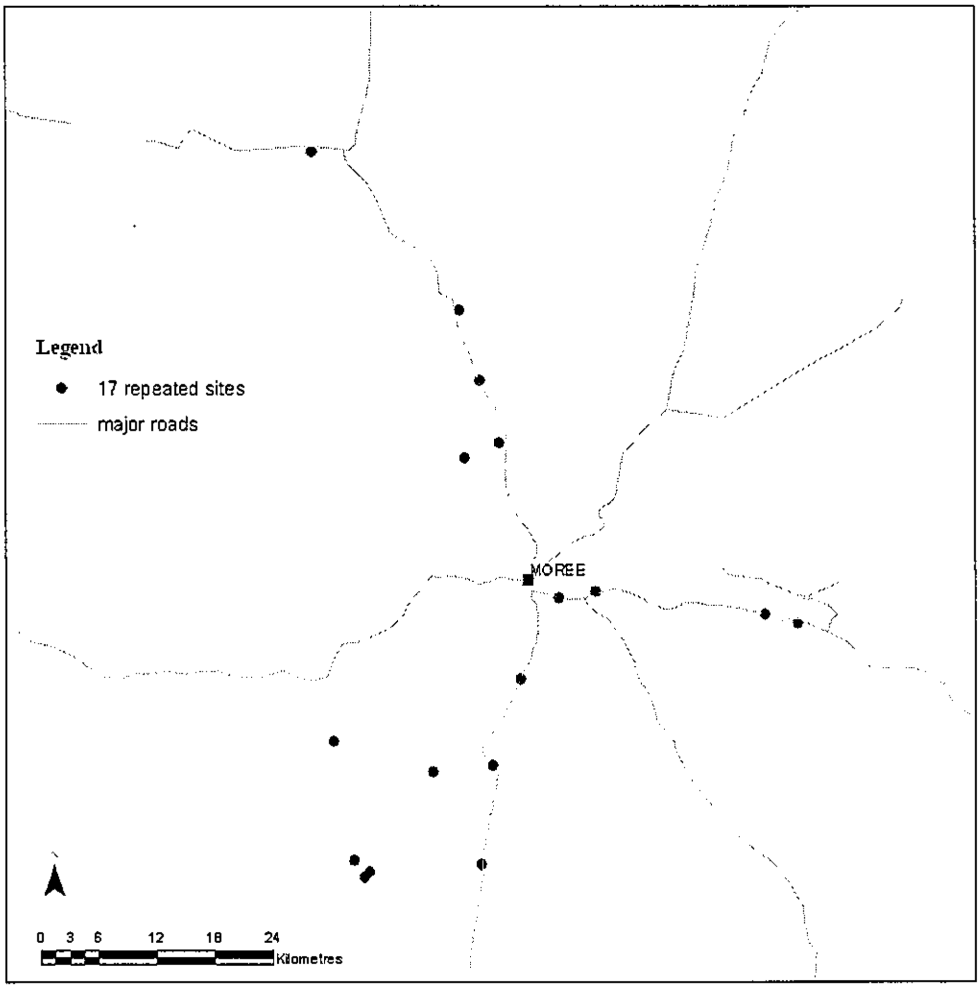


Fig. 3.2. Location of the 17 sites chosen for repeated surveys in spring and autumn of 2002 and 2003.

To be consistent with the vegetation survey by Hunter and Earl (1999), all vascular plants were recorded within 20 x 20 m (0.04 ha) quadrats. Each species was given a Braun-Blanquet cover abundance score using a six-point scale, by estimating its projected cover (Table 3.1). Error in the cover estimate was minimised by a single observer (Tom Lewis) making all estimates. Recording each species allowed later calculation of species richness and cover of natives, introduced species, grasses (Poaceae), forbs, graminoids (monocotyledons that are not Poaceae), sub-shrubs, shrubs and trees. This method is useful for rapidly assessing the composition of vegetation, and has been used widely in vegetation surveys (see Hunter & Earl 1999). Species were considered 'dominant' if they had a cover abundance score of ≥3 (i.e. ≥6% projected cover). While 6% cover may be considered low for a dominant species, higher cover abundance scores (i.e. ≥4) were not used to define dominance because some sites had no species with cover scores of greater than three. The area immediately outside each quadrat (within 10 m) was searched quickly (10 minutes) to record any additional species adjacent to the quadrat. A photograph of each site was taken at the time of sampling.

Geographical position was measured using a Garmin® GPS (eMap), with coordinates (easting and northing) for the WGS 84 map datum, recorded in the north-west corner of each site. Drought conditions in 2002 and 2003 influenced the location of sampling sites. In 2002 more sites were sampled in the east of the study region, due to the very dry conditions 30 km west of Moree. The first 98 sites surveyed were marked with a steel post in the north-western corner to allow these sites to be re-sampled in future surveys. Sites sampled in 2003 were not marked with posts, due to the cost and the fact that some posts used to mark TSR sites in 2002 were missing. A number of variables were recorded at each site, and these are discussed below.

Table 3.1. Braun-Blanquet cover abundance scores (adapted from Walker & Hopkins 1990)

Projected Cover (%)	Score
<5 (<3 individuals / quadrat)	1
<5 (≥3 individuals / quadrat)	2
6-25	3
26-50	4
51-75	5
>75	6

3.2.2 Site stratification

The influence of any one management variable on species composition needs to be treated with caution as its influence can be confounded by other management variables or environmental variables. Hence interpretation of the influence of one variable on species composition should be considered in light of the other variables. Prior to sampling, I decided on three main variables that were likely to influence species composition and abundance. They were (i) occurrence and type of livestock grazing (not grazed, continuously grazed or periodically grazed); (ii) the occurrence of cultivation (not cultivated or cultivated); and (iii) the occurrence of flooding (not flooded or flooded). Sampling was stratified based on these variables, and I attempted to sample a range of combinations of the variables (Table 3.2). Sampling ended when sufficient replication was achieved. Where possible an attempt was made to sample at least four sites in each cell of the stratification table. Where combinations had sufficient replication specific comparisons among variables could be made.

Table 3.2. Stratification table, showing the number of sites sampled within each of the defined combinations. Sites on non-clay soils are not included ($n = 175$). * refers to sample sizes that are too small for univariate analysis

	Not grazed	Periodically grazed	Continuously grazed
Not cultivated, not flooded	8	48	7
Not cultivated, flooded	7	32	13
Cultivated, not flooded	3*	18	4
Cultivated, flooded	2*	20	13

A total of 52 private properties were visited during the survey. For sites located on these properties, landholders were asked a series of questions regarding the management history of their grassland. These questions were based on history of cultivation, grazing, flooding and fire, along with a record of rainfall in the 4 months prior to sampling. The questions that were asked for each site are provided in Appendix 3.1. Landholders were generally able to provide the information on the day of sampling, but in some cases the questionnaire was left with the landholder if they agreed to send the completed questionnaire in the post.

Landholders were generally contacted a week or more in advance to ask whether they would participate in the research, then were called again closer to the sampling time, to arrange a time to meet. A number of landholders (approximately five) refused to participate in the research and some were concerned about the motives for the research. Information on the management history of TSRs and nature reserve sites was obtained through talking to Rural Lands Protection Board (RLPB) and National Parks and Wildlife Service (NPWS) rangers, respectively.

3.2.3 Disturbance-related variables

The main forms of grassland disturbance on the Moree Plains involve grazing, cultivation, flooding and fire. Drought may also be considered a disturbance but the effects of drought were investigated separately through repeated sampling of 17 sites over time.

Grazing variables

Sites were sampled on one of four different land tenures, with differing grazing histories: (1) Privately owned property (131 sites); (2) Public land (i.e. travelling stock routes and reserves) managed by the Moree RLPB (43 sites); (3) Nature Reserves, managed by the NSW NPWS or NSW State Forests (seven sites); and (4) Cemeteries, managed by local councils (three sites).

Time since livestock grazing was divided into four categories: (1) sites not grazed within 20 years prior to sampling; (2) sites not grazed within 5 years prior to sampling; (3) sites not grazed in the year immediately prior to sampling; and (4) sites that had been recently grazed, or had been grazed in the 12 months prior to sampling (Table 3.3). To my knowledge, there are no remnant grasslands in the study region that have been totally excluded from livestock grazing since European settlement.

Table 3.3. Number of sites in each category for time since last livestock grazing episode (*n* = 184)

Grazing variable	Number of sites
Not grazed for ≥20 years	6
Not grazed within 5 years	10
Not grazed within 1 year	7
Recent grazing	161

Livestock grazing intensity was estimated in two ways, using stocking rates, and determining whether or not sites were rested from grazing. For the latter, sites were divided into three regimes (Table 3.4):

(1) No recent grazing (i.e. no grazing within 2 years of sampling).

(2) Episodic or periodic grazing. This included areas where grazing was not continuous throughout an average year. Sites in this category had at least 2 months of spelling each year.

(3) Continuous grazing. These sites had a history of grazing throughout each year, without spelling. Such sites generally do get some rest from grazing during periods of drought, or at other times when there is not enough feed for stock.

For those sites that were periodically grazed, the percentage of the year that they were spelled from grazing was recorded as: (1) spelled for ≤3 months per year; or (2) spelled for >3 months per year (Table 3.4).

Table 3.4. Number of sites in each category of grazing regime (n = 184), and number of sites in each category of spelling from grazing for grazed sites only (n = 167)

Grazing variable	Number of sites
Grazing regime	
No recent grazing (within 2 years of sampling)	17
Periodic / episodic grazing	128
Continuous grazing	39
Spelling from grazing	
No spelling	39
Spelled ≤ 3 months / year	32
Spelled > 3 months / year	96

Stocking rates (dry sheep equivalents per hectare) were obtained or calculated for all sites with a history of recent grazing. For sites on private property, stocking rates were provided by landholders. In many cases, landholders did not keep accurate records of their stocking rates, in which case an overall yearly estimate was provided. An average stocking rate per year was calculated for each site, by taking into account periods of no grazing. Given that cattle and sheep have different metabolic requirements, stocking rates had to be converted for cattle-grazed sites by assuming that 10 dry sheep equivalents (DSE) was equivalent to one cow, bull or steer, and five DSE equivalent to one calf (NSW Agriculture 2002).

For sites on travelling stock routes, stocking rates were calculated using RLPB permit records in the months prior to sampling. This involved visiting the Moree RLPB office and looking through permit records to find those relevant to the TSR sites sampled. The permit records contain information on where the stock grazed (i.e. which stock route), the numbers of stock (sheep, cattle and horses), the number of days spent on each stock route and the distance travelled over that period. Using this information the total number of stock (cattle, sheep and horses) that used each route was calculated for a 5-month period prior to sampling, and the total number of days spent on the route was recorded. The average width of each stock route was calculated using RLPB maps, by measuring the width at 1-km intervals. For each permit record, a stocking rate was calculated by dividing the total number of stock grazing days (in DSE) by the area that they covered. The total stocking rate for the whole TSR could then be estimated by adding the individual stocking rates and dividing by the total number of days in the 5-month period.

The type of livestock grazing was also recorded for each site. This was done by recording whether sites were predominantly grazed by: (1) sheep; (2) cattle; or (3) both sheep and cattle, or a combination of sheep or cattle with other livestock (e.g. horses and goats) (Table 3.5). Grazing animal(s) were recorded for recent times (in the 3 years prior to sampling) and over a much longer time frame (based on the memories of landholders). Over a longer time frame, a higher number of sites had been predominantly grazed by sheep, and few had been predominantly grazed by cattle (Table 3.5). This suggests there may be a trend of decreasing sheep grazing and increasing cattle grazing over time.

Table 3.5. Number of sites in each category for type of grazing animal in recent time and over history (*n* = 183)

Grazing variable	Number of sites (recent)	Number of sites (over time)
No grazing	17	3
Predominantly sheep grazed	26	53
Predominantly cattle grazed	80	6
Sheep and cattle and/or other livestock	60	121

Cultivation variables

Sites were divided into one of four cultivation levels based on the information gained from land managers. The levels were: (1) never cultivated; (2) cultivated <5 years prior to sampling; (3) cultivated 5-15 years prior to sampling; and (4) cultivated >15 years prior to sampling (Table 3.6). Only sites where the history of cultivation was known were included. Sites that had been cultivated within 1 year of sampling were avoided, as such areas were generally depauperate in species and usually did not fit the definition of native grassland. Sites that had been cultivated were most commonly sown with cereal crops such as wheat, barely, oats and sorghum. Four cultivated sites were sown with introduced grasses or legumes (often after the last cereal crop cultivation). Sown introduced pasture species included Bambatsi Grass (**Panicum coloratum* var. *makarikariense*), Purple Pigeon Grass (**Setaria incrassata*), Green Panic (**Panicum maximum* var. *trichoglume*) and a variety of medics (e.g. Lucerne, **Medicago sativa*, and Snail Medic, **M. scutellata*). Two sites were sown with *Astrebla lappacea* in the 1980s, after the last cultivation.

For sites with a history of cultivation, the number of years of consecutive cultivations was recorded as: (1) ≤5 years of consecutive cultivations; or (2) >5 years of consecutive cultivations (Table 3.6).

Table 3.6. Number of sites in each category of time since last cultivation ($n = 184$), and length of consecutive cultivations ($n = 183$)

Cultivation variable	Number of sites
Time since cultivation	
Never cultivated	121
Cultivated <5 years ago	15
Cultivated 5-15 years ago	23
Cultivated >15 years ago	25
Length of cultivations	
≤5 years consecutive	20
>5 years consecutive	42

Flooding variables

A flood was defined as a period of time when the site would have been completely submerged by flowing water (usually from a nearby watercourse). Localised waterlogging from heavy rainfall events did not constitute a flood. Frequency of flooding at each site was divided into the following categories: (1) not flooded, or very rarely flooded (perhaps once in 30 years); (2) flooded annually, or at least once every 2 years on average; and (3) flooded, but with less than one flood every 2 years on average.

Sites that had a history of flooding were divided into three categories based on the amount of time that they were inundated during an average flood. These were: (1) inundated for <3 days; (2) inundated for 3-7 days; and (3) inundated for >1 week (Table 3.7). No information was available on the length of inundation for 12 sites, and for some TSR sites flood frequency was estimated from knowledge of nearby sites.

Table 3.7. Number of sites within each category of flood history (n = 184), and the length of time each flooded site was inundated (n = 81)

Flooding variable	Number of sites
Flood frequency	
Not flooded	94
Flooded at least every 2 years (high frequency)	37
Flooded, but less frequently (low frequency)	53
Inundation time	
Inundated <3 days	13
Inundated 3-7 days	36
Inundated >7 days	32

Fire variables

Sites were recorded as either: (1) not burnt (154 sites); (2) burnt within 5 years of sampling (22 sites); and (3) burnt, >5 years prior to sampling (seven sites). Some sites with a history of cultivation had been burnt to remove stubble, and such fires were not included in the above categories. A direct comparison between recently burnt sites (within 5 years) and long unburnt sites was made possible by sampling unburnt areas adjacent to the burnt area. Nine paired sites were sampled during the survey. Discussion with site managers indicated that each set of paired sites had received the same management history (apart from fire history).

3.2.4 Environmental variables

Rainfall

The amount of rainfall that had fallen in the 4 months prior to sampling, as well as average annual rainfall for the property, was obtained from most landholders. For sites that were not on private property, rainfall in the months prior to sampling and average annual rainfall were estimated using data obtained from the nearest rainfall recording station provided by the Commonwealth Bureau of Meteorology (2003), and nearby properties for which rainfall was known. The average annual rainfall for all sites ranged from 443 to 660 mm; while in the 4 months prior to sampling at each site, rainfall ranged from 75 to 357 mm (Table 3.8).

Table 3.8. Summary values for annual rainfall (mm) for each site and rainfall in the 4 months prior to sampling (mm)

Variable	Min	Max	Mean	SE	*n*
Annual rainfall	443	660	560.2	3.9	184
Rainfall prior to sampling	75	357	168.2	4.4	184

Sampling time

Sites were divided into two categories based on the year of sampling, with 98 sites sampled in 2002, and 86 new sites sampled in 2003. In addition the actual sampling date for each site was converted to Julian time (day 1 = 1 January 1900) and was included in analysis.

Soil variables

Soil type was broadly categorised in the field as either: (1) grey to black clay (132 sites); (2) brown clay (43 sites); or (3) other (sandy loams, red earthy soils, etc.) (nine sites). A soil sample was collected at each site using an auger with a 10 cm diameter and 10 cm depth. Two samples were taken at random points along the northern edge of each vegetation sampling quadrat, and were thoroughly mixed. After field collection, soils were dried in an oven at 40°C for approximately 48 hours, or until dry. Soils were stored in a cool environment for between 1 and 2 years before laboratory analysis. Immediately prior to analysis, samples from each site were repeatedly split using a sample divider until approximately 100 g remained. This sample was then placed through a sieving machine to ensure grain sizes <2 mm. The sieved sample was placed in a separate, airtight, labelled container for analysis. Soil samples were analysed for available phosphorous, organic carbon, pH, electrical conductivity and available sulphur.

Soil pH was measured in water, with a soil to solution ratio of 1:5 at 25°C, after 1 hour of shaking and a 20-30 minute settling time. The pH reading was taken using a standardised pH meter. Soil salt content was estimated by measuring the electrical conductivity of a 1:5 soil to water suspension at 25°C. Samples were mechanically shaken for 1 hour to dissolve soluble salts, and were rested for 20-30 minutes for the soil to settle. A measure of the amount of phosphorus available to plants was determined using the fluoride-extractable phosphorus, Bray method (Bray & Kurtz 1945, cited in Rayment & Higginson 1992). This method uses an extraction solution of 0.03 M ammonium fluoride in 0.025 M hydrochloric acid (HCl) and a very short extraction time (60 seconds) (Rayment & Higginson 1992). Available sulphur was measured using the liable and adsorbed sulphur method with potassium chloride (KCl) (Blair *et al.* 1991). This involved using a 0.25 M KCl extract heated to 40°C for 3 hours. The extracts were analysed using inductively coupled plasma (ICP) spectroscopy (Blair *et al.* 1991). Total nitrogen was not analysed in this study, as this variable is likely to be correlated with other soil elements (i.e. sulphur and phosphorus). More detailed methods for measuring pH, electrical conductivity, available phosphorus and available sulphur can be found in Rayment and Higginson (1992) and Blair *et al.* (1991).

Soil organic carbon was estimated by loss on ignition and follows the method of Allen *et al.* (1986). This provides a crude measure of the amount of organic matter in the soil. Approximately 1 g of soil was weighed and placed in a muffle furnace at 550°C for 2 hours. The sample was removed and allowed to cool to room temperature before reweighing. Loss in weight was calculated as a percentage of the dry weight. Percentage loss on ignition was converted to a percentage of organic carbon using the formula provided in Howard and Howard (1990).

Other environmental variables

At each site surveyed several other environmental attributes were recorded:

1. Altitude (metres above sea level) was recorded using a GPS. Altitude varied from 145 m in the west to 379 m in the east, with an average of 207 m for all sites.
2. Bare ground was estimated visually for each quadrat by recording one of three categories: (1) low bare ground (≤10% bare ground); (2) moderate bare ground (11-49% bare ground); and (3) high bare ground (≥50% bare ground). The majority of sites (77%) fell into the second category, while 15% of sites had high levels of bare ground, and 8% had low bare ground (Table 3.9).
3. Litter cover was also estimated at each site, as the percentage of ground covered by litter. This variable was also divided into three categories: (1) low litter cover (≤15%); (2) moderate litter cover (16-74%); and (3) high litter cover (≥75%). Half of the sites had moderate litter cover, 37.5% had low litter cover, and 12.5% high litter cover (Table 3.9).
4. Structural formation of the vegetation at each site was divided into four categories: (1) open grassland (only scattered trees or shrubs; 53% of sites); (2) shrubby grassland (5-25% cover of shrubs; 29% of sites); (3) open grassy woodland (5-25% cover of trees; 16% of sites); and (4) grassy woodland with shrubs (5-25% cover of trees with 5-25% cover of shrubs; 2% of sites).

Table 3.9. Number of sites within each category of estimated percentage bare ground and litter cover (*n* = 184)

Variable	Number of sites
Percentage bare ground	
Low	15
Moderate	141
High	28
Litter cover	
Low	69
Moderate	92
High	23

3.2.5 Grassland condition index

Sites were divided into three categories based on their condition. Sites in 'good condition' contained >36 native species, with ≤5 introduced species (a native to introduced ratio of >7.4), low cover (<10%) of introduced species, ≥3 native species (grasses, forbs, graminoids or sub-shrubs) with cover abundance scores ≥3 (i.e. ≥6% cover), and low or moderate levels of bare ground. Species of known conservation significance (listed for NSW or nationally) were a common feature of such sites, but were not a prerequisite for their definition. Sites in 'poor condition' were categorised by three or more of the following features: <20 native species, ≥10 introduced species, a native to introduced species ratio of ≤3, <3 native species (grasses, forbs, graminoids or sub-shrubs) with cover abundance scores ≥3 (i.e. ≥6% cover), high cover (>25%) of introduced species or high levels (≥50%) of bare ground. The occurrence of no species of conservation significance (listed for NSW or nationally) was also a requirement for such sites. Sites not defined within either of the above categories were considered to be of 'moderate condition'.

3.2.6 Taxonomy

Nomenclature for vascular plants follows 'Flora of NSW' (Harden 1992, 1993, 2000, 2002) and Wheeler *et al.* (2002) for all surveys in this thesis. An exception to this was for the Asteraceae genus *Ixiolaena*, which has recently been revised to *Leiocarpa* (Wilson 2001). Voucher specimens for most species were collected so that identifications could be verified. Specimens were labelled and pressed in the field. After each field trip, specimens were transferred to another press for at least 2 weeks. I identified most species, but each species identification was checked by at least one specialist, to ensure taxonomic accuracy. John Hunter, Wal Whalley or Lachlan Copeland checked the identification of most species collected. An attempt was made to identify specimens to genus where inadequate collections were made to allow identification to species, but several recently germinated specimens could not be identified and were record as 'unknown germinants'. One of the most common taxa, *Dichanthium sericeum*, could not be consistently identified to sub-species in the field and, hence, was not identified beyond the species level. The common name for each species is provided in Appendices (Appendix 3.2, this chapter; Appendix 4.1, Chapter 4; Appendix 5.1, Chapter 5; and Appendix 6.2 and 6.3, Chapter 6). At least one specimen of each taxon will be lodged at the University of New England's NCW Beadle Herbarium. Duplicate material will also be prepared for a specimen collection to be retained by the NSW National Parks and Wildlife Service.

3.2.7 Statistical analysis

Multivariate

Multivariate analyses were carried out using the program CANOCO, version 4.5 (ter Braak & Šmilauer 2002). Given the size and complexity of the regional survey data set, ordination analysis was chosen to provide insights into the effects of disturbance-related variables and environmental variables on species composition. The aim of ordination is to arrange points (in two-dimensional space) such that points that are close together on the ordination diagram correspond to sites that are similar in species composition (ter Braak 1995). Ordination involved two steps: indirect gradient analysis (unconstrained ordination) and direct gradient analysis (constrained ordination) (Lepš & Šmilauer 2003). Indirect gradient analysis was used initially to search for major gradients in the plant species data, irrespective of independent variables. Observation of the variability in species composition is important, because some of the variability may be missed if it is not related to the measured variables (Lepš & Šmilauer 2003). Direct gradient analysis was then used to explain the vegetation data in terms of specific explanatory variables of interest (e.g. disturbance-related and environmental variables) (ter Braak 1995).

Indirect gradient analysis using detrended correspondence analysis (DCA) produced a gradient length of 2.76 (after deletion of non-clay soils and non-significant variables) and no unmeasured gradient was detected, as the first DCA eigenvalue was not larger than the first canonical correspondence analysis (CCA) eigenvalue. The relatively short gradient length suggested species were responding roughly linearly to the explanatory variable gradients. However, for species data containing many zeros a unimodal method of analysis is recommended by ter Braak and Šmilauer (2002); hence, canonical correspondence analysis was selected as the most appropriate ordination technique. CCA has been widely used by ecologists (e.g. ter Braak 1986; ter Braak 1987; Palmer 1993; McIntyre & Lavorel 1994b; ter Braak & Verdonschot 1995; Oliver *et al.* 2000; Henderson & Keith 2002; McIntyre *et al.* 2002; McIntyre *et al.* 2003; Clarke 2003; Eilu *et al.* 2003; Barbaro *et al.* 2004; Yee 2004; Westbrooke *et al.* 2005). The statistical model underlying CCA is that species abundance is unimodal along environmental gradients (Palmer 1993). Whether it is best to use linear methods, such as redundancy analysis (RDA), instead of unimodal methods, such as CCA, has not yet been thoroughly studied (Palmer 2001). Hence the data were also analysed with RDA to confirm that relationships between variables and species were similar using both methods. The advantages of CCA, which is becoming the most widely used gradient analysis technique in ecology, are outlined by Palmer (1993).

Symmetric and biplot scaling options for CCA were selected in CANOCO. Symmetric scaling was selected because the relationships among samples and species were interpreted equally (ter Braak & Šmilauer 2002). Biplot scaling was used as it is better suited to short gradients (ter Braak & Šmilauer 2002). Species cover data were transformed since the data contained many zeros, by taking logarithms, using the transformation ln(10*X + 1), where X = species cover score. Transformation is suggested by ter Braak and Šmilauer (2002) as species abundance values often have a highly skewed distribution. The influence of explanatory variables on infrequent species may be misleading as their occurrence might be due to chance. Hence the option of 'down-weighting rare species' was chosen for the analyses and infrequent species (with two occurrences or less) were deleted from the data set prior to analysis. Species incidence data and species cover scores were analysed. Both analyses produced similar results and, hence, only results from analysis of species cover scores are presented.

Automatic forward selection was used to rank each explanatory variable and its relative importance in determining the species cover data and to reduce the number of variables in the data set (ter Braak & Šmilauer 2002). In this process explanatory variables are added one at a time, until no other variables are significant in explaining the variation in species composition (Palmer 2001). Monte Carlo permutation tests were used to test the significance of each variable, by repeatedly shuffling the samples and comparing the generated test statistics with the test statistic generated from the null hypothesis, which suggests species are unrelated to the explanatory variables. Each permutation leads to a new data set from which a test statistic can be calculated (ter Braak & Šmilauer 2002). The default option of 499 permutations was chosen, which is adequate for a test at the 5% significance level (ter Braak & Šmilauer 2002). With this number of permutations a significance level of P = 0.002 is the lowest possible. This means that none of the 499 ordinations based on permuted data sets achieved as good a result as the 'true' one (Lepš & Šmilauer 2003).

Variables that were not significant ($P > 0.05$) after forward selection were removed from the data set before re-analysis. All variables retained in the final analysis are shown in Table 3.10. Variables removed included length of spelling from grazing, predominant grazing animal over time, length of inundation time during a flood, time since the area was last burnt, soil pH and percentage bare ground. While these variables were removed to improve the clarity of ordinations, they may still have some influence on community composition. Given that there were still a relatively large number of variables (22) in the analysis it is likely that the effects of grazing, cultivation or flooding will be confounded by other variables. While this exploratory method of analysis limits interpretation of the effects of each variable on composition it has allowed identification of variables that have the most important influence on composition.

Many of the variables described earlier in this chapter are categorical, which is a problem since multivariate analyses cannot deal with categorical variables directly (Palmer 2001). To solve this problem all categorical variables were coded as 'dummy variables' within the data spreadsheet, where the dummy variable takes the value of one if the site belongs to that category, and zero if not (Palmer 2001). Forward selection with dummy variables is problematic because not all of the states of the dummy variable can be included in the analysis (Palmer 2001). CANOCO removes the superfluous dummy variables automatically, which does not result in any loss of information, but should be taken into account when interpreting the results. Problems in interpretation occur when there is a combination of significant and non-significant categories for a variable (J. Reseigh, unpublished data, 2004). When one or more categories of a variable were significant after forward selection all states belonging to that variable were retained in the analysis.

The aim of canonical ordination is to detect the main pattern in the relations between species and the explanatory variables (ter Braak 1995). A measure of the importance of each ordination axis is provided by an eigenvalue (λ). The first ordination axis has the largest eigenvalue, the second axis has the second largest eigenvalue, and so on (ter Braak 1995). An eigenvalue close to one suggests a high degree of correspondence between species and sites, while a value of zero suggests little correspondence. Biplots (ordination diagrams of two entities) were used to show explanatory variable vectors in the direction of maximum change, where important variables have vectors of greatest length. Dummy variables are represented as centroids on the ordination plots, which approximate the relative total abundances of species in the samples of that category (Lepš & Šmilauer 2003). CCA was carried out using data for all individual sites (not repeated sites), sampled in the autumn period of 2002 and 2003. Separate analyses were also carried out on the data for each of the years and for the 17 repeatedly sampled sites. The final (reduced) data set consisted of 14 environmental variables (with 21 categories), eight disturbance-related variables (with 23 categories), 175 sites and 260 species.

The conditional and marginal eigenvalues from automatic forward selection are presented for variables that contribute significantly (at the 5% level) to the model. For closely correlated variables, only a limited number are selected as conditional effects. After the best variable is selected, the conditional effects of variables correlated with it drop, sometimes dramatically (Lepš & Šmilauer 2003). Hence caution is necessary in interpretation of the conditional effects. Marginal effects provide the amount of variability in the species cover data that would be explained by a constrained ordination model using only that variable (Lepš & Šmilauer 2003). Deleting all but one collinear variable was not carried out, because even if there is a strong positive correlation between two variables, sites with high levels of one variable relative to the other may still have distinct species compositions (Palmer 1993). Fortunately CCA performs well with collinear and non-orthogonal gradients (Palmer 1993).

All ordination diagrams were created using species cover data and present linear combination scores (the default in CANOCO). Species richness variables (total, native, introduced, forb, grass, graminoid, sub-shrub, shrub and tree richness) were defined as supplementary variables in CANOCO and were added *post hoc* to the ordination by projection (by regressing the data on to the existing ordination axis) (ter Braak & Šmilauer 2002). To determine which species were correlated with different explanatory variables t-value biplots were constructed, where biplot projections are used to approximate the t-values of the regression coefficients that would be obtained from simple regression with one predictor (explanatory variable or variable category) and one response variable (species) (ter Braak & Šmilauer 2002). These plots can be used to determine statistically significant pair-wise relationships between species and explanatory variables, where significant relationships (positive or negative) occur when species arrows end within the Van Dobben Circles (Lepš & Šmilauer 2003). Species arrows that end within the circle have a corresponding t-value larger than two. The word 'associated' does not imply statistical significance when referring to ordinations in this thesis.

It is important to realise that while some variables have a significant influence on composition ($P < 0.05$) they might only explain a very small percentage of the variance and, hence, might not be considered biologically significant. As Palmer (2001) points out, even with random data, a number of falsely significant results will be obtained due to the problem of multiple comparisons, and the statistical power of the Monte Carlo tests is difficult to ascertain (Palmer 1993). CCA eigenvalues can in some senses be interpreted as variances (Palmer 2001). Partitioning of variance was used to determine the variance uniquely described by disturbance-related variables and environmental variables, the variance shared by both groups, and the unexplained variance. Palmer (2001) and Barbaro *et al.* (2004) described the method used to partition variance. It involves the use of partial ordination by 'factoring out' certain variables (e.g. all environmental variables) by setting them as covariates in the analysis. Hence three separate canonical correspondence analyses were carried out, one with no covariables, one with disturbance-related covariables and one with environmental covariables. Partial CCA (with all environmental variables defined as covariables) was used to search for patterns between disturbance-related variables and species composition.

Univariate

Paired t-tests were used to compare species richness at sites sampled in this survey with the previous survey of Hunter and Earl (1999), and to compare burnt and unburnt sites. Regression was used to investigate relationships between species richness and time since cultivation, stocking rate and rainfall. Single-factor analysis of variance (ANOVA) or t-tests (for two samples) were carried out to determine the effects of disturbance-related variables (e.g. stratification combinations in Table 3.2) on response variables (e.g. species richness variables). Square root transforms were applied to satisfy the assumptions of ANOVA for total, native, introduced, forb and grass species richness. The response variables graminoid, shrub and sub-shrub species richness contained many zero values and could not be transformed to meet the assumptions of ANOVA and were analysed using the nonparametric Kruskal-Wallis test or Mann-Whitney test (for two samples). Log transforms were applied to the soil variables, electrical conductivity, available phosphorus and available sulphur, while inverse transforms were applied to pH and organic carbon.

Table 3.10. Significant variables (disturbance-related and environmental) used in multivariate analysis after forward selection, their categories (if categorical) and codes used in figures and tables

Variable	Category and code in parentheses
Disturbance variables	
Land tenure	1) Privately owned (private) 2) Public land (TSR) 3) Nature Reserve (reserve) 4) Cemeteries (cemetery)
Time since grazing	1) Not grazed for ≥20 years (not grazed in 20 yr) 2) Not grazed within 5 years (not grazed in 5 yr) 3) Not grazed within 1 year (not grazed in 1 yr) 4) Recent grazing (recent grazing)
Grazing regime	1) No grazing within 2 years of sampling (no recent grazing) 2) Periodic / episodic grazing (periodic grazing) 3) Continuous grazing (continuous grazing)
Stocking rate	Continuous (stocking rate)
Grazing animal in recent time	1) No grazing within 2 years of sampling (no recent grazing) 2) Predominantly sheep grazed (sheep grazed) 3) Predominantly cattle grazed (cattle grazed) 4) Sheep and cattle and/or other livestock (mixed grazing)
Time since cultivation	1) Never cultivated (not cultivated) 2) Cultivated <5 years ago (cultivated <5 yr ago) 3) Cultivated 5-15 years ago (cultivated 5-15 yr ago) 4) Cultivated >15 years ago (cultivated >15 yr ago)
Length of cultivations	1) Never cultivated (not cultivated) 2) ≤5 years consecutive (≤5 yr consecutive cultivations) 3) >5 years consecutive (>5 yr consecutive cultivations)
Flood frequency	1) Not flooded (not flooded) 2) Flooded at least every 2 years (high flood frequency) 3) Flooded, but less frequently (low flood frequency)

Table 3.10. (continued)

Variable	Category and code in parentheses
Environmental variables	
Annual rainfall	Continuous (annual rainfall)
Rainfall in 4 months prior to sampling	Continuous (recent rainfall)
GPS easting	Continuous (easting)
GPS northing	Continuous (northing)
Julian time	Continuous (Julian time)
Year of sampling	1) 2002 (2002) 2) 2003 (2003)
Soil type	1) Grey to black clay (grey clay) 2) Brown clay (brown clay)
Soil electrical conductivity	Continuous (EC)
Available soil phosphorus	Continuous (avail P)
Available soil sulphur	Continuous (avail S)
Soil organic carbon	Continuous (organic C)
Altitude	Continuous (altitude)
Litter cover	1) Low (≤15%) litter cover (low litter) 2) Moderate (16-74%) litter cover (moderate litter) 3) High (≥75%) litter cover (high litter)
Structural formation of the vegetation	1) Open grassland (open grassland) 2) Shrubby grassland (shrubby grassland) 3) Open grassy woodland (open woodland) 4) Grassy woodland with shrubs (trees and shrubs)

Unplanned, one degree of freedom contrasts were used to identify the effects of grazing regime, where necessary. Means of response variables are recorded in the text with their standard error (i.e. mean ± standard error).

The Pearson chi-square test was used to identify differences in the frequencies of species associated with the categorical variables, grazing, cultivation and flooding (Table 3.2). For 2 x 2 tables the Fisher's exact test was used, as this method is preferable over the chi-square test (Zar 1999). Species with less than five occurrences for each test were ignored, because low frequencies may be due to chance. All responses significant at the 5% level were reported. However, given the large number of species, and small sample sizes for the variable combinations, it is possible that some of the observed responses were due to chance (i.e. type 1 error).

All of the above univariate analyses were carried out using the software package GenStat (6^{th} edition). Detailed univariate analyses were not carried out in this chapter because of the size and complexity of the data set, and the lack of replication for sites that share the same management history and environmental variables. Even for comparisons between the stratified variables for grazing, cultivation and flooding, the influence of these variables can be confounded by other disturbance-related or environmental variables not taken into account. The complex interactions with other variables were not explored in this analysis.

3.3 Results

3.3.1 General floristics

A total of 364 plant taxa were recorded in 235 sites surveyed (including sites repeatedly sampled), of which 77% (280) were native and 23% (84) introduced (Appendix 3.2). Most species (345) were recorded within quadrats and 19 further species were recorded opportunistically by searching the area immediately outside each quadrat.

The 364 taxa belonged to 55 different plant families (Appendix 3.2). Families with the greatest number of taxa were Poaceae (97), Asteraceae (54), Fabaceae (35), Chenopodiaceae (30), Malvaceae (14) and Cyperaceae (nine). Forbs were the most frequently recorded life form, followed by grasses (Table 3.11).

Table 3.11. Number of taxa belonging to different life forms in the regional vegetation survey (*n* = 364)

Life form	Number of taxa (%)
Forb	178 (49%)
Grass	97 (27%)
Sub-shrub	30 (8%)
Tree	18 (5%)
Graminoid	21 (6%)
Shrub	17 (5%)
Other (e.g. Pteridophytes)	3 (1%)

Eleven taxa occurred in at least 50% of the 184 individual sites. In order of decreasing frequency, these were *Dichanthium sericeum, Panicum decompositum, Enteropogon acicularis, Solanum esuriale, Chloris truncata, Acacia farnesiana, Sclerolaena muricata* var. *villosa, Boerhavia dominii, Neptunia gracilis, Sida trichopoda* and **Rapistrum rugosum.* Two of these species may be considered introduced (*Acacia farnesiana* and **Rapistrum rugosum*) although the origins of *A. farnesiana* are debatable (Clarke *et al.* 1998). The most common *Astrebla* species was *A. elymoides*, occurring in 76 of the 184 sites, and ranked 21st in terms of sample frequency. Many taxa were recorded only once or twice in the survey: 19% (69) occurred only once and 10% (36) were recorded twice. The sample frequencies of all 364 taxa are shown in Appendix 3.2.

Dichanthium sericeum was the most prevalent species in terms of mean cover score. The ten most cover-abundant species are shown in Figure 3.3. While *Astrebla* spp. were not the most prevalent, they were an important component of the flora, with *A. elymoides* ranked 5th and *A. lappacea* ranked 16th, based on mean cover. When present at a site, these species were often one of the most dominant (Fig. 3.4). Of the sites surveyed on clay soils *Dichanthium sericeum* or *Astrebla* spp. were dominant at 71% of sites, and occurred at a further 18% of sites.

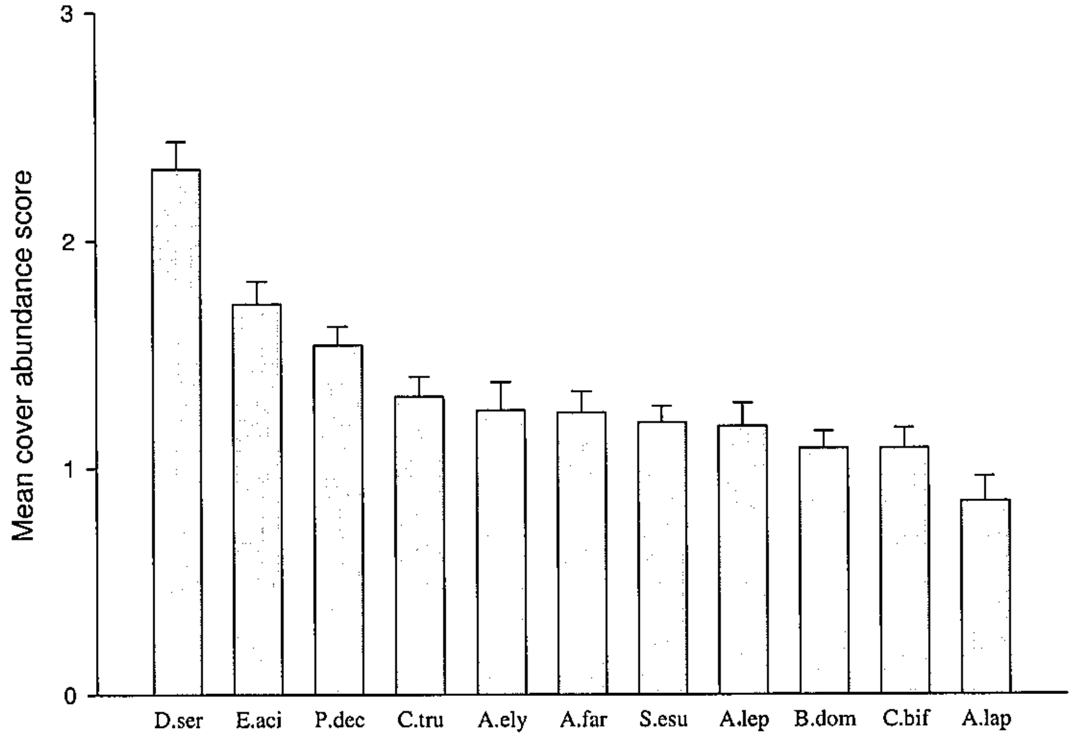


Fig. 3.3. Mean cover scores (+ SE) for the 10 most abundant taxa, and *Astrebla lappacea* (ranked 16th) in 184 sites. Species codes: D.ser, *Dichanthium sericeum*; E.aci, *Enteropogon acicularis*; P.dec, *Panicum decompositum*; C.tru, *Chloris truncata*; A.ely, *Astrebla elymoides*; A.far, *Acacia farnesiana*; S.esu, *Solanum esuriale*; A.lep, *Aristida leptopoda*; B.dom, *Boerhavia dominii*; C.bif, *Cyperus bifax*; and A.lap, *Astrebla lappacea.*

Total species richness ranged from 7 to 56 species per quadrat, with an average of 31 ± 0.64. Average native and introduced species richness was 26.3 ± 0.58 and 4.7 ± 0.20 per quadrat, respectively (Table 3.12). Sites were composed mostly of forbs (15.3 species per quadrat), followed by grasses (9.8 species per quadrat) and sub-shrubs (2.9 species per quadrat) (Table 3.12).

Four species of known conservation significance were recorded during this survey. *Desmodium campylocaulon* was the most common of these, occurring at 37 of the 184 sites sampled once only. This species is listed in the NSW *Threatened Species Conservation Act 1995* (*TSC Act*) as endangered. *Digitaria porrecta* was the next most common, occurring at 15 sites. This species is also listed as endangered in the *TSC Act* and the Commonwealth's *Endangered Species Protection Act 1992.* *Swainsona* *murrayana* is listed as vulnerable in NSW and nationally, and was recorded on three occasions (including repeatedly sampled sites) while *Phyllanthus maderaspatanus,* which is endangered in NSW, was recorded once. *Bothriochloa biloba,* a species that has recently been removed from the *TSC Act*, was also recorded in one quadrat.

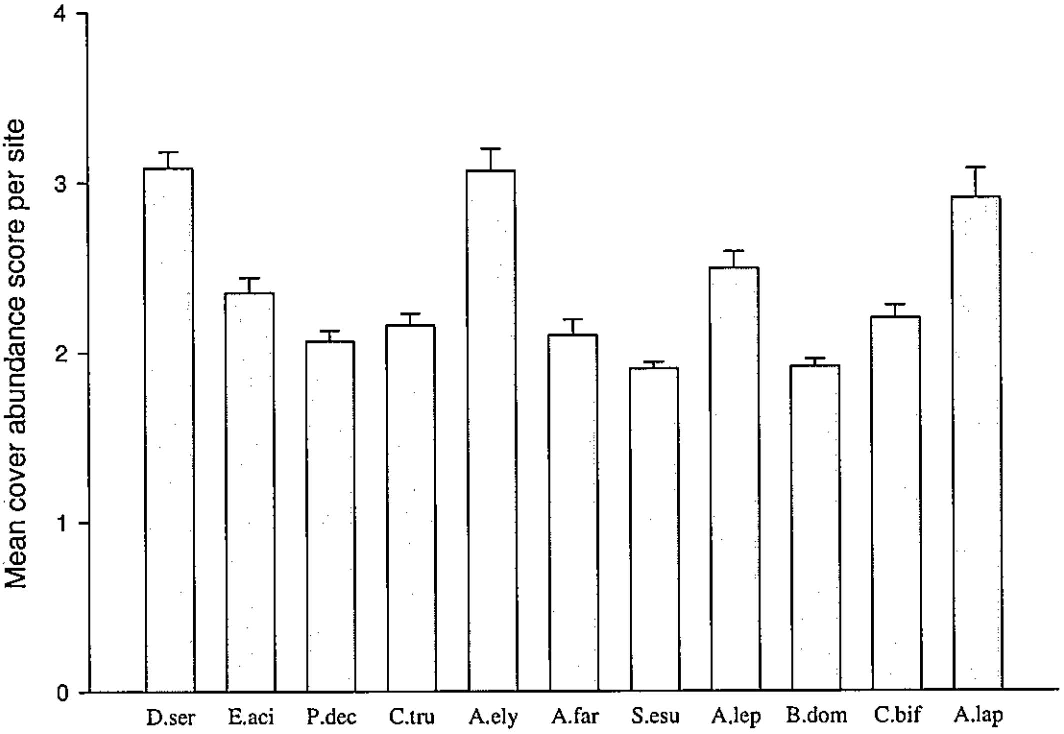


Fig. 3.4. Mean cover score per site (+ SE) considering only those sites in which the species occurred, for species with the highest mean cover scores. Species labels as for Figure 3.3, sample frequencies for the 11 species (left to right) were *n* = 138, 135, 138, 112, 76, 110, 116, 88, 105, 91, 55.

Table 3.12. Mean total, native, introduced and life form richness recorded at all sites surveyed (0.04 ha) during the regional survey (*n* = 235, including repeat samples at sites)

Species richness	Mean	SE
Total	31.0	0.64
Native	26.3	0.58
Introduced	4.7	0.20
Forb	15.3	0.39
Grass	9.8	0.26
Graminoid	1.5	0.08
Tree	0.2	0.03
Sub-shrub	2.9	0.11
Shrub	1.1	0.05

3.3.2 Variation in composition and richness over time

Species richness was similar for sites sampled in autumn 2002 and autumn 2003 (Table 3.13). There were, however, compositional differences between the 2 years (Fig. 3.5). In 2002, sampling was more concentrated towards the east of the study region (Fig. 3.1), due to lack of rainfall in the west. This included several sites on non-clay soils, at higher elevation than most other sites (Fig. 3.5). The most abundant taxa in 2002 and 2003 were similar, with six of the ten being most abundant in both years (Table 3.14). While *Dichanthium sericeum* was the most abundant species in both years, its mean cover abundance score declined considerably in 2003 (Table 3.14).

Table 3.13. Mean total, native, introduced and life form richness recorded at all regional sites surveyed in autumn 2002 (*n* = 98) and autumn 2003 (*n* = 103)

	2002		2003	
Species richness	Mean	SE	Mean	SE
Total	30.4	0.84	30.8	1.07
Native	26.3	0.83	26.1	0.94
Introduced	4.2	0.27	4.7	0.27
Forb	14.3	0.47	15.7	0.60
Grass	10.2	0.35	9.7	0.45
Sub-shrub	2.6	0.16	2.8	0.16
Graminoid	1.6	0.12	1.2	0.12
Shrub	1.0	0.08	1.0	0.08
Tree	0.3	0.06	0.2	0.04

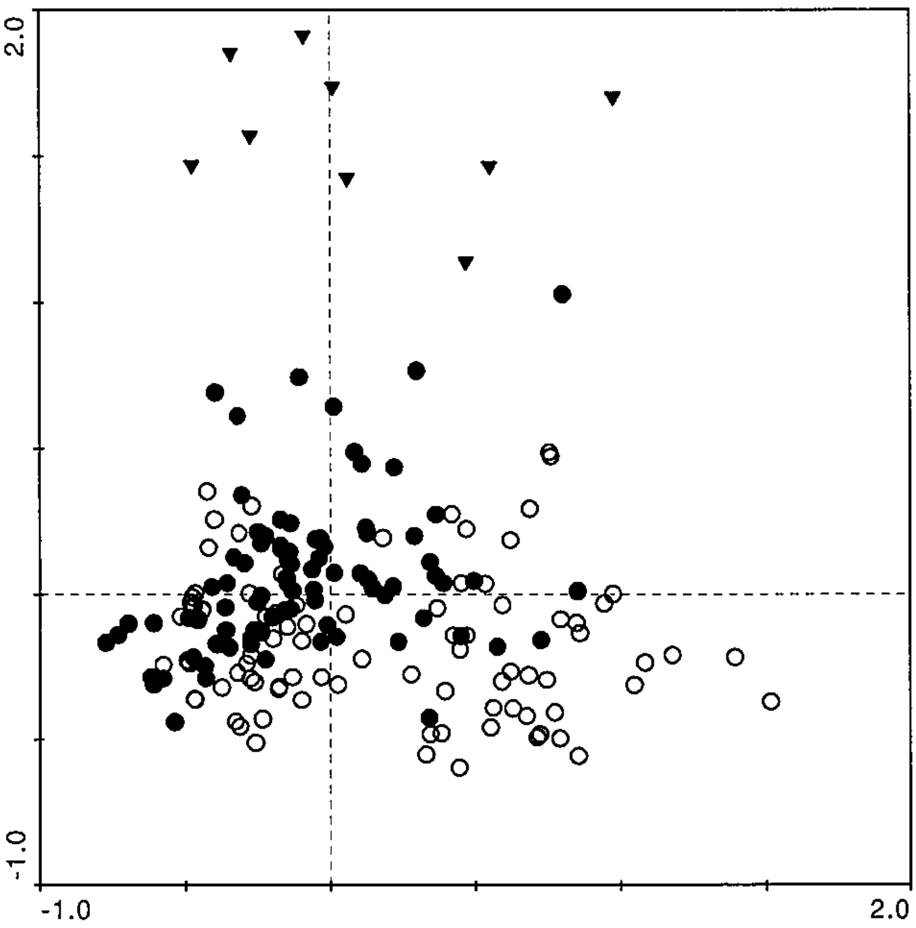


Fig. 3.5. Ordination (CCA) for species cover data, for all individual sites ($n = 184$) classified by year of sampling. Filled circles and down-triangles represent 2002 sites and non-filled circles 2003 sites. Down-triangles represent sites on non-clay soils.

Species richness varied greatly at the 17 sites that were sampled repeatedly (autumn 2002, spring 2002, autumn 2003 and spring 2003). Mean total species richness for these sites was lowest in spring 2002 (26.1 ± 1.56) and highest in spring 2003 (40.8 ± 1.80). Introduced species richness followed the same trend, but with a more pronounced increase in spring 2003 (Fig. 3.6). Native species richness followed a similar trend but peaked in autumn 2003 (Fig. 3.6).

The increase in introduced species richness in spring 2003 corresponded with an increase in forbs (Fig. 3.7). Species such as **Rapistrum rugosum, *Sonchus oleraceus, *Plantago cunninghamii, *Hedypnois rhagadioloides, Erodium crinitum* and **Medicago* spp. were more abundant or only occurred in spring. Grass species richness tended to decrease in spring (Fig. 3.7), which is not surprising given that most of the grass species (approximately 84%) were summer-growing species. Only six winter-growing grass species were recorded during spring surveys, and they were more frequent and abundant in 2003 (six species) than in 2002 (three species).

The changes in species richness corresponded with fluctuations in rainfall prior to sampling. There was a significant linear relationship between recent rainfall (i.e. rainfall in the 4 months prior to sampling) and total ($r^2 = 0.087$, $F_{1,183} = 17.51$, $P < 0.001$) and native species richness ($r^2 = 0.051$, $F_{1,183} = 9.76$, $P = 0.002$), over all sites sampled in the regional vegetation survey (n = 184). For the 17 repeated sites, rainfall was 178 ± 8.6 mm in the 4 months prior to autumn 2002, compared to 73 ± 5.2 mm prior to spring 2002. Rainfall was highest in the 4 months prior to autumn 2003 (246 ± 15.5 mm) but was still 143 ± 1.2 mm prior to spring 2003. Drought conditions (spring 2002) were associated with a considerable reduction in species richness (both native and introduced species). A 24% reduction in native species richness was recorded at TSR sites between autumn 2002 and spring 2002, and a 34% reduction in native species richness was recorded at ungrazed sites in the same period. Annual rainfall for Moree in 2002 was 277 mm, which is below half of the average annual rainfall (585 mm). Up until October 2003, however, rainfall was 381 mm at Moree, only 36 mm below average.

Table 3.14. Mean ± SE cover score (in parentheses) of the ten most abundant species during sampling in autumn 2002 (n = 98) and 2003 (n = 103)

2002 sampling	2003 sampling
Dichanthium sericeum (2.6 ± 0.15)	*Dichanthium sericeum* (2.0 ± 0.17)
Panicum decompositum (1.8 ± 0.10)	*Enteropogon acicularis* (1.9 ± 0.14)
Enteropogon acicularis (1.7 ± 0.13)	*Solanum esuriale* (1.4 ± 0.09)
Chloris truncata (1.5 ± 0.12)	*Astrebla elymoides* (1.4 ± 0.18)
Aristida leptopoda (1.3 ± 0.15)	*Panicum decompositum* (1.4 ± 0.11)
Acacia farnesiana (1.3 ± 0.12)	**Rapistrum rugosum* (1.3 ± 0.11)
Astrebla elymoides (1.2 ± 0.16)	*Acacia farnesiana* (1.3 ± 0.13)
Cyperus bifax (1.2 ± 0.12)	*Sclerolaena muricata* var. *villosa* (1.2 ± 0.10)
Neptunia gracilis (1.2 ± 0.10)	*Portulaca oleracea* (1.2 ± 0.09)
Boerhavia dominii (1.1 ± 0.10)	*Boerhavia dominii* (1.1 ± 0.10)

An ordination of species cover data showed that repeatedly sampled sites were not obviously grouped into separate sampling times and that site composition did not vary greatly between the four sampling periods (Appendix 3.3). The incidence and cover of dominant species did not change greatly between autumn 2002 and autumn 2003. The most abundant grass species (*Dichanthium sericeum, Enteropogon acicularis* and *Aristida leptopoda*) persisted through the dry conditions in 2002 (Table 3.15). However, certain species varied dramatically in cover abundance. For example, mean cover score for *Juncus usitatus* decreased greatly between spring 2002 and autumn 2003 (Table 3.15). There was a general increase in the abundance of winter-growing forbs (**Rapistrum rugosum, *Medicago polymorpha* and **Sonchus oleraceus*) after spring 2002, with highest abundances in spring 2003. These introduced species were not abundant in quadrats in spring 2002 due to low rainfall.

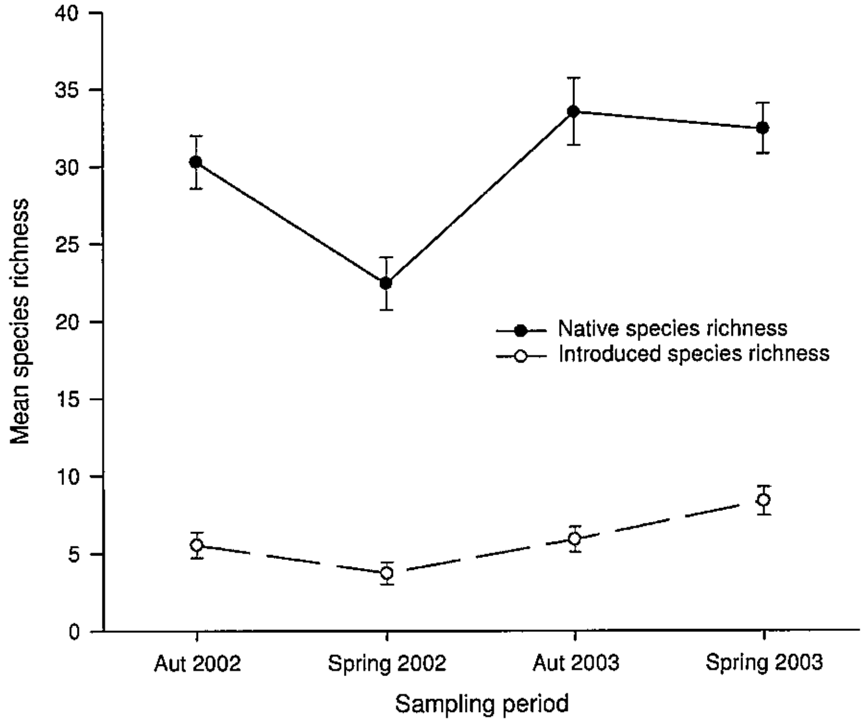


Fig. 3.6. Change in mean (± SE) native and introduced species richness over time, for sites revisited during the regional vegetation survey ($n = 17$).

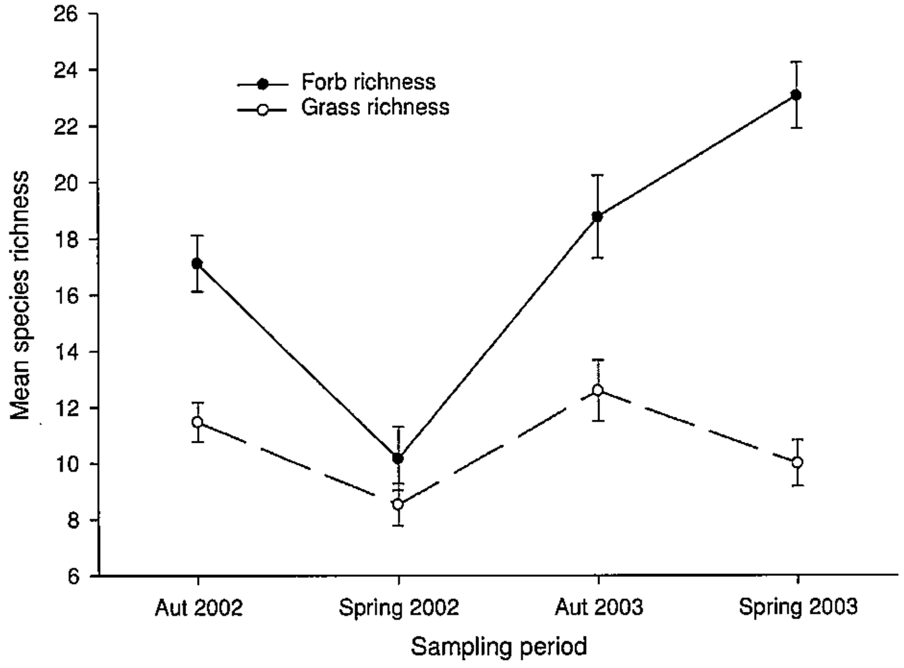


Fig. 3.7. Change in mean (± SE) forb and grass species richness, for sites revisited over time in the regional vegetation survey ($n = 17$).

3.3.3 General ordination output

Canonical correspondence analysis after forward selection to reduce the number of variables, produced eigenvalues for the first four ordination axes of 0.203, 0.161, 0.120 and 0.068. Soil type (non-clay soils) and altitude were the most important explanatory variables for predicting species composition, explaining 5.3% and 4.2% of the total variance respectively, and together display 27.9% of the total explainable variance. The strong influence of non-clay soils on species composition (Fig. 3.5) was taken into account when determining the influence of other variables on species composition, by only comparing sites located on clay soils. Fortunately, this was the majority of sites surveyed ($n = 175$).

Analysis of all individual sites on clay soils sampled in the autumn period of 2002 and 2003 ($n = 175$) explained approximately 32.4% of the variance in species cover data. Environmental variables accounted for 19.4% of the variance, while disturbance-related variables accounted for 10.5% of the variance, with a small percentage (2.5%) shared by both. After defining environmental variables as covariates, partial CCA produced eigenvalues of 0.051, 0.039, 0.036 and 0.031 for the first four ordination axes. Cultivation (number of consecutive cultivations) and stocking rate were the most important disturbance-related variables, explaining 1.9% and 1.4% of the total variance, respectively. Together these two variables accounted for 24.3% of the total explainable variance.

Table 3.15. Mean cover scores ± SE and rank (in parentheses) for the ten most cover-abundant species (italicised font) at each sampling time for the 17 repeated survey sites. Note: species ranked in order of abundance in autumn 2002 and alphabetical order when they have equal cover abundance

Species	Autumn 2002	Spring 2002	Autumn 2003	Spring 2003
Dichanthium sericeum	*2.8 ± 0.28 (1st)*	*2.3 ± 0.25 (1st)*	*2.1 ± 0.30 (2nd)*	*2.1 ± 0.30 (3rd)*
Acacia farnesiana	*2.2 ± 0.27 (2nd)*	*2.2 ± 0.27 (2nd)*	*1.9 ± 0.22(3rd)*	*1.9 ± 0.24(5th)*
Enteropogon acicularis	*2.1 ± 0.30 (3rd)*	*2.0 ± 0.19 (3rd)*	*2.4 ± 0.30 (1st)*	*2.5 ± 0.29 (2nd)*
Aristida leptopoda	*1.8 ± 0.36 (4th)*	*1.8 ± 0.37 (5th)*	*1.9 ± 0.41 (5th)*	*1.7 ± 0.41 (9th)*
Neptunia gracilis	*1.7 ± 0.17 (5th)*	0.9 ± 0.24 (16th)	1.1 ± 0.22 (17th)	0.7 ± 0.21 (31st)
Panicum decompositum	*1.7 ± 0.19 (6th)*	*1.9 ± 0.17 (4th)*	*1.9 ± 0.20 (4th)*	1.3 ± 0.28 (15th)
Verbena gaudichaudii	*1.5 ± 0.23 (7th)*	0.9 ± 0.21 (17th)	1.1 ± 0.20 (18th)	1.5 ± 0.19 (11th)
Boerhavia dominii	*1.4 ± 0.21 (8th)*	*1.4 ± 0.21 (6th)*	1.0 ± 0.23 (21st)	1.3 ± 0.21 (16th)
Eriochloa crebra	*1.4 ± 0.27 (9th)*	0.5 ± 0.21 (32nd)	0.9 ± 0.28 (29th)	0.5 ± 0.19 (40th)
Juncus usitatus	*1.4 ± 0.23 (10th)*	*1.2 ± 0.22 (9th)*	0.1 ± 0.06 (126th)	0.5 ± 0.19 (41st)
Solanum esuriale	*1.4 ± 0.23 (10th)*	0.6 ± 0.19 (27th)	*1.5 ± 0.23 (10th)*	0.9 ± 0.22 (25th)
Astrebla lappacea	1.3 ± 0.36 (13th)	*1.2 ± 0.36 (10th)*	1.2 ± 0.40 (12th)	0.9 ± 0.34 (24th)
Cyperus bifax	1.3 ± 0.31 (15th)	*1.4 ± 0.26 (7th)*	*1.5 ± 0.30 (9th)*	1.4 ± 0.27 (14th)
Astrebla elymoides	1.2 ± 0.35 (16th)	*1.2 ± 0.35 (8th)*	*1.8 ± 0.43 (6th)*	*1.6 ± 0.37 (10th)*
*Rapistrum rugosum	1.1 ± 0.22 (19th)	0.6 ± 0.21 (29th)	*1.7 ± 0.31 (8th)*	*2.6 ± 0.15 (1st)*
Cullen tenax	0.9 ± 0.22 (27th)	0.6 ± 0.21 (24th)	1.1 ± 0.24 (15th)	*1.9 ± 0.10 (4th)*
Convolvulus erubescens	0.8 ± 0.24 (32nd)	1.1 ± 0.22 (12th)	1.2 ± 0.23 (13th)	*1.8 ± 0.14 (8th)*
*Sonchus oleraceus	0.7 ± 0.23 (35th)	0.8 ± 0.23 (20th)	1.0 ± 0.27 (23rd)	*1.9 ± 0.22 (6th)*
*Medicago polymorpha	0.4 ± 0.19 (58th)	0.4 ± 0.19 (37th)	*1.8 ± 0.30 (7th)*	*1.8 ± 0.18 (7th)*

Forward selection of the species cover data identified 24 significant ($P < 0.05$) variable categories (Table 3.16). Nine of these were disturbance-related (Section 3.2.3). The first five variable categories in forward selection (i.e. with the highest conditional eigenvalues) were from environmental variables (Table 3.16).

Table 3.16. Forward selection results, showing conditional effects and marginal effects for significant ($P < 0.05$) variable categories in the CCA for species cover scores. Only sites on clay soils were analysed ($n = 175$). ME, marginal effects; CE, conditional effects; λ, eigenvalues. Note: a significance level estimate of $P = 0.002$ is the lowest achievable given the number of permutations

Variable category	ME λ	CE λ	F	P
brown clay	0.12	0.12	6.18	0.002
Julian time	0.09	0.09	4.52	0.002
altitude	0.10	0.08	4.22	0.002
organic C	0.09	0.07	3.92	0.002
northing	0.05	0.05	2.76	0.002
>5 yr consecutive cultivations	0.05	0.04	2.48	0.002
stocking rate	0.04	0.04	2.24	0.002
open woodland	0.04	0.04	2.05	0.002
TSR	0.04	0.03	1.85	0.002
recent rainfall	0.06	0.04	1.90	0.002
recent grazing	0.04	0.03	1.79	0.002
2002	0.10	0.03	1.73	0.004
shrubby grassland	0.04	0.03	1.72	0.002
low litter	0.07	0.03	1.61	0.002
high flood frequency	0.04	0.02	1.54	0.002
avail S	0.09	0.03	1.56	0.002
EC	0.11	0.03	1.65	0.002
high litter	0.05	0.02	1.55	0.002
not grazed in 5 yr	0.03	0.03	1.42	0.008
avail P	0.10	0.02	1.38	0.010
cultivated <5 years ago	0.03	0.03	1.34	0.016
annual rainfall	0.06	0.02	1.33	0.050
sheep grazed	0.02	0.02	1.30	0.020
not flooded	0.05	0.02	1.31	0.024

3.3.4 Disturbance-related determinants of composition and richness

Grazing

Grazing variables had an important influence on species composition, together accounting for 6.5% of the explained variance. Sites that had not recently been grazed (in the 2 years prior to sampling) were of similar composition to sites not grazed for 5-19 years and nature reserve sites (Fig. 3.8), because these three variable categories shared a number of sites (67% of not recently grazed sites were either nature reserve sites or other sites not grazed for 5-19 years). Sites for these three variable categories differed in composition to sites not grazed in 1 year and sites not grazed in ≥20 years (Fig. 3.8).

Across all sites on clay soils, total species richness was higher at ungrazed sites than at grazed sites (Appendix 3.4a; $F_{1,171} = 5.12$, $P = 0.025$), mostly due to higher forb richness at ungrazed sites (Appendix 3.4a; $F_{1,171} = 5.96$, $P = 0.016$). Sites not grazed for 5-19 years had higher total and native species richness than the other ungrazed sites (Table 3.17; $F_{1,171} = 6.22$, $P = 0.014$ and $F_{1,171} = 7.99$, $P = 0.005$, respectively). The four ungrazed, uncultivated, non-flooded sites at Kirramingly Nature Reserve had significantly greater native species richness than grazed, uncultivated, non-flooded sites over the region ($t = -3.38$, d.f. = 57, $P = 0.001$). These four sites at the Reserve had a mean native species richness of 43.5 ± 2.40, compared to 27.4 ± 1.12 at the grazed sites.

All cemetery sites were on non-clay soils and, hence, were removed from analysis because the absence of grazing at these sites was confounded with soil type. Unfortunately no ungrazed cemetery sites with grassland could be identified on the clay soils of the Moree Plains. After removal of cemetery sites only four sites not grazed in ≥20 years remained in the analysis. Contrasts identified that native species richness was marginally lower at these sites than at grazed sites ($F_{1,171} = 3.04$, $P = 0.083$). However, three of these sites were sampled on one property and the other on a nature reserve, and all four sites were in relatively close proximity (north-west of Moree) and had received low rainfall in the 4 months prior to sampling (76 mm for the three sites on private property). One of the sites not grazed in ≥20 years also had a history of cultivation and was in poor condition based on the grassland condition index (Section 3.2.5).

Sites in good condition were generally located to the left of the ordination space (Fig. 3.8), reflecting the higher total and native species richness (Fig. 3.9). Higher grass, forb, sub-shrub and shrub richness were associated with nature reserves (Fig. 3.9). Four sites in good condition were located at Kirramingly Nature Reserve (and, hence, had not been grazed within 5 years), two were located on TSRs, and 10 were on private properties. Sites in good condition on TSRs or privately owned properties often had lower stocking rates (3.2 ± 0.63 DSE/ha) than other grazed sites without a history of cultivation (4.8 ± 0.48 DSE/ha), although the difference was not statistically significant ($t = -1.08$, d.f. = 94, $P = 0.285$). Approximately 26% of all continuously grazed sites and 14% of periodically grazed sites were in poor condition. Of the 31 sites in poor condition, approximately 19% were on TSRs.

Table 3.17. Species richness for sites that had not been grazed for 1 year (NG1), 5-19 years (NG5) and ≥20 years (NG20), and sites that had been grazed (RG). Also shown are the probabilities from ANOVA (*F* statistic) comparing these groups of sites. Note: NA, sample sizes too small for Kruskal-Wallis ANOVA

	RG (n = 155)	NG1 (n = 6)	NG5 (n = 10)	NG20 (n = 4)		
Richness	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE	*F*	*P*
Total	29.3 ± 0.72	31.8 ± 3.32	38.5 ± 3.05	23.3 ± 2.39	3.64	0.014
Native	25.2 ± 0.67	26.0 ± 2.84	33.5 ± 3.21	17.8 ± 2.56	3.73	0.012
Introduced	4.1 ± 0.21	5.8 ± 0.98	5.0 ± 0.88	5.5 ± 1.04	1.67	0.175
Forb	14.4 ± 0.43	16.0 ± 1.00	19.5 ± 1.56	13.0 ± 2.27	2.80	0.042
Grass	9.5 ± 0.31	10.3 ± 1.63	12.8 ± 1.28	5.8 ± 1.55	3.59	0.015
Graminoid	1.3 ± 0.10	1.3 ± 0.49	1.4 ± 0.34	0.5 ± 0.50	NA	NA
Sub-shrub	2.5 ± 0.11	2.7 ± 1.12	3.5 ± 0.56	2.8 ± 0.85	NA	NA
Shrub	1.0 ± 0.06	1.0 ± 0.45	0.9 ± 0.28	0.8 ± 0.48	NA	NA

Stocking rate was identified as an important variable in explaining species composition (Table 3.16 and Fig. 3.8), although there was no significant relationship between stocking rate and native species richness ($r^2 = 0.0007$, $F_{1,179} = 0.12$, $P = 0.725$) or introduced species richness ($r^2 = 0.005$, $F_{1,179} = 0.89$, $P = 0.346$). Stocking rates varied considerably between sites, but were generally higher on TSRs than on privately owned properties (Table 3.18). Stocking rates for sites in the east of the study region (east of the Newell Highway) were significantly higher than those in the west ($t = 4.08$, d.f. = 94, $P < 0.001$). There were no significant differences in stocking rates between continuously and periodically grazed sites ($t = -0.33$, d.f. = 90, $P = 0.745$). Five out of the six sites grouped near the end of the stocking rate vector in Figure 3.8 were located on TSRs in the east of the study region and had very high stocking rates (≥16 DSE/ha). Partial CCA of species cover data recognised 14 species that were significantly correlated with increasing stocking rate, while two species were negatively correlated with this variable (Fig. 3.10). *Austrostipa verticillata*, *Digitaria brownii*, *Chrysopogon fallax*, *Sclerolaena birchii* and *Bothriochloa decipiens* were most strongly correlated with increasing stocking rate.

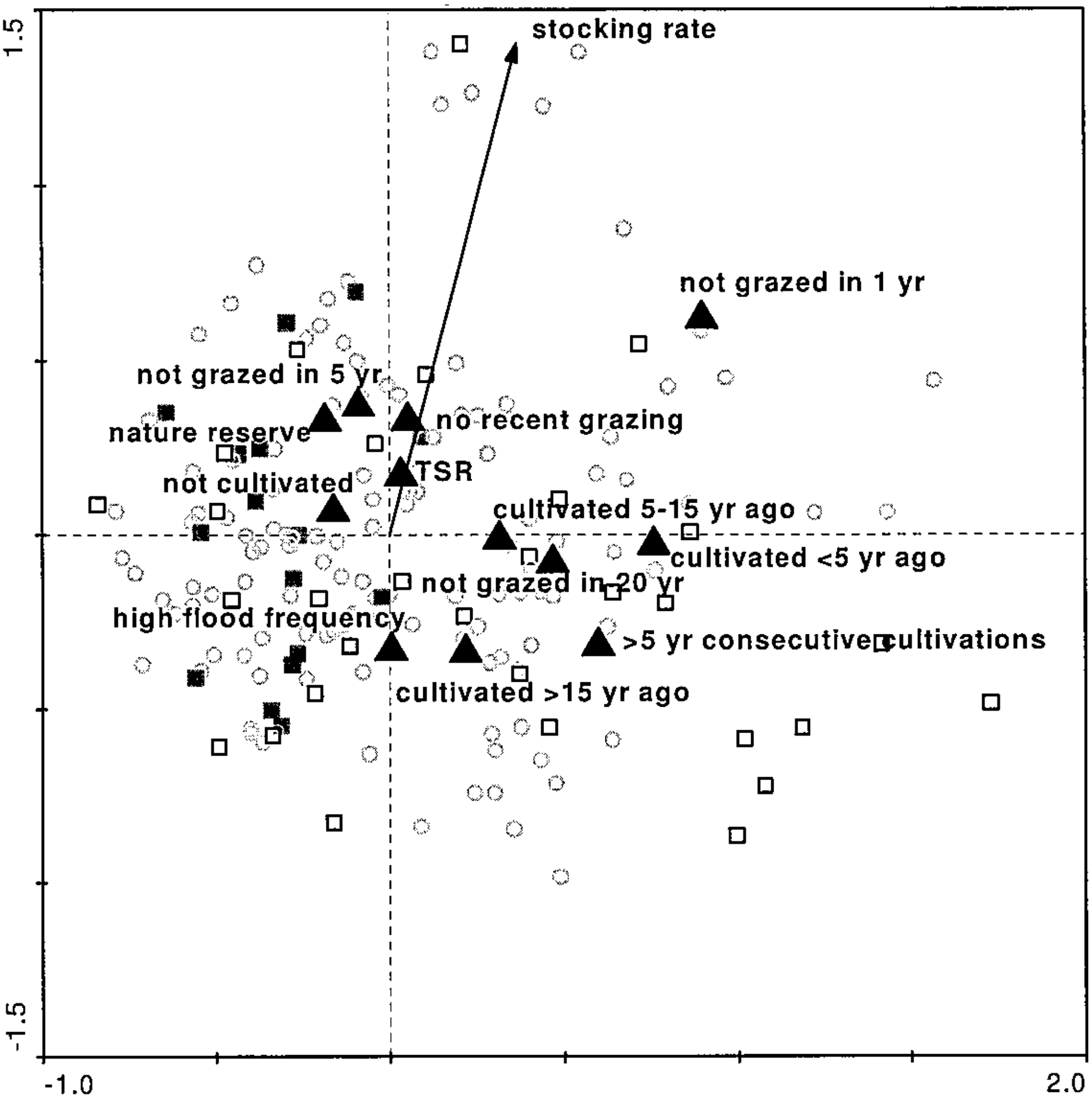


Fig. 3.8. Ordination biplot (partial CCA) for disturbance-related variables and individual sites sampled in autumn 2002 and 2003. Sites classified by grassland condition: filled squares, good condition; non-filled squares, poor condition; non-filled grey circles, moderate condition. Nominal variables categories represented by triangles and continuous variables by vectors.

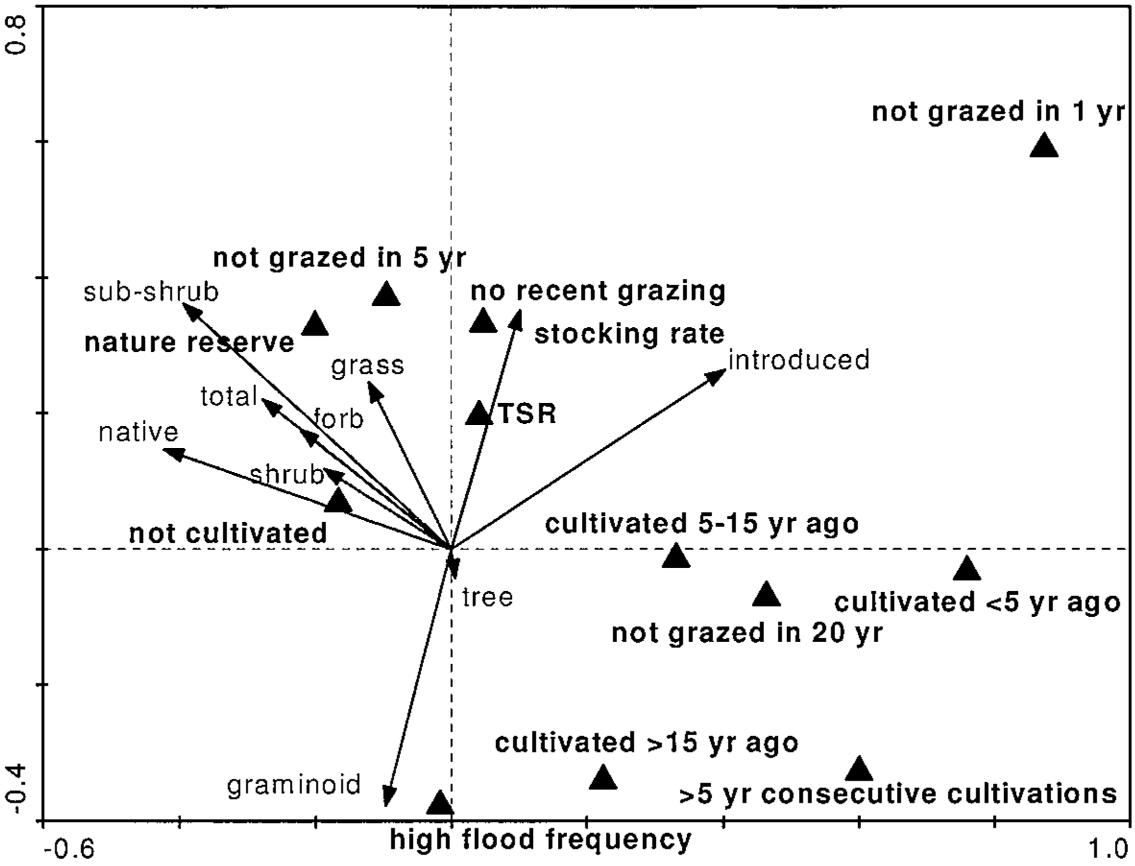


Fig. 3.9. Ordination (partial CCA) for disturbance-related variables showing trends in species richness (supplementary variables in grey font). Nominal variable categories represented by triangles and continuous variables by vectors.

Table 3.18. Variation in stocking rates between TSR sites and privately owned sites, and between the 2 years of sampling

Tenure and year	Min	Max	Mean	SE	*n*
TSR 2002	1.8	16.9	6.3	0.60	27
TSR 2003	0	16.9	8.1	2.00	13
Private 2002	0	13.9	3.4	0.44	57
Private 2003	0	12.4	2.4	0.27	71

Species richness varied greatly between grazed sites, suggesting that there are other factors influencing richness. When grazed sites that had never been cultivated were averaged across land tenure only minor differences were observed (Table 3.19). Native richness was marginally higher at privately owned sites than TSR sites (*P* = 0.092) while introduced richness was marginally higher at TSR sites (*P* = 0.073).

There were no statistically significant differences in total or native species richness between the different stratified grazing regimes (ungrazed, periodically grazed and continuously grazed)(Table 3.20). Only introduced species richness showed a significant response to grazing, with higher richness at ungrazed, uncultivated sites with a history of flooding (Table 3.20).

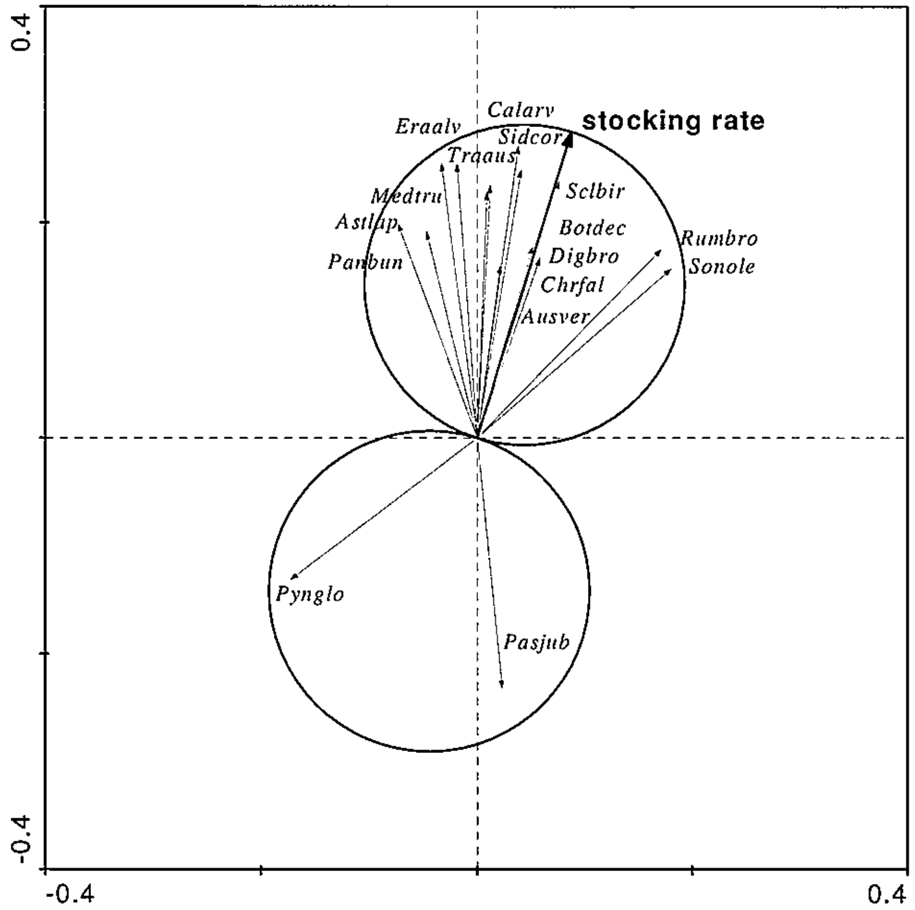


Fig. 3.10. A t-value biplot of significant correlations (positive and negative) between species and stocking rate. Only significant relationships are shown (i.e. where species arrows end within the Van Dobben Circles they have a corresponding t-value larger than two). Refer to Appendix 3.2 for species labels.

Table 3.19. Species richness at grazed sites on different land tenures. Also shown are the probabilities from ANOVA (*F* statistic) or nonparametric Mann-Whitney tests (*U* statistic) comparing the two land tenures

	Private (n = 63)		TSR (n = 39)			
Species richness	Mean	SE	Mean	SE	Statistic	*P*
Total	31.8	1.13	29.9	1.23	$F = 1.08$	0.302
Native	28.0	1.04	25.2	1.17	$F = 2.90$	0.092
Introduced	3.8	0.33	4.7	0.44	$F = 3.27$	0.073
Forb	15.9	0.68	14.0	0.72	$F = 2.65$	0.107
Grass	9.9	0.54	9.7	0.45	$F = 0.05$	0.826
Graminoid	1.3	0.13	1.4	0.21	$U = 1225$	0.984
Sub-shrub	2.9	0.17	2.9	0.24	$U = 1213$	0.913
Shrub	1.2	0.09	1.2	0.11	$U = 1226$	0.984

Table 3.20. Influence of grazing regime on species richness (mean ± SE (*n*)) for the different combinations of non-cultivated (NC), non-flooded (NF), flooded (F) and cultivated (C) sites. Also shown are the probabilities (and test statistics) from ANOVA (or nonparametric tests) comparing the different regimes. Kruskal-Wallis test (*H* statistic), Mann-Whitney test (*U* statistic). NA, sample sizes too small for comparisons

Total species richness

	Not grazed	Periodically grazed	Continuously grazed	*F*	*P*
NC, NF	39.5 ± 3.20 (8)	32.8 ± 1.30 (48)	29.9 ± 3.16 (7)	2.30	0.109
NC, F	30.3 ± 4.07 (7)	29.6 ± 1.50 (32)	28.5 ± 2.11 (13)	0.07	0.928
C, NF	NA	26.7 ± 2.30 (18)	24.8 ± 2.78 (4)	0.08	0.787
C, F	NA	26.8 ± 1.87 (20)	25.0 ± 2.57 (13)	0.41	0.527

Native species richness

	Not grazed	Periodically grazed	Continuously grazed	*F*	*P*
NC, NF	35.1 ± 3.48 (8)	27.5 ± 1.19 (48)	26.6 ± 3.48 (7)	2.53	0.088
NC, F	24.9 ± 3.36 (7)	26.7 ± 1.42 (32)	25.4 ± 2.25 (13)	0.22	0.805
C, NF	NA	21.3 ± 2.04 (18)	21.8 ± 2.87 (4)	0.05	0.819
C, F	NA	23.8 ± 1.67 (20)	20.7 ± 2.34 (13)	1.32	0.259

Table 3.20. (continued)

Introduced species richness

	Not grazed	Periodically grazed	Continuously grazed	*F*	*P*
NC, NF	4.4 ± 0.80 (8)	5.3 ± 0.43 (48)	3.3 ± 0.89 (7)	1.92	0.156
NC, F	5.4 ± 1.09 (7)	2.9 ± 0.32 (32)	3.2 ± 0.58 (13)	3.71	0.032
C, NF	NA	5.4 ± 0.61 (18)	3.0 ± 0.71 (4)	3.40	0.080
C, F	NA	3.0 ± 0.50 (20)	4.3 ± 0.62 (13)	3.10	0.088

Forb species richness

	Not grazed	Periodically grazed	Continuously grazed	*F*	*P*
NC, NF	20.5 ± 1.43 (8)	16.5 ± 0.79 (48)	16.0 ± 1.77 (7)	2.01	0.143
NC, F	14.6 ± 1.69 (7)	13.5 ± 0.87 (32)	13.9 ± 1.06 (13)	0.24	0.785
C, NF	NA	13.6 ± 1.58 (18)	12.8 ± 1.11 (4)	0.00	0.954
C, F	NA	13.0 ± 0.99 (20)	12.5 ± 1.68 (13)	0.17	0.684

Grass species richness

	Not grazed	Periodically grazed	Continuously grazed	*F*	*P*
NC, NF	13.1 ± 1.59 (8)	10.5 ± 0.54 (48)	9.1 ± 1.78 (7)	2.14	0.127
NC, F	9.0 ± 1.84 (7)	9.5 ± 0.63 (32)	8.2 ± 1.16 (13)	0.74	0.483
C, NF	NA	9.5 ± 0.90 (18)	8.5 ± 1.32 (4)	0.11	0.748
C, F	NA	8.9 ± 0.95 (20)	8.6 ± 1.00 (13)	0.00	0.972

Graminoid species richness

	Not grazed	Periodically grazed	Continuously grazed	Statistic	*P*
NC, NF	0.9 ± 0.30 (8)	1.04 ± 0.16 (48)	0.71 ± 0.29 (7)	$H = 0.31$	0.840
NC, F	1.6 ± 0.57 (7)	1.8 ± 0.20 (32)	1.9 ± 0.33 (13)	$H = 0.48$	0.777
C, NF	NA	0.6 ± 0.15 (18)	1.5 ± 0.65 (4)	$U = 19.0$	0.166
C, F	NA	2.0 ± 0.34 (20)	1.1 ± 0.18 (13)	$U = 81.5$	0.074

Table 3.20. (continued)

Sub-shrub species richness

	Not grazed	Periodically grazed	Continuously grazed	Statistic	*P*
NC, NF	4.1 ± 0.88 (8)	3.1 ± 0.21 (48)	2.6 ± 0.57 (7)	$H = 2.36$	0.292
NC, F	3.0 ± 0.49 (7)	2.7 ± 0.23 (32)	2.5 ± 0.31 (13)	$H = 0.78$	0.657
C, NF	NA	1.9 ± 0.37 (18)	1.8 ± 0.48 (4)	$U = 34.5$	0.902
C, F	NA	2.0 ± 0.23 (20)	1.9 ± 0.35 (13)	$U = 128.0$	0.957

Shrub species richness

	Not grazed	Periodically grazed	Continuously grazed	Statistic	*P*
NC, NF	0.75 ± 0.31 (8)	1.3 ± 0.10 (48)	1.0 ± 0.31(7)	$H = 2.71$	0.188
NC, F	1.4 ± 0.37 (7)	1.3 ± 0.11 (32)	1.2 ± 0.19 (13)	$H = 0.41$	0.776
C, NF	NA	0.8 ± 0.21 (18)	0.0 ± 0.00 (4)	$U = 16.0$	0.098
C, F	NA	0.5 ± 0.11 (20)	0.7 ± 0.31 (13)	$U = 125.0$	0.870

Individual species differed in frequency of occurrence between the differing grazing regimes (i.e. not grazed, periodically grazed or continuously grazed) at sites that had not been cultivated and were not flooded (Table 3.21). Species such as *Panicum queenslandicum, P. buncei, Themeda avenacea* and *Vittadinia cuneata* were more common at ungrazed sites. Other taxa such as *Sclerolaena muricata* var. *muricata* and **Tribulus terrestris* were more common at grazed sites. There were some species that did not respond consistently to grazing, with high percentage occurrence at both ungrazed sites and continuously grazed sites (e.g. *Boerhavia dominii* and *Sida trichopoda*).

Different species were influenced by grazing at sites that had flooded but had not been cultivated (Table 3.22). One native species (*Oxalis perennans*) and two introduced species, **Sonchus oleraceus* and **Urochloa panicoides*, were more common at such ungrazed sites. *Astrebla elymoides* and *Juncus aridicola* were more frequent at grazed sites in this category, with the latter species more common at continuously grazed sites (Table 3.22).

Two species (*Neptunia gracilis* and *Eleocharis pallens*) were more frequent at periodically grazed sites than at continuously grazed sites for those sites with a history of flooding and cultivation (Table 3.23).

Table 3.21. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (chi-square test, d.f. = 2) to grazing regime at sites that had not been cultivated or flooded

	Not grazed ($n = 8$)	Periodically grazed ($n = 48$)	Continuously grazed ($n = 7$)	χ^2	P
More frequent at ungrazed sites					
Einadia nutans subsp. *nutans*	63	17	29	8.0	0.018
Panicum buncei	63	23	14	6.1	0.047
Panicum queenslandicum	63	10	14	13.0	0.002
Ptilotus semilanatus	50	13	29	6.7	0.035
Themeda avenacea	50	6	29	12.0	0.002
Vittadinia cuneata	88	21	29	14.5	<0.001
More frequent at grazed sites					
Sclerolaena muricata var. *muricata*	25	42	86	6.2	0.045
**Tribulus terrestris*	0	13	43	6.1	0.047
More frequent at ungrazed and continuously grazed sites					
Boerhavia dominii	100	63	100	7.9	0.019
Sida trichopoda	100	52	86	8.6	0.014

Table 3.22. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (chi-square test, d.f. = 2) to grazing regime at sites that had flooded, but had not been cultivated

	Not grazed ($n = 7$)	Periodically grazed ($n = 32$)	Continuously grazed ($n = 13$)	χ^2	*P*
More frequent at ungrazed sites					
Oxalis perennans	100	44	69	8.39	0.015
**Sonchus oleraceus*	57	16	8	7.86	0.020
**Urochloa panicoides*	57	16	15	6.28	0.043
More frequent at grazed sites					
Astrebla elymoides	14	66	46	6.50	0.039
Juncus aridicola	0	6	31	6.50	0.039
More frequent at ungrazed and continuously grazed sites					
**Medicago polymorpha*	29	13	46	6.04	0.049

Table 3.23. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (Fisher's exact test, d.f. =1) to grazing regime at sites with a history of flooding and cultivation

	Periodically grazed ($n = 20$)	Continuously grazed ($n = 13$)	*P*
More frequent at periodically grazed sites			
Eleocharis pallens	35	0	0.027
Neptunia gracilis	75	23	0.005

Cultivation

Species composition was also influenced by cultivation variables, which accounted for 2.7% of the explained variance. Sites with a history of cultivation were generally positioned towards the right of the ordination diagram (Fig. 3.8). Time since last cultivation had some influence on composition, with sites cultivated within 5 years of sampling differing greatly from those that had not been cultivated (Fig. 3.8). Sites not cultivated within 15 years of sampling also differed in composition to uncultivated sites (Fig. 3.8). Some species associated positively and negatively with cultivated sites (in particular, sites cultivated within 5 years of sampling) are shown in Figure 3.11.

Across all sites on clay soils, uncultivated sites had higher total and native species richness (Fig. 3.9; Appendix 3.4b; $F_{1,169} = 12.59$, $P < 0.001$ and $F_{1,169} = 14.77$, $P < 0.001$, respectively). This was mainly because of higher forb, sub-shrub and shrub richness at uncultivated sites (Fig. 3.9; Appendix 3.4b). The effect of cultivation at periodically grazed sites that had not flooded was most obvious, with significantly higher total and native species richness at uncultivated sites (Table 3.24). Shrub, sub-shrub and forb species richness followed the same trend. Both shrub and sub-shrub richness were significantly ($P < 0.05$) or marginally ($P < 0.10$) higher at uncultivated sites with a history of periodic grazing, irrespective of the occurrence of flooding (Table 3.24).

Cultivation was associated with sites in poor condition (Fig. 3.8). In fact, 57% of sites in poor condition had a history of cultivation, and most of those (71%) were cultivated for >5 years consecutively.

Reductions in species richness due to cultivation at continuously grazed sites were less obvious (Table 3.24). Graminoid richness was the only statistically significant ($P < 0.05$) reduction associated with cultivation, at continuously grazed sites with a history of flooding.

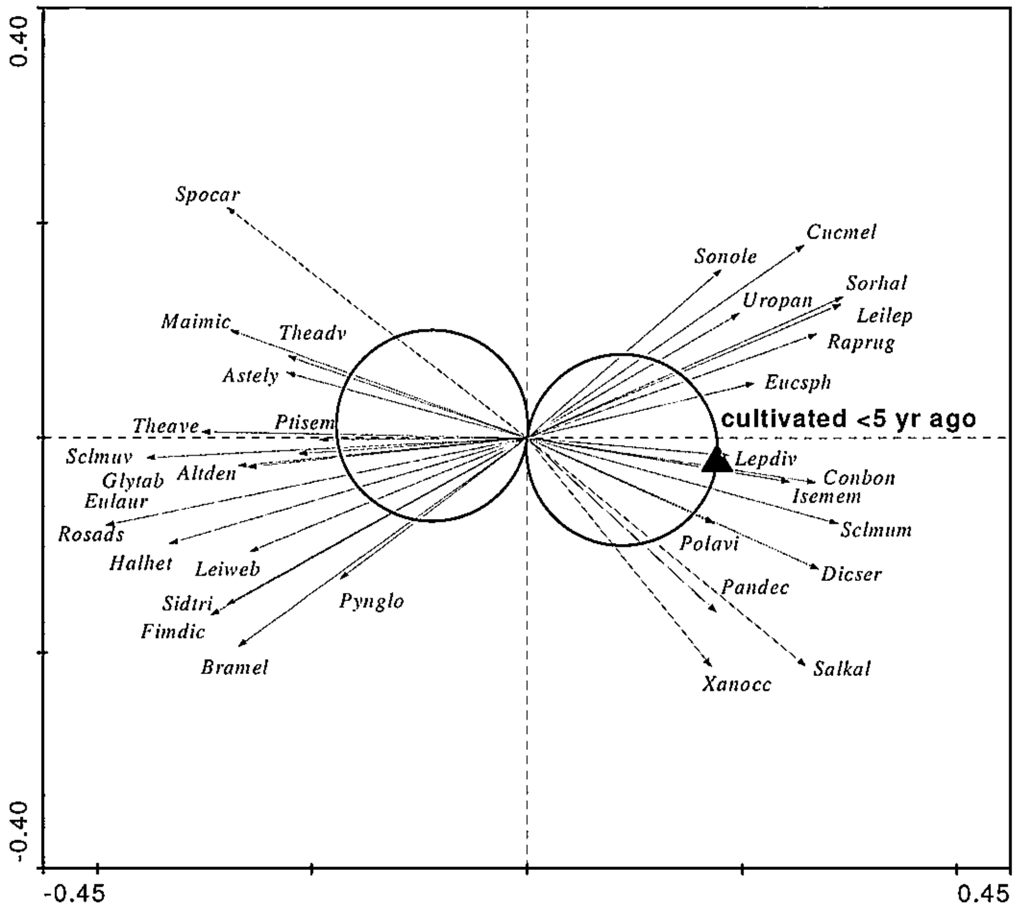


Fig. 3.11. A t-value biplot showing species associated with sites cultivated within 5 years of sampling. Species were not significantly associated with this variable category, as species arrows do not end within the Van Dobben Circles (corresponding t-values are less than two). Refer to Appendix 3.2 for species labels.

Differences in species richness were still evident for sites that had not been cultivated for at least 15 years prior to sampling, when compared to uncultivated sites (Table 3.25). Native species richness and shrub species richness were significantly lower at cultivated sites (Table 3.25), while total species richness, forb species richness and sub-shrub richness were marginally lower at these sites (Table 3.25). There was not a strong relationship between time since last cultivation and native species richness ($r^2 = 0.002$, $F_{1,61} = 0.14$, $P = 0.711$; Fig. 3.12). Interestingly, there were two sites with high native species richness (>35 species) within 5 years of the last cultivation. Both sites, however, had been conservatively grazed in the lead up to sampling and were only cultivated for 4 years.

Sites cultivated for greater than 5 years differed in composition to uncultivated sites (Table 3.16; Fig. 3.8). However, sites cultivated consecutively for greater than 5 years did not have significantly lower species richness than those that had received less than 5 years of consecutive cultivations (Appendix 3.5).

Table 3.24. Effect of cultivation on species richness (mean ± SE (*n*)) for the different combinations of ungrazed (NG), periodically grazed (PG), continuously grazed (CG), non-flooded (NF) and flooded (F) sites. Also shown are the probabilities (and test statistics) from ANOVA or the Mann-Whitney test (*U* statistic) comparing the two groups of sites. NA, sample sizes too small for comparisons

Total species richness

	Not cultivated	Cultivated	*F*	*P*
NG, NF	39.5 ± 3.20 (8)	NA	-	-
NG, F	30.3 ± 4.07 (7)	NA	-	-
PG, NF	32.8 ± 1.30 (48)	26.7 ± 2.30 (18)	6.19	0.015
PG, F	29.6 ± 1.50 (32)	26.8 ± 1.87 (20)	1.31	0.258
CG, NF	29.9 ± 3.16 (7)	24.8 ± 2.78 (4)	1.10	0.322
CG, F	28.5 ± 2.11 (13)	25.0 ± 2.57 (13)	1.30	0.266

Native species richness

	Not cultivated	Cultivated	*F*	*P*
NG, NF	35.1 ± 3.48 (8)	NA	-	-
NG, F	24.9 ± 3.36 (7)	NA	-	-
PG, NF	27.5 ± 1.19 (48)	21.3 ± 2.04 (18)	7.75	0.007
PG, F	26.7 ± 1.42 (32)	23.8 ± 1.67 (20)	1.62	0.209
CG, NF	26.6 ± 3.48 (7)	21.8 ± 2.87 (4)	0.80	0.395
CG, F	25.4 ± 2.25 (13)	20.7 ± 2.34 (13)	2.23	0.148

Table 3.24. (continued)

Introduced species richness

	Not cultivated	Cultivated	*F*	*P*
NG, NF	4.4 ± 0.80 (8)	NA	-	-
NG, F	5.4 ± 1.09 (7)	NA	-	-
PG, NF	5.3 ± 0.43 (48)	5.4 ± 0.61 (18)	0.09	0.768
PG, F	2.9 ± 0.32 (32)	3.0 ± 0.50 (20)	0.01	0.915
CG, NF	3.3 ± 0.89 (7)	3.0 ± 0.71 (4)	0.01	0.926
CG, F	3.2 ± 0.58 (13)	4.3 ± 0.62 (13)	1.93	0.177

Forb species richness

	Not cultivated	Cultivated	*F*	*P*
NG, NF	20.5 ± 1.43 (8)	NA	-	-
NG, F	14.6 ± 1.69 (7)	NA	-	-
PG, NF	16.5 ± 0.79 (48)	13.6 ± 1.58 (18)	3.65	0.061
PG, F	13.5 ± 0.87 (32)	13.0 ± 0.99 (20)	0.12	0.729
CG, NF	16.0 ± 1.77 (7)	12.8 ± 1.11 (4)	1.41	0.266
CG, F	13.9 ± 1.06 (13)	12.5 ± 1.68 (13)	0.95	0.339

Grass species richness

	Not cultivated	Cultivated	*F*	*P*
NG, NF	13.1 ± 1.59 (8)	NA	-	-
NG, F	9.0 ± 1.84 (7)	NA	-	-
PG, NF	10.5 ± 0.54 (48)	9.5 ± 0.90 (18)	1.11	0.297
PG, F	9.5 ± 0.63 (32)	8.9 ± 0.95 (20)	0.64	0.428
CG, NF	9.1 ± 1.78 (7)	8.5 ± 1.32 (4)	0.00	0.966
CG, F	8.2 ± 1.16 (13)	8.6 ± 1.00 (13)	0.14	0.715

Graminoid species richness

	Not cultivated	Cultivated	*U*	*P*
NG, NF	0.9 ± 0.30 (8)	NA	-	-
NG, F	1.6 ± 0.57 (7)	NA	-	-
PG, NF	1.04 ± 0.16 (48)	0.6 ± 0.15 (18)	336.0	0.171
PG, F	1.8 ± 0.20 (32)	2.0 ± 0.34 (20)	319.0	0.993
CG, NF	0.71 ± 0.29 (7)	1.5 ± 0.65 (4)	8.5	0.315
CG, F	1.9 ± 0.33 (13)	1.1 ± 0.18 (13)	45.5	0.044

Table 3.24. (continued)

Sub-shrub species richness

	Not cultivated	Cultivated	*U*	*P*
NG, NF	4.1 ± 0.88 (8)	NA	-	-
NG, F	3.0 ± 0.49 (7)	NA	-	-
PG, NF	3.1 ± 0.21 (48)	1.9 ± 0.37 (18)	253.3	0.009
PG, F	2.7 ± 0.23 (32)	2.0 ± 0.23 (20)	225.5	0.075
CG, NF	2.6 ± 0.57 (7)	1.8 ± 0.48 (4)	9.5	0.412
CG, F	2.5 ± 0.31 (13)	1.9 ± 0.35 (13)	65.0	0.336

Shrub species richness

	Not cultivated	Cultivated	*U*	*P*
NG, NF	0.75 ± 0.31 (8)	NA	-	-
NG, F	1.4 ± 0.37 (7)	NA	-	-
PG, NF	1.3 ± 0.10 (48)	0.8 ± 0.21 (18)	274.5	0.022
PG, F	1.3 ± 0.11 (32)	0.5 ± 0.11 (20)	130.0	<0.001
CG, NF	1.0 ± 0.31(7)	0.0 ± 0.00 (4)	4.0	0.073
CG, F	1.2 ± 0.19 (13)	0.7 ± 0.31 (13)	50.5	0.081

Table 3.25. Differences in species richness (detected using ANOVA or the Mann-Whitney test (*U* statistic)) between uncultivated sites and sites last cultivated >15 years prior to sampling

	Not cultivated ($n = 113$)		Cultivated >15 years ago ($n = 25$)			
Species richness	Mean	SE	Mean	SE	Statistic	*P*
Total	31.5	0.84	27.8	1.56	$F = 3.48$	0.064
Native	27.3	0.79	23.3	1.40	$F = 4.65$	0.033
Introduced	4.2	0.25	4.5	0.61	$F = 0.07$	0.798
Forb	15.5	0.48	13.7	1.00	$F = 2.78$	0.098
Grass	9.9	0.39	9.4	0.76	$F = 0.36$	0.550
Graminoid	1.4	0.11	1.6	0.28	$U = 1285.0$	0.485
Sub-shrub	3.0	0.14	2.3	0.17	$U = 1109.0$	0.094
Shrub	1.2	0.07	0.4	0.10	$U = 636.0$	<0.001

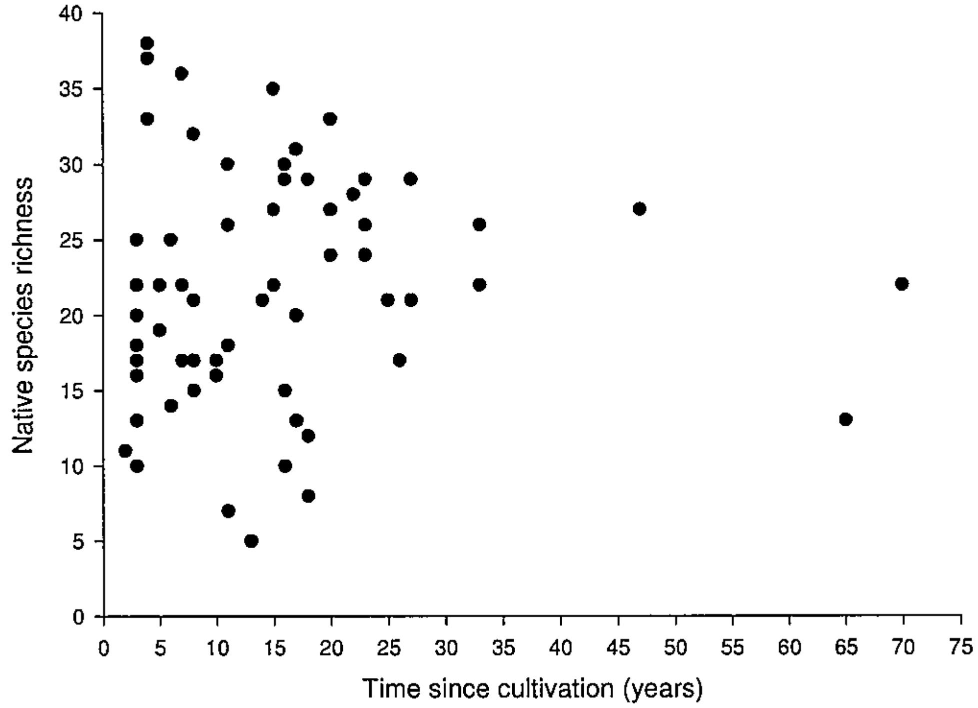


Fig. 3.12. Relationship between time since cultivation and native species richness for all sites with a history of cultivation.

Frequencies of individual species were influenced by cultivation at periodically and continuously grazed sites. At cultivated sites that were periodically grazed and not flooded, *Astrebla lappacea, Maireana decalvens* and *Solanum esuriale* were significantly less frequent (Table 3.26). The native sub-shrub, *Maireana decalvens*, was not present at any of the cultivated sites in this category. One introduced species, **Conyza bonariensis*, was also more common at uncultivated sites. One introduced forb, **Rapistrum rugosum*, and one native forb, *Rhynchosia minima*, were more frequent at previously cultivated sites, with no occurrences at uncultivated sites in this category. At periodically grazed sites that were flooded, a different group of species responded to cultivation (Table 3.27). Four native taxa, *Astrebla elymoides, Solanum esuriale, Verbena gaudichaudii* and *Sclerolaena muricata* var. *villosa*, were significantly less frequent at cultivated sites. *Acacia farnesiana* was also more common at such sites without a history of cultivation. Only one introduced forb, **Medicago polymorpha*, was significantly more common at these cultivated sites (Table 3.27).

At sites that were continuously grazed and not flooded, two native forb species (*Boerhavia dominii* and *Sida trichopoda*) were more common at uncultivated sites (Table 3.28). Another native forb (*Neptunia gracilis*) was also more frequent at uncultivated, continuously grazed sites that were subject to flooding. *Salsola kali* var. *kali* responded positively to cultivation at continuously grazed sites, and was not present at any uncultivated sites in this category (Table 3.28). *Sclerolaena muricata* var. *muricata* was more abundant at cultivated sites (mean cover abundance score of 1.1 ± 0.15 compared to 0.7 ± 0.08 at uncultivated sites), but its occurrence at uncultivated sites prevented recognition of a significant difference in frequency of occurrence.

Table 3.26. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (Fisher's exact test, d.f. = 1) to cultivation at sites that were periodically grazed and not flooded

	Cultivated ($n = 18$)	Not cultivated ($n = 48$)	P
More frequent at uncultivated sites			
Astrebla lappacea	11	48	0.009
**Conyza bonariensis*	0	23	0.028
Maireana decalvens	0	23	0.028
Solanum esuriale	44	77	0.017
More frequent at cultivated sites			
**Rapistrum rugosum*	78	0	<0.001
Rhynchosia minima	28	0	<0.001

The occurrence of past cultivation had an important influence on some soil variables. Available sulphur was significantly lower at cultivated sites, as was electrical conductivity and organic carbon (Table 3.29). Species cover scores that were correlated positively and negatively with topsoil organic carbon are shown in Figure 3.13. Five species (*Leptochloa divaricatissima, Austrostipa aristiglumis, *Conyza bonariensis, Verbena gaudichaudii* and *Boerhavia dominii*) were significantly correlated with increased organic carbon and eight species were negatively correlated (Fig. 3.13).

Table 3.27. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (Fisher's exact test, d.f. = 1) to cultivation at sites that were periodically grazed and subject to flooding

	Cultivated ($n = 20$)	Not cultivated ($n = 32$)	*P*
More frequent at uncultivated sites			
Acacia farnesiana	40	91	<0.001
Astrebla elymoides	35	66	0.046
Sclerolaena muricata var. *villosa*	45	78	0.019
Solanum esuriale	35	78	0.003
Verbena gaudichaudii	15	53	0.008
More frequent at cultivated sites			
**Medicago polymorpha*	50	13	0.005

Table 3.28. Frequency of occurrence (%) of species responding significantly ($P < 0.05$) (Fisher's exact test, d.f. = 1) to cultivation at sites that were continuously grazed

Not flooded

	Cultivated ($n = 4$)	Not cultivated ($n = 7$)	*P*
More frequent at uncultivated sites			
Boerhavia dominii	25	100	0.024
Sida trichopoda	0	86	0.015

Flooded

	Cultivated ($n = 13$)	Not cultivated ($n = 13$)	*P*
More frequent at uncultivated sites			
Neptunia gracilis	23	69	0.047
More frequent at cultivated sites			
Salsola kali var. *kali*	38	0	0.039

Table 3.29. The effects of cultivation on topsoil chemistry. Probabilities (and *F* statistic) are from ANOVA comparing cultivated and uncultivated sites

	Not cultivated ($n = 113$)		Cultivated ($n = 57$)			
Variable	Mean	SE	Mean	SE	*F*	*P*
Available P (mg/kg)	8.5	0.85	7.1	1.04	0.33	0.567
Available S (mg/kg)	5.8	0.34	4.7	0.49	9.21	0.003
Organic C (%)	5.4	0.13	4.9	0.17	4.54	0.035
EC (μS/cm)	81.4	4.62	65.5	4.76	6.40	0.012
pH_{H_2O}	7.1	0.05	7.2	0.06	1.33	0.251

General impacts of management disturbances

Sites in poor condition, with lower native species richness, tended to have histories of heavy grazing, recent cultivation, or were affected by the drought (usually with the combined influence of grazing or cultivation). Introduced species that were often present at such sites included **Tribulus terrestris, *Urochloa panicoides, *Rapistrum rugosum, *Physalis lanceifolia* and **Xanthium occidentale.* There were also several introduced species that occurred at lesser frequencies (and, hence, were not detected during analysis) in disturbed areas, such as **Cucumis myriocarpus, *Polygonum aviculare, *Gomphrena celosioides* and **Lactuca* spp. Native species were also frequent at such sites, including *Salsola kali* var. *kali, Echinochloa colona, Sclerolaena muricata* var. *muricata, Calotis scabiosifolia, Chloris truncata, Portulaca oleracea, Convolvulus erubescens, Rhynchosia minima, Sida trichopoda, Solanum esuriale* and *Boerhavia dominii*, and the less frequent *Sclerolaena birchii, Haloragis aspera, Polymeria pusilla* and *Ipomoea lonchophylla.* Obviously not all of these species would be present at any given disturbed site, however the presence of some of them was very likely.

Several native species were more commonly abundant at sites that had not been subject to frequent management-related disturbances, for example, *Panicum queenslandicum, Ptilotus semilanatus, Themeda avenacea, Vittadinia cuneata, Astrebla lappacea, Neptunia gracilis* and *Pycnosorus globosus.*

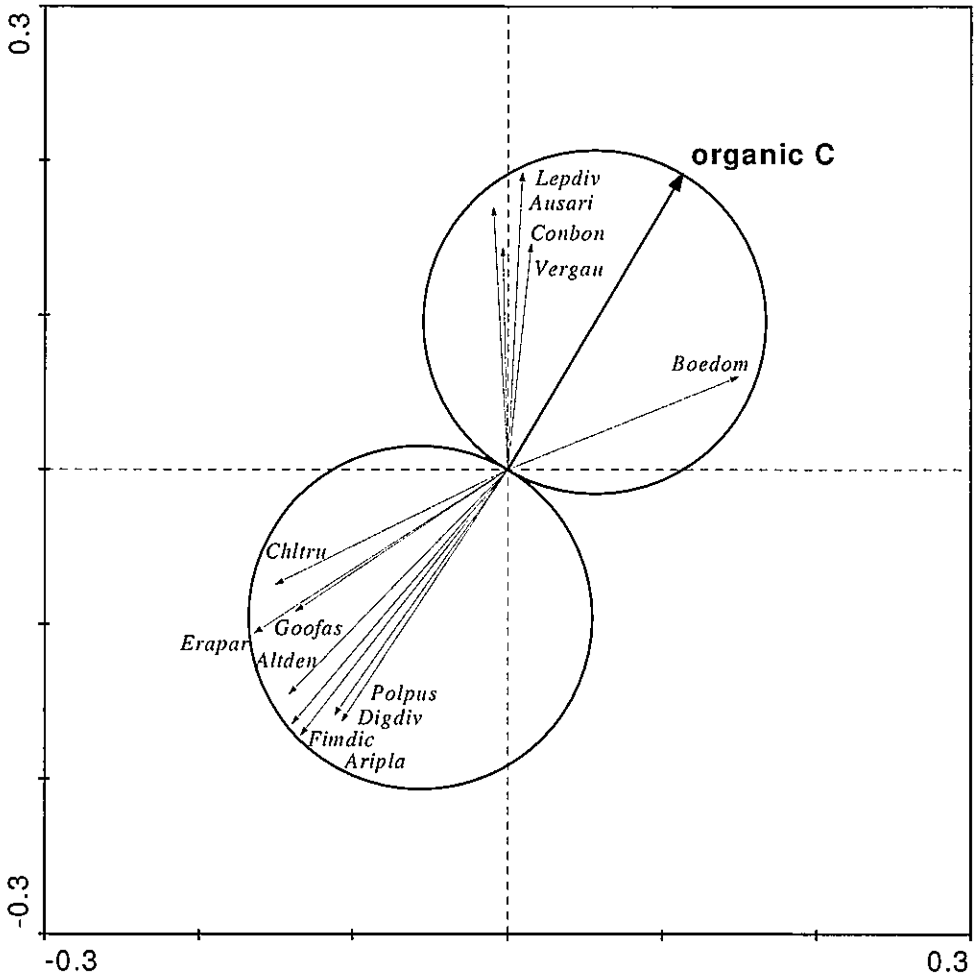


Fig. 3.13. A t-value biplot of the significant correlations (positive and negative) between species and topsoil organic carbon. Only significant relationships are shown (i.e. where species arrows end within the Van Dobben Circles they have a corresponding t-value larger than two). Refer to Appendix 3.2 for species labels.

Flooding

Frequency of flooding was identified using forward selection as having some influence on the composition of these grasslands (Table 3.16). Across all sites on clay soils, total, introduced, forb and grass species richness were significantly higher at sites were not been flooded, while graminoid richness was higher at flooded sites (Appendix 3.4c). Stratified comparisons recognised only a marginal influence of flooding on total species richness at sites that were ungrazed and uncultivated (Table 3.30). At such sites, native species richness was higher at sites that were not flooded (Table 3.30).

The occurrence of flooding had a significant effect on introduced species richness at sites that were periodically grazed and had not been cultivated, with higher introduced species richness at non-flooded sites (Table 3.30). Forb species richness was also negatively influenced by flooding, while graminoid richness responded positively (Table 3.30). For periodically grazed sites with a history of cultivation, introduced species richness was again higher at non-flooded sites, while graminoid richness was again higher at flooded sites (Table 3.30). At continuously grazed sites with no cultivation history, only graminoid richness was significantly higher at flooded sites (Table 3.30).

Partial CCA identified no species that were significantly correlated with the occurrence of flooding. However, some species were associated positively and negatively with this variable (Fig. 3.14).

Frequencies of individual species responded mostly to flooding at periodically grazed sites that had not been cultivated (Table 3.31). Only two native species responded positively to flooding at such sites. One of these species was a graminoid (*Cyperus bifax*) and the other a grass (*Paspalidium jubiflorum*). Six species were significantly less common at flooded sites in this category, including four introduced forbs (**Malvastrum americanum, *Medicago polymorpha, *M. truncatula,* and **Rapistrum rugosum*). This may explain the significantly higher introduced and forb species richness at such sites. Of the natives that responded negatively to flooding, one was a sub-shrub (*Maireana microphylla*) and the other a grass (*Paspalidium constrictum*). The threatened grass, *Digitaria porrecta*, was not present at any of the flooded sites in this category, although its absence was not statistically significant ($P = 0.080$).

At periodically grazed, cultivated sites, fewer species responded to flooding (Table 3.32). As recognised for uncultivated sites, *Cyperus bifax* increased in frequency while **Rapistrum rugosum* decreased in frequency at flooded sites. One other native graminoid (*Eleocharis pallens*) was also significantly favoured by flooding at such sites. At continuously grazed sites, only the native grass, *Astrebla lappacea,* was significantly more common at non-flooded sites (71% frequency) than at flooded sites (8% frequency) ($P = 0.007$). *Sida trichopoda* (a native forb) was also more common at non-flooded sites in areas that were not grazed ($P = 0.007$). This species was present at all eight non-flooded sites that had not been grazed or cultivated, and was only recorded at 29% of the flooded sites in this category.

Table 3.30. The effect of flooding on species richness (mean ± SE (*n*)) for the different combinations of ungrazed (NG), periodically grazed (PG), continuously grazed (CG), uncultivated (NC) and cultivated (C) sites. Also shown are the probabilities (and test statistics) from ANOVA or the Mann-Whitney test (*U* statistic) comparing the two groups of sites

Total species richness

	Not flooded	Flooded	*F*	*P*
NG, NC	39.5 ± 3.20 (8)	30.3 ± 4.07 (7)	3.45	0.086
PG, NC	32.8 ± 1.30 (48)	29.6 ± 1.50 (32)	2.47	0.120
PG, C	26.7 ± 2.30 (18)	26.8 ± 1.87 (20)	0.01	0.933
CG, NC	29.9 ± 3.16 (7)	28.5 ± 2.11 (13)	0.12	0.734
CG, C	24.8 ± 2.78 (4)	25.0 ± 2.57 (13)	0.01	0.930

Native species richness

	Not flooded	Flooded	*F*	*P*
NG, NC	35.1 ± 3.48 (8)	24.9 ± 3.36 (7)	4.68	0.050
PG, NC	27.5 ± 1.19 (48)	26.7 ± 1.42 (32)	0.16	0.691
PG, C	21.3 ± 2.04 (18)	23.8 ± 1.67 (20)	1.04	0.314
CG, NC	26.6 ± 3.48 (7)	25.4 ± 2.25 (13)	0.08	0.785
CG, C	21.8 ± 2.87 (4)	20.7 ± 2.34 (13)	0.13	0.727

Introduced species richness

	Not flooded	Flooded	*F*	*P*
NG, NC	4.4 ± 0.80 (8)	5.4 ± 1.09 (7)	0.54	0.474
PG, NC	5.3 ± 0.43 (48)	2.9 ± 0.32 (32)	18.34	<0.001
PG, C	5.4 ± 0.61 (18)	3.0 ± 0.50 (20)	10.57	0.002
CG, NC	3.3 ± 0.89 (7)	3.2 ± 0.58 (13)	0.01	0.926
CG, C	3.0 ± 0.71 (4)	4.3 ± 0.62 (13)	1.01	0.331

Table 3.30. (continued)

Forb species richness

	Not flooded	Flooded	*F*	*P*
NG, NC	20.5 ± 1.43 (8)	14.6 ± 1.69 (7)	7.30	0.018
PG, NC	16.5 ± 0.79 (48)	13.5 ± 0.87 (32)	5.55	0.021
PG, C	13.6 ± 1.58 (18)	13.0 ± 0.99 (20)	0.05	0.832
CG, NC	16.0 ± 1.77 (7)	13.9 ± 1.06 (13)	1.04	0.322
CG, C	12.8 ± 1.11 (4)	12.5 ± 1.68 (13)	0.09	0.763

Grass species richness

	Not flooded	Flooded	*F*	*P*
NG, NC	13.1 ± 1.59 (8)	9.0 ± 1.84 (7)	3.10	0.102
PG, NC	10.5 ± 0.54 (48)	9.5 ± 0.63 (32)	1.36	0.247
PG, C	9.5 ± 0.90 (18)	8.9 ± 0.95 (20)	0.31	0.583
CG, NC	9.1 ± 1.78 (7)	8.2 ± 1.16 (13)	0.16	0.698
CG, C	8.5 ± 1.32 (4)	8.6 ± 1.00 (13)	0.00	0.947

Graminoid species richness

	Not flooded	Flooded	*U*	*P*
NG, NC	0.9 ± 0.30 (8)	1.6 ± 0.57 (7)	21.0	0.463
PG, NC	1.04 ± 0.16 (48)	1.8 ± 0.20 (32)	475.0	0.004
PG, C	0.6 ± 0.15 (18)	2.0 ± 0.34 (20)	69.0	<0.001
CG, NC	0.71 ± 0.29 (7)	1.9 ± 0.33 (13)	18.5	0.030
CG, C	1.5 ± 0.65 (4)	1.1 ± 0.18 (13)	20.5	0.549

Sub-shrub species richness

	Not flooded	Flooded	*U*	*P*
NG, NC	4.1 ± 0.88 (8)	3.0 ± 0.49 (7)	19.5	0.336
PG, NC	3.1 ± 0.21 (48)	2.7 ± 0.23 (32)	622.0	0.154
PG, C	1.9 ± 0.37 (18)	2.0 ± 0.23 (20)	174.0	0.874
CG, NC	2.6 ± 0.57 (7)	2.5 ± 0.31 (13)	44.5	0.938
CG, C	1.8 ± 0.48 (4)	1.9 ± 0.35 (13)	23.5	0.785

Table 3.30. (continued)

Shrub species richness

	Not flooded	Flooded	*U*	*P*
NG, NC	0.75 ± 0.31 (8)	1.4 ± 0.37 (7)	17.0	0.232
PG, NC	1.3 ± 0.10 (48)	1.3 ± 0.11 (32)	733.0	0.736
PG, C	0.8 ± 0.21 (18)	0.5 ± 0.11 (20)	155.0	0.478
CG, NC	1.0 ± 0.31(7)	1.2 ± 0.19 (13)	40.5	0.699
CG, C	0.0 ± 0.00 (4)	0.7 ± 0.31 (13)	16.0	0.296

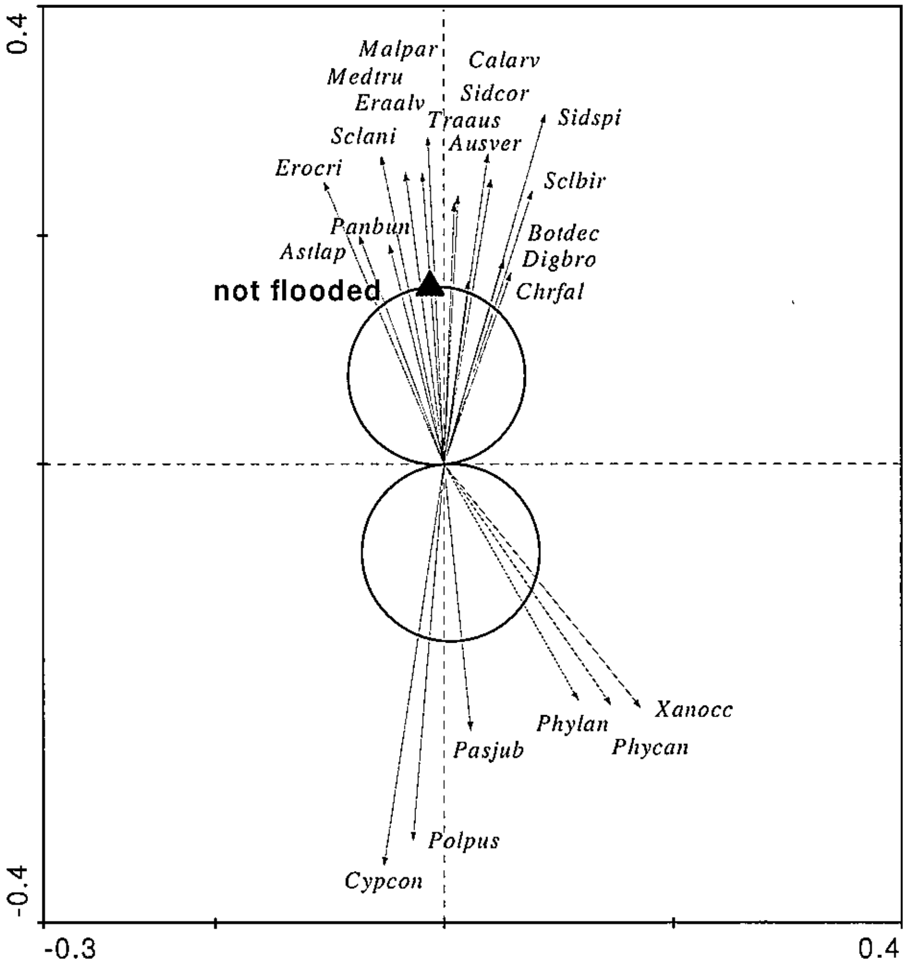


Fig. 3.14. A t-value biplot showing species associated with sites that were not flooded and sites that were flooded. Species were not significantly associated with this variable, as species arrows do not end within the Van Dobben Circles (corresponding t-values are less than two). Refer to Appendix 3.2 for species labels.

Table 3.31. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (Fisher's exact test, d.f. = 1) to flooding at sites that were periodically grazed, and not cultivated

	Flooded ($n = 32$)	Not flooded ($n = 48$)	P
More frequent at flooded sites			
Cyperus bifax	72	33	0.001
Paspalidium jubiflorum	22	2	0.006
More frequent at non-flooded sites			
Maireana microphylla	0	17	0.019
**Malvastrum americanum*	25	52	0.021
**Medicago polymorpha*	13	35	0.037
**Medicago truncatula*	3	19	0.045
Paspalidium constrictum	6	27	0.021
**Rapistrum rugosum*	34	65	0.012

Table 3.32. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (Fisher's exact test, d.f. =1) to flooding at sites that were periodically grazed, and had a history of cultivation

	Flooded ($n = 20$)	Not flooded ($n = 18$)	P
More frequent at flooded sites			
Cyperus bifax	70	33	0.050
Eleocharis pallens	35	0	0.009
More frequent at non-flooded sites			
**Rapistrum rugosum*	40	78	0.025

Fire

CCA of the species cover data showed that occurrence of fire did not have a significant influence on species composition. For the nine paired sites surveyed, only small differences in species richness were observed (Table 3.33). One unburnt site had 12 more species than the adjacent burnt site, although this site was burnt just 1 year before sampling in a high intensity burn. In contrast, however, another unburnt site had nine less species than the adjacent burnt site sampled 1 year after fire. Paired comparisons identified no significant difference in total species richness between burnt and unburnt sites (Table 3.33). The only marginal difference identified was for introduced species richness ($P = 0.076$), with a greater number of introduced species at unburnt sites (Table 3.33).

Table 3.33. Species richness for all nine burnt and unburnt sites and paired t-test results comparing the two sets of sites

	Burnt ($n = 9$)		Unburnt ($n = 9$)			
Species richness	Mean	SE	Mean	SE	t	P
Total	34.1	3.00	35.8	4.13	0.83	0.430
Native	29.9	2.97	30.1	3.88	0.15	0.886
Introduced	4.2	1.06	5.7	1.39	2.04	0.076
Forb	15.8	1.62	18.4	2.64	1.51	0.171
Grass	12.1	1.53	10.8	1.44	1.51	0.169
Graminoid	1.3	0.44	1.3	0.33	0.00	1.000
Sub-shrub	3.2	0.43	3.3	0.44	0.19	0.855
Shrub	1.3	0.24	1.6	0.18	0.61	0.559

3.3.5 Environmental determinants of composition

Environmental variables accounted for approximately 19% of the explained variance in the species cover data. Species composition was influenced by several environmental variables (Table 3.16; Fig. 3.15) mostly related to time of sampling (and, hence, rainfall prior to sampling), location of quadrat (northing, easting and altitude), soil type (and topsoil chemistry) and structural formation of the vegetation. Forward selection identified 15 environmental variable categories that were significant in explaining species composition (Table 3.16).

Rainfall and location

Date of sampling, year of sampling, geographical position and rainfall in the months prior to sampling influenced species composition (Figs 3.15, 3.16). Some changes in composition over time were reported in Section 3.2.2. There were several species associated with sites in the east and west of the study region (Fig. 3.17). Easting trended in a similar direction to altitude (sites in the east were generally at higher altitudes than sites in the west) and annual rainfall (sites in the east having greater annual rainfall). Grasslands in the east of the study region (east of the Newell Highway) were often dominated by the grasses *Aristida ramosa, Aristida leptopoda, Bothriochloa decipiens, Dichanthium sericeum* and *Austrostipa* spp. While some of these species were also dominant further west, *Astrebla lappacea, Enteropogon acicularis, Paspalidium jubiflorum* and **Urochloa panicoides* appeared only to be dominant in the west of the study region.

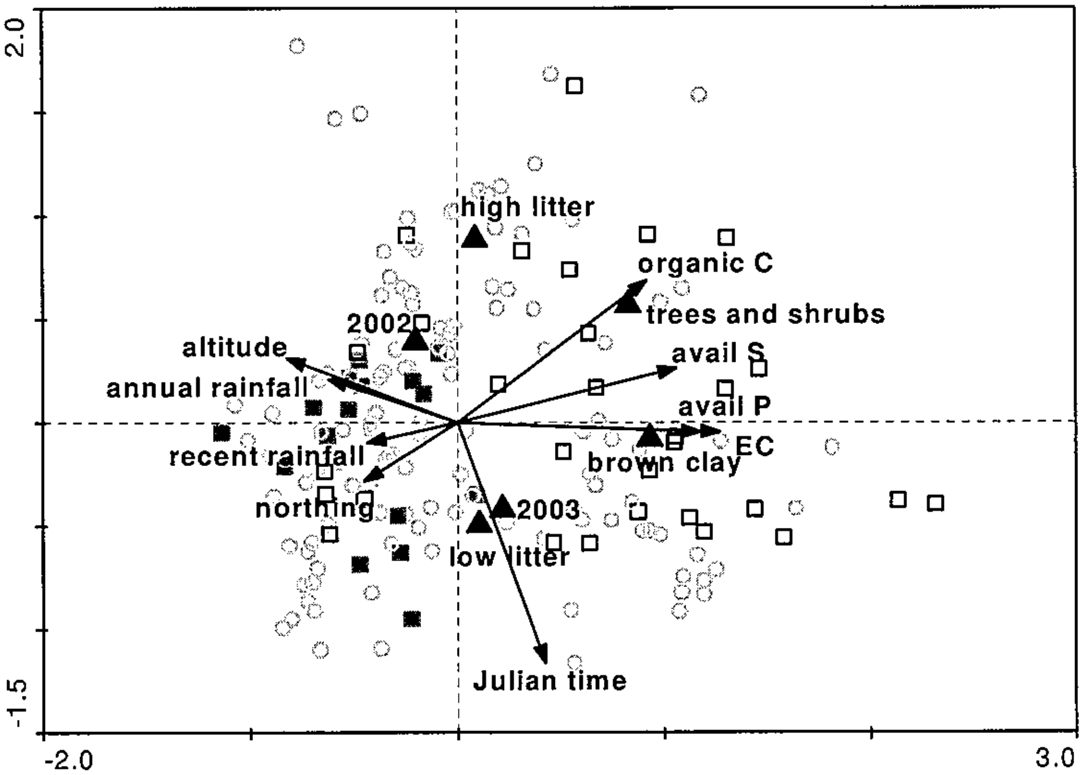


Fig. 3.15. Ordination biplot (partial CCA) for environmental variables and individual sites sampled in autumn 2002 and 2003. Sites classified by grassland condition: filled squares, good condition; non-filled squares, poor condition; non-filled grey circles, moderate condition. Nominal variable categories represented by triangles and continuous variables by vectors.

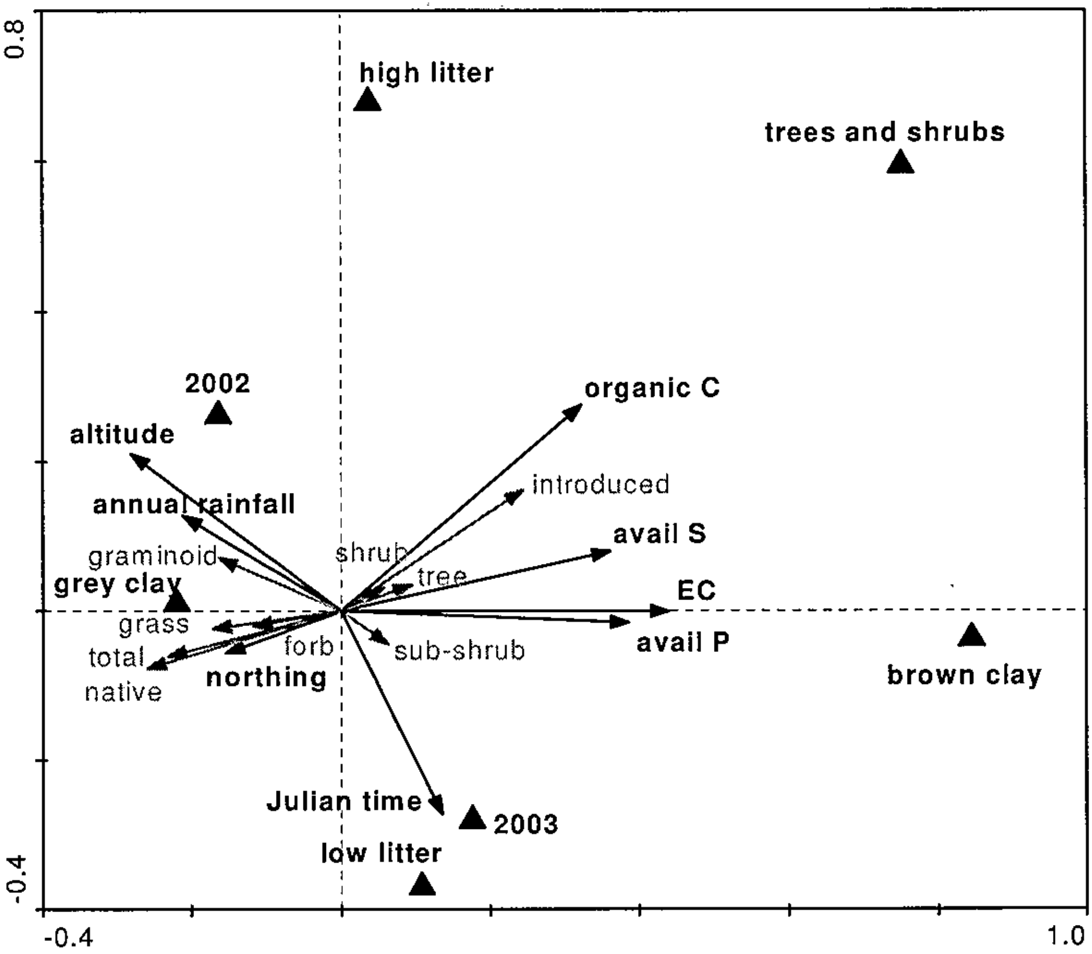


Fig. 3.16. Ordination (partial CCA) for environmental variables showing trends in species richness (i.e. supplementary variables in grey font). Nominal variable categories represented by triangles and continuous variables by vectors.

Sites in good condition were generally in the eastern half of the study region possibly reflecting the influence of higher rainfall (Fig. 3.15). Species associated with such sites include *Dichanthium sericeum, Aristida leptopoda, Vittadinia cuneata, Panicum queenslandicum, Leiocarpa websteri, Thellungia advena, Chamaesyce drummondii, Pycnosorus globosus, Desmodium campylocaulon, Eulalia aurea* and *Chloris truncata* (Fig. 3.17). Northing trended in a similar direction and was associated with similar species. Total, native, grass, and forb richness increased in the same direction on the ordination diagram, and were associated with northing and rainfall in the 4 months prior to sampling (Figs 3.15, 3.16).

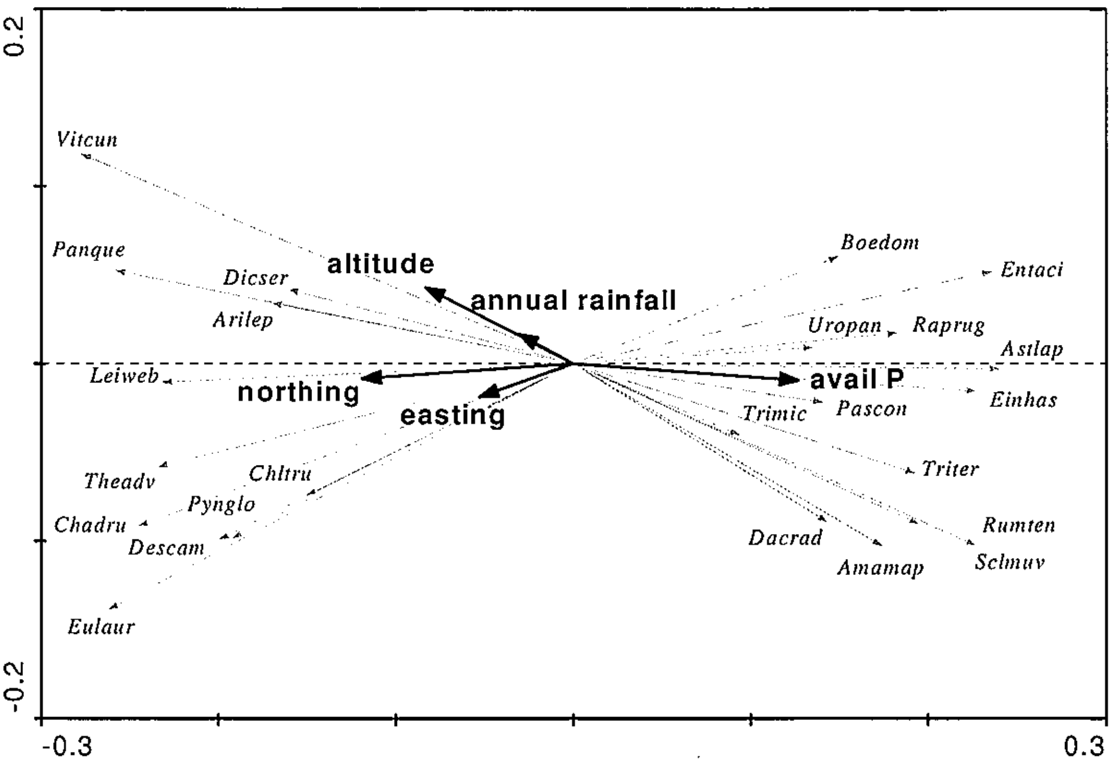


Fig. 3.17. Species associated (not significantly) with vectors for altitude, annual rainfall, northing, easting and soil fertility (available phosphorus). Refer to Appendix 3.2 for species labels.

Soil

Grassland species composition varied greatly between areas of clay soils and non-clay soils (Fig. 3.5), and there were obvious differences in composition between grey to black clays and brown clays (Fig. 3.16). Species associated with brown clay soils included **Tribulus* spp., *Portulaca oleracea, Dactyloctenium radulans,* **Urochloa panicoides*, *Paspalidium constrictum* and *Sclerolaena muricata* var. *villosa,* while *Dichanthium sericeum, Aristida leptopoda, Panicum queenslandicum* and *Vittadinia cuneata* were more common on grey to black clay soils. Soil type may also have an influence on grassland condition index, as all sites in good condition on clay soils were located on grey to black clay soils (Figs 3.15, 3.16). The measured soil properties also had an influence on composition (Fig. 3.15). Sites on brown clay soils were generally distributed towards the right of the ordination diagram and were associated with a soil fertility gradient (Fig. 3.16). Vectors for electrical conductivity, available phosphorus and available sulphur trended in a similar direction and were associated with similar species (e.g. Fig. 3.17). The vector for topsoil organic carbon was also to the right of the ordination diagram and was associated with introduced species richness (Fig. 3.16).

A summary of the soil properties measured for all sites is provided in Table 3.34.

Table 3.34. Summary values for topsoil chemistry at sites visited in the regional vegetation survey (*n* = 179)

Soil variable	Min	Max	Mean	SE
pH_{H_2O}	5.9	9.9	7.1	0.04
EC (µS/cm)	22.1	364.4	75.3	3.45
Available P (mg/kg)	0.9	50.4	8.4	0.69
Available S (mg/kg)	0.1	18.9	5.4	0.28
Organic C (%)	3.0	10.9	5.2	0.10

Other variables

Vegetation structural formation and litter also had an influence on species composition (Table 3.16; Fig. 3.15). However, there was no obvious grouping of species associated with the different vegetation structural formations (i.e. this variable had negligible variance). The main differences in composition associated with structural formation were due to the presence of trees (e.g. *Eucalyptus* spp.). Open woodlands and woodlands with shrubs were associated with higher topsoil organic carbon (Fig. 3.15) and introduced species richness (Fig. 3.16).

Sites with low and high litter levels were clearly separated in ordination (Fig. 3.15). Field observations suggest that litter cover varied greatly depending on the occurrence of recent grazing, and sometimes cultivation (i.e. where stubbles were not removed). Sites with high litter levels were commonly those with abundant *Aristida* spp. or *Austrostipa* spp. in the east of the study region (i.e. 43% of high litter sites).

3.4 Discussion

3.4.1 The effects of drought and seasonal variations in composition and richness

Winter-growing species were recorded in quadrats in spring 2002 and 2003. However, spring 2002 sampling was strongly influenced by drought. There was a reduction in both grass and forb species richness due to drought in the winter of 2002 (Fig. 3.7). Sampling in spring 2003 may have been closer to 'normal' spring conditions, although species composition and abundance at least at some sites were probably still influenced by drought. In spring 2003, there was an increase in winter-growing forbs (Fig. 3.7), including several introduced species. Increases in winter-growing forbs with the occurrence of winter rainfall in Mitchell grasslands are well known (e.g. Orr 1981; Campbell 1989; Clarke *et al.* 1998). In fact, sampling in winter to early spring in the Hunter and Earl (1999) survey increased species richness by about 20%, compared to sampling in summer-autumn.

While native species richness was reduced due to the drought conditions in spring 2002, rainfall in early 2003 resulted in increased richness by autumn 2003 (Fig. 3.6). Different results have been reported in Minnesota grasslands, where species richness did not recover in 2 years, despite more normal precipitation after a one in 50-year drought (Tilman & El Haddi 1992). These authors reported a recovery of biomass (after a 47% reduction) following drought and suggested that species richness became recruitment limited. The relatively rapid recovery of species richness on the Moree Plains may have been due to the naturally lesser basal area of grasses in these grasslands, less severe drought and differences in species composition. Given that droughts occur frequently in Mitchell grasslands (Partridge 1996) it is likely that many species have some resilience to drought (e.g. an ability to persist in the soil as seeds or storage organs until conditions are favourable).

The effects of drought are closely related to grazing, as defoliation through grazing may reduce a plant's ability to cope with drought (Busso & Richards 1995; Vesk & Westoby 2001). Anderson *et al.* (1996) reported low levels of germinable seed of perennial grasses after drought and suggested that areas subject to grazing may have less grass recruitment when suitable rain occurs. Therefore, we might expect TSR sites to decrease more in species richness in response to the drought than the four sites ungrazed by livestock. Surprisingly this was not the case, with a 24% reduction in native species richness at TSR sites between autumn 2002 and spring 2002, and a 34% reduction in native species richness at ungrazed sites in the same period. It is possible that grazed sites contain more drought-resistant species, as plant adaptations to drought and grazing may be similar, because both result in loss of individual plant organs (Milchunas *et al.* 1988). Hence grazing-sensitive species might also be less tolerant of drought. This suggests that higher species richness does not always lead to greater drought resistance, as proposed by Tilman and Downing (1994). They reported a decline in plant species richness of >35% due to severe drought in Minnesota grasslands. Differences in grassland response to drought are perhaps not surprising given differences in composition (and the ability of species to survive drought) and differences in the basal area of perennial grasses in these different grasslands.

**Rapistrum rugosum* had the highest mean cover score of all species recorded in the 17 repeatedly sampled sites, in spring 2003. This species appears to have been favoured by the dry conditions in 2002, which increased bare ground, allowing high levels of recruitment in autumn of 2003. The dominance of this introduced species may be of concern because of its ability to grow into late spring and compete with summer-growing species (Bellotti *et al.* 1986). One landholder reported a decrease in *Astrebla* spp. abundance due to competition with **Rapistrum rugosum* in the past. Rapid population growth of certain species after increases in the amount of bare ground during drought has been previously reported in grasslands (Bazzaz & Parrish 1982).

Despite the presence of winter-growing annual grasses (e.g. **Lolium perenne, *Avena fatua, *Bromus catharticus* and **Phalaris paradoxa*), there was a reduction in grass species richness each spring (Fig. 3.7). This may be related to the frost tolerance of summer-growing grasses. Taxa such as *Panicum buncei* and *Eriochloa* spp. were less common during the spring sampling times, and may have been reduced to rootstock during these times. Larger tussock grasses such as *Enteropogon acicularis, Panicum decompositum* and *Astrebla* spp. were able to persist through winter in a dormant and reduced state. Similar variations in frost tolerance were reported on the Northern Tablelands of NSW (J. Reseigh, unpublished data, 2004). While frosts on the Moree Plains are not severe, it is likely that they have impacts on species composition, due to specific differences in susceptibility. The impact of frosts was possibly exaggerated by the low winter rainfall in 2002.

3.4.2 Comparisons with previous surveys

Variations in species composition

The 364 taxa reported represent approximately 18% of the western NSW flora defined by Cunningham *et al.* (1992). Hunter and Earl (1999) listed a total of 544 plant taxa in the region from their survey (414 taxa), and other sources (130 taxa). The taxa recorded here represent a large proportion (53%) of their list. An additional 75 taxa not previously recorded by Hunter and Earl (1999) were identified in this study (Appendix 3.2). A similar proportion of introduced species was recorded during each of the surveys (21% in the Hunter and Earl (1999) survey compared with 23% in this study). In both surveys, the five most frequently represented families were the same. However, taxa belonging to the family Chenopodiaceae were more common in the Hunter and Earl (1999) survey (approximately 20 more species). Some differences between the two surveys are to be expected given that the Hunter and Earl (1999) survey extended approximately 110 km further west than the western-most site in the present study. While their survey concentrated on grassland communities, some of the communities described were not true grasslands. These authors found that true grasslands were generally restricted to the central and eastern regions of the Moree Plains, while open chenopod shrublands were more common in the west.

There were differences in species abundances between the Hunter and Earl (1999) survey and the current survey. Hunter and Earl (1999) reported *Eriochloa crebra* as having the highest cumulative cover. This species was a common component of the sites visited in the present study but was only ranked 25th based on mean cover abundance. *Marsilea drummondii* and *Sporobolus mitchellii* were also far more abundant in the Hunter and Earl (1999) survey. All three of these species were more common at sites that had received higher rainfall or some history of flooding in the current survey, and are known to prefer wetter conditions (Campbell 1989; Cunningham *et al.* 1992). For example, Campbell (1989) stated that *S. mitchellii* forms dense mats after flooding or heavy summer rains. The differences in cover scores of grassland taxa between surveys probably reflect differences in rainfall. In the 3-month period from January to March in 1999, a total of 243 mm of rainfall was recorded for Moree, while for the same period in 2002 and 2003 totals of 128 mm and 179 mm respectively, were recorded. According to graziers visited during the Hunter and Earl (1999) survey, seasonal conditions had been favourable in the

2 years prior to the survey. Hence, Hunter and Earl (1999) may be correct in saying that their findings are probably atypical of 'normal conditions' due to the quality of the season.

Fluctuations in *Dichanthium sericeum* and *Astrebla* abundance

Between sampling in autumn 2002 and autumn 2003 *Astrebla elymoides* increased in abundance, while *Dichanthium sericeum* decreased (Table 3.14). Variation in the abundances of dominant species between years in relation to rainfall has been previously reported (e.g. Fowler 1986). Fluctuations in the dominance of *Astrebla* spp. and *D. sericeum* were also reported by Williams and Roe (1975) and Groves and Williams (1981). The latter believed there is a process of 'fluctuating climaxes' in which an explosive recruitment of *Dichanthium* occurs episodically against the background of normal survival and recruitment of *Astrebla* spp. One landholder I spoke to during this research suggested the dominance of *Astrebla* spp. or *D. sericeum* depends on when the rainfall occurs (L. Boland, pers. comm., 2002). He stated that if rainfall occurs early in the growing season *D. sericeum* is more likely to dominate, while rainfall later in summer is more likely to promote *Astrebla* dominance. *Dichanthium sericeum* is known to respond well to spring and early summer rains (Campbell 1989).

Hunter and Earl (1999) reported that while *Astrebla* spp. were recorded in 40% of the quadrats surveyed, they only dominated a handful of sites and were only a minor component of the pasture. This was not the case in the current study. When present, *Astrebla* spp. were often among the dominant species (Fig. 3.4). Hence it is likely that *Astrebla* spp. were relatively less abundant during the Hunter and Earl (1999) survey due to seasonal conditions (i.e. above average rainfall) in the year that they sampled. Perhaps during favourable seasons (years of high rainfall) *Astrebla* spp. cover declines because of competition with species that become abundant under such conditions (e.g. *Dichanthium sericeum*). In support of this, Roberts (1978) suggested that the more drought-tolerant *Astrebla* communities may be dominated by *Dichanthium* after a series of good years.

Despite the decrease in abundance of *Dichanthium sericeum* between 2002 and 2003, it was still the most cover-abundant species in autumn 2003 and the second most cover-abundant in the repeat survey sites. This is surprising, as Campbell (1989) reported that it has little resistance to drought, and Phelps and Bosch (2002) reported removal of *D. sericeum* after drought. Field observations suggest that its ability to persist through drought was probably related to its ability to respond quickly from seed after relatively small rainfall events. Indeed, Campbell (1989) stated that it avoids droughts through its ability to readily recolonise under more favourable conditions.

3.4.3 The relative importance of disturbance-related and environmental variables

Floristic composition of the Moree Plains grasslands was explained more by environmental variables (19% of the variance) than disturbance-related variables (11% of the variance). This does not necessarily mean that environmental variables are more important than disturbance-related variables in determining plant composition, since there were a greater number of environmental variables (14) than disturbance-related variables (eight), and there are a number of problems associated with partitioning variance (Palmer 2001). However, forward selection revealed that five environmental variables explained more of the variance in plant species cover than disturbance-related variables (Table 3.16). The greater influence of environmental variables is related to the strong temporal influence on composition, as it is well known that Mitchell grasslands respond strongly to rainfall (Orr 1981; Orr & Holmes 1984; Campbell 1989; Orr & Evenson 1991a). These grasslands appear to differ from those on the Northern Tablelands of NSW where composition responds more to exogenous disturbance than to environmental factors (Henderson & Keith 2002; Clarke 2003).

3.4.4 Environmental variables

Rainfall was an important variable in explaining species richness and composition on the Moree Plains (Figs 3.6, 3.15) as has already been discussed (Sections 3.4.1 and 3.4.2). In a multivariate analysis of Mitchell grasslands in Queensland, Fensham *et al.* (2000) found that the longest floristic gradient was aligned with climatic variables associated with rainfall. Changes in Mitchell grasslands brought about by seasonal rainfall and drought were also reported by Orr and Holmes (1984), Campbell (1989), Orr and Evenson (1991a), Bowman *et al.* (1996) and Phelps and Bosch (2002). Other climatic variables such as temperature, moisture index, and solar radiation may have an additional influence on species composition (Risser 1985; J. Reseigh, unpublished data, 2004).

Date of sampling, year of sampling and geographical position had a significant influence on species composition (Table 3.16). The influence of sampling time on composition is not unexpected given that sites sampled on any one given day were often geographically close; up to four sites were sampled on any one property and nearby properties were often sampled on the same day. Nearby sites (especially on the same property) often had received similar climatic conditions (i.e. rainfall) and may have shared similar long-term management histories. For example, five out of ten sites not grazed for 5-19 years were located within Kirramingly Nature Reserve, and, hence, had a similar composition because of their close proximity (Fig. 3.8). Ideally, sampling time might have been randomised over all of the sites sampled, but this was not feasible in this study. Milchunas and Laurenroth (1993) reported that geographic location had a more important influence on species composition than grazing intensity. Although they investigated data from six regions of the world (Africa, Asia, Australia, Europe and North and South America), the same may be true at much smaller scales (Table 3.16).

Differences in composition from east to west across the study region were obvious, especially when the nine sites that occurred on non-clay soils were included in analysis (Fig. 3.5). Grasslands east of the Newell Highway were often dominated by *Aristida ramosa, Aristida leptopoda, Austrostipa* spp., *Bothriochloa decipiens* and *Dichanthium sericeum*. While some of these species were also dominant further west, some species (e.g. *Astrebla lappacea, Enteropogon acicularis, Paspalidium jubiflorum* and **Urochloa panicoides*) were only dominant in the west of the study region. Hunter and Earl (1999) similarly reported longitudinal variations in composition, with chenopod shrublands more common in the west and grasslands in the east. Floristic variation has also been reported for an east-west gradient in grassy woodlands in central NSW (Prober & Thiele 2004). Apart from the obvious soil type differences in the current study, other variations from east to west are probably related to altitude as well as the rainfall gradient over the study region. The east-west variation contributed to differences in composition between the 2 years of sampling, as more sites in the east of the study region were sampled in 2002. Seasonal conditions (i.e. winter drought in 2002) in the lead up to sampling also influenced differences between the 2 years (Section 3.4.1).

Sites in good condition were generally in the eastern half of the study region, possibly reflecting the influence of higher recent rainfall (Figs 3.15, 3.16) or a natural gradient in native species richness. Other environmental variables (or variable categories) that were associated with such sites include altitude, annual rainfall, northing and grey to black clay soils (Figs 3.15, 3.16). It would be interesting to see whether more sites in good condition appear in the west of the study region, in years of higher rainfall, or whether there is a natural gradient in richness from east to west. Sites with high native species richness have previously been recorded in the west of the region (Hunter & Earl 1999).

Soil type and chemistry had some influence on composition (Table 3.16). Sites on brown clay soils were associated with a soil fertility gradient (Fig. 3.16). Soil chemistry was also recognised as an important factor in determining grassland species composition on the Southern Tablelands of NSW (Prober *et al.* 2002). These authors reported dominance of introduced annuals in remnants with nutrient-rich topsoils, with a high correlation between topsoil nitrate and the abundance of introduced annuals. Introduced species richness was most strongly associated with organic carbon in the current study (Fig. 3.16). In fact, sites with higher organic carbon and fertility were generally in poorer condition in the current study (Fig. 3.16). This is not surprising given that introduced species are often more competitive than natives under conditions of enhanced soil fertility (Wijesuriya & Hocking 1998), with traits favouring rapid colonisation and effective reproduction (Trémont & McIntyre 1994). In the Walgett region, Bowman *et al.* (1996) reported that paddocks with 'good' pasture (higher densities of *Astrebla* spp.) differed in soil pH, electrical conductivity, organic carbon and mineralisable nitrogen, when compared to 'poor' pastures. Paddocks in poor condition had higher organic carbon and nitrogen, which was correlated with a greater component of annuals such as **Medicago* spp.

3.4.5 Disturbance-related variables

Grazing

While environmental variables explained more of the variance in species composition on the Moree Plains, livestock grazing was important, accounting for 6.5% of the explained variance. The effects of grazing on grassland species composition have been widely reported (e.g. McIntyre & Lavorel 2001; McIntyre *et al.* 2003; Barbaro *et al.* 2004). The importance of grazing in influencing composition varies between studies. In calcareous grasslands in France, grazing intensity was the second most important variable (explaining 6% of floristic variability) after lithology (Barbaro *et al.* 2004). Grazing was an important variable for grasslands of south-east Queensland, accounting for 4% of the explained variation (McIntyre *et al.* 2003), and grazing intensity was the strongest influence on composition of temperate grassy woodlands (Clarke 2003). However, McIntyre and

Lavorel (1994b) found lithology, altitude and soil disturbance to be the most important variables in explaining floristic composition, with grazing being of lesser importance.

Removal of livestock grazing for 5-19 years significantly increased species richness, especially at Kirramingly Nature Reserve where grazing had been excluded for 8 years. Several studies report increases in native plant diversity and abundance after long-term removal of livestock grazing (e.g. Lacey & Van Poollen 1981; Brady *et al.* 1989; Spooner *et al.* 2002). However, not all ungrazed sites in the present study had above average species richness. The four sites on clay soils that had not been grazed for 20 years had below average species richness, possibly owing to the dry conditions prior to sampling and a history of cultivation at one site (Section 3.3.4). The variability in richness between ungrazed sites highlights the influence of interactions with other variables, such as rainfall.

Despite the higher species richness at some ungrazed sites, there is substantial evidence (e.g. Collins & Barber 1985; Noy-Meir 1995; Bekker *et al.* 1997) that plant species diversity of undisturbed grasslands can deteriorate. This is because grazing reduces the competition from dominant tussock grasses that otherwise exclude inter-tussock species from surviving and regenerating (Trémont & McIntyre 1994) and, therefore, often increases the number of forbs in the inter-tussock spaces (Trémont 1994; Bai *et al.* 2001; McIntyre & Lavorel 2001). An increase in forb richness at grazed sites was not observed in the current study (Table 3.20). In fact, across all sites on clay soils, forb richness was lower at grazed sites than at ungrazed sites (Appendix 3.4a). It is likely that some native forbs are less resistant to grazing than native grasses due to the position of regenerating buds (McIntyre *et al.* 1995; Clarke 2003).

The evident lack of competitive exclusion in these grasslands may be due to the occurrence of spaces between the tussocks of the dominant grass species, with relatively high amounts of bare ground, allowing recruitment of inter-tussock species. Only 15 sites of the 184 surveyed had low levels of bare ground (≤10%), and most of these sites were in the east of the study region and were dominated by genera other than *Astrebla* and *Dichanthium*. Gaps between individual tussocks with a range of inter-tussock species are well known features of Mitchell grasslands (Everist & Webb 1975; Roberts 1978; Campbell 1989). Sites with less obvious inter-tussock spaces and higher litter accumulation were often dominated by *Aristida* spp. or *Austrostipa* spp., and species richness at such sites may well benefit more from low or moderate levels of livestock grazing (Trémont & McIntyre 1994; Fensham 1998).

Stocking rate was important in explaining species composition (Table 3.16; Fig. 3.10). It is possible this was partly an artefact of the east to west variations in composition, as sites east of the Newell Highway had significantly higher stocking rates ($P < 0.001$) and the species most correlated with increasing stocking rate (*Austrostipa verticillata*, *Digitaria brownii*, *Chrysopogon fallax*, *Sclerolaena birchii* and *Bothriochloa decipiens*) were more common in the east of the study region. However, the influence of livestock grazing intensity on species composition has been widely reported (e.g. Milchunas *et al.* 1988; Bullock *et al.* 1994; Garden *et al.* 2000; McIntyre & Lavorel 2001). Despite changes in composition, there was no obvious relationship between stocking rate and native species richness ($P > 0.05$) in the present study. Grazing intensity often affects species composition and abundance more than species richness in grasslands (McIntyre & Lavorel 1994b; Fensham 1998).

Species richness was not higher at intermediate grazing intensities as reported by McIntyre and Lavorel (1994a) probably because of the absence of strong competitive exclusion from the dominant grasses. However, native richness was marginally higher ($P = 0.092$) and introduced richness was marginally lower ($P = 0.073$) at privately owned sites, compared to TSR sites. These differences in richness were probably related to management differences because stocking rates were often higher for TSRs (Table 3.18). Bowman *et al.* (1996) suggested that the loss of species diversity in Mitchell grasslands in north-western NSW (Walgett Plains) is associated with pressure of livestock grazing. It is possible that the occurrence of past heavy grazing had an important influence on the species compositions observed in this study, as heavy livestock grazing for long periods can result in a decline in *Astrebla* spp. (e.g. Francis 1935; Beadle 1948; Campbell 1989; Phelps & Bosch 2002). The influence of stocking rate on composition and diversity requires experimental investigation on the Moree Plains.

Stocking rates recorded in this study were generally higher (especially for TSRs) than those recommended by Bowman (1992) for Mitchell grasslands of the Moree Plains (i.e. 1.6 to 2.5 DSE/ha, depending on rainfall) (Table 3.18). This may be partly because stocking rates for TSRs were calculated only for the period prior to sampling (rather than long-term averages). These estimates may not be representative of stocking rates over the full year, as stocking rates may vary between seasons, depending on feed availability. Further, these estimates assume that livestock grazed the entire stock route in question, when in reality sections of each route may have received widely differing grazing intensities within the 5-month period. A report on conservation of biodiversity in the TSRs of north-west NSW stated that changing patterns of TSR use over the past few decades has resulted in less of the traditional grazing methods involving short duration, intense, infrequent grazing over large areas, with large mobs of stock (Austen 2002). Instead there are more, smaller, local travelling mobs, with longer periods of grazing in smaller, fixed areas, aided by the use of electric fences. Austen (2002) pointed out that areas of former high environmental value have probably suffered from grazing-induced degradation and suggested that unmonitored use of electric fencing is one of the greatest threats to the native grasslands on public land in the region.

Approximately 26% of continuously grazed sites on the Moree Plains were considered to be in poor condition. Although there was a trend of higher species richness at periodically grazed sites, there were no significant differences in richness between periodically and continuously grazed areas (Table 3.20). The lack of significant differences may be due to the fact that the impacts of continuous and periodic grazing are complicated by the long history of grazing management. Despite the lack of differences, continuous grazing is generally not regarded as a sustainable grazing regime (Kemp *et al.* 2000). This is because continuous grazing with sheep at medium stocking rates results in continued selection of the most palatable plant species, and can lead to their loss from the paddock and, hence, a loss in production. For example, Francis (1935) reported that continuous overstocking is thought to have led to a decline in *Astrebla* spp. abundance. Stocking rates may have been low enough at most continuously grazed sites in the current study to prevent loss of species richness.

Major changes in composition do not always result in changes in total species richness (Stohlgren *et al.* 1999; Bai *et al.* 2001; McIntyre *et al.* 2003). If the number of species that increase in response to grazing is similar to the number that decrease, there is no net change in richness. Considering the significant ($P < 0.05$) responses to grazing reported here, more native species decreased in frequency than did introduced species (seven natives decreased and two introduced species decreased). However, only two introduced species (**Medicago polymorpha* and **Tribulus terrestris*) were more frequent at grazed sites. This is surprising given that McIntyre *et al.* (2003) reported that introduced species are more common amongst the group of species that increase in response to grazing, and similar results were reported by Prober and Thiele (1995). However, McIntyre and Lavorel (1994a) and Fensham (1998) suggested grazing does not always influence richness of introduced species.

High grazing intensities are known to encourage introduced species (McIntyre & Lavorel 1994a,b; McIntyre *et al.* 1995; McIntyre & Martin 2001), but there was no significant relationship between introduced species richness and stocking rate in the current study. In temperate grasslands, introduced species may replace natives where disturbance (fire or grazing) is frequent (Lunt 1990a; McIntyre & Lavorel 1994b; Lunt & Morgan 1999). This may be because many introduced species are pre-adapted to environments with long histories of livestock grazing, enhanced fertility and cultivation, and are able to quickly colonise disturbed areas (Trémont & McIntyre 1994). Trémont and McIntyre (1994) pointed out that in Australia there is a relatively low proportion of short-lived native forbs that can exploit conditions brought about by disturbance. Perhaps on the Moree Plains, a higher proportion of natives are well adapted to cope with grazing, due to seasonal variations (i.e. drought) and possibly fire regimes over evolutionary history. Some native forbs, for instance, are reduced to rootstock during dry conditions (or over winter) and resprout after sufficient spring or summer rain (e.g. *Desmodium campylocaulon*, *Neptunia gracilis*, *Phyllanthus virgatus*, *Sida trichopoda*, *Boerhavia dominii* and *Wahlenbergia communis*). Such species may be able to survive defoliation by grazing, but may not be well adapted to disturbances that kill the plant (i.e. cultivation).

Many species were common at both grazed and ungrazed sites and, hence, did not respond significantly to grazing. The current species composition at a site may represent a range of plant responses to environmental factors and historic land-uses (Díaz *et al.* 1994; Stohlgren *et al.* 1999). Thus, the interactions with cultivation and flooding history and a range of other variables complicate interpretations of the relative importance of grazing on the response of an individual species. For example, species that responded significantly to grazing at uncultivated sites not subject to flooding were different from those that responded at uncultivated, flooded sites (Tables 3.21, 3.22). Of the species that responded significantly to grazing, some (e.g. *Boerhavia dominii*, *Sida trichopoda* and **Medicago* *polymorpha*) responded inconsistently. For example, *Boerhavia dominii* was common at ungrazed and continuously grazed sites without a history of cultivation or flooding, but was significantly less common at periodically grazed sites in this category. In addition, Fensham *et al.* (1999) reported an increasing response along a grazing disturbance gradient for this species. Inconsistent responses of species are common in the literature (Risser *et al.* 1981; Noy-Meir *et al.* 1989; Stohlgren *et al.* 1999; Kemp *et al.* 2000; Vesk & Westoby 2001; Kirkpatrick *et al.* 2005). Vesk and Westoby (2001) reported that 41% of 324 species, Australia-wide, responded inconsistently to grazing and a similar percentage was recorded by Noy-Meir *et al.* (1989) in Mediterranean grasslands.

Further inconsistencies in grazing response are recognised when the results are compared to other studies. For example, *Panicum queenslandicum* was more frequent at ungrazed sites (63%) without a history of flooding or cultivation than grazed sites (12%) on the Moree Plains. However, this species was recognised as a generalist by McIntyre *et al.* (2003) in subtropical woodlands in south-east Queensland, and Fensham *et al.* (1999) reported a uni-modal response curve for this species along a grazing disturbance gradient in central Queensland. *Eriochloa crebra* was recognised as intolerant of high grazing by McIntyre *et al.* (2003), but appeared to tolerate a range of grazing regimes in this study. There were, on the other hand, similarities in species responses. For example, **Sonchus oleraceus* was recognised as a grazing decreaser by McIntyre *et al.* (2003), and was more common at ungrazed, non-flooded sites in this survey.

Dichanthium sericeum was not less frequent at livestock grazed sites in this study. This is surprising given that Vesk and Westoby (2001) reported that in six out of seven published studies this species was found to decrease in response to grazing. The lack of response of *D. sericeum* to grazing may be due to the large geographical scale of this survey, and the wide range of other variables influencing species responses. It is likely that this species is influenced by grazing intensity, as Fensham *et al.* (1999) reported that it is sensitive to high levels of grazing disturbance and may be more frequent at low grazing intensities. Vesk and Westoby (2001) also reported that *Chloris truncata* and **Urochloa panicoides* usually increase in frequency due to livestock grazing. Grazing occurrence had no real impact on the frequency of *C. truncata* in this study, while **U. panicoides* was more common at ungrazed, flooded sites. The higher frequency (57% occurrence) of **U. panicoides* at such sites was probably unrelated to grazing removal, since the species was generally common at grazed sites overall. The occurrence of this species in four (out of seven) ungrazed sites may have been related to the dry conditions prior to sampling, as these four sites had below average rainfall in the 4 months prior to sampling.

McIntyre *et al.* (2003) identified 'increaser' and 'decreaser' species in response to grazing intensity for subtropical woodlands in south-east Queensland. The varied responses of species to grazing suggest it may not be appropriate to label species as 'increasers' or 'decreasers' unless their responses are consistent. Stohlgren *et al.* (1999) believed that the inconsistent responses of plants to grazing at landscape scales undermine the concept of increaser and decreaser species. Given the concept is site and condition specific, repeated observations under different conditions are required to obtain trends in species response (McIntyre *et al.* 2003). This means collecting large amounts of data for individual species and collating them in a similar manner to Vesk and Westoby (2001).

Cultivation

Sites cultivated within 5 years of sampling differed in composition to uncultivated sites (Fig. 3.8) and the effects of cultivation on native species were still apparent after 15 years (Table 3.25). Given the variability in composition and richness of uncultivated areas it is difficult to estimate how long it takes these grasslands to recover from cultivation. A study by Römermann *et al.* (2005) in the Mediterranean steppe of France found that previously cultivated fields differed from undisturbed areas in terms of species composition, richness and evenness. They suggested that recovery of the original community might take decades or centuries, if at all. Similar findings are reported by Kindscher and Tieszen (1998) after tallgrass prairie restoration in Kansas. However, such a slow recovery would be partly a result of the combined effects of soil disturbance and enrichment or depletion of soil nutrients (depending on whether fertilisation of the soil occurred during cultivation), and not all ecosystems may take so long to recover. In Spanish Mediterranean grasslands, Montalvo *et al.* (1993) reported a short-term loss of species after ploughing, but restoration of the original diversity after 4 years in grazed (but not ungrazed) grasslands.

Species richness is expected to increase along a successional gradient with time since cultivation (Austrheim & Olsson 1999; Nadolny & Eldridge 1999). However, there was no obvious relationship between species richness and time since cultivation on the Moree Plains ($P = 0.711$). This is probably because species richness varied greatly after cultivation, due to influences such as rainfall, the number of consecutive cultivations, depth and fineness of cultivation and post-cultivation grazing. Despite the general negative impacts of cultivation on the Moree Plains, native richness at some sites without a long history of cultivation was high (>35 species) within 5 years of cultivation. Nadolny and Eldridge (1999) similarly reported that a high diversity of species was able to establish in as little as 5 years after cultivation. In addition, sites with a greater number of species were often closer to intact areas of vegetation, suggesting that such areas are important in providing a seed source for re-colonisation (Nadolny & Eldridge 1999). My observations support this finding. *Astrebla* spp. were sometimes observed around the edges of previously cultivated paddocks (presumably spreading inwards) when common in neighbouring paddocks. It is likely that after more than 5 years of consecutive cultivations, seed banks of many native species are depleted, resulting in different species composition at such sites (Table 3.16). The lack of propagules of new species in the seed bank or seed rain has previously been recognised as a limiting factor in the restoration of species diversity in grasslands (Bullock *et al.* 1994; Trémont & McIntyre 1994; Hutchings & Booth 1996; Yates & Hobbs 1997; Kindscher & Tieszen 1998; Coulson *et al.* 2001).

The scatter of sites in poor condition over the ordination diagrams (Figs 3.8, 3.15) suggests there are a number of variables that reduce grassland condition. However, the occurrence of past cultivation (especially sites cultivated for greater than 5 years consecutively) probably had the strongest influence on grassland condition index, as 57% of sites in poor condition had a history of cultivation. The negative effects of cultivation on species richness and grassland condition highlights the need to conserve areas that have not previously been cultivated. The *Native Vegetation Conservation Act 1997* requires landholders to lodge clearing applications before cultivating areas of native grassland that have not been cultivated in the last 10 years. However, the scientific basis of this policy has not been rigorously tested. The findings of this research suggest that protection of remnant grasslands that have never been cultivated may be more important than protection of grasslands that have been cultivated more than 10 years prior. Despite this, native species richness of previously cultivated sites can sometimes be high, even if such areas have not recovered to their original composition. Grassland condition of previously cultivated sites appears to vary greatly from site to site and, hence, individual site assessments over a period of time would be the best way to determine whether or not to allow an area to be returned to cultivation.

The decreases in species richness and changes in composition at previously cultivated sites are probably related to the differing abilities of species to colonise disturbed land and cope with changes in abiotic conditions. Changes in the measured soil properties were observed in this survey with decreases in available sulphur, organic carbon and electrical conductivity at cultivated sites (Table 3.29). Römermann *et al.* (2005) similarly reported changes in soil nitrogen, phosphorus, potassium and pH due to former cultivation, and Eldridge (1999) reported that organic carbon levels were significantly lower in cultivated paddocks on the Walgett Plains, despite a recovery period of 20 years. Gough and Marrs (1990) suggested that a reduction in available phosphorus in the soil might be necessary to restore cultivated areas to species-rich grassland, given the indirect effects of fertility through competition. By contrast, a return of available sulphur and organic carbon to cultivated areas might encourage recovery of species composition on the Moree Plains.

The variability in native species richness and composition at previously cultivated sites suggests several variables influence recovery. There can be dramatic differences in vegetation immediately after cultivation (Figs 3.18, 3.19). It is likely that grazing intensity after cultivation influences recovery. Paddocks may need to be rested from grazing initially to allow establishment of grasses and other plant species that are able to quickly colonise from seed that has remained in the soil or that is easily dispersed (i.e. wind-dispersed species) (Robson 1995). Following initial recruitment, grazing may then be important in the dispersal of other species that are not passively dispersed (Römermann *et al.* 2005). The importance of grazing in the recovery of grassland diversity after ploughing was also reported by Montalvo *et al.* (1993). Unfortunately the relationship between years of cultivation and the size and composition of the seed bank is unknown on the Moree Plains. Further research into this topic is recommended.

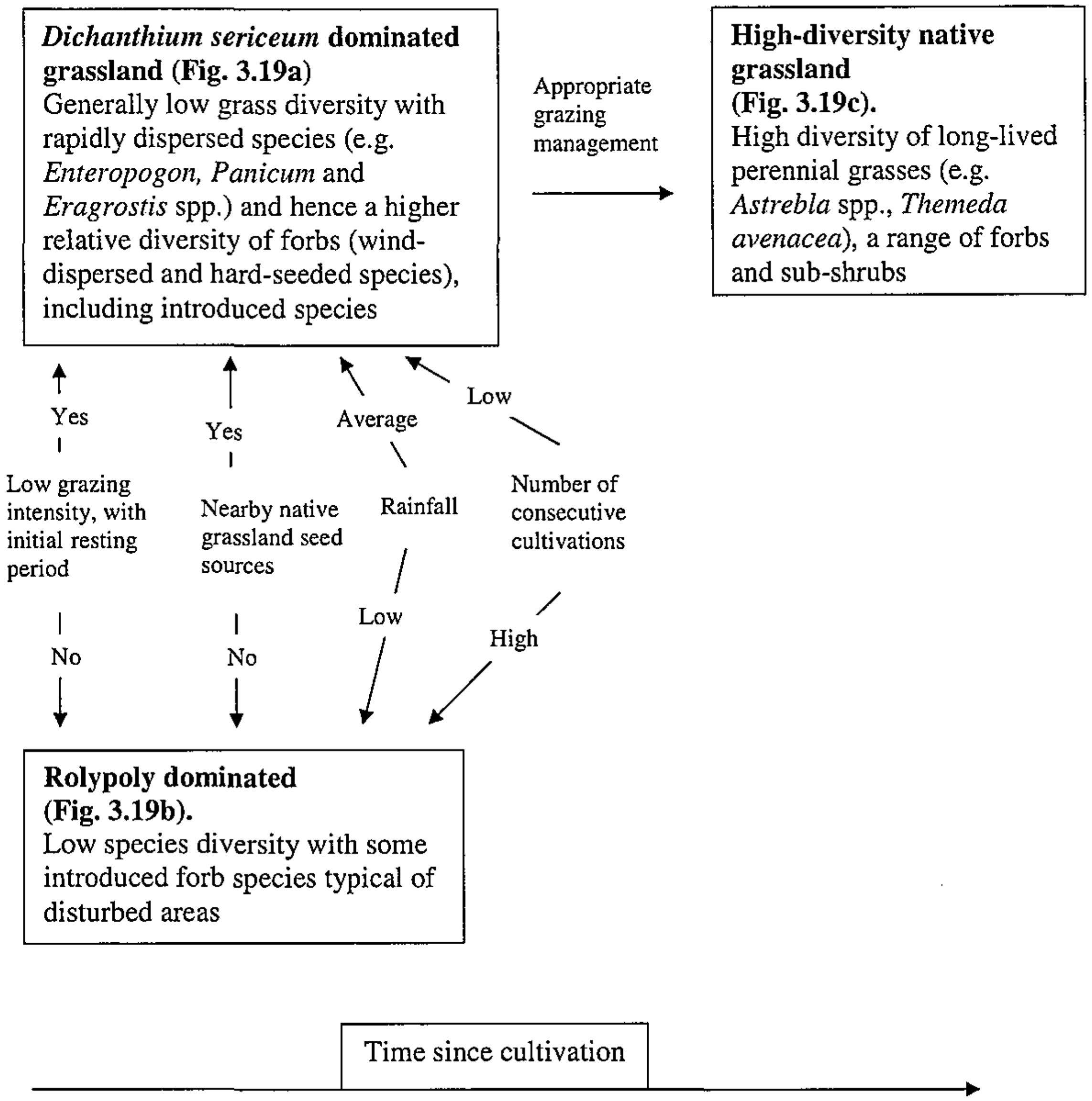


Fig. 3.18. Variables considered important in the recovery of native grasslands after cultivation on the cracking clay soils of the Moree Plains. This diagram depicts two extremes in composition that may follow immediately after cultivation of native grassland, and the desired end result. There may be a variety of compositional stages between each of these three community types. Composition may also vary from Rolypoly dominated communities to *Dichanthium sericeum* dominated grassland given appropriate grazing management and favourable seasons over time. Species composition prior to cultivation may also influence the type of community that develops following short periods of cultivation (where seeds can survive in the soil).

Fig. 3.19a. *Dichanthium sericeum* dominated grassland approximately seven years after cultivation. This site had a total of 19 species.

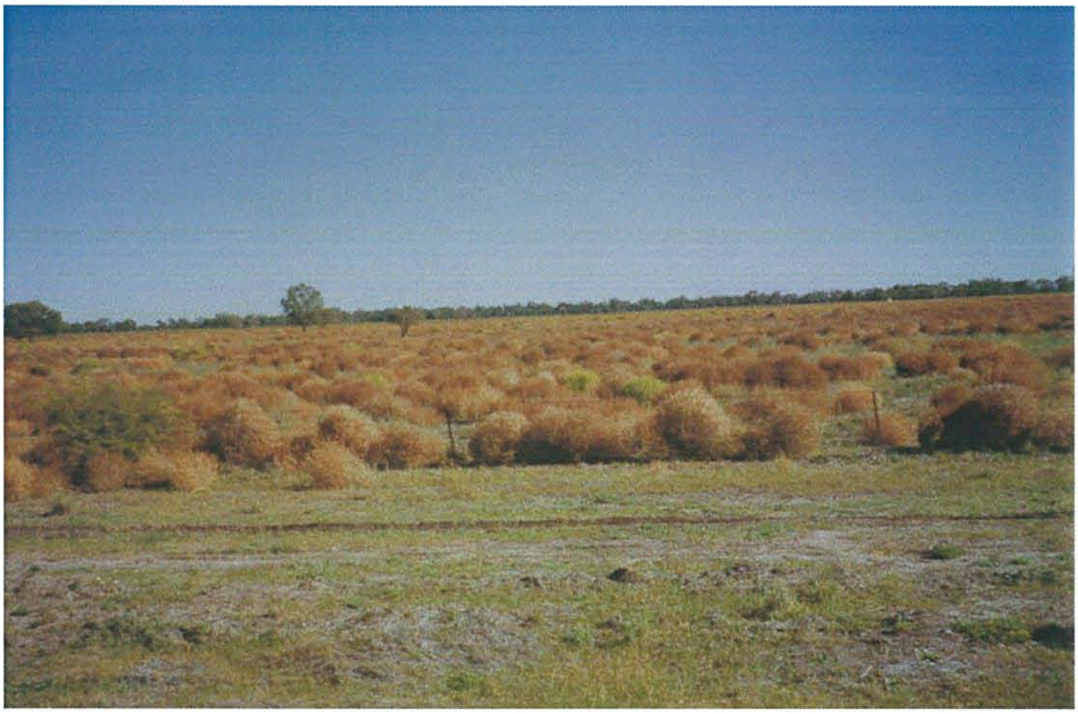


Fig. 3.19b. Recently cultivated paddock north-west of Moree dominated by Black Rolypoly (*Sclerolaena muricata* var. *muricata*).

Fig. 3.19c. Native grassland approximately 20 years after cultivation. This site had a total of 30 species, including a low abundance of *Astrebla elymoides*.

Mitchell grasses (*Astrebla elymoides* and *A. lappacea*) did not respond well to cultivation (Tables 3.26, 3.27). *Astrebla* spp. do not establish readily from seed and recruitment is episodic at intervals of 15 to 20 years or more (Bowman 1992). Hence, cultivation could be one of the reasons for the observed decline in *Astrebla* spp. abundance in north-west NSW (Bowman *et al.* 1996). Germination of *Astrebla* spp. requires adequate soil moisture and temperatures of 22°C to 38°C (Orr 1975; Roberts 1978; Orr & Holmes 1984). Rainfall after germination is also necessary to enable tillers and secondary roots to develop (Orr 1975). Seed may remain viable for up to 5 years (Bowman 1992) but maximum germination occurs after 12 months (Orr 1975), which suggests that repeated cultivation quickly exhausts seed reserves of these species. Grazing following germination may also influence seedling survival as young seedlings are easily pulled up by stock (Roberts 1978).

While the effects of past cultivation appear to be long lasting, I came across good examples of restoration following cultivation during the survey. The property 'Braden', managed by

Jim Burley, was almost all in cultivation when he bought it, but is now all native grassland. Jim (pers. comm., 2002) stressed that appropriate grazing management is important in the recovery process. Through management involving resting and grazing at strategic times, Jim has been able to establish paddocks dominated by *Dichanthium sericeum*, which can re-establish well at previously cultivated sites. Jim uses cattle to spread grass seed from paddock to paddock. In a good season, *D. sericeum* will produce new shoots and seed heads 4 weeks after sufficient rainfall. After the early seed heads have been grazed off, and spread by cattle, the species can produce more seed (given rain). By grazing paddocks intensively for short periods at appropriate times, then moving the herd to another paddock with less grass cover, Jim has been able to establish productive pasture with a high proportion of *D. sericeum*. He is now using the same principles to spread *Astrebla elymoides* seed from neighbouring stock routes, and, while the recovery of this species is slower, there has been an increase in abundance.

General impacts of management disturbances

Several species were common at disturbed sites (e.g. **Tribulus terrestris, *Urochloa panicoides, *Rapistrum rugosum, *Physalis lanceifolia, *Xanthium occidentale, Salsola kali* var. *kali, Echinochloa colona, Sclerolaena muricata* var. *muricata, *Cucumis myriocarpus, *Polygonum aviculare, Sclerolaena birchii* and *Calotis scabiosifolia*). Five of these species (**T. terrestris, *U. panicoides, *X. occidentale, *C. myriocarpus* and **P. aviculare*) were also recognised by Bean and Whalley (2001) as indicators of deterioration on the Liverpool Plains of NSW. Bowman *et al.* (1996) suggested that several of these taxa (**T. terrestris, S. kali* var. *kali, *C. myriocarpus, S. muricata* and *S. birchii*) are also negatively associated with *Astrebla* spp., and could replace *Astrebla* spp. where their density is low. Most of the species associated with degraded sites were forbs, and many were annuals or biennials (all of the introduced species except **P. aviculare*). McIntyre and Lavorel (2001) also reported increased frequency of annual forbs and grasses with increasing stocking rate, and McIntyre *et al.* (1995) reported higher proportions of therophytes at soil-disturbed sites.

The differing responses of individual native species to a range of management regimes highlights the importance of heterogeneity across the landscape. It may be important from a diversity perspective to maintain a range of different grazing regimes over the landscape. Some cultivation in the landscape may also be important, although few natives

(*Rhynchosia minima* and *Salsola kali* var. *kali*) occurred in higher frequency at cultivated sites. Animal species, such as grassland birds, are also influenced by various disturbances in different ways, and research suggests they would also benefit from a heterogeneous landscape (e.g. Boren *et al.* 1999; Fisher 1999; Barnett *et al.* 2004). The lack of grassland areas on the Moree Plains with a history of little or no grazing is a concern from a conservation perspective, because such areas are required for the persistence of grazing-sensitive species. Unfortunately grazing-sensitive species remaining in the region were not easily recognised in the current study, because their rarity prevented their detection in statistical analyses. However, some relatively common species were more frequent at ungrazed sites (e.g. Table 3.21).

Most of the statistically significant differences due to cultivation were identified within categories containing periodically grazed sites because periodically grazed sites were common on the Moree Plains, resulting in greater statistical power for detecting differences. The results suggest that large sample sizes are required to determine the impacts of past management (i.e. grazing and cultivation) given the variation in species responses and the likelihood that species responses are dependent on a range of other variables. However, this exploratory analysis has identified some trends that might be studied in more detail. Experimental manipulation would assist understanding of the changes in composition and abundance in these grasslands, and this is the focus of Chapters 5 and 6.

Flooding

Given that much of the Moree Plains are floodplains subject to varying degrees of flooding, it is not surprising that frequency of flooding influences species composition (Table 3.16). Capon (2005) recognised the importance of flood frequency in influencing plant community composition in central Australia. However, at low flood frequencies, local factors (e.g. soil type, local rainfall) had a stronger influence on community composition in both central Australia (Capon 2005) and on the Moree Plains (Table 3.16). Most of the species that were more frequent at flooded sites on the Moree Plains are known to thrive in wet conditions (e.g. *Cyperus bifax, Eleocharis pallens* and *Paspalidium jubiflorum*). Capon (2005) reported that frequently flooded areas were characterised by a high cover of annual monocots. Menges and Waller (1983) suggested that as flood frequency increased, smaller perennial forbs and tall annuals increased in importance.

Similar patterns in species composition were not recognised in the current study, probably due to the relatively low flood frequency of most flooded sites, and the confounding influences of other variables.

Floods may create an avenue for introduced species to invade (Florentine & Westbrooke 2005; Westbrooke *et al.* 2005). However, introduced species richness was higher at non-flooded sites, with higher frequencies of **Malvastrum americanum*, **Medicago polymorpha*, **M. truncatula* and **Rapistrum rugosum*. These species may be less well adapted than local native forbs for survival in flood prone areas. Apart from **M. americanum*, these species are generally winter-growing annuals, and it is possible that the less extensive root systems of annual forbs are more easily damaged by flooding. Boswell (1979) reported that perennial clovers are generally more tolerant of flooding than annual clovers, and the same may be true for medics. However, the annual, **M. polymorpha*, is common on river floodplains (Campbell 1989).

Astrebla lappacea decreased in frequency in response to flooding ($P = 0.007$; Table 3.16), supporting the findings of Bowman *et al.* (1996). *Astrebla elymoides* was not influenced by flooding in the same way. In fact, this species may benefit from some flooding. An example of this was seen on the Bunna Bunna watercourse where *A. elymoides* was the dominant species in a frequently flooded paddock. This watercourse floods every 3 years on average and generally inundates areas for at least 1 week (and up to 3 months) (N. Montgomery, pers. comm., 2002). *Astrebla elymoides* does not require flooding, however, as indicated by its dominance at Kirramingly Nature Reserve.

While inundation period did not significantly influence composition, the response of different grassland species to varying degrees of flooding and inundation requires further investigation, given the potential changes in natural flooding regimes brought about by development of irrigated agriculture. Season of flooding may also influence species composition as flooding in late summer is thought to have a more severe effect on perennial summer-growing grasses, than flooding at other times of the year (R. Whalley, pers. comm., 2004).

Fire

Fire appeared to have little impact on species composition or richness at the paired sites (Table 3.33). This is not surprising given the differing histories of the sites. Fires at these sites were started at differing times of the year, in different years and with differing intensities. However, it is important to realise that fire did not result in an influx of introduced species as reported by Lunt (1990a), and Milberg and Lamont (1995) in southern Australia. To better define the influence of fire on grassland species composition and abundance, a controlled experiment was carried out at Kirramingly Nature Reserve, and is reported in Chapter 5.

3.4.6 Grassland condition and taxa of conservation significance

Most of the nature reserve sites (four out of seven) were considered to be in good condition. Two of the nature reserve sites that were not in good condition had a history of cultivation or low rainfall prior to sampling. This is encouraging for the managers of reserves where the aim is to promote biological diversity (e.g. NPWS-managed reserves). Grassland condition in this study was primarily defined by plant species richness (native and introduced) and was determined after surveys had been carried out. While this index does not cover all aspects of condition (e.g. species evenness and productivity) it was considered adequate for the purposes of this study. On the Moree Plains grassland condition should ideally be assessed on several occasions to take into account reductions in richness during dry periods (e.g. Fig. 3.6).

Several plant taxa of high conservation significance occur within the Moree Plains grasslands. Two endangered species listed under the *Threatened Species Act* (NSW) were recorded during a previous survey at Kirramingly Nature Reserve (Clarke *et al.* 1998). *Desmodium campylocaulon* (a scrambling herbaceous pea) was recorded at 11 sites, and *Sida rohlenae* was collected at one location, although confirmation of its occurrence was suggested (Clarke *et al.* 1998). This species was not recorded in the present study. In addition to these, Hunter and Earl (1999) recorded four other taxa of high conservation significance: *Bothriochloa biloba*, *Swainsona murrayana* (both listed as Vulnerable at the time), *Lepidium hyssopifolium* and *Phyllanthus maderaspatanus* (both listed as Endangered). Neither of these surveys reported the presence of *Digitaria porrecta*, the only new threatened species recorded in the present study. However, this species was noted on the property 'Kirramingly', neighbouring Kirramingly Nature Reserve (R.

Whalley, pers. comm., 2002). Hunter and Earl (1999) reported eight species, which, they believed are of regional conservation significance. Two of these species, *Aristida leptopoda* and *Commelina ensifolia*, were recorded in the current survey. *Aristida leptopoda* was the 15th most frequent species recorded and is commonly a dominant species with *Astrebla elymoides* or *Dichanthium sericeum* on the Moree Plains.

The fact that a high proportion (29%) of species was recorded only once or twice indicates that there is a large number of rare species in these grasslands. However, as this survey did not cover a broad range of vegetation communities, it is likely that many of these species are more common components of other communities. It is also possible that a proportion of these species is only seasonally abundant, and sampling did not coincide with their period of abundance. Despite this, I would recommend further studies of these infrequently occurring species to determine whether they are in fact regionally rare. McIntyre *et al.* (1993) similarly reported that very few species were common (the majority were recorded at <5% of sites), but were usually widely occurring, in a survey on the Northern Tablelands. Such sparsely occurring species may be under-represented in lists of threatened plants (McIntyre 1992; McIntyre *et al.* 1993).

Desmodium campylocaulon was relatively common in this survey (ranked the 44th most frequent species) and appears to be a natural component of *Astrebla* spp. and *Dichanthium sericeum* dominated grasslands. Cavallaro (2001) suggested that this species may not be endangered, but rather is poorly known. Its rarity in NSW appears to be due to the major reductions and fragmentation of the communities in which it occurs. The presence or absence of grazing, cultivation or flooding did not explain the occurrence of *Desmodium campylocaulon*. However, this species is possibly influenced by grazing intensity, as it was more frequent at privately owned grazed sites (occurring at 29% of such sites) than grazed TSR sites (occurring at 10% of sites). In support of this, Fensham *et al.* (1999) reported a decrease in the abundance of this species at high levels of grazing disturbance.

Swainsona murrayana was only recorded at three sites during the regional survey. While this is not a large enough sample size to predict its response to disturbance, it was restricted to sites with low levels of grazing, or no grazing (including one nature reserve site, one site on private property that had not been grazed in 6 months and one TSR site that had not been grazed in the 3 months prior to sampling). The decline of *Swainsona* *murrayana* is probably a reflection of the palatability and upright habit of *Swainsona* spp., which makes them readily accessible to grazing animals (McDougall 1989).

The endangered grass, *Digitaria porrecta*, may be sensitive to flooding (P = 0.080), and I would recommend further work on this species to determine its ecological tolerances. The variable responses of species (particularly dominants like *Astrebla lappacea*) to differing flooding regimes supports the ordination findings which suggest that changing natural flooding occurrence, or frequencies, will have a significant influence on grassland composition. A number of landholders I spoke to raised concerns about the development of extensive flood irrigation farming and the influence of diversionary earthworks on natural flooding patterns.

3.4.7 Conclusions

Seasonal changes in composition are to be expected in these grasslands which are dynamic in their response to rainfall. Differences in herbaceous composition and richness between years and seasons are, hence, not surprising. The decline in species richness during the very dry winter of 2002 is of interest, but recovery of species richness soon took place after summer rains. This suggests that species that were not apparent in quadrats during the drought were able to survive drought conditions by either recruiting from seed or resprouting from rootstock or rhizomes after sufficient rainfall. Increased amounts of bare ground caused by drought are likely to have resulted in increased abundances of certain weed species (e.g. **Rapistrum rugosum*) that have an ability to quickly re-colonise such areas.

The fact that environmental variables have a greater influence on species composition than management has led many land managers in the region to believe that the native grasslands of the Moree Plains are resilient to disturbances like cultivation and grazing, and will 'bounce back' after rain. While many native species are able to persist in, or recolonise disturbed areas, this chapter has highlighted some of the more subtle changes in species composition and richness associated with anthropogenic disturbances and natural disturbances (i.e. flooding). While slight reductions in species richness, and decreases in abundance of certain native species in response to cultivation or grazing may not be of concern to graziers (assuming productivity is not decreased), such changes are of concern from a conservation perspective.

Both grazing and cultivation have an influence on species composition in Moree Plains grasslands. The effects on native and introduced richness are variable, as many species have varied responses, highlighting the need for a heterogenous landscape with a mixture of grazed and ungrazed (i.e. nature reserves) areas. Recovery of species richness after cultivation varies greatly in response to a range of variables, such as interactions with grazing after cultivation. The overall negative effect of cultivation on native species richness and composition is of concern, and it should not be assumed that cultivated grasslands will recover to grassland of good condition. Conservation of areas that have not already been cultivated should be a priority on the Moree Plains.

CHAPTER FOUR

The influences of grazing, cultivation and time of sampling on native grassland composition and richness at Kirramingly Nature Reserve

4.1 Introduction

It is generally accepted that long-term removal of grazing and other disturbances in grasslands results in a reduction in native plant diversity (Stuwe & Parsons 1977; Collins & Barber 1985; Trémont 1994; Trémont & McIntyre 1994; Noy-Meir 1995; Morgan & Lunt 1999). Many studies have investigated the influences of livestock grazing using exclosures (e.g. Williams & Roe 1975; Foran *et al.* 1982; Sneva *et al.* 1984; Brady *et al.* 1989; Noy-Meir *et al.* 1989; Orr & Evenson 1991a; Trémont 1994; Trémont & Whalley 1995; Stohlgren *et al.* 1999; Spooner *et al.* 2002; Valone & Sauter 2005). Studies that examined differences in diversity often report a reduction in plant diversity after removal of grazing (Noy-Meir *et al.* 1989; Belsky 1992, 1986b; Trémont 1994; Trémont & Whalley 1995) but there are some studies that report the opposite, or no effect (Brady *et al.* 1989; Stohlgren *et al.* 1999; Spooner *et al.* 2002). Different grasslands respond in different ways to grazing exclusions depending on variables like rainfall, soil fertility and the height and competitiveness of dominant grasses (Olff & Ritchie 1998).

Kirramingly Nature Reserve is the largest area of remnant Mitchell grassland in the Northern Outwash Province of western NSW (T. Waters, pers. comm., 2001). The Reserve presents a good location to study the impacts of grazing and cultivation on the Moree Plains at a property scale rather than a regional scale (i.e. Chapter 3). It is one of the few areas, and certainly the largest area, of native grassland on the Moree Plains that is not grazed by livestock. This provides an opportunity to study the impacts of livestock grazing removal over time, by comparing sites within the Reserve to grazed sites surrounding it. It was hypothesised that removal of grazing from the Reserve will have resulted in a reduction in species richness (i.e. interstitial species). This could present a problem because according to the *National Parks and Wildlife Act 1974*, the NPWS aims to manage the Reserve to conserve flora and fauna and maintain native biodiversity (Anon. 2002), but in Australia livestock grazing is usually excluded from national parks and reserves.

Kirramingly Nature Reserve was formerly a pastoral property and some paddocks were sown with cereal crops up until 1991 (P. Wall, pers. comm., 2002). A survey of the vegetation of the Reserve was carried out by Clarke *et al.* (1998), who established vegetation monitoring points in paddocks (referred to as management units) with different management histories. By revisiting monitoring sites, it is possible to follow the recovery of previously cultivated management units over time, in the absence of livestock grazing since 1996. Relatively few studies have investigated the effects of past cultivation on grassland diversity in Australia. While the effects of cultivation are variable, most studies report a decrease in plant diversity after extended periods of cultivation (Austrheim & Olsson 1999; McIntyre *et al.* 2002; Römermann *et al.* 2005; Chapter 3). The time it takes for grasslands to recover after several years of cultivation is largely unknown, but it is expected that native plant richness will increase with time since cultivation (Austrheim & Olsson 1999; Nadolny & Eldridge 1999).

Revisiting the sites previously established at the Reserve also allows investigation of the temporal variation in these grasslands, which is known to have an important influence on composition (Clarke *et al.* 1998; Hunter & Earl 1999) and richness (Section 3.3.2). It is important that this temporal variation is understood when determining the influence of other variables. When monitoring sites were established at the Reserve by Clarke *et al.* (1998) they were stratified to incorporate areas close to dams and areas with a tree canopy. These authors reported that dams might be a source of seeds for introduced species and suggested such areas be monitored. Two sites with *Casuarina cristata* canopy (i.e. Belah woodland) were recognised as being floristically distinctive (Clarke *et al.* 1998). In addition, Clarke *et al.* (1998) recommended that the abundance of *Acacia farnesiana* be monitored over time. Although this species did not have an influence on composition (Clarke *et al.* 1998), it may represent a potential threat to grassland diversity at higher abundances. Hence the influence of proximity to dams, tree cover and *A. farnesiana* abundance on species composition over time was of interest in the current research.

There is a lack of scientific information on how to best manage native grasslands on the Moree Plains for the conservation of biodiversity. This research aims to provide data to guide the management of Kirramingly Nature Reserve to enhance or maintain native plant diversity. The major objectives of this chapter are to determine:

1. The variability in grassland community composition and richness over time (between different years and months of sampling) at Kirramingly Nature Reserve.
2. The impacts of livestock grazing exclusion from Kirramingly Nature Reserve on native grassland composition and richness.
3. The impacts of past cultivation at Kirramingly Nature Reserve on native grassland composition and richness.
4. Whether *Acacia farnesiana* abundance, presence of tree canopy and proximity to dams has an influence on grassland composition.

4.2 Methods

4.2.1 Background and sampling methods

A vegetation survey of Kirramingly Nature Reserve and surrounding stock routes was conducted between February and May 1998 by Clarke *et al.* (1998). The survey was carried out under contract to the NSW NPWS. Clarke *et al.* (1998) established 22 monitoring sites representing the different vegetation communities of the Reserve, with 18 sites situated in the Reserve and four on the surrounding stock routes (Fig. 4.1). All sites were marked with a labelled steel post in the centre of the quadrat and their GPS locations recorded.

The frequency-score method of sampling was used to be consistent with previous surveys (Clarke *et al.* 1998). The theoretical basis for this method is provided by Morrison *et al.* (1995). The method is useful for providing quantitative information on species abundances and richness at different spatial scales. At each site a series of ten nested concentric sub-quadrats, each doubling in sample area, was sampled (i.e. 2, 4, 8, 16, 32, 64, 128, 256, 512 and 1024 m^2). Concentric sub-quadrats were surveyed using four strings, marked with coloured tape at the appropriate intervals. Strings were aligned north, south, east and west from the centre of each quadrat. The incidence of all vascular plant species was recorded in each of the concentric sub-quadrats, and a score out of ten assigned to each species in the 1024-m^2 quadrat, based on the number of sub-quadrats in which it occurred. Each species was also given a Braun-Blanquet cover abundance score (Table 3.1) for consistency with other data sets. Percentage of bare ground was estimated for each quadrat.

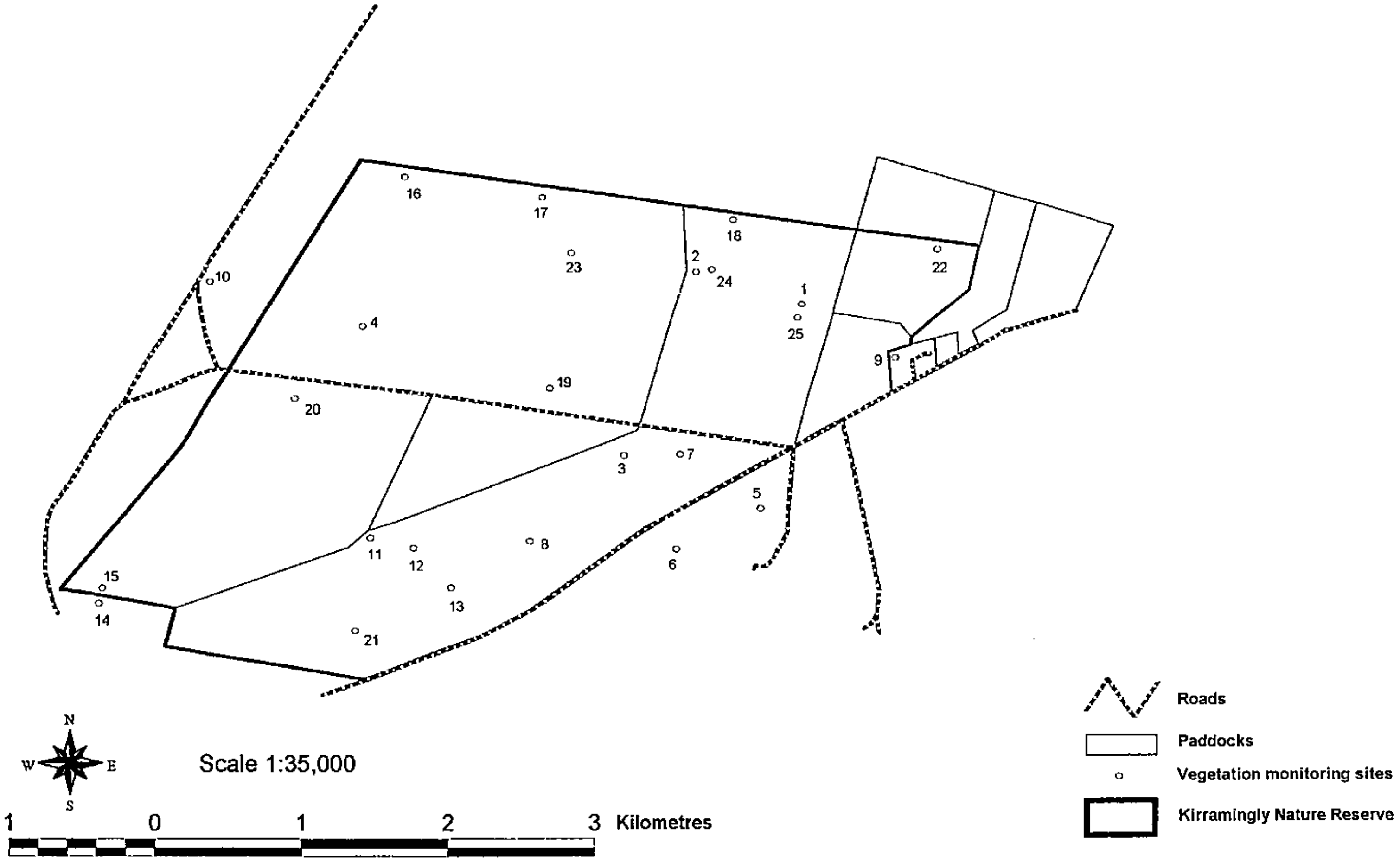


Fig. 4.1. Kirramingly Nature Reserve and location of the 22 monitoring points established by Clarke *et al.* (1998) and three additional points added in 1999 (23, 24 and 25). Sites 10, 14, 5 and 6 are on the surrounding TSR. Modified from Clarke *et al.* (1998).

4.2.2 Sampling times

The sites established by Clarke *et al.* (1998) were re-surveyed, to determine how vegetation composition and abundance varied over time at Kirramingly Nature Reserve. The NSW NPWS contracted the University of New England, Division of Botany to re-survey five sites in November of 1999 and establish a further three sites on the northern boundary of the Reserve (Fig. 4.1). Three separate surveys were carried out during the PhD research reported here. Six sites were revisited in December 2001, all 25 sites were sampled in February-March 2002 and 15 sites were surveyed in March 2004. Hence, there have been five vegetation survey periods at the Reserve: two small surveys (6-8 sites) between November and December, and three large surveys (15-25 sites) between February and March (Table 4.1). Data from all surveys were analysed.

Table 4.1. Vegetation surveys carried out at monitoring points at Kirramingly Nature Reserve (and surrounding TSR)

Survey	Sampling time	Number of sites
Botany / NPWS (Clarke *et al.* 1998)	Feb-Mar 1998	22
Botany / NPWS (unpublished)	Nov 1999	8 (including 3 new sites)
Tom Lewis (this research)	Dec 2001	6
Tom Lewis (this research)	Feb-Mar 2002	25
Tom Lewis (this research)	Mar 2004	15

4.2.3 Site stratification and management history

The original survey by Clarke *et al.* (1998) established sites to reflect the different vegetation types of the Reserve and a range of environmental attributes. Four sites were established in paddocks that had been previously cultivated, four sites were established on the surrounding TSRs, two sites were located within 100 m of dams on the Reserve, and three sites were established with tree canopy cover (two of which were TSR sites). After the 1998 survey one site (site 9) on the homestead block of Kirramingly was sold by NPWS. This site was subsequently grazed, and was grouped with the TSR sites when re-sampled in 2002. Of the three new sites added in November 1999, two were within previously cultivated paddocks, giving a total of six previously cultivated sites in the 2002 survey (Table 4.2). One of the sites added in November 1999 was also within 100 m of a dam, giving a total of three sites in close proximity to dams in the 2002 survey. Two other variables were added to the data set; survey period (Table 4.2) and the abundance of *Acacia farnesiana.* All these attributes were defined as environmental variables in multivariate analysis.

Sites were defined as having either abundant *Acacia farnesiana* (cover score ≥3, i.e. ≥6%) or little or no *A. farnesiana* (cover score <3). Six sites had abundant *A. farnesiana* at all sampling times, including one TSR site. The rate of population increase or decrease of *A. farnesiana* at Kirramingly Nature Reserve was estimated from frequency scores, by comparing scores from 1998 to those obtained in 2002 and 2004.

Table 4.2. Environmental variables used in multivariate analysis, and their codes used in tables and figures

Environmental variable	Variable categories	Code	Total *n*
Sampling period	1) Feb-May 1998	1) Mar 1998	22
	2) Nov 1999	2) Nov 1999	8
	3) Dec 2001	3) Dec 2001	6
	4) Feb-Mar 2002	4) Mar 2002	25
	5) Mar 2004	5) Mar 2004	15
Cultivation	1) Yes	1) Cultivated	6
	2) No	2) Not cultivated	19
Grazing	1) Yes	1) Grazed	5
	2) No	2) Reserve	20
Proximity to dam	1) <100 m	1) Dam <100 m	3
	2) ≥100 m	2) Dam ≥100 m	22
Presence of tree canopy	1) Yes	1) Canopy	3
	2) No	2) Open	22
Cover of *Acacia farnesiana*	1) Cover score ≥3	1) Acafar	6
	2) Cover score <3	2) Few Acafar	19

Kirramingly was grazed only by sheep until 1977, and was grazed only by a small number of cattle (40) until it was destocked in 1996. While stocking rates for the Reserve are unavailable, anecdotal evidence suggests it was overstocked from approximately 1929 to 1945 (P. Wall, pers. comm., 2002). After 1945, Kirramingly was not heavily stocked, and was destocked during dry periods. *Astrebla* spp. seed was spread by Sid Nicholson (former manager) in the 1920s. Seed from three introduced grasses, Rhodes Grass (**Chloris gayana*), Purple Pigeon Grass (**Setaria incrassata*) and Giant Panic (**Panicum antidotale*) was also spread through part of the Reserve, but did not establish (P. Wall, pers. comm., 2002). Over the course of this study the TSRs surrounding the Reserve have been subject to intermittent, heavy grazing. These TSRs differ to those on some other parts of the Moree Plains where grazing can be almost continuous (Austen 2002).

Five cultivated sites were first cultivated in 1967 and cropping ceased in 1982. The other was first cultivated in 1957 and last cultivated in 1991 (P. Wall, pers comm., 2003). All sites were only occasionally cultivated after 1977. Paddocks were sown with mainly oats and wheat. No fertiliser was used in paddocks in which sites are located (P. Wall, pers. comm., 2003).

The Reserve does not get flooded from nearby watercourses. However, some areas can be inundated for 3-4 days after heavy summer rains (P. Wall, pers. comm., 2002). Fire was not used in the management of the property and no monitoring sites had been burnt between 1960 and 1999. In 1999, a hazard reduction burn on the northern boundary of the Reserve, burnt four monitoring sites. The influence of fire on these grasslands will be the focus of the following chapter.

4.2.4 Rainfall

A total of 257 mm fell at Moree (35 km from Kirramingly) in the 4 months prior to sampling in February 1998, 238 mm fell prior to November 1999 sampling, 166 mm fell prior to December 2001 sampling, 196 mm fell prior to March 2002 sampling and 300 mm fell prior to March 2004 sampling (Fig. 4.2). The years of 1998, 1999 and 2001 had above average rainfall with 748 mm, 717 mm and 805 mm, respectively. Rainfall was particularly low in 2002 (277 mm for the year).

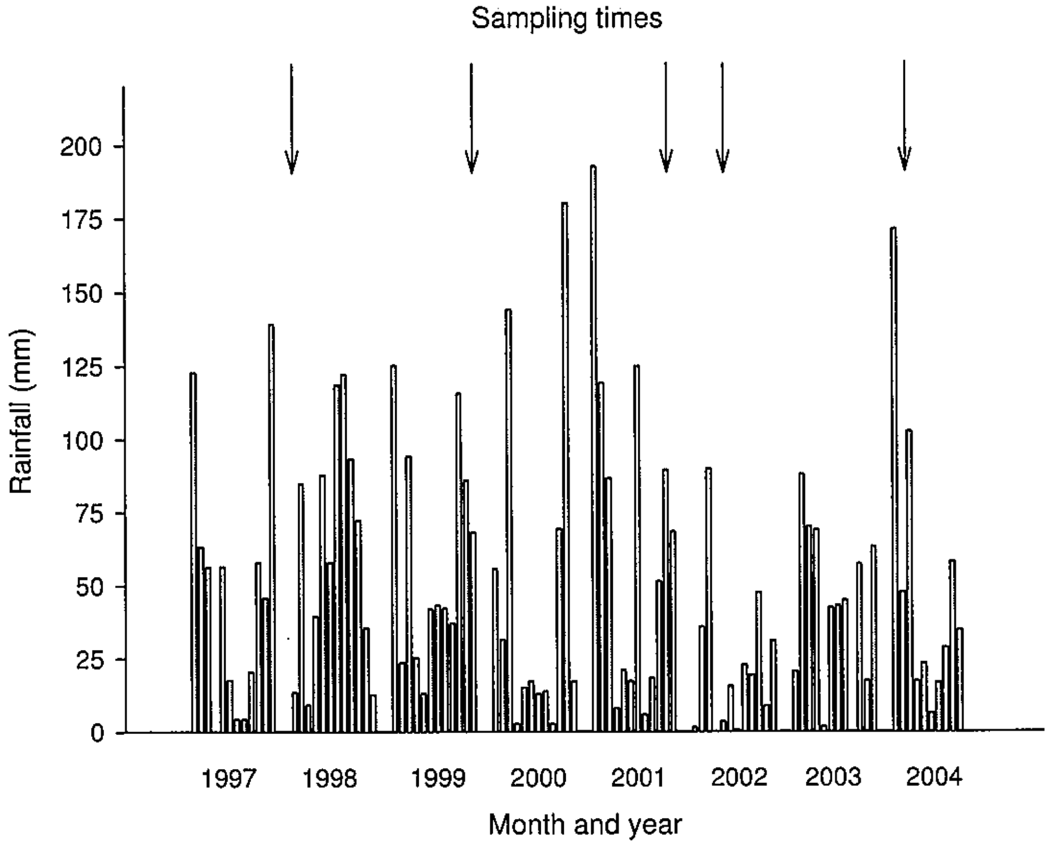


Fig. 4.2. Monthly rainfall (individual bars) for Moree from January 1997 to October 2004 with arrows showing sampling times for all surveys at Kirramingly Nature Reserve. Rainfall in each year is separated by a space (equivalent to 2 months). Data obtained from Commonwealth Bureau of Meteorology (2004).

4.2.5 Statistical analysis

Multivariate analysis was carried out using CANOCO, version 4.5 (ter Braak & Šmilauer 2002). All environmental variables were nominal (Table 4.2); hence, dummy variables were defined prior to analysis, as in Chapter 3. The final data set consisted of six environmental variables (with 15 categories), five surveys (76 sites sampled, spread over the 25 monitoring sites) and 153 species. Apart from sampling time, which had five categories, all other variables had only two categories (Table 4.2). Given the relatively small number of sites and the number of variable combinations when all categories of all variables are multiplied out (32, excluding sampling time), caution is required in interpreting the ordination results. It is likely that there will be confounding of variables because some variable combinations were common and others could not be sampled. A more balanced sampling design for all variables of interest would have provided a more unequivocal view of the effects of each variable.

Indirect gradient analysis using detrended correspondence analysis (DCA) produced a gradient length of 2.48. This is a short gradient length suggesting species were responding roughly linearly to the measured environmental gradients. No major unmeasured gradients were detected using indirect gradient analysis. Due to the short gradient length, redundancy analysis (RDA) was chosen as the most appropriate ordination technique. Canonical correspondence analysis (CCA) was also carried out to confirm that relationships between variables and species were similar using both methods. Section 3.2.7 outlines the options chosen for analysis. Infrequent species (with two occurrences or less) were removed from the data set prior to analysis. *Acacia farnesiana* was also deleted from the analysis, so as not to influence composition, when determining the influence of its abundance on other species. Species frequency data (score out of ten, described in Section 4.2.1) and species cover scores were analysed. Both analyses produced similar results and, hence, only results from analysis of species cover scores are presented. To determine the influence of environmental variables without sampling time, a partial RDA was carried out with sampling time as a covariable. This allowed partitioning of the variance to determine the relative influence of sampling time and the other environmental variables on species composition (Palmer 2001).

Automatic forward selection was used to determine the rank of each environmental variable and its relative importance in explaining the species cover data (ter Braak & Šmilauer 2002). The conditional and marginal eigenvalues from automatic forward selection are presented for variables that contributed significantly (at the 5% level) to the model. Biplots were constructed as outlined in Section 3.2.7. Total, native, introduced, forb, grass, graminoid, shrub, perennial and annual species richness were added as supplementary variables *post hoc*. To determine which species were correlated with different explanatory variables, t-value biplots were constructed, where biplot projections are used to approximate the t-values of the regression coefficients that would be obtained from simple regression with one predictor (explanatory variable or variable category) and one response (species) (ter Braak & Šmilauer 2002). Refer to Section 3.2.7 for more information on interpretation of t-value biplots.

Changes in total, native, introduced, annual, perennial, grass, forb, graminoid and shrub species richness over time were graphed for all sites sampled. Analysis of variance (ANOVA) was carried out using the software package GenStat (6th edition) to examine the influence of grazing, cultivation and sampling time. Response variables included total, native, introduced, forb, grass, graminoid and shrub species richness. Due to the low number of shrubs, shrub richness included both sub-shrubs and shrubs. Square root transforms were applied to satisfy the assumptions of ANOVA for total, native, introduced, forb, grass and shrub species richness. The response variable, graminoid richness, could not be transformed to meet the assumptions of ANOVA and was analysed using nonparametric Kruskal-Wallis ANOVA.

Repeated measures ANOVA was used because the same sampling quadrats were visited at successive data collections (i.e. sampling was not re-randomised for each sampling time). Degrees of freedom were multiplied by correction factors before calculating *F* probabilities in the analysis due to asymmetrical variance/covariance matrices over time. Only sites sampled in late summer-autumn (three sampling periods) were included in repeated measures analysis given the small number of sites sampled at other times and possible confounding effects of sampling season. The occurrence of tree canopy and proximity to dams were covariates in the analysis. Proximity to dams was only relevant in comparing cultivated and uncultivated sites because two of the previously cultivated sites were within 100 m of dams, but no grazed sites occurred near dams. Occurrence of tree canopy was only relevant in comparing grazed and ungrazed sites because two grazed sites were influenced by tree canopy, but no cultivated sites had tree canopy. It was not possible to investigate interactions between cultivation and grazing because none of the grazed sites (TSRs) had a history of cultivation. Unplanned, one degree of freedom contrasts were used to identify the effects of grazing and cultivation in 1998, 2002 and 2004 after looking at graphs to determine where contrasts were necessary. Contrasts were also used to identify differences between sampling times. The number of grazed, ungrazed, cultivated and uncultivated sites at each late summer-autumn sampling period are provided in Tables 4.3 and 4.4.

Fisher's exact test (for 2 x 2 contingency tables) or Pearson chi-square test was used to identify differences in the frequency of species between (1) the 15 quadrats repeatedly sampled in 1998, 2002 and 2004; (2) grazed and ungrazed quadrats; and (3) cultivated and uncultivated quadrats. Species with less than five occurrences for each test were ignored, because low frequencies could have been due to chance. For certain species, frequency scores per quadrat (frequency score out of ten) were used to compare changes in abundance between 1998 and 2002 using paired t-tests. All results significant at the 5% level are reported. Repeated measures ANOVA was used to determine whether *Acacia farnesiana* had increased or decreased in frequency per quadrat over time at cultivated and uncultivated sites within the Reserve and at grazed sites outside of the Reserve.

Table 4.3. Number of TSR (grazed) and Reserve (ungrazed) sites (only sites without a history of cultivation and >100 m from dams) at each late summer-autumn sampling period

Survey	Grazed	Number of sites
Mar 1998	Yes	4
Mar 1998	No	11
Mar 2002	Yes	5
Mar 2002	No	11
Mar 2004	Yes	3
Mar 2004	No	8

Table 4.4. Number of cultivated and uncultivated sites within Kirramingly Nature Reserve (only sites without canopy cover) at each late summer-autumn sampling period

Survey	Cultivated	Number of sites
Mar 1998	Yes	4
Mar 1998	No	12
Mar 2002	Yes	6
Mar 2002	No	13
Mar 2004	Yes	3
Mar 2004	No	9

4.3 Results

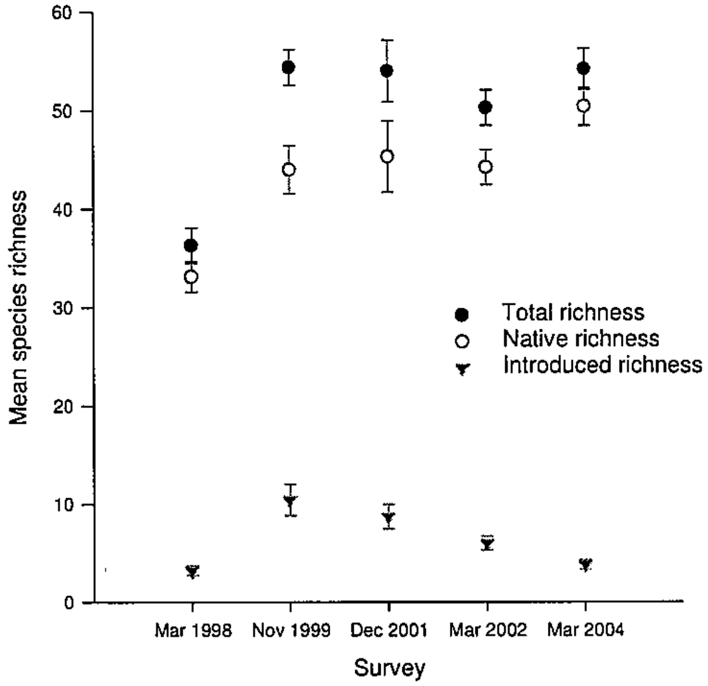
4.3.1 Changes in composition and richness over time

A total of 194 taxa and 49 plant families were recorded in quadrats at Kirramingly Nature Reserve and the surrounding stock route over all sites and times (Appendix 4.1). Most taxa (173) were recorded during surveys carried out in the current research (Appendix 4.1). Approximately 80% of all taxa were native. The ten most frequently occurring taxa (and their mean (± SE) frequency score out of 10) recorded during all surveys at the Reserve were: *Dichanthium sericeum* (7.1 ± 0.39), *Eriochloa pseudoacrotricha* (6.9 ± 0.42), *Chloris truncata* (6.9 ± 0.38), *Astrebla elymoides* (6.4 ± 0.45), *Panicum decompositum* (6.4 ± 0.40), *Boerhavia dominii* (6.3 ± 0.40), *Neptunia gracilis* (6.3 ± 0.44), *Sida trichopoda* (6.3 ± 0.38), *Sclerolaena muricata* var. *villosa* (6.1 ± 0.42) and *Aristida leptopoda* (6.0 ± 0.40).

Species richness varied greatly between sampling times at Kirramingly Nature Reserve (Fig. 4.3). Total richness ranged from 27 to 72 species per 0.1024-ha quadrat and was obviously lower in 1998 than at other sampling times (Fig. 4.3a). Total species richness significantly increased between sampling in 1998 and 2002 (Fig. 4.3a; $F_{1,21} = 59.02$, $P < 0.001$), but not between 2002 and 2004 (Fig. 4.3a; $F_{1,14} = 2.86$, $P = 0.113$). Native richness did, however, increase between 2002 and 2004 (Fig. 4.3a; $F_{1,14} = 8.79$, $P = 0.010$). Perennial species richness followed a similar trend to native species richness (Fig. 4.3a,b). The increase in native and perennial species richness appeared to be due to a gradual increase in grass species richness over time (Fig. 4.3c) and in shrub/sub-shrub richness after December 2001 (Fig. 4.3d). Graminoid species richness did not vary greatly over time (Fig. 4.3d).

Introduced and annual species richness was generally higher in November 1999 and December 2001 than in late summer-autumn (Fig. 4.3a,b). Most of the introduced and annual species that were common in November and December were forbs (Fig. 4.3c). Several species were common in November and December that were either not present, or not common, during the late summer-autumn sampling times. Such species included **Hedypnois rhagadioloides, *Medicago polymorpha, *Plantago cunninghamii* and **Rapistrum rugosum.* Several other less common species were only recorded in November or December, such as **Rumex crispus, *Phalaris paradoxa, Daucus glochidiatus, *Lactuca serriola, *Avena fatua* and **Medicago truncatula.* The four occurrences of *Swainsona murrayana* were also only at these times. This species was probably present at other times, but was not recorded because of its inconspicuous nature when not flowering.

(a)



(b)

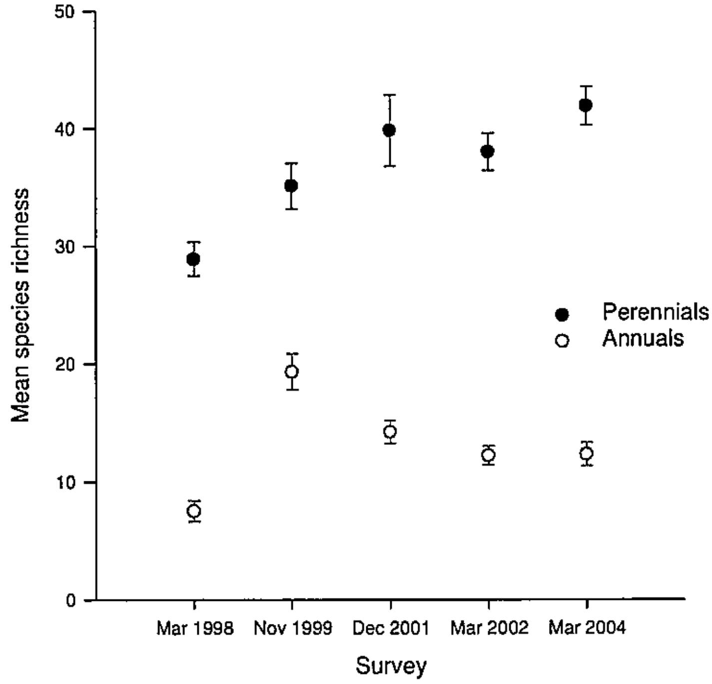
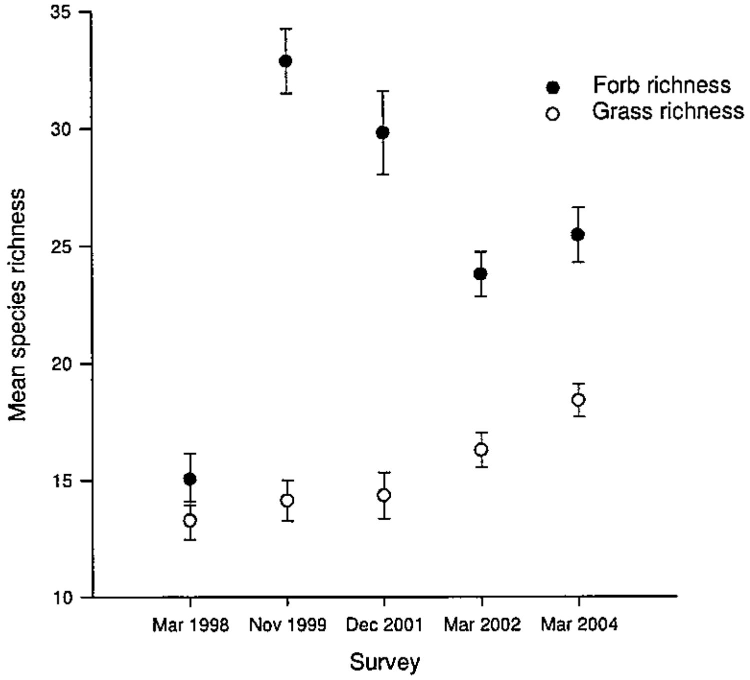


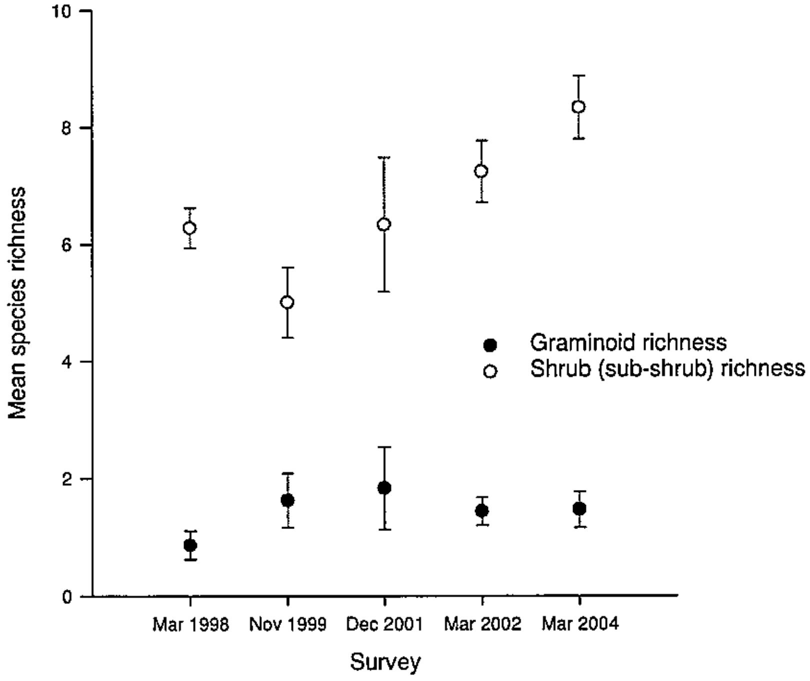
Fig. 4.3. Changes in species richness through time at all sites surveyed at Kirramingly Nature Reserve. (a) Total, native and introduced species richness; (b) Perennial and annual species richness; (c) Forb and grass species richness; and (d) Graminoid and shrub species richness. Number of sites sampled at each time are provided in Table 4.1.

Fig. 4.3. (continued)

(c)



(d)



Frequencies of all species recorded during surveys at Kirramingly Nature Reserve are provided in Appendix 4.2. Some 23 species varied significantly in frequency of occurrence between 1998 and 2004 (Table 4.5). Of these species, 21 increased after 1998 (Table 4.5). No species decreased consistently in frequency over time. Winter-active annual species such as **Conyza bonariensis, *Rapistrum rugosum* and **Sonchus oleraceus* were more common in 2002 only, while *Dactyloctenium radulans, Einadia polygonoides, Hibiscus trionum, *Physalis lanceifolia, Pimelea microcephala* and *Trianthema triquetra* were more common in 2004 (Table 4.5). Twelve species increased consistently in frequency at each sampling time between 1998 and 2004 (Table 4.5).

When comparing mean frequency scores per quadrat (i.e. out of 10 sub-quadrats) for the 22 sites visited in 1998 and 2002 some of the most abundant species also varied in frequency. *Dichanthium sericeum* increased significantly in frequency from 5.8 ± 0.88 scores per 0.1024-ha quadrat in 1998 to 7.5 ± 0.75 scores per quadrat in 2002 ($t = -2.67$, d.f. = 21, $P = 0.014$). *Aristida leptopoda* increased from 4.1 ± 0.67 to 7.1 ± 0.78 scores per quadrat ($t = -3.64$, d.f. = 21, $P = 0.002$) and *Neptunia gracilis* increased from 2.4 ± 0.67 to 7.5 ± 0.71 scores per quadrat ($t = -6.11$, d.f. = 21, $P < 0.001$). The threatened (*TSC Act*) forb, *Desmodium campylocaulon*, also increased significantly in frequency from 2.6 ± 0.76 to 4.1 ± 0.89 scores per quadrat ($t = -3.49$, d.f. = 21, $P = 0.002$).

4.3.2 Ordination output

Redundancy analysis with all variables produced eigenvalues for the first four ordination axes of 0.095, 0.073, 0.051 and 0.038. Sampling time and cultivation were the two most important explanatory variables for predicting species composition (Table 4.6). Partial RDA with time as a covariate produced eigenvalues of 0.069, 0.049, 0.031 and 0.026. Cultivation and tree canopy were the most important explanatory variables for predicting composition, explaining 8.4% and 5.9% of the total variance respectively. Environmental variables (other than sampling time) in partial ordination explained 26.7% of the variance. Most of the variance (65.1%) was unexplained and 8.2% was explained by sampling time. Forward selection suggested all variables had some influence on species composition (Table 4.6), although the variance explained by each variable may not be biologically significant (Section 3.2.7).

Table 4.5. Frequency of occurrence (%) for individual species varying significantly ($P < 0.05$) (chi-square test, d.f. = 2) over time at 15 sites visited in summer-autumn 1998, 2002 and 2004 at Kirramingly Nature Reserve (12 sites) and surrounding TSRs (3 sites)

	1998 ($n = 15$)	2002 ($n = 15$)	2004 ($n = 15$)	χ^2	P
Increased between 1998 and 2002 only					
**Conyza bonariensis*	33	87	7	20.40	<0.001
Enteropogon acicularis	67	100	100	11.25	0.004
Haloragis aspera	0	47	47	10.16	0.006
Neptunia gracilis	53	100	100	16.58	<0.001
Oxalis perennans	47	100	93	15.83	<0.001
**Rapistrum rugosum*	0	53	13	13.37	0.001
Solanum esuriale	60	100	100	13.85	<0.001
**Sonchus oleraceus*	13	60	0	16.12	<0.001
More frequent in 2004 only					
Dactyloctenium radulans	13	7	60	12.95	0.002
**Physalis lanceifolia*	0	0	53	19.46	<0.001
Portulaca oleracea	80	60	100	7.50	0.024
General increase over time					
Cyperus bifax	40	60	87	7.00	0.03
Einadia polygonoides	0	13	40	8.51	0.014
Eragrostis parviflora	13	60	73	11.92	0.003
Eremophila debilis	7	40	60	9.50	0.009
Goodenia fascicularis	20	80	93	19.98	<0.001
Hibiscus trionum	7	13	47	7.97	0.019
Panicum buncei	47	73	93	8.00	0.018
Paspalidium globoideum	7	33	67	11.83	0.003
Pimelea microcephala	0	13	33	6.43	0.04
Cullen tenax	13	60	67	10.18	0.006
Trianthema triquetra	0	7	33	8.08	0.018
Vittadinia cuneata	53	87	100	10.83	0.004

Table 4.6. Forward selection results, showing conditional effects and marginal effects for significant ($P < 0.05$) variable categories in RDA for species cover scores. ME, marginal effects; CE, conditional effects; λ, eigenvalues (refer to Section 3.2.6). Note: a significance level estimate of $P = 0.002$ is the lowest achievable given the number of permutations

Variable category	ME λ	CE λ	*F*	*P*
Mar 1998	0.06	0.06	5.05	0.002
Cultivated	0.06	0.06	4.97	0.002
Nov 1999	0.06	0.06	4.82	0.002
Open	0.05	0.05	4.64	0.002
Mar 2004	0.04	0.03	3.36	0.002
Dam <100 m	0.03	0.03	3.02	0.002
Reserve	0.05	0.03	2.84	0.002
Acafar	0.03	0.03	2.91	0.002
Dec 2001	0.03	0.02	2.26	0.002

The influence of cultivation on composition was obvious, with previously cultivated sites positioned mostly in the upper half of the ordination (Fig. 4.4). The influence of sampling time was also clear, with all sampling times spatially separated (Fig. 4.4). The eight sites sampled in November 1999 had different composition to the majority of sites (Fig. 4.4), but half of the sites sampled in November 1999 had been previously cultivated.

There was some correlation between sites that had not been cultivated and higher abundance of *Acacia farnesiana* (Fig. 4.5). Both these variable categories were associated with increasing native and perennial species richness. There were also similarities in composition for cultivated sites and those close to dams. Such sites were associated with increasing introduced and annual species richness (Fig. 4.5). Forb and graminoid richness were correlated and orthogonal to the perennial/native-annual/introduced axis. Although not strongly associated with any of the measured environmental variables, they were weakly associated with Reserve sites in the absence of tree canopy (Fig. 4.5).

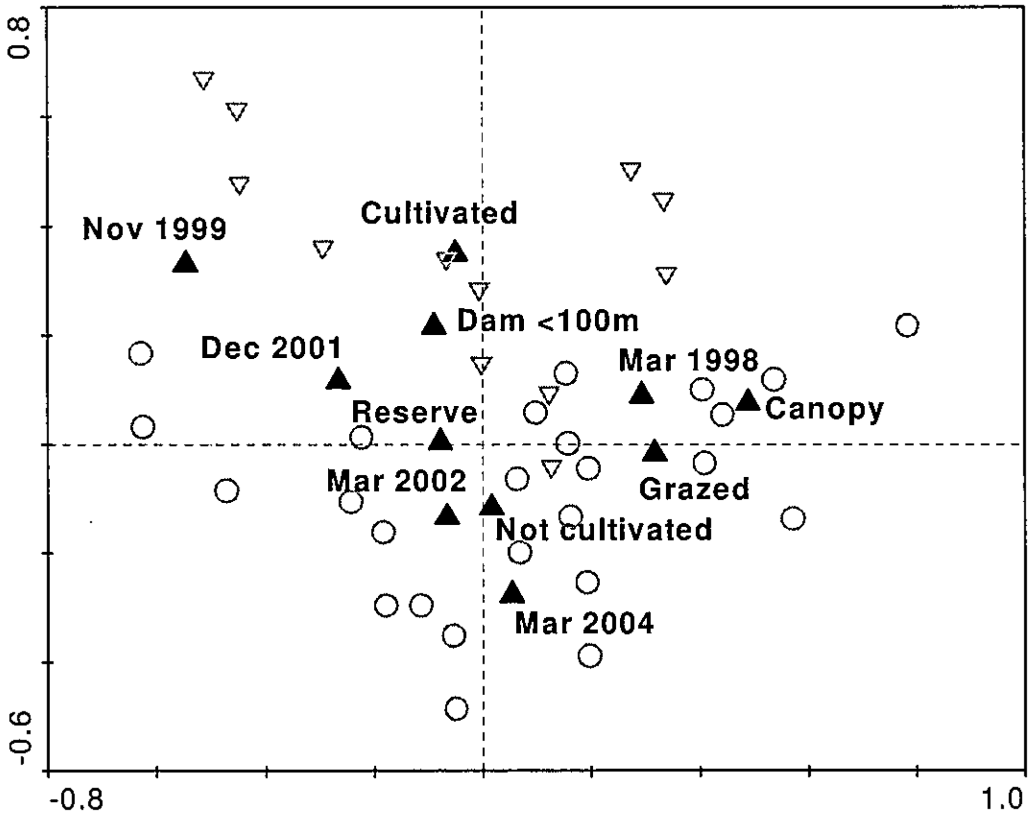


Fig. 4.4. RDA ordination biplot with all variable categories and sites for all sampling times. Filled triangles, nominal environmental variable categories (Table 4.2); circles, uncultivated sites; down-triangles, cultivated sites.

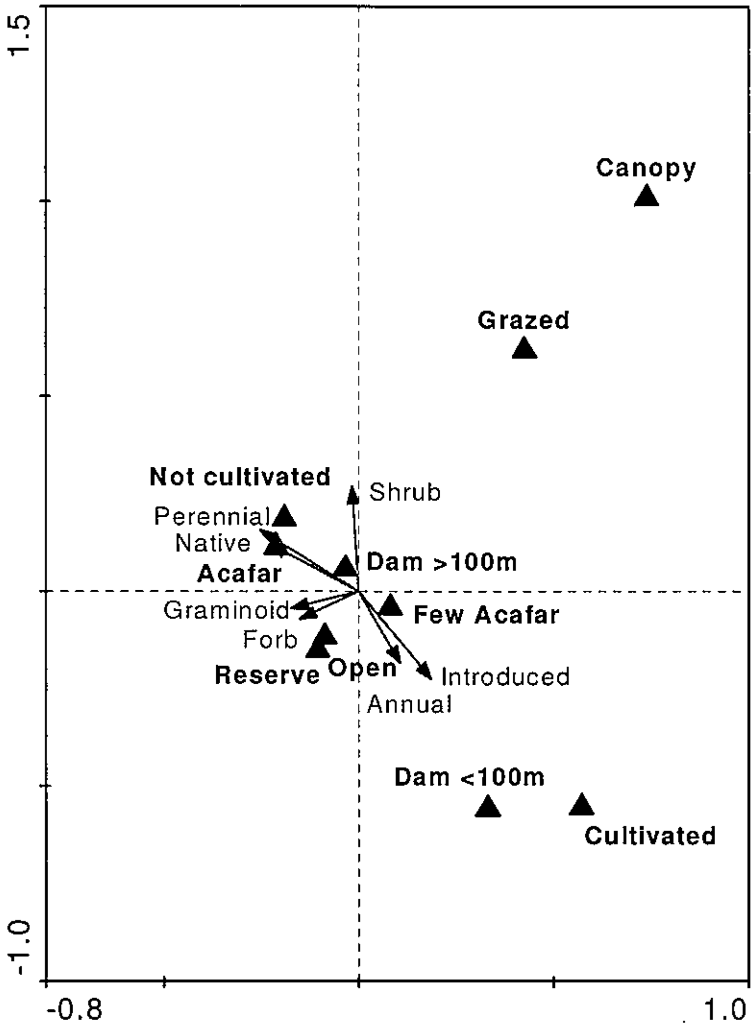


Fig. 4.5. Ordination (partial RDA), with time as a covariable. Filled triangles, nominal environmental variable categories; vectors for species richness, supplementary variables (grey font).

4.3.3 The effects of livestock grazing exclusion

There were no significant interactions between grazing and sampling time (Table 4.7). The occurrence of livestock grazing had no significant impact on total, native, introduced, forb, grass or shrub species richness (Table 4.7, Fig. 4.6a,b,c,d,e,g). Graminoid richness also did not vary significantly between grazed and ungrazed sites ($H = 0.629$, d.f. = 1, $P = 0.380$) in 2002. This variable could not be analysed at other sample times because Kruskal-Wallis ANOVA requires sample sizes of ≥5. Forb richness was higher in ungrazed sites in 2002 and 2003 (Fig. 4.6d; $F_{1,14} = 5.87$, $P = 0.030$ and $F_{1,9} = 5.49$, $P = 0.044$, respectively), although the presence of tree canopy appeared to have a stronger influence on forb richness ($P = 0.058$) than the occurrence of grazing (Table 4.7). Sampling time had a significant influence on all species richness measures (Table 4.7, Fig. 4.6 a,b,c,d,e,f,g).

To determine whether richness was responding to grazing at smaller sampling scales, total species richness at the 2-m² scale (the first of 10 concentric quadrats) was compared. In 2002, species richness was significantly higher at ungrazed sites (15.3 ± 1.34) than at grazed sites (9.4 ± 1.66)($F_{1,14} = 6.29$, $P = 0.025$). A similar result was observed in 2004. Ungrazed sites had a mean richness of 18.0 ± 0.96 species, while grazed sites had a richness of 10.7 ± 0.67 species per 2-m² quadrat ($F_{1,9} = 23.94$, $P < 0.001$).

Sites 14 and 15 were positioned either side of the Reserve boundary (Fig. 4.7) and provided the best comparison between grazed and ungrazed sites. These sites were surveyed four times (1998, 2001, 2002 and 2004). Mean richness over time for the grazed site was 40.8 ± 5.44 species per quadrat and 44.3 ± 6.34 species per quadrat for the ungrazed site, with no significant difference in richness between sites. However, in 2002 the ungrazed site had 13 more species than the grazed site due to recent heavy livestock grazing. At this time, the grazed site had a higher percentage of bare ground (approximately 80%) compared to the ungrazed site (approximately 15%). Differences in percentage bare ground and perennial grass density were also observed at other sampling times (Fig. 4.7).

Percentage bare ground estimates were consistently higher at grazed sites than ungrazed sites in 2002 and 2004. Percentage bare ground ranged from approximately 30-80% for the five grazed sites in 2002 and 25-50% for the three grazed sites in 2004. Most ungrazed sites had between 10% and 20% bare ground during both surveys, with a maximum estimate of 25%.

Partial redundancy analysis of species cover data identified four species (*Themeda avenacea, Asperula cunninghamii, Desmodium campylocaulon* and *Leiocarpa websteri*) that were significantly correlated with ungrazed sites (Fig. 4.8). Nine species were negatively correlated with ungrazed sites. Of those associated with grazed sites, two were tree species (*Acacia pendula* and *Casuarina cristata*) and three were shrubs (*Rhagodia spinescens, *Lycium ferocissimum* and *Capparis lasiantha*).

Table 4.7. Repeated measures ANOVA table (three summer-autumn sampling periods) showing differences between cultivated and uncultivated sites, grazed and ungrazed sites and the influences of covariates (proximity to dams and tree canopy) for different species richness variables (Intro. = introduced species richness)

Variable		Total		Native		Intro.		Forb		Grass		Shrub	
	d.f.	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*	*F*	*P*
Between-subjects													
Cultivation	1	7.70	0.012	9.56	0.006	1.17	0.292	4.51	0.046	1.81	0.193	2.02	0.170
Grazing	1	0.08	0.786	0.00	0.977	0.60	0.446	1.19	0.289	0.89	0.358	0.03	0.862
Covariates	2	3.00	0.072	1.10	0.353	6.21	0.008	5.08	0.016	1.24	0.310	0.64	0.536
Dam	1	5.95	0.024	1.86	0.188	11.25	0.003	6.12	0.022	1.75	0.201	0.00	0.983
Tree canopy	1	0.06	0.811	0.33	0.570	1.18	0.290	4.05	0.058	0.74	0.400	1.29	0.270
Residual	20												
Within-subjects													
Time	2	64.32	<0.001	53.58	<0.001	33.02	<0.001	61.33	<0.001	18.02	<0.001	8.59	0.001
Time*Cultivation	2	3.05	0.063	0.92	0.408	8.68	0.004	0.59	0.506	7.06	0.004	1.03	0.369
Time*Grazing	2	0.44	0.648	0.16	0.853	1.16	0.317	0.08	0.864	0.87	0.421	0.60	0.554
Residual	30												

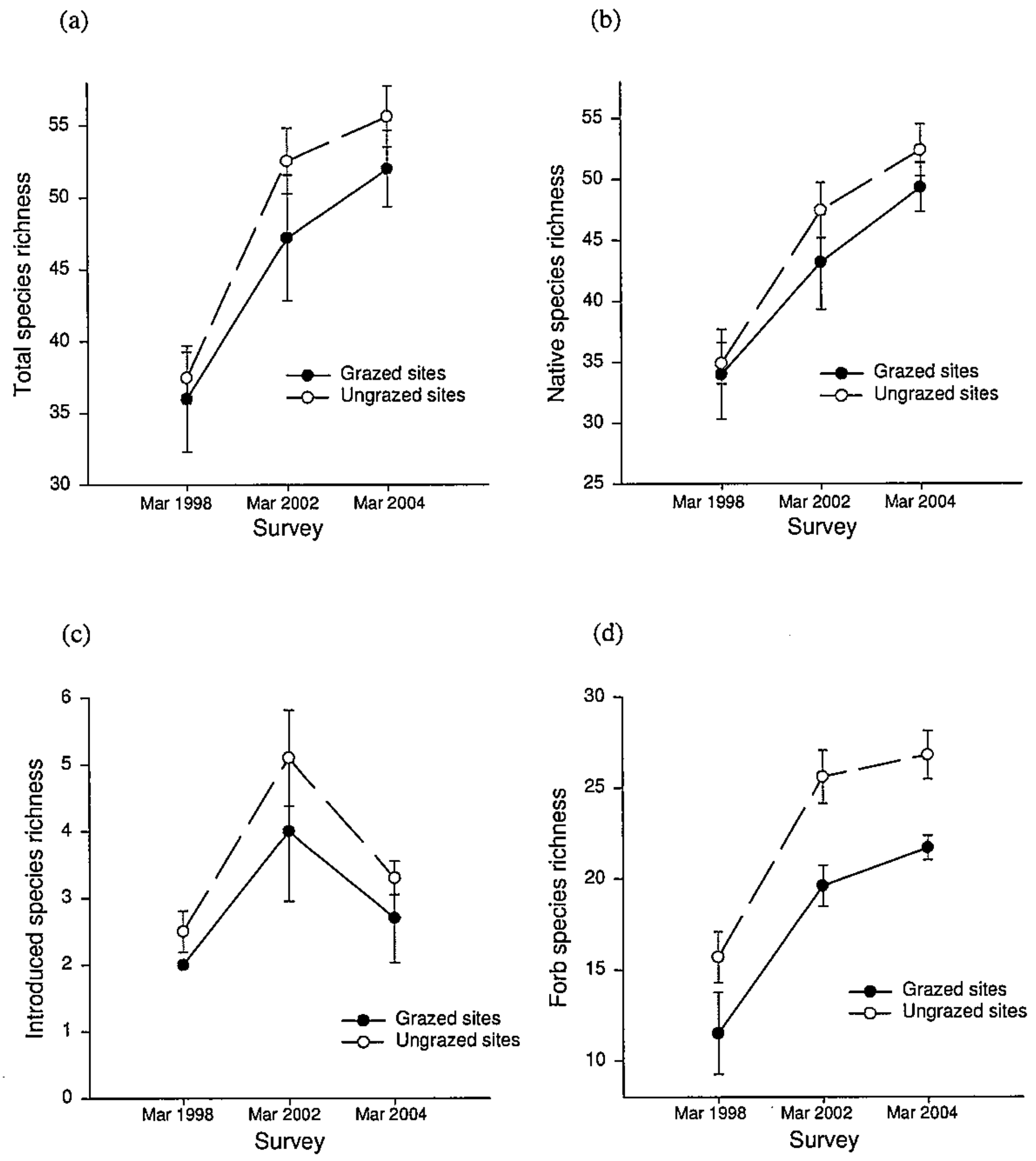
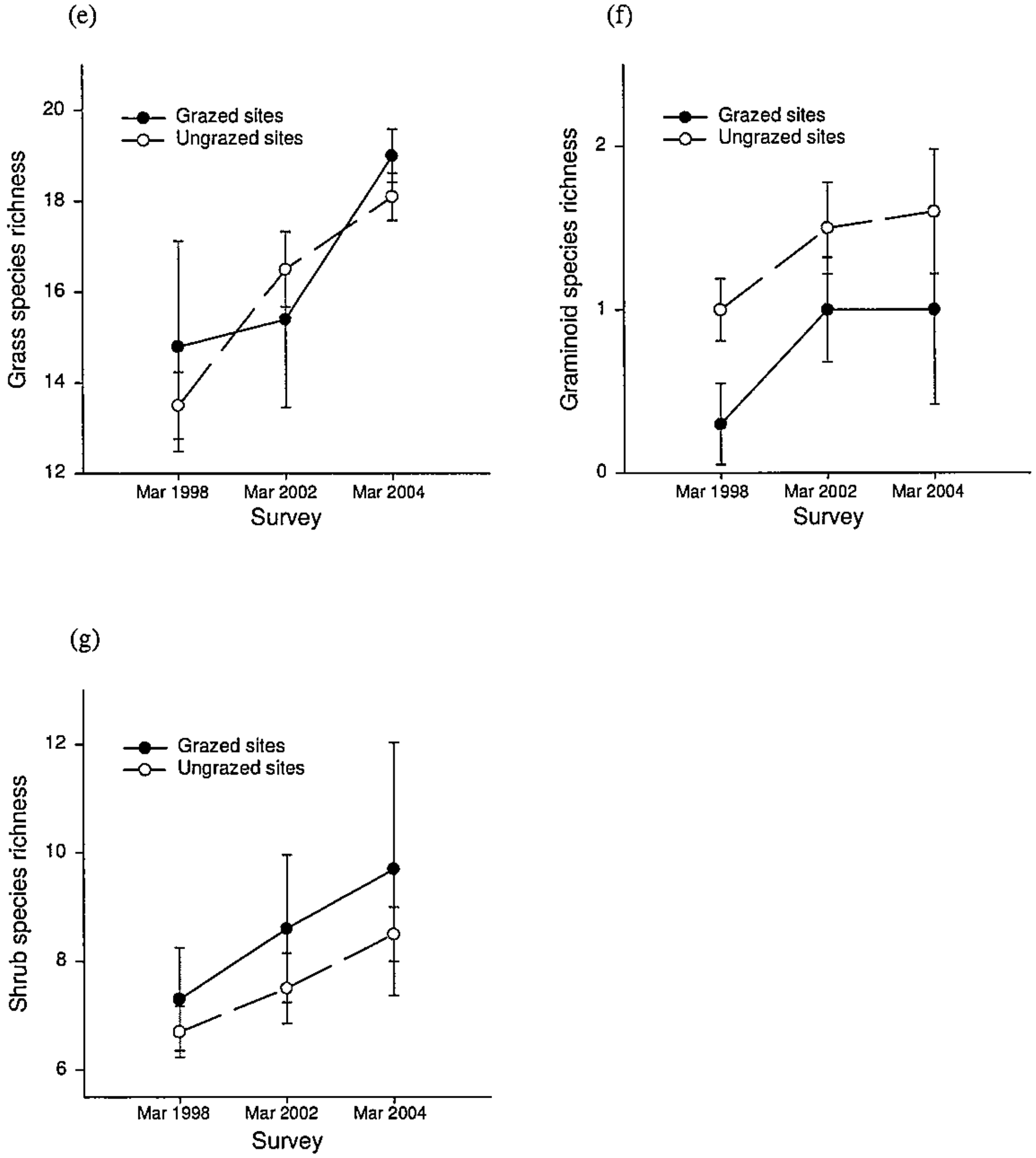


Fig. 4.6. Changes through time at ungrazed (Reserve) sites and grazed (TSR) sites in: (a) total; (b) native; (c) introduced; (d) forb; (e) grass; (f) graminoid and (g) shrub species richness. Refer to Table 4.3 for sample sizes at each time.

Fig. 4.6. (continued)



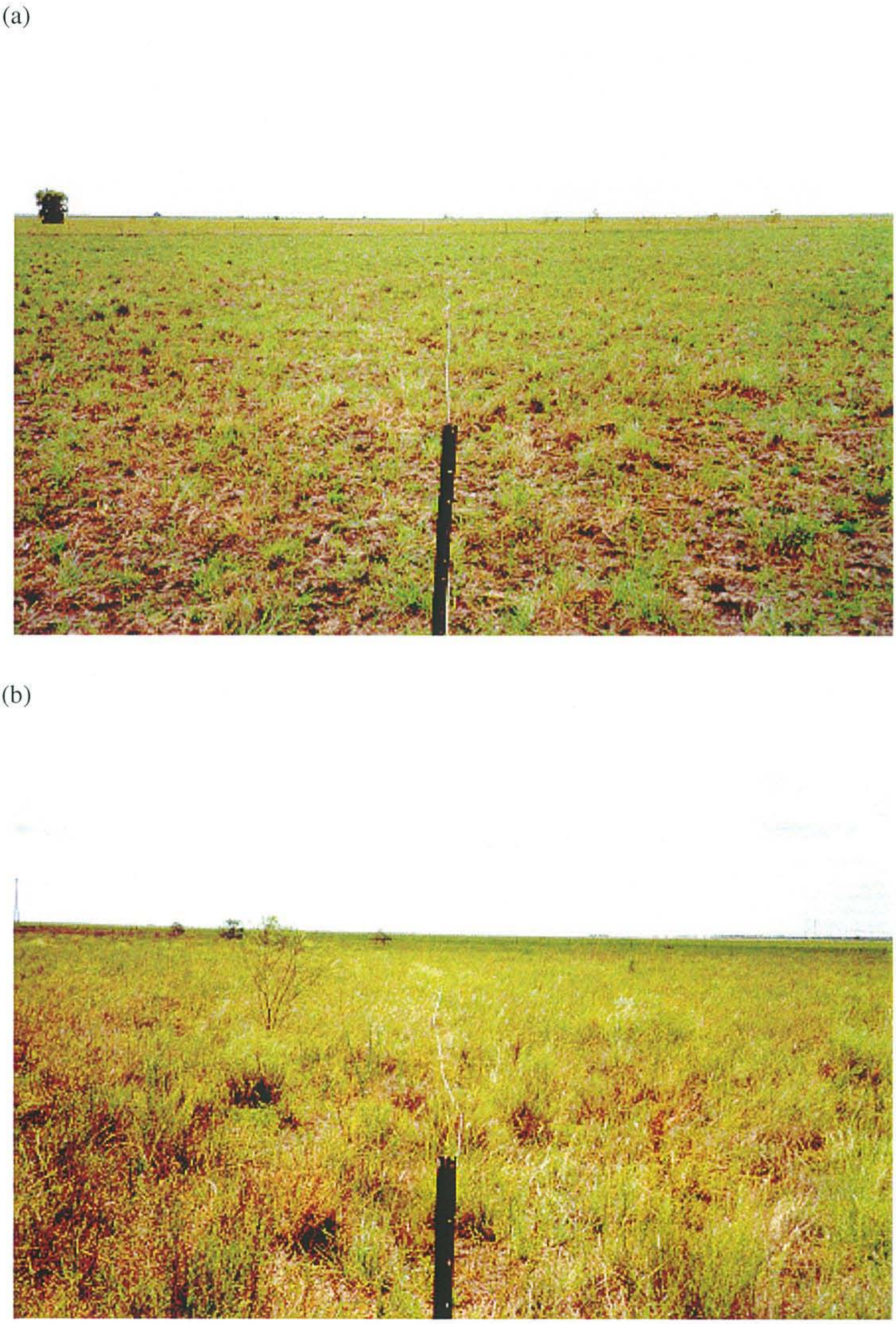


Fig. 4.7. Differences in bare ground and grass density between (a) site 14 (grazed TSR), and (b) site 15 (Kirramingly Nature Reserve) in December 2001. Total species richness at this time was 52 and 51 for sites 14 and 15, respectively.

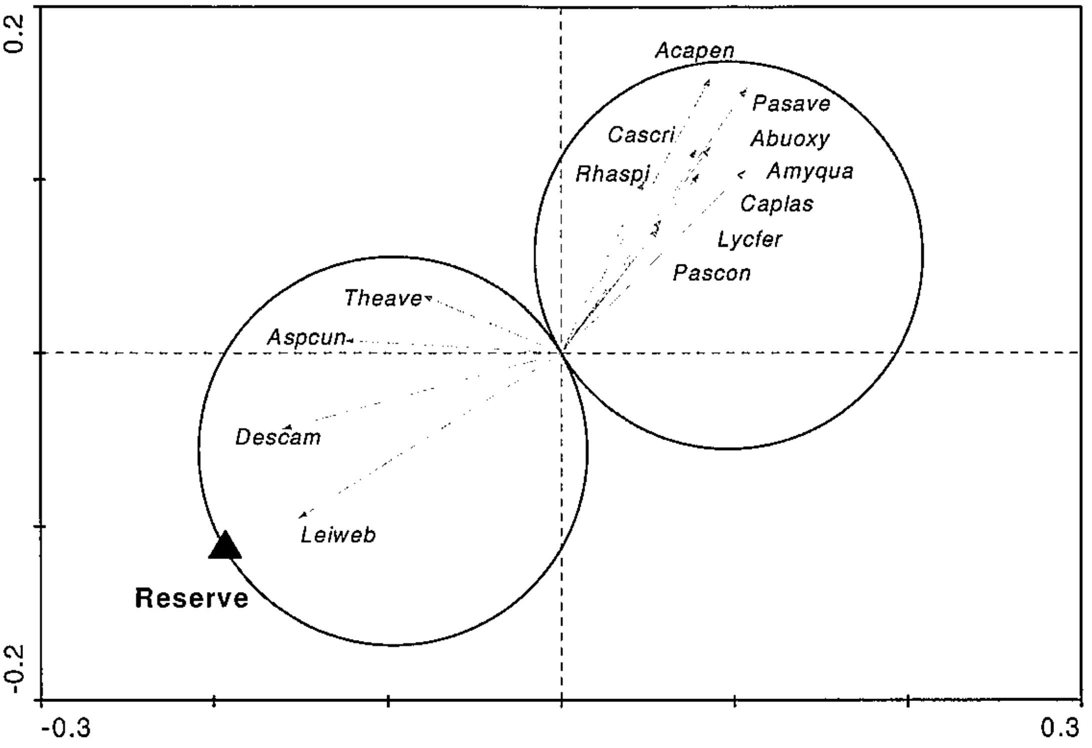


Fig. 4.8. A t-value biplot of the significant correlations (positive and negative) between species and their occurrence within Kirramingly Nature Reserve (ungrazed). Only significant relationships are shown (i.e. where species arrows end within the Van Dobben Circles they have a corresponding t-value larger than two). Refer to Appendix 4.1 for species labels.

Species varied in frequency between sites grazed by livestock and sites within the Reserve (Appendix 4.3). No species differed significantly in frequency of occurrence between grazed and ungrazed sites in 1998 or 2004. Ten species responded significantly to grazing in 2002 (Table 4.8): eight species were more common at ungrazed sites (Table 4.8), while two species (*Maireana decalvens* and *Portulaca oleracea*) were more common at grazed sites. The species more frequent at Reserve sites were mostly forbs (*Asperula cunninghamii, Desmodium campylocaulon, Phyllanthus virgatus, Ptilotus semilanatus, Wahlenbergia communis* and **Sonchus oleraceus*).

While no sites were established on the neighbouring grazed property for comparisons with the Reserve, field observations suggest that *Austrostipa aristiglumis* was common in the grazed paddock adjacent to the northern boundary of the Reserve. This species was rare within the Reserve.

Table 4.8. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (Fisher's exact test, d.f. = 1) to grazing at sites within and adjacent to Kirramingly Nature Reserve in 2002

	Grazed ($n = 5$)	Not grazed ($n = 11$)	P
More frequent at grazed sites			
Maireana decalvens	100	36	0.034
Portulaca oleracea	100	36	0.034
More frequent at ungrazed sites			
Asperula cunninghamii	20	82	0.036
Desmodium campylocaulon	20	91	0.013
Leiocarpa websteri	20	82	0.036
Panicum queenslandicum	20	82	0.036
Phyllanthus virgatus	20	91	0.013
Ptilotus semilanatus	20	82	0.036
**Sonchus oleraceus*	0	64	0.034
Wahlenbergia communis	20	82	0.036

4.3.4 The effects of past cultivation

There were significant interactions between cultivation and sampling time for introduced and grass species richness (Table 4.7). These interactions were due to a greater increase in introduced and grass species richness at cultivated sites relative to uncultivated sites in autumn 2002 (Fig. 4.9a,c,e). Introduced species richness was significantly higher at cultivated sites in 2002 (Fig. 4.9c; $F_{1,17} = 6.17$, $P = 0.024$). Despite this increase, total species richness was significantly higher at uncultivated sites over all three times (Table 4.7). Native and forb species richness was also significantly higher at uncultivated sites (Table 4.7, Fig. 4.9b,d). Grass species richness was higher at uncultivated sites in 1998 only (Fig. 4.9e; $F_{1,14} = 12.68$, $P = 0.003$). Proximity to dams (covariate) had a significant influence on total, introduced and forb species richness (Table 4.7; Section 4.3.5).

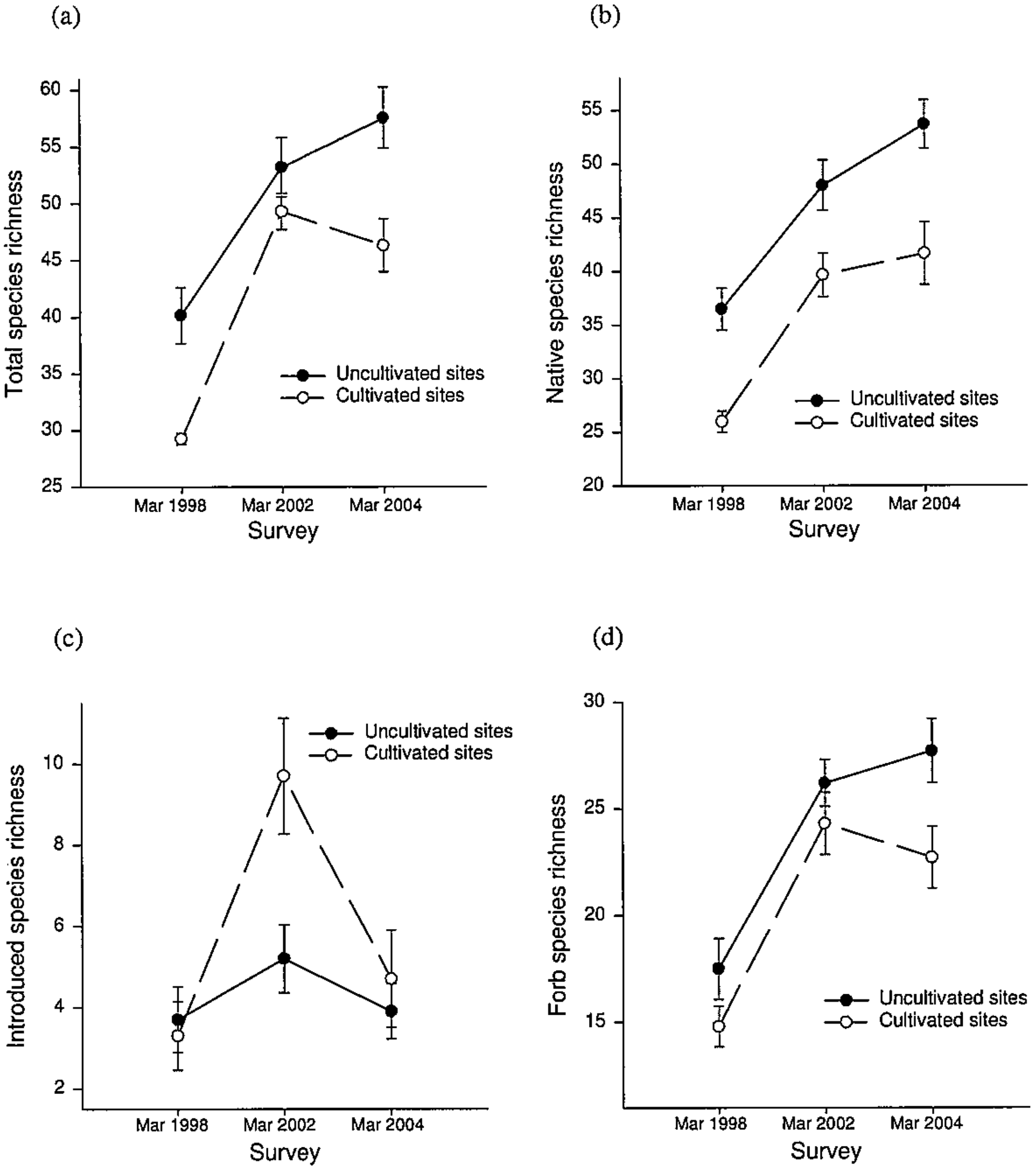
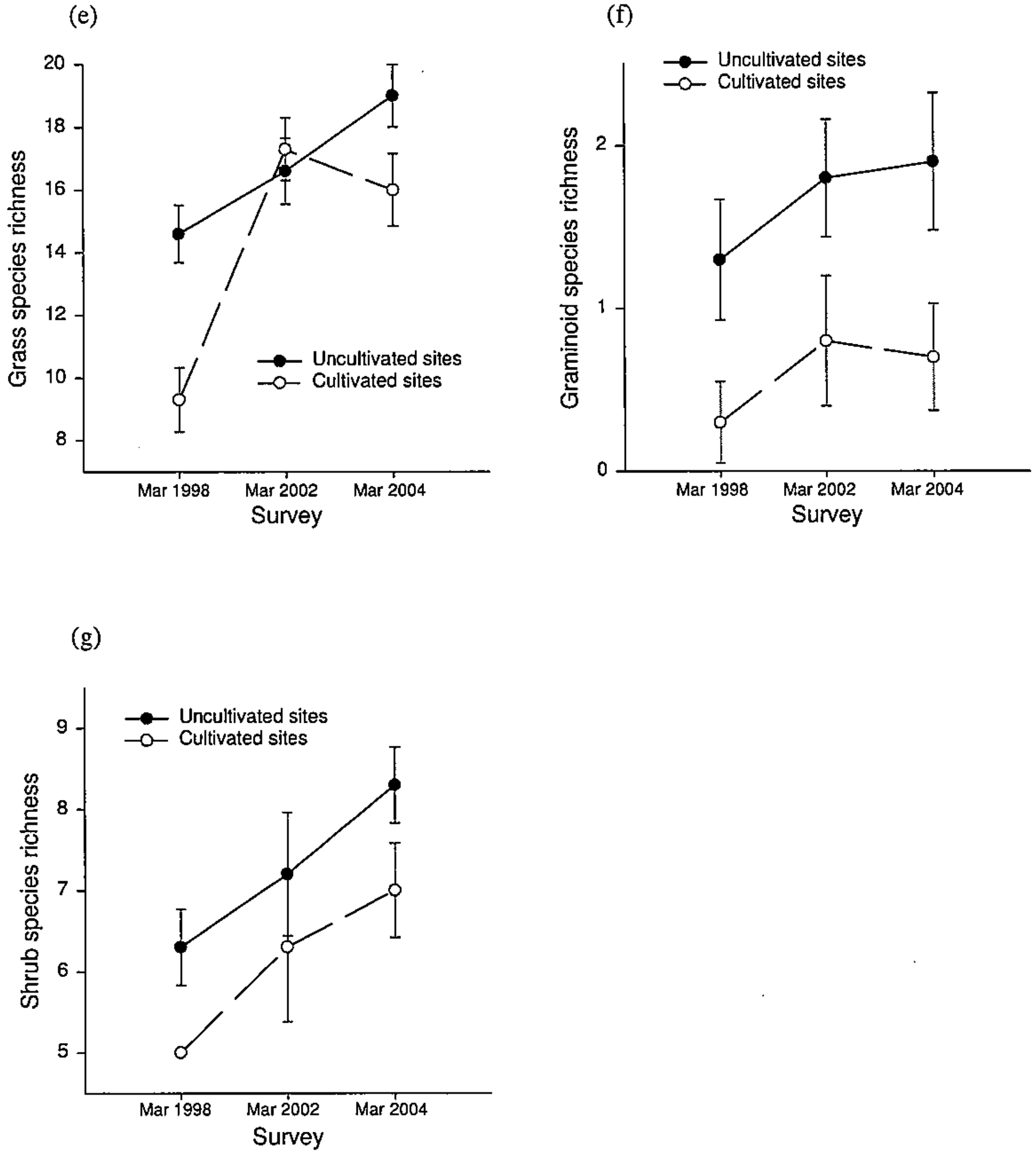


Fig. 4.9. Changes through time at uncultivated sites and cultivated sites in: (a) total; (b) native; (c) introduced; (d) forb; (e) grass; (f) graminoid and (g) shrub species richness. Refer to Table 4.4 for sample sizes at each time.

Fig. 4.9. (continued)



Previously cultivated sites were usually dominated by the grasses *Enteropogon acicularis, Dichanthium sericeum, Chloris truncata, Aristida leptopoda* and *Panicum decompositum* and the sub-shrub, *Sclerolaena muricata* var. *muricata* (Appendix 4.3). In November and December there were higher occurrences of annual grasses (e.g. **Avena fatua* and **Phalaris paradoxa*) at these sites.

Despite reductions in species richness, the abundance of 17 species was significantly correlated with cultivated sites (Fig. 4.10). Four of these species were introduced forbs (**Xanthium spinosum, *Lactuca serriola, *Cirsium vulgare* and **Rapistrum rugosum*) and four were introduced grasses (**Bromus cartharticus, *Phalaris paradoxa, *Setaria incrassata* and **Avena fatua*). Twenty-one species were negatively correlated with cultivated sites (Fig. 4.10). All of these were native, and were mainly forbs (11 species), grasses (six species) and sub-shrubs (three species). Both Mitchell grasses (*Astrebla elymoides* and *A. lappacea*) and the threatened (*TSC Act*) *Desmodium campylocaulon* were more abundant at uncultivated sites in the Reserve. However, *Astrebla elymoides* and *D. campylocaulon* did increase in frequency per quadrat in cultivated sites through time (Appendix 4.3).

In 1998 only one species (*Themeda avenacea*) was significantly more frequent at uncultivated sites than cultivated sites. This species was absent from cultivated sites and was significantly more common at uncultivated sites at each sampling time (Table 4.9). In 2002, nine taxa were more frequent at uncultivated sites, while three taxa (**Rapistrum rugosum, Leptochloa divaricatissima* and *Sclerolaena muricata* var. *muricata*) were more frequent at cultivated sites (Table 4.9). In 2004, six species were more frequent at uncultivated sites, while one species (*Cucumis melo*) was more common at cultivated sites. Two species (*Themeda avenacea* and *Asperula cunninghamii*) were consistently less frequent at cultivated sites in 2002 and 2004. For all three sampling times, the taxa more frequent at uncultivated sites were mostly forbs (six species) and grasses (five species).

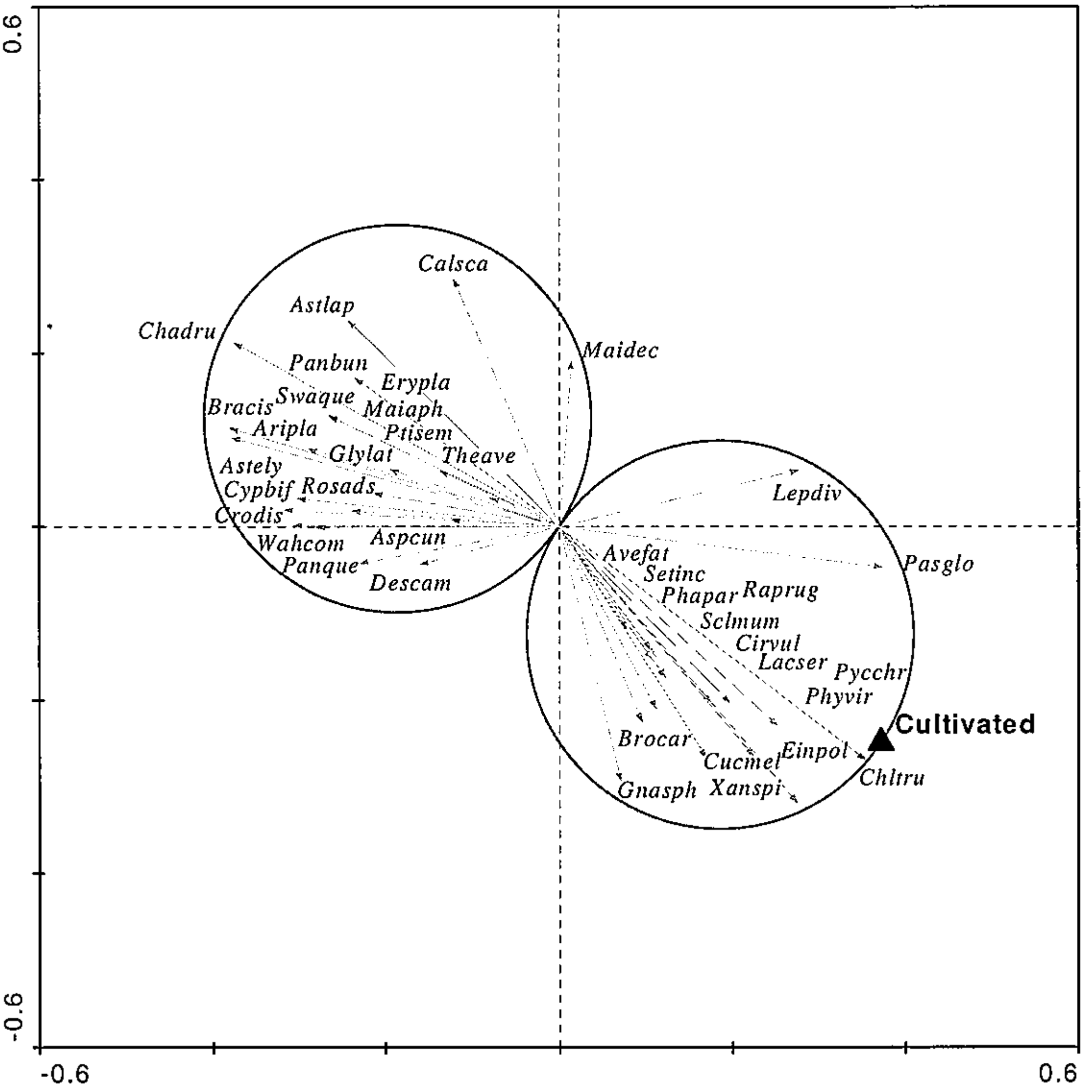


Fig. 4.10. A t-value biplot of the significant correlations (positive and negative) between species and occurrence of cultivation within Kirramingly Nature Reserve. Only significant relationships are shown (i.e. where species arrows end within the Van Dobben Circles they have a corresponding t-value larger than two). Refer to Appendix 4.1 for species labels.

Table 4.9. Frequency of occurrence (%) for individual species responding significantly ($P < 0.05$) (Fisher's exact test, d.f. = 1) to cultivation at sites within and adjacent to Kirramingly Nature Reserve

	Cultivated	Not cultivated	P
More frequent at cultivated sites			
2002 (n = 6 and 13)			
Leptochloa divaricatissima	83	23	0.041
*_Rapistrum rugosum_	100	38	0.018
Sclerolaena muricata var. _muricata_	100	38	0.018
2004 (n = 3 and 9)			
Cucumis melo	100	22	0.045
More frequent at uncultivated sites			
1998 (n = 4 and 12)			
Themeda avenacea	0	92	0.003
2002 (n = 6 and 13)			
Asperula cunninghamii	0	85	0.001
Calotis scabiosifolia	0	54	0.044
Chamaesyce drummondii	17	85	0.010
Eryngium plantagineum	0	54	0.044
Haloragis aspera	0	54	0.044
Maireana aphylla	33	85	0.046
Panicum buncei	33	85	0.046
Ptilotus semilanatus	17	77	0.041
Themeda avenacea	0	92	<0.001
2004 (n = 3 and 9)			
Asperula cunninghamii	0	78	0.045
Astrebla lappacea	33	100	0.045
Austrodanthonia bipartita	0	78	0.045
Leiocarpa websteri	33	100	0.045
Panicum queenslandicum	33	100	0.045
Themeda avenacea	0	100	0.005

4.3.5 The influence of tree canopy and proximity to dams on composition and richness

The three sites close to dams had higher total species richness, due to higher richness of introduced forbs. In 2002, when all three dam sites were sampled, total species richness was 57.3 ± 5.24 for sites near dams and 49.9 ± 2.08 (n = 17) for other Reserve sites. In the same year introduced species richness was 12.3 ± 0.88 (*n* = 3) for sites near dams and 5.5 ± 0.68 (*n* = 17) for other Reserve sites, and forb richness was 28.3 ± 0.88 (*n* = 3) for sites near dams and 24.2 ± 1.16 (*n* = 17) for other Reserve sites. Tree canopy had a marginal influence on forb species richness (Table 4.7), with lower richness at sites with canopy. For example, in 2002, forb richness was 18.3 ± 1.89 (*n* = 4) at sites with canopy and 25.2 ± 1.18 (*n* = 15) at uncultivated sites without canopy.

Two of the three sites <100 m from dams, had been previously cultivated. Hence it is not surprising these two variable categories were grouped closely in Figure 4.5, and had species in common (e.g. *Euchiton sphaericus*, **Bromus cartharticus*, **Xanthium spinosum*, *Cucumis melo*, **Phalaris paradoxa*, **Cirsium vulgare* and **Rapistrum rugosum*). Due to the confounding effect between these two variables it is difficult to say which species were responding to cultivation and which were influenced by proximity to dams. However, the one site near a dam that had not been previously cultivated had consistently higher species richness (63-72 species per quadrat compared to 30-59 species per quadrat for the cultivated sites) and differed in species composition to the other sites <100 m from dams (e.g. had higher abundance of *Astrebla* spp.).

There was also some correlation between the occurrence of tree canopy and grazed sites (Fig. 4.5), because two of the four TSR sites had tree canopy. Certain species were influenced by the occurrence of canopy (e.g. the mistletoe, *Amyema quandang*) and others may have benefited from grazing, but unfortunately these two groups of species cannot be easily differentiated. Some of the species aligned with these vectors are shown in Figure 4.8, (i.e. species negatively correlated with the Reserve) and include *Paspalidium constrictum*, **Lycium ferocissimum*, *Capparis lasiantha*, *Amyema quandang*, *Abutilon oxycarpum*, *Paspalidium aversum*, *Acacia pendula*, *Casuarina cristata* and *Rhagodia spinescens*. It is likely that most of these species were influenced by tree canopy because all except *Rhagodia spinescens* were absent from the grazed sites without trees. In fact, two of the species (*Acacia pendula* and *Casuarina cristata*) were responsible for providing the canopy on the neighbouring stock route. Sites with tree canopy had higher levels of bare ground, and appeared to have a lower density of native perennial grasses (e.g. *Astrebla* spp.) (Fig. 4.11).

Fig. 4.11. TSR site with tree canopy (site 5) in December 2001, illustrating bare ground and low perennial grass density beneath trees. This site had a total of 67 species in the 0.1024-ha quadrat on this occasion.

4.3.6 Abundance of Acacia farnesiana *and its influence on composition*

Abundance of *Acacia farnesiana* may have some influence on species composition (Table 4.6). However, sites with higher abundance of this species were generally uncultivated, suggesting cultivation influenced its abundance. *Acacia farnesiana* was most abundant in the uncultivated, north-west management unit of the Reserve (Fig. 4.1). It was also abundant on the TSR adjacent to this part of the Reserve, and on the neighbouring property. *Acacia farnesiana* had no negative impact on native species richness (Fig. 4.5). Species commonly associated with sites with abundant *A. farnesiana* included those negatively correlated with cultivation (Fig. 4.10), for example, *Ptilotus semilanatus, Maireana aphylla*, *Themeda avenacea, Aristida platychaeta* and *Glycine latifolia*.

Acacia farnesiana was ranked sixteenth in terms of mean frequency score per quadrat, averaged over all times and sites at the Reserve. It did not significantly increase or decrease in frequency at the Reserve or on the surrounding TSR between 1998 and 2004

(Table 4.10). Frequency score per quadrat varied from 4.8 ± 0.88 in 1998 to 4.4 ± 0.97 in 2002, for uncultivated sites at the Reserve. Frequency scores varied from 5.3 ± 2.60 in 1998 to 4.0 ± 0.58 in 2004, for cultivated sites in the Reserve, and from 4.3 ± 1.33 in 1998 to 5.7 ± 1.45 in 2002, for uncultivated, TSR sites. *Acacia farnesiana* has successfully established in areas that were cultivated but is not as abundant there as in some parts of the Reserve. For example, the four sites in the north-west management unit of the Reserve had frequencies of 7-10 occurrences per quadrat in 2002, while the six cultivated sites in the north-east of the Reserve had frequencies of 3-6 in the same year.

Table 4.10. Repeated measures ANOVA for the effect of land-use history (uncultivated, reserve; cultivated, reserve; and uncultivated, TSR) on *Acacia farnesiana* frequency score per quadrat over time (three summer-autumn sampling periods)

Variable	d.f.	*F*	*P*
Between-subjects			
Land-use history	2	0.09	0.916
Residual	12		
Within-subjects			
Time	2	0.01	0.942
Time*Land-use history	4	0.50	0.649
Residual	24		

4.4 Discussion

4.4.1 Seasonal variations in composition and richness

Changes in species richness over time were obvious at Kirramingly Nature Reserve. There were temporal changes in species richness at both seasonal and annual scales. Total species richness increased significantly between 1998 and 2002 ($P < 0.001$), but not between 2002 and 2004 ($P = 0.113$). However, native richness increased in each successive summer-autumn sampling period. Similar increases in richness were observed for both sites within the Reserve and the neighbouring stock route (Table 4.9). This suggests that the increases in richness were due to seasonal variations. Rainfall for Moree in the 4 months prior to sampling was 257 mm in 1998, 196 mm in 2002 and 300 mm in 2004 (Commonwealth Bureau of Meteorology 2004). The increase in native species richness between 2002 and 2004 may have been due to increased rainfall in the months prior to sampling. However, recent rainfall does not explain the differences between 1998 and 2002. Unfortunately there is no recent rainfall data for Kirramingly Nature Reserve, but given the variability in rainfall due to storms, local rainfall may have been better correlated with the changes in richness.

It is also possible that differences in richness were the result of different individuals carrying out the surveys. Two of the five surveys were carried out by individuals other than myself. However, richness was similar between the survey carried out in November 1999 (by contract for NPWS) and the survey I carried out in December 2001. Given that the methods used for all surveys were the same, and that the amount of time spent sampling 0.1024-ha quadrats was similar (P. Clarke, pers. comm., 2001), I think it is unlikely that surveyor bias was responsible for the differences in richness between 1998 and other sampling times. The significant increase in native species richness between 2002 and 2004 at sites sampled by myself highlights the temporal variability in richness.

Dichanthium sericeum increased significantly in frequency per quadrat between 1998 and 2002 ($P = 0.014$). This supports the findings in Chapter 3 (Section 3.4.2) that this species fluctuates in abundance from year to year. Similar fluctuations have been reported elsewhere (e.g. Williams & Roe 1975; Groves & Williams 1981). Campbell (1989) reported that *D. sericeum* responds well to spring and early summer rains. However, early summer rain at Moree did not appear to be responsible for the differences in frequencies between 1998 and 2002 (158 mm in November and December 2001, compared to 185 mm in November and December 1997). Perhaps other climatic variables (e.g. temperature) in addition to rainfall are important in determining year-to-year abundances of *D. sericeum* (e.g. Risser 1985; Fensham *et al.* 2000).

Annual and introduced species richness were lower during late summer-autumn, than in November 1999 and December 2001 (Fig. 4.3). This is not surprising given the seasonal changes in composition in these grasslands (Section 3.3.2). Even in December 2001, there were several winter-active species still common in the Reserve (e.g. introduced species like **Phalaris paradoxa* and **Hedypnois rhagadioloides*). Higher introduced species richness in February-March 2002 was mostly due to the early germination of certain winter-growing species (e.g. **Sonchus oleraceus* and **Rapistrum rugosum*). These times of transition between seasons may be the best time to sample to provide the highest estimate of total species richness at a site. However, once-off sampling at any time of the year is likely to miss some species.

4.4.2 Grazing and canopy cover

Removal of livestock grazing from Kirramingly Nature Reserve has not resulted in a reduction in species richness (Table 4.7; Fig. 4.6). Forb richness was not higher at grazed sites compared to Reserve sites (Fig. 4.6d). This suggests that exclusion of livestock grazing for approximately 8 years has not inhibited recruitment of forb species through competition with perennial grasses. Similar findings are reported for the regional vegetation survey (Section 3.3.4; Table 3.20) and by Williams and Mackey (1983) who reported that excluding sheep from *Astrebla* grassland in south-western Queensland for 29 years produced slightly more diverse grassland than under station management, with no obvious build up of litter. However, this contradicts many other Australian studies (Stuwe & Parsons 1977; Gibson & Kirkpatrick 1989; Lunt 1991; Trémont 1994; Trémont & McIntyre 1994; Morgan & Lunt 1999). The differences between studies are probably related to different climates (semi-arid *vs* temperate), competitiveness of the dominant grasses (Olff & Ritchie 1998) and differences in the amount of bare ground.

Intermediate grazing intensities did not enhance forb richness in temperate grassy woodlands on the New England Tablelands (Clarke 2003). Habitat heterogeneity at the 0.1 ha sampling scale was proposed as the reason for the lack of disturbance-related response in richness (Clarke 2003). Sampling scale had an important effect on the influence of livestock grazing in the current study. At the 2-m^2 sampling scale, grazed sites had significantly lower species richness than ungrazed sites ($P = 0.025$), whilst at the 0.1024-ha quadrat size there was no significant reduction in richness due to grazing ($P = 0.257$). However, this is opposite to what we would expect from the hypothesis that some livestock grazing is necessary to enhance richness. These findings highlight the differing effects of herbivores on plant diversity in different environments (Milchunas *et al.* 1988). Perhaps drought and other climatic influences, combined with kangaroo grazing, are providing a natural form of disturbance in these grasslands, preventing grass densities from becoming too high to limit growth of inter-tussock species.

The difference in richness at the 2-m^2 sampling scale supports the idea that sampling at small scales may be more likely to detect changes in species richness than at larger scales (e.g. total quadrat sizes sampled in the current research) (Stohlgren *et al.* 1999). Greater sensitivity in small quadrats is not surprising because changes in species frequency are more likely to equate to changes in richness in a small quadrat, where less common species are less likely to be recorded. However, very small scales might show differences in richness that are meaningless on a larger scale (i.e. local colonisation from a species pool at a larger spatial scale). For example, a seasonal increase in abundance of a certain species that already exists in a paddock may result in an increase in richness in small quadrats (i.e. by one species) but not an increase in richness at the paddock scale. Further, a decline in plant richness may reflect no loss of species from the community (or paddock, or quadrat), depending on the occurrence of species in the soil seed bank (Keith 1996).

The fact that grassland richness at Kirramingly is relatively insensitive to grazing at the 0.1024-ha scale suggests that these grasslands are similar to those in other parts of the world with long histories of grazing (Milchunas & Laurenroth 1993; Milchunas *et al.* 1998; Grace 1999). Many native species on the Moree Plains appear to be resilient to livestock grazing, perhaps because of their abilities to be reduced to rootstock during unfavourable conditions (e.g. during drought). Olff and Ritchie (1998) suggested that dry environments on fertile soils often support competitively dominant plant species that are able to tolerate grazing. In such areas, exclusion of herbivores may have a small positive effect on richness due to the persistence of a few plants that are intolerant of grazing (Olff & Ritchie 1998). While we have no records of the pre-grazing species richness, native species richness at Kirramingly Nature Reserve was considered high by Clarke *et al.* (1998) in comparison to other NSW grasslands, and has increased further since 1998. The evidence of past heavy livestock grazing at Kirramingly (P. Wall, pers. comm., 2002) suggests that these grasslands are able to recover well in terms of native species richness after such a disturbance and that some grazing-sensitive species are able to persist in grazed areas. However, it is possible that certain grazing-sensitive species (e.g. native peas, such as *Swainsona* spp.) have already been eliminated from the plant assemblage, given that Australia has a history of relatively light grazing compared to some other parts of the world (Landsberg *et al.* 1999).

Many plant species on the Moree Plains have traits that allow persistence of high species richness in undisturbed (i.e. ungrazed) grasslands. Such traits include: (1) The ability to reproduce vegetatively, by transferring stored resources from an established parent plant to offspring (Grime 1979; Belsky 1992). Many species on the Moree Plains have the ability to reproduce vegetatively (e.g. *Astrebla* spp., *Aristida leptopoda* and many other perennial grasses). (2) Climbing, twining or spreading habits that allow plants to either reach light by climbing on taller species, or find spaces in the canopy by spreading over large areas (e.g. *Desmodium campylocaulon, Neptunia gracilis, Rhynchosia minima, Convolvulus erubescens, Boerhavia dominii* and *Einadia nutans*). (3) Having large or long-lived seeds with dormancy cues (e.g. *Cullen tenax, Acacia spp., Goodenia fascicularis* and *Solanum esuriale*). Large seeds hold more resources and may give seedlings a better chance of emerging above the grass canopy. Long-lived seeds that are able to lie dormant until conditions are favourable for growth (e.g. after fire) would allow plant species to establish at times when competition is less intense (e.g. Gill 1981). (4) The ability of species to grow opportunistically or seasonally (e.g. *Wahlenbergia communis, Goodenia fascicularis, Phyllanthus virgatus, Sida trichopoda* and many other forb and grass species). Plants that are able to germinate and grow rapidly during times when conditions are favourable (i.e. after rainfall) are likely to suffer less intense competition. Hence, growth of species at different times of the year allows a greater number of species to coexist in the same area, as they do not compete directly for resources (Fowler 1986; Begon *et al.* 1996; Bullock 1996; Tokeshi 1999).

The cover of tall perennial grass species may increase following protection from grazing (e.g. Stuwe 1986; McDougall 1989; Noy-Meir *et al.* 1989; Belsky 1992). It will be interesting to see whether, over time, livestock grazing exclusion at Kirramingly Nature Reserve encourages tall species, such as *Themeda avenacea*, or other dominant grasses, like *Aristida leptopoda*, that might exclude smaller interstitial species. Frequency score per quadrat increased for *A. leptopoda* between 1998 and 2002 ($P = 0.002$). The frequency and abundance of dominant grasses, like *A. leptopoda*, and species richness, should be monitored in the future so that adaptive management can be implemented, if necessary, to prevent competitive exclusion and, hence, a reduction in species richness. This may be particularly important in the south-west management unit of the Reserve (Fig. 4.1) where grass density and litter accumulation appears higher than in other parts of the Reserve (although surveys to date have not recognised lower species richness at the two sites in this part of the Reserve).

It is not surprising that few species reacted to grazing (Table 4.8) given that native species richness did not vary significantly between Reserve sites and grazed sites (Table 4.7; Fig. 4.6b). Only in 2002 were seven native species (*Asperula cunninghamii, Desmodium campylocaulon, Leiocarpa websteri, Panicum queenslandicum, Phyllanthus virgatus, Ptilotus semilanatus* and *Wahlenbergia communis*) more frequent at ungrazed sites than at grazed sites. The increase in frequency of some species in successive summer-autumn samples (1998, 2002 and 2004) suggests they may be responding positively to the removal of grazing from the Reserve (Table 4.8). Species that increased in frequency at each sampling time included *Cyperus bifax, Einadia polygonoides, Eragrostis parviflora, Eremophila debilis, Goodenia fascicularis, Hibiscus trionum, Panicum buncei, Paspalidium globoideum, Pimelea microcephala, Vittadinia cuneata, Trianthema triquetra* and *Cullen tenax* (Table 4.5). However, none of these species were significantly more frequent at ungrazed sites than grazed sites (Table 4.8). Certain species did, however, increase in frequency per quadrat at ungrazed sites only. For example, *Desmodium campylocaulon* increased significantly in frequency per quadrat between 1998 and 2002 ($P = 0.002$), but decreased in frequency at the four grazed sites between these sampling times. While this threatened species appears to be able to tolerate some grazing (Chapter 3), grazing intensity may be influencing its abundance (Section 3.4.6).

Of the seven native species that were more frequent at ungrazed sites in 2002 (Table 4.8), three were also identified in multivariate analysis as more abundant in ungrazed sites. These were *Asperula cunninghamii, Leiocarpa websteri* and *Desmodium campylocaulon.* Four species that were less frequent or abundant at grazed sites adjacent to Kirramingly Nature Reserve were also identified as grazing-sensitive in the regional vegetation survey (Chapter 3). These were *Panicum queenslandicum, Ptilotus semilanatus, *Sonchus oleraceus* and *Themeda avenacea.* The negative response of *Themeda avenacea* to grazing is consistent with the negative response to heavy grazing by its more commonly studied congener, *T. australis,* in Australian and African grasslands (Hodgkinson *et al.* 1989). The relatively small number of consistent grazing responses between the Kirramingly surveys and the regional survey is not surprising given high levels (41%) of inconsistency to grazing in the literature (Vesk & Westoby 2001) and the range of other variables that may influence species response in the regional survey.

Everist and Webb (1975) suggested that many *Astrebla* grasslands are disclimax communities, once maintained by Aboriginal firing and native macropod grazing, and now maintained by livestock grazing. However, there was no evidence of changes in botanical composition leading to the development of different plant communities at Kirramingly Nature Reserve. Quadrat frequency and cover abundance of *Astrebla* spp. has not decreased as a result of livestock grazing exclusion at the Reserve. Long-term grazing exclusion in south-western Queensland reported similar results, with irregular recruitment over time and an increase in basal area of *Astrebla* spp. in exclosures (Williams & Roe 1975). In western NSW, Cunningham and Milthorpe (1981) reported that *Astrebla pectinata* was common in a regeneration area excluded from grazing, but was absent from the adjacent grazed area. Grazing exclusion had only a small effect on plant biomass over a 6-year period in *Astrebla* grassland in the Northern Territory and seasonal effects (i.e. rainfall) were more obvious than the effects of livestock exclusion (Foran & Bastin 1984). In contrast to these findings, Winders (1936) and Partridge (1996) suggested that under-utilisation of *Astrebla* spp. pastures may be deleterious to the vegetation. Further monitoring at Kirramingly is required to determine whether *Astrebla* spp. are negatively affected after a longer period of livestock exclusion.

Confounding effects between tree canopy and grazing may have prevented recognition of some grazing effects. Despite marginally lower forb richness at sites with tree canopy (Table 4.7), total species richness was not significantly different at grazed and Reserve sites at the 0.1024-ha scale, perhaps because the reduction in forbs was balanced by an increase in other species. Trees create sub-habitats, which differ from the open grassland and exert different influences on herbaceous vegetation (Belsky *et al.* 1989; Scanlan & Burrows 1990; Abule *et al.* 2005). Certain species may have been encouraged by tree canopy or grazing, while other species may have been negatively influenced by these variables. Hence, while composition was different at grazed sites with tree canopy (Fig. 4.8), there was no net increase or decrease in total richness compared to the Reserve sites. Differences in composition, without changes in richness, due to grazing have been reported elsewhere (e.g. Bai *et al.* 2001). Unfortunately, there is a lack of grazed sites with similar composition to the Reserve to allow accurate interpretations on the influence of livestock grazing removal. Sites 14 and 15 are positioned either side of the Reserve boundary (Figs 4.1, 4.7) and provide the best comparison between grazed and ungrazed sites of similar composition. Species richness at this TSR site varied greatly over time, probably due to differences in intensity of livestock grazing. Hence once-off surveys to determine the influence of livestock grazing may be inaccurate, as was suggested by Everist and Webb (1975). The establishment of further monitoring points with similar composition to the Reserve on TSRs and private properties neighbouring the Reserve, is recommended for further comparisons over time.

Despite confounding effects between livestock grazing and the presence of tree canopy, it is likely that the two variables are linked ecologically. This is because areas with tree canopy offer shade to grazing animals and, hence, are likely to receive higher intensities of grazing (Clarke 2003). It is possible that the reduction in perennial grass cover and higher levels of bare ground are related to higher grazing intensities. Irrespective of the occurrence of tree canopy, grazed sites had greater amounts of bare ground than Reserve sites. The one site with tree canopy on the Reserve was similar in composition to the grazed tree canopy sites. Some of the similarities may be due to utilisation of these areas by kangaroos, but it is more likely that these areas with tree canopy are inherently different because of their position in the landscape. These open woodland areas are common along the Little Bumble Creek running adjacent to the Reserve.

4.4.3 Cultivation and proximity to dams

Redundancy analysis revealed that previous cultivation had a strong influence on species composition (Table 4.6, Fig. 4.4). After sampling time, cultivation was the most important variable in explaining species composition, accounting for 7.3% of the total variance. Clarke *et al.* (1998) reported that sites did not cluster into groups associated with cultivation, although some groups were apparent at finer levels of clustering (e.g. Burr-grasslands). It is interesting that the effects of past cultivation were not more obvious in 1998, 7-16 years after the last cultivation.

Areas that had been cultivated were associated with higher introduced and annual species richness (Fig. 4.5). Clarke *et al.* (1998) also reported that cultivation was associated with the presence of annuals (weeds and grasses), and less perennial grass dominance. Perennial grasses had established dominance at cultivated sites by 2002, but some different species were dominant compared to uncultivated areas. Previously cultivated sites were usually dominated by the grasses *Enteropogon acicularis*, *Dichanthium sericeum, Chloris truncata*, *Aristida leptopoda* and *Panicum decompositum.* All of these species, except

Dichanthium sericeum, have open panicles that are easily moved by wind. McIntyre *et al.* (1995) also reported that wind-dispersed species occurred in higher proportions at soil-disturbed sites. *Astrebla* spp. were not common at cultivated sites, suggesting it may take some time (i.e. >22 years) for these species to reach abundances as in uncultivated areas. This supports findings from the regional survey (Chapter 3). The seeds of *Astrebla* spp. would not be easily dispersed by wind and may rely on animal dispersal.

Introduced forbs (**Xanthium spinosum, *Lactuca serriola, *Cirsium vulgare* and **Rapistrum rugosum*) and grasses (**Bromus cartharticus, *Phalaris paradoxa, *Setaria incrassata* and **Avena fatua*) were more abundant in cultivated areas. All of these species, except **Setaria incrassata*, which was sown after cultivation, are annual or short-lived. These findings are supported by McIntyre *et al.* (1995) who reported higher proportions of therophytes (annuals) at soil-disturbed sites. This is because cultivation opens space for colonisation through seed establishment and favours annual species that are able to disperse into cultivated areas and exploit resources rapidly (Clements 1916; McInyre *et al.* 1995). While many native species on the Moree Plains appear resilient to grazing, they may not be so well adapted to cultivation (which results in death of the plant). This is not surprising given that prior to European settlement, soils were generally free of disturbance. Effective means of propagule dispersal, long-lived seed banks and the ability to colonise bare ground may be more important characteristics for plants surviving in cultivated landscapes (Trémont & McIntyre 1994).

While species richness increased over time at cultivated sites, similar increases in richness were observed at uncultivated sites (Fig. 4.9). Hence native species richness was still lower at cultivated sites despite 13-22 years of recovery. Only grass species richness appeared to increase more at cultivated sites between 1998 and 2004 (Fig. 4.9e), probably due to colonisation of wind-dispersed species (mentioned above). The richness and abundance of introduced species in previously cultivated areas appears to fluctuate, probably depending on climatic conditions (Fig. 4.9c). Under certain conditions (e.g. 2002 sampling), introduced species were encouraged at such sites. It is likely that introduced, colonising species were common at cultivated sites immediately after cultivation and have remained in the seed bank (Lunt 1990b), fluctuating in abundance over time as seasonal opportunities permit. With increasing colonisation and competition from native species, introduced species richness should decrease in the previously cultivated areas over time.

Given that many of the introduced species are winter-active annuals, native perennial species have a competitive advantage because they can grow at times when there is less competition from the introduced species and persist in a dormant state through winter.

Cultivated management units at Kirramingly Nature Reserve were cropped for between 15 and 34 years. While there were some breaks between cultivation (P. Wall, pers. comm., 2002), the extent of these is unknown. The length of persistence of soil seed banks within previously cultivated areas is unknown for these grasslands, but given the length of cultivation, it is unlikely many perennial species survived, if any, as perennial species are poorly represented in the seed bank after cultivation (Graham & Hutchings 1988a,b). Studies on the seed banks of grasslands have shown ruderal species reappear after cultivation and that seeds of annual species can survive in the soil for long periods (Graham & Hutchings 1988a). The survival of seed banks in previously cultivated paddocks requires further investigation in Australia. As pointed out in the previous chapter, the recovery of seed banks after cultivation is probably heavily influenced by grazing animals. The influence of grazing on composition may be positive, through the spread of seed (Montalvo *et al.* 1993; Römermann *et al.* 2005), or negative, due to lack of re-establishment of palatable perennial species (Robson 1995).

Over the summer-autumn sampling times, forbs (six species) and grasses (five species) were generally more frequent at uncultivated sites. The decrease in forb richness at cultivated sites (Table 4.7) supports this finding. However, grass species richness was only lower at cultivated sites in 1998 (Fig. 4.9e). The reductions in total and native species richness at cultivated sites appear to be due to combined reductions in forb, grass, graminoid and shrub richness (Fig. 4.9d,e,f,g). Shrub (and sub-shrub) richness was not significantly lower at cultivated sites ($P = 0.170$). This does not support findings from the regional survey (Chapter 3) where shrub and sub-shrub richness was lower at cultivated sites with a history of periodic grazing and no flooding. Perhaps the recruitment of shrubs and sub-shrubs after cultivation is more likely after longer time intervals and in the absence of livestock grazing at the Reserve.

Only one species (*Themeda avenacea*) was consistently less frequent at uncultivated sites than cultivated sites at all summer-autumn sampling times. While this species was not significantly less frequent at cultivated sites in the regional vegetation survey, ordination showed that its abundance was associated with uncultivated sites (Fig. 3.11). In terms of frequency of occurrence, only two species in this survey and the regional survey showed similar responses to cultivation. They were, **Rapistrum rugosum*, which was more common at cultivated sites, and *Astrebla lappacea*, which was more frequent at uncultivated sites. Cover abundance of *Leptochloa divaricatissima* and *Sclerolaena muricata* var. *muricata* were also consistently associated with cultivated sites in ordination, while *Themeda avenacea, Ptilotus semilanatus* and *Astrebla elymoides* were consistently associated with uncultivated sites, in both surveys.

Clarke *et al.* (1998) stated that areas around the two dams at Kirramingly Nature Reserve might require managerial intervention, as these locations are potential sources of spread of introduced species (e.g. **Lycium ferocissimum, *Silybum marianum, *Xanthium occidentale* and **Rapistrum rugosum*). Introduced species (especially forbs) were common at sites close to dams in this study. For example, in 2002, introduced species richness was 12.3 ± 0.88 (*n* = 3) at sites near dams and just 5.5 ± 0.68 (*n* =17) at other Reserve sites. The above average introduced richness at such sites probably resulted from the combined effects of cultivation and past heavy grazing, as the influence of increased livestock grazing intensity due to distance from watering points is well known (e.g. Lange 1969; Landsberg *et al.* 1997). It is also possible that temporary waterlogging due to overflow from dams has influenced such areas.

4.4.4 The influence of Acacia farnesiana *on composition*

Clarke *et al.* (1998), and Hunter and Earl (1999) reported that *Acacia farnesiana* has no effect on botanical composition. Sites with abundant *A. farnesiana* had different species composition to those with low abundance in the current study (Fig. 4.5), although it is unlikely that *A. farnesiana* itself had a major influence on composition. The significant relationship with composition recognised is probably related to the fact that *A. farnesiana* was more abundant at uncultivated sites. Frequency of *A. farnesiana* was also influenced by cultivation in the regional survey (Table 3.27). Despite this, there may be certain species associated with *A. farnesiana.* For example, *Einadia nutans* var. *linifolia* was commonly observed growing on and beneath *A. farnesiana. Rhagodia spinescens* was also commonly observed growing amongst *A. farnesiana.* This may be due to dispersal of the fleshy fruits of these species by birds, which perch on the branches of *A. farnesiana.*

Shrub encroachment of grasslands has resulted in major changes in community composition and losses in production in Australia (e.g. Booth & Barker 1981; Booth 1986; Burrows *et al.* 1991) and overseas (e.g. Walker *et al.* 1981; Archer 1991; Brown & Archer 1999; Tobler *et al.* 2003). The main loss of production through shrub invasion is due to shrub competition with pastures. *Acacia farnesiana* does not, however, appear to limit herbaceous growth through competition, as herbaceous species grow beneath its canopy. The lack of obvious competition may be due to the relatively open canopy of *A. farnesiana* allowing light to penetrate, its deep taproot, reducing competition for soil moisture (Walker *et al.* 1981) and perhaps nutrients, and its nitrogen fixing ability, making a fertile micro-environment for competitive forbs. Clarke *et al.* (1998) reported that there is anecdotal evidence that dense thickets of *Acacia farnesiana* offer protection from heavy grazing to ground stratum plants (Surrey Jacobs, pers. comm.). At the same time, however, the shade provided by these shrubs, may lead to higher grazing intensities, than in open grasslands.

4.4.5 Conclusions

This research suggests that disturbance in the form of livestock grazing is not necessary to maintain plant species richness in Kirramingly Nature Reserve. In fact, at the 2-m² scale, grazing was associated with reduced species richness. While richness was not significantly lower in grazed 0.1024-ha quadrats, several native grazing-sensitive species were identified (e.g. *Asperula cunninghamii, Leiocarpa websteri, Desmodium campylocaulon, Panicum queenslandicum, Ptilotus semilanatus* and *Themeda avenacea*). Some of these species were also less abundant or absent in cultivated areas, suggesting they may be sensitive to anthropogenic disturbance in general. Cultivation has had a more pronounced influence on species composition and richness than removal of livestock grazing. Introduced species and wind-dispersed grasses were common at sites with a history of cultivation, although the richness of introduced species varied between years.

Long-term monitoring of quadrats at Kirramingly will hopefully provide information on how long it takes for these grasslands to reach a composition similar to nearby uncultivated grassland, if it happens at all. Further monitoring is also necessary, especially in certain parts of the Reserve, to ensure perennial grass density does not limit recruitment of other species after several more years of livestock exclusion. Results of the current study support findings of previous surveys in the region (Clarke *et al.* 1998; Hunter & Earl 1999) that *Acacia farnesiana* does not pose an immediate threat to native grasslands.