Implications of grazing management systems incorporating planned rest for biodiversity conservation and landscape function in rangelands

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Declaration

I certify that the substance of this thesis has not been submitted for any degree and is not currently being submitted for any other degree or qualification. I certify that, any assistance received in preparing this thesis, and all resources used, have been acknowledged in this thesis.

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Abstract

Livestock grazing is recognised as a major driver of biodiversity decline and land degradation in rangelands around the globe. Protected areas alone cannot conserve global biodiversity, and therefore off-reserve conservation is necessary to achieve biodiversity conservation outside reserves and improve connectivity between reserves. Grazing management strategies that promote both ecological and production outcomes have the potential to conserve biodiversity and maintain or improve landscape function in agricultural landscapes. However, there is a lack of understanding of the response of biodiversity and landscape function to different grazing management systems in arid and semi-arid rangelands. This thesis explored the effects of commercial grazing practices that incorporate frequent periods of rest from grazing on biodiversity and landscape function, and determined the potential for using these alternative grazing practices to achieve broad-scale conservation outcomes.

A systematic review and meta-analyses of scientific literature comparing grazing management incorporating periods of planned rest (strategic-rest grazing, SRG) with continuously grazed (CG) and ungrazed (UG) systems was undertaken to determine the effect of SRG on ecological and animal production variables. Where significant differences occurred, the trend analysis of ecological and animal production responses to grazing management predominantly favoured SRG over CG, except for animal weight gain, and favoured SRG over UG systems for plant, mammal and bird richness and diversity, but not invertebrate richness and diversity, biomass and ground cover. Most studies that compared plant species composition reported differences in response to grazing management. While we did not find any differences overall between grazing contrasts, meta-analyses of plant richness, diversity, animal weight gain and animal production per unit area indicated that management incorporating longer periods of rest compared to periods of grazing have the potential to improve animal weight gain and production per unit area, but reduce plant richness. The type of SRG system was also important, with multi-paddock SRG systems having lower plant richness relative to CG systems, and SRG systems based on seasonal or deferred grazing having greater diversity than CG systems. Most of the literature comparing SRG with CG or UG did not consider the response of ecological and animal production response variables simultaneously. Greater collaboration between ecological and animal production scientists is recommended to better understand the ecological and socio-economic trade-offs associated with different grazing management strategies.

Understorey floristic species composition and plant biodiversity measures were compared between commercial properties managed under alternative grazing management (incorporating frequent and long periods of rest), traditional (continuous) grazing management, and adjacent ungrazed areas managed for conservation across a broad region of the semi-arid rangelands in western NSW. Significant variation in understorey floristic composition was driven by soil type (clay and sand), season, preceding rainfall and geographic location. These variables were the major drivers of floristic composition. The effect of grazing treatment on floristic composition at the regional scale was comparatively small and not significant. However, infrequent species were more likely to be recorded in conservation areas. Measures of floristic biodiversity varied with the scale of observation, season of sampling and soil type. In comparison to traditional grazing management, alternative grazing management generally resulted in greater understorey floristic species richness and diversity, depending on the season and scale of sampling. Few differences were found in plant species richness, diversity or functional diversity between alternatively grazed properties and adjacent areas ungrazed by commercial livestock and managed for biodiversity conservation. This suggests that alternative grazing management may be compatible with biodiversity conservation on commercial livestock properties in western NSW rangelands, but potentially at the expense of rare species.

Ground cover, soil properties and landscape function were also compared between alternative grazing management, traditional grazing management and conservation management in semi-arid NSW. Alternative grazing management had greater total ground cover in comparison to traditional grazing management systems. However, both alternative and traditional grazing management treatments had significantly less ground cover than adjacent areas managed for conservation. Alternative grazing management properties did not differ significantly to areas managed for conservation in terms of landscape function, but many indices of landscape function (stability, nutrient cycling, landscape organisation index, patch area and average interpatch length) were significantly reduced under traditional grazing management compared to conservation. This suggests that alternative grazing management was more beneficial for landscape function than traditional grazing management.

Significant differences were observed in floristic biodiversity measures, ground cover, soil properties and landscape function between clay and sandy soils in the study region. Clay soils had greater soil organic carbon and organic nitrogen, and lower bulk density than sandy sites. Soil stability, nutrient cycling and landscape organisation indices were also greater on clay than sand soils, and average interpatch length was shorter on clay soils. There was no difference in total ground cover between sand and clay soils, although clay soils had greater vegetative cover than sand soils, while sandy soils had greater cryptogam cover. Floristic biodiversity measures (species richness, evenness, diversity, turnover) were significantly greater on sandy than clay soils at larger plot and site scales, but there was no difference in species richness at the finest scale of sampling (1 m^2 quadrats). Despite the common perception that clay soils are more resilient to disturbance than sand communities, we found no difference between sand and clay soils in floristic biodiversity measures, ground cover, landscape function, soil organic carbon, soil organic nitrogen, or bulk density in response to grazing management. This indicates that alternative grazing management may provide a sustainable option for conservation of biodiversity and landscape function across both sandy and clay soils in western NSW semi-arid rangelands.

Floristic composition, biodiversity measures and ground cover were also compared at a local scale between an ungrazed public nature reserve and an adjacent rotationally grazed commercial property in *Acacia aneura* woodland in semi-arid NSW. Significant differences in understorey floristic composition were observed between the two grazing treatments, including a greater frequency of palatable species in the nature reserve and more unpalatable species on the rotationally grazed property. There were no significant differences in understorey floristic species richness, diversity, functional diversity measures or ground cover between the nature reserve and rotationally grazed property. However, these measures increased with distance from water on the rotationally grazed property, highlighting the negative effects of increasing grazing intensity. These results suggest that at a whole-paddock scale (beyond the sacrifice zone of high grazing intensity surrounding water points), rotational grazing management, along with careful management of grazing intensity and stocking rates, has the potential to sustain biodiversity and ground cover and may offer an alternative to grazing exclusion to achieve broad-scale conservation objectives in semi-arid rangelands. However, management would still need to address the impacts on floristic composition.

In conclusion, I found improved understorey plant species richness, diversity, ground cover and landscape function under alternative grazing management compared to traditional grazing management, and few differences in these measures between alternatively grazed and ungrazed areas managed for conservation. These results provide support for utilisation of alternative grazing management practices to improve biodiversity conservation and landscape function outside of the public reserve system in semi-arid rangelands. Results also show incorporation of planned periods of rest in grazing management regimes has the potential to achieve dual ecological and animal production outcomes in grazing landscapes throughout the world. Further research is necessary to understand the circumstances in which commercial grazing is compatible with the conservation of biodiversity, landscape function and animal productivity, and to identify best grazing management practices for biodiversity conservation purposes.

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Note to the examiners

The thesis has been written in the style of a thesis by publications. The formatting aligns with that suggested by the University of New England for consistency. As Chapters 2 to 5 have been prepared as manuscripts, there is some repetition, for which I apologise in advance.

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Chapter 1. Introduction

1.1 Background

Approximately 38% of the Earth's land surface is utilised for agricultural production, with livestock grazing accounting for the greatest proportion of agricultural land use (~66%; FAOSTAT 2016). Pressure on agricultural land is expected to increase as the world population grows from 7.5 billion to reach nine billion by the year 2050, with a corresponding 70% increase in global demand for agricultural production (FAO 2011). Poorly managed livestock grazing is recognised as a significant contributor to global biodiversity loss and land degradation (see Appendix, Table A1.1 for glossary of key terms; MA 2005; Steinfeld *et al.* 2006). Improving the environmental sustainability of agricultural production whilst maintaining economic viability and increasing productivity to meet increasing demand for agricultural produce (sustainable intensification) is a major challenge for agricultural land management (Stafford Smith *et al.* 2000; MacLeod and McIvor 2006; Bell *et al.* 2014; Rockstrom *et al.* 2017).

Biodiversity refers to the diversity of life at all levels of organisation (e.g. genetic, population, species and ecosystem) and scales (e.g. local to global), and the associated ecological processes and interactions (MA 2005; Cresswell and Murphy 2017). Biodiversity is valued for intrinsic reasons and for the provision of ecosystem services, including supporting, regulatory, cultural and provisioning roles (MA 2005; Cresswell and Murphy 2017). As a result of human disturbance, global extinction rates of species are significantly greater than historical rates, and are projected to increase (MA 2005). In addition, approximately 25% of land throughout the world is classified as highly degraded and degrading as a result of unsustainable land management practices (FAO

2011). In this thesis, a distinction is made between plant biodiversity and plant diversity. Plant biodiversity refers to the plant component of biodiversity, as reflected by multiple different measures, including the richness, Shannon–Wiener diversity, evenness, turnover and abundance of plant species, for example. Plant diversity refers to the specific measures of the Shannon–Wiener or Simpson diversity indices in relation to plant communities (the distinction being made clear in the methods of each chapter).

Rangelands are areas of natural vegetation used primarily for livestock grazing, and include grasslands, shrublands, woodlands, tundra, deserts and forests, covering 25% of the global land surface (FAO 2011). The effects of livestock grazing in rangeland ecosystems vary with climate, the type and evolutionary history of the plants and animals in the grazing system, the duration, frequency and intensity of grazing, the type of livestock and soil type (Milchunas *et al.* 1988; Milchunas and Lauenroth 1993; Landsberg *et al.* 1997b; Olff and Ritchie 1998; Hickman *et al.* 2004; Tóth *et al.* 2016). Manipulation of grazing management, including the timing, intensity and distribution of grazing, and the type of livestock that graze pasture, can be used as a tool to achieve desired plant, animal, soil and economic outcomes (Vallentine 2000). While stocking rate is considered to be the most important means of manipulating grazing management, in recent decades, there has been growing interest in the adoption of grazing management systems that incorporate long periods of rest from grazing, or rotational grazing systems, to improve the productivity and sustainability of livestock enterprises (Walker 1995). However, the benefits of rotational grazing over systems in which grazing is more or less continuous, without frequent long periods of rest, are much debated (Briske *et al.* 2008, 2011; Teague *et al.* 2013).

Understanding and improving grazing management is important to achieve both ecological and economic sustainability of livestock production, and provision of reliable information for graziers is necessary to inform management and allow the integration of both ecological and socio-economic outcomes (Neilly *et al.* 2016). However, there is a lack of understanding of the response of biodiversity and landscape function in rangelands to different grazing management systems. Accordingly, this chapter (1) reviews literature concerning biodiversity conservation and grazing management in arid and semi-arid rangelands, and the conceptual paradigms describing ecological change in arid and semi-arid ecosystems in response to livestock grazing; (2) highlights research gaps; (3) outlines the aims and objectives of the research conducted in the remainder of the thesis to address these knowledge gaps, and (4) describes the structure of the thesis and how the structure allows the aims and objectives to be addressed in subsequent chapters.

1.2 Biodiversity conservation

Biodiversity is important for the provision of ecosystem services (MA 2005). These include provisioning services (e.g. food, fresh water, fuel); regulating services (e.g. climate regulation, water purification, flood regulation); supporting services (e.g. nutrient cycling, soil formation, primary production), and cultural services (e.g. aesthetic, spiritual, educational, recreational). Through the provision of these services, biodiversity can enhance ecosystem function and resilience to natural or human disturbances, and is crucial to supporting agricultural production (Fischer *et al.* 2006). Biodiversity loss and land degradation are associated with a loss in agricultural productivity (Fischer *et al.* 2006; Reynolds *et al.* 2007; Tilman *et al.* 2012) and human well-being (MA 2005). Therefore, effective management and conservation of biodiversity is essential.

1.3 Reserve network

Approximately 14.7% of the world’s terrestrial land surface is currently held in public reserves (UNEP-WCMC and IUCN 2016) for reasons including biodiversity conservation and the protection of areas of natural, recreational and scenic value (Margules and Pressey 2000). The global target is for 17% of terrestrial land and inland waters to be conserved and managed through protected areas or through other effective conservation measures (UNEP-WCMC and IUCN 2016). In 2014, 17.9% of Australia’s land area was covered by the national reserve system. This was expected to increase to 19.2% by 2016 (Cresswell and Murphy 2017). Reserves play an important role in biodiversity conservation (Margules and Pressey 2000), but on their own, are inadequate in representing and conserving all biodiversity (James *et al.* 1995; Margules and Pressey 2000; Rodrigues *et al.* 2004; Fischer *et al.* 2006; Lindenmayer *et al.* 2010). Reserves are often concentrated in remote or unproductive areas of low economic value, and they rarely include representative examples of natural ecosystems in agriculturally productive environments (Margules and Pressey 2000). Globally, it is estimated that up to 68% of ecoregions, 78% of important biodiversity sites and 57% of species are inadequately conserved or not represented at all in protected areas (Butchart *et al.* 2015). In addition, climate change poses a significant threat to biodiversity (Heller and Zavaleta 2009) and the location of existing reserves, as species requirements and ranges change but reserves remain fixed in space (Bengtsson *et al.* 2003; Araujo *et al.* 2004; Heller and Zavaleta 2009). Selection of reserves has not taken this into account. As species ranges and community dynamics change in response to climate change, reserves will become increasingly inadequate in protecting the biodiversity currently present (Heller and Zavaleta 2009).

1.3.1 Off-reserve conservation

As a result of these shortfalls in public conservation reserves and protected areas, alternative approaches to biodiversity conservation and conservation planning are required, including private conservation and off-reserve conservation (Morton *et al.* 1995; Fischer *et al.* 2006; Heller and Zavaleta 2009; Lindenmayer *et al.* 2010; Salmon and Gerritsen 2013; Butchart *et al.* 2015). Off-reserve conservation has an important role in conserving biodiversity and complementing the public reserve system by facilitating connectivity between reserves, accommodating changes in species distributions and extending conservation over a much broader area. There is evidence to suggest that appropriately managed livestock grazing is compatible with maintaining conservation objectives and can play an important role in enhancing the biodiversity value of agricultural landscapes (Curry and Hacker 1990; Fensham 1998; Dorrough *et al.* 2004; Lunt *et al.* 2007a; Waters and Hacker 2008; Papanastasis 2009; Fensham *et al.* 2011, 2014; Savory 2013; Silcock and Fensham 2013).

1.4 Rangeland ecosystem dynamics

Rangelands are also highly organised in terms of ecosystem function, consisting of patches (run-on zones) and interpatches (run-off zones) that differ in relation to infiltration and accumulation of resources (Tongway and Ludwig 1990). The 'trigger–transfer–reserve–pulse' framework (Ludwig and Tongway 1997) describes the redistribution of resources between patches and interpatches and resulting consequences for ecological processes (Figure 1.1). In this framework, the trigger refers to inputs to the system, such as rainfall, which are then transferred via runoff or wind and captured in patches that store resources, and this then results in a pulse of production. This redistribution of resources allows for resource concentration in patches, resulting in greater production pulses than would be possible if rainfall was evenly distributed, thus maintaining the patch structure (Tongway and Ludwig 2005). Landscape function refers to the ability of landscapes to capture, retain and utilise resources such as water and nutrients (Tongway and Ludwig 1997b). Functional landscapes are able to capture and store resources, whereas dysfunctional landscapes lose excessive amounts of resources.

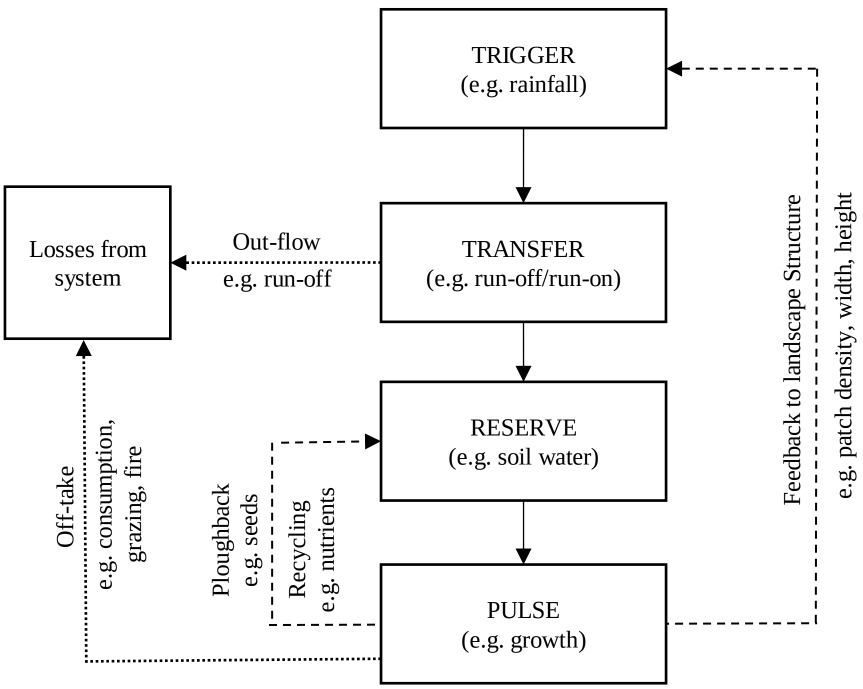

Figure 1.1. Trigger-transfer-pulse-reserve framework. Adapted from Ludwig and Tongway (1997).

1.5 Effects of grazing on biodiversity and landscape function

Livestock can have a direct effect on biodiversity and landscape function via herbivory and trampling, or indirectly via habitat modification and selective grazing, which can affect plant community composition. Livestock trampling can reduce plant, litter and cryptogam cover, and alter soil structure and function through breaking the soil surface crust, increasing bulk density (soil compaction) and reducing water infiltration (Yates *et al.* 2000). Soils impacted by grazing may be more vulnerable to soil erosion (Tongway *et al.* 2003). Livestock can also modify biodiversity and landscape function by facilitating seed dispersal, breaking the soil surface crust and enhancing seedling establishment, concentrating nutrients where dung is deposited, and altering competitive interactions between plant species (e.g. competition for light, moisture or nutrients; Olff and Ritchie 1998; Facelli and Springbett 2009).

Multiple methods exist to determine the effects of livestock grazing on biodiversity and landscape function, each highlighting different aspects. For example, while species richness and evenness indicate the number and relative contribution of species in an ecosystem, respectively, and can be combined to form a measure of diversity (e.g. the Shannon–Wiener diversity index), they provide little or no indication of other responses such as the abundance of rare or threatened species, or functional composition. Biodiversity is a complex concept comprised of multiple attributes (e.g. compositional, structural and functional diversity) at different scales of biological organisation (e.g. genetic, species, ecosystem), with many different response variables to inform the multiplicity of aspects (Duelli and Obrist 2003). While many indicators provide important information about biodiversity, it is often not economically or logistically possible to measure all. The choice of measurements and indicators is often guided by the goals of the management systems and research questions at hand. In this thesis, multiple measures of biodiversity (including plant species richness, evenness, diversity, turnover, composition, functional composition and functional diversity) and landscape function (including ground cover, bulk density, soil stability, infiltration, nutrient cycling, and patch–interpatch relationships) are discussed and reported on in order to provide broad insight into the possible effects of different grazing management systems. They are commonly used indicators in the ecological literature and are therefore comparable with many other studies.

1.5.1 Theory and conceptual models

A number of models have been developed to explain and predict vegetation response to livestock grazing. The models are based around two dominant theories: the equilibrium and non-equilibrium theories. The equilibrium theory suggests ecosystems exist in a single, stable climax state at equilibrium with climatic conditions (Ellis and Swift 1988). According to this theory, degradation of ecosystems is a result of overstocking and overgrazing (Ellis and Swift 1988). Models based on equilibrium theory include the Clementsian and range succession models, which predict that after a disturbance (e.g. grazing), vegetation returns to a predictable stable state dictated by the climatic conditions at the site (Dyksterhuis 1949; Milton *et al.* 1994). Under these models the objective is to match stocking rate with succession. Drought is recognised as a similar disturbance to grazing, and therefore stocking rate should be reduced in drought to keep the system in balance (Westoby *et al.* 1989). However, the equilibrium theory fails to account for the dominance of climatic variation, often making equilibrial conditions unattainable (Ellis and Swift 1988). In addition, vegetation change often occurs discontinuously, inconsistently and is not necessarily reversible (Westoby *et al.* 1989).

The non-equilibrium theory proposes that due to the environmental variability that drives arid and semi-arid rangeland dynamics, plant production and composition are predominantly a result of precipitation and extreme rare climatic events (Illius and O'connor 1999). The intermediate disturbance hypothesis (Grime 1973) predicts that the greatest plant species richness occurs at intermediate levels of disturbance (such as grazing intensity). Low disturbance results in competitive exclusion by dominant species, and high levels of disturbance limits plant species richness through stress, thus resulting in a bell-shaped response of plant species richness in relation to the level of disturbance. Huston (1979) proposed that variations in rates of competitive displacement between communities when recovering from disturbance determine differences in plant diversity (i.e. richness and evenness). Although communities are prevented from reaching equilibrium because of regular disturbance, biodiversity may be stable if species loss due to competitive displacement is balanced by an increase in species due to low to moderate levels of disturbance. Therefore, communities with low rates of competitive displacement should have higher plant biodiversity as the communities are further from reaching competitive equilibrium. This contrasts with communities with a high rate of competitive displacement that reach equilibrium faster (Huston 1979).

Some conceptual approaches combine equilibrium and non-equilibrium theory. For example, the state-and-transition model (Westoby *et al.* 1989) proposes that vegetation condition can exist as, and transition between, different ‘states’ on the same piece of land. Disturbances such as climatic events, fire or management actions such as overgrazing can trigger a transition between states (Westoby *et al.* 1989). Under the state-and-transition model, managers should aim to minimise hazards and seize opportunities in order to prevent degradation and to maximise productivity. The generalised model proposed by Milchunas *et al.* (1988) combines grazing history by livestock and environmental moisture or productivity gradients to explain vegetation response to grazing. Drought creates similar selection pressures to grazing. Milchunas *et al.* (1988) proposed that semi-arid grasslands with a short history of livestock grazing, such as those found in Australia, are dominated by drought-tolerant short and intermediate-height grasses. In these ecosystems, grazing at low intensity is expected to increase plant diversity (Shannon–Wiener), and to decrease diversity as grazing intensity increases. Grazing in semi-arid ecosystems with a long history of livestock is expected to have a smaller negative response as species have evolved in response to grazing pressure (Milchunas *et al.* 1988). Conversely, ecosystems in subhumid regions with a short history of grazing are expected to exhibit greater changes as species are not adapted to grazing or low soil moisture (Milchunas *et al.* 1988). Olff and Ritchie (1998) proposed a modification to the model of Milchunas *et al.* (1988), postulating that communities with large generalist herbivores (non-selective graziers) have higher plant biodiversity in comparison to those dominated by small herbivores (more selective graziers). Cingolani *et al.* (2005) also suggested a modification to the model of Milchunas *et al.* (1988), making it more compatible with the state-and-transition model by allowing for irreversible transitions as a result of varying grazing intensities.

1.5.2 Effect of rotational compared to continuous grazing

Continuous grazing is a method whereby livestock graze the same unit of land throughout the entire year or grazing season (Allen *et al.* 2011). In contrast, rotational grazing systems involve recurring periods of grazing and rest across a land unit by utilising multiple paddocks through which livestock are rotated (Allen *et al.* 2011). Rotational grazing management is claimed to be more environmentally sustainable in comparison to continuous grazing management, whilst increasing the productive potential of land (Norton 1998a; Teague *et al.* 2008). It is based on the assumption that continuous grazing leads to a decline in desirable perennial grasses as they are selectively grazed, over-utilised and do not have the opportunity to recover from grazing events (Norton 1998a). This gives a competitive advantage to less desirable and exotic or weedy species, which often increase at the expense of more desirable palatable species (Norton 1998a). In large paddocks, and at low grazing intensities, livestock do not graze the landscape uniformly (Fuls 1992; Barnes *et al.* 2008). Patch grazing is a significant problem in continuous grazing systems, even at low grazing intensities; patches previously grazed are often revisited, expanding these patches and neglecting areas not previously grazed (Teague *et al.* 2008). As a result the stocking rate of the grazed patches is much higher than the overall stocking rate of the paddock (Lange 1985; Norton 1998a; Teague *et al.* 2008). Rotational grazing can significantly reduce the negative effects of selective grazing and patch grazing as livestock graze at higher densities in smaller paddocks. This achieves a more even distribution of livestock grazing and utilisation of pasture, thus increasing the grazing pressure on species that would not normally be subject to defoliation. The vigour of palatable species is maintained by only grazing them for short periods and allowing species otherwise sensitive to grazing to recover during long periods of rest (Norton 1998a).

Research comparing rotational grazing management systems with continuous grazing systems in arid and semi-arid rangelands has produced conflicting results. There is evidence of greater plant species richness and biodiversity (Chillo *et al.* 2015), increased pasture biomass (Kahn *et al.* 2010), less bare ground (Kahn *et al.* 2010), less run-off (Teague *et al.* 2011), improved soil physical and chemical parameters (Sanjari 2008; Teague *et al.* 2011) and improved landscape function (soil stability, infiltration and nutrient cycling; Read *et al.* 2016) under grazing management incorporating planned periods of rest compared to continuous grazing. However, others report no advantage of rotational over continuous grazing (Roe and Allen 1993; Hacker and Richmond 1994; Briske *et al.* 2008; Bailey and Brown 2011; Hall *et al.* 2014). Reasons proposed as to why the expected benefits of rotational grazing have not been realised in many scientific studies include (Norton 1998b; Teague *et al.* 2013): (1) scientific grazing experiments fail to address landscape-scale issues, thus underestimating the impact of selective patch grazing; (2) the long-term consequences of management strategies are rarely examined in grazing experiments; (3) the impacts of grazing management systems are confounded by stocking rate; (4) the managers of grazing experiments fail to adapt to variable weather conditions, unlike ‘real-life’ managers, and (5) grazing experiments fail to employ adequate recovery periods.

1.5.3 Effect of grazing exclusion

Exclusion of commercial livestock is the most common approach used to achieve biodiversity conservation and landscape regeneration. Grazing exclusion for biodiversity conservation in arid and semi-arid environments is supported by a number of studies (Lunt *et al.* 2007b; Spooner and Briggs 2008; Legge *et al.* 2011; Schultz *et al.* 2011), in particular the protection of rare or grazing-sensitive species. However, others report no significant difference in plant biodiversity between grazed and ungrazed areas, supporting the integration of production and conservation in arid and semi-arid rangelands (Orr and Evenson 1991; Meissner and Facelli 1999; Beukes and Cowling 2000; Bowman *et al.* 2009; Souter and Milne 2009; Fensham *et al.* 2011, 2014; Silcock and Fensham 2013). These differences between studies may be a result of differences in scale, climate, soil type, livestock type or a combination of these. At small scales, livestock grazing can increase plant biodiversity by reducing competition and creating niches for germination; at larger scales, biodiversity declines as grazing-sensitive species are removed from communities, that is, the landscape becomes more homogeneous (Olff and Ritchie 1998; Landsberg *et al.* 2002; Kohyani *et al.* 2008). Negative effects of grazing on plant biodiversity are predicted to be greater in arid climate regions (Olff and Ritchie 1998; Proulx and Mazumder 1998; Bakker *et al.* 2006), although adaptations to frequent drought in arid environments are similar to those of herbivory and affords some resilience at low grazing intensities (Milchunas and Lauenroth 1993). Smaller herbivores are generally more selective grazers, which can also reduce plant biodiversity more in comparison to larger generalist species (Olff and Ritchie 1998; Rook *et al.* 2004; Bakker *et al.* 2006; Tóth *et al.* 2016).

There is evidence to suggest that carefully managed grazing can achieve similar conservation outcomes to those under grazing exclusion. For example, Teague *et al.* (2011) found no significant differences in soil physical and hydrological properties and bare ground between rotational grazing and grazing exclusion treatments. Daryanto and Eldridge (2010) found no difference in soil stability, nutrient cycling and infiltration between grazed and ungrazed areas, although in this study grazed areas had longer interpatch lengths and fewer patches. By contrast, Weltz and Wood (1986) reported increased bare ground, lower plant biomass and increased soil erosion under rotational grazing management compared to grazing exclusion, but rotational grazing treatments were only in place for 2–3 years, and were continuously grazed prior to implementation of the rotational grazing management. Often it can take decades for the effects of exclosures to become evident, especially in rangeland ecosystems where change is slow and often triggered by rare climatic events (Hall *et al.* 1964; Teague *et al.* 2013). In China, Cheng *et al.* (2012) reported an improvement in plant biodiversity as a result of rotational grazing in a degraded rangeland system, as opposed to grazing exclusion. Alemseged *et al.* (2011) compared rotationally grazed and ungrazed areas in conjunction with temporary cropping in an Australian semi-arid rangeland following removal of shrubs, and found low-intensity rotational grazing to have benefits for biomass and the restoration of perennial ground cover. A review by Holechek *et al.* (2006) of studies in the North American rangelands concluded that provided average utilisation of pasture biomass does not exceed 40%, grazing can have positive impacts on pasture in arid and semi-arid areas in comparison to exclusion. Few studies have been undertaken in the Australian rangelands examining the effects of alternative, well-managed, grazing strategies on biodiversity and landscape function compared to ungrazed areas, and none have set out to specifically test the potential of alternative grazing strategies as an alternate method of biodiversity conservation to grazing exclusion. A mosaic of grazing regimes has been recommended as the best approach to achieving optimal biodiversity conservation at regional scales (Leonard and Kirkpatrick 2004; Mavromihalis *et al.* 2013).

1.5.4 Effect of grazing intensity

Grazing intensity and stocking rate are often stated to be the predominant drivers of vegetation response to grazing, as opposed to grazing management strategy (Ash and Stafford Smith 1996; Stafford Smith *et al.* 2007; Briske *et al.* 2008). The challenge is to control stocking rate to maintain land in good condition through varying climatic fluctuations, production needs and land types (Hunt *et al.* 2014). Studies of the effect of grazing intensity on plant biodiversity and landscape function in Australian rangelands have produced variable results. Many have reported a negative correlation between plant biodiversity and increasing grazing pressure (Orr 1980; Landsberg *et al.* 2003). Fensham (1998) found greater plant species richness under light or intermediate grazing compared to high grazing intensity or ungrazed areas, while Landsberg *et al.* (1999) and Fensham *et al.* (2010) reported no significant difference in plant species richness with increasing grazing intensity, but a change in species composition. Similarly, a review of Australian studies by Eldridge *et al.* (2016) did not find a significant effect of grazing intensity on plant richness, although biomass, abundance and cover of plants declined with increased grazing intensity. Fewer studies have focussed on the response of native fauna to grazing intensity in Australian rangelands, although James (2003) reported increased abundance of reptiles and birds under light grazing compared to heavy grazing intensity. Similarly, Read and Cunningham (2010) found a lower abundance of small mammals and lower richness of reptiles under heavy grazing compared to light grazing intensity. However, Eldridge *et al.* (2016) did not find a significant difference in animal richness or abundance with increasing grazing intensity when reviewing Australian studies. No studies have investigated the effect of grazing intensity in rotational grazing systems in Australia, although studies overseas (Chillo and Ojeda 2014; Chillo *et al.* 2015, 2017) suggest fewer species have a negative response to a gradient of increasing grazing intensity under rotational grazing management, than under continuous grazing management.

Increasing grazing intensity has also been associated with degradation of soil structure and fertility, landscape function (stability, infiltration, nutrient cycling) and ground cover (Tongway *et al.* 2003; Eldridge *et al.* 2017). These changes are often most apparent closer to the soil surface (0–10 cm; Graetz and Tongway 1986; Greenwood and McKenzie 2001; Tongway *et al.* 2003). However, a review of Australian studies by Eldridge *et al.* (2016) found no significant effect of grazing intensity on soil functional response (index generated from soil carbon, nitrogen and phosphorus). In addition, at low stocking rates, infiltration may increase compared to ungrazed areas as soil crusts are broken by hoof action, but at higher stocking rates greater soil compaction and bulk density significantly reduce infiltration (du Toit *et al.* 2009).

1.5.5 Effect of soil type

Clay soils, such as those supporting Mitchell grasslands, are said to be resilient to livestock grazing (Orr and Holmes 1984) and other disturbance (Lewis *et al.* 2009b). Reasons for this include lower risk of soil erosion, greater fertility and the high-moisture holding capacity of clay soils, the persistence of Mitchell grass tussocks throughout drought and grazing, the role of Mitchell grass (*Astrebla* spp.) tussocks in stabilising soil, and the animal preference for ephemeral species over Mitchell grass, helping Mitchell grass persistence (Orr and Holmes 1984; Campbell 1989). By contrast, sandy and red earth communities are predicted to be less resilient to livestock grazing (Harrington *et al.* 1984).

The clay soils of Mitchell grassland communities are more fertile than the sandier soils of the semi-arid woodlands of eastern Australia (Harrington *et al.* 1984). Plant species richness and diversity responses to productivity gradients have been described as bell-shaped curves (Proulx and Mazumder 1998), where richness or diversity is limited in high-productivity environments as a result of litter accumulation and reduced light penetration, which reduce potential for seedling establishment and competitive displacement of existing species (Tilman 1993). In low-fertility environments, species germination is restricted by limited nutrients. Many studies have found that in nutrient-poor ecosystems, heavy grazing intensity can have a greater negative effect on species richness and diversity than at low grazing intensities, as limited resources in nutrient-poor ecosystems prevent regrowth after grazing, while this effect is less pronounced or reversed in nutrient-rich ecosystems (Proulx and Mazumder 1998; Bakker *et al.* 2006; Eldridge *et al.* 2016, 2017). Therefore, a greater negative response to grazing would be expected on sandy soils as opposed to more fertile clay soils.

1.6 Australian rangelands

Arid and semi-arid rangelands comprise approximately 81% of Australia's land mass, and livestock production in Australia's rangelands is a significant contributor to the Australian economy (Bastin 2008; Bell *et al.* 2014). Arid and semi-arid Australia is characterised by significant spatial and temporal variability in resource drivers, namely rainfall, soil moisture, nutrients and landscape geomorphology (Stafford Smith and McAllister 2008; Morgan *et al.* 2016). Rainfall in arid and semi-arid Australia is low, highly variable and unpredictable, and is the dominant driver of ecological processes (Noy-Meir 1973; Harrington *et al.* 1984; Stafford Smith and Morton 1990; Morton *et al.* 2011). Variability in rainfall increases as mean annual rainfall decreases (Beadle 1948). Temperature is also highly variable, with maxima above 45°C in summer and minima below 0°C in winter in many regions. In general, apart from the alluvial cracking clay floodplain communities, soils in arid and semi-arid Australia are predominantly highly weathered, highly sorted and infertile (low in phosphorus and nitrogen; Harrington *et al.* 1984; Stafford Smith and Morton 1990).

Australian rangelands have a relatively short history of ungulate grazing (<200 years), in contrast to rangelands in Africa, America and Eurasia. Prior to the introduction of European livestock, grazing pressure by native herbivores in Australian rangelands was limited by sparse water availability and predation by dingoes (*Canis lupus dingo*) and humans (Harrington *et al.* 1984).

European settlement changed the condition of the Australian rangelands significantly through the introduction of domestic livestock, development of artificial water sources, introduction of exotic plant species (e.g. buffel grass, *Cenchrus ciliaris*), elimination of dingoes from most sheep grazing areas, introduction of pest animals (e.g. rabbits and foxes), clearing of native vegetation, and changed fire regimes, in combination with droughts and floods (Harrington *et al.* 1984; James *et al.* 1999; Woinarski and Fisher 2003; Lunt *et al.* 2007a; Stafford Smith *et al.* 2007). Grazing by livestock and unmanaged feral herbivores has led to widespread land degradation, including a loss of palatable perennial species and an increase in unpalatable perennial shrubs, a decline in landscape function, increased soil erosion, degradation of soil structure and loss of production potential and biodiversity (Harrington *et al.* 1979, 1984; Morton 1990; James *et al.* 1995, 1999; Morton *et al.* 1995; McKeon 2004; Lunt *et al.* 2007a). In addition, extinction of rangeland mammals has also been significant over the past 200 years and is partly attributed to invasion by European herbivores (Morton 1990; Woinarski and Fisher 2003; McKeon 2004). This threat remains today (Woinarski *et al.* 2011; Cresswell and Murphy 2017).

Legacy effects of historical management practices have a significant effect on the floristic composition and biodiversity of Australia's rangelands (McIntyre *et al.* 2003; Monger *et al.* 2015). Recovery times from past degradation are considerably longer in arid and semi-arid regions than in temperate environments (Meissner and Facelli 1999;

Daryanto and Eldridge 2010; Seymour *et al*. 2010; Fensham *et al*. 2011), and recovery often requires significant rainfall events (Stafford Smith *et al*. 2007). Recovery of arid and semi-arid rangelands can take over 20 years (Hall *et al*. 1964; Fuhlendorf *et al*. 2001; Valone *et al*. 2002; Seymour *et al*. 2010), although some changes are irreversible such as soil loss and species extinctions. The effects of past grazing management are dependent upon the extent and intensity of the degradation event, time elapsed since the event, and the sensitivity of the ecosystem to change (Monger *et al*. 2015). It is important to understand legacy effects in order to effectively manage land into the future (Monger *et al*. 2015).

Approximately 12% of the Australian rangelands are currently protected in the national reserve system (Bastin 2011). Rangelands utilised for livestock grazing have the potential to complement the existing reserve network, due to the relative low-intensity nature of livestock production enterprises and the close relationship of biodiversity and landscape function with productivity in arid and semi-arid rangelands, compared to more temperate regions (Dorrough *et al.* 2004; Neilly *et al.* 2016). This provides producers with an incentive and opportunity to graze conservatively, in contrast to higher-productivity environments where biodiversity contributes little to production and functioning of landscapes (Kemp *et al.* 2003; Dorrough *et al.* 2004).

In recent decades there has been a significant increase in the uptake of cell grazing and holistic management strategies (McCosker 2000). These alternative grazing strategies have been promoted by commercial practitioners such as ‘Resource Consulting Services (RCS)’ and numerous anecdotal case studies have been published in the grey literature. However, while considerable research into cell grazing or holistic management practices has been undertaken in temperature Australia (e.g. Earl and

Jones 1996; Dowling et al. 2005), published peer-reviewed research is lacking for Australia’s arid and semi-arid rangelands.

1.6.1 Study region: western NSW semi-arid rangelands

Approximately 40% of New South Wales (NSW) is classified as arid or semi-arid. Livestock grazing is the dominant land use in this region, with the majority of land held under perpetual lease from the Crown and property sizes ranging from 5000 to 200 000 ha (DPI 2015). Approximately 4% of western NSW is held in public reserves (Figure 1.2; OEH 2015). Study sites for Chapters 3–5 of this thesis were concentrated in the semi-arid rangelands of north-western NSW, in the Mulga Lands and Darling Riverine Plains bioregions (Figure 1.2). Long-term mean annual rainfall in the study region ranges from approximately 400 mm at the eastern-most study site near Brewarrina, to 275 mm at the western-most site, near Wanaaring, and is spread relatively evenly throughout the year (Bureau of Meteorology 2017a). Soil and vegetation types across the study region are heterogeneous, consisting predominantly of heavy clay soils on floodplains along creeks, rivers and clay pans, and massive red earths and sandy soils with low dune development elsewhere (Harrington *et al.* 1984).

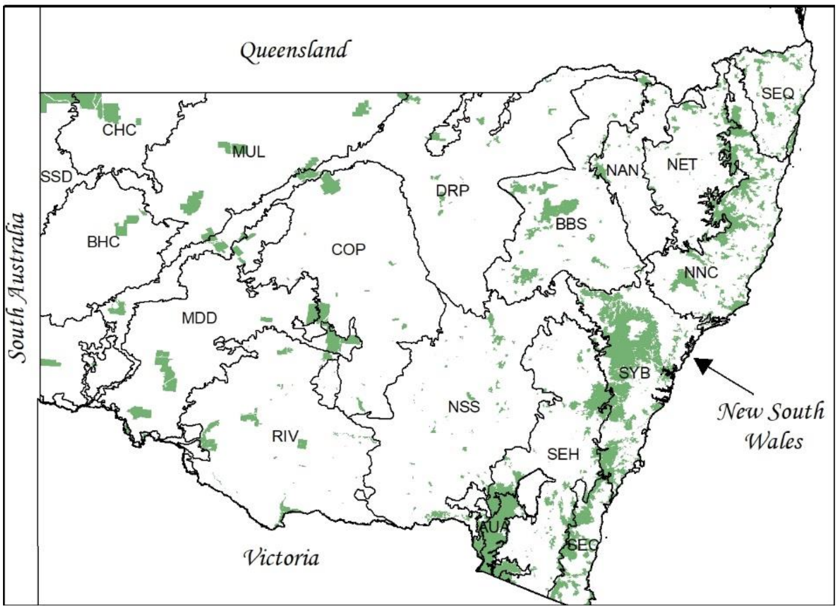

Figure 1.2. Location of protected areas in NSW (in green) and bioregions. SSD = Simpson Strzelecki Dunefields; CHC = Channel Country; BHC = Broken Hill Complex; MDD = Murray Darling Depression; COP = Cobar Peneplain; RIV = Riverina; DRP = Darling Riverine

As with other Australian rangelands, the rangelands of western NSW have undergone significant degradation since European settlement and are in a degraded but relatively stable state, largely a legacy of historical management (Green 1989). The most significant degradation event was the drought of 1896–1902, resulting in the Royal Commission of 1901 (Anon. 1901). Overstocking and rabbit plagues during this drought resulted in a loss of palatable perennial grasses, scalding and erosion, and significantly lowered the carrying capacity of land (McKeon 2004). Similar degradation events have occurred repeatedly since, due to drought (variability of rainfall and pasture growth), meat and wool price variability (affecting both build-up of livestock numbers and destocking), inadequate property sizes and government policy (McKeon 2004). In western NSW rangelands, approximately 40% of native mammal species have become extinct since European settlement, and 46% of the remaining species are threatened (Lunney 2001).

Additional challenges facing land managers in western NSW include unmanaged feral goats and native kangaroos and encroachment of invasive native scrub (INS). Feral goats are a significant threat to the conservation of western NSW rangelands, placing additional pressure on biodiversity and landscape function (Khairo *et al.* 2013). Over 2.5 million unmanaged goats are estimated to occur in the Western Division, and the population is expected to double by 2021 (Ballard *et al.* 2011). Kangaroo density is significantly higher than historic levels, and numbers are increasing (depending on seasonal conditions; Newsome 1975; OEH 2017), placing additional grazing pressure on understorey floristic species. The increase in kangaroo populations is attributed to a greater availability of water, modification of vegetation by domesticated ruminants, and elimination of dingoes (Newsome 1975; Caughley *et al.* 1987).

Since European settlement, it is widely believed that there has been a significant increase in scrub, partly as a result of changed fire regimes and overgrazing, which reduced biomass of perennial grasses that supported fuel for wild fires and competed with shrubs, in combination with rainfall events, which led to widespread shrub germination and establishment (Hodgkinson and Harrington 1985). INS creates competition for resources (water) between INS and understorey herbaceous species, grazing management difficulties (as it is difficult to travel through and muster stock from paddocks), and can result in a reduction in the carrying capacity of land as herbaceous biomass declines in response to competition with shrubs (Hodgkinson and Harrington 1985). However, a recent review by Eldridge and Soliveres (2014) highlights the positive effects of shrub encroachment on ecosystem function and ecosystem service provision, including increased biodiversity.

In western NSW, some progress has been made towards integrating livestock production and ecological goals. The West 2000 Plus Enterprise Based Conservation

Program involved landholders being provided a financial incentive to maintain ground cover at levels greater than 40%, as an alternative to payments for removing livestock for long periods (10 or more years; Hacker *et al.* 2010). This scheme provided an incentive for land managers to practise conservative grazing management at a landscape scale (Hacker *et al.* 2010). Other attempts to improve ecological outcomes on commercial properties in western NSW include providing financial assistance to develop infrastructure (e.g. subdivisional fencing or fencing to control total grazing pressure) and initiating landscape rehabilitation through water ponding or water spreading banks (Western Local Land Services 2016).

1.7 Research gaps and PhD aims and objectives

Previous research into the effects of grazing management on biodiversity in rangeland environments has produced inconsistent and inconclusive results. Moreover, little research in semi-arid Australian rangelands has compared grazed areas where rest plays an important role in the management regime, with continuously grazed areas, or areas managed for conservation where commercial livestock grazing has been excluded.

This thesis aims to explore the effects of alternative grazing practices on biodiversity and landscape function in rangelands, and determine the potential to utilise alternative grazing practices to achieve broad-scale conservation outcomes in grazed landscapes. The specific objectives of the thesis are to: (1) review the literature describing the ecological and animal production effects of utilising alternative grazing strategies compared to continuous grazing management and ungrazed areas, and determine the extent of integration between the production and ecological literature; (2) document the effect of alternative grazing practices on ground-layer floristic species composition and biodiversity measures (species richness, diversity, evenness, turnover and functional diversity) compared to traditional grazing management and conservation management strategies on contrasting soil types (clay and sand); (3) measure the effect of alternative grazing practices on soil, ground cover and landscape function compared to traditional grazing management and conservation management strategies, on contrasting soil types; and (4) determine the response of ground cover, floristic composition, biodiversity and ecosystem structure to alternative grazing management along a grazing intensity gradient.

1.8 Outline of thesis

This thesis consists of four research papers (Chapters 2–5) addressing comparisons between alternative grazing practices with traditional grazing practices and areas managed for conservation. A concluding chapter (Chapter 6) summarises findings from the thesis and provides management and recommendations for future research. Each research paper will be submitted for publication, and has been written as a stand-alone contribution.

Chapter 2 is a systematic review and meta-analysis comparing ecological and production outcomes under alternative management strategies with continuous grazing and ungrazed areas. This chapter presents a review of published literature utilising these contrasts, a meta-analysis comparing effects of alternative strategic-rest grazing management with continuous grazing and ungrazed systems on plant richness, diversity, animal weight gain and animal production per unit area, and examines the extent of integration between the production and ecological literature. This chapter highlights research gaps and provides direction for future research on these systems. It is a shared contribution principally between myself and another doctoral student with whom I collaborated.

Chapter 3 compares effects of alternative grazing management, traditional grazing management and conservation management on understorey floristic species composition and biodiversity (richness, evenness, Shannon–Wiener diversity, turnover and functional diversity measures) at three scales on contrasting soil types utilising adjacent-paddock contrasts across a large region of NSW semi-arid rangelands.

Chapter 4 compares effects of alternative grazing management, traditional grazing management and conservation management on soil properties (bulk density, pH, electrical conductivity, soil carbon and nitrogen), ground cover and landscape function indices (stability, nutrient cycling, infiltration, landscape organisation, patch area) on contrasting soil types in NSW semi-arid rangelands.

Chapter 5 reports a detailed study of one property that uses rotational grazing management and compares ground cover, floristic composition, understorey floristic biodiversity measures (richness, Shannon–Wiener diversity, evenness, turnover, functional diversity) and woody vegetation density with an adjacent nature reserve. Changes in response to a gradient of grazing intensity surrounding water points on the rotationally grazed property are also documented.

Chapter 6 presents a synthesis of Chapters 2–5 and draws conclusions regarding the potential to utilise alternative grazing management practices as alternative methods of off-reserve conservation in NSW semi-arid rangelands. Contributions of this research to current theoretical and practical knowledge are explained, limitations of the research are outlined and recommendations and directions for future research are suggested.

Chapter 2. Ecological and animal production outcomes under strategic rest grazing: A systematic review and meta-analysis

2.1 Abstract

Livestock grazing is important for food and economic security worldwide but can have considerable ecological impacts when managed inappropriately. Land sparing is often considered to be a solution, but with increasing human population growth and a growing demand for food, removing land from production may not be viable in many situations. Grazing management practices that incorporate periods of planned rest (i.e. strategic-rest grazing, SRG) may achieve both ecological and production goals simultaneously. We conducted a systematic review and meta-analyses to investigate the extent to which ecological and production attributes differ in response to SRG compared with continuously grazed (CG) and ungrazed (UG) areas across the world's biogeographic regions. Despite finding no overall differences between grazing treatments, the meta-analyses of plant richness, diversity, animal weight gain and animal production per unit area indicated that management incorporating longer periods of rest compared to length of the grazing period has the potential to improve animal weight gain and production per unit area, although at the expense of species richness. The type of SRG system was also important, with multi-paddock SRG having lower plant richness than CG, and SRG based on seasonal or deferred grazing having greater plant diversity than CG. Trend analyses of most ecological and animal production responses to grazing management predominantly favoured SRG over CG

and UG treatments, where significant differences occurred. The majority of studies that assessed plant species composition reported a difference between SRG and CG or UG, suggesting that plant richness and diversity do not effectively capture floristic changes. The relatively low number of negative effects of SRG compared to UG areas suggests that SRG may provide an alternative means of biodiversity conservation where grazing exclusion is unviable. Only a small proportion of studies considered the effect of SRG on both ecological and production outcomes simultaneously. Addressing this knowledge gap could assist in greater integration of conservation and production outcomes in agricultural landscapes globally.

2.2 Keywords

Biodiversity, livestock grazing, continuous grazing, grazing exclusion, grazing management, rotational grazing, species richness, weight gain

2.3 Introduction

Livestock grazing occupies ~25% of global land area (around 66% of the world's agricultural land; FAOSTAT 2016) and is the single most extensive use of land on the planet (Asner *et al.* 2004). On a global scale, the livestock industry is worth $1.4 trillion to the economy, employs over 1.3 billion people, and provides about 33% of human protein intake and 17% of dietary energy (Thornton 2010). However, the livestock sector is also a key driver of land-use change and degradation of ecosystem structure, function and composition (Dorrough *et al.* 2004; Eldridge *et al.* 2016), biodiversity loss (MA 2005; Steinfeld *et al.* 2006) and soil degradation (Yates *et al.* 2000; Greenwood and McKenzie 2001; MA 2005; Steinfeld *et al.* 2006). Increasing global population and corresponding demands for food production are placing increased pressure on grazing lands (Tilman *et al.* 2001; Foley *et al.* 2005; Steinfeld *et al.* 2006; FAO 2011). Excluding livestock from degraded areas is often seen as a means of achieving biodiversity conservation (Fleischner 1994; Pettit *et al.* 1995; Prober and Thiele 1995; Spooner *et al.* 2002; Eldridge *et al.* 2016). However, this inevitably comes with a loss of food and fibre production, and can be expensive and difficult to achieve across large scales (Neilly *et al.* 2016). In many parts of the world, large herbivores co-evolved with vegetation communities, allowing plant species to adapt to grazing pressure and develop mechanisms to tolerate herbivory (Coughenour 1985; Milchunas *et al.* 1988). Livestock can increase plant diversity by reducing the dominance of competitive plants (Grime 1973; Elias and Tischew 2016), providing opportunities for plant regeneration (Grubb 1977; Belsky 1992), facilitating seed dispersal throughout the landscape (Olff and Ritchie 1998; Rook and Tallowin 2003; Albert *et al.* 2015) and providing a management tool to achieve conservation objectives (Wallis De Vries *et al.* 1998). Livestock grazing managed with ecological as well as production goals provides a land-sharing option that is an alternative to grazing exclusion, potentially promoting biodiversity while avoiding or preventing degradation and the negative socio-economic consequences of removing land from production.

In contemporary grazing systems, a given area of land is commonly grazed continuously (year-long or throughout the entire grazing season) without periods of planned rest. This can result in heterogeneous patterns of use of the landscape by animals (i.e. patch grazing; Adler *et al.* 2001; Fuhlendorf and Engle 2001). Such grazing patterns can assist in maintaining diversity across the landscape by altering the relative dominance of plants through selective grazing, and leaving some areas little grazed (Bailey *et al.* 1996), as well as altering nutrient status across grazing units (Taylor *et al.* 1985). However, these heterogeneous grazing patterns can also exacerbate the negative effects of grazing because palatable and grazing-sensitive species can be overgrazed and lost from plant communities (Teague and Dowhower 2003; Teague *et al.* 2004, 2008; Norton *et al.* 2013). These patterns often increase with increasing size of management units (Stuth 1991; Bailey *et al.* 1996), and in extreme cases overgrazing exacerbates land degradation processes such as soil erosion (Blackburn 1984). Incorporating periods of planned rest into grazing regimes through the strategic movement of livestock, hereafter called strategic-rest grazing (SRG), is an alternative to continuous grazing management and may avoid some of the negative impacts that occur through the heterogeneous use of the landscape by livestock. While grazing practices that incorporate planned rest are commonly promoted to avoid environmental degradation and improve productivity (Norton 1998a; Teague *et al.* 2008), considerable debate exists around the benefits of these grazing management systems (Holechek *et al.* 2000; Briske *et al.* 2008; Brown 2009; Teague *et al.* 2013). If the claimed benefits are realised, strategic rest may provide an alternative approach to continuous grazing or livestock exclusion in some situations. This could potentially allow for continued animal production whilst simultaneously achieving improved ecological outcomes, including biodiversity conservation (Holechek *et al.* 2006; Lunt *et al.* 2007a; Papanastasis 2009; Metera *et al.* 2010).

Reviews to date comparing SRG with continuous grazing strategies have mainly focussed on how the former benefits animal production (e.g. Heady 1961; Holechek *et al.* 2000; Briske *et al.* 2008). Generally, these reviews have concluded that there is little difference in outcomes for animal production or rangeland sustainability (e.g. maintenance of biomass or ground cover to sustain long-term production) between contrasting management systems (Gammon 1978; O'Reagain and Turner 1992; Holechek *et al.* 2000; Briske *et al.* 2008), with little or no attention given to biodiversity conservation. Instead, biodiversity outcomes are typically considered in comparisons between grazed and ungrazed areas (Fleischner 1994; Yates *et al.* 2000; Spooner *et al.* 2002; Lunt *et al.* 2007a) or different grazing intensities (Holechek *et al.* 2006; Wallis De Vries *et al.* 2007; Eldridge *et al.* 2016). To understand the extent to which dual ecological and production outcomes can be achieved with SRG, it is necessary to consider both ecological and production outcomes simultaneously in research studies. Several authors in recent decades have called for greater communication, collaboration and integration between animal production research and ecological research to bridge these disciplinary silos (Jackson and Piper 1989; Fuhlendorf and Engle 2001; Watkinson and Ormerod 2001; Dorrough *et al.* 2004; Vavra 2005; Fischer *et al.* 2006; Metera *et al.* 2010; Glamann *et al.* 2015). If we are to gain an understanding of the potential for dual ecological and production outcomes under SRG in comparison with more conventional approaches, it is essential to address this knowledge gap.

This study aimed to examine both the ecological and animal production outcomes of incorporating strategic or planned rest into livestock grazing regimes. We investigated how ecological and animal production response variables compared under SRG to continuously grazed (CG) and ungrazed (UG) systems, and the effect of climate, type of SRG management and length of the graze and rest periods on these responses. We also explored the extent to which research has considered both ecological and animal production effects of SRG management simultaneously.

2.4 Methods

2.4.1 Literature review

A systematic literature review was conducted using Scopus, returning articles from 1950 until November 2016. We searched for studies that compared SRG systems (where land is rested for a planned period) with either CG systems or UG areas. Title, keywords and abstracts were searched for the following terms: (graz*) AND (*divers* OR biomass OR "carrying capacity" OR "weight gain" OR conserv* OR richness OR product*) AND (rotation* OR cell OR tactical OR holistic OR adaptive OR "short duration" OR planned OR continuous OR "set stocked" OR "set stocking" OR shepherd* OR "high intensity" OR "low frequency" OR "time controlled" OR "time control" OR "multi paddock" OR multipaddock OR "restorative" OR "grazing management" OR rest OR regenerat* OR "grazing system" OR "grazing regime" OR "grazing strategy" OR nomadic OR herding OR herder OR seasonal OR "active grazing"). Studies were only included if grazing animals were domesticated ruminants (e.g. cattle, sheep, goats, deer), the studies were published in English, and they reported above-ground biotic or animal production variables. Studies based on models or simulations were not included. In total, 250 articles were retained (see Appendix, Table A2.1).

2.4.2 Trend analysis

Two databases were constructed: one included all studies reporting SRG−CG contrasts, and the other included all studies reporting SRG−UG contrasts. For each study, we recorded the geographical region in which the study was undertaken (Europe, Eurasia, Middle East, Africa, North America, South America and

Australia/New Zealand), climatic zone (tropical, arid, temperate, cold) based on the Koppen–Geiger Climate Classification (Peel *et al.* 2007), and all above-ground biotic and animal production response variables reported for each SRG–CG and SRG–UG comparison. For each response variable, the effect of SRG relative to CG and UG treatments was recorded as either significantly greater ($P \leq 0.05$) or significantly lower, or no difference (neutral). Species composition was recorded as a difference or no difference between the grazing systems. Where the statistical significance of comparisons was not provided in studies, we determined trends based on the authors' interpretation. When opposing trends were present across multiple contrasts it was denoted as neutral. From this information, we calculated the proportion of studies conducted in different regions and climate zones, and the proportion of SRG–CG and SRG–UG comparisons reporting a greater, lesser or neutral response for each variable.

2.4.3 Meta-analysis – data extraction and synthesis

Meta-analyses were performed on subsets of studies reporting on the most frequent biodiversity and animal production response variables in the trend analysis. These were plant species richness, plant species diversity, weight gain per animal and animal production per unit of land area. While we recognise the limitations of total species richness and diversity measures in representing ecosystem function, their frequent use in the literature allowed for quantitative comparisons. We compiled a dataset for each of the four response variables, on which corresponding meta-analyses were conducted. In each dataset, information was collated for each independent grazing contrast (comparing an SRG treatment with a CG or UG treatment) about the mean, standard deviation and sample size of each response variable, along with the explanatory variables, climate zone, type of SRG system and the rest:graze ratio (see Table 2.1 for definitions). Information on geographic region, stock type, method of calculation of richness and diversity, and the type of diversity index was also recorded (Table 2.1). Where this information was not provided either in the text or as supplementary information, the study was not included in meta-analyses. Where the same data was reported in multiple papers, data from only one paper was included. We undertook all analyses using the metafor (v.1.9-6) and metagear (v.0.4) packages (Viechtbauer 2010; Lajeunesse 2016) within the R open-source software environment (Version 3.4.0; R core team 2017).

Table 2.1. Description and method of calculation of terms included in meta-analyses

Variable	Description	Method of calculation
Mean	Mean value of response variable presented in study	Obtained from text, tables or figures. When data were presented for multiple years, an average was taken
Standard deviation	Standard deviation (SD) of the mean response	When provided, obtained directly from text, tables or figures. If SD was not provided, SD was determined from the SE, *P* value or LSD. Imputation was used when no measure of variance was presented in the paper. Where studies were averaged across years, average SD was used
Sample size	Number of replicate measurements used to determine mean	Number of individual quadrats (for richness and diversity) or animals (for weight gain and animal production per unit area). Multiplied by number of years of study. When unclear, a best-decision was made based on available information
Climate zone	Climatic zone that study was undertaken in (four levels): (1) tropical; (2) arid, (3) temperate, and (4) cold	Based on Koppen-Geiger climate classification (Peel *et al.* 2007)
Grazing system	Type of SRG management system (two levels): (1) multi-paddock – stock moved between two of more paddocks, and (2) seasonal: grazed during certain seasons or during part of the grazing season (not rotated among paddocks)	Information provided in text
Rest:graze ratio	Length of time an area of land was rested relative to length of time an area of land grazed, during grazing season or year	Information provided in text. Length of rest time ÷ length of graze time

Variable	Description	Method of calculation
Geographical region	Region that study was undertaken in (seven levels): (1) Europe, (2) Eurasia, (3) Middle East, (4) Africa, (5) North America, (6) South America, (7) Australia/New Zealand	Information provided in text
Stock type	Type of livestock that grazed in study area (e.g. sheep, cattle, goats, deer, mixed)	Information provided in text
Calculation method	Method of calculation of species richness and diversity (3 levels): (1) mean of quadrats, (2) sum of quadrats, (3) point method	Information provided in text
Diversity index	Index of diversity used in study (2 levels): (1) Shannon–Wiener diversity index, and (2) Simpson's diversity index	Information provided in text
Animal production per unit area – unit type	The unit in which animal production per unit area was reported: (1) kg per ha^{-1}, and (2) kg per ha^{-1} per day	Calculated using information provided in text

We calculated the effect sizes of each comparison as the log response ratio ($\ln RR$) (Hedges *et al.* 1999):

$$\ln(\frac{X_T}{X_R})$$ **Equation 2.1**

where X_T was the mean value of the response variable (in either the UG or CG system) and X_R is the mean value for the SRG system. The $\ln RR$ quantified the log proportional change between means of each grazing system. If the $\ln RR > 0$ (positive), the response was greater for CG or UG systems, whereas if $\ln RR < 0$ (negative), the response outcome was greater under SRG. The $\ln RR$ has been widely used in the ecological literature and in comparable recent meta-analyses on grazing practices (e.g. Piñeiro *et al.* 2013; Eldridge *et al.* 2016). The statistical properties of the $\ln RR$ allow complex data structures to be modelled appropriately (Lajeunesse 2011). Although unweighted analyses are common in the ecological literature, such an approach can bias overall effects by giving equal weight to studies of differing precision (Koricheva *et al.* 2013). We undertook a weighted analysis to account for the heterogeneity in sample size and associated variance among studies. The sampling variance of $\ln RR$ was calculated as:

$$\mathrm{var}(RR) = \frac{(SD_T{}^2)}{N_T X_T^2} + \frac{(SD_{TC}{}^2)}{N_C X_C^2}$$ **Equation 2.2**

This variance helped to limit the influence of studies with low statistical power (i.e. those with a low sample size or large standard deviations; Hedges and Olkin 1985). Analyses that included studies with multiple contrasts, common treatments and a correlated error structure and were weighted with a variance–covariance matrix that accounted for this dependency (Lajeunesse 2011, 2016).

Multi-level random-effect (MLRE) models were fitted for each response variable for SRG–CG and SRG–UG comparisons separately. These model types are appropriate for ecological meta-analyses as they account for the non-independence among effect sizes through the inclusion of random effects and variance–covariance matrices (Viechtbauer 2010; Lajeunesse 2011; Nakagawa and Santos 2012; Koricheva *et al.* 2013). MLRE models were initially fitted without explanatory variables, to assess if the overall effect size differed significantly from zero (i.e. a null model). We included the random term of study, livestock type and geographical region in null models. In the null models of plant species richness and diversity, we added an additional random effect, calculation method, to account for differences in how species richness and diversity were estimated in each study. In null models of the effect size of plant diversity, a random term for the type of diversity index was also included (Table 2.1). In the animal production per unit area models, we added a random term for the type of unit (Table 2.1).

To explain variability in effect size, we fitted MLRE models including the explanatory variables of type of SRG grazing system, climate zone and rest:graze ratio as fixed effects. We tested individual factor levels by re-levelling each variable. Effect-size heterogeneity in each model was assessed using the least squares extension of Cochran's Q-test (Q_E; Hedges and Olkin, 1985; Veichtbauer, 2010). A significant Q_E-value indicated that effect size differed more than expected due to sampling variability (Hedges and Olkin 1985). It was not possible to calculate a rest:graze ratio for every study (see Appendix, Table A2.2). We undertook separate analyses of studies where we could calculate the rest:graze ratio. We assessed the influence of rest:graze ratio in conjunction with climate zone. Analysis of variances revealed that rest:graze ratio and grazing system were correlated in all response variable datasets except plant diversity, and therefore represented dependent quantitative and qualitative measures of grazing practice; they were therefore not analysed together. Rest:graze ratio was not analysed in SRG–UG comparisons as a large proportion of studies in this dataset failed to report data relating to the rest:graze ratio (see Appendix, Table A2.2).

MLRE models were fitted using maximum likelihood. We assessed the significance of the fixed effects using two tests: an omnibus test (Q_M) and likelihood-ratio tests (χ^2) (Viechtbauer 2010). Model selection was guided by assessment of the fit-statistics AIC, AIC_c, BIC and log-likelihood. In selecting models for plant species richness and diversity, we focussed on AIC_c, given the small sample sizes. A difference in AIC or AIC_c value of >2 was considered better than the null model. Homogeneity of variance was assessed by visualising model residuals against fitted values. For species richness with the SRG–UG comparison, homogeneity of variance was not satisfied in testing the effect of the full model in conjunction with the rest:graze ratio. Therefore, the effect of the rest:graze ratio for species richness was tested independently. Similarly, the full model for SRG–UG for plant diversity failed the model assumptions, so grazing system and climate were assessed individually. Model over-parameterisation was assessed by visualising likelihood-profile plots. Over-parameterisation was defined by the presence of ‘flat’ profile plots or gaps in likelihood profile due to lack of convergence (Viechtbauer 2010). Parameterisation was improved by either changing optimisation settings and re-checking profile plots or reducing the number of parameters.

Publication bias was assessed using Egger’s regression test (Egger *et al.* 1997; Sterne and Egger 2005), which is appropriate for use with MLRE models (Habeck and Schultz 2015). We re-ran each null model with the sampling variance as a fixed effect (see Appendix, Table A2.3). If the intercept differed significantly from zero in these tests, asymmetry in the relationship between sampling variance and effect size was demonstrated (Sterne and Egger 2005). A significance level of P = 0.1 was adopted following Egger *et al.* (1997) and Habeck and Schultz (2015).

2.4.4 Studies integrating ecological and animal production outcomes

Studies that reported on both ecological and animal production variables were classified as 'integrated'. We considered ground cover, biomass and plant species composition as relevant response variables to both ecological and animal production outcomes. Consequently, for each study, a decision was made whether these variables related to ecological or animal production outcomes.

2.5 Results

2.5.1 Literature review

We recorded 44 response variables in our total set of 250 articles. The most commonly reported response variables were biomass (117 studies), plant composition (97), livestock weight gain (82) and ground cover (58). Most studies were undertaken in North America (36%), followed by Australia/New Zealand (27%) and Europe (14%). A little more than half of the research (54% of studies) was conducted in temperate regions. Most of the remaining research was evenly split between arid (20%) and cold climates (24%). Very little comparative SRG research had been conducted in the tropics (2%).

2.5.2 Trend analysis

Most of studies comparing plant, mammal and bird species richness and diversity between SRG and CG systems reported no difference between grazing treatments

(Table 2.2). Where differences were observed, more studies reported greater plant species richness, plant diversity, mammal diversity and invertebrate richness and diversity under SRG than CG management. Of the studies that compared SRG and UG systems, the most commonly reported result for plant and invertebrate richness and diversity and bird richness was no difference. Where differences were reported, most reported greater plant species richness and mammal and bird richness and diversity under SRG than UG areas, but lower invertebrate richness and diversity, and equal numbers of lesser and greater responses for plant diversity. Only 37 studies (15%) considered faunal (e.g. birds, reptiles, mammals, invertebrates) response variables as opposed to plant response variables.

Increased animal production per unit area was more frequently reported under SRG than CG, whereas the most frequent result in SRG−CG comparisons of animal weight gain was no difference (Table 2.2). However, when differences did occur, animal weight gain was more often favoured under CG than SRG management (Table 2.2).

Ground cover and biomass were more frequently reported as greater under SRG compared to CG, than the reverse (Table 2.2). However, a large proportion of studies also reported no difference in ground cover and biomass between these grazing management systems. In the SRG–UG comparison, more studies reported neutral or lower biomass and ground cover under SRG than UG. Of the studies that reported impacts on plant species composition, most reported differences between SRG and CG or UG treatments (75% and 73%, respectively). Of the studies that reported both plant species composition and richness, 88% reported a difference in plant species composition between SRG and CG, and 86% reported a difference in plant species composition between SRG and UG areas.

Table 2.2. Trends in response variables (percent of total papers) in studies that compared strategic-rest grazing (SRG) with continuous grazing (CG) and in studies that compared strategic-rest grazing to ungrazed (UG) areas. Response variables are listed with overall number of studies beside it in parentheses (n). For each individual contrast the number of studies is shown in parentheses beside the parentheses (n).

Variable (n)	SRG–CG			SRG–UG		
	Lesser (%)	Greater (%)	Neutral (%)	Lesser (%)	Greater (%)	Neutral (%)
Plant richness (39)	12 (3)	34.5 (9)	54 (14)	17 (4)	39 (9)	44 (10)
Plant diversity (26)	6 (1)	41 (7)	53 (9)	26.5 (4)	26.5 (4)	47 (7)
Mammal richness (3)	0 (0)	0 (0)	100 (2)	0 (0)	100 (1)	0 (0)
Mammal diversity (2)	0 (0)	100 (1)	0 (0)	0 (0)	100 (1)	0 (0)
Bird richness (6)	20 (1)	20 (1)	60 (3)	0 (0)	0 (0)	100 (4)
Bird diversity (2)	0 (0)	0 (0)	100 (1)	0 (0)	100 (2)	0 (0)
Invertebrate richness (10)	0 (0)	86 (6)	14 (1)	29 (2)	14 (1)	57 (4)
Invertebrate diversity (4)	0 (0)	50 (2)	50 (2)	50 (1)	0 (0)	50 (0)
Biomass (117)	4 (4)	48 (51)	48 (51)	34 (11)	25 (8)	41 (13)
Ground cover (52)	6 (3)	48 (22)	46 (21)	40 (10)	4 (1)	56 (14)
Weight gain (82)	31 (25)	17 (14)	52 (43)	NA	NA	NA
Animal production per unit area (38)	16 (6)	47 (18)	37 (14)	NA	NA	NA

2.5.3 Meta-analyses

Between SRG and CG, meta-analyses revealed no significant difference overall in either plant species richness (z = 0.74, P = 0.460) or plant diversity (z = –0.81, P = 0.419; Figure 2.1). Similarly, no significant difference in plant richness (z = 0.71, P = 0.478) or diversity (z = 0.68, P = 0.499) was observed between SRG and UG areas. However, there was a significant amount of residual heterogeneity in models (plant richness in SRG–CG comparisons Q_E = 655.29, P < 0.001; plant richness, SRG–UG comparisons: Q_E = 1145.55, P < 0.001; plant diversity, SRG–CG comparisons:

$Q_E = 60.59$, $P < 0.001$; plant diversity, SRG–UG comparisons: $Q_E = 139.41$, $P < 0.001$).

Differences in plant species richness and diversity between CG and SRG were best explained by SRG type (richness: $\chi^2 = 8.24$, $P = 0.004$; $Q_M = 14.47$, $P < 0.001$; diversity: $\chi^2 = 6.12$, $P = 0.013$, $Q_M = 17.34$, $P < 0.001$). In multi-paddock SRG systems, the plant richness effect size was positive ($z = –2.47$, $P = 0.014$); in other words, richness was greater under CG than with multi-paddock SRG. However, the plant diversity effect size did not differ from zero ($z = 0.86$, $P = 0.392$). In seasonal SRG systems, plant richness effect size was not significant ($z = –0.41$, $P = 0.670$), but the plant diversity effect size was weakly negative ($z = –1.65$, $P = 0.099$); in other words, diversity was marginally greater under seasonal SRG than CG (Figure 2.1). Further, the rest:graze ratio was weakly positively related to plant richness effect size ($z = 1.74$, $P = 0.083$; Figure 2.2a). However, a significant amount of residual heterogeneity remained in these meta-analyses despite the inclusion of explanatory variables (SRG–CG, richness: $Q_E = 613.35$, $P < 0.001$; SRG–CG, diversity: $Q_E = 47.96$, $P < 0.001$; rest:graze ratio: $Q_E = 583.55$, $P < 0.001$).

Differences in plant richness between SRG and UG were best explained by climate ($\chi^2 = 10.92$, $P = 0.004$; $Q_M = 189.97$, $P < 0.001$). Studies in arid climates differed significantly from those in temperate and cold climates (arid versus temperate: $z = – 5.71$, $P < 0.001$; arid versus cold: $z = –13.67$, $P < 0.001$; Figure 2.1a). Arid climate zones showed a weak trend of lower species richness in SRG than UG areas ($z = 1.62$, $P = 0.104$). Temperate and cold climates both exhibited zero trend (temperate: $z = –0.77$, $P = 0.441$; cold: $z = –0.30$, $P = 0.768$). Again, the residual heterogeneity was significant ($Q_E = 702.56$, $P < 0.001$). There were no differences associated with explanatory variables in comparisons of plant diversity between SRG and UG systems.

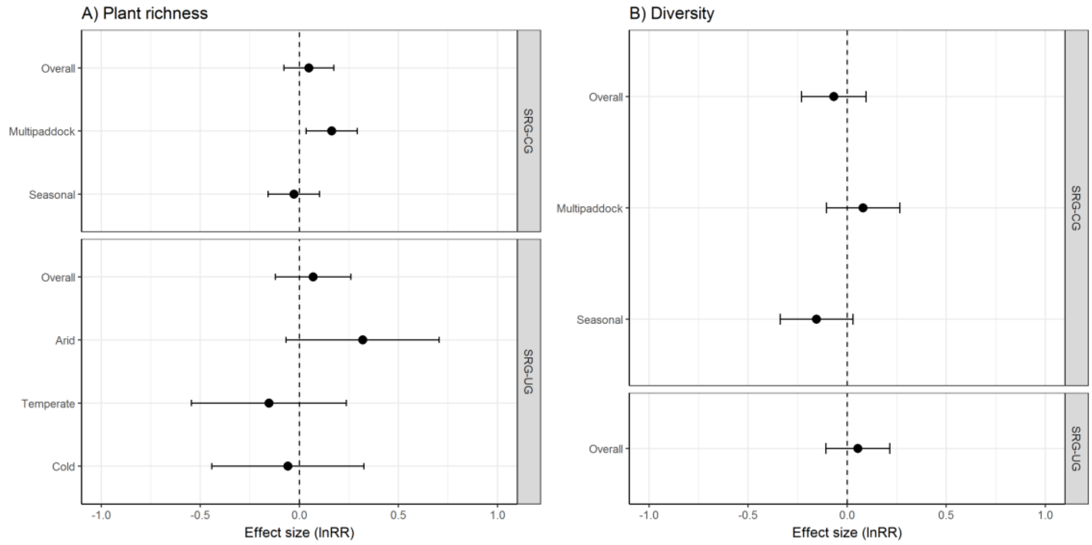

Figure 2.1. Estimated effects of differing grazing practices on species richness and diversity. (A) Species richness: SRG–CG (Top): overall effect (null model) and differences between multi-paddock and seasonal grazing systems; SRG–UG (bottom): overall effect and differences between different climate zones. (B) Diversity: SRG–CG (top): overall effect (null model) and between multi-paddock and seasonal grazing systems. SRG–UG (bottom): overall effect. (A and B) represent estimated effects (± C.I) from MLRE models. SRG = strategic-rest grazing, CG = continuous grazing, UG = ungrazed

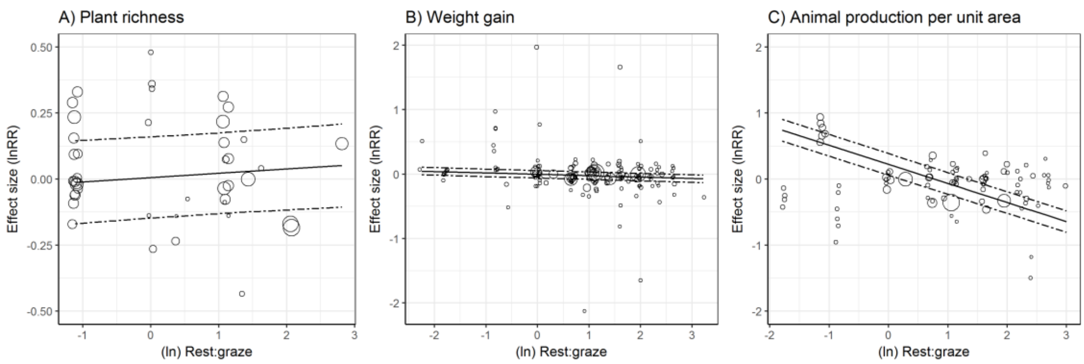

Figure 2.2. Relationships between the effect size and rest:graze ratio for the SRG–CG comparison for: A) plant richness; B) livestock weight gain; and C) animal production per unit area. Circle size represents the inverse of sampling variance. Larger circles indicate those contrasts with greater precision and higher weight in the analysis. Solid line = predicted values, dotted line = 95% confidence interval. SRG = strategic-rest grazing, CG = continuous grazing, UG = ungrazed

Effect sizes for livestock weight gain and animal production per unit area did not differ between SRG and CG (weight gain: $z = –0.84$, $P = 0.400$; animal production per unit area: $z = –1.42$, $P = 0.155$). However, there was a significant residual heterogeneity in the analyses of weight gain and animal production per unit area (weight gain: $Q_E = 10144.68$, $P < 0.001$; animal production per unit area: $Q_E = 85865.24$, $P < 0.001$). Relative differences in animal weight gain and production per unit area between CG and SRG were best explained by the rest:graze ratio (weight gain: $\chi^2 = 72.34$, $P < 0.001$; $Q_M = 72.40$, $P < 0.001$; animal production per unit area: $\chi^2 = 3969.16$, $P < 0.001$; $Q_M = 3971.76$, $P < 0.001$). In both analyses, the rest:graze period was negatively associated with effect size (weight gain: $z = –8.51$, $P < 0.001$; animal production per unit area: $z = –63.02$, $P < 0.001$; Figure 2.2b and 2.2c). Therefore, an increase in rest:graze period was associated with increased weight gain and animal production per unit area under SRG. However, for animal production per unit area, a low rest:graze ratio was associated with greater production under CG than SRG. Judging from the upper bounds of the confidence intervals, these responses were most pronounced with rest:graze ratios of seven to one in favour of rest for weight gain and four to one in favour of rest for animal production per unit area. There was still significant residual heterogeneity in effect sizes for both production variables (weight gain: $Q_E = 9897.94$, $P < 0 .001$; animal production per unit area: $Q_E = 34241.62$, $P < 0.001$).

2.5.4 Meta-analyses – model parameters

Multi-paddock SRG systems had a significantly higher rest:graze ratio than seasonal SRG systems in analyses of livestock weight gain ($F_{1,155} = 112.40$, $R^2 = 0.416$, $P < 0.001$), animal production per unit area ($F_{1,84} = 144.80$, $R^2 = 0.63$, $P < 0.001$) and plant richness (SRG–CG: $F_{1,41} = 4.00$, $R^2 = 0.067$, $P = 0.050$; SRG–UG: $F_{1,13} = 8.66$,

R^2 = 0.353, P = 0.011; see Appendix, Table A2.4). Rest:graze ratio was not significant in relation to type of SRG system and plant diversity (SRG–CG: $F_{1,12}$ = 0.89, R^2 = 0, P = 0.363; SRG–UG: $F_{1,13}$ = 2.73, R^2 = 0.110, P = 0.122).

Asymmetry between effect size and sampling variance was only observed in one dataset, the SRG–UG comparison for plant richness (P = 0.067). No other dataset displayed an indication of publication bias.

2.5.5 Studies reporting ecological and production outcomes

Of the 250 studies, similar proportions reported the effects of grazing on ecological variables (60%) and animal production variables (56%). Only 16% of studies reported the effects of grazing on both ecological and animal production response variables simultaneously. These integrated studies mostly reported ground cover and biomass responses. Of the studies that compared plant species richness, plant diversity, livestock weight gain and animal production per unit area between SRG and CG, 12%, 12%, 17% and 13%, respectively, were integrated. Of the studies comparing SRG with UG, no studies reporting plant richness or diversity results were integrated. Weight gain and animal production per unit area were not relevant in this analysis due to the nature of the comparison (no animal production in UG areas).

2.6 Discussion

2.6.1 Effect of strategic-rest grazing on ecological variables

The most frequent outcome in our trend analysis of 250 studies was that ecological outcomes did not differ with grazing treatment. Our meta-analyses also found no significant difference in plant species richness or diversity between grazing treatments. While there have been suggestions that SRG may lead to improved biodiversity (Norton 1998a; Provenza *et al.* 2003; Teague *et al.* 2008; Lindsay and Cunningham 2009), the lack of difference between SRG and CG is consistent with a large body of work that has concluded that stocking rate and overgrazing are more influential drivers of floristic diversity than grazing management system *per se* (Heitschmidt *et al.* 1989; O'Reagain and Turner 1992; Ash and Stafford Smith 1996; Manley *et al.* 1997; Provenza *et al.* 2003; Vermeire *et al.* 2008). However, differences became apparent when SRG was separated into seasonal and multi-paddock systems in meta-analyses. Multi-paddock SRG had significantly lower plant richness than CG, and seasonal SRG had greater plant diversity than CG. Lower richness associated with multi-paddock SRG may reflect greater heterogeneity under CG as animals heavily graze some areas of the landscape and under-utilise other areas (patch-grazing), leading to structural and compositional heterogeneity within large grazing units (Stuth 1991; Bailey *et al.* 1996; Adler *et al.* 2001; Fuhlendorf and Engle 2001; Teague *et al.* 2004). Compared to UG treatments, SRG generally favoured, or did not significantly alter, plant, mammal and bird species richness and diversity, and meta-analyses indicated no significant differences in plant richness or diversity between SRG and UG. Previous reviews have found that grazing has a negative effect on these biodiversity outcomes in arid rangelands (Holechek *et al.* 1999; Eldridge *et al.* 2016), but we found only invertebrate species richness and diversity to be more frequently greater under UG than under SRG systems.

Although a large proportion of studies reported no difference in plant species richness and diversity between SRG and CG or UG areas, the overwhelming majority of these studies reported a difference in plant community composition due to grazing treatment. Clearly, plant species richness and diversity do not effectively capture floristic changes. However, it is challenging to attribute plant community compositional change to a positive or negative response, given the dependence of the interpretation on the local and landscape context of the study and the goals of the grazing system (e.g. whether for ecological or animal production outcomes) and of the research. Winfree *et al.* (2011) and Seefeldt and McCoy (2003) suggested that species composition is a more informative indicator of community change than traditional measures of species richness and diversity. We also could not distinguish between native and exotic or invasive species when analysing plant species richness and diversity as this information was rarely reported in studies. Therefore, increased plant richness or diversity in the grazing studies reviewed here may not always reflect improved biodiversity conservation outcomes if the increase in richness and diversity were driven by introduced or invasive species.

No studies included in this review recorded measures of plant functional diversity. A greater emphasis on compositional changes and changes in functional diversity would be beneficial in future research to ensure that all relevant effects of grazing treatments are captured. Of the studies that reported ecological outcomes, most reported plant community impacts whereas impacts on mammals, birds, reptiles or invertebrates were rarely reported. This is a significant knowledge gap given that animal responses often differ to those of plants (Kruess and Tscharntke 2002; Zhu *et al.* 2012; Chillo *et al.* 2015; van Klink *et al.* 2015). Animals respond directly to trampling and vegetation changes induced by herbivory such as reduced herbage mass or unintentional consumption, but are also impacted by grazing-induced changes in plant diversity, microclimate and vegetation structure (van Klink *et al.* 2015).

While ground cover and biomass do not relate specifically to biodiversity conservation, they are important variables for maintaining ecological processes such as fauna and soil protection and nutrient cycling (Gardiner and Reid 2010). The trend analysis revealed more favourable responses in both these variables under SRG than CG, indicating more sustainable ecological outcomes under SRG. However, ground cover and biomass were reduced more often than not under SRG than in UG areas.

2.6.2 Effect of strategic-rest grazing on animal production variables

Continuous grazing is thought to increase individual animal weight gain compared to SRG as livestock can selectively graze preferred plants (Ellison 1960; Joseph *et al.* 2002; Briske *et al.* 2008). In contrast, under SRG, the smaller paddocks and higher animal densities necessary to maintain equivalent stocking rates lead to greater herbage utilisation and greater animal production per unit area (Hart *et al.* 1989; Joseph *et al.* 2002; Norton *et al.* 2013; Williamson *et al.* 2016). Although the trend analysis supported these generalisations, our meta-analyses did not. We found no significant difference between SRG and CG in either weight gain or animal production per unit area. However, as the amount of rest relative to grazing time increased, greater weight gain and animal production per unit area was observed under SRG than CG. Teague *et al.* (2015) demonstrated in a simulation study that increasing the number of paddocks in a rotational grazing system (and therefore the amount of rest between grazing events) can increase ecological condition and profitability. Many studies have shown that an adaptive grazing strategy according to seasonal and forage conditions, incorporating a short grazing period followed by a long period of rest, is optimal in grazing systems to achieve resource conservation and economic results (Jakoby *et al.* 2014, 2015; Teague *et al.* 2013, 2015). Adaptive management is difficult to incorporate in meta-analyses when defined by stocking rate and rest:graze times. However, greater exploration of this in future studies would be beneficial.

2.6.3 Other important influences of meta-analyses

Although we were able to incorporate the rest:graze ratio, climate zone and geographical region in our analyses, many other factors such as finer differences in SRG (e.g. high-density short-duration grazing, long rotations, deferred grazing, seasonal grazing or resting), management cues, sampling methods, and the length of time that the grazing treatment has been imposed prior to research being undertaken, should be included in future analyses. The complexity of the differences in grazing management between studies and the variability in the way data are reported may mask the effects of particular grazing management practices and confound results (Briske *et al.* 2008). For example, a large proportion of studies did not compare grazing regimes at equivalent stocking rates, or reported stocking rate differences poorly. This was particularly true of studies that focused on ecological outcomes.

Data quality limited the number of studies that could be included in meta-analyses. Many studies could not be included as measures of variation or mean response values were not reported. It was challenging to extract important grazing information from studies such as the timing of rest periods relative to periods of key pasture growth. The timing of rest is known to be important (Jones 1933; Lodge and Whalley 1985). The rest:graze ratio and type of SRG system (i.e. multi-paddock or seasonal) were able to be included in our meta-analyses. However, the large amount of residual heterogeneity in the meta-analyses indicated that other unexplained factors were likely influencing outcomes. This unexplained variation is unsurprising in the context of complex agro-ecological systems influenced by environmental, social and economic factors that are difficult to replicate or control for in field experiments (Briske *et al.* 2011; Teague *et al.* 2013). While this warrants further investigation, it may be difficult to adequately account for such complexity (Heady 1961; Briske *et al.* 2008).

2.6.4 Integration of ecological and production research

Our review highlighted a lack of integration in studies, with ecological and animal production variables rarely examined simultaneously in response to different grazing treatments. Most studies that measured the response of ecological response variables did not report animal production variables, and vice versa. Of the papers that were classified as integrated, many were so because they recorded ground cover, biomass or plant composition rather than animal production and biodiversity outcomes. Variables that were more relevant to biodiversity outcomes such as species richness and diversity measures, or a particular species or community of organisms for biodiversity conservation purposes, were rarely reported simultaneously with animal production outcomes such as animal weight gain and animal production per unit area. The small number of studies that explicitly considered both ecological (especially biodiversity) and production outcomes, exposed the lack of scientific understanding of the synergies and trade-offs in managing for production and biodiversity conservation in livestock grazing systems.

2.6.5 Conclusion

This systematic review and meta-analyses revealed predominantly neutral responses of ecological and animal production outcomes under SRG management compared to CG and UG areas. However, when differences occurred, they were more often in favour of SRG than CG or UG areas. In the meta-analyses, ecological and animal production responses varied with type of grazing system, climate zone and the length of rest relative to grazing time. CG had greater plant species richness than multi-paddock SRG systems, while plant diversity was greater under seasonal SRG than CG. However, greater exploration of the species compositional changes is necessary to understand whether the increased richness and diversity is ecologically beneficial or agriculturally beneficial, which in turn will depend on the objectives of management. The length of rest relative to graze period in SRG systems was positively related to weight gain and animal production per unit area, and negatively related to plant richness. Our review highlighted a lack of integration between ecological and animal production outcomes in the literature comparing SRG with CG and UG areas. Addressing this knowledge gap is important to further the integration of conservation and production outcomes in agricultural landscapes. An understanding of the ecological and production trade-offs associated with different grazing management strategies is essential to make informed decisions about best-management practices and for the sustainable management of the world's grazing lands for joint production and ecological outcomes.

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STATEMENT OF AUTHORS' CONTRIBUTION

(To appear at the end of each thesis chapter submitted as an article/paper)

We, the PhD candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated in the Statement of Originality.

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STATEMENT OF ORIGINALITY

(To appear at the end of each thesis chapter submitted as an article/paper)

We, the PhD candidate and the candidate's Principal Supervisor, certify that the following text, figures and diagrams are the candidate's original work.

Type of work	Page number/s
Figure 2.1	41
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Table 2.1	32
Table 2.2	39

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Chapter 3. Grazing management and biodiversity conservation in semi-arid rangelands

3.1 Abstract

Grazing management that promotes both biodiversity and production outcomes has the potential to improve conservation across a broad scale and complement the current protected area system. This study explored the potential to integrate commercial livestock grazing and conservation in a semi-arid rangeland system in south-eastern Australia. Understorey floristic composition and biodiversity measures at different scales were compared in relation to three grazing management treatments: alternative grazing management where livestock are frequently rotated and paddocks rested, traditional grazing management where paddocks are continuously grazed for the majority of the year, and protected areas managed for conservation where domestic livestock are excluded. Season of sampling, recent and long-term rainfall, soil characteristics and the spatial location of sites were the dominant drivers of species composition. The effect of grazing treatment on composition was relatively minor. Areas managed for conservation and under alternative grazing management had greater floristic richness and diversity than traditionally grazed areas, though results varied with season and soil type, particularly at the smallest scale. Results suggest that under certain seasons and soil types in semi-arid Australia, alternative grazing management which incorporates long periods of rest can achieve floristic conservation outcomes similar to areas where livestock have been removed, exceeding that of traditionally grazed areas. More research is necessary to determine whether alternative

grazing management is a viable option for conserving threatened species and communities in semi-arid rangelands.

3.2 Key words

Composition, continuous grazing, diversity, functional diversity, rotational grazing, soil type, spatial scale

3.3 Introduction

Pastoralism is recognised as an important factor in global rangeland degradation and domestic livestock grazing is generally considered to be incompatible with biodiversity conservation (Asner *et al.* 2004; MA 2005; Lunt *et al.* 2007a; Eldridge *et al.* 2016). While removing livestock and setting aside land for conservation is important for maintaining local biodiversity (Margules and Pressey 2000), such reserves are generally inadequate in representing regional biodiversity (Rodrigues *et al.* 2004; Watson *et al.* 2014). Despite the increasing extent of protected areas throughout the world, biodiversity declines continue (Butchart *et al.* 2010). Off-reserve conservation can play an important role in conserving biodiversity at a regional scale and complement the reserve system by facilitating connectivity between reserves and accommodating changes in species distributions (Fischer *et al.* 2006; Lindenmayer *et al.* 2010). As the need for global food security increases, livestock grazing management that promotes biodiversity values may offer an alternative approach for large-scale conservation while achieving dual production and ecological outcomes (Dorrough *et al.* 2004).

Grazing by domestic livestock is thought to have greater negative impacts on biodiversity in xeric environments with a short history of ungulate grazing, such as the Australian rangelands, than under more mesic conditions with a longer history of grazing (Milchunas *et al.* 1988). This is due to the lower regrowth potential of arid areas, species evolution in the absence of high or continuous grazing pressures, and greater susceptibility to soil erosion (Milchunas *et al.* 1988; Cingolani *et al.* 2005). As a result, grazing in more arid regions has been associated with declines in floristic biodiversity and changes in community structure and composition (Milchunas *et al.* 1988; Pettit *et al.* 1995; Bakker *et al.* 2006; Díaz *et al.* 2007; Carmona *et al.* 2012). These changes occur either directly as a result of herbivory, or indirectly through habitat modification (including soil disturbance and degradation, altered fire regimes, nutrient addition and seed dispersal) and the negative feedback effects associated with both (Hulme 1996; Olff and Ritchie 1998; Eldridge *et al.* 2016). However, there is evidence to suggest that certain types of grazing management can maintain or improve plant biodiversity and species composition, and enhance biodiversity in agricultural landscapes (Dorrough *et al.* 2004). Moreover, recent studies have found little impact of grazing on plant biodiversity measures in arid and semi-arid environments (Dostálek and Frantík 2008; Lewis *et al.* 2008; Fensham *et al.* 2011; Fensham *et al.* 2014; Oñatibia and Aguiar 2016)

Alternative grazing regimes, where livestock are frequently rotated and paddocks rested, have the potential to achieve both ecological and socio-economic outcomes (Teague *et al.* 2015). However, there is conflicting evidence as to the value of rotational grazing regimes over continuous grazing (Briske *et al.* 2008; Teague *et al.* 2013). Listed ecological benefits of rotational grazing systems include more even utilisation of pasture and less selective grazing (thereby avoiding overgrazing of patches and desirable species), maintenance of plant vigour, increased abundance and cover of perennial species, achieving desired species composition, increased plant biodiversity, and greater control over grazing pressure (Norton 1998a; Teague *et al.* 2008, 2013). However, few studies in arid and semi-arid regions have compared the effects of alternative grazing strategies on plant biodiversity and species composition with areas managed for conservation.

While 12% of Australian rangelands are held under formal public conservation tenure (Bastin 2008), only 3% of the Mulga Lands and Darling Riverine Plains bioregions in western NSW is currently held in public reserve (OEH 2016). Therefore integration of conservation and production goals on the commercial grazing lands that comprise these two bioregions is essential for the conservation of regional biodiversity. The semi-arid rangelands of western NSW have undergone significant degradation since domestic livestock were introduced in the mid - 1800s as a result of overstocking and poor management (Anon 1901; McKeon 2004).

Previous research on the impact of alternative grazing strategies on plant biodiversity and community composition has often not accounted for differences in soil, vegetation, season, or scale, and few studies have explored the potential of alternative grazing strategies for biodiversity conservation in Australian rangelands. This study aimed to determine whether alternative grazing strategies can improve biodiversity outcomes compared to traditional methods of commercial grazing and conservation management. Specifically, we compared understorey floristic species composition and biodiversity measures (richness, evenness, diversity, turnover, functional diversity) at three different scales between sites managed for nature conservation and those under traditional (continuous) and alternative (rotational) grazing management strategies on contrasting soil types (clay versus sand).

3.4 Methods

Thirteen grazing contrasts were sampled in six property clusters (groups of sites utilising adjacent properties) throughout the Mulga Lands and Darling Riverine Plains bioregions of western NSW (IBRA7 2012; Figure 3.1) on heavy clay (‘clay’, n = 7) and sandy-loam (‘sand’, n = 6) soils. Average rainfall declined from east (400 mm) to west (275 mm) across the study region. Rainfall in the preceding 3, 6 and 12 months to sampling was variable and showed no clear geographical trend among clusters (Figure 3.2).

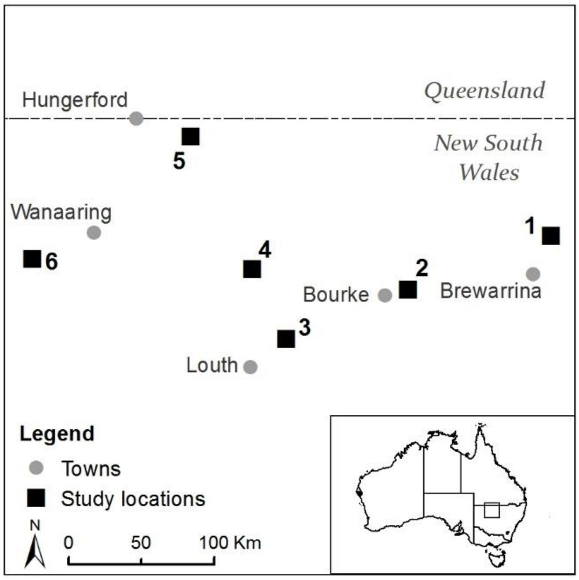

Figure 3.1. Location of property clusters (numbered) in western New South Wales, Australia.

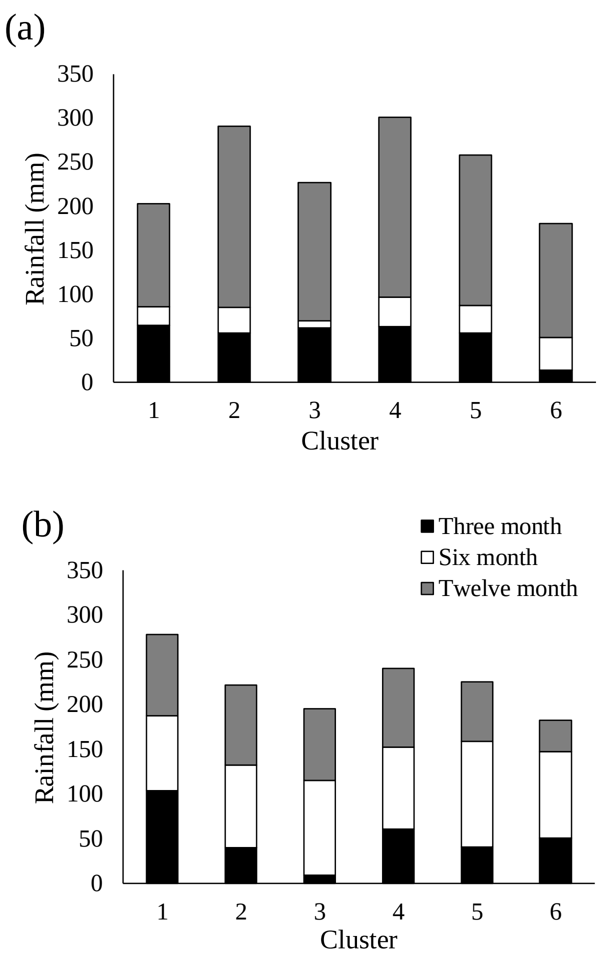

Figure 3.2. Preceding 3, 6 and 12 month rainfall (cumulative) for the site clusters sampled in (a) spring 2014 and (b) autumn 2015.

Three grazing treatments were studied: (1) areas currently managed for conservation (CON) where domestic livestock had been excluded, but feral and native herbivores remained; (2) alternative grazing management (AGM) on commercially grazed properties where paddocks are strategically rested for periods of time throughout the year and paddocks are grazed at an appropriate stocking rate to maintain ground cover, pasture biomass and biodiversity; and (3) traditional grazing management (TGM) on commercial properties, where paddocks are continuously grazed for most of the year. AGM treatments did not comply with strict rest–graze times; rather, stocking rates and grazing regimes were managed adaptively according to seasonal conditions and management constraints. As such, they were representative of current AGM management in Western NSW. Although very variable, average annual stocking rates were 4.5 DSE/ha and 3.5 DSE/ha for AGM and TGM treatments, respectively. Properties selected for this study had been undertaking their current grazing management for a minimum of 5 years prior to sampling. Information on current and historical management, grazing strategies and stocking rates was obtained for each site from the land managers (see Appendix, Table A3.1 and Table A3.2).

3.4.1 Sampling design

Each grazing contrast compared at least two different grazing treatments in adjacent paddocks in the same land system and vegetation community (see Appendix, Table A3.3). Three 100 × 100-m plots were selected over a distance of approximately 1 km within each grazing treatment (site), with different grazing treatments located on adjacent properties under different management. Understorey floristic measurements were recorded in twenty-five 1 × 1-m quadrats arranged systematically at 25-m intervals within plots (Figure 3.3).

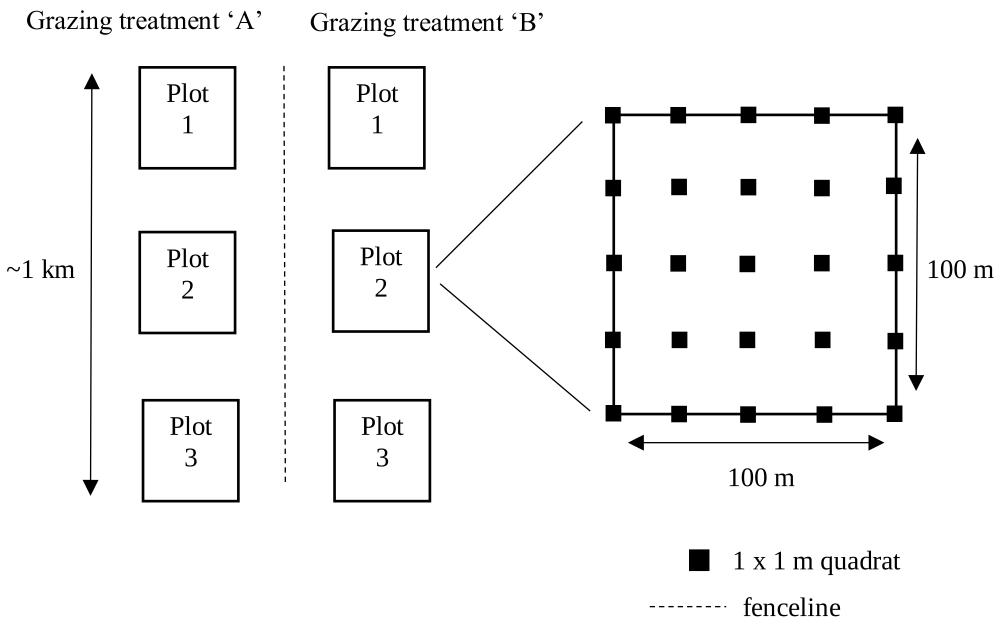

Figure 3.3. Diagram of typical paired-site plot and quadrat layout (not to scale).

3.4.2 Biodiversity measurements

Floristic surveys were undertaken in spring 2014 (20 September – 6 November) and autumn 2015 (21 April – 30 May). In each quadrat, all understorey plant species (<3 m tall at maturity) were identified to the lowest taxonomic level possible. Dung (number of pellets of goat and sheep combined, kangaroos and rabbits, and number of cattle pats) was also counted in each quadrat, to provide a measure of grazing intensity. Understorey floristic species richness, Pielou's evenness index (J'), Shannon–Wiener diversity index (H') and species turnover (pattern diversity) were calculated at two scales: site (75 quadrats pooled) and plot (25 quadrats pooled). Species richness was also calculated for each quadrat.

Functional trait information for each understorey plant species was obtained from online sources and literature (Table 3.1). Trait information included life history, origin, functional group, height, seed length, leaf area index and palatability. Species richness of each categorical functional trait was calculated for each quadrat, plot and site. Community weighted mean (CWM) scores were calculated for each trait at plot and site scale using species frequency data. Functional richness (FRic), functional evenness (FEve), functional dispersion (FDis), and functional diversity (Rao's quadratic entropy, RaoQ) were calculated from life history, origin, height, seed length, leaf area index and palatability information using species frequency in plots for each season. An index of species rarity within this study was also determined, calculated as the proportion of plots a species was absent from. Rarity was also analysed using CWM's.

Table 3.1. Description of functional trait data measured and reported in results

Variable	Description	Analysed as
Life history[1]	Category: total annual or total perennial	Richness and CWM frequency[2]
Origin[1]	Category: native or exotic	Richness and CWM frequency
Functional group[1]	Category: annual grass, annual forb, perennial grass, perennial forb	Richness and CWM frequency
Height[1]	Plant height at maturity	CWM frequency
Seed length[1]	Length of seed	CWM frequency
Leaf area index[1]	Leaf width × height	CWM frequency
Palatability[4]	Palatability to livestock. Category: unpalatable – not eaten unless little other food available; moderately palatable – eaten, but not readily or when mature; palatable – palatable and readily eaten[4]	Richness and CWM frequency
Functional richness[5]	Represents the range or variability of traits in a community (Villéger et al. 2008)	CWM frequency
Functional evenness[5]	Relates to the evenness of the abundance distribution of traits (Villéger et al. 2008)	CWM frequency
Functional dispersion[5]	The distance of a species in functional trait space to the centroid of all other species, weighted by abundance (Laliberté and Legendre 2010)	CWM frequency
Functional diversity[5]	RaoQ, measures the functional trait difference between species and weights by species abundance (Botta-Dukát 2005)	CWM frequency

[1] Information from PlantNet (The NSW Plant Information Network System), Royal Botanic Gardens and Domain Trust, Sydney. http://plantnet.rbgsyd.nsw.gov.au [15 October 2015]
[2] Community weighted means (CWM) calculated using the FD package in R (Laliberté *et al.* 2010) from species frequency data
[3] Forbs refer to all understorey species except grasses
[4] Palatability data from Cunningham *et al.* (2011)
[5] Calculated using the FD package in R (Laliberté *et al.* 2010) from the life history, origin, functional group, height, seed length, leaf area index, palatability and rarity index traits

3.4.3 Soil samples

Three soil cores (50 mm deep and 75 mm width) were taken in each plot in autumn 2015. Soil was dried at 40°C, and a subsample dried at 105°C for bulk density calculation. The three cores from each plot were bulked by equal volume, and sieved to 2 mm for pH and electrical conductivity (EC) analysis, and ground to 0.2 mm for

LECO analyses. Soil pH and EC were analysed using methods outlined by Rayment and Higginson (1992); pH was measured using a pH meter (Model 901-CP), and EC with an EC meter (Model labCHEM) in a 1:5 soil:water solution. Total organic carbon (TOC) and total organic nitrogen (TON) were measured using a LECO TruSpec Series Carbon and Nitrogen Analyser. Soils with pH > 7 were tested for carbonates by adding hydrochloric acid to a sample of the soil and looking for effervescence. Soils that contained carbonates were treated with sulphurous acid before LECO analysis.

3.4.4 Statistical analysis

Multivariate analysis was used to determine the variance in species composition attributable to all sources of measured variation (spatial, environmental and management) across plots and univariate analyses were used to investigate relationships between grazing treatments and biodiversity measures.

Detrended Canonical Analysis (DCA) was performed on species frequency data at the plot scale using CANOCO 5 (Ter Braak and Šmilauer 2012). The length of gradient produced from this analysis then determined whether a unimodal (gradient <3) or linear model (>4) was used for direct ordination analyses. Canonical Correspondence Analysis (CCA) was selected as the appropriate method. Monte-Carlo permutation tests (1000 permutations) and forward selection was used to determine inclusion of environmental variables. Environmental variables included average rainfall, preceding 3, 6 and 12 month rainfall, bulk density, pH, EC, TOC, TON, average months rested per year, average stocking rate when grazed, dung counts, and spatial variables (X, Y, X^2, XY, Y^2, X3, X^2Y, XY^2, Y^3) as outlined in Borcard *et al.* (1992). Factors soil (sand and clay), season (spring and autumn) and grazing treatment (CON, AGM and TGM) were also included as environmental variables. Holm's corrected *P*–values were used to adjust for type-1 error inflation; only variables with Holm's $P \leq 0.05$ were included in the model. Variance partitioning was performed on the CCA in three categories: (1) rainfall and soil variables (preceding 3, 6 and 12 month rainfall, annual rainfall, soil type, soil bulk density, pH, EC, TOC and TON); (2) grazing variables (treatment, average rest time, average stocking rate and dung counts), and (3) spatial variables.

Linear mixed-effects (LME) models using the lme4 package in R (Bates *et al.* 2016; R core team 2017) were used to analyse the effect of soil, season and grazing treatment on response variables at site, plot and quadrat scales. Differences between soil types or seasons for total species richness, diversity, evenness and turnover at the site scale were determined from LME models using soil type (sand/clay) or season (spring/autumn) as a fixed effect and cluster and site within cluster as nested random effects. To determine differences between grazing treatments at site scale, LME models included season of survey, soil type, and grazing treatment (CON/AGM/TGM) as fixed effects and cluster and site as nested random effects. LME models for plot and quadrat-scale data followed a similar structure to those at site scale, except that cluster, site within cluster, and plot within site, were included as nested random effects. All fixed effects and their interactions were initially included in the models, and non-significant interaction terms were removed one at a time, commencing with the highest-order interactions, until only significant interaction terms or main effects remained ($P \leq 0.05$; Ripley *et al.* 2013). Response variables were transformed as necessary to meet the assumptions of models. Predicted means were calculated and Tukey's HSD pairwise comparisons made using the lsmeans function in the lsmeans package (Lenth and Hervé 2013) to determine significant differences between factors ($P \leq 0.05$). Differences in dung counts between grazing treatments were also analysed using LME models at the plot scale. Pearson's correlations, stratified by soil type and season, were computed between species richness, evenness, diversity and turnover and explanatory variables using the Hmisc package in R (Harrell 2016) to help interpret the impact of explanatory variables on plant biodiversity measures.

3.5 Results

In total, 260 vascular understorey plant species (including subspecies) in 43 plant families were detected across all sites (see Appendix, Table A3.3). Of the 260 species recorded, 50% of species were perennials, 33% annuals, and 17% uncertain (plants lacking identifying features and seedlings too small to identify to species level, mostly due to recent rainfall at several sites prior to the autumn survey). Of the identified plant species, 52% occurred in just three families: Poaceae (19% of species), Chenopodiaceae (19%) and Asteraceae (14%). The majority of species identified were perennial forbs (48%), followed by annual forbs (33%), perennial grasses (13%) and annual grasses (6%). Fourteen species (5%) were exotic. In spring 2014, 209 species were recorded across all sites compared to only 162 species in autumn 2015, with 111 species present in both seasons. Conservation sites had higher mean counts of kangaroo dung than grazed treatments whereas grazed sites had higher mean counts of livestock dung than conservation sites (Table 3.2).

Table 3.2. Dung counts (predicted means, m^{-2}) in each grazing treatment. Different letters indicate significantly different means within rows ($P \leq 0.05$). CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Animal	**CON**	**AGM**	**TGM**	***P*-value**
Kangaroo (pellets/m^2)				<0.001†
Sand	3.40a	2.03a	2.91a	
Clay	4.60a	3.30b	1.46c	
Sheep/goat (pellets/m^2)‡				0.004†
Sand	5.47b	11.38a	6.61ab	
Clay	3.06b	5.49ab	7.73a	
Cow (pats/m^2)‡				0.002†
Sand	< 0.001b	0.01ab	0.03a	
Clay	< 0.001b	0.08a	0.07a	

† Significant soil:grazing treatment interaction

‡ Square-root transformation used in LME models; data in table have been back-transformed

3.5.1 Composition

CCA of species frequency within plots revealed a clear distinction in plant community composition between soil types and site clusters, and to a smaller extent, seasons (Figure 3.4). Some differences between grazing treatments within clusters were evident, but these patterns were not consistent among clusters or seasons. Grazing treatment was not significant; rather, soil type explained the greatest amount of variation, followed by spatial variables, rainfall variables, soil bulk density, season, soil pH, soil nitrogen, soil EC, sheep/ goat dung counts, average stocking rate under grazing, average months rested per year, and kangaroo dung. Combined, this set of variables accounted for 45% of the total variation. Variance partitioning revealed rainfall and soil variables accounted for 11.1% of all variation, closely followed by spatial variables, 9.5% (Figure 3.5). Grazing variables accounted for 1.4% of the floristic variation.

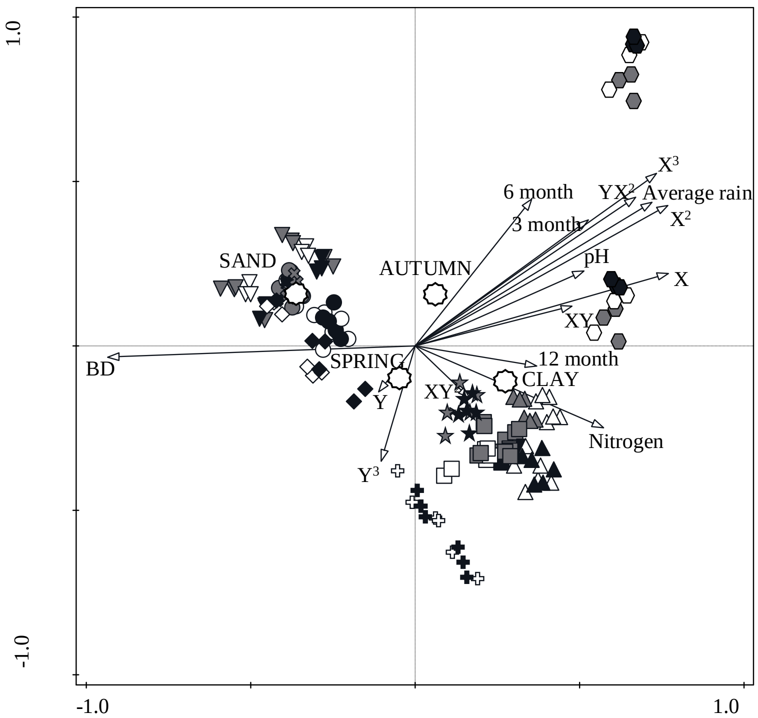

Figure 3.4. Constrained ordination of species frequency for both spring and autumn. White = conservation, grey = alternative grazing, black = traditional grazing.⬢ = cluster 1, clay; ■ = cluster 2, clay; ▲ = cluster 3, clay; ★ = cluster 4, clay; ✚ = cluster 5, clay; ● = cluster 3, sand; ✖ = cluster 4, sand; ◆ = cluster 5, sand; ▼ = cluster 6, sand. ✹ = factors. 6 month = previous 6 month rainfall; 3 month = previous 3 month rainfall; 12 month = previous 12 month rainfall; Average rain = average rainfall; BD = bulk density; Nitrogen = soil total organic nitrogen; see Borcard et al. (1992) for information on spatial variables. Significant variables explaining less than one percent of explained variation are not shown on graph (EC, average stocking rate under grazing, average number of months rested per year, sheep/ goat dung, kangaroo dung, Y^2)

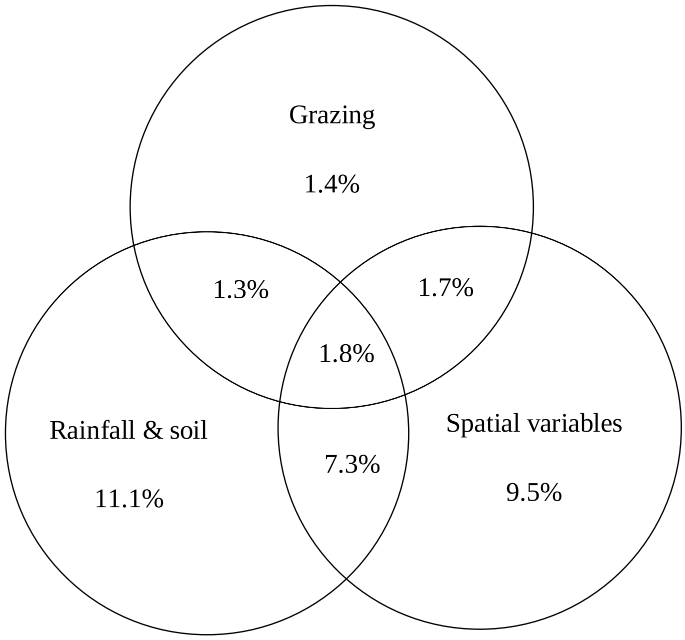

Figure 3.5. Variance partitioning of grazing, rainfall and soil, and spatial variables on species frequency in plots

3.5.2 Site-scale responses to soil type, season and grazing treatment

Total species richness and diversity of the understorey vegetation was significantly greater in sand sites than clay sites (Table 3.3), and species richness, diversity and turnover were greater in spring than autumn (Table 3.4). In spring, CON and AGM sites had greater species richness than TGM sites, and AGM sites had greater diversity than TGM sites (Table 3.5). In terms of differences in functional composition, AGM sites had greater richness of total perennial species, perennial forb and native species in spring, and greater richness of moderately palatable species richness in both seasons, than TGM sites. CON sites had greater native species richness and a greater frequency of rare species, than TGM sites. There were no significant differences in functional traits between grazing treatments when analysed as species frequency (CWM) or functional diversity (see Appendix, Table A3.4).

3.5.3 Plot-scale responses to soil type, season and grazing treatment

At the plot scale, species richness, diversity, evenness and turnover were significantly greater on sand than clay soils (Table 3.3), and richness and diversity were significantly greater in spring than autumn (Table 3.4). In spring, CON and AGM plots had significantly greater species richness than TGM plots, and CON plots had significantly greater diversity than TGM plots (Table 3.6). AGM plots had greater richness of total perennial species, native, and unpalatable species in spring, and of moderately palatable species in both seasons, than TGM plots. CON plots had greater richness of total annual species, annual and perennial forbs, native, unpalatable and palatable species than TGM plots in spring, as well as a greater frequency of rare species than TGM plots. There were no significant differences in functional traits between grazing treatments when analysed as species frequency (CWM) or functional diversity (see Appendix, Table A3.5).

3.5.4 Quadrat-scale responses to soil type, season and grazing treatment

At the quadrat scale, there was no difference in species richness between sand and clay soils (Table 3.3). However, there were significantly more species in spring than autumn (Table 3.4). Species richness was significantly greater under CON and AGM than TGM on clay soils in spring at quadrat scale (Table 3.7). Richness was also greater under CON than TGM on sand soils in spring, but this trend was reversed in autumn. AGM quadrats had greater richness of total annual species and annual forb species in spring on sand and in autumn on clay soils, and also of annual grass species in spring, compared to TGM quadrats. Total perennial species richness and perennial forb species richness were greater under AGM in spring on clay soils, and perennial forb species richness was also greater in autumn on clay soils under AGM than TGM.

CON management had greater richness of total annual species and annual forb and grass species in spring on both soil types than TGM, and greater richness of perennial forbs in spring on clay soils and of perennial grasses in autumn on clay soils than TGM. TGM quadrats had greater richness of perennial forbs on sand in autumn and of perennial grasses on clay soils in spring than CON management, and greater richness of perennial grass species in autumn than AGM. In terms of differences between CON and AGM, CON quadrats had greater richness of total annual species and annual forb species in spring on clay soils, and perennial grass species in autumn on clay soils, but AGM had a greater richness of total perennial species, and of perennial forb and grass species in spring on clay soils, and of perennial forb species in autumn on both soil types. Compared to TGM, native and exotic species richness was greater under AGM in spring on clay soils and CON quadrats had greater richness of natives in spring on sand and exotics in spring on clay soils. Compared to TGM, AGM had a greater richness of unpalatable species in both seasons and on both soil types, and a greater richness of moderately palatable species in spring and on clay soils. CON quadrats had a greater richness of unpalatable species in spring and on clay soils, moderately palatable species on clay soils, and palatable species in spring on sand soils than TGM quadrats. AGM quadrats also had a greater richness of moderately palatable species than CON quadrats.

Table 3.3. Predicted means for understorey floristic species richness, diversity, evenness and turnover at different scales for sand and clay soils

Variable	Scale	Sand	Clay	*P*-value
Richness	Site	39.12	28.16	0.007
Richness	Plot	25.51	17.68	0.015
Richness	Quadrat	5.41	5.19	0.826
Diversity[1]	Site	39.12	2.65	0.022
Diversity[1]	Plot	2.80	2.34	0.007
Evenness[2]	Site	0.83	0.81	0.183
Evenness[3]	Plot	0.87	0.85	0.043
Turnover	Site	0.75	0.68	0.618
Turnover	Plot	0.48	0.35	< 0.001

[1] Square-root transformation used in LME model, data presented have been back-transformed
[2] Transformed to the power of 6; data presented back-transformed
[3] Transformed to the power of 8; data presented have been back-transformed

Table 3.4. Predicted means for understorey floristic species richness, diversity, evenness and turnover at different scales, in spring 2014 and autumn 2015

Variable	Scale	Spring	Autumn	*P*-value
Richness	Site	38.63	27.82	< 0.001
Richness	Plot	24.50	17.96	< 0.001
Richness	Quadrat	6.23	4.39	< 0.001
Diversity[1]	Site	3.00	2.66	< 0.001
Diversity[1]	Plot	2.71	2.40	< 0.001
Evenness[2]	Site	0.83	0.82	0.224
Evenness[3]	Plot	0.86	0.85	0.344
Turnover	Site	1.06	0.37	< 0.001
Turnover	Plot	0.41	0.41	0.654

[1] Square-root transformation used in LME model; data presented have been back-transformed
[2] Transformed to the power of 6; data presented back-transformed
[3] Transformed to the power of 8; data presented back-transform

Table 3.5. Predicted means for site scale response variables for grazing treatments. Different superscript letters indicate significant differences within rows ($P \leq 0.05$). AF = annual forb, AG = annual grass, PF = perennial forb, PG = perennial grass, CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Response variable	**CON**	**AGM**	**TGM**	***P*-value**
Total richness				<0.001†
Spring	42.54^{a}	42.17^{a}	32.51^{b}	
Autumn	25.76^{a}	29.17^{a}	29.85^{a}	
Evenness‡	0.34^{a}	0.31^{a}	0.28^{a}	0.207
Shannon–Wiener Diversity§				0.010†
Spring	3.13^{ab}	3.13^{a}	2.76^{b}	
Autumn	2.65^{a}	2.67^{a}	2.72^{a}	
Turnover	0.637^{a}	0.81^{a}	0.70^{a}	0.318
Total annual richness				0.027†
Spring	17.39^{a}	15.16^{a}	12.20^{a}	
Autumn	6.17^{a}	7.38^{a}	7.42^{a}	
AF richness				0.026†
Spring	13.78^{a}	12.64^{a}	9.11^{a}	
Autumn	4.78^{a}	6.08^{a}	6.22^{a}	
AG richness	2.47^{a}	1.87^{a}	1.95^{a}	0.185
Total perennial richness				0.007†
Spring	23.75^{ab}	26.31^{a}	20.03^{b}	
Autumn	17.52^{a}	19.42^{a}	19.58^{a}	
PF richness				0.017†
Spring	17.86^{ab}	19.08^{a}	13.59^{b}	
Autumn	12.98^{a}	14.52^{a}	14.15^{a}	
PG richness	5.21^{a}	6.06^{a}	5.92^{a}	0.183
Native richness				0.002†
Spring	38.17^{a}	38.71^{a}	29.70^{b}	
Autumn	22.06^{a}	24.49^{a}	24.03^{a}	
Exotic richness	1.71^{a}	1.65^{a}	1.53^{a}	0.857
Unpalatable richness	9.12^{a}	9.69^{a}	7.74^{a}	0.079
Moderately palatable richness	6.39^{ab}	7.07^{a}	5.35^{b}	0.027
Palatable richness	14.36^{a}	14.30^{a}	13.80^{a}	0.883
Rarity index (CWM)	71.67^{a}	70.38^{ab}	69.13^{b}	0.009

† Significant season:grazing treatment interaction
‡ Transformed to power of 6; data presented back-transformed
§ Transformed to power of 2; data presented back-transformed

Table 3.6. Predicted means at the plot scale for response variables. Different superscript letters indicate significant differences within rows ($P \leq 0.05$). AF = annual forb, AG = annual grass, PF = perennial forb, PG = perennial grass, CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Response variable	**CON**	**AGM**	**TGM**	***P*-value**
Total richness				<0.001†
Spring	26.98^{a}	26.48^{a}	21.15^{b}	
Autumn	17.09^{a}	18.52^{a}	19.37^{a}	
Evenness‡	0.31^{a}	0.30^{a}	0.28^{a}	0.506
Shannon–Wiener Diversity§				0.006†
Spring	2.81^{a}	2.80^{ab}	2.56^{b}	
Autumn	2.38^{a}	2.41^{a}	2.46^{a}	
Turnover	0.42^{a}	0.42^{a}	0.41^{a}	0.854
Total annual richness				0.004†
Spring	11.22^{a}	9.83^{ab}	8.01^{b}	
Autumn	4.32^{a}	4.67^{a}	4.81^{a}	
AF richness				0.003†
Spring	8.45^{a}	7.71^{ab}	5.74^{b}	
Autumn	3.19^{a}	3.63^{a}	3.85^{a}	
AG richness	1.75^{a}	1.45^{a}	1.46^{a}	0.100
Total perennial richness				0.008†
Spring	15.27^{ab}	16.43^{a}	13.06^{b}	
Autumn	11.81^{a}	12.55^{a}	12.71^{a}	
PF richness*				0.008†
Spring	10.75^{a}	10.53^{ab}	7.57^{b}	
Autumn	7.25^{a}	7.77^{a}	8.14^{a}	
PG richness	3.86^{a}	4.29^{a}	4.24^{a}	0.196
Native richness				0.002†
Spring	24.71^{a}	24.38^{a}	19.70^{b}	
Autumn	14.90^{a}	15.86^{a}	15.55^{a}	
Exotic richness*	0.46^{a}	0.64^{a}	0.54^{a}	0.535
Unpalatable richness				0.003†
Spring	7.40^{a}	7.03^{a}	5.4^{b}	
Autumn	3.73^{a}	4.07^{a}	4.04^{a}	
Moderately palatable richness	3.93^{b}	4.66^{a}	3.53^{b}	0.001
Palatable richness				0.005†
Spring	12.30^{a}	11.41^{ab}	9.80^{b}	
Autumn	8.04^{a}	8.30^{a}	9.09^{a}	
Rarity index (CWM)	72.00^{a}	70.45^{ab}	69.61^{b}	0.005

† Significant season:grazing treatment interaction
‡ Transformed to power of 8; data presented have been back-transformed
§ Transformed to power of 2; data presented back-transformed
* Square-root transformed; data presented back-transformed

Table 3.7. Predicted means at the quadrat scale for response variables. Different superscript letters in rows indicate significant differences ($P \leq 0.05$). AF = annual forb, AG = annual grass, PF = perennial forb, PG = perennial grass, CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Response variable	**CON**	**AGM**	**TGM**	***P*-value**
Total richness				$0.003^{\dagger}$
Spring-Clay	6.53^{a}	6.70^{a}	5.71^{b}	
Spring-Sand	6.51^{a}	6.18^{ab}	5.60^{b}	
Autumn-Clay	4.62^{a}	4.59^{a}	4.24^{a}	
Autumn-Sand	3.78^{b}	4.44^{ab}	4.62^{a}	
Total annual richness‡				$0.008^{\dagger}$
Spring-Clay	1.95^{a}	1.28^{b}	1.01^{b}	
Spring-Sand	2.26^{a}	2.20^{a}	1.55^{b}	
Autumn-Clay	0.47^{ab}	0.62^{a}	0.33^{b}	
Autumn-Sand	0.69^{a}	0.85^{a}	0.80^{a}	
AF richness				$<0.001^{\dagger}$
Spring-Clay	2.14^{a}	1.58^{b}	1.51^{b}	
Spring-Sand	1.70^{a}	1.77^{a}	1.26^{b}	
Autumn-Clay	0.86^{ab}	1.13^{a}	0.76^{b}	
Autumn-Sand	0.52^{a}	0.72^{a}	0.60^{a}	
AG richness‡				$<0.001^{\S}$
Spring	0.35^{a}	0.30^{a}	0.22^{b}	
Autumn	0.12^{a}	0.14^{a}	0.11^{a}	
Total perennial richness				$<0.001^{\dagger}$
Spring-Clay	4.13^{b}	4.99^{a}	4.07^{b}	
Spring-Sand	3.49^{a}	3.24^{a}	3.28^{a}	
Autumn-Clay	3.22^{a}	3.09^{a}	2.85^{a}	
Autumn-Sand	2.50^{a}	2.85^{a}	2.98^{a}	
PF richness‡				$0.003^{\dagger}$
Spring-Clay	1.43^{b}	1.78^{a}	1.01^{c}	
Spring-Sand	1.83^{a}	1.55^{a}	1.48^{a}	
Autumn-Clay	0.91^{b}	1.14^{a}	0.86^{b}	
Autumn-Sand	1.09^{b}	1.53^{a}	1.50^{a}	
PG richness				$<0.001^{\dagger}$
Spring-Clay	1.91^{b}	2.38^{a}	2.42^{a}	
Spring-Sand	1.06^{a}	1.01^{a}	1.20^{a}	
Autumn-Clay	1.72^{a}	1.11^{c}	1.50^{b}	
Autumn-Sand	0.84^{a}	0.77^{a}	0.88^{a}	
Native richness				$<0.001^{\dagger}$
Spring-Clay	6.00^{ab}	6.24^{a}	5.54^{b}	
Spring-Sand	6.11^{a}	5.78^{ab}	5.21^{b}	
Autumn-Clay	3.78^{a}	3.74^{a}	3.26^{a}	
Autumn-Sand	3.42^{a}	3.95^{a}	4.00^{a}	
Exotic richness‡				$0.006^{\dagger}$
Spring-Clay	0.19^{a}	0.14^{a}	0.04^{b}	
Spring-Sand	0.002^{a}	0.005^{a}	0.003^{a}	
Autumn-Clay	0.10^{a}	0.13^{a}	0.07^{a}	
Autumn-Sand	$<0.001^{a}$	0.004^{a}	0.001^{a}	
Unpalatable richness				$<0.001^{\S}$
Spring	0.92^{a}	0.83^{a}	0.58^{b}	

Response variable	**CON**	**AGM**	**TGM**	***P*-value**
Autumn	0.27^{ab}	0.36^{a}	0.23^{b}	
Unpalatable richness				0.006*
Clay	0.38^{a}	0.33^{a}	0.19^{b}	
Sand	0.73^{ab}	0.87^{a}	0.66^{b}	
Moderately palatable richness				<0.001§
Spring	1.01^{b}	1.13^{a}	0.96^{b}	
Autumn	0.63^{a}	0.56^{a}	0.61^{a}	
Moderately palatable richness				0.006*
Clay	0.76^{a}	0.80^{a}	0.68^{b}	
Sand	0.88^{a}	0.89^{a}	0.90^{a}	
Palatable richness				<0.001†
Spring-Clay	3.91^{a}	4.10^{a}	3.86^{a}	
Spring-Sand	2.76^{a}	2.28^{ab}	2.13^{b}	
Autumn-Clay	2.84^{a}	2.81^{a}	2.62^{a}	
Autumn-Sand	1.83^{a}	2.13^{a}	2.26^{a}	

† Significant season:soil type:grazing treatment interaction
‡ Square-root transformed; data presented have been back-transformed
§ Significant season:grazing treatment interaction
*Significant soil type:grazing treatment interaction

3.5.5 Correlations with rainfall, soil, spatial and grazing variables

There were few significant correlations between plant biodiversity measures and grazing variables, and none were consistent across seasons and soil types (See Appendix, Table A3.6; A3.7). There were more significant correlations between rainfall, spatial and soil variables and the diversity measures, but again these depended on season and soil type. Previous 3, 6, and 12 month rainfall, soil organic carbon and soil organic nitrogen, were often positively correlated with richness, evenness and diversity. The effect (positive or negative) of average rainfall, spatial and soil variables depended on soil type and season.

3.6 Discussion

Plant species composition and biodiversity measures differed markedly between sand and clay sites, and to a lesser extent between spring 2014 and autumn 2015. Rainfall, soil type, season and geographical separation (>300 km between the further-most sites) were the main drivers of plant species composition and biodiversity measures. Grazing variables explained a lesser proportion of the variance in floristic composition and biodiversity measures, which is in agreement with other studies in similarly dry environments (Lunt *et al.* 2007b; Lewis *et al.* 2009b; Fensham *et al.* 2010, 2014, 2015). Sporadic and unpredictable rainfall events are the dominant drivers of the non-equilibrium community dynamics of arid and semi-arid rangeland systems, sometimes masking or reversing the effects of grazing and other disturbances (Ellis and Swift 1988; Westoby *et al.* 1989; Lewis *et al.* 2009a; Silcock and Fensham 2013). Despite this, differences in floristic biodiversity measures were detected between grazing treatments.

Areas under conservation and alternative grazing management generally had greater floristic diversity and richness than adjacent traditionally grazed areas in spring 2014, although at the quadrat scale differences were sometimes confined to one soil type. Measures of floristic richness rarely differed between conservation and alternative grazing management at the site and plot scale. At the quadrat scale, the few differences favoured alternative grazing management more often than conservation management. From a biodiversity conservation perspective, alternative grazing management registered similar levels of understorey floristic diversity and native species richness across a variety of scales to conservation areas where domestic livestock grazing had been removed, and was not associated with an increase in the proportion of exotic, annual or unpalatable species. These results suggest that commercial grazing management incorporating long periods of rest from grazing may be compatible with biodiversity conservation of ground-layer vegetation.

There are likely to be several explanations for the positive effect of conservation and alternative grazing management on understorey floristic richness and diversity compared with traditional grazing. Plant species in arid Australia have only been exposed to ungulate grazing for 150–200 years and have not evolved adaptations to cope with continuous grazing. They are therefore less resilient to livestock impacts and grazing-sensitive species are often removed from communities and replaced by grazing-tolerant species (Milchunas *et al.* 1988). Only at the smallest (quadrat) scale did traditional grazing management register greater floristic richness than conservation management (three instances) or alternative grazing management (one instance). This was probably a function of the scale of sampling, as at small scales grazing can increase richness by providing opportunities for species to establish and reducing competition from established plants. However, as grazing-sensitive species are lost from the species pool, richness declines at larger scales (Landsberg *et al.* 2002; Kohyani *et al.* 2008). This scale-related effect (increased richness at small scales under continuous grazing) is similar to that observed by Stohlgren *et al.* (1999), Adler *et al.* (2005), Kohyani *et al.* (2008), Landsberg *et al.* (2002) and Li *et al.* (2015), and describes homogenisation of landscapes.

Alternatively grazed sites and plots had greater floristic richness and diversity than traditional management in spring. Rotating livestock and resting pastures for most of the year allows recovery and persistence of species sensitive to continuous grazing and reduces overgrazing of patches (Norton 1998a; Teague *et al.* 2011). Moreover, some adaptations of desert species, such as rapid growth and short life cycles (ephemerality), small stature, tough leaves or a high salt content (less palatable species), can increase their resilience to short periods of grazing (Milchunas *et al.*

1988; Cingolani *et al.* 2005). Our findings are similar to recent studies in arid and semi-arid environments that found higher plant species richness where livestock grazing intensity was managed and interspersed with long periods of rest (Waters *et al.* 2016), and where greater negative effects of continuous grazing were found on plant species abundance than rotational grazing (Chillo *et al.* 2015).

Sustained grazing can favour annual, exotic, unpalatable and short-statured species at the expense of palatable, tall perennials and rare or grazing sensitive species (Noy-Meir *et al.* 1989; Belsky 1992; McIntyre and Lavorel 1994; McIntyre *et al.* 1999; Díaz *et al.* 2007), and such species may account for the positive richness and diversity responses to grazing sometimes observed. While more infrequent (rare) species were recorded under conservation management than traditional grazing management in this study, there was little other evidence to support this. At site and plot scales there was no difference in exotic species richness or species height between grazing treatments, and no consistent trends in palatability. At the quadrat scale, there was a greater number of exotic species, unpalatable and moderately palatable species under conservation and alternative grazing management than traditional management. The greater number of infrequent species recorded under conservation management highlights the importance of areas of grazing exclusion for the protection of rare, grazing sensitive species.

The lack of significant difference in richness, diversity, evenness or turnover between grazed and ungrazed areas observed at times is consistent with findings of previous studies (Adler *et al.* 2005; Lunt *et al.* 2007b; Lewis *et al.* 2008; Fensham *et al.* 2010, 2014). Lack of differences between grazing treatments in autumn were probably a reflection of the dry seasonal conditions experienced over the study period. Although rainfall at some sites in the preceding 3 months was similar to the spring survey, most recent rain had fallen just prior to the autumn survey, and germinating plants had not had a chance to establish. Lack of differences may also have been a reflection of the inherent variability and heterogeneous nature of the landscapes and vegetation communities in the study region. Grazing can also reduce the impacts of competitive exclusion by dominant species (Lunt *et al.* 2007a; Borer *et al.* 2014) and create opportunities for germination (Lunt *et al.* 2007a; Facelli and Springbett 2009) which can off-set the negative effects of grazing. In addition, grazing can increase the spatial heterogeneity of vegetation (turnover) via patch grazing, provided the spatial heterogeneity of the grazing is greater than that of the vegetation (Adler *et al.* 2001; De Bello *et al.* 2007). The findings of this study are consistent with the findings of Fensham *et al.* (2017) who reported that subtropical grasslands were tolerant of various disturbance regimes. While rotational grazing is expected to reduce patch grazing (Norton 1998a; Teague *et al.* 2011), the large paddock sizes and distances from water may still have resulted in a spatially variable grazing pattern, similar to the natural spatial heterogeneity in the vegetation. Lezama *et al.* (2014) also reported no significant effect of grazing on beta-diversity (turnover) in a low-productivity environment.

Heavy clay soils in western NSW are believed to be more resilient to the negative effects of grazing and disturbance than sandy soils due to higher fertility, higher water holding capacity and their occurrence in floodplain and claypan environments with negligible topographical relief, making them less susceptible to water erosion (Johns *et al.* 1984; Orr and Holmes 1984; Lewis *et al.* 2009b). However, this study found little evidence to support this; ground layer vegetation on heavy clay and sandy-loam soils responded in a similar way to the impact of different grazing treatments at site and plot scale. Only at the quadrat scale were there differences in relation to soil type, with more differences among grazing treatments on clay soils than sandy soils.

Despite effort made to control for potential environmental differences between adjacent sites, the nature of the study meant it was impossible to control for all factors. While representative of current conservation management in the region, conservation management was often associated with higher counts of kangaroo dung than adjacent commercially grazed properties, a consistent finding in arid Australia (Andrew and Lange 1986a; Landsberg *et al.* 2003), and there was evidence of unmanaged feral goats in conservation areas. Greater control of total grazing pressure in conservation reserves may improve conservation outcomes compared to commercially grazed areas (Fisher *et al.* 2005). The uneven pattern in which landscapes are grazed in large paddocks (Pringle and Landsberg 2004), differences in the grazing management techniques between landholders, and differences in the type of livestock between paired sites may have reduced the clarity of treatment effects. It is also acknowledged that 5 years may not have been enough time to respond to new management practices (AGM or CON), and legacy effects of past management and the long recovery times necessary for systems in dry environments (Meissner and Facelli 1999; Seymour *et al.* 2010; Fensham *et al.* 2011) may have affected results. Previous management can have a significant effect on the current composition and condition of communities (Waudby and Petit 2015) and some grazing-sensitive species may have already been lost from these communities (Silcock and Fensham 2013), as all conservation and alternatively grazed areas had been continuously grazed prior to implementation of the current management regimes. However, a minimum of 5 years under current management was chosen to ensure sufficient replicates of grazing treatments after consideration of the above-average rainfall conditions in the 5 years prior to surveys (Bureau of Meteorology 2017), which would have assisted in generating management responses. In addition, a greater focus on the timing and duration of rest periods would be beneficial to achieve optimum biodiversity outcomes under alternative grazing management (Müller *et al.* 2007). Despite these potential limitations, important differences were detected between grazing treatments.

3.6.1 Conclusion

Under certain environmental conditions, alternative grazing management incorporating long periods of planned rest from grazing achieved understorey plant composition and biodiversity measures similar to adjacent areas managed for conservation, and exceeded those of adjacent areas under traditional grazing management. The variability in our results with season, soil type and scale of sampling suggests that caution is required when interpreting data from one-off surveys at a single scale in semi-arid rangelands, and that the findings of such studies should not be extrapolated to other scales, sites or environmental conditions. Additional research is necessary to determine best grazing management practices in semi-arid rangelands, to understand under what environmental conditions and vegetation communities alternative grazing management is beneficial or detrimental to achieving conservation outcomes, and whether alternative grazing management is a viable option for conserving threatened species and communities in semi-arid rangelands.

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STATEMENT OF AUTHORS' CONTRIBUTION

(To appear at the end of each thesis chapter submitted as an article/paper)

We, the PhD candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated in the Statement of Originality.

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Table 4.1	88
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Chapter 5. Floristic diversity in rotationally grazed piospheres and an adjacent nature reserve in a semi-arid rangeland

5.1 Abstract

Rotational grazing, where paddocks are frequently rested for longer periods of time than they are grazed, may benefit soils, biodiversity and animal production in comparison to more conventional continuous grazing. Little research has investigated the potential to utilise alternative grazing management practices, such as rotational grazing, for large-scale biodiversity conservation and sustainable pastoral management of semi-arid rangelands. This study compared ground cover and floristic species composition and biodiversity at two scales (400-m^2 plots and 1-m^2 quadrats) in rotationally grazed paddocks managed for commercial livestock production with an adjacent public nature reserve (ungrazed by commercial livestock for 38 years), in semi-arid mulga (*Acacia aneura*) woodland in south-eastern Australia. Distance from water was used as an index of grazing intensity, and changes in ground cover, floristic composition and floristic biodiversity measures associated with this were also analysed for the livestock-grazed property. Although significant differences in understorey floristic species composition were observed between the rotationally grazed and ungrazed plots, there were no significant differences in understorey floristic species richness, diversity, functional richness, functional dispersion, functional diversity and ground cover between the rotationally grazed property and the nature reserve. However, these measures increased with distance from water on

the rotationally grazed property. The proportion and frequency of palatable species were greater in the ungrazed area, while unpalatable species showed the reverse trend. There were no differences in overstorey species richness or density between grazing treatments, or with increasing distance from water. These results show that while there was an impact associated with increasing grazing pressure close to water points under rotational grazing management, at a whole-paddock scale there were few differences in plant biodiversity and ground cover between the rotationally grazed and ungrazed treatment, although differences in floristic composition were apparent. This suggests the potential for alternative grazing strategies to sustain biodiversity and ground cover on commercial grazing properties, and may offer an alternative means of achieving broad-scale biodiversity conservation outcomes in semi-arid regions to protected areas and the exclosure of commercial livestock grazing.

5.2 Keywords

Biodiversity conservation, floristic composition, functional diversity, grazing gradient, grazing management, ground cover

5.3 Introduction

Incorporating frequent periods of rest from livestock grazing of greater duration than grazing periods (i.e. rotational grazing) reduces the negative impact of selective grazing by allowing recovery of preferred patches and plants after grazing, unlike continuous grazing systems (Earl and Jones 1996; Müller *et al.* 2007; Teague *et al.* 2008, 2013; Norton *et al.* 2013). Livestock grazing is a major driver of biodiversity loss and land degradation throughout the world (Steinfeld *et al.* 2006). Sustainable grazing management is required to maintain and improve biodiversity and productivity, and prevent degradation of landscape function in rangelands managed for commercial livestock production. Rotational grazing, along with careful management of grazing pressure, may improve sustainability and reduce the negative effects associated with livestock production (Chillo and Ojeda 2014; Chillo *et al.* 2015). However, recent reviews suggest that rotational grazing systems do not improve vegetation condition or animal production compared to continuous grazing systems (Briske *et al.* 2008; Bailey and Brown 2011). Differences in findings between studies are attributed to differences in spatial scale that rotational grazing treatments are examined under, the confounding effect of stocking rate, and the relatively short duration of studies that fail to show long-term effects of grazing management on variables such as species composition and soil health (Teague *et al.* 2013). In addition, many experimental studies of rotational grazing management do not reflect the flexible and adaptive management that is typically applied to commercial grazing practices (Teague *et al.* 2008, 2013).

Livestock are restricted by how far they can travel from water to graze, as they need to drink regularly (Pringle and Landsberg 2004). Therefore, in large paddocks where water sources are separated by large distances, grazing pressure near water sources is greater as livestock spend more time grazing or passing through the area immediately around the water (James *et al.* 1999; Todd 2006). These radial gradients in grazing intensity surrounding water points are termed ‘piospheres’ (Lange 1969). Many studies have utilised piospheres as grazing intensity gradients to understand grazing impacts in Australia (Andrew and Lange 1986b; Pickup and Chewings 1994; Friedel *et al.* 2003; Landsberg *et al.* 2003) and throughout the world (Fernandez-Gimenez and Allen-Diaz 2001; Todd 2006; Shahriary *et al.* 2012; Chillo and Ojeda 2014).

Patterns observed with increasing grazing pressure surrounding water points and elsewhere include an increase in small, prostrate, annual and forb species, and a decrease in palatable and perennial species (James *et al.* 1999; Heshmatti *et al.* 2002; Friedel *et al.* 2003; Landsberg *et al.* 2003; Hendricks *et al.* 2005), decreased plant species richness and diversity (Landsberg *et al.* 2003; Hendricks *et al.* 2005; Todd 2006), soil compaction (Andrew and Lange 1986b), reduced ground cover and cryptogam cover, increased soil erosion (Andrew and Lange 1986b; Bastin *et al.* 1993; Pickup and Chewings 1994; Tongway *et al.* 2003; Tabeni *et al.* 2014), and increased dung deposition (Andrew and Lange 1986b) and nutrient enrichment (Tolsma *et al.* 1987; Fernandez-Gimenez and Allen-Diaz 2001; Shahriary *et al.* 2012). In order to inform management that is compatible with biodiversity conservation goals, it is important to understand how piosphere patterns are influenced by alternative grazing systems, and differences between alternatively grazed areas and areas currently managed for conservation. We have found no published research undertaken in the semi-arid rangelands of south-eastern Australia comparing the effects of rotational grazing management with long-term ungrazed areas or the effects of grazing intensity under rotational grazing management on floristic biodiversity measures, ground cover and vegetation structure. Most research into alternative grazing management strategies in Australia has not examined large-scale paddock effects.

This study aimed to document differences in vegetation and ground cover in shrub-dominated woodland under commercial grazing and conservation systems using grazing intensity gradients surrounding water points. Patterns in understorey plant species biodiversity measures (Shannon–Wiener diversity, species richness, evenness, and turnover and functional diversity measures), composition, ground cover and community structure were compared between rotational grazing management and conservation management, and under different grazing intensities in the rotationally grazed property.

5.4 Methods

5.4.1 Study area

This study was undertaken in mulga (*Acacia anuera*) woodland in the Mulga Lands bioregion of north-western NSW, Australia (29°51'S, 143°53'E, ~150 m a.s.l.). The climate is semi-arid, with a mean annual precipitation of 282 mm (Bureau of Meteorology 2017b). More rainfall is recorded in the summer months on average, but rainfall is generally unpredictable and sporadic throughout the year. Mean maximum temperatures are 36.9°C in January and 18.2°C in July, and mean minimum temperatures are 22.0°C in January and 4.3°C in July (Bureau of Meteorology 2017b). Soils are deep sandy to sandy loam red earths, with relief to 5 m (Walker 1991). Mulga is the dominant tree or small shrub throughout the landscape, arranged in distinct groves and becoming denser along drainage lines. In addition, scattered ironwood (*Acacia excelsa*) and bimble box (*Eucalyptus populnea*), beefwood (*Grevillea striata*) and whitewood (*Atalaya hemiglauca*) are present, and areas of woody shrubs are common (*Eremophila* spp., *Dodonaea viscosa*, *Senna* spp.). *Eragrostis eriopoda* is the dominant understorey perennial grass. Only 186 mm was recorded in the 12 months prior to sampling.

5.4.2 Grazing management

Two adjacent properties were surveyed in April and May 2016: (1) a commercial grazing property, and (2) a public nature reserve. Both properties had historically been grazed by sheep and some cattle from the mid-1800s, with minimal fencing and water point development initially. Grazing intensity was relatively high for a period of time, before destocking due to drought and rabbits. Since 2004, the grazed property has been rotationally grazed by sheep and goats (past 10 years only) with stocking rates adjusted to match pasture availability and to prevent degradation of pasture and soils. Paddocks are rested for at least 6 months each year, with grazing periods ranging from 3 weeks to 4 months, depending upon seasonal conditions. Water points are fenced off when livestock are not in paddocks to reduce grazing pressure from other herbivores. The nature reserve was established in 1979, and commercial livestock grazing has been excluded since, but feral and native herbivores have access. Feral animals (goats, pigs, dogs, cats) are controlled as needed within the nature reserve.

5.4.3 Study design

Distance from water was used as an index of grazing intensity. Four paddocks (2079–3351 ha in size) were sampled on the rotationally grazed property, and a comparable geographical spread of sites was sampled on the adjacent reserve, which lacked paddocks and water points on comparable land systems (Table 5.1). All four paddocks sampled on the grazed property were rotationally grazed as described above. Each paddock represented an individual replicate of the rotational grazing treatment.

Table 5.1. Grazing information of replicates

Paddock	Size (ha)	Number of plots sampled	Number of water points	Livestock present?	Unmanaged herbivores present?
G1	2,427	10	2	Yes	Yes
G2	2,079	12	2	Yes	Yes
G3	2,165	9	2	Yes	Yes
G4	3,351	5	2	Yes	Yes
UG	74,728[†]	15	3*	No	Yes

* Two unfenced water points >7 km from nearest plots remained within the reserve, however they were in a different land system. One water point is located on an adjacent property without goat-proof fencing along the boundary, ~2.7 km from the nearest plot on the nature reserve

† Area of entire reserve (no paddocks within reserve)

Across the five paddocks, 70 points were randomly generated, with a minimum distance of 400 m between each point to achieve sample independence. Points were restricted to within 1000 m of fence lines and tracks on the rotationally grazed property, and within 2000 m of fence lines on the reserve, to allow timely completion of surveys and more time to survey plots, as it was not possible to drive to points. All points were ground-truthed, to ensure they were located on similar soil types and in the same vegetation community, and were at least 50 m from fence lines and tracks. Of the 70 points, 51 met these criteria and were sampled. At each point, a 20 × 20-m plot was established and nine 1 × 1-m quadrats were located systematically within each plot, with a 2-m buffer from the plot edge, and 6.5 m between each quadrat. The proportion of cover consisting of live plant material, litter (dead herbaceous material), cryptogam, coarse woody debris (>2 cm diameter), dung, rock and bare ground was estimated in each quadrat. In addition, the number of pellets of goat/sheep (these could not be differentiated), cattle, kangaroo and rabbit dung were counted in each quadrat to provide an additional measure of grazing intensity.

5.4.4 Vegetation surveys

All understorey floristic species (<3 m tall at maturity) were identified in each plot (400 m^2). The percent cover of each understorey floristic species was estimated in each quadrat (1 m^2). Understorey species richness in each plot and quadrat was calculated. Shannon–Wiener diversity and Pielou's evenness index were calculated for each plot using understorey species frequency data in plots. Understorey species turnover (pattern diversity) for each plot was calculated using the log of the slope of the understorey species accumulation curve generated from the quadrats in each plot. Overstorey woody plant species (>3 m in height) in each plot were identified and the numbers of each species ≤0.5 m in height (seedlings) and >0.5 m in height (large) were counted. Density of seedling and large overstorey woody species and species richness and Shannon–Wiener diversity of overstorey woody species were calculated for each plot.

The origin (native/exotic) and rarity (based on frequency of species encountered in this study) of each species identified at the plot scale was determined. Functional trait information was collected for each understorey species. Life history (annual/perennial), life form (graminoid/forb/shrub), functional group (annual grass/perennial grass/annual forb [all annual species except grasses]/perennial forb [all perennial species except grasses]) and palatability (unpalatable [not or rarely eaten]/moderately palatable [eaten but not readily or preferred, or when old]/palatable [eaten]; Cunningham *et al.* 2011) were analysed as categorical traits. Species height at maturity, seed length and leaf area index (leaf length × width) were analysed as continuous response variables. Richness and the proportion of total richness, cover and the proportion of total plant cover and frequency in plots were calculated for each categorical trait. Frequency data for species rarity and both categorical and continuous traits were analysed as community-weighted mean (CWM) scores where traits were weighted by each species' relative abundance (frequency) (Garnier *et al.* 2004). This was calculated as:

$$\text{CWM} = \sum_{i=1}^{n} \rho_i \times trait_i \qquad \textbf{Equation 5.1}$$

where ρ_i = the relative abundance of species i, and $trait_i$ = the trait value of species i.

Changes in functional trait diversity can be used to understand the effects of grazing management on ecological processes and ecosystem functioning, which may not be detected through analysis of floristic composition or traditional measures of biodiversity (richness, evenness, diversity; Cadotte *et al.* 2011). Functional diversity (Rao's quadratic entropy), functional richness, functional evenness and functional dispersion (Botta-Dukát 2005; Cadotte *et al.* 2011) were calculated for each plot using species frequency data and the above functional traits (excluding functional group as this was related to the life history and life form that was already included in the analysis), using the FD package in R (Laliberté *et al.* 2010).

5.4.5 Statistical analysis

5.4.5.1 Composition

Multivariate ordinations (unconstrained and constrained) were generated in CANOCO5 software (Ter Braak and Šmilauer 2012) using species frequency, cover and species incidence in plots. Goat and sheep dung, cattle dung, kangaroo dung, distance from water, density of woody species ≤0.5 m and >0.5 m in height and grazing treatment (rotationally grazed/ungrazed) were included as explanatory variables, without forward selection. Significant ($P \leq 0.05$) differences in composition between grazing treatments were determined using analysis of similarity (ANOSIM; vegan package in R; (Oksanen *et al.* 2007; R core team 2017), while similarity percentages (SIMPER; vegan) were used to highlight the significant species contributing to variation in species composition between the grazing treatments. Pearson's correlations were calculated to determine significant relationships between understorey floristic species frequency or cover and distance from water on the grazed property using the Hmisc package in R (Harrell 2016).

5.4.5.2 Univariate analysis

Linear mixed-effect models were generated in the lme4 package in R (Bates *et al.* 2016) to analyse dung counts, plant species richness, diversity, evenness, turnover, functional traits, functional diversity measures and overstorey seedling and large species density and richness. At plot scale, paddock was included as a random effect. At quadrat scale, paddock and plot within paddock were included as nested random effects. The effect of distance from water was analysed for rotationally grazed paddocks and was treated as a fixed-effect. Grazing treatment was analysed as a fixed effect in separate models using both rotationally grazed and ungrazed treatment data. If assumptions of models were not met a Generalized Linear Mixed-Effects Model with a Poisson error distribution was fitted (Bates *et al.* 2016).

5.5 Results

A total of 46 understorey floristic species were recorded in plots, 34 (74%) of which were also represented in quadrats (see Appendix, Table 5.1). The majority of species recorded were forbs (41% of species), followed by grasses (37%), and shrub or subshrub species (22%). Annual species totalled 22%, while 76% were perennial and one was unknown. One species recorded in plots, but not quadrats, was exotic. Fourteen species (30%) were recorded in only one plot. A species list is provided in the Appendix, Table A5.1.

More goat and sheep dung was recorded in rotationally grazed quadrats (25.8 ± 2.9 pellets m^{-2}, mean ± s.e.) than ungrazed quadrats (1.0 ± 0.4 pellets m^{-2}), and the amount of goat/sheep dung on the rotationally grazed property was negatively related to distance from water ($P < 0.001$; Figure 5.1a). More kangaroo dung was recorded in ungrazed quadrats (5.9 ± 1.2 pellets m^{-2}) than rotationally grazed quadrats (2.1 ± 0.3) and was positively related to distance from water on the rotationally grazed property ($P < 0.001$; Figure 5.1b).

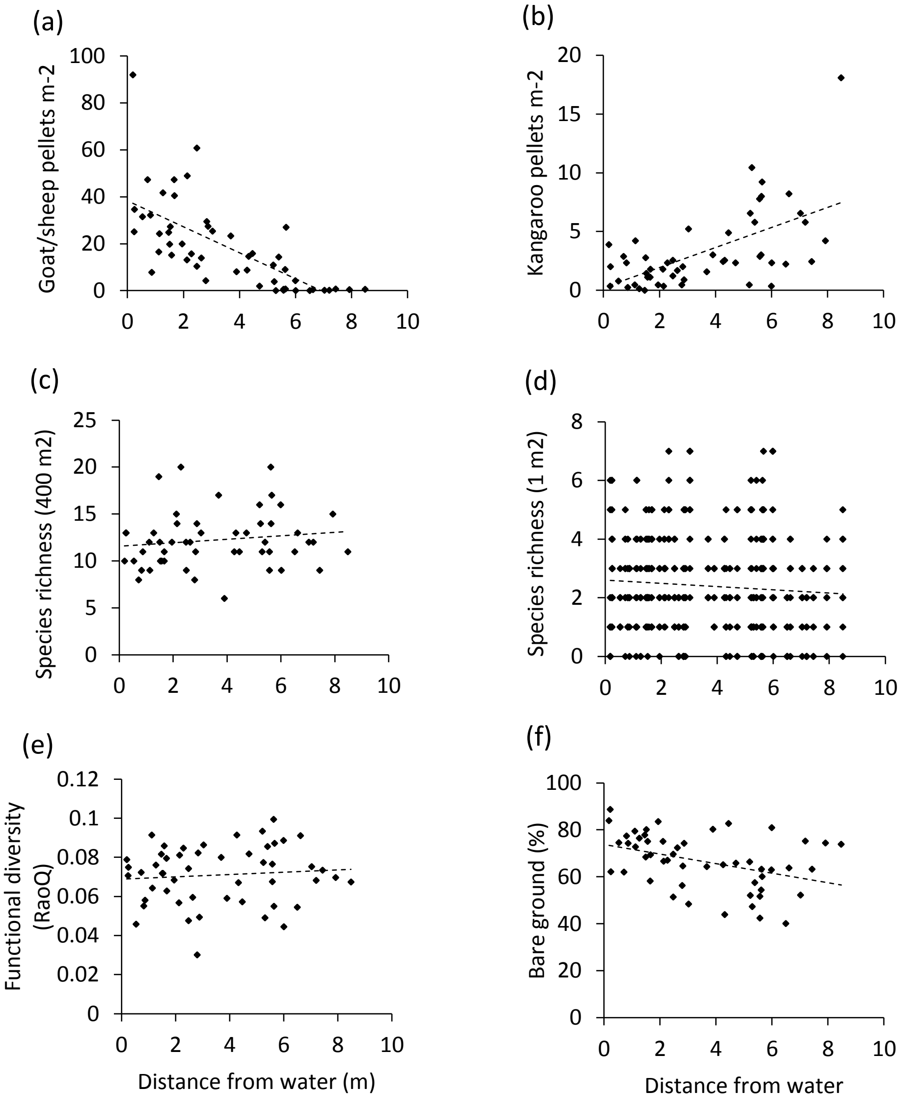

Figure 5.1. Relationship between distance from water and a) goat/sheep dung abundance (R^2 = 0.47); b) kangaroo dung abundance (R^2 = 0.35); c) plot species richness (R^2 = 0.02); d) quadrat species richness (R^2 = 0.01); e) functional diversity (RaoQ index) (R^2 = 0.01); and (f) bare ground (R^2 = 0.16).

5.5.1 Floristic species composition

Multivariate analyses showed a clear distinction in floristic composition between the rotationally grazed and ungrazed treatments (Figure 5.2). Unconstrained and constrained ordinations of understorey species frequency, cover and incidence data all showed similar patterns. For the constrained (RDA) species frequency ordination, the nominal variable, ‘grazing treatment’, explained the most variation (6.2%), followed by distance from water (3.9%), large (>0.5 m) overstorey species (3.4%), kangaroo dung (2.8%), cattle dung (1.7%), seedling (<0.5 m) overstorey species (1.2%) and goat/sheep dung (0.9%). Grazing treatment, distance from water and density of large overstorey species were all significant ($P \leq 0.05$) in this analysis.

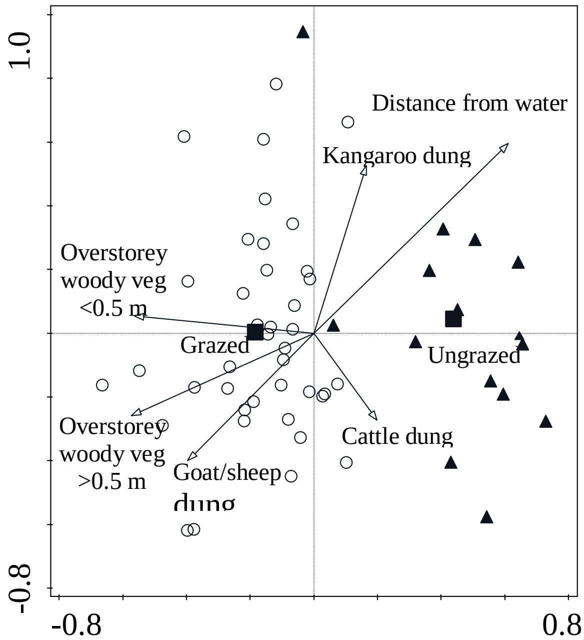

Figure 5.2. RDA ordination of species frequency in 1x1 m quadrats nested within 20 x 20 m plots ▲ = ungrazed plots, ○ = rotationally grazed plots; ■ = grazing treatment. Overstorey woody veg <0.5 m = density of woody vegetation species <0.5 m in height in plots; overstorey woody veg >0.5 m = density of woody vegetation species >0.5 m in height in plots; Grazed = nominal variable, grazed treatment; Ungrazed = nominal variable, ungrazed treatment; goat/sheep dung = abundance of goat/sheep pellets in quadrats; cattle dung = abundance of cow pats in quadrats; kangaroo dung = abundance of kangaroo pellets in quadrats; Distance from water = distance from nearest water point.

ANOSIM revealed a significant difference in species composition between the two grazing treatments (R = 0.13, *P* = 0.027). *Sclerolaena convexula* (SIMPER, *P* = 0.018) and *Eragrostis eriopoda* (*P* = 0.087) were more frequent in the rotationally grazed treatment and *Monachather paradoxa* (*P* = 0.022) and *Thyridolepis mitchellii* (*P* = 0.097) were more frequent in the ungrazed treatment (Table 5.2). Pearson's correlations of *Abutilon otocarpum, Aristida contorta, Portulaca oleracea, Sida* sp. (all for both frequency and cover) with distance from water were positive and significant ($P \leq 0.05$), as were *Tripogon lolliformis* (frequency only) and *Sclerolaena bicornis* (cover only; Table 5.3). No species were significantly negatively correlated with distance from water, though the frequency and cover of *Ptilotus sessifolius* was marginally significant ($P \leq 0.1$).

Table 5.2. Results from SIMPER analysis comparing rotationally grazed (RG) and ungrazed (UG) grazing treatments.

	Frequency in RG	**Frequency in UG**	**Cumulative sum of proportion of variance explained**	***P*-value**
Eragrostis eriopoda	0.494	0.370	0.147	0.087
Sclerolaena convexula	0.503	0.289	0.272	0.018
Tripogon loliiformis	0.352	0.348	0.364	0.410
Actinobole uliginosum	0.198	0.126	0.443	0.399
Austrostipa scabra	0.204	0.030	0.520	0.419
Aristida jerichoensis	0.179	0.022	0.592	0.554
Aristida contorta	0.191	0.081	0.659	0.622
Monachather paradoxa	0.034	0.119	0.708	0.022
Thyridolepis mitchelliana	0.056	0.081	0.750	0.097
Salsola australis	0.065	0.059	0.790	0.384
Digitaria sp.	0.068	0.037	0.820	0.502
Abutilon otocarpum	0.074	0.000	0.847	0.758
Maireana villosa	0.040	0.037	0.873	0.401
Chenopodium desertorum	0.025	0.052	0.898	0.163
Solanum ellipticum	0.040	0.044	0.923	0.221
Sida sp.	0.034	0.022	0.940	0.448
Portulaca oleracea	0.028	0.000	0.950	0.785
Triraphis mollis	0.015	0.000	0.957	0.294
Sclerolaena bicornis	0.019	0.000	0.962	0.475
Boerhavia dominii	0.012	0.000	0.967	0.647
Ptilotus polystachyus	0.003	0.007	0.972	0.294
Vittadinia sp.	0.012	0.000	0.977	0.694
Enteropogon acicularis	0.003	0.007	0.981	0.307
Einadia nutans	0.003	0.007	0.985	0.315
Eremophila latrobei	0.006	0.000	0.987	0.285
Evolvulus alsinoides	0.000	0.007	0.990	0.261
Ptilotus sessilifolius	0.006	0.000	0.992	0.521
Themeda australis	0.003	0.000	0.994	0.285
Chamaesyce drummondii	0.003	0.000	0.995	0.269
Sclerolaena muricata	0.003	0.000	0.996	0.312
Convolvulus erubescens	0.003	0.000	0.997	0.319
Goodenia fascicularis	0.003	0.000	0.998	0.282
Swainsona microphylla	0.003	0.000	0.999	0.282
Lepidium oxytrichum	0.003	0.000	1.000	0.276

Table 5.3. Pearson correlations between understorey floristic species frequency and distance from water. Values in bold ($P \leq 0.0.5$) are significant.

Species	Frequency		Cover	
	R	***P*-value**	**R**	***P*-value**
Abutilon otocarpum	0.39	0.020	0.39	0.020
Actinobole uliginosum	0.10	0.562	-0.03	0.843
Aristida contorta	0.47	0.004	0.39	0.017
Aristida jerichoensis	-0.22	0.203	-0.25	0.143
Austrostipa scabra	0.08	0.623	0.08	0.654
Boerhavia dominii	-0.10	0.544	-0.10	0.544
Chamaesyce drummondii	-0.17	0.324	-0.17	0.324
Chenopodium desertorum	-0.09	0.619	-0.09	0.619
Convolvulus erubescens	0.27	0.106	0.27	0.106
Digitaria sp.	0.03	0.849	-0.14	0.407
Einadia nutans	0.32	0.061	0.32	0.061
Enteropogon acicularis	0.04	0.825	0.04	0.825
Eragrostis eriopoda	-0.21	0.215	-0.26	0.125
Eremophila latrobei	-0.12	0.472	-0.12	0.472
Goodenia fascicularis	0.32	0.058	0.32	0.058
Lepidium oxytrichum	-0.02	0.899	-0.02	0.899
Maireana villosa	0.07	0.683	-0.13	0.443
Monachather paradoxa	-0.13	0.443	-0.12	0.496
Portulaca oleracea	0.48	0.003	0.48	0.003
Ptilotus polystachyus	-0.06	0.748	-0.06	0.748
Ptilotus sessilifolius	-0.33	0.052	-0.33	0.052
Salsola australis	0.11	0.531	0.09	0.609
Sclerolaena bicornis	0.31	0.066	0.35	0.038
Sclerolaena convexula	-0.08	0.627	-0.14	0.432
Sclerolaena muricata	0.32	0.061	0.32	0.061
Sida sp.	0.35	0.036	0.35	0.036
Solanum ellipticum	0.28	0.095	0.29	0.087
Swainsona microphylla	0.32	0.058	0.32	0.058
Themeda australis	0.20	0.247	0.20	0.247
Thyridolepis mitchelliana	-0.12	0.487	-0.07	0.665
Tripogon loliiformis	0.35	0.036	0.21	0.219
Triraphis mollis	0.04	0.825	0.04	0.825
Vittadinia sp.	0.27	0.117	0.27	0.117

5.5.2 Univariate analyses

5.5.2.1 Grazing treatment

At the plot scale, palatable species were more frequent ($P \leq 0.05$) in the ungrazed treatment, while unpalatable species were more frequent in the rotationally grazed treatment (Table 5.4), and understorey species turnover (pattern diversity) was significantly greater in the ungrazed treatment (0.053 ± 0.002 and 0.064 ± 0.005 in rotationally grazed and ungrazed treatments, respectively, $P = 0.021$). However, there were no significant differences between the two grazing treatments in understorey species richness or the richness or proportions of other functional trait categories (Table 5.5), Shannon–Wiener diversity, evenness or functional diversity measures ($P > 0.05$) at the plot scale.

Table 5.4. Community-weighted mean (CWM) (± 1 s.e.) of plant functional trait categories in the rotationally grazed (RG) and ungrazed (UG) treatments. Values in bold are significant ($P \leq 0.05$).

Functional trait category	CWM		*P*-value
	RG	UG	
Annual	0.16 (0.02)	0.15 (0.03)	0.711
Annual forb	0.10 (0.01)	0.10 (0.03)	0.804
Annual grass	0.06 (0.010	0.05 (0.02)	0.534
Perennial	0.84 (0.02)	0.85 (0.03)	0.711
Perennial forb	0.29 (0.02)	0.25 (0.04)	0.744
Perennial grass	0.54 (0.17)	0.60 (0.04)	0.796
Unpalatable	0.40 (0.02)	0.28 (0.03)	**0.004**
Moderately palatable	0.40 (0.02)	0.34 (0.04)	0.198
Palatable	0.20 (0.02)	0.38 (0.04)	**<0.001**
Height	0.50 (0.01)	0.46 (0.02)	0.122
Seed length	2.09 (0.10)	1.87 (0.13)	0.513
Leaf area	549.68 (72.57)	265.23 (64.35)	0.302
Rarity	0.33 (0.02)	0.31 (0.02)	0.794

At the quadrat scale, perennial grass and unpalatable species richness were greater in rotationally grazed quadrats than ungrazed quadrats (Table 5.5). There was also a significantly greater proportion of unpalatable species in rotationally grazed quadrats and a significantly greater proportion of palatable species in ungrazed quadrats, but again there was no difference in total species richness between rotationally grazed and ungrazed quadrats.

Within the plots, density of seedling and large overstorey species was greater in the rotationally grazed plots (seedlings = 23.19 ± 4.37; large = 31.08 ± 4.93) than ungrazed plots (seedlings = 4.07 ± 1.01; large = 17.00 ± 2.77), but the differences were not significant ($P > 0.05$). There was no significant difference in species richness of overstorey species in rotationally grazed and ungrazed plots (3.67 ± 0.19 and 5.53 ± 0.39, respectively), but diversity of overstorey species was greater in rotationally grazed plots (0.83 ± 0.05) than ungrazed plots (0.51 ± 0.11; $P = 0.004$). There was a greater proportion of palatable plant cover in the ungrazed compared to the rotationally grazed treatment (Table 5.6), and greater cover of unpalatable species in the rotationally grazed plots, but there were no significant differences in total ground cover, or of plant, litter or cryptogam cover between grazing treatments.

Table 5.5. Species richness and proportion of plant species by different functional groups (mean ± 1 s.e.) at plot and quadrat scales. Values in bold are significant (*P* ≤ 0.05). RG = rotationally grazed treatment, UG = ungrazed treatment.

	Plot scale						**Quadrat scale**					
	Species richness		***P*-value**	**Proportion of total species richness**		***P*-value**	**Species richness**		***P*-value**	**Proportion of total species richness**		***P*-value**
	RG	**UG**		**RG**	**UG**		**RG**	**UG**		**RG**	**UG**	
Total	12.50 (0.54)	11.47 (0.54)	0.78	–	–	–	2.69 (0.88)	1.75 (0.12)	0.265	–	–	–
Annual	2.33 (0.20)	1.60 (0.19)	0.523	0.18 (0.01)	0.14 (0.01)	0.372	0.50 (0.04)	0.27 (0.04)	0.408	0.18 (0.01)	0.11 (0.02)	0.066
Annual forb	1.56 (0.18)	0.93 (0.12)	0.606	0.12 (0.01)	0.08 (0.01)	0.671	0.31 (0.03)	0.19 (0.04)	0.127	0.11 (0.01)	0.07 (0.01)	0.109
Annual grass	0.78 (0.07)	0.67 (0.13)	NA*	0.07 (0.04)	0.06 (0.04)	NA*	0.19 (0.02)	0.08 (0.02)	NA*	0.07 (0.01)	0.05 (0.01)	NA*
Perennial	10.17 (0.42)	9.80 (0.47)	0.916	0.72 (0.01)	0.86 (0.02)	0.421	2.19 (0.08)	148 (0.10)	0.222	0.79 (0.02)	0.67 (0.04)	0.444
Perennial forb	4.89 (0.36)	4.93 (0.34)	0.904	0.38 (0.02)	0.43 (0.02)	0.445	0.78 (0.05)	0.47 (0.06)	0.513	0.26 (0.02)	0.19 (0.02)	0.058
Perennial grass	5.28 (0.20)	4.87 (0.27)	0.431	0.44 (0.02)	0.43 (0.02)	0.848	1.41 (0.05)	1.01 (0.08)	**0.013**	0.51 (0.02)	0.48 (0.03)	0.753
Unpalatable	3.92 (0.21)	3.13 (0.24)	0.363	0.32 (0.01)	0.27 (0.02)	0.094	1.03 (0.048)	0.47 (0.05)	**<0.001**	0.36 (0.02)	0.19 (0.02)	**<0.001**
Mod palatable	3.69 (0.25)	3.27 (0.24)	0.726	0.29 (0.01)	0.28 (0.03)	0.569	1.01 (0.05)	0.61 (0.06)	0.202	0.37 (0.02)	0.28 (0.03)	0.064
Palatable	3.39 (0.25)	3.27 (0.25)	0.969	0.27 (0.01)	0.29 (0.02)	0.563	0.53 (0.04)	0.59 (0.05)	0.753	0.17 (0.01)	0.28 (0.030	**0.007**

* Data did not meet assumptions of model and therefore was not analysed (very low richness)

Table 5.6. Percent ground cover (± 1 s.e.) by different components and proportion of plant cover by functional groups. RG = rotationally grazed treatment, UG = ungrazed treatment. Values in bold are significant (*P* ≤ 0.05).

	Cover		*P*-value	Proportion of total plant cover		*P*-value
	RG	UG		RG	UG	
Total	5.14 (0.62)	2.59 (0.73)	0.302	–	–	–
Bare ground	69.13 (1.75)	60.11 (3.24)	0.232	–	–	–
Litter cover	19.07 (1.48)	25.01 (3.21)	0.060	–	–	–
Cryptogam cover	3.64 (0.84)	10.76 (1.78)	0.189*	–	–	–
Annual	0.22 (0.05)	0.05 (0.02)	0.151*	0.08 (0.02)	0.05 (0.03)	0.175*
Annual forb	0.08 (0.02)	0.03 (0.01)	0.096*	0.03 (0.01)	0.02 (0.01)	0.754*
Annual grass	0.14 (0.04)	0.02 (0.01)	0.274*	0.05 (0.01)	0.03 (0.02)	0.186*
Perennial	4.98 (0.61)	2.56 (0.73)	0.318	0.94 (0.02)	0.96 (0.01)	1†
Perennial forb	0.24 (0.05)	0.13 (0.02)	0.752*	0.13 (0.03)	0.12 (0.03)	0.810*
Perennial grass	4.74 (0.63)	2.43 (0.74)	0.351	0.81 (0.04)	0.84 (0.04)	0.961‡
Unpalatable	0.40 (0.06)	0.13 (0.02)	**0.002***	0.16 (0.03)	0.13 (0.04)	0.531*
Moderately palatable	4.47 (0.63)	2.15 (0.74)	0.336	0.73 (0.05)	0.61 (0.08)	0.434*
Palatable	0.20 (0.05)	0.29 (0.78)	0.105	0.08 (0.03)	0.24 (0.06)	**0.001***

* Square-root transformation used in models
† Glmer model with a binomial family used in model
‡ Power of 2 transformation used in model

5.5.2.2 Distance from water

At the plot scale, species richness (Figure 5.1c), along with annual, perennial, perennial forb and palatable species richness, and frequency of annual and annual grasses, were positively related to distance from water ($P \leq 0.05$). Species diversity was also positively related to water (marginally significant, $P \leq 0.1$). In addition, functional richness and functional diversity (Figure 5.1e) were positively related to distance from water ($P \leq 0.05$). However, the proportion of perennial grass species and frequency of perennial species was negatively related to distance from water

($P \leq 0.05$). There were no significant relationships between distance from water and species diversity, evenness or turnover.

At the quadrat scale, the number and proportion of total annual and annual grass species was positively related to distance from water, while the proportion of perennial species was negatively related to distance from water ($P \leq 0.05$). Species richness was not significantly correlated with distance from water at the quadrat scale (Figure 5.1d). We found no significant relationships between distance from water and the density or richness of overstorey species in plots. The proportion of bare ground cover ($P = 0.002$) was negatively related to distance from water (Figure 5.1f), while cryptogam cover ($P < 0.001$) and litter cover (marginally significant, $P = 0.080$) were positively related to distance from water. There were no differences in the proportion of the cover of different plant functional groups with distance from water.

5.6 Discussion

In this study, we compared a commercial livestock grazing property that regularly rotates livestock and rests paddocks with an adjacent nature reserve where commercial livestock production had been excluded for 38 years, and examined trends associated with distance from water, to identify the impacts of alternative grazing management versus grazing exclusion and increasing intensity of rotational grazing on floristic composition, diversity, vegetation structure and ground cover. Significant differences in floristic composition, diversity and ground cover were observed with increasing grazing intensity and between rotationally grazed paddocks and the nature reserve.

5.6.1 Composition

Multivariate ordination revealed differences in understorey floristic species composition between rotationally grazed and ungrazed plots. Livestock can modify species composition through herbivory and soil disturbance, resulting in species vulnerable to grazing being lost from the species pool, while selection allows for increases in species with traits that avoid or benefit from grazing (Olff and Ritchie 1998). In our study, greater turnover was recorded in the ungrazed than rotationally grazed plots, indicating homogenisation of the understorey floristic community under grazing. More grazing-sensitive species may have been present in ungrazed plots which may explain why turnover was greater in ungrazed plots. *Thyridolepis mitchelliana, Monachather paradoxa* and *Chenopodium desertorum* are grazing-sensitive (decreaser) and palatable species that are often selectively grazed (Beadle 1948; Harrington *et al.* 1984; Cunningham *et al.* 2011). These species were more frequent in the ungrazed reserve, indicating they may have declined under grazing on the adjacent commercially grazed property. By contrast, *Sclerolaena convexula* was more frequent in the grazed treatment as it has spines that can deter livestock and is therefore more persistent under grazing. *Eragrostis eriopoda*, which was also more frequent on the grazed property, is a hardy perennial species tolerant of drought and grazing, provided grazing pressure is not too high (Cunningham *et al.* 2011).

Distance from water was a major factor explaining differences in floristic composition, as seen in the ordination analysis. The majority of species significantly ($P \leq 0.05$) or marginally significantly ($P \leq 0.05$) correlated with distance from water increased in cover or frequency further from water and were decreasers. Only one species, *Ptilolus sessifolius*, was negatively associated with distance from water ($P \leq 0.1$). This species is a grazing-tolerant perennial, is not readily grazed, and recovers well from grazing

(Cunningham *et al.* 2011). These results are similar to previous studies in semi-arid Australia, such as that of Landsberg *et al.* (2003), who found a greater number of decreaser than increaser species associated in response to grazing intensity.

5.6.2 Functional composition

Previous research suggests that unpalatable, prostrate, exotic, annual and forb species are more abundant under increasing livestock grazing intensity (Noy-Meir *et al.* 1989; Friedel *et al.* 2003; Landsberg *et al.* 2003; Hendricks *et al.* 2005; Díaz *et al.* 2007). In this study, unpalatable species were more frequent and had greater cover in rotationally grazed plots. At the quadrat scale, there was a greater richness and proportion of unpalatable species under rotational grazing. In contrast, palatable species were more frequent in and contributed a greater proportion of plant cover in the ungrazed plots. A greater proportion of palatable species were also recorded in the quadrats in the nature reserve. These patterns were expected as selective grazing by livestock increases grazing pressure on palatable species (Díaz *et al.* 2007). Friedel *et al.* (2003) recorded a similar reponse of palatable species along grazing gradients in Australian rangelands. Perennial grass richness was also greater in grazed quadrats. At a local scale, moderate livestock grazing pressure can reduce inter-species competition and create niches for plant establishment (Olff and Ritchie 1998), which may explain the greater richness of perennial species at the smallest scale. No other significant differences were detected in functional composition between the grazing treatments. The conservative grazing regime, which incorporated long periods of rest from grazing, may have minimised negative effects associated with livestock grazing, allowing the vegetation to recover between grazing events (Chillo and Ojeda 2014; Chillo *et al.* 2015). The frequencies of annual species and annual grasses increased with distance from water and the frequency of total perennial species and the proportion of perennial grass species decreased with distance from water. There were no other significant differences in richness, proportion or frequency of functional groups in relation to distance from water. By contrast, Landsberg *et al.* (1997a) and Díaz *et al.* (2007) reported more annuals and fewer perennials with increasing grazing intensity surrounding water points. The patterns observed in this study were a reflection of the dry season, with more palatable annual species having been selectively grazed and eliminated closer to water, but still present further from water at low grazing intensity. Perennial species have a more robust root system than annuals, allowing access to a larger volume of soil and moisture, which may assist them in persisting despite high grazing intensity, and in resprouting after grazing. Perennials also generally have tougher leaves, which may reduce the preferential grazing of perennial species compared to annual species and hence they comprise a greater proportion of the species pool under higher grazing intensities.

Greater functional diversity is beneficial for ecosystem function and resilience (Cadotte *et al.* 2011; Valencia *et al.* 2015). In this study, functional richness, diversity and dispersion significantly increased with distance from water. Few other studies have considered impacts of grazing management on functional diversity measures in rangelands. These results support those of Chillo *et al.* (2017), who also reported a loss of functional diversity associated with increasing grazing intensity close to water points. As livestock graze vegetation and modify habitat, selection favours traits that convey tolerance of grazing disturbance, such as prostrate, unpalatable or annual species with a short life cycle (Díaz *et al.* 2001). Species with traits that convey sensitivity to grazing are lost.

5.6.3 Richness, diversity, evenness, turnover

In dry, infertile environments, moderate to intense livestock grazing is expected to reduce plant biodiversity due to the local extinction of grazing-sensitive species, and the limited potential for colonisation due to infertility (Milchunas *et al.* 1988; Olff and Ritchie 1998). In addition, Australia's rangelands have a short history of livestock grazing and therefore may be more vulnerable to sustained ungulate grazing than rangelands with a long evolutionary history of ungulate grazing, such as the rangelands of Africa or the Americas (Milchunas *et al.* 1988; Olff and Ritchie 1998; Landsberg *et al.* 2003). Species richness increased with distance from water as grazing intensity decreased, as did diversity (marginally significant), consistent with previous studies in semi-arid regions (Landsberg *et al.* 2003; Hendricks *et al.* 2005; Todd 2006; Chillo *et al.* 2015). However, there was no significant difference in species richness or diversity between the rotationally grazed and ungrazed treatments in this study. Arid and semi-arid regions experience unpredictable rain events and frequent droughts. Both grazing and drought (water stress) result in loss of vegetative material, and adaptations to tolerate water stress can be comparable to adaptations for tolerance of grazing due to the similarity of the selection pressures (Milchunas *et al.* 1988). Rotational grazing with long rest periods can allow soil and vegetation to recover between grazing events (Müller *et al.* 2007; Teague *et al.* 2008), and careful management of stocking rates can reduce overgrazing and the associated negative effects, which may also explain the lack of difference between the rotationally grazed and ungrazed treatments. Silcock and Fensham (2013) also found no difference in plant species richness between grazed areas and areas excluded from livestock in long-term exclosures in arid Australia.

5.6.4 Vegetation structure and ground cover

Increased grazing pressure is often associated with shrub encroachment in semi-arid rangelands (Archer *et al.* 1995; Moleele and Perkins 1998; Van Auken 2000, 2009; Local Land Services 2014). However, in this study, woody vegetation density was not significantly related to grazing pressure, and there was no difference in woody vegetation density between rotationally grazed and ungrazed plots. Goats were the dominant livestock grazing the commercial property for 10 years prior to this study. Goats are known to consume a significant portion of their diet from shrub and tree species (Wilson *et al.* 1975; Harrington 1979; Dawson and Ellis 1996) and may have reduced the reproductive potential and establishment of new shrubs, balancing the tendency for livestock grazing to facilitate shrub encroachment. In addition, significant heterogeneity of woody vegetation patches across the landscape may have increased the variability of results and masked significant differences, as means of both seedlings and large woody vegetation density were greater in the rotationally grazed treatment. Rotational grazing of goats, allowing the paddocks to rest between grazing events, may have also reduced the impact of grazing on woody shrub encroachment.

As distance from water decreased, grazing intensity increased. Soil compaction, trampling and herbivory are expected to increase with grazing intensity and can all reduce ground cover. The increase in bare ground and decline in cryptogam and litter cover associated with proximity to water in this study were consistent with previous studies (Andrew and Lange 1986b; Tongway *et al.* 2003; Tabeni *et al.* 2014). Increased bare ground is associated with soil erosion and reduced landscape function (Freudenberger *et al.* 1997; Bartley *et al.* 2006; Muñoz-Robles *et al.* 2011). Thus, even conservative rotational grazing management can have an impact on ground cover close to water where grazing intensity is greater, than at the furthest points from water in large paddocks. However, when averaged at the paddock scale, including the plots further from water, there was no significant difference in the various components of ground cover between the rotationally grazed and ungrazed treatments – reduced ground cover was restricted to the high-impact areas near water points.

5.6.5 Dung

As expected, goat and sheep dung abundance was negatively correlated with distance from water, supporting our use of distance from water as a measure of grazing intensity. Kangaroo dung counts were positively correlated with distance from water. Previous research has found little relationship between kangaroo density and distance from water for up to 6 km from water (Montague-Drake and Croft 2004; Fensham and Fairfax 2008) and differing spatial distributions of livestock (sheep and cattle) and kangaroos (Andrew and Lange 1986a). Kangaroos are not constrained by commercial fencing and so prefer to avoid livestock and access otherwise untouched feed. Fensham and Fairfax (2008) suggested that sheep can travel up to 3 km from water, cattle 6 km and red kangaroos 7 km, but these distances depend on seasonal conditions and the availability of forage (James *et al.* 1999), landscape heterogeneity, the salinity of the drinking water, grazing history and provision of supplements (Pringle and Landsberg 2004). In this study, some goat and sheep dung was recorded in plots up to 6 km from water, and kangaroo dung >8 km from water.

5.6.6 Study design limitations

The design of this study was limited by the inability to find replicate properties of rotational grazing paddocks and more than one protected area on the same land system. It is therefore important that further comparisons of rotational grazing and traditional conservation management (livestock excluded) are undertaken to confirm whether the results found in this study are replicated across other land systems in western NSW rangelands. Previous studies have highlighted variation when large study areas have been utilised (see Chapter 3; Landsberg *et al.* 2003). The decision to focus on just one land system was justified to ensure the results of this study are relevant and applicable to the local surrounding area. In addition, random location of plots with a minimum distance of 400 m between plots ensured plots were independent.

5.6.7 Conclusion

Greater understanding of the impacts of alternative grazing regimes and grazing intensity on vegetation and ground cover in semi-arid rangelands is necessary to improve biodiversity and the environmental sustainability of livestock production enterprises. We found significant differences in floristic composition between a commercial rotationally grazed property and an adjacent nature reserve area and with increasing distance from water on the rotationally grazed property. Though results differed with the scale of analysis, the protected area usually had a greater frequency, richness and proportion of palatable species, while the rotationally grazed paddocks showed the reverse trend. Plant species richness, diversity, functional diversity measures and ground cover increased with distance from water (as grazing intensity decreased). However, there were no differences in these measures between the rotationally grazed property and the ungrazed protected area. While there was an impact associated with increasing grazing pressure close to water points under rotational grazing management, there were few differences in plant biodiversity and ground cover between the rotationally grazed and ungrazed treatments at the paddock scale. This suggests that rotational grazing and similar alternative grazing strategies may have the potential to sustain biodiversity and ground cover and offer an alternative method of large-scale conservation to exclusion of livestock from protected areas in semi-arid regions.

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STATEMENT OF AUTHORS' CONTRIBUTION

(To appear at the end of each thesis chapter submitted as an article/paper)

We, the PhD candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated in the Statement of Originality.

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Chapter 6. Synthesis and conclusions

6.1 Introduction

Protected areas are inadequate for conserving all biodiversity (James *et al.* 1995; Margules and Pressey 2000; Rodrigues *et al.* 2004; Fischer *et al.* 2006; Lindenmayer *et al.* 2010). Off-reserve conservation is necessary to assure regional and continental-scale biodiversity conservation, and may also improve connectivity between reserves (Morton *et al.* 1995; Fischer *et al.* 2006; Heller and Zavaleta 2009; Lindenmayer *et al.* 2010; Salmon and Gerritsen 2013; Butchart *et al.* 2015). Alternate grazing management strategies have the potential to maintain and improve biodiversity conservation and landscape function outside reserves in pastoral regions, compared to conventional grazing management (Norton 1998a; Dorrough *et al.* 2004; Teague *et al.* 2008; Papanastasis 2009). In recent decades, a considerable amount of research has been undertaken into grazing management to improve both the productivity and the environmental sustainability of grazing enterprises. However, detailed information on the effects of different grazing management strategies in semi-arid rangelands has been lacking. Landholders and resource managers need local, reliable scientific research to inform grazing management and conservation goals. In particular, little research has been undertaken in western NSW rangelands comparing alternative grazing management strategies that incorporate long periods of rest, with traditional continuous grazing strategies and areas managed for conservation.

6.1.1 Aims and objectives

Preceding chapters of this thesis have: (1) reviewed the global literature on the ecological and animal production effects of grazing management that incorporates periods of strategic rest, compared to continuous grazing strategies and ungrazed areas, and examined the extent to which the literature reports simultaneously on ecological and production outcomes; (2) investigated the potential for alternative livestock grazing practices to improve biodiversity, ground cover, soil properties and landscape function compared to traditional continuous grazing practices and areas managed for conservation on contrasting soil types in western NSW, and explored the potential of these alternative livestock grazing systems for off-reserve biodiversity conservation; and (3) examined changes in floristic species composition, biodiversity, ecosystem structure and ground cover associated with alternative grazing management along a grazing intensity gradient. This chapter provides a summary of the main findings of this research and explains how it contributes to current theoretical and practical knowledge of grazing management and biodiversity conservation in semi-arid rangelands. Limitations of the research are outlined and recommendations for future research to improve biodiversity conservation and landscape function in livestock grazing enterprises in semi-arid rangelands are suggested.

6.2 Summary of main findings

6.2.1 Review of ecological and production effects of incorporating periods of rest from grazing in grazing regimes

In Chapter 2, a systematic review and meta-analyses of scientific literature were undertaken describing the response of ecological and animal production variables to grazing management incorporating periods of planned rest (strategic-rest grazing, SRG) compared with continuously grazed (CG) and ungrazed (UG) systems. A trend analysis was undertaken to determine the proportion of papers that reported significantly greater, neutral or lesser outcomes in ecological and animal production response variables. The majority of studies reported greater or neutral plant, mammal and bird species richness and diversity under SRG compared to UG areas, but neutral or lower invertebrate richness and diversity, ground cover and biomass. Invertebrate species richness and diversity, and animal production per unit area were usually greater under SRG, than CG. Responses of plant, mammal and bird species richness and diversity, plant biomass, and livestock weight gain were predominantly neutral, when comparing SRG with CG management. However, when a difference was observed, these ecological and animal production variables were more often in favour of SRG than CG, with the exception of livestock weight gain, which showed the reverse trend. Most studies reported a difference in species composition between SRG areas and CG or UG systems, highlighting the importance of investigating changes in composition that may not otherwise be detected using traditional measures of richness and diversity.

Meta-analyses were undertaken to examine the effects of SRG on plant species richness, diversity, animal weight gain and animal production per unit area compared to CG or UG systems. Overall, no significant difference in plant species richness, diversity, livestock weight gain or animal production per unit area was found between SRG and CG or UG systems. The meta-analyses revealed that type of SRG management and the amount of rest relative to grazing time (rest:graze ratio) were important contributors to differences between SRG and CG or UG systems. Multi-paddock SRG systems resulted in lower plant richness relative to CG, but seasonal SRG systems had greater diversity than CG systems. As the rest:graze ratio increased, both weight gain and animal production per unit area increased under SRG compared to CG, but plant richness decreased. These differences highlight the importance of detailed reviews of the effects of different types of SRG management, to tease out potential grazing strategies that are beneficial or detrimental to achieving ecological as well as livestock production outcomes.

An understanding of both the ecological and economic (production) trade-offs associated with different grazing management strategies is necessary to make informed decisions about best-management practices (Metera *et al.* 2010). The extent to which ecological and animal production outcomes were considered simultaneously in the literature comparing SRG with CG or UG systems was investigated. The majority of studies did not report ecological and animal production response variables simultaneously. This indicates that information pertinent to one discipline is not being communicated effectively to the other, and vice versa, impeding progress towards a holistic understanding of how to integrate livestock production and biodiversity conservation outcomes in grazing lands.

6.2.2 Response of biodiversity to alternative grazing management systems

In Chapters 3 and 5, understorey floristic species composition and biodiversity measures were compared at different scales between alternative grazing management, traditional grazing management (Chapter 3 only), and areas currently managed for conservation. In comparison to traditional continuous grazing management, alternative grazing management resulted in greater understorey plant species richness and diversity, depending on the season and scale of sampling. Differences were more pronounced in spring and at larger scales (100 and 1000 m). At small (1 m) scales, grazing can reduce competition and provide niches for germination, but as grazing-sensitive species are lost from the species pool, richness declines at larger scales (Olff and Ritchie 1998; Landsberg *et al.* 2002; Kohyani *et al.* 2008). When comparing alternative grazing management with adjacent areas that were ungrazed by commercial livestock and managed for conservation, there were few differences in plant richness, Shannon–Wiener diversity or functional diversity measures (Chapters 3 and 5). However, more infrequent species were recorded in conservation areas (Chapter 3), and when examined at a local (property) scale, significant differences in composition were apparent. For example, an increase in palatable species such as *Thyridolepis mitchelliana*, *Monachather paradoxa* and *Chenopodium desertorum* was recorded in ungrazed areas and an increase in unpalatable species in areas under rotational grazing (Chapter 5). Selective grazing of palatable species can reduce their vigour and abundance in semi-arid rangelands (Milchunas *et al.* 1988; Olff and Ritchie 1998).

Soil type (clay and sand) explained the greatest amount of floristic variation in the ordination analysis (Chapter 3). The location (longitude and latitude) of sites, seasons and preceding rainfall also explained a significant proportion of variation in understorey floristic composition, whereas the effect of grazing treatment on floristic composition at the regional scale was not significant. In arid and semi-arid rangelands, sporadic rainfall events and non-equilibrium community dynamics are the dominant drivers of change, masking the effects of grazing and other disturbances (Ellis and Swift 1988; Westoby *et al.* 1989; Silcock and Fensham 2013), and these drivers may likely account for the lack of a grazing treatment effect on composition at the regional scale. These findings suggest that commercial grazing management incorporating long periods of rest is more compatible with biodiversity conservation than traditional grazing management, but potentially at the expense of palatable, rare and grazing-sensitive species.

6.2.3 Response of ground cover, soil and landscape function to alternative grazing management systems

Chapter 4 focused on the effects of alternative grazing management, traditional grazing management, and exclusion of commercial livestock grazing for conservation, on ground cover, soil attributes and landscape function indices. Areas managed under alternative grazing management had greater total ground cover in comparison to traditional grazing management treatments, but both alternative and traditional grazing management treatments had significantly less ground cover than adjacent areas managed for conservation. Herbivory and trampling reduce the amount of understorey plant biomass, increase the rate of litter decomposition, and destroy cryptogamic crusts (Graetz 1986; Eldridge and Greene 1994; Williams *et al.* 2008; Schönbach *et al.* 2010; Mofidi *et al.* 2012). Ground cover was also compared between a rotationally grazed property and an adjacent nature reserve in Chapter 5. However, unlike the results in Chapter 4, no difference in ground cover was detected between the rotationally grazed area and the nature reserve in this comparison, indicating that rotational grazing management can maintain ground cover under certain circumstances.

Evidence of the benefits of alternative grazing management for landscape function was also obtained. Alternative grazing management did not differ significantly from areas managed for conservation in terms of landscape function, whereas many indices of landscape function (stability, nutrient cycling, landscape organisation index, patch area and average interpatch length) were significantly lower under traditional grazing management than conservation treatments, indicating the impact of alternative grazing management on landscape function may have been lower than traditional grazing management. However, there were also no significant differences in landscape function indices between alternatively grazed areas and adjacent traditionally grazed areas. Greater ground cover and LFA indices under alternative grazing management compared to traditional grazing management were likely a result of rest from grazing allowing recovery of plants and soil structure between grazing events (Teague *et al.* 2011). Soil organic carbon, bulk density, pH and EC did not differ significantly between grazing treatments, but organic nitrogen was greater under traditional grazing management than under conservation management on sandy soils, presumably as a result of dung deposition (Eldridge *et al.* 2016).

6.2.4 Relationships between floristic diversity and ground cover and landscape function

In Chapter 4, the relationships between measures of floristic biodiversity (richness, evenness, Shannon–Wiener diversity and turnover) and ground cover and landscape function were examined. Ground cover components (plant, litter, cryptogam, total vegetative and total ground cover) were often positively correlated with understorey species richness, diversity and evenness. This suggests that improved plant biodiversity contributes to greater plant productivity, and vice versa, and that managing for improved ground cover can benefit biodiversity conservation. However, there were few correlations between floristic biodiversity measures and landscape function indices. In this semi-arid rangeland, understorey plant biodiversity was not closely linked to landscape function. The lack of significant relationships between plant biodiversity measures and landscape function indices may be a result of similarity in landscape function indices between sites and the lack of sensitivity in the LFA to pick up changes in landscape function, as has been found in other studies (Munro *et al.* 2012).

6.2.5 Patterns of biodiversity and ground cover along grazing intensity gradients

In Chapter 5, the patterns of floristic biodiversity and ground cover were examined with distance from water on an alternatively grazed property. Total plant species richness, Shannon–Wiener diversity, and plant functional richness, diversity and dispersion increased with distance from water under alternative grazing management, while the proportion and frequency of perennial species decreased. Increased grazing intensity close to water points (Lange 1969) increases soil disturbance and selection favours species tolerant of grazing, while grazing-intolerant and sensitive species are lost from the species pool, thus reducing overall biodiversity close to water (Milchunas *et al.* 1988; Díaz *et al.* 2007). However, no other differences in functional trait groups (e.g. annuals, annual and perennial grasses, annual and perennial forbs, palatable species, tall species, leaf area or seed length) were related to distance from water (i.e. the grazing intensity gradient). The conservative rotational grazing regime in the present study may have confined the negative effects of grazing to the sacrifice zone close to water (Müller *et al.* 2007; Teague *et al.* 2008), as well as reducing the impact of grazing intensity by allowing soil and vegetation to recover between grazing events (Chillo and Ojeda 2014).

No significant differences in density of woody shrub species were observed with increasing grazing intensity in Chapter 5, contrasting with suggestions that increasing grazing pressure results in shrub encroachment (Archer *et al.* 1995; Moleele and Perkins 1998; Van Auken 2000, 2009; Local Land Services 2014). Grazing by a commercial goat herd was the dominant livestock enterprise in the 10 years prior to this study. Goats consume a significant portion of their diets from shrubs or trees, and browsing by goats may have suppressed woody shrub increase in this instance (Wilson *et al.* 1975; Harrington 1979; Dawson and Ellis 1996). This indicates that when managed with regular periods of rest, livestock production can maintain vegetation structure. Total ground cover increased with distance to water as a result of increased cryptogam and litter cover. This indicates that even under alternative grazing management, livestock can significantly alter soil surface attributes at high grazing intensity.

6.2.6 Biodiversity and landscape function response to grazing management on contrasting soil types

In Chapters 3 and 4, the response of biodiversity measures and landscape function indices to contrasting grazing management regimes was compared on sand and clay soils. Significant differences in soil carbon, nitrogen, bulk density, landscape function indices and plant biodiversity measures were observed between sand and clay soil communities. Clay soils had greater organic carbon and organic nitrogen and lower bulk density than sandy sites. Soil stability, nutrient cycling and landscape organisation indices were also greater on clay than sand, and average interpatch length was shorter on clay soils. Clay soils had greater vegetative cover than sand soils, while sand soils had greater cryptogam cover. Floristic biodiversity measures (species richness, evenness, diversity, turnover) were significantly greater on sand than clay soils at the plot (100 m) and site (1000 m) scales, though there was no difference in species richness at the smallest scale (1 m). Despite the common perception that heavy clay soil communities are more resilient to disturbance than communities on sandy soils (Harrington *et al.* 1984; Lewis *et al.* 2009b), we found no difference between sand and clay soils in the response of plant biodiversity measures, total ground cover, landscape function, total organic carbon, total organic nitrogen or bulk density to alternative or traditional grazing management or conservation management. This result suggests that alternative grazing management may provide a sustainable option for biodiversity conservation on commercial livestock grazing properties across a wide range of rangeland communities in western NSW.

6.3 Contribution to scientific theory and practice

This research has contributed to scientific theory and practice by: (1) synthesising current knowledge of the effects of SRG management, compared to continuous grazing management and grazing exclusion on key ecological and animal production variables; (2) highlighting research gaps in the literature and the lack of integration between ecological and animal production research; (3) increasing knowledge of biodiversity and landscape function effects of grazing management incorporating planned rest in semi-arid rangelands, on contrasting soil types; (4) contributing new knowledge about relationships between landscape function and floristic biodiversity on sand and clay soils; and (5) highlighting the importance of alternative grazing management to improve biodiversity and ground cover compared to traditional grazing practices.

Previous reviews comparing grazing incorporating periods of rest with continuous grazing management systems have predominantly focussed on animal production (e.g. (Heady 1961; Holechek *et al.* 2000; Briske *et al.* 2008), with few reviews investigating the effects on biodiversity or taking a holistic perspective. This research provided a detailed synthesis of existing scientific literature, summarising trends in relation to both ecological and animal production outcomes across different geographic and climatic regions. Lack of differences in plant richness, diversity, animal weight gain or production per unit area are consistent with previous reviews comparing continuous or season-long grazing practices with SRG (O'Reagain and Turner 1992; Holechek *et* *al.* 2000; Briske *et al.* 2008). However, these reviews have generally not considered the effects of different types of SRG system. The results of our review indicate that seasonal SRG systems, where an otherwise continuously grazed area is rested for part of the grazing season often to achieve a strategic outcome, increase diversity and increasing rest relative to grazing time increases animal weight gain and production per unit area, justifying further research into SRG. Our review also outlined current knowledge gaps in relation to ecological and animal production effects of SRG and highlighted associated trade-offs, notably a lack of knowledge of the effects of SRG on (1) fauna, (2) management of SRG in the tropics and (3) of different types of SRG management, for example different intensities, timing, frequency and length of rest periods and species of livestock. Results of this study should guide future research and management of SRG systems for both conservation and production.

While there have been numerous calls for greater integration and collaboration between ecological research and animal production research in recent decades (Jackson and Piper 1989; Fuhlendorf and Engle 2001; Watkinson and Ormerod 2001; Dorrough *et al.* 2004; Vavra 2005; Fischer *et al.* 2006; Metera *et al.* 2010), no previous reviews of grazing management systems have considered both biodiversity and animal production effects simultaneously. This research addressed this gap in the literature, and highlights the need to improve communication and collaboration between ecological and agricultural production researchers in order to improve integration of ecological and animal production outcomes in grazing lands.

This study also contributed to the knowledge of the response of floristic biodiversity and landscape function to alternative grazing management in semi-arid rangelands. Previous research into grazing management, and in particular grazing management that incorporates rest, has been predominantly focussed in temperate regions and has not generally utilised ungrazed areas managed for nature conservation as contrasts to understand the trade-offs between different management approaches. No research has previously compared alternative grazing management to areas managed for conservation in the NSW semi-arid rangelands. Results of this research address these literature gaps and build on existing knowledge of the effects of alternative grazing practices for improved biodiversity conservation.

Theoretical models describing the effect of grazing in semi-arid rangelands have been generally based on the effects of varying grazing intensity, rather than different grazing management systems. However, the results of this study provide some support for theoretical models of grazing intensity. The significant influence of season and rainfall in ordination analysis compared to grazing treatment, and the significant correlations between previous rainfall and understorey floristic biodiversity measures provide support for the non-equilibrium theorem. These findings are similar to those of Lunt *et al.* (2007b); Lewis *et al.* (2009b); Fensham *et al.* (2010); Fensham *et al.* (2014). The lack of difference in plant species richness and diversity between alternatively grazed areas and areas managed for conservation, and lower richness and diversity in traditionally grazed areas and at high grazing intensities (close to water) also supports the theoretical model of Milchunas *et al.* (1988), which predicted low levels of grazing to increase diversity in semi-arid rangelands with a short evolutionary history of livestock grazing, but that further increases in grazing intensity would reduce diversity. This study did not observe an increase in plant richness or diversity under alternative grazing management. Milchunas *et al.* (1988) predicted an increase in plant biodiversity under low livestock grazing intensities in semi-arid regions with a short evolutionary history of ungulate grazing. In this study the grazing intensity under alternative grazing management may not have been low enough to achieve an increase in diversity. Functional diversity is an important indicator of ecosystem function and may highlight otherwise undetected differences (Cadotte *et al.* 2011). While the effects of alternative grazing management on functional diversity have been examined overseas (Chillo *et al.* 2017), no previous research in the Australian rangelands has examined this. Significant differences in functional diversity measures along the grazing intensity gradient in this study highlight the importance of the consideration of these measures in future research investigating the effects of grazing management.

While ecosystems on sandy soils in south-eastern Australian rangelands are considered to be less resilient to grazing and disturbance than heavy-clay communities (Harrington *et al.* 1984; Lewis *et al.* 2009b), little research has considered the effect of alternative grazing management on biodiversity and landscape function across contrasting soil types. Arid and semi-arid rangelands in south-eastern Australia are spatially heterogeneous with multiple soil and vegetation types (Walker 1991; Isbell 2016). It is important to understand the response of management strategies across different landscapes in order to identify areas more vulnerable than others to the impacts of grazing and to tailor management strategies appropriate to different rangeland types. This research goes some way towards achieving this at a broad scale in the NSW semi-arid rangelands. In contrast to previous suggestions, we found no difference in the response of heavy-clay and sandy-loam soil and vegetation communities to different grazing strategies, indicating that from a biodiversity conservation and landscape function perspective, similar management strategies may be appropriate for both soil types in western NSW rangelands.

Relationships between floristic biodiversity measures and landscape function indices have not previously been studied in detail. Understanding these relationships provides greater insight into the mechanisms behind floristic biodiversity and landscape function responses in semi-arid rangelands, and the potential to use one as an indicator of another. Previous studies have suggested that more functional landscapes have greater species biodiversity (Ludwig *et al.* 2004) and that greater biodiversity can contribute to landscape multifunctionality of landscapes (Maestre *et al.* 2012; Tilman *et al.* 2012, 2014; Pasari *et al.* 2013). In contrast to these expectations, we did not find many significant correlations between landscape function indices and floristic biodiversity measures, relative to the number of relationships between ground cover components and floristic biodiversity measures. The few relationships between LFA indices and plant biodiversity measures were not consistent across sand and clay soil types. This indicates that measures of floristic biodiversity and landscape function are not closely linked in western NSW semi-arid rangelands, and that LFA may provide an alternative measure of landscape change associated with different grazing management approaches that would be undetected following vegetation surveys alone.

The results of this study have demonstrated the important role that grazing management (in particular, grazing management incorporating frequent long periods of rest from grazing) can play in improving biodiversity, ground cover and to some extent, landscape function in semi-arid rangelands managed for livestock production, relative to traditional continuous grazing. These findings are consistent with recent studies (Teague *et al.* 2011; Deng *et al.* 2014; Chillo *et al.* 2015; Read *et al.* 2016; Sanjari *et al.* 2016). The research provides support for utilising alternative grazing management strategies to improve conservation outcomes in the rangelands, and for closer alignment of livestock and conservation outcomes in semi-arid rangelands. The findings of this research may also be beneficial to guide policy surrounding livestock enterprises in Australian rangelands, highlighting the benefits of incorporating periods of rest in grazing management to improve landscape and biodiversity conservation outcomes.

6.4 Research limitations

The research reported in this thesis was constrained by time. Greater replication of surveys across different seasonal conditions and different soil and vegetation types would have ensured that conclusions were relevant to more community types and enabled better-informed management recommendations to accommodate for natural climatic variation. Locating properties to achieve sufficient replication of grazing the different management strategies was an additional limitation. Subdivisional fencing for rotational grazing management requires significant economic investment in terms of infrastructure and management changes. Therefore, adoption rates of alternative grazing management in western NSW are low and finding suitable grazing contrasts in adjacent areas or in similar land systems and vegetation types was a challenge. No other sites employing alternative grazing practices adjacent to ungrazed areas managed for conservation were available in the study region, and therefore it was not possible to increase replication of these contrasts.

Grazing by feral and native herbivores is difficult to control across large paddocks in heterogeneous landscapes, and legacy effects of historical grazing management were not accounted for. Areas managed for conservation often had evidence of grazing impact, which may have masked differences between livestock grazing strategies and the benefits of grazing exclusion. Despite this, areas selected as ‘ungrazed’ treatments were representative of current conservation management within western NSW, where unmanaged feral goats have access to public reserves that predominantly do not have goat-proof boundary fencing and where goats have to be periodically removed from reserves (Ballard *et al.* 2011). Exclusion of all feral herbivores is costly and near-impossible to achieve in this region, and therefore these study sites provided suitable controls.

The meta-analysis comparing strategic-rest grazing with continuous grazing and ungrazed systems was limited by the data provided in individual studies. Many studies did not report mean values, and were therefore unable to be included in the meta-analyses. In addition, many studies did not provide measures of variance. Although we were able to impute missing variances, greater accuracy would have been possible if measures of variance were available for all studies. Interpretation of results from the meta-analyses was also limited by confounding stocking rates between grazing contrasts, and the large amount of unexplained variation which indicated that other factors influenced the results. As the systematic review was restricted to studies published in English in Scopus, and to specific search terms, not all studies of strategic-rest grazing were included in our systematic review and meta-analyses. However, it is considered that the subset of literature examined is likely to be a representative sample of all literature published on this topic.

6.5 Management recommendations

It is important to retain areas that are ungrazed by livestock in landscapes for the conservation of grazing-sensitive species and ecosystems at a regional scale (Margules and Pressey 2000), either through grazing exclusion or remoteness from water (Landsberg *et al.* 1997a; Fensham and Fairfax 2008). Despite the benefits of alternative grazing management compared to continuous grazing management, the results support the retention of areas excluded from livestock grazing for the protection of rare and grazing-sensitive species. Maintenance of areas ungrazed by livestock will also assist in increasing landscape heterogeneity, which is important to increase biodiversity at landscape scales (Fuhlendorf and Engle 2001; Fischer *et al.* 2006). Provision of financial incentives and support would increase establishment of grazing exclusion areas on commercial livestock grazing properties, especially for vulnerable areas of the landscape and to compensate for lost production. In addition, maintenance and upgrading of boundary fences around nature conservation areas and fencing of water points in reserves is necessary to achieve total grazing exclusion and water-remoteness in areas where feral herbivores have unrestricted access (Fensham and Fairfax 2008).

Exclusion of livestock from all rangelands is not a viable option in a socio-economic sense, and may not be necessary to achieve biodiversity conservation in arid and semi-arid rangelands at a regional scale (Fischer *et al.* 2011; Neilly *et al.* 2016). This study has shown benefits from incorporating periods of rest from livestock grazing for floristic biodiversity, ground cover and landscape function in semi-arid rangelands, relative to continuous grazing management (Chapters 3 and 4). Management strategies for improving biodiversity and landscape function in livestock production areas in semi-arid rangelands should therefore focus on greater adoption of alternative grazing management incorporating periods of rest from grazing, along with careful management of grazing intensity, especially in response to seasonal conditions. Consideration of economic factors is important in the development and implementation of alternative grazing strategies, as the greater amount of subdivisional fencing and water infrastructure associated with alternative grazing systems has significant monetary costs. In order to implement and increase adoption of rotational grazing systems, provision of financial support to erect subdivisional fencing and new water points would be valuable, along with restricting access to water points and erecting goat-proof fencing to ensure land is completely rested, allowing soil and vegetation to recover between grazing events. In addition, financial support for educational courses and the development of extension material to inform landholders of the importance of alternative grazing management for improving biodiversity conservation and landscape function would be useful in areas managed for commercial livestock production in western NSW. Increased fencing may also pose additional ecological consequences, such as disrupting movements and gene-flow of native wildlife; blocking access to water; altering predator–prey relationships, and increasing wildlife mortality as a result of collisions with fences (Pople *et al*. 2000; Haywood and Kerley 2009; Bradby *et al.* 2014).

Effective grazing management should aim to provide sufficient rest periods to allow recovery and persistence of grazing-sensitive species and maintain ground cover and landscape function. Increasing length of rest to graze time was associated with a decline in species richness (Chapter 2), and therefore careful management of timing is important. Rest from grazing during certain stages of plant development or during certain seasons (seasonal SRG management) can benefit plant diversity (Chapter 2). Identification and close monitoring of grazing-sensitive indicator species (Caro and O'doherty 1999) and adjustment of grazing intensity and the length of graze and rest periods is necessary to increase the abundance of these species. Identification and protection of vulnerable areas of the landscape is also necessary to develop appropriate grazing management strategies (or grazing exclusion) to promote the conservation of biodiversity and landscape function in these areas. Understanding and monitoring of thresholds at which negative effects of grazing are likely to occur – such as thresholds of grazing intensity or leading into a long dry period – are necessary to inform management decisions and prevent degradation under commercial grazing. In Chapter 5, increasing grazing intensity had negative effects on plant biodiversity measures and ground cover. Conservative grazing intensities are recommended to minimise this degradation.

6.6 Future research

Growing demand for livestock products will place greater pressure on rangelands to increase production (FAO 2011), thus increasing the potential for degradation. Degradation of biodiversity and landscape function may, in turn, result in reduced production (Fischer *et al.* 2006; Reynolds *et al.* 2007; Tilman *et al.* 2012). Therefore, there is significant incentive to increase understanding of and improve grazing management to maintain and improve biodiversity and landscape function.

The results of the systematic review (Chapter 2) highlighted a lack of research on alternative grazing management in tropical and arid regions, with the majority of research concentrated in the temperate zone. In addition, less research had been undertaken into alternative management strategies in developing countries. Additional research is necessary in these regions to improve sustainability and biodiversity conservation outcomes in grazing lands globally. The review also revealed a scarcity of knowledge of the effects of alternative grazing management on fauna. Effects of grazing management on fauna are likely to be different to those on vegetation (Kruess and Tscharntke 2002; Zhu *et al.* 2012; van Klink *et al.* 2015), and therefore the response of vegetation to grazing management is not necessarily a reliable indicator of the response of fauna. More research is necessary to understand grazing management variables (including differences in timing, frequency and duration of rest and graze periods, livestock type and grazing intensity) that have the potential to achieve better integration of ecological and production outcomes in grazing lands. Future research must also focus on greater collaboration between agricultural production scientists and ecologists, in order to improve both ecological and production outcomes, to understand trade-offs between these, and to increase adoption of more sustainable grazing practices by landholders.

Undertaking this study in different seasons and on contrasting soil types afforded greater generality and applicability of the results. Additional research is necessary to increase our understanding of the situations (e.g. soil and vegetation types, seasonal conditions) in which grazing is not appropriate for achieving biodiversity and landscape conservation, along with the management strategies that are most appropriate to achieve this (e.g. the timing, duration, and frequency of grazing and the type of livestock). Variability in biodiversity and landscape function responses to differing grazing management strategies and across different clusters and soil types (Chapters 3 and 4) indicates a need to undertake more research to ensure results and management recommendations are applicable to different soil and vegetation types. By repeating sampling in two different seasons, this study showed that season and rainfall were significant drivers of plant composition and biodiversity measures, and that effects of grazing management on composition and biodiversity differ with season. Many studies of grazing management in semi-arid rangelands utilise a one-off measurement only. It would be beneficial to replicate studies of grazing management across multiple years and seasons (including dry and wet seasons) to capture a complete understanding of grazing management impacts.

This research project analysed the response of understorey floristic biodiversity to alternative grazing management at differing scales, and found differences in response depending on the scale of measurement. Scale is widely recognised as influencing floristic biodiversity (Stohlgren *et al.* 1999; Landsberg *et al.* 2002; Adler *et al.* 2005; Kohyani *et al.* 2008; Li *et al.* 2015). However, little research in semi-arid rangelands has examined the response of floristic biodiversity measures to grazing management strategies at different scales. Studies utilising small-scale study plots may not detect negative effects of grazing on plant species biodiversity or composition at landscape scales, therefore it is important that the scale of the study is considered in interpreting results. To improve understanding of whole-ecosystem responses to grazing management, it is important that this matter is addressed.

This research investigated the effect of alternative grazing management on commercial properties and publically managed reserves. Although locating study sites on commercial properties and public reserves brought some additional variability into results compared to experimentally contrived grazing treatments, the grazing treatments reflected real-world conditions. Much of the previous research into grazing management has been undertaken as experiments in small plots or paddocks, and this is recognised as a key reason for differences observed in literature between rotational and continuous grazing systems. In small plots, patch-grazing dynamics are reduced in continuous grazing systems and grazing patterns become more similar to those of rotational systems (Teague *et al.* 2013). Utilising commercial grazing properties is important to understand effects at a paddock and landscape scale, and to ensure results are reliable and applicable under natural conditions.

The meta-analysis reported in Chapter 2 highlighted the significant effect of different types of strategic-rest grazing management on ecological and animal production response variables, in particular, the length of time of rest from grazing in relation to the length of time an area is grazed. Greater research into different types of adaptive rotational grazing management, at multiple stocking rates, would be beneficial to determine optimum management strategies for the ecological and socio-economic sustainability of livestock grazing enterprises. In particular, a focus on timing and duration of rest periods to achieve optimum biodiversity and landscape function outcomes in semi-arid NSW would be beneficial (Müller *et al.* 2007). In addition, climate-change is expected to alter ecosystem dynamics and vulnerability in response to grazing (Heller and Zavaleta 2009). Modelling the response of biodiversity and landscape function to different grazing management strategies under different climate change scenarios would assist in planning for the future. Research into the effects of different grazing strategies in conjunction with other rangeland management strategies (e.g. total grazing exclusion fencing) may also identify potential methods of improving biodiversity conservation in arid and semi-arid rangelands.

Historical management practices and degradation can result in legacy effects (Monger *et al.* 2015). Arid and semi-arid rangelands require long periods of time to recover from degradation events (Meissner and Facelli 1999; Daryanto and Eldridge 2010; Seymour *et al.* 2010; Fensham *et al.* 2011), and often require significant rainfall events to recover (Stafford Smith *et al.* 2007). Studies have shown that recovery of arid and semi-arid rangelands can take over 20 years (Hall *et al.* 1964; Fuhlendorf *et al.* 2001; Valone *et al.* 2002; Seymour *et al.* 2010). Some sites utilised in this study had only been under current management for five years, and therefore may not have had sufficient time to respond to new management practices (Teague *et al.* 2013). Many previous studies investigating effects of alternative grazing management have also not studied the effects of grazing management strategies beyond five years since the implementation of current management (Teague *et al.* 2013). Long-term studies and measurements of biodiversity and landscape function in response to changed grazing management may reveal additional benefits of grazing exclusion or alternative grazing management relative to traditional grazing management strategies, and should be established as a priority.

Finally, additional research into the relationships between floristic biodiversity and landscape function utilising a broader range of sites on the landscape function continuum may be useful to develop a better understanding of these relationships. The sites in this study were relatively similar with regards to landscape function and the LFA methodology may not have been sensitive enough to capture potential differences in landscape function between these sites.

6.7 Conclusions

Biodiversity conservation and maintenance of landscape function in agricultural landscapes is essential for the maintenance of ecosystem service provision and the production of food and fibre in semi-arid rangelands. Better understanding of the effects of different grazing strategies on biodiversity and landscape function is necessary to inform management and enhance biodiversity and landscape conservation. This research has provided important new insights into the effects of alternative grazing management on biodiversity and landscape function in the NSW semi-arid rangelands, across different scales and seasons and on contrasting soil types. It has highlighted the potential for alternative methods of grazing management to improve biodiversity and landscape conservation outcomes outside the public reserve system. Greater collaboration between ecologists and animal production scientists is needed to improve understanding of the ecological and agricultural trade-offs associated with different grazing management systems and in developing and extending sustainable commercial grazing management strategies to improve biodiversity conservation and landscape function in the semi-arid rangelands.

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Westoby, M., B. Walker, and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* **42**:266-274.

Williams, W. J., D. J. Eldridge, and B. M. Alchin. 2008. Grazing and drought reduce cyanobacterial soil crusts in an Australian *Acacia* woodland. *Journal of Arid Environments* **72**:1064-1075.

Williamson, J. A., G. E. Aiken, E. S. Flynn, and M. Barrett. 2016. Animal and pasture responses to grazing management of chemically suppressed tall fescue in mixed pastures. *Crop Science* **56**:2861-2869.

Wilson, A., J. Leigh, N. Hindley, and W. Mulham. 1975. Comparison of the diets of goats and sheep on a *Casuarina cristata – Heterodendrum oleifolium* woodland community in western New South Wales. *Australian Journal of Experimental Agriculture* **15**:45-53.

Winfree, R., I. Bartomeus, and D. P. Cariveau. 2011. Native pollinators in anthropogenic habitats. *Annual Review of Ecology, Evolution, and Systematics* **42**:1-22.

Witt, G. B., M. V. Noël, M. I. Bird, R. J. S. Beeton, and N. W. Menzies. 2011. Carbon sequestration and biodiversity restoration potential of semi-arid mulga lands of Australia interpreted from long-term grazing exclosures. *Agriculture, Ecosystems & Environment* **141**:108-118.

Woinarski, J., and A. Fisher. 2003. Conservation and the maintenance of biodiversity in the rangelands. *The Rangeland Journal* **25**:157-171.

Woinarski, J. C. Z., S. Legge, J. A. Fitzsimons, B. J. Traill, A. A. Burbidge, A. Fisher, R. S. C. Firth, I. J. Gordon, A. D. Griffiths, C. N. Johnson, N. L. McKenzie, C. Palmer, I. Radford, B. Rankmore, E. G. Ritchie, S. Ward, and M. Ziembicki. 2011. The disappearing mammal fauna of northern Australia: context, cause, and response. *Conservation Letters* **4**:192-201.

Yates, C. J., D. A. Norton, and R. J. Hobbs. 2000. Grazing effects on plant cover, soil and microclimate in fragmented woodlands in south-western Australia: implications for restoration. *Austral Ecology* **25**:36-47.

Zhu, H., D. Wang, L. Wang, Y. Bai, J. Fang, and J. Liu. 2012. The effects of large herbivore grazing on meadow steppe plant and insect diversity. *Journal of Applied Ecology* **49**:1075-1083.

Appendix 1

Table A1.1. Glossary of terms

Term	Definition
Biodiversity	The variety of life at all levels of organisation (Cresswell and Murphy 2017)
Carrying capacity	Number of animals that can be held on a unit of land over a defined time, without deterioration of land (Allen *et al.* 2011)
Composition	In the context of this thesis, floristic species composition refers to the identity of plant species within the defined area
Continuous grazing	Livestock have unrestricted and uninterrupted access to a unit of land throughout the grazing season or year (Allen *et al.* 2011)
Diversity	In the context of this thesis, floristic species diversity refers to the richness and evenness of different plant species within a defined area (Colwell 2009), as measured by the Shannon–Wiener diversity index or Simpson’s diversity index
Ecosystem service	Benefits provided by ecosystems for people. These include provisioning, regulating, supporting and cultural services. (MA 2005)
Evenness	In the context of this thesis, evenness refers to how equal the abundance of species is within a defined area (Colwell 2009), as measured by Pielou’s evenness index (J’)
Grazing intensity	The cumulative effects of grazing (Holechek *et al.* 1998)
Invasive Native Scrub (INS)	Woody plant species “that are encroaching or regenerating densely following disturbance” (Tighe *et al.* 2009)
Land degradation	Long-term loss of productivity and ecosystem function (Bai *et al.* 2008)
Landscape function	The ability of landscapes to capture, retain and utilise resources such as water and nutrients (Tongway and Ludwig 1997b)
Productivity	The capacity of land to support plant growth and animal production (Reynolds *et al.* 2007)
Rangeland	Areas of native vegetation grazed by livestock, including grasslands, savannas, shrublands, deserts, steppes, tundras, alpine and marsh communities (Allen *et al.* 2011)

Richness	In the context of this thesis, richness refers to the total number of different species present within the defined area (Colwell 2009)
Rotational grazing	Repeated periods of grazing an rest across a land unit, achieved by moving livestock throughout multiple paddocks (Allen *et al.* 2011)
Stocking rate	Number of animals held in a unit of land over a specified time (Allen *et al.* 2011)
Turnover	In the context of this thesis, floristic species turnover refers to the change in species composition between defined areas within the same vegetation community type (also known as pattern diversity; Tuomisto 2010)

Appendix 2

Table A2.1. List of papers included in systematic review. R = plant species richness, D = plant diversity, W = animal weight gain, P = animal production per unit area, I = integrated. Ticks in columns R, C, W and P = paper was included in meta-analyses for response variable, tick in column I = paper was considered integrated.

Reference	**R**	**D**	**W**	**P**	**I**
Aiken, G. 1998. Steer performance and nutritive values for continuously and rotationally stocked bermudagrass sod-seeded with wheat and ryegrass. *Journal of production agriculture* 11:185-190.			✓	✓	
Alemseged, Y., R. Hacker, W. Smith, and G. Melville. 2011. Temporary cropping in semi-arid shrublands increases native perennial grasses. *The Rangeland Journal* 33:67-78.					
Alemseged, Y., D. Kemp, G. King, D. Michalk, and M. Goodacre. 2003. The influence of grazing management on the competitiveness, persistence and productivity of chicory (Cichorium intybus L.). *Australian Journal of Experimental Agriculture* 43:127-133.					
Ali, K., M. K. Gullap, and H. I. Erkovan. 2013. The soil seed bank pattern in highland rangelands of Eastern Anatolian Region of Turkey under different grazing systems. *Turkish Journal Of Field Crops* 18:109-117.					
Allan, B. 1997. Grazing management of oversown tussock country 3. Effects on liveweight and wool growth of Merino wethers. *New Zealand Journal of Agricultural Research* 40:437-447.			✓	✓	
Angell, R. F. 1997. Crested wheatgrass and shrub response to continuous or rotational grazing. *Journal of Range Management*:160-164.					
Arthur, A. D., R. P. Pech, C. Davey, Z. Yanming, and L. Hui. 2008. Livestock grazing, plateau pikas and the conservation of avian biodiversity on the Tibetan plateau. *Biological Conservation* 141:1972-1981.					
Ash, A. J., J. P. Corfield, J. G. McIvor, and T. S. Ksiksi. 2011. Grazing Management in Tropical Savannas: Utilization and Rest Strategies to Manipulate Rangeland Condition. *Rangeland Ecology & Management* 64:223-239.					✓
Avery, A., D. Michalk, R. Thompson, P. Ball, T. Prance, C. Harris, D. FitzGerald, J. Ayres, and B. Orchard. 2000. Effects of sheep grazing management on cocksfoot herbage mass and persistence in temperate environments. *Australian Journal of Experimental Agriculture* 40:185-206.					
Bailey, J., S. Walkden-Brown, and L. Kahn. 2009. Comparison of strategies to provide lambing paddocks of low gastro-intestinal nematode infectivity in a summer rainfall region of Australia. *Veterinary Parasitology* 161:218-231.					
Barnes, D., and C. Dempsey. 1992. Towards optimum grazing management for sheep production on crownvetch (Coronilla varia L.). *Journal of the Grassland Society of Southern Africa* 9:83-89.					
Barthram, G., and S. Grant. 1995. Interactions between variety and the timing of conservation cuts on species balance in Lolium perenne-Trifolium repens swards. *Grass and Forage Science* 50:98-105.					

Reference	**R**	**D**	**W**	**P**	**I**
Beck, P., C. Stewart, M. Sims, M. Gadberry, and J. Jennings. 2016. Effects of stocking rate, forage management, and grazing management on performance and economics of cow–calf production in Southwest Arkansas. *Journal of animal science* 94:3996-4005.			✓		
Bertelsen, B., D. Faulkner, D. Buskirk, and J. Castree. 1993. Beef cattle performance and forage characteristics of continuous, 6-paddock, and 11-paddock grazing systems. *Journal of animal science* 71:1381-1389.			✓	✓	
Beukes, P., and R. Cowling. 2000. Impacts of non-selective grazing on cover, composition, and productivity of Nama-karoo grassy shrubland. *African journal of range and forage science* 17:27-35.					
Biondini, M. E., and L. Manske. 1996. Grazing frequency and ecosystem processes in a northern mixed prairie, USA. *Ecological Applications* 6:239-256.			✓		✓
Birrell, H., A. Bishop, A. Tew, and R. Plowright. 1978. Effect of stocking rate, fodder conservation and grazing management on the performance of wether sheep and pastures in south-west Victoria. 2. Seasonal wool growth rate, liveweight and herbage availability. *Australian Journal of Experimental Agriculture* 18:41-51.					
Bishop, A., and H. Birrell. 1975. Effect of stocking rate, fodder conservation and grazing management on the performance of wether sheep in south-west Victoria. 1. Wool production. *Australian Journal of Experimental Agriculture* 15:173-182.					
Boa, M., S. Thamsborg, A. Kassuku, and H. Bøgh. 2001. Comparison of worm control strategies in grazing sheep in Denmark. *Acta Veterinaria Scandinavica* 42:57.			✓		
Boswell, C., M. Monteath, N. Round-Turner, K. Lewis, and N. Cullen. 1974. Intensive lamb production under continuous and rotational grazing systems. *New Zealand journal of experimental agriculture* 2:403-408.			✓	✓	
Bowman, A., Y. Alemseged, G. Melville, W. Smith, and F. Syrch. 2009. Increasing the perennial grass component of native pastures through grazing management in the 400–600 mm rainfall zone of central western NSW. *The Rangeland Journal* 31:369-376.					
Bozkurt, Y., and I. Kaya. 2011. Effect of two different grazing systems on the performance of beef cattle grazing on hilly rangeland conditions. *Journal of Applied Animal Research* 39:94-96.			✓		
Bransby, D. 1990. Nitrogen fertilization, stocking rate and rotational grazing effects on steers grazing Pennisetum clandestinum. *Journal of the Grassland Society of Southern Africa* 7:261-264.					
Branson, D. H., and G. A. Sword. 2010. An experimental analysis of grasshopper community responses to fire and livestock grazing in a northern mixed-grass prairie. *Environmental entomology* 39:1441-1446.					
Brink, G., and D. Rowe. 1997. White clover clone response to alternative defoliation methods. *Crop science* 37:1832-1835.					
Broadbent, P. 1964a. THE USE OF GRAZING CONTROL FOR INTENSIVE FAT- LAMB PRODUCTION: II. The effect of stocking rates and grazing systems with a fixed severity of grazing on the output of fat lamb per acre. *Grass and Forage Science* 19:218-223.			✓	✓	
Broadbent, P. 1964b. THE USE OF GRAZING CONTROL FOR INTENSIVE FAT- LAMB PRODUCTION: III. the influence of systems of grazing, severity of grazing and stocking rates *Grass and Forage Science* 19:218-223.			✓		

Reference	R	D	W	P	I
Brown, T. 1976. The effect of stocking rate and deferred autumn grazing of pasture on liveweight and wool production of Merino wethers in a Mediterranean-type climate. *Australian Journal of Experimental Agriculture* 16:189-196.					
Bryant, H., R. Blaser, R. Hammes, and W. Hardison. 1961. Comparison of Continuous and Rotational Grazing of Three Forage Mixtures by Dairy Cows1. *Journal of Dairy Science* 44:1742-1750.				✓	
Bungenstab, E., A. Pereira, J. Lin, J. Holliman, and R. Muntifering. 2011. Productivity, utilization, and nutritive quality of dallisgrass () as influenced by stocking density and rest period under continuous or rotational stocking. *Journal of animal science* 89:571-580.			✓	✓	
Burke, J., J. Miller, and T. Terrill. 2009. Impact of rotational grazing on management of gastrointestinal nematodes in weaned lambs. *Veterinary Parasitology* 163:67-72.			✓		
Burns, J., and D. Fisher. 2010. Eastern gamagrass management for pasture in the mid-Atlantic region: II. diet and canopy characteristics, and stand persistence. *Agronomy journal* 102:179-186.			✓	✓	
Burns, J., and D. Fisher. 2011. Stocking strategies as related to animal and pasture productivity of endophyte-free tall fescue. *Crop science* 51:2868-2877.			✓	✓	
Burns, J., and D. Fisher. 2012. Grazing Management of Flaccidgrass Pastures in Autumn. *Crop science* 52:2392-2402.					
Carmona, C. P., F. M. Azcárate, E. Oteros-Rozas, J. A. González, and B. Peco. 2013. Assessing the effects of seasonal grazing on holm oak regeneration: Implications for the conservation of Mediterranean dehesas. *Biological Conservation* 159:240-247.					
Cassels, D. M., R. L. Gillen, F. T. McCollum, K. W. Tate, and M. E. Hodges. 1995. Effects of grazing management on standing crop dynamics in tallgrass prairie. *Journal of Range Management*:81-84.					✓
Chapman, D., D. Clark, C. Land, and N. Dymock. 1984. Leaf and tiller or stolon death of Lolium perenne, Agrostis spp., and Trifolium repens in set-stocked and rotationally grazed hill pastures. *New Zealand Journal of Agricultural Research* 27:303-312.					
Chapman, E. W., and C. A. Ribic. 2002. The impact of buffer strips and stream-side grazing on small mammals in southwestern Wisconsin. *Agriculture, ecosystems & environment* 88:49-59.					
Chen, J., Y. Yamamura, Y. Hori, M. Shiyomi, T. Yasuda, H.-k. Zhou, Y.-n. Li, and Y.-h. Tang. 2008. Small-scale species richness and its spatial variation in an alpine meadow on the Qinghai-Tibet Plateau. *Ecological research* 23:657-663.	✓				
Cheng, J., J. Cheng, T. Hu, H. Shao, and J. Zhang. 2012. Dynamic changes of Stipa bungeana steppe species diversity as better indicators for soil quality and sustainable utilization mode in Yunwu Mountain Nature Reserve, Ningxia, China. *CLEAN–Soil, Air, Water* 40:127-133.					
Chillo, V., R. A. Ojeda, M. Anand, and J. F. Reynolds. 2015. A novel approach to assess livestock management effects on biodiversity of drylands. *Ecological Indicators* 50:69-78.					
Coppa, M., I. Verdier-Metz, A. Ferlay, P. Pradel, R. Didienne, A. Farruggia, M. Montel, and B. Martin. 2011. Effect of different grazing systems on upland pastures compared with hay diet on cheese sensory					

Reference	R	D	W	P	I
properties evaluated at different ripening times. *International dairy journal* 21:815-822.					
Cui, S., X. Zhu, S. Wang, Z. Zhang, B. Xu, C. Luo, L. Zhao, and X. Zhao. 2014. Effects of seasonal grazing on soil respiration in alpine meadow on the Tibetan plateau. *Soil use and management* 30:435-443.					
Culvenor, R. 2000. Comparison of four phalaris cultivars under grazing: drought survival and subsequent performance under rotational grazing versus set stocking. *Australian Journal of Experimental Agriculture* 40:1047-1058.			✓		
Danley, R. F., R. K. Murphy, and E. M. Madden. 2004. Species diversity and habitat of grassland passerines during grazing of a prescribe-burned, mixed-grass prairie. *Western North American Naturalist*:72-77.					
Davies, H. L., and I. Southey. 2001. Effects of grazing management and stocking rate on pasture production, ewe liveweight, ewe fertility and lamb growth on subterranean clover-based pasture in Western Australia. *Australian Journal of Experimental Agriculture* 41:161-168.			✓		
Davies, K., J. Bates, and C. Boyd. 2016. Effects of Intermediate-Term Grazing Rest on Sagebrush Communities with Depleted Understories: Evidence of a Threshold. *Rangeland Ecology & Management* 69:173-178.	✓	✓			
Davis, S. C., L. A. Burkle, W. F. Cross, and K. A. Cutting. 2014. The effects of timing of grazing on plant and arthropod communities in high-elevation grasslands. *PloS one* 9:e110460.					
De Bruijn, S. L., and E. W. Bork. 2006. Biological control of Canada thistle in temperate pastures using high density rotational cattle grazing. *Biological Control* 36:305-315.					✓
Derner, J. D., and R. H. Hart. 2007a. Grazing-induced modifications to peak standing crop in northern mixed-grass prairie. *Rangeland Ecology & Management* 60:270-276.					
Derner, J. D., and R. H. Hart. 2007b. Livestock and vegetation responses to rotational grazing in short-grass steppe. *Western North American Naturalist* 67:359-367.			✓		✓
Derner, J. D., R. H. Hart, M. A. Smith, and J. W. Waggoner. 2008. Long-term cattle gain responses to stocking rate and grazing systems in northern mixed-grass prairie. *Livestock Science* 117:60-69.			✓		
Deuschle, J., and E. Gluck. 2008. Similarity of ground beetle communities of extensively managed orchards depending on management regimes. *Bulletin of Insectology* 61:73.					
Di Grigoli, A., M. Todaro, G. Di Miceli, V. Genna, G. Tornambè, M. L. Alicata, D. Giambalvo, and A. Bonanno. 2012. Effects of continuous and rotational grazing of different forage species on ewe milk production. *Small ruminant research* 106:S29-S36.					
Dimander, S., J. Höglund, E. Spörndly, and P. Waller. 2000. The impact of internal parasites on the productivity of young cattle organically reared on semi-natural pastures in Sweden. *Veterinary Parasitology* 90:271-284.			✓		
Donkor, N., J. Gedir, R. Hudson, E. Bork, D. Chanasyk, and M. Naeth. 2002. Impacts of grazing systems on soil compaction and pasture production in Alberta. *Canadian Journal of soil science* 82:1-8.					✓

Reference	**R**	**D**	**W**	**P**	**I**
Donnelly, J., A. Axelsen, and F. Morley. 1970. Effect of flock size and grazing management on sheep production. *Australian Journal of Experimental Agriculture* 10:271-278.			✓		
Dormaar, J. F., B. W. Adams, and W. D. Willms. 1997. Impacts of rotational grazing on mixed prairie soils and vegetation. *Journal of Range Management*:647-651.					
Dorrough, J., S. McIntyre, G. Brown, J. Stol, G. Barrett, and A. Brown. 2012. Differential responses of plants, reptiles and birds to grazing management, fertilizer and tree clearing. *Austral Ecology* 37:569-582.	✓				
Dorrough, J., and C. Moxham. 2005. Eucalypt establishment in agricultural landscapes and implications for landscape-scale restoration. *Biological Conservation* 123:55-66.					
Dorrough, J., C. Moxham, V. Turner, and G. Sutter. 2006. Soil phosphorus and tree cover modify the effects of livestock grazing on plant species richness in Australian grassy woodland. *Biological Conservation* 130:394-405.					
Dostálek, J., and T. Frantík. 2012. The impact of different grazing periods in dry grasslands on the expansive grass Arrhenatherum elatius L. and on woody species. *Environmental management* 49:855-861.					
Dowling, P., D. Kemp, P. Ball, C. Langford, D. Michalk, G. Millar, P. Simpson, and R. Thompson. 2005. Effect of continuous and time-control grazing on grassland components in south-eastern Australia. *Australian Journal of Experimental Agriculture* 45:369-382.					✓
Dowling, P., D. Michalk, D. Kemp, G. Millar, S. Priest, W. M. King, I. Packer, P. Holst, and J. Tarleton. 2006. Sustainable grazing systems for the Central Tablelands of New South Wales. 2. Effect of pasture type and grazing management on pasture productivity and composition. *Australian Journal of Experimental Agriculture* 46:457-469.					
Earl, J., and C. Jones. 1996. The need for a new approach to grazing management-is cell grazing the answer? *The Rangeland Journal* 18:327-350.					
Eaton, D. P., S. A. Santos, M. d. C. A. Santos, J. V. B. Lima, and A. Keuroghlian. 2011. Rotational grazing of native pasturelands in the Pantanal: an effective conservation tool. *Tropical Conservation Science* 4:39-52.					✓
Farruggia, A., B. Dumont, A. Scohier, T. Leroy, P. Pradel, and J. P. Garel. 2012. An alternative rotational stocking management designed to favour butterflies in permanent grasslands. *Grass and Forage Science* 67:136-149.			✓		✓
Fernández-Lugo, S., L. de Nascimento, M. Mellado, and J. Arévalo. 2011. Grazing effects on species richness depends on scale: a 5-year study in Tenerife pastures (Canary Islands). *Plant ecology* 212:423-432.	✓				
Fischer, J., J. Stott, A. Zerger, G. Warren, K. Sherren, and R. I. Forrester. 2009. Reversing a tree regeneration crisis in an endangered ecoregion. *Proceedings of the National Academy of Sciences* 106:10386-10391.					
Fonderflick, J., A. Besnard, A. Beuret, M. Dalmais, and B. Schatz. 2014. The impact of grazing management on Orthoptera abundance varies over the season in Mediterranean steppe-like grassland. *Acta oecologica* 60:7-16.					

Reference	R	D	W	P	I
Frame, H., L. Warn, and G. McLarty. 2002. Effects of grazing system and phosphorus application on pasture quality. *Wool Technology and Sheep Breeding* 50.					
Fritts, S. R., B. A. Grisham, D. A. Haukos, C. W. Boal, M. A. Patten, D. H. Wolfe, C. E. Dixon, R. D. Cox, and W. R. Heck. 2016. Long- term lesser prairie- chicken nest ecology in response to grassland management. *The Journal of Wildlife Management* 80:527-539.					
Fujita, N., N. Amartuvshin, Y. Yamada, K. Matsui, S. Sakai, and N. Yamamura. 2009. Positive and negative effects of livestock grazing on plant diversity of Mongolian nomadic pasturelands along a slope with soil moisture gradient. *Grassland science* 55:126-134.	✓	✓			
Gao, Y., M. Giese, X. Han, D. Wang, Z. Zhou, H. Brueck, S. Lin, and F. Taube. 2009. Land use and drought interactively affect interspecific competition and species diversity at the local scale in a semiarid steppe ecosystem. *Ecological research* 24:627-635.	✓	✓			
Gebeyehu, S., and M. J. Samways. 2003. Responses of grasshopper assemblages to long-term grazing management in a semi-arid African savanna. *Agriculture, ecosystems & environment* 95:613-622.					
Gillen, R. L., F. T. McCollum III, K. W. Tate, and M. E. Hodges. 1998. Tallgrass prairie response to grazing system and stocking rate. *Journal of Range Management*:139-146.					✓
Gilley, J. E., B. Patton, P. Nyren, and J. Simanton. 1996. Grazing and haying effects on runoff and erosion from a former conservation reserve program site. *Applied engineering in agriculture* 12:681-684.					
Girard-Cartier, C. B., and G. S. Kleppel. 2015. Grazing as a Control for the Spread of Mile-a-Minute (Persicaria perfoliata) and the Restoration of Biodiversity in Plant Communities in a Lower New York State Parkland. *Ecological Restoration* 33:82-89.	✓				
Glück, E., and J. Deutschle. 2005. Ground beetle distribution of distinct size and feeding type due to grassland management treatments in orchards (Coleoptera: Carabidae). *Entomologia generalis*:39-63.					
Goldberg, J., E. Wildman, J. Pankey, J. Kunkel, D. Howard, and B. Murphy. 1992. The influence of intensively managed rotational grazing, traditional continuous grazing, and confinement housing on bulk tank milk quality and udder health. *Journal of Dairy Science* 75:96-104.					
Goodenough, A. E., and M. H. Sharp. 2016. Managing calcareous grassland for the declining Duke of Burgundy Hamearis lucina butterfly: effects of grazing management on Primula host plants. *Journal of Insect Conservation* 20:1087-1098.					
Graham, J., B. Cullen, G. Lodge, M. Andrew, B. Christy, P. Holst, X. Wang, S. Murphy, and A. Thompson. 2003. SGS Animal Production Theme: effect of grazing system on animal productivity and sustainability across southern Australia. *Australian Journal of Experimental Agriculture* 43:977-991.				✓	
Grings, E., R. Heitschmidt, R. Short, and M. Haferkamp. 2002. Intensive-early stocking for yearling cattle in the Northern Great Plains. *Journal of Range Management*:135-138.					
Guevara, J., C. Stasi, and O. Estevez. 1996. Effect of cattle grazing on range perennial grasses in the Mendoza plain, Argentina. *Journal of Arid Environments* 34:205-213.					✓

Reference	R	D	W	P	I
Gunn, R., J. Doney, R. Agnew, W. Smith, and D. Sim. 1990. The effect of different pasture management strategies during the weaning-to-mating period on reproductive performance of Greyface ewes. *Animal Science* 51:163-171.			✓		
Guretzky, J. A., K. J. Moore, E. C. Brummer, and C. L. Burras. 2005. Species diversity and functional composition of pastures that vary in landscape position and grazing management. *Crop science* 45:282-289.					
Guretzky, J. A., K. J. Moore, C. L. Burras, and E. C. Brummer. 2007. Plant species richness in relation to pasture position, management, and scale. *Agriculture, ecosystems & environment* 122:387-391.	✓				
Gutman, M., Z. Holzer, H. Baram, I. Noy-Meir, and S. No'Am G. 1999. Heavy stocking and early-season deferment of grazing on Mediterranean-type grassland. *Journal of Range Management*:590-599.			✓	✓	
Gutman, M., and Seligman, N.G. 1979. Grazing management of Mediterranean foothill range in the upper Jordan river valley. *Journal of Range Management*:86-92.			✓	✓	
Gutman, M., N. G. Seligman, and I. Noy-Meir. 1990. Herbage production of Mediterranean grassland under seasonal and yearlong grazing systems. *Journal of Range Management*:64-68.			✓	✓	
Habtemicael, M., T. Yayneshet, and A. C. Treydte. 2015. Responses of vegetation and soils to three grazing management regimes in a semi- arid highland mixed crop- livestock system. *African Journal of Ecology* 53:75-82.		✓			
Hafley, J. 1996. Comparison of Marshall and Surrey ryegrass for continuous and rotational grazing. *Journal of animal science* 74:2269-2275.			✓	✓	
Haggar, R. J., W. Holmes, and P. Innes. 1963. WILD WHITE CLOVER SEED PRODUCTION I. The effects of defoliation and fertilizer treatment on flowering and seed yields from ryegrass/white-clover swards. *Grass and Forage Science* 18:97-103.					
Haider, M. S., A. Maclaurin, A. A. Chaudhry, M. Mushtaque, and S. Ullah. 2011. Effect of Grazing Systems on Range Condition in Pabbi Hills Reserve Forest, Kharian, Punjab, Pakistan. *Chilean journal of agricultural research* 71:560.					✓
Hall, T. J., J. G. McIvor, D. J. Reid, P. Jones, N. D. MacLeod, C. K. McDonald, and D. R. Smith. 2014. A comparison of stocking methods for beef production in northern Australia: pasture and soil surface condition responses. *The Rangeland Journal* 36:161-174.					✓
Hammond, A., M. Williams, T. Olson, L. Gasbarre, E. Leighton, and M. Menchaca. 1997. Effect of rotational vs continuous intensive stocking of bahiagrass on performance of Angus cows and calves and interaction with sire type on gastrointestinal nematode burden. *Journal of animal science* 75:2291-2299.					
Hao, J., U. Dickhoefer, L. Lin, K. Müller, T. Glindemann, P. Schönbach, A. Schiborra, C. Wang, and A. Susenbeth. 2013. Effects of rotational and continuous grazing on herbage quality, feed intake and performance of sheep on a semi-arid grassland steppe. *Archives of animal nutrition* 67:62-76.			✓	✓	
Harmoney, K. R., K. J. Moore, E. C. Brummer, C. L. Burras, and J. R. George. 2001. Spatial legume composition and diversity across seeded landscapes. *Agronomy journal* 93:992-1000.					

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Reference	**R**	**D**	**W**	**P**	**I**
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TOTAL	24	15	62	29	40

Table A2.2. Proportion of papers with missing data for each response variable dataset. 'n' = the number of studies, and the number of contrasts. 'SD' and 'rest:graze' = the number of contrasts with incomplete data (and percentages) for standard deviations (SD) and rest:graze ratios. SRG = strategic-rest grazing, CG = continuous grazing, UG = ungrazed

Response		**Missing data types**	
	n	**SD**	**Rest:graze**
Biodiversity variables			
Plant diversity			
SRG-CG	(8, 13)	(5, 38%)	(0, 0%)
SRG-UG	(9, 22)	(4, 18%)	(9, 41%)
Plant richness			
SRG-CG	(17, 50)	(31, 62%)	(6, 12%)
SRG-UG	(13, 23)	(6, 26%)	(8, 35%)
Production variables			
Animal production per unit area			
SRG-CG	(27, 86)	(37, 43%)	(3, 3.5%)
Weight gain			
SRG-CG	(62, 154)	(61, 40%)	(2, 1.3%)

Table A2.3. Publication bias within each of dataset of the four response variables and each comparison. 'n' = the number of studies, and the number of contrasts. β_0, z and P are output from Egger's regression test (Egger et al. 1997; Sterne and Egger 2005). A significant intercept term (β_0) is indicative of publication bias, and is analogous to funnel-plot asymmetry. SRG = strategic-rest grazing, CG = continuous grazing, UG = ungrazed

Response		**Model terms**		
	n	**β_0**	***z***	***P***
Biodiversity variables				
Plant diversity				
SRG–CG	(8, 13)	-0.144	-0.572	0.567
SRG–UG	(9, 22)	0.087	0.536	0.592
Plant richness				
SRG–CG	(17, 50)	0.046	0.791	0.429
SRG–UG	(13, 23)	0.191	1.827	0.068
Production variables				
Animal production per unit area				
SRG–CG	(27, 86)	-0.023	-0.443	0.658
Weight gain				
SRG–CG	(62, 154)	-0.031	-0.929	0.353

Table A2.4. Mean (± 1 s.e.) rest:graze ratio of multi-paddock and seasonal SRG systems for studies included in meta-analyses. SRG = strategic-rest grazing, CG = continuous grazing, UG = ungrazed

Biodiversity variables	**Multi-paddock**	**Seasonal**
Plant diversity		
SRG-CG	4.00 ± 3.928	2.222 ± 1.788
SRG-UG	3.077 ± 1.144	2.746 + 1.419
Plant richness		
SRG-CG	3.396 ± 5.032	1.864 ± 2.036
SRG-UG	2.186 ± 0.886	0.848 ± 0.568
Production variables		
Animal production per unit area		
SRG-CG	5.935 ± 4.392	0.556 ± 0.711
Weight gain		
SRG-CG	5.313 ±4.045	1.104 ± 1.062

Appendix 3

Table A3.1. Grazing information for sites as communicated by landholders. DSE = Dry sheep equivalent when grazed, 1 DSE = 0.2 animal units (AU). CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Cluster	Grazing contrast	Grazing treatment	DSE ha^{-1}	Months rest (per year)	Type of livestock grazing paddock	Length of current management (years)	Dung counts (pellets or pats per m^2) Kangaroo	Sheep/Goat	Cattle
1	1	AGM	30.00	11	Cattle	13	2.53	0.00	0.26
1	1	CON	0	12	NA	5.25	6.52	0.07	0.00
1	1	TGM	0.82	3	Sheep	>50	0.50	9.61	0.25
2	2	CON	0	12	NA	5.25	6.00	2.29	0.00
2	2	AGM	0.40	4	Sheep	30	4.63	4.46	0.00
2	3	AGM	0.50	4	Sheep	30	4.45	11.13	0.00
2	3	TGM	0.84	2.6	Sheep & cattle	6	1.21	25.84	0.00
3	4	CON	0	12	NA	6.7	4.45	0.01	0.00
3	4	TGM	0.20	3	Cattle	50	2.85	0.11	0.10
3	5	CON	0	12	NA	6.7	0.13	0.14	0.02
3	5	AGM	0.37	6	Sheep & Cattle	35	0.09	0.09	0.19
3	6	CON	0	12	NA	6.7	3.49	7.77	0.02
3	6	TGM	0.20	3	Cattle	>50	2.68	6.83	0.15
3	7	CON	0	12	NA	6.7	2.03	9.06	0.00
3	7	AGM	0.82	6	Sheep & Cattle	35	2.43	16.55	0.00
4	8	AGM	0.25	8	Sheep & Cattle	5	1.81	6.89	0.03
4	8	TGM	0.25	0	Sheep	>50	1.90	9.25	0.00
4	9	AGM	0.25	8	Sheep & Cattle	5	3.47	11.75	0.07
4	9	TGM	0.20	0	Cattle	14.5	2.23	0.54	0.02
5	10	CON	0	12	NA	6.6	4.98	0.28	0.03
5	10	TGM	1.19	4	Cattle	>50	2.71	1.14	0.15
5	11	CON	0	12	NA	6.6	3.73	7.98	0.01
5	11	TGM	0.16	4	Cattle	>50	3.93	5.45	0.06
6	12	AGM	0.15	6	Goats & cattle	7	1.11	11.66	0.03
6	12	CON	0	12	NA	35	4.52	0.89	0.00

Cluster	Grazing contrast	Grazing treatment	DSE ha^{-1}	Months rest (per year)	Type of livestock grazing paddock	Length of current management (years)	Dung counts (pellets or pats per m^2)		
							Kangaroo	Sheep/Goat	Cattle
6	13	AGM	0.15	6	Goats & cattle	7	2.65	10.55	0.01
6	13	TGM	0.30	3	Cattle	15	2.83	2.31	0.04

Table A3.2 Grazing regime information for sites as communicated by landholders. CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Cluster	Grazing contrast	Grazing treatment	Rotation frequency per year*
1	1	AGM	1-5 days, 8 month rest. Variable.
1	1	CON	NA
1	1	TGM	Nil
2	2	CON	NA
2	2	AGM	Depending on seasonal conditions
2	3	AGM	Depending on seasonal conditions
2	3	TGM	Nil
3	4	CON	NA
3	4	TGM	9 months to nil
3	5	CON	NA
3	5	AGM	4 months
3	6	CON	NA
3	6	TGM	9 months to nil
3	7	CON	NA
3	7	AGM	4 months
4	8	AGM	2 weeks, 3 month rest
4	8	TGM	Nil
4	9	AGM	2 weeks, 3 month rest
4	9	TGM	Nil
5	10	CON	NA
5	10	TGM	Nil
5	11	CON	NA
5	11	TGM	Nil
6	12	AGM	2 weeks to 3 months
6	12	CON	NA
6	13	AGM	2 weeks to 3 months
6	13	TGM	9 months to nil

* Typical rotation strategies. These strategies were not strictly employed on properties, and depended on seasonal conditions, all AGM managed adaptively according to seasonal conditions. Rest was employed on TGM properties when feed and water availability became too low to support livestock.

Table A3.3. Average rainfall (MAP), soil and vegetation characteristics of study clusters, and grazing contrasts compared within each cluster. CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Cluster	**MAP (mm)**	**Soil type[1]**	**ASC soil type[2]**	**Land System[3]**	**Bioregion**	**Vegetation community (overstorey)**	**Gazing treatments compared**		
							CON	**AGM**	**TGM**
Cluster 1	400			Long Meadow	Darling Riverine Plains	*Chenopodium nitrariaceum*			
Contrast 1		Clay	Vertosol				✓	✓	✓
Cluster 2	354			Long Meadow	Darling Riverine Plains	*Eucalyptus largiflorens, Euc. coolabah, Acacia stenophylla, Eremophila bignoniiflora, Duma florulenta*			
Contrast 2		Clay	Vertosol				✓	✓	
Contrast 3		Clay	Vertosol					✓	✓
Cluster 3	310				Darling Riverine Plains				
Contrast 4		Clay	Vertosol	Nelyambo		*Ere. bignoniiflora, Aca. stenophylla, Dum. florulenta*	✓		✓
Contrast 5		Clay	Vertosol				✓	✓	
Contrast 6		Sand	Calcarosol	East Toorale		*Euc. populnea, Grevillea striata, Atalaya hemiglauca, Aca. excelsa, Aca. cambagei, Alectryon oleifolius, Casuarina cristata, Ere. sturtii, Ere. mitchellii, Dodonaea viscosa*	✓		✓
Contrast 7		Sand	Calcarosol				✓	✓	
Cluster 4	303				Mulga Lands				
Contrast 8		Sand	Kandosol	Goonery		*Euc. populnea, Aca. cambagei, Aca. aneura, Aca. excelsa, Ata. hemiglauca, Flindersia maculosa, Ale. oleifolius, Dod. viscosa, Ere. sturtii*		✓	✓
Contrast 9		Clay	Vertosol	Walkdens		*Dum. florulenta*		✓	✓
Cluster 5	275			Warrego	Mulga Lands				
Contrast 10		Clay	Vertosol				✓		✓

Cluster	**MAP (mm)**	**Soil type**[1]	**ASC soil type**[2]	**Land System**[3]	**Bioregion**	**Vegetation community (overstorey)**	**Gazing treatments compared**		
							CON	**AGM**	**TGM**
Contrast 11		Sand	Kandosol			*Dum. florulenta, Eragrostis australasica* *Ata. hemiglauca, Aca. excelsa, Alectryon oleifolius, Aca. aneura, Euc. populnea, Ere. sturtii, Dod. viscosa, Cas. cristata*	✓		✓
Cluster 6	275			Waverly	Mulga Lands	*Aca. aneura, Euc. populnea, Aca. excelsa, Gre. striata, Senna spp., Dod. viscosa, Ere. longifolia, Ere. sturtii,*			
Contrast 12		Sand	Calcarosol				✓	✓	
Contrast 13		Clay	Calcarosol					✓	✓

[1] Grouped as sand or clay for analysis in this study

[2] Australian Soil Classification. Isbell, R. 2016. The Australian soil classification. CSIRO publishing, Clayton South, Australia.

[3] Walker, P. J. 1991. Land System of Western NSW, Technical Report No. 25. Soil Conservation Service of NSW, Sydney.

Table A3.3. Species frequency of occurrence and functional characteristics of understorey species recorded within quadrats. O = origin, N = native, E = exotic, LH = life history, A = annual, P = perennial, FG =functional group, AF = annual forb, PF = perennial forb, AG = annual grass, PG = perennial grass, Height = height at maturity, SL = seed length, LA = leaf area index, RI = rarity index, P = palatability, L = low palatability, M = moderate palatability, H = highly palatable

Species	Quadrat frequency - Spring		Quadrat frequency - Autumn		Functional traits							
	Sand	Clay	Sand	Clay	O	LH	FG	Height (cm)	SL (mm)	LA (mm^2)	RI	P
Abutilon otocarpum	0.189	0	0.169	<0.001	N	P	PF	0.6	3	4800	67.9	L
Actinobole uliginosum	0.042	0	0.041	0	N	A	AF	0.05	0.7	65	85.8	L
Alternanthera angustifolia	0.024	0.089	0.016	0.006	N	A	AF	0.05	1.5	60	84.57	H
Amaranthus mitchellii	0	0.228	0	0.005	N	A	AF	0.4	2.5	1250	85.19	H
Amphipogon caricinus v. caricinus	0.001	0	0.009	0	N	P	PG	0.5	3	450	98.15	L
Aristida anthoxanthoides	0	0.009	0	0	N	A	AG	0.5	6	135	98.77	H
Aristida contorta	0.477	0	0.210	0	N	A	AG	0.5	4	100	62.35	M
Aristida jerichoensis v. jerichoensis	0.069	0	0.003	0	N	P	PG	0.9	4.5	3450	90.74	L
Asperula conferta	0	<0.001	0	0	N	P	PF	0.2	1.5	8	99.38	M
Astrebla elymoides	0	0.004	0	0	N	P	PG	1	7	1375	98.77	H
Astrebla lappacea	0	0.169	0	0.288	N	P	PG	0.9	5	2100	79.01	H
Astrebla pectinata	0	0.020	0	0	N	A	AG	1.2	5	1500	98.15	H
Atriplex eardleyae	0	0.010	0	0.004	N	P	PF	0.3		180	96.91	
Atriplex holocarpa	0	0.034	0	0	N	A	AF	0.4		1400	95.68	L
Atriplex leptocarpa	0	<0.001	0	0	N	P	PF	0.3		240	99.38	L
Atriplex limbata	0	0	0.002	0	N	P	PF	0.4		540	98.77	M
Atriplex muelleri	0	0.004	0	0	N	A	AF	0.4		1250	98.77	L
Atriplex stipitata	0.020	0	0.017	0	N	P	PF	1		300	94.44	L
Austrostipa scabra	0.021	0	0.019	0	N	P	PG	0.6	2	500	91.36	M
Boerhavia dominii	0.193	0.050	0.282	0.091	N	P	PF	0.05	4.5	800	37.04	H
Brachyscome ciliaris v. lanuginosa	0.010	<0.001	0.007	0	N	P	PF	0.45	2	1200	93.21	M
Brachyscome curvicarpa	0	0.002	0	0	N	A	AF	0.4		900	98.77	H
Brachyscome lineariloba	0.003	<0.001	0	0	N	A	AF	0.15	2.5	800	98.15	H
Brachyscome whitei	0.010	0	0	0	N	A	AF	0.15		1120	98.15	
Bromus diandrus	0	0.008	0	0	N	A	AG	1		3300	97.53	M
Brunonia australis	0.002	0	0	0	N	P	PF	0.3	3	1800	99.38	M

Species	Quadrat frequency - Spring		Quadrat frequency - Autumn		Functional traits							
	Sand	Clay	Sand	Clay	O	LH	FG	Height (cm)	SL (mm)	LA (mm^2)	RI	P
Bulbine semibarbata	0.022	0.021	0.041	0.028	N	A	AF	0.5	3	1350	70.99	H
Calandrinia eremaea	0.092	<0.001	0.050	0	N	A	AF	0.05	0.5	440	72.84	H
Calandrinia ptychosperma	0.006	0	0	0	N	A	AF	0.01	0.5	210	98.15	H
Calostemma purpureum	0	0.002	0	0	N	P	PF	0.5		10800	98.77	L
Calotis ancyrocarpa	0	0.010	0	0	N	A	AF	0.22	3	240	98.77	H
Calotis hispidula	0.168	0.089	0	0	N	A	AF	0.1	3	140	72.22	M
Calotis scabiosifolia	0	0	0	0.004	N	P	PF	0.45	2	7200	98.77	M
Cenchrus ciliaris	0.003	0	0.003	0	E	P	PG	1.5	1.3	2960	98.15	H
Centipeda thespidioides	0.014	0.002	0.007	0	N	P	PF	0.2	2.5	175	94.44	H
Cheilanthes sieberi	0	0	0.010	0	N	P	PF	0.4	0.1	12250	96.91	L
Chenopodium auricomum	0	0.058	0	0.046	N	P	PF	2	1.5	1800	93.83	L
Chenopodium carinatum	0.001	0	0	0	N	A	AF	0.3	0.5	600	99.38	
Chenopodium curvispicatum	0.027	0	0.009	0	N	P	PF	1	1.5	225	96.3	L
Chenopodium desertorum	0.009	0.005	0.012	0.002	N	P	PF	0.3	1.5	200	90.12	M
Chenopodium melanocarpum	0.220	0.008	0	0	N	A	AF	0.05	0.5	900	79.63	L
Chloris truncata	0.017	0.145	0.007	0.074	N	P	PG	0.5	2.2	700	78.4	H
Convolvulus erubescens	0.092	0.267	0.060	0.132	N	P	PF	0.05	4	2400	55.56	H
Crassula colorata v. acuminata	0.031	0	0	0	N	A	AF	0.15	0.5	15	93.21	H
Cullen graveolens	0	0.016	0	0	N	A	AF	0.8	3	700	98.77	H
Cullen tenax	0	0.007	0	0	N	P	PF	0.15	3	300	97.53	H
Cuphonotus andraeanus	0.020	0	0	0	N	A	AF	0.25	1.5	150	97.53	
Cymbonotus maidenii	0	0.005	0	0.002	N	A	AF	0.4		24000	96.91	L
Cyperus bifax	0	<0.001	0	0	N	P	PF	0.9	1.5	2720	99.38	L
Cyperus gilesii	0	<0.001	0	0	N	A	AF	0.35	5	15750	99.38	L
Dactyloctenium radulans	0.029	0.029	0.001	0	N	A	AG	0.2	2.5	720	87.04	H
Daucus glochidiatus	0.029	0.068	0.012	0.130	N	A	AF	0.6		1500	70.37	H
Dichanthium sericeum	0.003	0.303	0	0.149	N	P	PG	1.2	3	600	74.07	H
Digitaria ammophila	0.009	0.003	0	<0.001	N	P	PG	0.8	3	1000	95.68	M
Digitaria coenicola	0.007	0	0	0	N	P	PG	0.8	3.5	900	98.77	H

Species	Quadrat frequency - Spring		Quadrat frequency - Autumn		Functional traits							
	Sand	Clay	Sand	Clay	O	LH	FG	Height (cm)	SL (mm)	LA (mm^2)	RI	P
Dissocarpus paradoxus	0	0	0.010	0	N	P	PF	0.5		45	98.15	L
Duma florulenta	0	<0.001	0	0.003	N	P	PF	3	4	700	97.53	L
Einadia nutans	0.009	0	0.009	<0.001	N	P	PF	0.2	1	600	91.36	H
Einadia polygonoides	0	<0.001	0	<0.001	N	A	AF	0.2	1	100	98.77	H
Enchylaena tomentosa	0.002	0.006	0.003	0	N	P	PF	1		30	95.06	L
Enneapogon avenaceus	0.463	0.123	0.458	0.096	N	A	AG	0.5	1.6	480	45.68	H
Enteropogon acicularis	0.083	0.004	0.069	0	N	P	PG	0.4		800	83.95	M
Eragrostis australasica	0	0.006	0	0	N	P	PG	3	1	600	98.15	L
Eragrostis cilianensis	0.001	0.003	0	0	E	A	AG	0.6	0.5	1400	98.15	L
Eragrostis dielsii	0.044	0.003	0.002	0	N	A	AG	0.4	2	160	90.12	H
Eragrostis eriopoda	0.204	0	0.214	0	N	P	PG	0.5	0.75	10	75.31	M
Eragrostis lacunaria	0	0.003	0	0	N	P	PG	0.5	0.5	140	98.15	H
Eragrostis microcarpa	0	<0.001	0	0	N	P	PG	0.6	0.4	200	99.38	L
Eragrostis setifolia	0.171	0.196	0.131	0.201	N	P	PG	0.6	0.4	260	31.48	H
Eremophila latrobei	0.019	0	0.004	0	N	P	PF	2		450	96.91	H
Eriochlamys squamata	0.002	0	0	0	N	A	AF			42	98.77	
Eriochloa crebra	0	0.183	0	0.047	N	P	PG	1	6	1500	79.01	H
Erodium crinitum	0.061	0.274	0.193	0.057	N	A	AF	0.5	10	1200	66.67	H
Euphorbia Drummondii	0.220	0.068	0.034	0.015	N	P	PF	0.05		50	61.11	L
Euphorbia planiticola	0	0	0	0.051	N	A	AF	0.5		250	95.06	H
Euphorbia tannensis	0	0	0.001	0	N	P	PF	1	3	490	99.38	L
Evolvulus alsinoides. v. villosicalyx	0	0	0.010	0	N	P	PF	0.4	1.5	150	95.68	M
Frankenia gracilis	0	0.002	0	0	N	P	PF	0.5		8.4	98.77	
Geranium solanderi v. solanderi	0	<0.001	0	0	N	P	PF	0.2		1500	99.38	M
Glossocardia bidens	0.004	0	0.001	0	N	P	PF	0.3	10	2700	96.91	L
Gnephosis tenuissima	0.004	0	0.001	0	N	A	AF	0.1	0.4	40	98.77	
Goodenia cycloptera	0.038	0	0.062	0	N	P	PF	0.3	6	1500	81.48	M
Goodenia fascicularis	0.090	0.041	0.194	0.223	N	P	PF	0.2	5	3500	55.56	H
Goodenia glauca	0	0.004	0	0.024	N	P	PF	0.3	4	800	95.06	H

Species	Quadrat frequency - Spring		Quadrat frequency - Autumn		Functional traits							
	Sand	Clay	Sand	Clay	O	LH	FG	Height (cm)	SL (mm)	LA (mm^2)	RI	P
Haloragis glauca	0	0.004	0	0	N	P	PF	0.4		360	98.15	H
Harmsiodoxa blennodioides	0.004	0	0	0	N	A	AF	0.3	2.2	2000	97.53	H
Hibiscus brachysiphonius	0.004	0.085	0	0.004	N	P	PF	0.5	3	2500	90.12	H
Hordeum leporinum	0	0.002	0	0	E	A	AG	0.2		1540	99.38	H
Iseilema membranaceum	0	0.028	0	0.005	N	A	AG	0.8	2.2	1000	95.06	H
Isoetopsis graminifolia	0.017	0	0	0	N	A	AF	0.05	2	100	95.68	H
Leiocarpa brevicompta	0	0.012	0	0.003	N	A	AF	0.6	2	175	96.91	M
Leiocarpa tomentosa	0	0.010	0	0.002	N	P	PF	0.7	5	120	96.3	M
Lepidium bonariense	0	0.008	0	0	E	A	AF	0.5	1.5	160	98.15	M
Lepidium oxytrichum	0.057	<0.001	0	0	N	A	AF	0.3	2	300	89.51	M
Lepidium papillosum	0	<0.001	0	0	N	A	AF	0.3	2	2000	99.38	L
Lepidium pseudohyssopifolium	0	0.002	0	0	N	P	PF	0.6	1.5	1350	99.38	
Leptorhynchos squamatus	0	0.003	0	0	N	P	PF	0.4	3	140	99.38	L
Lotus cruentus	0.003	0.036	0	0	N	P	PF	0.15	1.5	105	92.59	
Maireana aphylla	0	0.084	0	0.029	N	P	PF	1.5		8	91.98	L
Maireana coronata	0	0.103	0	0.106	N	P	PF	0.15		20	88.27	M
Maireana decalvans	0	0	0	0.004	N	P	PF	0.5		10	98.77	L
Maireana pyramidata	0.040	0	0.033	0	N	P	PF	1.5	1.5	5	92.59	L
Maireana villosa	0.066	0	0	0	N	P	PF	0.5		15	93.21	L
Malacocera tricornis	0	0.036	0	0.039	N	P	PF	0.8	2	20	93.83	M
Malvastrum americanum	0.002	0.124	0	0.054	E	P	PF	0.6		2100	76.54	L
Marsilea drummondii	0	0.050	0	0.020	N	P	PF	0.3		900	82.72	L
Medicago laciniata	0.006	0	0	0	E	A	AF	0.35	3	66	98.77	H
Medicago minima	0	0.038	0	0.089	E	A	AF	0.2	2	112	87.65	H
Medicago polymorpha	0	0.041	0	0.137	E	A	AF	0.2	4	540	90.12	H
Menkea australis	0	<0.001	0	0	N	A	AF	0.05	0.6	210	99.38	
Monachather paradoxus	0.038	0	0.008	0	N	P	PG	0.6	1	800	95.06	H
Neobassia proceriflora	0	0.044	0	0.020	N	P	PF	0.4		20	91.36	H
Neptunia gracilis	0.003	0	0	0	N	P	PF			250	99.38	H

Species	Quadrat frequency - Spring		Quadrat frequency - Autumn		Functional traits							
	Sand	Clay	Sand	Clay	O	LH	FG	Height (cm)	SL (mm)	LA (mm^2)	RI	P
Nicotiana megalosiphon v. megalosiphon	0.001	0	0	0	N	P	PF	0.9		13600	99.38	
Omphalolappula concava	0.002	0	0.002	0	N	A	AF	0.35	10	240	97.53	
Ophioglossum lusitanicum	0	0	0.009	0	N	A	AF	0.15		500	96.3	L
Osteocarpum acropterum v. deminuta	0.001	0.011	0	0.005	N	P	PF	0.2		6	95.68	M
Oxalis perennans	0.008	0.007	0.002	0.004	N	P	PF	0.1		60	88.89	
Panicum decompositum	0	0.355	0.001	0.371	N	P	PG	1	1.5	6000	64.2	H
Panicum effusum	0.038	0	0.006	0	N	P	PG	0.7	2	1500	91.98	H
Panicum queenslandicum	0	0.470	0	0.126	N	P	PG	0.8		12250	80.86	M
Parsonsia eucalyptophylla	0.002	0	0	0	N	P	PF			4800	98.77	M
Paspalidium constrictum	0.106	0	0.006	0	N	P	PG	0.6		450	94.44	H
Perotis rara	0.006	0	0	0	N	A	AG	0.4		200	97.53	M
Phlegmatospermum cochlearinum	0.004	0.013	0	0	N	A	AF	0.35	2	1800	95.68	L
Phyllanthus lacunarius	0	0.004	0	0	N	A	AF	0.25	1.5	140	98.77	L
Phyllanthus maderaspatensis	0	0.006	0	0.003	N	P	PF	0.5	1.5	200	96.91	L
Phyllanthus virgatus	0	0.002	0	0	N	P	PF	0.5	1.5	1400	99.38	L
Pimelea penicillaris	0.017	<0.001	0.017	0	N	P	PF	2	4	114	93.83	L
Pimelea simplex v. continua	0	0	0	0.003	N	A	AF	0.5	3.5	60	98.77	L
Pimelea trichostachya	0.087	0.003	0.010	0	N	P	PF	0.75	3.5	45	87.04	L
Plagiobothrys plurisepaleus	0	<0.001	0	0	N	A	AF	0.05	1.5	135	99.38	H
Plantago cunninghamii	0	<0.001	0	0	N	A	AF	0.15	3	1500	99.38	H
Plantago turrifera	0.023	0.093	0	0.002	N	A	AF	0.1	3	2500	85.8	H
Pluchea dentex	0	0.003	0	0	N	P	PF	0.6	1.5	360	98.15	
Podolepis jaceoides	0	0.012	0	0	N	P	PF	0.7	3	4000	98.77	L
Podolepis muelleri	0.013	0	0	0	N	A	AF	0.22	1.5	600	97.53	L
Polycarpaea corymbosa	0.007	0	0	0.002	N	A	AF	0.3		6	97.53	L
Portulaca oleracea	0.460	0.199	0.149	0.034	N	A	AF	0.05	1	375	34.57	H
Pseudognaphalium luteoalbum	0	0.003	0	0	N	A	AF	0.45		250	98.77	L
Ptilotus gaudichaudii v. parviflorus	0.017	0.004	0	0	N	A	AF	0.5	1.5	520	95.06	L
Ptilotus leucocoma	0.006	0	0	0	N	P	PF	0.25		210	98.77	L

Species	Quadrat frequency - Spring		Quadrat frequency - Autumn		Functional traits							
	Sand	Clay	Sand	Clay	O	LH	FG	Height (cm)	SL (mm)	LA (mm^2)	RI	P
Ptilotus nobilis v. nobilis	0	<0.001	0	0.005	N	P	PF	1	2	15000	97.53	H
Ptilotus nobilis v. semilanatus	0.001	0	0	0	N	P	PF	0.3	3	1200	99.38	H
Ptilotus polystachyus	0.062	0	0.003	0	N	P	PF	1	2	7560	89.51	H
Ptilotus sessilifolius v. sessilifolius	0.038	0	0.034	0	N	P	PF	1		2000	91.36	M
Pycnosorus chrysanthes	0	0.002	0	0	N	A	AF	0.6	2	700	98.77	L
Ranunculus pentandrus. v. platycarpus	0	0.002	0	0	N	A	AF	0.3	2	500	99.38	L
Rapistrum rugosum	0	0.018	0	0	E	A	AF	0.6	2	12500	98.77	M
Rhagodia spinescens	0.008	0	0	0	N	P	PF	3	1.5	300	96.91	M
Rhodanthe floribunda	0.053	0.065	0	0	N	A	AF	0.3	5	40	85.8	L
Rhodanthe uniflora	0	0.033	0	0	N	A	AF	0.07	3.5	5	98.15	L
Salsola australis	0.069	0.057	0.028	0.030	N	A	AF	1		90	69.14	M
Salvia verbenaca	0	0.006	0	0	E	P	PF	0.7	3	8000	98.77	L
Sauropus trachyspermus	0.049	<0.001	0.068	0.007	N	P	PF	0.2	5	60	75.93	
Schoenia ramosissima	0.042	0	0	0	N	A	AF	0.15	2.5	40	96.3	L
Sclerolaena parallelicuspis	0	0	0.006	0	N	P	PF	0.3		25	99.38	M
Scleroblitum atriplicinum	0	0.024	0	0	N	A	AF	0.15	1	1500	97.53	H
Sclerolaena articulata	0.006	<0.001	0	0	N	P	PF	0.4		15	98.77	L
Sclerolaena bicornis v. bicornis	0.191	0.096	0.102	0.074	N	P	PF	0.8		25	63.58	M
Sclerolaena brachyptera	0	0.030	0	0.030	N	P	PF	0.2		15	95.06	H
Sclerolaena calcarata	0	0.145	0	0.113	N	P	PF	0.3		15	82.1	M
Sclerolaena convexula	0.116	0	0.213	0	N	P	PF	0.4		10	75.93	L
Sclerolaena decurrens	0.002	0	0.007	0	N	P	PF	0.3		15	95.68	L
Sclerolaena divaricata	0.012	0.103	0.028	0.077	N	P	PF	0.75		12	78.4	L
Sclerolaena intricata	0.007	0	0	0	N	P	PF	0.7		50	98.77	L
Sclerolaena lanicuspis	0.037	0.036	0.049	0	N	P	PF	0.4		15	87.04	H
Sclerolaena muricata v. villosa	0.026	0.063	0.002	0.016	N	P	PF	1.5		40	79.01	L
Sclerolaena muricata v. semiglabra	0.027	0	0.033	<0.001	N	P	PF	1.5		60	88.89	L
Sclerolaena patenticuspis	0.001	0	0	0	N	P	PF	0.3		10	99.38	M
Sclerolaena stelligera	0.001	0.077	0	<0.001	N	P	PF	0.3		15	95.68	H

Species	Quadrat frequency - Spring		Quadrat frequency - Autumn		Functional traits							
	Sand	Clay	Sand	Clay	O	LH	FG	Height (cm)	SL (mm)	LA (mm^2)	RI	P
Sclerolaena tricuspis	0	0.003	0	0	N	P	PF	0.75		15	99.38	L
Senecio diaschides	0	0.002	0	0	N	P	PF	1		3000	98.77	
Sida trichopoda	0	0.193	0	0.102	N	P	PF	0.4	4	240	74.07	H
Sisymbrium erysimoides	0.001	0	0	0	E	A	AF	0.8		6000	99.38	L
Solanum ellipticum	0.018	0	0.030	0	N	P	PF	1	3	2400	85.8	
Solanum esuriale	0.119	0.059	0.089	0.106	N	P	PF	0.3	3	1200	59.88	L
Solanum parvifolium	0	0	0.001	0	N	P	PF	1	2.5	600	99.38	
Sonchus oleraceus	0.003	0.14	0	0.035	E	A	AF	1.1	3	42000	86.42	H
Sporobolus actinocladus	0	0.06	0.016	0.018	N	P	PG	0.8	0.8	450	91.36	H
Sporobolus caroli	0.021	0.217	0.021	0.074	N	P	PG	0.6	0.7	680	62.35	H
Sporobolus mitchellii	0	0.005	0	0	N	P	PG	1	1	200	98.15	M
Stemodia glabella	0	0.004	0	0	N	P	PF	0.4	0.5	250	96.91	L
Stenopetalum lineare	0	0.008	0	0	N	A	AF	0.5	1.2	700	98.77	H
Swainsona greyana	0	0.006	0	0.046	N	P	PF	1.5	4	9000	95.06	H
Swainsona microphylla	0.016	0.003	0.026	0	N	P	PF	0.6	1.5	1500	90.74	L
Tephrosia sphaerospora	0.010	0	0.007	0.002	N	P	PF	0.3	2.5	630	94.44	
Tetragonia tetragonioides	0.049	0.004	0.057	0.152	N	A	AF	0.05	3	5000	72.84	H
Teucrium racemosum	0.001	0.020	0	0.028	N	P	PF	0.4	2	120	87.04	H
Thyridolepis mitchelliana	0.090	0	0.070	0	N	P	PG	0.5	4	270	88.27	H
Tragus australianus	0.072	0.013	0.001	0	N	A	AG	0.4	2.1	360	82.72	M
Trianthema triquetra	0	0.037	0.007	0.035	N	A	AF	0.05	1.2	120	87.65	
Tribulus terrestris	0.040	0.006	0.036	0.071	E	P	PF	0.05		48	69.75	L
Triglochin calcitrapa	0	0.012	0	0	N	A	AF	0.12	1	880	98.77	M
Trigonella suavissima	0	0.002	0	0	N	A	AF	0.5	1	300	98.77	H
Tripogon loliiformis	0.028	0.067	0.233	0.052	N	P	PG	0.4	2.2	19.5	70.99	H
Triraphis mollis	0.182	0	0.013	0	N	P	PG	0.8	2.5	2000	84.57	L
Verbena officinalis	0	<0.001	0	0	E	P	PF	1		2100	99.38	L
Vittadinia cuneata. v. hirsuta	0.029	0	0	0	N	P	PF	0.3	3.5	125	93.21	L
Vittadinia cunneata	0.001	0.004	0.011	0	N	P	PF	0.4	3.5	125	94.44	L

Species	Quadrat frequency - Spring		Quadrat frequency - Autumn		Functional traits							
	Sand	**Clay**	**Sand**	**Clay**	**O**	**LH**	**FG**	**Height (cm)**	**SL (mm)**	**LA (mm^2)**	**RI**	**P**
Vittadinia sulcata	0.006	0.01	0.001	0	N	A	AF	0.3	4	1200	96.91	L
Vittadinia dissecta	0	0	0.002	0	N	P	PF	0.35		125	98.77	L
Vittadinia eremaea	0	0	0.001	0	N	A	AF	0.25	4	500	99.38	L
Wahlenbergia spp.	0.008	0.02	0	0.002	N	A	AF				91.98	
Zygophyllum ammophilum	0.001	0.036	0.006	0.044	N	A	AF	0.25		150	91.98	M
Zygophyllum iodocarpum	0	0.034	0	0	N	A	AF	0.2	4	240	96.3	H
Digitaria spp. #1	0.013	0	0	0							98.15	
Digitaria spp. #2	0.063	0	0	0							94.44	
Lepidium spp.	0	<0.001	0	0							99.38	
Sida spp. #1	0.251	0.060	0.231	0.022		P	PF				56.79	
Sida spp. #2	0	0	0.001	0.002		P	PF				95.06	
Sida spp. #3	0	0.039	0	0.014		P	PF				98.77	
Unidentified species (spring) x 13												
Unidentified species (autumn) x 35												

Table A3.4. Predicted (weighted) means for species frequency in sites, and functional diversity indices at the site scale. CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Response variable	**CON**	**AGM**	**TGM**	***P*-value**
Total annual	0.35	0.31	0.29	0.418
Annual forb	0.24	0.21	0.19	0.592
Annual grass	0.08	0.06	0.06	0.097
Total perennial	0.65	0.69	0.71	0.418
Perennial forb	0.36	0.39	0.36	0.585
Perennial grass	0.27	0.28	0.33	0.398
Native	0.95	0.95	0.96	0.489
Exotic	0.91	0.90	0.93	0.359
Unpalatable	0.20	0.20	0.18	0.45
Moderate palatable	0.25	0.23	0.22	0.789
Palatable	0.56	0.57	0.60	0.546
Height	0.54	0.52	0.54	0.759
Leaf area index	1599.00	1487.66	1760.54	0.505
Seed length	2.79	2.67	2.80	0.676
FRic				0.023
Spring	25.08	19.88	16.74	
Autumn	16.87	19.97	24.35	
FEve	0.65	0.61	0.60	0.068
FDiv	0.29	0.30	0.29	0.838
RaoQ	0.11	0.12	0.12	0.844

Table A3.5. Predicted (weighted) means for species frequency in plots and functional diversity indices at the plot scale. CON = conservation management, AGM = alternative grazing management, TGM = traditional grazing management

Response variable	**CON**	**AGM**	**TGM**	***P*-value**
Total annual	0.33	0.31	0.27	0.122
Annual forb	0.22	0.21	0.18	0.264
Annual grass	0.07	0.05	0.05	0.075
Total perennial	0.67	0.69	0.73	0.122
Perennial forb	0.35	0.37	0.35	0.372
Perennial grass	0.32	0.32	0.39	0.125
Native	0.97	0.97	0.98	0.352
Exotic	0.02	0.02	0.2	0.387
Unpalatable	0.20	0.20	0.17	0.130
Moderate palatable	0.20	0.19	0.19	0.887
Palatable	0.59	0.61	0.64	0.264
Height	0.55	0.52	0.54	0.408
Leaf area index	1658.61	1549.88	1804.02	0.265
Seed length				0.007
Sand	2.47	2.58	2.28	
Clay	3.05	2.84	3.24	
FRic				0.007
Spring	9.70	9.65	8.46	
Autumn	8.57	9.45	10.75	
FEve	0.70	0.66	0.67	0.11
FDis	0.28	0.28	0.28	0952
RaoQ	0.10	0.10	0.11	0.687

Table A3.6. Pearsons correlations for clay soils. * = significant at *P* ≤ 0.05.

		Spring				Autumn			
		Richness	Evenness	Diversity	Turnover	Richness	Evenness	Diversity	Turnover
Rainfall	Average rainfall	-0.485*	-0.502*	-0.538*	0.657*	-0.017	0.055	0.098	-0.269
	Three month	-0.162	-0.095	-0.178	0.229	0.448*	0.394*	0.558*	-0.353*
	Six month	0.348*	0.039	0.192	0.104	0.422*	0.454*	0.568*	-0.388*
	Twelve month	0.419*	0.188	0.348*	-0.267	0.444*	0.394*	0.558*	-0.354*
Spatial	Longitude	-0.498*	-0.491*	-0.544*	0.632*	0.022	0.088	0.159	-0.246
	Latitude	0.298*	0.169	0.239	-0.085	0.213	0.228	0.317*	-0.114
Soil	pH	–	–	–	–	-0.08	0.151	-0.031	-0.324
	EC	–	–	–	–	0.137	0.015	0.149	0.207
	Organic nitrogen	–	–	–	–	0.022*	0.088	0.159*	-0.246*
	Organic carbon	–	–	–	–	0.213*	0.228	0.317*	-0.114*
	Bulk density	–	–	–	–	0.049	-0.141	-0.01	0.409*
Grazing	Average DSE	-0.172	-0.109	-0.173	0.234	0.011	0.141	0.118	-0.221
	Ave rest	0.235	0.14	0.193	0.124	-0.027	0.321*	0.071	-0.279
	Kangaroo dung	0.119	-0.022	0.081	0.22	0.232	0.017	0.236	0.184
	Sheep/ goat dung	-0.011	-0.091	-0.058	0.064	0.06	-0.254	-0.028	0.178
	Cattle dung	-0.109	-0.011	-0.071	0.026	0.126	0.203	0.19	-0.295*

Table A3.7. Pearsons correlations for sand soils. * = significant at *P* ≤ 0.05.

		Spring				Autumn			
	Diversity measure	Richness	Evenness	Diversity	Turnover	Richness	Evenness	Diversity	Turnover
Rainfall	Average rainfall	0.618*	0.131	0.586*	-0.674*	0.071	-0.536*	-0.103	-0.17
	Three month	0.248	0.138	0.244	-0.504*	0.181	0.356*	0.262	0.018
	Six month	-0.122	-0.119	-0.139	-0.235	-0.072	0.329*	0.033	0.285
	Twelve month	-0.075	-0.153	-0.102	-0.27	-0.218	-0.312	-0.294	0.397*
Spatial	Longitude	0.387*	0.177	0.376*	-0.551*	-0.221	-0.537*	-0.357*	0.224
	Latitude	-0.726*	-0.12	-0.687*	0.545*	-0.293	0.287	-0.175	0.490*
Soil	pH					0.209	-0.345	0.058	-0.169
	EC					0.086	0.158*	0.175	-0.013
	Organic nitrogen					0.242	0.214	0.272	0.035
	Organic carbon					0.245	0.341*	0.336*	-0.024
	Bulk density					0.003	-0.325	-0.1	-0.299
Grazing	Average DSE	0.085	-0.068	0.041	-0.378*	0.016	-0.385*	-0.112	-0.159
	Ave rest	0.238	0.114	0.24	0.122	-0.311	-0.059	-0.258	0.221
	Kangaroo dung	-0.089	0.255	0.011	-0.06	-0.232	-0.112	-0.235	-0.117
	Sheep/ goat dung	0.407*	0.055	0.379*	-0.195	-0.133	-0.189	-0.145	0.102
	Cattle dung	0.18	0.172	0.182	-0.092	0.137	0.001	0.098	-0.03

Appendix 4

Table A4.1. Soil surface indicators assessed as part of the landscape function assessment, and contribution of indices to difference landscape function indices.

Indicator	Description and brief methodology	Stability	Infiltration	Nutrient cycling
Rainsplash Protection	The degree to which physical surface cover and projected plant cover ameliorate the effect of raindrops impacting on the soil surface. 5 classes; 1 = no rainsplash protection – 5 = very high rainsplash protection	X		
Perennial Vegetation Cover	The contribution of the below-ground biomass of perennial vegetation in contributing to nutrient cycling and infiltration processes. Determined from butt lengths of grasses and canopy cover and density of trees and shrubs. 4 classes; 1 = no below ground contribution – 4 = high below ground contribution.		X	X
Litter	The amount, origin and degree of decomposition of plant litter. 3 components; (1) percent cover of plant litter, 10 classes; 1 = <10% – 10 = 100%, >170 mm thick; (2) orgin, local or transported; (3) degree of decomposition/ incorporation, 4 classes, 1 = nil decomposition – 4 = extensive decomposition	X		
Cryptogam cover	The percentage cover of soil crust by cryptogams. 5 classes; 0 = no stable crust present – 4 = extensive contribution	X		X
Crust broken-ness	The extent to which the surface crust is broken, leaving loosely attached soil material available for erosion. 5 classes; 0 = no crust present – 4 = crust present but intact, smooth	X		
Erosion type and severity	The type and severity of recent/current soil erosion i.e. soil loss from the query zone. 5 forms of erosion (rills and gullies, terracettes, sheeting, scalding, pedestalling) and 4 severity classes (insignificant – severe)	X		
Deposited materials	The nature and amount of alluvium transported to and deposited on the query zone. 4 classes; 1 = extensive amount available – 4 = none or small amount of material available	X		

Surface roughness	The surface roughness for its capacity to capture and retain mobile resources such as water, propagules, topsoil and organic matter. 5 classes (<3 mm relief in soil surface, smooth – very deep depressions or cracks >100 mm, extensive retention		X	X
Surface Nature (resistance to disturbance)	The ease with which the soil can be mechanically disturbed to yield material suitable for erosion by wind or water. 5 classes; 5 = non-brittle – 1 = loose sandy surface	X	X	
Slake test	The stability of natural soil fragments to rapid wetting. 5 classes; 0 = not applicable – 4 = very stable	X	X	
Soil texture	Classify the texture of the surface soil, and relate this to permeability. Uses a pedologists' moist bolus test. 4 classes; 1 = silty clay to heavy clay – 4 = sandy to clayey sand		X	

Appendix 5

Table A5.1. Species list and functional trait information for all species recorded in surveys. N = native, E = exotic, PF = perennial forb, AF = annual forb, PG = perennial grass, AG = annual grass, L = low palatability, M = moderate palatability, H = high palatability

Species	Frequency in plots	Mean frequency in quadrats	Origin	Functional group	Life form	Life history	Palatability	Height (cm)	Seed length (mm)	Leaf area index (mm^2)	Rarity index
Abutilon otocarpum	0.373	0.052	N	PF	S	P	L	60	3.0	4800	0.78
Actinobole uliginosum	0.706	0.176	N	AF	F	A	L	5	0.7	65	0.41
Aristida behriana	0.020	0.000	N	PG	G	P	M	40	8.0	75	1.00
Aristida contorta	0.745	0.159	N	AG	G	A	M	50	4.0	100	0.41
Aristida jerichoensis	0.667	0.133	N	PG	G	P	L	90	4.5	3450	0.59
Austrostipa scabra	0.667	0.153	N	PG	G	P	M	60	2.0	500	0.53
Boerhaevia dominii	0.078	0.009	N	PF	F	P	H	5	4.5	800	0.94
Chenopodium desertorum	0.235	0.033	N	PF	S	P	M	30	1.5	200	0.86
Convolvulus erubescens	0.039	0.002	N	PF	F	P	H	5	3.7	2400	0.98
Digitaria sp.	0.647	0.059	N	PG	G	P					0.65
Eragrostis eriopoda	0.961	0.458	N	PG	G	P	M	50	0.8	10	0.08
Eragrostis setifolia	0.118	0.000	N	PG	G	P	H	60	0.4	260	1.00
Eremophila latrobei	0.137	0.004	N	PF	S	P	H	200		450	0.98
Lepidium oxytrichum	0.020	0.002	N	AF	F	A	M	30	2.0	300	0.98
Maireana villosa	0.569	0.039	N	PF	S	P	L	50		15	0.78
Monachather paradoxa	0.510	0.059	N	PG	G	P	H	60	1.0	800	0.71
Portulaca oleracea	0.255	0.020	N	AF	F	A	H	5	1.0	375	0.86
Ptilotus sessifolius	0.392	0.004	N	PF	F	P	M	100		2000	0.96
Salsola australis	0.333	0.063	N	AF	S	A	M	100		90	0.78

Species	Frequency in plots	Mean frequency in quadrats	Origin	Functional group	Life form	Life history	Palatability	Height (cm)	Seed length (mm)	Leaf area index (mm^2)	Rarity index
Sclerolaena bicornis	0.059	0.013	N	PF	S	P	M	80		25	0.96
Sclerolaena convexula	0.941	0.440	N	PF	S	P	L	40		10	0.04
Sida cunninghamii	0.529	0.031	N	PF	S	P	H				0.80
Solanum ellipticum	0.882	0.041	N	PF	F	P		100	3.0	2400	0.69
Thyridolepis mitchelliana	0.490	0.063	N	PG	G	P	H	50	4.0	270	0.73
Tribulus terrestris	0.078	0.000	E	PF	F	P	L	5		48	1.00
Tripogon lolliformis	0.941	0.351	N	PG	G	P	H	40	2.2	19.5	0.08
Triraphis mollis	0.039	0.011	N	PG	G	P	L	80	2.5	2000	0.98
Vittadinia cuneata	0.118	0.009	N	AF	F	A	L				0.94
Wahlenbergia communis	0.020	0.000	N	AF	F	A					1.00
Enteropogon acicularis	0.078	0.004	N	PG	G	P	M	40		800	0.96
Podolepis capillaris	0.020	0.000	N	AF	F	A	H	45		500	1.00
Einadia nutans	0.098	0.004	N	PF	F	P	H	20	1.0	600	0.96
Ptilotus polystachyus	0.098	0.004	N	PF	F	P	H	100	2.0	7560	0.96
Dianella porracea	0.020	0.000	N	PF	G	P		120	3.3		1.00
Unidentified 1	0.020	0.000			G						1.00
Swainsona microphylla	0.020	0.002	N	PF	F	P	L	60	1.5	1500	0.98
Goodenia fascicularis	0.020	0.002	N	PF	F	P	H	20	5.0	3500	0.98
Sclerolaena muricata, subsp. semiglabra	0.020	0.002	N	PF	S	P	L	150		60	0.98
Solanum esuriale	0.137	0.000	N	PF	F	P	L	30	3.0	1200	1.00
Evolvulus aslinoides, subsp. villosicalyx	0.020	0.002	N	PF	F	P	M	40	1.5	150	0.98
Daucus glochidiatus	0.020	0.000	N	AF	F	A	H	60		1500	1.00

Species	Frequency in plots	Mean frequency in quadrats	Origin	Functional group	Life form	Life history	Palatability	Height (cm)	Seed length (mm)	Leaf area index (mm^2)	Rarity index
Euphorbia drummondii	0.020	0.002	N	PF	F	P	L	5	1.5	50	0.98
Hibiscus sturtii	0.020	0.000	N	PF	S	P	M	600		1000	1.00
Themeda triandra	0.020	0.002	N	PG	G	P	M	120	10.0	2400	0.98